Technical Evaluation Report

For the Revised Performance Assessment for the Saltstone Disposal Facility at the Savannah River Site, South Carolina

Final Report

U.S. Nuclear Regulatory Commission
Office Of Federal And State Materials
And Environmental Management Programs
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ABBREVIATIONS/ACRONYMS

ACI	American Concrete Institute
ALARA	As Low As Is Reasonably Achievable
Am	americium
ARP	Actinide Removal Process
ASP	Alpha Strike Process
ASTM	American Society for Testing and Materials
Ba	barium
BGS	Below Ground Surface
C	carbon
CFR	Code of Federal Regulations
Cl	chlorine
CNWRA	Center for Nuclear Waste Regulatory Analyses
Cr	chromium
Cs	cesium
CSH	Calcium silicate hydrate
CSSX	Caustic Side Solvent Extraction
DCF	Dose Conversion Factor
DDA	Deliquification, Dissolution, and Adjustment
DNFSB	Defense Nuclear Facilities Safety Board
DOE	U.S. Department of Energy
DOE-SRS	U.S. Department of Energy, Savannah River Site
DRF	Dose release factor
DUST-MS	Disposal Unit Source Term – Multiple Species
DWPF	Defense Waste Processing Facility
E_h	Reduction Potential
EPA	U.S. Environmental Protection Agency
EXAFS	Extended X-ray absorption fine structure
FACT	Flow and Contaminant Transport
FDC	Future Disposal Cell
FTF	F-Tank Farm
GCL	Geosynthetic Clay Layer
GCU	Gordon Confining Unit
GSA	General Separations Area
HDPE	High Density Polyethylene
HELP	Hydrologic Evaluation of Landfill Performance
HLW	high-level radioactive waste
HTF	H-Tank Farm
I	iodine
IA	Interagency Agreement
IAEA	International Atomic Energy Agency
ICRP	International Commission on Radiological Protection
ITP	In-Tank Precipitation
K_d	distribution coefficient

K_h	hydraulic conductivity
LFRG	Low-Level Waste Disposal Facility Federal Review Group
LLW	low-level radioactive waste
MCU	Modular Caustic Side Solvent Extraction Unit (Modular CSSX Unit)
MEI	Maximally exposed individual
MOP	Member of the Public
MOU	Memorandum of Understanding
MST	Monosodium Titanate
NAS	National Academy of Sciences
NCRP	National Council on Radiation Protection and Measurements
NCRP	National Council on Radiological Protection
NDAA	Sect. 3116, National Defense Authorization Act for Fiscal Year 2005
Np	neptunium
NRC	U.S. Nuclear Regulatory Commission
PA	Performance Assessment
PGA	Peak Ground Acceleration
PMF	Probable Maximum Flood
PMP	Probable Maximum Precipitation
Pu	plutonium
QA	Quality Assurance
QAMP	Quality Assurance Management Plan
QC	Quality Control
Ra	radium
RAI	Request for Additional Information
Rh	rhodium
Sb	antimony
SCDHEC	South Carolina Department of Health and Environmental Control
SCHWMR	South Carolina Hazardous Waste Management Regulations
SDF	Saltstone Disposal Facility
Se	selenium
SPF	Saltstone Production Facility
SQAP	Software quality assurance plan
Sr	strontium
SRM	Staff Requirements Memorandum
SRS	Savannah River Site
SWPF	Salt Waste Processing Facility
Tc	technetium
TCCZ	Tan Clay Confining Zone
TCLP	Toxicity Characteristic Leaching Procedure
TEDE	Total Effective Dose Equivalent
TER	Technical Evaluation Report
TPB	tetraphenylborate
TRU	transuranic
U	uranium

UTR	Upper Three Runs
UTR-LZ	Upper Three Runs Lower Zone
UTR-UZ	Upper Three Runs Upper Zone
UV	Ultraviolet
WAC	Waste Acceptance Criteria
WCS	Waste Characterization System
WD	Waste Determination
WIR	waste-incidental-to-reprocessing
XANES	X-Ray absorption near edge structure
Yr	Year

EXECUTIVE SUMMARY

On November 23, 2009, the U.S. Department of Energy (DOE) submitted the "Performance Assessment for the Saltstone Disposal Facility at the Savannah River Site" (2009 PA) to the U.S. Nuclear Regulatory Commission (NRC) for review, as required by Section 3116(b) of the Ronald W. Reagan National Defense Authorization Act for Fiscal Year 2005 (NDAA or the Act). Section 3116(b) of the NDAA requires NRC to monitor DOE's disposal actions concerning certain wastes associated with spent fuel reprocessing that DOE, in consultation with NRC, has determined to be non-High Level Waste (HLW). Although radioactive material resulting from the reprocessing of spent nuclear fuel typically is defined as HLW, DOE may determine certain reprocessing waste is non-HLW, or Waste Incidental Reprocessing (WIR), if it does not need to be disposed of as HLW to manage the risks it poses. The 2009 PA is an update to DOE's February 28, 2005 Performance Assessment (PA) performed in support of the "Section 3116 Determination, Salt Waste Disposal, Savannah River Site" (DOE-WD-2005-001). The updated PA includes new information about issues addressed in the 2005 PA, and information about changes to the disposal cell design as well as an improved format for readability and technical clarity. Per the definition of NRC monitoring in the NDAA, this review focuses on DOE's compliance with the third criterion of the Act, which is that the disposal actions must comply with the performance objectives of NRC's LLW disposal facility regulations as presented in the Title 10 of the *Code of Federal Regulations* (CFR) Part 61, Subpart C. The purpose of DOE's PA is to demonstrate that its waste disposal strategy for the SDF remains in compliance with these performance objectives.

Approximately 136 million liters (36 million gallons) of liquid waste resulting from reprocessing spent nuclear fuel is stored in 49 underground carbon steel tanks at SRS. This waste is separated into two streams based on activity. The high activity fraction is HLW and is made into a glass waste form. The low activity fraction, called salt waste, is treated to reduce the concentrations of certain key radionuclides and then mixed with dry materials (i.e., cement, blast furnace slag, and fly ash) to form a grout waste form called saltstone. Saltstone is disposed of in underground disposal units in the Saltstone Disposal Facility (SDF). The performance of the SDF is the subject of DOE's revised PA and of this review. At the time of this review 24 million liters (6.3 million gallons) of salt waste has been disposed in Vault 4, equaling 42 million liters (11 million gallons) of saltstone (SRR-ESH-2011-00014).

NRC has conducted a review and confirmatory analysis of the 2009 PA and has documented this review and analysis in this Technical Evaluation Report (TER). NRC's review results are being provided to DOE in accordance with its monitoring role under the NDAA and are not intended to represent any regulatory authority related to DOE's disposal activities.

The NRC staff concludes it has reasonable assurance that waste disposal at the SDF meets the 10 CFR 61 performance objectives for protection of individuals against intrusion (§61.42), protection of individuals during operations (§61.43), and site stability (§61.44). However, based on its evaluation of DOE's results and independent sensitivity analyses conducted with DOE's models, the NRC staff no longer has reasonable assurance that DOE's disposal activities at the SDF meet the performance objective for protection of the general population from releases of radioactivity (§61.41). Although the NRC staff cannot conclude that the performance objective in §61.41 is met, based on DOE's results and NRC's own independent analyses, the potential dose to an off-site member of the public from DOE's disposal actions is still expected to be relatively low (i.e., approximately 1 mSv/yr [100 mrem/yr], the public dose limit in §20.1301)[1].

DOE concludes that any dose greater than 0.25 mSv/yr (25 mrem/yr) would occur more than 10,000 years after site closure; however, the staff disagrees with many of the assumptions in DOE's model. The staff expects that any exceedance of the §61.41 limit would occur many years after site closure but finds that DOE has not provided a sufficient basis for DOE's conclusion that any exceedances would occur beyond 10,000 years. In accordance with NUREG-1854, *NRC Staff Guidance for Activities Related to U.S. Department of Energy Waste Determinations*, the time for which the 0.25 mSv/yr (25 mrem/yr) dose limit in §61.41 must be met is generally[2] 10,000 years. Therefore, it seems likely that DOE could provide additional information or take mitigative actions in the short term that would provide reasonable assurance that salt waste disposal at the SDF meets the 10 CFR 61 Subpart C performance objectives. NRC has identified these items as monitoring factors in this TER.

In December 2005, the NRC staff documented a similar review of DOE's 2005 PA for the SDF. At that time, the NRC staff concluded that it had reasonable assurance that salt waste disposal at the SDF would meet the performance objectives of 10 CFR 61 provided certain assumptions in DOE's analyses were verified during monitoring. During the current review, the NRC staff carefully evaluated information related to these assumptions (i.e., information regarding saltstone oxidation, saltstone and disposal unit hydraulic conductivity, field-scale properties of as-emplaced saltstone, saltstone fracturing, numerical modeling of flow through fractures, radionuclide concentrations, moisture characteristic curves, and erosion control), as well as new factors of importance to the modified disposal plans and revised conceptual model. The outcome of the NRC staff's 2005 review was a list of 8 Key Monitoring Factors with which NRC tracks DOE's SDF disposal actions and development of supporting information. In the current review, staff has taken a similar approach to identify Key Monitoring Factors, in that staff has focused on risk-significant issues. However, based on staff's monitoring experience since the previous review, staff determined that making individual Key Monitoring Factors more specific (i.e., smaller in scope), though leading to a slightly larger number of individual Key Monitoring Factors, would facilitate monitoring. Following completion of this TER, the staff will revise its

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[1] §61.41 states *"Concentrations of radioactive material which may be released to the general environment in ground water, surface water, air, soil, plants, or animals must not result in an annual dose exceeding an equivalent of 25 millirems to the whole body, 75 millirems to the thyroid, and 25 millirems to any other organ of any member of the public."* NRC has evaluated compliance using a dose limit of 0.25 mSv/yr (25 mrem/yr) Total Effective Dose Equivalent (TEDE) consistent with the approach discussed in final rule (66 FR 55752) "Because each of the organs had the same limit under the older system even though each had a different level of radiosensitivity, it is very difficult to directly compare the old standards with the new standards. As noted in the proposed rule, the Commission considers 0.25 mSv/yr (25 mrem/yr) TEDE as the appropriate dose limit to compare with the range of potential doses represented by the older limits that had whole body dose limits of 0.25 mSv/yr (25 mrem/yr)." The DOE performance assessment and NRC's review have used the most updated dose factors consistent with Commission direction in SRM-SECY-01-0148 to calculate the potential dose.

[2] NUREG-1854 also indicates that assessments beyond 10,000 years may be necessary in some cases to demonstrate that the disposal of certain types of waste does not result in high impacts to future generations.

monitoring plan for the SDF. Appendix A of this TER lists the new Key Monitoring Factors and provides details about each.

Table 1 below presents the NRC staff's key review results related to the 10 CFR 61 performance objectives.

Table 1: Key Review Results from the NRC's Technical Evaluation of the 2009 PA

Performance Objective	Key Results
§61.41 Protection of the General Population from Releases of Radioactivity	• **Based on the results of the NRC's review, the NRC staff does not have reasonable assurance that DOE's disposal actions at the SDF meet this performance objective.** • The NRC staff does not find Case A to be an appropriate compliance case because it does not accurately reflect current site conditions, does not account for the full ranges of measured values of key parameters or expected differences between laboratory and as-emplaced values, and does not appropriately account for potential changes in parameter values with time. • In response to NRC concerns about DOE's Case A, DOE supplied Cases K, K1, and K2. Contrary to DOE's characterization of these new cases as overly-pessimistic sensitivity analyses, the NRC staff believes these cases are based on a combination of both overly-optimistic and apparently conservative assumptions. NRC staff has relied heavily on Case K1 in its review and conclusions because it resolves many of the concerns NRC staff has about using Case A as a compliance case. • The NRC staff finds that the timing of DOE's predicted peak dose to an off-site general member of the public in Case K1 (approximately 0.9 mSv/yr [90 mrem/yr] at 12,900 years after site closure) is sensitive to, and delayed by assumptions about, saltstone fracture growth, use of an average K_d value to simulate release of Tc-99, and seemingly overly-optimistic assumptions about Tc retention in disposal unit concrete. • Because of the large uncertainty in the predicted timing of the 0.9 mSv/yr (90 mrem/yr) peak dose from Tc-99 and the expectation that the predicted peak in DOE's Case K1 model was delayed by unsupported assumptions, the NRC staff could not conclude it had reasonable assurance that the dose would meet the 0.25 mSv/yr (25 mrem/yr) limit for 10,000 years after site closure based on DOE's Case K1 results. • The NRC staff performed additional analyses with DOE's PA model of source-term release to determine whether it could reduce the uncertainty in the timing or magnitude of the Case K1 peak dose. These additional analyses, however, still led to predicted Tc-99 peak doses that exceeded the performance objective dose limit of 0.25 mSv/yr (25 mrem/yr). • An additional attempt by NRC staff to develop a simplified release model to avoid an artifact of DOE's Case K1 Tc release model led to peak fractional release rates into the environment similar to the DOE's Case K1 fractional release rate.

Performance Objective	Key Results
§61.42 Protection of Individuals from Inadvertent Intrusion	• **Based on the results of the NRC's review, the NRC staff has reasonable assurance that DOE's disposal actions at the SDF meet this performance objective.** • The NRC staff finds that the scenarios and pathways analyzed by DOE for the assessment of this performance objective are appropriate based on the regional practices near SRS. • The NRC staff finds DOE's approach in its sensitivity analysis to evaluate the dose to an individual who drills into a disposal unit and spreads drill cuttings on the land surface to be reasonable. • The NRC staff concludes that the Case K1 chronic intruder dose predicted by DOE is likely to be conservative (i.e., overestimate the potential intruder dose). The NRC staff therefore concludes that the dose to an inadvertent intruder through the groundwater pathways is likely to be less than 5 mSv/yr (500 mrem/yr), provided that key assumptions in Case K1 analysis are true. • The results of the reviewed sensitivity cases indicate that the additional dose an intruder would receive from being exposed to drill cuttings containing saltstone would not be significant compared to the dose to an intruder from using groundwater on site, which the NRC staff expects to meet the 5 mSv/yr (500 mrem/yr) dose limit used in promulgation of §61.42. • The NRC staff expects the dose to an intruder who drills directly into saltstone from the drill cuttings to meet the performance objective for protection of an inadvertent intruder.
§61.43 Protection of Individuals during Operations	• **Based on the results of the NRC's review in its 2005 TER, onsite observations conducted in October 2007 and March 2008, and NRC's review of the annual SRS Environmental Reports, the NRC staff has reasonable assurance that DOE's disposal actions at the SDF meet this performance objective.**

Performance Objective	Key Results
§61.44 Stability of the Disposal Site after Closure	• **Based on the uncertainty in certain risks associated with this performance objective, aspects of site stability will be included in NRC's revised monitoring plan. However, based on the results of the NRC's review, the NRC staff has reasonable assurance that DOE's disposal actions at the SDF meet this performance objective.** • The NRC staff concludes the saltstone waste form will provide a monolithic structure, minimize void space, and prevent collapse and differential settlement that could occur due to consolidation of the waste. • NRC staff evaluated the dynamic settlement that would result from an earthquake with a 10,000-year return period and found it was unlikely to cause significant disruption to the SDF. • NRC staff determined that floods are unlikely to disrupt the SDF because the 10,000-year flood level for the Upper Three Runs basin near the SDF is significantly below the lowest planned elevation of a disposal unit at the SDF. • Much of SRS, including the SDF, is underlain by calcareous sediment in the Santee formation resulting in the presence of "soft zones." Historically, some of these zones have consolidated, resulting in depressions on the land surface. DOE concluded that consolidation of a soft zone would have minimal effects on the stability of a disposal unit at the SDF. • Sinks identified elsewhere at SRS are comparable in size to a four-pack of Future Disposal Cells (FDCs). If a sink developed under a four-pack, local infiltration could increase and disposal units could fracture. However, because the history of sink development at the SRS is unclear and the potential development of soft zones under the SDF also is uncertain, the probability a sink would develop at the SDF within 10,000 years of closure is uncertain. • Recent studies predict greater static settlement in the SDF than addressed by DOE in the PA. • In general, the NRC staff found more uncertainty in the potential effects of static settlement due to loading of the subsurface layers and settlement due to calcareous zones present or potentially developing under the SDF than that resulting from the potential effects of earthquakes, floods, and erosion at the SDF.

The following section provides a high-level overview of the staff's findings as described in the TER. Although the section offers helpful details of the staff's conclusions, for a broader and more detailed discussion of the 2009 PA and NRC staff's evaluation, please see the appropriate sections of the TER.

High-level Overview of NRC Conclusions on the 2009 PA

In 2005, the NRC reviewed the analyses the DOE performed to support its Waste Determination for the SDF. The main NRC findings for long-term performance in 2005 were that to contain the major risk driver, Tc, over the long-term, (1) the as-placed waste form would need to chemically reduce the Tc sufficiently to make it immobile, and (2) the waste form and other engineered features would need to limit water flow into the waste form. Water flow into the facility could bring oxygen into the waste form and oxidize the Tc, allowing release. In addition, the water would provide transport out of the waste form for any mobilized radionuclides, including Tc. After the waste determination was issued in 2006, DOE gathered additional information regarding their assumptions for long-term performance while producing saltstone operationally. After short-term performance issues with the existing vault design, DOE redesigned the vaults and referred to the new disposal units as “future disposal cells.” DOE developed revised analyses for the existing vaults and future disposal cells to address the new designs and the results of new experiments, as well as performance issues (such as cracking in the walls of Vault 4). DOE provided this revised PA to NRC in November 2009.

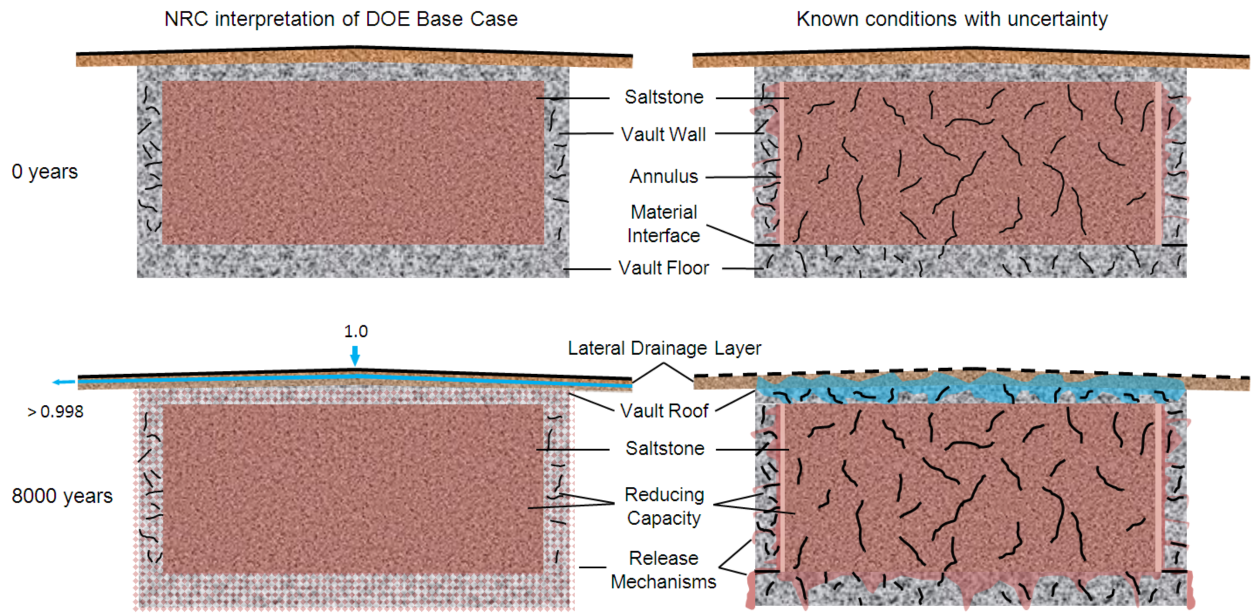

Figure 1: NRC Interpretation of DOE Base Case Conceptual Model

DOE’s revised PA evaluates a base case (alternately, called Case A), which DOE indicates reflects its expectation of long-term performance. In this case, the upper cover is assumed to degrade with time, but there is no degradation of the saltstone and limited degradation of the disposal units assumed over a 20,000 year evaluation period (Figure 1). The walls in Vaults 1 and 4 are assumed to be initially cracked, while the new future disposal cells walls, like the floor and roof experience only minimal degradation. Due to a combination of the material properties assumed and the lack of roof degradation, Case A continues to shed over 99% of the water around the disposal units from 400 to 10,000 years after site closure. This significant limitation on modeled water contact with the waste limits the modeled transport of radionuclides out of the waste. In addition, because oxidation from gas-phase transport of oxygen is not included in DOE’s base case, the flow restriction significantly limits the modeled oxidation of saltstone, further limiting Tc mobilization and release.

DOE also supplied a number of alternate assessments, which evaluated the impacts of increased degradation of one or more barriers (as compared to the base case). For example, in Cases B and C, DOE evaluates the impact of fast-flow paths, such as gaps between the saltstone and disposal unit walls. However, in almost all of the alternate assessments provided with the PA, the roof remained largely intact so the shedding of over 99% of the water occurred, which made the cases less useful for evaluating importance of the barriers. In one case, the Synergistic Case, DOE evaluated multiple degradation mechanisms, including roof degradation.

Although more water flows to the disposal unit in the synergistic case (as compared to the base case), because the saltstone matrix is assumed to have a very low hydraulic conductivity, most of the flow is diverted through the disposal unit walls (for the FDCs) or saltstone fractures (for Vault 4). Although the significant hydraulic degradation of the disposal unit walls appears to be pessimistic, intermediate model results show that the mathematical model does not reflect DOE's stated purpose for the Synergistic Case because the wall degradation lowers the predicted dose. Because almost all of the infiltrating water bypasses most of the saltstone inventory in the cases DOE supplied with the PA, these cases do not appear to realistically predict potential doses from saltstone. In addition to these deterministic cases, DOE also performed probabilistic analyses. However, because of concerns about the design and implementation of the probabilistic model, the NRC staff did not rely on the probabilistic model in its compliance evaluation.

NRC's evaluation focused largely[3] on the technical factors related to the oxidization and mobilization of Tc. The staff questioned support for several of DOE's assumptions in its base case analysis, including (1) the lack of saltstone fractures, given that cracking of saltstone already has been observed; (2) the performance provided by the roof and lower drainage layer in shedding over 99% of the water around the disposal units throughout a 10,000 year performance period; and (3) the basis for a number of parameters (e.g., hydraulic conductivity, Tc sorption coefficients), because recent research does not support DOE assumptions. The NRC sent two rounds of questions, called request for additional information. In the second round, NRC requested a revised base case to address assumptions in DOE's base case that the NRC staff had concluded were unrealistically optimistic or inadequately supported. For example, the NRC staff requested that the revised base case represent degradation of the engineered disposal units and saltstone over time (e.g., cracks developing which allows additional water to oxidize saltstone and release the mobilized Tc).

DOE responded to this request with Case K[4], which addresses a number of NRC concerns. The roof and disposal unit walls degrade over time, developing cracks and increased water flow. The saltstone develops a number of through-going cracks, which results in the saltstone becoming oxidized sooner than in earlier DOE cases. The fractures are not represented explicitly in the flow model, but water transport through the system is greater than DOE assumes in the base case because of greater assumed hydraulic conductivity in the saltstone and disposal unit concrete. Although more water flows through the disposal units and saltstone in Case K, DOE assumes that the disposal unit concrete acts as a significant barrier to Tc release. Because DOE changed the way Tc sorption was modeled from a discrete-fracture model to a model based on an average sorption coefficient, Tc does not leave the system

[3] A series of questions focused on Ra because it was a risk driver in Case A and several of the alternate analyses DOE submitted with the PA. Since that time, DOE has revised its projected inventory of Ra-226 and its ancestors, Th-230 and U-234, such that DOE no longer predicts radium to be a risk-driver.

[4] DOE provided the NRC three cases related to Case K: Case K, Case K1, and Case K2. The only differences between these three cases are the K_d values used to represent Tc sorption in oxidizing and reducing cementitious materials (saltstone and disposal unit concrete). When this distinction is not important (e.g., when discussing hydraulic properties, which are the same in all three cases), the NRC staff uses the term "Case K" to refer to all three cases. The NRC staff differentiates between Cases K, K1, and K2 when discussing values specific to Tc-99 (e.g., Tc-99 Kd values, release rates, or doses).

through oxidized pathways. Instead, in the Case K model, intermediate model outputs show Tc is retained in the disposal unit floor for thousands of years until it nears complete oxidation. Thus, in the Case K model, the disposal unit floor captures nearly all of the Tc, like a filter. For example, in the FDCs, this filtering effect leads to the disposal unit floor having a modeled Tc concentration approximately 13 times greater than the original concentration in the saltstone waste itself. The disposal unit floors also are modeled as releasing the Tc much more gradually than the saltstone does, which lowers the predicted peak dose. Like the Synergistic Case, the Case K mathematical model does not seem to represent DOE's stated purpose for the case. Specifically, although the description of the case indicates increased degradation of the disposal unit floor (compared to DOE's base case), the change in the mathematical model for Tc retention in the floor appears to have caused an unintended significant improvement in floor performance. There is no support provided for the modeled behavior of the disposal unit floor. In all likelihood, the disposal unit floor will develop flow pathways through cracks and joints. The NRC staff expects these pathways will oxidize quickly and will not significantly retain Tc.

Because of DOE assumptions about (1) the way cracks develop over time (i.e., most of the cracks develop after 8,000 years) and (2) Tc retention in the disposal unit floor, predicted peak doses are delayed by several thousand years to approximately 12,000 - 14,000 years. DOE calculated the peak doses to be between 0.5 mSv/yr (50 mrem/yr) and 0.9 mSv/yr (90 mrem/yr). The NRC staff concludes that the information supporting the delay of the peak by these two assumptions is weak and, therefore, there is not reasonable assurance that these peak doses will occur after 10,000 years. That is, with more supportable assumptions, it is likely the peak dose will be predicted to occur before 10,000 years. However, the NRC staff noted that certain modeling assumptions in Case K (e.g., the suddenness of fracturing and the suddenness of release in the Tc release model DOE used) appeared to be pessimistic (i.e., to over estimate dose). The NRC staff performed independent analyses to evaluate whether these assumptions, when combined with DOE's more optimistic assumptions (i.e., the superior chemical performance of the disposal unit floors), still led to an over estimate the predicted dose. The NRC staff concluded that, based on current information, the predicted dose is between approximately 0.25 mSv/yr (25 mrem/yr) and approximately 1 mSv/yr (100 mrem/yr).

DOE has indicated that it continues to believe that the limited degradation in Case A, their original base case, appropriately models the future behavior of the system. DOE also indicated it believes Case K is extremely pessimistic. NRC disagrees with this characterization and concludes that, with certain exceptions (e.g., the chemical performance of the disposal unit floors), Case K provides a more realistic estimate of the future behavior of the system than DOE's selected base case.

Conclusions in this TER are based on the NRC staff's review of the 2009 PA dated October 29, 2009, DOE responses to the NRC staff's requests for additional information, supporting references, and information provided during publicly-documented meetings and teleconferences between DOE and NRC, as well as NRC staff analyses as documented in this TER. If, in the future, DOE determines it is necessary to revise the assumptions, analysis, design, or waste management approach related to any aspects of the disposal strategy at the SDF in a way that may affect DOE's compliance with the 10 CFR 61 performance objectives, DOE should provide details of these changes to the NRC once approved by the sites regulatory body (South Carolina Department of Health and Environmental Control). This NRC staff assessment is not a precedent for any future decisions regarding non-HLW or incidental waste determinations at SRS or other DOE sites.

1. Introduction

1.1 Monitoring Background

In October 2004, the National Defense Authorization Act for Fiscal Year 2005 (NDAA) was signed into law. Section 3116 of the NDAA requires that (a) U.S. Department of Energy (DOE) consult with U.S. Nuclear Regulatory Commission (NRC) on its non-High-Level Waste (HLW) determinations[5] and, once a determination is complete, that (b) NRC monitor DOE's disposal actions to assess compliance with 10 CFR Part 61, Subpart C, low-level waste disposal facility performance objectives[6]. As stated in Section 3116(a), for waste to be determined to be incidental to reprocessing rather than HLW, DOE's disposal actions must meet the following criteria:

(1) The waste does not require permanent isolation in a deep geologic repository for spent fuel or HLW;
(2) The waste has had highly radioactive radionuclides removed to the maximum extent practical; and
(3) (A) does not exceed concentration limits for Class C low-level waste as set out in §61.55, and will be disposed of—
    (i) in compliance with the performance objectives set out in subpart C of 10 CFR Part 61; and
    (ii) pursuant to a State-approved closure plan or State-issued permit, authority for the approval or issuance of which is conferred on the State outside of this section; or

    (B) exceeds concentration limits for Class C low-level waste as set out in §61.55, but will be disposed of–

    (i) in compliance with the performance objectives set out in subpart C of 10 CFR Part 61;
    (ii) pursuant to a State-approved closure plan or State-issued permit, authority for the approval or issuance of which is conferred on the State outside of this section; and
    (iii) pursuant to plans developed by the Secretary in consultation with the Commission

Following DOE's completion of the basis documents for disposal, in consultation with the NRC, DOE begins disposal activities and NRC moves into the role of monitoring these disposal actions, per Section 3116(b). As monitor, NRC, in coordination with the covered State, is required to assess DOE's compliance with subparagraphs (A) and (B) of subsection (a)(3) listed above.

[5] A non-HLW determination (or waste determination [WD]) is a technical analysis that DOE uses to evaluate whether waste is incidental to reprocessing, or alternatively, is HLW, based on the risks it poses. A WD documents whether DOE's proposed disposal action will meet the applicable incidental waste criteria. A WD often is supplemented by a performance assessment (PA).

[6] Section 3116 of the NDAA is applicable only to South Carolina and Idaho and does not apply to waste transported out of those States.

As stated above, as monitor, NRC assesses compliance with 10 CFR 61, Subpart C, low-level waste disposal facility performance objectives. The objectives are as follows:

> **§61.40: General requirement -** Land disposal facilities must be sited, designed, operated, closed, and controlled after closure so that reasonable assurance exists that exposures to humans are within the limits established in the performance objectives in §61.41 through §61.44.
>
> **§61.41: Protection of the general population from releases of radioactivity -** Concentrations of radioactive material which may be released to the general environment in ground water, surface water, air, soil, plants, or animals must not result in an annual dose exceeding an equivalent of 25 millirems to the whole body, 75 millirems to the thyroid, and 25 millirems to any other organ of any member of the public. Reasonable effort should be made to maintain releases of radioactivity in effluents to the general environment as low as is reasonably achievable.
>
> **§61.42: Protection of individuals from inadvertent intrusion -** Design, operation, and closure of the land disposal facility must ensure protection of any individual inadvertently intruding into the disposal site and occupying the site or contacting the waste at any time after active institutional controls over the disposal site are removed.
>
> **§61.43: Protection of individuals during operations -** Operations at the land disposal facility must be conducted in compliance with the standards for radiation protection set out in part 20 of this chapter, except for releases of radioactivity in effluents from the land disposal facility, which shall be governed by §61.41 of this part. Every reasonable effort shall be made to maintain radiation exposures as low as is reasonably achievable.
>
> **§61.44: Stability of the disposal site after closure -** The disposal facility must be sited, designed, used, operated, and closed to achieve long-term stability of the disposal site and to eliminate to the extent practicable the need for ongoing active maintenance of the disposal site following closure so that only surveillance, monitoring, or minor custodial care are required.

In March 2005, DOE submitted its draft basis document for the Saltstone facility, initiating NRC's review under Section 3116(a) of the NDAA. In December 2005, following multiple interactions between the agencies, the NRC published its review of the draft basis document for the Saltstone facility in a technical evaluation report (NRC, 2005).

The NRC's 2005 review concluded with a finding of reasonable assurance for each of the Section 3116(a) criteria, with conditions (Table 1.1-1).

Table 1.1-1: Conditions of NRC's 2005 Findings Regarding the NDAA Section 3116(a) Criteria

Criterion	2005 Conclusions
One	Given the conclusion in the 2005 TER that the NRC staff has reasonable assurance that salt waste disposal at the Saltstone Disposal Facility (SDF) meets the performance objectives of 10 CFR 61, Subpart C, based on certain assumptions, and the lack of other considerations that necessitate disposal of the salt waste in a geologic repository, the NRC staff concluded it had reasonable assurance that salt waste meets Criterion One.
Two	DOE's conclusion that highly radioactive radionuclides would be removed to the extent practical by the proposed two-phase, three-part process, including the proposed process for the management of Tank 48 waste, was found to be reasonable.
Three	The NRC concluded it had reasonable assurance that salt waste disposal at the SDF would meet the performance objectives of 10 CFR Part 61, Subpart C, with the following provisions: *§61.41*: The assumptions [*as stated in Section 4.3 of the NRC's 2005 TER (NRC, 2005)*] relevant to the performance objective are verified during monitoring. *§61.42*: A design for long-term erosion control is implemented, which greatly reduces the likelihood of an agricultural intruder scenario occurring. *§61.43*: During operations, individuals are protected by DOE regulations, which were demonstrated to provide protection comparable to 10 CFR Part 20. In addition, a number of measures are applied to ensure that exposure of individuals are maintained as low as reasonably achievable including: (1) a documented radiation protection program, (2) a Documented Safety Analysis, (3) design of the Saltstone Production Facility (SPF) and SDF, (4) regulatory and contractual enforcement mechanisms, and (5) access controls, training, and dosimetry. *§61.44*: None.

One of the main outcomes of the 2005 review (NRC, 2005) was the development of monitoring factors, for use in NRCs monitoring role under NDAA Section 3116(b). In general, verification of DOE's assumptions made in assessing whether the 10 CFR 61 performance objectives can be met should be performed by DOE. However, because some of the assumptions made in the analysis, if incorrect, could lead to noncompliance with the performance objectives, NRC has monitored these assumptions as part of its responsibilities under the NDAA. These assumptions fall into the following general groups: waste form and vault degradation, the effectiveness of infiltration and erosion controls, and estimation of the radiological inventory. The NRC staff concluded that certain factors are important to assessing whether DOE's disposal actions will be compliant with the performance objectives. Based on the review described in the 2005 TER (NRC, 2005), the NRC staff crafted the following eight "key" factors critical to NRC monitoring of the Saltstone facility.

Table 1.1-2: Key Factors Resulting from NRC Review of 2005 Saltstone PA

Key Factor	Details
1. Oxidation of Saltstone	The rate of waste oxidation and release of technetium from an oxidized layer of saltstone will be a key determinant of the future performance of the SDF and therefore whether §61.41 can be met. More realistic modeling will be important to achieving the performance objectives, and adequate model support is essential to providing the technical basis for the model results. It will be important to ensure that gas phase transport of oxygen through fractures will not significantly increase oxidation of technetium in the saltstone.
2. Hydraulic Isolation of Saltstone	The extent of degradation that may influence the hydraulic isolation capabilities of the saltstone and vaults will be a key factor in assessing whether the SDF can meet §61.41. Degradation mechanisms that may result in the hydraulic conductivity of degraded saltstone and vault concrete being larger than $1x10^{-7}$ cm/s ($1x10^{-1}$ ft/yr) need to be evaluated with multiple sources of information (e.g., modeling, analogs, experiments [especially field scale and long-term], expert elicitation) to ensure that they are unlikely to occur. It will be important to ensure that field-scale physical properties (e.g., hydraulic conductivity, effective diffusivity) of as-emplaced saltstone are not significantly different from the results of laboratory tests of smaller-scale samples performed to date. It will be important to perform additional laboratory measurements of hydraulic conductivity because the data being relied upon represent limited samples that had a small range of curing times. In addition, because there was a fairly significant amount of variability in the Toxicity Characteristic Leaching Procedure (TCLP) test results, if DOE deviates significantly from the nominal saltstone composition, DOE should perform additional tests for hydraulic conductivity and effective diffusivity that justify the parameter values used over the range of compositions.
3. Model Support	Adequate model support is essential to assessing whether the SDF can meet §61.41. The model support for: (1) moisture flow through fractures in the concrete and saltstone located in the vadose zone, (2) realistic modeling of waste oxidation and release of technetium, (3) the extent and frequency of fractures in saltstone and vaults that will form over time, (4) the plugging rate of the lower drainage layer of the engineered cap, and (5) the long-term performance of the engineering cap as an infiltration barrier is key to confirming PA results.

Key Factor	Details
4. Erosion Control Design	The erosion control design is important to ensuring that §61.42 can be met because it eliminates pathways and scenarios for intruder dose assessments. Implementation of an adequate design that (1) does not deviate significantly from the information submitted to the NRC in CBU-PIT-2005-00203 Rev 1 and the associated references, or, (2) if it does deviate significantly, is reviewed by NRC staff to ensure that the revisions are consistent with long-term erosion control design principles is important.
5. Infiltration Barrier Performance	The infiltration control design is important to ensuring that §61.41 can be met because the release of contaminants to the groundwater is predicted to be sensitive to the large reduction in infiltration provided by the infiltration control. It is important to ensure that the design can be implemented and will perform as designed.
6. Feed Tank Sampling	Implementation of an adequate sampling plan is important to ensuring that §61.41 and §61.42 can be met. It is important to assess results of future sampling and confirm that current projections of the concentrations of highly radioactive radionuclides in treated salt waste (or grout) are greater than or equal to actual concentrations of highly radioactive radionuclides in treated salt waste (or grout).
7. Tank 48 Waste form	To ensure that Tank 48 waste can be safely managed, future tests of the physical properties of samples that contain organic materials similar to Tank 48 waste will need to confirm that the properties of the waste form made from this waste will provide for suitable waste form performance such that the disposal system will be able to meet the performance objectives. The technical basis should, at a minimum, include tests for hydraulic conductivity and effective diffusivity.
8. Removal Efficiencies	Predicted removal efficiencies of highly radioactive radionuclides by each of the planned salt waste treatment processes are a key factor in determining the radiological inventory disposed of in saltstone. The inventory, in turn, is an important factor in the determination that §61.41 and §61.42 can be met.

Following the January 2006 completion of the DOE's final waste determination for salt waste disposal (DOE-WD-2005-001), the NRC developed a monitoring plan (NRC, 2007a). The plan describes activities designed to monitor DOE's disposal activities as they relate to the eight key factors identified as part of NRC's PA review (Table 1.1-2) as well as other relevant activities that were identified early in the monitoring process (e.g., review of environmental monitoring data and worker dose records).

The NRC's monitoring plan, finalized in May 2007 (NRC, 2007a), describes two primary types of monitoring activities the NRC staff performs: (1) technical reviews of DOE data and analyses and (2) onsite visits to observe DOE's disposal actions. With regard to the role of technical reviews under monitoring, DOE's disposal activities at the site are often complex processes, which rely on technical documents as basis for disposal. Should any changes or revisions be made to key technical documents (e.g., to describe use of new technologies or methods, or to reflect changes requested by SC DHEC in its regulatory role or by NRC in its monitoring role), assumptions are reviewed again to confirm continued compliance with the performance objectives. Since the Saltstone PA is a critical element of DOE demonstration of compliance with the performance objectives, any updates to the Saltstone PA must also be assessed under NRC's monitoring role. In coordination with the covered State, NRC's onsite observation program involves trips to the site to observe and review certain operations that might affect compliance with the performance objectives as they are being performed. The NRC staff observes operational details, evaluates the implementation and purpose, and reviews documentation helpful to assessing the impact the process has on compliance.

Section 3.1.9 of the NRC monitoring plan for Saltstone, *Performance Assessment Process Review*, provides an explanation of how the NRC staff evaluates revisions made to the PA (NRC, 2007a). In brief, the staff uses an approach similar to that used in the original technical evaluation of the waste disposal actions (NRC, 2005). Where practical, the NRC staff performs its own independent assessments of the more risk-significant aspects of the performance assessment. Sensitivity and uncertainty analyses can be especially useful in determining the effects and importance of changes to the PA models. When a deterministic analysis is used for a PA, the NRC staff evaluates whether the analysis provides a demonstrably conservative estimate of potential doses, or DOE has provided appropriate support and bases for key assumptions or parameters in the PA. Determining the important features of a deterministic analysis may be difficult, which is one reason why it may be advantageous for the staff to undertake its own independent assessment. If a probabilistic analysis is used, the staff must ensure that appropriate parameter distributions are used and that the analyses appropriately account for parameter uncertainty and possible parameter correlation.

As of this report, NRC has conducted 11 onsite observations since inception of monitoring in 2006. Each observation covers a range of topics applicable to one or more performance objectives (Table 1.1-3). In addition, NRC has conducted numerous technical reviews topics covered in the monitoring plan.

On occasion, an issue of relatively high risk significance arises during monitoring activities that could affect compliance. These *Open Issues* require additional follow-up by the NRC staff or additional information from DOE to address questions that the NRC staff has raised regarding DOE disposal actions. Issues such as these retain the title of *Open* until adequately addressed by DOE.

Since the first onsite observation, a total of four Open Issues have been documented. Of these, one Open Issue has been closed (Open Issue 2007-3), and three remain open (Table 1.1-4). NRC monitoring of the SDF has continued during the development of this document. NRC staff has conducted five onsite observations and has participated in multiple meetings regarding DOE disposal activities since receiving the 2009 PA.

Table 1.1-3: History of Onsite Observation Activities

Date	Discussion Topics	Open Issue Activity
October 2007	Grout Formulation and Placement Vault Construction Waste Sampling Radiation Protection Program	2007-1 Opened 2007-2 Opened 2007-3 Opened
March 2008	Saltstone Characterization Vault Operation and Characterization Waste Sampling and Inventory Radiation Protection Program	2007-3 Closed
July 2008	Saltstone Characterization Waste Sampling NRC Staff Technical Reviews	
March 2009	Disposal Cell Construction Radionuclide Inventory NRC Staff Technical Reviews	2009-1 Opened
June 2009	Disposal Cell Construction	
August 2009	Performance Assessment Process Review	
February 2010	Disposal Cell Construction SPF Operation PA Process Review Radionuclide Inventory	
April 2010	Disposal Cell Construction	
July 2010	Saltstone Quality Assurance Plan Hydro-test results on Cell 2A and 2B Saltstone Core Samples May 19, 2010 Saltstone Inadvertent Transfer	
January 2011	Vault 4 Integrity SPF Operations	
April 2011	Radionuclide Inventory Waste Oxidation and Tc Release Research Disposal Cell Construction Topics from Previous Observations	

Table 1.1-4: Saltstone Disposal Facility Open Issues

Number	Brief Description of Issue	Identified
2007-1 (Open)[1]	DOE should determine the hydraulic and chemical properties of as-emplaced saltstone grout.	10/2007
2007-2 (Open)[1]	DOE should demonstrate that intrabatch variability, flush water additions to freshly poured saltstone grout at the end of each production run, and additives used to ensure processability are not adversely affecting the hydraulic and chemical properties of the final saltstone grout.	10/2007
2007-3 (Closed)	DOE should reassess the risk significance of the as-built conditions of Vault 4 in light of the presence of contaminated seeps on the exterior wall of Vault 4.	10/2007
2009-1 (Open)[1]	DOE should demonstrate that (1) Tc-99 in salt waste is strongly retained in saltstone grout and (2) the sorption of dissolved Tc-99 onto saltstone grout and vault concrete is consistent with the K_d values for Tc-99 assumed in the PA.	03/2009

[1] Issues remaining Open prior to review of the 2009 PA will be incorporated into the NRC staff's revision to its monitoring plan for the SDF. These actions will be closed upon issuance of this TER.

1.2 Current Review

In 2008, DOE decided to revise the PA for the SDF to reflect a new vault design, new vault and grout testing information, emergent facility conditions, and areas of uncertainty identified by NRC staff during its review of the 2005 Saltstone PA (NRC, 2008b). In May 2008, DOE hosted an interagency meeting between DOE, SC DHEC, and NRC in which DOE provided an overview of the revised design and received comments from both NRC and SC DHEC (NRC, 2008b). Following this meeting, once approved by SC DHEC, DOE began construction of the new disposal vaults (referred to as Future Disposal Cells [FDCs]).

In November 2009, DOE submitted a new PA[7] for the SDF (SRR-CWDA-2009-00017) to the NRC for review in the NRC's capacity as a monitor of DOE's disposal actions at the SDF in accordance with Section 3116(b) of the NDAA. In accordance with these responsibilities, the NRC staff began review of the 2009 Saltstone PA in November 2009. The purpose of the review is to assess whether DOE's disposal actions, as described in the PA, meet the performance objectives of 10 CFR 61, Subpart C. The review included two Requests for Additional Information (RAI). The RAI process under the NDAA typically involves the NRC preparing an RAI within 90 days of receiving the PA and DOE responding to that RAI within 90 days. The NRC sent the first RAI (RAI-2009-01) on March 31, 2010 (NRC, 2010b), to which DOE responded on July 22, 2010 (SRR-CWDA-2010-00033). After review of DOE's responses, the NRC staff concluded that some of these responses did not fully address the NRC's

[7] Throughout this TER, the staff references the new PA by either its DOE document number (SRR-CWDA-2009-00017), "the PA," or "the 2009 PA."

questions and concerns with assumptions in DOE's expected degradation case (or base case). The NRC sent a second RAI (RAI-2009-02) on December 15, 2010 (NRC, 2010i), to which DOE responded in draft form (SRR-CWDA-2011-00044 Rev. 0) and discussed at a public meeting on April 26, 2011, and then submitted a final version on August 26, 2011, (SRR-CWDA-2011-00044 Rev. 1). To respond to the NRC's concerns with DOE's base case, DOE developed a new case called Case K.

The NRC and DOE staffs met frequently throughout the review process, and held additional meetings to discuss the NRC's concerns and the details of the new Case K. The meetings in which the NRC staff conducted and/or took part are listed below in Table 1.2-1. All of the summaries of these meetings are available to the public and can be found via the associated Agencywide Documents Access and Management System (ADAMS) Accession numbers in the third column. With the exception of the technical exchange meetings, all of the meetings listed in Table 1.2-1 were open to members of the public.

In a letter dated May 20, 2011, following DOE's decision to respond to the NRC's concerns by developing the new Case K, DOE requested NRC acceptance of the new approach. On July 20, 2011, in response to this letter, the NRC stated that final conclusions on the new approach could only be reached after NRC staff completes its detailed review of DOE's written responses to RAI-2009-02 (NRC, 2011i).

Although Case K resolves many of the NRC staff's concerns with DOE's selected base case, the NRC staff identified both overly-optimistic and potentially overly-pessimistic assumptions in this new case. The impact of these assumptions was not immediately clear because the Case K analysis supplied by DOE did not include an uncertainty or sensitivity analysis that addressed these assumptions (NRC, 2011g). In addition, the staff noted unexpected intermediate results in the Case K computational model files. To better understand the uncertainty in the Case K results, the effects of conflicting optimistic and pessimistic assumptions, and the causes of the unexpected intermediate results, the staff performed its own confirmatory analyses related to Case K.

Table 1.2-1: NRC/DOE Meetings during PA Review

Meeting Date	Meeting Purpose	Meeting Summary (ADAMS Acc.)
December 8, 2009	Commencement Meeting for NRC Review of Revised PA	ML100050554
April 19, 2010	Clarification Discussion Following DOE Receipt of NRC's RAI	ML101440307
September 2, 2010	Clarification Discussion Following NRC Receipt of DOE's RAI Response	ML102980289
January 27, 2011	Detailed Discussion Following DOE Receipt of NRC's Second RAI	ML110341424

Meeting Date	Meeting Purpose	Meeting Summary (ADAMS Acc.)
April 27, 2011	Discussion Regarding DOE Potential Responses to Second RAI	ML111950042
May 3, 2011	Technical Exchange #1 – NRC/DOE Staff-Level Discussion Concerning Base Case [1]	ML111440310
May 5, 2011	Technical Exchange #2 – NRC/DOE Staff-Level Discussion Concerning Base Case [1]	ML111440270
May 12, 2011	Technical Exchange #3 – NRC/DOE Staff-Level Discussion Concerning Base Case [1]	ML111440857
May 17, 2011	Technical Exchange #4 – NRC/DOE Staff-Level Discussion Concerning Base Case [1]	ML111440864
May 19, 2011	Technical Exchange #5 – NRC/DOE Staff-Level Discussion Concerning Base Case [1]	ML111440873
June 2, 2011	Culmination of Technical Exchanges and Discussion of Path Forward	ML111780433

[1] A technical exchange is a staff-level, non-public discussion involving no decision-making. Summaries from technical exchanges are available to the public.

1.3 Site Overview

The 780 square kilometer (300 square mile) DOE SRS, located in south-central South Carolina, began operation in 1951 producing nuclear material for national defense, research, medical, and space programs. Waste produced at the site from spent nuclear fuel reprocessing for defense purposes has been commingled with non-reprocessing waste resulting from the production of targets for nuclear weapons and production of material for space missions. Significant quantities of radioactive waste are currently stored on-site in large underground waste storage tanks, which were placed into operation between 1954 and 1986. Of the original 51 storage tanks, 49 are still operational. The waste stored in the tanks at SRS is a mixture of insoluble metal hydroxide solids, referred to as sludge, and soluble salt supernate. The supernate volume has been reduced by evaporation, which also concentrates the soluble salts to their solubility limits. The resultant solution crystallizes as salts, and the resulting solid is referred to as saltcake. The saltcake and supernate combined are referred to as salt waste. DOE removes the salt waste, treats it to remove highly radioactive radionuclides to the maximum extent practical, and disposes the low activity fraction on site in the SDF.

The SDF is located in the Z-area of the SRS approximately 10 km (6.2 mi) from the nearest SRS site boundary on a well-drained local topographic high (Figure 1.3-1).

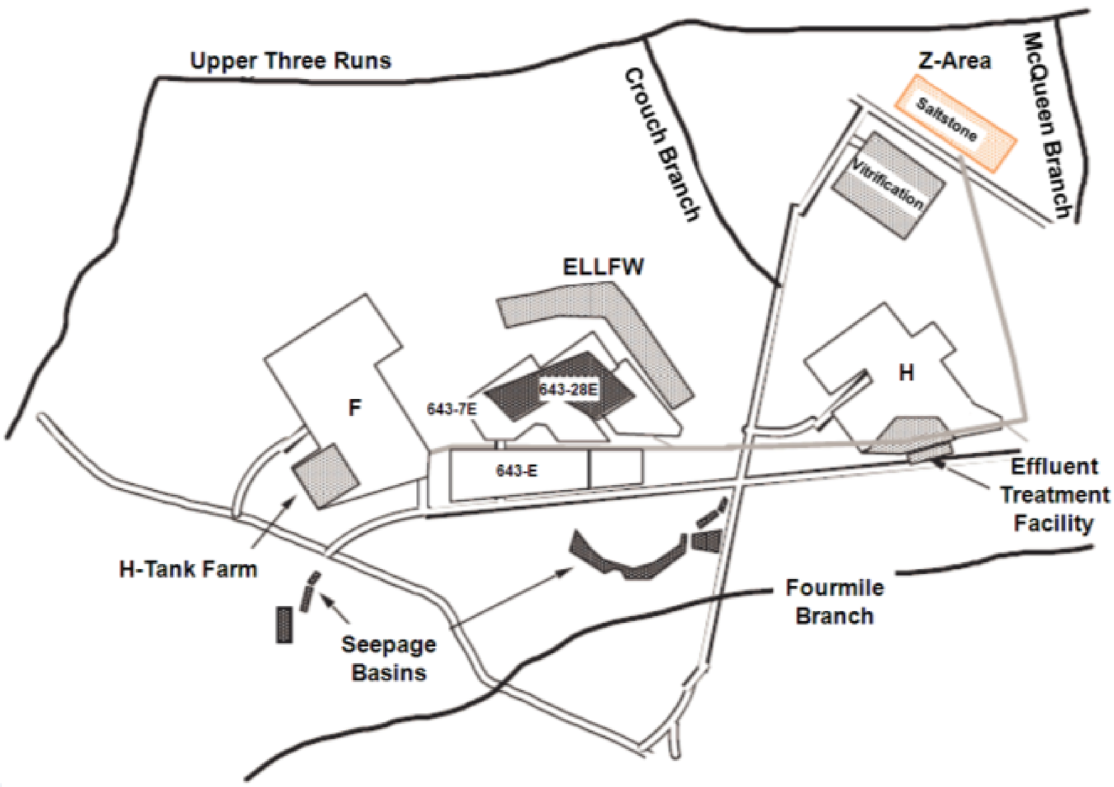

Figure 1.3-1: Location of the Z-Area (based on CBU-PIT-2005-00146)

The Saltstone Facility consists of two facility segments, the SPF, which receives and treats salt solution to produce solidified saltstone, and the SDF, which consists of existing vaults and projected FDCs used for the final disposal of the solidified saltstone. The SPF is permitted by the State of South Carolina as a wastewater treatment facility. The SDF is permitted by the State as a Class 3 Landfill.

Vaults 1 and 4, which were constructed in the 1980s, are constructed of reinforced concrete containing blast furnace slag. Vault 4 (Figure 1.3-2) is 60 m (197 ft) wide, by 180 m (590 ft) long, by 8 m (26 ft) high. The vault is divided into 12 cells of approximately 30 m (98 ft) by 30 m (98 ft). The vault is covered by a permanent roof with a minimum thickness of 10 cm (3.9 in). The roof is sloped, with a minimum slope of 0.19 cm/m (0.15 in/ft) from the middle of the roof parallel to the short axis. The vault walls are approximately 0.46 m (1.5 ft) thick and the base mat is 0.61 m (2.0 ft) thick. Vault 1 (Figure 1.3-3) is 30.5 m (100 ft) wide, by 183 m (600 ft) long, by 8.2 m (27 ft) high. It is divided into six cells of approximately 30 m (98 ft) by 30 m (98 ft). Vault 1 originally had a temporary rolling roof. This temporary roof was later removed, and three of the cells in Vault 1 have been filled with grout and covered with a permanent roof.

Nearly six of the FDCs (Figure 1.3-4) were fully constructed prior to publication of this TER. The diameter of the cells is 45.7 m (150 ft), with an interior height of 6.7 m (22 ft) (this height will increase to 7.16 m (23.5 ft) at the center of the cell). The cylindrical, reinforced concrete cells are constructed below grade. Native soil is used to backfill around the completed cells. The remaining capacity of Vault 4 and up to a total of 64 FDCs may be required to dispose of the total volume of saltstone produced. An engineered cap will be added to the site at closure to cover the disposal units. For perspective, Figure 1.3-5 shows the entire SDF.

Figure 1.3-2: Vault 4 at the SDF (SRR-CWDA-2010-00013)

Figure 1.3-3: Vault 1 at the SDF (2009 PA)

Figure 1.3-4: Disposal Cells 2A and 2B (and Approximate Design of FDC's) at the SDF (SRR-CWDA-2011-00082)

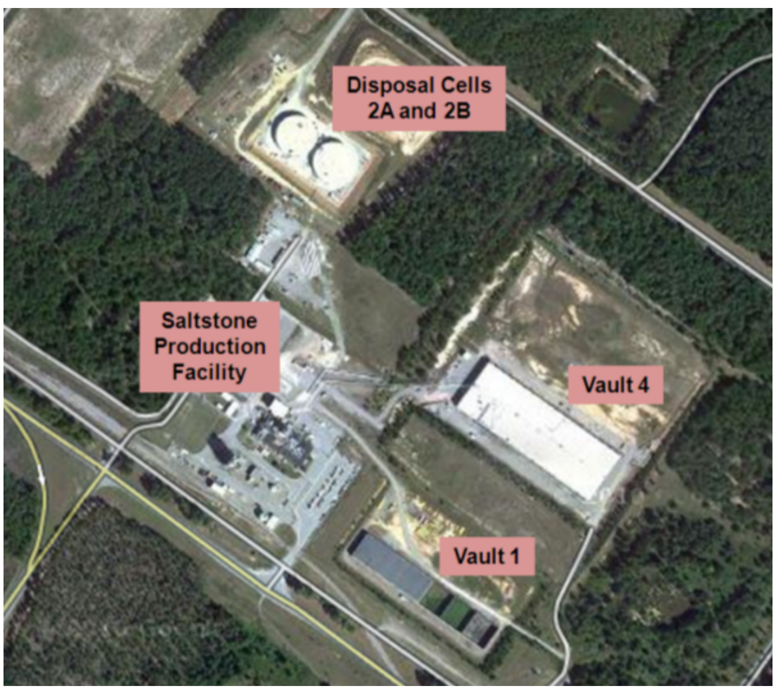

Figure 1.3-5: Areal Picture of SDF

2. Protection of the General Population

2.1. Performance Assessment Overview

DOE conducted the 2009 PA to evaluate the potential long-term dose to a member of the public (i.e., to assess compliance with the performance objective in §61.41). In this PA, the dose to an offsite receptor located 100 m from the disposal site was estimated using projected releases into the air and groundwater from waste disposed in Vaults 1, 4, and the FDCs. DOE also performed an intruder assessment (which DOE evaluated as part of the PA) to evaluate the potential dose to an inadvertent intruder who inhabits the site after the end of institutional controls (i.e., to assess compliance with §61.42). This evaluation is described in Chapter 3.

The 2009 PA consisted of a deterministic analysis for a case that DOE considers to represent expected future conditions, deterministic analyses of alternate cases, and a probabilistic sensitivity and uncertainty analysis. The infiltration rates through the upper layers of the closure cap (i.e., from the vegetative cover layer down to the lower backfill layer) were determined using the Hydrologic Evaluation of Landfill Performance (HELP) code. The flow through the lower layers of the closure cap (i.e., layers below the foundation layer in Figure 2.4-2) is modeled using the PORFLOW™ code with the infiltration rates from the HELP code acting as a boundary condition. In the deterministic cases, PORFLOW™ was also used to model the fluid flow and contaminant transport in the disposal units and the vadose and saturated zones to determine the concentrations of radionuclides in the groundwater. A GoldSim® model that was run in deterministic mode was then used to convert these groundwater concentrations to doses. The air pathway analysis was performed in a separate PORFLOW™ calculation and is described in Section 2.9. Sensitivity and uncertainty analyses were performed using a probabilistic GoldSim® model that was developed by DOE. As discussed in more detail in Section 2.11, the probabilistic model used near-field flow rate information that was calculated by PORFLOW™ in the deterministic cases, and factors affecting the near field flow rates were not evaluated probabilistically.

2.1.1 Description of Deterministic PORFLOW™ Cases

The deterministic cases evaluated by DOE using the PORFLOW™ code are described in Table 2.1-1. These cases consisted of the case that DOE considers to be the “base case” or expected case (i.e., Case A) as well as other cases that are considered by DOE to be sensitivity cases.

Table 2.1-1: Description of Deterministic Cases Modeled by DOE

Case	Description of Key Model Assumptions	Reference
Case A	DOE considers this the "base case," or expected case. Key assumptions include: – Saltstone intact for the duration of the assessment – Vault 1 and 4 walls degraded at t=0 – FDC walls intact for the duration of the assessment	SRR-CWDA-2009-00017 (p 249)
Case B	Same properties as Case A except: – Fast flow path present in the sheet drain system (i.e., along the walls from the roof through the floor) for Vault 4 and the FDCs	SRR-CWDA-2009-00017 (p 249)
Case C	Same properties as Case A except: – Fast flow path present in the sheet drain system (i.e., along the walls from the roof through the floor) for Vault 4 and the FDCs – Fast flow through cracks in saltstone	SRR-CWDA-2009-00017 (p 249)
Case C *PA-4 Case*	Same assumptions as Case C, but additional radionuclides were included	SRR-CWDA-2011-00044 (p 22)
Case D	Same properties as Case A except: – Capillary break present at sheet drains in Vault 4 and the FDCs	SRR-CWDA-2009-00017 (p 249)
Case E	Same properties as Case A except: – Saltstone severely degraded with a hydraulic conductivity of 1.7×10^{-3} cm/s	SRR-CWDA-2009-00017 (p 249)
No Closure Cap (with credit for the lower lateral drainage layer)	Same properties as Case A except: – Cap layers (composite hydraulic barrier, lateral drainage layers, and erosion control layer) modeled with the properties of the surrounding soil	SRR-CWDA-2009-00017 (p 541)
10x Sulfate Attack	Same properties as Case A except: – Diffusion coefficient used in prediction of sulfate attack on the concrete in the disposal unit walls and floors increased by factor of 10	SRR-CWDA-2009-00017 (p 542)
No Sulfate Attack	Same properties as Case A except: – There is assumed to be no damage from ettringite formation to the disposal unit walls and floors. Disposal units do not degrade over time	SRR-CWDA-2009-00017 (p 542)

Case	Description of Key Model Assumptions	Reference
Oxidized Concrete	Same properties as Case A except: – Vault 1 and 4 walls oxidized at closure	SRR-CWDA-2009-00017 (p 550)
Increased Saltstone Hydraulic Conductivity	Same properties as Case A except: – Hydraulic conductivity of saltstone increased to 1×10^{-7} cm/s (from 2×10^{-9} cm/s)	SRR-CWDA-2009-00017 (p 551)
Synergistic Case	Evaluates the effect of an assumed increase in several degradation mechanisms: – Closure cap degraded at a faster rate – Vault 1 and 4 walls initially middle aged oxidized concrete and transition to old aged concrete at 500 years – Vault 1 and 4 walls degraded hydraulically to soil properties at 500 years – FDC concrete degraded chemically and hydraulically at 500 years – Saltstone assumed cracked at closure and oxygen diffuses into the monolith via cracks	SRR-CWDA-2009-00017 (p 548)
Synergistic Case *VP-1*	Same assumptions as the Synergistic Case, but assumed a relative permeability of 1 (i.e., moisture characteristic curves were not used)	SRR-CWDA-2010-00033 VP-1
Synergistic Case *PA-9*	Same assumptions as Synergistic Case, but additional radionuclides were included	SRR-CWDA-2011-00044 (p 106)
Synergistic Case *Updated PA-9*	Same assumptions as Synergistic Case - PA-9, but updated dose methodology was used	SRR-CWDA-2011-00044 (p 107)

Case	Description of Key Model Assumptions	Reference
Case K	Evaluates the effect of revising multiple aspects of the PA to address concerns raised by NRC in the RAIs. Key assumptions in this case include: – Saltstone saturated hydraulic conductivity assumed to have an initial value of $1x10^{-7}$ cm/s (base case value $2x10^{-9}$ cm/s) and to degrade to $1x10^{-6}$ cm/s – Saltstone diffusivity assumed to degrade from $1x10^{-7}$ cm^2/s (base case value) to $5x10^{-6}$ cm^2/s – Moisture characteristic curves not used in determining flow through unsaturated cementitious material (i.e., relative permeability assumed to be 1) – Number of pore volumes required for E_h and pH transitions decreased – An average K_d model (referred to by DOE as a “single porosity” model) abstracted from shrinking core results is used to model Tc release instead of an explicit shrinking core model to accommodate an increase in the postulated number of saltstone fractures – Increased degradation of disposal unit concrete – Inventory of Pu-238, U-234, Th-230, and Ra-226 revised to lower values – K_d values for saltstone, disposal unit concrete, and soil updated – Tc K_d assumed to be 1000 mL/g for reduced saltstone and 10 mL/g for oxidized saltstone – Dose methodology updated (biotic transfer factors updated, chicken and egg pathway included, 25 year buildup in soil assumed, leafy vegetables included in plant transfer factor)	SRR-CWDA-2011-00044 (p 75)
Case K1	Same assumptions as Case K, but the K_d values for Tc were assumed to be 500 mL/g for reduced saltstone and 0.8 mL/g for oxidized saltstone	SRR-CWDA-2011-00044 (p 218)
Case K2	Same assumptions as Case K, but the K_d value for Tc was assumed to be 500 mL/g for reduced saltstone. The oxidized saltstone K_d remained 10 mL/g for this case	SRR-CWDA-2011-00044 (p 219)

2.1.2 Description of GoldSim® Probabilistic Analysis

In addition to the deterministic PORFLOW™ models, DOE also developed a probabilistic GoldSim® model to evaluate the uncertainty in model results and the sensitivity of predicted dose to various assumptions. The DOE GoldSim® model can either run Cases A - E individually or to run "all cases" (including Cases A - E). When the model is run in the "all cases" mode, a case is selected randomly for each realization based on DOE's predicted probabilities for each of these cases occurring, which were based on DOE judgment (2009 PA; Table 5.6-3). The other deterministic cases listed in Table 2.1-1 were developed after the GoldSim® model, and are therefore not included in this model. The probabilistic model relies on output from the deterministic models as input for some parameters and processes. For example, the one-dimensional near-field flow rates through the grouted systems are abstracted from two-dimensional, deterministic PORFLOW™ models.

As part of the development of the probabilistic model, DOE conducted a multi-step model adjustment process, which DOE referred to as benchmarking. During benchmarking, DOE modified the GoldSim® model to optimize the match between intermediate model results (i.e., flux rates and peak well doses) of corresponding simulations run in its deterministic and probabilistic models. GoldSim® parameters adjusted included (1) a pseudo-K_d for Tc-99, (2) flow multipliers for the upper aquifer zone, disposal unit floor, disposal unit wall, and grout, and (3) flow multipliers to adjust contributions from Vault 4, adjust flows to represent a flow divide at the site, and to adjust contributions from certain individual disposal units.

2.1.3 NRC Evaluation – PA Approach

NUREG-1854, the *NRC Staff Guidance for Activities Related to U.S. Department of Energy Waste Determinations* (NRC, 2007b) states that either a deterministic or probabilistic analysis can be used by DOE to demonstrate compliance with the performance objectives in 10 CFR 61, Subpart C, although probabilistic approaches are preferred for complex assessments. Probabilistic assessments are preferred because a deterministic model, without additional sensitivity analyses, gives no indication of the sensitivity of the results to certain parameters or of the importance of the uncertainty in the parameters. The use of a deterministic approach may result in the need for stronger justification of code input parameter values and may require further analysis of doses using upper or lower bounding conditions to gain insights into the range of dose estimates. Additionally, in a deterministic analysis, key parameters should be reasonably conservative or well justified unless adequate sensitivity analyses have been provided that demonstrate the overall risk significance of the parameters is small. These sensitivity analyses need to consider how uncertainties in different parameters are interrelated. If the sensitivity of interrelated parameters is only evaluated in "one-off" analyses in which one parameter is varied at a time, the potential effect of the parameters on the dose could be missed. For example, a degraded cap might also lead to an increased rate of degradation of the disposal units and waste form due to the increased infiltration. Additionally, the sensitivity analyses should consider the relative change in dose based on changes to the parameter, rather than absolute changes.

DOE's overall approach of performing a deterministic base case analysis, deterministic sensitivity cases, and a probabilistic sensitivity/uncertainty analysis is consistent with the NRC's guidance. However, NRC staff does not agree with some of the specific ways in which this approach was implemented, such as the selection of Case A as the base case. NRC staff also has significant concerns about key aspects of DOE's probabilistic analysis. In particular, the NRC staff finds that DOE's approach of using certain outputs of the deterministic model directly in the probabilistic model has obscured key factors that had the greatest effect on dose results. For example, the use of near-field flow rate information calculated by PORFLOW™ in the deterministic cases, limited the evaluation of factors affecting the near field flow rates in the probabilistic model.

NRC staff disagrees with DOE's determination that Case A represents the expected, or most probable, case (i.e., the base case). As described in the NRC RAIs (NRC, 2010b, i) the PA base case scenario is unrealistic and non-conservative for the following reasons:

(i) The base case model is inconsistent with known conditions. Significant site characteristics that have not been adequately incorporated into the model include the following:

- Fractured saltstone is not considered in the base case even though fracturing of saltstone has been observed. In addition, shrinkage has been observed and is not included in the model. (Section 2.6)

- The PA models appear to be inconsistent with observed, advective contaminant releases from Vault 4. (Section 2.5)

- Material interfaces have shown to be relevant to performance; however they are not considered in the PORFLOW™ model. (Section 2.5)

(ii) The base case model does not adequately account for the observed range of experimentally measured values representing saltstone and disposal unit initial conditions, or the uncertainty in the predicted temporal evolution of conditions. NRC concerns with parameter and conceptual model uncertainty include the following:

- The hydraulic conductivity and effective diffusion coefficient for saltstone are time-invariant as the base case model does not adequately account for temporal variation. (Section 2.6)

- The modeled initial hydraulic conductivity of saltstone does not fully account for or attempt to bound differences between the properties of laboratory-prepared and full-scale, as-emplaced saltstone. (Section 2.6)

- The PA does not account for the possible effects of potentially relevant disposal unit degradation mechanisms or provide a basis for discounting those mechanisms. (Section 2.5)

(iii) The base case does not have adequate technical bases. NRC concerns with the limited model support include the following:

- Model support for geotextile filter fabrics and the lateral drainage layers is not commensurate with their expected long-term performance and risk significance. (Section 2.4)

- The moisture characteristic curves implemented in the base case for intact and fractured cementitious materials, which significantly reduce flow, lack adequate support considering their risk significance. (Section 2.7)

- The chemical stability of saltstone provides a significant barrier to transport; however, the basis for the E_h-pH evolution of cementitious materials is very limited. (Section 2.6)

- The basis for the adopted Tc pseudo-K_d of 1,000 mL/g is inaccurate and insufficient. (Section 2.7)

Many of the concerns described above for Case A apply to the other PORFLOW™ deterministic cases as well. However, in DOE's response to comments PA-8 and SP-19 (SRR-CWDA-2011-00044), DOE created new PORFLOW™ cases, Cases K, K1, and K2, to address these concerns. Of the cases listed in Table 2.1-1, the NRC staff considers Cases K, K1, and K2 to best represent the current and future expected conditions of the disposal units and the waste form. Of these three cases, NRC staff considers Case K1 to contain K_d values for Tc in saltstone that are the most realistic based on experimental data (Section 2.7). NRC staff is therefore relying strongly on Case K1 in its determination of compliance. Although NRC staff believes that this case is the best representation of the current and future expected conditions of the cases DOE modeled, NRC staff finds that there are some aspects of the modeling performed for this case that do not reflect the expected future behavior of the system. For example, Cases K, K1, and K2 use an average K_d (referred to by DOE as a "single porosity" model) approach to modeling the Tc K_d values for saltstone (Section 2.7). This approach results in an underprediction of the release rates when the fraction of saltstone oxidized is low and an overprediction of the release rates when the fraction oxidized is high. Also, as described in more detail in Section 2.5, intermediate outputs from the PORFLOW™ model indicate that the modeled disposal unit performance in Cases K, K1, and K2 is not consistent with the expected performance, and the modeled disposal unit performance significantly lowers the predicted peak doses. There is also significant uncertainty in the assumptions regarding the timing, rate, and final amount of fracturing (Section 2.6). The values assumed for these parameters significantly affect the predicted maximum dose as well as the timing of the dose in Cases K, K1, and K2.

The NRC staff believes that the addition of a probabilistic model to the DOE PA is a valuable improvement to their PA and this type of modeling should be continued in the future. However, the NRC staff has a number of concerns with the specific probabilistic GoldSim® model used in this PA. As discussed in more detail in Section 2.11 and below, the concerns include difficulties caused by DOE's approach of basing parts of the probabilistic model on outputs of the deterministic model, adjustments made to the model based on the deterministic output, and apparent errors in the probabilistic model. As part of DOE's approach, the deterministic flow rates from PORFLOW™ for Cases A - E were "hard wired" into the probabilistic model. As described in the list presented above, the NRC staff has concerns about some of the key assumptions in Cases A - E (e.g., hydraulic conductivity, moisture characteristic curves), and, as a result, NRC staff finds that the flow rates generated using PORFLOW™ for these cases are optimistic. Additionally, because deterministic flow results are "hard wired" into the probabilistic models for the near-field, the probabilistic models could not provide information about the relative importance of assumptions and parameters affecting flow and near field release, which the NRC staff expects to have a significant effect on model results. As discussed in Section 2.11, NRC staff has concerns with the benchmarking process used in the development of the GoldSim® model (i.e., the process used to modify the GoldSim® model to optimize the match between intermediate model outputs in the PORFLOW™ and GoldSim® models). NRC staff also identified a number of potential errors in the GoldSim® model that could significantly affect the results (e.g., key features of Case C not being implemented when the model is run in "all cases" mode, incorrect linkages between the unsaturated zone cells, and an incorrect value for the Vault selector for Vault 4 [Section 2.11]). For these reasons, the NRC staff is not using the results of the DOE GoldSim® model as part of the assessment of compliance with the performance objectives.

NRC staff concludes that Case K1 best represents the current and future expected conditions of the SDF. Therefore, the NRC staff is relying strongly on Case K1 in its determination of compliance. The NRC staff is not using the uncertainty and sensitivity analyses performed using the probabilistic model as part of the assessment of compliance. Instead, the NRC evaluation of uncertainty in dose and the sensitivity of dose to input assumptions is based on: (1) an evaluation of the results of DOE's deterministic sensitivity cases, (2) an evaluation of intermediate model results from DOE's deterministic sensitivity analyses, and (3) independent sensitivity analyses focused on Cases K, K1, and K2 (Section 2.13).

2.2 Source Term and Inventory

2.2.1 Source Term and Inventory

The methodology used by DOE to estimate the closure inventory in the SDF is described in SRNS-J2100-2008-00004. The assumed inventory for Vault 1 is based on the current estimated inventory since there are no plans to add additional inventory to this vault. The assumed inventory for Vault 4 is based on the current inventory in Vault 4 plus the inventory intended for disposal in this vault in the future. The void spaces in the Vault 1 and Vault 4 walls are assumed to be filled with pore fluid containing the same concentration of radionuclides as in the saltstone. The purpose of this assumption is to evaluate the potential contamination in the vault wall weeping that occurred during saltstone placement in these vaults. In Vault 1, approximately 0.65% of the inventory is assumed to be in the wall, and in Vault 4, 0.5% of the inventory is assumed to be in the wall. The remainder of the inventory for these vaults is assumed to be located in the saltstone. For the FDCs, the inventory was determined based on the remaining tank farm inventory and assumptions about the removal efficiencies of the treatment processes. The total predicted inventory for the FDCs is assumed to be divided evenly over all of the FDCs.

The resulting inventory projected by DOE for the disposal units at the SDF is presented in Table 2.2-1. These projected inventories are based on the decay of the radionuclides to the date of October 1, 2030, the expected date of closure for the SDF.

Table 2.2-1: Projected Inventory at Time of Closure

	Vault 1 (Ci)	Vault 4 (Ci)	Individual FDC (64 FDCs total) (Ci)	Total SDF (Ci)
Ac-227		1.60×10^{-5}	1.70×10^{-7}	2.70×10^{-5}
Al-26		3.40×10^{-1}	1.90×10^{-1}	1.30×10^{1}
Am-241	4.70×10^{-4}	1.30×10^{2}	1.40	2.20×10^{2}
Am-242m		6.70×10^{-2}	5.90×10^{-4}	1.00×10^{-1}
Am-243		1.80	3.70×10^{-2}	4.20
Ba-137m	4.10	2.80×10^{5}	2.20×10^{1}	2.80×10^{5}
Bk-249		1.80×10^{-28}	1.80×10^{-28}	1.20×10^{-26}
C-14	1.30	2.70×10^{1}	2.00	1.60×10^{2}
Ce-144		1.80×10^{-9}	3.60×10^{-10}	2.50×10^{-8}
Cf-249		6.50×10^{-13}	6.70×10^{-13}	4.40×10^{-11}
Cf-251		1.20	2.30×10^{-14}	1.20
Cf-252		1.80×10^{-18}	1.80×10^{-18}	1.20×10^{-16}
Cl-36	7.60×10^{-4}	3.00×10^{-3}	4.20×10^{-4}	3.10×10^{-2}
Cm-242		6.70×10^{-2}	6.30×10^{-19}	6.70×10^{-2}
Cm-243		2.10×10^{-1}	2.10×10^{-4}	2.20×10^{-1}
Cm-244		1.30×10^{2}	9.50×10^{-1}	1.90×10^{2}
Cm-245		9.20×10^{-1}	2.40×10^{-4}	9.40×10^{-1}
Cm-247		3.90×10^{-6}	7.10×10^{-14}	3.90×10^{-6}
Cm-248		1.20×10^{-13}	7.40×10^{-14}	4.90×10^{-12}
Co-60	8.20×10^{-5}	4.60×10^{-1}	5.40×10^{-2}	3.90
Cs-134		5.20×10^{-1}	1.50×10^{-5}	5.20×10^{-1}
Cs-135		5.40	1.30×10^{-4}	5.40
Cs-137	4.30	3.00×10^{5}	2.30×10^{1}	3.00×10^{5}

	Vault 1 (Ci)	Vault 4 (Ci)	Individual FDC (64 FDCs total) (Ci)	Total SDF (Ci)
Eu-152	$1.80x10^{-3}$	$9.70x10^{-2}$	$9.80x10^{-2}$	6.40
Eu-154	$2.30x10^{-4}$	$1.20x10^{1}$	1.80	$1.30x10^{2}$
Eu-155		$6.80x10^{-1}$	$1.30x10^{-1}$	9.00
H-3	6.10	$2.60x10^{2}$	$3.00x10^{1}$	$2.20x10^{3}$
I-129	$1.10x10^{-1}$	$2.80x10^{-1}$	$3.80x10^{-1}$	$2.50x10^{1}$
K-40	$7.60x10^{-4}$	$3.00x10^{-3}$	$4.20x10^{-4}$	$3.10x10^{-2}$
Na-22		$1.50x10^{-1}$	$6.90x10^{-2}$	4.60
Nb-93m	$2.50x10^{-1}$	8.40	$3.70x10^{-1}$	$3.20x10^{1}$
Nb-94	$2.50x10^{-3}$	$8.70x10^{-2}$	$3.80x10^{-3}$	$3.30x10^{-1}$
Ni-59	$3.50x10^{-2}$	$4.00x10^{-1}$	$8.40x10^{-2}$	5.80
Ni-63	$7.80x10^{-1}$	$2.20x10^{1}$	2.40	$1.80x10^{2}$
Np-237	$4.50x10^{-3}$	$6.10x10^{-1}$	$5.00x10^{-2}$	3.80
Pa-231		$9.30x10^{-5}$	$9.80x10^{-7}$	$1.60x10^{-4}$
Pd-107	$1.90x10^{-3}$	$5.00x10^{-2}$	$5.60x10^{-3}$	$4.10x10^{-1}$
Pm-147		$4.10x10^{-1}$	$7.70x10^{-2}$	5.30
Pr-144		$1.80x10^{-9}$	$3.60x10^{-10}$	$2.50x10^{-8}$
Pt-193	$3.70x10^{-1}$	$1.00x10^{1}$	1.10	$8.10x10^{1}$
Pu-238	$7.80x10^{-3}$	$9.10x10^{3}$	$1.70x10^{2}$	$2.00x10^{4}$
Pu-239	$1.20x10^{-2}$	$3.80x10^{2}$	$1.50x10^{1}$	$1.30x10^{3}$
Pu-240	$1.20x10^{-2}$	$1.20x10^{2}$	4.10	$3.80x10^{2}$
Pu-241	$9.80x10^{-3}$	$2.40x10^{3}$	$4.20x10^{1}$	$5.10x10^{3}$
Pu-242	$9.00x10^{-4}$	$8.10x10^{-1}$	$3.90x10^{-3}$	1.10
Pu-244		$1.60x10^{-2}$	$1.60x10^{-5}$	$1.70x10^{-2}$
Ra-226	$6.40x10^{-7}$	4.10	$7.80x10^{-7}$	4.10
Ra-228		$1.60x10^{-6}$	$8.70x10^{-5}$	$5.60x10^{-3}$
Rh-106	$1.50x10^{-10}$	$9.10x10^{-7}$	$1.20x10^{-6}$	$7.80x10^{-5}$
Ru-106	$1.50x10^{-10}$	$9.10x10^{-7}$	$1.20x10^{-6}$	$7.80x10^{-5}$
Sb-125	$1.60x10^{-1}$	5.70	$2.40x10^{-1}$	$2.10x10^{1}$

	Vault 1 (Ci)	Vault 4 (Ci)	Individual FDC (64 FDCs total) (Ci)	Total SDF (Ci)
Sb-126	1.40×10^{-1}	9.00×10^{-1}	1.20	7.80×10^{1}
Sb-126m	1.00	6.40	8.20	5.30×10^{2}
Se-79	3.00×10^{-1}	4.60×10^{1}	1.40	1.40×10^{2}
Sm-151		4.20×10^{1}	5.90×10^{1}	3.80×10^{3}
Sn-126	1.00	6.40	8.20	5.30×10^{2}
Sr-90	6.90×10^{-3}	2.40×10^{5}	3.70×10^{1}	2.40×10^{5}
Tc-99	1.10×10^{2}	5.80×10^{2}	5.40×10^{2}	3.50×10^{4}
Te-125m	3.80×10^{-2}	1.40	5.80×10^{-2}	5.20
Th-229	3.00×10^{-1}	2.50×10^{1}	3.90×10^{-2}	2.80×10^{1}
Th-230	4.10×10^{-1}	7.50	1.90×10^{-1}	2.00×10^{1}
Th-232		3.20×10^{-4}	1.40×10^{-3}	9.00×10^{-2}
U-232		4.40×10^{-2}	3.10×10^{-4}	6.40×10^{-2}
U-233	2.80×10^{-1}	2.40×10^{1}	3.70×10^{-2}	2.70×10^{1}
U-234	2.80×10^{-1}	2.60×10^{1}	1.30×10^{-1}	3.50×10^{1}
U-235	3.20×10^{-3}	4.70×10^{-1}	3.00×10^{-3}	6.70×10^{-1}
U-236	3.20×10^{-3}	7.70×10^{-1}	1.60×10^{-2}	1.80
U-238	7.40×10^{-3}	5.90×10^{-1}	1.00×10^{-1}	7.00
Y-90	6.90×10^{-3}	2.40×10^{5}	3.70×10^{1}	2.40×10^{5}
Zr-93	2.50×10^{-1}	8.40	3.70×10^{-1}	3.20×10^{1}
Total	**1.30×10^{2}**	**1.10×10^{6}**	**1.00×10^{3}**	**1.10×10^{6}**

1.0 MBq is 3.7×10^{4} Ci

2009 PA; Tables 3.3-1, 3.3-3, 3.3-5, and 3.3-7

The inventory values listed in the above table were used in the PA calculations (2009 PA), with the exception of Cases K, K1, and K2, which used revised values for the inventory of Pu-238, U-234, Th-230, and Ra-226 (Table 2.2-2). Revised inventory values were used for Pu-238, Ra-226, Th-230, and U-234 for Vault 4 and for Ra-226 and Th-230 for the FDCs. The inventory of Pu-238 and U-234 in Cases K, K1, and K2 were revised for Vault 4 based on sample analyses for the waste already disposed in Vault 4 (SRR-CWDA-2011-00115). DOE stated that the inventories of Th-230 and Ra-226 were also revised in these cases to remove conservative assumptions made in the original estimation.

Table 2.2-2: Revised Inventory of Ra-226 and its ancestors in Cases K, K1, and K2

	Vault 4		Individual FDC	
	Original Inventory (Ci)	Case K Inventory (Ci)	Original Inventory (Ci)	Case K Inventory (Ci)
Ra-226	4.10	1.0×10^{-3}	7.80×10^{-7}	1.3×10^{-5}
Th-230	7.50	1.0×10^{-2}	1.90×10^{-1}	1.3×10^{-4}
U-234	2.60×10^{1}	1.0×10^{1}	1.30×10^{-1}	1.30×10^{-1}
Pu-238	9.10×10^{3}	1.0×10^{3}	1.70×10^{2}	1.70×10^{2}

1.0 MBq is 3.7×10^{4} Ci
SRR-CWDA-2011-00044; Response to PA-8.

In the estimation of the original inventory (SRNS-J2100-2008-00004), the inventory of Th-230 and Ra-226 was determined based on an assumption of transient equilibrium with U-234. However, DOE noted in the response to comment IN-5 (SRR-CWDA-2011-00044) that this is a significant conservatism based on the relatively long half-lives and the relatively young age of the waste. In SRR-CWDA-2011-00115, the inventory of Ra-226 and Th-230 was reevaluated using two different approaches. Because the sample results for Ra-226 and Th-230 were below the analytical detection limit in all samples, a predicted inventory of these radionuclides was generated based on the assumption that the radionuclides were present at the level of the detection limit. Additionally, an alternate inventory of these radionuclides was generated based on the in-growth of these radionuclides from U-234. The results of these estimates are presented in Table 2.2-3.

Table 2.2-3: Estimates of Ra-226 and Th-230 inventory in Vault 4

	Ra-226 (Ci)	Th-230 (Ci)
Estimated inventory based on analytical detection limits (SRR-CWDA-2011-00115; page 5)	6.6	20
Estimated inventory based on U-234 inventory and in-growth rates (SRR-CWDA-2011-00115; page 5)	7.6×10^{-4} (1)	7.6×10^{-3}
Reported Inventory in Vault 4 as of 9/30/10 based on an alternative determination (X-CLC-Z-00034)	9.29	2.82×10^{-2}

1.0 MBq is 3.7×10^{4} Ci
(1) Note that Table 5 of SRR-CWDA-2011-00115 also reports an inventory of 0.7 MBq (1.9×10^{-5} Ci) for Ra-226 based on a similar approach

Information on the inventory of Ra-226 and Th-230 disposed to date in Vault 4 has previously been reported by DOE in X-CLC-Z-00034 and X-CLC-Z-00027. The reported inventory of these radionuclides in these documents was generated using an alternative calculation based on the original waste tank radionuclide concentrations, the volumes of interstitial liquid removed and salt dissolved, and the volumes of the batches transferred to SDF. The inventory of Ra-226 and Th-230 generated using this method is also given in Table 2-4.

As can be seen from the information in Table 2-4, the inventory generated based on in-growth from U-234 is significantly smaller than the inventories based on the detection limit for both radionuclides. The reported inventory based on the special calculation was much lower than the one based on the detection limits for Th-230, but was higher for Ra-226. As discussed in the response to comment IN-1 (SRR-CWDA-2010-00033), DOE stated that the reported Ra-226 inventory to date was calculated from original waste tank inventories that were based on analytical detection limits.

In Section 5.2.2 of the 2009 PA, DOE identified key radionuclides, or the radionuclides which contribute the most significantly to dose, based on the peak all-pathways dose calculated at 100 m using the Base Case (Case A) assumptions over a period of 20,000 years. DOE included any radionuclides with a calculated peak dose of more than $5x10^{-4}$ mSv/yr (0.05 mrem/yr) in the list of key radionuclides. The resulting radionuclides that DOE identified as key radionuclides were Tc-99, I-129, Ra-226, Np-237, and Pa-231. The radionuclides that contributed most to the calculated dose were I-129, Tc-99, and Ra-226 and its progeny for Case A as well as for the sensitivity cases other than Case K. In Case K, Ra-226 contributed less to the calculated dose due to the revised inventory for Ra-226 and its ancestors, particularly Th-230.

The Case A and Cases K, K1, and K2 PORFLOW™ calculations included all of the radionuclides listed in Table 2-1. However, the other PORFLOW™ cases only included the key radionuclides (i.e., Tc-99, I-129, Ra-226, Np-237, and Pa-231) and the parents and progeny of these radionuclides (i.e., Ac-227, Pb-210, Pu-238, Th-229, Th-230, U-233, U-234, and U-235). In response to NRC RAIs, DOE reran the PORFLOW™ calculations for Case C and the Synergistic Case with 13 additional radionuclides to evaluate the potential dose from radionuclides not included in the initial calculations (SRR-CWDA-2011-00044). As described in the response to comment PA-4, the results for the Case C analysis with additional radionuclides were similar for Sector I for the 20,000 year time of analysis. The results for Sector B were similar until approximately 16,000 years after which the analysis that included additional radionuclides was higher. Similarly, the graphs presented in the response to comment PA-9 (SRR-CWDA-2011-00044) indicated that the doses from the Synergistic Case and the PA-9 case that included additional radionuclides were similar. However, the dose values presented for the Synergistic Case in this RAI response were higher than those presented elsewhere. A comparison of the PA-9 case to the Synergistic Case results presented in the original PA (DOE-WD-2005-001) shows a larger difference in dose between these two cases. The GoldSim® calculations included nearly all of the radionuclides listed in the above table, with the exception of radionuclides with short half-lives that did not require transport modeling (SRR-CWDA-2010-00033).

2.2.2 NRC Evaluation – Source Term and Inventory

Because the dose predicted by the 2009 PA is directly related to the assumed inventory, the inventory disposed of at SDF was identified as a key monitoring factor in NRC's previous review (NRC, 2005), and the NRC staff has been monitoring this inventory since that time. NRC staff will continue to monitor the inventory disposed of at the SDF by tracking the actual inventory disposed against the inventory in Table 2-2 (for Ra-226, Th-230, U-234, and Pu-238) and Table 2-1 (for all remaining radionuclides) as part of monitoring. The dose is a function of both the total dose, as well as the spatial distribution of the inventory among the different disposal units, so NRC staff will monitor both the total inventory disposed in the SDF as well as the inventory disposed in Vault 1, Vault 4, and in each individual FDC. If the total inventory or the inventory in an individual vault or FDC is higher than the assumed values, an analysis will need to be performed to understand the dose consequences of the increased inventory. Also, as part of monitoring, NRC staff will continue to review the methodology used to measure and determine the inventory to confirm that risk significant radionuclides are being assessed adequately.

NRC staff notes that there is significant uncertainty in the inventory of Ra-226 and Th-230 due to the fact that these radionuclides are present in the salt waste at levels that are below the detection limit of the analytical method used by DOE. As described in more detail in Section 2.13, the dose due to these radionuclides is potentially significant if they are present at levels that are as high as the detection limit. Because of this, it is important for the inventory of these radionuclides to be well understood. NRC staff finds that the alternate method of evaluating the Th-230 and Ra-226 inventory used in the Case K evaluation (i.e., determining the Th-230 and Ra-226 inventory from the in-growth from U-234) is appropriate for waste from tanks that have not had thorium inputs. However, this method would result in an underprediction of the Th-230 and Ra-226 inventory for waste that had sources of thorium other than the in-growth from U-234. In such waste, the inventory of Th-230 and Ra-226 should be based on the detection limits if the measured concentrations are below the detection limits or on the measured concentrations if the concentrations are above the detection limits. NRC staff will continue to monitor the methodology used to determine the Th-230 and Ra-226 inventory as part of the monitoring process, though as noted in Section 2.13, the inventory of these radionuclides may be less risk significant if assumptions used in the Case K modeling for the sorption coefficients prove to be true.

The NRC staff finds that the approach used by DOE to determine the key radionuclides using a case in which little degradation is assumed to occur, such as Case A, is potentially problematic because different radionuclides may be risk-significant in cases in which more degradation occurs. This approach is especially potentially problematic when the case used is not consistent with, and contains less degradation than, the known and future expected conditions. Additionally, the NRC staff was also concerned that the exclusion of the non-key radionuclides from many of the cases other than Case A could have led to an underestimation of the estimated dose. However, in response to NRC staff's concerns, DOE included all of the radionuclides in the Case K analysis. DOE also performed an evaluation of the effect of adding additional radionuclides to the Case C and Synergistic Case analyses. These evaluations indicated that the exclusion of the non-key radionuclides from many of the deterministic analyses likely did not have a large effect on the calculated dose. It therefore does not appear that DOE's approach for determining the key radionuclides led to the exclusion of risk significant radionuclides in the 2009 PA.

2.3 Scenario Selection and Receptor Group

2.3.1 Period of Performance and Institutional Controls

The period of performance assumed by DOE in the 2009 PA for the purpose of demonstrating compliance with the performance objectives was a period of 10,000 years following the closure of the site. For the purpose of the PA, DOE assumed that institutional controls that would last for a period of 100 years following closure of the site, which is assumed to occur in 2030. During the time of institutional controls, it is assumed that the site is owned and controlled by the federal government and access to the site is limited. Maintenance of the closure cap is also assumed to occur during the institutional control period. The assumption of a 100-year institutional control period was based on regulation (§61.59), not on DOE's future plans for the site. In reality, the site may be owned and controlled by the federal government for a longer period of time than 100 years.

DOE ran all transport models to at least 20,000 years for the purposes of determining peak concentrations that occur after the 10,000-year performance period. In addition, a deterministic PORFLOW™ analysis was performed for Case A for 40,000 years. This calculation only included the radionuclides identified as key dose contributing radionuclides (SRR-CWDA-2011-00044). A probabilistic analysis to 450,000 years was also run using the GoldSim® model. In the response to NRC RAIs, DOE noted that the long-term analyses are meant only to help identify the model's sensitivity to certain parameters and their uncertainty, and are provided for trend spotting only (SRR-CWDA-2010-00033).

2.3.2 Scenario Identification

The scenario considered by DOE in evaluating the dose to the member of the public was the resident farmer scenario. In this scenario, it is assumed that a receptor drills a well into a groundwater aquifer at a location 100 m from the disposal unit and uses this water as a potable water source as well as for agriculture (i.e., irrigation and water for livestock) following the end of active institutional controls. The resident farmer is also assumed to use nearby streams for recreational activities, such as swimming and fishing.

The point of compliance for the dose to the member of the public is the point of highest projected dose beyond a 100 m buffer zone surrounding the disposal facility. In the 2009 PA, the 100 m doses were determined in 12 sectors that are downgradient of the facility. A diagram of the SDF, the points of compliance for the member of the public and the intruder, and the sectors analyzed can be seen in Figure 2.3-1.

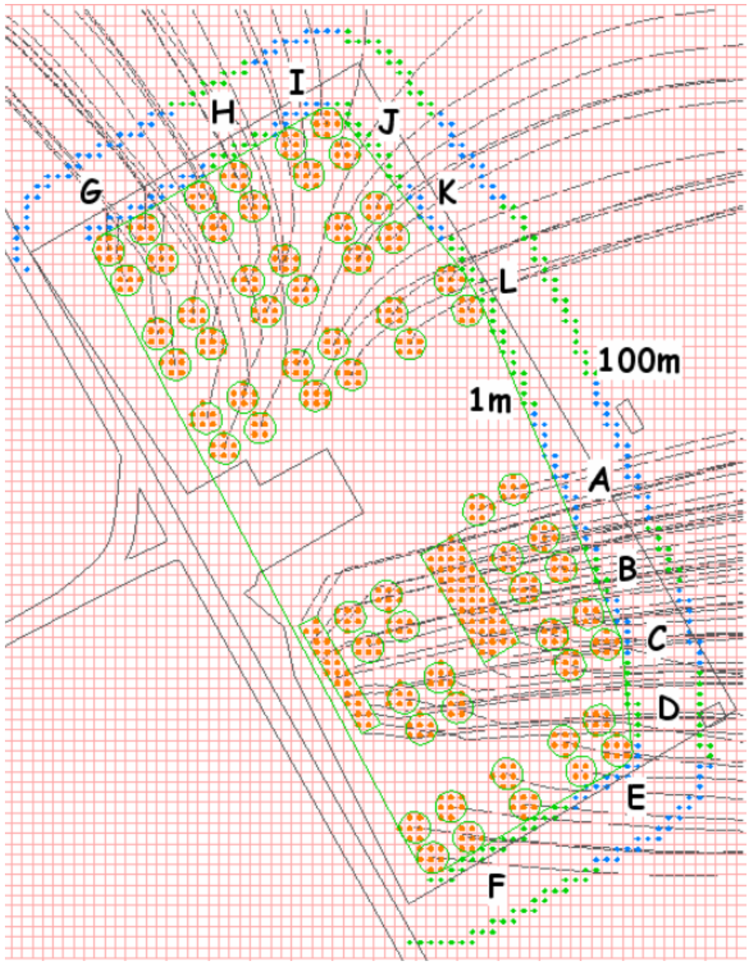

Figure 2.3-1: Locations of the Compliance Points and the Sectors Analyzed (2009 PA; Figure 4.4-70)

2.3.3 Identification of Relevant Features and Processes

The primary mechanism of transport evaluated in the assessment of the dose to the member of the public is leaching from the saltstone into the groundwater and subsequent human use. The gas phase diffusion of radionuclides from the waste to the surface was also evaluated. Contaminant disturbance caused by bio-intrusion or erosion was not considered in the PA because DOE did not believe these were credible mechanisms based on the depth waste (i.e., > 3.7 m (12 ft) below the erosion control barrier).

The transport of the radionuclides from the SDF to a well at 100 m from the disposal facility and to outcrops at two streams, the McQueen Branch to the east and the Upper Three Runs to the north was modeled as flow through porous media. The groundwater pathway dose for each sector was based on peak groundwater concentration in that sector. Additionally, the recreational dose for the receptor was based on the seepline location with the maximum concentration for each radionuclide. PORFLOW™ modeling of the transport of radionuclides to the stream seeplines was only performed for the radionuclides identified by DOE as being key radionuclides (i.e., Tc-99, I-129, Ra-226, Np-237, Pa-231). For the other radionuclides, the seepline concentrations were assumed to be equal to 35% of the 100 m concentration. This assumption was based on 35% being the bounding ratio of the seepline concentration to the 100 m concentration for any radionuclide.

The modeled release and transport of the radionuclides from the waste form is highly dependent on assumptions made regarding the release from the source term, the assumed degradation of the waste form and disposal units, the assumed degradation of the engineered cover, and the transport in the subsurface. More detail is provided about DOE's identification of the relevant features and processes for these areas in other sections of this TER. A description of the features and processes included in the modeling of the release of radionuclides from the waste form is in Section 2.7, descriptions of the features and processes involved in the degradation of the waste form and disposal units are located in Section 2.6 and 2.5 respectively, a description of the modeled long term performance of the engineered cover is in Section 2.4, and a description of the transport in the subsurface is discussed in Section 2.8.

2.3.4 Receptor Characteristics

The dose assessment for the member of the public evaluated well water, surface water, and air pathways. The exposure pathways related to the use of potentially contaminated well water included:

- direct ingestion of well water
- ingestion of vegetables grown in soil that is irrigated with well water
- ingestion of meat and milk from livestock that drink well water and eat fodder grown in soil irrigated with well water
- ingestion and inhalation of well water while showering
- dermal contact with well water while showering
- inhalation of well water during irrigation
- inhalation of dust from soil irrigated with well water
- ingestion of soil irrigated with well water
- direct radiation from soil irrigated with well water

The exposure pathways considered for the use of surface water included:

- direct radiation during recreational activities in the stream
- incidental inhalation of stream water during recreational activities
- incidental ingestion of stream water during recreational activities
- dermal contact with stream water during recreational activities
- ingestion of fish from the stream

The accidental ingestion of water while swimming and showering was considered as part of the total well water ingested and was not considered separately. The dose due to the dermal absorption of radionuclides was not considered quantitatively because the radionuclides are generally absorbed poorly and DOE therefore considered this dose to be insignificant.

The ingestion of shellfish was not included in the dose assessment because the Upper Three Runs and McQueen branch are not significant sources of edible shellfish. The ingestion of wild game was also not included because livestock raised on the site would be more affected by the disposed radionuclides than game, which are more transient. Additionally, the ingestion of game by the receptor would offset the amount of ingestion of livestock, which could result in a lower total dose. The ingestion of poultry and eggs was not included in the initial PA by DOE (DOE-WD-2005-001). However, an evaluation of this dose was performed by DOE in response to comment B-2 (SRR-CWDA-2011-00044), and the dose from this pathway was not found to be significant based on the parameter values selected in this analysis (Section 2.10).

The potential dose due to the diffusion of gaseous radionuclides through the waste form and overlaying materials was also evaluated (Section 2.9).

2.3.5 NRC Evaluation – Scenario Selection and Receptor Groups

In the PA, DOE assumed a period of performance of 10,000 years, though all analyses were run to 20,000 years, which was generally sufficient to capture the peak dose. NUREG-1854 (NRC, 2007b) states "Generally, a period of 10,000 years after closure is sufficient to capture the peak dose from the more mobile, long-lived radionuclides and to demonstrate the influence of the natural and engineered systems in achieving the performance objectives (NRC, 2000). However, assessments beyond 10,000 years may be necessary to ensure (1) that the disposal of certain types of waste does not result in markedly high impacts to future generations or (2) evaluate waste disposal at arid sites with extremely long groundwater travel times." DOE's approach is consistent with this guidance. DOE's assumption of a 100-year institutional control period is consistent with NRC guidance in NUREG-1854 and the regulations in §61.59(b).

The NRC staff finds that DOE's use of the resident farmer scenario and the pathways selected by DOE for this scenario are reasonable and appropriate based on regional practices near SRS. Additionally, NRC staff finds that the use of a 100 m point of compliance for the member of the public is consistent with the typical assumption of a 100 m buffer zone described in NUREG-1854 (NRC, 2007b). NRC's evaluation of the features and processes included in the modeled release and transport (i.e., the release from the source term, the assumed degradation of the waste form and disposal units, the assumed degradation of the engineered cover, and the transport in the subsurface) are discussed in the portions of this TER that specifically address those areas.

2.4 Infiltration and Erosion Control

This section addresses the engineered soil covers (closure caps) DOE plans to use to limit infiltration and erosion at the SDF. In addition, the erosion control layer of the closure cap is expected to deter potential inadvertent intruders. The current closure cap design is preliminary and will be finalized closer to the time of SDF closure.

2.4.1 Local Meteorology and Climatology

In the PA, DOE states that the climate at SRS is humid and subtropical. On average, approximately half of summer days have maximum temperatures exceeding 32°C (90°F). Less than one third of winter days have minimum temperatures below freezing on average, and days with temperatures below -7°C (20°F) are infrequent. Mean annual precipitation, consisting primarily of rainfall, is reported to be approximately 121 cm/yr [48 in/yr], with a range of 88 to 183 cm/yr [35 to 72 in/yr] (DP-MS-87-126, as reported in WSRC-STI-2008-00244).

2.4.2 Infiltration and Erosion Control

Closure of the SDF will include the construction of two engineered closure caps over Vault 1, Vault 4, and the FDCs, as illustrated in Figure 2.4-1 (2009 PA; WSRC-STI-2008-00244). The closure caps are designed to (1) provide physical stabilization, (2) limit infiltration by promoting runoff, evapotranspiration, and the shedding of water around the vaults and disposal cells and, (3) act as an intruder deterrent. The closure cap design (WSRC-STI-2008-00244) consists, from the surface downward, of topsoil and backfill, an erosion barrier of coarse stone, a middle backfill layer, an upper drainage layer over a high-density polyethylene (HDPE)/Geosynthetic clay liner (GCL) composite layer, and additional backfill (Figure 2.4-2). In addition, several geotextile filter fabric layers in the cover, while not directly influencing flow, serve to separate some of the engineered layers (e.g., filter fabric above the drainage layers helps to slow drainage layer clogging with soil particles from layers above). In addition to the site closure caps, a lower lateral drainage layer will be placed directly over each disposal unit. For the FDCs, the lower lateral drainage layer will be underlain by a second HDPE/GCL composite layer (Figure 2.4-2). The maximum slope of the upper surface of the closure caps is 1.5%. The side slopes, with maximum slopes of 10.5%, will be covered with large stones for erosion control.

After installation of the closure caps an initial 100 year institutional control period will begin, during which active maintenance will be conducted to prevent pine forest succession and to repair any significant erosion. In the PA, DOE does not assume any active maintenance will continue after the 100 year period post site closure. Although the closure caps are assumed to be initially vegetated with a persistent grass, such as Bahia (*Paspalum notatum*), the vegetation is assumed to evolve into a pine forest. The grass may later be replaced with bamboo, if DOE determines that bamboo will slow invasion of loblolly pine trees. DOE plans to slow the invasion of loblolly pines, if possible, because loblolly pines are expected to result in the degradation of the upper HDPE/GCL composite layer due to root depths up to 3.7 m (12 ft). DOE does not expect Loblolly pines to disrupt deeper layers (i.e., the lower hydraulic barrier layers) because the roots typically do not grow to that depth.

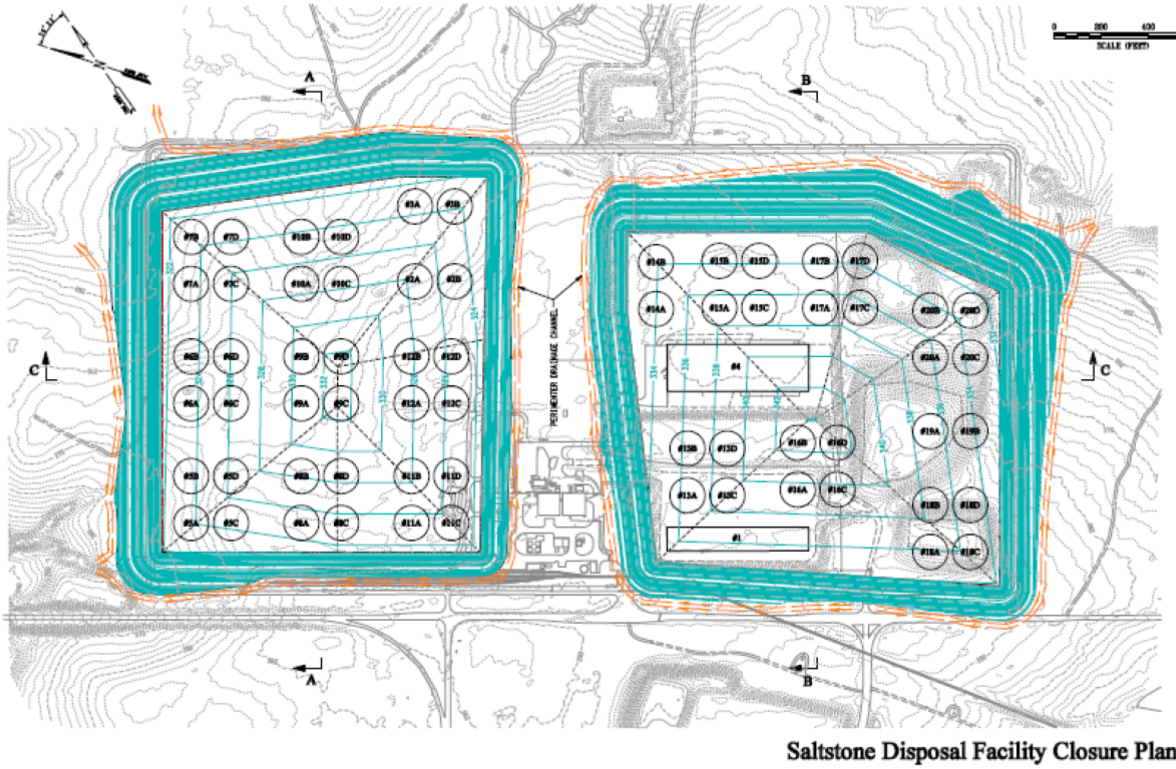

Figure 2.4-1: Conceptual illustration of the SDF Closure Cap (WSRC-STI-2008-00244; Figure 5)

Figure 2.4-2: Conceptual illustrations of the components of the SDF closure caps and additional engineered layers above the disposal units (2009 PA; Figure 3.2-20)

2.4.1.1 *Infiltration*

The potential for advective transport of radionuclides from saltstone is reduced by limiting infiltration. DOE used the HELP model (EPA-600-R-94-168a, EPA-600-R-94-168b) to estimate infiltration rates through the closure cap considering processes that degraded or altered the properties of the materials used to construct the closure cap, such as root penetration of the HDPE geomembrane and clogging of the drainage layer. Simulations of infiltration through the closure cap were based on a 100-year synthetic weather database for Augusta, Georgia that was modified with SRS-specific precipitation data. The annual precipitation in this data set ranged from 76 to 175 cm/yr [30 to 69 in/yr]. In both Case A and Case K, infiltration rates through the closure cap (i.e., from the surface through the foundation layer in Figure 2.4-2) reached steady-state conditions of 26.9 cm/yr [10.6 in/yr] at approximately 5,500 years after site closure. This steady-state value is approximately the same as the natural infiltration rate in the absence of a closure cap (WSRC-STI-2008-00244). Estimated average annual net infiltration through the closure cap was then used as an upper boundary condition to the SDF vadose zone model, which included the lower backfill layer, lower lateral drainage layer, HDPE and Geosynthetic Clay Liner (for the FDCs only), the disposal unit roofs, walls, and floors, and the saltstone waste form, as well as the underlying and surrounding backfill soil.

The SDF "no closure cap" sensitivity analysis case indicates that the closure cap (i.e., layers from the surface to the Foundation Layer in Figure 2.4-2) is expected to delay the release of radionuclides from the SDF and reduce peak radionuclide doses within 10,000 years by approximately 25% as compared to Case A (2009 PA; Tables 2-13 and 2-14 and Figure 5.6-75). Although this case is called the "no closure cap" case, it does include the effects of the lower lateral drainage layer.

2.4.1.2 *Erosion Control*

The ability of the closure cap to reduce infiltration and deter intrusion for long time periods is dependent on erosion controls. The closure cap design consists of a minimum of 3 m [10 ft] of material above the backfill covering the vaults and disposal cells. Typically, agricultural and resident intruder scenarios include a nominal excavation depth of 3 m [10 ft]. Therefore, proper design, construction, and performance of the erosion barrier should limit surface water erosion and direct contact of the waste by potential inadvertent intruders. In addition, DOE assumes that the erosion barrier will prevent animal intrusion into the lower layers. DOE does not assume the erosion layer will prevent root penetration from pine trees.

The erosion barrier is designed to limit erosion of the underlying cap layers, however the vegetative cover, topsoil, and upper backfill layer, which provide water storage and promote evapotranspiration, are susceptible to erosion. DOE performed scoping-level calculations to inform its design of the cap layers to prevent gully formation and ensure that soil loss would not impact closure cap performance.

DOE evaluated the physical stability of the closure cap with respect to a probable maximum precipitation (PMP) event (WSRC-STI-2008-00244) as is consistent with NUREG-1623

(NRC, 2002b). The PMP is defined as the theoretically greatest depth of precipitation that is physically possible during a given period of time over a given area at a particular geographic location. Based on the PMP, the design criteria for the vegetative cover, erosion barrier, side slopes, and toe of the side slopes were evaluated. Although the methodology presented in NUREG-1623 addresses a 1,000 year timeframe, DOE stated in an RAI response for the FTF review that the SRS-specific PMP event provides assurance of closure cap stability against gullying for the 10,000 year compliance period (SRR-CWDA-2009-00054). In response to NRC staff's concern with the long-term performance of the side slopes (NRC, 2010b; IEC-3), DOE indicated that slumping of the side slope and down-slope creep of the riprap were not evaluated explicitly. DOE also indicated that the conservatisms in its approach to designing the side slopes are expected to account for any potentially adverse effects of additional degradation mechanisms (SRR-CWDA-2010-00033). In addition, DOE indicated that slope stability will be considered as part of the closure cap final design.

Although the slopes and slope length for the topsoil and upper backfill layers of the closure caps have been designed to prevent gully formation due to a PMP event, these layers are still assumed to be subject to erosion (WSRC-STI-2008-00244). The projected long-term topsoil loss was determined according to the Universal Soil Loss Equation for both vegetative cover conditions (i.e., bahia grass and pine forest). DOE predicted approximately 3.3 cm [1.3 in] of soil loss for the topsoil and no reduction in the upper backfill layer within 10,000 years (WSRC-STI-2008-00244; Section 7.2).

Riprap for the integrated drainage system ditches has not yet been sized due to the early phase of the project and resultant lack of a detailed closure cap drainage system layout. Riprap material for the erosion barrier, side slope, and toe of the side slope will be selected from local granite or mylonitic quartzite quarries.

2.4.3 NRC-Evaluation – Infiltration and Erosion Control

The designs for long-term infiltration and erosion control have different objectives and are subject to different degradation mechanisms. Acceptability of a design for one does not ensure that an acceptable design has been achieved for the other (e.g., designing the vegetative cover and topsoil to promote runoff may reduce infiltration in the near term, but may increase long-term erosion).

2.4.3.1 NRC-Evaluation – Infiltration

Dose typically is sensitive to infiltration because infiltration is directly related to the flux of contaminants into the groundwater. However, the assumed performance of a series of additional engineered barriers limits the sensitivity of the overall performance of the SDF to closure cap performance (Section 2.13.3). In particular, due to the hydraulic performance of (1) the lower lateral drainage layer and disposal unit roofs; (2) saltstone grout; and (3) the composite HDPE/GCL layer above the FDCs, approximately 99.9% of the infiltrating water is modeled as being shed around the disposal units from 1,000 to 10,000 years (SRR-CWDA-2011-00044).

DOE's "no closure cap" sensitivity analysis is based on the assumption that the upper HDPE/GCL composite layer, the upper drainage layer, and the erosion control layer have the hydraulic properties of backfill soil at the time of site closure (2009 PA; Section 5.6.6.2). As a result, the infiltration rate is fixed at 41.8 cm/yr [16.5 in/yr] as a boundary condition into the PORFLOW™ vadose zone model (i.e., flow to layers below the foundation layer in Figure 2.4-2). However, the infiltrating water is still effectively shed around the disposal units (e.g., greater than 99.8% of the infiltrating water is predicted to be shed around the disposal units 8,000 years after closure). Consequently, the peak dose shifted only slightly earlier in time and the magnitude increased by less than 50%. If the assumptions regarding the hydraulic performance of the lower lateral drainage layer, disposal unit roofs, HDPE/GCL composite layer (for the FDCs only), and the saltstone grout are found to be optimistic during monitoring, the closure cap would become a more risk significant barrier.

The upper and lower lateral drainage layers are designed to divert a significant portion of the infiltrating water away from the underlying disposal units. A geotextile filter fabric will be placed on top of the drainage layers to provide filtration between the underlying sand and the overlying backfill layers. DOE assumed that the degradation of the drainage layers (i.e., a reduction in hydraulic conductivity) will be controlled by colloidal infilling of the pore spaces within the drainage layers from the overlying backfill. As there is limited data regarding the service life of filter fabrics, NRC requested additional information about potential infilling of the drainage layers with larger particles the resulting potential decrease in drainage layer hydraulic conductivity (NRC, 2010i; IEC-8). The NRC staff was concerned that decreased hydraulic conductivity could limit the ability of the lateral drainage layer to shed water and lead to more water reaching saltstone than is assumed in the PA. In response, DOE provided a flow budget showing that the lower lateral drainage layer significantly limits infiltrating water (e.g., approximately 99,9% of the water is modeled as being shed around Vaults 1, 4, and the FDCs at 10,000 years (SRR-CWDA-2011-00044; IEC-8). DOE also conducted a sensitivity analysis to assess the risk significance of filter fabric by doubling the modeled infilling of the drainage layers (SRR-CWDA-2011-00044; PA-10). DOE's analysis demonstrated that the shedding of the water around the disposal units is much more sensitive to assumptions about the disposal unit roofs and the HDPE/GCL composite layer above the FDCs than it is to assumptions about drainage layer infilling. Specifically, for Vault 4 and for an FDC, doubling the infilling of the drainage layers increased the Darcy velocity through saltstone by approximately a factor of 3 or less at 10,000 years in Case A, and caused smaller increases in Case K. In comparison, the Darcy velocity through saltstone at 10,000 years after closure is between a factor of 100 and 1,000 times greater in Case K than it is in Case A. The NRC staff agrees with DOE's assessment that the degradation of disposal unit concrete and saltstone is more risk significant than the degradation of the drainage layers. However, a factor of two to three in difference in the Darcy velocity is more significant to a compliance assessment if predicted doses approach the relevant dose limit.

As discussed in more detail in Chapter 5, the NRC staff determined the potential for differential settlement due to static settlement of the disposal units or the formation of sinks under the SDF may not have been fully evaluated. Differential settlement of disposal units could affect the performance of the closure cap and lower lateral drainage layer. Disruption of the upper

HDPE/GCL layer is not expected to affect system performance in ways not already addressed by increasing predicted infiltration to near natural infiltration rates. However, because the lower lateral drainage layer is modeled as performing well for the entire performance period, disruption of the disposal unit roofs or HDPE/GCL layers on the FDC roofs could cause the system behavior to deviate significantly from model predictions.

The assumed saturated hydraulic conductivity for the foundation layer of $1.0\text{x}10^{-6}$ cm/s ($3.3\text{x}10^{-8}$ ft/s) constrains the HELP model to a maximum infiltration rate of 31.6 cm/yr (12.4 in/yr) through the engineered cover (i.e., from the surface to the foundation layer). Consequently, the long-term steady-state infiltration rate, which ranges from 13.7 to 31.6 cm/yr (5.4 to 12.4 in/yr) is less than the background value of 37.7 cm/yr (14.9 in/yr). DOE indicated that the assumed saturated hydraulic conductivity for the foundation layer is appropriate, as the value is a design specification that is achievable for soil-bentonite blends (SRR-CWDA-2010-00033). An increase from DOE's assumed long-term infiltration rate to the background value would result in an increase in radionuclide release and a decrease in the timing of the chemical transitions in the saltstone waste form.

DOE modeling indicates that saturated conditions will occur above the upper composite layer in the closure cap. An initial hydraulic head of 9.55 cm (3.76 in) is predicted to develop on top of the HDPE geomembrane and increase until 5,400 years after closure, when it is predicted to range from 99.0 to 100 cm [39.0 to 40.0 in] until 10,000 years or more after closure (SRR-CWDA-2011-00044). DOE stated that conservative modeling assumptions (e.g., depth of evapotranspiration zone and degradation of the lateral drainage layer) resulted in estimates of head on the HDPE geomembrane that are bounding and conservative. Should the buildup of hydraulic head occur, DOE does not believe it would adversely impact the physical stability of the closure cap, vegetation, erosion, or the performance of the composite layer. Based on limited model support, it is difficult to assess: (1) the likelihood of hydraulic head buildup within the cover; or (2) its potential implications for closure cap performance. A more realistic representation of infiltration and saturation within the proposed closure cap is needed to assess the potential for buildup of hydraulic head. If an analysis containing a more realistic representation determines that the buildup of hydraulic head is realistic, an explicit evaluation of the physical stability of cover materials under this condition will be needed.

In addition, when the cap is constructed, it is important for there to be a robust quality assurance (QA)/quality control (QC) program. In particular, the ability of the composite hydraulic barrier to limit infiltration early in the compliance period is dependent on construction quality.

DOE indicated that the closure cap has a minimal impact on peak doses when other barriers to flow perform as designed (i.e., the lower lateral drainage layers above each disposal unit and the intended low permeability of disposal unit roofs and saltstone waste form). However, the NRC staff expects closure cap performance to be more risk significant if these other barriers do not perform as designed (Section 2.13.3). The long-term performance of the infiltration barrier is expected to be affected by several sources of uncertainty, including (1) degradation of the drainage layers and underlying disposal unit roofs or HDPE/GCL composite layers (for the

FDCs), (2) potential effects of differential settlement, (3) hydraulic degradation of the foundation layer, and (4) effects of hydraulic head buildup within the closure cap. The NRC staff concludes that more model support is needed for the long-term infiltration-limiting performance of the closure cap and infiltration-limiting engineered layers above the disposal units. Because of the importance of limiting the infiltration to system performance, the NRC staff will track these issues as monitoring factors.

2.4.3.2 NRC-Evaluation – Erosion Control

Erosion control is necessary to ensure that a thick cover of soil is maintained over the waste for protection of inadvertent intruders and to provide suitable conditions for the vegetative cover. To mitigate the potential effects of erosion by surface water, erosion protection designs must be based on an appropriately conservative rainfall event. DOE's determination of the PMP event and the corresponding design criteria for the vegetative layer, erosion barrier, side slopes, and toe of the side slopes were consistent with NUREG-1623 (NRC, 2002b). This guidance document specifically addresses a 1,000-year timeframe rather than the 10,000-year compliance period. DOE stated that the SRS-specific PMP has a low probability of occurrence and is a bounding event, thereby providing assurance of physical stability of the closure cap design for the 10,000-year compliance period (SRR-CWDA-2011-00054). NRC staff determined that DOE's estimates for the PMP are reasonable and that the probability of such an event being equaled or exceeded is very low. Accordingly, the PMP is considered by NRC staff to provide a reasonable design basis.

Long-term maintenance of the topsoil and vegetative cover is important to closure cap performance as the average evapotranspiration rate (82.7 to 85.4 cm/yr [32.6 to 33.6 in/yr]) dominates the modeled water balance distribution for SRS precipitation (125 cm [49.1 in]). An evaluation of the cumulative effects from precipitation events over long time periods with respect to gully formation is needed to support predictions of long-term performance of the topsoil and vegetative layers.

The NRC staff concludes that the closure cap, as designed, can provide adequate long-term erosion protection. However, because the cap will not be built until SDF closure, it remains for DOE to demonstrate that certain aspects of the designed performance can be achieved (e.g., evaluation of an acceptable rock source, the ability of an integrated drainage system to accommodate design features). Although the design will not be made final until closer to the time of site closure, verification that certain designed features can be implemented as designed is needed in advance of site closure to allow sufficient time to change the closure cap design or the assumptions regarding long-term erosion protection, if necessary. These modifications may be important in the overall evaluation of closure cap performance.

2.5 Disposal Unit Design and Performance

2.5.1 Design and Construction of the Disposal Units

2.5.1.1 Vault 1

Vault 1 is a rectangular, reinforced concrete disposal unit approximately 180 m (600 ft) long, 30 m (100 ft) wide, and 8.2 m (27 ft) high. It contains two units, each 30 m by 90 m (100 ft by 300 ft), which are separated by an 8 cm (3 in) gap. Each unit has three cells, each approximately 30 m by 30 m (100 ft by 100 ft). In total, Vault 1 comprises six cells total (Cells A to F), each 30 m by 30 m (100 ft by 100 ft). Vault 1 has a poured-in-place concrete roof with a minimum thickness of 15 cm (6 in). The walls are 46 cm (18 in) thick and the floor slab is 61 cm (24 in) thick. The walls and floor are constructed of reinforced concrete, using slightly different concrete formulations (Table 2.5-1). Underlying the floor slab is a 10 cm (4 in) thick concrete working slab. Vault 1 was built above grade and remains above grade, but it will be buried and covered with a closure cap at the time of site closure.

Table 2.5-1: Vault 1 Concrete Formulations

Ingredient	Quantity (lb/yd^3) [1]		
	Working Slab	Floor and Walls	Roof
ASTM C 150 Type II Cement	413	419	400
Grade 120 ASTM C 989 Blast Furnace Slag	0	278	0
Type F ASTM C 618 Fly Ash	73	0	70
ASTM C 33 Sand	1,356	1,133	1,149
No. 67 ASTM C 33 Aggregate (maximum 1.9 cm [0.75 in])	1,698	1,798	1,900
Water (maximum)	272	268	292
Water to Cementitious Material Ratio	0.56	0.385	0.62
Minimum Compressive Strength at 28 days	2,000 psi	4,000 psi	3,000 psi

2009 PA; Table 3.2-1
[1] 1.0 lb/yd^3 = 0.59 kg/m^3

Vault 1 Cells A, B, and C have been filled and are covered by a permanent roof. Vault 1 Cells D, E, and F are unfilled and have no roof. Vault 1 Cell A was filled with saltstone in 1994. While a temporary concrete roof was being installed, water was trapped between the saltstone and disposal unit walls (ESH-WPG-2006-00132). Pressure from the trapped water caused the wall to crack, at which point DOE observed liquid weeping from Vault 1 Cell A onto the ground. Vault 1 Cells B and C were closed in 1998. While a concrete roof was being installed over these cells, water also was trapped between the saltstone and disposal unit walls.

2.5.1.2 Vault 4

Vault 4 is a rectangular, reinforced concrete disposal unit, approximately 180 m (600 ft) long, 60 m (200 ft) wide, and 9 m (30 ft) high. It is divided into two units, each 60 m by 90 m (200 ft by 300 ft), which are separated by an 8 cm (3 in) gap. Each unit has 6 cells, each approximately 30 m by 30 m (100 ft by 100 ft). Thus, in total, Vault 4 comprises twelve cells (Cells A to L), each 30 m by 30 m (100 ft by 100 ft). Vault 4 Cells B to L are being filled with saltstone, whereas Cell A has drums of low-activity waste encapsulated by clean grout. Like Vault 1, Vault 4 was built and remains above grade, but will be buried and covered with a closure cap at site closure.

Table 2.5-2: Vault 4 Concrete Formulations

Ingredient	Quantity (lb/yd^3) [1]		
	Working Slab	Floor Slab and Walls to 7.6 m (25 ft)	Walls above 7.6 m (25 ft) and Roof
ASTM C 150 Type II Cement	413	419	466
Grade 120 ASTM C 989 Blast Furnace Slag	0	278	0
Type F ASTM C 618 Fly Ash	73	0	62
ASTM C 33 Sand	1,356	1,133	1,190
No. 67, ASTM C 33 Aggregate (maximum 1.8 cm [0.75 in])	1,698	1,798	1,800
Water (maximum)	273	254	296
Water to Cementitious Material Ratio	0.56	0.36	0.56
Minimum Compressive Strength at 28 days	2,000 psi	4,000 psi	4,000 psi

2009 PA Table 3.2-2

[1] 1.0 lb/yd^3 = 0.59 kg/m^3

Vault 4 was constructed in 1988 and initially included a 10 cm (4 in) thick concrete working slab, 0.6 m (2 ft) thick reinforced concrete floor slab, and 46 cm (18 in) thick reinforced concrete walls. Construction joints are located on 9 m (30 ft) centers in the floor slab and walls (T-CLC-Z-00006). The concrete formulations used for the Vault 4 floors and walls are listed in Table 2.5-2. In 1996, prior to filling the disposal unit cells with grout, a permanent roof (poured-in place concrete over steel decking) was installed to reduce potential rainwater intrusion. Nine equally spaced roof support columns (25 cm [10 in] diameter concrete-filled carbon steel pipes) also were installed in each of Vault 4 Cells B to L. The concrete roof was painted with a heat-dissipating coating, but no waterproof coat was applied. The roof joints allowed rainwater to enter the disposal unit cells, which DOE believes is responsible for wall cracking similar to that in Vault 1. Specifically, DOE concluded the cracking in the Vault 1 and 4 walls is due to hydrostatic pressure buildup of liquids in the annuli between the saltstone grout and the walls (SRR-CWDA-2010-00033, comment VP-1; C-CLC-Z-00016). Subsequently, the joints were sealed and sheet drain systems were installed in eight of the disposal unit cells to remediate the liquid accumulation (ESH-WPG-2006-00132).

2.5.1.3 Future Disposal Cells

As of 2009, DOE planned to construct 64 FDCs at the SDF (2009 PA). Each disposal unit is a separate 46 m (150 ft) diameter cylindrical tank primarily made of sulfate-resistant concrete. The walls are made of pre-cast panels of sulfate-resistant concrete, a steel shell diaphragm, reinforcing bars, pre-stressing wires, and shotcrete fill. The reinforcing steel, pre-stressing wire, and steel shell diaphragm are made of carbon steel. The shotcrete is intended to protect the reinforcing bars and pre-stressing wires from corrosion.

Construction on FDCs 2A and 2B began in 2009 and the cells began testing for water tightness beginning 2010 (Section 2.5.3.1). Both FDCs are 46 m (150 ft) in diameter and have an interior height of 6.7 m (22 ft) (7.2 m [23.5 ft] at the center). The current design for the remaining 62 FDCs is similar to the design of 2A and 2B; a few notable differences are mentioned in this section.

The concrete used in the FDC walls and floor serves as a chemical as well as a physical barrier to radionuclide release. The concrete includes blast furnace slag to create chemically reducing conditions to reduce the mobility of Tc-99 through the concrete. DOE stated that two potential FDC concrete mixes have been examined and produced acceptable results, but that further concrete mix testing is expected to be conducted during the operational phase of the SDF (2009 PA). One of these formulations, which may be used for the FDC floors and walls, is provided (Table 2.5-3). The concrete attributes DOE considers important to the FDC are (i) low hydraulic conductivity, (ii) high pH, (iii) low E_h, (iv) high degradation resistance, and (v) high sulfate attack resistance.

FDC construction begins with pouring a 10 cm (4 in) thick concrete base, called the lower mud mat, on site. The lower mud mat is covered with a geosynthetic clay liner (GCL), composed of a layer of bentonite between two geotextile layers (Section 2.5.2.2). The GCL is then covered with a 2.5-mm (100-mil) HDPE liner, onto which an upper mud mat, made of Class III sulfate-resistant concrete, is poured. The 10 cm (4 in) thick upper mud mat is designed to protect the HDPE liner during construction. Reinforcing steel bars are then put in place and a 20 cm (8 in) thick floor, also composed of Class III sulfate-resistant concrete, is poured.

Cell walls are constructed from pre-cast sulfate resistant concrete panels with a minimum thickness of 20 cm (8 in). The concrete panels are poured onto the steel diaphragm panels after reinforcing steel bars are put in place. DOE expects that rust that formed on the inside and outside (NRC, 2009b; photos 22, 23, 25, and 44 - 47) of the steel diaphragm panels during construction will not accelerate corrosion cracking because the steel will be passivated by the high pH environment created by the concrete wall, on the inside, and a layer of shotcrete on the outside of the steel shell.

Table 2.5-3: Potential FDC Concrete Formulation

Ingredient	Quantity (lb/yd^3) [1]
ASTM C 150 Type V Cement	213
Grade 100 ASTM C 989 Blast Furnace Slag	284
ASTM C 1240 Silica Fume	47.3
Type F ASTM C 618 Fly Ash	165.7
ASTM C 33 Sand	911
No. 67 Granite ASTM C 33 Aggregate	1,850
Water (maximum; gal/yd^3)	32.3
Grace WRDA 35 (oz.cwt c+p)	5
Grace Darex II (oz/cwt c+p)	0.4 to 0.5
Grace Adva 380 (oz/cwt c+p)	3 to 4
Maximum Water to Cementitious Material Ratio	0.38

2009 PA Table 3.2-3
[1] 1.0 lb/yd^3 = 0.59 kg/m^3

The wall panels are joined with steel closure strips. A polysulfide material is injected into the area where the closure strip meets the steel diaphragm panels to form a watertight seal. Joints between the panels are filled with sulfate resistant concrete. After the wall panels are erected and joined, the structure is covered with a 2.5 cm (1 in) minimum layer of concrete. Pretensioning wires are then wrapped around the structure and tensioned and another layer of shotcrete is applied to protect the pretensioning wires. The walls are then entirely covered with 2.5 mm (100 mil) HDPE sheets, which are sealed to each other and to the HDPE layer under the upper mud mat.

Inside an FDC, the floor and walls are coated with a mat-reinforced epoxy-novolac thermosetting lining. The liner is not expected to provide a long-term hydraulic barrier, but is designed to protect the floor and wall concrete from sulfate in saltstone bleed water during the initial saltstone curing and for several decades afterward. DOE has included sheet drains between the liner and saltstone waste form to remove saltstone bleed water and any infiltrating water from the disposal units. The sheet drain is fastened to the floor with anchor bolts. Removal of bleed water and infiltrating water from the FDCs is intended to protect the FDCs from the type of hydraulic loading that caused fracturing of Vault 4 walls. The sheet drain system is designed to function during the operational period. Drain pipes leading from the sheet drain system will be filled with grout at the time of site closure (NRC, 2010i; IN-4).

The 20 cm (8 in) thick FDC roof is supported by 48 36-cm (14-in) diameter columns made of carbon steel-reinforced sulfate-resistant concrete. The roof includes penetrations for ventilation, sheet drain, thermocouple trees, closed circuit television, and grout poring. During the operational period, these penetrations will be sealed with neoprene gaskets. The external surface of the roof will be covered with an HDPE/GCL composite layer once the FDC is filled (Section 2.5.2.2).

After testing cells 2A and 2B for water tightness, DOE made a number of design modifications to the FDCs that occurred after the PA was issued (WB00001K-058). In the second response comment VP-3 (SRR-CWDA-2011-00044), DOE discussed modifications to cells 2A and 2B that included cutting the anchor bolt penetrations flush with the floor, applying an interior coating, installing an additional 36 cm (14 in) of sulfate resistant concrete above the coating layer, and installing an exterior sulfate resistant curb around the perimeter of the disposal unit floor. In addition, DOE discussed that significant design modifications to the disposal units have been implemented based on the lessons learned from the hydrostatic testing (e.g., future disposal units do not include anchor bolt penetrations in the floor). Additional discussion of the hydrostatic testing and design modifications is provided in Section 2.5.3.1.

2.5.2 Modeling of Disposal Unit Performance

2.5.2.1 Cementitious Material Performance

The initial physical properties assumed in the PA for the concrete in Vault 1, Vault 4, and the FDCs are presented in Table 2.5-4. The Vault 1 and 4 walls are modeled with an increased initial hydraulic conductivity to represent the cracks that have been observed in these vault walls. As discussed in more detail in Section 2.7, DOE used moisture characteristic curves along with the hydraulic conductivity values listed below in the modeling of flow through the system.

DOE postulated that the physical degradation of disposal unit concrete will be dominated by sulfate attack from exposure to sulfate ions remaining in the saltstone porewater after curing. Sulfate reacts with cement paste and forms ettringite, an expansive mineral phase often associated with concrete spalling or cracking. DOE used the proprietary reactive transport code STADIUM® to simulate ettringite formation in disposal unit concretes exposed to solutions with three different sulfate concentrations (low, medium, and high sulfate concentration). The results of these simulations were used to derive an empirical equation for the position of the ettringite front as a function of sulfate concentration. Concrete degradation was estimated by assuming concrete ahead of the ettringite front retained its initial properties whereas concrete behind the front is cracked or otherwise degraded. Degradation due to sulfate attack was represented in the SDF performance assessment by increases in concrete hydraulic conductivity and diffusion coefficient. Completely degraded concrete was assumed to have properties similar to those of backfilled soil and, thus, not to be a barrier to contaminant release in comparison to the environment surrounding the disposal unit.

Table 2.5-4: Initial Disposal Unit Concrete Properties Assumed by DOE PA Cases and Cases K, K1, K2

	Material Porosity (%)	Dry Bulk Density (g/cm^3)	Particle Density (g/cm^3)	Hydraulic Conductivity (cm/s)	Effective Diffusion Coefficient (cm^2/s)
Medium quality concrete (Vault 1 roof)	14.5	2.20	2.57	5.0×10^{-9}	1.0×10^{-7}
Medium quality concrete (Vault 4 roof)	13.6	2.21	2.56	5.0×10^{-9}	1.0×10^{-7}
High quality concrete (Vault 1 and 4 base)	12.0	2.24	2.55	3.1×10^{-10}	5.0×10^{-8}
Fractured walls (Vault 1 and 4)	12.0	2.24	2.55	1.7×10^{-1} (1)	5.0×10^{-8}
Low quality concrete (lower mud mats of FDCs)	21.1	2.06	2.61	1.0×10^{-8}	8.0×10^{-7}
High quality concrete (FDCs)	11.0	2.22	2.49	9.3×10^{-11}	5.0×10^{-8}

Based on 2009 PA; Table 4.2-16
(1) In Case K, an initial hydraulic conductivity of 1×10^{-6} cm/s was used for Vault 1 and 4 walls

In Case A, sulfate degradation is assumed to cause the hydraulic conductivity of the Vault 1 roof and Vault 1 and 4 floors and walls to increase from their initial values by a factor of approximately 3 or less in 20,000 years. Sulfate degradation also is assumed to increase the effective diffusivity of the Vault 1 roof and Vault 1 and 4 floors and walls by a factor of approximately 4 or less in 20,000 years. However, sulfate attack traverses the full thickness (4 inches [10 cm]) of the Vault 4 concrete roof in 10,000 years, after which the Vault 4 roof hydraulic conductivity and effective diffusivity become equal to that of backfill soil (i.e., a hydraulic conductivity of 4.1×10^{-5} cm/s and a diffusivity of 5.3×10^{-6} cm^2/s [PA; Table 4.2-14]). Also, in Case A, sulfate attack causes the hydraulic conductivity and effective diffusivity of the FDC concrete to increase from their initial values by a factor of approximately 6 or less in 20,000 years (for the roof and floor) or 10,000 years (for the walls). The FDC wall hydraulic conductivity is modeled as equal to that of backfill soil after approximately 17,000 years. More detail on the modeled changes to the hydraulic conductivity and diffusivity with time is shown in Figures 4.2-36 to 4.2-40 of the PA. Additionally, Appendix E in SRNL-STI-2009-00115 provides more details on the assumed hydraulic conductivities at different time steps for Cases A - E.

Some of the deterministic cases modeled by DOE considered alternate assumptions for the disposal unit concrete degradation. In the 10x Sulfate Attack Case, DOE assumed that the diffusion coefficient used to predict the amount of sulfate attack was increased by a factor of 10, which led to a shorter modeled time to complete failure for the disposal unit concrete. In the No Sulfate Attack Case, the disposal unit properties were assumed to not degrade and to remain constant over time. In the Synergistic Case, the roof and floor of Vault 1 and 4 and the FDC concrete degrade hydraulically to soil properties 500 years after closure. As in Case A, the Vault 1 and 4 walls are assumed to be initially fractured (represented in the model as increased hydraulic conductivity and diffusivity).

Additionally, in Case K, the Vault 1 and 4 walls were assumed to be initially degraded such that the hydraulic conductivity was set equal to 1×10^{-6} cm/sec, which is similar to the hydraulic conductivity of the upper vadose zone soil (PA; Table 4.2-14). The initial hydraulic conductivity of the other cementitious materials in the disposal units was assumed to be the same as described in Table 2.5-4. In Case K, the degradation of the disposal unit concrete was modeled non-mechanistically with increases in the hydraulic conductivity and diffusivity. Except for the Vault 4 roof and walls, the disposal unit concrete was modeled with hydraulic conductivity increasing with a log-linear relationship to a final value of 1×10^{-6} cm/s at 10,000 years and diffusivity increasing from the values provided in Table 2.5-4 to a value of 5×10^{-6} cm^2/s at 10,000 years with a log-linear relationship. As previously noted, the Vault 4 walls were assumed to be degraded at closure. The Vault 4 roof was modeled similarly to the other disposal unit components but reached the degraded values (i.e., hydraulic conductivity of 1×10^{-6} cm/s and diffusivity of 5×10^{-6} cm^2/s) at 3,500 years after closure.

In Section 4.4.2 of the PA, DOE acknowledged that credible fast flow paths include large-scale cracks due to seismic events or differential settlement, shrinkage cracks, vertical sheet drains, degraded steel columns and roof supports, roof penetrations, and bleed water drain piping. To assess the effects of these and other disposal unit concrete degradation mechanisms that could result in the formation of fast flow paths, several alternative scenarios were considered in the DOE PA. DOE stated each case or scenario was meant to capture the outcome rather than predict the cause of the outcome. Cases B, C, and D assumed that a gap forms between saltstone and the walls of Vault 4 and of the FDCs at time zero due to the presence of a sheet drain at the wall and connecting breaches through the roof and floor. These gaps in Cases B and C function as fast pathways for water flow. In Case D, the gap between the disposal unit wall and saltstone is not a fast pathway because it is modeled as a capillary break (i.e., an impediment to flow). Case C contains additional fast flow paths through the disposal unit from the roof through the floor.

Chemical degradation of the vault and FDC concrete controls the chemical composition of pore fluids passing through the vault and FDC and ultimately influences the release of contaminants from the vault and FDC. DOE performed simulations using the Geochemist's Workbench geochemical code to estimate the chemical degradation of the concrete (SRNL-TR-2008-00283). The simulations estimated changes in E_h and pH as a function of the number of pore volumes of infiltrate water that reacts with the cementitious material. The concrete mineralogy that was used as input to the thermodynamic simulations was derived using a normative calculation that assigned chemical constituents determined by bulk chemical analysis to specific mineral phases. The derived vault and FDC concrete mineralogy is comprised of calcium-silicate-hydrate (CSH), hydrotalcite, gibbsite, quartz, hematite, and gypsum. To represent the reducing component of slag, pyrrhotite was included in the model, in an amount based on laboratory reducing capacity measurements (SRNS-STI-2008-00045). In the simulations, the concrete minerals were reacted with water having a composition based on an analysis of a sample from a water table monitoring well in the vicinity of the Saltstone Disposal Facility. The dissolved oxygen concentration in groundwater used in the simulations was 8 mg/L—higher than the 1.2 mg/L measured from the well water sample—based on assumed groundwater equilibrium with atmospheric oxygen.

For all radionuclides except Tc, sorption coefficient values (K_d values) were based on specific E_h and pH transitions in the disposal unit concrete. This process is similar to the process used to model radionuclides sorption in saltstone, which is described in more detail in Section 2.6 and 2.7. The results of the geochemical simulations described above were used to estimate the number of times the liquid in the pore volume of the disposal unit concrete must be replaced with infiltrating water to cause these E_h and pH transitions. The estimated pore volumes required to cause the transitions in the disposal unit concrete were the same in all cases. However, the times at which these transitions were predicted to occur were modified in certain cases based on the predicted flow through the disposal unit concrete (Table 2.5-5). The geochemical modeling indicated that the E_h transitions from reducing to oxidizing conditions (from –0.46 to +0.57 V) at 2,994 and 3,230 pore volumes for the Vault 1 and 4 concrete and the FDC concrete, respectively. The predicted pH gradually decreases from an initial value of 11.0 as the CSH phase in the cement dissolves. The modeled CSH phase in the vault concrete and FDC concrete is depleted at 3,660 and 4,206 pore volumes, respectively.

In the modeling of Tc transport in the PA cases (i.e., all cases except Cases K, K1, and K2), the E_h of the disposal unit concrete was modeled using a more sophisticated approach that DOE refers to as a shrinking-core model. In this model, the reducing capacity of the concrete is explicitly tracked in each grid cell in the PORFLOW™ model. When oxygen entering the cell by liquid phase transport consumes all of the reducing capacity, the Tc K_d for the cell is changed from a high value to a low value, greatly increasing the mobility of the Tc. Additional detail regarding the shrinking-core model is provided in Section 2.7.

In Cases K, K1, and K2, the E_h of the concrete was calculated externally to PORFLOW™ based on the diffusion of oxygen into the disposal unit concrete from fractures. The initial fracture spacing assumed in Cases K, K1, and K2 is presented in Table 2.5-6. In the RAI response describing Case K (SRR-CWDA-2011-00044, PA-8), DOE did not provide the final fracture spacing assumed for the disposal unit concrete or address whether the diffusion of oxygen from the exterior of the disposal units was included in the oxidation calculation. The fraction of the concrete oxidized with time was then used to determine an average K_d value for the concrete. A significant fraction of the disposal unit concrete is modeled as remaining reduced during a 20,000 year analysis time (Figure 2.5-1). Additional detail regarding the average-K_d model is located in Section 2.7.

Table 2.5-5: Chemical Transition Times for Cementitious Materials in Disposal Units

Cementitious Material	Vault 1		Vault 4		FDCs	
	E_h Transition (years)	pH Transition (years)	E_h Transition (years)	pH Transition (years)	E_h Transition (years)	pH Transition (years)
Case A						
Wall	20,781	21,043	15,519	16,018	16,334	16,753
Floor	>30,000	>30,000	>30,000	>30,000	22,498	23,274
Oxidized Concrete Case						
Wall	0	21,043	0	16,018	16,334	16,753
Floor	0	>30,000	0	>30,000	22,498	23,274
Synergistic Case						
Wall	0	500	0	500	500	500
Floor	0	500	0	500	500	500
Cases K, K1, K2						
Wall	8,297	9,545	9,219	10,671	7,756	8,232
Floor	7,740	8,227	>40,000	>40,000	7,970	8,462

2009 PA; Table 4.2-17 and Sections 5.6.6.5 and 5.6.6.6, and SRR-CWDA-2011-00044 Tables PA-8.8, PA-8.9, and PA-8.10

Table 2.5-6: Initial Fracture Spacing in Cases K, K1, and K2

Disposal Unit	Vault 1 (m)	Vault 4 (m)	FDCs (m)
Roof	10	10	41
Wall	10	10	7
Floor	10	10	41

From SRR-CWDA-2011-00044 Table PA-8.3

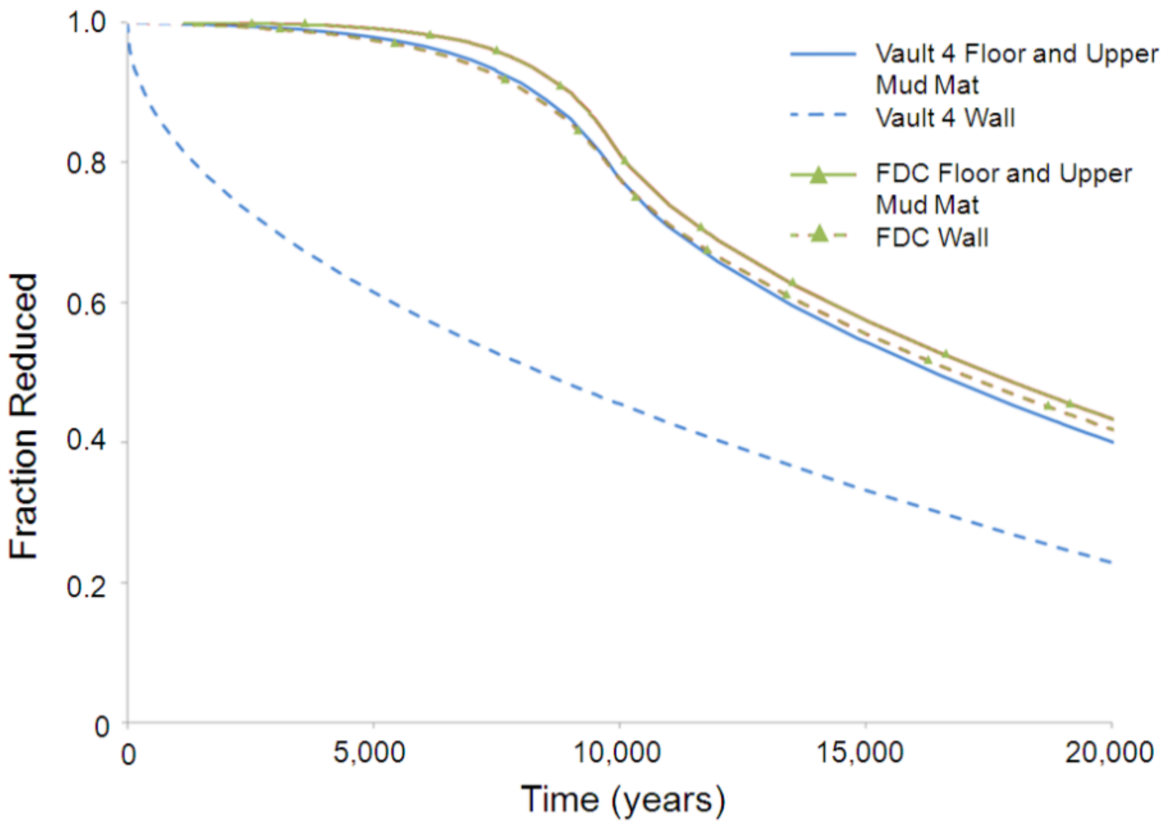

Figure 2.5-1: Fraction of Disposal Unit Concrete that Remains Reduced (calculated from K_d values in Case K PORFLOW™ files)

2.5.2.2 HDPE and HDPE/GCL Performance

DOE plans to use HDPE/GCL composite layers on the roof and below the floor (i.e., between the upper and lower mud mat) of the FDCs. In addition, DOE plans to cover the walls in a layer of 2.5-mm (100-mil) HDPE. Thus the HDPE will form an outer lining material for the FDCs[8]. The composite layer under the floor will extend 0.6 m (2 ft) up the side of the FDC walls and be welded to HDPE that is attached to the cylindrical sides of the FDCs. The HDPE on the sides will, in turn, be welded to the HDPE/GCL placed on the FDC roofs. The purpose of the HDPE surrounding the FDCs and the HDPE/GCL composite layer on the roof of the FDCs is to limit the flow of water and oxygen into the FDCs. The HDPE/GCL layer under the floors of the FDCs is designed to limit the flow of water out of the disposal units. In addition, DOE expects the HDPE to protect the FDC concrete from potentially corrosive soil components and carbonation. DOE also plans to include an HDPE/GCL composite layer in the closure cap. This section addresses the HDPE used to cover the FDCs and the HDPE/GCL composite layers on the roof and under the floor of the FDCs. The HDPE/GCL composite layer used in the closure cover is discussed in Section 2.4.

[8] DOE does not plan to use HDPE as an outer liner for either Vault 1 or Vault 4.

In the PA, DOE indicated Case A uses an initial hydraulic conductivity[9] of $2x10^{-13}$ cm/s and an initial diffusivity of $4x10^{-11}$ cm^2/s to represent the HDPE lining the FDC walls (PA page 217). The HDPE/GCL composite layer in the FDC roof and walls is modeled with an initial hydraulic conductivity[10] of $2.8x10^{-12}$ cm/s and an initial diffusivity of $1.2x10^{-10}$ cm^2/s (PA page 186). These values are modeled as degrading significantly during the first 1,000 years after closure and more slowly until 20,000 years after closure (Figure 2.5-2 and Figure 2.5-3). The same hydraulic conductivity and diffusivity used in Case A is used to model the HDPE in Cases B - E, except that Cases B, C, and D include breaks in the disposal unit roofs and floors (Section 2.5.2.1) that include breaks in the HDPE. Case K uses the same values of HDPE and HDPE/GCL hydraulic conductivity and diffusivity, except for apparent pathways through the floor. As previously discussed, the fractures in the Case K conceptual model are represented as a decrease in the bulk hydraulic conductivity and are not modeled explicitly in PORFLOW. Although the description of Case K does not include a discussion of fast pathways (SRR-CWDA-2011-00044), the PORFLOW™ output files indicate fast pathways are included in Case K from the sheet drains and columns extending through the floor, HDPE/GCL composite layer, and mud mats (i.e., to the unsaturated soil). DOE did not specifically discuss assumptions about HDPE performance in the other modeled case (i.e., any of the other cases described in Table 2.1).

To derive this projected HDPE performance (Figure 2.5-2), DOE considered various HDPE degradation mechanisms and selected those expected to cause the most significant degradation. DOE then estimated the number and size of defects that would form during the performance period (Table 2.5-7). Defects were classified as pinholes, holes, tears, small cracks, or cracks, depending on size. DOE then estimated hydraulic properties based on the total estimated area of defects per unit HDPE area (SRNL-STI-2009-00115, Table 23). In the PA, DOE indicates that the estimated degradation of the HDPE surrounding the FDCs is based on the same methods used to estimate HDPE degradation in the closure cap (Section 2.4). This procedure considered initial defects as well as degradation with time.

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[9] The reported initial hydraulic conductivity of HDPE is significantly different from the value of $5.9x10^{-10}$ cm/s used in the first modeling times step, as presented in SRNL-STI-2009-00115, Appendix E (unnumbered table on page 234). These values differ by more than the expected difference between the initial values and the average values for the first time step. Both of these values differ from the initial value depicted in Figure 2.5-2.

[10] The reported initial hydraulic conductivity of HDPE is different from the value of $2.2x10^{-11}$ cm/s used for the HDPE/CGL composite on the roof and under the floor in the first modeling times step, as presented in SRNL-STI-2009-00115, Appendix E (unnumbered table on pages 246 and 248). These values may differ because the table in SRNL-STI-2009-00115 presents the average for the first time step rather than the true initial value.

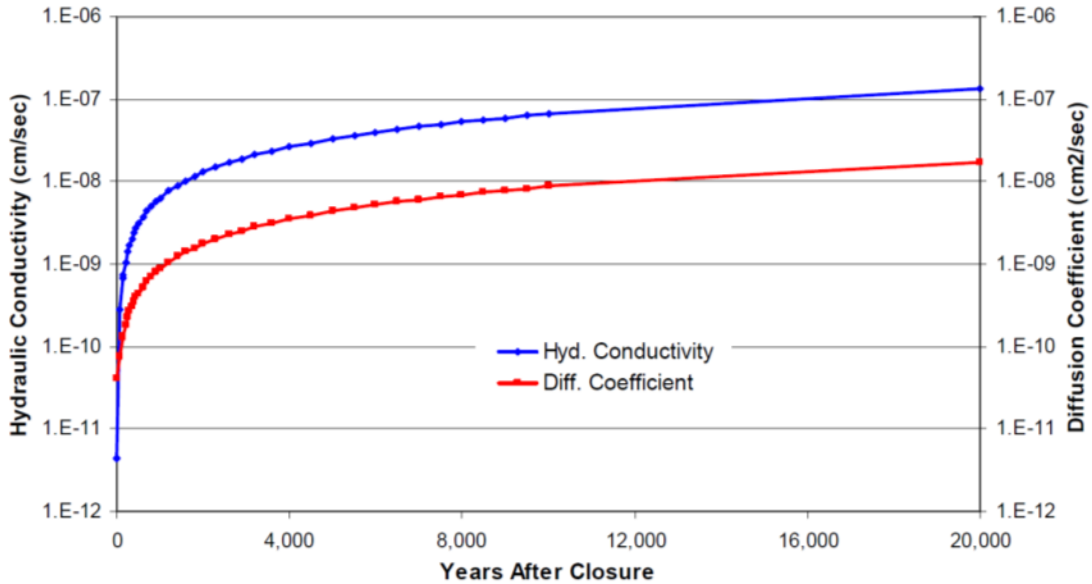

Figure 2.5-2: DOE Case A modeled Hydraulic Conductivity and Diffusion Coefficient for HDPE Lining of FDCs After Closure (2009 PA, Figure 4.2-42)

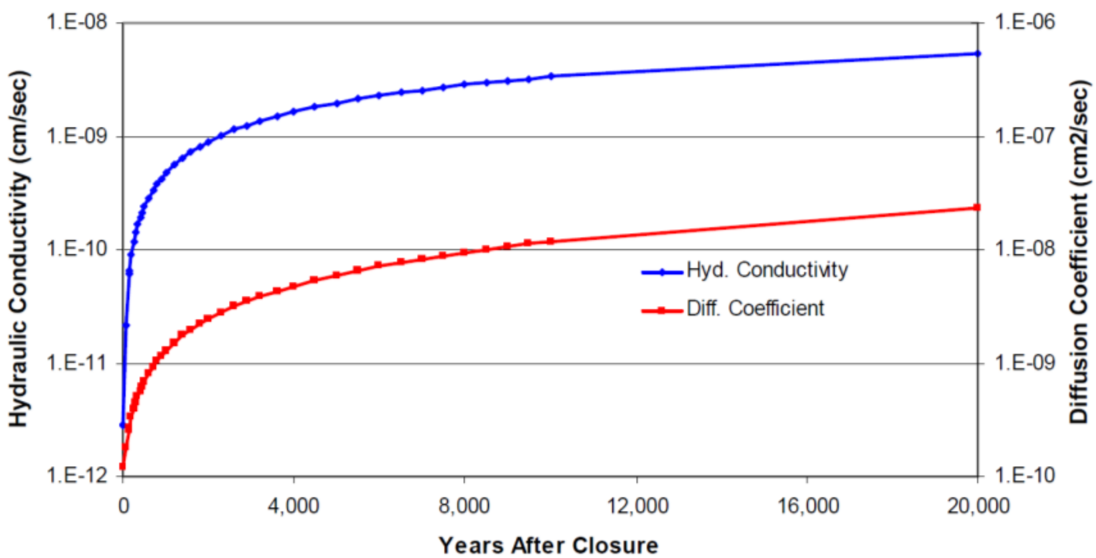

Figure 2.5-3: DOE Case A modeled Hydraulic Conductivity and Diffusion Coefficient for HDPE/GCL Composite Layer on the Roof and Below the Floors of the FDCs After Closure (2009 PA, Figure 4.2-19)

The estimate of initial defects is based on DOE's QA procedures for installing the HDPE. DOE provided a description of the QA procedures that would be used for the HDPE in the HDPE/GCL layer above the individual FDC roofs (WSRC-STI-2008-00244). These procedures include 100% visual inspection of the HDPE rolls prior to inspection and industry-recommended procedures for avoiding wrinkles. In addition, the QA procedures include 100% visual inspection of seams as well as 100% non-destructive vacuum or air pressure testing of seams. DOE also indicated that it would conduct periodic destructive testing of seams in accord with industry standards, and that the sites of destructive tests would be repaired and tested nondestructively after the repair. DOE did not indicate whether similar procedures will be used for the HDPE on the sides and in the floor of the FDCs.

Table 2.5-7: Cumulative Area of Holes in HDPE Encasing Disposal Cells

Year	Interpolated Cumulative Area of Holes[1]	Total Cumulative Hole Size[2] (mm^2/Hectare)	Fraction of HDPE Membrane with Holes[3]
0	550	5.50×10^{-8}	5.50×10^{-6}
100	46500	4.65×10^{-6}	4.65×10^{-4}
180	122220	1.22×10^{-5}	1.22×10^{-3}
220	157140	1.57×10^{-5}	1.57×10^{-3}
300	226980	2.27×10^{-5}	2.27×10^{-3}
380	296820	2.97×10^{-5}	2.97×10^{-3}
460	366660	3.67×10^{-5}	3.67×10^{-3}
560	453960	4.54×10^{-5}	4.54×10^{-3}
1,000	838080	8.38×10^{-5}	8.38×10^{-3}
1,800	1536480	1.54×10^{-4}	1.54×10^{-2}
3,200	2758680	2.76×10^{-4}	2.76×10^{-2}
5,412	4689756	4.69×10^{-4}	4.69×10^{-2}
5,600	4853880	4.85×10^{-4}	4.85×10^{-2}
10,000	8695080	8.70×10^{-4}	8.70×10^{-2}

Adapted from WSRC-STI-2008-00244 Table 50
[1] Using WSRC-TR-2005-00101, page D-6
[2] Total Cumulative Hole Size = earlier year hole size + (year interpolated - earlier year) / (later year - earlier year) * (later year hole size - earlier year hole size)
[3] Fraction of HDPE geomembrane with holes = Total cumulative hole size/10,000,000,000 mm^2/Hectare

DOE considered degradation of the HDPE layers by ultraviolet (UV) radiation, antioxidant depletion, thermal oxidation, high-energy irradiation, tensile stress cracking, attack from saltstone leachate, and biological degradation including microbial action, root penetration, and effects of burrowing animals. Of these degradation mechanisms, DOE concluded the mechanisms expected to cause the most degradation would be antioxidant depletion, thermal oxidation, and tensile stress cracking (WSRC-STI-2008-00244). As previously noted, in the PA DOE indicated that the degradation of the HDPE lining the exterior of the FDCs was based on the same evaluation of degradation made for the HDPE in the closure cap. For the HDPE in the closure cap, the estimated duration of antioxidant depletion, during which few defects are generated from oxidation, was based on recent studies (Mueller and Jakob, 2003). To estimate the creation of defects (i.e., pinholes, holes, tears, and cracks) in the closure cap HDPE from the combination of antioxidant depletion, thermal oxidation, and tensile stress, DOE used the method of Needham et al. (2004) which was developed by the Environment Agency of England and Wales to support PAs for landfills (WSRC-STI-2008-00244).

DOE expects the effects of other HDPE degradation mechanisms to be limited. For example, DOE expects degradation due to UV radiation during the time between FDC construction and backfilling to be limited by the incorporation of 2 to 3% carbon black and other ultraviolet chemical stabilizers into the HDPE. Attack from saltstone leachate is expected to be limited by use of the interior liner to limit bleed water penetration into the disposal units during curing (Section 2.5.1.3). DOE identified several methods to limit high-energy-irradiation induced damage of the HDPE layers, including shielding, lowering the level of oxygen exposed to HDPE, increasing antioxidant concentration in HDPE, use of thicker HDPE layer, and minimizing tensile stresses on HDPE layers. Of these, the current design calls for shielding (i.e., by the disposal unit walls), exposure to sub-surface (rather than atmospheric) oxygen concentrations, and use of thick (i.e., 2.5-mm [100-mil]) HDPE. DOE indicated that little information is available on long-term degradation of HDPE by fungi or bacteria, but that a common engineering textbook on geosynthetics (Koerner, 1998) describes HDPE as resistant to microbial degradation. DOE also expects the 2.5 mm (100 mil) HDPE used to cover the FDCs will be impervious to tree roots, except in areas with existing holes. This conclusion is based on Environmental Protection Agency (EPA) and industry experience with HDPE membranes used in landfill applications indicating that tree roots are effectively stopped even by thinner HDPE layers (e.g., approximately 8 mm [30 mil]). DOE also excludes the effects of burrowing animals from further consideration because DOE expects that burrowing animals will be deterred by the overlying erosion barrier (WSRC-STI-2008-00244).

With respect to GCL degradation, DOE considered the effects of slope stability, freeze-thaw cycles, dissolution, divalent cations (e.g., Ca^{+2}, Mg^{+2}), desiccation (wet-dry cycles), and biological degradation (e.g., root penetration, burrowing animals) (WSRC-STI-2008-00244).

Of these mechanisms, DOE concluded that degradation by divalent cations is likely to cause the most significant effects on the GCL performance. Divalent cations can cause GCL degradation because the divalent cations replace two Na^+ ions in the primary component of the GCL bentonite (i.e., sodium-montmorillonite). This replacement results initially in clays with approximately half the swelling capacity and poorer hydraulic conductivity, and subsequently in additional minerals that also have poorer hydraulic conductivity than sodium-montmorillonite. To estimate the effects of divalent cations on the GCL, DOE performed geochemical modeling with Geochemist's Workbench to evaluate the transformation of the sodium-montmorillonite in the GCL into other minerals as a function of the volume of water that has passed through the GCL. DOE used two different modeled water compositions: one representative of natural infiltrating water at the site and one representative of water equilibrated with portlandite ($Ca(OH)_2$). As expected, DOE found that the water equilibrated with portlandite caused much faster modeled degradation than the water equilibrated with site soil. Specifically, approximately 1,000 L/m^2 (100 L/ft^2) were required to cause significant transformation of the sodium-montmorillonite when water equilibrated with soil was modeled (WSRC-STI-2008-00244, Figure 17). Significant transformation of sodium-montmorillonite was seen after less than

50 L/m^2 (5 L/ft^2) of water equilibrated with portlandite was reacted with the same volume of the sodium-montmorillonite GCL (WSRC-STI-2008-00244, Figure 17). Because DOE judged there to be significant uncertainty in the geochemical modeling results, DOE did not directly apply the results and, instead, simply assumed the GCL layer to be degraded after the 100 year period of institutional controls. Specifically, DOE assumed the sodium montmorillonite GCL to be converted to calcium or magnesium montmorillonite with a saturated hydraulic conductivity approximately one order of magnitude higher (i.e., 5X10^{-8} cm/s). DOE used this hydraulic conductivity of the GCL alone in its determination of the hydraulic conductivity of the HDPE/GCL composite layer (Figure 2.5-3).

DOE provided rationale, summarized here, for excluding the other identified degradation mechanisms from modeling (WSRC-STI-2008-00244). DOE judged that slope stability will not significantly affect the HDPE/GCL composite layer performance because of the relatively low slope of installation. DOE expects that freeze-thaw cycling will not cause significant degradation for two reasons. The first is that the GCL layers on the roof and under the floors of the disposal units will be installed significantly below the 25 cm (10 in) depth to which soil freezes in South Carolina. The second is that DOE indicates that the clay in GCLs deforms easily around ice lenses and heals well after thawing. Regarding dissolution, DOE reasons that the relatively low amount of water DOE expects to reach the GCL layers will not lead to significant dissolution of the sodium-montmorillonite in the GCL into other minerals (e.g., kaolinite and quartz). DOE does not anticipate that burrowing animals will cause significant degradation because DOE expects they will be stopped by the erosion control layer in the closure cap. Similarly, DOE expects tree roots to be effectively deterred by the HDPE layer of the HDPE/GCL composite, and to cause GCL degradation only in areas of existing HDPE holes. In addition, DOE also expects the HDPE layer of the HDPE/GCL composite to prevent significant deterioration due to desiccation.

2.5.2.3 Integration and Interaction of Disposal Unit Materials

The FDCs contain complicated joints with many material interfaces. The FDC walls, in particular, are composed of multiple separate pre-cast wall panels that are joined in the field. DOE abstracted and simplified various components of Vaults 1 and 4 and the FDCs into the near-field PORFLOW™ model, as illustrated in Figure 2.5-4. As an example of the type of simplifications DOE made to model the disposal units, the complexity of the FDC wall-floor interface, including proposed changes to Cells 2A and 2B (as of November, 2010) (SRR-ESH-2010-00101), is shown in Figure 2.5-5. The abstraction of the wall-floor interface for the FDCs for the PORFLOW™ modeling is shown in Figure 2.5-6 (SRR-ESH-2010-00101). In addition to the physical complexity of the joints, the joints include a variety of different types of materials (e.g., epoxy, polysulfide material).

DOE developed Cases B and C (Section 2.5.2.1) to evaluate the dose impact of preferential pathways through the disposal units that may develop within a 10,000 year period of performance. As discussed in Section 4.4.2 of the PA, DOE designed these cases to capture the effect of preferential pathways through the disposal units, rather than explicitly represent all potential preferential pathways.

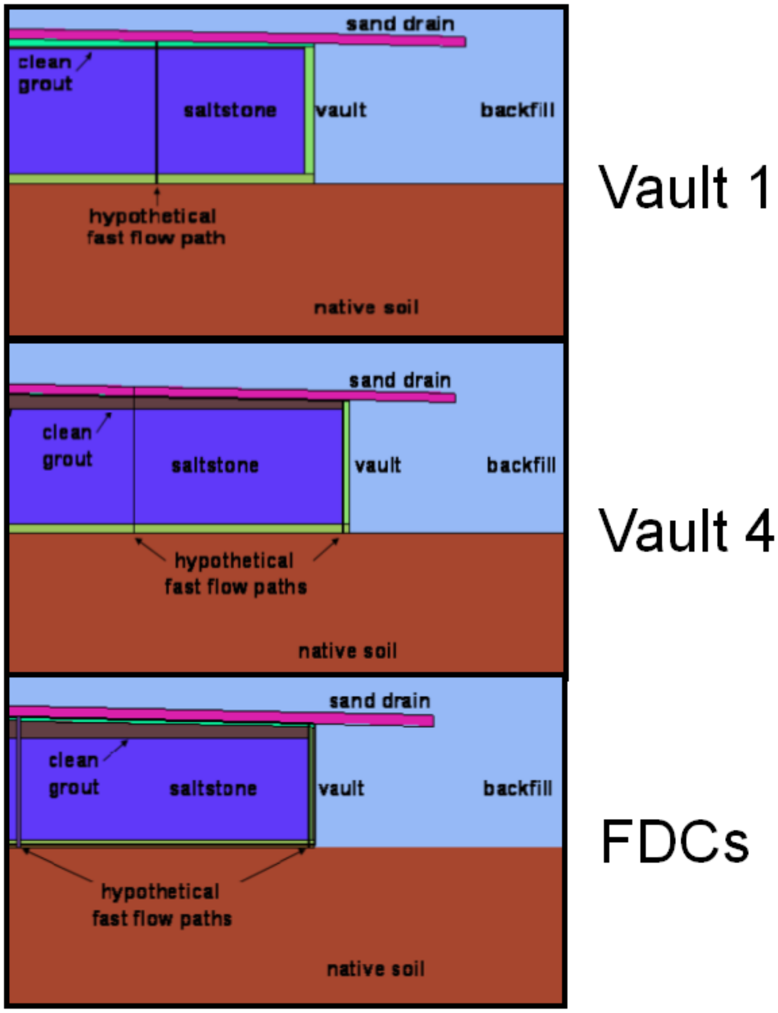

Figure 2.5-4: Respective Material Zones for PORFLOW™ modeling (adapted from 2009 PA)

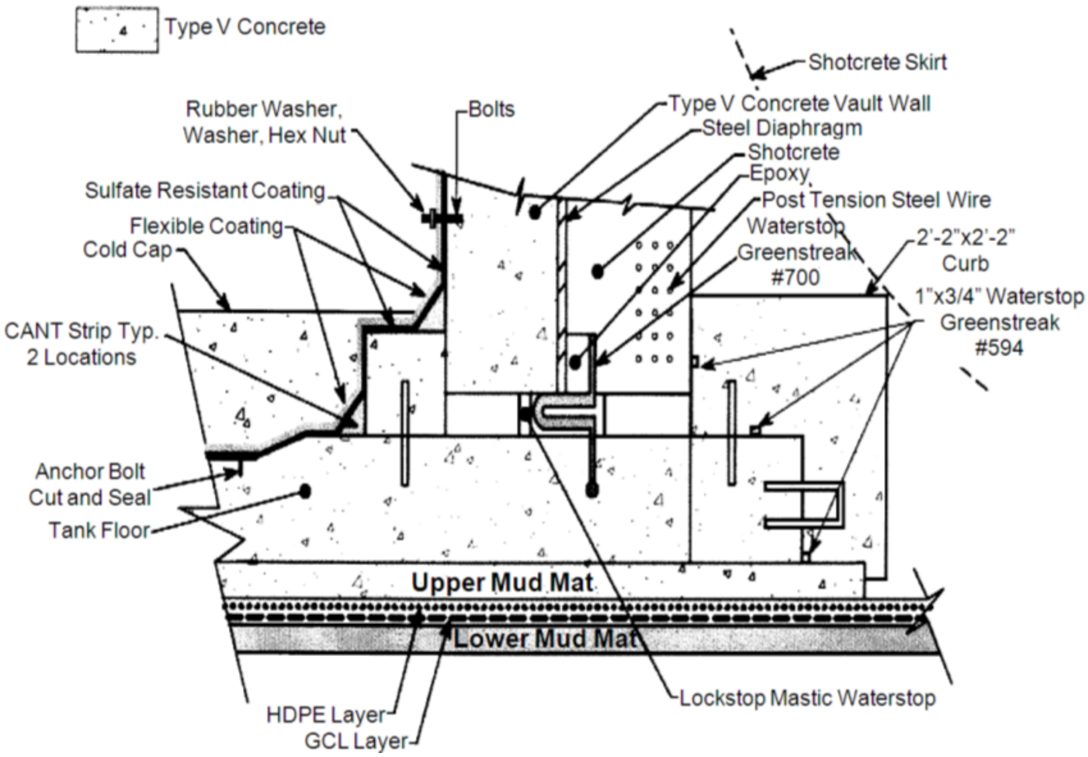

Figure 2.5-5: FDC wall-floor interface including proposed design enhancements (as of November, 2010) (adapted from SRR-ESH-2010-00101)

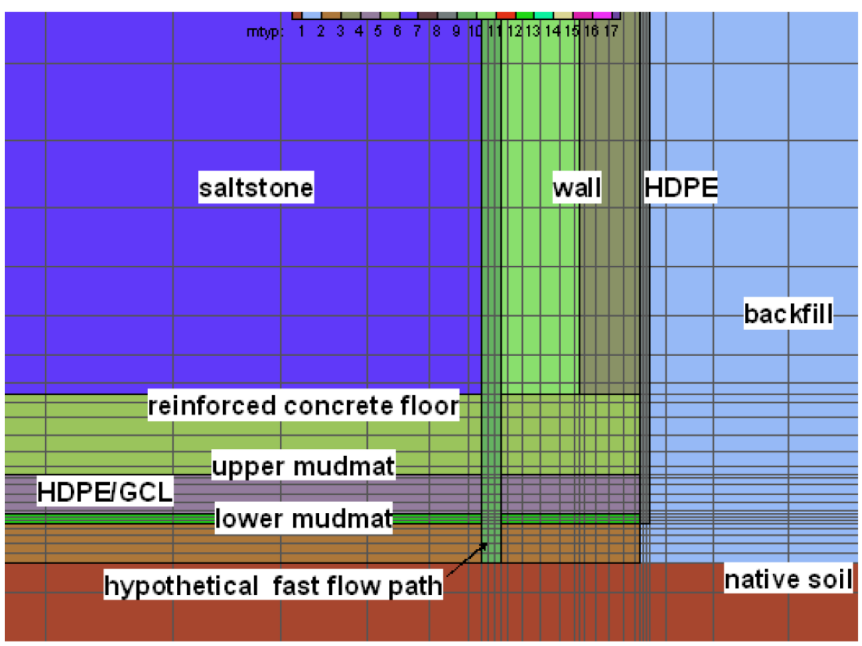

Figure 2.5-6: FDC lower corner detail for PORFLOW™ modeling (2009 PA)

2.5.3 NRC Evaluation – Disposal Unit Design and Performance

2.5.3.1 Construction Quality and Testing

The PA relies on the disposal units to provide secondary containment of the contaminants as well as limit waste form exposure to oxidizing chemical conditions. Therefore, inspection of the disposal units is important to identifying deviations from the disposal unit design and assumed performance within the PA, as discussed in the NRC's monitoring plan (NRC, 2007a).

In comment VP-1, NRC staff requested additional information to assess the applicability of the degradation mechanisms responsible for the observed fracturing of Vaults 1 and 4 to the FDCs and to other portions of Vaults 1 and 4. In its response (SRR-CWDA-2010-00033), DOE described design enhancements to the FDCs it believed would prevent the formation of cracks that were observed in Vaults 1 and 4 walls. These design enhancements include multiple engineered systems, structures, and components to address the potential buildup of liquids inside the FDCs, such as a system of sheet drains and associated drain water piping to drain liquids from the space between the saltstone monolith and FDC walls. Another is the cylindrical design that incorporates pre-stressing wires to counteract the stress of free-standing salt solution. Also, an interior epoxy-based coating is used to preclude liquids from penetrating the walls during filling operations and to protect the walls from sulfate attack during the approximately 30 year operational period when sulfate-bearing bleed water is the greatest concern. In addition, seismic restraints are included outside the concrete walls to address potential movement during a seismic event. Geotechnical investigations have been conducted (Chapter 5) as part of the overall FDC design and construction process to prevent potential differential settlement of the FDCs that might lead to cracking.

During hydrotests of disposal units 2A and 2B, damp spots were observed in several locations at the wall-floor joint and at the interface of the floor and mud mat (SRR-ESH-2010-00101). The observation of a fluorescent dye tracer confirmed that the damp spots were due to disposal unit leakage (SRR-ESH-2010-00101). Based on interior and exterior inspections, leakage was determined to have occurred through cracks beneath the anchor bolts and around the interior curb (SRR-ESH-2010-00101). Recurring delamination of the epoxy coating was determined to be a likely contributor to the leakage (SRR-ESH-2010-00101). NRC Onsite Observation Reports (NRC, 2010d, h) provide additional information on the hydrotesting of disposal units 2A and 2B. In its second response to NRC comment VP-3 (SRR-CWDA-2011-00044), DOE described significant FDC design modifications that have been implemented after the FDC 2A/2B hydrotesting. The modifications to disposal units 2A and 2B include cutting the anchor bolt penetrations flush with the floor, applying an interior coating, installing an additional 36 cm (14 in) of sulfate resistant concrete above the coating layer, and installing an exterior sulfate resistant curb around the perimeter of the disposal unit floor. DOE stated that these modifications were made to ensure the disposal units are constructed and operated in such a manner that they have properties consistent with the Case A assumptions. However, NRC staff has expressed concern that the installation of curbing to the exterior of the disposal units prior to the hydrotest would prevent the potentially weak interfaces that could have leaks from being visible during the hydrotest, thereby confounding any conclusions about the effectiveness of the disposal unit repairs (NRC, 2010h). In addition, DOE discussed that significant design modifications to the disposal units, including units 7A - D[11] have been implemented based on the lessons learned from the hydrostatic testing (e.g., future disposal units do not include anchor bolt penetrations in the floor).

In addition to the design modifications, the hydrotest procedures were also modified (SRR-ESH-2010-00101) to accommodate changes to the allowable stresses to the disposal unit resulting from the installation of the curb that was added on the exterior of the disposal units. The maximum test height of water was revised from 6.7 m to 3.7 m (22 ft to 12 ft) and the test duration was extended to a minimum of 98 hrs to provide a sufficient time for the hydrotest to equate to a hydraulic head of 6.7 m (22 ft). After the disposal units are backfilled, DOE anticipates that a fill height of 6.3 m (20.6 ft) for a liquid with a specific gravity of 1.3 will not exceed allowable stresses. NRC staff agrees that backfilling will help reduce stresses on the disposal unit walls. However, similar to the installation of curbing, the presence of backfill during the filling of the disposal units with saltstone grout will prevent visual inspection and result in additional uncertainty in the disposal unit integrity. The incorporation of a liquid-sample collection system from beneath certain disposal units (State of SC, 2007) may increase confidence in the short-term performance of the disposal units.

Based on the performance of the floors of FDC 2A and 2B during recent hydrotesting (due to cracking near anchor bolts), it is not clear that DOE's Case A assumption that there are no fractures in the floors of Vaults 1 and 4 is realistic, because Vaults 1 and 4 also contain anchor bolts. The NRC staff suggested (NRC, 2010b; VP-3) that DOE include vault floor fractures in its base case or provide a basis for not including this feature in light of limited vault floor characterization (SRR-CWDA-2011-00044). In response, DOE described a visual inspection that was done on drainwater system anchor bolts in Vault 4 Cells B and H, which indicated no evidence of voids or cracking at the embedded anchor bolt locations. DOE stated that plans and procedures have been developed to perform regular visual disposal unit inspections to identify any structural deterioration that may require evaluation. NRC staff considers DOE's response to cracking beneath the anchor bolts adequate to demonstrate that the floors do not currently have visible cracks and that DOE will monitor and address the potential development of visible cracks prior to filling the disposal units. However, significant uncertainty remains about the Case A assumption that the disposal unit floors will not fracture for thousands of years after closure.

2.5.3.2 NRC Evaluation – Disposal Unit Cementitious Materials Performance

The initial hydraulic conductivity and effective diffusion coefficient values assumed in the PA for the FDC are in the range of those observed in the literature for concretes for non-fractured concrete (CNWRA, 2008). The NRC staff finds that these values seem reasonable for the concrete in the FDCs. The Vaults 1 and 4 walls have already shown cracking and were modeled in the PA with an increased saturated hydraulic conductivity. A high hydraulic conductivity value (i.e., higher than the hydraulic conductivity of soil) was assumed for the

[11] DOE has changed the numbering of the FDCs since the PA was published. These FDCs are now referred to as 2A/2B, 3A/3B, and 5A/5B.

Vault 1 and 4 walls in the PA cases (i.e., cases other than Case K, K1, and K2). However, as described in more detail in Section 2.7, the modeled flow was limited by the assumed moisture characteristic curves. The Vault 1 and 4 roofs and floors are assumed to have properties of medium and high quality concrete, respectively. As described above, the Vault 4 roof has previously leaked. Repairs were made by DOE to seal the joints in the roof. However, it is unclear how well these repairs will perform in the future. Additionally, as described in the previous section, although DOE inspected the Vault 4 floors, uncertainty remains in their future hydraulic performance because the floors may have fractures that are not readily visible but transmit water.

In the PA cases, the degradation of the disposal unit concrete was based on sulfate attack modeling results performed using the STADIUM® code. The PA and supporting documents predict that all of the vault and FDC concrete components except the Vault 1 and 4 walls and Vault 4 roof would remain essentially intact with hydraulic conductivities increasing by less than an order of magnitude within 10,000 years. The estimated hydraulic conductivity of the Vault 4 roof increased by approximately three orders of magnitude within 10,000 years, and the Vault 1 and 4 walls were modeled as being degraded at closure.

In comment VP-2 (NRC, 2010i), NRC noted that degradation mechanisms other than sulfate attack (e.g., alkali silica reaction and corrosion-induced degradation) were not included in the evaluation of the degradation of the Vaults 1 and 4 roof and floor, or the FDC roof, walls, and floors. The NRC staff specifically noted that the design of the FDCs includes significant amounts of carbon steel components (e.g., rebar, prestressing wires, diaphragms), which could corrode and lead to expansive reactions that could cause cracks to form in the concrete.

In its second response to NRC comment VP-2 (SRR-CWDA-2011-00044), DOE provided additional information to support its assumption that degradation mechanisms other than sulfate attack will not significantly increase the degradation of the disposal unit concrete. DOE addressed (i) alkali–silica reaction, (ii) carbonation, (iii) chloride-induced corrosion, (iv) calcium leaching, (v) microbial degradation, (vi) freeze–thaw cycles, and (vii) cracking from seismic events, settlement, and external static loading. For example, DOE cited petrographic testing of concrete samples from the exterior and interior of the F- and H-Canyon structures that indicated alkali–silica reaction was not present in that concrete to date. DOE also indicated that, because the aggregate that will be used in the disposal unit concrete has a large component of relatively non-reactive granite, the potential for adverse alkali-silica reactions is limited.

DOE used a concrete carbonation rate derived from carbonation depths measured in P-reactor concrete to estimate the time interval for the Vault 1 and 4 concrete to be carbonated and for rebar corrosion to initiate. The estimated time interval was in excess of 10,000 years, which is similar to the PA estimated concrete degradation time by sulfate attack. DOE also provided an estimation of the amount of chloride induced corrosion expected. DOE predicted that it would take over 6,000 years to reduce 5% of the rebar and over 10,000 years to reduce 10% of the rebar. DOE noted that this reduction could result in the formation of preferential or fast flow paths.

In the response to NRC comment VP-2, DOE stated that calcium leaching is not expected to be significant in the saltstone disposal units, based on an assessment provided in NUREG/CR-5542. This is consistent with NRC staff's understanding of the potential degradation due to calcium leaching. However, the effects of calcium leaching may be important to the overall degradation of cementitious materials due to the potential coupling of multiple degradation mechanisms. DOE also stated that microbial organisms could potentially promote damage to cementitious materials and further research on this topic is planned. DOE stated that exposure of FDC concrete to freezing conditions is not expected after the operational period because, after the disposal units are filled, they will be surrounded with backfill and a closure cap will be installed. However, the NRC staff expects that some freeze-thaw damage could occur prior to the installation of the closure cap for Vaults 1, 4, and any portions of the FDCs that are covered by less than 25 cm (10 in) of soil, which is the depth to which soil freezes in South Carolina.

DOE indicated cracking from seismic events, settlement, and external static load have been considered in the design and construction of the vaults and FDCs and macroscopic cracks in the exterior Vault 1 and 4 walls were modeled in the 2009 PA. However, the NRC staff notes that the formation of cracks in the Vault 1 and 4 walls were modeled assuming bulk degradation of the walls, not discrete fractures, and the Case A modeling included the use of moisture characteristic curves that NRC staff questioned the basis (Section 2.7). Additionally, the Case A model did not include fracturing of the Vault 1 and 4 floors. The NRC staff also disagreed with some aspects of the settlement and structural integrity analysis (Chapter 5). Specifically, recent geotechnical investigations for FDCs 2A/2B, and 7A, 7B, 7C, and 7D (Figure 2.8-8])[12] predicted settlement values greater than the American Concrete Institute (ACI) standard (ACI-372) for the FDCs (K-ESR-Z-00001; K-ESR-Z-00002). Because neither the standard nor the DOE analysis addresses the consequences of exceeding these criteria, the risk-significance of this finding is unclear. Accordingly, the NRC staff will monitor the development of information related to settlement of the SDF due to static loading.

DOE predicted the effects of settlement to result in cracking of the floor and the overlying saltstone grout at the construction joints (T-CLC-Z-00006). These predictions are inconsistent with the hydraulic conductivities that were assumed in the PA for Vault 4 (e.g., 3.84×10^{-10} cm/s at 10,000 yrs) (SRNL-STI-2009-00115, Appendix E). In addition, this structural integrity analysis assumed that the strength of the concrete and saltstone do not change with time (T-CLC-Z-00006). However, the exclusion of chemical degradation mechanisms which tend to decrease the strength of cementitious materials (e.g., sulfate attack, rebar corrosion, alkali silica reaction) from the structural integrity analysis could result in an underestimation of the amount of fracturing. Both physical and chemical degradation mechanisms can affect the long-term performance of disposal units. Therefore, coupling these mechanisms in an integrity analysis is necessary to ensure that predictions of disposal unit performance are not overly optimistic.

[12] DOE has changed the numbering of the FDCs since the PA was published. These FDCs are now referred to as 2A/2B, 3A/3B, and 5A/5B.

Based on the information in this section, which describes a limited evaluation of degradation mechanisms for disposal units and the inconsistency between relevant studies and the PA, NRC staff finds the assumed long-term performance of disposal units in Case A to be unsupported.

In addition to the physical and hydraulic properties of the disposal unit concrete, the chemical properties can also significantly affect the transport of radionuclides out of the disposal unit and into the environment. The modeled transport of many elements depends on the E_h and pH of the disposal unit concrete, which affect the sorption coefficient (K_d value). The three different approaches DOE used to model the effects of E_h and pH on radionuclide release from saltstone and transport through the disposal units are discussed in more detail in Sections 2.7.1 and 2.7.4.1 and summarized here.

In the pore-volume modeling approach DOE used to model the chemical properties of the cementitious materials for radionuclides other than Tc (in all cases), the chemical conditions of the whole component (i.e., saltstone monolith or disposal unit floor, walls, or roof) transition in a step change. That is, the E_h and pH of the whole component transition only when until enough water has traveled through the entire component to completely consume the reducing capacity (for E_h) or to consume the buffering capacity to a certain pH. However, in the real system, portions of the concrete are likely to undergo earlier E_h and pH transitions due to relative position and preferential flow through intact and degraded regions. Earlier transitions in certain regions would affect the modeled transport of elements that have significantly different chemical properties (e.g., solubility limit or K_d value) depending on the E_h and pH of the system[13]. The NRC staff notes that the dissolved oxygen concentration used in the calculation of the number of pore volumes required to cause E_h transitions is conservatively assumed to be in equilibrium with atmospheric oxygen. The concentration of oxygen in the subsurface may be lower (e.g., DOE reports a measured value of 1.2 mg/L from a groundwater monitoring well near the SDF). Thus the NRC staff finds the pore-volume approach appropriate for elements with limited sensitivity to system E_h and pH.

The NRC staff finds that the explicit shrinking core approach that DOE used in Case A and the other PA cases to model the transport of Tc in the disposal units is more appropriate than the pore-volume approach appropriate for elements sensitive to system E_h or pH (e.g., Tc). In particular, the NRC staff finds that tracking releases from the oxidized and reduced fractions of saltstone separately appears to be the most appropriate implementation of DOE's conceptual model of Tc-99 release. Regarding the oxidation modeling, the NRC staff finds DOE's assumption that infiltrating water is equilibrated with atmospheric oxygen (i.e., 8 mg/L dissolved oxygen, equivalent to 1.06 meq/L) to be conservative because subsurface oxygen concentrations are likely to be lower (e.g., DOE reports a measured value of 1.2 mg/L from a groundwater monitoring well near the SDF). However, the NRC staff finds DOE's basis for neglecting gas-phase transport of oxygen to saltstone surfaces, which relies on saltstone saturation, to be insufficient (Section 2.7.4.1). In addition, the NRC staff finds DOE's omission

[13] The modeled release of the redox-sensitive radionuclide Tc-99 is not affected by this concern because, as previously described, Tc-99 release was modeled using a different approach.

of gas-phase oxygen transport in the PA cases is inconsistent with DOE's Case K, K1, and K2 results, which demonstrate that gas-phase oxygen can have a non-negligible effect on saltstone and disposal unit concrete oxidation. The relative importance of these two sources of oxygen (i.e., gas-phase oxygen and dissolved oxygen in infiltrating water) will depend on assumptions about water flow through the system as well as the concentrations of oxygen in soil, gas, and infiltrating water.

In Cases K, K1, and K2, DOE separates modeling of oxidation from modeling of Tc release and transport. Specifically, in Cases K, K1, and K2 DOE models saltstone and disposal unit oxidation by calculating the fraction of the concrete that is oxidized based on the diffusion of oxygen into the disposal unit concrete through fractures. Unlike the explicit shrinking core model implemented in PORFLOW, the average-K_d model does include diffusion of oxygen from the gas phase into saltstone or the disposal unit concrete, however it neglects oxygen introduced into the saltstone or disposal unit in infiltrating water. In Cases K, K1, and K2, DOE assumes that the soil gas in contact with saltstone has atmospheric concentrations of oxygen. The NRC staff finds this to be a conservative assumption. However, it is unclear if this conservative assumption entirely compensates for omitting water in infiltrating water from the oxidation model. As previously noted, the relative importance of dissolved oxygen in infiltrating water depends on assumptions about the dissolved oxygen concentration and flow rate.

In Cases K, K1, and K2, after the extent of oxidation is calculated, the sorption coefficient (K_d value) used in the model is then based on a weighted average of the oxidized and reduced K_d values and the fraction of the concrete that is oxidized (Section 2.7.1). As discussed in more detail in subsequent paragraphs and in Section 2.7.1.4, this "average-K_d" modeling approach results in significant unexpected modeled chemical performance.

DOE considered a number of cases that included various assumptions for the physical, hydraulic and chemical degradation of the disposal unit concrete. Case A, which DOE considers to be the base case, contains limited physical and hydraulic degradation. As noted above, the NRC staff does not consider the assumptions and parameter values used in the Case A modeling for the disposal unit concrete to represent the current and future expected performance. In particular, the NRC staff did not agree with the minimal degradation assumed for the FDC concrete and the Vault 1 and 4 floors and roofs. The Case A modeling assumptions for the disposal unit concrete did not include all of the relevant degradation mechanisms, particularly those mechanisms that result in the formation of discrete fractures in the concrete. The NRC staff also noted that the modeled diffusive release through the Vault 4 walls was inconsistent with the observed advective seeps that occurred during grout placement (NRC, 2007a).

The disposal unit assumptions in Case C were similar to those in Case A, except that Case C accounts for a discrete fracture through the disposal unit. Similarly, the 10x Sulfate Attack Case considers a faster rate of degradation of the concrete due to an increased rate of sulfate attack. The Oxidized Concrete Case evaluates the dose if the Vault 1 and 4 walls are assumed to be oxidized at the time of closure. However, like Case A, these cases contain other modeling assumptions that the NRC staff does not agree with, such as the use of overly optimistic moisture characteristic curves (Section 2.7), and overly optimistic assumptions for the chemical and hydraulic performance of the saltstone waste form (Section 2.6).

The Synergistic Case (PA Section 5.6.6.5) non-mechanistically accounts for early hydraulic and chemical degradation of disposal unit roofs, floors, and walls. However, the degree of overall conservatism assumed in the sensitivity analysis is unclear because the Synergistic Case incorporates overly optimistic assumptions for the chemical and hydraulic performance of the saltstone waste form (Section 2.6). In its response to NRC comment VP-2 (SRR-CWDA-2010-00033), DOE provided results of a revised Synergistic Case that demonstrated that use of the moisture characteristic curves did not significantly affect the synergistic case results (i.e., the assumption that the relative permeability is 1 did not significantly affect the results). However, this case still included the overly optimistic assumptions for the chemical and hydraulic performance of the saltstone waste form (Section 2.6).

As noted in Section 2.1, the NRC staff considers the modeling performed in Case K, K1, and K2 to most appropriately reflect current and future expected conditions at the SDF. These cases include significant bulk degradation of the disposal unit concrete, which is represented by an increase in the hydraulic conductivity of the cementitious materials to a hydraulic conductivity value that is typical for soil. However, it is unclear whether these cases include any discrete fractures in the disposal unit concrete. The response to NRC comment PA-8 (SRR-CWDA-2011-00044) does not describe the presence of discrete fractures in this model. However, the Case K PORFLOW™ files provided to the NRC indicate that minor fracturing (i.e., a single discrete fracture) may have been included. Regardless, Cases K, K1, and K2 derive significant chemical performance from the disposal unit concrete that is inconsistent with the assumption that the disposal unit concrete is fractured (Section 2.13). As seen in Figure 2.5-1, even though significant hydraulic failure of the disposal unit concrete occurred in Case K, K1, and K2, a significant fraction of the disposal unit concrete remained reduced, even at long evaluation times. This resulted in a large amount of modeled sorption of Tc-99, which resulted in a lower rate of release and a delayed time of release from the disposal unit for this radionuclide. In reality, the flow through the disposal unit concrete will likely occur through fractures rather than through the bulk concrete. The NRC staff expects these fractures to become oxidized (i.e., less sorptive for Tc-99) more quickly than the bulk concrete and will not provide as much chemical performance.

2.5.3.3 NRC Evaluation – HDPE and HDPE/GCL Composites

In general, the NRC staff concludes that the hydraulic properties assigned to the HDPE/GCL composite layers on the FDC roof and under the FDC floors and the HDPE on the FDC walls appear to be reasonable. The NRC staff evaluated DOE's consideration of potential sources of HDPE and GCL degradation and concluded most major potential degradation modes were considered. However, the NRC staff is concerned that the potential for differential settlement due to static settlement or consolidation of soft zones has not been evaluated fully and could impact the HDPE/GCL performance (Chapter 5).

DOE indicated that its estimated degradation of the HDPE lining the exterior of the FDCs was based on the same methods that were used to estimate degradation of the HDPE in the closure cap (PA page 217). The NRC staff finds the application of the methods used to estimate antioxidant depletion (Mueller and Jakob, 2003) were reasonable. Although considerable uncertainty in antioxidant depletion time is introduced by the uncertainty in the choice of an activation energy (e.g., WSRC-STI-2008-00244 Table 28), DOE's selection appears to be reasonable. Specifically, in the context of the closure cap, the NRC staff does not expect DOE's choice of the duration of antioxidant depletion (i.e., 275 years) (WSRC-STI-2008-00244 Appendix I) to have a significant effect on performance during a 10,000 year compliance period. Although it is unclear whether DOE used the same choice with respect to the HDPE surrounding the FDCs, the evolution of the hydraulic conductivity and diffusivity of the FDC HDPE (Figure 2.5-2) indicates that DOE appears to have chosen an appropriate value of the duration of antioxidant depletion.

With respect to estimation of the combination of the effects of antioxidant depletion, thermal oxidation and tensile stress cracking, in general, the NRC staff finds the application of the method of Needham et al. (2004) to be reasonable. The applicability of the method of estimating defects due to tensile stress to the vertical sections of HDPE on the FDC walls and the interfaces between the HDPE on the walls, roof, and under the floor is unclear. However, the NRC staff expects the method to apply to the HDPE in the HDPE/GCL layers on the roof and under the floor, which are expected to have the most significant effect on modeled HDPE performance (Section 2.7).

With respect to HDPE degradation from exposure to saltstone and concrete leachates, as well as site groundwater, the NRC staff concludes that short-term tests do not indicate the potential for HDPE corrosion, and that long-term information is lacking. The leachate from the saltstone grout is expected to be an aqueous solution containing aluminum nitrate, sodium carbonate, sodium nitrate, sodium nitrite, sodium sulfate, and sodium phosphate and from concrete is expected to be an aqueous solution containing predominantly calcium hydroxide. The groundwater is expected to contain predominantly sodium and calcium cations, and chloride and carbonate anions. Of these, calcium hydroxide, sodium hydroxide, aluminum nitrate, sodium carbonate, and sodium chloride aqueous solutions are not expected to be particularly corrosive to HDPE (Schweitzer, 2004). No literature information could be found on the effects of sodium nitrite, sodium nitrate, sodium sulfate, and sodium phosphate aqueous solutions on HDPE. However, HDPE is not expected to be adversely affected by calcium sulfate, calcium nitrate, up to 50 % phosphoric acid, and up to 20 % nitric acid aqueous solutions (Schweitzer, 2004) below 60 °C. Although there is limited information about the effects of these chemicals on HDPE performance in the long term (i.e., thousands of years), the NRC staff concludes that the potential effects of these chemicals on the HDPE are accounted for, at least in part, by DOE's modeled rapid degradation of HDPE hydraulic conductivity and diffusivity (Figure 2.5-2).

These chemicals may have a more significant effect on the performance of the HDPE/GCL composite because of the importance of divalent cations to GCL degradation. Specifically, DOE found that a relatively small amount of water equilibrated with portlandite was expected to cause significant transformation of the sodium-montmorillonite GCL into higher-hydraulic conductivity minerals (WSRC-STI-2008-00244, Figure 17). However, the effects of this transformation appear to be accounted for by DOE's choice to increase the modeled hydraulic conductivity of the GCL layer to 5 x10^{-8} cm/s at 100 years after site closure. The NRC staff does not expect DOE's selection of 100 years after site closure for this transformation to be important to modeled FDC performance.

With respect to the other degradation GCL mechanisms DOE considered, the NRC staff finds the discussions of slope stability, desiccation, and biological factors to be appropriate. Regarding freeze-thaw cycling, the NRC staff agrees that, once installed, the GCL layers should be protected from freezing, but notes some of the GCLs will be installed several years prior to installation of the closure cap. Neglect of freeze-thaw degradation therefore does rely in part of DOE's understanding that the GCLs will recover well from freezing. The discussion of the potential effects of sodium-montmorillonite dissolution did not appear to be complete. Specifically, DOE estimated the time required for complete dissolution of a unit area of the GCL under natural infiltration conditions (i.e., approximately 1,200 years). DOE then reasoned that since substantially less water was actually expected to reach the GCL layers over and under the FDCs, dissolution would not occur during the performance period. This reasoning appears to neglect the potential degradation of GCL performance from partial dissolution of the sodium-montmorillonite. However, the NRC staff concludes that dissolution is likely to cause less degradation than exposure to divalent cations and that the degradation is appropriately reflected in the evolution of hydraulic properties of the HDPE/GCL composite layer DOE used in its PA models (Figure 2.5-3).

Use of a material with which there is limited long-term engineering experience and no natural analogues, such as HDPE, introduces conceptual model uncertainty. For example, if the HDPE performs better than expected, and forms few defects for thousands of years after placement, the saltstone in the FDCs could oxidize substantially from gas-phase transport of oxygen while being exposed to very little water. If the HDPE were then to begin to fail several thousand years after placement, when the closure cover and FDC roofs are already significantly degraded, the oxidized saltstone in the FDCs could be suddenly exposed to a much larger flow of water that could cause the release of a significant fraction of the Tc-99 inventory in a relatively short amount of time. The sudden failure of the HDPE on the roof and under the floor of the FDCs is expected to be mitigated to some extent by the GCL, which the NRC staff expects to fail more gradually. However, if both layers fail as the result of a disruptive event (e.g., an earthquake or formation of a sink), water flow through the FDC could increase significantly in a relatively short time. Thus, information regarding the potential for sudden failure of the HDPE/GCL layers is important to an evaluation of predicted site performance.

2.5.3.4 NRC Evaluation – Integration and Interaction of Disposal Unit Materials

DOE's abstraction of the complex material interfaces present in the disposal units does not account for preferential pathways in Case A. The low hydraulic conductivity (e.g., 3.60x10^{-10} cm/s [1.42x10^{-10} in/s] for the FDCs at 10,000 yrs) of the modeled material interfaces in Case A are representative of seamless and undegraded high-quality concrete (hydraulic conductivity value from Appendix E of SRNL-STI-2009-00115). However, observations from disposal units 2A and 2B during the hydrotests and observations of seepage at interfaces in Vault 4 are indicative of the challenges associated with constructing water-tight joints. Furthermore, there is little support for the assumption that joints can remain essentially water tight over long periods of time. In particular, the inclusion of materials about which there is little information to support predictions of long-term performance (e.g., epoxy, polysulfide material) add uncertainty to the predictions of the long-term performance of the FDCs.

Because Cases B, C, and the Synergistic Case capture a range of potential disposal unit degradation mechanisms, the NRC staff concludes that these cases are more defensible with respect to potential preferential pathways than Case A, although they have other limitations (Sections 2.1, 2.6 2.7, and 2.13). The complex design of the disposal units, in particular the FDCs, and the limited operating experience with novel materials warrants the inclusion of preferential pathways through the joints in a compliance case. As previously discussed, the PORFLOW™ files for Cases K, K1, and K2, appear to include minor fast pathways through the disposal unit floors (Section 2.5.3.2). However, the amount of chemical performance from the disposal unit concrete in these cases is inconsistent with significant flow through fast pathways though disposal unit joints. Because of the importance of potential fast pathways through the disposal units, the NRC staff will monitor the development of model support for the long-term performance of the disposal unit material interfaces.

2.5.3.5 Disposal Unit Design and Performance Conclusion

The performance of the disposal units can significantly affect the release of radionuclides from saltstone into the environment. In the 2005 TER (NRC, 2005), the NRC staff stated that degradation that may influence the hydraulic isolation capabilities of the saltstone and disposal units is a key factor in assessing whether the SDF can meet the performance objective in §61.41. The 2005 TER also stated that model support for the extent and frequency of fractures that will form in the disposal units was essential to assessing whether the saltstone disposal facility can meet the performance objective.

With respect to the 2009 PA, the NRC staff concludes that the assumptions made in Case A, which DOE regards as its base case, for the performance of the disposal unit were not justified for the following reasons:

- Case A included only minimal degradation of the FDC concrete and the Vault 1 and 4 floors and roofs and did not account for all potentially relevant disposal unit degradation mechanisms;
- Case A did not include fast pathways that could occur at the material interfaces (e.g., joints between the floor and walls, or between the wall panels) in the disposal unit or fractures in the disposal unit concrete;
- Case A disposal unit performance assumptions were not consistent with the observed advective releases that occurred during saltstone placement in Vault 4.

The NRC staff finds that the modeling assumptions and parameters selected for the disposal unit concrete in Cases K, K1, and K2 are generally appropriate. However, as described above, while these cases are based on the assumption that the hydraulic conductivity of the disposal unit concrete degrades significantly, the chemical performance of the disposal units in these cases is inconsistent with any significant flow through fast pathways (e.g., flow through fractures in the concrete or joints). The NRC staff notes that DOE also performed a number of other sensitivity cases that included increased disposal unit degradation or the formation of fast paths out of the disposal unit. However, as described in other sections of this TER (Section 2.1, 2.6, 2.7, 2.13), the NRC staff determined that other aspects of the modeling in these cases were not appropriate and limited the use of these cases in the NRC assessment of compliance with the performance objectives.

DOE stated that PA maintenance activities include testing and research activities addressing (i) degradation mechanisms associated with cementitious hydraulic properties, (ii) moisture flow through fractures, and (iii) the extent and frequency of fractures in disposal units (SRR-CWDA-2011-00044). NRC staff expects that this research will be useful in providing model support for future revisions to the PA.

NRC staff concludes that additional information is needed regarding the expected performance of the disposal units and has identified key monitoring factors related to the disposal unit performance. For example, because of the importance of the large difference in hydraulic conductivity between the disposal unit roofs and lower lateral drainage layer to disposal unit performance (Sections 2.7 and 2.13), the NRC will monitor model support for the long-term roof performance. Similarly, because of the uncertainty in the long-term behavior of HDPE and the importance of the HDPE/GCL layer overlying the FDC roofs (Sections 2.7 and 2.13), the NRC staff will monitor the development of information about HDPE and GCL performance. In addition, NRC staff thinks that additional model support related to the long-term performance of the non-cementitious materials used in the disposal units (e.g., epoxy and neoprene seals) may be a useful part of an assessment of the potential for fast paths out of the disposal unit. In particular, the NRC will monitor the development of model support for the long-term chemical performance of the disposal units (e.g., sorption of radionuclides in disposal unit fractures or joints). As DOE constructs additional disposal units, the NRC staff will monitor any changes to the FDC design.

2.6 Waste Form

2.6.1 Description of Waste Form

Saltstone is a cementitious waste form made by mixing treated salt solutions from the SRS F- and H-Tank Farm liquid waste storage tanks with a dry mixture of blast furnace slag, fly ash, and cement. The treated salt solution is comprised mostly of sodium nitrate, sodium hydroxide, sodium nitrite, sodium aluminum hydroxide, sodium carbonate, and sodium sulfate. The solid components in the dry mixture are silicon dioxide, aluminum oxide, calcium oxide, magnesium oxide, and iron (III) oxide. Solidified saltstone forms a microporous matrix containing a salt solution, predominantly sodium, nitrate, and nitrite, within its pore structure. Chemically, saltstone is alkaline and reducing. Clean grout, which is added to the disposal units to cap the saltstone, is made from process water and has a bulk composition similar to saltstone (Table 2.6-1).

Table 2.6-1: Formula of Saltstone and Clean Grout

Ingredient	Quantity (wt%)	
	Saltstone	Clean Grout
Blast Furnace Slag (grade 100 or 120)	25	28
Cement (ASTM C 150 Type II) or lime	3	6
Fly Ash (Class F)	25	28
Salt Solution (average 28% by weight salts)	47	N/A
Water (maximum)	N/A	38

2009 PA; Table 4.2-7

One of the key assumptions of Case A is that the hydraulic and physical properties of the waste form (Table 2.6-2) will essentially eliminate any advective flux of radionuclides out of the waste and limit releases to diffusion-dominated fluxes. In Case A, the dry porosity, bulk density, particle density, hydraulic conductivity, and effective diffusion coefficient in the waste form are assumed to remain constant for the entire 20,000 year analysis period. As discussed in Section 2.1, DOE used deterministic sensitivity analyses to test the effects of fracturing and increased bulk hydraulic conductivity and diffusivity. Case K represents the effects of fracturing on the flow of fluid through saltstone by modeling an equivalent increase in the bulk hydraulic conductivity[14].

Table 2.6-2: Saltstone Hydraulic and Physical Properties Used in Case A

Property	Value
Hydraulic conductivity (cm/sec)	2.0×10^{-9}
Effective diffusion coefficient (cm^2/sec)	1.0×10^{-7}
Porosity (%)	58.0
Dry bulk density (g/cm^3)	1.01
Particle density (g/cm^3)	2.40

The properties were assumed to remain constant over time.
Values taken from 2009 PA (Table 4.2-16)

DOE expects that, with a few notable exceptions, release of most contaminants from saltstone is controlled by sorption, which DOE represents with linear distribution coefficients (K_d values).

[14] Although Case K represents the effects of fracturing on saltstone hydraulic properties with changes in the bulk hydraulic conductivity, the effects on the fraction of saltstone oxidized is modeled by considering oxidation as proceeding from fracture faces.

DOE expects release of Tc to be limited by precipitation of Tc(IV) with sulfur supplied by the blast furnace slag in the dry premix. However, as discussed in greater detail in Section 2.7, DOE also represents Tc release with a sorption process. In addition, DOE indicated that Pu may be solubility limited in saltstone, but it represented Pu release in the PA by a sorption process. K_d values are element specific and, for some elements, change as the saltstone E_h (oxidation potential or measure of reduction potential) and pH (measure of acidity or basicity) vary with time.

The release of redox sensitive elements, most notably Tc, is strongly influenced by E_h conditions. Tc is relatively insoluble and strongly sorbs under reducing conditions but is mobile under oxidized conditions. DOE included blast furnace slag in the saltstone dry mix to create reducing conditions that immobilize Tc-99. The use of blast furnace slag and the use of fly ash in the grout formulation also are known to have the added benefits of reducing heat evolution during hydration, decreasing hydraulic conductivity, and increasing strength and sulfate resistance. DOE measurements indicated slag has a specific reducing capacity of 820 μeq/g (SRNS-STI-2008-00045 and SRNL-STI-2009-00637) and solutions reacted with saltstone have redox potentials (E_h) as low as –585 mV (SRNL-STI-2010-00668). In addition, a field study conducted at SRS (Langton, 1988; WSRC–RP–92–1360) indicated that the addition of slag to saltstone essentially prevented Tc-99 leaching from a 210-liter (55-gallon) sample of saltstone for at least 2.5 years. The immobilization of Tc was ascribed to the formation of reducing conditions and the limited diffusion of oxygen within the core of the 210-liter (55-gallon) sample (Langton, 1988). The PA and supporting documents (e.g., SRNL-STI-2010-00668; also summarized in SRR-CWDA-2011-00044; SP-15 response) cite laboratory studies that suggest that Tc is in the reduced +4 oxidation state in saltstone and other cementitious materials containing blast furnace slag. However, the PA does not make an explicit assumption about the exact chemical forms of Tc in oxidizing or reducing saltstone.

Chemical degradation of saltstone controls the chemical composition of pore fluids passing through the saltstone, and ultimately influences the release of contaminants from the waste form. In the PA, chemical degradation of saltstone is modeled by transitions from one defined age–redox state to another. The “age” of the saltstone is determined based on the pH of the pore fluid traversing through the material. A pH value between ~13.5 and 12.5 indicates Region I (young age), a pH of 11 or greater indicates Region II (middle age), and a lower pH level indicates Region III (old age) (Bradbury and Sarott, 1995). In the model, the saltstone is assigned an age–redox state based on the pH and E_h of the system, which transitions as the pH decreases (from Region II to Region III) and the E_h increases (from reducing to oxidizing conditions). In the PA, for modeling of all elements except Tc, changes in pH or oxidation state are treated as functions of the pore water volume that travels through the saltstone. As discussed in more detail in Section 2.7, the release model from Tc uses a shrinking-core model of saltstone oxidation. The pore water volume traveling through saltstone is dependent on the infiltration rate and the saltstone hydraulic conductivity.

2.6.2 Quality Assurance for Waste Form

DOE does not directly discuss the QA program for the saltstone waste form in the PA. However, DOE has previously provided information on the QA for saltstone to the NRC during the monitoring process, particularly as part of addressing NRC Open Issues 2007-1 and 2007-2. A further discussion on DOE's QA program for the saltstone waste form is located in Section 2.6.4.1.

2.6.3 Modeling of Waste Form Degradation

The Case A simulation did not consider physical degradation of saltstone, such as fracturing or increased hydraulic conductivity of the saltstone matrix, even though physical degradation may ultimately affect the rate of chemical degradation by influencing the rate of infiltration through the waste form. The Case A assumption that physical degradation of the saltstone will not occur during the performance period is based on a thermodynamic and mass balance analysis (WSRC-STI-2008-00236) that DOE characterized as preliminary research (NRC, 2008e). This work was based on a simulation, implemented with the Geochemist's Workbench® geochemical code, of the formation of expansive mineral phases in saltstone due to exposure to rainwater or rainwater equilibrated with cement (WSRC-STI-2008-00236). The initial saltstone mineralogy assumed in the analysis comprised CSH, hydrotalcite, gibbsite, quartz, hematite, and gypsum, which is different from the mineralogy assumed in the analysis supporting the modeled E_h and pH transitions in saltstone (SRNL-TR-2008-00283). The author concluded that fracturing of saltstone is unlikely because expansive phase formation will cause a maximum 34% loss in porosity. However, this value is slightly higher than 30%, which is the median cement porosity for several cements that can be filled by expansive phases before fracturing can initiate (Tixier and Mobasher, 2003). Furthermore, the author acknowledged that the fraction of porosity that must be filled by expansive phases before fracturing can occur is uncertain (WSRC-STI-2008-00236).

Case A also neglects other types of degradation, such as shrinkage cracking, steel corrosion-induced cracking, or dissolution of salts and low solubility matrix phases. However, alternative scenarios were considered in PA sensitivity analyses that evaluated the consequences of a hypothetical degraded saltstone condition. Case E is similar to Case A in that no fast flow paths are assumed to exist, but the saltstone is assumed to be severely degraded at time zero. In this scenario, the degraded saltstone was modeled by increasing the saltstone hydraulic conductivity to 1.7×10^{-3} cm/sec and modifying the moisture characteristic curves based on separate DOE analyses (SRNL-STI-2009-00115). In Case C and the Synergistic Case, fractures through the saltstone were included.

Case K was constructed to provide additional understanding of the impacts of physical degradation of cementitious materials through time and other NRC concerns (Section 2.1). In Case K, complete physical degradation of saltstone was assumed to occur in 10,000 years, which was represented in the PA by increasing the saltstone saturated hydraulic conductivity and effective diffusivity with time. Case K applied a simplified, conservative assumption that saltstone saturation does not affect its hydraulic conductivity (i.e., the relative permeability is always 1). With respect to oxidation of slag-bearing grout in Tc-99 transport modeling, Case K also assumed the saltstone grout will form fractures, with spacing that decreases to 0.1 m in 10,000 years and a frequency that increases logarithmically with time. Values of saturated hydraulic conductivity and effective diffusivity assumed for Cases A, E, and K are listed in Table 2.6-3.

Table 2.6-3: Saturated Hydraulic Conductivity and Effective Diffusivity Used in Selected Cases

	Case A	Case E	Case K
Saturated Hydraulic Conductivity	2.0×10^{-9} cm/sec; invariant with time	1.7×10^{-3} cm/sec; invariant with time	Initial value: 1.0×10^{-8} cm/sec; Value at 10,000 years: 1.0×10^{-6} cm/sec
Effective Diffusivity	1.0×10^{-7} cm^2/sec; invariant with time	1.0×10^{-7} cm^2/sec; invariant with time	Initial value: 1.0×10^{-7} cm^2/sec; Value at 10,000 years: 5.0×10^{-6} cm^2/sec

Values taken from 2009 PA (Table 4.2-16) and SRR-CWDA-2010-00044

To determine the number of pore fluid volumes required to transition from one saltstone age–redox state to another, DOE conducted thermodynamic simulations using the Geochemist's Workbench® geochemical code (SRNL-TR-2008-00283). The simulations mimicked column experiments where at each simulation step a pore volume of infiltrate enters the column and equilibrates with the saltstone and an equal volume of reacted infiltrate is displaced and exits the column. The saltstone mineralogy that was used as input to the thermodynamic simulations was derived using a normative calculation that assigned chemical constituents determined by bulk chemical analysis to specific mineral phases. The derived saltstone mineralogy is comprised of calcium-silicate-hydrate (CSH), hydrotalcite, kaolinite, quartz, hematite, and gypsum. To represent the reducing component of slag, pyrrhotite was included in the model in an amount based on the measured saltstone reducing capacity[15] (SRNS-STI-2008-00045). Three different infiltrating fluid compositions were modeled as reacting with saltstone. The first is groundwater unreacted with concrete, which represents the scenario with a fast flow path through the concrete in which unreacted groundwater contacts the saltstone. The second composition is groundwater equilibrated with an oxidized, Region III (old age) disposal unit concrete defined as "groundwater equilibrated with calcite." The third infiltrating fluid composition was modeled as being equilibrated with an oxidized, Region II (middle age) future disposal cell concrete and defined as "groundwater equilibrated with CSH." The groundwater composition used in the simulations is based on an analysis of a sample from a water table monitoring well in the vicinity of the SDF. The dissolved oxygen concentration in groundwater used in the simulations was 8 mg/L - higher than the 1.2 mg/L measured from the well water sample - based on assumed groundwater equilibrium with atmospheric oxygen.

[15]The measured reducing capacity of simulated saltstone, which contained 23 wt% blast furnace slag, (SRNS-STI-2008-00045) was 822 μeq/g, which was approximately the same value as the reduction capacity Kaplan, et al. measured for 100% blast furnace slag.

The simulation results for Case A indicated that the E_h transitions from reducing to oxidizing conditions (from –0.45 to +0.6 V) at about the same number of pore volumes for the three infiltrate compositions (2,734, 2,775, and 2,806, respectively, for the first, second, and third infiltrating solution compositions). The pH transitions (from 11 to ~10) for the first and second fluid compositions occurred at 2,274 and 2,558 pore volumes, respectively. The simulation using the third infiltrating solution composition terminated before a pH transition occurred; a pH transition pore volume of 10,422 was derived based on extrapolation of the simulation results and an assumption that the pH transition occurs when the mass of CSH is completely consumed. The calculated fluid pore volumes for E_h transitions are similar for all three cases because the dissolved oxygen concentration in the three reacting fluids is the same, whereas the pH transitions are different because the pH values of the reacting fluids are different. The Case A results indicated that the saltstone remains in a reducing middle age state throughout the performance period. DOE considered an alternative scenario (Case K) in which the saltstone mineralogy has three-fourth less pyrrhotite and, therefore, three-fourth less reducing capacity than the saltstone considered in Case A[16]. The saltstone effective porosity used in determining the transition volumes for Case K was assumed to be 58%, which is based on measured values reported in SRNL-STI-2008-00421, compared to the value used in determining the transition volumes for Case A (42.3%) (SRNL-TR-2008-00283). Based on the lower reducing capacity and higher effective porosity of the saltstone compared to the Case A values, the pore volumes for the E_h and pH transitions used in the PA Case K scenario were decreased from Case A values to 505 and 7,608, respectively. The DOE analysis of E_h and pH transitions assumed that 100% of the minerals in the saltstone are available for reaction with infiltrating fluid. DOE acknowledged that if this assumption is not true, either because of fracturing or occlusion of portions of the cementitious materials from infiltrating fluid, the number of pore volumes required to reach E_h and pH transitions will be lower. In Case K, the fraction of Tc that is oxidized was determined based on the diffusion of oxygen into the monolith from fractures that form over time.

2.6.4 NRC Evaluation – Waste Form

2.6.4.1 NRC Evaluation – QA for Waste Form

In October 2007, NRC conducted an onsite observation that addressed, in part, DOE's quality assurance program related to the saltstone waste form (NRC, 2008a). As a result of that observation, the NRC staff concluded that certain aspects of DOE's quality assurance program were satisfactory but that certain others should be tracked as open issues related to waste form variability. Specifically, in 2007, the NRC staff determined that DOE had a program for verifying that the dry bulk grout materials conform to applicable American Society for Testing and Materials (ASTM) standards and that the program is implemented effectively. The NRC staff did recommend that, because of the importance of reducing capacity of saltstone to achieving the

[16]DOE reduced the saltstone reducing capacity in Case K because of an NRC staff concern regarding uncertainties in the saltstone reducing capacity. As stated in footnote-6, SRNS-STI-2008-00045 measured a reduction capacity of saltstone approximately equal to that of pure blast furnace slag, even though blast furnace slag comprised only approximately one-fourth of the saltstone mixture composition.

§61.41 performance objective, DOE should consider performing independent characterization of the slag upon receipt of the material rather than relying upon the vendor's documentation.

Although the quality assurance program for the composition of the dry bulk materials was determined to be implemented effectively, the NRC staff identified other aspects of the saltstone production and emplacement processes that may not be effectively controlled. Open issue 2007-1 indicates that, as a result of variations in the composition of saltstone grout actually produced at the SRS SPF, DOE should determine the hydraulic and chemical properties of as-emplaced saltstone grout. Open issue 2007-2 specifies that DOE should demonstrate that intra-batch[17] variability, flush water additions to freshly poured saltstone grout at the end of each production run, and additives used to ensure processability are not adversely affecting the hydraulic and chemical properties of the final saltstone grout. Open issue 2007-2 also indicates that DOE should show that the hydraulic and chemical properties of saltstone are consistent with the assumptions in the waste determination or show that any deviations are not significant with respect to demonstrating compliance with performance objectives.

To address the first part of Open Issue 2007-2 (i.e., related to intra-batch variability), DOE studied the variability in the fractions of the main components in 32[18] premix batches of the saltstone dry premix material (LWO-WSE-2009-00038). As part of its QA program, DOE controls the amount of fly ash, slag, and cement added to the saltstone dry premix material with "use every time" procedures. The saltstone dry premix material can be mixed with either automated or manual control. The 32 batches studied in 2009 (LWO-WSE-2009-00038) included 25 batches created with manual controls and 7 batches created with automated controls. There was no detectable difference between the batches created with automated and manual controls. In the PA, DOE reported that the 95% confidence interval for the fractional contribution of the main components extended ±3% from the nominal value for cement and ±2% of the nominal values for fly ash and blast furnace slag. Independent calculations that paired the ingredient masses with the total individual batch masses were essentially consistent with DOE's results and yielded 95% confidence intervals that vary ±3% from the nominal fraction for cement but only ±1% from the nominal fractions for slag and fly ash. These values reflect small uncertainties in the mean fractional values. However, the 95% confidence interval reflects uncertainty in the mean rather than batch-to-batch variability. With a sufficient number of measurements, the uncertainty in the mean can be quite small even if the batch-to-batch variability is large. A more appropriate measurement of the batch-to-batch variability is the relative standard deviation of the fractional composition of the premix from batch to batch. Pairing the ingredient weights with the individual batch weights, which reduces measured variability in the fractional contribution of each of the ingredients, yields relative standard deviations of ±2% in blast furnace slag and fly ash and ±9% in cement. The study did not

[17] In this context, "intra-batch variability" refers to the variability within a lift of saltstone. A lift of saltstone is typically comprised of many of the individual premix batches described in the following paragraph.
[18] In the PA, DOE indicated that the saltstone batch variability study (LWO-WSE-2009-00038) included 31 batches. Although 31 batches appear to have been included in the summary statistics, data for 32 batches are provided in the data sheets included in the report. No reason was given for excluding the last batch, which did not appear to differ significantly from the other batches. Including the last batch did not affect the summary statistics significantly.

include statistics regarding any other aspects of Open Issue 2007-2, including the variability in the composition and mass of admixtures added to each batch or the salt waste to premix ratios.

Like variations in the premix components, variations in the composition of salt waste may affect finished saltstone properties. To comply with South Carolina Hazardous Waste Management Regulations (SCHWMR), samples of saltstone made with samples of salt waste from the SPF feed tank (Tank 50) are checked quarterly with the EPA Toxic Characteristic Leaching Procedure (TCLP). Leachate taken with the TCLP is analyzed for the 8 listed toxic metals arsenic, barium, cadmium, chromium, mercury, lead, selenium, and silver. Because the unstabilized salt waste exceeds regulatory limits for several underlying hazardous constituents, the leachate also is tested for antimony, beryllium, nickel, and thallium. The TCLP results continue to show saltstone adequately retains the measured toxic metals and underlying hazardous constituents and is therefore a non-toxic, non-hazardous waste form in the context of the SCHWMR (e.g., SRNL-STI-2011-00262, SRNL-STI-2011-00561).

Like the chemical properties, physical properties of finished saltstone also are affected by variations in salt waste composition. Caustic sludge washing prior to sludge vitrification is expected to increase the concentration of aluminate in salt waste treated in the SWPF as compared to salt waste treated with the ARP/MCU process (SRNL-STI-2009-00184). Near baseline levels (0.11 M for SWPF waste and 0.05 M for ARP/MCU waste), increased aluminate appears to decrease saltstone porosity, which would be expected to decrease the hydraulic conductivity, if the curing temperature is controlled at approximately 22ºC (72ºF) (SRNL-STI-2009-00184). However, at higher concentrations (0.45 M to 0.65 M), increased aluminate in simulated SWPF samples appears to decrease the Young's modulus of cured saltstone, which is correlated with increased permeability (SRNL-STI-2009-00810). In addition, simulated ARP/MCU and SWPF samples with greater aluminate concentrations (i.e., greater than approximately 0.35 M in simulated SWPF samples and 0.22 M in simulated ARP/MCU simulated) appear to be more prone to cracking (SRNL-STI-2009-00184, SRNL-STI-2009-00546, SRNL-STI-2009-00810). Increasing the aluminate content of the salt waste also has been shown to increase the heat of hydration of saltstone (WSRC-STI-2007-00506 and SRNL-STI-2009-00546), which could increase curing temperatures if the curing temperature is not carefully controlled. Because curing temperature itself has a significant effect on saltstone properties, as discussed in greater detail later in this section, the net effect of increased aluminate concentrations is likely to depend in part on saltstone curing temperature control. This was confirmed by more recent research on the effect of aluminate on hydraulic conductivity (SRNL-STI-2011-00665) that found that a higher concentration of aluminate (0.28 M) reduced the hydraulic conductivity in samples that had the curing temperature controlled to 20ºC or 40ºC.

To begin to quantify the effects of these additional sources of variability, DOE studied the hydraulic and sorptive properties of laboratory-prepared saltstone samples prepared with different compositions and curing temperatures (SRNL-STI-2009-00419). Specifically, DOE examined the effects of admixtures (i.e., a set retarder and an anti-foam agent[19]), organic content, salt waste to premix ratio, aluminate concentration, and curing temperature on the porosity and saturated hydraulic conductivity of samples made with simulated ARP/MCU treated salt waste. Because of the small number of samples in each category (i.e., n = 3 or 6 for hydraulic conductivity and n=7 or 13 for porosity and density), the analysis of variance DOE performed had limited statistical power. In this study, DOE did not detect any effect of admixtures, organic content, or aluminate concentration. However, DOE did find statistically significant effects of salt waste to premix ratio and curing temperature.

Specifically, DOE found that varying the salt waste to dry premix ratio from the nominal value of 0.60 to 0.65 caused a six-fold increase in hydraulic conductivity (from 1.4×10^{-9} cm/s to 8.4×10^{-9} cm/s) (SRNL-STI-2009-00419). This result was consistent with the results of previous studies that demonstrated an increase in saltstone porosity, which is expected to cause an increase in hydraulic conductivity, with increasing liquid waste to dry premix ratio (WSRC-STI-2007-00352). This effect may be important to long-term waste form performance because DOE allows water used to flush the saltstone emplacement lines to enter the disposal cells immediately after the saltstone is emplaced (i.e., before it is cured). As identified in NRC's October 2007 monitoring report (NRC, 2008a), the extent to which this water mixes with the saltstone and affects the liquid to dry premix ratio is unknown. However, the results of DOE's laboratory tests (SRNL-STI-2009-00419) indicate that even a small amount of mixing is likely to degrade the saltstone hydraulic conductivity.

Curing temperature also appears to have a significant effect on the hydraulic properties of saltstone. Increased initial curing temperature from 22ºC to 60ºC (72ºF to 140ºF) appears to increase saltstone porosity (SRNL-STI-2009-00184) and decrease the Young's modulus of simulated saltstone samples (SRNL-STI-2009-00810), both of which are correlated with increased hydraulic conductivity. When hydraulic conductivity was measured directly, DOE found that samples cured at 60ºC (140ºF) had an average hydraulic conductivity of 8.0×10^{-7} cm/s, which is more than 500 times greater than similar samples prepared at 20ºC (72ºF) and 400 times greater than the base-case hydraulic conductivity assumed in the PA (SRNL-STI-2009-00419). DOE subsequently suggested that these results may have overrepresented the effects of curing temperature because the samples cured at 60ºC (140ºF) also were cured in a low-humidity environment. Grout cracking due to drying is well known (Pabalan et al, 2009). However, DOE previously found that saltstone samples cured at 90ºC (190ºF) in closed containers cracked during curing even though the samples did not dry (WSRC-TR-98-00337). Thus curing at 90ºC (190ºF) does appear to impact saltstone cracking

[19] DOE found that an anti-foam agent and set retarder were necessary to achieve the desired processability of saltstone made with H-canyon waste (WSRC-TR-2005-00149). A subsequent study confirmed the correlation between the fraction of H-canyon low-activity waste in the saltstone salt waste mixture and the need for an anti-foam agent and set retarder (SRNL-STI-2010-00522). The same study also found correlations between the fraction of Effluent Treatment Project waste and foam formation and General Purpose Evaporator waste and foam persistence (SRNL-STI-2010-00522).

through at least one mechanism in addition to drying. Although samples cured at 70ºC (160ºF) in a closed container were not visibly cracked, the effects on the hydraulic properties were not measured (WSRC-TR-98-00337), so it is unknown if the high temperature affected the hydraulic properties. Thus it is not certain that the elevated hydraulic conductivity DOE measured in samples cured at 60ºC (140ºF) in a low-humidity environment was entirely attributable to drying. Taken together, these results suggest the importance of good control of curing temperature to the hydraulic properties of as-emplaced saltstone. The results also suggest that laboratory samples will provide a good indication of the properties of field-scale saltstone only if the laboratory samples realistically reflect the curing temperature profile of field samples.

Newer research considered the effect of variations in the curing temperature, water to cement ratio, aluminate concentration, and amount of fly ash on the hydraulic conductivity of simulated ARP/MCU samples (SRNL-STI-2011-00665). It was found that the cure temperature had the largest effect on the hydraulic conductivity. As described above, there was also some effect of the aluminate concentration on the hydraulic conductivity. The measured hydraulic conductivity values ranged from 2.9×10^{-9} to 4.9×10^{-8} cm/s for samples cured at 20ºC, 1.0×10^{-9} to 1.9×10^{-6} cm/s for samples cured at 40ºC, and 6.2×10^{-8} to 1.3×10^{-6} cm/s for samples cured at 60ºC. The humidity was not controlled in these samples, so grout cracking due to drying could be contributing to the higher hydraulic conductivity in these samples.

In October 2007, DOE indicated that thermocouples are used within the vault and saltstone to monitor curing temperatures (NRC, 2008a). At that time, the maximum observed temperature in saltstone during curing was approximately 50ºC (120ºF). Because of the potential significance of curing temperatures on the hydraulic properties of saltstone, the NRC staff inquired about the curing temperatures for saltstone grout during the April 26, 2011, monitoring observation (NRC, 2011k). Specifically, the NRC staff asked about the potential impact of high-aluminate salt waste on curing temperatures. DOE stated that cure temperature profiles for saltstone are being compiled. As indicated in the summary of the NRC staff's April 2011 monitoring observation (NRC, 2011k), NRC staff will review the cure temperature profiles for saltstone when DOE compiles them following future testing.

To begin to understand potential differences between the properties of laboratory-prepared and field-emplaced saltstone (Open Issue 2007-1), DOE collected nine core samples from Vault 4, Cell E in September 2008. Of the nine samples collected from Vault 4 in September 2008, one was used for bulk density and porosity measurements and three were used for chemical phase measurements (SRL-STI-2009-00804). DOE reported that the measured bulk density after saturation with simulated salt solution was 1.9 g/cm^3 and the porosity was 0.599 (SRL-STI-2009-00804). X-ray diffraction measurements of three of the samples showed chemical phases typical of fly ash and calcium silicate hydrate (CSH). Based on these results, DOE concluded that the Vault 4 cored samples were consistent with laboratory-prepared samples (SRL-STI-2009-00804). However, permeability testing of five of the Vault 4 samples collected in September of 2008 showed that the saturated hydraulic conductivity of the Vault 4 samples was significantly greater than the hydraulic conductivity of laboratory-prepared samples (SRNL-STI-2010-00657). Specifically, the hydraulic conductivity of the field samples had an arithmetic mean of $4.0 \times 10^{-7} \pm 1.9 \times 10^{-7}$ cm/s (mean ± s.e.) and a median of 1.8×10^{-7} cm/s, which is approximately two orders of magnitude greater than the saturated hydraulic conductivity DOE used in its base-case PA model (i.e. 2.0×10^{-9} cm/s) and 18 times greater than the initial hydraulic conductivity DOE used in Case K (i.e., 1.0×10^{-8} cm/s).

In response to NRC's second RAI, DOE hypothesized that the relatively high hydraulic conductivity of the core samples taken from Vault 4 Cell E is an artifact of the sample collection method (SRR-CWDA-2011-00044; page 177). However, samples taken from laboratory-prepared blocks of simulated saltstone with the same coring method that was used to collect samples from Vault 4 had hydraulic conductivities similar to the hydraulic conductivities of laboratory samples prepared in molds (i.e., average hydraulic conductivity of 4.1×10^{-9} cm/s for cored samples from laboratory-prepared blocks and an average of 5.4×10^{-9} cm/s for samples made in molds) (SRNL-STI-2010-00657). Thus the relatively high hydraulic conductivity of Vault 4 samples may not be attributable to the coring method, unless the effects of coring in the field and laboratory are significantly different despite the same coring method being used.

In its PA, DOE identified Open Issues 2007-1 and 2007-2 related to the variability of as-emplaced Saltstone as areas of ongoing and future work. Because DOE hypothesizes that coring samples of saltstone in the field artificially increases the hydraulic conductivity of the samples, DOE is developing a formed-core sampling technique. The formed-core technique entails placing sampling tubes in the saltstone vaults prior to saltstone grout pours, and removing the sampling tubes (with samples) after the saltstone has cured (SRNL-STI-2010-00167). Hydraulic conductivity tests of these samples may provide information relevant to Open Issue 2007-1, related to the hydraulic properties of field-emplaced saltstone.

2.6.4.2 NRC Evaluation – Modeling of Physical and Hydraulic Properties of Waste Form

The NRC staff reviewed the DOE analysis of the initial properties of saltstone and waste form degradation, as described in PA Section 4.2 and supporting documents, the information DOE provided in its responses to NRC RAIs, as well as literature information. The values of hydraulic conductivity, porosity, dry bulk density, and particle density assumed in Case A are based on laboratory measurements on saltstone simulants (SRNL-STI-2008-00421). The effective diffusion coefficient is based on a value for concrete recommended in WSRC-STI-2006-00198. The PA values assumed for the saltstone porosity, bulk density, and particle density are reasonable based on comparison with measured values (Table 2.6-4).

The modeled hydraulic conductivity of saltstone in Case A (i.e., 2×10^{-9} cm/s) is generally in the range of the measured hydraulic conductivity of laboratory-prepared samples cured at low temperatures (Table 2.6-5). However, the hydraulic conductivity measured using groundwater equilibrated with vault concrete simulant as the permeating fluid instead of simulated saltstone pore fluid was almost an order of magnitude higher (1.5×10^{-8} cm/s) (WSRC-STI-2007-00649). The fluid flowing into and through the saltstone is expected to be more consistent with the groundwater equilibrated with vault concrete than simulated saltstone pore fluid, so this result may be more applicable than the hydraulic conductivity measured in other samples with simulated saltstone pore fluid. The hydraulic conductivity values reported in SRNL-STI-2011-00665 for simulated ARP/MCU saltstone cured at 20ºC (2.9×10^{-9} cm/s to 4.9×10^{-8} cm/s) were also higher than the Case A value. As noted in Section 2.6.4.1, samples cured at elevated temperatures had measured hydraulic conductivity values as high as 1.9×10^{-6} cm/s (SRNL-STI-2011-00665), though this may be due to drying cracking in the laboratory. Additionally, the hydraulic conductivity of 2×10^{-9} cm/s assumed in Case A appears to be optimistic as compared to the measured hydraulic conductivity of cores taken from Vault 4 (Table 2.6-5). As previously discussed, the hydraulic conductivity of the field samples had an arithmetic mean of 4.0×10^{-7} cm/s ± 1.9×10^{-7} cm/s (mean ± s.e.) and a median of 1.8×10^{-7} cm/s, which is approximately two orders of magnitude greater than the saturated hydraulic conductivity DOE used in its base-case PA model (i.e., 2.0×10^{-9} cm/s) and 18 times greater than the initial hydraulic conductivity DOE used in Case K (i.e., 1.0×10^{-8} cm/s). DOE has hypothesized that the measured values from Vault 4 samples are not representative of the actual hydraulic conductivity of saltstone in Vault 4 and instead represent an artifact of the coring method used. This hypothesis is not supported by the results of experiments that show samples cored from laboratory-prepared blocks using the same coring technique had measured hydraulic conductivity values similar to laboratory-prepared samples prepared in molds (i.e., not cored) (SRNL-STI-2010-00657). These results suggest the difference in the measured hydraulic conductivity of cored laboratory-prepared samples and cored field-emplaced samples may be attributable to the difference between laboratory and field saltstone preparation and curing rather than the coring method. Potential mechanisms for these differences include variations in curing temperature and the liquid to premix ratio in the field, as well as heterogeneity in physical properties at the field scale that is not captured in laboratory-scale samples. Alternately, the difference may be attributable to some difference between field and laboratory sampling even when the same sampling method is used (Section 2.6.4.1). If field-prepared saltstone has a significantly greater saturated hydraulic conductivity than laboratory-prepared saltstone, and, consequently, a significantly greater saturated hydraulic conductivity than DOE used in Case A, Case A would likely significantly under-represent radionuclide release from the saltstone. Although the measured hydraulic conductivity in core samples also exceeds the initial value of hydraulic conductivity used in Case K, the increase in hydraulic conductivity with time used in Case K is expected to capture an appropriate range of hydraulic conductivities.

The effective diffusivity value of 1×10^{-7} cm^2/s assumed by DOE in Case A was based on information in WSRC-STI-2006-00198. In this document, a literature review of effective diffusivity values reported values that ranged from 1.44×10^{-8} cm^2/s to 4×10^{-7} cm^2/s. Based on this data, the authors concluded that 1×10^{-7} cm^2/s was a representative value for the effective diffusivity in ordinary quality concrete. In this same report, research on the effective diffusivity of simulated saltstone, which was prepared with a slightly different formulation than is currently used, was also summarized. These effective diffusivity values ranged from 8×10^{-10} cm^2/s to 9×10^{-9} cm^2/s. A more recent study of the effective diffusion coefficient in simulated saltstone reported values less than 1×10^{-8} cm^2/s (SRNL-STI-2010-00515). Based on this information, NRC staff finds that the effective diffusion coefficient used by DOE is reasonable. However, NRC staff notes that research has not been performed on the effective diffusivity in as-emplaced saltstone, and, as noted above in the discussion on hydraulic conductivity, the physical properties of as-emplaced saltstone may differ significantly from lab-prepared simulated saltstone.

Table 2.6-4: Modeled and Measured Porosity (η), Bulk Density (ρ_b), and Particle Density (ρ_p) Values for Saltstone

Modeled Values		**Notes**
All Cases	η: 58% ρ_b: 1.01 g/cm^3 ρ_p: 2.40 g/cm^3	Assumed invariant with time
Measured Values		**Notes**
SRNL-STI-2008-00421; Table 31	η: 54 to 57% ρ_b: 1.04to 1.08 g/cm^3 ρ_p: 2.32 to 2.48 g/cm^3	Range of values measured for DDA saltstone cured for 28 or 90 days[1]
	η: 58 to 61% ρ_b: 0.95 to 1.01 g/cm^3 ρ_p: 2.35 to 2.49 g/cm^3	Range of values measured for ARP/MCU saltstone cured for 28 or 90 days
	η: 56 to 60% ρ_b: 1.00 to 1.05 g/cm^3 ρ_p: 2.35 to 2.53 g/cm^3	Range of values measured for SWPF saltstone cured for 28 or 90 days
Pabalan, et al. (2011)	η: 52 and 67%	Porosity of crushed and sieved DDA saltstone simulant used in two column leaching experiments
	ρ_p: 2.47 g/cm^3	Crushed and sieved DDA saltstone simulant

[1] DDA = Deliquification, Dissolution, and Adjustment salt simulant; ARP/MCU = Actinide Removal Process/Modular Caustic Side Solvent Extraction Unit salt simulant; SWPF = Salt Waste Processing Facility salt stimulant

The NRC staff believes that the Case A assumption that saltstone will be hydraulically undegraded for 20,000 years is unrealistically optimistic, and that this assumption is inconsistent with observations of existing cracks in Vault 4 saltstone (SRNL-ESB-2008-00017, SRR-CWDA-2011-00105). In the response to NRC comment SP-2 (SRR-CWDA-2011-00044), DOE stated that they do not believe that there is evidence of these cracks extending below the surface of the monolith. However, in SRR-CWDA-2011-00105, it is noted that the depth and thickness of these cracks is not known. Therefore NRC staff believes that there is insufficient evidence that these cracks do not extend below the surface. Furthermore, in the PA, DOE considered only sulfate attack and did not provide an adequate basis for neglecting other types of degradation such as shrinkage cracking, steel corrosion-induced cracking, cracking due to settlement or earthquakes, and dissolution of salts and low solubility matrix phases. Of these mechanisms, DOE estimates of fracturing were found only for static and dynamic settlement of Vault 4 saltstone (T-CLC-Z-00006). The settlement analysis assumed static settlement caused cracking in saltstone along Vault 4 construction joints (at 9.1 m [30 ft] intervals) and between the saltstone and vault walls. Dynamic settlement was assumed to cause cracking at 15.2 m (50 ft) intervals. The analysis calculated increases in crack length and width as a function of time, assuming 3 earthquakes occurred at random times between 100 and 10,000 years.

The NRC staff also is concerned about the DOE conclusion that saltstone fracturing by expansive phases due to sulfate attack is unlikely. The conclusion is based on geochemical modeling results that are unsupported by comparisons with empirical data or observations. Also, the geochemical equilibrium model ignored reaction kinetics that could result in metastable product formation often associated with an increase in volume. The effects of organic additives or pozzolanic replacement on the dissolution and precipitation of cement-related compounds, which may have an effect on the generation of expansive phases, also were not considered. NRC issued comment SP-1 (NRC, 2010b, i) asking for additional justification for this DOE conclusion. In its response to NRC comment SP-1 (SRR-CWDA-2011-00044), DOE stated that research into saltstone material degradation is planned or ongoing. NRC staff still has concerns associated with saltstone fracturing by expansion and suggests future research could (i) consider the effect that sulfide from the blast furnace slag might have on the phases and reactions present in this system, (ii) include experiments that are designed to collect data on initial mineralogical conditions, fundamental thermodynamic data and reaction kinetics, and (iii) consider expansive phases produced by intermediate or metastable reaction products.

The NRC staff understands fracturing was addressed non-mechanistically in the PA sensitivity analyses by Case C and the Synergistic Case, which consider fractured saltstone, and by Cases B and D, which postulate a gap between the saltstone and disposal unit wall. However, NRC believes these cases contain other assumptions, which are not representative of the expected performance of the saltstone (e.g., use of overly-optimistic moisture characteristic curves and potentially overly-optimistic saltstone hydraulic conductivity). In NRC comment PA-8 (NRC, 2010i), NRC staff expressed concern with DOE's use of Case A as the base case. In its response to NRC concerns (SRR-CWDA-2011-00044; PA-8), DOE maintained that Case A is valid, however, DOE proposed an alternative scenario (Case K) in which saltstone grout will develop fractures with spacing that decreases and frequency that increases with time such that within 10,000 years after closure the saltstone has a hydraulic conductivity and diffusivity similar to soil. The Case K final fracture spacing at 10,000 years after closure is one thoroughgoing fracture every 10 cm (4 in). The semi-log fracture growth relationship used in Case K results in minimal cracking at early times followed by a rapid increase in the number of fractures between 8,000 and 10,000 years after site closure. There is significant uncertainty in the rate and extent of fracturing because of the lack of experience with engineered sub-surface cementitious materials several thousand years old or applicable natural analogs. The NRC staff believes that a modeling approach in which the rate of fracturing increases with time is reasonable because, as the saltstone degrades, the increase in fractures would lead to increased flow through the monolith, which could then accelerate the formation of new fractures. However, a number of alternate fracture growth curves other that the semi-log model DOE used could be valid. These models could result in a different rate of fracture formation and, potentially, earlier significant fracturing. If earlier fracturing occurred, the dose peaks would move to earlier times. The NRC staff evaluated the model sensitivity to the assumed rate of fracture growth, as discussed in Section 2.13.

Table 2.6-5: Modeled and Measured Initial Hydraulic Conductivity Values for Saltstone

	Hydraulic Conductivity (cm/sec)	Notes
Modeled Values		
Case A	2.0×10^{-9}	Assumed invariant with time
Case E	1.7×10^{-3}	Assumed invariant with time
Case K	1.0×10^{-8}	Value at 10,000 years: 1.0×10^{-6} cm/sec
Measured Values – Lab Prepared Samples		
WSRC-STI-2007-00056	1.4×10^{-9} to 3.4×10^{-9}	MCU saltstone; measured using a beam-bending technique
SRNL-STI-2008-00421; Table 30	2.5×10^{-9} to 9.3×10^{-9}	Range of logarithmic averages of hydraulic conductivities measured for DDA, ARP/MCU, and SWPF saltstone; 28-day curing period[1]
	9.6×10^{-11} to 6.0×10^{-9}	Range of logarithmic averages of hydraulic conductivities measured for DDA, ARP/MCU, and SWPF saltstone; 90-day curing period
WSRC-STI-2007-00649	1.5×10^{-8}	MCU saltstone; groundwater equilibrated with vault concrete simulant was used as permeating fluid
	5.3×10^{-9}	MCU saltstone; saltstone pore fluid simulant was used as permeating fluid
SRNL-STI-2009-00419	1.7×10^{-10} to 9.9×10^{-9}	Range of hydraulic conductivities measured for various formulations of simulated saltstone
	8.0×10^{-7}	Sample cured at 60ºC at low relative humidity
SRNL-STI-2011-00665	2.9×10^{-9} to 4.9×10^{-8}	ARP/MCU saltstone cured at 20ºC without control of humidity; permeating fluid is low aluminate salt solution [2]
	1.0×10^{-9} to 1.9×10^{-6}	ARP/MCU saltstone cured at 40ºC without control of humidity; permeating fluid is low aluminate salt solution [2]
	6.2×10^{-8} to 1.3×10^{-6}	ARP/MCU saltstone cured at 60ºC without control of humidity; permeating fluid is low aluminate salt solution [2]
Measured Values – Saltstone Core Samples		
SRNL-STI-2010-00657	3.9×10^{-7}	Arithmetic average of 10 measurements of 5 cores taken from Vault 4, Cell E

[1] DDA = Deliquification, Dissolution, and Adjustment salt simulant; ARP/MCU = Actinide Removal Process/Modular Caustic Side Solvent Extraction Unit salt simulant; SWPF = Salt Waste Processing Facility salt stimulant

[2] Samples prepared using a range of water to cement ratios, aluminate concentration, and fly ash

2.6.4.3 NRC Evaluation – Modeling of Chemical Properties of Waste Form

The NRC staff is concerned that the DOE conclusion the saltstone will remain reducing and middle-aged through the performance period relies on the results of geochemical equilibrium modeling (SRNL-TR-2008-00283) that lacks adequate model support. NRC staff is concerned that the E_h transition to oxidizing conditions may occur sooner than DOE derived from its geochemical modeling. The model results have not been validated by comparison to experimental or other data independent of model calculations. In its response to NRC comment SP-12 (SRR-CWDA-2010-00033), DOE acknowledged that its geochemical modeling results lack experimental verification and that source term model support is one area of future DOE work, potentially including physical testing to address issues that were evaluated only through Geochemist's Workbench® simulations. DOE also indicated in its response to NRC comment SP-15 (SRR-CWDA-2011-00044) that future work on saltstone degradation will include focus on the timing of saltstone transitions from reducing to oxidizing state. NRC staff is supportive of additional work to provide model support (e.g., pH and E_h measurements in accelerated physical testing using higher flow rates than anticipated in full-scale saltstone).

Depending on which initial mineralogy is more appropriate, a different conclusion could be reached regarding the likelihood of expansive phase formation or the calculated pore volumes for E_h and pH transitions. In its second response to NRC comment SP-8 (SRR-CWDA-2011-00044), DOE provided additional information on E_h and pH transitions times and variability in grout mineralogy. The Geochemist's Workbench® code was shown to use very similar saltstone mineralogies in reaction path calculations even if the input mineralogies are different because the code seeks an initial equilibrium assemblage prior to proceeding with the reaction path calculation. In addition to differences in saltstone formulations, differences in bulk density and porosity also can lead to differences in the predicted initial mineralogy. Additional Geochemist's Workbench® calculations, using two saltstone compositions with similar formulae but different bulk densities and porosities, showed that the pore volumes to reach the major E_h transition was 39% different between the two mineralogies and the pore volumes to reach the major pH transition was 42% different

The geochemical modeling of E_h transitions assumed all the reducing capacity in the saltstone is available for reaction with the infiltrate. Also, the reducing capacity of saltstone used in the model is based on laboratory measurements using slag samples that were finely ground to increase the reactive surface area (SRNS-STI-2008-00045). The NRC staff is concerned that these DOE measurements of saltstone reducing capacity (SRNS-STI-2008-00045) indicate the reducing capacity of saltstone is equivalent to the reducing capacity measured for pure blast furnace slag, even though the saltstone sample only contained 23 weight percent (wt%) blast furnace slag. The NRC staff also is concerned that, in actual field conditions, only a fraction of the slag will be accessible for reaction with the infiltrate and that the reactive surface area and reducing capacity of saltstone emplaced in the field are likely to be much smaller than that of finely ground laboratory samples. Thus, the longevity of reducing chemical conditions and the timing of release of redox-sensitive elements such as Tc, Pu, and Np, may be overestimated in the PA. In response to NRC comment PA-8 (SRR-CWDA-2011-00044) regarding the saltstone reducing capacity and E_h transition times assumed in the PA, DOE conducted additional analysis (Case K) in which the saltstone reducing capacity was reduced by a factor of four to account for the saltstone composition (25% blast furnace slag) and the saltstone porosity was changed from 42.3% to 58% to correct the porosity used in the initial E_h and pH transition estimate (SRNL-TR-2008-00283) to the measured porosity used in the PA. Based on the lower reducing capacity and higher effective porosity of the saltstone compared to the Case A values, the E_h transition used in the PA Case K scenario was decreased from the Case A value of 2,806 to 505 pore volumes. The NRC staff believes the decreased E_h transition time in Case K more reasonably reflects the expected evolution of the saltstone chemical environment.

The DOE conclusion that Tc release will be mitigated by the addition of blast furnace slag to the saltstone mixture appears to be supported by data from DOE studies (Langton, 1988) and other published literature (Brodda, 1988; Gilliam, et al., 1990; Aloy, et al., 2007). Leaching experiments indicated that blast furnace slag cement spiked with Tc-99 retained 99.9 percent of the Tc after 500 days of leaching in a simulated medium-level waste solution (Brodda, 1988), while others showed Tc leachability decreased by several orders of magnitude by the addition of blast furnace slag to the grout (Gilliam, et al., 1990). The reduction in Tc leachability in the presence of blast furnace slag has been attributed to the formation of relatively insoluble TcO_2 (Langton, 1988), Tc_3S_{10} (Lukens, et al., 2005), or Tc_2S_7 (Liu, et al., 2007). The Tc sulfide solids are considered to have formed by reaction of Tc with sulfide species released into solution by slag hydration.

However, as described in more detail below, other experiments have shown that a fraction of Tc may remain oxidized in saltstone formulations containing blast furnace slag. Thus, the NRC staff is concerned about the ability of saltstone to effectively reduce Tc, which is initially in the highly mobile Tc(VII) oxidation state, to its low solubility and highly sorptive Tc(IV) oxidation state. NRC staff has been monitoring information regarding Tc reduction in saltstone as part of Open Issue 2009-1 (NRC, 2009a). A DOE study (SRNS-STI-2008-00045) indicated that Tc apparently was not reduced in the experiments and the measured Tc K_d values (6.5 and 13.0 mL/g) were significantly less than the value (1,000 mL/g) assumed for Tc(IV) species and used in the PA. Although DOE recently measured higher K_d values (711 mL/g after 22 days of equilibrium and 581 mL/g after 56 days) (SRNL-STI-2010-00668), H_2(g), a strong reductant that could contribute to Tc reduction, was present in the experimental system. Because H_2(g) is not expected to be representative of SRS field conditions, any reduction attributable to H_2(g) would be an experimental artifact. In a different system, DOE measured a relatively low Tc K_d of 139 mL/g under reducing conditions using an actual Vault 4 saltstone sample (SRNL-STI-2010-00667). DOE attributed the low K_d to the saltstone sample not being fully reduced because of the presence of low concentrations of atmospheric oxygen (30 ppm to 60 ppm). In SRNL-STI-2010-00668, the authors also hypothesize that the sample color, which was olive green as opposed to black, indicated that the sample had become partially oxidized. The NRC staff believes that the color of the saltstone is insufficient evidence regarding the redox state of Tc because species other than Tc are likely responsible for the color change. Given that the saltstone sample was taken from Vault 4 nine months after emplacement, and the Tc was not fully reduced, the DOE result reinforces the NRC staff concern regarding the degree and the rate of Tc reduction by saltstone.

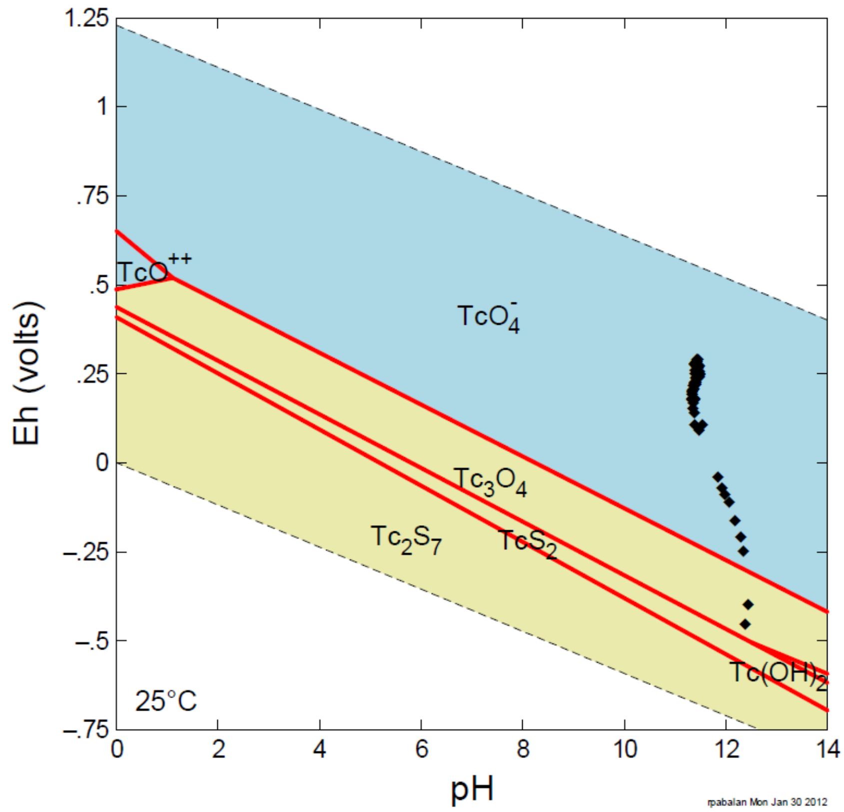

Figure 2.6-1: E_h–pH Diagram Showing Tc Speciation[20]

Tc is predicted to form a variety of different solids with sulfur depending on the system E_h and pH (Figure 2.6-1). In this diagram, the yellow region represents the stability field for Tc(IV) (i.e., the form of Tc that has low solubility and high sorption), and the blue region represents the stability field for Tc(VII). As noted in Section 2.6.1, DOE's measurements of initial reducing conditions were also as low as -585 mV, which is in the region of stability for Tc(IV). Additionally, Center for Nuclear Waste Regulatory Analyses (CNWRA) column experiments using simulated saltstone determined initial E_h values as low as -460 mV and a pH of approximately 12, which is well within the stability field for Tc(IV) (Figure 2.6-1). However, after approximately 30 pore volumes flowed through the column, the E_h and pH values for the simulated saltstone transitioned to the stability region for the highly mobile Tc(VII) (TcO_4^-). The applicability of these column experiments to the expected performance of the as-emplaced saltstone is described in greater detail in a subsequent paragraph.

Several studies of the redox state and speciation of Tc have shown the transformation of oxidized Tc(VII) to reduced Tc(IV) in blast furnace slag-containing grouts to be incomplete. Allen, et al. (1997) observed that, in blast furnace slag-containing cement mixtures with simulated radioactive waste solutions, $Tc(VII)O_4^-$ indicators persisted in XANES and EXAFS X-ray absorption spectra. Thirty to 90 percent of the Tc remained in the oxidized state even

[20] Calculated using Geochemist's Workbench® ACT2. (TcO_4^- Activity =10^{-8}; SO_4^{2-} Activity = 10^{-3}). Diamond symbols are experimental E_h and pH values for simulated saltstone measured by CNWRA. Yellow areas indicate solid phase stability fields and blue areas indicate aqueous phase stability fields.

after one month (Allen, et al., 1997, Figure 2). The $Tc(VII)O_4^-$ was eliminated only when Na_2S or FeS was added to the mixtures instead of blast furnace slag. The work of Lukens, et al. (2005) suggests that Tc reduction by blast furnace slag is chiefly due to reaction with sulfide, leading to precipitation of a Tc(IV) sulfide with the likely formula Tc_3S_{10}. Lukens, et al. (2005) prepared some of their Tc-doped grout samples with added Na_2S to ensure complete Tc reduction; the Tc in these samples was initially entirely Tc(IV) sulfide, according to EXAFS data. This pretreatment is not consistent with expected conditions for emplaced blast furnace slag-containing saltstone. Lukens, et al. (2005) prepared three additional samples using a Na_2S solution that had become oxidized prior to the experiment. The specific sulfur species present in that solution is unknown. However, the authors estimated that the solution contained SO_3^{2-} (0.1 M) and $S_2O_3^{2-}$ (1.4 M). In SRNL-STI-2010-00668, it was stated that the reductant in these samples can not be the Na_2S solution. However, the NRC staff believes that it is possible that the solution may have still contained some Na_2S that could have reduced some of the Tc. Additionally, Lukens, et al. (2005) note that SO_3^{2-} can react with Tc to form $TcO_2•2H_2O$. In these samples, approximately 40 percent of the Tc was initially in the form of $TcO_2•2H_2O$, indicating that the SO_3^{2-} may have reacted with the Tc. Also, in these samples, 16 to 20 percent of the Tc was initially in the form of the oxidized pertechnetate ion (TcO_4^-) (Lukens, et al., 2005, Figure 4 upper right; details of these results may be found in Shuh, et al., 2003). Even though there is some uncertainty in the effect of the presence of the oxidized Na_2S solution on the reduction of Tc in these samples, the results show that blast furnace slag alone is not sufficient to ensure full Tc reduction.

As pointed out by Kaplan, et al. (2011), in discussing the results found in Lukens, et al. (2005), the proportion of Tc that was in the oxidized form of TcO_4^- subsequently decreased over time when the samples prepared using the oxidized Na_2S solution were isolated from air. Lukens, et al. (2005) reported that TcO_4^- fell below 10 percent of total Tc in 18 months in all three samples and down to 3 percent in 30 months in the one sample that was not exposed to air. Lukens, et al. attributed this reduction to the blast furnace slag in the grout mixture, though NRC staff believes the presence of the oxidized Na_2S solution creates some uncertainty in this conclusion. The TcO_4^- was not observed to be completely reduced in these samples. This is an important point because even a small amount of mobile Tc under nominally reducing conditions could lead to higher Tc releases than DOE has modeled. Additionally, although the length of time found by Lukens, et al (2005) for Tc to become 97% reduced (i.e., 30 months) is small compared to the analysis period in the PA (i.e., 20,000 years), the initial presence of oxidized Tc could result in larger releases at early times than predicted. It also should be noted that these samples were not exposed to air, while actual saltstone is not isolated from the environment. The extent of reduction may therefore be less in the as-emplaced saltstone.

The NRC staff concludes that, while blast furnace slag certainly helps reduce Tc release by immobilizing it as Tc(IV) solids, it is not clear that all Tc will be initially in the reduced state in emplaced saltstone. There appears to be considerable uncertainty in the quantity of oxidized Tc potentially present in saltstone and other blast furnace slag-containing grouts with no additional reductants (see discussion in this section of Allen, et al., 1997, Lukens, et al., 2005). In addition, there remains uncertainty not only about the thermodynamic characteristics of the

Lukens, et al. (2005) proposed Tc(IV) sulfide, but also the relative quantities of different Tc(IV) solids in fresh saltstone.

DOE also acknowledged that only a small concentration of O_2(g) (e.g., 30 to 60 ppm) is needed to oxidize Tc(IV) to Tc(VII) (SRNL-STI-2010-00667). The oxidation rate will depend on the groundwater infiltration rate, the hydraulic properties of the saltstone, the ability of blast furnace slag to consume oxygen, and the kinetics of Tc oxidation. For example, bench-scale column experiments on Tc leaching from simulated saltstone performed at CNWRA indicated the system transitioned from reducing to oxidizing conditions after only tens of fluid pore volumes have passed through the column, which resulted in an enhanced release of Tc from the simulated saltstone. Although the bench-scale tests do not completely represent field conditions because of the relatively high flow rate through the system, the high reactive surface area of the crushed saltstone simulant, and the interaction of the particles with atmospheric oxygen during grinding, the results still indicate that the DOE model may be overestimating the length of time required for the E_h transition to oxidizing conditions and the model may be underestimating release at earlier times. Additional research that more closely represents actual conditions is needed to reduce the uncertainty in the release of Tc-99 from the saltstone waste form.

2.6.4.4 Waste Form Conclusions

The NRC staff concludes that the representation of the waste form in Case A is not realistic and does not represent current conditions or expected future degradation. Of the cases analyzed by DOE, NRC staff considers the modeled initial conditions and degradation of the waste form in Cases K, K1, and K2 to best reflect the current and future expected conditions of the waste form based on currently available data. However, NRC staff recognizes that there is significant uncertainty in the rate and extent of fracturing and there is considerable uncertainty in some of the key parameters related to the waste form performance. For example, little is known regarding the potential performance of the actual as-emplaced waste form. DOE performed research on some of these parameters since the 2006 Waste Determination (DOE-WD-2005-001) was issued. Additionally, in Section 8.2 of the PA, and in the response to NRC comment PA-7 (SRR-CWDA-2011-00044), DOE described their planned future research in these areas. NRC staff views this research as extremely useful and encourages continued research in these areas to ensure that the estimated dose does not under-predict the actual expected dose.

NRC staff concludes that additional information is needed on the performance of the as-emplaced saltstone waste form. Specific areas in which continued research is needed include the:

- hydraulic conductivity of as-emplaced saltstone
- potential variability of as-emplaced saltstone properties with variations in the composition (e.g., water to cement ratio, presence of aluminate, changes to admixtures)
- applicability of measured hydraulic properties (e.g., hydraulic conductivity, diffusivity) for laboratory-prepared samples to field-scale, as-emplaced saltstone (e.g., effects of scale, temperature, presence of admixtures, and reducing conditions)
- effect of curing temperature profile for on the hydraulic properties of as-emplaced saltstone
- expected fracturing in saltstone with time
- leaching of radionuclides from as-emplaced saltstone
- reduction of Tc in saltstone
- expected current and future E_h and pH conditions in the saltstone

NRC staff concludes that the items listed above are key monitoring factors (Appendix A). NRC staff had previously identified aspects of the performance of the waste form as key monitoring factors in the 2005 NRC TER and associated monitoring plan for saltstone (NRC, 2005, 2007a). Additionally, during monitoring NRC staff identified three Open Issues related to the performance of the waste form (Chapter 1). Due to the risk significance of the performance of the waste form, the NRC staff will continue monitoring this area.

2.7 Source Term Release and Near-Field Transport

Source term modeling estimates the partitioning in and release of radionuclides from the disposal unit. The near field is generally defined as the area surrounding the waste that may have moisture flow and chemical conditions (e.g., due to the presence of the waste or engineered barriers) significantly different from the natural system in which the waste disposal facility is located. In this document, the term “Near Field” is used to reference the closure cap, saltstone waste, disposal units, backfill, and natural soil above the water table.

2.7.1 Source Term Release Models

Radionuclide release from saltstone is modeled in all cases is diffusive and advective, with aqueous concentrations controlled by linear partitioning (represented with K_d values) from saltstone solids into pore water. Chemical properties of saltstone are expected to change over time as infiltrating groundwater depletes the reductive and buffering capacity of the saltstone matrix, causing the E_h to fall and the pH to rise. Because sorption of many radionuclides depends on these chemical properties, release of most radionuclides is represented with K_d values that depend on step changes in the modeled redox state and pH of saltstone (Section 2.7.1.1). As discussed in Section 2.6, Tc-99 mobility is more sensitive to redox conditions than the mobility of other key radionuclides. Because of this sensitivity, Tc release was modeled differently from the release of other elements (Sections 2.7.1.2 and 2.7.1.3).

2.7.1.1. Pore-Volume Model

For elements other than Tc, DOE models radionuclide release by applying a single K_d value to the entire saltstone monolith or disposal unit component (i.e., floor, roof, or walls). DOE models the evolution of these K_d values as step changes corresponding to certain changes in the saltstone or disposal unit E_h and pH. The redox state is broadly divided into “reduced” and “oxidized” conditions and the pH is divided into three categories associated with the age of the cementitious material. DOE indicates that the initial condition of saltstone is expected to be middle-age, so the “new” values are not actually used in the source term release model. The material is "middle" if the pH is above 11 and “old” if the pH drops below 11. Elements other than Tc are assigned a single K_d value in the saltstone or disposal unit component based on six redox-age pairs: reducing-new, reducing-middle, reducing-old, oxidizing-new, oxidizing-middle, and oxidizing-old.

As discussed in Section 2.6.3, DOE performed geochemical modeling to determine the number of pore volumes required to transition from reduced to oxidized conditions and from high pH to lower pH conditions. Cases A and K use the same type of release model for elements other than Tc, but chemical transitions in saltstone are predicted to occur at fewer pore volume replacements in Case K because of higher porosity and lower modeled reducing capacity. Based on the lower reducing capacity and higher effective porosity of the saltstone compared to the Case A values, the pore volumes for the E_h and pH transitions used in saltstone in Case K decreased from Case A values to about 500 and 7,600 pore volumes, respectively.

Because the relevant properties of the disposal unit concrete (e.g., porosity, reducing capacity) are the same in Cases A and K, the number of pore volume replacements required to cause the chemical transitions are the same. However, because more water flows through saltstone and disposal unit in Case K, the same number of pore volume replacements are achieved in less time (Table 2.5-5). Additional details regarding chemical transition times for disposal unit concrete and saltstone are found in Sections 2.5 and 2.6.

2.7.1.2 Explicit Shrinking-Core Model for Tc Release

As discussed in Section 2.6, Tc is much more mobile in its reduced Tc(IV) form than it is in its oxidized Tc(VII) state. Because release of Tc is more redox-sensitive than release of other key radionuclides, DOE modeled its release based on the gradual oxidation of saltstone with time rather than the whole-monolith step changes used for other elements modeled with the pore-volume model. In Case A and the sensitivity cases included in the PA, DOE used what it calls a “shrinking core” model that it implemented explicitly in PORFLOW. In this model, the redox state of individual finite elements is tracked in PORFLOW™ as dissolved oxygen in inflowing water consumes the reducing capacity in the cementitious material. DOE assumed the infiltrating water has oxygen concentrations that would be in equilibrium with atmospheric oxygen (i.e., 8 mg/L or 1.06 meq e^-/L) (the NRC evaluation of this assumption is included in Section 2.7.3). Oxidation from gas-phase transport of oxygen is not included (i.e., the amount of reducing capacity consumed is limited by the water flowing into the waste form or disposal unit component). Tc is assigned a “pseudo-K_d” in each finite element based on the amount of reducing capacity consumed. The "pseudo-K_d" ranges from 1,000 mL/g to 0.8 mL/g and essentially represents Tc as mobile in oxidized areas of the saltstone and immobile in reduced areas.

2.7.1.3 Average-K_d Model for Tc Release

In Cases K, K1, and K2, DOE assumed saltstone has an initial fracture spacing of 30 m in Vault 1, 61 m in Vault 4, and 41 m in the FDCs. The saltstone in each disposal cells is assumed to decrease to 0.1 m at 10,000 years after closure (SRR-CWDA-2011-00044, Tables PA-8.3). However, PORFLOW™ cannot easily be used to model either the high fracture frequency at 10,000 years or a fracture frequency that increases with time. Therefore, the Case K PORFLOW™ model did not model Tc release from oxidized and reduced areas explicitly, as was done for Case A and the other cases presented in the original PA (Section 2.7.1.2). Instead, for Case K, saltstone oxidation was calculated separately from the PORFLOW™ model using the method of Smith and Walton (1990). In this method, oxidation is modeled as proceeding from saltstone edges and fracture faces using the oxygen concentration at these surfaces as a boundary condition. The calculation results in the fraction of saltstone that is oxidized as a function of time. The fraction of saltstone oxidized is then is used to calculate a weighted average of the K_d values for oxidizing and reducing saltstone. This gradually-decreasing average K_d is assigned to Tc throughout the entire saltstone monolith. DOE refers to this approach as a "single-porosity" approach. To avoid confusion with the use of the term "single-porosity" in fracture flow modeling, the NRC staff refers to the Case K Tc release model as an "average-K_d" approach.

An analytical model (external to PORFLOW) is used to estimate fracture growth over time using a semi-log fracture growth relationship with fracturing assumed to terminate at 10,000 years at a final fracture spacing of 10 cm (4 in). In Cases K, K1, and K2, the reducing capacity in saltstone is assumed to be 25% of the Case A value (i.e., 0.206 meq e^-/g). The reducing capacity in the disposal units is unchanged from its Case A value (i.e., 0.240 meq e^-/g). The oxygen concentration at the saltstone or disposal unit surfaces is assumed to remain constant at atmospheric levels (i.e., 1.06 meq e^-/L). The reduction capacity and dissolved oxygen concentrations are used in combination with a diffusion coefficient of 1.0×10^{-7} cm^2/s to parameterize an oxidation model to estimate progression of the oxidation front along each fracture face for each fracture (or partial fracture) that grows in over time.

For Cases K, K1, and K2, oxidation of disposal unit concrete was modeled as proceeding from floor and wall edges, similar to the modeling of saltstone oxidation. In addition, like the modeling of saltstone oxidation, oxidation is affected by an assumed increase in diffusivity with time (from the values provided in Table 2.5-2 for the various disposal unit components to 5×10^{-6} cm^2/s at 10,000 years). As discussed in Section 2.5, DOE did not provide information about the fracturing assumptions used in the disposal unit concrete in these cases.

2.7.2 Sorption Coefficients

2.7.2.1 Cementitious Materials

DOE developed sets of sorption coefficients (K_d) that were used in the modeling radionuclide release and transport in saltstone and concrete in the PA for all radionuclides except for Tc-99 (2009 PA; Table 4.2-18). In the PA cases (i.e., all cases except for Cases K, K1, and K2) the same sets of sorption coefficient data were used for both saltstone and the disposal unit concrete.

Table 4.2-18 in the PA included six sets of K_d values that represent various ages and oxidation states of the cementitious materials (i.e., young age, middle age, and old age for both reduced and oxidized cementitious materials). However, DOE only used three of these sets in the PORFLOW™ calculations (i.e., reduced middle age, oxidized middle age, and oxidized old age). The cementitious materials that do not contain slag are initially assigned oxidized middle age K_d values, while those that contain slag are initially assigned the reduced middle age K_d value. The derivation of the times required for the E_h and pH transitions to occur are discussed in Section 2.7.1.1. When the E_h transition time is reached, the K_d values corresponding to oxidized middle age are used, and when the pH transition time is reached, the oxidized old age K_d values are used (SRR-STI-2009-00115).

DOE stated that the K_d values were based, where possible, on site-specific information, but literature reviews or engineering judgment were relied on in the absence of such data (2009 PA; page 213). The reference cited for the majority of the radionuclides was WSCR-STI-2007-00640, which contained a compendium of literature information on the K_d values for cementitious materials as well as some site-specific K_d measurements for a reducing grout and an aged cement. Other references cited include SRNS-STI-2008-00045, which reports site-specific measurements of sorption onto simulated saltstone and FDC concrete, and WSRC-STI-2007-00640, which reports site-specific measurements for Pu K_d values for cementitious materials.

For Cases K, K1, and K2, DOE developed revised K_d values for cementitious materials for many elements (SRR-CWDA-2011-00044, Tables PA-8.6 and PA-8.7). DOE stated that these values are based on recent site-specific experimental data and new analyses of existing data. For most elements the citation for the revised K_d values is the 2010 compilation of geochemical data applicable to SRS performance assessments (SRNL-STI-2009-00473; Table 18). The revised K_d values for Ba and Sr are based on site-specific and literature information summarized in SRNL-STI-2010-00667, and the revised K_d values for U are based on a site-specific K_d measurements onto reducing grout and a weathered concrete sample (SRNL-STI-2010-00493). Additionally, for Cases K, K1, and K2, DOE developed saltstone specific K_d values for Ba and Sr based on K_d values measured in leaching from a core sample of saltstone from Vault 4 (SRNL-STI-2010-00667). In the majority of cases, the new analyses resulted in the selection of higher K_d values compared to the 2009 PA base case, meaning that the newer calculations will result in delayed radionuclide release and slower transport.

The Tc K_d values for the PA cases (i.e., cases other than Cases K, K1, and K2), were used in a shrinking core model explicitly implemented in PORFLOW™ (Section 2.7.1.2). In the PA cases, the Tc K_d value for reduced cementitious materials was 1,000 mL/g. The PA notes that this K_d value is intended to immobilize the Tc in the reduced cementitious materials. As the reducing capacity is consumed by dissolved oxygen in infiltrating water, the Tc is made mobile by using a K_d value of 0.8 mL/g. The transition of the Tc K_d value from 1,000 mL/g to 0.8 mL/g as the reducing capacity is consumed is shown in Figure 4.2-41 of the PA.

As described in Section 2.7.1.3, in Cases K, K1, and K2, the K_d for Tc was implemented in the model using an "average-K_d approach." In this approach, a weighted average of the reduced and oxidized K_d values for the cementitious materials was determined based on the fraction of the material that was oxidized during the given time step. In Case K, a Tc K_d value of 1,000 mL/g is assumed for reduced cementitious materials, and a value of 10 mL/g is assumed for oxidized cementitious materials. DOE based the reduced value of 1,000 mL/g on sorption experiments performed for simulated saltstone in a reducing environment (2% $H_2(g)$) (SRNL-STI-2010-00668). The oxidized value of 10 mL/g was based on desorption of Tc from a saltstone core sample under oxidizing conditions (SRNL-STI-2010-00667). In Case K1, a Tc K_d value of 500 mL/g is assumed for reduced cementitious materials and a value of 0.8 mL/g is assumed for oxidized materials. In Case K2, a Tc K_d value of 500 mL/g is assumed for reduced cementitious materials and a value of 10 mL/g is assumed for oxidized materials.

2.7.2.2 Soil K_d Values

The K_d values DOE used to model radionuclide transport through (i) backfill, (ii) the vadose zone, and (iii) the saturated zone are presented in Table 4.2-15 in the PA. The values tabulated for backfill soil are applied also to "clayey" soils in the saturated zone, that is, soils with a hydraulic conductivity less than 1×10^{-7} cm/sec (2009 PA). The values tabulated for the vadose zone are applied also to "sandy" soils in the saturated zone, that is, soils with a hydraulic conductivity greater than 1×10^{-7} cm/sec (2009 PA).

DOE stated that the K_d values were selected on the basis of site-specific sorption data, from compilations of literature data, or on expert judgment; preference was always given to SRS site-specific data (2009 PA). In some cases, chemical homologues were used. The main reference for the K_d values was WSRC-TR-2006-00004 (Table 10), which contains a compendium of SRS site-specific and literature K_d values. Another DOE report, SRNL-TR-2009-00019, is cited for the Tc values in PA Table 4.2-15. This report summarizes recent data from an E-Area borehole and recommends new Tc K_d values for soils.

For Cases K, K1, and K2, DOE adopted modified soil K_d values for some elements on the basis of new data or new analyses of the literature (SRR-CWDA-2011-00044; Table PA-8.4). The new references for the modified values are SRNL-STI-2009-00473, SRNL-STI-2010-00493, and SRNL-STI-2011-00011. SRNL-STI-2009-00473 is a new compilation of geochemical data, prepared as part of the PA maintenance program, relevant to a number of performance assessment efforts at SRS. The relevance of this new report to the data in SRR-CWDA-2011-00044, Table PA-8.4, is that the new compilation includes consideration of more recent site-specific sorption experiments. SRNL-STI-2010-00493 provides new Cl and U K_d values based on recent sorption experiments using SRS soil samples. SRNL-STI-2011-00011 cites new site-specific Ra data and calculates K_d values for Ba as being midway between measured values for Sr and Ra.

2.7.3 Modeling of Flow in the Waste Form and Near Field

DOE used the PORFLOW™ computer code to model unsaturated flow and contaminant transport to determine the source-specific, time-dependent radionuclide contaminant flux entering the water table aquifer at SDF. Three separate near-field (or vadose zone) models were constructed representing Vault 1, Vault 4, and the FDCs. This section summarizes DOE's approaches for near-field flow and transport model construction, material properties, and model validation.

2.7.3.1 Near-Field and Vadose Zone Model Construction and Boundary Conditions

The discrete features of SDF Vaults 1 and 4, and the sixty-four FDCs, are described in Section 2.5 of this TER. The discrete features of Vaults 1 and 4, and the FDCs were necessarily simplified for the purposes of PA modeling (Figure 2.5-6). A description of the simplified conceptual models represented in the PORFLOW™ model for each SDF source is discussed in Section 4.4.1 of the 2009 PA (page 243). Section 4.4.4.1.2, "Disposal Unit Modeling in PORFLOW" (2009 PA; page 277) also provides details on the numerical model construction and flow results for each vadose zone model.

The long rectangular, Vaults 1 and 4, were represented in 2-D Cartesian coordinates as a transverse slice with inventory assigned to a nominal thickness of 1 length unit (i.e., 0.3 m [1 ft]). Only half of Vault 1 and 4 in the short dimension is modeled, taking advantage of symmetry about the centerline. The FDCs were modeled in 2-D cylindrical coordinates with an assumed "thickness" of one unit radian, implicitly assuming symmetry about the centerline axis.

Cracks in Vault 1 and 4 walls are not modeled explicitly but rather are reflected in the initial and degraded hydraulic property assignments. To account for the potential for contaminants to be located in the vault walls, vault wall pore fluid is assumed to contain concentrations similar to those found in the vault cells with approximately 0.65 and 0.5 percent of the inventory estimated to be present in the Vault 1 and 4 walls, respectively.

DOE places no-flow boundary conditions at the disposal unit centerlines and at the outer perimeter or radius of the Vault 1, 4, and FDC model domains, which extend at least 9.1 m (30 ft) beyond the perimeter of each disposal unit (2009 PA; page 283). Net infiltration rate and vadose zone thickness are naturally variable with time; however, short term variations (e.g., seasonal effects) in these parameters are not simulated. A time-dependent net infiltration flux that varies as a function of the degradation state of the engineered closure cap (Section 2.4 of this TER and 2009 PA; Table 3.2.7) is prescribed as the upper boundary condition to the near-field and vadose zone PORFLOW™ model. The lower boundary of the model coincides with the water table where pressure head is set to zero. DOE specifies an outflow boundary condition for radionuclide transport, whereby the flux leaving the vadose zone is by advection only.

The vadose zone PORFLOW™ model domains use approximately 7,000 cells to represent up to 18 different material types each with their own hydraulic and transport properties. Different material types are also used to evaluate various scenarios such as flow through preferential pathways that may form in a portion of a single material, in addition to representing variation in hydraulic and transport properties of the various materials themselves. Grid resolution is stated to be a compromise between two competing objectives: (1) resolution of thin geometric features (e.g., sheet drains, HDPE-GCL liners) and sharp flow field transitions (e.g., ponded water flowing over roof edge), and (2) achieving reasonable computer storage and runtimes.

The upper boundary of the near-field model also includes materials located physically above the top of the disposal units (e.g., lower lateral drainage layer and HDPE-GCL). The lower lateral drainage layer and HDPE-GCL are assumed to degrade over time (Sections 2.4 and 2.5). After approximately 19,000 years, the hydraulic conductivity and porosity of the lower lateral drainage layer are estimated to be similar to those for the overlying backfill layer (2009 PA; page 183). However, the maximum hydraulic conductivity of the HDPE-GCL at 20,000 years is still only assumed to be about a fraction of an inch per year (i.e., $<1x10^{-8}$ cm/s [<0.1 inches/yr]) after 20,000 years (Figure 2.5-3). Additionally, an HDPE layer is applied to the walls of the FDCs with a hydraulic conductivity that peaks at around $8x10^{-8}$ cm/s (1 inch/yr) at 20,000 years (Figure 2.5-2). The vadose zone (below the top of the disposal units) comprises an upper and lower zone described in more detail below.

Time step sizes were selected as a compromise between two competing objectives:
(i) resolution of concentration peaks from relatively mobile species that migrate as a pulse, and (ii) achieving reasonable computer runtimes. A transport step size of 1 year was selected for vadose zone flux simulations with the exception of Tc-99 simulations that were run using a 0.05 year transport step size to avoid numerical inaccuracies. A recording frequency of 1 year was used for the vadose zone transport simulations. A 2.5 year transport step size and 20 year recording frequency was selected for aquifer concentrations.

2.7.3.2 Near-Field and Vadose Zone Transport Material Properties

The cementitious materials simulated in the SDF PA modeling can be grouped into five categories as follows:

1. Low quality concrete associated with the lower mud mats for the FDCs.
2. Medium quality or ordinary concrete associated with the roof of Vaults 1 and 4.
3. High quality concrete associated with the base of Vaults 1 and 4, and the concrete for the FDCs, including the upper mud mats.
4. “Fractured concrete” associated with the existing walls of Vaults 1 and 4, which have experienced macroscopic cracking.
5. Saltstone.

The initial hydraulic properties associated with each of the cementitious material categories are presented in Table 4.2-16 of the PA, and are summarized in Table 2.7-1

Preferential pathways through cracks, fractures, or other discrete features were either implicitly represented as an increase in saturated hydraulic conductivity within a porous medium, or were explicitly represented in a porous medium formulation as discrete zones of high permeability depending on the feature and the case being analyzed. The materials palette used in PORFLOW™ SDF modeling is located in Table 4.4-10 of the 2009 PA (page 291).

Non-zero diffusivities were assigned in the vadose zone transport modeling, while dispersivities were set to zero. A value of zero for the dispersivity would tend to minimize plume spreading leading to higher peak fluxes to the water table.

DOE assumed that the disposal unit concrete will degrade with time due to exposure to sulfate in the salt waste (Section 2.5). In Case A, degradation of the hydraulic properties of cementitious materials such as the disposal unit roofs, walls, and floors is assumed to coincide with the creation of expansive phases (e.g., ettringite) that leads to cracking. Effective hydraulic properties are calculated based on a weighted average of degraded and intact material properties. In Cases K, K1, and K2, the degradation of the disposal unit concrete was modeled non-mechanistically with increases in the hydraulic conductivity and diffusivity. Increased degradation of the disposal unit concrete also is assumed in several of the deterministic sensitivity cases (Section 2.5).

As discussed in detail in Section 2.6, the NRC staff concluded that DOE's Case A assumption that saltstone does not degrade during a 20,000 year evaluation period is unrealistic. Cases K, K1, and K2 consider a scenario in which the saltstone monolith degrades hydraulically with an initial hydraulic conductivity of 1×10^{-8} cm/s that increases over time to 1×10^{-6} cm/s at 10,000 years. Increased degradation of saltstone is also assumed in some of the deterministic sensitivity cases (Section 2.6). The Case A assumptions regarding engineered barrier and saltstone hydraulic performance result in greater than 99 percent of meteoric water that infiltrates through the cover being shed through the lower lateral drainage layer (Sections 2.5 and 2.13). The result of the revised parameters in Cases K, K1, and K2 is a significant increase in flow through the saltstone matrix compared to the Case A scenario with average Darcy velocities around 10 cm/yr (4 in/yr) at 10,000 years and 20 cm/yr (8 in/yr) at 20,000 years.

The near-field PORFLOW™ models utilize characteristic curves (relative permeability and suction head as functions of saturation) for cementitious materials and soils. The data used to develop these curves are taken from WSRC-STI-2006-00198, WSRC-STI-2007-00649, or SRNL-STI-2009-00115. Moisture characteristic curves used for cementitious materials in PA modeling are based on WSRC-STI-2006-00198, except for fractured concrete which is based on SRNL-STI-2009-00115. In the fractured cases, the fractures become unsaturated under certain conditions. Moisture characteristic curves are illustrated in Figures 4.2-23 through 4.2-27 of the 2009 PA. The NRC staff's evaluation of DOE's approach to modeling unsaturated flow including development of moisture characteristic curves is discussed further in Section 2.7.4.4. In Cases K, K1, and K2, it was assumed that relative permeability and saturation were equal to 1.0 for all suction levels and moisture characteristic curves were not used.

Table 2.7-1: Initial (or for Some Materials Time Invariant) Material Properties Used in PORFLOW™ Vadose Zone Modeling

Material	Saturated Effective Diffusion Coefficient (cm^2/s)	Average Total Porosity (%)	Average Dry Bulk Density (g/cm^3)	Average Particle Density (g/cm^3)	Saturated Horizontal Hydraulic Conductivity (cm/s)	Saturated Vertical Hydraulic Conductivity (cm/s)
Backfill	5.3×10^{-6}	35	1.71	2.63	7.6×10^{-5}	4.1×10^{-5}
Undisturbed Vadose Zone	5.3×10^{-6}	39	1.62	2.66	3.3×10^{-4}	9.1×10^{-5}
Saltstone and Clean Grout Cap	1.0×10^{-7}	58	1.01	2.4	2.0×10^{-9}	2.0×10^{-9}
High Quality Concrete	5.0×10^{-8}	12[1] 11[2]	2.24[1] 2.22[2]	2.55[1] 2.49[2]	3.1×10^{-10} (1) 9.3×10^{-11} (2)	3.1×10^{-10} (1) 9.3×10^{-11} (2)
Fractured Walls in Vaults 1 and 4	5.0×10^{-8}	12.0	2.24	2.55	1.7×10^{-1}	1.7×10^{-1}
Medium Quality Concrete	1.0×10^{-7}	14.5[3] 13.6[4]	2.20[3] 2.21[4]	2.57[3] 2.56[4]	5.0×10^{-9}	5.0×10^{-9}
Low Quality Concrete[5]	8.0×10^{-7}	21.1	2.06	2.61	1.0×10^{-8}	1.0×10^{-8}

2009 PA (Tables 4.2-14 and 4.2-16) and WSRC-STI-2006-00198 (Table 5-18).
[1] Vaults 1 and 4 base.
[2] FDCs.
[3] Vault 1 roof.
[4] Vault 4 roof.
[5] Lower mud mats of the FDCs.

The vadose zone thickness between the vaults and the underlying Upper Three Runs (Upper Zone) water table ranges from 10.9 m (35.6 ft) (for future disposal cells 10A to 10D [Figure 2.8-8]) to 14.6 m (48 ft) (for Vault 1). Although the vadose zone thickness is variable for FDCs as provided in Table 4.2-13 in the 2009 PA, page 191, the thickness is modeled as a constant based on the average thickness for all 64 FDCs of 12.8 m (42 ft) (2009 PA; page 188).

The upper vadose zone is assumed to be composed of backfill materials with properties more similar to clay than sand. The undisturbed lower zone is assumed to have material properties similar to sand. In reality, the undisturbed materials below the vaults may consist of both native upper and lower vadose zone materials. Because the properties of the upper vadose zone are more clayey and would tend to retard contaminant transport compared to backfill or lower vadose zone materials, DOE indicates the modeling assumptions that backfilled (rather than upper vadose zone) material surrounds the vaults and that all of the soil underneath the disposal unit is similar to lower vadose zone (rather than upper vadose zone) material is conservative. Vadose zone material properties are listed in Table 2.7-1. DOE assumes vadose zone material properties are constant, and do not change with time. Soil moisture characteristic curves are presented in the PA for backfill (Figure 3.2-17) and for lower vadose zone soil (Figure 4.2-22). A complete list of figures that illustrate moisture characteristic curves used in the SDF PORFLOW™ modeling is provided in Table I3-2 of the PA, along with the source of the data used to construct each curve.

2.7.3.3 Near-field Model Validation

DOE used characterization and monitoring data from uncapped E-Area (adjacent to F-Area) to validate aspects of the PORFLOW™ vadose zone model. These data included soil suction and water content, tracer test pore velocity, and tritium plume concentration. Soil suction data from E-Area indicate a range from 50 to 200 cm (20 to 79 in), and PORFLOW™ vadose zone modeling produces upper vadose zone soil suction values of 83 cm (33 in) and lower vadose zone values of 170 cm (67 in) (SRS-REG-2007-00002). Water content data suggest saturation ranges from 35 to 75 percent, and PORFLOW™ vadose zone modeling produces upper vadose zone saturation values of 91 percent and lower vadose zone values of 72 percent (SRS-REG-2007-00002). Field and laboratory tracer test experiments indicate a pore velocity of 114 cm/yr (45 in/yr), and PORFLOW™ vadose zone modeling produces upper vadose zone pore velocities of 86 cm/yr (34 in/yr) and lower vadose zone velocities of 109 cm/yr (43 in/yr) (SRS-REG-2007-00002). Comparison between the measured tritium plume concentration from a disposal trench at E-Area and PORFLOW™ vadose zone model results do not serve to validate the model, but PORFLOW™ results are generally consistent with measured data and the vadose zone model is not invalidated by the data (SRS-REG-2007-00002).

2.7.4 NRC Evaluation – Release and Near-Field Transport

2.7.4.1 NRC Evaluation – Release Models

The three radionuclide release models DOE used in the PA and RAI responses (i.e., the pore volume model, explicit shrinking-core model, and average-K_d model) all are based on linear partitioning of radionuclides from saltstone solids into the pore water. Differences between these three approaches are discussed in Section 2.7.1 and evaluated later in this section. First, this section addresses the common assumption that radionuclide release and transport can be represented with linear partitioning (represented by K_d values). This section addresses two main implications of the linear partitioning assumption for release and transport modeling: (1) effects if laboratory tests are actually solubility limited, and (2) effects if radionuclides are solubility limited in the emplaced waste.

If the concentration of radionuclides dissolved in water is actually limited by their solubility instead of by sorption in laboratory tests intended to measure linear sorption coefficients (K_d values), K_d values may be overestimated. Values may be overestimated because, at equilibrium, the dissolved radionuclide concentration cannot exceed its solubility limit (by definition). Therefore, if dissolved radionuclide concentrations in sorption experiments are actually limited by solubility, they will not increase as a radionuclide is added to the test system, even though the amount of the radionuclide in the solid phase increases. Because the sorption coefficient is the ratio of the solid concentration to the dissolved concentration, increases in the solid concentration without increases in the dissolved concentration cause the measured sorption coefficient to increase. This phenomenon is a particular concern for Pu-239 and Np-237, which were found to be solubility-controlled in experiments designed to measure sorption in cementitious material (SRNL-STI-2009-00636). In an RAI response (SRR-CWDA-2010-00033, SP-9), DOE indicated that sensitivity tests performed with the SDF GoldSim® model suggested the potential effect of overestimating these K_d values is small. Specifically, DOE found that setting K_d values for Pu-239 and Np-237 in cementitious material equal to zero increased their contribution to dose by less than a factor of three. However, because of NRC concerns about the GoldSim® model (Sections 2.11.4.2 and 2.11.4.3), the NRC staff concluded that a better basis is needed for Pu-239 and Np-237 sorption in cementitious materials. The NRC staff included improved support for K_d values in saltstone as a PA maintenance item.

If a radionuclide is actually solubility limited in emplaced waste but its release is represented with a K_d value, the accuracy of the modeled release depends on the solid concentration in the actual waste. For the SDF, representing radionuclide release with a K_d instead of a solubility limit is likely to be of the most concern for Tc-99, because it is anticipated to be solubility-limited in reducing saltstone but is represented with a K_d value. DOE has not shown whether the aqueous concentrations calculated with the sorption coefficients used in the model are reasonable and reflect the physical processes that DOE believes actually control release (i.e., if the aqueous concentrations calculated with a K_d match the aqueous concentrations that would result from a solubility limitation). Therefore, the NRC staff performed a simple calculation to evaluate the reasonableness of the aqueous Tc concentration given a K_d of 1,000 mL/g (the appropriateness of the selected K_d values is evaluated in subsequent paragraphs). The parameters needed for this calculation are as follows:

- Average FDC Tc inventory of 540 Ci (3.18×10^4 g) (PA Table 3.3-5)
- FDC saltstone diameter of 45.7 m (150 ft) and height of 6.10 m (20 ft) (PA Section 3.2.1.3.2)
- Saltstone porosity of 0.58 and particle density of 2.42 g/cm^3 (PA Table 4.5-5)

The NRC staff first considered a saturated system at equilibrium under reducing conditions. In this case, the Tc inventory is modeled as being distributed between the saltstone solid and the pore water according to a K_d of 1,000 mL/g. The resulting calculated aqueous Tc concentration would be 3×10^{-8} M. This value is within the range of measured Tc(IV) solubility limits in reducing, high-pH systems (e.g., Pilkington, 1990, Greenfield, et al., 1998, and Warwick, et al., 2007). Therefore, it appears that, the high K_d would yield aqueous concentrations roughly consistent with solubility control until the initial Tc K_d is significantly depleted. After the Tc inventory is depleted, continued solubility control would predict a constant release rate while modeling with a K_d value will predict a decreasing release rate. This is a general problem associated with modeling solubility-limited release with a K_d model. However, because Tc release is likely to be more significantly affected by the rate of saltstone oxidation than the difference between a K_d model and solubility–limited release, the NRC staff concluded using a K_d model was appropriate in this version of the PA.

Pore Volume Model

There are two types of concerns about the pore volume model: (1) concerns about the basis for the estimated number of pore volume flushes required to cause redox (E_h) and age (pH) transition times, and (2) concerns about using a whole-monolith step change to model changes in radionuclide mobility instead of modeling more gradual changes in the saltstone and disposal unit concrete. As discussed in more detail in Section 2.6, the NRC staff found the basis for predicted number of pore volumes required to cause certain pH and E_h changes to be insufficient in Case A. However, the NRC staff expects the transition times in Case K to more reasonably reflect the expected evolution of the saltstone chemical environment.

The NRC staff found the pore-volume model DOE used for radionuclides other than Tc (in all cases) to be appropriate for radionuclides with little sensitivity to E_h or pH because they are generally insensitive to how E_h and pH evolution is modeled. The NRC staff questioned the applicability of the model to other radionuclides (i.e., redox or pH sensitive radionuclides) such as Np-237 and Pu-239 (NRC, 2010b, i; SP-13). In response, DOE indicated that only Cr, Rh, Sb, and Tc had large enough differences in their sorption coefficients under different pH and redox conditions to be sensitive to the use of a pore-volume model as compared to a model implementing a more gradual transition. DOE indicated that, except for Tc, none of these elements is likely to be dose significant. The NRC staff agrees that changing the source term release model for these radionuclides will not have a significant effect on peak dose unless other assumptions (e.g., regarding inventory or sorption coefficients) about these radionuclides change.

Explicit Shrinking Core Model

In general, the NRC staff concludes the explicit shrinking-core model DOE used to model Tc-99 release from saltstone in the PA Cases (i.e., all cases except Cases K, K1, and K2) is appropriate. In particular, the NRC staff finds that tracking releases from the oxidized and reduced fractions of saltstone separately appears to be the most appropriate implementation of DOE’s conceptual model of Tc-99 release. Regarding the oxidation modeling, the NRC staff finds DOE’s assumption that infiltrating water is equilibrated with atmospheric oxygen (i.e., 8 mg/L dissolved oxygen, equivalent to 1.06 meq/L) to be conservative because subsurface oxygen concentrations are likely to be lower (e.g., DOE reports a measured value of 1.2 mg/L from a groundwater monitoring well near the SDF).

The NRC staff finds DOE’s basis for neglecting gas-phase transport of oxygen to saltstone surfaces to be insufficient. Specifically, DOE indicated that gas-phase transport of oxygen to saltstone faces is neglected in the PA Cases because saltstone is nearly completely saturated with water. The NRC staff does not believe this is an adequate basis for neglecting gas-phase transport of oxygen to saltstone faces because the significance of gas-phase transport is not necessarily that gas permeates saltstone, but that gas-phase oxygen serves as another source of oxygen beyond the finite quantity of oxygen in infiltrating water. The NRC staff expects that, in cases with sufficient water flow (e.g., the Synergistic Case) neglecting gas phase oxygen transport is mitigated by DOE's assumption that the infiltrating water is saturated as if in equilibrium with atmospheric concentrations of oxygen. However, the NRC scoping calculations suggest that in Case A, water flow into saltstone is so limited that the amount of oxidation from inflowing water would be small relative to the potential oxidation from diffusion of gas-phase oxygen into saltstone.

Average-K_d Model

DOE addressed some of the uncertainties of the shrinking core model and the Tc K_d values in Case K presented in SRR-CWDA-2011-00044 (Table PA-8.1). For example, the model allowed for more extensive cracking of the saltstone, providing a greater available surface area for oxidation. Unlike the explicit shrinking core model implemented in PORFLOW, the average-K_d model does include diffusion of oxygen from the gas phase into saltstone or the disposal unit concrete, however it neglects oxygen introduced into the saltstone or disposal unit in infiltrating water.

As explained in more detail in Section 2.7.1.3, the weighted average K_d is based on a calculation of saltstone oxidation performed outside of the PORFLOW™ near-field model. The average K_d value, which gradually decreases as saltstone is modeled as becoming more oxidized, is then applied to the whole monolith. In general, the NRC staff finds the average-K_d approach yields results that are not consistent with DOE's conceptual model of Tc release, in which Tc is relatively mobile in oxidized portions of saltstone and immobile in the reduced fraction. In particular, the NRC staff finds that with certain relevant parameter combinations, the average-K_d model predicts very little release of Tc as oxidation proceeds, and predicts a relatively sudden release of Tc when saltstone oxidation is nearly complete. This tends to both delay and overestimate the peak Tc release (Section 2.13.3).

In theory, the average-K_d model rests on the assumption that the concentration of Tc in saltstone pore water will equilibrate between oxidized and reduced regions more rapidly than Tc is released from the oxidized region. In DOE's description of the average-K_d model (SRR-CWDA-2011-00044) DOE indicates that "...the liquid concentration has been assumed to be in equilibrium between the oxidized and reduced regions," that is, the Tc pore liquid concentration is the same in both the oxidized and reduced regions. This assumption is necessary to make the simplification from Equation 9 to Equation 10 in DOE's mathematical development of the average-K_d model (SRR-CWDA-2011-00044, page 84). The NRC staff finds that this concentration equivalence to be unrealistic. First, the assumption that the Tc concentration in saltstone equilibrates between the oxidized and reduced regions before a significant amount of Tc is released from the oxidized region is inconsistent with the same mechanism (i.e., diffusion) governing both processes. That is, diffusion controls both the release of Tc from the oxidized region and the diffusion of Tc from the oxidized to the reduced region. If Tc is removed advectively from the fractures, as the NRC staff expects, the aqueous concentration gradient from the oxidized region to the fracture would be similar (and slightly greater than) the aqueous concentration gradient from the oxidized region to the reduced region. Thus, Tc release would be expected to occur on a similar time scale as Tc equilibration between the oxidized and reduced regions. The NRC staff also finds that DOE has not provided a basis for assuming that the diffusion of Tc between the oxidized and reduced regions of saltstone is significantly faster than the diffusion of dissolved oxygen into saltstone.

In response to NRC concerns about the average-K_d model (NRC, 2011g, h, I); DOE compared two simple GoldSim® models (SRR-CWDA-2011-00114). The models were designed to enable a comparison between the results of an average-K_d approach and an approach in which Tc release from the oxidized and reduced portions of saltstone are modeled with two separate K_d values. DOE refers to this approach as a "dual-porosity" approach. To avoid potential confusion with the use of the terms "dual-continuum" and "double-porosity" used in flow modeling, the NRC staff refers to a model in which Tc release from reduced and oxidized fractions of saltstone is tracked with two K_d values as a "dual-K_d" approach.

As a result of this comparison, DOE concluded that the average-K_d model provides a good approximation of the results of a more complex dual-K_d model. DOE also concluded that the average-K_d model underestimates release at early times and overestimates release at later times (SRR-CWDA-2011-00114). The NRC staff has three main concerns with DOE's conclusions, each of which is described in more detail below:

- The results of the two GoldSim® models presented by DOE do not agree as well as DOE suggests.
- Although the results did not agree with each other well in many cases, the agreement would have been worse if the results were not artificially made more similar by a modeling artifact related to DOE's method of representing decreasing fracture spacing.
- Agreement between two GoldSim® models does not demonstrate that DOE's PORFLOW™ model accurately represents Tc release.

In addition, the NRC staff notes that a comparison between two GoldSim® models, or two PORFLOW™ models, is not a substitute for a comparison between the model output and experimental results.

DOE presented the results of the two models on log scales, which minimizes the apparent difference between the results. However, in many cases, the results of the two models differed by approximately an order of magnitude. In all cases, the peak release from the average-K_d model exceeded the peak release from the dual-K_d model. While apparently conservative, this modeling artifact can lead to inconclusive results if the peak dose exceeds the dose limit.

DOE's model representing the average-K_d approach was a simple one-dimensional model containing a column of 10 cells that releases from the last cell into the unsaturated zone. The flow rate through this column is based on DOE's Case K Vault 4 PORFLOW™ model.

DOE's dual-K_d model is a 20 x 20 grid of cells representing the intact saltstone, with an additional column of 20 cells representing a fracture. An oxidation front was assumed to proceed from the fracture face. As the oxidation front reaches the furthest edge of a column of cells, the K_d of Tc in the column of cells changes from its value in reduced material to its value in oxidized material; resulting in the mobilization of the Tc in the column. Tc was released from the oxidized column through diffusion to the fracture followed by advective transport out of the fracture and through diffusion back into the reduced region. As the system fractures, the modeled cell size decreases to represent smaller fracture spacing, but the inventory in the cells did not change. At later times and with larger amounts of degradation, the amount of back diffusion from the oxidized columns into the reduced ones was significant and rapid because of the very small diffusive lengths. This caused the dual-K_d model increasingly to mimic a stirred tank (i.e., equilibration of aqueous concentrations in different parts of the model), and, thereby, to increasingly mimic the average-K_d model.

The NRC staff determined that modeling flow through the oxidized regions in the dual-K_d model, which the NRC staff considers reasonable, would limit the amount of diffusion from the oxidized region to the reduced region because the oxidized Tc is advectively transported out of the monolith. Limiting diffusion from the oxidized to the reduced region would cause less concentration of Tc in the shrinking reduced region, which would mitigate the pulse release characteristic of the average-K_d model (Figures 2.13-6 and 2.13-7). DOE performed a sensitivity analysis in which they did include flow through the oxidized region, but the results did not show much of a difference (i.e., compare SRR-CWDA-2011-00114; Figure 11 with Figure 1). This result was contrary to NRC results with a similar GoldSim® model, which showed a significant difference in the peak release rate from the dual-K_d model if advective flow was modeled through the oxidized region. The flow rate used in DOE's sensitivity analysis was not provided in SRR-CWDA-2011-00114. The difference in the NRC and DOE results may be due to differences in the assumed flow rates.

2.7.4.2 NRC Evaluation – Cementitious Material Sorption Coefficients (K_d values)

In general, the approach of defining sets of K_d values that apply to both saltstone and other cementitious materials is reasonable. Like saltstone, the formulations for floors and walls of Vault 1, Vault 4, and the FDCs include the key reactive components blast furnace slag and cement (Tables 2.5-4 and 2.6-3). On the other hand, there are enough differences in the formulations that K_d differences must be assessed if using data on one type for modeling another. For example, unlike the concretes, saltstone contains no sand or aggregate, and was formed using a salt solution (PA; Table 4.2-7). DOE has been conducting laboratory studies to obtain saltstone-specific K_d values (e.g., SRNS-STI-2008-00045). The NRC staff believes that performing this type of research to determining medium-specific parameter values that recognize differences in formulation, texture, and processing will provide useful model support for the performance assessment.

In addition, when conducting research on sorption coefficients, it is important to assess whether, given the laboratory results, a K_d approach will be appropriate for modeling release from saltstone throughout the simulated time period, particularly when solubility may be a factor in controlling concentration. When performing K_d experiments it is also important to understand if the measured values reflect precipitation rather than sorption. The results of K_d or leaching experiments can be misleading and inappropriate to apply to transport modeling if the sorption is controlled by solubility in the experiment.

In comment SP-11 (NRC, 2010i), the NRC staff raised the issue of K_d measurements for old age materials that were conducted using a calcite-saturated solution but did not account for the mineralogical characteristics of long-aged cementitious materials. As discussed in the staff's review (NRC, 2010i) of the original DOE response (SRR-CWDA-2010-00033, SP-11), the NRC staff does not agree that the measured K_d values have been demonstrated to be appropriate or conservative for an extensively aged material. In response, DOE noted the challenge of identifying an appropriate surrogate for aged cementitious material, acknowledged the uncertainty in their approach, and implied that their long-range program plan for testing of cementitious material will consider additional experiments that could address sorption coefficients for aged materials. The NRC staff acknowledges the difficulty in finding appropriate surrogate materials and concludes that, because the radionuclides expected to be most significant to off-site dose from the SDF have relatively low sorption coefficients, the effect of using K_d values specific to old-age material may be limited. The NRC staff will review any information DOE develops on this issue as part of its PA maintenance program.

A related and important issue is the question of the applicability for modeling oxidizing conditions of sorption data obtained from cementitious materials that still contain blast furnace slag. As discussed in the context of saltstone Tc release in Section 2.7.1, the DOE conceptual model for oxidizing conditions in any cementitious material assumes that the reducing agent (e.g., blast furnace slag) has been exhausted. Therefore, sorption experiments on material originally containing a reducing agent may not faithfully reflect modeled oxidizing conditions, even if conducted under air using an initially oxidizing aqueous solution. This observation has potential implications for measurements such as those in SRNS-STI-2008-00045, used as the basis for oxidizing conditions K_d values of redox-sensitive elements such as Tc and Pu.

The NRC review of the K_d values focused on the radionuclides that had a greater potential to be risk significant. The NRC review of those radionuclides (i.e., Tc, I, Ra, Se, Sr) is described below.

Technetium K_d Value for Cementitious Materials

The release of Tc from the saltstone monolith is sensitive to the oxidation state of the Tc (Section 2.6). The NRC staff has been monitoring information regarding Tc reduction in saltstone as part of Open Issue 2009-1. The modeling of the reduction and sorption of Tc (i.e., the K_d value for Tc) is therefore one of the most risk-significant portions of the PA.

In the PA cases (i.e., the cases other than Cases K, K1, and K2), DOE used a "shrinking core" model for the release of Tc. In this model, a value of 1,000 mL/g was used for the K_d for Tc under reducing conditions to immobilize the Tc. Case K also uses a value of 1,000 mL/g Tc K_d

value for cementitious materials under reducing conditions, while Cases K1 and K2 assume a value of 500 mL/g (Table 2.7-2).

In comment SP-15 (NRC, 2010b, i), the NRC staff asked DOE to provide further support for the adopted Tc pseudo-K_d of 1,000 mL/g because this value was not consistent with K_d values measured for simulated saltstone under reducing conditions (e.g., values of 6.5 mL/g and 91.3 mL/g reported in SRNS-STI-2008-00045). In the initial response to the comment (SRR-CWDA-2010-00033), DOE cited a “recommended” value of 5,000 mL/g as the basis for selecting a value of 1,000 mL/g. In the second RAI (NRC, 2010i), NRC staff noted that the 5,000 mL/g value was measured for a formulation that included a strong reducing agent and is very different than the saltstone formulation.

In the second response comment SP-15 (SRR-CWDA-2011-00044), DOE referred to a recent report summarizing literature and new laboratory data (SRNL-STI-2010-00668). The new sorption data reported in SRNL-STI-2010-00668 (originally reported by Lilley, 2010) are used by DOE to justify a Tc reducing K_d of 1,000 mL/g. In this report, Tc K_d values for simulated saltstone under reducing conditions were measured to be 711 mL/g after 22 days and 518 mL/g after 56 days. However, NRC staff notes that this measurement was performed in a 2% H_2 atmosphere, so this result may not be applicable to the actual saltstone system. In fact, in SRNL-STI-2010-00668, the reported K_d values for Tc sorption onto concrete that does not contain slag (i.e., concrete that does not contain a reducing agent) were comparable to those seen for the simulated saltstone. DOE subsequently concluded that the 2% H_2 atmosphere was responsible for the observed Tc reduction and sorption in these experiments (SRNL-STI-2011-00716). In a more recent report (SRNL-STI-2011-00716), a Tc K_d value of 1,258 mL/g was reported. This measurement was also made in an atmosphere containing 0.1% H_2 in the presence of a palladium catalyst to convert O_2 to water through reaction with H_2. Unlike the results of the study using 2% H_2 (SRNL-STI-2010-00668), a concrete sample without slag in the 0.1% H_2 atmosphere did not show Tc reduction or sorption (SRNL-STI-2011-00716). This result indicates that the observed reduction and sorption of Tc should be attributable to saltstone rather than an experimental artifact (SRNL-STI-2011-00716). However, the Tc K_d for saltstone under reducing conditions was also measured in a leaching experiment using a saltstone core sample (SRNL-STI-2010-00667). In this experiment, a K_d value of 139 mL/g was measured for Tc. This measurement was performed under an atmosphere of 30 to 60 ppm O_2. Because DOE has not shown that the SDF will be completely free of even trace levels of O_2, it is unclear if the results of the study performed in the 1% H_2 atmosphere with a palladium catalyst (SRNL-STI-2011-00716) are applicable to the actual saltstone system.

The Tc K_d information described above is summarized in Table 2.7-2. Based on this information, the NRC staff concludes that the assignment of a K_d of 1,000 mL/g to Tc in saltstone under reducing conditions is still not adequately supported. The Tc K_d measurements for simulated saltstone that have been performed to date have either not supported a Tc K_d value of 1,000 mL/g for saltstone under reducing conditions, or the experiments have been performed in an atmosphere that is significantly different than for the actual as-emplaced saltstone. Furthermore, the use of a reducing K_d value requires that oxygen is essentially absent from the system, because studies have shown that even very small quantities of oxygen (i.e., 30 to 60 ppm O_2), in a system oxidize Tc in saltstone (e.g., SRNL-STI-2010-00668). DOE has not demonstrated that oxygen will be absent from the water-saltstone system when release is possible. Although the NRC staff finds that there is also insufficient basis for the Tc K_d value of 500 mL/g assumed in Cases K1 and K2, this K_d value is more defensible than the value of 1,000 mL/g.

In the PA cases (i.e., cases other than Cases K, K1, and K2), a Tc K_d value of 0.8 mL/g was assumed for the sorption of Tc to saltstone under oxidizing conditions. A value of 0.8 mL/g was also assumed in Case K1, while Cases K and K2 use a value of 10 mL/g (Table 2.7-3).

Measurements of the Tc K_d value for simulated saltstone under oxidizing conditions show little sorption occurring. SRNS-STI-2008-00045 measured average K_d values of 0.25 and -0.02 mL/g, respectively, for partially oxidized saltstone in $Ca(OH)_2$ and $CaCO_3$ solutions, and the corresponding measurements for reducing saltstone under oxidizing conditions were also below 1 mL/g. In SRNL-STI-2009-00636, a measured Tc K_d value of 5 mL/g was measured for simulated saltstone under oxidizing conditions. This research is consistent with frequent observations of Tc mobility in non-reducing cementitious materials (e.g., WSRC-RP-2007-01122; Langton, 1988; Serne, et al., 1992).

In SRR-CWDA-2011-00044 response SP-19, DOE stated that the new oxidizing saltstone Tc K_d of 10 mL/g used for Cases K and K2 was based on a single measurement of 12 mL/g for adsorption from a saltstone core from Vault 4 (SRNL-STI-2010-00667). The saltstone core sample was only partially oxidized, meaning that blast furnace slag was likely still present in the solid. The staff notes that Tc release is affected not only by the redox character of the water used in the experiment; the effect of any remaining blast furnace slag in the saltstone could be to continue to sequester a fraction of the Tc even if the water itself is oxidizing. Because the conceptual model for saltstone Tc release assumes that oxidizing conditions are in effect when all the reducing agent (i.e., blast furnace slag) in the grout is exhausted, experiments that use saltstone still containing blast furnace slag do not faithfully reflect the oxidizing conditions simulated in the PA. This caveat applies also to the interpretation of other Tc release experiments such as column or monolith leaching (e.g., Langton, 1986, 1988; Harbour and Aloy, 2007; Gilliam, et al., 1990; Pabalan, et al., 2010).

Based upon the above information (as summarized in Table 2.7-3), the NRC staff concludes that a Tc K_d value of 0.8 mL/g is more appropriate than 10 mL/g for saltstone under oxidizing conditions. The NRC staff therefore concludes that of Case K, K1, and K2, the case that has the most appropriate Tc K_d values is Case K1.

Table 2.7-2: Saltstone K_d Value for Tc under reducing conditions

	K_d Value (mL/g)	Notes
Modeled Values		
Case A	1000	
Case K	1000	
Case K1	500	
Case K2	500	
Measured K_d Values for Simulated Saltstone		
SRNS-STI-2008-00045	6.5 to 91.3	Measurements made in solutions of $Ca(OH)_2$ and $CaCO_3$ that were purged with N_2. DOE noted that the quality of the spectra in these measurements was compromised by a U-233 shift.
SRNL-STI-2009-00636	32 [(2)] (9.1 to 56)	Experiments performed in a glovebox with a 2% H_2 environment. Measurement taken after 4 d and may not have reached steady state.[1]
SRNL-STI-2010-00668	711 [(2)]	Experiments performed in a glovebox with a 2% H_2 environment. Measurement taken after 22 d.[1]
	518 [(2)]	Experiments performed in a glovebox with a 2% H_2 environment. Measurement taken after 56 d.[1]
SRNL-STI-2011-00716	1258 [(2)] (757 to 1759)	Experiments performed in 0.1% H_2 atmosphere in the presence of a palladium catalyst to convert O_2 to water through reaction with H_2). Measurement taken after 56 days.[1]
Measured K_d Values for Saltstone Core Sample		
SRNL-STI-2010-00667	139	Leached under N_2 using a $Ca(OH)_2$ solution. DOE noted that the nitrogen glovebag contained small amounts of O_2. Measurement taken after 20 d.

[1] Measurements performed using a solution of $CaCO_3$ unless otherwise reported
[2] Measurements performed in an environment containing H_2 (g)

The Tc K_d value for saltstone under reducing conditions is one of the most important modeling parameters in the PA for the SDF. Because the NRC staff does not find that the value DOE used in the PA is adequately justified, the NRC staff will monitor the development of additional information regarding sorption-reduced saltstone (in addition to sorption in oxidized saltstone, as previously discussed). Also, the same K_d values were assumed for the disposal unit concrete as for the saltstone. Because the disposal unit concrete has a smaller fraction of slag than saltstone, it is expected that the sorption will be less for the disposal unit concrete. The NRC staff will therefore also monitor the development of additional justification for sorption in reduced disposal unit concrete (in addition to the K_d in oxidized concrete).

Table 2.7-3: Saltstone K_d Value for Tc under oxidizing conditions

	K_d Value (mL/g)	Notes
Modeled Values		
Case A	0.8	
Case K	10	
Case K1	0.8	
Case K2	10	
Measured K_d Values for Simulated Saltstone		
SRNS-STI-2008-00045	0.16 to 0.93	Measurements made in solutions of $Ca(OH)_2$ and $CaCO_3$.
	-0.02 to 0.25	Simulated saltstone was partially oxidized prior to experiment. Measurements made in solutions of $Ca(OH)_2$ and $CaCO_3$.
SRNL-STI-2009-00636	5.0 (3.1 to 6.3)	Measurement taken after 4 d and may not have reached steady state.[1]
Measured K_d Values for Saltstone Core Sample		
SRNL-STI-2010-00667	12	Leached using a $Ca(OH)_2$ solution. Purged with air continuously. Measurement taken after 20 d.

[1] Measurements performed using a solution of $CaCO_3$ unless otherwise reported.

Iodine K_d Value for Cementitious Materials

The K_d values DOE assumed for I in the performance assessment are presented in Table 2.7-4. Cementitious material I K_d values were the subject of an NRC RAI (NRC, 2010b, i; SP-14). The main NRC concerns were that laboratory measurements suggesting no I sorption were neglected and that grout formulations used in experiments differed substantially from saltstone. The reducing middle age value (9 mL/g) initially appears reasonable in light of experimental data in SRNS-STI-2008-00045, and as discussed in the DOE response in comment SP-14 in SRR-CWDA-2011-00044, a later study using more appropriate saltstone and concrete samples that yielded values similar to 9 mL/g (SRNL-STI-2009-00636). The NRC staff notes, however, that the short-term I experiments presented in SRNL-STI-2009-00636 and cited in the comment SP-14 response (SRR-CWDA-2010-00033) appear to have been considered unreliable in the later compilation in the Lilley (2010) thesis. The long-term reducing experiments gave somewhat lower I K_d values of less than 5 mL/g (Lilley, 2010, Figure 4.26). Similarly, data on cementitious materials under oxidizing conditions included in SRNL-STI-2009-00636 were later not included in Lilley (2010). These data were not cited as the source for the oxidized values in the PA (Table 4.2-18); rather, the oxidized middle and old age values of 15 and 4 mL/g, respectively, were ascribed to Kaplan and Coates (2007) (WSRC-RP-2007-01122). The Lilley (2010, Figure 4.23) long-term I experiments on more appropriate cement and saltstone samples yielded K_d clustering in the range 2 to 8 mL/g—lower than the adopted oxidized middle age value of 15 mL/g.

Table 2.7-4: K_d values (mL/g) assumed by DOE for I for cementitious materials

	Young Aged	Middle Aged	Old Aged
Reducing Conditions			
PA Cases (i.e., cases other than Cases K, K1, and K2)	5	**9**	0
Cases K, K1, and K2	5	**9**	4
Oxidizing Conditions			
PA Cases (i.e., cases other than Cases K, K1, and K2)	8	**15**	**4**
Cases K, K1, and K2	8	**15**	**4**

From Table 4.2-18 in the PA and Table PA-8.6 in SRR-CWDA-2011-00044
Bold values indicate K_d values that were used in the PORFLOW™ model

The NRC staff notes that literature data on I retention are variable. Fuhrmann, et al. (2006) found very low I K_d values (0.7 and 0.8 mL/g) for fresh reducing grouts under air and showed that Portland cement was the main I sorptive material. Other sorption and diffusion studies have found that I can be effectively partially retained on cementitious materials (e.g., Atkins and Glasser, 1990; Bonhoure, et al., 2002; Mattigod, et al., 2001).

The NRC staff concludes that the K_d values assumed for I for cementitious materials seem to be slightly higher than measured values. This could result in a minor underestimation of the predicted dose. The NRC staff notes that model support for the K_d values in the PA is important and NRC staff will be monitoring this under PA maintenance (i.e., NRC staff will monitor the K_d values selected for I, and their related model support, in the next revision to the PA).

Radium K_d Value for Cementitious Materials

The K_d values assumed by DOE for Ra for cementitious materials in the performance assessment are listed in Table 2.7-5. Whereas the Ra cementitious material K_d values for oxidizing conditions are based on the literature, the reducing values in the PA are set equal to Sr values. The contrasting sources of information are responsible for the unexpectedly large differences in the two sets of values for an element that is generally insensitive to oxidation state. In the DOE responses to the comment SP-14 (SRR-CWDA-2010-00033 and SRR-CWDA-2011-00044), DOE explained the source of the contrast and stated that they will consider future studies on the transport properties of this potentially risk-significant element.

The NRC staff agrees that the potential future studies would be useful, particularly because directly applicable literature data on Ra are lacking.

Table 2.7-5: K_d values (mL/g) assumed by DOE for Ra for cementitious materials

	Young Aged	Middle Aged	Old Aged
Reducing Conditions			
PA Cases (i.e., cases other than Cases K, K1, and K2)	0.5	**3**	20
Cases K, K1, and K2	100	**100**	70
Oxidizing Conditions			
PA Cases (i.e., cases other than Cases K, K1, and K2)	100	**100**	**70**
Cases K, K1, and K2	100	**100**	**70**

From Table 4.2-18 in the PA and Table PA-8.6 in SRR-CWDA-2011-00044
Bold values indicate K_d values that were used in the PORFLOW™ model

In SRR-CWDA-2011-00044 (Table PA-8.6) DOE presented Case K K_d values for Ra on reducing cementitious materials that are much higher than the previous values. The young K_d was increased from 0.5 to 100 mL/g, the middle value from 3 to 100 mL/g, and the old value from 20 to 70 mL/g. These changes made the reducing K_d values identical to the oxidizing K_d values (PA; Table 4.2-18). As discussed in the SP-14 comment and response (SRR-CWDA-2011-00044), Ra is not expected to be redox sensitive and the reducing values were previously based only on Sr data. This approach is reasonable, but it must also be pointed out that there appears to still be no experimental Ra sorption data on saltstone or a similar formulation. Bayliss, et al., (1989) measured K_d values of 860 mL/g and higher on a mixture of ordinary Portland cement, blast furnace slag, and limestone aggregate. The inclusion of the aggregate and the absence of fly ash call into question the applicability of the data.

In Case A, and many of the other cases, the dose from Ra-226 dominates the dose. The dose from Ra-226 in Case K was much smaller due to this case including a much smaller inventory of Ra-226 and its ancestors, Th-230 and U-234.

Because of the potential risk significance of this radionuclide and the lack of applicable data, the NRC staff will monitor measurements made of the K_d of Ra for saltstone and for reducing disposal unit concrete. However, as discussed in Section 2.2, the inventory of Ra-226 and its ancestors is uncertain. If strong justification is provided for the reduced inventory, then this radionuclide might no longer be risk significant and this monitoring factor might no longer be necessary.

Selenium K_d Value for Cementitious Materials

In the PA a Se K_d value of 300 mL/g was assumed for young age, middle age, and old age reduced cementitious materials (Table 2.7-6). However, a footnote to this table states that it was discovered after the PORFLOW™ analysis were completed that the old aged K_d for reduced cementitious materials should have been 150 mL/g. In Cases K, K1, and K2, the Se K_d for old aged reduced cementitious materials was revised from 300 to 150 mL/g. The previous value for young and middle aged material of 300 mL/g was retained. The basis for these values is provided in SRNL-STI-2009-00473 (Table 17), which refers to selenate sorption measurements (250 to 930 mL/g) by Johnson, et al. (2000). However, that study actually concluded that its measurements were for selenite, a more reduced form of aqueous Se. The Johnson, et al. (2000) data are, therefore, reasonable to consider for moderately reducing conditions. A number of studies (Sugiyama and Fujita, 1999; Ochs, et al., 2002; Baur and Johnson (2003); Pointeau, et al. 2004) support the strongly sorptive behavior of selenite onto cementitious materials, with typical K_d values of 100 mL/g and higher. Based on this information, the NRC staff concludes that the Se K_d values assumed for reduced cementitious materials appear to be reasonable.

Table 2.7-6: K_d values (mL/g) assumed by DOE for Se for cementitious materials

	Young Aged	Middle Aged	Old Aged
Reducing Conditions			
PA Cases (i.e., cases other than Cases K, K1, and K2)	300	**300**	300
Cases K, K1, and K2	300	**300**	150
Oxidizing Conditions			
PA Cases (i.e., cases other than Cases K, K1, and K2)	300	**300**	**150**
Cases K, K1, and K2	300	**300**	**30**

From Table 4.2-18 in the PA and Tables PA-8.6 and PA-8.7 in SRR-CWDA-2011-00044
Bold values indicate K_d values that were used in the PORFLOW™ model

However, the NRC staff does not find that there is an adequate technical basis for the Se K_d values of 300 and 150 mL/g for oxidized middle and old age conditions, respectively. The DOE response (SRR-CWDA-2011-00044, comment C-4) to a follow-up to an original RAI response (SRR-CWDA-2010-00033, comment C-4) did not provide sufficient information to show that selenite would be the important species under oxidizing conditions or that selenate, the expected oxidized species, would have K_d values in excess of the 30 to 79 mL/g range measured by DOE for simulated saltstone and FDC concrete (SRNS-STI-2008-00045). The NRC staff notes that, in Cases K, K1, and K2, DOE used an oxidized old age Se K_d of 30 mL/g (SRR-CWDA-2011-00044, Table PA-8.7) and Se-79 did not become an important dose contributor. However, in these cases, the saltstone did not transition to old age within 20,000 years, and the disposal unit concrete did not transition to old age until 7,300 to 10,700 years. DOE did not test the sensitivity of Se dose to a lower Se K_d for oxidized middle age conditions. The NRC staff therefore concludes that better supported information is needed for the K_d for Se for saltstone and disposal unit concrete under oxidizing conditions. The NRC staff will monitor the development of this information.

Strontium K_d Value for Cementitious Materials

The Sr K_d values assumed in the original PA cases (Table 2.7-7) were based on site specific measurements as documented in WSRC-STI-2007-00640. The NRC staff finds that these values are appropriate.

Table 2.7-7: K_d values (mL/g) assumed by DOE for Sr for cementitious materials

	Young Aged	Middle Aged	Old Aged
Reducing Conditions			
PA Cases (i.e., cases other than Cases K, K1, and K2)	0.5	**3**	20
Cases K, K1, and K2 (disposal unit concrete)	15	**15**	5
Cases K, K1, and K2 (saltstone)	1000	**1000**	NA
Oxidizing Conditions			
PA Cases (i.e., cases other than Cases K, K1, and K2)	3	**30**	**15**
Cases K, K1, and K2 (disposal unit concrete)	15	**15**	**5**
Cases K, K1, and K2 (saltstone)	1000	**1000**	**NA**

From Table 4.2-18 in the PA and Tables PA -8.5, PA-8.6, and PA-8.7 in SRR-CWDA-2011-00044
Bold values indicate K_d values that were used in the PORFLOW™ model

In Cases K, K1, and K2, the Sr K_d values for reducing and oxidizing cementitious materials were revised (Table 2.7-7). This revision was based on recommended values in SRNL-STI-2010-00667. This report cited the same site-specific measurements that were used to determine the K_d values in the original PA calculations, so it is not clear to the NRC staff what the basis was for changing the recommended values. The NRC staff also notes that the revised value of 15 mL/g for middle aged reduced materials is higher (i.e., less conservative) than the site-specific measured value of 2.9 mL/g (WSRC-STI-2007-00640; Table 11).

In Cases K, K1, and K2, the K_d values for Sr for saltstone were also changed upward on the order of two to three orders of magnitude to 1,000 mL/g. The new value was based on desorption experiments on an actual ground saltstone core sample from Vault 4, which had evidence for partial oxidation (SRNL-STI-2010-00667). The measured K_d values, were 5,728 and 737 mL/g for oxidizing and reducing conditions, respectively. The measurements of desorption K_d values on an actual Vault 4 saltstone sample provide an important source of information on radionuclide release for performance assessment. However, it is not clear to the NRC staff that sufficient data have been obtained to support using the derived Sr values in performance assessments. For example, the investigators speculate that the dissolved Sr measurements may, in fact, be controlled by $SrSO_4$ solubility rather than sorption. There is a suggestion in the text that $SrSO_3$ could also be a controlling solid, perhaps under more reducing conditions. In any case, if the amount of desorption observed in a leaching experiment is controlled by solubility, the experimental artifact will result in the calculated K_d value calculated being artificially high. Furthermore, in the measurement of the K_d for oxidizing conditions, the saltstone sample was only partially oxidized (SRNL-STI-2010-00667), meaning that blast furnace slag was likely still present in the solid. Because the conceptual model for saltstone radionuclide release assumes that oxidizing conditions are in effect when all the reducing agent (i.e., blast furnace slag) in the grout is exhausted, experiments that use saltstone still containing blast furnace slag do not necessarily reflect the oxidizing conditions simulated in the performance assessment.

The revised K_d value for Sr for saltstone has the potential to significantly limit the modeled Sr release from the saltstone. Because of the importance of Sr releases to the potential dose to an inadvertent intruder (Chapter 3), the NRC staff will monitor the development of additional measurements for the sorption or leaching of Sr from saltstone. Additionally, the NRC staff will monitor the basis for the selection of the reduced middle age K_d value for the disposal unit concrete in Cases K, K1, and K2 as part of PA maintenance (Appendix A).

2.7.4.3 NRC Evaluation - Soil Sorption Coefficients (K_d values)

The NRC staff review of the soil K_d values used for the saltstone performance assessment (PA; Table 4.2-15 and SRR-CWDA-2011-00044; Table PA-8.4) focused on elements considered potentially risk significant. The NRC staff finds that the overall DOE approach to establishing K_d values for soil is appropriate (i.e., employing site-specific data when possible and relying on literature data when site-specific data are lacking or sparse). However, the NRC staff notes that it is preferable to use site-specific K_d values for radionuclides that are potentially risk significant due to the large variability in K_d values from site to site.

The Tc K_d values in PA Table 4.2-15 were based on the analysis in SRNL-TR-2009-00019, which relied on the median of existing data and more recent K_d measurements on 24 samples from different depths in an E-Area borehole (SRNS-STI-2008-00286). The PA values of 1.8 and 0.6 mL/g for clayey and sandy soils, respectively, appear reasonable in light of the new borehole dataset; in fact, these values are lower than nearly all the SRNS-STI-2008-00286 measurements.

In the PA calculations (i.e., in cases except for Cases K, K1, and K2), a K_d value of 0 mL/g (i.e., assumption of no sorption) was used for I in vadose/sandy soil. The NRC finds this assumption of no sorption to be appropriate for these potentially mobile elements in the absence of site-specific data for vadose/sandy soil. The backfill/clayey soil K_d value of 0.6 mL/g for I was reasonably based on site-specific data. In Cases K, K1, and K2, for I, the soil K_d values were changed from 0 to 0.3 mL/g for sandy soil, and from 0.6 to 0.9 mL/g for clayey soil (SRR-CWDA-2011-00044, Table PA-8.4). DOE cited recent studies conducted with I and SRS soils (SRNL-STI-2009-00473, Table 16) as the basis for the revised values. This report describes research performed by Schwehr, et al. (2009) that showed that I sorbs to SRS soils more readily at lower aqueous I concentrations (i.e., those more reflective of environmental conditions) and that more sorptive organo-I and iodate can be more abundant at low I concentrations. These observations support the somewhat higher K_d values for I assumed in Cases K, K1, and K2.

The NRC staff finds that the Sr K_d soil values are appropriately based on site data as described in WSRC-TR-2006-00004. In this reference it is noted that other measurements of the Sr K_d on site found a very strong pH effect on the Sr K_d values, with lower K_d values being measured at lower pH values. Because the SDF will have large amounts of cementitious materials that are highly alkaline, NRC staff expects that the pH values downgradient of the SDF will not be low.

In the original PA calculations, the Ra soil K_d values were set equal to the Sr K_d values using the chemical homologue argument. The Ra values are reasonable because subsequent site-specific measurements yielded significantly higher values (SRNL-STI-2010-00527). The revised sorption dataset used in Cases K, K1, and K2 also incorporates increases in Ra K_d values, with the sandy value changed from 5 to 25 mL/g and the clayey value from 17 to 185 mL/g (SRR-CWDA-2011-00044, Table PA-8.4). The changes were recommended in SRNL-STI-2011-00011 on the basis of measurements reported in SRNL-STI-2010-00527. The two Ra K_d values resulted from measurements at an ionic strength of 0.02 M (i.e., the approximate ionic strength of SRS groundwater) that included added Sr and considered a range of starting aqueous Ra concentrations. While there was some scatter in the sorption isotherms (SRNL-STI-2010-00527, Figures 3-1 and 3-2), the results are appropriately interpreted as central value numbers based on site-specific analyses.

Se K_d values of 1,000 mL/g for both clayey and sandy soils were the subject of an NRC RAI (NRC, 2010b, i; FFT-3). In the RAI, NRC staff noted this K_d value was representative of low-pH soil that the Se sorption is less at neutral and high pH values. In the initial response to the RAI, DOE provided the results of a sensitivity analysis performed using the probabilistic GoldSim® model in which the Se K_d values for the sandy and clayey soils set to 0 mL/g (SRR-CWDA-2010-00033). The results of this sensitivity analysis showed that changing the Se K_d values resulted in less than a 3% increase in the peak dose within 20,000 years. However, although 3% represents a small absolute increase in dose, it represents a large relative increase in the dose derived from Se-79. Also, as discussed in more detail in Section 2.11.4.1, the NRC staff has number of concerns about the GoldSim® model. In the response to the second RAI (SRR-CWDA-2011-00044), DOE provided additional information regarding the pH dependence of the Se K_d for soils and the pH in monitoring wells in Z area. DOE stated that Se K_d values will decrease sharply as the pH increases above pH 6; and decrease an order of magnitude as the pH value approaches 7. DOE further noted that individual pH readings for 72 samples collected over a 5-year time period from 9 wells ranged from 4.0 to 7.8 and that out of the 72 measured pH values only six readings were above a pH of 7. However, a total of 30 out of the 72 samples have pH values that are greater than or equal to 6. Furthermore, the pH of water in the near field may be elevated by the presence of saltstone and the cementitious disposal units. It is therefore not clear to the NRC staff that there is a basis for assuming that the soils in Z area will have a low pH. Additionally, the NRC staff notes that the multiple measurements of a Se K_d of 1,041 mL/g in the original report cited by DOE for the Se K_d values (WSRC-STI-2006-00037; Table 6) appears unusual and is suggestive of an experimental artifact that has not been explained. Based on above information, the NRC staff finds that the basis for choosing a K_d representative of low-pH soil as compared to a more neutral soil is unclear, especially in light of the potential for alkaline buffering of the vadose zone soils by the significant quantity of cementitious materials in the SDF. The NRC staff will therefore monitor the K_d value assumed for Se in sand and clay soils (Appendix A).

2.7.4.4 NRC Evaluation of PORFLOW™ Modeling of Near-Field Flow

The NRC staff reviewed DOE's near-field model constructed with the PORFLOW™ modeling code and generally found DOE's approach to near-field model to be reasonable. However, the NRC staff found that, in several cases, parameter values and modeling assumptions resulted in severely constrained flow through saltstone. Specifically, in Vault 4, the Darcy velocity through saltstone in Case A remains below 0.05 cm/yr for more than 10,000 years after closure (Figure 2.7-1). The flow through the saltstone matrix remains even lower in the Synergistic Case, in part because almost all of the water that reaches saltstone is predicted to flow through fractures (Figure 2.13-4). In the FDCs, the Darcy velocity through saltstone in the Increased Hydraulic Conductivity Case and Case E remain less than 0.1 cm/yr for 10,000 years after closure, and the flow through saltstone in Case A remains less than 0.01 cm/yr for 10,000 years after closure. As in Vault 4, the flow through the saltstone matrix in the Synergistic Case is similar to the flow through saltstone in Case A because almost all of the water moving through saltstone in the Synergistic Case flows through fractures.

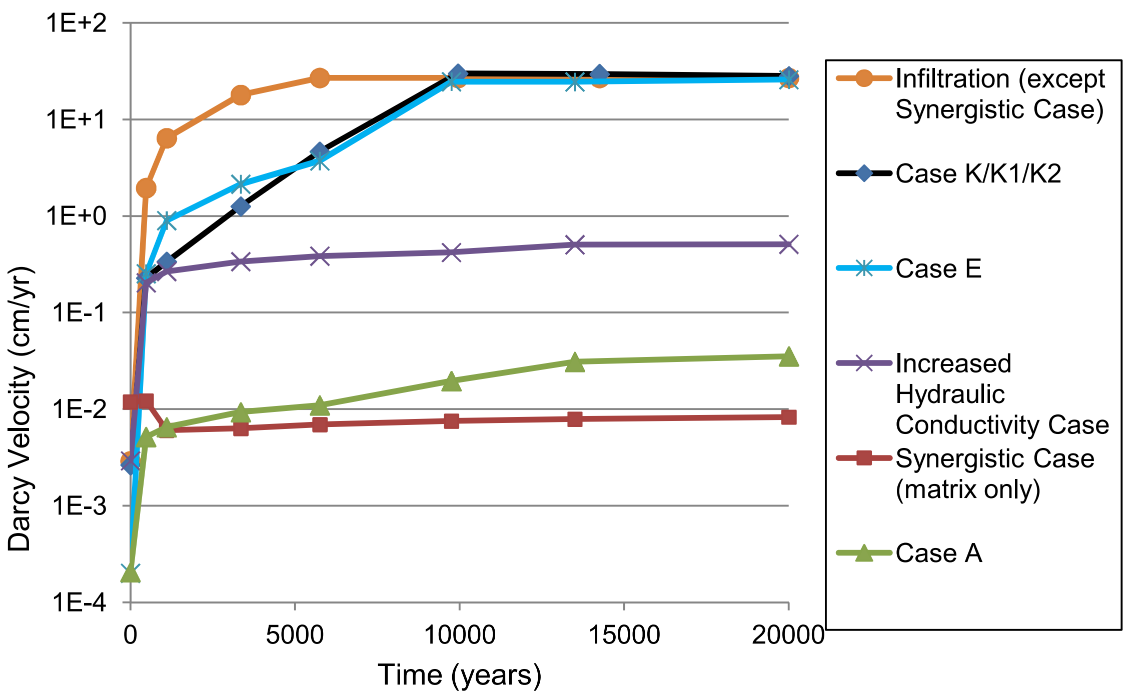

Figure 2.7-1: Vertical Darcy velocity through saltstone in Vault 4 predicted with the DOE PORFLOW™ model. Data for the Synergistic Case excludes fractures (other cases do not represent fractures explicitly). Data taken from DOE's STAT.out files (NRC, 2010g).

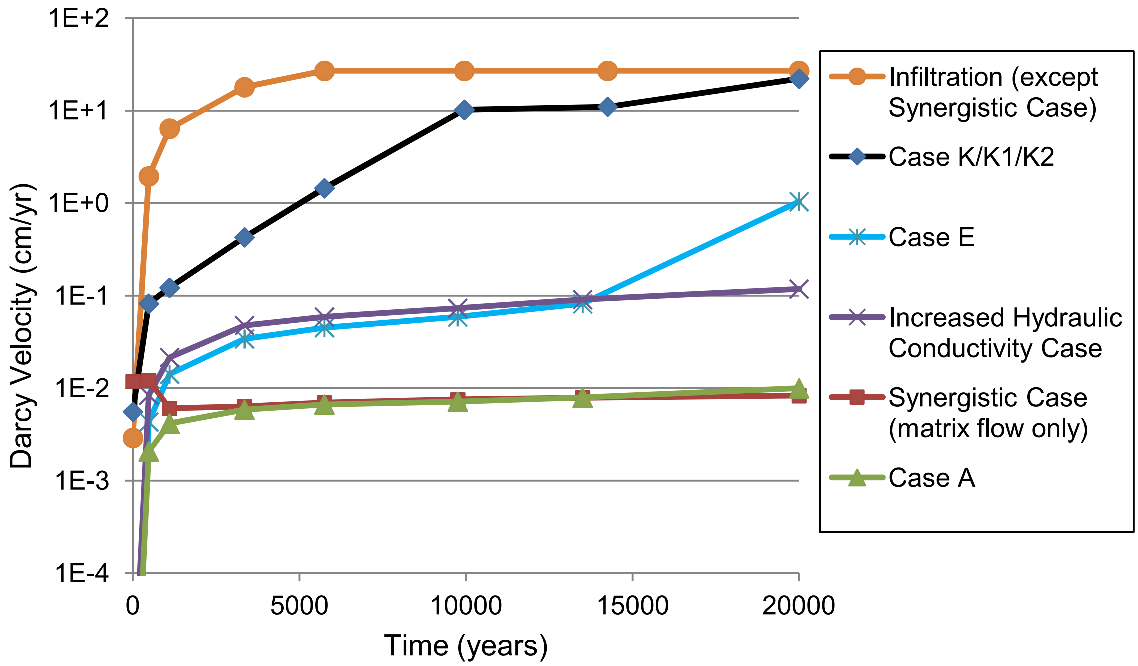

Figure 2.7-2: Vertical Darcy velocity through saltstone in an FDC predicted with the DOE PORFLOW™ model. Data for the Synergistic Case excludes fractures (other cases do not represent fractures explicitly). Data taken from DOE's STAT.out files (NRC, 2010g).

As discussed in more detail in Sections 2.5 and 2.6, the NRC staff found that the assumed low hydraulic conductivities of saltstone and the disposal unit concrete used in Case A and many of the deterministic sensitivity cases were overly optimistic. Similarly, assumptions regarding the performance of the lateral drainage layer, the disposal unit roofs, and the HDPE/GCL layer (for the FDCs) resulted in a significant amount (i.e., greater than 99% of the infiltrating water being shed around the disposal units even 10,000 years after site closure in Case A and several of the deterministic sensitivity analyses (Section 2.5) (SRR-CWDA-2011-00044; IEC-8). As discussed in Sections 2.5 and 2.6, the NRC staff finds that Cases K, K1, and K2 more realistically represent the current and expected future hydraulic and physical properties of the saltstone and disposal units. In these cases, the modeled flow through the saltstone and disposal units does not appear to be inappropriately constrained (Figures 2.7-1, 2.7-2, and 2.13-1).

The NRC staff also found that the moisture characteristic curves DOE used in the PA cases (i.e., cases other than Cases K, K1, and K2) to represent flow through unsaturated cementitious porous media appear to constrain flow unrealistically. In particular, in the PA cases, DOE assumes the relative permeabilities of intact saltstone and fractured cementitious materials decrease by several orders of magnitude with minor decreases in moisture content (NRC, 2010b; SP-3 and SP-4) (Figures 2.7-3 and 2.7-4). As a result, small decreases in the saturation of the material cause unrealistically large decreases in the modeled flow. The large decrease in the relative permeability with decreasing saturation appears to be unjustified because the moisture characteristic curves assumed in the PA for intact cementitious material are substantially different from those found in the literature (NRC, 2010b; SP-3). For this reason, the NRC staff is concerned that the large decreases in the modeled flow with small changes in saturation could lead to an underestimate of radionuclide release from the waste form and an underestimate of dose.

In response to NRC's concern about the moisture characteristic curves in intact saltstone, DOE agreed that the moisture characteristic curve utilized in the PA for intact saltstone is inconsistent with the literature (SRR-CWDA-2010-00033; SP-3). To evaluate the impact of moisture characteristic curves, DOE performed a sensitivity case based on Case A with the relative permeability for intact saltstone fixed at 1.0 (i.e., the hydraulic conductivity was not reduced based on moisture characteristic curves) for the most risk significant radionuclides. The resulting contaminant release rate was approximately twice that of the original Case A result for an FDC within 10,000 years after site closure. For Vault 4, with the relative permeability equal to 1.0, the release rate of Tc-99 was almost doubled, while the I-129 and Ra-226 rates increased by less than 30% as compared to the original Case A values. DOE stated that these increases in release rates would not significantly impact the resulting dose to the member of the public. In addition, DOE performed a sensitivity case based on the Synergistic Case with a revised saltstone relative permeability of 1.0 and concluded that the sensitivity case showed no significant impact on the resulting dose (SRR-CWDA-2010-00033, VP-2). Based on these analyses, DOE concluded that the moisture characteristic curve used in the PA for intact saltstone does not appreciably impact the estimated dose to the member of the public.

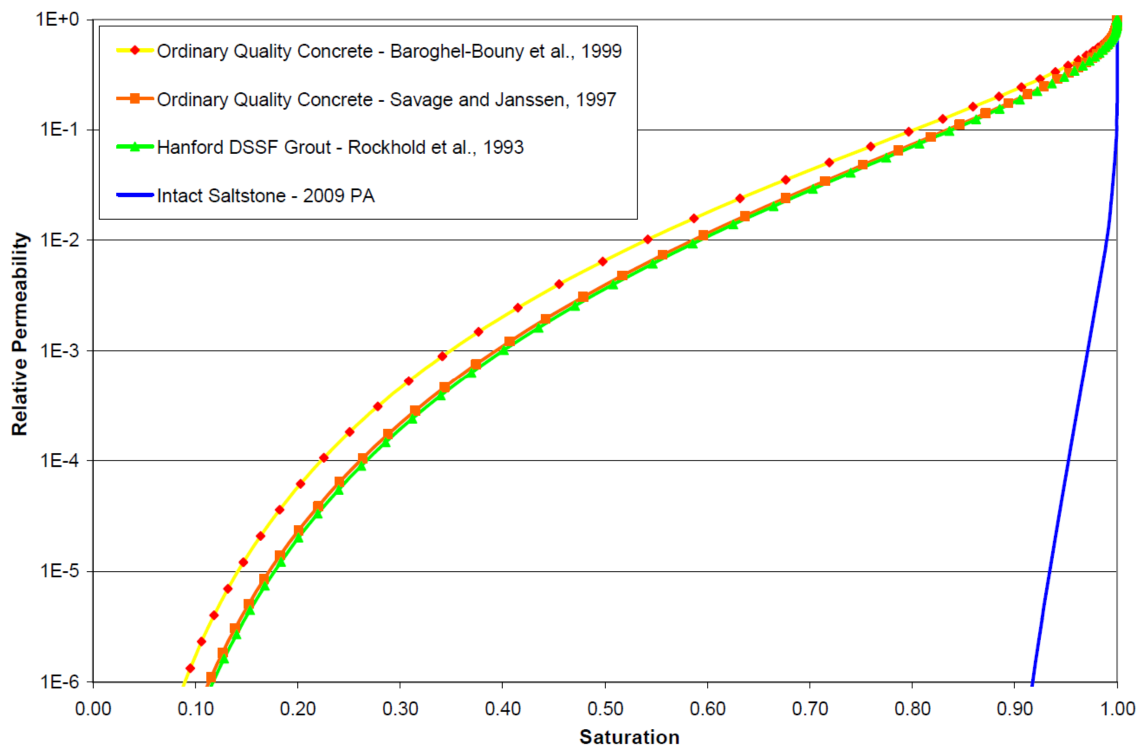

Figure 2.7-3: Moisture Characteristic Curves Adapted from the PA and WSRC-STI-2006-00198 (SRR-CWDA-2010-00033; Figure 1)

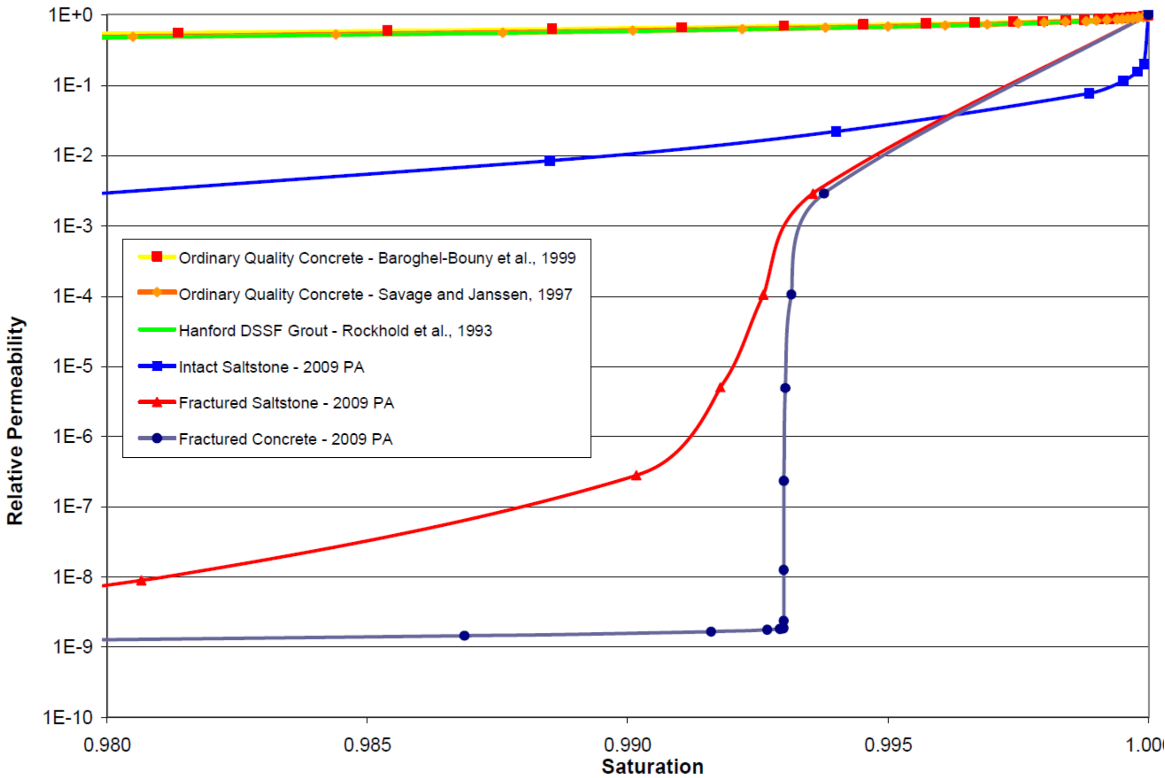

Figure 2.7-4: Characteristic curves for the intact and fractured saltstone and concrete as adapted from the 2009 PA and SRNL-STI-2009-00115 (SRR-CWDA-2010-00033; Figure 2)

However, as NRC indicated in response to these DOE analyses (NRC, 2010i; SP-3), one-off sensitivity analyses may result in insignificant increases in the dose on an absolute basis; however, the effect on dose results may be significant when there are many outstanding uncertainties that are evaluated on a cumulative basis. In response to the second RAI (SRR-CWDA-2011-00044), the DOE provided a more comprehensive analysis, Case K, to address the NRC staff's concern with DOE's moisture characteristic curve for intact saltstone as well as other concerns. In this analysis, DOE fixed the relative permeability to one for the cementitious materials (i.e., moisture characteristic curves were not used). NRC staff finds that this approach is reasonable due to the (i) uncertainty in unsaturated flow through cementitious materials over long periods of time and (ii) limited amount of reduction in flow that would be expected for cementitious materials that are predicted by DOE to be near saturation. Case K also addressed NRC staff's concern with the moisture characteristic curves that were used to represent unsaturated flow through fractured saltstone and concrete.

DOE used an analytical approach to develop curves for fractured cementitious materials in the PA based on the moisture characteristic curves that were to represent intact saltstone. Figure 2 of the first RAI response (SRR-CWDA-2010-00033) shows the curves for fractured saltstone (used to represent the saltstone grout in Case E) and fracture concrete (used to represent the walls of Vaults 1 and 4 in all of the cases in the PA except the Synergistic Case) (Appendix E of SRNL-STI-2009-00115; SRR-CWDA-2010-00033). The figure illustrates that a decrease in the saturation of the walls of Vaults 1 and 4 from 100% to 99% results in an unrealistic decrease of nine orders of magnitude in relative permeability. The NRC staff's concern, as discussed in the second comment SP-4 (NRC, 2010i), is that DOE's abstraction of unsaturated fracture flow significantly underestimates actual flow rates through a fractured system. As discussed in the previous paragraph, Case K addresses NRC staff's concern about the use of use moisture characteristic curves to represent fractured cementitious materials by setting the relative permeability equal to 1.0.

2.7.4.5 Source Term Release and Near Field Transport Conclusions

The NRC staff concludes that DOE’s Case A model appears to underestimate the release of radionuclides from saltstone because of overly optimistic assumptions about the physical and chemical properties of saltstone and disposal unit concrete. The NRC staff found the assumptions related to near-field flow in Cases K, K1, and K2 to be appropriate. However, the NRC staff concludes the average-K_d model DOE used to model Tc release in Cases K, K1, and K2 appears to delay and overestimate the peak release rate in a manner that makes it difficult to interpret the results of these cases. Most significantly, the NRC staff finds the K_d values DOE uses to represent Tc sorption in reduced cementitious materials, and the K_d DOE uses to represent Tc sorption in oxidized saltstone in Cases K and K2 to be unsupported. Of Cases K, K1, and K2, the NRC staff finds that Case K1 has K_d values for Tc for cementitious materials that are the most supported. Thus, because Cases K, K1, and K2 differ only in the selected K_d values for Tc in saltstone and the disposal units, the NRC staff considers Case K1 to best represent the system.

With respect to flow modeling, the NRC’s analysis of DOE’s near-field model indicates that Case A appears to severely constrain flow through saltstone (Figures 2.7-1 and 2.7-2). As discussed in detail in Section 2.6, the saltstone hydraulic conductivity that contributes to this low flow value is inconsistent with recent measurements and does not reflect any degradation of the hydraulic conductivity as the saltstone ages. Flow through the saltstone and disposal unit concrete in the PA Cases (i.e., cases other than Cases K, K1, and K2) were further limited by moisture characteristic curves that are inconsistent with curves found in the literature for similar materials. DOE demonstrated that the effect of the suspect moisture characteristic curves is limited to a factor of 2 in Case A (Section 2.7.3), however, the effect may be greater if other engineered barriers do not perform as modeled. Because of the large uncertainty in the potential degradation of saltstone, disposal unit concrete, and the HDPE/GCL layers on the FDCs, and because of the importance of these barriers to system performance (Section 2.13), the NRC staff will monitor the development of model support for the long-term performance of these materials.

The NRC staff found the pore-volume model DOE used for elements other than Tc to be appropriate for less redox and pH sensitive elements. However, as described in Section 2.6, the NRC staff found the basis for the number of pore volumes required to cause certain E_h and pH transitions in saltstone and disposal unit concrete in the pore-volume model to be insufficient in the PA cases. The NRC staff found that the basis for the number of pore volumes assumed in Cases K, K1, and K2 to be reasonable. The NRC staff found the explicit shrinking core model DOE used to model Tc release from saltstone in PA cases (i.e., all cases but Case K, K1, and K2) to be appropriate for redox sensitive elements. However, as discussed below, the NRC staff finds the K_d value used for Tc sorption in reducing conditions to be unsupported and concludes that the average-K_d model DOE used to represent Tc release in Cases K, K1, and K2 is inconsistent with DOE's conceptual model of Tc release. Furthermore, the NRC staff finds that the average-K_d model overestimates peak release rates to different extents with different parameter values, in a manner that makes it difficult to interpret the model results. That is, the extent to which the peak release rate is overestimated is not consistent between different cases or readily predictable.

The NRC staff finds the K_d value used to represent Tc sorption in reduced saltstone and disposal unit concrete to be unsupported. The assumed K_d values are based on experiments that included a strong reducing agent that is not consistent with the current saltstone formulation or expected field conditions. Moreover, the value is inconsistent with DOE's conclusion that only trace levels of oxygen (i.e., 30 to 60 ppm) are required to maintain Tc in its oxidized and mobile form. DOE has not provided a basis for assuming that these trace levels of oxygen will not be present in the unsaturated soils surrounding saltstone after closure. Furthermore, DOE has not shown that the potential oxidation of saltstone during operations (when the saltstone is exposed to atmospheric oxygen) is negligible. For these reasons, and because of the importance of the K_d value of Tc in reducing cementitious materials to SDF performance (Section 2.13), the chemical reduction and sorption of Tc in saltstone is a key monitoring factor. Specifically, the NRC staff will monitor the development of model support for the assumption that saltstone will chemically reduce Tc(VII) to Tc(IV) to the extent assumed in DOE's PA model under the range of conditions to which saltstone is expected to be subjected during the compliance period.

Similarly, as described in Section 2.7.4.2, the NRC staff finds that the K_d values for cementitious materials were not supported for Ra and Se. Additionally, the NRC staff finds that the K_d value assumed for Sr for saltstone in Cases K, K1, and K2 were not supported. The NRC staff also found that the K_d value for Se for sand and clay soils was not supported. Because of the potential risk significance of these radionuclides, NRC staff will monitor the development of additional model support for the saltstone K_d value for Ra, Se, and Sr, the disposal unit concrete K_d value for Ra and Se, and the sand and clay K_d values for Se.

2.8 Hydrology and Transport

2.8.1 Description of Site Hydrology

The SDF hydrological system and far-field modeling approach and results are discussed in several PA sections (Section 3.1.5 "Hydrogeology", Section 4.2.3.1.3 "Transport Model—Saturated Zone", Section 4.2.3.2.5 "Saturated Zone Hydraulic Properties", Section 4.3.1.2, "PORFLOW", Section 4.4.4.1 "PORFLOW™ Modeling Process", and Section 5.2.1, "Groundwater Concentrations at 100 m"). Data used to define and parameterize the three-dimensional groundwater flow and contaminant transport models for the General Separations Area (GSA) are found in the GSA database. This database includes elevations for tops of hydrostratigraphic units, sediment core descriptions, water levels in the upper and lower zones of the Upper Three Runs (UTR) aquifer (or UTR-UZ and UTR-LZ), water levels in the Gordon aquifer, and permeability data from laboratory tests, multiple and single well pump tests, and slug tests. The location of the SDF on the GSA (Z-Area) and nearby surface water locations are depicted in Figure 2.8-1.

In the area of the SDF, the hydrological system consists of three aquifers of interest: (1) the water table or UTR aquifer, which is split into upper and lower zones, (2) the Gordon aquifer, and (3) the Crouch Branch aquifer. The UTR and Gordon aquifers are expected to be impacted by radionuclides from the SDF facility. Contamination is not expected to affect the deeper Crouch Branch aquifer, however, because of an upward flow gradient between the Crouch Branch and Gordon aquifers near Upper Three Runs creek. Groundwater flow in the UTR aquifer is driven by recharge. In the vicinity of the SDF, groundwater from the UTR aquifer discharges or seeps to McQueen Branch to the east or Upper Three Runs creek to the north and west. The underlying Gordon aquifer is strongly influenced by its discharge to Upper Three Runs creek. The Gordon aquifer is recharged by downward leakage from the UTR aquifer above and upward leakage from the Crouch Branch aquifer below. Conceptually, recharge of the Gordon aquifer by the Crouch Branch aquifer is very small in comparison to recharge from the Upper Three Runs aquifer and is thus neglected in modeling. DOE's conceptual model for flow in affected aquifers on the GSA is presented in Figure 2.8-2. The northwest half of the SDF straddles a groundwater divide between Upper Three Runs creek and McQueen Branch causing SDF contaminants to discharge to either creek depending on the SDF source location (2009 PA, pg 172).

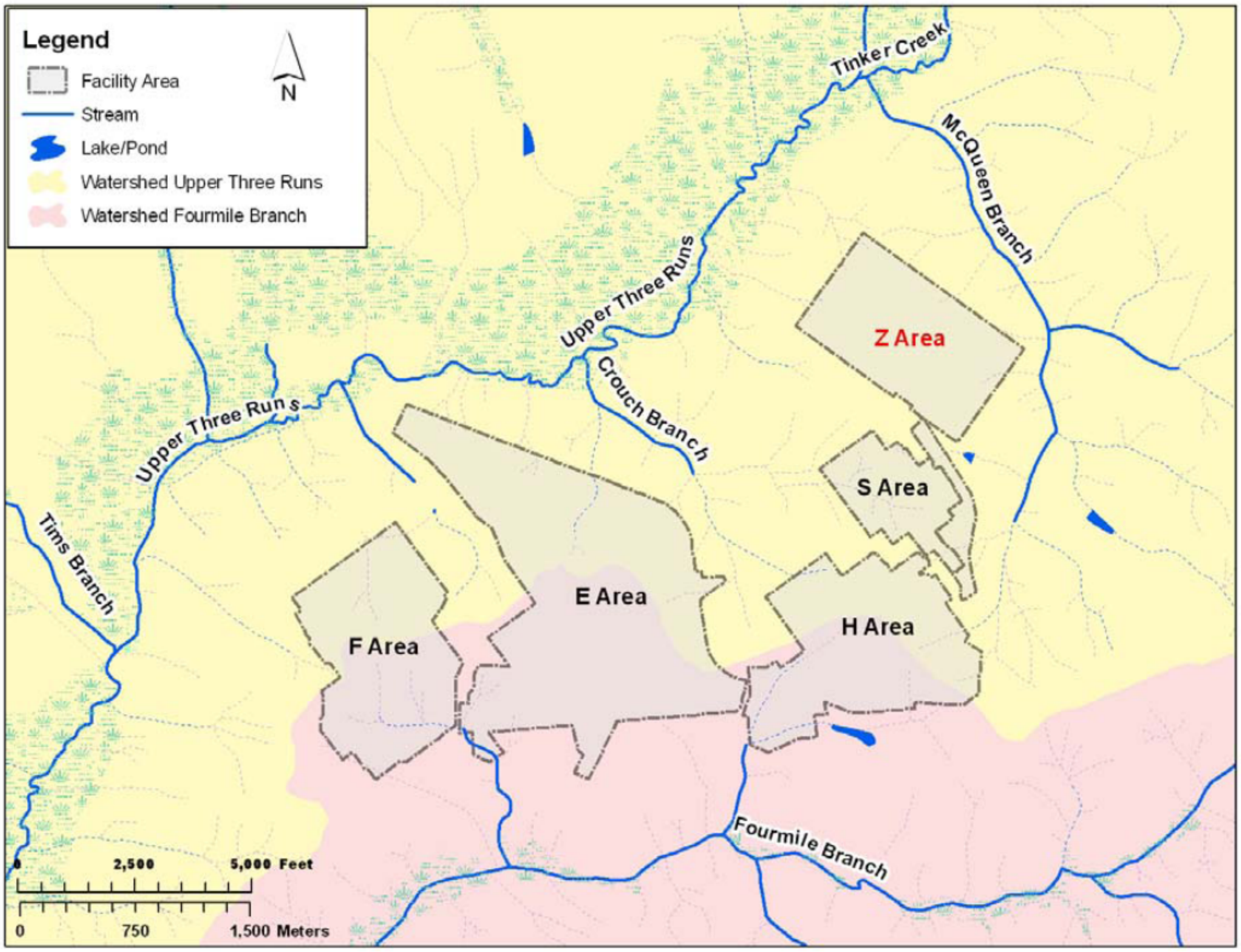

Figure 2.8-1: Location of Z-Area within the GSA and Surface Water Locations (2009 PA, Figure 4.2-1)

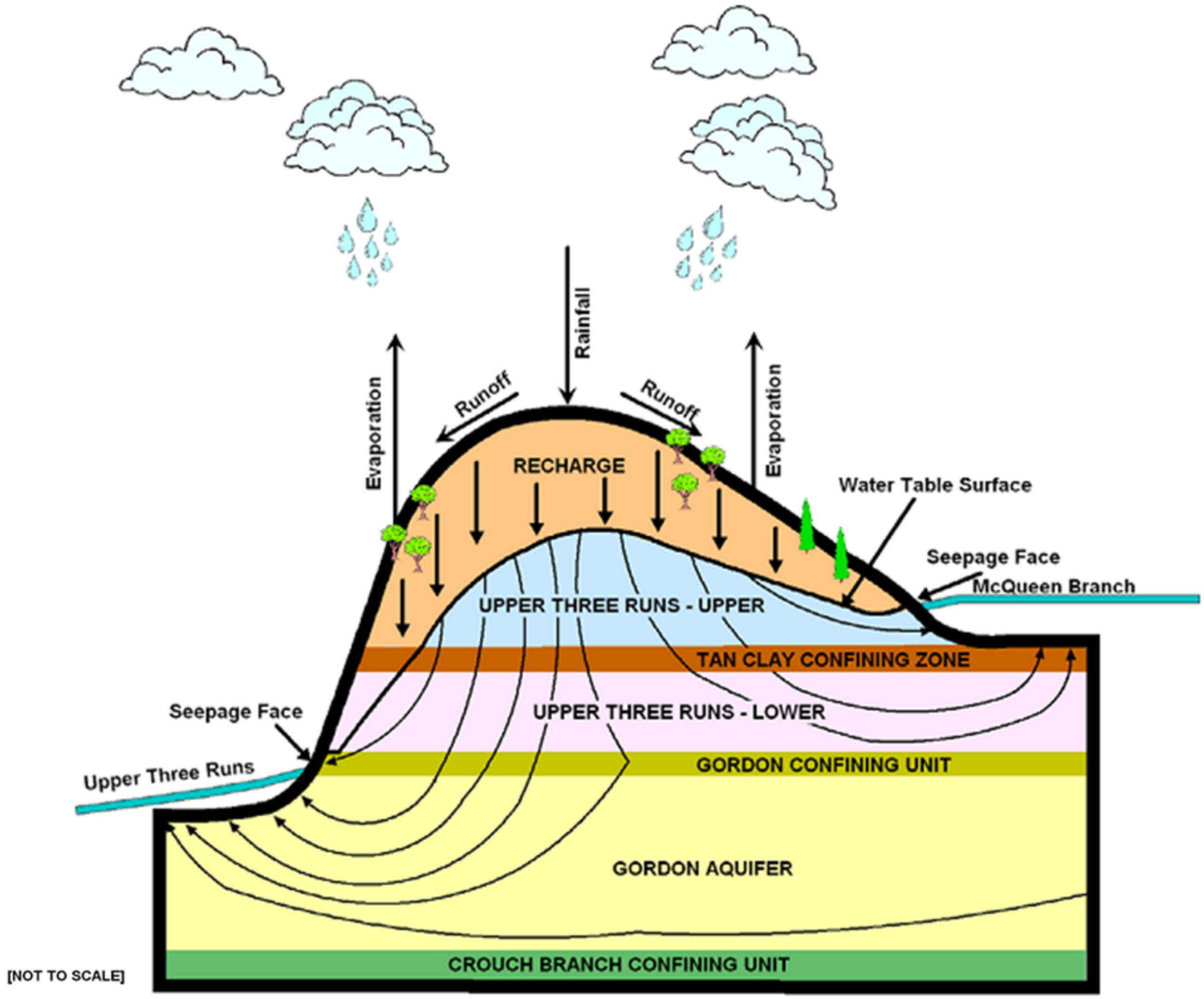

Figure 2.8-2: Conceptual Model for Flow on the GSA near SDF. (2009 PA, Figure 3.1-24 [Adapted from WSRC-TR-96-0399 Rev. 1, Vol. 2, Figure 13])

2.8.2 Modeling of Hydrology and Far-Field Transport

2.8.2.1 Regional GSA/PORFLOW™ Model

DOE previously constructed a regional GSA groundwater flow model to provide a common framework to perform various hydrogeological investigations for the GSA as documented in WSRC-TR-96-0399, Volumes 1 and 2 using the FACT computer code. The regional GSA flow model was migrated to PORFLOW™ version 5.95.0 in 2004 for use in various GSA PAs as documented in WSRC-TR-2004-00106. The PORFLOW™ version of the GSA regional flow model is hereinafter referred to as the GSA/PORFLOW™ model. Figure 2.8-3 shows a perspective view of the regional GSA/PORFLOW™ model. The GSA/PORFLOW™ saturated zone flow and transport models are used to simulate flow and contaminant transport of constituents released from the SDF to surface water. Seepline concentrations are extracted from the GSA/PORFLOW™ model for use in calculating surface water pathway doses in the PA.

The interior areal resolution of the GSA/PORFLOW™ model is 61x61 m^2 (200x200 ft^2); peripheral grid cells at the margin of the model domain are larger. A maximum of 108-grid cells span the domain from east to west, and a maximum of 77 grid cells span the domain from north to south. Vertical resolution varies with hydrostratigraphic picks and topography. The UTR-UZ aquifer is represented by up to 10 cells in the vertical direction, and the vadose zone is considered to be part of this unit. The tan clay-confining zone (TCCZ) is represented by 2 cells in the vertical direction, and is assumed to be laterally continuous. The UTR-LZ aquifer is represented by 5 cells in the vertical direction. The underlying Gordon Confining Unit (GCU) and Gordon aquifer are each represented by 2 cells in the vertical direction. Thus, a maximum of 21 grid cells represent the GSA hydrogeology from ground surface to the base of the Gordon aquifer. The regional model domain comprises 102,294 cells (2009 PA).

The hydrostratigraphy of the GSA/FACT and GSA/PORFLOW™ models are similar with some notable exceptions (WSRC-TR-2004-00106). Adjustments to the GSA/FACT model mesh were necessary to overcome limitations of the PORFLOW™ model in accurately representing the velocity field for highly distorted elements present in the FACT model or to accommodate specific project needs. Most notably, GSA/PORFLOW™ model layers are assumed to be truncated by the ground surface to prevent excessively thin layers that are present in the GSA/FACT model. Additionally, GSA/PORFLOW™ model layers above the TCCZ are non-uniformly distributed compared to their counterparts in the GSA/FACT model. Thinner layers just above the TCCZ were desirable for E-Area modeling where the water table is located just above the TCCZ. The layering below the TCCZ is essentially the same between the two models.

2.8.2.2 Local SDF/PORFLOW™ Model

The primary focus of the local SDF/PORFLOW™ saturated zone transport model is contaminant concentrations at locations 1 m (3 ft) and 100 m (330 ft) from the SDF site boundary. These concentrations are used for calculating groundwater pathway doses for the intruder (§61.42) and members of the public (§61.41), respectively. The areal resolution of the local SDF model is 15×15 m^2 (50×50 ft^2). In the horizontal plane, each GSA/PORFLOW™ regional model grid cell is divided four ways in each coordinate direction into 16 local SDF transport model grid cells; vertical resolution, however, is preserved. This constitutes a 4×4×1 mesh refinement.

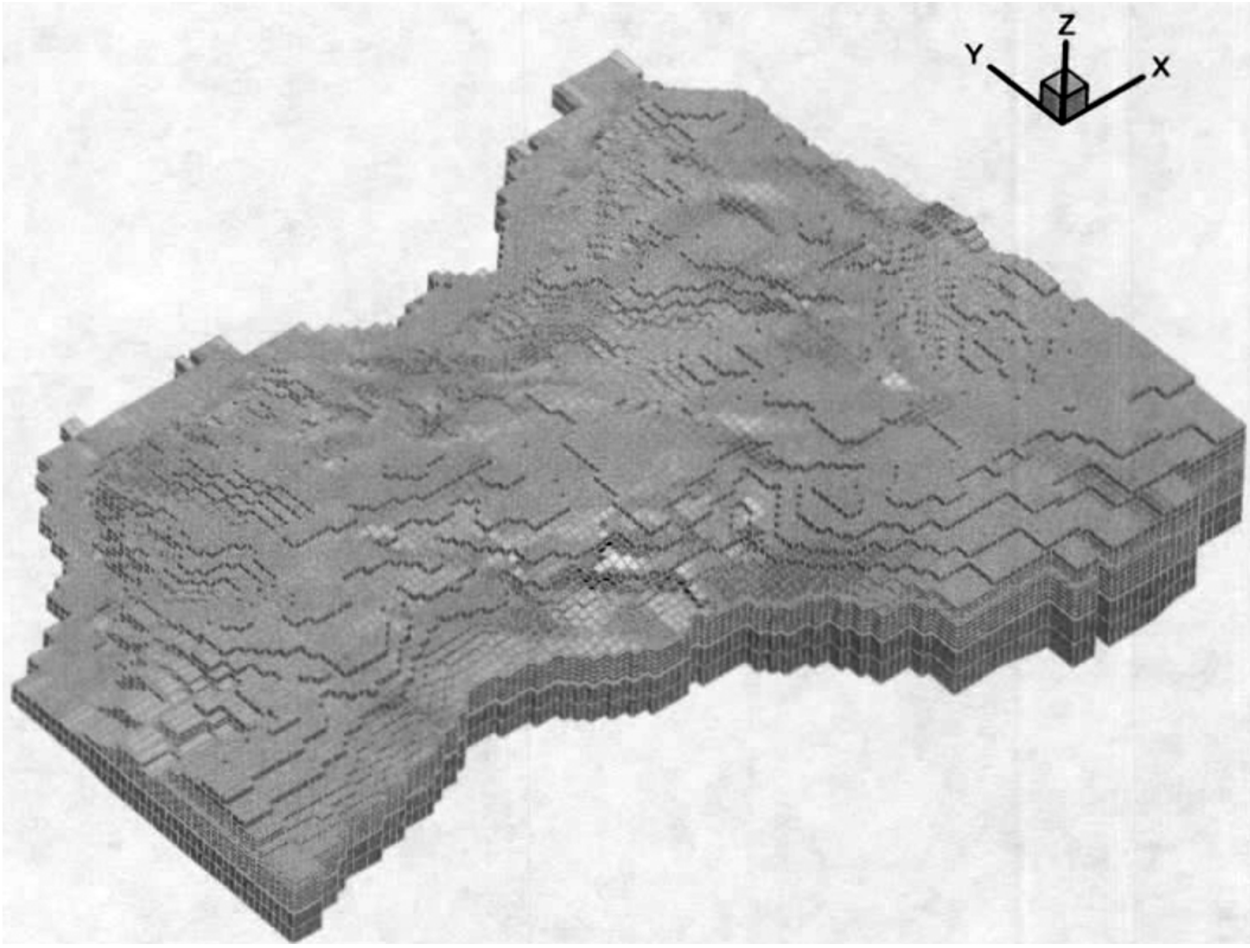

Figure 2.8-3: Perspective View of Regional GSA/PORFLOW™ Flow and Transport Model Domain. (2009 PA, Figure 4.2-9 [taken from WSRC-TR-2004-00106, Figure 2-3])

The velocity field for the local SDF/PORFLOW™ model is generated with a mass-conserving linear interpolation scheme directly from the regional GSA/PORFLOW™ velocity model; thus, the local SDF/PORFLOW™ model does not require a separate flow model with its own boundary conditions and properties. Within the lateral confines of the local SDF/PORFLOW™ model, the velocity field includes the complete vertical extent of the regional GSA/PORFLOW™ model. Figure 2.8-4 illustrates the 100-m point-of-compliance used to evaluate the §61.41 performance objectives, as well as particle tracks to provide information on groundwater flow directions and rates for conservative species released at various SDF sources.

Red Streamlines Represent Each Plume Centerline. A Series of Five-Year Time Markers is Noted with Blue Dots on the Red Streamlines. Alternating Blue and Green Dots Mark 100-m Compliance Boundaries to Indicate Evaluation Sectors [Results are Binned by Sector].

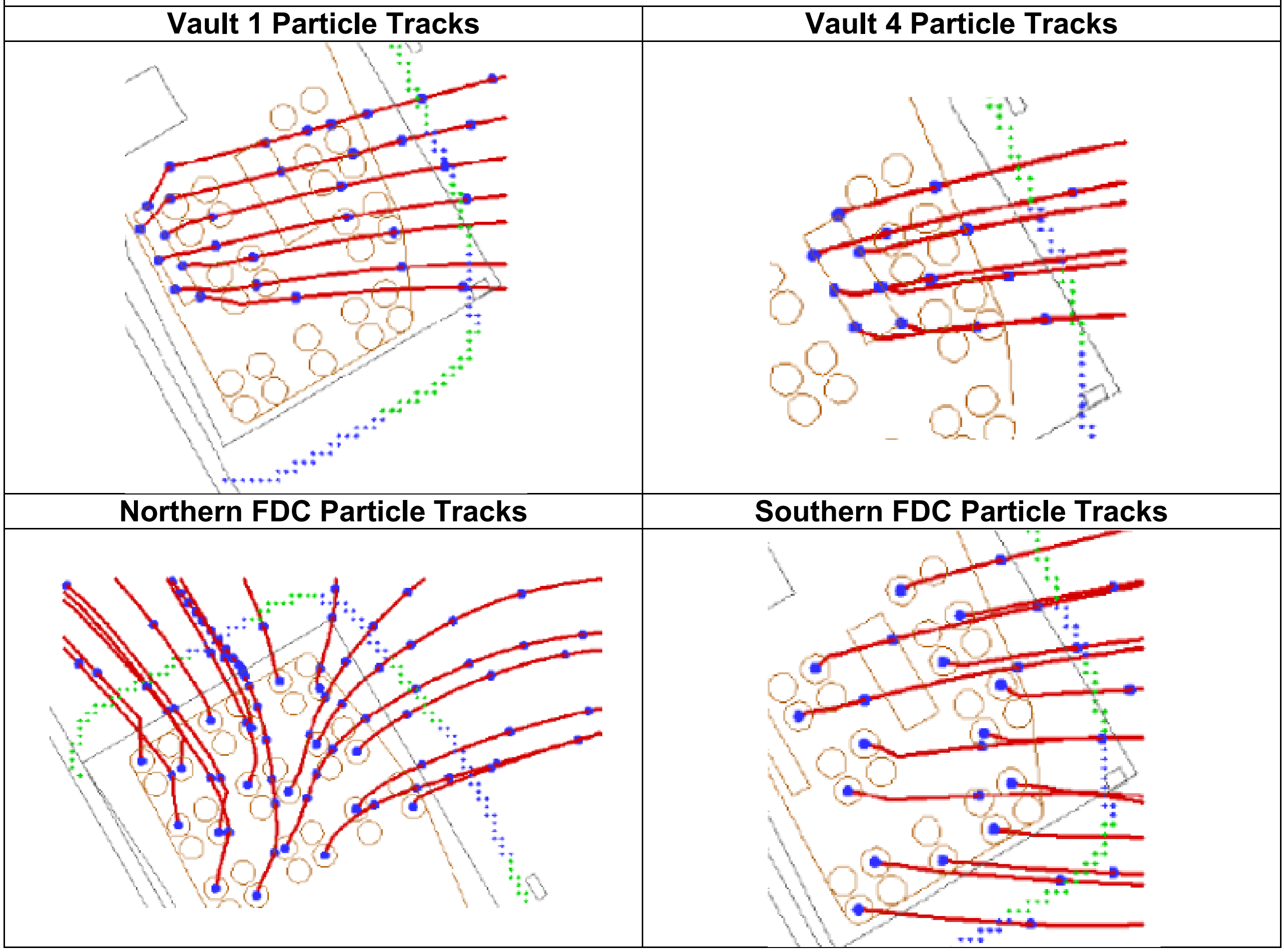

Figure 2.8-4: SDF/PORFLOW™ Saturated Zone Model with 100-m (330-ft) Compliance Evaluation Sectors (2009 PA, Figures 5.2-2 though 5.2-4)

2.8.2.3 Boundary Conditions

The GSA/PORFLOW™ saturated zone model domain encompasses the GSA and its surface water discharge points. Streams define three lateral domain boundaries: Upper Three Runs forms the northern boundary; McQueen Branch forms the eastern boundary; and Four Mile Branch forms the southern boundary (Figure 2.8-1). Four Mile Branch and McQueen Branch provide natural, no-flow boundary conditions for the Upper Three Runs aquifer (Figure 2.8-5). This aquifer unit is absent at the northern model boundary due to Upper Three Runs stream incision. The western boundary is arbitrarily chosen where hydraulic head values from a contour map of measured water elevations are prescribed. The Gordon aquifer only discharges to Upper Three Runs creek; Upper Three Runs creek provides a natural no-flow boundary for the Gordon aquifer on the north face of the model. Gordon aquifer hydraulic head values from a contour map are prescribed over the west, south and east faces of the model. The grid cells that represent the base of the Gordon aquifer are prescribed a general head boundary condition; upwelling recharge from the Crouch Branch aquifer below is considered negligible. The upper surface of the PORFLOW™ saturated zone model is prescribed a spatially variable recharge/drain boundary condition (implemented using a prescribed flux boundary with the flux being a function of pressure head) with an average recharge rate of 37.3 cm/yr (14.7 in/yr) (WSRC-TR-2004-00106, pg 14 and Figure 3-7) representative of non-capped GSA sediments.

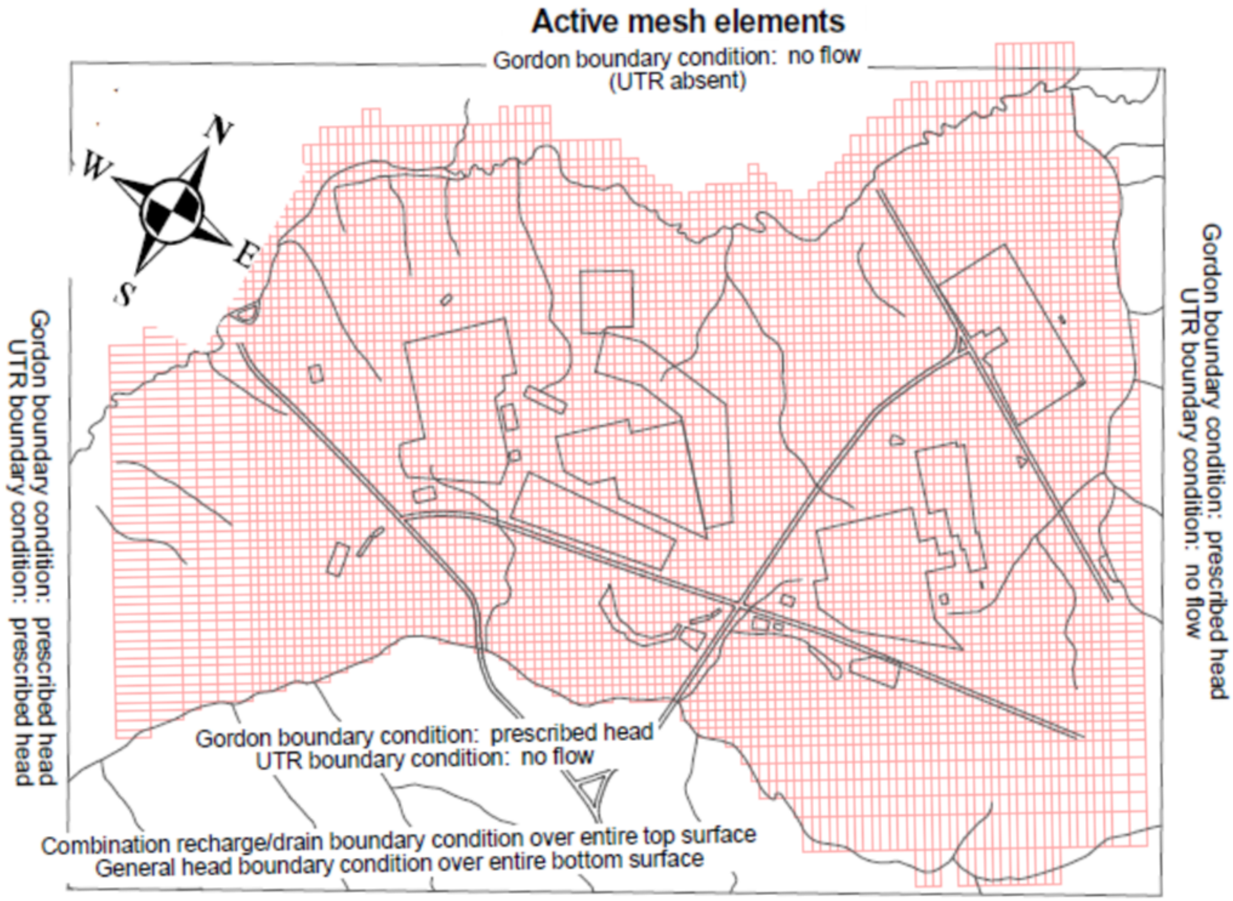

Figure 2.8-5: GSA/PORFLOW™ Model Boundary Conditions

Contaminant fluxes that exit the individual vadose zone transport models (i.e., one each for disposal cell) become an upper boundary condition for the local SDF saturated zone model with fluxes from the representative Vault 2 (or FDC) vadose zone model multiplied by 64 and loaded homogeneously under the footprints of the 64 individual FDCs. Each water table flux contribution from an individual SDF disposal unit is assigned to the aquifer transport grid by uniformly distributing the flux to those water table cells with centroids lying within the footprint of the disposal unit. Inspection of PORFLOW™ modeling files reveals that the fewest and largest number of elements representing the footprint of an individual FDC is four and nine, respectively, with the overwhelming majority of FDCs represented by seven elements.

2.8.2.4 Material Properties and Parameters

The regional GSA/PORFLOW™ model used for the PA generally inherited its parameter set from that of the predecessor GSA/FACT model. The GSA/FACT parameter set was derived from the GSA database of hydrologic data, which includes hydraulic conductivity data from laboratory tests, pump tests (multiple and single well tests), and slug tests. The same approach used in the GSA/FACT modeling to assign initial hydraulic conductivity fields was also used during GSA/PORFLOW™ model construction.

The initial (uncalibrated) hydraulic conductivity assignments to GSA/FACT model grid cells were based upon a correlation between hydraulic conductivity and total mud fraction (both calcareous and siliciclastic), where mud is the summation of silt and clay fractions (corrected Figures CC-FF-5.1 and 5.2 provided in the FTF RAI responses [SRR-SWDA-2011-00054]). Apparently, for both GSA/FACT and GSA/PORFLOW, known core lithologies (e.g., mud fraction) from discrete intervals were correlated with known field- or laboratory-measured hydraulic conductivities from the same intervals, although DOE does not clearly state this in the FTF RAI responses (SRR-SWDA-2011-00054). There were slight differences in the initial definition of GSA/FACT and GSA/PORFLOW™ model hydraulic conductivity fields due to differences in the model meshes and in the manner in which property assignments are made (i.e., at element nodes in FACT versus on element faces in PORFLOW), as described in WSRC-TR-2004-00106 (pg 5). The initial GSA/PORFLOW™ hydraulic conductivity field was subsequently modified during model calibration. Modifications made to the hydraulic conductivity field during GSA/PORFLOW™ model calibration were different than those made during GSA/FACT model calibration, leading to differences between the two models in terms of the final calibrated hydraulic conductivities.

Given that the vadose zone is included within the regional GSA/PORFLOW™ model mesh, water retention characteristic curves are also required. Pseudo-soil water retention characteristic curves, which exhibit greater linearity than real soil water curves, are used to simulate unsaturated flow from the land surface to the water table under steady-state conditions (WSRC-TR-2004-00106; Figure 2-7). Horizontal and vertical saturated hydraulic conductivities of cells with computed saturation less than 90 percent are set to 3.5×10^{-5} cm/s (0.1 ft/d) to minimize lateral flow and thereby ensure that modeled water movement in the vadose zone is vertically downward (WSRC-TR-2004-00106, pg 5).

With respect to transport properties, effective diffusion coefficients (Table 2.8-1) are assigned to saturated zone sediments based upon whether they are defined as sandy or clayey (WSRC-STI-2006-00198, pg 128). GSA/PORFLOW™ transport modeling assumes an effective porosity value (Table 2.8-1) for all aquifers and aquitards to account for dead-end pores that do not participate in radionuclide transport (WSRC-STI-2006-00198). Because the Upper Three Runs and Gordon aquifer material properties are similar to the lower vadose zone material properties, DOE converts average values for lower vadose zone bulk density 1.62 g/cm^3 and particle density 2.66 g/cm^3 to effective values for saturated zone bulk density (1.04 g/cm^3) and particle density (1.39 g/cm^3) using the average total value (39 percent) and effective value (25 percent) of lower vadose zone porosity (2009 PA, page 218; WSRC-STI-2006-00198, pg 128). DOE also assumes these same effective material property values for the TCCZ and GCU.

Table 2.8-1: Saturated Zone Material Properties DOE used in PORFLOW™ Modeling

Saturated Sediments	Saturated Effective Diffusion Coefficient (cm^2/s)	Effective Porosity (%)	Effective Dry Bulk Density (g/cm^3)	Effective Particle Density (g/cm^3)
Sandy Clayey	$5.3x10^{-6}$ $4.0x10^{-6}$	25	1.04	1.39

K_d values used for radionuclide transport through the sandy saturated zone are the same as values assigned in the PORFLOW™ vadose zone modeling for undisturbed vadose zone material. The entire UTR aquifer [i.e., the UTR-UZ, UTR-LZ, and tan clay confining unit] is assigned sandy K_d values. K_d values used for radionuclide transport through the clayey saturated zone are the same as values assigned in the PORFLOW™ vadose zone modeling for backfill. Clayey K_d values are only assigned to the GCU (SRNL-STI-2009-00115, pages 129 - 130). K_d values are listed in Table 4.2-15 of the PA.

DOE indicates that hydrodynamic dispersion in the local SDF/PORFLOW™ transport model is represented by a (i) longitudinal dispersivity of 10 m (33 ft), (ii) transverse horizontal dispersivity of 1 m (3 ft), and (iii) an apparent transverse vertical dispersion of 0 m (0 ft) that are 10, 1, and 0 percent of a nominal 100 m (330 ft) plume travel distance, respectively. Although DOE indicates vertical dispersivity is set to 0 m, DOE also indicates that some numerical dispersion occurs, nonetheless (SRNL-STI-2009-00115, pg 130). It is significant to note that PORFLOW™ version 6.10.3 (G-TR-G-00002) used for simulation of all cases in the PA only allows specification of a single longitudinal and transverse dispersivity, while PORFLOW™ version 6.30.2 used to simulate Case K in the RAI responses is able to provide separate specification of longitudinal and transverse dispersivities in the horizontal and vertical direction. However, the dispersivities for longitudinal and transverse vertical for Case K are set at values of 1 and 0.1 m (3.28 and 0.328 ft), respectively. Therefore, while it appears longitudinal vertical dispersivity is lower in Case K compared to the base case (and all other cases), the transverse vertical dispersivity appears to be higher than the value DOE intended to use for the base case in the PA. Effective diffusivities for sand and clay materials in the aquifer are assigned values of $5.3x10^{-6}$ cm^2/s (2009 PA; Table 4.4-10).

2.8.2.5 Model Verification, Calibration, and Validation

PORFLOW™ acceptance testing on version 5.95.0 used to construct the GSA/PORFLOW™ model confirmed that the code conserves mass and satisfies Darcy's Law (WSRC-TR-2004-00106, pg 21). DOE indicates in its PA that software quality assurance for the version of PORFLOW™ used for the local SDF/PORFLOW™ calculations (version 6.10.3) is covered by WSRC-SQP-A-00028 and G-TR-G-00002.

The initial hydraulic conductivity fields of the GSA/PORFLOW™ model were modified to the final calibrated parameter set (e.g., Figure 2.8-6 and Figure 2.8-7) by matching model results with measured hydraulic heads (WSRC-TR-2004-00106, pg 7). Particle tracking simulations were also performed to compare groundwater travel times for the GSA/FACT and GSA/PORFLOW™ models. The GSA/PORFLOW™ modeled travel times were generally longer; the maximum recharge rate was, therefore, increased from 0.46 to 0.48 m (18 to 19 in/yr) in the GSA/PORFLOW™ model. Relative to the GSA/FACT model, modifications to the GSA/PORFLOW™ models that were needed to obtain similar calibration results included:

- Increasing horizontal hydraulic conductivity in the UTR-UZ by 25 percent
- Decreasing vertical hydraulic conductivity in the TCCZ by 50 percent
- Increasing horizontal hydraulic conductivity in the UTR-LZ by 35 percent

The average calibrated horizontal conductivities in the saturated UTR-UZ, UTR-LZ, and Gordon Aquifer are approximately 3.0, 4.0, 11.6 m/d (10, 13, and 38 ft/d), respectively. The average calibrated vertical conductivities for the TCCZ and the Gordon Confining Unit are 2.1×10^{-6} and 3.5×10^{-9} cm/s (6.0×10^{-3} and 1.0×10^{-5} ft/d), respectively (2009 PA, pg 176).

DOE evaluated the need to update the GSA/PORFLOW™ model with more recent data (2009 PA, pg 177). Two investigations were conducted including (i) comparisons of GSA/PORFLOW-generated head contours to hand-drawn contours using data from 1995, 1998, and 2003, and (ii) comparison of head residuals using data through 1995 and considering more recent data collected through 2006. The investigations suggest that the additional data collected since construction of the original GSA/PORFLOW™ model has not significantly altered DOE’s understanding of the GSA groundwater system.

Summary statistics for hydraulic head residuals were provided in the GSA/PORFLOW™ documentation (WSRC-TR-2004-00106, Table 3-1); however, no calibration goals were specified. GSA/FACT documentation (WSRC-TR-96-0399, Vol 2 Rev. 1) indicates that calibration criteria should be no more stringent than the uncertainty in the data. Hydraulic head data was reported as having a maximum uncertainty of 1 m (3 ft) (WSRC-TR-96-0399, Vol 2 Rev. 1). Therefore, a calibration goal of 1 m (3 ft) for the root-mean-square hydraulic head residual was set for the regional GSA/FACT model. WSRC-TR-96-0399, Vol. 2 Rev. 1 also defines a calibration goal for the maximum head residual as 5 to 10 percent of the total hydraulic head variation in a given aquifer. Total hydraulic head variation within the Gordon aquifer is given as ~24 m (~80 ft). Therefore, a maximum target head residual would be no greater than 1.2 to 2.4 m (4 to 8 ft) for the Gordon aquifer. Total head variation within the UTR aquifer is given as ~49 m (~160 ft). Therefore, a maximum target head residual would be no greater than 2.4 to 4.9 m (8 to 16 ft) for the UTR aquifer. While no specific calibration goals are set for the GSA/PORFLOW™ model, the maximum residual in the UTR-LZ of 8.2 m (27 ft) near H-Area in the GSA/PORFLOW™ model greatly exceeds the calibration goal indicated in the GSA/FACT model. WSRC-TR-2004-00106 indicates that head residuals for the GSA/PORLOW model are somewhat larger than those for GSA/FACT for various reasons (e.g., use of an artificial recharge zone near H-Area and finer vertical discretization in the GSA/FACT model) and that more extensive calibration efforts would likely improve the GSA/PORFLOW™ model.

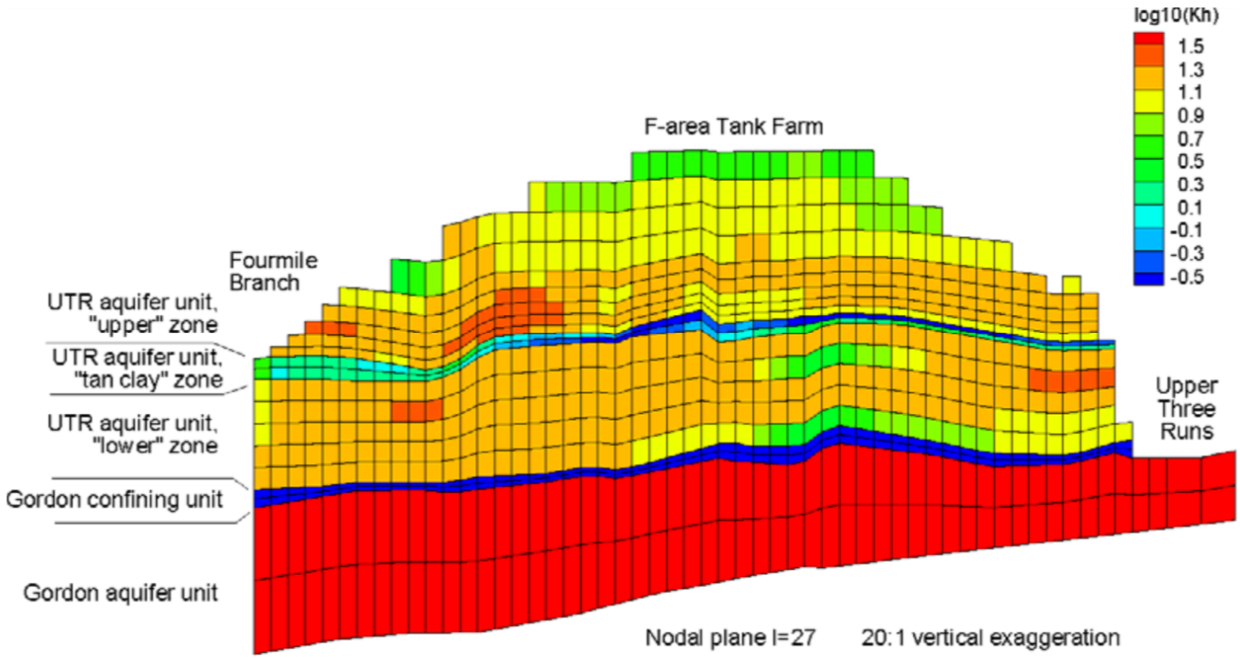

Figure 2.8-6: Calibrated Horizontal Hydraulic Conductivity Assignments to the Regional GSA/PORFLOW™ Model (Fourmile Branch is to the South and Upper Three Runs Creek is to the North on the GSA [2009 PA, Figure 4.2-10, pg 176])

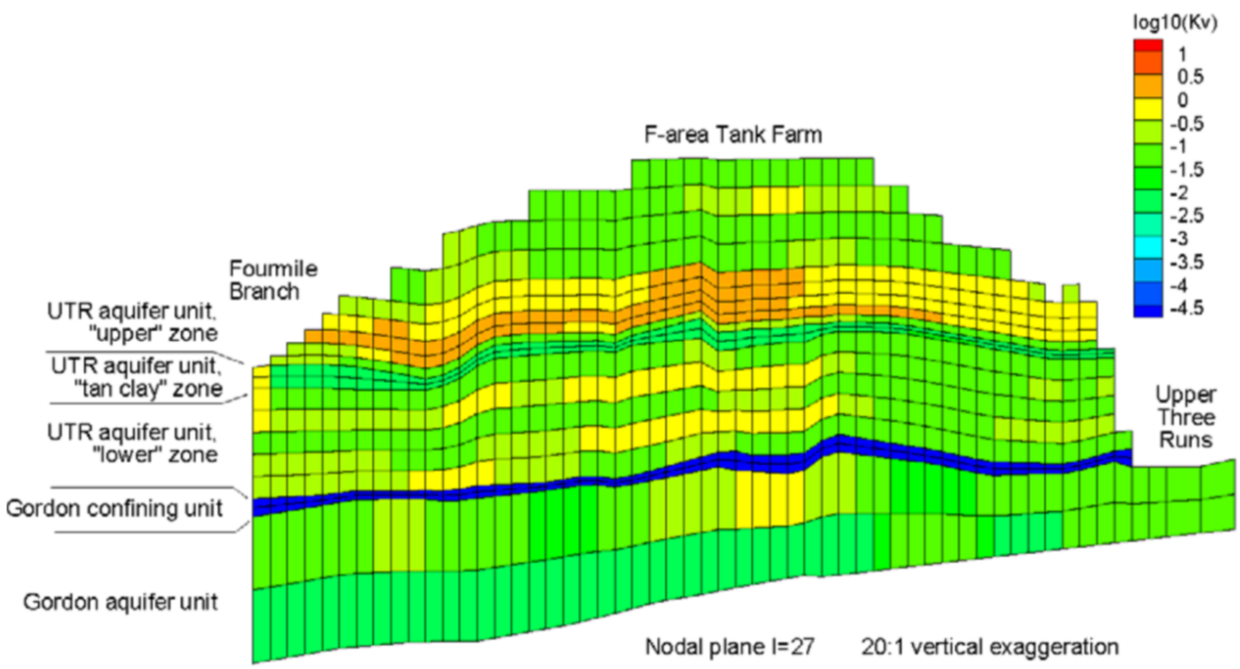

Figure 2.8-7: Calibrated Vertical Hydraulic Conductivity Assignments to the Regional GSA/PORFLOW™ Model (Fourmile Branch is to the South and Upper Three Runs Creek is to the North on the GSA [2009 PA, Figure 4.2-11, pg 177])

With regard to calibration in the area of interest, few calibration targets appear to be located in the vicinity of the SDF. Although Figure 4.2-14 in the 2009 PA provides comparisons of hand-contoured water table elevations based on monitoring data versus modeled heads (pg 180), comparisons of monitored versus modeled heads at the scale of the SDF are not provided and visual comparison of heads in the vicinity of SDF reveal significant differences (i.e., water table data reveal a greater hydraulic gradient at SDF than modeled). Text in the 2009 PA, reports a base case Darcy velocity of 45.7 m/yr (150 ft/yr) in the saturated zone (pg 495) for use in the probabilistic assessment, while the 5-year time markers (pages 274 to 276) seem to indicate variability across the SDF (e.g., relatively higher velocities are present towards the south of the SDF). For the purposes of probabilistic assessment, a normal distribution was selected with a mean Darcy velocity of 46 m/yr and standard deviation of 0.5 m/yr (pg 495) and a minimum value of 80 percent and a maximum value of 120 percent of the mean (pg 1980).

Although no specific calibration criteria related to baseflow are provided, WSRC-TR-96-0399, Vol. 2 Rev. 1 also suggests that agreement between measured and modeled baseflow is acceptable within the uncertainty range of the data or ±20 to 50 percent. GSA/PORFLOW™ simulated baseflow values are provided in WSRC-TR-2004-00106 (Table 3-2, pg 15) and are reproduced in Table 2.8-2 below. The simulated values appear to be in good agreement given the level of uncertainty in the data. McQueen Branch baseflow values are slightly higher than the upper bound target of 150 percent or ~4.2 m^3/s (2.25 ft^3/s). GSA/FACT and GSA/PORFLOW™ stream baseflows are similar with the largest differences occurring for Upper Three Runs creek and McQueen Branch (WSRC-TR-2004-00106).

Table 2.8-2: Comparison of Estimated and GSA/PORFLOW-Modeled Baseflows

Stream	Estimated Baseflow m^3/s (ft^3/s)	Modeled Baseflow m^3/s (ft^3/s)
Upper Three Runs & Tributaries	0.52 (18.2)	0.32 (11.4)
Fourmile Branch	0.07 (2.6)	0.11 (3.8)
McQueen Branch	0.04 (1.5)	0.07 (2.4)
Crouch Branch	0.05 (1.8)	0.05 (1.7)

WSRC-TR-2004-00106

WSRC-TR-2004-00106 (pg 25) provides additional benchmarking information, including comparisons of results of particle tracking simulations and tritium transport simulations performed for E-Area (WSRC-TR-2003-00432) with the two models. Peak concentrations between the models are similar, but individual runs vary on the order of ±25 percent (WSRC-TR-2004-00106, pg 25). WSRC-TR-2004-00106, states that particle tracking and transport comparisons indicate the velocity field of the two GSA models is similar.

In the 2009 PA, DOE also compares GSA/PORFLOW™ modeled path lines with E-Area tritium plume distribution data and F-Area plume data in map view to show that the regional GSA/PORFLOW™ saturated zone model reproduces known plume trajectories. GSA/PORFLOW™ model comparisons to data in the vicinity of the SDF are not provided. Likewise, no local SDF/PORFLOW™ model comparisons to data are provided. Presumably, such data are either not available or too limited to validate the local SDF/PORFLOW™ model.

2.8.2.6 Point of Compliance

The NRC guidance found in NUREG-1854 (NRC, 2007b) indicates that after the end of the institutional control period, the receptor evaluated to demonstrate compliance with the §61.41 performance objective is assumed to be located at the point of maximum exposure located outside of the disposal site (Section 2.3.2). DOE calculates potential groundwater exposure concentrations at a point located 100 m (330 ft) from the SDF boundary using the local SDF model. However, because (i) the transport paths away from SDF sources do not occur along a straight line, (ii) the transport paths traverse up to three aquifers, and (iii) sources are located at a variety of distances from the down gradient edge of the disposal site, the actual travel distances to reach a 100 m (330 ft) location beyond the SDF site are, in fact, greater than 100 m (330 ft) for many SDF sources. Table 5.2-2 of the PA shows the approximate travel distances for constituents released from the SDF vaults and FDCs to the 100 m (330 ft) boundary (2009 PA).

As stated above, DOE assumes the groundwater concentrations 100 m (330 ft) from the SDF site are the highest concentrations in the area 100 m (330 ft) or farther from the site. DOE indicates that this assumption is supported by Figures 5.2-5 through 5.2-7 in 2009 PA (pages 389 and 390), which presents the plume (in plan view and cross-sectional maps) that would result from a continuous and conservative (no decay or sorption) tracer. Peak concentration is observed to decrease monotonically with travel distance from the source zone, as a result of hydrodynamic dispersion. DOE indicates that no physical mechanism exists to concentrate contamination beyond the source zone in the fully three-dimensional PORFLOW™ simulations. Therefore, DOE concludes the 100 m (330 ft) assessment points from the disposal site are adequate to capture the peak concentration that can occur at or beyond 100 m (330 ft).

DOE calculates 100 m (330 ft) concentrations for twelve sectors (Sectors A – L) in the local SDF model, as shown on Figure 5.2-1 of the 2009 PA. The peak concentration values for the 100 m (330 ft) results are recorded for the three aquifers of concern (i.e., UTR-UZ, UTR-LZ, and Gordon aquifers). The concentration for each aquifer represents peak concentration in any vertical computational mesh within the aquifer. The vertical thicknesses of the computational grid are on average ~2 m (6.6 ft) in the UTR-UZ, and ~3 m (12.7 ft) in the UTR-LZ. Well screen averaging was not used to determine the concentrations for dose calculations because the typical well screen length of 6.10 m (20 ft) is similar to the thickness of the computational grid but conservative with respect to being lower in vertical thickness than the average well screen length. Dividing the results into sectors was necessary to allow the large amount of concentration data from PORFLOW™ to be stored and used by the GoldSim® dose calculator model, and to allow evaluation of variability in peak concentration for different source areas of the SDF. The twelve sectors are analyzed for each radionuclide and chemical to find the maximum groundwater concentrations 100 m (330 ft) from the SDF. DOE also calculates 1-m (3-ft) concentrations for four sectors (Sector 1A – 1L) for the inadvertent intruder dose calculations, as shown in Figure 5.2-1 of the 2009 PA. Using the sectors to determine the highest groundwater concentrations causes the calculated peak doses to be higher than they actually would be, because the peak concentrations are determined for each radionuclide independent of the location within the sector.

2.8.3 NRC Evaluation – Hydrology and Far-Field Transport

2.8.3.1 NRC Evaluation of Model Construction Including Boundary Conditions

It is commendable that DOE developed the hydraulic conductivity parameter set from field and laboratory characterization data (WSRC-TR-2004-00106), rather than starting with a uniform parameter set and adding variation as needed to match measured hydraulic heads and estimated stream baseflows. However, the process of migrating the GSA/FACT to the GSA/PORFLOW™ model could be more transparent, and as discussed in more detail below, calibration in the area of interest could be improved.

With respect to source loading, DOE appears to distribute the convective flux of contaminants out of the near-field model for Vaults 1, 4, and the 64 FDCs to a number of source elements located underneath the SDF. DOE intends to flag elements located near the water table with centroids located within the footprint of individual SDF sources as source cells for the purposes of far-field modeling. With respect to FDCs, DOE appears to distribute the cumulative contributions of the 64 FDCs amongst 457 source cells. The minimum number of source cells used to model releases from any single FDC is 4 cells for FDC 20C[21] and the maximum number of source cells used to model releases from any single FDC is 9 cells for FDC 10C. If DOE were to change the manner in which disposal unit releases are loaded into the far-field model by (i) ensuring that the same number of source cells are used for each FDC, (ii) ensuring that the same amount of mass is loaded into the saturated zone underneath each individual FDC, (iii) loading contaminant mass as a recharge concentration, or (iv) otherwise ensuring that source loading occurs at the water table in the far-field model, the results of the simulations could be significantly different. Scoping simulations performed by NRC staff using Tc-99 fluxes from Case K indicate that the peak sector concentrations at the 100 m boundary could be significantly higher for some sectors. Therefore, the manner in which contaminant fluxes are loaded in the far-field model should be further evaluated by DOE to ensure that the dose estimates are not significantly under-predicted in future PA analyses.

With regard to grid discretization, DOE provided information during the FTF review to support the finite element size used in the local FTF model to ensure that the level of numerical dispersion is at an acceptable level (SRR-SWDA-2011-00054 Rev. 1). The FTF analysis is expected to be relevant to the SDF because the local FTF and SDF models have the same grid resolution. DOE also presented results in the FTF RAI responses that indicate additional grid refinement may be necessary to reduce numerical dispersion in cases of very low to no assumed physical dispersion. For example, if no physical dispersion is assumed, then the peak concentrations associated with a pulse release of a conservative tracer are shown to be a factor of approximately three to four times higher with a grid refined by a factor of two in each dimension (or a factor of 8 times more elements). Therefore, NRC staff reviewed DOE's basis for the assumed level of physical dispersion in its local SDF PORFLOW™ model in the next

[21] Disposal unit designations used in the TER (Figure 2.8-8) correspond to designations used in the PA, which may not correspond to current designations.

section to determine if numerical dispersion is at an acceptable level for the assumed level of physical dispersion.

2.8.3.2 NRC Evaluation of Material Properties and Parameters

As discussed above, DOE satisfactorily demonstrated that physical dispersion dominates numerical dispersion in the local FTF/PORFLOW™ saturated zone model and that numerical dispersion is at an acceptable level for the level of physical dispersion assumed (SRR-SWDA-2011-00054 Rev. 1, Figure RAI-FF-3.2). However, if the level of physical dispersion assumed in the modeling is not supported, a finer grid discretization may be needed to ensure numerical dispersion is at an acceptable level. The results presented in response to NRC RAI-FF-3 (SRR-SWDA-2011-00054 Rev. 1) also indicate that physical and numerical dispersion combined in the FTF/PORFLOW™ model accounts for significantly lower concentrations compared to modeling simulations where little to no physical dispersion is assumed and/or less numerical dispersion is simulated over the range of grid resolutions studied. Scoping simulations for SDF indicate that if no physical dispersion is assumed (only numerical dispersion is simulated), concentrations at the 100-meter boundary could increase by a factor of 2 for certain sectors for a relatively non-sorbing constituent. The impact of lower physical *and* numerical dispersion is unknown but is expected to be similar to the local FTF/PORFLOW™ model response (around a factor of 3 to 4).

Hydrodynamic dispersion in the various local SDF/PORFLOW™ models discussed in the PA for various scenarios (or PA Cases) is represented by a longitudinal dispersivity of 10 m (33 ft) and transverse dispersivities of 1 m (3 ft) for horizontal transverse and 0 m (0 ft) for vertical transverse[22], which are 10, 1, and 0 percent of a nominal 100 m (330 ft) plume travel distance, respectively (Table 2.8-3). Although DOE attempts to set transverse vertical dispersivity to 0 m in the PA Cases, DOE indicates that some numerical dispersion occurs, nonetheless (SRNL-STI-2009-00115, pg 130). It is significant to note that PORFLOW™ version 6.30.2 used for Case K, prepared after the PA was completed, does allow separate specification of longitudinal and transverse dispersivity in the vertical direction. For Case K, DOE specifies non-zero longitudinal and transverse vertical dispersivities of 1 m and 0.1 m for the longitudinal and transverse vertical dispersivities, respectively. Therefore, although DOE specifies its intent in the PA Cases to set transverse vertical dispersion to 0, DOE assigns a higher value of 0.1 m in Case K.

In general, the literature supports significantly lower vertical dispersivities compared to horizontal dispersivities. Gelhar (1992) indicates that very low vertical dispersivities have been observed at several sites and are typically two orders of magnitude lower than longitudinal dispersivities. Although DOE attempted to limit vertical dispersion in Case A and other PA Cases, vertical dispersion may, nonetheless, be overstated in Case A and other PA cases.

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[22] NRC staff could not confirm, prior to issuance of this TER that a PORFLOW™ command line “set w 0” noted in comment to “omit transverse vertical dispersion” for the SDF/PORFLOW™ PA Cases was executed in PORFLOW™ as intended. Based on review of the PORFLOW™ user’s manual (v. 6.12.3), the version used by NRC staff, it appears that while setting the velocity vector “w” will generally lower dispersion, some vertical dispersion will nonetheless occur.

Likewise, vertical dispersion may be overstated in Case K as a value of 1 m that is only 10 times less than the horizontal longitudinal dispersivity is used for vertical dispersivity in the longitudinal direction.

Table 2.8-3: Dispersivities Used in DOE's SDF/PORFLOW™ models

Case	Longitudinal[1]	Transverse[1]	Longitudinal (vertical)[2]	Transverse (vertical)[2]
Case A and all other PA Cases	10 m	1 m	not applicable	not applicable
Case K	10 m	1 m	1 m	0.1 m

[1] Case A (and other PA case) simulations were performed using PORFLOW™ (v 6.10.3) that only allows specification of a single longitudinal and transverse dispersivity that applies to both the horizontal and vertical directions. Case K simulations were performed using PORFLOW™ (v 6.30.2) that allows separate specification of longitudinal and transverse dispersivity in the vertical direction. Therefore, the longitudinal and transverse dispersivities specified in columns 2, 3 above apply to the horizontal direction for Case K.
[2] DOE indicates in the PA that it sets vertical transverse dispersivity to zero for Case A and all other PA cases. NRC staff review of the PORFLOW™ input files reveals that an additional command line in PORFLOW™ was added in an attempt to "omit transverse vertical dispersion", because PORFLOW™ (v 6.10.3) does not allow direct specification of longitudinal and transverse vertical dispersivity as indicated in the note above.

With regard to horizontal longitudinal dispersivity, DOE indicated during the FTF review that the specified dispersivity values selected from scale-dependent correlation data (Gelhar 1992) are supported by an optimization study for F-Area and H-Area seepage basins (WSRC-TR-2002-00291). Because the optimization study was based on (1) calibration to contaminant breakthrough data (less reliable than calibration to plume distributions), and (2) preferential flow pathways away from the seepage basins in the UTR-UZ are thought to exist that may not be reflective of flow and transport away from SDF sources, F-Area and H-Area seepage basin modeling results may not fully corroborate or be entirely relevant to field-scale dispersion at the SDF facility. Lower dispersivities [e.g., α_L = 1.50 m (5 ft), α_{Th} = 0.10 m (0.33 ft), α_{Tv} = 0.01 m (0.03 ft)] from the tritium optimization study at E-Area Old Burial Ground (WSRC-TR-96-0037), although considered less certain by DOE, may be more representative of dispersion along flow paths away from the SDF. In general, Gelhar (1992) supports the conclusion that the most reliable field data are typically associated with relatively low field-scale dispersivities; lower dispersivities would tend to lessen plume spread and result in higher doses at the point of compliance. A number of modeling studies with similar grid discretization as the SDF/PORFLOW™ model for the larger SRS site also support lower dispersivities based on calibration to three-dimensional plume data, although numerical dispersion may have been dominant in these exercises. In general, dispersivities derived from calibration to three-dimensional plume distribution data are expected to be more reliable than calibration to breakthrough curve data (Gelhar, 1992). Additionally, because NRC staff is unable to conclude that physical dispersion is adequately supported, NRC staff is unable to conclude that horizontal and vertical discretization is adequate to ensure that numerical dispersion is at an acceptable level in the SDF/PORFLOW™ model.

In summary, DOE has not provided sufficient information to support the level of physical dispersion assumed in SDF/PORFLOW™ modeling. It is significant to note that changes to the SDF/PORFLOW™ model that may be made to address other issues identified by NRC staff as discussed in the model calibration section may result in changes in hydraulic gradients and lessen the impact of potentially excessive levels of vertical dispersion in the SDF/PORFLOW™ model. Evaluation of any relevant contaminant transport data in the vicinity of SDF or tracer experiments may provide additional support for the levels of dispersion simulated in SDF/PORFLOW™ models.

NRC staff's evaluation of K_ds is discussed in the near-field section (Section 2.7.4), as the same K_ds are used for both vadose zone and saturated zone materials.

2.8.3.3 NRC Evaluation of Model Calibration

With respect to GSA/PORFLOW™ model calibration, the maximum hydraulic head residual in the UTR-LZ of 8.2 m (27 ft) (WSRC-TR-2004-00106) greatly exceeds the GSA/FACT model calibration goal of no more than 5 to 10 percent of the total hydraulic head variation in the UTR aquifer for the maximum head residual. The UTR-LZ exhibits large residuals in the vicinity of H-Area (WSRC-TR-2004-00106), but the well locations having the largest residuals are not in the vicinity of SDF.

In fact, very few calibration targets are located at the margins of the GSA/PORFLOW™ model in the vicinity of the SDF. It is not clear to what extent data collected from recent geological investigations or well construction at or near the SDF agree with the GSA/PORFLOW™ model and specific details regarding the adequacy of GSA/PORFLOW™ model calibration in the vicinity of the SDF are not provided in the 2009 PA Table 4.2-10, pg 181). PA data comparing calibration statistics over time are not particularly helpful as these statistics are biased by the large number of wells on the greater GSA that would dominate calibration statistics for the area of interest.

Comparisons of observed to modeled heads at SDF are provided in Figure 4.2-14 of the 2009 PA and generally reveal higher, modeled heads at the SDF and a greater horizontal gradient from the SDF to surface water. It is possible that lower hydraulic conductivity assignments in the SDF subsurface or higher hydraulic conductivity assignments nearer surface water might improve model calibration if the local SDF area was the focus of model calibration. Flach (WSRC-TR-2004-00106) suggests that more extensive calibration efforts would likely improve the hydraulic head residuals of the GSA/PORFLOW™ model. Nonetheless, more extensive calibration in the area of interest is desirable both for transparency and model performance. Additionally, the SDF/PORFLOW™ model does not consider capped conditions consistent with closure plans for the site. Given the apparent importance of the location of the groundwater divide and hydraulic gradient to contaminant fate and transport at SDF, it would be prudent for DOE to evaluate the impact of reduced recharge underneath the SDF due to the presence of a cover, if it has not already done so.

The impact of improved model calibration in the area of interest on PA dose predictions is not clear. However, lower vertical gradients at the SDF could lead to higher peak sector doses due to (i) less dispersion vertically, (ii) less loss of activity to the Gordon Confining Unit and Gordon aquifer, or (iii) greater plume overlap of SDF sources. Lower hydraulic conductivities in the area of interest could potentially lead to lower dilution rates of SDF plumes. The location of the SDF, on a groundwater divide, also significantly affects modeled plume overlap. While 64 FDC sources could theoretically contribute to groundwater dose at the point of maximum exposure, it appears that sources nearest the 100 m boundary contribute more significantly to peak sector doses at the SDF and that most SDF sources do not, in fact, overlap. Rather, SDF sources appear to spread out almost in a radial fashion, thereby minimizing plume overlap laterally compared to a situation where flow is primarily in the same direction. In conclusion, slight changes to the location of the groundwater divide or hydraulic gradients could lead to greater cumulative impacts and/or less aquifer dilution than assumed in the 2009 PA. Improved calibration in the area of interest would serve to provide additional support for the assumed level of dilution/dispersion of SDF sources.

2.8.3.4 NRC Evaluation of Data and Model Uncertainty

NRC staff attempted to evaluate the contributions of various sources to the peak dose in Case K. Although NRC staff used an earlier version of PORFLOW™ (v 6.12.3) that is not able to specify vertical dispersivities separately and the sources simulated may not contribute to the exact point in the UTR aquifer that the peak Case K dose in DOE's SDF/PORFLOW™ model occurs from all sources, the results should provide a relative indication of individual source (or source group) contributions to dose at the 100 m boundary. Table 2.8-4 presents the results of this evaluation. Figure 2.8-8 illustrates the source locations simulated at the SDF.

As stated above, many factors contribute to what appears to be an optimal configuration of sources at the SDF (i.e., minimal overlap of and cumulative impacts from SDF sources). A shift in the location of the groundwater divide could lead to variations in cumulative impacts from multiple SDF sources. A lower hydraulic conductivity (to better match the horizontal gradient) may lead to less dilution of SDF source plumes, or a lower vertical hydraulic gradient could lead to less vertical dispersion, greater plume overlap in the vertical direction, or less loss of mass to the GCU. Therefore, an adequate set of calibration targets and acceptable calibration statistics in the area of interest are needed to provide additional confidence that the level of dilution realized in DOE's SDF PA model is not over-stated.

NRC staff is also concerned with the potential impact of calcareous zones on contaminant flow and transport at the SDF. Many SDF sources traverse the lower zone of the UTR aquifer where calcareous materials are known to be more pervasive in the subsurface at SRS. The GSA/PORFLOW™ and SDF/PORFLOW™ models do not explicitly account for the impact of these zones on contaminant flow and transport (other than considering the impact of mud fraction on hydraulic conductivity). During the SDF review, NRC staff expressed concerns regarding the potential impact of these zones on contaminant flow and transport. Thayer, et al. (1995) indicate that contaminant data for the Burial Ground Complex (located in E-Area at the GSA) and the Chemical, Metals, and Pesticide Pits (located off the GSA) support the hypothesis that contaminants may be preferentially transported within these zones. These contaminant data were not provided nor assessed during this evaluation. Furthermore, sorption of key radionuclides may be overstated in the SDF PA for calcareous materials as no site-specific K_ds have been developed specifically for this zone and material in the UTR aquifer. However, because Tc and I are expected to be the key risk drivers in Case K and little to no credit is taken for sorption for these radionuclides, the need for additional site-specific information in this area may be limited.

Table 2.8-4: Ratio of All Sources to Individual Source Peak Doses

Case	Source[1]	Normalized Concentration[2]	Location of Maximum Concentration	Preparer
Case K	All	1	Sector I	DOE
Case K	All	0.83 (1)	Sector I	NRC[3]
Case K	7A, B, C, D	0.58 (0.70)	Sector I	NRC[3]
Case K	7A	0.2 (0.24)	Sector I	NRC[3]
Case K	2AB	0.2 (0.24) (0.34) [(4)]	Sector L[5]	NRC[3]

[1] Figure 2.8-8for location of sources from the 2009 PA. DOE has recently renumbered the FDCs.
[2] Normalized to the maximum concentration in Sector I from DOE Case K. Values in parentheses are normalized to the maximum concentration in Sector I from a similar NRC staff PORFLOW™ simulation of Case K using an older version of PORFLOW. The results of the NRC simulation are 83 percent of the peak concentrations from DOE's Case K simulation.
[3] NRC simulations are performed with PORFLOW™ version 6.12.3. PORFLOW™ (v 6.12.3) does not allow separate specification of vertical dispersivities; therefore, dispersion is expected to be overestimated in the NRC simulations (concentrations are lower).
[4] The centerline of the plumes emanating from Vaults 2A and 2B traverses compliance evaluation Sector L. The peak concentration from Vaults 2A and 2B is 34 percent of the peak concentration in Sector L from all sources.
[5] Sector L peak concentration is 57 percent of the peak concentration in Sector I.

No formal mapping to identify calcareous zone seeps along stream valleys has been conducted. Nonetheless, DOE indicated in an FTF RAI resolution meeting (NRC, 2011d) that a field mapping activity such as this could be incorporated in the future. NRC staff support such an activity to evaluate the impact on the hydrogeologic system of the SDF facility of potentially highly porous and conductive soft zones in the UTR-LZ. DOE progress in this area will be evaluated during monitoring. If calcareous zone seeps are identified, tracer studies in the SDF UTR-LZ using innocuous tracers that are commonly used to understand preferential flow and transport could be conducted to better understand the effect of these zones on contaminant flow and transport. Results from any such tracer studies would be evaluated by NRC during the monitoring period.

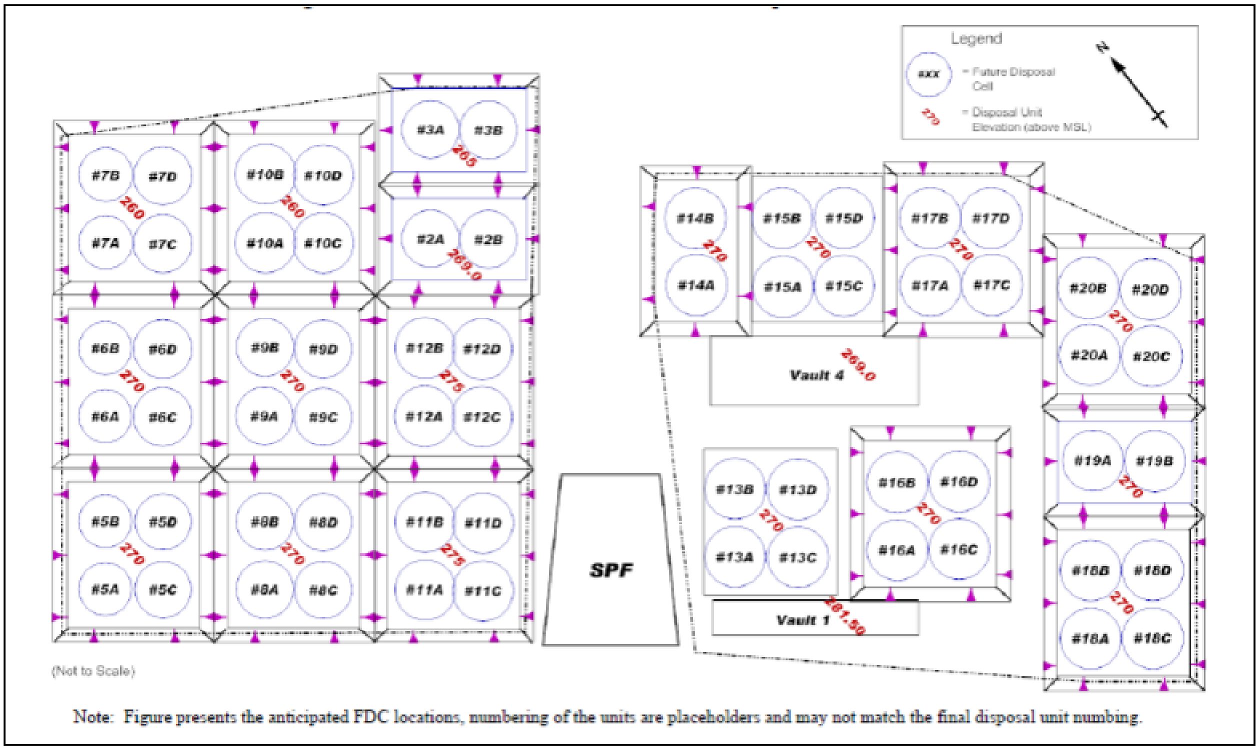

Figure 2.8-8: Locations of Sources at the Saltstone Disposal Facility (2009 PA, Figure 4.2-20)

2.8.3.5 Far-field Conclusions

NRC staff concludes that DOE's far-field model presents an acceptable framework to facilitate decision-making regarding SDF operations. However, as described in Section 2.8.3, the NRC staff also concludes that modeled far-field performance results in potentially overly optimistic levels of dilution and dispersion of SDF source plumes and limited cumulative impacts from the facility. Changes to key parameters (e.g., hydraulic conductivities, recharge rates reflective of capped conditions) and improved model calibration in the area of interest could lead to (i) variability in the location of the groundwater divide, (ii) changes to the hydraulic gradient, or (iii) changes to Darcy velocities that could result in significantly lower levels of modeled dilution and dispersion, which could result in larger predicted doses. Likewise, lower dispersivities or finer grid discretization could lead to lower levels of modeled dispersion and higher predicted doses. Other areas of uncertainty in the far-field model include: (i) the impact of DOE's source loading approach on predicted doses and (ii) the impact of calcareous zones on contaminant flow and transport. The NRC staff concludes that in future PA updates, DOE could improve the current far-field model to reduce uncertainty in dose modeling predictions. For example:

- The calibration process could be improved and made more transparent, particularly in the area of interest, to provide confidence that the level of dilution in DOE's model is not overstated.

- Loading of the contaminant-flux source terms from the vaults could be evaluated to ensure that the dose estimates are not significantly under-predicted.

- The appropriateness of selected dispersivities and the need for additional vertical or horizontal mesh refinement could be evaluated to ensure that contaminant plumes are not artificially dispersed in the far-field model.

- Additional information could be collected during the monitoring period to support DOE's modeling treatment of the calcareous zones in the lower portion of the UTR aquifer. DOE could consider additional data collection related to calcareous zone outcrop locations and tracer tests to provide further support for the adequacy of its modeling treatment of the lower zone of the UTR aquifer. Site-specific K_ds may be developed for the lower zone of the UTR, if deemed risk-significant.

As a result of the uncertainty in the far-field model, NRC staff will monitor these items when the PA is revised as part of DOE's PA maintenance program. DOE can address this monitoring factor by making appropriate revisions during future PA updates.

2.9 Air Transport of Radionuclides

2.9.1 Modeling of Transport of Radionuclides in the Air Pathway

The airborne and radon analysis evaluated the potential dose due to the diffusion of radionuclides from the waste form to the surface through the air-filled pore space in the overlaying material (i.e., the disposal unit roof and the engineered cover). The air pathway analysis was performed for the three different disposal unit designs (i.e., Vault 1, Vault 4, and the FDCs) and for waste forms created with DDA and ARP/MCU waste (for Vaults 1 and 4) and SWPF waste (for the FDCs). The radionuclides considered in the airborne pathways included only radionuclides that were assumed to have a measureable inventory in the disposal units at the time of closure and that have the potential to form a vapor phase in the waste form. These radionuclides included C-14, Cl-36, H-3, I-129, Sb-125, Se-79, and Sn-126. Tc-99 was also added to the list of radionuclides evaluated in the air transport assessment subsequent to the screening analysis. The flux and associated dose from Rn-222 was also modeled.

The diffusion of the radionuclides from the waste form to the surface and the resulting flux at the land surface was modeled using PORFLOW. In this calculation, boundary conditions were assumed that forced the modeled diffusion to go upwards to the land surface. This assumption increases the estimated flux at the land surface because in reality diffusion will also occur in the downwards and sideways directions. The leaching of radionuclides out of the waste form to the groundwater was not included in the air pathways calculation. Advective transport due to fluctuations in atmospheric pressure that could cause air movement into and out of the soil (i.e., barometric pumping) was also not included in their analysis.

The apparent Henry's law constants used to model the partitioning of the radionuclides between the liquid and gas phases were estimated using the Geochemist's Workbench software. These apparent Henry's law constants were then converted to pseudo-partitioning coefficients for use in the PORFLOW™ code. In the PA, DOE noted that the ionic strengths assumed in the estimation of the Henry's constants were much lower than the expected ionic strength expected in the saltstone pore fluid because a more sophisticated model would be required to assess high ionic strengths and data to support these models is limited. For all of the radionuclides but tritium, higher ionic strengths result in higher activity coefficients and consequently higher partitioning of the radionuclide into the gas phase (i.e., "salting out" effect). In the response to comment AP-2 (SRR-CWDA-2010-00033), DOE discussed the potential effect of the increased ionic strength on the calculated fluxes. Based on experimental data reported in the literature, DOE concluded that the activity coefficients would be unlikely to increase by more than a factor of ten due to the higher ionic strengths.

The PORFLOW™ calculations assumed the minimum closure cap thickness. Additionally, the portions of the closure layer located above the erosion barrier (i.e., the top soil layer and upper backfill layer) were not included in the analysis because these portions of the closure cap are subject to erosion over the time of the simulation. The potential reduction in gaseous flux caused by the presence of the geotextile fabrics, HDPE layer, and the GCL layer in the cap was also not included in this analysis. However, the components of the closure cap located below the erosion barrier were assumed to remain intact for the duration of the simulation.

Once the fluxes of gaseous radionuclides through the cover were determined, these fluxes were used to estimate doses. The dose to an intruder who lives directly above the disposal site was calculated based on the highest peak radon flux into a basement sized control volume (500 m^3) that has an air exchange rate of 1 hr^{-1}. A DCF of 1.4×10^{-5} mSv/Bq (5.0×10^{-5} mrem/pCi) was assumed for Rn-222 (SRR-CWDA-2010-00033). The dose from the other radionuclides to the intruder was based on the assumption that the conservative assumption that the intruder's annual intake is equal to the emission rate from the disposal units (SRR-CWDA-2010-00033). With the exception of Rn-222, the doses to the member of the public were determined from the fluxes using dose-release factors (DRFs) calculated from effective dose equivalents generated using the CAP88 computer code. Receptors at 100 m from the SDF and at the site boundary were considered. The CAP88 computer code used site-specific meteorological information to determine the air concentration of the radionuclides, and then calculated doses from the ingestion, inhalation, air immersion, and ground shine pathways (SRNL-STI-2008-00415). The dose from Rn-222 to a member of the public located at 100 m from the disposal units was not calculated, but this dose would be lower than the dose to the intruder and is therefore bound by the intruder calculation.

Using the analysis described above, the dose to the intruder from Rn-222 was estimated to be 8.2×10^{-13} mSv/yr (8.2×10^{-11} mrem/yr) above the disposal cell with the highest peak radon flux. The dose to the intruder from the other gaseous radionuclides was calculated to be 2.3×10^{-8} mSv/yr (2.3×10^{-6} mrem/yr) above Vault 1, 5.2×10^{-7} mSv/yr (5.2×10^{-5} mrem/yr) above Vault 4, and 3.0×10^{-8} mSv/yr (3.0×10^{-6} mrem/yr) above a FDC. The total dose to a member of the public at 100 m from the disposal units from gaseous radionuclides other than Rn-222 was determined to be 8.14×10^{-16} mSv/yr (8.14×10^{-14} mrem/yr) (SRR-CWDA-2010-00033).

A sensitivity analysis was performed by DOE for the air pathways analysis to assess the effect of uncertainty in the saturation conditions of the cap and the emanation rate for radon (SRNL-L6200-2010-00019). Based on this sensitivity analysis, it was found that the dose to the intruder from Rn-222 was 5×10^{-11} mSv/yr (5×10^{-9} mrem/yr) assuming the minimum moisture content for the cap and overlying materials. The member of the public dose assuming minimum moisture saturation was 1×10^{-15} mSv/yr (1×10^{-13} mrem/yr) from radionuclides other than Rn-222. Additionally, the calculated radon flux is directly proportional to the assumed emanation rate, so increasing the emanation from 0.25 to 0.7, the upper end of reported literature values, would only increase the flux by a factor of 2.8 (SRR-CWDA-2010-00033).

2.9.2 NRC Evaluation – Air Transport

DOE's air transport analyses estimated that the potential doses from gaseous radionuclides will be much lower than the performance objectives. Additionally, DOE also evaluated uncertainty in the moisture content of the soil and the emanation of radon and found that the dose was small over the range of these values evaluated. DOE's analysis included several conservative assumptions, including the use of the minimum cover thickness in the analysis, and the exclusion of the performance of the geotextile, HDPE, and GCL layers, and the exclusion of the layers above the erosion barrier. An additional conservatism in this calculation was that the leaching of radionuclides out of the waste form to the groundwater was not included in the air pathways assessment. This is conservative because leaching is expected to remove some radionuclides from the waste form over time, which would result in a smaller source term. The use of boundary conditions that forced the modeled diffusive flux upwards was another conservatism in DOE's modeling. An additional potential conservatism in the assessment of Rn-222 dose is that the inventory of its ancestors, Ra-226 and Th-230, used in this analysis could be higher than the actual inventory due to the use of conservative assumptions in estimating this inventory (Section 2.2).

An area of uncertainty in DOE's analysis is the exclusion of advective transport due to fluctuations in atmospheric pressure. This type of transport can be a significant source of transport of gaseous material such as radon into a house located on site (Nazaroff, 1992). However, this type of transport can be difficult to predict and model. Additionally, there is some uncertainty in the effect of the assumed ionic strength on the model results. As described in the previous section, DOE provided some literature information about the effect of ionic strength on activity coefficients, but measurements have not been performed for the actual saltstone. The assumed range of moisture conditions is also potentially a source of uncertainty in the air pathway assessment. As noted above, DOE evaluated the effect of setting the moisture content to the lowest end of their predicted range and found that the dose was still small. However, if the actual saturations are lower than the assumed range, the doses could be higher than predicted. Even though there are some uncertainties in DOE's air pathway analysis, the NRC staff concludes that the calculated doses from the air pathway were so small that the doses from this pathway are still likely to be less than the performance objectives, even when accounting for uncertainty.

2.10 Biosphere Characteristics and Dose Assessment

DOE's biosphere model converts calculated groundwater concentrations at the receptor location to public doses considering various exposure pathways (Section 2.3). This modeling involves numerous input parameters that describe the characteristics of the biosphere and receptor. The DOE biosphere model calculates the transport of radionuclides within the biosphere pathways that lead to human exposures. Dosimetry calculations then convert these human exposures to annual doses to a member of the public. Dose modeling of air releases from the Saltstone facility was conducted as a separate modeling activity (Section 2.9).

2.10.1 DOE Approach to Modeling Exposure Pathways and Public Dose

The DOE PA calculates the biosphere pathway transport and receptor dose using a deterministic GoldSim® biosphere model that calculates the dose results for comparison with performance measures. This deterministic biosphere model is separate from the probabilistic GoldSim® model DOE developed to evaluate the sensitivity of dose results to various input parameters. The DOE biosphere model converted groundwater radionuclide concentrations (calculated by the PORFLOW™ transport model) at various locations to all-pathway doses considering the biosphere exposure pathways described in Section 2.3. DOE cited Regulatory Guide 1.109 as the basis for the pathway models (2009 PA). Pathways described in Regulatory Guide 1.109 as applicable to liquid release dose calculations include potable drinking water, aquatic foods, shoreline deposits, and irrigated food. Irrigated food pathways described in the guide include produce (as a combination of non-leafy vegetables, fruit, and grains), milk, meat and poultry, and leafy vegetables. Plant concentrations are calculated in Regulatory Guide 1.109 based on irrigation deposition on plant surfaces and uptake of radionuclides to plants from soil. The Regulatory Guide 1.109 dose calculations involve ingestion, direct radiation, and inhalation routes of exposure.

For Case K, DOE ran the deterministic GoldSim® model with updated biotic transfer factors, additional food sources (chicken and egg), and 25-year buildup and leaching of radionuclides in irrigated soil. The DOE Case K model calculated the receptor dose from ingestion of meat, milk, and poultry-based food products by assuming the animals drink water and consume fodder irrigated with water from a well at the point of maximum exposure 100 m downgradient of the SDF. The effects of buildup and leaching were calculated using a model that modifies the calculated annual soil deposition of radionuclides by the years of irrigation while accounting for losses due to radioactive decay and leaching. Leaching was addressed by applying a root zone soil leaching model (Baes and Sharp, 1983) that evaluates sorption of contaminants in soil based on the water infiltration rate and soil properties including depth, water content, density, and geochemistry represented by distribution coefficients.

2.10.2 DOE Biosphere Model Input Parameter Values

DOE's documentation of biosphere input parameters describes the technical bases for bioaccumulation factors, human health and exposure parameters, and dose analysis parameters. Additional details on these groups of input parameters are provided in the following paragraphs.

The bioaccumulation factors (also commonly known as transfer factors) used by DOE in the initial PA analysis (i.e., in all cases other than Cases K, K1, and K2) have been documented by WSRC–STI–2007-00004. The PA uses factors that represent the following transport pathways: soil-to-vegetable, feed-to-milk, feed-to-beef, and water-to-fish. DOE selected input parameters based on a review of available technical information with a preference for site-specific and more recent compilations. The DOE selection methodology placed priority on site-specific (Friday et al., 1996; Jannik, 2003) or region-specific values (if available) followed by more general literature compilations (Staven et al., 2003; Baes et al., 1984; and the National Council on Radiological Protection [NCRP], 1996). DOE selected site-specific and general literature values without modification, however, the geometric mean of the latest general literature value and the previous SRS selected value was used when the latest value differed from the prior SRS selected value by more than two orders of magnitude (2009 PA).

For Case K, DOE derived updated plant bioaccumulation factors by calculating a weighted average of crop-specific bioaccumulation factors from the most recent International Atomic Energy Agency (IAEA) compilation (IAEA, 2010). DOE weighting factors were based on crop production data (i.e., leafy vegetables 20%, legumes 15%, tubers and roots 10%, and non-leafy vegetables 55%) for South Carolina. Element-specific animal product bioaccumulation factors for feed-to-milk, feed-to-beef, and water-to-fish were updated based on the most recent IAEA compilation (IAEA, 2010). Previous plant and animal product bioaccumulation factors were used for elements that were not included in the referenced IAEA report. Chicken and egg bioaccumulation factors were selected from the most recent source (IAEA, 2010) or Staven et al. (2003) for elements not reported by IAEA, or set to zero if no values were reported in either reference.

The Case K updated soil concentration model introduced additional input parameters to the biosphere model to calculate the soil leaching rate constant. These additional input parameters included the irrigation rate (131.3 cm/yr [51.7 in/yr]) (2009 PA), precipitation and evapotranspiration rates (124.8 cm/yr [49.14 in/yr]; 82.7 cm/yr [32.57 in/yr]) (Jones and Phifer, 2008), and soil moisture content (0.039) (2009 PA), and radionuclide-specific soil distribution coefficients (2009 PA; Table 4.2-15). The soil distribution coefficients used in the calculation of the soil rate leaching constant were assumed to be the same as the K_d values in the vadose zone.

DOE PA human health and exposure parameters also were documented by WSRC-STI-2007-00004. This group of inputs includes food and water consumption rates, the air inhalation rate, exposure times, and local food production and gardening related inputs. DOE selected values from a hierarchy of data sources with preference given to site-specific (if available) and more recent sources of information (2009 PA). Site-specific studies included Hamby (1991; 1992), and regional information was obtained from a recent dose reconstruction study of the SRS site (CDC, 2006). When site-specific information was not available, DOE selected input parameter values from national or international organizations EPA (1997), ANL-EAIS-8, ANL-EAD-4, and NUREG/CR-5512 and assumed the information was applicable to SRS area practices. For Case K, the DOE selected additional chicken and egg pathway input parameters from a prior NRC-sponsored analysis (Simpkins et al., 2008). These inputs included livestock feed and water consumption rates, human consumption rates of chicken and egg food products, and fractions of chicken and egg food that are locally obtained.

DOE dose analysis parameters include dose coefficients that convert human intakes of radionuclides and human exposure to radioactivity outside the body to dose. For the initial PA and for Cases K, K1, and K2, DOE selected adult internal dose coefficients from the International Commission on Radiological Protection, Publication 72 (ICRP, 1996). DOE selected inhalation dose coefficients based on the ICRP recommended lung absorption type, where available, and the most conservative lung absorption type for radionuclides where no ICRP recommendation is provided in the source document (2009 PA). DOE selected external dose coefficients for uniformly distributed ground surface contamination (at an infinite depth with no shielding and at 15 cm) and for water immersion from Federal Guidance Report No. 12 (EPA, 1993). For Case K, DOE made corrections to the values for Pb-210 and U-232 to account for short half-life decay products and updated the value for Pt-193.

2.10.3 NRC Evaluation – Biosphere Pathway and Dose Calculations

To review the DOE PA biosphere pathway and dose calculations, the NRC staff evaluated the general methodology of the DOE biosphere modeling approach and the DOE selection of key input parameters with particular attention focused on the most recent DOE responses to NRC comments on the PA.

2.10.3.1 NRC Evaluation of DOE Biosphere Modeling Approach

The dose methodology of the DOE PA involves executing biosphere pathway and dose calculations with a deterministic GoldSim® model that DOE developed based on the pathway models in Regulatory Guide 1.109 (2009 PA). The NRC staff considers the pathway calculations in Regulatory Guide 1.109 to be a reasonable basis for developing the pathway and dose calculations. The NRC staff verified that the DOE biosphere modeling approach includes the Regulatory Guide 1.109 pathways, with a few exceptions. The DOE biosphere model does not explicitly evaluate separate pathways for leafy vegetables, fruit, or grains. DOE combines a subset of food products from the produce category into a single food category (i.e., vegetables). The NRC staff finds that the use of a single category for vegetables to be technically acceptable provided related input parameters are internally consistent and the analysis is complete (Section 2.10.3.2). Case K includes consideration of leafy and other vegetables in the derivation of plant transfer factors but the analysis does not provide sufficient information to determine whether fruit and grain transfer factors were included in the data that DOE averaged. The Case K analysis also includes the poultry and egg ingestion pathways, which were not included in the initial PA analyses. Based on the preceding analysis, the staff concludes the biosphere pathways considered by DOE are complete.

The dosimetry methodology applied by DOE differs from what is described in Regulatory Guide 1.109. This is expected by the NRC staff because methods and data have been revised since the regulatory guide was published. The methodology used by DOE is based on adult internal dose coefficients from the ICRP (1996) and external (i.e., direct radiation exposure) dose coefficients from Federal Guidance Report No. 12 (EPA, 1993). These dose coefficients are acceptable to NRC staff because they incorporate the most current and appropriate scientific models and methodologies for an adult receptor and are consistent with Commission direction in SRM-SECY-01-0148 (NRC, 2002a).

The NRC staff finds that the approach used in Case K for evaluating soil build-up from a land irrigation scenario in the DOE biosphere model is appropriate. While current radionuclides that contribute most to the calculated all-pathway PA doses involve elements that are generally more mobile in the environment and are less affected by soil buildup, potential future changes to the PA could change the radionuclides that contribute most to calculated all-pathway doses. As a result, NRC staff finds the DOE incorporation of the soil buildup modeling capability into Case K to be a useful improvement.

2.10.3.2 NRC Evaluation of DOE Biosphere Input Parameters

The NRC staff reviewed the parameters used in the biosphere calculations in the original PA and in Case K and finds the basis for the parameter selection as well as the parameters selected to be generally acceptable, with the exception of the items described in the following paragraphs. Additionally, NRC staff finds that the DOE emphasis on modeling site-specific pathways and selecting site-specific or site-applicable input parameters, where available, is generally consistent with past NRC practices that have been described, for example, for demonstrating compliance with NRC decommissioning regulations (NRC, 2003).

The NRC staff asked DOE for more information about its basis for excluding bioaccumulation factors from the uncertainty analysis and the DOE approach to deriving bioaccumulation factors from available data sources (NRC, 2010b, i; B-1). DOE's response included a comparison of dose calculation results using (1) the PA soil-to-vegetable bioaccumulation factors, and (2) the maximum values presented by DOE in Tables 4.6-1 to 4.6-4 of the 2009 PA soil-to-vegetable bioaccumulation factors (SRR-CWDA-2010-00033). While NRC staff agreed the absolute changes to dose were small, they noted in the response (NRC, 2010i) that the relative changes were moderate to significant. Additionally, NRC staff notes that the true maximum values for these factors are larger than the values used in this analysis. For example, the maximum soil-to-vegetable transfer factor assumed for Tc was 5.46 in Table 4.6-1 of the 2009 PA, which is lower than the value assumed for Case K. Due to the wide variability in some of the available transfer factor data and the direct influence on pathway-specific dose results, NRC staff believes that it is important to use technically defensible methods to analyze and propagate transfer factor variability into the PA calculations, in particular, for those elements that exhibit high variability are or have the potential to be important contributors to the total dose results.

In the most recent RAI response (SRR-CWDA-2011-00044), DOE discussed the basis for their use of the geometric mean to average the prior DOE average value with a new DOE estimate in their calculation of the bioaccumulation factor inputs used in the original PA analysis, citing other past uses of a geometric mean by various practitioners. However, NRC staff's principal concern with the DOE's approach was that no basis was provided for averaging the prior DOE average value with a new DOE estimate. DOE's revised Case K used biosphere parameters taken directly from a recent IAEA compilation that did not include any data manipulations using a geometric mean. The NRC staff finds the approach used for the Case K bioaccumulation factors to be an improvement over the past averaging approach.

In the second RAI, NRC staff asked for more information about DOE's basis for excluding leafy vegetables from the derivation of the soil-to-vegetable bioaccumulation factors (NRC, 2010i; B-4). The NRC staff commented that the DOE derivation of the vegetable bioaccumulation factor based solely on bioaccumulation factors for root vegetables (assuming limited local leafy vegetable productivity information) would produce a lower bioaccumulation factor than one that included leafy vegetable bioaccumulation factors. In the RAI response (SRR-CWDA-2011-00044), DOE derived updated bioaccumulation factors for Case K from available data by taking a weighted average of crop-specific bioaccumulation factors from the most recent IAEA compilation (IAEA, 2010) using crop production data (i.e., leafy vegetables 20%, legumes 15%, tubers and roots 10%, non-leafy 55%) for the SRS region. The DOE approach used previous bioaccumulation factors for those elements that were not included in the latest IAEA report. The NRC staff finds the weighted-average approach for updating the vegetable bioaccumulation factors provided in the response to be better than the approach that treated all vegetables as root vegetables, which was used in the DOE PA. The NRC staff finds the approach used in Case K addresses the principal concern of comment B-4 (NRC, 2010i).

The NRC staff finds that the DOE input parameters used in the soil buildup analysis for Case K were referenced to acceptable sources and had sufficient technical bases with a few exceptions. The irrigation rate used by DOE for calculating the leaching rate constant (which DOE refers to as the soil buildup rate) of 131.3 cm/yr (51.7 in/yr) may be overestimated because the value has not been adjusted by the DOE fraction of time vegetation is irrigated (0.2, 2009, PA; Table 4.6-6). Because a higher rate of irrigation also leads to a larger amount of leaching, a high value overestimates leaching and underestimates the buildup factor. However, a higher rate of irrigation also leads to larger amount of deposition of radionuclides on the soil. The overall soil concentration is determined by multiplying the buildup factor described above by the irrigation rate and the concentration in the water. It is not clear to the NRC staff if DOE used the high irrigation rate of 131.3 cm/yr (51.7 in/yr) in the calculation of the soil concentration, or if DOE used an irrigation rate that was adjusted for the fraction of time vegetation is irrigated (26.2 cm/yr [10.3 in/yr]). If DOE used the high irrigation rate in both places, then the net effect would be an overestimation of the soil concentration. However, if a high irrigation rate was used in estimating the leaching and a low rate was used in the calculation of the soil concentration, then the soil concentration and the dose would be underestimated. For example, NRC staff recalculation of the soil buildup factors from the DOE analysis using the adjusted irrigation rate of 26.2 cm/yr (10.3 in/yr) increased the buildup factor estimate (for 25 years with leaching) by approximately 2.6 times for I-129, Ra-226, and Tc-99 relative to the current DOE comment response analysis.

The NRC staff also finds that the distribution coefficients used in the buildup analysis were referenced to a report that selected values conservatively within the context of hydrologic transport modeling (WSRC-TR-2006-00004). Selecting distribution coefficients in this manner is non-conservative when applied to irrigation and soil sorption modeling because lower sorption values are typically regarded as conservative in the context of transport modeling (increases predicted mobility) but lower values of sorption coefficients could underestimate radionuclide build up for the same reason (i.e., by increasing predicted mobility). Additionally, the geochemical environment at the surface may differ significantly from the subsurface (e.g., increased organic content). The estimated buildup in soil for some elements could increase if further review caused distribution coefficients to be updated to higher values. For example, NRC staff evaluated the effect of changing the K_d for Ra from the value of 5 mL/g assumed in the buildup analysis to a value of 25 mL/g which was measured in sandy site soils (SRNL-STI-2010-00527). This change to the K_d value increased the calculated the buildup factor by a factor of 4.8. This increase in the buildup factor would result in a higher concentration in the soil and a corresponding higher concentration from the ingestion of plant and animal products pathways. The K_d values for Tc and I used in the buildup calculation (0.6 mL/g and 0 mL/g respectively) are also less than some of the measured site specific K_d values (SRNS-STI-2008-00286, WSRC-TR-2006-00004). However, the range of K_d values measured for Tc and I was smaller than for Ra, so the potential effect on the calculated buildup factor is also smaller. Additionally, some of the higher K_d measurements for Tc and I were made in soil that had a high clay content, and these values might not be as applicable to the surface.

In the NRC RAIs (NRC, 2010b, i) NRC staff commented that the DOE selected value for the drinking water ingestion rate of 337 L/yr (0.92 L/d) is inconsistent with an average member of the critical group definition. The water ingestion pathway is an important pathway and has consistently been the largest contributor to DOE PA public all-pathway calculated doses. In its latest response (SRR-CWDA-2011-00044) DOE provided additional description of the technical basis for the selected value, which is from an EPA analysis of community water survey results (EPA, 2004). The value chosen by DOE is the national mean per capita community water consumption rate for all ages. However, because the DOE receptors are defined as adults, the selected consumption rate value for all ages is low relative to the value reported for adults (e.g., 401 L/yr relative to the selected DOE value of 337 L/yr). Historically, in establishing the drinking water standards, EPA has accepted a single value for drinking water consumption rate of 2 L/d as a reasonably conservative estimate. NRC has used values of approximately 1.3 L/yr and greater in various previous reviews (NRC, 1977). Considering the site-specific nature of the warmer climate at SRS, the drinking water consumption for the average member of the critical group could be above the EPA national average for adults of 401 L/yr. The water ingestion rate has a linear effect on the predicted dose from the drinking water pathway, so impacts of changing the parameter are therefore easy to evaluate. Because drinking water contributes approximately half of the dose to an off-site member of the public in most of the cases DOE evaluated, changing the water consumption rate to 401 L/yr would increase the total predicted dose by approximately 10% and increasing the consumption rate to 2 L/d would increase the total dose by approximately 50%. For future stochastic analyses, another defensible alternative could be to define a drinking water consumption rate distribution that reflects the range of variability in the national survey data. Use of a distribution is expected to encompass the variability in a population of individuals with different activity levels and consumption patterns.

NRC staff concludes that the Case K biosphere modeling included changes to the modeling approach and parameters that addressed most of the concerns NRC staff identified in the RAIs. However, NRC staff also concludes that some of the parameter values selected in Case K may have resulted in an underestimation of dose, such as the drinking water ingestion rate as well as the irrigation rate and K_d values assumed in the soil buildup calculations. As a result, NRC staff will monitor these items when the PA is revised as part of DOE's PA maintenance program. DOE can address these PA maintenance items by making appropriate revisions and/or providing additional justification for these biosphere parameters when the PA for the SDF is next revised.

2.11 Computer Models and Computer Codes

2.11.1 Computer Models and Computer Codes Used by DOE

DOE used a variety of codes to assess long-term risk from the SDF (2009 PA; Section 4.3.1). The 2009 PA models are integrated to provide long-term dose estimates for comparison against dose-based standards in 10 CFR Part 61, Subpart C (Figure 2.11-1). Primary codes used in the 2009 PA are marked in bold, black text in Figure 2.11-1. This section provides summary description of the various codes used in the 2009 PA modeling including information regarding code capabilities, documentation, verification, and validation testing, as applicable. In addition, DOE's quality assurance program, including software quality assurance, also is described.

HELP

HELP (Hydrologic Evaluation of Landfill Performance) is a quasi-two dimensional model designed by the United States Army Core of Engineers under an interagency agreement with the United States Environmental Protection Agency with the primary objective of performing landfill water balance calculations. HELP version 3.07 (1997) is used to calculate infiltration through the proposed SDF cover. Additional information regarding the use of the HELP computer code to model infiltration through the SDF cover is provided in Section 2.4. Data inputs to HELP include weather, soil, and cover design information. Modeling outputs include runoff, evapotranspiration, lateral drainage, hydraulic head, water storage, and infiltration.

HELP documentation includes a User's Manual, engineering documentation providing source code information and HELP calculation methodology, and verification testing (see http://www.wes.army.mil/el/elmodels/helpinfo.html). Verification consists of testing the ability of the code to adequately predict lateral drainage and other water balance outputs against measured data from large-scale physical models and actual disposal cells at seven sites across the United States.

The HELP code has been extensively used by DOE for various SRS applications including area closure projects under the Comprehensive Environmental Response, Compensation and Liabilities Act.

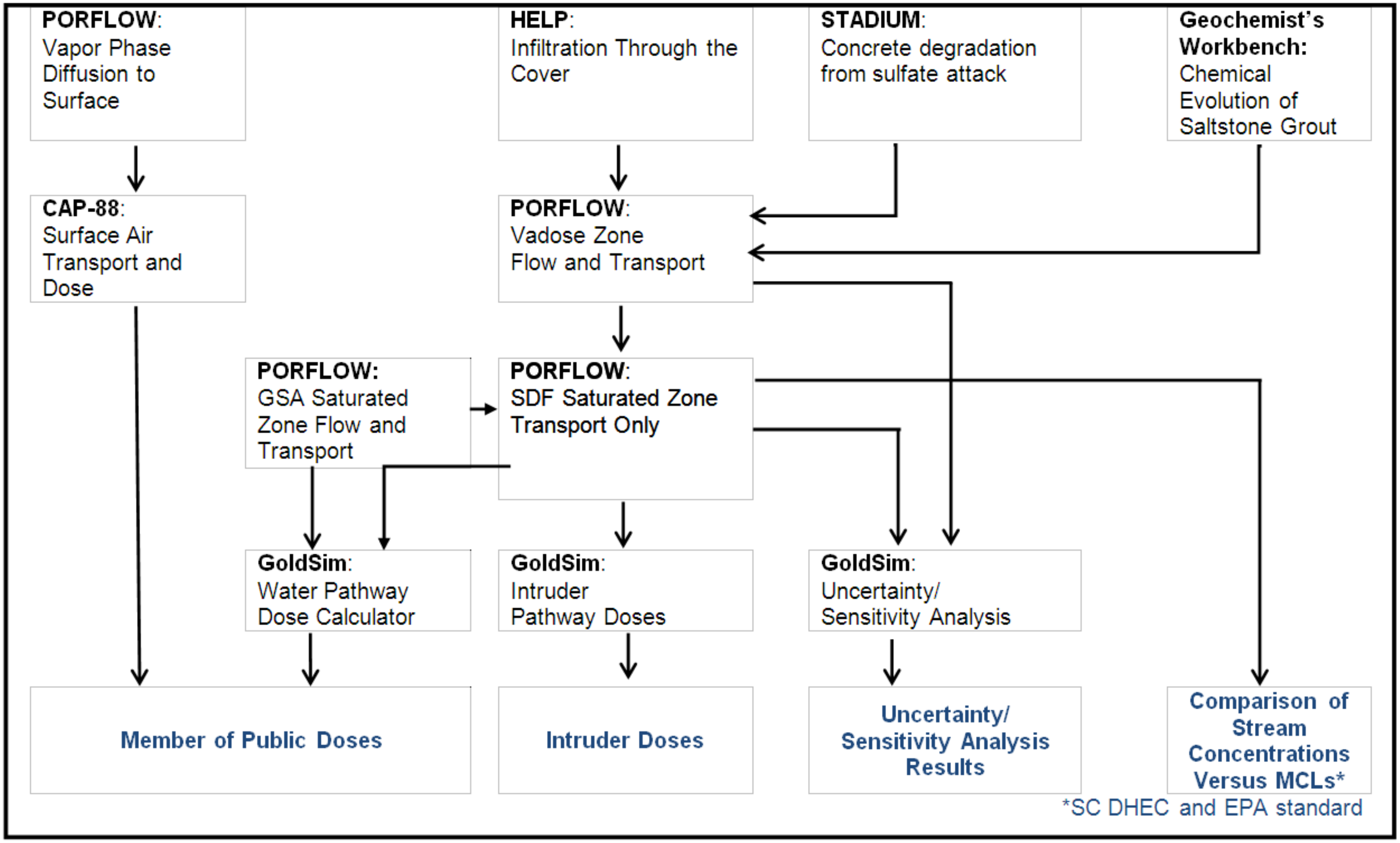

Figure 2.11-1: Model Linkages in DOE's Performance Assessment for the SDF

PORFLOW

PORFLOW™ is a commercially available multi-phase computational fluid dynamics tool developed by Analytic Computational Research, Inc. PORFLOW™ uses numerical methods to solve steady state or transient fluid flow, heat, salinity and mass transport problems in variably saturated, porous, or fractured media. PORFLOW™ version 5.95.0 was used to construct the GSA/ PORFLOW™ model described in greater detail in Section 2.8. PORFLOW™ version 6.10.3 was used in the 2009 PA to perform unsaturated and saturated flow and transport simulations described in greater detail in Section 2.7.3 and 2.8 (Figure 2.11-1). PORFLOW™ version 6.30.2 was used to perform Case K simulations as described in RAI response (SRR-CWDA-2011-00044). PORFLOW™ is also used to perform vapor phase diffusion of contaminants from SDF sources to the ground surface for input in the air transport code CAP-88 (Figure 2.11-1). Flow outputs from PORFLOW™ are also used in DOE's probabilistic assessment executed within the GoldSim® modeling platform discussed in more detail below.

A user's manual and numerous validation reports are available for the PORFLOW™ computer code (www.acricfd.com/software/porflow/). Acceptance testing on PORFLOW™ version 5.95.0 used to construct the GSA/ PORFLOW™ model confirmed that the code conserves mass and satisfies Darcy's Law (WSRC-TR-2004-00106). DOE indicates in the 2009 PA (page 238) that software quality assurance for the version of PORFLOW™ used for the local SDF/PORFLOW™ calculations (version 6.10.3) is covered by software quality assurance plans (WSRC-SQP-A-00028 and G-TR-G-00002) that cover things such as installation and maintenance, acceptance testing, configuration control, and quality control of software models. DOE provides details on the data used to construct the local SDF/PORFLOW™ models and the overall approach to modeling contaminant flow and transport from SDF sources to various points of assessment (SRNL-STI-2009-00115).

PORFLOW™ was selected for use in the 2009 PA modeling for multiple reasons including its capability to perform necessary project functions, extensive verification testing by the vendor and QA testing by site personnel, and given the familiarity of project personnel with use of the code. One key example of PORFLOW's capability to meet project needs is its ability to model first-order decay and progeny in-growth necessary for simulation of radionuclide transport.

GoldSim

The GoldSim® code, developed by the GoldSim® Technology Group, is a commercially available program described by the developer as a user-friendly and highly graphical program for carrying out dynamic, probabilistic simulations of complex systems to support management and decision-making in engineering, science, and business. GoldSim® version 9.6 was used to perform the probabilistic uncertainty and sensitivity analysis in support of the 2009 PA as described in Section 2.1.2. DOE also uses GoldSim® to calculate the dose associated with (i) use of contaminated groundwater by a potential receptor and (ii) direct intrusion into the disposal site for comparison against dose based standards in §61.41 and §61.42. Figure 2.11-1 illustrates model linkages to GoldSim.

GoldSim® Technology Group LLC provides a user's guide and separate data validation guide (GTG-2007 Vol. 1 and Vol. 2, GTG-2006b). DOE provides information about GoldSim® software quality assurance (G-SQA-A-00011).

GoldSim® was selected for use by DOE due to its sufficiency in meeting project needs, input and output capabilities, and given its use in other similar projects (Nevada Test Site and Yucca Mountain).

CAP-88

The Clean Air Act Assessment Package – 1988 (CAP-88) computer model (EPA, 1993) is used to estimate dose from radionuclide emissions to air. CAP-88-PC version 1.0 was used to assess risk to a member of the public from inhalation of radionuclides that diffuse to the ground surface from SDF disposal cells. CAP-88 uses a modified Gaussian plume equation to estimate the average dispersion of radionuclides released from up to six sources at the same release location with different release heights. Assessments are performed with a circular grid with a radius up to 80 kilometers (50 miles). CAP-88 provides estimates of annual dose to Maximally Exposed Individuals (MEI) considering plume and ground shine (gamma radiation), inhalation, and foodstuff ingestion pathways using the results of the vapor phase radionuclide diffusion to surface calculations output from PORFLOW™ (model linkages in Figure 2.11-1) as described in Section 2.9 of this TER.

A user's manual, which provides instruction for use of the CAP-88-PC Version 1.0 is available (EPA-402-B-92-001). The software quality assurance plan (SQAP) for the CAP-88 version used for the 2009 PA calculations is covered by Q-SQP-A-00002.

STADIUM®

The STADIUM® code (Software for Transport and Degradation in Unsaturated Materials) is a proprietary code developed by SIMCO Technologies Inc. to predict short and long-term behavior and service life of concrete. STADIUM® was used by SIMCO Technologies Inc. as a subcontractor to SRS for the 2009 PA to support the calculations of the ettringite front movement from sulfate reactions, and thus degradation front, through the disposal unit concrete. The results of the degradation modeling are abstracted for use in the PORFLOW™ model as described in Section 2.5 of this TER (Figure 2.11-1).

A user guide (STADIUM® User Guide), which provides information on calculation methodology and instructions for use, is available at http://www.mslexperts.com/slm/stadium_help/index.html. STADIUM® has been validated via comparison both to laboratory tests and to field conditions of existing structures. The comparisons are provided in the user guide and show that STADIUM® reproduced the results for both the laboratory and field materials tests (STADIUM® User Guide). The work by SIMCO Technologies Inc. as reported in SRNS-STI-2008-00050 used testing of laboratory prepared samples of the SDF concrete formulations to input the material parameters into STADIUM.

Geochemist's Workbench

Geochemist's Workbench is a commercially available computer program developed by RockWare. Geochemist's Workbench Release 6.0 was used by DOE to model the change in pore water composition (i.e., transition to higher E_h and pH), resulting in K_d changes in the saltstone waste form and disposal units as described in Section 2.6 and 2.7 of this TER. Geochemist's Workbench was also used in simulations of the formation of expansive mineral phases in saltstone (Section 2.6) and in modeling of degradation of the GCL (Section 2.5).

2.11.2 Probabilistic Uncertainty and Sensitivity Analysis

DOE included an uncertainty and sensitivity analysis in its PA documentation. DOE used the software program GoldSim® to construct probabilistic models to perform these analyses. Output data is extracted from PORFLOW™ for use in the probabilistic GoldSim® models. For example, velocity profiles were extracted from each two-dimensional PORFLOW™ near-field model (for each disposal unit type and case) for use in GoldSim® for Cases A - E. Because the same velocity profile is used for each case for each realization, uncertainty in flow through the system is evaluated in a limited sense (i.e., uncertainty from case to case is assessed, but uncertainty in flow in a particular case is not). Furthermore as discussed in Section 2.7, flow through saltstone is constrained to low values in Case A and other PA cases.

As mentioned above, the GoldSim® model does not independently compute flow. Flows through the cover system, saltstone disposal cells, unsaturated, and saturated zones are extracted and simplified from more complex models such as PORFLOW™ for use in GoldSim®. Major components of the engineered system explicitly simulated in GoldSim® for three different disposal cell types: Vault 1, Vault 2 (FDCs), and Vault 4, include the saltstone disposal cells, disposal unit walls, and disposal unit floors. The GoldSim® representation also considers diffusion of radionuclides to "dirt" in contact with the disposal units. Diffusion of contaminants between the saltstone, disposal unit walls, and surrounding soil was solved analytically, rather than numerically, in GoldSim. Direct transfer pathways were used in GoldSim® to transfer mass between cells based on the analytical solutions to the diffusion equation. After transport through the saltstone disposal cells or into "dirt" surrounding the disposal unit walls, radionuclides are transported through the SDF vadose zone located below the disposal units. Twenty cells each are used to represent transport through the (i) saltstone, (ii) disposal unit walls, and (iii) dirt to which radionuclides diffuse from the waste zone. Ten cells are used to simulate transport from the SDF disposal units through the vadose zone underneath the disposal units to the saturated zone. A set of ten waste footprint cells are used to except flux from the vadose zone and transport contaminants to the downgradient edge of the SDF sources in the saturated zone. "Near well" cells are used to transport radionuclides from the waste footprint to the 100 meter boundary. In the probabilistic assessment, the specific aquifer used by the receptor is determined stochastically based on assumed probabilities for a potential receptor completing a well in each aquifer.

A plume function is used to account for plume overlap due to transverse dispersion. The plume function calculates a factor to be applied to centerline plume concentrations at the 100 m boundary for each individual disposal unit to reflect plume concentrations at various reference locations. The center node of each of twelve sectors (sectors are illustrated in Figure 2.3-1) are used as reference points where the contributions of the various disposal unit plumes are calculated (PA, page 320). Values for plume overlap are calculated only for plumes which may reasonably be thought to interact. For example, the model did not simulate any plume overlap between the north and south sections of the SDF, nor were plumes on opposite sides of the flow divide assumed to coalesce.

Inherent differences between the multi-dimensional, PORFLOW™ models and the one-dimensional, analytical (GoldSim®) model make it difficult to compare modeling results. DOE attempts to alleviate this problem through the benchmarking process. Benchmarking was used to change GoldSim® modeling parameters so that certain GoldSim® intermediate results better matched the corresponding intermediate results from PORFLOW™ for the same system and scenario being simulated. DOE deemed benchmarking necessary to allow a common point from which comparisons between the two models could be made. Only Case A and C were initially benchmarked for only three radionuclides: Ra-226, I-129, and Tc-99. Additional cases were benchmarked subsequent to initial efforts. Benchmarking occurred at two locations in the model domain: (i) the flux at the interface between the unsaturated and saturated zones, and (ii) concentrations at the 100 m boundary. DOE also compares member of the public doses at the 100 m compliance boundary for each sector during benchmarking to assess how well the two models match with respect to the overall dose from all radionuclides and disposal units, rather than individual radionuclides and disposal units that might not contribute significantly to dose. For a detailed description of the benchmarking process see Section 5.6.2 in the PA (page 450).

A benchmarking factor was used to attempt to alleviate model divergence associated with simulation of transport of Tc-99. PORFLOW™ uses what is termed a "shrinking core model" to simulate oxidation of saltstone along edges and fractures with oxidation rates controlling release of Tc-99. The "shrinking core model" is not explicitly simulated in GoldSim. Therefore, DOE uses the Tc-99 K_d as a benchmarking factor to control Tc-99 release. An increase by a factor of 500 in the oxidized K_d and a reduction by a factor of 0.3 in the reduced K_d was selected during benchmarking.

Adjustments made to GoldSim® to match flux from the vadose zone to saturated groundwater were constrained to flow parameters and included flow factors for the following: (i) saltstone grout, (ii) disposal unit walls, (iii) "dirt" located adjacent to disposal unit walls, and (iv) vadose zone soils located underneath the disposal units.

Saturated zone transport was complicated by the contributions of various SDF sources to the 100 m boundary locations. Adjustments made to GoldSim® saturated zone parameters included a (i) plume factor to account for groundwater divide effects in Sector J, and (ii) a factor to account for contributions from Vaults 1 and 4 to concentrations at the 100 m boundary in northern sectors.

2.11.3 DOE's QA Program for Computer Codes

The Quality Assurance (QA) program is an integral part of SRS's Integrated Safety Management System (Management Policies (MP), WSRC 1-01, Policy 4.2 Quality Assurance). The QA policy requires that the SRS QA program comply with DOE O 414.1C, 10 CFR 830, Subpart A, and the Quality Assurance Management Plan (QAMP, WSRC-RP-92-225). The QAMP provides for the prevention, identification, and correction of any identified errors, as well as establishing an evaluation process to further continuous improvement.

General requirements for QA are described in Quality Assurance Manual, Manual 1Q, Procedure 2-1 Quality Assurance Program. The software quality assurance plan (SQAP) requirements are described in Procedure 20-1, Software Quality Assurance Manual (Manual 1Q 20-1). DOE indicates that Procedure 20-1 fulfills the requirements of DOE Order 414.1.C and 10 CFR 830, Subpart A.

Conduct of Engineering and Technical Support Procedure Manual E-7, Procedure 2.60, Technical Reviews (E7 Manual 2.60) is the QA implementing procedure for performing technical reviews. Various levels of rigor are required based on the risk associated with the end use of the data and calculations. With respect to PA activities, DOE indicates that a design checker assures the technical accuracy of the design document by performing design check activities such as mathematical checks, if appropriate, review for correct use of inputs, including quality requirements, review of approach used, and reasonableness of the output. Design checkers must be knowledgeable in the area being reviewed and be able to perform similar designs or analyses as those being checked but must also have independence from the actual work being checked.

DOE indicates that assurance that the input data to the various codes is verified to be accurate is integral to the model integration process. DOE documented the verification of the model input traced from source documents to modeling input and finally to appropriate sections in the PA (Appendix I).

2.11.4 NRC Evaluation – Computer Models and Computer Codes

2.11.4.1 NRC Evaluation – Models, Modeling Approach, and Model Integration

With noted exceptions, NRC staff generally finds the codes and models used by DOE to be adequate for the purpose of evaluating disposal facility compliance with the performance objectives in 10 CFR Part 61, Subpart C. Vendor verification and validation testing exists for many of the codes selected and a software quality assurance program is in place to help ensure the integrity of model calculations (i.e., ensure that model equations are solved consistently and correctly). Data input quality is also addressed in DOE's quality assurance program. Data input quality is essential to ensuring the quality of model outputs. However, independent peer review of FTF geochemical modeling identified issues with DOE's quality assurance program (LA-UR-12-00079). FTF geochemical modeling is relevant to SDF as geochemical modeling and data inputs are similar. The independent peer review found issues with the quality of thermodynamic data used in the calculations, as well as issues with transparency and traceability of documentation, and project integration issues (LA-UR-12-00079). NRC staff expects DOE to address the quality assurance issues identified by the independent peer review group, as applicable to the SDF PA modeling inputs, models, and calculations in future updates to its PA.

With respect to code selection, the codes used by DOE appear to be thoughtfully considered. However, the use of a more sophisticated code to model infiltration through the closure cap may provide more realistic estimates of infiltration. Although the HELP code may be suitable for estimating long-term water balances, short-term events and trends may not be adequately represented (Bonaparte et al., 2002). Bonaparte et al. (2002) state, “the model will generally not be adequate for use in a predictive or simulation mode, unless calibration is performed using site-specific measured (not default) material properties and actual leachate generation data.” Since calibration data over the lifetime of the planned closure cap is unavailable, the use of an alternative code may provide more defensible infiltration estimates.

With respect to model integration, the NRC staff notes a few irregularities that result from the use of more than one PA model to evaluate various performance measures. For example, the GSA/PORFLOW™ model used to calculate stream concentrations to evaluate dose impacts associated with recreational use of stream water has a coarser grid resolution then the SDF/PORFLOW™ model. Because (i) sources are loaded at the element closest to the SDF disposal units, (ii) the selection of source loading locations are fewer in number for the GSA/PORFLOW™ model given its coarser resolution (compared to the SDF/PORFLOW™ model), and (iii) slight variations in source loading locations can lead to variations in flow paths away from the SDF (given divergent flow associated with the groundwater divide at SDF), flow paths away from some SDF sources to GSA streams appear to differ between models (flow paths in Figure 4.4-12 [produced from GSA/PORFLOW™ model] compared to 4.4-19 [produced from SDF/PORFLOW™ model]). Other source loading issues are discussed in Section 2.8 on far-field modeling. For example, code limitations (e.g., inability to load flux as a recharge concentration) or modeling approach (e.g., use of a local SDF/PORFLOW™ model that does not include a recharge boundary condition) may limit options to addressing source loading issues; however, if deemed risk-significant, a change to DOE’s modeling approach may be warranted.

The NRC staff concludes that increased transparency of modeling results and evaluation of intermediate modeling outputs could lead to a greater understanding of total system performance, as well as facilitate problem identification and resolution. For example, the decision to break up the cover system and model separately upper and lower cover components in HELP and PORFLOW™ makes it difficult to evaluate total cover system performance. In fact, the No Closure Cap Case addresses under-performance of most of the cover but allows the lower lateral drainage layer (located above all disposal units) and HDPE/GCL (located directly above the FDCs) to operate at the same level of performance as in Case A (Section 2.13). In general, barrier performance often is obscured in PA sensitivity cases due to the redundancy of barriers and the tendency to “switch off” some barriers or parts of barrier systems while others remain “on” (NRC, 2010b; PA-8). In another example, the Synergistic

Case appears to be a highly conservative sensitivity case (i.e., represents a highly degraded state of the entire engineered system). However, the dose impact of this case is limited given the limited flow of water through the saltstone matrix with the bulk of the flow going through the cracks in the saltstone and the disposal unit walls. Therefore, rather than representing a seemingly bounding case as one might surmise based on its label, the doses associated with the Synergistic Case appear to be limited. Evaluation and presentation of intermediate outputs (e.g., flow rates through saltstone) may have led to a change in the approach used to model this PA case and faster resolution of technical issues.

Information regarding performance of the FDCs, which the NRC staff found to be one of the two most important barriers affecting engineered system performance in Case K, was not provided. As discussed in Section 2.7 and 2.13, the FDC disposal unit floors served to substantially attenuate release of Tc-99 with over 90 percent of the Tc-99 retained in the disposal unit concrete shortly after release from the saltstone monolith. Although the disposal unit concrete significantly decreases the release rates from the engineered system, DOE did not provide details regarding key modeling assumptions and results related to disposal unit performance for this very important barrier in Case K.

2.11.4.2 NRC Evaluation - Probabilistic Uncertainty and Sensitivity Analysis

The NRC staff has concerns with how the flow-field was "hard-wired" into the probabilistic GoldSim® model. As previously discussed, flows were not independently modeled in the probabilistic model. Deterministic PORFLOW™ flow outputs were used to parameterize the GoldSim® models. Flow of water into the saltstone matrix is substantially limited in most of the deterministic PA Cases evaluated in the probabilistic model (Section 2.7). Because flow rates in each case are not varied in the probabilistic analysis, the peak dose is expected to be biased low in the probabilistic analysis. NRC staff concludes that the probabilistic GoldSim® model does not appropriately consider uncertainty in dose predictions due to potential uncertainty in the magnitude of flow of water and oxygen into the saltstone waste form. While flow variability could have been considered in the probabilistic model through evaluation of alternative conceptual models with higher flow rates, the probabilistic model only included a single PA case representing higher flow rates through the saltstone matrix (i.e., Case E), and this case is assigned a significantly lower probability (ten percent for Case E compared to eighty-five to ninety-five percent for Case A). Additionally, other cases with higher flow rates through the saltstone matrix that the NRC staff expects to be more likely (e.g., Case K) were developed after PA preparation and are, therefore, not considered in the probabilistic assessment.

With regard to benchmarking, the NRC staff concludes that documentation of the benchmarking process used to align the GoldSim® probabilistic modeling results with deterministic PORFLOW™ modeling results could have been more complete and transparent in the PA. However, in response to NRC comment, DOE provided a well-developed comment PA-5 response (SRR-CWDA-2011-00044) that attempted to explain and illustrate the need for and effect of each benchmarking factor or set of benchmarking factors added to the GoldSim® model to improve the match between PORFLOW™ and GoldSim® modeling results. However, because the benchmarking process inherently assumes that PORFLOW™ modeling results are somehow better or more valid than GoldSim® modeling results, PORFLOW™ models must first be shown to be well calibrated, well understood, and sensible to prevent the inadvertent transfer of modeling artifacts or inaccuracies from the PORFLOW™ model to the GoldSim® model. The NRC staff observed that the SDF/PORLOW model does not, in fact, appear to be well calibrated (Section 2.8). Furthermore, it is not clear that the complicated flow patterns observed in deterministic near-field modeling are well understood and supported, as details are lacking in PA documentation. Because PORFLOW™ modeling results lack transparency and validation, it is difficult to conclude that the multitude of adjustments made to the GoldSim® model to mimic the complex flow behavior of water and contaminants through the engineered and natural systems are beneficial or well supported. DOE appears to take the approach of significantly simplifying the representation of certain processes in GoldSim® and adding in adjustment factors to enable reproduction of the complex system behavior observed in PORFLOW. If the PORFLOW™ model is determined to be well calibrated and validated, this approach might be acceptable, but it also limits the ability of the GoldSim® model to serve as an independent check of the PORFLOW™ modeling results. An independent model from which comparisons can be made and distinct differences in system behavior can be explained can only add value to the overall modeling process.

As previously discussed, DOE's response to NRC comment PA-5, (SRR-CWDA-2011-00044) illustrates the effect of adding in benchmarking factors or sets of benchmarking factors in a similar sequence to what was done during the actual benchmarking process. In some cases, adjustments lead to order of magnitude or more changes in near-field GoldSim® parameters. While some notable improvements were made (e.g., peak dose from Ra-226 from FDCs), the benefit of these adjustments is not always clear. For example, the Ra-226 fluxes out of the unsaturated zone from Vault 1 seem to benefit little from these adjustments with fluxes differing by an order of magnitude between models after benchmarking. Vault 4 flux comparisons are generally worse following the benchmarking process with a lowering of GoldSim® fluxes out of the unsaturated zone.

With regard to saturated zone benchmarking, the NRC staff recognizes improvements in the match between modeling results after the application of benchmarking factors; however, the basis for the benchmarking factors is not always intuitive. DOE used very large adjustments to the contributions of Vault 4 to the Sectors A - C concentrations of Ra-226: factors of 70, 100, and 50 that DOE indicates account for heterogeneities in the flow rates (only a single Darcy velocity is used in GoldSim) and possibly due to vertical mixing. NRC staff reviewed PORFLOW™ modeling files and noted that plumes emanating from Vault 4 actually appear to be located relatively low in the Upper Three Runs aquifer at the 100 m boundary. Therefore, it appears that the Vault 4 plumes are vertically mixed in a larger aquifer thickness compared to the 12 m (39 ft) assumed in GoldSim® modeling necessitating a decrease, rather than an increase, in the concentrations of Ra-226 in Sectors A - C in GoldSim. DOE assigns an average Darcy velocity of 45 m/yr in the probabilistic GoldSim® model. NRC staff agree that the flow field is heterogeneous across the SDF; however, particle tracks from Vault 4 (PA, page 386) do not seem to be significantly different than average flows across the site that would account for the rather significant factors used to adjust the Vault 4 100 m concentrations in Sectors A - C, again making it difficult to explain the large factors used in this adjustment (i.e., differences in aquifer thicknesses and velocities are on the order of factors of 2 to 4, respectively, and would not explain the factor of 50 to 100 increase in Ra-226 concentrations in Sectors A - C). DOE uses a benchmarking factor of 5 to adjust the concentrations from FDCs 7A - D[23] (Figure 2.8-8) due to minimal mixing and possible vertical dispersion effects. The NRC staff agrees that dilution factors near the groundwater divide near Sector I are low; however, contaminant plumes emanating from FDCs 7A - D are well-dispersed vertically, potentially leading to an offset of low dilution associated with low flow (i.e., the adjustment may have been greater than 5 had it not been for significant vertical mixing). DOE also uses an adjustment factor to account for contributions from Vault 1 to Sectors E and F due to transverse dispersion. Review of PORFLOW™ model outputs indicates that while small amounts of contamination from Vault 1 are present in Sectors E and F, the contributions to Sector F are orders of magnitude lower than contributions to other Sectors. DOE uses an adjustment factor of 0.5 to decrease 100 m well concentrations to account for additional transverse vertical dispersion. Given the small aquifer thickness assumed in the GoldSim® modeling (i.e., 12 m), the NRC staff agrees that the amount of vertical mixing is expected to be larger particularly for northern sector plumes and sources located further from the 100 m boundary. However, the figures showing the effect of the benchmarking factor on the results seemed to show doses further out of alignment for some key radionuclides and sectors (e.g., Ra-226 and Sectors A - C). Nonetheless, the overall effect of all of the adjustments seemed to improve matches between the two models.

Finally, DOE indicates that an adjustment had to be made to the GoldSim® model to account for contributions of Vaults 1 and 4 to northern sectors due to northerly flow from these vaults when the plumes are transported vertically downward to the Gordon aquifer. It was not obvious to the NRC staff why adjustments needed to be made during the benchmarking process to account for flow from these vaults to the north in the Gordon aquifer, an aquifer that is not explicitly represented in the GoldSim® model. As discussed below, DOE assumes that Gordon aquifer concentrations are 5 percent of the concentrations in the Upper Three Runs aquifer and states in the PA that DOE believes the factor to be conservative (a factor of 600 lower concentrations have actually been calculated for conservative tracers in the Gordon aquifer compared to the Upper Three Runs aquifer based on PORFLOW™ modeling simulations). Therefore, it is not clear why contamination of the Gordon aquifer from Vault 1 and 4 sources was significant enough to warrant adjustments to the GoldSim® model in representing Gordon aquifer concentrations. NRC staff hypothesize that very low concentrations of key radionuclides in Sectors G - L that occur very early (prior to FDC failure and therefore attributable to inventory in Vaults 1 and 4) represent the peak concentrations in those sectors but actually occur in the Gordon aquifer during early simulation timeframes. Thus, inventory from Vaults 1 and 4 may be used in benchmarking to increase 100 m concentrations in Sectors G - L in the Upper Three Runs aquifer, the only aquifer simulated in GoldSim, because the concentrations would otherwise be zero in the Gordon aquifer (0 multiplied by 5 percent is 0). Therefore, it appears DOE may have allowed nominal concentrations of key radionuclides to be present in the Upper Three Runs aquifer to allow an even smaller amount of these key radionuclides to be present in

[23] Disposal unit designations used in the TER (Figure 2.8-8) correspond to designations used in the PA, which may not correspond to current designations.

the Gordon aquifer (5 percent of the Upper Three Runs aquifer concentration). While physically inconsistent with the hydrogeological conceptual model (the Upper Three Runs should not be contaminated from Vaults 1 and 4 in the northern sectors), the need for the adjustment factor can be explained.

The NRC staff concludes that the benchmarking process could be more transparent in PA documentation and appears in some cases to be more like a fitting routine rather than a process by which physical differences between two models can be explained and addressed. In some cases benchmarking factors are used to compensate for lack of complexity in the GoldSim® probabilistic modeling (e.g., shrinking core model implementation in PORFLOW™ that is not explicitly represented in GoldSim® modeling or differences in the dimensionality of flow fields between the two models). In some instances, benchmarking factors often appear at odds with one another and some factors appear to lack physical basis. Given the issues identified by the NRC staff in the probabilistic model discussed below, it is likely that some adjustments may have been made to the GoldSim® model to compensate for errors in the probabilistic model. Furthermore, as discussed in comment PA-4 (NRC, 2010i), benchmarking is limited to just three key radionuclides: Ra-226, Tc-99, and I-129, potentially limiting the GoldSim® model's ability to accurately evaluate of contributions of other constituents with different transport behavior to peak dose in the probabilistic analysis. If additional isotopes that were not originally benchmarked are potentially significant dose contributors for the SDF, the benchmarking process could benefit consideration of additional radionuclides not previously considered.

In its probabilistic assessment, DOE considers the probability of a potential receptor completing a well in various SDF aquifers based on regional data. Concentrations in the Gordon aquifer from SDF source release are expected to be significantly less than those in the Upper Three Runs due to attenuation in the Gordon Confining Unit and increased dilution in the Gordon Aquifer. Therefore, consideration of doses due to exposure of a potential receptor to various groundwater dependent pathways resulting from extraction of contaminated well water from the Gordon aquifer (in lieu of the Upper Three Runs aquifer) will significantly lower the overall peak of the mean dose in the 2009 PA given the relatively high probability of well completion in the Gordon aquifer (Table 2.11-1). The NRC staff communicated its concern with DOE's use of exposure point concentrations that did not represent the point of maximum exposure in a FTF scoping meeting (DOE, 2008) and in the FTF TER (NRC, 2011m). Thus, while averaging doses based on use of well water from various aquifers and probability of well completion may provide valuable risk insights, use of the dose at the point of maximum exposure in the UTR aquifer is more appropriate for comparison against 10 CFR Part 61, Subpart C performance objectives.

Although NRC is unable to rely on DOE's probabilistic analysis due to issues discussed in this section, if DOE intends to rely on its probabilistic analysis in the future, the point of assessment within the Gordon Aquifer may also be risk-significant. To determine the Gordon aquifer concentrations for use in the probabilistic model, aquifer concentrations at the 100 m boundary in the Gordon aquifer were compared to concentrations at the 100 m boundary in the Upper Three Runs aquifer. Based on this comparison, Gordon aquifer concentrations were assigned a relative concentration of a factor of 20 lower than Upper Three Runs aquifer concentrations for use in the probabilistic assessment. However, if higher concentrations are present at points beyond the 100 m boundary in the Gordon aquifer that were not considered in the comparison, then the default factor of 20 lower concentrations assigned to wells that may be completed in the Gordon aquifer may under-predict doses for relatively non-sorbing constituents that tend to drive the peak dose at SDF.

Table 2.11-1: Aquifer Exposure Probabilities and Relative Concentrations

Aquifer	Exposure Probability	Fraction of UTR-UZ Concentration	Weighted Concentration
UTR-UZ	0.13	1.0	0.13
UTR-LZ	0.44	1.0	0.44
Gordon	0.43	0.05	0.02
Total	1.0	NA	0.59

Data in Columns Two and Three are from the 2009 PA pages 488-489.

In fact, it is not clear that the exposure point concentrations at the 100 m compliance boundary in the Gordon aquifer are the maximum concentrations at or beyond the 100 m boundary. DOE presents PORFLOW™ simulation results that indicate plume concentrations decrease with distance from the source (2009 PA, pages 389, 390). However, the figures presented by DOE in cross section (e.g., Figures 5.2-6 and 5.2-7) actually seem to indicate that the maximum concentration in the Gordon aquifer occurs beyond the 100 m compliance boundary. This would be especially true for sources located near the SDF boundary and further away from the groundwater divide. For other sources located further away from the SDF boundary or on the groundwater divide, vertical gradients and transport distance may be sufficient to allow transport of the centerline of SDF source plumes to the Gordon Confining Unit within the 100-m compliance boundary.

In an FTF RAI resolution meeting held on July 21, 2011, (NRC, 2011d), DOE indicated that it thought a factor of 20 reduction in concentration in the Gordon Aquifer is sufficiently conservative for locations of higher concentration that may occur beyond the 100 m (330 ft) boundary at FTF. It is not clear to the NRC staff if a factor 20 reduction in concentrations of Upper Three Runs source concentrations at the 100 m point of compliance also is bounding for concentrations in the Gordon aquifer at SDF considering that Gordon aquifer concentrations may be higher beyond the 100 m boundary.

In addition to the concerns described above, NRC staff found that sufficient justification was not provided for many of the parameter distributions included in DOE's probabilistic GoldSim® model. The parameters discussed below are examples of areas that NRC staff have identified as needing more justification is not intended to be a comprehensive list. One example where a stronger basis is needed is the assumed probabilities of Cases A - E. When DOE's GoldSim® model was run for "All Cases," the model probabilistically selected a case among five cases, 2009, PA Cases A - E. In the response to NRC comment PA-11 (SRR-CWDA-2011-00044), DOE stated that these probabilities were developed by engineers and scientists on the 2009 PA development team using a systematic, step-by-step approach.

However, as noted in this RAI (NRC, 2010i) NRC staff finds that the probabilities are unrealistic and do not reflect the current and expected future conditions of the system.

Additionally, NRC staff does not find the ±50 percent uncertainty range assumed for E_h and pH transition times to adequately capture the transition time uncertainty. DOE provided additional justification for this range in the response to NRC comment SP-8 (SRR-CWDA-2011-00044). Based on the additional information, NRC considers the ±50 percent uncertainty range likely to capture uncertainty that could result from variability in initial saltstone mineralogy, bulk density, and porosity adequately. However, it appears unlikely that the ±50 percent uncertainty accounts for differences caused by the choice of initial mineralogy in addition to uncertainty caused by other factors, such as the assumed reducing capacity of saltstone, which could cause the E_h and pH transition pore volumes to be lower than DOE's Case A values.

The NRC staff also concludes that there is insufficient justification for the uncertainty distributions for cementitious material K_d values. In developing these uncertainty distributions, DOE used a statistical analysis of data on SRS sediments from a single borehole in E-Area. In response to an original and follow-up NRC RAI on this topic, DOE acknowledged that the approach is not optimal and implied that it will consider studies that may help elucidate uncertainty and variability in cementitious material K_d values (SRR-CWDA-2010-00033, SRR-CWDA-2011-00044; SP-18). The NRC staff raised the same issue during its review of the F-Area Tank Farm PA (SRS-REG-2007-00002 Rev. 1). As discussed in that review (NRC, 2011m; pages 128, 130), the NRC staff does not think it is sufficient to note that variation in sediment K_d values is likely to exceed variation in cementitious materials. Additionally, recent research at SRNL (SRNL-STI-2011-00672), found that the range of measured K_d values for the sorption of Cs, Sr, and I onto a variety of formulations of simulated saltstone was much larger than the K_d range used by DOE in stochastic models. However, the authors of this study suggested that the large range could be due to experimental conditions that do not prevail in the field and the small number of samples analyzed.

Finally, NRC staff also noted in comment B-1 (NRC, 2010i) that the basis for excluding the biotic transfer factors from the uncertainty analysis was unclear. DOE stated in their RAI response (SRR-CWDA-2011-00044) that they believed that analyses performed by DOE (SRR-CWDA-2010-00033; B-1) and CNWRA (2008) demonstrated that these parameters did not have a significant effect on dose and that the use of uncertainty distributions for these parameters could result in risk dilution. NRC staff notes that while DOE's assessment showed small absolute changes in the dose due to these parameters, the relative changes were large. Additionally, the analysis performed by CNWRA was intended to be a preliminary sensitivity analysis to identify the most important parameters in the CNWRA BDOSE model. Additional justification beyond this preliminary sensitivity analysis is needed for excluding the biotic transfer factors from the uncertainty analysis. Finally, since these parameters affect the magnitude of the peak dose, not the timing of the peak, using uncertainty distributions would not result in risk dilution when calculating the peak of the mean. As discussed in more detail in Section 2.10, it is important to use technically defensible methods to analyze and propagate transfer factor variability into the 2009 PA calculations due to the wide variability in the transfer factor data.

2.11.4.3 NRC Evaluation - Quality Assurance

NRC staff has evaluated DOE's quality assurance program and concludes that DOE has an adequate quality assurance program, procedures, and plan in place to ensure the quality of PA documentation, modeling, and calculations. Nonetheless, execution of the quality assurance program appears to be sub-optimal in some key areas. Significant deficiencies in probabilistic PA models and documentation have been identified. Specifically, NRC staff noted several irregularities in GoldSim® probabilistic modeling as described below.

As an example of a potential break-down in quality controls, the NRC staff concludes that design checks were not effective in ensuring the reasonableness of model outputs. DOE provided results in its PA that showed that the peak of the mean dose for the SDF would occur at around 450,000 years in Sector B (Figure 5.6-39 in PA). Upon further investigation of GoldSim® model outputs, NRC staff concluded that the overall peak of the mean dose is attributable to Ra-226 and Pu-239. Given the half-life of Pu-239 (24,000 years), the NRC staff believes a more thorough quality check of the results would have caused DOE to question the timing of the peak dose. The NRC staff found that unit inventories are used in the GoldSim® model and then inventory multipliers are applied to model outputs to calculate doses based on the assumed SDF inventory. The GoldSim® model inaccurately multiplies the Pu-239 concentrations at the 100 m boundary by the Pu-239 inventory, rather than by the virtually negligible Cm-247 inventory, predecessor to Pu-239, responsible for the Pu-239 found at the 100 m assessment point a few hundreds of thousands of years after closure. A careful review of the model output would have likely led to identification of this issue in the GoldSim® calculations, as significant quantities of Pu-239 would not be expected at the 100 m boundary at the time of the peak dose (which was greater than 10 half-lives of Pu-239), unless through production from a predecessor radionuclide. Furthermore, peak dose from Ra-226 appears to be underestimated by this approach (long-term Ra-226 dose appears to be attributable to in-growth from U-238 and U-234 with the U-234 inventory significantly greater than the Ra-226 inventory but being multiplied by the lower Ra-226 inventory). More detailed evaluation and interrogation of a key PA model output, the overall peak of the mean dose, was expected based on DOE's documented QA procedures.

Furthermore, as noted in Section 2.1, the NRC staff did not fully review the DOE probabilistic GoldSim® model due to NRC staff concern regarding the use of outputs from deterministic cases (Cases A - E) which NRC staff did not consider to be appropriate inputs to the GoldSim® model. NRC staff also had concerns about the adjustments made to the model during the benchmarking process. However, in performing a brief review of this model, NRC staff identified a number of items that seemed to be errors in the model. For example, the DOE GoldSim® model is designed to either run Cases A - E individually or to run "all cases" (including Cases A to E). When the model is run in the "all cases" mode (i.e., when the FixedRunCase_switch is set to false), a case is selected probabilistically for each realization based on DOE's predicted probabilities for each case. The flows for Cases A - E seem to be selected correctly based on the case that was randomly selected. However, parameters related to fractures in Case C (BypassFrac, BypassWallDiff, CaseCTurnOffDiff) are not linked to the stochastic "Configuration" element and consequently do not change based on the case that is probabilistically selected. Because of this, the “all cases” results do not seem to include key features of Case C, which likely led to the all cases probabilistic dose being underestimated.

The flows in the unsaturated zone cells are also linked incorrectly. According to the documentation in DOE’s GoldSim® model, the outflow from UZCell_Out is intended to be evenly divided between the ten footprint cells below the disposal units. However, one ninth of the outflow from UZCell_Out goes to the Footprint_In cell and the remainder is split between UZCells 2 - 9. This results in a loop in which the contaminated water is routed back up into the UZ instead of going to the cells below the unsaturated zone. The effect of this contaminant routing is that the timing of the peak is delayed and the magnitude of the peak is decreased.

In addition, the Vault selector for Vault 4 (data element “Vault” under \TheVaults\Vault 4) seems to be set incorrectly. The value for this selector is 2, which seems to correspond to an FDC. This selector affects the modeled vault area and the “Vault Index” (which affects the thickness of the unsaturated zone).

2.11.4.4 Computer Codes and Models Conclusions

The NRC staff finds the computer codes used by DOE in the performance assessment to be generally adequate for demonstrating compliance with the performance objectives. The NRC staff also believes that increased transparency in PA documentation and a shared understanding of total system performance facilitated by presentation and evaluation of intermediate modeling outputs could lead to more efficient resolution of key technical issues. NRC staff notes that some of the 2009 PA cases appear to be inconsistent with the conceptual model for the case. In these instances, presentation of intermediate outputs could facilitate identification of potential issues and provide early opportunity for adjustments, if determined to be necessary.

Additionally, as discussed in Section 2.1, the NRC staff is unable to use DOE’s probabilistic analysis to assess facility compliance with the performance objectives in 10 CFR 61, Subpart C. DOE’s probabilistic model is based on a set of PA cases (Cases A - E) that NRC staff does not think appropriately represent the system based on currently available information. The NRC staff also has concerns regarding the transparency of and basis for some of the factors applied in the benchmarking process used to adjust the outputs of the probabilistic model to more closely match those of deterministic PORFLOW™ models. Improvements to the benchmarking process could be made in the future. Additionally, key parameters in the probabilistic assessment are “hard wired” (i.e., are not treated as uncertain parameters) limiting the evaluation of uncertainty in peak dose due to parameters such as flow rates through the saltstone waste form. The NRC staff also concludes that some of the uncertainty distributions used by DOE are not adequately supported. Finally, as described above, the NRC staff finds a number of possible errors in DOE’s probabilistic GoldSim® model that further limit the usefulness of probabilistic modeling results.

2.12 ALARA Analysis

2.12.1 ALARA Analysis

The method DOE used to demonstrate that doses to the off-site member of the public be maintained As Low As Reasonably Achievable (ALARA), as required by the 10 CFR 61, Subpart C performance objectives, is presented in Section 5.7 of the 2009 PA. The discussion that follows was provided in that section and is supplemented by DOE's responses to NRC's RAI's (SRR-CWDA-2011-00033 and SRR-CWDA-2011-00044).

DOE's ALARA program is based on DOE Order 435.1-1. The NRC's ALARA requirement, as it applies to §61.41, is as follows:

"Reasonable effort should be made to maintain releases of radioactivity in effluents to the general environment as low as is reasonably achievable."

As noted in DOE's second RAI response (SRR-CWDA-2011-00044, comment A-1) DOE's guidance document for DOE Order 435.1-1 states in part: "*…that the goal of the ALARA process is not the attainment of a particular dose level (or, in this case, level of release), but rather the attainment of the lowest practical dose level after taking into account social, technical, economic, and public policy considerations. The PA should include assessments that focus on alternatives for LLW disposal. ALARA is meant to provide a documented answer to the question: 'Have I done all that I can reasonably do to reduce radiation doses or releases to the environment?'"*

In the 2009 PA, DOE stated that the goal of its ALARA process is to attain the lowest practical dose level after taking into account social, technical, economic, and public policy considerations. DOE also notes that the ALARA program at the SRS is well-documented. DOE assumed in the PA that using the SRS ALARA program, processes, and typical protocols to meet the ALARA requirement of §20.1003 would be sufficient to demonstrate that salt waste disposal at the SDF meets the §61.41 ALARA requirement.

The PA states that the design of the FDCs was intended to improve the demonstration of ALARA at the SDF by including features that were expected to reduce releases of radioactivity below levels necessary to meet the dose limit. The PA lists some of these features, including those listed below.

- Carbon steel walls to ensure watertight containment
- Minimum 20 cm (8 in) thick pre-cast walls of Class III sulfate resistant concrete
- Interior coating
- GCL in place above and below the FDCs
- HDPE liner completely surrounding the cells
- Thin layer of Class III sulfate resistant concrete on top to protect the GCL-HDPE during construction

DOE notes in SRR-CWDA-2011-00044 that, in addition to evaluating and improving disposal cell design, DOE's ALARA process also includes (1) making conservative assumptions when modeling, and (2) evaluating and implementing alternative salt processing processes that could reduce the SDF inventory. For example, DOE notes the Small Column Ion Exchange technology as an alternative considered for better salt processing.

An additional example of a conservatism noted by DOE (SRR-CWDA-2011-00044) is that in the PA models, DOE establishes a 100-meter perimeter around the SDF that it evaluates after a 100-year institutional control period. Since DOE owns the land, it is assumed that the site boundary will not change at any point during this institutional control period. DOE's current access point for any off-site member of the public is approximately 8 kilometers (5 miles) from the SDF site boundary. DOE states that this distance is sufficient for assuming that any dispersion of radionuclides from the SDF would be negligible to any MOP at the site boundary. In Section 7.2 of the PA, DOE provides a summary of all of the assumptions they believe to be conservative.

DOE also states (SRR-CWDA-2010-00033) that since the calculated dose as presented in the 2009 PA (Case A) is well below the limits specified by the 10 CFR 61 performance objectives, DOE believes that a qualitative analysis (in lieu of any additional rigorous quantitative analysis) is reasonable for demonstrating ALARA.

2.12.2 NRC Evaluation – ALARA Analysis

DOE's discussion of its ALARA analysis (2009 PA; Section 5.7) provides several examples of technical issues that DOE considered. In response to comments in RAI-2009-01 (NRC, 2010b) and RAI-2009-02 (NRC, 2010i), DOE provides further description of technical, social, economic, and public policy considerations taken in its ALARA analysis (SRR-CWDA-2010-00033 and SRR-CWDA-2011-00044).

As discussed in more detail in Chapter 4, NRC staff believes that DOE's radiation protection program limits releases to the environment and subsequent dose to members of the public while the site remains under DOE control. The NRC staff also agrees that while DOE maintains control of the site, the potential dose to a member of the public is limited due to the distance from SDF to the site boundary. The staff also agrees that, in the near-term, improvements made to the disposal unit design reduce the potential release to the environment. However, the amount that these improvements affect long-term performance is not clear (Section 2.5). The NRC staff concludes that DOE has taken appropriate actions to ensure that near-term doses are ALARA.

In its second response to comment A-1 (SRR-CWDA-2011-00044), DOE states that a cost-benefit analysis is untimely due to ongoing operations and currently unavailable costs of new technology. However, NRC staff believes that changes to various aspects of the disposal process could be considered prior to completion of salt operations. In its second response to comment A-1 (SRR-CWDA-2011-00044), DOE's consideration of introducing Small Column Ion Exchange technology is an example of a reasonable consideration.

Also in its response to comment A-1 (SRR-CWDA-2011-00044), DOE states that consideration of many alternatives to the current disposal process was taken in DOE's 2001 EIS for salt processing alternatives, DOE-EIS-0082-S2. NRC staff agrees that this reference provides a good analysis of potentially dose limiting alternatives and considers aspects of alternatives not discussed in the 2009 PA. However, the NRC staff believes that potential design changes and technological advances are not sufficiently considered in the DOE's ALARA analysis. Alternatives that could be considered could include:

- Evaluation of practicality of additional Tc removal,
- Evaluation of practicality of adding a stronger Tc-reducing agent,
- Evaluation of practicality of improving waste form quality, and
- Evaluation of practicality of maintaining better quality control on water to cement ratios and curing temperatures (Section 2.6).

The NRC staff agrees with the concept that a less detailed ALARA analysis is required when the predicted doses are low, however, the staff believes that only the short-term aspects of DOE's ALARA demonstration are currently considered sufficient. As discussed in Section 2.1, the NRC staff does not believe that Case A, which DOE used as their compliance case, adequately represents the expected current and future conditions of the SDF. Additionally, the NRC staff believes that many of the assumptions DOE included in this case are non-conservative. As discussed in more detail in Section 2.13, the NRC staff does not agree with DOE's statement that the expected dose to an off-site member of the public is well below the 10 CFR 61 performance objectives. Because of NRC staff concerns regarding DOE demonstration of compliance with §61.41 (Section 2.13), the NRC staff has not made a conclusion regarding DOE's long-term ALARA demonstration for this performance objective.

2.13 Protection of the General Population from Releases of Radioactivity

2.13.1 DOE Dose Calculations

To evaluate the potential dose to a member of the public, DOE evaluated the dose to an adult assumed to live and extract groundwater at the point of maximum exposure beyond a 100 m (330 ft) buffer zone around the SDF. The point of greatest exposure was calculated by splitting the SDF into 12 sectors and evaluating the dose downgradient of the SDF in each sector (Figure 2.3-1). In each sector, DOE used the maximum groundwater concentration from any depth (i.e., from the Upper Three Runs upper or lower zones or the Gordon aquifer). Peak concentrations typically occurred in the Upper Three Runs lower zone. Modeled peak radionuclide concentrations in the Upper Three Runs aquifer were significantly greater than modeled concentrations in the Gordon aquifer because of (1) attenuation in the Gordon Confining Unit, (2) limited release to the Gordon aquifer, and (3) significant dilution in the Gordon aquifer. Because predicted concentrations decreased with increasing distance from the SDF in the UTR, the point of maximum exposure in each case was 100 m (330 ft) downgradient of the SDF, although the sector with the maximum exposure depended on the particular case being evaluated. Groundwater was assumed to be used for drinking water consumption, plant irrigation, and watering livestock. The nearby stream water was assumed to be used for fishing and swimming (Section 2.3.2). For most cases, the dominant pathways were drinking water ingestion, fish ingestion, and vegetable ingestion. Air concentrations also were considered but did not contribute appreciably to predicted doses (Section 2.9).

In Case A, which DOE considers the base case, the predicted peak dose was greatest in Sector B (0.014 mSv/yr [1.4 mrem/yr]) within 10,000 years of site closure and Sector I (0.031 mSv/yr [3.1 mrem/yr]) within 20,000 years of closure. The primary radionuclides contributing to the Sector B peak dose include Ra-226 (94%), and I-129 (4%), with other radionuclides contributing less than 1% each to the total predicted dose. For Section B, the dominant pathways for the 10,000 year peak dose include drinking water ingestion (49%), fish ingestion (22%), vegetable ingestion (22%) and others (6%). The peak dose in Sector I within 20,000 years of closure was primarily attributable to I-129 (87%) and Ra-226 (13%), with all other nuclides contributing less than 1% to the total dose. Pathway contributions for the Sector I peak dose within 20,000 years include drinking water ingestion (45%), fish ingestion (28%), vegetable ingestion (21%) and others (7%).

As previously discussed (Section 2.1), in response to an NRC request for a revised base case (NRC, 2010i; PA-8), DOE supplied Cases K, K1, and K2. The main differences between Case A and Cases K, K1, and K2, which are summarized in Table 2-1, include consideration of increased saltstone hydraulic conductivity, oxidation of saltstone proceeding from fractures, the assumption that the relative permeability in the saltstone and disposal unit concrete is always 1 (i.e., the level of saturation does not influence hydraulic conductivity or saturation is equal to 1), increased hydraulic conductivity and diffusivity of disposal unit concrete, reduced inventory of Ra-226 and its ancestors, and updated biosphere modeling. The only differences between Cases K, K1, and K2 are the coefficients used to model Tc sorption in oxidized and reduced saltstone (i.e., K_d values for Tc-99 in saltstone).

DOE Case K peak dose results were highest for Sector I. The peak dose within 10,000 years of site closure was 0.13 mSv/yr (13 mrem/yr). DOE predicted that a peak dose of 0.55 mSv/yr (55 mrem/yr) would occur within 20,000 years of site closure. Radionuclides contributing to the Case K Sector I 10,000 yr peak dose include I-129 (71%), Cs-135 (28%), and Tc-99 (1.6%), with all other radionuclides contributing less than 1% to the total dose. Pathway contributions to the peak dose within 10,000 years of closure include drinking water ingestion (49%), fish ingestion (40%), vegetable ingestion (8%) and others (3%). Of these pathways, dose from the fish ingestion pathway was primarily attributable to Cs-135 while the dose from the drinking water and vegetable ingestion pathways was primarily due to I-129. In contrast, the Case K peak dose within 20,000 years is dominated by Tc-99, with all other radionuclides contributing less than 0.4% of the peak dose. No description of the primary dose pathways was provided for the peak dose within 20,000 years for Case K. DOE projected the peak dose from Case K1, which uses different K_d values for Tc in saltstone (i.e., 500 mL/g in reducing saltstone and 0.8 mL/g oxidized saltstone for Case K1, as compared to 1,000 mL/g in reducing saltstone and 10 mL/g in oxidized saltstone in Case K), to be approximately 0.09 mSv/yr (90 mrem/yr) and to occur at approximately 12,900 years after site closure (Figure 2.13-1). DOE projected the peak dose from Case K2 (500 mL/g in reducing saltstone and 0.8 mL/g oxidized saltstone) to be about 0.80 mSv/yr (80 mrem/yr) and, like the peak dose from Case K1, to occur at approximately 12,900 years after closure.

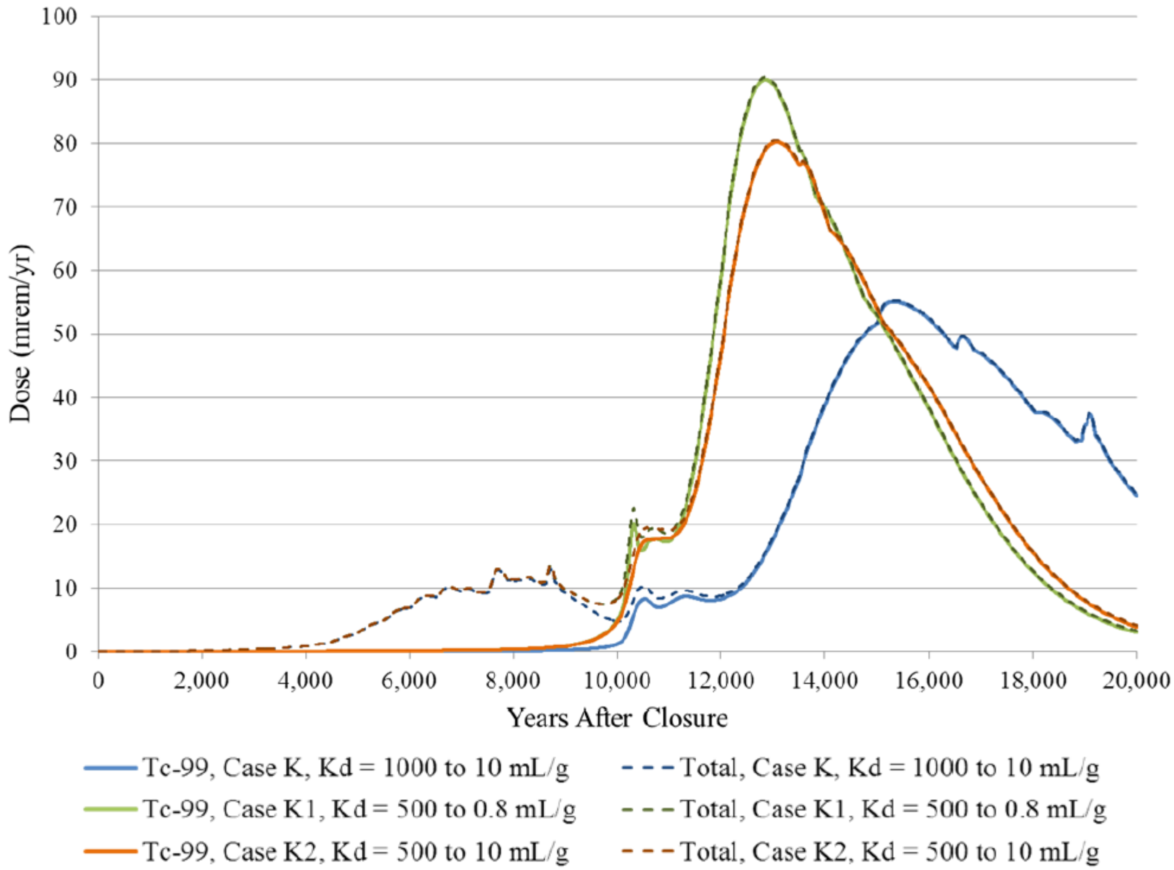

Figure 2.13-1: Doses from DOE Case K with variations in assumed K_ds for Tc in oxidized and reducing saltstone (SRR-CWDA-2011-00044; Figure SP-19.4).[24]

Peak doses from Cases K, K1, and K2 are dominated by release of Tc-99 in Sector I, with the majority of the dose attributable to releases from FDCs 7A-D[25] (Figure 2.8-8). Because the FDCs were all assumed to have an identical inventory in DOE's deterministic analysis, the contribution from these disposal units reflects their proximity to the 100 m (330 ft) boundary and far-field flow effects (Section 2.8). The projected peak doses from Ra-226 in Cases K, K1, and K1 are smaller than the projected peak dose from Ra-226 in Case A because the inventory of Ra-226 and its ancestors Th-230 and U-234 were lower in Case K, K1, and K2 than they are in the other cases. The projected peak doses from Tc-99 are greater in Cases K, K1, and K2 than in Case A because Cases K, K1, and K2 include the effects of oxidation of saltstone proceeding from fractures and because these cases allow significantly more water to flow through the saltstone matrix than Case A does (Section 2.7).

[24] To convert mrem/yr to mSv/yr, divide values on the vertical axis by 100.
[25] Disposal unit designations used in the TER correspond to designations used in the 2009 PA (as repeated in this TER; Figure 2.8-8), which may not correspond to current designations.

2.13.2 DOE Uncertainty and Sensitivity Analysis for Protection of the General Population

In the initial PA, DOE submitted deterministic PORFLOW™ analyses of Case A, four "alternative disposal unit configuration" sensitivity analyses (Cases B - E), and six additional deterministic sensitivity analyses, which DOE called "single parameter" sensitivity cases. These cases are described in more detail in Section 2.1.1. In brief, Case A assumes saltstone is unfractured with constant hydraulic conductivity (2.0×10^{-9} cm/s) and diffusivity (1×10^{-7} cm^2/s) for the 20,000 year evaluation period. Case B and C both model fast flow paths through the saltstone, disposal unit floor, and basemat, with more flow paths in Case C than in Case B. Case D is similar to Case A except that the sheet drain in the FDCs and Vault 4 is modeled as a capillary break (i.e., an impediment to flow) instead of as a fast flow path. Case E does not include fast flow paths, but assumes the saltstone has a high bulk hydraulic conductivity (1.7×10^{-3} cm/s) and a moisture characteristic curve representing degraded material. In addition to these alternative disposal unit configuration sensitivity analyses, DOE included six additional sensitivity analyses in the PA. DOE referred to these as "single parameter" sensitivity analyses (2009 PA; Section 5.6.6), although several parameters are actually modified in two of the six cases. These cases include evaluations of the effects of the closure cap, increased or decreased sulfate attack on the Vault 1 and 4 concrete (relative to Case A), oxidation of disposal unit concrete, increased saltstone hydraulic conductivity, and a synergistic case that includes decreased cap performance, chemically and hydraulically degraded disposal unit concrete, water flow through fractures in saltstone, and chemical oxidation of saltstone from fractures (Table 2-1).

In most of the cases in the PA, including Case A, the majority of peak doses DOE predicted to occur within 10,000 or 20,000 years of site closure occur in Sector B (Table 2-13), with the remaining peaks occurring in Sector I (Table 2-14). The majority of peak doses DOE predicted to occur within either 10,000 or 20,000 years in Sector B are dominated by Ra-226 (SRR-CWDA-2010-00033; Tables PA-1.1 and PA-1.3). Exceptions include the peak dose from Case A within 20,000 years, which occurs in Sector I and is split almost evenly between Ra-226 and I-129, and the peak doses in the Oxidized Concrete Case (within 10,000 years) the 10X Sulfate Attack Case (within 20,000 years), which occur in Sector B but are dominated by Tc-99. The peak dose DOE predicts within 20,000 years of closure in the "no cap" sensitivity analysis occurs in Sector I and is largely attributable to I-129. As previously discussed, the peak doses within either 10,000 or 20,000 years of closure in Cases K, K1, and K2 are attributable to Tc-99 in Sector I.

Table 2.13-1: Sector B Doses from DOE's Deterministic Analyses

Case	Sector B				Reference
	within 10,000 yrs		within 20,000 yrs		
	Dose (mrem/yr)	Time (yrs)	Dose (mrem/yr)	Time (yrs)	
Case A	**1.4**	10,000	2.9	15,080	SRR-CWDA-2009-00017 Table 5.5-1
Case B	**3.9** [1]	4,180	**6.0** [1]	15,740	SRR-CWDA-2010-00033 Table PA-1.1
Case C	**5.1** [1]	7,360	**5.5** [1]	15,420	SRR-CWDA-2010-00033 Table PA-1.1
Case C – PA-4 Case	**~5.1** [2]	7,360	**~7.4** [2]	20,000	SRR-CWDA-2011-00044 Figure PA-4.1
Case D	**1.3** [1]	9,800	**2.1**[1]	16,180	SRR-CWDA-2010-00033 Table PA-1.1
Case E	**56** [1]	9,400	**56** [1]	9,400	SRR-CWDA-2010-00033 Table PA-1.1
No Closure Cap (with credit for lower lateral drainage layer)	**2.1** [1]	9,800	3.5 [1]	15,060	SRR-CWDA-2009-00017 Table 5.6-17
10x Sulfate Attack	**1.7** [1]	10,000	**4.8** [1]	12,140	SRR-CWDA-2009-00017 Table 5.6-19
No Sulfate Attack	**0.8** [1]	10,000	**2.3** [1]	15,600	SRR-CWDA-2009-00017 Table 5.6-19
Oxidized Concrete	**0.4** [1]	860	**2.5** [1]	15,820	SRR-CWDA-2009-00017 Table 5.6-21
Increased Saltstone Hydraulic Conductivity	**5.4** [1]	9,860	**11** [1,3]	15,680	SRR-CWDA-2009-00017 Table 5.6-22
Synergistic Case (as reported in PA) [4]	**18** [1,3]	6,380	**28** [1,3]	15,760	SRR-CWDA-2009-00017 Table 5.6-20
Synergistic Case (as reported in SRR-CWDA-2011-00044) [4]	**~23** [1]	~6,500	**~36** [1]	~16,000	SRR-CWDA-2011-00044 Figure PA-9.1
Synergistic Case - VP-1	**~19**	not reported	not reported	not reported	SRR-CWDA-2010-00033 VP-1

Case	Sector B				Reference
	within 10,000 yrs		within 20,000 yrs		
	Dose (mrem/yr)	Time (yrs)	Dose (mrem/yr)	Time (yrs)	
Synergistic Case - PA-9	**~24** [2]	~8,500	**~37** [2]	~16,000	SRR-CWDA-2011-00044 Figure PA-9.1
Synergistic Case - Updated PA-9	**~15** [2,3]	~8,500	**~24** [2]	~16,000	SRR-CWDA-2011-00044 Figure PA-9.1
Case K	8.8	8,710	29[†]	15,258	SRR-CWDA-2011-00044 Table PA-8.11
Case K1	not reported		not reported		NA
Case K2	not reported		not reported		NA

Bold values indicate the peak dose from any sector in the indicated time frame (all peaks occur either in Sector B or Sector I)

To convert mrem/yr to mSv/yr, divide values in either dose column by 100.

[1] Dose only includes key radionuclides

[2] Dose includes more radionuclides than the key radionuclides, but does not include the entire list considered in Case A

[3] Value rounded to two significant digits by NRC staff

[4] Different values were reported for the unmodified synergistic case in the PA and second RAI response

Table 2.13-2: Sector I Dose Results from DOE's Deterministic Analyses

Case	Sector I				Reference
	within 10,000 yrs		within 20,000 yrs		
	Dose (mrem/yr)	Time (yrs)	Dose (mrem/yr)	Time (yrs)	
Case A	0.4	10,000	**3.1**	15,080	SRR-CWDA-2009-00017 Table 5.5-1
Case B	1.6 [1]	4,180	1.9 [1]	15,740	SRR-CWDA-2010-00033 Table PA-1.2
Case C	1.9 [1]	7,360	2.2 [1]	15,500	SRR-CWDA-2010-00033 Table PA-1.2
Case C – PA-4	~2.0 [2]	7,360	~2.4 [2]	20,000	SRR-CWDA-2011-00044 Figure PA-4.4
Case D	0.3 [1]	9,840	0.6 [1]	15,580	SRR-CWDA-2010-00033 Table PA-1.2
Case E	14 [1]	9,400	19 [1]	15,060	SRR-CWDA-2010-00033 Table PA-1.2
No Closure Cap (with credit for lower lateral drainage layer)	0.6 [1]	10,000	**3.6** [1]	15,060	SRR-CWDA-2009-00017 Table 5.6-17

Case	Sector I				Reference
	within 10,000 yrs		within 20,000 yrs		
	Dose (mrem/yr)	Time (yrs)	Dose (mrem/yr)	Time (yrs)	
10x Sulfate Attack	0.9 [1]	2,420	1.4 [1]	12,600	SRR-CWDA-2009-00017 Table 5.6-19
No Sulfate Attack	0.2 [1]	10,000	0.6 [1]	15,600	SRR-CWDA-2009-00017 Table 5.6-19
Oxidized Concrete	0.1 [1]	10,000	2.3 [1]	15,080	SRR-CWDA-2009-00017 Table 5.6-21
Increased Saltstone Hydraulic Conductivity	0.3 [1]	10,000	3.9 [1]	15,040	SRR-CWDA-2009-00017 Table 5.6-22
Synergistic Case (as reported in PA) [4]	2.1 [1]	10,000	5.4 [1]	17,320	SRR-CWDA-2009-00017 Table 5.6-20
Synergistic Case (as reported in SRR-CWDA-2011-00044) [4]	~7 [1]	~9,000	~14 [1]	~16,000	SRR-CWDA-2011-00044 Figure PA-9.4
Synergistic Case – VP-1	~3	not reported	not reported	not reported	SRR-CWDA-2010-00033 Text, VP-1
Synergistic Case - PA-9 Case	~8 [2]	~9,000	~15 [2,3]	~16,000	SRR-CWDA-2011-00044 Figure PA-9.4
Synergistic Case - Updated PA-9	~3 [2]	10,000	~5.5 [2]	~15,500	SRR-CWDA-2011-00044 Figure PA-9.4
Case K	**13**†	8,720	**55**†	15,348	SRR-CWDA-2011-00044 Table PA-8.11
Case K1	**~13**	~9,000	**~90**	~12,800	SRR-CWDA-2011-00044 Figure SP-19.4
Case K2	**~13**	~9,000	**~80**	~13,200	SRR-CWDA-2011-00044 Figure SP-19.4

Bold values indicate the peak dose from any sector in the indicated time frame (all peaks occur either in Sector B or Sector I).
To convert mrem/yr to mSv/yr, divide values in either dose column by 100.
[1] Dose only includes key radionuclides
[2] Dose includes more radionuclides than the key radionuclides, but does not include the entire list considered in Case A
[3] Value rounded to two significant digits by NRC staff
[4] Different values were reported for the unmodified synergistic case in the PA and second RAI response

To support its demonstration of compliance with the performance objective for protection of a member of the general public, DOE also submitted a probabilistic model that was intended to evaluate the weighted average of GoldSim® representations of Cases A through E (2009 PA; Table 5.6-3). However, because near-field flow results were incorporated from the deterministic model, DOE's probabilistic analysis did not allow variation in parameters affecting flow in the near field (i.e., the saltstone, disposal units, or backfill in the unsaturated zone) within individual cases.

DOE's original probabilistic analysis included sensitivity results from Case A and Case C. The Case A probabilistic analysis indicated the peak dose within 10,000 years was most sensitive to the sorption coefficient for Ra-226 in sandy soil, followed by the local fraction of vegetable consumption and the saturated zone thickness (2009 PA; Table 5.6-14). The Case A peak dose within 20,000 years of closure was sensitive to the K_d for I-129 in reducing middle aged concrete, vegetable production yield, and the number of pore volumes required for concrete to transition to middle age (2009 PA; Table 5.6-14). Results for Case C, which reflected the effect of water flow through fast pathways through the saltstone, floor, and basemat, showed the dose within 10,000 years is sensitive to the K_d for Pu in clayey soil, the unsaturated zone thickness below the FDCs, followed by the K_d for Pu in sandy soil. The peak dose within 20,000 years was sensitive to the same parameters, but more sensitive to the K_d for Pu in sandy soil than clayey soil (2009 PA; Table 5.6-15).

In response to an NRC RAI (NRC, 2010i) DOE provided sensitivity results for Case E, which is the only case reflecting hydraulic degradation of bulk saltstone that was included in DOE's probabilistic analysis. The Case E probabilistic results had a peak of the mean value within 10,000 years of 1.32 mSv/yr (132 mrem/yr) in Sector B (SRR-CWDA-2010-00033), in comparison to the deterministic results for Case E, which have a projected peak of 0.56 mSv/yr (56 mrem/yr) within either 10,000 or 20,000 years of closure. The probabilistic results indicate that the Case E peak dose within 10,000 years is sensitive to the K_d for Ra in sandy soil, the unsaturated zone thickness, and the local fraction of vegetable consumption. Consistent with the sensitivity of dose to the sandy soil K_d for Ra, the peak dose in both the deterministic and probabilistic analyses is primarily attributable to Ra-226 (SRR-CWDA-2010-00033; PA-1 and PA-2).

2.13.3 NRC Evaluation – Evaluation of Performance Assessment Results for Protection of the General Public

In support of its compliance demonstration, DOE submitted a deterministic base case, several deterministic sensitivity analyses, and a probabilistic uncertainty and sensitivity analysis. Because of concerns about the design and implementation of the probabilistic model (Section 2.11), the NRC staff did not rely on the probabilistic model in its compliance evaluation. Thus the NRC evaluation of uncertainty in dose and the sensitivity of dose to input assumptions included three main factors: (1) an evaluation of the results of DOE's deterministic sensitivity cases, (2) an evaluation of intermediate model results from DOE's deterministic sensitivity analyses, and (3) independent sensitivity analyses focused on Cases K, K1, and K2. The NRC staff conducted independent sensitivity analyses to evaluate and attempt to isolate the effects of optimistic and pessimistic assumptions used in DOE's analyses, and to investigate the effects of unexpected barrier performance indicated by intermediate results. In addition, the NRC staff conducted sensitivity analyses focused on Cases K, K1, and K2 because the NRC staff relied on these cases heavily in its review and these cases were supplied by DOE without accompanying sensitivity or uncertainty analyses. This section includes an evaluation of DOE's sensitivity and uncertainty analyses, an evaluation of modeled barrier performance, a description of the NRC staff's sensitivity analyses, and a comparison of the dose results to the performance objectives.

2.13.3.1 Consideration of DOE's Uncertainty and Sensitivity Analyses

Use of a set of deterministic cases, including deterministic sensitivity analyses, supported by a probabilistic uncertainty model, which DOE refers to as a "hybrid" approach, could provide useful complementary information about the uncertainty in predicted doses and the sensitivity of predicted dose to various input parameters. However, the utility of a model depends on its scope and implementation. In DOE's 2009 SDF PA analysis, near-field flow was outside of the scope of the probabilistic model and was, instead, directly incorporated from deterministic PORFLOW™ results. As a result, the sensitivity analysis conducted with the probabilistic model excluded several of the near field parameters that often have a significant impact on dose (e.g., hydraulic conductivity in saltstone or disposal unit concrete, diffusivity in saltstone).

The sensitivity of the probabilistic (i.e., GoldSim® model) Case A and Case E to parameters affecting the dose from Ra-226 (i.e., Ra sorption coefficients and fraction of local vegetable uptake) is consistent with the importance of Ra-226 to the peak dose in these cases. However, the importance of Ra to dose in these cases, like all cases included in the original PA, is likely to be overstated because of potentially conservative assumptions about the inventory of Ra-226 and its ancestors that were used in these cases but revised in Case K. Thus these results are of interest primarily in that they indicate that the uncertainty in inventory may change the projected dose driver and associated parameters to which dose is most sensitive.

Beyond the way deterministic near-field flow results were used, the NRC staff has concerns about other aspects of the approach taken in the probabilistic model. For example, DOE did not provide model support (e.g., documented expert elicitation) for the probabilities assigned to each of the sensitivity cases considered in the probabilistic analysis (NRC, 2010i; PA-11). In addition, in the probabilistic model, DOE modifies the aquifer concentrations by the probability of well completion in each aquifer. Although this approach is appropriate if DOE uses the probabilistic model solely for risk information, as discussed in the context of the review for the FTF (NRC, 2011m), the NRC staff believes aquifer concentrations should not be modified by the probability of well completion if the dose results are compared to a performance objective. For comparison to a performance objective, it is most appropriate to locate the point of maximum exposure in the highest-concentration aquifer unit that could support the groundwater dependent pathways evaluated in the biosphere modeling.

In addition to concerns about the approach taken in the probabilistic model, the NRC staff has specific concerns about the implementation of the DOE probabilistic analysis (Section 2.11). These concerns include, but are not limited to, the exclusion of key features of Case C when the case selection is run probabilistically, the modeled flows in the unsaturated zone, the area assumed for Vault 4, and the methodology used to estimate the ingrowth of radionuclides. For example, the Case C sensitivity to Pu sorption in the environment may be due to a modeling artifact that overestimates the long-term ingrowth of Pu-239 (Section 2.11.4.3). Because of these concerns with the approach to the probabilistic modeling and the specific implementation of the probabilistic model, the NRC staff did not rely on DOE's probabilistic modeling results in its compliance evaluation.

Deterministic PORFLOW™ Cases B through E, which DOE refers to as alternate disposal unit configuration cases, essentially vary assumptions about flow through saltstone and the disposal units. Whereas Cases B, C, and D vary assumptions about flow through potential fractures in the saltstone and disposal units, Case E is used to evaluate the effects of elevated flow through the saltstone matrix (i.e., increased hydraulic conductivity of the bulk saltstone rather than discrete fractures). Of these cases, the case with the largest projected peak dose (Tables 2-13 and 2-14) is Case E (i.e., peak dose within 10,000 years of 0.56 mSv/yr [56 mrem/yr]), which allows the most flow through the saltstone matrix. Cases B and C, which each allow flow through fractures in the saltstone and the disposal unit floors, have more modestly increased doses compared to Case A, which represents saltsone as an uncracked monolith for the entire performance period. That is, the predicted peak dose within 10,000 years from Case B is 0.039 mSv/yr [3.9 mrem/yr] and the predicted peak dose from Case C in 10,000 years is 0.051 mSv/yr [5.1 mrem/yr]. Case D, which DOE based on the assumption that the sheet drain between the saltstone and disposal unit wall in Vault 4 and the FDCs would act as a capillary break (i.e., a barrier to flow), predicted a slightly lower peak dose within 10,000 years than Case A (i.e., 0.013 mSv/yr [1.3 mrem/yr]) (Tables 2-13 and 2-14). While the absolute values of the results of these case are difficult to interpret because they are affected by overly optimistic assumptions (e.g., unrealistic moisture characteristic curves (NRC, 2010b; SP-3 and SP-4), limited saltstone fracturing (NRC, 2010b; SP-1), and exceptional performance of the lower lateral drainage layer for thousands of years (NRC, 2010b)), the relative magnitude of the results indicates the importance of flow through the saltstone matrix to the predicted dose.

Comparison of the results of Case E and the Increased Hydraulic Conductivity Case (2009 PA; Section 5.6.6.7) with the Case A results helps to illustrate the effects of flow through the saltstone matrix. Both Case E and the Increased Hydraulic Conductivity Case represent increased hydraulic conductivity of the saltstone matrix with all other aspects of the disposal facility performing as designed. Although saltstone in Case E has a nominal saturated hydraulic conductivity of $1.7x10^{-3}$ cm/s, the flow is actually limited by infiltration to a flow of approximately $1x10^{-6}$ cm/s. The Increased Hydraulic Conductivity Case is essentially the same as Cases A and E, except that the saturated hydraulic conductivity of saltstone is assumed to be $1x10^{-7}$ cm/s. In all three cases, the saturated hydraulic conductivity is assumed to remain constant during the analysis period (i.e., 20,000 years). These differences in saltstone hydraulic conductivity result in an increase in the peak dose within 10,000 years of closure from 0.014 mSv/yr (1.4 mrem/yr) in Case A ($K_h = 2x10^{-9}$ cm/s) to 0.054 mSv/yr (5.4 mrem/yr) in the increased hydraulic conductivity case ($K_h = 1x10^{-7}$ cm/s) and 0.56 mSv/yr (56 mrem/yr) in Case E (flow limited by infiltration at an effective K_h of $1x10^{-6}$ cm/s).

DOE used the Sulfate X10 Case and the No Sulfate Attack case to explore the effects of different degrees of hydraulic degradation of the disposal unit roofs, floors, and walls. The primary effect of increased sulfate attack was to shorten the time in which the disposal unit concrete becomes hydraulically equivalent to the backfill soil. In the 10X sulfate attack case, the predicted failure time for the FDC roof, floor, and walls and the predicted failure time for the Vault 4 walls moved from beyond 10,000 years to within 10,000 years after closure (i.e., from 40,000 years to 5,000 years for the FDC floor, from 40,000 years to 7,000 years for the FDC roof, from 18,000 years to 3,000 years for the FDC wall, and from 10,000 to 3,000 years for the Vault 4 roof).

Predicted doses from the No Sulfate Attack and 10X Sulfate Attack cases are greatest in Sector B (Table 2.14). As in Case A, peak doses within 10,000 years were due primarily to Ra-226 (85%) and I-129 (9.7%), although, unlike in Case A, the peak from the 10X Sulfate Attack case within 10,000 years of closure did have a non-negligible contribution from Tc-99 (3.2%). In further contrast to Case A, peak doses within 20,000 years in the 10X sulfate attack case have a significant contribution from Tc-99 (70%) with the remaining dose primarily attributable to Ra-226 (24%) and I-129 (4%) (SRR-CWDA-2010-00033; Table PA-1.3). DOE's predicted peak doses from the 10X Sulfate Attack Case within 10,000 and 20,000 years of site closure are only slightly greater than the corresponding peaks in Case A (i.e., 21% greater and 55% greater, respectively). This result indicates that the dose is not very sensitive to the hydraulic performance of the disposal units when the remainder of the system performs as designed. Hydraulic performance of the disposal units may have a greater effect on dose if flow is not limited by the hydraulic performance of the cap and saltstone waste form.

To test the effects of simultaneous degradation of multiple barriers, DOE developed the Synergistic Case (Table 2.1). Key features include early hydraulic degradation of the cap and disposal unit concrete, early oxidation of the Vault 1 and 4 walls and FDC concrete, and fractures through saltstone. Saltstone fractures are assumed to occur at approximately 1.5 m (5.0 ft) intervals[26] at the time of closure and not to increase during the performance period. The fractures allow water flow through saltstone. The fractures also allow saltstone oxidation to proceed from the fracture faces. In the PA, DOE reported a predicted peak dose of 0.18 mSv/yr (18 mrem/yr) within 10,000 years of closure and a predicted peak of 0.28 mSv/yr (28 mrem/yr) within 20,000 years. In the second RAI response (SRR-CWDA-2011-00044 Figure PA-9.1), DOE reported different doses for the unmodified synergistic case: approximately 0.23 mSv/yr (23 mrem/yr) within 10,000 years and approximately 0.36 mSv/yr (36 mrem/yr) within 20,000 years. As in Case A, peak doses within either 10,000 years or 20,000 years of closure are predicted to be due primarily to Ra-226 with a small contribution from I-129 (SRR-CWDA-2010-00033; Table PA-1.3).

In response to the NRC RAIs (NRC, 2010b, i), DOE provided additional sensitivity analyses based on the Synergistic Case (Table 2.1). In response to NRC concerns about the effects of unrealistically optimistic moisture characteristic curves, DOE recalculated the fluxes of Ra-226, I-129, and Tc-99 to the unsaturated zone using an assumed relative permeability of one in saltstone (SRR-CWDA-2010-00033, comment VP-2). The recalculation indicated that the Synergistic Case results were not sensitive to the difference between the moisture characteristic curves and a fixed relative permeability of one in saltstone. In response to an NRC concern about the limited set of radionuclides analyzed in the Synergistic Case (i.e., an initial inventory assumed for I-129, Np-237, Pu-238, Tc-99, Th-230, U-234, and U-238), DOE re-ran the Synergistic Case PORFLOW™ model with thirteen additional radionuclides in the initial

[26] Fracture spacing is described as 1.5 m (5 ft) intervals in the *Review Team Report on the Performance Assessment for the Saltstone Disposal Facility at the Savannah River Site*. However, fractures appear to occur at 0.75 m (2.5 ft) interval in the PORFLOW™ model file.

inventory (i.e., Am-241, Am-243, Cm-244, Cm- 245, Cs-135, Nb-93m, Pu-239, Pu-240, Pu-241, Pu-244, Th-229, U-233, and U-235) (SRR-CWDA-2011-00044, PA-9). The results of this analysis showed the additional radionuclides did not contribute significantly to dose in either Sector B, Sector I, or the adjacent sectors. In addition, for consistency with RAI responses regarding biosphere parameters, DOE also ran the Synergistic Case (including the additional radionuclides) with updated biosphere parameters (SRR-CWDA-2011-00044, PA-9). This modification resulted in slightly lower predicted peak doses within 10,000 and 20,000 years of closure, largely due to a reduction of the water-to-fish transfer factor for Ra-226 and an increase in the garden productivity factor.

The additional sensitivity cases based on the Synergistic Case (i.e., saltstone relative permeability of one, additional radionuclides, and updated biosphere parameters) aid in the interpretation of the Synergistic Case results. However, additional factors that could lead to underestimates of radionuclide release make it difficult to determine if the Synergistic Case is likely to over- or under-predict the peak dose. For example, the assumption that the hydraulic conductivity of the unfractured saltstone remains constant at the value used in DOE's Case A ($2x10^{-9}$ cm/s) appears to be unrealistically optimistic in light of recent experimental results. These experimental results show hydraulic conductivity values in both laboratory samples prepared under conditions relevant to saltstone as well as core samples taken from emplaced saltstone (Section 2.6). Furthermore, even if a hydraulic conductivity of $2x10^{-9}$ cm/s were achieved in field-emplaced saltstone, it does not appear to be realistic that the hydraulic conductivity of the saltstone matrix would not degrade within 10,000 years. Although fracturing does reflect one degradation mode of the saltstone waste form, the result of the modeled low hydraulic conductivity of the saltstone matrix is to route flow almost entirely through the fractures. In addition, the selection of hydraulic properties similar to soil for the disposal unit walls, while it appears to be conservative, serves to route additional flow away from the saltstone (e.g., the drop in flow through the saltstone at 500 years is attributable to hydraulic degradation of the disposal unit walls) (Figures 2.13-1 and 2.13-2). As a result of the assumed low hydraulic conductivity of the saltstone and high hydraulic conductivity of the fractures and walls, even less water flows through the saltstone matrix in Vault 4 in the Synergistic Case than flows through the saltstone matrix in Case A (Figure 2.13-2), while the flows are approximately equal in the FDCs (Figure 2.13-3).

Flow through the matrix is expected to be important to the predicted dose because flow through the matrix exposes significantly more radionuclide inventory to infiltrating water than fracture flow. The sensitivity of the predicted peak dose from the SDF to the hydraulic conductivity of the saltstone matrix is shown clearly by the results of Case E and the Increased Hydraulic Conductivity Case, as previously discussed in this section. In particular, the Increased Hydraulic Conductivity Case, which used a saturated hydraulic conductivity in saltstone similar to values measured for field saltstone samples ($1x10^{-7}$ cm/s), has a predicted peak dose within 10,000 years approximately 4 times greater than the predicted peak dose in Case A.

The NRC staff understands that sensitivity analyses are designed to test the response of the system to various changes in input, and do not necessarily yield peak doses that are appropriate to compare to compliance limits. However, if major degradation modes are considered and if dose predictions do not exceed the relevant limits, alternate cases based on enhanced degradation of multiple barriers can provide added confidence in a compliance determination. In this case, the Synergistic Case results approach, but do not exceed the compliance limits. However, because of the apparent sensitivity of dose to the saturated hydraulic conductivity of the saltstone matrix, it appears that a Synergistic Case run with a more realistic value of this parameter would yield peak dose predictions that exceed the relevant dose limits. In addition, more realistic values of the sorption coefficient for Tc in reducing and oxidizing saltstone and disposal unit concrete are likely to further increase the predicted release of Tc (Section 2.7). Thus, because of the flow-limiting effects of the low hydraulic conductivity in the saltstone matrix, the flow-diverting effects of the disposal unit walls, and the Tc release-limiting effects of the high sorption coefficients chosen for Tc, the NRC staff cannot conclude that the results of the Synergistic Case necessarily bound the predicted dose from the SDF.

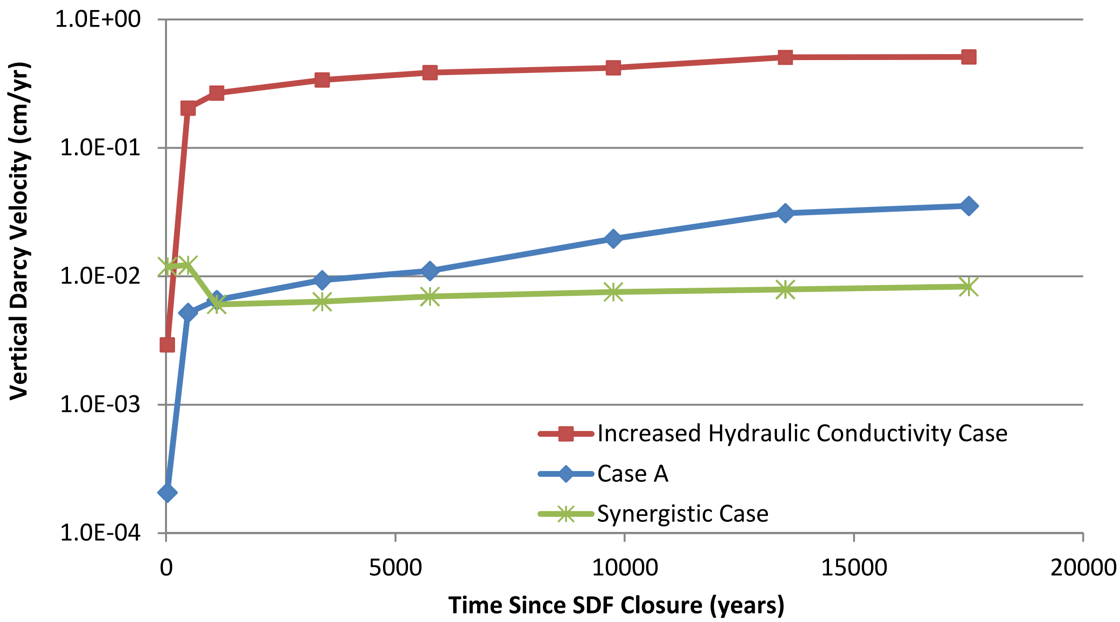

Figure 2.13-2 Vault 4 vertical Darcy velocity through the saltstone matrix (excludes fractures) predicted with the DOE PORFLOW™ model for Case A, the Synergistic Case, and the Increased Hydraulic Conductivity Case (Data taken from the STAT.out files [NRC, 2010g]).

The NRC staff concludes that the types of deterministic sensitivity analyses performed by DOE, including those sensitivity analyses performed in response to NRC's RAIs, were reasonable in the technical areas they addressed, but were too limited in scope. In general, deterministic sensitivity analyses have a limited capacity to evaluate potentially unexpected effects from parameters that are not explicitly varied in the deterministic test. For example, there is uncertainty in the potential dose from Se-79 due to concerns about the sorption coefficients used for Se in cementitious materials and site soil (Section 2.7). Because the potential effect of this uncertainty has not been captured in the current sensitivity analysis, it has been identified as an issue to be monitored (Table A-1). These types of effects often may be captured with a more comprehensive probabilistic uncertainty analysis, such as the GoldSim® analysis submitted by DOE, if correctly implemented. As previously discussed, the NRC staff was unable to rely on DOE's probabilistic GoldSim® analysis as part of its compliance demonstration (Sections 2.11.4.2 and 2.11.4.3).

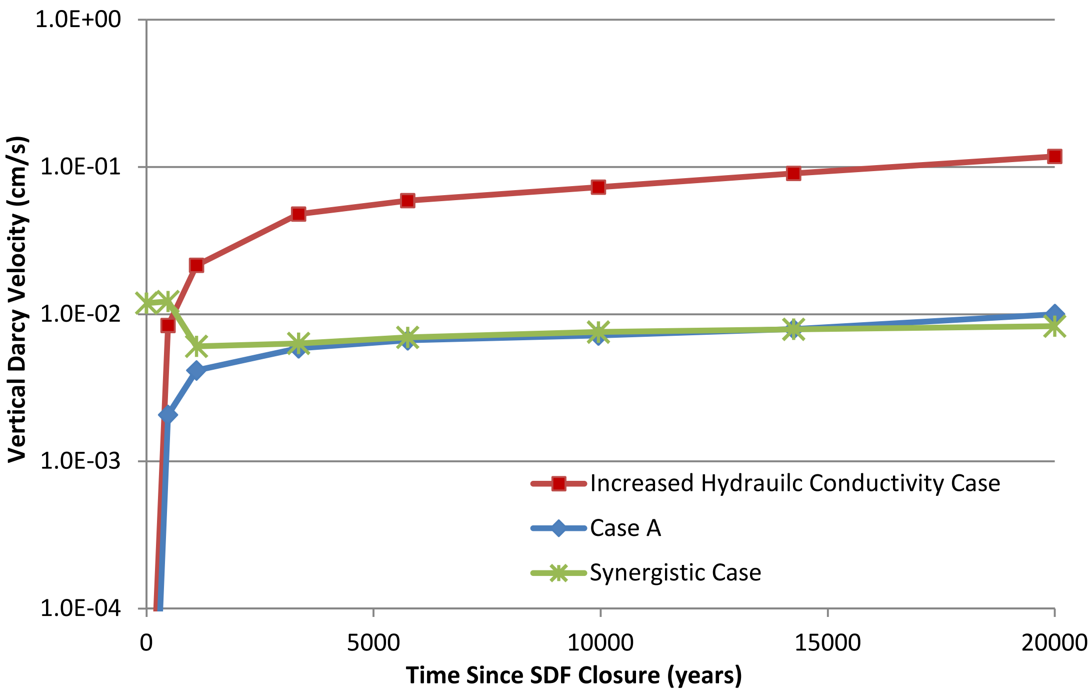

Figure 2.13-3 FDC vertical Darcy velocity through the saltstone matrix (excludes fractures) predicted with the DOE PORFLOW™ model for Case A, the Synergistic Case, and the Increased Hydraulic Conductivity Case (data taken from the STAT.out files [NRC, 2010g]).

Finally, both deterministic and probabilistic models may fail to capture conceptual model uncertainty if it is not specifically considered and incorporated into an analytical model. During the SDF review, the NRC staff identified possible alternative conceptual models in which significant oxidation of saltstone occurred prior to significant release of water from the disposal units. For example, if oxygen-bearing water infiltrates into the FDCs but is retained by the HDPE layer on the outside of these units, a pulsed release of Tc could occur if the HDPE experienced a period of rapid deterioration. Similarly, if Tc is oxidized by the infiltration of oxygen-bearing water from the top of a disposal unit and is re-reduced and concentrated as it moves downward (e.g., in the manner of a uranium roll-front), oxidation of the bottom layer of the disposal unit could cause a release greater than the releases considered in current conceptual model. Because of the potentially large doses that could be predicted with this conceptual model, consideration of this possibility has been identified as a PA maintenance item (Table A-2).

2.13.3.2 Barrier Contribution to Dose Reduction

PA Cases

DOE's initial sensitivity analysis demonstrates that the disposal system is designed with multiple barriers to radionuclide release. The primary barriers either act to limit water flow through the waste form or to control the water chemistry to slow radionuclide release from the waste form or through the disposal unit floor and walls. By design, primary barriers to water flow are (1) the closure cap and the engineered layers above each disposal unit (including the lower lateral drainage layer), (2) disposal unit roof, floor, and walls and (3) the intended low permeability of the saltstone waste form. DOE's initial sensitivity analyses tested the effects of (1) the upper layers of the closure cap, (2) permeability of the disposal unit roof, floors, and walls, (3) permeability of the saltstone, and (4) the chemical condition of Vault 1 and 4 walls and floor.

Several of the initial sensitivity analyses demonstrate that the degradation of a single barrier has little effect on SDF performance because of the designed performance of other barriers. For example, DOE's "No Cap" sensitivity analysis tested the performance of the SDF if the upper layers of the closure cap (i.e., the composite hydraulic barrier, the upper lateral drainage layer[27], and the erosion control layer) did not limit flow, resulting in infiltration of 41.8 cm/yr (16.4 in/yr) through these layers. The scenario also assumes that the engineered layers on top of the disposal units (i.e., the geotextile filter fabric and lower lateral drainage layer for Vaults 1 and 4, and the geotextile filter fabric, lower lateral drainage layer, HDPE geomembrane and geosynthetic clay liner for the FDCs) the disposal unit concrete, and the saltstone use Case A hydraulic performance. DOE's analysis indicates that there is little effect of removing the credit for the upper layers of the closure cap if the engineered layers above each disposal unit and other features of the facility perform as designed (Table 2-13 and 2-14).

The NRC staff evaluation of DOE PORFLOW™ intermediate outputs indicates the significant barrier to flow DOE expects to achieve with the combination of a high hydraulic conductivity lower lateral drainage layer and low hydraulic conductivity disposal unit roof (for Vaults 1 and 4) or low hydraulic conductivity HDPE layer, geosynthetic clay liner, and roof (for the FDCs). This contrast of low and high hydraulic conductivity layers is intended to divert water around the disposal units. For example, in Case A, 99.9% of the water infiltrating through the cap is diverted around the Vault 4 and FDC roofs up to 10,000 years after SDF closure (Figure 2.13-4). In the "no cap" scenario, 8,000 years after closure these layers divert 99.7% of the water around an FDC, while 99.98% is diverted around Vault 4. Although there is a significant difference in the infiltration through the upper cap layers in Case A and "no cap" analyses, there is not a significant difference in the predicted peak dose from each scenario (i.e., 0.14 mSv/yr [1.4 mrem/yr] in Case A and 0.021 mSv/yr [2.1 mrem/yr] in the "no cap" scenario within 10,000 years and 0.031 mSv/yr [3.1 mrem/yr] in Case A as compared to 0.036 mSv/yr [3.6 mrem/yr] in the "no cap" case within 20,000 years). This dose result, in combination with the intermediate model results illustrating water diversion in the lower lateral

[27] Although the 2009 PA (p. 541) indicates that the "no cap" scenario models performance as if neither of the drainage layers is present, the DOE PORFLOW™ file used to model the scenario includes credit for the lower lateral drainage layer.

drainage layer, demonstrates the importance of the large contrast between the modeled hydraulic conductivity of the lower lateral drainage layer and the layers it overlies above each disposal unit (i.e., the roof for Vault 1 and 4, the HDPE, geosynthetic clay liner, and roof for the FDCs).

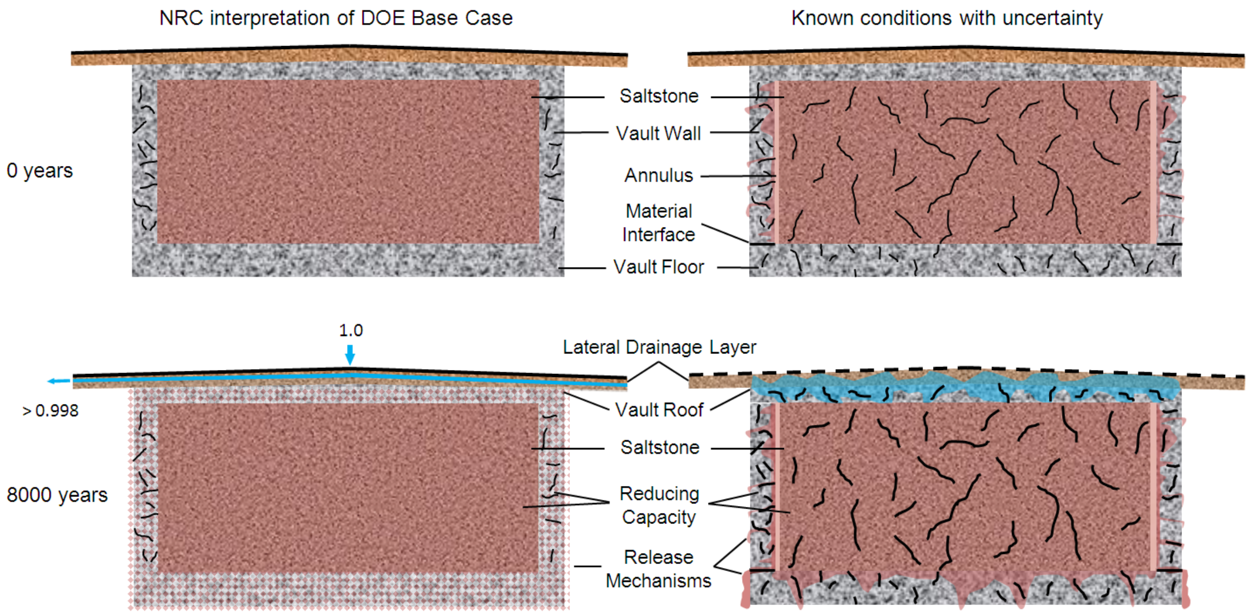

Figure 2.13-4: Water shedding around Vault 4 in DOE Case A at 0 and 8,000 years after site closure (left top and bottom) as compared to an NRC interpretation of know conditions (top right) or potential future conditions considering uncertainty (bottom right)

As discussed in the previous section, DOE developed the Synergistic Case to evaluate the impact of simultaneously changing multiple material parameters to account for several potential increased degradation mechanisms as compared to Case A. The PA describes this case as overly pessimistic. However, in the Synergistic Case, although the saltstone is modeled as fractured, the hydraulic conductivity of the saltstone matrix is the same as it is in Case A (i.e., a static value of $2x10^{-9}$ cm/s). Water moves primarily through fractures in the saltstone, and the flow through the saltstone matrix is very similar to the flow through the saltstone matrix in the Case A (Figures 2.13-2 and 2.13-4). Comparison with Case E and the Increased Hydraulic Conductivity Case suggests that a case with higher saltstone matrix hydraulic conductivity would predict a larger dose, presumably because water flowing through saltstone would come into contact with more of the saltstone inventory of radionuclides. This comparison is of interest particularly because DOE results suggest that the hydraulic conductivity of field-emplaced saltstone may be considerably greater than DOE modeled in the Synergistic Case (Section 2.6).

In addition to these tests of various hydraulic barriers, DOE used the Oxidized Concrete Case to evaluate the effects of earlier oxidation of Vault 1 and 4 concrete. Specifically, the Oxidized Concrete case is based on the assumption that the Vault 1 and 4 floor and walls are completely oxidized at closure. The FDC floor and walls are modeled just as they are in Case A. In the Oxidized Concrete Case, the peak doses in Sectors B and I within either 10,000 or 20,000 years of site closure decrease slightly (Tables 2-13 and 2-14). This decrease does not completely clarify the importance of the Vault 1 and 4 floor and walls as a chemical barrier, because it is attributable to a somewhat artificial difference in the K_d values for Ra in oxidizing and reducing saltstone (Section 2.7). The more significant conclusion from the Oxidized Concrete case in the context of the barrier analysis may be that complete oxidation of the Vault 1 and 4 walls is predicted to generate a small dose from Tc-99 at approximately 900 years after SDF closure in Sector B due to the assumed initial inventory in the walls (rather than the inventory in saltstone itself) (2009 PA; Figure 5.6-84). While the initial inventory in the Vault 1 and 4 walls is based on the apparently conservative assumption that the pore spaces in the walls are completely saturated with pore fluid containing the same concentrations of radionuclides as in saltstone, the timing of the peak is of interest. Because the transport in the saturated zone is expected to take only tens of years, the peak at 900 years is expected to be due primarily to the time needed for release from Tc from the oxidized disposal unit floor and walls (K_d of 0.8 mL/g) under Case A flow conditions during the first 1,000 years following closure (i.e., with significant cap performance).

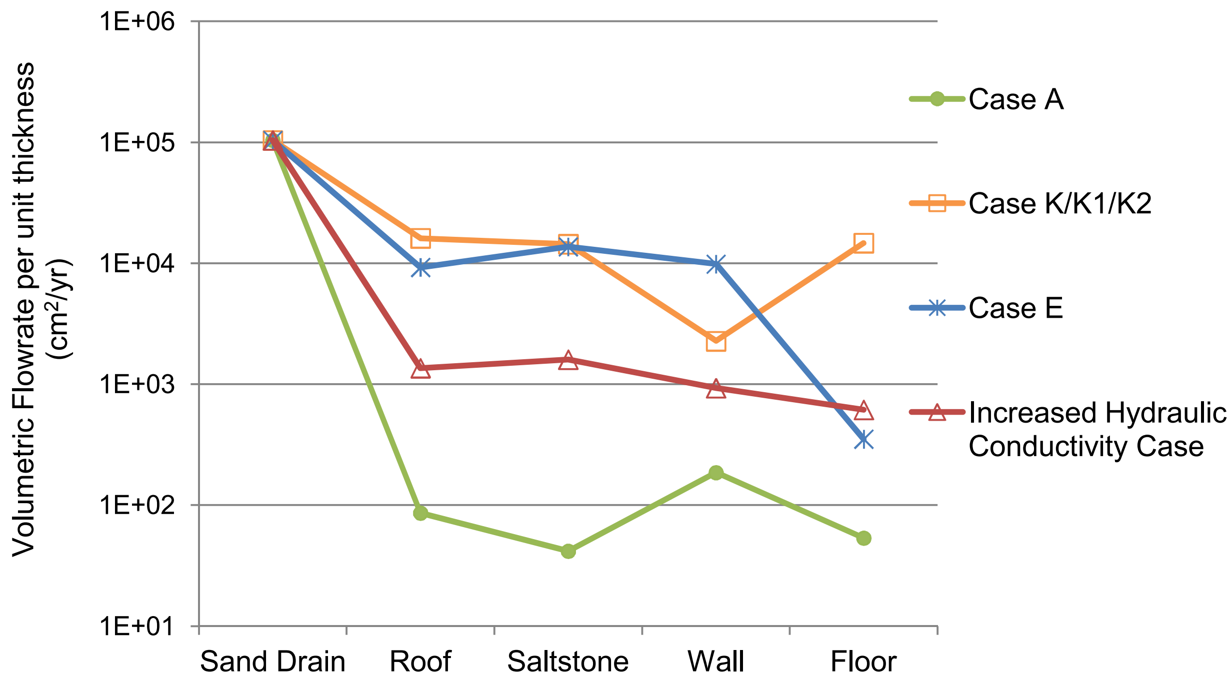

Figure 2.13-5: Modeled volumetric flow per unit thickness through Vault 4 from 5,500 to 6,000 years after SFD closure predicted by the DOE PORFLOW™ model for Case A, Cases K/K1/K2, and select DOE sensitivity analyses (data taken from the STAT.out files [NRC, 2010g]).

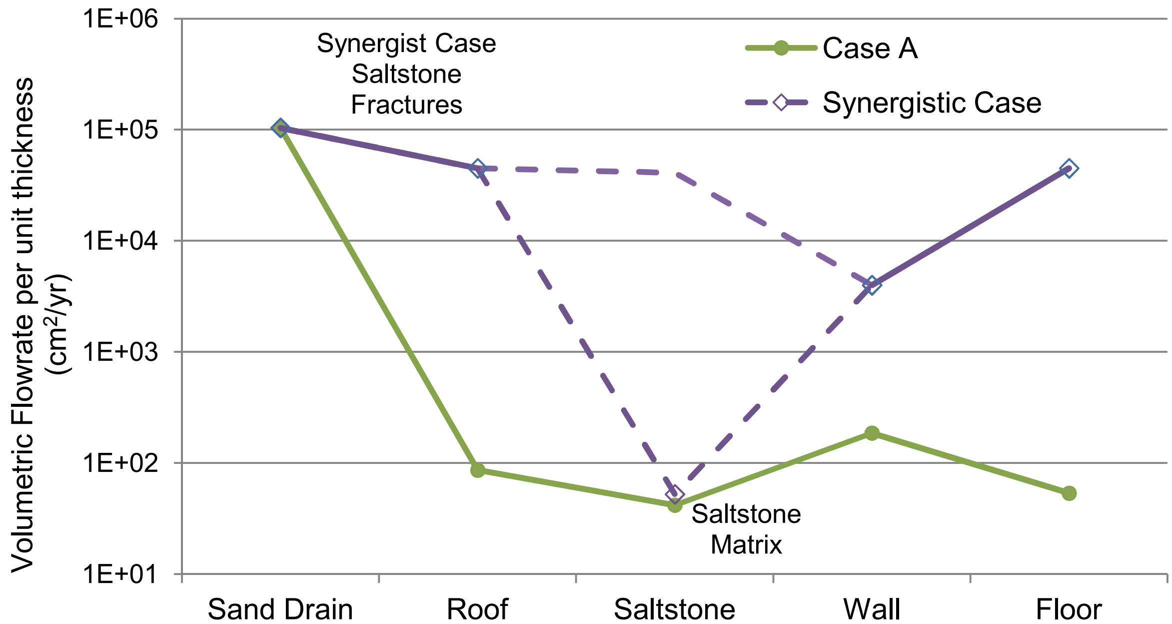

Figure 2.13-6: Modeled volumetric flow per unit thickness through Vault 4 from 5,500 to 6,000 years after SFD closure predicted by the DOE PORFLOW™ model for the synergistic case (data from Case A repeated from Figure 2.13-3 for comparison) (data taken from the STAT.out files [NRC, 2010g]).

After release from the disposal units, the final dose-reducing barrier is dilution and dispersion in the saturated zone. As described in Section 2.8, the modeled far-field performance potentially overestimates dilution and dispersion and may significantly limit estimated cumulative impacts from the facility. In the southern portion of the SDF, where the peak dose occurs in Case A and in most of the sensitivity analyses discussed in this section, most plumes tend to spread away from one another both horizontally and vertically, limiting the cumulative effect of multiple sources. Changes to key parameters and improved local SDF/PORFLOW™ model calibration could result in significantly lower levels of modeled dilution and dispersion, which could result in larger predicted doses. For these reasons, the NRC staff concludes it is more likely that predicted doses would be increased, rather than reduced, by changes to the SDF far-field modeling.

Cases K, K1, and K2

Because of very different assumptions about saltstone degradation and oxidation and Tc sorption, Cases K, K1, and K2 emphasize different barriers than the cases initially evaluated in the DOE PA.

As previously discussed (Section 2.13.3.1), the magnitude and timing of the Case K peak dose is sensitive to assumptions about saltstone fracturing. In addition, the magnitude and timing of Tc release also are sensitive to assumptions about disposal unit performance. DOE's Cases K, K1, and K2, display an unexpected behavior of reconcentration of Tc in the disposal unit floor and walls. By approximately 13,000 years after site closure over 90% of the Tc originally in the saltstone has been released from the saltstone and reconcentrated in the disposal unit concrete (Figure 2.13-4). Because the disposal unit floor and walls have a much lower volume than the saltstone, this process, as modeled, would result in a 13-fold increase in the original Tc concentration in saltstone. This unexpected behavior occurs because the disposal unit floors and walls are modeled with much higher K_d values than saltstone as saltstone becomes more oxidized (e.g., at 10,000 years after closure, saltstone is modeled with a K_d of 0.8 mL/g as compared to the FDC floor at 406 mL/g, FDC walls at 388 mL/g, Vault 4 floor at 388 mL/g and Vault 4 walls at 228 mL/g) (Section 2.7). The NRC staff does not believe this modeled behavior is realistic because (1) the disposal unit concrete is expected to have about 40% of the specific reducing capacity of saltstone because of its lower blast furnace slag concentration; (2) the disposal unit floors and walls are more exposed to environmental conditions than saltstone is and would be expected to fracture within 10,000 years; (3) water flowing through fractures in the disposal unit floors and walls would be expected to create oxidized conduits in which Tc is not expected to be re-reduced and immobilized after release from the saltstone, and (4) water may flow out of the disposal unit through joints and consequently may not interact significantly with the disposal unit concrete.

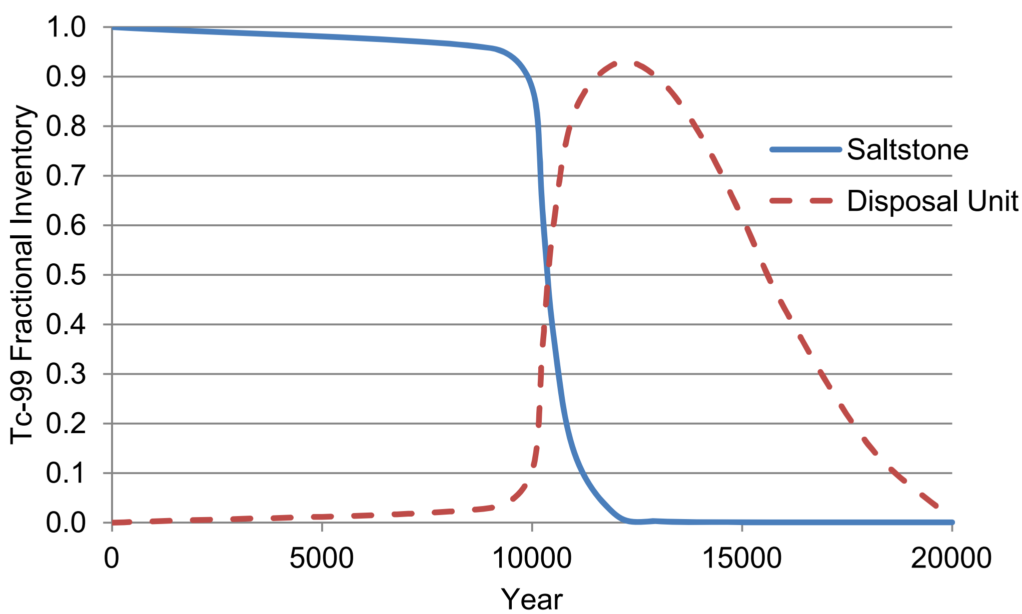

Figure 2.13-7 Intermediate results from DOE PORFLOW™ model for Case K, showing Tc release from Saltstone and re-concentration in the FDC floor and walls. At approximately 13,000 years after closure, over 90% of the Tc inventory originally in saltstone is present in the FDC floor and walls.

The two primary effects of this reconcentration in the disposal unit floor and walls are to delay the peak dose by approximately 5,000 years and to significantly reduce the magnitude of the peak dose. Peak doses are reduced because the peak dose is very sensitive to the rate at which Tc is released into the environment, and the modeled Tc release from the disposal units is much slower than the modeled Tc release from the saltstone.

Cases K, K1, and K2 were run with the same assumptions about far-field flow used in Case A and the deterministic sensitivity analyses discussed in this section. The maximum aquifer concentrations for Cases K, K1, and K2 occur in Sector I compared to Sector B for Case A and most of the other sensitivity cases evaluated in the PA. FDC sources in the northern portion of the SDF that contribute most to Sector I concentrations appear to be significantly affected by the groundwater divide that tends to spread plumes laterally in the Upper Three Runs aquifer, thereby limiting plume overlap (Figure 2.8-4). Additionally, the strong vertical gradient that appears to exist at the SDF also tends to spread contaminant plumes vertically leading to enhanced vertical dispersion and potential loss of contaminant mass to the Gordon Confining Unit. Thus, like the cases originally evaluated in the PA, dilution and dispersion in the saturated zone appear to be significant to performance for DOE's Cases K, K1, and K2. The NRC staff expects that it is more likely that predicted doses would be increased, rather than decreased, by further adjustment of SDF far-field modeling.

2.13.3.3 NRC Independent Sensitivity Analyses

As previously discussed, Case K, K1, and K2 resolve many of the concerns the NRC staff has with the use of Case A as a case to compare to the performance objectives. In response to a DOE request to confirm that development of Case K was an acceptable approach to NRC's request for a revised base case, the NRC staff agreed that development of Case K appeared to be an acceptable approach (NRC, 2011c). However, as discussed in Section 1.2, because of the complexity of the system being modeled, the NRC staff also indicated to DOE that it could not make any specific conclusions about the assumptions used in Case K until the staff reviewed DOE's written responses to RAI-2009-02 (NRC, 2011c). Upon review of Cases K, K1, and K2, the NRC staff concluded that, while some of the assumptions DOE used appear to be conservative or realistic, others appear to be overly optimistic. Because of the combination of assumptions used, the NRC staff is unable to conclude whether these cases are likely to over- or under-predict the potential dose from the SDF. Furthermore, unlike Cases A - E, Case K1 was supplied to NRC with a very limited sensitivity analysis (i.e., the only sensitivity analysis provided was a comparison of the results of Cases K, K1, and K2, which use different K_d values to represent Tc sorption in saltstone). As NRC indicated to DOE during technical exchanges regarding the development of Case K, if Case K is to be a significant part of DOE's compliance demonstration, NRC needs to understand the effects of parameter uncertainties on Case K results (NRC, 2011e). Thus, as described in this section, the NRC staff developed sensitivity analyses to better understand the potential dose from two radionuclides: (1) Ra-226, which dominates the dose from Case A and many of DOE's deterministic sensitivity analyses, and (2) Tc-99, which dominates the predicted dose in Cases K, K1, and K2.

<u>*Analyses related to Ra-226*</u>

In many of the deterministic analyses performed by DOE, the dose from Ra-226 contributed a significant portion of the peak dose (SRR-CWDA-2010-00033; Tables PA-1.1 to 1.4). Most of the Ra-226 dose is due to the ingrowth of Ra-226 from its parent Th-230. As noted in Section 2.2, the inventory of Th-230 is uncertain because it is below the analytical detection limit in tank waste samples taken to date (SRR-CWDA-2011-00115). In the Case K analysis, DOE stated that it revised the inventories of Th-230 and Ra-226 to much lower values to remove conservative assumptions used in the initial inventory estimate. The Case K analysis also included changes to a number of parameters (Section 2.1), including the Ra K_d values assumed for reducing cementitious materials (i.e., saltstone and disposal unit concrete) and for sand and clay in the subsurface. As described in Section 2.7, the NRC staff believes that more justification is needed for the Ra K_d values assumed for the saltstone and disposal unit concrete in Case K. Although Ra-226 is not a primary contributor to peak dose in Cases K, K1, or K2, the importance of Ra-226 to the final dose depends on uncertain assumptions about the inventory of its ancestors, including Th-230 and U-234, in saltstone (Section 2.2). As noted in Section 2.7, the NRC staff believes that the revised K_d values for the sand and clay are appropriate because they are based on site-specific measurements.

To understand the potential dose from Ra-226, the NRC staff used DOE's PORFLOW™ models to evaluate the dose from Ra-226 due to in-growth from Th-230 for four different cases: Case A, Case K, Case K with edited saltstone and disposal unit concrete K_d values for Ra-226, and Case K with edited saltstone, disposal unit concrete, clay, and sand K_d values for Ra-226. This assessment was performed for Vault 4 because the Vault 4 Th-230 inventory is expected to be much higher than the inventory in an FDC (Table 2.13-3). The version of PORFLOW™ used by the NRC staff (PORFLOW™ Version 6.12.3) in performing this assessment was different than the version used by DOE for Cases K, K1, and K2 (PORFLOW™ Version 6.30.2), which may have resulted in differences in the calculated dose due to differences in the way PORFLOW™ implements dispersivity in these two versions. Based on a comparison of DOE's results with the results of equivalent cases run in NRC's version of PORFLOW, these differences in the representation of dispersivity in the two versions of PORFLOW™ are expected to cause a difference of less than a factor of two in the dose results.. The results of these analyses indicate the large difference in projected dose that can be attributed to the difference in inventory estimates as compared to the difference in assumed K_d values (Table 2.13-3). Peak dose within 20,000 years of site closure are reported because, as noted in Section 2.7, the NRC staff believes that there is significant uncertainty in the timing of the peak dose due to uncertainty in the timing of the degradation of the system. Specifically, the NRC believes that the peak dose for Ra-226 that DOE predicts will occur within 20,000 years of site closure in Case K may occur within 10,000 years of SDF closure because of the uncertainty in the timing of increases in saltstone hydraulic conductivity. In Case K, saltstone hydraulic conductivity is based on a DOE assumption that most of saltstone occurs approximately 9,000 years, which results in a peak dose beyond 10,000 years. DOE did not provide a basis for this assumption about the timing of saltstone fracturing. NRC staff analyses related to the peak dose from Tc-99, discussed later in this section, indicate that different fracturing models could lead to earlier peak releases. As a result, the NRC staff believes it is possible that the peak dose from Ra-226 may occur within 10,000 years of SDF closure.

In all cases where the revised Th-230 inventory was assumed, the estimated dose was significantly less than 0.01 mSv/yr (1 mrem/yr). Thus, if the revised inventory of Th-230 is accurate, the Ra-226 dose is likely to be small regardless of the other parameter values assumed. The dose in the cases in which the original Th-230 inventory was used was higher, but the only one of these cases that had a peak dose greater than 0.25 mSv/yr (25 mrem/yr) was the case in which both the cementitious K_d values and the subsurface K_d values were changed to the original Case A values. As noted in Section 2.7, the NRC staff believes that DOE has provided a basis for use of the site-specific clay and sand K_d values used in Case K, so the NRC staff does not believe that this case is applicable. The doses estimated using the site-specific subsurface K_d values and the Case A or Case K cementitious material K_d values were all less than 0.1 mSv/yr (10 mrem/yr). The NRC staff determined that, with the K_d values assumed in Case A or Case K, the Ra-226 peak is not likely to exceed 0.25 mSv/yr (25 mrem/yr). However, because the peak from Ra-226 could overlap with the peak dose from other radionuclides, the Ra-226 dose could still contribute significantly to the total peak dose from the SDF and impact the determination of whether the total dose meets the limit in the performance objective (i.e., 0.25 mSv/yr [25 mrem/yr]). Additionally, DOE has not presented site-specific information on the cementitious material Ra-226 K_d values, which was shown to significantly affect the predicted peak Ra-226 dose. The NRC staff concludes that site-specific information on the cementitious material Ra-226 K_d values is needed. Therefore, the NRC staff will monitor the development of site specific K_d values for Ra-226 in saltstone and the disposal unit concrete.

Table 2.13-3: Estimated Peak Vault 4 Dose from Ra-226 that Ingrows from Th-230

	Vault 4 Dose to Source Ratio (mrem/yr Ra-226)/ (initial Ci of Th-230)[1]	**Dose Based on Original Th-230 Inventory (7.5 Ci) (mrem/yr)**	**Dose Based on Revised Th-230 Inventory (0.01 Ci) (mrem/yr)**
Case A [3]	< 0.29	<2.2	$< 2.9 \times 10^{-3}$
Case K (reducing cementitious K_d value of 100 mL/g [3], clay K_d of 185 mL/g, and sand K_d of 25 mL/g)	0.75	5.6	7.5×10^{-3}
Case K with a changed reducing cementitious K_d value of 3 mL/g [(3)]	1.2	9.2	1.2×10^{-2}
Case K with a changed reducing cementitious K_d value of 3 mL/g [(3)], and with a changed clay K_d of 17 mL/g and sand K_d of 5 mL/g	3.7	28	3.7×10^{-2}

[1] Based on peak dose in 20,000 years.
[2] Dose to source ratio calculated based on Ra-226 dose reported in PA Figure 5.4-4. This dose includes contributions from FDCs and from Ra-226 from sources other than ingrowth from Tc-230 and therefore bounds the Case A estimated dose from Vault 4 from Ra-226 ingrown from Th-230.
[3] Only middle-age reducing conditions are provided because only middle-age conditions were predicted to occur during the evaluation period. The K_d values for young and old-age cementitious material changed as well, but were not used because these conditions did not occur. Case K uses K_d values of 100 mL/g for young- and middle-age reducing cementitious material and 70 mL/g for old-age reducing material. The Case K tests with changed reducing cementitious K_d values used 0.5 mL/g for young reducing cementitious material, 3 mL/g for middle-age reducing cementitious material, and 20 mL/g for old-age reducing cementitious material.

Analyses Related to Tc-99

NRC analyses related to Tc-99 are based on Case K1 because the NRC staff concludes the K_d values for Tc in reducing and oxidizing grout are most realistic in this case (Section 2.7). However, because of apparently pessimistic assumptions about the rate of saltstone fracturing but unrealistically optimistic assumptions about Tc-99 retention in the disposal unit floors and walls (Figure 2.13-4); the NRC staff was unable to conclude whether Case K1 is likely to over- or under-predict the peak dose from Tc-99. To clarify the potential dose from Tc-99 in a case with fractured Saltstone, the NRC staff performed two types of independent sensitivity analyses. First, the NRC staff used DOE's Case K1 PORFLOW™ model to re-calculate Tc transport based on different assumptions about saltstone and disposal unit oxidation (but unchanged Case K1 flow velocities[28]). Second, the staff developed analytical calculations, implemented in Microsoft Excel®, to simulate (1) a simple average-K_d model and (2) a simple "dual-K_d" analytical model that simulates release of Tc from oxidized saltstone separately from release from reducing saltstone (NRC, 2012a, b). The analytical calculations were developed principally to compare results of the average-K_d model with the dual-K_d model, because this comparison could not be made within DOE's PORFLOW™ model. In addition, the analytical calculations were used as scoping calculations to allow the effects of variations in certain model parameters to be evaluated quickly.

NRC evaluated six cases with DOE's PORFLOW™ model (Table 2.13-3). First, the NRC staff reran Case K1 to ensure the results matched DOE's reported results. Then the NRC staff ran five sensitivity cases to evaluate the following factors: (Test 1) less sudden fracturing (i.e., fractures developing as a quadratic function of time rather than a logarithmic function of time); (Tests 2 and 3) the effects of the chemical retention of Tc in the disposal unit floors and walls; (Test 4) less fracturing occurring within 10,000 years of closure; and (Test 5) a lower K_d value for Tc in reducing saltstone. In DOE's Case K1, the fracture rate only affects saltstone oxidation. Saltstone hydraulic properties degrade to soil properties at 10,000 years and are not explicitly based on assumptions about fracturing. Therefore, to test the importance of the fracture rate, the NRC staff re-evaluated the oxidation of saltstone proceeding from fractures with the same saltstone oxidation model and oxidation parameters used by DOE, with the exception that fractures were assumed to appear as a quadratic function of time rather than a logarithmic function of time. The NRC staff then re-calculated the weighted average K_d value that DOE used in Case K1 based on the same K_d-averaging process used by DOE but with the new projection of the amount of oxidized saltstone as a function of time. The primary output of the staff's variations of DOE's Case K1 model was the Tc flux into the unsaturated zone. Dose predictions were based on the ratio of Tc flux to the unsaturated zone to final dose in DOE Case K1. Use of this ratio is expected to provide a reasonable dose estimate in cases with slower oxidation (e.g., Tests 4 and 5 in Table 2.13-3). NRC scoping calculations indicate that the use of this ratio may over-predict dose by approximately a factor of 3 in cases with more rapid oxidation (e.g., Tests 2 and 3 in Table 2.13-3) because the flux-to-peak-dose ratio diminishes when the peak release rate is brief.

[28] Flow velocities used in DOE's Cases K, K1, and K2 are identical.

Dose sensitivity to the fracture growth rate is expected to occur because, in this modeling approach, more Tc is released from new fractures than old fractures. Tc release from a fracture diminishes with time as the oxidized region near a fracture face grows and longer diffusion lengths into the matrix (for dissolved O_2) and out of the matrix (for Tc release) slow release from older fractures. Thus the rate at which Tc release from a fracture slows with time is sensitive to assumptions about diffusivity, including the assumption that the diffusivity remains constant instead of increasing as saltstone degrades and the particular value of the diffusion constant. Although diffusion typically is modeled as proportional to the square root of time, in some cases diffusion in a cementitious medium has been found empirically to have a different functional form (Kennedy and Strenge, 1992). NRC staff calculations confirm that, in the Tc release model used by DOE in Cases K, K1, and K2, Tc release is dominated by new fractures. The NRC staff therefore expects the magnitude and timing Tc release predicted by DOE's model to be sensitive to assumptions about rate at which new fracture faces develop.

Table 2.13-4: Variations of DOE Case K1 PORFLOW™ model.
(Highlighted values indicate values changed since the previous test.)

	Saltstone K_d (mL/g)		Disposal Unit K_d (mL/g)	Fracturing Scheme	Final Fracture Spacing (m)	Time of Peak Release Rate (yr)	Dose Estimate[1] (mrem/yr)
	Reduced	Oxidized					
DOE Case K1	500	0.8	500 to 217	Log	0.1	12,800	90
Test 1	500	0.8	500 to 217	**Quadratic**	0.1	12,100	86
Test 2	500	0.8	**0.8**	Quadratic	0.1	8730	680[2]
Test 3	500	0.8	0.8	**Log**	0.1	10,300	930[2]
Test 4	500	0.8	0.8	Quadratic	**1**	19,100	25
Test 5	**139**	0.8	0.8	Quadratic	1	10,100	35

There are 100 mSv/yr in 1 mrem/yr.

[1] Dose estimated based on the annual fraction Tc inventory released from the near-field domain in each case scaled by the ratio of the peak annual fractional Tc inventory released from the near-field domain in DOE's Case K1 to peak dose in Case K1.

[2] NRC staff believes these doses are significant overestimates attributable to an artifact of DOE's "single-porosity" average-K_d model that primarily affects cases with rapid oxidation.

Comparison of the results of DOE's Case K1 with the results of Test 1 (Table 2.13-4) indicates a very small effect of using a more gradual fracture rate. However, the true difference in the release rates of Tc from saltstone between DOE Case K1 and the Test 1 Case is obscured by the re-immobilization and gradual re-release from the disposal unit floor and walls. To evaluate the effect of retention in the disposal unit, the NRC staff re-evaluated DOE's Case K1 PORFLOW™ files using a K_d value for the disposal unit floor and walls more typical of oxidized concrete. This analysis does not assume that the disposal floors and walls are completely oxidized; rather, it simulates the performance of the disposal unit floors and walls if the pathways along which Tc would be released from the disposal unit are oxidized.

Comparison of DOE's Case K1 with Test 3, which differ only in the K_d used in the disposal unit floor and walls, shows that the modeled Tc retention in the disposal unit floor and walls lowered the predicted peak doses by slightly more than an order of magnitude (Table 2.13-4). Similarly, comparison of Test 1 with Test 2, which also differ only in the modeled retention of Tc in the disposal unit floors and walls, also show a similar order-of-magnitude dose reduction attributable to retention in the disposal unit concrete. Re-immobilization in the disposal unit floor and walls decreases the peak dose because release from the disposal unit floor and walls is significantly more gradual than the modeled Tc release from saltstone (Figure 2.13-5). This result is of interest principally because it demonstrates the large effect of the assumed disposal unit performance in DOE's Case K1 dose prediction. The predicted peak dose in Tests 2 and 3, however, are not expected to be realistic because NRC staff analytical calculations, described later in this section, indicate that this result is likely a significant overestimate due to an artifact of DOE's average-K_d model. The artifact related to the average-K_d model appears to be most significant in cases with faster rates of saltstone oxidation. Therefore, the dose predictions from Tests 4 and 5 are expected to be less effected by the artifact.

Comparison of Test 2 with Test 3, which differ only in the assumed rate of fracture growth, shows the expected decrease in peak dose if a more gradual fracture rate is used (i.e., quadratic instead of logarithmic fracture growth). Comparison of Test 2 and Test 3 shows a larger effect of the slower fracture growth rate than comparison of DOE's Case K1 and Test 1 because, in Tests 2 and 3, the release of Tc from saltstone is not obscured by gradual re-release from the disposal unit floors and walls. Significantly, Test 2 also shows that if Tc is released through oxidized fractures in the disposal unit floor and walls, and a more gradual fracture rate is assumed, the predicted peak dose could occur within 10,000 years of site closure.

Test 4 and Test 5 demonstrate the significant effect of the final fracture spacing. In particular, comparison of Test 4 with Test 2, which differ only in the final fracture spacing, shows a change in fracture spacing from 10 cm (4 in) to 1 m (3 ft) decreases the predicted peak dose by approximately a factor of 27 and delays the peak dose from approximately 8,730 to 19,100 years after closure. In comparison to Test 4, Test 5 shows that using a K_d for reducing grout that is similar to values measured by DOE in tests of desorption of Tc from saltstone core sample in the presence of trace quantities of oxygen (i.e., 30 to 60 ppm) (SRNL-STI-2010-00667) causes a modest increase in dose but a significant advance in the timing of the peak dose (i.e., from 19,100 years to 10,100 years after closure). This result is of interest because it remains unclear whether saltstone could be exposed to trace quantities of oxygen.

To further test the applicability of the results of the variations in DOE's Case K1 PORFLOW™ model (Table 2.13-4), the staff developed analytical calculations to evaluate the behavior of simple average-K_d and dual-K_d models. Both models were based on the same saltstone oxidation calculations the NRC staff used to estimate saltstone oxidation for alternate fracture spacings and growth rates in its variations of the DOE Case K1 PORFLOW™ model. In addition to oxidation proceeding from fracture faces, oxidation also was assumed to take place because of oxygen in water flowing through the saltstone. The fraction of oxidation proceeding from fracture faces was reduced to account for areas already oxidized by infiltrating water to avoid double-counting of the oxidation. In the simple analytical average-K_d model, the fraction of oxidized saltstone was used to develop a weighted-average K_d value the same way DOE developed the weighted average K_d value in Cases K, K1, and K2. In the dual-K_d model, the fraction of saltstone oxidized at each time step was used to predict release from the oxidized and reduced fractions separately. The amount of Tc released in each time step was tracked so that the oxidized areas would cease to release Tc once the inventory in the oxidized volume was depleted. In both models, the primary output was the annual fractional release of Tc from saltstone, and no credit was provided for retention of Tc in the disposal unit floors and walls.

Comparison of the simple average-K_d and dual-K_d models indicates that the average-K_d model can under-predict releases in the short-term and significantly over-predict peak release rates in some cases (Figure 2.13-5). The NRC staff calculations indicate the average-K_d model over-predicts release rates most significantly in cases in which most of the release comes from the oxidized, rather than the reduced, fraction of saltstone. Thus the effect is most pronounced when there is relatively low water flow through saltstone to release Tc from the reduced region before significant oxidation occurs. Comparison of model results indicates that the dual-K_d model allows a more gradual release of radionuclides, resulting in a smaller available inventory at times when the average-K_d model would predict nearly the entire initial inventory is released with a relatively low average sorption coefficient. This result is similar to the result of DOE's comparison of an average-K_d model and a dual-K_d model (which DOE refers to as single- and dual-porosity models) (SRR-CWDA-2011-00114), although the NRC staff's simplified models predicted a more substantial difference between the two approaches than predicted by DOE. The magnitude of the overprediction of peak dose that both NRC and DOE expect the average-K_d model to cause is unclear. DOE's conclusion that the average-K_d model provides a good representation of Tc release is based on a comparison of two GoldSim® models rather than a comparison involving the actual PORFLOW™ model DOE used to model Tc release. NRC staff concluded DOE's GoldSim® comparison was likely to underestimate the difference between the models, as compared to a similar comparison performed in PORFLOW, because of certain implementation details in DOE's GoldSim® models (Section 2.7). However, the DOE models include diffusion between the oxidized and reduced regions, which is an important process not included in the simplified NRC staff models. In DOE's Case K, K1, and K2 analyses, the expected over-prediction of peak dose attributable to the average-K_d model is further obscured by the re-concentration and gradual re-release of Tc from the disposal unit floor and walls (Figure 2.13-7).

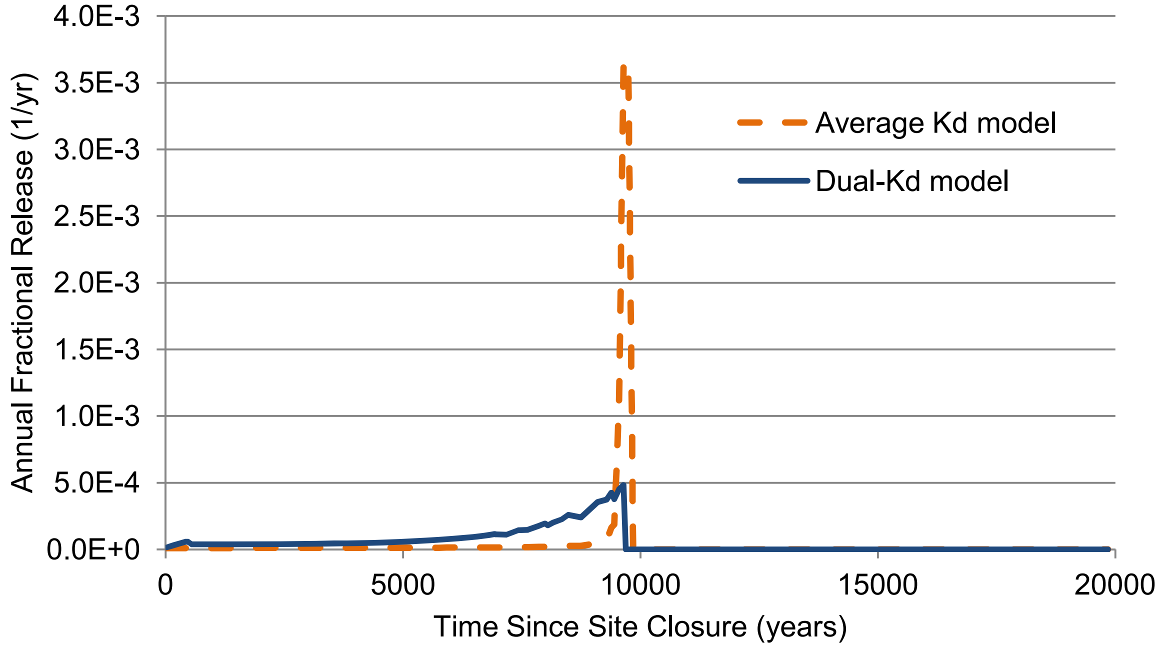

Figure 2.13-8 Comparison of annual fractional release from saltstone predicted by NRC's simple analytical average-K_d and dual-K_d models, using DOE Case K1 fracturing, near-field flow, and K_d values. Both models simulate the same cumulative release.

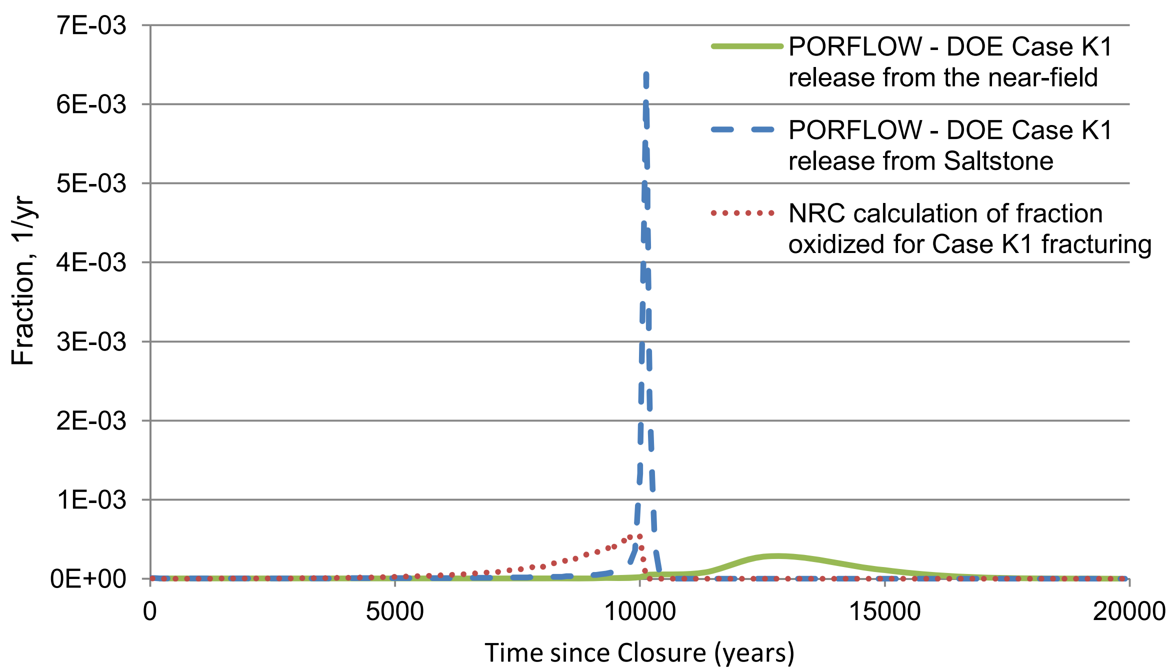

Figure 2.13-9: Comparison of annual fractional release from saltstone and from the near-field domain (i.e., saltstone, disposal units, and unsaturated soil) from DOE Case K1 model output files (STAT.out files [NRC, 2010g]) with the NRC predicted saltstone oxidation for Case K1 fracturing.

In addition to evaluating the applicability of the average-K_d model, the NRC staff's analytical calculations based on the simple dual-K_d model were used to map a greater range of parameter variation than could be addressed efficiently with variations in the DOE Case K1 PORFLOW™ model. The NRC staff did not use these values for comparison to the performance objectives. Rather, these values were used to indicate factors to which the Case K1 dose may be sensitive and to develop an estimate of the uncertainty in the Case K1 value. As previously discussed, this effort was undertaken because the NRC staff concluded the DOE Case K1 average-K_d model overestimated the expected annual fractional release from the saltstone waste form in a way that was difficult to isolate using the PORFLOW™ model. In addition, the effort was necessary because, unlike Cases A - E, Case K1 was supplied without an uncertainty analysis and with limited deterministic sensitivity analysis (i.e., the only sensitivity analysis provided was a comparison of the results of Cases K, K1, and K2, which used different K_d values to represent Tc sorption in saltstone).

Specifically, the NRC staff used the analytical calculations to evaluate the effects of changing (1) saltstone fracture spacing at 10,000 years after closure, (2) the function describing the rate of fracture growth with time, (3) flow through saltstone between zero and 20,000 years after site closure, (4) sorption coefficients for Tc in oxidizing and reducing saltstone, and (5) the reducing capacity of saltstone. As in the variations on DOE's Case K1 PORFLOW™ model, results were evaluated for final fracture spacings of 1 fracture every 10 cm (4 in) and 1 fracture every 1 m (3 ft). Similarly, the fracture growth rate also was alternated between logarithmic growth and quadratic growth with time. Three different options were evaluated with respect to flow through saltstone. One set of calculations used the near-field flow through saltstone from DOE's Case K1 PORFLOW™ model. Another two sets of calculations assumed flows were limited by the hydraulic conductivity of saltstone. First, the hydraulic conductivity of saltstone was assumed to be 1×10^{-7} cm/s for the first 10,000 years after closure and to increase linearly to 1×10^{-6} cm/s by 20,000 years after closure. Second, the hydraulic conductivity of saltstone was assumed to be 1×10^{-8} cm/s at closure, increasing linearly to 1×10^{-7} cm/s by 10,000 years after closure and 1×10^{-6} cm/s by 20,000 years after closure. Sorption coefficient (K_d) values in oxidizing saltstone were alternated between 0.8 mL/g and a nominal value of 1×10^{-4} mL/g to test the importance of sorption in the oxidized region. Sorption coefficient (K_d) values in the reduced region were alternated between 500 mL/g and 139 mL/g. The reducing capacity of saltstone was alternated between the Case A value (0.822 meq e^-/g) and the Case K1 value (0.206 meq e^-/g). The matrix of results from varying each of these parameters is provided in Appendix B.

In general, selecting combinations of parameters as described above yielded fractional release rates corresponding to doses between 0.1 and 1.7 mSv/yr (10 and 170 mrem/yr). In most cases, the peak release rate within 10,000 years was equivalent to the peak release rate within 20,000 years of site closure. The largest consistent exception to that generality is cases based on a Case A reducing capacity (0.822 meq e^-/g) and 1 m (3 ft) final fracture spacing, which typically had peak doses within 20,000 years that were approximately three times greater than the peak doses within 10,000 years of closure because the slower oxidation resulting from less fracturing and more reducing capacity lead to later release peaks.

Of the factors evaluated, the factor with the largest apparent effect on the fractional release rate was the final fracture spacing. The calculations most likely to yield relatively low release rates were calculations with a final fracture spacing of 1 m (3 ft) and Case A reducing capacity (0.822 meq e^-/g) (See Appendix B for release rate results). In general, for cases with a 1 m (3 ft) fracture spacing and Case A reducing capacity, cases in which water flow was based on DOE's Case K1 PORFLOW™ flow yielded slightly larger release rates than cases in which water flow was limited by saltstone hydraulic conductivity (i.e., limited to $1x10^{-7}$ or $1x10^{-8}$ cm/s at closure and $1x10^{-7}$ cm/s at 10,000 years after closure). The PORFLOW™ flow yielded larger peaks even though choosing an initial hydraulic conductivity of $1x10^{-7}$ cm/s allowed more water flow through for the first several thousand years after closure. The NRC staff hypothesizes that the cases based on DOE's Case K1 flow resulted in larger peak doses because the flow remains low while oxidation from fracture faces proceeds and increases when the saltstone is relatively oxidized.

Variations in the fracture growth rate had a modest effect in most cases; although in cases with a low final fracture spacing (i.e., 10 cm [4 in]) and Case K reducing capacity (i.e., 0.206 meq e^-/g) logarithmic fracture growth could result in approximately twice the fractional release rate of quadratic growth. The difference between using a K_d value of 0.8 mL/g and modeling the oxidizing saltstone with essential no sorption in oxidizing saltstone (i.e., K_d of $1x10^{-4}$ mL/g) was imperceptible in most cases and small in others. The difference between using a K_d of 500 mL/g and 139 mL/g in the reducing grout was slightly larger but still within a factor of two in the cases evaluated.

While the results of these analyses were not used as a basis for assessing compliance, comparison of the average-K_d and dual-K_d models supported the NRC conclusion that the tests run with the DOE PORFLOW™ Case K1 model modified to reflect Tc release through oxidized pathways in the disposal unit floors and walls (i.e., Tests 2 and 3 in Table 2.13-4) produced artificially large dose predictions due to an artificially large modeled pulse release from the waste form. In addition, comparison of the average-K_d and dual-K_d models showed that, in many cases, average-K_d model tended to delay the peak release rate as compared to the dual-K_d model. This result is of interest because variations of DOE's PORFLOW™ Case K1 model (Table 2.13-4) showed that, if Tc is assumed to be released through oxidized pathways in the disposal unit floors and walls, the peak dose can occur within (Test 2) or around (Test 3 and 5) 10,000 years after site closure. If the average-K_d model tends to delay the dose, it appears that the peak dose predicted in Tests 3 and 5, as well as Test 2, all may occur within 10,000 years of site closure.

Similarly, while the results of sensitivity analyses performed with the dual-K_d model were not used as a basis for a compliance determination, these analyses highlighted the importance of the final fracture spacing and the assumed specific reducing capacity for saltstone.

In summary, the staff conducted two types of sensitivity analyses related to DOE's Cases K, K1, and K2. First, the staff used DOE's PORFLOW™ model to test the effects of variations in model parameters on the peak doses from Ra-226 and Tc-99. With respect to Ra-226, the NRC staff concludes that site-specific information on the cementitious material Ra-226 K_d

values is needed. Therefore, the NRC staff will monitor the development of site-specific K_d values for Ra-226 in saltstone and the disposal unit concrete. With respect to Tc-99, based on variations of DOE's Case K1 PORFLOW™ model, the staff concluded that that changes in assumptions about fracture spacing and chemical retention of Tc-99 in disposal unit concrete significantly affected the peak dose. The NRC staff also concluded that peak doses from Tc-99 could occur within 10,000 years of disposal. The second type of sensitivity analysis the NRC staff conducted was based on simple analytical representations of a dual-K_d and average-K_d model for Tc-99 release. Based on the results of these models, the NRC staff concluded that the average-K_d model tends to delay and overestimate the peak Tc-99 release in DOE's Case K PORFLOW™ model. The saltstone release rates calculated with the simplified dual-K_d model generally supported the saltstone release rates calculated with variations of DOE's Case K PORFLOW™ model.

2.13.3.4 Comparison of Results to Performance Objectives

The Performance Objective for Protection of the General Population from Releases of Radioactivity in §61.41 states:

> *Concentrations of radioactive material which may be released to the general environment in ground water, surface water, air, soil, plants, or animals must not result in an annual dose exceeding an equivalent of 25 millirems to the whole body, 75 millirems to the thyroid, and 25 millirems to any other organ of any member of the public.*

The NRC has evaluated compliance using a dose limit of 0.25 mSv/yr (25 mrem/yr) Total Effective Dose Equivalent (TEDE), consistent with the approach discussed in the Final Rule for Disposal of High-Level Radioactive Wastes in a Proposed Geologic Repository at Yucca Mountain, Nevada (66 FR 55752):

> *Because each of the organs had the same limit under the older system even though each had a different level of radiosensitivity, it is very difficult to directly compare the old standards with the new standards. As noted in the proposed rule, the Commission considers 0.25 mSv/yr (25 mrem/yr) TEDE as the appropriate dose limit to compare with the range of potential doses represented by the older limits that had whole body dose limits of 0.25 mSv/yr (25 mrem/yr).*

To evaluate doses for comparison to a 0.25 mSv/yr (25 mrem/yr) TEDE limit, DOE and NRC have used the updated internal dose factors of the International Commission on Radiological Protection (ICRP) Publication 72 (ICRP, 1996), consistent with Commission direction in the Staff Requirements Memorandum on Processes for Revision of 10 CFR 20 Regarding Adoption of ICRP Recommendations on Occupational Dose Limits and Dosimetric Models and Parameters (SRM-SECY-01-0148).

As discussed in Section 2.3.1, compliance with the performance objectives typically is based on the predicted peak dose within 10,000 years of site closure. DOE predicted the peak dose to a member of the general public within 10,000 years of site closure for deterministic Case A, which DOE considers its base case, to be 0.014 mSv/yr (1.4 mrem/yr) in Sector B. DOE's peak predicted dose within 20,000 years for deterministic Case A is 0.031 mSv/yr (3.1 mrem/yr) in Sector I. The peak of the mean dose from DOE's "all cases" probabilistic analysis (i.e., a weighted average of Cases A through E) within 10,000 years of closure is approximately 0.04 mSv/yr (4 mrem/yr) in Sector B and the all cases peak of the mean result within 20,000 years is approximately 0.10 mSv/yr (10 mrem/yr) in Sector J (2009 PA; Figure 5.6-33). All of these predicted doses are below the dose limit of 0.25 mSv/yr (25 mrem/yr) specified for protection of the general population in §61.41. As discussed in Section 2.1, the NRC staff does not consider Case A to be an acceptable compliance case because (1) it is not reflective of current site conditions, (2) it does not account for measured ranges of parameter values or expected differences between laboratory and as-emplaced values, and (3) it does not adequately account for potential changes in site conditions during the compliance period. Additionally, because the probabilistic analysis incorporated near-field flow values from the deterministic PORFLOW™ analyses, because the subjective weights given to each alternate case in the probabilistic model were unsupported, and because of apparent modeling inconsistencies in the probabilistic analysis, the NRC staff has not used the results of DOE's probabilistic analysis in its compliance evaluation.

In DOE's base case (Case A) and most of its sensitivity analyses, Ra-226 dominates the dose to a member of the general population. Based on DOE analyses and the results of NRC's sensitivity analyses related to Ra-226 (Section 2.13.3.3), the NRC staff concludes that the dose from Ra-226 is not likely to exceed 0.25 mSv/yr (25 mrem/yr). However, if key model parameters vary from the values assumed in these cases, the estimated dose may increase. The NRC staff concludes that the Ra-226 dose is less risk significant than the Tc-99 dose. However, the NRC staff believes that the collection of better information on the Th-230 inventory or on key parameters, such as the Ra-226 K_d value in saltstone and disposal unit concrete, would be useful in confirming that the Ra-226 dose will not exceed the 0.25 mSv/yr (25 mrem/yr) dose limit consistent with the §61.41 performance objective.

In response to NRC concerns about using Case A as the base case (NRC, 2010b, i; PA-8), DOE developed Cases K, K1, and K2. Because they resolve many of the specific concerns the NRC staff has about using Case A as the compliance case (Section 2.1), the NRC staff has relied heavily on these cases in its review. In its response to NRC concerns about Case A (SRR-CWDA-2011-00044; PA-8), DOE expressed concern that relying heavily on Case K would create the impression that Case A was not the most likely case. The NRC staff understands DOE's desire to develop a case that provides a "best estimate" of the dose. However, because Case A appears to be overly optimistic compared to present day conditions (e.g., by assuming no fracturing of saltstone (Section 2.6), a hydraulic conductivity lower than measured values for field samples (Section 2.6), and Tc sorption coefficients greater than relevant measured values (Section 2.7)), the NRC staff believes Case A is not the most likely case and does not provide a best estimate of the dose from the SDF.

As previously discussed, Case K, K1, and K2 resolve many of the concerns the NRC staff has with the use of Case A as a case to compare to the performance objectives. However, as discussed in Section 1.2, because of the complexity of the system being modeled, the NRC staff indicated to DOE that it could not make any specific conclusions about the assumptions used in Case K until the staff reviewed DOE's written responses to RAI-2009-02 (NRC, 2011c). Although DOE characterized Case K as a non-mechanistic, pessimistic sensitivity analysis, upon review of Cases K, K1, and K2, the NRC staff concluded that, while some of the assumptions DOE used appear to be conservative, others appear to be realistic or overly optimistic. One apparently realistic (as opposed to pessimistic) change is that Cases K, K1, and K2 use lower inventories of Ra-226 and its parent Th-230 than Case A, which lowers the peak dose from Ra-226. Similarly, Cases K, K1, and K2 use updated biosphere parameters that also lower the dose from Ra-226 and other radionuclides (SRR-CWDA-2011-00044 response PA-9). Other changes do not appear to be as pessimistic as DOE indicates. For example, DOE characterized the reduction of the specific reducing capacity of saltstone to 25% of its Case A value as "arbitrary" (SRR-CWDA-2011-00044 response PA-8). However, the factor was based on the fraction of blast furnace slag in saltstone because it is unclear why the measured specific reducing capacity for saltstone would be equivalent to the measured reducing capacity of pure blast furnace slag (Section 2.6). Similarly, DOE characterizes the initial saturated hydraulic conductivity of $1x10^{-8}$ cm/s used in Case K as pessimistic, but the value is actually less than (i.e., more favorable than) values measured for present-day as-emplaced saltstone (Section 2.6). Although DOE discounts the measured values in as-emplaced saltstone as an artifact of the sampling method, the same sampling method used on laboratory-prepared samples did not elevate the measured hydraulic conductivity as compared to cast samples (SRNL-STI-2010-00657). In summary, the NRC staff does not agree with DOE's characterization of Cases K, K1, and K2 as overly pessimistic sensitivity analyses and, instead, believes these cases provide a more probable estimate of the peak dose from the SDF than Case A.

In all three cases, DOE projected a dose peak at approximately 8,000 years due primarily to I-129, while a larger peak due to Tc-99 is predicted to occur at different times in Case K, K1, and K2 (Figure 2.13-1). In Case K, DOE predicts a 0.55 mSv/yr (55 mrem/yr) dose peak from Tc-99 to occur at approximately 15,300 years after site closure. However, Case K uses sorption coefficients (K_d values) in reduced (1,000 mL/g) and oxidized (10 mL/g) cementitious material that the NRC staff does not believe are consistent with DOE experimental results (Section 2.7). DOE predicts the dose from Case K1, which uses better-supported K_d values for Tc in the saltstone (i.e., 500 mL/g in reducing saltstone and 0.8 mL/g in oxidized saltstone), to be approximately 0.90 mSv/yr (90 mrem/yr) and to occur at approximately 12,900 years after site closure (Figure 2.13-1).

As described in Section 2.13.3.2 and 2.13.3.3, the magnitude and timing of predicted peak dose from DOE Case K1 is sensitive to and delayed by (1) DOE assumptions about the timing and rate of saltstone fracture growth, (2) DOE's use of the average-K_d model for Tc release, and (3) seemingly optimistic assumptions about Tc re-reduction and re-immobilization in disposal unit floors and walls. Because of the considerable resulting uncertainties in the timing of the peak Tc dose, the NRC staff does not have reasonable assurance that the maximum dose from

Case K1 (i.e., 0.9 mSv/yr [90 mrem/yr]) will occur beyond 10,000 years after site closure as the DOE analysis concludes. Thus, based on NRC's use of Case K1 in its compliance evaluation (Section 2.1), the value of the predicted peak dose of 0.90 mSv/yr (90 mrem/yr), and the large uncertainty in the predicted timing of the dose, the NRC staff could not conclude it had reasonable assurance that the dose would not exceed the 0.25 mSv/yr (25 mrem/yr) dose limit specified in the performance objective within 10,000 years of site closure.

Rather than base the compliance demonstration on DOE's Case K1 alone, the NRC staff performed additional analyses to determine whether it could develop additional assurance about the timing or magnitude of the Case K1 peak dose. As described in Section 2.13.3.2 and 2.13.3.3, the NRC staff believes the projected dose from Case K1 is artificially increased by DOE's average-K_d approach to Tc sorption and artificially decreased by unrealistic assumptions about Tc retention in and gradual release from the disposal unit floors and walls. However, as described in Section 2.13.3.3, the NRC staff sensitivity analyses performed with DOE's Case K1 model, as well as NRC staff calculations with simplified models of Tc release, support a dose prediction within an approximate range that is between the limits established in the §61.41 performance objective (0.25 mSv/yr [25 mrem/yr]) and the public dose limit found in §20.1301 (i.e., 1 mSv/yr [100 mrem/yr] total effective dose equivalent). Depending on information developed to support assumptions about the inventory of Ra-226 and its ancestors, the uncertainty in the potential dose from Ra-226 and its timing could slightly increase dose predictions, but is not expected to significantly affect these estimates. Thus, based on current available information and model support, the NRC staff does not have reasonable assurance that salt waste disposal at the SDF meets the 10 CFR 61 Performance Objective for Protection of the General Population from Releases of Radioactivity. Although the NRC staff cannot conclude that the performance objective in §61.41 is met, the potential dose to an off-site member of the public from DOE's disposal actions is still expected to be relatively low (i.e., approximately 1 mSv/yr [100 mrem/yr], the public dose limit[29] in §20.1301).

Furthermore, because the staff expects that any exceedance of the §61.41 limit would occur long after site closure, DOE may be able to develop additional information or take mitigative actions in the short term (e.g., add a strong chemical reducing agent to the saltstone formula and improve the quality assurance for saltstone grout) that could provide reasonable assurance that salt waste disposal at the SDF meets the 10 CFR 61 Subpart C performance objectives. The NRC staff has identified these issues factors it plans to use as a basis for further SDF monitoring (Section 6.2 and Table A-1). As described in Section 6.3, the NRC staff will review information DOE provides during monitoring to determine if the information addresses the technical concerns identified by the staff in the TER and provides reasonable assurance that salt waste disposal at the SDF meets the §61.41 Performance Objective.

[29] As indicated in the Statements of Consideration for 10 CFR 20 (56 FR 23374), the 1 mSv/yr (100 mrem/yr) value in the final rule represents the primary dose limit for protection of the public. The Statements of Consideration also indicate that the dose limit "applies only to doses from radiation and radioactive materials under the licensee's control." At the SRS, there may be multiple sources of radioactivity that could contribute to dose. The 1 mSv/yr (100 mrem/yr) value is referenced here not as a regulatory limit, but as an indication of the relative magnitude of the potential dose.

3. Protection of Inadvertent Intruders

3.1 Assessment of Inadvertent Intrusion

To demonstrate that the performance objective in §61.42, protection of inadvertent intruders, is met, DOE performed an evaluation of the potential long term dose to an individual who inadvertently intrudes into the disposal site following the assumed institutional control period (i.e., 100 years after site closure) to assess compliance with the performance objective in §61.42. DOE considered both acute and chronic intruder scenarios and used the chronic intruder agriculture scenario as the basis for the compliance evaluation. In the chronic intruder agriculture scenario, a farmer is assumed to live on site and use water from an on-site well as a drinking water source as well as a source for agriculture (i.e., irrigation and livestock water). DOE assumes that the well will not go directly through the disposal units because the disposal units contain long-lasting materials which are clearly distinguishable from the surrounding soil. DOE noted that because the local soil is generally sandy, local well drillers do not expect to need to drill through high-strength geologic materials when constructing a drinking water well, and consequently the well driller would stop operations and move to a different location upon encountering engineered barriers, such as the closure cap erosion barrier or the disposal unit concrete roof. Although DOE does not consider the construction of a well through saltstone to be a credible scenario, DOE performed two sensitivity analyses to assess the potential dose resulting from drilling directly into a disposal unit.

The pathways assumed in the chronic intruder agriculture calculation were the same as those assumed for the member of the public (Section 2.3) except that the groundwater concentrations were based on different well locations (on the SDF instead of outside of a 100 m boundary). The biosphere modeling approach and parameters used in this assessment were also the same as those used for the member of the public (Section 2.10). As with the dose assessment for the member of the public, the concentrations of radionuclides in the groundwater were determined using the PORFLOW™ code (Section 2.7), and the dose was calculated based on these concentrations using DOE's deterministic model GoldSim®(Section 2.10). However, the external dose due to direct exposure to the waste form through the engineered cover and the air pathway dose from diffusion of radionuclides from the waste form to the surface were evaluated in separate calculations (SRNS-J2100-2009-00006 and Section 2.9, respectively). The doses from both the direct exposure and air pathways were both predicted to be negligible. As was true for the dose assessment for the member of the public, in the intruder assessment DOE also assumes that the inventory in the FDCs is evenly distributed among the 64 FDCs.

The cases evaluated by DOE as part of the intruder assessment are summarized in Table 3-1. The Case A PORFLOW™ analysis was used in the initial evaluation of the dose to an inadvertent intruder. In this case, DOE used the highest average groundwater concentration within the 15 m by 15 m (50 ft by 50 ft) grid cells that encroached on the 1 m perimeter from the SDF (Figure 2.3-1). In the first NRC RAI (NRC, 2010b), the NRC staff expressed concern that the average concentration over the entire grid cell might be much less than the concentration at 1 m from a disposal unit, especially for slow moving and relatively fast decaying radionuclides. In the initial DOE RAI response (SRR-CWDA-2010-00033), DOE performed an analysis of the dose to an individual who uses water from directly below Vault 4. The concentration of radionuclides in this water was determined based on the flux of radionuclides entering the water table below Vault 4 that was calculated using the Case A PORFLOW™ model. This flux was converted to a concentration based on the area of the Vault 4 footprint and the Case A Darcy velocity of water into the water table from the vadose zone directly below the vault. In the second RAI response (SRR-CWDA-2011-00044), DOE performed a similar calculation for an FDC. However, this analysis used the updated biosphere parameters from Case K. DOE also provided a revised estimate of the dose to an individual using water taken from directly under Vault 4 based on the updated biosphere parameters.

In the initial DOE RAI response (SRR-CWDA-2010-00033), DOE also used the Case A model to calculate a dose from the groundwater concentration in a 15 m by 15 m (50 ft by 50 ft) grid cell that encroached on the 1 m perimeter from an individual FDC. In this case, the calculated groundwater concentrations were multiplied by 10 to account for the maximum expected inventory in an individual FDC.

DOE also provided predicted intruder doses based on the Case K PORFLOW™ model (SRR-CWDA-2011-00044). In this analysis, the dose was estimated based on the concentration of radionuclides in the groundwater beneath Vault 4 and an FDC. For the FDCs, the dose was based on the maximum concentration of each radionuclide in the groundwater under any FDC. This analysis also included the dose from stream pathways (e.g., swimming, fish ingestion), and used the updated Case K biosphere parameters.

In the acute intruder sensitivity analysis, DOE assumed that a well is drilled directly through a disposal unit and the well cuttings are spread across a garden. An individual is then exposed to the cuttings while spending time in the garden (20 hours/year). This individual receives a dose from the ingestion and inhalation of soil containing drill cuttings, and from direct exposure to the soil containing drill cuttings. The inventory in the cuttings was determined based on a hypothetical 20 cm (8 in) core drilled directly through Vault 1, Vault 4, or an FDC. The highest value from the three disposal unit types was assumed for each radionuclide. DOE also performed a chronic intruder sensitivity analysis that used the same drilling scenario use in the acute intruder analysis. This scenario considered both the exposure to the drill cuttings, as well as the pathways considered in the chronic intruder agriculture scenario (e.g., use of groundwater as a drinking water source as well as a source for agriculture).

The period of performance assumed by DOE for the inadvertent intruder for the purpose of demonstrating compliance with the performance objective in §61.42 was a period of 10,000 years following the closure of the site, as was true for the assessment of compliance with the performance objective in §61.41 (Section 2.3). DOE also ran all transport models to at least 20,000 years to determine peak concentrations that occur after the 10,000-year performance period.

In the Chronic Intruder: 1 m from the SDF Case, 91% of dose reported by DOE as occurring within 10,000 years is from Ra-226. The main pathways that contribute to the dose are drinking water consumption (47%), vegetable consumption (37%), and fish consumption (16%).

Table 3.1-1: Description of Intruder Cases Evaluated by DOE

Case	Source Term	Used updated biosphere parameters?	Reference
Chronic Intruder: 1 m from the SDF ***Case A***	Groundwater in grid cells that encroached on the 1 m perimeter of the SDF	no	SRR-CWDA-2009-00017 (p597)
Chronic Intruder: Vault 4 ***Case A***	Groundwater directly under Vault 4	no	SRR-CWDA-2010-00033 (p96)
Chronic Intruder: Vault 4 Case A Revised Biosphere ***Case A***	Groundwater directly under Vault 4	yes	SRR-CWDA-2011-00044 (p258)
Chronic Intruder: Vault 4 ***Case K***	Groundwater directly under Vault 4	yes	SRR-CWDA-2011-00044 (p102)
Chronic Intruder: FDC Case A Increased Concentrations ***Case A***	Groundwater in grid cells that encroached on the 1 m perimeter of the SDF. Calculated concentrations were multiplied by factor of 10 to account for maximum expected inventory.	no	SRR-CWDA-2010-00033 (p96)
Chronic Intruder: FDC Case A Revised Biosphere ***Case A***	Groundwater directly under a FDC	yes	SRR-CWDA-2011-00044 (p258)
Chronic Intruder: FDC ***Case K***	Groundwater directly under a FDC	yes	SRR-CWDA-2011-00044 (p102)
Acute Intruder: Drill Cuttings Sensitivity Analysis ***NA***	Drill cuttings from drilling through saltstone. The highest concentration of Vault 1, Vault 4, or the FDCs was assumed for each radionuclide.	no	SRR-CWDA-2009-00017 (p604)
Chronic Intruder: Drill Cuttings Sensitivity Analysis ***NA***	Drill cuttings from drilling through saltstone. The highest concentration of Vault 1, Vault 4, or the FDCs was assumed for each radionuclide.	no	SRR-CWDA-2009-00017 (p605)

The doses calculated for each of these cases reported by DOE as occurring within 10,000 years and 20,000 years are presented in Table 3-2.

Table 3.1-2: Results of Intruder Assessment Performed by DOE

Case	within 10,000 yrs		within 20,000 yrs		Reference
	Dose (mrem/yr)[1]	Time (yrs)	Dose (mrem/yr)[1]	Time (yrs)	
Chronic Intruder: 1 m from the SDF[2]	1.9	10,000	7.2	15,060	SRR-CWDA-2009-00017 (p597)
Chronic Intruder: Vault 4 Case A	35	9,760			SRR-CWDA-2010-00033 (p96)
Chronic Intruder: Vault 4 Case A Revised Biosphere	48	~9,950	~92	~16,500	SRR-CWDA-2011-00044 (p258)
Chronic Intruder: Vault 4 Case K	8.1	~8,700	~60	~13,000	SRR-CWDA-2011-00044 (p102)
Chronic Intruder: FDC Case A Increased Concentrations	1.5	10,000			SRR-CWDA-2010-00033 (p96)
Chronic Intruder: FDC Case A Revised Biosphere	0.96	~10,000	~20	~15,100	SRR-CWDA-2011-00044 (p258)
Chronic Intruder: FDC Case K	53	~7,900	~290	~15,500	SRR-CWDA-2011-00044 (p102)
Acute Intruder: Drill Cuttings Sensitivity Analysis	3.8[3]	500	3.8[3]	500	SRR-CWDA-2009-00017 (p604)
Chronic Intruder: Drill Cuttings Sensitivity Analysis	15[3]	500	15[3]	500	SRR-CWDA-2009-00017 (p605)

[1] 1 mrem/yr = 0.01 mSv/yr

[2] This case included the dose from plumes originating from multiple disposal units

[3] The dose prior to 500 years was not considered in this analysis because DOE expects the disposal unit roofs and clean cap to provide a robust barrier to intrusion directly into the saltstone waste form for at least 500 years after site closure.

In the Chronic Intruder: Vault 4 Case K analysis, the dose reported as occurring within 10,000 years is primarily due to Cs-135 (60.2%) and I-129 (34.4%), and the main pathways that contribute to the dose are fish consumption (63.3%), drinking water consumption (26.2%), and vegetable consumption (7.4%).

In the Chronic Intruder: FDC Case K analysis, 94.9% of the dose reported as occurring within 10,000 years is from I-129, and the main pathways that contribute to the dose are drinking water consumption (68.7%), and vegetable consumption (18.8%). DOE did not provide dose by pathway or radionuclide for the Case K doses within 20,000 years, but based on Figures PA-8.8

and PA-8.9 in SRR-CWDA-2011-00044, which show the member of the public dose, this dose is likely to be predominately from Tc-99. The dose calculated in the acute intruder sensitivity analysis was primarily due to Sn-126 and direct external exposure.

DOE performed an uncertainty analysis using the probabilistic GoldSim®model described in Section 2.1. In this analysis, the 1 m concentrations were estimated from the 100 m GoldSim®concentrations using a multiplier of 1.6. This multiplier was based on the maximum ratio of the maximum Case A 1 m concentration to the maximum Case A 100 m concentration for any of the key radionuclides.

3.2 NRC Evaluation – Protection of Intruders

A specific numerical performance objective is not provided in §61.42 for intruder assessment, however a dose limit of 5 mSv (500 mrem) per year was described in the Draft Environmental Impact Statement for 10 CFR 61 for development of waste classification requirements (NRC, 1981), which were developed to provide for inadvertent intruder protection. Consistent with the review procedures in NUREG-1854 (NRC, 2007b), Chapter 5, the staff uses this 5 mSv (500 mrem) annual dose value in evaluating the intruder scenarios.

The NRC staff finds that the scenarios and pathways analyzed by DOE for the assessment of the performance objective in §61.42 (protection of an inadvertent intruder) are appropriate based on the regional practices near SRS. The NRC staff finds that DOE's revised approach of taking the groundwater concentration from directly below the disposal unit is preferable to the use of the average concentration in the 15 m by 15 m (50 ft by 50 ft) cells that begin at a distance of 1 m from the disposal unit, although this revised approach is conservative, because the average concentration over this large grid cell could be much less than the concentration at 1 m from a disposal unit. Dilution in a grid cell could be especially problematic for slow moving and relatively short-lived radionuclides. This type of radionuclide was not identified as being significant in DOE's analysis of the concentration of radionuclides under the disposal units, but they could become important if key parameter values change. As was true for the member of the public analysis, the NRC staff believes that Case K1 best represents the current and expected future conditions (Sections 2.7 and 2.13). DOE did not perform an intruder analysis for Case K1, but Case K was analyzed, which differed from Case K1 only in the K_d values selected for Tc.

As noted in Sections 2.7 and 2.13, although NRC staff believes that Case K1 is the DOE case that best represents the system, this case does contain aspects that are likely conservative as well as aspects that are likely non-conservative. The aspects of this case that are potentially conservative include the assumed rapid rate of fracturing of the saltstone, the large amount of fracturing assumed at 10,000 years (i.e., a through-going fracture every 0.1 m), and the modeled faster peak rate of release caused by the use of the average-K_d approach (referred to by DOE as a "single porosity" model). Additionally, in the case of the intruder analysis, the assumed use of the groundwater directly below the disposal unit without dispersion or dilution in the saturated zone may be conservative. A non-conservatism in Case K1 is the modeled performance of the disposal unit wall and floor (Section 2.13). Also, as noted below, the NRC staff considers DOE's assumption regarding the timing of the fracturing (i.e., limited fracturing until almost 10,000 years) to be non-conservative. The NRC staff also finds that the K_d values used for saltstone in Case K (i.e., 1000 mL/g for reduced saltstone and 10 mL/g for oxidized saltstone versus 500 mL/g and 0.8 mL/g used for Case K1), which DOE used for the intruder analysis, is not conservative (Section 2.6). This combination of conservatisms and non-conservatisms clouds the conclusion of whether the outcome is likely to over- or under-predict potential doses.

NRC staff does not find that the method used by DOE to perform the uncertainty analysis for the intruder (i.e., the use of a multiplier based on the ratios of the 1 m to 100 m concentration of key radionuclides) to be appropriate because the ratio of the 1 m to 100 m concentrations is probably higher than the assumed multiplier for some radionuclides. Because DOE determined the key radionuclides based on the radionuclides that cause the highest dose at 100 m, radionuclides that travel slowly enough and have short enough half-lives to decay before reaching 100 m were not considered in the development of the multiplier. The ratio of the groundwater concentration at 1 m from the SDF to the concentration at 100 m from the SDF would be considerably higher for slow-moving, short-lived radionuclides than for the key radionuclides. None of the radionuclides that contributed significantly to the deterministic Case A and Case K intruder doses were ones that sorb significantly (and are, therefore, slow moving). However, when a range of parameter values is considered in a probabilistic uncertainty analysis there may be some realizations in which such radionuclides are more significant. Additionally, as discussed in Section 2.11, NRC staff has concerns regarding the probabilistic GoldSim®model that was used to perform this uncertainty analysis.

As was true for the member of the public assessment, the NRC staff is focusing on the Case K results in its evaluation to determine whether there is reasonable assurance that the performance objective in §61.42 is met. The Case K results for Vault 4 were 8.1×10^{-2} mSv/yr (8.1 mrem/yr) within 10,000 years and approximately 0.6 mSv/yr (60 mrem/yr) within 20,000 years. The Case K dose results for an FDC were 0.53 mSv/yr (53 mrem/yr) within 10,000 years, and approximately 2.9 mSv/yr (290 mrem/yr) in 20,000 years. As described in Sections 2.6 and 2.13, NRC staff does not believe that there is a basis for assuming that there will not be significant fracturing, and therefore oxidation, of the saltstone until nearly 10,000 years. Therefore, as in the case of the member of the public assessment, the NRC staff assumes that the Case K, K1, and K2 doses DOE reported as occurring between 10,000 and 20,000 years could occur prior to 10,000 years.

The Case K dose from Vault 4 DOE reported as being within 10,000 years was largely due to Cs-135 and the fish pathway. NRC staff expects that the reported fish consumption dose from Cs-135 is likely an artifact of the conservative assumptions DOE made in determining the concentration of radionuclides in the stream as well as in the amount of fish ingested, and the true dose from Vault 4 is likely to be less. The dose from an FDC reported by DOE as being within 10,000 years was primarily due to I-129. DOE did not report the contributions from individual radionuclides for the peak that they predict to occur after 10,000 years, but NRC staff expects that this peak is likely to be primarily due to Tc-99 since this radionuclide was responsible for most of the dose to the member of the public during that time period. Unlike

Vault 4, the predicted FDC dose does not have as significant of a contribution from the fish pathway, so this dose was not affected much by the conservative fish pathway assumptions.

The NRC staff estimated the Tc-99 dose to an intruder for Case K and Case K1 from PORFLOW™ files provided by DOE. Based on DOE's PORFLOW™ files[30], the maximum estimated concentration of Tc-99 under an FDC in Case K was 1.16×10^{-4} mol/m^3 (3.29×10^{-6} mol/ft^3) under Cell 11D (Figure 2.8-8). NRC staff estimated the all pathway dose conversion factor (DCF) for Tc-99 assuming the Case K biosphere parameters to be approximately 2.23×10^4 (mSv/yr)/(mol/m^3) [7.89×10^7 (mrem/yr)/(mol/ft^3)] based on the Case K member of the public peak dose of 0.55 mSv/yr (55 mrem/yr) and the Case K peak 100 m concentration of 2.46×10^{-5} mol/m^3 (6.97×10^{-7} mol/ft^3)[31]. Based on this all pathway DCF, the Case K Tc-99 intruder dose is approximately 2.6 mSv/yr (260 mrem/yr). The maximum estimated concentration of Tc-99 under an FDC in Case K1 was 1.89×10^{-4} mol/m^3 (5.34×10^{-6} mol/ft^3)[32], which, using the same pathway DCF, would be equivalent to a dose of approximately 4.2 mSv/yr (420 mrem/yr).

As noted above, Case K1 includes both conservative and non-conservative assumptions. As part of the evaluation of the member of the public dose, the NRC staff performed a series of evaluations to estimate the effect of the various assumptions (e.g., extent of fracturing, rate of fracturing) in Case K1 on the predicted dose to the member of the public (Section 2.13). Because the dose to a chronic intruder is attributable to groundwater pathways (i.e., rather than direct exposure) and because the doses from the groundwater pathway to the member of the public and the intruder are driven by the release of radionuclides from the saltstone, conclusions from these evaluations regarding the net effect of changing particular parameter values on the overall release and dose are also applicable to the intruder. Additionally, because Tc does not sorb strongly to soils and has a long half-life, it is not expected that there would be a significant difference between the concentration in the aquifer directly below a disposal unit and at a 100 m well.

In the member of the public sensitivity cases evaluated by NRC staff based on the DOE PORFLOW™ model (Table 2.13-4), it was found that the dose increased by almost an order of magnitude over DOE's reported Case K1 value of 0.9 mSv/yr (90 mrem/yr) when the model parameters were corrected for the disposal unit performance optimism. NRC staff expects that this high dose increase is largely due to an artifact of the release calculated using the average-K_d approach (Section 2.13). Additionally, the dose calculated by NRC using DOE's PORFLOW™ model was less than the original Case K1 dose of 0.9 mSv/yr (90 mrem/yr) when the model parameters were corrected for the disposal unit optimism and assuming a final fracture spacing of 1 m instead of 0.1 m. Similarly, as documented in Appendix B, the NRC staff calculated fractional release rates using a simple analytical dual-K_d model that did not include the disposal unit optimism and corrected for the average-K_d model conservatism. The calculated fractional release rates (Tables B-1 and B-2) were less than or equivalent to the Case K1 fractional release rate from the disposal unit (2.9×10^{-4} yr^{-1}) (Table B-3) if the final

[30] PORFLOW™ files\AquiferZ_rev1\transport_original\CaseK_rev2_spacing\All\Tc-99\STAT_V2 (NRC, 2010g)
[31] PORFLOW™ files\AquiferZ_rev1\transport_original\CaseK_rev2_spacing\All\Tc-99\STAT_V2 (NRC, 2010g)
[32] PORFLOW™ files\AquiferZ_rev1\transport_original\CaseK_rev2_spacing_Kd\All\Tc-99\STAT_V2 (NRC, 2010g)

fracture spacing was 1 m or if the rate of fracturing was less sudden. However, the fractional release rates calculated with the simple analytical dual-K_d model were higher than the Case K1 fractional release rate from the disposal unit by a factor of approximately 2 when the original fracturing assumptions were used (i.e., log based fracture growth and a final fracture spacing of 0.1 m). The results of these sensitivity analyses indicate that when the model parameters are only corrected for the disposal unit optimism and the average-K_d conservatism, the overall release, and therefore dose, would be expected to increase by a factor of approximately 2. However, when a lower rate and extent of fracturing is also considered, the overall predicted release decreases from the Case K1 value. Therefore, if the actual rate or extent of fracturing is less than what was assumed by DOE (i.e., log based fracture growth and a fracture spacing of 0.1 m at 10,000 years), then the actual rate of release, and resulting dose, will be less than the amount calculated using the original Case K1 model.

As noted in Section 2.13, the available model support for the expected amount and rate of fracturing is limited. However, the NRC staff finds that the rate of fracturing and fracture spacing at 10,000 years might be less than what was assumed by DOE in Case K. Additionally, the dilution of the radionuclides in the aquifer is likely to be greater than DOE assumed in the intruder calculations. Based on these considerations, the NRC staff concludes that the net effect of the conservatisms and non-conservatisms in Case K1 would be to lower the groundwater pathways dose slightly from the Case K1 dose of approximately 4.2 mSv/yr (420 mrem/yr). The NRC staff therefore concludes that the dose to an inadvertent intruder through the groundwater pathways is likely to be less than 5 mSv/yr (500 mrem/yr).

The NRC staff agrees that, in the short term, the drilling of a well directly through saltstone would be unlikely considering the physical properties of the disposal units, saltstone, and engineered cover as compared to the natural materials in the region. However, as these engineered systems degrade over time, these materials might not serve as strongly as an engineered barrier, and these materials might not be as distinguishable from the natural material. In the DOE drill cuttings scenario sensitivity analyses, the doses to acute and chronic intruders who intrude directly into saltstone after 500 years were estimated to be 3.8×10^{-2} mSv/yr (3.8 mrem/yr) and 0.15 mSv/yr (15 mrem/yr) respectively. The dose prior to 500 years could be significantly higher due to the presence of short-lived radionuclides, but intrusion during this time is not likely due to the expected performance of the cover and disposal units. Although the NRC staff thinks the dose from the chronic drill cuttings sensitivity case is underestimated because it is based on the groundwater concentration determined using Case A, rather than the larger groundwater concentrations in Case K or K1, the results of these sensitivity cases are useful in that they indicate that the additional dose an intruder would receive from being exposed to drill cuttings containing saltstone would not be significant compared to the dose to an intruder from using groundwater on site. The NRC staff therefore concludes that the dose to an intruder who drills directly into saltstone from the drill cuttings would be consistent with the performance objective for protection of an inadvertent intruder.

As was true for the member of the public analysis, the dose results estimated for the intruder are a function of the inventory assumed in the analysis. Section 2.2.2 describes the monitoring of the inventory that NRC staff has performed and the monitoring that will be performed in the future. In this monitoring, the NRC staff will track the actual inventory disposed in Vault 1, Vault 4, and the individual FDCs against the inventories listed in Tables 2.2-1 and 2.2-2. If the inventory in any of these disposal units is greater than the inventories listed in these tables, then an analysis will need to be performed to evaluate the potential dose consequences. In the case Chronic Intruder: FDC Case A Increased Concentrations, DOE evaluated the potential dose to an intruder from an individual FDC containing an increased inventory of radionuclides. However, this analysis was done using the Case A assumptions, which NRC staff does not believe are realistic (Section 2.1), so the NRC staff does not believe that this analysis adequately assesses the potential dose from an FDC containing an increased inventory. In the event that an individual FDC has an inventory that is greater than the Tables 2.2-1 and 2.2-2 inventories, the dose consequences should be analyzed using a more realistic case, such as Case K, or a bounding case.

Considering the projected intruder dose based on Case K, as well as the drill cuttings sensitivity analyses, the NRC staff concludes the dose to an inadvertent intruder is likely to be below 5 mSv/yr (500 mrem/yr). NRC, therefore, has reasonable assurance that the disposal plans will satisfy the performance objective in §61.42.

4. Protection of Individuals during Operations

4.1 Protection of Individuals during Operations

The performance objective in §61.43, *protection of individuals during operations,* states the following:

Operations at the land disposal facility must be conducted in compliance with the standards for radiation protection set out in part 20 of this chapter, except for releases of radioactivity in effluents from the land disposal facility, which shall be governed by §61.41 of this part. Every reasonable effort shall be made to maintain radiation exposures as low as is reasonably achievable.

This performance objective cross-references *the standards for radiation protection* in 10 CFR 20. In DOE-WD-2005-001 (DOE's 2005 waste determination for the saltstone facility), DOE provided a crosswalk of the relevant DOE regulation or limit consistent with that provided in 10 CFR 20 to demonstrate that the DOE regulation provides an equivalent level of protection. As stated in Section 2.5.2 of the PA, the performance objective in §61.43 is not addressed in the PA and DOE does not intend to address this subject in this or any future revisions of the PA. Measures taken in DOE-WD-2005-001 addressed this performance objective and annual reports on worker dose exposure and SRS environmental compliance reports account for a current and continual review of this section of the performance objectives.

In cross-referencing the regulations in 10 CFR 20, DOE only considered sections of the regulation containing the dose limits for the public and the workers during disposal operations. These sections are the following:

- §20.1101(d),
- §20.1201(a, e),
- §20.1208(a), and
- §20.1301(a, b).

The considered dose limits listed above correspond to the dose limits in 10 CFR 835 and relevant DOE Orders, which establish DOE regulatory and contractual requirements for DOE facilities and activities.

A number of measures will ensure that exposure of individuals during operations are maintained ALARA. These include:

(1) a documented Radiation Protection Program (RPP),
(2) a Documented Safety Analysis (DSA),
(3) design of the SDF and SPF,
(4) regulatory and contractual enforcement mechanisms, and
(5) access controls, training, and dosimetry (CBU-PIT-2005-00146).

Details of the measures taken by DOE to account for the list above can be found in CBU-PIT-2005-00146 and WSRC-IM-2004-00008.

4.2 NRC Evaluation – Protection of Individuals during Operations

In the 2005 TER (NRC, 2005), the NRC concluded that, during operations, individuals are protected by DOE regulations which were demonstrated to provide protection comparable to 10 CFR 20, and thus concluded that there is reasonable assurance that the performance objective of §61.43 for protection of individuals during operations can be met. Since the NRC's 2005 TER (NRC, 2005), not much has changed in DOE's demonstration of this performance objective.

As stated in the NRC monitoring plan (NRC, 2007a), to verify that DOE's radiation protection program is in place for operations at the SPF and the SDF to assess compliance with §61.43, *protection of individuals during operations*, NRC staff monitors activities at the SDF in the areas of worker dose monitoring, groundwater and air effluent monitoring, and other topics associated DOE's radiation protection program at the SDF. Since monitoring began in 2007, the NRC has conducted two onsite observations of the SDF that have included discussions related to this performance objective.

In October 2007 (NRC, 2008a), the NRC staff incorporated a review and discussion of two aspects of DOE's Radiation Protection Program (RPP): (i) worker dose monitoring and (ii) groundwater and air effluent monitoring. In verifying DOE's RPP was in place for operations at the SDF to assess compliance with §61.43, *protection of individuals during operations*, NRC staff (i) interviewed DOE's contractor environmental monitoring personnel, (ii) reviewed records and radiological control documents associated with saltstone operations, and (iii) reviewed associated worker dose records. The staff also focused specifically on the 2007 groundwater monitoring program results for three of the groundwater monitoring wells installed downgradient of Vault 4, and the 2007 air-effluent monitoring program for the SDF.

In March 2008 (NRC, 2008c), the NRC staff conducted another review of the DOE's RPP at the SDF. During this observation, the staff again focused on the 2007 groundwater monitoring program results, however, this time considering eight (instead of three, as before) groundwater monitoring wells installed in or near the salt waste disposal area. NRC staff also focused on the 2007 air effluent monitoring program for the SDF; reviewed soil-sample results in the vicinity of Vault 4; and, observed a groundwater sampling event from a background monitoring well.

Following the review of this topic in both of the onsite observations, NRC staff found that there is no conclusive indication of groundwater contamination in the vicinity of Vault 4 resulting from salt waste disposal operations, but stated that it will continue to monitor groundwater data. NRC staff also found that the air effluent sampling results for Vault 4 during filling operations indicate that doses to nearby workers and members of the public from air effluents were well below DOE regulatory limits. NRC staff also learned that personnel from SC DHEC periodically collect sediment samples from a nearby sedimentation basin. Also following this observation, NRC staff began including these independent data collected by SC DHEC as part of ongoing monitoring activities at the SDF.

The conclusion of both of the onsite observations was that NRC staff determined that DOE has an adequate RPP in place for SDF operations. Therefore, the results of the NRC's review in its 2005 TER (NRC, 2005), multiple onsite observations, and NRC's review of the annual SRS Environmental Reports provide reasonable assurance that the §61.43 performance objective will be met during facility operation. NRC will continue to assess DOE's radiation protection program through future monitoring activities.

5. Site Stability

5.1 Site Stability

The performance objective in §61.44, *stability of the disposal site after closure*, states the following:

> *The disposal facility must be sited, designed, used, operated, and closed to achieve long-term stability of the disposal site and to eliminate to the extent practicable the need for ongoing active maintenance of the disposal site following closure so that only surveillance, monitoring, or minor custodial care are required.*

In Section 2.5.2 of the PA, DOE states, "§61.43 for radiation protection during operations and §61.44 for site stability are not subjects that will use the PA as a technical basis in future closure documents and are therefore not discussed." However, DOE has developed additional information related to site stability which includes the effects of earthquakes, floods, and settlement. As the stability of the disposal facility is also important to the performance objectives in §61.41 and §61.42, this section includes discussion of site stability with respect to the performance of the disposal units, waste form, and the closure cap. Further discussion on the effect of site stability is included in the respective sections for the disposal units (Section 2.5), waste form (Section 2.6), and closure cap (Section 2.4).

The use of grout to stabilize salt waste at the SDF is designed to provide a monolithic structure, minimize void space, and prevent collapse. Following the closure of the disposal units, a closure cap will be installed over the SDF and a 100-year period of institutional controls will begin. During this period, active maintenance of the SDF will include prevention of pine forest succession and reparation of any significant erosion. No active SDF maintenance is assumed to be conducted beyond the institutional control period. The closure cap will be designed to provide a minimum of 3 m (10 ft) of clean material above the disposal units for the protection of inadvertent intruders. DOE's "SRS End State Vision" includes ownership and control of the entire site by the federal government in perpetuity and prohibits residential use of the site (2009 PA).

SRS is located on the Atlantic Coastal Plain within the Aiken Plateau. This region has a relatively low seismic activity (WSRC-IM-2004-00008). The largest known earthquake in the vicinity of the site occurred in Charleston, SC in 1886 with a magnitude of 7.3 on the Richter Scale (USGS, 2011). In addition, the SDF is sited on a well-drained topographic high (NRC, 2005). The 10,000 yr flood level for the Upper Three Runs basin near the SDF is approximately 48 m (157 ft) above mean sea-level (MSL) which is significantly below the lowest planned elevation of a disposal unit at the SDF, which is 79 m [260 ft] MSL (2009 PA).

Much of SRS, including the SDF, is underlain by calcareous sediment that has resulted in the presence of under-consolidated "soft zones" in the Santee formation (WSRC-TR-99-4083). In conjunction with these soft zones, layers of hardened sediment are commonly observed. These layers have been characterized as bridges or arches in a honeycomb-like structure that acts to redistribute stresses. Historically, some of the soft zones have consolidated, resulting in depressions on the land surface. These depressions, or "sinks", at the SRS site typically are 3 to 5 m [10 to 15 ft] deep and, in F Area, up to 300 m [1,000 ft] long (WSRC-TR-2007-00283). A stereoscopic examination of aerial photography of the SRS illustrated the potential presence of several basins and sinks located in and near the SDF (USACE, 1952; Figure A8). However, a subsequent investigation in 1986 stated that no evidence of ground subsidence was observed in the vicinity of where SDF is now located (87814-PT1).

In 2006 and 2009, geotechnical investigations beneath FDC 2A/2B and 7A - D[33] project sites indicated that the top of the Santee formation was approximately 24 m to 34 m Below Ground Surface (BGS) (80 ft to 110 ft BGS) with soft zones observed beneath the FDC 2A/2B site between approximately 30 m to 37 m BGS (100 ft to 120 ft BGS) (K-ESR-Z-00001; K-ESR-Z-00002). The sediment in this formation was described as being calcareous fine- to medium-grained sand with some clay and silt in addition to occasional fragments of shells and limestone. In response to NRC comment PA-14, DOE stated that the average thickness of the soft zones at the SDF is generally a few feet, although a soft zone approximately 4.3 m (14 ft) thick was identified beneath the FDC 2A/2B site (SRR-CWDA-2011-00044). DOE's postulated maximum lateral dimension of these features is 3 m to 6 m (10 ft to 20 ft) (SRR-CWDA-2011-00044). The stability of these soft zones has been investigated since the 1950s (USACE, 1952) with the understanding and treatment of these features evolving since preliminary evaluations (WSRC-TR-99-4083). Early approaches to stabilizing structures built in these regions included subsurface grouting, however it was determined that these grouting campaigns provided limited benefit in mitigating the potential settlement from the soft zones (WSRC-TR-99-4083). To evaluate the effect of consolidation of the subsurface layers beneath the SDF, DOE has conducted geotechnical and structural integrity analyses.

DOE conducted a structural integrity analysis to evaluate the effects of static and dynamic settlement on Vault 4 under closure conditions, which included disposal unit loading and 4.0 m (13 ft) of closure cap material (T-CLC-Z-00006). Static settlement displacements were based on soil properties and actual settlement measurements from the Defense Waste Processing Facility (DWPF). DOE considers this data to be applicable to Vault 4 because of similarities in the soil profiles and the close proximity of the DWPF to the SDF. The maximum displacement for Vault 4 was calculated to be 19 cm (7.3 in). Dynamic settlement estimates relied on site-specific geotechnical data which indicated that no soft zone settlement is expected because the Santee Formation is very dense beneath Vault 4. Settlement due to liquefaction from earthquakes with return periods of 2,500 yrs and 10,000 yrs ranged from 0.8 cm to 2.5 cm (0.25 in to 1.0 in) and 3.8 cm to 10.2 cm (1.5 in to 4.0 in), respectively. Mean dynamic settlement values of 1.9 cm (0.75 in) and 7.0 cm (2.75 in) were used in the structural analysis for earthquake return periods of 2,500 yrs and 10,000 yrs, respectively. The results of the analysis predicted disposal unit cracking to occur at construction joints, which are located on 9 m (30 ft) centers in the base slab and walls. Cracking in the much weaker saltstone monolith was predicted to follow the joint-initiated disposal unit cracking.

[33] Disposal unit designations used in the TER correspond to designations used in the 2009 PA (Figure 2.8-8), which may not correspond to current designations.

Table 5.1-1: Summary of predicted static settlements for the FDC 2A/2B project site (adapted from K-ESR-Z-00001; Section 5.2.4)

Phase	Minimum Settlement (in)	Maximum Settlement (in)	Average Settlement (in)
Immediately after operations completed	2	17	4
Immediately after closure cap completed	5	13	7
30 years after closure cap completed	6	18	9

2.5 cm are in one inch.

DOE conducted geotechnical investigations for FDC 2A/2B and 7A – D (Figure 2.8-8) to evaluate the settlement of the subsurface layers due to static loading (i.e., the presence of disposal units, saltstone grout, and the closure cap) and the dynamic settlement due to potential liquefaction from earthquakes and the settlement due to the compression of the soft zones (K-ESR-Z-00001; K-ESR-Z-00002). Based on the similarities of the DWPF to the SDF, DOE estimated the total static settlement for the FDC project sites (Table 5.1-1 and 5.1-2). Differential settlement at the various phases was estimated to be of the same magnitude as the average uniform settlement in Table 5.1-2 (K-ESR-Z-00002). Estimated dynamic settlement for FDCs 2A/2B due to liquefaction from a design basis earthquake with a return period of 2500 yrs and a peak ground acceleration (PGA) of 0.21 g ranged from 0.0 cm to 5.8 cm (0.0 in to 2.3 in). The maximum surface settlement resulting from the compression of the 4.3 m (14 ft) thick soft zone was calculated to be approximately 1 cm (0.5 in) (K-ESR-Z-00001). The estimated dynamic settlement for FDCs 7A - D due to liquefaction from an earthquake with a PGA of 0.20 g ranged from 0.8 cm to 5.1 cm (0.3 in to 2 in). Although no significant soft zones were identified beneath the FDCs 7A - D sites, the soft zone configuration from the FDC 2A/2B site was used in a conservative estimate of potential settlement from compression of soft zones, which similarly resulted in a maximum estimated surface settlement of 1 cm (0.5 in) (K-ESR-Z-00002). In response to NRC comment VP-1, DOE stated that seismic events and differential settlement are incorporated into the design calculations for the FDCs (SRR-CWDA-2010-00033).

Table 5.1-2: Summary of predicted static settlements for the FDC 7A – D[34] project site (adapted from K-ESR-Z-00002; Table 5)

Phase	Minimum Settlement (in)	Maximum Settlement (in)	Average Settlement (in)
Immediately after operations completed	2	16	4
Immediately after closure cap completed	3	13	9
30 years after closure cap completed	5	18	11

2.5 cm are in one inch.

5.2 NRC Evaluation – Site Stability

The NRC staff agrees that stabilizing salt waste with grout and completely filling the disposal units will provide a monolithic structure, minimize void space, and prevent collapse and differential settlement that could occur due to consolidation of the waste. Thus, the main processes evaluated with respect to site stability at the SDF are earthquakes, floods, erosion, and settlement. The NRC staff determined that the dynamic settlement expected to be caused by an earthquake with a 10,000 yr return period is unlikely to cause significant disruption to the SDF. Similarly, the staff concludes that floods are unlikely to disrupt the SDF because the 10,000 yr flood level for the Upper Three Runs basin near the SDF is significantly below the lowest planned elevation of a disposal unit at the SDF. As discussed in Section 2.4, NRC staff concluded that the closure cap can provide adequate long-term erosion protection.

With respect to settlement due to static loading, the NRC staff concluded DOE's approach to the geotechnical investigations at the SDF documented in K-ESR-Z-00001 and K-ESR-Z-00002 is reasonable, however, it is not clear that the conclusions of these investigations are consistent with the assumptions in the PA. NRC staff requested additional information in IEC-5 regarding the assumption that settlement due to static loading will only be a few inches and will be uniformly distributed over the closure cap (NRC, 2010b; RAI-2009-01). In response, DOE stated that the maximum static settlement estimated for the F-Area closure cap is estimated to be 5.6 cm (2.2 in) at 10,000 yrs post closure (SRR-CWDA-2010-00033). As the estimated pressure from the planned F-Area closure cap is more than twice the pressure of the estimated SDF closure cap, DOE concluded that a range of 5 cm to 8 cm (2 in to 3 in) is conservative. In addition, DOE stated that the assumption of uniform settlement was supported by the differential settlement at FTF of less than 0.3 cm (0.1 in) for any given tank. However, the predicted settlement for FDCs 2A/2B and 7A – D (Figure 2.8-8) in SDF-specific calculations (Table 5.1-1 and 5.1-2) exceed these values, and also exceed the values assumed in *Saltstone Disposal Facility Closure Cap Concept and Infiltration Estimates* (WSRC-STI-2008-00244). Because the

[34] DOE has changed the numbering of the FDCs since the PA was published. These FDCs are now referred to as 2A/2B, 3A/3B, and 5A/5B.

magnitude of settlement (uniform and differential) assumed in the performance assessment (Section 6.1 of WSRC-STI-2008-00244) was significantly less than the settlement predicted in recent investigations, the NRC staff finds that DOE has not shown whether settlement due to static loading may disrupt the performance of the synthetic materials and drainage layers. In addition, relevant American Concrete Institute (ACI) standard (ACI-372) for the FDCs suggests the maximum recommended limits for uniform and differential settlement are 15 cm (6 in) and 5.3 cm (2.1 in), respectively, which are also exceeded by the predicted settlement for FDCs 2A/2B and 7A – D. Because neither the standard nor the DOE analysis addresses the consequences of exceeding these criteria, the risk-significance of this finding is unclear. Accordingly, the NRC staff will monitor the development of information related to settlement of the SDF due to static loading.

The NRC staff also requested additional information on the implications of calcareous material in the subsurface at SRS and how these features were addressed in the PA (NRC, 2010i; PA-14). With respect to site stability, DOE referenced a geotechnical evaluation report (K-ESR-Z-00002), which considered the effects of consolidation of the soft zones at the SDF (SRR-CWDA-2011-00044). However, DOE's analyses of the potential settlement due to consolidation of these zones do not account for the potential removal of subsurface material which has resulted in subsidence observed at SRS. There is evidence of sinks near the SDF that are much more significant than DOE's analysis of consolidation of the current soft zones suggests. It is not clear whether the process of sinkhole formation will continue. In response to NRC comment PA-14, DOE discussed the hypothesis that soft zone formation occurred by the dissolution of carbonate material by meteoric water when the Santee Formation was located in the vadose zone (SRR-CWDA-2011-00044). The potential for mass removal of carbonate material leading to subsidence within 100 years was dismissed due to the Santee formation currently being located beneath the water table, in a relatively stable geochemical environment (WSRC-TR-99-4083). Due to the location of the soft zones at the SDF (30 – 37 m BGS [100 – 120 ft BGS]), the NRC staff does not expect the closure cap to result in the lowering of the water table to below the calcareous sediment, which would likely accelerate the dissolution of this material. However, evidence of dissolution based on elevated bicarbonate ion concentrations and relatively high pH values for groundwater samples collected in or near the Santee formation (WSRC-RP-92-450), demonstrates an ongoing evolution of the subsurface (USACE, 1952). Although dissolution of the calcareous sediment in the saturated zone is likely to be a very slow process, it has not been demonstrated that dissolution is insignificant with respect to site stability over the course of a 10,000 yr performance period. For example, subsidence beneath the SDF could result in fracturing in the saltstone grout and disposal units and increased localized infiltration because of run in from the surrounding area. Accordingly, the NRC staff will monitor the development of information related to the potential for sink development at the SDF. The NRC staff also will monitor the development of model support for the assumption that reasonably predicted future dissolution of calcareous sediment is insignificant to site performance.

The disposal of salt waste as solidified saltstone in combination with the erosion controls DOE has included in its closure cap design provides reasonable assurance that salt waste disposal at the SDF will meet the site stability performance objective (§61.44). The use of saltstone grout will limit void space in the disposal units and therefore the adverse effects of differential settlement of the waste form will be avoided (e.g., collapse of the disposal unit). However, the NRC staff has identified concerns related to site stability that could affect the long-term performance of the SDF with respect to §61.41. Based on the uncertainty in the potential risk associated with settlement due to static loading and the calcareous zones, these aspects of site stability will be included in NRC's revised monitoring plan.

6. Conclusions

The NRC staff reviewed the *2009 Performance Assessment for the Saltstone Disposal Facility at the Savannah River Site*, (2009 PA) dated October 2009 (SRR-CWDA-2009-000017), and associated documentation. The 2009 PA is an update to DOE's January 31, 2006 PA performed in support of the *Section 3116 Determination, Salt Waste Disposal, Savannah River Site* (DOE-WD-2005-001 Rev. 0). In its December 2005 TER (NRC, 2005) documenting NRC's review of the PA in support of DOE's draft Section 3116 determination for the SDF (DOE-WD-2005-001 Rev. 1), the NRC staff concluded that it had reasonable assurance that salt waste disposal at the SDF would meet the performance objectives of 10 CFR 61 provided certain assumptions in DOE's analyses were verified during monitoring. During the current review, the NRC staff carefully evaluated information related to these assumptions (i.e., information regarding saltstone oxidation, saltstone and disposal unit hydraulic conductivity, field-scale properties of as-emplaced saltstone, saltstone fracturing, numerical modeling of flow through fractures, radionuclide concentrations, moisture characteristic curves, and erosion control), as well as new factors of importance to the modified disposal plans and revised conceptual model.

The NRC staff concludes it has reasonable assurance that waste disposal at the SDF meets the 10 CFR 61 performance objectives for protection of individuals against intrusion (Chapter 3), protection of individuals during operations (Chapter 4), and site stability (Chapter 5). However, the staff no longer has reasonable assurance that DOE's disposal activities at the SDF meet the performance objective for protection of the general population from releases of radioactivity[35]. Based on NRC's review of the DOE PA and its own independent analyses, the NRC staff concludes that predicted doses in many of the scenarios the NRC staff considers reasonable fall within a range of approximately 0.25 mSv/yr (25 mrem/yr) [the limit established in the §61.41 performance objective] to approximately 1 mSv/yr (100 mrem/yr) [the public dose limit found in §20.1301]. Thus, although the NRC staff cannot conclude that the performance objective in §61.41 is met, the potential dose to an off-site member of the public from DOE's disposal actions is still expected to be relatively low. Furthermore, the staff expects that any exceedance of the §61.41 limit would occur long after site closure. The NRC staff has identified additional information DOE could develop and mitigative actions DOE could take in the short term that might provide reasonable assurance that salt waste disposal at the SDF meets the 10 CFR 61 Subpart C performance objectives. This information and these actions are described as part of the Key Monitoring Factors NRC plans to use as a basis for further SDF monitoring (Section 6.2 and Table A-1). As described in Section 6.3, the NRC staff will review information DOE provides during monitoring and determine if the information addresses the technical concerns identified by the staff in the TER.

[35] As discussed further in Section 2.13, NRC has evaluated compliance using a dose limit of 0.25 mSv/yr (25 mrem/yr) Total Effective Dose Equivalent (TEDE) consistent with the approach discussed in the Final Rule for Disposal of High-Level Radioactive Wastes in a Proposed Geologic Repository at Yucca Mountain, Nevada 66 FR 55752. DOE and NRC have used the updated internal dose factors of ICRP Publication 72 (ICRP, 1996), consistent with Commission direction in SRM-SECY-01-0148 (NRC, 2002).

6.1 Compliance with the Performance Objectives of 10 CFR Part 61, Subpart C

6.1.1 Protection of the General Population from Releases of Radioactivity

In the PA and in subsequent RAI responses (SRR-CWDA-2011-00044, PA-8), DOE indicated that it believes Case A is an appropriate base case because it reflects DOE's expected site performance. Based on this deterministic Case A analysis DOE's expected peak dose to a member of the general population from the SDF is 0.014 mSv/yr (1.4 mrem/yr) within 10,000 years of site closure and 0.31 mSv/yr (3.1 mrem/yr) within 20,000 years of closure. As described in this TER (Sections 2.1 and 2.13), the NRC staff does not find Case A to be an appropriate case to compare to the performance objectives because it does not accurately reflect current site conditions, does not account for the full ranges of measured values of key parameters or expected differences between laboratory and as-emplaced values, and does not appropriately account for potential changes in parameter values with time.

In response to NRC concerns about Case A, DOE supplied Cases K, K1, and K2. The main differences between these cases and Case A include (1) consideration of increased saltstone hydraulic conductivity, (2) the assumption of oxidation of saltstone proceeding from fractures for the purpose of modeling Tc release, (3) the assumption that the relative permeability in the saltstone and disposal unit concrete is always 1, (4) increased degradation of disposal unit concrete, (5) reduced inventory of Ra-226 and its ancestors Th-230, U-234 and Pu-238, and (6) updated biosphere modeling (Table 2-1). Cases K, K1, and K2 differ from one another only in the K_d values used to represent Tc sorption in oxidizing and reducing saltstone and disposal unit concrete. Of these three cases, the NRC staff concludes Case K1 uses the best-supported K_d values. The NRC staff disagrees with DOE's characterization of Cases K, K1, and K2 as overly-pessimistic sensitivity analyses (Section 2.13.3.4). Because it resolves many of the concerns NRC identified with Case A, the NRC staff finds Case K1 is more appropriate for determining whether salt waste disposal at the SDF meets the performance objectives than Case A is. Therefore, the NRC staff has relied heavily on Case K1 in its review.

DOE predicts Case K1 will lead to a dose to an off-site member of the general public of approximately 0.9 mSv/yr (90 mrem/yr) at approximately 12,900 years after site closure. However, the timing of this peak dose is very sensitive to, and delayed by the following factors:

(1) the assumed timing and rate of saltstone fracture growth,

(2) use of an average K_d value to track Tc release from saltstone instead of tracing release from oxidized and reduced regions separately, and

(3) seemingly overly-optimistic assumptions about Tc re-reduction and re-immobilization in disposal unit floors and walls (see Sections 2.13.3.2 and 2.13.3.3).

Because of the large uncertainty in the predicted timing of the 0.9 mSv/yr (90 mrem/yr) peak dose from Tc-99 and the expectation that the predicted peak in DOE's Case K1 model was delayed by unsupported assumptions, the NRC staff could not conclude it had reasonable assurance that the dose would meet the 0.25 mSv/yr (25 mrem/yr) limit for 10,000 years after site closure based on DOE's Case K1 results.

Rather than base the compliance demonstration on DOE's Case K1 alone, the NRC staff performed additional analyses to determine whether it could reduce the uncertainty in the timing or magnitude of the Case K1 peak dose. However, as explained in Section 2.13.3.3, NRC modifications of DOE's Case K1 PORFLOW™ model that were expected to lower the predicted dose led to predicted Tc-99 peak doses greater than 0.25 mSv/yr (25 mrem/yr) at approximately 10,000 years after site closure (Figure 2.13-1). Most of these predicted doses fell within a range of approximately 0.25 mSv/yr (25 mrem/yr) (the §61.41 performance objective) to approximately 1 mSv/yr (100 mrem/yr) total effective dose equivalent (the public dose limit found in §20.1301. As described further in Section 2.13.3.3, NRC evaluations of a simplified dual-K_d release model, which the NRC staff developed to avoid an artifact of DOE's average-K_d model that increases the dose in DOE's Case K1,also led to annual fractional release rates for Tc-99 similar to DOE's results (Appendix B).

Because the staff expects that any exceedance of the §61.41 limit would occur long after site closure, the staff has not identified an immediate safety concern at the SDF. The NRC staff has identified additional information DOE could develop and mitigative actions DOE could take in the short term that might provide reasonable assurance that salt waste disposal at the SDF meets the 10 CFR 61 Subpart C performance objectives. The NRC staff has described this information in its Key Factors it plans to use as a basis for further SDF monitoring (Section 6.1.3 and Table A-1).

6.1.2 Protection of Individuals from Inadvertent Intrusion

As described in Chapter 3, the NRC staff finds that the scenarios and pathways analyzed by DOE for the assessment of the performance objective in §61.42 (protection of an inadvertent intruder) are appropriate based on the regional practices near SRS. Specifically, DOE expects an individual is unlikely to inadvertently intrude into a disposal unit (e.g., by drilling a well) because the cementitious disposal unit and waste form, even at long times after closure, would be substantially different than and more difficult to disturb than the natural soil in the region. Therefore, DOE assumed an inadvertent intruder would drill a well near, rather than through, a disposal unit. In a sensitivity analysis, DOE also evaluated the dose to an individual who drills into a disposal unit and spreads drill cuttings on the land surface. The NRC staff finds this approach to be reasonable.

DOE evaluated the doses to an acute and chronic intruder and found the chronic intruder dose to be greater. DOE estimated a peak chronic intruder dose within 10,000 years based on Case A to be 0.019 mSv/yr (1.9 mrem/yr) and a peak chronic intruder dose based on Case K within 20,000 years[36] to be 2.9 mSv (290 mrem/yr). As explained in more detail in Chapter 3, the NRC staff estimated a peak chronic intruder dose based on DOE's Case K1 to be

[36] As discussed in Section 2.13, because of the uncertainty in the timing of fracturing and the resulting uncertainty in radionuclide release from Cases K, K1, and K2, the NRC staff has determined that doses DOE predicts to occur within 20,000 years after site closure may occur within 10,000 years of closure.

approximately 4.2 mSv/yr (420 mrem/yr). Because this dose is dominated by groundwater pathways, it is subject to the same optimisms and pessimisms described for the Case K1 dose to an off-site member of the public (Section 2.13.3). Specifically, it appears to be artificially increased by DOE's average-K_d model but unjustifiably diminished by unsupported credit for chemical retention of Tc in the disposal unit concrete. In Case K1, these factors appear to have similar effects on dose (Section 2.13.3.3). In addition to the assumptions affecting the dose to an off-site member of the public, the chronic intruder dose incorporates apparently conservative assumptions. Specifically, the NRC staff concludes that the Case K1 chronic intruder dose predicted by DOE is likely to be conservative (i.e., overestimate the potential intruder dose) for the following reasons: (1) the rate of fracturing and final fracture spacing may be less than assumed by DOE in Case K1, which would lower expected groundwater concentrations (Section 2.13.3.3) and, (2) radionuclides are likely to encounter more dilution in the aquifer than assumed by DOE in the intruder analysis. The NRC staff therefore concludes that the dose to an inadvertent intruder through the groundwater pathways is likely to be less than 5 mSv/yr (500 mrem/yr), provided that key assumptions in the Case K1 analysis are true. These key assumptions are identified as monitoring issues (Section 6.2 and Table A-1).

In the DOE drill cuttings scenario sensitivity analyses, the doses to acute and chronic intruders who intrude directly into saltstone after 500 years were estimated to be 0.038 mSv/yr (3.8 mrem/yr) and 0.15 mSv/yr (15 mrem/yr) respectively. The dose prior to 500 years could be significantly higher due to the presence of short lived radionuclides, but intrusion during this time is not likely due to the expected performance of the erosion barrier layer in the closure cover that acts as an intruder deterrent and robust disposal unit roofs and clean cap layers. The results of these sensitivity cases indicate that the additional dose an intruder would receive from being exposed to drill cuttings containing saltstone would not be significant compared to the dose to an intruder from using groundwater on site, which the NRC staff expects to meet the 5 mSv/yr (500 mrem/yr) dose limit specified in §61.42. Therefore, the NRC staff concludes that the dose to an intruder who drills directly into saltstone meets the performance objective for protection of an inadvertent intruder.

6.1.3 Protection of Individuals during Operations

In the 2005 TER (NRC, 2005), the NRC concluded that, during operations, individuals are protected by DOE regulations that provide protection comparable to 10 CFR 20. Since that time, the NRC staff has conducted two onsite observations of the SDF related to this performance objective. NRC staff reviewed aspects of DOE's Radiation Protection Program (RPP) related to (1) worker dose monitoring and (2) groundwater and air effluent monitoring. The conclusion of both of the onsite observations was that DOE has an adequate RPP in place for SDF operations. Based on the results of the NRC's review in its 2005 TER (NRC, 2005), onsite observations, and NRC's review of the annual SRS Environmental Reports the NRC staff has reasonable assurance that DOE's disposal actions at the SDF meet the §61.43 performance objective for protection of individuals during operations. NRC will continue to assess DOE's radiation protection program through future monitoring activities.

6.1.4 Stability of the Disposal Site after Closure

The NRC Staff concludes the saltstone waste form will provide a monolithic structure, minimize void space, and prevent collapse and differential settlement that could occur due to consolidation of the waste. Thus, the main processes evaluated with respect to site stability at the SDF are earthquakes, floods, erosion, and settlement. NRC staff evaluated the dynamic settlement expected to be caused by an earthquake with a 10,000 year return period and found it was unlikely to cause significant disruption to the SDF (Section 5.1). Similarly, the staff determined that floods are unlikely to disrupt the SDF because the 10,000 year flood level for the Upper Three Runs basin near the SDF is significantly below the lowest planned elevation of a disposal unit at the SDF (Section 5.1). Although the design for the infiltration and erosion control cover will be made final closer to the time of site closure, the NRC staff concluded that, if implemented as preliminarily designed, the cap will include enough depth and a sufficiently robust erosion control layer to prevent exposure of the saltstone waste.

As compared to the potential effects of earthquakes, floods, and erosion, there is more uncertainty in the potential effects of static settlement due to loading of the subsurface layers and settlement due to calcareous zones present or potentially developing under the SDF. With respect to static settlement, recent studies predict greater static settlement in the SDF than addressed by DOE in the PA (Chapter 5). Because updated modeling may be needed to account for the new information in the PA, this item has been identified as a monitoring factor.

Much of SRS, including the SDF, is underlain by calcareous sediment in the Santee formation resulting in the presence of “soft zones.” Historically, some of these zones have consolidated, resulting in depressions on the land surface. An evaluation of aerial photography of the SRS illustrated the potential presence of several basins and sinks located in and near the SDF. Although the potential for mass removal of carbonate material leading to subsidence within 100 years was dismissed due to the Santee formation being located beneath the water table, it is unclear whether there could be a change in the water table elevation leading to additional mass removal within 10,000 years of site closure. The potential for sinks to form in the future cannot be ruled out because of these site features: (2) the presence of calcareous material in the SDF subsurface, (2) groundwater samples indicating ongoing dissolution of calcareous material near the SDF, and (3) evidence of sinks in the surrounding area.

To support its analysis of site stability, DOE noted that the average thickness of a soft zone at the SDF is generally a few feet, with a postulated maximum lateral dimension of about 3 m to 6 m (10 ft to 20 ft). DOE concluded that consolidation of such a soft zone would cause a maximum differential settlement of approximately 1 cm (0.5 in), which DOE concluded would have minimal effects on the stability of a disposal unit (Chapter 5). In contrast, observed sinks at the SRS site are typically 3 m to 5 m (10 ft - 15 ft) deep and in F-Area, up to 300 m (1,000 ft) long. A sink of this magnitude, if it were to occur at the SDF, could result in fracturing in the saltstone grout and disposal units and increased localized infiltration because of run in from the surrounding area. Although sinks have been identified in the surrounding area, the time over which the sinks developed is unclear. Thus the likelihood that a sink could develop at the SDF within 10,000 years after disposal also is unclear. Because of the significant uncertainty in the potential for settlement due to consolidation of soft zones under the SDF, the development of additional information about the potential effects of calcareous zones on site stability has been identified as a key factor for SDF monitoring (Section 6.2 and Table A-1).

The disposal of salt waste as solidified saltstone in combination with the erosion controls DOE has included in its closure cap design provides reasonable assurance that salt waste disposal at the SDF will meet the site stability performance objective (§61.44). The use of saltstone grout will limit void space in the disposal units and therefore the adverse effects of differential settlement of the waste form will be avoided (e.g., collapse of the disposal unit). However, the NRC staff has identified concerns related to site stability that could affect the long-term performance of the SDF with respect to §61.41. Based on the uncertainty in the potential risk associated with settlement due to static loading and the calcareous zones, these aspects of site stability will be included in NRC's revised monitoring plan.

6.2 Key Monitoring Factors

Based on its review of DOE's updated PA (SRR-CWDA-2009-000017) and associated documentation, as well as its independent analyses, the NRC staff has identified factors that are important to assessing whether DOE's disposal actions meet the performance objectives of 10 CFR 61, Subpart C. Many, though not all, of these factors are similar to factors identified in the NRC staff's 2005 TER for salt waste disposal at the SDF (NRC, 2005), which the staff used as the basis of its current monitoring plan (NRC, 2007). Additionally, DOE has identified areas of ongoing and future work that are similar to many of these factors (PA Section 8.2 and SRR-CWDA-2011-00052). Key areas of interest continue to be waste form and disposal unit degradation, the effectiveness of infiltration and erosion controls, and estimation of the radiological inventory. In addition, additional monitoring factors related to site stability will be part of monitoring as a result of this review. The factors developed during this review will serve as the basis for a revised monitoring plan. A summary of the monitoring factors for the key areas of the PA is provided below. A more extensive list of the monitoring factors is provided in Appendix A.

Inventory:	NRC will monitor the inventory in Vault 1, Vault 4, and in each FDC in comparison to the values in Table 2-2 (with the exception of Ra-226 and Th-230). NRC will monitor the inventory of Ra-226 and Th-230 as compared to the values in Table 2-3. NRC will also monitor the methods DOE uses to assess radionuclide inventories.
Infiltration and Erosion Control:	NRC will monitor issues related to long-term closure cap performance, including any changes to the design or planned implementation of the closure cover, as its design develops and is made final closer to the time of site closure. NRC will monitor the development of model support for the

performance of the lower lateral drainage layer. NRC will monitor updates to DOE's assessment of the effects of settlement on the long-term performance of the composite layer in the closure cap, the composite layers overlying the FDC roofs, and the lateral drainage layers.

Waste Form Hydraulic Performance: NRC will monitor the hydraulic conductivity in field-emplaced saltstone samples, the potential variability of field-emplaced saltstone properties and DOE's quality assessment program for grout. NRC also will monitor the applicability of data measured using laboratory-produced samples to field-emplaced saltstone and the effect of the curing temperature profile on the hydraulic properties of field-emplaced saltstone.

Waste Form Physical Degradation: NRC will monitor the development of model support for assumptions about saltstone fracturing.

Waste Form Chemical Performance: NRC will monitor the radionuclide release from field-emplaced saltstone, the chemical reduction of Tc in saltstone, the reducing capacity of saltstone and the expected evolution of redox conditions in saltstone. NRC also will monitor the sorption of Ra-226, Se-79, and Sr-90 in saltstone and the potential for an initial short-term release from saltstone (i.e., a rinse-release).

Disposal Unit Performance: NRC will monitor the sorption of Tc-99, Ra-226, and Se-79 in disposal unit concrete. NRC will also monitor the development of model support for the long-term physical integrity of cementious materials forming most of the disposal unit structure as well as non-cementitious materials (e.g., epoxy, and neoprene seals) used in disposal unit joints.

Subsurface K_d Values: NRC will monitor the K_d value assumed for Se-79 in sand and clay.

Environmental Monitoring: NRC will monitor results from leak detectors installed under select FDCs, and NRC will continue to review groundwater monitoring data collected near the SDF.

Radiation Protection Program: NRC will continue to monitor annual worker dose reports and the results of DOE's air monitoring program.

Site Stability: NRC will monitor the development of model support for the potential settlement of the SDF due to the consolidation of subsurface layers and potential formation of sinks under the SDF.

In addition to the monitoring factors, NRC staff identified a number of items for review when the PA is revised as part of DOE's PA maintenance program (Table A-2). These items include aspects of model development (e.g., consideration of alternate conceptual models, evaluation of intermediate results), the diffusivity in saltstone, model support for the use of moisture characteristic curves, parameters related to the far-field flow model, and biosphere parameters. NRC staff considers these items to be potentially important to the predicted dose, though these items are not as risk-significant as the key monitoring factors.

This is not an all-inclusive list of factors that may need to be monitored by the NRC to assess compliance with the performance objectives, but rather is based on DOE's current planned approach and NRC's current analysis of DOE's approach. Therefore, factors to be monitored may change as DOE implements its disposal plans. NRC staff will revise its plan to monitor DOE's disposal actions at the SDF in coordination with SC DHEC. This revised plan will present the details of NRC's planned future monitoring activities.

6.3 Path Forward

Section 3116(b) of the NDAA requires the NRC to monitor certain disposal actions taken by the DOE for the purpose of assessing compliance with the performance objectives set out in 10 CFR Part 61, Subpart C. NRC monitoring activities of the Saltstone facility began following the 2007 completion of the *U.S. Nuclear Regulatory Commission Plan for Monitoring the U.S. Department of Energy Salt Waste Disposal at the Savannah River Site in Accordance with the National Defense Authorization Act for Fiscal Year 2005*, and have included a range of onsite observations, technical reviews, and meetings with DOE related to assessing DOE's compliance with the 10 CFR Part 61, Subpart C performance objectives. The NDAA also requires that NRC report any noncompliance to Congress, the State, and the DOE as soon as practicable after discovery of the noncompliant conditions and states that NRC's monitoring is subject to judicial review.

As stated in NUREG-1854, there are three primary reasons that DOE disposal actions could be found noncompliant: (1) if there are sufficient indications that the requirements of the performance objectives are currently not being met, (2) if there are sufficient indications that there is no longer reasonable assurance that the dose limits specified in the performance objectives will be met in the future, or (3) if key aspects relied upon to demonstrate compliance with one or more performance objectives are no longer supported due to the lack of supporting information obtained during the monitoring period. As documented and explained in the TER, the NRC is not stating that releases have occurred from the disposal facility that could lead to annual doses that exceed the limits established in §61.41 (item 1 listed above). The NRC staff is, however, concerned that (i) information collected during the monitoring period (e.g., hydraulic conductivity assessments, Tc sorption measurements) does not support DOE's compliance demonstration and (ii) sufficient information has not been provided to support many key modeling assumptions relied on for performance (items 2 and 3 above).

Per the results of the NRC staff's review, and in accordance with the reasons listed above for identifying reasons for noncompliance, the NRC is sending a letter of concern to both DOE and the SC DHEC so that SC DHEC is kept informed and has an opportunity to provide input and comments, and to provide DOE with an opportunity to furnish information that demonstrates its disposal actions are in compliance with the performance objectives. For example, DOE may present new or additional information or make design changes that would enable NRC to conclude with reasonable assurance that salt waste disposal at the SDF meets the 10 CFR 61 performance objectives (Appendix A). If the staff determines that, based on the information provided, there is a sufficient basis to conclude that the performance objectives are met, NRC will send a notification of resolution letter. This letter and the potential resolution letter will be made publicly available on NRC's website as they formally document NRC's concern and its resolution.

If, after having reviewed any additional information received from DOE, the staff determines that it still cannot conclude there is reasonable assurance the performance objectives will be met, NRC will issue a noncompliance notification letter to the DOE in accordance with the NDAA. Also in accordance with the NDAA, the NRC is required to inform DOE, the covered State, and Congress if it considers any of DOE's waste disposal actions to be noncompliant with the performance objectives of 10 CFR 61, Subpart C.

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Appendix A Monitoring Factors

Table A-1: Key Monitoring Factors

Monitoring Topic	Performance Objective	Factor	Basis	TER Section
Inventory	§61.41 and §61.42	NRC will monitor the inventory in Vault 1, Vault 4, and in each FDC in comparison to the values in Table 2.2-1, with the exception of Ra-226 and Th-230. NRC will monitor the inventory of Ra-226 and Th-230 as compared to the values in Table 2.2-2.	The distribution of inventory among the disposal units will be important in determining potential doses to both an off-site member of the public and an inadvertent intruder. Because a single FDC can dominate the dose to an inadvertent intruder, and because the projected intruder dose was a significant fraction of the 5 mSv/yr (500 mrem/yr) limit, each FDC inventory must be consistent with PA assumptions (2009 PA) for NRC to retain reasonable assurance disposal meets the performance objective for intruder protection. Because a small number of FDCs can dominate the dose to an off-site member of the public, the inventory in individual FDCs also is relevant to protection of the general public. Although certain disposal units dominate the dose projected in DOE's updated performance assessment, the location of the peak dose and degree of plume overlap can change with changing assumptions about far-field transport.	2.2
Inventory	§61.41 and §61.42	NRC will monitor the methods DOE uses to assess radionuclide inventories.	Because of the considerable uncertainty in inventory estimates and the importance of radionuclide inventory to dose to an off-site member of the public as well as an inadvertent intruder, NRC will monitor the methods DOE uses to assess radionuclide inventories. NRC will focus on radionuclides that are currently identified as risk significant as well as relevant ancestors (e.g., Tc-99, Ra-226, Th-230, I-129) but also will consider radionuclides that could become more risk significant if the inventory increases significantly or if modeling assumptions change (e.g., Se-79). In particular, NRC will monitor assumptions about the inventory of Th-230 in tanks that are known to have Th-230 bearing waste and any methods used to estimate the concentration of Th-230 when Th-230 is below DOE's detection limit.	2.2

Monitoring Topic	Performance Objective	Factor	Basis	TER Section
Infiltration and Erosion Control	§61.41 and §61.42	NRC will monitor updates to DOE's assessment of the effects of settlement on the long-term performance of the composite layer in the closure cap and overlying the FDC roofs.	Recent studies predict greater settlement in the SDF than addressed by DOE in the PA. Because settlement has the potential to disrupt the composite layer in the closure cap and the lateral drainage layers that play a key role in diverting water around the FDCs, NRC will monitor updated DOE evaluations of the expected effects of settlement on composite layers in the closure cap and overlying the FDC roofs.	2.4, 2.5, and 5
Infiltration and Erosion Control	§61.41, §61.42, and §61.44	NRC will monitor issues related to long-term closure cap performance.	DOE expects the SDF closure cap to limit infiltration and erosion for thousands of years after site closure. Because the cap design will be made final closer to the time of site closure, NRC will monitor whether any design changes or specific aspects of the cap implementation (e.g., sources of erosion barrier source material) are expected to affect the predicted cap performance assumed in the PA. Before closure cap installation, additional information is needed about the potential effects of head build up above the composite layer on slope stability, the potential for gully formation due to the cumulative effects of smaller, more frequent flood events, and the predicted hydraulic conductivity of the foundation layer. Additional issues may be identified as the cap design develops.	2.4
Waste Form Hydraulic Performance	§61.41 and §61.42	NRC will monitor the measured values of hydraulic conductivity in field-emplaced saltstone samples, values from other representative studies, and the appropriateness of any new sampling technique.	The hydraulic performance of field-emplaced saltstone is critical to adequate SDF performance. Recently DOE has measured saturated hydraulic conductivity values in saltstone in a range greater than assumed in the PA. Because of DOE concerns about the sampling technique, DOE is developing new methods to sample field-emplaced saltstone. This issue previously has been tracked as part of Open Issue 2007-1.	2.6

Monitoring Topic	Performance Objective	Factor	Basis	TER Section
Waste Form Hydraulic Performance	§61.41 and §61.42	NRC will monitor the potential variability of as-emplaced saltstone properties and DOE's quality assessment program for grout.	Variations in the composition of saltstone grout produced at the SPF and emplaced in the disposal units (e.g., variations in the water-to-cementitious material ratio, aluminate concentration, presence of admixtures) may affect hydraulic properties of saltstone grout. Because of the sensitivity of dose predictions to saltstone hydraulic properties, quality assurance for the emplaced saltstone waste form will be important for building reasonable assurance in SDF performance. This factor includes both determining the potential variability in the hydraulic properties due to variations in the saltstone composition and ensuring that factors determined to significantly affect the hydraulic properties are well controlled in the production of saltstone. This factor is related to issues previously tracked as Open Issue 2007-1 and Open Issue 2007-2.	2.6
Waste Form Hydraulic Performance	§61.41 and §61.42	NRC will monitor the applicability of data on the hydraulic properties measured using laboratory-produced samples to ensure that it adequately reflects the properties of field-emplaced saltstone.	Because laboratory-produced samples have been used as a basis for the value of the saltstone saturated hydraulic conductivity and diffusivity used in the PA, the applicability of the measurements made using laboratory-produced samples to field conditions could affect dose predictions. For example, it is important to consider the potential effects of scale, temperature, the presence of admixtures, and reducing conditions in the lab sample when estimating properties of as-emplaced saltstone from laboratory-produced samples.	2.6

Monitoring Topic	Performance Objective	Factor	Basis	TER Section
Waste Form Hydraulic Performance	§61.41 and §61.42	NRC will monitor the effect of the curing temperature profile on the hydraulic properties of as-emplaced saltstone.	DOE studies have demonstrated the potential importance of the curing temperature profile to the saturated hydraulic conductivity of saltstone. Saltstone hydraulic conductivity, in turn, significantly affects dose predictions. It is therefore important to understand the potential effect of curing temperature on the hydraulic properties and to verify that the cure temperatures in the saltstone monolith will not result in a waste form that has higher hydraulic conductivities.	2.6
Waste Form Physical Degradation	§61.41 and §61.42	NRC will monitor the development of model support for assumptions about saltstone fracturing.	DOE evaluations of alternative cases and NRC independent analyses indicate the sensitivity of dose projections to the assumed rate and extent of saltstone fracturing because of effects on saltstone oxidation as well as water flow through the disposal system.	2.6
Waste Form Chemical Performance	§61.41 and §61.42	NRC will monitor whether measured radionuclide release from samples of field-emplaced saltstone are consistent with assumptions used in the PA.	Because of the importance of radionuclide release rates to the projected dose, it will be important to determine whether the release of radionuclides from field-emplaced saltstone is consistent with assumptions in the PA. This issue previously has been tracked as part of Open Issue 2007-2. Additionally, leaching experiments conducted to date have reflected the bulk constituents of saltstone and simulated waste, but have not included admixtures used in the production of saltstone at the SPF. Because certain admixtures, such as such as the anti-foam agent Tributyl Phosphate (TBP) may form chemical complexes with radionuclides that limit radionuclide sorption or increase solubility, it will be important to determine whether radionuclide leaching from samples containing these admixtures is consistent with assumptions in the PA.	2.6

Monitoring Topic	Performance Objective	Factor	Basis	TER Section
Waste Form Chemical Performance	§61.41 and §61.42	NRC will monitor the development of a robust demonstration that saltstone will chemically reduce Tc(VII) to Tc(IV) to the extent assumed in DOE's PA model under the range of conditions to which saltstone is expected to be subjected during the compliance period.	Studies DOE has relied on to demonstrate Tc retention in saltstone have included experimental artifacts [i.e., the presence of sodium thiosulfate or $H_2(g)$] that have made it difficult to interpret the results of these experiments. Based on these experiments, it is unclear whether saltstone itself can reduce Tc and maintain Tc in a reduced state. Furthermore, recent DOE studies have shown the sensitivity of Tc retention in saltstone to trace quantities of oxygen. The peak dose to an off-site member of the public and the inadvertent intruder is sensitive to Tc release, which in turn is sensitive to Tc redox state. For these reasons, there must be a robust demonstration of the ability of saltstone to reduce Tc and maintain Tc in a reduced state in environmental conditions similar to the expected environmental conditions of the emplaced waste form will be important to demonstrating there is reasonable assurance that SDF disposal meets the performance objectives for protection of a member of the general population. The reduction of Tc in saltstone previously has been tracked as Open Issue 2009-1.	2.6
Waste Form Chemical Performance	§61.41 and §61.42	NRC staff will monitor the development of additional information regarding the initial reducing capacity of saltstone as compared to the value assumed in the PA (i.e., 0.82 meq e^-/g) and the expected evolution of redox conditions over time.	The DOE Case K model and NRC sensitivity analyses demonstrate the importance of saltstone reducing capacity to the projected Tc release rate. However, it is unclear why the measured value of the specific reducing capacity of saltstone, which contains only 25% blast furnace slag, is equivalent to the measured reducing capacity of pure blast furnace slag. Additionally, there is uncertainty in the E_h transition times assumed in the PA, which affects the predicted release of redox sensitive radionuclides other than Tc (because they were modeled with a pore-volume step-change release model).	2.6 and 2.7

Monitoring Topic	Performance Objective	Factor	Basis	TER Section
Waste Form Chemical Performance	§61.41 and §61.42	NRC will monitor measurements of Ra-226, Se-79, and Sr-90 sorption in or leaching from saltstone.	Although the dose from Ra-226 dominates the dose from Case A and most of the DOE sensitivity analyses, DOE relied on measured values of Sr sorption to estimate Ra sorption in oxidizing saltstone in the PA. If Ra-226 continues to make a significant contribution to DOE's projected does from the SDF, it will be important to reduce the uncertainty in Ra sorption by collecting element-specific information about Ra sorption in and release from saltstone. Additionally, as discussed in Section 2.7, the NRC staff does not believe that there is an adequate basis for the K_d values DOE uses to represent Se or Sr sorption in saltstone. Lower saltstone K_d values could result in greater dose contributions to an off-site member of the public (from Se-79) or an inadvertent intruder (from Sr-90).	2.6 and 2.7
Waste Form Chemical Performance	§61.41 and §61.42	The NRC staff will monitor the development of support for the assumption that short-term rinse release of radionuclide's from saltstone seen in laboratory experiments will not significantly affect projected peak doses from groundwater pathways at the SDF.	Studies of Tc release from saltstone samples often demonstrate an initial relatively rapid release of Tc that is characterized as a "rinse-release" phenomenon and excluded from calculated release rates. If water is excluded from the SDF for extended periods after site closure, this rinse-release, if applicable to as-emplaced saltstone, would not occur until well after the sheet-drain system is grouted and closed.	2.7

Monitoring Topic	Performance Objective	Factor	Basis	TER Section
Disposal Unit Performance	§61.41 and §61.42	NRC will monitor the development of information about Ra and Se sorption in disposal unit concrete.	Although the dose from Ra-226 dominates the predicted dose in Case A and most of the DOE sensitivity analyses, DOE relied on measured values of Sr sorption as a surrogate for Ra sorption in the PA. If Ra-226 continues to make a significant contribution to DOE's projected does from the SDF, it will be important to reduce the uncertainty in Ra sorption by collecting radionuclide-specific information about Ra sorption in disposal unit concrete. Additionally, the NRC staff does not believe that there is an adequate basis for the K_d value DOE used in the PA to estimate Se sorption in disposal unit concrete.	2.5
Disposal Unit Performance	§61.41 and §61.42	NRC will monitor the development of information about Tc sorption in disposal unit concrete.	NRC analyses of intermediate results from DOE's Case K PORFLOW™ model demonstrate the importance of Tc retention in disposal unit concrete to the timing and magnitude of the dose from fractured saltstone.	2.7 and 2.13
Disposal Unit Performance	§61.41 and §61.42	NRC will monitor the development of model support for the performance of the lower lateral drainage layer, which depends on the performance of the disposal unit roofs and the HDPE/GCL layers above the FDCs.	In most of DOE's modeled cases, the lower lateral drainage layer above each disposal unit diverts nearly all of the infiltrating water around the disposal units. This modeled diversion of infiltrating water is due to the large difference in the hydraulic conductivity between the lower lateral drainage layer and the disposal unit roofs (for Vaults 1 and 4) and the HDPE/GCL layer (for the FDCs). Because an increase in the amount of infiltrating water will increase the amount of leaching from and the rate of degradation of the waste form, an increase in the amount of infiltrating water would likely result in a higher dose. Therefore, the NRC will monitor model support for the long-term performance of the disposal unit roofs and HDPE/GCL layers overlying the FDCs.	2.5 and 2.7

Monitoring Topic	Performance Objective	Factor	Basis	TER Section
Disposal Unit Performance	§61.41 and §61.42	NRC will monitor the development of model support regarding the long-term fracturing of disposal unit concrete.	NRC analyses of intermediate results from DOE's Case K PORFLOW™ model demonstrate the importance of Tc retention in disposal unit concrete to the timing and magnitude of the dose from fractured saltstone. Radionuclides that flow through fast-pathways created by fractures in the disposal unit floors or walls are not expected to experience as much sorption as radionuclides moving through an unfractured cementitious matrix.	2.5
Disposal Unit Performance	§61.41 and §61.42	NRC will monitor the development of model support for the long-term physical integrity of non-cementitious materials (e.g., epoxy, and neoprene seals) used in disposal unit joints.	There is uncertainty associated with the performance of novel components in the design, as a result of a lack of information on long-term performance of novel components (e.g., epoxy, and neoprene seals). DOE may need to review its experience base with these materials in similar facilities, or, for example, perform accelerated testing to obtain long-term performance data. NRC analyses of intermediate results from DOE's Case K PORFLOW™ model demonstrate the importance of Tc retention in disposal unit concrete to the timing and magnitude of the dose from fractured saltstone. If radionuclides flow through fast-pathways created by degradation of joint material instead of through the disposal unit cementitious material, radionuclides may not be effectively retained in the disposal unit concrete.	2.5
Subsurface K_d values	§61.41 and §61.42	NRC will monitor the K_d value assumed for Se in sand and clay soils.	As discussed in Section 2.7, the NRC staff does not believe that the K_d value assumed in the PA for Se for sand and clay was adequately supported.	2.7
Environmental Monitoring	§61.41 and §61.42	NRC will monitor results from leak detectors installed under select FDCs.	DOE's Consent Order of Dismissal with the SC DHEC requires DOE to install leak detection on Cell 3A and every fifth cell constructed thereafter. Monitoring information from these leak detectors will ensure NRC staff is aware of any early hydraulic failure of the FDCs.	NA

Monitoring Topic	Performance Objective	Factor	Basis	TER Section
Environmental Monitoring	§61.41	NRC will continue to review groundwater monitoring data collected near the SDF.	Reviewing groundwater monitoring data will help to ensure NRC staff is aware of any early release of radionuclides from saltstone. It may also provide staff with other indicators of SDF performance, such as unexpected plumes of nitrate or increased alkalinity.	NA
Radiation Protection Program	§61.43	NRC will continue to monitor annual worker dose reports.	NRC has previously determined that DOE has an effective program in place to protect individuals during operations. As part of its monitoring responsibilities under the NDAA NRC will continue to monitor annual worker dose reports.	4
Radiation Protection Program	§61.43	NRC will continue to monitor results of DOE's air monitoring system.	Releases in the air pathway during operations can contribute to worker dose during operations.	4
Site Stability	§61.41, §61.42, and §61.44	NRC will monitor the development of modeling support for the potential for settlement of the SDF due to consolidation of subsurface layers and the potential formation of sinks under the SDF.	The potential for differential settlement due to consolidation of soft zones or the formation of sinks under the SDF is important to the assessment of site stability because it appears formation of a sink at the SDF could cause significant saltstone fracturing and disruption to the infiltration controls and disposal units.	2.4, 2.5, 2.6, and 5
Site Stability	§61.41, §61.42, and §61.44	NRC will monitor updates to DOE modeling of the effects of static settlement on site stability.	Recent studies predict greater static settlement in the SDF than addressed by DOE in the PA. Updated modeling is necessary to support DOE's conclusion that static settlement will not adversely affect the performance of the closure cap, disposal units, or saltstone grout.	2.4, 2.5, 2.6, and 5

Table A-2: PA Maintenance Items

Topic	Performance Objective	Factor	Basis	TER Section
Modeling	§61.41 and §61.42	NRC staff will continue to monitor any changes to the DOE PA for SDF, including the implementation of the conceptual model, consistency of intermediate model results with the conceptual model, quality assurance of models and codes used, and the appropriate use of probabilistic factors, when used.	NRC's review of the updated 2009 PA for the SDF, as documented in this TER, indicates the importance of these PA factors in the NRC staff's development of reasonable assurance that waste disposal at the SDF meets the performance objectives for protection of the general population and protection of individuals against inadvertent intrusion.	2.11, 2.13, 3
Modeling	§61.41 and §61.42	NRC staff will monitor the defensibility of DOE's conceptual models for releases of radionuclides from the SDF and potential exposures of off-site members of the public and potential inadvertent intruders.	Conceptual model uncertainty is difficult to capture in dose models but can dominate the uncertainty in the dose predictions. For example, an alternate conceptual model in which saltstone oxidizes for a long period of time in which little or no water flows into the waste and then is suddenly exposed to increased water flow (e.g., though HDPE failure) could generate a much larger peak dose than a more gradual failure. Because of the potential importance of alternate conceptual models to dose predictions, NRC staff will monitor DOE's consideration of alternate conceptual models in future PA development.	2.13, 3

Topic	Performance Objective	Factor	Basis	TER Section
Waste Form	§61.41 and §61.42	NRC will monitor measurements of intrinsic diffusivity in degraded saltstone and model support for assumptions about intrinsic diffusivity in saltstone used in future PA revisions.	DOE Case K results and NRC analyses demonstrate the sensitivity of the magnitude and timing of the dose from Tc-99 to the rate of saltstone oxidation. In DOE Case K, the movement of an oxidation front is modeled as proceeding as a function of the square root of time, which limits the progression of oxidation from older fractures. However, other functional relationships are possible if saltstone degrades and the diffusivity increases with time.	2.6
Waste Form	§61.41 and §61.42	NRC will monitor model support for K_d values used to represent sorption of radionuclides in saltstone in future PA revisions.	Based on DOE and NRC sensitivity analyses for Case A and Case K, saltstone K_d values are expected to affect predicted doses significantly. Because K_d values can affect which radionuclides are the primary dose contributors, the NRC staff will monitor changes in K_d values in saltstone. The NRC staff will monitor model support for saltstone K_d values for radionuclides that are risk-significant based on the K_d values and the uncertainty in those values.	2.7

Topic	Performance Objective	Factor	Basis	TER Section
Near Field Flow and Transport	§61.41 and §61.42	NRC will monitor the development of model support for any moisture characteristic curves used to modify the permeability of saltstone or disposal unit concrete in a revised PA.	DOE's Case A and the sensitivity analyses included in the PA relied on moisture characteristic curves that reduced water flow through saltstone, fracture, and disposal unit concrete more than expected based on comparison to curves published for similar materials. DOE's Case K analysis does not take credit for decreasing permeability with decreasing saturation (as described by moisture characteristic curves) and instead assumes the relative permeability always is 1. Because of the sensitivity of projected doses to the modeled permeability of saltstone, the development of appropriate model support for any moisture characteristic curves used in a revised base case is expected to be important to the development of reasonable assurance that waste disposal meets the performance objectives.	2.7
Near Field Flow and Transport	§61.41 and §61.42	NRC will monitor model support for K_d values used to represent sorption of radionuclides in disposal unit concrete.	As discussed in NRC's barrier analysis (Section 2.13.3), the K_d values used in disposal unit concrete can have a significant effect on the modeled release rate of radionuclides into the near-field environment. Release rates, in turn, can directly affect the predicted dose.	2.7, 2.13
Far-Field Flow	§61.41	NRC will monitor the adequacy of DOE's far-field model calibration.	The calibration process could be improved and made more transparent, particularly in the area near to the SDF, to provide confidence that the level of dilution and dispersion in DOE's SDF transport models is not overstated.	2.8

Topic	Performance Objective	Factor	Basis	TER Section
Far-Field Flow	§61.41	NRC will monitor DOE's source loading approach to ensure that the dose estimates are not significantly under-predicted.	If DOE were to change how disposal unit releases are loaded into the far-field model to ensure that (i) the same amount of mass is loaded into the saturated zone underneath each individual FDC, and (ii) that source loading occurs at the water table in the far-field model, the results of the simulations could be significantly different. Scoping simulations performed by NRC staff using Tc-99 fluxes from Case K indicate that the peak sector concentrations at the 100 m boundary could be significantly higher for some sectors.	2.8
Far-Field Flow	§61.41	NRC will monitor the appropriateness of selected dispersivities and the need for additional vertical or horizontal mesh refinement to ensure that contaminant plumes are not artificially dispersed in the far-field model.	DOE presented results in the FTF RAI responses (SRR-CWDA-2009-00054) that indicate additional grid refinement may be necessary to reduce numerical dispersion in cases of very low to no assumed physical dispersion; this analysis is expected to be relevant to the SDF because the local FTF and SDF models have the same grid resolution. For example, if no physical dispersion is assumed, then the peak concentrations associated with a pulse release of a conservative tracer are shown to be a factor of approximately three to four times higher with a grid refined by a factor of two in each dimension (or a factor of 8 times more elements).	2.8

Topic	Performance Objective	Factor	Basis	TER Section
Far-Field Flow	§61.41	NRC will monitor DOE's efforts to collect data and information to evaluate the potential impact of calcareous zones on contaminant flow and transport at the SDF.	Many SDF sources traverse the lower zone of the UTR aquifer where calcareous materials are more pervasive in the subsurface at SRS. Some evidence exists that contaminants in Burial Ground Complex (located in E-Area at the GSA) and the Chemical, Metals, and Pesticide Pits (located off the GSA) may be preferentially transported within these zones. In an FTF RAI resolution meeting (NRC, 2011x) DOE indicated that a field mapping activity could be performed. If calcareous zone seeps are identified, tracer studies in the SDF UTR-LZ using innocuous tracers that are commonly used to understand preferential flow and transport could be conducted to better understand the effect of these zones on contaminant flow and transport.	2.8
Far-Field Flow	§61.41	NRC will monitor model support for K_d values used to represent sorption of radionuclides in site soil.	Sorption in soil can significantly affect doses from short- to moderate- half-life radionuclides if they are retained long enough to experience significant decay. Thus NRC will monitor support for K_d values for risk-significant radionuclides or radionuclides that may become risk-significant based on the choice of K_d values.	2.7

Topic	Performance Objective	Factor	Basis	TER Section
Biosphere	§61.41 and §61.42	NRC will monitor whether K_d values for key radionuclides in surface soil are expected to significantly increase predicted radionuclide build-up in biosphere soil.	Distribution coefficients used in the build-up analysis were based on values selected conservatively in the context of hydrologic transport modeling (i.e., values were purposefully biased low). This selection process is non-conservative when applied to irrigation and soil sorption modeling because lower sorption values could underestimate radionuclide build-up. NRC staff scoping calculations show using site-specific K_d measurements could increase estimated build-up by approximately a factor of 2 to 5. This increase would result in a higher predicted soil concentration and a higher projected dose from ingestion of plant and animal products.	2.10
Biosphere	§61.41 and §61.42	NRC will monitor consumption factors and uncertainty distributions for transfer factors.	The assumed consumption of drinking water in the 2009 PA is approximately a factor of 2 less than EPA recommended assumption of 2 L/d and is not supported by site-specific information. Transfer factors can have considerable uncertainty, which should be evaluated in any future probabilistic model.	2.10

Appendix A: References

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Appendix B Results of NRC Staff Sensitivity Analyses Performed with a Simple Analytical Dual-K_d Model

DOE and NRC observed that, compared to a model that tracks release from oxidized and reduced grout separately, the average-K_d model DOE used to estimate Tc release predicts less release at early times and more release when saltstone nears full oxidation (SRR-CWDA-2011-00114, TER Sections 2.7 and 2.13). Because of this behavior, the average-K_d model appears to significantly over-estimate Tc peak fractional release rates from saltstone under certain conditions (Section 2.13). Because this apparent overestimate was difficult to avoid using the DOE PORFLOW™ model, the NRC staff could not conduct all of the necessary sensitivity analyses for Case K1 by modifying DOE's PORFLOW™ model. Thus, the NRC staff developed a simple analytical dual-K_d model to conduct certain sensitivity analyses based on Case K1 (NRC, 2012a, b). These NRC sensitivity analyses were necessary in part because, unlike Cases A - E, DOE submitted Case K with only a limited deterministic sensitivity analysis (i.e., evaluation of certain Tc K_d values by comparison of Cases K, K1,and K2). As discussed in Section 2.13, the NRC staff did not use the results of the NRC model for comparison to the performance objectives. Rather, the NRC staff used the model to identify risk-significant assumptions and to develop an estimate of the uncertainty in the Case K1 value.

Sensitivity and uncertainty were evaluated only with respect to source-term release. Specifically, the staff evaluated the effects of fracture spacing, fracture growth rate, K_d values for Tc in oxidizing and reducing saltstone, reducing capacity, and flow assumptions. Assumptions about water flow were evaluated with cases in which the flow thorough saltstone was assumed to be limited by the hydraulic conductivity of saltstone (Table B-1) and cases in which flow was based on the flow used in the DOE Case K1 PORFLOW™ model (Table B-2). Results were evaluated in terms of the peak annual fractional release rate of Tc from saltstone. The annual fractional release rate is the activity of Tc that leaves saltstone in a year divided by the initial Tc inventory. The annual fractional release rate is not a direct indication of dose, because releases of Tc from saltstone are attenuated by transport through the disposal unit concrete, site soils, and biosphere. However, in general, higher annual fractional release rates lead to higher doses to off-site members of the public[37].

Model sensitivities were evaluated by comparing the predicted peak fractional release rate in various test cases. For example, in general, cases with a fracture spacing of 1.0 m (3.3 ft) at 10,000 years have lower release rates than cases with a fracture spacing of 10 cm (4 in) at 10,000 years (i.e., compare tests 1 though 8 to tests 9 through 16 in Table B-1 and tests 17 though 20 to tests 21 through 24 in Table B-2). Similarly, cases in which fractures develop as a logarithmic function of time have greater peak release rates than cases with the same flow and sorption parameters in which fractures develop as a quadratic function of time (e.g., compare test 1 with 3; test 2 with 4; or test 3 with 5 in Table B-1).

[37] Because the projected dose to an individual inadvertently intruding on the SDF is expected to be dominated by groundwater pathways instead of direct exposure (Chapter 3), higher fractional release rates also would lead to higher projected doses to an inadvertent intruder at the SDF.

Model sensitivities to flow were evaluated by comparing the effects of using different initial saltstone hydraulic conductivities (Table B-1) and by using the flow rates DOE used in the PORFLOW™ model in Case K1, which account for effects of the closure cap (Table B-2). In general, for the flow rates that were evaluated, cases with a constant flow have lower release rates than cases with increasing flow (if fracturing and sorption are not changed). This effect can be seen with two different types of comparisons. First, because the tests shown in Table B-1 all use a hydraulic conductivity of $1x10^{-7}$ cm/s at 10,000 years, cases with an initial hydraulic conductivity of $1x10^{-7}$ cm/s are modeled with a constant flow for the first 10,000 years after closure[38]. In contrast, cases with an initial hydraulic conductivity of $1x10^{-8}$ cm/s are modeled with an increasing flow. By comparing results for the first 10,000 years for cases that differ only in the initial flow rate (e.g., tests 1 and 2; tests 3 and 4; tests 5 and 6 in Table B-1), it is possible to compare otherwise equivalent cases that have either constant or increasing flow. Second, a similar effect is seen by comparing the constant flow cases (i.e., odd-numbered tests in Table B-1) with the results of cases based on the DOE Case K1 PORFLOW™ model (Table B-2). In each case, tests with comparable fracture growth, final fracture spacing, and sorption coefficients lead to a greater peak annual fractional release rate within 10,000 years if flow is based on DOE Case K1 PORFLOW™ model, which increases with time. The result that peak predicted release rate is greater if the initial hydraulic conductivity is lower may appear to be counterintuitive. However, this observation is consistent with the general trend that a low level of release at early times leads lower peak release rates than cases in which saltstone oxidizes during a period of low water flow and is subject to higher water flow only when saltstone is more significantly oxidized. While this discussion of observations is not exhaustive, it provides examples of how the NRC staff used the results of the simple analytical model.

The model includes oxidation proceeding from fracture faces due to gas-phase transport of oxygen into the fractures, as well as matrix oxidation attributable to oxygenated water flowing through saltstone. In contrast, the DOE Case K, K1, and K2 models include oxidation proceeding from fracture faces, but not oxidation of the matrix from inflowing water. In the NRC simple analytical model, the fraction of oxidation proceeding from fracture faces was reduced to account for areas already oxidized by infiltrating water to avoid double-counting. The fraction of saltstone oxidized at each time step was used to predict release from the oxidized and reduced saltstone fractions separately. Tc release was based on the assumption that Tc comes to equilibrium within the oxidized and reduced regions separately based on the K_d for reduced or oxidized saltstone, as appropriate. Diffusion between the reduced and oxidized regions was neglected. The amount of Tc released from both the oxidized and reduced regions was tracked separately for each time step. In particular, inventory in the oxidized region in each time step was tracked so that the oxidized areas would cease to release Tc once the inventory in the oxidized volume was depleted). The inventory of Tc in the reduced region was tracked as well; however, the inventory in the reduced region typically was not exhausted until the inventory in the entire monolith was exhausted. Details of the analytical model are provided in a separate document (NRC, 2012b).

[38] In all cases in Table B-1, the modeled flow through saltstone increased from $1x10^{-7}$ cm/s to $1x10^{-6}$ cm/s from 10,000 to 20,000 years after closure.

The simple analytical model estimates the release from saltstone, and does not account for attenuation (i.e., lowering of the peak release rate) in the disposal unit floor and walls. The amount of attenuation the disposal unit provides depends in part[39] on the magnitude of the peak release from saltstone and on the K_d values used to model Tc transport in the disposal unit concrete. Comparison of peak fractional release rates from saltstone and from the disposal unit in DOE's PORFLOW™ model for Cases K, K1, and K2 illustrates the attenuation in the disposal unit (Table B-3). For example, the effect of the magnitude of the peak release from saltstone is demonstrated by comparison of Cases K and K1. Although Case K uses larger disposal unit K_d values for Tc than Case K1, (i.e., decreasing from 1,000 mL/g to approximately 420 mL/g over 20,000 years in Case K and from 500 mL/g to approximately 210 mL/g in 20,000 years in Case K1), more attenuation occurs in K1 because it has a larger peak saltstone release rate. The effect of the disposal unit K_d values is seen by comparison of Case K1 and a modified version of K1 in which the K_d value used in the disposal unit concrete is changed to a value representative of oxidized concrete (i.e., 0.8 mL/g). Although both cases have the same peak saltstone release rate, there is more than an order of magnitude more attenuation in the disposal unit concrete in Case K1 than the modified Case K1 because of the difference in disposal unit K_d values (i.e., factor of 22 attenuation in Case K1 and factor of 1.1 in the modified Case K1). In Cases K, K1, and K2, the peak release rate from the unsaturated soil is similar to the peak release from the disposal unit, indicating that little additional attenuation of Tc release takes place in the unsaturated soil. In the modified version of Case K1, little attenuation occurs in the disposal unit concrete but more attenuation is seen in the unsaturated soil. More attenuation occurs in the unsaturated zone in the modified version of Case K1 than in Cases K, K1, and K2, presumably because the modified version of Case K1 has a higher release rate from the disposal unit into the unsaturated soil than the other cases do.

[39] The amount of attenuation also depends in part on the shape of the peak (i.e., magnitude as a function of time).

Table B-1: Peak annual fractional release rates estimated with a simple analytical dual-K_d model (NRC, 2012a, b) for cases in which flow through saltstone is limited by the saltstone hydraulic conductivity

Input Values[1]				Output Values[2]			
Test Number	Fracture Growth Rate (function of time)	Hydraulic Conductivity at time of closure (cm/s)	Sorption Coefficient (K_d) Reducing Saltstone (mL/g)	Case K Saltstone Reducing capacity (0.206 meq e^-/g)		Case A Saltstone Reducing capacity (0.822 meq e^-/g)	
				In 10,000 years	In 20,000 years	In 10,000 years	In 20,000 years
Final Fracture Spacing of 10 cm (3.9 in)							
1	Logarithmic	$1x10^{-7}$	500	$4.8x10^{-4}$	$4.8x10^{-4}$	$3.1x10^{-4}$	$3.1x10^{-4}$
2	Logarithmic	$1x10^{-8}$	500	$5.6x10^{-4}$	$5.6x10^{-4}$	$3.1x10^{-4}$	$3.1x10^{-4}$
3	Quadratic	$1x10^{-7}$	500	$2.7x10^{-4}$	$2.7x10^{-4}$	$2.9x10^{-4}$	$2.9x10^{-4}$
4	Quadratic	$1x10^{-8}$	500	$2.8x10^{-4}$	$2.8x10^{-4}$	$2.9x10^{-4}$	$2.9x10^{-4}$
5	Logarithmic	$1x10^{-7}$	139	$3.8x10^{-4}$	$3.8x10^{-4}$	$3.3x10^{-4}$	$3.3x10^{-4}$
6	Logarithmic	$1x10^{-8}$	139	$5.1x10^{-4}$	$5.1x10^{-4}$	$3.3x10^{-4}$	$3.3x10^{-4}$
7	Quadratic	$1x10^{-7}$	139	$2.4x10^{-4}$	$2.4x10^{-4}$	$2.9x10^{-4}$	$2.9x10^{-4}$
8	Quadratic	$1x10^{-8}$	139	$2.8x10^{-4}$	$2.8x10^{-4}$	$2.9x10^{-4}$	$2.9x10^{-4}$
Final Fracture Spacing of 1.0 m (3.3 ft)							
9	Logarithmic	$1x10^{-7}$	500	$8.3x10^{-5}$	$1.6x10^{-4}$	$4.1x10^{-5}$	$1.2x10^{-4}$
10	Logarithmic	$1x10^{-8}$	500	$8.4x10^{-5}$	$1.7x10^{-4}$	$4.1x10^{-5}$	$1.2x10^{-4}$
11	Quadratic	$1x10^{-7}$	500	$6.7x10^{-5}$	$1.6x10^{-4}$	$4.2x10^{-5}$	$1.1x10^{-4}$
12	Quadratic	$1x10^{-8}$	500	$6.8x10^{-5}$	$1.7x10^{-4}$	$4.1x10^{-5}$	$1.2x10^{-4}$
13	Logarithmic	$1x10^{-7}$	139	$1.0x10^{-4}$	$1.6x10^{-4}$	$6.5x10^{-5}$	$1.7x10^{-4}$
14	Logarithmic	$1x10^{-8}$	139	$1.0x10^{-4}$	$1.9x10^{-4}$	$6.6x10^{-5}$	$2.0x10^{-4}$
15	Quadratic	$1x10^{-7}$	139	$8.3x10^{-5}$	$1.5x10^{-4}$	$6.5x10^{-5}$	$1.7x10^{-4}$
16	Quadratic	$1x10^{-8}$	139	$8.6x10^{-5}$	$1.9x10^{-4}$	$6.3x10^{-5}$	$1.9x10^{-4}$

[1]All cases use an oxidizing saltstone K_d of 0.8 mL/g, a hydraulic conductivity of $1x10^{-7}$ cm/s at 10,000 years and $1x10^{-6}$ cm/s at 20,000 years after closure
[2]Peak Annual Fractional Release ($year^{-1}$)

Table B-2: Peak annual fractional release rates estimated with a simple analytical dual-K_d model (NRC, 2012a, b) for cases in which flow is based on DOE PORFLOW™ Case K1 flow through saltstone

Input Values[1]			Output Values[2]	
Test Number	Fracture Growth Rate (function of time)	Sorption Coefficient (K_d) Reducing Saltstone (mL/g)	within 10,000 years of closure	within 20,000 years of closure
Final Fracture Spacing of 10 cm (3.9 in)				
17	Logarithmic	500	$5.6x10^{-4}$	$5.6x10^{-4}$
18	Quadratic	500	$3.1x10^{-4}$	$3.1x10^{-4}$
19	Logarithmic	139	$5.6x10^{-4}$	$5.6x10^{-4}$
20	Quadratic	139	$3.1x10^{-4}$	$3.1x10^{-4}$
Final Fracture Spacing of 1.0 m (3.3 ft)				
21	Logarithmic	500	$1.6x10^{-4}$	$1.6x10^{-4}$
22	Quadratic	500	$1.4x10^{-4}$	$1.4x10^{-4}$
23	Logarithmic	139	$2.2x10^{-4}$	$2.2x10^{-4}$
24	Quadratic	139	$2.0x10^{-4}$	$2.0x10^{-4}$

[1]All cases use an oxidizing saltstone K_d of 0.8 mL/g and Case K Saltstone Reducing capacity (0.206 meq e^-/g)
[2]Peak Annual Fractional Release Rate ($year^{-1}$)

Table B-3: Peak annual fractional release rates predicted with DOE's Case K, K1, and K2 PORFLOW™ models and an NRC-modified version of Case K1

Release	Peak Annual Fractional Release Rate ($year^{-1}$)			
	Case K	**Case K1**	**Case K2**	**Modified[1] K1**
from Saltstone	$1.3x10^{-3}$	$6.4x10^{-3}$	$1.2x10^{-3}$	$6.4x10^{-3}$
from Disposal Unit	$1.8x10^{-4}$	$2.9x10^{-4}$	$2.6x10^{-4}$	$5.7x10^{-3}$
from Unsaturated Zone (UZ)	$1.8x10^{-4}$	$2.9x10^{-4}$	$2.6x10^{-4}$	$3.0x10^{-3}$
Attenuation	**Ratio of Saltstone Release to Disposal Unit or UZ Release**			
In Disposal Unit	7.0	22	4.5	1.1
In Disposal Unit and Unsaturated Zone	7.1	22	4.6	2.1

data for Cases K, K1 and K2 taken from DOE STAT.out files [NRC, 2010]
[1] Case identical to K1 except that the K_d for Tc in disposal unit concrete represents oxidized concrete conditions (i.e., 0.8 mL/g)

Appendix B: References

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