CHEMICAL QUALITY OF LANDFILL LEACHATE IN TREATMENT PONDS AND MIGRATION OF LEACHATE IN THE SURFICIAL AQUIFER, PINELLAS COUNTY, FLORIDA

By Mario Fernandez, Jr., and G. L. Barr

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ABBREVIATIONS AND CONVERSION FACTORS

Factors for converting inch-pound units to International System (SI) of Units and abbreviation of units

Multiply	By	To obtain
inch (in)	25.40	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
acre	4,047	square meter (m^2)
square foot (ft^2)	0.0929	square meter (m^2)
cubic foot (ft^3)	0.02832	cubic meter (m^3)
gallon (gal)	3.785	liter (L)
gallon per minute (gal/min)	0.06309	liter per second (L/s)
pound (lb)	0.4536	kilogram (kg)
ton, short	0.9072	metric ton (t)
degree Fahrenheit (°F)	(°F - 32)/1.8	degree Celsius (°C)
micromho per centimeter at 25° Celsius (umho/cm at 25°C)	1.000	microsiemens per centimeter at 25° Celsius (uS/cm at 25°C)

* * * * * * * * * * * * * *

National Geodetic Vertical Datum of 1929 (NGVD of 1929): A geodetic datum derived from a general adjustment of the first-order level nets of both the United States and Canada, formerly called mean sea level. NGVD of 1929 is referred to as sea level in the text of this report.

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ABSTRACT

Pinellas County, in west-central Florida, utilizes mechanical aeration to treat landfill leachate. The leachate-treatment and disposal site encompasses about 8 acres within the 220 acres of the county's Bridgeway Acres landfill. The site has a shallow water table and is subject to inundation during tidal flooding and major storms.

Fresh leachate is pumped from V-shaped trenches at pumping rates that range from 150 to 500 gallons per minute for an average of about 3.8 hours per day. The leachate is aerated for about 2 days in a lined basin, then discharged to a stabilization pond where it is permitted to infiltrate into the surficial aquifer. The rate of leachate movement from the pond is dependent on the water levels in the pond and ground water. If the pond and ground-water levels were to remain high, as observed for January 24, 1980, water moving through the surficial aquifer from the pond could be expected to take about 620 days to reach within 20 feet of the east boundary of the site.

Two chemical constituents, ammonia nitrogen and potassium, were used as indicators of leachate migration in the shallow aquifer. No apparent nitrification occurred within the stabilization pond. What probably occurred was deamination or ammonification. Concentrations of the indicator constituents in the pond were dependent on water levels in the operating trench and rainfall. Leachate had migrated about 75 to 80 feet along the upper 5 feet of the aquifer during the period of study. Vertical migration was about 4 feet beneath the bottom of the pond into the aquifer.

INTRODUCTION

The Bridgeway Acres landfill (formerly the Pinellas County landfill) disposes of about 800 tons of solid waste (refuse) per day utilizing the trench method (Sorg and Hickman, 1970) combined with the high-rise method (Florida Division of Health, 1972). The trench method, also called "cut and cover," is a method whereby solid waste is deposited in V-shaped or flat-bottomed excavations. The excavated earth is subsequently used as daily cover material. In areas where the water table is near land surface, such as at Bridgeway Acres, ground water flows into a trench while it is being filled with refuse, making compaction difficult in addition to creating odor problems. Dewatering of the trenches is required before and during deposition of solid waste. Disposal of this water that may contain leachate must be accomplished in an environmentally safe manner.

The Pinellas County Department of Solid Waste Management developed a method whereby trench water or raw leachate from dewatering operations is pumped into a lined aeration basin and the leachate is mechanically aerated for about 2 days (Barr and Fernandez, 1981, p. 7). The treated leachate is then discharged by gravity into an unlined stabilization pond for final treatment and eventual disposal by infiltration into the surficial aquifer.

The U.S. Geological Survey, in cooperation with Pinellas County, undertook this investigation to determine water-quality changes that might result from aeration and ponding of leachate and to determine the rate of movement of the leachate in the shallow aquifer. The investigation began in January 1978 and was completed in September 1981.

Purpose and Scope

The purpose of this investigation was to provide analysis and supporting data to evaluate: (1) the extent and rate of leachate movement in the aquifer and (2) the chemical quality and phytoplankton and bacterial content of water within the stabilization pond. Aspects investigated included changes in the physical, chemical, and biological characteristics of the ponds and tracing of the leachate migration in the ground-water system beneath and adjacent to the ponds with ammonia nitrogen and potassium. Water-level measurements were made to define the direction and use in calculating rates of ground-water movement. Samples of raw and treated leachate were collected periodically for chemical and microbiological analysis to determine the efficacy of treatment. Samples of pond water and ground water beneath and near the ponds were also collected periodically for analysis of selected constituents to determine physical or chemical changes that occurred in order to identify and delineate the extent of migration of treated leachate from the pond through the aquifer. Samples of pond water for biological analysis were collected twice during the investigation to compare changes in numbers and types of phytoplankton that occurred due to increased treated leachate loading of the pond.

Acknowledgments

This report was prepared as part of the cooperative program of water-resources investigations between Pinellas County and the U.S. Geological Survey. Special appreciation is extended to Donald Acenbrack, Director of Solid Waste Management, Pinellas County, for his assistance and support.

STUDY SETTING

Location and Description of Study Area

The leachate-treatment and disposal site is within the Bridgeway Acres landfill, about 2 miles southwest of Old Tampa Bay and about 7 miles north of St. Petersburg (fig. 1). The landfill was formerly part of the Pinellas County landfill (Fernandez, 1982, p. 11). Land-surface altitudes range from 7 to 9 feet above sea level. Part of the site is subject to tidal flooding (fig. 1).

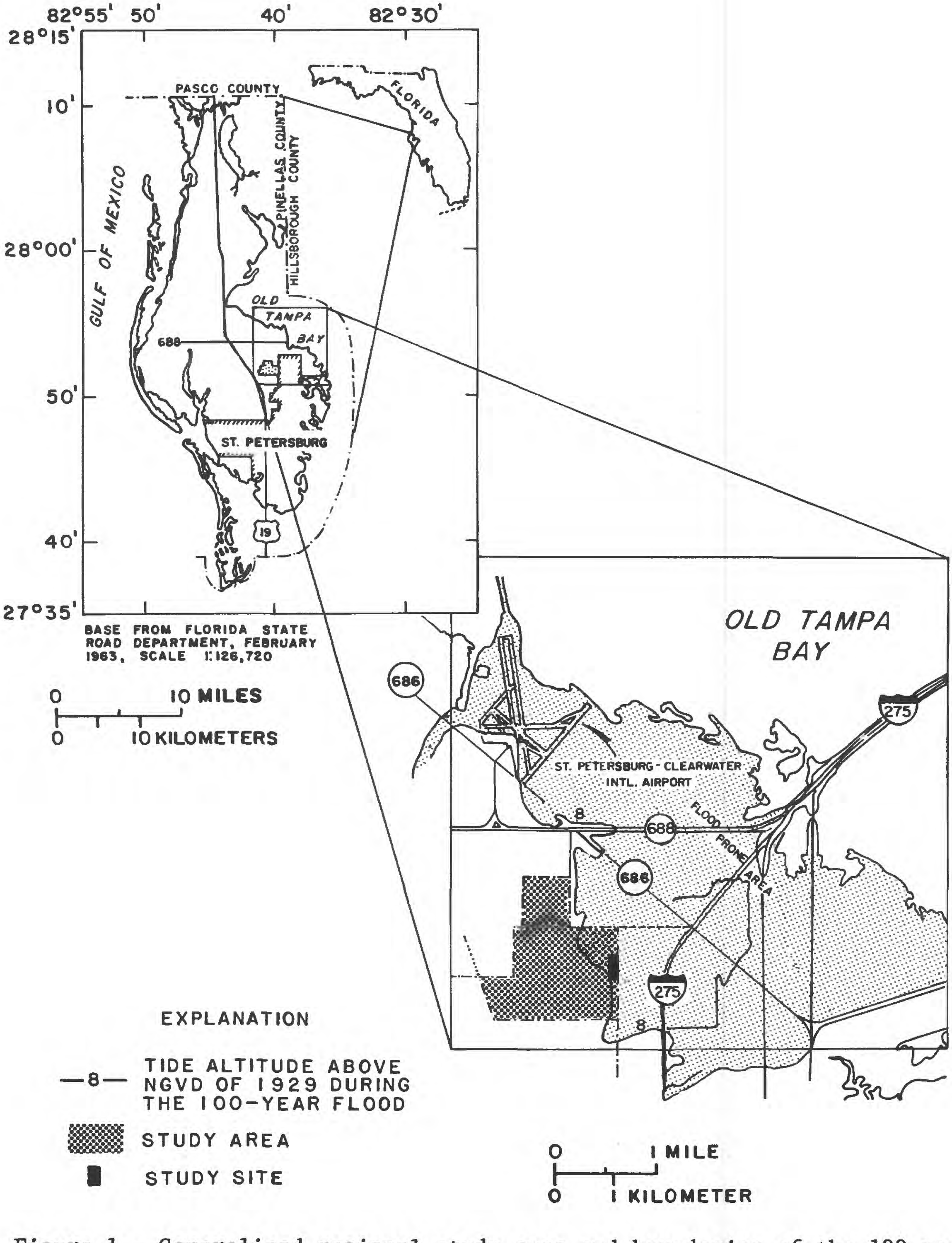

Figure 1.--Generalized regional study area and boundaries of the 100-year tidal flood.

The tidal-flood altitude (8 feet) is the altitude that is expected to occur on an average of once every 100 years. The flood-prone area shown in figure 1 has a 1 percent chance of being inundated during any year (U.S. Geological Survey, 1973a; 1973b).

The study area has little topographic relief and is covered with palmetto and scrub pine. Soils, which are poorly developed, are of the Elred soil series. This soil series is characterized by poorly drained, sandy soils that were formed in thick beds of sandy and loamy marine deposits (U.S. Department of Agriculture, 1972, p. 10). The soil series occurs in areas characterized by a near-surface water table.

Pinellas County has a subtropical climate that is characterized by warm, humid summers with frequent thundershowers and by mild, dry winters with occasional showers associated with cold fronts. The average yearly rainfall, based on the period 1941-80, is 52.48 inches (National Oceanic and Atmospheric Administration, 1941-1980) and for the period December 1979 through December 1980 was 61 inches. Some rainfall occurs during each month of the year, but about 70 percent occurs between June and October. The greatest monthly rainfall that occurred during the period of study was 23.2 inches that occurred in May 1979.

Heavy rains flood the area because of poor natural drainage. For example, during June 22-24, 1974, about 20 inches of rain fell in the county. Extensive flooding of the landfill site and surrounding area was reported by landfill operators.

The average annual temperature for the area is about 73.5°F. Average monthly temperatures range from 61°F in January to 82°F in August.

The Bridgeway Acres landfill is in an area (fig. 2) of mixed land use comprised of light industry, private homes, and waste-disposal sites (one sludge-disposal and three landfill sites). The sludge-disposal site and the Toytown landfill are operated by the city of St. Petersburg. A private landfill is located at the northwest side of the Bridgeway Acres landfill. Information on the waste-disposal sites, except for the private landfill, have been reported by Fernandez (1979a; 1979b; 1982) and Hutchinson and Stewart (1978).

Pinellas County is presently (1982) in the process of building a resource-recovery plant on the Bridgeway Acres landfill site. The plant will burn an estimated 2,000 tons of refuse daily thereby reducing the volume of refuse by 95 percent and producing electricity as a by-product (Donald Acenbrack, Pinellas County, oral commun., 1980).

Design and Operation of the Landfill Leachate-Treatment Site

Landfill operations began in November 1977 and have been continuous until the present (1982). Solid waste from municipal collection systems and private collectors is deposited daily, Monday through Saturday. The operation handles about 800 tons of refuse daily, consisting of municipal garbage, trash, brush, and construction debris. The refuse is buried in V-shaped trenches that range from 100 to 200 feet in width, 500 to 1,000 feet in length, and 15 to 20 feet in depth. The trenches are dug into the surficial sand and underlying marl.

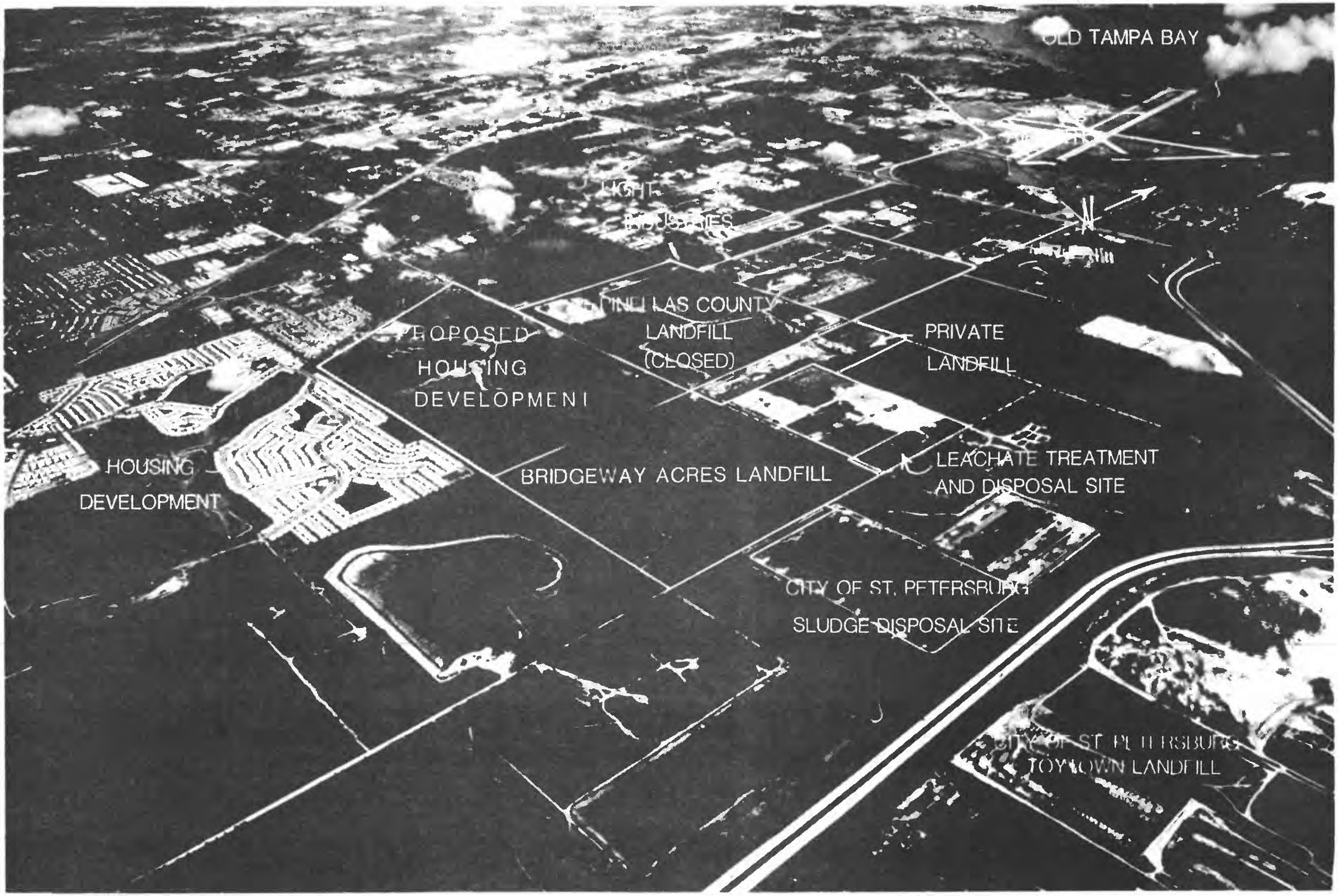

Figure 2.--Aerial view of the Bridgeway Acres landfill, leachate-treatment and disposal site, and surrounding area.

The depth of excavation is limited by thickness of the sand. The trenches are excavated with a dragline and the overburden is deposited adjacent to the trenches. Operators at the landfill have estimated that the average time of operation for a trench is about 2 months with an additional 4 months for the high-rise fill. Total time of operation at a trench site is estimated at about 6 months (Jesse Wells, Wells Bros. Inc., oral commun., 1981). Individual trenches must be dewatered during the landfilling operation due to the shallow water table. There is no dewatering during emplacement of the high-rise fill. During the process of covering the refuse, ground water that has infiltrated into the trench and mixed with the refuse (leachate) is pumped into the treatment system (fig. 3). Three trenches were excavated and dewatered during the period of study. Dewatering of the trenches was usually carried out Monday through Saturday or as needed during the period that the trenches were being filled. Pumping time over a 30-day period averages about 3.8 hours per day. The rate of leachate pumping ranges from about 150 to 500 gal/min. A conservative estimate for the rate and total volume of leachate pumped into the treatment system during the approximate 2-month operation of a trench is about 470 gal/min (110,000 gal/d) and 5,400,000 gallons, respectively.

The leachate-treatment and disposal site encompasses about 8 acres within the 220 acres of the Bridgeway Acres landfill as shown in figure 4. The treatment system consists of a lined aeration basin (fig. 4) and two unlined stabilization ponds. A description of the ponds follows:

Aeration Basin.--In this basin, a floating aerator transfers oxygen into the leachate by means of turbulent mixing, thereby treating the leachate the equivalent of secondary sewage treatment. The detention time in the basin is about 2 days (Donald Acenbrack, Pinellas County, oral commun., 1980).

North Oxidation Pond.--This stabilization pond is essentially facultative. The treatment process depends upon bacteria and algae for the degradation of putrescible organic material in conjunction with algal production of oxygen. This type of treatment is suitable where climatic conditions are favorable and where organic loadings may fluctuate considerably (Gloyna, 1968, p. 397). The leachate is generally retained in the pond until lost by evaporation and infiltration. When the pond reaches capacity, discharge of leachate from the pond goes into the south oxidation pond or is recirculated by pumping through a chlorinator and discharged into a nearby field.

South Oxidation Pond.--The south oxidation pond was designed to hold overflow from the north pond. During the first few months of the study, it was used as a control for determining background concentrations of chemical and biological constituents. During those months, overflow from the north pond had not occurred.

MONITORING NETWORK

A network of cluster wells was established in the surficial aquifer to obtain: (1) hydrologic information used in determining the direction and rate of ground-water movement and (2) water-quality data to define background conditions

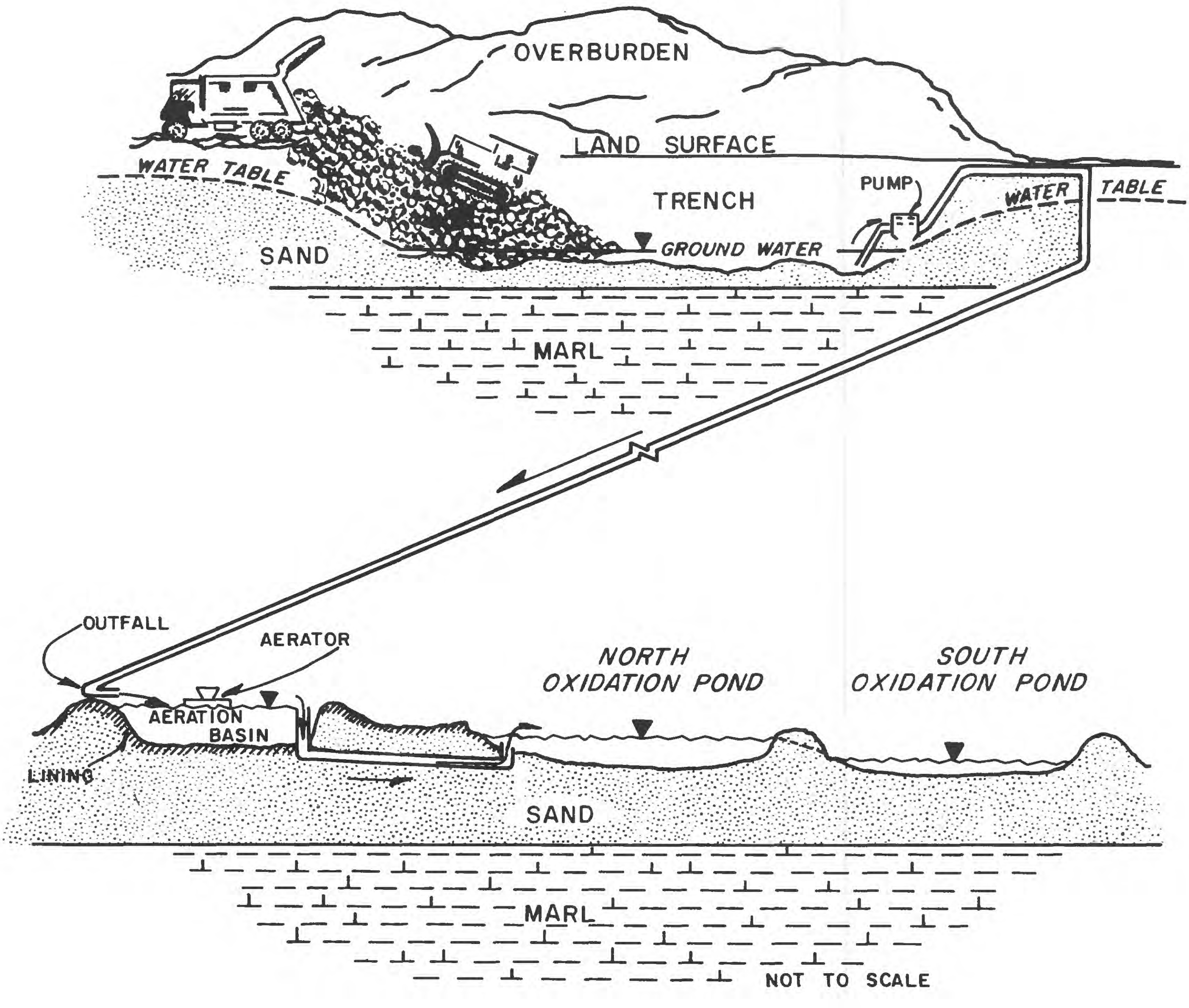

Figure 3.--Landfilling method and leachate-collection, treatment, and disposal system.

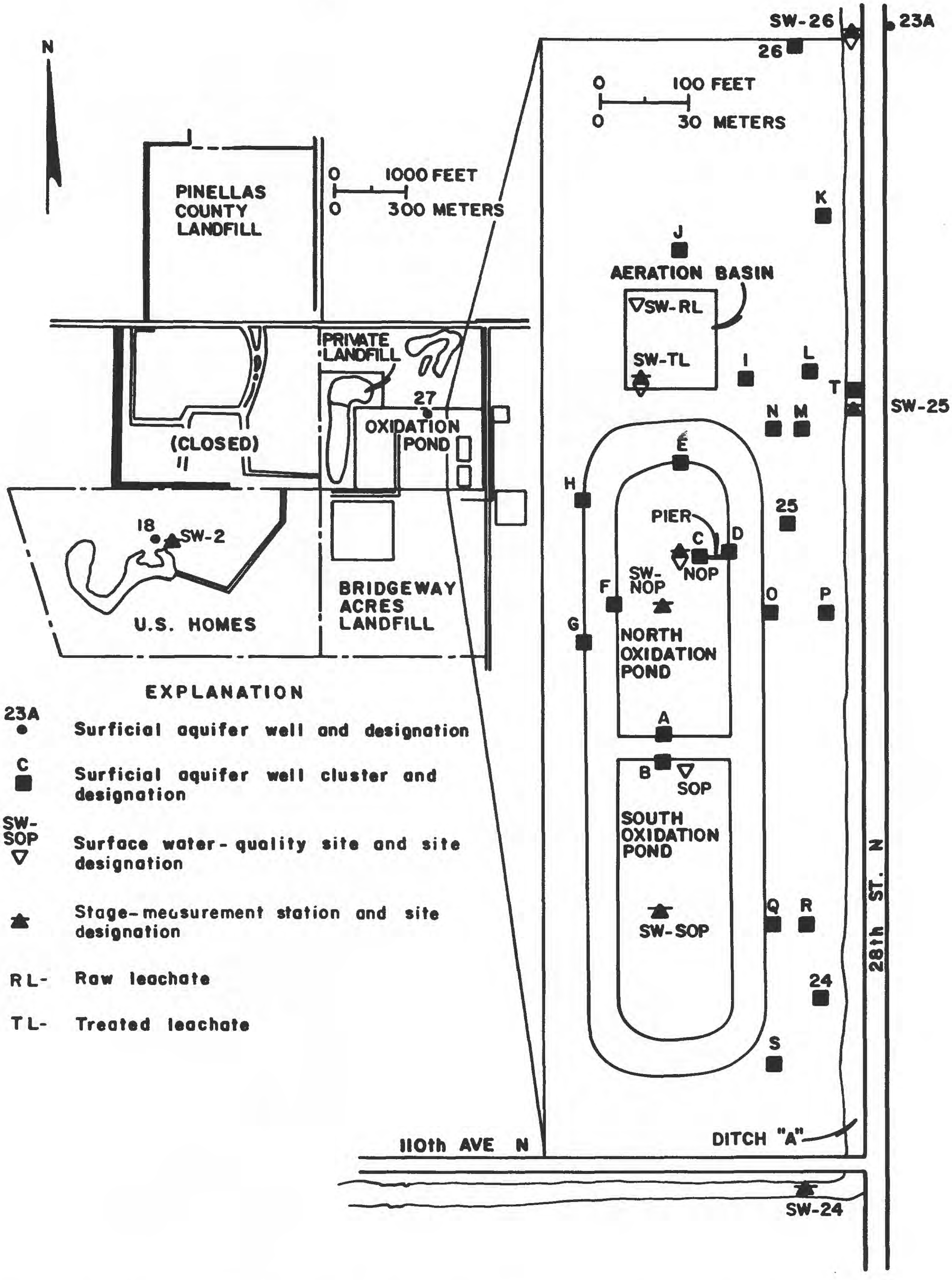

Figure 4.--Locations of wells and surface-water quality monitoring sites at the Bridgeway Acres landfill and leachate-treatment and disposal site.

and vertical and lateral changes in ground-water quality that are occuring due to leachate disposal operations. Eighty test holes, in clusters of four wells each, were drilled into the surficial aquifer. Well depths were the same for all clusters. Five clusters were jetted within the pond that received the treated leachate to monitor changes in the aquifer beneath the pond. Water levels were measured in wells to determine the depth to ground water and changes that occurred during ponding operations. Water-level contour maps for the ponding operations were prepared to determine the general direction and slope of ground-water movement. Chemical changes in water of selected well clusters were also used to determine rate of leachate migration during the period of study. Four surface-water stations were established to obtain data on surface-water quality and water levels in the ponds and the 28th Street ditch (ditch A). Existing wells and nearby surface-water sites were used as reference for background water-quality conditions. Monitoring ground and surface waters at the pond site was initiated prior to ponding of leachate to establish baseline water-quality conditions. Chemical analyses of water samples were made to determine concentrations of major ions, heavy metals, nutrients, chemical oxygen demand (COD), coliforms, phytoplankton, and benthic invertebrae before and during ponding operations. The network was completed and operational in 1979. Locations of the monitoring sites are shown in figure 4.

Test holes were drilled using a hollow-stem auger. Wells established in the ponds and ditch A (along 28th Street) were installed by jetting. All wells were cased and completed using 2-inch PVC (polyvinylchloride) pipe and finished with 1-1/2-inch diameter PVC screen. Wells were developed by pumping water until the specific conductance of the water stabilized. Well depths ranged from 1 to 10 feet, and screen lengths ranged from 0.5 to 1 foot. Descriptions of the wells and surface-water monitoring sites are given in a previous report by Barr and Fernandez (1981).

Water samples from wells were collected by first pumping out the equivalent volume of two casings using a centrifugal pump attached to a drop pipe in the well and then using a portable peristaltic pump to collect the water samples. Water samples from wells 18 and 23A and well clusters 24 through 27 (fig. 4) were used to establish background water-quality conditions unaffected by the landfill and leachate disposal operations. After the ponding operation began, water from wells 23A and 24 through 27 were no longer considered to reflect background water-quality conditions. Water samples from the surface-water site at the south oxidation pond (SW-SOP) were also used to establish background conditions. Water samples from well clusters C and D were used to determine ground-water quality changes in the surficial aquifer beneath the north pond. Water samples from clusters B, H, I, O, and P were used to determine the extent of lateral and vertical migration from the north oxidation pond (NOP).

GEOHYDROLOGY

Geologic Framework

Three stratigraphic units that underlie the landfill area were investigated (fig. 5). These units, in descending order of deposition, are the following: (1) undifferentiated surficial deposits of Pleistocene age; (2) the Hawthorn

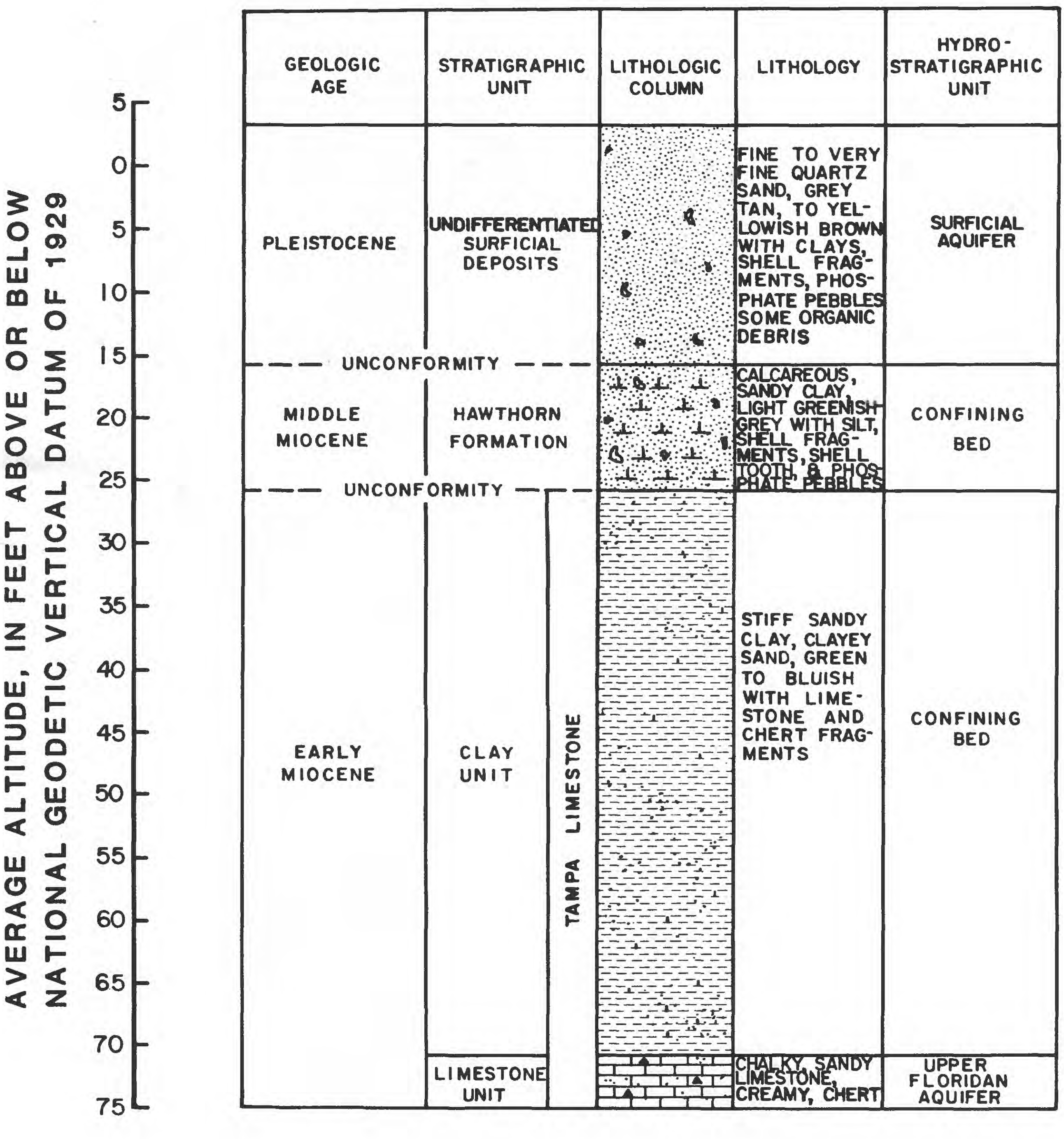

Figure 5.--Generalized geologic framework of the Pinellas County landfill study area (Heath and Smith, 1954).

Formation of middle Miocene age; and (3) the Tampa Limestone of early Miocene age (Heath and Smith, 1954). The surficial deposits, the calcareous sandy clay of the Hawthorn Formation, and clays of the Tampa Limestone are generally unconsolidated. The consolidated Tampa Limestone unit is the first of the persistent limestone section and is the top of the Floridan aquifer (Buono and Rutledge, 1978). Comparable deposits occur at a site 0.5 mile to the east, as described by Hutchinson and Stewart (1978).

The undifferentiated surficial deposits consist of a permeable bed of grey, tan, and yellowish brown, fine to very fine quartz sand that averages 19 feet in thickness (Fernandez, 1982). Quartz sand is the chief constituent of these deposits. Some shell and pelecypod fragments, phosphate pebbles, and organic debris occur with traces of calcite, aragonite, kaolinite, montmorillinite, and other mixed clay minerals. These components were described in the field from auger-drill cuttings and from split-spoon samples analyzed by the U.S. Geological Survey hydrologic laboratories in Denver, Colo. (Fernandez, 1982). As described by Heath and Smith (1954, p. 16), the Hawthorn Formation unconformably underlies the surficial deposits and unconformably overlies the Tampa Limestone.

The Hawthorn Formation averages about 10 feet in thickness and is an unconsolidated, light greenish-grey, calcareous sandy clay that contains silt, shell, and shark-tooth fragments and phosphate pebbles. Eleven selected samples of the formation from near the study area were analyzed using x-ray diffraction. The major constituents of the components were calcite, aragonite, and quartz and lesser amounts of mixed clay minerals. This formation material becomes sticky when wetted and is commonly referred to as marl. Unconformably underlying this marl layer is the clay unit of the Tampa Formation that is approximately 45 feet thick. It is a green to bluish, stiff, sandy clay and sometimes a clayey sand that contains limestone and chert fragments. At depth, the formation materials become stiffer and grade into the Tampa Limestone, a consolidated limestone and chert (Fernandez, 1979a; 1979b). The Tampa Limestone is cream colored, chalky, sandy, fossiliferous, and cherty with chert grading out with depth. Consolidated limestone occurs at about 70 feet below sea level and is estimated to be about 180 feet thick (Hickey, 1982).

Water-Bearing Characteristics

The hydrogeologic units (fig. 5) that underlie the study area are: (1) surficial aquifer comprised of sand; (2) confining beds comprised of the Hawthorn Formation and the clay unit of the Tampa Limestone; and (3) the upper part of the Floridan aquifer comprised of the Tampa Limestone. The Floridan aquifer serves as the major source of freshwater for municipal and domestic use and the surficial aquifer is primarily used for lawn irrigation in Pinellas County.

A previous study by Fernandez (1982) showed that vertical migration of chemical constituents through the confining layer would take about 7,000 years to reach the Floridan aquifer. Therefore, in this study, emphasis was placed on the hydrologic character of the surficial aquifer as defined by grain size and compaction of the sediments, effective porosity, hydraulic conductivity, transmissivity, and storage coefficient. Comparisons were made among four surficial-aquifer samples from investigations by Hutchinson and Stewart (1978) at the Toytown landfill

Table 1.--Hydraulic characteristics of the surficial aquifer from selected sites at the study area and Toytown landfill

[K_v, vertical hydraulic conductivity; K_h, horizontal hydraulic conductivity; mm, millimeter; ft/d, feet per day]

Well	Identification number	Median grain size (mm)	Effective porosity (percent)	K_v (ft/d)	K_h (ft/d)
11	2752120824058.01	0.16	28.6	8.56×10^{-5}	0.886
14	2752120824101.01	.14	29.9	1.44×10^{-3}	3.28
[1]TT-13	2752240823948.01	.15	29.2	--	2.0
[1]CW-1	(Toytown landfill perimeter canal wall)	.14	32.2	--	1.2
Average		.15	30.0	7.63×10^{-4}	1.8

[1]Wells TT-13 and CW-1 are located about 2,700 feet east of the study area at the Toytown landfill site (Hutchinson and Stewart, 1978).

and by Fernandez (1979a) at the Pinellas County landfill, both within half a mile of the study site. These samples, assumed to be representative of the study area, were analyzed for grain size, effective porosity, and vertical and horizontal hydraulic conductivity (table 1). The average grain size was 0.15 mm (millimeters); effective porosity, 30 percent; vertical hydraulic conductivity, 7.6×10^{-4} ft/d; and horizontal hydraulic conductivity, 1.8 ft/d. These hydraulic properties influence the flow of ground water horizontally and vertically through the surficial aquifer.

Water Levels

To determine changes in water levels in the surficial aquifer, observations of water levels were made in 80 surficial aquifer wells and at 4 surface-water sites. Water levels were measured daily from October 30, 1979, to November 10, 1979. Subsequently, water levels were usually measured once a month until December 1980. Water-level recorders were used at the north oxidation pond and aeration basin to determine inflow volumes of leachate.

Fluctuations in water levels in the surficial aquifer are due to seasonal changes caused by precipitation, evapotranspiration, downward confining-bed leakance, infiltration of leachate from the oxidation ponds, surface runoff from landfill mounds northwest of the oxidation ponds, and runoff from ditch A. Figure 6 shows weekly rainfall from January 1979 to December 1980 at the St. Petersburg-Clearwater Airport, about 2.5 miles north of the study site.

Before construction of the leachate-treatment system, a previous study by Fernandez (1982) in this general area reported that average water levels in the surficial aquifer in May, following the dry season, were about 5 feet above sea level. In September, following the wet season, the levels were about 8 feet above sea level.

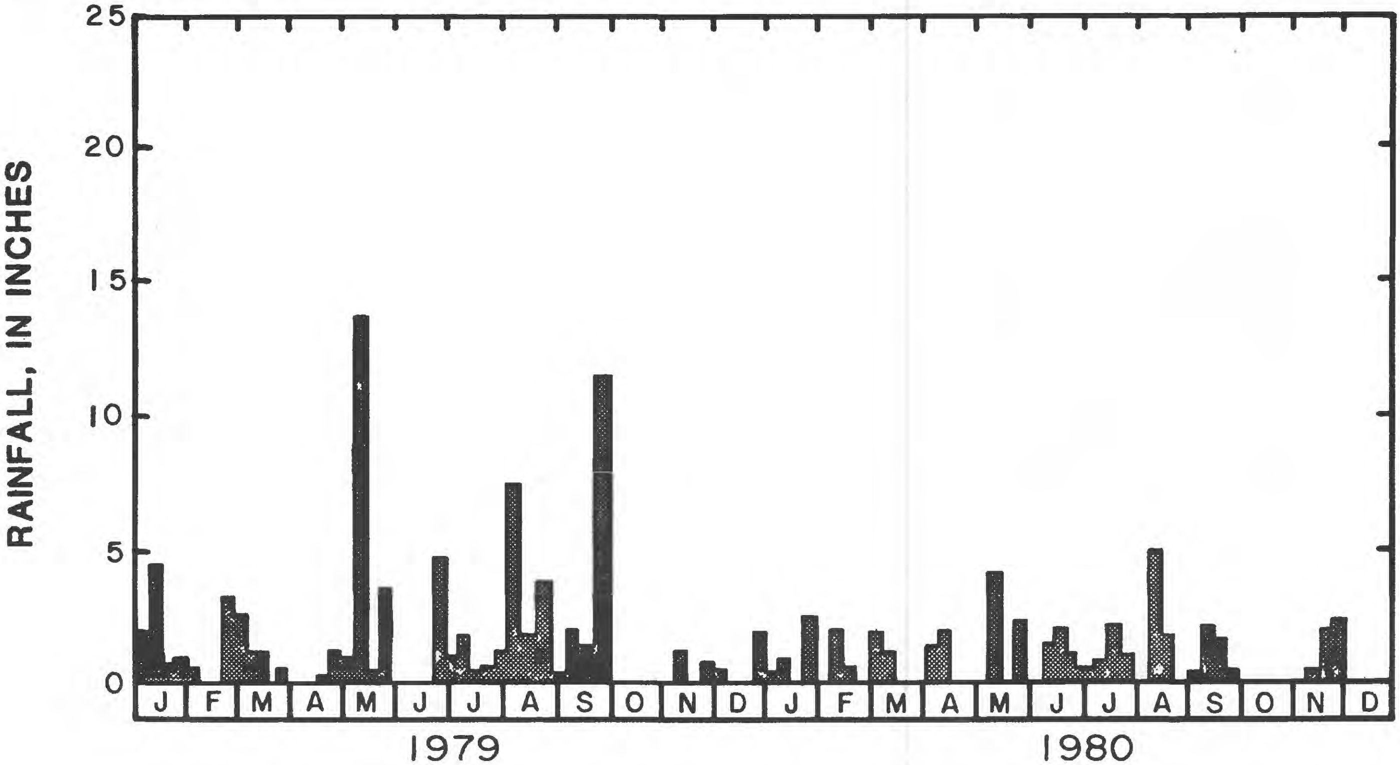

Figure 6.--Weekly rainfall at the St. Petersburg-Clearwater Airport, 1979-80.

Changes in water levels were used to determine the direction of ground-water flow, flow rate, and hydraulic gradient in the surficial aquifer. Water levels in the north oxidation pond and in well clusters O and P from October 1979 to December 1980 are shown in figure 7. Water levels in the four wells at each of the two well clusters were averaged to obtain the values plotted. As shown, the water levels at well clusters O and P did not fluctuate as much as those in the pond and were about 2 to 3 feet lower during the greater part of the period. Water levels in the oxidation pond had a downward trend, while levels in the two wells (O and P) had upward trends during April, June, September, November, and December 1980. This was due to a rising water table that had an effect on water levels in the wells but not in the pond. During May 1980, the water level of the pond increased while the water-table dropped. Possible causes for this condition include: large inputs to the pond while large losses occurred from the aquifer through evapotranspiration and a large lag time between water-level changes in the pond and water-level changes in the surficial aquifer.

To determine ground-water flow directions, water-level contour maps were developed (figs. 8-12). As shown on the maps, water levels were influenced by variations in water levels within the oxidation ponds. Figure 13 shows water levels along a section through the north oxidation pond for two periods. The high water level in the north oxidation pond on January 24, 1980, created large hydraulic gradients near the ponds. The pond level on December 1, 1980, caused a small hydraulic gradient and one similar to that of background gradient conditions. Hydraulic gradients between the north oxidation pond and well cluster O are larger than those between well clusters O and P, and therefore, ground-water velocities near the pond are larger than those velocities determined among the selected well clusters.

Figures 8-12 show changes in the water table in the surficial aquifer near the oxidation pond for selected periods from January 1980 through December 1980, changes that are due primarily to the pumping of leachate from the landfill through the aeration basin and into the oxidation pond. Water-table mounding at the ponds acted as a driving force to move ground water away from the ponds in all directions. During the rainy season (fig. 10), following periods of abnormally high rainfall (figs. 9 and 11) or during periods of extensive pumping of leachate into the ponds (fig. 8), the mounding effect was greatest and the most widespread effect was seen. Figure 12 shows conditions during a period after little rainfall and when leachate was not being pumped. As shown by figures 8-12, higher water levels in the surficial aquifer were maintained by ground-water mounding. During these high water-level conditions, a 7-foot water-level contour was situated between ditch A and the ponds. This was similar to wet-season water-level conditions prior to the operation of the oxidation ponds (Fernandez, 1982).

In the report by Fernandez (1982), it was shown that during May 1977, following a dry season, a 5-foot water-level contour was situated between ditch A and where the ponds would eventually be constructed. During September 1977, following a wet season, the water level in this area was raised to about 7 feet. During the leachate-treatment study, when leachate was being pumped into the ponds, the 5- and 7-foot water-level contours shifted farther away from the pond as water-level mounding at the ponds took place. Throughout the study, water levels in the north oxidation pond were at a higher stage than those of the south oxidation pond.

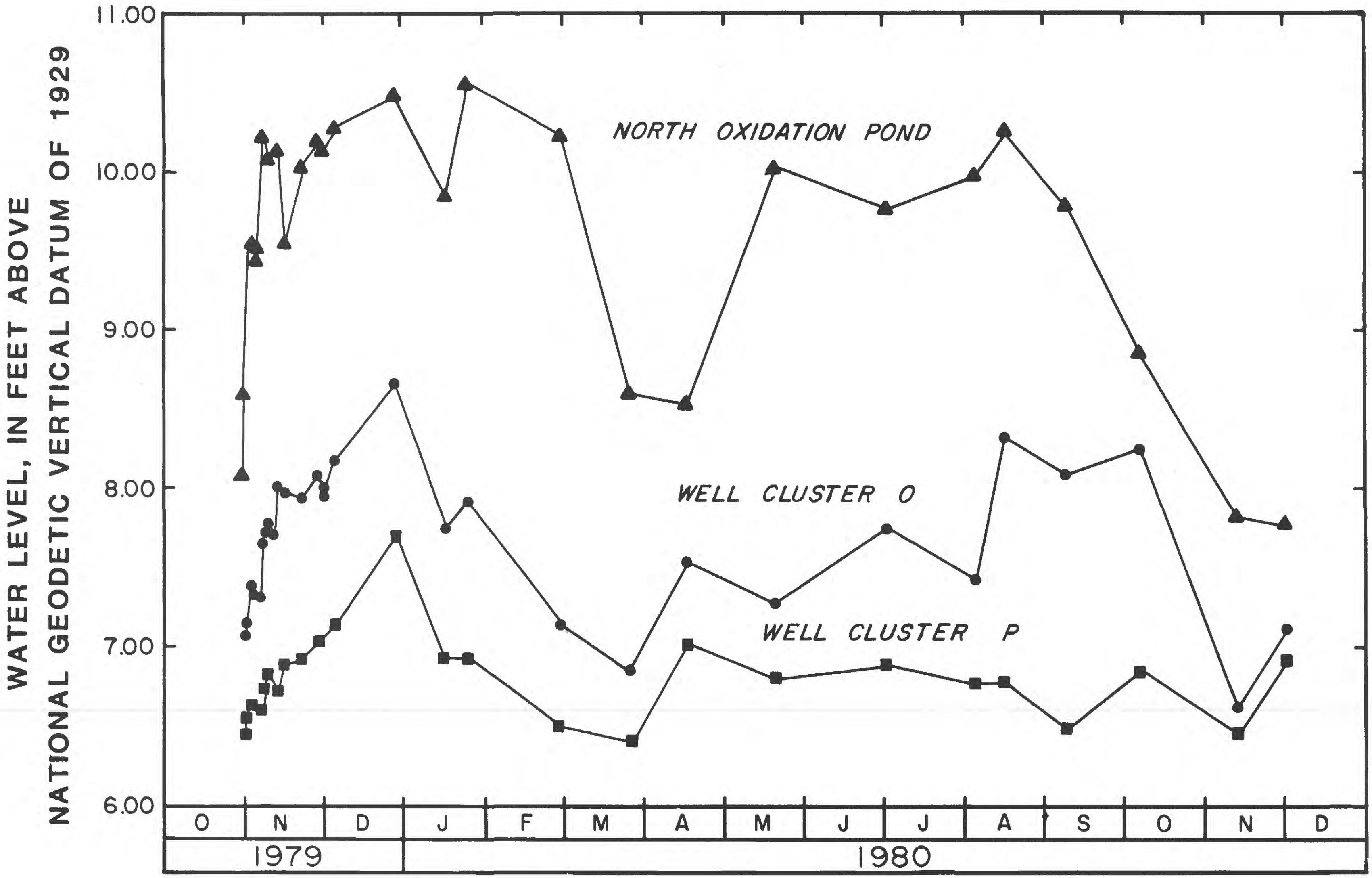

Figure 7.--Water levels in the north oxidation pond and average water levels in well clusters O and P.

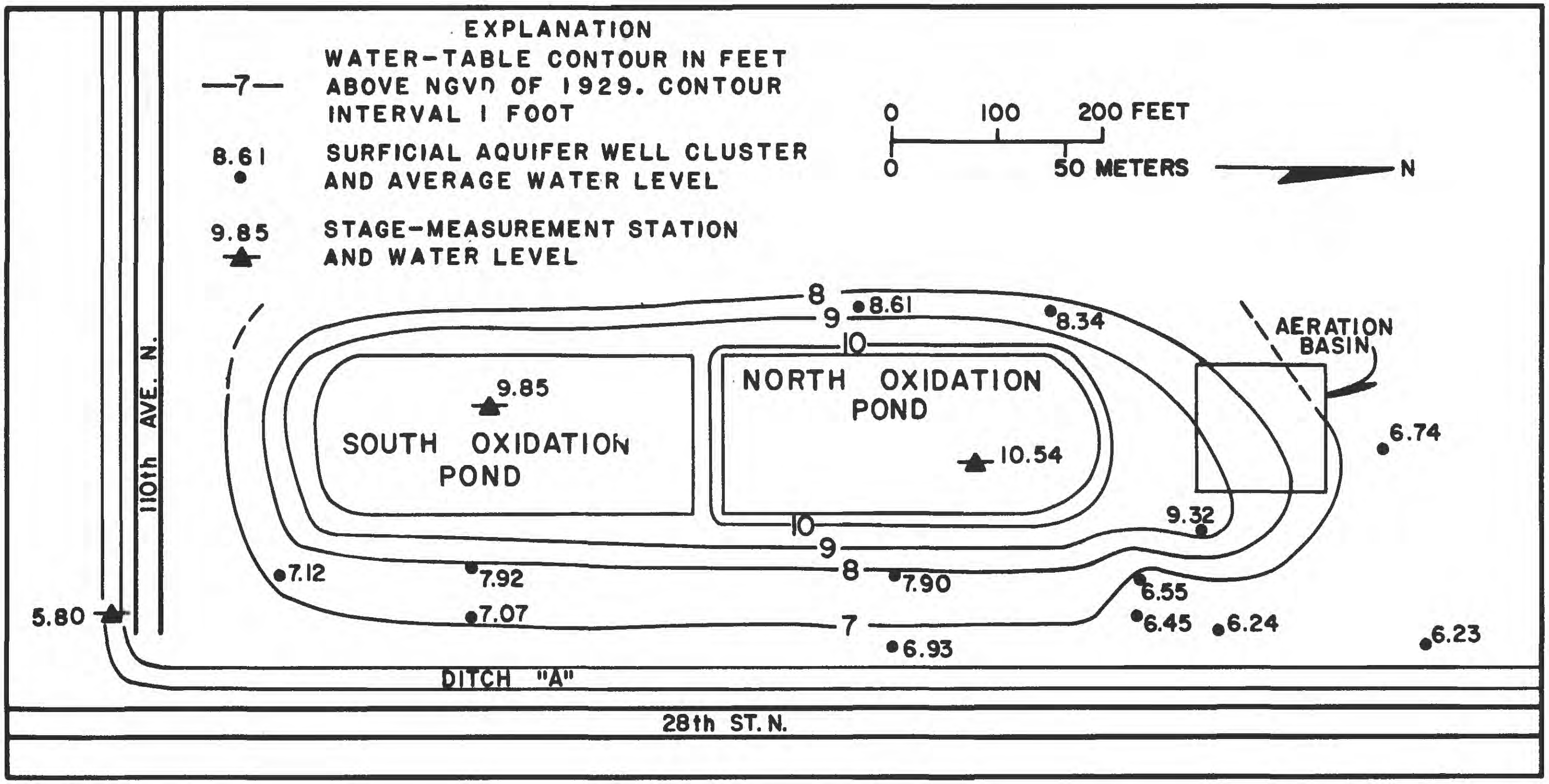

Figure 8.--Water table at the Pinellas County leachate-treatment and disposal site for January 24, 1980.

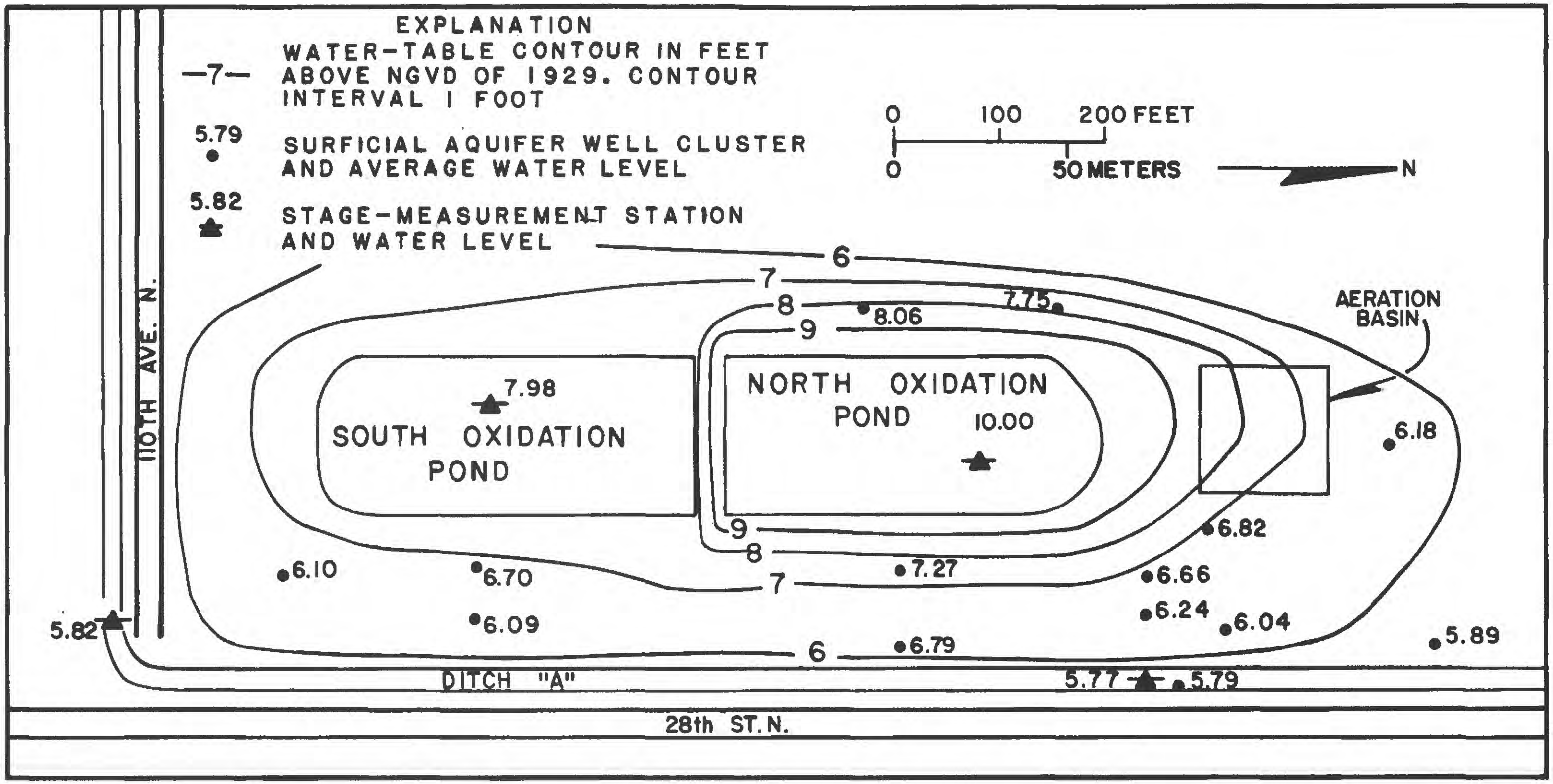

Figure 9.--Water table at the Pinellas County leachate-treatment and disposal site for May 19, 1980.

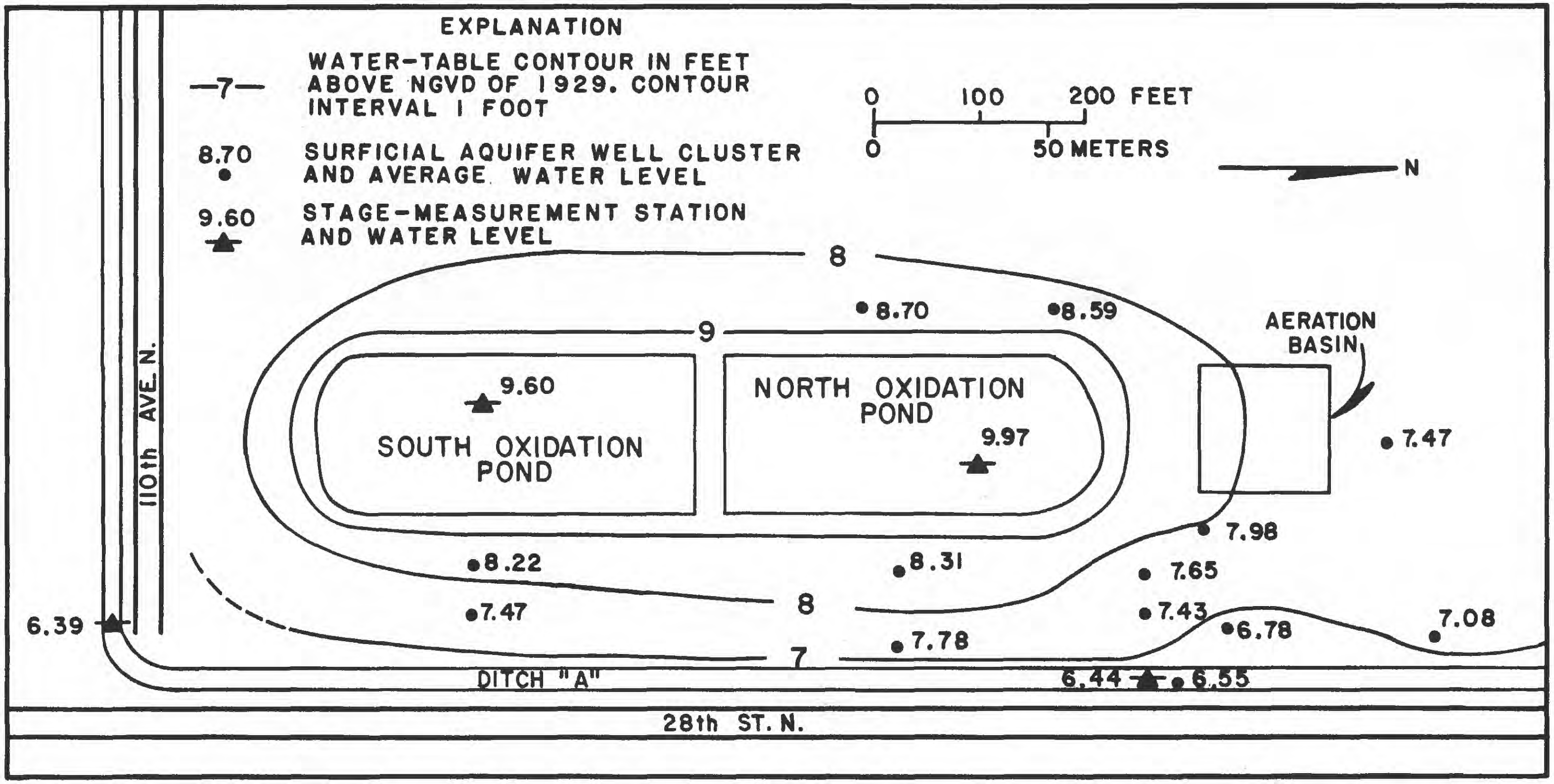

Figure 10.--Water table at the Pinellas County leachate-treatment and disposal site for August 15, 1980.

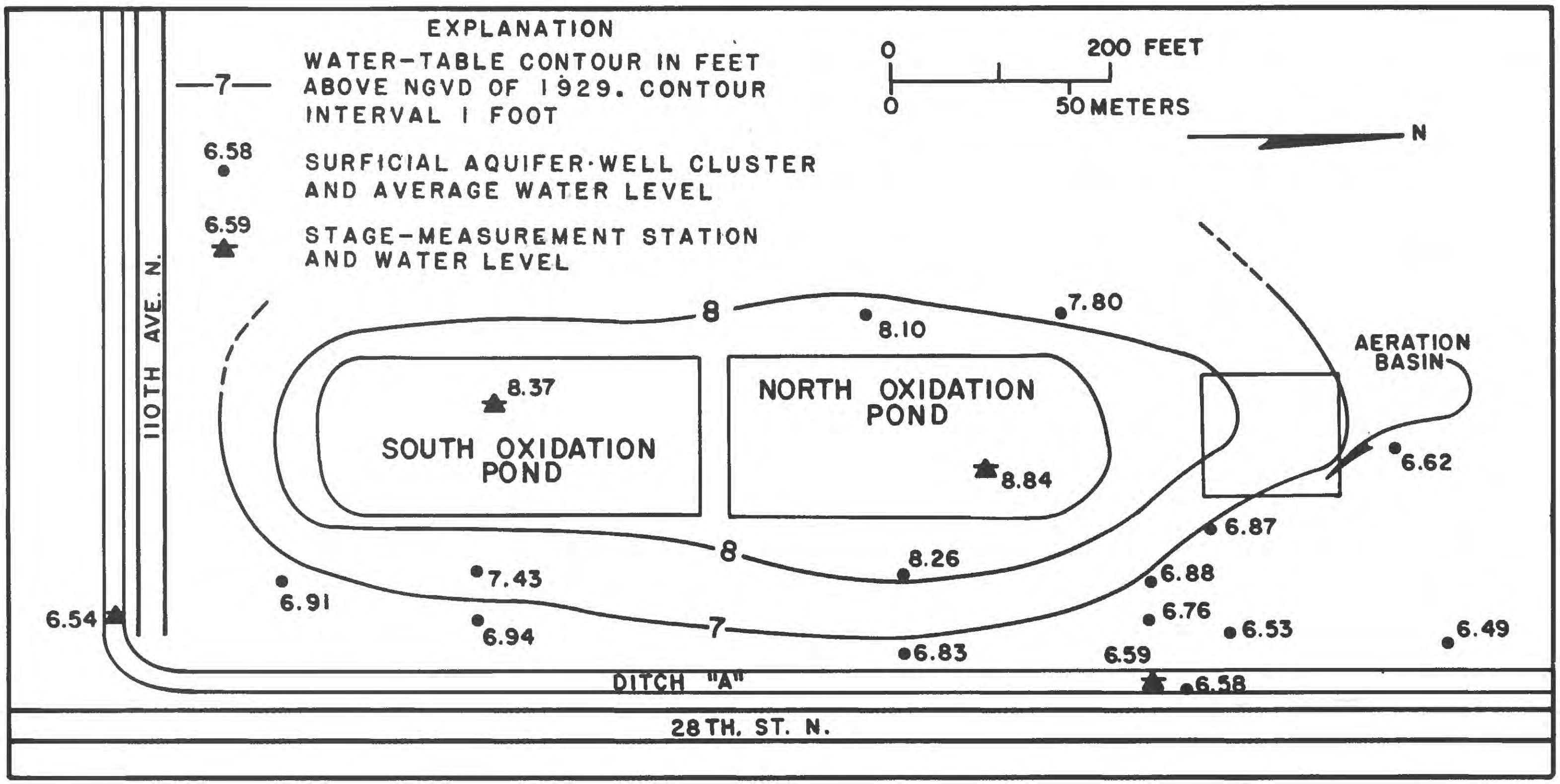

Figure 11.--Water table at the Pinellas County leachate-treatment and disposal site for October 6, 1980.

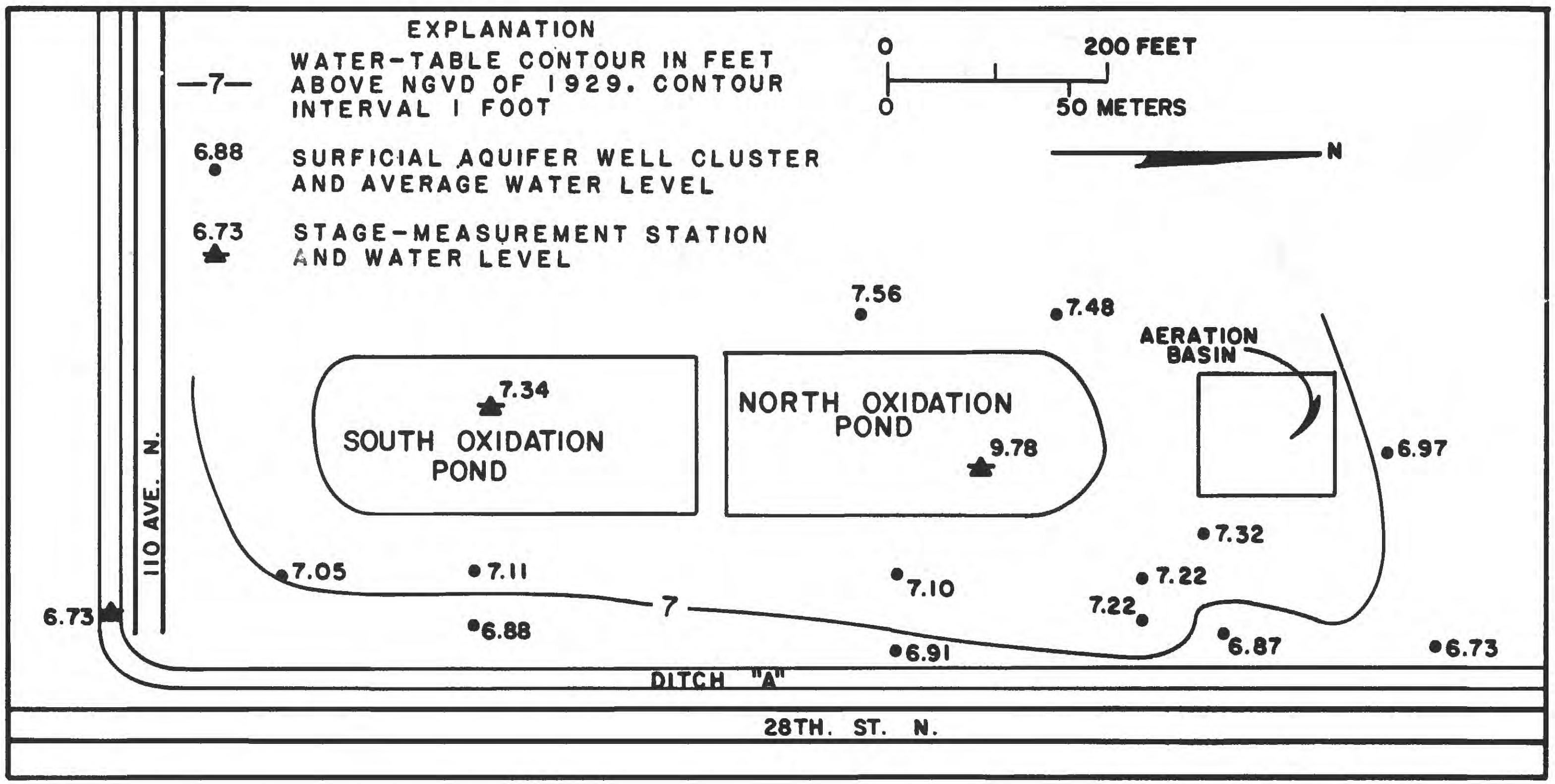

Figure 12.--Water table at the Pinellas County leachate-treatment and disposal site for December 1, 1980.

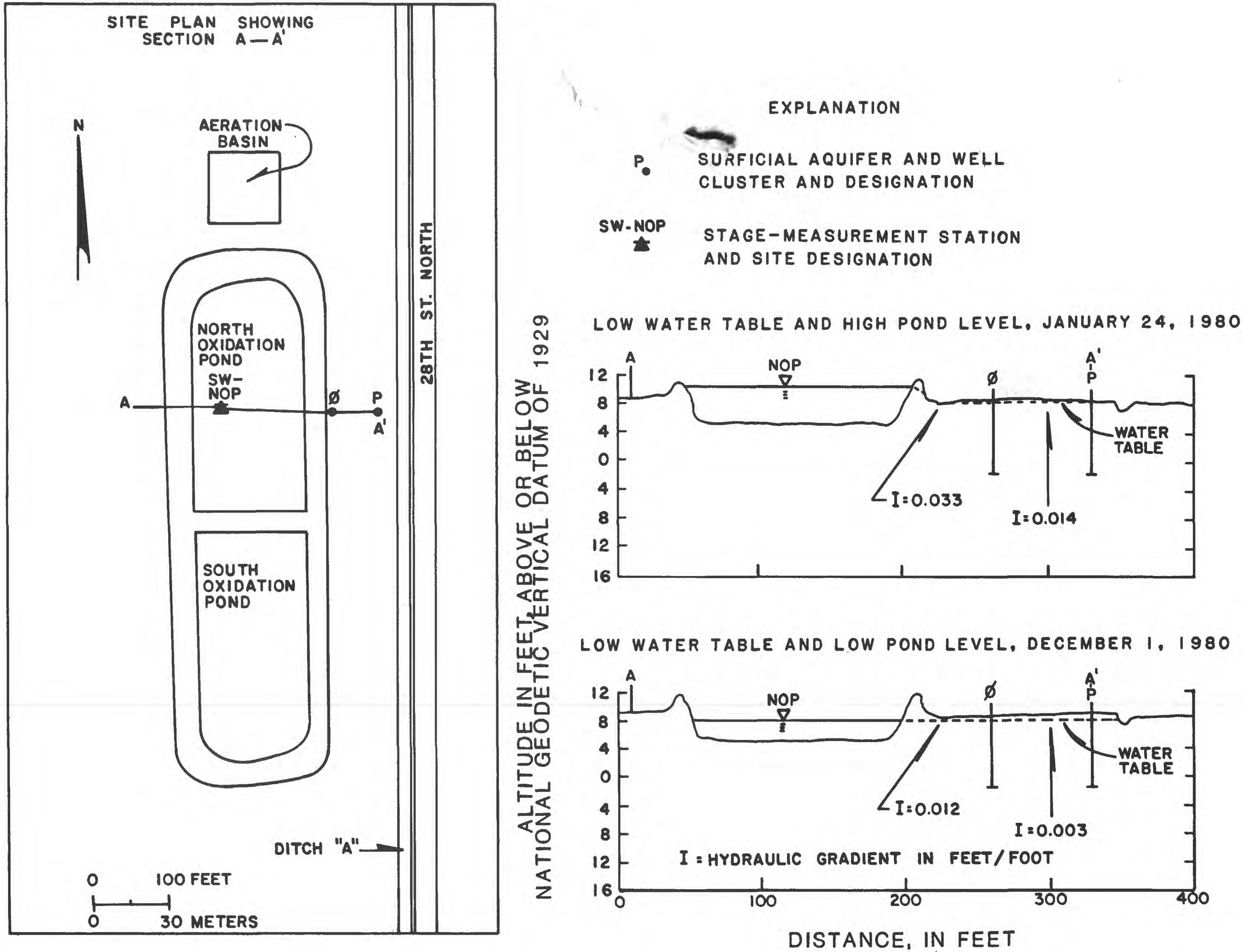

Figure 13.--Hydraulic gradients between the north oxidation pond and well clusters O and P during selected high and low water-level periods.

Surface-water runoff flowing in a northeasterly or easterly direction is intercepted by ditch A that acts as a conveyance for surface water along the east side of the study area (fig. 4). Water in the ditch flows east, parallel to 110th Avenue North, and then north along ditch A, parallel to 28th Street North, and discharges into Old Tampa Bay. During the period of study, flow of water in a section of ditch A, just east of the oxidation ponds between SW-25 and SW-24 (fig. 3), had changed direction from north to south. The southerly flow was a result of the mounding effect upon ground-water discharge to the ditch. However, the resulting effects of the reversed hydraulic gradient in ditch A for these periods on the overall flow pattern for the study area were insignificant.

WATER QUALITY

To evaluate the migration of leachate in the aquifer, water samples were collected before and during the operation of the treatment facility. Background ground-water quality conditions were determined by combining data from wells 18 and 23A and well clusters 24 through 27 (fig. 4) so that changes resulting from infiltration of leachate could be distinguished from natural conditions. Water from selected clusters of wells in and near the pond and from well 18 were sampled periodically before and during ponding for nutrients, chemical oxygen demand, major cations, and trace metals. Well 18 (fig. 4) is upgradient from the oxidation ponds and had been established as a background monitor well for a previous study. Well 23A and well clusters 24 through 27 (fig. 4) were established to detect leachate migration from the landfill (Fernandez, 1982).

The Floridan aquifer was not monitored because the possibilities of contamination from the leachate ponding site were considered to be slight. An earlier study showed that it would require about 7,000 years for leachate to travel vertically through the confining layer into the Floridan aquifer (Fernandez, 1982).

Samples of raw and treated leachate and samples from the north oxidation pond were also analyzed for the above constituents. In addition, the north oxidation pond was sampled for phytoplankton, chlorophyll *a*, and benthic invertebrae.

Surface-water site SW-2, near well 18, was selected as the reference for background surface-water quality after the south oxidation pond became contaminated by treated leachate from the north oxidation pond.

Background water-quality data for ground- and surface-water sites are listed in table 2. These data include the mean and, if sufficient data were available, the standard deviation and range.

Discussions that follow describe the quality of background surface and ground water, raw and treated leachate, water in the north oxidation pond, and leachate-affected water in the surficial aquifer.

Table 2.--Statistics of selected water-quality constituents and physical characteristics

[umho, micromhos per centimeter at 25 degrees Celsius; mg/L, milligrams per liter; ug/L, micrograms per liter]

Constituents and physical properties	"t" test 1/	Degrees of freedom		Background ground water 2/		
		df*	df**	Mean	Standard deviation	Range
Specific conductance, umho	D	12	16	1,600*	±521	800-2,310
				1,080**	±310	725-1,630
Chemical oxygen demand, mg/L	ND	11	12	97	±56	10-250
pH	ND	10	16	7.0	±0.4	6.1-7.7
Nitrogen, organic (total), mg/L	D	11	16	1.6*	±0.67	0.88-3.4
				0.50**	±0.24	0.20-0.95
Nitrogen, ammonia (total and dissolved), mg/L	ND	11	16	0.39	±0.61	0.08-3.4
Nitrogen, nitrite (total and dissolved), mg/L	ND	11	16	0.01	±0.01	0.00-0.03
Nitrogen, nitrate (total and dissolved), mg/L	ND	11	16	0.00	±0.01	0.00-0.02
Calcium, dissolved, mg/L	ND	10	14	150	±42	94-250
Magnesium, dissolved, mg/L	D	10	14	17*	±6	3.6-28
				12**	±5	3.7-20
Sodium, dissolved, mg/L	D	10	14	190*	±68	34-280
				68**	±48	24-190
Potassium, dissolved, mg/L	ND	10	14	1.4	±0.8	0.6-3.9
Chloride, dissolved, mg/L	D	12	16	260*	±100	63-439
				130**	±80	7.1-290
Arsenic (total), ug/L	ND	9	14	5	±7	0-33
Cadmium (total), ug/L	ND	9	14	2	±5	0-22
Chromium (total), ug/L	ND	9	14	18	±20	4-100
Copper (total), ug/L	ND	9	14	9	±13	0-55
Iron (total), ug/L	ND	6	4	8,900	±4,700	990-18,000
Iron (dissolved)***, ug/L	ND	-	-	1,600	±2,800	0-7,500
Lead (total), ug/L	ND	8	14	17	±19	0-72
Mercury (total), ug/L	ND	8	14	0.3	±0.2	0-1
Zinc (total), ug/L	ND	9	14	38	±33	10-150

Footnotes are at end of table.

Table 2.--Statistics of selected water-quality constituents and physical characteristics--Continued

Constituents and physical properties	Background surface water, SW-2			Raw leachate		
	Mean	Standard deviation	Range	Mean	Standard deviation	Range
Specific conductance, umho	615	±216	398-860	1,810	±1,170	1,010-3,550
Chemical oxygen demand, mg/L	60	±13	44-81	--	--	-
pH	7.8	±0.6	6.9-8.5	6.9	±0.5	6.2-7.6
Nitrogen, organic (total), mg/L	0.78	±0.09	0.67-0.88	7.3	--	0.72-14
Nitrogen, ammonia (total and dissolved), mg/L	0.05	±0.06	0.00-0.15	0.06	--	0.05-0.08
Nitrogen, nitrite (total and dissolved), mg/L	0.00	±0.00	0.00-0.01	0.02	--	0.01-0.03
Nitrogen, nitrate (total and dissolved), mg/L	0.01	±0.03	0.00-0.07	0.4	--	0.01-0.81
Calcium, dissolved, mg/L	54	±13	44-78	220	±110	150-380
Magnesium, dissolved, mg/L	11	±4.4	6.7-17	20	±14	7.1-40
Sodium, dissolved, mg/L	60	±31	30-100	98	±45	53-160
Potassium, dissolved, mg/L	1.6	±0.6	0.7-2.4	42	±60	2-130
Chloride, dissolved, mg/L	90	±46	44-150	110	±30	74-130
Arsenic (total), ug/L	1	±2	0-4	25	±31	4-61
Cadmium (total), ug/L	<1	--	0-1	2	±2	1-4
Chromium (total), ug/L	16	±18	10-30	200	--	50-350
Copper (total), ug/L	2	±1	0-3	25	±36	1-66
Iron (total), ug/L	120	±56	70-180	58,000	--	58,000
Iron (dissolved)***, ug/L	95	--	30-160	30	--	30
Lead (total), ug/L	6	±5	1-14	36	±53	2-97
Mercury (total), ug/L	0.3	±0.2	0.0-0.5	0.5	--	0.1-0.1
Zinc (total), ug/L	17	±14	0-40	370	±360	110-790

Table 2.--Statistics of selected water-quality constituents and physical characteristics--Continued

Constituents and physical properties	Treated leachate			North oxidation pond		
	Mean	Standard deviation	Range	Mean	Standard deviation	Range
Specific conductance, umho	1,060	±730	430-2,700	945	±330	480-1,520
Chemical oxygen demand, mg/L	440	±780	6-2,000	120	±110	3.5-360
pH	8.2	±0.9	6.9-9.3	7.7	±0.6	6.9-8.9
Nitrogen, organic (total), mg/L	7.0	±8.0	0.12-19	6.0	±8.2	0.12-32
Nitrogen, ammonia (total and dissolved), mg/L	3.9	±8.4	0.00-21	2.3	±4.7	0.01-16
Nitrogen, nitrite (total and dissolved), mg/L	0.02	±0.01	0.00-0.04	0.01	±0.01	0.01-0.03
Nitrogen, nitrate (total and dissolved), mg/L	0.04	±0.06	0.00-0.15	0.03	±0.06	0.00-0.26
Calcium, dissolved, mg/L	120	±93	23-270	77	±48	24-210
Magnesium, dissolved, mg/L	13	±9.1	4.1-30	14	±6.7	5.2-27
Sodium, dissolved, mg/L	90	±31	56-140	87	±23	43-130
Potassium, dissolved, mg/L	23	±35	2.3-100	36	±38	0.10-120
Chloride, dissolved, mg/L	120	±33	94-200	130	±35	79-200
Arsenic (total), ug/L	3	±5	0-8	2	±1	1-3
Cadmium (total), ug/L	1	±1	0-2	<1	--	0-1
Chromium (total), ug/L	33	±23	20-60	23	±12	5-37
Copper (total), ug/L	4	±2	2-6	2	±1	1-2
Iron (total), ug/L	1,900	±3,200	50-5,700	420	±430	50-1,100
Iron (dissolved)***, ug/L	--	--	-	250	±220	30-460
Lead (total), ug/L	6	±4	2-9	2	±1	0-4
Mercury (total), ug/L	0.7	±0.3	0.5-1	0.3	±0.2	0.1-0.5
Zinc (total), ug/L	200	±280	30-530	30	±22	10-70

1/ Student's "t" test: null hypothesis $H_0: X_1 = X_2$. Average for well 18 (X_1) is equal to the cumulative average for well 23A and well clusters 24 through 27 (X_2). Yes, no significant difference (ND) exists between X and X_2 at the 95 percent confidence level; no, a difference (D) exists between X and X_2 at the 95 percent confidence level.

2/ Background includes wells 18, 23A, and 24 through 27. Only data prior to July 1979 for wells 23A and 24 through 27 were used.

*Statistics for well 18.

**Statistics for cumulative average for well 23A and well clusters 24 through 27.

***Insufficient data; 5 and 0 degrees of freedom.

Background Quality of the Surficial Aquifer

The quality of water from well 18 (near surface-water site SW-2) was compared to that from well 23A and from well clusters 24 through 27 near the oxidation pond to establish background water quality in the surficial aquifer. A t-test of the hypothesis that the means of two groups of data are equal (SAS Institute, Inc., 1979) was run for comparison of mean constituent levels to determine whether the data from the two sites could be combined and used as a single background quality reference. Results of the test are presented in table 2. The values for specific conductance, organic nitrogen (total), magnesium, sodium, and chloride could not be combined to use as background water quality. The average values of each of the five constituents for water from well 18 and the average for water from well 23A and well clusters 24 through 27 are presented for comparison. The average for all five constituents in water from well 18 was greater than the average from the wells near the oxidation pond. The reason for the difference is probably due to the alteration of the land. Well 23A and well clusters 24 through 27 are at sites with natural vegetation, whereas well 18 is at a site where trees were cleared leaving only shrub and grass.

Raw Leachate

Samples of raw leachate were collected at the orifice (outfall) of the pipe delivering leachate to the aeration basin (fig. 4) and were analyzed for chemical, physical, microbiological constituents, and characteristics. Results of the analyses are presented in a report by Barr and Fernandez (1981) and in table 2. Results indicate that levels of the following constituents and characteristics were greater, by up to a factor of about 30, in the leachate than in background groundwater samples (table 2): specific conductance, organic nitrogen, nitrate nitrogen, potassium, arsenic, chromium, copper, total iron, lead, and zinc. The remaining constituents occurred at concentrations that were either lower than or approximately equal to background levels (table 2).

The comparatively low levels of dissolved constituents in the leachate are probably due to the relatively short period of contact between refuse and ground water when compared to completed trenches. The ground water that percolates into the operational trench is kept at the lowest level possible by pumping the water out of the trench, thereby reducing the contact time of ground water with the refuse. The ground water that is pumped is not truly leachate, but rather a dilute mixture of leachate and ground water.

Treated Leachate

When leachate is aerated, dissolved gases are stripped, and reduced cations in solution, especially iron, are oxidized and precipitated. A discussion on aeration and mass transfer of oxygen can be found in Eckenfelder (1966).

Samples of aerated leachate were collected near the discharge orifice. The results of aerating can be seen by comparing values of constituents in raw leachate with treated leachate (table 2). The reduction in specific conductance and concentrations of trace metals is attributed to coprecipitation of dissolved cations by ferric hydroxide formed during aeration and to formation of calcium carbonate ($CaCO_3$) by the loss of carbon dioxide (CO_2). Ion exchange by clay particles in the trenches is probably negligible because clay content is low in strata exposed by the trenches. The mineralogical analysis of corings from a site about 4,000 feet west of the ponds (Fernandez, 1982) indicates that at 13 to 18 feet below land surface, the clay content of the soil is as follows: kaolinite, 1 percent; montmorillonite, 5 to 12 percent; and mixed clay minerals, 2 to 7 percent.

The highly soluble ferrous ion precipitates out of solution when oxidized as colloid ferric hydroxide. Below pH 5.2, the ferric hydroxide colloid has a surface charge that is positive; at pH greater than 5.2, the colloid becomes negatively charged and attracts and coprecipitates heavy metal ions. Coprecipitation effects of ferrous and ferric solutions is presented by Hem and Skougstad (1960). Dissolved calcium precipitates out of solution as calcium carbonate when the dissolved carbon dioxide is stripped during aeration. The loss of carbon dioxide is accompanied by an increase in pH (table 2).

Average specific conductance decreased from 1,810 to 1,060 micromhos per centimeter (umho/cm) at 25°C in the aeration process. Specific conductance in micromhos per centimeter at 25°C divided by 100 approximates the milliequivalents per liter of anions and cations (Skougstad and others, 1972, p. 9). The loss in specific conductance can be approximated by multiplying the difference in cations or anions between raw and aerated leachate by 100. This decrease is attributed mainly to the precipitation of iron and calcium. Loss in milliequivalents in iron and calcium amounts to about 7 milliequivalents, and multiplying by 100 gives a loss of 700 micromhos. This value is comparable to the loss of 750 micromhos obtained from averaged measured specific conductance.

Aeration Basin

A bench-scale test, performed before the aeration basin became operational, was made using leachate-laden ground water to define changes in quality that might occur through treatment of leachate by aeration. Since raw leachate was not available, ground-water samples were obtained from a well open to the surficial aquifer. A previous study by Fernandez (1982) indicated that leachate had migrated into this well from completed trenches about 85 feet upgradient from the well. A test was also performed on the raw leachate collected at the aeration basin after it became operational. The results of both tests are presented in figure 14.

Figure 14 shows that the greatest reduction in specific conductance of leachate-laden ground water, about 36 percent, occurred within the first 16 hours of aeration, conductance then increased by about 16 percent followed by a gradual decrease. Figure 14 also shows that the specific conductance of the

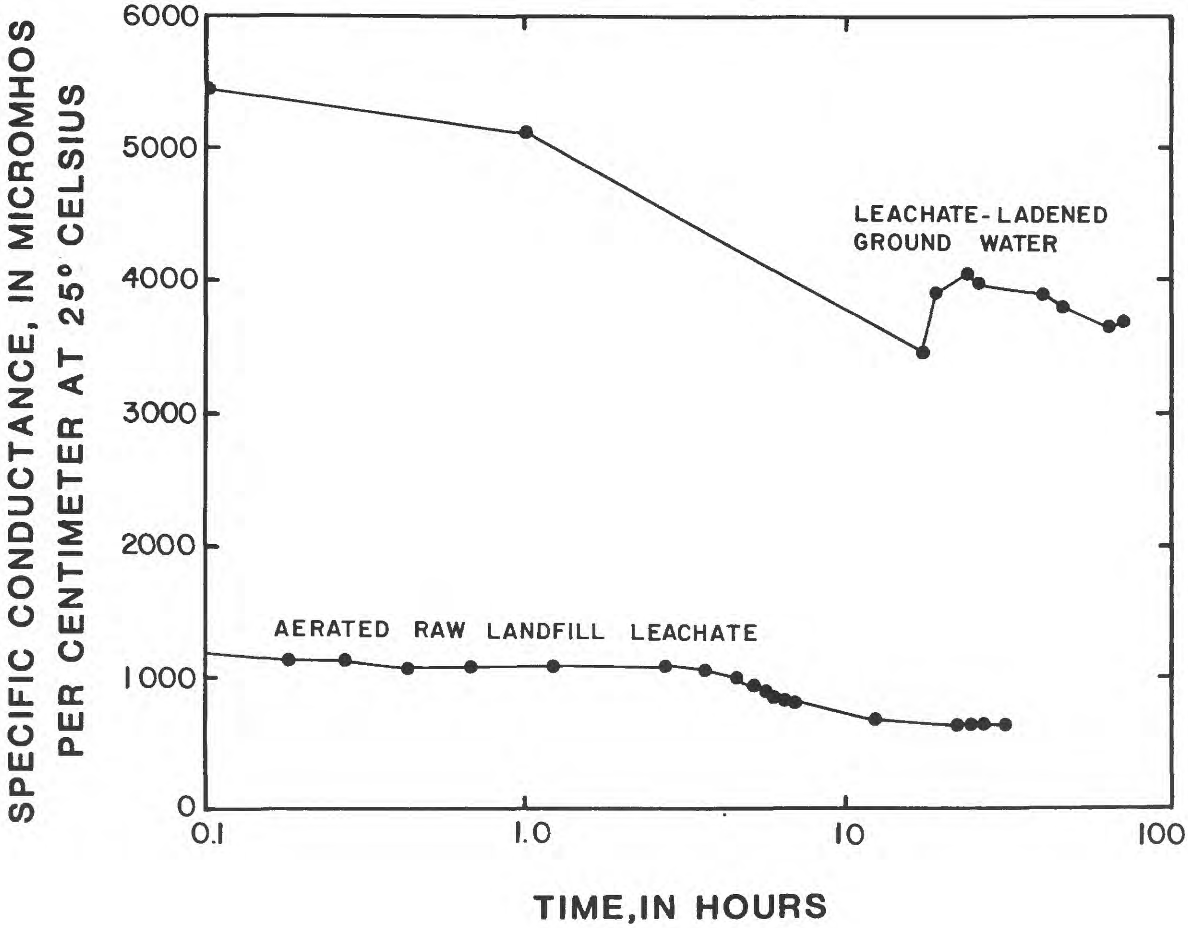

Figure 14.--Effects of aeration time on specific conductance for landfill leachate-laden ground water and for raw landfill leachate.

raw leachate decreased by about 46 percent after about 20 hours of aeration. Findings for the efficacy of the aeration basin under operating conditions showed that the specific conductance of the raw leachate was reduced about 41 percent after 2 days detention in the basin (table 2). The decrease in specific conductance is probably due to precipitation of dissolved calcium and iron in the form of calcium carbonate and ferric oxide, respectively. Based on the bench test, it appears that aeration is a viable method for treatment of leachate. Where leachate has infiltrated from a landfill and percolated through the surficial aquifer, it could be possible to withdraw the leachate-laden ground water by using wells that are spaced sufficiently close to each other and then treat the leachate by aeration.

North Oxidation Pond

The north oxidation pond is a stabilization pond that receives treated landfill leachate by gravity flow from the aeration basin for further treatment before discharging through infiltration into the surficial aquifer. Treatment in the pond is basically a biological process that is dependent upon bacteria to break down putrescible organic compounds and upon the algae for oxygenation. Analyses of water samples collected periodically from the north oxidation pond were reported by Barr and Fernandez (1981). The pond is a dynamic system and concentrations of selected constituents vary with each inflow of treated leachate.

Data for the pond show that nitrate nitrogen concentrations, 0.03 ± 0.06 mg/L (milligrams per liter) (modified from Barr and Fernandez, 1981), are near the 0.01 mg/L detection limit for the precision of the method of analysis used (Beetem and others, 1981). It appears that little or no nitrification, conversion of ammonium to nitrite and then nitrate, occurs within the north oxidation pond. What is probably occurring is deamination or ammonification, a process by which bacteria (usually saprophytic) transforms organic nitrogen (protein matter) from animal or plant matter into ammonia under aerobic or anaerobic conditions; the undigestible part becomes the detritus in water (Sawyer and McCarty, 1978, p. 441). The ammonia produced is probably assimilated by phytoplankton and aquatic plants into cell material (Boney, 1976, p. 29).

Two constituents were selected to illustrate trends in ground- and surface-water quality that are caused by landfill operation. The constituents representative of the organic and inorganic chemical character of leachate are ammonia nitrogen and potassium, respectively. These indicators are commonly used by researchers in monitoring the movement of leachate in the environment. These constituents also are generally among the parameters that must be monitored as required by many regulatory agencies (U.S. Environmental Protection Agency, 1975).

The quality of water in the north oxidation pond was compared to that at a reference background surface-water site, SW-2 (table 2), to determine how water quality in the north oxidation pond changed in relation to background conditions. Graphs for ammonia nitrogen and potassium concentrations for water samples collected from the north oxidation pond for the period of study compared to background surface-water quality are presented in figure 15. The background concentrations of 0.17 mg/L for ammonia nitrogen and 2.8 mg/L of potassium (table 2) represent the upper limits of background surface-water quality and were set at two standard deviations that represent 95 percent of the observed data.

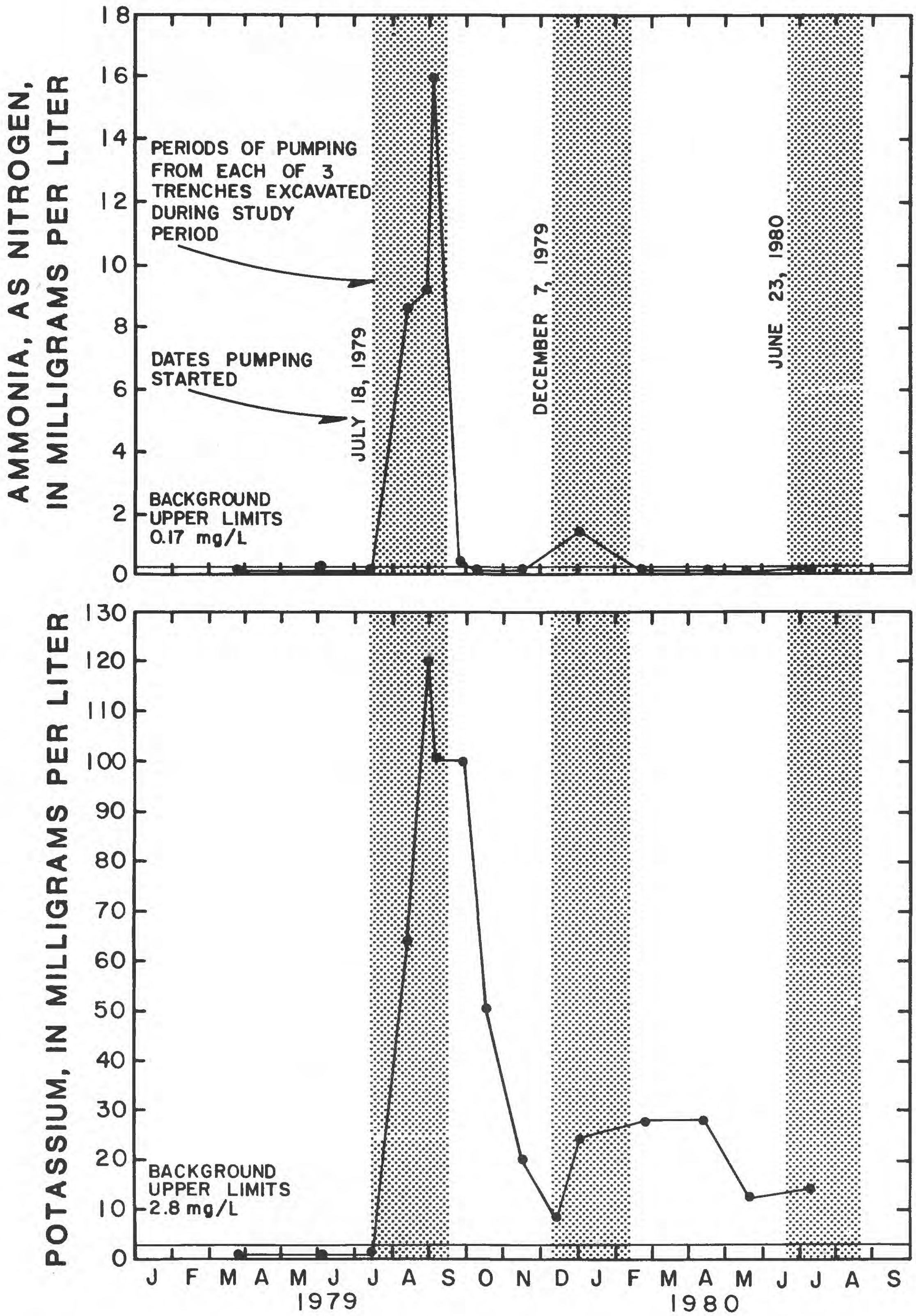

Figure 15.--Ammonia nitrogen (top) and potassium (bottom) concentrations in water from the north oxidation pond.

Treated leachate was discharged into the pond for durations of approximately 2 months beginning in July 1979, December 1979, and June 1980. The increased concentrations of ammonia nitrogen and potassium in July through September 1979 (fig. 15) reflect the leaching of the readily soluble ammonia nitrogen and potassium in the landfill refuse by rainfall that occurred during May 1979 (fig. 6) prior to pumping the leachate from the trench. During the period May through July, the refuse near the bottom of the trench was beneath the water table.

The decline in ammonia nitrogen and potassium concentrations that occurred during the pumping period in September is probably due to rainfall occurring in September that lowered concentration levels in the pond. The decline is a result of dilution of pond water and lower concentrations in the leachate, since most of the readily soluble materials had previously been leached from the refuse. Thus, weaker leachate was being pumped into the pond. The rise in concentrations within the pond during December is probably due to pumping leachate from a new trench. In this instance, however, the levels of ammonia nitrogen and potassium were not as high as during the previous pumping period because (1) the water level in the new trench was at a minimum because water was pumped out of the trench prior to the deposition of refuse, thereby decreasing the amount of ground water in contact with the refuse, and (2) the rainfall was considerably less during this period than the previous summer, thereby probably reducing leaching.

A comparison was made between the leachate indicators--organic nitrogen, ammonia nitrogen, calcium, and potassium--for the north oxidation pond (table 2) and the background ground-water quality data (table 2) using a t-test of hypothesis on two means. The data indicated that there was a significant difference between the means for organic nitrogen, calcium, and potassium from the pond and background ground-water quality at the 95 percent confidence level. Although there was not any difference between the ammonia nitrogen values at the 95 percent level for both sites, there was a difference at the 90 percent level. It was decided to keep ammonia nitrogen as an indicator in comparing these two sites. Though organic nitrogen and calcium were also tested (U.S. Environmental Protection Agency, 1975), they were not selected because organic nitrogen is easily converted to ammonia by bacterial action and also because levels of organic nitrogen are relatively high in concentration in the background ground-water samples (table 2). Calcium was not selected because it may precipitate out of solution with an increase in pH.

Phytoplankton and Bacterial Data for the North Oxidation Pond

Water samples were collected from the north oxidation pond for analysis of phytoplankton, chlorophyll *a*, total coliforms, and fecal coliforms. The purpose of the sampling was to determine: (1) changes in phytoplankton as treated leachate was introduced into the pond and (2) the effects of aeration and ponding on coliform densities. Results from analyses of samples from the north oxidation pond were compared to those from the south oxidation pond and surface-water site SW-2 (fig. 4). Surface-water site SW-2 was used as an alternate to the south oxidation pond during the second sampling due to a cross connection occurring between the north and south oxidation ponds at some period after the first sampling.

Water samples were collected at two random points within each site. Phytoplankton samples were collected in November 1979 and August 1980. Bacterial samples were collected in July and August 1979 and July and August 1980.

Samples for coliform analysis were collected from raw leachate, aeration basin, north oxidation pond, south oxidation pond, and surface-water site SW-2. Coliform density comparisons were made between the sites. A discussion of the findings follows.

Phytoplankton

Although populations of phytoplankton (algae) appear to be higher in the control pond (south oxidation pond) than in the north oxidation pond in the November 1979 sampling of the ponds, there is probably no significant difference biologically between the control and north oxidation ponds when the total number of taxa and the diversity are considered (table 3). After November 16, 1979, the pond became cross contaminated by the north oxidation pond when the north pond filled and water was pumped into the south pond to relieve high water-level conditions. The August 1980 sampling of the ponds indicated that the north oxidation pond had a population greater than the control (now the pond at SW-2) by a factor over 10 and the total number of taxa and diversity was less in the control pond than in the north oxidation pond (table 4). A low number of taxa and diversity is usually an indication of a body of water under stress; the north oxidation pond received treated landfill leachate.

Concentrations of chlorophyll _a_ in November 1979 (table 3) and August 1980 (table 4) were significantly higher in the north oxidation pond that received leachate than in either the south oxidation pond (control pond in 1979) or the control pond, SW-2 (past November 1979). In the north oxidation pond, the south oxidation pond, and the control ponds, chlorophyll _a_ increased by several orders of magnitude in August 1980. The higher concentration of chlorophyll _a_ in the north oxidation pond indicates that this pond supported a larger algal biomass and probably more algal productivity. The large biomass probably results from a high concentration of nutrients. For example, concentrations of nitrogen and phosphorus in the north oxidation pond averaged 8.3 and 0.54 mg/L, respectively, for the period of study, whereas in the control pond at SW-2, nitrogen and phosphorus averaged 0.83 and 0.16 mg/L, respectively.

Coliform Bacteria

Water samples were collected from a control site (south oxidation pond), an active trench, a lined aeration basin, and the north oxidation pond. Samples were analyzed for total and fecal coliforms and fecal streptococci. The results, and the ratio of fecal coliform to fecal streptococci (FC/FS), are presented in table 5 and discussed as follows:

South Oxidation Pond: Initially used as a control, the pond water was altered when excess water was pumped from the north to the south pond.

Table 3.--Phytoplankton data for control pond (south oxidation pond) and north oxidation pond, November 16, 1979

[Analysis by Environmental Science and Engineering, Inc., 1979, Tampa, Florida. Data in cells per milliliter (cells/mL) except as noted; mg/m^3, milligrams per cubic meter]

Species	Control pond		North oxidation pond	
	1	2	3	4
Division Chlorophyta				
Order Chlorococcales				
Ankistrodesmus falcatus	-	147	-	75
A. f. var. spiriliformis	665	84	36	350
Chlorella sp.	190	147	144	-
Crucigenia tetrapedia	-	-	-	25
Scenedesmus biguga	95	63	36	-
S. opolienis	95	-	-	-
S. quadricauda	-	-	72	50
S. sp.	-	-	-	25
Selenastrum minutum	5,798	1,155	684	425
Tetraedron caudatum	95	21	-	-
Order subtotal	6,938	1,617	972	950
Order Volvocales				
Chlamydomonas botrys	-	63	72	425
C. fenestrata	-	21	-	-
Order subtotal	0	84	72	425
Order Zynemateles				
Mougeotia spp.	-	1	-	-
Division subtotal	6,938	1,702	1,044	1,375
Division Cyanophyta				
Order Chroococales				
Agmenellum quadrduplicatum	-	21	-	-
Anacystis incerta	380	-	-	-
Order subtotal	380	21	0	0
Order Nostocales				
Anabaena sphaerica	-	21	-	25
A. spp.	95	21	-	-
Order subtotal	95	42	0	25
Order Oscillatoriales				
Oscillatoria geminata	380	-	-	-
Phormidium angustissimum	95	36	-	-
P. spp.	-	-	36	-
Order subtotal	475	36	36	0
Division subtotal	950	99	36	25

Table 3.--Phytoplankton data for control pond (south oxidation pond) and north oxidation pond, November 16, 1979--Continued

Species	Control pond		North oxidation pond	
	1	2	3	4
Division Cryptophyta				
Family Cryptochrysidaceae				
Cryptomonas coerulea	-	63	36	-
C. erosa	-	-	-	25
Rhodomonas minuta	2,281	714	2,088	1,225
Division subtotal	2,281	777	2,124	1,250
Division Chrysophyta				
Family Bacillariophyceae				
Cocconeis placentula	-	42	-	-
Cyclotella glomerata	-	-	144	150
C. meneghiniana	95	-	-	-
Navicula cryptocephala	190	-	-	-
N. spp.	-	21	144	25
Nitzschia palea	-	21	-	-
N. thermalis	-	-	-	50
Family subtotal	285	84	288	225
Order Chromulinales				
Cercobodo angustus	-	-	72	-
Monochrysis vesiculifera	95	-	-	-
Order subtotal	95	0	72	0
Order Ochromonadales				
Mallomonas caudata	-	-	72	-
M. spp.	-	-	-	25
Synura spp.	-	-	-	75
Order subtotal	0	0	72	100
Division subtotal	380	84	432	325
Division Euglenophyta				
Family Euglenaceae				
Euglena proxima	-	-	-	25
Phacus spp.	-	-	36	-
Trachelomonas volvocina	190	63	36	-
Division subtotal	190	63	72	25
Division Phyrrhophyta				
Family Peridiniacea				
Peridinium inconspicuum	190	21	36	-
Division subtotal	190	21	36	0
Total cells/mL	10,929	2,746	3,744	3,000
Total number of taxa	16	20	16	16
Diversity (H')	2.36	2.74	2.35	2.77
Evenness (E)	.59	.64	.59	.69

Table 3.--Phytoplankton data for control pond (south oxidation pond) and north oxidation pond, November 16, 1979--Continued

Species	Control pond		North oxidation pond	
	1	2	3	4
Chlorophyll _a_ (mg/m^3)	0.41	0.80	2.41	1.74
Average	0.61		2.08	

Active Trench: Samples of leachate from an active trench were obtained from a 6-inch pipe delivering leachate-laden ground water into a lined aeration basin. Analysis indicates the total coliforms, fecal coliforms, and fecal streptococci averaged about $3.1x10^6$, $1.3x10^6$, and $8.2x10^5$ colonies per 100 milliliters (col/100 mL), respectively. The average FC/FS ratio was 1.6. The FC/FS ratio for warm-blooded animals is 0.6 or less, whereas for man, it is 4.4 or greater (Geldreich, 1966, p. 101). The results in table 5 indicate that not all coliforms and fecal streptococci reported were derived from human sources, but had to include animal waste. Because dead animals were not accepted at the landfill, the animal waste probably originates from the numerous seagulls and other birds that scavenge the landfill and roost on the overburden next to the operating trenches.

Aeration Basin: Samples were collected near the outlet of the aeration basin. Total coliforms, fecal coliforms, and fecal streptococci averaged $1.8x10^6$, $6.7x10^5$, and $1.2x10^5$ col/100 mL, respectively. The average FC/FS ratio was 5.6, which indicates contamination by human waste. The difference in ratios between the leachate and samples from near the basin outlet cannot be explained with the amount of data collected.

North Oxidation Pond: Samples were collected from the pond near well cluster C (fig. 5). Total coliforms ranged from 120 to 180,000 col/100 mL for the period of study. Leachate was initially introduced into the pond on July 18, 1979. The 120 col/100 mL reflects the coliform levels in the ponds prior to receiving leachate. During the first month following the introduction of leachate, the FC/FS ratio ranged from 1.0 to 23. Results of samples collected for the period of study are presented in table 5. Although the FC/FS ratio for the aeration basin and the pond on July 7, 1980, is about the same, 52 and 55, respectively, the population counts were considerably less in the pond, which indicates the pond was thoroughly mixed and the microbial population was being diluted in the pond. Similar findings were found for other sampling dates (table 5). The trend in FC/FS ratio in the pond indicates that an increase probably has occurred in the amount of warm-blooded animal waste in the pond. The increase of the FC/FS ratio to 55 on July 7, 1980, and then down to 4.8 and 9.4 on July 14 and August 13, 1980, respectively, is probably due to the treated leachate that was introduced into the pond beginning on June 23, 1980.

Table 4.--Phytoplankton data for control pond (pond at SW-2) and north oxidation pond, August 12, 1980

[Analysis by Environmental Science and Engineering, Inc., 1980, Tampa, Florida. Data in cells per milliliter (cells/mL) except as noted: mg/m^3, milligrams per cubic meter]

Species	Control pond 1	Control pond 2	North oxidation pond 1	North oxidation pond 2
Division Chlorophyta				
Order Chlorococcales				
Actinastrum spp.	-	-	75	-
Ankistrodesmus falcatus	36	36	792	-
Bottryococcus spp.	-	-	38	-
Coelastrum spp.	-	-	-	113
Crucigenia spp.	-	4	-	-
Elakothrix spp.	-	-	-	30
Golenkinia spp.	-	4	-	-
Kirchneriella spp.	-	12	-	-
Micratinium spp.	-	-	75	-
Oocystis spp.	9	20	-	38
Scenedesmus spp.	12	32	-	-
Tetraedron spp.	9	24	-	38
Order subtotal	66	132	980	219
Order Tetrasporales				
Gloeocystis spp.	9	-	-	-
Order subtotal	9	0	0	0
Order Volvocales				
Chlamydomonas	93	164	-	38
Eudorina elegans	-	-	38	113
Pandorina spp.	-	-	-	38
Spondylomorum spp.	12	16	302	151
Order subtotal	105	180	340	340
Order Zynemateles				
Mougeotia spp.	3	-	-	-
Order subtotal	3	0	0	0
Unidentified coccoids	-	-	302	264
Unidentified flagellates	-	-	-	76
Subtotal	0	0	302	340
Division subtotal	183	312	1,622	899
Division Euglenophyta				
Order Euglenales				
Euglena acus	-	-	2,865	4,147
Euglena spp.	-	-	75	113
Lepocinclis spp.	-	-	-	38
Pincus longicauda	-	-	226	716
P. orbicularis	-	-	38	792
Phacus spp.	-	-	75	226
Division subtotal	0	0	3,279	6,032

Table 4.--Phytoplankton data for control pond (pond at SW-2) and north oxidation pond, August 12, 1980--Continued

Species	Control pond		North oxidation pond	
	1	2	1	2
Division Cyanophyta				
Order Nostocales				
Anabaena spp.	-	8	-	-
Order subtotal	0	8	0	0
Order Chroococales				
Anacystis spp.	-	1	-	113
Order subtotal	0	1	0	113
Order Oscillatoriales				
Oscillatoria geminata	42	60	-	-
Oscillatoria spp.	87	80	75	113
Order subtotal	129	140	75	113
Division subtotal	129	149	75	226
Division Cryptophyta				
Family Cryptochrysidaceae				
Chroomonas norstedtii	-	76	-	-
Chroomonas spp.	3	8	-	-
Cryptomonas spp.	-	-	-	38
Division subtotal	3	84	0	38
Division Chrysophyta				
Family Bacillariophyceae				
Acnanthes spp.	9	4	-	-
Amphora spp.	9	4	-	-
Cyclotella spp.	-	20	-	-
Diatoma spp.	6	-	-	-
Diploneis spp.	6	-	-	-
Gyrosigma spp.	3	-	-	-
Mastogloia spp.	6	12	-	-
Navicula spp.	120	92	-	-
Neidium spp.	-	4	-	-
Nitzschia spp.	135	80	-	-
Rhopalodia spp.	6	-	-	-
Synedra spp.	24	4	-	-
Division subtotal	324	220	0	0
Division Phyrrhophyta				
Family Glenodiniaceae				
Glenodinium pluvisculus	-	-	1,697	3,770
Dinoflagellate cysts	-	-	1,621	943
Division subtotal	0	0	3,318	4,713

Table 4.--Phytoplankton data for control pond (pond at SW-2) and north oxidation pond, August 12, 1980--Continued

Species	Control pond		North oxidation pond	
	1	2	1	2
Total cells/mL	639	765	8,294	11,908
Total number of taxa	21	23	15	21
Diversity (H')	3.4	3.67	2.69	2.64
Evenness (E)	.77	.81	.69	.61
Chlorophyll *a* (mg/m^3)	16.93	16.04	607.31	525.25
Average	16.49		568.25	

Table 5.--Bacteria data for surface waters

[All data except ratios, which are dimensionless, are in colonies per 100 milliliters]

Site	Date sampled	Coliform		Fecal streptococcus (FS)	Ratio FC/FS[1]
		Total	Fecal (FC)		
[2]South oxidation pond	8-01-79	[3]K 380	210	[3]E $1.5x10^4$	0.01
	7-07-80	$1.2x10^4$	[3]K 50	<10	>5
Active trench	7-18-79	$4.8x10^6$	$2.0x10^6$	$5.2x10^5$	3.8
	7-19-79	$6.9x10^6$	[3]K $2.9x10^6$	$2.7x10^6$	1.1
	7-07-80	$4.0x10^5$	[3]K $1.9x10^5$	$2.0x10^3$	95
	7-14-80	$2.2x10^5$	[3]K $1.8x10^5$	[3]K $5.0x10^4$	3.6
Aeration basin	7-19-79	$2.5x10^6$	$6.0x10^5$	$2.7x10^5$	2.2
	7-20-79	$4.5x10^6$	$2.0x10^6$	$2.1x10^5$	9.5
	7-07-80	$6.1x10^4$	[3]K $6.2x10^4$	[3]K $1.2x10^3$	52
	7-14-80	[3]K $5.0x10^4$	$3.3x10^4$	$1.0x10^4$	3.3
North oxidation pond	7-18-79	[3]K 120	290	[3]K 26	11
	7-19-79	360	230	92	2.5
	8-01-79	$2.0x10^4$	$<1x10^4$	$<1x10^4$	1.0
	8-08-79	$4.2x10^4$	$4.6x10^4$	$2.0x10^3$	23
	7-07-80	$2.7x10^3$	550	[3]K 10	55
	7-14-80	$1.0x10^4$	[3]K $1.1x10^3$	230	4.8
	8-13-80	$1.8x10^5$	$3.3x10^3$	350	9.4

[1]Ratio of fecal coliform to fecal streptococcus. Ratio greater than 4.0 usually indicates the source is domestic wastes, for example, human body waste, laundry waste, and food refuse (Geldreich, 1966, p. 103).

[2]South pond was initially used to obtain background conditions.

[3]K indicates that the reported data are based on colony count outside acceptable range. E indicates that the reported data are estimated.

MIGRATION OF LEACHATE IN THE SURFICIAL AQUIFER

The approximate distance the leachate migrated was determined using water-quality data. The approach and findings are discussed, and comparison is made of the time it took the leachate to migrate with the computed time of ground-water movement for the same distance.

Aerated (treated) landfill leachate, when ponded, infiltrates into the surficial aquifer and carries with it dissolved chemical constituents such as ammonia nitrogen and potassium. Water from well clusters C, O, and P (fig. 4) were used to identify changes in water quality in the surficial aquifer as a result of water infiltrating the aquifer from the north oxidation pond. Well cluster C is in the pond. The wells are opened at depths of 1, 3, 5, and 7 feet below the bottom of the pond and are assumed to represent water-quality conditions in the surficial aquifer beneath the entire pond. Well clusters O and P are located 55 and 125 feet east of the pond, respectively. Each well cluster was opened to 3, 5, 7, and 10 feet below land surface, and water levels were used in determining the rate of lateral migration. The upper limits for background ground-water quality were set at two standard deviations to represent about 95 percent of the observed data. The upper limits for concentrations of ammonia nitrogen and potassium were set at 2.0 and 3.0 mg/L (table 2), respectively. Any concentrations greater than 2.0 and 3.0 mg/L for ammonia nitrogen and potassium, respectively, were considered to reflect the effect of the leachate-laden north oxidation pond on the ground water. Data from well cluster D were not used because leakage may be occurring through the well annulae.

The concentrations of ammonia nitrogen in water from well C1 (1 foot below the bottom of the pond) increased within a month after the leachate was added to the pond (fig. 16) and continued to increase throughout the period of study. Concentrations of ammonia nitrogen in water from well C2 (3 feet below the bottom of the pond) showed a similar rate of increase as C1 after January 1980. The reduction in ammonia nitrogen concentrations between March 1979 and December 1979 probably reflects a natural decline in concentration between the two sample periods. The concentrations in water from well C2 remained lower than those observed in water from well C1 throughout the study. Based on the interception of the graph for well C2 with the background ground-water quality average concentration baseline, it took about 4 months from the date leachate was added to the pond for the leachate to migrate vertically 3 feet.

The peaks in concentrations of ammonia nitrogen observed in water from well C3 (5 feet below the bottom of the pond) may reflect short circuiting through the annular space of the well since the peaks (fig. 16) coincide with the high ammonia levels in the pond (fig. 16). Water from well C4 (7 feet below the bottom of the pond) did not show any effects of the leachate migration from July 1979, the date of first pumping, to July 1980, the date of last sample.

The initial ammonia nitrogen levels in water from well O1, 3 feet below land surface (fig. 17), probably reflects the effect of decaying vegetation on the land surface. The decrease in concentrations of ammonia nitrogen between September 1979 and January 1980 is probably due to the effects of clearing vegetation and flushing from rainfall. Concentrations appear to be increasing in

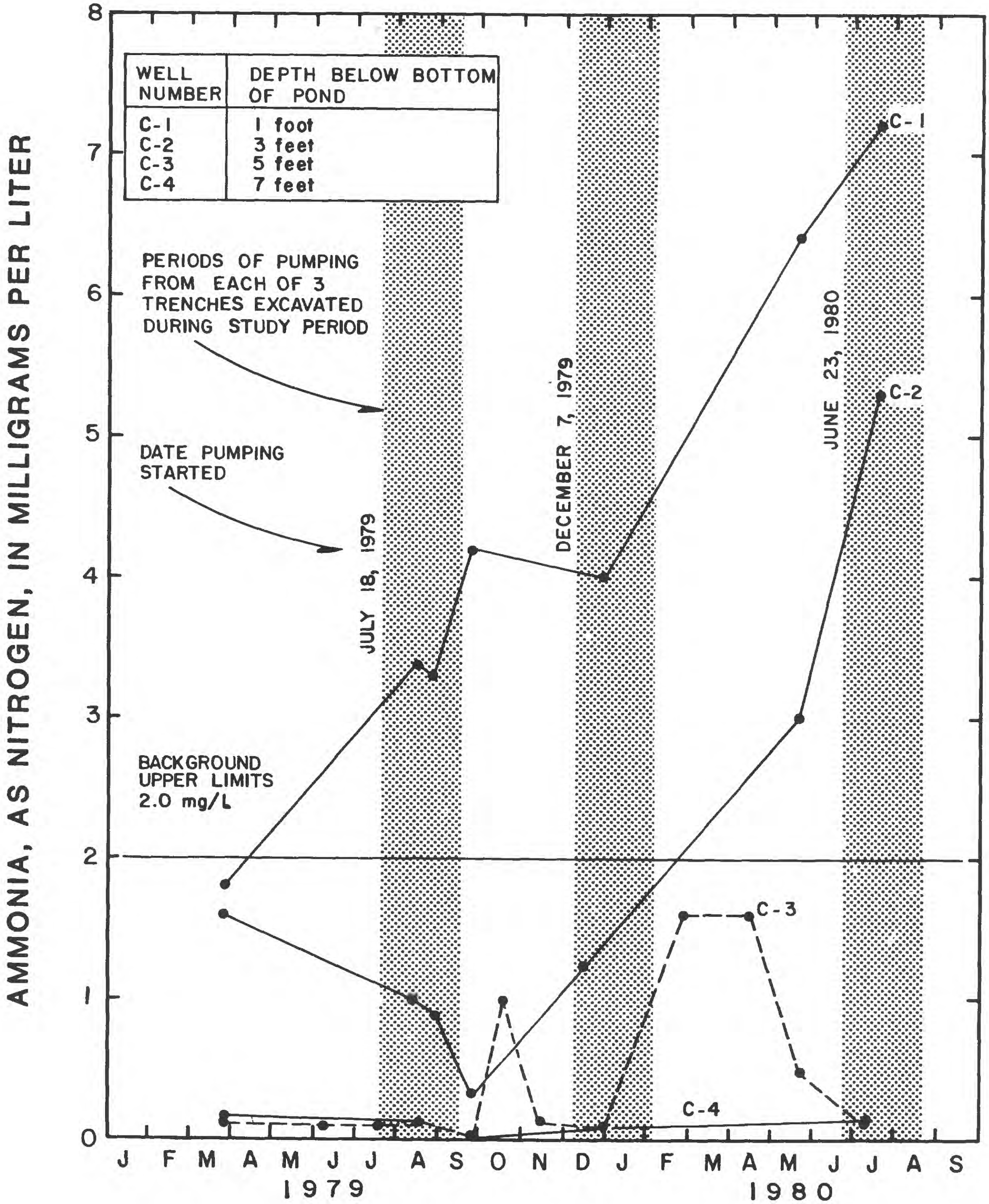

Figure 16.--Concentrations of ammonia nitrogen in water from cluster wells C1 through C4.

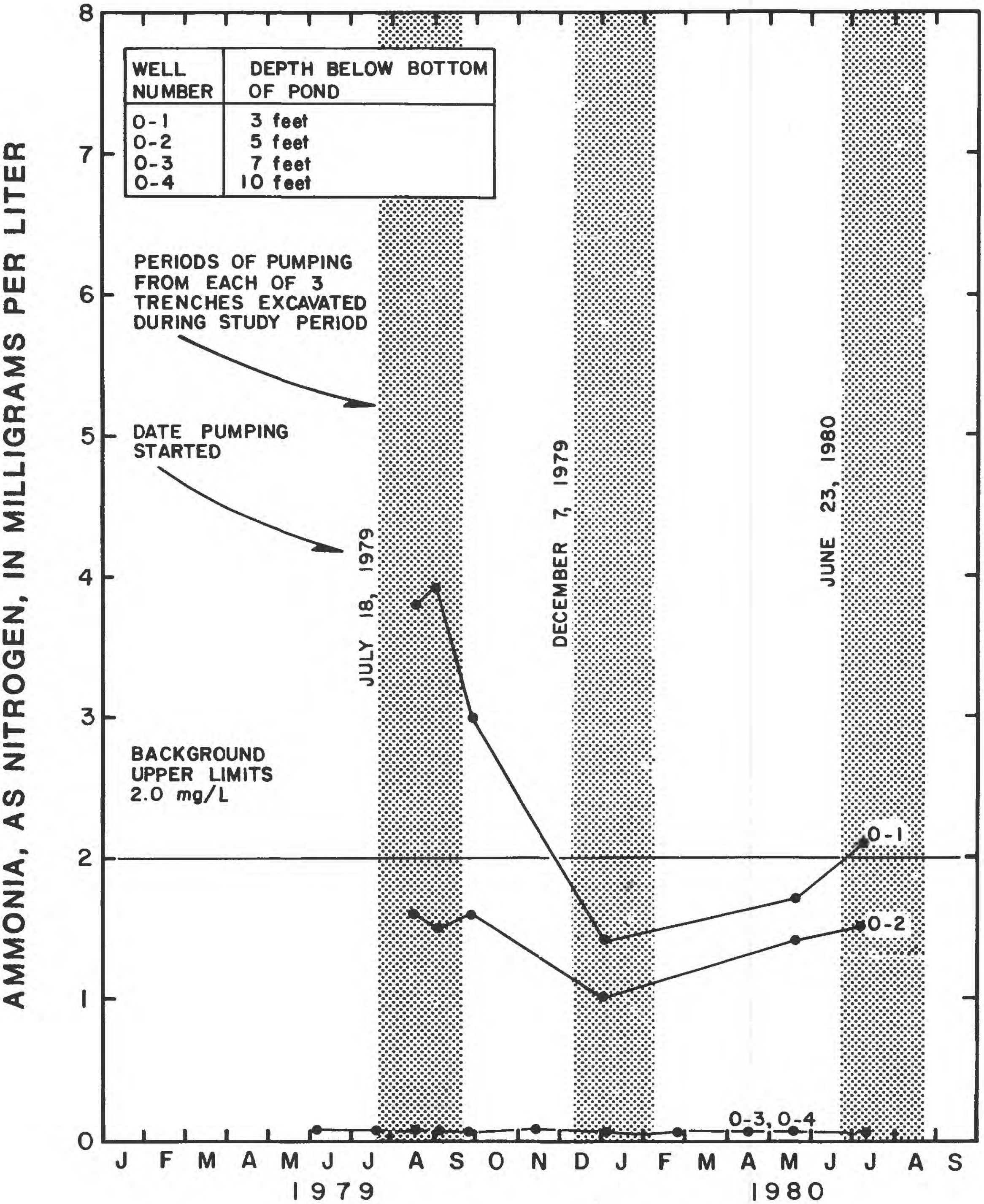

Figure 17.--Concentrations of ammonia nitrogen in water from cluster wells 01 through 04.

water from wells 01 and 02 starting in January 1980, probably reflecting movement of ammonia nitrogen through that strata. Water from wells 03 and 04, 7 and 10 feet below land surface, respectively, do not show any effects of leachate migration.

A section between well cluster C and 28th Street North (fig. 4) that shows concentrations of ammonia nitrogen at various depths on July 8, 1980, is presented in figure 18. As shown, the treated leachate, which had a range in concentration of NH_4-N of 0.01 to 16 mg/L during the study, had migrated about 75 feet from the pond in the upper 5 feet of the surficial aquifer between July 1979 and July 1980. Further migration will probably be along this strata with eventual partial interception by ditch A. Ditch A drains into a system of drainage ditches and retaining ponds that eventually empty into Old Tampa Bay. Figure 18 shows that the extent of vertical migration of the ammonia nitrogen below the pond after 1 year is about 4 feet.

The concentrations of potassium in water samples from well C1 exceeded background limits within 4 months after the treated leachate was ponded, reached a peak in January 1980, and then decreased slightly (fig. 19). Potassium concentrations in water from well C2 were less than the maximum expected limit (mean plus two standard deviations) of 3.0 mg/L for background water quality until February or March 1980, about 7 months after leachate had been added to the pond. In water from wells C3 and C4, an increase in concentrations of potassium occurred between January and May 1980; however, the magnitude of change is within the expected maximum range of background levels.

There appears to be an increasing trend in potassium concentration in well 01 (fig. 20) after December 1979, about 6 months after leachate was introduced into the pond. The analysis of water from well 02 reflects the same increasing trend between January 1980 and May 1980 as water from well 01; however, all values are less than the expected maximum background level (fig. 20). Water samples from wells 03 and 04 show little, if any, effects from the leachate ponding; variations in concentrations probably reflect natural variations in ground-water quality.

Variations in potassium concentrations with depth between well cluster C and 28th Street North are presented in figure 21. As shown, potassium had migrated about 80 feet from the edge of the oxidation pond along the upper 5-foot strata of the surficial aquifer between July 1979 and July 1980. Further migration will probably continue along this strata and eventually be partially intercepted by ditch A along 28th Street North. Vertical migration for potassium after about 1 year is about 4 feet.

A modified version of the Darcy equation was used to compute the velocity and time required for the north oxidation pond water to migrate an observed distance through the surficial aquifer:

$$V_h = \frac{K_h I}{N_e}$$

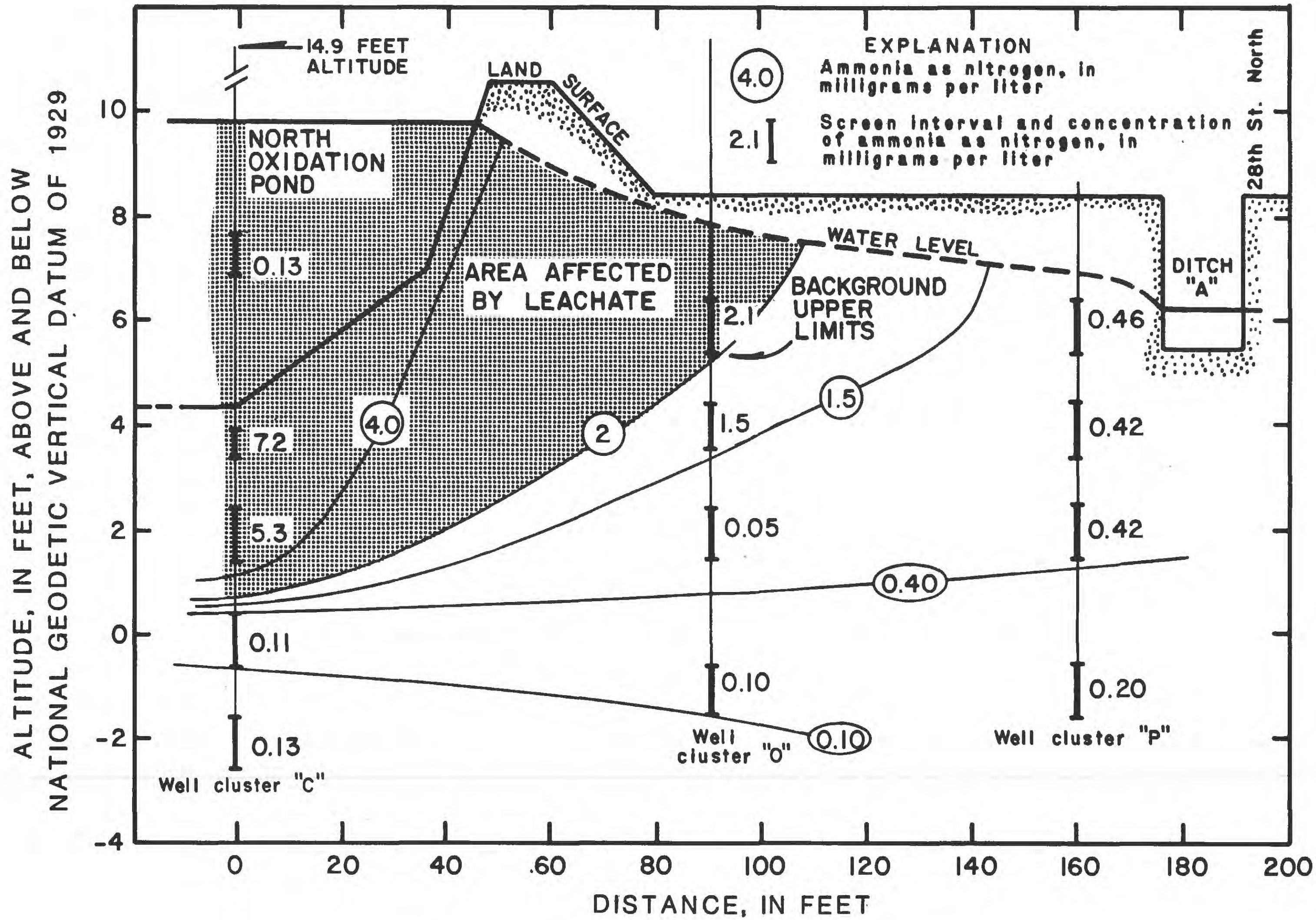

Figure 18.--Section of surficial aquifer showing concentrations of ammonia as nitrogen, July 8, 1980. (See figure 4 for locations of well clusters.)

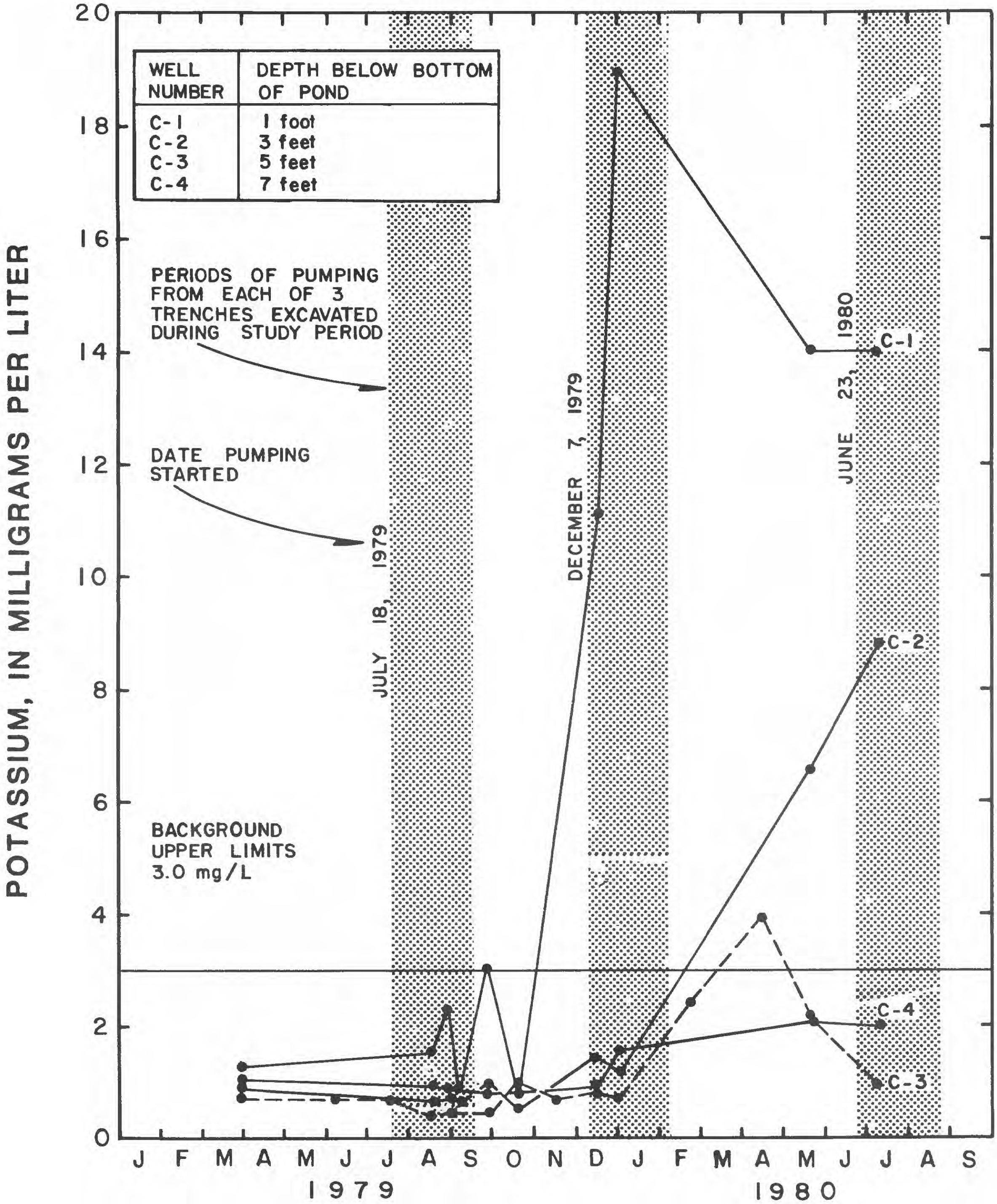

Figure 19.--Concentrations of potassium in water from cluster wells C1 through C4.

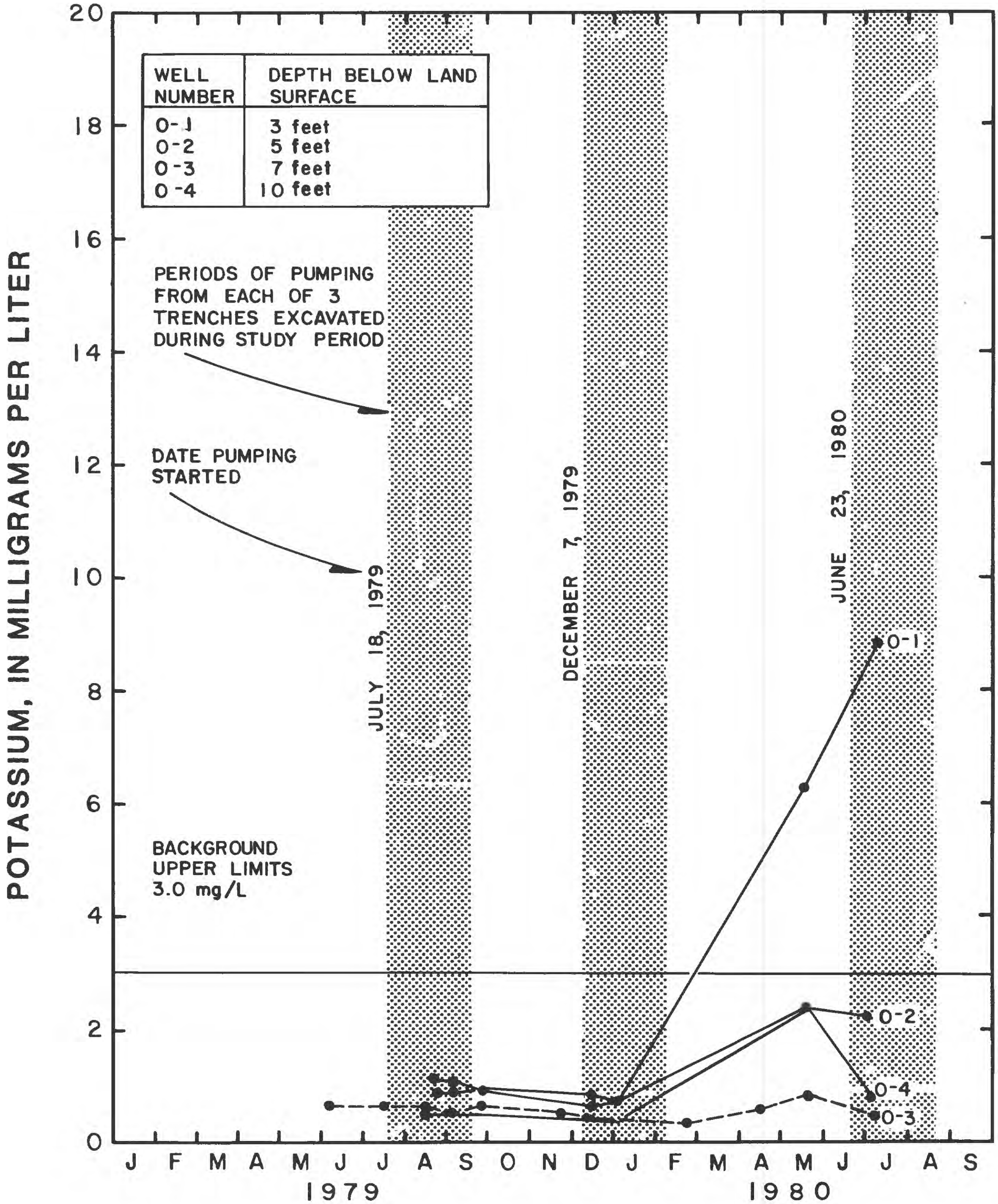

Figure 20.--Concentrations of potassium in water from cluster wells 01 through 04.

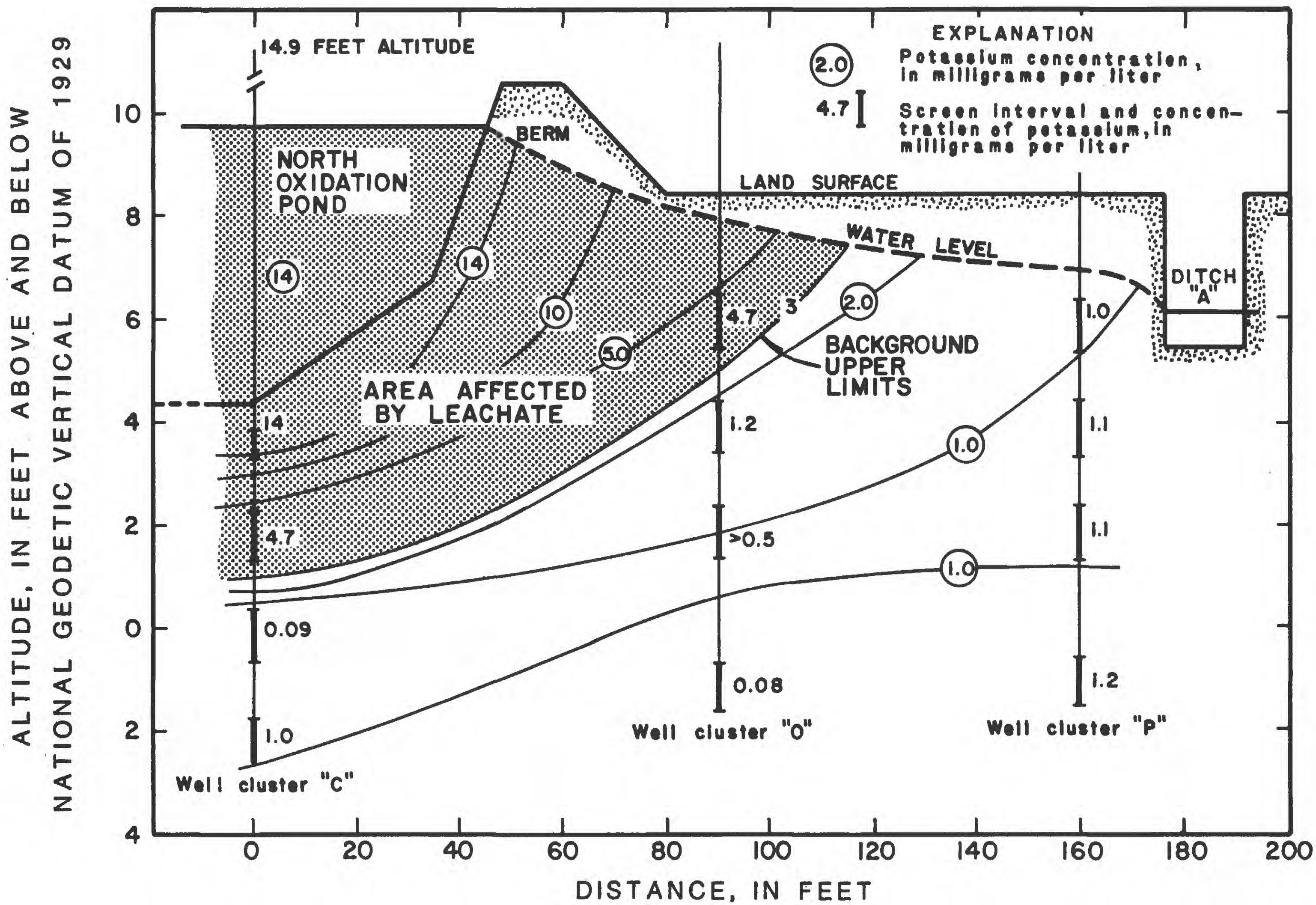

Figure 21.--Section of surficial aquifer showing concentrations of potassium, July 8, 1980. (See figure 4 for locations of well clusters.)

where V_h = horizontal velocity, in feet per day;

K_h = horizontal hydraulic conductivity, in feet per day;

I = hydraulic gradient, in feet per foot; and

N_e = effective porosity, dimensionless.

$$T = \frac{L}{V_h}$$

where T = estimated leachate traveltime, in days; and

L = distance leachate was observed to travel, in feet.

Values of hydraulic conductivity and effective porosity, 3.3 ft/d and 0.33, respectively, for similar strata, were previously reported for the study area (Fernandez, 1982, p. 12). Hydraulic gradients and horizontal velocities between the north oxidation pond and well cluster O and between well clusters O and P for periods of high and low water levels in the pond, January 24 and December 1, 1980 (figs. 8 and 12), respectively, are presented in figure 22. During high water, the rate of ground-water movement between the pond and well cluster O and between well clusters O and P were 0.48 and 0.14 ft/d, respectively. During low water levels, the rates were 0.12 and 0.03 ft/d, respectively. Darcy's equation was used to estimate the leachate traveltime based on the assumption that ammonia nitrogen and potassium behaved as conservatives and were not adsorbed by the clays found in the sands (Fernandez, 1982, p. 11).

If ground-water levels remain fairly constant at the level shown for January 24, 1980 (fig. 8), ground water moving through the surficial aquifer from the pond could be expected to take about 120 days to reach well cluster O and arrive at well cluster P in an additional 500 days, for a total of 620 days traveltime. Well cluster P is within 20 feet of ditch A, the east boundary of the site. In most cases, a leachate will move through an aquifer at a slower rate than ground water. The leachate tends to undergo chemical reactions and is absorbed in or adsorbed on the material through which it passes, all of which tend to slow the rate of movement of the leachate relative to ground water.

The computed traveltime of the leading edge of the leachate plume to well cluster P, based on the observed distances traveled for ammonia nitrogen and potassium (figs. 18 and 21), are 600 and 560 days, respectively. The computations were based on observed distances traveled for a time period of 355 days, beginning with the introduction of leachate into the pond on July 18, 1979, and ending July 8, 1980, with the final sampling for water quality. The possible reasons that the traveltime is somewhat shorter when compared to results using Darcy's equation are (1) the Darcy equation assumes that the aquifer is homogenous and isotropic, (2) physical properties of nearby soils (hydraulic conductivity and effective porosity) used in computations are not the same as those at the pond site, and (3) the computations do not consider the possible effects of adsorption by clays and absorption of ammonia nitrogen and potassium by plant root systems.

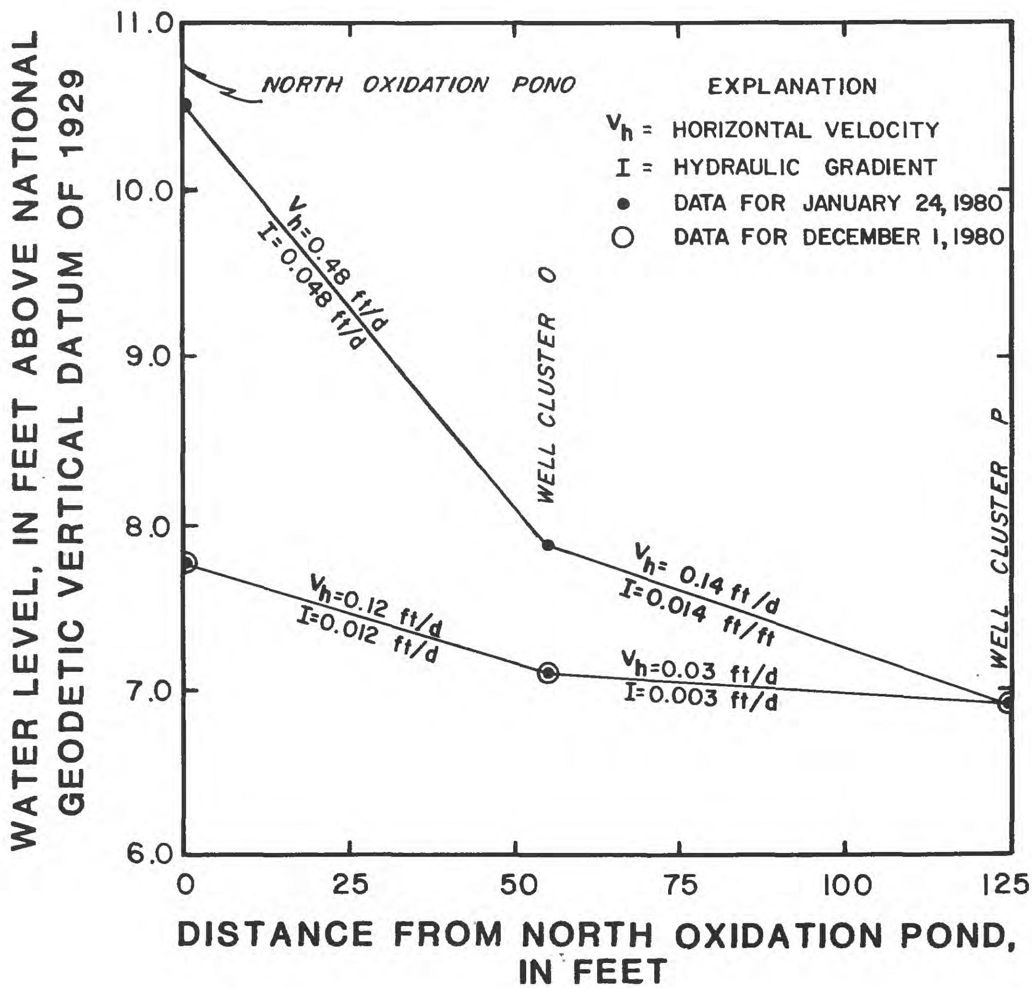

Figure 22.--Horizontal velocities and hydraulic gradients between north oxidation pond, well cluster O, and well cluster P.

SUMMARY AND CONCLUSIONS

The Bridgeway Acres landfill receives about 800 tons of refuse per day that is disposed of by the trench and high-rise methods of landfilling. The high water table at the site requires dewatering of the operating trenches. The Pinellas County Department of Solid Waste Management developed a treatment method whereby trench water was pumped into a lined aeration basin, with a 2-day detention, and then discharged through gravity flow into an unlined stabilization pond for further treatment and eventual disposal by evaporation and infiltration to the surficial aquifer.

Eighty test holes, commonly in clusters of four wells at different depths, were constructed into the surficial aquifer for water-level measurements. Selected clusters were used for water-quality monitoring. Well depths ranged from 1 to 10 feet and screen lengths ranged from 0.5 to 1 foot.

Three stratigraphic units underlie the study area: (1) undifferentiated surficial deposits of Pleistocene age that compose the surficial aquifer and average 19 feet in thickness; (2) the Hawthorn Formation of middle Miocene age that underlies the surficial aquifer composes the confining layer and is about 10 feet thick; and (3) the Tampa Limestone of early Miocene age that is the upper part of the Floridan aquifer and averages about 180 feet in thickness.

Localized changes of water levels in the surficial aquifer were observed when levels in the oxidation ponds changed. A mounding effect due to storage and leakage of water from the ponds results in movement of water away from the ponds in all directions.

Results of a bench-scale test in which raw leachate was aerated showed that the specific conductance (used as a reflectance of dissolved solids) was reduced by about 46 percent after about 20 hours of aeration. The aeration of fresh leachate under operating field conditions reduced the dissolved solids by about 41 percent.

Analysis of samples from the aeration basin and north oxidation pond indicated that measureable nitrification of ammonia nitrogen did not occur in either system. What probably occurred was the deamination or ammonification of the organic nitrogen by saprophytic bacteria. Potassium, a conservative substance, was used as an indicator of movement throughout the entire treatment process.

Water samples were collected from the south oxidation pond and surface-water site SW-2 and used as control for analyses of phytoplankton, chlorophyll a, total coliforms, and fecal coliforms. Although populations of phytoplankton appeared to be higher in the control pond (south oxidation pond) than in the north oxidation pond in November 1979, there is probably no difference biologically between the control and north oxidation ponds when the total number of taxa and the diversity are considered. In the August 1980 sampling when the pond at SW-2 was the control pond, the population density at the north oxidation pond exceeded that of the control pond by a factor of 10. Concentrations of chlorophyll a were significantly higher in the north oxidation pond that received leachate than in the control pond in both November 1979 and August 1980. The higher concentrations of chlorophyll a in the north oxidation pond than in the control pond indicate that the leachate pond supported a larger algal biomass and probably more algal productivity than did the control pond because of a high concentration of nutrients.

Analysis of raw leachate from an active trench for coliforms and fecal streptococci indicated that total coliform, fecal coliforms, and fecal streptococci averaged about $3.1x10^6$, $1.3x10^6$, and $8.2x10^5$ col/100 mL, respectively. The average FC/FS ratio of 2.6 for the raw leachate indicated some effect from waste of warm-blooded animals (soiled baby diapers). Analysis of treated leachate from the aeration basin for total and fecal coliforms and fecal streptococci averaged $1.8x10^6$, $6.7x10^5$, and $1.2x10^5$ col/100 mL, respectively. The average FC/FS ratio was 5.6. Analysis of water samples from the north oxidation pond showed that total coliforms ranged from 120 to 180,000 col/100 mL. The 120 col/100 mL reflects the initial stage of the pond prior to receiving leachate. An increase in the FC/FS ratio in the pond indicates that an increase has occurred in the quantity of warm-blooded animal waste in the pond.

The chemical constituents, ammonia nitrogen and potassium, were also used as indicators of migration of the leachate in the aquifer from the ponded, treated leachate. The upper limits for background ground-water quality were set at two standard deviations to represent 95 percent of the observed data. The upper limits for concentrations of ammonia nitrogen and potassium were set at 2.0 and 3.0 mg/L, respectively. Any concentrations greater than 2.0 and 3.0 mg/L for ammonia nitrogen and potassium, respectively, were considered to reflect the effect of the leachate-laden north oxidation pond on the ground water.

Ground-water quality beneath and near the pond appears to have been affected by infiltration and percolation of treated leachate through the surficial aquifer. Analyses indicate that between 6 months and 1 year was required for treated leachate to move 4 feet vertically beneath the pond into the surficial aquifer and to migrate horizontally between 75 and 80 feet along the upper 5 feet of the surficial aquifer. During high-water levels in the pond (January 24, 1980), the rates of ground-water movement between the pond and well cluster O and between well clusters O and P were 0.48 and 0.14 ft/d, respectively; during low-water levels in the pond (December 1, 1980), the rates were 0.12 and 0.03 ft/d, respectively. The rate of leachate movement from the pond is dependent on the water levels in the pond and ground water and the physical properties of the soil.

Darcy's equation was used to estimate the leachate traveltime based on the assumption that ammonia nitrogen and potassium behaved as conservatives and were not adsorbed by the clays found in the sands. The computed traveltime of the leading edge of the leachate plume to well cluster P, based on the observed distances traveled for ammonia nitrogen and potassium, are 600 and 560 days, respectively. The observed distances were for a time period of 355 days, beginning with the introduction of leachate into the pond on July 18, 1979, and ending July 8, 1980, with the final sampling for water quality. The possible reasons that the traveltime is somewhat shorter when compared to results using Darcy's equation are (1) the Darcy equation assumes that the aquifer is homogenous and isotropic, (2) physical properties of nearby soils (hydraulic conductivity and effective porosity) used in the computations are not the same as those at the pond site, and (3) the computations do not consider the possible effects of adsorption by the clays and absorption of ammonia nitrogen and potassium by the plant root systems.

REFERENCES

Barr, G. L., and Fernandez, Mario, Jr., 1981, Water-quality data from a landfill leachate treatment and disposal site, Pinellas County, Florida, January 1979-August 1980: U.S. Geological Survey Open-File Report 81-1064, 76 p.

Beetem, W. A., Freedman, L. C., Perryman, G. R., and Watterson, C. A., editors, 1981, Water quality laboratory services catalog: U.S. Geological Survey Open-File Report 80-1279.

Boney, A. D., 1976, Phytoplankton, in The Institute of Biology's Studies in Biology no. 52: London, Edward Arnold, 116 p.

Buono, Anthony, and Rutledge, A. T., 1978, Configuration of the top of the Floridan aquifer, Southwest Florida Water Management District and adjacent areas: U.S. Geological Survey Water-Resources Investigations Open-File Report 78-34, 1 sheet.

Eckenfelder, W. W., 1966, Industrial water pollution control: New York, McGraw Hill, 275 p.

Fernandez, Mario, Jr., 1979a, Ground-water quality data from a sludge disposal test site, St. Petersburg, Florida, November 1973-July 1977: U.S. Geological Survey Open-File Report 78-821, 101 p.

______ 1979b, Water-quality data from a landfill, Pinellas County, Florida, May 1975-October 1977: U.S. Geological Survey Open-File Report 78-822, 97 p.

______ 1982, Hydrogeology of a landfill operation, Pinellas County, Florida: U.S. Geological Survey Water-Resources Investigations 82-30, 41 p.

Florida Division of Health, 1972, A handbook for sanitary landfills in Florida —Supplement to solid waste management plan for Florida: Florida Department of Health and Rehabilitative Services publication.

Geldreich, E. E., 1966, Sanitary significance of fecal coliforms in the environment: U.S. Department of the Interior, Federal Water Pollution Control Administration WP-20-3, p. 101.

Gloyna, E. F., 1968, Basis for waste stabilization pond design, in Gloyna, E. F., and Eckenfelder, W. W., Jr., eds., Advances in water quality improvement: University of Texas Press, 513 p.

Heath, R. C., and Smith, P. C., 1954, Ground-water resources of Pinellas County, Florida: Florida Geological Survey Report of Investigations 12, 122 p.

Hem, J. D., and Skougstad, M. W., 1960, Coprecipitation effects in solutions containing ferrous, ferric and cupric ions: U.S. Geological Survey Water-Supply Paper 1459-E, p. 95-110.

Hickey, J. J., 1982, Hydrogeology and results of injection tests at waste-injection test sites in Pinellas County, Florida: U.S. Geological Survey Water-Supply Paper 2183, 42 p.

Hutchinson, C. B., and Stewart, J. W., 1978, Geohydrologic evaluation of a landfill in a coastal area, St. Petersburg, Florida: U.S. Geological Survey Water-Resources Investigations 77-78, 40 p.

National Oceanic and Atmospheric Administration, Climatological data, annual summary, Florida, 1941 through 1980.

SAS Institute, Inc., 1979, SAS user's guide: North Carolina, 464 p.

Sawyer, C. N., and McCarty, P. C., 1978, Chemistry for environmental engineering, 3rd edition: New York, McGraw-Hill, 532 p.

Skougstad, M. S., Fishman, M. J., Friedman, L. C., Erdman, D. E., and Duncan, S. S., eds., 1972, Methods for determination of inorganic substances in water and fluvial sediments: U.S. Geological Survey Techniques of Water-Resources Investigations, Chapter Al, Book 5, 626 p.

Sorg, T. V., and Hickman, H. L., Jr., 1970, Sanitary landfill facts: U.S. Public Health Service Publication 1792, 30 p.

U.S. Department of Agriculture, Soil Conservation Service, 1972, Soil survey of Pinellas County: p. 10.

U.S. Environmental Protection Agency, 1975, Compilation of methodology used for measuring pollution parameters of sanitary landfill leachate: Ecological Series, EPA-600/3-73-011, 174 p.

U.S. Geological Survey, 1973a, Map of flood-prone areas, Safety Harbor Quadrangle, Florida: Tallahassee, Florida, U.S. Geological Survey, WRD.

______ 1973b, Map of flood-prone areas, St. Petersburg quadrangle, Florida: Tallahassee, Florida, U.S. Geological Survey, WRD.