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U.S. Geological Survey
Open-File Report 95-381

Prepared in cooperation with the
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By DONALD A. WALTER, BRIGID A. REA, KENNETH G. STOLLENWERK, and JENNIFER SAVOIE

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U.S. DEPARTMENT OF THE INTERIOR
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CONVERSION FACTORS, VERTICAL DATUM, AND WATER-QUALITY INFORMATION

Multiply	by	To obtain
acre	4,047	square meter
cubic foot per day (ft^3/d)	0.02832	cubic meter per day
foot (ft)	0.3048	meter
foot per day (ft/d)	0.3048	meter per day
gallon per minute (gal/min)	0.06309	liter per second
gallon per year	0.003785	cubic meter per day
inch (in.)	2.54	centimeter
mile (mi)	1.609	kilometer

Air temperature is given in degrees Fahrenheit (˚F), which can be converted to degrees Celsius (˚C)by the following equation:

˚C = (˚F-32)/1.8.

Vertical Datum

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

Water-Quality Information

Chemical concentration is given in units of milligrams per liter (mg/L) or micrograms per liter (µg/L). Milligrams and micrograms per liter are units expressing the mass of the solute per unit volume (liter) of water. One thousand micrograms per liter is equivalent to 1 milligram per liter. Micrograms per liter is equivalent to "parts per billion." Milligrams per liter is equivalent to "parts per million." Chemical concentration is also given in units of milligrams per kilograms (mg/kg) or milligrams per gram (mg/g). One milligram per kilogram is equivalent to 1 microgram per gram (µg/g). Milligram per kilogram is equivalent to "parts per million."

Geochemical and Hydrologic Controls on Phosphorus Transport in a Sewage-Contaminated Sand and Gravel Aquifer Near Ashumet Pond, Cape Cod, Massachusetts

By Donald A. Walter, Brigid A. Rea, Kenneth G. Stollenwerk, *and* Jennifer Savoie

Abstract

The disposal of secondarily treated sewage onto rapid infiltration sand beds at the Massachusetts Military Reservation, Cape Cod, Massachusetts, has created a plume of sewage-contaminated ground water in the underlying sand and gravel aquifer; a part of the sewage plume that contains dissolved phosphorus extends about 2,500 feet downgradient of the sewage-disposal beds. A part of the plume that contains nearly 2 milligrams per liter of phosphorus currently (1993) discharges into Ashumet Pond along about 700 feet of shoreline. The sewage plume discharges from about 59 to about 76 kilograms of phosphorus per year into the pond. Hydraulic-head measurements indicate that the north end of Ashumet Pond is a ground-water sink and an increased component of ground-water discharge and phosphorus flux into the pond occurs at higher water levels. Phosphorus was mobile in ground water in two distinct geochemical environments—an anoxic zone that contains no dissolved oxygen and as much as 25 milligrams per liter of dissolved iron, and a more areally extensive suboxic zone that contains little or no iron, low but detectable dissolved oxygen, and as much as 12 milligrams per liter of dissolved manganese. Dissolved phosphorus is mobile in the suboxic geochemical environment because continued phosphorus loading has filled available sorption sites in the aquifer.

Continued disposal of sewage since 1936 has created a large reservoir of sorbed phosphorus that is much greater than the mass of dissolved phosphorus in the ground water; the average ratio of sorbed to dissolved phosphorus in the anoxic and suboxic parts of the sewage plume were 31:1 and 155:1, respectively. Column experiments indicate that phosphorus in the anoxic core of the plume containing dissolved iron may be immobilized within 17 years by sorption and coprecipitation with iron oxyhydroxides following the cessation of sewage disposal and the introduction of uncontaminated oxygenated ground water into the aquifer. Residual oxygen demand associated with sorbed organic compounds and ammonia could retard the movement of oxygenated water into the aquifer. Sorbed phosphorus in the suboxic part of the aquifer will continue to desorb into the ground water and will remain mobile for perhaps hundreds of years. Also, the introduction of uncontaminated water into the aquifer may cause dissolved-phosphorus concentrations in the suboxic zone of the aquifer to increase sharply and remain higher than precessation levels for many years due to the desorption of loosely bound phosphorus. Data from three sampling sites, located along the eastern and western boundaries of the sewage plume and downgradient of abandoned sewage-disposal beds, indicate that ground-water mixing and phosphorus desorption may already be occurring in the aquifer in response to the introduction of uncontaminated recharge water into previously contaminated parts of the aquifer.

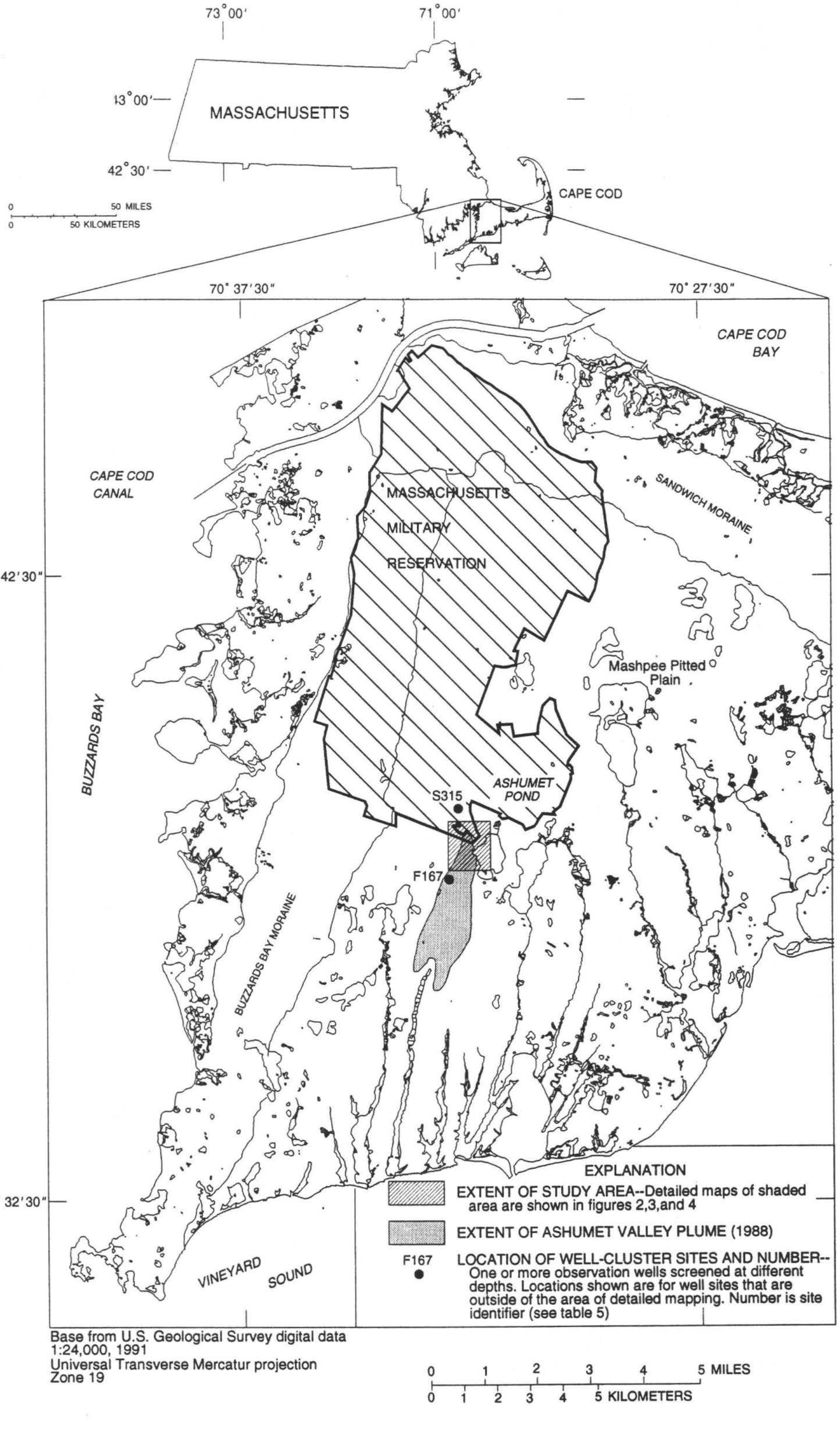
INTRODUCTION

The use of rapid-infiltration sand beds as a means of sewage disposal has been the cause of ground-water contamination in many localities (Bouwer, 1973; Aulenbach and Tofflemire, 1975; Koerner and Haws, 1979; Scorca, 1990). Since 1936, secondarily treated sewage has been discharged onto 12 acres of rapid-infiltration sand beds at the Massachusetts Military Reservation (MMR), Cape Cod, Massachusetts; the MMR is located on a broad, glacial outwash plain on western Cape Cod that is bounded to the north and east by morainal ridges and to the south by Vineyard Sound (fig. 1). The disposal of treated sewage on the military reservation has created a plume of sewage-contaminated ground water, known as the Ashumet Valley sewage plume, in the underlying sand and gravel aquifer (fig. 1). LeBlanc (1984) found that the sewage plume, as defined by conservative constituents such as specific conductance and chloride, was nearly 11,000 ft long and 3,000 ft wide. Phosphorus concentrations are high in a part of the sewage plume just downgradient of the sewage-disposal beds. Phosphorus transport in ground water is retarded by the tendency of phosphorus to sorb onto sediment surfaces, particularly the surfaces of iron and manganese oxyhydroxides, and to coprecipitate with metal oxides (Russell, 1973; Goldberg and Sposito, 1984a, 1984b; Hem, 1992). Despite these adsorption properties, phosphorus can be transported in ground water for considerable distances (Atkinson, 1974; Lund and others, 1976; Beek and others, 1977a, 1977b). LeBlanc (1984) found that the leading edge of phosphorus-contaminated ground water, as defined by phosphorus concentrations greater than 0.05 mg/L, had advanced about 2,500 ft downgradient of the sewage-disposal beds. Two different geochemical environments are found within the phosphorus-containing part of the plume: an anoxic zone that has no dissolved oxygen and has dissolved iron and a suboxic zone that has low but detectable dissolved oxygen and no dissolved iron.

Phosphorus-contaminated ground water discharges along part of the northwestern shore of Ashumet Pond. There is concern that discharge of phosphorus into the pond, which is currently considered mesotrophic, will increase nutrient concentrations and cause the pond to become eutrophic. Eutrophication is believed to be the cause of at least two fish kills in the pond in July 1985 and June 1986 (E.C. Jordan Co., 1988; K.V. Associates, 1991). The National Guard Bureau (NGB) is considering possible remediation strategies to minimize phosphorus loading to Ashumet Pond. At present, little data are available on the distribution of phosphorus in the aquifer and the geochemical and hydrogeologic factors affecting the transport of phosphorus to the pond. Sewage disposal is scheduled to end on the military reservation in December 1995. The fate of phosphorus sorbed to aquifer sediments following the introduction of uncontaminated water into the aquifer is of particular concern to the local community; the possible desorption and remobilization of sorbed phosphorus may adversely affect the ecology of Ashumet Pond for many years.

A 1-year cooperative study between the U.S. Geological Survey (USGS) and the NGB was started in June 1993 to develop a conceptual understanding of the hydrogeologic and geochemical factors affecting the transport of phosphorus from the sewage-disposal beds to Ashumet Pond. The primary objectives of the study were to delineate the part of the Ashumet Valley sewage plume containing phosphorus, to describe ground-water-flow patterns near Ashumet Pond, to determine the geochemical processes that affect the mobility of phosphorus in the sand and gravel aquifer, and to determine how phosphorus mobility may change after sewage disposal ceases. The location of the study area on western Cape Cod and the location of observation wells outside the study area are shown in figure 1.

This report discusses ground-water quality, the geochemistry of phosphorus on aquifer sediments, and the physical and chemical factors affecting the transport of phosphorus in the aquifer near Ashumet Pond. The report describes the distribution of phosphorus and related constituents in the sewage plume on the basis of maps and sections. The report also discusses how phosphorus transport may change after sewage disposal is discontinued in December 1995.



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Zone 19

Figure 1. Extent of study area on western Cape Cod, location of Massachusetts Military Reservation, Ashumet Valley sewage plume, and observation-well sites not shown in study-area maps, Massachusetts (figs. 2 and 3).

METHODS OF INVESTIGATION

Hydrogeologic Data Collection

An existing network of 69 wells augmented with 6 additional wells and 10 permanent drive-point piezometers installed during this study was used to measure water levels in the sand and gravel aquifer to assess ground-water-flow patterns near Ashumet Pond (fig. 2). The wells were installed with a hollow-stem auger, and the drive-point piezometers were installed using a tripod and jackhammer. Water levels were measured in observation wells bimonthly from June 1993 through January 1994 to determine ground-water-flow patterns near the pond. The network of 75 observation wells included 45 different cluster sites; each cluster site had from 1 to 7 wells. The 10 permanent drive-point piezometers were located at three different sites. Physical data for the observation wells and drive-point piezometers are presented in tables 5 and 6 (at back of report). Pond-bottom hydraulic gradients were measured at 15 sites along the northwest shore of Ashumet Pond by use of a drive-point minipiezometer and a manometer board; a detailed description of the method is given in Winter and others (1988). The location of the pond-bottom drive-point minipiezometers are shown in figure 2; physical data for the minipiezometers are included in table 6. Locations of the pond-bottom drive points are relative to the Massachusetts Fish and Wildlife boat landing at Fisherman's Cove (fig. 2).

Water-Quality Sampling

The water-quality sampling network included 23 sites; each site had from 2 to 51 individual sample points (fig. 3). Ground-water samples collected from the network of 80 observation wells, and 19 multilevel samplers were used to assess the spatial distribution of phosphorus and related constituents. The existing network of wells and multilevel samplers was augmented with three observation wells and seven multilevel samplers installed during this study; each multilevel sampler had 15 individual sampling ports. The new wells and multilevel samplers were installed using a hollow-stem auger. Physical data for observation wells and multilevel samplers are shown in tables 5 and 7 (at back of report). Pond-bottom water-quality samples were collected at 16 sites along the northwest shore of Ashumet Pond (fig. 3). Physical data for the pond-bottom drive points are shown in table 6.

Eighty-one water-quality samples were collected from the observation wells and 276 water-quality samples were collected from the multilevel samplers from August through November 1993. Water-quality samples from cluster site F254 were collected in January 1994. Sixteen ground-water samples were collected from the pond-bottom minipiezometers in December 1993 (fig. 3). Ground-water samples were collected from the observation wells using a Keck[1] stainless-steel submersible pump equipped with a packer. The packer was set 2 ft above the top of the screen, and pumping rates were from 150 to 200 mL/min. Ground-water samples were collected from the multilevel samplers and piezometers using a peristaltic pump. A minimum of three casing volumes were evacuated before field parameters (specific conductance, pH, dissolved oxygen, and field alkalinity) were measured and samples were collected. Samples for dissolved constituents at all sampling sites were filtered with 0.45–µm in-line filters, and samples from six of the sampling sites also were filtered with 0.1–µm in-line filters. Phosphorus samples were preserved with mercuric chloride, and metal samples were acidified onsite. Field blanks were collected for sampling equipment used for wells and multilevel samplers. Laboratory duplicates were collected from six wells and seven multilevel sampler ports.

Specific conductance, pH, water temperature, and dissolved oxygen were measured onsite at all sampling sites and field alkalinity was measured onsite at all observation wells. Specific conductance was measured in a flow-through cell with a Hach specific-conductance meter. Measurements of pH were made in a glass beaker with a Beckman pH meter and an Orion pH electrode; calibration of the pH electrode and meter was done using 4.0 and 7.0 pH standard buffers that were cooled to the same temperature as the sample. Water temperature was measured with a mercury thermometer. Dissolved oxygen was

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[1] Any use of trade, product, or firm names in this publication is for identification purposes only and does not constitute endorsement by the U.S. Geological Survey.

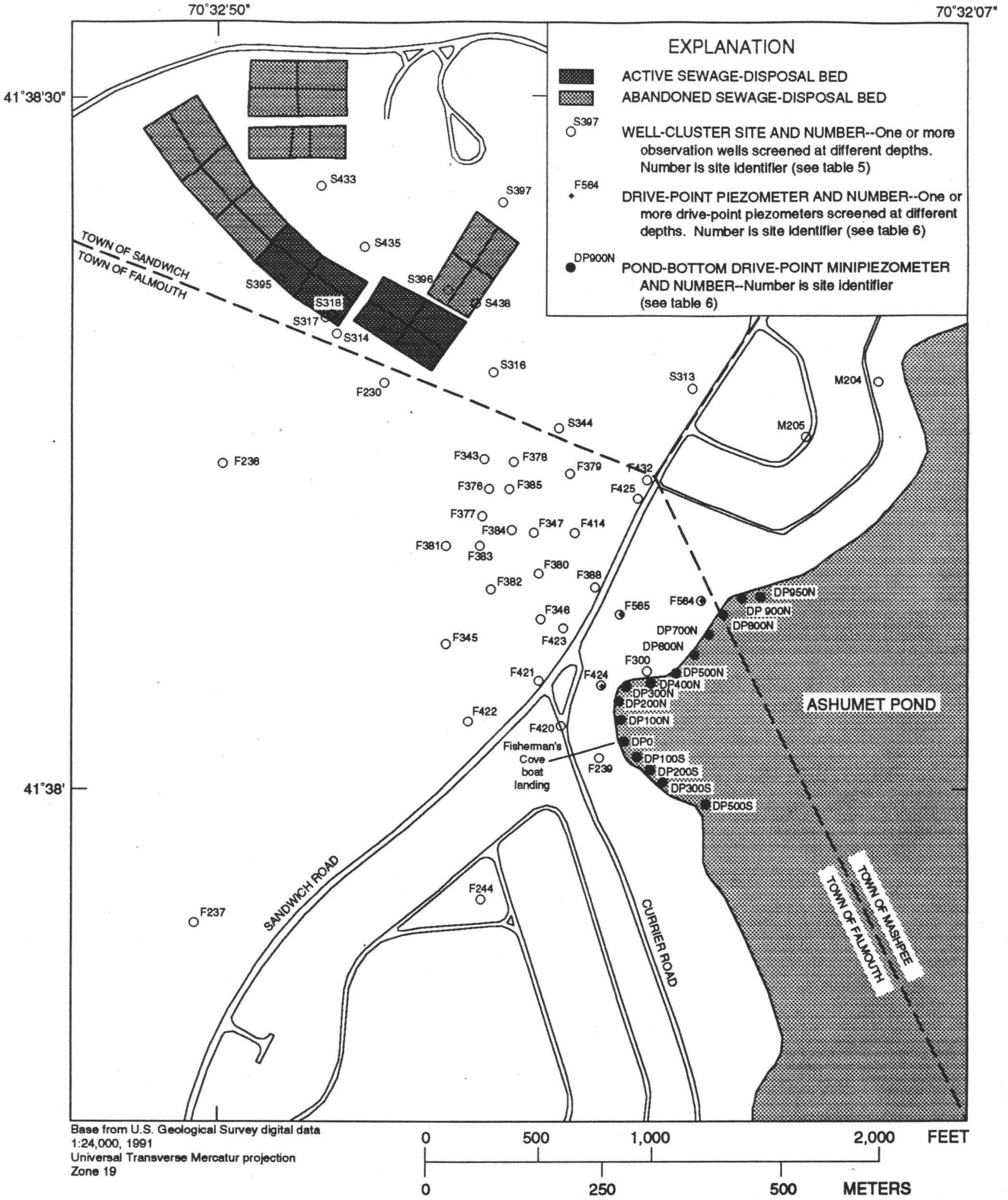

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Figure 2. Location of well-cluster sites, drive-point piezometers, and pond-bottom drive points used in hydrologic data collection near Ashumet Pond, Massachusetts.

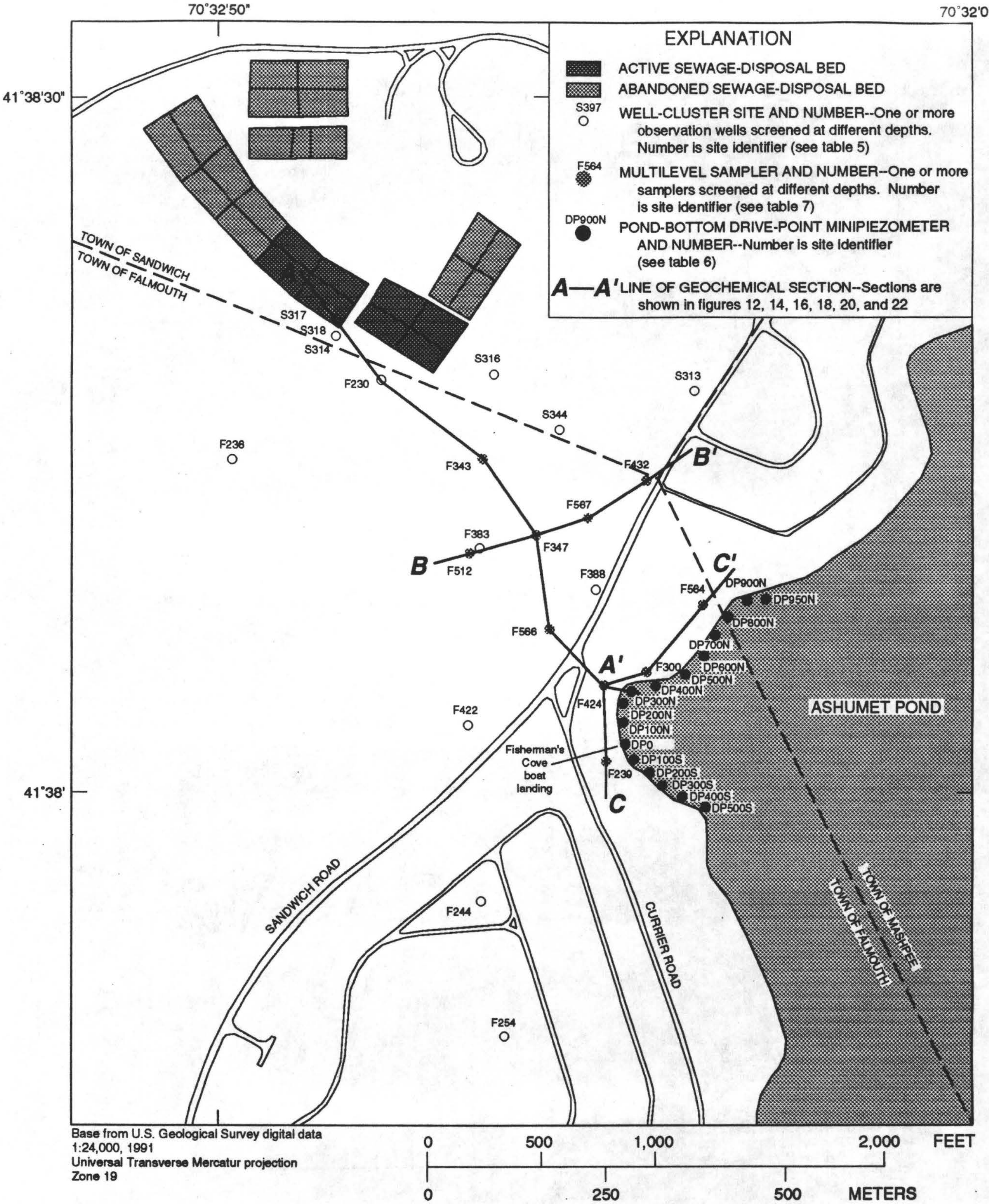

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Figure 3. Location of well-cluster sites, multilevel samplers, and pond-bottom drive points used for water-quality sampling and location of geochemical sections near Ashumet Pond, Massachusetts.

measured in a flow-through cell with a YSI dissolved-oxygen meter. The dissolved oxygen was measured again with a Chemetrix DO chemet if dissolved-oxygen concentrations were less than 1 mg/L. Field alkalinities were measured with a Hach field-alkalinity titration unit.

Dissolved sulfate, phosphorus, iron, and manganese in ground-water samples from all sites were analyzed at the USGS National Water-Quality Laboratory (NWQL) in Arvada, Colo. In this report, the term phosphorus is interchangeable with orthophosphate (PO_4^{3-}); this species of phosphorus is the predominant species of phosphorus in ground water (Hem, 1992). Additional samples for analyses of dissolved nitrate, nitrite plus nitrate, ammonia, boron, dissolved organic carbon, and methylene blue active substances (MBAS) (detergents) were collected at a subset of 35 observation wells. At the NWQL, sulfate was analyzed by inductively coupled plasma atomic emission; nitrogen and phosphorus species were analyzed by automated-segments flow colorimetry; boron was analyzed by direct-current plasma atomic emission; iron and manganese were analyzed using atomic absorption; dissolved organic carbon was analyzed by wet oxidation; and MBAS was analyzed using colorimetry. Additional samples for phosphorus and iron were collected at all sites and analyzed at an onsite laboratory by USGS personnel; phosphorus was analyzed using the ascorbic acid/molybdate-blue colorimetric method (Fishman and Friedman, 1985) and iron was analyzed using the ferrozine colorimetric method.

Sediment Core Collection and Screened Auger Borings

Sediment samples were collected for geochemical analysis at 10 sites (fig. 4); sites were selected to sample the range of geochemical conditions in the sewage plume. The 10 sites include 9 core sites and 1 trench site (site F510). Cores were collected using a Waterloo wire-line and piston core barrel (Zapico and others, 1987), sealed in a polycarbonate lining during collection, and frozen to preserve geochemical conditions in the cores. At two sites, the wire-line and piston-core barrel was augmented with a liquid carbon-dioxide freezing-core shoe to improve core recovery. Aquifer sediments from an uncontaminated part of the unsaturated zone (site F510) were collected from a shallow trench.

A total of 113 split-spoon cores were collected from six sites near Ashumet Pond (fig. 4) to estimate the hydraulic conductivity of the aquifer sediments. Grain-size analyses were done on the cores using an automated sieve shaker. The grain-size analyses were used to estimate hydraulic conductivities according to the methods discussed in Hazen (1893) and Krumbein and Monk (1943).

Ground-water samples were collected from screened auger borings at nine sites (fig. 4) and analyzed for specific conductance, phosphorus, and boron. The sampling was done in cooperation with ABB Environmental, Inc., as part of an ongoing remedial investigation of the sewage plume. The samples were collected from 5-foot screened augers with a submersible pump. Samples were analyzed for specific conductance with a Hach specific-conductance meter; phosphorus was analyzed at an onsite laboratory with the ascorbic acid/molybdate-blue colorimetric method (Fishman and Friedman, 1985), and boron samples were sent to the NWQL for analysis by direct-current plasma atomic emission.

Geochemical Analysis of Core Samples

Sediment cores from different geochemical environments in the sewage plume were analyzed using batch experiments, core extractions, and column experiments to determine the geochemistry of phosphorus on aquifer sediments.

Batch Experiments and Core Extractions

Two composite sediment samples were used in batch experiments; one composite sample was from the unsaturated zone vertically above the sewage plume (site F510), and another composite sample was from the unsaturated zone beneath the active disposal beds (site S317). The composite sediment samples from sites F510 and S317 are referred to as uncontaminated and contaminated composite sediment, respectively. The composites were created using subsections of core material that were air dried and sieved to retain a grain size of less than 2 mm.

An experiment to determine the rate of phosphorus sorption onto uncontaminated sediments was conducted for two initial phosphorus concentrations, 31 and 3.1 mg/L. Sixteen polycarbonate centrifuge tubes for each phosphorus concentration were prepared with 15 g of uncontaminated composite

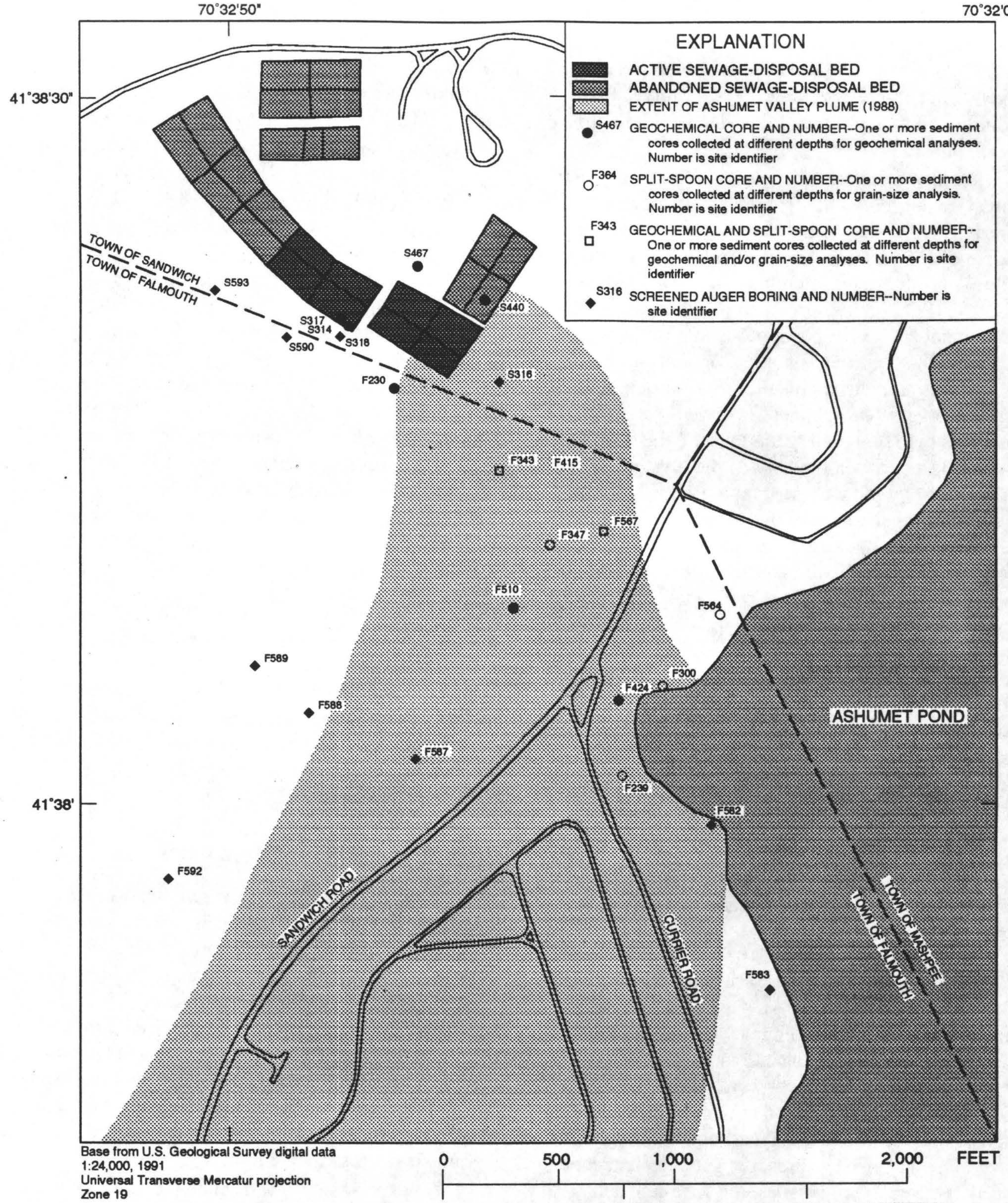

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Figure 4. Location of sediment-sampling and screened auger boring sites near Ashumet Pond and location of active and abandoned sewage-disposal beds at the Massachusetts Military Reservation sewage-treatment plant near Ashumet Pond, Massachusetts.

sediment, and 25 mL of phosphorous-spiked artificial ground water; artificial ground water refers to deionized water that has been altered to approximate uncontaminated ground water at the site. The tubes were placed on an end-over-end mixer. At specified time intervals (from 4 to 240 hours), two tubes of each phosphorus concentration were removed and centrifuged at 13,000 revolutions per minute for 15 minutes. Sixteen milliliters of supernatant solution were removed, placed in a 30-mL high-density polyethylene bottle, and preserved with 100 μL of 6 N (normal) hydrochloric acid (HCl). The supernatant solution was analyzed for phosphorus using the ascorbic acid/molybdate-blue colorimetric method (Fishman and Friedman, 1985).

Adsorption isotherms for phosphorus were constructed for uncontaminated and contaminated sediments. Solutions of artificial plume water containing concentrations of phosphorus ranging from 0.15 to 31 mg/L were created by adding potassium dihydrogen phosphate to artificial ground water. Twenty-five milliliters of phosphorus-contaminated ground water was added to 15 g of sediment, and the pH of the solutions was adjusted to 6.0 standard units with sodium hydroxide. The solutions, which were in 50-mL centrifuge tubes, were placed on an end-over-end mixer for 48 hours; the pH of the solutions was measured after mixing, and then the solutions were centrifuged at 13,000 rpm for 15 minutes. Sixteen milliliters of the supernant solution were removed from each tube, placed in a 30-mL bottle, preserved with 100 μL of 6 N HCl, and analyzed for phosphorus using the same method as described above.

Core extractions were done on core material collected from nine locations in the sewage plume to estimate the amount of extractable phosphorus on the sediment. Several methods for the extraction of phosphorus from specific mineral phases have been used in previous investigations (Bray and Kurtz, 1945; Seyers and others, 1968; Pattersson and others, 1988; Ruttenberg, 1992). The purpose of the extractions used in this study was to determine the amount of extractable phosphorus sorbed to iron-and aluminum-oxide coatings in the aquifer. Core extractions were done using 1 N HCl as the extractant; preliminary experiments showed that 1 N HCl extractions were rapid, able to extract recrystallized or tightly bound phosphorus, and free of artifacts. Preliminary extractions using 0.05 M (molar) ethylenediaminetetraacetic acid (EDTA), and 0.1 N HCl were ineffective at dissolving iron- and aluminum-oxide coatings and yielded phosphorus concentrations that were too low and did not stabilize in sufficient time (72 hours). Extractions with 1 N and 2 N HCl both provided a similar amount of phosphorus in an acceptable period of time (24 to 72 hours); 1 N HCl was chosen as an extractant because it effectively dissolved the iron- and aluminum-oxide coatings and promoted less dissolution of accessory minerals. Individual core subsections were air dried in separate plastic bags. Samples were not sieved, but the grain size greater than 5-mm size fraction was removed. Fifteen grams of sediment and 25 mL of 1 N HCl were placed in 50-mL polycarbonate centrifuge tubes and sealed; the sealed tubes were placed on an end-over-end mixer. After 48 hours, the tubes were removed and centrifuged at 13,000 rpm for 15 minutes; 16 mL of supernatant solution were removed, placed in a 30-mL high-density polyethylene bottle, and analyzed for phosphorus. Core extractions were done in triplicate.

A batch desorption experiment was conducted to determine the rate and extent of phosphorus release under conditions similar to those found in the uncontaminated part of the sand and gravel aquifer. Phosphorus was desorbed from eight subsections of core from beneath the active sewage-disposal beds using artificial ground water with about the same chemical composition as water from the oxic, uncontaminated part of the aquifer; splits from the eight subsections also were extracted with 1 N HCl. Fifteen grams of sediment were placed in a 50-mL polycarbonate centrifuge tube, 25 mL of artificial ground water was added to the tube, and the pH was measured but not adjusted. The tubes were placed on an end-over-end mixer. At the end of specific desorption periods, the tubes were removed from the mixer and centrifuged at 13,000 rpm for 15 minutes. Sixteen milliliters of the centrifuged supernatant solution were removed, placed in a 30-mL high-density polyethylene bottle, and preserved with 100 μL of 6N HCl. As much of the remaining supernatant solution that could be removed was drawn off and discarded; artificial ground water was added back into the tube such that the final weight equaled the weight of the tube, sand, and solution prior to the removal of any supernatant. This last step served to replace as much of the supernatant as was possible with new artificial ground-water solution to allow for the continued desorption of phosphorus and to maintain the sand:solution ratio at 15 g per 25 mL. Weights of the tubes were recorded with and without supernatant solution so that the value of phosphorus measured could be corrected for the amount of phosphorus that was carried over in the entrained supernatant solution to the next desorption step. The tubes then were placed back on the mixer to continue the desorption. The supernatant solution was collected and changed at specific time periods for a total of 32 days. Phosphate concentrations were analyzed colorimetrically as described previously in the extraction section. The pH of all supernatant solutions was measured before centrifugation and after addition of new artificial ground water but was not adjusted. The amount of phosphorus desorbed after 32 days with artificial ground water was compared to the amount extracted with 1 N HCl from splits of the same sediment.

Column Experiments

Column experiments were conducted on uncontaminated and contaminated core samples to evaluate phosphorus sorption and desorption in the aquifer under conditions of one-dimensional flow. Plexiglass columns, 12 in. long by 1.0 in. inside diameter, were packed with either uncontaminated or contaminated core material. A peristaltic pump was used to drive solutions through the columns at a velocity of about 1.38 ft/d—the average velocity of ground water in the sand and gravel aquifer (LeBlanc and others, 1991). Effluent was collected in a fraction collector that was sealed and humidified to minimize evaporation. The experimental conditions for the column experiments are compared to conditions in the aquifer in table 1.

Uncontaminated sediment was collected from a depth of 4.3 to 5.2 ft below land surface in a shallow trench excavated in the unsaturated zone (site F510, fig. 4). The sediment was dry sieved through a 2-mm screen; the less than 2-mm grain-size fraction was used in the experiments. Four columns containing uncontaminated sediment from the unsaturated zone were eluted with uncontaminated ground water to establish precontamination conditions in the column. Bromide was added to the columns as a conservative tracer to estimate longitudinal dispersivity, which was calculated using the standard-probability graph-paper technique (Levenspiel, 1972). After bromide was eluted from the columns, the influent was changed to contaminated ground water from the suboxic part of the sewage plume. Phosphorus concentration in two of the columns was 0.49 mg/L. In the other two columns, the phosphorus concentration was increased to 6.1 mg/L. These concentrations bracket most of the phosphorus concentrations measured in the suboxic part of the sewage plume. In all columns, sewage-contaminated ground water was eluted through the columns until phosphorus concentrations in the effluent approached influent concentrations and stabilized. Influent then was changed back to uncontaminated ground water to evaluate phosphorus desorption. The

Table 1. Experimental conditions for column experiments and conditions in the sand and gravel aquifer, near Ashumet Pond, Cape Cod, Massachusetts

[Column experiment: Experiment refers to influent phosphorus concentration in milligrams per liter. g/cm^3, gram per cubic centimeter; ft/d, foot per day; ft, foot; mg/L, milligram per liter; nd, not determined]

Experiment	Porosity	Bulk density (g/cm^3)	Average velocity (ft/d)	Longitudinal dispersivity (ft)
Uncontaminated Column Experiments				
0.49 mg/L influent, A	0.33	1.68	1.48	0.072
0.49 mg/L influent, B	.33	1.68	1.41	.072
6.1 mg/L influent, A	.34	1.68	1.41	.072
6.1 mg/L influent, B	.32	1.58	1.38	.072
Contaminated Column Experiments				
Site F424 (suboxic)	0.29	1.68	1.48	nd
Site F343 (anoxic)	.33	1.63	1.31	nd
Site F415 (anoxic)	.37	1.61	1.34	nd
Aquifer Conditions				
Sand and gravel aquifer	0.39	1.67	1.38	3.15

chemical characteristics of the uncontaminated and sewage-contaminated ground water used in the experiments are shown in table 2.

Cores from the sewage-contaminated part of the aquifer—one core from the suboxic zone and two cores from the anoxic zone—were eluted with uncontaminated ground water to evaluate phosphorus desorption from contaminated sediments. The suboxic zone refers to that part of the sewage plume that contains low but detectable dissolved oxygen and no dissolved iron, and the anoxic zone refers to the part of the plume that contains no dissolved oxygen and dissolved iron. Core-material samples from the suboxic part of the aquifer were thawed, homogenized, and repacked into columns under atmospheric conditions, whereas core-material samples collected from the anoxic zone were thawed, homogenized, and repacked into columns in a nitrogen glove box to maintain reducing conditions in the cores. The fraction collector tubes used to collect effluent from the anoxic core contained acid to minimize iron oxidation and precipitation.The effluent from the suboxic and anoxic columns were analyzed for phosphorus according to the ascorbic acid/molybdate-blue colorimetric method discussed previously. Effluent from the anoxic column was analyzed for iron according to a ferrozine colorimetric method. Effluent pH was measured in effluent from the suboxic column.

Table 2. Chemical characteristics of uncontaminated and sewage-contaminated ground water used in column experiments, near Ashumet Pond, Cape Cod, Massachusetts, October 1993

[Location of sites shown in figure 4. Concentrations are in milligrams per liter unless otherwise noted. µS/cm, microsiemen per centimeter at 25°C. <, actual value is less than value shown]

Properties and constituents	Well F347 M1-PT (above sewage plume)	Well F343 M3-GY (in sewage plume)
Specific conductance (µS/cm)	51	309
pH (standard units)	5.5	6.5
Calcium (Ca^{2+})	2.9	10
Magnesium (Mg^{2+})	.8	4.1
Sodium (Na^+)	5.5	34
Potassium (K^+)	1.2	8.0
Alkalinity (as $CaCO_3$)	1.4	16
Sulfate (SO_4^{2-})	4.3	25
Chloride (Cl^-)	15	34
Nitrate (NO_3^-)	<.40	48
Phosphate (PO_4^{3-})	<.02	.49 and 6.1
Iron (Fe)	<.003	<.003
Ionic balance (percent)	-1.0	-.9

The geochemical computer model HYTEQ (Kool, 1990) was used to simulate the transport of phosphorus and the nonreactive solute bromide in the sediment columns. HYTEQ is a coupling of the one-dimensional solute-transport code HYDRUS (Kool and van Genuchten, 1990) and the geochemical speciation code MINTEQA2 (Allison and others, 1991), which contains a diffuse-layer surface-complexation model that was used to simulate phosphorus sorption onto aquifer sediments. Input parameters and equilibrium constants used in HYTEQ are listed in table 3. The concentration of sorption sites was determined by potentiometric titration of the sediment with hydrogen ions to a pH of 3.5. Acidity constants for reactions 1 and 2 were fit to potentiometric titrations. Diffuse-layer model parameters and equilibrium constants for the three phosphorus reactions listed in table 3 were fit by trial-and-error to batch-sorption data from Stollenwerk (1995). Surface area was measured utilizing the BET method (Brunauer and others, 1938).

Table 3. Diffuse-layer model parameters and equilibrium constants used to simulate phosphate sorption on sediment, near Ashumet Pond, Cape Cod, Massachusetts

[Reactions 1 and 2 detemined by hydrogen titration; reactions 3, 4, and 5 determined from sorption data from Stollenwerk (1995). Surface area was measured using the BET method (Brunauer and others, 1938). µm, micrometer; g, gram; m², square meter]

Surface-complexation reactions and numbers	Conditional equilibrium constants (log K)
1 $\equiv SOH + H^+ = \equiv SOH_2^+$	6.3
2 $\equiv SOH = \equiv SO^- + H^+$	-7.1
3 $\equiv SOH + PO_4^{3-} + 3H^+ = \equiv SH_2PO_4^0 + H_2O$	27.8
4 $\equiv SOH + PO_4^{3-} + 2H^+ = \equiv SHPO_4^- + H_2O$	21.6
5 $\equiv SOH + PO_4^{3-} + H^+ \equiv SPO_4^{2-} + H_2O$	17.3
Sediment Properties	
Sorption-site density for sediment in laboratory experiments = 1.1 µm g^{-1}	
Surface area = 0.33 m^2 g^{-1}	

HYDROGEOLOGIC SETTING

Regional Hydrogeologic Setting

The study area is located on a glacial outwash plain on western Cape Cod known as the Mashpee Pitted Plain. The outwash plain is bounded to the north by the Sandwich Moraine, to the west by the Buzzards Bay Moraine, to the east by an adjacent outwash plain, and to the south by Vineyard Sound (fig. 1). Collapse structures and kettle ponds are numerous in the outwash plain. Western Cape Cod is underlain by unconsolidated glacial sediments; the sediments consist of gravel, sand, silt, and clay that was deposited during the Pleistocene Epoch as part of a large glacial delta. The deltaic sediments were deposited in a large proglacial lake that formed to the south of the retreating Laurentide Ice Sheet. The underlying bedrock consists of granodiorite. The deposits can be divided into topset, foreset, and bottomset beds (Byron Stone, U.S. Geological Survey, oral commun., 1993). Topset deposits are glaciofluvial sediments that consist of coarse sand and gravel. The underlying foreset deposits are glaciolacustrine deposits of medium to fine sand. Bottomset deposits are glaciolacustrine deposits that consist of fine sand and silt. Most ground-water flow is in the coarser grained topset and foreset beds. All three depositional units become finer with increasing distance from the sediment source, located near the Cape Cod Canal. The deltaic sediments are underlain locally by glacial till consisting of poorly sorted sand, silt, and clay.

The regional ground-water-flow system of western Cape Cod is bounded to the north, south, and west by saltwater, to the east by a ground-water divide, below by impermeable bedrock, and above by the water table. The ground-water system generally is unconfined; however, confined conditions can occur locally beneath fine-grained deposits. Precipitation is the sole source of recharge to the aquifer and ground water discharges primarily to streams and saltwater embayments. The flow system is dominated by a water-table mound located to the north of the study area. Ground water flows radially from the mound, which has a maximum hydraulic head of 70 ft. In the study area, land-surface altitude ranges from about 40 to 90 ft above sea level, and the water-table altitude ranges from about 44 to 49 ft above sea level. Ground-water flow, which generally is southward near the study area, is primarily in the topset and foreset deposits and is strongly affected by ponds. A flow-through condition is indicated in ponds on Cape Cod where shallow ground water discharges to a pond at the upgradient side and pond water recharges the aquifer at the downgradient side. Ground water deep in the system flows beneath the ponds. Ashumet Pond is a flow-through pond that has a long-term average stage of 44 ft and a maximum depth of about 60 ft.

Aquifer Structure and Properties Near Ashumet Pond

The glacial silt, sand, and gravel underlying the study area can be separated into three depositional units as described previously. The contact between the glaciofluvial coarse sand and gravel (topset) and the glaciolacustrine sand (foreset) is at an altitude of about 10 ft below sea level near Ashumet Pond. The contact between the glaciolacustrine sand (foreset) and the underlying glaciolacustrine fine sand and silt (bottomset) is at an altitude of about 70 ft below sea level. The bedrock surface is at an altitude of about 250 ft below sea level in the study area, which gives a total saturated thickness of about 300 ft; the entire saturated thickness beneath the study area contains freshwater. Ground-water flow at the site occurs primarily within the topset and foreset beds, resulting in an effective saturated thickness of about 120 ft. Ashumet Pond is located within a collapse structure that formed when a buried ice block melted and the overlying sediments collapsed. The subsurface geology within collapse structures is typically different than the surrounding aquifer; the contact between the medium sand and the fine sand and silt deposits beneath the Ashumet Pond probably is deeper than in the surrounding outwash plain (Byron Stone, U.S. Geological Survey, oral commun., 1993).

Analysis of an aquifer test conducted in sand and gravel deposits by the U.S. Geological Survey about 1,000 ft downgradient of the sewage-disposal beds, yielded an average hydraulic conductivity of 380 ft/d and a porosity of 0.39 (LeBlanc and others, 1988). Moench (1994) reported a hydraulic conductivity of the sand and gravel aquifer of 302 ft/d from an aquifer test conducted near Ashumet Pond. Hydraulic conductivities measured at 16 sites between the disposal beds and the pond using a borehole flowmeter had a geometric mean of 312 ft/d (Hess and others, 1992), whereas estimates of hydraulic conductivity made from grain-size distributions in the same area, as determined from 33 core samples, were about 112 ft/d (Wolf and others, 1991). Sand and gravel deposits elsewhere on western Cape Cod typically have hydraulic conductivities from 250 to 350 ft/d (Barlow and Hess, 1993; Masterson and Barlow, 1994).

Split-spoon samples were collected at varying depths in the aquifer at three sites along Ashumet Pond, F239, F300, and F564, and at three upgradient sites, F567, F343, and F347 (fig. 4). Grain-size distributions were determined for split-spoon cores from sites F239, F564, and F567 according to standard sieving practices (Folk, 1974), and estimates of hydraulic conductivity were obtained from the grain-size distributions based on the methods of Hazen (1893) and Krumbein and Monk (1943). Hydraulic conductivities were estimated based on the Hazen method for sites F300, F343, and F347 as part of a separate study (Thompson, 1993).

The vertical distribution of estimated hydraulic conductivity at the six split-spoon sample sites is shown in figure 5. The geometric mean of the hydraulic conductivities estimated from the Hazen and the Krumbein and Monk methods were 136 and 103 ft/d, respectively. The hydraulic-conductivity values estimated with the Hazen method were on average about 29 percent higher than hydraulic conductivities estimated from the Krumbein and Monk method. The estimates of hydraulic conductivity at the six sites (fig. 5) are useful for evaluating vertical trends in hydraulic conductivity even though estimates from grain-size distribution are commonly significantly lower than values determined from other methods (Wolf and others, 1991). At the sites along Ashumet Pond, a zone of high hydraulic conductivity is present from about 15 to 35 ft above sea level. Hydraulic-conductivity values generally decrease with depth. Glacial sediments on Cape Cod typically display a coarsening upwards sequence consistent with a rapidly aggrading proglacial delta (Byron Stone, U.S. Geological Survey, oral commun., 1993). The zone of high hydraulic conductivity may represent coarse sand and gravel deposits associated with glaciofluvial topset sediments. The grain-size distributions from sites F343 and F347, located upgradient of the pond, suggest a contact between coarse-grained, homogeneous deposits and underlying deposits that are finer grained and more heterogeneous, as delineated by large variations in hydraulic conductivity at an altitude from 30 to 40 ft below sea level. The zone of highly variable hydraulic conductivities was not observed at the sites located along the pond. The hydraulic-conductivity distributions suggest that the overlying coarser, more homogeneous deposits extend to a greater depth along the pond. The location of Ashumet Pond within a collapse structure—areas where coarse-grained sediments typically extend to greater depths—could explain the different trends in the hydraulic-conductivity distributions between sites located upgradient of the pond and along the pond. The mean hydraulic conductivity estimated from the grain sizes also was higher at sites along the pond than at sites upgradient of the pond, which indicates that aquifer sediments are coarser grained in the collapse structure.

Ground-Water Flow

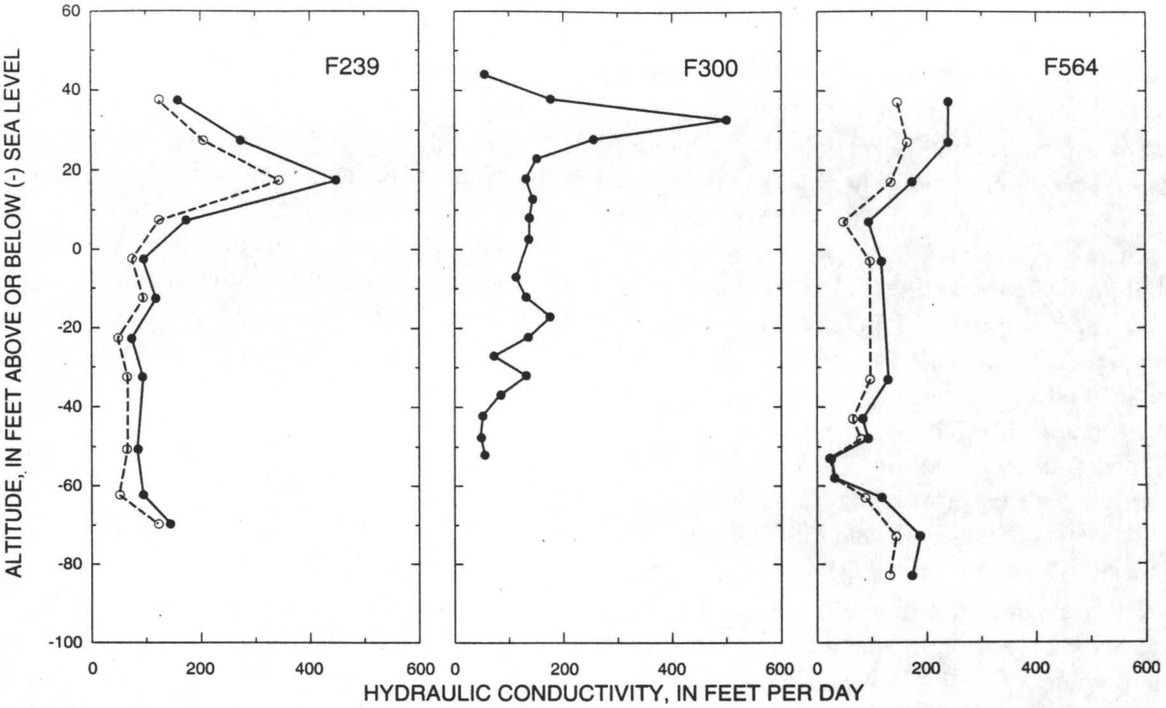
Hydraulic data collected during the study were used to assess the ground-water-flow system between the sewage-disposal beds and Ashumet Pond. The data were used to evaluate flow directions, ground-water velocities, advective traveltimes, and ground-water fluxes and to assess ground-water/surface-water interaction along Ashumet Pond.

Hydraulic Heads and Gradients

Water levels in the sand and gravel aquifer and in Ashumet Pond fluctuate seasonally; both are higher in the spring and summer months than in the autumn and winter months (fig. 6A). Pond-level altitudes in Ashumet Pond ranged from 45.18 ft above sea level in May 1993 to 43.11 ft above sea level in November 1993. Water levels in well FSW167-55 fluctuated about 3 ft over the same period; the well is located about 2 mi southwest of Ashumet Pond (fig. 1). Data show that hydrologic conditions during the study generally were typical of the lake and aquifer conditions during the previous 21 years of record (fig. 6*B*). Water-level altitudes in well FSW167-55 had a median value of 41.37 ft and a mean value of 41.46 ft during the study. The median and mean water-level altitudes in well FSW167-55 since 1975 were 41.21 and 40.91 ft, respectively. The fluctuations in ground-water levels and pond levels in Ashumet Pond showed similar trends, which indicate that Ashumet Pond is in good hydrologic connection with the aquifer and that the pond is an expression of the local water table.

Ashumet Pond shows a flow-through condition with respect to ground water. Ground water discharges to the northern part of the pond and the pond acts as a local ground-water sink. As ground-water levels increase in the aquifer, the lower storativity of the

A. Sites along Ashumet Pond



B. Sites upgradient of Ashumet Pond

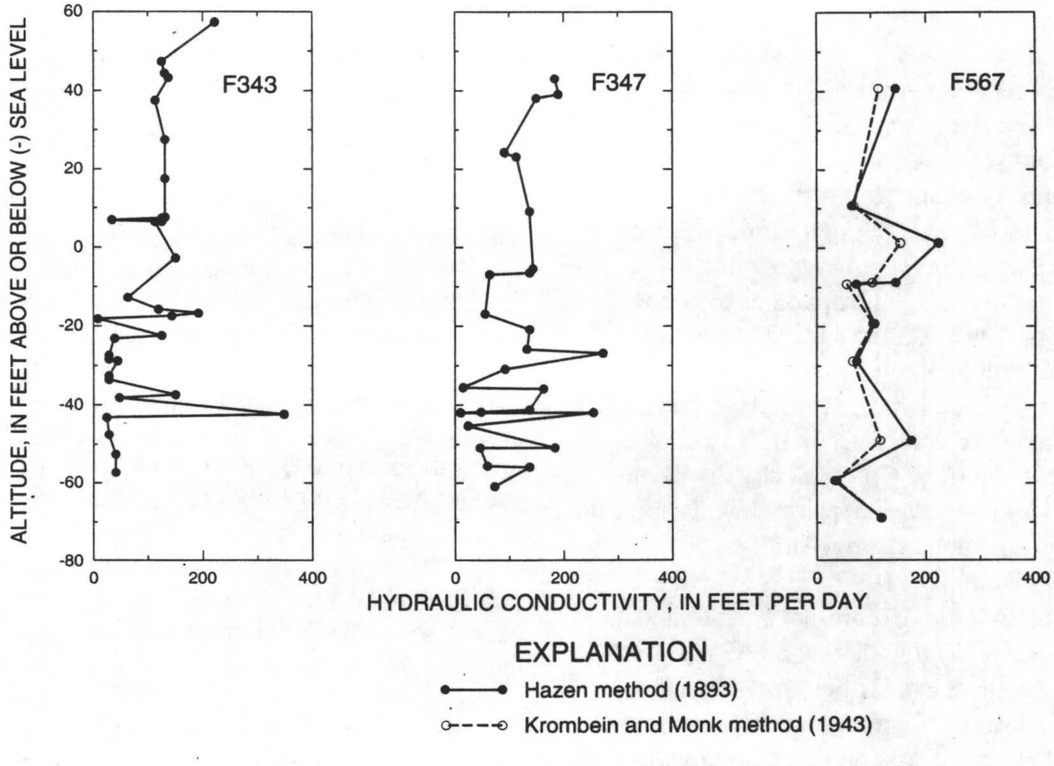


Figure 5. Vertical distribution of hydraulic conductivity estimated on the basis of grain-size analysis of split-spoon samples from (*A*) three sites along Ashumet Pond and (*B*) three sites upgradient of Ashumet Pond, Massachusetts. (Location of sites shown in fig. 4.)

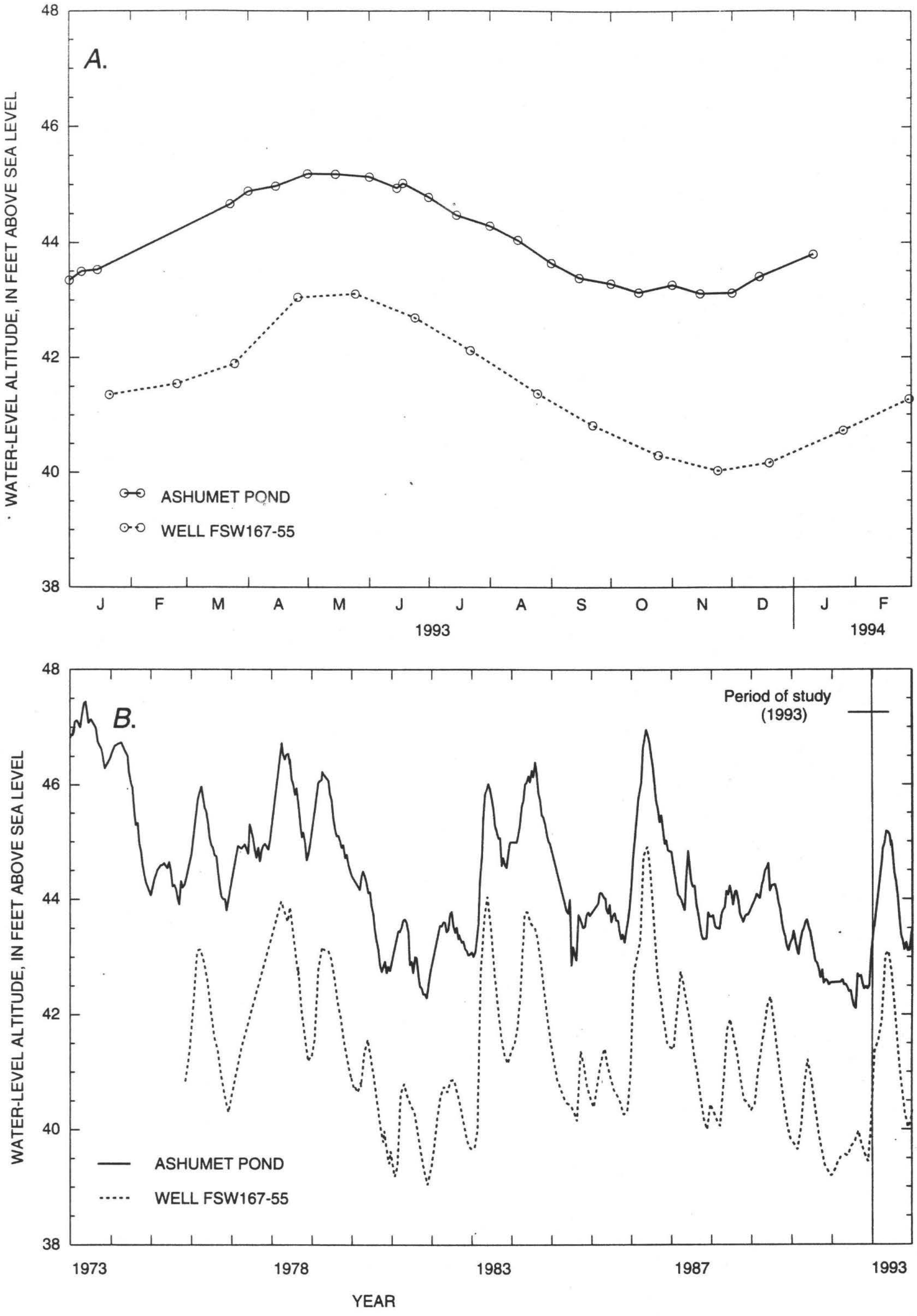


Figure 6. Water-level altitudes in Ashumet Pond, Massachusetts, and in well FSW167-55 (*A*) from December 1992 through February 1994 and (*B*) from 1973 through 1993. (Location of site F167 shown in fig. 1.)

aquifer causes water levels to rise more rapidly in the aquifer than water levels in the pond. The rapid rise in water levels in the aquifer causes hydraulic gradients to increase and ground water to flow more directly toward the pond, which results in an increased flux of ground water to the pond during rising water levels. The effect of Ashumet Pond on local ground-water-flow patterns is shown in figures 7*A* and 7*B*. In August 1993, when water levels were high, the direction of ground-water flow was to the southeast, directly toward the pond, and the hydraulic gradient was steeper (fig. 7*A*). In January 1994, when water levels were lower, ground-water flow was more southerly, away from the pond, and hydraulic gradients were less steep (fig. 7*B*).

Horizontal gradient directions and magnitudes have been measured at three sites in the study area since November 1984; gradients were calculated by a linear interpolation of water-level altitudes at wells FSW343-36, FSW382-32, and FSW414-36. During the period of record, the mean gradient magnitude and standard deviation were 0.162 and 0.013 percent, respectively, and the mean gradient direction and standard deviation were 166.4° east of north and 4.4°, respectively. Gradient magnitude and direction varied with changes in water-level altitude (fig. 8). The data show that during periods of high water levels in the aquifer, gradients were higher, and flow was more easterly, or towards Ashumet Pond. During this investigation, gradient direction ranged from 159.3° east of north in July 1993 to 174.4° east of north in December 1993, averaged 164.8° east of north, and had a standard deviation of 4.8°. Gradient magnitude ranged from 0.191 percent in July 1993 to 0.147 percent in January 1994; the average gradient magnitude and standard deviation were 0.169 and 0.014 percent, respectively. The data show that the gradient increases and shifts more directly toward the pond when water levels are high, resulting in a larger component of ground-water discharge to the pond than when water levels are low.

Ground-water flow upgradient of the pond is predominately horizontal. The data show that vertical-head gradients were negligible and that vertical flow components in the aquifer were minor. Vertical gradients in the aquifer ranged from 0.01 to 0.09 percent, whereas the horizontal gradient averaged 0.169 percent. Vertical gradients near the pond shore are significantly higher because of the strong local effect of the pond.

Traveltimes and Ground-Water Fluxes

Gradient magnitudes and hydraulic conductivities were used to estimate ground-water traveltimes and fluxes to Ashumet Pond. Assuming an average hydraulic conductivity of 300 ft/d, estimated from previous aquifer tests, an average gradient at 0.00169, and a porosity of 0.39 (LeBlanc and others, 1988), ground-water velocities averaged about 1.3 ft/d during the course of the study. The average traveltime of ground water from the center of the sewage-disposal beds to the pond, a distance of about 2,000 ft, was about 4.2 years.

The direction and magnitude of the hydraulic gradient affects the flux of ground water into Ashumet Pond. The data indicate that ground-water fluxes and the amount of phosphorus entering the pond fluctuate seasonally; the fluxes are highest at higher water levels. Gradient data were used to estimate the flux of ground water into the pond within the area of the sewage plume. Saturated thicknesses were estimated on the basis of water-table altitudes and the contact between coarse- and fine-grained deposits at about 70 ft below sea level, and the width of the part of the plume discharging into the pond was estimated from water-table contours. Assuming a hydraulic conductivity of 300 ft/d, a porosity of 0.39, and the gradient data discussed previously, fluxes ranged from 15,200 ft^3/d in January 1994 to 37,300 ft^3/d in July 1993. The flux estimates, which are based on aerial flow lines, do not take into account vertical gradients and negligible ground-water underflow in the coarse-grained deposits is assumed.

Ground-Water/Surface-Water Interactions

Pond stage and the vertical distribution of hydraulic heads in wells at four sites within 100 ft of Ashumet Pond (fig. 9) show that vertical head gradients are small between wells and that almost all measured head loss occurs close to the pond shore. At well F300, the vertical gradient was 0.03 percent, whereas the horizontal gradient between the water table and Ashumet Pond was 0.625 percent. The locally high gradients near the pond shore are due to the high storage coefficient of the pond and the relatively slow rise in pond stage during periods of rising ground-water levels.

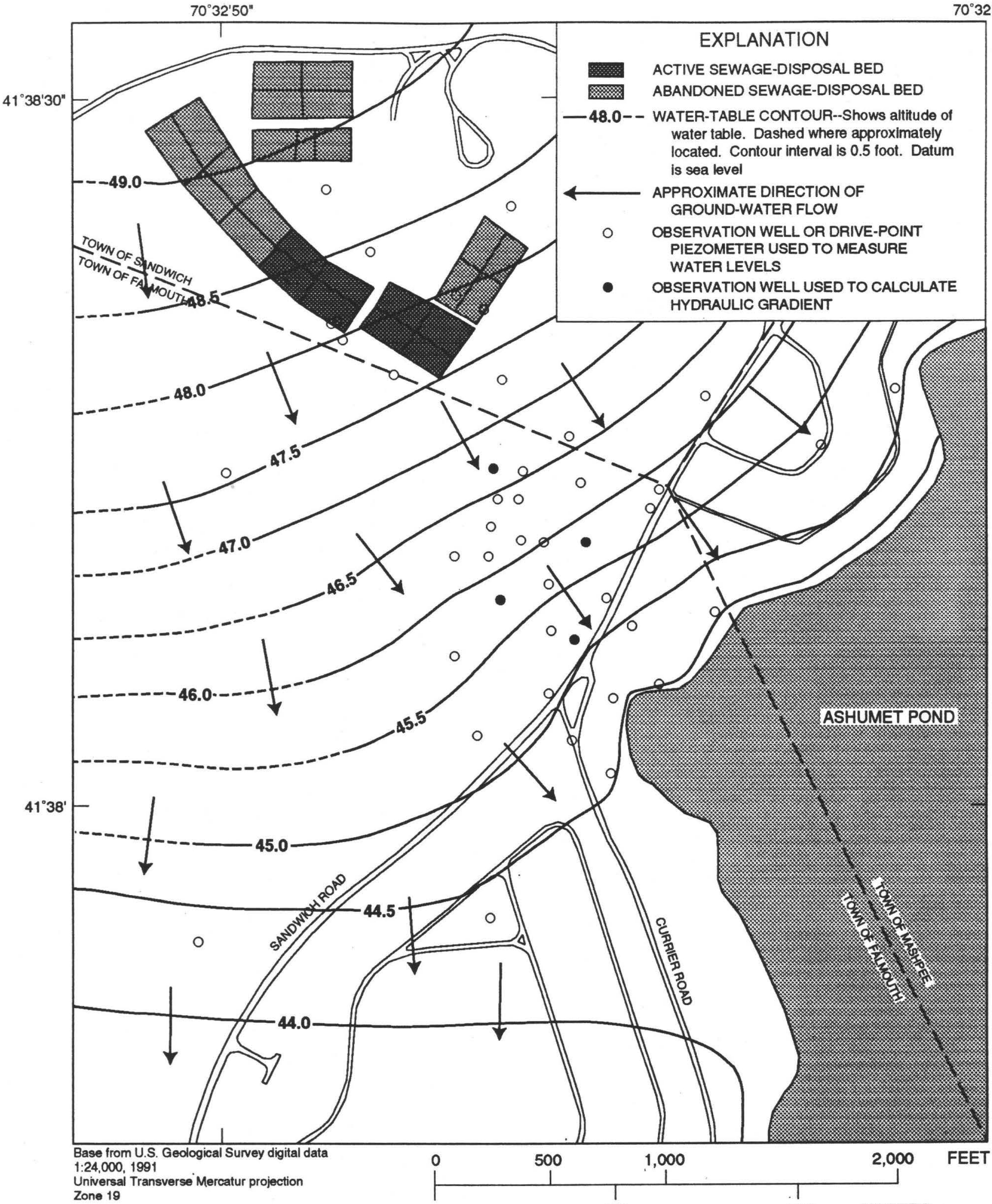


Figure 7. Altitude of the water table and approximate direction of ground-water flow for (*A*) August 1993 and (*B*) January 1994 near Ashumet Pond, Massachusetts.

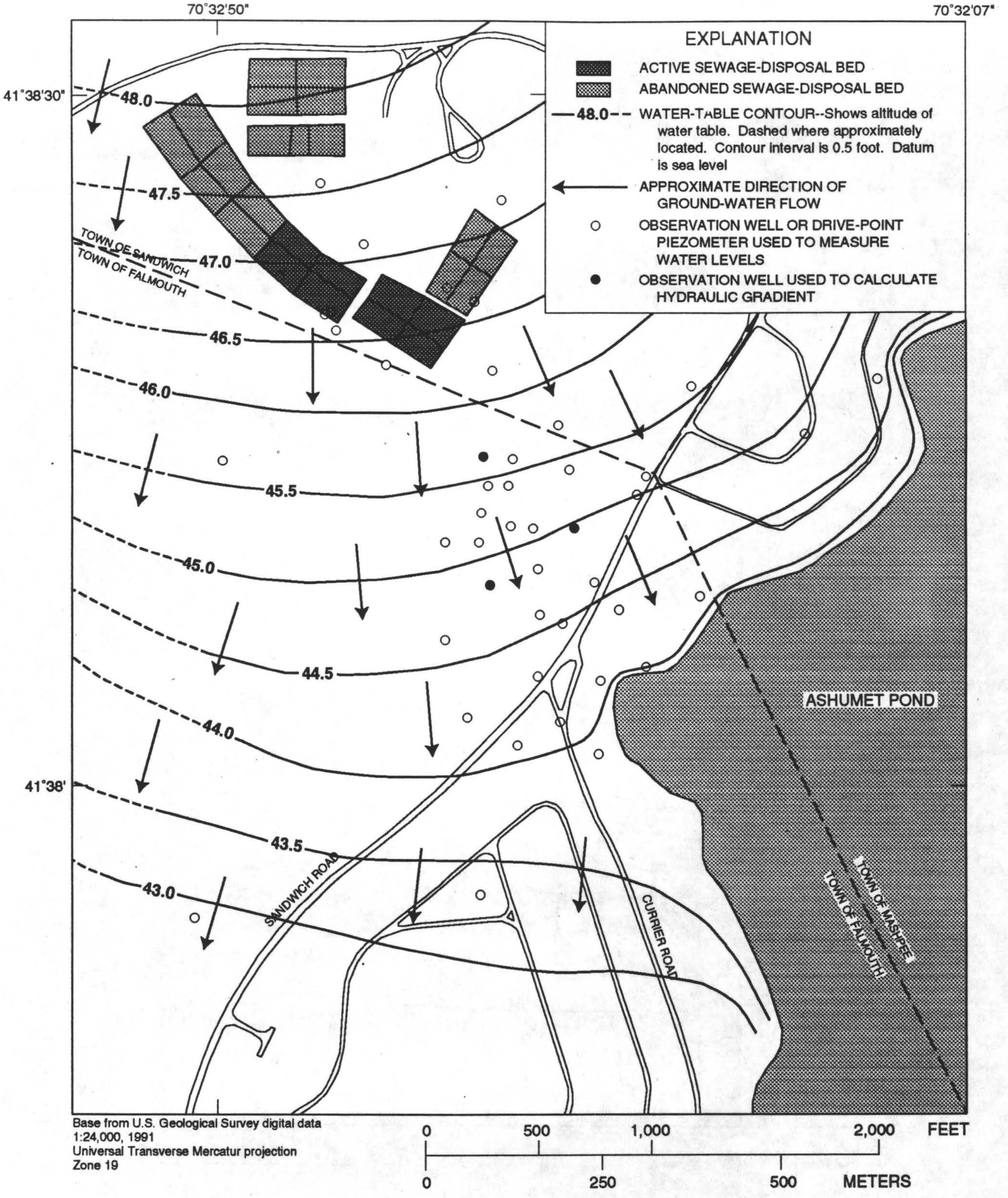


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Figure 7. *Continued.*

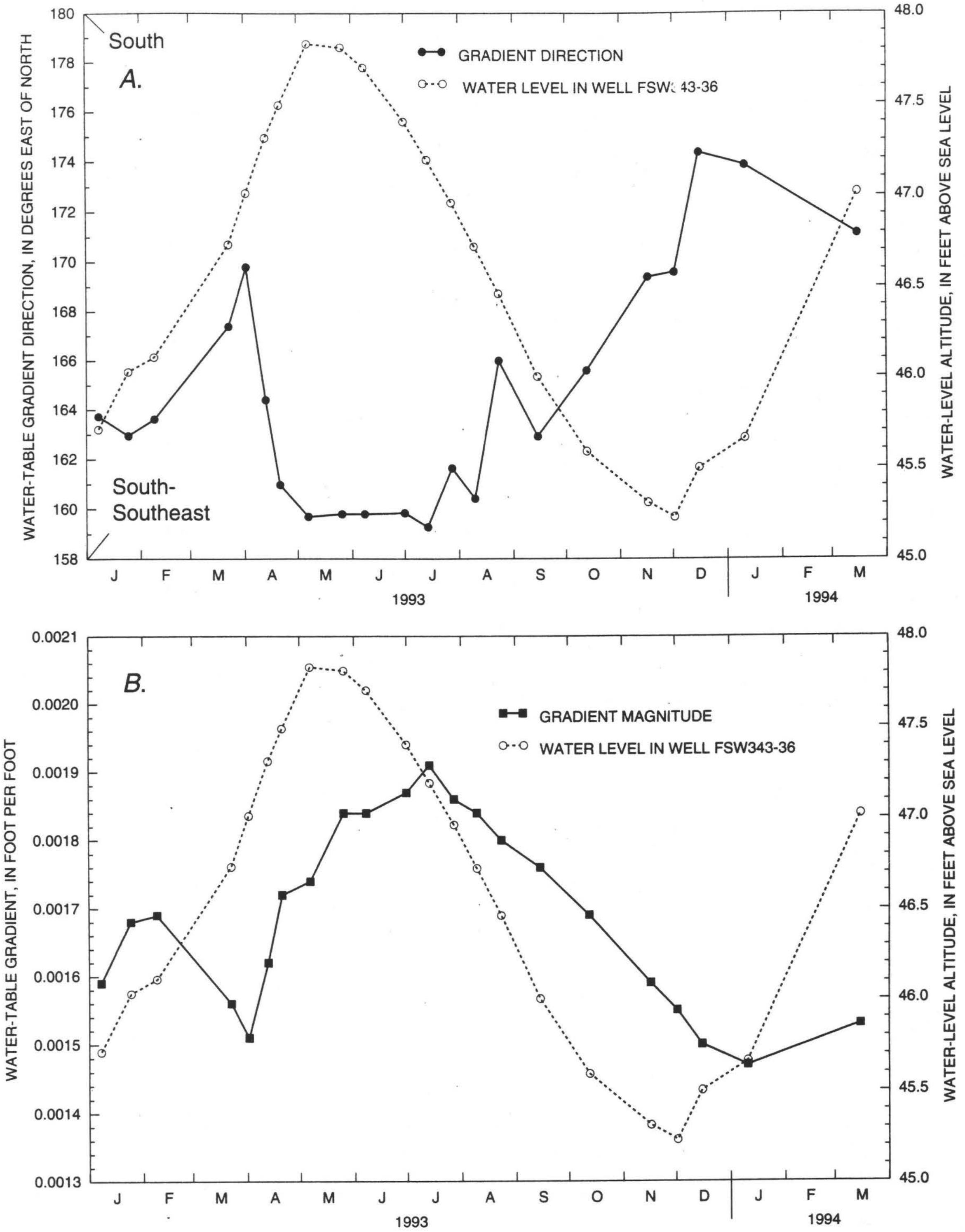


Figure 8. Changes in (*A*) hydraulic-gradient direction and (*B*) hydraulic-gradient magnitude with water-level altitude in well FSW343-36 (site F343) near Ashumet Pond, Massachusetts, January 1993 through March 1994.

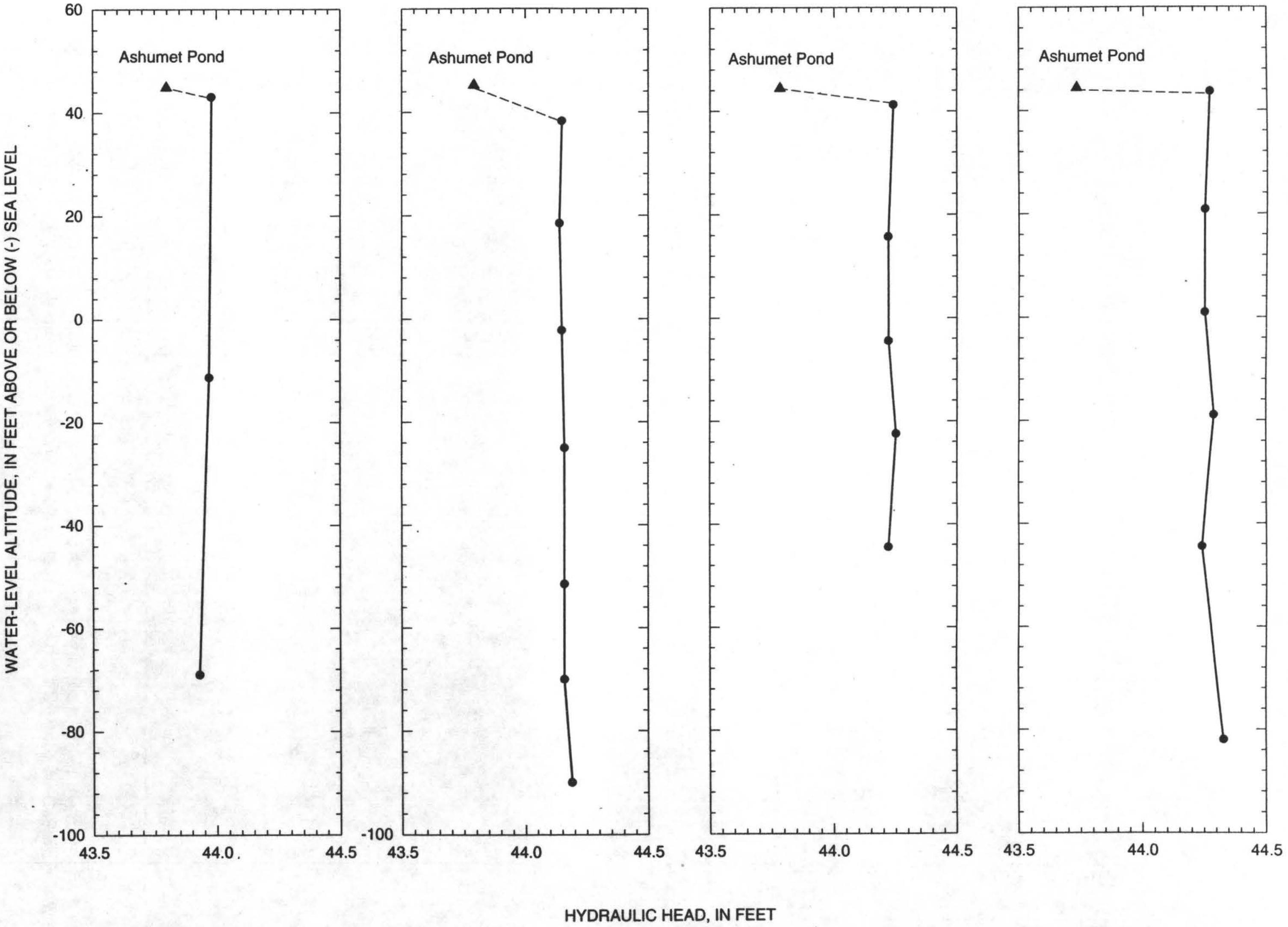


Figure 9. Pond stage and vertical distribution of hydraulic head in four wells along Ashumet Pond, Massachusetts, January 1994.

Pond-bottom minipiezometers, at 15 sites, were installed at a distance of 10 ft from the shoreline and at an altitude of about 38 ft above sea level (fig. 2). Hydraulic gradients between the pond and the aquifer were measured using a manometer board. Pond-bottom gradients were as high as 6.7 percent, indicating a strong upward gradient and ground-water discharge into the pond (fig. 10); vertical gradients in the aquifer adjacent to the pond averaged about 0.02 percent. The large variations in the pond-bottom hydraulic gradients are a function of the pond-bottom hydraulic conductivity. The data indicate a strong upward component of ground-water flow along the shore of the pond; McBride and Pfannkuch (1975) found that most discharge to lakes is along the shore and that gradients across the lake bottom decrease exponentially with increasing distance from the shore. The vertical gradient reached 0 at distance of about 500 ft south of the Fisherman's Cove boat landing, indicating no vertical flow between the aquifer and the pond. The location of the zero gradient in a flow-through pond is an area of transition from areas of ground-water discharge to the pond to areas of pond-water recharge to the aquifer and would be expected to change seasonally.

The strong upward gradient along the pond shore indicates that ground-water-flow lines curve sharply upward as they near the pond. Comparison of specific-conductance measurements in pond-bottom minipiezometers and in adjacent multilevel samplers along the pond shore indicate that the top of the sewage plume, as delineated by specific-conductance values that were higher than background values, rises steeply within 100 ft of the pond. The angle of rise associated with nearshore discharge into the pond ranged from 9.0° in the area of well F564 to 22° in the area of well F239. Water-quality data from pond-bottom minipiezometers and multilevel samplers are discussed in the section "Ground-Water Quality."

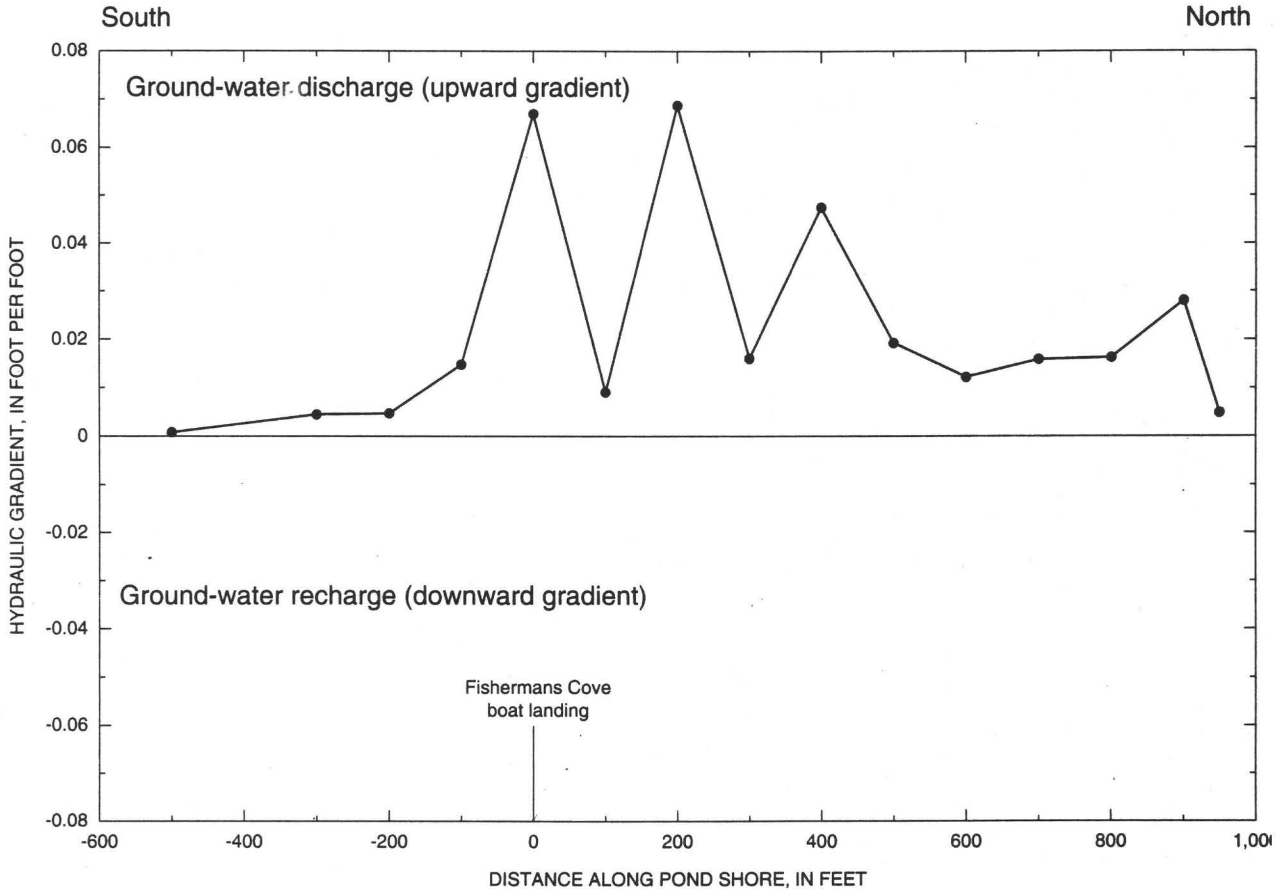


Figure 10. Pond-bottom hydraulic gradients at 15 minipiezometer sites along the shore of Ashumet Pond, Massachusetts, December 1993.

GROUND-WATER QUALITY

The chemical quality of ground water in the study area varies both areally and vertically; the greatest contrast in water quality is between uncontaminated and sewage-contaminated ground water. LeBlanc (1984) conducted extensive sampling of the entire sewage plume and found that the plume was at least 11,000 ft long, 3,000 ft wide, and as much as 75 ft thick; the part of the plume that contained phosphorus was about 2,500 ft long. Sewage-derived contaminants are transported by advection and the shape of plume is affected by ground-water-flow directions in the aquifer. The history and distribution of sewage disposal onto the disposal beds also affect the distribution of sewage-derived contaminants.

Water-quality samples collected during the study were used to assess the distribution of important chemical constituents in ground water for August through December 1993. Areal distributions of specific conductance, pH, dissolved oxygen, iron, manganese, and phosphorus are illustrated with maps; these constituents were selected because they may be related to the transport of phosphorus in the aquifer. The distributions are defined on the basis of the maximum values of specific conductance, pH, iron, manganese, phosphorus, and minimum values of dissolved oxygen at each of the 23 water-quality sampling sites. Vertical distributions are illustrated with geochemical sections, the locations of which are shown in figure 3. Section *A-A´* is a longitudinal section that coincides with the general direction of ground-water flow and extends from the active sewage-disposal beds to Ashumet Pond. Section *B-B´* is a transverse section about 1,350 ft downgradient of the disposal beds. Section *C-C´* is a transverse section near the shore of Ashumet Pond, about 2,000 ft downgradient of the disposal beds. Locations and distances described in this section of the report are relative to the center of the eight active disposal beds.

Comparison of Background Ground-Water Quality and Sewage-Plume Water Quality

Water-quality data collected from observation wells, multilevel samplers, and minipiezometers are presented in tables 8, 9, and 10 (at back of report). The results of chemical analyses of uncontaminated and sewage-contaminated ground-water samples from selected wells are given in table 4.

The quality of water from well FSW300-10 (shallow well) located vertically above the sewage plume, and from well FSW432-92 (deep well), located to the east of the plume, is typical of uncontaminated ground water in the study area. Uncontaminated ground water typically is slightly acidic and has a low specific conductance and alkalinity. Dissolved oxygen is typically high and approaches saturation near the water table. Phosphorus and dissolved nitrogen species are typically at or less than the detection limit and iron and manganese are less than the detection limit. Locally, naturally reducing conditions associated with fine-grained deposits are observed elsewhere in the aquifer; these environments are characterized by high specific conductance, low dissolved oxygen, and elevated concentrations of iron and manganese. Anthropogenic constituents, such as boron and surfactants (MBAS), are less than the detection limit in uncontaminated ground water. Uncontaminated ground water typically has low concentrations of sulfate and organic carbon.

The quality of sewage-contaminated ground water varies according to geochemical conditions in the plume, the distance from the disposal beds, and the loading history at individual beds. The quality of water from well FSW343-57 is typical of contaminated ground water in the anoxic zone; the anoxic zone refers to that part of the sewage plume that contains no dissolved oxygen and high concentrations of dissolved iron. The quality of water from well FSW300-30 is typical of contaminated ground water in the suboxic zone; the suboxic zone refers to that part of the plume that contains low but detectable dissolved oxygen, contains little or no dissolved iron, and may contain high concentrations of dissolved manganese. Contaminated ground water typically has higher specific-conductance and pH values than uncontaminated ground water. Nitrogen concentrations typically are higher in the sewage plume than in uncontaminated ground water; the predominate species is related to redox conditions, with ammonia predominate in reducing environments and nitrate predominate in more oxidizing environments in the plume. Concentrations of phosphorus also are typically higher in the plume than in uncontaminated ground water; phosphate is the predominate species dissolved in the ground water. Contaminated ground water has higher concentrations of anthropogenic constituents such as boron and surfactants than uncontaminated ground water. Concentrations of sulfate and organic carbon also are higher in sewage-contaminated ground water than in uncontaminated ground water.

Table 4. Chemical analyses of water from selected wells between the sewage-disposal beds and Ashumet Pond, Massachusetts, August and September 1993

[Concentrations are in milligrams per liter unless otherwise noted. µS/cm, microsiemen per centimeter at 25°C; °C, degrees Celsius; MBAS, methylene blue active substances. --, no data; <, actual value is less than value shown]

Properties and constituents	Uncontaminated ground water		Sewage-contaminated ground water	
	Well FSW300-10 (shallow)	Well FSW432-92 (deep)	Well FSW343-57 (anoxic zone)	Well FSW300-30 (suboxic zone)
Date of sample (month-day-year)	9-08-93	9-14-93	8-24-93	9-08-93
Elevation of top of screen (feet)	39.3	[1]-21.5	14.1	19.5
Elevation of bottom of screen (feet)	37.3	[1]-23.5	12.1	17.5
Specific conductance (µS/cm)	80	102	310	400
pH (standard units)	5.2	5.7	6.6	6.3
Temperature, water (°C)	15.0	12.0	14.0	13
Oxygen, dissolved	9.2	12.5	<.05	.15
Alkalinity (as $CaCO_3$)	3	4	84	43
Sulfate, dissolved	7.7	19	23	39
Nitrate, dissolved (as N)	<.01	<.01	<.01	<.01
Nitrite plus nitrate, dissolved (as N)	.22	<.35	<.01	<14
Ammonia, dissolved, organic (as N)	.02	<.01	1.2	3.1
Phosphorus, dissolved (as P)	<.01	.03	3.0	1.1
Boron, dissolved	<10	<10	360	250
Iron, dissolved	<.01	<.01	14.0	<.01
Manganese, dissolved	<.01	<.01	.25	13
Carbon, dissolved, organic (as C)	.30	--	2.7	2.0
MBAS (detergents)	<.02	<.02	.10	<.02

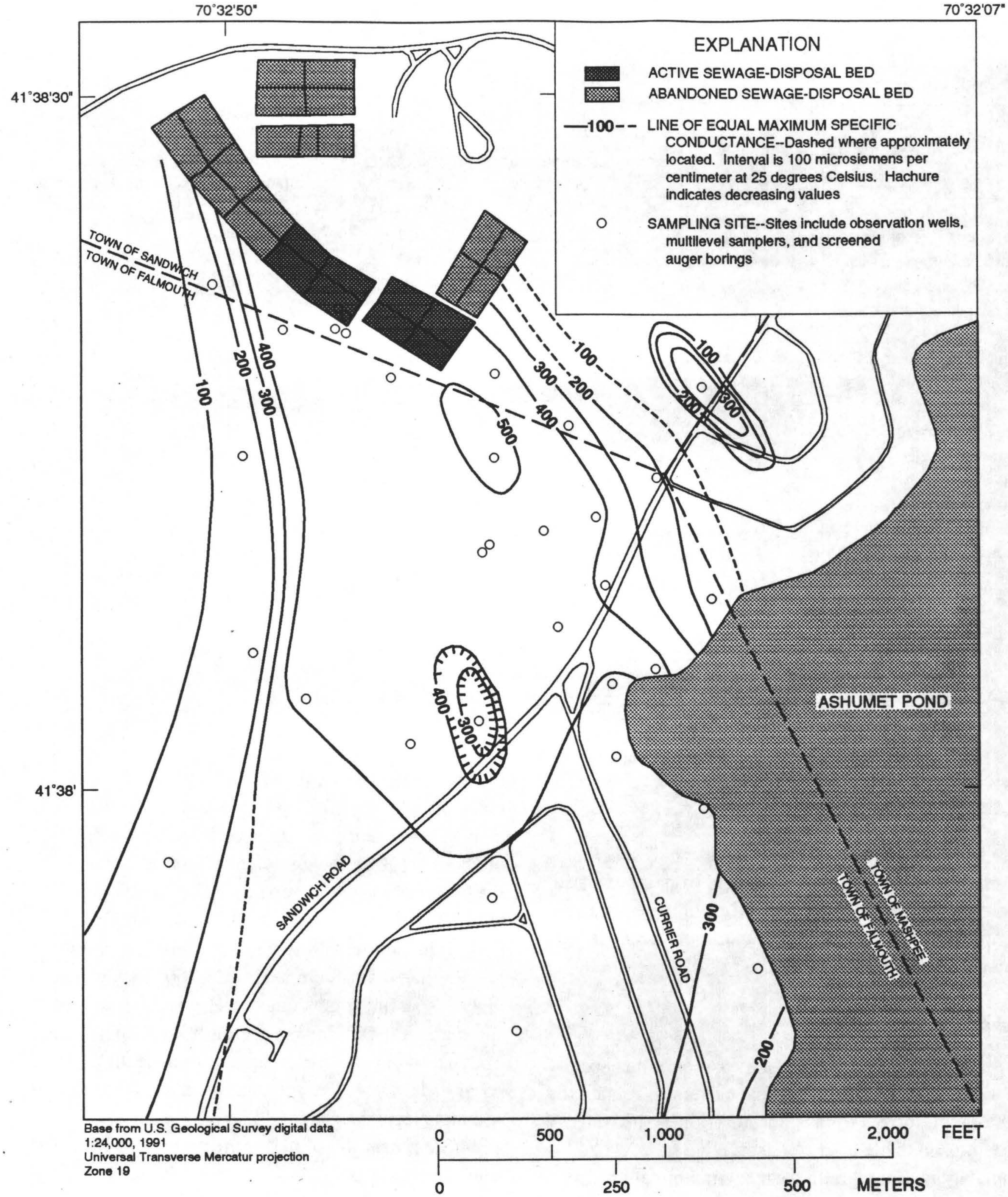
[1] Minus sign (-) indicates elevation is below sea level.

Selected Water-Quality Characteristics of the Sewage Plume

The areal and vertical distribution of specific conductance, pH, dissolved oxygen, iron, manganese, and phosphorus were determined from water-quality sampling and are illustrated in maps and geochemical cross sections.

Specific Conductance

Specific conductance, a measure of the capacity of water to conduct an electrical current, is related to the total dissolved-solids concentration in the water. Specific conductance is a conservative property and is a good indicator of ground-water contamination (Hem, 1992); sewage-contaminated ground water in the study area typically has higher specific-conductance values than uncontaminated ground water. The specific conductance of the sewage effluent was 497 µS/cm in October 1993. The specific conductance of the ground water exceeded 300 µS/cm for a distance of about 2,950 ft downgradient of the sewage-disposal beds and exceeded 400 µS/cm for a distance of about 2,150 ft downgradient (fig. 11). The part of the plume with specific-conductance values greater than 300 µS/cm is about 1,600 ft wide, whereas the part of the plume with specific-conductance values greater than 400 µS/cm is about 1,350 ft wide. The highest specific conductance, 541 µS/cm, was measured about 800 ft downgradient of the disposal beds. The maximum specific conductance beneath the disposal beds was 491 µS/cm, and maximum specific-conductance values along the pond ranged from 420 µS/cm near the center of the plume to 176 µS/cm near the eastern edge of the plume. The vertical distribution of specific conductance is shown in figure 12; lines of equal concentration beneath the pond on these and all subsequent sections bend upwards near the pond, which illustrates the local effect of nearshore, upward gradients on plume migration. The zone of



Base from U.S. Geological Survey digital data
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Figure 11. Areal distribution of maximum specific conductance in ground water near Ashumet Pond, Massachusetts, August through November 1993.

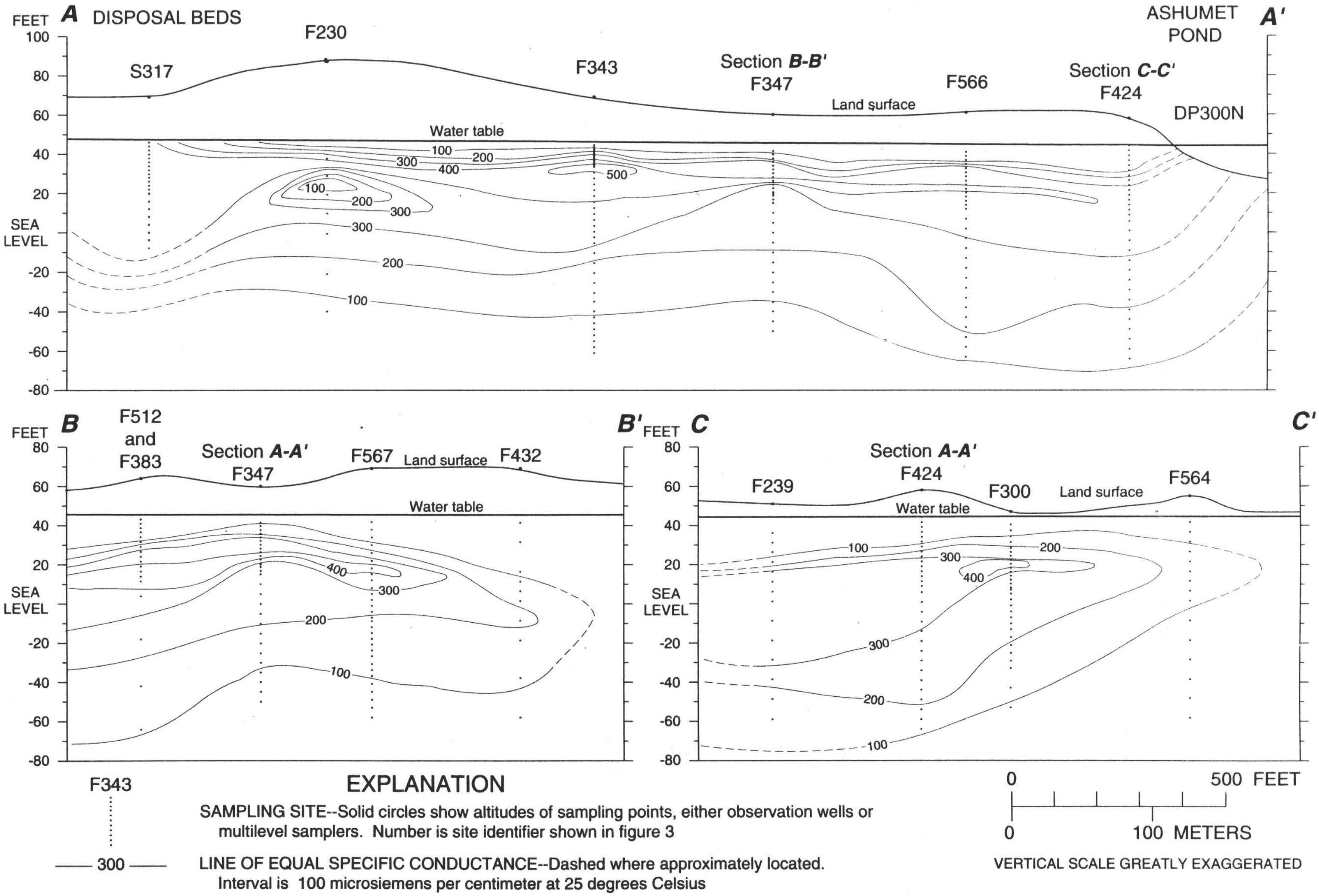


Figure 12. Vertical distribution of specific conductance in ground water along geochemical sections *A-A′*, *B-B′*, and *C-C′* near Ashumet Pond, Massachusetts, August through November 1993. (Location of geochemical sections are shown in fig. 3.)

contaminated ground water with specific-conductance values greater than 300 μS/cm ranged in thickness from about 7 ft along the pond to more than 45 ft beneath the sewage-disposal beds. Along Ashumet Pond, the zone of ground water with specific-conductance values greater than 300 μS/cm had a maximum thickness of about 40 ft. As the plume is transported downgradient of the disposal beds, areal recharge causes the plume to sink; the sewage plume is overlain by a zone of uncontaminated ground water characterized by specific-conductance values less than 100 μS/cm. The thickness of the zone of uncontaminated recharge water ranged from 0 ft beneath the disposal beds to about 22 ft along the pond.

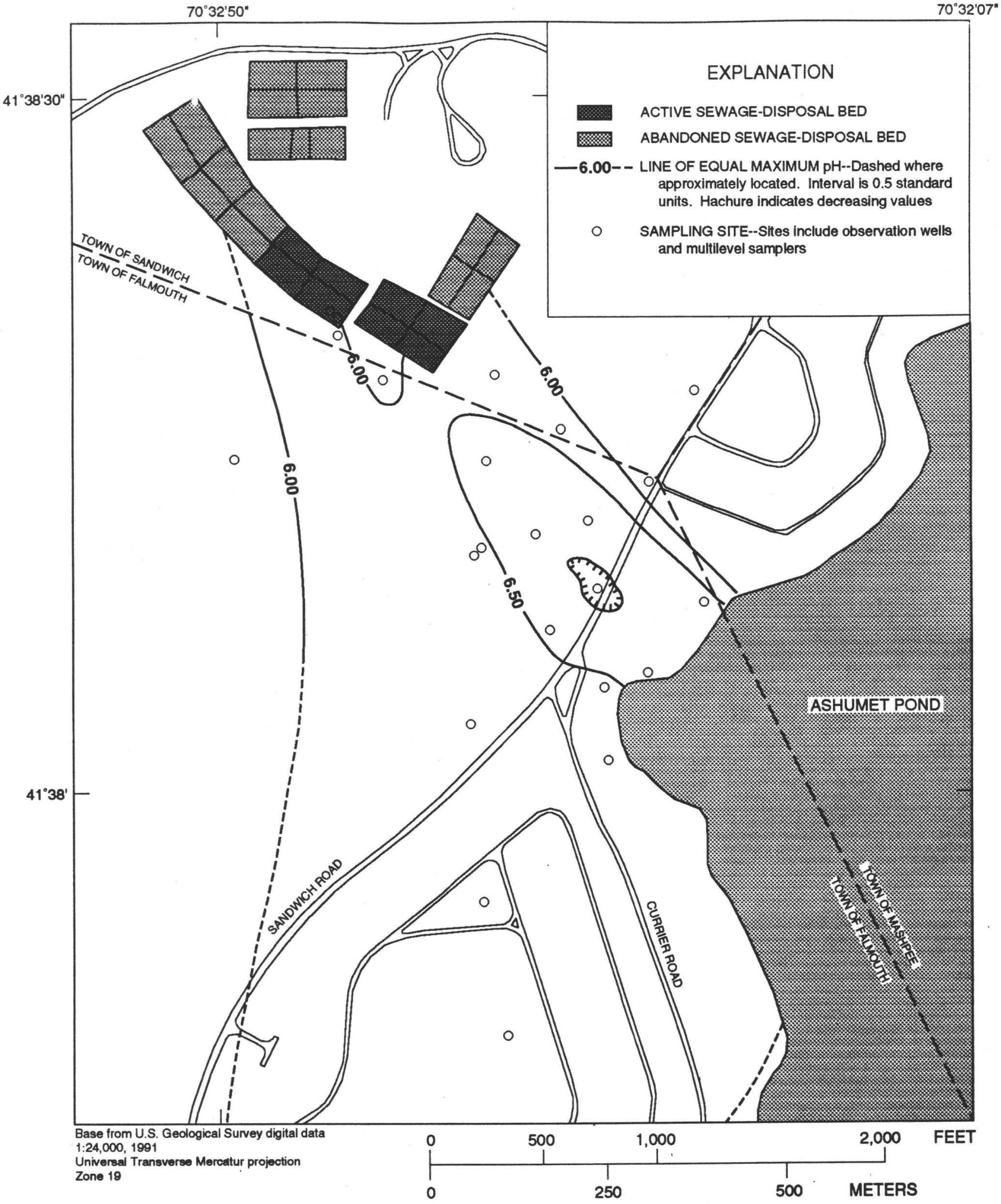
pH and Dissolved Oxygen

The disposal of sewage effluent with near-neutral pH values causes sewage-contaminated ground water to have a higher pH than uncontaminated water; the pH of the sewage effluent was 7.0 in October 1993. The pH of the ground water exceeded 6.0 in an area that was more than 3,600 ft long and 2,000 ft wide and exceeded 6.5 in an area that was about 1,500 ft long and about 700 ft wide (fig. 13). The highest pH was 6.9 at a distance of about 1,350 ft downgradient of the disposal beds. The vertical distribution of pH between the sewage-disposal beds and Ashumet Pond is shown in figure 14. The highest pH beneath the active disposal bed was 6.2, and the highest pH along the pond was 6.6. The zone of contaminated ground water with pH values greater than 6.0 ranged in thickness from about 6 ft beneath the disposal beds to more than 80 ft along the pond. Values of pH exceeded 6.5 in a zone that was about 20 ft thick along the pond.

The oxidation of organic carbon and reduced-nitrogen species, which are constituents in sewage effluent, consumes dissolved oxygen and has created zones of anoxic and suboxic ground water downgradient of the sewage-disposal beds. A zone of anoxic ground water, which contains less than 0.05 mg/L dissolved oxygen, extends about 1,500 ft downgradient of the disposal beds and has a maximum width of about 750 ft (fig. 15); dissolved-oxygen concentrations less than 0.05 mg/L, which was the detection limit of the analytical method, are given values of 0 on the map. A zone of suboxic ground water, which contains from 0.05 to 1.0 mg/L of dissolved oxygen, extends more than 3,600 ft downgradient of the disposal beds and has a maximum width of about 1,450 ft. The vertical distribution of dissolved oxygen in the aquifer is shown in figure 16. The anoxic zone has a maximum thickness of about 40 ft near the center of the plume. The suboxic zone ranges in thickness from about 36 ft beneath the active disposal beds to more than 80 ft thick along Ashumet Pond. Dissolved oxygen beneath the active disposal beds indicates that insufficient time has elapsed to allow for the consumption of all available dissolved oxygen by the degradation of organic carbon; this kinetically controlled process explains why, in some areas, the upgradient extent of the anoxic zone is downgradient of the disposal beds.

Iron and Manganese

In the anoxic zone, where all available dissolved oxygen has been consumed, conditions are reducing enough to favor the dissolution of the iron oxyhydroxide coatings that are found naturally on aquifer sediments, which results in the release of ferrous iron into the ground water. Natural concentrations of dissolved iron in the aquifer typically are less than 0.01 mg/L. A zone of dissolved iron greater than 0.1 mg/L extends about 1,650 ft downgradient of the disposal beds (fig. 17). Concentrations of dissolved iron exceed 1.0 mg/L in an area that extends about 1,450 ft downgradient and has a maximum width of about 750 ft, which corresponds to the extent of the anoxic zone. The highest concentration of dissolved iron, which was 25 mg/L, was in the center of the sewage plume, about 950 ft downgradient of the disposal beds. The zone of dissolved iron begins about 300 ft downgradient of the western four active disposal beds because suboxic conditions persist beneath the active beds. The vertical distribution of dissolved iron in the aquifer is shown in figure 18. The zone of ground water with iron concentrations greater than 1.0 mg/L had a maximum thickness of about 40 ft near the center of the plume. The altitude of the high dissolved iron corresponds to the altitude of the anoxic zone. Dissolved iron is less than 0.1 mg/L beneath the active disposal beds and along Ashumet Pond.



Base from U.S. Geological Survey digital data 1:24,000, 1991
Universal Transverse Mercatur projection
Zone 19

Figure 13. Areal distribution of maximum pH in ground water near Ashumet Pond, Massachusetts, August through November 1993.

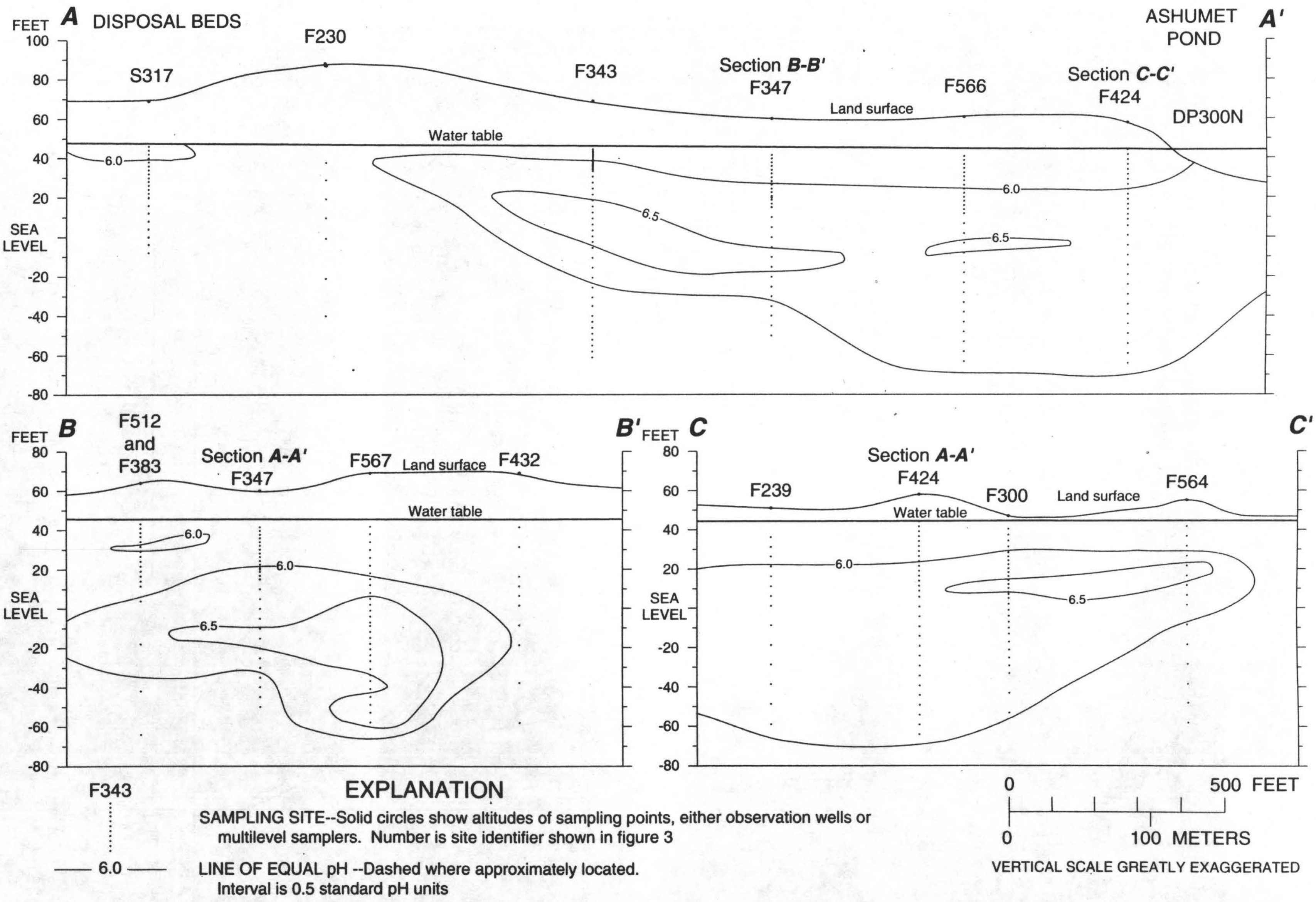
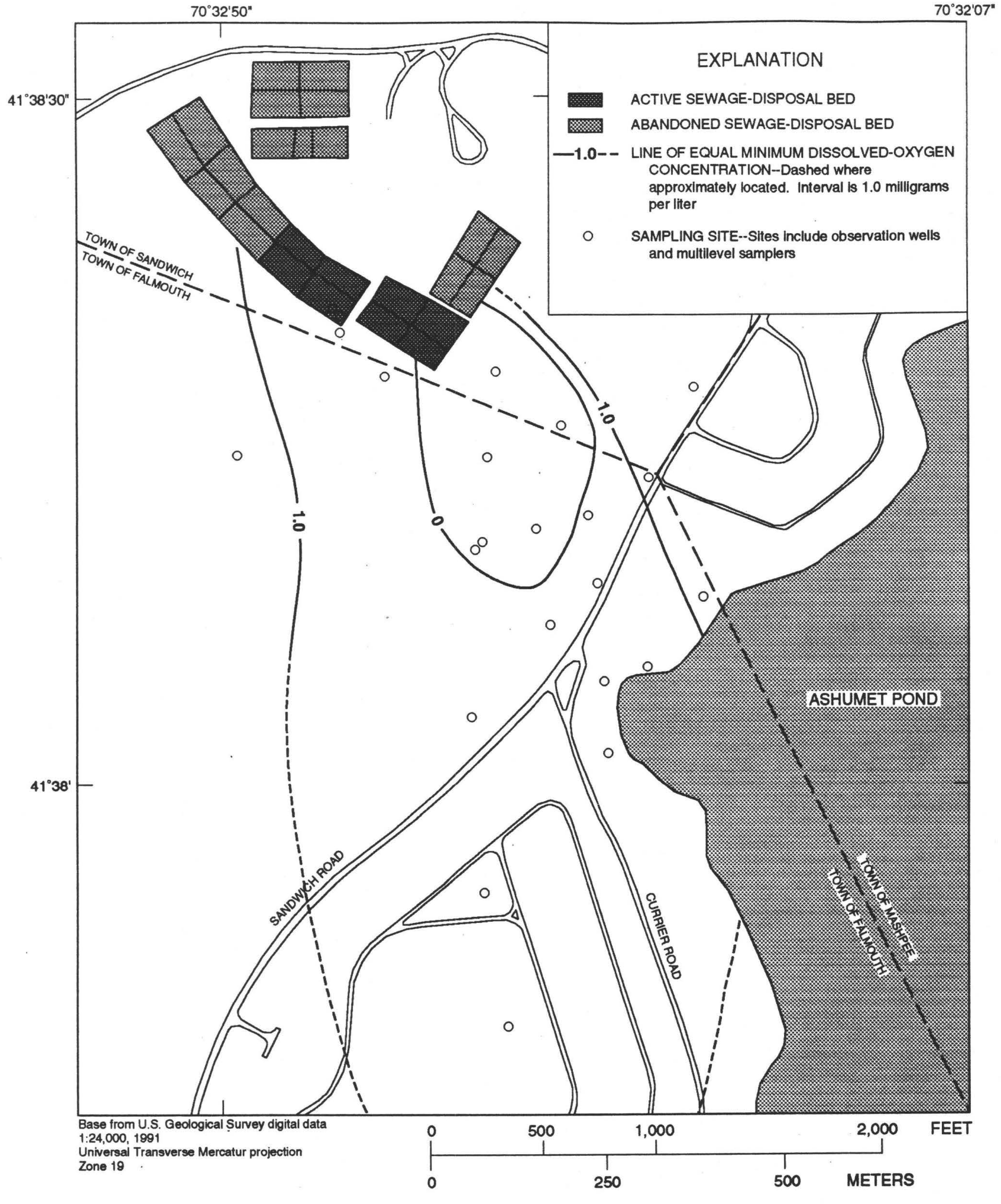


Figure 14. Vertical distribution of pH in ground water along geochemical sections *A-A′*, *B-B′*, and *C-C′* near Ashumet Pond, Massachusetts, August through November 1993. (Location of geochemical sections are shown in fig. 3.)



Base from U.S. Geological Survey digital data
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Figure 15. Areal distribution of minimum dissolved-oxygen concentrations in ground water near Ashumet Pond, Massachusetts, August through November 1993.

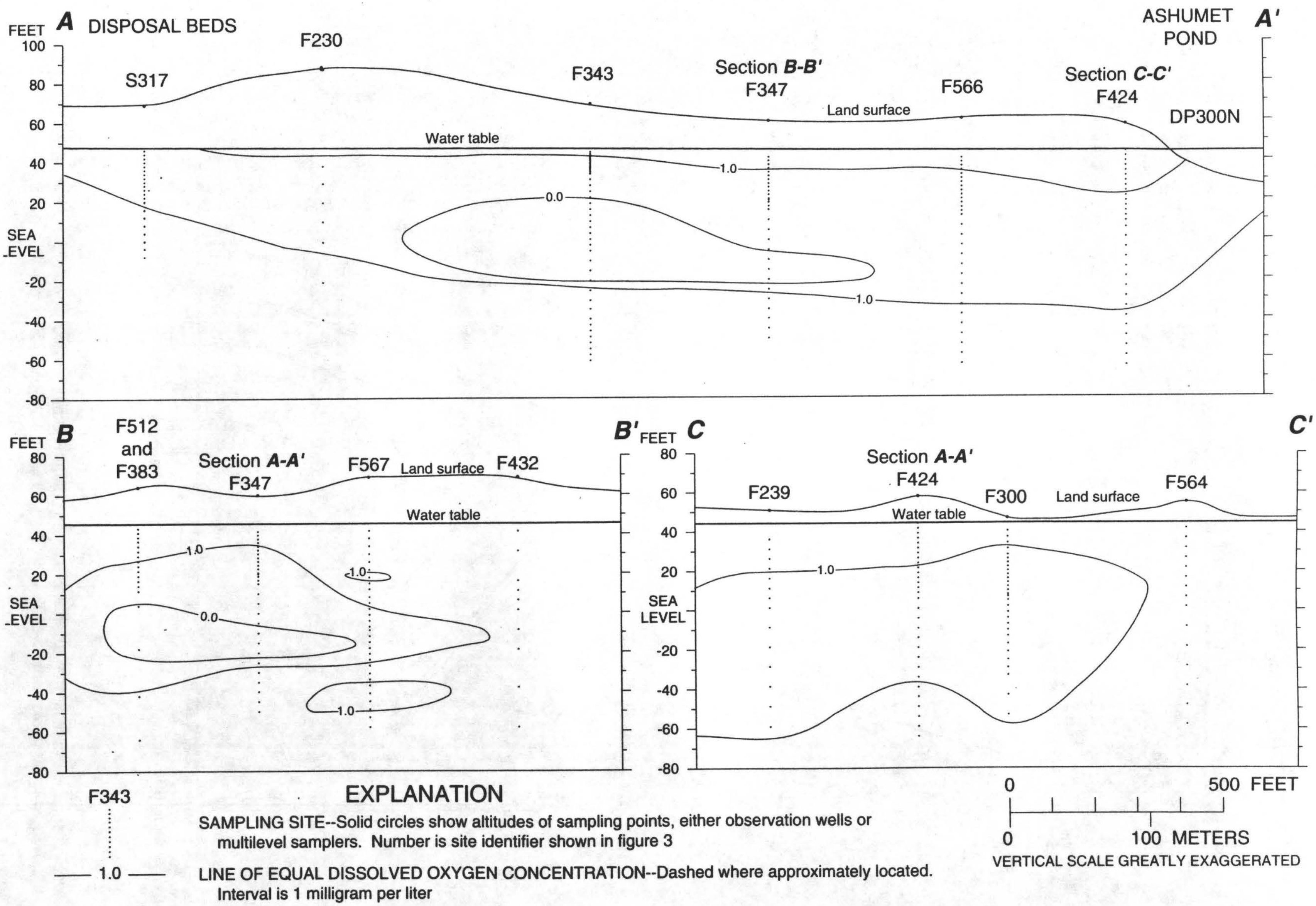
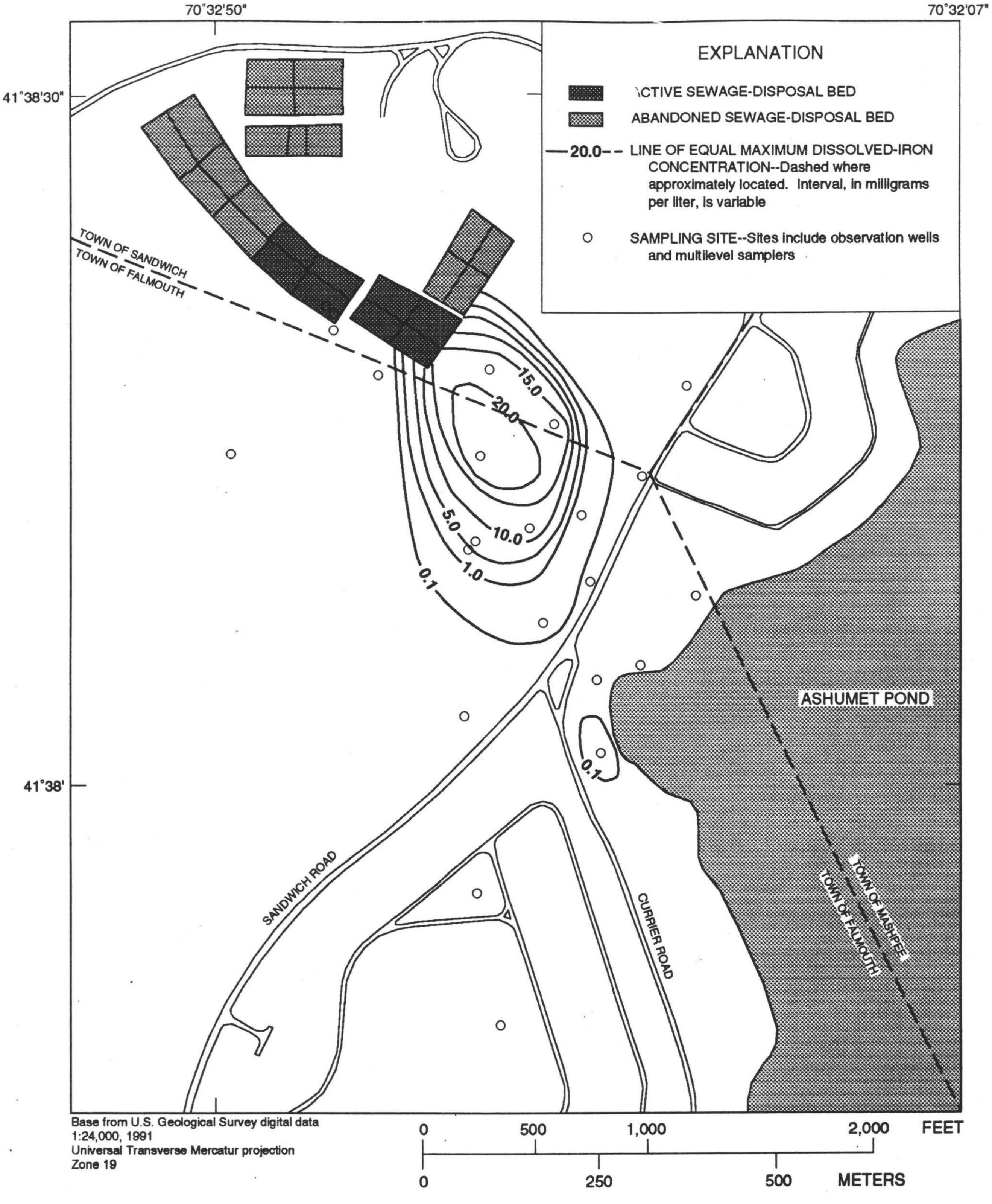


Figure 16. Vertical distribution of minimum dissolved-oxygen concentrations in ground water along geochemical sections *A-A′*, *B-B′*, and C-C′ near Ashumet Pond, Massachusetts, August through November 1993. (Location of geochemical sections are shown in fig. 3.)



Base from U.S. Geological Survey digital data
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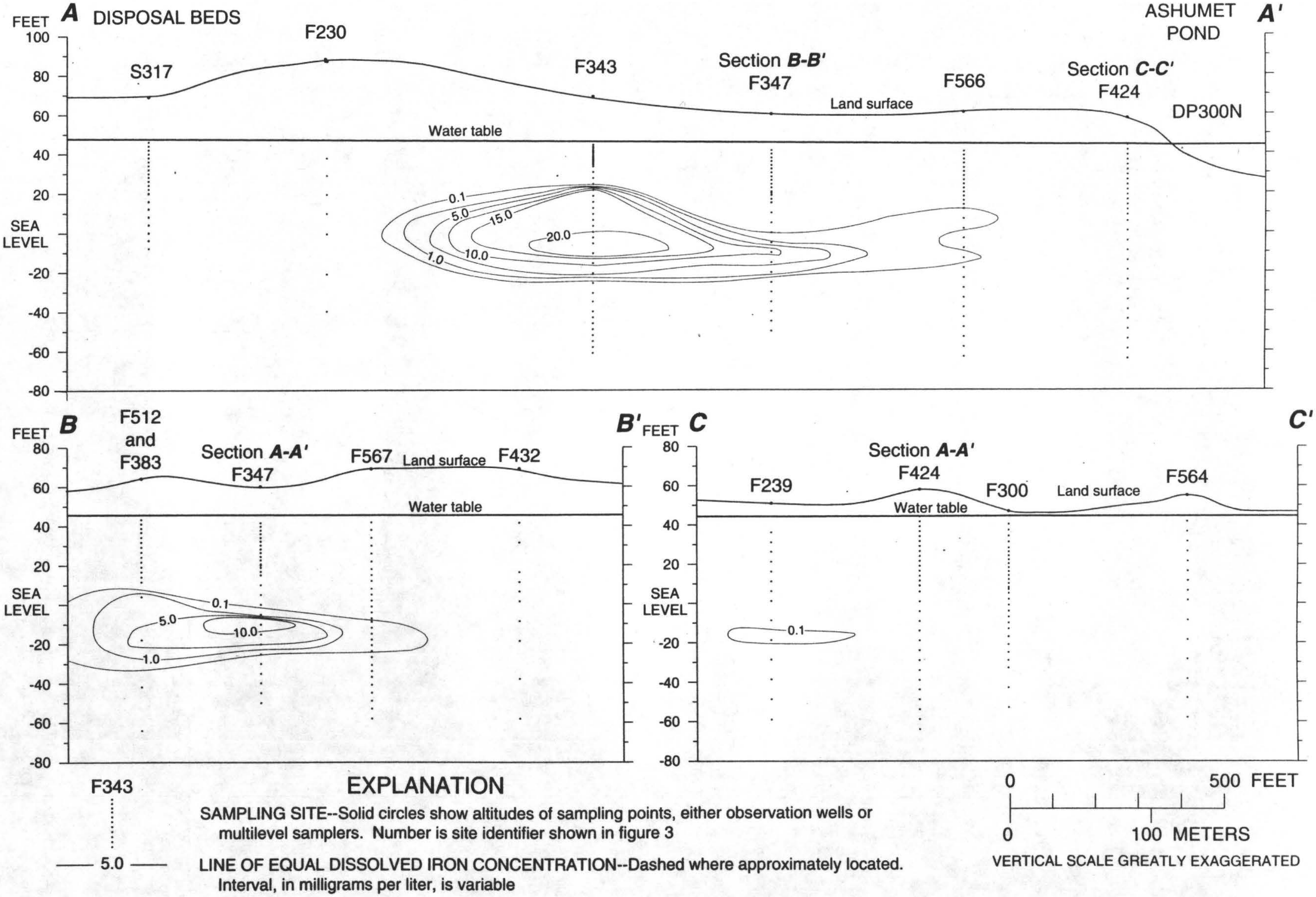
Universal Transverse Mercatur projection
Zone 19

Figure 17. Areal distribution of maximum dissolved-iron concentrations in ground water near Ashumet Pond, Massachusetts, August through November 1993.



EXPLANATION

SAMPLING SITE--Solid circles show altitudes of sampling points, either observation wells or multilevel samplers. Number is site identifier shown in figure 3

LINE OF EQUAL DISSOLVED IRON CONCENTRATION--Dashed where approximately located. Interval, in milligrams per liter, is variable

Figure 18. Vertical distribution of dissolved-iron concentrations in ground water along geochemical sections *A-A'*, *B-B'*, and *C-C'* near Ashumet Pond, Massachusetts, August through November 1993. (Location of geochemical sections are shown in fig. 3.)

Manganese, which has a chemistry similar to iron, is mobilized in ground water through the reductive dissolution of manganese oxyhydroxides but at higher redox potentials than iron (Hem, 1992). As conditions become more reducing, manganese tends to mobilize sooner, and as conditions become more oxidizing, manganese tends to remain in solution longer (Stumm and Morgan, 1981). The tendency for manganese to remain dissolved under suboxic conditions and at higher pH values than iron results in a more extensive plume of dissolved manganese. Background concentrations of manganese in the sand and gravel aquifer typically are less than 0.01 mg/L. A zone of ground water with manganese concentrations greater than 0.10 mg/L extends more than 3,600 ft downgradient of the sewage-disposal beds; the area with manganese concentrations greater than 1.0 mg/L extends about 2,900 ft downgradient (fig. 19). A zone of ground water with dissolved manganese greater than 10 mg/L extends from 1,850 to 2,200 ft downgradient of the disposal beds. The highest concentration (12.0 mg/L) was along Ashumet Pond. Dissolved manganese is higher along the pond and at downgradient sites than at sites closer to the disposal beds. The lower dissolved-manganese concentrations near the disposal beds may be due to the depletion of manganese oxyhydroxides in the aquifer sediments; the tendency of manganese oxyhydroxides to dissolve at higher redox potentials than iron oxyhydroxides suggests that dissolution of manganese coatings has been occurring for a longer period of time than dissolution of iron coatings. The vertical distribution of manganese is shown in figure 20. The zone of ground water containing greater than 1.0 mg/L dissolved manganese ranges in thickness from about 6 ft near the eastern boundary of the sewage plume to about 80 ft along Ashumet Pond. Dissolved-manganese concentrations were less than 1.0 mg/L in samples from beneath the disposal beds and in the core of the anoxic zone.

Phosphorus

Phosphorus is a common constituent of sewage effluent, and sewage-contaminated ground water can contain high concentrations of phosphorus (LeBlanc, 1984). Phosphorus mobility is retarded in ground water by the processes of coprecipitation and adsorption onto aluminum, iron, and manganese oxyhydroxides (Hem, 1992). Dissolved phosphorus in ground water is present in the form of orthophosphate. Phosphorus concentrations used in this report are for orthophosphate reported in milligrams per liter as phosphorus. Background phosphorus concentrations in the aquifer typically are less than 0.05 mg/L. The distribution of dissolved phosphorus in the aquifer is affected by interaction with the aquifer sediments and by changes in disposal-bed loading at the wastewater-treatment facility.

Phosphorus concentrations in the ground water exceeded 0.05 mg/L for a distance of 3,250 ft, exceeded 0.10 mg/L for a distance of about 2,500 ft, and exceeded 1.0 mg/L for a distance of 2,200 ft downgradient of the active disposal beds (fig. 21). The highest phosphorus concentration was 11 mg/L at a distance of 100 ft downgradient of the four westernmost active disposal beds; phosphorus concentrations in this area exceeded 5.0 mg/L for a distance of about 350 ft downgradient of the beds. The effect of disposal-bed loading on phosphorus distribution is shown by the low concentrations just downgradient of the eastern and western abandoned disposal beds. The highest phosphorus concentration observed at significant distances downgradient of the disposal beds was 6.2 mg/L at site F567. Site F567, which is near the eastern boundary of the sewage plume, is located about 1,350 ft downgradient of the beds and is in an area where phosphorus concentrations typically are greater than 3.0 mg/L. This area of high phosphorus concentrations is about 1,150 ft long, about 450 ft wide, and is about 750 ft upgradient of Ashumet Pond. The vertical distribution of phosphorus in the aquifer is shown in figure 22. Maximum phosphorus concentrations from the active disposal beds to the pond, decreased from 4.4 mg/L beneath the beds to 1.3 mg/L near the pond; concentrations in the center of the sewage plume ranged from 3.0 to 3.5 mg/L (fig. 22). The zone of ground water containing greater than 1.0 mg/L of phosphorus ranged in thickness from about 10 ft along the pond to 83 ft at site F567 near the eastern boundary of the sewage plume. Concentrations at site F567 exceeded 4.4 mg/L, which was the highest concentration beneath the active disposal beds, in a zone that was about 26 ft thick. The zone of ground water with phosphorus concentrations greater than 1.0 mg/L was about 45 ft thick beneath the disposal beds. Along the shore of the pond, the zone of ground water with phosphorus concentrations greater than 1.0 mg/L was about 700 ft wide and ranged in thickness from about 10 to about 25 ft. Maximum phosphorus concentrations along the pond ranged from 1.7 to 1.3 mg/L.

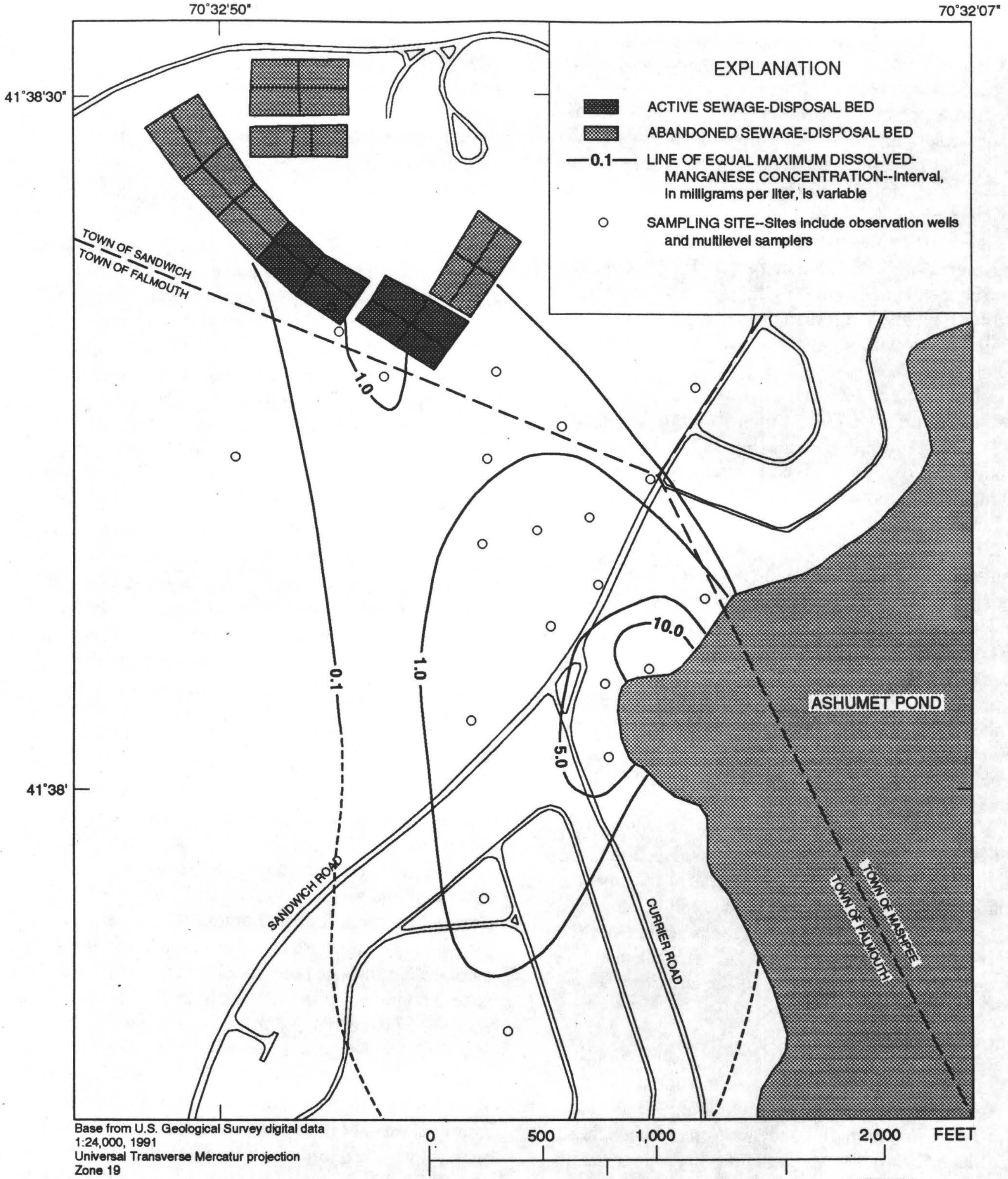
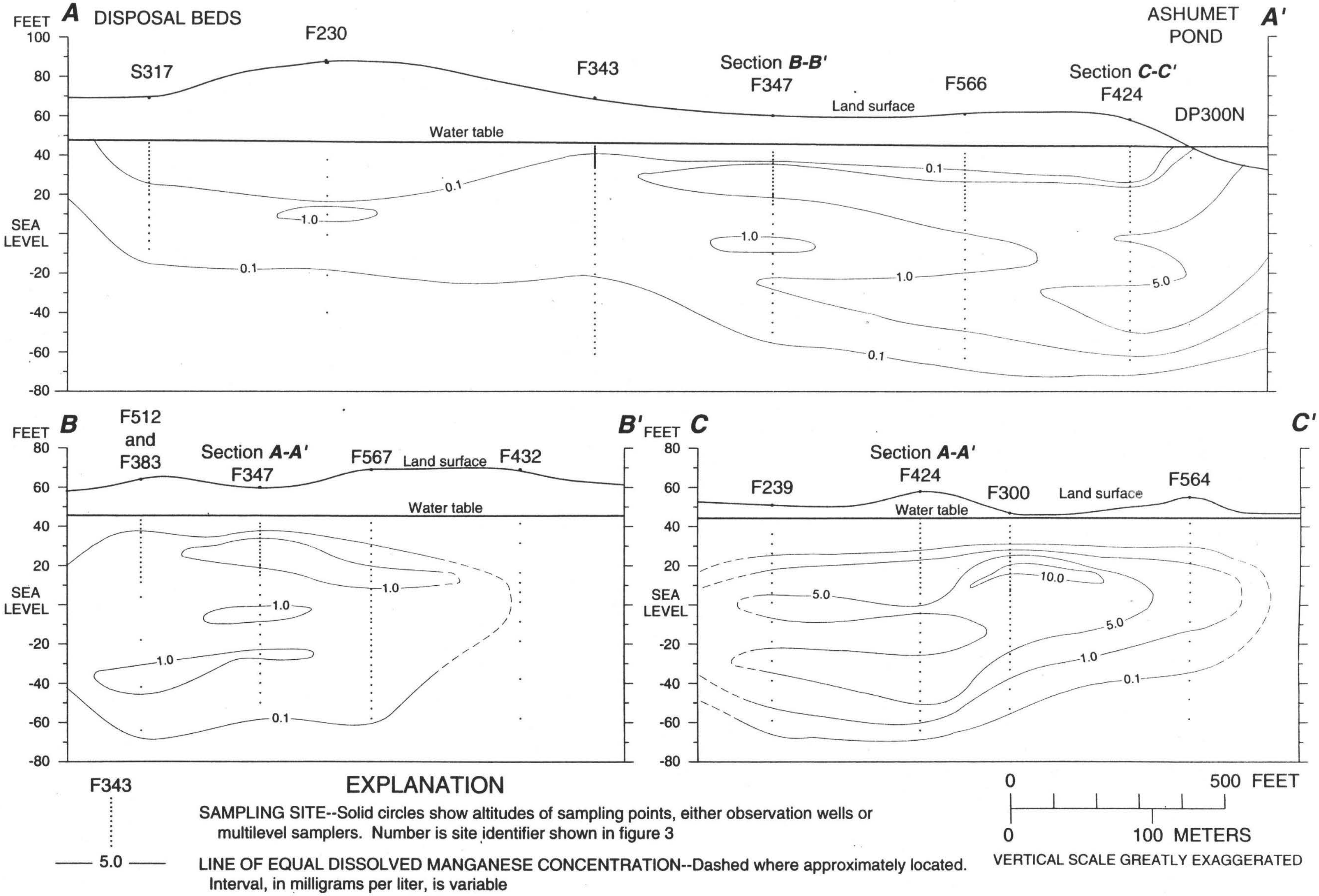


Figure 19. Areal distribution of maximum dissolved-manganese concentrations in ground water near Ashumet Pond, Massachusetts, August through November 1993.

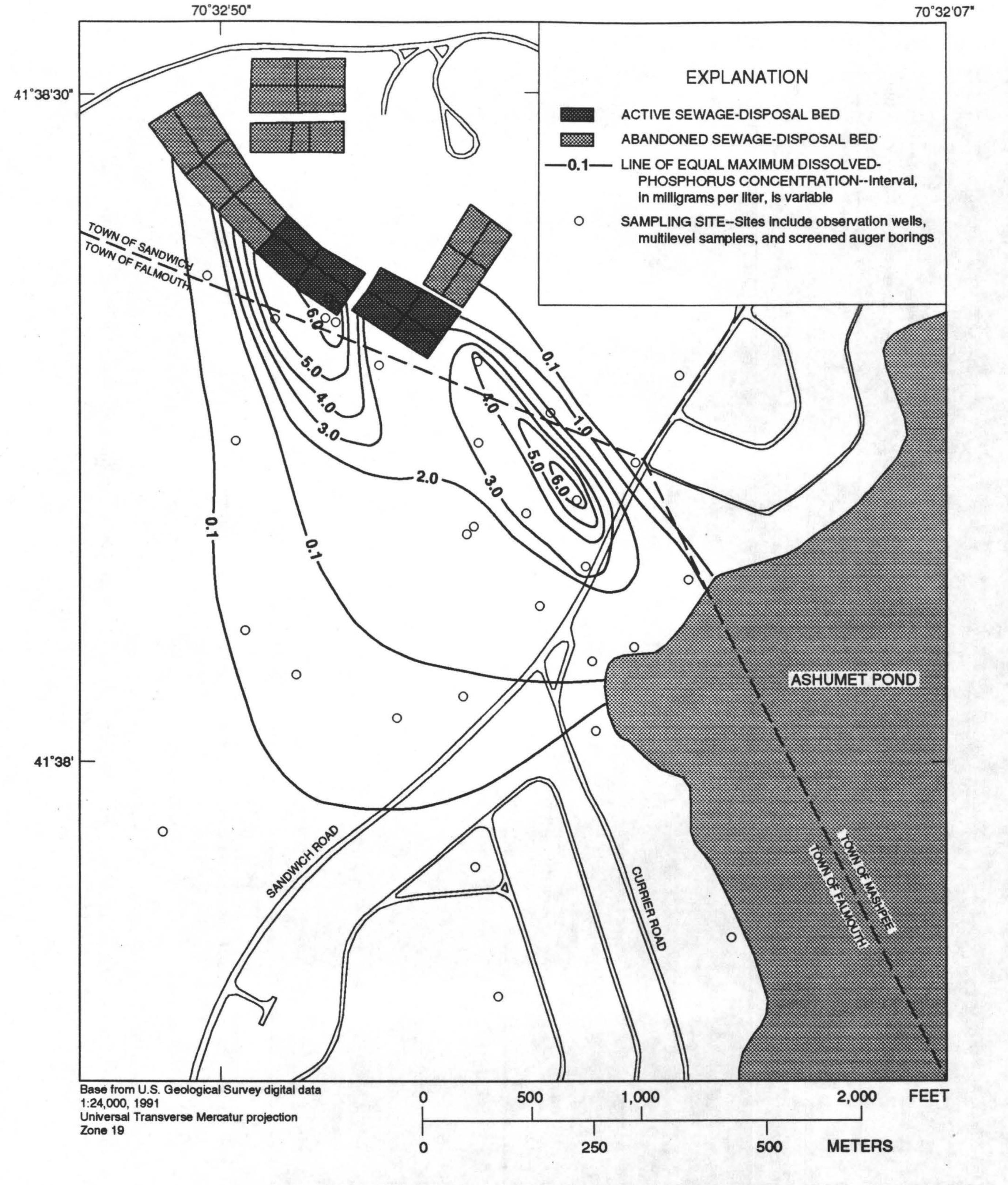


EXPLANATION

F343 SAMPLING SITE--Solid circles show altitudes of sampling points, either observation wells or multilevel samplers. Number is site identifier shown in figure 3

5.0 LINE OF EQUAL DISSOLVED MANGANESE CONCENTRATION--Dashed where approximately located. Interval, in milligrams per liter, is variable

Figure 20. Vertical distribution of dissolved-manganese concentrations in ground water along geochemical sections *A-A′*, *B-B′*, and *C-C′* near Ashumet Pond, Massachusetts, August through November 1993. (Location of geochemical sections are shown in fig. 3.)



Base from U.S. Geological Survey digital data
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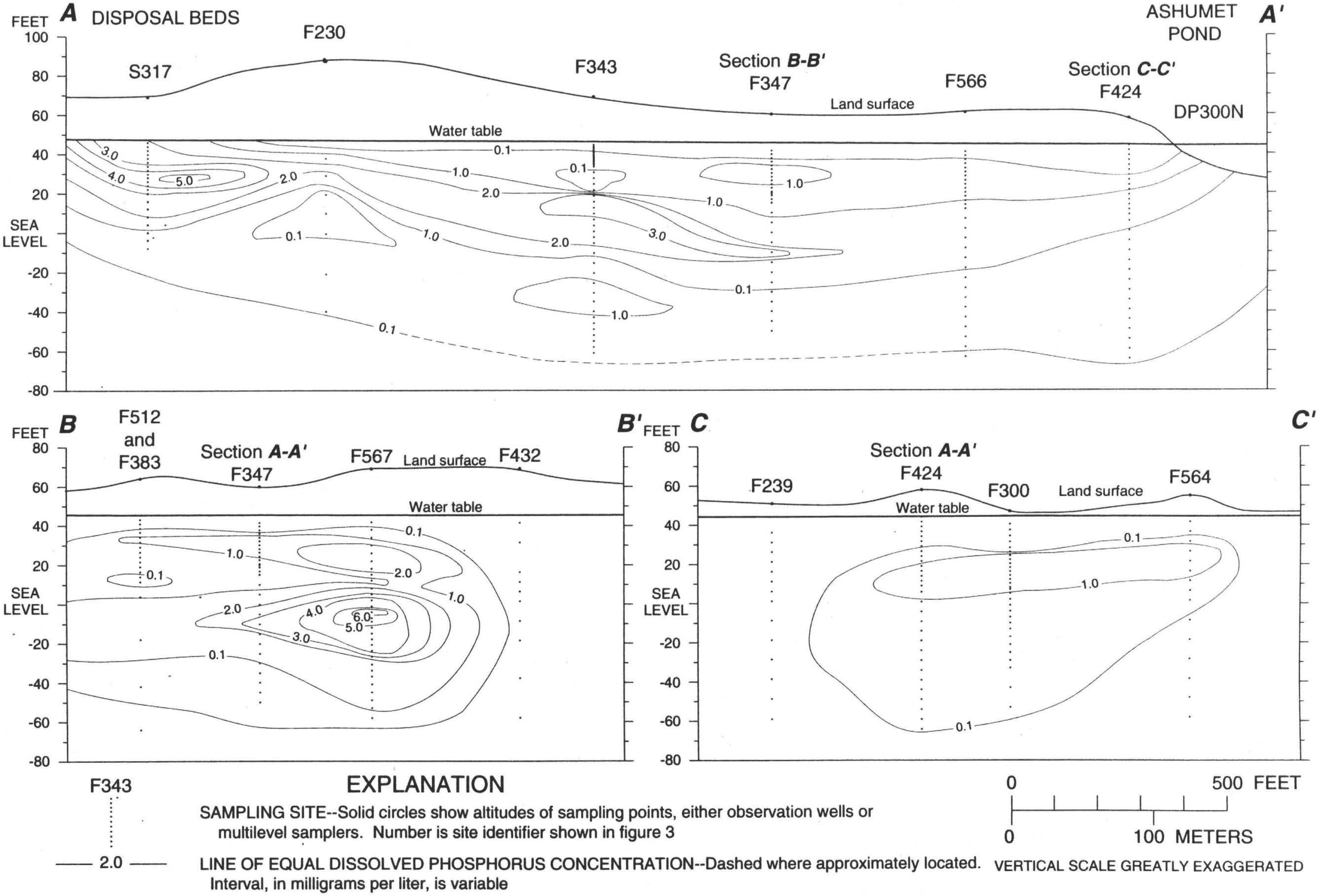
Figure 21. Areal distribution of maximum dissolved-phosphorus concentrations in ground water near Ashumet Pond, Massachusetts, August to November 1993.



EXPLANATION

SAMPLING SITE--Solid circles show altitudes of sampling points, either observation wells or multilevel samplers. Number is site identifier shown in figure 3

— 2.0 — LINE OF EQUAL DISSOLVED PHOSPHORUS CONCENTRATION--Dashed where approximately located. Interval, in milligrams per liter, is variable

Figure 22. Vertical distribution of dissolved-phosphorus concentrations in ground water along geochemical sections *A-A′*, *B-B′*, and *C-C′* near Ashumet Pond, Massachusetts, August through November 1993. (Location of geochemical sections are shown in fig. 3.)

GEOCHEMISTRY OF PHOSPHORUS ON AQUIFER SEDIMENTS

Water-quality data indicate that the distribution of dissolved phosphorus in the sewage plume is limited in extent as compared to more conservative constituents such as specific conductance. LeBlanc (1984) found that phosphorus in the plume had been transported about one-fifth as far as chloride. Phosphorus in ground water readily sorbs onto mineral surfaces with high zero points of charge (ZPC), such as iron and aluminum-oxides (Stumm and Morgan, 1981). Previous studies have shown that dissolved phosphorus is strongly adsorbed to metal-oxide coatings on aquifer sediments, making phosphorus relatively immobile in ground water (Russell, 1973; Goldberg and Sposito, 1984a, 1984b). Davis and others (1993) found that sediments in the study area contained extensive iron- and aluminum-oxide coatings; the surfaces of iron and aluminum coatings are an effective sorption substrate for phosphorus (Hem, 1992). A series of experiments, including sorption and desorption rate experiments, equilibrium isotherms, core extractions, and column experiments, were used to evaluate the geochemistry of phosphorus on aquifer sediments. The laboratory data were used to develop an improved understanding of the processes of sorption and desorption of phosphorus in aquifer sediments and to assess the role of reversible phosphorus sorption in the transport of phosphorus in the sewage plume.

Phosphorus Sorption onto Aquifer Sediments

Batch Experiments

Batch experiments were conducted to evaluate the rate and extent of phosphorus sorption onto uncontaminated aquifer sediments; a detailed discussion of the experimental design is given in the section "Methods of Investigation." Changes in dissolved phosphorus with time are shown in figure 23; the difference between the phosphorus concentrations at a given time and the initial phosphorus-solution concentrations of 31 and 3.1 mg/L represents the amount of phosphorus sorbed onto the sediments. At an initial phosphorus concentration of 31 mg/L, the plot of concentration and time had two different slopes, indicating two sorption processes. The data show that phosphorus sorption consists of an initial rapid sorption process that was complete in about 48 hours and a slower sorption process that did not reach equilibrium in 10 days. Similar trends have been reported for various substrates including pure mineral phases. The rapid sorption process may represent the adsorption of phosphorus at a discrete set of surface sorption sites on the aquifer material. This phosphorus may represent phosphorus bound by the formation of bridging complexes. The slow sorption process is poorly understood; various authors attribute it to such processes as surface precipitation, incorporation of phosphorus into recrystallizing solids such as iron and aluminum oxyhydroxides, or diffusion into porous adsorbents (Davis and Hem, 1989; Davis and Kent, 1990).

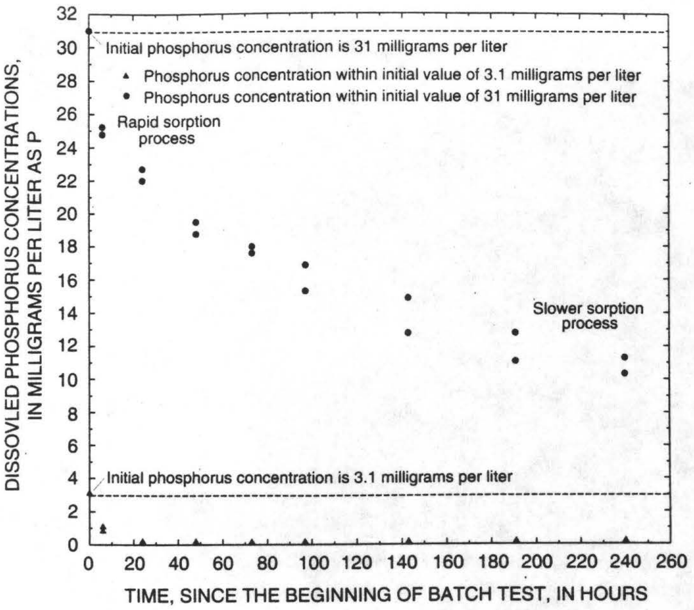


Figure 23. Dissolved-phosphorus concentrations and time in uncontaminated-sediment batch tests using initial solutions with phosphorus concentrations of 3.1 and 31 milligrams per liter.

At the lower initial phosphorus concentration of 3.1 mg/L, which is similar to concentrations measured in the sewage plume, all the dissolved phosphorus was completely sorbed, indicating a high affinity of the sediments for phosphorus. The data indicate that the sorption of phosphorus at the lower concentration was complete within 24 hours. The data indicate that uncontaminated aquifer sediments initially have the capacity to completely sorb all or most available dissolved phosphorus at concentrations similar to those measured in the sewage effluent; phosphours in the sewage effluent is typically between 6 and 7 mg/L as P.

Phosphorus adsorption isotherms were developed for uncontaminated and contaminated sediments (fig. 24); a detailed discussion of the experiment design is given in the section "Methods of Investigation." When phosphorus is introduced into aquifer sediments under constant chemical and hydrologic conditions, the sediments will sorb an amount of phosphorus that is in equilibrium with the influent concentration. Although the direct application of an isotherm to determine phosphorus behavior in the aquifer is difficult, isotherms can be useful in evaluating phosphorus sorption under a specific set of experimental conditions. The shape of an isotherm is determined by the density of available sorption sites, pH, and the concentrations of competing ions.

The data indicate that phosphorus sorption onto uncontaminated and contaminated sediments is nonlinear for the concentration range used in the experiment. The slopes of both isotherms decrease with increasing dissolved phosphorus concentrations, indicating that the ratio of sorbed to dissolved phosphorus decreases with increasing dissolved concentrations. The isotherm for uncontaminated sediments plots higher than the isotherm for the contaminated sediments, indicating that uncontaminated sediments have a greater capacity to sorb phosphorus than previously contaminated sediments. The presence of fewer available phosphorus sorption sites on sediments previously contaminated with phosphorus diminishes the capacity of contaminated sediments to sorb additional phosphorus. Data indicate that as phosphorus is continuously loaded onto aquifer sediments, the shape of the adsorption isotherm at successive times will become less steep and that the slope will approach zero; this reflects a reduction in the capacity of more contaminated sediments to sorb additional phosphorus. The data also indicate that after prolonged loading of phosphorus, aquifer sediments can become saturated with phosphorus, resulting in a loss of their capacity to sorb any additional phosphorus.

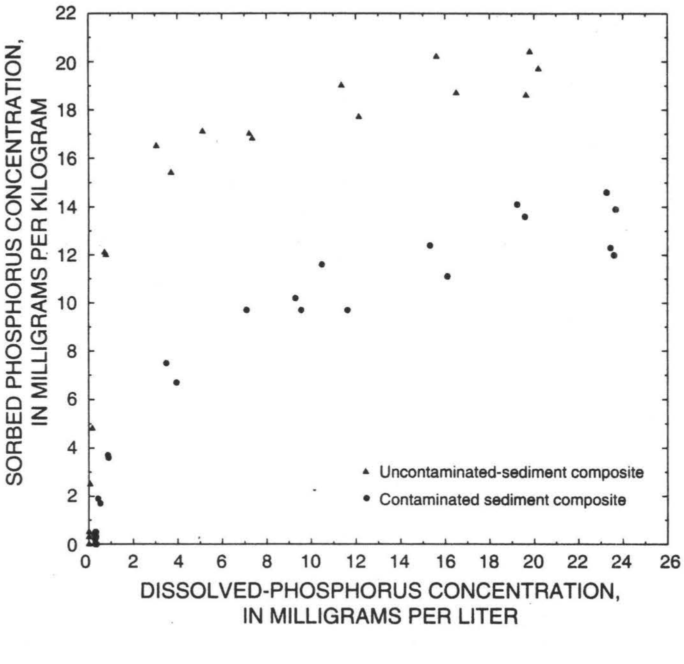


Figure 24. Phosphorus sorption isotherms for uncontaminated- and contaminated-sediment composites.

The data indicate that phosphorus may become more mobile in the ground water as the capacity of the sediments to sorb additional phosphorus decreases. The presence of dissolved phosphorus in ground water, which has been transported more than 2,500 ft downgradient of the sewage-disposal beds, suggests that the continued disposal of phosphorus at the wastewater-treatment plant since 1936 has diminished the capacity of aquifer sediments to sorb additional phosphorus and has allowed phosphorus to become mobile in the ground water. The slow sorption process seen in the rate experiment could allow sorbed phosphorus concentrations to increase beyond the equilibrium values shown in the adsorption isotherm. Spatial variability in sorption properties of the aquifer solids also may result in variation of the shape of the adsorption isotherms. The sorptive properties of aquifer sediments from cores previously collected from the site with respect to lead and zinc varied by more than a factor of five (Davis and others, 1993). The phosphorus adsorption isotherms were determined on a large composite sample in order to minimize this source of variability.

Core Extractions

The high affinity of the aquifer sediments for phosphorus and the continued disposal of phosphorus at the wastewater-treatment plant have resulted in a large reservoir of sorbed phosphorus in the aquifer. Extractions of core material were used to determine the total amount of extractable phosphorus sorbed onto the aquifer sediments at nine locations. Phosphorus extractions were performed using 1 N HCl; a detailed discussion of the extraction methodology is given in the section "Methods of Investigation."

The vertical distribution of extractable phosphorus from uncontaminated sediment cores collected from a location upgradient of the sewage-disposal beds (sampling site S467) is shown in figure 25. Extractable phosphorus in a sample from the site ranged from 0.014 to 0.24 mg/g, averaged 0.019 mg/g, and had a standard deviation of 0.003 mg/g. These values were assumed to represent an approximate range for background concentrations of extractable phosphorus in the aquifer. This phosphorus probably is derived from the partial dissolution of accessory minerals in the aquifer material, such as apatite $[Ca_5(F, OH, Cl)_3 (PO_4)_3]$, by the 1N hydrochloric acid used as an extractant.

The vertical distributions of extractable phosphorus in samples from three sites in the sewage-disposal beds also are included in figure 25. Concentrations of extractable phosphorus were highest in saturated sediments from beneath the abandoned disposal beds at sampling site S440. Extractable phosphorus in samples from this site ranged from 0.051 to 0.152 mg/g of sand and remained fairly constant with depth. Extractable phosphorus in cores from sampling site S318, which is in the active disposal beds, ranged from 0.024 to 0.100 mg/g. Extractable phosphorus in samples from the site generally decreased with depth and reached background concentrations near the bottom of the cored interval. Cores taken from the unsaturated zone at site S317, also in the active disposal beds and located adjacent to site S318, yielded extractable phosphorus that ranged from 0.085 to 0.416 mg/g. These data indicate an additional reservoir of phosphorus in the unsaturated zone. Concentrations of extractable phosphorus in water-saturated core material from beneath the abandoned disposal beds (sampling site S440) were generally higher and phosphorus contamination extended to a greater depth than in samples from beneath the currently active beds (sites S318 and S317). This may be related to the four abandoned beds near site S440 that were used exclusively for a long period of time compared to the eight beds that are currently used for shorter periods of time. Using fewer beds for a longer period of time also might explain why phosphorus contamination is deeper at site S440 than at site S318. Greater vertical transport due to the conditional site-saturation effects, discussed in the previous section, and the greater vertical flow component from more concentrated mounding associated with more concentrated disposal could allow phosphorus to be transported deeper into the aquifer. The vertical distribution of desorbed phosphorus in samples from site S318 that were exposed to uncontaminated ground water also is included in figure 25; comparison between the two extraction results are presented in the next section, "Batch Phosphorus Desorption from Aquifer Sediments."

The vertical distributions of extractable phosphorus in samples from four sites downgradient of the sewage-disposal beds are shown in figure 26. Extractable phosphorus was highest in core samples at site F230, about 300 ft downgradient of the disposal beds. Extractable phosphorus at the site ranged from 0.055 to 0.150 mg/g. Extractable phosphorus exceeded 0.100 mg/g from an altitude of 42.5 to 20.7 ft above sea level. Extractable phosphorus concentrations in samples from site F230 were higher than those from site S318, possibly due to the closer proximity of site F230 to the center of historical contamination. Extractable phosphorus in samples from site F415 ranged from 0.022 to 0.046 mg/g, and extractable phosphorus in samples from site F343 ranged from 0.018 to 0.026 mg/g. The amount of extractable phosphorus in cores from the anoxic zone at site F343 was lower than in cores from the suboxic zone, but phosphorus extracted from cores from the same depth at site F415, about 20 ft away, showed no difference between the two zones. This suggests that the differences may be due to sample variability. Extractable phosphorus from cores collected at site F567, near the eastern boundary of the sewage plume, ranged from 0.039 to 0.018 mg/g. Concentrations of extractable phosphorus in cores from along the edge of Ashumet Pond, at site F424, ranged from 0.022 to 0.046 mg/g of sand.

Concentrations of extractable phosphorus were compared to dissolved concentrations in samples from adjacent multilevel samplers at six sites in that part of the sewage plume that had dissolved-phosphorus concentrations greater than 1.0 mg/L (fig. 27). A porosity of 0.39 and a sediment density of 2.74 g/cm^3 were used to convert aqueous and solid-phase concentrations of phosphorus to total concentrations in the aquifer. Sediment density was used to convert milligrams per gram of sand to milligrams per liter of

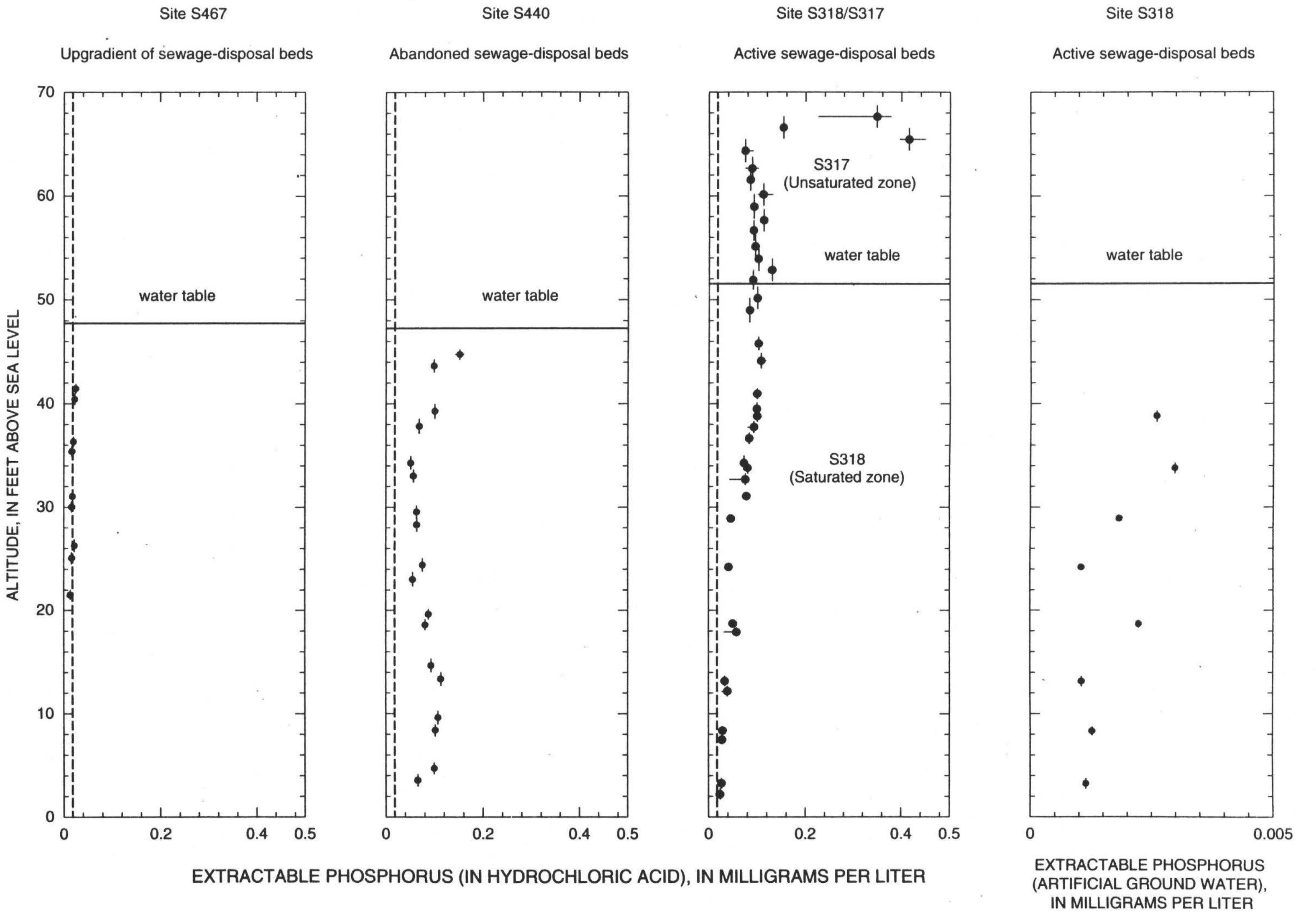


Figure 25. Vertical distributions of extractable phosphorus using 1 N hydrochloric acid (HCl) at three sites in and near the Massachusetts Military Reservation sewage-treatment plant disposal beds, Massachusetts, June 1991 through August 1993. Extractions were done using 1 N HCl and artificial ground water at site S318. Vertical error bars represent cored interval. Horizontal error bars represent variability among replicate extractions. Dashed line represents background extractable phosphorus concentrations collected from extractions at upgradient sampling site S467.

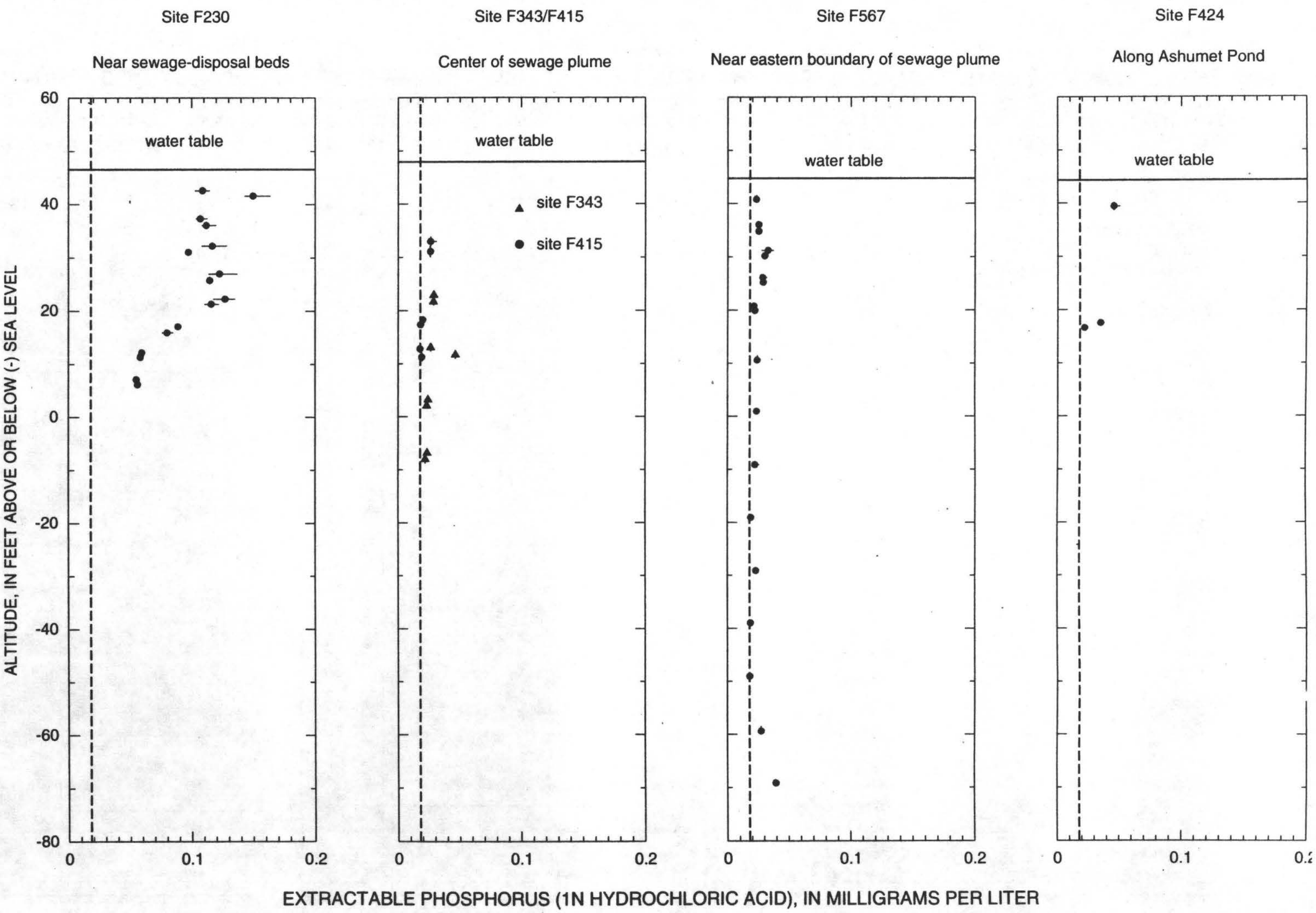


Figure 26. Vertical distributions of extractable phosphorus in samples at four sites downgradient of the sewage-disposal beds near Ashumet Pond, Massachusetts, September 1992 through August 1993. Vertical error bars represent cored interval. Horizontal error bars represent variability among replicate extractions. Dashed line represents background extractable phosphorus concentrations at upgradient site S467.

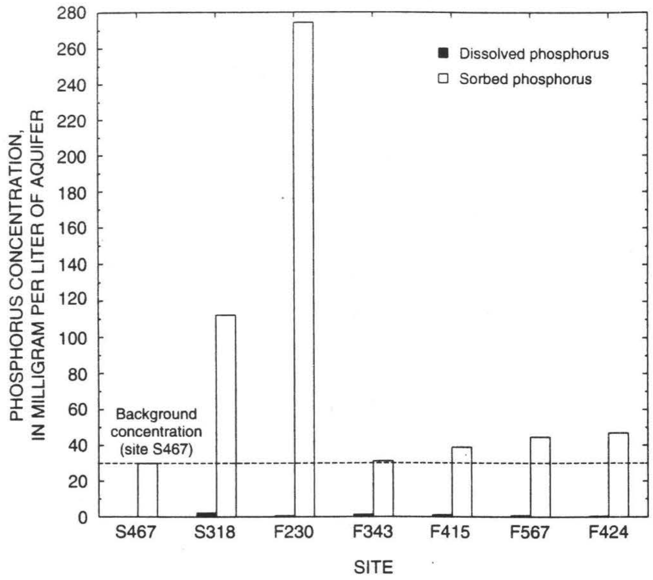


Figure 27. Sorbed and dissolved-phosphorus concentrations in samples from six sites within the sewage plume and one site upgradient of the plume (site S467) near Ashumet Pond, Massachusetts. Core extractions from sampling site S457 represent background concentrations.

sand; the porosity then was used to convert milligrams per liter of sand and milligrams per liter of water to milligrams per liter of aquifer for sorbed and dissolved phosphorus. The data indicate that a large reservoir of sorbed phosphorus exists in the aquifer and that solid-phase phosphorus concentrations are much higher than concentrations in the ground water. The mean ratio of sorbed to dissolved phosphorus in samples from six sites ranged from 25:1 at site F343 to 410:1 at site F230. The ratio in samples from beneath the active disposal beds, at site S318, was 55:1. The ratios in samples from sites F415, F567, and F424 were 37:1, 53:1, and 104:1, respectively. The data indicate that the ratio in samples from sites F343 and F415, which were in the anoxic, dissolved-iron zone were significantly lower than the ratio in samples from the other sites, which were in the suboxic part of the aquifer. The lower ratio in samples from the anoxic zone indicate that more phosphorus was dissolved in the ground water relative to total concentrations in the aquifer. This suggests that phosphorus sorption may be less extensive and that phosphorus may be more mobile in the anoxic zone. The greater mobility of phosphorus in the anoxic zone may be due to the lesser amount of iron oxyhydroxides on the sediment surfaces; a lesser amount of iron oxyhydroxides would result in fewer available sorption sites for phosphorus.

Extractable phosphorus concentrations in the cores were considerably different from concentrations determined from the adsorption isotherm. The measured concentrations exceeded the estimated concentrations at some sites and were less than equilibrium concentrations at other sites. The lack of agreement is attributed to the natural complexities in the aquifer; the shape of the adsorption isotherm at any given location is affected by the number of adsorption sites, pH, loading history, and the concentrations of competing ions and, therefore, would be expected to vary considerably. Natural variability in the capacity of aquifer sediments to sorb solutes is an important factor that can complicate direct application of an equilibrium isotherm (Robin and others, 1991; Davis and others, 1993). Davis and others (1993) found that the sorptive properties of sediment cores collected at the site, with respect to lead and zinc, varied over a factor of five.

Batch Phosphorus Desorption from Aquifer Sediments

The large reservoir of sorbed phosphorus in the aquifer could remain an important source of phosphorus contamination after sewage disposal is ceased in December 1995. Although 1N HCl extractions are useful for estimating the total reservoir of extractable phosphorus in the aquifer sediments, phosphorus in the aquifer may be removed by natural flushing with uncontaminated ground water, which is much less acidic than 1N HCl. A series of batch experiments were designed to determine the rate and extent of phosphorus removal under natural conditions. Artificial ground water, with a similar chemical composition as uncontaminated ground water in the study area was used to desorb phosphorus from contaminated sediments. Eight vertical core subsections from site S318, which is beneath the active disposal beds, were used in the experiments. A detailed discussion of the methods used in the experiments are given in the section "Methods of Investigation."

Cumulative changes in dissolved phosphorus with time in the eight core subsections are shown in figure 28. Phosphorus initially was released quickly followed by a slower, steady release of phosphorus from aquifer sediments. This is consistent with the rate experiments; the initial part of the desorption curves represent the rapid desorption of loosely bound phosphorus, whereas the later part of the curves corresponds to the desorption of a reservoir of sorbed phosphorus that is more strongly bound. The desorption curves show that the reservoir of phosphorus that is loosely bound to the sediments decreases with depth, indicating that the aquifer sediments near the water table have the largest reservoir of rapidly desorbable phosphorus. Rapidly desorbable phosphorus may represent phosphorus bound by bridging complexes and may not be present in deeper cores due to the predominance of stronger, monodentate-bonding mechanisms, where less phosphorus has been added to the sediments. Core-extraction data from sampling site S318 (fig. 25) confirm that the amount of extractable phosphorus decreases with depth at this site.

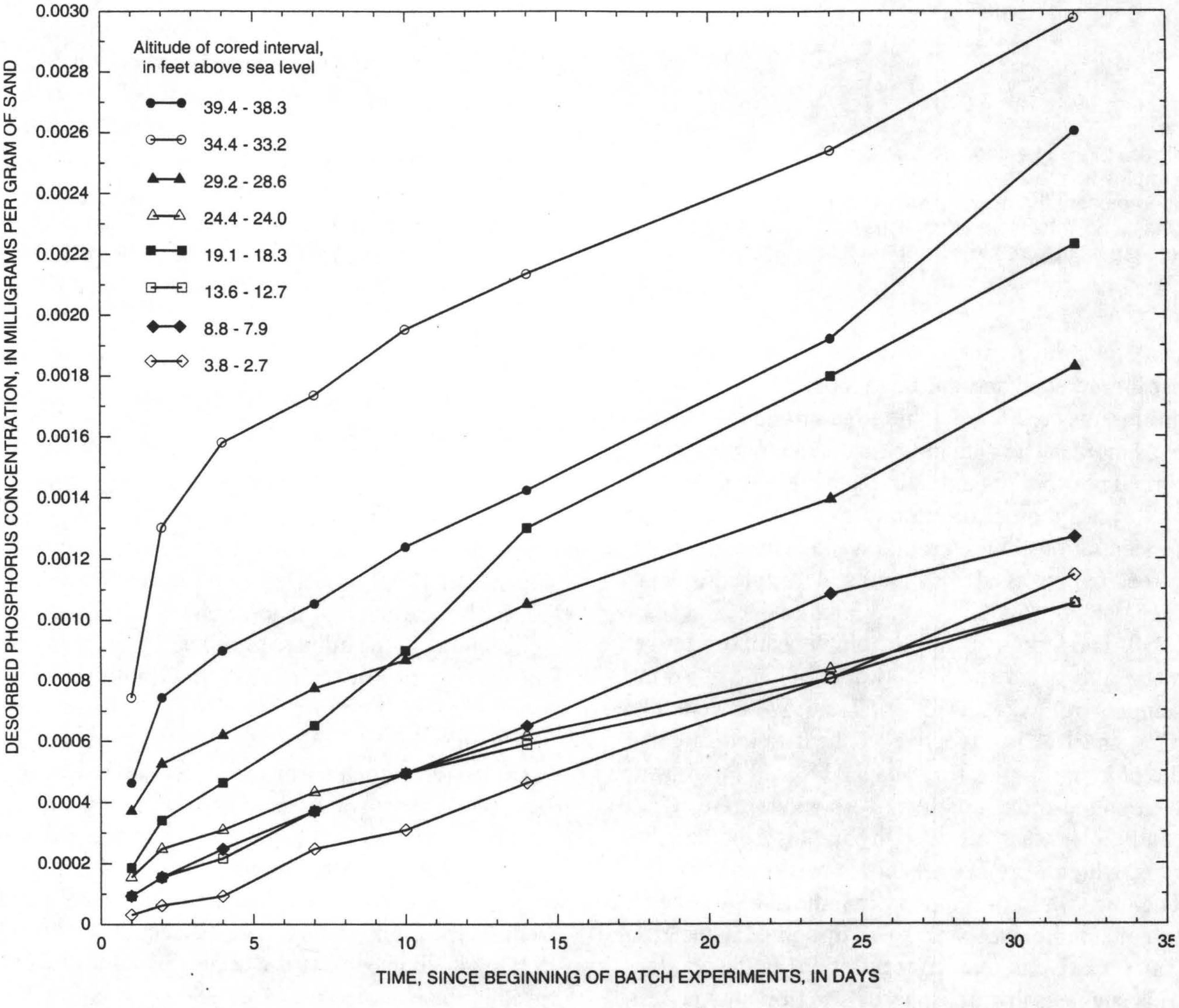


Figure 28. Cumulative changes in desorbed-phosphorus concentrations and time in artificial ground water for eight core sections from beneath active sewage-disposal beds (site S318), Massachusetts Military Reservation near Ashumet Pond, Massachusetts.

Stollenwerk (1995) found that the amount of phosphorus sorbed onto sediment surfaces decreased with increasing pH. The rate and extent of phosphorus desorption in the aquifer may depend on the rate at which pH of the ground water decreases to that of uncontaminated ground water and the rate at which additional phosphorus is added to the aquifer. The data indicate that, as uncontaminated ground water moves through the aquifer after sewage disposal ceases, the reservoir of sorbed phosphorus that has accumulated during the 59 years of sewage disposal will continue to release phosphorus into the ground water for many years. The endpoints of the desorption curves are included in figure 25; these values represent desorbable phosphorus after 32 days of exposure to artfiicial ground water with the same chemical composition as uncontaminated ground water at the site. Comparison with the 1N HCl extractions shows that after 32 days, uncontaminated ground water desorbed between 2.6 and 4.3 percent of the total amount of phosphorus that was extracted with 1N HCl; the mean was 3.6 percent. The data indicate that the introduction of uncontaminated ground water into the aquifer will cause a slow and prolonged release of phosphorus from aquifer sediments.

Phosphorus Sorption and Desorption in Sediment Columns

The rate and extent of sorption and desorption processes and the effect of reversible sorption on phosphorus transport were evaluated under simple one-dimensional flow conditions using column experiments. Column experiments were done using uncontaminated and contaminated sediments to assess phosphorus transport in the aquifer under constant geochemical and hydrologic conditions and to determine how phosphorus transport may change following cessation of sewage disposal. A description of the experimental design is given in the section "Methods of Investigation."

Uncontaminated Sediments

Previous investigations have found that the concentration of dissolved phosphorus in ground water is a function of the contact time between ground water and sediment, indicating that phosphorus sorption is a rate-controlled physiochemical process (Barrow and Shaw, 1957; Ryden and others, 1977; Lijklema, 1980). Column experiments with uncontaminated sediment were done using influent phosphorus concentrations of 0.49 and 6.1 mg/L; these concentrations bracket most of the phosphorus concentrations measured in the sewage plume.

Phosphorus

Changes in phosphorus concentration in column effluent with pore volume for the two uncontaminated-sediment columns—referred to as phosphorus breakthrough curves—are shown in figure 29. A pore volume refers to the volume of pore water in the column at any one time. The asymmetric shape of the breakthrough curves indicates that phosphorus sorption is rate controlled (James and Rubin, 1979; Grove and Stollenwerk, 1985). The pore volume where phosphorus was first detected in the column effluent and the rate at which effluent concentrations approached the influent concentration, indicating saturation of the sediments with phosphorus, were a function of the influent concentration and were more rapid at high phosphorus concentrations. For an initial influent concentration of 0.49 mg/L, phosphorus was detected in the effluent after 44 pore volumes (fig. 29*A*). When the influent phosphorus concentration was 6.1 mg/L, phosphorus was detected in the effluent after nine pore volumes (fig. 29*B*). The more rapid breakthrough in the latter column can be attributed to the capacity of the influent with higher phosphorus concentrations to more rapidly fill available sorption sites in the column. The data indicate that all of the phosphorus initially introduced into the uncontaminated sediments was sorbed and that phosphorus became mobile after available sorption sites became filled with phosphorus.

Effluent from both columns showed rapid increases in phosphorus following breakthrough followed by much slower increases at later pore volumes. For an influent concentration of 6.1 mg/L, the effluent concentration increased from 0 to 5.0 mg/L after six pore volumes (fig. 29*B*). An additional 25 pore volumes were required to reach the influent concentration of 6.1 mg/L, indicating saturation of available sorption sites; van der Zee and others (1989) observed similar phosphorus breakthrough curves in soil columns. At equilibrium, as indicated by equal effluent and influent concentrations, phosphorus moves through the column as a nonreactive solute. Phosphorus concentrations in the effluent in the

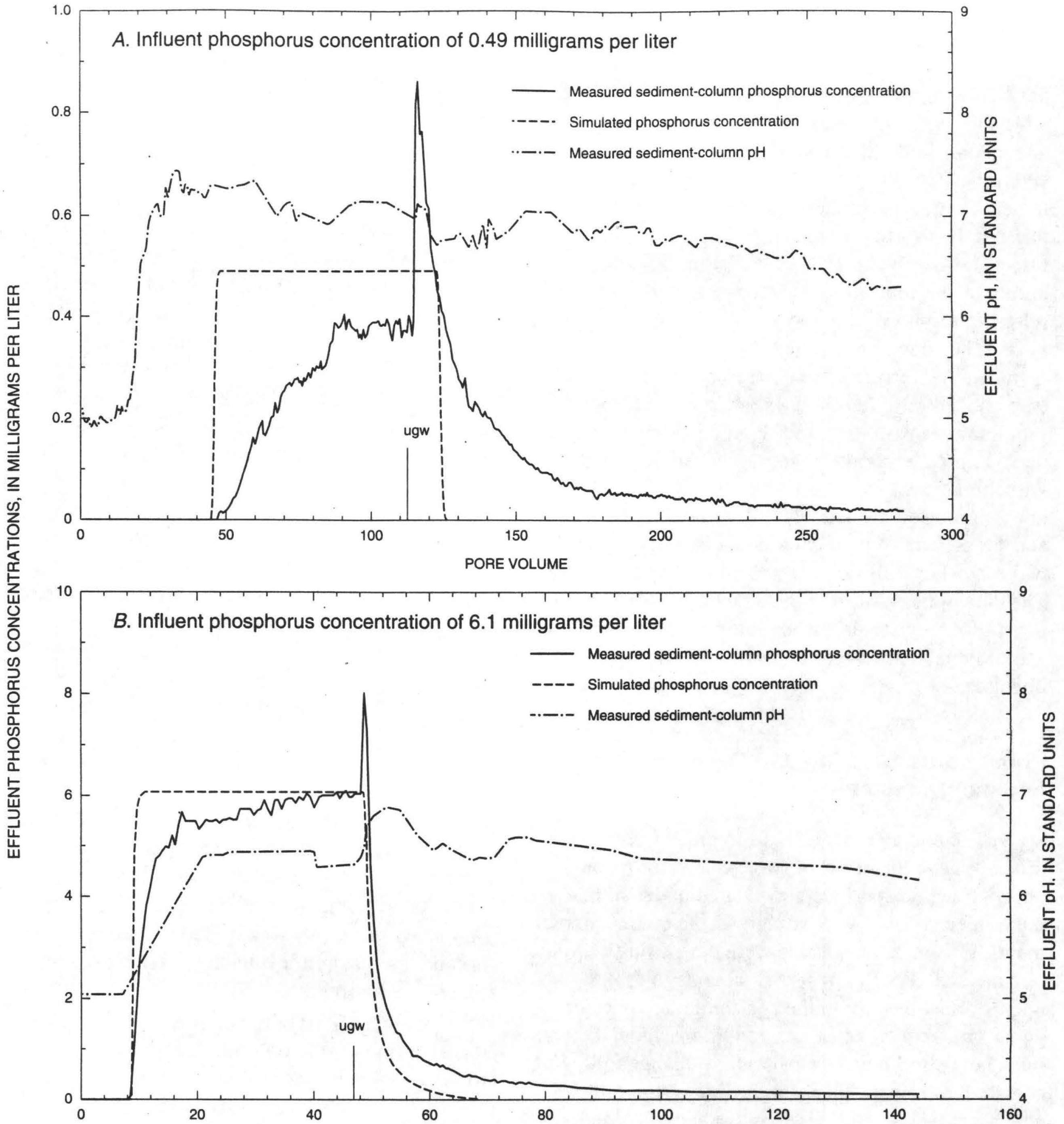


Figure 29. Changes in phosphorus and pH in uncontaminated sediment-column effluent using solutions with phosphorus concentrations of (*A*) 0.49 and (*B*) 6.1 milligrams per liter followed by uncontaminated ground water as influent, and simulated phosphorus effluent concentrations using the surface-complexation model in the solute-transport code HYTEQ. ("ugw" indicates point at which uncontaminated ground water was eluted through the columns.)

0.49-mg/L column never reached the influent concentration, even after 120 pore volumes (fig. 29*A*). This type of nonequilibrium transport of solutes through porous media occurs when ground-water velocities are rapid enough to prevent attainment of chemical and physical equilibria. The nonequilibrium condition observed at low influent phosphorus concentrations can be attributed to several factors. The rate at which phosphorus is sorbed is partly a function of the time required for phosphorus to diffuse to less-accessible sorption sites in areas of immobile water. Water can be immobile in a layer around grains, in dead-end pore spaces between tightly packed grains, or within crevices or pits on grain surfaces. If the diffusion rate is slow compared to the interstitial velocity of the ground water, nonequilibrium will result. Wood and others (1990) observed fractures in grains of feldspar and quartz from the sand and gravel aquifer. Davis and others (1993) observed thick coatings of hydrous metal oxides on aquifer sand in the study area. Diffusion of phosphorus to reaction sites within these fractures and oxide coatings could explain some of the slow rate of approach to equilibrium. The slow approach to equilibrium also could be explained by chemical reactions between phosphorus and metal oxides. Martin and others (1988) present evidence for the formation of crystallites of griphite [$Fe_3Mn_2\,(PO_4)_{2.5}\,(OH)_2$] on the surfaces of naturally occurring geothite. They concluded that a monolayer of phosphorus was initially adsorbed rapidly, followed by slow coprecipitation. In the absence of manganese, which was supplied by manganese impurities in the geothite, an iron-phosphate hydroxide mineral phase would form. Isenbeck-Schroter and others (1993) proposed precipitation of variscite ($AlPO_4$) to explain the slow rate of approach to complete breakthrough in their columns.

When influent was changed to uncontaminated ground water, simulating cessation of sewage disposal, the concentration of phosphorus in both columns increased sharply (fig. 29). Phosphorus concentrations in the 0.49-and 6.1-mg/L columns sharply increased to 0.82 and 8.03 mg/L, respectively, following the introduction of uncontaminated water. The initial increase in phosphorus can be attributed to the rapid desorption of more weakly bound phosphorus on sediment surfaces. Maximum concentrations in the 0.49-mg/L column were 1.7 times greater than influent concentrations, and the period where effluent concentrations remained greater than influent concentrations, was nine pore volumes. The peak in the 6.1-mg/L column had a maximum concentration that was 1.3 times greater than the influent concentration, and effluent concentrations exceeded influent concentrations for two pore volumes. The data indicate that the relative magnitude and duration of the rapid phosphorus desorption were smaller when phosphorus concentrations were higher. Following the rapid-desorption concentration peaks, the phosphorus concentrations in both columns rapidly decreased for several pore volumes; the rate of phosphorus desorption slowed significantly with time. The slow rate of phosphorus desorption at later pore volumes may be due either to the slow diffusion of phosphorus from immobile water or to the dissolution of a phosphate mineral phase. Phosphorus in the 0.49-mg/L column remained greater than 0.02 mg/L until pore volume 280; that is, after 160 pore volumes of uncontaminated water had moved through the column. Phosphorus concentrations in the 6.1-mg/L column decreased to a concentration of 0.09 mg/L at pore volume 144, or after about 100 pore volumes of uncontaminated ground water had moved through the column.

Simulated phosphorus breakthrough curves for the two uncontaminated-sediment columns also are included in figure 29. Phosphorus breakthrough was simulated with the diffuse-layer surface-complexation model in the solute-transport code HYTEQ; breakthrough refers to the pore volume where phosphorus is first detected in the effluent. Assymmetry of the breakthrough curves could not adequately be simulated because the model lacks an adequate diffusion mechanism and can not account for the formation of secondary phases. Model simulation indicated that sorbed phosphorus should desorb rapidly and that effluent concentrations would reach detection limits within a few pore volumes. Also, the rapid-desorption concentration peak in the effluent following the introduction of uncontaminated ground water was not simulated.

The amount of phosphorus sorbed to the sediments also was a function of influent concentration. During the sorptive part of the 0.49- and 6.1-mg/L experiments, a total of 0.007 and 0.015 mg/g of sand, respectively, were sorbed to the column sediments. Extractable phosphorus in cores from the suboxic zone of the aquifer, which ranged from 0.015 to 0.16 mg of phosphorus per gram of sand, generally was higher than those in the artificially contaminated column sediments. As was noted previously, some of the phosphorus extracted from core material may be derived from partial dissolution of secondary minerals. Some of the difference also may be due to the longer contact time between aquifer sediments and sewage-contaminated ground water; phosphorus in the sewage-contaminated ground water has been in contact with the aquifer for years, whereas the adsorptive phase of the column experiments lasted only a few weeks. More time would be available for rate-controlled chemical reactions to attain equilibrium in the aquifer. Following the introduction of uncontaminated water into the columns, 50 percent of the sorbed phosphorus in the 0.49-mg/L column and 55 percent of the sorbed phosphorus in the 6.1-mg/L column eventually desorbed from the sediments. Extrapolation of both desorption curves to the point where phosphorus concentrations decrease to the detection limit indicates that at least one-third of the initial reservoir of sorbed phosphorus will remain sorbed. This suggests either the presence of a strong sorption complex or the formation of a relatively insoluble mineral phase.

pH

Initially, the pH in the two uncontaminated-sediment columns was in equilibrium with uncontaminated ground water, with a pH of about 5.3. Column data indicate that as contaminated ground water moved through the columns, the pH remained at 5.3 for about 10 pore volumes then increased sharply to a pH of about 7.0, which was the pH of the influent (fig. 29). The pH remained at 7.0 for several pore volumes following the introduction of uncontaminated water before slowly beginning to decrease and was still greater than 6.0 at the end of the experiment. The data indicate that the aquifer sediments have the capacity to buffer the pH. Theoretical and experimental evidence is present for the sorption and desorption of hydrogen ions from oxide surfaces (Dzomback and Morel, 1990). Initial hydrogen desorption increased the hydrogen-ion concentration and maintained a low pH early in the experiment. The pH remained high following the introduction of uncontaminated ground water because of hydrogen-ion sorption onto oxide surfaces; these reactions are illustrated in table 3. This is significant for the transport of phosphorus because the ratio of sorbed to dissolved phosphorus decreases at higher pH values (Hingston and others, 1972). Stollenwerk (1995) reported that phosphorus in the dissolved phase increased with higher pH values. The data indicate that, as uncontaminated ground water moves through the aquifer after sewage disposal ceases, the pH of the ground water may remain significantly higher than that of uncontaminated water and that the capacity of the sediments to sorb phosphorus will remain low resulting in the increased mobility of phosphorus in ground water.

Bromide

Bromide was added to the uncontaminated-sediment columns as a conservative tracer to assess physical transport in the columns. Bromide is nonreactive and the asymmetry of bromide breakthrough curves can be attributed to physical processes. The influent concentration of bromide was 18 mg/L. The asymmetry of the bromide breakthrough curves indicate that diffusion is important in the columns (fig. 30). When bromide transport in the cores was simulated using a calculated dispersivity of 0.072 ft and no diffusion, a sharp concentration front was simulated in the model. Stollenwerk and Kipp (1990) found that adding a side-pore diffusion mechanism to a solute-transport model resulted in an excellent fit to experimental data. The amount of asymmetry caused by diffusion increases when a solute reacts with a solid (Stollenwerk and Kipp, 1990); the magnitude of the increase is affected by the extent of the reaction. Stollenwerk (1995) found that breakthrough curves for molybdate were more asymmetrical at a pH of 5.7 where molybdate sorption was greater; the breakthrough curves for molybdate in those experiments approached those of bromide at high pH values where molybdate sorption was less. Phosphorus in the columns shows a pattern similar to the previous observations of molybdate. The phosphorus breakthrough curve is more asymmetrical in the 0.49-mg/L column than in the 6.1-mg/L column. Concentration gradients are smaller in the 0.49-mg/L column than in the 6.1 mg/L column and result in slower diffusion. Also, the rate of kinetically controlled phosphorus reactions will decrease as concentrations decrease.

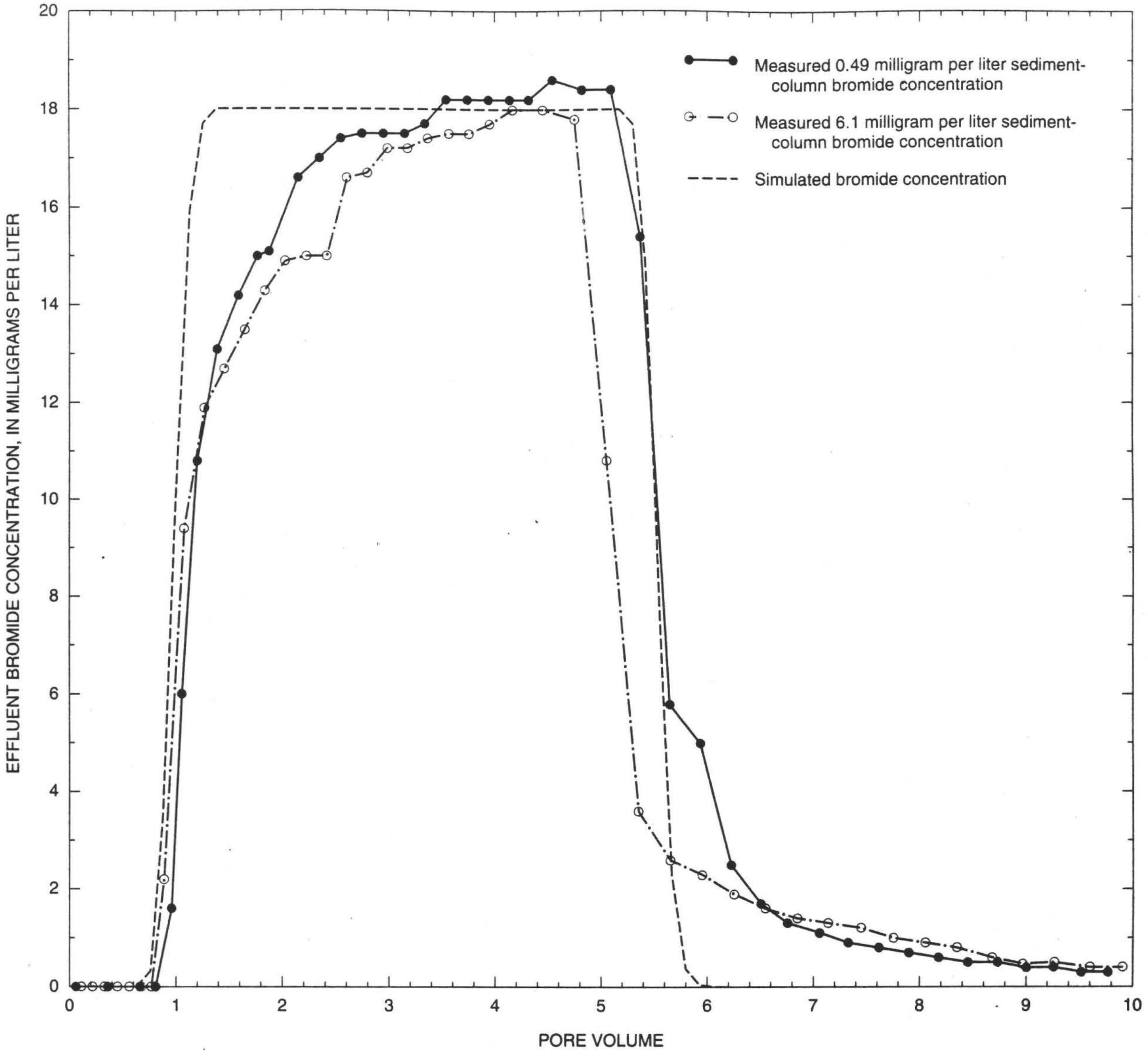


Figure 30. Changes in effluent bromide concentrations in two uncontaminated-sediment columns and simulated bromide breakthrough curve based on the surface-complexation model HYTEQ.

Contaminated Sediments

Column experiments using sediment from the contaminated part of the aquifer were used to evaluate phosphorus mobility in contaminated aquifer sediments after the introduction of uncontaminated oxygenated ground water. Sediment cores from the anoxic and suboxic zones within the aquifer were used in the experiments.

Anoxic Zone.—Changes in phosphorus and iron with pore volume in two anoxic columns are shown in figure 31*A*. Phosphorus in the initial pore water was 0.78 mg/L in sediments from anoxic core F415. After the introduction of uncontaminated water, phosphorus concentrations rapidly decreased to 0.03 mg/L by pore volume 5 and were less than detection by pore volume 18. In sediments from anoxic core F343, almost no

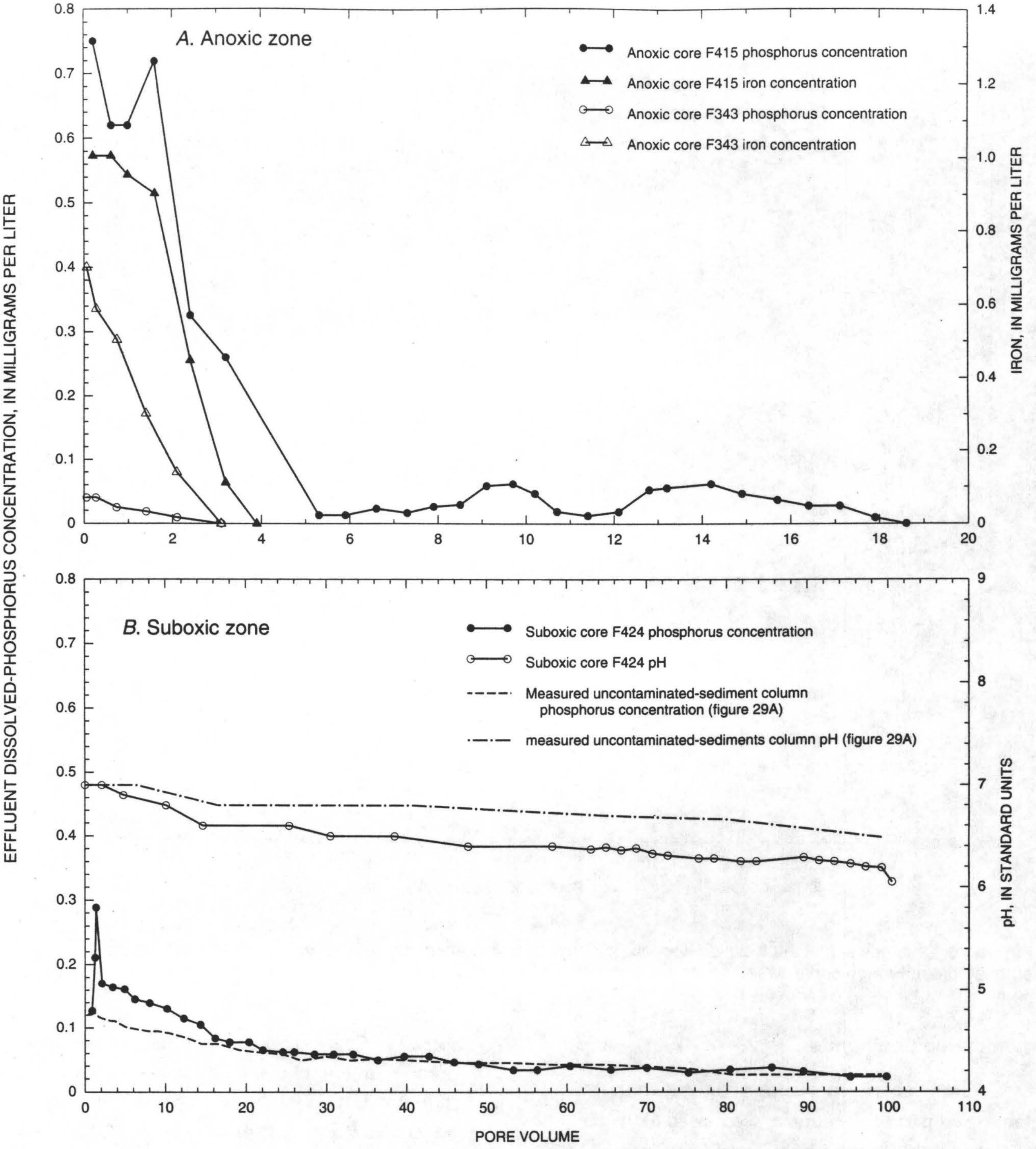


Figure 31. Changes in dissolved-phosphorus and iron concentrations in two contaminated-sediment columns from the (*A*) anoxic zone and changes in dissolved-phosphorus concentrations and pH in a contaminated sediment column from the (*B*) suboxic zone, both using uncontaminated ground water as influent.

phosphorus desorbed and the aqueous concentration decreased to less than detection limits by pore volume 3. The data indicate that phosphorus is relatively immobile in the anoxic sediments after uncontaminated, oxygenated water is introduced into the columns.

Concentrations of dissolved iron in the effluent from cores F343 and F415, which were initially 0.70 and 1.0 mg/L, respectively, decreased to less than detection limits by pore volume 4. Influent water was saturated with atmospheric oxygen, and ferrous iron in the interstitial water was rapidly oxidized to ferric iron or flushed from the columns. When ferric iron precipitated in the columns, much of the dissolved phosphorus either coprecipitated with or was sorbed onto the new iron oxyhydroxides. Willett and others (1988) present evidence that phosphorus rapidly adsorbs on all available sites on freshly precipitated ferric hydroxide. Phosphorus in the sorbed phase may have been present as an insoluble mineral, or the precipitation of ferric hydroxide may have coated previously sorbed phosphorus and limited any phosphorus desorption.

The laboratory experiments may not accurately reflect processes that will occur in the aquifer after sewage disposal is stopped. The laboratory eluting of anoxic sediments with oxygenated water may not allow adequate time for bacterial processes to become established in the columns. The microbially mediated oxidation of ammonia to nitrate in the anoxic zone could consume oxygen introduced by recharge water and could maintain reducing conditions in the aquifer for some time (Richard Smith, U.S. Geological Survey, oral commun., 1993). Dissolved oxygen in recharge water also could be removed by the oxidation of sorbed organic carbon or by the in-situ biomass. If dissolved oxygen is effectively removed from recharge water in this way, the mobility of phosphorus in the anoxic zone following the cessation of sewage disposal could be greater than the mobility observed in the columns.

Suboxic Zone.—Phosphorus was much more mobile in sediments from the suboxic zone of the aquifer following the introduction of uncontaminated ground water (fig. 31*B*). Phosphorus in the initial pore water of the sediment core from the suboxic zone was 0.12 mg/L. After uncontaminated water was introduced into the core, the concentration of phosphorus sharply increased to 0.30 mg/L. The concentration of phosphorus in the effluent remained greater than the initial concentration for about 11 pore volumes. This period of increased phosphorus concentrations may be analagous to the periods of high phosphorus concentration observed in the uncontaminated sediment columns following the introduction of uncontaminated ground water. The rate of phosphorus desorption then slowly decreased; 0.02 mg/L of phosphorus remained in the effluent after 100 pore volumes. This pattern is similar to the later part of the column experiment using uncontaminated sediments (fig. 29, pore volumes 150-250), which is replotted in figure 31*B* for comparison. The data indicate that phosphorus desorption in the suboxic column followed a similar trend as phosphorus desorption in the artificially contaminated column and suggest that similar processes affected phosphorus mobility in both columns. Phosphorous desorption in the more contaminated parts of the aquifer would be expected to yield higher concentrations of phosphorous than those observed in the suboxic column. Phosphorus in the initial pore water of the suboxic core used in the column core was 0.12 mg/L, whereas dissolved-phosphorous concentrations elsewhere in the suboxic zone of the aquifer were as high as 6.2 mg/L.

The trend in pH from the suboxic column was similar to the trend in the uncontaminated columns, indicating that the reversible adsorption of hydrogen ions effectively buffers pH in the suboxic zone of the aquifer. The initial pH in the suboxic column was 6.5. Although the influent pH was 5.5, the pH only decreased to 6.2 after 100 pore volumes of uncontaminated water had moved through the column. The slow decrease in pH slowly decreases the subsequent release of phosphorus as the sorbed phase becomes more favored in the equilibrium reaction between sorbed and dissolved phosphorus.

TRANSPORT OF PHOSPHORUS

Although the mobility of phosphorus in ground water is retarded by sorption, phosphorus can be transported by advection for substantial distances in ground water. Advective transport has caused the front of dissolved phosphorus to reach Ashumet Pond. Laboratory and field data were used to estimate phosphorus loading to the pond under current (1993) conditions, to develop a conceptual model of phosphorus transport in different geochemical environments, and to determine the change in phosphorus transport after sewage disposal ceases in December 1995.

Phosphorus Loading into Ashumet Pond

The extent of the Ashumet Valley sewage plume along the shore of Ashumet Pond was delineated with an EM34 surface electromagnetic induction tool; the instrument measures ground conductivity and can detect ground-water contamination when the water table is close to land surface and background conductivities are low. Ground conductivity along the shore of Ashumet Pond for January 1994 is shown in figure 32; the 0 datum is the U.S. Fish and Wildlife boat landing at Fisherman's Cove (fig. 2). Data indicate that the sewage plume, as defined by ground conductivity, underlies the shore from about 1,000 ft south to about 800 ft north of the boat landing. The extent of the part of the sewage plume that contains phosphorus, as determined from pond-bottom minipiezometers, is included in figure 32.

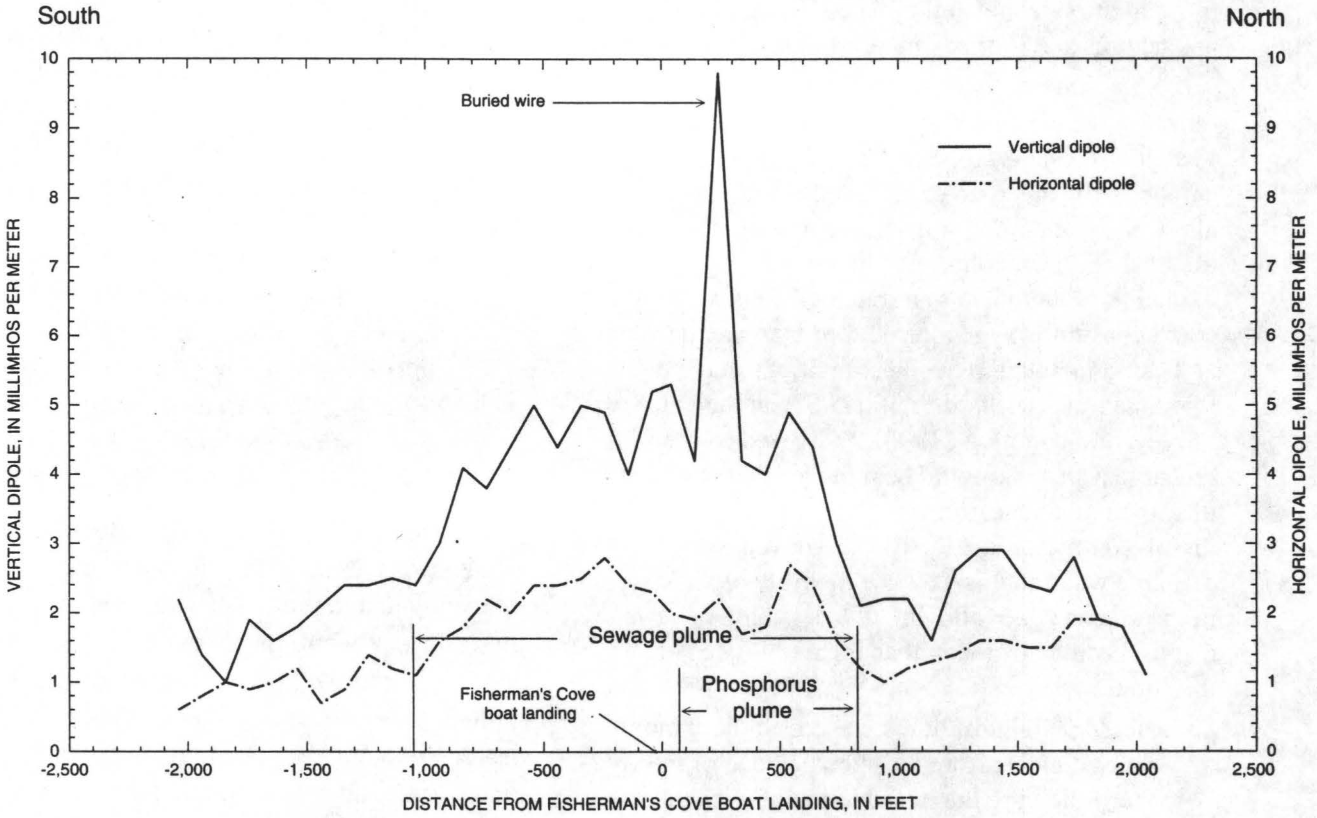


Figure 32. Ground conductivity along the northwestern shore of Ashumet Pond, Massachusetts, as measured using electromagnetic induction (EM34), January 1994.

Water-quality samples collected from 15 pond-bottom minipiezometers were used to delineate the distribution of phosphorus and related constituents in ground water discharging into Ashumet Pond (fig. 33). The data indicate that phosphorus currently discharges into the pond along about 700 ft of shoreline: from about 150 ft to about 850 ft north of the Fisherman's Cove boat landing. Phosphorus concentrations along the shore ranged from less than 0.01 mg/L in uncontaminated water to 1.9 mg/L in the sewage plume. The extent of the part of the plume that contained phosphorus in comparison to the extent of the entire sewage plume, as defined by conductivity, is shown in figure 32. The northern extent of the sewage plume along the shore, as delineated by specific conductance, pH, and dissolved-oxygen concentration was about 850 ft north of the boat landing. Dissolved-oxygen concentrations within the plume were low but detectable indicating suboxic conditions. Dissolved-iron concentrations typically were less than 0.05 mg/L in ground water discharging into the pond. Manganese, which was as high as 8.3 mg/L beneath the shore of the pond, discharged into the pond along about 1,000 ft of shoreline—from about 200 ft south to about 800 ft north of the boat landing. Black encrustation consisting of manganese oxyhydroxides was observed on cobbles along the pond shore.

Water-quality data from four sites along the shore of Ashumet Pond suggest that most or all of the phosphorus plume discharges to the pond. Multilevel-sampler data from sites F300, F424, and F564 (fig. 3) indicate that the part of the plume that contains phosphorus is primarily within the upper 75 ft of the sewage plume. Specific conductance and boron data from a screened auger boring at sampling site F583 indicate the presence of uncontaminated ground water to a depth of about 80 ft. Site F583 is in an area where pond water is recharging the aquifer and uncontaminated water overlies the plume. Phosphorus was not detected at any depth at site F583. Data indicate that the upper part of the plume is removed by the pond and that the part of the plume that contains phosphorus is effectively captured by the pond.

The flux of water entering the pond from the part of the aquifer contaminated by phosphorus was calculated using Darcy's Law:

$$Q_w = K\, i\, A_p,$$

where Q_w is the flux of water, K is hydraulic conductivity, i is hydraulic gradient, A_p is the cross-sectional area of the phosphorus-containing part of the sewage plume along the pond. Assuming conservative transport of phosphorus across the boundary between the aquifer and the pond, the mass flux of phosphorus (Q_p) entering the pond can be calculated by multiplying the flux of water (Q_w) by the weighted average concentration of phosphours (C_p) along the pond ($Q_p = Q_w C_p$). An average hydraulic conductivity (K) of 300 ft/d, a porosity of 0.39, and the previously discussed minimum, maximum, and average hydraulic gradients (i) were used in the calculations. The cross-sectional area (A_e)was calculated from geochemical section *C-C′* (fig. 22), and the weighted concentrations of phosphorus (C_p) in different cross-sectional subareas along the pond were calculated from multilevel-sampler data (table 9).

Water-level data indicate an increased component of ground-water flow to the pond during periods of rising water levels and that the mass of phosphorus discharging to the pond increases in the spring and summer months. The mass of phosphorus discharging to the pond during the study ranged from 59 kg/yr in January 1994 to 76 kg/yr in August 1993 and averaged 67 kg/yr. These estimates would vary with hydraulic conductivity, porosity, hydraulic gradient, and phosphorus concentrations along the pond. At present, phosphorus concentrations upgradient of the pond are significantly higher than those along the pond indicating that the mass flux of phosphorus into Ashumet Pond may increase significantly in the future.

K.V. Associates (1991) estimated that phosphorus loading to Ashumet Pond from the sewage plume averaged 37 kg/yr. Estimates made by E.C. Jordan Co. (1988), using the same data, ranged from 18 to 56 kg/yr and averaged 37 kg/yr. These previous estimates were made with phosphorus concentrations along the pond ranging from 0.10 to 0.30 mg/L; data collected during this study show that phosphorus concentrations along the pond were as high as 1.9 mg/L.

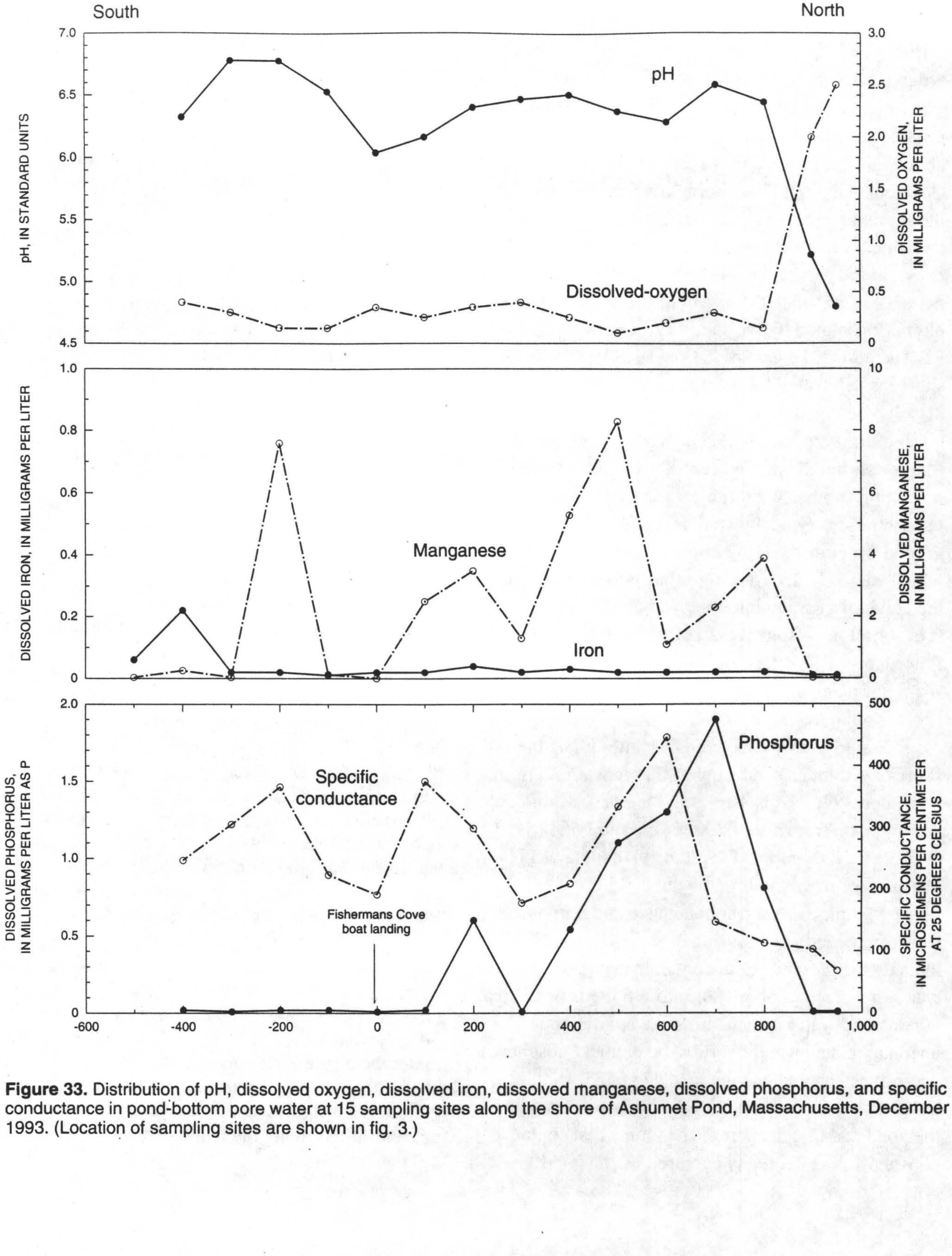


Figure 33. Distribution of pH, dissolved oxygen, dissolved iron, dissolved manganese, dissolved phosphorus, and specific conductance in pond-bottom pore water at 15 sampling sites along the shore of Ashumet Pond, Massachusetts, December 1993. (Location of sampling sites are shown in fig. 3.)

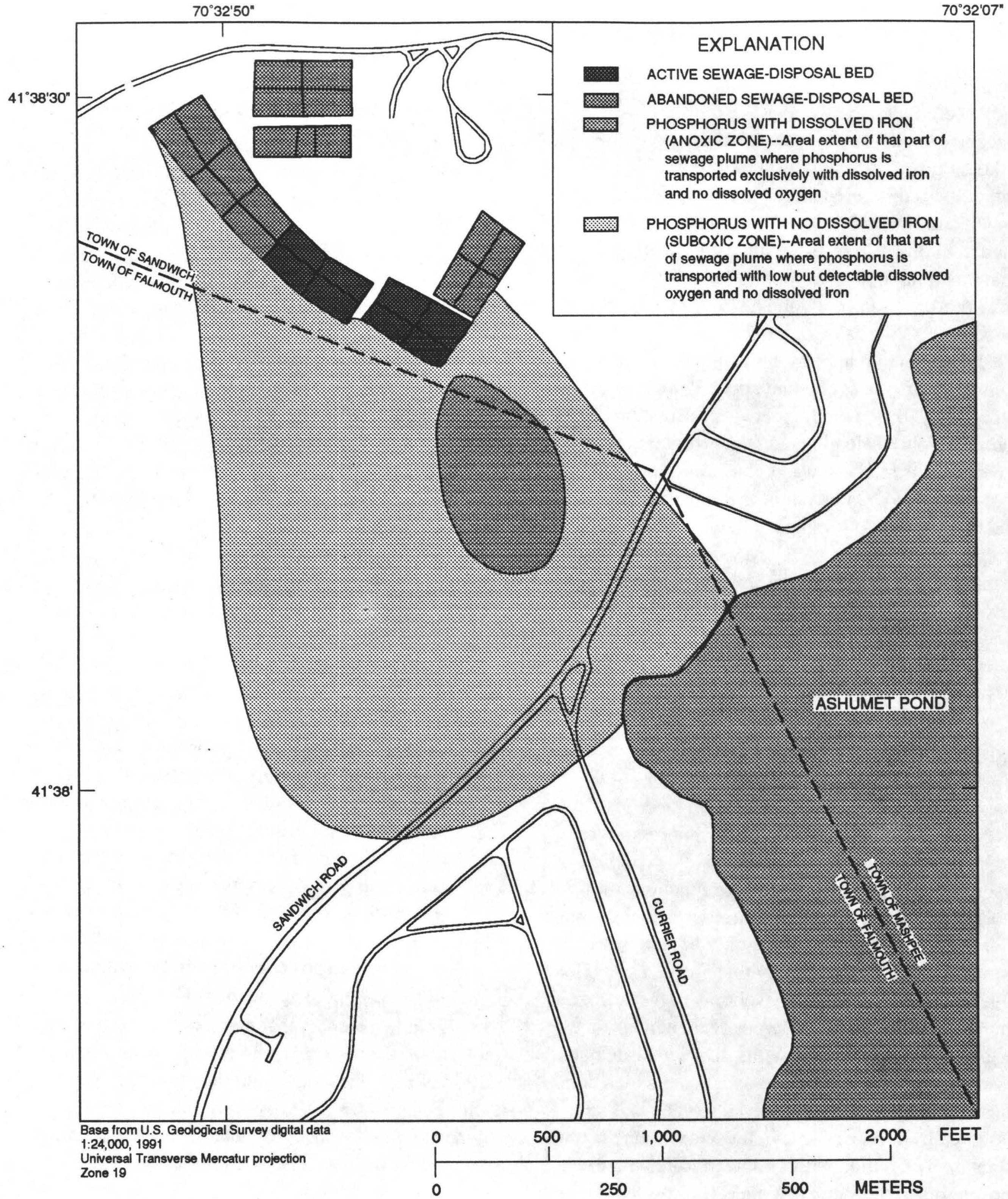
Assuming no other sources of phosphorus and a total ground-water input to the pond of 7.68 billion gal/yr (E.C. Jordan Co., 1988), input to the pond from the sewage plume, as delineated during this study, would increase the average phosphorus concentration in ground water discharging to the pond to 0.02 mg/L. A concentration of 0.02 mg/L generally is considered to be the critical limit for algal growth and eutrophication (Vollenweider, 1968). Ashumet Pond also receives inputs of phosphorus from precipitation, background ground water, septic tanks, stormwater runoff, and a bog at the north end of the pond. E.C. Jordan Co. (1988) and K.V. Associates (1991) estimated that phosphorus inputs to the pond from all other sources totaled about 70 and 88 kg/yr, respectively. Based on the previous estimation of additional phosphorus loading and data from this study, phosphorus from the sewage plume currently constitutes from 40 to 52 percent of the total phosphorus load to the pond. Assuming a total phosphorus input of 88 kg/yr from other sources (K.V. Associates, 1991), an input of 67 kg/yr from the sewage plume, and complete flushing of the pond, the corresponding average concentration of phosphorus for surface and ground water discharging into the pond would be about 0.053 mg/L.

Geochemical Factors Affecting Phosphorus Transport

The fate and transport of phosphorus, which are largely affected by the processes of sorption/desorption and coprecipitation, are functions of geochemical conditions in the aquifer. Typically, phosphorus is thought to be mobile only in reducing environments and in association with dissolved iron (Hem, 1992). Data indicate that phosphorus in the sewage plume is mobile in two different geochemical zones; the anoxic zone, which contains dissolved iron but no dissolved oxygen, and the suboxic zone, which contains low but detectable dissolved oxygen and no dissolved iron. Phosphorus in the anoxic zone is transported in association with dissolved iron, whereas phosphorus in the suboxic zone is transported without dissolved iron. The areal distribution of zones where phosphorus is and is not associated exclusively with dissolved iron is shown in figure 34. Laboratory and field data were the basis for developing a conceptual model of phosphorus transport in the two geochemical environments.

Anoxic Zone.—The sorption of phosphorus onto the surfaces of iron oxyhydroxides is an important factor affecting the mobility of phosphorus in the aquifer. In the anoxic zone, where redox potentials are low enough for the reductive dissolution of iron oxyhydroxides to occur, fewer sorption sites are available on aquifer sediments, and phosphorus is more mobile. The vertical distribution of selected water-quality constituents at three sampling sites in the sewage plume are shown in figure 35. Data from site F343, which is about 950 ft downgradient of the sewage-disposal beds and has sampling points in the anoxic zone of the aquifer, indicate that dissolved-phosphorus concentrations are high in the part of the aquifer where dissolved oxygen is at or near zero and dissolved iron is high. The average ratio of the concentration of sorbed to dissolved phosphorus in cores from the anoxic zone was about 31:1 as compared to about 155:1 for cores from the suboxic zone. This indicates that more phosphorus is dissolved in the anoxic ground water relative to total aquifer concentrations than in suboxic ground water and that phosphorus is more mobile in the anoxic, dissolved-iron zone than in the suboxic zone. The part of the sewage plume in which phosphorus is transported only in association with dissolved iron is areally less extensive and currently extends about 1,500 ft downgradient of the sewage-disposal beds (fig. 34).

Another process that may affect phosphorus mobility in the anoxic part of the aquifer is the precipitation of iron-phosphate mineral phases such as vivianite [$Fe_3(PO_4)_2 \cdot 8H_2O$]. Saturation indexes for vivianite, which were determined using the chemical speciation model WATEQF (Plummer and others, 1984), indicate that ground water from the anoxic



Base from U.S. Geological Survey digital data
1:24,000, 1991
Universal Transverse Mercatur projection
Zone 19

Figure 34. Areal extent of anoxic and suboxic zones where dissolved phosphorus in ground water is associated with and without dissolved iron near Ashumet Pond, Massachusetts, August through November 1993.

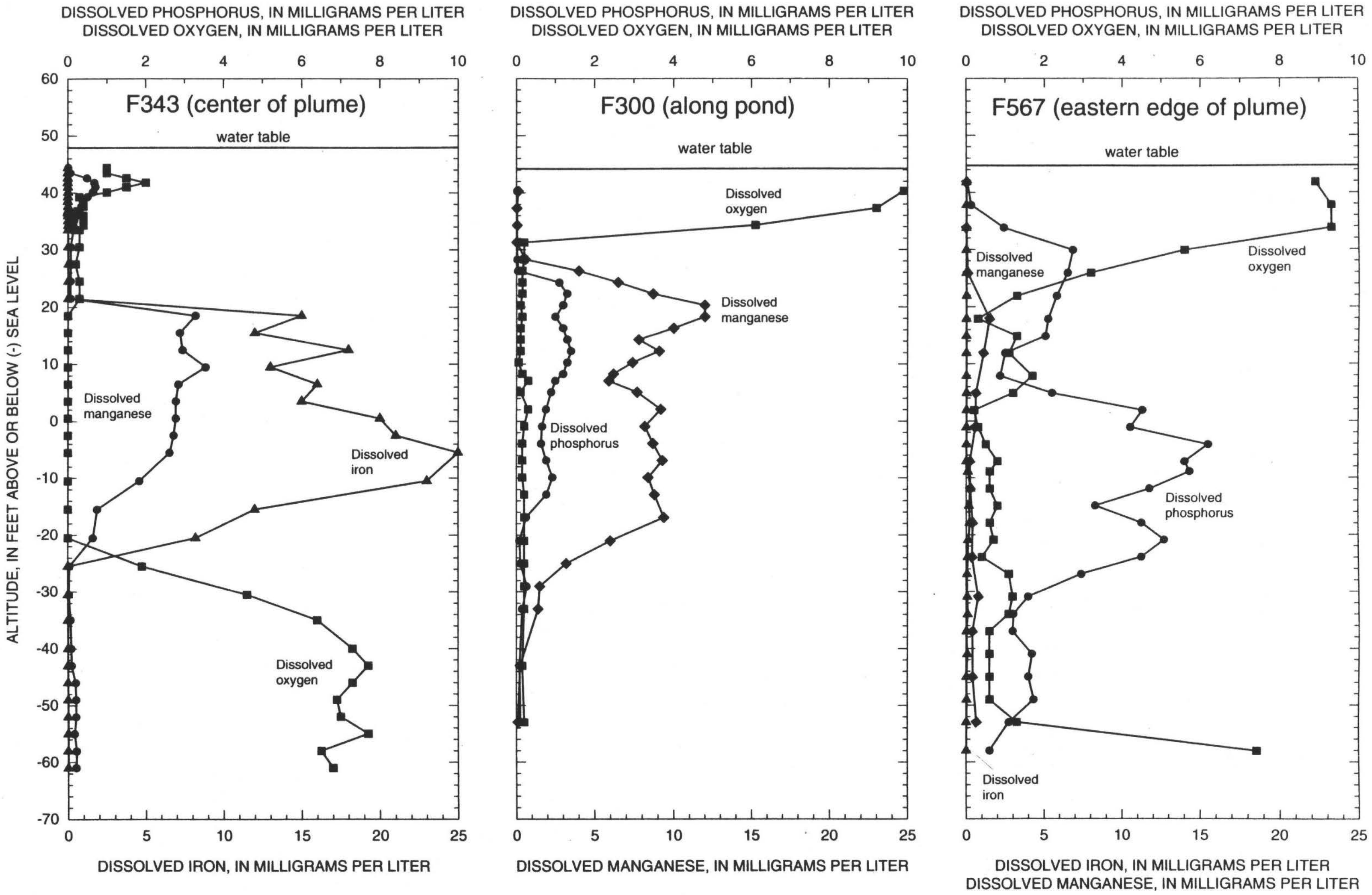


Figure 35. Vertical distribution of selected water-quality constituents at three sampling sites in the sewage plume near Ashumet Pond, Massachusetts, August through November 1993.

zone is supersaturated with respect to vivianite and that vivianite is thermodynamically favored to precipitate. Gschwend and Reynolds (1987) found colloids of vivianite in anoxic ground water from site F343, which suggests that colloidal transport of phosphorus may occur in the aquifer; the most common colloid size was in the 0.1-µm size fraction. Comparison of whole water samples and 0.1- and 0.45-µm filtered samples collected during the study from two sites in the anoxic zone showed no differences in phosphorus or iron. If colloids of iron phosphate minerals are immobile in the aquifer, the formation of iron-phosphate colloids may decrease phosphorus mobility in the anoxic zone.

Suboxic Zone.—The suboxic zone of the sewage plume is more areally extensive than the anoxic zone and currently (1993) the suboxic part of the plume is discharging into Ashumet Pond. Data from batch and column experiments indicate that sorption of phosphorus onto the surfaces of iron oxyhydroxides is the primary factor affecting phosphorus mobility in the suboxic zone. Davis and others (1993) found that aquifer sediments contained thick coatings of iron oxyhydroxides. The column and phosphorus data show that as available sorption sites are filled, phosphorus can be mobile in the suboxic and oxic parts of the aquifer. At present, the leading edge of phosphorus greater than 0.10 mg/L has migrated about 2,500 ft downgradient of the disposal beds. Maximum concentrations at sampling sites along the pond currently range from 1.0 to 2.0 mg/L. Data from site F300, which is in the suboxic zone of the aquifer along the pond, is shown in figure 35. Phosphorus in this part of the plume is associated with high specific conductances, low but detectable dissolved-oxygen concentrations, no dissolved iron, and high dissolved-manganese concentrations. Data from sampling site F567 show dissolved phosphorus in an environment with appreciable dissolved oxygen and no dissolved iron and low dissolved manganese (fig. 35). Concentrations of phosphorus at this site were the highest observed at substantial distances downgradient of the disposal beds; dissolved-oxygen concentrations associated with the phosphorus were as high as 5.5 mg/L, and dissolved iron typically was less than 0.1 mg/L. The presence of phosphorus in suboxic or oxic zones of the sewage plume, environments where phosphorus typically is not mobile in ground water, is attributed to prolonged loading of phosphorus in the aquifer that has filled available sorption sites and diminished the capacity of the aquifer sediments to sorb additional phosphorus.

Column experiments using uncontaminated sediments and influent phosphorus concentrations similar to sewage-effluent concentrations showed that appreciable time (about eight pore volumes) passed before any phosphorus transport occurred in the sediments. Saturation of phosphorus, indicating that all available sorption sites are filled with phosphorus, occurred after about 40 pore volumes; a single pore volume in the aquifer between the disposal beds and the pond corresponds to a traveltime of about 4.2 years. The column data indicate that, as sorption sites in the aquifer become filled, phosphorus breakthrough occurs and dissolved phosphorus is transported downgradient. After all available sorption sites are filled, phosphorus may be transported conservatively in the ground water. Assuming a traveltime of 4.2 years and a sewage-effluent concentration of 6.0 mg/L, phosphorus breakthrough and transport may not have occurred in the aquifer until about 30 years after sewage disposal began. The relatively slow movement of phosphorus through the columns indicates why phosphorus has been transported for a relatively short distance in the sewage plume. LeBlanc (1984) found that phosphorus had been transported only about one-fifth as far as more conservative constituents such as specific conductance and chloride. The presence of dissolved phosphorus as much as 2,500 ft downgradient of the sewage-disposal beds indicates that phosphorus breakthrough has occurred in the aquifer and that the capacity of the sediments to sorb phosphorus has been diminished. The disposal of sewage effluent has created a large reservoir of sorbed phosphorus in the suboxic zone of the aquifer that is as much as 410 times greater than phosphorus concentrations in the ground water.

Effects of Cessation of Sewage Disposal on Phosphorus Transport

The disposal of treated sewage at the wastewater-treatment plant will be discontinued in December 1995 (Michael Minior, Air National Guard Installation Restoration Program, personal commun., 1993). Following the cessation of sewage disposal, uncontaminated recharge water will move through previously contaminated parts of the aquifer, and geochemical conditions in the sewage plume will change. The introduction of uncontaminated ground water may greatly affect the mobility of phosphorus in the aquifer, and these effects may vary considerably according to prevailing geochemical conditions in the plume.

Anoxic Zone.—Results of the column experiments indicate that phosphorus in the anoxic zone will become relatively immobile following the introduction of oxygenated, uncontaminated ground water into the aquifer after sewage disposal is ceased. The oxidation of ferrous iron and the precipitation of ferric oxyhydroxides, such as geothite, hematite, and ferric hydroxide, will cause most of the dissolved phosphorus to sorb onto or coprecipitate with the iron oxyhydroxides. Phosphorus that is already sorbed onto aquifer sediments will be coated with new iron hydroxide and should remain immobile in the aquifer. In column experiments, all the dissolved phosphorus in the sediment cores from the anoxic zone was immobilized within four pore volumes. Ground water in the aquifer moves at a rate of about one pore volume every 4.2 years, indicating that phosphorus in the anoxic zone may be immobilized after about 17 years.

The continued consumption of oxygen by sorbed ammonia and organic carbon and by the in-situ biomass could allow anoxic conditions to persist in the aquifer for some time after sewage disposal is discontinued (Richard Smith, U.S. Geological Survey, oral commun., 1994). This would delay the arrival of oxygenated water into the anoxic zone and could allow the anoxic zone to move downgradient toward Ashumet Pond. In addition, the introduction of phosphorus-free ground water into the anoxic zone could dissolve any colloidal vivianite present in the aquifer, resulting in increased phosphorus concentrations. The distribution of oxygen in the aquifer should be monitored following sewage-disposal cessation to assess upgradient oxygen consumption.

Suboxic Zone.—Laboratory experiments show that, as uncontaminated ground water recharges suboxic parts of the aquifer after sewage disposal is ceased, a significant amount of sorbed phosphorus could desorb into the ground water and that the reservoir of sorbed phosphorus could remain a source of phosphorus for many years. Data from the artificially contaminated, suboxic-column experiments and from batch desorption experiments indicate that, following the introduction of uncontaminated water, phosphorus continues to desorb from aquifer sediments for many pore volumes. Data from column experiments indicate that phosphorus concentrations in the suboxic zone of the sewage plume could remain greater than 0.02 mg/L for as many 160 pore volumes and that concentrations in more contaminated parts of the aquifer could remain greater than 0.10 mg/L for as many as 95 pore volumes. The data also show that assuming a pore-volume traveltime of 4.2 years, phosphorus could remain greater than 1.0 mg/L for more than 30 years in the more contaminated parts of the aquifer.

The data also indicate that immediately after the introduction of uncontaminated ground water, dissolved phosphorus concentrations in the aquifer may sharply increase to concentrations greater than the initial concentrations. The maximum concentration of phosphorus in the column experiments occurred immediately after the introduction of uncontaminated water. The relative magnitude and duration of the period of maximum concentration was larger at low phosphorus concentrations. Assuming a pore-volume traveltime of 4.2 years, phosphorus concentrations in the ground water may increase above precessation levels for 8 to 30 years. Column data also indicate that about 50 percent of the reservoir of sorbed phosphorus may eventually desorb after sewage disposal is discontinued.

Data from the study suggest that geochemical conditions along the eastern boundary of the plume are changing in response to the abandonment of the eastern disposal beds. Data from sampling site F564, which is along Ashumet Pond, indicate that sewage-contaminated ground water is mixing with uncontaminated ground water. The site is downgradient of the easternmost group of four disposal beds (fig. 4), which were abandoned in 1984, and is along the eastern boundary of the sewage plume. The sewage plume at the site is about 25 ft thick and extends from an altitude of 26.6 to 1.6 ft above sea level. Specific-conductance values range from 110 to 176 µS/cm and dissolved-oxygen concentrations range from 2.9 to 4.0 mg/L. Phosphorus concentrations range from 0.21 to 1.70 mg/L and manganese concentrations range from 0.88 to 4.30 mg/L. The low specific-conductance values and high concentrations of dissolved oxygen, manganese, and phosphorus indicate that suboxic sewage-plume water that contains high concentrations of phosphorus and manganese is mixing with uncontaminated, oxygenated ground water. Dissolved-manganese concentrations should be less than the detection limit (0.01 mg/L) in oxygenated ground water (Hem, 1992).

Specific conductance and phosphorus data from two sites downgradient of abandoned sewage-disposal beds indicate that rapid phosphorus desorption analogous to the concentration peak observed in the column data may be occurring in some parts the aquifer (fig. 36). Specific conductance and phosphorus profiles from sampling site F567, upgradient of site F564 and along the eastern boundary of the plume, indicate that phosphorus exceeds 4.0 mg/L in a 26-foot thick zone that extends from an altitude of about 2 ft above sea level to about 24 ft below sea level. Phosphorus concentrations in water from sampling site F567, which is about 1,350 ft downgradient of the abandoned sewage-disposal beds, were as high as 6.2 mg/L. Specific conductance, which is an indicator of ground-water contamination, in the same zone was less than 200 µS/cm, whereas specific conductance in contaminated parts of the aquifer typically range from 300 to 500 µS/cm. The data indicate that the highest phosphorus concentrations are in a zone that has relatively moderate specific-conductance values indicating that phosphorus may be desorbing from aquifer sediments in response to the introduction of uncontaminated ground water into the eastern part of the sewage plume. The high phosphorus concentrations in ground water from site F567 may be the result of rapid phosphorus desorption similar to the concentration peak observed in the uncontaminated column experiments, in response to the introduction of uncontaminated water. About two pore volumes of contaminated water had moved through the aquifer prior to the time of sampling. This suggests that concentrations may be similar in other parts of the aquifer after sewage disposal is ceased.

The high concentrations measured in ground water from site F567 also may be related to the previous high phosphorus loading at the easternmost four abandoned disposal beds; site F567 may be on a more phosphorus-contaminated flow line as suggested by the high extractable phosphorus observed beneath the abandoned disposal beds (site S440). The high pH measured in the zone also may contribute to the high phosphorus concentrations. Stollenwerk (1995) found that less phosphorus was sorbed to aquifer sediments at high pH.

Specific conductance and phosphorus data from a screened auger boring at sampling site S590 also show evidence of rapid phosphorus desorption following the introduction of uncontaminated ground water (fig. 36). The site is downgradient of the western abandoned disposal beds (fig. 4); the loading history at these disposal beds and when the beds were last used are unknown. The data indicate a zone with phosphorus concentrations as high as about 8 mg/L from an altitude of about 2 to 22 ft above sea level. The specific conductance in the same zone ranged from 100 to 130 µS/cm, indicating near background specific-conductance values. High phosphorus concentrations in a zone with low specific-conductance values suggests that phosphorus desorption is occurring at the site in response to the introduction of uncontaminated ground water.

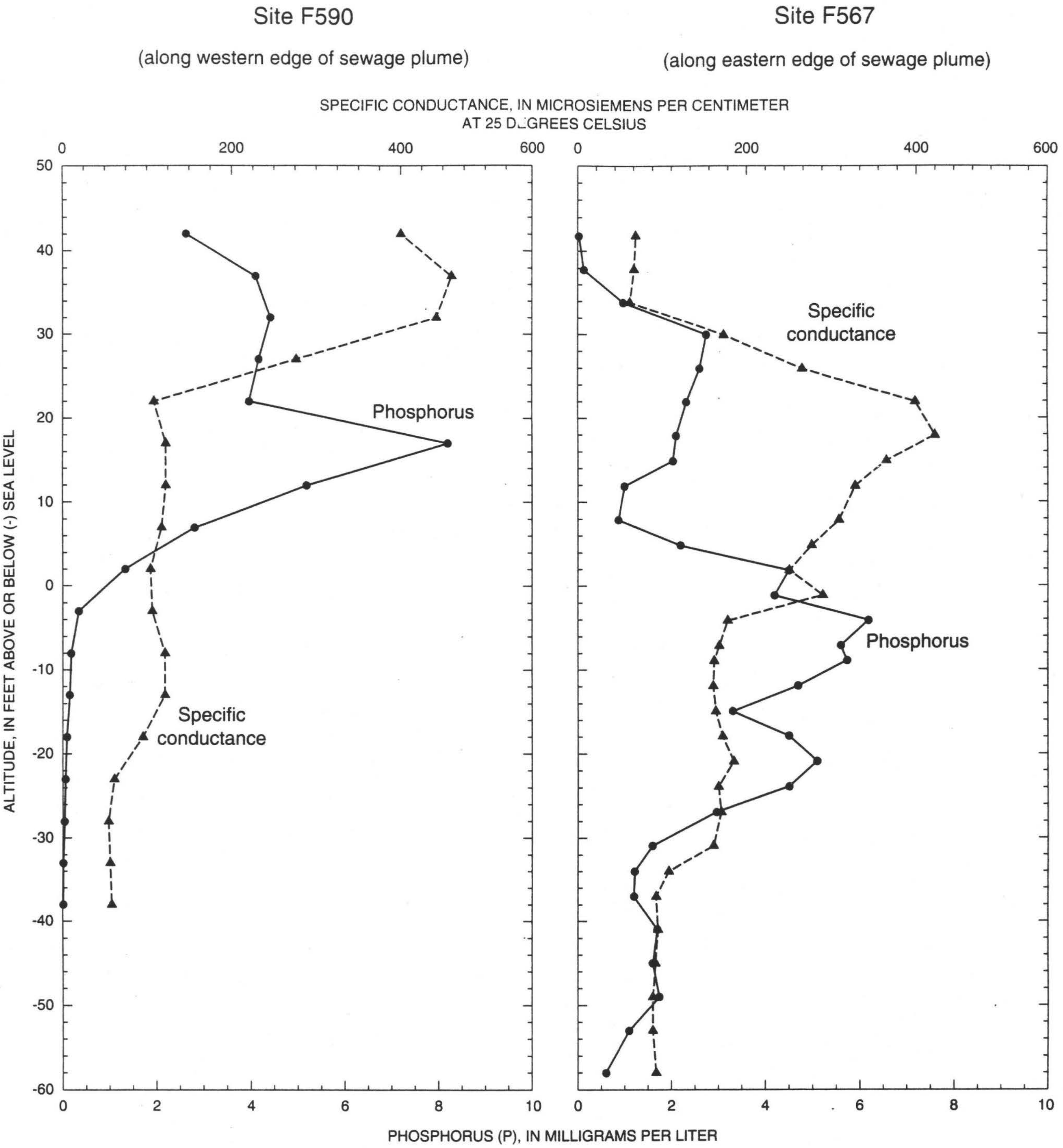


Figure 36. Vertical distribution of specific conductance and dissolved phosphorus at two sites downgradient of abandoned sewage-disposal beds at the Massachusetts Military Reservation sewage-treatment plant near Ashumet Pond, Massachusetts, October 1993 (site F567) and February 1994 (site F590).

SUMMARY

The disposal of secondarily treated sewage at the Massachusetts Military Reservation, Cape Cod, Massachusetts, has created a sewage plume of phosphorus-contaminated ground water in the underlying sand and gravel aquifer. Part of the sewage plume discharges into nearby Ashumet Pond, and there is concern that the continued discharge of phosphorus-contaminated ground water into the pond will degrade the ecology of the pond. Little data were available regarding the distribution and transport of phosphorus in the aquifer. A 1-year study was initiated in 1993 in cooperation with the National Guard Bureau to investigate the hydrologic and geochemical factors affecting phosphorus transport in the aquifer using field and laboratory data. Ground-water samples, sediment cores, and hydrologic data were collected during the study to develop a conceptual model of phosphorus transport between the disposal beds and Ashumet Pond.

Ground water in the aquifer is unconfined and flows in a southeasterly direction between the disposal beds and the pond. The gradient direction, which shifts to the east during rising water levels, averaged 164.8° east of north. The hydraulic gradient, which increases during rising water levels, averaged 0.00169 ft/ft. During this investigation, the average ground-water velocity was about 1.3 ft/d, and the corresponding traveltime between the disposal beds and the pond was about 4.2 years. Vertical hydraulic-head gradients in the aquifer are small, and flow is predominately horizontal. Data from wells and pond-bottom minipiezometers indicate that almost all the hydraulic-head loss between the aquifer and the pond is near the pond shore. Hydraulic gradients across the pond bottom, in the area of the sewage plume, ranged from 0 ft/ft where the pond and aquifer were in hydraulic equilibrium to 0.067 ft/ft where ground water was discharging into the pond. Mean hydraulic conductivities were higher at sites along the pond than those upgradient of the pond because of the location of the pond within a collapse structure. Previous aquifer-test results in the surrounding sand and aquifer yielded hydraulic conductivities that ranged from 300 to 380 ft/d.

Uncontaminated ground water typically is slightly acidic, has a low specific conductance and alkalinity, and has high dissolved-oxygen concentrations; concentrations of phosphorus, iron, and manganese are typically less than detection limits. Sewage-contaminated ground water has a higher pH and specific conductance, and lower dissolved-oxygen concentrations; contaminated ground water also may contain high concentrations of iron and manganese and typically has elevated concentrations of phosphorus. Specific conductance and pH in the sewage plume were as high as 541 μS/cm and 6.9, respectively. Dissolved oxygen was near zero in the center of the plume and concentrations were low but detectable along the pond indicating suboxic conditions. Dissolved iron ranged from less than 0.01 mg/L along the pond to 25 mg/L in the center of the plume. Dissolved manganese ranged from less than 0.01 mg/L at upgradient locations to 12 mg/L along the pond. Dissolved-phosphorus concentrations in the plume ranged from 0.10 to 11 mg/L, and exceeded 0.10 mg/L for a distance of about 2,500 ft downgradient of the disposal beds. Phosphorus concentrations were high in an area along the eastern edge of the plume, directly upgradient of the pond; concentrations in this area exceeded 4.0 mg/L. Along the shore of Ashumet Pond, the part of the plume that contained greater than 1.0 mg/L of phosphorus ranged in thickness from 10 to 25 ft. Data indicate that phosphorus discharges into Ashumet Pond along about 700 ft of shoreline and that phosphorus concentrations beneath the pond bottom were as high as 1.9 mg/L. The sewage plume currently discharges an average of about 67 kg of phosphorus into the pond annually; the plume contributes from 43 to 49 percent of the total phosphorus load into the pond. Phosphorus concentrations upgradient of the pond are higher than concentrations along the pond, indicating that the flux of phosphorus into Ashumet Pond could increase substantially.

The processes of sorption and coprecipitation limit the mobility of phosphorus in ground water. Core extractions and batch experiments were conducted to assess how phosphorus is distributed and transported in the aquifer. A large reservoir of sorbed phosphorus has accumulated in the aquifer. The ratio of sorbed to dissolved phosphorus ranged from 25:1 in the anoxic zone to 410:1 in the suboxic zone just downgradient of the disposal beds. Field and laboratory data indicate that phosphorus may be more mobile in the anoxic, dissolved-iron zone. Isotherm and rate-experiment data indicate that the advective transport of phosphorus under suboxic conditions occurs when the continued loading of phosphorus into the aquifer saturates available sorption sites on aquifer sediments. Phosphorus desorption occurs in the sediments when uncontaminated ground water is introduced.

Uncontaminated sediment-column experiments using different influent concentrations of phosphorus showed that uncontaminated aquifer sediments initially effectively sorbed phosphorus for several pore volumes. After several pore volumes of phosphorus loading, phosphorus moved through the column indicating that available sorption sites were becoming filled and that phosphorus was more mobile in the columns. Effluent concentrations eventually reached influent concentrations and stabilized indicating phosphorus saturation. After influent was changed to uncontaminated water, phosphorus concentrations sharply increased and exceeded influent concentrations for a few pore volumes because of the desorption of loosely bound phosphorus. Phosphorus continued to desorb for as many as 160 pore volumes. At higher influent concentrations of phosphorus, phosphorus in the effluent was detected earlier, approached influent concentrations more rapidly, and desorbed longer. Adsorption of hydrogen ions onto metal-oxide coatings caused the pH to remain high after uncontaminated water was introduced into the columns; the higher pH could result in increased phosphorus mobility.

Contaminated column experiments using uncontaminated water as influent showed that phosphorus desorption in the suboxic column had a similar trend to phosphorus desorption in the uncontaminated column—a sharp increase in phosphorus concentrations followed by slow and steady desorption. Phosphorus continued to desorb from the suboxic column for more than 100 pore volumes. Phosphorus in sediments from the anoxic column containing dissolved iron was immobilized within four pore volumes. Phosphorus in the pore water either sorbed onto or coprecipitated with iron oxyhydroxides after the introduction of oxygenated water into the core; phosphorus already sorbed to sediment surfaces probably was coated with iron oxyhydroxides and remained immobile. Column data indicate that following the cessation of sewage disposal in December 1995, phosphorus concentrations could greatly increase for 8 to 30 years and that the reservoir of sorbed phosphorus may remain a source of phosphorus for hundreds of years.

Data from three sampling sites downgradient of abandoned sewage-disposal beds and along the eastern and western boundaries of the sewage plume indicate that ground-water mixing and phosphorus desorption may be occurring in the aquifer in response to the introduction of uncontaminated ground water into previously contaminated parts of the aquifer. The data indicate high phosphorus concentrations in areas where specific conductance, a general indicator of ground-water contamination, was low.

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Table 5. Physical data for well-cluster sites used to collect water-quality samples and water-level measurements, near Ashumet Pond, Massachusetts

[Well locations shown in figures 1, 2, or 3. Altitudes are in feet above or below (-) sea level. Use: QW, water-quality sampling; WL, water-level measurement. ft, foot]

Cluster site	Well depth (ft)	Altitude of land surface	Altitude of top of screen	Altitude of bottom of screen	Use
F167	55	54.38	4.4	-0.6	WL
F230	42	87.06	46.6	44.6	WL
	49	86.14	38.7	36.7	QW, WL
	58	85.93	29.9	27.9	QW, WL
	68	86.33	20.2	18.2	QW, WL
	78	86.46	10.6	8.6	QW, WL
	88	86.35	.3	-1.7	QW, WL
	108	86.13	-20.0	-22.0	QW, WL
	127	86.06	-39.0	-41.0	QW, WL
F236	70	98.05	30.1	28.1	QW, WL
	89	98.16	11.2	9.2	QW
	106	98.00	-5.0	-8.0	QW
	121	98.09	-21.0	-23.0	QW
	141	97.69	-41.0	-43.0	QW
F237	88	90.46	5.4	2.4	WL
F239	10	51.32	45.0	41.1	WL
	64	50.98	-9.7	-12.7	QW, WL
	121	51.47	-68.0	-70.0	QW, WL
F244	70	69.88	4.3	2.3	QW, WL
	90	69.50	-17.7	-20.7	QW
	119	69.64	-47.0	-49.0	QW
F254	26	56.05	32.9	29.9	QW
	54	56.05	5.3	2.3	QW
	72	56.05	-12.8	-15.8	QW
	107	56.05	-48.0	-51.0	QW
F300	10	47.23	39.3	37.3	QW, WL
	30	47.14	19.5	17.5	QW, WL
	50	47.22	-1.2	-3.2	QW, WL
	73	47.11	-24.0	-26.0	QW, WL
	99	46.88	-50.5	-52.5	QW, WL
	118	46.94	-69.0	-71.0	QW, WL
	138	47.13	-89.0	-91.0	QW, WL
F343	36	68.91	34.5	32.5	QW, WL
	57	68.94	14.1	12.1	QW, WL
	79	68.82	-8.4	-10.4	QW, WL
	99	68.81	-28.1	-30.1	QW, WL
	114	69.20	-43.0	-45.0	QW, WL
	145	69.10	-74.0	-76.0	QW, WL
F345	31	68.45	38.9	36.9	WL
F346	32	59.08	28.9	26.9	WL
F347	20	50.54	32.0	30.0	WL
	31	59.71	30.9	28.9	QW, WL
F347	38	59.53	23.9	21.9	QW, WL
	46	59.75	15.9	13.9	QW, WL
	67	59.97	-4.4	-7.4	QW, WL
	101	59.73	-39.2	-41.0	QW, WL
	116	59.95	-54.0	-56.0	QW, WL
	131	60.27	-69.0	-71.0	QW, WL
F376	37	68.09	32.8	30.8	WL
F377	37	67.41	32.6	30.6	WL
F378	40	70.10	32.5	30.5	WL
F379	34	68.00	36.4	34.4	WL
F380	32	59.18	29.0	27.0	WL
F381	56	80.02	25.8	23.8	WL
F382	32	61.60	31.3	29.3	WL
F383	23	64.56	43.6	41.6	QW, WL
	30	64.58	36.8	34.8	QW
	32	64.05	33.5	31.5	WL
	40	64.58	26.2	24.2	QW
	61	64.39	5.8	3.8	QW
	82	64.32	-16.1	-18.1	QW
	106	64.87	-40.0	-42.0	QW
	129	64.74	-62.0	-64.0	QW
F384	33	60.46	29.8	27.8	WL
F385	32	66.23	35.9	33.9	WL
F388	37	68.78	34.2	32.2	QW, WL
	72	68.81	-1.4	-3.4	QW, WL
F414	36	67.97	33.8	31.8	WL
F420	36	60.37	26.1	24.1	WL
F421	31	56.66	27.5	25.5	WL
F422	45	73.33	30.0	28.0	QW, WL
	65	73.40	10.3	8.3	QW
	85	73.41	-9.7	-11.7	QW
	105	73.26	-30.0	-32.0	QW
F423	28	63.65	37.7	35.7	WL
F424	20	58.03	43.8	38.8	QW, WL
F425	33	68.24	37.2	35.2	WL
F432	26	68.66	46.5	42.5	WL
	59	68.34	11.2	9.2	QW
	79	68.41	-9.0	-11.0	QW
	92	68.52	-21.5	-23.5	QW

Table 5. Physical data for well-cluster sites used to collect water-quality samples and water-level measurements, near Ashumet Pond, Massachusetts--*Continued*

Cluster site	Well depth (ft)	Altitude of land surface	Altitude of top of screen	Altitude of bottom of screen	Use
F564	16	54.90	48.2	39.2	WL
	137	55.08	-81.0	-83.0	QW
F565	22	61.00	48.3	39.3	WL
F567	136	68.76	-65.0	-67.0	QW
S313	20	55.52	37.3	35.3	QW, WL
	38	55.52	19.4	17.4	QW
	60	55.44	-1.7	-4.7	QW
	80	55.42	-22.3	-24.3	QW
S314	35	78.00	45.7	43.7	QW, WL
	51	78.09	28.8	26.8	QW
	75	77.85	5.3	3.3	QW
	98	77.78	-17.9	-19.9	QW
S315	82	108.37	28.2	26.2	QW
	126	108.34	-16.0	-18.0	QW
	149	108.18	-39.0	-41.0	QW
S316	51	95.50	46.3	44.3	QW, WL
	66	95.47	31.7	29.7	QW
	82	95.42	15.1	13.1	QW
	100	95.39	-3.1	-5.0	QW
S316	134	95.07	-37.0	-39.0	QW
S317	27	68.90	51.1	42.1	QW, WL
	51	68.77	20.0	18.0	WL
S318	36	68.58	34.6	32.6	QW, WL
	64	68.77	7.1	5.1	QW
S344	38	79.87	43.4	41.4	QW, WL
	61	80.13	21.1	19.1	QW
	80	80.04	1.3	-0.7	QW
	100	79.99	-18.1	-20.0	QW
S395	28	68.72	50.5	40.5	WL
S396	41	79.31	48.3	38.3	WL
S397	52	93.60	51.6	41.6	WL
S433	16	58.27	51.7	41.7	WL
S435	17	58.83	51.4	41.4	WL
S438	41	79.14	48.1	38.1	WL
M204	9	49.68	49.5	40.5	WL
M205	32	71.80	48.3	39.3	WL

Table 6. Physical data for drive-point piezometers and minipiezometers used to collect water-quality samples and water-level measurements, near Ashumet Pond, Massachusetts, December 1993

[Drive-point sites shown in figures 1 and 2. Altitudes are in feet above or below (-) sea level. Use: QW, water-quality sampling; WL, water-level measurement. ft, foot]

Drive-point site	Well depth (ft)	Altitude of land surface	Altitude of top of screen	Altitude of bottom of screen	Use
Drive-Point Piezometers (Permanent)					
F424 P01	40	55.00	16.2	15.3	WL
	60	54.91	-3.9	-4.8	WL
	80	56.15	-22.9	-23.9	WL
	10	54.59	-43.9	-44.8	WL
F564 P01	35	55.09	21.3	20.4	WL
	55	55.64	1.4	.5	WL
	75	55.77	-18.4	-19.3	WL
	10	55.25	-43.8	-45.0	WL
F565 P01	40	61.60	22.7	21.8	WL
	65	61.64	-2.4	-3.3	WL
Pond-Bottom Drive-Point Minipiezometers (Temporary)					
DP 500S		42.82	38.77	38.94	QW, WL
DP 400S		42.92	40.43	40.60	QW
DP 300S		42.90	38.48	38.65	QW, WL
Pond-Bottom Drive-Point Minipiezometers (Temporary)--*Continued*					
DP 200S		42.65	38.48	38.65	QW, WL
DP 100S		42.89	38.48	38.65	QW, WL
DP 0		41.46	38.20	38.37	QW, WL
DP 100N		42.48	38.51	38.68	QW, WL
DP 200N		42.57	38.51	38.68	QW, WL
DP 300N		42.58	38.51	38.68	QW, WL
DP 400N		42.51	38.51	38.68	QW, WL
DP 500N		42.74	38.51	38.68	QW, WL
DP 600N		42.44	38.40	38.57	QW, WL
DP 700N		42.51	38.40	38.57	QW, WL
DP 800N		42.61	38.40	38.57	QW, WL
DP 900N		42.35	38.40	38.57	QW, WL
DP 950N		42.70	38.69	38.86	QW, WL

Table 7. Physical data for multilevel samplers used to collect water-quality samples, near Ashumet Pond, Massachusetts

[Locations of multilevel samplers shown in figure 3. Altitudes are in feet above or below (-) sea level]

Multilevel sampler		Altitude of land surface	Altitude of top of screen	Altitude of bottom of screen
F239	M01-02	51.10	36.3	36.2
	M01-03	51.10	31.3	31.2
	M01-04	51.10	26.3	26.2
	M01-05	51.10	21.3	21.2
	M01-06	51.10	16.3	16.2
	M01-07	51.10	11.3	11.2
	M01-08	51.10	6.3	6.2
	M01-09	51.10	1.3	1.2
	M01-10	51.10	-8.6	-8.7
	M01-11	51.10	-18.6	-18.7
	M01-12	51.10	-28.6	-28.7
	M01-13	51.10	-38.6	-38.7
	M01-14	51.10	-48.6	-48.7
	M01-15	51.10	-59.0	-59.0
F300	M02-01	46.90	7.2	7.1
	M02-02	46.90	5.2	5.1
	M02-03	46.90	2.2	2.1
	M02-04	46.90	-0.8	-0.9
	M02-05	46.90	-3.8	-3.9
	M02-06	46.90	-6.8	-6.9
	M02-07	46.90	-9.8	-9.9
	M02-08	46.90	-12.8	-12.9
	M02-09	46.90	-16.8	-16.9
	M02-10	46.90	-20.9	-21.0
	M02-11	46.90	-24.9	-25.0
	M02-12	46.90	-28.9	-29.0
	M02-13	46.90	-32.9	-33.0
	M02-14	46.90	-42.9	-43.0
	M02-15	46.90	-52.9	-53.0
	M03-01	46.90	40.4	40.3
	M03-02	46.90	37.4	37.3
	M03-03	46.90	34.4	34.3
	M03-04	46.90	31.4	31.3
	M03-05	46.90	28.4	28.3
	M03-06	46.90	26.4	26.3
	M03-07	46.90	24.4	24.3
	M03-08	46.90	22.4	22.3
	M03-09	46.90	20.4	20.3
	M03-10	46.90	18.4	18.3
	M03-11	46.90	16.4	16.3
	M03-12	46.90	14.4	14.3
	M03-13	46.90	12.4	12.3
	M03-14	46.90	10.4	10.3
	M03-15	46.90	8.4	8.3
F343	M01-01	68.90	-5.4	-5.5
	M01-02	68.90	-10.4	-10.5
	M01-03	68.90	-15.4	-15.5
F343	M01-04	68.90	-20.4	-20.5
	M01-05	68.90	-25.4	-25.5
	M01-06	68.90	-30.4	-30.5
	M01-07	68.90	-35.0	-35.0
	M01-08	68.90	-40.0	-40.0
	M01-09	68.90	-43.0	-43.0
	M01-10	68.90	-46.0	-46.0
	M01-11	68.90	-49.0	-49.0
	M01-12	68.90	-52.0	-52.0
	M01-13	68.90	-55.0	-55.0
	M01-14	68.90	-58.0	-58.0
	M01-15	68.90	-61.0	-61.0
	M02-01	68.90	36.6	36.5
	M02-02	68.90	33.6	33.5
	M02-03	68.90	30.6	30.5
	M02-04	68.90	27.6	27.5
	M02-05	68.90	24.6	24.5
	M02-06	68.90	21.6	21.5
	M02-07	68.90	18.6	18.5
	M02-08	68.90	15.6	15.5
	M02-09	68.90	12.6	12.5
	M02-10	68.90	9.6	9.5
	M02-11	68.90	6.6	6.5
	M02-12	68.90	3.6	3.5
	M02-13	68.90	0.6	0.5
	M02-14	68.90	-2.4	-2.5
	M02-15	68.90	-5.4	-5.5
	M03-01	68.65	45.3	45.2
	M03-02	68.65	44.5	44.4
	M03-03	68.65	43.6	43.5
	M03-04	68.65	42.7	42.6
	M03-05	68.65	41.9	41.8
	M03-06	68.65	41.1	41.0
	M03-07	68.65	40.2	40.1
	M03-08	68.65	39.4	39.3
	M03-09	68.65	38.6	38.5
	M03-10	68.65	37.7	37.6
	M03-11	68.65	36.9	36.8
	M03-12	68.65	36.1	36.0
	M03-13	68.65	35.3	35.1
	M03-14	68.65	34.4	34.3
	M03-15	68.65	33.6	33.5
F347	M01-01	59.90	41.6	41.5
	M01-02	59.90	39.7	39.6
	M01-03	59.90	37.8	37.7
	M01-04	59.90	35.9	35.8
	M01-05	59.90	34.0	33.9
	M01-06	59.90	32.1	32.0

Table 7. Physical data for multilevel samplers used to collect water-quality samples, near Ashumet Pond, Massachusetts *--Continued*

Multilevel sampler		Altitude of land surface	Altitude of top of screen	Altitude of bottom of screen
F347	M01-07	59.90	30.2	30.1
	M01-08	59.90	28.3	28.2
	M01-09	59.90	26.4	26.3
	M01-10	59.90	24.5	24.4
	M01-11	59.90	22.6	22.5
	M01-12	59.90	20.7	20.6
	M01-13	59.90	18.8	18.7
	M01-14	59.90	16.9	16.8
	M01-15	59.90	15.0	14.9
	M06-01	59.90	20.0	19.9
	M06-02	59.90	15.0	14.9
	M06-03	59.90	10.0	9.9
	M06-04	59.90	5.0	4.9
	M06-05	59.90	0	-.1
	M06-06	59.90	-5.0	-5.1
	M06-07	59.90	-10.0	-10.1
	M06-08	59.90	-15.1	-15.2
	M06-09	59.90	-20.1	-20.2
	M06-10	59.90	-25.1	-25.2
	M06-11	59.90	-30.1	-30.2
	M06-12	59.90	-35.1	-35.2
	M06-13	59.90	-40.0	-40.0
	M06-14	59.90	-45.0	-45.0
	M06-15	59.90	-50.0	-50.0
F424	M01-01	57.88	5.9	5.8
	M01-02	57.88	.9	.8
	M01-03	57.88	-4.1	-4.2
	M01-04	57.88	-9.0	-9.1
	M01-05	57.88	-14.0	-14.1
	M01-06	57.88	-19.0	-19.1
	M01-07	57.88	-24.0	-24.1
	M01-08	57.88	-29.0	-29.1
	M01-09	57.88	-34.0	-34.1
	M01-10	57.88	-39.0	-39.1
	M01-11	57.88	-44.0	-44.0
	M01-12	57.88	-49.0	-49.0
	M01-13	57.88	-54.0	-54.0
	M01-14	57.88	-59.0	-59.0
	M01-15	57.88	-64.0	-64.0
	M02-01	56.87	44.2	44.1
	M02-02	56.87	41.7	41.6
	M02-03	56.87	39.2	39.1
	M02-04	56.87	36.7	36.6
	M02-05	56.87	34.2	34.1
	M02-06	56.87	31.7	31.6
	M02-07	56.87	29.2	29.1
	M02-08	56.87	26.6	26.5
	M02-09	56.87	24.1	24.0
	M02-10	56.87	21.6	21.5
F424	M02-11	56.87	19.1	19.0
	M02-12	56.87	16.6	16.5
	M02-13	56.87	14.1	14.0
	M02-14	56.87	11.6	11.5
	M02-15	56.87	9.1	9.0
F432	M01-01	68.54	41.4	41.3
	M01-02	68.54	36.4	36.3
	M01-03	68.54	31.4	31.3
	M01-04	68.54	26.4	26.3
	M01-05	68.54	21.4	21.3
	M01-06	68.54	16.4	16.3
	M01-07	68.54	11.4	11.3
	M01-08	68.54	6.4	6.3
	M01-09	68.54	1.4	1.3
	M01-10	68.54	-8.6	-8.7
	M01-11	68.54	-18.6	-18.7
	M01-12	68.54	-28.6	-28.7
	M01-13	68.54	-38.0	-38.0
	M01-14	68.54	-48.0	-48.0
	M01-15	68.54	-58.0	-58.0
F512	M01-01	64.52	45.1	45.0
	M01-02	64.52	43.0	43.1
	M01-03	64.52	40.9	41.0
	M01-04	64.52	38.8	38.7
	M01-05	64.52	36.7	36.6
	M01-06	64.52	34.2	34.1
	M01-07	64.52	31.7	31.6
	M01-08	64.52	29.2	29.1
	M01-09	64.52	26.7	26.6
	M01-10	64.52	24.2	24.1
	M01-11	64.52	21.7	21.6
	M01-12	64.52	19.2	19.1
	M01-13	64.52	16.7	16.6
	M01-14	64.52	14.1	14.0
	M01-15	64.52	11.6	11.4
F564	M01-01	54.71	41.7	41.6
	M01-02	54.71	36.7	36.6
	M01-03	54.71	31.7	31.6
	M01-04	54.71	26.7	26.6
	M01-05	54.71	21.7	21.6
	M01-06	54.71	16.7	16.6
	M01-07	54.71	11.7	11.6
	M01-08	54.71	6.7	6.6
	M01-09	54.71	1.7	1.6
	M01-10	54.71	-8.3	-8.4
	M01-11	54.71	-18.3	-18.4
	M01-12	54.71	-28.3	-28.4

Table 7. Physical data for multilevel samplers used to collect water-quality samples, near Ashumet Pond, Massachusetts --*Continued*

Multilevel sampler		Altitude of land surface	Altitude of top of screen	Altitude of bottom of screen
F564	M01-13	54.71	-38.3	-38.4
	M01-14	54.71	-48.0	-48.0
	M01-15	54.71	-58.0	-58.0
F566	M01-01	60.08	7.1	7.0
	M01-02	60.08	2.1	2.0
	M01-03	60.08	-2.9	-3.0
	M01-04	60.08	-7.9	-8.0
	M01-05	60.08	-12.9	-13.0
	M01-06	60.08	-17.9	-18.0
	M01-07	60.08	-22.9	-23.0
	M01-08	60.08	-27.9	-28.0
	M01-09	60.08	-32.9	-33.0
	M01-10	60.08	-37.9	-38.0
	M01-11	60.08	-43.0	-43.0
	M01-12	60.08	-48.0	-48.0
	M01-13	60.08	-53.0	-53.0
	M01-14	60.08	-58.0	-58.0
	M01-15	60.08	-63.0	-63.0
	M02-01	60.08	41.1	41.0
	M02-02	60.08	39.0	38.9
	M02-03	60.08	37.0	36.9
	M02-04	60.08	34.9	34.8
	M02-05	60.08	32.8	32.7
	M02-06	60.08	30.7	30.6
	M02-07	60.08	28.6	28.5
	M02-08	60.08	26.5	26.4
	M02-09	60.08	24.4	24.3
	M02-10	60.08	22.3	22.2
	M02-11	60.08	20.3	20.2
	M02-12	60.08	18.3	18.2
	M02-13	60.08	16.2	16.1
	M02-14	60.08	14.1	14.0
	M02-15	60.08	12.0	11.9
F567	M01-01	68.57	41.9	41.8
	M01-02	68.57	37.9	37.8
	M01-03	68.57	33.9	33.8
	M01-04	68.57	30.0	29.9
	M01-05	68.57	26.0	25.9
	M01-06	68.57	22.0	21.9
	M01-07	68.57	18.0	17.9
	M01-08	68.57	15.0	14.9
	M01-09	68.57	12.0	11.9
	M01-10	68.57	8.0	7.9
	M01-11	68.57	5.0	4.9
	M01-12	68.57	2.0	1.9
	M01-13	68.57	-1.0	-1.1
F567	M01-14	68.57	-4.0	-4.1
	M01-15	68.57	-7.0	-7.1
	M02-01	68.35	-8.8	-8.9
	M02-02	68.35	-11.8	-11.9
	M02-03	68.35	-14.8	-14.9
	M02-04	68.35	-17.8	-17.9
	M02-05	68.35	-20.8	-20.9
	M02-06	68.35	-23.8	-23.9
	M02-07	68.35	-26.8	-26.9
	M02-08	68.35	-30.8	-30.9
	M02-09	68.35	-34.0	-34.0
	M02-10	68.35	-37.0	-37.0
	M02-11	68.35	-41.0	-41.0
	M02-12	68.35	-45.0	-45.0
	M02-13	68.35	-49.0	-49.0
	M02-14	68.35	-53.0	-53.0
	M02-15	68.35	-58.0	-58.0
S317	M01-02	68.76	46.0	45.9
	M01-03	68.76	44.0	43.9
	M01-04	68.76	42.0	41.9
	M01-05	68.76	40.0	39.9
	M01-06	68.76	38.0	37.9
	M01-07	68.76	36.0	35.9
	M01-08	68.76	34.0	33.9
	M01-09	68.76	32.0	31.9
	M01-10	68.76	30.0	29.9
	M01-11	68.76	28.1	28.0
	M01-12	68.76	26.1	26.0
	M01-13	68.76	24.1	24.0
	M01-14	68.76	22.1	22.0
	M01-15	68.76	20.1	20.0
	M02-01	68.76	20.0	19.9
	M02-02	68.76	20.0	19.9
	M02-03	68.76	16.0	15.9
	M02-04	68.76	16.0	15.9
	M02-05	68.76	12.1	12.0
	M02-06	68.76	12.1	12.0
	M02-07	68.76	8.1	8.0
	M02-08	68.76	8.1	8.0
	M02-09	68.76	4.1	4.0
	M02-10	68.76	4.1	4.0
	M02-11	68.76	0.2	0.1
	M02-12	68.76	-3.8	-3.9
	M02-13	68.76	-3.8	-3.9
	M02-14	68.76	-5.8	-5.9
	M02-15	68.76	-7.8	-7.9

Table 8. Chemical analyses of ground-water samples from well-cluster sites, near Ashumet Pond, Massachusetts,

[Analyses were done by the U.S. Geological Survey National Water-Quality Laboratory in Arvada, Colo. Location of cluster site shown in figure 3. ft, foot;

Cluster site	Well depth (ft)	Date	Specific conductance (µS/cm)	pH (standard units)	Oxygen, dissolved (mg/L)	Alkalinity, dissolved, field (mg/L as $CaCO_3$)	Sulfate, dissolved (mg/L)	Nitrogen, nitrate, dissolved (mg/L as N)	Nitrogen, nitrite plus nitrate, dissolved (mg/L as N)
F230	49	8-27-93	400	5.7	0.20	26	28	<0.01	17.90
	58	8-27-93	350	5.9	.20	31	29	<.01	12
	68	8-27-93	115	5.6	.10	7	17	<.01	1.6
	78	8-27-93	380	5.2	.30	7	23	<.01	20
	88	8-27-93	205	5.6	.60	95	11	.03	6.5
	108	8-27-93	115	5.5	8.2	3	<.10	<.01	.90
	127	8-27-93	62	5.4	9.6	3	12	<.01	.73
F236	70	8-13-93	76	5.6	7.7	6	7.4	--	--
	89	8-13-93	82	5.6	8.4	5	11	--	--
	106	8-13-93	127	5.4	9.5	3	17	--	--
	121	8-13-93	63	5.3	11.2	2	13	--	--
	141	8-13-93	75	5.6	11.3	3	16	--	--
F239	64	8-19-93	315	6.4	.05	51	47	--	--
	121	8-19-93	83	6.5	.15	13	7.2	--	--
F244	70	11-23-93	311	6.3	.20	39	--	--	--
	90	11-23-93	246	6.4	.25	36	--	--	--
	119	11-23-93	158	6.3	.15	21	--	--	--
F254	26	1-24-94	111	4.9	2.5	--	5.3	--	--
	54	1-24-94	220	5.9	.30	--	23	--	--
	72	1-24-94	246	6.5	.30	--	29	--	--
	107	1-24-94	125	6.3	.20	--	12	--	--
F300	10	9-08-93	80	5.2	9.2	3	7.7	<.01	.22
	30	9-08-93	400	6.3	.15	43	39	<.01	14
	50	9-08-93	240	6.4	.08	44	20	<.01	<.05
	73	9-08-93	225	6.2	.08	39	21	<.01	<.05
	99	9-08-93	98	6.3	4.3	12	11	<.01	1.8
	118	9-08-93	93	6.4	7.8	8	14	<.01	1.4
	138	9-08-93	84	6.5	9.5	11	12	<.01	1.0
F343	36	8-24-93	520	6.0	.90	29	27	.26	50
	57	8-24-93	310	6.6	<.05	84	23	<.01	<.01
	79	8-24-93	132	6.6	<.05	35	12	<.01	<.05
	99	8-24-93	100	5.9	5.50	4	12	<.01	.76
	114	8-24-93	68	5.7	11.0	4	12	<0.01	0.66
	145	8-24-93	92	5.6	9.8	--	17	<.01	.83
F347	31	8-31-93	280	5.9	.10	21	18	.02	15
	38	8-31-93	440	5.9	.15	28	31	.17	22
	46	8-31-93	410	6.0	.15	33	31	.36	15
	67	8-31-93	210	6.4	<.05	50	17	<.01	<.05
	101	8-31-93	82	5.8	3.4	4	14	<.01	.76
	116	8-31-93	68	5.8	7.1	5	12	<.01	.87
	131	8-31-93	85	5.7	7.7	2	19	<.01	1.0

August 1993 through January 1994

mg/L, milligram per liter; µg/L, microgram per liter; µS/cm, microsiemen per centimeter at 25°C. <, actual value is less than value shown. --, no analysis)

Cluster site	Well depth (ft)	Nitrogen, ammonia, dissolved, organic (mg/L as N)	Phosphorus, dissolved (mg/L as P)	Boron, dissolved (mg/L)	Iron, dissolved (mg/L)	Manganese, dissolved (mg/L)	Carbon, organic, dissolved (mg/L as C)	Methylene-blue active substances (mg/L)
F230	49	0.63	1.9	390	<0.01	0.06	2.8	0.04
	58	.59	1.5	390	.01	.05	2.8	.07
	68	1.0	.07	60	<.01	.05	1.4	<.02
	78	1.2	.06	270	.02	1.1	1.0	<.02
	88	.02	.05	150	.04	.90	.80	<.02
	108	.02	.12	40	<.01	.06	.20	<.02
	127	.02	.13	20	.02	.04	.30	<.02
F236	70	--	.11	--	<.01	.02	--	--
	89	--	.09	--	<.01	.02	--	--
	106	--	.01	--	.08	.05	--	--
	121	--	.03	--	<.01	.10	--	--
	141	--	.05	--	<.01	.01	--	--
F239	64	--	.05	--	.10	<.01	1.6	--
	121	--	.02	--	.01	<.01	--	--
F244	70	--	.03	--	.01	3.0	--	--
	90	--	.07	--	.02	.20	--	--
	119	--	.05	--	.01	2.5	--	--
F254	26	--	.01	--	.02	.02	--	--
	54	--	.03	--	.02	.39	--	--
	72	--	.03	--	.52	.85	--	--
	107	--	.04	--	.02	.01	--	--
F300	10	.02	<.01	<10	<.01	<.01	.30	<.02
	30	3.1	1.1	250	<.01	13	2.0	<.02
	50	3.3	.77	160	<.01	7.3	1.4	.04
	73	2.4	.18	30	<.01	4.3	1.2	.05
	99	1.7	.10	30	<.01	.11	.30	<.02
	118	.03	.04	20	<.01	<.01	.10	<.02
	138	.02	.03	10	.01	<.01	.10	<.02
F343	36	.04	.10	380	.02	.78	2.7	.09
	57	1.2	3.0	360	14.0	.25	2.7	.10
	79	.33	3.0	50	14.0	.21	.80	<.02
	99	.03	.04	40	<.01	.03	.40	<.02
	114	.02	.10	20	<.01	<.01	.30	<.02
	145	.02	.16	20	< .01	.05	.40	<.02
F347	31	.03	3.5	170	<.01	1.30	2.1	.09
	38	.02	.97	370	<.01	2.40	2.3	.11
	46	.01	.45	430	<.01	1.00	2.0	.11
	67	.96	1.6	190	3.90	.59	1.20	.03
	101	.02	.40	30	<.01	.17	.30	<.02
	116	.02	.26	20	.13	1.20	.20	<.02
	131	.02	.10	40	<.01	.30	.30	<.02

Table 8. Chemical analyses of ground-water samples from well-cluster sites, near Ashumet Pond, Massachusetts,

Cluster site	Well depth (ft)	Date	Specific conductance (μS/cm)	pH (standard units)	Oxygen, dissolved (mg/L)	Alkalinity, dissolved, field (mg/L as $CaCO_3$)	Sulfate, dissolved (mg/L)	Nitrogen, nitrate, dissolved (mg/L as N)	Nitrogen, nitrite plus nitrate, dissolved (mg/L as N)
F383	23	9-02-93	82	5.9	11.0	5	7.3	--	--
	30	9-02-93	300	6.1	1.6	3	25	--	--
	40	9-02-93	410	5.5	.10	11	36	--	--
	61	9-02-93	348	6.5	<.05	58	32	--	--
	82	9-02-93	290	6.3	<.05	50	29	--	--
	106	9-02-93	115	5.9	.10	5	18	--	--
	129	9-02-93	150	5.8	4.1	--	20	--	--
F388	37	1-24-94	368	5.8	.30	--	27	--	--
	72	1-24-94	186	6.4	.20	--	14	--	--
F422	45	11-23-93	92	5.7	10.0	5	--	--	--
	65	11-23-93	277	6.2	.10	33	--	--	--
	85	11-23-93	255	6.2	.08	37	--	--	--
	105	11-23-93	140	6.1	.50	29	--	--	--
F432	59	9-14-93	135	5.7	10.5	5	15	<.01	.46
	79	9-14-93	202	5.5	11.5	4	12	<.01	.78
	92	9-14-93	102	5.7	12.5	4	19	<.01	.35
F564	137	9-30-93	73	6.4	10.2	13	4.8	--	--
F567	136	10-20-93	90	6.5	1.5	--	--	--	--
S313	20	8-20-93	77	5.5	9.7	2	6.2	--	--
	38	8-20-93	143	5.5	11.0	2	3.3	--	--
	60	8-20-93	132	5.5	11.8	3	14	--	--
	80	8-20-93	368	5.4	10.5	1	--	--	--
S314	35	9-23-93	448	5.4	1.8	10	26	--	--
	51	9-23-93	400	6.0	.25	59	25	--	--
	75	9-23-93	350	5.8	.15	37	27	--	--
	98	9-23-93	197	5.7	2.7	8	21	--	--
S315	82	8-11-93	76	5.6	11.0	4	9.8	--	--
	126	8-11-93	89	5.5	11.1	5	19	--	--
	149	8-11-93	76	5.8	11.8	8	14	--	--
S316	51	9-24-93	104	4.8	9.2	2	13	<.01	1.1
	66	9-24-93	177	6.0	.10	26	17	.32	2.9
	82	9-24-93	116	5.9	.10	12	12	.08	.18
	100	9-24-93	145	6.4	<.05	36	17	<.01	<.05
	134	9-24-93	162	5.7	3.0	7	11	<.01	3.4
S318	36	9-28-93	410	6.0	.50	78	--	--	--
	64	9-28-93	430	5.9	.35	48	--	--	--
S344	38	9-03-93	160	5.6	8.8	9	14	--	--
	61	9-03-93	348	6.0	.20	5	25	--	--
	80	9-03-93	192	6.5	<.05	44	13	--	--
	100	9-03-93	345	6.4	<.05	79	23	--	--

Cluster site	Well depth (ft)	Nitrogen, ammonia, dissolved, organic (mg/L as N)	Phosphorus, dissolved (mg/L as P)	Boron, dissolved (mg/L)	Iron, dissolved (mg/L)	Manganese, dissolved (mg/L)	Carbon, organic, dissolved (mg/L as C)	Methylene-blue active substances (mg/L)
F383	23	--	0.20	--	<0.01	<0.01	0.40	--
	30	--	.54	--	<.01	.27	1.8	--
	40	--	.05	--	.02	.69	1.9	--
	61	--	1.1	--	1.50	.37	1.3	--
	82	--	.29	--	9.00	.4	1.2	--
	106	--	.22	--	<.01	1.9	5.0	--
	129	--	.05	--	<.01	.19	.30	--
F388	37	--	1.4	--	.02	.24	--	--
	72	--	3.9	--	.01	.71	--	--
F422	45	--	.01	--	.01	.02	--	--
	65	--	.03	--	.01	2.0	--	--
	85	--	.54	--	.01	2.4	--	--
	105	--	.10	--	.01	2.2	--	--
F432	59	<0.01	.02	40	<.01	<.01	--	<0.02
	79	<.01	.01	10	<.01	.02	--	<.02
	92	<.01	.03	<10	<.01	<.01	--	<.02
F564	137	--	.04	--	.02	<.01	--	--
F567	136	--	1.1	--	.01	--	--	--
S313	20	--	.01	--	<.01	<.01	--	--
	38	--	.03	--	<.01	<.01	--	--
	60	--	<.01	--	<.01	<.01	--	--
	80	--	<.01	--	<.01	.05	--	--
S314	35	--	3.8	--	<.01	.03	--	--
	51	--	1.0	--	.02	.02	--	--
	75	--	4.0	--	.02	.13	--	--
	98	--	.47	--	<.01	.31	--	--
S315	82	--	<.01	--	<.01	<.01	--	--
	126	--	.01	--	<.01	<.01	--	--
	149	--	.02	--	<.01	<.01	--	--
S316	51	.02	.31	20	.01	.09	--	<0.02
	66	.01	1.6	120	.02	.10	--	.02
	82	.02	.58	50	.05	.10	--	<.02
	100	.07	1.4	40	17	.16	--	<.02
	134	.01	.36	90	.02	.03	--	<.02
S318	36	--	2.7	--	.01	--	--	--
	64	--	3.7	--	.01	--	--	--
S344	38	--	1.4	--	<.01	.01	--	--
	61	--	1.8	--	.02	.10	--	--
	80	--	1.0	--	9.3	.20	--	--
	100	--	.37	--	17	.49	--	--

Table 9. Chemical analyses of ground-water samples from multilevel samplers, near Ashumet Pond, Massachusetts,

[Analyses were done by the USGS National Water Quality Laboratory in Arvada, Colo. Location of multilevel samplers shown in figure 3. mg/L, milligram

Multilevel sampler		Date	Specific conductance (μS/cm)	pH (standard units)	Oxygen, dissolved (mg/L)	Alkalinity, dissolved, field (mg/L as $CaCO_3$)	Sulfate, dissolved (mg/L)	Nitrogen, nitrate, dissolved (mg/L as N)	Nitrogen, nitrite plus nitrate, dissolved (mg/L as N)
F239	M01-02	9-14-93	88	5.4	8.7	--	5.8	--	--
	M01-03	9-14-93	84	5.4	9.0	--	6.0	--	--
	M01-04	9-14-93	88	5.9	7.9	--	8.7	--	--
	M01-05	9-14-93	170	6.3	2.8	--	16	--	--
	M01-06	9-14-93	340	6.4	.25	--	34	--	--
	M01-07	9-14-93	325	6.4	.30	--	36	--	--
	M01-08	9-14-93	310	6.5	.33	--	37	--	--
	M01-09	9-14-93	300	6.4	.25	--	46	--	--
	M01-10	9-14-93	298	6.4	.30	--	47	--	--
	M01-11	9-14-93	340	6.4	.30	--	46	--	--
	M01-12	9-14-93	310	6.4	.25	--	48	--	--
	M01-13	9-14-93	220	6.2	.30	--	29	--	--
	M01-14	9-14-93	157	6.3	.30	--	15	--	--
	M01-15	9-14-93	107	6.3	.30	--	14	--	--
F300	M02-01	10-04-93	225	6.3	.10	--	9.0	--	--
	M02-02	10-04-93	237	6.4	.30	--	14	--	--
	M02-03	10-04-93	276	6.3	.20	--	25	--	--
	M02-04	10-04-93	250	6.4	.20	--	19	--	--
	M02-05	10-04-93	259	6.4	.15	--	18	--	--
	M02-06	10-04-93	255	6.4	.15	--	25	--	--
	M02-07	10-04-93	228	6.4	.15	--	24	--	--
	M02-08	10-04-93	229	6.3	.20	--	23	--	--
	M02-09	10-04-93	241	6.3	.20	--	24	--	--
	M02-10	10-04-93	188	6.4	.20	--	17	--	--
	M02-11	10-04-93	197	6.3	.20	--	21	--	--
	M02-12	10-04-93	157	6.3	.20	--	12	--	--
	M02-13	10-04-93	172	6.1	.20	--	15	--	--
	M02-14	10-04-93	134	6.3	.15	--	8.6	--	--
	M02-15	10-04-93	97	6.5	.20	--	12	--	--
	M03-01	10-05-93	119	5.0	.20	--	8.6	--	--
	M03-02	10-05-93	89	5.5	9.2	--	8.1	--	--
	M03-03	10-05-93	120	5.5	6.1	--	6.9	--	--
	M03-04	10-05-93	177	5.9	.20	--	15	--	--
	M03-05	10-05-93	211	6.0	.20	--	18	--	--
	M03-06	10-05-93	269	6.3	.15	--	21	--	--
	M03-07	10-05-93	298	6.4	.15	--	24	--	--
	M03-08	10-05-93	337	6.4	.15	--	29	--	--
	M03-09	10-05-93	420	6.4	.10	--	30	--	--
	M03-10	10-05-93	412	6.4	.15	--	32	--	--
	M03-11	10-05-93	347	6.4	.10	--	30	--	--
	M03-12	10-05-93	271	6.5	.10	--	17	--	--
	M03-13	10-05-93	275	6.5	.10	--	20	--	--
	M03-14	10-05-93	270	6.5	.05	--	19	--	--
	M03-15	10-05-93	254	6.6	.15	--	14	--	--
F343	M01-01	10-14-93	368	6.4	<.05	--	30	--	--
	M01-02	10-14-93	256	6.4	<.05	--	22	--	--

September and October 1993

per liter; µg/L, microgram per liter; µS/cm, microsiemen per centimeter at 25 degrees Celsius. <, actual value is less than value shown. --, no analysis]

Multilevel sampler		Nitrogen, ammonia, dissolved, organic (mg/L as N)	Phosphorus, dissolved (mg/L as P)	Boron, dissolved (mg/L)	Iron, dissolved (mg/L)	Manganese, dissolved (mg/L)	Carbon, organic, dissolved (mg/L as C)	Methylene-blue active substances (mg/L)
F239	M01-02	--	<0.01	--	<0.01	0.04	--	--
	M01-03	--	<.01	--	.01	.03	--	--
	M01-04	--	<.01	--	<.01	.02	--	--
	M01-05	--	<.01	--	<.01	.31	--	--
	M01-06	--	<.01	--	.05	3.6	--	--
	M01-07	--	.02	--	.02	4.9	--	--
	M01-08	--	.05	--	.02	4.3	--	--
	M01-09	--	.05	--	.01	5.2	--	--
	M01-10	--	.04	--	.01	4.3	--	--
	M01-11	--	.03	--	.11	2.9	--	--
	M01-12	--	.05	--	<.01	5.8	--	--
	M01-13	--	.06	--	.01	3.0	--	--
	M01-14	--	.08	--	.02	1.5	--	--
	M01-15	--	.08	--	<.01	.49	--	--
F300	M02-01	--	1.0	--	<.01	5.9	--	--
	M02-02	--	.89	--	<.01	7.7	--	--
	M02-03	--	.76	--	<.01	9.2	--	--
	M02-04	--	.66	--	<.01	8.2	--	--
	M02-05	--	.64	--	<.01	8.7	--	--
	M02-06	--	.77	--	<.01	9.3	--	--
	M02-07	--	.93	--	<.01	8.4	--	--
	M02-08	--	.77	--	<.01	8.8	--	--
	M02-09	--	.25	--	<.01	9.4	--	--
	M02-10	--	.08	--	<.01	6.0	--	--
	M02-11	--	.10	--	<.01	3.2	--	--
	M02-12	--	.27	--	<.01	1.5	--	--
	M02-13	--	.15	--	<.01	1.4	--	--
	M02-14	--	.08	--	<.01	.24	--	--
	M02-15	--	.10	--	<.01	.09	--	--
	M03-01	--	<.01	--	.01	.12	--	--
	M03-02	--	<.01	--	<.01	.02	--	--
	M03-03	--	<.01	--	<.01	.05	--	--
	M03-04	--	.01	--	<.01	.01	--	--
	M03-05	--	.02	--	<.01	.57	--	--
	M03-06	--	.04	--	<.01	4.0	--	--
	M03-07	--	1.1	--	<.01	6.5	--	--
	M03-08	--	1.3	--	<.01	8.7	--	--
	M03-09	--	1.2	--	<.01	12	--	--
	M03-10	--	1.0	--	<.01	12	--	--
	M03-11	--	1.2	--	<.01	10	--	--
	M03-12	--	1.3	--	<.01	7.8	--	--
	M03-13	--	1.4	--	<.01	9.1	--	--
	M03-14	--	1.3	--	<.01	7.4	--	--
	M03-15	--	1.2	--	<.01	6.2	--	--
F343	M01-01	--	2.6	--	25	.32	--	--
	M01-02	--	1.8	--	23	.36	--	--

Table 9. Chemical analyses of ground-water samples from multi-level samplers, near Ashumet Pond, Massachusetts,

Multilevel sampler		Date	Specific conductance (μS/cm)	pH (standard units)	Oxygen, dissolved (mg/L)	Alkalinity, dissolved, field (mg/L as $CaCO_3$)	Sulfate, dissolved (mg/L)	Nitrogen, nitrate, dissolved (mg/L as N)	Nitrogen, nitrite plus nitrate, dissolved (mg/L as N)
F343	M01-03	10-14-93	162	6.6	<0.05	--	14	--	--
	M01-04	10-14-93	127	6.4	<.05	--	11	--	--
	M01-05	10-14-93	92	5.9	1.9	--	7.2	--	--
	M01-06	10-14-93	92	5.8	4.6	--	8.5	--	--
	M01-07	10-14-93	120	5.7	6.4	--	13	--	--
	M01-08	10-14-93	129	5.6	7.3	--	16	--	--
	M01-09	10-14-93	77	5.6	7.7	--	14	--	--
	M01-10	10-14-93	68	5.6	7.3	--	12	--	--
	M01-11	10-14-93	65	5.6	6.9	--	12	--	--
	M01-12	10-14-93	77	5.6	7.0	--	15	--	--
	M01-13	10-14-93	82	5.6	7.7	--	16	--	--
	M01-14	10-14-93	85	5.7	6.5	--	14	--	--
	M01-15	10-14-93	80	5.8	6.8	--	16	--	--
	M02-01	10-15-93	317	6.1	.20	--	24	--	--
	M02-02	10-15-93	427	6.2	.30	--	26	--	--
	M02-03	10-15-93	403	6.2	.30	--	26	--	--
	M02-04	10-15-93	340	6.3	.20	--	27	--	--
	M02-05	10-15-93	345	6.4	.30	--	26	--	--
	M02-06	10-15-93	427	6.4	.30	--	28	--	--
	M02-07	10-15-93	334	6.7	<.05	--	26	--	--
	M02-08	10-15-93	343	6.8	<.05	--	30	--	--
	M02-09	10-15-93	374	6.7	<.05	--	33	--	--
	M02-10	10-15-93	355	6.8	<.05	--	35	--	--
	M02-11	10-15-93	334	6.8	<.05	--	28	--	--
	M02-12	10-15-93	340	6.7	<.05	--	29	--	--
	M02-13	10-15-93	318	6.7	<.05	--	28	--	--
	M02-14	10-15-93	303	6.6	<.05	--	27	--	--
	M02-15	10-15-93	233	6.7	<.05	--	19	--	--
	M03-02	10-22-93	77	5.4	1.0	--	5.5	--	--
	M03-03	10-22-93	74	5.6	1.0	--	4.7	--	--
	M03-04	10-22-93	120	5.6	1.5	--	7.4	--	--
	M03-05	10-22-93	206	5.5	2.0	--	14	--	--
	M03-06	10-22-93	269	5.6	1.5	--	18	--	--
	M03-07	10-22-93	322	5.7	1.0	--	21	--	--
	M03-08	10-22-93	342	5.8	.30	--	22	--	--
	M03-09	10-22-93	346	5.9	.40	--	21	--	--
	M03-10	10-22-93	360	6.0	.40	--	22	--	--
	M03-11	10-22-93	362	6.1	.25	--	22	--	--
	M03-12	10-22-93	394	6.2	.40	--	22	--	--
	M03-13	10-22-93	503	6.1	.40	--	25	--	--
	M03-14	10-22-93	541	6.1	.40	--	27	--	--
	M03-15	10-22-93	523	6.3	.20	--	27	--	--
F347	M01-01	10-19-93	74	5.1	10	--	5.8	--	--
	M01-02	10-19-93	105	5.2	8.1	--	--	--	--
	M01-03	10-19-93	123	5.4	6.2	--	11	--	--
	M01-04	10-19-93	140	5.6	4.3	--	--	--	--
	M01-05	10-19-93	302	5.7	.20	--	24	--	--
	M01-06	10-19-93	336	5.8	.20	--	--	--	--

September and October 1993--*Continued*

Multilevel sampler		Nitrogen, ammonia, dissolved, organic (mg/L as N)	Phosphorus, dissolved (mg/L as P)	Boron, dissolved (mg/L)	Iron, dissolved (mg/L)	Manganese, dissolved (mg/L)	Carbon, organic, dissolved (mg/L as C)	Methylene-blue active substances (mg/L)
F343	M01-03	--	<0.75	--	12	0.21	--	--
	M01-04	--	.64	--	8.2	.11	--	--
	M01-05	--	.04	--	<.01	.04	--	--
	M01-06	--	.03	--	<.01	.02	--	--
	M01-07	--	.07	--	<.01	.05	--	--
	M01-08	--	.09	--	<.01	.03	--	--
	M01-09	--	.10	--	.01	.01	--	--
	M01-10	--	.20	--	.01	<.01	--	--
	M01-11	--	.21	--	<.01	<.01	--	--
	M01-12	--	.21	--	<.01	.28	--	--
	M01-13	--	.17	--	.01	.01	--	--
	M01-14	--	.22	--	<.01	.03	--	--
	M01-15	--	.21	--	<.01	.25	--	--
	M02-01	--	.28	--	.01	.31	--	--
	M02-02	--	.15	--	.02	.39	--	--
	M02-03	--	.08	--	.03	.31	--	--
	M02-04	--	.08	--	.04	.18	--	--
	M02-05	--	.07	--	.03	.18	--	--
	M02-06	--	.07	--	.03	.20	--	--
	M02-07	--	3.3	--	15	.21	--	--
	M02-08	--	2.9	--	12	.22	--	--
	M02-09	--	3.0	--	18	.22	--	--
	M02-10	--	3.5	--	13	.19	--	--
	M02-11	--	2.8	--	16	.19	--	--
	M02-12	--	<2.8	--	15	.21	--	--
	M02-13	--	2.8	--	20	.22	--	--
	M02-14	--	2.7	--	21	.27	--	--
	M02-15	--	2.6	--	17	.19	--	--
	M03-02	--	.02	--	<.01	.02	--	--
	M03-03	--	.07	--	<.01	.09	--	--
	M03-04	--	.50	--	<.01	<.01	--	--
	M03-05	--	.68	--	<.01	<.01	--	--
	M03-06	--	.71	--	.02	.07	--	--
	M03-07	--	4.3	--	.03	.24	--	--
	M03-08	--	.51	--	.02	.42	--	--
	M03-09	--	.39	--	.01	.51	--	--
	M03-10	--	.35	--	.02	.53	--	--
	M03-11	--	.29	--	.02	.50	--	--
	M03-12	--	.23	--	.02	.50	--	--
	M03-13	--	.16	--	.02	.65	--	--
	M03-14	--	.14	--	.02	.65	--	--
	M03-15	--	.14	--	.01	.58	--	--
F347	M01-01	--	.01	--	<.01	.08	--	--
	M01-02	--	.01	--	.01	--	--	--
	M01-03	--	.09	--	<.01	.03	--	--
	M01-04	--	.95	--	.01	--	--	--
	M01-05	--	1.2	--	<.01	1.4	--	--
	M01-06	--	1.6	--	.01	--	--	--

Table 9. Chemical analyses of ground-water samples from multi-level samplers, near Ashumet Pond, Massachusetts,

Multilevel sampler		Date	Specific conductance (μS/cm)	pH (standard units)	Oxygen, dissolved (mg/L)	Alkalinity, dissolved, field (mg/L as $CaCO_3$)	Sulfate, dissolved (mg/L)	Nitrogen, nitrate, dissolved (mg/L as N)	Nitrogen, nitrite plus nitrate, dissolved (mg/L as N)
F347	M01-07	10-19-93	330	5.9	0.20	--	19	--	--
	M01-08	10-19-93	362	5.9	.10	--	--	--	--
	M01-09	10-19-93	353	6.0	.35	--	22	--	--
	M01-10	10-19-93	413	5.9	.15	--	--	--	--
	M01-11	10-19-93	335	6.0	1.2	--	31	--	--
	M01-12	10-19-93	266	6.1	.40	--	--	--	--
	M01-13	10-19-93	238	6.1	.20	--	28	--	--
	M01-14	10-19-93	215	6.2	.20	--	--	--	--
	M01-15	10-19-93	271	6.2	.30	--	26	--	--
	M06-01	10-19-93	234	6.3	.30	--	--	--	--
	M06-02	10-19-93	351	6.3	.40	--	26	--	--
	M06-03	10-19-93	244	6.4	.90	--	--	--	--
	M06-04	10-19-93	229	6.5	.55	--	18	--	--
	M06-05	10-19-93	240	6.3	.70	--	--	--	--
	M06-06	10-19-93	246	6.4	.40	--	27	--	--
	M06-07	10-19-93	184	6.6	<.05	--	--	--	--
	M06-08	10-19-93	136	6.6	<.05	--	18	--	--
	M06-09	10-19-93	128	6.4	<.05	--	--	--	--
	M06-10	10-19-93	117	6.3	.80	--	16	--	--
	M06-11	10-19-93	104	6.3	1.7	--	--	--	--
	M06-12	10-19-93	84	6.0	3.1	--	13	--	--
	M06-13	10-19-93	87	5.8	3.2	--	--	--	--
	M06-14	10-19-93	66	5.8	7.2	--	11	--	--
	M06-15	10-19-93	74	6.1	9.9	--	--	--	--
F424	M01-01	10-06-93	371	6.4	.70	--	36	--	--
	M01-02	10-06-93	386	6.4	.70	--	45	--	--
	M01-03	10-06-93	377	6.4	.40	--	47	--	--
	M01-04	10-06-93	328	6.4	1.0	--	38	--	--
	M01-05	10-06-93	261	6.4	.85	--	29	--	--
	M01-06	10-06-93	295	6.3	.50	--	35	--	--
	M01-07	10-06-93	300	6.4	.70	--	30	--	--
	M01-08	10-06-93	273	6.3	1.1	--	29	--	--
	M01-09	10-06-93	277	6.2	.70	--	26	--	--
	M01-10	10-06-93	255	6.0	4.0	--	23	--	--
	M01-11	10-06-93	243	5.9	3.3	--	24	--	--
	M01-12	10-06-93	205	5.9	3.3	--	24	--	--
	M01-13	10-06-93	180	5.9	3.6	--	21	--	--
	M01-14	10-06-93	161	6.0	3.5	--	15	--	--
	M01-15	10-06-93	125	6.3	4.2	--	11	--	--
	M02-01	10-06-93	108	5.4	6.1	--	12	--	--
	M02-02	10-06-93	67	5.3	5.1	--	8.6	--	--
	M02-03	10-06-93	59	5.4	7.9	--	8.1	--	--
	M02-04	10-06-93	56	5.6	8.2	--	8.0	--	--
	M02-05	10-06-93	54	5.8	8.4	--	7.9	--	--
	M02-06	10-06-93	74	5.9	8.2	--	7.9	--	--
	M02-07	10-06-93	130	5.9	6.3	--	9.4	--	--
	M02-08	10-06-93	204	5.9	5.8	--	15	--	--
	M02-09	10-06-93	291	6.0	3.0	--	23	--	--

Multilevel sampler		Nitrogen, ammonia, dissolved, organic (mg/L as N)	Phosphorus, dissolved (mg/L as P)	Boron, dissolved (mg/L)	Iron, dissolved (mg/L)	Manganese, dissolved (mg/L)	Carbon, organic, dissolved (mg/L as C)	Methylene-blue active substances (mg/L)
F347	M01-07	--	1.9	--	<0.01	1.3	--	--
	M01-08	--	1.9	--	.02	--	--	--
	M01-09	--	1.5	--	<.01	1.7	--	--
	M01-10	--	1.0	--	.01	--	--	--
	M01-11	--	.79	--	.01	1.9	--	--
	M01-12	--	.61	--	.01	--	--	--
	M01-13	--	.54	--	.01	.57	--	--
	M01-14	--	.53	--	.01	--	--	--
	M01-15	--	.41	--	<.01	.49	--	--
	M06-01	--	.64	--	.01	--	--	--
	M06-02	--	.47	--	<.01	.94	--	--
	M06-03	--	.16	--	.01	--	--	--
	M06-04	--	1.8	--	.02	.77	--	--
	M06-05	--	1.2	--	.04	--	--	--
	M06-06	--	1.3	--	.15	1.3	--	--
	M06-07	--	3.6	--	1.1	--	--	--
	M06-08	--	1.6	--	6.9	.33	--	--
	M06-09	--	1.6	--	4.4	--	--	--
	M06-10	--	1.0	--	.01	1.2	--	--
	M06-11	--	.55	--	.01	--	--	--
	M06-12	--	.37	--	<.01	.68	--	--
	M06-13	--	.37	--	.01	--	--	--
	M06-14	--	.24	--	.01	.14	--	--
	M06-15	--	.27	--	.03	--	--	--
F424	M01-01	--	1.1	--	<.01	2.8	--	--
	M01-02	--	.94	--	.04	3.9	--	--
	M01-03	--	.82	--	<.01	5.0	--	--
	M01-04	--	.79	--	<.01	4.9	--	--
	M01-05	--	.78	--	<.01	3.7	--	--
	M01-06	--	.62	--	.04	4.4	--	--
	M01-07	--	.55	--	.03	4.3	--	--
	M01-08	--	.41	--	.04	6.3	--	--
	M01-09	--	.37	--	<.01	6.5	--	--
	M01-10	--	.34	--	<.01	6.5	--	--
	M01-11	--	.32	--	<.01	6.2	--	--
	M01-12	--	.30	--	<.01	5.9	--	--
	M01-13	--	.27	--	<.01	4.3	--	--
	M01-14	--	.21	--	<.01	2.0	--	--
	M01-15	--	.10	--	<.01	.45	--	--
	M02-01	--	.01	--	.06	.05	--	--
	M02-02	--	<.01	--	<.01	<.01	--	--
	M02-03	--	.01	--	<.01	<.01	--	--
	M02-04	--	<.01	--	<.01	<.01	--	--
	M02-05	--	.02	--	<.01	<.01	--	--
	M02-06	--	.02	--	<.01	<.01	--	--
	M02-07	--	.03	--	<.01	<.01	--	--
	M02-08	--	.39	--	<.01	<.01	--	--
	M02-09	--	.62	--	<.01	.66	--	--

Table 9. Chemical analyses of ground-water samples from multi-level samplers, near Ashumet Pond, Massachusetts,

Multilevel sampler		Date	Specific conductance (μS/cm)	pH (standard units)	Oxygen, dissolved (mg/L)	Alkalinity, dissolved, field (mg/L as $CaCO_3$)	Sulfate, dissolved (mg/L)	Nitrogen, nitrate, dissolved (mg/L as N)	Nitrogen, nitrite plus nitrate, dissolved (mg/L as N)
F424	M02-10	10-06-93	351	6.0	0.50	--	32	--	--
	M02-11	10-06-93	358	6.2	.40	--	37	--	--
	M02-12	10-06-93	360	6.3	.60	--	33	--	--
	M02-13	10-06-93	384	6.3	.50	--	31	--	--
	M02-14	10-06-93	389	6.3	.20	--	34	--	--
	M02-15	10-06-93	397	6.3	.70	--	34	--	--
F432	M01-01	10-01-93	99	4.7	8.2	--	7.0	--	--
	M01-03	10-01-93	63	5.1	8.6	--	7.5	--	--
	M01-06	10-01-93	97	5.7	8.4	--	11	--	--
	M01-07	10-01-93	143	5.8	8.9	--	15	--	--
	M01-08	10-01-93	139	5.9	8.5	--	17	--	--
	M01-09	10-01-93	147	5.8	8.4	--	13	--	--
	M01-10	10-01-93	248	5.6	9.3	--	11	--	--
	M01-11	10-01-93	128	5.8	8.8	--	22	--	--
	M01-13	10-01-93	113	5.8	9.2	--	18	--	--
	M01-15	10-01-93	77	6.0	9.0	--	13	--	--
F512	M01-02	10-18-93	64	5.6	9.1	--	--	--	--
	M01-03	10-18-93	58	5.7	8.7	--	--	--	--
	M01-04	10-18-93	60	5.7	9.3	--	--	--	--
	M01-05	10-18-93	64	5.8	11.3	--	--	--	--
	M01-06	10-18-93	87	6.0	11.3	--	--	--	--
	M01-07	10-18-93	140	6.0	7.3	--	--	--	--
	M01-08	10-18-93	245	6.0	2.4	--	--	--	--
	M01-09	10-18-93	318	6.0	.70	--	--	--	--
	M01-10	10-18-93	337	6.0	.50	--	--	--	--
	M01-11	10-18-93	390	5.8	.60	--	--	--	--
	M01-12	10-18-93	408	5.6	.40	--	--	--	--
	M01-13	10-18-93	402	5.6	.60	--	--	--	--
	M01-14	10-18-93	421	5.6	.60	--	--	--	--
	M01-15	10-18-93	410	5.4	.50	--	--	--	--
F564	M01-01	9-30-93	148	5.4	4.0	--	13	--	--
	M01-02	9-30-93	70	5.6	8.4	--	7.5	--	--
	M01-03	9-30-93	84	5.8	7.2	--	6.5	--	--
	M01-04	9-30-93	151	6.4	3.7	--	3.8	--	--
	M01-05	9-30-93	110	6.6	2.9	--	5.2	--	--
	M01-06	9-30-93	112	6.6	4.3	--	8.5	--	--
	M01-07	9-30-93	123	6.2	3.0	--	11	--	--
	M01-08	9-30-93	160	6.4	3.7	--	15	--	--
	M01-09	9-30-93	176	6.3	3.3	--	16	--	--
	M01-10	9-30-93	92	5.7	9.0	--	16	--	--
	M01-11	9-30-93	64	5.9	8.8	--	12	--	--
	M01-12	9-30-93	87	5.7	3.6	--	13	--	--
	M01-13	9-30-93	112	6.4	2.7	--	11	--	--
	M01-14	9-30-93	69	6.3	9.3	--	7.6	--	--
	M01-15	9-30-93	62	6.4	9.2	--	4.5	--	--
F566	M01-01	10-07-94	305	6.3	.40	--	38	--	--

September and October 1993--*Continued*

Multilevel sampler		Nitrogen, ammonia, dissolved, organic (mg/L as N)	Phosphorus, dissolved (mg/L as P)	Boron, dissolved (mg/L)	Iron, dissolved (mg/L)	Manganese, dissolved (mg/L)	Carbon, organic, dissolved (mg/L as C)	Methylene-blue active substances (mg/L)
F424	M02-10	--	0.93	--	<0.01	2.3	--	--
	M02-11	--	1.1	--	.01	2.5	--	--
	M02-12	--	1.2	--	<.01	3.7	--	--
	M02-13	--	1.3	--	<.01	3.0	--	--
	M02-14	--	1.2	--	<.01	2.5	--	--
	M02-15	--	1.2	--	<.01	3.1	--	--
F432	M01-01	--	<.01	--	<.01	.12	--	--
	M01-03	--	<.01	--	<.01	.05	--	--
	M01-06	--	<.01	--	<.01	<.01	--	--
	M01-07	--	<.01	--	<.01	<.01	--	--
	M01-08	--	<.01	--	<.01	.01	--	--
	M01-09	--	<.01	--	<.01	.02	--	--
	M01-10	--	<.01	--	<.01	.01	--	--
	M01-11	--	.02	--	<.01	.01	--	--
	M01-13	--	<.01	--	<.01	.08	--	--
	M01-15	--	.02	--	<.01	<.01	--	--
F512	M01-02	--	<.05	--	.01	--	--	--
	M01-03	--	<.05	--	.01	--	--	--
	M01-04	--	.04	--	.01	--	--	--
	M01-05	--	.14	--	.01	--	--	--
	M01-06	--	.37	--	.01	--	--	--
	M01-07	--	1.6	--	.02	--	--	--
	M01-08	--	.49	--	.02	--	--	--
	M01-09	--	.41	--	.01	--	--	--
	M01-10	--	.34	--	.01	--	--	--
	M01-11	--	.24	--	.01	--	--	--
	M01-12	--	.14	--	.01	--	--	--
	M01-13	--	.15	--	.01	--	--	--
	M01-14	--	.05	--	.02	--	--	--
	M01-15	--	.02	--	.02	--	--	--
F564	M01-01	--	<.01	--	<.01	.18	--	--
	M01-02	--	<.01	--	<.01	.05	--	--
	M01-03	--	.34	--	<.01	.04	--	--
	M01-04	--	1.4	--	<.01	.88	--	--
	M01-05	--	1.7	--	<.01	.84	--	--
	M01-06	--	1.1	--	<.01	1.2	--	--
	M01-07	--	.26	--	<.01	4.0	--	--
	M01-08	--	.15	--	<.01	4.3	--	--
	M01-09	--	.21	--	<.01	4.0	--	--
	M01-10	--	.04	--	<.01	1.4	--	--
	M01-11	--	.06	--	<.01	.14	--	--
	M01-12	--	.01	--	<.01	.12	--	--
	M01-13	--	.06	--	<.01	.02	--	--
	M01-14	--	.03	--	<.01	<.01	--	--
	M01-15	--	.04	--	<.01	<.01	--	--
F566	M01-01	--	1.2	--	.49	1.3	--	--

Table 9. Chemical analyses of ground-water samples from multi-level samplers, near Ashumet Pond, Massachusetts,

Multilevel sampler		Date	Specific conductance (μS/cm)	pH (standard units)	Oxygen, dissolved (mg/L)	Alkalinity, dissolved, field (mg/L as $CaCO_3$)	Sulfate, dissolved (mg/L)	Nitrogen, nitrate, dissolved (mg/L as N)	Nitrogen, nitrite plus nitrate, dissolved (mg/L as N)
F566	M01-02	10-07-94	315	6.3	0.50	--	40	--	--
	M01-03	10-07-94	291	6.5	9.9	--	33	--	--
	M01-04	10-07-94	251	6.4	.40	--	23	--	--
	M01-05	10-07-94	263	6.3	.60	--	25	--	--
	M01-06	10-07-94	168	6.4	.30	--	12	--	--
	M01-07	10-07-94	146	6.3	.70	--	14	--	--
	M01-08	10-07-94	162	6.0	.50	--	21	--	--
	M01-09	10-07-94	127	6.0	.80	--	10	--	--
	M01-10	10-07-94	112	6.0	1.5	--	12	--	--
	M01-11	10-07-94	103	6.2	3.5	--	14	--	--
	M01-12	10-07-94	106	5.7	1.0	--	13	--	--
	M01-13	10-07-94	103	6.2	.40	--	12	--	--
	M01-14	10-07-94	123	6.4	.50	--	13	--	--
	M01-15	10-07-94	119	6.3	.90	--	16	--	--
	M02-01	10-08-93	41	4.8	1.9	--	5.0	--	--
	M02-02	10-08-93	46	5.0	3.4	--	5.3	--	--
	M02-03	10-08-93	79	5.5	5.3	--	6.5	--	--
	M02-04	10-08-93	220	5.7	2.4	--	19	--	--
	M02-05	10-08-93	303	5.7	.40	--	29	--	--
	M02-06	10-08-93	357	5.8	.30	--	34	--	--
	M02-07	10-08-93	389	5.9	.20	--	34	--	--
	M02-08	10-08-93	393	5.9	.20	--	34.	--	--
	M02-09	10-08-93	406	6.0	.60	--	35	--	--
	M02-10	10-08-93	390	6.1	.50	--	36	--	--
	M02-11	10-08-93	398	6.1	.60	--	36	--	--
	M02-12	10-08-93	370	6.1	.40	--	37	--	--
	M02-13	10-08-93	305	6.2	.50	--	39	--	--
	M02-14	10-08-93	317	6.3	.30	--	41	--	--
	M02-15	10-08-93	331	6.3	.70	--	42	--	--
F567	M01-01	10-20-93	69	5.3	8.9	--	7.2	--	--
	M01-02	10-20-93	67	5.6	9.3	--	--	--	--
	M01-03	10-20-93	62	5.9	9.3	--	5.6	--	--
	M01-04	10-20-93	173	5.9	5.6	--	--	--	--
	M01-05	10-20-93	266	5.9	3.2	--	18	--	--
	M01-06	10-20-93	399	5.9	1.3	--	--	--	--
	M01-07	10-20-93	422	6.0	.30	--	29	--	--
	M01-08	10-20-93	365	6.2	1.3	--	--	--	--
	M01-09	10-20-93	328	6.2	1.1	--	18	--	--
	M01-10	10-20-93	309	6.3	1.7	--	--	--	--
	M01-11	10-20-93	277	6.4	1.2	--	18	--	--
	M01-12	10-20-93	251	6.6	.20	--	--	--	--
	M01-13	10-20-93	290	6.6	.30	--	29	--	--
	M01-14	10-20-93	178	6.7	.50	--	--	--	--
	M01-15	10-20-93	168	6.8	.80	--	9.5	--	--
	M02-01	10-20-93	162	6.9	.60	--	--	--	--
	M02-02	10-20-93	161	6.8	.60	--	9.1	--	--
	M02-03	10-20-93	164	6.7	.80	--	--	--	--
	M02-04	10-20-93	172	6.7	.60	--	13	--	--

September and October 1993--*Continued*

Multilevel sampler		Nitrogen, ammonia, dissolved, organic (mg/L as N)	Phosphorus, dissolved (mg/L as P)	Boron, dissolved (mg/L)	Iron, dissolved (mg/L)	Manganese, dissolved (mg/L)	Carbon, organic, dissolved (mg/L as C)	Methylene-blue active substances (mg/L)
F566	M01-02	--	1.2	--	0.10	1.3	--	--
	M01-03	--	1.6	--	.06	.8	--	--
	M01-04	--	1.3	--	<.01	.65	--	--
	M01-05	--	1.2	--	.14	.79	--	--
	M01-06	--	1.4	--	.09	.58	--	--
	M01-07	--	.85	--	<.01	1.0	--	--
	M01-08	--	.45	--	<.01	1.7	--	--
	M01-09	--	.39	--	<.01	1.4	--	--
	M01-10	--	.33	--	<.01	.92	--	--
	M01-11	--	.28	--	.32	.71	--	--
	M01-12	--	.21	--	<.01	1.6	--	--
	M01-13	--	.27	--	<.01	.49	--	--
	M01-14	--	.14	--	<.01	.07	--	--
	M01-15	--	.08	--	<.01	.05	--	--
	M02-01	--	<.01	--	<.01	<.01	--	--
	M02-02	--	.02	--	<.01	<.01	--	--
	M02-03	--	.02	--	<.01	<.01	--	--
	M02-04	--	.14	--	<.01	.01	--	--
	M02-05	--	.43	--	<.01	.04	--	--
	M02-06	--	.64	--	<.01	.47	--	--
	M02-07	--	.72	--	.03	1.0	--	--
	M02-08	--	.79	--	.02	1.1	--	--
	M02-09	--	.84	--	<.01	1.2	--	--
	M02-10	--	.99	--	<.01	1.4	--	--
	M02-11	--	.91	--	<.01	1.8	--	--
	M02-12	--	.97	--	<.01	1.6	--	--
	M02-13	--	.95	--	.01	.94	--	--
	M02-14	--	1.5	--	.02	.99	--	--
	M02-15	--	1.3	--	<.01	1.2	--	--
F567	M01-01	--	.03	--	<.01	.02	--	--
	M01-02	--	.13	--	.02	--	--	--
	M01-03	--	.97	--	<.01	<.01	--	--
	M01-04	--	2.7	--	.02	--	--	--
	M01-05	--	2.6	--	.02	.1	--	--
	M01-06	--	2.3	--	.03	--	--	--
	M01-07	--	2.1	--	.02	1.5	--	--
	M01-08	--	2.0	--	.02	--	--	--
	M01-09	--	1.0	--	.02	1.1	--	--
	M01-10	--	.87	--	.02	--	--	--
	M01-11	--	2.2	--	.03	.62	--	--
	M01-12	--	4.5	--	.03	--	--	--
	M01-13	--	4.2	--	.02	.58	--	--
	M01-14	--	6.2	--	.03	--	--	--
	M01-15	--	5.6	--	.02	.24	--	--
	M02-01	--	5.7	--	.12	--	--	--
	M02-02	--	4.7	--	.27	.27	--	--
	M02-03	--	3.3	--	.18	--	--	--
	M02-04	--	4.5	--	.23	.41	--	--

Table 9. Chemical analyses of ground-water samples from multi-level samplers, near Ashumet Pond, Massachusetts,

Multilevel sampler		Date	Specific conductance (μS/cm)	pH (standard units)	Oxygen, dissolved (mg/L)	Alkalinity, dissolved, field (mg/L as $CaCO_3$)	Sulfate, dissolved (mg/L)	Nitrogen, nitrate, dissolved (mg/L as N)	Nitrogen, nitrite plus nitrate, dissolved (mg/L as N)
F567	M02-05	10-20-93	185	6.7	0.70	--	--	--	--
	M02-06	10-20-93	167	6.7	.40	--	13	--	--
	M02-07	10-20-93	170	6.6	1.1	--	--	--	--
	M02-08	10-20-93	161	6.5	1.2	--	14	--	--
	M02-09	10-20-93	108	6.5	1.1	--	--	--	--
	M02-10	10-20-93	93	6.4	.60	--	10	--	--
	M02-11G	10-20-93	95	6.6	.60	--	--	--	--
	M02-12	10-20-93	92	6.6	.60	--	9.7	--	--
	M02-13	10-20-93	89	6.6	.60	--	--	--	--
	M02-14	10-20-93	89	6.5	1.3	--	10	--	--
	M02-15	10-20-93	93	6.5	7.4	--	--	--	--
S317	M01-02	10-21-93	452	6.2	6.2	--	36	--	--
	M01-03	10-21-93	430	6.1	.20	--	30	--	--
	M01-04	10-21-93	452	6.0	.25	--	26	--	--
	M01-05	10-21-93	445	6.0	.30	--	26	--	--
	M01-06	10-21-93	446	5.9	.50	--	26	--	--
	M01-07	10-21-93	443	5.9	.40	--	26	--	--
	M01-08	10-21-93	449	5.8	.60	--	26	--	--
	M01-09	10-21-93	430	5.5	.40	--	25	--	--
	M01-10	10-21-93	423	5.4	.40	--	22	--	--
	M01-11	10-21-93	427	5.4	.30	--	23	--	--
	M01-12	10-21-93	410	5.4	.20	--	24	--	--
	M01-13	10-21-93	410	5.6	.30	--	25	--	--
	M01-14	10-21-93	444	5.8	.70	--	27	--	--
	M01-15	10-21-93	445	5.8	.15	--	30	--	--
	M02-01	10-21-93	447	5.8	.40	--	27	--	--
	M02-02	10-21-93	480	5.8	.20	--	--	--	--
	M02-03	10-21-93	491	5.9	.30	--	26	--	--
	M02-04	10-21-93	476	6.0	.80	--	--	--	--
	M02-05	10-21-93	470	6.0	.60	--	25	--	--
	M02-06	10-21-93	439	5.8	.30	--	--	--	--
	M02-07	10-21-93	445	5.8	.40	--	24	--	--
	M02-08	10-21-93	413	5.8	.80	--	--	--	--
	M02-09	10-21-93	407	5.8	5.8	--	24	--	--
	M02-10	10-21-93	394	5.6	5.8	--	--	--	--
	M02-11	10-21-93	448	5.6	8.1	--	--	--	--
	M02-12	10-21-93	430	5.4	8.4	--	--	--	--
	M02-13	10-21-93	461	5.6	9.4	--	23	--	--
	M02-11	10-21-93	430	5.6	7.2	--	23	--	--
	M02-15	10-21-93	403	5.4	8.2	--	20	--	--

September and October 1993--*Continued*

Multilevel sampler		Nitrogen, ammonia, dissolved, organic (mg/L as N)	Phosphorus, dissolved (mg/L as P)	Boron, dissolved (mg/L)	Iron, dissolved (mg/L)	Manganese, dissolved (mg/L)	Carbon, organic, dissolved (mg/L as C)	Methylene-blue active substances (mg/L)
F567	M02-05	--	5.1	--	0.12	--	--	--
	M02-06	--	4.5	--	.10	0.38	--	--
	M02-07	--	3.0	--	.06	--	--	--
	M02-08	--	1.6	--	.08	.79	--	--
	M02-09	--	1.2	--	.10	--	--	--
	M02-10	--	1.2	--	.03	.40	--	--
	M02-11	--	1.7	--	.09	--	--	--
	M02-12	--	1.6	--	.03	.42	--	--
	M02-13	--	1.7	--	.04	--	--	--
	M02-14	--	1.1	--	.02	.65	--	--
	M02-15	--	.60	--	.02	--	--	--
S317	M01-02	--	2.7	--	.01	.03	--	--
	M02-02	--	2.6	--	.01	--	--	--
	M01-03	--	2.1	--	.02	.04	--	--
	M01-04	--	1.8	--	.02	.04	--	--
	M01-05	--	2.0	--	<.01	.03	--	--
	M01-06	--	2.1	--	<.01	.03	--	--
	M01-07	--	2.1	--	.02	.03	--	--
	M01-08	--	3.3	--	<.01	.03	--	--
	M01-09	--	4.1	--	.03	.04	--	--
	M01-10	--	4.1	--	.02	.05	--	--
	M01-11	--	4.4	--	.02	.04	--	--
	M01-12	--	3.2	--	.02	.04	--	--
	M01-13	--	2.7	--	<.01	.04	--	--
	M01-14	--	2.5	--	.01	.03	--	--
	M01-15	--	2.4	--	.02	.04	--	--
	M02-01	--	2.5	--	.01	.04	--	--
	M02-02	--	2.6	--	.01	--	--	--
	M02-03	--	2.3	--	<.01	.07	--	--
	M02-04	--	2.4	--	.01	--	--	--
	M02-05	--	2.1	--	<.01	.11	--	--
	M02-06	--	2.6	--	.01	--	--	--
	M02-07	--	2.4	--	<.01	.22	--	--
	M02-08	--	2.4	--	.01	--	--	--
	M02-09	--	2.3	--	<.01	.29	--	--
	M02-10	--	2.7	--	.01	--	--	--
	M02-11	--	1.4	--	.01	--	--	--
	M02-12	--	2.3	--	.01	--	--	--
	M02-13	--	.59	--	<.01	.24	--	--
	M02-11	--	1.7	--	<.01	.18	--	--
	M02-15	--	.41	--	<.01	.43	--	--

Table 10. Chemical analyses of ground-water samples from pond-bottom drive-point minipiezometers, near Ashumet Pond,

[Analyses were done by the USGS National Water-Quality Laboratory in Arvada, Colo. Drive-point site shown in figure 3. mg/L, milligram per liter;

Drive-point site	Date	Specific conductance (μS/cm)	pH (standard units)	Oxygen, dissolved (mg/L)	Alkalinity, dissolved, field (mg/L as $CaCO_3$)	Sulfate, dissolved (mg/L)	Nitrogen, nitrate, dissolved (mg/L as N)	Nitrogen, nitrite plus nitrate, dissolved (mg/L as N)
DP 500S	12-15-93	99	--	--	--	--	--	--
DP 400S	12-09-93	247	6.3	0.40	--	36	--	--
DP 300S	12-09-93	305	6.8	.30	--	33	--	--
DP 200S	12-09-93	366	6.8	.15	--	35	--	--
DP 100S	12-09-93	224	6.5	.15	--	16	--	--
DP 0	12-03-93	192	6.0	.35	--	16	--	--
DP 100N	12-07-93	375	6.2	.25	--	27	--	--
DP 200N	12-07-93	299	6.4	.35	--	24	--	--
DP 300N	12-07-93	178	6.5	.40	--	15	--	--
DP 400N	12-07-93	210	6.5	.40	--	17	--	--
DP 500N	12-07-93	334	6.4	.10	--	27	--	--
DP 600N	12-08-93	446	6.3	.20	--	30	--	--
DP 700N	12-08-93	147	6.6	.30	--	11	--	--
DP 800N	12-08-93	113	6.4	.15	--	9.3	--	--
DP 900N	12-08-93	103	5.2	2.0	--	8.2	--	--
DP 950N	12-08-93	68	4.8	2.5	--	7.8	--	--

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µg/L microgram per liter; µS/cm, microsiemen per centimeter at 25 degrees Celsius. <, actual value is less than value shown. --, no analysis]

Drive-point site	Nitrogen, ammonia, dissolved, organic (mg/L as N)	Phosphorus, dissolved (mg/L as P)	Boron, dissolved (mg/L)	Iron, dissolved (mg/L)	Manganese, dissolved (mg/L)	Carbon, organic, dissolved (mg/L as C)	Methylene-blue active substances (mg/L)
DP 500S	--	--	--	0.06	0.04	--	--
DP 400S	0.02	--	--	.22	.25	--	--
DP 300S	.01	--	--	.02	.04	--	--
DP 200S	.02	--	--	.02	7.6	--	--
DP 100S	.02	--	--	.01	.13	--	--
DP 0	--	0.01	--	.02	.01	--	--
DP 100N	.02	--	--	.02	2.5	--	--
DP 200N	.6	--	--	.04	3.5	--	--
DP 300N	.01	--	--	.02	1.3	--	--
DP 400N	.54	--	--	.03	5.3	--	--
DP 500N	1.1	--	--	.02	8.3	--	--
DP 600N	1.3	--	--	.02	1.1	--	--
DP 700N	1.9	--	--	.02	2.3	--	--
DP 800N	.81	--	--	.02	3.9	--	--
DP 900N	.01	--	--	.01	.02	--	--
DP 950N	.01	--	--	.01	.01	--	--

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