MOVEMENT AND FATE OF SOLUTES IN A PLUME OF SEWAGE-CONTAMINATED GROUND WATER, CAPE COD, MASSACHUSETTS: U.S. GEOLOGICAL SURVEY TOXIC WASTE GROUND-WATER CONTAMINATION PROGRAM

U.S. GEOLOGICAL SURVEY

Open-File Report 84-475

Papers presented at the Toxic Waste Technical Meeting
Tucson, Arizona, March 20-22, 1984

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Denis R. LeBlanc, Editor

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WILLIAM P. CLARK, Secretary

GEOLOGICAL SURVEY

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PREFACE

Over the past decade, the need for understanding the mechanisms of contamination of ground water by toxic wastes has become exceedingly important. Many private and government studies of contaminated aquifers have described the movement, chemical alteration, and dispersal of toxic chemicals in ground-water systems. These studies have also shown, however, that transport of contaminants in aquifers is a very complex process. Many technical questions are yet to be answered about the behavior of specific chemicals under different hydrologic and geologic conditions. Answers to these questions are needed to guide the evaluation and cleanup of contaminated ground-water supplies and to ensure safe use and disposal of toxic chemicals.

The U.S. Geological Survey has begun a nationwide program to study the fate of toxic wastes in ground water. Several sites where ground water is known to be contaminated are being studied by interdisciplinary teams of geohydrologists, chemists, and microbiologists. The objective of these studies is to obtain a thorough quantitative understanding of the physical, chemical, and biological processes of contaminant generation, migration, and attenuation in aquifers. The knowledge obtained from these site-specific studies will contribute greatly to successful evaluation, monitoring, and remedial action at similar sites where contamination by toxic chemicals occurs.

One of the sites being studied by the U.S. Geological Survey under this program is a plume of sewage-contaminated ground water on Cape Cod, Massachusetts. The plume was formed by land disposal of treated sewage to a glacial outwash aquifer since 1936. Although sewage generally is not considered a hazardous waste, even relatively "clean" domestic sewage contains many inorganic and organic chemicals such as sodium, nitrate, detergents, and volatile organic compounds which can be toxic and render a ground-water supply unfit for use. Research on how these sewage-derived contaminants move in the Cape Cod aquifer will add to the understanding of the complex interactions of geohydrologic, chemical, and microbiological processes that affect contaminant migration.

This report summarizes results obtained during the first year of research at the Cape Cod site under the U.S. Geological Survey Toxic-Waste Ground-Water Contamination Program. The seven papers included in this volume were presented at the Toxic Waste Technical Meeting, Tucson, Arizona, in March 1984. They provide an integrated view of the subsurface distribution of contaminants based on the first year of research and discuss hypotheses concerning the transport processes that affect the movement of contaminants in the plume.

- A mathematical model of solute transport is used to simulate and evaluate the movement of dissolved constituents within the ground-water-flow system.

- The distributions of inorganic and organic chemicals in the plume are described and preliminary conclusions are presented about the behavior of selected contaminants in the aquifer.

- A special technique to determine concentrations of semi-volatile organic compounds in the plume at the nanogram per liter level is evaluated and used to identify possible organic tracers of the contaminated ground water.

- The distributions of inorganic nitrogen, organic carbon, and bacterial populations in the aquifer are used to show that microbial activity significantly affects the fate of some contaminants.

- The abundance and nature of the bacterial population in sediment and water samples are described and preliminary conclusions are presented about the reaction of ground-water bacteria to the subsurface sewage contamination.

- Measurements of the denitrification potential of sediment and water samples are used to show that a zone of bacterially mediated denitrification probably has been established in the aquifer in response to the increased quantities of nitrate and organic carbon in the plume.

The authors thank the many persons who have kindly given time, information, and guidance to the project team. Particular thanks are given to Richard Quadri, Stephen Garabedian, Virginia de Lima, Albert Augustine, John Organek, Peter Haeni, Joseph Newell, and other persons in the Geological Survey who assisted with data collection and project logistics. The authors also gratefully acknowledge Col. Philip McNamara and George Sundstrom of Otis Air National Guard Base for their help and cooperation.

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ABBREVIATIONS AND CONVERSION FACTORS

The inch-pound units used in this report may be converted to metric (International System) units by the following factors.

Multiply inch-pound unit	By	To obtain metric unit
inch (in.)	2.540	centimeter (cm)
foot (ft)	0.3048	meter (m)
foot per second per foot [(ft/s)/ft]	0.3048	meter per second per meter [(m/s)/m]
mile (mi)	1.609	kilometer (km)
square mile (mi^2)	2.590	square kilometer (km^2)
acre	4,047	square meter (m^2)
ounce, fluid (oz)	0.02957	liter (L)
gallon (gal)	3.785	liter (L)
million gallons (Mgal)	3,785	cubic meter (m^3)
billion gallons (Bgal)	3,785,000	cubic meter (m^3)
ounce, avoirdupois (oz)	28.35	gram (g)
inch per year (in/yr)	25.4	millimeter per year (mm/a)
foot per day (ft/d)	0.3048	meter per day (m/d)
foot per mile (ft/mi)	0.1894	meter per kilometer (m/km)
million gallons per day (Mgal/d)	3,785	cubic meters per day (m^3/d)
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)
foot squared per day (ft^2/d)	0.09290	meter squared per day (m^2/d)

Temperature, in degrees Fahrenheit (°F), can be converted to degrees Celsius (°C) as follows:

$$°F = 1.8\ °C + 32$$

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."

Use of brand/firm/trade names in this report is for identification purposes only and does not constitute endorsement by the U.S. Geological Survey.

CHAPTER A

Many sites where ground water has been contaminated by toxic wastes are located on sandy, permeable, water-table aquifers. This introductory paper describes the source and hydrogeologic setting of a plume of sewage-contaminated ground water which is located in a sandy aquifer. Research on how sewage contaminants have moved in the aquifer on Cape Cod will help scientists and engineers to understand the fate of toxic wastes in ground water. This research is described in subsequent chapters of this volume.

DESCRIPTION OF THE HAZARDOUS-WASTE RESEARCH SITE

By Denis R. LeBlanc

ABSTRACT

Disposal of treated sewage since 1936 to a sand and gravel aquifer at Otis Air Base, Cape Cod, Massachusetts, has formed a plume of contaminated ground water that is more than 11,000 feet long. The plume is being studied by the U.S. Geological Survey as part of a nationwide program to understand the fate of toxic wastes in ground water.

More than 8 billion gallons of secondarily treated sewage have been discharged to the aquifer at the Otis Air Base sewage plant since 1936. Disposal is by rapid infiltration through sand beds. The aquifer that receives the treated sewage is composed of 90 to 140 feet of stratified sand and gravel outwash underlain by silty sand and by till. Ground water in the outwash is unconfined and moves southward toward Nantucket Sound at a rate of about 1 foot per day.

INTRODUCTION

Disposal of treated sewage since 1936 through infiltration beds to a sand and gravel aquifer at Otis Air Base, Cape Cod, Massachusetts (fig. 1), has formed a plume of contaminated ground water that is 3,000 feet wide, 75 feet thick, and more than 11,000 feet long. Water in the plume contains elevated concentrations of chloride, sodium, boron, nitrogen, detergents, and other constituents of the treated sewage. The plume was previously mapped and described in a study by the U.S. Geological Survey, in cooperation with the DWPC (Massachusetts Department of Environmental Quality Engineering, Division of Water Pollution Control), to determine the impacts on ground-water quality of land disposal of sewage (LeBlanc, 1982; 1984). The sewage-disposal method used at Otis Air Base is used at more than 350 sites in the United States and has been proposed for use at other sites in Massachusetts.

DESCRIPTION OF STUDY AREA

The Otis Air Base sewage-treatment facility is located 6 miles north of Nantucket Sound on a broad glacial outwash plain of Pleistocene age (fig. 1). The study area includes 20 mi^2 of the outwash plain in the towns of Falmouth, Sandwich, and Mashpee, and is generally bounded by Otis Air Base to the north, the heads of several saltwater bays to the south, Coonamessett Pond to the west, and Johns Pond to the east.

The study area south of Otis Air Base is mostly rural, although many homes have been built since the plume was first mapped in 1978-79. Otis Air Base has been a military reservation since at least 1936. During World War II, the reservation housed as many as 70,000 troops, and between 1948 and 1973 the base was a major installation of the U.S. Air Force. Since 1973, the base has been used by the Massachusetts National Guard and the U.S. Coast Guard.

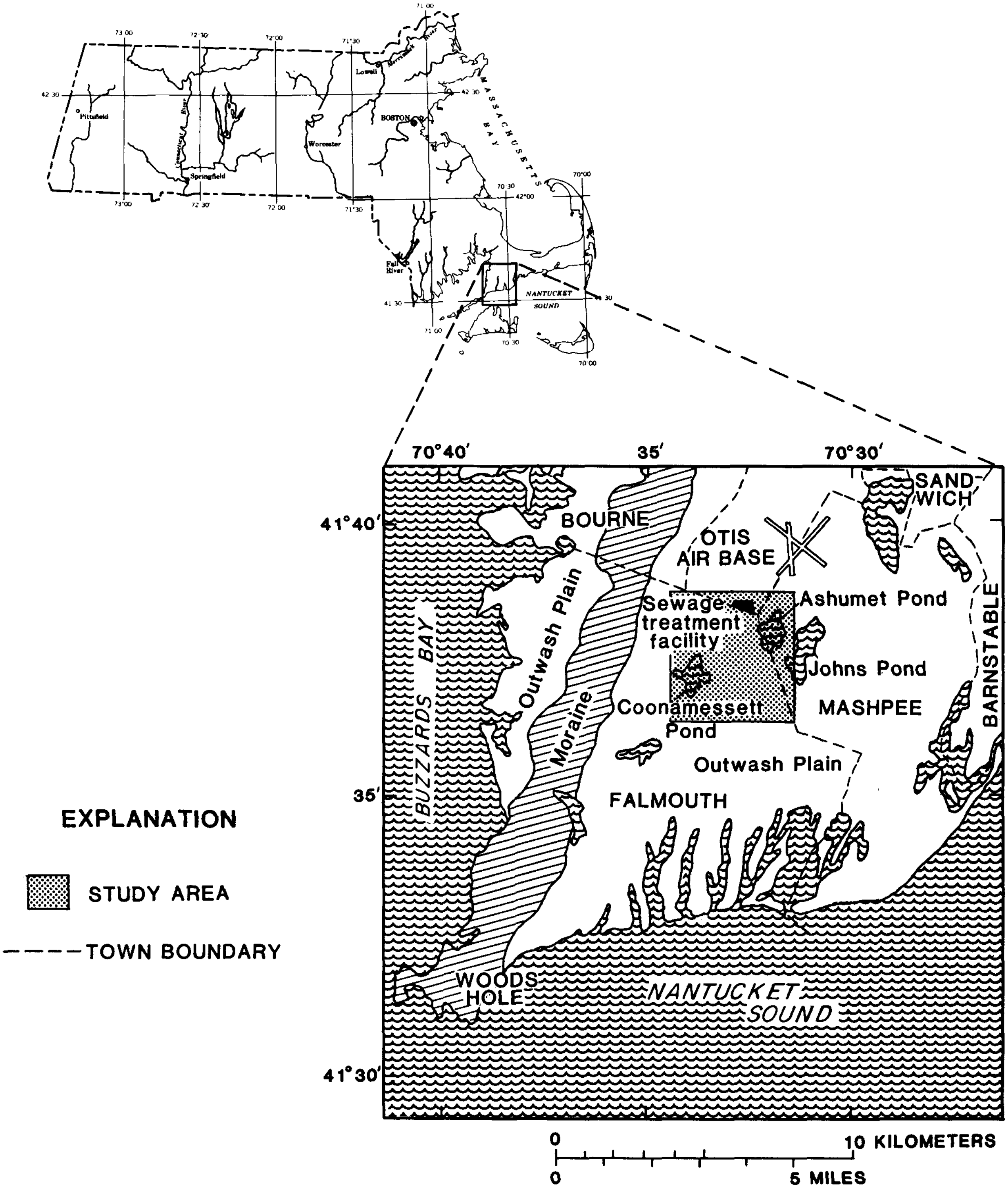


Figure 1.--Study area.

DISPOSAL OF TREATED SEWAGE

The Otis Air Base sewage-treatment facility provides secondary treatment to sewage from the base. A small treatment plant with 4 acres of sand beds served the base between 1936 and 1941. The present treatment plant was constructed in 1941 at the site of the original plant and is designed to treat an average flow of 3 Mgal/d.

The treated sewage can be discharged to 24 one-half acre sand beds (fig. 2) which are rectangular and have flat sandy surfaces. The treated sewage infiltrates into the ground at the beds and percolates to the water table 20 feet below the bed surfaces. The 24 beds have not all been actively used because actual sewage flows during 1936-80 have been much less than design rates, but historical data on bed usage are not available. Since 1977, the treated sewage has been applied exclusively to two beds at the northeast corner of the bed area. Eight other beds are being renovated and will be used for sewage disposal starting in mid-1984 (George Sundstrom, Otis Air National Guard, oral commun., 1983).

The estimated average daily volume of sewage treated at the base from 1936 to 1980 is 0.46 Mgal/d. Average flows have varied from 0.15 Mgal/d prior to World War II to 1.4 Mgal/d during 1941-44 (fig. 3). Records of sewage flows were available for only part of this period and are of questionable accuracy (George Sundstrom, Otis Air National Guard, oral commun., 1983). Sewage flows for the remainder of the period were estimated from records of pumpage for drinking water and from estimates of historical base population.

Analyses of the treated sewage in 1979-80 showed that it contains 150-180 mg/L dissolved solids, about four times the dissolved solids content of uncontaminated ground water in the study area (LeBlanc, 1982, p. 13). Few data are available on the chemical quality of the treated sewage prior to 1974. The estimated average concentrations of boron, chloride, and detergents in the treated sewage, based on analyses of the treated sewage during 1974-80 and analyses reported for earlier periods at similar treatment facilities, are given in table 1. The chemistry of the treated sewage is discussed by Thurman and others (1984).

Table 1.—Estimated average concentrations of boron, chloride, and detergents in the treated sewage and in the uncontaminated ground water

[Concentrations in milligrams per liter unless otherwise noted.]

Constituent	Treated sewage		Uncontaminated ground water
	Period	Concentration	Concentration
Boron (micrograms per liter)	1936-79	500	<50
Chloride	1936-79	30	9
Detergents (MBAS)	1936-47	.0	.0
	1948-64	3	.0
	1965-79	.3	.0

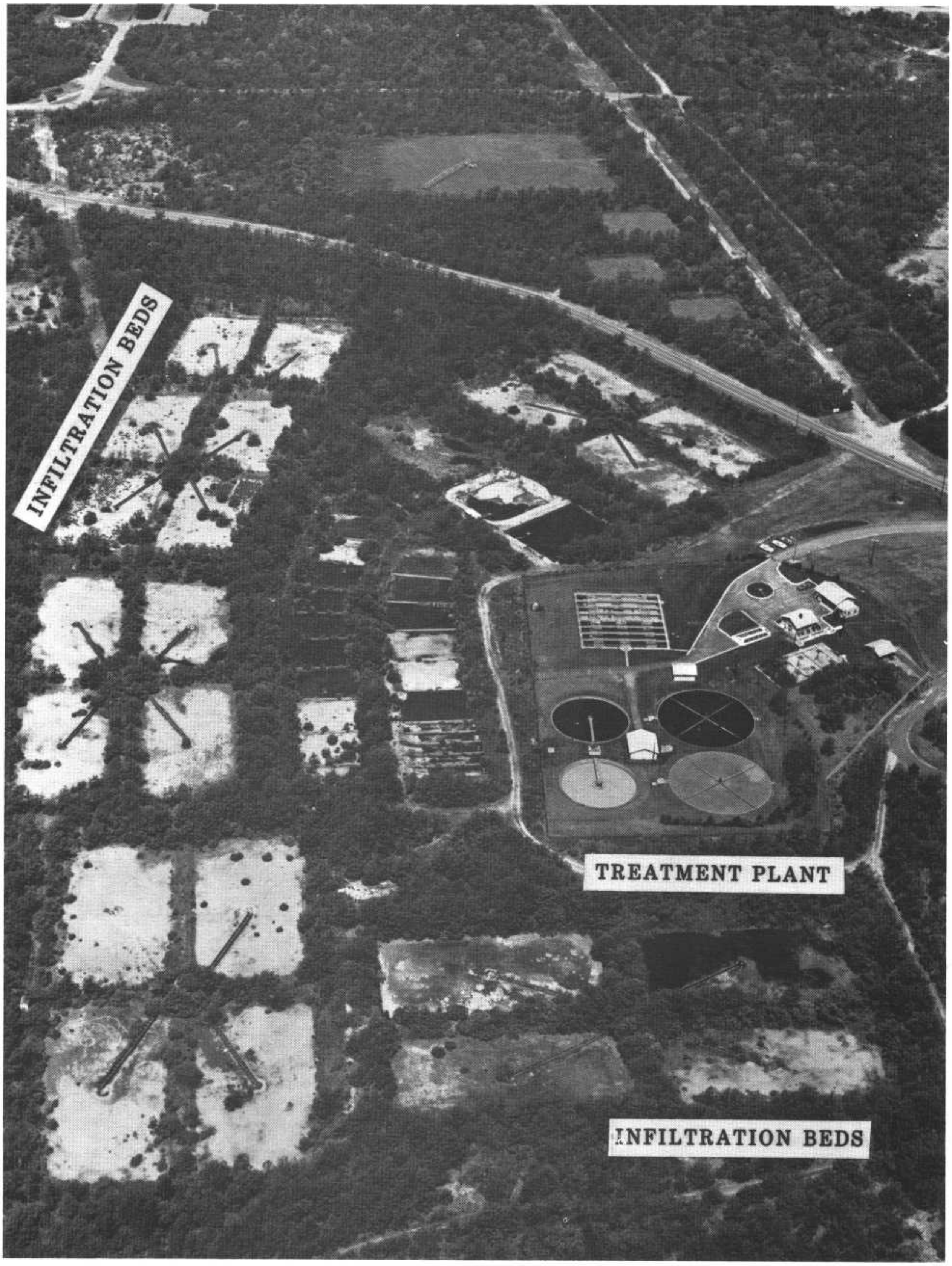


Figure 2.--View west of the Otis Air Base sewage-treatment plant. Photograph courtesy of the U.S. Air National Guard.

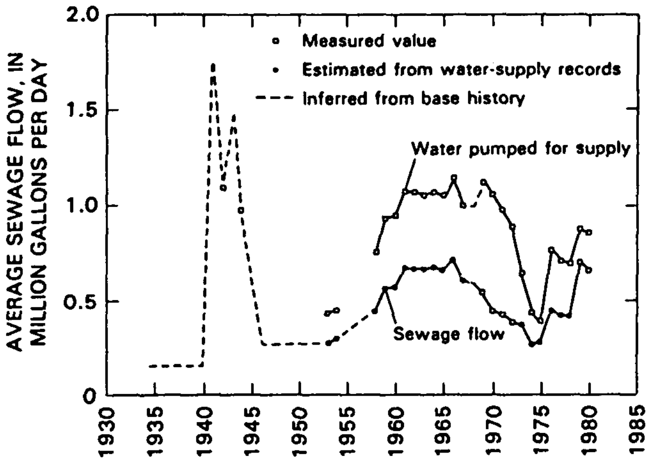


Figure 3.--Volume of sewage treated at the Otis Air Base sewage-treatment plant between 1936 and 1980 (from LeBlanc, 1982).

HYDROLOGIC SYSTEM

Hydrogeology

The aquifer that receives the treated sewage is composed of Pleistocene glacial deposits of sand, gravel, silt, and clay that overlie crystalline bedrock (fig. 4). The top 90 to 140 feet of the aquifer is composed of stratified sand and gravel outwash. These sands and gravels contain only trace quantities of silt and clay sized particles and are composed predominantly of quartz and feldspar. The saturated thickness of the outwash decreases to the south, based on examination of logs of seven boreholes that penetrate into the underlying sediments.

In the northern half of the study area, outwash overlies fine to very fine sand and silt. In the southern half, outwash overlies fine to very fine sand and silt, and dense sandy till. The till contains lenses of silt and clay and lenses of sand and gravel. The bedrock surface slopes gently from west to east across the study area (Oldale, 1969). Bedrock has been mapped as granodiorite, a gray crystalline igneous rock (Oldale and Tuttle, 1964, p. D121).

Hydrology

Ground water in the aquifer is unconfined, and the water table slopes uniformly to the south except where it is distorted by ponds (fig. 5). The water-table contour map was prepared from water levels measured in November 1979. Water levels in November 1979 were near average for the period 1963-76 (Guswa and LeBlanc, 1985) at 10 long-term monitoring sites on Cape Cod.

The hydrograph for well FSW 167 (fig. 6) is typical of water-level fluctuations in the study area. The water table fluctuates 1 to 3 feet each year due to seasonal variations in recharge from precipitation. No long-term rise or decline of average annual water levels has been observed since monitoring began in the study area in 1975. The hydrograph of well A1W 230 (fig. 6), located in a geohydrologically similar outwash deposit 14 miles east of the study area, shows no long-term rise or decline of water levels since 1960. Water levels in the study area correlate well with levels measured in well A1W 230 (Frimpter, 1980, p. 6 and plate 1). Therefore, the water-table map shown in figure 5 represents a steady-flow system.

The only natural source of water to the aquifer is recharge from precipitation. The estimated average annual recharge rate is 21 in/yr (LeBlanc, 1982, p. 10). Recharge occurs over most of the study area. Direct surface runoff is negligible because the sandy soils are very permeable.

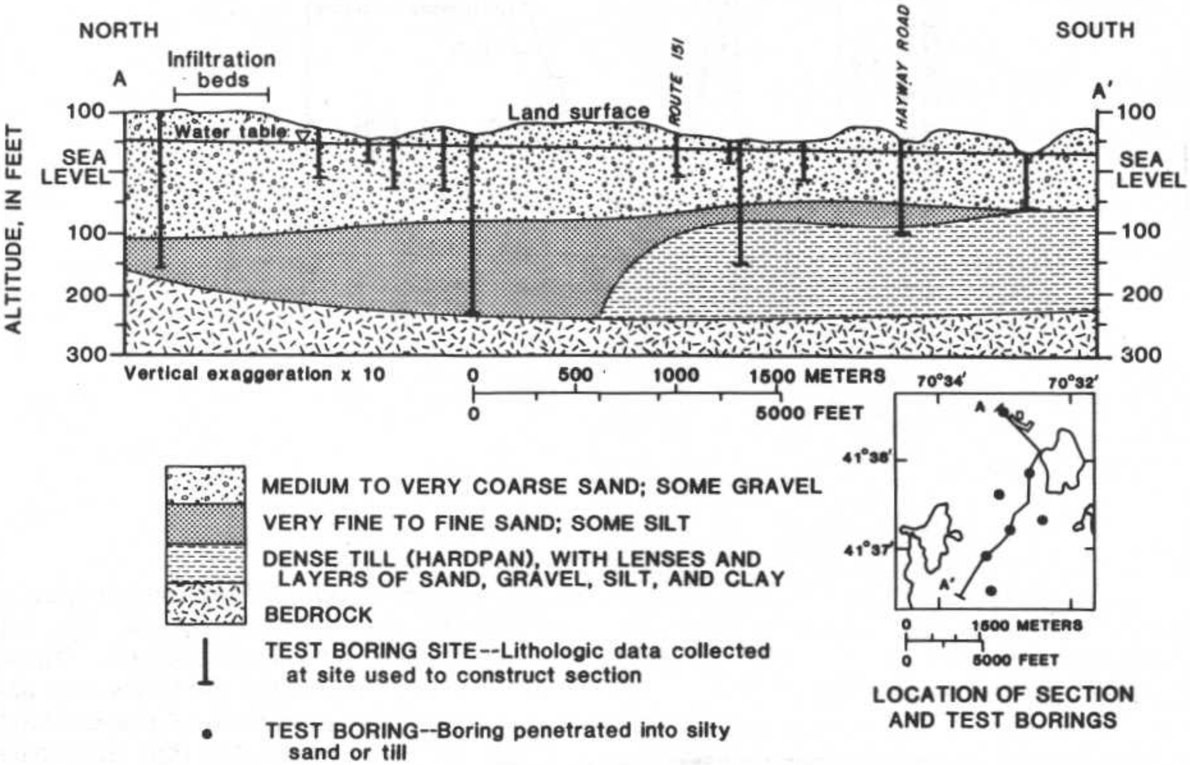
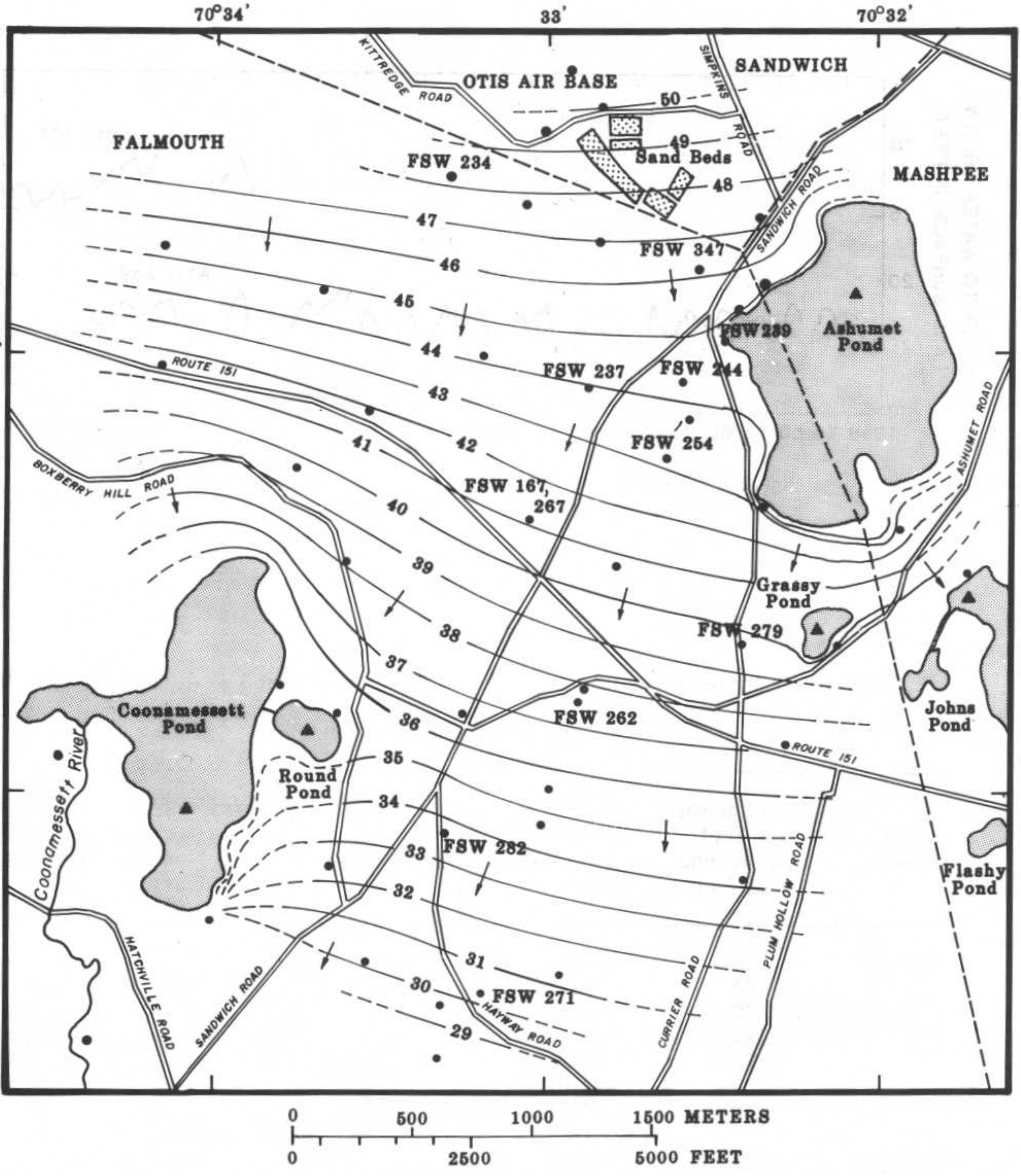


Figure 4.--Geologic section showing hydrogeologic units in the study area (from LeBlanc, 1982).

Ground water in the study area generally flows south as inferred from the water-table map and from the path of the plume. LeBlanc (1982, p. 10) estimated that ground water moves through the sand and gravel at 0.8 to 2.3 ft/d. Ground-water flow is nearly horizontal except near the ponds and presumably near the infiltration beds. Vertical variations in hydraulic head within the outwash are very small (table 2) at the five sites (fig. 5) where water levels were measured in well clusters. The observed vertical differences in head are not significantly larger than the accuracy of the measurements. There may be larger transient vertical head gradients during recharge events, but continuous water-level records at the well clusters are needed to detect these gradients.

Most ground water flowing through the study area discharges to streams, ponds, and wetlands in southern Falmouth and to Nantucket Sound. The net discharge from the aquifer by pumping wells is small because most water is returned to the aquifer through irrigation and septic systems. Water also flows between the aquifer and the three large kettle-hole ponds. Ashumet Pond, which is located 1,700 feet southeast of the infiltration beds (fig. 5), has no surface inlet or outlet. Johns Pond and Coonamessett Pond are drained by streams. Ground-water levels south of the Otis treatment plant are controlled, in part, by the relatively constant water levels along Johns and Coonamessett Ponds and the streams that flow from the ponds.



35 --- WATER-TABLE CONTOUR, NOVEMBER, 1979-- Shows altitude of water table. Contour interval 1 foot. Datum is sea level. Arrows show direction of ground-water movement. Contours dashed where inferred.

FSW 234 • WATER-LEVEL OBSERVATION WELL -- Number is well designation used in tables 2 and 4.

▲ POND AT WHICH WATER LEVEL WAS MEASURED

Figure 5.--Water-table contour map and direction of ground-water flow (from LeBlanc, 1982).

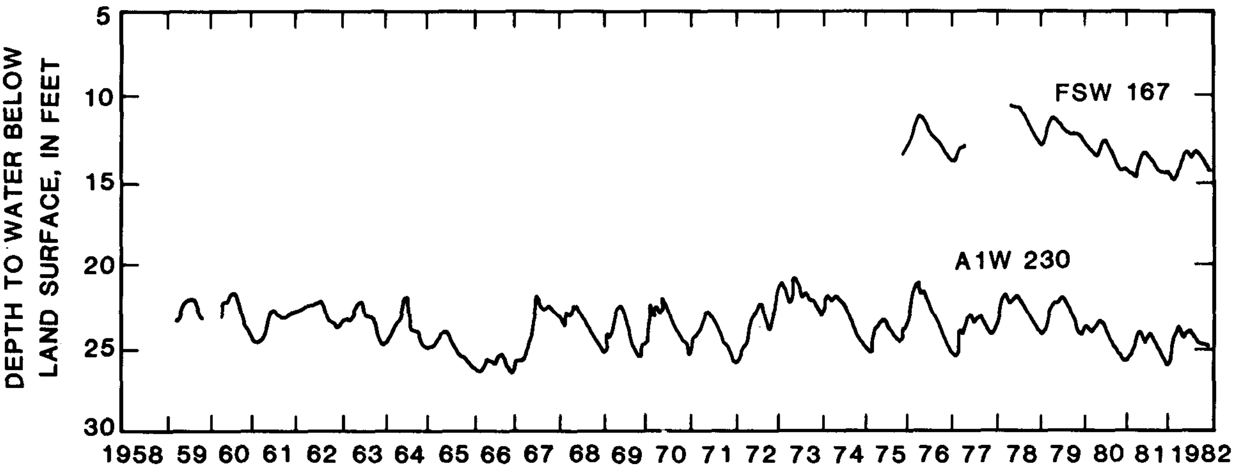


Figure 6.--Hydrograph of observed water levels in wells FSW 167 and A1W 230 for the period 1959-82.

Table 2.—Observed hydraulic head in clusters of observation wells, November 1979

[Well sites are shown in figure 5.]

Well cluster	Well depth, in feet below land surface	Hydraulic head, in feet above sea level	Well cluster	Well depth, in feet below land surface	Hydraulic head, in feet above sea level
FSW 254	26	43.84	FSW 167, 267	55	41.33
	54	43.88		88	41.33
	72	43.89		[1]111	41.30
	107	43.88		[1]136	41.32
	[1]140	43.93		[1]155	41.26
	[1]168	43.91			
	[1]216	43.91	FSW 271	41	30.45
				[1]85	30.45
FSW 262	41	37.69		[1]141	30.49
	69	37.69		[1]165	30.49
	85	37.68			
	[1]109	37.72	FSW 282	49	33.67
	[1]159	37.53		70	33.68
				94	33.70
				[1]123	33.69

[1] Well screened below the bottom of the sand and gravel outwash.

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CHAPTER B

Contaminants in aquifers move primarily by advection with flowing ground water. Consequently, a key to understanding the fate of contaminants in ground water is a good description of the physical transport system. This paper describes the use of a two-dimensional solute-transport model of the plume to determine the geohydrologic factors that affect movement of solutes in the aquifer. This model will be used as a tool to guide data collection and to test and revise hypotheses about contaminant transport developed from these data.

DIGITAL MODELING OF SOLUTE TRANSPORT IN A PLUME OF SEWAGE-CONTAMINATED GROUND WATER

By Denis R. LeBlanc

ABSTRACT

Conservative solute transport of boron in a sewage plume in a sand and gravel aquifer was simulated with a ground-water flow and solute-transport model. The plume was formed by disposal of treated sewage through infiltration beds to an unconsolidated water-table aquifer at Otis Air Base, Cape Cod, Massachusetts, since 1936. The two-dimensional, method-of-characteristics model was used to study the hydrologic and geologic factors that affect movement of contaminants in the aquifer.

Flow was assumed to be independent of solute concentration and was simulated separately from solute transport. During calibration of the steady-state flow model, the best match between observed and computed water levels was obtained with a hydraulic conductivity of about 190 feet per day. Ground-water velocities, which averaged about 1 foot per day, were computed from the flow model results and an estimated effective porosity of 0.35.

The areal location of the simulated boron plume agreed reasonably well with the location of the plume observed in 1978-79 after 40 years of disposal. Differences between the observed and simulated locations of the plume were due, in part, to inaccurate preliminary delineation of the plume from analyses of water samples from wells. A subsequent drilling program verified the predicted location of the eastern boundary of the plume. The two-dimensional model computes average concentrations for the aquifer's entire thickness, whereas the plume occupies only part of the saturated thickness. Consequently, the model was unable to simulate the observed concentrations accurately. In these simulations, as much as 65 percent of the treated sewage discharges to a pond located near the infiltration beds. The beds cover an area that extends from 1,500 to 2,500 feet from the pond, and the rate of mass discharge to the pond is very sensitive to the distance between the pond and the active beds.

INTRODUCTION

The U.S. Geological Survey is studying the geologic, hydrologic, and chemical processes that affect the movement of contaminants in the sewage plume at Otis Air Base[1] as part of a nationwide program to describe the fate of contaminants in the subsurface. Because ground water is the sole source of drinking water on Cape Cod (U.S.

[1] A description of the site, including its geologic and hydrologic setting, is given in chapter A of this volume.

Environmental Protection Agency, 1982), there is concern that land disposal of treated sewage at the base and at other sites may adversely affect ground-water quality on the Cape. An understanding of the processes that affect solute transport is needed to evaluate and predict the movement of contaminants in the plume. The understanding of these processes that will be gained by detailed study at this site will also be useful for evaluation of contamination by similar wastes in other sandy aquifers. In this report, a mathematical model of solute transport in the plume, prepared in cooperation with the Massachusetts Division of Water Pollution Control, is presented as a first step in the quantitative description of these transport processes.

In this study, a mathematical model was used to examine the relationship of the plume to its geologic and hydrologic setting. The objectives of the study were to: (1) Investigate the feasibility of simulating the plume with a mathematical model, (2) develop a predictive tool for describing and evaluating the behavior and movement of solutes in the plume, and (3) examine the geologic and hydrologic processes that affect solute transport in sand and gravel outwash.

The initial description of the plume (LeBlanc, 1982) was used as input for the modeling effort, and a documented computer code (Konikow and Bredehoeft, 1978) was used for the simulations. The study included: (1) Evaluation of hydrologic, geologic, and chemical data for its applicability to modeling; (2) estimation of aquifer properties that affect solute transport; and (3) simulation of transport in the plume with a two-dimensional model.

DESCRIPTION OF THE SEWAGE PLUME

The location and chemical composition of the plume were determined by collection and chemical analysis of water samples from 66 wells in 1978-79. The chemical composition of the plume is described by LeBlanc (1982) and Thurman and others (1984). Three constituents were identified by LeBlanc (1982) as possible tracers for modeling solute transport: Chloride, detergents, and boron.

Chloride generally is a good indicator of the contaminated zone. The conservative behavior of chloride has been described by Hem (1970, p. 172). However, chloride concentrations in the treated sewage are only 3 times greater than concentrations in uncontaminated ground water (table 1). Also, other chloride sources, especially road salts, have locally increased chloride concentrations and obscured the boundaries of the plume.

Detergents clearly delineate the plume because they are absent in uncontaminated ground water. Nonbiodegradable detergents used between 1946 and 1964 move conservatively in ground water and do not degrade chemically in most ground-water environments (Wayman and others, 1965, p. 49-96). The biodegradable detergents used since 1964 are more likely to degrade chemically (Wayman and others, 1965) or to be adsorbed (Freeze and Cherry, 1979, p. 440), but the rate of biodegradation may be insignificant in ground-water environments similar to the plume (Wayman and others, 1965, p. 56). The concentration of detergents in the sewage entering the aquifer has varied with time, however, and the pattern of variations could be only roughly estimated (table 1) from historical detergent concentrations reported for treated sewage from similar facilities.

Boron also is a good indicator of the contaminated zone. Boron concentrations in the treated sewage between 1974 and 1980 were 10 to 50 times greater than boron concentrations in the uncontaminated ground water. The major sources of boron in the sewage are cleaning agents and detergents. The concentrations of boron in the sewage prior to 1974 are unknown, but concentrations of boron in treated sewage from a similar plant (Kardos and Sopper, 1973, p. 150-151) remained constant from 1964 to at least 1973.

Boron probably moves conservatively through the outwash. Because boron forms a weak acid in ground water, it can be sorbed on weakly basic sites such as aluminum-oxide coatings or perhaps clay minerals. However, the sediments may contain few sorption sites for boron because (1) the aquifer contains less than 1 percent clay, (2) quartz and feldspar are the principal minerals, and (3) available sites may be used by other contaminants that exist at much higher concentrations (Thurman and others, 1984). Conservative movement of boron through sand and gravel aquifers was also reported by Kimmel and Braids (1980, p. 20), Koerner and Haws (1979, p. 80), and Bouwer (1973, p. 172).

The ratio of chloride to boron in water samples of the treated sewage and ground water collected in 1978-79 also suggests that boron behaves conservatively. A plot of boron versus chloride concentrations in these samples is shown in figure 7. A theoretical dilution line connects points that represent uncontaminated ground water and the point that represents the treated sewage. Points that represent samples collected from the plume generally fall along the dilution line. This trend suggests that boron concentrations are attenuated primarily by mixing of contaminated and uncontaminated water. The scatter of points along the line is caused partly by measurement inaccuracy. Past variations of boron and chloride concentrations in the treated sewage, and chloride contamination from other sources could also cause deviations from the dilution line. Several analyses plotted in the lower right part of figure 7 contain high chloride concentrations without corresponding high boron concentrations. These samples were collected from wells located south of Route 151 that are affected by road salting (LeBlanc, 1982, p. 18-21).

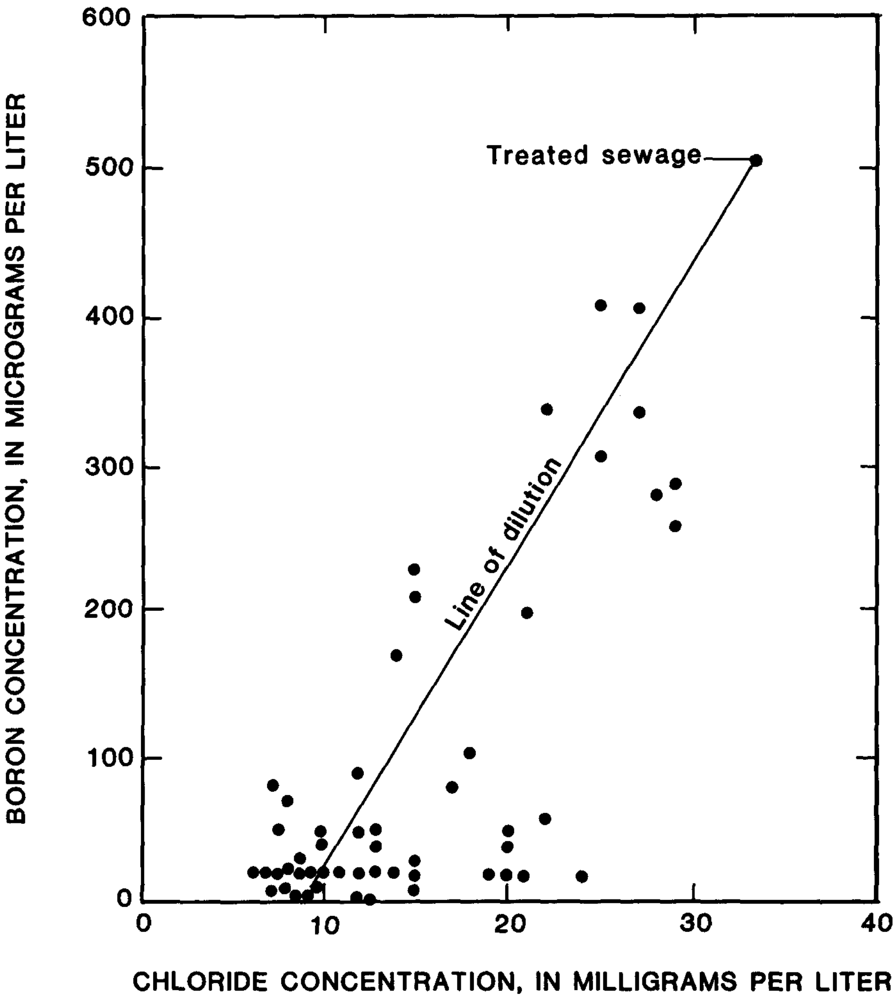


Figure 7.--Concentrations of boron and chloride in samples of ground water and the treated sewage collected in 1978-79.

The observed areal and vertical distributions of boron, chloride, and detergents are shown in figures 8 and 9. The vertical distributions of these contaminants were determined from clusters of wells screened at various depths in the aquifer. The map views were prepared only with data from wells screened at or near the depth of the center of the contaminated zone.

The plume delineated by the elevated concentrations of these contaminants is 3,000 feet wide and more than 11,000 feet long (fig. 8). Contaminants from the disposal site may have moved farther than 11,000 feet downgradient of the infiltration beds, but water samples were not collected beyond this distance in 1978-79. The large extent of the contaminated zone shows that advection of solutes with the flowing ground water is very effective in moving contaminants through the aquifer. The longitudinal axis of the plume is oriented in the direction of ground-water flow shown in figure 5. Spreading and dilution by hydrodynamic dispersion is evident along the toe and sides of the plume, but the contaminant concentrations in the center remain high as far as 8,000 feet downgradient of the sand beds. The amount of spreading could not be determined precisely because the observation wells were spaced several thousand feet apart.

The apparent limits of the plume shown in figure 8 differ for each constituent. This difference is due partly to the relation between source concentrations and concentrations above which the plume was identifiable. As illustrated in table 3, the chloride plume was delineated to a dilution ratio of 1:1.5; whereas, the detergents plume was delineated to a dilution ratio of 1:30. Consequently, the detergents plume seems to be more extensive than the chloride plume.

Table 3.—Lower limit of dilution of treated sewage used to map the plume

[Concentrations in milligrams per liter unless otherwise noted.]

Constituent	Concentration in treated sewage	Lower limit of concentration used to map plume	Lower limit of dilution used to map plume
Chloride	30	20	1.5
Detergents (MBAS)	3	.1	30
Boron (micrograms per liter)	500	100	5

Although the plume is areally extensive, it is only about 75 feet thick (fig. 9) and is contained almost entirely in the sand and gravel outwash. Its bottom boundary generally coincides with the contact between the permeable sand and gravel and the less permeable silty sand and till. A zone of uncontaminated ground water that is 20 to 50 feet thick overlies the plume (fig. 9). The source of water is areal recharge from precipitation (LeBlanc, 1982, p. 28). In the southern part of the study area, it is believed that road salting has increased chloride concentrations in the zone above the plume (fig. 9).

The treated sewage moves downward through the sand and gravel near the infiltration beds, then it moves laterally with little vertical mixing. Vertical movement of the contaminated water near the beds is probably due to vertical head gradients caused by recharge of treated sewage to the aquifer at the disposal site. Because the concentrations of total dissolved solids in the treated sewage (155 to 178 mg/L) and in uncontaminated ground water (39 mg/L) are not significantly different, vertical movement of contaminated ground water probably is not due to density contrasts.

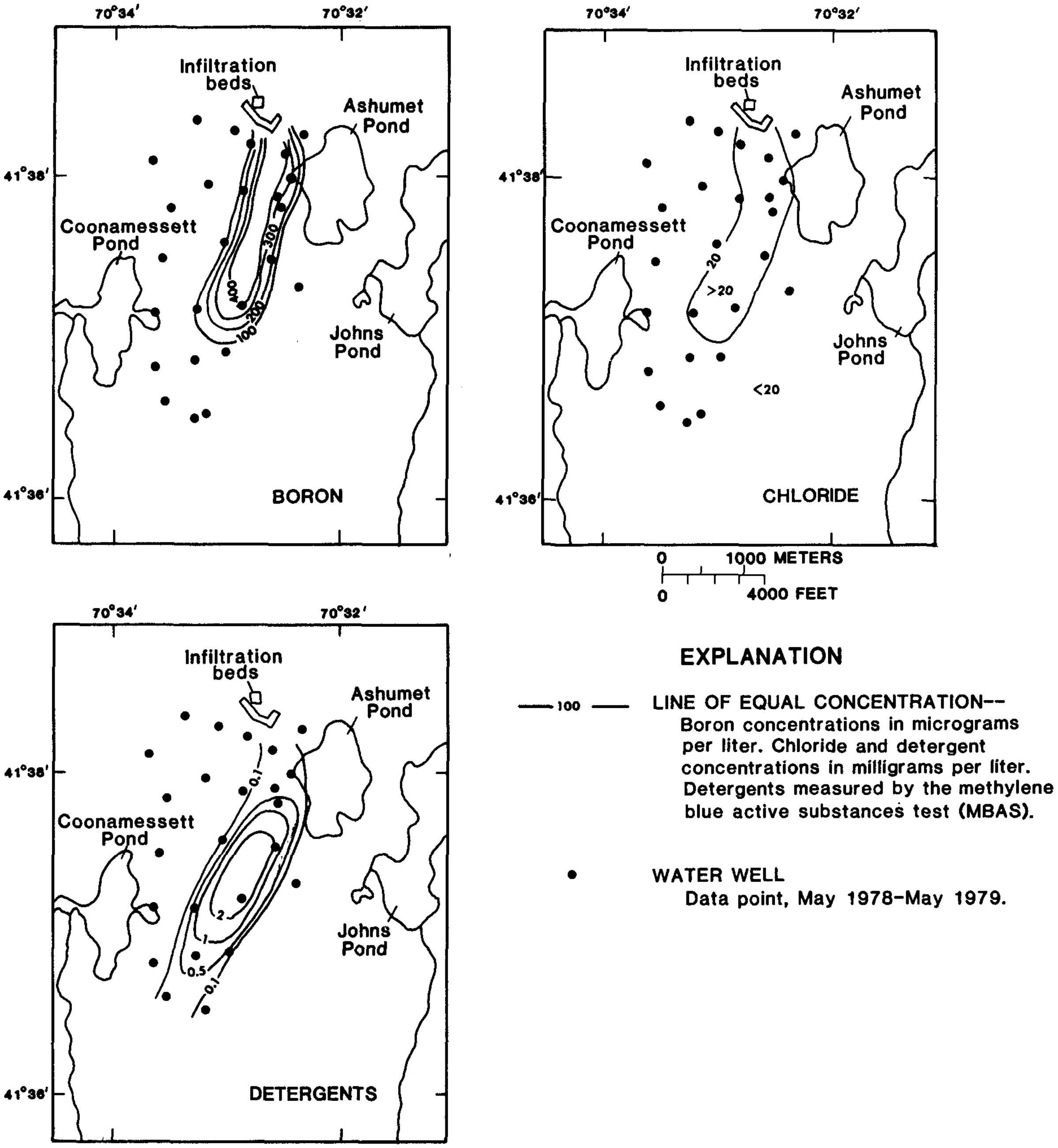


Figure 8.--Observed areal distributions of boron, chloride, and detergents in ground water, May 1978 through May 1979 (from LeBlanc, 1982).

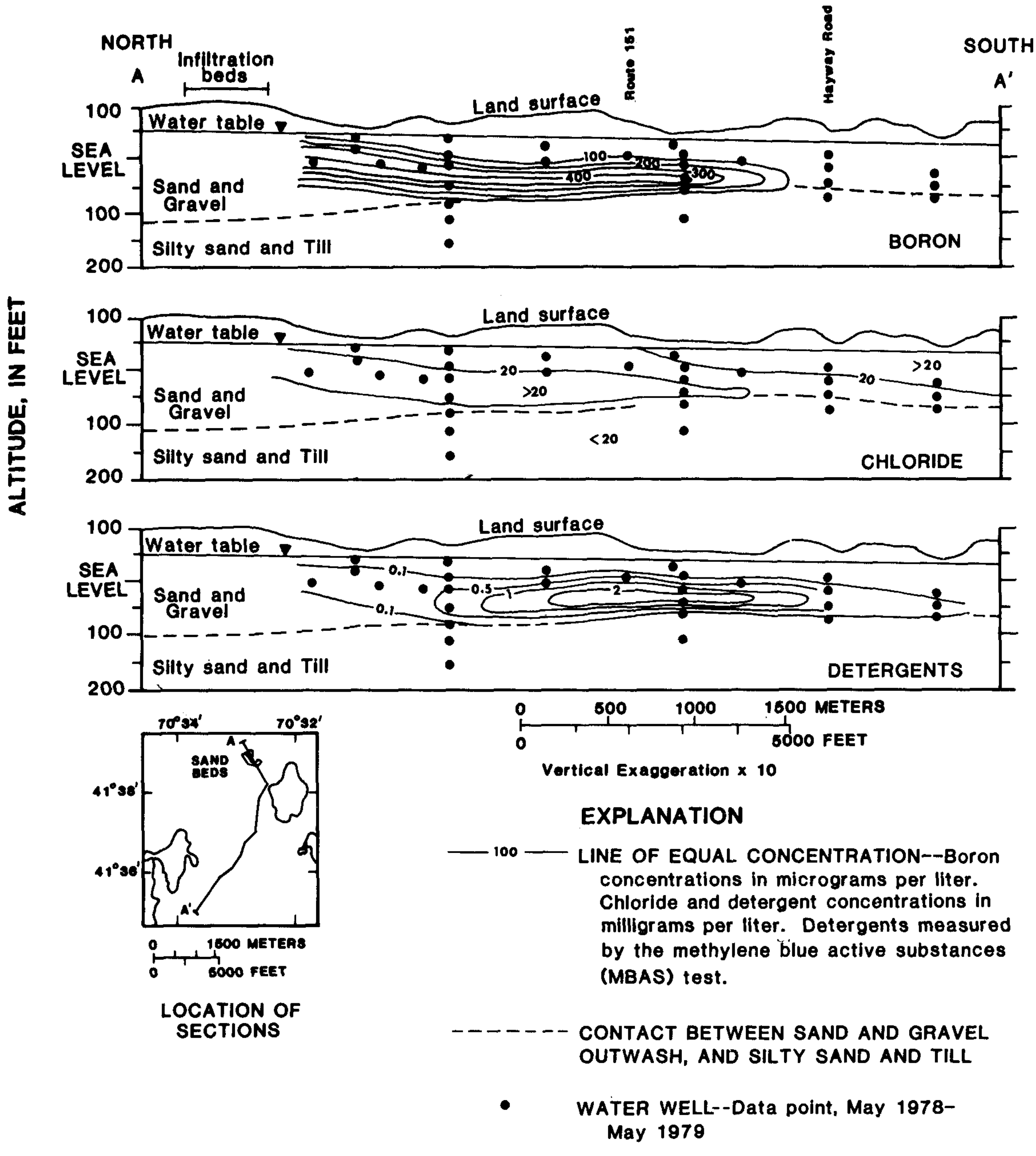


Figure 9.--Observed vertical distributions of boron, chloride, and detergents in ground water, May 1978 through May 1979 (from LeBlanc, 1982).

DIGITAL MODEL OF THE PLUME

A mathematical model was used to simulate solute transport in the aquifer to better understand the hydrologic and geologic influences on the movement of contaminants in the plume. The solute-transport model is a computer program that calculates ground-water flow and water-quality conditions in the aquifer based on the (1) hydraulic properties of the aquifer (transmissivity, porosity), (2) quantity and quality of water entering and leaving the aquifer (recharge and discharge), and (3) potential of the aquifer to disperse contaminants (dispersivity). A digital model was chosen which can simulate the complex aquifer boundaries, spatially variable hydraulic properties, variable flow, and complex hydraulic stresses that characterize the flow and transport system.

Description

The digital model that was selected was developed by Konikow and Bredehoeft (1978) and simulates both flow and transport in a two-dimensional aquifer. Differential equations describing ground-water flow and solute transport are solved numerically with a digital computer code. The flow and transport equations are solved separately because fluid properties are assumed to be independent of concentration.

The flow portion of the model solves the partial differential equation that describes two-dimensional flow through a heterogeneous, anisotropic aquifer (Konikow and Bredehoeft, 1978, p. 2). The equation, written in Cartesian tensor notation, is

$$\frac{\delta}{\delta x_i}\left(T_{ij}\,\frac{\delta h}{\delta x_j}\right) = S\,\frac{\delta h}{\delta t} + W \qquad i,j = 1,2$$

where

T_{ij} = the transmissivity tensor, L^2/T;
h = the hydraulic head, L;
S = the storage coefficient (dimensionless);
t = the time, T;
$W = W(x,y,t)$ = is the volumetric flux per unit area (source term), L/T; and
x_i and x_j = are the Cartesian coordinates, L.

In the model, $W(x,y,t)$ is expressed as

$$W(x,y,t) = Q(x,y,t) - \frac{K_z}{m}(H_s - h)$$

where

Q = the rate of withdrawal or recharge per unit area, L/T;
K_z = the vertical hydraulic conductivity of the streambed or pond bottom, L/T;
m = the thickness of the streambed or pond bottom, L; and
H_s = the hydraulic head in the stream or pond, L.

The area of interest is subdivided into a grid, and a system of finite-difference equations that approximate the differential equation is solved to obtain the hydraulic head at each node in the grid (Konikow and Bredehoeft, 1978, p. 4-5). Darcy's law is then used to calculate the ground-water velocity distribution for input into the solute-transport model (Konikow and Bredehoeft, 1978, p. 2). This expression can be written in Cartesian tensor notation as

$$V_i = -\frac{K_{ij}}{n}\,\frac{\delta h}{\delta x_j} \qquad i,j = 1,2$$

where

V_i = the seepage velocity in the direction of x_i, L/T;
K_{ij} = the hydraulic conductivity tensor, L/T; and
n = the effective porosity (dimensionless).

The solute-transport portion of the model solves the partial differential equation that describes two-dimensional transport of a nonreactive dissolved contaminant in flowing ground water (Konikow and Bredehoeft, 1978, p. 3).

The equation may be written as

$$\frac{\delta(C\ b)}{\delta t} = \frac{\delta}{\delta x_i}\left(b\ D_{ij}\ \frac{\delta C}{\delta x_j}\right) - \frac{\delta}{\delta x_i}\left(bCV_i\right) - \frac{C'W}{n} \qquad i,j = 1,2$$

where

C = the concentration of the dissolved chemical species, M/L^3;

D_{ij} = the coefficient of hydrodynamic dispersion related to aquifer dispersivity and seepage velocity, L^2/T;

b = the saturated thickness of the aquifer, L; and

C' = the concentration of dissolved chemical in a fluid source or sink, M/L^3.

The first term on the right side of this equation represents the change in concentration due to hydrodynamic dispersion. The second term describes the effects of advective transport, while the third term represents a fluid source or sink.

The method of characteristics is used to solve the differential equation. In this method, reference particles are distributed throughout the modeled area and are tracked over time as they move with the flowing ground water to simulate advective transport. Concentrations assigned to each particle are used to calculate the concentrations in each block of the finite-difference grid. Concentrations are adjusted by a finite-difference method to reflect the effects of hydrodynamic dispersion and mixing due to fluid inflow and outflow (Konikow and Bredehoeft, 1978, p. 5-11).

Simplifying Assumptions

Application of the model requires some simplification of the real system. The simulations must be interpreted with consideration of the assumptions made in the modeling procedure. Four major assumptions are discussed below.

1. The aquifer is formed only by the sand and gravel outwash: The underlying silty sand and till are at least 10 to 20 times less permeable than the outwash (table 4), and vertical hydraulic-head gradients across the contact between the outwash and fine-grained sediments are small (table 2). Also, the plume remains mostly in the sand and gravel, although contaminants penetrate 10 to 15 feet into the silty sand near the toe of the plume. Therefore, it is reasonable to assume that the silty sand and till approximate an impermeable bottom boundary to the aquifer.

2. The aquifer can be represented by a single, two-dimensional layer in which vertical variations in hydraulic head and solute concentration are negligible: The assumption of two-dimensional flow is reasonable because ground-water flow in the outwash is nearly horizontal. Vertical flow occurs at the infiltration beds, kettle-hole ponds, and streams, but the hydraulic influence of recharge and discharge at these sites is roughly equivalent to uniform, vertically distributed inflow and outflow because these areas are small or are located at the flow-system boundaries. The assumption of complete vertical mixing of contaminants does not agree with observations that the plume occupies only part of the thickness of the aquifer. The effect on the simulations of this difference between assumed and actual conditions is discussed later in this report.

3. The density and viscosity of the contaminated and uncontaminated ground water are essentially identical, so only hydraulic-head gradients affect the velocity distribution: The difference in total dissolved solids concentration between the treated sewage (155 to 178 mg/L) and uncontaminated ground water (39 mg/L) is small, and ground-water temperatures vary only slightly. Therefore, density differences due to solute-concentration and temperature variations are negligible.

4. Ground-water levels and the velocity distribution do not change with time and represent a steady-flow system: Although water levels fluctuate 1 to 3 feet seasonally, no long-term rise or decline of water levels has been observed since observations began in 1975. The short-term fluctuations are relatively uniform throughout the area and have little effect on the hydraulic gradient. Therefore, the flow model is solved only once to obtain hydraulic heads for a given set of aquifer properties, inflow and outflow rates, and aquifer boundaries.

Table 4.—Estimated hydraulic conductivity of sediment samples of sand and gravel, silty sand, and sandy till

Geohydrologic unit shown in figure 4	Well number (sites shown in figure 5)	Depth of sample below land surface, in feet	Grain-size diameter, in millimeters Percent finer by weight 10 percent	 50 percent	Estimated hydraulic conductivity,[1] in feet per day[2]
Sand and gravel	FSW 347	41	0.32	0.80	254
		42	.51	1.03	562
		72	.29	.54	182
	FSW 234	72	.30	.63	181
	FSW 237	63	.27	.58	155
	FSW 239	83	.16	.36	60
	FSW 244	98	.28	.58	172
	FSW 254	107	.26	.66	141
	FSW 262	87	.07	.34	15
		88	.29	.65	170
	FSW 279	148	.09	.21	20
	FSW 282	79	.15	.66	57
Average			.35	.59	164
Silty sand	FSW 254	138	.014	.125	1.6
		168	.092	.20	19
		198	.078	.16	13
		268	—	.155	5.8
Average			—	.16	10
Till	FSW 262	155	.115	.22	28
		160	.068	.26	13
Average		179	—	.23	14

[1] Assumes ground-water temperature of 10°C.
[2] From Krumbein and Monk, 1943.

Finite-Difference Grid

The limits of the modeled area were selected to include the entire area of the mapped plume and the areas downgradient into which the plume would likely spread (fig. 10). The modeled area, approximately 12.9 mi^2, was subdivided into a finite-difference grid of uniformly spaced rectangles arranged in 40 rows and 36 columns (fig. 11). The grid contains 1,440 rectangles, but only 957 rectangles fall in the active model area. Each rectangle, or block, has dimensions of 500 feet by 750 feet. The block size was chosen so that (1) the width of the plume was spanned by at least 7 blocks and (2) boundaries and sources could be represented in sufficient detail.

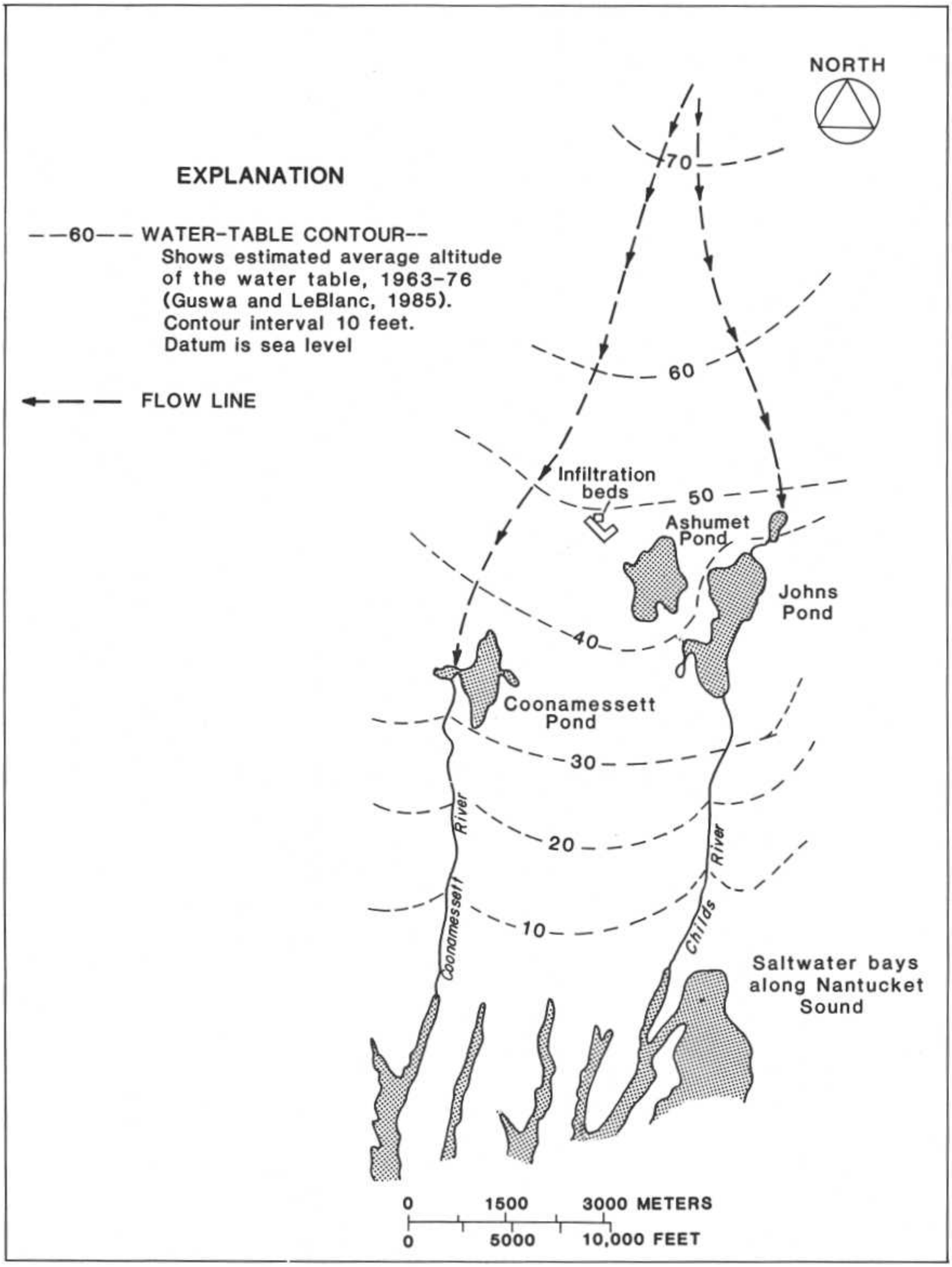
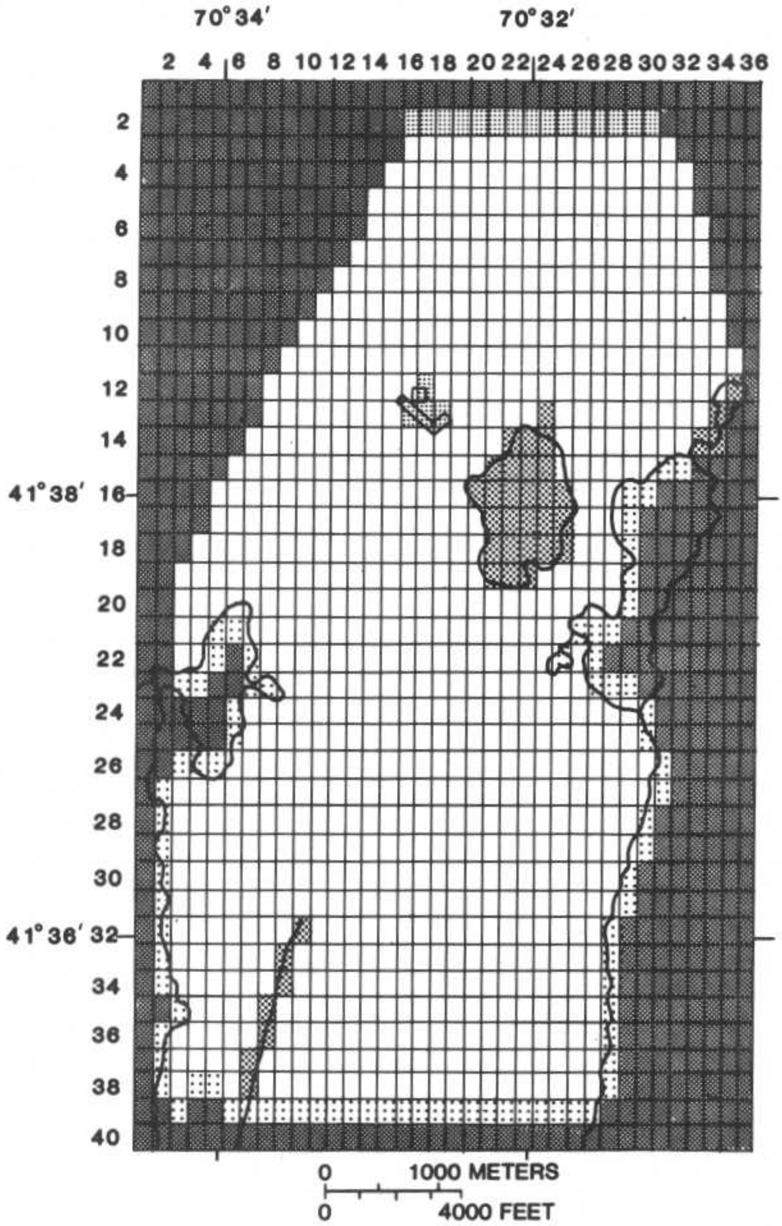


Figure 10.--Major hydrologic features and boundaries of the modeled area.



EXPLANATION

ZERO-TRANSMISSIVITY NODE
Outside active modeled area.

LEAKAGE NODE
Leakance set equal to 0.0001 per second or less.

SPECIFIED-INFLOW NODE
Constant rate of inflow at infiltration beds and northern boundary.

SPECIFIED HEAD NODE
Leakance set equal to 1.0 per second. Constant head during simulations.

Figure 11.--Finite-difference grid used to model the study area.

Parameter Estimation

From the 1978-79 description of the plume and its hydrologic and geologic setting, LeBlanc (1982, p. 28) showed that solutes in the aquifer move primarily by advection with the flowing ground water and are diluted by hydrodynamic dispersion. As a first step toward digital simulation of the movement of contaminants in the plume, estimates of hydraulic conductivity, porosity, and dispersivity were obtained because these factors are fundamental aquifer properties that control advective-dispersive transport of solutes. Estimates of these factors must be reasonably accurate if mathematical modeling is to be successful.

Hydraulic Conductivity

The average hydraulic conductivity of the sediments was estimated from: (1) Flow-net analysis of the regional water-table map, (2) measured hydraulic conductivity at four aquifer-test sites near the study area (Palmer, 1977, p. 21-45) and at three sites in similar sediments elsewhere on Cape Cod (Guswa and LeBlanc, 1985, p. 6), (3) aquifer parameters used in a digital model of regional ground-water flow on Cape Cod (Guswa and LeBlanc, 1985, p. 32-36), and (4) an empirical equation relating grain-size distribution to permeability. The empirical equation (Krumbein and Monk, 1943) was applied to grain-size analyses of 12 split-spoon cores of the sand and gravel, 4 cores of the fine to very fine sand and silt, and 3 cores of the till (table 4).

The estimates of average hydraulic conductivity of the sand and gravel, obtained by all the above methods, ranged from 140 to 220 ft/d. Estimates of hydraulic conductivity from grain-size data for core samples of the sand and gravel varied over an order of magnitude (table 4) because the split-spoon cores sample individual fine-grained and coarse-grained layers that form the stratified outwash. The average hydraulic conductivity for the cores of the outwash was 164 ft/d. Although Mather and others (1942, p. 1153) reported an areal trend in the grain-size distribution of the outwash, an areal trend in hydraulic conductivity was not evident from the estimates shown in table 4. In the digital model described in this report, a hydraulic conductivity of 170 ft/d was initially assigned to the outwash. This value was modified slightly (increased 10 percent) during calibration of the flow model.

Estimates of hydraulic conductivity from grain-size data for the silty sand and till are 10 to 20 times lower than estimates for the sand and gravel. Several cores contained mostly silt and clay, and estimates of hydraulic conductivity of these cores could not be made with the empirical relations. The hydraulic conductivity of these silts and clays may be less than 1 ft/d (Todd, 1980, p. 72).

Porosity

Porosity of the sand and gravel was estimated from (1) measured porosity of the outwash near the sewage-treatment plant and (2) measured porosity of similar outwash on Long Island, New York. The average porosity of samples collected from shallow wells near the sewage-treatment plant was 0.32 (Kerfoot and Ketchum, 1974). The porosities of two core samples of outwash on Long Island were 0.34 and 0.38 (Perlmutter and Lieber, 1970, p. G12). The grain-size distributions of the Long Island samples were very similar to the grain-size distributions for outwash in the study area. Porosities of 0.36 to 0.42 were reported by Morris and Johnson (1967, p. D20-D29) for undisturbed samples of sandy stratified glacial deposits.

From these data, the average porosity of the sand and gravel was estimated to be about 0.35. Although the total pore space may not be available for flow due to dead-end pores and adhesion of water to the sediment grains (Bear, 1979, p. 63), the effective porosity available for flow is essentially equal to total porosity in coarse-grained unconsolidated media (Todd, 1980, p. 27).

Dispersivity

Hydrodynamic dispersion causes the plume to spread and mix with uncontaminated ground water in the direction of flow and, to a lesser extent, perpendicular to flow. It is a function of ground-water velocity and dispersivity, a property of the aquifer (Freeze and Cherry, 1979, p. 390). Initial estimates of dispersivity were obtained for the outwash from values determined for similar aquifers in other areas. These values, summarized in Anderson (1979, p. 127) and Pickens and Grisak (1981, p. 1192), were determined by matching observed plumes with mathematical models by trial-and-error adjustment of dispersivity and other parameters. Based on these estimates, reasonable values for the outwash at Otis Air Base are (1) 40 to 100 feet for longitudinal dispersivity, and (2) 13 to 30 feet for horizontal transverse dispersivity.

Data Requirements

Aquifer properties and stresses must be defined at all active nodes of the finite-difference grid, and boundary conditions must be specified for nodes along the edges of the grid. Initial estimates of properties, stresses, and boundary characteristics described in previous sections of this report were adjusted during the simulation procedure so the simulated system reasonably approximated the observed system. The estimates were adjusted within limits consistent with the accuracy of the data and uncertainty of the estimation procedures. Because the adjustments required were generally small, only the final input data for boundaries, aquifer properties, and inflows and outflows are described in this report.

Aquifer Boundaries

The lateral boundaries to the ground-water-flow system are shown in figure 10, and their representations in the finite-difference model grid are shown in figure 11. The northern boundary, at the 60-foot water-table contour, was specified as a constant-inflow boundary. Extension of the modeled area to the ground-water divide would have greatly increased the size of the finite-difference grid, and reasonable estimates of inflow across the boundary were obtained by several methods.

Coonamessett Pond and the Coonamessett River and Johns Pond and the Childs River (fig. 10) act as drains to the ground-water-flow system along which water levels are relatively constant. These ponds and streams were specified as constant-head boundaries in the model (fig. 11). This is accomplished by representing the boundaries as leakage nodes at which leakance is set to a high value [1.0 (ft/s)/ft]. Leakance is the vertical hydraulic conductivity of the streambed or pond bottom divided by bed thickness. The pond water levels were measured in November 1979 (fig. 5) and the stream water levels were estimated from topographic maps.

Because there are no physical boundaries to flow extending north of the two ponds, no-flow boundaries were delineated by tracing flow lines on the regional water-table map from the northern ends of the ponds to the 60-foot water-table contour (fig. 10). In the model, flow does not cross these lines, although in the real system their positions are not fixed and move in response to changes in ground-water levels.

The southern boundary to the modeled area, along the 10-foot water-table contour (figs. 10 and 11), was specified as a constant-head boundary. Because of the proximity of this boundary to sea-level saltwater bays, water levels in the aquifer remain nearly constant along the 10-foot water-table contour.

Aquifer Properties

Transmissivity, a measure of the rate at which water will flow through the aquifer under a unit hydraulic gradient, was determined from saturated thickness and hydraulic conductivity of the outwash. The transmissivity for the modeled area ranged from 24,300 ft^2/d north of the infiltration beds to 16,800 ft^2/d south of Coonamessett and

Johns Ponds. These values are based on an average hydraulic conductivity of 186 ft/d and a saturated thickness that decreases from 130 feet north of the infiltration beds to 90 feet at the southern boundary of the model. At the kettle-hole ponds, transmissivity was decreased to reflect lower saturated thickness of the outwash beneath the ponds. Although the simulated position of the water table differed slightly from the observed water levels used to calculate saturated thickness, transmissivities were not adjusted during the simulations because the differences between observed and computed saturated thicknesses generally were less than 2 percent.

Effective porosity was set equal to 0.35 everywhere in the modeled area. A longitudinal dispersivity of 40 feet and a transverse dispersivity of 13 feet were used in the model. These values are at the low end of the range of dispersivities reported in other modeling studies.

Inflow and Outflow

The rate of areal recharge from precipitation was estimated by application of the Thornthwaite and Mather (1957) method to climatic data for Falmouth. The original recharge estimate, 21 in/yr, was adjusted downward to 19.8 in/yr during model calibration. Recharge directly to the aquifer was assumed to be zero over Coonamessett, Ashumet, and Johns Ponds. The net gain of water to the ponds from precipitation minus evaporation was accounted for in the model by a net leakage from the ponds to the aquifer. The recharge rate was decreased slightly to 18.9 in/yr at nodes where the water table is within 5 feet of land surface. The total annual evapotranspiration in these low-lying areas where plant roots reach the water table should be higher than in upland areas where soil moisture typically becomes depleted at the end of the summer. However, the Thornthwaite estimates of potential evapotranspiration do not greatly exceed the estimates of actual evapotranspiration on Cape Cod because precipitation is evenly distributed through the year and, in most months, exceeds the evapotranspiration demand. Therefore, significant recharge probably still occurs where the water table is near land surface.

Inflow across the northern boundary of the study area was estimated by two methods. First, an inflow rate of 2.5 ft^3/s was obtained from Darcy's law using estimates of aquifer properties and observed hydraulic gradient at the 60-foot water-table contour. Second, an inflow rate of 2.2 ft^3/s was obtained by multiplying the area between the 60-foot contour and the water-table divide by the estimated areal recharge rate (19.8 in/yr), then decreasing the calculated rate to account for water pumped from a supply well located north of the 60-foot contour and for water that moves down into the fine-grained sediments that underlie the sand and gravel. An average of the two estimates, 2.3 ft^3/s, was evenly distributed along the model's northern boundary.

The rate of recharge of treated sewage at the infiltration beds was set equal to 0.72 ft^3/s, the average rate of sewage inflow at the plant for the period 1936-78. Because the pattern of distribution to the 24 beds is unknown, the recharge was distributed uniformly over four nodes that include all the beds (fig. 11).

The interaction between the ground-water-flow system and Ashumet Pond was simulated by leakage nodes (fig. 11) which lie entirely within the active modeled area. At these nodes, inflow and outflow are simulated by head-dependent flow through a pond-bottom layer. The pond water level used in the model equals the estimated average water level of Ashumet Pond. The vertical hydraulic conductivity and thickness of the bottom sediments of the pond have not been measured. The pond has a sandy bottom near the shore, and marine-reflection profiles indicate that fine-grained sediments may cover the bottom of the center of the pond. On this basis, two zones of leakage nodes represent Ashumet Pond. The outer zone was assigned a leakance of 0.0001 (ft/s)/ft, equivalent to a 2-foot-thick layer with a vertical hydraulic conductivity of 19 ft/d. Leakance for the inner zone was 0.000002 (ft/s)/ft, equivalent to a 10-foot-thick layer with a conductivity of 2 ft/d.

A small stream, known locally as the Backus River, that drains a series of cranberry bogs in the southwest corner of the modeled area (fig. 11) was also simulated with leakage nodes. Leakance for the Backus River nodes was set equal to 0.00001 (ft/s)/ft.

Calibration

The path and shape of the plume are functions of the ground-water velocity distribution, which in turn is partly determined by the distribution of hydraulic head in the aquifer. The flow model computes the head distribution from the specified aquifer properties, boundaries, and inflows and outflows. To be a useful tool for studying the plume, the model should be able to simulate the flow system with reasonable accuracy, given the accuracy of the input data and assumptions made in applying the model. Therefore, the flow model was calibrated by comparing observed water-table altitudes with corresponding calculations by the model. During the calibration process, a "best-fit" between observations and computations was obtained by adjusting the input data within limits based on the precision and accuracy of the data.

The flow model was calibrated to the average water-table conditions shown in figure 5. A comparison of water-table maps drawn from the observed and computed water levels (fig. 12) shows close agreement in most areas. Direct comparison of observed and computed water levels was made at 49 wells. The computed heads were within 1 foot of the observed heads at 31 wells, and within 2 feet of the observed heads at 44 wells.

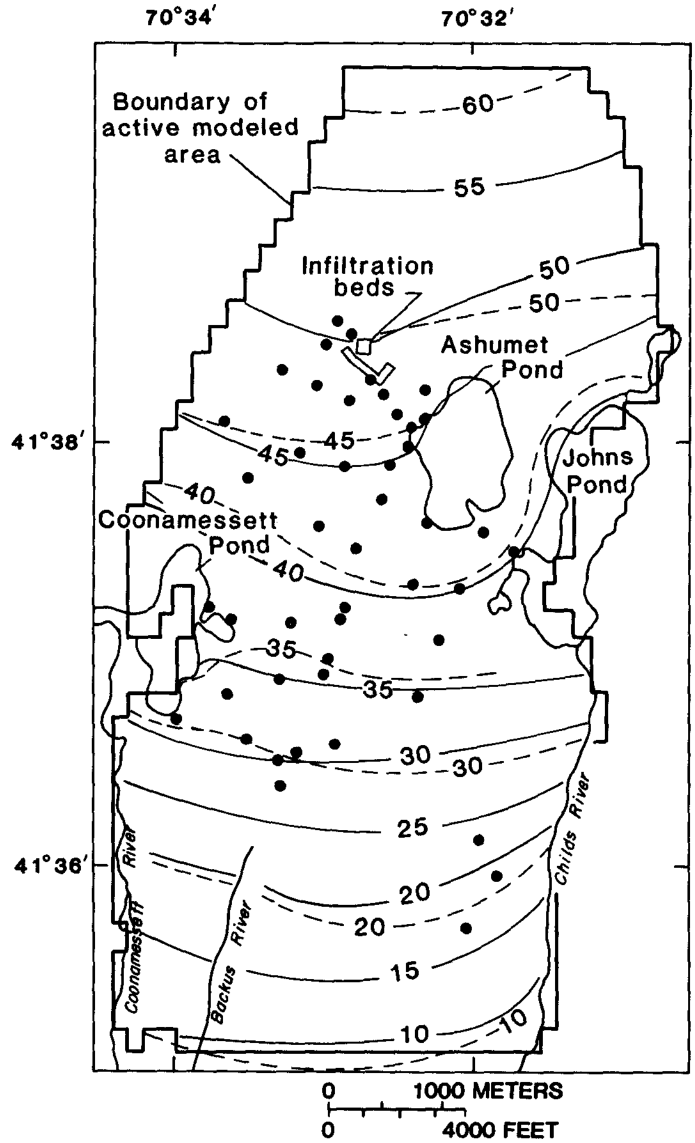
The computed heads are slightly higher than the observed heads north and east of Coonamessett Pond. At the three southernmost observation wells (fig. 12), the computed heads were 4 to 5 feet lower than the observed water levels. Little data on water levels, aquifer properties, and boundary characteristics were available for the southern part of the modeled area, so no adjustments were made to improve the water-level match in this area.

As an additional check on the accuracy of the calibrated flow model, the balance of inflow and outflow to the leakage nodes at Ashumet Pond was computed. Because the pond has no surface inlet or outlet, leakage from the pond to the aquifer should exceed leakage from the aquifer to the pond by the net input to the pond from precipitation. Based on the average annual precipitation, free-water-surface evaporation (Farnsworth and others, 1982, pl. 3), and the surface area of the pond, the net input from precipitation to the pond is estimated to be 0.4 ft^3/s. The net leakage of water from the pond to the aquifer calculated by the model is 0.77 ft^3/s, which is reasonably close to 0.4 ft^3/s.

The balance of inflow and outflow computed by the model is shown in table 5. Note that outflows to the Coonamessett River and the Childs River represent only contributions from the modeled area and cannot be compared directly to field measurements of total flow in the streams. As expected, however, the computed flows were less than the total measured flows in the streams.

During flow-model calibration, the sensitivity of the computed hydraulic heads and leakage-node flows to model input data was tested by varying hydraulic parameters, inflow and outflow specifications, and boundary characteristics. Several conclusions were reached from the sensitivity analysis:

1. The computed heads are sensitive to hydraulic conductivity and to areal recharge rate. If both parameters are changed simultaneously in the same proportion, however, the computed water levels change very little. Although the observed and computed water levels match closely, the hydraulic conductivity is not uniquely determined by the calibration process because the recharge rate is also imprecisely known. Because the ground-water velocity is directly proportional to hydraulic conductivity, better independent estimates of the conductivity are needed.



EXPLANATION

---40--- WATER-TABLE CONTOUR, NOVEMBER 1979--
Shows altitude of water table. Contour interval 5, 10 feet. Datum is sea level.

——40—— COMPUTED WATER-TABLE CONTOUR, FLOW-MODEL CALIBRATION--
Shows altitude of simulated water table. Contour interval 5 feet. Datum is sea level.

• WATER-LEVEL OBSERVATION WELL--
Site where computed and observed water levels compared during calibration.

Figure 12.--Observed and computed water-table map and location of water-level observation wells for flow model calibration.

2. The computed head distribution in the area of the plume is insensitive to reasonable adjustment of the positions of the no-flow boundaries north of Coonamessett Pond and Johns Pond. Although the flow lines are not fixed physical boundaries, their specification in the model has little effect on the transport simulations.

3. The computed heads are sensitive to the assigned location and specified heads of the leakage nodes at Ashumet Pond and along the eastern and western model boundaries. The direction of the hydraulic gradient south of the infiltration beds is determined, in large part, by the heads specified at the leakage nodes. Changes in the relative specified heads along the Coonamessett and Childs Rivers within the accuracy of the topographic maps used to estimate the heads cause shifts in the direction of the hydraulic gradient. Consequently, the simulated path of the plume shifts eastward or westward. The leakage nodes at Ashumet Pond act as specified-head nodes unless the leakance is less than 0.00001 (ft/s)/ft. Specification of lower leakances causes distortion of the flow system around the pond, and solutes tend to spread over a broad front south of the pond and, at very low leakances [0.0000001 (ft/s)/ft], travel to Johns Pond.

Table 5.--Ground-water inflow and outflow rates computed by the calibrated flow model

	Cubic feet per second	Million gallons per day
Inflow		
Across northern model boundary	2.31	1.5
Recharge from precipitation	17.32	11.2
Recharge of treated sewage	.72	.5
Leakage from Ashumet Pond	3.75	2.4
Total inflow	24.10	15.6
Outflow		
Leakage to Ashumet Pond	2.92	1.9
Net leakage to Coonamessett Pond and Coonamessett River	4.60	3.0
Net leakage to Johns Pond and Childs River	11.10	7.2
Net leakage to Backus River	1.04	.7
Across southern model boundary	4.7	2.9
Total outflow	24.13	15.6
Inflow minus outflow (numerical error)	-0.03	

SIMULATION OF SOLUTE TRANSPORT

The solute-transport model computes concentrations of a given solute at each node in the active modeled area at specified time intervals. The ground-water velocities, inflows, and outflows computed by the flow model are used as input for the solute-transport simulations. The distribution of boron in the aquifer after 40 years of transport (1940 through 1979) was simulated first to test the ability of the two-dimensional model to reproduce the plume observed in 1978-79. Boron was selected for simulation because it seems to behave conservatively in the outwash, enters the aquifer in significant quantities only at the infiltration beds, and clearly delineates an extensive plume. The assumptions about the source history and concentrations are discussed below. Additional simulations were then run to test hypotheses about the plume's relationship to the flow system, to examine the movement of contaminants such as detergents that have a variable source history, and to predict future movement of the plume.

Model Application

In the method of characteristics, advective transport of solutes is simulated by tracking reference particles that move with the flowing ground water. The accuracy and precision of the solution generally are improved by using more particles (Konikow and Bredehoeft, 1978, p. 32-35). For all simulations, five particles were initially assigned to each active node. The solutions seemed to be reasonably accurate, and computer costs were reduced by limiting the number of particles that had to be tracked.

Because flow diverged strongly downgradient of Ashumet Pond due to leakage of pond water to the aquifer, nodes in this region tended to become void of particles. Regeneration of the initial distribution of particles, done automatically by the model code (Konikow and Bredehoeft, 1978, p. 18-19), occurred periodically during the simulations to minimize the number of void nodes. The regeneration did not introduce any perturbations or obvious inaccuracies in the solutions.

The numerical solution of the transport equations has several stability criteria that limit the size of the time step (Konikow and Bredehoeft, 1978, p. 11-13). For these simulations, the limiting stability criterion was the rate of inflow from leakage nodes along the downgradient sides of the ponds. To reduce the number of time increments needed to simulate the transport period, specified heads at the critical nodes were decreased 0.5 to 2.0 feet. These small adjustments had little effect on the computed flow system.

To help check the accuracy and precision of the numerical solution, solute-mass balance calculations are performed after specified time increments. The net sum of solute-mass inflow and outflow is compared to the mass stored in the aquifer. The percent difference, which should be small, is one measure of the numerical accuracy of the solution (Konikow and Bredehoeft, 1978, p. 14). For these simulations, the difference generally was less than 7 percent, although it tended to be slightly higher at early time steps. This error was acceptable given the nature of the solution method, the relatively coarse grid spacing, and the small number of particles.

Input Data and Assumptions

The solute, boron, was assumed to be conservative and to be absent in uncontaminated ground water, recharge from precipitation, and inflow from ponds, streams, and upgradient parts of the aquifer. Therefore, the only source of boron in the model was the treated sewage recharged at the infiltration beds. Although part of the plume discharges to Ashumet Pond in the simulations, the boron concentration in leakage from the pond to the aquifer was assumed to be zero. To test the validity of this assumption, the potential increase in solute concentration in the pond due to inflow of contaminated ground water was estimated with a simple reservoir model. The pond was assumed to act as a fully mixed reservoir, and 60 percent of the treated sewage was assumed to discharge to the pond. The solute concentration in the pond as a function of time was computed from ground-water inflow and outflow rates obtained from the calibrated flow model. The computed concentrations in the pond water did not exceed 10 percent of the concentration in the treated sewage because the rate of inflow of contaminated ground water is much less than the total flow through the pond (table 5). This assumption is further substantiated by the relatively low concentrations of boron, 80 µg/L, measured in the pond water during the period 1975-78 (Vaccaro and others, 1979, p. 88-89).

For the initial simulations of the plume as observed in 1978-79, treated sewage recharged the aquifer at a rate of 0.72 ft^3/s at four nodes that represent the location of the infiltration beds. The boron concentration of the source was set at 500 µg/L (table 1). The concentration and rate of recharge were assumed to remain constant during the simulation period. Because sewage flows were small prior to World War II (fig. 3), the simulation period extended from 1940 to 1979 (40 years).

Simulated Plume

The computed concentrations of boron after 9.8, 29.3, and 40 years of simulated disposal to four nodes at the infiltration beds are shown in figure 13. The simulated plume spreads longitudinally and, to a lesser extent, laterally from the source, and first intersects the western side of Ashumet Pond within 6 years. It continues to grow in a southerly direction, and at 40 years extends more than 11,000 feet downgradient of the beds.

The simulated path of the plume agrees reasonably well with the location of the plume observed in 1978-79. The location of the observed plume in figure 13c represents the area in which detergents were detected or concentrations exceeded 20 mg/L chloride or 100 µg/L boron. The simulated plume generally is wider than the observed plume, although the center of the simulated plume in which computed concentrations exceed 50 µg/L boron (10 percent of the source concentration) matches the observed path more closely. The simulated plume also diverges eastward of the observed path and spreads farther southward than the observed plume.

The simulated plume intersects the leakage nodes that represent Ashumet Pond in an area where ground water discharges to the pond. Solutes discharge to the pond with the ground water, and the rate of mass discharge was calculated during the mass-balance computations. This rate, expressed as a fraction of the rate of mass input at the infiltration beds, increases rapidly as the simulated solute first reaches the pond (fig. 14) and gradually stabilizes at about 65 percent of the rate of inflow at the beds.

Differences Between Observed and Computed Paths

The simulated plume of boron generally seems wider and longer and diverges eastward of the observed plume. Several hypotheses were considered to explain these differences.

The greater width of the simulated plume compared to the observed plume (fig. 13c) may be due to several causes: (1) The apparent boundaries in 1978-79, delineated by the concentration contours, may not reflect the maximum extent of the contaminated zone. Identification of the boundaries of the plume from chemical analyses of ground water becomes uncertain as concentrations are attenuated by dilution near the boundaries (table 3) or as concentrations approach the detection limits of the laboratory analytical methods. The digital model, however, has a high apparent resolution of contaminated versus uncontaminated water. (2) The simulated flow system is only an approximation of the real flow system and may predict greater divergence of flow at the infiltration beds and toward the Coonamessett and Childs Rivers than occurs in the real

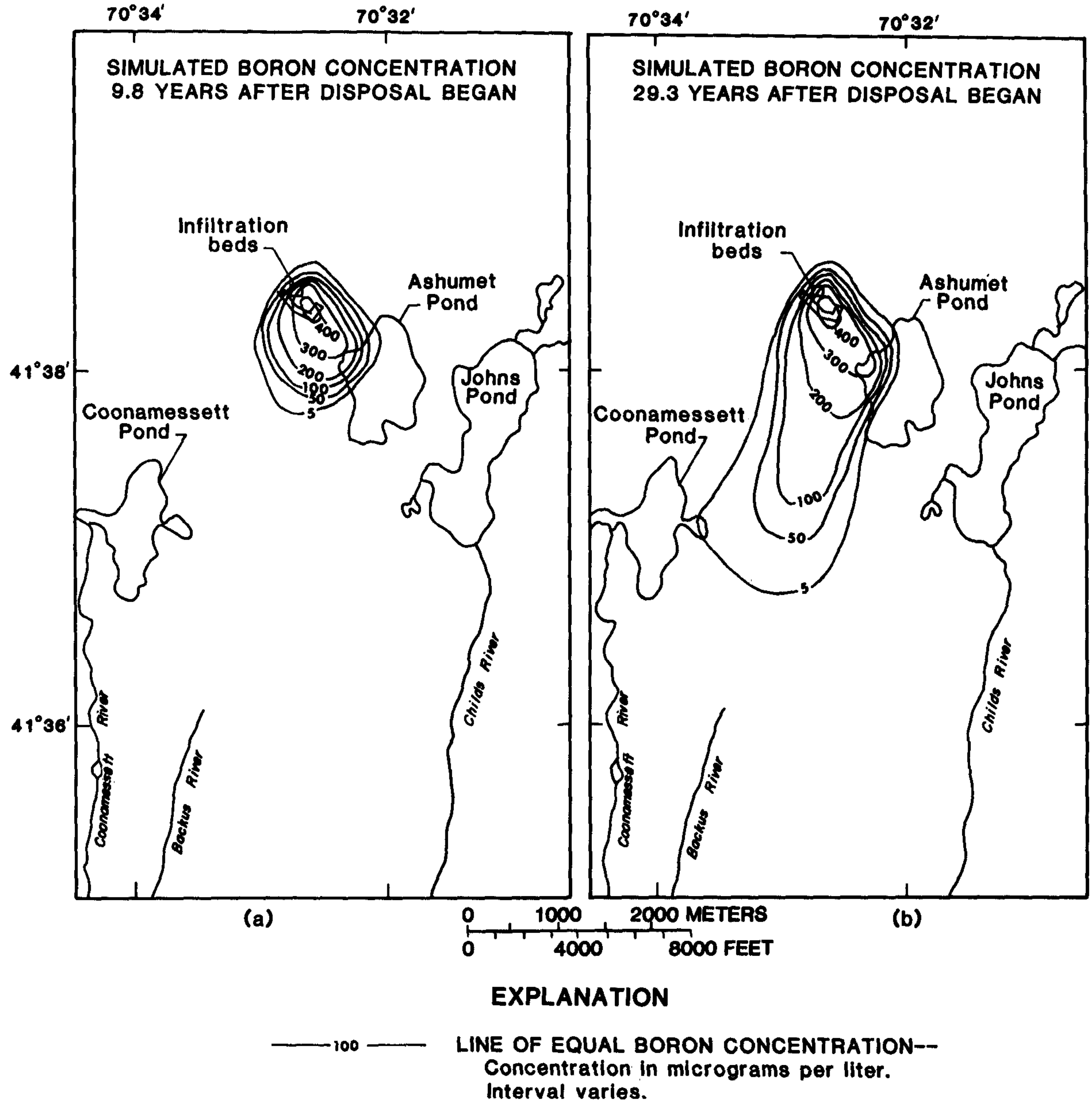
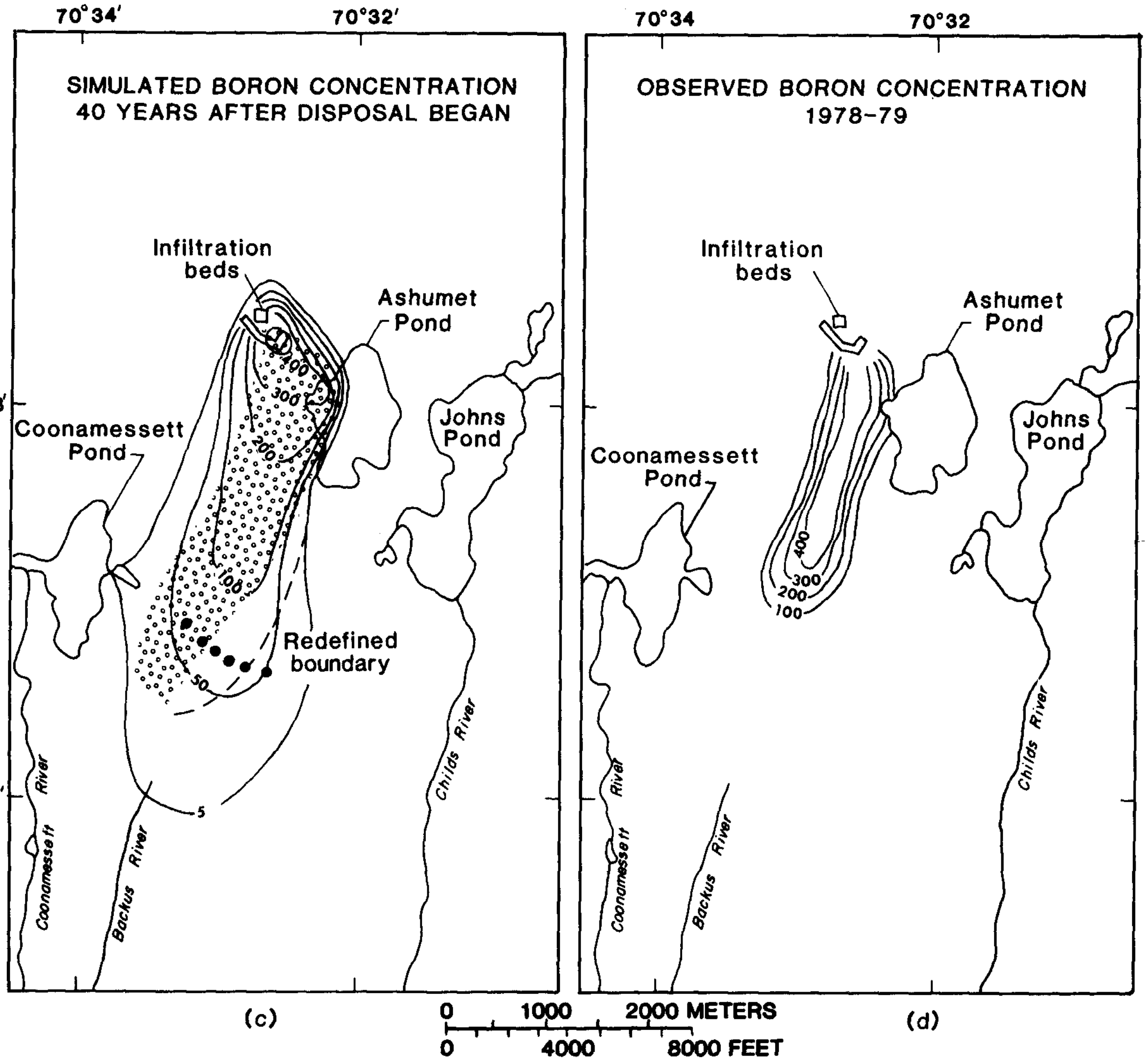


Figure 13.--Computed boron concentrations after (a) 9.8 years, (b) 29.3 years, and (c) 40 years of disposal of treated sewage containing 500 µg/L boron, and (d) observed areal distribution of boron in 1978-79. Assumes that all sand infiltration beds (four nodes) were used for disposal.



EXPLANATION

——100—— LINE OF EQUAL BORON CONCENTRATION-- Concentrations in micrograms per liter. Interval varies.

— — — — — EASTERN BOUNDARY OF PLUME REDEFINED BY NOVEMBER 1983 TEST-WELL DRILLING

AREA OF PLUME DELINEATED IN 1978-79 -- Denoted by detection of detergents, or concentrations exceeding 20 milligrams per liter chloride or 100 micrograms per liter boron.

● OBSERVATION WELL-- Drilled in November 1983 to delineate plume boundaries.

Figure 13.--(continued).

system. (3) The dispersivity used in the model may be larger than the actual dispersivity and may predict more spreading of solutes by hydrodynamic dispersion than occurs in the aquifer. (4) The simulated spreading may be due in part to model error, especially numerical dispersion. Effects of dispersion are discussed in a later section of this report.

The apparent eastward divergence of the simulated plume from the observed plume may be due to several causes: (1) The model may simulate too much discharge to the Childs River and too little discharge to the Backus and Coonamessett Rivers. The computed heads are sensitive to inaccurate specification of heads along these boundaries, and an eastward bias in the water-table gradient may have been introduced in the model. Computed water levels at three wells in the southeastern corner of the modeled area are 4-5 feet lower than observed water levels. (2) Hydraulic conductivity may be laterally anisotropic or heterogeneous due to areal trends in the grain-size distributions in the outwash. Such variations could not be detected from the field data. Several hypothetical simulations were run in which hydraulic conductivity was assumed to decrease from 210 ft/d in the northwest corner of the modeled area to 150 ft/d in the southeast corner of the modeled area. Such a pattern would be consistent with observations by Mather and others (1942, p. 1153) that the outwash tends to become finer from northwest to southeast. However, these variations resulted in little change in computed hydraulic heads and did not affect the simulated path of the plume. (3) The apparent eastward divergence of the simulated plume may also be due partly to inaccurate field delineation of the plume along its eastern boundary. The wells that defined the eastern boundary in 1978-79 were widely spaced. This third hypothesis was tested by additional data collection in 1983. Ground electrical conductivity measurements, made in June 1983 with an electromagnetic induction device, suggested that the contaminated zone extends farther eastward than earlier thought (Gary Olhoeft, U.S. Geological Survey, oral commun., 1984). Also, additional wells were drilled in November 1983 at the sites shown in figure 13c. Water samples were collected and analyzed during drilling. Specific conductances greater than 190 μmhos/cm and the presence of significant foaming at all but the easternmost drilling site show that the plume's eastern boundary lies 1,300 feet east of the original delineation and agrees more closely with the simulated plume (fig. 13c).

The length of the simulated plume generally agrees with the length of the zone in which detergents were detected or concentrations exceeded 20 mg/L chloride or 100 μg/L boron (fig. 13c). Additional chemical analyses of water samples collected in 1983 (Thurman and others, 1984) confirm that contaminants from the disposal site are present in wells 11,000 feet from the infiltration beds. The observed boron plume seems to be shorter than the simulated boron plume, however (fig. 13d). This apparent difference may be due to several causes: (1) The apparent boundaries of the observed plume in 1978-79, shown as the 100 μg/L concentration contour in figure 13d, may not reflect the maximum extent of boron movement in the aquifer. (2) Boron may not have been introduced into the treated sewage at present concentrations until some time after 1940. Data are not available to evaluate this hypothesis, but the association of boron with cleaners and detergents suggests that such a source variation may have occurred.

Two-Dimensional Simulation of a Three-Dimensional Plume

The computed concentrations in the center of the simulated plume decrease gradually from the infiltration beds downgradient to the toe (fig. 13c). In contrast, the observed concentrations of boron and other constituents, such as chloride, remain relatively unchanged in the center of the plume for 8,000 feet (fig. 13d). The inability of the model to simulate the observed concentrations, despite the good match between the observed and computed paths, is mostly due to the inherent limitations of the two-dimensional approach.

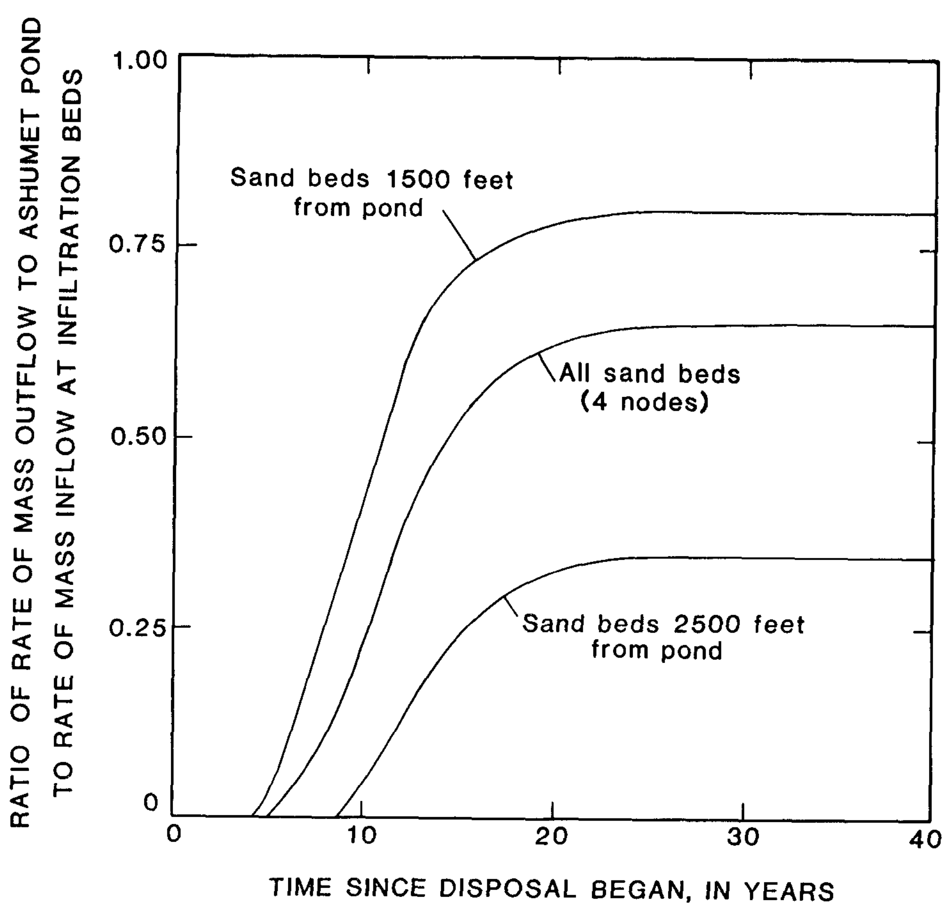


Figure 14.--Portion of total solute mass applied to the infiltration beds that discharges to Ashumet Pond in the simulations.

Typical observed and calculated vertical profiles of concentration through the plume are shown in figure 15. In the field, the contaminated ground water occupies only part of the aquifer thickness and is overlain by uncontaminated water from precipitation. Although some mixing occurs in a thin zone between contaminated and uncontaminated water, the two remain essentially separate. In the two-dimensional model, however, vertical variations of concentration are assumed to be negligible. The solute is evenly distributed through the thickness of the aquifer (fig. 15). Rather than overlying the plume, inflow from areal recharge or pond leakage immediately mixes through the full thickness of the aquifer in the model. Therefore, in the simulated plume, concentrations are diluted by recharge from precipitation and decrease away from the source.

An attempt was made to calculate equivalent vertically averaged concentrations from the field data for comparison with the simulated concentrations. In practice, however, concentration profiles and aquifer thickness were well defined at only a few observation sites, and reasonable estimates could not be obtained at a sufficient number of sites.