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Probabilistic Performance-Assessment Modeling of the Mixed Waste Landfill at Sandia National Laboratories (2nd Edition)

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Abstract

A probabilistic performance assessment has been conducted to evaluate the fate and transport of radionuclides (amercium-241, cesium-137, cobalt-60, plutonium-238, plutonium-239, radium-226, radon-222, strontium-90, thorium-232, tritium, uranium-238), heavy metals (lead and cadmium), and volatile organic compounds (VOCs) at the Mixed Waste Landfill (MWL). Probabilistic analyses were performed to quantify uncertainties inherent in the system and models for a 1,000-year period, and sensitivity analyses were performed to identify parameters and processes that were most important to the simulated performance metrics. Comparisons between simulated results and measured values at the MWL were made to gain confidence in the models and perform calibrations when data were available. In addition, long-term monitoring requirements and triggers were recommended based on the results of the quantified uncertainty and sensitivity analyses.

At least one-hundred realizations were simulated for each scenario defined in the performance assessment. Conservative values and assumptions were used to define values and distributions of uncertain input parameters when site data were not available. Results showed that exposure to tritium via the air pathway exceeded the regulatory metric of 10 mrem/year in about 2% of the

simulated realizations when the receptor was located at the MWL (continuously exposed to the air directly above the MWL). Simulations showed that peak radon gas fluxes exceeded the design standard of 20 pCi/m^2/s in about 3% of the realizations if up to 1% of the containers of sealed radium-226 sources were assumed to completely degrade in the future. If up to 100% of the containers of radium-226 sources were assumed to completely degrade, 30% of the realizations yielded radon surface fluxes that exceeded the design standard. For the groundwater pathway, simulations showed that none of the radionuclides or heavy metals (lead and cadmium) reached the groundwater during the 1,000-year evaluation period. Tetrachloroethylene (PCE) was used as a proxy for other VOCs because of its mobility and potential to exceed maximum contaminant levels in the groundwater relative to other VOCs. Simulations showed that PCE reached the groundwater, but only 1% of the realizations yielded aquifer concentrations that exceeded the regulatory metric of 5 μg/L.

Based on these results, monitoring triggers have been proposed for the air, surface soil, vadose zone, and groundwater at the MWL. Specific triggers include numerical thresholds for radon concentrations in the air, radionuclide and heavy-metal concentrations in surface soil, soil-gas concentrations of VOCs in the vadose zone, moisture content in the vadose zone, and uranium and VOC concentrations in groundwater. The proposed triggers are based on U.S. Environmental Protection Agency and Department of Energy regulatory standards. If a trigger is exceeded, then a trigger evaluation process will be initiated which will allow sufficient data to be collected to assess trends and recommend corrective actions, if necessary.

Acknowledgments

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Preface to the 2nd Edition

This 2nd Edition includes revisions to address comments made by the New Mexico Environment Department (NMED, November 2006).

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1. Introduction

1.1 Background and Objectives

The Corrective Measures Implementation (CMI) Plan for the Mixed Waste Landfill (MWL) at Sandia National Laboratories, Albuquerque, NM, is being submitted to the New Mexico Environment Department (NMED). As part of the final order selecting a remedy for the MWL (NMED May 2005), NMED required that the CMI Plan include a comprehensive fate and transport model to determine if contaminants will move from the MWL down through the vadose zone to groundwater. In addition, the NMED required that the CMI Plan include triggers for future action that identify and detail specific monitoring results that will require additional testing or implementation of an additional or different remedy.

This report presents the probabilistic fate and transport models that were used to assess the performance of the MWL. Relevant contaminants of concern at the site were included, and site-specific models and parameters were used in a probabilistic analysis. Results of the analysis were compared to regulatory performance metrics, and sensitivity analyses were performed to determine the most important parameters and processes that impacted the variability of the simulated performance metrics. Based on these simulations and results, appropriate triggers were identified and defined to address long-term monitoring requirements at the site.

A period of 1,000 years was selected for the probabilistic analysis to be consistent with DOE Order 435.1. DOE Order 435.1 requires that performance assessments be conducted for low-level radioactive waste disposed after September 26, 1988, and that performance objectives be evaluated for a 1,000-year period to determine potential risk impacts to the public and environment. Although most of the MWL wastes were disposed of prior to September 26, 1988, a 1,000 year period was nonetheless determined to be appropriate for assessment of regulatory performance metrics.

1.2 Overview of the Mixed Waste Landfill

The Mixed Waste Landfill (MWL) is located approximately five miles southeast of Albuquerque International Sunport and four miles south of Sandia National Laboratories' (SNL) central facilities (Figure 1). The landfill is a fenced, 2.6-acre area in the north-central portion of Technical Area 3 (TA-3). The mean elevation at the MWL is 5381 feet.

The MWL was established in 1959 as a disposal area for low-level radioactive and mixed waste that was generated at SNL research facilities. Originally, the landfill was opened as the "Area 3 Low-level Radioactive Dump," when the low-level radioactive disposal area in Technical Area 2 was closed in March 1959. The MWL accepted low-level radioactive waste and minor amounts of mixed waste from March 1959 through December 1988. Approximately 100,000 cubic ft of low-level radioactive waste containing approximately 6,300 curies of activity was disposed of at the landfill.

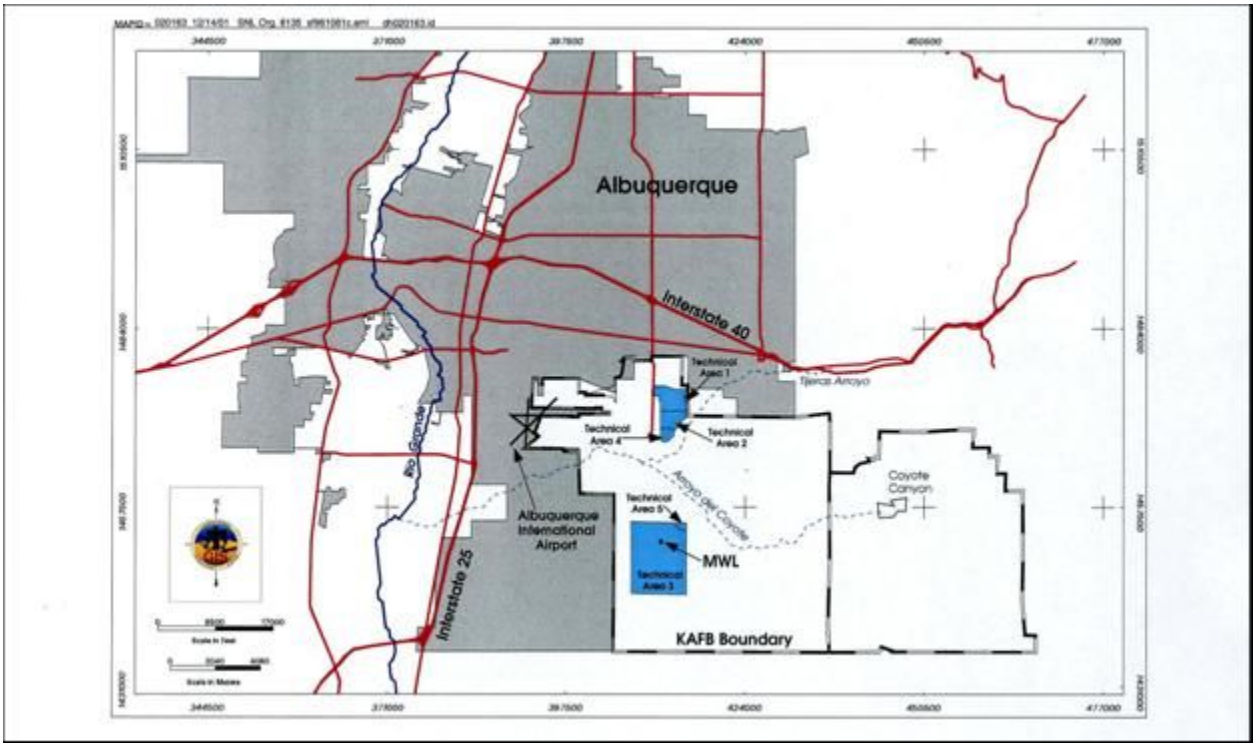

Figure 1. Location of the Mixed Waste Landfill relative to Albuquerque, NM, and Kirtland Air Force Base.

1.2.1 Site Description

The MWL consists of two distinct disposal areas: the classified area, occupying 0.6 acres, and the unclassified area, occupying 2.0 acres (Figure 2). Low-level radioactive and mixed waste has been disposed of in each area. Wastes in the classified area were buried in unlined, vertical pits. Wastes in the unclassified area were buried in unlined, shallow trenches.

A Phase 1 RCRA facility investigation was conducted in 1989 and 1990 to determine if a release of RCRA contaminants had occurred at the MWL and to begin characterizing the nature and extent of any such release. The Phase 1 facility investigation indicated that tritium was the primary contaminant of concern. No organic contaminants were identified. A Phase 2 RCRA facility investigation was initiated in 1992 to determine contaminant source, define the nature and extent of contamination, identify potential contaminant transport pathways, evaluate potential risks posed by the levels of contamination identified, and recommend remedial action, if warranted, for the landfill.

The Phase 2 RCRA facility investigation incorporated the streamlining approach, combining data quality objectives and the observational approach. Nonintrusive field activities were conducted first to facilitate the efficiency and cost-effectiveness of intrusive field activities. Data collected during the Phase 2 RCRA facility investigation were evaluated using U.S. Environmental Protection Agency-approved methods. Initially, a constituent population was statistically compared to natural background. Any constituent failing the statistical comparison was further

analyzed for spatial distribution. Constituents that failed the statistical comparison to background and showed a strong spatial correlation were identified as potential contaminants of concern.

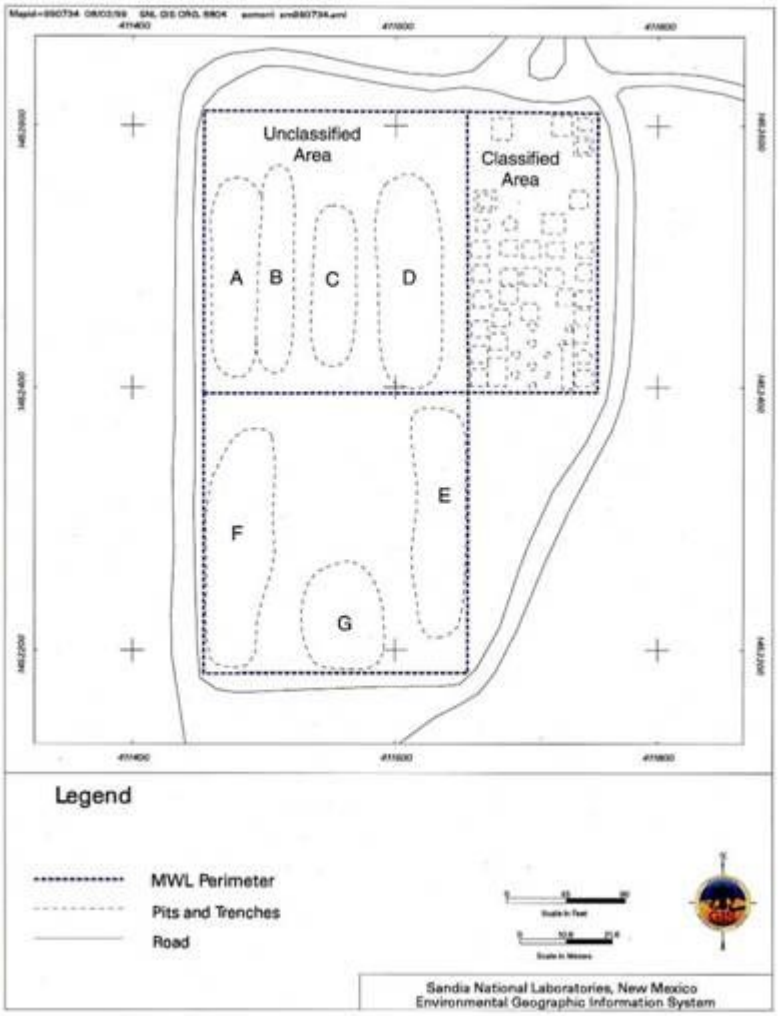

Figure 2. Map of the Mixed Waste Landfill.

The Phase 2 RCRA facility investigation was completed in 1995. This investigation included surface radiological surveys; ambient air sampling; soil sampling for background metals and radionuclides; soil sampling for volatile organic compounds (VOCs), semivolatile organic compounds, target analyte list metals, and radionuclides; nonintrusive geophysical surveys; passive and active soil gas sampling; borehole drilling; installation of groundwater monitoring wells; groundwater sampling; vadose zone tests; aquifer tests; and risk assessment. The Phase 2 RCRA facility investigation confirmed the findings of the Phase 1 RCRA facility investigation.

1.2.2 Contaminants of Concern

Based on the results of the Phase 1 and Phase 2 RCRA Facility Investigations, tritium was found to be the primary contaminant of concern that has been released from the MWL. An estimated 2400 curies of tritium were disposed of in the MWL. Tritium is extremely mobile when incorporated in water in liquid and vapor form, moving easily through the vadose zone and into the atmosphere.

Tritium levels range from 1100 picocuries/gram in surface soils to 206 picocuries/gram in subsurface soils in the classified area of the landfill. The highest tritium levels are found within 30 feet of the surface in soils adjacent to and directly below classified area disposal pits. At depths greater than 30 feet below ground surface, tritium levels fall off rapidly to a few picocuries/gram of soil.

Tritium also occurs as a diffuse air emission from the landfill. Tritium emissions from the MWL are diminishing with time due to its half-life of 12.3 years. Total tritium emissions to the atmosphere were measured at 0.294 curies/year in 1993 and at 0.090 curies/year in 2003 (Peace et al., 2002; Anderson, 2004).

An estimated 27,900 kg (9.3 curies) of uranium-238 (depleted uranium) are present in the MWL inventory. Based on the results from the Phase 1 and Phase 2 RFIs, there is no indication that uranium has been released from the MWL. However, because of the large quantity of depleted uranium disposed of in the MWL, the fate and transport of uranium was modeled in this study.

Other radionuclides present in the MWL inventory include cobalt-60, strontium-90, cesium-137, plutonium-238 and -239, americium-241, radium-226, and thorium-232. The fate and transport of these radionuclides was modeled, although there is no evidence that these radionuclides have been released from the MWL.

There is an estimated 128,000 kg of lead disposed of within various pits and trenches in the landfill. Most of the lead is in the form of shielding (i.e. lead bricks, casks, pigs, and shipping canisters). Smaller lead items include containers commonly used to dispose of radioactive sources. The lead containers were typically placed in concrete-filled A/N cans or 55-gallon drums. Larger lead items include five massive stainless steel and lead casks disposed of in Trench F, each weighing up to 40 tons. The fate and transport of lead was modeled, although there is no evidence that lead has been released from the MWL.

Cadmium is not specifically listed in the MWL inventory. However, slightly-elevated cadmium has been detected in five boreholes along the west side of the MWL to depths of up to at least 104 ft bgs. The cadmium concentrations in MWL soils range from non-detect to 1.97 mg/kg, approximately two times the NMED maximum background value of 0.9 mg/kg. The source of cadmium in MWL soils is unknown.

Cadmium has occasionally been detected in MWL groundwater at concentrations above the EPA MCL, although these detections are sporadic and unpredictable. Because the cadmium detections above the MCL are inconsistent, it is believed that these detections do not indicate contamination from the MWL. Nevertheless, cadmium is considered a contaminant of concern, and the fate and transport of cadmium was modeled.

During the Phase 2 RCRA Facility Investigations, low levels of VOCs were detected in soil gas samples obtained from the landfill. The primary VOCs detected in soil gas at the MWL include tetrachloroethene (PCE), trichloroethene (TCE), dichloro-difluoromethane, 1,1,1-trichloroethane (1,1,1-TCA), trichlorofluoromethane, and 1,1,2-trichloro,1,2,2-trifluoroethane. Of these VOCs, PCE was determined to have the highest potential to reach groundwater at concentrations near its maximum contaminant level (Klavetter, 1995a). Other VOCs were either not as mobile or did not have sufficiently high initial soil gas concentrations. For this reason, PCE is a contaminant of concern, and the fate and transport of PCE was modeled. However, because the remaining VOCs still have some potential to contaminate groundwater, PCE was modeled in this study as a proxy for all of the VOCs.

Radon gas generation from the landfill is based on the estimated 6 curies of radium-226 in the MWL inventory. Most of the radium-226 in the MWL is in the form of sealed sources. Emission of radon gas from the MWL was investigated in 1997. No significant difference between the MWL and the background measurements in terms of median, mean, and standard deviation was observed (Haaker, 1998). However, at the request of the NMED, radon was included in the MWL fate and transport model.

In summary, the following list of actual and potential contaminants was included in the MWL fate and transport model: tritium, americium-241, cesium-137, cobalt-60, plutonium-238 plutonium-239, radium-226, radon-222, strontium-90, thorium-232, uranium-238, lead, cadmium, and PCE.

2. Modeling Approach

2.1 Previous Modeling Studies

This section summarizes previous modeling studies conducted for the MWL. These studies include fate and transport modeling studies conducted by Argonne National Laboratory, Sandia, and WERC (Consortium for Environmental Education & Technology Development). Cover performance modeling studies were conducted by Sandia in support of the MWL cover design, and are summarized in this section as well.

2.1.1 Fate and Transport Modeling Studies

Previous fate and transport modeling studies conducted for the MWL include a study by Argonne National Laboratory in 1995 as part of a preliminary human health risk assessment for the MWL; a subsequent study conducted by Sandia in 1995 regarding the potential migration of radionuclides and organic compounds from the MWL; a 1997 study to model the infiltration of reactor coolant water discharged into an MWL trench in 1967; and a study conducted in 2001 by WERC of tritium migration through the vadose zone beneath the MWL.

Argonne National Laboratory Modeling Study

One of the earlier modeling studies on the MWL was conducted by Johnson et al. (1995) at Argonne National Laboratory (ANL). The ANL study used a "worst case" scenario approach in which they took conservative values of parameters at different levels of model complexity to ascertain the probable fate and transport of, as well as risk from, the contaminants. The study used a tiered approach for modeling the fate and transport of contaminants, with increasing model complexity and more justifiable simplifying assumptions.

The first-tier screen was a geometric approach in which tritium from the MWL was distributed evenly throughout the vadose zone. This first-tier screening suggested that tritiated water from the MWL could potentially reach groundwater, although the likelihood was considered small.

The second-tier analysis utilized a one-dimensional analytical solution for flow and transport in the vadose zone, but did not include lateral dispersion, which would reduce concentrations of tritium and the distance traveled by tritium from the landfill. This analysis showed that tritium concentrations could exceed the EPA drinking water guideline of 20,000 pCi/L after 57 years if the underlying soils were fully saturated. However, because of the uncertainty of the input parameters (particularly velocity, which was considered too high), the analysis over-predicted tritium concentrations in subsurface soils.

The final tier utilized a three-dimensional numerical code, TRACR3D, which still is extensively used for flow and transport calculations. This code is relatively complex, utilizing finite-element solutions for both the saturated and unsaturated zones. Tritium was the primary contaminant modeled because of its assumed higher mobility compared to other radionuclides and organic contaminants. Conservative assumptions were used in the model, boundary conditions, and hydrologic parameters to bound the probable extent and concentration of tritium. The model predicted that 27 years after disposal, the maximum tritium contamination reaches 184 ft below ground surface (bgs) with a maximum concentration of 2.8×10^6 pCi/L, significantly higher than measured field values. After an additional 100 years, the tritium was predicted to have traveled to a depth of 230 ft bgs, with a maximum tritium concentration of 5,400 pCi/L. The ANL study concluded that no detectable tritium concentrations would be likely to reach groundwater at the MWL.

The study also included screening calculations for aqueous-phase transport of PCE and TCE, and predicted that these VOCs could reach the water table approximately 250 years from time of disposal. No calculations were conducted for vapor-phase transport, which has proven to be the most significant transport mechanism for organic compounds in the vadose zone at nearby ER sites, including the Chemical Waste Landfill.

Sandia Modeling of Radionuclide and Organic Compound Transport

A subsequent study was conducted by Sandia in August 1995 to simulate potential contaminant flow and transport from the MWL. The study was conducted using the code Borehole Optimization Support System (BOSS), originally developed to determine the optimum number and location of boreholes and monitoring wells necessary to define the nature and extent of contamination. Monte Carlo uncertainty analysis of flow and transport was used to simulate the

migration of radionuclides and organic compounds from the MWL. (Klavetter, 1995a; Klavetter, 1995b).

BOSS was first used to simulate the migration of radionuclides, including tritium, cesium-137, and strontium-90 from the MWL, using more representative hydrologic property values than were applied in the ANL study. The modeling study predicted that no detectable tritium would reach groundwater at the MWL, and that detectable tritium would not migrate below a depth of 40 m (131 ft). These results are consistent with the actual tritium distribution data for subsurface soils collected during the Phase 2 RFI. The model also predicted that no detectable activity of cesium-137 and strontium-90 would migrate even 10 m below the MWL pits and trenches.

The code BOSS was also used to simulate the vapor-phase and aqueous-phase transport of the six VOCs detected in MWL soil gas (Section 1.2.2). The modeling results demonstrated that aqueous-phase transport of organic contaminants from the MWL was not a significant transport mechanism. The modeling results also demonstrated that vapor-phase transport of five of the six organic compounds was not significant, due to the low concentrations of these contaminants detected in the soil gas.

Concentrations of PCE detected in soil gas near the MWL surface were calculated to be high enough to result in concentrations of sub-ppb to a few ppb in groundwater within 50 years. The model predicted that the lateral extent of PCE in the groundwater would be limited, with PCE at concentrations greater than 1 ppb extending less than 130 m (426 feet) downgradient of the MWL. The study recommended that further evaluation of the fate and transport of PCE be considered, including a review of PCE concentrations in borehole soil samples collected during the Phase 2 RFI. PCE was detected at low concentrations in soil samples from 2 of the 16 boreholes drilled during the Phase 2 RFI. PCE was detected in BH-3 at a maximum concentration of 2.45 *J* µg/kg, and in MW-4 at a maximum concentration of 5.4 µg/kg (Peace et. al., 2002).

Modeling Study of Reactor Coolant Water Infiltration

In 1997, a modeling study was conducted to simulate the infiltration of 271,500 gallons of reactor coolant water from a trench at the MWL (Wolford1997). The objective of the study was to evaluate the potential migration of coolant water discharged into Trench D of the MWL in May and June, 1967. The water originated from the Sandia Engineering Reactor Facility in Technical Area 5, and contained approximately 1 Ci of total radioactivity, primarily short-lived fission products. Trench D was an active disposal trench at the time, and was believed to be the most likely source for contaminant release and migration from the MWL.

The modeling study used the code VS2DT (Healy, 1990), a finite difference unsaturated flow and transport model developed by the U.S. Geological Survey. The modeling results indicated that the reactor coolant water, and any tritium mobilized by the water, would not have migrated beyond a depth of approximately 120 ft, based on a 30-year simulation. The modeling results were consistent with Phase 2 RFI field measurements of tritium activities in subsurface soils, which showed tritium detected to a maximum depth of 120 ft bgs.

The study also simulated the fate and transport of the coolant water and tritium for a period of 90 years into the future. The study predicted that the coolant water and any tritium in the water would not migrate more than 5 to 10 ft below its current predicted depth of 120 ft. Due to radioactive decay, tritium concentrations in the water were predicted to decrease at a faster rate than the downward movement of the wetting front.

WERC Modeling of Tritium Migration through the Vadose Zone

In January 2001, WERC was requested by the U.S. Congress to perform an independent peer review of the performance of the MWL. The results of the study are presented in WERC (2001).

As part of this study, members of the WERC review team developed a fate and transport model of tritium migration in the vadose zone beneath the MWL. The code GoldSim, a generalized object-oriented probabilistic spreadsheet, was used to model tritium contaminant concentrations and fluxes at various depths beneath the MWL over time. The model incorporated mass transport from a source (inventory), various release mechanisms, transport processes, migration pathways, and radionuclide decay.

The WERC team concluded that based on their model results, the spatial and temporal distribution of tritium activities measured in the vadose zone appear to be consistent with those expected, given the inventory, regional meteorology, subsurface soil conditions, and hydrologic parameters. Their modeling results showed good agreement with the Phase 2 RFI data regarding tritium distributions in subsurface soils beneath the MWL. The WERC team also concluded that future concentrations of tritium in subsurface soils at the MWL should decrease over the next 10 years, based on diffusion and natural decay of tritium.

2.1.2 Cover Performance Modeling

In addition to the fate and transport models discussed above, Sandia has conducted extensive cover performance modeling to predict infiltration through various thicknesses of alternative covers. The results from these studies were used to develop the MWL alternative cover design.

Early Cover Performance Modeling

Sandia's early cover performance modeling studies utilized multiple codes to assess infiltration through various thicknesses of alternative covers. The codes used included the water balance model, HELP-3 (Schroeder et al. 1994), and two unsaturated flow models, UNSAT-H (Fayer and Jones 1990) and VS2DT (Healy 1990).

The earlier modeling studies are documented in Wolford (1998); SNL (April 1999); and culminate with the modeling results presented in the original MWL design document, "Deployment of an Alternative Cover and Final Closure of the Mixed Waste Landfill, Sandia National Laboratories, New Mexico" (SNL September 1999). This report was submitted to the NMED in September 1999 for technical review and comment, and was later published as a SAND report by Peace et al. in 2003. The cover performance modeling results from the report are also presented in Section 5.3 of the main text of the MWL Corrective Measures Implementation Plan.

In order to demonstrate that the MWL alternative cover design complies with regulatory guidance, the hydrologic performance of the cover was modeled using HELP-3, UNSAT-H and VS2DT. These codes were used to predict infiltration through soil covers ranging in thickness from 1 to 5 ft. All three models demonstrated that deployment of a vegetated soil cover for final closure of the MWL would reduce infiltration into the landfill to a small percentage of the total precipitation. The models also demonstrated that a 3-ft-thick vegetated soil cover meets the intent of RCRA Subtitle C regulations. Additional cover thicknesses did not lead to significantly better performance. Additional details on the cover performance modeling using HELP-3, UNSAT-H and VS2DT are presented in Section 5.3 of the MWL CMI Plan.

Recent Cover Performance Modeling

The most recent cover performance modeling was conducted in 2003 and 2004 using site-specific climate, hydrologic, and vegetation input parameters. The modeling simulated infiltration of water through the MWL soil cover using the one-dimensional, numerical code UNSAT-H. UNSAT-H is a Richards' equation-based model that simulates infiltration, unsaturated flow, redistribution, evaporation, plant transpiration, and deep infiltration of water. The modeling results corroborated the results from earlier modeling studies. The recent modeling results are published in the SAND report entitled, "Calculation Set for Design and Optimization of Vegetative Soil Covers" (Peace and Goering, 2005). The modeling results were used to determine infiltration input parameters for the MWL probabilistic performance-assessment model.

One of the objectives of the modeling was to assess whether a 3-ft soil cover would meet the EPA-prescribed technical equivalency criteria. The EPA performance-based, technical equivalency criteria used are 31.5 millimeter (mm)/year (yr), or less, for net annual infiltration and 1×10^{-7} centimeter (cm)/second (s) average infiltration rate, based on a hydraulic conductivity of 1×10^{-7} cm/s and the assumption of unit-gradient conditions. The modeling results verified that the 3-ft MWL cover will meet the EPA-prescribed technical equivalency criteria for RCRA landfills under both present and future conditions.

Present conditions were simulated by modeling infiltration through various thicknesses of an engineered cover, while future conditions were simulated by modeling infiltration through various thicknesses of soil under natural conditions (i.e. the "natural analog"). The recent cover modeling results are discussed further in Section 3.4 below. Complete modeling input parameters, boundary conditions, and results are presented in Peace and Goering (2005).

2.2 Probabilistic Performance-Assessment Modeling Approach

This section summarizes the approach used in this study to provide a comprehensive performance assessment of the MWL. Previous studies have looked at individual components of the landfill performance, and nearly all of the studies relied on deterministic evaluations. This study describes a probabilistic performance-assessment approach that captures the inherent uncertainties in the system while honoring site-specific features, processes, and parameters. Sensitivity analyses are also introduced that utilize the probabilistic results to identify the parameters and processes that are most important to the simulated performance metrics.

A performance assessment is defined in DOE M 435.1-1 as "an analysis of a radioactive waste disposal facility conducted to demonstrate there is a reasonable expectation that performance objectives established for the long-term protection of the public and the environment will not be exceeded following closure of the facility." In addition, DOE M 435.1-1 states that the method used for the performance assessment must include uncertainty analyses. A method that addresses these requirements has been used for the Waste Isolation Pilot Plant (DOE, 1996), the Yucca Mountain Project (DOE, 1998), and the intermediate-depth Greater Confinement Disposal Boreholes (Cochran et al., 2001) to assess the long-term performance of nuclear waste repositories. Probabilistic performance assessments have also been used for sites with uranium mill tailings (Ho et al., 2004). A similar systematic approach has been used here to conduct a performance assessment of the MWL. The approach is outlined as follows:

1. Develop and screen scenarios based on regulatory requirements (performance objectives) and relevant features, events, and processes

2. Develop models of relevant features, events, and processes

3. Develop values and/or uncertainty distributions for input parameters

4. Perform calculations and sensitivity/uncertainty analyses

5. Compare results to performance objectives, identify important parameters and processes, and provide feedback to improve calculations, as needed

In Step 1, a scenario is identified as a well-defined sequence of features, events and processes that describes possible future conditions at the disposal site. An example of a scenario is the release of radionuclides from a landfill via the vadose zone to the aquifer, where water is pumped from a well and ingested by an individual. The decision to evaluate various scenarios depends, in part, on relevant performance objectives set forth by regulatory requirements. In addition, scenarios should be chosen that represent features, events, and processes that are relevant to the specific site being evaluated.

Step 2 develops the models that are necessary to simulate the chosen scenarios in the performance assessment. The models that are used vary in complexity, and a hierarchy of models can exist. A conceptual model of each scenario is developed to guide the development of more detailed mechanistic models of individual features, events, and processes that comprise the scenario. These detailed models are then integrated into a total-system model of the entire scenario. The integration of the more detailed models may include the models themselves or a simplified abstraction of the model results.

In Step 3, values are assigned to the parameters to populate the models. If the parameter is well-characterized, a single deterministic value may be assigned. However, uncertainty and/or variability in the parameter may require the use of distributions (e.g., log-normal, uniform) to define the values. Experimental data, literature sources, and professional judgment are often used to determine these distributions. The development of uncertainty distributions for parameters used in this study is described in Section 3.3.

In Step 4, calculations are performed using the integrated models. Because stochastic parameters are used, a Monte Carlo approach is taken to create an ensemble of simulations that use different combinations of the input parameters. For each run (realization), a value for each input parameter is sampled from the uncertainty distribution, and the simulation is performed. The results of each realization are equally probable, and the collection of simulation results yields an uncertainty distribution that can be compared to performance objectives to assess the risk of exceeding those performance objectives or metrics. Sensitivity analyses can also be performed to determine which parameters the performance metrics are most sensitive to (see Section 2.2.1).

The last step (Step 5) is to analyze and compare the results with relevant performance objectives. The findings are typically documented as cumulative distribution functions that present the probability of exceeding a performance objective. Important parameters and processes are also identified through sensitivity analyses. Together, these results may be used to assess the overall performance, prioritize site characterization, evaluate alternative designs, or identify triggers for future actions to address long-term monitoring requirements for regulatory compliance. In this study, the primary purpose of the performance assessment is to determine which contaminants and performance objectives are at risk based on the simulated performance of the MWL. This information will then provide a basis for the triggers that are identified and recommended for the site.

2.2.1 Sensitivity Analyses

A probabilistic performance assessment provides not only a quantification of uncertainties in the simulated performance metrics, it also allows for a quantified sensitivity analysis to be performed. A sensitivity analysis of the probabilistic assessment results can provide valuable information regarding the processes and parameters that are most important to the simulated performance metric(s). This information provides understanding about the relationship between uncertainty in individual input parameters and the uncertainty in the performance of the system. In addition, knowledge of the parameters having the greatest influence on future performance can be used to help prioritize site characterization activities, to help optimize landfill cover design, and to assist in the design of monitoring systems and triggers. Using a sensitivity analysis provides the quantitative information necessary to ensure that resources are directed to those aspects of the cover system that “drive” performance and not on those aspects of cover design that have little significance.

The sensitivity of the performance-assessment model can be determined from the Monte Carlo probabilistic realizations using regression analysis. Multiple regression analysis involves construction of a linear regression model of the simulated output (the dependent variable) and the stochastic input variables (independent variables) using a least-squares procedure. Stepwise linear regression is a modified version of multiple regression that selectively adds input parameters to the regression model in successive steps (Helton and Davis, 2000). In this method, a sequence of regression models is constructed that successively adds the most important input parameters to the regression to improve the overall correlation. In the end, the sensitivity analysis identifies those parameters that are significantly correlated to the performance metric, and omits those parameters that are not. This study uses a stepwise linear rank regression to perform sensitivity analyses on simulated performance metrics that are at risk of being exceeded.

3. Performance-Assessment Modeling of the Mixed Waste Landfill

3.1 Scenarios and Performance Objectives

In this study, relevant contaminants of concern were grouped into the following categories: (1) radionuclides, (2) heavy metals, and (3) VOCs. Table 1 summarizes the specific contaminants, scenarios, and performance objectives that were considered in this study. In general, the two pathways of concern include transport of volatile or gas-phase contaminants from the MWL to the atmosphere, and migration of aqueous-phase or vapor-phase contaminants through the vadose zone to the groundwater. For each of these primary pathways, relevant performance objectives and metrics were identified for each of the contaminants of concern. The chosen scenarios represent the most likely releases of contaminants from the MWL based on estimated inventories, contaminant properties, and previous studies.

Table 1. Summary of scenarios and performance objectives used in the performance assessment of the MWL.

Scenario	Description	Performance Objectives[a]
1	Water percolates through the cover to the waste	• Infiltration through the cover shall be less than 10^{-7} cm/s (a unit-gradient flow is assumed to equate infiltration to hydraulic conductivity) (U.S. EPA 40 CFR 264.301)
2	Tritium diffuses to the atmosphere and migrates via gas and aqueous phases through the vadose zone to the groundwater	• Dose to the public via the air pathway shall be less than 10 mrem/yr (excludes radon) (U.S. EPA 40 CFR 61.92) • Dose from beta particles and photon emitters shall be less than 4 mrem/yr (U.S. EPA 40 CFR 141.66; U.S. EPA, 2003) • Tritium concentrations in groundwater shall not exceed 20,000 pCi/L (40 CFR 141.66 Table A; tied to 4 mrem/yr)
3	Radon steadily diffuses to the atmosphere and migrates via gas and aqueous phases through the vadose zone to the groundwater	• The average flux of radon-222 gas shall be less than 20 pCi/m^2/s at the surface of the landfill (U.S. EPA 40 CFR 192) • Radon concentrations in groundwater shall not exceed 300 pCi/L (proposed EPA rules, Federal Register: November 2, 1999 (Volume 64, Number 211) Pages 59345-59378)
4	One or more radionuclides migrate via the aqueous phase through the vadose zone to the groundwater	• Maximum concentrations in groundwater of gross alpha particle activity (including radium-226 but excluding radon and uranium) is 15 pCi/L (U.S. EPA 40 CFR 141.66; U.S. EPA, 2003) • Uranium concentrations in groundwater shall not exceed EPA MCL of 30 μg/L (U.S. EPA 40 CFR 141.66; U.S. EPA, 2003) • Dose from beta particles and photon emitters shall be less than 4 mrem/yr (U.S. EPA 40 CFR 141.66, U.S. EPA, 2003)
5	Lead and cadmium migrate via the aqueous phase through the vadose zone to the groundwater	• Lead concentrations in groundwater shall not exceed the EPA action level of 15 μg/L (U.S. EPA, 2003) • Cadmium concentrations in groundwater shall not exceed the EPA MCL of 5 μg/L (U.S. EPA, 2003)
6	PCE migrates through the vadose zone to the groundwater	• PCE concentrations in groundwater shall not exceed the EPA MCL of 5 μg/L (U.S. EPA 40 CFR 141.61; U.S. EPA, 2003

MCL = Maximum Contaminant Level

[a]The point of compliance is taken at the boundary of the waste site. The period of performance was specified as 1,000 years in the regulations for some of the performance metrics, but for many of the performance metrics, the period of performance was not specified. In this study, a 1,000 -year period was simulated.

3.2 Performance-Assessment Models

The following sections describe the models that were developed and used to simulate the fate and transport of the different contaminants in the various scenarios summarized in Table 1.

3.2.1 FRAMES/MEPAS

The aqueous transport of heavy metals (lead and cadmium) and the radionuclides were simulated using the probabilistic simulation tools FRAMES[1] (Framework for Risk Analysis in Multimedia Environmental Systems; Whelan et al., 1997) and MEPAS[2] (Multimedia Environmental Pollutant Assessment System; Whelan et al., 1992), developed by Pacific Northwest National Laboratory. The FRAMES system, which integrates the fate and transport models comprising MEPAS, allows for a holistic approach to modeling in which models of different type (i.e., source, fate and transport, exposure, health impact), resolution (i.e., analytical, semi-analytical, and numerical), and operating platforms can be combined as part of the overall assessment of contaminant fate and transport in the environment. The FRAMES system employs a graphical user interface for integrating computer models, an extensive contaminant database, a probabilistic sensitivity/uncertainty module, and textual and graphical viewers for presenting modeling outputs.

Existing models in FRAMES include those derived from MEPAS (Whelan et al., 1992). MEPAS is a physics-based environmental analysis code that integrates source-term, transport, and exposure models for endpoints such as concentration, dose, or risk. MEPAS is capable of computing contaminant fluxes for multiple routes, which include leaching to groundwater, overland runoff, volatilization, suspension, radioactive decay, constituent degradation, and source/sink terms. In this study, only the source-term and vadose-zone models were implemented. The source-term model conservatively simulates leaching from the waste zone (assuming no containment) based on either the solubility or the inventory-limited concentration (Streile et al., 1996). Decay of constituents can also occur within the source-term model. The transport of the contaminant through the vadose-zone is then simulated assuming liquid-phase advection, dispersion, adsorption, and decay of the contaminant (Whelan et al., 1996). It should be noted that gas-phase transport is not assessed in FRAMES/MEPAS. Separate models were used to evaluate the gas-phase transport of tritium, radon, and VOCs.

In this study, the aquifer concentration and subsequent dose, if applicable, were conservatively estimated based on the simulated concentration of the constituent in the groundwater at the interface of the vadose-zone and the water table (e.g., dilution caused by transport in the saturated zone was ignored). Section 3.3 presents the input parameters that were used in the radionuclide-transport models.

Uncertainty analyses are performed in FRAMES using the sensitivity module. The sensitivity module can be attached to any model that has been integrated into FRAMES and allows the user to stochastically vary any input parameter that is identified in the process models. Input parameters can be stochastically varied by a distribution, correlation coefficient, an equation, or

[1] http://mepas.pnl.gov/FRAMESV1 (FRAMES v. 1.5)
[2] http://mepas.pnl.gov/earth/mepasmain.html (MEPAS v. 4.1.1)

any combination of these three options. Four distributions are currently available: (1) uniform, (2) log uniform, (3) normal, and (4) log normal. The sensitivity module utilizes the Latin Hypercube Sampling (Wyss and Jorgensen, 1998) technique to minimize the number of modeling runs that must be performed to accurately represent distributions selected by the user. In this study, 100 realizations were simulated for each scenario (a sensitivity analysis was performed using 100 vs. 200 realizations in Section 3.5.2.2, and results showed that 100 realizations were sufficient to adequately represent the distribution of the simulated output).

3.2.2 Transient Gas- and Liquid-Phase Transport

A separate model was used to model the transient transport of tritium in both the gas and liquid phases at the MWL. As stated in the previous section, FRAMES/MEPAS was used to simulate the transport of radionuclides such as tritium, but only in the liquid phase. Tritium, in the form of tritiated water, is volatile and can be transported via both the gas and liquid phases. Regulatory metrics exist for dose caused by exposure to tritium (a beta particle emitter) in both the air and groundwater pathways (see Table 1). Also, because the half-life of tritium is relatively short (12.3 years), a transient analysis was required. Therefore, the transport of tritium was modeled using a transient model that accounts for advective liquid-phase transport, diffusive gas-phase transport, decay, and adsorption (if applicable) in the vadose zone (Jury et al., 1983; Jury et al., 1990). This same model was also used to model the transport of PCE. In this model, a contaminated zone is assumed to initially exist with a defined thickness and concentration. Over time, the contaminant migrates and decays (if applicable) assuming a flux boundary condition at the surface, defined by an atmospheric boundary layer thickness (see Jury et al., 1983) and a zero concentration boundary beneath the waste zone at a location infinitely far away from the source. Superposition is used to account for a clean overburden (cover) above the waste zone (Jury et al., 1990). The analytical solution to this model was implemented in Mathcad,® and a Monte Carlo analysis was implemented with the uncertain variables using 100 realizations. Section 3.3 presents the input parameters and distributions that were used in the tritium- and PCE-transport models.

3.2.3 Steady-State Gas- and Liquid-Phase Transport

Radon-222 is generated from the decay of radium-226, which is a decay product of uranium-238. Because these parent constituents have long half lives, the source of radon-222 production is assumed to last indefinitely. Therefore, the transient model described in the previous section that accounts for a finite source of contaminant is not appropriate. Instead, a steady-state model of radon transport was developed to account for steady generation of radon-222, advective liquid-phase transport, diffusive gas-phase transport, and decay (see Appendix A in Section 7). Mathcad® was used to provide a Monte Carlo analysis of the analytical solution using 100 realizations. Section 3.3 presents the input parameters and distributions that were used in the radon-transport model.

3.3 Input Parameters and Distributions

The constituents that were included in the performance assessment of the MWL are summarized in Table 2. The parameter values and distributions that were used are also summarized in the table. The adsorption coefficient (K_d) was assumed to be an uncertain parameter, so a range of values was obtained from the literature for the constituent and soil type (sandy loam) at the MWL. A log-uniform distribution was used to emphasize the lower values in the distribution. The inventory of each constituent was also assumed to be an uncertain variable. The estimated inventory from previous reports and studies was used as the lower bound in a uniform distribution for each constituent. The lower bound was multiplied by two to obtain the upper bound for the assumed uniform distribution. The maximum solubility obtained from the literature for each constituent was used. All other parameters were obtained from site-specific reports, scientific literature, or EPA recommendations.

Table 3 summarizes the parameters and distributions used to define the contaminated waste zone (source term) in the models. The waste-zone length, width, and thickness is based on the size of the pits, trenches, and dimensions of the MWL. The maximum thickness of the cover is based on the design specifications given in Peace et al. (2005). The minimum thickness of the cover is set equal to zero as a bounding value to account for the possibility that complete erosion of the cover may occur in the future. This is a conservative bounding assumption since the intent is to maintain the integrity of the cover at the MWL.

Table 4 summarizes the parameters and distributions used to describe the vadose-zone in the models. Uncertainty was included for a number of variables including thickness of the vadose zone, infiltration rate, hydraulic conductivity, and site-specific transport parameters. The distributions used for the various vadose-zone parameters were derived from site-specific data or literature pertaining to the constituents and scenarios evaluated in this study. The liquid- and gas-phase tortuosity coefficients are used to calculate effective diffusion coefficients in porous media. The tortuosity coefficient accounts for the increased tortuosity and reduced area available for diffusion in porous media. The minimum value is based on formulation by Millington (1959), and the maximum value is assumed to be equal to one (the upper bound), which yields the maximum diffusion. Studies of enhanced vapor diffusion have shown that large values of the tortuosity coefficient (yielding diffusion rates equivalent to those in free space) are possible in unsaturated porous media because of evaporation and condensation mechanisms across liquid islands in pores (Ho and Webb, 1998).

Finally, Table 5 summarizes the parameters and distributions used to estimate dose due to exposure via the atmospheric (e.g., inhalation) or groundwater pathway. Dose via inhalation and dermal adsorption of gas-phase tritium was calculated based on the surface flux ($pCi/m^2/s$) of tritium determined in the models.[3] The length and width of the waste zone was used to determine the flux rate of tritium at the surface (pCi/s), and the average wind speed and vertical mixing height was used to determine the average concentration above the landfill. The inhalation rate was then used to estimate the human intake of gas-phase tritium, and the dose-

[3] Inhalation and dermal adsorption of gas-phase radon and PCE were not used as performance metrics in this analysis because the enforceable regulatory metrics pertaining to radon and PCE do not use dose (surface flux of radon and groundwater concentration of PCE was used). Table 1 summarizes the performance metrics that were used for these constituents.

conversion factor (Table 2) was used to determine the dose. For groundwater exposure, a conservative estimate for water ingestion (10 L/day) was used together with the simulated groundwater concentrations to determine intake. The assumed water ingestion rate of 10 L/day is five times greater than the EPA drinking-water standard of 2 L/day and is intended to account for indirect sources of water ingestion and absorption such as consumption of vegetables and fruits irrigated by contaminated water. The dose-conversion factor was then used to estimate dose via the groundwater pathway.

Table 2. Summary of input parameters and distributions for constituents used in the models.

Constituent and Molecular Weight	Inventory[a]	Half-Life[b]	Specific Activity (Ci/g)[c]	Adsorption Coefficient, K_d (mL/g)[d]	Max Solubility (mg/L)[e]	Liquid-Phase Diffusion Coefficient (m^2/s)[f]	Gas-Phase Diffusion Coefficient (m^2/s)[f]	Henry's Constant (C_g/C_l)[g]	Dose Conversion Factor (rem/pCi)[h]
Americium-241$^{\alpha}$	Uniform: 0.04 - 0.08 Ci	433 yrs	3.43	Log-Uniform: 1900 – 9600	2.4×10^{4}	6×10^{-10}	N/A	N/A	3.64×10^{-6}
Cesium-137$^{\beta}$	Uniform: 410 – 820 Ci	30.2 yrs	86.4	Log-Uniform: 30 – 4600	137,000	6×10^{-10}	N/A	N/A	5.0×10^{-8}
Cobalt-60$^{\beta}$	Uniform: 3500 – 7000 Ci	5.27 yrs	1130	Log-Uniform: 60 – 1300	600	6×10^{-10}	N/A	N/A	2.69×10^{-8}
Plutonium-238$^{\alpha}$	Uniform: 0.0012 - 0.0024 Ci	87.7 yrs	17.1	Log-Uniform: 80 – 520	0.24	6×10^{-10}	N/A	N/A	3.2×10^{-6}
Plutonium-239$^{\alpha}$	Uniform: 0.0012 - 0.0024 Ci	2.41×10^{4} yrs	0.0621	Log-Uniform: 80 – 470	0.24	6×10^{-10}	N/A	N/A	3.54×10^{-6}
Radium-226$^{\alpha}$	Uniform: 6-12 Ci	1,600 yrs	0.989	Log-Uniform: 500 – 36,000	0.45	6×10^{-10}	N/A	N/A	1.32×10^{-6}
Radon-222$^{\alpha}$	Constant generation from Radium-226	3.82 days	1.54×10^{5}	0	N/A	$0.07\exp[-4(S - S\phi^2 + S^5)]$ where S=liquid saturation, ϕ=porosity		0.26^{-1}	1.44×10^{-8} (inhalation)
Strontium-90$^{\beta}$	Uniform: 410 -820 Ci	29.1 yrs	137	Log-Uniform: 15 – 20	90,000	6×10^{-10}	N/A	N/A	1.42×10^{-7}
Thorium-232$^{\alpha}$	Uniform: 1 – 2 Ci	1.4×10^{10} yrs	1.10×10^{-7}	Log-Uniform: 20 – 2000	23	6×10^{-10}	N/A	N/A	2.73×10^{-6}
Tritium$^{\beta}$ H-3	Uniform: 2400 – 4800 Ci	12.3 yrs	9690	0	N/A	2.3×10^{-9}	2.6×10^{-5}	1.7×10^{-5}	6.4×10^{-11} (inhalation; x1.5 to include dermal absorption)

Constituent and Molecular Weight	Inventory[a]	Half-Life[b]	Specific Activity (Ci/g)[c]	Adsorption Coefficient, K_d (mL/g)[d]	Max Solubility (mg/L)[e]	Liquid-Phase Diffusion Coefficient (m^2/s)[f]	Gas-Phase Diffusion Coefficient (m^2/s)[f]	Henry's Constant (C_g/C_l)[g]	Dose Conversion Factor (rem/pCi)[h]
Uranium-238[α]	Uniform: 9.3 – 18.6 Ci	4.47×10^9 yrs	3.35×10^{-7}	Log-Uniform: 0.4 – 15	24	6×10^{-10}	N/A	N/A	2.55×10^{-7}
Cadmium 112.41	Uniform: 1350 – 2700 kg	stable	N/A	Log-Uniform: 8 – 80	1.4×10^6	6×10^{-10}	N/A	N/A	N/A
Lead 207.2	Uniform: 128,000 – 256,000 kg	stable	N/A	Log-Uniform: 270 – 4360	4.43×10^5	6×10^{-10}	N/A	N/A	N/A
PCE 165.83	Uniform: 5 – 70 kg	Log-Uniform: 9 mos – 10^{10} yrs	N/A	Log-Uniform: 0.038 - 2	N/A	9.2×10^{-10}	9.5×10^{-6}	0.42	N/A

N/A–Not Applicable or not used in the model; for solubility, this indicates that the value is not limiting

[α]Alpha particle; [β]Beta particle

[a]Minimum inventory of all constituents except cadmium and PCE was estimated from values in SNL (1993); maximum value was assumed to be twice the minimum value. Cadmium inventory was estimated from measured soil concentrations (Peace et al., 2002) and maximum simulated penetration depth (120 feet) of coolant water potentially carrying the cadmium (Wolford, 1997). PCE inventory is estimated from measured soil-gas concentrations (Peace et al., 2002); the maximum measured gas concentration (5,900 ppb) was used as a minimum value in a uniform distribution increasing to ten times this value (calibrated to available data). The maximum areal extent of the MWL was used (430 feet x 300 feet) along with an uncertain thickness ranging from 10-27 feet (see Table 3 for waste-zone description).

[b]Lide (2005); half-life of PCE is assumed to range from 9 months (EPA fact sheet: www.epa.gov/OGWDW/dwh/t-voc/tetrachl.html) to 10^{10} yrs (no degradation)

[c]Specific activity is calculated as 3.575×10^5/(half-life (yrs) x molecular weight)

[d]U.S. EPA (1999), Sheppard and Thibault (1990), Looney et al. (1997), EPA fact sheet: www.epa.gov/OGWDW/dwh/t-voc/tetrachl.html

[e]Looney et al. (1997), Chen et al. (2002), Ohe et al. (2002), Elless and Lee (1998), BSC (2005), and EPA Online Fact Sheets (www.epa.gov/safewater/dwh/t-ioc/cadmium.html; www.epa.gov/safewater/dwh/t-ioc/lead.html). Based on the maximum inventory and minimum waste volume possible, the solubility may potentially limit the maximum aqueous source concentration for radium-226, thorium-232, uranium-238, and lead; all other constituents are not limited by the solubility.

[f]Whelan et al. (1996), Smiles et al. (1995), Rogers et al. (1994), U.S. NRC (1989), Reid et al. (1987)

[g]Rogers et al. (1984), U.S. NRC (1989), Smiles et al. (1995), steam tables, and EPA's online Henry's Constant calculator (www.epa.gov/athens/learn2model/part-two/onsite/esthenry.htm)

[h]U.S. EPA (1988)

Table 3. Summary of input parameters and distributions for the waste zone.

Input Parameter	Value or Distribution	Basis and Comments
Waste-Zone Length [m]	Uniform 3.05 – 131	Minimum value determined by size of individual pit (10'). Maximum value determined by extent of Mixed Waste Landfill.
Waste-Zone Width [m]	Uniform 3.05 – 91.4	Minimum value determined by size of individual pit (10'). Maximum value determined by extent of Mixed Waste Landfill.
Waste-Zone Thickness [m]	Uniform 3.05 – 8.23	The thickness of the waste zone for all constituents except for cadmium is based on the depth of the trenches and pits, which range from 3 – 8 m (10 – 27 feet). The thickness of the cadmium contamination zone is assumed to be equal to 36.6 m (120 feet), which is the maximum simulated penetration depth of the coolant water that may have carried the cadmium (Wolford, 1997).
Thickness of Cover and Clean Overburden [m]	Uniform 0 – 4.88	Minimum value is assumed to be zero due to erosion.[a] Maximum value is based on maximum thickness of the cover at various locations (Peace et al., 2005).

[a]The intent is to maintain the integrity of the cover at the MWL. Complete erosion of the cover is a conservative bounding assumption for modeling purposes.

Table 4. Summary of input parameters and distributions for the vadose zone.

Input Parameter	Value or Distribution	Basis and Comments
Thickness of Vadose Zone[a] [m]	Uniform 133 - 148	Thickness of the vadose zone for all constituents except for cadmium is based on measured depths to the water table. The depth to the water table from the surface ranges from 141 – 151 m (461 - 495 feet) (Goering et al., 2002). The range of vadose-zone thicknesses accounts for the waste-zone thickness. For cadmium, the thickness is assumed to be 104 m (461 – 120 = 341 feet).
Infiltration Rate [m/s]	Uniform 1.18×10^{-11} – 6.12×10^{-11}	Minimum value based on infiltration through 2 ft of engineered cover under current climate (Peace and Goering, 2005); maximum value based on two times the current maximum precipitation in a natural analog vegetative cover to account for future climates (Waugh, 1997; Menking et al., 2004).
Saturated Hydraulic Conductivity [cm/day]	Log-Normal Mean log: 1.039 S.D. log: 0.705 Upper bound: 173 Lower bound: 0.38	Peace et al. (2003)
Porosity [-]	Uniform 0.302 – 0.445	Peace and Goering (2005)
Volumetric Moisture Content [-]	Uniform 0.053 – 0.225	Peace and Goering (2005)
Longitudinal dispersivity [m]	0.1 times the travel distance (vadose-zone thickness)	Based on field data reported in Gelhar et al. (1992). This is used in the FRAMES/MEPAS models for liquid transport to the groundwater.
Liquid-Phase Tortuosity Factor [-]	Uniform 0.001 – 1	Lower bound based on formulation of Millington (1959); upper bound is physical limit. This is used in the tritium and PCE models.
Gas-Phase Tortuosity Factor [-]	Uniform 0.1 – 1	Lower bound based on formulation of Millington (1959); upper bound is physical limit. This is used in the tritium and PCE models.

[a]Used only in FRAMES/MEPAS. For all other models, the depth to the water table (141-151 m) is used.

Table 5. Summary of input parameters and distributions for the biosphere.

Input Parameter	Value or Distribution	Basis and Comments
Atmospheric Boundary Layer Thickness [m]	Uniform 0.001 – 1	Minimum is based on values reported by Jury et al. (1983). Maximum is a conservative upper value.
Vertical Atmospheric Mixing Length [m]	2	Conservative value to encompass volume occupied by a human (Yu et al., 1993).
Average Wind Speed [m/s]	3.63	Average value based on seven years of site data (SNL Site Environmental Monitoring Reports 1990-1996).
Inhalation Rate [m^3/day]	20	U.S. EPA (1991)
Water Intake [L/day]	10	Conservative estimate to account for drinking water and indirect ingestion or absorption via plants, animals, showering, etc. Recommended value for drinking water is 2 L/day (U.S. EPA, 2000).
Distance to Receptor [m]	0	The point of compliance for groundwater concentrations is assumed to be at the boundary of the landfill. Receptor is assumed to be located adjacent to landfill for inhalation, and water used for drinking, irrigation, etc. is assumed to be drawn from the aquifer directly beneath the MWL.

Key Assumptions:

The key assumptions regarding the models and input parameters used in the performance assessment of the MWL are summarized below:

- Receptor located adjacent to MWL
    - Tritium dose caused by continuous inhalation and exposure of tritium flux directly above MWL.
    - Groundwater dose calculated based on concentrations in aquifer directly beneath MWL. Water intake assumed to be 10 L/day (five times EPA standard of 2 L/day for drinking water).
- Maximum waste inventory set equal to twice estimated values based on historical records.
- Sealed sources of radium-226 allowed to degrade in 1,000 years (emanation factor for radon-222 allowed to increase).
- Cover allowed to completely erode in 1,000 years.
- 1-D model: yields maximum transport to surface and groundwater.
- Bounding tortuosity coefficients: yields maximum diffusion rates.

3.4 Water Infiltration through the Cover

Infiltration of water through a proposed soil cover for the MWL was modeled using the one-dimensional, numerical code UNSAT-H (Peace and Goering 2005). UNSAT-H is a Richards' equation-based model that simulates infiltration, unsaturated flow, redistribution, evaporation, plant transpiration, and deep infiltration of water. The modeling was conducted in 2003 and 2004 using site-specific climate, hydrologic, and vegetation input parameters. The modeling results corroborated the results from earlier modeling studies presented in Section 5.3 of the MWL Corrective Measures Implementation Plan. Complete modeling input parameters, boundary conditions, and results are discussed in Peace and Goering (2005).

One of the objectives of the modeling was to assess whether the proposed 3-ft cover will meet the EPA-prescribed technical equivalency criteria. The EPA performance-based, technical equivalency criteria used in this study are 31.5 millimeter (mm)/year (yr), or less, for net annual infiltration and 1×10^{-7} centimeter (cm)/second (s) average infiltration rate, based on a hydraulic conductivity of 1×10^{-7} cm/s and the assumption of constant unit gradient conditions. The modeling results demonstrate that the proposed 3-ft MWL cover will meet the EPA-prescribed technical equivalency criteria for RCRA landfills under both present and future conditions.

3.4.1 Model Description

The modeling study was formulated in one dimension, vertically, and was discretized by placing computational nodes at predetermined vertical spacing in a conceptual soil profile to evaluate the performance of a cover 3 ft in thickness. Figure 3 shows a cross-section of the conceptual soil profile and its numerical discretization. A total of 30 nodes were used to discretize a conceptual soil profile 6 ft in thickness. A thickness of 6 ft is used so that the overlying nodes of interest are not adversely impacted by the lowermost boundary conditions.

The conceptual soil profile was simulated as a lithologic monolayer. A soil profile with uniform soil and hydrologic properties translates into a significant conservative estimate of liquid water flow. If multiple layers are simulated, the water potential in the underlying layer must equal the water potential in the overlying layer before flow into the lower layer occurs. Multiple layering in performance modeling as well as multiple layers in nature attenuate the downward flow of liquid water (e.g., multiple capillary barriers). UNSAT-H input parameters for the cover are summarized in Table 6-1 in Peace and Goering (2005). All parameters are site-specific and were carefully measured to obtain the most accurate estimate of infiltration possible.

Climatic data represent the site-specific conditions to the maximum extent possible. The historical rainfall record from Albuquerque International Sunport, dating from 1919 to 1996, was used to input precipitation and simulate infiltration through the cover. Two discrete sets of precipitation data were compiled from the historical record. The first data set, the "historical precipitation data," included 65 years of daily rainfall recorded from 1932 to 1996. The second data set, the "maximum precipitation data," included the 8 heaviest years' rainfall recorded between 1919 and 1996, repeated 8 times for a total of 64 years. The heaviest rainfall years were 1919, 1929, 1940, 1941, 1982, 1986, 1988, and 1992. These maximum precipitation data represent a climate change of 50% more precipitation overall (1.5 times the current level).

Precipitation during these years ranged from 12 in. to over 15 in. The current average annual precipitation for the Albuquerque area is 8.65 in./yr.

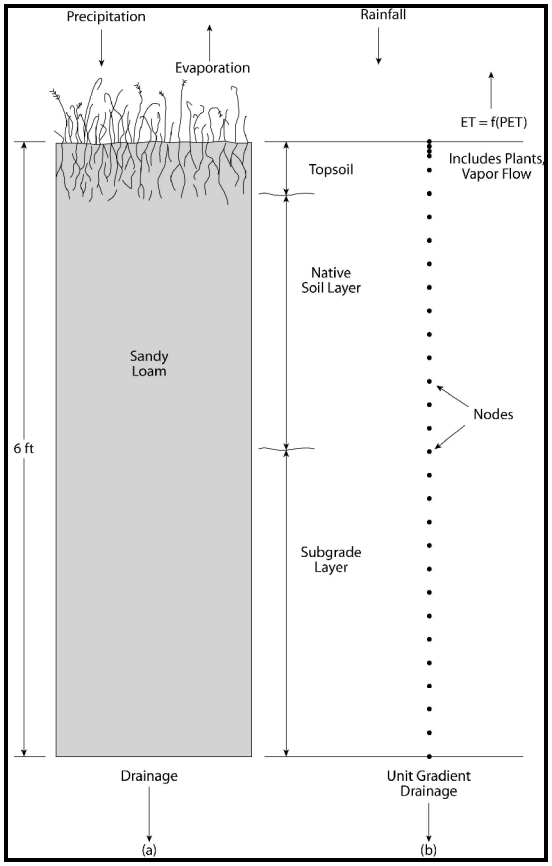

Figure 3. (a) Conceptual model for infiltration model. (b) Nodal discretization in UNSAT-H.

Literature evidence suggests that wetter conditions probably occurred during the last glacial episodes in the Southwest. Studies of paleoclimate during the Last Glacial Maximum suggest that precipitation in the Estancia basin, located west of the Manzano Mountains, nearly doubled relative to modern levels during brief, decade- to century-long episodes of colder and wetter climate (Menking et al. 2004). Farther west, studies of floral assemblages in late Pleistocene packrat middens near Yucca Mountain, Nevada, indicate that precipitation was an estimated 2.4 times modern levels during the Last Glacial Maximum (Menking et al. 2004).

Because precipitation in the southwest may have been significantly higher in the past, a precipitation multiplier of 2X was used to estimate maximum infiltration levels in the future through the MWL cover. A polynomial extrapolation of infiltration was developed using the results from modeling the “historical precipitation data” and the “maximum precipitation data”, and assuming that hydrologic properties of the cover are at equilibrium with the natural system.

Plant transpiration is the primary mechanism in removing water from a cover. Without plants, covers would only depend on evaporation to remove water from the soil profile. Vegetative input for the UNSAT-H code included root depth, root length density, leaf area index, growing season, and percent bare area. Root depth, root length density, leaf area index, growing season, and percent bare area for a climax community were measured in the field (Peace and Goering, 2005).

3.4.2 Model Results

The UNSAT-H code simulated infiltration through a soil cover with a climax community of native vegetation. The range of average infiltration rates for the MWL was predicted under current and future climate conditions. For both the current and future scenarios, the estimated infiltration rates through a 2-ft cover rather than a 3-ft cover were used to be conservative, as the model predicted infiltration through a 3 ft cover to be slightly negative, i.e. a net upward flux (Peace and Goering 2005).

Under present climate conditions, the model predicted the average infiltration rate through the proposed MWL cover to be 1.18 X 10^{-9} cm/s for the historical precipitation scenario and 5.34 X 10^{-9} cm/s for the maximum precipitation scenario.

Under future climate conditions, the properties of the MWL cover soils will gradually revert towards those of the natural soils around the landfill, as the bulk density and porosity of the soil equilibrate with natural conditions. Under these conditions, the model predicted the average infiltration rates to be 2.44 X 10^{-10} cm/s for the historical precipitation scenario and 1.04 X 10^{-9} cm/s for the maximum precipitation scenario.

Since the maximum precipitation scenario represents a 50% increase in precipitation over the historical precipitation scenario, a polynomial regression for infiltration as a function of precipitation can be determined (assuming that zero infiltration occurs with zero precipitation). We assign a normalized precipitation value of one to the historical precipitation scenario and a value of 1.5 to the maximum precipitation scenario. The quadratic regression then allows extrapolation to future climates where the precipitation is expected to be twice as high as present values. If the future precipitation is twice as high as current precipitation, the precipitation multipliers will increase to 2X for the historical scenario and 3X for the maximum scenario. Applying these multipliers to the quadratic regression yields estimated future infiltration rates of 2.29 X 10^{-9} cm/s for the historical precipitation scenario and 6.12 X 10^{-9} cm/s for the maximum precipitation scenario (Figure 4). We use 6.12×10^{-9} cm/s as an upper bound for the infiltration distribution to represent maximum precipitation conditions in the future, and we use 1.18×10^{-9} cm/s as a lower bound for the infiltration distribution to represent current precipitation conditions with the engineered cover design.

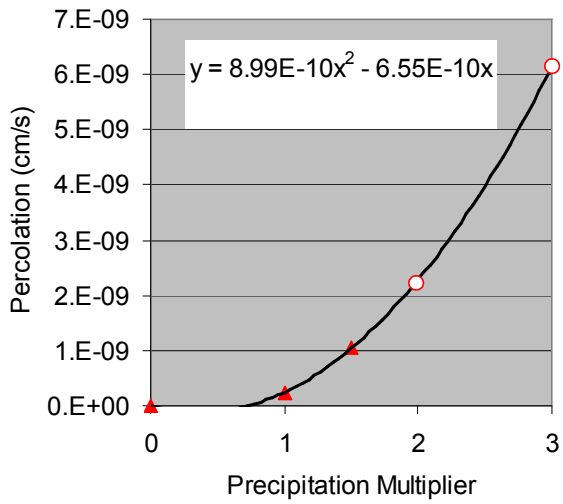

Figure 4. Polynomial regression used to estimate future infiltration values as a function of precipitation multipliers. Triangles denote simulated values; circles denote extrapolated values.

In summary, the modeling results demonstrate that the proposed 3-ft soil cover will meet the EPA-prescribed technical equivalency criteria for both present and future climate conditions, even if precipitation is significantly higher. The EPA performance-based, technical equivalency criteria are 31.5 mm/yr or less for net annual infiltration and 1 x 10^{-7} cm/s average infiltration rate, based on a hydraulic conductivity of 1 x 10^{-7} cm/s and the assumption of constant unit gradient conditions. Predicted average infiltration rates through the MWL cover are expected to range from 1.18 X 10^{-9} cm/s for present conditions to 6.12 X 10^{-9} cm/s for future conditions, under the assumption of significantly higher precipitation. These infiltration rates are considerably lower than the EPA performance-based, technical equivalency criterion of 1 x 10^{-7} cm/s.

3.4.3 Summary of Key Results and Assumptions

- Simulations of infiltration through the engineered cover at the MWL show that the net annual infiltration will be less than the regulatory metric of 10^{-7} cm/s.
- Predicted average infiltration rates through the MWL cover are expected to range from 1.18 X 10^{-9} cm/s for present conditions to 6.12 X 10^{-9} cm/s for future conditions.
- Key Assumption:
    - Predicted range of infiltration rates was based on simulated infiltration averaged over 64 years of data (as opposed to selected annual or daily averages).

3.5 Fate and Transport of Tritium

3.5.1 Model Description

As described in Section 3.2.2, the fate and transport of tritium was simulated using a model that accounts for transient liquid advection, gas diffusion, and decay (Jury et al., 1983; Jury et al., 1990). The upper boundary condition at the surface allowed for gas-phase transport of tritium to the atmosphere across a prescribed (uncertain) boundary-layer thickness. The concentration at the bottom of the model was specified as zero infinitely far away from the source.

The initial inventory of tritium was estimated from past records (SNL, 1993), and the extent of the contaminated waste zone was allowed to vary from the size of an individual pit to the entire size of the MWL. The inventory was allowed to vary between the estimated value (as a lower bound) and an upper bound equal to twice the estimated value. The simulations were run until tritium concentrations decreased to negligible values in the system. One hundred realizations were used in the simulations.

3.5.2 Model Results

3.5.2.1 Comparison to Field Data

In 1990 and 1993, measurements of tritium at the surface and at locations in the subsurface were measured at the MWL (Johnson et al., 1995). These measurements were used as a reference to check the simulated results of the model. Figure 5 shows the simulated tritium surface flux as a function of time for 100 realizations. The minimum and maximum measured tritium surface flux values taken in 1993 are also shown in the figure. The measured values are shown spanning 5 to 33 years because the actual time elapsed since the tritium was emplaced is uncertain. Emplacement of waste at the MWL began in 1960 and ended in 1988; therefore, the measured values sampled in 1993 could have occurred between 5 and 33 years after emplacement. Results show that the simulated results during this span of time are either within or above the measured bounding values. Figure 6, Figure 7, and Figure 8 show similar plots and results for different locations in the subsurface. In most cases, the simulated fluxes and concentrations are higher than the measured values. These results and comparisons provide evidence that the models can provide realistic values for the simulated outputs. In addition, the comparisons confirm that the model is producing conservatively high results for surface fluxes and subsurface concentration because of the conservative values and distributions used for the model parameters.

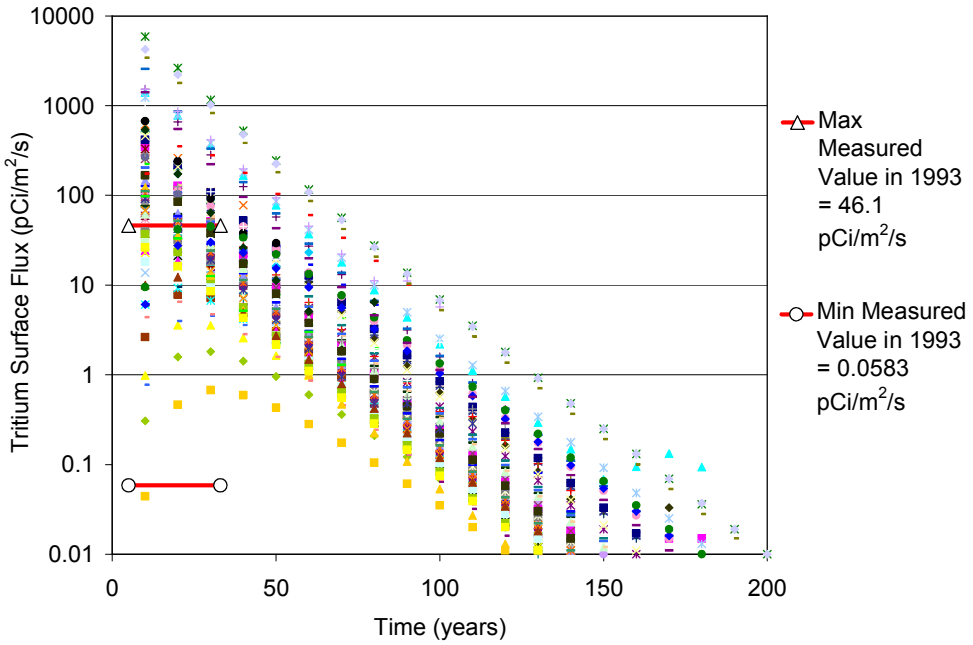

Figure 5. Comparison of simulated tritium surface flux as a function of time for 100 realizations with range of measured values in 1993.

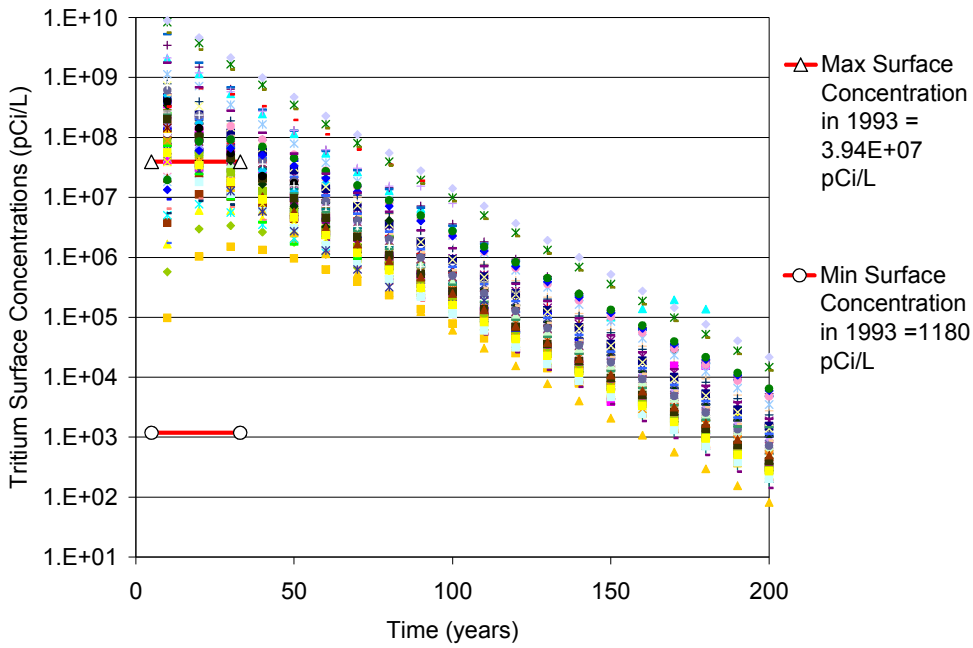

Figure 6. Comparison of simulated tritium surface concentration as a function of time for 100 realizations with range of measured values in 1993.

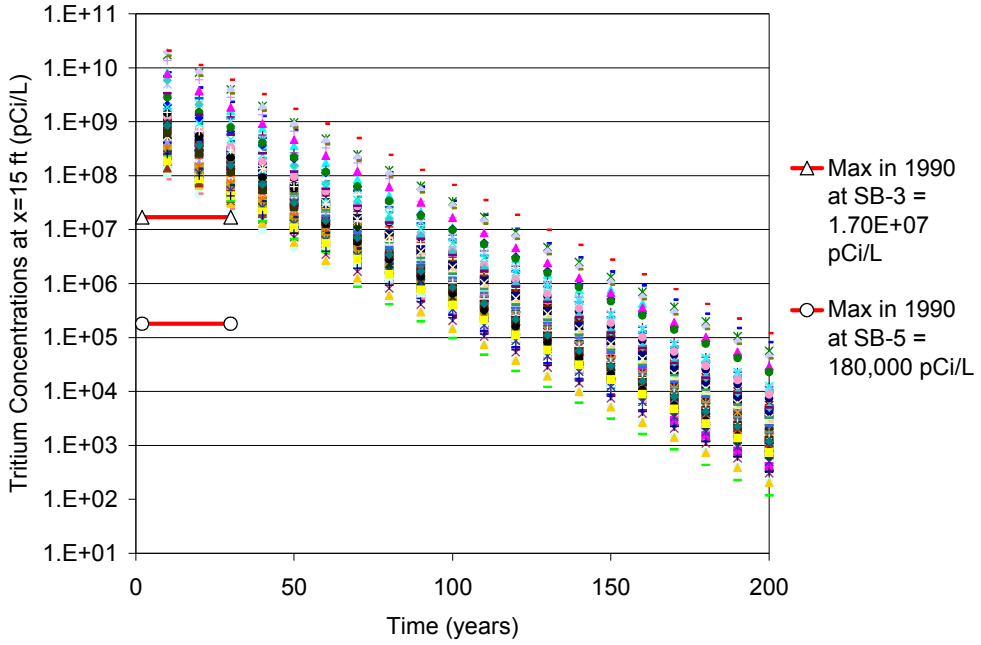

Figure 7. Comparison of simulated tritium concentration at a depth of 15 feet as a function of time for 100 realizations with measured maximum values in 1990.

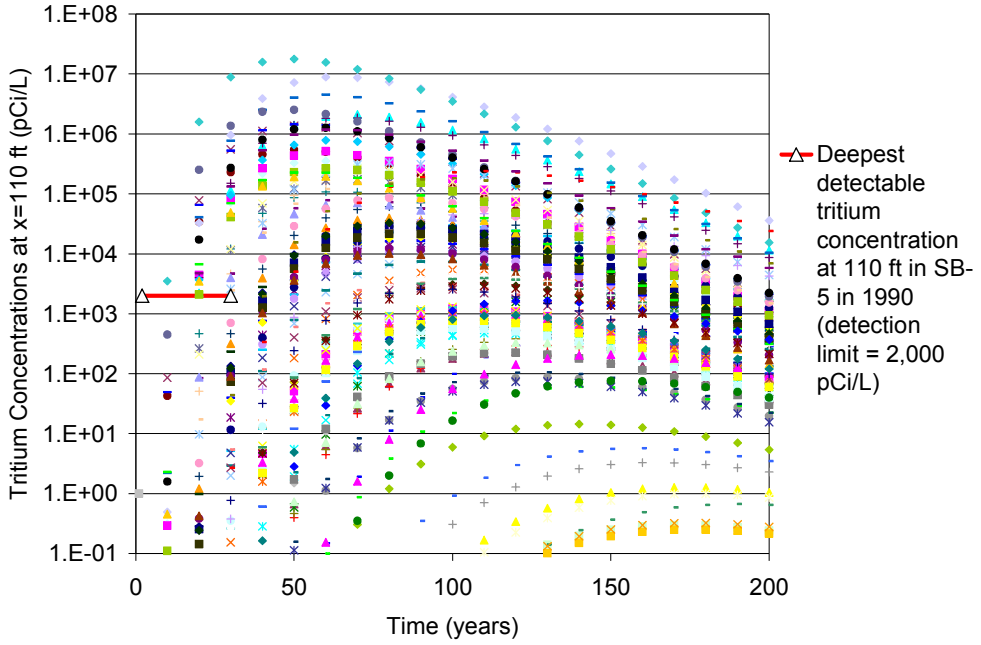

Figure 8. Comparison of simulated tritium concentration at a depth of 110 feet as a function of time for 100 realizations with measured value in 1990.

3.5.2.2 Comparison to Performance Objectives

The simulated tritium concentrations reaching the groundwater are shown in Figure 9 for all 100 realizations as a function of time. The peak tritium groundwater concentrations are all small, and Figure 10 shows the cumulative probability of the peak concentrations for 100 realizations and 200 realizations. The results show that the simulated tritium groundwater concentrations are all well below 20,000 pCi/L. In addition, the distribution resulting from 100 realizations is nearly the same as the distribution resulting from 200 realizations (therefore, all subsequent analyses only use 100 realizations).

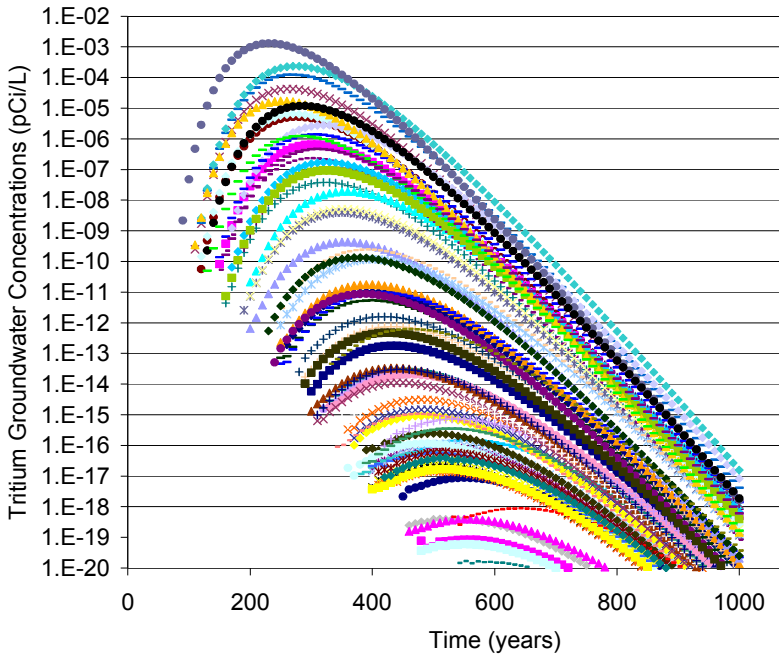

Figure 9. Simulated tritium concentrations in the aquifer as a function of time for 100 realizations.

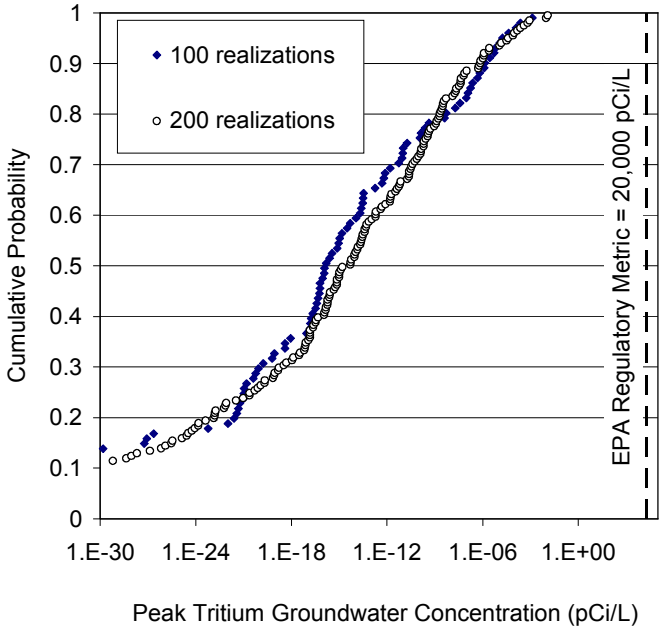

Figure 10. Cumulative probability for simulated peak tritium groundwater concentrations using 100 and 200 realizations.

Figure 11 shows the cumulative probability for the simulated peak tritium dose via groundwater, which is calculated based on the simulated aquifer concentrations and a conservative water intake of 10 L/day (accounts for drinking water, indirect ingestion via plants and animals, absorption and inhalation via showering, etc.). The results shows that all realizations are well below the EPA metric of 4 mrem/year.

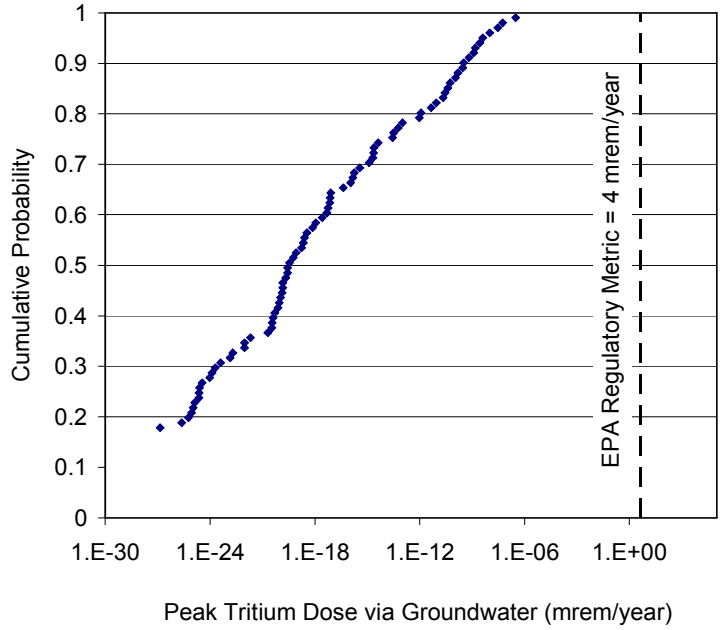

Figure 11. Cumulative probability for simulated peak tritium dose via the groundwater pathway using 100 realizations.

Figure 12 shows the cumulative probability for the simulated peak tritium dose via the air pathway for 100 realizations. The simulated dose due to inhalation (and skin absorption) is based on the concentration of gas-phase tritium immediately above the MWL. The average wind velocity, vertical mixing length, and surface flux of tritium are used to calculate the air concentration above the MWL, and the inhalation rate is used to calculate the intake (Table 5). The dose conversion factor (Table 1) is then used to calculate the dose rate. Because the simulated surface flux of tritium for several realizations was quite high (Figure 5), a small percentage (~2%) of the realizations yield a dose via the air pathway that exceeds the EPA metric of 10 mrem/year.

It should be noted, however, that Figure 5 shows the peak tritium surface fluxes occurring before 50 years due to the natural decay of tritium. The simulated maximum surface concentrations of tritium that yielded the peak fluxes are on the order of 10^{10} pCi/L. If measured values of tritium vapor concentrations at the surface over the next few decades are not shown to increase from previously measured values, which are several orders of magnitude less than maximum simulated values, the dose due to tritium via the air pathway is not likely to be exceeded.

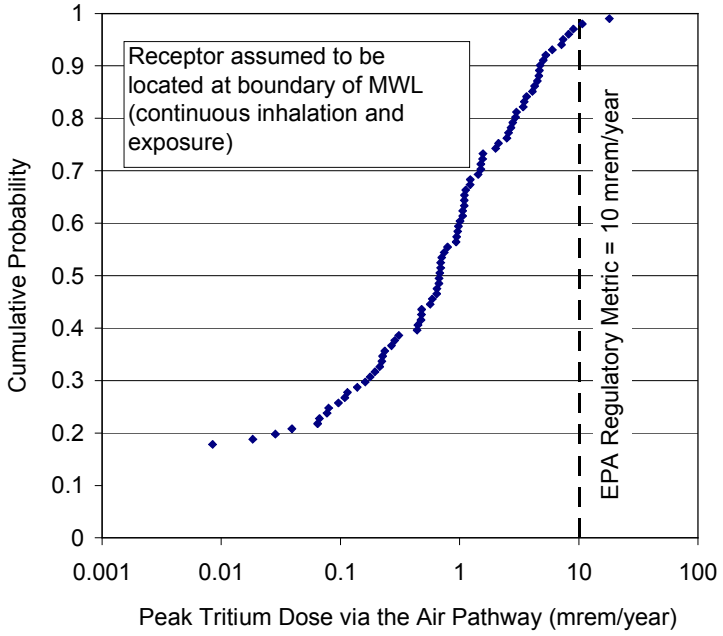

Figure 12. Cumulative probability for simulated peak tritium dose via the air pathway for 100 realizations.

3.5.2.3 Sensitivity Analysis

A sensitivity analysis (as described in Section 2.2.1) was performed to determine the parameters that were most important to the simulated performance metrics of aquifer concentration and inhalation dose. Figure 13 presents a chart that summarizes the results of the stepwise linear rank regression analysis. All of the uncertain input variables summarized in Table 2 through Table 5 relevant to tritium transport were evaluated, but only the most important input variables are shown in Figure 13. The ΔR^2 values in Figure 13 provide a measure of the incremental contributions from each input variable to the variability in the simulated performance metric. For example, the uncertainty in the liquid-phase tortuosity accounts for about 60% of the variability in the simulated tritium aquifer concentration

The sensitivity of the inhalation dose to liquid-phase tortuosity and moisture content indicates that the transport of tritium is dependent on upward diffusion through the liquid phase as well as the gas phase. A conservative upper bound for the liquid- and gas-phase tortuosity coefficients was implemented in this study (Table 4) to account for the possible effects of enhanced vapor diffusion (Ho and Webb, 1998). The dependence on cover thickness and atmospheric boundary-layer thickness indicates that the inhalation dose is also dependent on the upper boundary conditions of the landfill. Therefore, the thickness and integrity of the cover should be monitored and maintained to mitigate tritium migration to the surface. Finally, although not included as an uncertain parameter, the location and disposition of the receptor played an

important role in the simulated inhalation dose. In this study, the receptor was assumed to be located adjacent to the MWL, continuously inhaling air directly above the MWL (24 hours a day, 365 days a year). If the receptor were located further away from the site, or if the exposure were not continuous, the simulated dose via the air pathway would be considerably less.

The variability of the tritium aquifer concentration is shown to be dependent on the liquid-phase mobility parameters, indicating that diffusion of liquid-phase tritium is important. A separate ("one-off") sensitivity analysis of infiltration revealed that the infiltration would have to be increased by several orders of magnitude (close to the saturated hydraulic conductivity of the vadose zone) in order for the tritium to reach substantial concentrations in the groundwater.

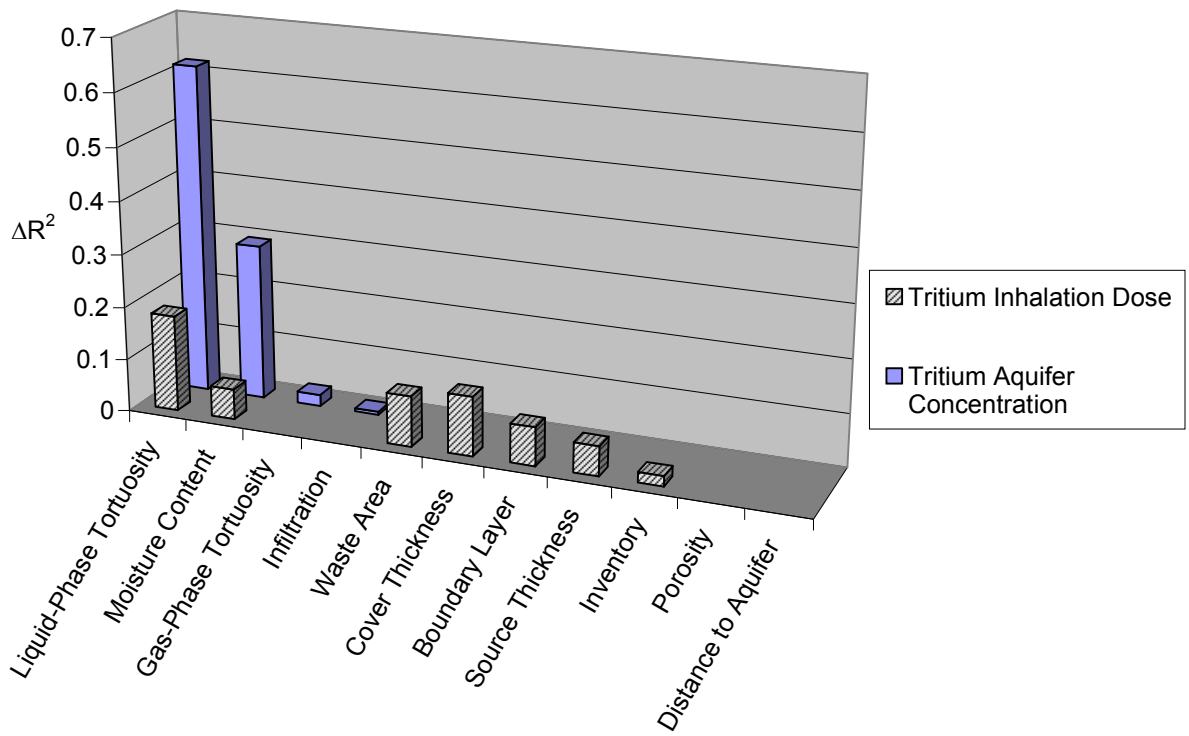

Figure 13. Analysis of sensitivity of simulated tritium inhalation dose and aquifer concentration to uncertain input parameters.

3.5.3 Summary of Key Results and Assumptions

- All simulated realizations of tritium aquifer concentration and dose via the groundwater pathway were well below the regulatory metrics of 20,000 pCi/L and 4 mrem/year, respectively.

- A small percentage (2%) of the simulated dose due to tritium via the air pathway exceeded the regulatory metric of 10 mrem/year.

- Parameters impacting tritium diffusion through both the liquid and gas phases (e.g., tortuosity coefficient, moisture content, cover thickness, atmospheric boundary-layer thickness) were found to be important to the simulated inhalation dose.

- Key Assumptions:

    - Receptor located at MWL; continuous inhalation and exposure of tritium flux from subsurface
    - Cover allowed to erode completely
    - 1-D model: maximum transport to surface
    - Bounding tortuosity coefficients: maximum diffusion rate
    - Maximum waste inventory set equal to twice estimated value of 2,400 Ci

3.6 Fate and Transport of Radon

3.6.1 Model Description

Section 3.2.3 and Appendix A describe the steady-state radon transport model that was developed for this study. Diffusion, advection, and decay of radon is included in the model. A constant generation of radon is assumed to occur in the prescribed waste zone, which can vary in size. A significant difference between the current model and previous models of radon transport in geological media (see, for example, Rogers et al., 1984) is the nature of the radium-226 source. In previous studies, the radium-226 originated from ore deposits containing uranium. At the MWL, pure radium-226 was disposed of in sealed containers. Therefore, the overall concentration of radium-226 can be much higher in the current analysis, but the emanation factor, *E*, which governs how much radon-222 gas can be released from the radium-226, can be significantly lower because of the containment. Generally speaking, the integrity of radioactive sealed sources is very robust. The radium-226 sealed sources disposed of in the MWL were most likely fabricated according to design standards that required tests to evaluate the integrity of the sources subject to extreme temperature, impact, pressure, and vibration (see, for example, 10 CFR 39.41). Radon-222 originating from uranium-238 was not considered in the radon-transport model because the activity of radium-226 (parent of radon-222) resulting from the decay of uranium-238 is negligible (15 microCuries after the first 1,000 years) relative to the radium-226 activity assumed in the model (6-12 Curies). However, radon-222 was included as a decay product of uranium-238 in the FRAMES/MEPAS liquid-phase transport simulations of the radionuclides (see Section 3.7.2.2).

3.6.2 Model Results

3.6.2.1 Comparison to Field Data

Radon surface fluxes at the MWL were measured in 1997 (Haaker, 1998). A total of 89 four-inch-diameter activated charcoal radon canisters were used to evaluate the radon surface fluxes in the vicinity of the MWL, as well as background values. Results showed that the measured radon fluxes above the MWL were not significantly different than the background values. The median flux in the vicinity of the MWL was 0.33 pCi/m^2/s while the median background flux was 0.35 pCi/m^2/s. The maximum measured fluxes for the MWL and background were 1.02 and 0.664 pCi/m^2/s, respectively. This difference in maximum values was used to calibrate the emanation factor in the radon transport model. The emanation factor governs how much radon is released to the immediate surroundings from the radium-226 source. A factor of zero represents no emission (complete containment), and a factor of one represents total emission (no containment).

The potential sources of radon-222 (radium-226) were sealed and contained, and the sealed sources were likely tested for integrity before disposal in the MWL. Therefore, the containment is assumed to be generally intact at present, but defects or breaks may still be present. The minimum emanation factor, which accounts for present-day emissions, was adjusted to yield a radon flux between 0.1 and 1 pCi/m^2/s (equivalent to the difference in maximum measured and background fluxes). The resulting minimum emanation factor used in the probabilistic simulations was 10^{-6}. The maximum emanation factor was estimated based on the possibility that the sealed containers may degrade in the future. The integrity of the containers is expected to last well beyond 1,000 years, but an upper value of the emanation factor was set equal to 0.01 to represent the possibility that 1% of the containers will completely degrade within 1,000 years. An evaluation was also performed assuming that the maximum emanation factor was equal to one, which is equivalent to complete degradation of the containment of all the radon sources within 1,000 years. A log-uniform distribution between 10^{-6} and the maximum value was used for the emanation factor.

3.6.2.2 Comparison to Performance Objectives

Figure 14 shows the cumulative probability for the simulated peak radon-222 surface flux for 100 realizations. For the scenario with a maximum emanation factor of 0.01 (1% of the radon-source containers degrades completely), the results show that 97% of the simulated radon surface fluxes are below the design standard of 20 pCi/m^2/s (3% of the realizations yield radon surface fluxes that exceed the design standard). In the bounding scenario, where we allow all of the containment of the sealed sources to completely degrade, nearly 30% of the realizations exceed the design standard of 20 pCi/m^2/s. As shown in the sensitivity analysis in the next section, the large uncertainty in the emanation factor allowed significant variations in the simulated radon surface flux. It is unlikely that the sealed sources and containers for radium-226 will degrade significantly over the next few hundred years, but because the half-life of radium-226 and uranium-238 is extremely long, radon-222 will continue to be generated from these parent products indefinitely. Therefore, degradation of the containers may eventually cause the emanation factor for radon-222 to increase at some point in the future. For a 1,000-year

evaluation period, however, the probability of exceeding the radon surface-flux design standard is very small if the sealed sources and containers do not degrade significantly and the emanation factor remains below 0.01.

Simulated radon concentrations in groundwater were negligible (<10^{-20} pCi/L). The short half-life of radon (3.8 days) and the large thickness of the vadose zone prohibit radon from migrating significant distances to the water table when the source originates from the landfill. However, in Section 3.7, small amounts of radon are shown to reach the groundwater after 10,000 years when radon is included as progeny of uranium-238, which is fairly mobile (relative to the other non-volatile radionuclides). This effectively mobilizes the source of radon toward the groundwater. However, the decay chain for uranium-238 to radium-226 to radon-222 is an extremely long process (billions of years). Therefore, the amount of radon-222 produced from uranium-238 in 1,000 years is extremely small; no radon-222 is simulated to reach the groundwater in 1,000 years, even when it is included as progeny of uranium-238.

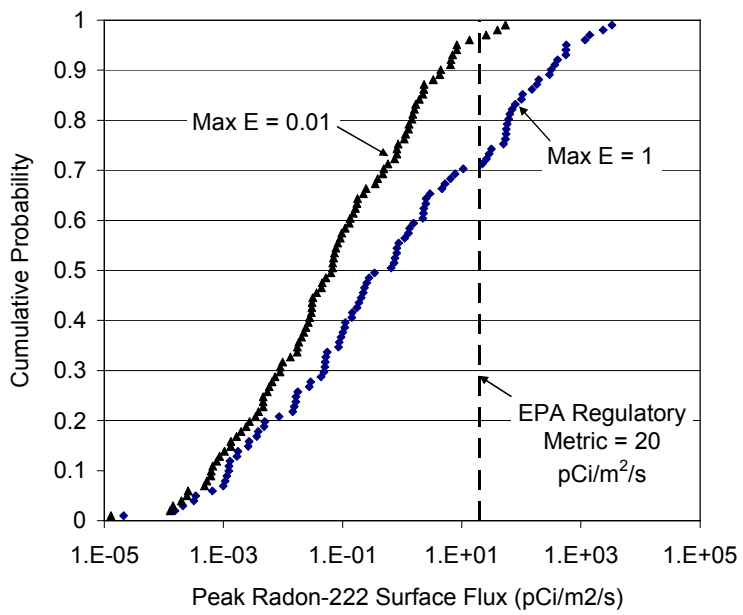

Figure 14. Cumulative probability for simulated peak radon-222 surface flux for 100 realizations using two different maximum values for the emanation factor, E.

3.6.2.3 Sensitivity Analysis

A sensitivity analysis (as described in Section 2.2.1) was performed to determine the stochastic input parameters that were most important to the simulated radon surface flux. Figure 15 presents a chart that summarizes the results of the stepwise linear rank regression analysis. The emanation factor was by far the most significant variable that influenced the variability in the simulated radon surface flux. The waste volume, cover thickness, and effective diffusion coefficient were also shown to be statistically correlated to the simulated radon surface flux, but to a much lower degree.

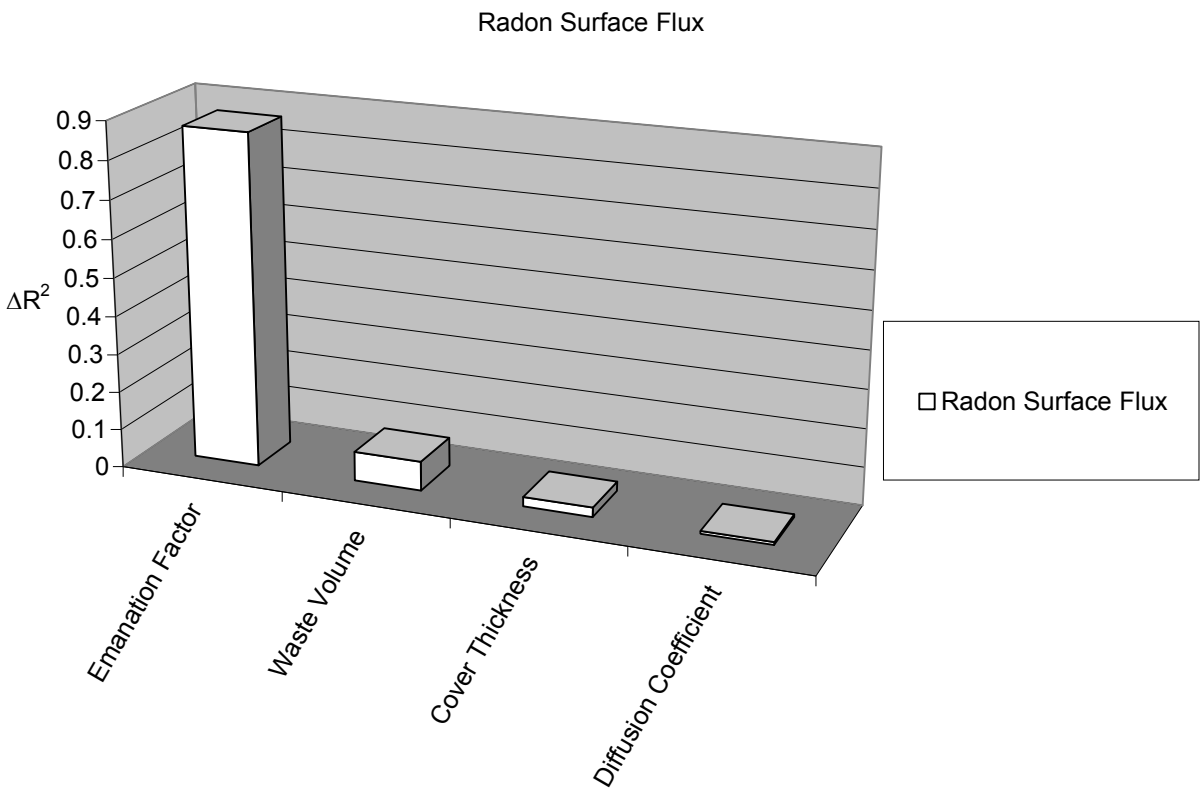

Figure 15. Analysis of sensitivity of simulated radon surface flux to uncertain input parameters.

3.6.3 Summary of Key Results and Assumptions

- Sensitivity studies show that the emanation factor, which depends on the integrity of the radium-226 containment, is important to the performance of the landfill with regard to surface radon fluxes.

- For a maximum radon emanation factor of 0.01 (1% of the radium-226 containers fail), the simulated radon surface fluxes exceed the design standard of 20 pCi/m^2/s in about 3% of the realizations. For a maximum radon emanation factor of 1 (100% of the radium-226 containers fail), the simulated radon surface fluxes exceed the design standard in about 30% of the realizations.

- Simulated radon concentrations in the groundwater were negligible.

- Key Assumptions:

    - Sealed sources of radium-226 allowed to degrade in 1,000 years (emanation factor allowed to increase)
    - Cover allowed to erode completely
    - 1-D model: maximum transport to surface

3.7 Fate and Transport of Other Radionuclides

3.7.1 Model Description

The FRAMES/MEPAS source-term and vadose-zone models (see Section 3.2.1) were used to evaluate the aqueous-phase transport of the following radionuclides to the groundwater: amercium-241, cesium-137, cobalt-60, plutonium-238, plutonium-239, radium-226, strontium-90, thorium-232, tritium, and uranium-238. Although tritium was simulated separately using the model of Jury et al. (1983, 1990), it was also included in the FRAMES/MEPAS model. Decay products of plutonium-238 (e.g., uranium-234), radium-226 (e.g., radon-222), and uranium-238 (e.g., uranium-234, radium-226) are also simulated in the FRAMES/MEPAS model (see Whelan et al., 1996).

3.7.2 Model Results

3.7.2.1 Comparison to Field Data

Other than the detection of tritium and radon in the atmosphere and subsurface as discussed in previous sections, no other radionuclides have been detected at the surface or in the subsurface beyond the extent of the landfill. The inventory for each of the radionuclides shown in Table 2 was estimated based on past records regarding the content of the MWL (SNL, 1993). The upper value for the inventory distribution of each radionuclide was conservatively assumed to be equal to twice the estimated value from past records.

3.7.2.2 Comparison to Performance Objectives

In all realizations, none of the radionuclides were simulated to reach the groundwater in 1,000 years.[4] All of the radionuclides were retarded sufficiently by adsorption to prevent significant migration in 1,000 years, even with the realistically conservative distributions used for model inputs (Table 2). In order to assess potential failure mechanisms, additional scenarios were performed.

Alternative Scenario: Increased Infiltration

First, the infiltration was increased while holding all other input parameters at fixed, conservative values. After 1,000 years, uranium (uranium-238, uranium-234) reached the groundwater when the Darcy infiltration through the vadose-zone was increased by an order of magnitude over its maximum stochastic value (6.12×10^{-11} m/s) to 6.12×10^{-10} m/s, but the groundwater concentrations were still less than the regulatory metric of 30 μg/L. Groundwater concentrations of uranium exceeded the regulatory metric when the simulated Darcy infiltration increased by two orders of magnitude over the maximum stochastic value to 6.12×10^{-9} m/s.

[4] Tritium was simulated to reach the groundwater when vapor-phase transport was included in Section 3.5, but simulated tritium groundwater concentrations and dose were well below the regulatory metrics.

Alternative Scenario: Increased Simulation Period

FRAMES/MEPAS was allowed to run past 1,000 years to assess the potential travel times of the different radionuclides to the groundwater using the original distributions and parameter values (Table 2). Only uranium-238 and its decay products (uranium-234, radon-222) were simulated to reach the groundwater after ~10,000 years. The other radionuclides were retarded by their relatively large adsorption coefficients. The radon-222 that reached the groundwater was a decay product of uranium-238. As shown in previous simulations of radon originating from the waste zone (Section 3.6), radon originating from the MWL was not simulated to reach the water table because of its short half-life (3.8 days). However, since uranium-238 has a small distribution coefficient (K_d) and long half-life, a number of realizations showed that uranium-238 and some of its daughter products (uranium-234 and radon-222) could reach the water table after ~10,000 years. Although the decay of uranium-238 to radon-222 is extremely slow, some small but finite amount of radon-222 is generated from uranium-238 as it moves toward the water table. In MEPAS, the Bateman equation (Bateman, 1910) is used to estimate the relative concentrations of the daughter products as a function of the concentration of the parent, the half lives of the parent and daughter products, and the time elapsed.

Figure 16 shows the cumulative probability for simulated peak radon-222 (progeny from uranium-238) aquifer concentrations for 100 realizations after a simulated period greater than 10,000 years. Although the radon-222 reached the water table as a result of the transport of its parent product, uranium-238, the concentration of radon-222 in the groundwater is still well below the proposed limit of 300 pCi/L.

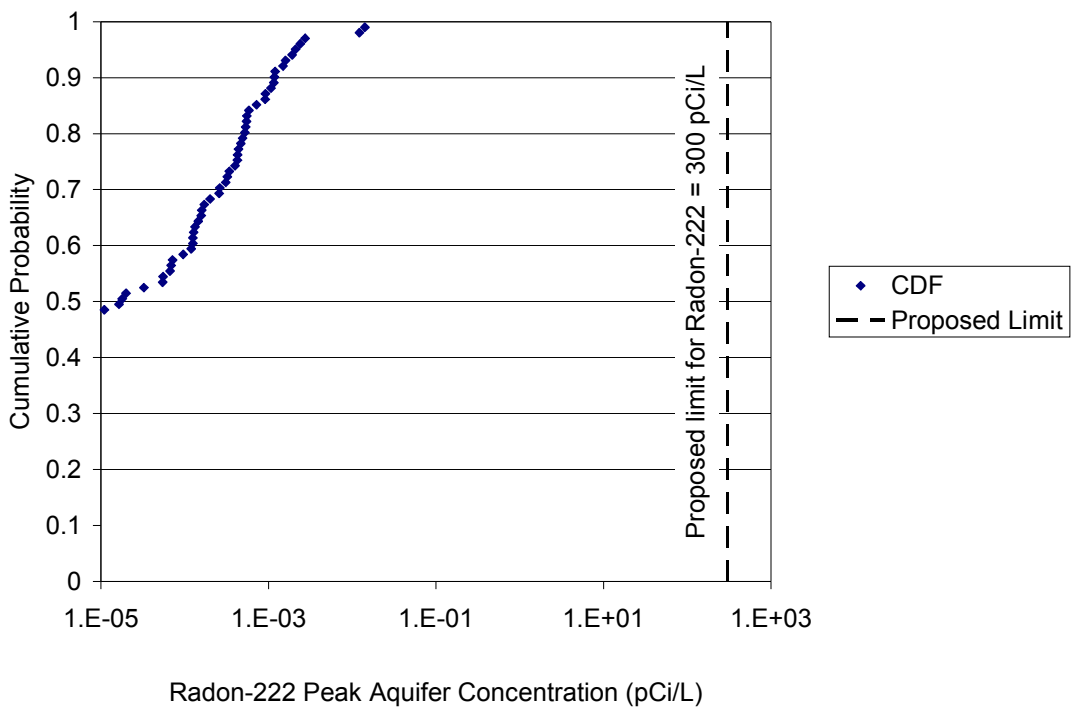

Figure 16. Cumulative probability for simulated peak radon-222 (progeny from U-238) aquifer concentrations for 100 realizations for a time period extending beyond 10,000 years.

Figure 17 shows the cumulative probability for the simulated peak uranium concentration in the groundwater for 100 realizations after a simulated time period greater than 10,000 years. The total uranium concentration is comprised of both uranium-234 (decay product of plutonium-238 and uranium-238) and uranium-238. All realizations yielded peak uranium aquifer concentrations that were less than the EPA regulatory metric of 30 μg/L.

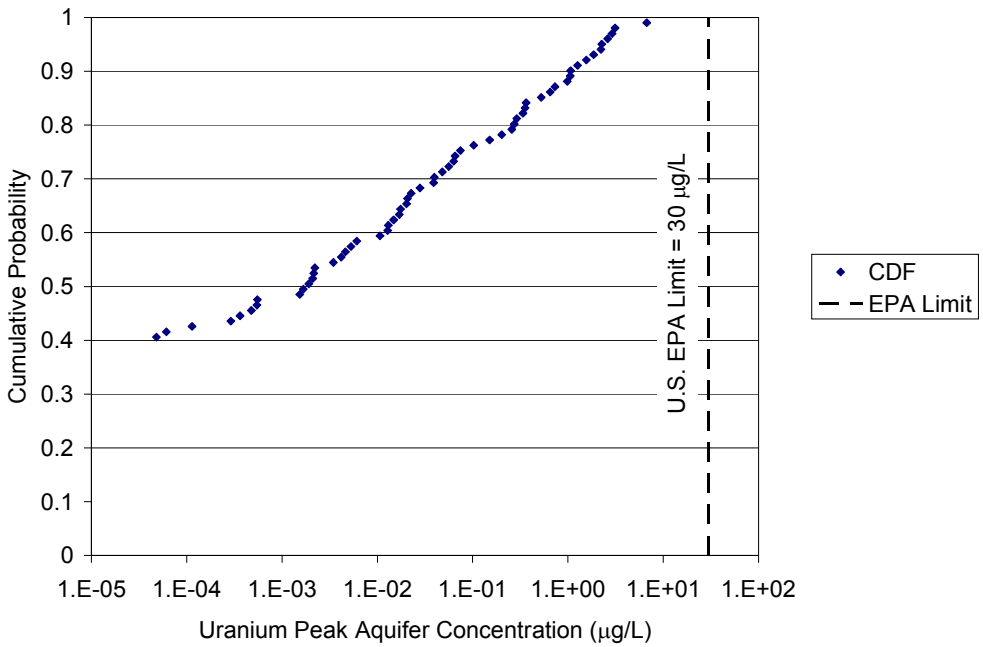

Figure 17. Cumulative probability for simulated peak uranium aquifer concentrations for 100 realizations for a time period extending beyond 10,000 years.

The total groundwater dose for extended periods of time (past 10,000 years) is calculated from the peak aquifer concentrations of uranium (uranium-234 and uranium-238) and radon. The groundwater consumption is assumed to be a conservative 10 L/day to account for drinking water, indirect ingestion through irrigation of vegetables and intake by food-producing animals, and absorption via showering. Figure 18 shows the cumulative probability for the simulated total peak groundwater dose for 100 realizations after a simulated period greater than 10,000 years. The EPA regulatory metric of 4 mrem/year (for beta particles) is shown for reference, but it does not actually apply to the primary constituents contributing to the dose, uranium-234 and uranium-238, which are alpha particles.

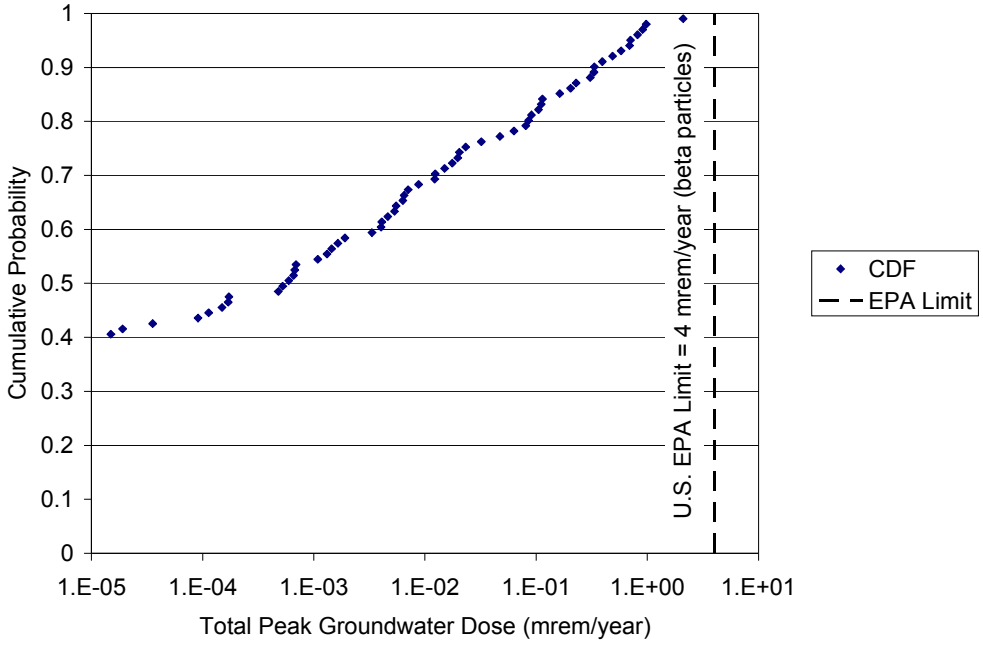

Figure 18. Cumulative probability for simulated peak groundwater dose for all radionuclides for 100 realizations for time periods extending beyond 10,000 years.

3.7.2.3 Sensitivity Analysis

Although no radionuclides were simulated to reach the groundwater within 1,000 years, sensitivity analyses were performed on the extended simulations (>10,000 years) to identify important parameters and processes (Figure 19). Sensitivity analyses show that the infiltration is the primary parameter impacting the variability in the simulated aquifer concentrations for uranium-238, its decay products (uranium-234, radon-222), and the simulated dose via groundwater. A "one-off" sensitivity analysis showed that the infiltration would have to be increased by two orders of magnitude to increase the uranium concentrations above the regulatory metric of 30 μg/L within 1,000 years. Other parameters that were found to be statistically correlated to the variability in the simulated performance metrics were waste length and width, uranium-238 K_d, and the bulk density (which, together with the K_d value, impacts the retardation).

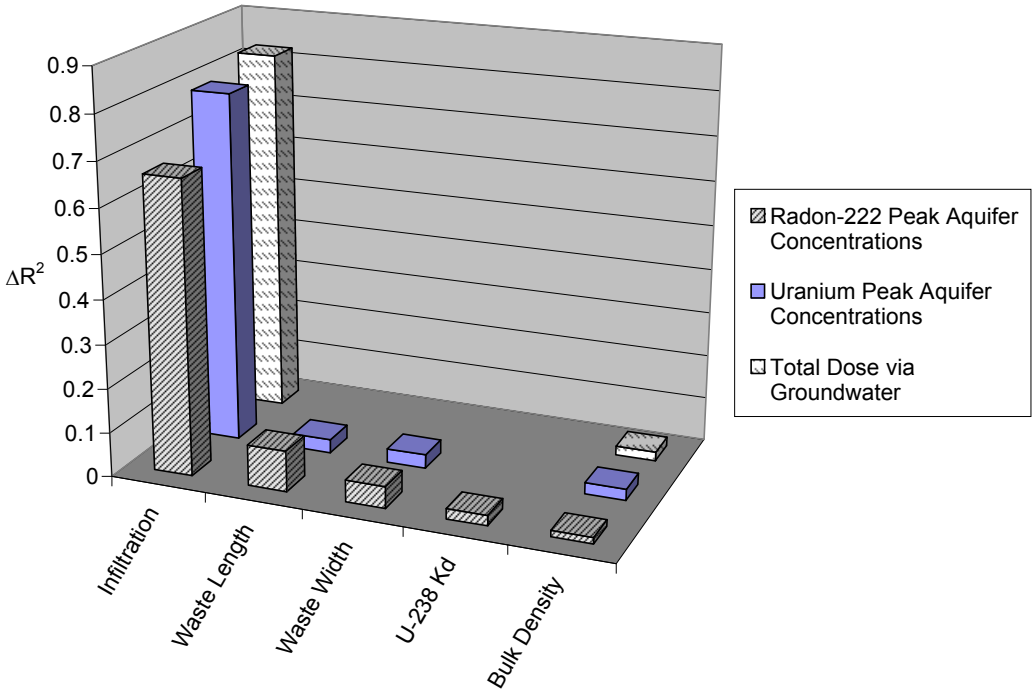

Figure 19. Analysis of sensitivity of simulated peak radon aquifer concentrations, peak uranium aquifer concentrations, and total dose via groundwater to uncertain input parameters for a time period extending beyond 10,000 years.

3.7.3 Summary of Key Results and Assumptions

- None of the radionuclides were simulated to reach the groundwater within 1,000 years for all realizations.

- Only uranium-238 (and some of its decay products) were simulated to reach the water table for extended periods (>10,000 years). All peak aquifer concentrations were still less than the EPA regulatory metric of 30 µg/L.

- Infiltration rate was found to be the most significant parameter impacting the variability in the simulated groundwater concentrations and dose via groundwater. Uranium groundwater concentrations were simulated to exceed the regulatory metric of 30 µg/L if the infiltration increased two orders of magnitude above the maximum stochastic value to 6.12×10^{-9} m/s.

- Key Assumptions:
    - 1-D model: maximum transport to groundwater
    - Receptor assumed to be located at MWL. Water intake assumed to be 10 L/d (5 times greater than EPA standards)

3.8 Fate and Transport of Heavy Metals

3.8.1 Model Description

The fate and transport of two heavy metals, lead and cadmium, were simulated using FRAMES/MEPAS (see Section 3.2.1). The inventory of lead was estimated from previous records (SNL, 1993), and uncertainty in the inventory was captured by using a uniform distribution with the estimated value as a lower bound (see Table 2). There were no records of cadmium being disposed of at the MWL, but soil samples revealed concentrations of cadmium in the subsurface (Peace et al., 2002). The maximum soil concentrations of cadmium were used with the bulk density of the soil and maximum simulated penetration of coolant water (Wolford, 1997) to estimate the mass of cadmium in the MWL. This value was then used as a lower bound in a uniform distribution (see Table 3).

3.8.2 Model Results

Neither lead nor cadmium were simulated to reach the groundwater in all 100 realizations for 1,000 years. Extended simulation periods (>10,000 years) also did not yield any breakthrough of lead or cadmium to the water table. Therefore, comparisons to the regulatory metrics of 15 μg/L and 5 μg/L for lead and cadmium, respectively, are not plotted. Both lead and cadmium have relatively large adsorption coefficients (see Table 2), which retard their transport through the thick vadose zone.

3.8.2.1 Sensitivity Analysis

A "one-off" sensitivity analysis was performed to determine the impact of infiltration on the transport of lead and cadmium while holding all other parameters at constant conservative values. Results showed that cadmium could reach the groundwater in 1,000 years and exceed its regulatory metric if the Darcy infiltration were increased by three orders of magnitude over the maximum expected infiltration, which is based on future climate scenarios (i.e., from $6x10^{-11}$ m/s to $6x10^{-8}$ m/s). Lead was simulated to reach the water table in 1,000 years if the infiltration were increased by four orders of magnitude over the maximum expected infiltration. Although this additional increase in infiltration is not expected to occur based on detailed infiltration simulations (see Section 3.4), the infiltration at the MWL should be monitored in the future. Significant increases (by several orders of magnitude or more) may lead to increased potential for migration of heavy metals and other contaminants to the groundwater.

3.8.3 Summary of Key Results and Assumptions

- Neither lead nor cadmium were simulated to reach the groundwater in 1,000 years (or extended periods past 10,000 years)
- Additional increases in infiltration would (3-4 orders of magnitude over expected maximum infiltration rates) allow cadmium and lead to reach the groundwater in 1,000 years.
- Key Assumptions:
    - 1-D model: maximum transport to groundwater

3.9 Fate and Transport of Volatile Organic Compounds

3.9.1 Model Description

Volatile organic compounds (VOCs) were used as cleaners and solvents for machining and other industrial processes at Sandia National Laboratories. Rags, residual containers, and other wastes contaminated with these contaminants were disposed of at the MWL. Although no quantitative estimates of the volumes of these contaminants disposed of in the MWL exists, soil samples provide an estimate of the extent and concentration of the region contaminated with VOCs at the MWL. Previous studies have shown that VOCs such as trichloroethylene (TCE) and tetrachloroethylene (PCE) can migrate long distances in the vapor phase. Klavetter (1995a) showed that among the VOCs of concern at the MWL, PCE was the only VOC that posed a threat to exceeding regulatory metrics in the groundwater (PCE has a greater Henry's constant and, hence, greater gas-phase transport rate than TCE for the same aqueous source concentration). However, because there is still a potential for other VOCs from the MWL to migrate to groundwater due to their mobility, PCE was modeled in this study as a proxy for other VOCs detected in soil gas and in soils beneath the MWL.

In this study, PCE is simulated using the transient model of Jury et al. (1983, 1990), which accounts for aqueous-phase advection, gas-phase diffusion, adsorption, and decay (see Section 3.2.2). Table 2 summarizes the uncertainty distributions that were used in the model. The inventory was calculated based on the maximum measured soil gas concentration (5,900 ppb) at 30 feet (Peace et al., 2002). We assumed that the PCE vapor was in equilibrium with its aqueous phase (using Henry's constant). The maximum measured gas concentration (5,900 ppb) was used as a minimum value in a uniform distribution increasing to ten times this value to develop a range of equilibrium aqueous concentrations. The maximum value was based on calibrations with measured data (see next section). The total mass of PCE was then calculated using the moisture content, maximum areal extent of the MWL (430 feet x 300 feet), and an uncertain thickness ranging from 10-27 feet. Other values in Table 2 were taken from conservative values and ranges found in the literature for PCE.

3.9.2 Model Results

3.9.2.1 Comparison to Field Data

Samples of PCE soil-gas concentrations were taken at the MWL in 1993 (Johnson et al., 1995). The ranges of measured values at two different depths (10 feet and 30 feet) were compared to simulated soil-gas concentrations using the transient PCE transport model described in the previous section. Figure 20 and Figure 21 show the comparisons for all 100 simulated realizations. As discussed in previous sections, the measured values in 1993 are shown spanning a time period between 5 and 33 years, which accounts for the uncertainty in the time of emplacement. Results show the majority of simulated soil-gas concentrations during this time period at the two depths are between the maximum and minimum values measured in 1993.

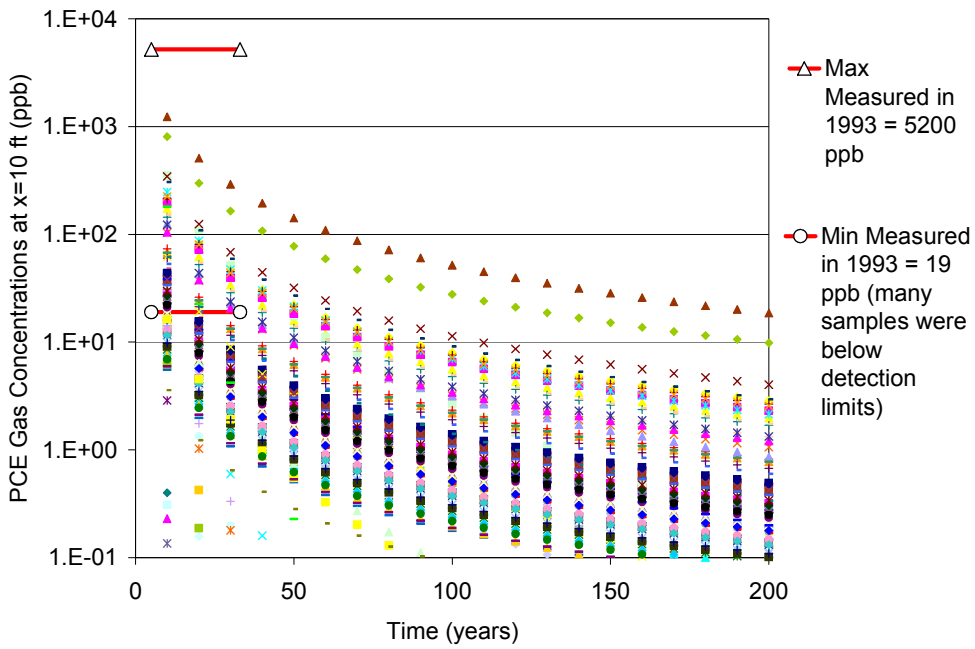

Figure 20. Simulated PCE gas concentration at a depth of 10 feet as a function of time for 100 realizations with a range of measured values in 1993.

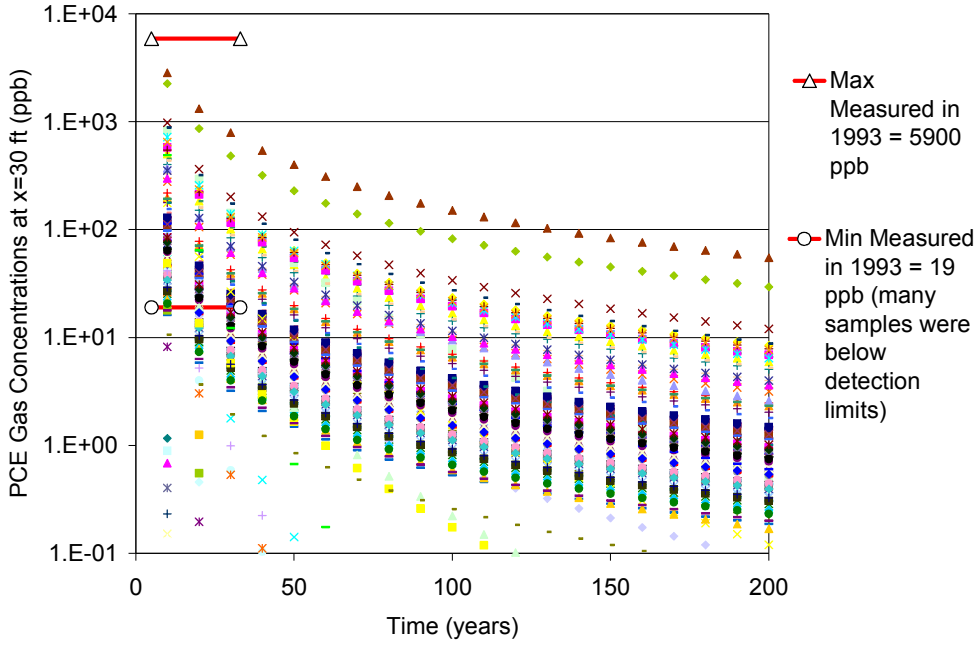

Figure 21. Simulated PCE gas concentration at a depth of 30 feet as a function of time for 100 realizations with a range of measured values in 1993.

3.9.2.2 Comparison to Performance Objectives

Figure 22 shows the simulated PCE concentrations in the groundwater as a function of time for all 100 realizations. The majority of the realizations show the aquifer concentrations peaking before 50 years. Depending on the time of disposal, this corresponds to peak concentrations occurring by 2010 – 2040. So far, no detectable amounts of PCE have been found in the groundwater at the MWL. This is still consistent with the simulations, which show a large amount of variability in the simulated concentrations resulting from uncertainty included in the input parameters (see next section).

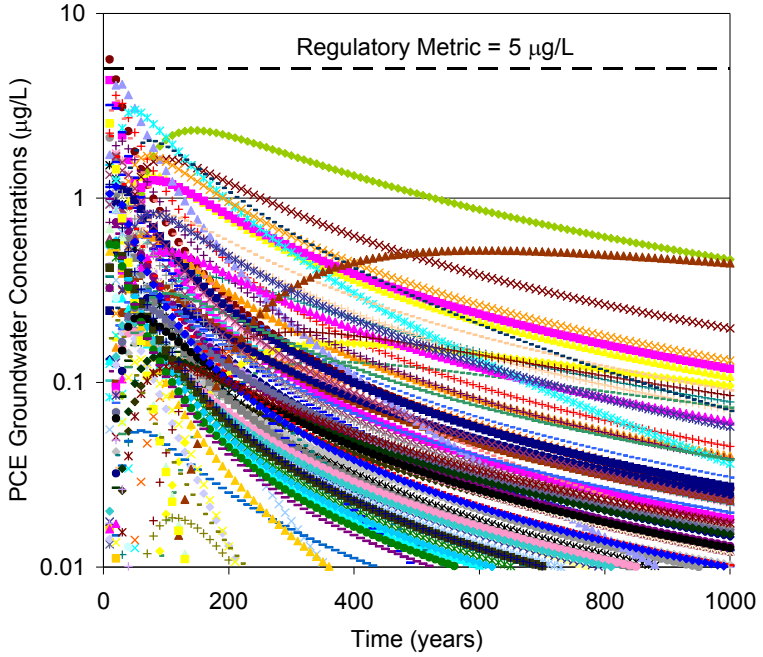

Figure 22. Simulated PCE groundwater concentrations for 100 realizations.

The cumulative probability of the peak PCE groundwater concentration for all 100 realizations is shown in Figure 23. The results show that approximately 99% of the realizations yield groundwater concentrations less than the regulatory metric of 5 μg/L. Only 1% of the realizations yielded groundwater concentrations that exceeded the regulatory metric.

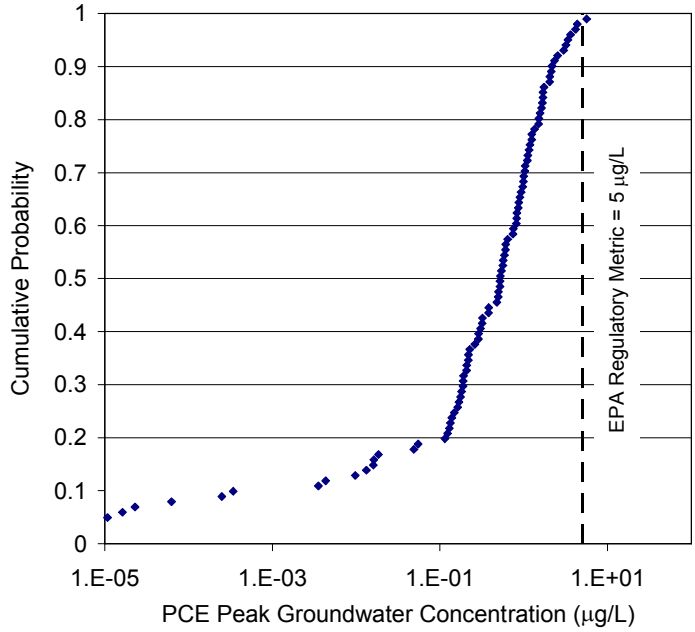

Figure 23. Cumulative probability for simulated PCE peak groundwater concentrations for 100 realizations.

3.9.2.3 Sensitivity Analysis

The uncertainty in the PCE K_d, half-life (degradation), inventory concentration, source thickness, and cover thickness values were found to be the most statistically significant parameters that impacted the variability in the simulated PCE aquifer concentrations. As stated in previous sections, the adsorption coefficient, K_d, plays an important role in the retardation and mobility of the constituent. The half-life and inventory both govern the persistence and availability of the PCE during migration to the groundwater. The source thickness also contributes to the overall inventory of PCE since the inventory concentration is applied to the entire source volume.

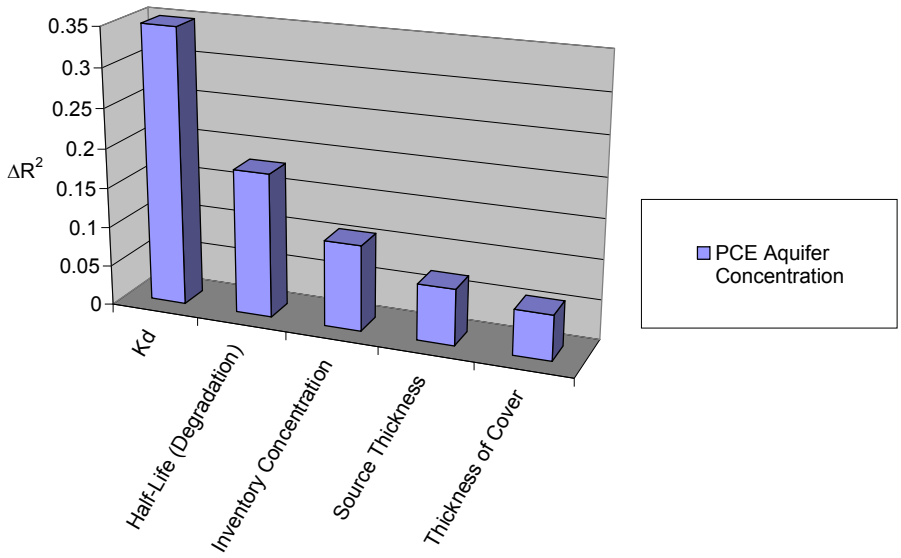

Figure 24. Analysis of sensitivity of simulated PCE peak aquifer concentrations to uncertain input parameters.

3.9.3 Summary of Key Results and Assumptions

- 99% of the realizations yielded peak PCE concentrations in the groundwater that were less than the regulatory metric of 5 µg/L. The majority of the realizations showed that the peak PCE groundwater concentration occurred within 100 years.
- Uncertainty in the PCE adsorption coefficient, half-life, inventory concentration, source thickness, and cover thickness were found to be significantly correlated to the simulated groundwater concentrations.
- Key Assumptions:
    - 1-D model: maximum transport to groundwater

4. Recommended Triggers for Long-Term Monitoring

The NMED's Class 3 permit modification (NMED, May 2005) requires that the MWL CMI Plan include triggers for future action that identify and detail specific monitoring results that will require additional testing or the implementation of an additional or different remedy. Based on the results of the probabilistic performance-assessment modeling for the MWL, the following parameters were identified as important for meeting the performance metrics:

- Surface emissions of tritium and radon
- Infiltration through the MWL cover
- Concentrations of uranium in groundwater
- Concentrations of VOCs in the vadose zone and in groundwater

Monitoring triggers are proposed for these parameters to ensure that the MWL performance metrics and corrective action objectives are met. The proposed triggers are based on EPA and DOE regulatory standards, and are discussed in Section 4.2. To address concerns regarding potential mobilization of contaminants by biota, additional monitoring triggers are proposed for metals and radionuclides in surface soil near animal burrows and ant nests.

A trigger evaluation process is proposed in Section 4.1. This process will be initiated if a trigger is exceeded during long-term monitoring at the MWL. The logic and rationale behind specific triggers are presented in Section 4.2.

Additional details regarding long-term monitoring at the MWL will be presented in the MWL Long Term Monitoring and Maintenance Plan. This plan will be submitted within 180 days after the NMED's approval of the MWL CMI Report. The plan will include all necessary physical and institutional controls to be implemented in the future, and will also include contingency procedures to be implemented if the MWL remedy fails to be protective of human health and the environment.

4.1 Trigger Evaluation Process

A trigger evaluation process is recommended for the MWL during long-term monitoring activities at the site. The process will be a phased approach designed to ensure the protection of human health and the environment, while allowing adequate data collection to evaluate whether corrective action is warranted. This process is based upon the "Conceptual Corrective Measure Evaluation Process' proposed in the Post-Closure Care Plan for the Chemical Waste Landfill (SNL, September 2005).

In the event that a trigger level is exceeded, the process shown in Figure 25 will be used to ensure that adequate data are collected to determine whether additional corrective action is warranted. The increased frequency of data collection proposed in the trigger evaluation process (see Step 3 in Figure 25 and the corresponding explanation on the reverse side of the figure) will ensure that adequate data are collected to eliminate field sampling error, laboratory error, or short-term exceedances that do not reflect long-term trends. Thus, any recommendations for corrective action because of trigger exceedances will be based upon data trends rather than upon single detection values above the trigger level. If data trends in the monitored parameters indicate an established trend above the proposed trigger value, the process requires that a technical letter report be submitted to the NMED recommending whether or not corrective action should be implemented.

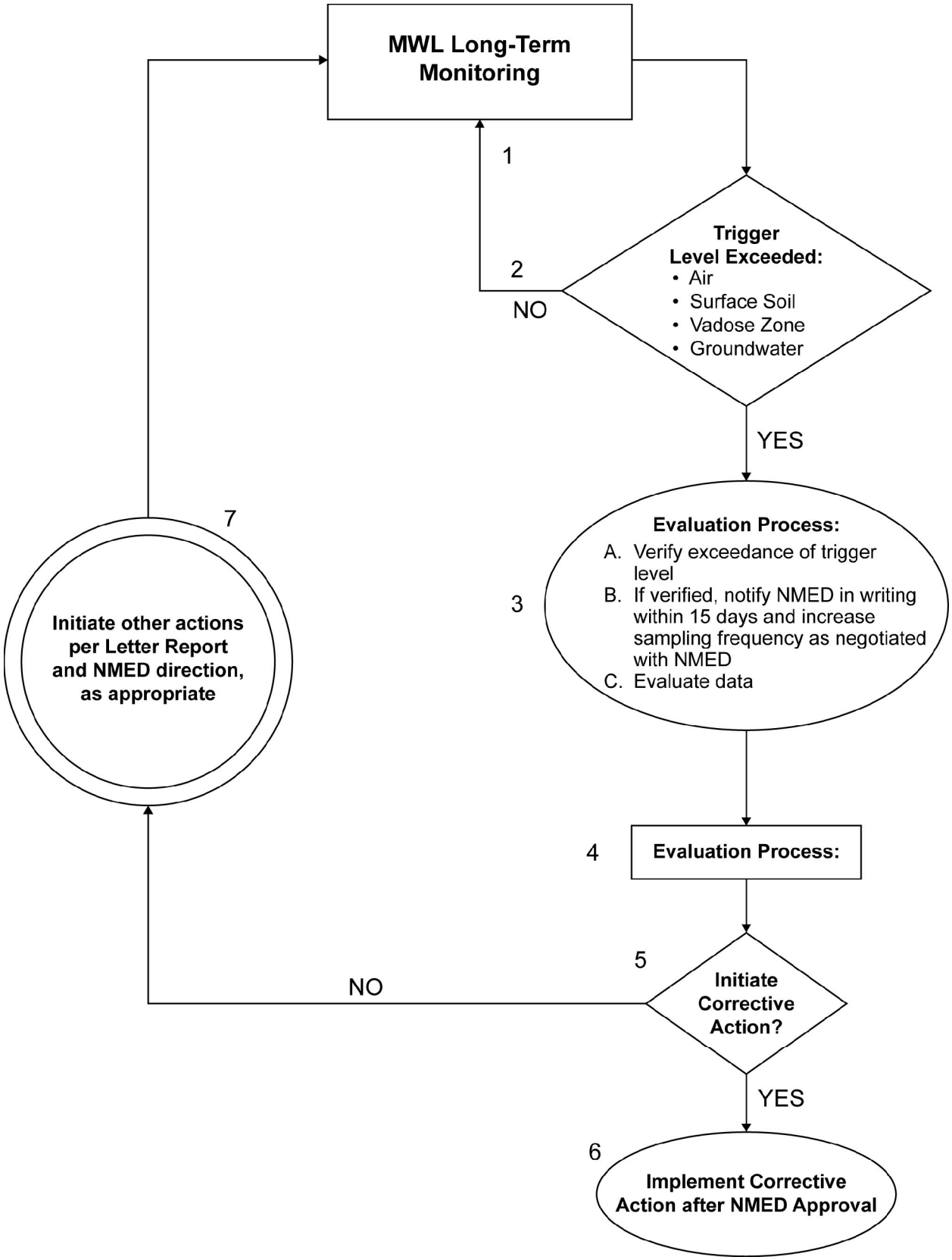

Figure 25. Trigger evaluation process for the Mixed Waste Landfill.

The steps outlined in Figure 25 are explained below:

1. Long-term monitoring of the air, surface soil, vadose zone, and groundwater at the MWL.

2. Exceedance of one or more trigger levels initiates the specific actions described below.

3. Step A of the evaluation process initiates resampling to verify the result(s) that exceeded the trigger level. Step B is based upon the conceptual model for the MWL. Because infiltration through the MWL cover is expected to be very low, and contaminant transport times in the vadose zone and groundwater are anticipated to be relatively slow, a longer period for data collection at an increased sampling frequency is recommended to determine trends. The length of this period and the increased sampling frequency will be negotiated with the NMED. Once the increased sampling data have been collected, the data and any resulting trends will be evaluated to determine the significance of the exceedance (Step C).

4. After the resulting trends have been evaluated, a brief technical letter report will be prepared and submitted to the NMED within three months of receiving the final data set that summarizes the trigger exceedance(s), presents the results of the increased monitoring, and provides recommendations regarding corrective action.

5. NMED Decision Point: after the technical letter report is submitted to the NMED, a meeting will be held to discuss the data evaluation and the recommendations regarding corrective action. If the NMED determines that further investigation of the trigger exceedance is needed, NMED may require corrective action based on a finding that releases of contaminants have occurred, are occurring, or are likely to occur.

6. If the data trend is increasing and higher than the proposed trigger value, corrective action may be necessary. The technical letter report will address appropriate options and form the basis for further discussion with NMED to determine the final corrective action.

7. If the data trend is not clear or is decreasing, corrective action may not be necessary, but other actions may be required as proposed in the technical letter report or requested by the NMED.

4.2 Proposed Triggers

Based on the results of the probabilistic performance-assessment modeling conducted for the MWL, and on subsequent input received from the NMED, monitoring triggers are proposed for the air, surface soil, vadose zone, and groundwater at the MWL These triggers are listed in Table 6, and are discussed below.

Table 6. Proposed Monitoring Triggers for the Mixed Waste Landfill.

Environmental Medium	Monitoring Parameter	Main Potential Receptors	Proposed Trigger Value	Sampling Points	Performance Objective	Applicable Guideline or Regulation
Air	Radon	Humans	4 pCi/L (measured by Track-Etch radon detectors)	MWL Perimeter	Average flux of radon-222 gas shall be less than 20 pCi/m^2/s at the landfill surface (design standard)	EPA Action Threshold for radon in air (U.S. EPA 2005)
Surface Soil	Tritium	Humans and ecological receptors	20,000 pCi/L tritium in soil moisture	MWL Perimeter	Dose to the public via the air pathway shall be less than 10 mrem/yr	DOE Order 5400.5, 10 CFR 61 Subpart H, 40 CFR 141.66
Surface Soil	Cs-137	Humans and ecological receptors	0.664 pCi/g	Animal burrows & ant nests on the cover	Radionuclide concentrations in soil shall not exceed NMED-Approved Maximum Background Concentrations	NMED-Approved Maximum Background Concentrations (Dinwiddie 1997)
Surface Soil	Ra-226	Humans and ecological receptors	2.30 pCi/g	Animal burrows & ant nests on the cover	Radionuclide concentrations in soil shall not exceed NMED-Approved Maximum Background Concentrations	NMED-Approved Maximum Background Concentrations (Dinwiddie 1997)
Surface Soil	Th-232	Humans and ecological receptors	1.01 pCi/g	Animal burrows & ant nests on the cover	Radionuclide concentrations in soil shall not exceed NMED-Approved Maximum Background Concentrations	NMED-Approved Maximum Background Concentrations (Dinwiddie 1997)
Surface Soil	U-235	Humans and ecological receptors	0.16 pCi/g	Animal burrows & ant nests on the cover	Radionuclide concentrations in soil shall not exceed NMED-Approved Maximum Background Concentrations	NMED-Approved Maximum Background Concentrations (Dinwiddie 1997)

Table 6 (continued)

Environmental Medium	Monitoring Parameter	Main Potential Receptors	Proposed Trigger Value	Sampling Points	Performance Objective	Applicable Guideline or Regulation
Surface Soil	U-238	Humans and ecological receptors	1.4 pCi/g	Animal burrows & ant nests on the cover	Radionuclide concentrations in soil shall not exceed NMED-Approved Maximum Background Concentrations	NMED-Approved Maximum Background Concentrations (Dinwiddie 1997)
Surface Soil	Arsenic	Humans and ecological receptors	17.7 mg/kg	Animal burrows & ant nests on the cover	RCRA metal concentrations in soil shall not exceed NMED Industrial/Occupational Soil Screening Levels	NMED Industrial/Occupational Soil Screening Levels (NMED 2006)
Surface Soil	Barium	Humans and ecological receptors	100,000 mg/kg	Animal burrows & ant nests on the cover	RCRA metal concentrations in soil shall not exceed NMED Industrial/Occupational Soil Screening Levels	NMED Industrial/Occupational Soil Screening Levels (NMED 2006)
Surface Soil	Cadmium	Humans and ecological receptors	56.4 mg/kg	Animal burrows & ant nests on the cover	RCRA metal concentrations in soil shall not exceed NMED Industrial/Occupational Soil Screening Levels	NMED Industrial/Occupational Soil Screening Levels (NMED 2006)
Surface Soil	Chromium	Humans and ecological receptors	3400 mg/kg	Animal burrows & ant nests on the cover	RCRA metal concentrations in soil shall not exceed NMED Industrial/Occupational Soil Screening Levels	NMED Industrial/Occupational Soil Screening Levels (NMED 2006)
Surface Soil	Lead	Humans and ecological receptors	800 mg/kg	Animal burrows & ant nests on the cover	RCRA metal concentrations in soil shall not exceed NMED Industrial/Occupational Soil Screening Levels	NMED Industrial/Occupational Soil Screening Levels (NMED 2006)
Surface Soil	Mercury	Humans and ecological receptors	100,000 mg/kg	Animal burrows & ant nests on the cover	RCRA metal concentrations in soil shall not exceed NMED Industrial/Occupational Soil Screening Levels	NMED Industrial/Occupational Soil Screening Levels (NMED 2006)

Table 6 (continued)

Environmental Medium	Monitoring Parameter	Main Potential Receptors	Proposed Trigger Value	Sampling Points	Performance Objective	Applicable Guideline or Regulation
Surface Soil	Selenium	Humans and ecological receptors	5680 mg/kg	Animal burrows & ant nests on the cover	RCRA metal concentrations in soil shall not exceed NMED Industrial/Occupational Soil Screening Levels	NMED Industrial/Occupational Soil Screening Levels (NMED 2006)
Surface Soil	Silver	Humans and ecological receptors	5680 mg/kg	Animal burrows & ant nests on the cover	RCRA metal concentrations in soil shall not exceed NMED Industrial/Occupational Soil Screening Levels	NMED Industrial/Occupational Soil Screening Levels (NMED 2006)
Subsurface Soil	Moisture Content	Humans via groundwater	23 percent by volume	Linear depths of 10 ft to 100 ft along neutron probe access holes beneath the MWL	Infiltration through the cover shall be less than the EPA-prescribed technical equivalence criterion of 31.5 mm/yr [10E-7 cm/s]	RCRA 40 CFR Part 264.301
Subsurface Soil Gas	PCE	Humans via groundwater	20 ppmv	Deepest FLUTe Sampling Port	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Subsurface Soil Gas	TCE	Humans via groundwater	20 ppmv	Deepest FLUTe Sampling Port	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Subsurface Soil Gas	Total Volatile Organic Compounds	Humans via groundwater	25 ppmv	Deepest FLUTe Sampling Port	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	Uranium	Humans via groundwater	15 µg/L	Downgradient monitoring well locations	Uranium concentrations in groundwater shall not exceed the EPA MCL of 30 µg/L	EPA Primary Drinking Water Standard
Groundwater	1,1,1-Trichloroethane (1,1,1-TCA)	Humans via groundwater	100 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	1,1,2-Trichloroethane	Humans via groundwater	2.5 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard

Table 6 (continued)

Environmental Medium	Monitoring Parameter	Main Potential Receptors	Proposed Trigger Value	Sampling Points	Performance Objective	Applicable Guideline or Regulation
Groundwater	1,1-Dichloroethene	Humans via groundwater	3.5 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	1,2-Dichloroethane	Humans via groundwater	2.5 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	1,2-Dichloropropane	Humans via groundwater	2.5 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	Benzene	Humans via groundwater	2.5 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	Carbon tetrachloride	Humans via groundwater	2.5 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	Chlorobenzene	Humans via groundwater	50 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	Ethyl benzene	Humans via groundwater	350 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	Methylene chloride	Humans via groundwater	2.5 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	Styrene	Humans via groundwater	50 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	Tetrachloroethene (PCE)	Humans via groundwater	2.5 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	Toluene	Humans via groundwater	500 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard

Table 6 (continued)

Environmental Medium	Monitoring Parameter	Main Potential Receptors	Proposed Trigger Value	Sampling Points	Performance Objective	Applicable Guideline or Regulation
Groundwater	Trichloroethene (TCE)	Humans via groundwater	2.5 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	Vinyl Chloride	Humans via groundwater	1.0 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	Xylenes (Total)	Humans via groundwater	5,000 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	cis-1,2-Dichloroethene	Humans via groundwater	35 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	Trans-1,2-Dichloroethene	Humans via groundwater	50 µg/L	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA MCLs	EPA Primary Drinking Water Standard
Groundwater	Method 8260 VOCs with no MCLs	Humans via groundwater	EPA Region 6 Human Health Medium-Specific Screening Levels	Downgradient monitoring well locations	VOC concentrations in groundwater shall not exceed EPA Region 6 Human Health Medium-Specific Screening Levels	EPA Region 6 Human Health Medium-Specific Screening Levels

CFR = Code of Federal Regulations.
cm = Centimeter(s).
DOE = U.S. Department of Energy.
EPA = U.S. Environmental Protection Agency.
ft = Foot (feet).
L = Liter(s).
m = Meter(s).
m^2 = Square meter(s).
µg = Microgram(s).
MCL = Maximum contaminant level.
mm = Millimeter(s).
mrem = Millirem.
MWL = Mixed Waste Landfill.
pCi = Picocurie(s).
RCRA = Resource Conservation and Recovery Act.
s = Second(s).
TCA = Trichloroethane.
VOC = Volatile organic compound.
yr = Year(s).

4.2.1 Surface Soil and Air Monitoring Triggers

Proposed surface soil and air monitoring triggers include a trigger for tritium concentrations in soil collected at select locations along the MWL perimeter, and a trigger for radon emissions from the MWL. Triggers are also proposed for radionuclides and metals in surface soil near animal burrows and ant nests to address concerns regarding potential mobilization of contaminants by biota.

4.2.1.1 Tritium

Tritium is the most mobile radionuclide disposed of at the MWL, and the performance-assessment modeling indicates that there is a possibility that tritium emitted from the MWL may exceed the performance objective of 10 mrem/yr dose to the public via the air pathway. For this reason, a trigger is proposed for tritium emitted from the MWL. Figure 12 shows that the simulated peak tritium dose via air exceeded the performance objective in only 2% of the realizations. Figure 6 reveals that the maximum simulated surface concentration of tritium for the realizations that yielded the peak doses via air are on the order of 10^9-10^{10} pCi/L. Therefore, we propose a conservative trigger value of 20,000 pCi/L in surface soils at the MWL perimeter.

The proposed tritium trigger would apply to surface soil samples currently collected annually at select locations along the MWL perimeter by Sandia's Environmental Monitoring group. Soil samples have been collected from these locations and analyzed for tritium on an annual basis since 1985. Soil moisture is extracted from these samples, and tritium concentrations in the soil moisture are determined using liquid scintillation. Any increase in tritium emissions from the MWL would be indicated by elevated tritium concentrations in these soil samples.

Figure 26 shows a comparison between historical tritium concentrations measured in samples from the four perimeter locations, and the proposed trigger value of 20,000 pCi/L. All exceedances of the trigger value occurred prior to 1998, and exceedances are not anticipated in the future due to radioactive decay and the relatively short (12.3 year) half-life of tritium. If measured concentrations of tritium at the surface exceed 20,000 pCi/L, this would indicate a significant increase relative to present-day values, and the trigger evaluation process (Figure 25) would be followed. Because the proposed trigger value is 4-5 orders of magnitude less than simulated concentrations that yielded exceedances in the dose via air, the proposed trigger value serves as a conservative early-warning indicator for potential exceedances of tritium dose via air.

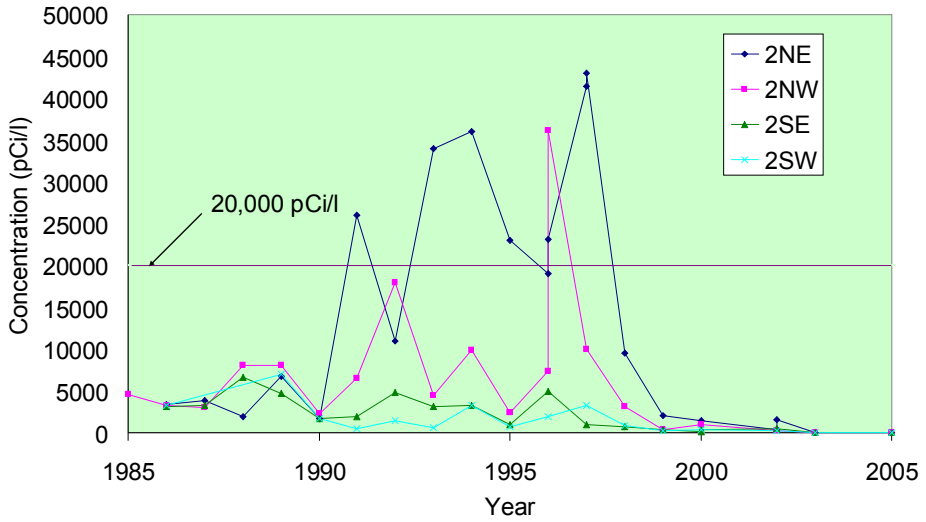

Figure 26. Comparison between historical tritium concentrations measured in samples from the four perimeter locations, and the proposed trigger value of 20,000 pCi/L.

4.2.1.2 Gamma-Emitting Radionuclides and Heavy Metals

NMED has requested that surface soil near animal burrows and ant nests be monitored for radionuclides and heavy metals (NMED, Nov 2006). Triggers proposed for gamma-emitting radionuclides are the NMED-HWB Approved Background Values (Dinwiddie 1997). Triggers proposed for RCRA metals concentrations in surface soil are the NMED Industrial/Occupational Soil Screening Levels (NMED June 2006).

Triggers for gamma-emitting radionuclides and RCRA metals are listed in Table 6. Specific details regarding monitoring frequency and locations will be included in the MWL Long Term Monitoring and Maintenance Plan, to be submitted following completion of the MWL cover.

4.2.1.3 Radon

A trigger for radon is also recommended based on the results of the probabilistic performance-assessment modeling. The modeling indicates that there is a possibility that the radon-222 flux from the MWL to the atmosphere will exceed the design standard of 20 pCi/m^2/s at the landfill surface. Commercially-available Track-Etch radon detectors are recommended to measure the radon concentration in air along the MWL perimeter. These detectors provide an integrated average concentration of radon in air over long exposure periods, on the order of 3 to 6 months. The alternative monitoring detectors, charcoal canisters, are useful only for short exposure periods, on the order of a few days.

The proposed trigger for radon in air is 4 pCi/L, and the proposed point of compliance is the MWL perimeter. The 4 pCi/L value is the EPA "action threshold" for radon in household air (U.S. EPA, 2005). This proposed value is much lower than the simulated radon-gas

concentrations (>10,000 pCi/L) at the surface of the MWL that yielded fluxes that exceeded the design standard of 20 pCi/m^2/s. Should the radon trigger of 4 pCi/L be exceeded in air at the MWL point of compliance, then the trigger evaluation process shown in Figure 25 will be implemented. Additional details regarding long-term monitoring of radon at the MWL will be presented in the MWL Long Term Monitoring and Maintenance Plan.

4.2.2 Vadose Zone Monitoring Triggers

The vadose zone beneath the MWL extends nearly 500 ft from ground surface to groundwater. Because VOCs released from the MWL have the potential to migrate to groundwater, a robust monitoring system is planned for the vadose zone at the MWL to serve as an early warning system for protecting groundwater. This system will provide early evidence of potential threats to groundwater, and it will allow corrective action to be initiated long before groundwater contamination occurs.

Long-term-monitoring of the vadose zone is planned for VOCs and for moisture content to ensure that the MWL remedy remains protective of human health and the environment. Details of the proposed monitoring systems for VOCs, moisture content, and trigger values are discussed below. Additional details regarding the frequency and extent of long-term monitoring activities will be included in the MWL Long-Term Monitoring and Maintenance Plan.

4.2.2.1 Volatile Organic Compounds

Volatile organic compounds (VOCs) are the most mobile of the hazardous constituents detected in soils beneath the MWL. Two passive and three active soil-gas surveys at the MWL have shown the presence of low concentrations of VOCs in soil gas (Peace et al., 2002). In addition, low concentrations of VOCs were detected in a 1993 study of VOC and tritium fluxes to the atmosphere from MWL soils (Radian Corp., 1993). Low concentrations of VOCs were also detected in subsurface soil samples collected from boreholes drilled during the MWL Phase 2 RFI.

VOC concentrations with depth will be monitored using three Flexible Liner Underground Technologies (FLUTe™) sampling wells. The FLUTes™ are proposed to be constructed in vertical boreholes located immediately outside the perimeter of the MWL cover with the locations selected near areas where the highest concentrations of VOCs were detected during earlier studies at the MWL. Actual locations of the FLUTe™ boreholes will be selected in conjunction with NMED. Soil gas sampling ports are proposed to be installed in each FLUTe™ at depths of 50 ft, 100 ft, 200 ft, 300 ft, and 400 ft below ground surface. Soil gas data collected from the FLUTes™ will be used to assess current VOC distributions with depth, and to monitor VOC concentrations over time, allowing early identification of any potential threats to groundwater.

Triggers are proposed for PCE, TCE, and total VOCs in soil gas at the MWL. The proposed triggers are 20 ppmv for PCE and TCE, and 25 ppmv for total VOCs. These triggers, although not based on risk or regulatory limits, are sufficiently low to protect groundwater quality of the

aquifer. All triggers would apply to samples collected from the deepest sampling port in each FLUTe™. Triggers would not apply to samples collected from shallower ports.

4.2.2.2 Moisture Content

Moisture content with depth will be monitored using a neutron moisture meter in three neutron probe access boreholes that were installed to a linear depth of 200 ft at a 30 degree angle directly below the waste disposal cells. The moisture content data will be used to evaluate infiltration through the MWL disposal cell cover. Infiltration is an important parameter for determining whether or not MWL performance objectives are met.

Infiltration through the cover will be indirectly monitored by monitoring the moisture content in the vadose zone beneath the MWL. A significant increase in moisture content beneath the landfill may indicate that the disposal cell cover may not be performing as originally designed, and that infiltration through the cover is greater than originally predicted.

Moisture contents will be measured using neutron logging, and data will be compared to baseline moisture content data collected prior to deployment of the MWL cover. A significant increase in moisture content within the vadose zone may indicate that corrective action is warranted in order to prevent the downward movement of liquid water through the disposal cell. Moisture content data will be evaluated to ensure that the performance objective of infiltration through the MWL cover is less than the EPA-prescribed technical equivalence criteria of 10^{-7} cm/s (31.5 mm/yr), as detailed below.

Infiltration may be estimated indirectly using Darcy's Law. The method is based on soil-physics and the relationship between unsaturated hydraulic conductivity and volumetric moisture content of subsurface soils. The method is described in detail in the MWL Phase 2 RFI SAND Report (Peace et al., 2002). Assumptions required for this method include one-dimensional, steady-state flow, a vertical hydraulic gradient of unity, and the assumption that the downward flux of water beneath the root zone will eventually reach groundwater.

If one applies these assumptions, then the downward flux at a particular depth is equivalent to the unsaturated hydraulic conductivity as a function of the moisture content at that depth. Thus, by monitoring the moisture content of the vadose zone beneath the MWL, one can also indirectly monitor the downward flux through the vadose zone. If infiltration through the cover increases significantly, than the downward flux through the vadose zone would increase as well, resulting in higher moisture content in the vadose zone beneath the landfill. Hence, by monitoring moisture content in the vadose zone, one can indirectly monitor the performance of the MWL cover. A significant increase in moisture content beneath the MWL may indicate that the cover is not performing as designed.

Figure 27 shows the calculated unsaturated hydraulic conductivity curves for 18 subsurface soil samples collected from the IP Test site, located approximately 500 ft west of the MWL. Based on this figure, and assuming a unit gradient in the vadose zone, if infiltration through the MWL cover exceeds the EPA-prescribed technical equivalence criteria of 10^{-7} cm/s (31.5 mm/yr), then volumetric moisture content in the underlying soils will exceed approximately 23 percent.

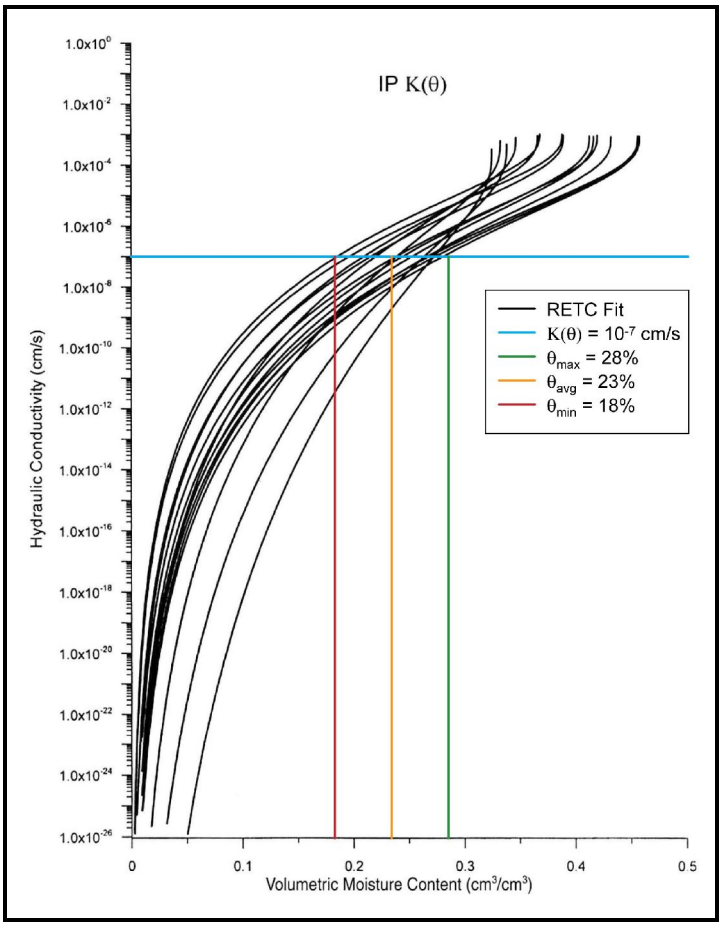

Figure 27. Hydraulic conductivity as a function of volumetric moisture content for different soil samples at the site

The recommended trigger level is the moisture content which corresponds to an unsaturated hydraulic conductivity equal to the EPA-prescribed technical equivalence criteria of 10^{-7} cm/s (31.5 mm/yr). The moisture content at which this occurs is 23 percent by volume, and the proposed trigger level for moisture content in the vadose zone is, therefore, 23 percent by volume. This value is based arbitrarily on the EPA-prescribed technical equivalence criteria, and does not necessarily indicate that hazardous constituents or radionuclides are migrating from the landfill.

The 23-percent trigger is proposed for linear depths of 10 ft and 100 ft (vertical depths of 8.7 ft to 86.6 ft) along the neutron probe access holes in the vadose zone beneath the MWL. This interval is proposed as the "regulated interval" because it lies beneath the root zone, and yet is shallow enough that a response would be detected fairly rapidly if there is a significant increase in infiltration through the cover. Should this 23-percent trigger level be exceeded in the regulated interval, then the process shown in Figure 25 will be implemented. Additional details regarding vadose zone monitoring at the MWL will be presented in the MWL Long Term Monitoring and Maintenance Plan.

4.2.3 Groundwater Monitoring Triggers

Based on the results of the probabilistic performance-assessment modeling, monitoring triggers are proposed for uranium and VOCs in groundwater at the MWL. These proposed triggers are discussed below.

4.2.3.1 Uranium

Uranium occurs naturally in MWL groundwater at concentrations ranging from 1.34 to 9.23 µg/L, and averaging 5.97 µg/L. Total uranium concentrations in groundwater beneath the MWL are well within the total uranium ranges established by the United States Geological Survey (USGS) for the Middle Rio Grande Basin (USGS, 2002). Isotopic analyses of uranium have demonstrated that it is of natural origin (Goering et al., 2002).

The probabilistic performance-assessment modeling for the MWL indicates that there is a possibility that uranium will reach the groundwater (although none of the simulations showed the uranium concentrations exceeding the regulatory metric of 30 µg/L). For this reason, a monitoring trigger of 15 µg/L (1/2 of the EPA MCL) is proposed for uranium in MWL groundwater at the point of compliance. The proposed point of compliance is at the downgradient monitoring wells. Should the uranium trigger value be exceeded in MWL groundwater at the point of compliance, then the trigger evaluation process shown in Figure 25 will be implemented. Additional details regarding long-term monitoring of uranium in groundwater will be presented in the MWL Long Term Monitoring and Maintenance Plan.

4.2.3.2 Volatile Organic Compound Triggers for Groundwater

Groundwater monitoring for VOCs at the MWL has been conducted for sixteen years, since September 1990, and there is no evidence that wastes from the MWL have contaminated groundwater. However, earlier studies as well as the current probabilistic performance-assessment modeling have shown that there is a potential for VOCs to contaminate groundwater at the MWL.

The potential downward vertical transport of six organic compounds to groundwater by both aqueous-phase transport and vapor-phase transport was evaluated in 1995 (Klavetter, 1995). The study showed that PCE could eventually migrate to groundwater through vapor-phase transport. Although the modeling predicted that the most likely PCE concentrations in groundwater would be considerably lower than the detection limit of 0.5 ppb, sensitivity analyses suggested that PCE concentrations could potentially reach 1 to 5 ppb within 50 years (Klavetter, 1995a).

The current probabilistic performance-assessment modeling also simulated the migration of PCE to groundwater and arrived at similar conclusions regarding the potential contamination of groundwater by PCE through vapor-phase transport. Because PCE was modeled in this study as a proxy for other VOCs detected in soil gas and in soils beneath the MWL, there is a potential for other VOCs from the MWL to also migrate to groundwater in the future. For this reason, continued groundwater monitoring for VOCs at the MWL is recommended.

Groundwater trigger levels are proposed for all Target Compound List VOCs for which there are primary EPA MCLs, or for which there are EPA Region 6 Human Health Medium-Specific Screening Levels. The proposed groundwater trigger levels for VOCs with MCLs are equal to ½ of the EPA MCLs; concentrations of VOCs with no corresponding MCLs will be compared to the EPA Region 6 Human Health Medium-Specific Screening Levels.

The proposed point of compliance is the downgradient monitoring wells. Should any VOC trigger values be exceeded in MWL groundwater at the point of compliance, then the trigger evaluation process shown in Figure 25 will be implemented. Additional details regarding long-term monitoring of VOCs in groundwater will be presented in the MWL Long Term Monitoring and Maintenance Plan.

4.3 Summary of Recommended Triggers

Based on the results of the probabilistic performance-assessment modeling conducted for the MWL, monitoring triggers have been proposed for the air, surface soil, vadose zone, and groundwater at the MWL. Specific triggers include numerical thresholds for (1) radon concentrations in the air, (2) tritium, gamma-emitting radionuclides, and heavy metal concentrations in surface soil, (3) VOC concentrations and moisture content in the vadose zone, and (4) uranium and VOC concentrations in groundwater. The proposed triggers are based on EPA, DOE and NMED regulatory standards, as well as on NMED maximum background concentrations for select radionuclides. If a trigger is exceeded, then SNL/DOE will initiate a trigger evaluation process which will allow sufficient data to be collected to assess trends and recommend corrective action, if necessary.

By utilizing these triggers during long-term monitoring at the MWL, SNL/DOE will ensure that the MWL remedy continues to protect human health and the environment, while meeting the performance objectives for the cover and the corrective action objectives established in the MWL Corrective Measures Study.

5. Summary and Conclusions

A probabilistic performance assessment has been conducted to evaluate the fate and transport of contaminants of concern at the Mixed Waste Landfill. The contaminants that were simulated include radionuclides (amercium-241, cesium-137, cobalt-60, plutonium-238, plutonium-239, radium-226, radon-222, strontium-90, thorium-232, tritium, and uranium-238), heavy metals (lead and cadmium), and a volatile organic compound (PCE). The current analysis differs from previous analyses in several ways: (1) probabilistic analyses[5] were performed to quantify uncertainties inherent in the system and models; (2) a comprehensive analysis of the performance of the MWL was evaluated and compared against relevant regulatory metrics; (3) sensitivity analyses were performed to identify parameters and processes that were most important to the simulated performance metrics; and (4) long-term monitoring requirements and

[5] One hundred realizations were used in the probabilistic analyses. A preliminary comparison between the results of 100 vs. 200 realizations revealed that the output distribution was adequately represented by 100 realizations.

triggers were recommended based on the results of the quantified uncertainty and sensitivity analyses. The key results of this study are summarized below:

Infiltration through the Cover:

- Net infiltration through the engineered cover at the MWL was simulated to be less than the regulatory metric of 10^{-7} cm/s for all conditions and scenarios.

- Predicted average infiltration rates through the MWL cover are expected to range from 1.18×10^{-9} cm/s for present conditions to 6.12×10^{-9} cm/s for future conditions. These values were used in a uniform distribution for the performance-assessment simulations.

- To ensure that future infiltration rates will not exceed the regulatory metric of 10^{-7} cm/s, the moisture content of the vadose zone will be monitored. Based on the site-specific two-phase characteristic curves of the soil, a moisture content of 23 percent by volume will be used as a trigger to indicate if the infiltration metric is exceeded.

Release of Radionuclides to the Atmosphere:

- A small percentage (2%) of the simulated dose due to exposure to tritium via the air pathway exceeded the regulatory metric of 10 mrem/year.

- Parameters impacting tritium diffusion through both the liquid and gas phases (e.g., tortuosity coefficient, moisture content, cover thickness, atmospheric boundary-layer thickness) were found to be important to the simulated inhalation dose.

- Sensitivity studies show that the emanation factor, which depends on the integrity of the radium-226 containment, is important to the performance of the landfill with regard to surface radon fluxes.

- For a maximum radon emanation factor of 0.01 (1% of the radium-226 containers fail), the simulated radon surface fluxes exceed the design standard of 20 pCi/m^2/s in about 3% of the realizations. For a maximum radon emanation factor of 1 (100% of the radium-226 containers fail), the simulated radon surface fluxes exceed the design standard in about 30% of the realizations.

- Based on these results, both radon and tritium concentrations are recommended to be monitored at the surface of the MWL in the future. In addition, other radionuclides that may be brought to the surface by burrowing animals or insects are also proposed to be monitored. Specific triggers are identified in Table 6.

Release of Radionuclides to the Groundwater:

- None of the radionuclides were simulated to reach the groundwater within 1,000 years for all realizations.

- Only uranium-238 (and some of its decay products) were simulated to reach the water table for extended periods (>10,000 years). All peak aquifer concentrations were still less than the EPA regulatory metric of 30 μg/L.

- Infiltration rate was found to be the most significant parameter impacting the variability in the simulated groundwater concentrations and dose via groundwater. Uranium groundwater concentrations were simulated to exceed the regulatory metric of 30 μg/L if the infiltration increased two orders of magnitude above the maximum stochastic value to 6.12×10^{-9} m/s.

- Uranium in the groundwater will be monitored in the future and a trigger value of 15 μg/L, equal to ½ of the U.S. EPA maximum contaminant level in drinking water, is proposed.

Release of Heavy Metals to the Groundwater:

- Neither lead nor cadmium were simulated to reach the groundwater in 1,000 years (or extended periods past 10,000 years)

- Additional increases in infiltration (3-4 orders of magnitude over expected maximum infiltration rates) allowed cadmium and lead to reach the groundwater in 1,000 years.

- No triggers are recommended for lead or cadmium in groundwater at this time. However, heavy metals are proposed to be monitored in surface soils to account for transport by burrowing animals or instects (see Table 6).

Release of VOCs to the Groundwater:

- Only 1% of the realizations yielded peak PCE concentrations in the groundwater that exceeded the regulatory metric of 5 μg/L. The majority of the realizations showed that the peak PCE groundwater concentration occurred within 100 years.

- Uncertainty in the PCE adsorption coefficient, half-life (degradation), inventory concentration, source thickness, and cover thickness were found to be significantly correlated to the simulated groundwater concentrations.

- Based on these results, PCE and other volatile organic compounds are recommended to be monitored in the vadose zone and in groundwater at the MWL in the future (see Table 6). Trigger values for the soil gas in the vadose zone will be 20 ppmv for TCE and PCE, and 25 ppmv for total VOCs. Trigger values in groundwater will be based on values equal to ½ of the U.S. EPA maximum contaminant levels in drinking water. Concentrations of VOCs with no corresponding MCLs will be compared to the EPA Region 6 Human Health Medium-Specific Screening Levels.

Key Assumptions:

- Receptor located adjacent to MWL
    - Tritium dose caused by continuous inhalation and exposure of tritium flux directly above MWL.
    - Groundwater dose calculated based on concentrations in aquifer directly beneath MWL. Water intake assumed to be 10 L/day (five times EPA standard of 2 L/day for drinking water).
- Maximum waste inventory set equal to twice estimated values based on historical records.
- Sealed sources of radium-226 allowed to degrade in 1,000 years (emanation factor for radon-222 allowed to increase).
- Cover allowed to completely erode in 1,000 years.
- 1-D model: yields maximum transport to surface and groundwater.
- Bounding tortuosity coefficients: yields maximum diffusion rates.

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7. Appendix A: Derivation of a Steady-State Gas and Liquid-Phase Radon Transport Model

A steady-state radon transport model is derived here to account for advection in the liquid phase, diffusion in both the liquid and gas phases, and decay of radon-222. Because radium-226, which is the source of radon-222, has a half-life of 1,600 years, we assume steady-state conditions (e.g., the source of radon-222 is constant and the resulting long-term radon-222 concentration profile does not change with time). Assuming steady-state conditions is conservative because the radon-222 concentration profile is assumed to develop instantaneously.

We define three regions in the model: (1) a clean overburden (or cover) free of radium-226 that extends to a depth, L_1, beneath the surface; (2) a contaminated source zone of radium-226 that extends to a depth, L_2, from the surface; and (3) a vadose zone free of radium-226 that extends a distance, L_3, to the water table (see Figure 28). The radon-222 generated by the radium-226 is free to diffuse and advect upward to the atmosphere and downward toward the water table. Downward liquid advection also carries aqueous-phase radon toward the water table. Partitioning of radon between the gas and liquid phases is assumed to occur instantaneously and can be described by a liquid/gas partitioning coefficient, k (this is the inverse of Henry's constant, K_H). The steady-state governing equations for the transport of radon-222 in these two regions is as follows:

$$D_{eff}^{(1)} \frac{d^2 C_g^{(1)}}{dx^2} - kq \frac{dC_g^{(1)}}{dx} - \lambda C_g^{(1)} \left(\theta_g^{(1)} + k\theta_w^{(1)} \right) = 0 \tag{1}$$

$$D_{eff}^{(2)} \frac{d^2 C_g^{(2)}}{dx^2} - kq \frac{dC_g^{(2)}}{dx} - \lambda C_g^{(2)} \left(\theta_g^{(2)} + k\theta_w^{(2)} \right) + \dot{Q} = 0 \tag{2}$$

$$D_{eff}^{(3)} \frac{d^2 C_g^{(3)}}{dx^2} - kq \frac{dC_g^{(3)}}{dx} - \lambda C_g^{(3)} \left(\theta_g^{(3)} + k\theta_w^{(3)} \right) = 0 \tag{3}$$

where

$$D_{eff} = \frac{0.07}{10^4} e^{\left[-4\left(S_l - S_l\phi^2 + S_l^5\right)\right]} \tag{4}$$

$$\dot{Q} = \frac{E\ Ci_{226}}{SA_{226} 1000} \frac{\lambda_{226}}{V} \tag{5}$$

where the superscripts (1), (2), and (3) denote the three regions shown in Figure 28, C_g is the radon gas-phase concentration [kg/m^3], x is the distance from the surface [m] (positive downward), D_{eff} is the effective diffusion coefficient [m^2/s] for combined gas and aqueous phases (Rogers et al., 1984), S_l is the liquid saturation [-], k is the water/gas partitioning coefficient (i.e., water concentration/gas concentration) [-], q is the Darcy infiltration rate [m/s], λ is the decay

coefficient for radon-222 and is calculated as ln(2)/half-life [1/s], θ_g and θ_w are the gas and moisture volumetric contents, respectively, $\dot{Q}$ is the volumetric generation rate of radon-222 [kg/m^3/s], E is the emanation factor for radon-222 that accounts for containment of the radium-226 (0 = complete containment; 1 = no containment), Ci_{226} is the concentration of radium-226 in curies, SA_{226} is the specific activity of radium-226 [Ci/g], λ_{226} is the decay coefficient for radium-226 [1/s], and V is the total volume of the contaminated waste zone (region 2). In this derivation, we assume local equilibrium between the gas and aqueous phases; therefore, the equation can be expressed entirely in terms of the gas concentration, C_g, and the partitioning coefficient, k, is used to convert between the gas concentration and aqueous concentration.

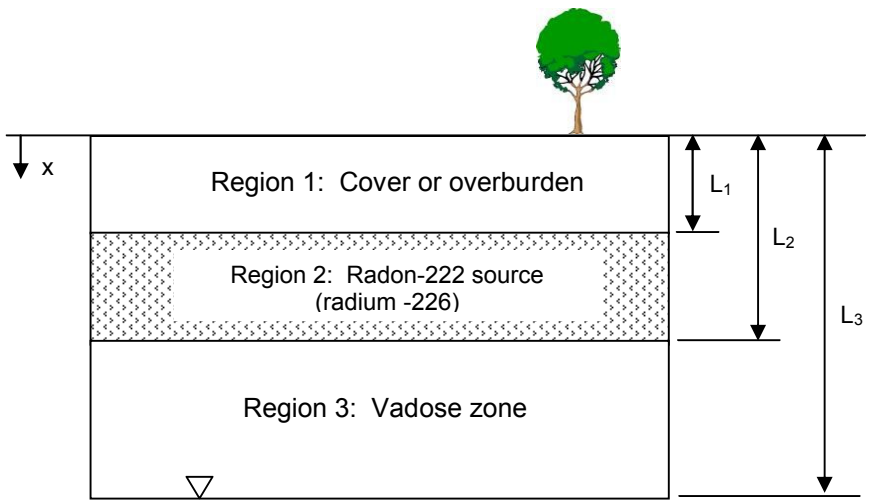

Figure 28. Conceptual model of three-region radon-transport model.

The boundary conditions for this system are as follows: (1) the radon concentration at the surface in region 1 is zero (this is conservative because it creates the largest gradient for radon flux to the atmosphere); (2) the radon concentration in region 1 is equal to the radon concentration in region 2 at the interface of regions 1 and 2; (3) the radon flux in region 1 reaching the interface between regions 1 and 2 must be equal to the radon flux entering region 2; (4) the radon concentration in region 2 is equal to the radon concentration in region 3 at the interface of regions 2 and 3; (5) the radon flux in region 2 reaching the interface between regions 2 and 3 must be equal to the radon flux entering region 3; and (6) the radon concentration infinitely far away from the source (as $x \rightarrow \infty$) goes to zero. These boundary conditions can be expressed as follows:

$$C_g^{(1)}(x=0)=0 \tag{6}$$

$$\begin{aligned} C_g^{(1)}(x=L_s) &= C_g^{(2)}(x=L_s) \\ C_w^{(1)}(x=L_s) &= C_w^{(2)}(x=L_s) \end{aligned} \tag{7}$$

$$D_{eff}^{(1)} \left. \frac{dC_g^{(1)}}{dx} \right|_{x=L_s} = D_{eff}^{(2)} \left. \frac{dC_g^{(2)}}{dx} \right|_{x=L_s} \tag{8}$$

$$\begin{aligned} C_g^{(2)}(x = L_s) &= C_g^{(3)}(x = L_s) \\ C_w^{(2)}(x = L_s) &= C_w^{(3)}(x = L_s) \end{aligned} \tag{9}$$

$$D_{eff}^{(2)} \left. \frac{dC_g^{(2)}}{dx} \right|_{x=L_s} = D_{eff}^{(3)} \left. \frac{dC_g^{(3)}}{dx} \right|_{x=L_s} \tag{10}$$

$$C_g^{(3)}(x \to \infty) = 0 \tag{11}$$

If we assume that the soil properties and hydrologic conditions are the same in all three regions, the solutions to the ordinary differential equations (1) - (3) for the three regions can be expressed as follows:

$$C_g^{(1)} = c_1 e^{r_1 x} + c_2 e^{r_2 x} \tag{12}$$

$$C_g^{(2)} = c_3 e^{r_1 x} + c_4 e^{r_2 x} + Q_{source} \tag{13}$$

$$C_g^{(3)} = c_5 e^{r_1 x} + c_6 e^{r_2 x} \tag{14}$$

where

$$c_1 = -c_2 \tag{15}$$

$$c_2 = \frac{c_3 e^{r_1 L_1} \left(r_1 - r_2 \right) - r_2 Q_{source}}{e^{r_1 L_1} \left(r_2 - r_1 \right)} \tag{16}$$

$$c_3 = \frac{r_2 Q_{source}}{e^{r_1 L_2} \left(r_1 - r_2 \right)} \tag{17}$$

$$c_4 = \frac{c_2 \left(e^{r_2 L_1} - e^{r_1 L_1} \right) - c_3 e^{r_1 L_1} - Q_{source}}{e^{r_2 L_1}} \tag{18}$$

$$c_5 = 0 \tag{19}$$

$$c_6 = \frac{c_3 e^{r_1 L_2} + c_4 e^{r_2 L_2} Q_{source}}{e^{r_2 L_2}} \tag{20}$$

$$r_1 = \frac{kq + \sqrt{(kq)^2 + 4D_{eff}\lambda_{eff}}}{2D_{eff}} \tag{21}$$

$$r_2 = \frac{kq - \sqrt{(kq)^2 + 4D_{eff}\lambda_{eff}}}{2D_{eff}} \tag{22}$$

$$\lambda_{eff} = \lambda\left(\theta_g + k\theta_w\right) \tag{23}$$

$$Q_{source} = \frac{\dot{Q}}{\lambda_{eff}} \tag{24}$$

Equations (12) - (24) yield the solutions for the gas concentrations in the three regions defined in Figure 28. The aqueous concentration can be obtained by multiplying the gas concentration at any location by the liquid/gas partition coefficient, *k*. The groundwater concentration at the interface of the vadose zone and the water table, $C_w^{(3)}(L_3)$, can be expressed as follows:

$$C_w^{(3)}(L_3) = k\, C_g^{(3)}(L_3) \tag{25}$$

The upward flux of radon-222 gas at the surface, q_s [kg/m^2/s] can be determined by evaluating the gas-phase concentration gradient at the surface (region 1) using Fick's Law:

$$q_s = -\left(-D_{eff}\left.\frac{dC_g^{(1)}}{dx}\right|_{x=0}\right) = D_{eff}c_2(r_2 - r_1) \tag{26}$$

The negative sign preceding the term in parentheses is to account for the positive downward direction of *x*. Equation (26) is used to estimate the radon gas flux at the surface in the performance assessment, and Equation (25) is used to determine the radon groundwater concentration. The concentration and flux of radon can be converted to pCi/L and pCi/m^2/s using the specific activity of radon (see Table 2) and appropriate unit conversions.

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