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Mass Transport Modeling in Contaminated Buried-Valley Aquifer

By Paul F. Hudak[1] and Hugo A. Loaiciga,[2] Member, ASCE

Abstract: The U.S. Geological Survey's two-dimensional solute-transport and dispersion model (the method of characteristics, or MOC, model) accurately reproduces the chloride-concentration distribution observed in a buried-valley aquifer contaminated by a solid-waste landfill in Butler County, Ohio. The predictive capacity of the calibrated model has utility in water resources planning in regions of ground-water discharge downgradient of the landfill. Calibration of the ground-water-flow model to an observed steady-state head field indicates a ratio of hydraulic conductivity to diffuse recharge of 8.8×10^{-4}. Mass transport calibration was achieved with a longitudinal dispersivity of 53 m, a transverse dispersivity of 5.3 m, and a chloride concentration of 320 ppm at injection wells used in simulating the contaminant source. The model was calibrated to a contaminant distribution observed 16.7 years after initial landfill operations; 10 and 50 year contaminant-distribution predictions indicate that the plume will reach a steady-state equilibrium with the ambient ground-water flow system after 10 years. The attainment of such a condition could significantly decrease the possibility of future contamination of downgradient domestic supply wells.

Introduction

Significant advances have been made in the design of liners and leachate collection systems for new solid-waste landfills. These advances greatly reduce the potential for pervasive aquifer contamination from modern facilities. Older landfills (those constructed in the 1970s and earlier), however, continue to pose a significant threat to the ground-water environment. Aquifer contamination from percolating leachate from such landfills may be viewed as the inevitable consequence of insufficient and improperly constructed containment structures. In many instances the areal extent of contaminant plumes emanating from these older facilities have rendered effective cleanup nearly impossible. The potential for widespread contamination is enhanced in cases where the older facilities have been located above relatively permeable geologic deposits (e.g., glacial outwash). The massive leachate plumes originating from the Babylon and Islip landfills on Long Island, N.Y., are well-known examples of such pervasive contamination (Kimmel and Braids 1974, 1980). In cases where older solid-waste facilities have caused extensive contamination of productive aquifers, numerical simulation of contaminant transport may provide a useful tool in water resources planning and management. Existing plume geometry may provide a means for contaminant-transport model calibration. Model verification may be achieved by comparing model-calculated plumes with contaminant distributions observed at different times in the evolution of the leachate plume. A calibrated and verified model may then be used in a predictive capacity. Contaminant distributions derived in the predictive modeling phase may be used qualitatively

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to assess risk of contamination of downgradient water-supply wells.

The study described herein involves an application of the U.S. Geological Survey's two-dimensional contaminant transport model (MOC) (Konikow and Bredehoeft 1978) to a contaminated buried-valley aquifer in southwest Ohio. The model was used to simulate contaminant transport from an existing solid-waste facility. Model predictions were used to assess contamination potential for water-supply wells located downgradient from the landfill source.

BACKGROUND

Leachate generated from a solid-waste landfill in Butler County, Ohio (Fig. 1), has caused widespread contamination of highly permeable underlying buried-valley aquifer. The spatial distribution of chloride ion concentrations in ground-water samples taken from 12 monitoring wells located throughout

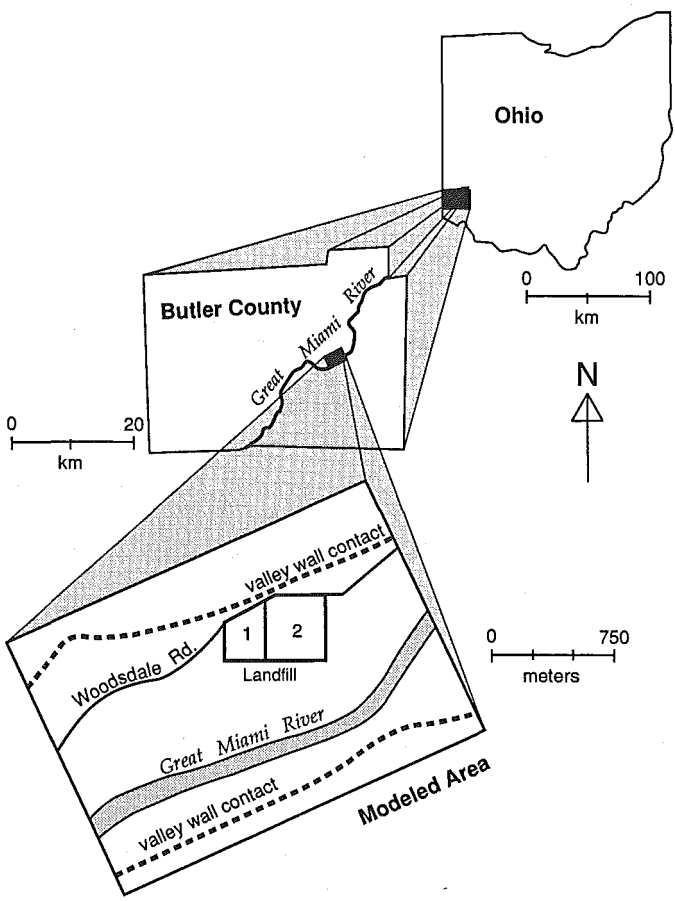

FIG. 1. Location of Study Area

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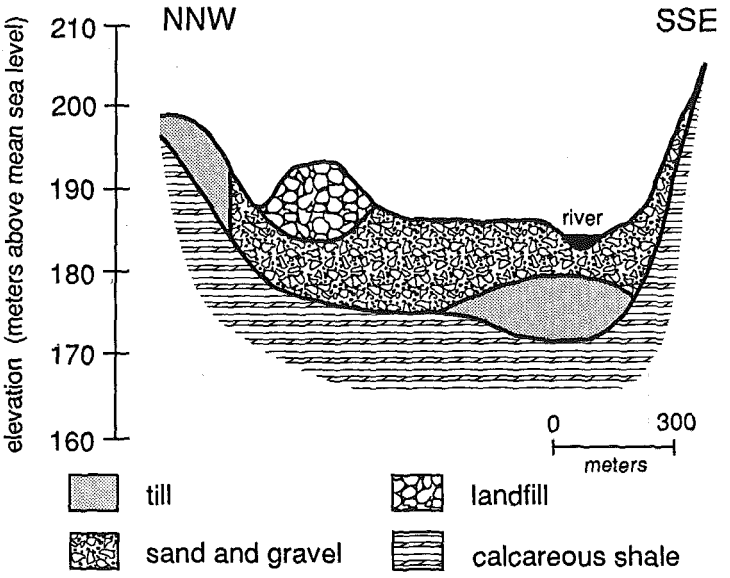

FIG. 2. Schematic Geologic Cross Section through Study Area

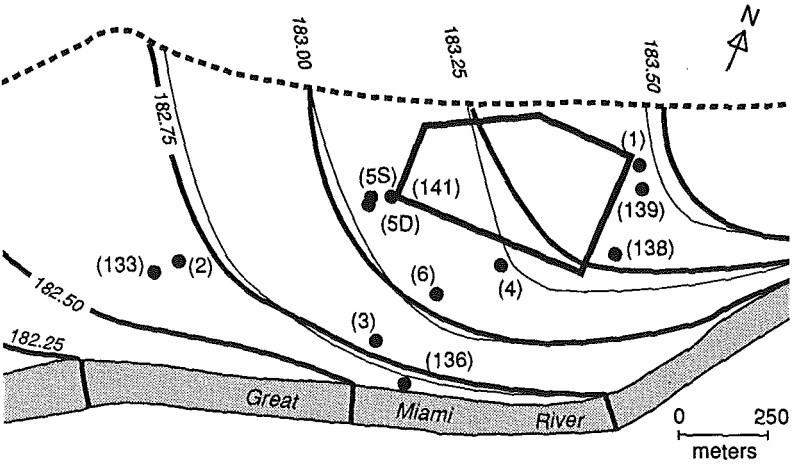

FIG. 3. Monitor-Well Locations (Black Circles), Observed Head Distribution (Thin Contour Lines), and Model-Calculated Steady-State Head Distribution (Thick Contour Lines)

the study area suggests that the contaminant plume is approximately 1,500 m long. The distal portion of the chloride plume, inferred from a set of samples collected on September 25, 1987, is approximately 300 m upgradient of the nearest downgradient domestic water-supply wells.

The solid-waste landfill is located in east-central Butler County, Ohio, on the floodplain of the Great Miami River (Figs. 1 and 2). The landfill received domestic and commercial waste from January 6, 1971, to June 1, 1985. Landfill development proceeded in two phases of operation. The phase 1 portion of the landfill (see area 1 in Fig. 1) was active from 1/6/71 to 4/3/75. Refuse was placed in the phase 2 area from 4/3/75 to 1/1/85. Landfill closure was recommended by the Ohio Environmental Protection Agency on the basis of several findings, including elevated levels of lead and arsenic in a downgradient monitoring well (well 141, Fig. 3) and the presence of leachate seeps on the sides of the landfill. (In Fig. 3 the contours are in meters above mean sea level. Monitor-well identification numbers are in parentheses.)

Hydrogeologic Setting

The landfill overlies a buried-valley aquifer. The aquifer consists of glaciofluvial deposits comprised predominantly of coarse sand and gravel. Silt and clay deposits are present in thin, laterally discontinuous lenses. Unconsolidated deposits range from 6 to 12 m in thickness and overlie shale bedrock. The regions to the immediate northwest and southeast of the study area (Figs. 1 and 2) are upland surfaces underlain by glacial till. Till deposits range from 1 to 12 m in thickness and directly overlie the shale bedrock. The shale bedrock and glacial till are characterized by low hydraulic conductivities (estimated less than 10^{-8} m/s). Approximate boundaries between the upland surfaces and the unconfined aquifer are represented by the "valley wall contacts" depicted in Fig. 1. No geologic boundaries are present transverse to the axis of the buried valley at the northeast and southwest margins of the study area. Topography is uniformly flat, with an average elevation of about 185 m above mean sea level. Depth to ground water characteristically ranges from 3 to 5 m. The river is effluent (gaining) and in hydraulic connection with ground water in the underlying aquifer.

Data Acquisition

The monitoring network from which head levels were measured and groundwater samples were collected is shown in Fig. 3. The network includes five 1.25-in. (3.2-cm) diameter steel drive-point piezometers (wells labeled with three digits) and seven 4-in. (10.2-cm) diameter PVC monitoring wells (wells labeled with one digit). The bottom 3.1 m of the PVC wells is screened. The wells sample ground water from intermediate depths within the saturated zone (i.e., within the 3–7-m depth interval below ground surface), with the exception of well 5D (screened over 7–10-m depth interval). River-stage measurements were made at a gage on the north bank of the river near well 136. Head level, chloride concentration, and river-stage data were collected biweekly during the period of mid-June to late-November, 1987. The sampling program was designed to provide enough contemporaneous sets of measurements to make a representative approximation of the steady-state head distribution and the areal distribution of the contaminant plume. A "contemporaneous" set of measurements is used herein to designate a set of measurements taken on the same day. Chloride ion concentrations were used as a conservative tracer for delineating the spatial extent of the plume. Nu-

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TABLE 1. Chloride Concentrations (ppm)

Date (1)	Well (Fig. 3) 1 (2)	2 (3)	3 (4)	4 (5)	5S (6)	5D (7)	6 (8)	133 (9)	136 (10)	138 (11)	139 (12)	141 (13)	Temporally averaged concentration (14)
5/29/87	27	—[a]	98	194	138	90	85	—[a]	—[a]	—[a]	—[a]	—[a]	22
7/16/87	22	79	66	181	61	96	66	36	68	22	19	36	78
7/31/87	22	81	78	201	125	90	50	50	109	27	24	130	104
8/14/87	25	88	95	166	149	102	46	48	101	25	24	146	198
9/8/87	17	73	69	75	76	51	16	34	72	17	13	89	149
9/11/87	21	81	123	120	146	80	66	47	75	23	21	110	91
9/25/87	21	77	115	210	146	91	72	55	77	20	21	128	68
10/9/87	24	74	114	193	134	88	56	47	87	21	21	130	48
10/21/87	23	72	122	252	263	97	58	—[a]	—[a]	—[a]	—[a]	149	81
10/23/87	18	75	116	220	144	89	75	48	65	20	18	129	23
11/6/87	21	76	117	244	181	88	105	51	63	24	21	123	21
													120

[a]No data collected.

merical simulations provide a worse-case scenario, because predicted plumes defined by chloride are likely greater in areal extent than plumes of toxic chemicals, which may be retarded as a result of sorption or chemical processes.

Steady-State Head Distribution

A "steady state" water table map was constructed from head measurements taken on 6/26/87 (Fig. 3), during a period of gradual baseflow recession, when water table fluctuations were minimal (less than 5 cm/week). These fluctuations are low relative to fluctuations observed during transient conditions following river flood wave passage and aquifer recharge during rainfall events (e.g., fluctuations in excess of 50 cm/week).

Contaminant Plume

Concentration contours for contaminant plumes were constructed from several contemporaneous data sets. For a given set of contemporaneous data, plume contours of equal chloride concentration were constructed on the basis of data collected at wells 1, 3, 4, 5S, 133, 136, 138, and 139 (Fig. 3). On the basis of depth interval of sampling horizon, these wells were assumed to provide samples representative of the center portion of the plume along a vertical transect at their respective locations. Some vertical variability in chloride concentrations was observed at well nest 5S-5D, located immediately downgradient of the landfill. Although the absence of monitor-well nests precluded quantitative assessment of vertical differentiation of chloride concentrations at locations more distal to the landfill, it is likely that vertical concentration gradients were less extreme there due to dispersion. Several studies attest to increasingly uniform vertical concentration profiles with increasing distance from a landfill [e.g., MacFarlane et al. (1983)].

From review of 11 contemporaneous sets of chloride concentrations, a representative sampling date was chosen. To assess which sampling date included concentrations that were most representative of the entire set, temporal averages were calculated for each well. The results are given in Table

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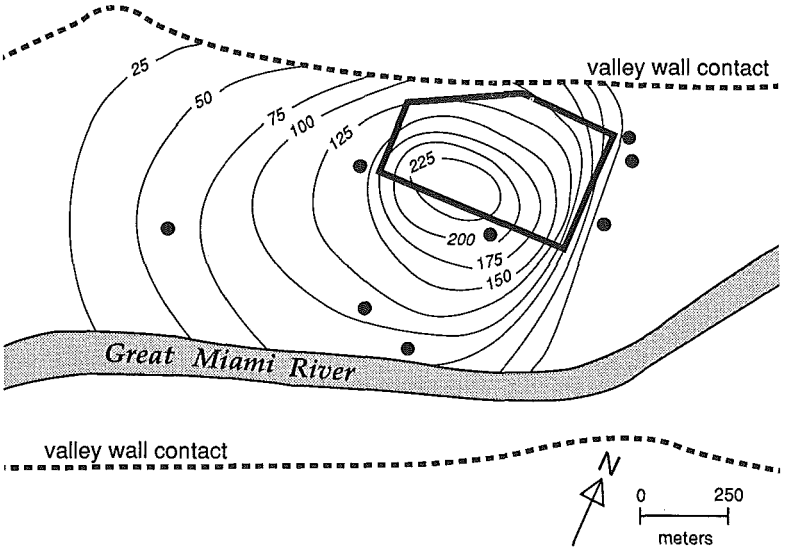

FIG. 4. Chloride Concentration Distribution Inferred from 9/25/87 Data (Contours in ppm); Black Circles Represent Control Points (see Table 1 for Data)

1. The contemporaneous set that best approximated the temporally averaged set was chosen by calculating, for each sampling date, the sum of the squares of the differences (SSD) of chloride concentration values between observed and temporally averaged data for corresponding wells. The 9/25/87 data set yielded the minimum SSD. The chloride-concentration distribution inferred from this data set is shown in Fig. 4.

Parameter Estimation

Falling-head piezometer tests were conducted at wells 1, 2, 3, 4, 5D, and 6 (Fig. 3). A pressure transducer was used to obtain frequent data during the tests. Data were analyzed using the Hvorslev (1951) technique. A summary of hydraulic conductivity (K) estimates are listed in Table 2. The relatively low value for well 2 is attributed to insufficient development of the well subsequent to installation. The average (arithmetic) piezometer test-determined K value, excluding well 2, is 1.2×10^{-3} m/s.

Additional K estimates were obtained by analysis of grain-size distribution

TABLE 2. Slug Test Results

Well (1)	Depth of screened interval (2)	Hydraulic conductivity (K) (3)
1	12–22 ft (3.7–6.7 m)	1.7×10^{-3} ft/sec (5.2×10^{-4} m/s)
2	8–18.5 ft (2.4–5.6 m)	2.9×10^{-5} ft/sec (8.9×10^{-6} m/s)
3	10–15 ft (3.0–4.6 m)	1.1×10^{-3} ft/sec (3.3×10^{-4} m/s)
4	8–18 ft (2.4–5.5 m)	9.2×10^{-4} ft/sec (2.8×10^{-4} m/s)
5D	22–32 ft (6.7–9.8 m)	4.6×10^{-3} ft/sec (1.4×10^{-3} m/s)
6	25–35 ft (7.6–10.7 m)	1.1×10^{-2} ft/sec (3.2×10^{-3} m/s)

TABLE 3. Results of Grain Size Analyses

Well (1)	Depth[a] (ft) (2)	Sample description[b] (3)	K[c] (4)	K[d] (5)	Porosity (6)
1	15–16.5	S, G	8.5×10^{-3}	9.4×10^{-3}	0.36
1	20–21.5	S, G	4.1×10^{-4}	1.4×10^{-3}	0.28
2	10–11.5	S, G	1.6×10^{-4}	1.1×10^{-2}	0.32
2	20–21.5	S, G	1.3×10^{-4}	1.1×10^{-2}	0.31
3	5–6.5	S	5.1×10^{-5}	5.2×10^{-4}	0.45
3	10–11.5	S, G	2.5×10^{-3}	6.6×10^{-3}	0.22
3	15–16.5	S, G	5.3×10^{-4}	2.5×10^{-4}	0.28
4	9–10.5	S, G	3.1×10^{-4}	1.0×10^{-3}	0.27
4	15–16.5	S, G	4.7×10^{-4}	1.4×10^{-3}	0.29
5	7.5–9	S, G	2.5×10^{-3}	4.7×10^{-3}	0.27
5	15–16.5	S, G	1.1×10^{-3}	2.8×10^{-3}	0.30
5	20–21.5	S, G	3.6×10^{-4}	1.4×10^{-3}	0.45
6	5–6.5	S, G	6.3×10^{-4}	2.1×10^{-3}	0.28
6	7.5–9	S, G	3.3×10^{-3}	5.6×10^{-3}	0.33

[a]1 ft = 0.305 m.
[b]S = sand; G = gravel.
[c]Hazen approximation (m/s) (1 m/s = 3.28 ft/s).
[d]Rose and Smith (1957) technique (m/s).

curves drawn from sieve data for representative split-spoon samples taken during the drilling of the wells at which piezometer tests were conducted. K estimates were made using the techniques of Hazen [see for example Freeze and Cherry (1979)] and Rose and Smith (1957). Results are summarized in Table 3. From the values summarized in Tables 2 and 3, a range of 1.5×10^{-4} to 5.0×10^{-3} m/s was established for K.

Model Representation

Ground-Water Flow

A 20 by 22 (rows by columns) finite-difference grid with a uniform nodal spacing of 114.3 m along rows and 76.2 m along columns was utilized in the simulation of a single, unconfined layer (Fig. 5). The aquifer was modeled as homogenous and isotropic with respect to hydraulic conductivity. Input saturated thickness was varied spatially. The modeled area corresponds to the region to the northwest of the river in Fig. 1.

No-flow boundary conditions were established along the valley-wall contact at the northwest margin of the study area. Boundaries at the northeast and southwest margins of the modeled region and the river were represented by constant-head cells. Head levels for these cells were estimated from the 6/26/87 steady-state head distribution. The locations of the no-flow valley-wall boundaries were determined from field observation, topographic contours, and hydrologic maps by Spieker (1968a).

Contaminant Transport

The governing equation for mass transport simulation with the advection-dispersion model is time-dependent and therefore requires initial conditions.

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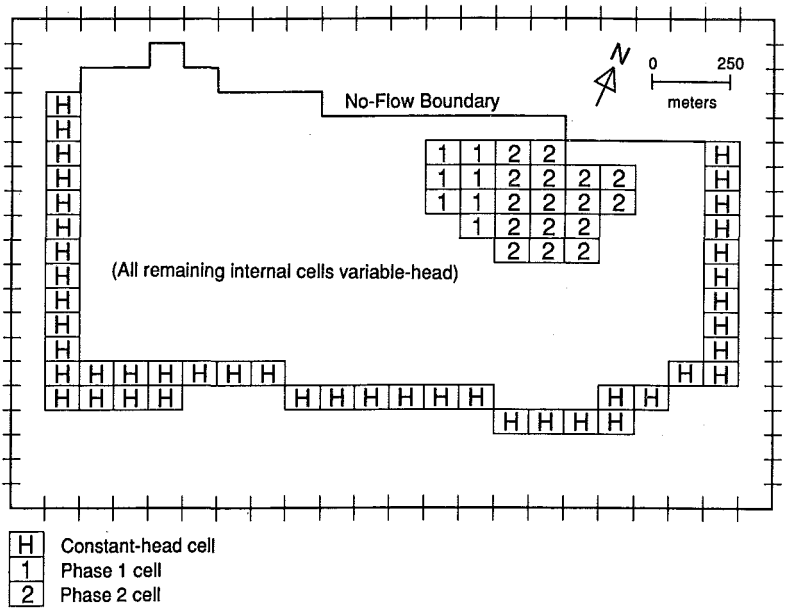

FIG. 5. Model Representation

A background chloride concentration (uncontaminated ground water) of 20 parts per million (ppm) was inferred on the basis of chemical analyses made on samples from upgradient wells 1, 138, and 139 (Fig. 3). An initial concentration of 20 ppm was input for all model cells.

Contaminant influx to the aquifer was simulated with a series of injection wells, one at each landfill cell (Fig. 5), injecting at a rate equal to (cell area) × (areal recharge rate). Areal recharge rate was determined in model calibration (next section). Wells at the cells contained in the phase 1 part of the landfill injected contaminant throughout the 16.7 year simulation (1/6/71–9/25/87). Wells at phase 2 cells injected contaminant during the interval from the initiation of operations in this portion of the landfill (4/3/75) to the end of the simulation (9/25/87). The temporal simulation of contaminant flux rather than assuming an instantaneous release of a "contaminant slug" is appropriate because the contaminant originates from the solid-waste leachate, which is gradually produced and released to the ground-water flow system.

Model Calibration

Ground-Water Flow

Hydraulic conductivity (K) and areal recharge (R) were adjusted in calibrating the flow model to the 6/26/87 water table surface. Prior to calibration, independent estimates of the magnitude of these parameters were established, as explained previously. In calibrating the model, recharge was varied within the range of 15–60 cm/year. This range incorporates independent estimates made by previous investigators in similar buried-valley-

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aquifer settings in southwest Ohio [e.g., 31 cm/years, Walton and Scudder (1960), 31–33 cm/year, Norris and Spieker (1966) and 15–53 cm/year, Spieker (1968b)].

K and R values were input in increments of 3.0×10^{-4} m/s and 1 cm/yr, respectively. The degree to which a given combination of K and R produced a match between calculated and observed water table configurations was assessed through qualitative comparison of shape and relative position of contours of various head values. A calibration summary of matching parameter combinations is as follows:

$$K = 7.0 \times 10^{-4} \text{ m/s} \quad (1a)$$

$$R = 25 \text{ cm/year} \quad (1b)$$

$$K = 1.0 \times 10^{-3} \text{ m/s} \quad (2a)$$

$$R = 36 \text{ cm/year} \quad (2b)$$

$$K = 1.3 \times 10^{-3} \text{ m/s} \quad (3a)$$

$$R = 46 \text{ cm/year} \quad (3b)$$

$$K = 1.6 \times 10^{-3} \text{ m/s} \quad (4a)$$

$$R = 57 \text{ cm/year} \quad (4b)$$

These combinations resulted in virtually identical model-calculated water table configurations. The model-calculated hydraulic head surface closely approximates the observed steady-state surface (Fig. 3). Note that each calibrated set has the same ratio of K to R, approximately 2.8×10^{-5} [(m/s)/cm/yr)]. The calibration process thus identifies the ratio of K to R, but not the actual value of either parameter. It is well known that K and R cannot both be identified if only head measurements are available, as in this study [e.g., Bear (1979)]. The independent field estimates of K can, however, help determine which calibrated sets of K and R include values for K that are the most reasonable.

Because the piezometer tests provide K estimates for aquifer deposits along the entire well screen (3.1 m), the results are more indicative (than grain-size estimates) of a bulk aquifer K. The parameter combinations of Eqs. 2 and 3 include K values that bracket, and are very close to, the piezometer-test average ($1.2 \times$, 10^{-3} m/s). Further, they include values for R that are reasonable for this aquifer-flow system. It is probable that actual recharge values for the aquifer are relatively high because of the permeable nature of near-surface deposits and the proximity of the water table to ground surface during periods of recharge. Substantial recharge may also be facilitated by the absence of a surface drainage system (runoff is minimal). The parameter combinations of Eqs. 2 and 3, were thus considered most reasonable.

Contaminant Transport

The ground-water-flow parameter combinations of Eqs. 2 and 3 were used alternately in concentration calibration with the advection-dispersion model. The porosity input was 0.32, the average porosity determined from lab measurements of bulk and particle mass densities on each of the split-spoon samples for which K values were determined (see Table 3).

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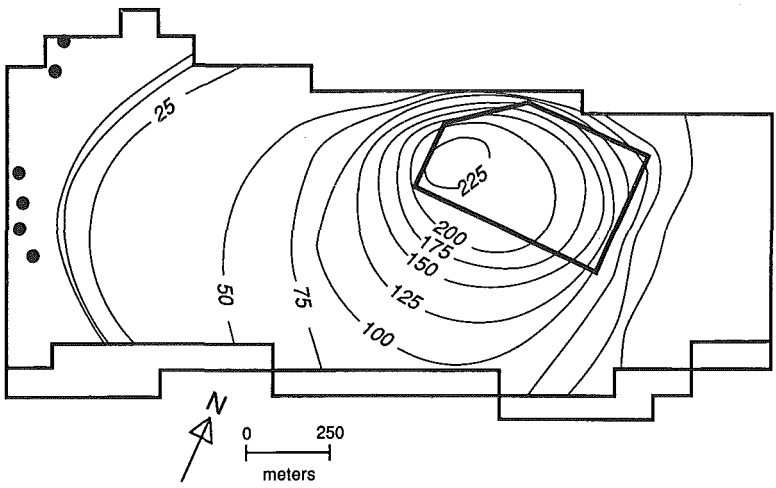

FIG. 6. Model-Calculated 9/25/87 Chloride Concentration Distribution (Contours in ppm); Thicker Lines Represent 25 ppm Contours in 10 Year and 50 Year Model Predictions; Black Circles Represent Nearest Downgradient Domestic Wells

Chloride concentrations were calibrated to the 9/25/87 distribution shown in Fig. 4. Adjusted parameters included concentration of injected water at landfill source nodes (C), longitudinal dispersivity (α_l) and ratio of transverse to longitudinal dispersivity (α_t/α_l). The heterogeneous distribution of waste and corresponding variation in leachate concentration and quantity are such that accurate determinations of the spatial distribution of C are virtually impossible. A single value for C was used for the entire landfill. A range of 200–600 ppm was established on the basis of chloride concentrations in samples taken from leachate seeps along the boundary slopes of the landfill. C was adjusted upward from an initial value of 200 ppm until the calculated concentrations immediately downgradient of the landfill were close in magnitude to observed values for wells 4, 5S, and 141 (Fig. 3). Values for α_l and α_t/α_l were then varied (one parameter between runs) to adjust the shape of the calculated plume. The best match was achieved with $C = 320$ ppm; $\alpha_l = 53$ m; $\alpha_t/\alpha_l = 0.10$; and the ground-water-flow parameter combination of Eq. 3 ($K = 1.3 \times 10^{-3}$ m/s; $R = 46$ cm/year) (Figs. 4 and 6).

Model Verification

Ground-Water Flow

Available field data were insufficient to perform a verification of the ground-water-flow model. Such verification is ideally conducted with an observed perturbation of the "steady state" head field resulting from the operation of a known, quantifiable stress (e.g., pumping). Unfortunately, permission was not granted to perform such a test.

Contaminant Transport

The degree to which the model reproduced the 9/25/87 plume in calibration suggests its potential capability in simulating long-term contaminant

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TABLE 4. Observed versus Calculated Chloride Concentrations (ppm) for Historical Sampling Periods

Sampling date (1)	Well Number 133 (2)	136 (3)	138 (4)	139 (5)	141 (6)
Observed 11/23/83	—[a]	—[a]	31	32	132
Calculated	32	88	31	36	183
Observed 4/20/79	—[a]	57	33	18	89
Calculated	23	45	26	31	130
Observed 5/20/75	16	—[a]	—[a]	19	79
Calculated	20	23	20	20	95

[a]No observed data.

transport. The validity of the model was further assessed by testing its ability to reproduce contemporaneous concentration data sets from historical sampling periods, using the ground-water-flow and mass transport model parameters established in calibration. Three concentration data sets were chosen at approximately four-year increments prior to 1987 (11/23/83, 4/20/79, and 5/20/75). Because of the limited number of control points for the historical data sets, no attempt was made to reconstruct the associated plumes. Instead, for the various wells for which data are available, the observed and calculated chloride concentrations were compared.

The verification results are summarized in Table 4. The data indicate that there is reasonable agreement between observed and model-calculated chloride concentrations for most wells. Because mass transport parameters were calibrated on the basis of chloride measurements made in the present study (i.e., calibration 9/25/87 plume, Fig. 4), there is potential for discrepancy in measured and calculated chloride concentrations attributable to inconsistencies in past and present field sampling and/or analytical procedures. Considering the possibility than an individual sampling set may not be indicative of concentrations representative for the period it was taken but rather may reflect some extreme condition of the hydrologic regime (e.g., flooding and associated high water table levels and contaminant dilution), the observed and calculated values are reasonably close.

Model Predictions

The final phase of the modeling study involved model predictions of contaminant plume configuration at time periods of 10 and 50 years beyond the 9/25/87 sampling date. Predictions were made for the purpose of assessing the possibility of future contamination of downgradient domestic supply wells. Contaminant transport was modeled under steady-state flow conditions (see Fig. 3).

The downgradient extent of the model-predicted contaminant plumes, defined by the 25 ppm concentration contour, are shown in Fig. 6. The plume undergoes a slight increase in overall size over the time interval from the end of the 9/25/87 calibration to the end of the 10 year prediction. Very little change in plume geometry occurs between the 10 year and 50 year simulations. The location of the model-calculated 25 ppm contour, at the

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west margin of the study region, remains essentially the same. The plume reaches a steady-state equilibrium with the ambient ground-water-flow system after 10 years and does not undergo significant growth beyond this time. The attainment of a condition of plume equilibrium could effectively decrease the potential for contamination of the downgradient supply wells. The results of the simulation may thus be used in a tentative manner in guiding future water use in the town of Woodsdale, Ohio, near the downgradient margin of the modeled region. Conditions of contaminant-plume equilibrium have been suggested in previous landfill studies. Palmquist and Sendlien (1975), for example, analyzed temporal variation in water-quality data for five floodplain aquifers ranging in size from 13 to 47 acres (53,000–190,000 m^2) and in age from 9 to 47 years. They found that in the majority of cases, contaminant plumes were not increasing in size but had achieved a maximum size and were in a steady-state condition.

Importantly, the simulations made in the present study provide a worse-case scenario in that future chloride-plume configurations are likely greater in areal extent than corresponding plumes of more toxic, reactive species. Furthermore, a progressive decrease in the concentration of leachate that might occur in time would result in smaller plumes than those predicted.

Conclusions

The MOC transport model accurately fits chloride concentration contours observed nearly 17 years after initial waste deposition at the Butler County landfill. Additionally, the model reproduces with reasonable accuracy chloride concentrations observed during earlier time periods. These results suggest that the model may accurately simulate long-term contaminant transport under steady-state ground-water-flow conditions in this aquifer-flow system, and thus provide a tool in guiding the future use of downgradient water supply wells potentially affected by the landfill. An important initial step in the application of the contaminant-transport model was the construction of a representative observed contaminant-plume configuration to which the model was to be calibrated. Frequent chemical sampling over the course of the field portion of the study allowed determination of a representative contemporaneous set of chloride concentrations for the wells sampled. The observed concentrations were reproduced by the model with reasonable accuracy. This may attest to the validity of the approach employed in this study in simulating the contaminant source. Exact definition of the nature of temporal release of contaminants from the landfill into the ground-water-flow system is impossible. Results of this study suggest, however, that for long-term simulations, source representation by a series of recharge wells injecting contaminant with an average leachate concentration at a rate governed by average annual areal recharge may be sufficient.

Model predictions of the future extent of the chloride plume suggest the possibility of a condition of plume equilibrium beyond a period of approximately 10 years. The attainment of such an equilibrium condition would significantly decrease the potential for future contamination of downgradient domestic supply wells from landfill-derived contaminant.

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APPENDIX II. NOTATION

The following symbols are used in this paper:

C = concentration of injected water at landfill source nodes (ppm);
K = hydraulic conductivity (m/s);
R = areal recharge (cm/year);
α_l = longitudinal dispersivity (m); and
α_t = transverse dispersivity (m).

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