EPA 600/3-84-97

EFFECT OF DREDGING
LILLY LAKE, WISCONSIN

By

Russell C. Dunst and James G. Vennie
Wisconsin Department of Natural Resources
Madison, Wisconsin 53707

and

R. B. Corey and A. E. Peterson
Department of Soil Science
University of Wisconsin
Madison, Wisconsin 53701

R804875

Project Officer
Spencer A. Peterson
Environmental Research Laboratory
Corvallis, Oregon 97333

ENVIRONMENTAL RESEARCH LABORATORY
OFFICE OF RESEARCH AND DEVELOPMENT
U.S. ENVIRONMENTAL PROTECTION AGENCY
CORVALLIS, OREGON 97333

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ABSTRACT

Lilly Lake is located in southeastern Wisconsin. It has a surface area of 37 ha and in 1977 had a maximum depth of 1.8 m and a calculated infilling rate of 0.5 cm per year. The basin contained up to 10.7 m of light-weight, organic sediments. Recreational activity was severely restricted due to periodic winter fish kills and dense growths of macrophytes throughout the summer. During the open water periods of 1978 and 1979, 683,900 m^3 of sediment were removed with a 30-cm cutterhead dredge and transported via pipeline to two disposal sites. The dredging operation deepened the lake to a maximum of 6.6 m and afforded an excellent opportunity to evaluate the inlake and disposal site effects of the project.

The inlake portion of the investigation included an assessment of water quality, aquatic biology, sediments, and hydrology before, during, and after completion of dredging. The evaluation of sediment disposal emphasized the impact on the nearby groundwater system and the value of using hydrosoils to enhance agricultural crop production. The study began in July, 1976 and extended through 1981, with some work continuing into the summer of 1982.

This report was submitted in fulfillment of grant number R804875 to the Wisconsin Department of Natural Resources under the sponsorship of the U.S. Environmental Protection Agency.

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ACKNOWLEDGEMENTS

Many people provided valuable contribution to this investigative effort. Rick Beauheim was instrumental in establishing the groundwater monitoring program and in analyzing the information associated with lake hydrology. Dave Janes was responsible for coordinating the data collection team and tabulating the information for the inlake and sediment disposal portions of the project. Sampling assistance was primarily provided by Debbie Young, Muriel Guy, and Jody Parent. The sampling and data manipulation activities associated with the greenhouse and field crop studies were performed by Joseph Tuschl, Thomas Wright, and particularly, Bradley Russell. Analytical services were furnished by the Wisconsin State Laboratory of Hygiene (Bob Schuknecht, George Bowman and their staff). In addition, the U.S. EPA Project Officer, Spencer Peterson, supplied much helpful advice throughout the duration of the project.

SUMMARY

Lilly Lake is a 37 ha, natural, seepage lake located in southeastern Wisconsin. In the early 1970s, it was 1.8 m deep and infilling at a rate of 0.5 cm/yr. The basin contained up to 10.7 m of light-weight, organic sediments, with a water content of 90-98%. During July-October, 1978 and May-August, 1979, 683,900 m^3 of sediment were removed with a 30-cm cutterhead dredge, deepening the lake to 6.6 m. Inlake water quality, aquatic biology, sediment characteristics, and hydrology were investigated before, during, and after completion of the project (1976-82).

Before dredging began, winter fish kills were common and rooted macrophytes grew over the entire basin, reaching the surface in most areas. The fish population was dominated by slow growing bluegills, and natural reproduction of the gamefish was poor. If oxygen resupply/production would have been eliminated during the winter, dissolved oxygen concentrations would have approached zero within 25 days. Water quality was satisfactory in the summer. Chlorophyll a levels averaged under 5 ug/L, and at least 22 genera of algae were present. Chlorophyceae were dominant in terms of numbers and biomass. Macrophytes were represented by 12 submergent species, with a combined lakewide weighted average biomass of 685 and 335 g/m^2 in 1976 and 1977, respectively. These plants strongly influenced inlake dynamics and were a major problem in recreational usage of the lake. In 1977 groundwater inflow contributed 17 and 2% of the water and phosphorus loading to the lake, respectively.

During the dredging activity, lake levels dropped 1.5 m. This produced an increased groundwater inflow, including a halt in the outflow, with flow reversal in all of the previous outflow areas. As a result inlake water chemistry showed more similarity to that of the groundwater, illustrated by increases in conductivity and total alkalinity. Because of sediment disturbance, there were also increases in ammonia nitrogen, total phosphorus, and biological oxygen demand (5 day) levels. However, inlake dissolved oxygen concentrations were not affected, and, in general, water quality remained within tolerable limits. The fish population did not appear to be stressed by the prevailing conditions. Initially the algal population increased, peaking at 33 ug/L of chlorophyll a. Changes were also measured for gross primary productivity, numbers of algal cells, and total biomass; however, the population did not undergo any major change in diversity or relative composition. Apparently in response to the algal expansion, the zooplankter _Bosmina longirostris_ exhibited greatly increased numbers during the same period.

Since completion of the dredging project, the lake has refilled to pre-treatment levels. Magnesium, sodium, potassium, chloride, conductivity, and total alkalinity were higher while total phosphorus was lower as compared to earlier concentrations. In 1981 sediment oxygen demand and ammonia nitrogen release rates were similar to 1977. Nevertheless, the inlake water storage was increased by 2.3X, and winter dissolved oxygen levels remained above 7 mg/L at all depths. The ammonia nitrogen was apparently being quickly converted to other nitrogen forms (as in 1977), because only insignificant levels were found in the water column. Soluble reactive phosphorus release into the water column was minimal due to the oxic conditions, although it would become a significant phosphorus source under anoxia. In 1981 the lake was still polymictic, but thermal gradients up to 8°C were present on occasion. Low dissolved oxygen levels were found near the bottom during these periods of reduced mixing. Bottom temperatures reached 24°C; however, summer maxima were depressed at all depths due to the increased water volume.

By 1981 average summer chlorophyll a levels were again under 5 ug/L. Algal diversity and density were similar to 1978 (before dredging), but there was a shift in species composition, represented by the appearance of six genera of Chrysophyceae. These were co-dominant with the Chlorophyceae in terms of biomass. The benthic invertebrate community was relatively sparse both before as well as after dredging; however, numbers and diversity were greater in 1981. The populations of *Hyalella azteca* and *Pericoma* sp. were down, but Oligochaetes, *Caenis* sp., and several genera of chironomids responded to the improved inlake conditions. Many of these are known to be good fish food organisms. Dredging and drawdown was a major disruption for the rooted submergent vegetation. In 1980 species diversity was reduced to six, and the lake bottom was inhabited primarily by a macrophytic algae, *Chara* sp. This is often a pioneer species for new habitat. By 1982 the *Chara* sp. was being replaced by rooted macrophytes, and the diversity of species had increased somewhat. The plant community was still evolving, but the existing characteristics included: 1) growth over 75% of the lake area, to a depth of 3.7 m, 2) top 1.2-1.8 m of the water column weed-free, except in the shallow, near shore area, 3) lakewide weighted average biomass under 100 g/m^2, and 4) lower biomass over sand versus muck bottom. The lower biomass over sand was also true before dredging but is important because of the greatly enlarged area of sand bottom post-treatment.

Although dredging removed the phosphorus-rich upper sediments, resulting in a major reduction in most phosphorus forms (especially sodium hydroxide extractable phosphorus), the sediments were still the most important reservoir of phosphorus within the lake. However, phosphorus transport directly into the water column was minimal, based on a combination of measurement and predictive equation. Groundwaters furnished 46% of the water loading in 1981 but only 9% of the phosphorus. Several models were used with the water and phosphorus loadings before and after dredging to predict a chlorophyll a level for the lake. All of the models suggested that low chlorophyll a concentrations would be present in the lake, correlating well with actual measurements.

The lake sediments were transported via pipeline to two disposal sites. The primary site, a modified abandoned gravel pit, was used in 1978 and 1979, eventually receiving about 540,000 m^3 of sediment. The remainder of the sediments were placed in a low diked area on agricultural land. This site was used in 1979 only. Changes to the groundwater systems were monitored with observation wells at both sites plus private drinking water wells at the gravel pit. Parameters of interest at both sites included water levels, nitrate plus nitrite nitrogen, ammonia nitrogen, pH, and conductivity. In addition, chemical oxygen demand was measured at the diked area and the following at the gravel pit site: chloride, total organic nitrogen, total dissolved phosphorus, total phosphorus, arsenic, barium, cadmium, copper, iron, lead, mercury, selenium, silver, zinc, and chromium.

The results were similar at each of the disposal sites. The response time for a particular well was related to soil permeability and distance from the diked area or gravel pit basin. The greatest impact was observed in wells located adjacent to the sites, especially where permeable soils were present. Impact duration was short because the lake sediments quickly inhibited continued seepage of water away from the sites. In general, the surrounding groundwater systems were not affected significantly as a result of lake sediment disposal. In 1983 the modified gravel pit was still retaining water; however, the sediments at the diked area were landspread, dried, and incorporated into the terrestrial soils during 1980-81. By 1983 the area was growing corn.

Laboratory, greenhouse and field studies, begun in 1980 and terminated in 1982, were set up to determine the effects of applications of sediment from the lake on agricultural crops. The laboratory studies included a survey of sediments from 11 other Wisconsin lakes, and the greenhouse study included three additional sediments so that chemical properties and plant responses to this sediment could be compared with chemical properties and crop responses from other sediments.

The sediment survey included analyses of the 12 sediments for pH, total carbon, chemical oxygen demand, loss on ignition, total nitrogen, ammonia nitrogen, total phosphorus, organic phosphorus, phosphorus extracted with 0.5M sodium bicarbonate, phosphorus equilibrated in 0.01M calcium chloride, and total zinc, manganese, copper, cadmium and lead. A study of nitrogen and phosphorus released or immobilized on incubation was also included. Lilly Lake sediment had the highest pH, total carbon, chemical oxygen demand, loss on ignition, total nitrogen and nitrogen release on three month's incubation. It also showed the greatest decrease in phosphorus equilibrating in calcium chloride after incubation. The sediments showed wide ranges in many of the factors, particularly in those associated with pH and organic matter content.

The four sediments used in the greenhouse study were selected to give wide ranges in pH (5.1 to 7.7), chemical oxygen demand (3.5 to 28.7 mg/L), total nitrogen (0.35 to 2.69 mg/L) and carbon/phosphorus ratio (64-1420). Sediment concentrations equivalent to 0, 10, 30, 90 and 270 mT/ha were established in 1.5 dm^3 pots of Plainfield sand and Withee silt loam. Corn was planted and harvested after eight weeks.

Factors investigated in the greenhouse experiment included yield, tissue concentrations of nitrogen, phosphorus, potassium, calcium, magnesium, sulfur, iron, manganese, copper, zinc, boron, and plant uptake of nitrogen, phosphorus and potassium. Statistical analyses showed highly significant differences associated with the soil used for all factors except yield and copper concentration. Differences associated with sediment source were significant at the 5% level or greater for all factors except phosphorus, potassium and boron concentrations. However, only concentrations and uptakes of nitrogen showed differences that were great enough to be important from a practical standpoint. The New Richmond Flowage sediment did increase the zinc concentration in the plant tissue more than the other sediments, but the concentration was not excessive.

Although there was a sizable, significant variation in nitrogen concentration and uptake associated with the different sediments, there was no effect of application rate beyond the first increment. Application rate did significantly affect calcium, magnesium, sulfur and manganese concentrations, but except for manganese the effects were not great. For manganese, increasing rates of high pH Lilly Lake sediment decreased manganese concentrations in the plant while increasing rates of the acid Lake Tomah sediment increased them.

The field studies were set up on a Fox silt loam in 1980. Lilly Lake sediment additions equivalent to 0, 22.4, 44.8 and 89.6 mT/ha of dry sediments were made. There were four replicates in a latin square design. Sudan grass was planted and was harvested twice. Even though the site was an old alfalfa field, significant yield and nitrogen uptake responses over the control were obtained in both harvests. There were no significant differences between different rates of sediment application.

The field experiment of 1981 and 1982 was on a Hebron silt loam at a different site because sediment from an adjoining lagoon was spread over the original site. The same experimental design was used at the new site which was previously in pasture, and corn was grown both years. No significant differences in yield or in nitrogen or phosphorus concentrations or uptakes occurred in 1981. Grain yields could not be obtained in 1982 because of cow damage, but concentrations of the following elements were determined in ear leaves at silking and in the corn grain: nitrogen, phosphorus, potassium, calcium, magnesium, sulfur, zinc, iron, copper, aluminum, manganese, cobalt, arsenic, cadmium and lead. The only significant effect associated with sediment addition was an increase in the nitrogen concentration in the grain at the highest sediment rate.

Both the greenhouse and field results suggest that, of the factors measured, the only beneficial effect to crops from the application of Lilly Lake sediment would be an increase in nitrogen availability which would probably continue to be effective for a number of years. No harmful effects of sediment application were apparent.

INTRODUCTION

Lake aging is a naturally occurring, continual process which eventually leads to lake extinction (Hasler, 1947). Sediment accumulation will ultimately fill in the basin and convert it into a dry land environment. Thousands of years would normally be required for completion of this process; however, it can be greatly accelerated by man's activities.

A wide variety of watershed land use practices can markedly increase the sediment influx to a lake. Rapid infilling can, however, be caused by autochthonous, as well as allochthonous materials. In a lake with a small watershed, sediment influx may be minimal, but nutrients can enter via groundwaters, atmospheric deposition and/or surface runoff. Sufficient nutrient loading can support dense algal populations and/or macrophyte growths, with plant residue subsequently settling to the lake bottom. As organic materials accumulate, anaerobic conditions develop in the sediments and bottom waters. Due to the slower rate of anaerobic decomposition, there is an acceleration in the sediment infilling process. Decreased water depth is a problem in itself, and the shallower waters allow rooted vegetation to invade a greater share of the lake basin. Also, because of the increased oxygen demand, winter fish kills occur with more regularity. Lake use problems are intensified throughout the process, causing severe alterations and restrictions in recreation. Eventually, the open water areas of the lake will disappear.

Lilly Lake, Wisconsin was in an advanced stage of the aging process. Maximum depth of water was 1.8 m, with up to 10.7 m of underlying organic sediments. Recreational usage was severely restricted due to intense macrophyte growth over the entire basin and winter fish kills. In an effort to alleviate these problems a project was implemented in 1978-79 to deepen the lake by hydraulic dredging. The environmental objectives included:

1. elimination of both winter and summer fish kills;
2. elimination of about 80 percent of the dense macrophyte growth;
3. enhancement of fish habitat; and
4. enhancement of recreational usage, such as fishing, swimming, boating, and water skiing.

Implementation of this project presented an excellent opportunity to evaluate the inlake conditions before, during, and after rehabilitation. The physical effects of dredging are known and the various equipment, techniques and costs have been reviewed (Pierce, 1970), but little information is available concerning biochemical effects on lake environments (U.S. EPA, 1973 and Dunst <u>et al</u>., 1974). Therefore, an evaluation program was initiated in 1976 to determine the impact on water quality, sediment characteristics, hydrology, and biota. Monitoring encompassed two years following completion of dredging activities, extending into 1982.

Rehabilitation projects in Wisconsin's Inland Lake Renewal program have utilized various techniques such as aeration, dredging, drawdown, storm sewer diversion, alum treatment, aquatic macrophyte harvesting, improved animal manure handling, stream bank erosion control, and several upland conservation methods. However, dredging has been the primary technique. The most serious technical limitation for this approach has been the identification of adequate, low cost sediment disposal methods. Therefore, there is the need to establish guidelines for sediment disposal without environmental damage, and, ideally, to document beneficial uses for the sediments. The Lilly Lake project also furnished a good opportunity to explore this aspect of lake deepening. Consequently, the evaluation program included the monitoring of groundwater quality and quantity around both disposal sites and the suitability of these highly organic sediments for enhancing agricultural production.

Application of lake sediment to agricultural land, where feasible, would have a number of potential advantages. First, the nutrients, particularly nitrogen and phosphorus, in the sediment would be recycled and put to good use; second, the addition of organic matter to the soils should improve soil structure and increase water infiltration in fine to medium-textured soils and increase water-holding capacity in sandy soils; third, the extra water applied could act as supplemental irrigation in periods of available-water deficits.

The amount of sediment suspension that could be applied might possibly be limited by the rate and extent of nitrogen mineralization from the sediment but more probably by the ability of the soil to accept the extra water during the pumping period. Nitrogen is mineralized to ammonium and nitrate forms during the decomposition of an organic sediment, and groundwater contamination with nitrate in excess of 10 mg nitrogen per liter could result if the rate of mineralization greatly exceeded the rate at which nitrogen was taken up by the crop or lost by denitrification. Therefore, the maximum application rate might possibly be limited by excessive nitrogen availability as is the case with sewage sludge (Keeney _et al._, 1975).

It is unlikely that a crop could be grown successfully during the period of application if the sediment were applied directly from the lake. Therefore, the cost of the operation would have to include such crop losses. If a crop were grown, application rates would have to be controlled so that anaerobic conditions were not induced, and the permissible rates would depend on the permeability of the soil and effects of surface deposits on the plants. In any case, application rates would have to be matched to soil permeability so that surface runoff does not become a problem. Possible effects of the applied sediment on plugging soil pores and lowering infiltration rates would have to be determined to arrive at an acceptable rate of application.

There is little information available in the literature on the effects of lake sediments on soil properties and nutrient uptake by plants. Konrad _et at._ (1970) analyzed sediments of lakes Mendota and LaBelle and found the carbon:nitrogen ratios to be in the range of 8 to 12, and available forms of nitrogen quite high and increasing with deoth. Keeney _et al._ (1970) found the available nitrogen to be higher in sediments from eutrophic than oligotrophic lakes. These studies suggest that the lake sediments would be a good sustained source of nitrogen for crops.

The potential availability of phosphorus in sediments is much less certain than in the case of nitrogen. Wildung _et al._ (1974) found that sediments from three lakes in Oregon contained from about 600 to 1000 mg/L of total phosphorus with about half being inorganic and half organic. Under the aerobic conditions at the soil surface, the inorganic forms would probably be relatively unavailable. The carbon:phosphorus ratios of 160 to 200 reported by Wildung _et al._ (1974) would suggest a possible temporary immobilization of phosphorus by microorganisms during the decomposition process, and thus possibly a need for applying additional fertilizer phosphorus for good crop growth. The possible deficiency of phosphorus is not surprising, considering the fact that phosphorus is the element most often limiting algae growth in natural waters (Schindler, 1977).

The entire evaluation effort was supported largely through EPA Grant No. R804875, with sampling and interpretive activities performed by the Wisconsin Department of Natural Resources and the University of Wisconsin.

OBJECTIVES

The evaluation project encompassed three categories of interest - inlake conditions, sediment disposal, and usage of the sediments as a soil additive. The major objectives reflect these concerns:

1) to determine the degree of environmental disturbance caused by dredge operation in the lake,

2) to document the alterations in water quality, hydrology, sediments, and biota resulting from the deepened lake,

3) to measure the magnitude and duration of impact on groundwater levels and quality near the sediment disposal sites. Drinking water quality was also monitored in private wells near the major disposal site,

4) to compare the effects of applying sediments from Lilly Lake and 11 other Wisconsin lakes on soil properties and crop responses under laboratory and greenhouse conditions, and

5) to determine the agricultural crop response to the usage of Lilly Lake sediments as a soil additive under field conditions.

PROJECT DESCRIPTION

Lilly Lake is a natural, seepage lake located in southeastern Wisconsin possessing no surface inlets or outlets. The lake covers 37 ha and in 1977 had a mean depth of 1.4 m. Maximum water depth was 1.8 m, with up to 10.7 m of underlying organic sediments. The water content of the sediments ranged from 90 to 98 percent (Table 1). The lake bottom and water column were mostly filled with dense, rooted macrophytic growth. Winter fish kills were common, and recreational opportunities were severely restricted. Some fishing, boating, and swimming activities were possible in limited areas, but the recreational value was considered of poor quality.

TABLE 1: Water Content of the Sediments; September, 1977 (4 locations)

Depth Into Sediments	Percent Water
1.5 meters	96.4 - 97.6
3.7 meters	95.7 - 96.9
6.1 meters	90.6 - 95.2

Prior to the 1930s little development had taken place around Lilly Lake, although it provided some fishing, swimming, boating, and ice for ice boxes. In the 1930s the north shore and adjacent areas were developed, mainly for summer cottages. About that time a modest dragline-type dredging project was also completed to remove roughly 1230 m^3 of material for creation of a public beach. Spoils were deposited adjacent to the shore, with houses now located on this area. Residential development continued and at least two different lake associations annually cut weeds using their own weed cutters. Removal of the weeds that floated to shore was taken care of by the lakefront property owners. In the 1960s a weed harvesting program was adopted for the entire lake, and two aerators were operated each winter in an attempt to prevent the fish kills. Generally this operation was successful.

Prior to the 1970s, the Wisconsin Conservation Department conducted a fish management program on the lake including total fish eradication followed by restocking of desirable species; however, the beneficial impact of this effort was temporary due to the existing poor habitat. Then in 1974, the Lilly Lake Restoration District was formed by town resolution after advisory petitions and public hearings indicated more than 50% of the residents and property owners desired such a district. This lead to the development and implementation of a lake deepening project to permanently alter and improve the habitat and recreational usability of the lake.

The project was designed to remove 683,900 m^3 of sediment, increasing the maximum depth to 6.6 m. Two disposal sites were used during the dredging operation - a modified gravel pit about 3.0 km from the lake (plus a 27 m rise in elevation) and a 15 ha area of farmland approximately 0.5 km from the lake. Six shallow (about 1.2-1.8 m depth) settling basins were constructed on roughly 15 ha by scraping topsoil into low dikes. Spray irrigation of the sediments was also attempted, albeit unsuccessfully, on the farmland. A project location map is given in Figure 1.

Dredging began on July 17, 1978, with a 30 cm cutterhead, hydraulic suction dredge and continued until October 26 at which time the equipment was shut down for the winter. In 1978, sediment removal was concentrated near the lake center to attain the desired maximum depth. The material had been expected to slough toward the excavation, resulting in little need for dredge movement. Also, the sediment was primarily water, and it was expected to pump like water without requiring much additional carriage water. These assumptions later proved incorrect. The light-weight nature of the sediments (density of 1.02) and their cohesiveness prevented sloughing. As a result only about 382,400 m^3 were taken out in 1978 along with 415,700 m^3 of lake water. These were primarily deposited in the gravel pit site.

The remaining sediment was removed between May 1 and August 28, 1979. Dredging operations started on the east shore in order to clean the shoreline region while lake levels were still high. The south shore was dredged next. Both of these regions contained muck underlain mostly by sand with a very gradual slope. Cleaning was then begun on the west side. This was an easy area to operate in because the underlying sand sloped sharply and the desired 3.0 m water depth was achievable relatively close to shore. The north shore was left until last and was difficult to clean due to the lower water level by that time. A pocket of muck about 30 m wide and 90 m long had to be left because it was not possible to get to it. Both disposal sites were heavily utilized in 1979. The lake's depth contour and depth-volume relationship before and after dredging are given in Figure 2 and Table 2, respectively.

TABLE 2: Depth - Water Storage Relationship Before and After Dredging; Lilly Lake

	Before		After	
Depth (m)	Area (ha)	Water Storage (m^3)	Area (ha)	Water Storage (m^3)
0	37	532,300	37	1,216,200
0.6	32	320,500	35	995,800
1.2	29	132,200	33	787,100
1.8	14	0	31	590,000
2.4			30	403,700
3.0			28	228,600
3.6			9	115,700
4.2			6	70,900
4.8			5	40,300
5.4			4	15,400
6.0			1	2,300
6.6			1	0

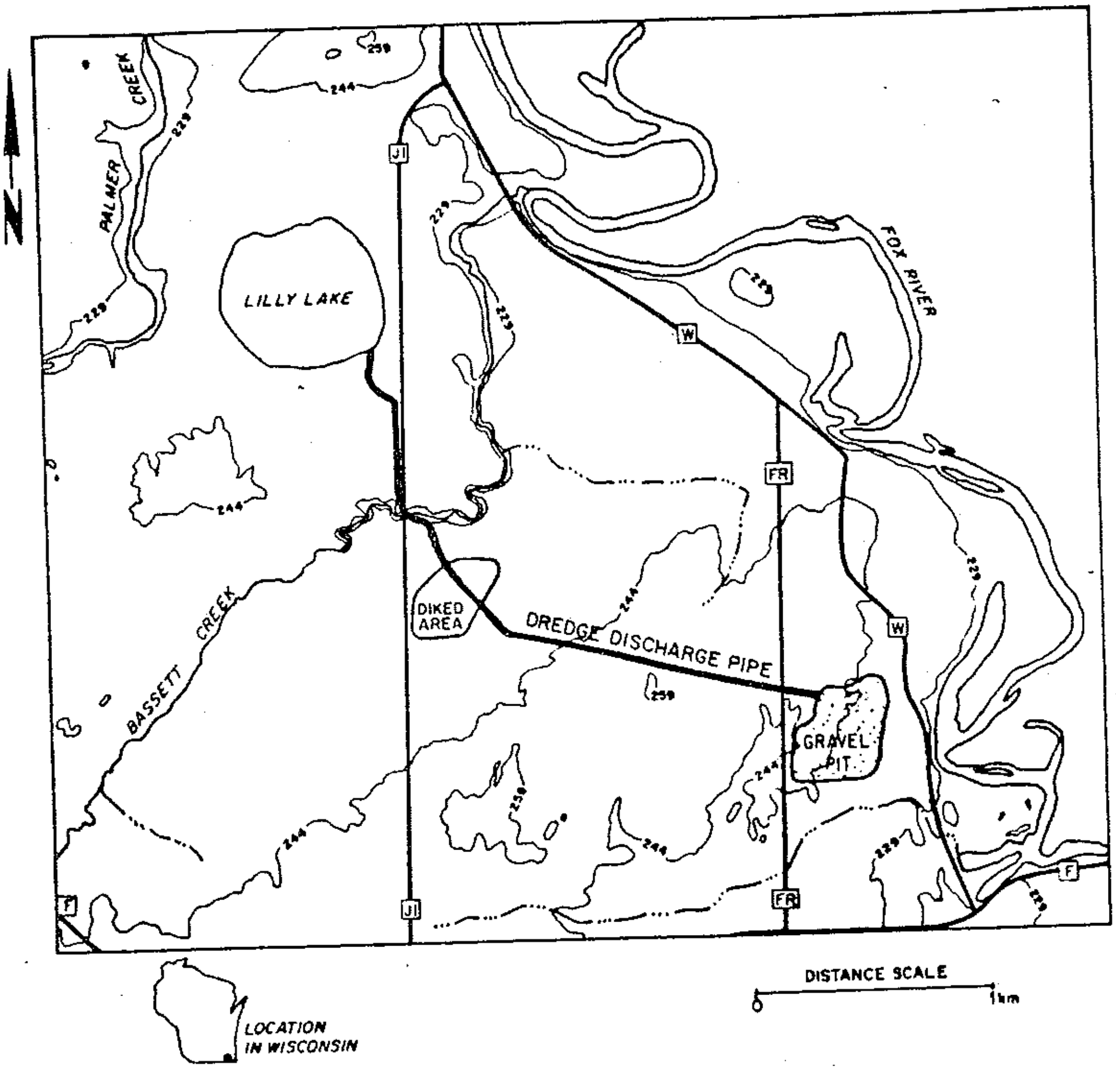

FIGURE 1. Location Map for the Lilly Lake Project

BEFORE

AFTER

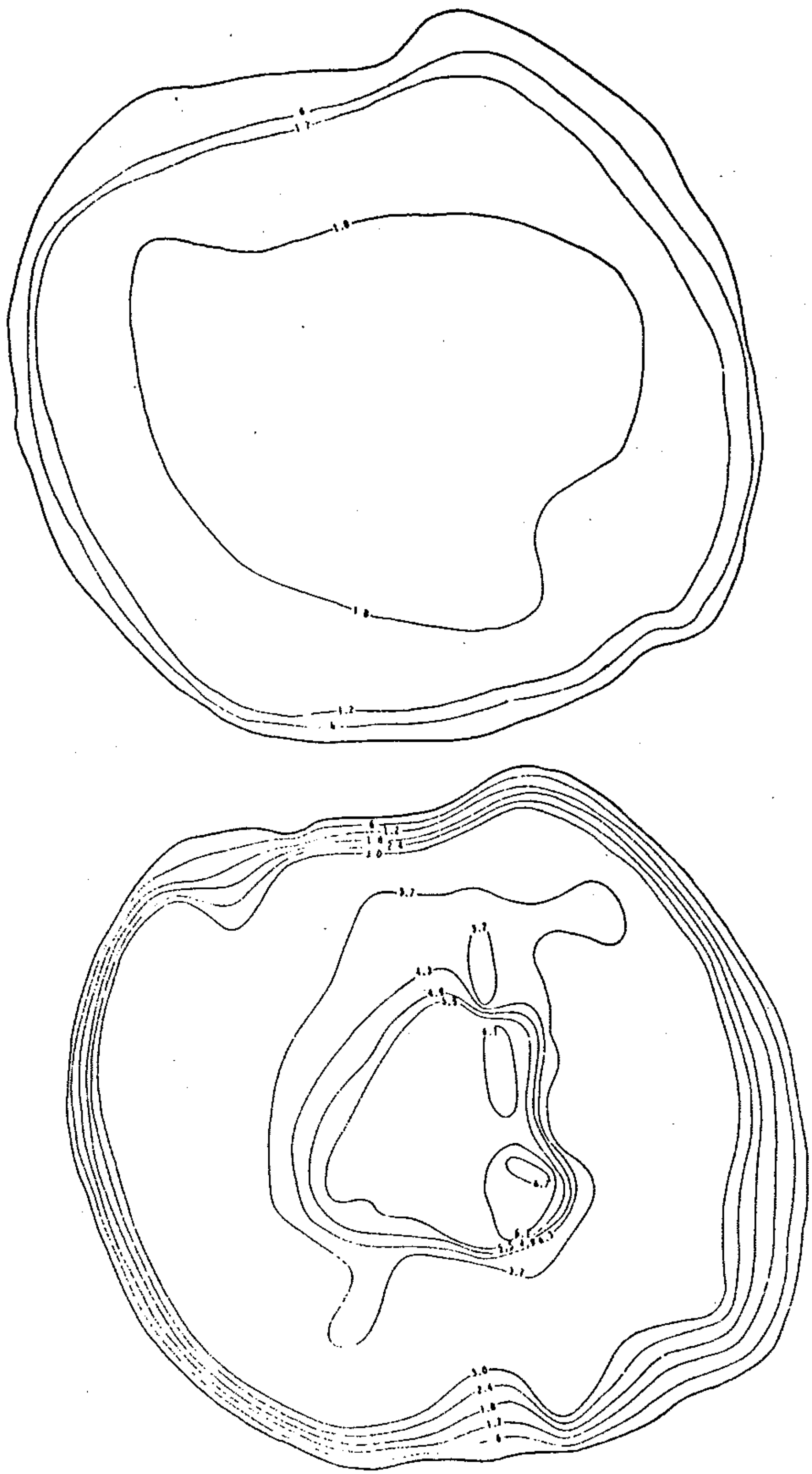

FIGURE 2. Lilly Lake Depth Contour Maps -- Before and After Dredging

The gravel pit was originally an excavation into the hillside, resulting in a steep incline on three sides and a flat exit area on the fourth side. The earth fill dams were constructed in the late spring of 1978 using existing soil from the gravel pit area. A schematic of the area is shown in Figure 3. Basin I (the upper one) was established with a capacity of 370,000 m^3 while Basin II was given 185,000 m^3. Basin I was built with a depth of almost 9 m. Basin II has a maximum depth of about 7 m. The water level difference between Basin I and II was 5 m and the drop from Basin II to the Fox River was approximately 11 m. The groundwater gradients from the basin's water level down to the river were much steeper than the usual regional groundwater gradient. Approximately 540,000 m^3 of lake sediment was deposited in the gravel pit area, with Basin I filled before usage of Basin II.

The secondary disposal site consisted of six individual cells constructed using topsoil and mineral subsoil located onsite. A schematic is shown in Figure 4 and the cell characteristics are given in Table 3. Cell 3 had the largest volume at 52, 340 m^3, while cell 5 was the smallest at 10,490 m^3. The deepest basin was cell 5 at 1.52 m followed by cells 1, 3, and 4 at 1.22 m and cells 2 and 6 at 0.91 m. Approximately 140,000 m^3 of lake sediment was deposited in the diked area. A rotational procedure was used, with sediment pumped into each cell until it was filled to capacity. The discharge was then directed into the next cell. After the sediment had settled, the overlying water was drained, thereby allowing the cells to be at least partially reused through several rotations. Later, after completion of the lake dredging activity, the sediments at this site were dried and incorporated into the topsoil. The fields were put back into agricultural production, growing a corn crop, in 1981.

TABLE 3: Basic Characteristics of the Diked Area

	CELLS						TOTAL
	1	2	3	4	5	6	
Area							
ha	2.95	1.34	4.29	1.27	0.69	1.46	12.0
ac	7.3	3.3	10.6	3.2	1.7	3.6	29.7
Top Elevation							
m	30.8	32.0	32.6	33.2	32.0	32.3	-
ft	101.0	105.0	107.0	109.0	105.0	106.0	-
Max. Water Level Elevation							
m	30.5	31.7	32.3	32.9	31.7	32.0	-
ft	100.0	104.0	106.0	108.0	104.0	105.0	-
Depth							
m	1.22	0.91	1.22	1.22	1.52	0.91	-
ft	4.0	3.0	4.0	4.0	5.0	3.0	-
Volume							
m^3	35,990	12,190	52,340	15,800	10,490	13,290	140,100
Ac-ft	29.2	9.9	42.4	12.8	8.5	10.8	113.6

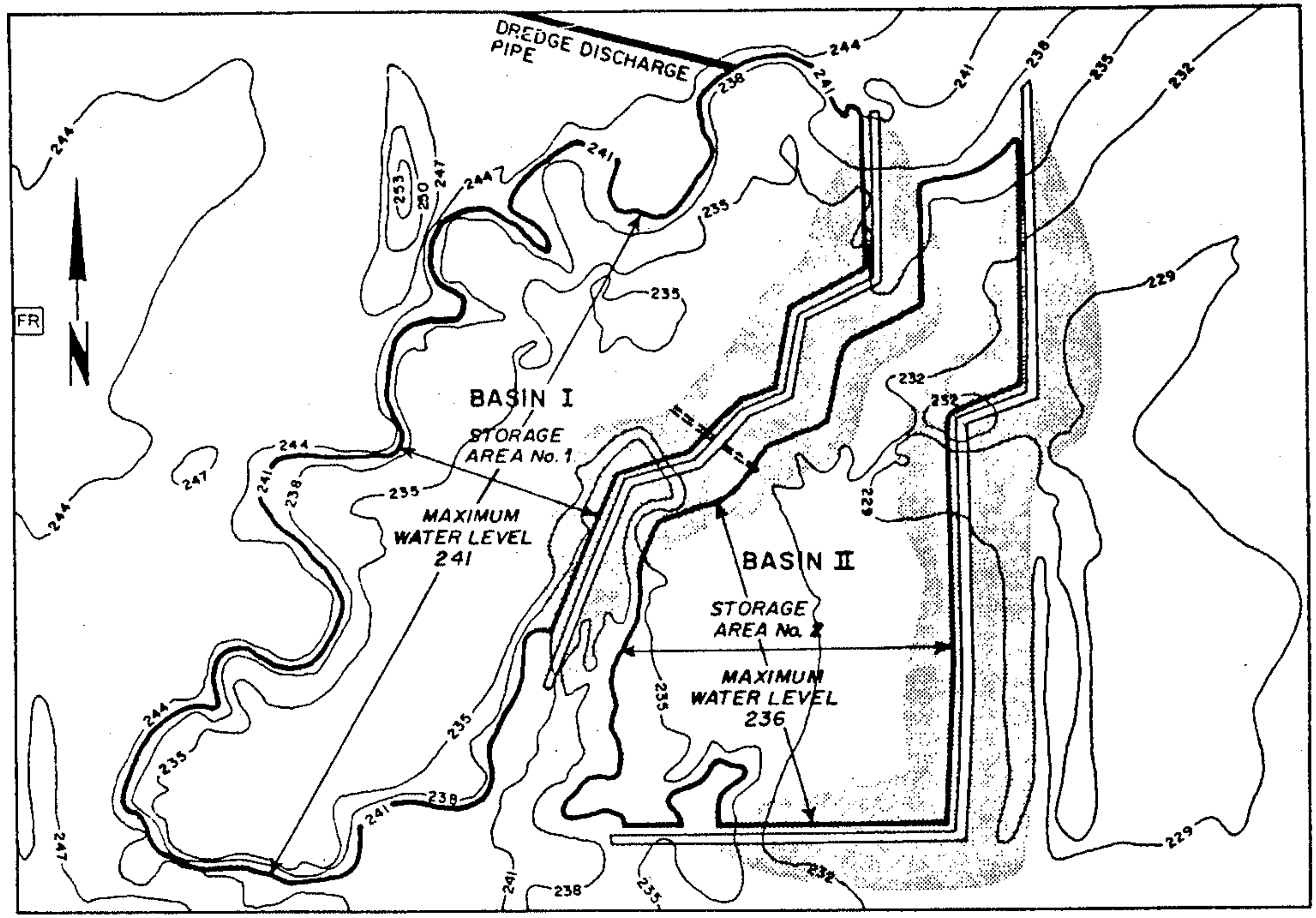

FIGURE 3. Gravel Pit Site Plan

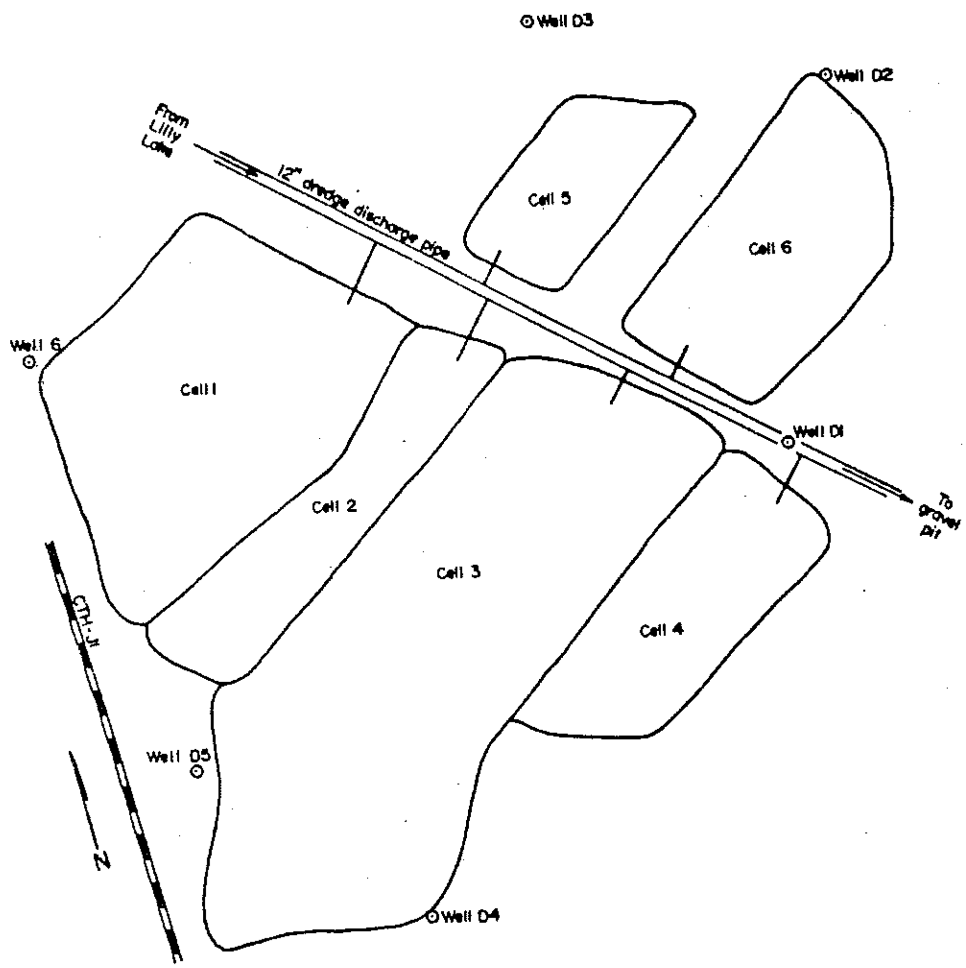

FIGURE 4. Diked Area Site Plan (with observation well locations)

SITE DESCRIPTION

Topography (see Figure 1)

Lilly Lake is located in the Town of Wheatland, Kenosha County, Wisconsin. It occupies a kettle-like depression in a topographically high area between the drainage basins of Palmer Creek, to the north and west, and Bassett Creek, to the south and east of the lake. Both of the streams flow northeasterly into the Fox River. The lake is at an elevation of 230.4 m above mean sea level. Hills to the north and southwest of the lake rise from 21.3 to 30.5 m above the lake. The floodplains of Palmer and Bassett creeks are at approximately an elevation of 228.6 m mean sea level. The terrain is gently rolling and irregular.

The gravel pit area was located in the NW 1/4 of the SE 1/4, Section 18, Township of Salem (T1N, R20E), Kenosha County, Wisconsin. It was constructed 3 km southeast of Lilly Lake. Local topographical relief in the area was 17 m on a ridge above the Fox River.

The diked area was located on parts of the Olson and Topczewski farms. It was constructed on 15 ha of land, 0.5 km southeast of Lilly Lake. Local topographical relief in the area was 3.6 m on an outwash bench above Bassett Creek.

Geology

Lilly Lake lies on the Niagara upland of the eastern ridge and lowland province of Wisconsin. The area is underlain by dolomitic limestone and was covered by the Lake Michigan lobe of the Wisconsin glacial stage, which is primarily responsible for the surface deposits (Alden, 1918). The lake lies in an area of pitted outwash near one of the terminal moraines. Prior to dredging, sand predominated along 65% of the shoreline, gravel and rubble covered 6%, and soft sediments covered 29%. Sand and gravel usually dominated along wave-washed shores, but finer sediments appeared beyond a depth of about 0.5 m. The entire lake center was composed of mucky sediments. In 1977 a sediment core was taken and analyzed by Battelle Laboratories (Richland, Washington) for 210 lead content. The results indicated that the surface sediments (upper 1 m meter) were being deposited at a rate of 0.5 cm/yr.

Bedrock under the gravel pit site is a Silurian age dolomite. Prior to gravel removal, approximately 50 m of unconsolidated glacial deposits made up the aquifers and parent material for the topsoil at the site. Some 14 m of gravel were removed between 1955 and 1971. Both ice-contact and outwash deposits in the northwest one half and southeast one half of the site, respectively, have been identified by Hutchinson (1970). Both deposits are characterized as having high permeability. Ice-contact deposits are composed of coarse gravel, boulders, sand and some silts which are fair to well sorted and may contain large masses of low permeability material. Outwash, sometimes covered by alluvium, is well sorted sands and fine gravel in nearly horizontal beds. The logs of six soil borings taken in the gravel pit prior to dam building show a sand and gravel layer over silt and clay with sand and gravel again under the clay.

Bedrock is approximately 34 m under the unconsolidated glacial deposits which make up the aquifers and parent material for the topsoil at the diked area. Both outwash and ice-contact deposits have also been identified in the area by Hutchinson (1970). The boring logs of the six observation wells are shown in a fence diagram in Figure 5.

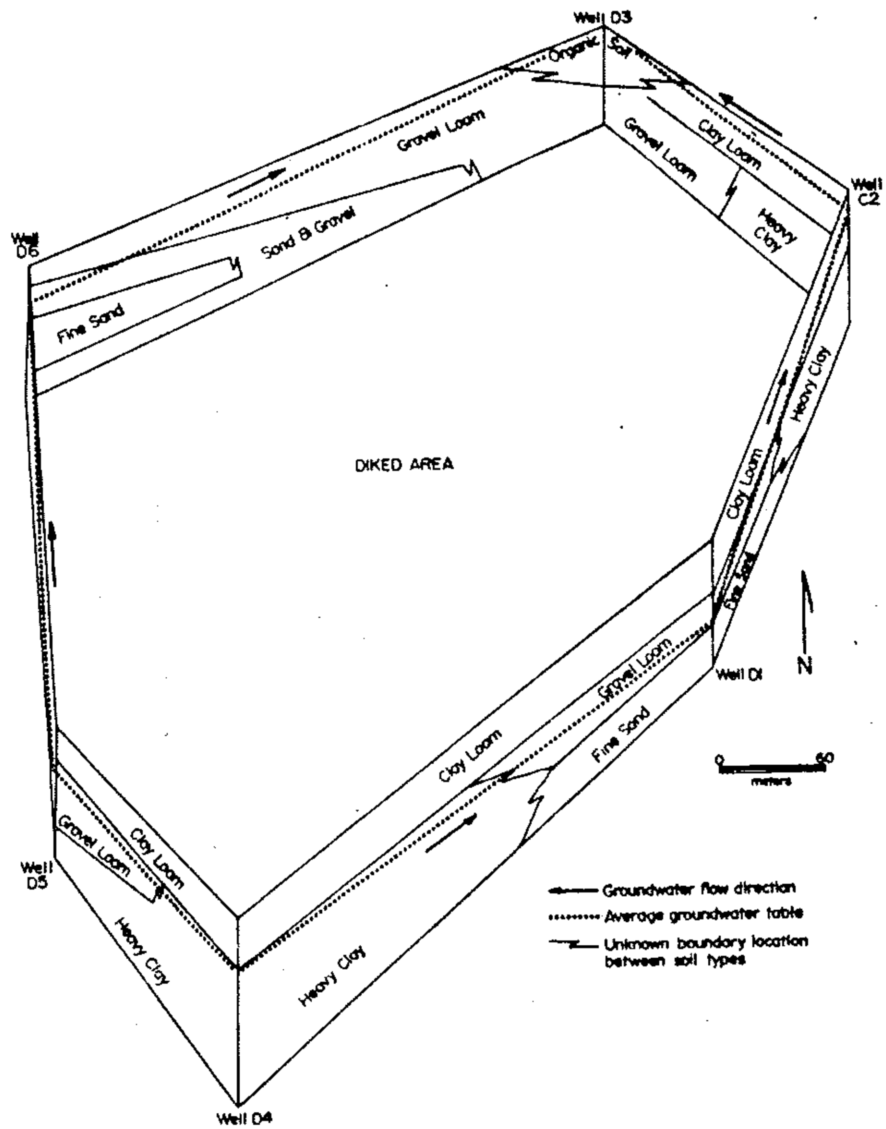

FIGURE 5. Fence Diagram of the Well Logs Around the Diked Area

Soils

The Lilly Lake watershed is presently 155 ha in size. The drainage pattern is somewhat irregular, showing influences of recent erosional activity. The soils in the area are Miami loam, Miami fine sandy loam, Rodman gravelly loam, peat, Fox silt loam, Fox loam, Miami silt loam and Clyde silt loam.

The topsoil was removed over the gravel pit area, but the undisturbed soils near the gravel pit give some useful information. Miami loam is the primary soil type to the north and upland of the pit. It is a well drained soil which formed from a silt mantle over loamy glacial till. The soil types which are south and east and generally downhill of the Miami loam are the Casco-Miami loams, Fox silt loams and Fox loams. These soils formed on an outwash sand and gravel. They are well drained soils. Other soil types in the adjacent areas, include Hebron sandy loam, Navan silt loam and Saylesville silt loam.

Hebron loam is the primary soil type in the diked area. It is a loamy, well drained soil which developed from outwash material underlain by laminated lacustrine silts and clays on the flat sections of old lake beds. Other types in the diked area are Matherton loam, Fox loam, and Navan silt loam.

A comprehensive description of the distribution and characteristics of these soil types can be found in USDA, SCS (1970).

Precipitation/Groundwater

Rainfall amounts were estimated for the project area using the average of four nearby U.S. Weather Stations: Burlington, Union Grove, and Lake Geneva, Wisconsin; and Antioch, Illinois (Table 4). During the five year period 1977 through 1981, 1978 had the highest annual precipitation (106.53 cm). Based on these stations, 1977 and 1981 had precipitation below the long-term average (e.g. 85.55 cm, 1952-81). However, onsite monitoring at Lilly Lake in 1981 indicated precipitation of about 86.5 cm, suggesting that 1977 was the only below normal year.

Lilly Lake has no permanent surface inlets or outlets, although historical drainage patterns have been dramatically different. The regional movement of groundwater is northeast toward the Fox River, roughly paralleling the surface drainage. Test wells have indicated that local groundwater discharge from Lilly Lake travels to the southeast toward Bassett Creek, with groundwater entering the lake from the south, west and north.

Groundwater flow patterns near the gravel pit run from the west and south, a groundwater recharge zone, to the east, a groundwater discharge zone (Hutchinson, 1970). The rate of water table drop across the gravel pit area is estimated to be about 1 m per 260 m. Groundwater flow patterns near the diked area also run from the south and west to the north and east (Hutchinson, 1970). Soil permeability tests were done on four wells using a slug test method (Bouwer, 1978). Hydraulic conductivities of 0.16×10^{-4}, 0.65×10^{-5}, 0.22×10^{-4} and 0.13×10^{-5} cm/sec were determined for wells D2 through D5, respectively. Well D4 was the most permeable while well D5 was the least permeable of the four wells tested. The soil boring data and field observations during sampling indicated that wells D1 and D6 had larger permeabilities than well D4.

TABLE 4: Average Precipitation for the General Lilly Lake Project Area, 1977-1981*

MONTH	YEAR 1977	1978	1979	1980	1981	MEAN (1952-1981)
January	1.96	4.58	7.49	3.02	0.28	3.86
February	1.37	0.84	2.69	2.44	5.11	3.07
March	10.77	2.06	9.68	1.95	1.75	6.07
April	5.18	8.99	14.93	10.29	12.17	9.17
May	5.16	7.72	4.22	5.05	7.11	7.67
June	11.48	15.16	11.71	12.19	9.80	10.72
July	9.45	22.68	4.88	16.15	8.56	10.80
August	12.42	19.00	25.30	15.31	14.53	9.96
September	8.10	10.72	0.20	14.73	9.75	8.15
October	6.63	4.04	4.60	3.99	5.33	5.92
November	6.12	5.33	7.31	3.12	4.50	5.44
December	5.74	5.41	4.65	8.36	2.18	4.72
YEAR	84.38	106.53	97.66	96.60**	81.07**	85.55

*Average precipitation in centimeters calculated from Burlington, Lake Geneva, and Union Grove, Wisconsin and Antioch, Illinois.

**Precipitation was monitored at a site on the south side of Lilly Lake during April-October, 1980 and 1981. These measurements indicated that about 92.7 and 86.5 cm fell in the first and second year, respectively.

METHODS

Inlake Conditions

Sampling began for some parameters in July 1976; however, a comprehensive effort was not initiated until the spring of 1977. As monitoring objectives were achieved, sampling for that particular parameter was discontinued with all sampling terminated in 1982. Unless indicated otherwise, inlake sampling was performed at 1-2 m depth intervals every 1-4 weeks throughout the year at a central location.

Dissolved oxygen (D.O.) and temperature measurements were taken by the Winkler method and with an electronic thermistor thermometer, respectively. Sampling was conducted throughout the project period at the central location, with additional sites used during some winters.

The following chemical parameters were monitored in the lake throughout the year: biological oxygen demand (B.O.D.; 5 day), turbidity, soluble reactive phosphorus (SRP), total dissolved phosphorus (TDP), total phosphorus (TP), nitrite/nitrate nitrogen (NO_2/NO_3-N), ammonia nitrogen (NH_4-N), organic nitrogen (TON), conductivity (Sp. C), pH, and total alkalinity (TA). Additional parameters measured on a quarterly basis were calcium (Ca), magnesium (Mg), sodium (Na), potassium (K), chloride (Cl), and sulfate (SO_4). B.O.D. samples were incubated for 5 days at 20°C without inoculation; all other chemical analyses were performed by the Wisconsin State Laboratory of Hygiene (SLOH) using approved procedures (U.S. EPA, 1979).

Algal population characteristics were determined in terms of chlorophyll _a_, water clarity, species composition and enumeration, biovolume, and productivity. Sampling was limited to the period of optimal growth, May through October. Water clarity was noted using a 20 cm diameter white Secchi disk. Chlorophyll _a_ samples were kept in a cool, dark container and processed the following day. Productivity was determined by the light and dark bottle, D.O. method, with suspension from noon to 6 P.M. The results were converted into mg C/m^3/day as per Megard (1972). Inlake D.O. levels were also measured at the start and end of the suspension period in order to assess the impact of the algae on the net change observed in the water column. Whole water samples were removed from the 0-1 m depth and preserved with acidic Lugol's solution until the time of analysis for species composition, enumeration, and biovolume. Biovolume was determined by approximating the geometric forms of individual plankters and applying the estimated volume to cell counts. Species composition was estimated using the Utermohl technique (Lund _et al._, 1958) with an inverted phase contrast microscope. A minimum of 100 cells of the most dominant organism were counted. Phytoplankton organisms were identified to the lowest practical taxonomic division. Taxonomic keys used included Hustedt (1930), Huber-Pestalozzi (1938), Skuja (1948), Smith (1950), Prescott (1962), Patrick and Reimer (1966 and 1975), and Weber (1971).

Zooplankton sampling consisted of a single, vertical tow from the 2 m depth (1.8 m prior to dredging). A #20 mesh cone net was used during May through October and lifted at a rate of 0.3 m/sec. The samples were preserved in 5% formalin until species identification and enumeration. One percent of the total sample was counted using a Sedgwick-Rafter cell and compound microscope. Taxonomic keys included Brooks (1957), Deevy and Deevy (1971), Brandlova _et al._ (1972), Yeatman (1959), Torke (1976), and Smith and Fernando (1978).

Benthic invertebrates were collected with an Eckman dredge in April. Eight samples (0.25 m^2 each) were taken each year, four near shore over sand bottom in water 0-0.5 m deep and four offshore over organic sediments in water 1.5-3 m deep. The invertebrates were separated from the sediments with a #90 mesh sieve and preserved in 5% formalin until analysis. Hilsenhoff (1975) and Pennak (1953) were used for taxonomic identifications.

Fish analyses were limited to determination of species composition and age-growth for the primary species. This effort was simplified by the severe winter fish kills in 1976-77 and 1977-78. Immediately after ice-out dead fish were identified, enumerated, and measured to develop a length-frequency relationship. Scale samples were collected from the primary species and the age-growth relationship was established by the methodology of Lagler (1961). Species identifications were made according to Hubbs and Lagler (1958).

Aquatic macrophytes were sampled biweekly, May through September. Ten 0.1 m^2 whole plant biomass samples were removed from each area of interest. Prior to dredging there were 10 areas - four at the 0 to 0.5 m depth, four at the 0.5 to 1.5 m depth, and two at maximum depth. After dredging additional sampling was conducted at the 3 m depth. After removal, the samples were placed in plastic bags, brought back to the laboratory, cooled overnight, and processed the next day. The samples were separated according to individual species using the taxonomic classifications of Fassett (1960) and Voss (1972). Drying was accomplished at 105° C. Phosphorus tissue content was determined monthly for the major species. Analyses were conducted by the University of Wisconsin Soil and Plant Analysis Laboratory as per procedures described in Jackson (1958).

Eighteen observation wells were established around the lake (Figure 6). Water levels were monitored monthly using a Johnson well light attached to a steel tape calibrated in 0.3 cm intervals (an alternative was to use water level indicator paste on the tape). Water levels were also noted on the same days at a staff gauge installed in the lake.

Every 1 to 3 months water samples were taken from the wells located near the lakeshore. This was done with a tygon tube or plastic garden hose attached to a "Guzzler" pump. Each well was pumped to remove the equivalent of about three volumes of water, or until the well went dry whichever occurred first. As soon as the well refilled, a sample was collected in a plastic bottle pre-rinsed with groundwater. Chemical analyses were conducted at SLOH.

Nine seepage meters were emplaced about equidistant around the shoreline in water depths of 0.1 to 0.5 m. These were constructed and installed as per Lee (1977). The seepage meters were located at a particular site for the entire study, and water was allowed to pass in or out of the meter between samplings. Monitoring was undertaken at 1 to 4 week intervals during the open water period. Plastic bags containing 0 to 800 ml of water, and having a total capacity of about 1500 ml, were affixed to each seepage meter. After 4 to 6 hours, the plastic bags were removed and the change in water volume noted.

A record of precipitation at the project site was obtained primarily by averaging the records from four nearby U.S. Climatological Stations; however, measurements were taken by an onsite cooperator for part of the study period. Due to watershed characteristics, storm events rarely produced channelized flow into the lake; therefore, runoff sampling was limited to periodic collections with analyses at SLOH for nitrogen (N) and phosphorus (P) forms.

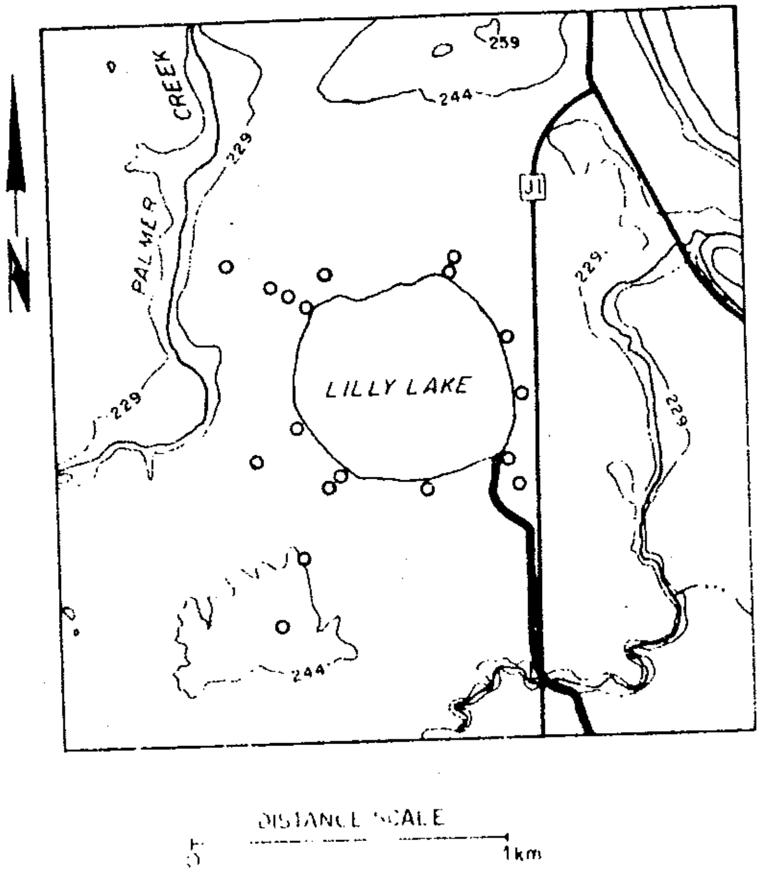

FIGURE 6. Location of Groundwater Monitoring Wells - Lilly Lake

In 1978 a Barton Differential Pressure Recorder was placed on the dredge pipeline to measure the amount of water and sediment leaving the lake. The Barton recorded head losses over a 5 m section of the pipeline. Head losses were then converted to flow using the Manning and Hazen-Williams equations and friction coefficients supplied by the pipeline manufacturer (ITT Barton, Monterey Park, California). Once the volume leaving the lake through the pipeline was known, a relationship was established between the volume and the number of hours of dredge operation.

Four opaque sediment oxygen demand (SOD) chambers were used to determine the N and P release rates. These chambers were similar to those used by Sonzogni et al. (1977). In 1977 they were installed at mid-lake in July and sampled 2-4 times per month through October. Only two chambers were used in 1980 and 1981, and these were placed at the 3 m depth. Each chamber covered 0.96 m^2 of lake bottom and contained 0.44 m^3 of water. The chamber top was 46 cm above the sediment surface. The chambers were sampled with a 500 ml stainless steel syringe, with collections from 10 and 43 cm below the top of the chamber. Analyses were routinely performed for P and N forms by SLOH. Manganese (Mn), iron (Fe), pH, D.O., and bacteria levels were also noted periodically.

A piston corer was used to determine P and N profiles in the sediments and interstitial waters. A 1 m core was collected in June, 1978 and every 2 to 4 weeks during May through September in 1980 and 1981. Analyses were conducted for the following depth intervals: 0 - 7.6 cm, 7.6 - 15.2 cm, 15.2 - 22.9 cm, 22.9 - 30.5 cm, 60 - 70 cm, and 90 - 100 cm. In collecting the interstitial waters, sediment contact with air was minimized by using a 50 ml syringe to remove the sediments from the desired depth within the piston corer and transferring them directly into a centrifuge tube. In 1978 the interstitial waters were extracted by centrifuge and filtered at lakeside. In later years the tubes were covered, kept in a cool, dark container, and transported to the laboratory for processing further. SLOH examined the samples for NH_4-N, NO_2/NO_3-N, and SRP and/or TDP. A portable meter was used to measure pH in the field.

In addition, sediment samples were taken from the piston corer, placed directly into plastic bags, and dried on 23 cm diameter watch glasses at 60°C in the laboratory. Bulk chemical analyses were conducted by the University of Wisconsin Soil and Plant Analysis Laboratory as per Jackson (1958). This was also performed on sediment samples taken from the 1.5 m, 3.0 m, 4.6 m, 6.1 m, 7.6 m, and 8.5 m depths in February, 1977. Phosphorus mobility was also measured in a separate sediment subsample using the sodium hydroxide (NaOH-P) and hydrochloric acid (HCl-P) extraction procedures as described in Peterson and Corey (1966). The extractions were performed at the University of Wisconsin Soils Department laboratory, with analyses of the resultant water conducted by SLOH.

Lake water depth mapping was performed before and after dredging using a laser-operated EDM. Measurements were taken at about 75 sites distributed throughout the lake basin. Hydrographic maps were prepared from this information by experienced personnel in the Bureau of Engineering, Wisconsin Department of Natural Resources.

Sediment Disposal

Eight observation wells were installed in the spring of 1978 to supplement the one existing well at the gravel pit (Figure 7). Six of the wells were piezometric nests of two wells each (sites G1-G3) with one screen installed 3.4 to 4.8 m deeper than the other (Table 5). At the diked area, six observation wells were installed in late April, 1979 to measure the impact of sediment disposal at that site (see Figure 4). In both cases the wells were established by using a truck-mounted drill rig. Galvanized steel pipe with a diameter of 3.2 cm was used. The well screens were 0.9 m drive points. Well tops projected above the land surface from 0.3 to 1.2 m. Each well was backfilled with drill cuttings, and a bentonite seal was placed at the land surface. The well pipes were capped with a threaded cap which had a hole in the side to allow for air exchange.

TABLE 5: Location and Depth of Groundwater Observation Wells - Gravel Pit*

Well Number	Approximate Distance From Basin	Meters	Depth of Well (Meters)
G1	II	91	9.0
G1A	II	91	13.8
G2	II	47	7.3
G2A	II	47	10.7
G3	II	64	9.0
G3A	II	64	13.7
G5	I	46	15.6
G6	I	343	8.0
G7	I	76	26.8

*A 0.91 m well point screen was used for wells G1-G6. Well G7 was an existing well; the length of its screen is unknown.

The following procedure was used when sampling observation wells. First, the water level was measured by lowering a Johnson well light attached to a steel tape calibrated in 0.3 cm intervals (an alternative method was to use water level indicator paste on the tape). Then, the well was pumped to remove the equivalent of about three volumes of water, or until the well went dry depending on which occurred first. This was done with a tygon tube or plastic garden hose attached to a "Guzzler" pump. As soon as the well refilled, a water sample was collected in a acid washed, MILLQ (distilled) water rinsed bottle provided by SLOH. The plastic bottle was rinsed with 50-100 ml aliquot of groundwater before a water sample was retained for analysis. Analyses generally done in the field were pH and Sp. C. All other chemical analyses were done in the laboratory by SLOH.

In addition to the observation wells, water samples were collected from nine houses located near the gravel pit (Figure 8). Their distances from the gravel pit and depths ranged from 76 to 360 m and 15 to 91 m, respectively (Table 6). The water from these wells was sampled from a faucet as near as possible to the groundwater source. In some wells this was an outside faucet. The faucet was allowed to run for five to ten minutes prior to taking the sample.

In the summer of 1978, six samples were collected from the sediments deposited in the gravel pit site. The samples (including pore waters) were analyzed for percent solids, N forms, and pH according to the procedures and methods described in the Inlake Condition section.

Beneficial Uses

The study included three major components: (1) a survey of several chemical properties for 12 sediments from lakes throughout Wisconsin; (2) a greenhouse study comparing the effects of different rates of four lake sediments on yield and chemical composition of corn plants; and (3) field studies to determine the effects of different rates of sediment from Lilly Lake in Kenosha County on yield and chemical composition of corn.

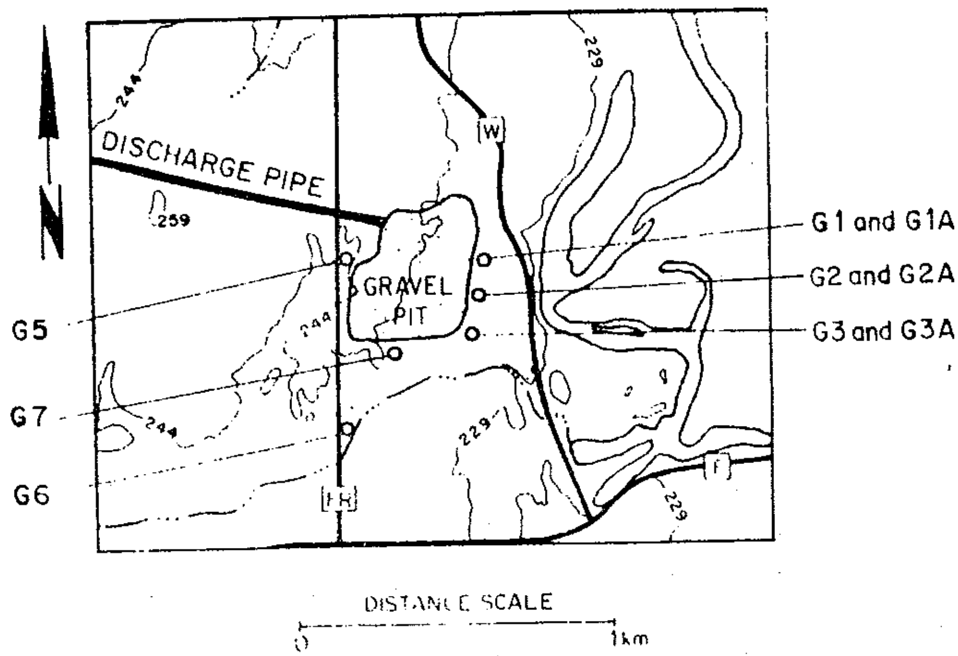

FIGURE 7. Groundwater Observation Well Location Map - Gravel Pit

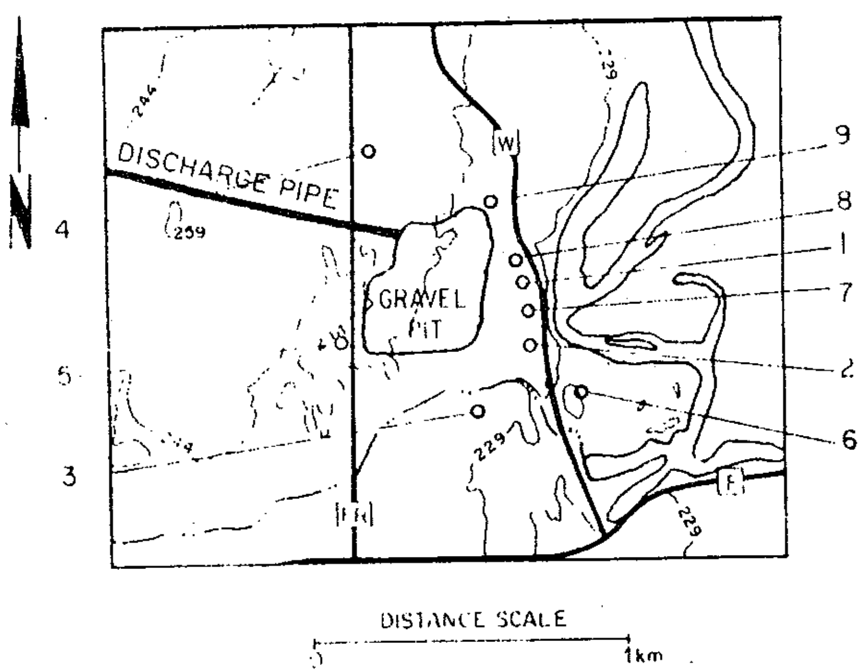

FIGURE 8. Private Home Well Location Map - Gravel Pit

TABLE 6: Location and Depth of Private Home Wells - Gravel Pit

Well Number	Approximate Distance from Basin I (Meters)	Depth of Well (Meters)
1	168	Shallow (depth unknown)
2	189	15
3	244	26
4	189	91
5	189	50
6	360	15
7	168	Deep (depth unknown)
8	94	(depth unknown)
9	76	47

A. Sediment Survey

Samples for the sediment survey were collected by personnel from the Wisconsin Department of Natural Resources; these samples were stored in a frozen condition. The lake names and their corresponding counties are: Comus, Walworth Co.; Half-Moon, Bayfield Co.; Leota, Rock Co.; Lilly, Kenosha Co.; Lily, Marathon Co.; New Richmond, St. Croix Co.; North Ash, Polk Co.; Pigeon, Waupaca Co.; Pine, Forest Co.; Rib, Taylor Co.; Tomah, Monroe Co.; and Wilson, Waushara Co. These sediment samples were later thawed and air-dried. They were then ground and passed through a 0.64 cm screen. Subsamples taken from each lake sediment were ground, passed through a 2 mm screen and stored for chemical analyses. All sediments were analyzed for pH, total carbon (C), chemical oxygen demand (COD), loss on ignition, total N, NH_4-N, NO_3-N, TP, organic P, P extracted with 0.5M sodium bicarbonate ($NaHCO_3$), P equilibrated in 0.01M calcium chloride ($CaCl_2$), and total zinc (Zn), Mn, copper (Cu), cadmium (Cd), and lead (Pb). The sediments were also incubated for a total of 6.5 months at approximate field moisture capacity and room temperature to determine changes in mineral N and $CaCl_2$- and $NaHCO_3$-extractable P.

Sediment pH was determined on a 7.5:10 sediment to solution ratio in a 0.01M $CaCl_2$ extract. Weight loss on ignition was determined by the percent weight change of oven-dried sediments when heated for two hours at 900°C. COD was determined as outlined by Water Standard Methods (Am. Public Health Assoc., 1975), except that a 1 g sediment sample plus 20 ml of water was used in place of the 20 ml water sample. Total C was estimated by the Walkley-Black wet combustion method. Schulte (1980) outlines the procedure which uses one volume 1N potassium chromate (K_2CrO_7) with two volumes sulfuric acid (H_2SO_4) mixed with sediment sample. Excess $Cr_2O_7^{-2}$ is determined by titration with a standard ferric sulfate ($FeSO_4$) solution. Total P in sediments was extracted by a nitric-perchloric digestion method. Organic P in sediments was estimated by subtracting the inorganic P from the total P (Mehta et al., 1954). Concentration of P was determined by the method outlined by Murphy and Riley (1962). Total N was determined by the semi-micro Kjeldahl method, and NO_3-N + NH_4-N and available plant nutrients were determined by methods outlined by Liegel et al. (1980). Trace elements in sediments as well as in plant tissue were determined using Inductively-Coupled Plasma (ICP) emission spectroscopy following a nitric-perchloric acid digestion.

The incubation study began on July 23, 1980. Seventy grams of each lake sediment was mixed with silica sand to a total moist volume of 500 cm^3 (this was equivalent to the high rate in the greenhouse study, 270 mT/ha). The sediment mixtures were then leached with 0.01M $CaCl_2$ and filtered under suction through a Buchner funnel with No. 42 Whatman filter paper until the leachate stopped flowing. Sediment mixtures were then transferred to plastic bags and sealed with rubberbands. The moisture content was maintained at this level throughout the study, and the temperature was that of the laboratory, ranging from 18-25°C.

Sediment mixtures were sampled at the start and on October 26, 1980, January 28, 1981 and May 8, 1981. Samples were analyzed for soil solution P by equilibrating with 0.01M $CaCl_2$ (Kamprath and Watson, 1980). Surface P was determined by a 0.5M $NaHCO_3$ extraction (Olsen et al., 1954). Concentrations of P were determined by the method of Murphy and Riley (1962). The pH of sediment mixtures was determined by equilibrating 4.7 cm^3 of sample with 10 ml of 0.01M $CaCl_2$. The analysis of NO_3-N + NH_4-N in sediment mixtures was performed by methods outlined by Liegel et al. (1980).

B. Greenhouse Study

The two soils used in the greenhouse experiment were Withee silt loam (Aeric Glossaqualf) and Plainfield sand (Typic Udipsamment). Soils were spread, dried and sieved through a 0.64 cm screen.

Four air-dried sediments were used in the greenhouse experiment. The sediments were selected to give a range in organic matter, pH, N content and C/P ratio. Sediments from Lilly Lake in Kenosha County and Lily Lake in Marathon County were ground and passed through a 0.64 cm screen. Sediments from New Richmond and Tomah lakes (St. Croix and Monroe counties, respectively) contained less organic matter and could be passed through a 2 mm screen. These sediments were mixed with each of the two soils at four rates equivalent to 10, 30, 90 and 270 mT of sediment per ha, or 7.5, 22.5, 67.5, and 202.5 g per 1.5 dm^3. The sediment-soil mixtures were split into four replicates and placed into four separate pots each having a final volume of 1.5 dm^3. Each pot received 50 mg K/dm^3 as potassium chloride (KCl). The pots with Plainfield sand received 1.2 g calcium carbonate ($CaCO_3$) per dm^3 to raise the pH of the soil. In addition to the above, the fertilized controls received 50 mg N/dm^3 as ammonium nitrate (NH_4NO_3) and 25 mg sulfur(S) per dm^3 as calcium sulfate ($CaSO_4$). The unfertilized control received 50 mg K/dm^3 as KCl.

The field moisture capacity of the sediment soil mixtures was determined by pouring an amount of water into a soil-sediment column that was not enough to wet all of the soil. The percent moisture in the wetted portion of the soil after 24 hours was taken as the field capacity of that sediment-soil mixture.

All pots were watered to field moisture capacity and incubated for 17 days before planting. The pots were sown to corn (*Zea mays*, variety Blaney '606') at eight seeds per pot. Two weeks after planting, the pots were thinned to four plants per pot.

Two weeks after emergence, chlorotic tissue, indicating N deficiencies, was first observed on the unfertilized control in the Plainfield sand. Nitrogen deficiencies were also noted on plants in the second rate of Lilly Lake-Kenosha (Withee silt) and Lily Lake-Marathon fourth rate (Plainfield sand). Phosphorus deficiencies were observed with all sediments but were most pronounced among the high rates of Lilly Lake-Kenosha and Lily Lake-Marathon sediments.

After eight weeks of growth, the plants were harvested 2.54 cm above the soil surface, rinsed with distilled water, dried at 60°C and ground in a stainless steel Wiley mill for analysis. Soil samples were taken from each pot the day following the harvest, air-dried, ground and sieved through a 2 mm screen for analysis.

Soil organic matter, NO_3-N + NH_4-N, pH and essential plant nutrients in both soil and plant tissue were then determined by methods outlined by Liegel et al. (1980). Plant tissue harvested from the high sediment rates and controls were also analyzed for trace elements on the ICP following a nitric acid-perchloric acid digestion. Greenhouse soils were analyzed for P by the 0.01M $CaCl_2$ and 0.5M $NaHCO_3$ extraction methods previously mentioned in the incubation study.

C. Field Study

The field study was set up on June 18, 1980 on the Topczewski farm near Lilly Lake. The soil was a Fox silt loam (Typic Hapludalf). Target rates of 0, 22.4, 44.8 and 89.6 mT/ha were applied to four replicates of each treatment. The design was a latin square. Plot size was 3.7 m x 6 m with border areas between plots. Lake sediment was obtained from a lagoon adjacent to the plot with a bucket mounted on the rear of a tractor. The sediment contained in five leveled buckets was weighed to determine the average amount of sediment per bucket. The sediment was then applied by volume (number of buckets) rather than by weight. The 89.6 mT/ha rate required 10 buckets per plot. Sediment in each bucket applied was sampled with a soil sampling tube to determine average moisture content, loss on ignition and chemical analysis of the sediment applied to each plot. For determining the amounts to be applied, the sediment was assumed to have 18% dry matter, which was the amount measured in samples taken a few days before from the same lagoon.

Prior to sludge application, the alfalfa growing on the plots was removed and the land was worked twice with a chisel plow. After sediment application, the land was worked immediately with the chisel plow and later it was disked, plowed and disked again prior to planting. Sudan grass, var. Sudax, was then planted.

Two crops of Sudan grass were harvested in 1980 (July 31 and September 3). Dry matter yields were determined and samples were dried, ground and subsampled for analysis. Analyses for both macronutrient and trace elements were made, including N, P, Zn, Mn, Ca, Cd, arsenic (As), and Pb (Liegel et al., 1980).

In the spring of 1981, the field plots were reestablished on the Russell Olson farm (Hebron silt loam, Typic Hapludalf) with the same experimental design. Prior to planting, soil samples were taken from all plots and analyzed for pH, organic matter, P, K, Ca and Mg (Liegel et al., 1980). Corn was then planted that spring. Ear leaf samples were taken along with yields and subsamples of grain and stover at harvest. Elemental analyses were performed for N and P on plant tissue.

The following year corn was grown again, and plant nutrient concentrations were determined on ear leaf and corn grain samples. Analytical testing included N, P, K, Ca, Mg, S, Zn, boron (B), Fe, Cu, aluminum (Al), Mn, cobalt (Co), As, Cd and Pb, with trace element analyses limited to the ear leaf tissue only (Liegel et al., 1980).

Statistical analyses of greenhouse data were performed using the Statistical Analysis Service (SAS) computer system (Helwig and Council, 1979). Statistical tests of field data were performed by using the MINITAB statistical computing system.

INLAKE EFFECTS OF DREDGING

The monitoring program included measurement of a wide array of physical, chemical, and biological parameters. The information provided a simple description of the conditions prior to, during, and after dredging; and some insight into inlake nutrient dynamics and cause and effect relationships. The findings were separated into report sections entitled sediments, hydrology, inlake water quality, and aquatic biota.

Sediments

The sediments were characterized by rate of deposition, water content, chemistry, oxygen demand, rate of nutrient release, and quality of the interstitial waters. Prior to dredging, lead 210 dating procedures were utilized to determine the recent deposition rate of 0.5 cm per year. The sediments were light-weight (density of 1.02) and highly organic (62 percent). The water content was about 97 percent near the sediment-water interface but decreased with depth (age) of the sediments. In mid-lake the sediment depth was 10.7 m. Rooted aquatic plant debris was prominent in the upper sediments, becoming progressively smaller in particle size with increased depth into the sediments. Infilling of the lake has, therefore, occurred primarily through autochonous processes. General sediment chemistry is shown in Table 7.

Dissolved oxygen depletion rates inside the SOD chambers were measured at 1.0 to 1.3 mg/L/day in 1977 and 1980. The rates were calculated using initial concentrations versus levels still present on the following sampling date. In 1981, the starting concentration was only 4.1 mg/L, resulting in total depletion by the next sampling. In 1977 and 1980, initial concentrations were 8 to 10 mg/L, and subsequent sampling showed the presence of about 1.0 mg/L. Based on these measurements the sediments were consuming 458 to 596 mg of oxygen from the water column per m^2 of lake bottom per day.

Using Table 2, and assuming an ice thickness of 0.6 m and starting concentration of 10 mg/l, this rate would have produced anoxic conditions within 17-22 days in the pre-deepened Lilly Lake. During one period in the winter of 1977-78 the whole lake oxygen depletion rate was measured at 0.4 mg/L/day (in the absence of oxygen production) which would be equivalent to anoxia within 25 days. The difference between these two methodologies is slight and can be attributed to the colder water temperatures in winter.

Actual inlake depletion rates could not be determined due to the mild winters following the deepening project. However, water volume was increased by a factor of 2.3, and given similar oxygen demand from the sediments, about 58 days would now be required (without oxygen resupply) before anoxic conditions would develop within the lake. Winter fish kills are therefore unlikely in future years.

TABLE 7: Composition of the Lilly Lake Sediments*
(dry weight basis)

Depth Interval:	0–7.6 cm	7.6–15.2 cm	15.2–22.9 cm	22.9–30.5 cm	60–70 cm	90–100 cm	1.5 m	3.0 m	4.6 m	6.1 m	7.6 m	8.5 m
Percent Solids	2.8	3.1	3.5	3.2	2.8	3.3	3.0	-	-	7.2	-	-
pH	6.7	6.9	7.0	7.0	6.7	6.8	6.9	7.4	6.7	6.8	7.0	6.9
Percent Nitrogen	3.31	3.10	3.03	3.15	3.76	3.33	2.53	2.80	2.86	2.96	2.02	1.71
Percent Phosphorus	0.12	0.11	0.10	-	0.15	-	0.07	-	0.06	0.07	0.09	0.09
Percent Potassium	-	-	-	-	-	-	1.64	-	1.23	1.17	1.80	1.89
Percent Calcium	2.80	1.84	1.95	1.58	1.57	1.38	1.04	-	1.94	2.80	1.43	1.93
Percent Magnesium	0.58	0.58	0.52	0.48	0.43	0.50	0.41	-	0.45	1.34	0.45	0.70
Percent Sodium	-	-	-	-	-	-	0.44	-	0.30	0.36	0.37	0.39
Percent Sulfur	1.90	1.76	1.68	1.48	0.39	0.32	-	-	-	-	-	-
Aluminum; ug/g	17,481	18,270	16,851	16,070	10,249	12,299	18,400	-	11,000	16,000	20,500	19,300
Barium; ug/g	198.3	190.7	183.5	171.9	132.2	139.2	223.0	-	185.0	212.0	266.0	317.0
Iron; ug/g	15,206	16,942	15,814	14,575	5,371	6,052	10,200	-	8,020	9,840	14,600	15,200
Strontium; ug/g	40.3	35.0	31.2	28.2	19.6	21.3	18.8	-	20.6	31.3	41.0	50.2
Boron; ug/g	435.0	388.8	385.6	223.0	61.2	155.9	76.0	-	63.4	83.2	102.0	104.0
Copper; ug/g	28.7	39.2	14.4	22.5	11.2	10.5	17.0	-	17.0	15.4	17.0	29.3
Zinc; ug/g	280.6	282.8	216.0	187.4	63.0	38.5	409.0	-	97.4	120.0	89.4	97.4
Manganese; ug/g	143.1	127.8	115.5	104.2	102.9	83.2	217.0	-	125.0	146.0	125.0	146.0
Chromium; ug/g	18.1	16.0	19.0	14.1	-	13.1	34.8	-	35.8	32.2	45.2	52.5

*samples for the bottom six depth intervals were collected February, 1977; the others were taken in June, 1978

Ammonia-N concentrations inside the chambers were monitored in 1977, 1980, and 1981. As shown in Figure 9, the concentrations progressively increased throughout the period of measurement in each year. The average release rate was 29.3 mg/m^2/day in 1977 (see Table 8). Release rates were much lower in the summer following project completion but increased to near pre-project rates again in 1981. The reason for the depressed rate in 1980 is unclear; the interstitial water concentrations for NH_4-N remained about the same, as indicated by the pre-dredging measurement in 1978 versus 1980 and 1981 information (see Table 9). In any case the rate of release was much higher in every year as compared to predictions based on Fick's first law of diffusion:

$$F = -D\ (C/Z)\ \phi$$

where
- D = diffusion coefficient of 3×10^{-6} cm^2/sec (Imboden and Lerman, 1978).
- C = observed concentration difference of -3 ug/cm^3
- Z = distance of 3.8 cm
- ϕ = porosity of 0.99

and therefore F = 2.34×10^{-6} ug/cm^2/sec
or 2.0 mg/m^2/day.

Consequently, the sediments are an important contributor of NH_4-N to the overlying lake waters where it is presumably being quickly converted into NO_3-N.

TABLE 8: Ammonia-N Release Rates from the Sediment (mg/m^2/day)

SOD Chamber	1977	1980	1981
#1	22.6	9.8	26.8
#2	32.3	15.3	22.9
#3	25.3	-	-
#4	36.9	-	-
Average	29.3	12.6	24.9

TABLE 9: Interstitial Water Concentrations of Ammonia-N in Lilly Lake Sediments*

Depth Interval (cm)	1978	1980	1981
0-7.6	2.1	3.8 ± 0.8	2.1 ± 0.8
7.6-15.2	4.7	6.3 ± 1.6	4.5 ± 1.2
15.2-22.9	6.7	8.2 ± 1.2	6.9 ± 1.0
22.9-30.5	7.0	10.9 ± 1.8	9.7 ± 1.2
60-70	12.0	18.2 ± 4.5	19.1 ± 3.7

*In mg/L, and with standard deviation where available

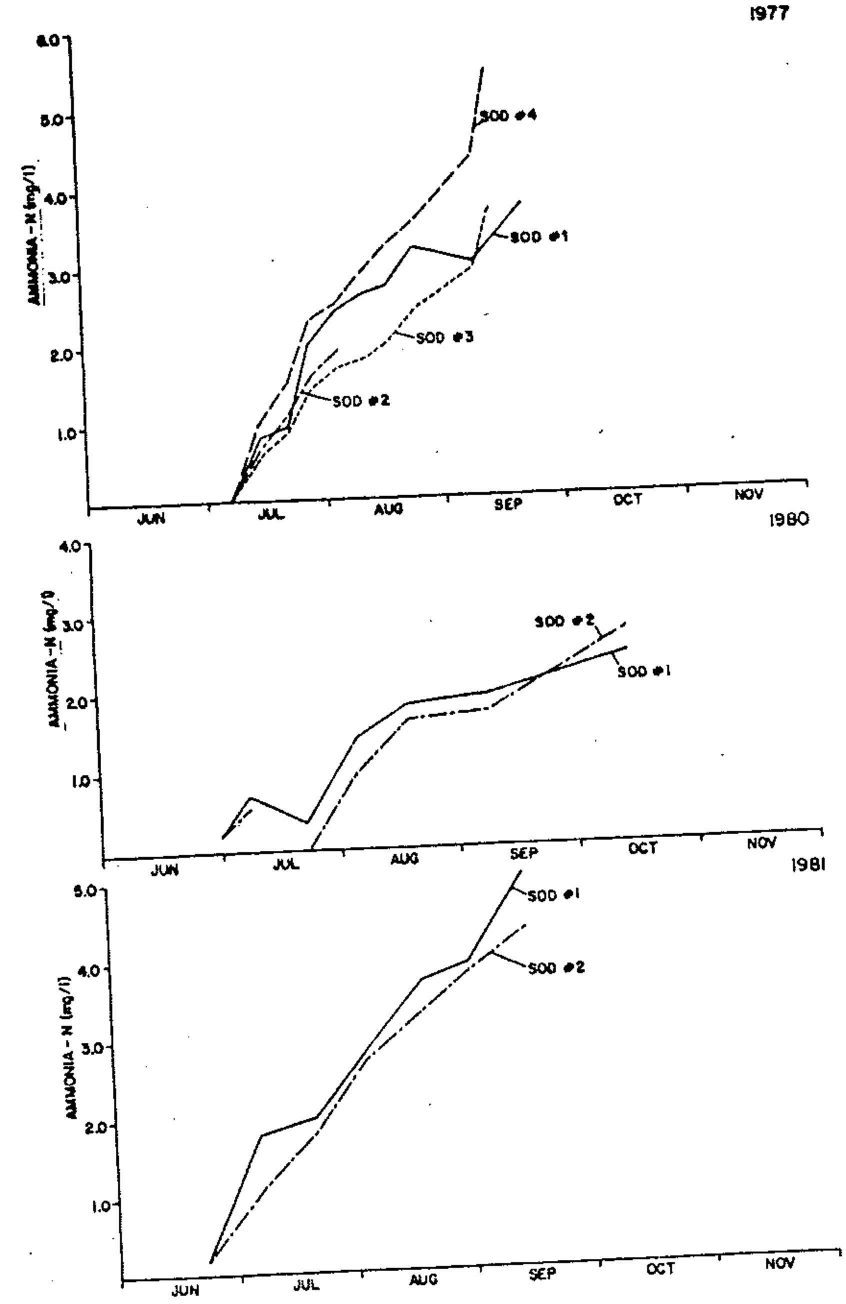

FIGURE 9. Concentration of Ammonia-N Inside the Sediment Oxygen Demand Chambers, 1977-81

7.6 7.6cm held

The sediments also contained a considerable reservoir of P. The top 7.6 cm held about 1,060 kg of P in 1978. Although the P-rich upper sediments were removed by dredging (Table 10), roughly 560 kg were still present in the top 7.6 cm as measured in 1980 and 1981. NaOH and HCl extraction procedures indicated that 53% of the P was in a relatively available state in 1978 in contrast to 34-38% in the two latter years. This was due primarily to the NaOH-P fraction which concentrated in the upper layers (Table 11). The NaOH-P fraction was in general much higher in the upper layers prior to dredging. These sediments were later removed, and NaOH-P levels were low at all depths in 1980 and 1981. There was, however, slight indication of a trend toward increased concentrations in the upper layer by 1981.

TABLE 10: Sediment Phosphorus Content*

Depth Interval (cm)	1978	1980	1981
0-7.6	1240	628 ± 22	667 ± 22
7.6-15.2	1090	610 ± 14	597 ± 12
15.2-22.9	960	573 ± 73	583 ± 35
22.9-30.5	840**	-	-

*In ug/g, with standard deviation where available
**less than

TABLE 11: Concentration of NaOH and HCl Extractable Phosphorus in the Lake Sediments*

Depth Interval (cm)	1978	NaOH-P Fraction 1980	1981
0-7.6	390	13 ± 1	21 ± 3
7.6-15.2	284	13 ± 3	15 ± 3
15.2-22.9	208	12 ± 3	13 ± 3
22.9-30.5	126	13 ± 2	13
60-70	28	14 ± 3	13

Depth Interval (cm)	1978	HCl-P Fraction 1980	1981
0-7.6	270	199 ± 18	230 ± 19
7.6-15.2	138	186 ± 20	195 ± 10
15.2-22.9	153	169 ± 33	195 ± 27
22.9-30.5	95	176 ± 24	174
60-70	-	167 ± 25	179

*In ug/g, with standard deviation where available

The HCl-P content also tended to be higher near the sediment-water interface in 1978, with a reduction in 1980 followed by an increase the next year. But the differences between depths and particularly between years were less pronounced, and sediment removal had a much smaller impact on this constituent.

Two forms of P were monitored in the interstitial waters - SRP and TDP. The results are shown in Table 12. For each parameter in each year the concentration increased with depth. The concentrations at each depth interval remained similar from 1980 to 1981, but there were reduced levels from pre- to post-dredging, at least for TDP. Using Fick's equation, the rate of release for SRP was estimated to be 6.7 X 10^{-4}mg/m^2/day where:

$D = 10^{-6}$cm^2/sec (Stumm and Leckie, 1971)
$C = -3 \times 10^{-3}$ ug/cm^3
$Z = 3.8$ cm
$\phi = 0.99$

Even if this predicted rate is lower than the actual rate by a factor of 10 as was shown for NH_4-N, the P loading to the lake from the sediments would still be less than 1 kg per year.

TABLE 12: Interstitial Water Concentrations of SRP and TDP in Lilly Lake Sediments*

Depth Interval (cm)	1978	Soluble Reactive Phosphorus 1980	1981
0-7.6	-	4**	4**
7.6-15.2	-	4**	4**
15.2-22.9	-	16 ± 20	12 ± 21
22.9-30.5	-	13 ± 9	14 ± 23
60-70	-	19 ± 14	9 ± 5

Depth Interval (cm)	1978	Total Dissolved Phosphorus 1980	1981
0-7.6	105	7 ± 2	6 ± 1
7.6-15.2	128	7 ± 2	7 ± 3
15.2-22.9	118	17 ± 12	16 ± 21
22.9-30.5	198	16 ± 7	19 ± 22
60-70	202	19 ± 7	15 ± 5

*In ug/L, with standard deviation where available
**less than

However, under anoxic conditions inside the SOD chambers the rate of release was much higher. Presumably, this provides some insight into what would happen in the hypolimnion if oxygen levels become depleted during summer stratification. The average release rate was 0.33 $mg/m^2/day$ in 1977 (Table 13). The measured rate was similar in 1981, but for some unknown reason was lower in 1981 (0.09 $mg/m^2/day$). At these rates the sediments would represent a major P source to the lake waters.

TABLE 13: SRP Release Rates from the Sediment ($mg/m^2/day$)

SOD Chamber	1977	1980	1981
#1	0.23	0.14	0.44
#2	0.24	0.03	0.32
#3	0.49	-	-
#4	0.37	-	-
Average	0.33	0.09	0.38

Figure 10 illustrates the concentrations inside the chambers during 1977, 1980, and 1981, and indicates an important difference in the P release before versus after dredging. In 1977 the P concentrations peaked and then stabilized/declined slightly (the release rates in Table 13 apply from startup to peak values only). The concentrations remained below 20 ug/L throughout the summer. In contrast the values continued to increase through the entire summer in 1981 and had reached 60-80 ug/L by the end of sampling in mid-September.

Monitoring for Fe, Mn, bacteria, and pH was also performed within the chambers. The results are shown in Figures 11-14. In general, Fe and Mn tended to increase while pH decreased during the period of isolation. In 1981 bacterial counts increased from initial levels. Presumably this occurred each year but is not well-demonstrated because early samples were not taken in 1977 or 1980.

Hydrology

The hydrologic system for the lake includes inputs from direct precipitation, groundwater, and storm runoff while the outputs involve groundwater and evaporation. There are no streams entering or leaving during normal conditions. The lake is completely landlocked, although as discussed earlier this may not have always been the situation. During 1978 and 1979 there was an additional outflow, the dredge pipeline.

Prior to dredging the lake levels oscillated about 0.6 m, with a normal elevation of 230.4 m (see Figure 15). However, between mid-July and November, 1978 the dredge removed 798,100 m^3 of sediment-lake water mixture. As a result the lake level dropped 1.5 m. During the cessation of dredging activity it again rose 1.2 m. Less material was removed from the lake between May and September, 1979 and the level only fell 1.1 m. After dredging, the lake level continued a gradual increase through 1980. Stabilization at normal elevations was reached in 1981.

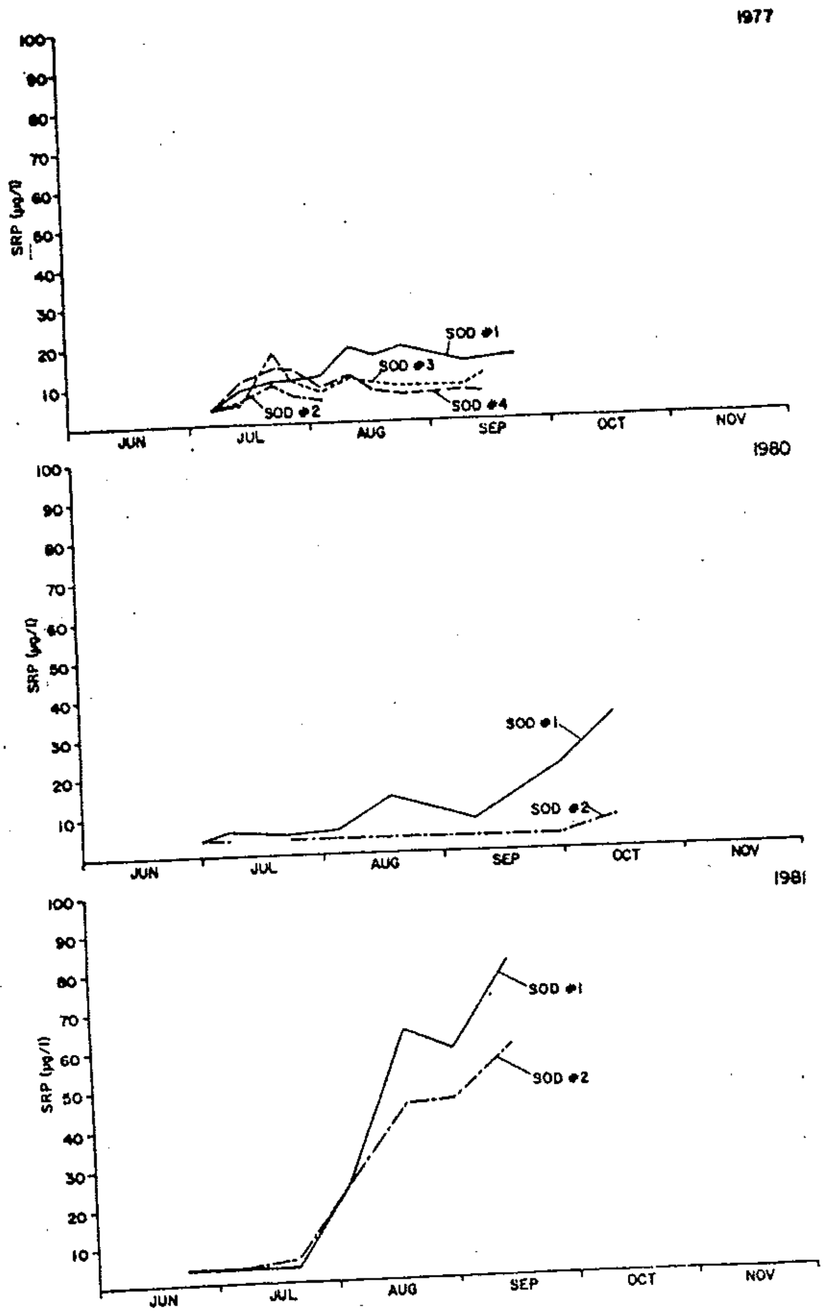

FIGURE 10. Concentration of SRP Inside the Sediment Oxygen Demand Chambers, 1977-81

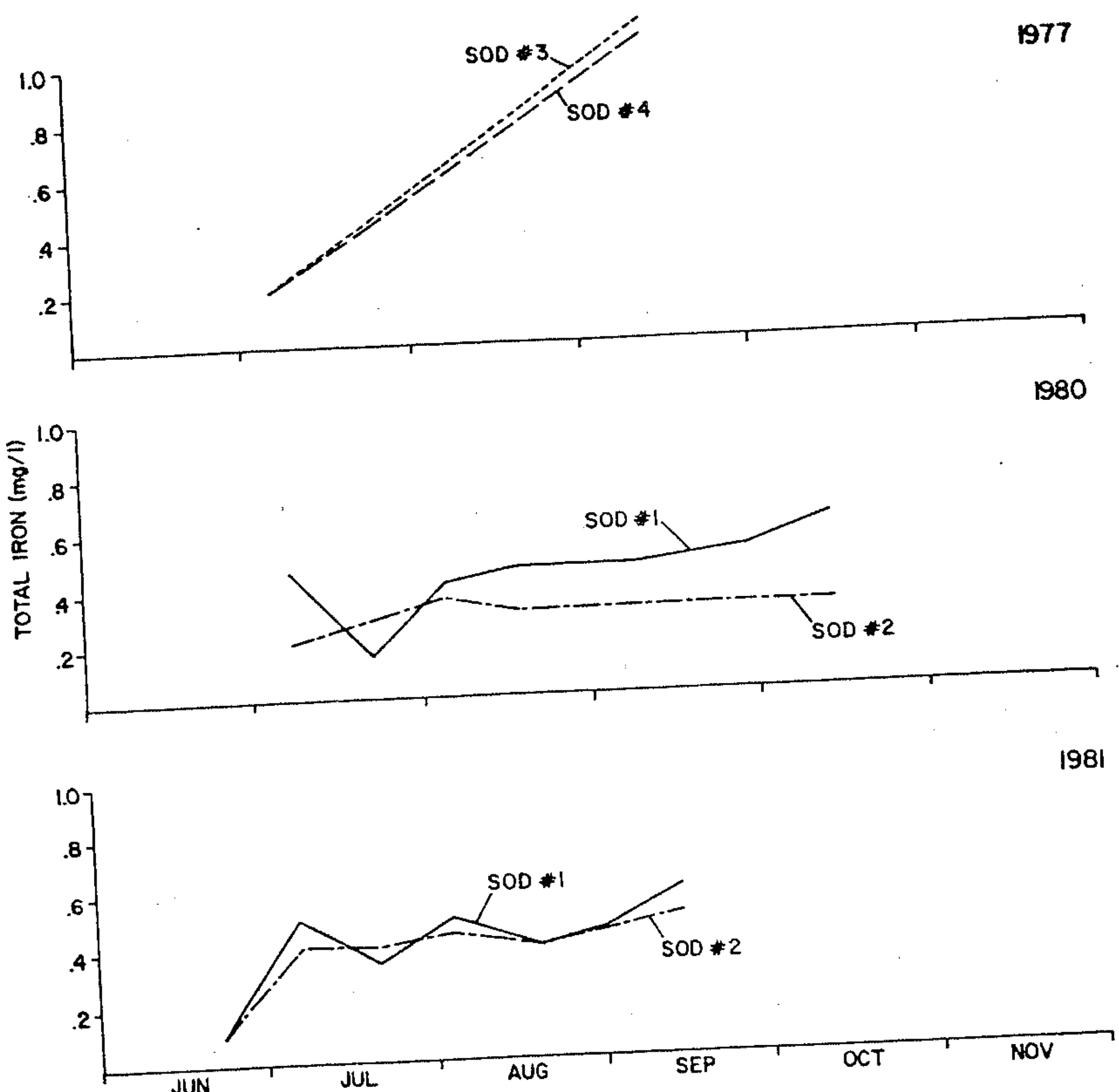

FIGURE 11. Concentration of Total Iron Inside the Sediment Oxygen Demand Chambers, 1977-81

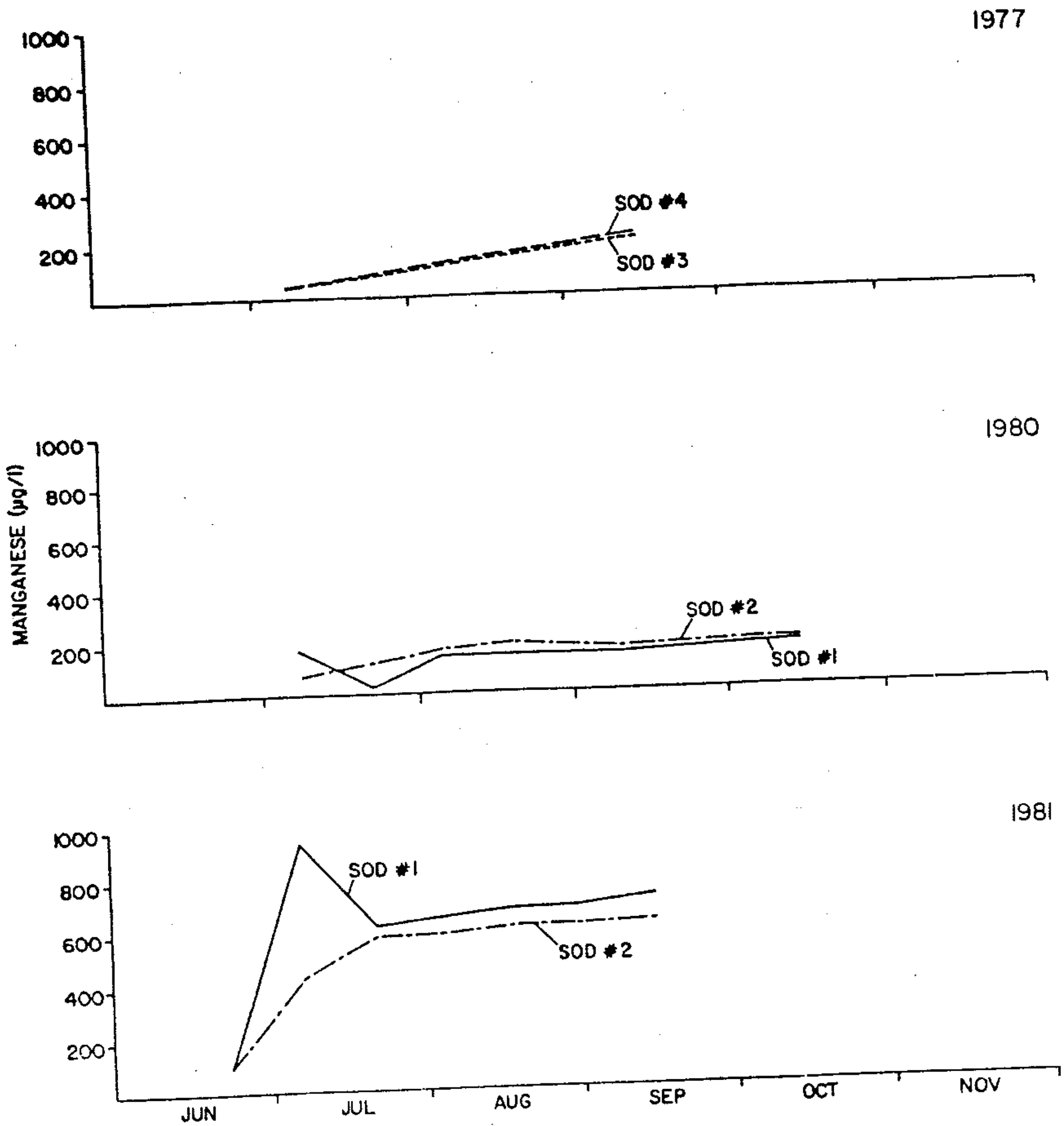

FIGURE 12. Concentration of Manganese Inside the Sediment Oxygen Demand Chambers, 1977-81

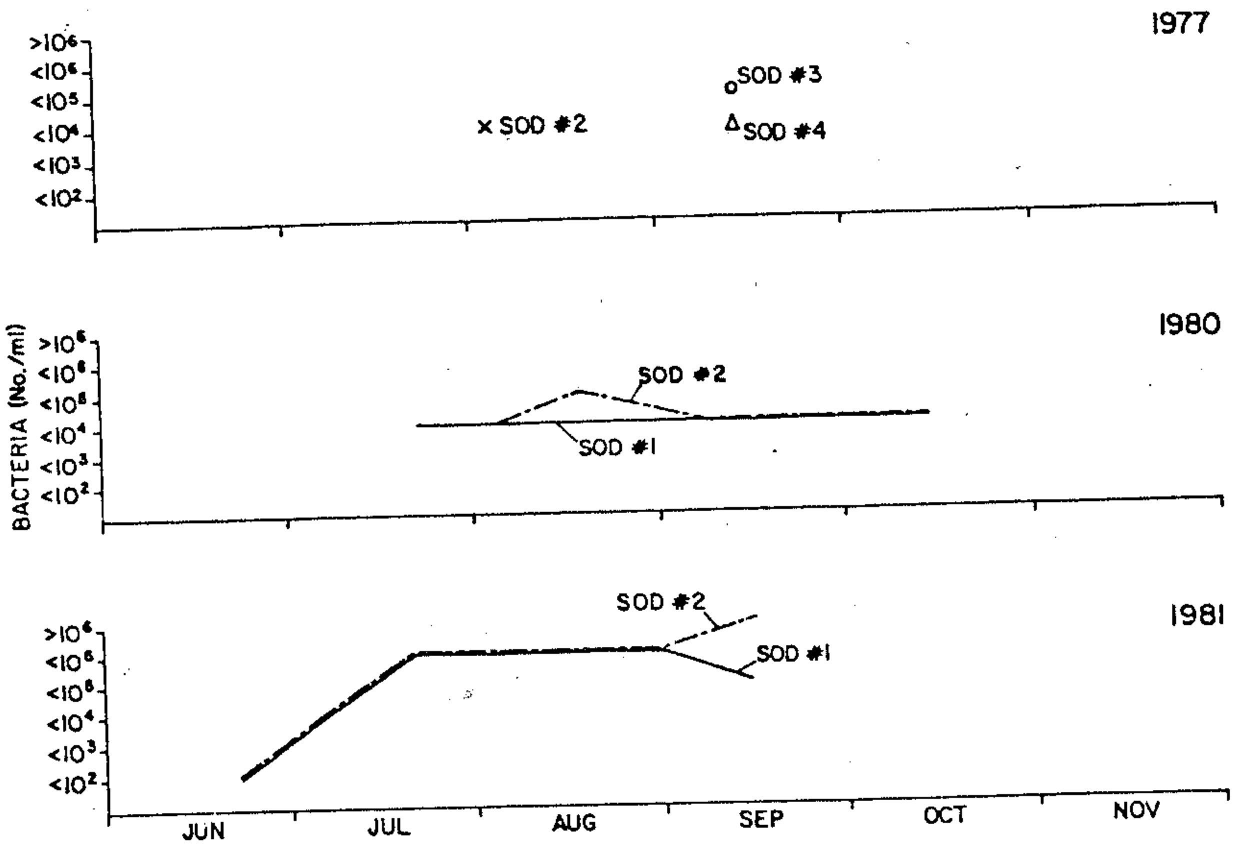

FIGURE 13. Concentration of Bacteria Inside the Sediment Oxygen Demand Chambers, 1977-81

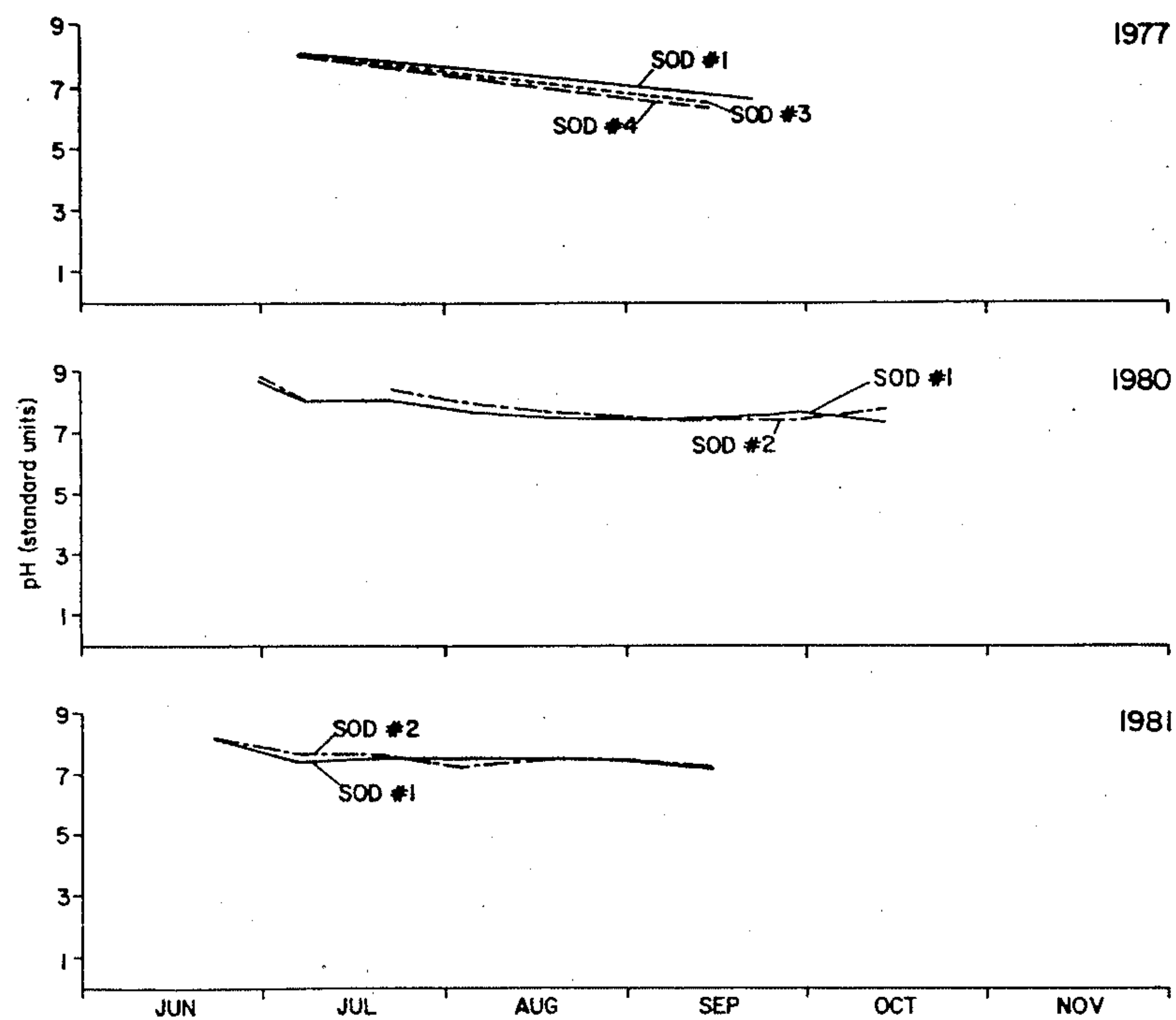

FIGURE 14. pH Changes Inside the Sediment Oxygen Demand Chambers, 1977-81

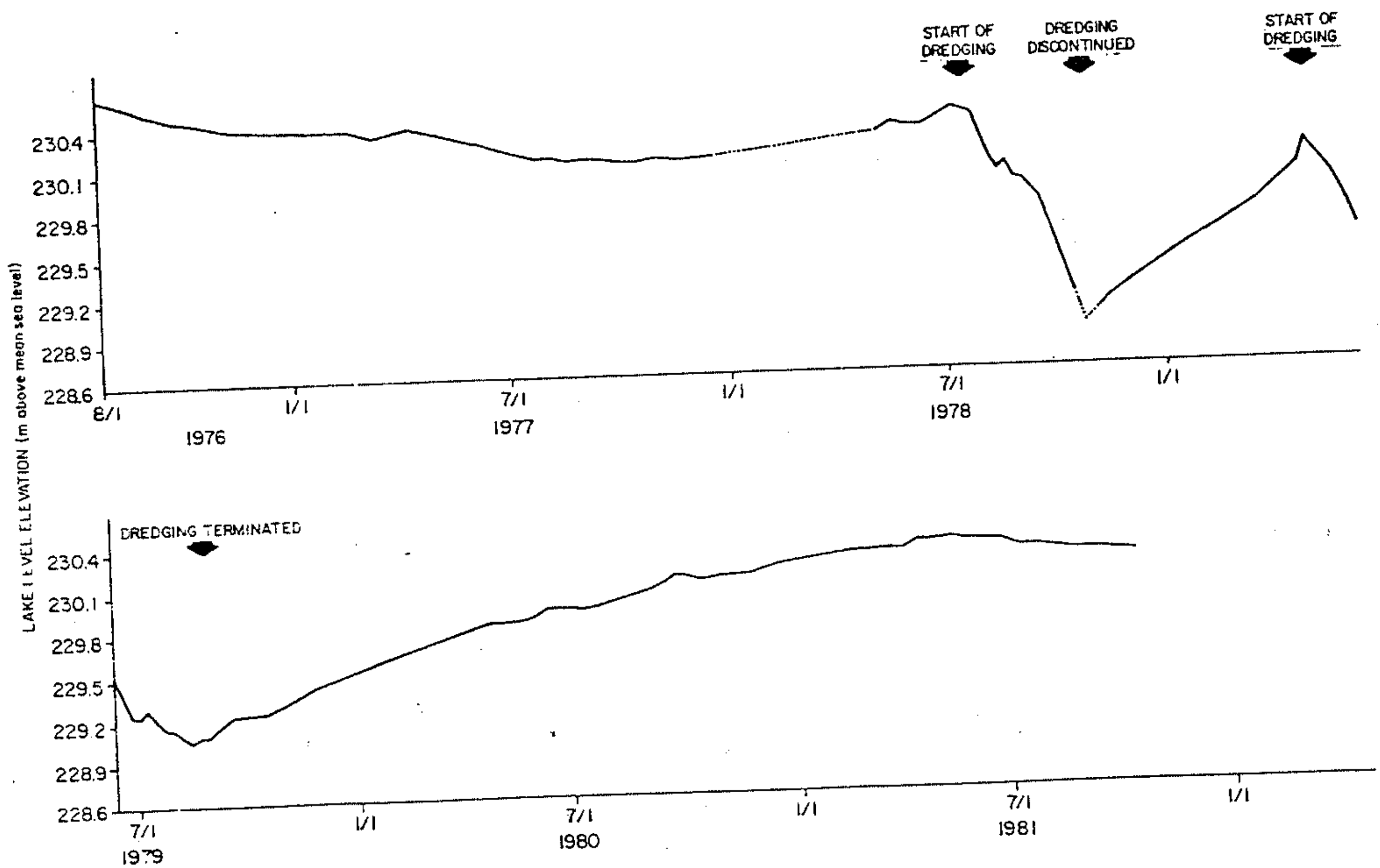

FIGURE 15. Fluctuation in Lilly Lake Water Level, 1976-81

Seepage meters placed equidistant around the lake in the near shore regions were utilized to show the impact of drawdown on the groundwater flow pattern. Before dredging, groundwater inflow was noted along approximately one-half of the shoreline; however, when lake levels began to decline the groundwater system responded with increased flow into the lake around the entire perimeter (including reversal of previous outflow regions, see Table 14). Consequently outflow was negligible from late July, 1978 to spring, 1980, while inflow was roughly 10X higher than pre-dredging values (Table 15).

TABLE 14: Mean Seepage Magnitudes Before and During Dredging*
(ml/m^2/hr)

Seepage Meter	Before	During
1	-7	+87
2	+748	+929
3	+247	+821
4	+8	+26
5	-63	+146
6	-8	+93
7	-122	+458
8	-21	+107
9	+742	+1997

*the seepage meters were used until the water level decline reached about 0.5 m, thereby exposing the sampling sites.

TABLE 15: Annual Water Budget; Lilly Lake,
1977-81*

Inflows (X 10^{6}L)	1977	1978	1979	1980	1981
Precipitation	312	394	361	343	320
Surface Runoff	6	7	7	6	6
Groundwater	65	630	734	301	276
TOTAL	383	1031	1102	650	602
Outflows (X 10^{6}L)					
Groundwater	192			140	344
Evaporation	258			258	258
TOTAL	450			398	602
Change in lake storage (X 10^{6}L)	-67			+252	0

*dredge in operation during 1978 and 1979; measurements taken from the system of monitoring wells were used for the groundwater calculations.

Table 15 also illustrates how deepening the lake and removal of the low permeability organic sediments has altered the groundwater system. In 1977 the groundwater inflow was only about one-third of the outflow volume. And groundwaters contributed 17% of the inflow from all sources. However, in 1981 the inflow was nearly equal to the outflow and supplied about 46% of the water to the lake. Groundwater inflow and outflow both increased post- versus pre-dredging, by 425 and 179%, respectively. This is a result of the dredging project and is expected to be a permanent change as discussed more fully in Beauheim (1980).

Although groundwater has now become a major component of the water inflow to the lake, it still supplies a relatively small share of the P input (Table 16). It now furnishes 9% versus 2% in 1977 but is over-shadowed by each of the other sources. Monitoring of the groundwater observation wells on the inflow side of the lake revealed an average concentration of 6 ug/L of P in the water. This value was used with the inflow estimates in Table 15 to yield the groundwater quantities for Table 16.

TABLE 16: Annual Phosphorus Loading to Lilly Lake

Source (kg)	1977	1980	1981
Surface Runoff	8.3	8.3	8.3
Groundwater	0.4	1.8	1.7
Precipitation	4.7	5.1	4.8
Dry Fallout	4.0	3.8	3.9
TOTAL	17.4	19.0	18.7

Phosphorus values for direct precipitation (15 ug/L) and dry fallout (108 g/ha/yr) were obtained from recent studies at a nearby location (Andren and Stolzenburg, 1978). The values for storm runoff were determined by in-the-field observations during storms and collections of runoff samples for analyses at the SLOH. Runoff was typically limited to an area of about 9228 m^2. The quantity of water discharge from the area was established from the record of direct precipitation adjusted downward as per the expected runoff coefficient for that type of land surface. There were two subareas with coefficients of 0.8 and 0.5 for 7,078 and 2,150 m^2, respectively. Phosphorus analyses of the water indicated an average concentration of 1.39 mg/L with a range of 0.13-4.76 mg/L. This procedure generated an estimate of 8.3 kg/yr for storm runoff, which is in close agreement with the value (e.g., 7.3 kg/yr) resulting from multiplying the area in urban and forest times the appropriate published P loss coefficients for those land use practices (Uttormark et al., 1974).

The water and P loadings were next used with various predictive models (see Table 17 plus Sakamoto, 1966) to estimate the inlake water quality in terms of chlorophyll a concentration. All of the models predicted low chlorophyll a levels both before as well as after lake deepening. However, the Vollenweider (1976) model in particular is sensitive to the changed hydraulics resulting from lake deepening. This model indicated that chlorophyll a levels would be cut in half by lake deepening. Improved water quality was anticipated even without an increase in groundwater inflow. In actuality the chlorophyll a concentrations have not been reduced by 50%, although the levels were low and near the prediction for 1981.

TABLE 17: Inlake Chlorophyll a Levels - Actual Versus Predicted

	1977	1980	1981
Lake Size (ha)	37	34.1	36.4
Lake Volume (m^3)	532,300	896,800	1,149,200
Phosphorus Loading (kg)	17.4	19.0	18.7
Water Loading (m^3/sec)	0.012	0.021	0.019
Regressed Phosphorus Retention Coefficient	0.89	0.82	0.84
Predicted Chlorophyll a by:			
Dillon and Rigler, 1974	0.7	0.8	0.8
Vollenweider, 1975	0.6	0.7	0.6
Vollenweider, 1976	6.0	3.1	3.0
Bachmann and Canfield, 1979	6.3	3.7	3.6
Reckhow et al., 1980	0.5	0.5	0.5
Actual Chlorophyll a (ug/L)	3.3	5.9	5.0*
Adjusted Phosphorus Retention Coefficient	0.70	0.28	0.4 - 0.5
Predicted Chlorophyll a by: Dillon and Rigler, 1974 (ug/L)	3.3	5.8	3.9 - 5.1
Hydraulic Residence Time (yrs)	1.4	1.4	1.9

*less than

The Dillon and Rigler (1974) model is not sensitive to the changed hydraulics; however, it is affected by the P retention coefficient. The Vollenweider (1976) model, in contrast, is not affected by the coefficient. Because the Dillon and Rigler model has proven useful in other lake studies in Wisconsin's lake management program, this model was used with the retention coefficient adjusted as necessary to yield chlorophyll a predictions identical to the concentrations actually found in the lake. The resultant coefficients were 0.7, 0.28, and 0.4-0.5 for 1977, 1980, and 1981, respectively. Manipulation of the coefficient is not appropriate under the assumptions inherent to this model; however, the calculations illustrate the difference between expected (regressed) and necessary (adjusted) P retention coefficients for a precise estimation of chlorophyll a levels found in this macrophyte dominated lake.

Inlake Water Quality

As indicated in the methods section a wide diversity and quantity of chemical information was collected on this project. Most of the analyses were conducted at the SLOH, and all of these results were filed on the U.S. EPA STORET system. Therefore, the reader has potential access to each individual value and only the most pertinent tabulations are presented herein.

Basic water quality at spring overturn is given in Table 18. In general the lake can be categorized as hardwater with moderate fertility. The concentrations for each parameter are within the range expected for a lake in southeastern Wisconsin (Lillie and Mason, 1983). Although the values are, therefore, "normal" 1977 through 1981, dredging did have a noticeable impact on water quality. After dredging activities were completed, concentrations were increased for the following parameters: Mg, Na, K, Sp. C, Cl, and TA. Values were also higher at spring overturn for SO_4, NO_2/NO_3-N, and NH_4-N, but these apparent increases were not observed during other sampling periods. Total phosphorus was the only constituent that was reduced in 1980 and 1981 versus the pre-dredging years.

TABLE 18: Inlake Water Chemistry During Spring Overturn*

	Pre-Dredging		Post-Dredging	
	1977	1978	1980	1981
Calcium	25	32	56	31
Magnesium	14	15	23	23
Sodium	1**	4	8	8
Potassium	0.5**	1.0	2.1	1.9
Conductivity	245	260	560	350
Sulfate	-	9	16	15
Chloride	8	7	15	18
pH	7.8	8.5	8.2	8.4
Alkalinity	102	128	182	150
Turbidity	4.1	0.8	1.7	0.7
Nitrite/Nitrate-N	0.02**	0.02**	0.19	0.05
Ammonia-N	0.04**	0.02**	0.81	0.11
Organic N	0.61	1.0	1.1	0.9
SRP	9	4**	4**	4**
TDP	-	-	6	4
TP	25	20	16	9

*units in mg/L except: 1) SRP, TDP, and TP in ug/L; 2) pH in Standard units; 3) conductivity in umho/cm at 20°C; and 4) turbidity in Formazin units.

**less than

A. Temperature .

Prior to dredging, the lake mixed continuously to its 1.8 m depth throughout the summer. Temperature changed rapidly in response to climatic conditions. Maximum water temperature reached 30°C in mid-lake.

Since dredging, the greater storage appears to have dampened the summer temperature maxima; the highest post-dredging temperature has been 27°C (observed in both 1980 and 1981). The lake is still polymictic, although mixing is not continuous (Figure 16). In 1980 slight temperature gradients were occasionally present but mixing occurred through most of the summer. In 1981, the gradients were more persistent and of greater magnitude, up to 8°C. There was, however, no evidence of a stable thermocline, and the bottom temperature still reached 24°C (versus 27°C in 1980).

B. Dissolved Oxygen

Before dredging there was a history of low winter D.O. levels, resulting in significant fish kills. This was due to the small water storage capacity of the lake (e.g. shallow water) and the high content of organic matter in the sediments. A major die-off was noted during the winter of 1977-78. Since completion of the dredging project, D.O. concentrations have not dropped below 7 mg/L at any depth during the winter (Figure 17).

Although this was a major dredging operation on a relatively small lake, D.O. values were not lowered during the summer. The disturbance of highly organic sediments in a reduced state might have been expected to stress the lake's oxygen content. However, oxygen reduction was not observed in either 1978 or 1979. Apparently, the processes of oxygen input to the lake waters (e.g. atmospheric and photosynthetic) superseded the effect of dredging. In this case a 30 cm diameter cutterhead dredge was used by an experienced operator.

In 1980 the lake waters were generally mixing from top to bottom and the D.O. levels were also similar throughout the water column. In contrast, as a result of the reduced degree of mixing in 1981 the bottom D.O. dropped to 0.6 mg/L on one occasion. Although this existed for a short period only and oxygen was resupplied by the following sampling date, the measurement indicates that depletion will occur if the lake becomes dimictic.

C. pH and Alkalinity

Lilly Lake is a hardwater calcareous lake that is fairly well-buffered. The pH generally tends to be higher in the summer and lower in winter (Figure 18). The surface water values ranged from 7.5 to 8.9 during the period of record. These levels are satisfactory and pose no problem for aquatic life in the lake. During summer, pH values will now drop near the bottom as oxygen becomes depleted; however, due to the natural buffering capacity of the lake, pH values are not anticipated below approximately 7.0.

In contrast to the pH, TA concentrations tend to be highest in winter and lower in summer (Figure 19). During the summer, photosynthetic activity causes utilization of carbon dioxide, resulting in the precipitation of calcium carbonate and a drop in TA. In the winter, carbon dioxide is still produced through sediment decomposition; however, there is little photosynthetic activity, and the ice cover prevents exchange with the atmosphere. The increase in carbon dioxide in the water causes dissolution of calcium carbonate from the bottom muds and an increase in TA. The equation is:

$$CaCO_3 + CO_2 + H_2O \rightleftharpoons Ca^{+2} + 2HCO_3^{-1}$$

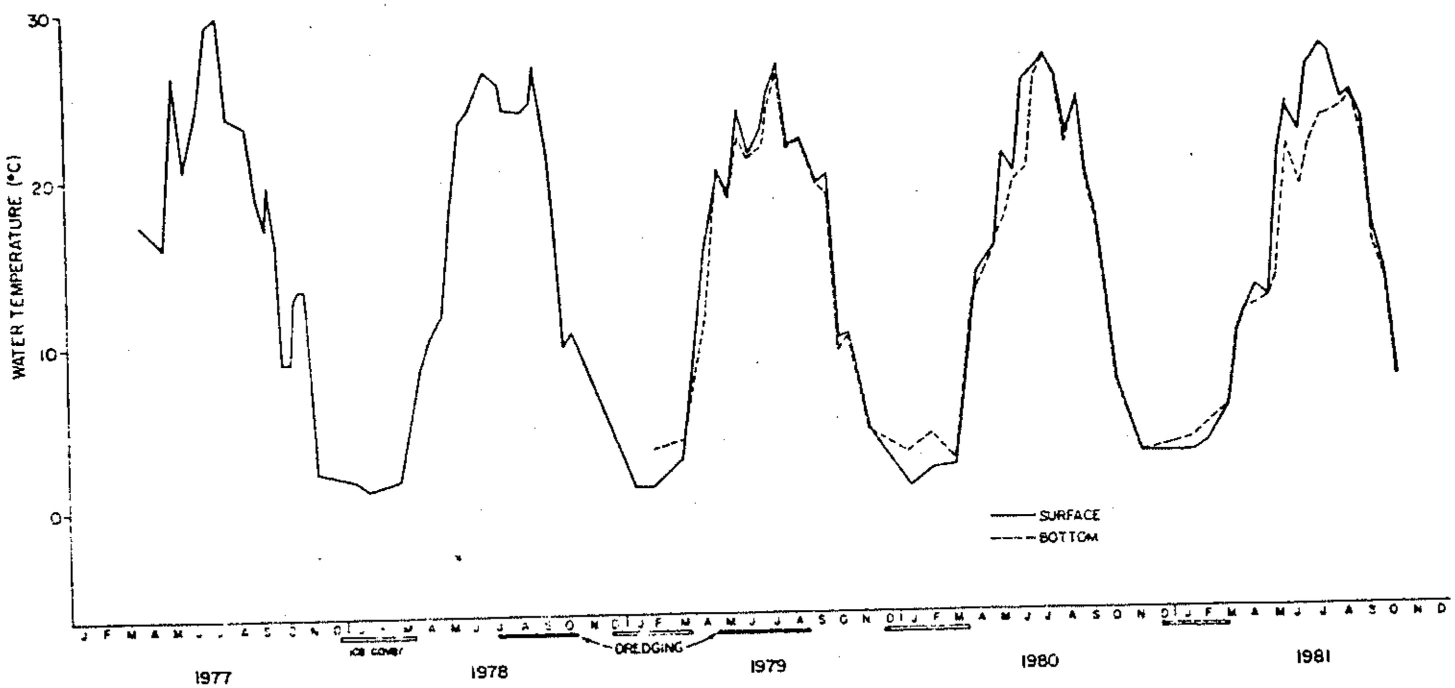

FIGURE 16. Fluctuation in Water Temperature at Lilly Lake, 1977-81

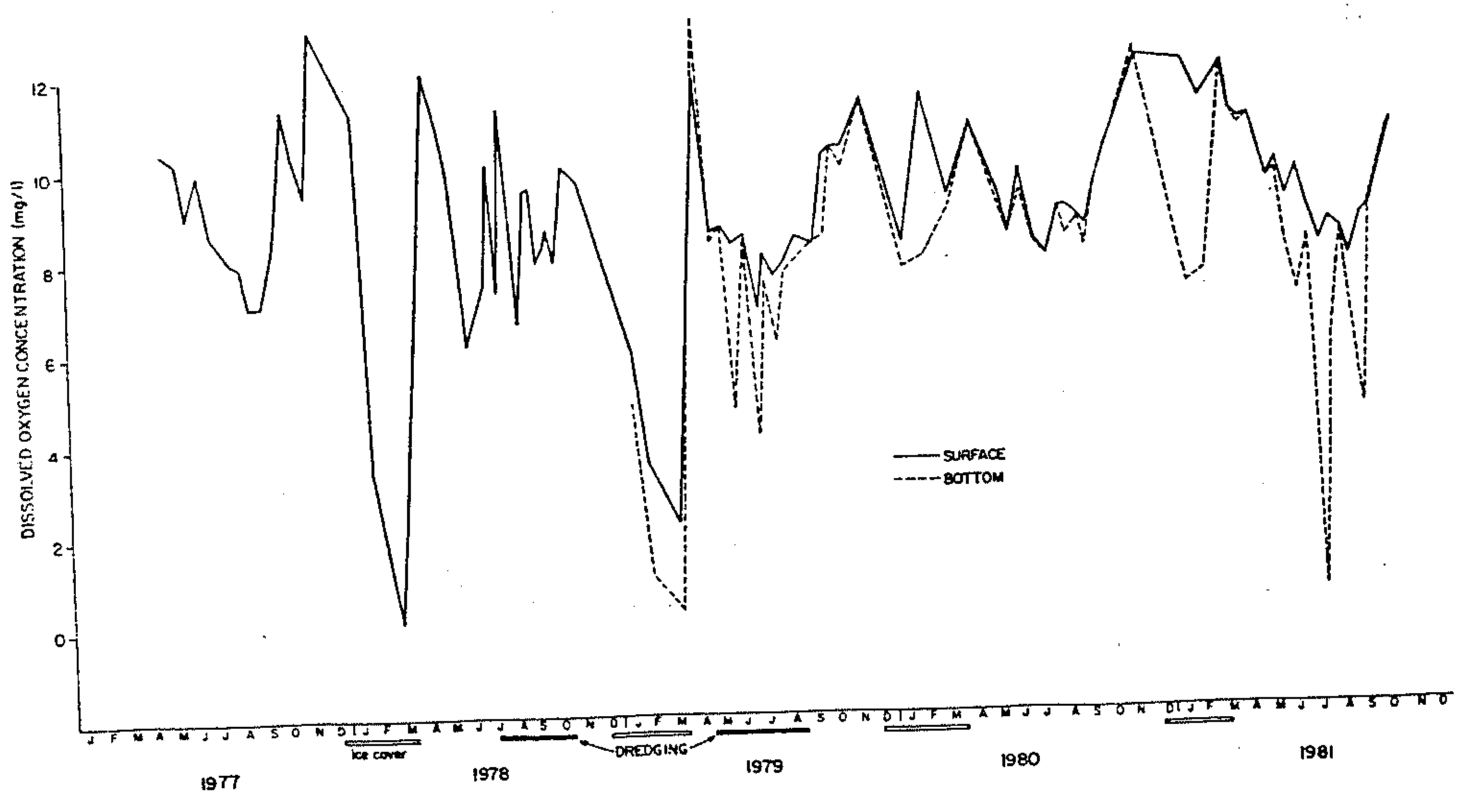

FIGURE 17. Fluctuation in Dissolved Oxygen, Lilly Lake, 1977-81

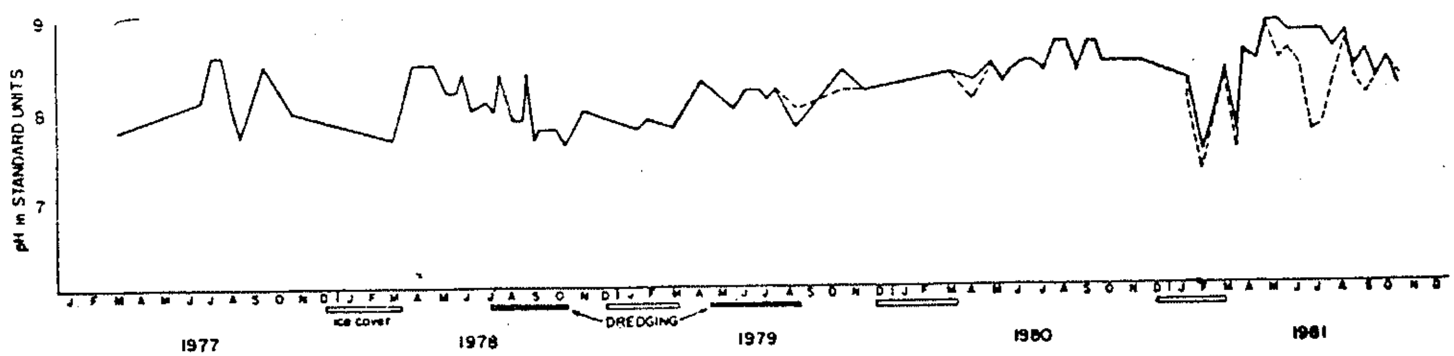

FIGURE 18. Fluctuation in pH, Lilly Lake, 1977-81

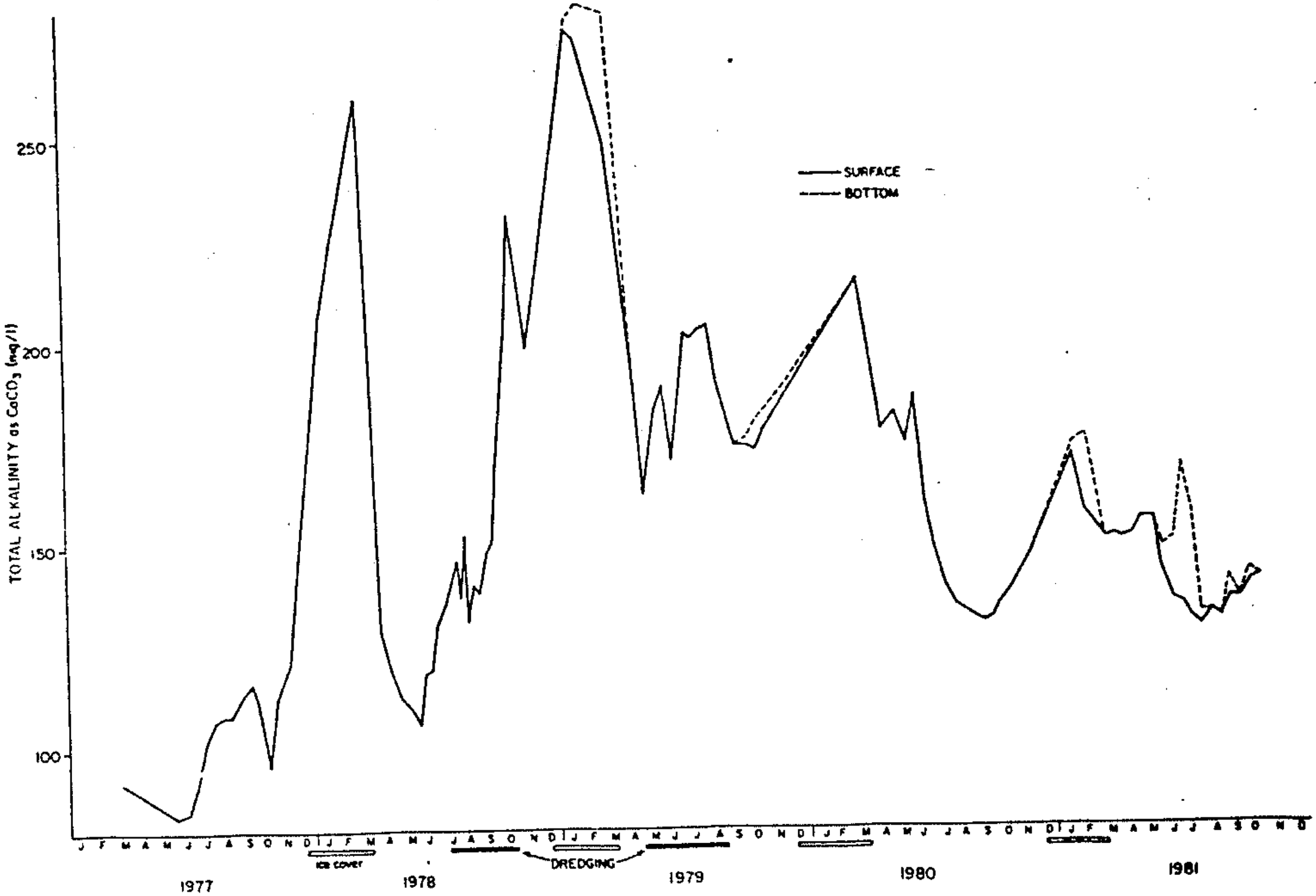

FIGURE 19. Fluctuation in Total Alkalinity, 1977-81

A comparison of TA values before and after dredging shows much higher concentrations during the winters of 1977-78 and 1978-79. This is presumably due to the low water volume to sediment ratio before completion of the lake deepening project. Any calcium carbonate released from the sediment in recent and future winters will be diluted by the now increased water volume within the lake. An examination of summer concentrations also shows a difference pre- versus post-dredging. The levels are now higher, probably due to the increased groundwater inflow. The groundwater has a higher TA than the lake water, and, because low permeability sediments were removed by dredging, there has been a significant increase in the rate of inflow.

D. Nitrogen

Organic N concentrations were about 1 to 2 mg/L through the entire 1977-81 study period. The levels tended to be slightly increased during the first dredging season (1978); however, the significance was minimal. There were major changes in the N content of the lake water during dredging, but this was due to NH_4-N (Figure 20).

In the first year of dredging, NH_4-N levels increased from near zero to almost 6 mg/L (NO_2/NO_3-N concentrations were below 0.1 mg/L). The disturbed sediments were the primary source of this N form because pre-dredging investigations had shown: 1) interstitial sediment NH_4-N concentrations up to at least 19 mg/L, and 2) NH_4-N release rates from the sediment of 22-37 mg per m^2 of lake bottom per day. The high levels persisted through the first winter and then gradually declined, returning to pre-dredging levels about one year after completion of the project. It appears that NH_4-N concentrations will now remain low except when D.O. levels approach zero.

Despite the high NH_4-N concentrations in 1978-79, fish survival was not threatened. The short duration lethal level of unionized NH_4-N is 0.2-2.0 mg/L for a variety of fish species (U.S. EPA, 1976). Using the pH and temperature measurements, the highest unionized ammonia level was calculated to be 0.13 mg/L in May, 1979 (calculation procedure described in Thurston et al., 1974). This was well below the toxic level. The existing fish population did not appear to be stressed at any time during the dredging activities.

E. Phosphorus

Lilly Lake was low in SRP prior to dredging (Figure 21). Summer values were 4 ug/L or less (the analytical procedure had a detection limit of 4 ug/L). Later measurements revealed little change either during or after the dredging project. The only rise occurred during the winter of 1978-79.

Pre-dredging measurements of TDP showed concentrations up to at least 200 ug/L in the sediment interstitial waters. Most of this was presumably in the inorganic form (e.g. SRP). Initially this would have been released into the water column during dredging as was noted for NH_4-N. However, much of this disappeared quickly due to the oxidizing conditions present in the water column. Also, during the summer, biological uptake would have been rapid. The increase to 12 ug/L in February, 1979 was apparently due to the lack of biological uptake at that time of year.

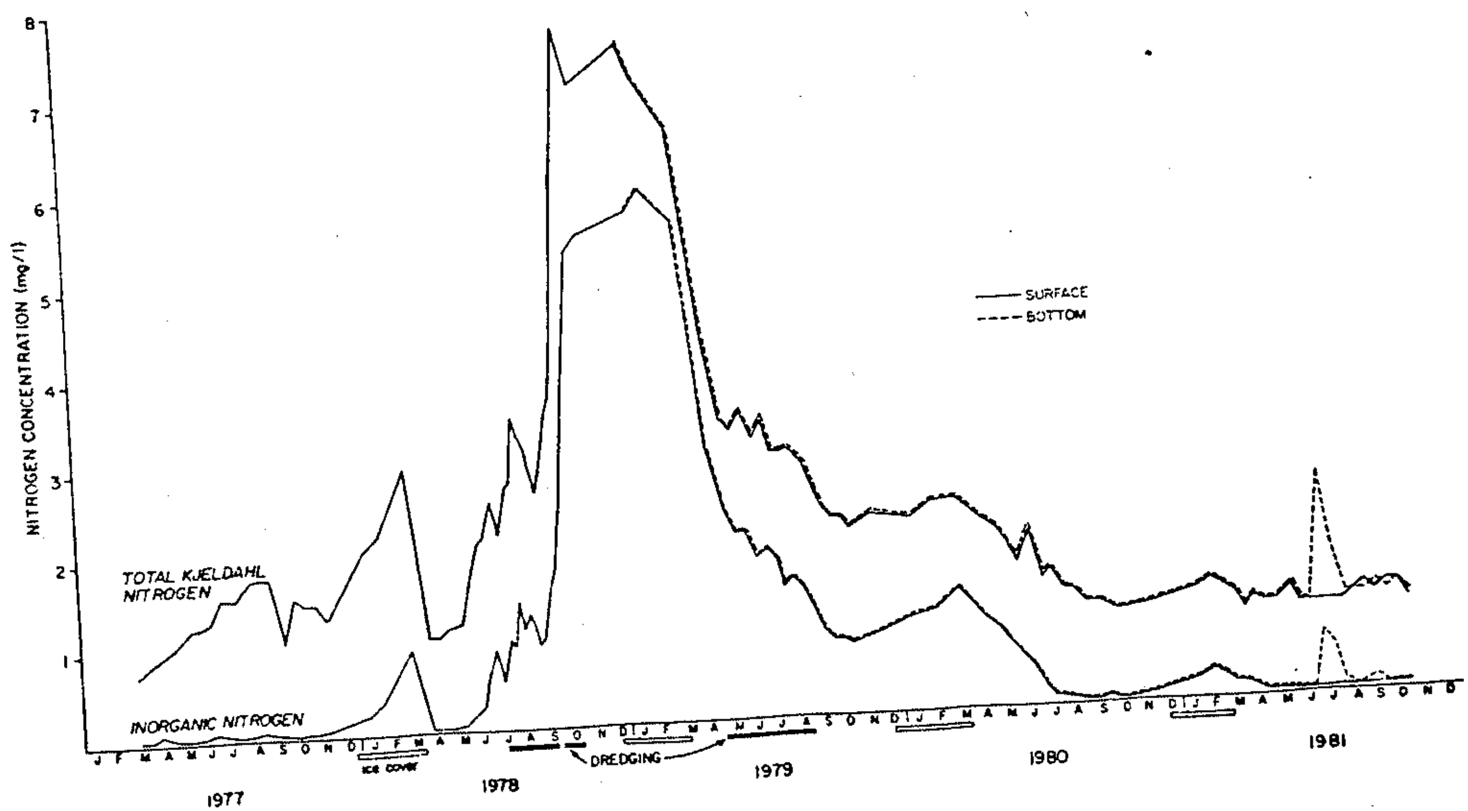

FIGURE 20. Fluctuation in Nitrogen, Lilly Lake, 1977-81

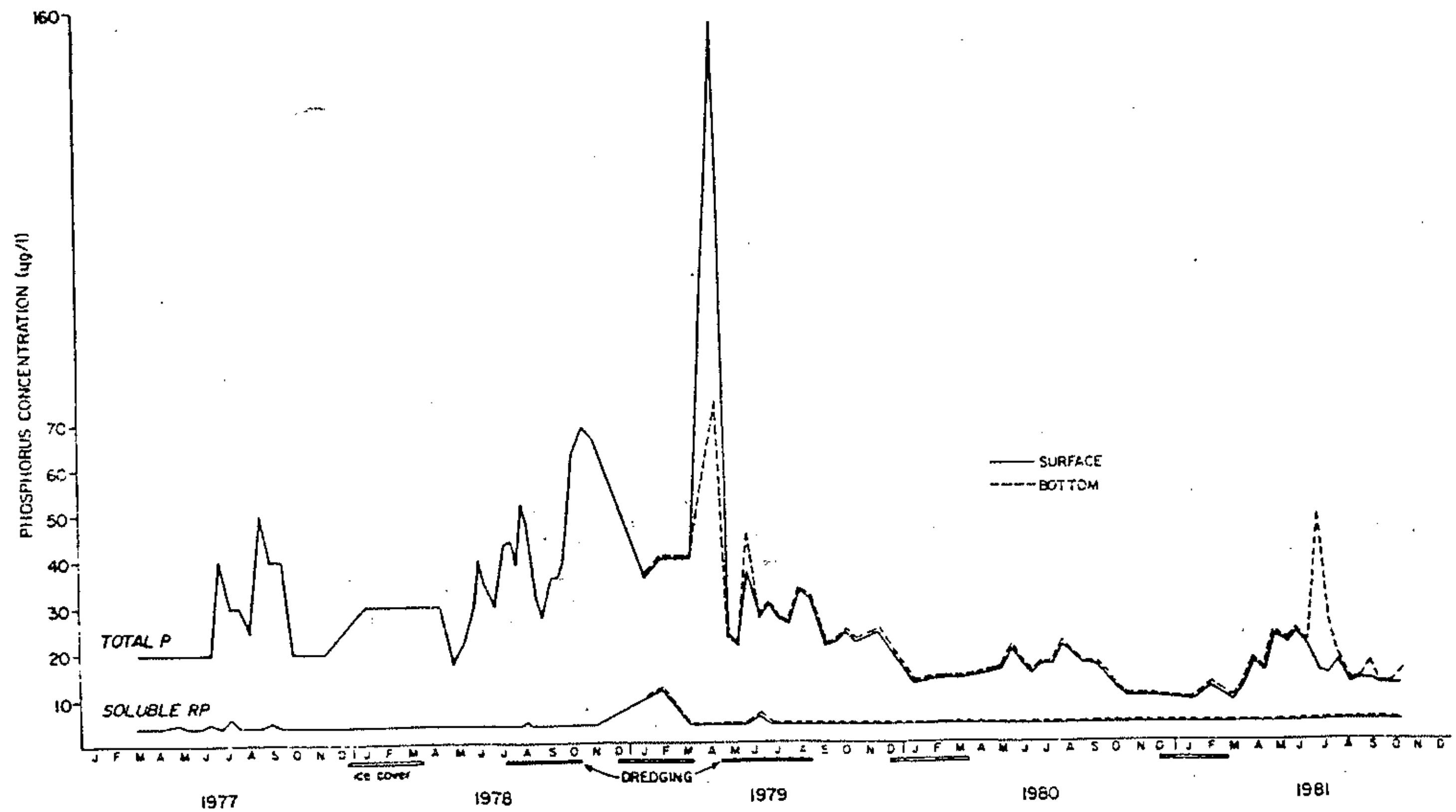

FIGURE 21. Fluctuation in Phosphorus, Lilly Lake, 1977-81

In contrast to SRP, there was an initial increase in TP during the first year of dredging. This reflects the release of dissolved phosphorus via sediment disturbance and its rapid transformation into the particulate form. The P in the water column existed as phytoplankton tissue, inorganic phosphorus compounds, and phosphorus sorbed on any suspended sediments. As similarly noted for NH_4-N, high release of P did not occur during the second dredging season. With the exception of one sampling date prior to the initiation of dredging in 1979, TP gradually declined after 1978, and levels fell below pre-dredging concentrations within one year of project completion.

The impact of dredging on inlake water quality is summarized in Table 19. As discussed earlier, the deepened lake had increased concentrations of Sp. C and TA while TP decreased. However, there were some additional changes observed during the dredging activity itself. Ammonia-N, Sp. C, TA, and BOD (5 day) values were higher in 1978 and/or 1979 than either before or after dredging. Although TP concentrations showed an increase in 1978 (Figure 21), the average July/September value was statistically similar to 1977. These results indicate that dredging was causing increased groundwater inflow (Sp. C and TA) and sediment disturbance (BOD and NH_4-N). The latter impact would be considered a negative, albeit temporary effect. Nevertheless, this direct consequence of dredging activity was not sufficiently great to degrade the water quality below criteria (or standards) established for aquatic life (U.S. EPA, 1976).

TABLE 19: Inlake Conditions During July/September, 1977-81*

	1977	1978	1979	1980	1981
Nitrite/ Nitrate-N	0.02**	0.03 ± 0.01	0.05 ± 0.01	0.05 ± 0.04	0.01
Ammonia-N	0.03 ± 0.01	1.11 ± 0.33	1.44 ± 0.32	0.05 ± 0.03	0.02 ± 0.01
Organic N	1.53 ± 0.25	1.82 ± 0.15	1.38 ± 0.10	1.17 ± 0.13	1.09 ± 0.09
SRP	4**	4**	4**	4**	4**
TDP	19 ± 5	10 ± 1	9 ± 2	7 ± 1	5 ± 1
TP	36 ± 9	40 ± 7	28 ± 4	18 ± 2	14 ± 2
Conductivity	247 ± 13	317 ± 22	433 ± 23	337 ± 16	327 ± 7
pH	8.3 ± 0.4	8.0 ± 0.3	8.1 ± 0.2	8.6 ± 0.1	8.6 ± 0.2
Alkalinity	107 ± 8	142 ± 7	196 ± 12	135 ± 6	132 ± 3
Turbidity	1.2 ± 0.5	3.0 ± 0.8	4.1 ± 0.7	2.6 ± 1.7	2.0 ± 2.9
BOD(5 day)	1.7 ± 0.2	3.6 ± 0.6	2.0 ± 0.1	1.5 ± 0.1	1.2

*4 to 11 analyses were conducted for each parameter each year; units are in mg/L except: 1) phosphorus forms are in ug/L, 2) pH is in Standard units, 3) turbidity is in Formazin units, and 4) conductivity is in umho/cm at 20°C.

**less than

Aquatic Biota

Biotic monitoring included phytoplankton, zooplankton, benthos, macrophytes, and fish, although emphasis was placed on the two vegetative groups.

Prior to dredging, summer chlorophyll a levels were low in the lake. In 1976 and 1977, the concentration ranged from 0 to 8 ug/L, with July/September averages of 2.5 and 3.3 ug/L, respectively (Table 20). During May/June 1978, the levels were 2 to 7 ug/L with a mean of 4.6 ug/L. However, after dredging began, phytoplankton densities increased greatly. The chlorophyll a concentration jumped to 33 ug/L on July 17, 1978 and averaged 18.5 ug/L through September (Figure 22).

TABLE 20: Inlake Chlorophyll a Levels (ug/L)

Time Period	Average Concentration	Range In Concentration
July/September, 1976	2.5	0 - 7
July/September, 1977	3.3	2 - 8
May/June, 1978	4.6	2 - 7
July/September, 1978	18.5	10 - 33
July/September, 1979	9.5	6* - 13
July/September, 1980	5.9	5* - 8
July/September, 1981	5*	5* on 7 dates

*less than

In 1979 dredging continued, but the average chlorophyll a fell to 9.5 ug/L (range of about 6 to 13 ug/L; July/September). The concentrations continued to decline in 1980 and 1981 in response to the lowered P levels. The July/September average chlorophyll a was 5.9 and under 5.0 ug/L in 1980 and 1981, respectively (laboratory procedures used in 1981 had a detection limit of 5.0 ug/L). Measurements of gross primary productivity indicate a similar trend, although algal productivity did not continue to decline in 1981 versus 1980 (Table 21).

TABLE 21: Gross Primary Productivity of Phytoplankton in Lilly Lake (mg C/m^3/day)

Time Period	Average Value
July/September, 1976	185
July/September, 1977	140
May/June, 1978	260
July/September, 1978	1,005
July/September, 1979	395
July/September, 1980	225
July/September, 1981	270

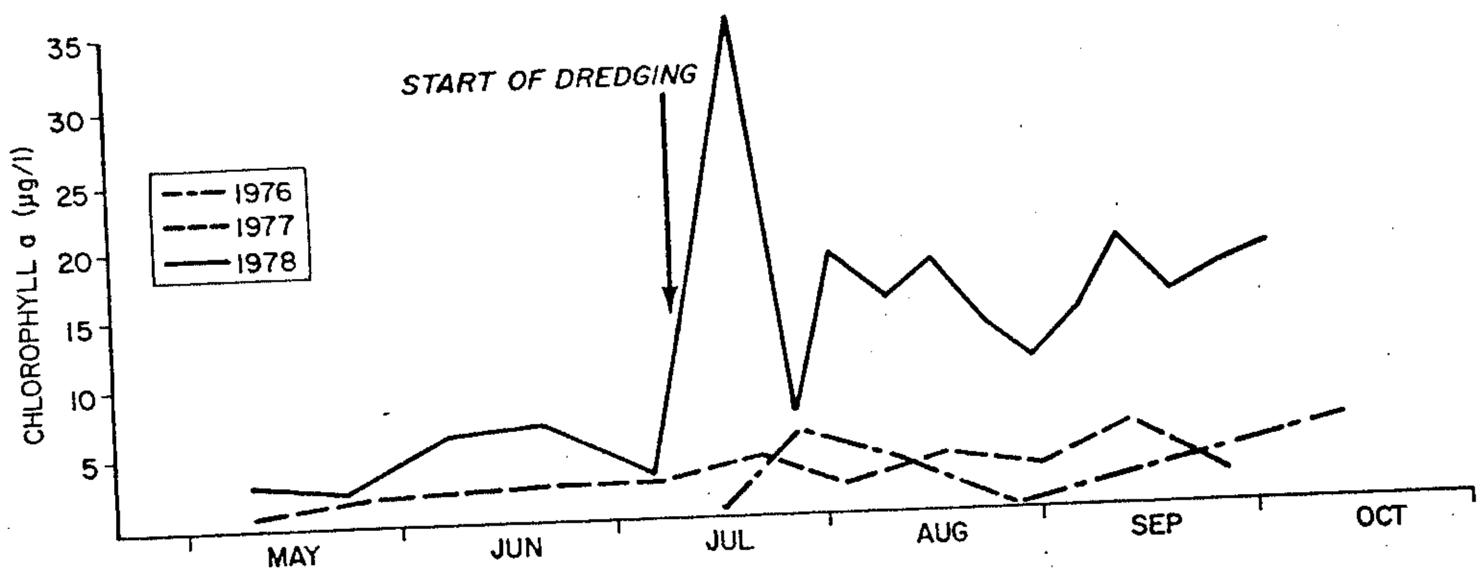

FIGURE 22. Inlake Chlorophyll a levels, 1976-78

Twenty-two genera of phytoplankton were known to be present in the lake during May/June, 1978 (Table 22). The numbers and biomass ranged from 2,600-29,100 cells/ml and 420,000-2,850,000 u^3/ml, respectively. Chlorophyceae (green algae) were predominant, comprising over 70% of the algal community. After dredging began in 1978, the numbers and biomass eventually peaked at 111,400 cells/ml and 16,290,000 u^3/ml, respectively; however, diversity and relative composition exhibited little change.

TABLE 22: Phytoplankton Genera Present; May/June, 1978 Versus 1980

Phytoplankton	1978	1980
Chlorophyta		
Chlorophyceae		
Carteria sp.	X	-
Cerasterias sp.	X	-
Chlamydomonas sp.	X	X
Chlorella sp.	X	-
Cosmarium sp.	-	X
Crucigenia sp.	X	X
Elakatothrix sp.	X	-
Gloeocystis sp.	X	X
Kirchneriella sp.	X	X
Lagerheimia sp.	-	X
Oocystis sp.	X	X
Pediastrum sp.	X	-
Planktosphaeria sp.	X	-
Scenedesmus sp.	X	X
Selenastrum sp.	-	X
Sphaerocystis sp.	X	X
Tetraedron sp.	X	X
Chrysophyta		
Bacillariophyceae		
Cyclotella sp.	X	X
Navicula sp.	X	X
Nitzschia sp.	X	-
Synedra sp.	X	X
Chrysophyceae		
Chrysococcus sp.	-	X
Dinobryon sp.	-	X
Erkenia sp.	-	X
Kephyrion sp.	-	X
Monosiga sp.	-	X
Ochromonas sp.	-	X
Pyrrhophyta		
Cryptophyceae		
Chroomonas sp.	X	X
Cryptomonas sp.	X	X
Cyanophyta		
Myxophyceae		
Anabaena sp.	X	-
Chroococcus sp.	-	X
Coelosphaerium sp.	X	X

Phytoplankton were again examined in detail in May/June, 1980. At that time densities were 9,000-60,200 cells/ml for numbers and 610,000-3,080,000 u^3/ml for biomass, and 24 genera were found in the samples. Therefore, overall diversity and density were very similar to pre-treatment conditions in 1978. There was, however, some shift in relative abundance between algal groups. As shown in Table 22, six genera of Chrysophyceae appeared in the community in 1980 and, although numerically inferior to the green algae, they contributed a nearly equal biomass.

The zooplankton community was evaluated 1977 through 1979. Sixteen species were found in 1977 as compared to 21 in 1978 and 17 in 1979. These included two species not previously collected in Wisconsin, Mesocyclops leuckarti and Daphnia laevis (Mace, 1979). Table 23 provides a comparison of population levels for July/September between the three years. Much year to year fluctuation occurred, making evaluation difficult. The only conclusions that can be made with some confidence are: 1) a major dredging operation in a small shallow lake did not stress the zooplankton community, and 2) dredging initially caused a rapid population growth for Bosmina longirostris (see Figure 23). This was presumably a result of the increased algal densities as discussed earlier.

TABLE 23: Zooplankton Present in Lilly Lake; July/September, 1977-79
(number/liter)

Zooplankton	1977	1978	1979
Cladocera			
Sididae:			
Diaphanosoma leuchtenbergianum	1	1	26
Daphnidae:			
Ceriodaphnia sp.	20	11	21
Daphnia spp. (4)	pres.	pres.	pres.
Bosminidae:			
Bosmina longirostris	56	274	81
Macrothricidae:			
Macrothrix laticornis	--	9	pres.
Chydoridae:			
Acroperus harpae	pres.	pres.	pres.
Pleuroxus denticulatus	--	pres.	pres.
Chydorus sphaericus	2	12	5
Eurycercus lamellatus	pres.	pres.	pres.
Copepoda			
Cyclopidae:			
Macrocyclops albidus	pres.	pres.	pres.
Tropocyclops prasinus	5	9	pres.
Mesocyclops spp. (2)	pres.	pres.	pres.
Eucyclops spp. (2)	--	4	pres.
Diaptomidae:			
Skistodiaptomus oregonensis	2	pres.	10
Copepodids	10	23	22
Naplii	57	27	22
Ostracoda	pres.	--	7

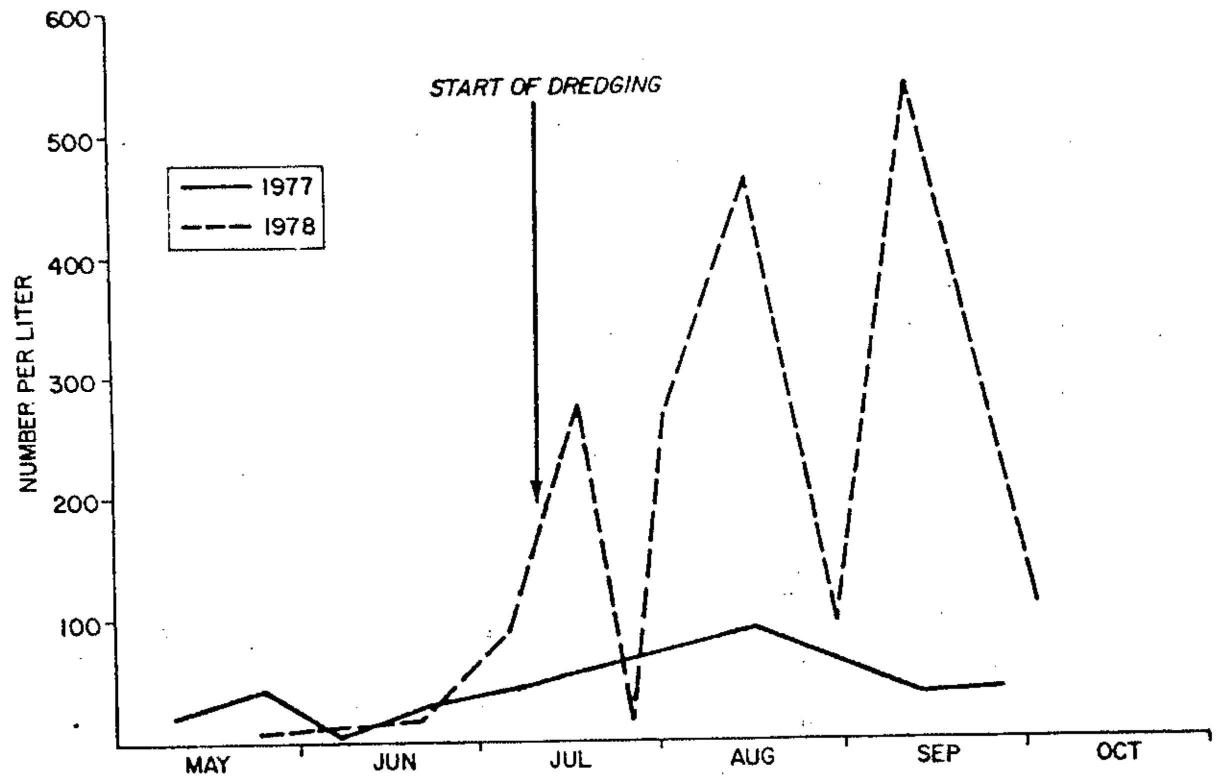

FIGURE 23. Population of _Bosmina_ _Longirostris_, 1977-78

Benthic invertebrates were collected during spring overturn for three years - before dredging began in April, 1977 and 1978, and after dredging was completed and the lake refilled in April 1981. As shown in Table 24, both species diversity and numerical abundance increased in the deepened lake. Major shifts in abundance included, in particular, Hyalella azteca, Pericoma sp., Caenis sp., Oligochaeta, and chironomids.

The population changes can be related to either surface sediment removal directly and/or improved winter D.O. levels as an indirect benefit of deepening (Hilsenhoff, 1975; McCafferty, 1982; and Pennak, 1953). Hyalella azteca is a bottom deposit feeder, and its decline is probably due to removal of the more readily digestible surficial plant material. On the other hand, Pericoma sp. is tolerant of low D.O. conditions, and its disappearance was related to the improved water quality. The pronounced increases in Caenis sp. and chironomids were also most likely due to this factor. Their greater abundance should benefit the fish population as they are considered good food organisms. The increase in Oligochaetes is apparently due to the same factors, but these organisms were not speciated and in general less is known about their habitat requirements.

Originally, planned fishery investigations included establishment of a species list and calculation of growth rates prior to, during, and following dredging. Changes in growth could then be related to habitat alterations. However, severe fish kills in the winters of 1976-77 and 1977-78 resulted in a major population decline and prompted plan cancellation. The following fish population statistics are, therefore, presented to describe pre- dredging conditions only.

A species list was generated from observations in March, 1977 of dead fish along the shoreline. This survey was conducted immediately after ice-out and included the collection of scale samples from the primary fish species for age-growth analyses. Eleven species were identified, and these were characteristic of a shallow, weedy, warmwater lake (Table 25). Panfish, especially bluegills, dominated the population. Only two gamefish species were present, with largemouth bass being the most abundant. Young fish were not found for either of the gamefish species, indicating poor natural reproduction. A comparison of growth rates with expectations shows that the growth of bluegills was very poor, suggesting an overabundance due to the dense rooted vegetation (Table 26).

Dredging has resulted in greater depth, increased abundance of fish food organisms, improved winter D.O. levels, higher TA and Sp. C concentrations, and in general better water quality and habitat. Predictive methodologies indicate that this will lead to the establishment of an improved fishery in the lake; however, verification will not be possible for several additional years, until the re-estabilization of the fish community.

Because of the shallow depth and high water transparency, macrophyte growth covered the entire lake bottom prior to dredging. Most of the lake surface was also invaded by early to mid-summer. A diversity of plant species was present, but the community was heavily dominated by Potamogeton Robbinsii. This growth was, however, completely disrupted during the dredging activities.

TABLE 24: Benthic Invertebrates Present in Lilly Lake (Individuals/m^2)

Invertebrate	April 1977 Sand	April 1977 Muck	April 1978 Sand	April 1978 Muck	April 1981 Sand	April 1981 Muck
Mollusca						
Gyraulus sp.	0	0	0	1	1	1
Helisoma sp.	0	0	0	1	0	0
Physa sp.	0	0	1	0	0	0
Sphaerium sp.	0	0	4	0	0	0
Oligochaeta	0	0	1	0	16	0
Amphipoda						
Hyalella azteca	20	4	2	21	7	0
Hirudinea	0	0	0	2	0	0
Hydrachnidae	0	0	0	1	0	2
Coleoptera						
Rhantus sp.	1	0	0	0	0	0
Ephemeroptera						
Caenis sp.	0	0	0	3	12	24
Odonata						
Gomphidae	2	0	0	0	0	0
Trichoptera						
Agraylea sp.	0	0	0	0	0	4
Oecetis sp.	6	0	0	4	0	0
Nectopsyche sp.	10	2	0	0	0	0
Diptera						
Psychodidae						
Pericoma sp.	0	0	22	0	0	0
Tipulidae						
Tipula sp.	2	0	0	0	0	0
Chaoboridae						
Chaoborus albatus	0	1	5	0	0	9
Chaoborus punctipennis	0	0	0	0	0	2
Chironomidae						
Dicroptendipes sp.	0	4	0	5	0	1
Einfeldia sp.	0	0	0	0	1	0
Harnischia sp.	0	0	0	0	0	1
Microchironomus sp.	0	0	0	0	0	25
Parachironomus sp.	0	0	0	0	0	1
Paralauterborniella sp.	0	0	0	0	0	1
Paratanytarsus sp.	0	0	0	0	92	8
Polypedilum sp.	0	0	0	0	0	2
Procladius sp.	0	0	0	2	0	3
Rheotanytarsus sp.	0	0	0	0	0	1
Stempellinella sp.	0	0	0	0	0	20
Stictochironomus sp.	0	0	0	0	135	2
Tanytarsus sp.	0	0	0	0	0	29
Chironomus sp.	2	0	0	0	0	0
Clinotanypus sp.	0	1	0	0	0	0
Ablabesmyia sp.	1	2	0	0	0	0
Conchapelopia sp.	0	1	0	0	0	0
Glyptotendipes sp.	0	1	0	0	0	0
Cryptochironomus sp.	0	2	0	0	0	0
Psectrocladius sp.	0	4	0	0	0	0
Ceratopogonidae						
Probezzia sp.	0	0	4	2	4	1
TOTAL	44	22	39	42	268	137

TABLE 25: Fish Species Present; March, 1977

Species	Common Name
Lepomis macrochirus	Bluegill
Micropterus salmoides	Largemouth bass
Chaenobryttus gulosus	Warmouth
Pomoxis nigromaculatus	Black crappie
Ictalurus natalis	Yellow bullhead
Esox lucius	Northern pike
Perca flavescens	Yellow perch
Morone interrupta	Yellow bass
Lepomis cyanellus	Green sunfish
Ambloplites rupestris	Rock bass
Catostomus commersonnii	White sucker

TABLE 26: Age-Growth Analyses for the Primary Fish Species*

Age	Bluegill	Black Crappie	Largemouth Bass
III	8.9 (16.0)		
IV	11.2 (18.5)	17.5 (23.6)	33.8 (29.7)
V	13.0 (19.8)	24.4 (25.7)	37.8 (32.5)
VI	14.7 (21.8)	28.7 (25.7)	38.6 (37.8)
VII	16.5 (24.4)	32.5 (32.0)	41.7 (41.1)
VIII			43.4 (42.7)
IX			46.2 (46.7)
X			49.0 (49.3)

*Length in centimeters. Sampled March 1977. Expectation based on Druckenmiller (1972) in parentheses.

In 1980, the first year after termination of dredging, the lake was quickly reinvaded by vegetation. A mixture of species was present; however, *Chara* sp. was by far the dominant plant. This macrophytic algae is often a pioneer species in new aquatic habitat. Growth occurred in about 25% of the lake, primarily around the shoreline out to a depth of roughly 3 m (Figure 24). This was in agreement with expectations based on the following relationship between water transparency and maximum depth of weed growth:

$$X = 0.83 + 1.22\,Y$$

where X = depth of growth in meters
and Y = water transparency in meters

(Dunst, 1982)

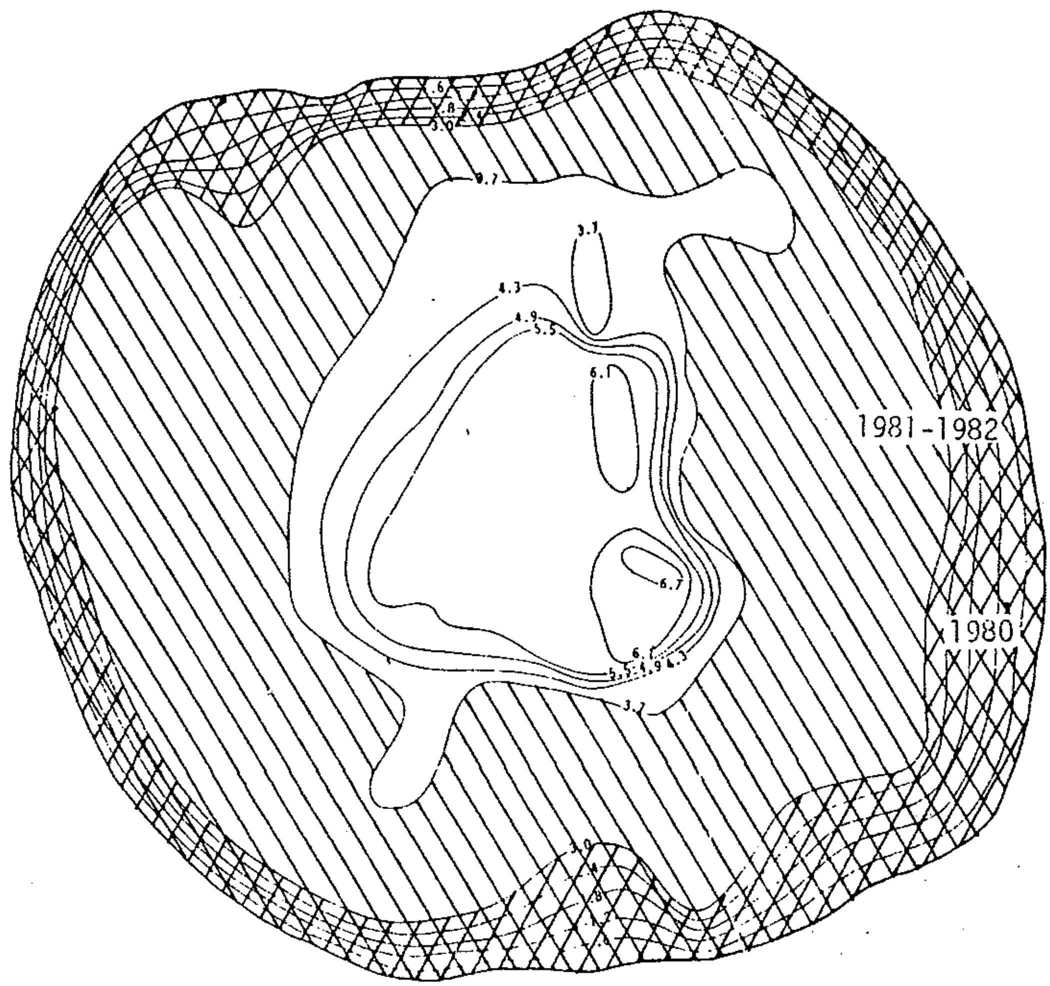

FIGURE 24. Extent of Macrophyte Growth in Lilly Lake, 1980-82

In 1981, the transparency was 2.2 m, predicting a maximum depth of growth of 3.5 m. The actual depth was about 3.7 m and, therefore, growth covered about 75% of the lake bottom. Chara sp. was still abundant, but Myriophyllum spp. was nearly co-dominant. Growth was within 0 to 0.6 m of the lake surface over the entire area of habitation. The vegetation was also examined periodically in 1982. At that time, the area of growth was similar to 1981, but the macrophytes remained 1.2-1.8 m below the lake surface except near the shoreline.

The macrophyte community was composed primarily of submergent species throughout the 1976-82 period of study. No floating leaf species were observed in the lake. Some emergent species, such as Sagittaria sp. and Scirpus sp. were present, especially after dredging, but the number of plants and their biomass was negligible. In contrast, there were at least 12 submergent species in 1976 and 1977, with a combined lakewide weighted average biomass of up to 840 g/m^2 (dry weight basis) on one sampling date in 1976 (Table 27). During dredging in 1978 and 1979 the entire lake bottom was affected, causing a total disruption of the macrophyte community. As a result only six species were found in the lake in 1980. A new species of pondweed - Potamogeton foliosus - was present, but all of the pre-dredging pondweeds along with Megalodonta Beckii were absent. In 1981-82, species diversity increased somewhat; however, it remained below 1976-77 levels. P. foliosus was no longer present, and M. Beckii and Potamogeton amplifolius had still not reinvaded the lake.

TABLE 27: Submergent Plant Species Present in Lilly Lake Before and After Dredging

	Before		After		
Species	1976	1977	1980	1981	1982
Potamogeton Robbinsii	X	X	-	X	X
P. amplifolius	X	X	-	-	-
P. praelongus	X	X	-	X	X
P. illinoensis	X	X	-	X	X
P. pectinatus	X	X	-	X	-
P. zosteriformis	X	X	-	X	-
P. foliosus	-	-	X	X	-
Megalodonta Beckii	X	X	-	-	-
Myriophyllum spp. (spicatum & exalbescens)	X	X	X	X	X
Elodea canadensis	X	X	X	X	X
Heteranthera dubia	X	X	X	-	X
Chara sp.	X	X	X	X	X
Najas flexilis	X	X	X	X	X
TOTAL	12	12	6	10	8

Biomass was also much lower after completion of the dredging project. The lakewide weighted average biomass was 685 and 335 g/m^2 in 1976 and 1977, respectively. However, this dropped to about 100 g/m^2 in 1980 and 1981. As a result, macrophytes now are less dominant in influencing inlake dynamics. This is indicated by the noon to 6 P.M. D.O. measurements. In 1977 the inlake concentrations increased by an average of 1.6 mg/L during this period, only 0.1 mg/L of which could be attributed to phytoplankton. In 1981 the average inlake increase was 0.2 mg/L, a value comparable to the change noted in the light bottle (e.g. algal productivity as measured by D.O. content in suspended light and dark bottles).

Rooted macrophytes were impacted by several changes brought about by the dredging project. These include the water quality and sediment chemistry as discussed earlier. In addition:

1) 25% of the lake is now deeper than the maximum depth of growth possible due to prevailing water clarity,

2) almost all of the near shore region has been converted from muck to sand bottom, and

3) *Chara* sp., although a macrophyte, is an alga and not a rooted plant. This species provided much of the total biomass in 1980-81 but contains a high inorganic content versus the typical rooted species; therefore, the 100 g/m^2 is somewhat misleading when compared to the biomass estimates for 1976-77. Also, this is considered a pioneer species and is expected to gradually diminish in importance in future years.

The shift in macrophyte species dominance is illustrated for muck and sand bottom in Figures 25 and 26, respectively. The change from *Potamogeton Robbinsii* in 1977 to *Chara* sp. in 1980 and then *Myriophyllum* spp. in 1981 is very pronounced over muck bottom which still occurs over roughly 75% of the lake bottom. *Chara* sp. remains dominant in the sand areas, but *Myriophyllum* spp. was becoming more important by 1981. The biomass of macrophytes over muck versus sand bottom (with and without *Chara* sp.) is shown in Table 28. Without the presence of *Chara* sp., the lakewide weighted average of 100 g/m^2 would be much lower. Also, muck bottom is clearly able to support higher biomass of rooted macrophytes than sandy areas.

TABLE 28: Macrophyte Biomass in Lilly Lake - Sand Versus Muck Bottom*

	With Chara				Without Chara			
Depth (m):	0-0.5	0.5-1.5	1.5-2.5	Over 2.5	0-0.5	0.5-1.5	1.5-2.5	Over 2.5
1976								
Muck	790	898	659	-	790	898	659	-
Sand	38	71	-	-	38	71	-	-
1977								
Muck	345	432	335	-	345	432	335	-
Sand	76	125	-	-	76	125	-	-
1980								
Muck	-	283	-	9	-	82	-	2
Sand	194	485	-	-	10	49	-	-
1981								
Muck	-	309	292	94	-	210	190	75
Sand	49	237	224	-	9	50	18	-

*units in g/m^2 on a dry weight basis.

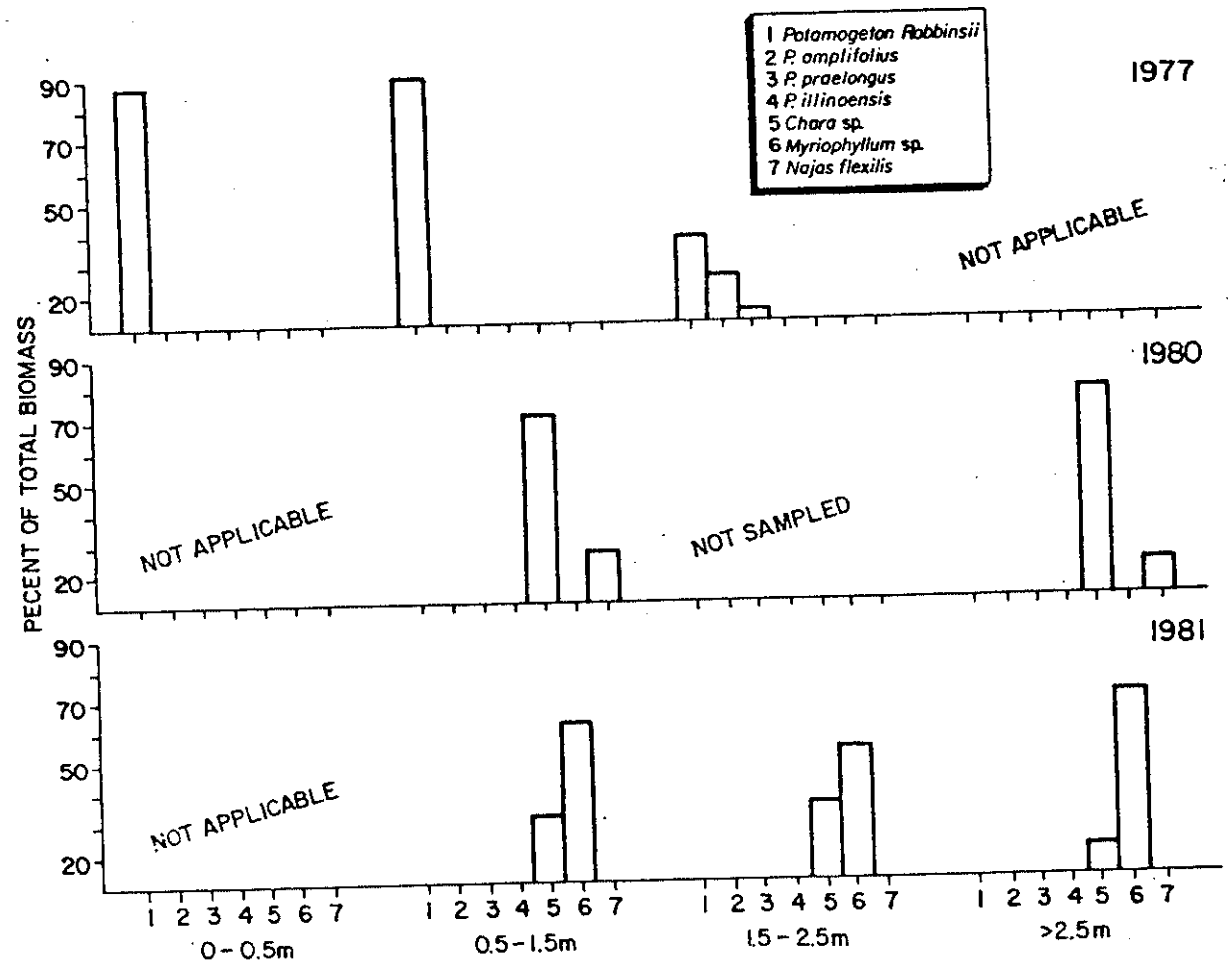

FIGURE 25. Dominant Macrophyte Species Growing on Muck Bottom, 1977-81

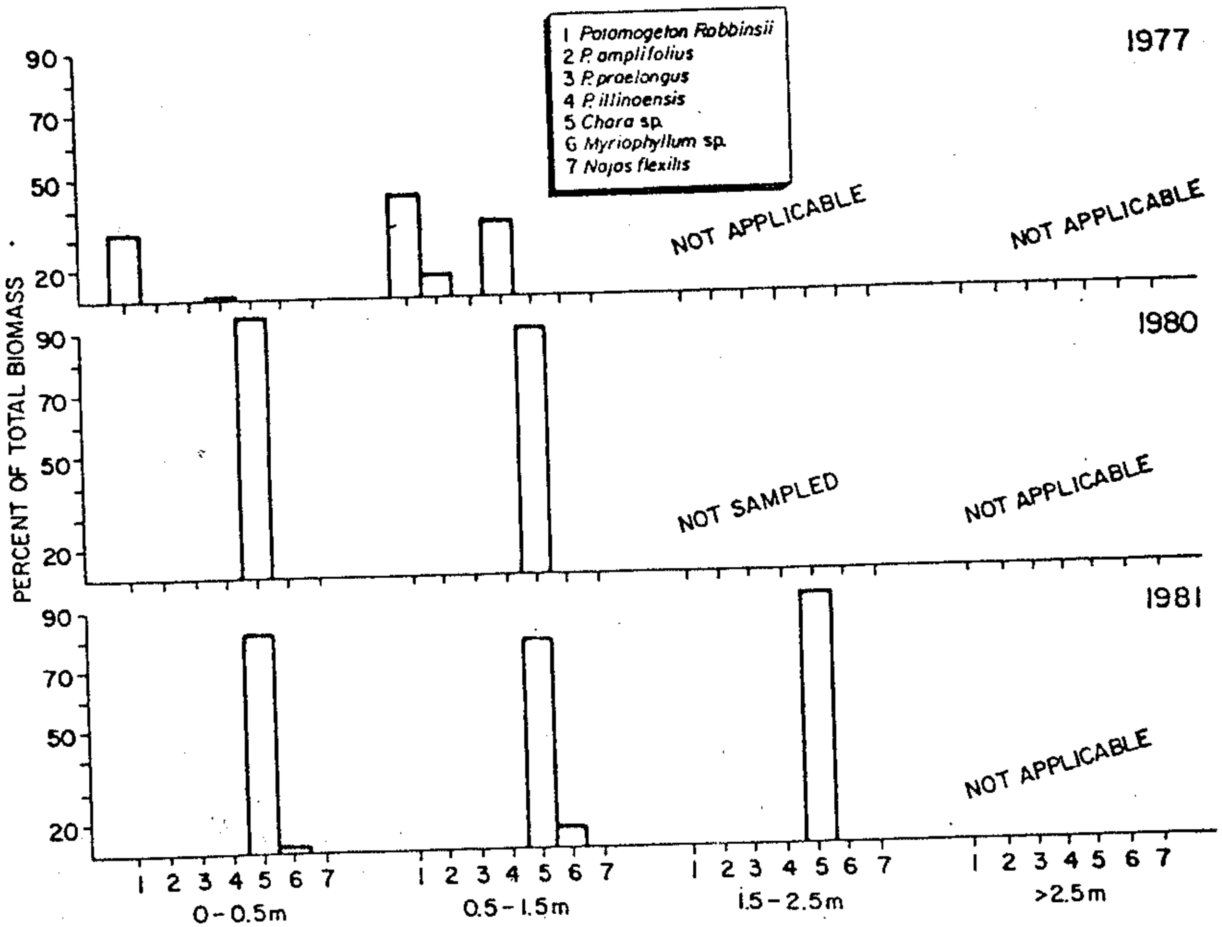

FIGURE 26. Dominant Macrophyte Species Growing on Sand Bottom, 1977-81

The P content of whole plant tissue samples was also examined during 1977-81. There were differences between species and between years but no indication that changes were related to the dredging project (Table 29). In general the content ranged from 0.1 to 0.2% for the rooted species. *Chara* sp. was slightly lower due to its high inorganic composition. The quantity of P contained in macrophyte tissue as compared to other sources/pools is shown in Table 30. Phosphorus in macrophytes was much lower in 1980-81 versus 1977; however, it was still greater than either the annual loading from external sources or the amount contained in the water column. Sediments were by far the largest reservoir of P throughout 1977-81, but the NaOH-P fraction was reduced greatly by dredging.

TABLE 29: Macrophyte Tissue Phosphorus (summer average percent)*

1977	
Potamogeton Robbinsii	0.08
P. amplifolius	0.13
P. praelongus	0.18
P. illinoensis	0.15
Megalondonta Beckii	0.14
Myriophyllum spicatum	0.41
1978	
Potamogeton Robbinsii	0.07
P. amplifolius	0.16
P. praelongus	0.28
Megalondonta Beckii	0.16
Elodea canadensis	0.20
1980	
Chara sp.	0.07 (0.81)
Najas flexilis	0.18 (1.65)
Myriophyllum spicatum	0.11 (2.12)
Elodea canadensis	0.16 (1.97)
1981	
Chara sp.	0.08 (0.94)
Najas flexilis	0.20 (1.90)
Myriophyllum spicatum	0.10 (1.27)
Elodea canadensis	0.13 (1.85)
Potamogeton praelongus	0.17 (1.67)
P. Robbinsii	0.12 (2.12)

*Percent nitrogen is in parentheses.

TABLE 30: Phosphorus Sources/Pools in Lilly Lake, 1977-81*

	1977	1980	1981
External Loading (TP)	17.4	19.0	18.7
Water Column (TP)	12.3	18.8	19.1
Macrophyte Tissue (TP)	119.9	24.8	32.6
Interstitial Waters:			
as SRP	-	0.2**	0.2**
as TDP	6.4	0.4	0.4
Total Sediment Phosphorus (TP)	1,981	990	1,074
NaOH-P Fraction	(573)	(21)	(31)
HCl-P Fraction	(347)	(308)	(361)

*units in kilograms. The 1977 interstitial and sediment values are based on the sampling conducted in June, 1978. The values for total sediment phosphorus represent the quantity contained in the top 7.6 centimeters.

**less than

IMPACT OF SEDIMENT DISPOSAL

Lilly Lake sediment was deposited in the gravel pit area during 1978 and 1979 while the diked area was used in 1979 only. The environmental effects of deposition on the groundwater system were evaluated at both sites in terms of well water levels, NH_4-N, NO_3/NO_2-N, pH, and Sp. C. In addition, COD was monitored at the diked area and the following at the gravel pit: Cl, TON, TDP, TP, As, Ba, Cd, Cu, Fe, Pb, Mg, Se, Ag, Zn, and Cr.

Modified Gravel Pit

A. Groundwater Levels

Groundwater levels were monitored from June 13, 1978 through September 29, 1981 (well G6 was destroyed September, 1980). Precipitation during 1977 was 1.2 cm below the 30 year average of 85.6 cm. This was followed by February through May, 1978 which had below average monthly precipitation values. Therefore, the water levels should have been lower to start with and increased in 1978-80, because these years had annual precipitation amounts above average; 1978 had a rate of 106.53 cm/yr which was 21 cm above average.

The effect of sediment and water disposal on groundwater levels showed up in all the observation wells in both years, 1978 and 1979. Well G5 showed the largest impact, which occurred in 1978. It was the closest well to Basin I, 46 m, and it had the highest water level rise at approximately 6 m (Figure 27). Well G7 was the next closest well, 76 m, and it had a water level rise of 2.4 m. Groundwater levels increased in wells G1 - G3 approximately 2 m in 1978. They were about 150 m from Basin I and 50 - 90 m from Basin II. Part of this increase was due to natural recharge from precipitation and the rest from artificial recharge from the sediment disposal.

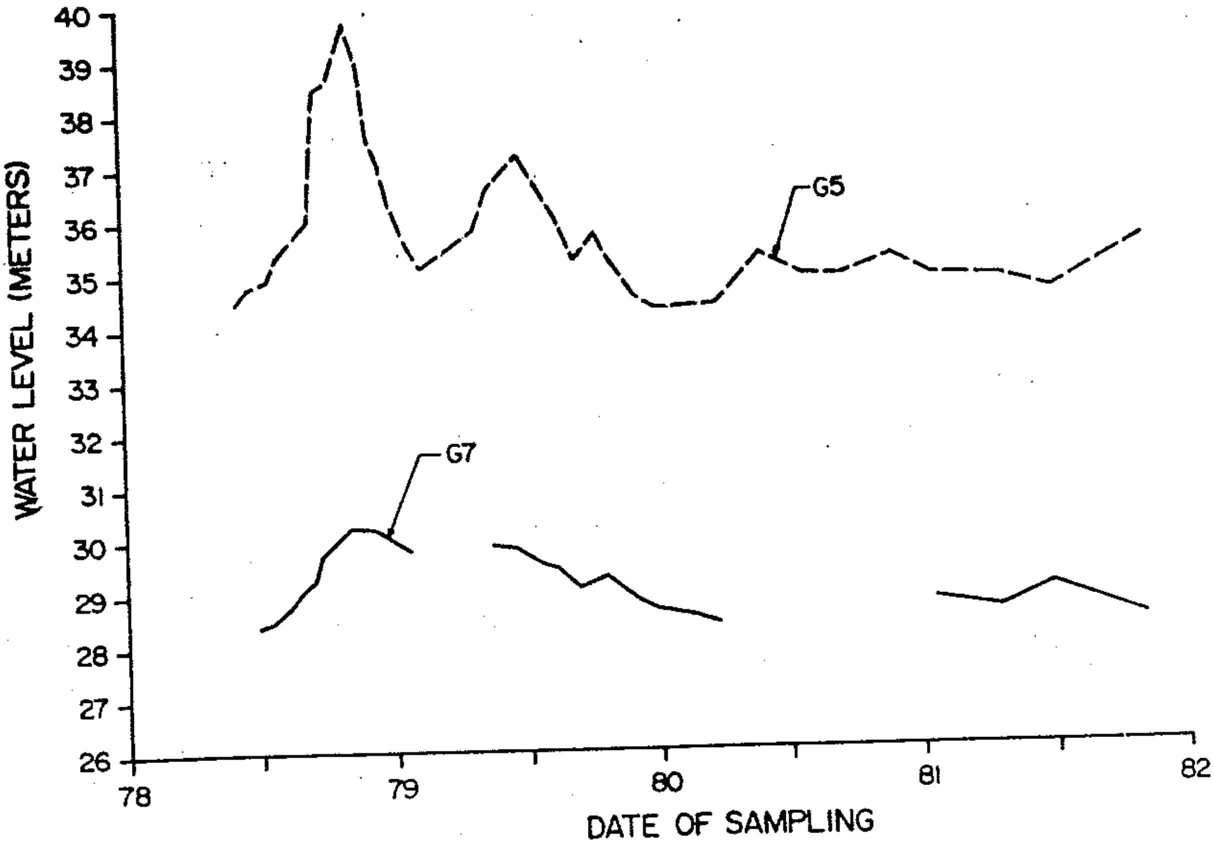

FIGURE 27. Groundwater Levels for the Monitoring Wells G5 and G7 Gravel Pit, 1978-82

During 1978, sediment plus water was put in Basin I. There was a standing pipe that drained relatively clear surface water from Basin I to Basin II. This is important because of the ability of Lilly Lake's sediment to prevent large amounts of groundwater recharge by preventing leakage from the bottom or lower sides of Basin I. Most of the lake water recharge to groundwater probably occurred near the water's surface at points away from the sediment discharge pipe where the water was relatively clear of sediment particles. This is corroborated by the extremely slow rates at which levels in the pits declined, 1.1 m in Basin I during the winter of 1978-79 (Table 31) compared to the more rapid 5 m drop in well G5 (Figure 27). This was also shown by the lack of draining over the longer term when there was only a 1 m drop in level in Basin I from August, 1979 through August, 1981.

TABLE 31: Selected Basin Water Level Data for the Gravel Pit Disposal Area

Date	Basin I Depth Below Drain Pipe Top	Basin II Depth Below Horizontal Tube	Notes
10/16/78	Over Drain Pipe Top 0.15 meters from Dike Top	At Top of Tube	
10/26/78	-	-	Dredging Stopped
10/30/78	0.1 meters	-	
11/13/78	0.3-0.5 meters	-	
11/27/78	0.6-0.8 meters	0.6 meters	
12/29/78	Same	Same	
3/20/79	1.2 meters	-	
5/1/79	-	-	Dredging Started
5/2/79	0.5 meters	-	
5/9/79	Over Drain Top	-	
8/29/79	-	-	Dredging Finished
8/3/81	0.84 meters (Wetland)	(2.5 meters Below Basin II Berm Top) (Shallow Pond)	

B. Vertical Component of Groundwater Flow

Wells G1, G1A, G2, G2A, G3 and G3A were nests of wells. Both wells at each site were located within a meter of each other. The screen for the shallow well was near the water table surface, and the deeper well's screen was 3.4 to 4.8 m below that of the shallow well. These wells were used to indicate the vertical direction of groundwater flow.

Figure 28 shows groundwater level data for well G1, G1A, G2, and G2A. Generally the flow at these sites was downward during the major parts of the study period. On the other hand, at well G3 the flow was more horizontal and slightly upward (Figure 29). This interpretation of well site G3 is consistent with its location between Basin II and the intermittent stream and Fox River to the south and east, respectively.

An interpretation of the data indicates that during and after sediment disposal, groundwater had a tendency to flow from well G2 to G1 and G3. Also, the vertical flow was down in G1 and G2 and horizontal to slightly upward in G3.

C. Ammonia Nitrogen

Although NH_4-N was found at elevated levels in the lake sediments (Table 32), concentrations remained low in the observation wells. The one exception to this occurred on October 2, 1978 when 10 mg/L was noted in well G1. For this same date well G1A, the deeper well, had a concentration of 1.4 mg/L. Ammonia-N in the home well samples were all below 0.7 mg/L . The range of concentrations found in the home wells is illustrated by wells 1 and 2 in Figure 30.

TABLE 32: Average Composition of the Sediment and Pore Waters for Lilly Lake as Found in Basin I

Parameters	Sediments Mean	Pore Water Mean	Pore Water Range
Solids (%)	3.1	-	-
TKN (mg/L)	1018.2	11.0	3.3-19.5
NH_4-N (mg/L)	-	9.5	2.1-18.0
NO_2/NO_3-N (mg/L)	-	0.1*	-
Lab pH (Standard units)	6.8	-	-

*less than

Ammonia is absorbed to clay particles under anaerobic conditions or is oxidized to NO_3-N under aerobic conditions. So, even though NH_4-N was the dominant inorganic N form in the lake sediments, nitrification to NO_3-N was generally necessary for significant transport of the N into the groundwater system.

D. Nitrate Plus Nitrite Nitrogen

Nitrate plus nitrite-N was of concern due to the drinking water standard of 10 mg/L for NO_3-N (U.S. Public Health Service, 1962). During the study period, 231 well samples and three gravel pit basin samples were analyzed. The average concentration for the well samples was 0.94 mg/L, with a maximum of 8.10 mg/L and a minimum of less than the detectable limit of 0.02 mg/L.

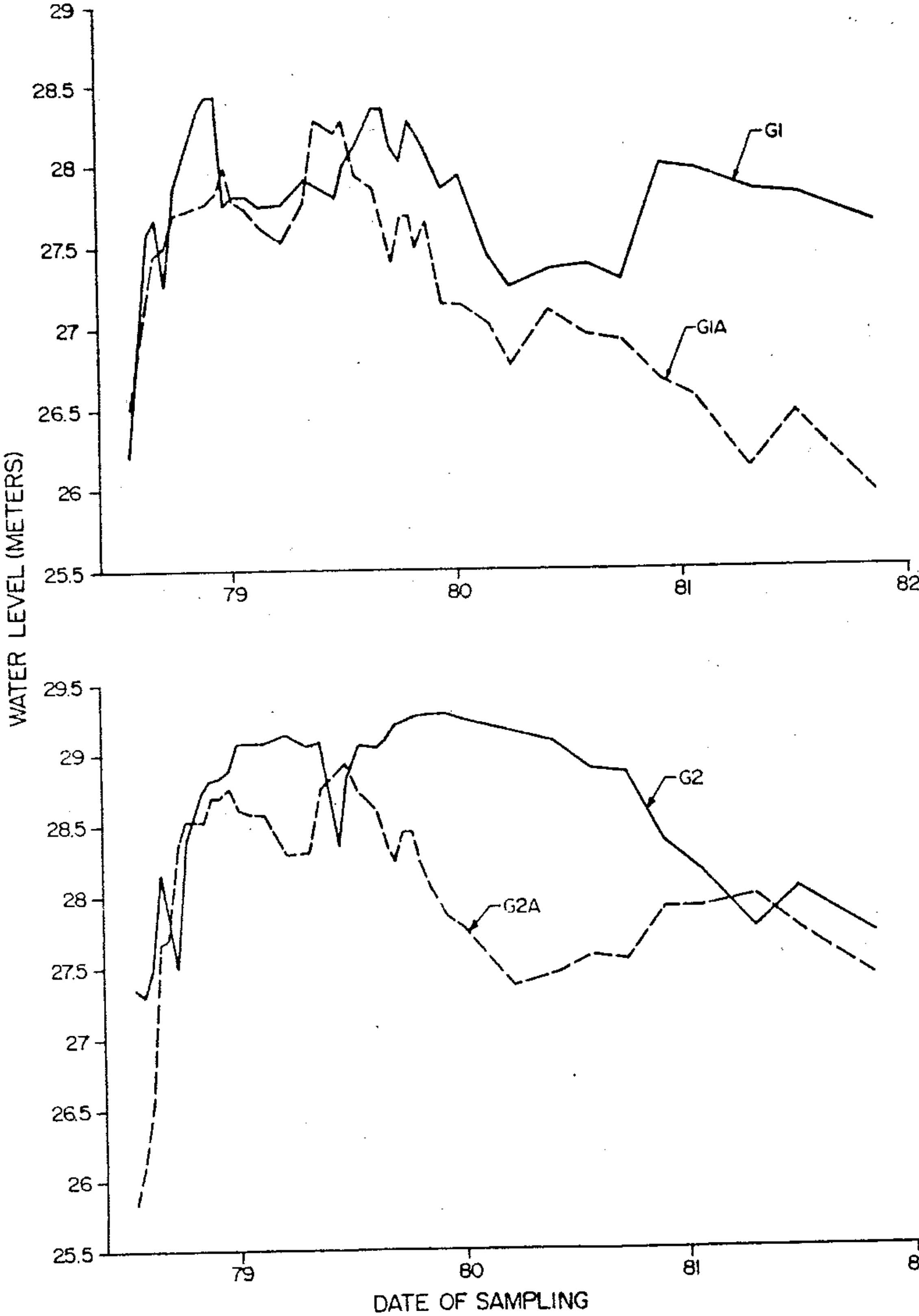

FIGURE 28. Comparison of Water Levels in Piezometer Well Nests G1 and G2, 1978-82

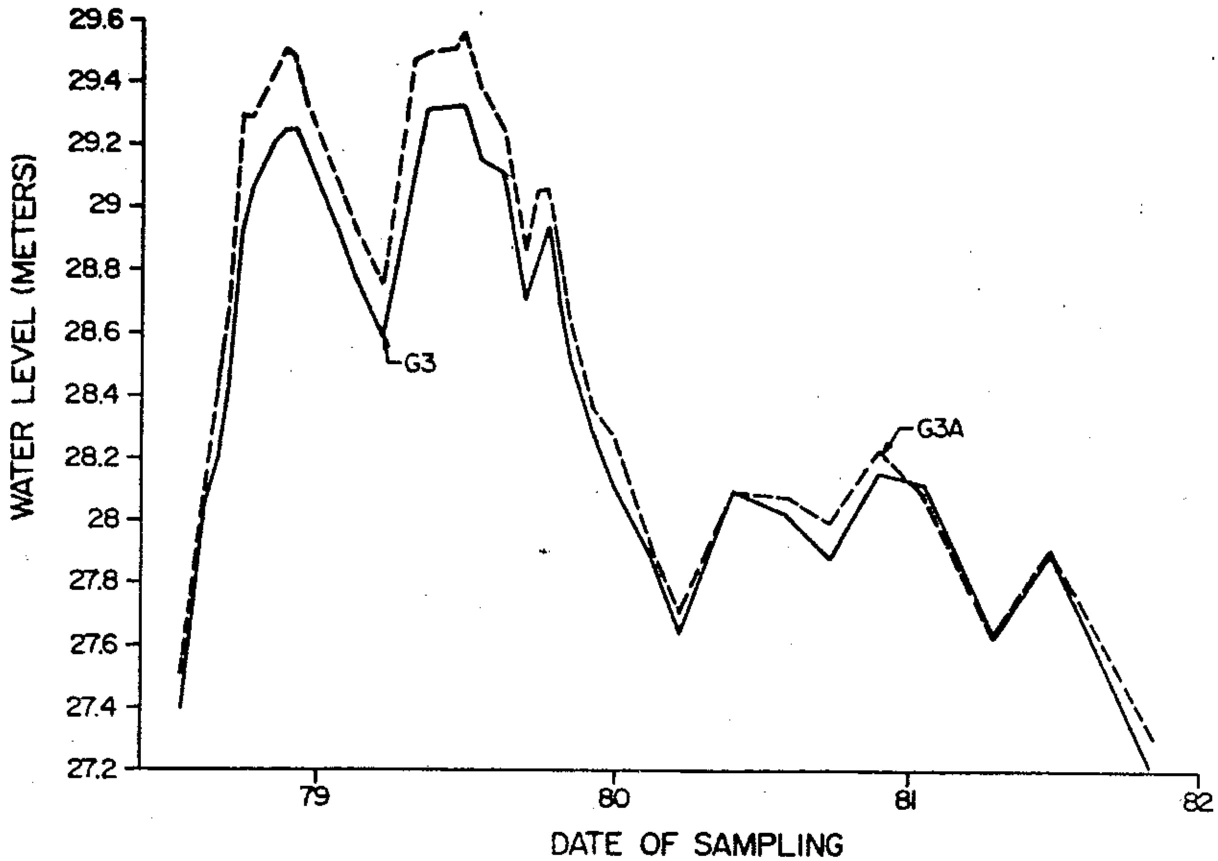

FIGURE 29. Comparison of Water Levels in Piezometer Well Nest G3, 1978-82

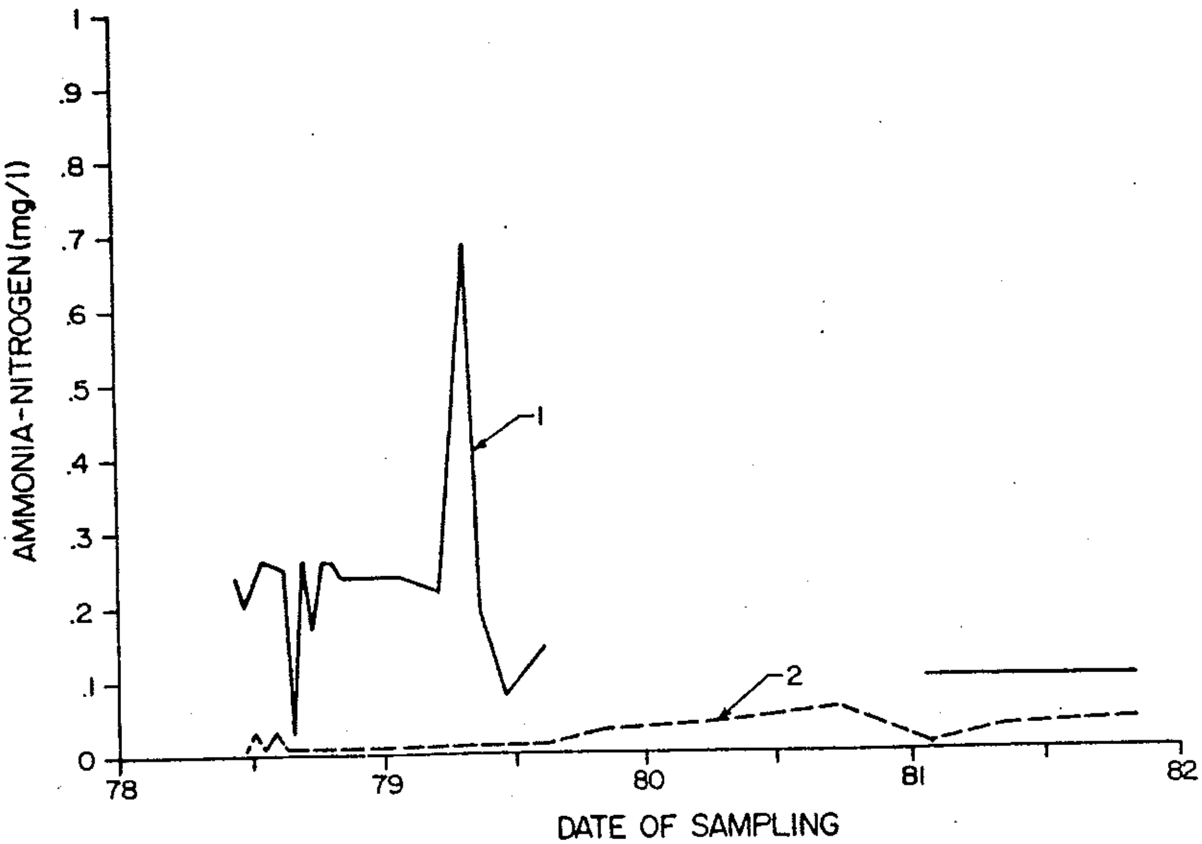

FIGURE 30. Ammonia Nitrogen in Home Wells 1 and 2, 1978-82

Figure 31 exhibits the range in concentrations found in the home wells. Only one home well, #2, showed a significant change in concentration that may have been related to the disposal activity. This well was relatively shallow, 15.2 m deep. Therefore, NO_2/NO_3-N concentrations in the groundwater used for drinking water in homes around the gravel pit disposal area did not exceed the 10 mg/L standard.

E. Other Parameters

The wells were sampled for a variety of indicator parameters such as Sp. C, Cl, and others as described earlier. None of these demonstrated changes that could be related to the sediment disposal activity.

Low Dikes on Agricultural Land

A. Groundwater Levels

Groundwater levels were monitored from May 2, 1979 through October 25, 1981 (well D1 was destroyed in May, 1980). The spring snowmelt in 1979 and the unusually wet late summer conditions of 1978 caused elevated groundwater levels prior to sediment disposal. During this period water levels were declining as would be expected after the spring snowmelt recharge.

Disposal of lake sediment to the diked area began May 21 and continued until July 24, 1979. Clear water was also pumped from the other sediment disposal site to the diked area from June 11 through August 29, 1979. The effect of sediment and water disposal on groundwater level first showed up in the sampling on June 4, 1979. This rise was due primarily to seepage of water out of the diked area. Only 1.6 cm of rainfall fell during the eight day period prior to June 4, 1979 (Table 33). Well D6 was one of the first to respond because it was close to the cells and its screen was in sand or sand and gravel with good permeability (Figure 32). Well D3 was the slowest to respond. Its well point was located in a loamy gravel soil and the slow response was due to greater distance from the disposal field and lower permeability.

Groundwater levels in June, 1979 were higher than at any other time during the study period. During the latter half of June and July the water levels declined, indicating a reduced rate of recharge. This was during the period of sediment disposal when the cells were full of sediment/water. Therefore, the soil pores must have been plugged with low permeability lake sediments which significantly slowed the rate of water recharge to the groundwater table from the cells.

Increased water levels were observed occasionally later during the study period. These were, however, related to rainfall or snowmelt recharge events. For instance, the elevated levels in late August, 1979 were in response to a rainfall accumulation of 25.3 cm which was 16.3 cm above the monthly mean of 9.0 cm.

B. Ammonia Nitrogen

Ammonia-N concentrations ranged from 0.01 to 0.70 mg/L, with a mean of 0.14 mg/L for all of the well water samples taken during the study period. Prior to sediment disposal, NH_4-N ranged from 0.06 to 0.49 mg/L, with a mean of 0.20 mg/L. During and after sediment disposal, concentrations ranged from 0.01 to 0.70 mg/L, with a mean of 0.13 mg/L (Table 34).

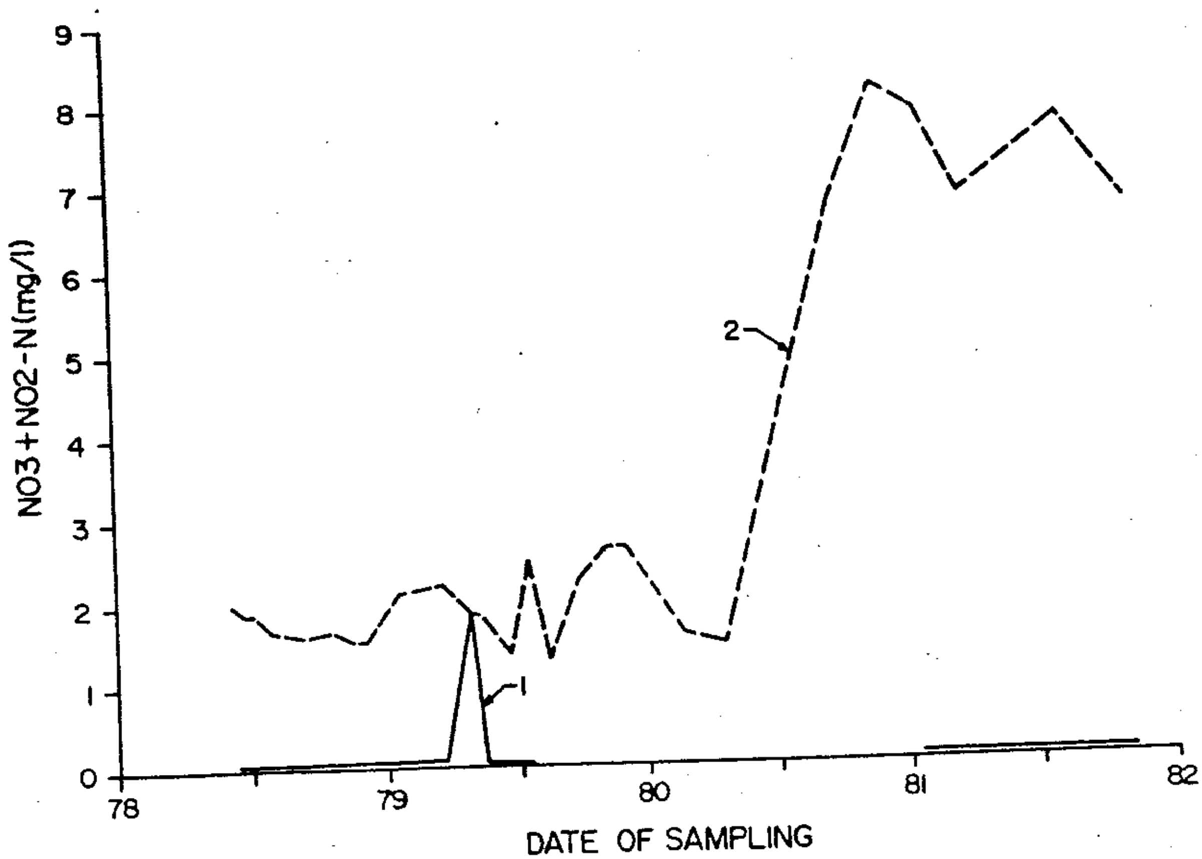

FIGURE 31. Nitrate Plus Nitrite Nitrogen in Home Wells 1 and 2, 1978-82

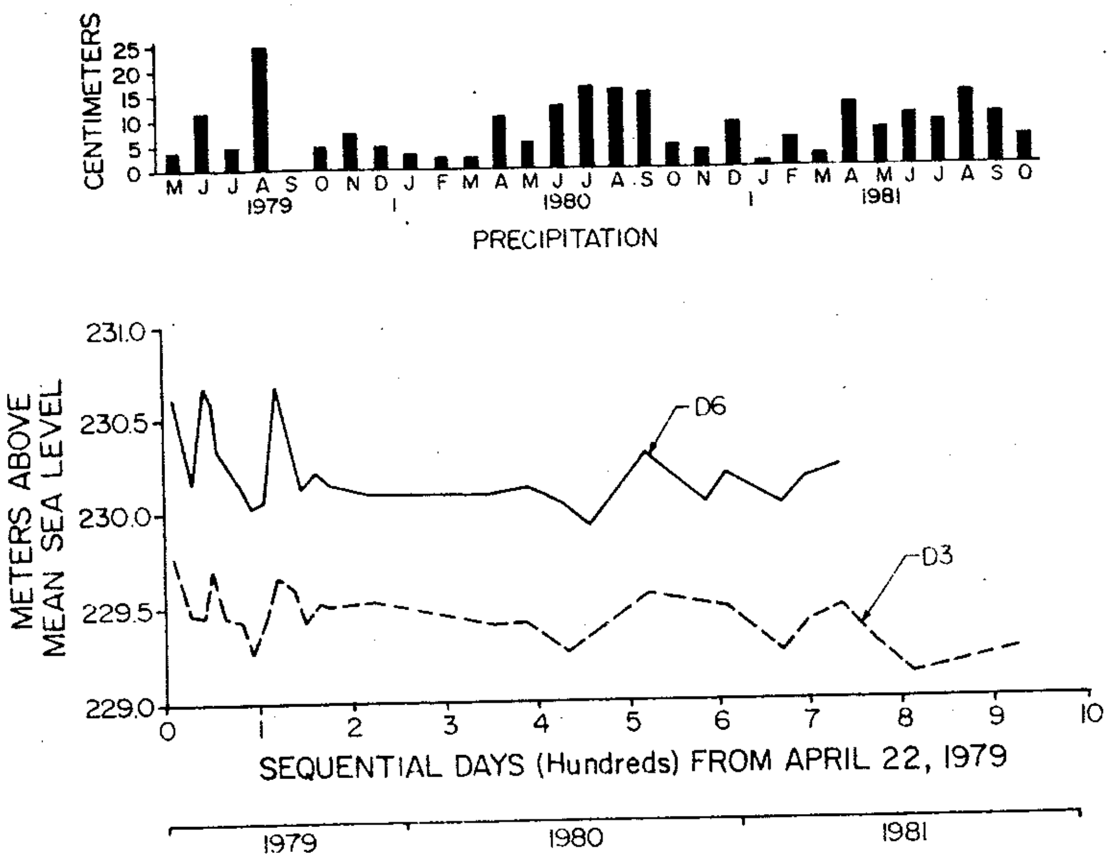

FIGURE 32. Monthly Precipitation and Groundwater Levels Around the Diked Area, 1979-81

TABLE 33: Daily Rainfall Data During Initial Filling of the Diked Area, 1979

Month	Day	Well Sampling Dates	Centimeters of Rainfall, Weather Station: Burlington	Lake Geneva	Union Grove	Antioch	Mean
May	19		.38	.08	.13	.03	.15
	20		.05				.03
	21	X					
	22						
	23						
	24						
	25						
	26						
	27						
	28					.38	.10
	29			.03		.94	.25
	30		.43	.38	1.37	.94	.79
	31		.10		.05		.05
June	1					T*	T
	2						
	3					1.60	.41
	4	X					
	5		.71	.13	.43	.33	.33
	6				.03		T
	7			4.11			1.04
	8		.94	1.32	1.60		.97
	9		.97	.58	4.09		1.42
	10		.84	.56	.89		.58
	11	X	.03		.03	.43	.13
	12					.18	.05
	13					T	T

*T stands for trace of precipitation.

TABLE 34: Average Groundwater Quality in the Diked Area Wells Before and After Sediment Disposal

Parameter	Before: Number of Samples	Before: Mean	Before: Range	During and After: Number of Samples	During and After: Mean	During and After: Range
Nitrate Plus Nitrite-N (mg/L)	11	0.68	0.01 - 2.7	90	1.38	0.01 - 9.6
Ammonia-N (mg/L)	7	0.20	0.06 - 0.49	90	0.13	0.01 - 0.70
Chemical Oxygen Demand (mg/L)	9	38	13 - 85	61	35	5.0 - 150
ph (Standard units)	13	7.53	7.1 - 8.4	68	7.28	6.6 - 9.7
Conductivity (umho/cm at 20°C)	11	1006	516 - 3240	74	651	302-1310

Changes in concentration observed during the study period are illustrated in Figure 33. The one peak which may have been related to sediment disposal occurred at well D6. The peak at well D4 in 1981 was probably related to ammonia fertilization on a nearby field.

C. Nitrate Plus Nitrite Nitrogen

Nitrate plus nitrite-N samples were taken on three dates prior to sediment disposal in the diked area. The highest concentration in the wells prior to sediment disposal was 2.7 mg/L in well D2. During and after disposal, wells D2 and D6 had peak values of 8.6 and 9.6 mg/L, respectively (Figure 34). No other well had concentrations above 3 mg/L at any time. Overall there was an average concentration increase from 0.68 to 1.38 mg/L (Table 34).

Because of the limited pre-disposal sampling period of 19 days in May of 1979, it is difficult to assess how much of this was due to sediment disposal versus natural seasonal fluctuations under an agricultural field; however, the results from water samples collected at well D5 should represent typical groundwater quality. This well was located upgradient in clay soils of poor permeability. Nitrate plus nitrite-N concentrations remained low throughout the study period, at least suggesting that the increase observed in other wells was due to sediment disposal. In any case the 10 mg/L standard for NO_3-N in drinking water was not even exceeded in well D6, a downgradient well located in permeable soils immediately adjacent to the diked area.

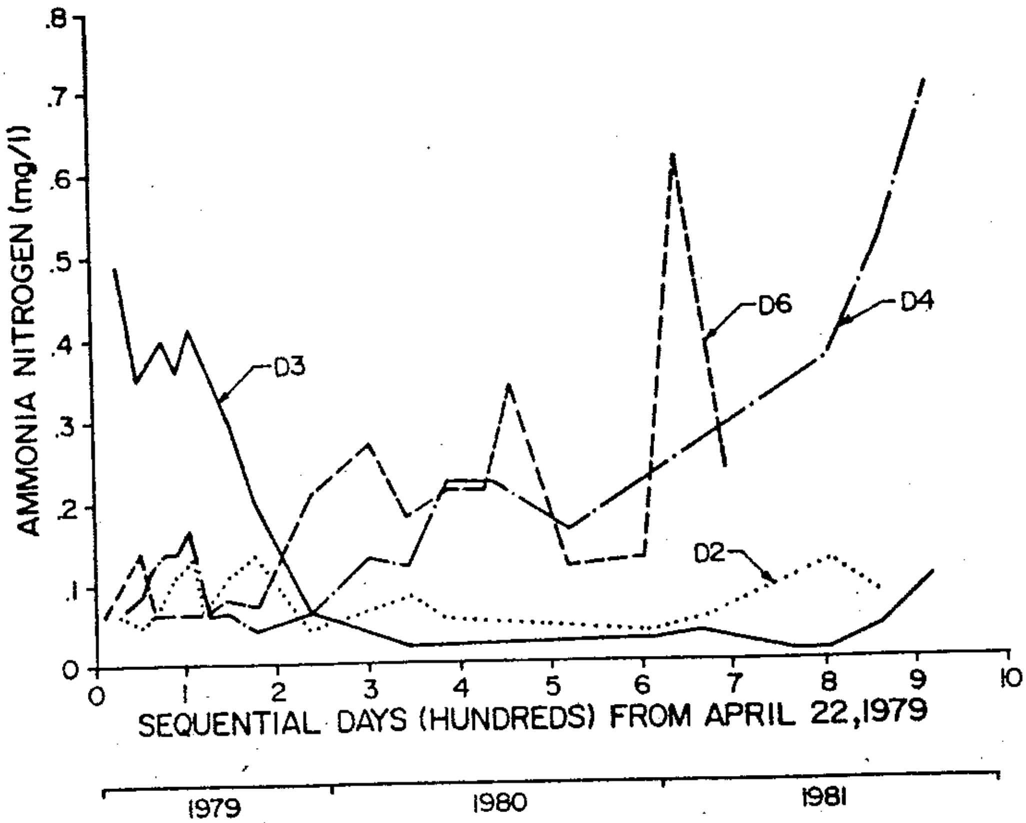

FIGURE 33. Ammonia Nitrogen Concentrations in the Diked Area Wells, 1979-81

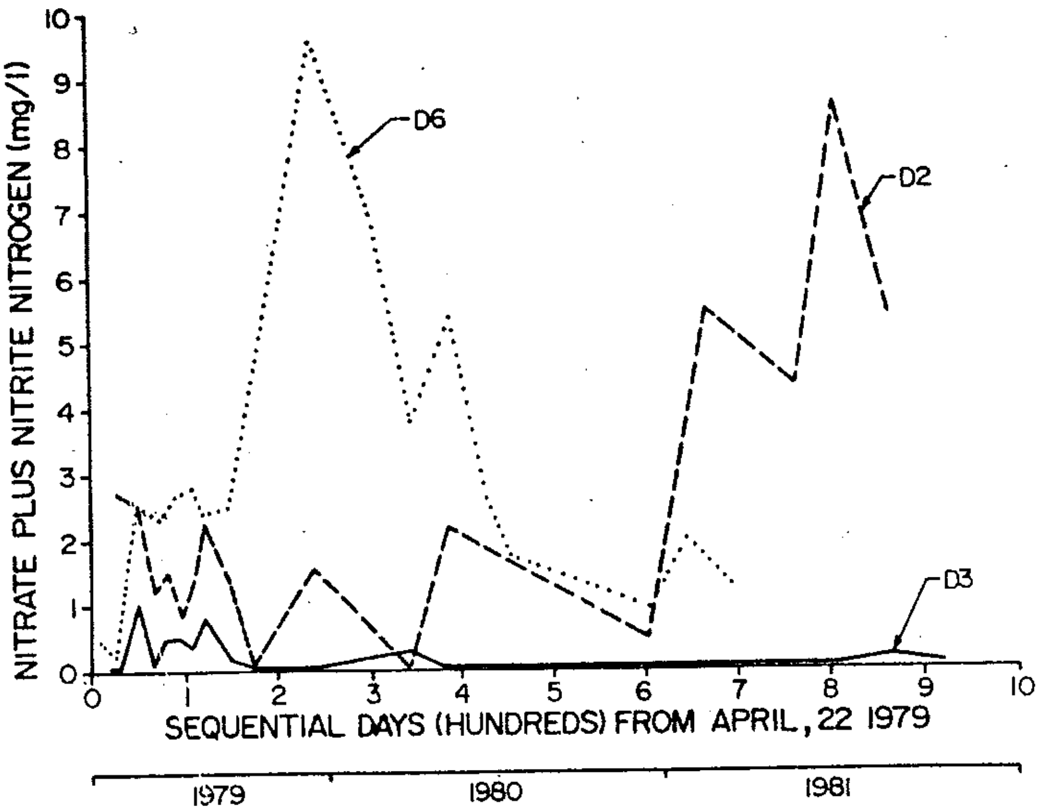

FIGURE 34. Nitrate Plus Nitrite Nitrogen in the Diked Area Wells, 1979-81

D. Chemical Oxygen Demand

Chemical oxygen demand ranged from 5 to 150 mg/L, with a mean of 36 mg/L. Well D3 had the highest average at 48.2 mg/L (excluding well D1 which had only two samples); however, the wetland soils and setting of this well were contributing factors. The highest COD (150 mg/L) occurred at well D6 on June 26, 1979 and was probably related to sediment disposal (Figure 35).

E. pH

The pH ranged from 6.6 to 9.7, with a mean of 7.3. The most basic well was D1 with a mean of 8.0. The most acidic well was D4 with a mean of 7.0. The most acidic months were July through September. At well D3 when the pH was between 7.5 and 8.2, the total inorganic N was high; and when the pH ranged from 8.2-9.7, the inorganic N concentrations were low.

F. Conductivity

Conductivity ranged from 302 to 3,240 umhos/cm for all the wells (Table 34). The highest mean at an individual site, well D6, was 992 umhos/cm. A maximum value of 3,240 occurred in the first sample from well D6; however, it may have been contaminated due to the well installation process (Figure 36). Well D4 had the next highest mean of 839, and well D2 had the lowest mean of 516 umhos/cm.

BENEFICIAL USES OF DREDGED LAKE SEDIMENT

Sediment Survey

Analytical data from the sediment survey are summarized in Table 35. Of the 12 sediments, Lilly Lake-Kenosha is highest in pH, total C, COD, loss on ignition, total N and N release on three months incubation. It showed the greatest decrease in P equilibrated in 0.01M $CaCl_2$, and was intermediate in the other categories. Thus, it would appear that this sediment should be a reasonably good source of N, a poor source of P, and should tend to raise the soil pH if applied to acid soils.

The 12 sediments showed wide ranges in most of the factors studied. The pH values ranged from 5.1 (Tomah) to 7.7 (Lilly Lake-Kenosha), and the COD, which reflects the organic matter content, ranged from 3.5 (Leota) to 31.9 (Lilly Lake-Kenosha). Other factors of particular interest from the soil fertility standpoint are the rather narrow range in C/N ratios (8.9 in Pigeon to 13.3 in Lily Lake-Marathon), the very wide range in C/P ratios (49 in Leota to 1420 in Lily Lake-Marathon), the range in N immobilized (43 ug/g in Rib) to N mineralized (14.5 ug/g in Lilly Lake-Kenosha) in the incubation experiment, and the decrease (4.4 ug/g in Rib) to increase (1.0 ug/g in New Richmond) of $NaHCO_3$-extractable P on incubation.

Except for Mn, trace element contents showed fairly small ranges. Arsenic values in the sediments could not be obtained because of Al interference on the ICP. Cadmium values were just above the 2 ug/g level established as the allowable limit for unrestricted application of sewage sludges (U.S. EPA, 1979). However, there is a serious question as to whether such a low limit can be justified on the basis of toxicological data (Chaney, 1983).

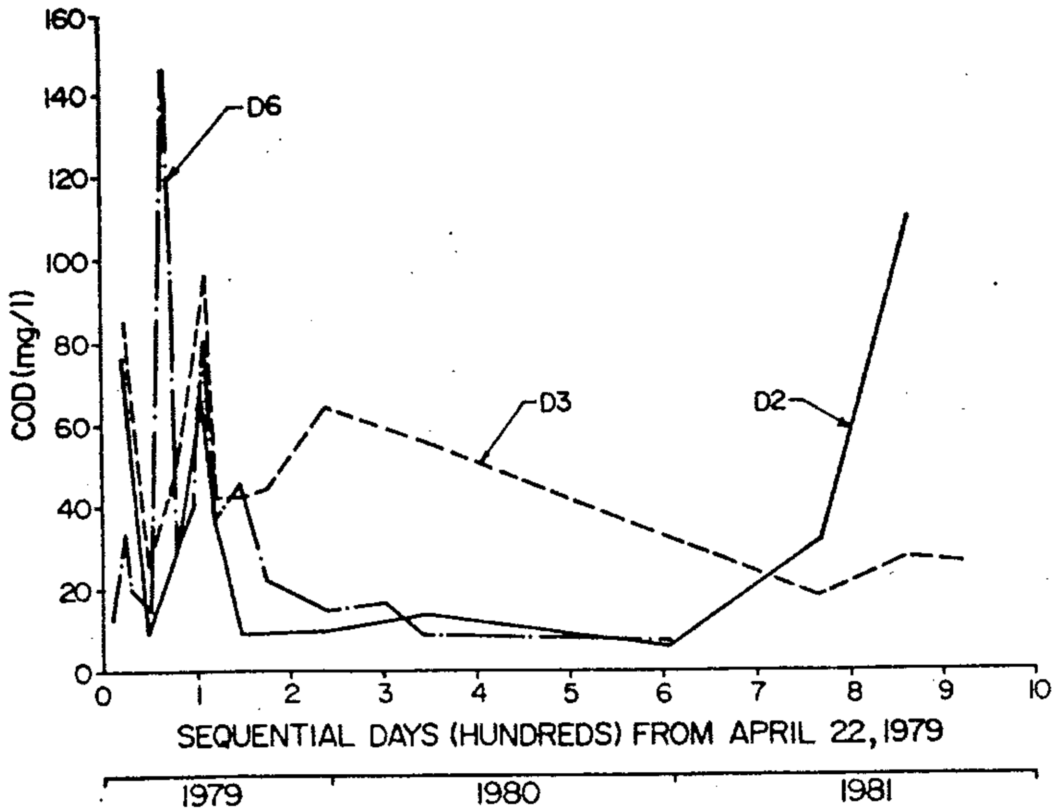

FIGURE 35. COD Concentrations in the Diked Area Wells, 1979-81

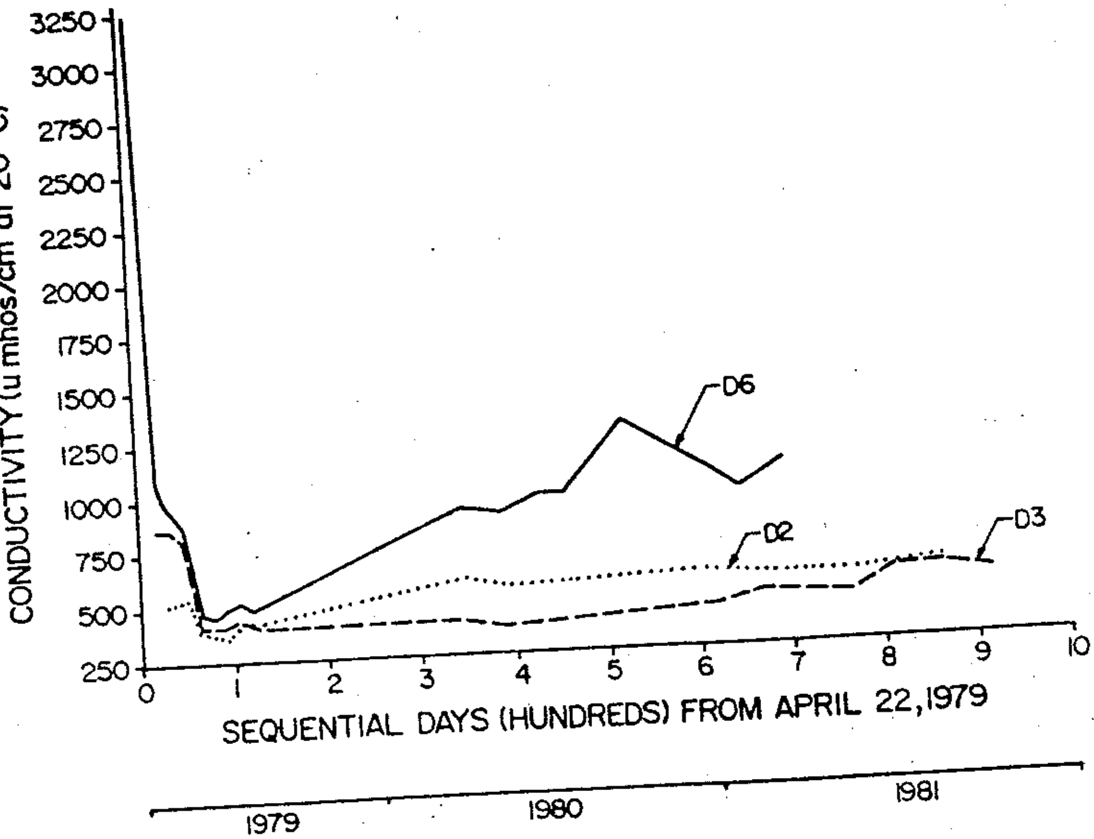

FIGURE 36. Conductivity in the Diked Area Wells, 1979-81

TABLE 35: Ranges for Chemical Factors Determined on 12 Wisconsin Lake Sediments Compared With Values Obtained on the Sediment from Lilly Lake in Kenosha County

Variable	Range		Lilly Lake Kenosha*	
	Low	High	I	II
pH	5.1	7.7	7.7	7.2
Total C, %	5.5	34.2	32.4	34.2
COD, %	3.5	31.9	28.7	31.9
Ign. loss, %	12	66	66	66
Total N, %	0.35	2.69	2.69	2.69
NH_4-N, ug/g	18	671	38	70
NO_3-N, ug/g	9	64	17	37
Total P, ug/g	366	1520	621	635
Organic P, ug/g	176	1080	355	338
Total C/N	8.9	13.3	10.7	11.9
Total C/P	49	1420	808	944
Zn, ug/g	30	100	88	--
Mn, ug/g	103	967	123	--
Cu, ug/g	22	58	31	--
Cd, ug/g	2.1	5.9	2.4	--
Pb, ug/g	13	60	25	--
$CaCl_2$-P, ug/ml	0.005	0.044	0.016	--
$NaHCO_3$-P, ug/g	2.1	14.6	3.3	--
N release, ug/g**	-43	14.5	14.5	--
$CaCl_2$-P release, ug/ml**	-0.0.12	0.025	-0.012	--
$NaHCO_3$-P release, ug/g**	-4.4	1.0	-3.1	--

*Sample I was taken from the lagoon adjacent to the field site. It was used for the greenhouse and field studies. Sample II was taken from a frozen sample of sediment collected from Basin I at the gravel pit site.

** Released during three months incubation after leaching with 0.01M $CaCl_2$.

Greenhouse Study.

On the basis of the sediment survey results (and also the availability of sufficient sediment), three sediments in addition to that from Lilly Lake-Kenosha were selected to give wide ranges in pH, organic matter, total N and C/P ratios. These sediments were Lilly Lake-Kenosha (LLK) with high pH, organic matter, total N and C/P ratio; Lily Lake-Marathon (LLM) with intermediate pH, high organic matter, moderately high total N and very high C/P ratio; New Richmond (NR) with neutral pH, low organic matter, low total N and low C/P ratio; and Tomah (T) with low pH, low organic matter, low total N and low C/P ratio. These sediments were applied to a Plainfield sand and a Withee silt loam.

Statistical analyses of the effects of soil, sediment source and sediment rate are given in Table 36. Differences due to soil factors related to yield or concentration in tissue were all significant at the 1% level except for yield and Cu concentration. As can be seen from Table 37, these differences due to soil, though significant, were not large except for B and K concentrations which differed by an average factor of about two.

Differences in yield and element concentrations and uptakes associated with sediment source were significant at the 1 or 5% levels for all factors except B, P and K concentrations (Table 36). For purposes of illustration, the values for the controls and for the averages of all treatments for a given sediment applied to the Plainfield sand are given in Table 38. The data for the sand are given here because the low buffering capability should allow the greatest impact of sediment characteristics.

The two highly organic soils, LLK and LLM, showed the greatest N uptake. The high C/P ratios of these soils did not seem to have a depressing effect on P uptake. Only N and K concentrations showed consistent increases over the control when compared with all sediments on both soils. The Zn in the NR sediment appeared to be more available to the corn than that in the other sediments even though total concentrations were equivalent. Other differences among sediments, while significant statistically, were not of much consequence practically.

Although sediment source had a significant effect on most of the element concentrations, the effect of the rate at which the sediment was applied was significant at 5% or better only for Ca, Mg, S and Mn. Calcium concentration increased with rate for all sediments except Tomah. For Mg, addition of all rates of all sediments depressed the concentration below that of the fertilized control on the sand. This did not happen on the silt loam. Increasing rates of the LLK sediment generally resulted in increasing Mg concentrations. The lower Mg concentration associated with sediment addition on the sand is probably associated with uptake antagonism from increased K availability.

As with Mg, S concentration tended to drop from the control to the lowest sediment rate and then rise with increasing additions. This was least apparent with the LLK sediment.

Concentrations of Mn in the corn tissue were markedly depressed by increasing rates of the high pH LLK sediment and increased with rate in the highly acidic T sediment. There was a tendency for higher Mn concentrations at the higher rates of the other two sediments, particularly on the sand.

Lack of an effect of sediment rate on concentration of the other elements is somewhat surprising, particularly for N where there is a marked effect of sediment source on N concentration in tissue.

TABLE 36: F Values Showing Significance of the Effects of Soil, Sediment Source and Sediment Rate on Variables Determined in the Greenhouse Study

Variable	Soil	Sediment Source	Sediment Rate
Yield	0.65	2.68**	1.39
N conc.	37.79***	79.58***	0.39
P conc.	89.94***	2.52*	0.33
K conc.	1528.96***	2.24*	0.47
Ca conc.	206.58***	6.71***	29.14***
Mg conc.	476.92***	27.52***	9.22***
S conc.	43.74***	9.27***	12.80***
Fe conc.	64.32***	3.03**	0.94
Mn conc.	139.12***	52.38***	8.05***
Cu conc.	0.81	25.44***	2.24*
Zn conc.	11.38***	5.79***	1.56
B conc.	991.27***	0.81	0.53
N uptake	21.83***	28.21***	0.39
P uptake	13.55***	3.15**	0.65
K uptake	675.99***	10.90**	1.17

*Significant at the 10% level.
**Significant at the 5% level.
***Significant at the 1% level.

TABLE 37: Effect of Soil on Average Values of Variables Determined in the Greenhouse Study

Variable	Plainfield Sand	Withee Silt Loam	Significance*
Yield, g/pot	2.90	3.04	ns
N, %	2.94	3.27	***
N uptake, mg/pot	85	99	***
P, %	0.144	0.124	***
P uptake, mg/pot	4.2	3.8	***
K, %	2.3	4.4	***
Ca, %	0.87	0.69	***
Mg, %	0.63	0.44	***
S, %	0.183	0.167	***
Zn, ug/g	46	38	***
B, ug/g	15	30	***
Fe, ug/g	59	52	***
Mn, ug/g	82	63	***
Cu, ug/g	10.7	9.6	ns

*ns = not significant.
***Significant at the 1% level.

TABLE 38: Effect of Sediment Source on Average Values of Variables Determined in the Greenhouse Study

Variable	Control	LLK	LLM	NR	T	Significance
Yield, g/pot	2.5	3.2	2.8	3.1	2.9	**
N, %	2.8	3.1	4.1	2.2	2.6	***
N uptake, mg/pot	71	97	113	66	77	***
P, %	0.15	0.14	0.14	0.14	0.15	*
P uptake, mg/pot	3.9	4.5	3.9	4.3	4.3	**
K, %	1.6	2.4	2.7	2.4	2.7	*
K uptake, mg/pot	40	75	74	73	80	**
Ca, %	0.89	0.89	0.92	0.85	0.78	***
Mg, %	0.73	0.65	0.62	0.58	0.56	***
S, %	0.19	0.18	0.19	0.17	0.19	***
Zn, ug/g	39	46	39	61	44	***
B, ug/g	12	16	16	15	15	ns
Fe, ug/g	62	64	56	57	58	**
Cu, ug/g	10	9	16	8	11	***

*Effect of sediment source significant at the 10% level.
**Effect of sediment source significant at the 5% level.
***Effect of sediment source significant at the 1% level.
ns = not significant at the 10% level.

Field Study

The original field study was sited on an old alfalfa field. Because plots were not established until June, Sudan grass was planted instead of corn. The yields and elemental composition of the plant tissue from the two harvests taken in 1980 are given in Table 39.

TABLE 39: Effect of Sediment Rate on Yield, N and P Uptake, and Concentration of Various Elements in Sudan Grass Tissue from the 1980 Field Study*

	1st harvest (mT/ha)				2nd harvest (mT/ha)			
Variable	0	22.4	44.8	89.6	0	22.4	44.8	89.6
Yield, mT/ha	2.40a	3.27b	2.96ab	2.96ab	2.31a	2.96b	2.91ab	3.09b
N, %	3.31	3.24	2.99	3.23	2.48	2.84	2.56	2.94
N uptake, kg/ha	79a	105b	87ab	96ab	57a	84ab	75ab	91b
P, %	0.33	0.31	0.31	0.31	0.35	0.37	0.36	0.35
P uptake, kg/ha	7.8	10.3	9.3	8.9	8.0	11.0	10.5	10.8
Zn, ug/g	48	53	54	56	42	42	43	42
Mn, ug/g	47	45	42	38	62	56	60	59
Cu, ug/g	16	18	16	17	16	18	16	17
Cd, ug/g	0.28	0.25	0.16	0.22	0.28	0.25	0.16	0.22
As, ug/g**								
Pb, ug/g**								

*Values followed by the same letter in a given row for the same harvest are not significantly different at the 5% level. If a row contains no letter, the differences are not significant.

**Below detection.

Even though the plots were established on an old alfalfa field, there was a significant increase in both yield and N uptake with the first sediment rate in the first harvest and from the third sediment rate in the second. Yield but not N uptake was increased significantly by the first sediment rate. These were the only significant effects of sediment addition.

Because the whole plot area was covered with sediment from a nearby sediment lagoon in the fall of 1980, the treatments were reestablished on a different site that had been in pasture for many years. Corn was planted in 1981 and the values for N and P in ear leaf, stover and grain and for yield and N and P uptake of stover and grain are given in Table 40. No significant effects of sediment application were noted.

TABLE 40: Effect of Sediment Rate on N and P Concentrations in Corn Ear Leaves at Silking and in Stover and Grain, and on Yield and N and P Uptake for Stover and Grain from the 1981 Field Study

Element	Ear Leaf (mT/ha)				Stover (mT/ha)				Grain (mT/ha)			
	0	22.4	44.8	89.6	0	22.4	44.8	89.6	0	22.4	44.8	89.6
Yield					6.03	7.12	5.89	6.03	7.77	7.80	7.84	8.31
N, %	2.62	2.80	2.48	2.75	0.53	0.55	0.51	0.53	1.22	1.21	1.18	1.24
N, kg/ha					32	39	39	32	94	94	93	103
P, %	0.32	0.33	0.31	0.32	0.07	0.07	0.09	0.06	0.26	0.25	0.26	0.25
P, kg/ha					4.2	5.0	5.3	3.6	20.2	19.5	20.4	20.8
K, %	1.7	1.7	1.7	1.8	0.78	0.88	0.90	0.96	0.34	0.34	0.35	0.33
Ca, %	0.52	0.53	0.52	0.52	0.27	0.28	0.25	0.25	0.005	0.004	0.003	0.004
Mg, %	0.53	0.51	0.51	0.49	0.33	0.36	0.31	0.31	0.099	0.097	0.098	0.094
S, %	0.20	0.23	0.22	0.26	0.077	0.088	0.088	0.097	0.10	0.10	0.10	0.10
Zn, ppm	41	40	40	47	24	23	27	32	18	17	18	18
B, ppm	6.1	6.2	6.4	7.0	6.8	7.4	7.5	7.2	4.3	3.7	3.7	4.7
Fe, ppm	83	88	80	87	120	90	120	71	10	5.9	5.7	6.3
Cu, ppm	12	12	12	13	6.0	7.1	5.5	6.3	4.2	2.3	2.3	1.9*
Al, ppm	71	59*	74	80	98	108	119	51	55*	41*	41*	35*
Mn, ppm	44	42	39	40	23	28	29	20	3.6	3.4	3.4	3.3

* one or more blocks reported are below detection limit.

In 1982, corn was planted again, but cows broke through the fence after the ear leaf samples had been taken and trampled a large portion of the plot area to the point that grain yields could not be obtained. Samples of grain were taken from standing stalks within the plots for chemical analysis, however. The results of the analyses of the ear leaf and grain samples are given in Table 41. Because this was the final field trial, as many elements were run on the tissue as was possible with the analytical programs available on the ICP.

TABLE 41: Effect of Sediment Rate on the Concentrations of 16 Elements in Samples of Corn Ear Leaves at Silking and in Grain from the 1982 Field Study

	Ear Leaf (mT/ha)				Grain (mT/ha)			
Element	0	22.4	44.8	89.6	0	22.4	44.8	89.6
N, %*	2.56	3.12	2.71	2.60	1.36a	1.42ab	1.50ab	1.68b
P, %	0.30	0.29	0.31	0.30	0.30	0.29	0.29	0.33
K, %	1.6	1.5	1.5	1.6	0.39	0.40	0.39	0.46
Ca, %	0.69	0.67	0.72	0.64	0.01	0.01	0.01	0.01
Mg, %	0.51	0.55	0.57	0.50	0.12	0.12	0.12	0.13
S, %	0.29	0.28	0.30	0.28	0.12	0.12	0.13	0.14
Zn, ppm	25	25	31	26	23	23	23	26
B, ppm	6.0	3.9	6.2	4.2	4.0	1.9	4.3	2.7
Fe, ppm	88	86	89	87	19	19	19	22
Cu, ppm	12	11	13	12	2.6	2.8	2.5	2.9
Al, ppm	49	47	49	49	13	9	7	10
Mn, ppm	43	41	47	38	6.5	5.9	6.4	7.5
Co, ppm	5.5	5.1	5.9	5.3	5.2	4.8	6.2	4.6
As, ppm	0.6	0.5	0.9	1.1	--	--	--	--
Cd, ppm	0.34	0.37	0.37	0.39	--	--	--	--
Pb, ppm	1.6	1.5	2.0	1.8	--	--	--	--

*For N in corn grain, values not followed by the same letter differ significantly at the 5% level. None of the other analyses show significant differences with sediment rate.

Statistical analysis showed that there was a significant increase in N concentration in the corn grain at the highest sediment addition. This increase was not apparent in the ear leaf samples. This appeared to be the only significant effect of sediment application found in two years of field studies on corn at this site.

CONCLUSIONS

The Lilly Lake project represented a major dredging operation on a small lake. Water quality degradation resulted from disturbance of the flocculent, organic sediments; however, water chemistry remained within tolerable limits. In this case, the sediments did not contain any particularly problematic pollutants. High NH_4-N concentrations in the sediment interstitial waters were the most serious concern.

During dredging, density increased for algae and a species of zooplankton. Casual observations of the fish population revealed no adverse impact. Although not specifically monitored in the summers of dredge activity, physical disruption of the entire lake basin would presumably have caused near decimation of the benthic invertebrates and rooted macrophytes. The benthos were reexamined about 1.5 years after project completion. At that time both numbers and diversity were greater than in the pre-treated lake. Macrophytes were investigated the summer following dredge removal. Diversity was down about 50%, with most of the inhabitable lake bottom invaded by Chara sp., a pioneer species. By 1982 this plant was being out-competed by a mixture of rooted macrophytes, and diversity was improving.

The study was terminated about 2.5 years after lake deepening. Dissolved oxygen levels had not dropped below 7 mg/L at any depth during the three intervening winters, whereas fish kills had been a common occurrence previously. General water chemistry was good and reflected the enhanced groundwater inflow. Chlorophyll a levels averaged under 5 ug/L. Rooted macrophytes occupied 75% of the lake basin but were not growing to the lake surface except near shore. Plant biomass was reduced significantly, and, although the situation was still evolving, biomass was expected to remain down because of the conversion of muck to sand bottom. Project duration limited the ability to assess the impact on the fish population; however, the various measures of habitat suitability indicate that an improved fishery will develop in the future.

An initial effect of sediment disposal in the modified gravel pit was an elevated groundwater table. The largest increase in groundwater level occurred in the wells closest to the disposal basins. However, the elevated water levels generally declined soon after the disposal activity stopped in 1978 and 1979. Water and sediment levels in the basins did not show such large declines. This was probably due to the sediment's ability to seal the bottom and sides of each basin and restrict the outward seepage of water. Therefore, most water seepage occurred near the water surface in the basins as they filled.

In terms of groundwater quality, NH_4-N and NO_3-N may have entered the groundwater during 1978 and 1979. However, the impacts were relatively minor and localized in the immediate vicinity of the gravel pit area. The general conclusion was that no significant adverse effect on groundwater quantity or quality occurred as a result of sediment disposal into the gravel pit.

Similar effects were observed, and conclusions reached at the diked disposal area. An elevated groundwater table showed up in the wells situated in the most permeable soil first and in the least permeable soil later. However, the water levels again declined soon afterward due to the sediment's ability to seal the bottom and sides of the diked area and restrict the outward seepage of water. In terms of groundwater quality, NO_3/NO_2-N concentrations rose to 9.6 mg/L in the most permeable well (D6). Also, in two cases, the NH_4-N concentrations increased a short time after the NO_3/NO_2-N. In one well the change may have been related to sediment disposal; however, the other well probably responded to ammonia fertilization of a nearby farm field.

The sediment from Lilly Lake in Kenosha County, Wisconsin is higher in pH, total C, COD, loss on ignition, total N and N release on three months' incubation than the 11 other lake sediments included in the study. Its potential for increasing P availability in soils appears to be very low compared with most of the other sediments. Laboratory data suggest that some P immobilization might actually result from application of this sediment, but field data do not show any effect on P availability.

From a practical standpoint, the only beneficial effect found was an increase in N availability following sediment application. This occurred in both greenhouse and field trials. No other beneficial or detrimental effect showed up in the field trials. Tests for changes in physical properties of the soil associated with sediment applications were not made.

From the data obtained in this study, it would appear that there are no harmful effects from applications up to and probably exceeding 89.6 mT/ha (40 T/A) on a dry-weight base. Small but significant increases in available N should result from these additions through mineralization of organic N, and the increases should persist over a number of years.

REFERENCES

Alden, W. C. 1918. Quaternary geology of southeastern Wisconsin. U.S. Geological Survey, Professional Paper 106. U.S. Government Printing Office, Washington, D.C. 356 p.

American Public Health Association. 1975. Standard methods for the examination of water and wastewater, 14th edition. Washington, D.C. 874 p.

Andern, A. and J. Stolzenburg. 1978. Atmospheric deposition of lead and phosphorus on the Menomonee River watershed. Water Resources Center, Univ. of Wis., Madison.

Bachmann, R. W. and D. E. Canfield. 1979. Role of sedimentation in the phosphorus budget of natural and artificial Iowa lakes. Completion Report, Proj. No. A-063-IA. Iowa State Univ., Ames. 95 p.

Beauheim, R. 1980. The effects of dredging on groundwater-lake interactions at Lilly Lake, Wis. M.S. Thesis. Dept. Geology & Geophysics. Univ. of Wis., Madison. 178 p.

Bouwer, H. 1978. Groundwater hydrology. McGraw-Hill Book Company, New York. 480 p.

Brandlova et al. 1972. The Cladocera of Ontario with remarks on some species and distribution. Can. J. Zool. 50:1373-1403.

Brooks, J. Y. 1957. The systematics of North American Daphnia. Mem. Connecticut Acad. Arts Sci. 13:5-180.

Chaney, R. L. 1983. Potential effects of waste constituents on the food chain, pp. 152-240. in Land treatment of hazardous wastes. J. F. Paco, P. B. Marsh and J. M. Kla (eds). Noyes Data Corp., Park Ridge, N.J.

Deevy, E. S. and G. B. Deevy. 1971. The American species of Eubosmina Seligo. Limnology and Oceanography 16:201-218.

Dillon, P. J. and F. H. Rigler. 1974. A test of a simple nutrient budget model predicting the phosphorus concentrations in lake water. Jour. Fish. Res. Bd. Canada 31:1771-1778.

Druckenmiller, H. S. 1972. Updated age-length averages for southeastern Wisconsin game fishes. Wis. Dept. Natural Resources, Management Report No. 55. Madison, Wis. 5 p.

Dunst et al. 1974. Survey of lake rehabilitation techniques and experiences. Wis. Dept. Natural Resources, Technical Bulletin 75. Madison, Wis. 179 p.

Dunst, R. C. 1982. Sediment problems and lake restoration in Wisconsin. Environment International 7:87-92.

Fassett, N. C. 1960. A manual of aquatic plants. Univ. of Wisconsin Press. Madison, Wis. 405 p.

Hasler, A. D. 1947. Eutrophication of lakes by domestic drainage. Ecology 28:383-395.

Helwig, J. T. and K. A. Council. 1979. SAS user's guide - 1979 edition. SAS Inst., Inc. Raleigh, N.C.

Hilsenhoff, W. L. 1975. Aquatic insects of Wisconsin. Wis. Dept. Natural Resources, Technical Bulletin 89. Madison, Wis. 52 p.

Hubbs, C. L. and K. F. Lagler. 1958. Fishes of the Great Lakes region. Cranbrook Institute of Science, Bulletin 26. Bloomfield Hills, Michigan. 213 p.

Huber - Pestalozzi, G. 1938. Das Phytoplankton des Susswassers. Die Binnengewasser. Bd. XVI. Teil 1.

Hustedt, F. 1930. Bacillariophyta (diatomae). Heft. 10. In Die Susswasser - Flora Mitteleuropas. A. Paschner (ed.). Gustau Fisher, Jena, Germany.

Hutchinson, R. D. 1970. Water resources of Racine and Kenosha counties, southeastern Wisconsin. U.S. Geological Survey, Water Supply Paper 1878. U.S. Government Printing Office, Washington, D.C. 63 p.

Imboden, D. M. and A. Lerman. 1978. Chemical models of lakes. In Lakes: chemistry, geology, physics. A. Lerman (ed.). Springer - Verlag, New York.

Jackson, M. L. 1958. Soil chemical analysis. Prentice-Hall, Englewood Cliffs, N. J.

Kamprath, E. J. and M. E. Watson. 1980. Conventional soil and tissue tests for assessing the phosphorus status of soils, pp. 433-469. In The role of phosphorus in agriculture. F. E. Khasawneh, E. C. Sample and E. J. Kamprath (eds.). Am. Soc. of Agron., Madison, Wis.

Keeney, D. R., J. G. Konrad, and G. Chesters. 1970. Nitrogen distribution in some Wisconsin lake sediments. J. Water Pollut. Control Fed. 42:411-417.

Keeney, D. R., K. W. Lee, and L. M. Walsh. 1975. Guidelines for the application of wastewater sludge to agricultural land in Wisconsin. Wis. Dept. of Natural Resources, Technical Bulletin 88. Madison, Wis.

Konrad, J. G., D. R. Keeney, G. Chesters, and K. L. Chen. 1970. Nitrogen and carbon distribution in sediment cores of selected Wisconsin lakes. J. Water Pollut. Control Fed. 42:2094-2101.

Lagler, K. F. 1961. Freshwater fishery biology. W. C. Brown Company. Dubuque, Iowa. 421 p.

Lee, D. R. 1977. A device for measuring seepage flux in lakes and estuaries. Limnology and Oceanography 22(1):140-147.

Liegel, E. A., C. R. Simson, and E. E. Schulte. 1980. Wisconsin procedures for soil testing, plant analysis and feed and forage analysis. Univ. of Wisconsin Extension, Soil Fertility Series No. 6.

Lillie, R. A. and J. W. Mason. 1983. Limnological characteristics of Wisconsin lakes. Wis. Dept. Natural Resources, Technical Bulletin 138. Madison, Wis. 116 p.

Lund et al. 1958. The inverted microscope method of estimating algal numbers and statistical basis of estimations by counting. Hydrobiol. 11:143-170.

Mace, S. E. 1979. Senecella calanoides Juday (Calanoida, Copepoda), Mesocyclops leuckarti Claus (Cyclopoida, Copepoda), and Daphnia laevis Birge (Cladocera) in inland Wisconsin lakes. Wis. Acad. Sci., Arts and Letters 67:171-173.

McCafferty, M. P. 1982. Aquatic entomology. Science Books International. Boston, Massachusetts. 448 p.

Megard, R. O. 1972. Phytoplankton, photosynthesis, and phosphorus in Lake Minnetonka, Minnesota. Limnology and Oceanography 17(1):68-87.

Mehta, N. C., J. O. Legg, C. A. I. Goring, and C. A. Black. 1954. Determination of organic phosphorus in soils: I. Extraction method. Soil Sci. Soc. Am. Proc. 18:443-448.

Murphy, J. and J. P. Riley. 1962. A modified simple solution method for the determination of phosphate in natural waters. Anal. Chim. Acta 27:31-36.

Olsen, S. R., C. V. Cole, F. S. Watanabe, and L. A. Dean. 1954. Estimation of available phosphorus in soils by extraction with sodium bicarbonate. U.S. Dep. Agric. Circ. 939.

Patrick, R. and C. W. Reimer. 1966. The diatoms of the United States. Vol. I. Academy of Natural Sciences, Philadelphia.

Patrick, R. and C. W. Reimer. 1975. The diatoms of the United States. Vol. II. Academy of Natural Sciences, Philadelphia.

Pennak, R. W. 1953. Freshwater invertebrates of the United States. Ronald Press, New York. 769 p.

Peterson, G. W. and R. B. Corey. 1966. A modified Change and Jackson procedure for routine fractionation of inorganic soil phosphates. Soil Sci. Soc. Amer. Proc. 30:563-565.

Pierce, N. D. 1970. Inland lake dredging evaluation. Wis. Dept. Natural Resources, Technical Bulletin 46. Madison, Wis. 68 p.

Prescott, G. W. . 1962. Algae of the western Great Lakes area. W. C. Brown Company. Dubuque, Iowa.

Reckhow et al. 1980. Modeling phosphorus loading and lake response under uncertainty: a manual and compilation of export coefficients. EPA 440/5-80-011. Washington, D.C. 214 p.

Sakamoto, M. 1966. Primary production by phytoplankton community in some Japanese lakes and its dependence on lake depth. Arch. Hydrobiol. Bd. 62:1.

Schindler, D. W. 1977. Evolution of phosphorus limitations in lakes. Science 195:260-262.

Schulte, E. E. 1980. Recommended soil organic matter tests, pp. 28-31. In Recommended chemical soil test procedures for the North Central Region. W. C. Dahnke (ed.). Pub. No. 221 (Revised), Oct. 1980, North Dakota Agric. Exp. Stn. Fargo, N.D.

Skuja, H. 1948. Taxonomie des Phytoplanktons einiger Seen in Uppland, Schweden. Symb. Bot. Upsal. 9(3):1-339.

Smith, G. M. 1950. The freshwater algae of the United States. 2nd edition. McGraw-Hill, New York. 719 p.

Smith, K. and C. H. Fernando. 1978. A guide to the freshwater calanoid and cyclopoid copepod Crustacea of Ontario. Univ. of Waterloo, Biology Series #18. Ontario, Canada.

Sonzogni et al. 1977. Use of large submerged chambers to measure sediment-water interactions. Water Research 11:461-464.

Stumm, W. and J. O. Leckie. 1971. Phosphate exchange with sediments: its role in the productivity of surface waters. Proc. 5th Internat. Wat. Poll. Res. Conf. Pergamon Press, London.

Thurston et al. 1974. Aqueous ammonia equilibrium calculations. Fisheries Bioassay Laboratory, Technical Report No. 74-1. Montana State Univ., Bozeman.

Torke, B. 1976. A key to the identification of the cyclopoid copepods of Wisconsin, with notes on their distribution and ecology. Wis. Dept. Natural Resources, Research Report 88. Madison, Wis.

U.S. Department of Agriculture, Soil Conservation Service. 1970. Soil Survey - Kenosha and Racine counties, Wisconsin. U.S. Government Printing Office, Washington, D.C. 113 p.

U.S. Environmental Protection Agency. 1973. Measures for the restoration and enhancement of quality of freshwater lakes. EPA-430/9-73-005. Washington, D.C. 238 p.

U.S. Environmental Protection Agency. 1976. Quality criteria for water. U.S. Environ. Prot. Agency. Washington, D.C. 256 p.

U.S. Environmental Protection Agency. 1979. Criteria for classification of solid waste disposal facilities and practices. Federal Register 44(179):53438-53464.

U.S. Environmental Protection Agency. 1979. Methods for chemical analysis of water and wastes. EPA/4-79-020. Washington, D.C. 490 p.

U.S. Public Health Service. 1962. Drinking water standards. Department of Health, Education and Welfare, Publ. No. 956. U.S. Govt. Printing Office, Washington, D.C. 61 p.

Uttormark et al. 1974. Estimating nutrient loadings of lakes from nonpoint sources. EPA-660/3-74-020. Washington, D.C. 112 p.

Vollenweider, R. A. 1975. Input-output models with special reference to the phosphorus loading concept in limnology. Schweiz. A. Hydrol. 37:53-83.

Vollenweider, R. A. 1976. Advances in defining critical loading levels for phosphorus in lake eutrophication. Mem. Ist. Ital. Idrobiol. 33:53-83.

Voss, E. G. 1972. Michigan flora. Part I: gymnosperms and monocots. Cranbrook Institute of Science, Bulletin 55. Bloomfield Hills, Michigan. 488 p.

Weber, C. I. 1971. A guide to the common diatoms at water pollution surveillance system stations. U.S. Environ. Prot. Agency, National Environmental Research Center. Cincinnati, Ohio. 101 p.

Wildung, R. E., R. L. Schmidt, and A. R. Gahler. 1974. The phosphorus status of eutrophic lake sediments as related to changes in limnological conditions--total, inorganic, and organic phosphorus. J. Environ. Qual. 3:133-138.

Yeatman, H. C. 1959. Cyclopoida, pp. 795-814. In Freshwater biology, 2nd edition. W. T. Edmondson (ed.). Wiley, New York.