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Technical Report

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ABSTRACT

This report models the transport of a simply reactive contaminant through a landfill and an underlying shallow, one dimensional, unconfined aquifer with a plane, sloping bottom in the assumed absence of dispersion and downgradient dilution. The constant user population and a presumedly constant contaminant loading factor determine the pollution input to the "near field" groundwater region under the landfill. This near field is modeled as a linear reservoir whose output at the downgradient edge of the facility comprises the source plane for "far field" transport through the unconfined aquifer. The far field analysis describes temporal concentration variation experienced by observers moving away from the source plane at speeds modified by recharge, head loss, bottom slope, and linear adsorption. The temporal concentration variation witnessed by the observers reflects linear adsorption and first order reaction kinetics, and yields a prediction of contamination at any far field place and time.

The basic theory underlying the model has been published in Water Resources Research [Ostendorf et al. (1984)], using an observed landfill leachate plume in Babylon, Long Island for calibration and testing. This report includes an application of the analysis to the plume emanating from the decommissioned landfill of Amherst, Massachusetts as an example of model applicability in the Commonwealth. A simple extension of the model permits the analysis of plumes from infiltration beds [Ostendorf (1985,1986)], as

illustrated by contamination from infiltration beds at Otis Air Force Base in Barnstable County, Massachusetts. A series of applications illustrates the inference of source history from an existing plume, future plume prediction from an existing source, and recovery of groundwater quality after the shutdown of the facility. An infiltration bed design example demonstrates the poor mixing characteristics of the groundwater environment compared to surface waters: dilution factors of 5 or less are feasible in the subsurface, while factors from 10 to 100 are common in rivers and lakes. Thus, infiltration bed effluents contaminate the unconfined aquifer at essentially full strength concentrations in the absence of reaction.

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Symbol	Definition
A	recharge area, m^2.
B	boron.
b	landfill width, m.
C_i	miscellaneous constants.
Cl	chloride.
c	contaminant concentration.
c_a	ambient contaminant concentration.
c_e	effluent concentration.
c_{sf}	steady state source concentration above ambient.
D	dilution factor.
g	gravitational acceleration, m/s^2.
h	aquifer thickness, m.
k	permeability, m^2.
MBAS	methyl active blue substances.
N	total nitrogen.
n	porosity.
P	user population, cap.
Q_e	effluent discharge, m^3/s.
q	discharge per unit aquifer width, m^2/s.
R	retardation factor.
r_e	infiltration bed radius, m.

S	contaminant loading factor.
T	year of measurement, yr.
T_o	year of startup, yr.
t	time, s.
t_c	near field response time, s.
v	average linear velocity, m/s.
x	far field distance downgradient of pollutant source, m.
y	lateral distance, m.
α	effluent ratio.
α_T	transverse dispersivity, m.
β	underlying aquiclude slope angle.
γ	velocity modification factor.
δ	error.
$\bar{\delta}$	mean error.
ε	recharge rate, m/s.
ζ	landfill length, m.
η	water table depression below source position, m.
λ	decay constant, 1/s.
σ	error standard deviation.
ν	water kinematic viscosity, m^2/s.
ψ	near field stream line.
ψ_{max}	stream line separating pure and contaminated water.

SUBSCRIPTS

s	conditions at the source plane.
sd	source conditions at time of shutdown.

CHAPTER 1

INTRODUCTION

1.1 Objectives and Relevance to the Commonwealth

This report models the transport of a simply reactive contaminant through a landfill and an underlying shallow, one dimensional, unconfined aquifer with a plane, sloping bottom. Dispersion and downgradient dilution are assumed to be negligible. The resulting quantitative appreciation of the physical transport mechanisms and time scales associated with unconfined aquifer pollution identifies the source history of an existing plume, estimates the future trajectory of an existing source, and predicts the water quality recovery after the shutdown of the landfill. This understanding is prerequisite for the assessment of the emerging evidence of subsurface water pollution downgradient of existing landfills [Garland and Mosher (1975), Kimmel and Braids (1980)], and the proper design and operation of the future facilities necessitated by ongoing waste generation. The landfill leachate analysis is simply modified to account for contamination from infiltration beds, which are important sources of groundwater pollution in their own right [LeBlanc (1984), Perlmutter and Leiber (1970), Bedient et al. (1983)]. The modified solution will aid in the assessment and design of these related facilities as well.

The relevance of the transport model to the groundwater resources of Massachusetts is demonstrated by case studies of plumes induced by a sanitary landfill in Amherst and an infiltration bed at Otis Air Force Base in Barnstable County. The former contamination resulted in the closure of municipal wells [Lederer (1983)] while the latter threatens public and private water supply wells along its projected path [LeBlanc (1984)].

1.2 Outline of Present Approach

The unconfined aquifer contamination analysis follows surface water quality modelers [Fischer et al. (1979)] by schematizing the environment into "near" and "far field" flow regions in an attempt to relate subsurface pollution to surface application of solid waste, as indicated by Figure 1. The near and far field regions are linked by a "source plane" located at the downgradient boundary of the landfill: the source plane receives output contamination from the near field region and delivers input contamination to the uniform flow of the far field.

The far field, consisting of a one dimensional, unconfined aquifer with a plane sloping bottom, is modeled first using a method of characteristics approach in the assumed absence of dispersion and downgradient dilution. The far field model describes the temporal concentration variation of linearly adsorptive, first order reactive contaminants experienced by an observer moving away from the source plane at a speed modified by recharge, head loss, bottom slope, and linear adsorption. The far field concentration at a given place and time is then a function of the observer's time of departure and departing concentration at the source plane.

These source conditions correspond to output conditions for the near field region under the landfill. This more complicated zone is idealized as a single linear reservoir receiving input from a constant user population and a constant per capita contaminant generation rate, or loading factor. The resulting linear reservoir routing equation expresses the far field source concentration in terms of near field time and the contaminant loading factor, so that the coupling of near and far field analyses relates contaminant plume distribution to prior surface application of solid waste. A simple modification of the theory to simulate contamination from an infiltration bed completes the model development. In the latter regard, infiltration beds alter the local groundwater hydraulics by injecting a locally significant flow of water into the subsurface environment [Hantush (1967), Hanson and Brock (1984)]. Following Ostendorf (1986), an "equivalent landfill" is used to represent these sources of pollution, with dimensions dependent on the amount of flow injected. The report subsequently cites data requirements for contaminant transport model usage and suggests possible sources of information.

Two case studies illustrate the applicability of the contaminant transport model to sites in the Commonwealth. The decommissioned sanitary landfill in Amherst, Massachusetts is successfully modeled for chloride and specific conductance, using the relatively sparse data reported by Metcalf and Eddy (1976), Lederer (1983), and the Commonwealth of Massachusetts [DEQE (1986)]. LeBlanc (1984) presents a more comprehensive set of measurements describing chloride, total nitrogen, and boron concentrations downgradient of the infiltration beds at Otis Air Force Base in Barnstable County, Massachusetts. The modified transport model is tested against these data with reasonably accurate results as well. A series of applications further illustrates the modeling procedure:

Source History from an Existing Reactive Plume

Post Shutdown Recovery of Groundwater Quality

Infiltration Bed Dilution Constraint

The last application demonstrates the relatively poor mixing characteristics of the groundwater flow field, compared to its surface water counterpart.

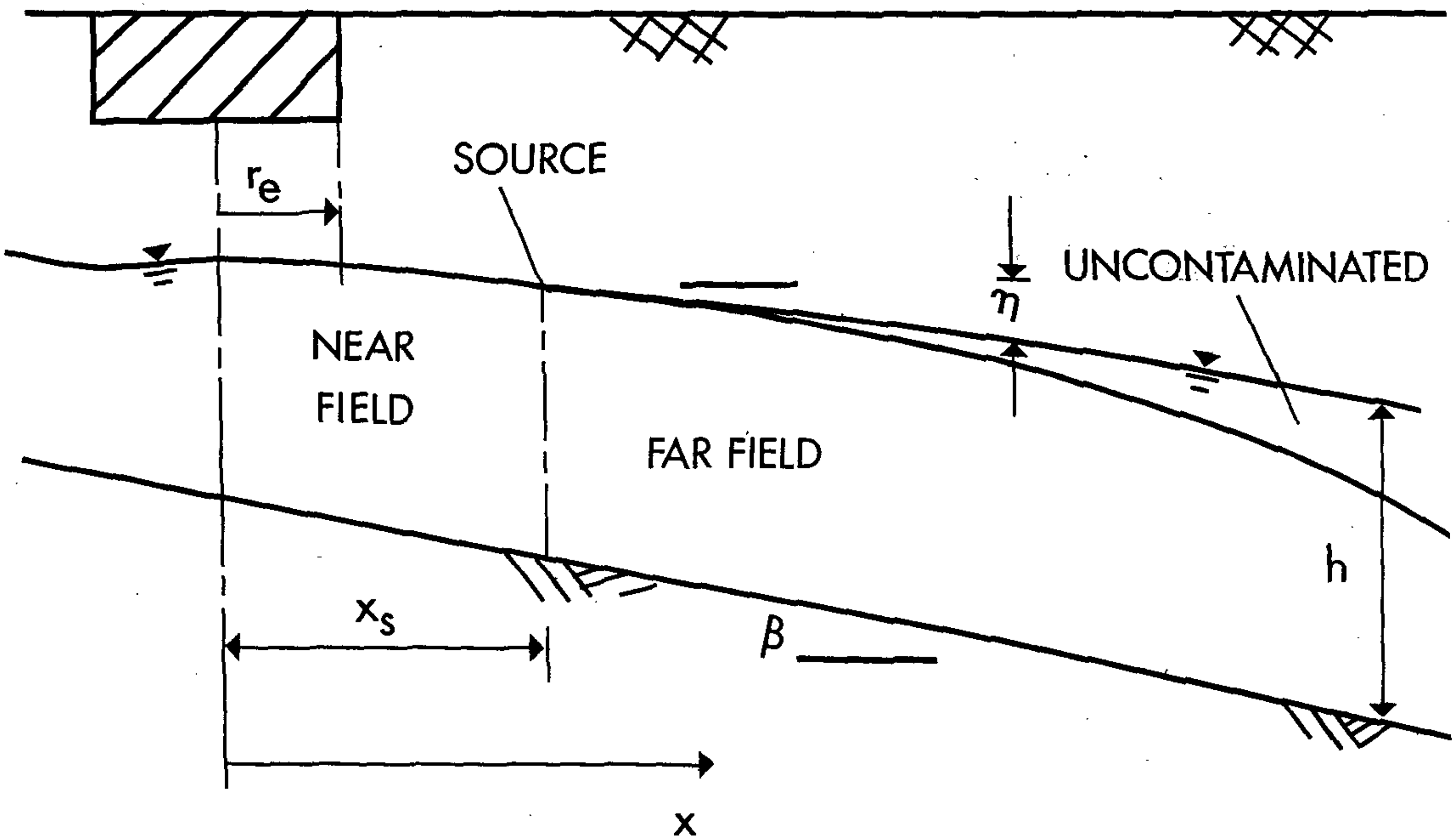


FIGURE 1 - DEFINITION SKETCH

Dilution factors less than 5 in magnitude are feasible in the subsurface, while factors from 10 to 100 are common in rivers and lakes [Fischer et al. (1979)].

The report concludes with a model summary and some suggested avenues of future research.

1.3 Literature Review

The use of linear reservoirs in surface water hydrology has been long established [Dooge (1973)] and the proposed groundwater application of the concept in the near field has more recent precedence in the work of Gelhar and Wilson (1974) and Mercardo (1976), who successfully describe groundwater pollution due to distributed contaminant input.

The literature distinguishes analytical and numerical descriptions of far field subsurface contaminant transport. The numerical modelers [Pinder (1973), Bachmat et al. (1980), Hwang et al. (1985), Huyakorn et al. (1986)] retain all terms in the conservation equations governing the process by simulating differential equations 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. The complexity of these models is warranted for detailed study of well documented episodes of pollution, with data sufficient to resolve the plume in two or three dimensions. An analytical approach, on the other hand, is called for in the usual case of a sparse data set describing contamination in one dimension. This method solves simplified differential equations explicitly, obviates the computer, and yields a generic, simple, and physically valid model appropriate in the preliminary planning and assessment context of the present investigation.

There are 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 contamination 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.

The continuous, spatially distributed landfill leachate contamination considered in this report yields small concentration gradients and consequent domination of dispersion by the advective and reactive transport mechanisms. The resulting neglect of dispersion yields a method of characteristics analysis that permits additional realism either in the nature of the reactive constituents [Charbeneau (1981)] or in the hydraulics of the aquifer [Ostendorf et al. (1984)]. The latter approach is adopted in the present model, and the effects of recharge, head loss, and a sloping underlying aquiclude are incorporated into the unconfined aquifer flow field. Like Wilson and Miller (1978,1979), Bear (1979), and Prakash (1982), the reactive contaminants under study are assumed to be linearly adsorptive or to exhibit first order decay in the far field. The method of characteristics approach adopted in this study has also been used by Wilson and Gelhar (1981) and Bredehoeft and Pinder (1973) in analytical and numerical accounts of contaminant migration through the unsaturated and saturated zones, respectively.

CHAPTER 2

CONTAMINANT TRANSPORT MODEL

2.1 Far Field Governing Equations

The transport of contaminants through the far field is described by coupled equations of the groundwater flow (hydraulics) and pollutant flow. The hydraulic component must be described first, since groundwater carries the contamination through the subsurface environment by advection and dispersion. This hydraulic model must be simple, since it will be input to a more complicated contaminant counterpart, and analytical solutions are desired for both governing equations.

The groundwater flow is accordingly assumed to be steady in this report. The assumption is quite reasonable since the unconfined aquifer time scale is well over a year in magnitude, and seasonal fluctuations of subsurface hydraulics cancel out over this period. The steady conservation of water mass which governs the one dimensional unconfined aquifer flow is a balance of net efflux and recharge ε through a control volume

$$\frac{dq}{dx} - \varepsilon = 0 \qquad (1)$$

with discharge q per unit width and distance x downgradient of the center of the landfill, as suggested by Figure 1. With a constant recharge, equation 1 may be integrated from the source plane location, where conditions are denoted by an s subscript, to any position in the far field with the result

$$q = q_s + \varepsilon(x-x_s) \qquad (2)$$

The discharge per unit width and the average linear velocity v in the x direction are related by definition [Freeze and Cherry (1979)]

$$v = \frac{q}{nh} \tag{3}$$

with porosity n defined as the void volume divided by the total volume, and aquifer depth h. The average linear velocity is the actual speed of the water molecules through the unconfined aquifer, and will consequently be the speed of the contaminant molecules as well. Figure 1 indicates that the aquifer depth is comprised of its source value, modified by sloping top and bottom surfaces

$$h = h_s + (x-x_s)\tan\beta - \eta \tag{4}$$

The bottom slope angle β allows for a sloping underlying aquiclude, while the water table depression η below its source position is due to head losses incurred by the flow through the aquifer. The depression may be approximated using the observed water table slope at the source plane in accordance with

$$\eta = (x - x_s)\left(\frac{d\eta}{dx}\right)_s \tag{5}$$

The hydraulic component of the the far field model reduces to a description of the average linear velocity, obtained by combining equations 2-5 and solving for v, with the result

$$v = v_s\left[\frac{1 + \dfrac{\varepsilon(x-x_s)}{q_s}}{1 + \dfrac{(x-x_s)\gamma}{h_s}}\right] \tag{6a}$$

$$\gamma = \tan\beta - \left(\frac{d\eta}{dx}\right)_s \tag{6b}$$

The far field transport of a simply reactive contaminant may be considered once the hydraulics are specified. The analysis simplifies considerably when the complicating effects of dispersion are neglected. This transport process smooths out sharp discontinuities of contaminant concentration and is accordingly important in the study of accidental "spikes" of pollution. The concentration gradients will be small for the continuous, long term sources of contamination considered in the present investigation however, so that dispersion can be safely neglected [Ostendorf et al. (1984)]. Far field dilution of the plume will also be ignored, following the suggestion of Kimmel and Braids (1980), who note the presence of an uncontaminated recharge lens of slightly lower density above the contaminated flow. Ostendorf et al. (1984) state that the resulting one dimensional conservation of contaminant mass equation reduces to a balance of storage change, advection, and reaction

$$\frac{\partial c}{\partial t} + \frac{v}{R}\frac{\partial c}{\partial x} = -\frac{\lambda c}{R} \tag{7}$$

with concentration c and time t. The retardation factor R characterizes linear adsorption of the contaminant onto the soil grains [Freeze and Cherry (1979)], while the first order decay constant λ reflects the assumed reactive kinetics of the contaminant. The behavior of a conservative (nonreactive) substance may be recovered from the governing equation by setting R and λ equal to 1 and 0 respectively in the corresponding solution.

The pollutant transport process posed by equation 7 can be conveniently considered by the method of characteristics [Eagleson (1970)], which effectively divides the analysis into two simpler problems of observer trajectory and observer concentration. The concentrations governed by equation 7 are simply experienced by an observer leaving the source plane at time t_s with a speed dx/dt given by

$$\frac{dx}{dt} = \frac{v}{R} \tag{8}$$

The path of the moving observer follows upon of integration of equations 6 and 8 from starting conditions (x_s, t_s) at the source plane to any subsequent place x and time t in the far field. The resulting observer trajectory is given by [Gradshteyn and Ryzhik (1965)]

$$t-t_s = \frac{Rn}{\varepsilon} \{(x-x_s)\gamma + (h_s - \frac{q_s\gamma}{\varepsilon})\ln[1 + \frac{\varepsilon(x-x_s)}{q_s}]\} \tag{9}$$

The moving observer occupying this path will witness a time variation dc/dt of contaminant concentration described by

$$\frac{dc}{dt} = -\frac{\lambda c}{R} \tag{10}$$

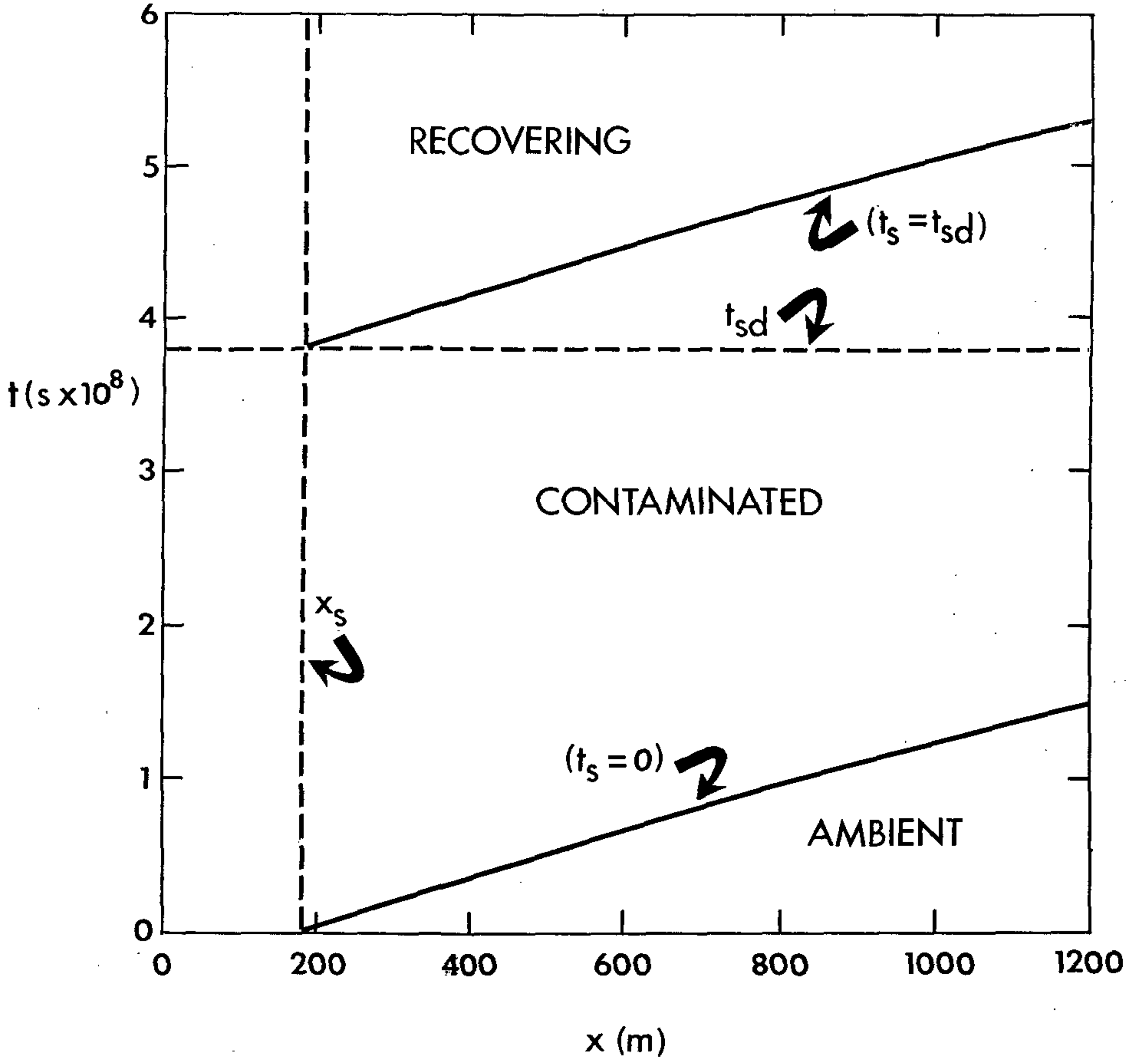
The concentration accompanying the observer into the far field likewise depends on starting conditions at the source plane, as equation 10 is integrated from c_s, t_s to any subsequent c,t with the simple result

$$c = c_s \exp\left[\frac{\lambda(t_s - t)}{R}\right] \tag{11}$$

Figure 2 displays representative paths established by equation 9 for two observers of interest, each identified by a time of departure t_s on the t axis. The observer leaving at the onset (t_s equals zero) of source contamination follows the front of pollution and traces out a path separating pure and contaminated groundwater in the far field. Equation 9 suggests that this observer, who brings the message of impending aquifer pollution, follows a path specified by

$$t = \frac{Rn}{\varepsilon} \{(x-x_s)\gamma + (h_s - \frac{q_s\gamma}{\varepsilon})\ln[1 + \frac{\varepsilon(x-x_s)}{q_s}]\} \quad (t_s=0) \tag{12}$$

Source concentrations will intensify until the time of shutdown t_{sd}. The observer departing the source plane at this time describes a path marking



Note: Observers leave x_s at time t_s.

Amherst landfill paths.

FIGURE 2 - PATHS OF FAR FIELD OBSERVERS

the beginning of aquifer recovery, as indicated by appropriate substitution into equation 9

$$t = t_{sd} + \frac{Rn}{\varepsilon}\{(x-x_s)\gamma + (h_s - \frac{q_s\gamma}{\varepsilon})\ln[1 + \frac{\varepsilon(x-x_s)}{q_s}]\} \qquad (t_s=t_{sd}) \qquad (13)$$

Restating matters, equations 9 and 11 describe the far field concentration c at a far field location x and far field time t for a specified observer who left the source plane (location x_s) at time t_s with source concentration c_s. These latter quantities depend on the near field behavior of the landfill and aquifer.

2.2 Near Field Reservoir

The landfill constitutes a distributed input of width b and length ζ in the direction of groundwater flow to a linear reservoir whose output comprises c_s. Following Ostendorf et al. (1984), the near field conservation of contaminant mass equation is a balance of storage change, pollutant outflux, and pollutant influx

$$\zeta n h_s R \frac{dc_s}{dt_s} + q_s c_s - q_s c_a - \frac{SP}{b} = 0 \qquad (0<t_s<t_{sd}) \qquad (14)$$

The influx of contamination from upgradient ambient flow has an ambient concentration c_a, assumed constant for this study. The polluted landfill input is assumed to be simply related to the user population P by a constant contaminant loading factor S, representing the contaminant generation rate per capita. The contaminant inflow begins at source time t_s equals 0 and flow persists until the time of shutdown t_{sd}. The loading factor constancy reflects presumedly rapid leachate generation due to precipitation and solid waste interaction, in contrast to the slower near field response time governing the hydraulics under the landfill. The near field response time is in turn much smaller than both the far field time scale and the reactive decay time scale represented by $1/\lambda$. The latter two periods must be of the same order of magnitude if concentrations are appreciable in the far field; the first order reactions are therefore negligibly slow in the relatively fast flow field under the landfill, and λ is ignored in this region.

Equation 14 may be conveniently expressed in a more compact form

$$\frac{dc_s}{dt_s} + \frac{c_s}{t_c} = \frac{c_{sf}}{t_c} + \frac{c_a}{t_c} \qquad (0<t_s<t_{sd}) \qquad (15)$$

with near field response time t_c and steady state source concentration c_{sf} (above ambient) defined by

$$t_c = \frac{R\zeta}{v_s} \qquad (16a)$$

$$c_{sf} = \frac{SP}{bq_s} \qquad (16b)$$

This linear first order differential equation is solved subject to the initial condition of ambient concentration

$$c_s = c_a \qquad (t_s=0) \qquad (17)$$

The solution to this nonhomogeneous problem may be obtained from Rainville and Bedient (1969)

$$c_s = c_{sf}\,[1-\exp(-\frac{t_s}{t_c})] + c_a \qquad (0<t_s<t_{sd}) \qquad (18)$$

Equation 18 suggests that concentrations leaving the near field will increase until the time of shutdown t_{sd}. At this time, the governing near field equation will change to reflect the absence of pollutant input from the landfill; thus equation 15 will become

$$\frac{dc_s}{dt_s} + \frac{c_s}{t_c} = \frac{c_a}{t_c} \qquad (t_s>t_{sd}) \qquad (19)$$

This homogeneous, first order differential equation must be solved subject to a matching condition at the time of shutdown

$$c_s = c_{sd} + c_a \qquad (t_s = t_{sd}) \qquad (20a)$$

$$c_{sd} = c_{sf}[1 - \exp(-\frac{t_{sd}}{t_c})] \qquad (20b)$$

The shutdown concentration c_{sd} at the source plane follows from equation 18; it marks the onset of groundwater quality recovery and is also the highest concentration of pollution in the aquifer.

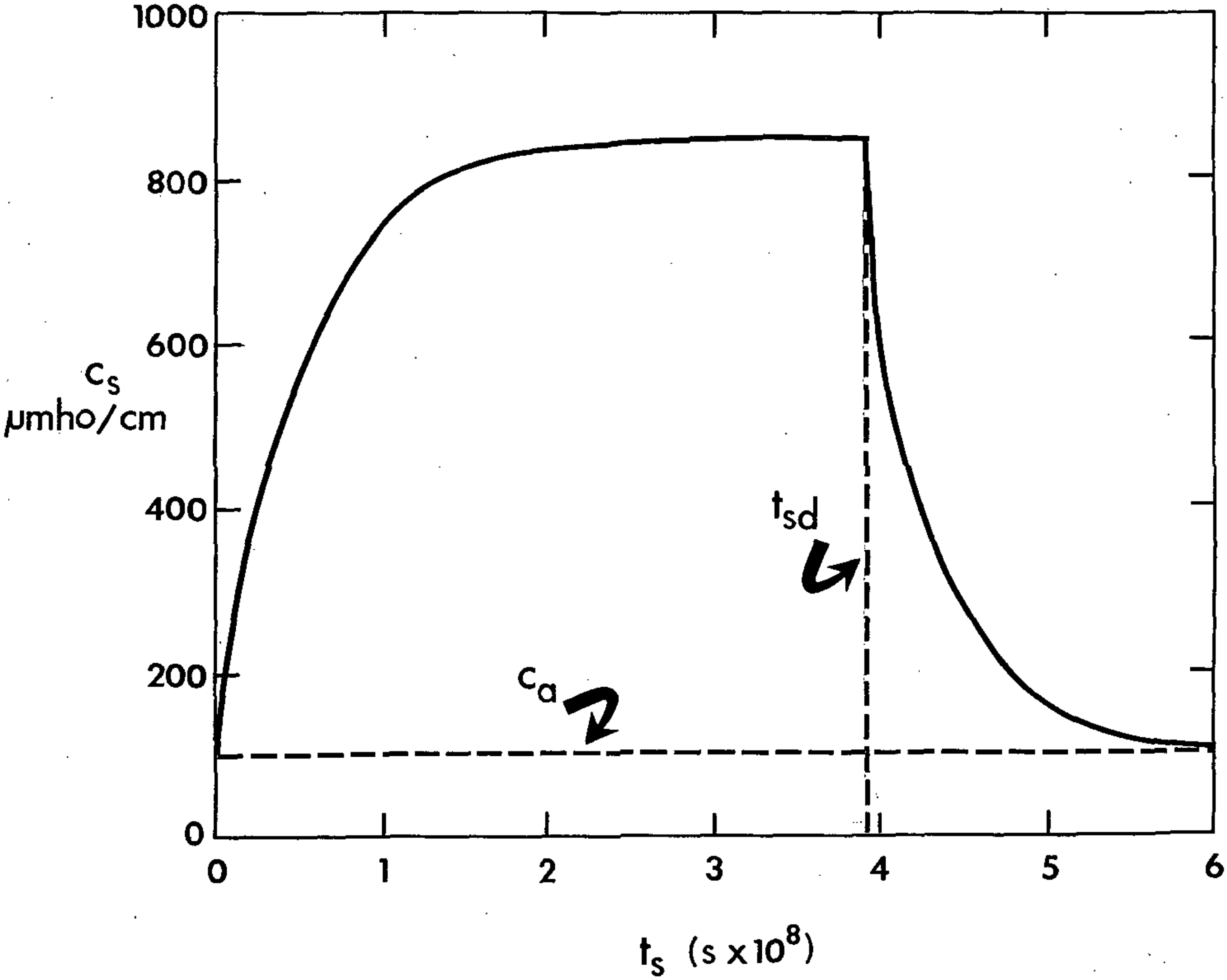
The solution to equations 19 and 20 is straightforward

$$c_s = c_{sd} \exp[\frac{(t_{sd} - t_s)}{t_c}] + c_a \qquad (t_s > t_{sd}) \qquad (21)$$

Equation 21 suggests that the recovery of aquifer water quality after shutdown of the landfill will be gradual due to the slow discharge of contamination accumulated in the near field reservoir. Figure 3 displays the source plane concentration predicted by equations 18 and 21 for a typical landfill, while Figure 4 indicates its subsequent use in the far field. Physically speaking, Figure 3 records concentrations experienced by a stationary observer located at the downgradient edge of the landfill, where near and far fields meet. It may also be construed as the locus of starting conditions for moving observers traveling into the far field. Figure 4 shows the concentration experienced by one such observer, leaving the source plane at time t_s and moving into the far field along a path described by equation 9. The sketched concentration variation is simply equation 11.

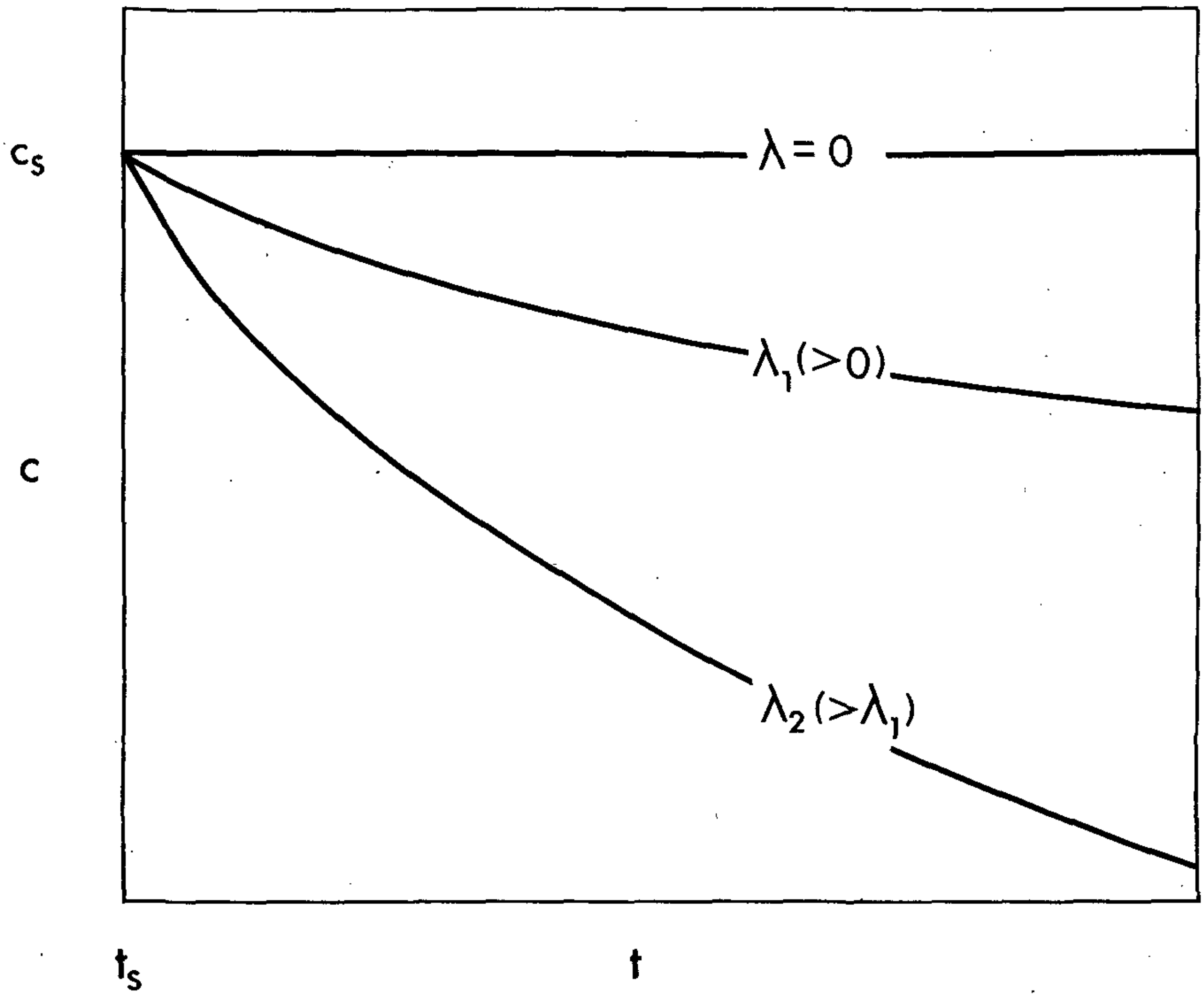
2.3 Infiltration Bed Modification

Ostendorf (1986) demonstrates that the foregoing near field reservoir model may be simply modified to account for an artificially high rate of



Note: Concentrations are for specific conductance.

FIGURE 3 - SOURCE PLANE CONCENTRATIONS
(AMHERST LANDFILL)



Note: Observer departs source plane at time t_s.

FIGURE 4 - FAR FIELD CONCENTRATIONS
FOR A MOVING OBSERVER

recharge, representing the effluent from an infiltration bed. The strong artificial recharge from the infiltration bed alters the hydraulics of the near field by superimposing a radial velocity component onto the unidirectional ambient flow. The resulting stream lines ψ, which describe the path of water and contaminant particles, are governed by

$$\psi = - \frac{y}{r_e} + \alpha \tan^{-1} \frac{x}{y} \tag{22}$$

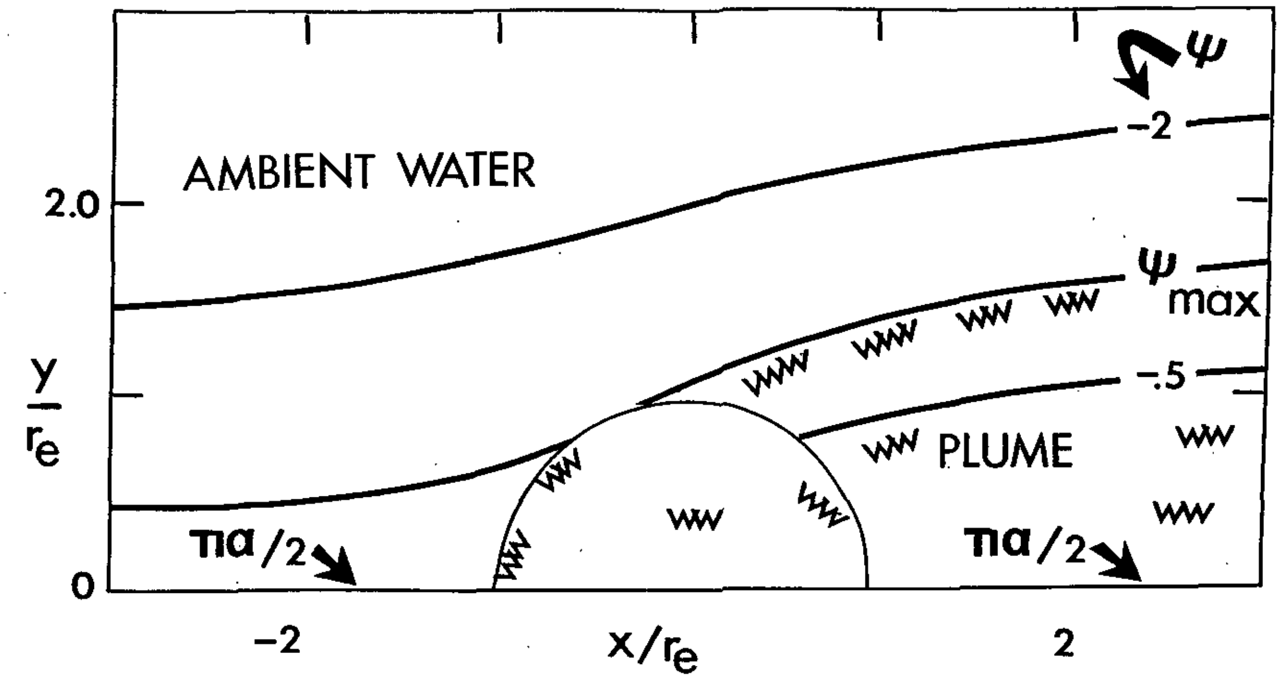
and are sketched in Figure 5. Here r_e is the radius of the circle which circumscribes the beds, y is lateral distance from the plume centerline, and α is the effluent ratio, a measure of the relative strength of the effluent volumetric discharge rate Q_e

$$\alpha = \frac{Q_e}{2\pi r_e q_s} \tag{23}$$

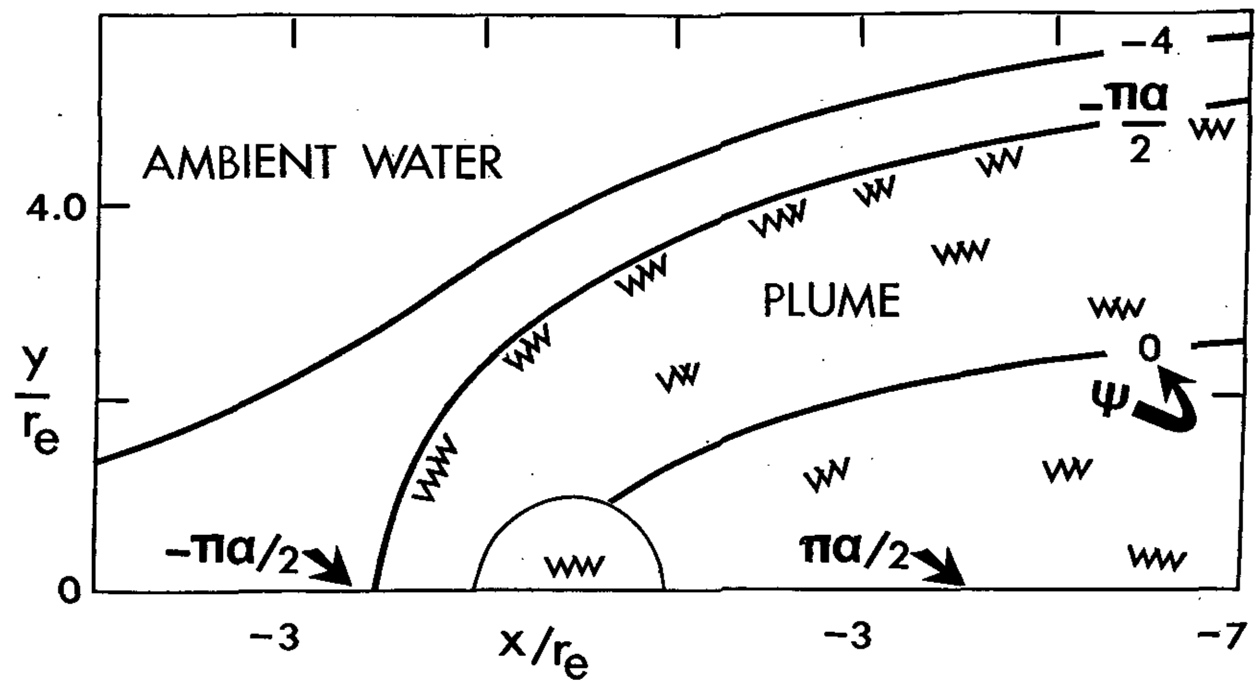
The use of a circular infiltration bed yields a simpler flow field than its rectangular counterpart [Hantush (1967)] and is accordingly in keeping with the approximate spirit of the present model. The source is distant from the origin and is properly represented by such an assumption, since the superimposed hydraulics become insensitive to bed geometry with increasing radial distance from the facility.

Figure 5 indicates that the largest negative stream line ψ_{max} emerging from the infiltration bed marks the lateral extent of pollution. ψ_{max} thus separates clean and polluted water, and defines the plume in the near field. Ostendorf (1986) derives an analytical estimate of the value of this important stream line

$$\psi_{max} = -(1-\alpha^2)^{1/2} - \alpha \sin^{-1}\alpha \qquad (\alpha<1) \tag{24a}$$



5a - EFFLUENT RATIO = 0.50



5b - EFFLUENT RATIO = 2.00

FIGURE 5 - INFILTRATION BED STREAMLINES

$$\psi_{max} = -\frac{\pi\alpha}{2} \qquad (1<\alpha) \qquad (24b)$$

The trajectory of ψ_{max} follows upon substitution of equation 24 in 22. As sketched in Figure 5, the plume boundary becomes parallel to the x axis with increasing distance away from the infiltration bed, reflecting the diminishing importance of the radial velocity component induced by the effluent stream and a return to one dimensional flow. Ostendorf (1986) uses this behavior to arbitrarily define a source plane location by the implicit relations

$$\frac{y_s}{r_e} = -\psi_{max} + \alpha \cos^{-1}\left[\left(\frac{y_s}{5\alpha r_e}\right)^{1/2}\right] \qquad (25a)$$

$$\frac{x_s}{r_e} = \frac{y_s}{r_e}\tan\left(\frac{\psi_{max}}{\alpha} + \frac{y_s}{r_e\alpha}\right) \qquad (x_{max}>r_e) \qquad (25b)$$

$$\frac{x_s}{r_e} = 1 \qquad (x_{max}<r_e) \qquad (25c)$$

The resulting source plane location is sketched in Figure 6 as a function of the effluent ratio and the infiltration bed radius for ease in model usage.

The plume width at this location corresponds to b, which can be considered as the width of an "equivalent" landfill. Ostendorf (1986) deduces a value for this parameter as well

$$b = (\pi\alpha - 2\psi_{max})\, r_e \qquad (26)$$

The plume width is also sketched in Figure 6 as a function of the effluent ratio and the infiltration bed radius: stronger effluent flows will generate plumes with widths many times greater than the bed radii. The length of the equivalent landfill used to represent the infiltration bed may be simply represented by

$$\zeta = 2x_s \qquad (27)$$

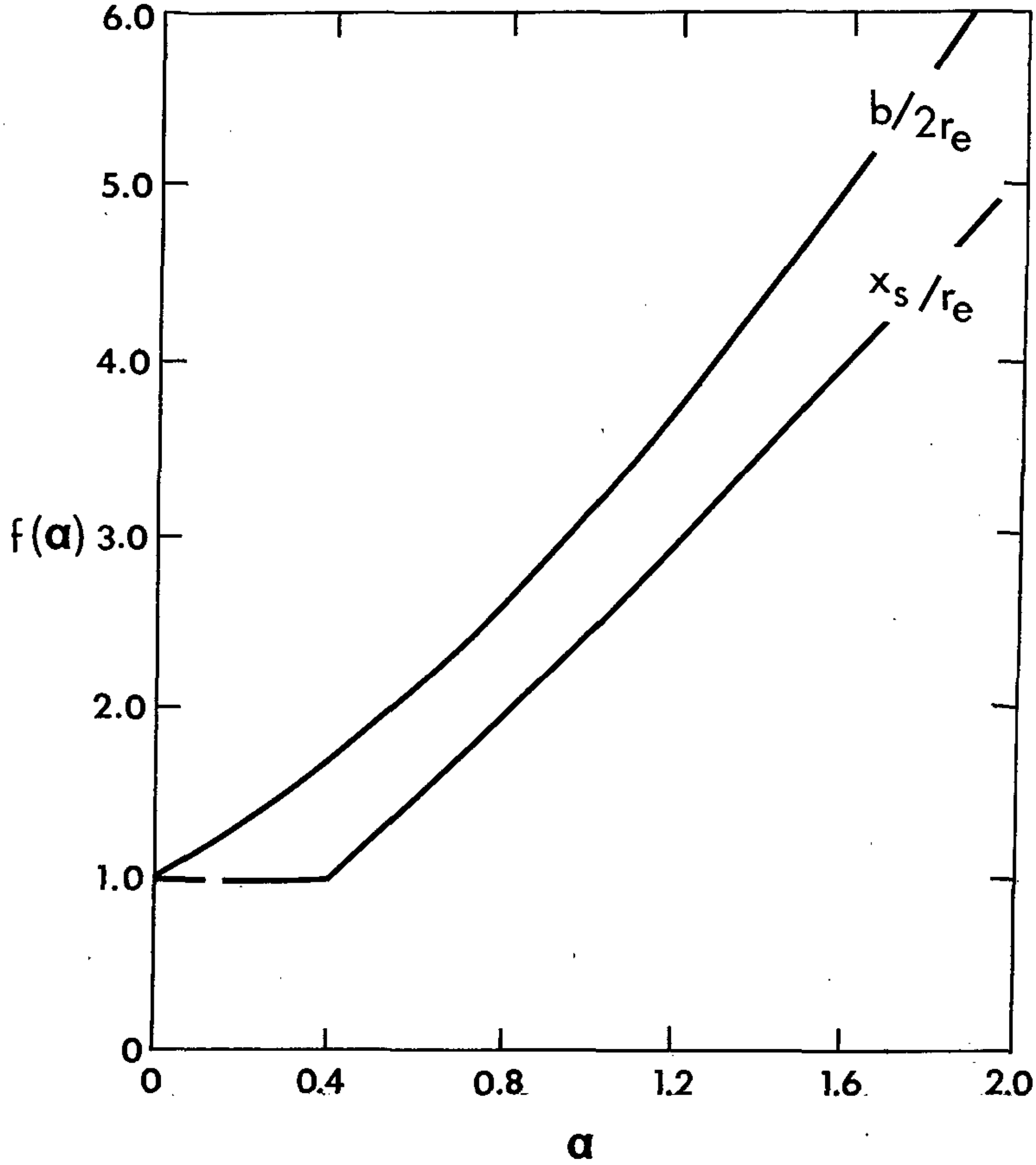


FIGURE 6 - EQUIVALENT LANDFILL DIMENSIONS

The equivalent landfill dimensions represented by equations 26 and 27 reduce to the actual geometry of the pollutant source as the effluent ratio goes to zero. Thus, a landfill may be regarded as a limiting case of weak infiltration.

The infiltration bed analysis corresponds closely to its landfill counterpart from this point forward. The only difference lies in the nature of the input to the reservoir in the conservation of contaminant mass equation

$$\zeta n h_s R \frac{dc_s}{dt_s} + q_s c_s - \frac{Q_e c_e}{b} - (q_s - \frac{Q_e}{b}) c_a = 0 \qquad (28)$$

In this case, the effluent concentration c_e and volumetric discharge are specified, and the latter quantity contributes appreciably to the groundwater flow. Equation 28 reduces to equation 15, provided c_{sf} is redefined as

$$c_{sf} = \frac{c_e - c_a}{D} \qquad (29a)$$

$$D = \frac{bq_s}{Q_e} \qquad (29b)$$

Since the initial and matching conditions for the infiltration bed and landfill analyses are identical, as are the governing equations, the solutions will correspond exactly. Thus equations 18 and 21 describe the infiltration bed source plane condition, with c_{sf} given by equation 29a.

D is a dilution factor representing the ratio of effluent inflow and source plane outflow, as indicated by equation 29b. The mixing, which takes place in the near field reservoir, reduces the steady state concentration leaving the near field in accordance with equation 29a. The distinctly different mixing behavior of surface and groundwater environments is illustrated by the behavior of the strong infiltration case, in which α is greater than unity. Equation 24b corresponds to the case of a strong source of infiltration, distinguished by the high effluent ratio. This is the usual case in view of the modest discharges associated with gradual flow of water in the subsurface zone. Consideration of equations 23, 26, and 29b reveals that the effluent Q_e and source plane bq_s discharges will be equal when α exceeds unity, so that strong infiltration will experience no dilution in the near field. This case is illustrated in Figure 5b: all polluted stream lines originate from within the infiltration bed. A strong pollutant source will experience little or no mixing in the far field either, so that effluent concentrations will persist at full strength in the groundwater environment, unless modified by reactions. This case strongly contrasts with conventional surface water effluent behavior, which relies strongly on mixing with the ambient river, lake, or ocean water to reduce effluent concentrations by dilution factors from 10 to 100 in magnitude [Fischer et al. (1979)].

CHAPTER 3

DATA REQUIREMENTS AND SOURCES

3.1 Geology

The unconfined aquifer porosity, permeability, and bottom slope specify the geology of the site area. Surficial geology maps published by the United States Geological Survey [Clarke et al. (1982)] identify the rock or soil type of the unconfined aquifer and provide rough estimates of the porosity and permeability with the aid of Table 1 [Freeze and Cherry (1979)]. The latter parameter relates the water table slope to the average linear velocity, and plays an important role in establishing aquifer hydraulics, as discussed below. The bottom slope marking the interface between the overlying permeable unconfined aquifer and underlying impermeable aquiclude must be inferred from cross-sectional information obtained from deep wells and bedrock geology maps [Clarke et al. (1982)]. Existing USGS data are summarized on a computerized system WATSTORE [Clarke et al. (1982)], while state water resource agencies [Giefer and Todd (1972)] provide additional information as well. Local studies offer a third source of data.

3.2 Hydraulics

The aquifer thickness, water table slope, and recharge area establish site hydraulics. With the elevation of the aquifer/aquiclude interface estimated as a geologic parameter, the aquifer thickness requires shallow well data to elucidate the water table elevation. The USGS shallow well data, which is summarized on the WATSTORE computer system [Clarke et al. (1982)],

TABLE 1 - AQUIFER POROSITY AND PERMEABILITY ESTIMATES

Soil or Rock Type	Porosity n	Permeability k (m^2)
Unconsolidated Deposits		
Gravel	0.25 - 0.40	10^{-7} - 10^{-10}
Sand	0.25 - 0.50	10^{-9} - 10^{-12}
Silt	0.35 - 0.50	10^{-12} - 10^{-16}
Clay	0.40 - 0.70	10^{-16} - 10^{-19}
Consolidated Material		
Fractured Basalt	0.05 - 0.50	10^{-11} - 10^{-15}
Karst Limestone	0.05 - 0.50	10^{-9} - 10^{-13}
Sandstone	0.05 - 0.30	10^{-13} - 10^{-16}
Limestone, Dolomite	<0.01 - 0.20	10^{-13} - 10^{-16}
Shale	<0.01 - 0.10	10^{-16} - 10^{-20}
Fractured Crystalline Rock	<0.01 - 0.10	10^{-11} - 10^{-15}
Dense Crystalline Rock	<0.01 - 0.05	10^{-17} - 10^{-22}

Source: Freeze and Cherry (1979).

will also provide the water table slope, supplemented by municipal and private well records. The slope yields an estimate of the discharge per unit width through the source plane through Darcy's law

$$q_s = \frac{kgh_s}{\nu} \left(\frac{dn}{dx}\right)_s \qquad (30)$$

with gravitational acceleration g, permeability k, and fluid kinematic viscosity ν. The gravitational acceleration is a physical constant equal to 9.81 m^2/s in magnitude, while the kinematic viscosity varies directly with the temperature of the groundwater between 1×10^{-6} and 1.4×10^{-6} m^2/s [Streeter and Wylie (1979)].

The recharge area A upgradient of the source plane may be used to estimate the recharge ε when the discharge leaving the near field is known. The steady conservation of water mass requires that the source plane discharge and recharge flow be equal, so that

$$q_s b = \varepsilon A \qquad (31)$$

Since the source discharge may be independently computed from Darcy's law, equation 31 yields an estimate of ε. The resulting value should be checked against the average rainfall rate of 3.6×10^{-8} m/s (44 inches/year) for Massachusetts [Motts and O'Brien (1981)] to ensure a reasonable estimate. The rainfall rate corresponds to a maximum natural recharge value, since surface runoff, water use, and evapotranspiration all withdraw water from the unconfined flow field before its arrival at the landfill [Linsley et al. (1982)].

Surficial geology maps [Clarke et al. (1982)] may be used to delineate the recharge area and to identify streams for purposes of surface runoff estimation. In the latter regard, Water Supply Papers [(USGS 1986)] document major stream and tributary flow values. The evapotranspiration rate in general will not exceed the rate of evaporation from a water surface, and Linsley et al. (1982) suggest a value of 2.2×10^{-8} m/s (27 inches/year) for this parameter in the Commonwealth. Many factors, such as wind speed, net radiation, plant type, relative humidity, and air temperature determine evapotranspiration [Eagleson (1970)], so that considerable effort would be needed to refine this estimate of water loss. Groundwater withdrawals from the recharge area may be inferred from the distribution of public and domestic shallow supply wells in the upgradient zone. In the absence of this data, a crude estimate may be obtained by multiplying the water user population by a domestic usage rate of 7.0×10^{-6} m^3/sec-cap (160 gallons per capita per day) [Viessman and Welty (1985)].

The best check of aquifer hydraulics, however, is the observed distribution of the plume itself, since the front of the conservative contaminant plume is advected by the average linear velocity. The predicted path of the pollutant front is traveled by the observer leaving the source plane at time t_s equals zero, and is given by equation 12. With x set at the observed present location of the leading edge of the plume, and t established by the present time, equation 12 yields an implicit check on both ε and q_s. In practice, Darcy's law, the recharge equation, and the plume trajectory are used concurrently to establish an optimal estimate of the aquifer hydraulics.

3.3 Source Conditions

Landfill life and dimensions, user population, and contaminant loading factors determine the source conditions for the landfill model, while effluent discharge and concentration characterize the infiltration bed modification, along with the bed radius. The reactive constants of the non-conservative contaminants complete the set of parameter values needed for transport model use.

The size of the landfill may be readily measured in the field or from topographical maps [Clarke et al. (1982)], while US Census (1977) data, supplemented by local information, specifies user population for the history or projected life of the facility. The infiltration bed equivalent radius corresponds to the radius of a circumscribing circle around the unit [Ostendorf (1986)]. The effluent flow from the infiltration bed is a commonly reported parameter, particularly in view of current MDWPC initiatives towards standardized reporting procedures [Noss et al. (1984)]. In the absence of definitive effluent flow data, Q_e may be estimated by multiplying the user population by a wastewater generation rate of 5.5×10^{-6} m^3/s-cap (125 gallons per capita per day) [Tchobanoglous and Schroeder (1985)], excluding industrial and institutional users. Effluent concentrations should be read from operating data [Noss et al. (1984)], but in the absence of such information, the untreated sewage data cited in Table 2 [Metcalf and Eddy (1972)] may be considered as an upper bound on the domestic effluent pollutant levels.

The contaminant loading factor for an existing landfill should be calibrated from groundwater quality samples over the depth of the unconfined aquifer, preferably at the downgradient boundary of the facility. The resulting source concentration c_s at time t_s may be substituted into the

TABLE 2 - UNTREATED WASTEWATER CONCENTRATIONS

Constituent	Strong	Concentration c (mg/l) Medium	Weak
Biochemical Oxygen Demand (5 d, 20°C)	300	200	100
Total Organic Carbon	300	200	100
Chemical Oxygen Demand	1000	500	250
Total Nitrogen (as N)	85	40	20
Organic	35	15	8
Free Ammonia	50	25	12
Nitrites	0	0	0
Nitrates	0	0	0
Total Phosphorus (as P)	20	10	6
Organic	5	3	2
Inorganic	15	7	4
Chlorides (above Ambient)	100	50	30
Alkalinity (as $CaCO_3$)	200	100	50

Notes: Domestic Wastewater.

Source: Metcalf and Eddy (1972).

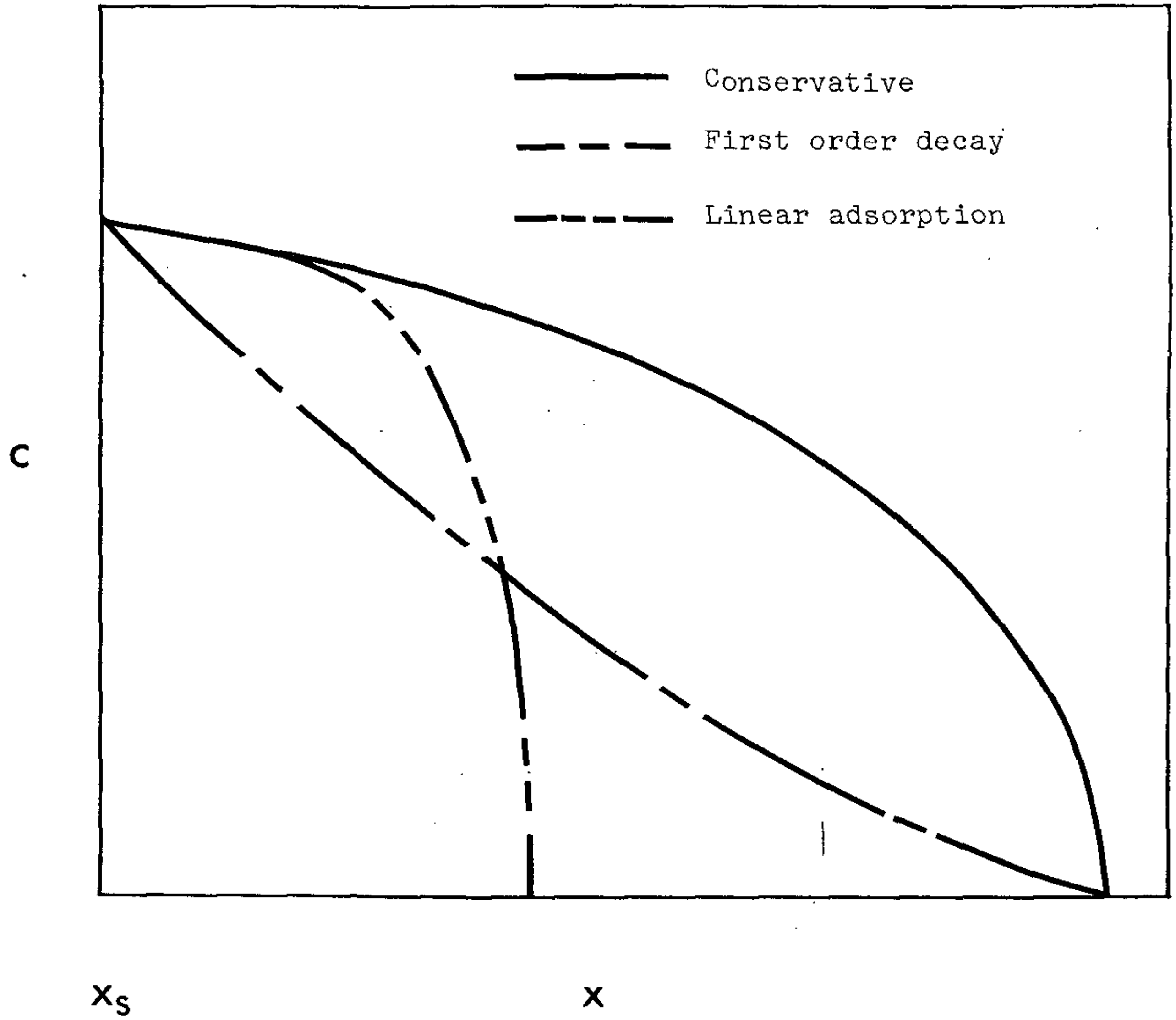
source relation (equations 16b, 18, and 21) to yield an estimate of the contaminant loading factor

$$S = \frac{bq_s(c_s - c_a)}{P[1-\exp(-\frac{t_s}{t_c})]} \qquad (t_s<t_{sd}) \qquad (32a)$$

$$S = \frac{bq_s(c_s-c_a)\exp(t_s/t_c)}{P[\exp(\frac{t_{sd}}{t_c}) - 1]} \qquad (t_{sd}<t_s) \qquad (32b)$$

Ostendorf et al. (1984) follow this procedure for a sanitary landfill at Babylon, New York for chloride and bicarbonate pollution with the resulting values listed in Table 3. The Table also includes the chloride and specific conductance loading factors derived in this study for Amherst, Massachusetts. The values provide an order of magnitude estimate of the parameters, but it should be stressed that actual loading factors are functions of refuse composition, and are accordingly site specific.

The reactive nature of the contaminant completes the specification of the pollutant source. Chloride and specific conductance are generally regarded as conservative in the groundwater environment [Kimmel and Braids (1980)] and their observed distribution consequently serves as a good test of the contaminant transport model. In the absence of detailed knowledge of the potentially complex leachate geochemistry describing reaction kinetics, the first order decay constant and retardation factor basically function as calibration constants fitting the model to observed contaminant concentrations. Ostendorf (1986) notes that two distinctly different far field distributions are represented by decay and adsorption as suggested by Figure 7. A first order decaying substance will persist over the entire length of the plume generated by a coexisting conservative pollutant, but at progressively lower relative concentrations. A linearly adsorbed substance,



Note: Time is fixed.

FIGURE 7 - REACTIVE CONTAMINANT CONCENTRATIONS

on the other hand, will exhibit full strength concentrations, but will extend a much shorter downgradient distance, due to the reduction of velocity by the retardation factor.

Table 3 cites calibrated decay constants for bicarbonate in Babylon, Long Island [Ostendorf et al. (1984)] and total nitrogen at Otis AFB [Ostendorf (1986)], along with a boron retardation factor, also observed at Otis AFB [Ostendorf (1986)]. The low (order 10^{-9} s^{-1}) magnitude of the decay constants strongly suggests that the reactions are rate limited by mixing processes with the plume. Ostendorf et al. (1984) put forth a preliminary estimate of the transverse mixing thought to be responsible for the loss of bicarbonate concentrations

$$\lambda = \frac{\alpha_T v_s}{h_s^2} \qquad (33)$$

The transverse dispersivity α_T characterizes lateral mixing in the aquifer, and varies from 0.1 m [Sykes et al. (1982)] to 10 m [Pinder (1973)] in magnitude, with a value dependent primarily on the homogeneity of the aquifer material [Freeze and Cherry (1979)]. The study of dispersivities, geochemical reactions, and mixing is the focus of much current research, so that improved estimates of α_T and physically valid λ estimates should be forthcoming in the scientific literature.

TABLE 3 - CALIBRATED REACTIVE CONSTANTS

Parameter	Site	Value
Loading Factors (S)		
Bicarbonate	Babylon Landfill[a]	2.66×10^{-5} mg-m^3/l-cap-s
Chloride	Babylon Landfill[a]	1.40×10^{-5} mg-m^3/l-cap-s
Chloride	Amherst Landfill[b]	1.62×10^{-5} mg-m^3/l-cap-s
Specific Conductance	Amherst Landfill[b]	2.09×10^{-4} μmho-m^3/cm-cap-s
First Order Decay Constants (λ)		
Bicarbonate	Babylon Landfill[a]	6.70×10^{-10} s^{-1}
Total Nitrogen	Otis AFB Bed[b]	1.71×10^{-9} s^{-1}
Methyl Blue Active Substances	Otis AFB Bed[c]	2.35×10^{-9} s^{-1}
Retardation Factors (R)		
Boron	Otis AFB Bed[b]	1.33

Source: [a]Ostendorf et al. (1984).
[b]This report.
[c]Ostendorf (1986).

CHAPTER 4

CASE STUDIES

4.1 Amherst, Massachusetts Decommissioned Landfill

The contaminant transport model may be applied to the sparse data base describing the leachate plume emanating from the decommissioned sanitary landfill in Amherst, Massachusetts. The application is an exercise in calibration of loading factors for two conservative species observed at the site: specific conductance and chloride. Success of the model calibration rests in the smallness of the error standard deviation associated with the calibration and the correlation of values with loading factors from other sites.

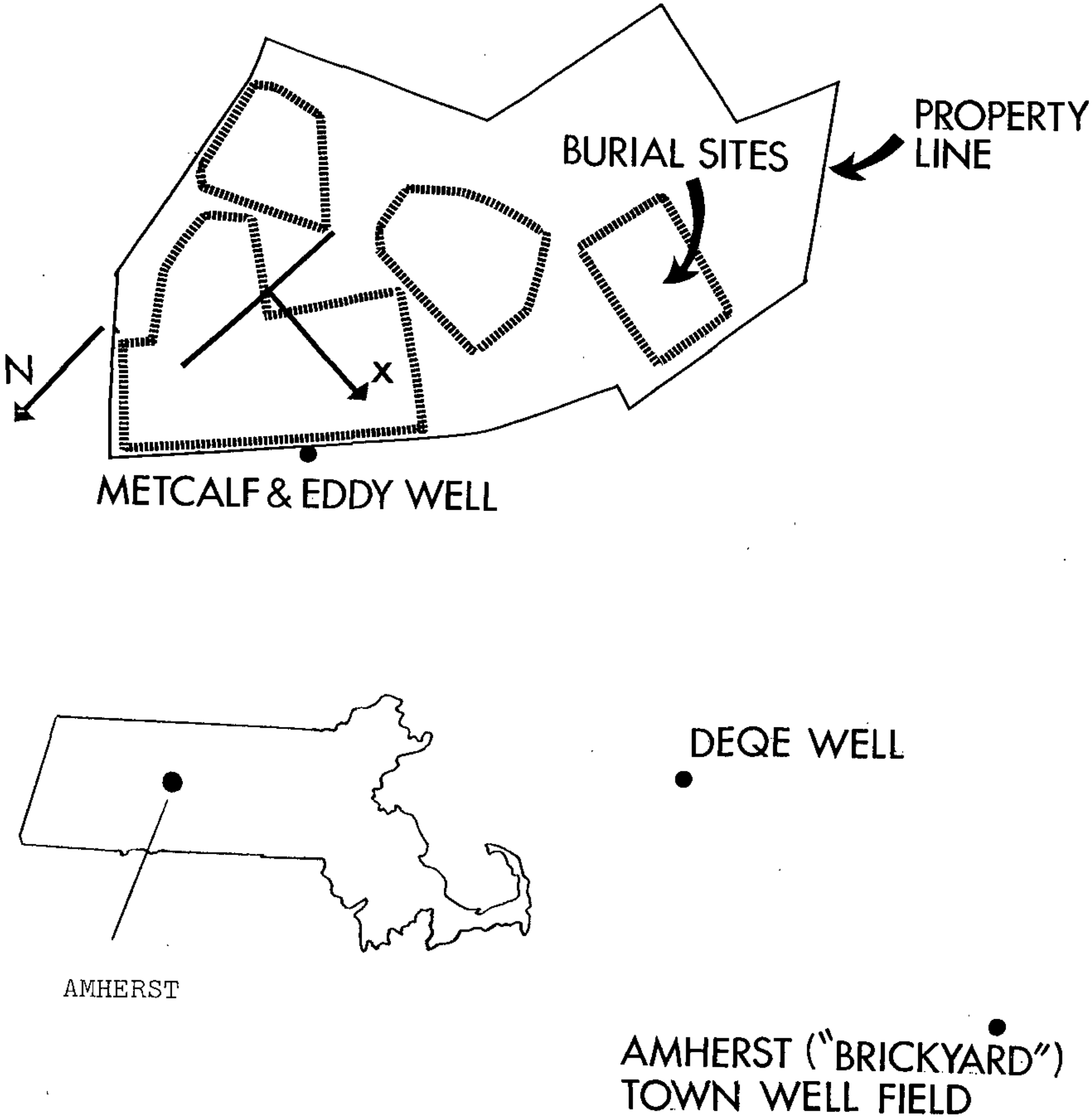
The decommissioned sanitary landfill of the Town of Amherst, Massachusetts is located on the southwest side of Massachusetts Route 9, near the Belchertown border. The landfill began operations in 1971 serving both the Town population of 34,500 [Lederer (1983)] and the University community of 25,000, half of whom live on campus. The population figures are representative for the life of the landfill, which ceased operation in December 1982, and is presently (June 1986) in the final phases of a capping procedure designed to minimize infiltration. A user population (P) of 47,000 will consequently be used in the model application, along with an assumed January 1971 temporal origin and a time of shutdown (t_{sd}) equal to 3.78×10^8 s.

The landfill is located atop an upland terrace comprised of predominantly fine sand, with subordinate amounts of coarse sand and gravel, according to Metcalf and Eddy (1976), who conducted a study of the landfill and associated plume for the Town in response to synthetic organic chemical contamination observed in the municipal wells located 1000 m downgradient of the facility [Lederer (1983)]. Lederer (1983) estimates a porosity (n) value of 0.30 and a permeability (k) of 1.43×10^{-11} m^2, based on the soil type (using Table 1) and reported hydraulics, respectively. In the latter regard, Metcalf and Eddy (1976) cite a water table elevation of about 66 m above mean sea level under the landfill, with a steep slope $(d\eta/dx)_s$ of 0.015 to the west, and a rapid average linear velocity (v_s) of 7.03×10^{-6} m/s. The permeability value follows from Darcy's law (equations 3 and 30), and is also consistent with Table 1. The fluid kinematic viscosity (ν) is taken as 1.0×10^{-6} m^2/s [Lederer (1983)].

Motts and O'Brien (1981) present a geologic cross-section through the site area, aligned in a westerly, downgradient direction from which the bedrock elevation and recharge area can be estimated. These authors suggest that the bedrock surface is about 53 m above mean sea level under the landfill, so that the unconfined aquifer thickness at the source plane (h_s) is taken to be 13 m in the present study, as indicated by Figure 8. In view of equation 3, the source discharge per unit width (q_s) is thus equal to 2.74×10^{-5} m^2/s at Amherst. The bedrock surface slope ($\tan\beta$) is about 0.0262 in magnitude, while the recharge area (A) is roughly equal to 7×10^5 m^2. Equation 6b specifies the velocity modification factor (γ) value of 0.0112, while the recharge rate, in view of equation 31 and the known landfill geometry, is approximately equal to 1.88×10^{-8} m/s. The latter parameter corresponds to 52% of the precipitation falling on the sand and gravel recharge area, in good agreement with the 50% recharge rate suggested by Motts and O'Brien (1981), and reasonably close to the 1.4×10^{-8} m/s difference between annual precipitation and evaporation for the Commonwealth [Linsley et al. (1982)].

The geometry of the landfill characterizes the near field reservoir. Lederer's (1983) sketch of the facility is reproduced in Figure 8: a length (ζ) of 350 m and a width (b) of 480 m are adopted in the present investigation. Thus, with the spatial origin in the center of the landfill, as indicated by Figure 1, the source plane coordinate (x_s) is equal to 175 m, while the landfill response time (t_c) is 4.98×10^7 s. The latter value follows from equation 16a, with the retardation factor (R) set equal to unity for the conservative contaminants under study at the site. The near field response time is over a year and a half in magnitude, and represents the time required for the reservoir under the landfill to respond to a change in input conditions, such as capping or the cessation of dumping operations. A longer time will be required for these near field changes to be noticed in the slower responding far field.

The observed spread of contamination constitutes the final set of data for the Amherst site. The Commonwealth of Massachusetts [DEQE (1986)] established a set of monitoring wells in the vicinity of the landfill in 1974, and one of the wells lies in the plume, as indicated in Figure 8. The well is screened through the upper 3 m of the unconfined aquifer, and lies 650 m downgradient of the origin. Data are available for 1977, 1979, and 1984 for specific conductance and chloride at the DEQE well, along with a 1976 Metcalf and Eddy (1976) specific conductance observation at the source



Note: Scale 1" = 163 m.
Based on Lederer (1983).

FIGURE 8 - AMHERST LANDFILL AND
MONITORING LOCATIONS

plane, as reported by Lederer (1983). Ambient concentrations (c_a) of chloride and specific conductance are estimated at 15 mg/l and 100 μmhos/cm, respectively [Lederer (1983)].

The measured and predicted concentrations for the two constituents are summarized in Table 4 and Figure 9. The predicted values follow from equations 9, 11, 18, and 21, using procedures described in detail in the subsequent chapter of this report. Data and theory are compared using statistics of the error δ defined by

$$\delta = \frac{c(\text{measured}) - c(\text{predicted})}{c(\text{predicted})} \tag{34}$$

with mean error $\bar{\delta}$ and standard deviation σ computed in accordance with [Benjamin and Cornell (1970)]

$$\bar{\delta} = \frac{1}{J} \Sigma \delta \tag{35a}$$

$$\sigma = (\frac{1}{J} \Sigma \delta^2 - \bar{\delta}^2)^{1/2} \tag{35b}$$

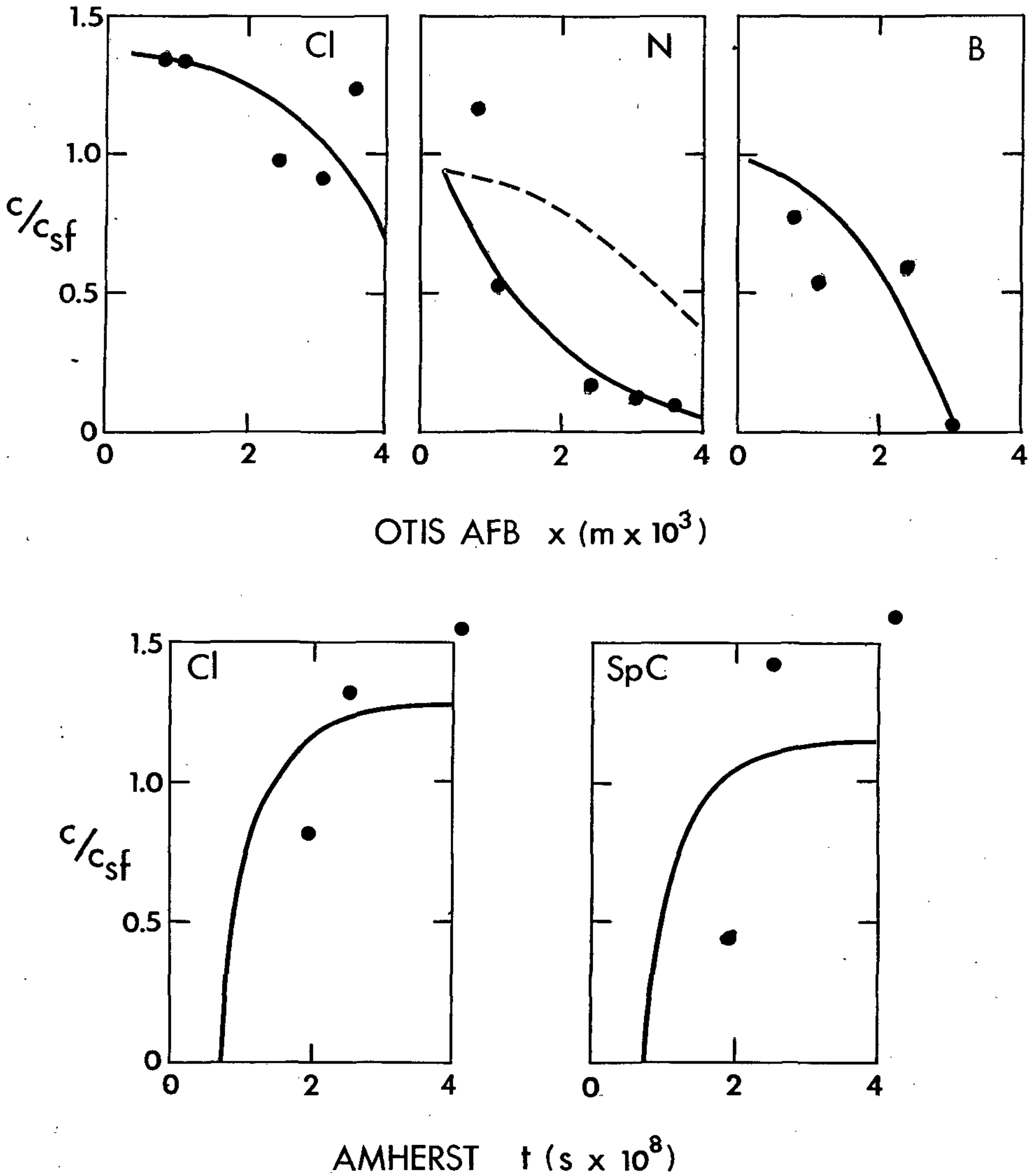
The sign of δ indicates model over or underprediction and is accordingly useful in identifying systematic model errors in calibration and testing. The mean value is reserved for calibration of model parameters, through a search technique which minimizes $\bar{\delta}$. The error standard deviation is based on the absolute value of the individual errors and consequently measures the magnitude of the error, so that an accurate calibration would have a zero mean error and a low standard deviation. In this regard, about 2/3 of the predictions lie within σ of their measured values for a zero mean error.

The data are used to calibrate loading factors for chloride and specific conductance at the Amherst landfill. S values yield predicted con-

TABLE 4 - SUMMARY OF CASE STUDY RESULTS

Constituent	Site	Ambient Concentration c_a	Effluent Concentration c_e	Mean Error $\bar{\delta}$	Standard Deviation σ
Chloride	Amherst Landfill	15 mg/l	---	---[a]	0.23
Specific Conductance	Amherst Landfill	100 μmho/cm	---	---[a]	0.38
Chloride	Otis AFB Bed	8.1 mg/l	31.3 mg/l	0.02	0.21
Total Nitrogen	Otis AFB Bed	0.4 mg/l	21.1 mg/l	---[a]	0.34
Boron	Otis AFB Bed	7.0 μg/l	500.0 μg/l	---[a]	0.21

Notes: [a]Calibration by minimizing mean error.



Note: N dashed line is conservative prediction.
Metcalf and Eddy (1976) data not shown.

FIGURE 9 - CASE STUDY RESULTS

centrations by virtue of equation 16b, and values of 1.62×10^{-5} mg-m^3/l-cap-sec and 2.09×10^{-4} μmho-m^3/cm-cap-sec zero the mean errors for chloride and specific conductivity. The respective standard deviations of 23 and 38% represent reasonable calibration accuracy, particularly in view of the sparse nature of the data set, and the partial penetration of the monitoring wells. The calibrated chloride loading factor compares surprisingly well with the comparable value of 1.40×10^{-5} mg-m^3/l-cap-sec obtained by Ostendorf et al. (1984) at the Babylon, Long Island site. Figure 9 does indicate a systematic underprediction of the far field concentrations with increasing time. This may perhaps be attributed to an overestimation of the average linear velocity, so that the observed far field concentrations are in fact carried by older observers who left the source plane while concentrations were still rising appreciably. The behavior could also be explained by a delay in leachate chemistry, which would impose an additional delay in the response of the near field reservoir. Clearly, additional data are needed to support any further analysis of the Amherst plume. Nonetheless, the relatively low standard deviations and close correspondence of the chloride loading factors for Amherst and Babylon endorse the present approach, particularly in view of the model simplicity.

4.2 Otis AFB Infiltration Beds

Ostendorf (1986) successfully calibrates and tests the infiltration bed modification against chloride (Cl), total nitrogen (N), boron (B), and methyl active blue substances (MBAS) concentrations downgradient of infiltration beds at Otis Air Force Base in Barnstable County, Massachusetts, as reported by LeBlanc (1984). The tests of the first three species are restated here: chloride is treated as conservative and offers a true test of the model, while total nitrogen and boron exhibit first order reactive and linearly adsorptive behavior, respectively. The data are used to calibrate the reactive constants, so that the standard deviations reflect model accuracy. The MBAS data are accurately matched by the model, but the predictions are necessarily based on a nonzero shutdown concentration at the source plane. Since this condition is excluded from the present solution in the interests of model simplicity, the MBAS run is deleted from this study; interested readers may refer to the Ostendorf (1986) paper for details of this test.

The Otis Air Force Base wastewater treatment plant began operating in 1941, and has discharged an average flow (Q_e) of 0.0231 m^3/s through a bed of approximate radius (r_e) equal to 250 m into a sand and gravel aquifer of porosity 0.30 [LeBlanc (1984)], as sketched in Figure 10. LeBlanc (1984) reports an aquifer thickness of about 47 m under the infiltration beds, along with a rising bottom slope of -0.00348. The observed water table slope of 0.0015 under the infiltration beds is then substituted into equation 6b to yield a value of -0.00498 for the velocity modification factor.

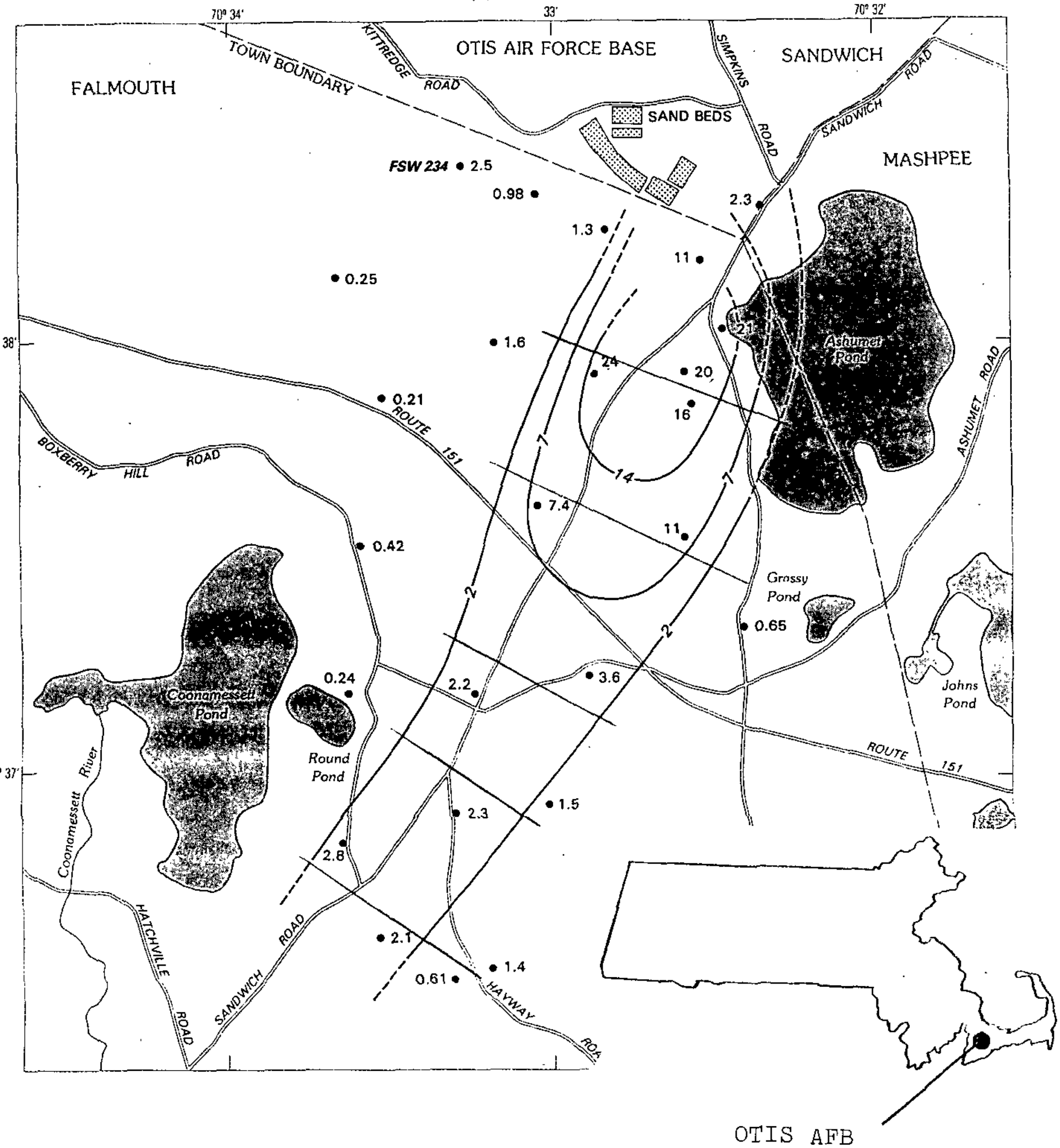
In the absence of locally definitive values for permeability and local recharge, Ostendorf (1986) uses the observed 3700 m extent of the MBAS plume in 1978 to calibrate the far field hydraulics. The position and time of the front position are substituted into equation 12, which is then solved implicitly for recharge, making use of Darcy's law (equation 30) and the recharge equation (31). In the latter regard, Ostendorf (1986) reports a recharge area of 3.91×10^6 m^2 and the resulting recharge estimate adopted in the present study is 7.14×10^{-9} m/s. This figure is about 20% of the annual average precipitation, perhaps indicative of upgradient withdrawals for the base water supply, and is slightly lower than the value appearing in the Ostendorf (1986) analysis due to the exclusion of (small) head loss effects in that study.

The discharge per unit width follows from the recharge equation, and is set equal to 2.89×10^{-5} m^2/s, so that the average linear velocity is equal to 2.05×10^{-6} m/s, much slower than its Amherst counterpart. The infiltration bed effluent from the Base results in warmer water temperatures and a higher kinematic viscosity of 1.3×10^{-6} m^2/s; thus the permeability is estimated to be 5.44×10^{-11} m^2.

The natural and artificial flows are substituted into equation 23 to derive a value of 0.51 for the effluent ratio (α) characterizing the equivalent landfill of the near field. Figure 6 then yields a plume width, landfill length, and source plane location of 966, 628, and 314 m respectively. The estimated plume width compares favorably with LeBlanc's (1984) reported range of 760 to 1060 m, in support of the simple modeling approach of the present investigation. Equation 29b suggests that the dilution factor at Otis AFB is 1.21, so that only a modest reduction of effluent concentration is to be expected in the subsurface flow field, unless reactions are occurring. In view of equation 16a, the near field response time for a conservative contaminant is roughly equal to 3.06×10^8 s, or nearly ten years. The large near field reservoir thus exhibits a sluggish response to changes in effluent conditions due to massive contaminant storage upgradient of the source plane. As with the Amherst landfill, a considerable period of time must pass before the groundwater environment benefits from remedial measures instituted at the ground surface. By the same token, many years elapsed before the far field waters of the aquifer were degraded by the infiltration bed or landfill operation.

With the hydraulics established, contaminant transport modeling can proceed, first with conservative chloride. LeBlanc (1984) cites recent effluent (c_e) and ambient concentration data which yield values of 31.3 and 8.1 mg/l, respectively. Chloride concentrations are reported for 1978 at transects located 760, 1020, 2360, 2990, and 3590 m downgradient of the infiltration beds as suggested by Figure 10. The values represent depth and lateral averages across the plume, which may be regarded as well-defined compared to the Amherst contamination. The salient difference between landfills and infiltration beds lies in the relatively accurate specification of the source pollution for the latter facility. Thus the reported effluent data replace the artifice of a contaminant loading factor, and the chloride data pose a true, uncalibrated test of the transport model. The results, as summarized by Table 4 and Figure 9, are encouraging indeed: the (uncalibrated) mean error of 2% and standard deviation of 21% represent excellent model accuracy.

LeBlanc (1984) measured total nitrogen (N) across the chloride transects as well, and Figure 9 displays observed and conservatively computed values for total nitrogen in the Otis AFB plume. The predictions are based on LeBlanc's (1984) reported values of 21.1 and 0.4 mg/l for effluent and ambient concentrations, respectively. The conservative concentrations exceed the data with a systematic increase in error with downgradient distance, and the contaminant is found over the entire length of the conservative plume, as delineated by chloride. Recalling Figure 7, such behavior may be explained by the postulation of a first order reaction, with the decay rate treated as a calibration factor. The far field concentration



Note: After LeBlanc (1984).
Scale 1" = 683 m.
Wells are averaged at 5 transects shown.
Shown are total nitrogen concentrations.

FIGURE 10 - OTIS AFB INFILTRATION BED AND WELL LOCATIONS

experienced by a moving observer will decrease with time in the moving frame, in accordance with equation 11, so that the far field c and source plane c_s concentrations differ, as suggested by Figure 4. A decay constant of λ equals 1.71×10^{-9} s^{-1} zeros the mean error for total nitrogen, as suggested by Figure 10. The corresponding standard deviation of 34% indicates reasonable calibration accuracy.

Ambient and effluent boron concentrations are equal to 7 and 500 µg/l, respectively [LeBlanc (1984)]. The boron plume extends to 3000 m at full strength and, in view of Figure 7, exhibits linearly adsorptive behavior in the far field. The decay constant is accordingly set equal to 0, and the retardation factor becomes a calibration factor instead. The four transects nearest the source plane yield a modest retardation factor of 1.33 and generate a low standard deviation of 31%, as sketched in Figure 9. The adsorptive behavior is consistent with limited independent evidence cited by LeBlanc (1984). The near field response time (equation 16a) and far field trajectory (9) equations must be modified to allow for R greater than unity in the course of calculations. Sample calculations for both reactive contaminants will be included in the following applications chapter.

CHAPTER 5

APPLICATIONS

5.1 Source History of Existing Reactive Plume

The observed far field distribution of total nitrogen downgradient of the Otis AFB infiltration bed will be used to illustrate the inference of source history from an existing plume of first order reactive contamination. Given data for this application is assumed to consist of the following parameters, with values appropriate for the site:

<u>Geology</u>

Porosity $n = 0.30$

Permeability $k = 5.44 \times 10^{-11}\ m^2$

Bottom Slope $\tan\beta = -0.00348$

<u>Hydraulics</u>

Aquifer Thickness at Source Plane $h_s = 47m$

Water Table Slope at Source Plane $(\frac{dh}{dx})_s = 0.0015$

Recharge Area $A = 3.91 \times 10^6\ m^2$

<u>Existing Plume Data</u>

Infiltration Bed Radius $r_e = 250$ m

Plume Width $b = 966$ m

First Order Decay Constant for Total Nitrogen $\lambda = 1.71 \times 10^{-9}\ s^{-1}$

Retardation Factor for Total Nitrogen $R = 1$

Far Field Concentrations c in Table 5

TABLE 5 - SOURCE CONDITIONS FOR REACTIVE PLUME APPLICATION

Downgradient Distance x (m)	Travel Time[a] $t-t_s$ (s x 10^8)	Far Field Concentration[b] c (mg/l)	Source Concentration c_s (mg/l)
760	2.02	20.0	28.3
1020	3.06	9.2	15.5
2360	7.26	2.9	10.0
2990	8.72	1.9	8.4
3590	9.88	1.7	9.2

Notes: [a]Trip time for observer to travel from source plane to x.
[b]Total nitrogen data at Otis AFB [LeBlanc (1984)].

Ambient Concentration $c_a = 0.4$ mg/l

Year of Measurement $T = 1978$

Physical Constants

Gravitational Acceleration $g = 9.81\ m/s^2$

Kinematic Viscosity $\nu = 1.3 \times 10^{-6}\ m^2/s$

For the purposes of this example, the source conditions are assumed to be unknown, reflecting either a lack of records or a lack of cooperation by the waste generator. The required output parameters for this problem characterize the flow through the infiltration bed responsible for the observed contamination:

Unknown Source Conditions

Year of Startup T_o

Effluent Discharge Q_e

Effluent Concentration c_e

The solution consists of sequential sets of hydraulic and contaminant calculations. The effluent discharge is explicitly estimated, while the concentration data yield a set of values for the concentration and year of startup, from which averages are taken.

Explicit equations developed in prior chapters are invoked to generate additional parameters characterizing the hydraulics of the unconfined aquifer. The Otis values, along with the appropriate equation numbers are as follows:

Hydraulic Calculations

Discharge per Unit Width through the Source Plane (30) $q_s = 2.89 \times 10^{-5} m^2/s$

Average Linear Velocity through the Source Plane (3) $v_s = 2.05 \times 10^{-6}$ m/s

Velocity Modification Factor (6b) $\gamma = -.00498$

Recharge Rate (31) $\varepsilon = 7.14 \times 10^{-9}$ m/s

The observed plume width and infiltration bed radius yield a ratio b/r_e equal to 3.86 in magnitude, and Figure 6 may consequently be consulted to deduce data that will characterize the source plane configuration:

Source Plane Configuration

Effluent Ratio (Figure 6) $\alpha = 0.51$

Source Plane Location (Figure 6) $x_s = 314$ m

Equivalent Landfill Length (27) $\zeta = 628$ m

Effluent Discharge (23) $Q_e = \underline{0.0231 \text{ m/s}}$

The hydraulic parameters yield the far field observer travel times that in turn specify contamination at the source plane through the method of characteristics. The travel time $t-t_s$ represents the duration of the trip from the source plane to the present position x associated with a given far field concentration data point, as specified by equation 9. The first order decay constant is inserted into equation 11 to trace the observer's concentration back to its source value c_s, which is in force at the time of departure t_s. The appropriate calculations are explicit, as summarized below:

Source Plane Concentrations

Observer Travel Time (9) $t-t_s$ in Table 5

Source Plane Concentration at Time t_s (11) c_s in Table 5

Near Field Reponse Time (16a) $t_c = 3.06 \times 10^8$s

The computed $c_s(t-t_s)$ values appearing in Table 5 may be manipulated to yield estimates of the steady state output concentration c_{sf} and effluent concentration. *Equation 18 may be rearranged in the following fashion*

$$1 - \frac{c_s - c_a}{c_{sf}} = \exp(- \frac{t_s}{t_c}) \qquad (36)$$

Equation 36 is valid at two data points, denoted by 1 and 2 subscripts and representing two independent equations. The two equations may be divided and solved simultaneously for the unknown output concentration with the result

$$c_{sf} = \frac{c_{s1} - c_a - C_1(c_{s2} - c_a)}{1 - C_1} \qquad (37a)$$

$$C_1 = \exp(\frac{t_{s2} - t_{s1}}{t_c}) \qquad (37b)$$

Each pair of data points yields a different estimate of c_{sf}, so that the five entries in Table 5 provide 10 values of the output concentration; the average of these estimates is adopted, and completes the inference of source history:

<u>Effluent Concentration</u>

Steady State Output Concentration above Ambient c_{sf} = 22.9 mg/l

<u>Effluent Concentration</u> (29a) c_e = <u>28.1 mg/l</u>

<u>Year of Startup</u> T_o = <u>1945</u>

The pair of equations leading to equation 37 also specifies the year of startup, which is related to the present (T=1978) far field time when the observations were made by

$$t = C_2(T - T_o) \qquad (38a)$$

$$C_2 = 3.15 \times 10^7 \text{ s/yr} \tag{38b}$$

Equation 38a leads to an expression for the source time

$$t_s = C_2(T-T_o)-(t-t_s) \tag{39}$$

which may be compared with another equation (36) based on the source concentration

$$t_s = -t_c \ln[1-(\frac{c_s-c_a}{c_{sf}})] \tag{40}$$

An estimate of the startup year follows from equations 39 and 40

$$T_o = T - \frac{(t-t_s) - t_c \ln[1-(\frac{c_s-c_a}{c_{sf}})]}{C_2} \tag{41}$$

The pairs of data points, along with the individual c_{sf} estimates, yield estimates of T_o, so long as the later c_s value in the pair is lower in magnitude. This is the case for 9 of the 10 pairs, and the average value is cited as the year of startup. The reasonable correspondence of the computed and actual values appearing in the previous chapter is not surprising, since a common decay constant is used.

5.2 Post Shutdown Recovery of Groundwater Quality

The calibrated loading factor for specific conductance at the decommissioned landfill in Amherst is used to predict future contamination from an existing source. The projected recovery of groundwater quality after closure of the facility will be computed as well. Future distributions at a fixed time and at a fixed location illustrate the predicted behavior using contaminant "photograph" and "hydrograph" concepts, respectively. Given data for this application consists of the following parameters:

<u>Geology</u>

Porosity $n = 0.30$

Permeability $k = 1.43 \times 10^{-11}\ m^2$

Bottom Slope $\tan\beta = 0.0262$

Hydraulics

Aquifer Thickness at Source Plane $h_s = 13$ m

Water Table Slope at Source Plane $(\frac{d\eta}{dx})_s = 0.015$

Recharge Area $A = 7 \times 10^5\ m^2$

Existing Source Data

Landfill Length $\zeta = 350$ m

Landfill Width $b = 480$ m

User Population $P = 47000$

Year of Startup $T_o = 1971$

Time of Shutdown $t_{sd} = 3.78 \times 10^8 s$

Specific Conductance Loading Factor $S = 2.09 \times 10^{-4}$ μmho-m^3/cm-cap-s

Retardation Factor for Specific Conductance $R = 1$

Decay Constant for Specific Conductance $\lambda = 0\ s^{-1}$

Ambient Concentration $c_a = 100$ μmho/cm

Physical Constants

Gravitational Acceleration $g = 9.81\ m/s^2$

Kinematic Viscosity $\nu = 1.0 \times 10^{-6}\ m^2/s$

Two types of predictions of contaminant concentration are required: a fixed time photograph of the spatial variation of pollution, and a fixed position contaminant hydrograph describing temporal variation. Both predictions use a series of moving observers and rest on common hydraulic and source plane calculations. Proceeding with the hydraulics first:

Hydraulic Calculations

Discharge per Unit Width through the Source Plane (30) $q_s = 2.74 \times 10^{-5} m^2/s$

Average Linear Velocity through the Source Plane (3) $v_s = 7.03 \times 10^{-6}$ m/s

Velocity Modification Factor (6b) $\gamma = 0.0112$

Recharge Rate (31) $\varepsilon = 1.88 \times 10^{-8}$ m/s

The source plane concentrations follow from the known hydraulics, and are sketched in Figure 3 as specific conductance levels at the downgradient boundary of the landfill:

Source Plane Calculations

Source Plane Location (27) $x_s = 175$ m

Near Field Response Time (16a) $t_c = 4.98 \times 10^{7}$ s

Steady State Output Concentration above Ambient (16b) $c_{sf} = 747$ μmho/cm

Shutdown Source Concentration above Ambient (20b) $c_{sd} = 747$ μmho/cm

Source Plane Concentrations (18,21) c_s in Figure 3

The spatial distribution of pollution in 1990 is established by setting t equal to 5.99×10^{8} s and computing the departure times t_s and source concentrations c_s of observers that will occupy various far field positions x at that future time. Since specific conductance is conservative, the observers will experience no change in concentration in their moving reference frames, and the source and far field concentrations will be equal. The results are summarized in Table 6 and Figure 11:

Contaminant Photograph

TABLE 6 - CALCULATIONS FOR CONTAMINANT PHOTOGRAPH

Far Field Position x (m)	Observer Time of Departure t_s (s x 10^8)	Observer Source Concentration c_s (μmho/cm)	Far Field Concentration c(μmho/cm)
175[a]	5.99	109[b]	109[c]
275	5.85	112	112
375	5.70	116	116
475	5.55	121	121
575	5.40	129	129
675	5.25	139	139
775	5.10	153	153
875	4.95	172	172
975	4.79	198	198
1075	4.64	234	234

Notes: [a]Source plane location.

[b]Source and far field concentrations equal for conservative plume.

[c]Predicted specific conductance concentrations at Amherst, in 1990.

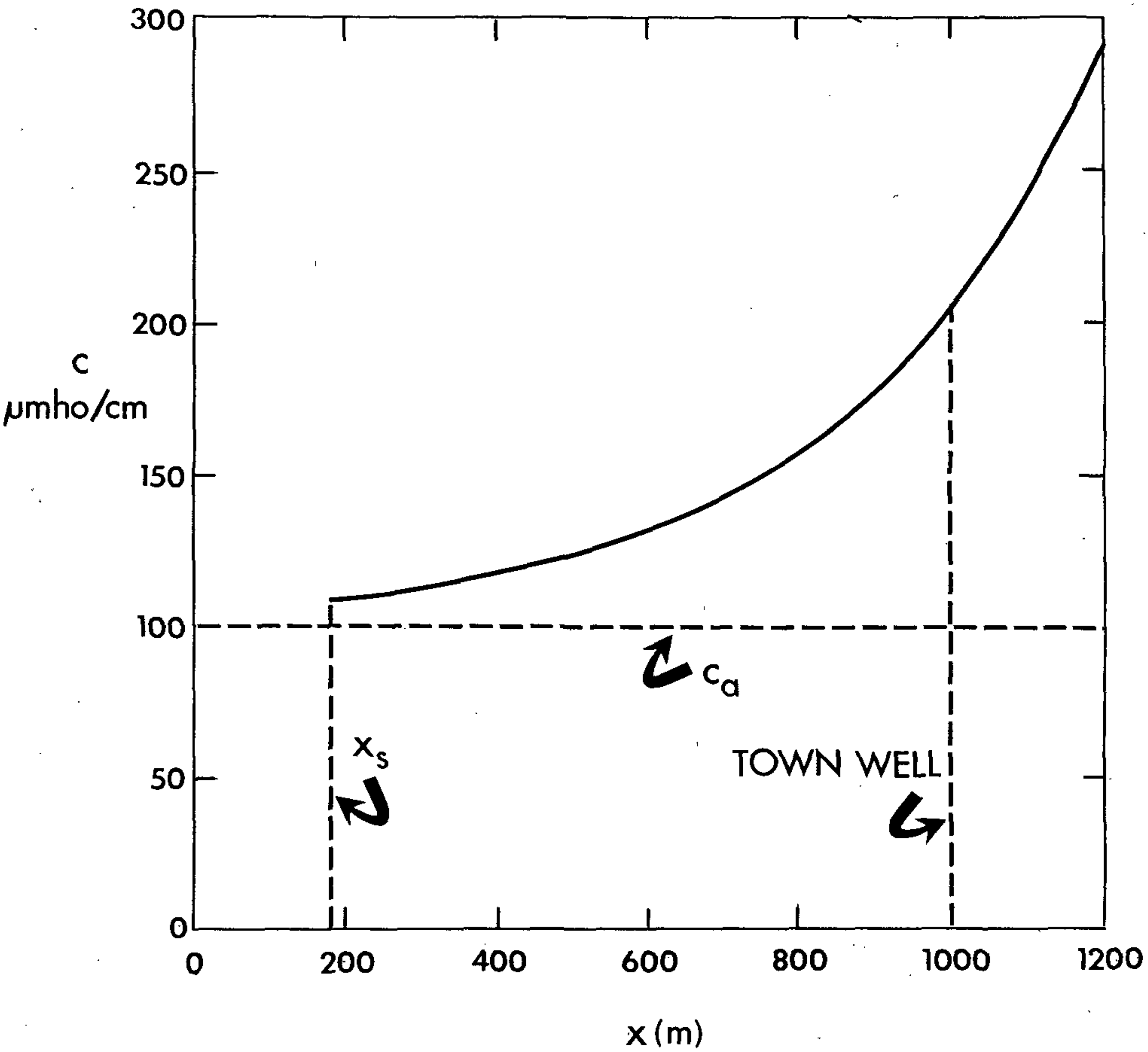


FIGURE 11 - SPECIFIC CONDUCTANCE AT AMHERST IN 1990

Year $T = 1990$, $t = 5.99 \times 10^8$ s

Far Field Position x in Table 6

Observer Departure Time (9) t_s in Table 6

Observer Source Concentration (Figure 3) c_s in Table 6

Far Field Concentration at Arbitrary Position x (11) c in Table 6

Figure 6 suggests that the pollution will return to essentially ambient levels at the landfill by 1990, while contamination will still be appreciable at the Town wells 1000 m downgradient of the origin.

The contaminant hydrograph describes the temporal variation of pollution at the Town well. In this case, arbitrary values of time t are selected instead of x, but the procedure is otherwise identical to the photograph calculation:

Contaminant Hydrograph

Far Field Position $x = 1000$ m

Far Field Time t in Table 7

Observer Departure Time (9) t_s in Table 7

Observer Source Concentration (Figure 3) c_s in Table 7

Far Field Concentration at Arbitrary Time t (11) c in Table 7

The results are summarized in Table 7 and sketched in Figure 12: concentrations of specific conductance should return to background levels by the year 1995, reflecting delays induced by the near field response time and travel time through the far field.

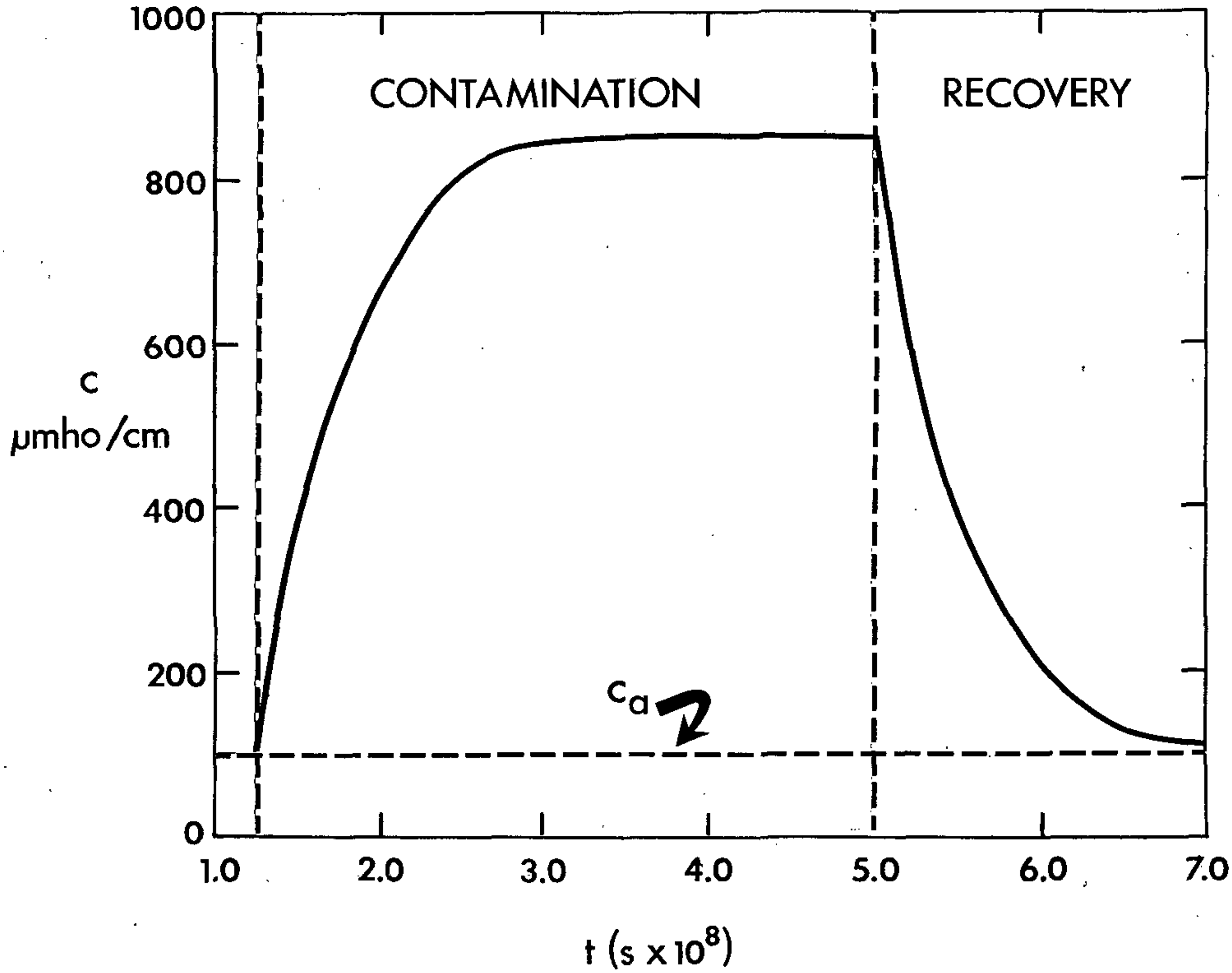
5.3 Infiltration Bed Dilution Constraint

The final application illustrates the use of the dilution factor in the design of an infiltration bed, and underscores the poor mixing characteristics of the groundwater flow field. A hypothetical unconfined aquifer is

TABLE 7 - CALCULATION FOR CONTAMINANT HYDROGRAPH

Far Field Time t (s x 10^8)	Observer Time of Departure[a] t_s (s x 10^8)	Observer Source Concentration c_s (μmho/cm)	Far Field Concentration c (μmho/cm)
1.24[b]	0.00	0	0
1.51	0.27	417[c]	417[d]
2.27	1.03	753	753
3.02	1.79	826	826
3.78	2.54	842	842
4.54	3.30	846	846
5.02[e]	3.78	847	847
5.29	4.06	530	530
6.05	4.81	194	194
6.80	5.57	121	121
7.56	6.32	105	105

Notes: [a]From source plane.
[b]Plume arrival time at the town well.
[c]Source and far field concentrations equal for conservative plume.
[d]Specific conductance concentrations at Amherst town well, x=1000 m.
[e]Shutdown observer arrives at well, onset of falling concentrations.



Note: x = 1000 m.

FIGURE 12 - SPECIFIC CONDUCTANCE AT AMHERST TOWN WELL

assumed with known geologic and hydraulic properties, and the infiltration bed receives effluent from a municipal wastewater treatment plant serving 8000 people with a desired dilution factor of 2:

Known Parameters

Permeability $k = 5 \times 10^{-11} m^2$

Aquifer Thickness at Source Plane $h_s = 30$ m

Water Table Slope at Source Plane $(\frac{d\eta}{dx})_s = 0.005$

Gravitational Acceleration $g = 9.81$ m^2/s

Kinematic Viscosity $\nu = 1.2 \times 10^{-6}$ m^2/s

User Population $P = 8000$

Dilution Factor $D = 2$

Sought are the effluent discharge, infiltration bed radius and associated plume width required to accomodate the effluent:

Unknown Source Conditions

Effluent Discharge Q_e

Infiltration Bed Radius r_e

Plume Width b

This problem is an exercise in near field hydraulics involving explicit and implicit (trial and error) equations. The effluent discharge estimate is straightforward, following the estimate of Tchobanoglous and Schroeder (1985) cited in Chapter 3, so that the dilution factor specifies the plume width:

Source Conditions

Effluent Discharge [Tchobanoglous and Schroeder (1985)] $Q_e = \underline{0.044 \ m^3/s}$

Discharge per Unit Width through the Source Plane (30) $q_s = 6.13 \times 10^{-5} m^2/s$

Plume Width (29b) $b = 1435$ m

Effluent Ratio $\alpha = 0.21$

Infiltration Bed Radius $r_e = 552$ m

The infiltration bed radius is the result of an iterative solution involving equation 23 and Figure 6:

Infiltration Bed Iteration

1 Assume $r_e = 100$ m.

2 Compute α from equation 23.

3 Read b/r_e from Figure 6 as a function of α.

4 Compute a new r_e value from the b/r_e ratio, with known b.

5 Go to step 2 and iterate.

The numerical values suggest that even a small user population can generate a wide contaminant plume, particularly if dilution is required in the groundwater environment and the ambient flows are modest. This is a consequence of the poor mixing characteristics of the subsurface flow field.

CHAPTER 6

CONCLUSIONS

This report models the transport of a simply reactive contaminant through a landfill and an underlying shallow, one dimensional, unconfined aquifer with a plane, sloping bottom in the assumed absence of dispersion and downgradient dilution. The constant user population and a presumedly constant contaminant loading factor determine the pollutant input to the near field groundwater region under the landfill. This near field is modeled as a linear reservoir whose output at the downgradient edge of the facility comprises the source plane for far field transport through the unconfined aquifer. The far field analysis describes temporal concentration variation experienced by observers moving away from the source plane at speeds modified by recharge, head loss, bottom slope, and linear adsorption. The temporal concentration variation witnessed by the observers reflects linear adsorption and first order reaction kinetics, and yields a prediction of contamination at any far field place and time. A simple extension of the model permits the analysis of plumes from infiltration beds, using the concept of an equivalent landfill with dimensions dependent in part on the strength of the bed effluent relative to the ambient discharge in the aquifer.

The landfill model, which has been previously tested with good accuracy against an extensively measured plume in Long Island, describes the contamination downgradient of the decommissioned facility in Amherst, Massachusetts, to demonstrate Commonwealth applicability of the analysis.

The sparse Amherst data base is used to calibrate loading factors for specific conductance and chloride, an exercise yielding respective S values of 2.09×10^{-4} mho-m^3/cm-cap-s and 1.62×10^{-5} mg-m^3/l-cap-s. The error standard deviations of 23 and 37% constitute reasonable calibration accuracy, in view of the sparse nature of the measurements, and the partial penetration of the monitoring wells. The calibrated chloride loading factor compares favorably with the 1.40×10^{-5} mg-m^3/l-cap-s value obtained in Long Island, perhaps indicative of a common value for this parameter in similar waste and geologic settings.

The salient difference between an infiltration bed and a landfill lies in the relatively accurate specification of the input pollution for the former facility. Thus the reported effluent discharge and concentration data replace the artifice of a contaminant loading factor, removing a calibration parameter from comparisons of data and theory. The extensive set of conservative chloride observations at Otis AFB in Barnstable County, Massachusetts accordingly offers a true uncalibrated test of model accuracy and illustrates a second application of the analysis to a Commonwealth site. The results are a strong endorsement of the model approach, with mean error and standard deviation values of 2 and 21%, respectively. Total nitrogen and boron data are also available for the Otis plume. These constituents exhibit first order reactive and linearly adsorptive behavior in the far field, and the corresponding reactive constants are used as calibration coefficients. The total nitrogen observations support a first order decay constant of 1.71×10^{-9} s^{-1}, and the boron measurements indicate a modest retardation factor of 1.33. The slow nature of the decay kinetics suggests that reactions are rate limited by mixing processes within the plume. The respective standard deviations of 34 and 31% suggest reasonable calibration accuracy, particularly in view of the simple nature of the model.

The three model applications demonstrate the inference of source history from an existing plume, recovery of groundwater quality after shutdown of the landfill, and dilution constraints in the design of infiltration beds. The source inference application may be used to assign responsibility for present pollution to past dischargers of effluent into the subsurface environment, while the recovery example might be useful in the assessment of potential effectiveness of remedial measures in mitigating future impacts of groundwater pollution. The last application demonstrates the relatively poor mixing characteristics of the unconfined aquifer flow field, compared to its surface water counterpart. Dilution factors less than 5 in magnitude are feasible in the subsurface, while factors from 10 to 100 are common in rivers and lakes. Thus effluent concentrations will persist at essentially full strength in the groundwater environment in the absence of reactions.

Future research may proceed on several fronts. Differential plume density may affect far field hydraulics and more realistic chemistry should be studied in attempt to put the calibrated constants of the near and far field analyses on a better physical basis. In the latter regard, the loading factor model adopted in the near field should be modified in future studies to accomodate slow reactions between precipitation and solid waste, using available lysimeter data. Such a study would bear on the need for impervious capping of these facilities, which presupposes continued generation of leachate after the time of shutdown. The Amherst data cited in this report does not yield information on this matter, because the source times associated with available information all predate the closure of the facility. A more complete investigation, including fully penetrating monitoring wells, would provide valuable insight into capping effectiveness. Far field reactions require more sophisticated study as well. Total nitrogen is the expression of a coupled transport system involving ammonia, nitrate, and dissolved oxygen, which may yield analytical solutions in the absence of dispersion. Such a modeling effort must remain as simple as allowed by the available data however, and a continuing need is evident for historically documented pollutant sources and spatially resolved contaminant plumes in this regard.

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