JOURNAL OF GEOPHY[illegible]AL RESEARCH, VOL. 33, NO. 11, PAGES 2[illegible]-2545 NOVEMBER 1997

102730

Modeling and risk assessment of a 30-year-old subsurface radioactive-liquid drain field

Lon A. Dawson and Phillip I. Pohl

Energy and Environment Division, Sandia National Laboratories, Albuquerque, New Mexico

Abstract. The contamination from a 30-year-old radioactive liquid drain field was assessed for movement in the subsurface and potential risks to humans. This assessment included determining field concentrations of cesium 137 (^{137}Cs) and other inorganic contaminants and modeling of the flow and transport of the liquid waste that was sent to the drain field. The field investigation detected no contamination deeper than 15 feet (4.6 m) from the bottom of the drain field. Prediction of the water content of the vadose zone showed no saturated conditions for times greater than 10 years after the known infiltration. Sensitivity analysis of the modeling parameters showed the equilibrium sorption coefficient to be the most important factor in predicting the contaminant plumes. Calibration of modeling results with field data gave a ^{137}Cs sorption coefficient that is within the range of values found in the literature. The risk assessment for the site showed that the contamination poses no significant risk to human health.

1. Introduction

In 1980 Congress enacted the Low-Level Radioactive Waste Policy Act of 1980, Public Law 96-573 [*U.S. Congress*, 1980], to promote the safe and efficient disposal of low-level radioactive waste. In 1985 there was an amendment to this act, Public Law 99-240 [*U.S. Congress*, 1980]. Because of this act, many states are currently addressing siting issues within either their state or a compact of states. The main scientific and safety issue for these states is to address potential contamination from radioactive waste disposal and the need for good scientific data and predictions on migration of radionuclides in the subsurface. In this article, contamination from a radioactive liquid drain field was assessed for movement in the subsurface and potential risks to humans.

A significant portion of the nuclear radiation–effects testing on weapons and equipment, as well as nuclear design and engineering, performed at Sandia National Laboratories/New Mexico (SNL/NM) was conducted at the Sandia Engineering Reactor Facility. Operation of this facility resulted in the generation of industrial waste water, some of which contained low concentrations of radionuclides. The Liquid Waste Disposal System (LWDS) was designed to dispose this waste water by infiltration into underlying soils. The LWDS (Figure 1) consists of three holding tanks and the associated pumping system, a drain field, and two surface impoundments. The drain field was used from 1963 to 1967; it collapsed in 1967 and was replaced with the two surface impoundments [*Sandia National Laboratories*, 1993]. The drain field was a long cement pipe filled with gravel and buried in the ground. The drain field received a total of more than 6 million gallons (22,700 m^3) of radioactive cooling water. The precise chemical nature of the discharged water is not known; however, the water was analyzed for gamma-radiation activity prior to discharge to the drain field.

The sediments underlying the LWDS facility consist of alluvial fan deposits derived from the mountains to the east [*Sandia National Laboratories*, 1993]. The geophysical and lithologic information in the area indicates that coarse-grained sediments dominate the upper 100 feet (30.5 m) and that grain size decreases downward to the groundwater table, at a depth of approximately 495 feet (150.9 m). Table 1 summarizes the volumes and radioactive content of the drain field discharges. The activities listed in Table 1 are the annual sums of gross activities measured at the times of discharge from the holding tanks. The discharge sampling was usually done within a few days of the liquid's irradiation at the reactor facility. The activity measured is assumed to have been heavily dominated by relatively short-lived activation products. It is not practical to correlate the original activities to current activity levels because the latter would be dominated by longer-lived radionuclides.

In March 1994 a site investigation was completed that included drilling and sampling of four boreholes in and adjacent to the drain field. More than 40 soil samples were collected and analyzed for gamma-emitting radionuclides, tritium, volatile and semivolatile organic contaminants, and metals at an offsite contract laboratory following the analytical procedures required in the Environmental Protection Agency (EPA)–approved investigation workplan [*Sandia National Laboratories*, 1993]. The soil was photographed and lithologically described, but no other soil characteristics were measured [*Dawson*, 1995].

Initial infiltration modeling of the drain field was conducted in 1991 prior to any field investigation (D. M. Peterson, unpublished report to SNL, 1991). The present article reports on a study which expanded the earlier infiltration modeling to examine the transport of contamination, principally ^{137}Cs, from the site. Cesium 137 was chosen because it is the more widely distributed of the radioactive contaminants. It also has the longest half-life and is present at the highest concentrations. Flow and transport modeling began with a baseline simulation that incorporated a set of conservative input parameters that were selected on the basis of engineering judgment. The baseline scenario then was subjected to a sensitivity anal-

Copyright 1997 by the American Geophysical Union.

Paper number 97WR01700.
0043-1397/97/97WR-01700$09.00

31151

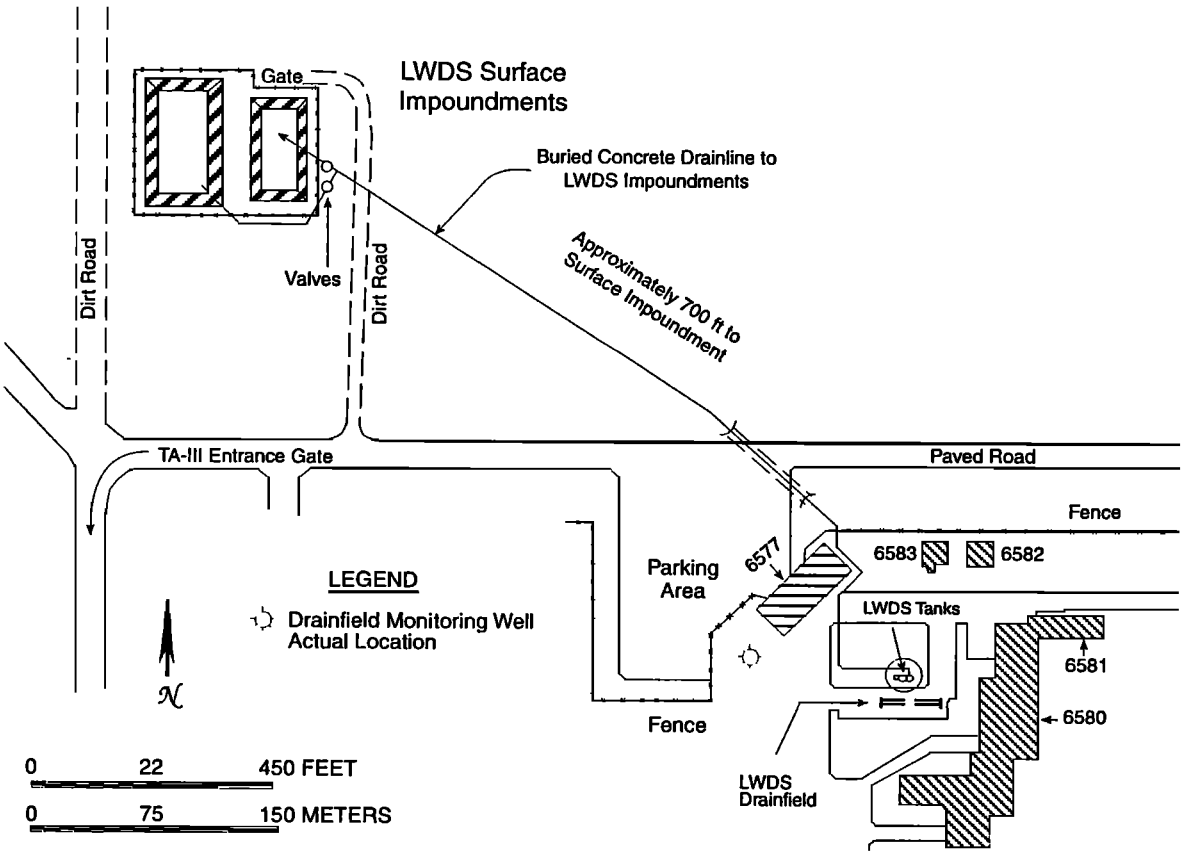

Figure 1. Diagram of Sandia National Laboratories/New Mexico's Liquid Waste Disposal System (LWDS).

ysis to determine the relative importance of the input parameters. Next, the LWDS drain field investigation field data were used to calibrate the model. Finally, the model was used to estimate risks to human health and the environment.

2. Technical Approach

2.1. Experimental Data Collection

In response to applicable environmental regulations (Environmental Protection Agency, Identification and Listing of Hazardous Wastes, *U.S. Code of Federal Regulations 40, Part 261, 1990)*), the SNL/NM Environmental Restoration (ER) Project completed a Resource Conservation and Recovery Act (RCRA) facility investigation of the LWDS [*Dawson*, 1995]. This work was conducted specifically to define the nature and extent of contamination from the LWDS ER sites and to evaluate any associated potential risks to human health. The results are summarized here and used later in model calibration and evaluation. A complete report of the investigation is available in the LWDS RCRA facility investigation report [*Dawson*, 1995].

Table 1. LWDS Discharges to Drainfield

Year	Activity *Ci*	Volume, Gallons
1963	6.60	1,493,500
1964	0.92	1,806,500
1965	5.86	912,000
1966	2.98	1,442,000
1967	3.40	832,000
Summary	19.76	6,486,000

One gallon equals 0.003785 m^3, or 3.785 L.

Four soil boreholes were installed at the LWDS drain field. Figure 2 shows the borehole locations. Samples were collected from the core at discrete intervals using a stainless steel trowel that was decontaminated between samples. The samples were placed directly into appropriate sample containers and analyzed for gamma-emitting radionuclides, tritium, volatile and semivolatile organic contaminants, and metals at an offsite contract laboratory following the analytical procedures required in the EPA-approved investigation workplan [*Sandia National Laboratories*, 1993].

The investigation was successful in that the extent of site contamination was determined. However, more insight regarding the nature of the various transport processes involved was needed to predict the future transport of contamination and associated risk to human health and the environment.

2.2. Infiltration Modeling

The LWDS drain field was first modeled before any intrusive investigation to provide rough estimates of the hydraulic potentials and moisture contents that might be encountered during drilling (D. L. Peterson, unpublished report to SNL, 1991). The infiltration modeling consisted of a series of variably saturated flow simulations. The modeling provided estimates of the distributions of hydraulic head and moisture content below the LWDS drain field between 1963 and 1994. No attempt was made to calibrate the model to field data since moisture content data were not available at the time. Rather, the simula-

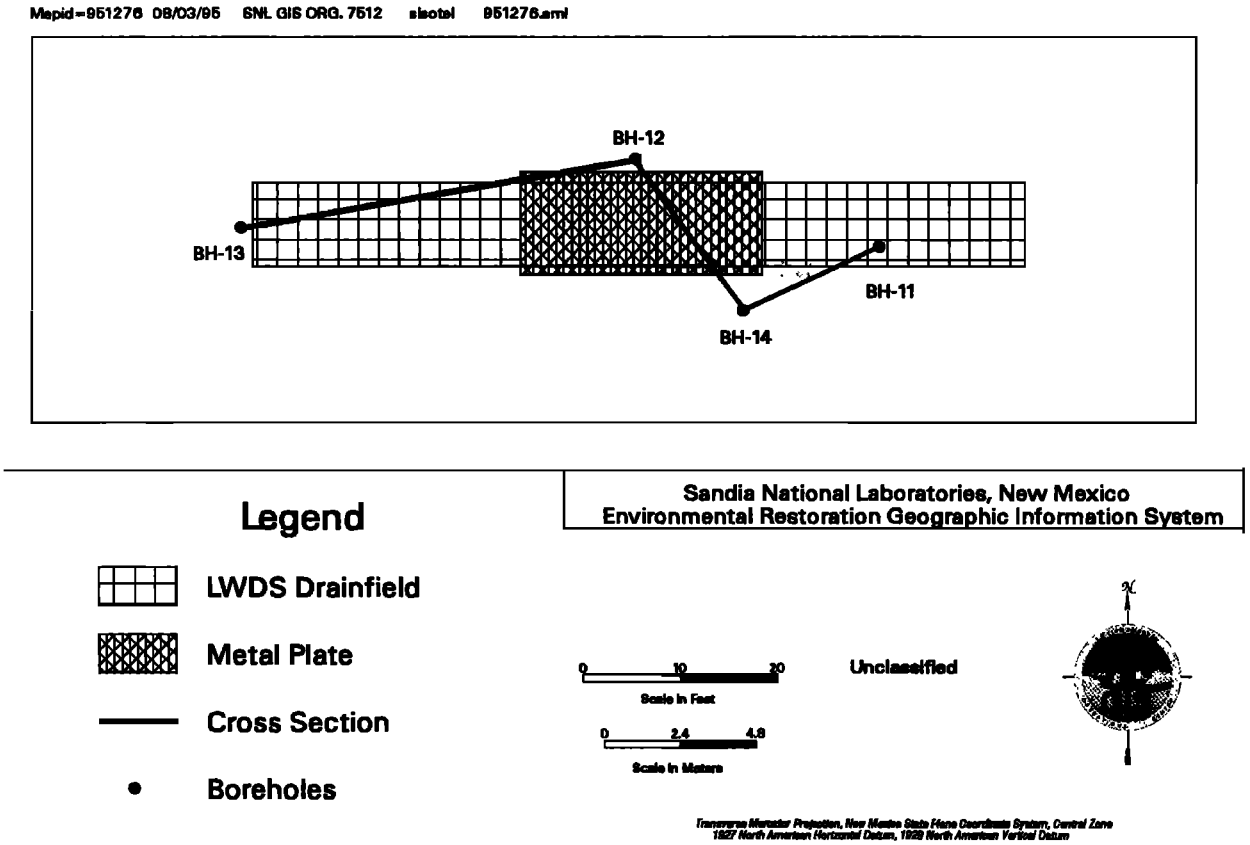

Figure 2. Borehole locations at the LWDS drain field.

tions were based on relatively simple flow systems that were assumed to consist of mostly homogeneous materials.

The U.S. Geological Survey (USGS) flow and transport code VS2DT was selected for modeling. The flow portion of this code was initially developed by *Lapalla et al.* [1987]. It assumes that subsurface fluid flux is described by Darcy's law extended to variably saturated conditions. The numerical solution code is based upon the finite difference method.

VS2DT is designed to solve for flow and transport in two-dimensional (2-D) planar or axisymmetric cylindrical domains. In the case of the LWDS modeling, VS2DT was used to model 2-D water flow in a vertical cross section that intersects the LWDS drain field. Symmetry was invoked along the drain field centerline. Thus the variably saturated flow simulation accounted for water movement on only one side of the drain field under the assumption that computed hydraulic potential and moisture content distribution would be mirror images of those occurring on the opposite side. The maximum grid size was limited to 10 feet (3.1 m) by 10 feet. The finite difference grid was designed such that the smallest grid blocks were located near the drain field, and dimensions became progressively larger with distance from the drain field until the 10-foot maximum was reached.

The drain field is estimated to be 6 feet (1.8 m) wide and 6 feet thick, with the top at a depth of 30 feet (9.1 m). Infiltration was simulated by introducing water at a depth of 30 feet in the three 1-foot (0.3 m) nodes horizontally adjacent to the drain field centerline (i.e., the left model boundary). The estimated annual water discharges to the LWDS drain field during the years 1963 through 1967 are presented in Table 1. This infiltration was assumed to have been uniformly distributed over the entire drain field and is accounted for in the model using a constant prescribed flux, which was calculated by dividing the volumes in Table 1 by the area of the drain field at the centerline (6 feet wide by 70 feet (21.3 m) long) and the time of 1 year. Each year in Table 1 was accounted for by a separate 1-year infiltration period.

It is important to note that on the basis of the field investigation, the actual infiltration occurred at the base of the drain field (36 feet (11 m)). The difference between the simulation and actual field conditions is accounted for by comparing the transport distances (i.e., the model-predicted distances), beginning at 30 feet (9.1 m), to the actual measured distances, beginning at 36 feet. The impacts of higher, localized flow were also examined and are discussed in the section on high flow rate simulations.

Boundary conditions were required at the base, top, and sides of the model domain. A prescribed capillary pressure head of zero was assigned to the bottom row of model nodes to represent the water table at a depth of approximately 495 feet (150.9 m). The left model boundary (i.e., drain field centerline) and right model limit were treated as no flow boundaries. The top layer was also treated as a no flow boundary because the drain field is covered by asphalt that effectively eliminates precipitation and evapotranspiration.

The material properties used in the infiltration modeling were taken from data and studies presented for the Technical Area III Chemical Waste Landfill [*Phelan et al.*, 1991; D. F. McTigue and C. L. Stein, unpublished SNL memorandum, 1990] and a geologic characterization study of the Technical Area III Mixed Waste Landfill [*Sandia National Laboratories and Ecology and Environment, Inc.*, 1990]. On the basis of this

Table 2. Infiltration and Transport Parameters

Parameter	Symbol	Baseline	Range	Ref.
Hydraulic conductivity, feet/day	K_{s_x}	3.0	0.3–30	5, 6, 7
Anisotropy [–]	K_{s_x}/K_{s_y}	10	10–25	5, 6, 7
Longitudinal dispersivity, feet	α_L	0.314	0.173–0.738	12
Transverse dispersivity, feet	α_T	0.063	0.016–0.205	12
Diffusion coefficient, feet^2/day	D_m	2.7 (10^{-5})	8.1 (10^{-6})–1.6 (10^{-4})	12
Equilibrium distribution coefficient, feet^3/kg	K_d	9.8	0.18–350	11

One foot equals 0.3048 m, 1 foot^2 equals 0.093 m^2, and 1 foot^3 equals 0.028 m^3.

information, a range in horizontal hydraulic conductivity, K_{s_x}, of 0.30–30 feet (0.09–9.1 m) per day was selected. This range was chosen to represent sandy alluvial materials of low to high conductivity. The high end of this range is commensurate with the highest conductivities measured in the area. However, the low end of the range is well above many of the lowest measured values. Thus the range selection is conservative in the sense that predicted infiltration distances will be greater than would be expected using the lowest values of K_{s_x}. Also, although the alluvial deposits may have small amounts of various materials from clay to gravel, any low-conductivity deposits are localized (i.e., very limited in areal extent) [*Dawson*, 1995] and the overall conductivity along flow paths is assumed to be dominated by the sandy, more permeable materials.

Owing to the stratification that is commonly observed in alluvial fill materials like those found at Sandia sites, water flow from the drain field was expected to be affected by anisotropy of hydraulic conductivity (K_{s_x}/K_{s_y}). On the basis of the previously mentioned local studies [*Phelan et al.*, 1991; *Sandia National Laboratories and Ecology and Environment, Inc.*, 1990; D. F. McTigue and C. L. Stein, unpublished SNL memorandum, 1990] anisotropy was predicted to range from 10 to 25. Table 2 summarizes the soil properties studied as variables in the infiltration modeling.

Functional relationships between pressure head, moisture content, and unsaturated hydraulic conductivity, as presented by *van Genuchten* [1978], were used to develop material property inputs for the VS2DT simulations [*Healy*, 1990]. The infiltration modeling used mean values of van Genuchten parameters (θ_s, θ_r, α, and N) from the previously mentioned local studies [*Phelan et al.*, 1991; *Sandia National Laboratories and Ecology and Environment, Inc.*, 1990; D. F. McTigue and C. L. Stein, unpublished SNL memorandum, 1990]. Residual moisture content (θ_R) was assumed to be 0.114, saturated moisture content (θ_s) was assumed to be 0.388, and parameters α and N were assigned values of 0.463 foot^{-1} (1.52 m^{-1}) and 1.871, respectively. These input parameters were treated as constants in the modeling.

2.3. Transport Modeling

The infiltration modeling was later expanded to examine the transport of contamination from the drain field. Cesium 137 was the main focus of the modeling because it was shown in the field investigation to be the most widely distributed radioactive contaminant and is present at the highest concentrations relative to background. Some supporting parallel calculations also were made for cadmium, another site contaminant. Actual field data were used to calibrate the transport model and to predict the fate of, and risks associated with, site contamination.

Similar to the infiltration modeling, the transport modeling was conducted under many simplifying assumptions; however, the effect of varying several input parameters over a large range was also tested. Solute transport was assumed to be described by the advection-dispersion transport equation [*Healy*, 1990]. Fluid viscosity, density, and temperature were assumed constant. Temperature and explicit concentration effects on diffusion were ignored. Scale and moisture content effects on dispersivity were ignored. The impacts of these assumptions are discussed in the sensitivity analysis.

Contaminant mass was introduced into the model domain by the infiltrating water. The initial concentration (C_0) in the infiltrating water is not precisely known, so C_0 was set to 1. This initial unit concentration automatically normalizes the solute distribution in the model. For comparison, field data were similarly normalized by dividing each measured concentration by the highest projected field concentration.

The radioactive decay of ^{137}Cs was included in the model with a half-life of 30.1 years. The model also included contaminant retention in soil. Since the movement of water in soils is often slow compared to the rate of adsorption [*Travis and Etnier*, 1981], it was assumed that equilibrium adsorption occurred. Our model included adsorption as described by a linear isotherm. Such isotherms are characterized by an unlimited capacity of the solid to adsorb the solute [*Travis and Etnier*, 1981]. This is an acceptable limitation for this study in that at the expected contaminant concentrations, the soil limit of sorption is unlikely to be reached. Using the linear isotherm also allows experimental results to be incorporated from other studies where this common simplification was assumed. The equilibrium distribution coefficient, K_d, values for this study were obtained for a sandy soil from *Sheppard and Thibault* [1990].

Values for the remaining transport parameters were taken from literature sources and are consistent with similar work conducted by the SNL/NM site-wide Hydrogeologic Characterization Project [*Sandia National Laboratories*, 1995]. These transport parameters are summarized in Table 2.

Modeling was conducted in four separate phases. First, a single scenario was simulated with a "best guess" set of input parameters, henceforth referred to as "baseline." The baseline parameters are shown in Table 2. The baseline simulation results represent an initial modeling approximation. These results compared quite favorably with actual contamination profiles, as discussed in the results section.

In the second phase of transport modeling, a sensitivity analysis was completed using ranges for parameters to represent the uncertainty of the actual field conditions. The ranges are provided in Table 2. The input parameters were individually varied to the low and high end of the predicted ranges. For example, while studying longitudinal dispersivity, α_L, all other parameters were held to their baseline values. Separate simu-

lations were completed with α_L at 0.738 feet (0.23 m) and then 0.173 feet (0.05 m); α_L then was returned to 0.314 feet (0.96 m) while another variable was manipulated.

The third step of transport modeling was calibration of the model. Because the sensitivity analysis indicated that the variability of only one parameter, K_d, has a significant effect on the modeling results, K_d was adjusted such that the model predictions of contaminant distribution at 31 years matched the distribution measured in the 1994 field investigation.

Finally, additional contaminants were briefly considered and the results of some high–flow rate simulations evaluated to better assess the field results.

3. Results

3.1. Experimental Data

The field investigation found soil contamination from several metals, radionuclides, and toluene. The contamination is limited to the drain field and the surrounding soil, and no contamination was detected below 45 feet (13.7 m). Metal contaminants included cadmium, chromium, and beryllium. The contaminant profile for cadmium is shown in Figure 3a. The contour plot was generated by linear interpolation of sample results in the four drain field boreholes along the fence diagram depicted in Figure 2.

The total amount of radioactive contamination was fairly low; cobalt 60 (^{60}Co) and ^{137}Cs were the only detected anthropogenic radionuclides. Cobalt 60 was detected in only one sample, most likely having decayed away due to its short half-life, and will not be further discussed. Cesium 137 was present at higher concentrations and was the more widely distributed of the radioactive contaminants. It was present at 40 feet (12.2 m) below ground surface but not detected in any of the 45-foot samples. Figure 3b shows the distribution of ^{137}Cs contamination.

In general, the profiles for the various drain field contaminants were very similar. The highest contaminant concentrations were consistently located just beneath the drain field bottom's elevation, which was approximately 36 feet (11 m). Borehole 12 was located just north of the drain field (Figure 2) and consistently contained the highest levels of contamination. This is likely the result of the original drain field construction excavation which was extended to the north for installing the LWDS holding tanks. The drain field discharge is expected to have followed the path of least resistance, which would be through the less-compacted fill materials.

3.2. Baseline Modeling Results

The baseline simulations were conducted using a saturated horizontal hydraulic conductivity (K_{s_x}) of 3.0 feet/day (0.9 m/d) and an anisotropy ratio (K_{s_x}/K_{s_y}) of 10. Predicted subsurface moisture profiles for 5 and 31 years are shown in Figures 4a and 4b. Linear interpolation was performed between the grid blocks to produce the contours. The infiltration front created by the drain field operation simulation is observed at a depth of at least 370 feet (112.8 m) and about 125 feet (38.1 m) laterally from the drain field centerline after 5 years. Moisture contents at this time range from the background value ($\theta = 0.114$) to saturation ($\theta = 0.388$). Saturated conditions are no longer observed after 10 years. After 31 years, lateral propagation has reached a horizontal distance of 180 feet (54.9 m), and the greatest concentration of moisture is observed between the depths of 300 feet (91.4 m) and 495 feet (150.9 m), directly below the drain field.

Baseline transport modeling results for 31 years are presented in Figure 5 (note the different length scale from Figure 4). These contours represent the distribution of the contamination (^{137}Cs) as it migrates from the drain field. In discussing the transport modeling results, we are typically interested in how far the contamination traveled (the contamination wavefront). Henceforth the term "5% wave front" refers to the farthest vertical transport distance at which the predicted concentration equals 5% of the maximum plume concentration. At 31 years the 5% wave front predicted by the model has traveled approximately 3 feet (0.9 m) below the drain field. Figure 3b shows that measurable ^{137}Cs contamination stops somewhere between 4 feet (1.2 m) and 9 feet (2.7 m) beneath the drain field. This is somewhat farther but still in the same order of magnitude as the baseline simulation results.

The model output was also examined at different times in the 31-year simulation period. Contaminant transport during the first 5 years was drastically different than the subsequent 26 years. The first five years of the modeling simulate the time from 1963 through 1967, when the drain field was in operation. During this time, the contaminant transport was subject to advection from the large amounts of water that were pumped through the drain field. In 1967, flow to the drain field stopped. Contaminant transport still occurs through unsaturated flow, but its contribution to plume growth is insignificant, and the radioactive decay of ^{137}Cs begins to dominate. The contaminant plume begins to recede slowly.

3.3. Sensitivity Analysis

A sensitivity analysis was performed to study the relative impacts of each of the input parameters listed in Table 2. Table 3 summarizes the results of the 12 sensitivity analysis simulations. For each simulation the vertical distance of the 5% wave front was determined at the end of the 5-year infiltration period. The impacts of each of the six parameters studied in the sensitivity analysis is evident.

Considering Table 3, the sensitivity of transport to variations in dispersion is insignificant. Manipulating the dispersion-related input parameters (transverse and longitudinal dispersivity, α_T and α_L, and coefficient of diffusion, D_m) to the full extent of their predicted ranges has no measurable effect on the 5-year, 5% wave front. In a period of high flow, such as the first 5 years of the simulation, it is expected that advection would dominate dispersion in the contaminant transport. These results show that the assumptions and limitations regarding dispersion and diffusion-related modeling parameters have comparatively little impact on the model output. It is likely that dispersion during the unsaturated flow period becomes relatively more important, compared to advective transport. However, as shown in the baseline results, all ^{137}Cs transport during this time is more than offset by the radioactive decay.

The advection-related input parameters (hydraulic conductivity, K_{s_x}, and anisotropy, K_{s_x}/K_{s_y}) are comparatively more important, as demonstrated in Table 3. Changing the hydraulic conductivity from 0.3 feet/day (0.09 m/d) to 30 feet/day (9.1 m/d) caused the wave front to change from 1.9 feet (0.6 m) to 3.4 feet (1.04 m). Increasing the anisotropy of the soil causes the wave front to drop to 1.5 feet (0.5 m). This is expected, because as the anisotropy is increased, more water flows horizontally and comparatively less water infiltrates vertically. Al-

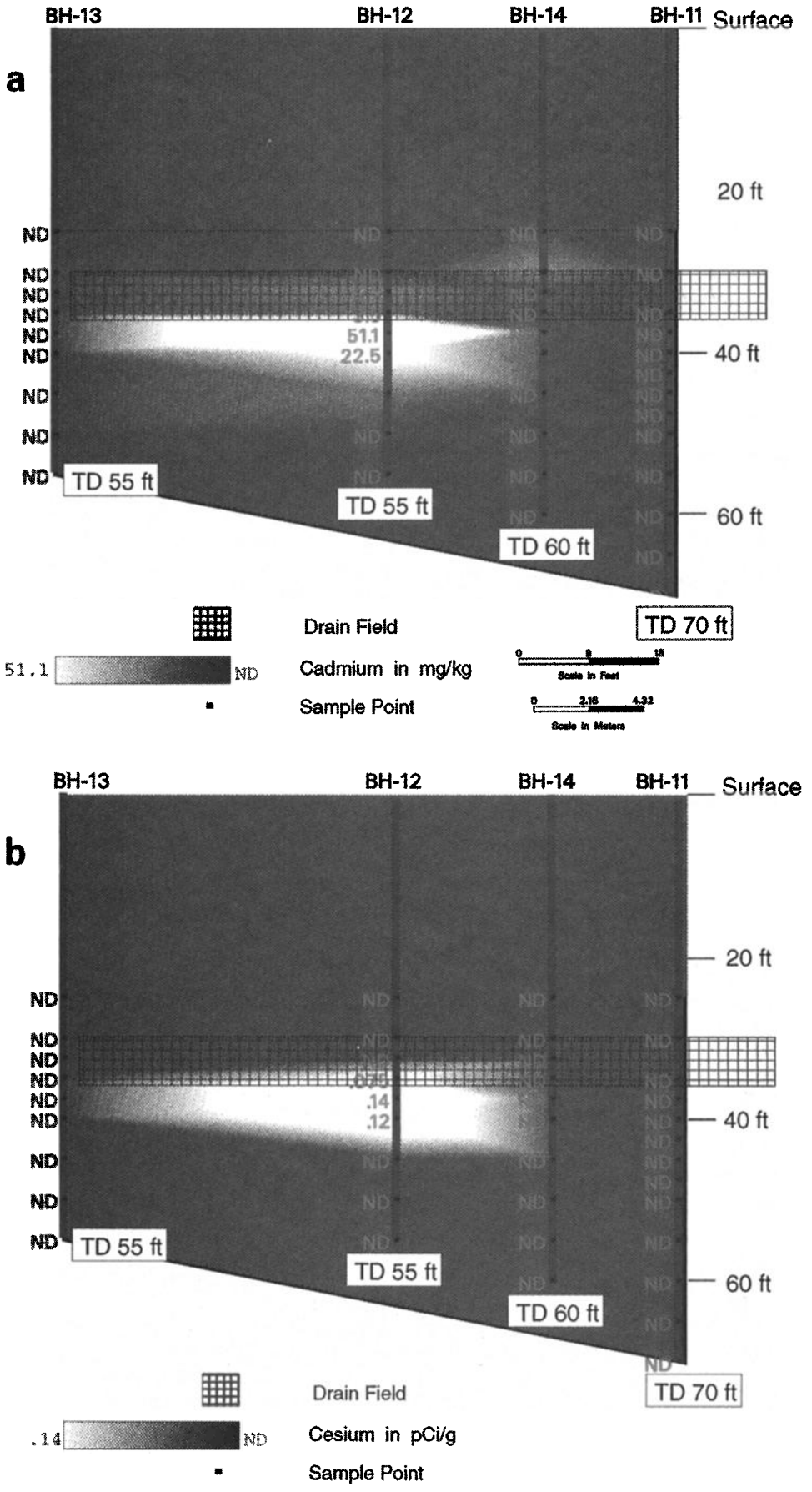

Figure 3. Contaminant profiles. Cross sections of (a) Cd and (b) ^{137}Cs at LDWS drain field.

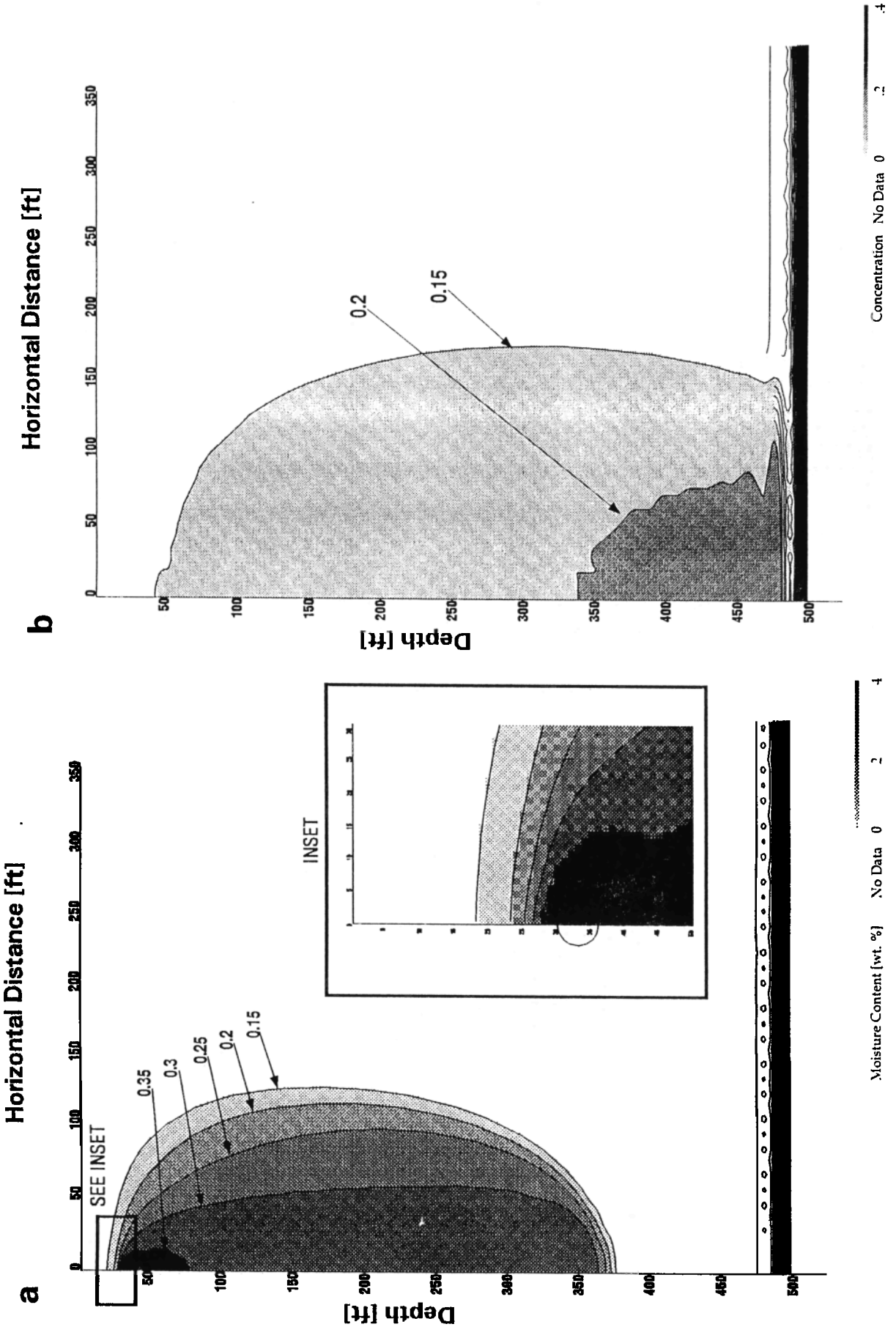

Figure 4. Moisture content from simulations after (a) a 5-year discharge period and (b) 31 years.

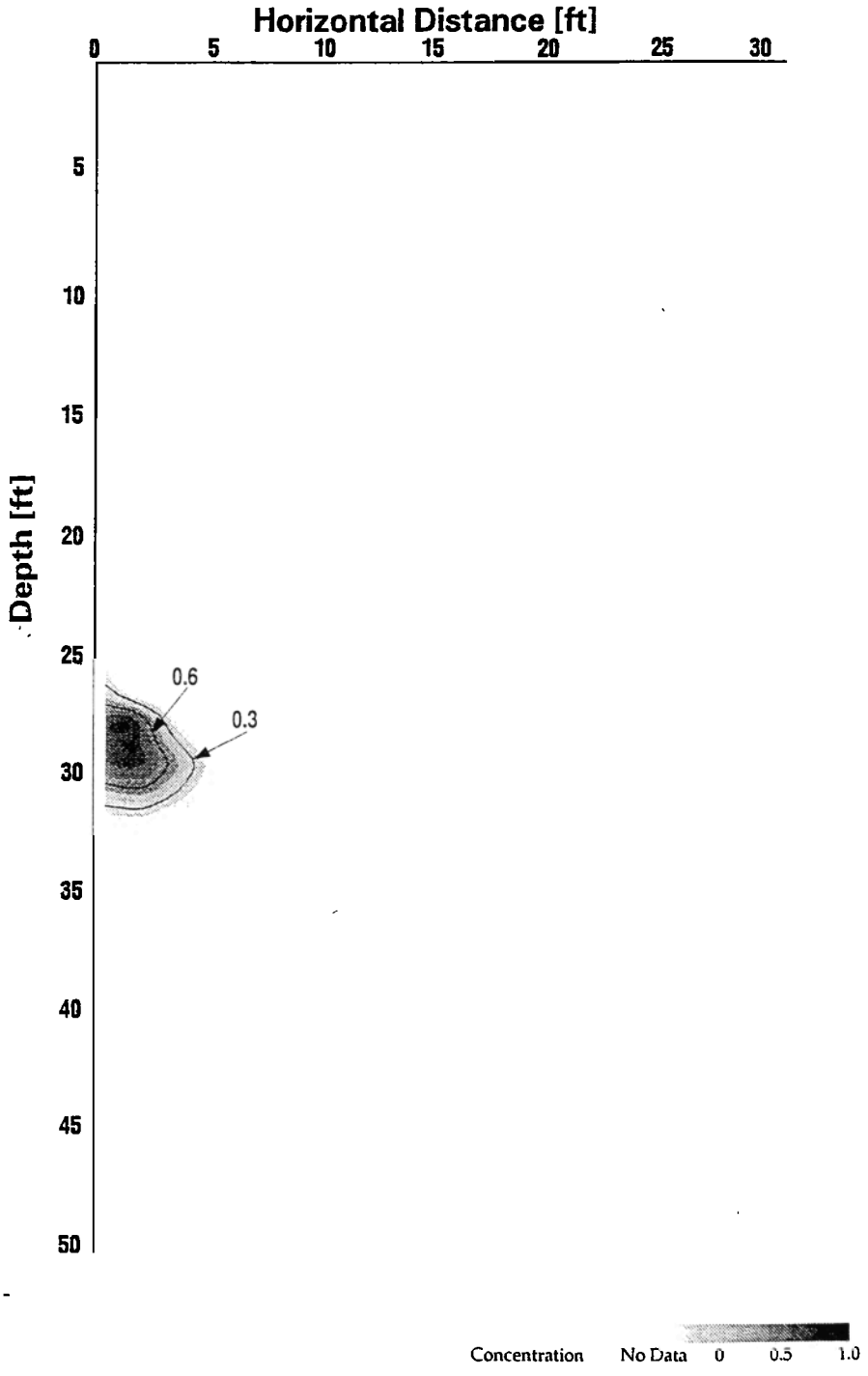

Figure 5. Transport of ^{137}Cs from simulations after 31 years.

though the advection-related input parameters caused greater changes than the dispersion-related input parameters, there was still very little effect. These results suggest that the assumptions and limitations regarding advection-related modeling parameters would have relatively little impact on the model output.

The equilibrium distribution coefficient, K_d, was the parameter that most affected the predicted movement of ^{137}Cs. Changing K_d across the entire range reported by Sheppard and Thibault for sandy soil caused the location of the wave front to change from 0.5 feet (0.2 m) to 41.7 feet (12.7 m). The reason that the K_d had such a dramatic impact in the sensitivity analysis is probably related primarily to the large range (greater than 3 orders of magnitude) of Cs-137 K_d values assumed for this study. The importance of the equilibrium distribution coefficient is not surprising. This coefficient is a measure of how the contamination is sorbed onto the soil and is a direct indication of how the soil retards the contaminant migration.

3.4. Model Calibration and Evaluation

The field data were normalized for comparison to the calibrated model output. Table 4 shows the field data as they have

Table 3. Sensitivity of 5-year, 5% Wave Front to Modeling Parameters

	K_{s_x}	K_{s_x}/K_{s_y}	α_L	α_T	D_m	K_d
Distance at low end of range, feet	1.9	1.5	2.9	3.0	3.0	0.5
Distance at high end of range, feet	3.4	3.0	3.0	3.0	3.0	41.7

Baseline distance equals 3 feet. One foot equals 0.3048 m.

Table 4. Normalized Field Data

Depth, feet	Transport Distance, feet	Concentration, pCi/g	Normalized Concentration, pCi/g
37.5	1.5	0.14	0.98
40	4	0.12	0.84
45	9	ND (≤0.04)	≤0.28

One foot equals 0.3048 m.

been transformed for comparison to the normalized concentration output of the model. The normalized transport distances were found by subtracting 36 feet (11 m) from the sample depths to account for the depth of the bottom of the drain field. The concentrations were divided by 0.143 pCi/g, which is the highest existing concentration extrapolated from the field data. The normalized field data thus vary from 1.0 to 0.28 (the 0.04 pCi/g detection limit), as depicted in Figure 6. The vertical arrows on the field data represent analytical uncertainty. Unfortunately, the 0.04 pCi/g detection limit does not lend itself to exact detection of the contamination wave front. The contamination profile was visually extrapolated to yield an approximate wave front of 11 feet (3.4 m) (Figure 6).

K_d, the most critical parameter as determined by the sensitivity analysis, was varied to calibrate the model, with all other parameters held to their baseline values. A K_d of 1.3 was found to yield approximately the 11-foot (3.4 m) wave front, as shown on Figure 6. No statistical fit was justified based on the low number of data.

Some insight as to the actual transport processes may be gained by analyzing the final calibrated K_d value. Recall that the baseline transport input parameters were chosen to be somewhat conservative. For K_d a conservative value would be low, representing weak contaminant adsorption and deeper migration. The final calibrated K_d was 1.3, which is lower than the supposedly conservative baseline K_d (9.8) but still within the suggested range (0.18–350). Three possible reasons are offered for this difference:

1. Most likely, the energetic flushing below the drain field not only caused advective transfer of contamination through the soil but also moved soil particles. At its end of life the drain field collapsed and would no longer accept water, further evidence suggesting that the soil beneath had been washed away.
2. Another possibility for the low K_d may be the composition of the soils beneath the drain field. Soils with a higher percentage of organic content, silt, or clay are expected to retain more contamination [*Travis and Etnier*, 1981]. Soil at this site is characteristic of the region's arid climate and has very little organic content, although there is a small amount of clay. These "desert" sandy soils may not compare well to those sandy soils of the literature experiments. This possibility could be conclusively evaluated by measuring the local K_d values.
3. Finally, the simple linear adsorption model assumed may be incorrect. Recall that the linear model implies equilibrium-based kinetics, but the sorption could be sufficiently slow such that the contaminants migrate some distance before sorbing.

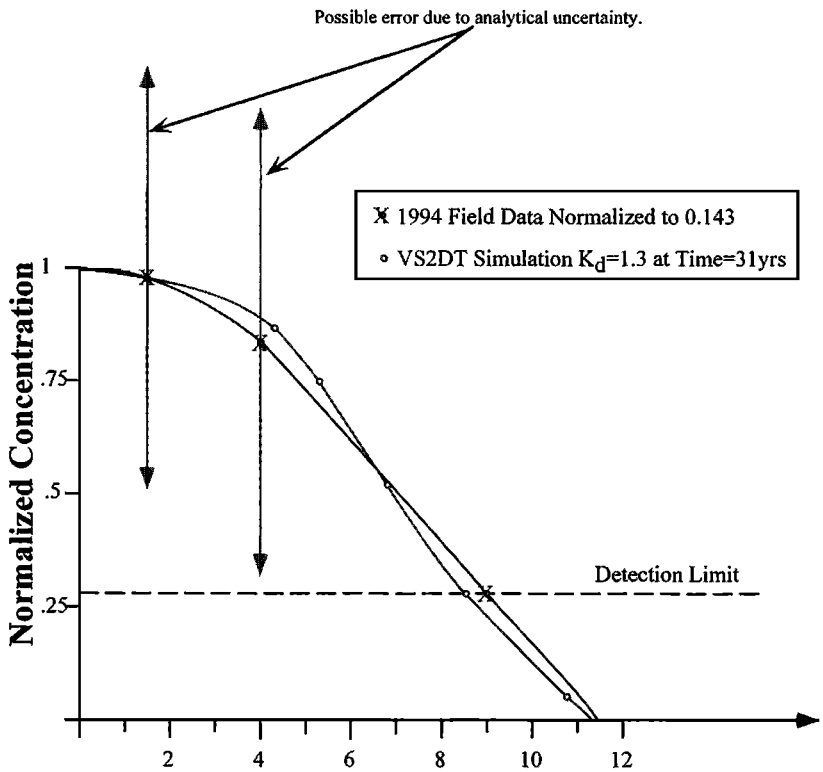

Figure 6. Comparison of normalized field data to calibrated model predictions.

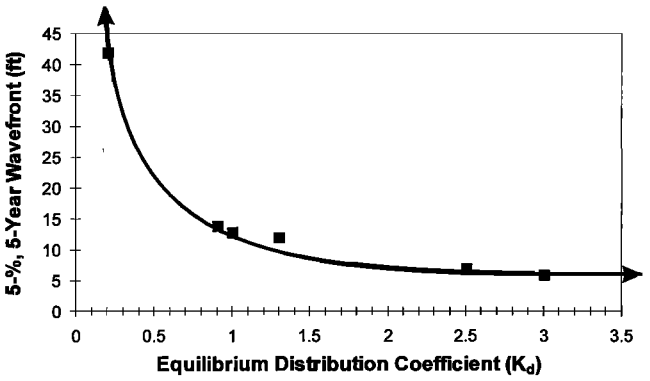

Figure 7. Contaminant transport as function of K_d value for 5-year simulations.

3.5. High Flow Rate Simulations

The drain field is constructed of sections of cement pipe that are 6–8 feet (1.8–2.4 m) long. Infiltrating water from the drain field was modeled as being evenly distributed to the underlying soil along the entire drain field length and highest diameter. In reality, this water may have been focused at the pipe joints, thus causing higher local infiltration rates than those derived assuming an even distribution. An additional simulation was completed to examine the effects of higher localized flow. The boundary condition fluxes, originally estimated for a 1-foot (0.3 m) length, were multiplied by 6 to account for the most likely length of drain field pipe. All parameters were held to their baseline values except for K_d, which was simulated at the calibration value of 1.3.

As expected, the higher flow rate caused greater contaminant migration. The 31-year, 5% wave front predicted for the higher flow rates increased to 17 feet (5.2 m). A plume length of approximately 11 feet (3.4 m) was achieved in another simulation by increasing K_d to 2.5. This is closer to, but still less than, the baseline K_d of 9.8. This work shows that higher flow could, in part, account for the contaminant plume moving to a greater depth than predicted by the baseline simulation.

3.6. Additional Drain Field Contaminants

In the course of the model sensitivity analysis and calibration, multiple simulations were completed with a range of K_d values. These simulations are summarized in Figure 7, which shows the 5-year transport distances predicted for K_d values ranging from 0.18 to 3.0. Simulations conducted without contaminant decay showed that the transport distances for 5 and 31 years would be virtually identical because the migration due to unsaturated flow in this time frame is negligible.

Figure 7 allows the transport study to be applied to other drain field contaminants. Figure 3a shows the contaminant profile for cadmium measured beneath the drain field and a plume length of approximately 19 feet (5.8 m). Based on the relationship depicted in Figure 7, a similar result would be produced with cadmium K_d of 0.6. From the experimental compendium [*Sheppard and Thibault*, 1990], the mean cadmium K_d is 2.8, and the range is 0.1 to 21.9.

Not surprisingly, the cadmium migration distance beneath the drain field is farther (thus implying lower K_d) than that predicted using mean values obtained from the literature search. This finding is consistent with the previous analysis for ^{137}Cs.

3.7. Potential Risks to Human Health

Exposure from the drain field contaminants must be considered from two pathways: direct exposure at the ground surface and groundwater contamination. The calibrated model for ^{137}Cs transport indicates that at the end of the 5-year infiltration period, the contaminant plume is at its maximum size. A simulation completed with no radioactive decay showed no detectable transport after 5 years. Although unsaturated flow continues, adsorption of ^{137}Cs allows no significant migration. With no infiltration source, radioactive decay causes the plume to shrink. The steady decay of the ^{137}Cs plume coupled with the approximately 500 feet (152.4 m) to groundwater suggests that groundwater contamination will not occur.

The low concentrations of radioactive contaminants do not pose a significant threat to human health at the surface because of the greater than 30 feet (9.1 m) of soil cover. An attempt to calculate the soil activity that would yield 15 mrem/yr to an occupationally exposed individual at the surface could not be achieved because of the shielding from the soil. The negligible risk is especially evident considering the current regulatory standard of 30 mrem/yr total effective dose equivalent (TEDE) [*Yu et al.*, 1993].

4. Conclusions

The LWDS drain field received approximately 6 million gallons (22,700 m^3) of waste water contaminated with 19.8 curies of radioactivity. The movement of water and of the contamination have been studied with the VS2DT computer program and verified with a field investigation. On the basis of this work, radioactive contamination from the LWDS drain field poses no significant risk to human health.

Water infiltrating from the drain field moved hundreds of feet into the subsurface and may have reached the regional aquifer, some 500 feet (152 m) below. The initial flow modeling showed that, even under very conservative assumptions, infiltrating water is predicted to have traveled to the regional water table before the end of the drain field's operational history in 1967. However, under any reasonable condition, this water is not predicted to have moved the radioactive contamination more than about 6 feet (1.8 m). The groundwater pathway is thus removed as a potential exposure source. The more than 30 feet (9.1 m) soil covering the drain field effectively blocks all direct exposure from existing contamination.

The risk assessment is strengthened by the consistent results produced by the various simulations. The absence of significant downward migration was shown not only in the most conservative analysis but also in all the simulations. Advective transport by the infiltrating water was shown to be the dominant transport mechanism. Unsaturated flow still continues following the 5-year infiltration period, but its contribution to contaminant migration is insignificant. Following the infiltration period, radioactive decay causes the plume to shrink. For nonradioactive contaminants the migration distance is also at an approximate maximum at 5 years and is essentially stationary afterwards.

The migration distances for all contaminants studied were consistently higher than those expected using published K_d values. Several reasons could account for the increased transport. Recommended follow-up work consists of experimentally measuring K_d values using local soils. The benefits of this work would be twofold. First, the exact reason for the higher-than-expected migration distances would be evident. Second, K_d

was shown to be the most important input parameter in this application. Accurate K_d values would reduce the uncertainty in any similar modeling because the general understanding of where local K_d values lie in relation to literature could be extrapolated to other contaminants.

As mentioned above, the main scientific and safety issue for states in siting low-level waste disposal facilities is to address potential contamination from radioactive waste disposal and the need for good scientific data and predictions on migration of radionuclides in the subsurface. In this and other recent articles [*Antonopoulos-Domis et al.*, 1995; *Monte*, 1996], the understanding of ^{137}Cs migration is improved by comparison of modeling results with field data. The low migration of ^{137}Cs, especially for arid environments, should lessen concerns about breaches in containment barriers for radioisotopes which have similar characteristics of cesium.

Acknowledgments. This work was supported by the United States Department of Energy under contract DE-AC04-94AL85000. Sandia is a multiprogram laboratory operated by Sandia Corporation, a Lockheed Martin Company, for the Unites States Department of Energy.

References

Antonopoulos-Domis, M., A. Clouvas, A. Hiladakis, and S. Kadi, Radiocesium distribution in undisturbed soil: Measurements and diffusion-advection model, *Health Phys.*, *69*(6), 949–953, 1995.

Dawson, L. A., Results of the Liquid Waste Disposal System RCRA facility investigation, Sandia Natl. Lab., Albuquerque, N. M., 1995.

Healy, R. W., Simulation of solute transport in variably saturated porous media with supplemental information on modifications of the U.S. Geological Survey's computer program VS2DT, *U.S. Geol. Surv. Water Resour. Invest. Rep. 90-4025*, 1990.

Lappala, R. W., R. W. Healy, and E. P. Weeks, Documentation of computer program VS2D to solve the equations of fluid flow in variably saturated porous media, *U.S. Geol. Surv. Water Resour. Invest. Rep. 83-4099*, 1987.

Monte, L., Analysis of models assessing the radionuclide migration from catchments to water bodies, *Health Phys.*, *70*(2), 227–237, 1996.

Phelan, J., G. Neuman, and N. Durand, Variation of soil hydraulic properties within a profile, *Soils, CE 447 Class Proj.*, Univ. of N. M., Albuquerque, 1991.

Sandia National Laboratories, Liquid Waste Disposal System RCRA facility investigation workplan, *Plan 92-12*, rev. 1, Albuquerque, N. M., 1993.

Sandia National Laboratories, Site-wide hydrogeologic characterization project calendar year 1994 annual report, 1995, Albuquerque, N. M., 1995.

Sandia National Laboratories and Ecology and Environment, Inc., Report of the phase I RCRA facility investigation of the mixed waste landfill, Albuquerque, N. M., 1990.

Sheppard, M. I., and D. H. Thibault, Default soil solid/liquid partition coefficients, K_ds, for major soil types: A compendium, *Health Phys.*, *59*(4), 471–482, 1990.

Travis, C. C., and E. L. Etnier, A survey of sorption relationships for reactive solutes in soil, *J. Environ. Qual.*, *10*(1), 8–17, 1981.

U.S. Congress, Low-Level Radioactive Waste Policy Act of 1980, *Public Law 96-573*, 1980. (Amended as *Public Law 99-240*, 1985.)

van Genuchten, R., Calculating the unsaturated hydraulic conductivity with a new closed-form analytical model, *78-WR-08*, Water Resour. Prog., Dep. of Civ. Eng., Princeton Univ., Princeton, N. J., 1978.

Yu, C., et al., A manual for implementing residual radioactive material guidelines, *ANL/ES-160 DOE/CH-8901*, Environ. Assess. Div., Argonne, Ill., 1993.

L. A. Dawson and P. I. Pohl, Sandia National Laboratories, Mail Stop 720, Albuquerque, NM 87185-0720. (e-mail: pipohl@sandia.gov)

(Received February 6, 1997; revised May 20, 1997; accepted June 8, 1997).