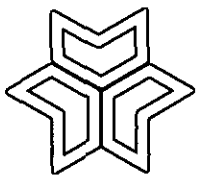

Fluids and Hydraulics

LANDFILL LEACHATE MIGRATION THROUGH
SHALLOW UNCONFINED AQUIFERS

David W. Ostendorf, Richard R. Noss and
David O. Lederer

Water Resources Research

In Press, 1984

Department of Civil Engineering
University of Massachusetts at Amherst

LANDFILL LEACHATE MIGRATION THROUGH SHALLOW UNCONFINED AQUIFERS

David W. Ostendorf, Richard R. Noss and
David O. Lederer

Water Resources Research

In Press, 1984

WATER RESOURCES RESEARCH, VOL. 20, NO. 2, PAGES 291–296, FEBRUARY 1984

Landfill Leachate Migration Through Shallow Unconfined Aquifers

DAVID W. OSTENDORF, RICHARD R. NOSS, AND DAVID O. LEDERER

Department of Civil Engineering, University of Massachusetts

We model the transport of a simply reactive contaminant through a landfill and initially pure, underlying, shallow, one-dimensional unconfined aquifer with a plane, sloping bottom under steady hydraulic conditions in the assumed absence of dispersion and downgradient dilution. The user population and a presumedly constant contaminant loading factor determine the pollution input to the groundwater system, and we model the near field response as a single linear reservoir whose output comprises the far field source term at the downgradient edge of the landfill. The far field analysis yields a method of characteristics solution valid in the vicinity of the source location with frame speeds modified by recharge, head loss, bottom slope, and linear adsorption, and concentrations reflecting first-order reaction kinetics. We calibrate and test the near and far field models against conservative chloride and first-order reactive bicarbonate data at the Babylon, New York landfill with accurate and physically plausible results.

INTRODUCTION

We model the transport of a simply reactive contaminant through a landfill and initially pure, underlying, shallow, one-dimensional, unconfined aquifer with a plane, sloping bottom under steady hydraulic conditions in the assumed absence of dispersion and downgradient dilution. The resulting quantitative appreciation of the physical transport mechanisms and time scales associated with unconfined aquifer pollution enables us to identify the source history of existing plumes and predict trajectories of future contamination. This understanding is prerequisite for the assessment of the emerging evidence of subsurface water pollution downgradient of existing landfills [*Garland and Mosher*, 1975] and the proper design and operation of the future facilities necessitated by ongoing waste generation.

We follow surface water quality modelers [*Fischer et al.*, 1979] by distinguishing a near field region under the landfill, where mixing of leachate and groundwater is presumed to occur, and a far field region of fully mixed, one-dimensional flow downgradient of the landfill in an attempt to relate subsurface contaminant migration to surface application of solid waste. We describe the near field with an initially pure linear reservoir that converts leachate input to a contaminated baseflow output which comprises the far field source term at the downgradient end of the facility. The lumped parameter near field approach follows *Gelhar and Wilson* [1974] and *Mercardo* [1976], who successfully describe regional pollution due to distributed inputs with the linear reservoir analysis. We complete the near field model by relating the input to the per capita contaminant generation rate and solve the case of a linearly increasing user population, although the near field–far field decomposition accommodates other input variation with the convolution integral response common to linear systems analysis [*Dooge*, 1973].

We distinguish analytical and numerical descriptions of far field subsurface contaminant transport in the literature. The numerical modelers [*Bachmat et al.*, 1980] retain all terms in the conservation equations governing the process by simulating differential equations and boundary and initial conditions with numerical equivalents over a temporal and spatial grid; the resulting models properly represent physics at the expense of site specific computer programs with attendant documentation requirements. An analytical approach, where problem geometry permits, solves simplified differential equations, and boundary and initial conditions explicitly, obviates the computer, and yields a flexible, simple, and physically valid model appropriate in a preliminary planning and assessment context. We may cite several analytical studies of advective-dispersive transport of a contaminant injected into a uniform flow field on a spatially or temporally discontinuous basis: *Lenau* [1972] postulates a steady state, conservative injection from a recharge well, while *Wilson and Miller* [1978, 1979] consider unsteady pollution due to a constant vertical line source of reactive contaminant. *Bear* [1979] summarizes unsteady contaminant migration due to a series of one-dimensional reactive source conditions, and *Prakash* [1982] models steady state reactive pollution in three dimensions due to point, line, and volume sources. We suggest that the continuous, spatially distributed contamination generated by landfill leachate yields small gradients and consequent dominance of dispersion by the advective and reactive transport mechanisms. The resulting neglect of dispersion permits us to consider a simply varying, one-dimensional flow field in the vicinity of the source location by using the method of characteristics; as in the work of *Wilson and Miller* [1978, 1979], *Bear* [1979], and *Prakash* [1982], the reactive contaminants under study are assumed to be linearly adsorptive or exhibit first-order decay in the far field. We note that *Wilson and Gelhar* [1981] and *Bredehoeft and Pinder* [1973] use analytical and numerical versions of the method of characteristics to describe contaminant migration in the unsaturated and saturated zones, respectively.

FAR FIELD GOVERNING EQUATIONS

The steady conservation of water mass in a one dimensional, unconfined aquifer subject to constant recharge ε is simply

$$q = q_s + \varepsilon x \tag{1}$$

with horizontal distance x downstream of the pollutant source, where conditions are denoted by an s subscript, as indicated in Figure 1. The discharge q per unit width and average linear velocity v are related by definition

$$v = q/nh \tag{2}$$

Paper number 3W1936.
0043-1397/84/003W-1936$05.00

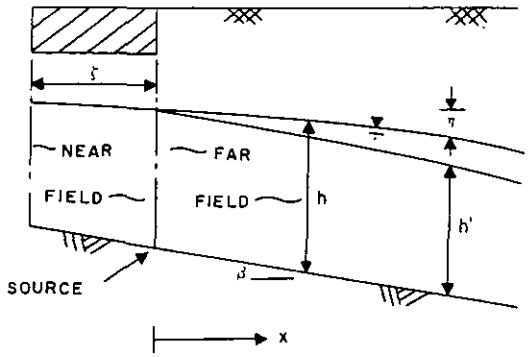

Fig. 1. Definition sketch.

with porosity n and aquifer thickness h given by

$$h = h_s + x \tan \beta - \eta \tag{3}$$

Equation (3) reflects the assumption of a plane, sloping, underlying aquiclude of small angle β to the horizontal; η is the water table elevation below its source position, and its horizontal gradient obeys Darcy's law when β is small:

$$v = \frac{kg}{\nu n} \frac{d\eta}{dx} \tag{4}$$

with gravitational acceleration g, permeability k, and fluid viscosity ν. We collect (1), (2), and (4) and discover

$$\frac{q}{h} = \frac{kg}{\nu} \frac{d\eta}{dx} \tag{5}$$

which, in the vicinity of the far field source, yields the first-order relation

$$\eta = \frac{q_s \nu x}{kgh_s} \qquad x \ll \frac{h_s}{\tan \beta}, \frac{q_s}{\varepsilon} \tag{6}$$

In view of (1), (3), and (6), we may approximate the average linear velocity by modifying (2) with the result

$$v = v_s\left(1 + \gamma \frac{x}{h_s}\right) \tag{7}$$

where the second-order factor γ reflects recharge, head loss, and bottom slope effects

$$\gamma = \frac{\varepsilon h_s}{q_s} + \frac{q_s \nu}{kgh_s} - \tan \beta \tag{8}$$

Next we consider the conservation of contaminant mass for steady hydraulics in the far field as a balance of retarded storage change, advection, recharge, reaction, and dispersion [*Pinder*, 1973; *Freeze and Cherry*, 1979]

$$nh'R \frac{\partial c}{\partial t} + \frac{\partial}{\partial x}(q'c) - \varepsilon' c_\varepsilon = -nh'\lambda c + n \frac{\partial}{\partial x}\left(h' D_L \frac{\partial c}{\partial x}\right) \tag{9}$$

with time t and recharge concentration c_ε. We follow *Wilson and Miller* [1978, 1979] by postulating a shallow aquifer so that c represents depth averaged plume concentration and mild differential density effects, which tend to establish vertical concentration gradients within the plume, are ignored. Field measurements [*Kimmel and Braids*, 1980] do suggest that gross differential density separates the plume from lighter downgradient recharge ε', and we consequently assume the fluids to be immiscible, so that the contaminated discharge q' equals its source value, i.e.,

$$\varepsilon' = 0 \qquad q' = q_s \tag{10}$$

and the plume is overlain by a freshwater lens as suggested by Figure 1. A common pressure gradient $d\eta/dx$ drives the lens and plume at the same velocity however, so that the contaminated aquifer thickness h' will be given by

$$h' = q_s/n\nu \tag{11}$$

We postulate a first-order reactive contaminant with decay constant λ subject to linear adsorption with corresponding retardation factor R given by *Freeze and Cherry* [1979]

$$R = 1 + K \frac{\rho}{n} \tag{12}$$

where ρ is the solid mass per unit solid matrix volume and K is the distribution coefficient characterizing linear adsorption. The longitudinal coefficient of hydrodynamic dispersion D_L is the product of the aquifer longitudinal dispersivity α_L and the average linear velocity [*Freeze and Cherry*, 1979]

$$D_L = \alpha_L v \tag{13}$$

so that, in view of (2) and (9), dispersion is negligible when

$$x_c \gg \alpha_L \tag{14}$$

where x_c is the distance over which concentration varies appreciably. When (14) is satisfied, (9)–(11) yield a simple advection equation describing contaminant migration in the far field, valid for nonuniform, immiscible flow

$$\frac{\partial c}{\partial t} + \frac{v}{R} \frac{\partial c}{\partial x} = \frac{-\lambda c}{R} \tag{15}$$

which is subject to specified source conditions

$$c = c_s \qquad x = 0 \tag{16}$$

CONTAMINANT CHARACTERISTICS

We solve (15) and (16) with the method of characteristics [*Eagleson*, 1970] which rests on the chain rule

$$\frac{dc}{dt} = \frac{\partial c}{\partial t} + \frac{\partial c}{\partial x} \frac{dx}{dt} \tag{17}$$

where dc/dt represents temporal change in a frame of reference moving at speed dx/dt. Equations (7), (15), and (17) yield the frame speed

$$\frac{dx}{dt} = \frac{v_s}{R}\left(1 + \gamma \frac{x}{h_s}\right) \tag{18}$$

in which contaminant concentration obeys

$$\frac{dc}{dt} = -\frac{\lambda c}{R} \tag{19}$$

The paths, or characteristics, of the moving frames follow upon integration of (18) from a starting time t_s at the source $x_s = 0$ to any subsequent place x and time t in the far field, subject to the constraints of (6). Recalling the smallness of $\gamma x/h_s$, we have

$$\int_0^x \left(1 - \frac{\gamma \chi}{h_s}\right) d\chi = \frac{v_s}{R} \int_{t_s}^t d\tau \tag{20}$$

so that the contaminant characteristics in the vicinity of the source are approximated by

$$t - t_s = \frac{Rx}{v_s}\left(1 - \frac{\gamma x}{2h_s}\right) \qquad x \ll \frac{h_s}{\tan \beta}, \frac{q_s}{\varepsilon} \tag{21}$$

The frame starting at $t_s = 0$ marks the arrival x_a, t_a of the contaminant plume; (21) suggests that

$$t_a = \frac{Rx_a}{v_s}\left(1 - \frac{\gamma x_a}{2h_s}\right) \tag{22}$$

Equation (19) indicates that the reactive nature and source strength of the contaminant determine its temporal behavior in the moving frame; we integrate from c_s, t_s starting conditions to any subsequent c, t with the result

$$c = c_s \exp\left[\frac{\lambda}{R}(t_s - t)\right] \tag{23}$$

The far field source strength and starting time also represent near field output and accordingly link our two model regions together, as indicated schematically by Figure 1.

NEAR FIELD SYSTEM

The landfill constitutes a distributed input of width b and length ζ in the direction of groundwater flow to an initially pure, linear reservoir whose output comprises c_s. The conservation of contaminant mass per unit width is given by

$$\zeta n h_s R \frac{dc_s}{dt_s} + q_s c_s = \frac{SP}{b} \tag{24}$$

where, in keeping with the simple near field system approach, we assume that the pollution input is simply related to user population P by a constant contaminant loading factor per capita S. The loading factor constancy reflects presumedly rapid contaminant generation due to precipitation and solid waste interaction, in contrast to the slower time scale governing contaminant decay. This decay time scale must be comparable to the far field time scale if concentrations are appreciable in the far field; the first-order reactions are therefore negligibly slow in the fast flow field under the landfill, and we consequently set $\lambda = 0$ in the near field. We also assume linked linear segments of user population, i.e.,

$$P = P_i + G_i(t_s - t_{si}) \qquad t_{si} \le t_s \le t_{si+1} \tag{25}$$

with population P_i at time t_{si} and growth rate G_i valid for the ith segment of time.

We combine (24) and (25) and derive the near field response equation

$$\frac{dc_s}{dt_s} + \frac{c_s}{t_c} = \frac{c_i}{t_c}\left(1 + \frac{t_s}{t_i}\right) \tag{26}$$

with landfill response time t_c and segment concentration c_i and time t_i obeying

$$t_c = R\zeta/v_s \tag{27a}$$

$$c_i = \frac{S(P_i - G_i t_{si})}{bq_s} \tag{27b}$$

$$t_i = \frac{P_i}{G_i} - t_{si} \tag{27c}$$

We solve (26) subject to pure initial and matching conditions

TABLE 1. Population Segment Parameters, Babylon Landfill

	i		
	1	2	3
t_{si}, s × 10^8	0	4.10	5.68
G_i, cap/s × 10^{-4}	1.06	7.11	3.05
P_i, cap × 10^4	5.44	9.79	21.0
t_i, s × 10^8	5.13	−2.72	1.21
c_i, kg/m³ × 10^6	5.25S	−18.7S	3.53S
c_{si}, kg/m³ × 10^6	0	6.93S	11.6S

c_{si} between segments of linear growth

$$c_s = 0 \qquad t_s = 0 \tag{28a}$$

$$c_s = c_{si} \qquad t_s = t_{si} \tag{28b}$$

This nonhomogeneous, linear, first-order ordinary differential equation with constant coefficients has the solution [*Rainville and Bedient*, 1969]

$$c_s = c_1\left\{\left(1 - \frac{t_c}{t_1}\right)\left[1 - \exp\left(-\frac{t_s}{t_c}\right)\right] + \frac{t_s}{t_1}\right\} \qquad 0 \le t_s \le t_{s1} \tag{29a}$$

$$c_s = c_i\left\{1 + \frac{t_s - t_c}{t_i} + \left(\frac{c_{si}}{c_i} - 1 - \frac{t_{si} - t_c}{t_i}\right) \cdot \exp\left[(t_{si} - t_s)/t_c\right]\right\} \qquad t_{si} \le t_s \le t_{si+1} \tag{29b}$$

$$c_s = c_{sd} \exp\left[(t_{sd} - t_s)/t_c\right] \qquad t_{sd} \le t_s \tag{29c}$$

with source concentration c_{sd} at the time of shutdown t_{sd}. We illustrate near and far field calculations with a case study.

BABYLON LEACHATE PLUME

Our case study is the well-documented leachate plume downstream of the sanitary landfill in Babylon, New York. The contaminant flows into the upper glacial aquifer in southern Long Island, an unconfined aquifer of porosity $n = 0.27$ and permeability $k = 6.34 \times 10^{-11}$ m² underlain by the relatively impervious Gardiners Clay and Magothy formations, which intersect the water table a distance 6300 m upstream of the landfill [*Collins et al.*, 1972] with a local bottom slope $\tan \beta = 0.0027$ [*Kimmel and Braids*, 1975, 1980]. *Kimmel and Braids* [1980] present water table elevations under the landfill, and we accordingly estimate $h_s = 22.5$ m and $(d\eta/dx)_s = 0.00161$; (2) and (4) then yield $v_s = 3.37 \times 10^{-6}$ m/s and $q_s = 2.05 \times 10^{-5}$ m²/s, where $\nu = 1.1 \times 10^{-6}$ m²/s. We consider the upstream recharge area in light of this discharge and estimate $\varepsilon = 3.25 \times 10^{-9}$ m/s so that (8) yields $\gamma = 0.00248$. *Kimmel and Braids* [1980] suggest that the landfill opened in 1947 and served a 1973 population of 287,000; the user population data of Table 1 reflect this value and census [*U.S. Department of Commerce*, 1977] figures for Suffolk County. The anomalous surge of growth in the early 1960's cited by the table forces us to consider three growth segments for the Babylon area. *Kimmel and Braids* [1975] also specify landfill dimensions $b = 505$ m and $\zeta = 689$ m.

Kimmel and Braids [1980] suggest that the chloride ion is conservative in the relatively pure upper glacial aquifer and present the 1974 depth averaged concentrations c_m presented in Table 2, so that we specify $t = 8.52 \times 10^8$ s, $R = 1$ and $\lambda = 0$. Wells 1–3 and 8 lie at the downstream boundary of the

TABLE 2. Observed Chloride Concentration and Error, From *Kimmel and Braids* [1980]

Well	x, m	c_m, kg/m³	t_c, s × 10⁸	c_c, kg/m³	δ, %	i
127	360	0.245	7.47	0.256	5	3
6	900	0.190	5.98	0.180	−6	3
10	920	0.170	5.93	0.176	4	3
12	1570	0.175	4.26	0.100	−43	2
124	1580	0.058	4.24	0.100	72	2
118	2180	0.055	2.83	0.073	34	1
122	2230	0.048	2.71	0.071	49	1
35	2810	0.057	1.47	0.044	−23	1
29	3190	0.044	0.71	0.023	−47	1

Babylon landfill and, as suggested by Table 3, their average value accordingly yields a source concentration $c_s = 0.305$ kg/m³ at time $t_s = 8.52 \times 10^8$ s. We consult (27)–(29) to compute $t_c = 2.04 \times 10^8$ s and the t_i, c_i, and c_{si} values cited in Table 1; the chloride loading factor $S_{Cl} = 1.40 \times 10^{-8}$ kg/cap-s (where "cap-s" denotes "capita seconds") calibrates the near field model with the 1974 data point.

Table 2 also lists far field model error δ defined by

$$\delta = \frac{c - c_m}{c_m} \tag{30}$$

We predict far field concentration at x, t by identifying the starting time t_s of the frame of reference occupying x, t from (21); the t_s value also identifies the appropriate population segment constants in Table 1. The starting strength c_s follows from (29) and far field concentration $c(c_s, t_s)$ from (23). We assess model accuracy with the mean error δ and standard deviation σ defined by [*Benjamin and Cornell*, 1970]

$$\delta = \frac{1}{j} \sum \delta \tag{31a}$$

$$\sigma = \left[\frac{1}{j} \sum \delta^2 - \delta^2 \right]^{1/2} \tag{31b}$$

The mean error $\delta_{Cl} = 5\%$ and standard deviation $\sigma_{Cl} = 38\%$ indicate a good fit of far field model and data and accordingly endorse the conservative contaminant transport analysis.

A similar conservative analysis of the *Kimmel and Braids* [1980] bicarbonate data of Table 4 yields a consistently strong far field overprediction, and we accordingly look to a simple account of the reactive nature of the bicarbonate ion. *Kimmel and Braids* [1980] suggest that high gaseous carbon dioxide pressure at the landfill generates high bicarbonate leachate concentrations; we postulate falling carbon dioxide pressures downgradient of the landfill due to transverse dispersive transport to the freshwater lens, where pressures equilibrate with low atmospheric values. A transverse hydrodynamic dispersion coefficient D_T sends the gas down a

TABLE 3. Source Data, From *Kimmel and Braids* [1980]

Well	c_{Cl}, kg/m³	c_{HCO_3}, kg/m³
1	0.625	0.898
2	0.054	···
3	0.385	0.470
8	0.157	···
114	···	0.371*

*1973 concentration 0.342.

TABLE 4. Observed Bicarbonate Concentration and Error, From *Kimmel and Braids* [1980]

Well	x, m	c_m, kg/m³	t_c, s × 10⁸	c_c, kg/m³	δ, %	i
127	360	0.540	7.47	0.487	−16	3
128	630	0.277	6.72	0.416	33	3
6	900	0.665	5.98	0.341	−57	3
10	920	0.154	5.93	0.336	83	3
124	1580	0.158	4.24	0.190	−10	2
118	2180	0.086	2.83	0.140	10	1
122	2230	0.138*	2.19	0.115	−44	1
35	2810	0.054	1.47	0.083	−4	1
29	3190	0.020	0.71	0.044	31	1
···	3320	0.023	0.47	0.030	−24	1

*1973 Data.

vertical concentration gradient $\partial c'/\partial y$ through the voids at the freshwater interface so that, equating this transport to the reaction term in (9), we deduce

$$-nh'\lambda c = -nD_T \left. \frac{\partial c'}{\partial y} \right|_{\text{interface}} \tag{32}$$

Equation (32) yields a rough decay constant estimate

$$\lambda = \frac{\alpha_T v_s}{h_s^2} \tag{33}$$

where c/h_s characterizes the vertical concentration gradient and the transverse dispersivity α_T is analogous to the longitudinal parameter in (13). *Bear* [1979] discusses a similar approach describing diffusive contaminant transport into dead-end pore space.

Wells 1, 3, and 114 specify a source value $c_s = 0.580$ kg/m³ at $t_s = 8.52 \times 10^8$ s so that the bicarbonate loading factor, in view of Table 1 and (29), is $S_{HCO_3} = 2.66 \times 10^{-8}$ kg/cap-s. Table 4 indicates that the calibrated decay constant $\lambda_{HCO_3} = 6.7 \times 10^{-10}$ s^{-1} zeros the mean far field error, with starting conditions c_s, t_s computed in accordance with the previous chloride procedure and 1974 predicted concentrations reflecting first-order decay in the moving frame, as suggested by (23). The error standard deviation $\sigma_{HCO_3} = 39\%$ is particularly encouraging in view of the possible sampling errors in wells 6, 10, and 122.

The calibrated bicarbonate decay constant and postulated mechanism of (33) yield a transverse dispersivity estimate $\alpha_T = 0.10$ m for the upper glacial aquifer. This value, while considerably less than *Pinder*'s [1973] numerical model estimate of 4.2 m obtained from upper glacial aquifer contamination in nearby Nassau County, is in keeping with *Kimmel and Braids*' [1980] reported concentration contours at Babylon, which retain their original source widths. The discrepancy may in part reflect the presence of silt lenses in the upper glacial aquifer at *Pinder*'s [1973] site, which would act to increase dispersion [*Freeze and Cherry*, 1979]. *Pinder* [1973] also estimates longitudinal dispersivity to be equal to 20 m at Nassau, and we satisfy (14)'s neglect of dispersion even with his high parameter value.

CONCLUSIONS

We model near and far field migration of a simply reactive contaminant through a landfill and initially pure, underlying, shallow, one-dimensional, unconfined aquifer with a plane,

sloping bottom under steady hydraulic conditions in the assumed absence of dispersion and downgradient dilution. The near field is construed as a single linear reservoir with an input governed by a constant contaminant loading factor reflecting per capita waste generation and an output specifying the depth averaged source term for the far field model. We analyze the advective transport in this latter region in the vicinity of the source with the method of characteristics and discover that recharge, head loss, bottom slope, and linear adsorption modify the reference frame speeds, while first-order reactions influence pollutant concentration in the frames. Observed 1974 chloride and bicarbonate concentrations at the Babylon, New York landfill calibrate the near field linear reservoir model with contaminant loading factors $S_{Cl} = 1.40 \times 10^{-8}$ kg/cap-s and $S_{HCO_3} = 2.66 \times 10^{-8}$ kg/cap-s, respectively. We test the far field advective transport model against downgradient conservative chloride measurements with a mean error of 5% and standard deviation of 38%, indicative of good accuracy. A postulated first-order bicarbonate reaction zeros the far field model error with a decay constant $\lambda_{HCO_3} = 6.7 \times 10^{-10}$ s^{-1}, which implies a transverse dispersivity $\alpha_T = 0.10$ m consistent with observed nondispersive plume behavior; the 39% bicarbonate error standard deviation suggests reasonable model accuracy as well.

Future research may proceed on several fronts. We suggest that the contaminant loading factor include temporal variability representative of precipitation–solid waste interaction at the landfill surface, the near field parameter estimation reflect a distributed parameter analysis of contamination under the landfill, and the far field model accommodate more realistic reactions. These model improvements must consider the long time scale of the subsurface environment and preserve the simplicity of the present approach; in the latter regard, the near field–far field decomposition decouples the improvements as well, facilitating separate analytical investigations. Additionally, the calibrated loading factors and decay constant representing Babylon conditions should be compared to comparable plumes in geologically similar aquifers as data becomes available.

NOTATION

b landfill width, m.
c contaminant concentration, kg/m^3.
D dispersion coefficient, m^2/s.
G population growth rate, cap/s.
g gravitational acceleration, m/s^2.
h aquifer thickness, m.
K distribution coefficient, m^3/kg.
k permeability, m^2.
n porosity.
P user population, cap.
q discharge of water per unit aquifer width, m^2/s.
R retardation factor.
S contaminant loading factor, kg/s cap.
t time, s.
v average linear velocity, m/s.
x far field horizontal distance downstream of pollutant source, m.
y vertical distance, m.
α dispersivity, m.
β underlying aquiclude slope angle.
γ velocity modification factor.
δ error.
ε recharge velocity, m/s.
ζ landfill length, m.
η water table elevation below source position, m.
λ decay constant, 1/s.
ν water kinematic viscosity, m^2/s.
ρ solid matrix mass per unit volume, kg/m^3.
σ standard deviation.

Subscripts

a arrival trajectory.
c characteristic quantity.
Cl chloride property.
d shutdown condition.
HCO_3 bicarbonate property.
i population growth segment condition.
L longitudinal transport property.
m measured quantity.
s conditions at far field source.
T transverse transport property.
ε recharge conditions.

Superscript

$'$ contaminated discharge property.

Acknowledgments. The Commonwealth of Massachusetts Division of Water Pollution Control funded this research under grant MDWPC 80-32; we acknowledge and appreciate their support.

REFERENCES

Bachmat, Y., J. Bredehoeft, B. Andrews, D. Holtz, and S. Sebastian, *Groundwater Management: The Use of Numerical Models, Water Resour. Monogr. Ser.*, vol. 5, AGU, Washington, D. C., 1980.

Bear, J., *Hydraulics of Groundwater*, pp. 263–276, McGraw-Hill, New York, 1979.

Benjamin, J. R., and C. A. Cornell, *Probability, Statistics and Decision for Civil Engineers*, p. 12, McGraw-Hill, New York, 1970.

Bredehoeft, J. D., and G. F. Pinder, Mass transport in flowing groundwater, *Water Resour. Res.*, 9(1), 194–210, 1973.

Collins, M. A., L. W. Gelhar, and J. L. Wilson, Hele-Shaw model of Long Island aquifer system, *J. Hydraul. Div. Am. Soc. Civ. Eng.*, *98*(HY9), 1701–1714, 1972.

Dooge, J. C. I., Linear theory of hydrologic systems, *Tech. Bull. 1468*, pp. 20–25, U.S. Dep. of Agric., Washington, D. C., 1973.

Eagleson, P. S., *Dynamic Hydrology*, pp. 337–344, McGraw-Hill, New York, 1970.

Fischer, H. B., E. J. List, R. C. Y. Koh, J. Imberger, and N. H. Brooks, *Mixing in Inland and Coastal Waters*, pp. 10–11, Academic, New York, 1979.

Freeze, R. A., and J. A. Cherry, *Groundwater*, pp. 388–408, Prentice-Hall, Englewood Cliffs, N. J., 1979.

Garland, G. A., and D. C. Mosher, Leachate effects from improper land disposal, *J. Wasteage*, *6*, 42–48, 1975.

Gelhar, L. W., and J. L. Wilson, Groundwater quality modeling, *Groundwater*, *12*(6), 399–408, 1974.

Kimmel, G. E., and O. C. Braids, Preliminary findings of a leachate study on two landfills in Suffolk County, New York, *J. Res. U.S. Geol Surv.*, *3*(3), 273–280, 1975.

Kimmel, G. E., and O. C. Braids, Leachate plumes in groundwater from Babylon and Islip landfills, Long Island, New York, *U.S. Geol. Surv. Prof. Pap. 1085*, 1980.

Lenau, C. W., Dispersion from recharge well, *J. Eng. Mech. Div. Am. Soc. Civ. Eng.*, *98*(EM2), 331–344, 1972.

Mercardo, A., Nitrate and chloride pollution of aquifers: A regional study with the aid of a single cell model, *Water Resour. Res.*, *12*(4), 731–747, 1976.

Pinder, G. F., A Galerkin-finite element simulation of groundwater contamination on Long Island, New York, *Water Resour. Res.*, 9(6), 1657–1669, 1973.

Prakash, A., Groundwater contamination due to vanishing and finite size continuous sources, *J. Hydraul. Div. Am. Soc. Civ. Eng.*, *108*(HY4), 572–590, 1982.

Rainville, E. D., and P. E. Bedient, *Elementary Differential Equations*, pp. 110–125, MacMillan, New York, 1969.
U.S. Department of Commerce, *County and City Data Book*, p. 330, Bureau of the Census, Washington, D. C., 1977.
Wilson, J. L., and L. W. Gelhar, Analysis of longitudinal dispersion in unsaturated flow, 1, The analytical method, *Water Resour. Res.*, *17*(1), 122–130, 1981.
Wilson, J. L., and P. J. Miller, Two-dimensional plume in uniform groundwater flow, *J. Hydraul. Div. Am. Soc. Civ. Eng.*, *104*(HY4), 503–514, 1978.
Wilson, J. L., and P. J. Miller, Two-dimensional plume in uniform groundwater flow-closure, *J. Hydraul. Div. Am. Soc. Civ. Eng.*, *105*(HY12), 1567–1570, 1979.

D. O. Lederer, R. R. Noss, and D. W. Ostendorf, Civil Engineering Department, University of Massachusetts, Amherst, MA 01003.

(Received July 12, 1983;
revised November 22, 1983;
accepted December 6, 1983.)