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

U.S. GEOLOGICAL SURVEY
Open-File Report 84-475

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

USGS
science for a changing world

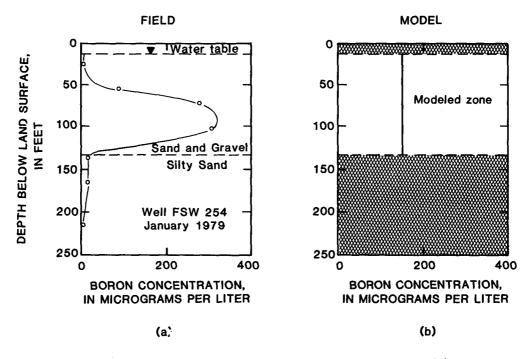FIELD   MODEL

Figure 15.--Profiles of boron concentration with depth showing (a) typical
vertical variation of concentration in the plume and (b) simulated
vertically averaged concentration in the model. Location
of well FSW 254 shown in figure 5.

Interaction of the Plume and Ashumet Pond

Ashumet Pond affects the simulated plume in two ways. First, the pond intercepts a portion of the treated sewage as it moves southward from the sand beds. In the calibration simulation, with the treated sewage applied to four nodes representing the entire sand-bed area, as much as 65 percent of the treated sewage discharges to the pond (fig. 14). The presence of contaminated water at the pond bottom was confirmed by two methods. Water samples collected from shallow wells driven into the pond bottom along the northwestern shore had specific conductances of over 400 μ mhos/cm and con-tained as much as 10 mg/L total nitrogen (William B. Kerfoot, K-V Associates, written commun., 1983). These levels are typical of wells screened in the plume within 200 feet of the pond. In addition, geophysical electromagnetic measurements of apparent ground conductivity along the shoreline show the presence of a contaminated zone having an elevated electrical conductance in the presumed area of discharge (Francis Frishneckt, U.S. Geological Survey, written commun., 1983).

The computed discharge of the plume to Ashumet Pond is very sensitive to the distance between the active infiltration beds and the pond. The sand beds cover a total area of 1,500 feet by 1,000 feet, and little data are available on past bed usage. The plume is likely a product of disposal to different beds at different times. Two simula-tions were run to test the effect of source location on the plume's path and rate of discharge to Ashumet Pond. In one simulation, all the treated sewage was applied to node (18,13) located 2,500 feet from the pond. In the other, the treated sewage was

applied to node (16,13) located 1,500 feet from the pond. The simulated plumes, after 40 years of transport, are shown in figure 16. The plume formed by disposal to the beds closest to the pond is much less extensive than the plume formed by disposal to the beds farthest from the pond. The difference in extent reflects the much greater rate of discharge of treated sewage to Ashumet Pond (fig. 14) when the closest beds are used.

These predicted effects of source location are reflected in the field data collected between 1978 and 1983. Since 1977, the treated sewage has been applied exclusively to two beds within 1,700 feet of the pond. Between 1978 and 1983, wells located west of the pond within 3,000 feet of the beds generally showed a drop in specific conductance and concentrations of constituents like boron (Thurman and others, 1984). Concentrations in wells located directly between the beds and Ashumet Pond remained at levels near the concentrations in the treated sewage. A similar pattern of lower apparent ground electrical conductivity west of the pond and elevated conductivity between the beds and the pond was inferred from geophysical measurements made in June 1983 (Gary Olhoeft, U.S. Geological Survey, written commun., 1984). As indicated by the simulations, the use of beds close to the pond since 1977 probably has caused contaminants to move more directly toward the pond. Concentrations have decreased west of the pond as contaminated water, representing disposal to more westerly beds prior to 1977, moves downgradient. In August 1984, the U.S. Air National Guard plans to shift disposal to eight sand beds located 1,700 to 2,600 feet from the pond. As a result, concentrations in wells west of Ashumet Pond should increase again as less treated sewage discharges to the pond.

Ashumet Pond also affects the simulated plume by discharge of pond water to the aquifer along its southwest shoreline. A well cluster drilled to a depth of 100 feet below pond level at the southwestern corner of the pond did not detect the plume. The recharging pond water, in effect, creates a flow line that bounds the plume along its eastern side south of Ashumet Pond.

A more rigorous evaluation of the interaction between Ashumet Pond and the plume must consider the three-dimensional flow and transport system. In the two-dimensional model, concentrations of solute in ground water discharging to the pond are vertically averaged through the full thickness of the aquifer. Vertical flow patterns must be considered because discharge to the pond may include more of the uncontaminated water above the plume and less of the deeper contaminated water in the plume.

Sensitivity to Dispersivity

Dispersivity may be adjusted during transport simulations like other model parameters until an acceptable match is obtained between observed and computed solute concentrations. In these two-dimensional simulations, concentrations could not be compared directly, and no attempt was made to adjust dispersivity during comparison to the observed plume. Dispersivity was varied, however, to test its effect on computed concentrations.

With dispersivity set equal to zero, concentrations in the simulated plume decrease with distance from the infiltration beds due to dilution by areal recharge, divergence of flow toward the discharge boundaries, and perhaps numerical dispersion (fig. 17a). Additional spreading and dilution of the plume occurs when longitudinal dispersivity is set to 40 feet (fig. 13c) and 100 feet (fig. 17b), the range of dispersivity reported by other studies of large plumes (Anderson, 1979, p. 127). The changes in concentration which result by varying longitudinal dispersivity from 40 to 100 feet are small, however. The contours of computed concentration shift less than 500 feet (4 percent of the length of the plume) as longitudinal dispersivity is increased from 40 feet to 100 feet. Estimation of dispersivity by matching observed and computed concentrations would be hindered by this insensitivity of concentration to dispersivity within the range of expected values. Matching concentrations, if model-generated concentrations were not vertically averaged and could be compared directly to field values, would be further hindered by the wide spacing between observation points, the uncertainties associated with other aquifer characteristics, and the unknown source variations.

70°34' 70°32' 70°34' 70°32'

SIMULATED BORON CONCENTRATION

DISCHARGE TO NODE (18,13)
2,500 FEET FROM POND

SIMULATED BORON CONCENTRATION

DISCHARGE TO NODE (16,13)
1,500 FEET FROM POND

Infiltration
beds

Infiltration
beds

Ashumet
Pond

Ashumet
Pond

Johns
Pond

Johns
Pond

41°38'

400

300

200

100

50

5

Coonamessett
Pond

Coonamessett
River

Backus River

Childs River

41°36'

400
300
200
100

50

5

Coonamessett
Pond

Coonamessett
River

Backus River

Childs River

(a)

0   1000   2000 METERS

0   4000   8000 FEET

(b)

EXPLANATION

——————— 100 ———————

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

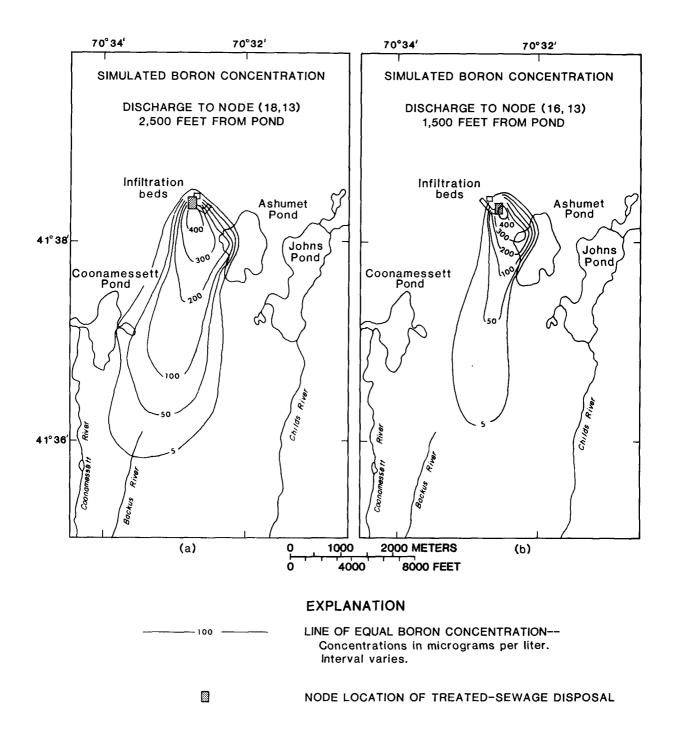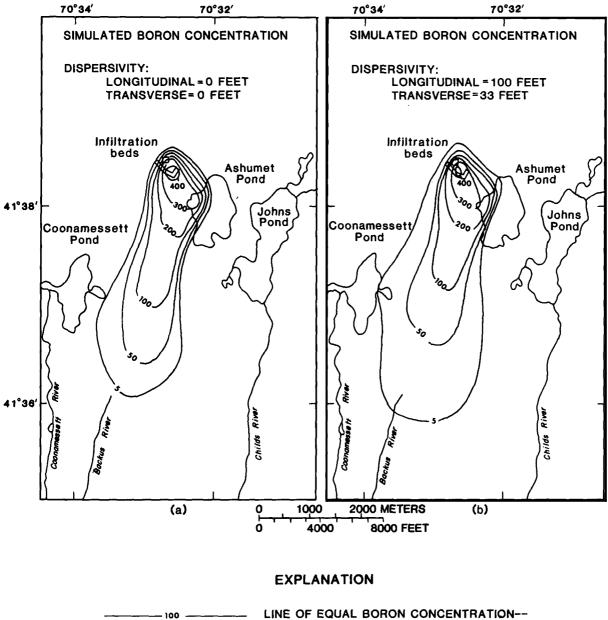▓

NODE LOCATION OF TREATED-SEWAGE DISPOSAL

Figure 16.--Computed boron concentrations after 40 years of disposal of treated
sewage containing 500 µg/L to infiltration beds (a) 2,500 feet
and (b) 1,500 feet from Ashumet Pond.

Figure 17.--Computed boron concentrations after 40 years of disposal of treated sewage containing 500 µg/L, assuming (a) zero dispersivity and (b) longitudinal dispersivity of 100 feet and transverse dispersivity of 33 feet.

Some dispersion in the simulations may be a result of the numerical methods used to solve the transport equation. Numerical dispersion may be caused by two factors: (1) The relatively coarse grid may result in some smearing of concentrations as reference particles are moved through the simulated flow system. (2) The regeneration of particles to minimize the number of void nodes also tends to smear concentrations within a given node, although the program includes an optimization routine to minimize this cause of numerical dispersion (Konikow and Bredehoeft, 1978, p. 19). The degree of numerical dispersion could not be evaluated from the simulated results.

PREDICTIVE CAPABILITY OF THE MODEL

If the transport model reasonably represents the observed field conditions, it also can be used to evaluate the effects of future source changes or other stresses on the plume. Two projections beyond the 40 years of transport shown in figure 13 were made to illustrate the predictive capability of the model. It is important to note, however, that the model was calibrated against a single set of observations made in 1978-79. Its ability to reproduce past changes is unverified because historical data on solute concentrations in the aquifer are unavailable. Also, little hydrologic and no chemical data are available for the area farther than 11,000 feet from the infiltration beds into which the contaminants move. Therefore, the reliability of the predictions cannot be assessed.

Constant and Continuous Source of Solute

In the original set of simulations, treated sewage containing 500 μg/L of boron continuously recharged the aquifer for 40 years at a rate of 0.72 ft^3/s. If disposal continues for 60 more years and all other hydrologic conditions remain the same, the simulated plume spreads along a broad front before discharging to the southern boundary of the modeled area (fig. 18). Concentrations at the toe are greatly diluted, although the dilution is due largely to mixing with areal recharge. In the actual three-dimensional flow and transport system, concentrations would be diluted much more slowly, and higher concentrations would reach the discharge areas.

Under the constant conditions assumed in the 100-year simulation, the simulated plume reaches a steady configuration. Concentrations stabilize after about 60 years from the start of disposal (fig. 19). Once a steady state is reached, solutes move continuously from the infiltration beds to the discharge areas, but the concentration pattern does not change with increasing time. This condition will probably never be observed because of variations at the source and changes in other stresses in the aquifer.

Changes in Concentrations of Solutes in the Source

Concentrations of some solutes in the treated sewage, such as detergents, have varied considerably since disposal began in 1936. The sewage also could receive additional treatment to remove selected contaminants that enter the aquifer at undesirable levels. To illustrate the change of concentrations of a constituent in the aquifer in response to the elimination of the constituent at the source, a simulation was run in which the concentration of boron in the treated sewage was suddenly and permanently decreased from 500 μg/L to 0.0 μg/L. The simulated boron plume is gradually flushed from the aquifer (fig. 20). Because of the two-dimensional approximation, the concentrations of boron previously introduced into the aquifer also are diluted by mixing with recharge from precipitation as the contaminated ground water moves downgradient. Therefore, the simulated concentrations shown in figure 20 decrease more rapidly than they would in the real system where little vertical mixing occurs.

Boron concentrations remain relatively high in the simulated plume, even 20 years after boron is eliminated from the source. In 20 years, the simulated peak concentration is 6,000 feet downgradient of the infiltration beds. A second area of elevated concentration occurs at Ashumet Pond (fig. 20) that persists 40 years after boron is eliminated
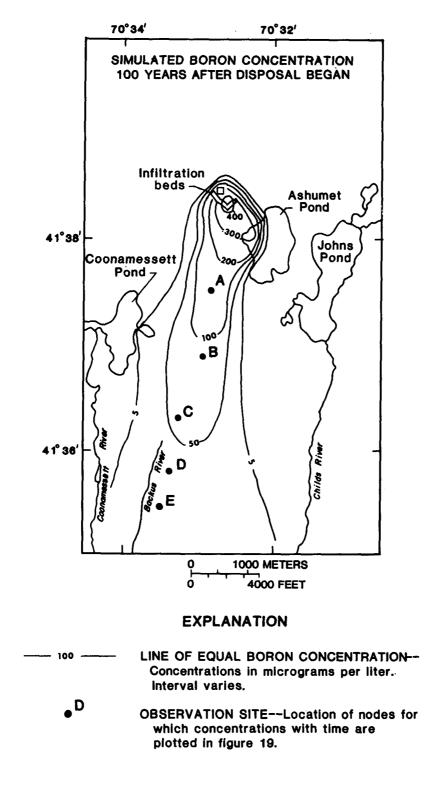
Figure 18.--Computed boron concentrations after 100 years of continuous disposal of treated sewage containing 500 µg/L. Assumes constant hydrologic conditions.

from the treated sewage. This zone coincides with a leakage node at Ashumet Pond that is located at the boundary between areas where water leaks into and water leaks out of the pond. At this point, ground-water velocities generated by the model are very small, and the rate of solute transfer between this node and adjacent nodes is very slow. As the plume reaches the pond, solute concentrations increase much more slowly in the node containing the stagnant water than in adjacent nodes. As the plume is subsequently flushed from the aquifer, concentrations in the node containing the stagnant water decrease more slowly than in adjacent nodes. Therefore, a small area of high concentrations remains that is slowly attenuated. This zone may be an artifact of the two-dimensional simulation and the representation of Ashumet Pond essentially as a constant-head boundary. In the field situation, it is not known if such a stagnant area exists.

The simulated distribution of boron in figure 20 is very similar to the distribution of detergents observed in 1978-79 (fig. 8). Detergents, however, were never completely eliminated from the source because some foaming agents (MBAS) remain in the treated sewage (table 1). Detergent concentrations in 1978-79 in the plume exceeded 0.5 mg/L MBAS, the recommended concentration limit for foaming agents in drinking water (U.S. Environmental Protection Agency, 1979, p. 42198). A simulation of the transport of detergents was run to determine how rapidly the residual high concentration of detergents located 3,000 to 10,000 feet downgradient of the infiltration beds will be flushed out of the aquifer. The approximate history of concentration of detergents in the treated sewage, shown in table 1, was used for the simulations.

The simulated distribution of detergents (fig. 21) after 32 years of transport (1948 through 1979) is very similar to the observed distribution of detergents in 1978-79 (fig. 8). The computed concentrations are lower than the observed concentrations, partly due to the two-dimensional approximation, but exceed the 0.5 mg/L MBAS drinking water standard. After 20 years of additional transport, the simulated zone of elevated detergent concentrations has moved an additional 7,500 feet away from the infiltration beds. Although the maximum computed concentration of detergents in figure 21b is 0.35 mg/L MBAS, concentrations in the three-dimensional system would be diluted much more slowly.

The distributions of other contaminants in the plume seem to reflect a variable source history. These contaminants include strontium and volatile organic compounds (Thurman and others, 1984). The model may be a useful tool to evaluate the processes that affect the movement of these solutes in the aquifer and to test hypotheses concerning the source and period of disposal of these contaminants.
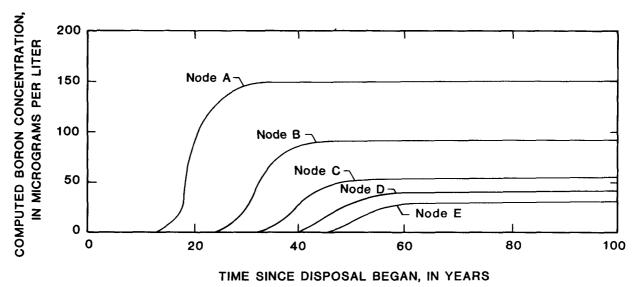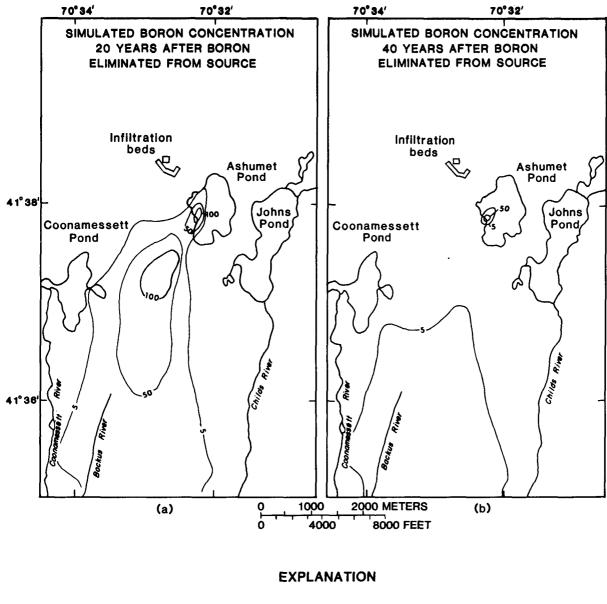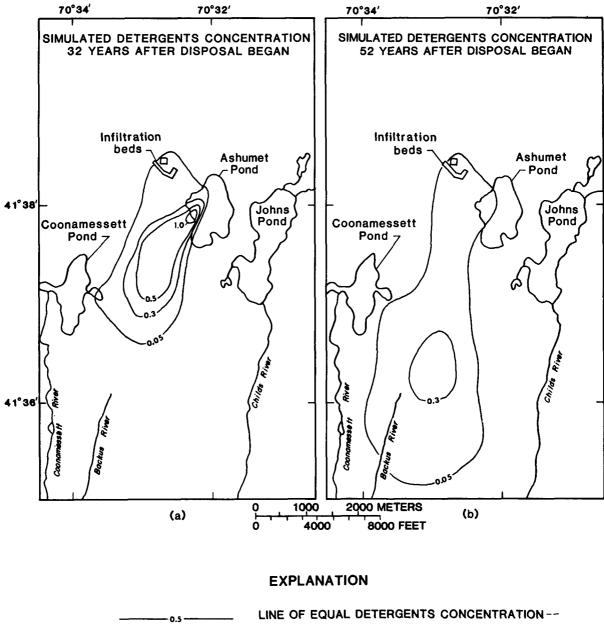

Figure 19.--Change in computed boron concentration at selected nodes if disposal of treated sewage continues for 100 years (node locations shown in figure 18).

SIMULATED BORON CONCENTRATION
20 YEARS AFTER BORON
ELIMINATED FROM SOURCE

SIMULATED BORON CONCENTRATION
40 YEARS AFTER BORON
ELIMINATED FROM SOURCE

Infiltration
beds

Ashumet
Pond

Johns
Pond

Coonamessett
Pond

Infiltration
beds

Ashumet
Pond

Johns
Pond

Coonamessett
Pond

41°38'

41°36'

Coonamessett River

Backus River

Childs River

(a)

(b)

0   1000   2000 METERS
0   4000   8000 FEET

EXPLANATION

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

Figure 20.--Computed boron concentrations assuming 40 years of disposal of treated
sewage containing 500 µg/L, followed by disposal of treated sewage containing
no boron for (a) 20 years and (b) 40 years.

Figure 21.--Computed detergent (MBAS) concentrations assuming 18 years of disposal of treated sewage containing 3.0 mg/L MBAS, followed by disposal of treated sewage containing 0.3 mg/L MBAS for (a) 14 years and (b) 34 years.

SUMMARY AND CONCLUSIONS

A method-of-characteristics computer code (Konikow and Bredehoeft, 1978) was used to simulate two-dimensional solute transport in the sewage plume at Otis Air Base. The first step in the modeling process was calibration of a steady-state flow model against average water-table conditions. A hydraulic conductivity of about 190 ft/d and an average areal recharge rate of about 20 in/yr yielded the best match between observed and computed water levels. The ground-water velocity distribution from the flow model was used as input to the transport simulations.

The second step in the modeling process was simulation of solute transport. Boron was simulated because it is a good tracer of the plume and was believed to behave conservatively in the plume. The simulated plume was compared to the plume delineated by chemical analyses of water samples collected from wells in 1978-79, approximately 40 years after sewage disposal to the sand beds began at the base. The locations of the simulated and observed paths agree reasonably well, although the simulated plume is wider and spreads eastward of the observed plume. The differences may be due, in part, to inaccurate field delineation of the plume. Additional drilling and sampling should be coupled with geophysical methods to locate and characterize the plume more precisely, especially along its southern and eastern boundaries. The modeling results also suggest that boron may not delineate the entire plume. A better description of boron's source history, and identification of other tracers, such as trace-level nonreactive organic compounds, is needed for future modeling efforts.

The approximation of the aquifer as a two-dimensional system resulted in inadequate simulation of observed concentrations in the plume. The two-dimensional model vertically averages concentrations and mixes areal recharge from precipitation through the full aquifer thickness. The plume occupies only part of the aquifer thickness, however, and areal recharge forms an uncontaminated zone above the plume. The assumptions of the two-dimensional approach are not completely met at the Otis site, and the effects of three-dimensional flow on transport are not well represented. Future modeling efforts should focus on simulation of the three-dimensional features of the sewage plume.

Ashumet Pond affects the plume by intercepting a portion of the treated sewage as it moves southward from the sand beds. In the simulations, as much as 65 percent of the treated sewage discharges to the pond. The percentage of the plume that discharges to the pond is sensitive to the distance between the infiltration beds and the pond. Use of beds located farther from the pond than the beds in use during 1977-83 should reduce the rate of plume discharge to the pond and increase the western extent of the contaminated zone. The observed plume is the result of disposal to different beds at different times. The simulations were used to show how the plume-pond interaction has a significant effect on the response of the plume to these changes. Additional research is needed to understand the potential impacts of the three-dimensional movement of contaminants at Ashumet Pond on the pond water quality and on the extent and path of the plume.

The transport model is a useful tool for analyzing and interpreting field data and for evaluating possible changes in the plume. Although the simplifications inherent in the modeling process result in only rough agreement with the field situation, the model simulations give considerable insight into the complex interactions of geologic and hydrologic processes that affect the transport of solutes in the aquifer. Transport models should be an integral part of future research at the Otis site.

REFERENCES CITED

Anderson, M. P., 1979, Using models to simulate the movement of contaminants through ground-water flow systems: CRC Critical Reviews in Environmental Control, v. 9, no. 2, p. 97-156.

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

Bouwer, Herman, 1973, Renovating secondary effluent by groundwater recharge with infiltration basins, in Sopper, W. E., and Kardos, L. T., eds., Recycling treated municipal wastewater and sludge through forest and cropland: University Park, Pa., The Pennsylvania State University Press, p. 164-175.

Farnsworth, R. K., Thompson, E. S., and Peck, E. L., 1982, Evaporation atlas for the contiguous 48 United States: National Oceanic and Atmospheric Administration Technical Report NWS 33, 26 p.

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

Guswa, J. H., and LeBlanc, D. R., 1985, Digital models of ground-water flow in the Cape Cod aquifer system, Massachusetts: U.S. Geological Survey Water-Supply Paper 2209, 112 p.

Hem, J. D., 1970, Study and interpretation of the chemical characteristics of natural water: U.S. Geological Survey Water-Supply Paper 1475, 2d ed., 363 p.

Kardos, L. T., and Sopper, W. E., 1973, Renovation of municipal wastewater through land disposal by spray irrigation, in Sopper, W. E., and Kardos, L. T., eds., Recycling treated municipal wastewater through forest and cropland: University Park, The Pennsylvania State University Press, p. 164-175.

Kerfoot, W. B., and Ketchum, B. T., 1974, Cape Cod waste water renovation and retrieval system, a study of water treatment and conservation, annual report, June 11, 1973-June 10, 1974: Woods Hole Oceanographic Institution Technical Report WHOI-74-13, 67 p.

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

Koerner, E. L., and Haws, D. A., 1979, Long-term effects of land application of domestic wastewater, Vineland, New Jersey, rapid infiltration site: U.S. Environmental Protection Agency Report EPA-600/2-79-072, 166 p.

Konikow, L. F., and Bredehoeft, J. D., 1978, Computer model of two-dimensional solute transport and dispersion in ground water: U.S. Geological Survey Techniques of Water-Resources Investigations, book 7, chap. C2, 90 p.

Krumbein, W. C., and Monk, G. D., 1943, Permeability as a function of the size parameters of unconsolidated sand: Transactions of the American Institute of Mining, Metallurgical, and Petroleum Engineers, v. 151, p. 153-163.

LeBlanc, D. R., 1982, Sewage plume in a sand and gravel aquifer, Cape Cod, Massachusetts: U.S. Geological Survey Open-File Report 82-274, 35 p. (Superseded by WSP 2218)

_____1984, Sewage plume in a sand and gravel aquifer, Cape Cod, Massachusetts: U.S. Geological Survey Water-Supply Paper 2218, 28 p.

Mather, K. F., Goldthwait, R. P., and Thiesmeyer, L. R., 1942, Pleistocene geology of western Cape Cod, Massachusetts: Geological Society of America Bulletin, v. 53, p. 1127-1174.

Morris, D. A., and Johnson, A. J., 1967, Summary of hydrologic and physical properties of rock and soil materials as analyzed by the Hydrologic Laboratory of the U.S. Geological Survey, 1948-60: U.S. Geological Survey Water-Supply Paper 1839-D, 42 p.

Palmer, C. D., 1977, Hydrogeological implications of various wastewater management proposals for the Falmouth area of Cape Cod, Massachusetts: Woods Hole Oceanographic Institution Technical Report WHOI-77-32 (appendix), 142 p.

Perlmutter, N. M., and Lieber, Maxim, 1970, Dispersal of plating wastes and sewage contaminants in ground water and surface water, South Farmingdale-Massapequa area, Nassau County, New York: U.S. Geological Survey Water-Supply Paper 1879G, 67 p.

Pickens, J. F., and Grisak, G. E., 1981, Scale-dependent dispersion in a stratified granular aquifer: Water Resources Research, v. 17, no. 4, p. 1191-1211.

Thornthwaite, C. W., and Mather, J. R., 1957, Instructions and tables for computing potential evapotranspiration and the water balance, in Publications in climatology: Enterton, N.J., Drexel Institute of Technology, v. 10, no. 3, p. 185-311.

Thurman, E. M., Barber, L. B., II, Ceazan, M. L., Smith, R. L., Brooks, M. G., Schroeder, M. P., Keck, R. J., Driscoll, A. J., LeBlanc, D. R., and Nichols, W. J., Jr., 1984, Sewage contaminants in ground water, in LeBlanc, D. R., ed., 1984, Movement and fate of solutes in a plume of sewage-contaminated ground water, Cape Cod, Massachusetts: U.S. Geological Survey Toxic Waste Ground-Water Contamination Program: U.S. Geological Survey Open-File Report 84-475.

Todd, D. K., 1980, Groundwater hydrology: New York, John Wiley and Sons, 535 p.

U.S. Environmental Protection Agency, 1979, National secondary drinking water regulations: Federal Register, v. 44, no. 140, p. 42195-42202.

_____1982, Cape Cod aquifer determination: Federal Register, v. 47, no. 134, p. 30282-30284.

Vaccaro, R. F., and others, 1979, Wastewater renovation and retrieval on Cape Cod: U.S. Environmental Protection Agency Report EPA-600/2-79-176, 174 p.

Wayman, Cooper, Page, H. L., and Robertson, J. B., 1965, Behavior of surfactants and other detergent components in water and soil-water environments: Federal Housing Administration FHA no. 532, Washington, D.C., 142 p.

CHAPTER C

The plume was originally described in 1978-79 by chemical analyses of water samples for major cations and anions, nutrients, and detergents. In this paper, additional chemical data collected in 1983 are presented that show movement of selected contaminants in the 4 years since 1979 and that describe the distributions of metals and organic compounds in the plume. This information adds to the base upon which hypotheses about the physical and chemical transport processes can be formulated and tested.

SEWAGE CONTAMINANTS IN GROUND WATER

by

E. M. Thurman, L. B. Barber, II, M. L. Ceazan, R. L. Smith,
M. G. Brooks, M. P. Schroeder, R. J. Keck, A. J. Driscoll,
D. R. LeBlanc, and W. J. Nichols, Jr.

ABSTRACT

The major cations and anions in the sewage plume at Otis Air Base, Massachusetts, are transported at the front of the plume and do not appear to be retained by the sand and gravel aquifer. Nutrients behave conservatively and nonconservatively. Nitrate moves rapidly while ammonia and phosphate are retarded. Organic constituents are present in the contaminated area of the aquifer, and detergents are a major component of the dissolved organic carbon. Volatile organic compounds are present in the aquifer, although not as far downgradient as the detergents. Also, the volatiles make only a small contribution to the dissolved organic carbon (less than 2 percent).

INTRODUCTION

Each year the Nation discharges 1 trillion gallons of wastewater and millions of tons of sludge to ground and surface waters. In coastal areas, wastewater is commonly discharged onto sand beds, and sludge is dried on land and disposed into landfills. Because many coastal communities dispose of wastewater and sludge onto sand beds, these wastes and their by-products, such as trace metals, nitrate, detergents, and nondegradable organic compounds, are continually being added to ground water. The fate of these compounds in an aquifer is an area of research in environmental chemistry of national significance and of specific interest to those who drink ground water in these areas. For these reasons we are studying geologic, hydrologic, chemical, and microbiological processes that control the fate and transport of treated sewage at Otis Air Base,[1], Falmouth, Massachusetts. This report summarizes the distribution and movement of inorganic solutes, nutrients, and organic compounds in ground water of this area in 1983.

EXPERIMENTAL PROCEDURES

Samples were collected from approximately 60 wells constructed of polyvinylchloride (PVC), which varied from 1.25 to 2.00 inches in diameter. Four of the wells were constructed of 2.50-inch diameter steel. The 1.25-and 2.50-inch wells were evacuated with a gasoline pump, then sampled with a peristaltic pump. The 2.00-inch wells were evacuated and sampled with a submersible stainless-steel pump. All wells were pumped until at least three casing volumes had been removed, and specific conductance was monitored to insure that stagnant water had been flushed from the well. Teflon tubing was used for discharge tubing for the peristaltic and submersible pumps.

Specific conductance, temperature, dissolved oxygen, and pH were measured in the field. Samples for major cations and anions were filtered through 0.45 micron Nucleopore filters. Samples for cations were preserved with nitric acid and stored in plastic bottles. After filtration both anions and nutrients were preserved without additives at 4 degrees C. Nutrients were analyzed within 3 days at a field laboratory near the site. Volatiles were sampled in a 40-mL sample bottle by removing all headspace and storing on ice at 4 degrees C. Samples for dissolved organic carbon and detergents were filtered through 0.45 micron silver filters and were stored on ice. Samples for semi-volatiles were taken by filling 1-gallon amber bottles with sample and spiking with three surrogate standards (see Barber and others, 1984). Samples were extracted by trapping the semi-volatiles on charcoal within 24 hours of sampling and were analyzed within one week at the laboratory.

Inorganic analyses, nutrients, and organic carbon were analyzed according to standard procedures of the U.S. Geological Survey (Skougstad and others, 1979), and trace metals were analyzed by inductively coupled plasma emission spectroscopy. Volatiles were analyzed by purge and trap followed by gas chromatography/mass spectrometry (Wershaw and others, 1983).

---

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

DISTRIBUTION AND MOVEMENT OF INORGANIC SOLUTES

Water samples were collected during the summer of 1983 and were analyzed for major cations and anions, trace metals, nutrients, and various organic constituents (dissolved organic carbon (DOC), volatiles, semi-volatiles, and detergents). The results of these analyses verified the general location of the plume that was reported previously (LeBlanc, 1982), but indicated that in three years the plume had moved southwest approximately 1000 feet, which is approximately one foot per day. The results of the chemical analyses for the 1983 field season confirm the hypothesis by LeBlanc (1982) that boron and detergents are nearly conservative tracers of the plume in this sand and gravel aquifer. Column experiments are planned for major and minor constituents to determine if field interpretations of conservative behavior can be confirmed in the laboratory.

Furthermore, a series of different zones have been identified within the sewage plume. These results are discussed in two sections, inorganic and organic solutes. In each section there is a discussion of solute distribution, possible origins, and movement of the solutes in the aquifer. There are four tables of data at the back of this report. Table 6 contains the concentration of inorganic solutes found in the samples of ground water, table 7 contains the concentration of organic solutes, table 8 contains DOC fractionation analyses, and table 9 includes the concentration of volatile organic compounds that are present in the ground water. Finally, figure 22 shows the location of wells that were sampled during the 1983 field season.

Specific Conductance

Because the specific conductance of the effluent is 400 µmhos and the specific conductance of the native ground water is 50 to 80 µmhos, specific conductance is a good indicator of the zone of sewage-contaminated water of the aquifer. Using specific conductance as indicator of the plume, we found that conductances greater than 200 µmhos (50 percent effluent and 50 percent native ground water) extend 8000 feet from the beds and are 2000 feet wide (fig. 23). The concentration contours form an ellipse that is elongated in the direction of ground-water flow, which is from northeast to southwest.

Because specific conductance may come from any inorganic ions in the ground water, it is a nonspecific indicator of the sewage plume. The major usefulness of conductivity is as a field tool to locate the general vertical and horizontal pattern of the plume. When new wells are drilled, specific conductance is used by the field geologist to locate screens in the core of the plume. Also we used a stable reading of specific conductance during sampling of wells as an indicator of water from the aquifer rather than stagnant water from the casing of the well.

Since the last sampling period 3 years ago, the specific conductance has increased at the toe of the plume. Wells FSW 282-70, FSW 282-94, FSW 282-123, and FSW 294-89 increased slightly in conductance with increases in major cations and anions. This suggests that contaminated ground water has moved southwest over the past few years. This is chemical evidence showing plume migration. Later sections will show this movement in more dramatic terms (see section on boron and detergents).
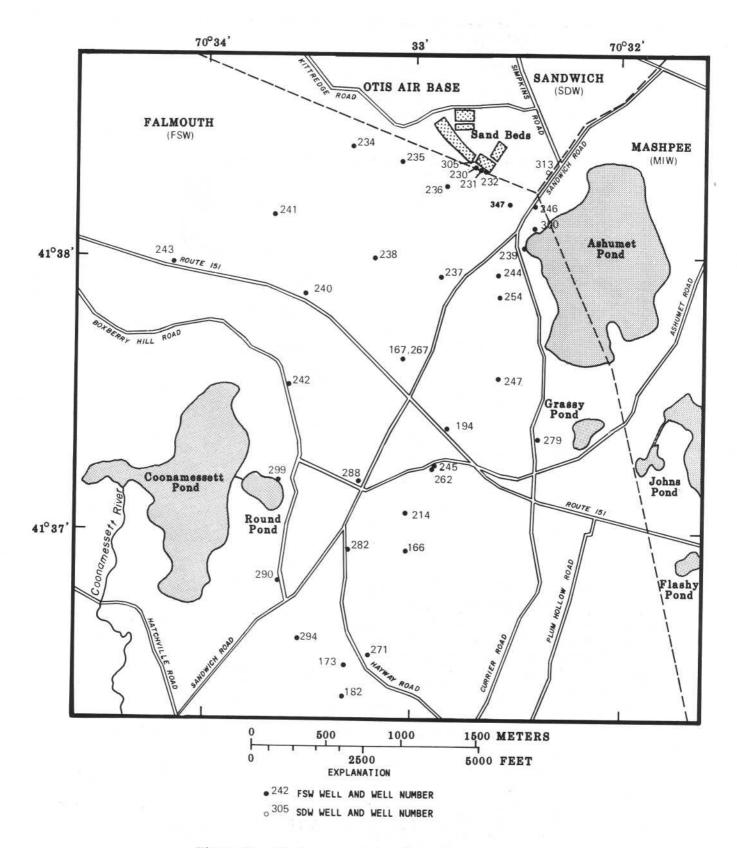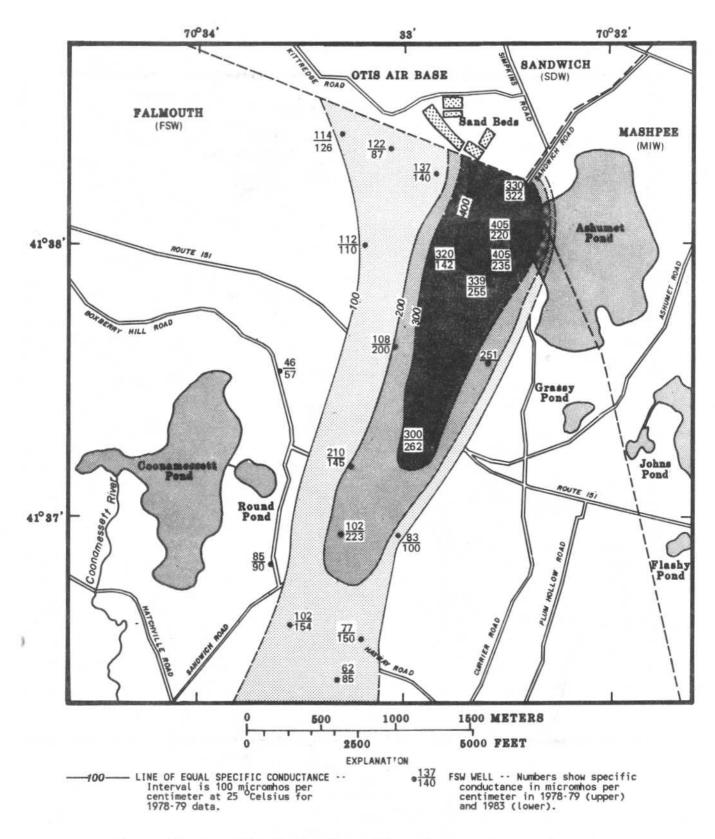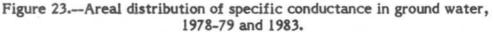
Figure 22.--Study area and locations of wells sampled.

70°34' 33' 70°32'

OTIS AIR BASE SANDWICH
(SDW)

KITTREDGE ROAD
SIMPKINS ROAD

FALMOUTH
(FSW)

Sand Beds

SANDWICH ROAD

MASHPEE
(MIW)

$\frac{114}{126}$ $\frac{122}{87}$

$\frac{137}{140}$

$\frac{330}{322}$

400

$\frac{405}{220}$

Ashumet Pond

41°38'

ROUTE 151

$\frac{112}{110}$

$\frac{320}{142}$ $\frac{405}{235}$

$\frac{339}{255}$

BOXBERRY HILL ROAD

100

200

300

$\frac{108}{200}$

$\frac{46}{57}$

251

Grassy
Pond

ASHUMET ROAD

Coonamessett
Pond

$\frac{300}{262}$

Coonamessett River

$\frac{210}{145}$

Johns
Pond

Round
Pond

41°37'

$\frac{102}{223}$

$\frac{83}{100}$

$\frac{85}{90}$

ROUTE 151

PLUM HOLLOW ROAD

Flashy
Pond

HATCHVILLE ROAD

SANDWICH ROAD

$\frac{102}{154}$

$\frac{77}{150}$

CURRIER ROAD

MAILAV ROAD

$\frac{62}{85}$

0 500 1000 1500 METERS
0 2500 5000 FEET

EXPLANATION

—100— LINE OF EQUAL SPECIFIC CONDUCTANCE --
Interval is 100 micromhos per
centimeter at 25 °Celsius for
1978-79 data.

•$\frac{137}{140}$ FSW WELL -- Numbers show specific
conductance in micromhos per
centimeter in 1978-79 (upper)
and 1983 (lower).

Figure 23.—Areal distribution of specific conductance in ground water,
1978-79 and 1983.

Not only has there been an increase in specific conductance at the toe of the plume, there has also been a decrease in conductance in wells near the sand beds just west of Ashumet Pond (see wells FSW 237-88, FSW 239-64, FSW 244-90, FSW 254 series). Figure 23 shows the ratio of conductances collected during the 1979 and 1983 field seasons. The lower conductances probably reflect the current use of the beds closest to Ashumet Pond (LeBlanc, 1984). The plume is moving more directly toward Ashumet Pond, and the conductance in wells just west of Ashumet Pond is decreasing as contaminated water moves toward the south. The decrease in conductance in these wells just west of Ashumet Pond is further corroborated by decreases in the concentrations of major cations, anions, and boron, which has been used as a tracer of the plume by LeBlanc (1982).
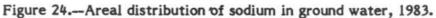
Major Cations

The ratio of major cations (Na/Ca), although not distributed identically in the effluent and ground water, does not show a trend in the plume of contaminated ground water that would indicate that cation exchange is occurring. The concentration of sodium is generally greater than the concentration of calcium, which is not a typical distribution for ground water. Commonly, the concentration of calcium is greater than the concentration of sodium. Because the concentration of sodium is greater than calcium both in the effluent and in the native ground water, there are multiple sources for sodium. Sodium is a major cation in the sewage and may also come from road salt introduced into the aquifer (LeBlanc, 1982). A natural source for sodium in the aquifer is sea spray that makes its way into the aquifer with recharging groundwater from precipitation. For these reasons, sodium and calcium are of limited use as tracers of the effluent plume in the ground water. Figure 24 and 25 show the distributions of sodium and chloride, which do follow the general location of the conductivity plume. Because there is only a five-fold change in concentration from the core of the plume to the native ground water, it is difficult to locate the boundaries of the plume from the distributions of sodium and chloride.

Major Anions

Major anions (SO_4, Cl, and HCO_3) are present in approximately equal distributions in the effluent and contaminated and native ground waters. The major difference is an increase in alkalinity just downgradient from the infiltration beds. Alkalinity (as HCO_3) increases from 0.5 to 1.0 meq in the first 1000-2000 feet of contaminated ground water. Organic carbon is converted to inorganic carbon in this zone through microbial respiration (carbon dioxide, which then forms carbonate alkalinity), and approximately 0.75 meq of organic carbon is removed. Thus, nearly a 1:1 stoichiometry is maintained in this process. Microbiological studies are underway to better understand these conversions (Ceazan and others, 1984; Smith and Duff, 1984).

Chloride and sulfate are moving conservatively in the aquifer based on Cl/SO_4 ratios of 1.0 to 1.5 throughout the core of the plume. There are no significant trends in these ratios with distance from the beds. For example, if preferential sorption of sulfate or sulfate reduction were occurring, the ratios of Cl/SO_4 should increase with distance from the beds. Sulfate removal in ground water commonly occurs through microbial sulfate reduction to sulfide. Although

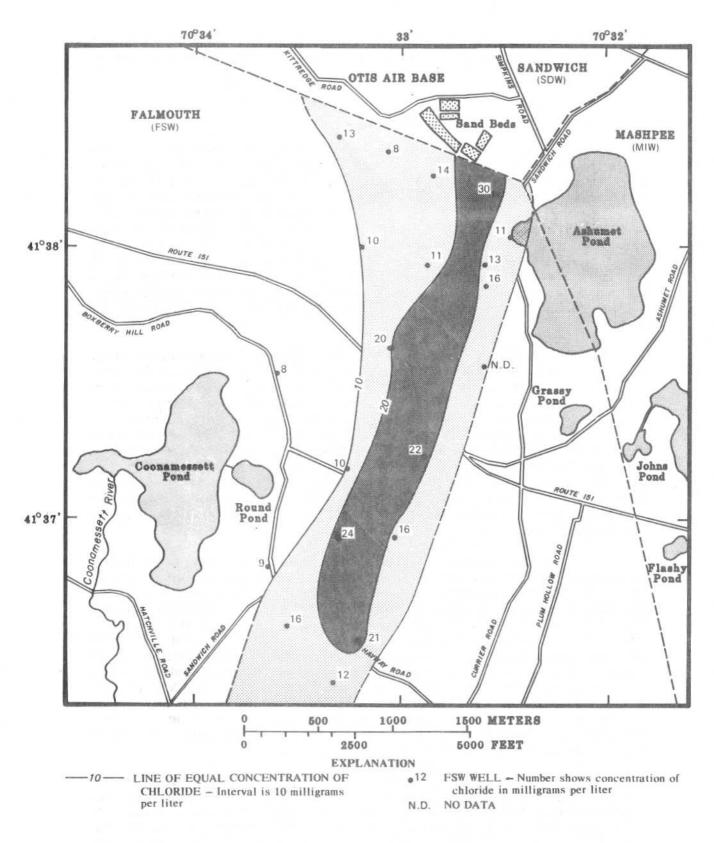Figure 24.—Areal distribution of sodium in ground water, 1983.

Figure 25.--Areal distribution of chloride in ground water, 1983.

hydrogen sulfide was not detected, sulfate reduction may be occurring near the sand-filtration beds, and new wells have been drilled near the beds to determine if this is the case.

Trace Elements

Trace metals are present in low concentration in the effluent and in the aquifer. Locally, there are increased concentrations of iron and manganese, which appear to be leaching from the sediment of the aquifer. This conclusion was reached from three lines of evidence. First, manganese concentrations are low in the treated sewage (20 µg/L), but concentrations in certain wells in the plume are 1-3 mg/L. Second, there is no systematic distribution of iron and manganese in the plume that follows the concentration of other solutes in the plume, such as boron, detergents, and conductivity. Finally, field evidence of leaching of manganese and iron from cobbles along the beach of Ashumet Pond and in gravel pits near the infiltration beds suggest that the source of manganese is from leaching of the sediments by the anaerobic (reducing) waters of the plume.

Strontium, which previously had not been measured in water samples from the aquifer, forms a plume similar to sodium, chloride, and specific conductance (fig. 26). Strontium is a minor alkaline earth cation with concentrations that vary from approximately 10-100 µg/L in the aquifer. The distribution of strontium suggests that it may behave conservatively, but this conclusion is based only on the distribution of strontium in the plume, and column experiments are underway to test this hypothesis. From theoretical considerations of strontium, which is a divalent cation of large size, one would predict that strontium would be retained by ion exchange. The fact that the strontium distribution resembles the distribution of detergents suggests that there may be relationship between these two constituents, which may be related to origin in the sewage effluent.

The trace element that has been a good indicator of the plume is boron, which is present in the effluent at 500 µg/L and is present at less than 30 ug/L in the native ground water. Boron is used as an additive in soaps and detergents, and based on several analyses of the effluent seems to be of relatively constant concentration over the short history of our sampling (from 1979 to 1983). Thus, boron is being used as a conservative tracer of the plume for solute-transport modeling (LeBlanc, 1984). In fact, boron may generally be a good tracer of sewage in ground water. However, column studies are underway to determine if boron behaves conservatively in the ground water. Because boron is a weak acid, it will be sorbed by weakly basic sites, such as aluminum-oxide coatings or perhaps on clay minerals. The aquifer solids contain less than 1 percent clay, and quartz and feldspar are the principal minerals of the sand. For these reasons, we hypothesize that boron moves conservatively.

Figure 27 shows the distribution of boron in 1978-1979 and again in the 1983 sampling. Notice that the concentrations of boron have increased near the toe of the plume from less than 20 µg/L to 380 µg/L (for example, wells FSW 166-67, FSW 282, FSW 290, FSW 294, and FSW 182-69). These data show that the plume has moved approximately 1000 feet over the past 3 years. Perhaps more interesting is the sharpness of the boron-concentration front, where concentrations increase from 10 to 400 µg/L over a distance of 1000 feet. This
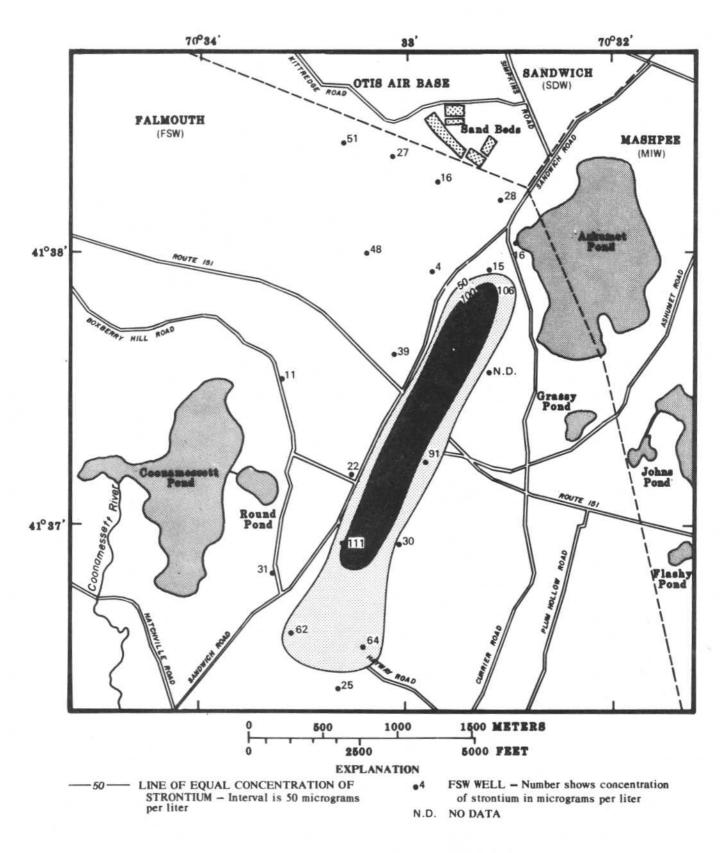
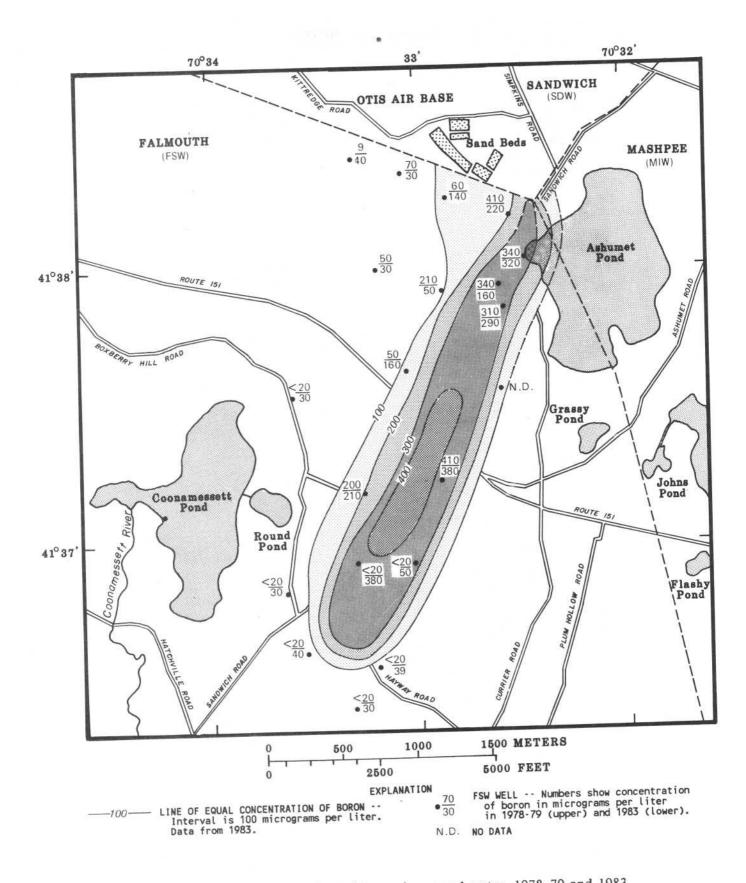Figure 26.—Areal distribution of strontium in ground water, 1983.

Figure 27.--Areal distribution of boron in ground water, 1978-79 and 1983.

suggests that longitudinal dispersion is small throughout the flow path of the boron plume.

Nutrients

The distribution of nutrients (N and P) are discussed in a separate section (Ceazan and others, 1984). Two major findings that should be emphasized are the nonconservative behavior of phosphate and ammonia. On the other hand, nitrate does not appear to be retained by the aquifer. This conclusion is based on the finding of nitrate at the furthest toe of the plume. Phosphate shows the greatest retardation and has moved less than 1000 feet from the infiltration beds. Ammonia has traveled further downgradient and the core of the ammonia plume is approximately 3000 to 4000 feet from the beds. Finally, nitrate is present as far as 10,000 feet from the infiltration beds. Because of microbiological transformations of nitrogen, such as nitrification from ammonia to nitrate, it is difficult to separate the mechanisms that are the cause of the retention of ammonia. Both sorption and nitrification may be responsible for the apparent retardation of ammonia in the aquifer.

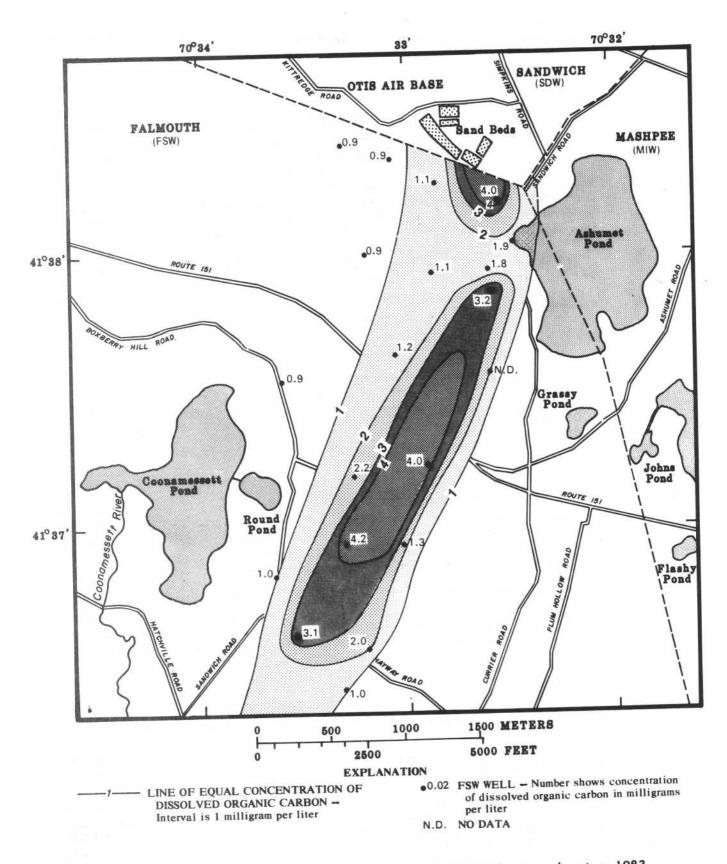## DISTRIBUTION AND MOVEMENT OF ORGANIC SOLUTES

This section discusses organic solutes and is divided into three parts: dissolved organic carbon (DOC), detergents (MBAS), and volatiles. Semi-volatiles are discussed in a separate report by Barber and others (1984).

Dissolved Organic Carbon

Dissolved organic carbon is a good indicator of the amount of organic matter in the plume. It is a measure of all nonvolatile organic compounds in the ground water which includes the natural organic matter as well as the organic compounds from the treated sewage. The concentration of dissolved organic carbon varied from 0.7 mg/L, which was a background concentration for natural organic matter in the ground water, to 4.2 mg/L at the center of the plume at well FSW 282. Table 7 shows the concentration of dissolved organic carbon in all the wells sampled during the 1983 field season. The range in concentration of DOC found in the plume (from 0.7 to 4.2 mg/L), is within the range of dissolved organic carbon found naturally in ground waters. These low concentrations of organic carbon in the plume present a challenge for the analytical work of identification of specific organic compounds.

In spite of low concentrations of dissolved organic carbon in the ground water, the plume is easily recognizable. Figure 28 shows the distribution of dissolved organic carbon in the ground water. There are two areas in which DOC concentrations exceed 2 mg/L. The first is located within 1500 ft of the infiltration beds at well FSW 347-67. The DOC in this zone is as high as 4 mg/L, which is well below the DOC of the treated sewage (12 mg/L). The decrease in DOC is caused by microbial decomposition, which converts organic carbon to inorganic carbon. This conversion is suggested by increasing alkalinity in the waters of the plume. The second zone of increased DOC is located at wells FSW 262 and 282, which corresponds to the location of the detergent maximum (see figure 29). The concentration of DOC increases to 4 mg/L in this second zone. This increase in DOC is caused by the nondegradable detergents that are present

Figure 28.--Areal distribution of dissolved organic carbon in ground water, 1983.

in this part of the plume. Detergents account for approximately 50 percent of the DOC in this zone based on DOC fractionation analyses, which separate the detergent component of the DOC. DOC fractionation analysis is explained in the following section.

Detergents

The use of nonbiodegradable detergents began in approximately 1946 and continued through 1964. After 1964 biodegradable detergents were used. Because of the switch to degradable detergents, there is an 18-year pulse of nondegradable detergents that forms a plume centered at well FSW 282 (LeBlanc, 1982) and is shown in figure 29. The purpose of the 1983 sampling was to measure the amount of detergents present, their contribution to DOC, and any movement since the last sampling in 1979.

The detergents were measured this past field season by two methods. First was the methylene blue active substances (MBAS) test, which was also used by LeBlanc in the 1978-1979 sampling period. The second method was DOC fractionation analysis to measure the amount of organic carbon contributed by the detergents. Using DOC fractionation, we found that the hydrophobic neutral fraction contained the detergents. For example, table 8 shows the difference in hydrophobic neutrals between the native ground water (FSW 242-77) and the core of the detergents (FSW 282-70). Well FSW 282-70 contained 2.8 mg/L as DOC in the hydrophobic neutral fraction and the native ground water contained no DOC in the hydrophobic neutral fraction. Combination of the DOC-fractionation and MBAS analyses shows that approximately 50 percent of the dissolved organic is contributed by the detergents. By difference, 2 mg/L of organic carbon is not accounted for. Identification of this organic carbon is the subject of further research.

Volatile Organic Compounds

Approximately 32 samples were analyzed for volatile organic compounds (table 9). The purge and trap technique, which is reported in this section, identified from 1-10 compounds in the samples with a detection limit of 0.1 µg/L. Table 9 shows the compounds identified and their concentration in the ground water. The compounds that were detected frequently include: trichloroethene (trichloroethylene), tetrachloroethene (tetrachloroethylene), and tetra- and trichloroethanes. Nine of the wells had concentrations exceeding 10 ug/L (fig. 30). The core of the volatiles was found approximately 3000 feet downgradient from the sand beds at well cluster FSW 254. This core of volatiles is well upgradient of the toe of the plume delineated by boron and detergents. This difference may be caused by several factors. First, the volatiles may be coming from a different source than the boron and detergents. Another source might be a spill or septic tank, where volatile organic compounds are commonly used as degreasing solvents. However, the distribution of the volatile organics in the ground water suggests an origin that is similar in location to the other compounds in the plume.

A second factor that may explain the difference in the distribution of volatiles is sorption of the volatile compounds on the sediments of the aquifer. Given the center of the volatile plume near well FSW 254, the retardation of
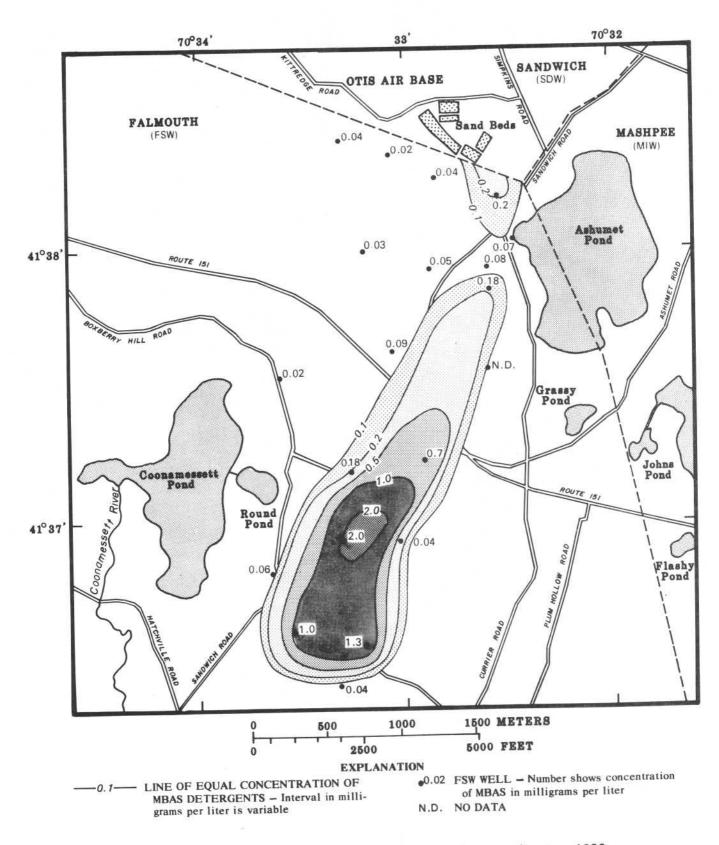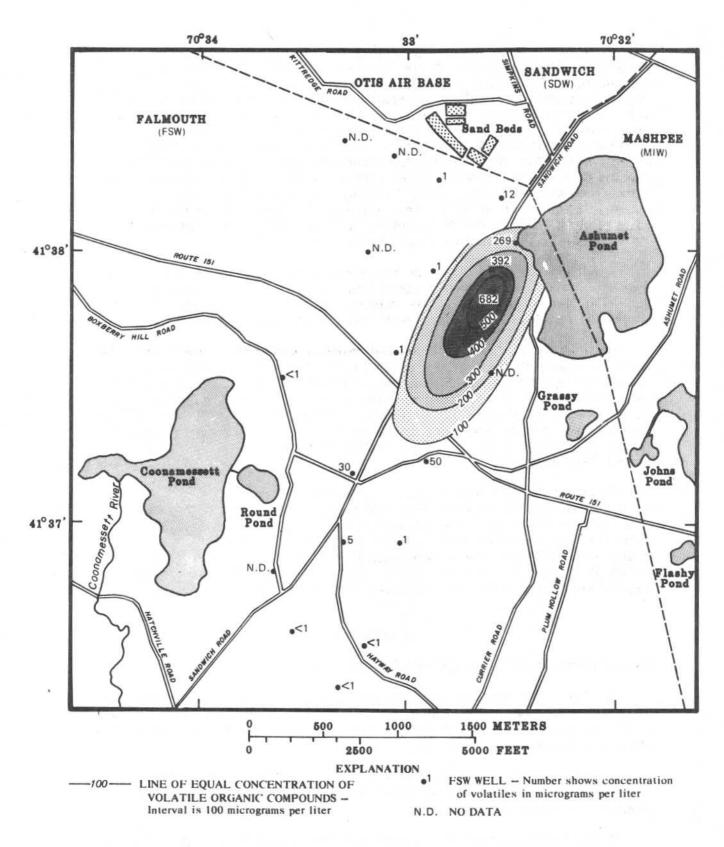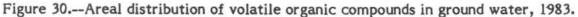
Figure 29.--Areal distribution of MBAS detergents in ground water, 1983.

Figure 30.--Areal distribution of volatile organic compounds in ground water, 1983.

volatiles would be a factor of 3 to 4 times that of the boron and detergents in the plume. Previous studies on the removal of tetrachloroethene, and similar compounds, by bank filtration (Schwartzenbach and others, 1983), showed that tetrachloroethene and similar compounds are mobile in ground water and do not sorb onto the aquifer sediments. This is expected from theoretical considerations. Water solubility is a simple measure of sorptive potential of an organic compound. Because these volatile compounds have large water solubilities (greater than 300 mg/L), theory predicts that they would not be retained on a sand and gravel aquifer with less than 0.5 percent organic carbon (Schwartzenbach and others, 1983). Therefore, this second hypothesis of sorption and retardation seems unlikely. The fact that detergents, which are of similar water solubility and, by inference, similar sorptive capacity, are not retained by the aquifer, is further evidence that retention is not occurring.

A third possibility, and the one that seems most likely, is that the volatiles have only more recently been disposed into the sewage. Given the distance of 3000-4000 feet and a ground-water velocity of 1 ft/d, we estimate that these compounds may have entered the ground water in the past 10-15 years.

Because there are no wells located between wells FSW 254 and FSW 262, which are separated by a distance of 3000 feet, the center of the volatile plume may be further downgradient than the data now indicate. The area between these two wells is an important zone for more sampling in order to determine the exact location of the volatile plume. Also, additional wells east and up-gradient from the sand beds may give more exact information on the source of the volatile contaminants. Therefore, further sampling for volatiles should focus on these two locations.

Finally, we found that sampling by peristaltic pump loses approximately one half of the volatile organic compounds present in the sample when compared with a submersible pump. This means that concentrations in the volatile organic plume may be 1.5 to 2.0 times greater than shown in figure 30, because the central well, FSW 254, was sampled with a peristaltic pump. This difference is caused by degassing of the sample. In future work we plan to investigate a simple down-hole sampler, such as a bailer versus the submersible pump.

CONCLUSIONS

The major conclusions from the 1983 field study are:

1) The concentrations of inorganic solutes in the ground water are consistent with the previous study of LeBlanc (1982) with respect to the location of the plume based on major cations and anions and boron. A new tracer, strontium, was found in the plume.

2) Metals, such as manganese and iron, are being mobilized from the aquifer solids near the sand-infiltration beds by the anaerobic (reducing) conditions of the ground water.

3) The plume has moved approximately 1000 feet over the past 3 years in the direction of ground-water flow, from northeast to southwest.

4) The organic plume was investigated by DOC, DOC fractionation, and volatile organic analysis. Detergents accounted for 50-60 percent of the dissolved organic carbon in the plume at its most southern extent. Tetrachloroethene and other volatile organic solutes were discovered in well cluster FSW 254.

5) The type of pump used for sampling affects the concentration of volatile organic compounds in the sample. Because of degassing of the sample, the submersible pump gives a greater concentration of volatiles by a factor of two than the peristaltic pump.

6) Finally, the organic compounds in the plume are affected more by microbial processes than by sorption, which seems to play a small role, if any, in the movement of organic solutes in the ground water.

REFERENCES CITED

Barber, L. B. II, Thurman, E. M., and Schroeder, M.P., 1984, Closed-loop stripping, combined with GC-MS analysis, to define a semi-volatile organic contaminant plume: U. S. Geological Survey Open-File Report (this volume).

Ceazan, M., Updegraff, D. M., and Thurman, E. M., 1984, Evidence of microbial processes in sewage-contaminated ground water: U.S. Geological Survey Open-File Report (this volume).

Kerfoot, W. B., and others, 1975, Cape Cod waste water renovation and retrieval system, a study of water treatment and conservation, first year of operation, annual report, June 11, 1974-June 10, 1975: Woods Hole Oceanographic Institution Technical Report WHOI-75-32, 194 p.

LeBlanc, D. R., 1982, Sewage plume in a sand and gravel aquifer, Cape Cod, Massachusetts, U.S. Geological Survey Open-File Report 82-274, 35 p. Superseded by U.S. Geological Survey Water-Supply Paper 2218, 28 p.
_____ 1984, Digital modeling of solute transport in a plume of sewage-contaminated ground water: U.S. Geological Survey Open-File Report (this volume).

Leenheer, J. A. and Huffman, E. W. D., Jr., 1979, Analytical method for dissolved-organic carbon fractionation, U.S. Geological Survey Water-Resources Investigations 79-4, 16 p.

Oldale, R. N., 1976, Notes on the generalized geologic map of Cape Cod: U. S. Geological Survey, Open-File Report 76-765, 23 p.

Schwartzenbach, R. P., Giger, W., Hoehn, E., and Schneider, J. K., 1983, Behavior of organic compounds during infiltration of river water to groundwater, Field studies, Environmental Science and Technology, v. 17, p. 472-479.

Skougstad, M., and others, 1979, Methods for determination of inorganic substances in water and fluvial sediments, U.S. Geological Survey, Techniques of Water-Resources Investigations, Book 5, Chapter A-1, 626 p.

Smith, R.L. and Duff, J. H., 1984, A preliminary study of denitrification in a plume of sewage-contaminated ground water: U.S. Geological Survey Open-File Report (this volume).

Wershaw, R. L., Fishman, M. J., Grabbe, R. R., and Lowe, L. E., 1983, Methods for the determination of organic substances in water and fluvial sediments: U. S. Geological Survey Techniques of Water-Resources Investigations, Book 5, Laboratory Analysis, Chapter A3, 173 p.