

Leachate Plumes in Ground Water From Babylon and Islip Landfills, Long Island, New York

GEOLOGICAL SURVEY PROFESSIONAL PAPER 1085

*Prepared in cooperation with the
Suffolk County Department of
Environmental Control*

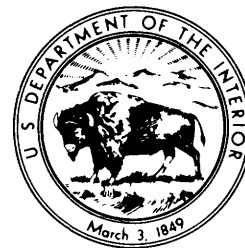


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By GRANT E. KIMMEL *and* OLIN C. BRAIDS

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CONVERSION FACTORS AND ABBREVIATIONS OF UNITS

Factors for converting U.S. customary units to metric units are shown to four significant figures. However, in the text the metric equivalents are shown only to the number of significant figures consistent with the values for the U.S. customary units.

<i>U.S. customary units</i>	<i>Multiply by</i>	<i>Metric units</i>
acres	0.4047	hectares (ha)
feet (ft)	.3049	meters (m)
cubic feet (ft ³)	.02832	cubic meters (m ³)
feet per day (ft/d)	.3049	meters per day (m/d)
feet squared per day (ft ² /d)	.01075	centimeters squared per second (cm ² /sec)
gallons (gal)	3.785	liters (L)
gallons per day per square foot (gal/d) /ft ²	.041	meters per day (m/d)
gallons per foot (gal/ft)	12.418	liters per meter (L/m)
gallons per minute (gal/min)	.06309	liters per second (L/s)
gallons per minute per foot		
inches (in)	25.40	millimeters (mm)
miles (mi)	1.609	kilometers (k)
cubic yards (yd ³)	.7646	cubic meters (m ³)
degrees Fahrenheit (°F)	.5556	degrees Celsius (°C)

LEACHATE PLUMES IN GROUND WATER FROM BABYLON AND ISLIP LANDFILLS, LONG ISLAND, NEW YORK

By GRANT E. KIMMEL and OLIN C. BRAIDS

ABSTRACT

Landfills operated by the Towns of Babylon and Islip in southwest and central Suffolk County contain urban refuse, incinerated garbage, and scavenger (cesspool) waste; some industrial refuse is deposited at the Babylon site. The Islip landfill was started in 1933, the Babylon landfill in 1947. The landfills are in contact with and discharge leachate into the highly permeable upper glacial aquifer (hydraulic conductivity 190–500 feet per day). The aquifer is 74 feet thick at the Babylon landfill and 170 feet thick at the Islip landfill. The leachate-enriched water occupies the entire thickness of the aquifer beneath both landfills, but hydrologic boundaries retard downward migration of the plumes to deeper aquifers. The Babylon plume is 1,900 feet wide at the landfill and narrows to about 700 feet near its terminus 10,000 feet from the landfill. The Islip plume is 1,400 feet wide at the landfill and narrows to 500 feet near its terminus 5,000 feet from the landfill.

Hydrochemical maps and sections show the distribution of the major chemical constituents of the plumes. The most highly leachate-enriched ground water obtained was from the Babylon site; it contained 860 mg/L (milligrams per liter) sodium, 110 mg/L potassium, 565 mg/L calcium, 100 mg/L magnesium, 2,700 mg/L bicarbonate, and 1,300 mg/L chloride. Sulfate was notably absent or in low concentration in most parts of both plumes. Nitrogen in plume water was mostly in the form of ammonium, and concentrations as high as 90 mg/L were found; concentrations of nitrogen as N in the plume were less than 10 mg/L. As much as 440 mg/L iron and 190 mg/L manganese were found in the leachate-enriched water. Samples were also tested for arsenic, boron, cadmium, cobalt, chromium, copper, mercury, nickel, lead, selenium, strontium, and zinc. Boron was more or less ubiquitous and was found in concentrations as high as 2 mg/L. Organic carbon was found in concentrations as high as 2,250 mg/L in the most highly leachate-enriched water but attenuated rapidly to less than 20 mg/L. Dissolved-solids concentrations near the landfills were between 400 and 3,000 mg/L at Babylon and between 500 and 1,500 mg/L at Islip.

Ground-water temperatures near the landfills exceed those in ambient water by as much as 7°C at Babylon and 16°C at Islip. Heat contributed by the landfills was mostly dissipated with 0.4 mi of the landfill, but at Islip, the warm leachate-enriched water extended 0.5 mi downgradient.

The entrance of leachate into the less dense ground water as pulsations after rainfall may explain the presence of high leachate enrichment near the bottom of the aquifer. A comparison of the physical characteristics of leachate-enriched ground water with those of ambient water suggests that the downward movement of leachate results from its greater density.

Simulation of the movement and dispersion of the Babylon plume with a mathematical dispersion model indicated the coefficient of longitudinal dispersion to be about 60 ft²/d (feet squared per day) and the ground-water velocity to be 1 ft/d. However, the velocity determined from the hydraulic gradient and public-supply wells in the

area was 4 ft/d; this velocity would cause a plume four times as long as that predicted by the mathematical dispersion model. At the Islip site, the plume was one-third the length calculated on the basis of the age of the landfill. The shortness of the plumes has not been explained; it may be a result of the leachate's having been too dilute to form a plume during the early years of the landfills.

INTRODUCTION

ALTERATION OF GROUND WATER BY LANDFILL LEACHATE

Reports of ground-water alteration from landfills in humid environments (Hughes and others, 1969), semihumid environments (Andersen and Dornbush, 1968) and arid environments (State of California, 1954) generally conclude that landfill leachate deteriorates ground-water quality, sometimes even where the landfill is hundreds of feet above the water table (Apgar and Langmuir, 1971).

Deterioration of ground water and surface water by leachate from solid-waste landfills has become a problem of general concern. Where precipitation causes leachate to seep from a landfill to ground water, the high dissolved-solids concentrations and injurious substances in the solution may adversely affect local water supplies.

The deterioration of water resources, especially of public drinking supplies, can threaten the health and economy of an area. The drinking water for most of Long Island is pumped from a large ground-water reservoir separated from the mainland by saltwater, and this reservoir is the source of freshwater for several million inhabitants over about 80 percent of the island. Contamination by wastewater and other materials deposited on or beneath the land surface is a major concern in the management of Long Island's ground-water reservoir (Perlmutter and Koch, 1971, 1972). Deterioration of ground water by landfills on Long Island was surmised but undocumented until this study.

PURPOSE AND SCOPE OF REPORT

In 1971, a cooperative program between the U.S. Geological Survey and the Suffolk County Department of Environmental Control was established to investi-

gate the deterioration of ground water on Long Island from solid-waste landfills. The study evaluated the type and extent of leachate enrichment caused by two active, long-established landfills in the Towns of Babylon and Islip. Wells were installed, and water samples were analyzed to obtain data for a 3-year period in order to map the chemical properties and movement of leachate in the area of the landfills. The purpose of this report is to describe the effect of these landfills on the Long Island ground-water reservoir, to evaluate, to the extent possible, the chemical effect of landfill leachates on the ground water, and to investigate the flow characteristics of ground water and leachate in the region of the landfills. The interpretations of leachate formation and movement may be applicable to similar hydrogeologic settings elsewhere.

LOCATION OF STUDY

The landfills selected for this study are operated by the Towns of Babylon and Islip, in the south-central part of Long Island, New York. The landfills are 4.6 mi and 3.9 mi respectively, north of Great South Bay (fig. 1). The Babylon landfill is surrounded by cemeteries and some light industry; the Islip landfill is surrounded by unused land as well as some light industry.

Babylon had a population of 287,000 and Islip 300,000 in 1973 (LILCO, 1974.) Most of the population growth has been since 1950. Babylon has about four times as much industry as Islip; however, most of the industry of both towns is a fabrication type that yields little decomposable refuse to landfills.

HYDROLOGY

GROUND-WATER FLOW SYSTEM

A southward-sloping wedge of unconsolidated sediment underlies the entire island and overlies crystalline bedrock (Veatch and others, 1906). These sediments provide an extensive fresh ground-water reservoir. A regional ground-water flow system, recharged by precipitation, extends from the ground-water divide near the center of the island outward toward saltwater, which surrounds the island. Throughout the island, shallow ground water discharges into nearby stream channels. South of the center of the island, ground water in the upper part of the system discharges directly into Great South Bay (fig. 1), but that in the lower part of the system may discharge by upward leakage into the bays and ocean. Generalized descriptions of the ground-water reservoir are given in Cohen, Franke, and Foxworthy (1968) and McClymonds and Franke (1972).

RECHARGE

The ground-water reservoir is recharged by precipitation that infiltrates the soil, which is generally very permeable. Long Island receives an average of 3–4 in. of precipitation monthly throughout the year, but most of the recharge probably takes place in the cool, non-growing season. As shown in table 1, precipitation is only slightly higher in the cool season than in the warm season. During the warm season, from April through September, nearly all precipitation is con-

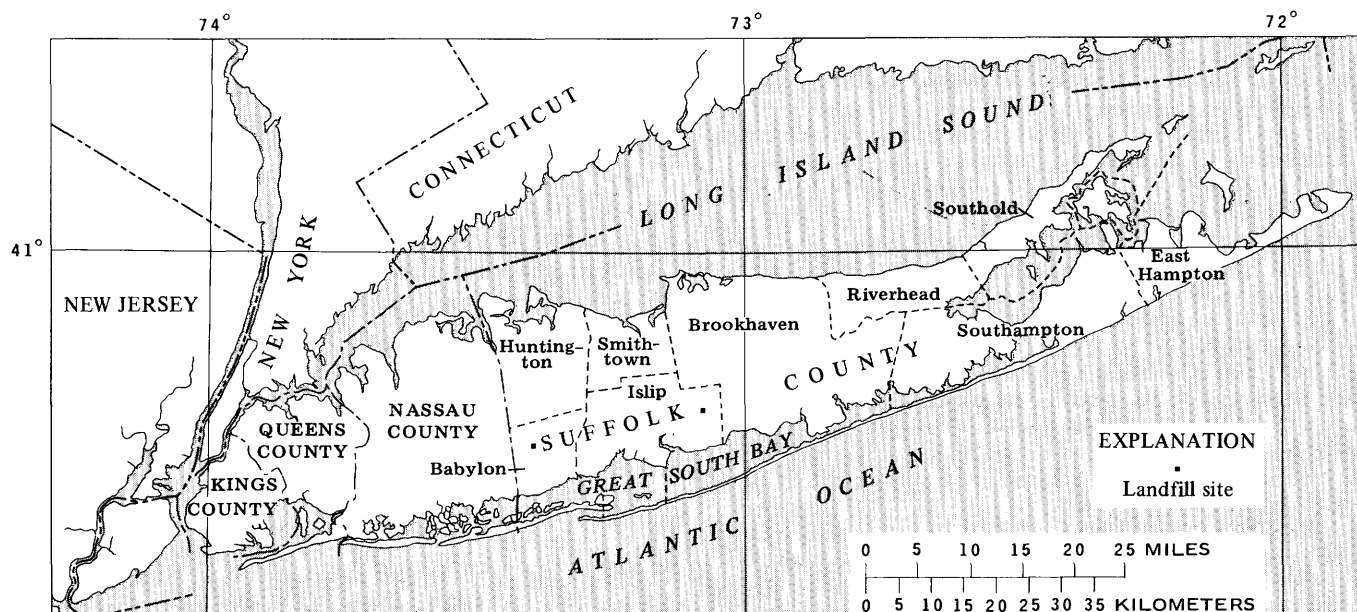


FIGURE 1.—Location of Babylon and Islip landfills.

TABLE 1.—*Fifteen-year (1951–65) mean seasonal and annual precipitation (in inches), Long Island, New York*
[From Miller and Frederick, 1969]

Period	Babylon	Islip
October 1–March 31	24	23
April 1–September 30	23	20
October 1–September 30	47	43

sumed through evapotranspiration, and generally ground-water recharge is negligible. During summer storms, however, several inches of precipitation may recharge the ground-water reservoir, especially through landfills, which lack significant vegetation and may have highly permeable surfaces.

Between 5 and 10 percent of the cool season's precipitation is snow (Miller and Frederick, 1969), but the snow usually does not stay long and is not a significant aspect of the hydrologic system.

The amount of precipitation infiltrating to the ground-water reservoir throughout the island is estimated to be 23 in. annually, with possibly 25- to 50-percent error (Cohen and others, 1968, p. 45).

HYDROGEOLOGY

The southern half of Long Island consists of a plain mantled by outwash associated with the terminus of a Wisconsin-age glacier. The outwash plain is underlain by deposits of stratified sand containing some gravel. The landfills studied are on the outwash plain about midway between the center of the Island and Great South Bay. Both landfills are about 60 ft above mean sea level. The outwash deposits are 90 and 210 ft thick at the Babylon and Islip sites, respectively. The saturated part of the outwash is called the upper glacial aquifer (Cohen and others, 1968). The distance between land surface and the water table is less than 30 ft at both sites. The water table fluctuates with local recharge from precipitation and with regional southward movement of ground water. Hydrographs of water levels in wells near the landfills show the extent of these fluctuations (fig. 2). At both sites, the landfill deposits are at or near the water table; consequently, there is no significant zone of aeration beneath the landfill, and the water table undoubtedly rises into the landfill at times.

Normally the water table recedes during the warm months, presumably because most of the precipitation during the growing season is consumed through evapotranspiration before it reaches the water table and also because some water is removed from the upper glacial aquifer by pumping and by transpiration. Recharge does occur during very wet periods of the warm months, though. In June 1972, 9.1 in. of rain fell near

the Babylon landfill, and, as a result, the water level in July was 0.8 ft higher than it had been in May. In the early summer of 1973, the upper glacial aquifer was again recharged because rainfall from April through July was 20 percent above the long-term average of 19 in. The following fall and winter were drier than usual, so the water table fell through the first half of 1974 with only a minor rise in late winter. Water levels near the two landfills during the 3-year study period fluctuated 6 ft at Babylon and 4 ft at Islip.

During the study period, the water table was between 12 and 18 ft below land surface at the Babylon site and between 12 and 16 ft below land surface at the Islip site. The water table (figs. 3 and 4) has a gradient of 0.0021 at Babylon and 0.0016 at Islip, except where intercepted by streams. The land surface slopes uniformly toward the south shore. In the Babylon area it intersects the water table at the headwaters of Santipogue Creek, about 2 mi south of the Babylon landfill, and, in the Islip area, at the headwaters of Brown Creek, 1.5 mi south of the landfill. In the area investigated, regional ground-water flow in the upper glacial aquifer is essentially parallel to the water table.

In the area investigated, the upper glacial aquifer, which lies between the water table and the Gardiners Clay or the Magothy aquifer, consists of coarse quartz sand, a small amount of heavy minerals, and some gravel. Quartz grains are generally coated with iron oxide, which probably has considerable sorptive capacity (S. Ragone, written commun., 1972). The deposit appears unusually uniform for outwash. Beds of materials finer than sand size were not found in the aquifer at any of the 34 Islip sites or the 35 Babylon sites drilled. Auger cuttings indicate that the outwash deposits at the two sites are lithologically similar.

At Babylon, the outwash deposits are underlain by Gardiners Clay, which consists of 10–13 ft of silty, gray clay. This deposit was found at every site where drilling was at least 90 ft deep and is very likely to be present everywhere in the area of Babylon that was studied. It is a barrier to the downward movement of water because of its low hydraulic conductivity. A cross-sectional analog model study of Long Island (Franke and Getzen, 1976) indicated a vertical hydraulic conductivity of 10^{-2} ft/d for the Gardiners Clay.

The top of the upper glacial aquifer is considered to be the water table, which fluctuates by at least 5 ft at both sites (fig. 2). At the Babylon landfill, the bottom of the aquifer is well defined by the top of the Gardiners Clay about 90 ft below land surface. At the Islip site the clay is not present, but at site I-19 (fig. 4), augering revealed gray, silty, micaceous fine sand 187 ft below the floor of the landfill, or 203 ft below land surface. A gamma-ray log of the hole shows a lithologic change

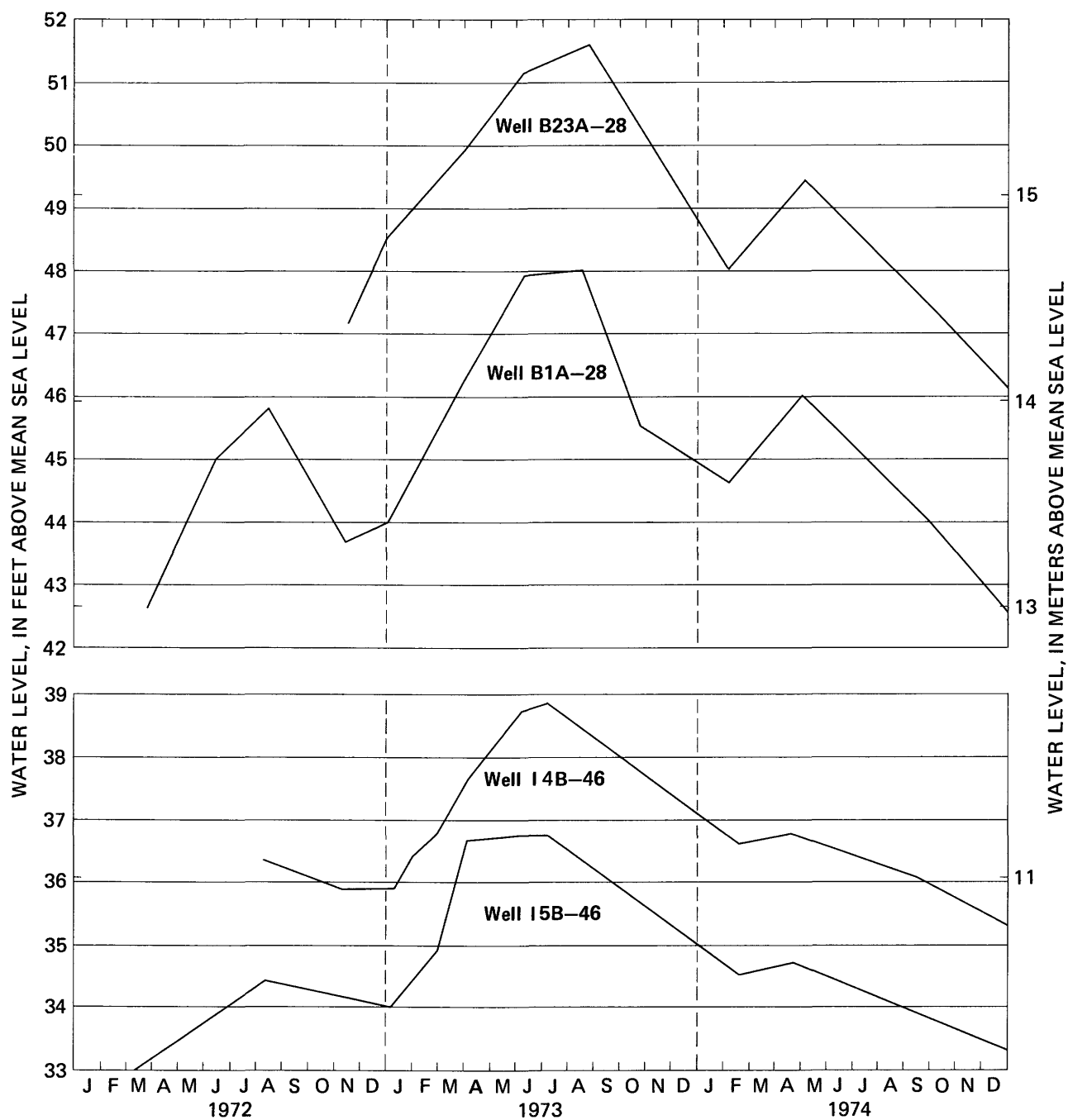


FIGURE 2.—Hydrographs of water-table wells at Babylon and Islip landfills, 1972-74.

170 ft below the pit floor. This fine-grained deposit seems to act as a hydrologic boundary; therefore, the bottom of the upper glacial was placed at 186 ft below land surface. The thickness of the upper glacial aquifer varies slightly with the altitude of the water table; it was 74 ft and 170 ft thick at Babylon and Islip, respectively.

The Gardiners Clay is overlain by a grayish-green sand that contains clam shells; the clay pinches out somewhere between 1 and 1.4 mi south of the landfill. This deposit is about 3 ft thick at site B-15 (fig. 3), 1.4 mi south of the landfill. Although this deposit may be closely related to the Gardiners Clay, its hydraulic conductivity is closer to that of the overlying outwash

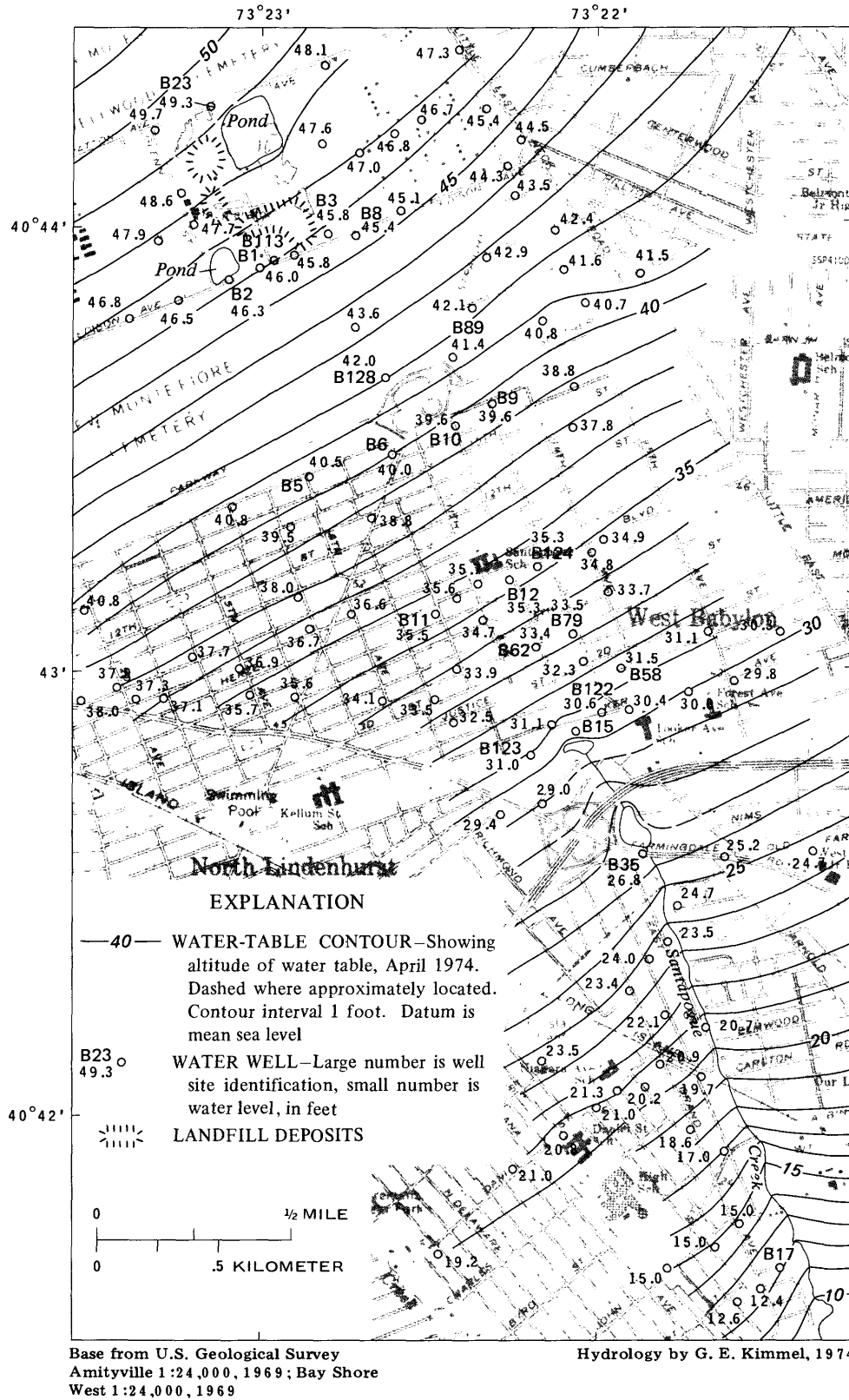


FIGURE 3.—Water table in April 1974 and location of wells referred to in text in vicinity of Babylon landfill.

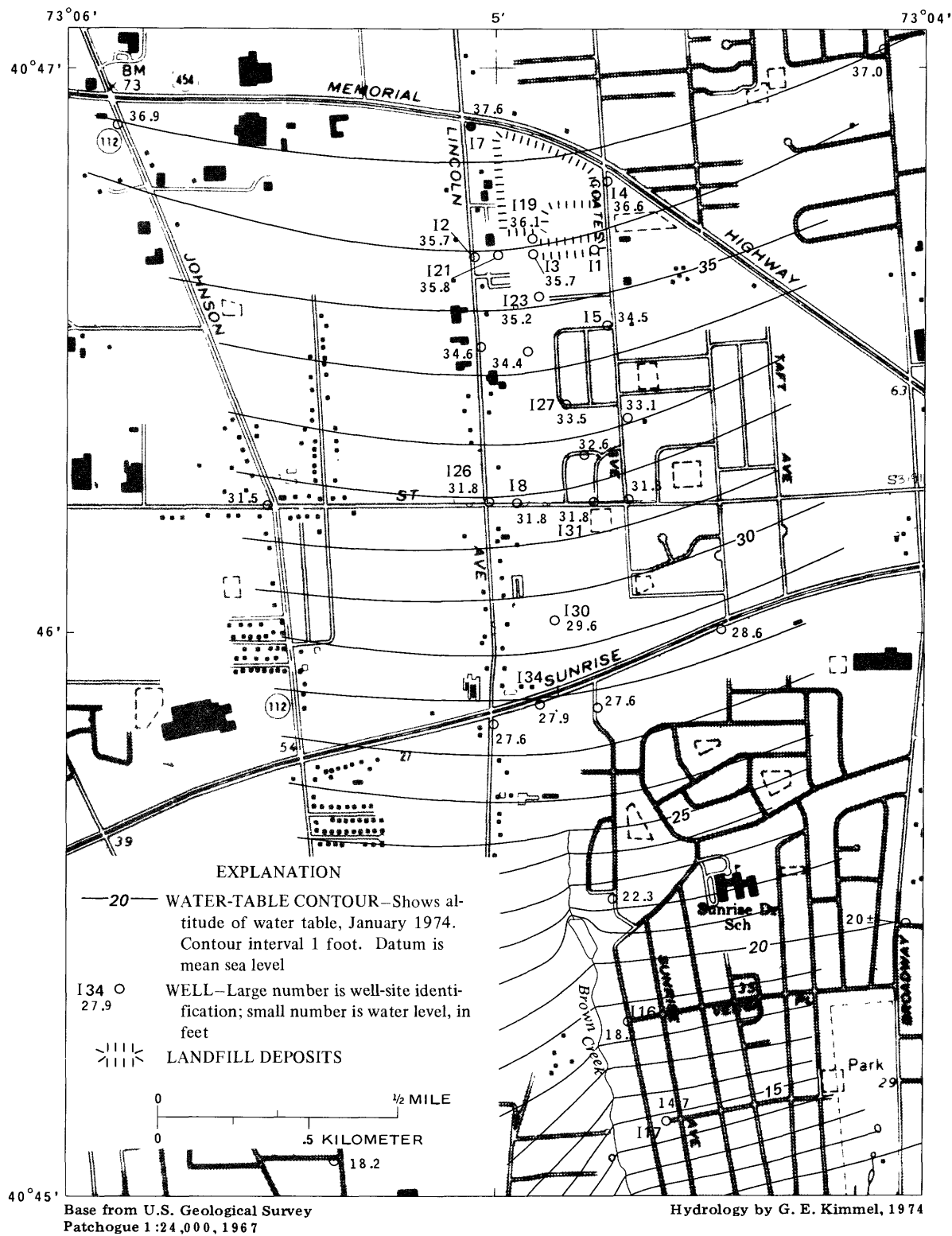


FIGURE 4.—Water table in January 1974 and location of wells referred to in text in vicinity of Islip landfill.

deposits, therefore, it is included in the upper glacial aquifer in this report.

Except near points of ground-water discharge to

streams, ground-water flow in the upper glacial aquifer is nearly horizontal (gradient about 0.002). This is supported by the observation that hydraulic heads,

measured at wells of different depths at the same site and at many locations in both areas, were virtually identical.

The hydraulic conductivity of the upper glacial aquifer was studied regionally by McClymonds and Franke (1972) with specific-capacity tests from wells. They used the relation proposed by Theis, Brown, and Meyer (1963, p. 332):

$$\bar{K} \approx 2,000 (Q/s)L$$

where \bar{K} = average hydraulic conductivity of materials opposite the well screen, in gallons per day per foot squared,

Q/s = specific capacity of the well, in gallons per minute per foot of drawdown, and

L = length of the well screen, in feet.

The constant factor 2,000 was found to be average for conditions on Long Island. Values of hydraulic conductivity contoured on maps (McClymonds and Franke, 1972, pl. 1) indicate that the average hydraulic conductivity (\bar{K}) in the area of the Babylon and Islip landfills is about 2,000 (gal/d)/ft² or 270 ft/d.

In the upper glacial aquifer south of the Babylon landfill, there are seven public-supply wells at four stations and, in the vicinity of the Islip landfill, five public-supply wells at two stations (fig. 5). These wells partly penetrate the aquifer, and screens are set near the bottom of the aquifer. About 35 percent of the aquifer is screened at Babylon and about 25 percent at Islip. Specific capacities ranged from 17 to 104 gal/ft of drawdown when the wells were new.

A computer analysis by T. E. Reilly (written commun., 1976) to determine the vertical and horizontal hydraulic conductivity took into consideration the partial penetration of the aquifer and used a 10:1 ratio of horizontal to vertical hydraulic conductivity. This ratio seems valid for regional models (Getzen, 1975; Franke and Getzen, 1976, p. 78). Where data were available from more than one supply well at a given station, only data from the well with the highest specific capacity were used; it was assumed that wells with the higher value would indicate more accurately the hydraulic conductivity in the area because those with lower values could be reflecting screen losses or other losses due to variations in well construction. Hydraulic conductivities obtained from three stations at Babylon and two stations at Islip are given in figure 5. Data from the two wells at the Albans Road station near Babylon were not included because their specific capacity was 17 and 18 gal/ft—only 20 percent of the next lowest value in the area. This amount suggests that the data from the wells would not accurately reflect the hydraulic conductivity at the well site. Values of hydraulic conductivity ranged from 470 to 500 ft/d for the

Babylon area and from 190 to 360 ft/d for the Islip area (fig. 5). The values for the Babylon area are about twice the values obtained by the method used in McClymonds and Franke (1972), and those values probably more nearly reflect the hydraulic conductivity in the area of the landfill than those of McClymonds and Franke.

Fine-grained deposits in the Magothy aquifer (Cohen and others, 1968) occur directly beneath the Gardiners Clay at Babylon and the outwash deposits at Islip. The Magothy aquifer is composed of beds of sand, silt, and clay and is between 800 and 1,000 ft thick in the area. At Babylon, water in the Magothy is currently (1974) 3 ft below the water in the upper glacial aquifer at the site of the Babylon landfill; it is probably lower because of public-supply withdrawals. Samples from 9 ft of drilling below the Gardiners Clay at Babylon indicate that the Magothy aquifer consists of lignitic and micaceous, silty, fine to medium sand. Pyrite is associated with the lignite. Observation wells of the kind used in this study that are screened in the Magothy aquifer at Babylon yield very little water—1 to 2 gal/min. The average hydraulic conductivity of the Magothy aquifer in the region of the two sites is estimated to be 50 ft/d (McClymonds and Franke, 1972). Although one hole is hardly sufficient to characterize the Magothy in the report area, logs for the three public-supply-well stations near Babylon (fig. 5) indicate that the sampling point was characteristic of the top of the Magothy in that area.

QUALITY OF NATIVE AND AMBIENT GROUND WATER

Fresh, uncontaminated ground water—native ground water—generally contains only small amounts of dissolved mineral matter on Long Island. Water from several wells in the vicinity of the landfills (table 2) had dissolved-solids concentrations of 51 mg/L (milligrams per liter) or less, which indicates a chemical character close to that of native ground water in at least some parts of the upper glacial aquifer.

In native ground water on Long Island, the concentration of individual species of the major cations (calcium, magnesium, sodium, potassium) and anions (bicarbonate, sulfate, chloride) is usually less than 10 mg/L, and only minor amounts of other ions are present. However, the quality of ground water in this region has been altered to various degrees, mostly by cesspool and septic-tank wastewater; thus, the ambient water in areas affected by landfill leachate can be different from native water.

Analyses of water samples listed in table 2 (well locations are shown in figs. 3 and 4) indicate that during the time of sampling (1972–73), the ground water in some places was of essentially native quality. Sub-

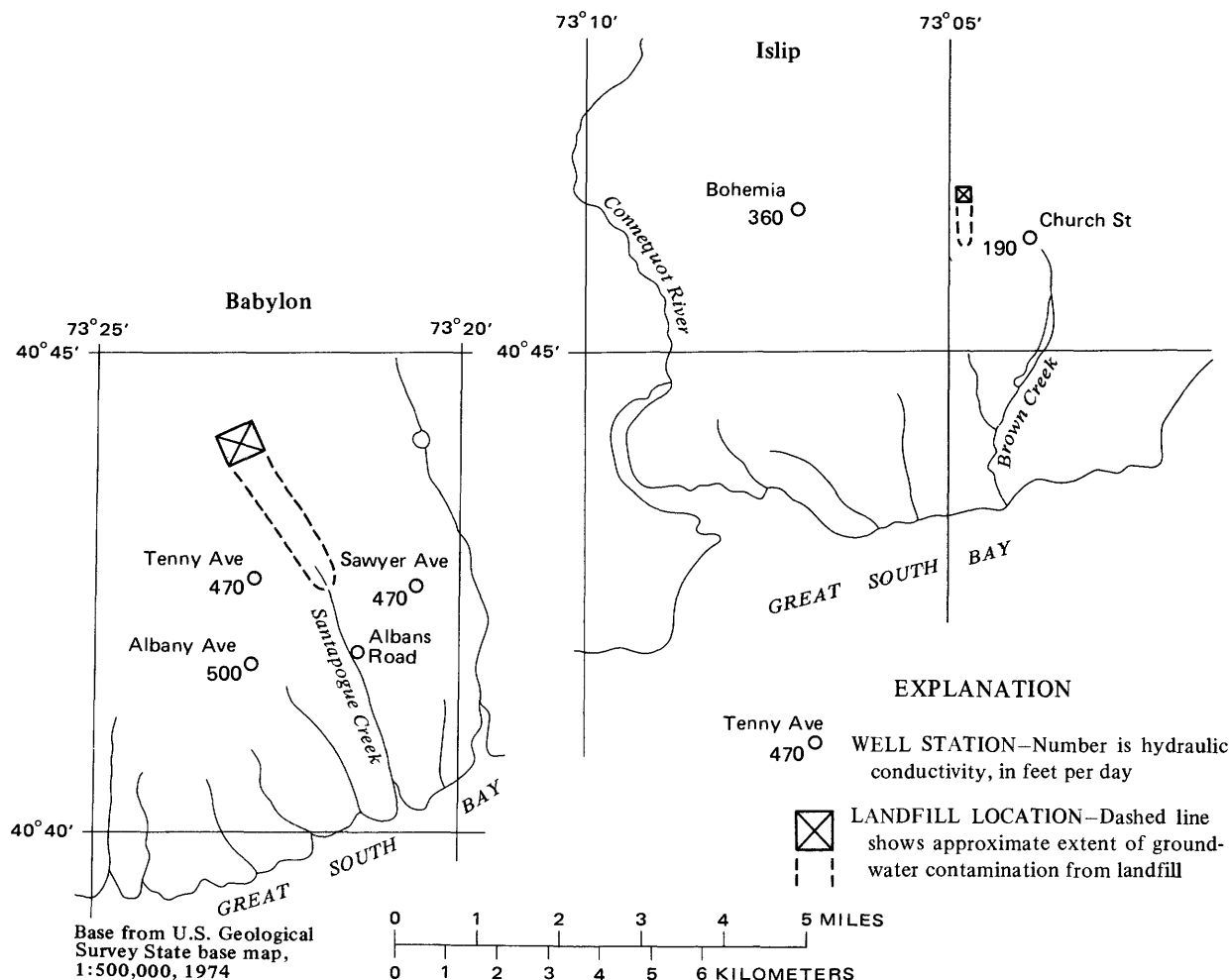


FIGURE 5.—Hydraulic conductivity at selected public-supply-well stations in upper glacial aquifer in vicinity of Babylon and Islip landfills.

sequent and previous samples from shallow wells I15A-28 and I7A-33 indicate intermittent contamination by domestic wastewater rich in ammonia, nitrate, calcium, sodium, sulfate, and chloride. Contamination of the ground water by domestic and possibly other kinds of wastes and by road salt has also altered the regional quality of the ground water to various degrees. The chemical character of water in the region of the plumes created by landfill leachate can be intermediate between that of plume water and native water. Although identification of the source of altered water is not always possible, leachate-enriched water in areas affected by the landfills generally has a much greater dissolved-solids content and hence a greater specific conductance than ambient ground water.

At Islip, regional alteration of water in the top part of the upper glacial aquifer has increased the dissolved-solids concentration to as much as 120 mg/L and the specific conductance to as much as 200 μmho (micromho). Consequently, at Islip a specific conduc-

tance greater than 200 μmho in the top of the aquifer indicates leachate enrichment. In the bottom part of the aquifer, where regional enrichment from domestic waste is small, a specific conductance as low as 60 μmho could indicate leachate-enriched water, and there the definition of areas of leachate-enriched water is even better than near the top of the aquifer.

Water samples from the Babylon area indicate that water from the upper glacial aquifer has been altered through virtually its entire depth by domestic waste. Only one sample, from well B17C-68 (table 2 and fig. 3), was of native quality. Specific conductance of the water in the upper glacial aquifer around the toe, or end, of the Babylon plume commonly ranged from 200 to 400 μmho . Because ambient ground water in the area of the plume seldom exceeded 400 μmho , this value was taken as the basis for detection of the leachate plume at Babylon.

The concentrations of trace elements given in table 2 were assumed to be those of native water because no

TABLE 2.—Chemical analyses showing quality of native water in the upper glacial aquifer in study areas

Constituents and characteristics	Well No. and date of sample collection						
	B17C-68	I1C-99	I5A-28	I7A-33	I8A-25		I34D-130
	2-9-72	3-9-72	2-17-72	11-16-73	2-24-72	5-16-73	10-19-73
Major constituents (mg/L)							
Silica (SiO ₂)	13	12	7.9	4.6	6.1	6.0	14
Iron (Fe), total	2.2	.1	.05	.58	.05	.15	.21
Manganese (Mn)	.04	.07	.0	.0	.02	.10	.02
Calcium (Ca)	4.2	2.0	3.5	5.0	1.7	2.2	2.2
Magnesium (Mg)	1.0	.8	1.8	1.7	.8	.9	1.2
Sodium (Na)	4.7	4.0	5.6	4.5	3.5	3.7	4.0
Potassium (K)	.70	.4	1.5	.6	.6	.4	.5
Bicarbonate (HCO ₃)	16	10	9.0	7.0	2.5	3.0	15
Sulfate (SO ₄)	8.0	.5	13	10	7.5	8.0	.4
Chloride (Cl)	6.7	7.0	6.7	6.2	5.5	6.0	4.5
Fluoride (F)	.0	.1	.0	.1	.0	.6	.2
Nitrate as N	.27	.20	.80	.54	.10	-----	.03
Nitrite as N	.02	.005	.24	.0	.002	-----	.0
Ammonia as N	.13	.04	.54	.01	.080	-----	.01
Nitrogen, organic as N	.29	.35	.45	.09	.520	.0	.0
Phosphorus, as P	.04	.06	.28	.0	.10	.0	.01
Trace elements (μg/L)							
Arsenic (As)	0.0	3	11	0.0	0.0	-----	-----
Copper (Cu)	0	0	0	0	0	-----	-----
Lead (Pb)	4.0	.8	4.0	0	2	-----	-----
Mercury (Hg)	<.5	<.5	<.5	.8	.8	-----	-----
Nickel (Ni)	2.0	4.0	4.0	-----	7	-----	-----
Selenium (Se)	0.0	2.0	.0	-----	8	-----	-----
Zinc (Zn)	20	.0	40	30	20	.0	50
Other characteristics							
Dissolved solids (sum, mg/L)	47	33	51	36	28	29	34
Specific conductance (μmho/cm at 25°C)	66	45	73	72	43	46	73
pH	6.8	6.7	6.4	5.9	5.4	5.3	7.2
Temperature (°C)	11	11	13	12	11	11	11
Depth of screen below water table (ft)	65	67	12	13	14	14	110

comparisons for these elements were available. Copper, which was not found in any of the native-quality samples, is virtually ubiquitous in ground water enriched with septic-tank waste (table 8) in the Towns of Babylon and Islip.

DESCRIPTION OF LANDFILLS

The Babylon and Islip landfills have been described in Kimmel and Braids (1975). The Babylon site contains about twice as much refuse as the Islip site even though it is 14 years younger. Both landfill sites have incinerators, both were used for gravel-mining operations before deposition of refuse, and both accumulated about the same kind of refuse. Scavenger waste, which is the collected effluent from septic tanks and cesspools, was discharged at both sites. In summary, the Babylon landfill had a much greater rate of accumulation than the Islip landfill.

BABYLON

Refuse has been deposited at the Babylon site in three separate piles. A photograph of the site (fig. 6) shows two piles; a third has been partly removed and is not distinguishable in the photograph. The third pile was the original refuse heap and is about 500 ft east of the incinerator building. The north pile was completed in 1965, but some refuse has been added since. The southeast pile is currently (1974) active and is being built northward into the sand and gravel mining operation (large pond in photo). Sand and gravel is excavated to depths of at least tens of feet below the water table in the northeast and southwest quadrant of the site. Currently (1974), only noncompostable rubbish is deposited below the water table, but other types of refuse may have been deposited there previously.

About 40,000 gal/d of scavenger waste was treated to some extent in a facility on the west side of the site and



FIGURE 6.—View of Babylon landfill, looking south. Incinerator facility is at right. Sand-and-gravel operation, center left, is in water-table pond. Scavenger-waste-facility ponds are to the front and right (north and west) of large refuse pile in center. Photograph taken June 13, 1974.

was then discharged into lagoons along the north and west parts of the site. These lagoons are above the water table and drain poorly because their bottoms have become clogged with sludge. Periodically, bottom sludge was excavated from the lagoons and placed over the landfill refuse.

The refuse was at least 60 ft thick and possibly as much as 80 ft thick except in the earliest pile, which was about 30 ft thick in 1974. The total volume of refuse in 1974 was estimated to be 2.3×10^6 yd³. The landfills were covered with sand and gravel and, more recently (1973), with sewage sludge. Compaction occurs both naturally and by the weight of heavy equipment being driven over the surface.

The refuse comes from homes and light manufacturing. Much of the domestic garbage was incinerated before deposition on the landfills. A large part of the manufacturing refuse consists of plastics or other inert material. The landfill operation began in 1947; it covered 25 acres in 1973.

ISLIP

The Islip landfill occupies a pit from which sand has been mined. Landfilling reportedly began in 1933, and, by 1938, the landfill covered a little less than 1 acre. At that time there was no incinerator, but a 1947 aerial photograph shows an incinerator building and 1.5 acres of refuse. In 1973, refuse covered 17 acres, mostly in the north part of the landfill site. A narrow strip of refuse covering about 2 acres was deposited on the south side of the pit in 1972. A photograph of the site (fig. 7) shows the extent of the landfilling as of 1974.

Most of the main pile of refuse was about 40 feet thick, with gently sloping sides. The refuse is similar to



FIGURE 7.—View of Islip landfill, looking north. Incinerator facility is at center left with large smokestack. Old incinerator beyond it (to north) has stack without smoke. Sand and gravel is not currently (1974) mined here, but a gravel-washing operation is at lower right of landfill. Water-table ponds are in right center. Photograph taken June 13, 1974.

that in the Babylon landfill, but scavenger waste, which was once discharged here, has not been discharged since the late 1960's. The landfill is covered with sand and has some grassy vegetation on the north side. The infiltration rate is probably high.

CHEMICAL AND PHYSICAL PROPERTIES OF LEACHATE PLUMES

METHODS OF STUDY

Ground-water quality near and downgradient from the landfills was determined by installing 2-in. wells screened at various depths. A letter-and-number code was used to indicate screen setting, well depth, and water-quality data on maps. Well numbers are preceded by B or I (for Babylon or Islip) and are followed by a screen-depth code letter and the well depth. Table 3 gives the letter code for the screen settings. For example, in well B2C-86, B2 indicates Babylon site 2, and C-86 indicates that the well is screened at some interval between 60 and 80 ft below the water table and is 86 ft deep. Some wells have 3-ft screens, others have 4-ft screens. At Babylon, 90 wells were installed at 40 sites, and, at Islip, 76 wells were installed at 34 sites. In addition, water samples from 26 firewells at Babylon supplemented the water-quality data, and water-level measurements from 80 firewells supplemented measurements from 2-in. wells. The firewells were installed prior to this study for fire protection and consist of 4-in.-long screens that bottom at 45 ft. The firewells were screened at B depth (table 3). The designation BFW in this report indicates Babylon firewells.

TABLE 3.—Letter code for well-screen depths

Letter code	Depth of screen setting below average water table (in ft)	
	Babylon wells	Islip wells
A-----	0-20	5-15
AA-----	---	16-28
B-----	30-40	30-44
BB-----	47-57	50-60
C-----	60-80	65-87
D-----	---	94-114
E-----	---	123-146
F-----	---	186

¹One well at site 19.

Sampling of the ground water for chemical analysis began in the winter of 1972 and was continued annually through 1974. Water from most wells, therefore, had three chemical analyses as well as several additional analyses for pH and conductivity. Chemical analyses for inorganic species were made at the U.S. Geological Survey laboratory in Albany, N.Y. Analyses of 91 water samples for total organic carbon were made at the Geological Survey laboratory in Denver, Colo. In 1973 and 1974, concentrations of nitrate and ammonium were analyzed at the Geological Survey in Mineola, N.Y., by specific-ion electrodes.

Procedures for sampling were predicted on the assumption that removing as much water as possible in a short time would flush the well bore, pump, and hoses and would thus minimize contamination from the sampling equipment and well. Yields of as much as 60 gal/min were attained. Withdrawals for some samples were large enough that water could have come from as much as 6 ft beyond the well screen. It is assumed that, because of the high hydraulic conductivity and uniformity of the aquifer, withdrawals of this magnitude did not significantly distort the shape of the leachate plume.

Techniques for preservation of nitrogen species in water samples between the time of sampling and the time of analysis have been questioned in recent years. Several procedures were selected for this study on the basis of current information. Variations included chilling at the sampling site, adding mercuric chloride as a preservative, analyzing the nitrogen components of the sample within 3 days of sampling, and analyzing nitrate and ammonium by specific-ion electrode. Results from the electrode method differed significantly from the standard laboratory-analysis results at times, especially at very low concentrations.

COMPOSITION OF PLUMES

MAJOR COMPONENTS

Leachate-enriched ground water is characterized by enhanced concentrations of most ions that form the bulk of dissolved solids in regional ground water. Thus,

the concentration of cations sodium (Na^+), potassium (K^+), calcium (Ca^{+2}), and magnesium (Mg^{+2})—and of anions chloride (Cl^-), bicarbonate (HCO_3^-), and sulfate (SO_4^{-2})—in leachate plumes are higher than in ambient water. These ions are dissolved directly from refuse by percolating water and are released from the organic matrix by biological decomposition. Results of chemical analyses of leachate-enriched water are given in table 4.

The landfills consist of incinerator ash, unincinerated refuse, and scavenger waste (septic-tank and cesspool contents). Unincinerated refuse includes demolition waste, compostable materials, metals, textiles, and plastics.

As part of the effort to identify the source of the components of leachate, 50-gram samples of fresh incinerator ash were taken from both landfills and were placed in distilled water for 4 weeks to simulate the leaching of ash by rainwater. The suspension was then filtered and analyzed (table 5). Because rainwater contains some dissolved solids (about 20 mg/L at Mineola, Long Island, N.Y.) and is characteristically acidic (pH about 5), this method is not exact, but the low concentration of soluble material and the considerable amount of HCO_3^- that dissolved out of the ash samples strongly suggest that, chemically, the distilled water is similar enough to rainwater to justify its use in these tests.

In the Babylon ash sample, Ca^{+2} and HCO_3^- were the major ions present, with 425 and 1,160 mg/L, respectively. In the Islip ash sample, the ion pair Ca^{+2} and SO_4^{-2} were the dominant cation and anion, with 275 and 500 mg/L, respectively. The analyses indicate that all major components of leachate-enriched water were also major components of ash leachate, although the SO_4^{-2} concentration in the plumes was erratic. Minor amounts of zinc were detected in the samples. Because the incinerator ash, which readily contributes ions when placed in water, constitutes a large proportion of the refuse in both landfills, it is assumed that it is also a major chemical contributor to landfill leachate.

The pH of ambient water in the upper glacial aquifer in the vicinity of the landfills has been found to be as high as 7.2 (well I34D-130, table 2), but it is commonly about 5. In the plume water, it is about one pH unit higher than that generally found in ambient water. The pH range of leachate-enriched water at Babylon was from 4.9-7.3 and at Islip was from 4.6-6.8.

BICARBONATE

Bicarbonate was the dominant anion in most of the leachate-enriched ground-water samples. Its concen-

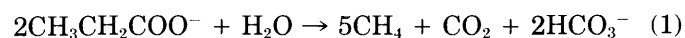
TABLE 4.—Chemical analyses of leachate-enriched ground water in study areas

Constituents and characteristics	Well No. and date of collection								
	Babylon landfill					Islip landfill			
	B1BB-69 7-3-74	B128C-78 7-31-74	B12C-70 5-3-73	B62BFW 7-26-74	B35C-67 7-29-74	I19B-61 8-1-74	I23E-144 6-25-73	I8D-22 8-13-74	I30D-120 8-9-74
Major constituents (mg/L)									
Silica (SiO ₂)	<1	8.9	5.3	4.1	8.8	11	7.1	14	16
Calcium (Ca)	100	93	---	0	---	0	---	20	---
Magnesium (Mg)	28	53	22	8.8	5.5	17	55	7.3	6.2
Sodium (Na)	700	280	160	120	43	68	260	170	39
Potassium (K)	110	16	25	29	5.3	52	75	3	1.1
Bicarbonate (HCO ₃)	898	277	426	208	54	392	1,010	59	31
Carbonate (CO ₃)	0	0	0	0	0	0	0	0	0
Alkalinity as CaCO ₃	737	227	349	171	44	322	828	48	25
Sulfate (SO ₄)	100	19	61	36	60	26	57	150	74
Chloride (Cl)	910	410	200	150	55	92	400	150	29
Fluoride (F)	.3	.1	.2	0	0	.4	.2	0	.1
Bromide (Br)	.15	4.2	---	.85	.6	1.3	---	2.3	.36
Nitrate (NO ₃)	2.2	2.7	.05	1.6	.67	.69	.34	1.1	.45
Nitrite (NO ₂)	---	---	0	---	---	---	.17	---	---
Nitrogen, ammonia as N	43	7.8	35	15	5.3	14	12	0.08	0
Minor constituents (μg/L)									
Arsenic (As)	14	1	---	0	---	29	---	2	---
Boron (B)	2,500	770	1,000	780	300	690	1,200	430	240
Copper (Cu)	0	20	---	10	---	0	---	0	---
Lead (Pb)	1	0	---	0	---	6	---	0	---
Selenium (Se)	1	1	---	2	---	0	---	2	---
Strontium (Sr)	680	1,400	---	160	180	370	---	230	130
Zinc (Zn)	20	80	10	10	---	10	190	10	---
Aluminum (Al)	0	50	---	0	---	0	---	20	---
Iron (Fe)	24,000	1,200	210	180	1,200	39,000	3,400	100	110
Other characteristics									
Dissolved solids									
Residue at 180°C (mg/L)	2,300	1,220	652	508	219	490	1,560	555	223
Calculated (mg/L)	2,475	1,030	778	486	230	575	1,548	539	196
Hardness as CaCO ₃ (mg/L)	370	450	213	120	65	210	676	60	61
Noncarbonate hardness (mg/L)	0	220	0	0	21	---	---	---	---

tration ranged from 2,700 mg/L (analysis from well B3C-86, January 1972) near the Babylon landfill to about 50 mg/L near the toe of the plume at Babylon (fig. 8), which was about the maximum concentration of bicarbonate in ground water altered by domestic

waste in the Babylon area. At Islip, bicarbonate concentrations ranged from 1,010 mg/L (well I23E-144, table 4) near the landfill to about 30 mg/L near the toe of the plume. Ambient water near the top of the Islip plume was somewhat altered but was of virtually native quality near the bottom part of the plume. The top part of the plume was distinguishable by bicarbonate concentrations greater than 20 mg/L.

Bicarbonate is one of the best indicators for tracing the leachate plumes. It is derived from the leaching of incinerator ash (table 5), anaerobic decomposition of organic matter, and dissolution of carbon dioxide. The mechanism of bicarbonate formation in incinerator ash is not understood, but it probably is related to incomplete combustion of organic matter before it is doused with water. An example of anaerobic decomposition is provided by butyric acid, a common microbial metabolite:



where methane, carbon dioxide, and bicarbonate are

TABLE 5.—Chemical analyses of solutions derived from incinerator ash

Constituents	Babylon ash		Islip ash	
	mg/L	meq/L	mg/L	meq/L
Silica (SiO ₂)	20	---	1.5	---
Calcium (Ca)	425	21.21	275	13.72
Magnesium (Mg)	61	5.02	18	13.72
Sodium (Na)	11	.48	37	1.61
Potassium (K)	77	1.97	11	.28
Bicarbonate (HCO ₃)	1,160	19.01	260	4.26
Sulfate (SO ₄)	300	6.25	500	10.41
Chloride (Cl)	210	5.92	38	1.07
Fluoride (F)	.7	.04	.4	.02
Nitrate (NO ₃) as N	0	.00	.1	.01
Ammonia (NH ₄) as N	6.5	.46	---	---
Copper (Cu)	0	---	.02	---
Zinc (Zn)	.15	---	.09	---

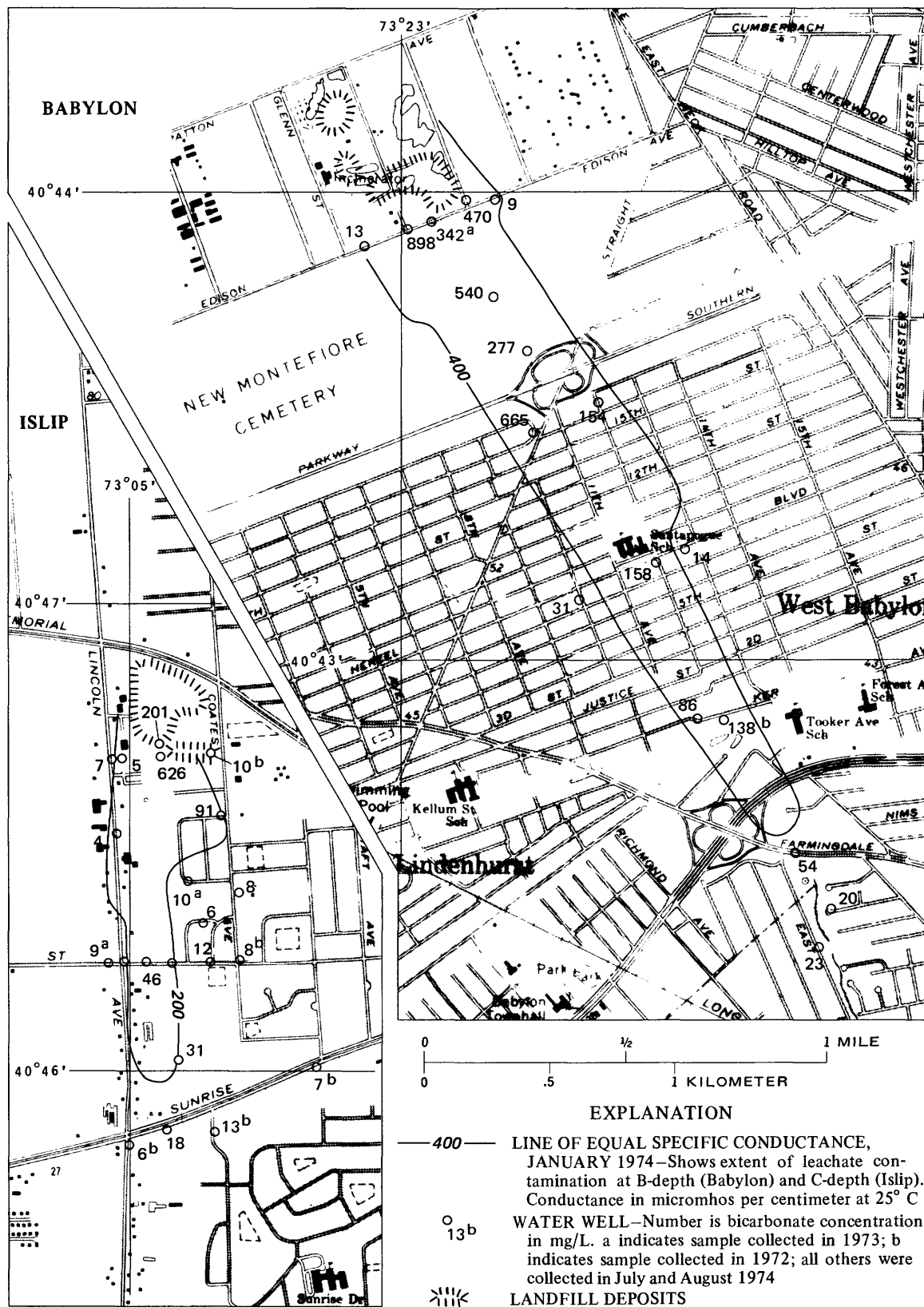
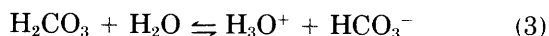


FIGURE 8.—Bicarbonate concentrations at C-depth wells in vicinity of Babylon and Islip landfills.

the products (Lawrence, 1971, p. 173). Dissolution of carbon dioxide in water



yields carbonic acid (eq. 2), which dissociates to hydronium and bicarbonate ions. The concentration of carbonic acid in equation 2 is proportional to the partial pressure of carbon dioxide in the atmosphere in contact with water. Inside the landfills, the partial pressure of carbon dioxide is much greater than normal because carbon dioxide is part of the organic decomposition process (eq.1), therefore, the equilibrium (eq. 2 and 3) is shifted to the right, producing more bicarbonate. Together, these sources of bicarbonate result in concentrations many times greater in leachate than in ambient ground water.

SULFATE

The distribution of sulfate in the plume is shown in figure 9. The sulfate probably comes from solutions of incinerator ash that travel aerobically (table 5) to ground water. Thus, sulfur that originates in the sulfate form is transported to the ground-water body unchanged. Unincinerated organic refuse that was decomposing aerobically would also contribute sulfate.

The combination of an aerobic environment, suitable bacteria, and a carbonaceous substrate provides an environment favorable for the reduction of sulfate to sulfide. These conditions are found in the interior of refuse piles and continue for several feet beneath and beyond in the ground water. These conditions also favor reduction of iron to the more soluble ferrous state. The presence of iron and sulfide under conditions of low oxidation-reduction potential and near-neutral pH results in the precipitation of ferrous sulfide, which removes these ions from solution. The wide range in sulfate concentration in leachate-enriched ground water near the landfills may result from this chemical reduction and from ion precipitation. Iron concentrations as high as 440 mg/L and sulfate concentrations as high as 290 mg/L were found in the plume; iron concentrations were generally higher than sulfate concentrations. The sulfur, after reduction to sulfide, is chemically bound; this fact may account for the low sulfate content at many locations near the landfills.

Incinerator ash and cesspool and septic-tank effluents are other major sources of sulfate in the leachate (table 5), and the uneven distribution of incinerator ash in the landfill may be another reason for the wide variation in sulfate concentrations in

leachate-enriched ground water near the Babylon landfill. However, even where the plume is highly enriched with leachate, the sulfate concentration is likely to constitute less than 15 percent of the total anions (fig. 9). In the Babylon plume, the proportion of sulfate to total anions increases to approximately 30 percent as the plume water moves downgradient (fig. 9). Simultaneously, bicarbonate becomes diluted relative to sulfate, further increasing the sulfate percentage.

Sulfate concentration in both plumes is variable, but in ambient ground water it is much lower at Islip than at Babylon. Concentrations greater than 10 mg/L at Islip probably indicate leachate enrichment (fig. 9).

CHLORIDE

As shown in table 6, chloride concentrations in leachate-enriched water at Babylon in 1973 were as high as 1,300 mg/L—five times the recommended drinking-water limit (U.S. Public Health Service, 1962). Chloride concentrations mapped in figure 10 are listed in table 6. Chloride is a conservative ion; that is, it does not interact appreciably with other chemical species in ground water and the sediments. The high correlation coefficient of sodium to chloride at Babylon, 0.99 (fig.11), suggests that both ions are conservative and that chloride would be a good ion for tracing leachate if secondary sources could be subtracted from existing concentrations in ground water. The close correlation between chloride concentration and specific conductance further indicates that chloride is conservative and a constant contributor to the conductance of the water compared with other ions in the water (correlation coefficient is 0.783). Because of this property (see section on "Dispersion Model"), chloride concentration is useful for calculation of the dispersion coefficient for the aquifer.

MAJOR CATIONS

Maximum concentrations of the cations Na^+ , K^+ , Ca^{+2} , and Mg^{+2} at the two sites given in table 7. The analyses for these ions show heterogeneity in the addition of these constituents to ground water. Concentrations of cations plotted against conductance had a correlation coefficient of 0.8 for sodium at both sites and of 0.77 for calcium and 0.8 for magnesium at Babylon, whereas concentrations of calcium, magnesium, and potassium had correlation coefficients of less than 0.7. The major cations are contributed by leachate in a nonuniform way, thus, at high concentrations there is considerable scatter in the correlation of ions and conductivity. Water samples taken outside the plumes were not included in the statistical analyses.

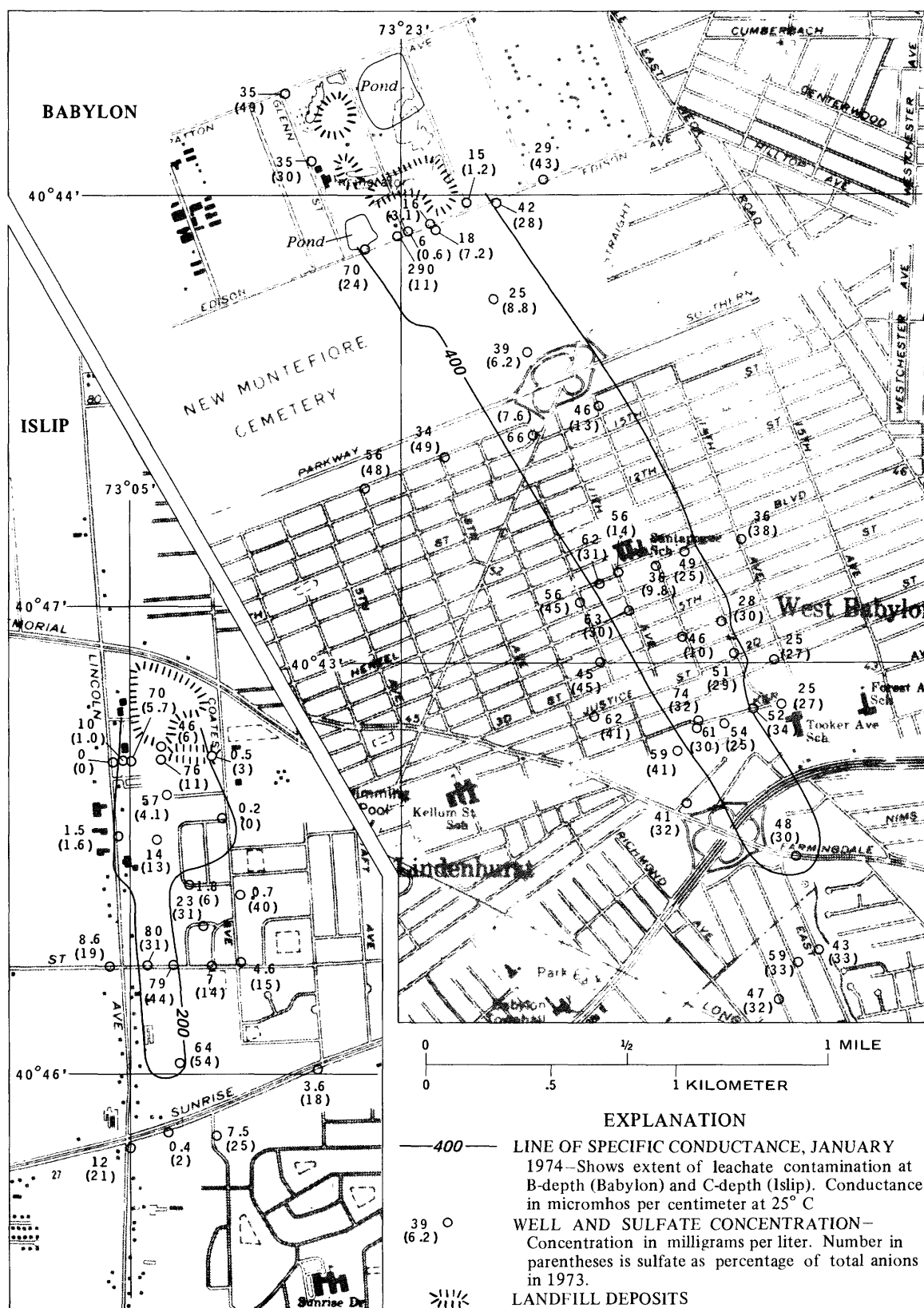


FIGURE 9.—Sulfate concentrations in ground water in and near leachate plumes at Babylon and Islip landfills.

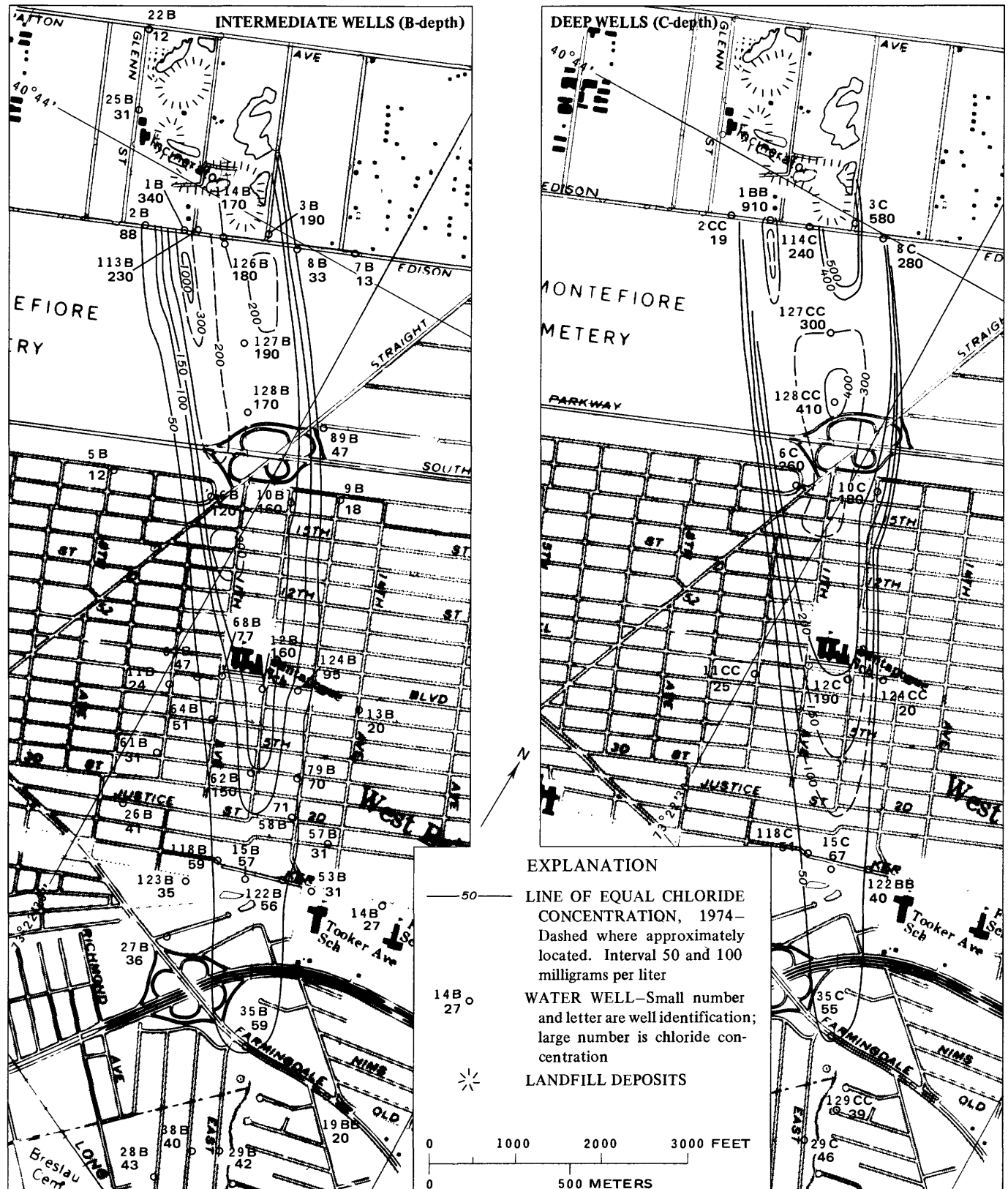


FIGURE 10.—Chloride concentrations in ground water in vicinity of Babylon landfill plume.

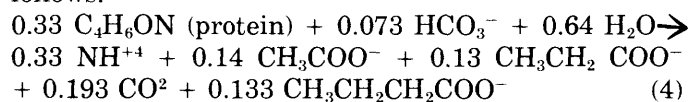
TABLE 6.—Chloride concentrations in upper glacial aquifer at Babylon landfill

[In milligrams per liter]

Well-site No.	Medium-depth wells (B depth)			Deep wells (C depth)		
	1972	1973	1974	1972	1973	1974
1	160	1300	340	130	950	910
2		99	88		20	19
3	240	290	190	380	220	580
5	12	12				
6	220	200	120	220	360	260
7	9.5	13				
8	65	68	33	14	170	280
9	18					
10	120	130	160	21	180	180
11	31	27	24	12	15	25
12	140	140	160	170	200	190
13	20					
14	27	27				
15	57			61	67	
19	20	20				
22	12					
25	31					
26	39	43	41			
27	36					
28	45	42				
29	42	44	42	35	43	46
35	57	48	59	58	45	55
38		46	40			
53		31				
57		31				
58		72	71			
61		31				
62		150	150			
64		50	51			
67		50	47			
68		110	77			
79		22	70			
89			47			
113		300	230			
114		160	170	240		
118		60	59	39	51	
122		52	56	61	40	
123		35				
124		90	95	13	20	
127		190	190	270	300	
128		180	170	410		
129					39	

NITROGEN COMPOUNDS

Nitrogen is present in refuse as a component of organic matter. It is especially available in unincinerated garbage and scavenger waste. Upon decomposition of organic matter, proteinaceous nitrogen is converted to nitrate under aerobic (oxidizing) conditions or to ammonia under anaerobic (reducing) conditions. Because anaerobic conditions prevail in landfills, ammonium-nitrogen is the major nitrogen species in leachate. Ammonium can be produced from proteinaceous wastes by microbiological decomposition as follows:



(McCarty, 1971, p. 102). Carbon dioxide produced in this reaction dissolves in water to create more bicarbonate (eq. 2 and 3). The organic acid thus produced are easily metabolized and probably remain in solution for only a short time.

Ammonium concentrations as high as 46 mg/L (well I19C-96, 6/20/73) were found near the Islip landfill, and as high as 90 mg/L (well B3B-51, 5/20/73) near the

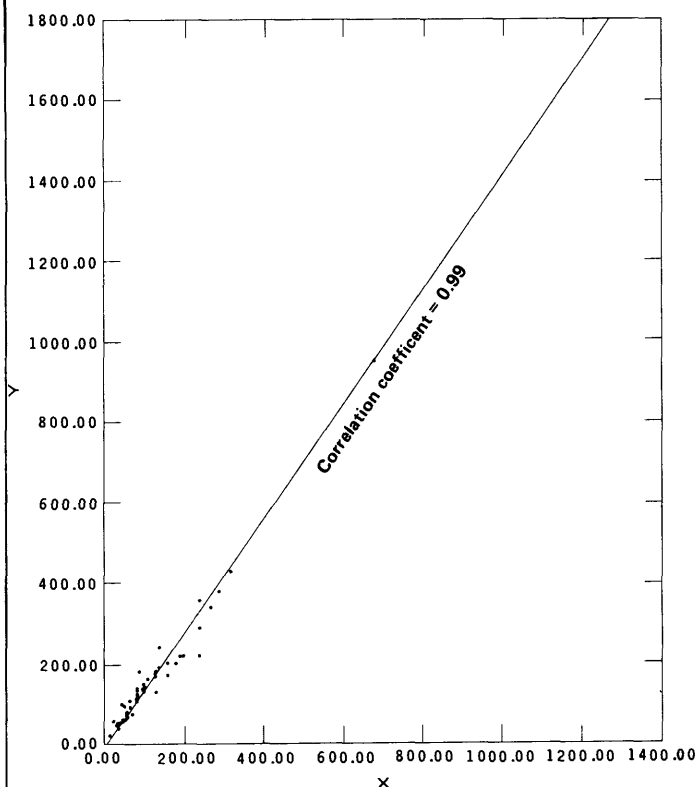


FIGURE 11.—Relations of sodium and chloride content in leachate plume at Babylon.

Babylon landfill. Data are insufficient to determine the concentration gradient at Islip, but the ammonium concentration at Babylon between the landfill and site B12 (fig. 3) ranged from 35 to 90 mg/L and tapered off to about 10 mg/L near the toe of the plume.

Nitrate-nitrogen is present in refuse such as lawn cuttings and other vegetable matter. When leached through reducing zones, it may be denitrified to nitrogen gas; this may explain the low nitrate concentrations in leachate plumes near their point of origin where reducing conditions prevail. Nitrate as a percentage of total nitrogen increases downgradient to about 40 percent of the total nitrogen near the toe of the plumes (fig. 12). Contamination from septic tanks

TABLE 7.—Concentration of cations in samples from Islip and Babylon plumes, in milligrams per liter

[Location of sampling sites is shown in figs. 3 and 4]

Well	Date of collection	Cations			
		Na ⁺	K ⁺	Ca ⁺⁺	Mg ⁺⁺
Islip:					
121E-146	11-15-73	560	11	45	26
123E-144	6-25-73	260	75	180	55
Babylon:					
B1B-50	4-19-73	860	65	170	36
B1C-69	7-3-74	700	110	100	28
B3C-86	1-25-72	290	30	565	

*Maximum concentration at landfill.

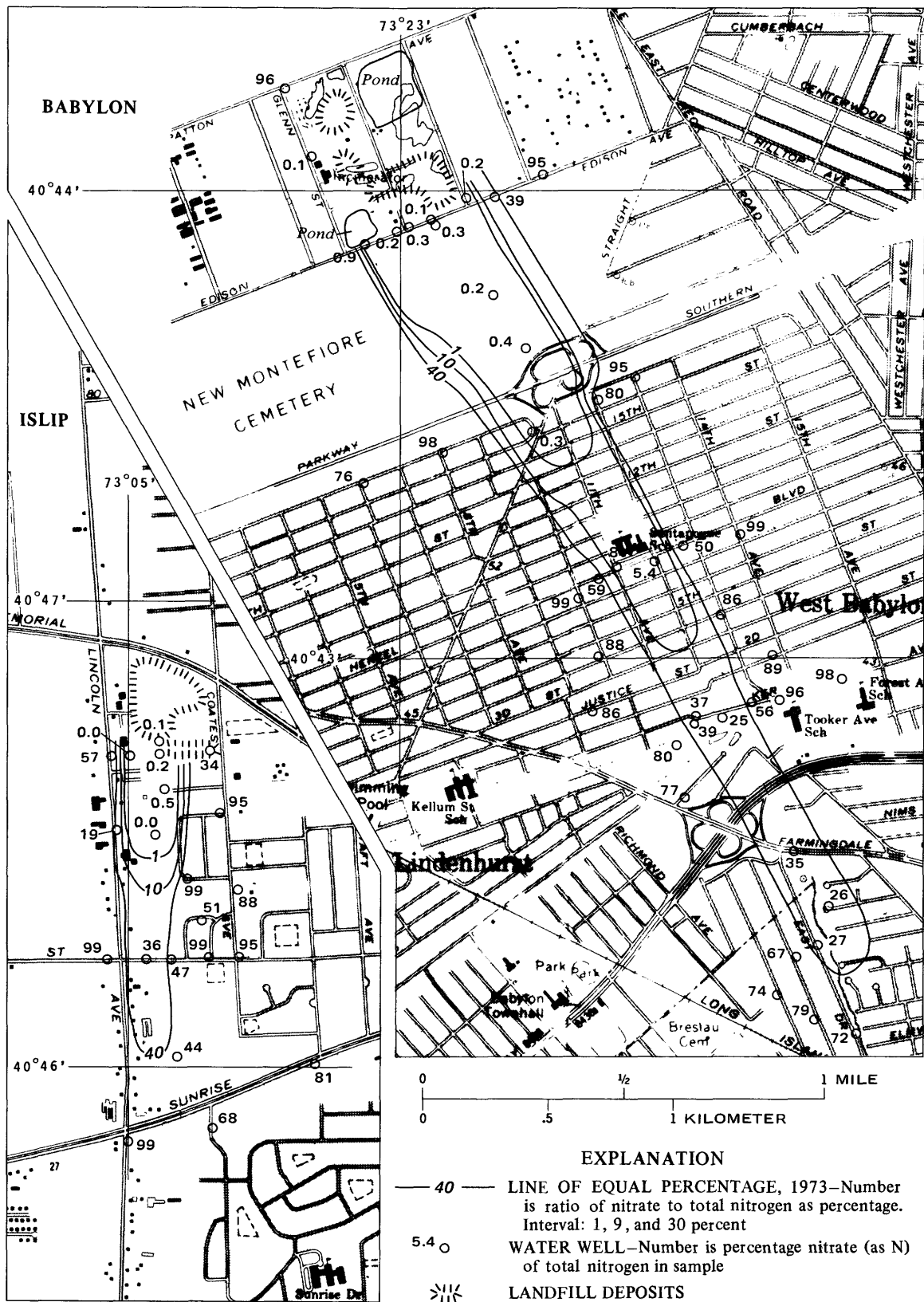


FIGURE 12.—Nitrate concentrations as percentage of total nitrogen in water from B-depth wells at Babylon landfill and C-depth wells at Islip.

at Babylon, and possibly oxidation of ammonium to nitrate, cause the increase in nitrate content downgradient from the landfill.

Within the plumes, nitrate concentrations did not exceed the recommended limit (U.S. Public Health Service, 1962) of 10 mg/L as nitrogen. Outside the plume at Babylon, the limit was equaled or exceeded in 11 samples.

SUMMARY OF MAJOR COMPONENTS

Stiff diagrams of the major ionic components of leachate-enriched water (fig. 13) show that the chemical character of plume water changes between the refuse pile and the toe of the plumes. Generally, water highly enriched with leachate was the sodium bicarbonate type, but in many samples, calcium and chloride were equally dominant or exceeded the sodium and bicarbonate concentrations. Toward the toe of both plumes, bicarbonate was greatly attenuated relative to

the other major ions. Bicarbonate concentration, which tends to change directly with pH, decreases as the pH falls below about 6 and would also be expected to decrease in the more acid ambient water. Attenuation of other ions toward the toe may take place by chemical reactions such as oxidation of ammonium to nitrate or by sorption on the hydrous iron oxide coating of the sand grains and clay components of the sediments. The proportion of sulfate appears to increase downgradient at both sides, possibly as a result of oxidation of an unknown, reduced form of sulfur, and at Babylon it is increased by local domestic waste contamination.

The character of leachate-enriched water is modified by enrichment from other sources, particularly at the Babylon site. The Stiff diagrams for the Babylon plumes (fig. 13) represent C-depth wells, but analyses of samples from wells at this depth outside the plume show that ambient water was enriched by septic-tank and cesspool waste and was not significantly different

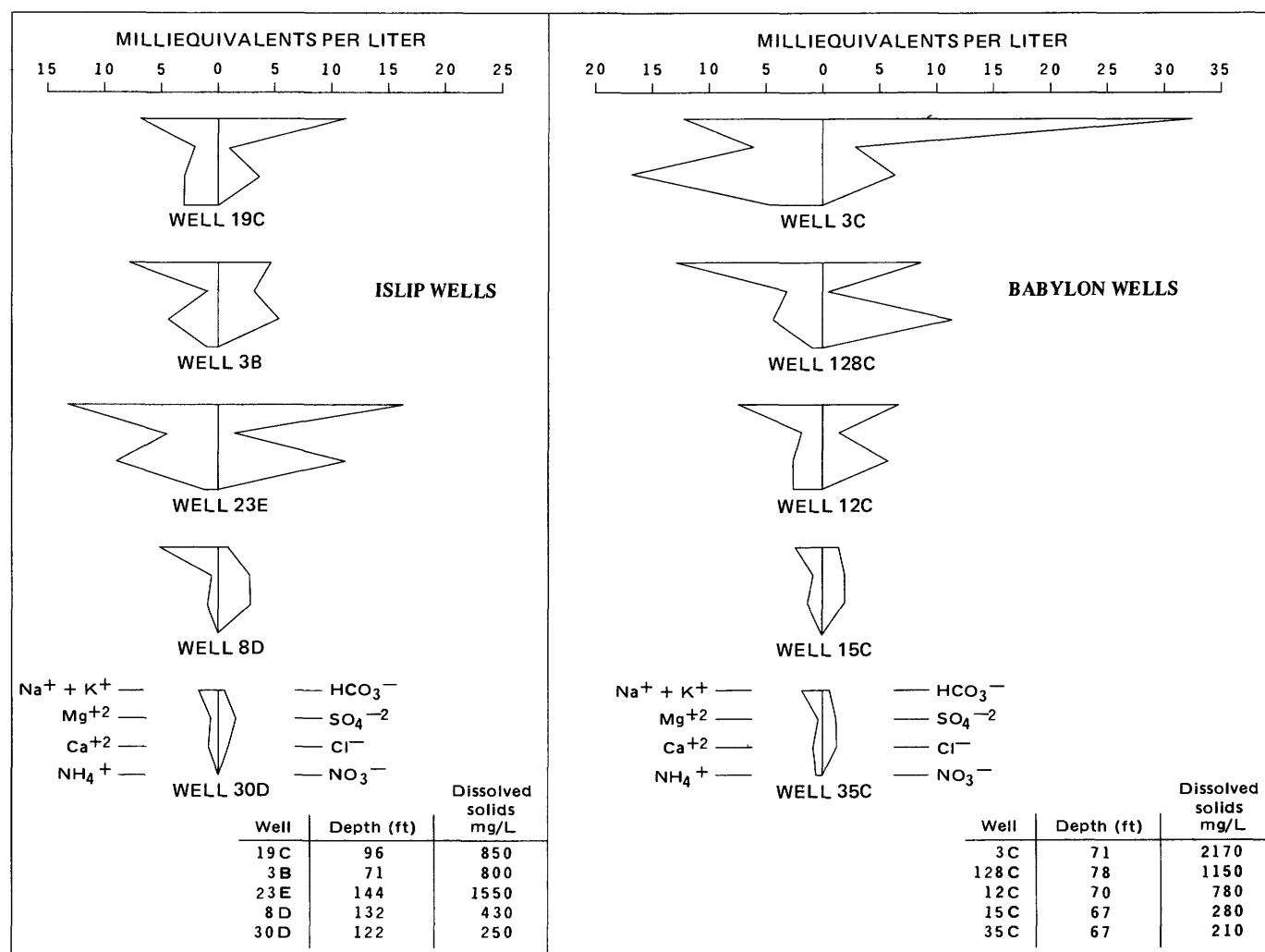


FIGURE 13.—Concentration of chemical constituents of water from selected wells in Babylon and Islip plumes.

from highly dispersed leachate-enriched water such as that found in well B35C-67 (fig. 13). Thus, some of the changes in the chemical character of water that take place downgradient from the Babylon landfill may result from enrichment by outside sources.

At the Islip site, however, ambient water at the depths of samples given in figure 13 differs significantly from leachate-enriched water. Changes in the chemical character of water in the plume (fig. 13) relate either to initial differences in plume-water quality (leachate entering the ground water) or to some attenuating chemical effect on the leachate-enriched water as it moves through the aquifer. Although chloride is the most obviously attenuated ion except for bicarbonate at the Islip site, the linear-correlation coefficient for the sodium-to-chloride ratio is 0.99 for all Islip plume-water analyses; this coefficient suggests that the mass of the chloride ion is conserved as the water moves through the aquifer. The chloride concentration in well I30D-120 (fig. 13) was low relative to that of the other ions, probably because of differences in the initial concentration of the leachate.

MINOR COMPONENTS

A suite of trace elements, including most of the common heavy metals, was included in the chemical analysis of ground water from the study sites. The elements, with their maximum concentrations and the percentage of samples in which they were detected, are listed in table 8.

IRON AND MANGANESE

The heavy metals that were found in highest concentrations and frequencies in the leachate plumes were iron and manganese. Their concentrations exceeded the U.S. Public Health Service (1962) recommended

TABLE 8.—Maximum concentration and occurrence of minor chemical constituents in the upper glacial aquifer in both study areas

Constituent	Maximum concentration (micrograms per liter)		Percentage of samples having concentration above detection level	
	All samples	Plume samples	All samples	Plume samples
Arsenic	11	7	32	47
Boron	2,500	2,500	95	93
Bromine	4.2	4.2	100	100
Cadmium	1	1	3	4
Cobalt	370	370	52	58
Chromium (VI)	10	10	10	7
Copper	100	50	34	41
Iron	440,000	440,000	99	98
Mercury	4	4	23	18
Manganese	190,000	190,000	81	88
Nickel	27	7	69	73
Lead	70	28	94	87
Selenium	38	38	83	81
Strontium	1,400	1,400	100	100
Zinc	220	180	78	80

¹Exceeds U.S. Public Health Service (1962) recommended drinking-water standards.

standards of 300 and 50 mg/L, respectively, throughout most of the plumes and the surrounding areas. Concentrations of iron as high as 440,000 mg/L at pH 6.9 (table 8) and of manganese as high as 190,000 mg/L at pH 6.3 were found in water from well B3C-86 at different times. The iron and manganese concentrations at this and other wells varied through an order of magnitude from time to time. During the study period at well B6C-78 (fig. 3), 3,000 ft south of the refuse pile, iron content ranged from 320 to 690 mg/L, and manganese content ranged from 80 to 600 mg/L. Although well B6C-78 was in highly leachate-enriched water (conductance ranged between 1,600 and 2,100 μ mho), iron and manganese concentrations were about equal to those in ambient water in the Babylon area. Because the concentration of these constituents throughout much of the plume was no greater than in ambient water, they apparently were reacting with the aquifer by sorption or were removed by precipitation with other unknown chemical species or bacteria in the aquifer.

The high concentrations of iron and manganese in ground water near the landfills suggest that these constituents are largely derived from the refuse. Additional iron may be added from the glacial outwash sediments, which are coated with hydrous oxides of iron that can contribute iron under the reducing conditions present in the plume water. The sand and gravel of the upper glacial aquifer, typically orange-brown from the hydrous oxide coating, was gray for a few hundred feet beyond the landfills where leachate enrichment was severe; this condition suggests that the iron in the formation has been stripped away by the leachate-laden water.

BORON

Boron was present in both plumes and reached a maximum concentration of 2,500 μ g/L (table 8). Because its concentrations were higher in plume water than in ambient ground water, it too may be used to define the plumes. It occurs as the highly soluble and conservative borate anion, which accounts for its relatively high concentration and frequency. Boron enters the ground water from septic-tank effluent; therefore, its concentration near the toe of the Babylon plume equals that of ambient water. Although most of the boron in leachate probably comes from laundry products, small amounts could also be contributed by refuse such as printing inks, ceramics, and rubber.

SELENIUM AND ARSENIC

These elements occur as the anions selenate and arsenate. Both species are soluble and, as anions, are

not adsorbed on materials in the aquifer. Selenium occurred in higher concentrations and was more prevalent than arsenic (table 8) because selenium is used in common products such as rubber and ink, whereas arsenic, which occurs chiefly as a contaminant in impure phosphorus and selenium salts, is used mainly in toxic products such as insecticides and rodenticides and therefore is not deposited in appreciable concentrations in the landfills studied.

OTHER HEAVY METALS

The heavy metals cadmium, chromium, cobalt, copper, mercury, nickel, lead, strontium, and zinc were all detected (table 8). Some of the cobalt and zinc was contributed by the landfill. Concentrations of the remaining metals were no higher in the plume than in ambient water. Cobalt was found in concentrations as high as 370 $\mu\text{g/L}$ in water from well B3C-86 and 21 $\mu\text{g/L}$ in water from well B1B-69. Elsewhere in leachate-rich water, cobalt was undetected or in very low concentrations. Zinc concentration was generally higher in leachate-enriched water (180 $\mu\text{g/L}$, well B3C-86) than in ambient water, but at some locations in ambient water, zinc concentrations approached those found in the plumes.

Lead was the most frequently found heavy metal; it was detected in 94 percent of the samples (table 8). The high percentage of samples containing lead may reflect the ubiquitous dispersion of lead into the atmosphere and soils by combustion of tetra-alkyl lead in fuels (Page and others, 1971; Chow and Earl, 1970).

The generally low concentrations of heavy metals in leachate can be attributed to their occurrence in refuse primarily in the elemental (metallic) state. In the elemental form, heavy metals are virtually insoluble in water and are not readily changed to a soluble state; therefore, water percolating through refuse is not likely to dissolve more than trace quantities. Moreover, heavy-metal ions have a greater affinity for exchange on soil and sediment colloids than protons or monovalent cations. Thus, heavy metals are likely to be adsorbed from solution by organic or clay colloids and, to a slight degree, by hydrous iron oxide coatings on the sediment. Chelates formed with organic acid and phenolic ligands produced in the decomposition process are probably the major transport mechanism of heavy metals through the landfill materials. Potential ligands are among compounds reported by Robertson, Toussaint, and Jorque (1974, p. 33, 34, and 37) in landfill leachate.

ORGANIC CARBON

Organic carbon analysis can be used to estimate the organic contribution in ground water from waste wa-

ter. For this purpose it is superior to BOD (biochemical oxygen demand) because it gives a truer measure of organic matter in solution (Malcolm and Leenheer, 1973). One disadvantage of this method is that it cannot be used to identify the specific organic compounds present; however, it is a good indicator of the amount of organic matter available for further analysis. Identification of individual organic compounds was not undertaken in this study.

The concentration of organic carbon at various points in the upper glacial aquifer in April 1972 is shown in figure 14. Ground water outside the plume contained as much as 5 mg/L organic carbon, probably as a result of septic-tank waste. Organic carbon concentrations as high as 2,250 mg/L were found in water from well B3C-86. Although other samples from the plume contained considerably less organic carbon, the data suggest that organic carbon concentrations greater than 5 mg/L are indicative of plume water. Except near the Babylon landfill, where organic carbon concentrations were relatively high, organic carbon concentrations in the plumes were less than 22 mg/L. The lower part of the plume at Islip, which contained the highest dissolved-solids concentration, was not sampled for organic carbon.

Some compounds in the highly contaminated water are volatile, as indicated by a strong odor. Much of the organic matter is of low molecular weight and consists of decomposition and metabolic products resulting from the intense microbial activity in the refuse (Robertson and others, 1974). Undoubtedly some of the organic carbon comes from soluble organic materials in the landfill, such as detergents, solvents, and petroleum products.

The rapid attenuation of organic carbon in the plume is not surprising. Bacteria in the plume decompose soluble organic materials and thereby remove organic carbon from the plume water, and other polar organic compounds (molecules having functional groups with positive or negative charges) may be adsorbed on the surfaces of minerals in the aquifer. Together, these phenomena effect a net removal of organic matter from the leachate-enriched ground water.

BABYLON PLUME

As described earlier in the section on the "Quality of Native and Ambient Ground Water," the plume of leachate-enriched water south of the Babylon landfill can be delineated where specific conductance is greater than 400 μmho (pl. 1). From 1947 or 1948 (when leachate was probably first generated by the landfill) to 1974, the plume advanced southward 10,000 ft (3,000 m) to Farmingdale Road (site B35, pl. 1). The already

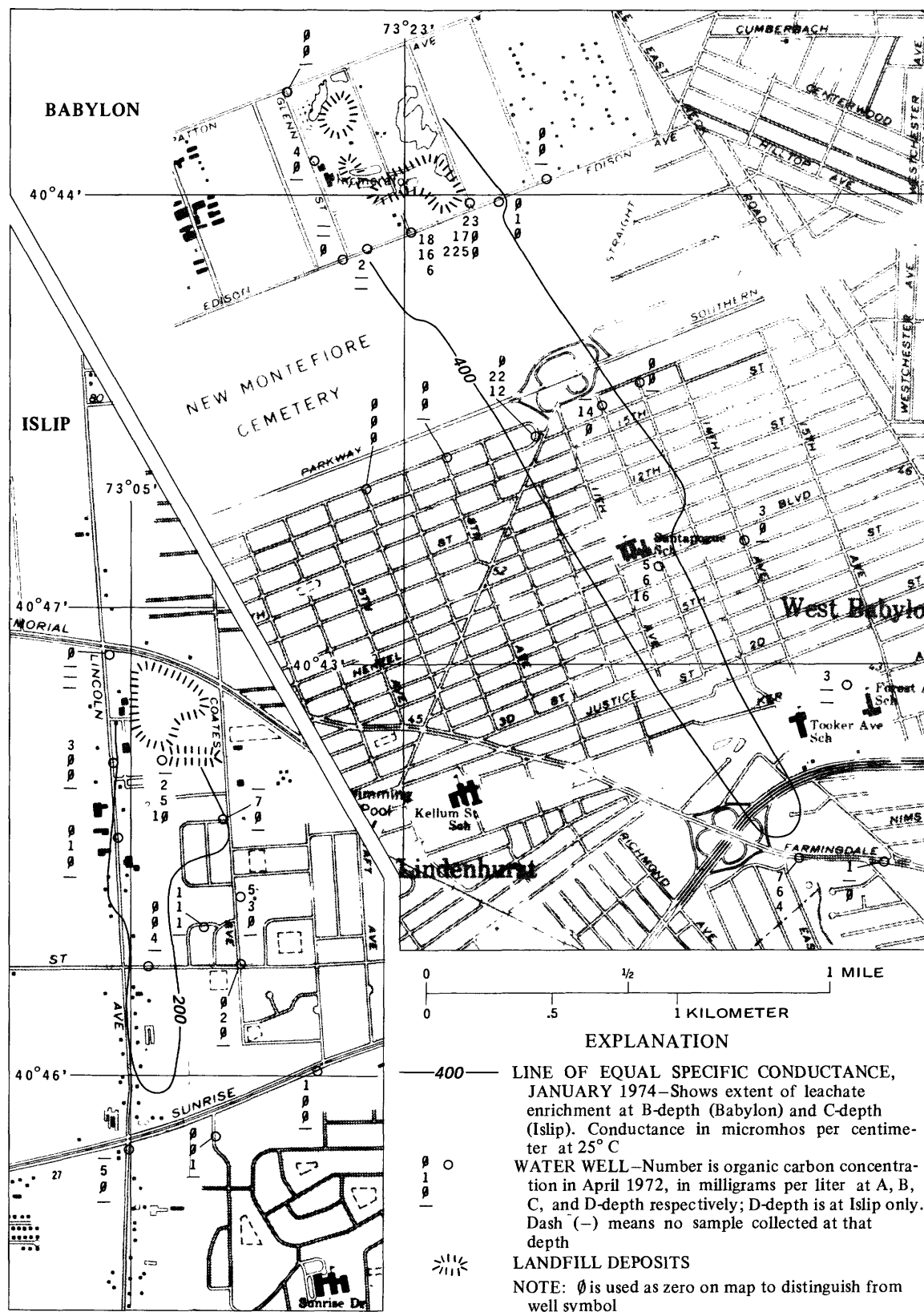


FIGURE 14.—Organic carbon concentration in upper glacial aquifer at selected sites in vicinity of Babylon and Islip landfills, April 1972.

degraded quality of ambient water in this region precludes tracing the leachate-enriched water at lower solute concentrations. Leachate-rich water sinks to the bottom of the aquifer beneath the landfill, and the concentration of leachate in the plume varies with depth and distance from the refuse pile (pl. 2A). Downgradient, the top of the plume was below the water table so that ground-water quality at shallow depths was not significantly affected. For example, water at well B128A-7 was of native quality because no septic-tank waste or other waste was in that area to alter water near the water table. At its terminus near the headwaters of Santapogue Creek, the top of the plume was at least 30 ft below the water table in 1972. The better quality of water above the plume was due to the infiltration of precipitation in the region south of the landfill.

The specific conductance of ground water at the toe of the plume (site B35) is shown in table 9. The increase in conductance in water from well B35A-25 since the summer of 1972 may be the result of sewer construction in the area south of site B35, during which time frequent dewatering near the water table from 1972 to 1974 may have disturbed ground-water flow paths sufficiently to bring water of higher conductivity from the deeper part of the aquifer up to the shallow level of the well screen in B35A-25. During the study period (1971-74), no significant increase in specific conductance was observed at well site 35. This apparent lack of increase reflects both the dispersion of the leachate-enriched water and the presence of domestic waste in the ground water, which together obscure the subtler changes in conductance during leachate dispersion. Thus, significant changes in specific conductance at the plume front would require a longer observation period to become apparent.

Near the refuse pile, the bottom of the plume was 1,900 ft wide (pl. 2A, section B-B'), and the top was a few hundred feet narrower, even though the west side of the landfill has been widened somewhat by a gravel-excavation pond that dilutes the leachate-enriched water. Conductance of ground water near the landfill indicated that leachate enrichment of water in the plume varies (pl. 1).

TABLE 9.—Specific conductance of ground-water samples at well site B35

[In $\mu\text{mho/cm}$ at 25°C]

Date	Well No.		
	B35A-25	B35B-46	B35C-67
04-07-72	190	360	350
10-26-72	380	380	410
06-06-73	370	380	400
12-26-73	370	410	340
07-29-74	420	315	450

At some locations, the concentrations of dissolved constituents changed over the period of study. Water near the bottom of the aquifer at well B8C-75 (pl.1) had a specific conductance of 140 μmho in February 1972 and 1,000 μmho in July 1974. Specific conductance of water near the top of the aquifer, at well 8A, decreased from 420 μmho to 230 μmho during the same period. High nitrate concentrations (as much as 8.8 mg/L as N) suggest contributions from septic-tank contamination in shallow ground water at this site. Water from well B8C-75 in July 1974 was unusual in that only the sodium chloride concentrations were high; although no source for the salt is known in the area, it is likely that it originates somewhere other than the landfill.

The irregular, stepped outline of the east side of the plume reflects the eastward growth of the landfill and the subsequent discharge of leachate east of the initial solid-waste pile (fig. 10 and pl. 1). The location of the lateral steps in the plume at C depth was obtained by extrapolation of the change in water quality with time. For example, the increase in chloride concentration in water from deep wells (table 6 and 10) indicates the plume's movement past sites 10 and 8 during the period of investigation, and this movement downgradient from sites 10 and 8 (especially B10C-78) can be attributed to the rate and direction of ground-water flow.

The chloride content (table 6) and other data (table 10) indicate that the plume lacked measurable lateral dispersion. Comparison of samples from the edge of the plume (table 10) with those from outside and the center of the plume indicates that the quality of water at the edge was either close to that of ambient water or that the characteristics of plume water cannot be distinguished at these well sites. Near its terminus, the Babylon plume was about 700 ft wide.

Throughout most of the plume, specific conductance ranged between 2,000 and 1,000 μmho but decreased abruptly at the edges. In the last 2,000 ft of the plume, specific conductance decreased gradually. Pockets of more concentrated leachate-enriched water apparently occur at various places in the plume; their shape and extent, as depicted by lines of equal conductance, are generalized (pl. 2A). Although the contours are not precise, they provide an indication of the degree of leachate enrichment. The lateral placement of the 400- μmho equal conductance line is probably accurate to within several hundred feet, for the lateral limits of the plume appear to be well defined by the ground-water-flow path.

Well B113D-107 (pl. 1) was installed to determine the quality of water in beds of silty, fine sand at the top of the Magothy aquifer. Although a 3-foot drop in head

TABLE 10.---Chemical analyses of ground water along lateral edge of plume at Babylon

[Concentrations in milligrams per liter]

Constituents and characteristics	Well No. and date of collection												
	B5B-46 1-26-72	B9B-46 1-31-72	B11B-46 7-22-74	B11C-88 7-22-74	B48BFW 5-7-73	B79BFW 5-5-73	B79BFW 7-26-74	B89BFW 7-16-74	B122B-37 7-29-74	B122BB-65 7-31-74	B123B-37 6-8-73	B124C-90 6-8-73	B124-90 7-26-74
Major constituents:													
Silica (SiO ₂)	8.6	9.1	8.4	12	7.9	10	8.4	8.2	8.2	6.8	8.8	15	14
Iron (Fe)	.25	.09	.14	1.2	1.3	1.5	.97	.18	.06	.27	.16	.17	1.3
Manganese (Mn)	.0	.0	.38	.04	1.1	.9	.95	.06	1.1	.29			.1
Calcium (Ca)	14	8	20	21	14	18	16	10	13	10	22	7.1	10
Magnesium (Mg)	3.6	6	4.8	9.9	5.3	3.2	6.8	4.7	4.8	5.5	4.6	2.3	4
Sodium (Na)	7.7	8	17	15	47	16	48	35	36	28	25	7.6	9.5
Potassium (K)	1.8	1.2	5	1.2	4.1	4.1	3.8	2.1	5.1	2.6	6.3	.8	.8
Bicarbonate (HCO ₃)	6	7	14	31	16	8	16	10	16	14	16	17	14
Carbonate (CO ₃)	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0
Alkalinity as CaCO ₃	.0	.0	11	25	13	7	13	8	13	11	13	14	11
Sulfate (SO ₄)	31	19	49	62	56	28	57	36	48	36	59	7.9	16
Chloride (Cl)	12	18	24	25	71	22	70	47	56	40	35	13	20
Nitrate (NO ₃) as N	4.3	3.6	6.2	5.2	1.9	8.1	1.6	2.7	3.2	4.1	6.8	2.0	1.5
Nitrogen, ammonia as N	.12	.10	.06	.04	2.0	1.3	1.8	.09	2.2	2.2	1.7	.1	.03
Nitrogen total as N	4.42	3.77				9.4					8.5	2.1	
Other characteristics:													
Dissolved solids (sum)	101	89	136	163	216	143	221	149	182	137	201	71	83
Hardness total	50	45	70	94	57	58	68	44	52	48	74	27	42
Specific conductance, μmho/cm at 25°C	164	146	270	320	420	233	440	330	380	310	318	100	142
pH	5.8	5.9	5.1	5.7	5.2	5.2	5.1	5.4	5.3	5.1	6.1	6.5	5.7

occurs across the Gardiners Clay at this location, indicating a downward gradient, characteristics of leachate were not found in water in the uppermost part of the Magothy. Analyses of water from this well, results of which are given in table 11, show that the dominant cation is calcium and the dominant anion is bicarbonate. Although similar ionic relationships can be found in leachate-enriched water, the concentrations in well B113D-107 were very low and virtually the same as, or only slightly greater than, in native water (table 2). Moreover, sodium and chloride concentrations, which are usually high in leachate-enriched water, were at about the level of native water in well B113D-107. Conductivities as much as twice that of native water in the upper glacial aquifer and the relatively high bicarbonate concentration may be characteristic of native ground water in this part of the Magothy aquifer. In any case, the results of analyses shown in table 11 do not indicate enrichment of the Magothy aquifer by landfill leachate.

TABLE 11.—Composition of water from well B113D-107

(Constituents in milligrams per liter unless otherwise noted)

Constituent	Date of collection	
	2-25-74	7-3-74
Major Constituents:		
Silica (SiO ₂)	20	17
Aluminum (Al), $\mu\text{g/L}$		80
Iron (Fe), $\mu\text{g/L}$		1,100
Manganese (Mn), $\mu\text{g/L}$		60
Calcium (Ca)	10	11
Magnesium (Mg)	3.9	4.0
Sodium (Na)	5.7	7.0
Potassium (K)	1.4	1.0
Bicarbonate (HCO ₃)	40	54
Alkalinity total as CaCO ₃	33	44
Sulfate (SO ₄)	8.0	2.4
Chloride (Cl)	12	9.4
Fluoride (F)	.0	.1
Bromide (Br)		.1
Nitrate (NO ₃) as N	.01	1.00
Nitrite (NO ₂) as N	.04	
Nitrogen, ammonia, total as N	.01	1.25
Nitrogen, total as N	.39	
Trace Elements:		
Arsenic (As), $\mu\text{g/L}$		2
Boron (B), $\mu\text{g/L}$		20
Copper (Cu), $\mu\text{g/L}$		0
Lead (Pb), $\mu\text{g/L}$		1
Selenium (Se), $\mu\text{g/L}$	1	3
Strontium (Sr), $\mu\text{g/L}$		90
Zinc (Zn), $\mu\text{g/L}$		100
Other Characteristics:		
Alkalinity total as CaCO ₃	33	44
Dissolved solids:		
Residue at 180°C	107	84
Calculated	81	80
Hardness as CaCO ₃	41	44
Noncarbonate hardness	8	0
Specific conductance, $\mu\text{mho/cm at } 25^\circ\text{C}$	112	130
pH	7.0	8.1

¹Data obtained by specific ion electrodes.

ISLIP PLUME

The plume of leachate-enriched water emanating from the landfill at Islip is illustrated on plates 2 and 3. Directly below the landfill, the plume permeated the entire aquifer, as at Babylon. The main body of the plume extended 5,000 ft downgradient from the original landfill. At the landfill the plume was 1,400 ft wide. An eastern lobe of the plume (pl. 1) reflects recent eastward extension of the landfill. Water from well I1C-95 indicated that leachate-enriched water had not yet penetrated the full thickness of the aquifer below the eastern part of the landfill, and that therefore, the bottom of the plume was probably not more than 70 ft below the water table (section *D-D'*, pl. 2*B*). The greatest concentration of dissolved solids was in the western part of the plume, near the bottom of the aquifer. From section *D-D'*, a tongue of the plume 500 ft wide extended 4,100 ft downgradient (south) at a depth of from 60 to 130 ft below the water table (section *E-E'*, (pl. 2*B*).

Leachate-enriched ground water flows in the direction of the hydraulic gradient (normal to the water-table contours, fig. 4) from beneath the landfill; therefore, the water of highest conductance (up to 1,100 μmho , AA-depth wells, pl. 3) west of the refuse deposition was probably contaminated from a source other than the landfill. Water from wells along the west end of section *D-D'* (pl. 2*B*) ranged from a completely sodium chloride character (I21E-146) to a sodium-calcium, chloride-sulfate character (wells I22AA-41, I22C-98). Sulfate, which is usually low in water highly enriched with leachate at Babylon, was the dominant anion in two of the wells in this locality. The relatively high sodium, chloride, and sulfate concentrations and the low iron and manganese concentrations (generally 0.3 mg/L) suggest that the wells at site 22 tapped water altered by a source other than the landfill. That source may be road salt stored in piles 1,230 ft upgradient from section *D-D'* or from an unknown discharge from the incinerator, also directly upgradient.

The bottom of the Islip plume was determined from water samples taken at site I1 in 1972 and site I19 and I8 in 1973. In 1972, dissolved-solids concentration of water from well I1C-99 was 33 mg/L (table 2). Unfortunately, the well was destroyed by the addition of more refuse in 1972, so no further sampling at this site was attempted. If the shape of the plume did not change significantly after 1972, the base of the plume is probably about as shown on section *D-D'* (pl. 2*B*).

Site I19 (section *C-C'*, pl. 2*B*) was installed in 1973. At well I19E-167, the conductance was 390 μmho in June 1973 and 410 μmho in January 1974; the conductance at I19F-206, screened 39 ft lower, was 55 μmho in October 1973 and in February 1974. Water levels

from site I19 wells, given in table 12, showed head decreases of as much as 0.2 ft between the E and F screen depths. The water levels indicated no significant upward or downward gradients above the E-depth screen, whereas below it, a downward vertical component of flow existed. Because leachate enrichment was not apparent at the lower screen setting, it is concluded that the beds of silty, fine sand between the two screens (see section on "Hydrogeology") formed a hydrologic boundary sufficient to retard downward passage of water.

Wells at site I8 did not fully penetrate the upper glacial aquifer. The conductance at screen depth E (section *E-E'*, pl. 2*B*) approached the limit for detection (200 μ mho); thus, the bottom of the plume was drawn at that depth about 150 ft below land surface.

The outline of the plume (pl. 3) shows that the sides do not extend beyond the streamline of ground-water flow from the extremities of the landfill. Evidence for this restriction on the width of the leachate-enriched zone and for its placement between 60 and 130 ft below the water table was obtained through wells shown in section *E-E'* (pl. 2*B*) and from well I27D-180 (pl. 3). Analyses of water from wells along this section are given in table 13. Water from well sites I11, I26, I31, and I32 showed no evidence of alteration specifically by landfill leachate. Well I8B-46, screened above the delineated leachate plume (pl. 2*B*), contained a sodium chloride-type water that was unusually high (for plume water) in nitrate (3.9 mg/L) and low in bicarbonate (7 mg/L). Water from well I8C-99 contained a sodium-type water, also high in chloride and bicarbonate, but low in nitrogen and with ammonia (0.52 mg/L) exceeding nitrate (0.34 mg/L) in one analysis. The deeper water seemed to be enriched by leachate, the shallower by domestic waste. The conductance of water from I8C-99 also exceeded 400 μ mho, which was unusually high for water enriched by domestic waste in this locality; thus, the top of the plume was placed between 46 and 99 ft, the depths of the two well screens.

Evidence for lateral restriction on the width of the plume was also shown by water analyses from wells I26D-130, I27D-130, I31C-87, and I33D-119 (table

13, pls. 2*B* and 3). These wells were screened either outside the plume (I26D-130, I27D-130, I31C-87) or on its periphery (I33D-119). Water from wells I26D-130 and I27D-130 was of virtually native character, and that from well I31C-87 had a somewhat elevated sulfate concentration and conductance, which could indicate slight domestic-waste enrichment. The analysis from well I33D-119 indicates dilute (dispersed) leachate-enriched water near the edge of the plume; this water was enriched in sodium, calcium, sulfate, chloride, and bicarbonate, and it was similar in chemical character to that in the plume at site 8. The analyses indicated that lateral spreading of the leachate-enriched water could not be much greater than that shown on pl. 3 and pl. 2*B* and that the edges of the plume were directly downgradient from the landfill area, as was the situation at Babylon landfill.

Movement of plume water near its periphery was noted in several wells during the course of this study. For example, conductance of water at well I6C-99 was 210 μ mho in November 1972 and 520 μ mho in January 1974; conductance of water from well I5C-98 was 47 μ mho in January 1972, 324 μ mho April 1972, and 750 μ mho in August 1974. Water quality near the toe of the plume, however, did not change significantly during the period of measurement. The analyses given in table 14 for water from wells I30C-82 and I30-120 (pl. 3) indicate that in areas where plume water was highly dispersed and diluted, more than 1 year of observation would be necessary to reveal significant changes in water quality.

GROUND-WATER TEMPERATURE

Biological activity in the organic material of the landfill elevates temperatures in leachate. In other studies, temperatures as high as 74°C (Stutzenberger and others, 1970) and 68°C (Fungaroli and Steiner, 1971) were found in composting refuse subject to aerobic digestion; however, Stutzenberger, Kaufman, and Lassin (1970) found that after an initial high period of several weeks, temperatures fell to 40°–55°C, presumably after conditions had become anaerobic. Fungaroli and Steiner (1971) found temperatures in the range of 15°–25°C under similar anaerobic conditions. Warm leachate flowing from the composting refuse would naturally raise the temperature of ground water with which it came into contact.

Ground-water temperatures near the Babylon and Islip landfills are indicated in figure 15. Heat generated in the landfill and carried out by the leachate causes a rise in ground-water temperatures beneath and downgradient from the landfills. In the area adjacent to the south sides of the Babylon landfill, the temperature rise was about 7°C; at Islip it was about 16°C.

TABLE 12.—Water levels at site I19

Well	Screen setting below water table (ft)	Altitude of water table, in feet above mean sea level		
		1-17-74	3-14-74	3-19-74
I 19B-61	36-40	36.11	36.12	36.20
I 19C-96	72-76	36.13	36.13	36.21
I 19E-167	142-146	36.16	36.07	36.18
I 19F-206	183-186	35.95	35.91	36.04
Level F minus level E		0.21	0.16	0.14

TABLE 13.—*Chemical Analyses of Water from Selected Wells in and near Isip Plume*
 [Concentrations in milligrams per liter unless otherwise noted]

Constituents and Characteristics	Well No. and date of sample collection											
	18B-46 ¹	18C-99	18D-122	18E-147	126B-61	126D-130	127D-130	131C-87	133D-119			
	2-23-72	5-16-73	7-19-73	8-13-74	6-29-73	8-13-74	8-13-74	6-26-73	8-12-74	6-28-73	10-18-73	8-13-74
Major Constituents:												
Silica (SiO ₂)	9.4	9.6	12	13	17	14	16	9.1	12	11	14	12
Iron (Fe), $\mu\text{g/L}$	80	150	15,000	8,300	30	100	20	95	40	0	220	130
Manganese (Mn), $\mu\text{g/L}$	460	650	140	150	16	30	10	530	10	10	40	50
Calcium (Ca)	8.2	12	6.3	6.5	16	12	10	10	3.0	2.9	15	15
Magnesium (Mg)	2.6	2.4	3.7	4.2	6.1	7.3	3.9	2.6	1.4	1.4	8.6	7.6
Sodium (Na)	18	29	68	90	115	170	10	15	4.6	9.0	49	60
Potassium (K)	5.7	6.4	10	15	2.8	3	.8	2.5	.5	.4	4.4	3.3
Bicarbonate (HCO ₃)	7.0	7.0	31	46	48	59	20	5.0	10	10	28	28
Alkalinity as CaCO ₃	6.0	6.0	25	38	39	48	16	4.0	8	8	23	23
Sulfate (SO ₄)	12	9.5	80	100	130	150	13	10	.2	1.8	79	98
Chloride (Cl)	33	52	56	83	100	150	18	31	10	9.0	57	53
Fluoride (F)	0	.2	.10	.2	.20	0	.1	.20	.1	.2	.2	0
Bromide (Br), $\mu\text{g/L}$	4.1	3.9	.34	1.9	0	2.3	.17	3.4	.04	3.4	.2	.4
Nitrate (NO ₃) as N	.17	.39	.52	.76	0	1.1	.23	.05	.02	0	.23	.71
Nitrogen, ammonia as N		83	250	290	190	430	70	30	40	0	390	2.07
Boron (B), $\mu\text{g/L}$				160		230	80					
Strontium (Sr), $\mu\text{g/L}$												
Other Characteristics:												
Dissolved solids:												
Residue at 180°C	164	277	352	428	428	555	172	133	44	63	264	180
Calculated	111	142	254	345	411	539	128	98	37	56	241	299
Total hardness	31	40	31	34	65	60	75	36	13	13	73	265
Noncarbonate hardness	25	34	6.0	0	26	12	49	32	5	5.0	50	46
Specific conductance								178	50	78	423	430
($\mu\text{mho at } 25^\circ\text{C}$)	195	258	446	565	709	850	231	5.8	6.4	5.9	5.5	5.3
pH	6.2	5.8	6.5	5.8	6.2	5.5	6.8					

¹See table 2 for analysis of well 18A-25.

²Analyses by selective ion electrodes.

TABLE 14.—*Chemical analyses of water from wells I30C-82 and I30D-120*

[Concentrations in milligrams per liter unless otherwise noted]

Constituents and Characteristics	Well No. and date of collection			
	I30C-82		I30D-120	
	6-29-73	8-9-74	10-19-73	8-9-74
Major Constituents:				
Silica (SiO ₂)	8.2	6.8	16	16
Iron (Fe), mg/L	12	30	110	110
Manganese (Mn), mg/L	40	50	50	20
Calcium (Ca)	8.1	9.6	15	14
Magnesium (Mg)	3.1	4.6	6.9	6.2
Sodium (Na)	28	24	34	39
Potassium (K)	12	10	1.8	1.1
Bicarbonate (HCO ₃)	13	7	36	31
Alkalinity as CaCO ₃	11	6	30	25
Sulfate (SO ₄)	64	63	70	74
Chloride (Cl)	29	19	29	29
Fluoride (F)	.2	0	.2	1
Bromide (Br)		.28		.36
Nitrate (NO ₃) as N	3.2	2.8	.47	.45
Nitrogen, ammonia as N	4.1	2.6	.07	0
Boron (B), mg/L	250	310	160	240
Strontium (Sr), mg/L		40		130
Other Characteristics:				
Dissolved solids:				
Residue at 180°C	166	176	191	223
Calculated	179	141	250	196
Hardness as CaCO ₃	33	43	66	61
Noncarbonate hardness	22	37	36	35
Specific conductance (μmho/cm at 25°C)	294	295	325	340
pH	6.0	5.1	6.5	5.8

This difference could be due to well location rather than any major difference in the temperature of the landfills. When the highest temperature, 27.7°C, was recorded at site I19, refuse was being deposited within a few feet of the well site. If this refuse were undergoing rapid decomposition, warm leachate would be entering the ground-water near the well site, whereas at Babylon, where the wells were not as close to the landfill, some of the heat may have dissipated before reaching them. Temperatures were measured with an underwater thermistor at 2-foot intervals from the water table down to 170 ft (fig. 16). The thermal gradient was most pronounced at site I19. As water moves away from the landfill, the vertical temperature gradient decreases, as indicated by the difference between temperature profile for wells B113D-107 and I19F-206 (fig. 16).

Heat from the leachate was dissipated within less than 0.4 mi downgradient from the Babylon landfill, but at Islip some heat was retained downgradient as far as site I8 (fig. 15), 0.5 mi from the landfill. At this site, the temperature of water from well I8C-99 was 5°C warmer than that from wells both above and below. Nearly the same temperature difference was noted on several occasions when the wells were pumped.

In December and January 1973, when all but a few of the measurements in figure 15 were made, water near the water table was 1–2°C higher than water deeper in the aquifer. The water table was declining from the previous May high owing to lack of recharge (fig. 2),

but the cooler winter precipitation apparently had not yet reached the water table.

DENSITY

Water with a high dissolved-solids concentration of several thousand milligrams per liter is more dense than water of the same temperature with a low dissolved-solids concentration. Thus the water of greater density will move downward through the lighter ambient water by gravity. Below the landfills, the temperature of ground water is sufficiently elevated to lower its density to approximately that of the cooler ambient ground water. It is possible, therefore, for warm water of high solute content to have the same density as cool (10°C) native water. The effect of temperature on the density of sodium chloride solutions of varying strength is shown as a curve in figure 17. Sodium chloride solutions have properties of density similar to those of sodium bicarbonate solutions, sea water, and probably also leachate. For example, a 10,000-μg/L solution of sodium chloride at 40°C is slightly less dense than native ground water at 10°C (fig. 17). Solutions having higher concentrations of sodium chloride (or leachate, presumably) are more dense than native water at 40°C or lower. It is conceivable that warm leachate-enriched water with a dissolved-solids concentration as high as 30,000 mg/L may be more dense than cool, native ground water by as much as two parts in a hundred.

De Laguna (1966, p. E11) found that water exceeding the density of pure water by 3.5 parts per thousand moved downward through pure water contained in a beaker of sand. Thus, however slight the difference between density of leachate and ambient ground water is, it may be enough to cause downward movement of leachate through the aquifer if warm, leachate-rich water is in excess of about 10,000 mg/L dissolved solids.

VISCOSITY

Temperature has a much greater effect on the viscosity, or resistance to flow, of water than it does on its density. The influence of viscosity on ground-water flow is shown by its effect on hydraulic conductivity:

$$K = \frac{kg}{m/p} \quad (5)$$

where K = hydraulic conductivity,

k = intrinsic permeability, a property of the aquifer framework,

g = acceleration due to gravity

m = absolute viscosity, and

p = density

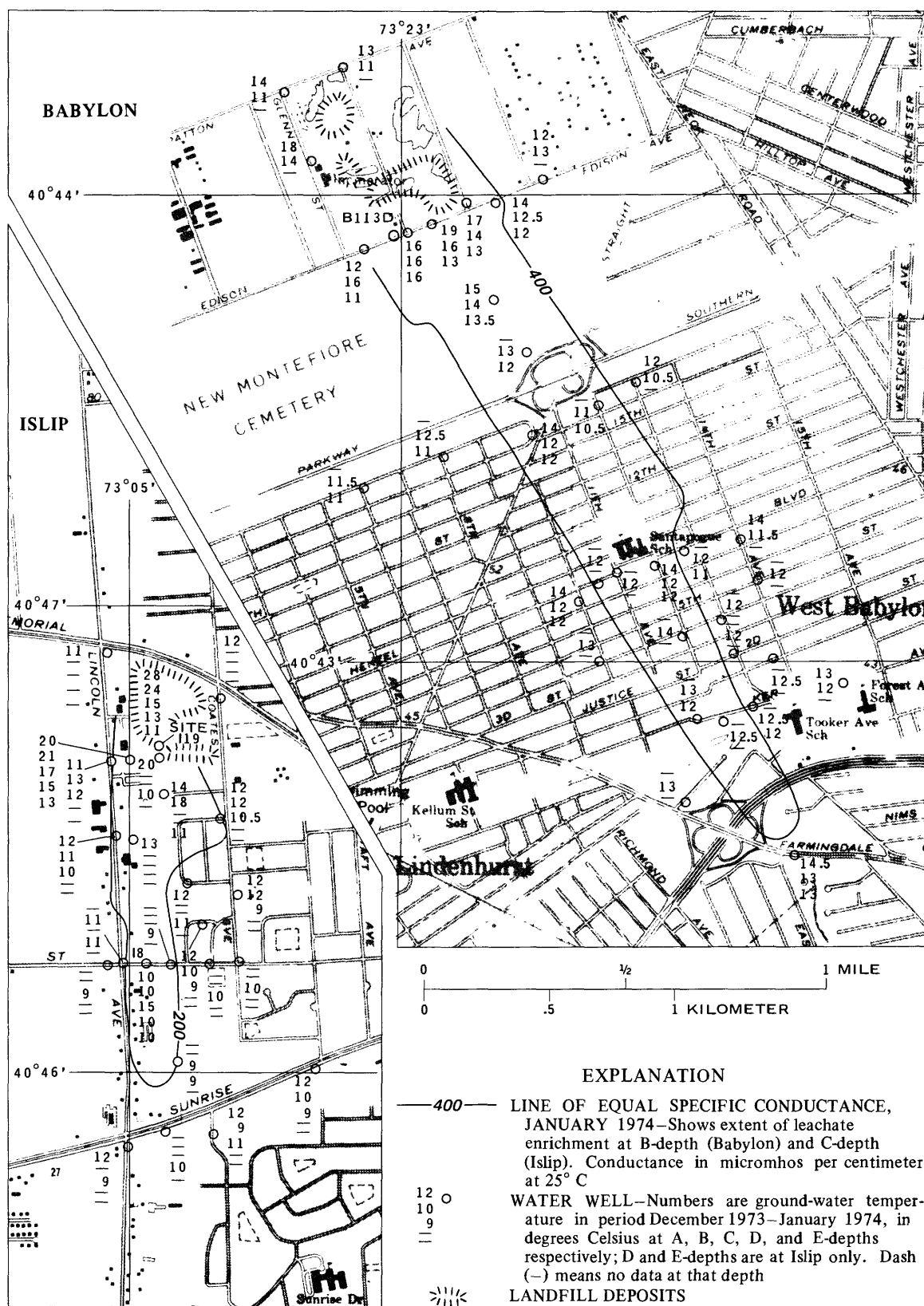


FIGURE 15.—Temperature of water in the upper glacial aquifer in vicinity of Babylon and Islip landfills.

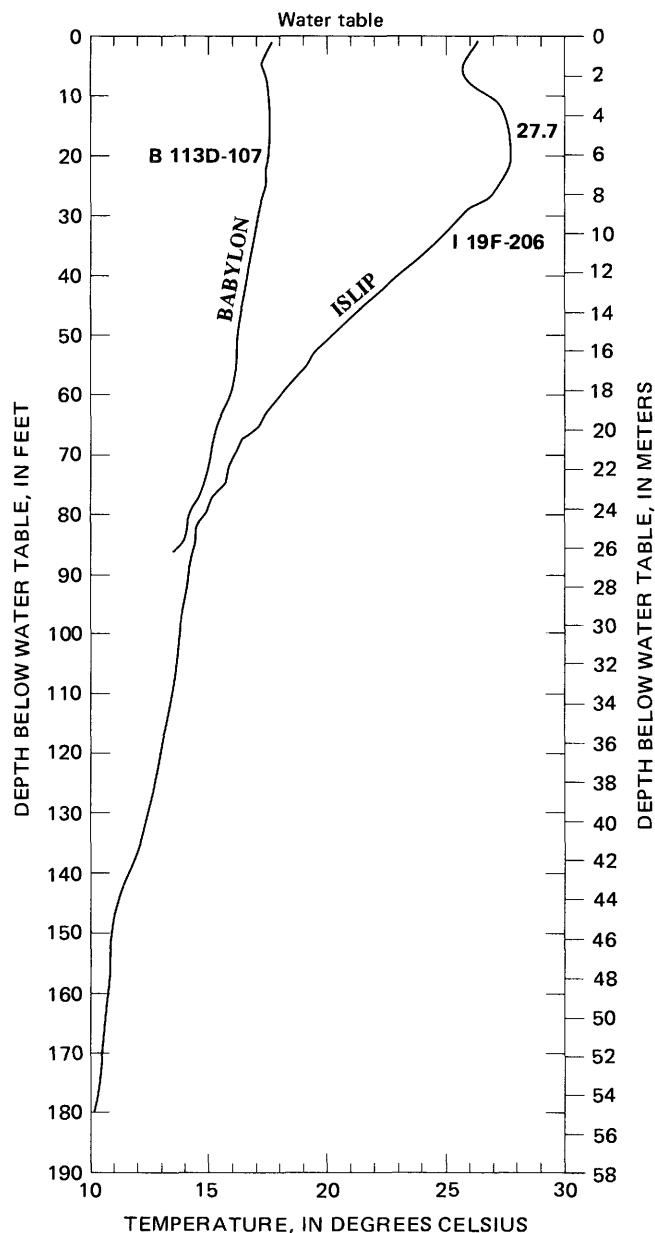


FIGURE 16.—Temperature profiles in August 1974 from wells in vicinity of Babylon and Islip landfills. Location of wells is shown on plate 2.

The intrinsic permeability and acceleration of gravity are considered constant in this discussion. The quotient of m/p , the kinematic viscosity, is the actual measure of resistance to flow. This is graphed in figure 18 for solutions of sodium chloride and for pure water in the temperature range 10°–40°C. The kinematic viscosity of water decreases by about half from 10°–40°C, but the addition of a strong electrolyte (in this case sodium chloride) to the water stabilized the kinematic viscosity at 0.01 stokes. Kinematic viscosity differs by as little as 4 percent between a sodium chloride solution with 6,000 mg/L at 10°C and one with

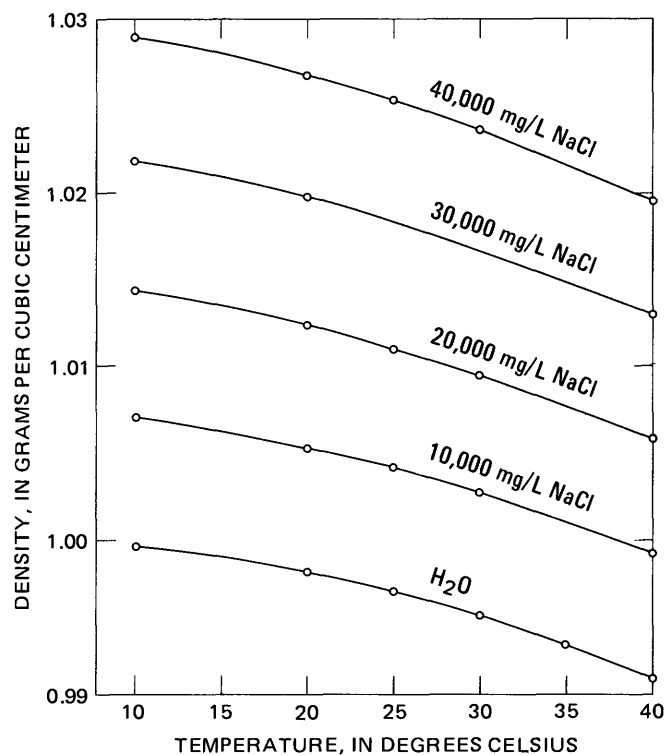


FIGURE 17.—Change in density with temperature for sodium chloride (NaCl) solutions of different strengths and for water. Data from Washburn (1926, v. III).

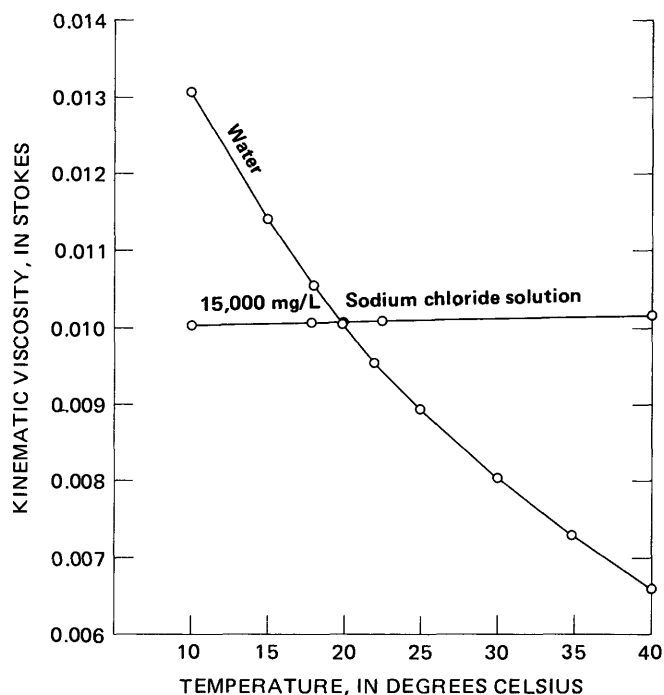


FIGURE 18.—Variation of kinematic viscosity with temperature for sodium chloride solution and for water. Values of kinematic viscosity were calculated from density and viscosity data in Washburn (1926, v. III) and Weast (1972).

30,000 mg/L at 40°C. The change in kinematic viscosity for a solution of sodium bicarbonate is twice that of sodium chloride, but the effect is still small.

Because a strong electrolyte tends to stabilize the kinematic viscosity through a wide range of temperatures, it was concluded that when the solute content of water is high, as in leachate-rich water, the hydraulic conductivity does not significantly vary from cold to warm regions of the aquifer, but the density difference between ambient and leachate-rich water may be sufficient to cause the leachate to sink to the bottom of the aquifer.

FLOW AND DISPERSION OF LEACHATE PLUMES

Although dispersion of the plumes into ambient ground water of somewhat similar chemical quality makes the exact extent of leachate-enrichment uncertain, the practical limits of detection—400 and 200 μ mho at the Babylon and Islip sites respectively—probably define the lateral extent of the plume accurately. The characteristic of dispersion, explained further on, describes the distribution of leachate enrichment near the edges of the plume, and dispersion was found to be greater in the primary flow direction than laterally.

Regardless of what dispersion patterns and conductances are used to delineate the plumes, water-quality data indicate that the plume lengths do not correspond with those predicted from estimated ground-water velocity and the lengths of time since the beginning of landfill operation. The plume at Babylon is twice the length of the Islip plume; yet the Islip landfill is reported to be 14 years older than the Babylon landfill. Although the rate of ground-water flow is probably somewhat greater at Babylon, the water-quality data seem to suggest that the Islip landfill did not produce detectable quantities of leachate until some time after its inception. The following discussion analyzes the relation of leachate dispersion to ground-water flow patterns beneath and downgradient from the landfills.

FLOW

The quantity and composition of leachate from landfills are determined by many factors, the major ones of which are thickness, areal extent, and age of the landfill; cover and compaction of refuse; and climatic regime. Composition of the refuse and biological activity within it also influence the composition of the leachate. With the exception of unusual industrial wastes, the kinds of refuse that go into municipal landfills are fairly similar from place to place. Because no unusual refuse is deposited at the landfills studied, the determining factors in leachate composition at Babylon and

Islip are probably the same as those at other landfills in similar environments.

To calculate initial concentrations of leachate and predict its movement, it is necessary to quantify the infiltration and dilution factors. Most of the liquid fraction of the leachate comes from infiltration of precipitation on the landfill. Because the landfill surfaces are probably more permeable than the surrounding terrain, rate of recharge through them is probably greater than the annual average of 23 in. estimated for Long Island. Although it is possible that recharge through the landfill could be as much as 40 in. per year, an average rate of 30 in. annually seems reasonable. The 25 acres of the Babylon landfill would then capture about 2.7×10^6 ft³, and the 17 acres of the Islip landfill would capture 1.9×10^6 ft³ of precipitation annually and would also yield this amount of leachate. The estimated volume of ground water flowing annually below the landfills is 48×10^6 ft³ at Babylon and from $(25-46) \times 10^6$ ft³ at Islip.¹ If the leachate were dispersed evenly throughout the aquifer beneath the landfills, the ratio of leachate to ground water would be 1:17 at Babylon and between 1:13 and 1:25 at Islip.

If recharge through the landfills were greater than in surrounding areas, a ground-water mound might form beneath the landfills that would cause a vertical component of flow greater there than elsewhere, and this could also result in downward movement of the leachate beneath the landfills. Despite the likelihood of greater recharge through the landfills than through surrounding areas, no mounding of the water table beneath the landfill was observed. Careful measurements of the water table in wells at various depths near the landfill indicated no shift in the water-table contours that could be attributed to increased recharge through the landfills (figs. 3 and 4). If mounding occurred, it would cause an observable lateral flow of ground water away from the landfill. Because no lateral spreading of leachate-enriched water from the landfill was observed, greater recharge through the landfills and the consequent mounding are probably not significant causes of downward movement of leachate at either landfill.

The shape of the plumes beyond the landfill area is determined by the regional flow of ground water, which is horizontal and southward—normal to the water-table contours in figures 3 and 4. Infiltration of precipitation outside the leachate's point of entry into the aquifer forms a body of water over the plume that is not enriched by leachate. This added water does not depress the plume, but only overlies it, and therefore it is not a

¹Annual volume of underflow = hydraulic conductivity (in ft/d) \times gradient \times area (in ft²) \times 365 days. Thus,
At Babylon: $500 \text{ ft/d} \times 0.0021 \times (1,800 \text{ ft} \times 70 \text{ ft}) \times 365 \text{ d} = 48 \times 10^6 \text{ ft}^3$, and
At Islip: $(190 \text{ to } 360 \text{ ft/d}) \times 0.0016 \times (1,300 \text{ ft} \times 170 \text{ ft}) \times 365 \text{ d} = (25 \text{ to } 46) \times 10^6 \text{ ft}^3$

factor in the plume's presence at the bottom of the aquifer just south of the landfill. Probably the main reason that leachate enrichment was highest near the bottom of the aquifer beneath the landfill is that the heavier, leachate-rich water sinks by gravity as it moves out of the refuse, and not because it is displaced downward by freshwater from the surface. Infiltration of precipitation downgradient from the landfill is sufficient, though, to account for a few feet of freshwater over the plume at well B128A-7 (pl. 24, section A-A') about 2,000 ft south of the Babylon landfill.

The leachate-enriched water differs from the ambient ground water mainly in its high dissolved-solids content (or greater density) and its higher temperature. It was pointed out in the section on "Viscosity" that the kinematic viscosity of leachate is not greatly different from that of ambient ground water, regardless of the temperature differences, because its electrolyte content decreases the effect of temperature; hence the flow characteristics of the fluid are probably much the same as that of ambient water. It is likely, though, that in spite of its elevated temperature, the leachate is somewhat more dense than ambient water and would thus have a downward driving force or vertical gradient in addition to that provided by whatever mounding was created by the increased recharge through the landfill. Because only a small amount of mixing with ambient water would be needed to equalize the density and thereby prevent the downward movement of leachate through the aquifer, it is concluded that relatively little mixing does, in fact, take place before the leachate reaches the bottom of the aquifer.

Leachate could not flow directly from the landfill base to the bottom of the aquifer without interfering with the continuity of the regional flow of ground water. Because the water-table contours give no evidence that the regional flow is disturbed by the inflow of leachate at the landfill, it is assumed that the leachate flows out of the landfill as pulsations of high-density fluid after periods of recharge, and moves as pockets, or slugs, diagonally downward through the regional ground-water flow lines toward the bottom of the aquifer. Their vertical movement is more rapid than their horizontal flow, otherwise they would not reach the bottom of the aquifer beneath the landfill but would be strung out downgradient from it. As the pockets of high-density leachate move downward, they alter the surrounding ground water and produce plumes of leachate-enriched water throughout their vertical course (pls. 1 and 3). It is assumed that the pockets of leachate tend to retain their original density, otherwise there would be insufficient density gradient to carry them downward, and the most leachate-rich water would not be near the bottom of the aquifer.

Some mixing does take place in the upper part of the aquifer, though, which accounts for the leachate enrichment in the middle and upper parts of the aquifer.

The movement of leachate is depicted in figure 19. Ground water underflowing the landfills mixes with pockets of leachate sinking through the aquifer so that plumes of diluted leachate are formed. Concentration of leachate is high near the bottom of the aquifer, but it probably varies greatly from time to time. After an initial formation period, the concentration of leachate probably does not fall below a relatively high value near the downgradient side of the landfill (C_0 in fig. 19). The time necessary for the pockets of leachate to reach the bottom of the aquifer is less than that necessary for ground water to flow from the upgradient to the downgradient edge of the landfill.

An average velocity of ground water moving through the aquifer can be estimated from the relationship

$$u = \frac{Ki}{n}$$

where u = average ground-water velocity,

K = hydraulic conductivity,

i = hydraulic gradient, and

n = porosity.

With an assumed effective porosity of 0.25 at both sites, hydraulic conductivity of 470 ft/d at Babylon and 190–360 ft/d at Islip, and a gradient of 0.0021 at Babylon and 0.0016 at Islip (see section on "Hydrogeology"), the average ground-water velocity is 4.0 ft/d at Babylon and from 1.2–2.3 ft/d at Islip. At these average velocities, the time of travel for ground water from the upgradient edge to the downgradient edge of a refuse pile 1,000 ft across would be 0.7 year at Babylon and up to 1.4 years at Islip (using $u = 2$ ft/d at Islip.)

After the plume is formed, the concentration established in the region of initial concentration is assumed to remain more or less constant. The distance from the source to the dispersed fronts of plumes observed in this study was great enough that, generally, fluctuations in dissolved-solids concentration in the C_0 region would tend to balance out, but pockets of more highly enriched ground water did occur. Chloride concentrations discussed in the later section, "Dispersion Model," are based on this concept of leachate-plume formation.

As a result of the pulsating nature of leachate entering the ground water, the concentration of dissolved constituents has a wide range near the head of the plume. Observed values of dissolved-solids concentration at the sampling points in the main part of the plume at Babylon and Islip are given in table 15. The

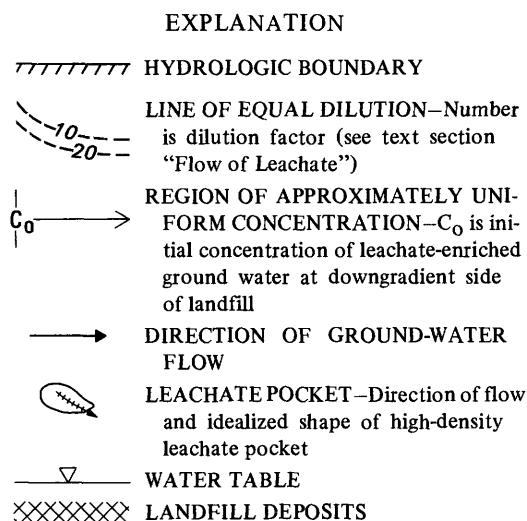
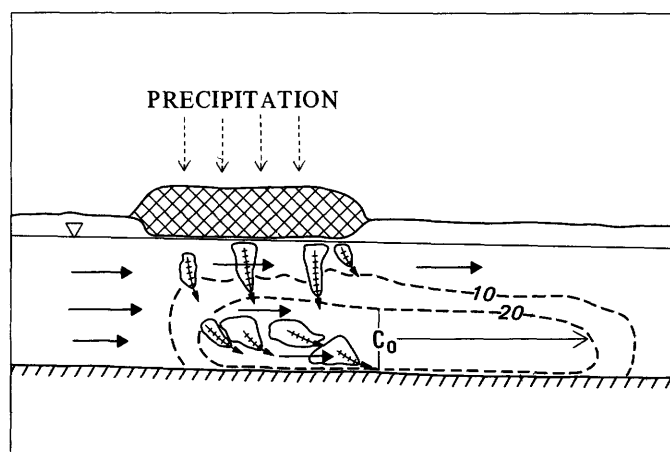


FIGURE 19.—Leachate movement and dispersion in ground water beneath a landfill.

data show that the concentration at one sampling horizon (well B1B-50) increased as much as fivefold in a 1-year period. Dissolved-solids concentrations ranging between about 400 and 3,000 mg/L at Babylon and 200 and 1,500 mg/L at Islip are indicated. Dissolved-solids concentrations of from 10,000 to 20,000 mg/L are reported to have been observed after 1 to 2 years of refuse emplacement in controlled studies of landfill materials (Fungaroli and Steiner, 1971; Rovers and Farquhar, 1973). If these high concentrations are typical for landfills and are necessary to produce the density difference between ambient ground water and leachate, the concentrations in table 14 suggest that leachate is diluted considerably between the landfill and the observation wells used in this study.

Hydrologically, the two landfill sites are similar in many respects. Hydraulic conductivity at both sites was found to range from 190 to 500 ft/d, and porosities are probably quite similar also. Sand samples from

TABLE 15.—Dissolved-solids concentration (sum) in selected wells in plume near Babylon and Islip landfills

[In milligrams per liter]

Well	1972	1973	1974
Babylon Wells			
B1A-28	1,260	1,170	1,140
B1B-50	642	3,070	1,340
B1BB-69	702	2,530	2,480
B3A-30	289	911	
B3B-51	743	1,400	696
B3C-86	2,780	2,170	1,440
B113BFW		1,110	1,020
B114A-22		417	
B114B-42		613	692
B114CC-92		796	
1972-74 Average			1,278
Islip Wells			
I3AA-45	530	948	1,160
I3B-71	801		
I3D-120	563	782	
I19B-61		640	575
I19C-96		850	280
I19E-167		217	425
I20AA-36		484	674
I20BB-77		711	854
I21AA-42		1,050	664
I21E-146		1,510	1,346
1972-74 Average			753

both sites are virtually the same, and, at both sites, the lithology from top to bottom of the aquifer is fairly homogeneous. Furthermore, the shape of the plumes and their lack of lateral dispersion indicate similarity of the aquifer at the two sites. The regular spacing and uniformity of the water-table contours south of the landfills (fig. 3 and 4) suggest that the upper glacial aquifer has a fairly uniform hydraulic conductivity because regional variations in hydraulic conductivity would produce irregularly spaced potentiometric contours. The sinking of leachate-enriched ground water directly below the landfill indicates that the ratio of vertical to horizontal hydraulic conductivity is not relatively great; for example, Getzen (1975) and Franke and Getzen (1976), in a verified electric-analog model of ground-water flow on Long Island, obtained a ratio of 10:1 for the upper glacial aquifer, but a ratio of 40:1 for the underlying Magothy aquifer.

DISPERSION MODEL

The fate of solute flowing into a saturated, permeable medium in relation to the dispersion phenomena is discussed comprehensively by Bear (1972). A mathematical treatment of dispersion phenomena is given in Ogata and Banks (1961) for a simple hydrologic system with one-directional flow. The ratio of observed concentration at the front (dispersed) part of the plume to the initial concentration is related according to the equation:

$$\frac{C}{C_0} = \frac{1}{2} \operatorname{erfc} \frac{x - ut}{2\sqrt{Dt}} \quad (6)$$

where C = concentration of tracer at observation point,

C_0 = initial concentration of tracer at source,

D = coefficient of dispersion,

erfc = complementary error function ($1 - \operatorname{erf}$)

erf = error function,

t = time since tracer was released,

u = average ground-water velocity, and

x = distance from source of tracer to observed concentration (C).

The coefficient of dispersion, as determined in laboratory studies for ideal conditions, is usually on the order of 1 ft²/d (Ogata and Banks, 1961). Few determinations of the dispersion coefficient for actual field conditions have been made, but the field coefficient has been found to be several orders of magnitude higher than the laboratory coefficient in at least some cases (Bredhoeft and Pinder, 1973; Pinder, 1973).

Assumptions of the dispersion equation are that (1) the contaminant is a vertical, planar source at $x = 0$ that began discharging into the full thickness of the aquifer at the initial time, (2) the initial concentration of the contaminant is known, (3) contaminant concentration at $x = 0$ has remained constant through time, (4) the aquifer is homogeneous and isotropic, and (5) contaminated water has the same flow characteristics as ambient water. Although the hydrologic situation at Babylon does not fully meet these conditions, the similarity was considered adequate for application of the dispersion equation to the plume in the upper glacial aquifer.

The initial concentration, C_0 , must be estimated within reasonable limits. Chloride concentrations for the deep, north part of the plume were as high as 1,000 mg/L (fig. 9); however, values greater than 300 mg/L occurred at scattered locations and seem to be the result of chloride slugs of greater than average concentration resulting from either the inhomogeneity of the refuse, the addition of new refuse, or undetermined factors. Accordingly, initial leachate concentrations of between 200 and 300 mg/L were used.

With this range in initial concentration, the dispersed front of the plume extends from the 200- or 300-mg/L line to the 50-mg/L line of equal chloride concentration and beyond (fig. 10). Chloride concentration, given in table 5, was used as a tracer to evaluate dispersion. Because the bottom part of the plume seems to have the greatest concentration of leachate, it is assumed that the chloride concentration near the bottom more nearly approaches the one-dimensional model described above (eq. 6). Therefore, chloride data from the

deeper part of the plume were used. The chloride-concentration values given in table 6 for wells B12C-70, B15C-77, and B35C-67 were decreased because waste from domestic sources in the surrounding area adds chloride to the overall concentration. The data suggest (fig. 10) that background chloride content contributed by cesspool sources was about 20 mg/L at well B12C-70, 30 mg/L at B15C-77, and 40 mg/L at B35C-67. Chloride concentrations for 1973 were available for all three wells, and, accordingly, the values given in table 8 for 1973 were reduced by these amounts and were then used to calculate the dispersion coefficient. If 4 years are allowed for formation of the leachate and its entry into ground water beneath the landfill, traveltime for the Babylon plume from the south side of landfill pile 1 (pl. 1) to its 1973 position is 22 years (1951-73).

The dispersion equation (eq. 6) relates the spread of the dispersed front [$(x-ut) > 0$, $(x-ut) < 0$] to the strength in the region of initial concentration (C_0). The equation states that where the initial concentration is reduced by 50 percent, $x = ut$ or the distance to the observation point divided by the traveltime is equal to the average ground-water velocity.

If this landfill-leachate situation can effectively be modeled by equation 6, it should be possible to verify the ground-water velocity calculated from the hydraulic conductivity and, thereby, have a better estimate of the regional hydraulic conductivity. On the other hand, if the hydraulic conductivity estimated from the dispersion equation (eq. 6) is not close to that estimated from well-test data, then either or both methods may have a faulty premise.

From trials of a range of initial concentration between 200 and 300 mg/L and the following field data, the dispersion coefficient was calculated (table 16). For this range in initial concentration, the region of initial concentration extends from the initial landfill (pile 1) to a point just north of site B12 (fig. 10) where the dispersed front begins.

The concentration gradient shown by the contours on the map in figure 10 falls between 6,000 and 10,000 ft from the initial landfill. Accordingly, the concentration range of 200 to 300 mg/L (C_0), distances from the initial refuse pile to the corresponding locations of $0.5 C_0$ (ut from fig. 10), and adjusted chloride concentration from wells B12C-70, B15C-77, and B35C-67 (C from table 6), were used to determine the coefficient of dispersion.

With these initial concentrations, the dispersion coefficient ranges between 350 and 50 ft²/d (table 16). The dispersion coefficients are most similar at wells B12C-70 and B15C-77 at an initial chloride concentration of 200 mg/L. Thus, an average dispersion coefficient of about 60 ft²/d is indicated. Higher values of the

TABLE 16.—*Values of dispersion coefficient for plume at Babylon after 22 years (8,000 days) of traveltime*

Well	Distance from source to well (ft)	Chloride concentration (mg/L)	Concentration adjusted (mg/L)	Dispersion coefficient (ft ² /d)		
				300 mg/L	Initial concentration 250 mg/L	200 mg/L
B12C-70 -----	6,200	200	180	350	190	70
B15C-77 -----	8,400	67	37	120	80	50
B35C-67 -----	10,300	45	5	170	140	120
Traveltime (<i>ut</i>) for corresponding initial concentration -----				6,800	7,200	7,600

dispersion coefficient determined by the chloride concentration from well B35C-67 probably indicate that the chloride value used for this site is too high, even though reduced by 40 mg/L, because of the addition of domestic sewage. The dispersion coefficient for well B35C-67 coincides with those of the northerly wells only if the effective chloride concentration is reduced to about 1 mg/L. In a nearby area, a dispersion coefficient equivalent to 100 ft²/d was used (Pinder, 1973, p. 1665), which is reasonably close to the 60 ft²/d estimated above.

At the rate of movement determined by the dispersion model, the 200-mg/L chloride concentration should reach Sunrise Highway (site 35) by the year 2010 and be near the shoreline by the year 2040. Depending on the depth to which ground-water inflow to streams affects the ground-water gradient around Santapogue Creek, the leachate-enriched water may or may not flow into Santapogue Creek. Pluhowski and Kantrowitz (1964, p. 48-50) found that seepage to streams had no effect on regional flow below about 5 ft along Champlin Creek, whose hydrologic relations are similar to those of Santapogue Creek, 8 mi to the west. Thus, because the leachate-enriched water is generally 5 ft or more below the water table, the plume may not affect the quality of water in Santapogue Creek.

As a tracer moves through a porous medium, the lateral, or transverse, dispersion is less than longitudinal dispersion and is possibly in the same order of magnitude as ionic diffusion in sandy materials (Ogata and Banks, 1964, p. G9). At least part of the lateral dispersion of leachate-enriched water has been obscured by a gradual widening of the plume at its source. Leachate produced when the landfill was young is probably now (1974) in the vicinity of well B15, where the plume is 700 ft wide, as opposed to only a few hundred feet at the source. Thus, a few hundred feet of lateral dispersion may have taken place. However, the west sides of the Babylon and Islip plumes extend directly downgradient from the west sides of the landfills, which have remained fixed as the landfills grew eastward (pls. 1 and 3). Water analyses (tables 10 and 13) do not show measurable amounts of leachate-

enriched water beyond about 200 ft west of the boundaries shown on plates 1 and 3.

COMPARISON OF PLUMES

The average velocity of ground water (*u*) in the region is determined from the dispersion model by the time of travel (*ut*) for the nonsorbing component, divided by the time for the component to flow from the source to a location where the concentration is 50 percent of the initial concentration. At the Babylon site this is

$$\frac{ut}{t} = \frac{7,600}{8,000}, \text{ or about } 1 \text{ ft/d}$$

This velocity is exceeded fourfold by the average ground-water velocity of 4 ft/d estimated for Babylon from the hydraulic conductivity and gradient. (See section on "Hydrogeology.") This range in velocities casts doubt on the validity of one or the other method of determining ground-water velocity.

Data are insufficient to provide, independently, a dispersion coefficient at the Islip site. The range in ground-water velocity at Islip, based on the hydraulic conductivities obtained from wells in the area (see section on 'Hydrogeology') is from 1.2-2.3 ft/d. If we assume an initial chloride concentration of 200 mg/L, the 50-percent concentration of 100 mg/L occurs at well 18D-122, 3,400 ft south of the landfill. The calculated time of arrival of this concentration at this site is from 1,500-2,800 days, or 4-8 years after initial leachate formation—which is 28-32 years short of the time of arrival estimated from the age of the landfill (36 years), even if 4 years are allowed for the formation and accumulation of leachate below the landfill. Even when water quality at the most distant part of the plume (5,000 ft) is used as a tracer, the time needed to form the entire Islip plume, as measured in 1973, is only about 11 years, or about a third that expected from the age of the landfill. In order for the present length of the plume to conform to the assumed time of initial leachate production, the hydraulic conductivity

would have to be one-third less than the value found from the analysis of partially penetrating wells.

It seems reasonable to assume that, in a region such as Long Island, where precipitation produces leachate from landfills, leachate would enter the ground-water system within about 2 years after the landfill began if the refuse were in contact with the water table and that the landfill would continue to yield leachate throughout its period of operation and beyond. Although the amount and concentration of leachate produced is related to the size and thickness of the landfill, the length of the plume is determined by the amount of time that leachate was being discharged and by the ground-water velocity, not by the size of the landfill. If the size of the landfill were to increase with time, the head of the plume would become wider but the plume's rate of advance should not increase.

Neither the Babylon nor the Islip landfill's pattern of growth is known in detail, but a general concept can be gained from aerial photographs. Table 17 gives the acreage occupied by the landfills at various times, as estimated from aerial photographs and from recent maps. The data indicate that the Babylon landfill has grown rapidly in comparison to that at Islip.

The ground-water-velocity estimates given by the dispersion model are clearly not in accord with those obtained from independent aquifer data. If the ground-water velocity of 4 ft/d calculated from the hydraulic conductivity at Babylon is nearly correct, it suggests that the determination of ground-water velocity on the basis of 22 years of plume movement (at 1 ft/d) may be in error and that a plume was not produced until many years after the landfill began. However, the accuracy of ground-water velocity is difficult to evaluate. It may be that the conditions assumed for the dispersion model are not matched by actual field conditions and also may not be appropriate for evaluation of the effect of landfills on the ground water. The dispersion-model analysis suggests that the landfills may not have yielded leachate until several years after they were started. Although the Babylon landfill grew rapidly during the 1950's as the town's population was expanding, the landfill still covered only about 1 acre in 1953 (table 14), 6 years after it began. Ground-water

velocities determined at both landfills from hydraulic-conductivity data indicate that the plumes are both only 5–8 years old.

It was learned in this study that the time of travel for leachate plumes from landfills is difficult or impossible to predict without data on the time of entrance of leachate into the aquifer. Unless the public-supply wells from which the hydraulic conductivity was estimated are in locations where the hydraulic conductivity is unusually high, or the plumes occupy regions of particularly low hydraulic conductivity, or the landfills did not yield leachate until recently, the discrepancy between leachate-plume movement and ground-water velocities determined both from well data and the dispersion model at Babylon and Islip cannot be explained with the data at hand.

SUMMARY AND DISCUSSION

The two plumes of leachate-enriched ground water have been determined by specific conductance to extend at present (1974) 10,000 ft and 5,000 ft, respectively, from the initial landfill areas in the towns of Babylon and Islip. The highest major cationic concentration measured in the plumes was 860 mg/L Na^+ , 565 mg/L Ca^{+2} , 100 mg/L Mg^{+2} , and 110 mg/L K^+ . The highest major anionic concentration measured was 2,700 mg/L HCO_3^- and 900 mg/L Cl^- . Sulfate concentrations as low as 6 mg/L and as high as 290 mg/L, were found in leachate-rich solutions near the landfills. The low sulfate concentration probably results from the prevalent reducing conditions there. Most nitrogen in leachate-enriched water occurs as ammonium and was found to be as high as 90 mg/L. Nitrate did not exceed 10 mg/L, or 40 percent of the total nitrogen, in the plume. Iron concentrations as high as 440 mg/L and manganese as high as 190 mg/L were found near the landfills, but these concentrations decreased within about 3,000 ft of the landfills. Boron, common in the upper glacial aquifer in the areas studied, was found in 95 percent of all samples and in 95 percent of all leachate-enriched water samples. It occurs in concentrations as high as 2.5 mg/L in leachate-enriched water near the landfills, but values above 0.5 mg/L are indicative of plume water. Nickel, lead, selenium, strontium, and zinc were also found in most leachate-enriched water samples tested. Only selenium, at 38 $\mu\text{g/L}$, exceeded the U.S. Public Health Service (1962) recommended drinking-water standards. As much as 2,250 mg/L organic carbon was found in one water sample near the Babylon landfill, but most samples contained less than 20 mg/L organic carbon. Leachate-enriched ground water was found to be as

TABLE 17.—*Estimated acreage of Babylon and Islip landfills from 1938 to 1973*

Year	Babylon	Islip
1938	0	0.7
1947	0	1.5
1953	1.1	
1961		5.2
1969		10.2
1972	23	14
1973	25	17

much as 18°C warmer than ambient ground water near the landfills, but this effect was dissipated within a short distance of landfills.

The plumes are confined to the upper glacial aquifer. Downward movement of the leachate-enriched water is restricted by a clay layer (the Gardiners Clay) at Babylon and by silty sand at Islip. The upper glacial aquifer is 74 ft thick at Babylon and 170 ft thick at Islip but the plume at Islip is not significantly thicker than at Babylon. The plumes are overlain by freshwater downgradient from the landfills. At Babylon, the plume occupies almost the full thickness of the aquifer, and it has higher concentrations of dissolved material for a greater distance downgradient than the plume at Islip.

The distribution of ions and particulate matter in plume water at the Babylon and Islip landfills results from several processes within the landfills and the ground-water system. Among these are (1) recharge from precipitation at the landfill, (2) sorption, desorption, and biological activity in the landfills, (3) mixing of leachate and incoming ground water, (4) sorption of dissolved and particulate matter in the aquifer, (5) changes in physical properties of the water, which influence its hydraulic characteristics, and (6) dispersion of the contaminants. The sinking of leachate-enriched ground water and the lack of lateral dispersion in ground water downgradient from the landfills studied may be typical for landfill leachate plumes in hydrogeologic conditions similar to those at Babylon and Islip; namely, (1) the landfill is essentially contiguous with the water table in a highly permeable aquifer that has relatively uniform vertical hydraulic conductivity, (2) recharge through the landfill is rapid, and (3) ground-water velocity below and around the landfill is relatively uniform. The chemical quality of leachate-enriched water is improved by dilution and by sorption of suspended organic material on the aquifer framework. Slower ground-water movement than anticipated and possibly delayed formation of leachate in the landfills limits the spread of leachate in ground water. The plumes are not only shorter than predicted from two independent determinations of ground-water movement, they also have no observable lateral dispersion, even though longitudinal dispersion is extensive.

The movement of leachate to the bottom of the aquifer may be typical of aquifers of high hydraulic conductivity. A zone of aeration between the bottom of the landfill and the water table would probably significantly alter the conditions surrounding the movement of leachate into the ground water and would significantly change the chemistry and characteristics of flow of the leachate before it joined the ground water. Such a condition might result in leachate enrichment

only near the water table. Although no evidence of faster recharge through the landfill was obtained, downward movement of leachate through the aquifer may result partly from recharge being greater through the landfill than through the surrounding area and partly from density differences between leachate and ground water. Impedance of recharge by covers of poorly permeable materials on the landfills might significantly alter the production of leachate, reduce ground-water alteration significantly, and restrict leachate to the top part of the aquifer.

Data presented in this report show the extent to which the quality of ground water has been changed by landfill leachate at two sites on Long Island. The change is serious near the landfill. As the water moves away from the landfill, dilution and sorption of the leachate-rich water reduce the severity of the change, but the size of the leachate-enriched region will increase for many years after the accumulation of refuse ceases. Leachate formation could be reduced by covering the landfill, but little can be done to reduce leachate enrichment of the ground water once it has begun. It would seem advisable to cover or seal new and old landfills to minimize the production of leachate so that little or none can enter the ground water.

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