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Landfill Performance Assessment at a Semiarid Site: Modeling and Validation

by Daniel B. Stephens and Larry M. Coons

Abstract

The HELP model (Version 2.05) was applied to simulate the long-term percolation from a proposed landfill in southern New Mexico. The model predicted percolation would be about 0.0012 in/yr (10^{-10} cm/s). This result compared very favorably with independent estimates of recharge at the site which used the chloride mass balance method and hydrogeologic properties. The recharge estimates at this site are also quite similar to values obtained at other sites in New Mexico and west Texas. The long-term percolation through a closed landfill at this site is very small and would be nearly 1000-fold smaller than the saturated hydraulic conductivity of a typical clay liner.

Introduction

Before permitting solid waste landfills, the U.S. Environmental Protection Agency (EPA) or the authorized states generally require an assessment of the seepage or percolation rate beneath the landfill, in accord with the federal standards and regulations for solid waste (National Archives and Records Administration 1992). This seepage analysis evaluates landfill designs and predicts impacts to ground water quality. Numerical and analytical models are more commonly being used to predict seepage rates from landfills. Development of one such model was sponsored by the U.S. EPA: Hydrologic Evaluation of Landfill Performance (HELP) (Schroeder et al. 1984). This model allows for lateral and vertical flows within the cover and liner systems, and it routes water vertically in the vadose zone below the landfill if the soil water content exceeds the so-called field capacity of the soil.

The predictions of any model achieve validity through field results. Unfortunately, in arid and semiarid areas there are few published field results of landfill performance that can be used to validate the models. This may be due in part to the absence of significant leachate generated to date by landfills in dry climates. Peyton and Schroeder (1988) ran the HELP model to simulate 17 landfill cells from 6 sites in California, Kentucky, and Wisconsin. None of these sites are in a semiarid climate. Although we concluded that the HELP model generates reasonable water balance results, percola-

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tion through the landfill clay liner was only measured at the Kentucky site where the HELP model overpredicted percolation by approximately 35 percent. Nichols (1991) evaluated the HELP model by comparing it to a more physically based code called UNSAT-H for a landfill design in the semiarid environment of Hanford, Washington. Nichols found that HELP predicted more deep percolation of water below the root zone than did UNSAT-H. Because of the manner in which HELP deals with unsaturated flow, Thompson and Tyler (1984) found that the early version of UNSAT-H (UNSAT1D) produced more respresentative results than the early version of HELP (Version 1.0) for semihumid and arid (e.g., Phoenix, Arizona) climates.

This paper will present a case study on applying the HELP model to predict deep percolation from a proposed expansion of an existing solid waste landfill in southern New Mexico. We also present the results of field investigations designed to validate the long-term rate of leachate generation predicted by the HELP model.

Site Description

The site is located near Sunland Park, New Mexico, along the U.S.-Mexico border about 2 miles (3 km) northwest of El Paso, Texas, and 40 miles (64 km) south of Las Cruces, New Mexico (Figures 1 and 2). The site is in the Chihuahuan Desert physiographic province; the climate is semiarid. Mean monthly temperatures range from about 44 F (7 C) in January to 82 F (28 C) in July. In nearby El Paso, Texas, the mean annual precipitation is about 8 inches (20 cm) and the potential evapotranspiration is about 50 inches (127 cm). The sparse vegetation at the site consists principally of creosote bush.

The site is situated on the western edge of the valley of the Rio Grande in a cusp incised into the La Mesa escarpment (Figure 2). The topography of the landfill area gently slopes to the northeast at an average of about 300 ft/mile (57 m/km). There are a few broad, poorly defined drainage channels that traverse the site from southwest to northeast.

The site is underlain by unconsolidated alluvial sediments, including the Camp Rice and other Quaternary units of the Santa Fe Group. The depth to ground water is about 250 feet (76 m) below the site and the ground water gradient is about 0.002 to the northeast.

While the existing landfill occupies about 20 acres (8 hectares) at the site, the proposed new area would expand the landfill to about 500 acres (202 hectares). The proposed new landfill designs would place refuse in four lifts, each about 18 feet (6 m) thick. There would be three intermediate lifts to cover the refuse with 6 inches (15 cm) of compacted native sandy soil, in addition to the daily soil covers within the 18-foot-thick (6-m-thick) refuse lifts. A 24-inch-thick (61-cm-thick) final soil cover would be placed above the last refuse lift. The design would include berms to control run-on, as well as grading to divert runoff from the landfill. Figure 3 is a simplified conceptual model of the final landfill in vertical section.

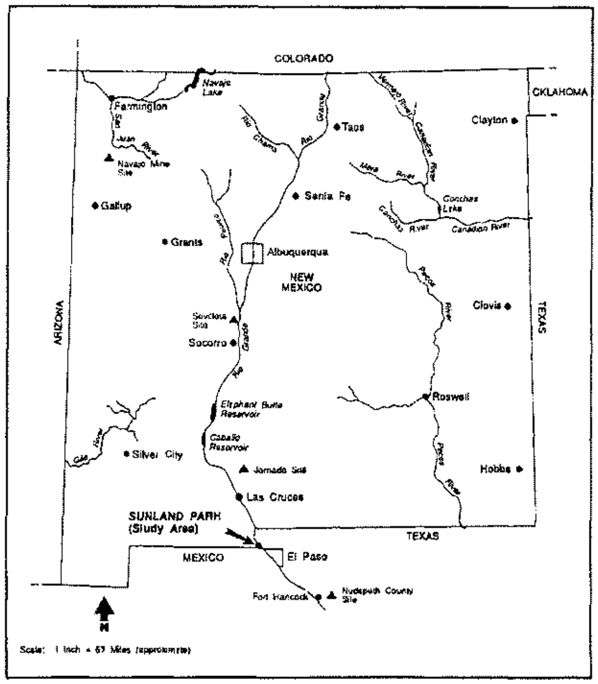

Figure 1. Location of study area in New Mexico.

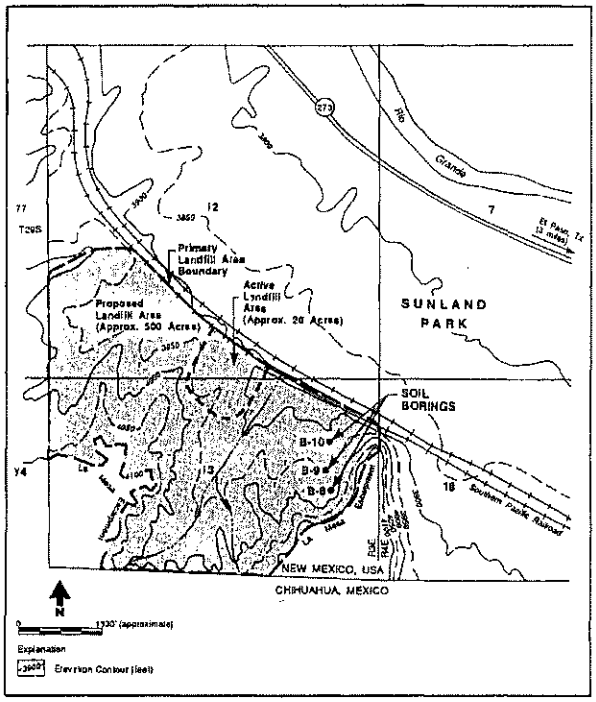

Figure 2. Location of study area near Sunland Park, New Mexico.

Approach

To support the application for a solid waste disposal permit, the New Mexico Environment Department asked the applicant to predict the rate of seepage that could develop from the proposed landfill with the HELP model. We applied HELP Version 2.05 (Schroeder et al. 1989) to the design shown schematically in Figure 3.

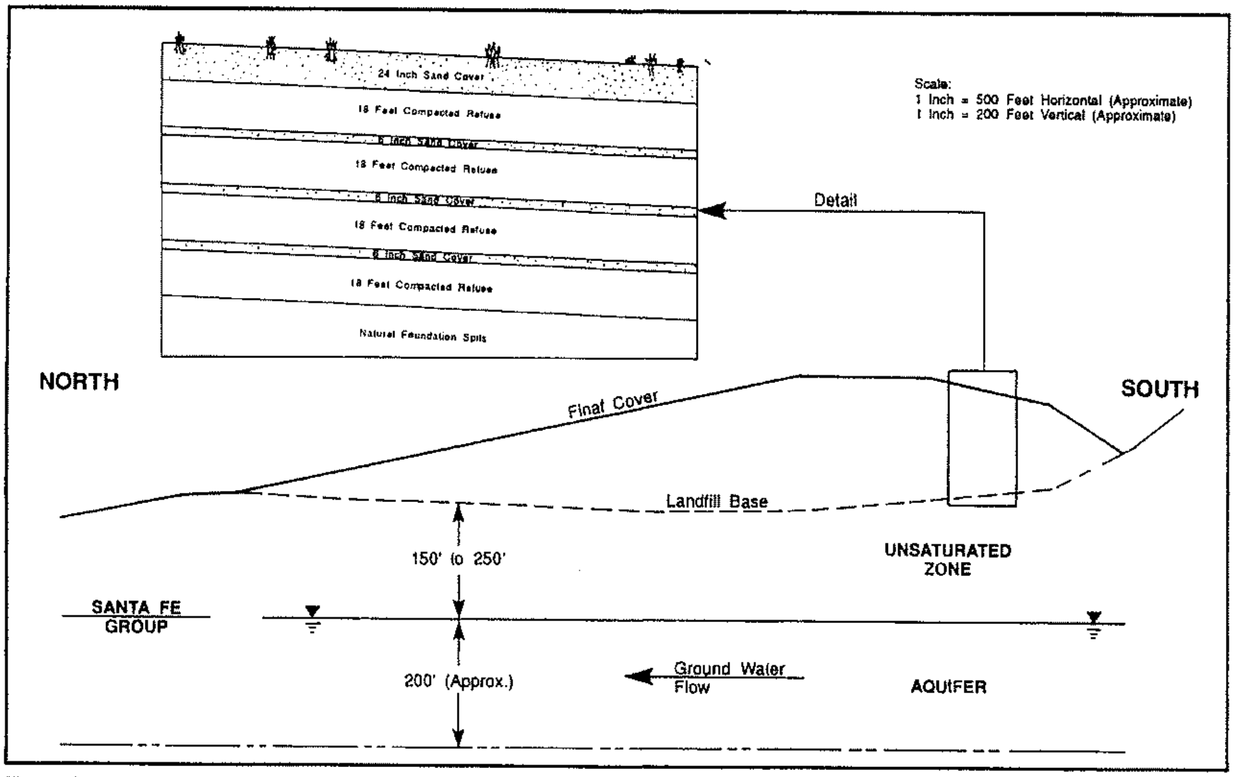

Figure 3. Simplified vertical section of final proposed landfill.

We assumed the landfill would operate for 80 years and during this time there would be 6 inches (15 cm) of compacted bare soil comprising the top layer. At the end of 80 years, the landfill would be capped with 24 inches (61 cm) of compacted soil. We also assumed that a poor grass cover would be established on the capping layer during the closure period. The input data for the HELP model are summarized in Table 1.

Because there were no field monitoring data available to evaluate seepage from the existing landfill, there were no means of validating the predictions from the model with a site-specific data set. An attempt was made to partially validate the model by comparing the model-predicted deep percolation component of the water balance equation to the long-term diffuse areal recharge at the site under natural conditions.

Natural recharge was quantified for this site using the chloride mass balance method described by Allison and Hughes (1978). This method has been widely applied in semiarid climates (e.g., Stone 1984; Phillips et al. 1987; Dettinger 1988; Scanlon and Richter 1990). The method recognizes that the principal source of chloride ions in the soil water is due to chloride in precipitation. As water infiltrates, chloride accumulates in the soil and is concentrated by evapotranspiration within the root zone. At equilibrium, the mass rate of chloride input to the soil from precipitation will equal the mass rate of chloride output of water percolating below the root zone:

$$R = (Cl_p/Cl_{sw}) \times P \tag{1}$$

where:

R = Recharge rate (L/T)
Cl_p = Chloride concentration in precipitation (M/L^3)
Cl_{sw} = Chloride concentration in soil water (M/L^3)
P = Average annual precipitation (L/T).

In this model, precipitation rate, chloride in the precipitation, and evapotranspiration rates are assumed to represent long-term averages; hence, the flow and transport are steady state. Application of the chloride mass balance method assumes that water percolating below the root zone eventually becomes recharge. Under ideal conditions, chloride would be expected to increase in concentration with increasing depth and reach a constant value below the depth where neither evaporation nor evapotranspiration are significant.

For our analysis of recharge by the chloride mass balance method, we considered 0.34 mg/L representative of the concentration of chloride in precipitation in New Mexico (Stone 1984; Phillips et al. 1987). Average annual precipitation was assumed to be 8 in/yr (20 cm/yr). Soil chloride concentration was determined from 50-foot (15-m) soil cores collected from three borings located in the southeast section of the proposed landfill at undisturbed topographic settings considered representative of the site (Figure 2).

The soil cores were collected in the field using a hollow-stem flight auger rig and split-barrel sampler.

Table 1
HELP Model Input Data Summary

Feature	Measurement	Notes
Site Description		
Thickness	75 feet	Average after 80 years of operation
Top layer	6 inches 24 inches	During operations (0–80 years) After closure
Refuse	18 feet	Four layers
Intermediate life cover	6 inches	Three covers
Base layer	6 inches	Natural soil
Layer type	—	Vertical percolation (no lateral drainage or barrier layers)
Climatic data	—	El Paso, Texas (model-generated, 20-year sets of evaporation and precipitation data)
Surface conditions, during operations	6 inches 18 inches 0.95	Compacted sand, bare ground cover Evaporative depth (default value) Potential runoff fraction (exposed working face)
Surface conditions, after closure	24 inches 24 inches 1.0	Compacted sand, poor grass cover Evaporative depth (default value) Potential runoff fraction (final cover in place)
SCS Curve No.	82	SCS soil type is Doña Ana (hydrologic soil group B); poor range conditions; lifts are compacted to an estimated 90–95% standard proctor density with sheeps-foot roller; 4–12% slope is maintained for drainage
Initial Cap, Cover, and Native Material Conditions		
Porosity	0.357	Default value for loamy fine sand
Field capacity	0.113	Default value
Initial moisture content	0.072	Wilting point value plus 25% of the difference between wilting point and field capacity. Typical starting point is wilting point plus 10–20% of the difference (Shroeder 1991)
Wilting point	0.058	Default value
Saturated hydraulic conductivity	7.5×10^{-4} cm/s	From remolded samples tested in the laboratory
Initial Refuse Conditions		
Porosity	0.520	Default value
Field capacity	0.290	Default value
Initial moisture content	0.178	25% of difference value
Wilting point	0.140	Default value
Saturated hydraulic conductivity	2×10^{-4} cm/s	Default value

Samples were placed in 4-ounce (114-cm^3) jars that were capped and taped to prevent evaporation of soil water. Upon arrival in the laboratory, the samples were weighed and oven dried to determine gravimetric water content. After cooling, 50 mL of deionized water was added to each sample. The samples were mechanically shaken for four hours and then were filtered. The chloride concentration of the decant liquid was determined by ion chromatography, and the concentration of chloride in the soil water was calculated after correcting for the deionized water additions and field water content.

Hydraulic properties of selected soil core samples were measured in our in-house laboratory to check if the recharge calculation was reasonable. Laboratory analyses included gravimetric moisture content by oven drying, soil-water potential by psychrometry, porosity calculated from dry bulk density, saturated hydraulic conductivity in falling and constant head permeameters, moisture retention from desorption on a pressure plate with psychrometry, and particle size by sieving and hydrometer tests.

The moisture retention and saturated hydraulic conductivity were used to calculate the unsaturated hydraulic conductivity with the van Genuchten three-parameter model of the Mualem equation (van Genuchten 1980).

The soil hydraulic properties were used to estimate the recharge rate by calculating the Darcy velocity in the vadose zone. The Darcy velocity is the product of the unsaturated hydraulic conductivity at the in situ volumetric moisture content and the hydraulic gradient. Because our data set was not sufficient to calculate the hydraulic gradient, we assumed the gradient was unity in the vertical downward direction. A unit downward gradient simply represents a gravity-dominated flow field.

"water fall" gradient

Results

The HELP model predicted that deep percolation from the landfill would be approximately 5×10^{-4} in/yr (4×10^{-11} cm/s) after 80 years of operation (Figure 4). After closure of the landfill and placement of a final, 24-inch-thick (61-cm-thick) vegetated cover, the predicted percolation continues to increase, reaching a maximum of 0.0084 in/yr (6.8×10^{-10} cm/s) nearly 1200 years after operations began. Following this period of moisture drainage from the landfill, HELP model results (Figure 6) indicate that for approximately the next 3000 years, the percolation rate decreases and eventually equilibrates at 0.0027 in/yr (2.2×10^{-10} cm/s). In the overall water balance, the deep percolation that would become recharge represents approximately 0.034 percent of the mean annual precipitation.

The chloride profiles for the three borings are shown in Figures 5, 6, and 7. These figures also show the moisture content profiles and geologic logs of the borings. All three borings exhibit strikingly similar features. The soil is mostly sand with gravel, but there is a distinct

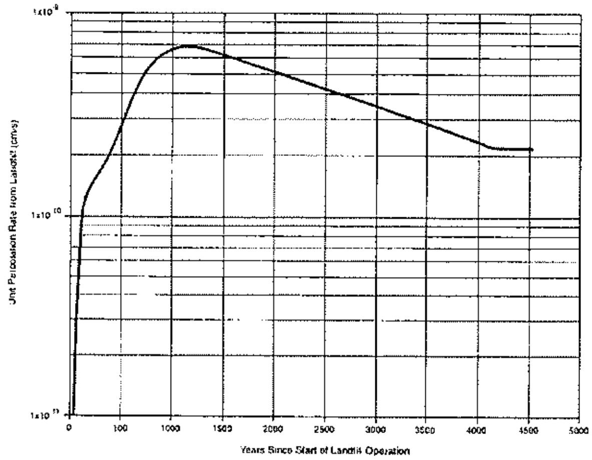

Figure 4. Landfill percolation calculated by the HELP model.

clay layer at least 3 feet (0.9 m) thick at depths of 28 to 35 feet (8.5 to 11 m) below land surface. In the sand above the clay layer, gravimetric moisture content is only about 2 percent, but in the clay layer the moisture content is about 15 to 18 percent by weight. In each boring, the chloride concentration ranges from 130 to 320 mg/L within the upper 6.5 feet (2 m) of the profile. The chloride concentration increases below this depth to a maximum of about 3000 mg/L at about 21 feet (6.4 m) below land surface. At greater depths, particularly in boring B-9 (Figure 6), the chloride concentration decreases and approaches a near constant concentra-

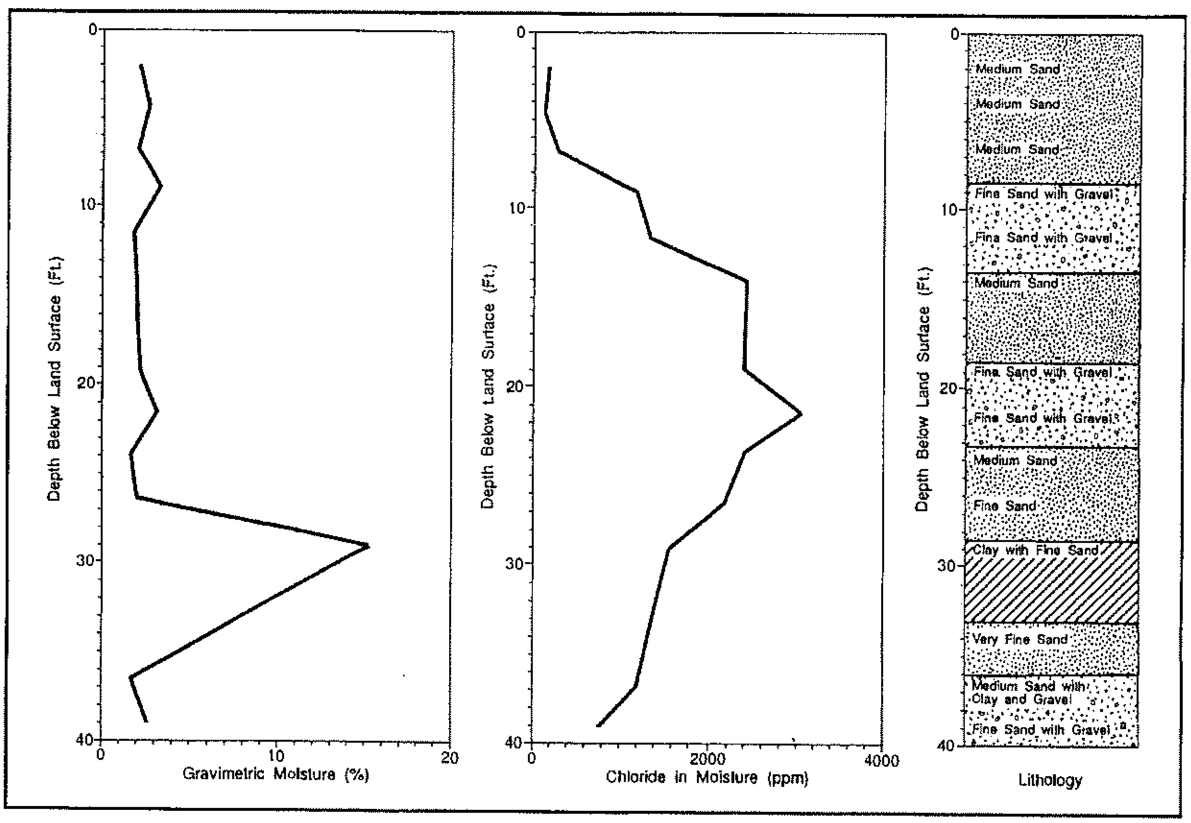

Figure 5. Profile of moisture content, chloride concentration, and lithology at soil boring B-8.

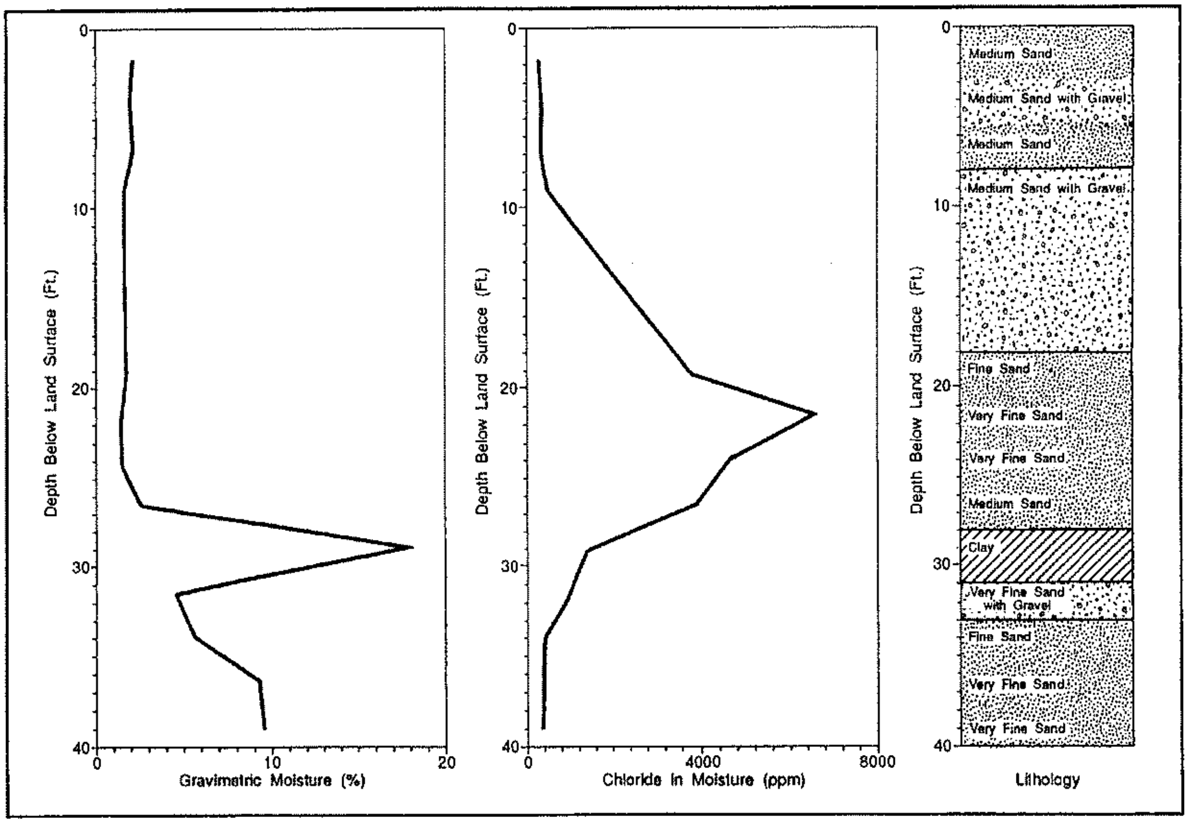

Figure 6. Profile of moisture content, chloride concentration, and lithology at soil boring B-9.

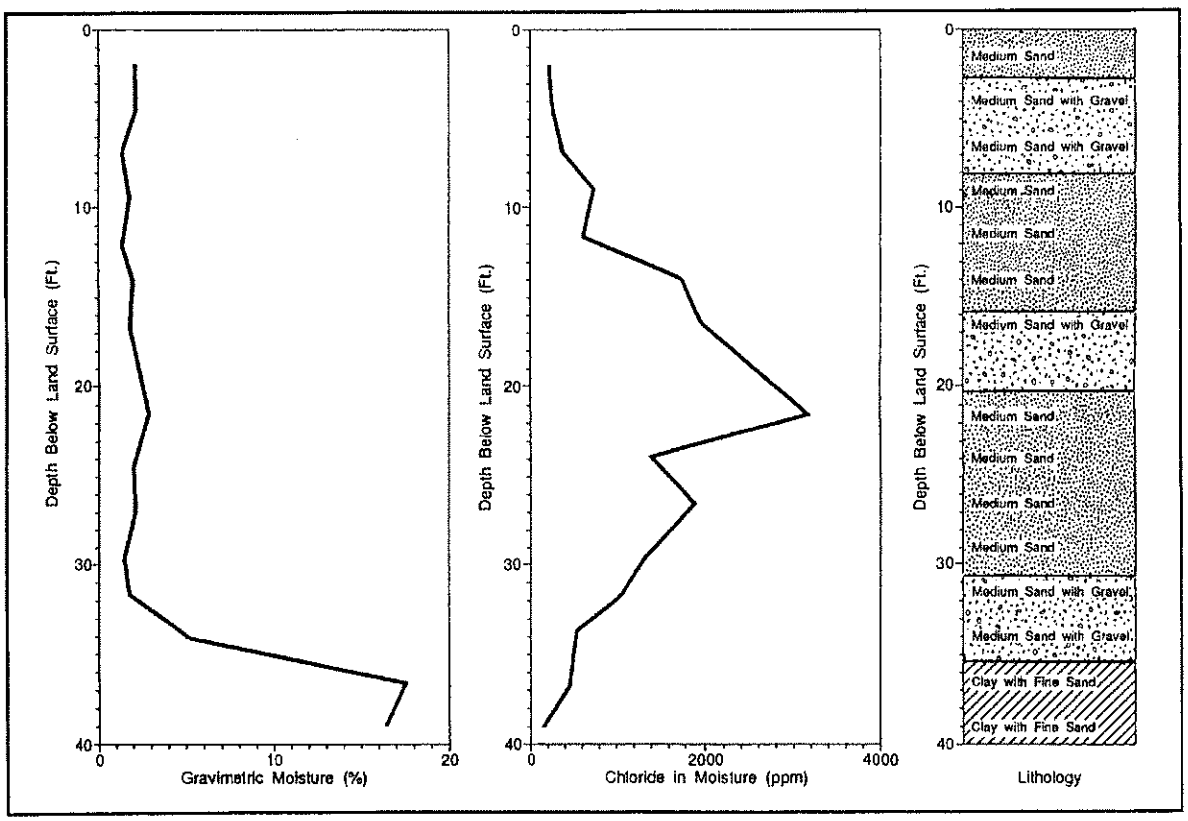

Figure 7. Profile of moisture content, chloride concentration, and lithology at soil boring B-10.

tion. The average chloride concentration of the three deepest samples in borings B-9 and B-10 is 353 mg/L and 377 mg/L, respectively. The corresponding recharge rates calculated from Equation 1 are 0.0077 in/yr (6.2×10^{-10} cm/s) and 0.0072 in/yr (5.8×10^{-10} cm/s). These results agree surprisingly well with the long-term deep percolation predicted by the HELP model.

Soil hydraulic properties were determined in the laboratory for four samples collected at the site (Table 2). The Darcy velocity was calculated from the laboratory data for two of the samples. Darcy velocities for samples B-8 and B-9 are 0.00022 in/yr (1.8×10^{-11} cm/s) and 0.1736 in/yr (1.4×10^{-8} cm/s), respectively. The geometric mean of the two values is 0.0062 in/yr (5.0×10^{-10} cm/s), which is also very close to the long-term infiltration rate calculated by the HELP model and is virtually the same rate of natural recharge estimated using the chloride mass balance method. Although the significant variability in the calculated Darcy velocity and the mean of the two values is not representative of the population mean, the sample mean result is consistent with other results and suggests that additional calculations of the Darcy velocity may be justified.

As a final check on the soil-water fluxes, we examined the saturated hydraulic conductivity of the clay layer in boring B-10 (Table 2). The saturated hydraulic conductivity of the clay sample is 0.0756 in/yr (6.1×10^{-9} cm/s), but the clay was only about 70 percent saturated when the sample was collected (Table 2). Therefore, the flux through the clay was much less than the magnitude of the saturated hydraulic conductivity, probably one or two orders of magnitude less. There is no evidence of increased moisture or perching which would be expected if the soil-water flux were equal to or greater than the saturated hydraulic conductivity of the clay. This analysis further substantiates a deep soil water flux under present conditions on the order of 0.0012 in/yr (10^{-10} cm/s) or less.

Discussion

The HELP model appeared to produce results that were consistent with field information and independent analyses of deep percolation. However, it is important to point out some of the uncertainties in selecting parameters for the HELP model. One of the uncertain parameters is the evaporative depth. This parameter controls in part the amount of moisture that can be stored in the soil. The greater the evaporative depth, the more moisture is available for evapotranspiration. The smaller the evaporative depth, the greater is the potential for deep percolation. In our analysis, we used an evaporative depth of 18 inches (46 cm) while the landfill was operating, and 24 inches (61 cm) after closure of the landfill and placement of a final cover. In desert climates with sandy soils, this depth seems reasonable. Extensive petrocalcic horizons (caliche deposits) occur beneath the La Mesa surface to the west and north of the site which are as much as 45 inches (114 cm) deep (Gile et al. 1981). Significant soil-water evaporation apparently occurs at least to this depth. It is also uncertain what role the presence of refuse below the final cover plays in effective evaporative depth. Therefore, our estimate of evaporative depth may be low, and the long-term deep percolation predicted by HELP may be conservatively high.

We also note that the HELP model uses the concept of field capacity to predict deep percolation. In the model, the moisture in the refuse or soil must exceed field capacity before deep percolation will occur. Field capacity is not a soil characteristic that is particularly well grounded in physics. It most commonly is defined as either the water content remaining in the field soil after two or three days following a thorough irrigation or the moisture content retained in a laboratory core at −1⁄10 or −1⁄3 bars of soil-water potential. It is widely recognized by soil physicists that soil water movement

Table 2
Summary of Laboratory Soil Hydraulic Properties

Sample[a]	Texture	Residual Moisture Content[e] (%)	Saturated Moisture Content[b] (%)	Calculated Porosity (%)	Saturated Hydraulic Conductivity (cm/s)	Initial Moisture Content[b] (% volume)	Potential[c] (−cm)	Unsaturated Hydraulic Conductivity[d] (cm/s)
B-8, 19-B	Fine sand	5.9	31.6	37.5	1.1×10^{-3}	3.9	53,000	1.8×10^{-11} [f]
B-9, 31.5-B	Very fine sand	11.7	39.0	38.2	1.6×10^{-4}	7.6	33,500	1.4×10^{-8} [g]
B-10, 9-B	Medium sand	5.0	32.8	38.1	2.0×10^{-3}	3.1	49,100	NC
B-10, 39-D	Clay	NM	NM	42.6	6.1×10^{-3}	30.5	24,100	NC
								$\bar{x} = 5 \times 10^{-10}$ [h]

[a] See Figures 5, 6, and 7. Depth of the sample below ground surface is indicated by the value following the comments.
[b] Volumetric moisture content.
[c] Potential at initial moisture content.
[d] Calculated unsaturated hydraulic conductivity at initial (in situ) volumetric moisture content using three-parameter van Genuchten analysis.
[e] Volumetric moisture content at −15,000 cm potential.
[f] $\alpha = 0.07206$; $N = 1.219$; residual volumetric moisture content = 2.75%.
[g] $\alpha = 0.00505$; $N = 1.728$; residual volumetric moisture content = 8.73%.
[h] $\bar{x}$ = Geometric mean value of samples B-8 and B-9.

occurs at moisture contents much less than field capacity (e.g., Hillel 1980). For example, in the native soils at our site we found that the soil water potential is less than −24 bars (Table 2), yet under these conditions, which are much drier than field capacity, it is clear that soil water has percolated to significant depths based upon chloride and caliche distributions. Consequently, unless field capacity is defined as equal to the specific retention (water content at which liquid transport is virtually nil), this concept is inappropriate to deal with infiltration and drainage through native soils. Additionally, with the field capacity concept in the HELP model, upward flow of soil water from below the evaporative depth is precluded. As a result, the HELP model may overestimate deep percolation. For other more detailed comparisons of HELP with more physically based models, refer to Thompson and Tyler (1984) and Nichols (1991).

It is important to recognize the limitations of models used to predict vadose zone processes and to make regulatory decisions. The advantage of the HELP model lies in its simplicity and ease of use for a variety of conditions, including those aspects other than deep percolation such as surface runoff and lateral flows within the landfill.

Despite these limitations of the model, the predicted deep percolation is realistic. Our chloride analysis and hydrogeologic data for the site certainly lend credibility to the HELP model. However, these are uncertainties in the analysis. In particular, the chloride data are not what would be expected under ideal conditions, as described previously. The chloride concentration reached a maximum at about 21 feet (6 m) and then decreased. If the profile were at equilibrium, the chloride concentration should remain constant at depths below 21 feet (6 m). We suggest that one possible explanation for this behavior is attributed to paleoclimatic changes. The chloride found at about 21 feet (6 m) may have entered the soil several thousand years ago or more, perhaps when the climate was even more arid or when vegetation was more aggressive in water uptake. However, it is also possible that the peak chloride accumulations are in part due to upward-flowing soil water which gradually evaporates or is transpired. Limited soil-water potential measurements (Table 2) indicate that the hydraulic head gradient is upward, at least within the upper 40 feet (12 m) of soil at the time of sampling. However, the field data are too limited to do more than speculate on directions of soil-water movement at this site. If the peak chloride concentration is used to compute recharge, then the recharge rate would be about 10-fold smaller than that computed from the deeper chloride measurements. On the other hand, if we used the lowest values of chloride in the profiles, the recharge rate would only increase three-fold from that computed from the average chloride in the three deepest soil samples. These values bracket the likely recharge rates estimated by the chloride mass balance method.

Although there may be some uncertainty in the recharge rates calculated from chloride, the recharge results from our site are consistent with those from research sites in the region (Table 3). This table shows that in general, diffuse areal recharge in several locations nearby is on the order of 10^{-11} to 10^{-9} cm/s. Note in particular the consistency between the results of Phillips et al. (1987) at the Jornada site about 50 miles (80 km) to the north of Sunland Park, New Mexico, and the results of Scanlon and Richter (1990) at the Hudspeth County site about 50 miles (80 km) to the southeast of El Paso, Texas.

Table 3

Comparison of Sunland Park Percolation Rates with Regional Percolation Rates

Location[a]	Percolation Rate (cm/s)	Method	Reference
Sunland Park, NM	2.1×10^{-10}	HELP model	This study
	6×10^{-10}	Chloride mass balance	This study
Navajo Mine, NM	1.6×10^{-10} to 8.1×10^{-10}	Chloride mass balance	Stone 1984
Sevilleta Wildlife Refuge, NM	8.1×10^{-10}	Chloride mass balance	Phillips et al. 1987
Hudspeth County, TX	3.2×10^{-7} to 3.2×10^{-9}	Chloride mass balance	Scanlon and Richter 1990

[a] See Figure 1.

The analyses presented herein and the results of recharge studies elsewhere suggest that long-term fluxes through landfills in this area would be very small. This conclusion is valid for properly designed and managed solid waste landfills that do not allow free liquids in the refuse or run-on to accumulate excess moisture in the landfill. Under these conditions, it is apparent that a clay liner compacted to a saturated hydraulic conductivity of 1.24 in/yr (10^{-7} cm/s) at the base of a landfill will not have a significant effect, if any, on reducing the long-term seepage rate at this site. Recall that a native clay layer beneath the site having a hydraulic conductivity of 0.0124 in/yr (10^{-9} cm/s) did not appear to impede deep soil-water movement. Flux of water from the landfill would have to increase by about 100-fold beyond the HELP model predictions before a clay liner would be effective. Even a 40-mL synthetic liner having a primary hydraulic conductivity of 1.24×10^{-4} in/yr (10^{-11} cm/s) would only be marginally effective in this environment for this landfill design (because of potentially inherent material and seaming defects in the liner). It is our opinion that the primary utility of the landfill liner will be during operations to collect liquid drainage from the refuse.

It is also important to bear in mind that we have assumed that, after the post-closure care period, the engineered landfill will perform at least as well as the native soil and vegetation in limiting seepage through the vadose zone. The combined effects of compacted capping materials, final grading, and re-established vegetation will likely cause less seepage to occur through the landfill than through native soils. It is difficult to identify many mechanisms that could cause the opposite to occur. One possible scenario in which the landfill could enhance seepage is if the refuse consolidates and the final cover collapses in such a way as to focus runoff and infiltration within the landfill. If there is potential for such an extreme problem to occur, a clay liner and drain system below the landfill could indeed significantly reduce the impacts to ground water. Otherwise, in areas of low precipitation, clay liners and subdrains may not have much effect on reducing seepage unless there are major climate changes.

Conclusions

1. The chloride mass balance method is a convenient method to evaluate long-term diffuse natural recharge.
2. The long-term diffuse natural recharge at a proposed landfill expansion in southern New Mexico is consistent with the flux through the bottom of the landfill predicted by the HELP model.
3. The HELP model and other predictors of landfill performance should be validated using actual, site-specific information from operating landfills in semi-arid climates.
4. In semiarid areas, clay liners may not appreciably reduce long-term, post-closure seepage, except for in extreme events.

References

Allison, L.B., and M.W. Hughes. 1978. The use of environmental chloride and tritium to estimate total recharge to an unconfined aquifer. *Aus. J. Soil Res.* 16: 181-95.

Dettinger, M.D. 1988. Reconnaissance estimates of natural recharge to desert basins in Nevada, U.S., by using chloride-balance calculations. *J. Hydro.* 106: 55-78.

Gile, L.H., J.W. Hawley, and R.B. Grossman. 1981. Soils and geomorphology in the basin and range area of southern New Mexico: Guidebook to the desert project. Memoir 39. Socorro, New Mexico: New Mexico Bureau of Mines and Mineral Resources.

Hillel, D. 1980. *Fundamentals of soil physics.* 1st ed. New York: Academic Press Inc.

National Archives and Records Administration. 1992. 40 CFR (Code of Federal Regulations), Subchapter I (Solid Wastes), Parts 240 through 259. Washington, D.C.: U.S. Government Printing Office.

Nichols, W.E. 1991. Comparative simulations of a two-layer landfill barrier using the HELP version 2.0 and UNSAT-H version 2.0 computer codes. Richland, Washington: Pacific Northwest Laboratories. Doc. No. PNL-7583.

Peyton, R.L., and P.R. Schroeder. 1988. Field verification of HELP model for landfills. *ASCE J. Env. Eng.* 114, no. 2: 247-69.

Phillips, F.M., J.L. Mattick, and T.A. Duval. 1987. Quantification of ground-water recharge rates in New Mexico using bomb-36 CL, bomb-34 and chloride as soil-water tracers. NMWRRI Tech. Completion Rept., Proj. Nos. 1423638 and 1423654. Las Cruces, New Mexico: New Mexico Water Resources Research Institute.

Scanlon, B.R., and B.C. Richter. 1990. Analysis of unsaturated flow based on chemical tracers (chloride, CL-36, H-3 and bromide) and comparison with physical data, Chihuahuan Desert, Texas. Bur. Econ. Geol., Univ. Texas at Austin. Prepared for the Texas Low-Level Radioactive Waste Disposal Authority, IAC (90-91) 0268.

Schroeder, P.R. 1991. Personal communication to M. Darr (DBS&A) regarding initial moisture content, 17 March, Albuquerque, New Mexico.

Schroeder, P.R., A.C. Gibson, and M.D. Smolen. 1984. The hydrologic evaluation of landfill performance (HELP) model. USEPA Doc. No. EPA/530-SW-84-010. Washington, D.C.: U.S. EPA, Office of Solid Waste and Emergency Response.

Schroeder, P.R., B.M. McEnroe, R.L. Peyton, and J.W. Sjostrom. 1989. The hydrologic evaluation of landfill performance (HELP) model, version 2.05. Vicksburg, Mississippi: U.S. Army Engineer Waterways Experiment Station, Environmental Laboratory.

Stone, W.J. 1984. Preliminary estimates of recharge at the Navajo Mine based on chloride in the unsaturated zone. NMBMMR Open-file Rept. 213. Socorro, New Mexico: New Mexico Bureau of Mines and Mineral Resources.

Thompson, F.L., and S.W. Tyler. 1984. Comparison of two groundwater flow models: UNSAT1D and HELP. Topical Rept. CS-3695, Research Project 1406-1. Richland, Washington: Battelle, Pacific Northwest Laboratories.

van Genuchten, M. Th. 1980. A closed-form equation for predicting the hydraulic conductivity of unsaturated soils. *Soil Sci. Soc. Am. J.* 44: 892-98.

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Biographical Sketches

Dr. Daniel B. Stephens *(6020 Academy Rd., NE, Ste. 100, Albuquerque, NM 87109), formerly chairman of the Geoscience Department at New Mexico Institute of Mining and Technology (NMIMT) in Socorro, New Mexico, began private consulting in 1976 and founded Daniel B. Stephens & Associates Inc. (DBS&A) in 1984. Stephens is the principal hydrologist and president of DBS&A, an adjunct professor of geology at the University of New Mexico in Albuquerque, and an adjunct professor of hydrology at NMIMT. He received his bachelor's degree in geological science from Penn State University, his master's degree in hydrology from Stanford University, and his doctorate in hydrology from the University of Arizona. Stephens, a certified Professional Hydrogeologist, has nearly 20 years experience in a variety of hydrogeologic problems ranging from site characterization to mathematical modeling of flow and transport.*

Larry M. Coons *is presently systems operations manager and senior engineer at Daniel B. Stephens & Associates Inc. (6020 Academy Rd., NE, Ste. 100, Albuquerque, NM 87109). Coons began private consulting in New Mexico in 1981 after receiving his bachelor's degree in geological engineering and master's degree in civil engineering from New Mexico State University. Coons is a registered Professional Engineer in New Mexico, Utah, Colorado, and Arizona and is a certified Professional Hydrogeologist. Coons has broad experience in hydrogeology and hydrological engineering, but specializes in design and performance assessment of waste repositories for municipal waste, radioactive waste, and mining and milling waste.*