GROUND-WATER FLOW AND SOLUTE TRANSPORT AT A MUNICIPAL LANDFILL SITE ON LONG ISLAND, NEW YORK

PART 3: SIMULATION OF SOLUTE TRANSPORT

by Eliezer J. Wexler

U.S. GEOLOGICAL SURVEY

Water-Resources Investigations Report 86-4207

Prepared in cooperation with the

TOWN OF BROOKHAVEN

Syosset, New York

1988

DEPARTMENT OF THE INTERIOR

DONALD PAUL HODEL, Secretary

U.S. GEOLOGICAL SURVEY

Dallas L. Peck, Director

For additional information
write to:

U.S. Geological Survey
5 Aerial Way
Syosset, New York 11791

Copies of this report may be
purchased from:

U.S. Geological Survey
Books and Open-File Reports
Federal Center, Bldg. 810
Box 25425,
Denver, Colo. 80225

CONTENTS

ILLUSTRATIONS

ILLUSTRATIONS (continued)

TABLES

CONVERSION FACTORS

For the convenience of readers who prefer metric (International System) units rather than the inch-pound units in this report, the following conversion factors may be used:

Multiply inch-pound units	by	To obtain SI (metric units)
	Length	
inch (in)	2.54	centimeter (cm)
inch (in)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
acre	0.4047	hectare
square foot (ft^2)	0.09294	square meter (m^2)
square mile (mi^2)	2.59	square kilometer (km^2)
	Volume	
gallon (gal)	3.785	liter (L)
	Flow	
foot per second (ft/s)	0.3048	meter per second (m/s)
foot per day (ft/d)	0.3048	meter per day (m/d)
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)
gallon per minute (gal/min)	0.06308	liter per second (L/s)
gallon per day (gal/d)	0.003785	cubic meter per day (m^3/d)
million gallons per day (Mgal/d)	0.04381	cubic meter per second (m^3/s)

Equivalent concentration terms

milligrams per liter (mg/L) equals parts per million (ppm)

micrograms per liter ($\mu g/L$) equals parts per billion (ppb)

Sea level: In this report "sea level" refers to the National Geodetic Vertical Datum of 1929 (NGVD of 1929)--a geodetic datum derived from a general adjustment of the first-order level nets of both the United States and Canada, formerly called "Mean Sea Level of 1929."

GROUND-WATER FLOW AND SOLUTE TRANSPORT AT A MUNICIPAL LANDFILL SITE ON LONG ISLAND, NEW YORK --
Part 3. Simulation of Solute Transport

by Eliezer J. Wexler

ABSTRACT

A solute-transport model representing a 2.3-square-mile (mi^2) area surrounding and downgradient from a municipal landfill site in the Town of Brookhaven was developed to simulate advective-dispersive migration of a conservative solute (chloride) through the upper glacial aquifer. Aquifer-property values used in the model were: hydraulic conductivity, 200 feet per day; effective porosity, 0.30; longitudinal dispersivity, 100 feet; and transverse dispersivity, 20 feet. Concentration of chloride ion in leachate entering the upper glacial aquifer was set at 875.0 milligrams per liter (mg/L). The rate of leachate entry to the aquifer was set equal to the average annual rate of recharge, 24.6 inches per year. Entry of leachate was assumed to have started in 1977. Chloride concentrations in recharge and ambient ground water were set at 10 mg/L.

The model simulated 12 years of plume travel, beginning in 1977, when leachate first entered the aquifer. Simulated chloride concentrations after 6 years of travel matched chloride data collected in October-December 1982 reasonably well. After 12 years of travel, the simulated plume extended 6,200 feet and was 2,600 feet wide. Maximum predicted chloride concentrations at the boundary of the landfill site were 160 mg/L at the end of year 12. The time necessary to reach dynamic equilibrium--a steady-state condition whereby solute concentrations stabilize at some maximum distance from the point of solute introduction--was predicted to be approximately 30 years. The maximum distance of simulated solute movement before reaching dynamic equilibrium was 8,800 feet.

Sensitivity analyses showed that the simulated plume configuration was highly sensitive to changes in effective porosity but less sensitive to changes in longitudinal and transverse dispersivity. Concentrations within the plume were most sensitive to changes in influent concentrations.

Additional simulations were made to test the model's ability to predict the effect of three remedial actions on the movement of solutes; these include capping the sanitary landfill, pumping a line of recovery wells, and a combination of capping and pumping.

INTRODUCTION

Sanitary landfills in humid environments generate leachate, a fluid produced when precipitation infiltrates the refuse and dissolves organic and inorganic material in the waste. If leachate percolates into an underlying aquifer, it can flow through the aquifer, mix with native ground water, and migrate to areas downgradient of the landfill.

Sanitary landfills are currently the primary method of solid-waste disposal on Long Island, N.Y. Studies by Kimmel and Braids (1980) and Wexler (1987) have shown that landfill leachate has degraded ground-water quality downgradient from three sanitary landfills on Long Island.

The ability to accurately predict the movement of contaminants through the ground-water flow system is an important element in the management and protection of ground-water resources. Numerical models can be developed to simulate the flow of ground water and the transport of ground-water solutes. Such models are useful for (1) estimating the spatial and temporal variation in concentration of chemical constituents; (2) estimating the traveltime of a contaminant from its source to a point of discharge; (3) aiding in the design of effective water-quality-monitoring systems; and (4) evaluating the feasibility and effectiveness of remedial procedures to remove contaminants from the aquifer or to prevent their spread (Konikow, 1977).

The Town of Brookhaven, in central Suffolk County, N.Y. (fig. 1), operates a lined sanitary landfill for the disposal of municipal solid waste. The landfill was excavated in glacial outwash deposits that form the upper glacial (water-table) aquifer and is lined with an 0.02-inch-thick polyvinyl chloride (PVC) membrane. Landfilling began in 1974, and, by 1983, the landfill covered 60 of the site's 180 acres (fig. 2).

In 1981, the U. S. Geological Survey, in cooperation with the Town of Brookhaven, began a hydrologic investigation at the landfill site vicinity to develop a digital solute-transport model capable of simulating advective-dispersive transport of conservative contaminants associated with landfill leachate through the underlying upper glacial aquifer.

The investigation was divided into three phases. The first entailed collection of hydrogeologic and water-quality data. A report by Wexler (1988) describes that phase and concludes that leachate has entered the underlying aquifer despite the PVC liner. A report by Pearsall and Wexler (1986) describes the distribution of organic contaminants in the vicinity of the landfill site.

The second phase of the investigation entailed the development and calibration of a steady-state ground-water-flow model to represent a 26-mi^2 area centered about the Brookhaven landfill site (fig. 2). The boundaries of the area modeled coincide with the natural hydrogeologic boundaries closest to the site.

The purpose of the ground-water flow model was twofold. First, development and calibration of the model yielded refined estimates of aquifer and confining-unit properties and flows across the model boundaries. Second, the calibrated model was used to calculate rates and direction of ground-water flow downgradient of the landfill under long-term average conditions. The flow model and simulations are described by Wexler and Maus (1988).

The third phase of the investigation entailed the development of a solute-transport model representing a rectangular 2.3-mi^2 area extending southeastward from the landfill site (fig. 2). The model was used to simulate advective-dispersive migration of conservative ground-water solutes through the upper glacial aquifer.

Purpose and Scope

This report is the last in a three-part series that describes the 4-year study to observe and simulate the transport of chemical solutes from the solid-waste-disposal facility through the underlying glacial outwash deposits. The three reports summarize the extent of solute movement at the site, define the geometry, hydraulic properties, and boundary conditions of the affected area, describe the predicted future movement of solute from the site, and present results of various simulated remedial actions designed to protect the ground water downgradient of the site.

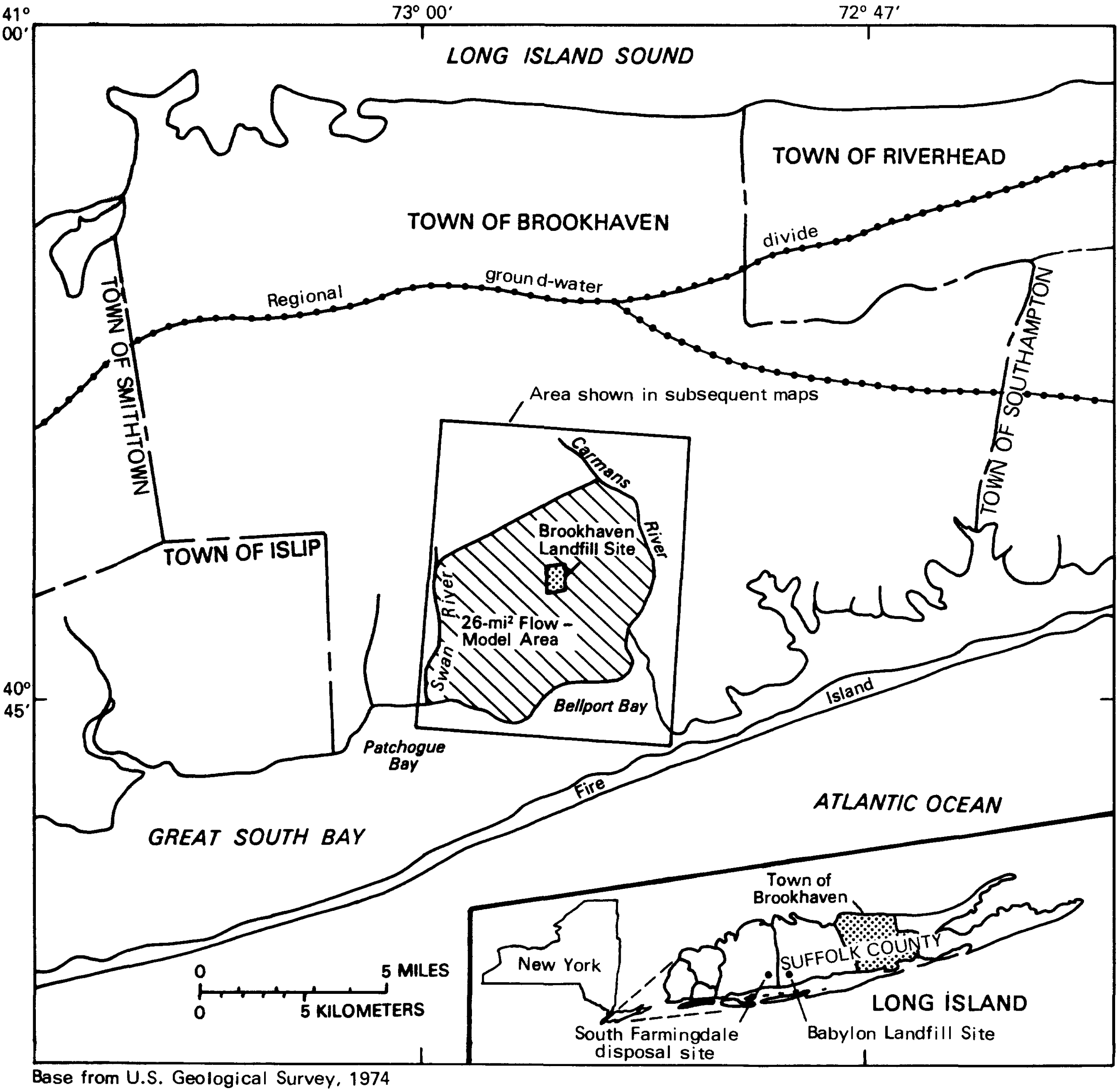
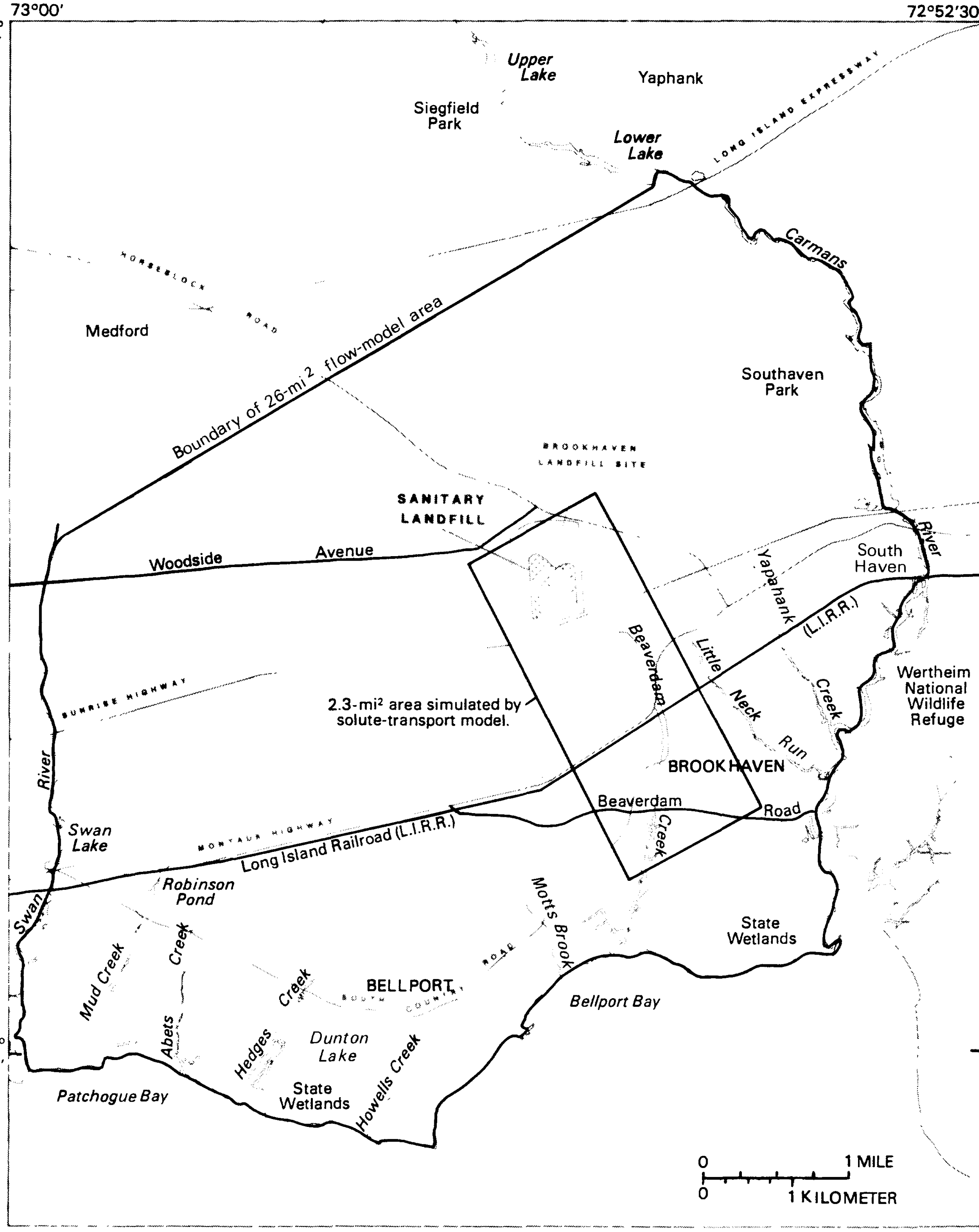


Figure 1.--Location of municipal landfill site in the Town of Brookhaven, N.Y. (Modified from Wexler and Maus, 1988.)



Base from NYSDOT, Bellport, and Howells Point, 1981, NY, 1:24,000

Figure 2.--Location of the 2.3-mi² area simulated by the solute-transport model. (Modified from Wexler and Maus, 1988.)

This report describes the third phase of the investigation and presents results of model simulations, sensitivity analyses, and simulations of three proposed remedial actions to prevent or retard leachate migration.

Previous Investigations

Solute-transport models have been applied to two waste-disposal sites on Long Island. Pinder (1973) used a Galerkin finite-element ground-water flow and solute-transport model to simulate the movement of hexavalent chromium from waste-disposal pits in South Farmingdale (fig. 1), and Gureghian and others (1981) used a Galerkin finite-element, solute-transport model to simulate the movement of a leachate plume at an unlined sanitary landfill in the Town of Babylon (fig. 1). Other recent studies that used numerical models to simulate contaminant transport in unconsolidated aquifers include Konikow (1977), Pickens and Lennox (1976), Gray and Hoffman (1983), and Sykes and others (1982).

Acknowledgments

The computer code for the solute-transport model used in this study, titled SUTRA, was developed by Clifford Voss of the U.S. Geological Survey, who also advised in its application to the Brookhaven landfill site. The author also thanks Leonard Konikow, David Pollock, Thomas Reilly, and Herbert Buxton, all of the U.S. Geological Survey, for their help and advice during this phase of the investigation.

DESCRIPTION OF STUDY AREA

The Brookhaven landfill site is in the northwestern part of the 2.3-mi^2 study area (fig. 2). The land surrounding the site is mostly vacant and is covered with pitch pine and scrub oak forest. Two major roads, Sunrise Highway and Montauk Highway, pass through the study area. South of Montauk Highway is the residential area of the hamlet of Brookhaven (fig. 2).

Hydrogeologic Setting

Long Island is composed of a series of unconsolidated sedimentary deposits of Late Cretaceous and Pleistocene age resting on crystalline bedrock (fig. 3). A description of the principal aquifers and confining units underlying the Town of Brookhaven is given in table 1.

Upper Glacial Aquifer

The uppermost unit, the upper glacial aquifer, consists primarily of sand and gravel associated with outwash from the Wisconsin glaciation. Aquifer thickness ranges between 100 to 180 ft in the study area. The lower 10 to 15 ft of the aquifer consists of silt and silty sand. The approximate configuration of the lower surface of the aquifer is depicted in Wexler and Maus (1988, fig. 5).

Previously published estimates of aquifer hydraulic conductivity in the region range between 187 and 268 ft/d (Warren and others, 1968; Franke and McClymonds, 1972). Calibration of the ground-water flow model developed in the second phase of the study yielded a value of 200 ft/d (Wexler and Maus, 1987), which was adopted in all simulations discussed herein. Published estimates of effective porosity for the sand and gravel in the south-shore area of Long Island range between 0.25 and 0.30 (Kimmel and Braids, 1980, p. 32; Gureghian and others, 1981). A value of 0.30 was selected for use in the solute-transport model.

Table 1.--Principal aquifers and confining units underlying the Town of Brookhaven.

[Modified from Jensen and Soren, 1971. Relative positions are shown in fig. 3]

Hydrogeologic unit	Geologic name	Approximate thickness (ft)	Description and water-bearing character
Upper glacial aquifer	Upper Pleistocene deposits	0 - 750	Mainly brown and gray sand and gravel of medium to high hydraulic conductivity; also includes deposits of clayey till and lacustrine clay of low hydraulic conductivity. A major aquifer.
Gardiners Clay	Gardiners Clay	0 - 75	Green and gray clay, silt, clayey and silty sand, and some interbedded clayey and silty gravel. Unit has low hydraulic conductivity and confines water in underlying aquifer.
Monmouth greensand	Monmouth Group, undifferentiated	0 - 200	Interbedded marine deposits of dark gray, olive-green, dark greenish-gray, and greenish-black glauconitic and lignitic clay, silt, and clayey and silty sand. Unit has low hydraulic conductivity and tends to confine water in underlying aquifer.
Magothy aquifer	Matawan Group and Magothy Formation, undifferentiated	0 - 1,100	Gray and white fine-to-coarse sand of medium hydraulic conductivity. Generally contains sand and gravel beds of low to high hydraulic conductivity in basal 100 to 200 ft. Contains much interstitial clay and silt, and beds and lenses of clay with low hydraulic conductivity. A major aquifer though not developed in study area.
Raritan clay	Unnamed clay member of the Raritan Formation	0 - 200	Gray, black, and multicolored clay and some silt and fine sand. Unit has low hydraulic conductivity and tends to confine water in underlying aquifer.
Lloyd aquifer	Lloyd Sand Member of the Raritan Formation	0 - 500	White and gray fine- to-coarse sand and gravel of medium hydraulic conductivity and some clayey beds of low hydraulic conductivity. Not developed as a source of water in study area.
Bedrock	Undifferentiated crystalline rocks	Unknown	Mainly metamorphic rocks of low hydraulic conductivity; surface generally weathered; considered to be the bottom of the ground-water reservoir.

Gardiners Clay

Along the south shore of Long Island, the upper glacial aquifer is underlain by the Pleistocene Gardiners Clay (fig. 3), which lies approximately 100 ft below sea level and thins gradually northward. Its thickness in the study area is 10 to 15 ft. The unit confines water in lower formations, although vertical leakage does occur. Estimates of the vertical hydraulic conductivity of the Gardiners Clay range between 0.01 ft/d (Franke and Getzen, 1976) and 0.04 ft/d (Warren and others, 1968). Simulations made with the ground-water flow model (phase 2) indicated a value of 0.007 ft/d, which was used in the solute-transport simulations described herein.

Monmouth Greensand

In the southernmost part of the study area, the Gardiners Clay is underlain by the Monmouth greensand unit, a marine deposit of Cretaceous age that consists of interbedded clay, silt, and silty sand layers. The hydraulic conductivity of the unit is assumed to be similar to that of the Gardiners Clay. Its thickness within the study area ranges between 0 and 10 ft. Maps showing the combined thickness of the two confining units--the Gardiners Clay and Monmouth greensand--are given in Wexler and Maus (1988, fig. 4).

Magothy Aquifer

The Gardiners Clay and Monmouth greensand, where present, rest upon the the Magothy aquifer (fig. 3), which is approximately 800 ft thick in the study area and of moderate hydraulic conductivity. The Magothy was treated as a source of steady leakage to the upper glacial aquifer in both the ground-water-flow and solute-transport models because it is overlain by the two

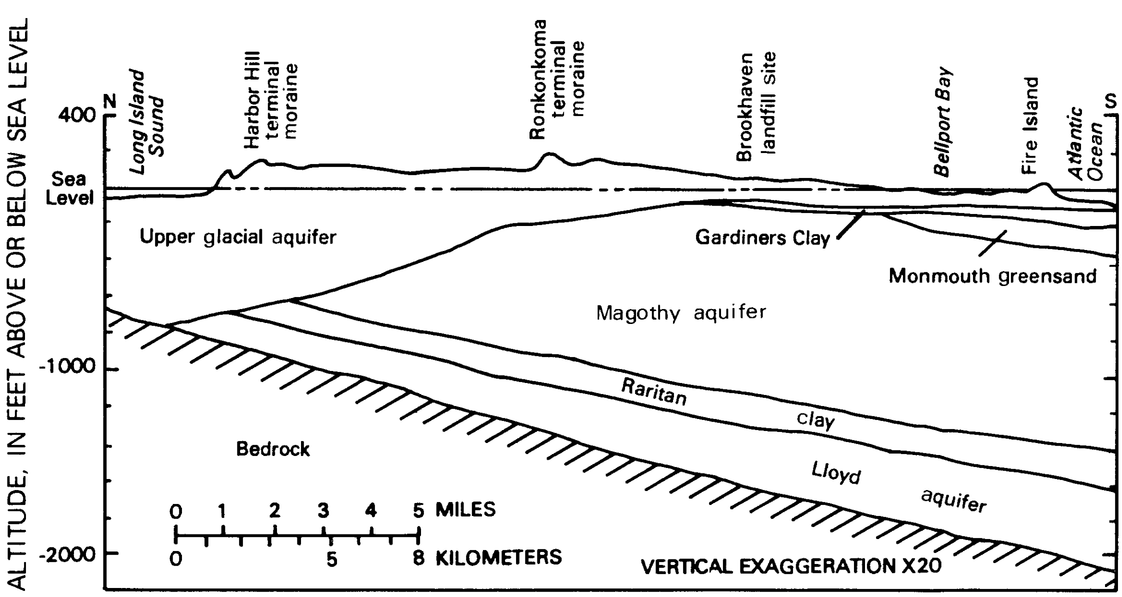


Figure 3.--Generalized north-south hydrogeologic section through Town of Brookhaven showing relative position of the principal aquifers and confining units. (Modified from Jensen and Soren, 1974.)

confining units. Head values in the Magothy aquifer are greater than the corresponding heads in the upper glacial aquifer throughout most of the study area; thus, most leakage is vertically upward. Some downward leakage occurs in the northwest corner of the 2.3-mi^2 study area, however, where gradients are reversed.

Ground-Water Flow in the Upper Glacial Aquifer

Ground water in the upper glacial aquifer is under water-table (unconfined) conditions. Saturated thickness of the aquifer ranges between 100 and 130 ft in the 2.3-mi^2 study area. Water levels were measured at 164 observation wells and 23 stream-stage-measurement sites in the landfill vicinity. A map showing the water table in September 1982, a period of average water levels, is presented in figure 4. Ground-water flow in the upper glacial aquifer is predominantly horizontal and perpendicular to lines of equal water-table altitude.

The aquifer is recharged by precipitation and, to a lesser degree, by upward leakage from the Magothy aquifer. About 24.6 inches, which represents 52 percent of the annual average precipitation of 47.4 inches, is available for ground-water recharge (Wexler and Maus, 1988). Water also enters the study area as underflow across the northwestern and southwestern boundaries.

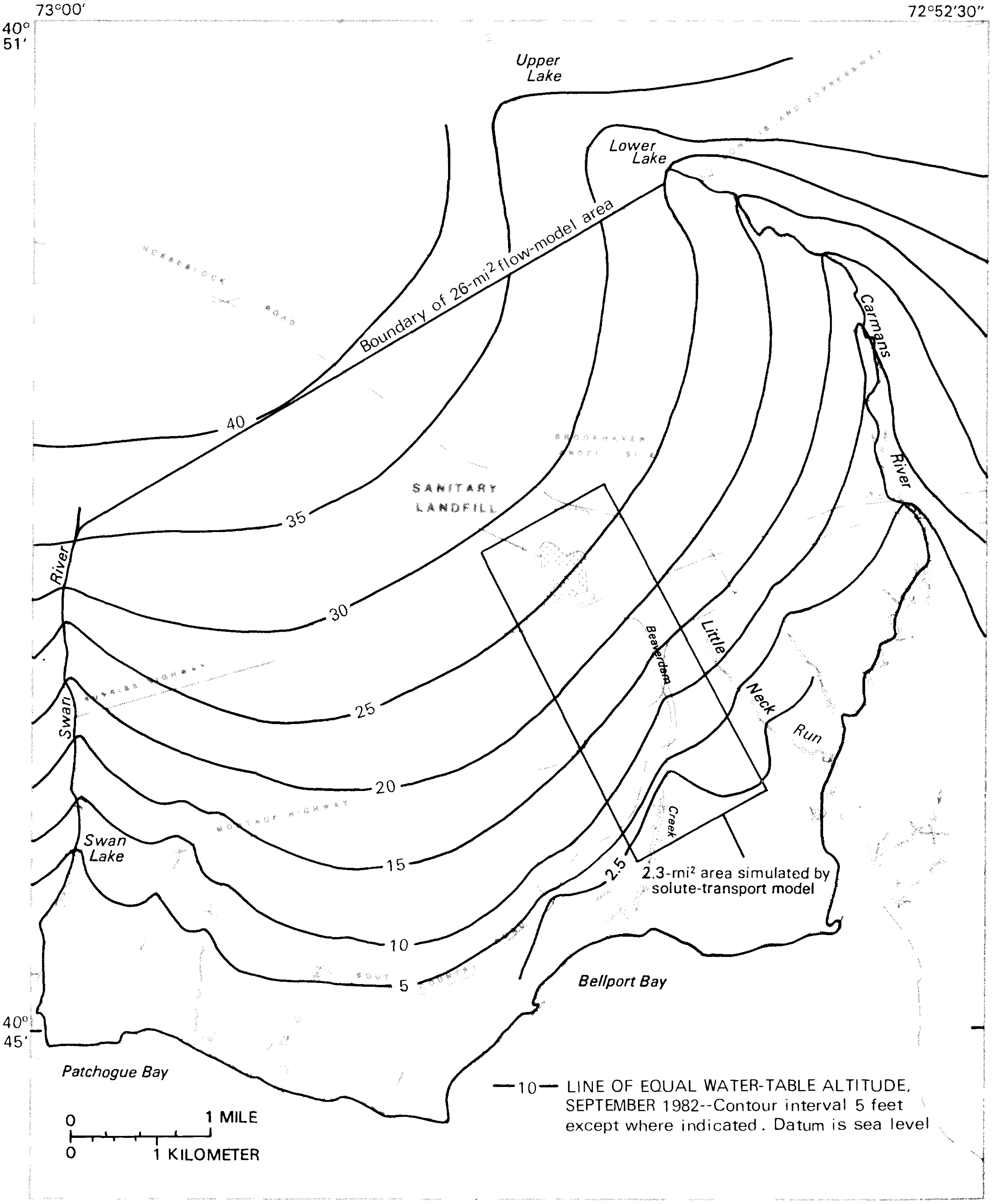
Ground-water discharge within the study area occurs principally as discharge to Beaverdam Creek, which flows southward toward Bellport Bay (fig. 4). Pumpage within the area is for domestic use and irrigation. Consumptive losses (water not returned through cesspools or as irrigation return flow) are assumed to be minor and were not simulated in this study. Ground-water evapotranspiration in areas where the water table is close to land surface was also assumed to be minor. Ground water not discharged within the study area moves southward beyond the area's boundaries, where it eventually discharges to tidal reaches of streams or directly to Bellport Bay.

Surface Water

Beaverdam Creek is the only surface-water body of significance within the study area (fig. 2). The creek is fed primarily by ground water, although stormwater runoff may contribute to flow within the developed areas of Brookhaven hamlet. Average annual base flow at a partial-record discharge-measurement site at South Country Road (fig. 2) is estimated to be 1.35 ft^3/s (D. S. Peterson, U.S. Geological Survey, written commun., 1983).

The exact point at which flow begins in Beaverdam Creek is determined by the point at which the water table first intersects the stream-channel bottom. Flow usually starts in the area between Sunrise and Montauk Highways (fig. 2). The stream becomes tidal approximately 900 ft south of South Country Road.

Little Neck Run, just east of the study area, is a point of discharge for ground water that flows eastward out of the study area. The upper reach of the stream, north of the Long Island Railroad right-of-way (fig. 2), is ponded behind a clogged culvert. Flow gradually resumes south of the railroad for about 1,000 ft, where the stream becomes tidal.



Base from NYSDOT, Bellport, and Howells Point, 1981, NY, 1:24,000

Figure 4.--Water-table altitude in September 1982. (Location is shown in fig. 1.)

Water Quality

Ground water flowing across the northwestern and southwestern boundaries of the study area is generally of excellent quality and is similar in composition to precipitation. It is generally soft (low calcium and magnesium concentrations), slightly acidic, and has a low dissolved-solids concentration. Chloride-ion concentrations range between 8 and 10 mg/L, although use of road salt has increased chloride concentrations downgradient from stormwater basins and along Horseblock Road (fig. 2) and Sunrise Highway (Wexler, 1988). Use of lawn and agricultural fertilizers and discharge from cesspools may have degraded water quality in parts of Brookhaven hamlet.

Leachate Migration

Despite the PVC liner beneath the sanitary landfill, leachate has entered the upper glacial aquifer. Samples of leachate and leachate-contaminated ground water collected in October-December 1982 showed elevated values of specific conductance, pH, and temperature; elevated concentrations of sodium, potassium, calcium, magnesium, ammonium, bicarbonate, chloride, iron, and manganese; and lowered concentrations of sulfate and nitrate.

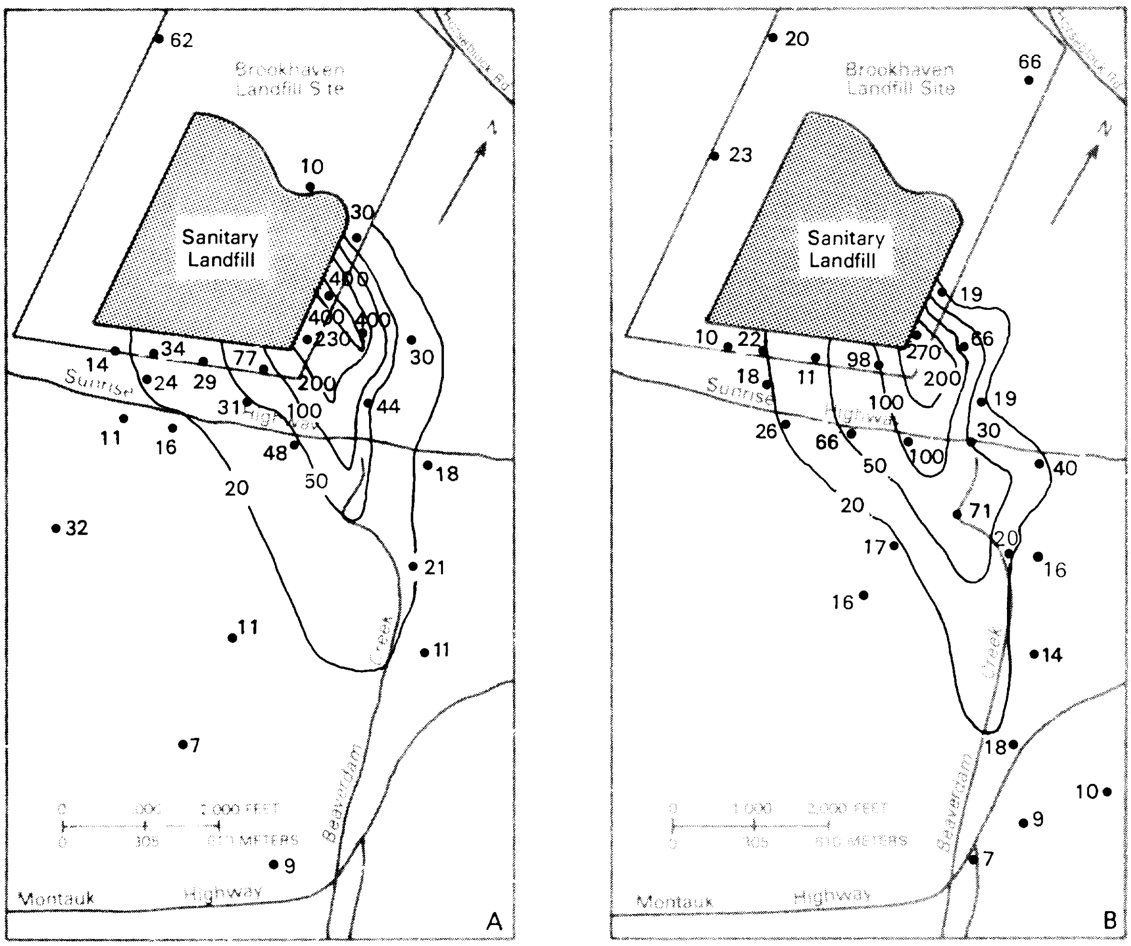
A plume of leachate-contaminated ground water extends southeastward from the landfill site (Wexler, 1988). At a depth between 20 and 40 ft below the water table, the plume extends 3,700 ft downgradient from the southeast corner of the landfill site and is 2,400 ft wide. Along the southern boundary of the site, the plume extends to at least 90 ft below the water table. It appears to be confined to the upper glacial aquifer because a well screened just below the Gardiners Clay showed no evidence of leachate contamination. Additional data on the chemical composition of leachate-contaminated ground water and the distribution of inorganic and organic constituents downgradient of the site are given in Wexler (1988) and in Pearsall and Wexler (1986).

The means of leachate entry into ground water beneath the liner was not determined, although data from monitoring wells along the southeastern boundary of the site indicated that a combination of two mechanisms may have been responsible. The first is leakage through tears in the liner or along separated seams; the second is overflow of leachate that has ponded above the liner. Overflow is likely only along the east side of the sanitary landfill (Wexler, 1988).

Distribution of Chloride Downgradient from the Landfill

Data from monitoring wells in the landfill-site vicinity were used to delineate the distribution of chloride in leachate-contaminated ground water. Monitoring wells were grouped according to depth of screen below the average water level. Most wells were screened between 8 and 20 ft and between 20 and 40 ft below the water table. The distribution of chloride in these two depth intervals in October-December 1982 is shown in figures 5A and 5B, respectively. A slight widening of the plume at Sunrise Highway is due to contamination by road salt. Chloride concentrations exceed New York State and Federal recommended limits of 250 mg/L only along the eastern boundary of the landfill site.

Fewer data are available on the chloride distribution at other depths within the upper glacial aquifer. Data from monitoring wells along the southern boundary of the landfill site indicated elevated chloride concentrations at all depths and slight increases with depth. Lenses of clay or silt seem to have prevented the downward movement of chloride and other leachate contaminants at the eastern boundary of the site. Elevated chloride concentrations were detected at depths of at least 60 ft below the water table in wells 400 ft downgradient of the eastern boundary of the site.



EXPLANATION

•16 OBSERVATION WELL--Number is chloride concentration, in milligrams per liter, October -December, 1982

— 50— LINE OF EQUAL CHLORIDE CONCENTRATION, OCTOBER-DECEMBER, 1982--Contour interval as indicated

Figure 5.--Chloride concentrations, October-December 1982:
A. In wells screened 8 to 20 ft below the water table.
B. In wells screened 20 to 40 ft below the water table.
(Location is shown in fig. 4.)

SIMULATION OF SOLUTE TRANSPORT

Data on hydrogeologic conditions and water quality in the study area (Wexler, 1988, and Wexler and Maus, 1988) were used to develop a digital model to simulate transport of conservative contaminants under average ground-water flow conditions in the upper glacial aquifer. Simulation of solute transport in ground water required solution of two governing partial differential equations subject to appropriate boundary and initial conditions. The first equation describes the flow of ground water under steady-state conditions; the second describes the movement and spread of solutes within the flowing ground water. A brief discussion of the theoretical background relating to the governing equations is provided below.

Theoretical Background

Ground-Water-Flow Equation

The partial differential equation governing steady-state two-dimensional (areal) ground-water flow in a heterogeneous, isotropic water-table aquifer is given by Bear (1979) as:

$$\frac{\partial}{\partial x}\left[K(h-b)\frac{\partial h}{\partial x}\right] + \frac{\partial}{\partial y}\left[K(h-b)\frac{\partial h}{\partial y}\right] + Q = 0 \qquad (1)$$

where:

K = hydraulic conductivity at a point in the aquifer $[LT^{-1}]$,
h = hydraulic head (or water-table altitude) above a datum $[L]$,
b = altitude of the aquifer bottom above the same datum $[L]$, and
Q = rate of recharge per unit area of aquifer $[LT^{-1}]$.

In deriving equation 1, it is assumed that flow in the water-table aquifer is horizontal (the Dupuit assumption) and that fluid density is uniform. The term Q in equation 1 is used to represent both distributed and point sources of water.

If changes in saturated thickness of the water-table aquifer are small compared to the overall saturated thickness, equation 1 can be linearized by replacing the term $K(h-b)$ with $\bar{T}$ such that:

$$\frac{\partial}{\partial x}\left[\bar{T}\frac{\partial h}{\partial x}\right] + \frac{\partial}{\partial y}\left[\bar{T}\frac{\partial h}{\partial y}\right] + Q = 0 \qquad (2)$$

where $\bar{T}$ is the effective transmissivity (hydraulic conductivity multiplied by the average saturated thickness of the aquifer). Additional discussions on the ground-water flow equation and the associated boundary conditions can be found in Bear (1979).

Equations 1 or 2 can be solved for a given set of boundary conditions, aquifer properties, and rates of recharge (or withdrawals from the aquifer). The solution will be in terms of the hydraulic head at all points in the aquifer. The average pore velocity, $\vec{v}$, can then be determined from the distribution of hydraulic head by Darcy's Law, such that:

$$\vec{v} = -\frac{K}{n} \cdot \left[\frac{\partial h}{\partial x} i + \frac{\partial h}{\partial y} j\right] \tag{3}$$

where:

n = effective porosity of the aquifer [dimensionless]; and

i, j = unit vectors in the x and y directions, respectively.

Advection and Hydrodynamic Dispersion

Movement of solutes through a porous medium is controlled by two physical processes--advection and hydrodynamic dispersion. Advective transport describes the movement of solute particles along the mean direction of fluid flow at a rate equal to the average pore velocity. Hydrodynamic dispersion describes the spread of solute particles along and transverse to the direction of average fluid flow.

Hydrodynamic dispersion accounts for the spreading of solute particles by the combined effects of two processes--molecular diffusion and mechanical dispersion. Molecular diffusion produces a flux of solute particles from areas of high solute concentrations to areas of low solute concentrations; it is independent of the rate of fluid movement. Mechanical dispersion is the mixing of solutes by variations in fluid velocities at the microscopic scale. These velocity variations result from several factors, including (1) velocity distributions within the pore space; (2) variations in pore size; (3) differences in path lengths for different solute particles; and (4) the effect of converging and diverging flow paths (Bear, 1979.) Mechanical dispersion is velocity dependent, and, except at low pore velocities, its effects are greater than those of molecular diffusion.

Dispersive flux, $\vec{J}$, is described by Fick's first law:

$$\vec{J} = -\bar{\bar{D}}_m \cdot \vec{\nabla} c \tag{4}$$

where:

c = volumetric concentration of solute,

$\vec{\nabla}$ = gradient operator (where $\vec{\nabla} = \frac{\partial}{\partial x}\vec{i} + \frac{\partial}{\partial y}\vec{j} + \frac{\partial}{\partial z}\vec{k}$)

$\vec{\nabla} c$ = concentration gradient, and

$\bar{\bar{D}}_m$ = second-rank tensor containing the coefficients of mechanical dispersion.

Mechanical-dispersion coefficients are related to the average pore velocity by the dispersivity of the medium (Scheidegger, 1961). The coefficients of dispersivity are dependent on properties of the medium, including permeability, length of a characteristic flow path, and tortuosity. In an isotropic medium (with respect to dispersion), the coefficients of mechanical dispersion can be expressed in terms of two components--the coefficient of longitudinal dispersivity, a_l, and the coefficient of transverse dispersivity, a_t. Mechanical dispersion in the transverse direction is much weaker than dispersion in the longitudinal direction; transverse dispersivities are normally lower by a factor of 5 to 20 (Freeze and Cherry, 1979).

The nine components of the symmetric mechanical dispersion tensor can be expressed in terms of v, the magnitude of the average pore velocity and the velocity components v_x, v_y, and v_z (Bear, 1979). In a system where ground-water flow is horizontal ($v_z = 0$), the components of the mechanical dispersion tensor are written as:

$$
\begin{aligned}
D_{xx} &= (a_l \, v_x^2 + a_t \, v_y^2) \, / \, v \\
D_{xy} &= D_{yx} = (a_l - a_t) \, v_x \, v_y \, / \, v \\
D_{yy} &= (a_l \, v_y^2 + a_t \, v_x^2) \, / \, v \\
D_{xz} &= D_{zx} = D_{yz} = D_{zy} = 0 \\
D_{zz} &= a_t \, v
\end{aligned}
\qquad (5)
$$

The hydrodynamic dispersion tensor can be written as:

$$\bar{\bar{D}}_h = \bar{\bar{D}}_m + D_d \, \bar{\bar{I}} \qquad (6)$$

where:

$\bar{\bar{D}}_h$ = second-order hydrodynamic dispersion tensor,
$\bar{\bar{D}}_m$ = mechanical dispersion tensor,
D_d = coefficient of molecular diffusion, and
$\bar{\bar{I}}$ = identity tensor.

Macrodispersion

In laboratory experiments with homogeneous materials, longitudinal dispersivities typically range between .004 and 0.4 in, whereas in field studies of contaminant plumes, longitudinal dispersivities of up to 300 ft have been calculated (Freeze and Cherry, 1979). The larger values in field studies are related to increased mixing (on a macroscopic scale) due to local variations in aquifer hydraulic conductivity.

Most studies of contaminant transport at waste-disposal sites assumed that macrodispersive fluxes can be represented by Fick's law (for example, Pinder, 1973; Gureghian and others, 1981; Konikow, 1977; Pickens and Lennox, 1976; Gray and Hoffman, 1983; and Sykes and others, 1982). However, more recent studies have suggested that the concept may not be valid within short distances of the contaminant source, but only after travel distances of hundreds or thousands of feet (Gelhar and Axness, 1981). Within short distances, dispersivities may be scale-dependent, with values increasing with distance from the contaminant source as larger scale heterogeneities are encountered (Gelhar and others, 1979). Additional discussions of recent developments in macrodispersion theory can be found in Anderson (1984).

In general, values of dispersivity obtained in the laboratory or in small-scale tracer tests are not useful in predicting plume migration over distances of hundreds or thousands of feet. Conversely, values determined through calibration of predictive models against observations of plumes that have traveled these distances cannot be applied to simulation of contaminant behavior within short distances (tens of feet) of the source. It may also be inappropriate to use these values for long-distance travel (tens of thousands of feet) if larger scale variations are to be encountered.

For the purpose of this study, the dispersive transport of solutes is assumed to obey Fick's law. However, limitations of using the model to predict solute travel within short distances from the landfill or for long-distance travel must be recognized. Values of aquifer dispersivity used in the study are described in a later section.

Advective-Dispersive Solute-Transport Equation

The advective-dispersive solute-transport equation has been presented in many texts, including Bear (1972) and Fried (1975). The two-dimensional form of the equation (1984), can be written in vector form as:

$$n\,B\,\frac{\partial c}{\partial t} = -\,n\,B\,\vec{v}\cdot\vec{\nabla}c + \vec{\nabla}\cdot(n\,B\,\bar{\bar{D}}_h\cdot\vec{\nabla}c) + Q\,(c - c') \qquad (7)$$

where:

n = effective porosity [dimensionless],
B = saturated thickness of the aquifer [L],
c = volumetric solute concentration [M/L^3],
$\vec{v}$ = average pore velocity [L/T],
$\bar{\bar{D}}_h$ = hydrodynamic dispersion tensor [L^2/T],
Q = rate of recharge per unit area of aquifer [L/T], and
c' = volumetric solute concentration in recharge [M/L^3].

In equation 7, it is assumed that flow is horizontal and that fluid density is uniform throughout the aquifer. It is further assumed that the solute is uniformly mixed in the vertical direction--that is, the term $\partial c/\partial z$ is equal to zero. Thus, the term c represents the vertically averaged concentration at a point in the aquifer. In studies of contaminant plumes in predominantly horizontal flow systems on Long Island (Pinder, 1973; Gureghian and others, 1981), simulations that used a two-dimensional approach produced reasonable matches with observed contaminant movement.

Equation 7 can be expanded as:

$$nB\,\frac{\partial c}{\partial t} = -\,nBv_x\,\frac{\partial c}{\partial x} - nBv_y\,\frac{\partial c}{\partial y} + \frac{\partial nBD_{xx}}{\partial x}\,\frac{\partial c}{\partial x} + \frac{\partial nBD_{xy}}{\partial y}\,\frac{\partial c}{\partial x} + \frac{\partial nBD_{yx}}{\partial x}\,\frac{\partial c}{\partial y}$$

$$+ \frac{\partial nBD_{yy}}{\partial y}\,\frac{\partial c}{\partial y} + Q\,(c'-c) \qquad (8)$$

where D_{xx}, D_{xy}, D_{yx} and D_{yy} are defined in equation 5. The fluid source term, $Q(c-c')$, represents the sum of all distributed and point sources of fluid. Withdrawals of fluid from the aquifer are not considered in equations 7 and 8 because the concentration of solute in the fluid withdrawn from the aquifer, c', is identical to the solute concentration, c. Solution of equation 8 with a given set of boundary and initial conditions and with values specified for the velocity distribution, aquifer dispersivity, and solute inflow concentrations yields the vertically averaged concentration of solute at all points in the aquifer as a function of time.

Except under certain idealized conditions, the solution of the two governing partial-differential equations must be done by numerical methods. These methods involve numerical approximation techniques that determine the aquifer head and concentration values at a finite number of points in the aquifer and at specified time intervals. Computer codes based on the Galerkin finite-element technique were used in this study. Descriptions of the technique as applied to the solution of the ground-water flow and solute-transport equation can be found in many texts, including Bear (1979), Pinder and Gray (1977), and Wang and Anderson (1982).

Simulation of Ground-Water Flow

The area of the local ground-water flow system, as defined by the location of hydrologic boundaries nearest the landfill site, may differ considerably from the area likely to be affected by landfill leachate. For this reason, ground-water flow in the 2.3-mi^2 study area was analyzed by coupling two steady-state flow models of differing scale. The first model was developed to simulate flow within the upper glacial aquifer in the regional, 26-mi^2 area shown in figure 2; the second simulated the 2.3-mi^2 area shown in figure 4. Hydrogeologic conditions and model development are described in Wexler and Maus (1988); basic features of the model and model results are summarized below.

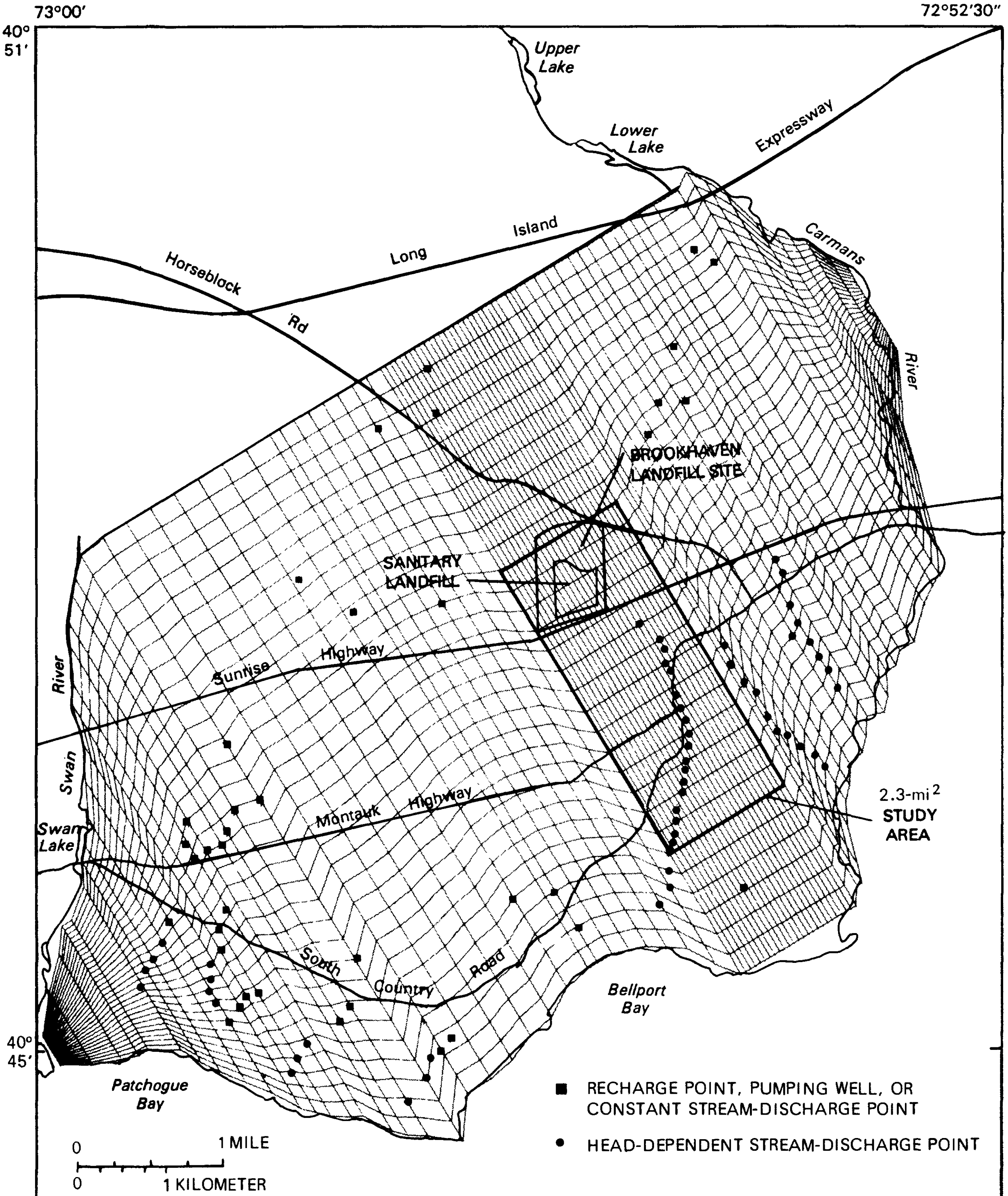
Regional Flow Model

The size and shape of the flow-model area was based on the location of natural hydrologic boundaries closest to the landfill site. The lateral boundaries include the Carmans River, Swan River, and the freshwater/saltwater interface along the shore. The northern boundary is defined by an arbitrary line extending from the headwaters of the Swan River to the downstream end of Lower Lake on the Carmans River (fig. 2). The water table and clay units that underlie the upper glacial aquifer form the upper and lower boundaries, respectively (fig. 3).

The 26-mi^2 area was represented by a finite-element grid composed of 2,490 quadrilateral elements with 2,604 nodes (fig. 6). Application of boundary conditions and the rates of recharge and discharge used in the model are described by Wexler and Maus (1988). The computer code used in the study, referred to herein as the "modified Tracy code," applies the Galerkin finite-element technique to solve equation 1. The code is documented by Dunlap and others (1984).

Accurate simulation of ground-water discharge to streams is important because several streams are close to the landfill site. Discharge to Beaverdam Creek, Little Neck Run, and Yapahank Creek (fig. 2) was simulated as head-dependent discharge through the streambed material (Wexler and Maus, 1988). Hydraulic conductivity of the streambed material was adjusted during model calibration until discharge matched average base-flow values for the streams.

Aquifer-property values were adjusted until simulated water-table altitudes closely matched those observed in September 1982--a period when streamflow and aquifer heads were close to long-term average values. The best match was achieved with an aquifer hydraulic conductivity of 200 ft/d, a confining-unit hydraulic conductivity of 7.0×10^{-3} ft/d, and streambed hydraulic conductivities of 6.5 ft/d in flowing reaches of streams and 3.25 ft/d in the tidal reaches. Average streambed thickness was assumed to be 3.0 ft. The simulated steady-state water-table altitudes are shown in figure 7.



Base from NYSDOT, Bellport, and Howells Point, 1981, NY, 1:24,000

Figure 6.--Finite-element grid used to represent the 26-mi² flow-model area. (Modified from Wexler and Maus, 1988; location is shown in fig. 2.)

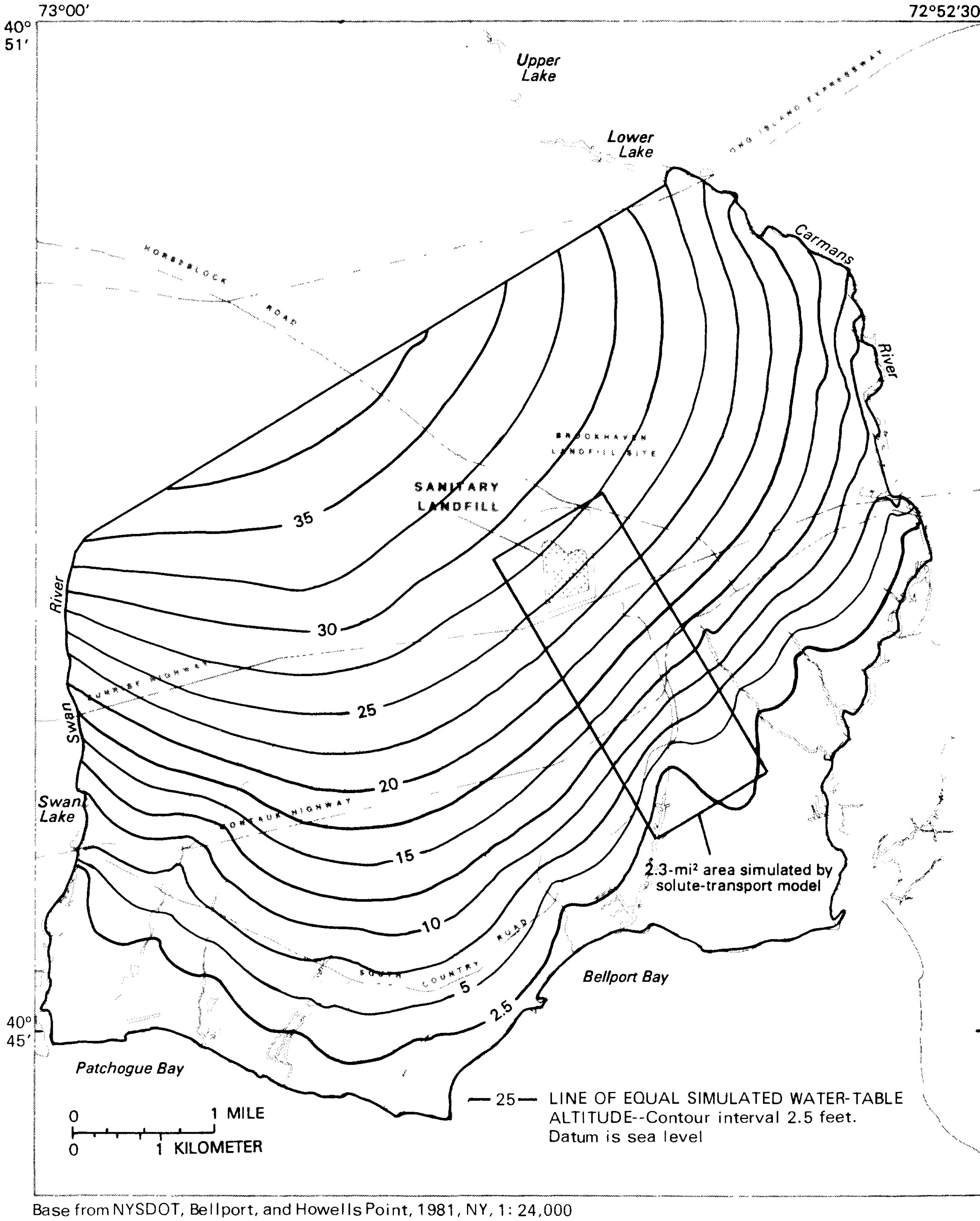


Figure 7.--Simulated water-table altitude in the 26-mi² flow-model area. (Modified from Wexler and Maus, 1988.)

Study-Area Flow Model

The first ground-water flow model was constructed such that the 2.3-mi^2 study area overlapping and downgradient of the landfill site was represented by a block of 384 elements, each representing an 840-ft by 200-ft area (fig. 6). An even finer discretization of this area was needed for the solute-transport simulation to (1) provide a more accurate representation of the ground-water velocity in the landfill vicinity, and (2) to avoid the problem of numerical dispersion associated with large elements. Thus, each of the 384 elements was divided into five smaller elements, each representing a 168-ft by 200-ft rectangle. The fine-scale grid (fig. 8) contains 1,920 elements and 2,025 nodes and represents the area in greater detail.

All data required for simulating ground-water flow in the study area were obtained from the larger scale model, thus effectively coupling the two models. Prescribed-head boundary conditions were applied to all nodes along the boundary of the fine-scale grid. Values at the nodes were interpolated from water-table altitudes calculated for corresponding points in the large-scale model. Values for the basal altitude of the upper glacial aquifer, the confining-unit thickness, and the Magothy aquifer heads were also interpolated from values used in the large-scale flow model. Rates of areal recharge and ground-water pumpage were identical in both models. Hydraulic conductivity of the upper glacial aquifer was set at 200 ft/d; that of the confining units was set equal to 0.007 ft/d.

Locations of the 15 nodes representing the flowing reaches of Beaverdam Creek and the 8 nodes representing the tidal reach are shown in figure 8. Hydraulic conductivity of the streambed material was set at 6.5 ft/d in the flowing reaches and 3.25 ft/d in the tidal reaches; streambed thickness was set at 3 ft.

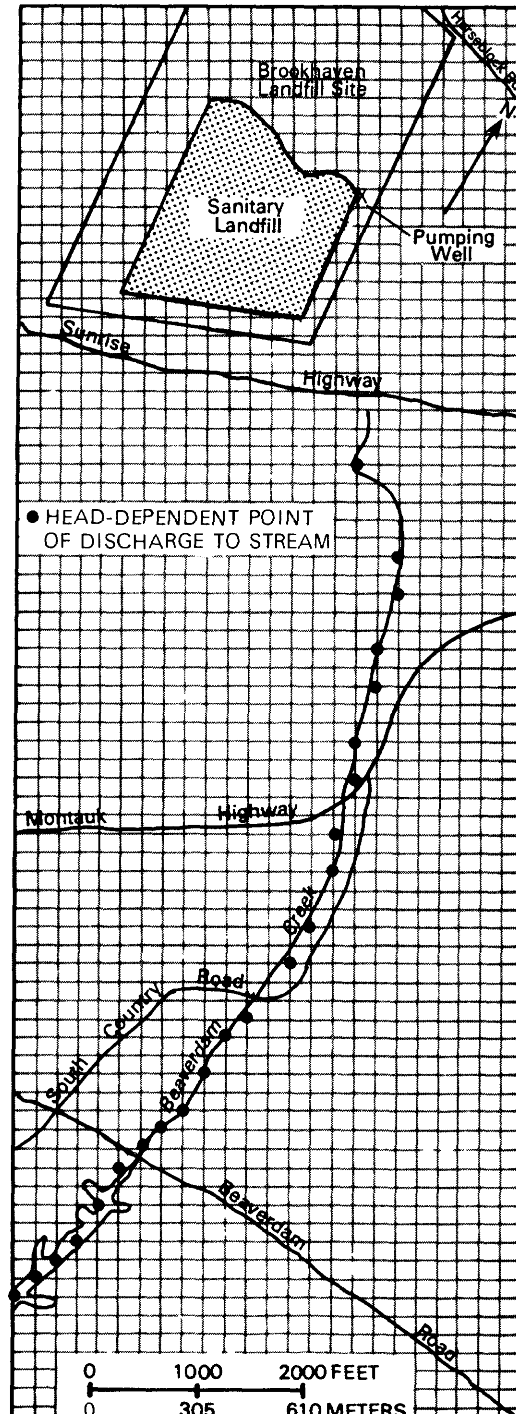
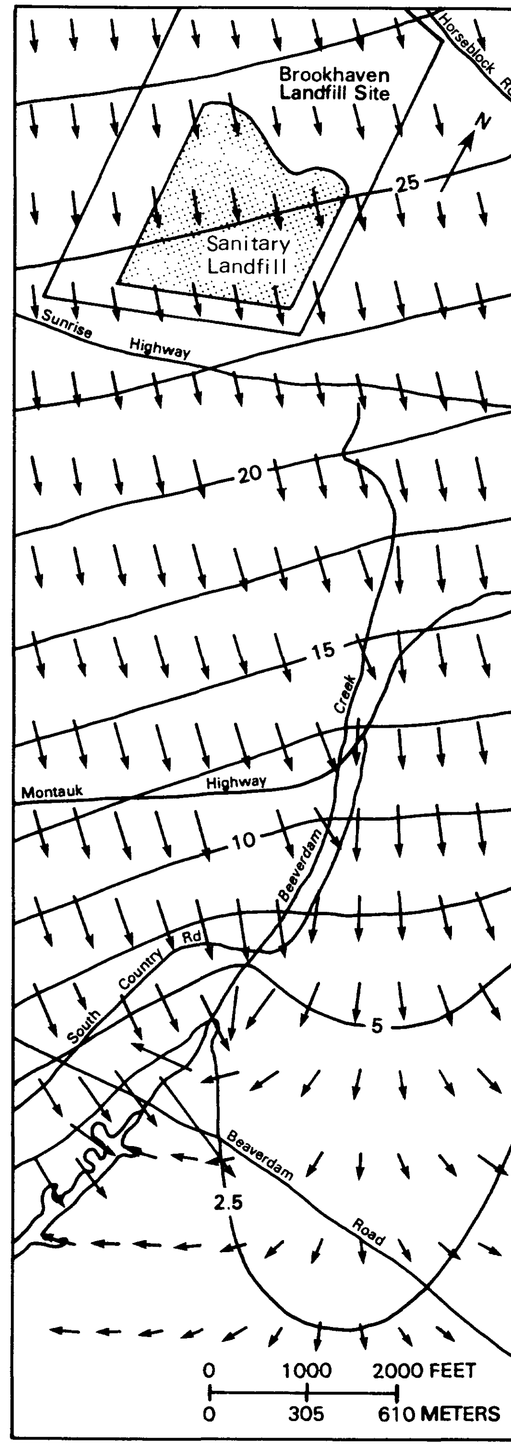


Figure 8.--Fine-scale finite-element grid used to represent the study area. (Location is shown in fig. 4.)

The modified Tracy code was used to solve equation 1 and calculate aquifer heads. Lines of equal water-table altitude shown in figure 9 closely match those of the larger scale model (fig. 7), except that the fine-scale model produces a smoother solution in the vicinity of Beaverdam Creek. A summary of inflows and discharges from the study area as computed by the fine-scale flow model is given in table 2. Flow rates differ slightly from those obtained in the first simulation; this is due primarily to the more accurate representation of discharge to Beaverdam Creek in the fine-scale model.



EXPLANATION

SCALED GROUND-WATER-VELOCITY VECTOR

0 2 4 6 FEET PER DAY

0 1 METER PER DAY

—15— LINE OF EQUAL SIMULATED WATER-TABLE ALTITUDE--Contour interval 2.5 feet. Datum is sea level

Figure 9.--Simulated water-table altitude and scaled vector plot of simulated ground-water velocities in the study area.

Table 2.--Simulated inflows to and discharges from study area.

[Values are in cubic feet per second;
inflows are positive, discharges are negative.
Locations are shown in fig. 8.]

Inflows and discharges	Flow rates (ft^3/s)
Inflows	
Inflow across model boundaries	4.76
Upward leakage into upper glacial aquifer	2.48
Recharge from precipitation	4.20
Outflows	
Outflow across model boundaries	-4.13
Downward leakage out of upper glacial aquifer	<-0.01
Withdrawal at pumping wells	-0.01
Discharge to flowing reaches of Beaverdam Creek	-5.18
Discharge to tidal reaches of Beaverdam Creek	-2.12

Simulation of Chloride Migration in the Study Area

A computer code, titled SUTRA (for Saturated-Unsaturated Transport), was used to simulate the advective-dispersive transport of a conservative solute downgradient from the landfill site. SUTRA is capable of simulating transient-state, two-dimensional, fluid-density-dependent, saturated or unsaturated ground-water flow and solute transport and can easily simulate transient solute movement in a steady-state, saturated ground-water-flow system where fluid density is constant. SUTRA uses a modified Galerkin finite-element technique to solve equations 2 and 8 and calculates the steady-state head values and time-dependent concentrations at nodes of the finite-element grid. The computer code is documented in a report by Voss (1984).

Chloride ion was selected as the conservative solute because it is not subject to adsorption, biological decay, or chemical precipitation and because it is present in high concentrations in leachate and in leachate-contaminated ground water near the landfill. Other conservative solutes are expected to behave in a similar fashion and could be simulated, given data on influent and background concentrations. The SUTRA code can also simulate transport of nonconservative solutes subject to linear adsorption or first-order decay, but this was beyond the scope of the study.

Model Input Data

Solution of the solute-transport equation (eq. 8) requires data on distribution of ground-water velocities and the specification of boundary and initial conditions. Values for coefficients of longitudinal and transverse dispersivity and data on the configuration, rates of inflow, and solute concentrations associated with the contaminant source are also required.

Ground-water-velocity distribution.--To determine the distribution of ground-water velocities, the SUTRA code first solves the linearized ground-water flow equation (eq. 2). SUTRA cannot be used to simulate water-table conditions when simulating areal ground-water flow, head-dependent flux-boundary conditions, or head-dependent discharges. Instead, data obtained in simulations of ground-water flow in the study area with the modified Tracy code (described in a previous section) were used to determine the effective aquifer-transmissivity values (T in eq. 2) for each element and the net recharge (Q) at each node. Prescribed head values at nodes along the lateral boundaries of the grid (fig. 8) were the same in both simulations. The steady-state head values, as determined by the SUTRA code, were identical (to the nearest 0.005 ft) to those obtained with the modified Tracy code.

The SUTRA code computed steady-state pore velocities for centers of all elements in the fine-scale grid from simulated head values and an effective porosity of 0.30. A scaled-vector plot of the simulated velocities is shown in figure 9. Resulting velocities range between 0.3 and 5.8 ft/d, with a velocity of 1.13 ft/d in the center of the landfill site. Velocities differ slightly from those reported by Wexler and Maus (1988) because the finer scale grid provides better representation of discharge to Beaverdam Creek.

Boundary and initial conditions.--Initial conditions were specified with chloride concentration equal to 10 mg/L throughout the study area at the start of the simulation. Chloride concentrations in ground water flowing across model boundaries were set equal to 10 mg/L to represent ambient chloride concentrations in the upper glacial aquifer. Rates of inflow across the lateral boundaries of the study area (which were treated as prescribed-head boundaries in the ground-water flow simulation) are calculated internally by the SUTRA code. Solute fluxes are determined from the calculated inflow rate and the inflow concentration.

Chloride concentrations in areal recharge and in upward leakage from the Magothy aquifer were also set equal to a uniform value of 10 mg/L. Leachate entry into the upper glacial aquifer is discussed further on. No other point sources or distributed sources of chloride ion were simulated in this study.

Solute concentrations in outflow from the study area, including discharge at prescribed-head boundaries, seepage to streams, and downward leakage to the Magothy aquifer, were equal to the simulated concentrations calculated at discharge nodes and were not specified as boundary conditions. Solute fluxes can be determined from the specified or computed discharge rate and simulated nodal concentrations.

An important consideration is that, in the two-dimensional simulation, discharge from the aquifer has a concentration equal to the vertically averaged concentration at that point in the aquifer. In actuality, differences in the vertical distribution of contaminants would cause the concentrations in discharge to partially penetrating wells or shallow streams to differ from the vertically averaged value.

Solute source.--The rate at which leachate is generated within the landfill is a function of the rate of recharge to the landfill and the capacity of the refuse and soil cover to store moisture. The rate of recharge to the landfill is unknown but is assumed to be greater than that to the surrounding area because the sand and gravel used as cover material allows rapid infiltration and minimal evapotranspiration (Wexler, 1988). Channeling of water through the refuse has allowed leachate to move down toward the base of the landfill even though the capacity of the material to retain moisture may not yet have been reached. Leachate was found to have ponded above the liner as early as 1976 (Wexler, 1988).

As indicated earlier, the mechanism of leachate entry into the aquifer beneath the lined landfill is unknown. Water-quality data indicate that it may be flowing out through holes in the liner or along separated seams. The rate of leakage out of the liner bottom is assumed to be less than the infiltration rate to account for ponding of leachate above the liner. Sufficient leachate accumulated to fill the shallow basin formed by the liner and overflowed along the eastern side of the landfill.

In this study, the rate of leachate entry to the aquifer was assumed to equal the rate of average annual recharge for the surrounding area (24.6 in/yr). Overflow along the eastern side of the landfill was not simulated because it probably started only 1 to 2 years before the October-December 1982 sampling period (Wexler, 1988). Pumping of leachate and transport to a treatment facility was begun in August 1982 to prevent further overflow.

A total of 74 nodes representing a 60-acre area beneath the sanitary landfill were used to simulate the inflow of leachate to the aquifer. The staged growth of the landfill since 1974 was simulated by increasing the number of nodes representing the landfill. Locations of nodes and the years at which they began to contribute leachate are shown in figure 10. Leachate is assumed to have entered the upper glacial aquifer since the end of 1976,

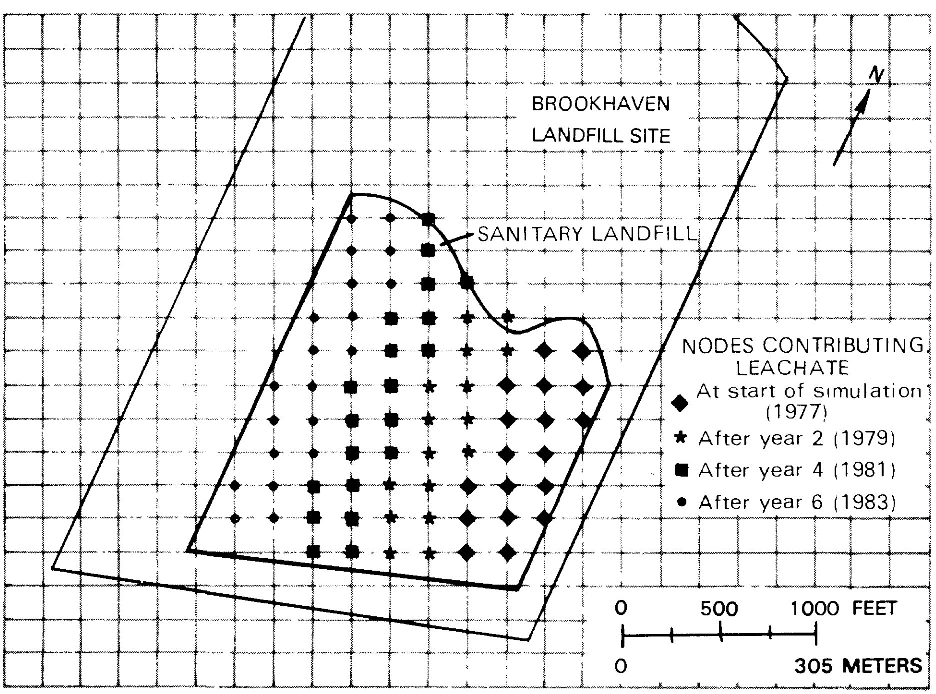


Figure 10.--Location of nodes used to simulate discharge of leachate from the sanitary landfill to the upper glacial aquifer. (Site location is shown in fig. 2.)

when ponding of leachate above the liner was first noticed. Thus, the year 1977 corresponds to year 1 in the simulation. The landfill excavation in the southwest part of the site was expanded in 1982 and was lined with a double liner. Leakage of leachate through the double liner was considered improbable and therefore was not simulated in this study.

Chloride concentrations in leachate samples collected during the study ranged from 690 to 1,100 mg/L. A value of 875 mg/L was used as an average source concentration in the simulation. Sensitivity analyses of the effect of source concentration on model predictions are discussed further on.

In the model, leachate is assumed to be mixed throughout the entire thickness of the aquifer beneath each point of entry. Water-quality data from monitoring-well clusters along the perimeter of the landfill site show that contamination is present at almost all depths (Wexler, 1988). As an additional simplifying assumption, concentrations were assumed to be uniform with depth. Monitoring data showed that concentrations did vary slightly with both depth and time, however.

The density differences between leachate entering the aquifer and ambient water may be an important factor in the vertical movement of leachate throughout the aquifer thickness within a few hundred feet of the sanitary landfill; however, the effects of density differences could not be considered in this areal simulation, nor could localized mounding beneath the points of leachate entry be examined. The effect of these factors on vertical movement of leachate could be analyzed with either a fully three-dimensional or a cross-sectional model in which density-dependent ground-water flow can be simulated, but this was beyond the scope of the study.

Aquifer-dispersivity values.--Results of two previous simulations of contaminant plumes on the south shore of Long Island (Pinder, 1973, and Gureghian and others, 1981) were used as guidelines in selecting appropriate values of aquifer dispersivity. Hydrogeologic conditions at these sites are similar to those at the Brookhaven landfill site, and the scales of the simulations are also similar. In the study by Pinder (1973) of a plume of hexavalent chromium in South Farmingdale (fig. 1), model calibration yielded values of 70 ft for longitudinal dispersivity, a_l, and 14 ft for transverse dispersivity, a_t. The study by Gureghian and others (1981) of the leachate plume from the Babylon landfill (fig. 1) yielded higher values--140 ft for a_l and 25 ft for a_t. Ratios of transverse to longitudinal dispersivity obtained in the two studies are 1:5.0 and 1:5.6, respectively.

This study used values of 100 ft for longitudinal dispersivity and 20 ft for transverse dispersivity. Sensitivity analyses, in which the effect of varying these values was studied, are described in a subsequent section.

Model Results

Plume migration over a 12-year period, wherein the first year of leachate entry into the water-table aquifer is designated year 1 (1977) was simulated with the SUTRA model. Each 1-year interval consisted of 10 equal time steps of 36.5 days. Reducing the time step by half caused no difference in concentrations in a 1-year test simulation.

A graphics routine was developed to contour the nodal concentration data; results of the simulations, plotted as lines of equal chloride concentration at the end of years 2 through 12 (December 1978 through December 1988) are presented in figure 11. The 20-mg/L contour is used to delineate the extent of the leachate plume. Limitations of the two-dimensional approach and assumptions and simplification made in the model development must be considered in evaluations of these results, however.

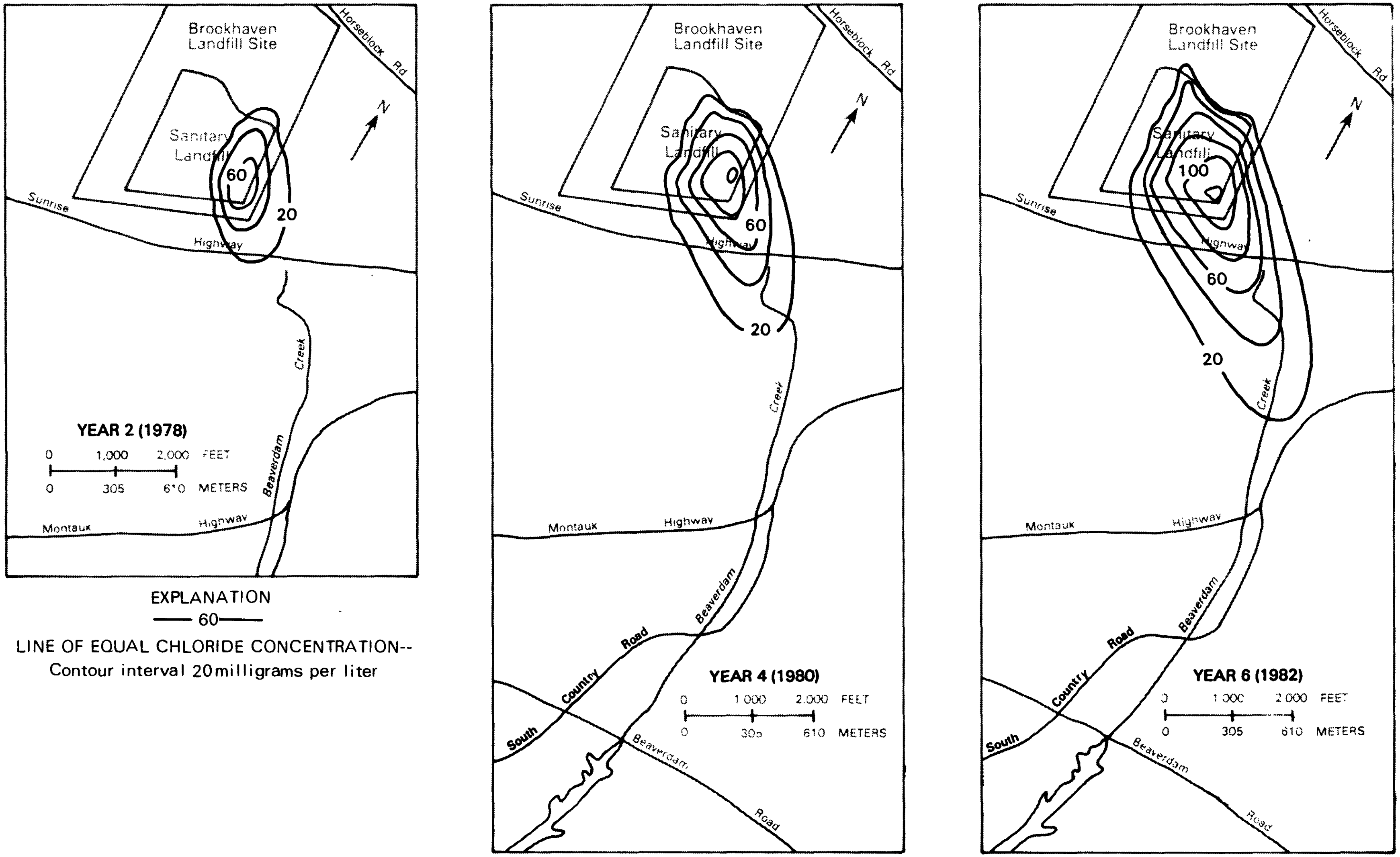
Model calibration.--Water-quality data collected during October-December 1982 were compared with the simulated chloride concentrations at the end of year 6 (December 1982). Although data on the vertically averaged field concentrations downgradient of the landfill are lacking, the simulated areal configuration of the plume matches the plume's observed distance traveled and width fairly well, as indicated in figures 5A and 5B (p. 11).

The observed and simulated chloride concentrations differ most sharply along the eastern boundary of the landfill. The higher observed concentrations are due to overflow of leachate rather than leakage beneath the landfill, which was not simulated in the model. The match between the observed and simulated concentrations elsewhere indicates that, despite uncertainties in values for aquifer properties and simplifying assumptions made in the model, the values used in the simulation are reasonable, and the model can be considered calibrated.

Predicted plume migration in years 7 through 12.--Simulations made with the calibrated model for years 7 through 12 (fig. 11) represent the predicted configuration of the plume from the start of 1983 through the end of 1988. The toe of the plume (as defined by the 20-mg/L chloride contour) crosses Montauk Highway and enters the residential area of Brookhaven hamlet by the end of year 7 (December 1983). After 12 years of travel, the plume extends 6,200 ft downgradient of the site and has a maximum width, perpendicular to the plume centerline, of 2,600 ft. The maximum simulated chloride concentrations at the southeastern corner of the landfill site and at Montauk Highway are 160 mg/L and 55 mg/L, respectively.

Steady-state simulation.--If model simulations were done for additional years with all conditions remaining constant, steady-state conditions would eventually be achieved. Under steady-state conditions, a dynamic equilibrium develops in which solute continues to migrate from the landfill to points of discharge, but solute concentrations remain constant. To determine the concentration distribution under these conditions, the model was run for a single large time step (10,000 years). The steady-state plume, shown in figure 12, extends 8,800 ft downgradient of the landfill site and passes beyond the northeastern boundary of the study area, where it intersects Little Neck Run. Lateral spreading near the toe of the plume is primarily due to the divergence of ground-water flow in the area between the two streams (Beaverdam Creek and Little Neck Run).

Additional simulations were done to determine the amount of time needed to reach a steady-state condition. The plume configuration after a 30-year period closely matched that of the steady-state plume.



EXPLANATION

—— 60 ——

LINE OF EQUAL CHLORIDE CONCENTRATION--
Contour interval 20 milligrams per liter

Figure 11.--Predicted extent of chloride plume 2, 4, and 6 years after assumed start of leachate entry into upper glacial aquifer. (Location is shown in fig. 2.)

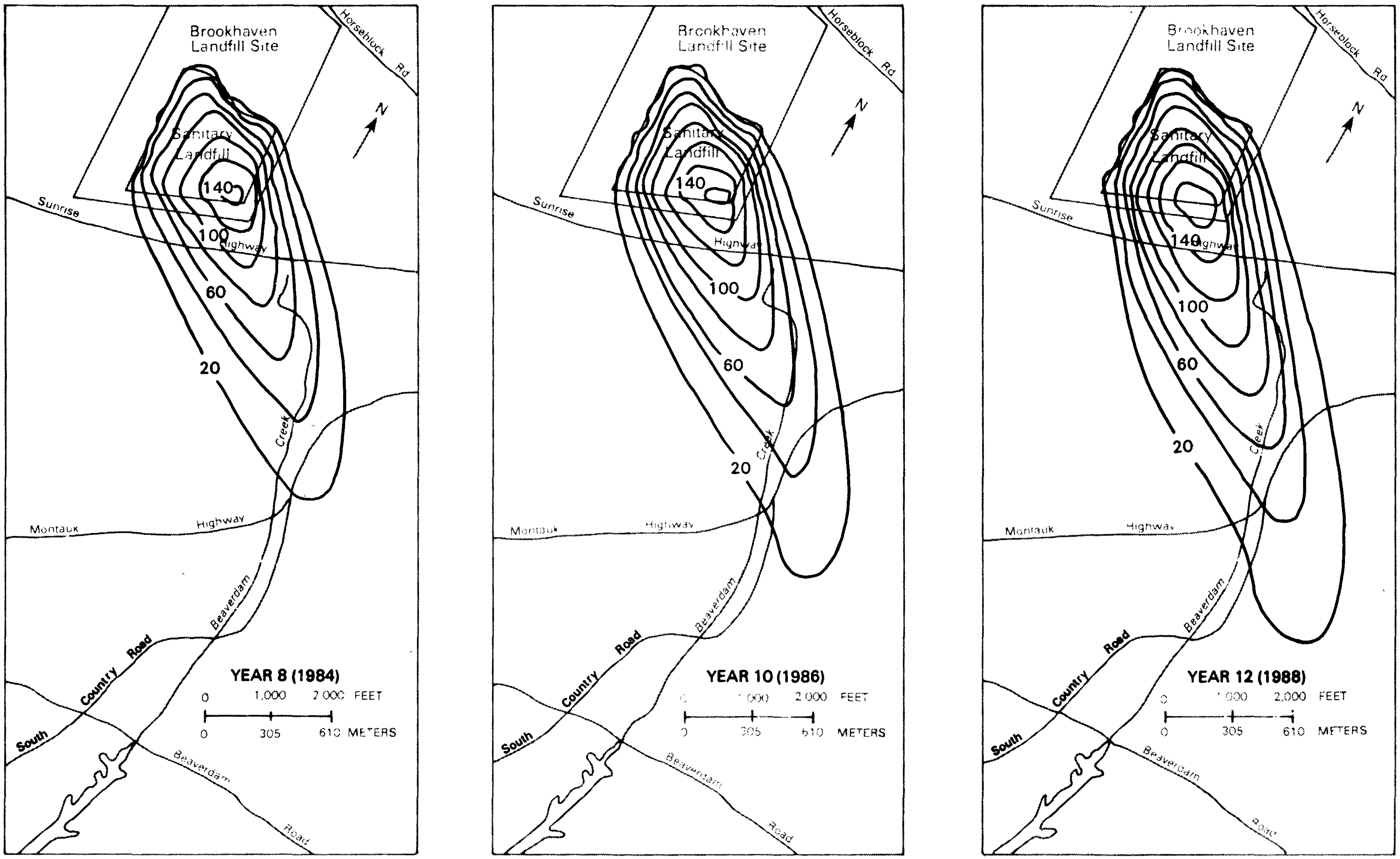
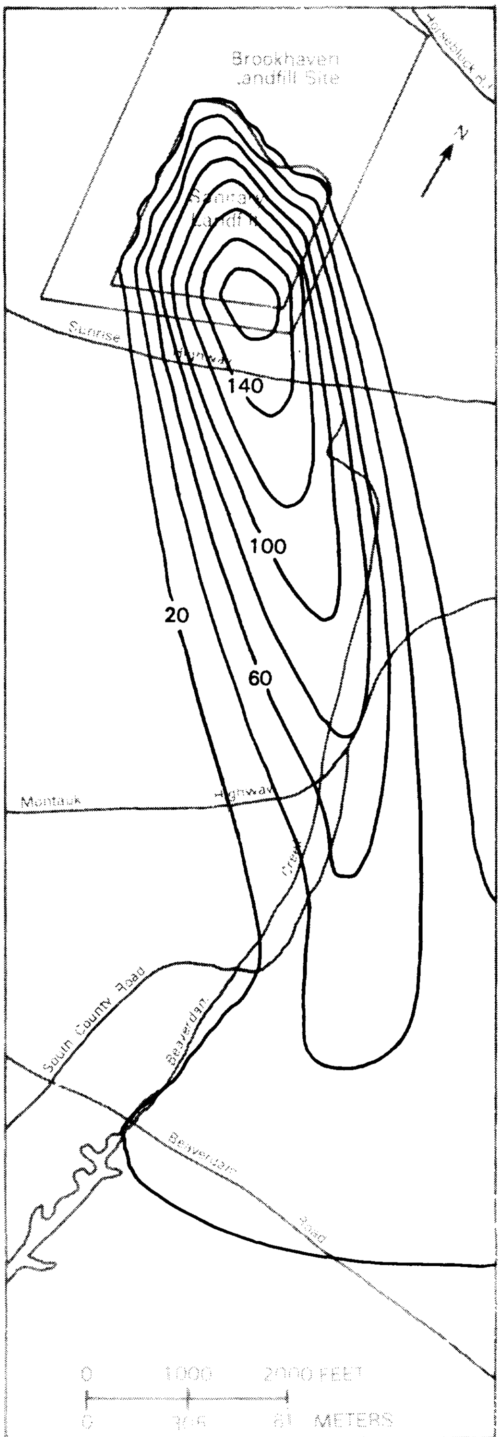


Figure 11 (continued).--Predicted extent of chloride plume 8, 10, and 12 years after assumed start of leachate entry into upper glacial aquifer. (Location is shown in fig. 2.)

Sensitivity Analyses

A degree of uncertainty is associated with the aquifer properties and solute concentrations used in the solute-transport model. Additional simulations were made to evaluate the sensitivity of predicted solute concentrations to changes in the values of several model variables. In each simulation, a single variable was changed within a selected range of values while all other model variables were held constant. Sensitivity analyses were made for effective porosity, longitudinal and transverse dispersivity, and chloride concentration in the leachate. Values used in the sensitivity analyses are given in table 3.

In addition to the variables mentioned above, model predictions would be particularly sensitive to changes in values of hydraulic conductivity of the upper glacial aquifer and confining units and to rates of recharge and discharge because these terms affect both the rate and direction of ground-water flow. Reasonable values for these variables had been obtained through calibration of the ground-water flow model as discussed previously; therefore, model variables associated with the ground-water-flow simulation were not included among the sensitivity analyses described below.



EXPLANATION

—— 60 ——

LINE OF EQUAL CHLORIDE CONCENTRATION--
Contour interval 20 milligrams per liter

Figure 12.--Simulated steady-state chloride concentrations.

Aquifer porosity.--Simulated chloride concentrations at the end of years 4, 8, and 12, based on an effective porosity equal to 0.24, are plotted in figure 13A; those for the same years with an effective porosity equal to 0.36 are shown in figure 13B. This range is considered reasonable for the upper glacial aquifer. Comparison of the six maps shows that model results are highly sensitive to changes in porosity because (1) pore velocity is inversely proportional to the effective porosity; (2) dispersion coefficients, which are linearly proportional to the pore velocities, are greater at lower porosity values, and (3) a larger volume of aquifer is occupied by the influent leachate at lower porosity values. The plume with effective porosity equal to 0.24 (fig. 13A) is wider and has moved farther downgradient than the plume with porosity equal to 0.36. The maximum difference in plume travel in the 12th year of the simulation is about 1,500 ft. Maximum chloride concentrations in the area are about the same in both simulations.

Longitudinal and transverse dispersivity.--Several sensitivity runs were made with longitudinal dispersivity values (a_l) ranging from 50 to 150 ft. The ratios of transverse to longitudinal dispersivity (a_t) were held constant at 1:50. The lower value of a_l (50 ft) is equal to one-fourth the length of the elements in the fine-scale grid (which are 168 ft by 200 ft). Simulations with the SUTRA code with dispersivity values smaller than one-fourth the grid size could be subject to numerical dispersion errors (Voss, 1984, p. 232). The upper a_l value in these analyses (150 ft) is based on an analysis of previous studies and is therefore close to the maximum value for the upper glacial aquifer on the south shore of Long Island.

Simulated chloride concentrations at the ends of years 4, 8, and 12 with a_l and a_t equal to 50 ft and 10 ft, respectively, are shown in figure 14A; those at the ends of years 4, 8, and 12 with a_l and a_t equal to 150 ft and 30 ft, respectively, are shown in figure 14B. The resulting chloride concentrations do not differ significantly, which indicates that, at least over the ranges simulated, chloride concentrations are not particularly sensitive to the dispersivity value used.

Influent chloride concentration.--Simulations were made with chloride concentrations in the leachate equal to 700 and 1,050 mg/L. These values represent reasonable upper and lower limits for the long-term average chloride concentration in leachate entering the upper glacial aquifer. As expected, the simulated concentrations (figs. 15A and 15B) were sensitive to this term. The maximum concentrations near the landfill site at the end of year 12 differ by about 60 mg/L. The two simulated plumes do not differ greatly in overall extent, however.

Table 3.--Range of values used in the sensitivity analyses.

Term	Model value	Range: Maximum	Range: Minimum	Percent difference
Effective porosity	0.30	0.36	0.24	20
Longitudinal dispersivity	100 ft	150 ft	50 ft	50
Transverse dispersivity	20 ft	30 ft	10 ft	50
Concentration of solute source	875 mg/L	1050 mg/L	700 mg/L	20

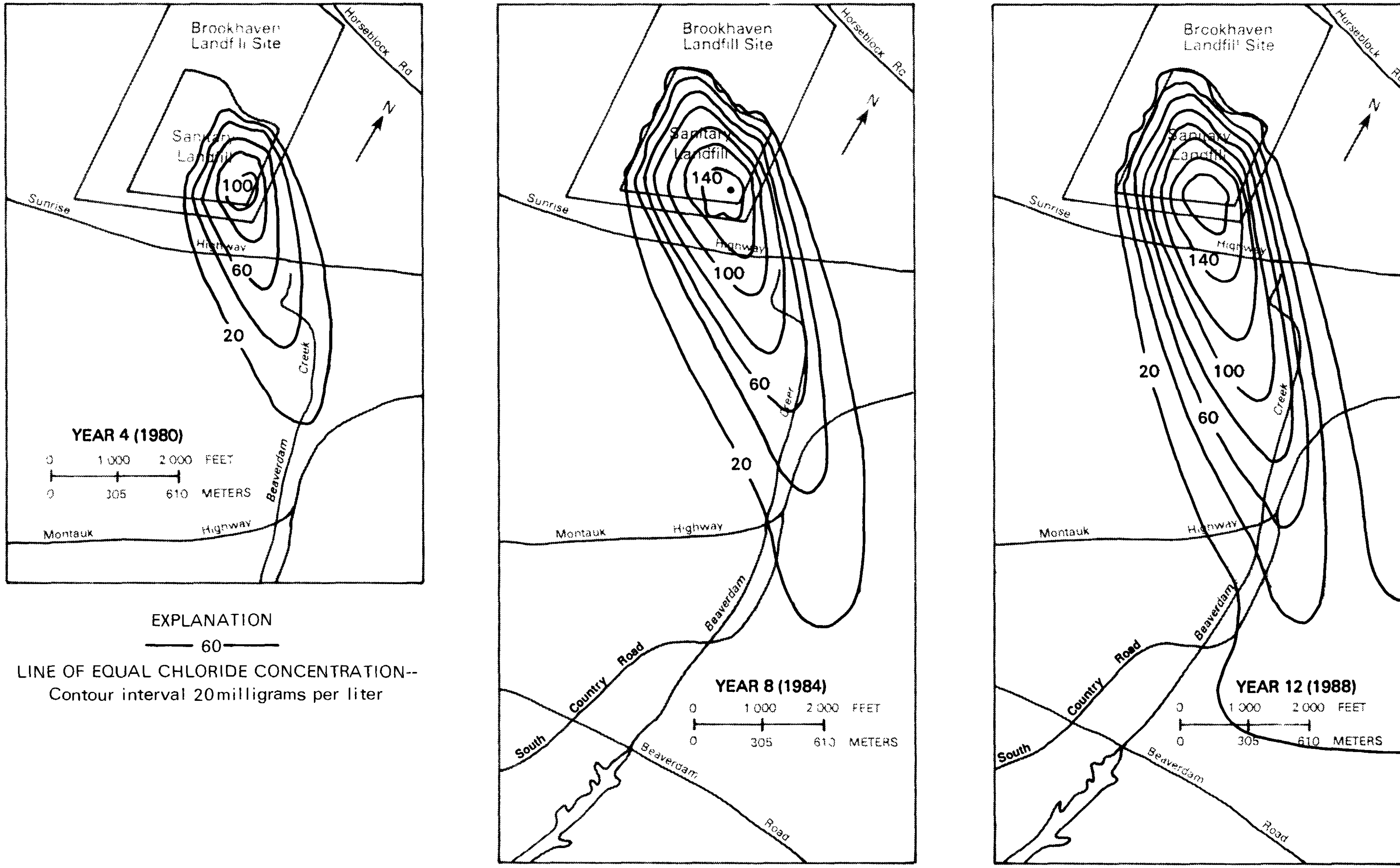


Figure 13A.--Simulated chloride concentrations at end of years 4, 8, and 12 with effective porosity equal to 0.24.

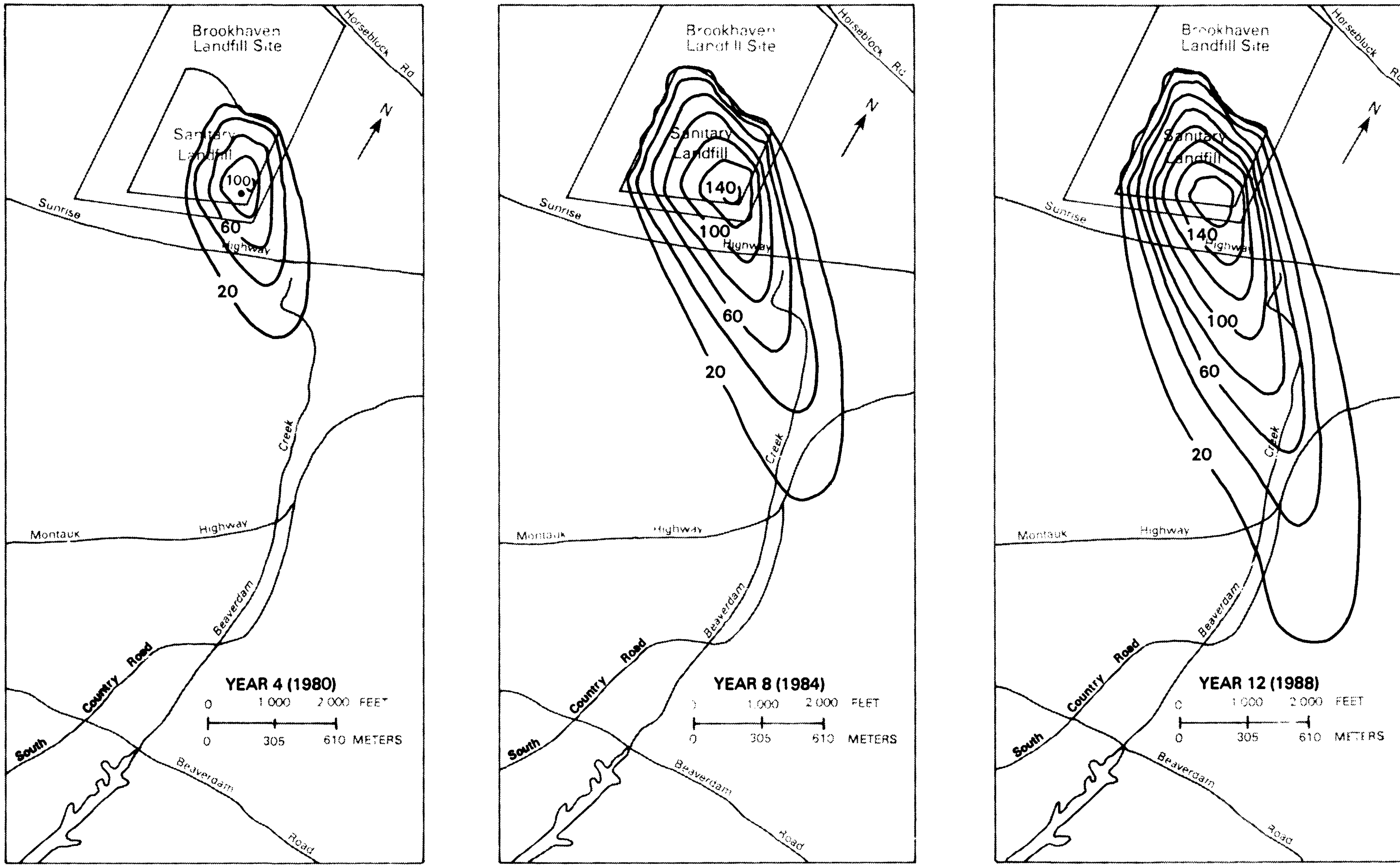


Figure 13B.--Simulated chloride concentrations at end of years 4, 8, and 12 with effective porosity equal to 0.36.

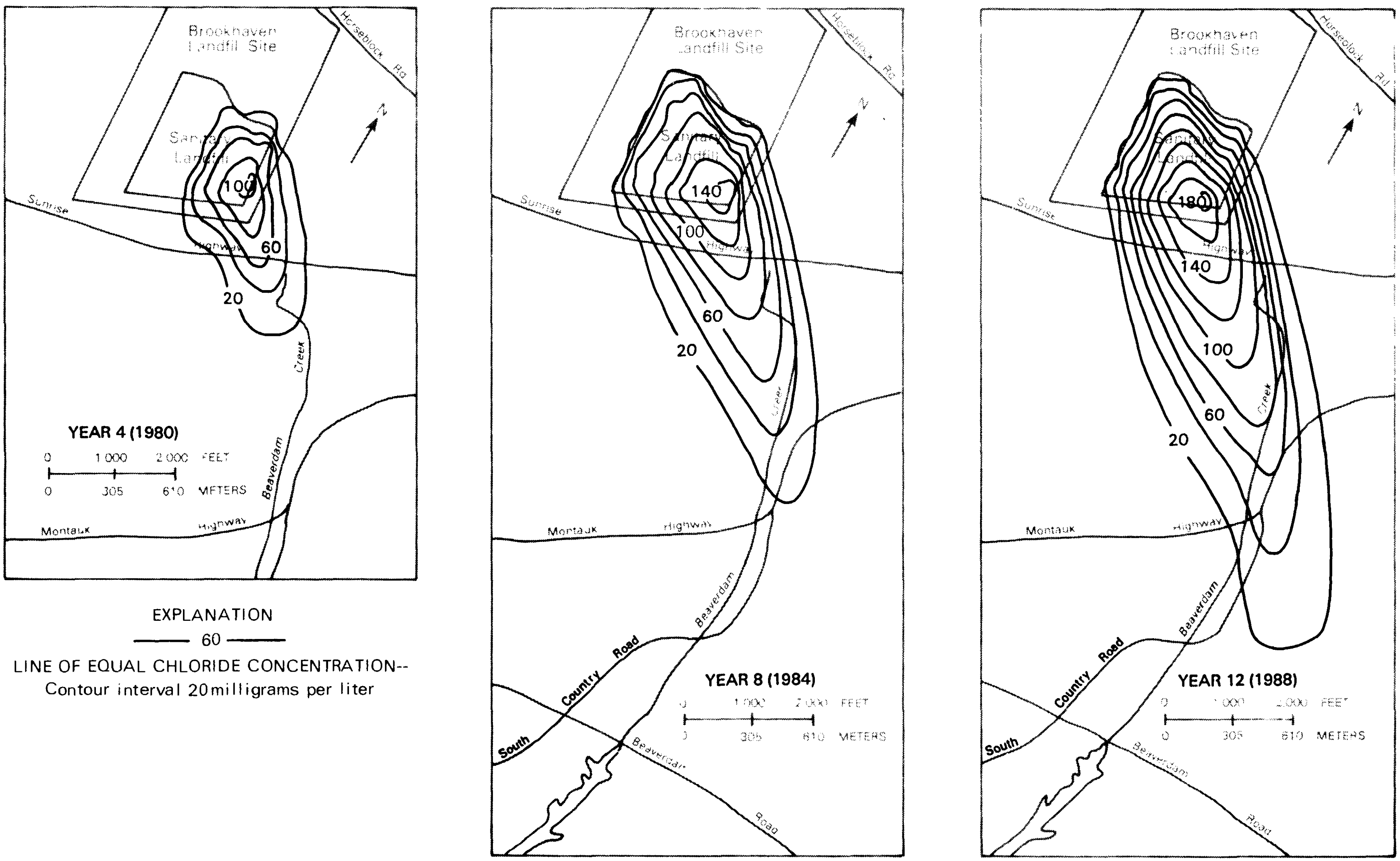


Figure 14A.--Simulated chloride concentrations at end of years 4, 8, and 12 with longitudinal dispersivity equal to 50 ft and transverse dispersivity equal to 10 ft.

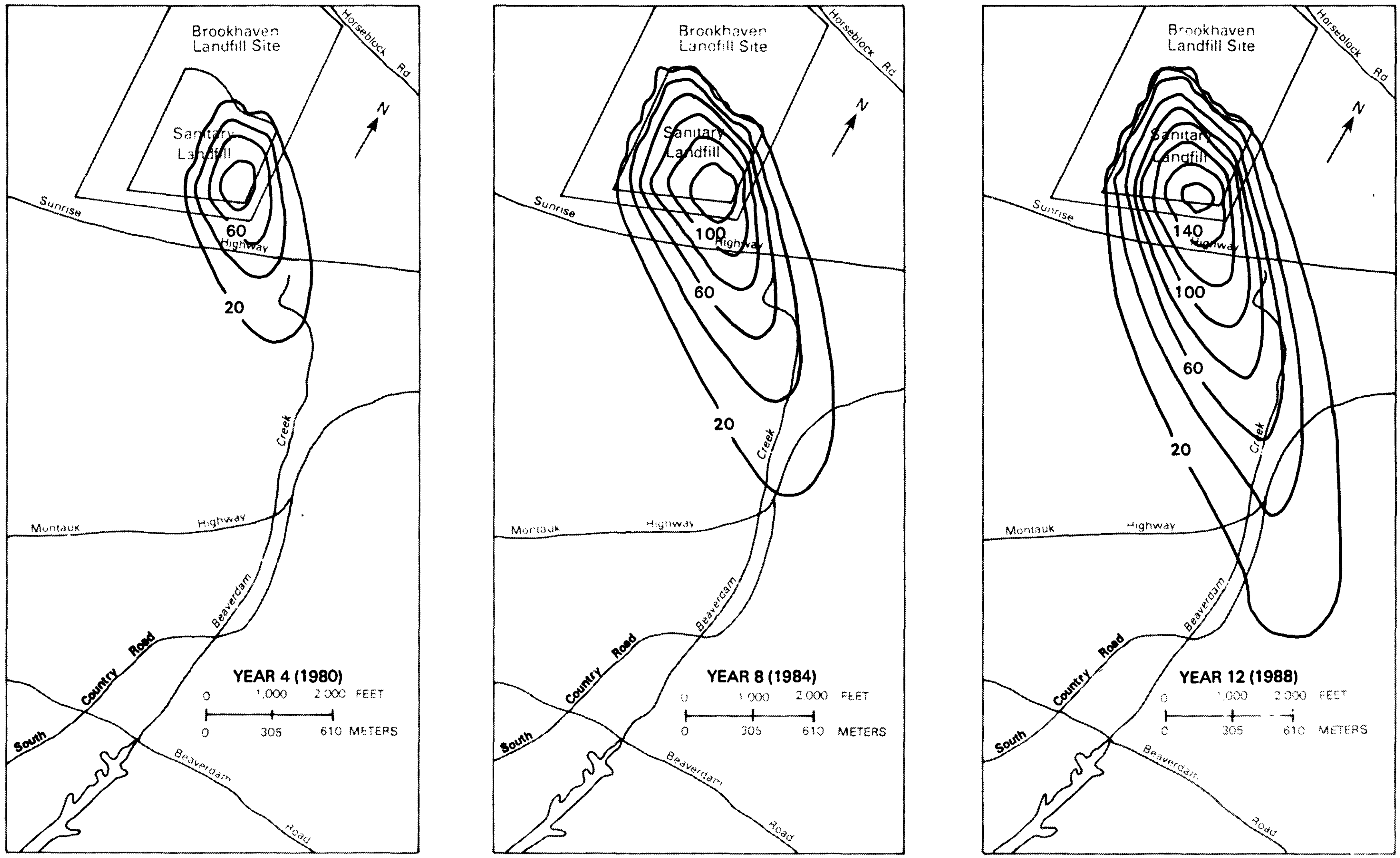


Figure 14B.--Simulated chloride concentrations at end of years 4, 8, and 12 with longitudinal dispersivity equal to 150 ft and transverse dispersivity equal to 30 ft.

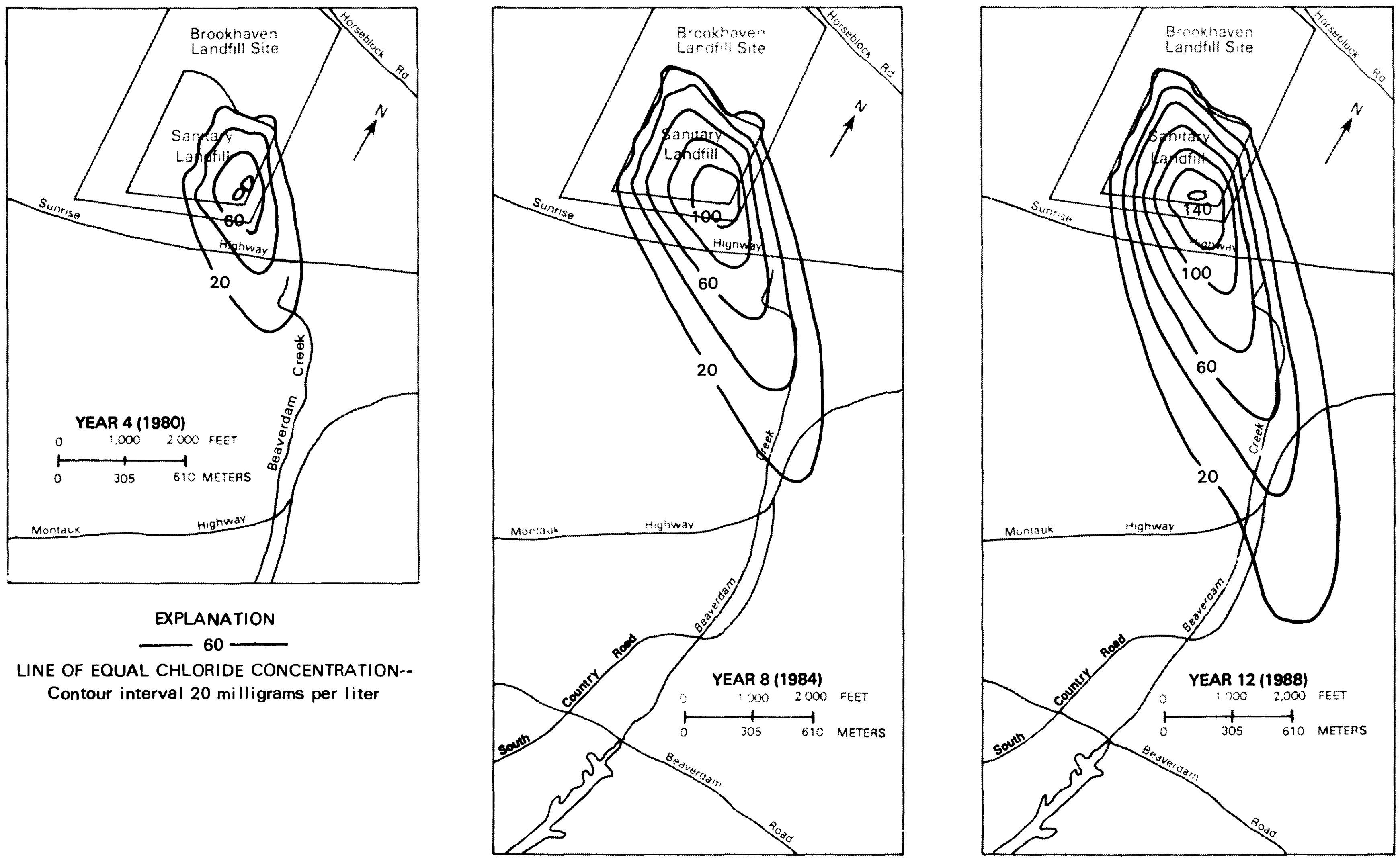


Figure 15A.--Simulated chloride concentrations at end of years 4, 8, and 12 with chloride concentration in leachate equal to 700 mg/L.

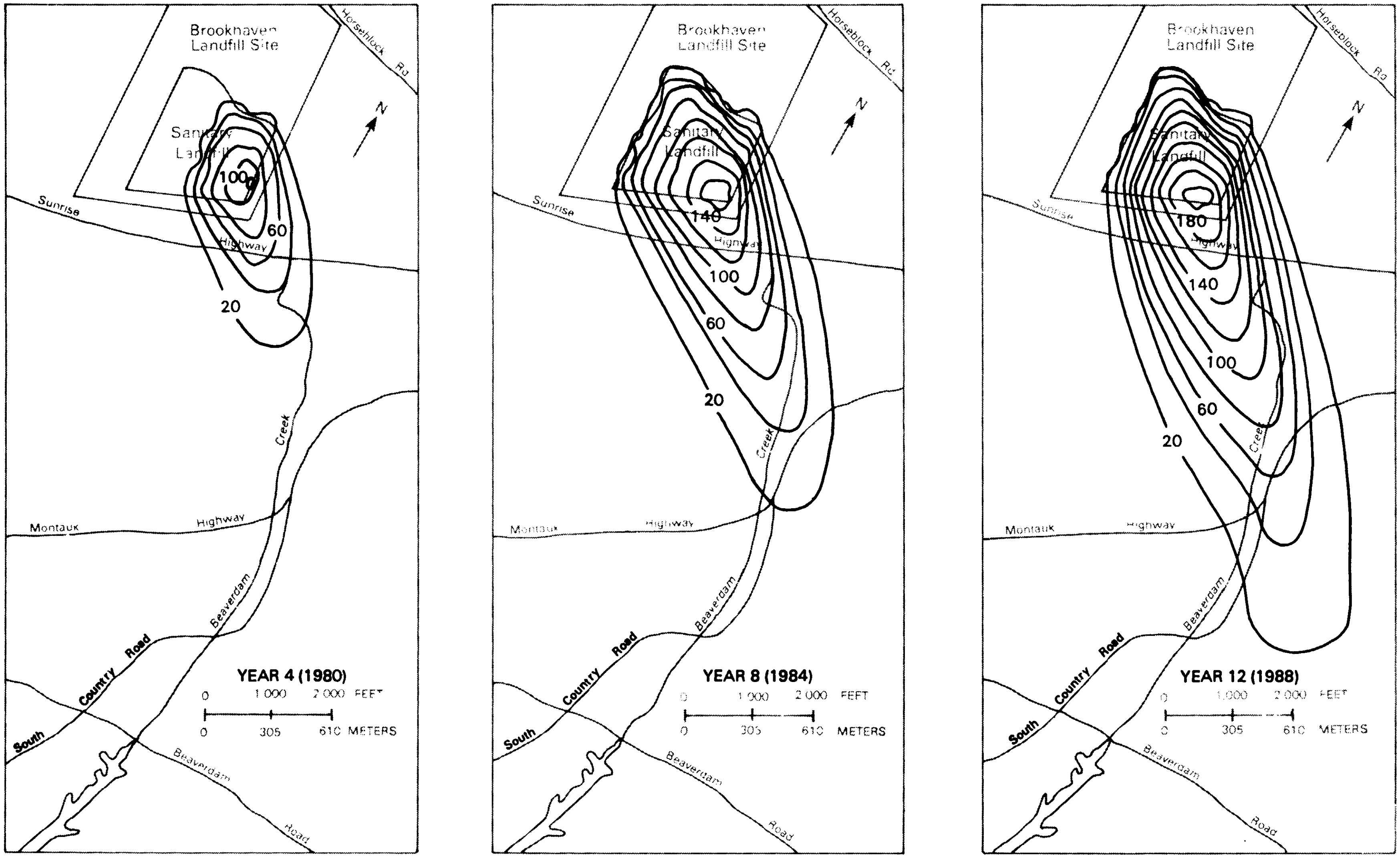


Figure 15B.--Simulated chloride concentrations at end of years 4, 8, and 12 with chloride concentration in leachate equal to 1,050 mg/L.

Predicted Effects of Remedial Actions

In addition to simulating the migration of contaminants under natural ground-water-flow conditions, the solute-transport model can be used to evaluate remedial actions in which contaminants are removed from the aquifer or their spread is prevented. Three hypothetical remedial strategies were examined with the solute-transport model; the simulations were intended for illustrative purposes only and are not meant as recommendations or as endorsements of a particular option. The results are subject to the limitations imposed by the assumptions and simplifications discussed in the earlier sections on model development.

In all simulations, remedial actions are assumed to take place at the start of the 8th year of the simulation, in January 1984. Aquifer properties are the same as in the predictive simulations, but all simulated remedial actions create changes in the ground-water flow system, either by pumping or through diversion of recharge that normally enters the aquifer beneath the landfill. This change in flow made it necessary to recalculate the distribution of pore velocities by the procedures described earlier. In the simulation of remedial actions that included pumping, the effects were sufficient to require recalculation of ground-water levels throughout the 26-mi^2 flow-model area. Results from that model were used to update the prescribed-head values along the boundaries of the fine-scale grid. Ground-water flow in the study area was first simulated with the modified Tracy code, and the updated velocity distribution was determined by the SUTRA code.

Method 1: Capping the Landfill

Capping the landfill with an impermeable surface would prevent further generation of leachate and halt the discharge of leachate to the upper glacial aquifer. Capping of the landfill was simulated to determine the time required for chloride concentrations in the plume to reach background levels under natural conditions. Neither removal of contaminated ground water nor the prevention of chloride migration by pumping was simulated.

Capping of the sanitary landfill was assumed to be complete at the end of year 7, and entry of landfill leachate into the aquifer was halted immediately. Precipitation that would normally infiltrate the landfill is diverted to a stormwater basin along the east edge of the landfill and recharges the water table at that point. Changes in the ground-water flow system as a result of the diversion were calculated by the ground-water flow model for the study area and found to be minor.

The predicted chloride concentrations at the ends of years 8, 10, 12, 14, and 16 are plotted in figure 16. Dilution by uncontaminated recharge, dispersive mixing, and discharge to Beaverdam Creek are the principal mechanisms for reducing the concentration in the plume over time. The plume leaves the landfill site at the end of year 12 and becomes more elliptical. Because leachate migration is not prevented, the plume passes beneath residential areas of Brookhaven hamlet south of Montauk highway. The simulated plume has a maximum chloride concentration of 50 mg/L as it crosses Montauk Highway at the end of year 13. Chloride concentrations within Brookhaven hamlet are generally below 40 mg/L.

The benefits of this remedial action are that it does not incur costs for pumping and treatment of water. A cost that may be incurred, however, aside from capping the landfill, is that of providing an alternative water supply to residential areas in Brookhaven hamlet. Even though simulated chloride concentrations are well below the New York State drinking-water standard of 250 mg/L, the presence of other contaminants and a disagreeable taste and odor may make the leachate-contaminated water unfit for domestic use.

Method 2: Pumping of Leachate-Recovery Wells

The second simulation analyzed the effect that pumping a line of recovery wells would have on the leachate plume. The landfill was assumed to remain uncapped and to produce leachate at a constant rate and concentration. Four wells were placed perpendicular to the plume centerline (fig. 17) and pumped to recover contaminated water. A pumping rate of 0.75 ft^3/s per well was selected, for a total pumpage of 3.0 ft^3/s. Pumped water is assumed to be treated and discharged at some point on Beaverdam Creek. The resulting water-table altitudes are shown in figure 17.

The simulated chloride concentrations at the end of years 8, 10, 12, and 14 are plotted in figure 18. Southward migration of contaminants is effectively halted at the line of wells at the end of year 8, although a small part of the plume (with chloride concentrations below 40 mg/L) does break away. By the end of year 14, steady-state conditions are achieved. The principal advantage in this remedial action is that costs of capping and providing water supply to parts of Brookhaven hamlet are avoided. Construction and operation costs associated with pumpage and treatment facilities will be incurred, however. Recovery and treatment of leachate-contaminated ground water would have to continue as long as leachate is produced.

The pumping rates tested and the number of wells and their locations used in the simulation are reasonable but probably do not represent optimum values. The SUTRA model has been coupled with optimization programs (Gorelick and others, 1984) and could be applied to the study area as a means of developing a more efficient remedial strategy, but this was beyond the scope of the study.

Method 3: Capping the Landfill and Pumping Recovery Wells

The third simulation analyzed the effect of combining the first two methods. In this simulation, the landfill was assumed to be completely capped at the start of year 8, and each well in the line of recovery wells was pumped at 0.75 ft^3/s. All other assumptions governing the first two simulations were applied to this simulation.

Simulated chloride concentrations at the end of years 8, 10, and 12 are plotted in figure 19. Plume migration is halted at the line of recovery wells, and, by the end of year 12, the bulk of the contamination has been removed. The principal advantage of this action is that costs of providing water supply to parts of Brookhaven hamlet and of pumping and treating the water can be avoided after year 12. Construction and operation costs for pumpage and treatment facilities would be the major costs incurred.

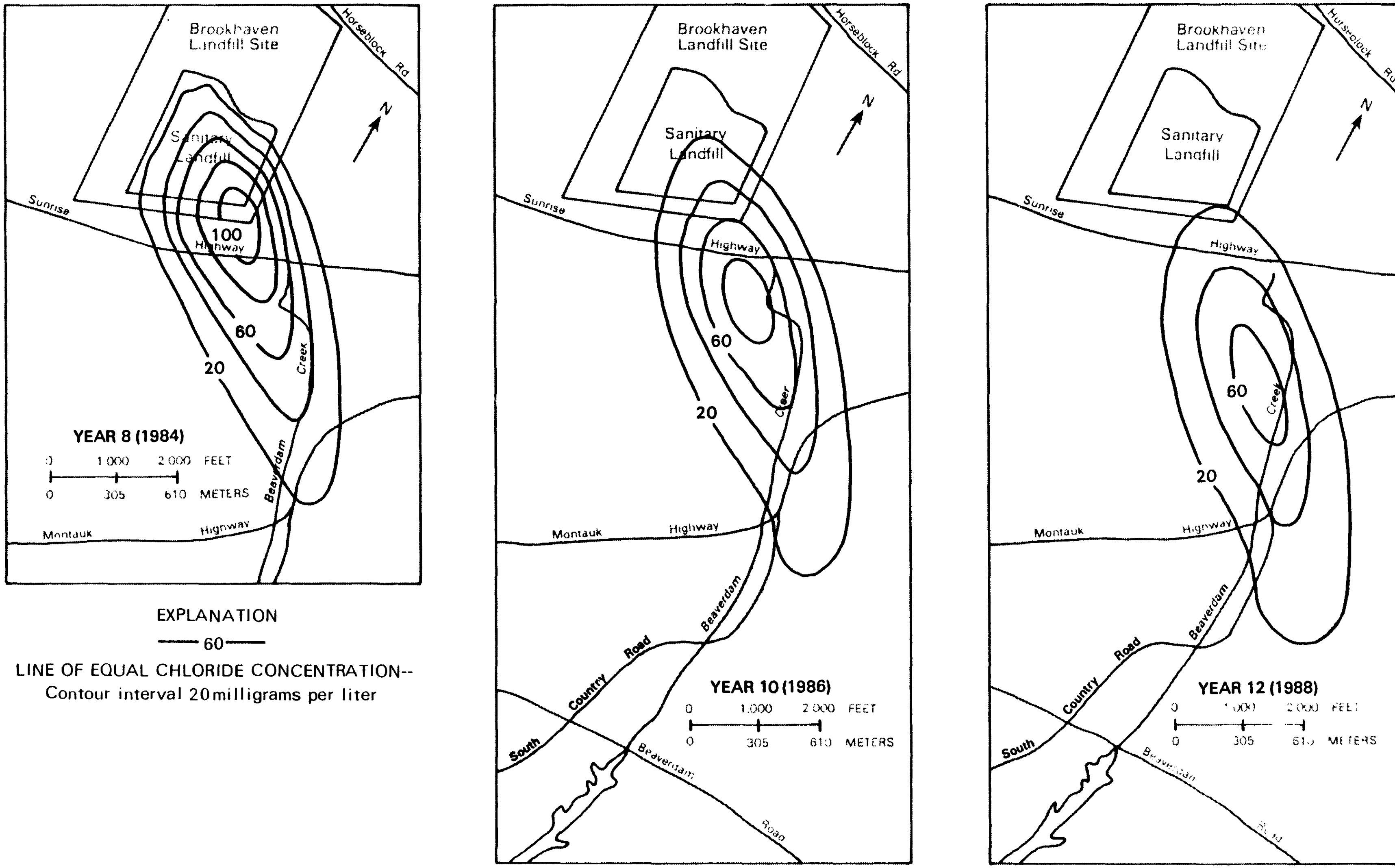


Figure 16.--Simulated chloride concentrations at end of years 8, 10, and 12 with the landfill capped at the start of year 8 (1984).

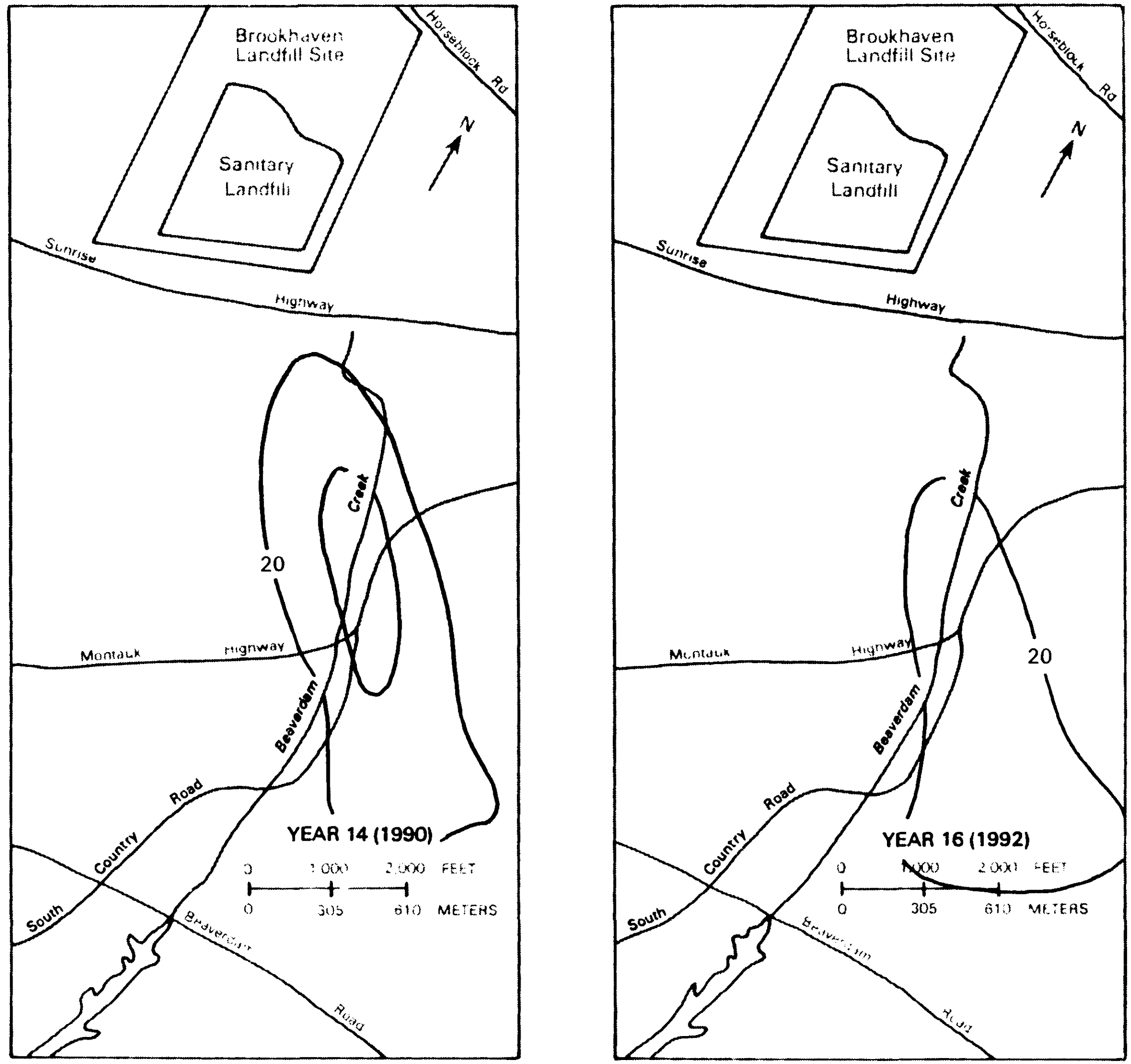
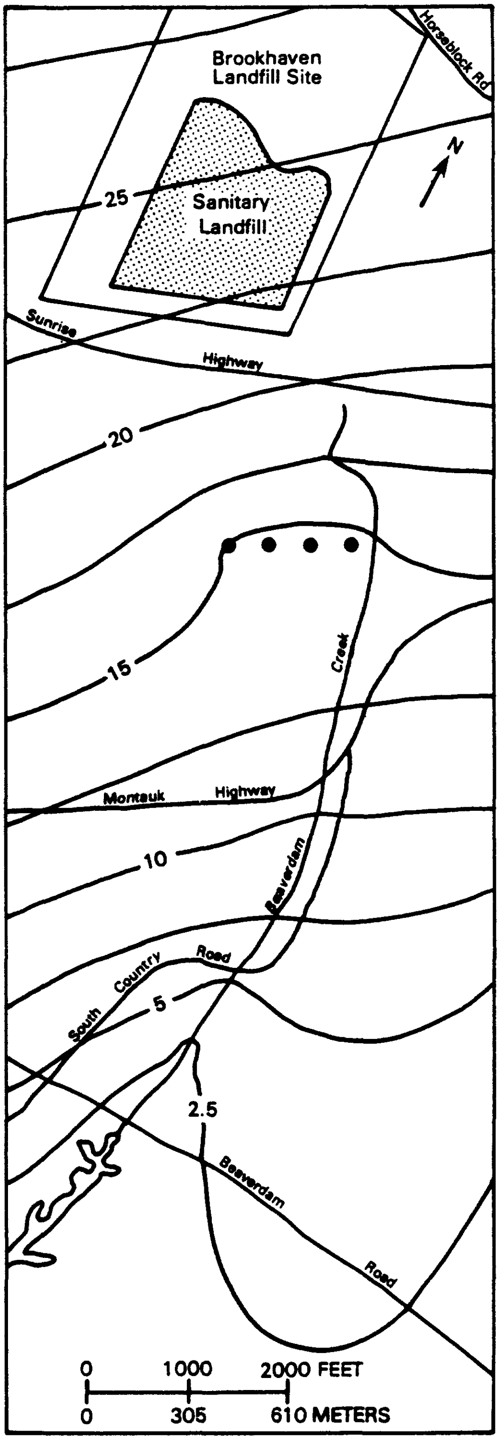


Figure 16 (continued).--Simulated chloride concentrations at end of years 14 and 16 with the landfill capped at the start of year 8 (1984).



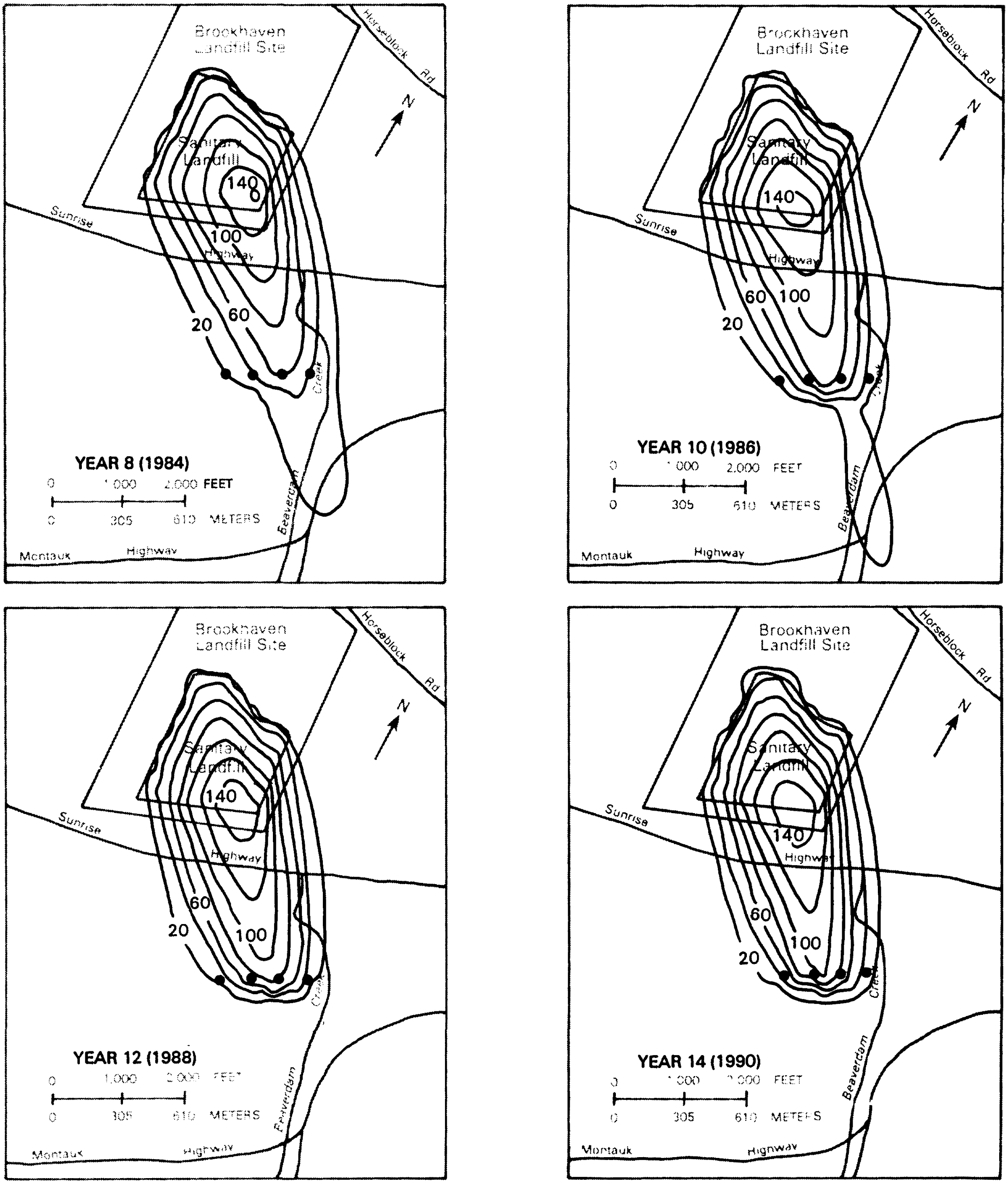
EXPLANATION

● LOCATION OF RECOVERY WELL

—10— LINE OF EQUAL SIMULATED WATER-TABLE ALTITUDE--Contour interval 2.5 feet. Datum is sea level

Figure 17.

Simulated water-table altitude in the study area with each of four recovery wells pumping at 0.75 ft^3/s.

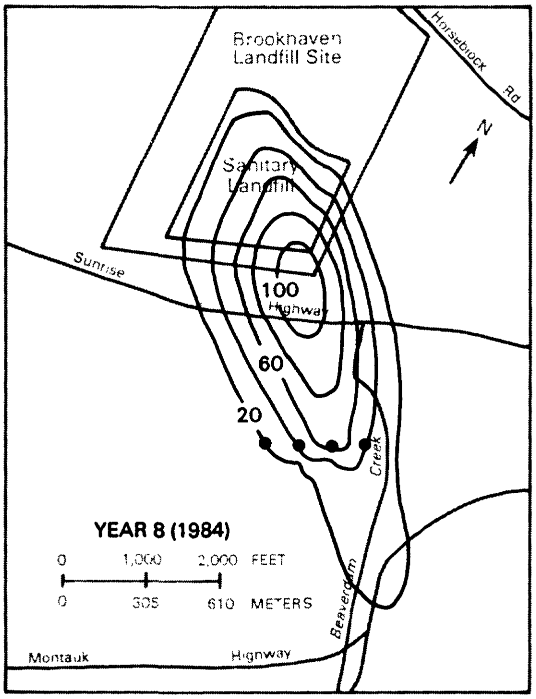


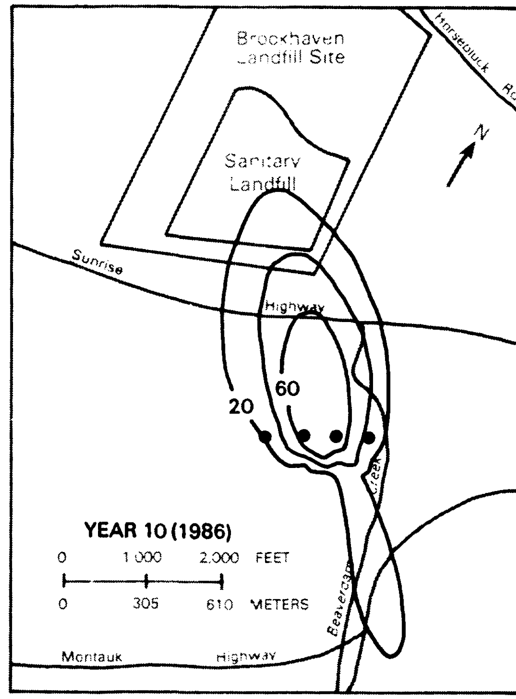
EXPLANATION

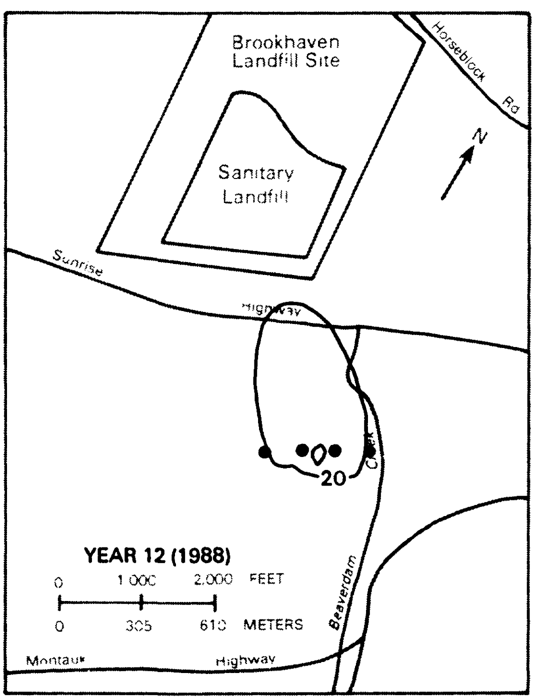
—— 60 —— LINE OF EQUAL CHLORIDE CONCENTRATION--
Contour interval 20 milligrams per liter

● RECOVERY WELL

Figure 18.--Simulated chloride concentrations at end of years 8, 10, 12, and 14 with each of four recovery wells pumping at 0.75 ft^3/s at the start of year 8 (1984).







EXPLANATION

—— 60 ——

LINE OF EQUAL CHLORIDE CONCENTRATION--
Contour interval 20 milligrams per liter

●

Recovery well

Figure 19.

Simulated chloride concentrations at the end of years 8, 10, and 12 with the landfill capped and each of four recovery wells pumping at 0.75 ft^3/s at the start of year 8 (1984).

SUMMARY AND CONCLUSIONS

A solute-transport model was developed in a hydrologic investigation of the Brookhaven landfill site in the Town of Brookhaven. The model represents a 2.3 mi^2 area surrounding and extending downgradient from the landfill and was used to simulate advective-dispersive migration of a conservative solute from the landfill through the upper glacial aquifer.

The model was developed in three phases. First, steady-state ground-water flow in the 26 mi^2 area encompassing the landfill was simulated as described in a companion report by Wexler and Maus (1988). Second, part of the finite-element grid was discretized more finely over a 2.3-mi^2 area adjacent and downgradient from the site for detailed ground-water flow simulations. Values of prescribed head along the lateral boundaries of the fine-scale grid were interpolated from water-table altitudes generated in the initial simulation. Hydraulic properties of the aquifer and confining unit were identical to those obtained through calibration of the larger model. Ground-water flow within the 2.3-mi^2 study area under steady-state conditions was simulated, and water-table altitudes, ground-water discharge to Beaverdam Creek, and head-dependent leakage rates at grid nodes were calculated.

In the third phase, the SUTRA code was used to simulate solute transport in the 2.3-mi^2 study area. Ground-water velocities were computed by the SUTRA code from the linearized form of the ground-water flow equation (eq. 2). Values for effective transmissivity and net recharge to the aquifer were obtained from the flow simulation in step 2. Effective porosity of the upper glacial aquifer was assumed to equal 0.30.

The solute-transport model simulated 12 years of chloride migration. Chloride was selected because it is present at high concentrations in the leachate and is universally accepted as a conservative solute. Chloride concentration in leachate entering the upper glacial aquifer was set equal to 875.0 mg/L. The rate of leachate entry was assumed equivalent to the average annual rate of recharge, 24.6 inches per year. Background chloride concentration and concentrations in recharge were set at 10 mg/L. Longitudinal and transverse dispersity in the upper glacial aquifer were set equal to 100 ft and 20 ft, respectively.

The model was judged to be calibrated when simulated chloride concentrations at the end of year 6 (corresponding to December 1982) matched reasonably well the chloride data collected in October-December 1982. At the end of year 7 in the simulation (December 1983), the plume crossed Montauk Highway and entered the residential part of Brookhaven hamlet. By the end of year 12 (December 1988), the plume had a maximum length of 6,200 ft and a maximum width of 2,600 ft. Maximum chloride concentrations at the landfill were 160 mg/L and, at Montauk Highway, 55 mg/L. In this simulation, chloride leaves the study area in the ground-water discharge to Beaverdam Creek.

A steady-state analysis showed that discharge to Beaverdam Creek, together with dilution by dispersion and mixing with recharge water of lower concentration, tends to stabilize the plume configuration. Maximum plume travel in the steady-state simulation was approximately 8,800 ft. Additional simulations indicated that equilibrium would be approached after about 30 years.

Sensitivity analyses in which aquifer porosity, longitudinal and transverse dispersivities, and influent chloride concentration were varied over selected ranges showed that the plume configuration was highly sensitive to changes in effective porosity but not longitudinal and transverse dispersivity. Chloride concentrations within the plume were most sensitive to changes in influent concentrations.

Three remedial strategies--capping the landfill, pumping a line of recovery wells, and a combination of the two procedures--were simulated. The first simulation indicated that if the landfill were capped, chloride concentrations would be reduced to near background levels at the end of year 16 through dispersion, dilution, and discharge to Beaverdam Creek. Simulated concentrations were generally below 40 mg/L as the plume moved through residential areas of Brookhaven hamlet south of Montauk Highway. Simulation of the four recovery wells, each pumped at 0.75 ft^3/s, prevented leachate migration beyond the line of the wells. Pumping would have to continue as long as leachate continued to enter the aquifer, however. Capping, in conjunction with pumping each of the four wells at 0.75 ft^3/s, would also prevent plume migration beneath the residential areas of the hamlet. The simulation indicated that the bulk of contaminants would be removed at the end of year 12, at which point pumping could be discontinued.

Results of these simulations indicate that the model can simulate solute transport under conditions both natural and altered by remedial actions. Thus, it can be used to predict movement of conservative solutes downgradient of the landfill site, calculate concentrations at specific times and locations, and analyze ground-water-quality management alternatives. Interpretation of model results, however, should always include consideration of the limitations of the two-dimensional approach and the other simplifying assumptions made in the development of the transport model.

REFERENCES CITED

Anderson, M. P., 1984, Movement of contaminants in groundwater--Groundwater transport-advection and dispersion, in Groundwater Contamination: Washington, D.C., National Academy Press, 179 p.

Bear, Jacob, 1979, Hydraulics of groundwater: New York, McGraw-Hill, 569 p.

Dunlap, L. E., Lindgren, R. J., and Carr, J. E., 1984, Projected effects of ground-water withdrawals in the Arkansas River Valley, 1980-89, Hamilton and Kearny Counties, southwestern Kansas: U.S. Geological Survey Water-Resources Investigations Report 84-4082, 168 p.

Franke, O. L., and Getzen, R. T., 1976, Evaluation of hydrologic properties of the Long Island ground-water reservoir using cross-sectional analog models: U.S. Geological Survey Open-File Report 75-679, 80 p.

Franke, O. L., and McClymonds, N. E., 1972, Summary of the hydrologic situation on Long Island, New York, as a guide to water-management alternatives: U.S. Geological Survey Professional Paper 627-F, 59 p.

REFERENCES CITED (Continued)

Freeze, R. A., and Cherry, J. A., 1979, Groundwater: Englewood Cliffs, N.J., Prentice-Hall, 604 p.

Fried, J. J., 1975, Ground water pollution--theory, methodology, modeling, and practical rules: New York, Elsevier, 330 p.

Gelhar, L. W., and Axness, C. L., 1981, Stochastic analysis of macrodispersion in three-dimensionally heterogeneous aquifers: Socorro, N.M., New Mexico Institute of Mining and Technology, Geophysical Research Center, Hydrology Research Program, Report No. H8, 140 p.

Gelhar, L. W., Gutjahr, A. L., and Naff, R. L., 1979, Stochastic analysis of macrodispersion in a stratified aquifer: Water Resources Research, v. 15, p. 1387-1397.

Gorelick, S. M., Voss, C. I., Gill, P. E., Murray, Walter, Saunders, M. A., and Wright, M. H., 1984, Aquifer reclamation design--the use of contaminant transport simulation combined with nonlinear programming: Water Resources Research, v. 20, no. 4, p. 415-427.

Gray, W. G., and Hoffman, J. L., 1983, A numerical study of ground-water contamination from Price's Landfill, New Jersey. II. Sensitivity analysis and plume simulation: Ground Water, v. 21, no. 1, p. 15-21.

Gureghian, A. B., Ward, D. S., and Cleary, R. W., 1981, A finite-element model for the migration of leachate from a sanitary landfill in Long Island, New York, Part II, Application: Water Resources Bulletin, v. 17, no. 1, p. 62-66.

Jensen, H. M., and Soren, Julian, 1971, Hydrogeologic data from selected wells and test holes in Suffolk County, Long Island, New York: Suffolk County Department of Environmental Control, Long Island Water Resources Bulletin 3, 35 p.

__________ 1974, Hydrogeology of Suffolk County, Long Island, New York: U.S. Geological Survey Hydrologic Investigation Atlas HA-501, 2 sheets.

Kimmel, G. E., and Braids, O. C., 1980, Leachate plumes in ground water from Babylon and Islip landfills, Long Island, New York: U.S. Geological Survey Professional Paper 1085, 38 p.

Konikow, L. F., 1977, Modeling chloride movement in the alluvial aquifer at the Rocky Mountain Arsenal, Colorado: U.S. Geological Survey Water-Supply Paper 2044, 43 p.

Pearsall, K. A., and Wexler, E. J., 1986, Organic compounds in ground water near a sanitary landfill in the Town of Brookhaven, Long Island, New York: U.S. Geological Water-Resources Investigations Report 85-4218, 36 p.

REFERENCES CITED (Continued)

Pickens, J. F., and Lennox, W. C., 1976, Numerical simulation of waste movement in steady ground-water flow systems: Water Resources Research, v. 12, no. 2, p. 171-180.

Pinder, G. F., 1973, A Galerkin finite-element simulation of ground-water contamination on Long Island, New York: Water Resources Research, v. 9, no. 6, p. 1657-1669.

Pinder, G. F., and Gray, W. G., 1977, Finite-element simulation in surface and subsurface hydrology: New York, Academic Press, 259 p.

Scheidegger, A. E., 1961, General theory of dispersion in porous media: Journal of Geophysical Research, v. 66, no. 10, p. 3273-3278.

Sykes, J. F., Pahwa, S. B., Lantz, R. B., and Ward, D. S., 1982, Numerical simulation of flow and contaminant migration at an extensively monitored landfill: Water Resources Research, v. 18, no. 6, p. 1687-1704.

Voss, C. I., 1984, A finite-element simulation model for saturated-unsaturated, fluid-density-dependent ground-water flow with energy transport or chemically-reactive single-species solute transport: U.S. Geological Survey Water-Resources Investigations Report 84-4369, 409 p.

Warren, M. A., DeLaguna, Wallace, and Lusczynski, N. J., 1968, Hydrology of Brookhaven National Laboratory and vicinity, Suffolk County, New York: U.S. Geological Survey Bulletin 1156-C, 127 p.

Wang, H. F., and Anderson, M. P., 1982, Introduction to ground-water modeling--finite difference and finite element methods: San Francisco, W. H. Freeman, 237 p.

Wexler, E. J., 1988, Ground-water flow and solute transport at a municipal landfill site on Long Island, New York--part 1, hydrogeology and water quality: U.S. Geological Survey Water-Resources Investigations Report 86-4070, 43 p.

Wexler, E. J., and Maus, P. E., 1988, Ground-water flow and solute transport at a municipal landfill site on Long Island, New York--part 2, simulation of ground-water flow: U.S. Geological Survey Water-Resources Investigations Report 86-4106, 44 p.

☆ U.S. GOVERNMENT PRINTING OFFICE: 1988—201 - 979 / 60004