FOX CHAIN OF LAKES INVESTIGATION AND WATER QUALITY MANAGEMENT PLAN

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CONTENTS

(Concluded on next page)

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INTRODUCTION

The Fox Chain of Lakes consists of nine major internally connected lakes located in Lake and McHenry Counties in northern Illinois. The main inflow into the lake system is the Fox River. Other tributaries of importance are the Sequoit, Nippersink, and Squaw Creeks, and Lily Lake Drain. The drainage area of the Fox River at Johnsburg, immediately downstream of the Fox Chain of Lakes, is 1184 square miles. Approximately 75 percent of the drainage area lies in Wisconsin. *[For locations, see figure 13, page 34.]*

The surface water levels of the system are maintained by a dam at McHenry. At normal pool elevation, the water surface area of the lake system is about 6850 acres. The shoreline extends 48 miles and there is an additional 30 miles of dredged channel shoreline to accommodate demands for lake frontage property.

Cultural Development and Land Use

The lands surrounding the Fox Chain of Lakes were being cleared of forests by 1840. The cleared land was converted to row-crop agriculture. From that original agricultural base, the shores of the Chain were steadily converted to residential and recreational uses. The interconnected bodies of water, natural shoreline, picturesque beauty, and proximity to Chicago have combined to make the Chain a popular area for swimming, boating, water skiing, fishing, and resort development.

The shores of these lakes were first dotted by summer homes but those have long been converted to year-round occupancy. All of the shore naturally suitable (and much of it not suitable) for construction of homes, restaurants, marinas, and other recreational facilities has been developed.

Artificial channels have been dredged to accommodate demands for lake frontage property. The recreational value of the lakes is a major economic influence in the area. Regional planners report there is no area in the surrounding six counties with greater potential for meeting the varied recreational needs of people in the region. However, the full potential of the Chain area is not being realized because of the eutrophic conditions existing in the system's waters.

The Problem

The lakes and their associated shorelands constitute a valuable natural resource that is showing the effects of use and abuse by more and more people. A population with growing leisure time, greater affluence, and increased mobility has imposed spiraling demands on the recreational resources of the region. The Illinois Department of Conservation (1964) in a special report indicated that 4000 boats existed in the lake waters in 1915. Today, Lake County officials estimate that 35,000 boats are harbored on the Chain and that over 2 million gallons of gasoline are used annually by boaters.

Nuisance algal blooms have been a source of recorded complaints on the Chain for at least the past 35 years. Such protests expressed apprehension regarding the curtailment of bathing, boating, water skiing, fishing, lowering of property values, cancellation of resort trade, and impairment of the picturesque beauty of the Chain. Luxuriant and prolonged algal blooms, periodic fish kills, and offensive odors have been reported during many investigations (Illinois Department of Conservation, 1964; Lake County Health Department, 1962; and U.S. Public Health Service, 1963; *see References in Part 3*) .

With the exception of a 14-month study by the Lake County Health Department, all earlier investigations have been limited in scope and have been predicated upon complaints. An occasional investigation has been undertaken with a specific objective in mind such as a fish population count (Illinois Department of Conservation, 1964) or bacterial density enumeration (Lake County Health Department, 1962).

The eutrophic conditions existing annually in the middle and lower Fox River in Illinois are related to the Fox Chain of Lakes. The Chain, being upstream, contributes. massive concentrations of algae to the river waters, impairing the recreational and aesthetic enjoyment sought by the residents of the valley. The benefits of needed waste treatment facilities along the course of the Fox River are not likely to be realized so long as its headwaters remain grossly eutrophic.

Clearly, a thorough evaluation was needed of the processes governing algal growth in order to develop technically sound and economically feasible management plans for controlling the problems of excessive fertilization of the Chain of Lakes. The successful application of a strategy for restoring desirable water quality within the Chain would serve two purposes. It would offer relief consistent with the needs of sportsmen, swimmers, boaters, and sightseers as well as persons depending upon the active use of the Chain for their livelihood; and it would permit the subsequent development of a water quality management plan for the Fox River in Illinois which is greatly influenced by the lake system.

Objectives and Scope

Primary objective of this investigation was to gain an understanding of the Fox Chain of Lakes' hydrological, physical, chemical, and biological characteristics. This knowledge would be used to develop short-term and long-term strategies leading to restoration and preservation of desirable water quality in the Chain. Efforts were directed to updating hydrologic maps for the lake system, determining the water circulation patterns, defining the extent of thermal and dissolved oxygen stratifications in the lakes, documenting the temporal and spatial variations in chemical and biological water quality, investigating the geology and geochemistry of the lake bottoms, identifying the critical elements responsible for the profuse algal growth in the lake system, and developing a nutrient budget for the lake system.

The State Water Survey in cooperation with the State Geological Survey began the 18-month detailed investigation of the Fox Chain of Lakes in October 1974. The investigation was encouraged and funded by the Illinois Institute for Environmental Quality. The U.S. Geological Survey collected the interflow data between lakes and developed rating curves for three newly installed gaging stations in the Chain of Lakes region. Water samples from the tributaries and outflow from the system were collected by the Lake County Health Department. The Illinois Department of Conservation assisted in collecting rainwater samples.

Report Plan

This report is a cooperative project of the Illinois State Water Survey and State Geological Survey. Part 1, prepared by the Geological Survey, discusses the geologic history and character of bottom sediments. Parts 2 and 3 were prepared by the Water Survey. Part 2 presents the hydraulic and hydrologic conditions of the Chain. Part 3 discusses the water quality and sources of nutrients and the living organisms. Part 3 also evaluates remedial measures found effective in other locations and proposes a reliable water management program.

Acknowledgments

This investigation, sponsored and financially supported by the Illinois Institute of Environmental Quality, was conducted under the general supervision and guidance of Dr. William C. Ackermann, Chief of the Illinois State Water Survey. Many Water Survey personnel contributed to the final preparation of this report. John W. Brother, Jr., and William Motherway, Jr., prepared the illustrations for Parts 2 and 3. J. Loreena Ivens and Patricia A. Motherway edited the manuscript. Suzi O'Connor typed the camera copy.

The authors of Part 1 are indebted to the following members of the Illinois State Geological Survey staff for their performance of chemical analyses: John D. Steele and Jim F. Ashby, atomic absorption spectrometry analyses; Eleanor F. Hopke, optical emission spectography; Joyce Kennedy Frost, instrumental neutron activation analysis; Larry R. Camp, radiochemical neutron activation analysis; David B. Heck and Mark A. Seifrid, inorganic carbon and boron tests; Lawrence Kohlenberger and Renee M. Gracon, total carbon and nitrogen analysis; John K. Kuhn and Leonard R. Henderson, X-ray fluorescence spectrometry; and Dennis D. Coleman and Chao-Li Lui, radiocarbon dating. Grain-size analyses of sediments were supervised by Susan K. Friesen. Statistical analyses were performed by John A. Schleicher. Figures in Part 1 were prepared by members of the Geological Survey staff.

James King, Illinois State Museum, did the palynology and contributed to the understanding of the sedimentation rates. Jerry T. Wickham assisted with all field work and data compilation. The Illinois Division of Water Resources furnished a boat from which to collect samples and allowed the use of their warehouse and dock facility at McHenry Lock and Dam where Lockmaster Frank Novak was most helpful.

The following Water Survey Hydraulic and Hydrology personnel participated in the Part 2 investigation under the supervision of John B. Stall, Head of the Hydrology Section. Ji-Ang Song and Jeffery C. Elledge assisted in analysis of the data. David J. Kisser, Gary L. Ahlberg, and Steven Glenn carried out the field work and collected the circulation data. Wilbur Debolt, Jr., assisted in the installation of the wind-set.

All horizontal and vertical control surveying was performed by the Engineers Surveying and Calculating Service of Mount Prospect through a subcontract. This firm also collected all of the sounding information. The Illinois Division of Water Resources supplied all long-term stage records in the Chain and flow records at McHenry Lock and Dam. The U.S. Geological Survey (Madison, Wisconsin, office) supplied flow records for the Fox River at Wilmot, Wisconsin. The Ranger's office at Chain of Lakes State Park furnished a boat whenever needed, and stored instruments for our investigators. All property owners in the Chain of Lakes area were helpful, cooperative, and pleasant to those collecting data for the study.

The following personnel, under the supervision of Ralph L. Evans, Head of the Water Quality Section, participated in various phases of the Part 3 investigations, namely, sample collection, analyses, data reduction and evaluation, and report writing: Gary Benker, Davis B. Beuscher, Thomas A. Butts, David L. Hullinger, Melbern E. Jannett, Shundar Lin, Meri Phillips, Dorothy L. Richey, Donald Roseboom, Donald H. Schnepper, Wun-Cheng Wang, and Jack W. Williams, Jr. Rebecca Phillips typed the original manuscript. Dr. J. C. Neill, Water Survey statistician, performed the statistical analyses and Calcomp plotting.

Special thanks go to Eugene Theios, William Garrison, Charles Bernard, Glenn Zink, and William Mellon of the Lake County Health Department for providing assistance in the collection of tributary and rainwater samples, and in furnishing information on septic tank disposal systems in the Chain area.

The wastewater treatment superintendents and operators provided assistance in obtaining effluent samples. The assistance of Joe Halbmeir, E. W. Bates, Gary Bill, Robert R. Leber, John F. McMahon, Charles Maplethorpe, Thomas Evans, and Robert Martz is appreciated. Michael Gaudiello of the Woodstock Die Casting Company furnished trade effluent samples. Gary Parisi provided access and a boat for water sampling in Cedar Lake.

Part 1. Geology and Sediment Geochemistry of the Lakes

GEOLOGIC HISTORY

The geologic history of the Fox Chain of Lakes essentially began with the melting of the last of the great continental glaciers from Illinois. That last episode of glaciation, the Woodfordian substage of the Wisconsinan stage, had begun when ice flowed southwestward from the Lake Michigan Basin 23,000 years ago. The time has been documented by a radiocarbon date (I-2783) on the organic Robein Silt which was penetrated by a boring made 20 miles south of the lakes. The Woodfordian glaciers flowed southward, covering the northeastern quarter of Illinois, and reached their maximum extent at Peoria and Shelbyville 20,000 years ago. For the next 6000 years the glacial front retreated northward, a withdrawal interrupted by many minor readvances.

The final episode of Woodfordian glaciation occurred about 14,000 years ago, when two minor sublobes of the glacier met along what is now the western side of the Fox Chain of Lakes. Ice flowed into McHenry County from the north, bringing from the northern Silurian dolomite area the sandy yellow-brown till now named the Haeger Till Member of the Wedron Formation (Willman and Frye, 1970). Meltwater from the ice deposited great thicknesses of sand and gravel outwash. At the same time the other lobe of glacial ice flowed westward or southwestward into Lake County from the Lake Michigan Basin and deposited the olive-gray clayey till now referred to as the Wadsworth Till Member of the Wedron Formation (Willman and Frye, 1970). That till was derived from erosion of Devonian shales in the Lake Michigan Basin.

The two minor sublobes met along what is now the Lake-McHenry County line just west of the Chain of Lakes. The line is marked by a prominent escarpment separating the uplands (elevation 268 meters) (880 feet) of McHenry County from the lowlands (elevation 229 meters) (750 feet) surrounding the lakes. The line also separates the coarse-grained surficial till, sand, and gravel materials of McHenry County from the clayey till, silts, clays, and peats of the lake basins and regions to the east in Lake County. The glacial lobe coming from the north melted first, possibly 13,000 years ago, allowing the Fox River to begin flowing southward as an ice marginal stream on the western front of the lobe coming from the east. The eastern lobe then melted quickly, leaving a hummocky kame-and-kettle topography formed by the deposition of vast quantities of glacial debris. Buried in that debris were many large blocks of ice that, as they melted, left depressions filled with meltwater. It was the melting of these buried ice blocks that formed the Chain of Lakes in the random pattern that we see today. A detailed map of the surficial deposits in Lake County was published in Larsen (1973), and a similar map of the surficial deposits in McHenry County may be found in Hackett and McComas (1969).

The original depths of the lakes are not known but they probably were not much deeper than the 12 meters (40 feet) maximum observed today. Just after the lakes were formed, sedimentation rates were very high because surrounding slopes slumped into more stable positions and the Fox River brought in muddy glacial meltwater. After the glacial ice melted and the climate began to warm, sedimentation rates slowed, but they have risen again in the past 150 years as human activities in the surrounding watershed have increased.

Volo Bog, 4 kilometers (2.4 miles) south of Pistakee Lake, was another lake formed at the same time and by the same processes as the Chain, but it is now almost completely filled by muck, peat, and marl. McComas et al. (1972) described in considerable detail the formation and history of that bog, and most of their discussion applies equally well to the Chain. The Fox Chain of Lakes differs from the bog principally in the amount of filling that has taken place.

It is no accident that the lakes along the Fox River (Grass, Nippersink, and northern Pistakee Lakes) are shallow today and that those away from the river channel (Marie, Channel, and Catherine Lakes) are relatively deep. Where the river's course is through deep lakes, the rate of flow drops and much of the river's sediment load is deposited. Thus the sedimentation rates are high in the first deep-water areas encountered as the river flows through the Chain. Natural infilling of the lakes undoubtedly progressed from north to south and is largely complete in Grass, Nippersink, and northern Pistakee Lakes.

In 1907, the construction of the first dam across the Fox River at McHenry increased water levels in the Chain and also reduced water velocities, causing an increase in sedimentation rates. Sedimentation also increases after each episode of dredging. Grass Lake is an exception because it has an erosional bottom today. No significant quantity of recent sediment has been deposited in Grass Lake, and its shore is eroding slightly. Possibly the slight rise of water level caused by the dam allowed wave action to scour the bottom of this the shallowest lake in the Chain.

The area surrounding the Chain of Lakes is included in four major soil associations which are described in detail in the excellent soil reports for Lake and McHenry Counties by Paschke and Alexander (1970) and Ray and Wascher (1965). The area west of Channel, Marie, and Fox Lakes and the areas surrounding Grass, Nippersink, and Pistakee Lakes are included in the Marsh-Fox-Boyer soil association. It is a marshy area, level to rolling, with well to moderately well drained soils that are moderately deep, lie over sand and gravel, and have rapid to moderate permeability.

The areas east of Lakes Catherine and Channel, those southwest of Lake Marie, and those west of Bluff, Spring, and Petite Lakes are in the Nappanee-Montgomery soil association. This area is made up of level to moderately sloping soils that are somewhat poorly drained and deep soils that have slow permeability, are level to depressional, and poorly to very poorly drained.

The east side of Bluff, Spring, Petite, and Fox Lakes is in the Morley-Markham-Haughton soil association, which includes deep soils of moderately slow permeability that are gently sloping to steep and well to moderately well drained. Level to depressional, very dark colored, very poorly drained organic soils also are in that association. The south side of Fox Lake is included in the ZurichGrays-Wauconda soil association, which consists of nearly level to moderately steep, well drained to somewhat poorly drained, deep soils that have moderate permeability.

TYPES OF SEDIMENT

Silt, sand, peat, marl, muck, glacial-lacustrine clay, and glacial till were found in 80 cores and 25 surface grab samples from the bottom of the Fox River Chain of Lakes (figure 1). Cores were taken with a Benthos open-tube gravity corer (Gross et al., 1970). Each core was collected in a plastic tube that had an inside diameter of 6.7 centimeters (2.6 inches). Most cores were from 80 to 120 centimeters (2.6 to 3.9 feet) long.

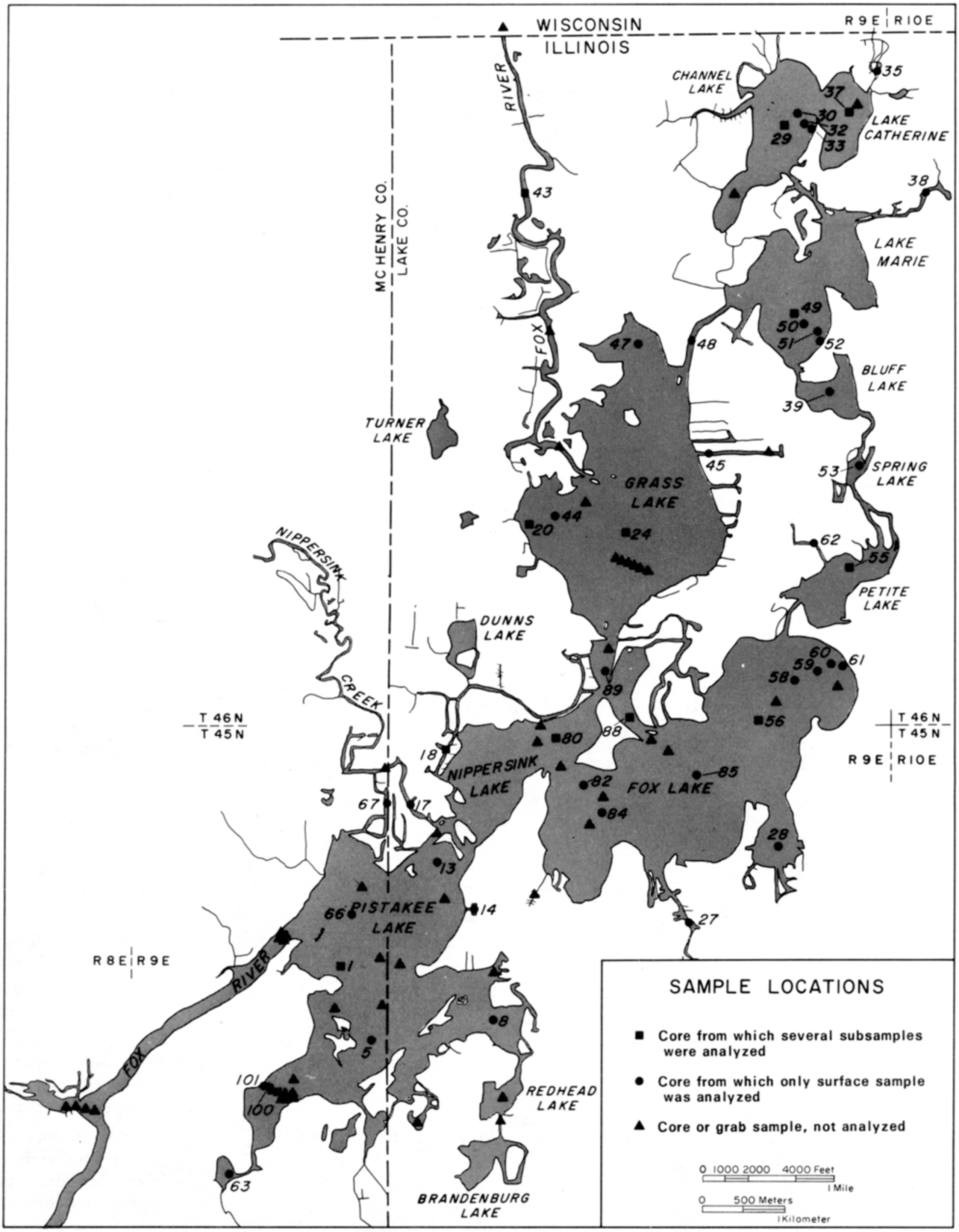

Figure 1. Sampling locations for bottom sediment samples and cores

The coring was done from a pontoon boat operated by the Division of Water Resources (Illinois Department of Transportation) at McHenry Dam. Some cores were extruded from their tubes and subsampled on the boat; others were returned to the Illinois State Geological Survey laboratories in Urbana for subsampling. The depth of penetration of the cores was not sufficient to permit examination of the entire postglacial sediment sequence in the lakes, and, therefore, little can be said about the deeper stratigraphic record. However, the cores and bottom samples taken furnish a relatively detailed picture of the types of sediment now accumulating in the Chain of Lakes system.

Nine different types of sediment can be identified in the top meter (3.3 feet) of sediment (table 1) in the Chain of Lakes, and those units have been simplified to the seven sediments shown in figure 2. Each sediment type represents a particular environment within the lake-river system.

Table 1. Types of Sediment in the Fox River Chain of Lakes

Type	*Major component*	*Minor component*	*Type of organic debris*	*Color*	*Type of environment*
Silt with sand and/or shells	Silt	Sand and/or shells	Coarse debris from marsh vegetation and algal remains	Black to brownish black	Open lake, shallow < 2 meters of water
Peaty silt	Silt	Plant fragments	Mostly peaty debris from marsh vegetation	Black	Open lake, shallow < 2 meters of water near marshes
Algal silt	Silt	Algal fragments	Algal filaments	Black	Open lake, deeper than 2 meters of water
Marl	Calcarenite and shells		Little	Light gray	Open lake, generally 2-4 meters deep in narrow zone between nearshore silty sediments and deep water algal silts
Muck	Silt	Clay	Fine to coarse debris from marshes, trees, other channel-side vegetation, and septic tank outfalls		Dredged channels
Peat	Plant debris		Coarse marsh vegetation	Brownish black	Peat marshes and places where lakes have transgressed older marshes
Sand	Sand	Shells, silt	Some coarse plant debris	Dark gray	Channel of Fox River and other tributary channels
Glaciolacustrine clay	Clay	Sand, silt	Very little debris	Gray	Areas of steep slope and other places where recent sediments have not accumulated or have been eroded
Till	Clay	Sand, silt	Very little debris	Gray	Areas of steep slope and other places where recent sediments have not accumulated or have been eroded

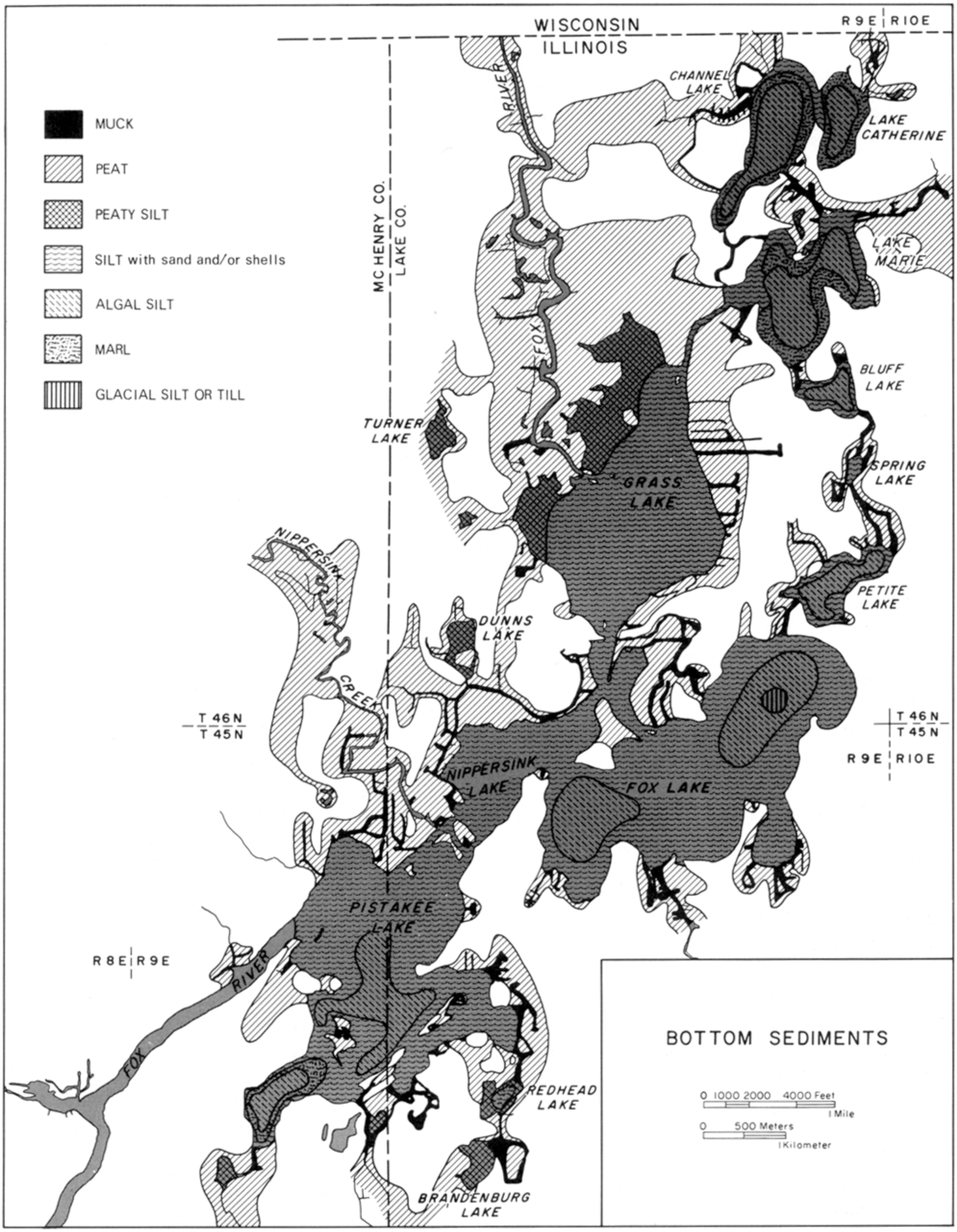

Figure 2. Distribution of various types of bottom sediments

Sixty-three selected sediment samples were analyzed chemically for 27 major and minor components (tables 2 and 3). Grain-size determinations were made on 61 samples. Pollen from three cores was examined to determine the percentage of *Ambrosia* (ragweed) pollen present. The geochemistry of each sediment type is distinctive and is related both to the environment of deposition as well as to the sediment components.

Lake Sediments

Black silt containing abundant peat-like plant debris has the widest distribution on the floors of the nine lakes studied. Three subtypes of black silt were identified on the basis of the kinds of plant debris they contain and on their sand or shell content.

Silt with Sand and/or Shells

Silt containing shells and/or sand is the most common type of sediment now accumulating in the lakes (figure 2). This type of sediment is generally found in water less than 2 meters (6.6 feet) deep (figure 3) in the larger channels connecting the lakes, in the shallow lakes, and in the near-shore areas of deep lakes. This sediment averages 71 percent silt (table 2). The sand content ranges from 0 to 15 percent and is highest in Nippersink Lake and in the portions of Grass and Pistakee Lakes that are most affected by the through-flow of Fox River water. Sand is also found in near-shore sediments where the shores are steep. Shells (mostly gastropods and a few clams) are abundant in shallow water away from the main channels between the lakes.

The sediment contains an average of 5 percent organic carbon. Coarse and fine plant debris is present in all samples. This type of silt has the most heterogeneous origin of all the lake sediments. It is a mixture of quartz silt and sand, and clay minerals carried into the lakes by creeks, rivers, and shore erosion; algal debris; plant debris derived from marshes and shore vegetation; and calcium carbonate from calcareous algae and whole and broken shells of gastropods and clams that lived on the lake bottom or on vegetation in the lakes.

Peaty Silt

A second sub-variety of black silt sediment is peaty silt, which differs from the sandy and shelly silt in having considerably fewer shells and proportionately more peaty organic matter (up to 26.5 percent organic carbon) (table 2). Samples of peaty silt were collected from areas into which the lakes have recently expanded at the expense of the surrounding peat bogs and marshes. Grass Lake is a good example. A comparison of present day maps of Grass Lake with those made over 50 years ago shows considerable portions of marsh now covered with 0.5 to 1 meter (1.6 to 3.3 feet) of water. The installation of McHenry Dam on the Fox River, which resulted in higher water levels and increased natural wave erosion of the shore, along with the activities of motor boats, has been responsible for the extension of the lake. Most of the added organic material in the sediment is coarse plant debris from near-by marshes.

Algal Silt

The third type of black silt is present in the deeper water areas of all the lakes except Nippersink and Grass (figure 2). It is generally found in water deeper than 2 meters (6.6 feet), but may accumulate in shallow water if located sufficiently far from shore, as

Table 2. Geochemistry of the Surface Sediments in the Fox Chain of Lakes Region

Core	*Inorganic carbon*	*Organic carbon*	*Total carbo*	*N*	*P_2O_5*	*S*	*SiO_2*	*Al_2O_3*	*K_2O*	*CaO*	*MgO*	*Fe_2O_3*	*TiO_2*	*MnO*	*Sand*	*Silt*	*Clay*
								(percent)									
Silt with sand and/or shells																	
1	8.20	4.12	12.32	.46	.22	.69	18.16	2.84	.60	36.36	3.00	1.99	.19	.047	48.1	51.7	0.2
5	10.00	2.30	12.30	.19	.11	.66	6.79	1.20	.21	46.80	0.26	1.14	.04	.028			
8	7.46	7.01	14.47	.54	.20	.64	15.08	2.41	.44	33.10	2.25	1.51	.13	.020			
13	6.76	3.00	9.76	.44	.29	.30	32.46	3.90	.95	25.86	4.46	1.87	.24	.045			
24B	5.02	4.39	9.41	.48	.23	.23	39.78	5.57	1.30	18.56	6.36	2.56	.34	.045	8.0	72.0	20.0
44	6.45	7.95	14.40	.74	.15	.45	22.89	4.06	.91	25.66	3.78	2.26	.03	.048			
61	4.68	1.78	6.46	.23	.08	.50	57.15	5.66	1.58	15.15	6.59	1.39	.20	.015			
66	4.59	6.33	10.92	.72	.30	.44	36.82	7.18	1.28	21.27	3.45	3.14	.35	.043			
80	4.81	7.60	12.41	.75	.31	.53	32.46	5.93	1.19	19.69	4.16	3.01	.35	.053	5.7	71.9	22.4
85	7.13	4.71	11.84	.52	.18	.77	22.38	4.90	.79	35.75	2.53	2.16	.15	.035	2.4	81.6	16.0
88	4.46	7.49	11.95	.80	.25	.39	34.06	5.95	1.25	20.42	3.60	2.87	.32	.048	3.5	81.4	15.1
89	4.51	9.16	13.67	.70	.26	.50	32.52	6.55	1.20	21.29	3.77	2.91	.29	.049	15.0	69.7	15.3
Average	6.17	5.49	11.64	.55	.22	.46	29.20	4.68	.98	26.66	3.68	2.23	.22	.040	13.8	71.4	14.8
Peaty silt																	
20	3.43	16.87	20.30	1.66	.23	.96	37.62	5.28	1.23	18.36	6.21	2.36	.36	.037			
28	3.01	14.10	17.11	1.35	.13	1.02	30.61	5.97	1.35	14.52	3.24	3.03	.33	.053			
47	3.21	17.40	20.61	1.50	.31	1.01	24.33	4.88	1.04	14.03	3.11	2.74	.05	.045			
59	0.43	11.60	12.03	.70	.17	.65	28.28	4.71	1.11	24.54	4.59	2.29	.25	.023			
60	0.32	11.55	11.87	.69	.16	.56	30.35	4.55	1.14	19.55	4.90	2.12	.23	.035			
Average	2.08	14.30	16.38	1.18	.20	.84	30.24	5.08	1.17	18.20	4.41	2.51	.24	.039			
Algal silt																	
29B	4.66	7.57	12.23	.60	.22	.60	33.22	6.33	1.52	18.25	4.55	2.39	.36	.046	1.2	73.5	25.3
30	5.08	7.27	12.35	.75	.00	.64	31.32	5.79	1.49	19.47	4.54	2.48	.33	.046			
37	4.72	5.76	10.48	.60	.23	.57	36.44	7.13	1.75	17.66	5.02	2.75	.40	.044	1.5	71.7	26.8
39	3.50	7.15	10.65	.85	.38	.48	40.25	8.11	1.72	13.77	3.30	3.12	.07	.049			
49B	3.83	7.95	11.78	.33	.42	.46	36.39	7.07	1.51	15.69	3.84	3.04	.07	.060	1.4	94.2	4.4
50	4.40	6.76	11.16	.88	.25	.56	34.19	6.49	1.42	17.58	3.81	2.92	.37	.047			

Core	*Cl*	*V*	*B*	*Cu*	*Co*	*Ni*	*Be*	*Mo*	*Ge*	*Zr*	*Pb*	*Zn*	*Cd*
						(parts per million)							
Silt with sand and/or shells													
1	223	<30	10	18.9	3.7	40.7	<1.2	5.2	<5.7	62	60	75.4	< .5
5	177	<30	4	10.7	<3.8	31.1	<1.2	5.1	<5.7	22	51	28.4	< .5
8	266	<30	13	16.4	<3.9	19.9	<1.2	<3.3	<5.7	18	61	51.1	< .5
13	224	<30	9	12.6	<4.1	26.2	<1.2	<3.3	<5.7	47	35	40.3	< .5
24B	210	47	18	18.2	6.6	30.9	<1	2.2	<3	158	20	67.8	< .3
44	153	30	26	13.8	13.0	23.4	<1.2	<3.7	<5.7	83	45	84.3	<1.1
61	171	13	13	5.1	<3.4	<9.8	<1	<1.3	<3	40	<10	23.8	<1.1
66	105	38	27	27.5	<3.4	43.5	<1	<1.3	<3	132	26	90.7	<1.1
80	198	39	22	27.9	<3.4	29.0	<1	<1.3	<3	121	32	94.8	<1.1
85	161	<30	9	15.0	<3.5	16.7	<1.2	5.6	<5.7	39	29	80.0	< .6
88	235	34	16	24.6	3.6	29.3	<1	<1.3	<3	102	28	107	< .6
89	147	33	20	23.3	<4.1	31.8	<1	<1.3	<3	111	36	103	< .6
Average	189	<32	16	17.8	<4.7	27.7	<1.1	<2.9	<4.4	78	36	70.6	< .7
Peaty silt													
20	243	46	15	24.1	<3.6	31.8	<1.2	2.5	< 3	85	36	85.7	< .3
28	259	44	36	21.4	3.5	21.2	<1	1.9	< 3	72	53	81.3	< .3
47	227	<33	23	24.0	4.1	28.2	<1	1.4	< 3	64	37	108	< .6
59	241	<30	16	16.9	<3.3	17.1	<1.2	4.2	< 5.7	64	36	74.2	< 1.1
60	187	21	17	17.4	<3.4	17.6	<1	< 1.3	< 3	79	37	69.4	< 1.1
Average	231	<35	21	20.8	<3.6	23.2	<1.1	2.3	< 3	73	40	83.7	< .7
Algal silt													
29B	178	53	34	20.0	4.4	13.9	<1.0	2.6	<3	139	71	93.7	< .3
30	261	59	28	18.5	<3.4	15.3	1.0	2.5	<3	130	88	127	< .5
37	185	69	35	23.0	5.2	15.9	1.3	3.6	<3	175	90	133	< .5
39	237	84	37	25.2	4.3	21.7	1.5	3.3	<3	100	103	149	< 1.1
49B	176	45	23	24.7	<3.6	24.2	<1.0	1.9	<3	87	52	117	< .6
50	194	66	29	23.2	5.0	22.3	2.5	2.3	<3	121	51	125	< .6

(Continued on next page)

Table 2. (Continued)

Core	Inorganic carbon	Organic carbon	Total carbon	N	P_2O_5	S	SiO_2	Al_2O_3	K_2O	CaO	MgO	Fe_2O_3	TiO_2	MnO	Sand	Silt	Clay
								(percent)									
Algal silt (continued)																	
51	5.82	5.96	11.78	.77	.22	.82	28.09	5.16	1.16	22.53	4.22	2.61	.27	.046			
55	3.42	12.44	15.86	.92	.31	.62	37.74	8.11	1.82	13.89	3.41	2.94	.42	.050			
56	5.15	7.80	12.95	.85	.29	.69	30.08	5.83	1.20	21.86	3.85	2.83	.30	.044			
58	6.09	6.45	12.54	.81	.20	.70	27.97	4.98	1.15	24.62	4.50	2.54	.28	.040			
82	4.96	7.74	12.70	.69	.27	.63	30.75	6.31	1.27	20.70	4.01	2.86	.33	.047	0.9	67.3	31.8
84	5.40	6.65	12.05	.85	.28	.62	28.53	6.19	1.30	26.12	3.22	2.65	.27	.040	2.9	66.7	30.4
98B	5.41	6.43	11.84	.76	.45	.45	32.45	5.58	1.02	27.67	2.38	2.30	.22	.037	0.6	71.5	27.9
Average	4.80	7.38	12.18	.74	.29	.60	32.88	6.39	1.41	19.96	3.90	2.73	.28	.046	1.4	74.2	24.4
Muck																	
14	3.81	9.83	13.64	.70	.42	.40	36.54	6.26	1.35	14.23	4.79	3.68	.41	.054			
18B	3.30	14.13	17.43	1.38	.25	.69	29.78	6.55	1.20	14.35	3.37	3.06	.37	.050	3.9	66.1	30.0
27	1.92	11.97	13.89	1.19	.24	.75	44.24	7.75	1.83	6.99	4.27	5.01	.45	.070			
35	1.30	15.91	17.21	1.60	.30	1.24	39.62	8.05	1.71	6.30	3.23	4.27	.48	.062			
38	2.10	9.20	11.30	.76	.71	.24	44.21	8.87	2.10	6.40	4.80	4.23	.11	.065			
48	3.54	13.14	16.68	1.33	.27	.61	30.65	6.00	1.19	13.82	3.96	2.87	.06	.048			
53	3.04	14.60	17.64	1.25	.09	.83	29.76	5.03	1.23	10.00	4.34	2.59	.30	.043			
62	0.90	26.53	27.43	2.54	.31	1.49	23.99	5.24	1.09	7.47	1.98	2.95	.35	.040			
67	3.19	7.35	10.54	.68	.48	.30	43.08	7.94	1.39	13.20	3.43	3.92	.49	.058			
Average	2.57	13.60	16.20	1.27	.34	.73	35.76	6.85	1.45	10.31	3.79	3.62	.34	.054	3.9	66.1	30.0
Marl																	
32	10.61	1.46	12.07	.15	.04	.28	5.68	0.46	0.16	49.74	1.63	0.29	.01	.007			
33	4.60	1.48	6.08	.14	.004	.42	46.09	3.40	1.07	22.68	3.77	1.10	.11	.008			
52	9.33	1.25	10.58	.03	.02	.28	14.71	1.33	0.36	42.64	2.69	0.48	.05	.007			
100	9.88	1.86	11.74	.22	.06	.62	10.14	1.34	0.16	52.27	1.51	0.55	.01	.012	34.1	54.3	11.6
101	8.05	0.65	8.70	.09	.36	.44	36.12	5.57	0.96	27.37	2.36	2.28	.22	.042	41.6	48.4	10.0
Average	8.49	1.34	9.83	.13	.10	.41	22.55	2.42	0.54	38.94	2.39	0.94	.08	.015	37.9	51.3	10.8

Core	Cl	V	B	Cu	Co	Ni	Be	Mo	Ge	Zr	Pb	Zn	Cd
						(parts per million)							
Algal silt (continued)													
51	175	65	20	18.0	5.6	19.8	<2.3	2.3	<3	112	46	79.5	< .7
55	197	90	38	28.6	6.0	24.7	1.5	3.2	<3	142	69	127	< .7
56	223	34	20	23.0	<3.4	24.1	<1	<1.3	<3	94	49	91.3	<1.1
58	247	34	21	19.8	5.6	19.2	<1.2	5.4	<5.7	74	44	79.4	<1.1
82	189	40	32	31.9	3.4	26.9	<1	<1.3	<3	97	48	104	<1.1
84	192	34	29	23.3	<3.2	24.6	<1.2	5.0	<5.7	60	37	97.6	< .7
98B	149	<30	16	21.0	<3.0	21.7	<1.2	<4.3	<5.7	40	48	94.4	< .6
Average	200	54	28	23.1	<4.3	21.1	<1.4	<3.0	<3	105	61	109	< .7
Muck													
14	433	64	21	84.7	4.7	29.0	1.3	2.0	<3	100	442	268	< .5
18B	188	55	25	27.7	<3.5	30.6	1.2	<1.4	<3	93	59	110	< .5
27	182	44	36	98.8	4.8	16.0	<1	2.0	<3	89	60	126	< .3
35	229	62	37	33.3	5.7	21.6	<1.2	3.5	<3	97	67	162	< .5
38	182	98	39	74.1	11.0	18.3	2.0	4.8	<3	209	282	267	<1.1
48	183	34	20	21.9	<3.5	29.1	<1	<1.3	<3	86	32	95.4	< .6
53	280	50	22	24.1	<3.4	20.4	<1	1.5	<3	88	58	95.8	< .7
62	170	30	37	21.4	<3.5	20.0	<1	<1.3	<3	37	41	101	<1.1
67	176	47	22	38.2	<3.4	90.4	<1.0	<1.3	<3	120	19	111	<1.1
Average	225	54	29	47.1	<4.8	30.6	<1.2	<2.1	<3	102	117	148	< .7
Marl													
32	238	<30	2	1.9	15.0	<4.9	<1.2	<3.3	<5.7	17	34	74.0	< .5
33	288	31	6	3.6	<3.4	<4.9	<1	<1.3	<3	35	28	40.4	< .5
52	162	<30	3	3.2	7.7	3.6	<1.2	<3.3	<5.7	<11	9	16.8	< .7
100	209	<31	10	3.4	<3.0	<6.8	<1.2	<3.3	<5.7	<11	7	40.4	< .6
101	195	<30	10	3.3	<3.0	<6.8	<1.2	<3.3	<5.7	<11	<6	32.2	< .6
Average	218	<30	6	3.1	<6.4	<5.4	<1.2	<2.9	<5.2	<17	17	40.8	< .6

(Concluded on next page)

Table 2. (Concluded)

Core	Inorganic carbon	Organic	Total carbon	N	P_2O_5	S	SiO_2	Al_2O_3	K_2O	CaO	MgO	Fe_2O_3	TiO_2	MnO	Sand	Silt	Clay
								(percent)									
River sand																	
17	3.20	2.63	5.83	.41	.22	.12	62.16	5.90	1.47	10.13	4.91	2.34	.35	.035			
40	3.14	8.26	11.40	.82	.13	1.87	44.79	4.83	1.27	11.23	4.82	3.64	.05	.052			
43	3.50	1.87	5.01	.22	.11	.11	60.82	5.41	1.43	9.47	5.87	1.56	.04	.034			
72	3.94	3.45	7.39	.30	.19	.40	52.27	5.57	1.36	17.55	3.19	2.24	.24	.034			
Average	3.45	4.05	7.50	.44	.16	.63	55.01	5.43	1.38	12.10	4.70	2.45	.17	.039			
Peat																	
45	3.46	20.22	26.38	1.90	.20	.67	15.94	3.10	0.71	14.40	2.61	2.07	.03	.028			
63	3.72	15.14	18.86	1.86	.34	1.40	24.66	4.43	0.82	22.16	1.53	3.46	.20	.049			
Average	3.59	17.68	21.27	1.88	.27	1.04	20.30	3.77	0.77	18.28	2.07	2.77	.12	.039			

Core	Cl	V	B	Cu	Co	Ni	Be	Mo	Ge	Zr	Pb	Zn	Cd
River sand						(parts per million)							
17	209	37	14	21.0	3.6	46.5	<1	<1.3	<3	141	28	52.5	<.5
40	152	37	19	10.2	<3.4	12.9	<1	5.7	<3	92	33	83.3	<1.1
43	177	17	17	6.3	<3.4	<7.4	<1	<1.3	<3	99	33	60.6	<1.1
72	104	24	11	14.8	<3.4	20.3	<1	<1.3	<3	43	28	84.3	<1.1
Average	161	29	15	13.1	<3.5	21.8	<1	<2.4	<3	94	31	70.2	<1
Peat													
45	330	34	20	18.0	<3.4	21.6	<1	<1.3	<3	44	37	76.2	< .6
63	116	32	24	15.6	3.6	11.1	<1	<1.3	<3	42	27	97.6	<1.1
Average	223	33	22	16.8	<3.5	16.4	<1	<1.3	<3	43	32	86.9	< .9

Table 3. Vertical Changes in Geochemistry of Cores of Bottom Sediments in Fox Chain of Lakes

Core depth (cm)	Inorganic carbon	Organic carbon	Total	N	P_2O_5	S	SiO_2	Al_2O_3	K_2O	CaO (percent)	MgO	Fe_2O_3	TiO_2	MnO	Sand	Silt	Clay
Core 1 - Pistakee Lake																	
0-5	8.20	4.12	12.32	.46	.22	.69	18.16	2.84	0.60	36.36	3.00	1.99	.19	.047	48.1	51.7	0.2
20-25	11.21	1.79	13.00	.14	.11	.23	3.17	0.11	0.02	49.15	2.48	0.32	.01	.010	63.2	27.8	9.0
57-63	11.48	1.07	12.55	.22	.01	.27	1.46	0.01	0.01	49.95	2.47	0.42	.01	.010	45.3	54.7	0.0
Core 18B - Dredged Channel																	
0-6	3.30	14.13	17.43	1.38	.25	.69	29.78	6.55	1.20	14.35	3.37	3.06	.37	.050	3.9	66.1	30.0
25-30	3.43	13.15	16.58	.42	.41	.77	32.89	6.39	1.22	15.41	3.09	2.78	.39	.040	3.7	68.3	28.0
50-57	1.87	20.83	22.70	2.14	.29	1.07	27.11	5.81	1.08	9.98	2.61	2.82	.37	.042	14.6	59.4	26.0
70-75	7.52	3.55	11.07	.45	.07	.62	24.18	2.60	.62	33.42	2.37	1.38	.16	.015	8.2	72.8	19.0
90-94	6.66	4.24	10.90	.40	.09	.79	26.44	4.36	1.00	28.44	3.20	2.24	.24	.030	10.0	66.0	24.0
Core20 - Grass Lake																	
0-6	3.43	16.87	20.30	1.66	.23	.96	37.62	5.28	1.23	18.36	6.21	2.36	.36	.037			
6-12	3.42	16.32	19.14	1.56	.33	.98	25.26	5.20	1.06	15.78	3.12	2.94	.32	.040			
25-30	2.44	19.93	22.37	1.85	.27	1.37	22.32	4.40	0.83	11.63	2.45	2.68	.25	.041			
50-55	2.52	24.47	26.99	2.15	.23	.68	21.23	3.47	0.70	13.28	2.37	2.07	.21	.030			
71-75	9.77	4.68	14.45	.41	.01	.55	6.77	0.99	0.18	44.55	2.64	0.83	.02	.014			
Core 24B - Grass Lake																	
0-7	5.02	4.39	9.41	.48	.23	.23	39.78	5.57	1.30	18.56	6.36	2.56	.34	.045	8.0	72.0	20.0
7-14	5.22	4.33	9.55	.43	.19	.27	37.22	5.30	1.23	18.31	6.16	2.45	.36	.039	10.0	74.0	16.0
14-21	5.56	4.17	9.73	.31	.18	.30	36.22	4.78	1.20	20.52	6.27	2.43	.30	.036	12.7	70.3	17.0
30-35	5.44	7.86	13.30	.39	.18	.25	28.20	5.10	1.15	22.92	4.47	2.41	.31	.044	8.6	69.6	22.0
54-59	5.50	8.05	13.55	.66	.]2	.27	26.89	5.19	1.16	22.89	4.10	2.52	.26	.045	13.6	69.4	17.0

Core depth (cm)	Cl	V	B	Cu	Co	Ni	Be	Mo (parts per million)	Ge	Zr	Pb	Zn	Cd	Hg
Core 1 - Pistakee Lake														
0-5	223	<30	10	18.9	3.7	40.7	<1.2	5.2	<5.7	62	60	75.4	<.5	
20-25	295	<30	0.3	3.1	<3.7	< 3.3	<1.2	<3.3	<5.7	<11	30	1.9	<.5	
57-63	175	<30	2	3.3	<4.1	< 3.3	<1.2	<3.3	<5.7	<11	27	17.5	<.5	
Core 18B - Dredged Channel														
0-6	188	55	25	27.7	<3.5	30.6	1.2	<1.4	<3	93	59	110	<.5	
25-30	286	56	25	27.8	4.6	30.8	1.0	<1.3	<3	86	40	89.3	<.5	
50-57	289	46	29	22.4	<3.8	26.3	<1	1.3	<3	46	51	82.9	<.5	
70-75	150	<30	15	7.5	12.0	< 4.1	<1.2	4.8	<5.7	62	<4	16.1	<.3	
90-94	131	<30	17	11.5	14.0	8.6	<1.2	8.1	<5.7	70	<4	59.0	<.3	
Core 20 - Grass Lake														
0-6	243	46	15	24.1	<3.6	31.8	1.2	2.5	3	85	36	85.7	<.3	
6-12	268	45	19	25.1	4.2	33.3	<1	<1.3	<3	66	36	86.0	<.3	
25-30	278	33	17	20.3	<3.4	13.1	<1	1.9	<3	46	773	70.0	<.3	
50-55	258	29	13	13.9	4.1	8.1	<1	<1.3	<3	35	<4	29.5	<.3	
71-75	221	<30	10	4.8	15.0	4.3	<1.2	<3.3	<5.7	< 11	<4	5.7	<.3	
Core 24B - Grass Lake														
0-7	210	47	18	18.2	6.6	30.9	<1	2.2	<3	158	20	67.8	<.3	
7-14	302	48	21	18.0	<3.7	30.6	<1	<1.3	<3	149	20	63.3	<.3	
14-21	221	47	19	14.6	6.1	24.3	<1	2.2	<3	142	18	62.3	<.3	
30-35	168	65	24	10.9	4.5	8.8	<1	2.2	<3	128	<3	46.2	<.3	
54-59	140	54	22	12.6	<4.1	9.6	<1	2.2	<3	113	<3	90.8	<.3	

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Table 3. (Continued)

Core depth (cm)	Inorganic carbon	Organic carbon	Total	N	P_2O_5	S	SiO_2	Al_2O_3	K_2O	CaO	MgO	Fe_2O_3	TiO_2	MnO	Sand	Silt	Clay
										(percent)							
Core 29B - Channel Lake																	
0-5	4.66	7.57	12.23	.60	.22	.60	33.22	6.33	1.52	18.25	4.55	2.39	.36	.046	1.2	73.5	25.3
5-10	4.81	7.66	12.47	.74	.22	.58	34.52	6.46	1.63	18.20	4.50	2.46	.38	.035	1.1	64.6	34.3
10-15	4.53	5.63	10.16	.68	.22	.57	37.36	7.04	1.72	16.56	4.41	2.57	.37	.047	1.2	68.3	30.5
20-25	3.98	6.36	10.34	.70	.24	.58	38.59	7.37	1.77	15.56	4.43	2.92	.40	.052	1.4	65.0	33.6
40-45	2.72	8.86	11.58	1.10	.20	.29	39.87	8.44	2.24	8.42	5.01	5.37	.50	.066	6.0	59.9	34.1
65-70	3.10	7.43	10.53	.61	.25	.24	40.33	7.84	2.18	11.32	4.80	4.35	.46	.064	3.5	68.7	27.8
Core 37 - Lake Catherine																	
0-6	4.72	5.76	10.48	.60	.23	.57	36.44	7.13	1.75	17.66	5.02	2.75	.40	.044	1.5	71.7	26.8
6-12	4.65	4.73	9.38	2.40	.13	.54	37.45	7.56	1.86	17.34	5.83	1.86	.41	.045	1.1	71.5	27.4
12-18	4.68	1.86	6.54	.43	.16	.21	42.02	9.09	2.39	13.96	7.06	3.21	.55	.045	0	63.2	36.8
18-25	4.26	3.97	8.23	.45	.18	.48	40.32	8.68	2.05	14.63	4.98	3.17	.09	.049	1.3	67.8	30.9
40-45	3.92	5.85	9.77	.59	.23	.40	39.28	8.24	1.94	14.57	4.48	3.17	.08	.055	2.7	70.5	26.8
60-65	3.40	6.75	9.97	.71	.20	.26	42.00	8.58	2.14	11.62	5.01	4.34	.09	.069	1.0	64.6	34.4
Core 43 - Fox River																	
0-8	3.50	1.87	5.01	.22	.11	.11	60.82	5.41	1.43	9.47	5.87	1.56	.04	.034			
15-20	3.84	3.00	6.84	.25	.17	.14	50.23	5.77	1.44	10.96	6.33	2.18	.06	.034			
25-30	3.99	4.05	8.04	.40	.29	.16	47.38	6.24	1.44	11.86	6.49	2.52	.07	.035			
35-40	4.27	3.49	7.76	.28	.18	.15	45.88	6.02	1.44	12.73	6.52	2.25	.07	.037			
Core 49B - Lake Marie																	
0-5	3.83	7.95	11.78	.33	.42	.46	36.39	7.07	1.51	15.69	3.84	3.04	.07	.060	1.4	94.2	4.4
5-9	3.95	7.76	11.71	.87	.43	.48	36.93	7.33	1.55	15.98	3.76	2.96	.07	.048	0.8	94.6	4.6
9-13	4.21	7.72	11.93	.95	.36	.52	34.94	6.76	1.47	16.32	3.44	2.84	.07	.049	1.7	93.9	4.4
20-25	4.27	7.35	11.62	.88	.30	.52	34.39	6.04	1.34	17.10	3.56	2.67	.06	.047	0.7	56.7	42.6
50-55	4.36	5.49	9.85	.69	.18	.56	38.30	7.53	1.75	16.33	4.21	2.92	.39	.051	0.4	64.0	35.6
79-84	3.79	7.90	11.69	.80	.18	.39	36.52	7.53	1.94	14.11	4.24	3.96	.43	.068	1.6	64.8	33.6

Core depth (cm)	Cl	V	B	Cu	Co	Ni	Be	Mo	Ge	Zr	Pb	Zn	Cd	Hg
								(parts per million)						
Core 29B - Channel Lake														
0-5	178	53	34	20.0	4.4	13.9	<1.0	2.6	<3	139	71	93.7	< .3	
5-10	315	63	30	19.6	4.6	14.1	1.2	3.5	<3	138	71	92.8	< .3	
10-15	280	55	32	19.1	4.4	13.9	1.3	3.3	<3	136	57	95.4	< .3	
20-25	175	67	34	17.0	5.4	11.3	1.4	3.6	<3	162	33	82.5	< .3	
40-45	202	67	45	18.5	7.6	15.2	1.4	4.3	<3	118	< 3	62.1	< .3	
65-70	295	66	43	18.5	6.0	19.2	1.5	3.9	<3	143	20	87.3	< .5	
Core 37 - Lake Catherine														
0-6	185	69	35	23.0	5.2	15.9	1.3	3.6	<3	175	90	133	< .5	
6-12	286	70	37	21.4	6.2	16.6	1.4	3.9	<3	179	76	114	< .5	
12-18	136	82	39	20.7	7.0	20.1	1.6	4.1	<3	233	42	79.8	< .5	
18-25	162	82	43	21.2	6.7	16.1	1.5	2.6	<3	197	74	117	< .5	
40-45	153	104	36	17.6	9.5	17.6	1.6	4.7	<3	169	39	87.3	<1.1	
60-65	236	94	42	16.0	7.4	19.0	1.7	5.2	<3	169	24	73.5	<1.1	
Core 43 - Fox River														
0-8	177	17	17	6.3	<3.4	<7.4	<1	<1.3	<3	99	33	60.6	<1.1	
15-20	209	47	23	14.2	<3.6	14.9	<1	<1.3	<3	130	39	85.7	<1.1	
25-30	235	42	20	18.0	4.8	26.8	<1	2.4	<3	173	43	101	<1.1	
35-40	191	36	18	12.8	4.0	17.9	<1	1.5	<3	121	40	79.5	<1.1	
Core 49B - Lake Marie														
0-5	176	45	23	24.7	<3.6	24.2	<1	1.9	<3	87	52	117	< .6	
5-9	186	39	21	24.6	4.2	24.5	<1	2.2	<3	104	55	120	< .6	
9-13	201	55	26	25.9	<3.4	25.4	<1	2.0	<3	104	114	164	< .6	
20-25	257	60	25	22.8	3.5	26.8	<1	2.0	<3	94	56	110	< .8	
50-55	204	90	29	20.5	6.0	18.5	<1.9	4.4	<3	163	45	112	< .6	
79-84	207	86	35	20.4	7.4	15.3	2.9	4.1	<3	156	7	82.4	< .6	

(Continued on next page)

Table 3. (Continued)

Core depth (cm)	Inorganic carbon	Organic carbon	Total	N	P_2O_5	S	SiO_2	Al_2O_3	K_2O	CaO	MgO	Fe_2O_3	TiO_2	MnO	Sand	Silt	Clay
Core 55 - Petite Lake										(percent)							
0-2	3.42	12.44	15.86	.92	.31	.62	37.74	8.11	1.82	13.89	3.41	2.94	.42	.050			
2-6	3.40	9.31	12.71	.96	.31	.70	38.29	7.77	1.75	14.46	3.59	3.20	.46	.050			
6-10	3.39	8.95	12.34	.65	.36	.63	35.63	7.75	1.72	14.14	3.41	3.20	.52	.046			
15-20	4.14	8.05	12.19	.92	.32	.64	35.09	6.90	1.59	16.94	3.45	2.64	.34	.045			
35-40	3.42	7.14	10.56	1.99	.19	.70	41.66	8.76	1.98	14.09	3.04	3.44	.46	.065			
80-85	2.75	15.57	18.32	1.68	.22	1.07	33.43	4.28	0.72	13.90	3.04	2.58	.24	.029			
Core 56 - Fox Lake																	
0-4	5.15	7.80	12.95	.85	.29	.69	30.08	5.83	1.20	21.86	3.85	2.83	.30	.044			
4-8	5.10	8.23	13.33	.89	.27	.71	29.45	5.68	1.17	21.41	4.06	2.92	.30	.049			
8-13	5.08	7.83	12.91	.83	.26	.72	31.00	6.00	1.26	23.09	4.13	3.21	.31	.059			
15-20	5.28	6.86	12.14	.95	.16	.70	29.24	5.89	1.27	21.93	4.29	2.80	.31	.048			
35-40	4.93	7.21	12.14	.93	.13	.72	32.73	6.38	1.42	19.93	3.67	3.12	.33	.054			
55-60	5.92	7.77	13.69	1.00	.12	.59	26.16	4.88	1.05	25.71	3.50	2.34	.23	.043			
Core 80 - Nippersink Lake																	
0-5	4.81	7.60	12.41	.75	.31	.53	32.46	5.93	1.19	19.69	4.16	3.01	.35	.053	5.7	71.9	22.4
10-15	4.67	8.19	12.86	.45	.28	.52	33.66	5.85	1.24	19.52	4.16	2.91	.35	.041	7.6	70.0	22.4
20-25	5.94	7.10	13.04	.66	.32	.71	25.89	4.96	1.00	25.76	3.51	2.71	.23	.046	15.2	59.8	25.0
40-45	4.49	16.42	20.91	.95	.12	.57	22.15	3.79	0.76	20.63	2.57	1.96	.18	.028	7.0	66.8	26.2
Core 88 - Channel between Fox and Nippersink Lakes																	
0-5	4.46	7.49	11.95	.80	.25	.39	34.06	5.95	1.25	20.42	3.60	2.87	.32	.048	3.5	81.4	15.1
45-50	4.04	9.43	13.47	.13	.34	.49	34.84	6.39	1.35	18.61	3.33	3.28	.36	.047	4.8	77.1	18.1

Core depth (cm)	Cl	V	B	Cu	Co	Ni	Be	Mo	Ge	Zr	Pb	Zn	Cd	Hg
Core 55 - Petite Lake								(parts per million)						
0-2	197	90	38	28.6	6.0	24.7	1.5	3.2	<3	142	69	127	< .7	
2-6	218	30	35	28.2	<3.7	23.8	1.1	<1.4	<3	98	69	126	< .7	
6-10	271	41	33	27.5	3.6	24.1	1.3	<1.3	<3	104	79	121	< .7	
15-20	253	40	28	25.3	<3.4	24.1	<1	<1.3	<3	103	64	109	< .7	
35-40	239	51	40	26.5	<3.4	20.9	1.3	<1.3	<3	116	62	114	< .7	
80-85	303	18	31	16.1	<3.4	10.2	<1	<1.3	<3	24	< 7	464	< .7	
Core 56 - Fox Lake														
0-4	223	34	20	23.0	<3.4	24.1	<1	<1.3	<3	94	49	91.3	<1.1	.22
4-8	191	36	24	23.0	<3.4	27.3	<1	<1.3	<3	94	43	90.2	<1.1	.22
8-13	214	35	18	21.8	<3.4	25.5	<1	<1.3	<3	96	42	89.3	<1.1	.19
15-20	197	38	22	20.1	<3.4	23.5	<1	<1.3	<3	95	30	87.7	<1.1	.20
35-40	208	39	33	19.3	<3.4	15.2	<1	<1.3	<3	94	29	82.9	<1.1	.26
55-60	201	<35	22	14.3	<4.1	11.4	<1.2	<4.2	<5.7	45	15	51.6	<1.1	.15
Core 80 - Nippersink Lake														
0-5	198	39	22	27.9	<3.4	29.0	<1	<1.3	<3	121	32	94.8	<1.1	
10-15	192	35	19	24.5	<3.8	30.9	<1	<1.3	<3	129	33	97.4	<1.1	
20-25	240	<32	19	18.5	<3.9	24.0	<1.2	5.9	<5.7	61	32	77.0	<1.1	
40-45	231	25	13	14.3	4.4	12.4	<1	<1.3	<3	32	<12	42.4	<1.1	
Core 88 - Channel between Fox and Nippersink Lakes														
0-5	35	34	16	24.6	3.6	29.3	<1	<1.3	<3	102	28	107	< .6	
45-50	154	33	20	25.7	3.5	32.4	<1	<1.3	<3	101	37	111	< .6	

(Concluded on next page)

Table 3. (Continued)

Core depth (cm)	*Inorganic carbon*	*Organic carbon*	*Total*	*N*	*P_2O_5*	*S*	*SiO_2*	*Al_2O_3*	*K_2O*	*CaO*	*MgO*	*Fe_2O_3*	*TiO_2*	*MnO*	*Sand*	*Silt*	*Clay*
Core 98B - Pistakee Bay										*(percent)*							
0-4	5.41	6.43	11.84	.76	.45	.45	32.45	5.58	1.02	27.67	2.38	2.30	.22	.037	0.6	71.5	27.9
4-8	5.48	6.91	12.39	.46	.47	.48	31.50	5.31	1.05	27.13	2.09	2.34	.23	.041	1.3	63.2	35.5
8-12	5.53	6.44	11.97	.72	.47	.48	31.13	6.30	1.02	27.50	1.92	2.33	.23	.037	2.5	59.9	37.6
35-40	5.34	5.66	11.00	.65	.04	.62	38.37	1.60	0.53	35.39	1.37	0.67	.01	.010	1.6	66.5	31.9
70-75	6.41	7.17	13.58	.79	.21	.36	26.85	2.37	0.52	31.69	1.80	1.93	.07	.033	5.1	81.1	13.8

Core depth (cm)	*Cl*	*V*	*B*	*Cu*	*Co*	*Ni*	*Be*	*Mo*	*Ge*	*Zr*	*Pb*	*Zn*	*Cd*	*Hg*
Core 98B - Pistakee Bay							*(parts per million)*							
0-4	149	<30	16	21.0	<3.0	21.7	<1.2	<4.3	<5.7	40	48	94.4	<.6	.11
4-8	182	<31	29	21.9	<3.3	24.3	<1.2	4.6	<5.7	49	47	106	<.6	.23
8-12	163	<30	28	21.4	<3.1	25.3	<1.2	4.2	<5.7	43	47	104	<.6	.22
35-40	138	<33	22	22.5	<3.0	14.9	<1.2	4.9	<5.7	36	59	113	<.6	.50
70-75	134	<30	9	7.0	4.2	<6.8	<1.2	<3.6	<5.7	25	<6	36.9	<.6	.29

it does in Fox and Pistakee Lakes. The algal silt averages 74 percent silt and contains less than 3 percent sand (table 2). The organic components constitute about 7 percent of the sediment and, except for the fine, thread-like algal filaments that are abundant, the organic debris is very finely comminuted.

Marl

A type of sediment found in small areas of some lakes is marl. Marl is a light gray sediment consisting of a mixture of calcarenite (sand-sized fragments of organically precipitated calcium carbonate), shell debris, and, commonly, minor amounts of plant debris (table 2). Marl generally occurs in a narrow band around the margin of lake basins that are more than 4 meters (13.1 feet) deep (figure 2). The marl forms in a zone with water depths of between 2 and 4 meters (6.6 and 13.1 feet) lying between the algal silts of the deep water and the near-shore silty sediments.

Marl deposition is best developed in Channel Lake, Lakes Catherine and Marie, and in Pistakee Bay, all of which have deep water basins. It may also be present in Bluff and Petite Lakes [both contain water over 4 meters (13.1 feet) deep] but no samples were taken in the marl zones in those lakes. No active marl deposition is known to be taking place in the shallower lakes - Fox, Grass, Nippersink, and Pistakee (except Pistakee Bay) - where organic sedimentation predominates. However, older marls were encountered in these shallower lakes under more recent peaty sediments, indicating that marl deposition may have been more widespread in the past.

River Sediments

Samples from the channel of the Fox River and Nippersink Creek contain coarse-grained quartz sand. The river sediments are much coarser grained than any of the lake sediments and they also contain coarse shell debris, including large bivalves. The organic content of the river sands is generally low, consisting of a small amount of coarse plant debris. The silica content, most of it in the form of quartz, is higher in this type of sediment than in the other types (table 2).

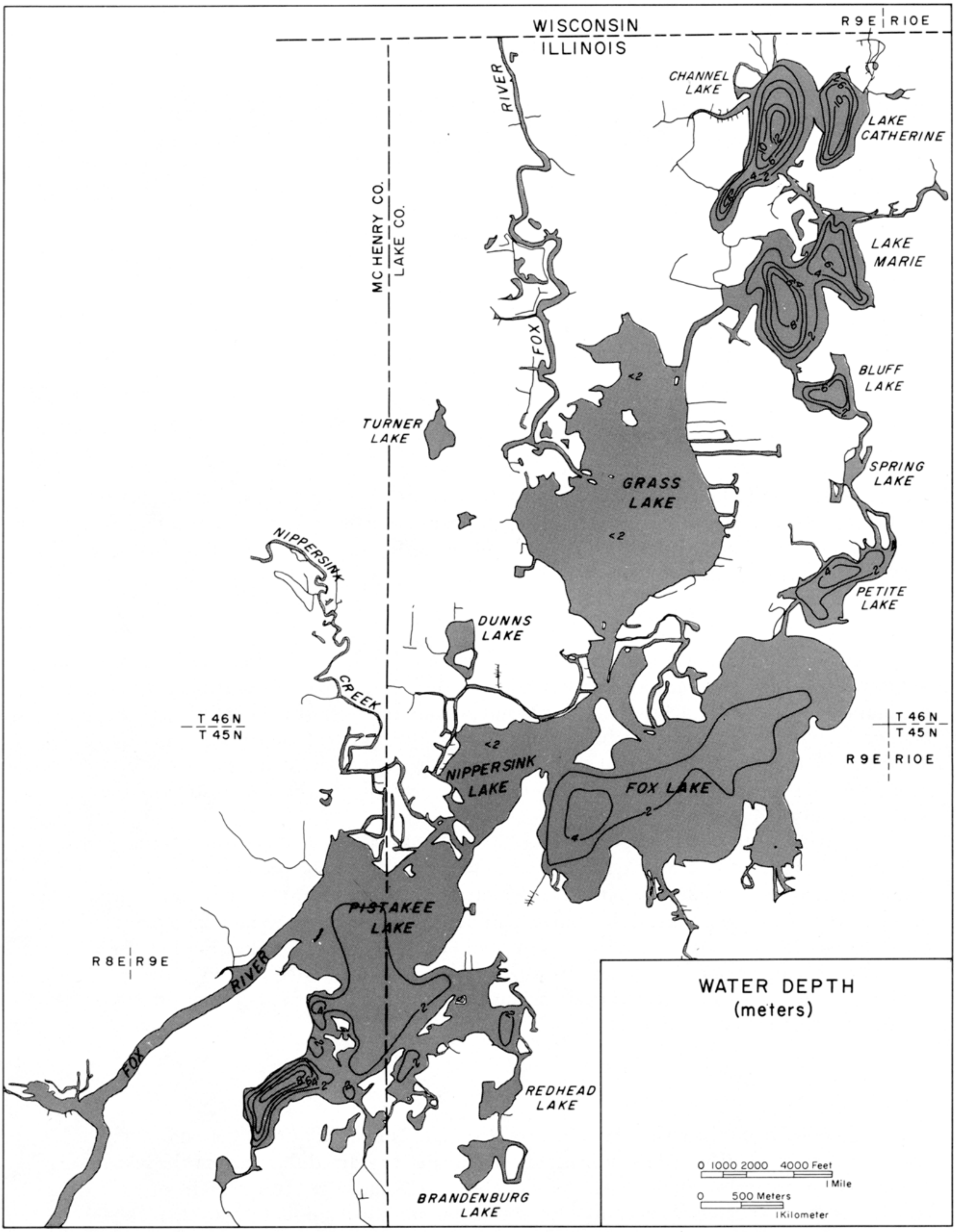

Figure 3. Water depth in the Fox Chain of Lakes

Dredged Channel Sediments

Dead-end channels, dredged to provide water access to the lake for property owners, are typically filled with a soft black muck (figure 2). The muck is mainly silt but contains up to 26 percent organic carbon (table 2). The organic components consist of algae, comminuted and decomposed leaves and marsh plant fragments from channel-side vegetation, and organic material from septic tank outfalls. The carbonate content of muck is low and shells are scarce (table 2). Because the dredging dates are known for many of the channels, the accumulation of sediment can be measured. In one channel, dredged 10 years before, 0.36 meters (1.2 feet) of sediment had accumulated.

Peat

In Grass Lake and in other places where older peat was found, peat is generally covered by a thin layer of peaty silt. However, in two cores, the older peat was exposed at the lake floor. Peat, which contains less inorganic sediment and more coarse organic fragments than the peaty silt (table 1), is now accumulating in the widespread cattail marshes that surround parts of the Chain of Lakes (figure 2).

Glacial Deposits

Gray glacial till underlies most of the lakes. As the glacial ice melted, glacial-lacustrine clay accumulated above the glacial till. These sediments probably occur deep under the younger silts and marls in the Chain of Lakes but were encountered in only a few places during coring. The deeper lakes commonly have steep underwater slopes, which may be too steep for sediments to collect. The older glacial-sediment floor of the lake is exposed in these places. Where the lake floor has been eroded, or where no deposition has occurred, the old glacial sediments may be at or near the lake floor (figure 2). None of the glacial sediments was analyzed. The older sediments, particularly the till compressed by the weight of the glacial ice, are compact and provide the best foundations for construction of large buildings, docks, or sheet pilings.

PALYNOLOGY

Ambrosia (ragweed) pollen generally constitutes less than 10 percent of the total pollen found in samples of Holocene lake sediments. However, in the most recent sediments *Ambrosia* pollen content may form up to 40 percent of the pollen preserved. The dramatic increase in *Ambrosia* pollen takes place over only a few centimeters vertically in the typical core. The sudden increase marks the first effects of large-scale clearing of forest cover for the establishment of agriculture in the Midwest. This is generally considered, on the basis of historical records, to have taken place between 1830 and 1840 in northern Illinois. When a forest is cut, the abundance of ragweed increases dramatically because it grows well on cleared and disturbed land.

The base of the *Ambrosia* zone as preserved in the sediments of the Chain of Lakes is therefore an important marker horizon, indicating the position of the sediment-water interface about 1840 (James King, personal communication, 1976). This date can be used to compute sedimentation rates in the lakes from 1840 to 1975 in three cores that were analyzed (figures 4, 5, and 6).

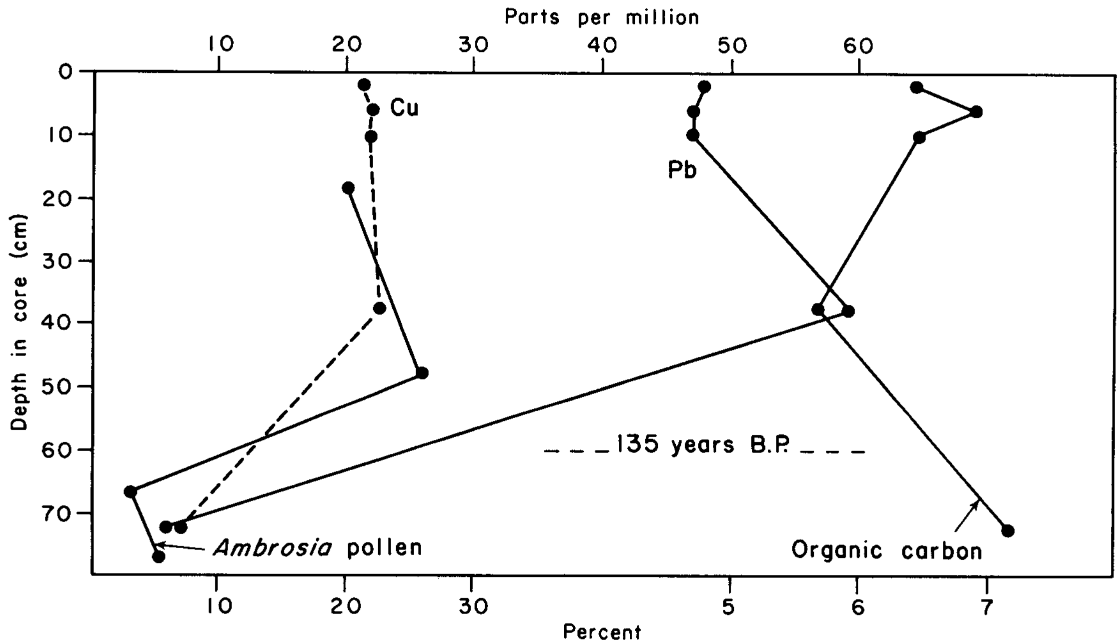

Figure 4. Depth profile for core 98B from Pistakee Bay

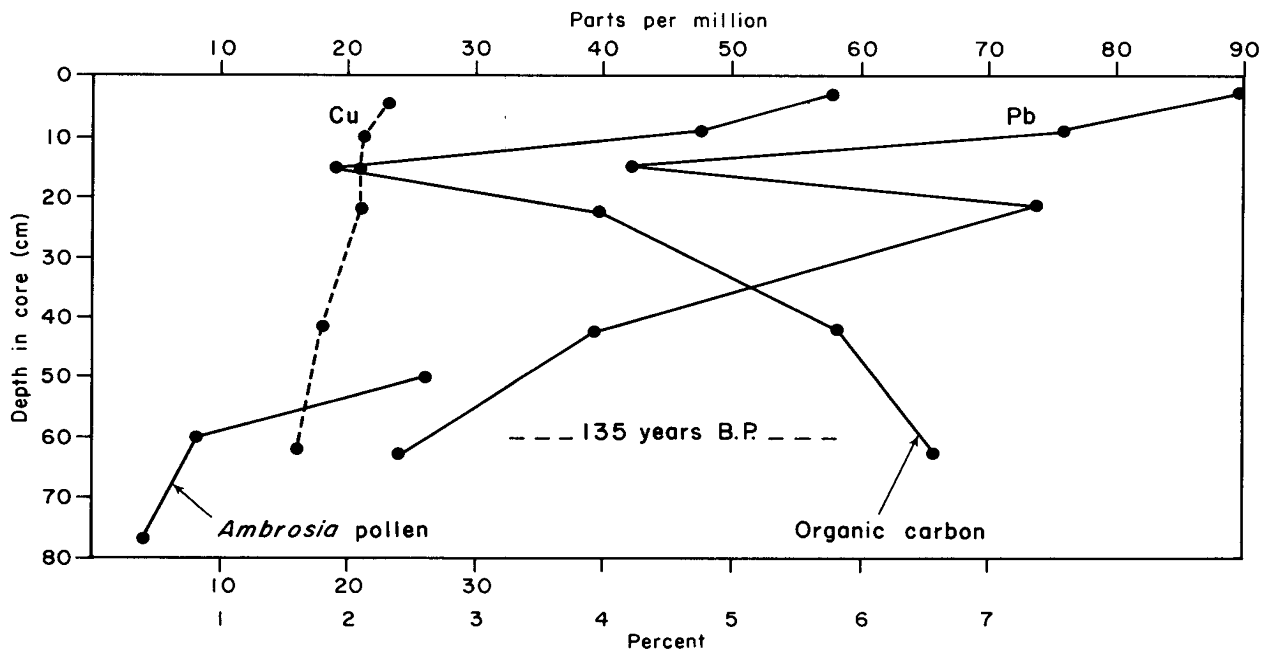

Figure 5. Depth profile for core 37 from Lake Catherine

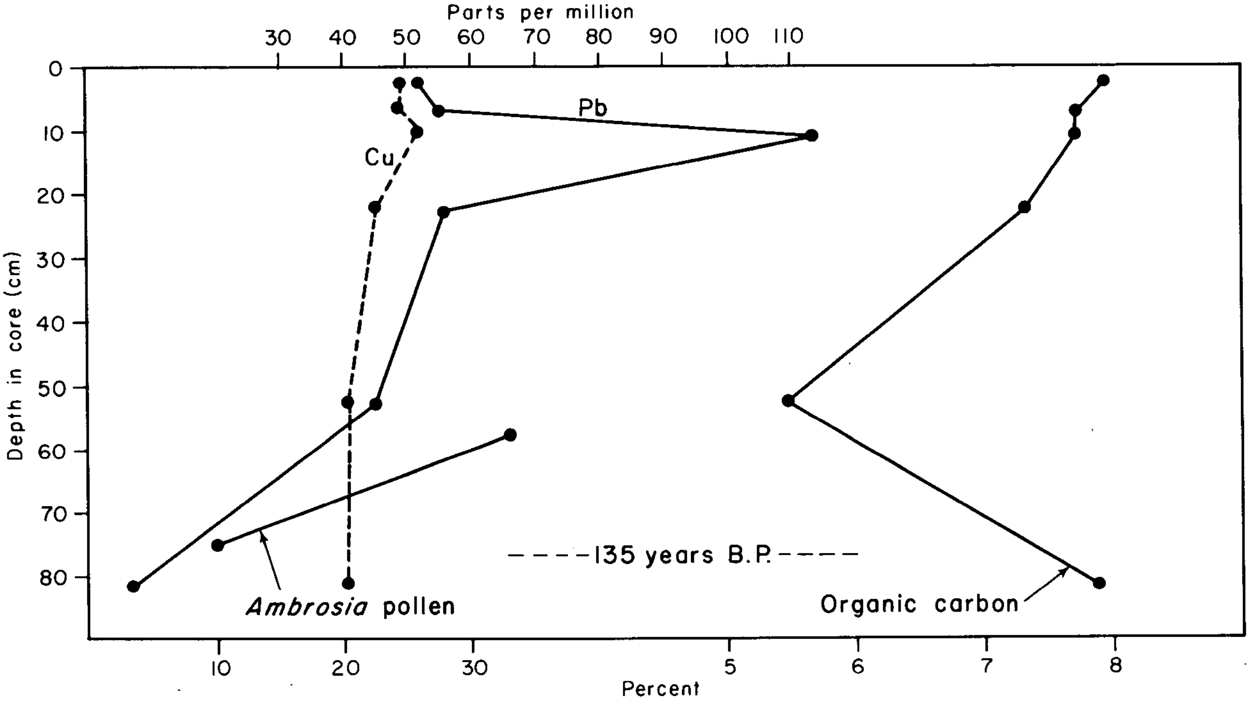

Figure 6. Depth profile for core 49B from Lake Marie

Sedimentation rates in Lake Catherine are about 0.41 centimeters per year (cm/yr) (0.16 inch), in Lake Marie about 0.48 cm/yr (0.19 inch), and in Pistakee Bay about 0.44 cm/yr (0.17 inch). These rates are several times higher than Lake Michigan rates of 0.05 to 0.10 cm/yr (0.02 to 0.04 inch/year) (King, et al., 1976).

SEDIMENT GEOCHEMISTRY

Sediment samples from several levels in 12 cores were analyzed geochemically for 27 major and minor components to determine the vertical variations in the recent sediments (table 3). The base of the *Ambrosia* (ragweed) pollen zone was determined for three cores. Samples from the surface layer at an additional 37 locations were analyzed to determine areal distribution of the components (table 2).

Geochemical studies of Lake Michigan sediments (Shimp et al., 1971; Lineback and Gross, 1972) show that some elements that occur in lake sediments in trace amounts (e.g., lead, zinc, and copper) are more abundant in the surface sediment layers than in older layers. This pattern of accumulation in recently deposited sediments has been interpreted as being the result of man-caused pollution of the lake environment. Heavy metals and other pollutants may enter a lake via the air (e.g., lead from automobile exhaust) or via water from industrial, agricultural or domestic sources. In Lake Michigan all excessive concentrations occur above the base of the *Ambrosia* pollen zone — that is, in sediment deposited since 1840 and the highest concentrations are in the most recent sediments. The depth to the base of the *Ambrosia* zone in Lake Michigan is 5 to 15 centimeters (2 to 6 inches).

Analytical Techniques

X-ray Fluorescence Spectrometry and Carbon Determination

The analytical procedure followed for the determination of major and minor elements in sediments by x-ray fluorescence spectrometry (XRF) and the methods used to determine total carbon, organic carbon, and inorganic carbon (CO_2) have been reported by Shimp et al. (1970). Since their work, the sensitivity of the XRF equipment has been increased for the light elements by using a thallium acid phthalate (TIAP) crystal in place of the previously used ammonium dihydrogen phosphate (ADP) and ethylene diamine ditartrate (EDDT) crystals.

Optical Emission Spectrography

Two synthetic standards were prepared for use in determining trace-element concentrations in these sediments by optical emission spectrography. It was felt that the sediments would be represented by two general matrix types, those containing low concentrations of calcium carbonate ($CaCO_3$) and those with high calcium carbonate concentrations. After the concentrations of major constituents in approximately two-thirds of the samples had been determined by x-ray fluorescence spectrometry, the calcium carbonate concentrations were listed in increasing order. A natural division in $CaCO_3$ concentrations occurred between 36.64 percent and 40.23 percent. The first set of synthetic standards, which had been prepared previously for the analysis of other lake bottom sediments, contains 21.48 percent $CaCO_3$. Concentrations of the major constituents are listed in table 4.

Table 4. Major Constituents in Synthetic Standard Matrices for Sediments

Constituent	*Concentrations (%)* Low $CaCO_3$	*Concentrations (%)* High $CaCO_3$
Al_2O_3	7.38	2.24
$CaCO_3$	21.48	76.44
SiO_2	54.06	16.36
Fe_2O_3	3.44	1.35
MgO	8.98	2.87
Na_2CO_3	1.18	
K_2CO_3	3.48	0.74

The composition of the matrix low in calcium carbonate was determined by knowing the mean composition of several hundred previously analyzed bottom sediment samples from Lake Michigan. The matrix with a composition high in calcium carbonate was determined from the mean composition of the Chain of Lakes bottom sediments that contain more than 40.23 percent $CaCO_3$.

To the first (low $CaCO_3$) matrix type, amounts of Spex Industries, Inc., alumina (Al_2O_3) and silica (SiO_2) Spex Time Saver Standards containing 1000, 333, 100, and 33 parts per million (ppm) of 49 different elements were added in the same ratio as the SiO_2:Al_2O_3 ratio in the matrix. The final concentrations of the trace metals were 333, 100, 33.3, 10.0, 3.3, and 1.0 ppm. There was a corresponding decrease in concentrations of major elements due to additions of the Spex standards, but it did not significantly affect the behavior of the standards in the arc nor the ability of the standards to approximate the samples. The procedure used for the high calcium carbonate matrix was similar, but $CaCO_3$ Spex Time Saver Standards were used. The weights of each standard used are listed in table 5.

Each standard mixture was ground thoroughly with a mullite mortar and pestle under absolute ethanol to increase the homogeneity of the individual standards. Studies of time vs. intensity spectra were made for both standard matrices to determine the proper exposure times. The experimental parameters chosen are listed in table 6.

The percent transmittance values of the desired spectrographic lines on the developed photographic plates were determined by standard densitometry. Intensities were determined

Table 5. Compositions of Standards

Weight of synthetic matrix (grams)	*Low $CaCO_3$ matrix* SiO_2 *Spex time saver standard (mg)*	*(ppm)*	*Al_2O_3 Spex time saver standard (mg)*	*(ppm)*	*High $CaCO_3$ matrix* $CaCO_3$ *Spex time saver standard (mg)*	*(ppm)*	*Final standard concentrations (ppm)*
0.33333	146	1000	20	1000	166	1000	333
0.45000	44	1000	6	1000	50	1000	100
0.45000	44	333	6	333	50	333	33.3
0.45000	44	100	6	100	50	100	10.0
0.45000	44	33.3	6	33.3	50	33.3	3.3
0.45000	44	10.0	6	10.0			1.0

Table 6. Experimental Spectrographic Parameters

Arc gap	4 mm
Exposure time	100 sec for low $CaCO_3$, 110 sec for high $CaCO_3$
Arc current	10 A
Sample electrode	National L-3903
Counter electrode	National SP-1009
Atmosphere and flow rate	80% argon, 20% oxygen at 14 SCFH
Photographic emulsion	Eastman-Kodak SA-1
Photographic developer	Eastman-Kodak D-19
Step sector	6 step, 2:1 step ratio
Slit width	10 µm
Internal standard	none
Electrode charge	20 mg
Sample mixture	1 part sample, 4 parts SP-2X graphite powder

via Hurter-Driffield emulsion calibration curves. However, the data reduction steps have been computer programmed, by means of a spline function routine to fit the Hurter-Driffield curves with points spaced every 2 percent transmittance. The individual element working curves (intensity vs. concentration) have been fitted by either first or second degree least-squares regressions or by combinations of the two types.

Atomic Absorption Spectrometry

The atomic absorption spectrometric techniques used were similar to those described by Ruch et al. (1974), except that 125-ml or 60-ml linear polyethylene bottles that had been washed in HNO_3 were substituted for the stainless steel, Teflon-lined pressure decomposition bombs. The sample and reagents were added to the bottle, which was sealed and then heated on a steam bath for 3 to s4 hours, being swirled occasionally. The only difference noted between duplicate coal-ash samples decomposed in polyethylene bottles and those decomposed in Teflon-lined bombs was a zinc blank of about 0.02 mg/ml.

Instrumental Neutron Activation Analysis

The samples were irradiated in the Advanced TRIGA reactor at the University of Illinois in a flux of 1.4 x 10^{12} thermal neutrons cm^{-2} sec^{-1}. The sediment samples (0.6 to 1 g) were weighed into two-fifths dram polyethylene snap-cap vials which had been previously cleaned with deionized water and acetone. The vials were heat-sealed. Multi-element standards were prepared by pipetting aliquots of standard solutions (made from a spectroscopically pure grade of each element or its compound) onto standard cellulose ash-free filter paper. The dried filters were sealed in small polyethylene bags, and these in turn were heat-sealed in clean 2-dram polyethylene vials. The vials containing the sediments were also encapsulated in 2-dram vials. During irradiation the vials were rotated at 1 rpm to ensure all samples and standards had equal neutron flux.

To determine elements that give rise to short-lived isotopes by (n, γ) reactions upon irradiation, the samples and a standard were irradiated for 10 minutes. After a 3-hour decay period, the gamma activities were counted for 2000 to 3000 seconds. The following elements were measured via their isotopes (the γ-ray lines used are in parentheses): ^{165}Dy (94.7, 361.5 keV), ^{139}Ba (165.8), ^{87m}Sr (388.4), ^{69m}Zn (438.7), ^{56}Mn (846.8, 1810.9), ^{24}Na (1368.6), and ^{42}K (1524.7). [In the symbol defining an isotope of an element, the mass number (protons + neutrons) is written as a superscript to the elemental symbol. Metastable isotopes (e.g., ^{69m}Zn) occur in energy states above the ground state (i.e., ^{69}Zn). Each radioactive isotope in decay emits γ rays of characteristic energy, measured in keV.]

To determine those elements that give rise to intermediate-lived isotopes upon irradiation, a set of samples and standards was irradiated for 2 hours. After a 26-hour decay, 5000 to 8000 second counts of the gamma activities of the samples and standards were made in a 2-day period. The following elements were determined via their isotopes as listed: ^{153}Sm (103.2 keV), ^{177}Lu (208.4, 113), ^{152m}Eu (121.8, 963.1), ^{47}Ca (1297) and Ca via the daughter ^{47}Sc (160.0), ^{175}Yb (396.1, 282.6), ^{140}La (1596.2, 487.0, 328.7), ^{69m}Zn (438.7), ^{187}W (479.5, 685.7), ^{82}Br (619.0, 776.5, 554.2), ^{76}As (559.1, 657.0), ^{122}Sb (563.9), ^{72}Ga (834, 630), ^{24}Na (1368.6), and ^{42}K (1524.7). Data with unsatisfactory accuracy and precision were collected for uranium via the daughter ^{239}Np (228.1, 277.9).

The samples, with their standards, that had been irradiated for 2 hours were allowed to decay for 30 days, after which the activities due to long-lived isotopes were counted in a 30,000 to 50,000 second counting interval and the following elements were determined: ^{141}Ce (145.4 keV), ^{182}Ta (1221.4, 222.1), ^{169}Yb (177.0, 197.9), ^{131}Ba (216.0, 496.3), Th via the daughter ^{233}Pa (311.9), ^{51}Cr (320.0), ^{181}Hf (482.0), ^{85}Sr (514.0), ^{134}Cs (795.8, 596.3), ^{124}Sb (1691.0), ^{152}Eu (1408.1, 779.1, 1086.0), Ni via ^{58}Co from the ^{58}Ni (n,p) reaction (810.8), ^{160}Tb (879.3), ^{46}Sc (889.2, 1120.5), ^{86}Rb (1078), ^{59}Fe (1099.3, 1291.6) and Fe via ^{54}Mn from the ^{54}Fe (n,p) reaction (824.8), ^{65}Zn (1115.5), and ^{60}Co (1173.2, 1332.4). The data collected for ^{75}Se (264.6, 279.5) were unsatisfactory in accuracy and precision.

The γ-ray spectrometer system used consists of a 55-cm^3 coaxial Ge (Li) detector coupled with a 4096-channel analyzer and a magnetic tape unit. The detector has an efficiency rating of 10.1 percent, a resolution of 2.2 keV (FWHM) at 1332 keV, and a peak/compton ratio of 30.5 at that energy. The data were analyzed by computer in which a modified Gamanal (P. K. Hopke, personal communication) program was used that fits the peaks with gaussian and exponential functions and smoothed background functions. [These data are not included in tables 2 and 3 but will be available from the Illinois State Geological Survey.]

Vertical Distribution

In the Chain of Lakes copper, lead, and zinc increase in abundance upward in the top 20 to 50 centimeters (8 to 20 inches) of most cores. Zirconium and nickel also show upward increases in two or more cores. Other elements that increase in the surface sample of at least one core include molybdenum, boron, cobalt, vanadium, and manganese. The upward increase in trace elements

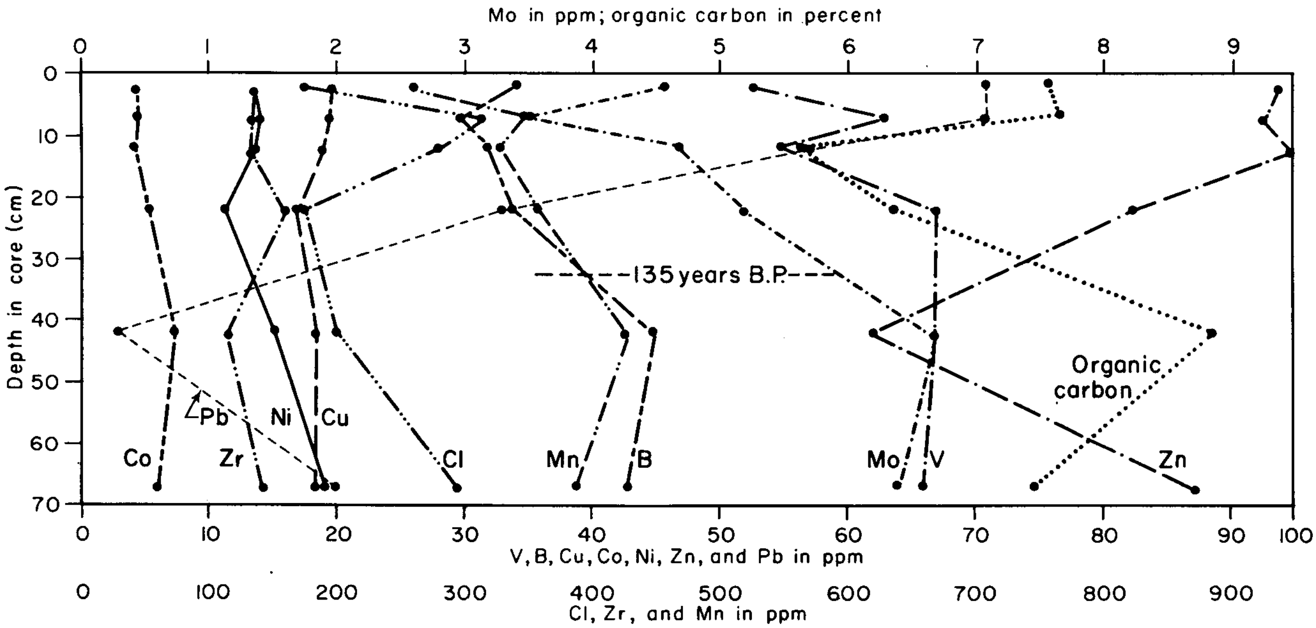

Figure 7. Depth profile for core 29B from Channel Lake

is most clearly seen in cores from the algal silts in the center of the deeper lakes where sedimentation has been slow and continuous (figures 4, 5, 6, and 7). Lead shows the greatest and most consistent increase. Background levels of lead in older sediments is about 20 ppm but increases to an average of 60 ppm at the top of the cores. In each of the three cores on which pollen data were determined, the increase in lead begins above the base of the *Ambrosia* horizon, that is, in sediment deposited after 1840. As most lead is believed to reach the lake environment through the atmosphere from automobiles burning leaded gasoline, this increase in lead probably has taken place within the past 50 to 70 years.

Lead values as high as 442 ppm were found in muck dredged from a boat channel occupied by a marina off Pistakee Lake. Some of the lead may come from gasoline spills or other activities associated with watercraft. One value exceeding 700 ppm recorded from Grass Lake was believed due to the presence of lead from lead shot used in duck hunting.

Areal Distribution and Sediment Type

In addition to the vertical variations within a single core mentioned previously, the geochemical content of the sediments varies from lake to lake and from sediment type to sediment type.

Among the major sediment components, aluminum, iron, silicon, and potassium vary with the proportion of quartz sand, silt, and clay minerals in the sediment. The calcium content of the sediment is proportional to the amount of shells or marl present. The sulfur and nitrogen contents increase with increasing organic content. The highest phosphorus content is in the muck sediments filling dredged channels.

Trace-element distribution in the different types of lake sediment also varies (table 2). In the surface samples, the highest concentrations of vanadium, boron, copper, nickel, lead, and zinc occur in the muck that fills dredged channels. The algal silts in the deep-water

Table 7. Mean Trace-Element* Concentrations of Sediment in the Various Lakes in the Fox Chain of Lakes

(Surface samples only)

Lake	*Number of samples*	*V*	*B*	*Cu*	*Co*	*Ni*	*Be*	*Mo*	*Ge*	*Zr*	*Pb*	*Zn*	*Cd*
Pistakee Lake	6	<30	13	14.8	<3.7	30.5	<1.2	<3.8	<5.3	54	45	63.4	<.6
Nippersink Lake	2	36	21	25.6	<3.8	30.4	<1	<1.3	<3	116	34	98.9	<.9
Fox Lake	10	31	21	19.8	<3.6	20.7	<1	<2.9	<4	72	37	81.7	<.9
Grass Lake	4	39	17	20.0	<6.9	28.6	<1	<2.5	<3	98	35	86.5	<.6
Upper lakes combined (Channel, Catherine, Marie, Petite, Bluff)	8	66	31	23.0	<4.7	19.7	< 1.5	<2.7	<3	126	71	118.9	<.6

*Elements expressed in parts per million

basins contain high concentrations of these elements and also the highest levels of zirconium, molybdenum, and beryllium. Marls contain the highest amounts of cobalt and germanium and high values of molybdenum, but they have the lowest concentrations of most other trace elements, including lead and zinc. The muck contains the highest total average content of trace elements in the lake system, and the sample with the highest total concentration of combined trace elements (1090 ppm) is the muck sample from a marina in a dredged channel off Pistakee Lake.

The upper lakes (Channel, Catherine, Marie, Bluff, and Petite) have higher average concentrations of zirconium, lead, zinc, vanadium, and boron than the other lakes (table 7 and figures 8 and 9). Copper is most abundant in Nippersink Lake; nickel is most abundant in Pistakee, high in Nippersink and Grass Lakes, but low in Fox and the upper lakes. The areal distribution of nickel, therefore, appears as the inverse of the distribution of lead (figure 10).

It is somewhat puzzling that concentrations of most trace elements are higher in the upper lakes than in the lower lakes. The upper lakes are deeper and appear, on superficial examination, to be cleaner than the shallower lower lakes. The samples from the upper lakes, however, are mostly from algal silts that contain higher average concentrations everywhere in the lake system than do the shelly silts more common in the lower lakes. Also, current studies show that water currents are moving from Grass Lake into Lake Marie and then into Channel Lake. This northward movement of water may move sediment out of Grass Lake, and the sedimentary material carried in by the Fox River may be concentrating in the upper lakes rather than being carried through the lake system and out the Fox River outlet.

Organic Carbon

The organic carbon content of the algal silt in all cores shows a decrease of about 1 percent between depths of 12 and 52 centimeters (5 and 20 inches) (figures 4, 5, 6, and 7). This dip in organic carbon may represent a system-wide change in the lake chain that has taken place within the historical past. The decrease may be related to the increase in water depth and wave action that occurred with the damming of the Fox River at McHenry. These changes could have resulted in a temporary lower production of organic matter.

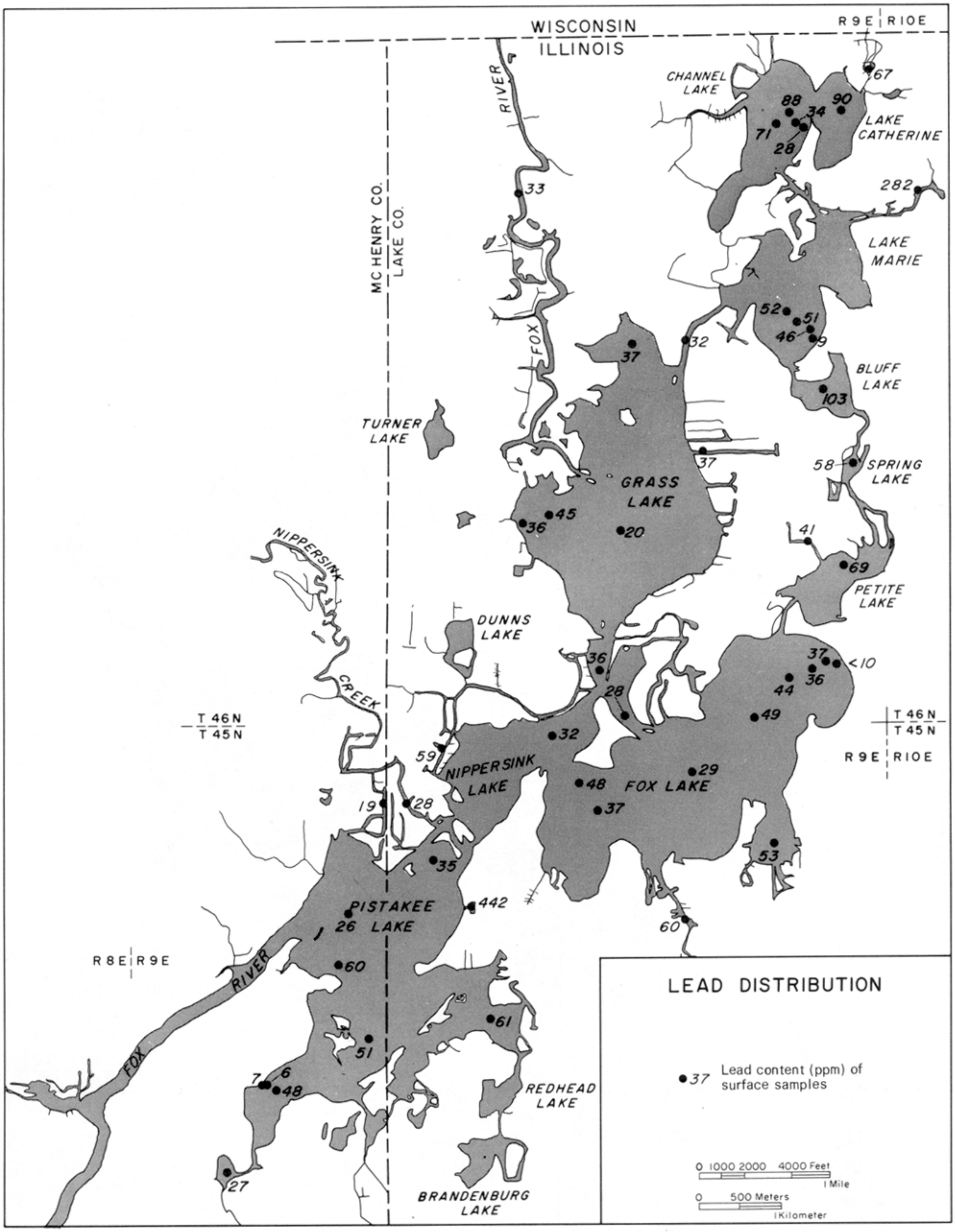

Figure 8. Distribution of lead (ppm) in surface sediment samples

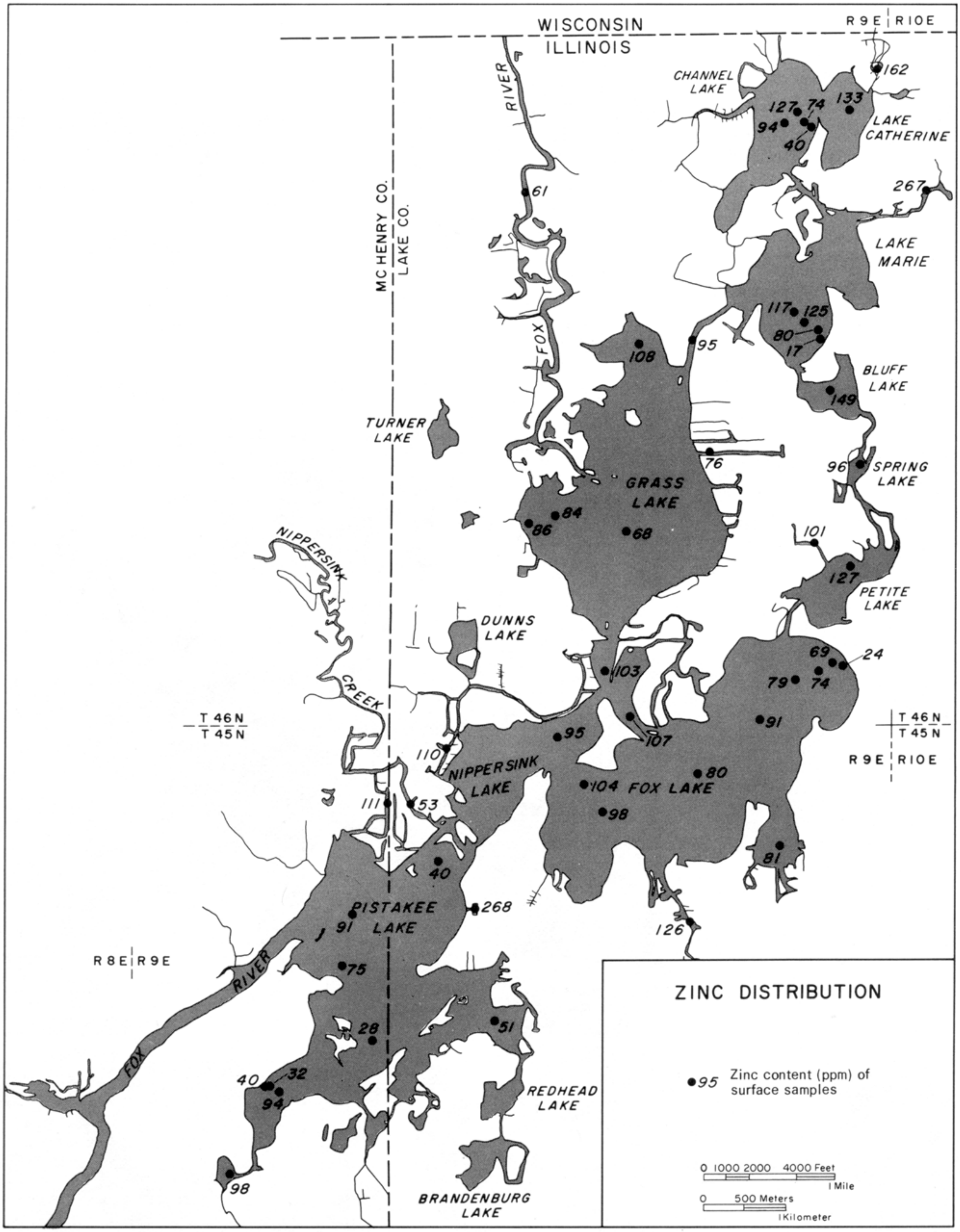

Figure 9. Distribution of zinc (ppm) in surface sediment samples

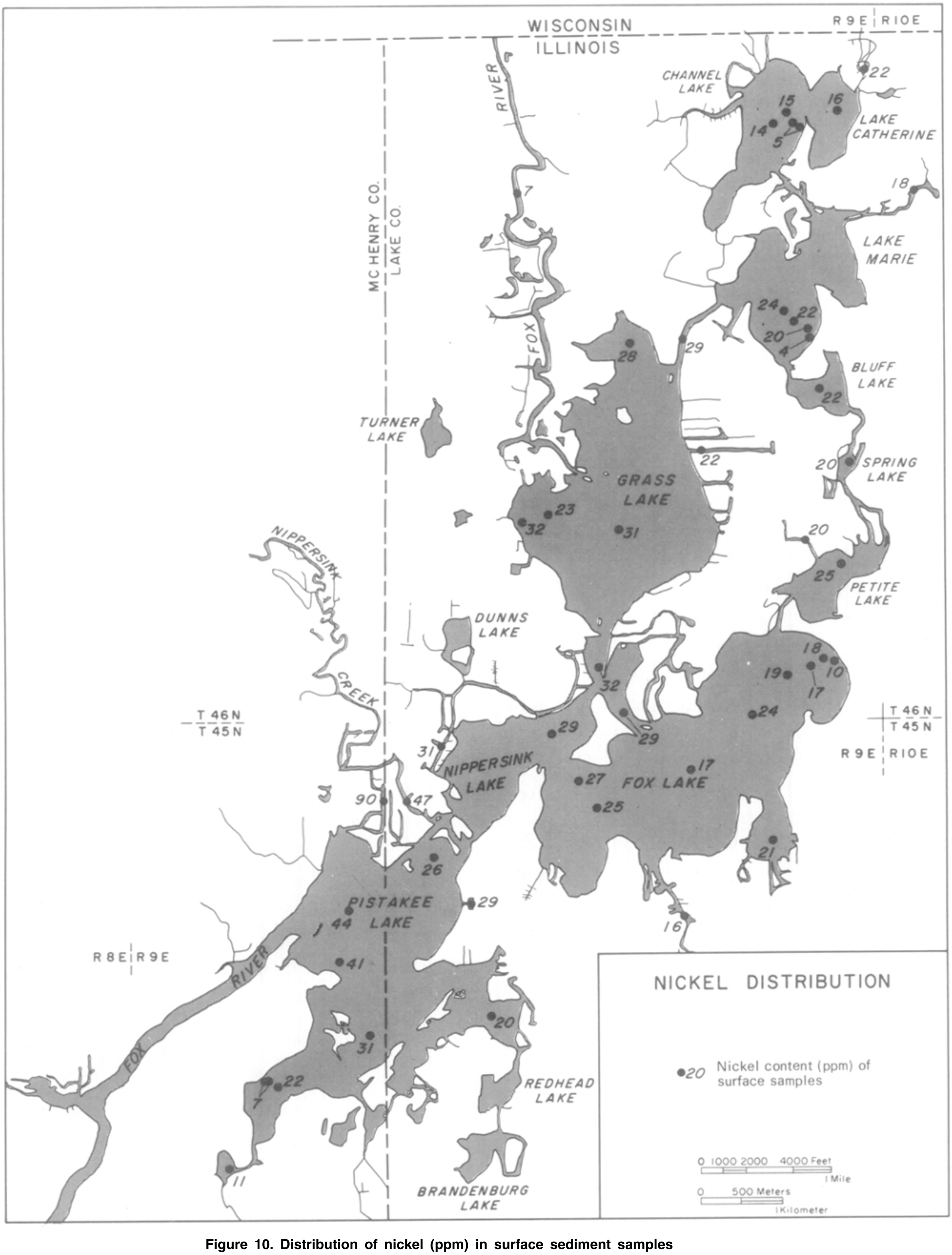

Figure 10. Distribution of nickel (ppm) in surface sediment samples

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Part 2. Hydraulics and Hydrology of the Lakes

BACKGROUND DATA AND FIELD DATA COLLECTION

The Hydrology Section of the Illinois State Water Survey investigated the hydrologic and hydraulic aspects of the Fox Chain of Lakes. The major areas of concern were:

1) Water budget including all inflows and outflows
2) Scour and deposition patterns in the lakes
3) Circulation patterns in the lakes and the interflow between the lakes
4) Theoretical detention times in all the lakes

Field data related to these major areas of concern were collected from December 1974 to November 1975, and these were supplemented by data available from other sources. However, in order to get a feel of the representativeness of this information, the data were compared with long-term variations of certain parameters.

Climate

The climatological parameters that were compared are wind velocity and direction, precipitation, evaporation, and temperature. Long-term climatological data in the vicinity of the Chain of Lakes were available only from the National Weather Service station at Chicago.

Figure 11 shows the wind velocity variation from July 1974 through June 1975. Average daily wind velocities remained in the vicinity of 8 to 10 miles per hour (mph) and the wind generally blew from the southwest.

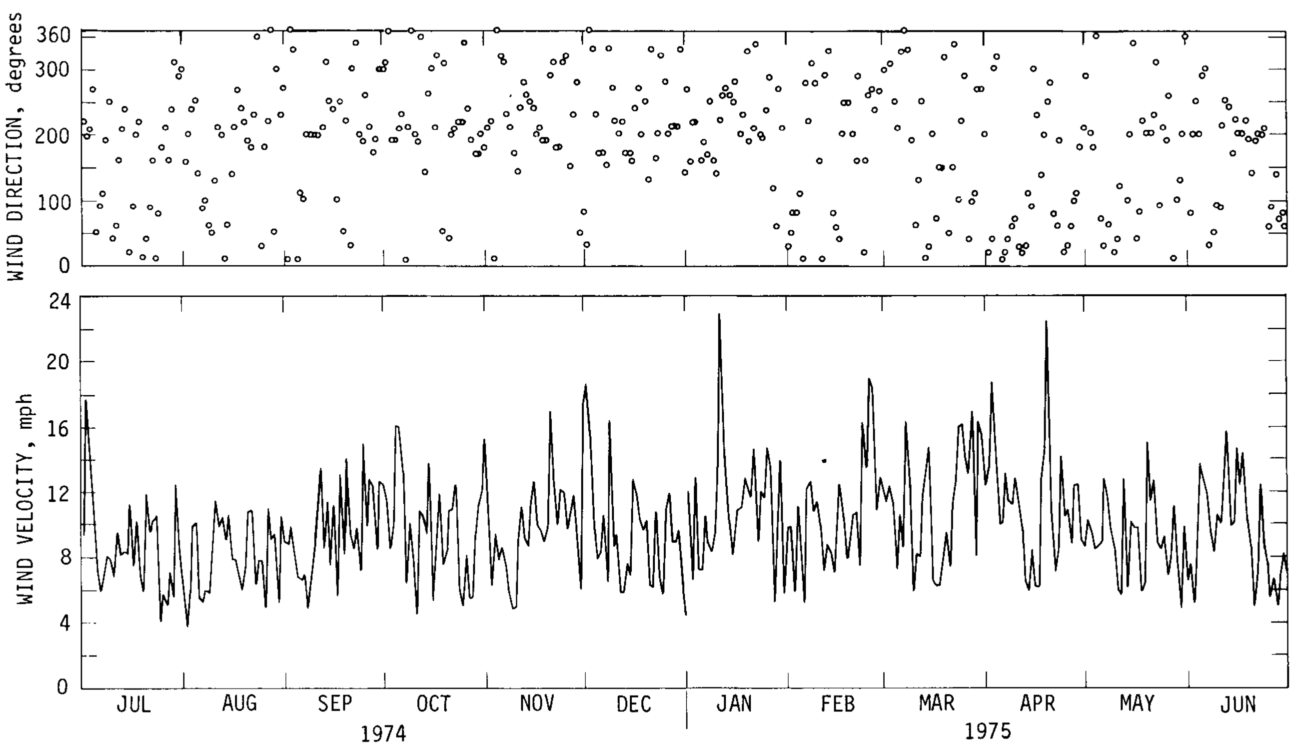


Figure 11. Wind velocity and direction, July 19, 1974 through June 19, 1975 *(Data from Chicago Weather Station)*

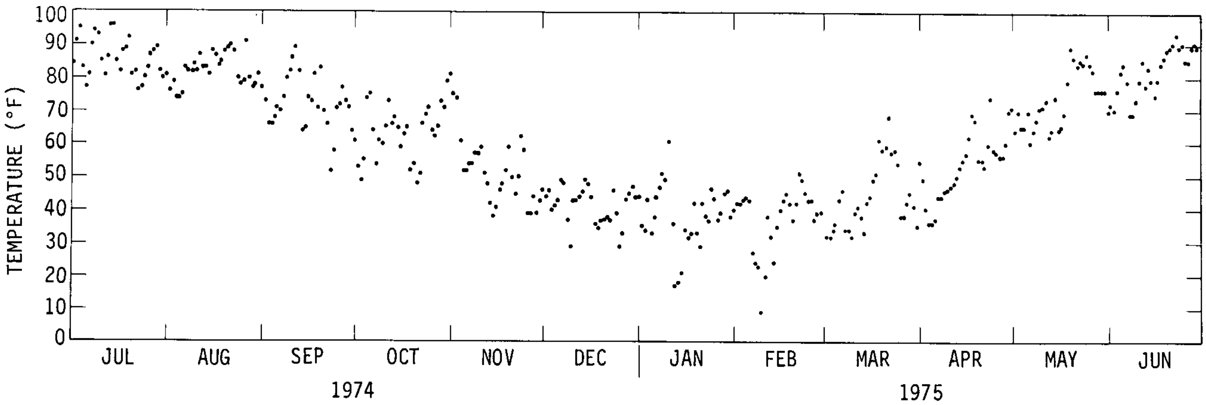


Figure 12. Temperature variation from July 1974 through June 1975 *(Data from Chicago Weather Station)*

Table 8. Comparison of Long-Term and Short-Term Values of Climatological Parameters

	Wind velocity (mph)		*Predominant wind*	*Temperature (°F)*			*Evaporation*	*Precipitation*
	Max	*Avg*	*direction*	*Max*	*Min*	*Avg*	*(inches)*	*(inches)*
November 1974	26	9.7	SW	78	22	40.9	0.66	2.82
November 1960-1974	36	10.6	SW	64	20	41.4	0.85	2.19
December 1974	26	9.3	SW	47	11	31.2	0.24	2.35
December 1960-1974	33	11.4	SW	58	0	26.6	0.36	2.76
January 1975	42	11.0	SW	60	2	27.9	0.33	3.68
January 1960-1974	33	12.0	WSW	54	-8	22.9	0.32	1.93
February 1975	35	10.7	W	51	-6	26.9	0.55	2.29
February 1960-1974	35	12.0	W	54	0	26.9	0.58	1.43
March 1975	31	11.3	NW	74	12	34.2	1.56	2.44
March 1960-1974	37	12.4	NW	72	13	37.7	1.65	2.69
April 1975	35	11.1	NE	76	16	43.5	2.44	7.84
April 1960-1974	39	12.5	variable	76	29	50.0	2.75	4.08
May 1975	35	9.1	E	94	39	63.0	1.39	5.31
May 1960-1974	34	11.1	SW	88	35	60.0	4.10	3.01
June 1975	33	9.6	S	92	49	72.1	5.49	4.63
June 1960-1974	36	10.1	SW	95	48	70.5	4.70	4.43
July 1975	19	8.0	SW	93	52	76.1	5.95	1.53
July 1960-1974	34	9.0	SW	95	53	74.4	5.80	4.30
August 1975	23	8.8	S	94	60	76.4	4.49	5.51
August 1960-1974	31	8.8	SW	94	53	73.3	4.55	2.68
September 1975	29	8.9	SW	86	39	61.4	3.06	1.09
September 1960-1974	30	9.4	variable	91	42	66.4	3.00	3.90
October 1975	27	10.6	S	89	33	56.6	1.88	2.26
October 1960-1974	29	9.5	SW	84	31	55.7	2.10	2.25

Figure 12 shows the variations of the average daily temperatures from July 1974 to June 1975. A cyclic variation in the temperature range is evident with the maximum in the months of May through August and minimum in the months of December through February. Short monthly cyclic variations are also evident in this figure.

Table 8 compares the long-term (1960-1974) monthly average values of various climatological parameters with monthly average values for November 1974 through October

1975. This table shows that the average wind velocity and direction for the year 1975 are generally comparable with the long-term wind data. Both short- and long-term temperatures also compare favorably, and evaporation in 1975 basically remained the same as the long-term average. The precipitation data indicate that 1975 was slightly wetter than the long-term average precipitation.

Long-Term Stage Records

The U.S. Geological Survey and the Illinois Division of Water Resources (DOWR) maintain recording and nonrecording stage gages in and around the Chain of Lakes (figure 13). Long-term stage records from 1959 to 1975 were obtained from DOWR and an analysis was made to check any systematic variations of the lake levels between these stage records at various locations. It must be pointed out that since most of these gages are non-recording types, the stages that were analyzed were not collected at the same time of day.

It was thought that wind might have some effect on the variation of the water levels at various parts of these lakes. All the lakes are oriented in more or less southerly or southwesterly directions. The long-term wind analysis (figure 11 and table 8) indicates that the wind normally blows from the south or southwest. This predominant wind can pile up water on the eastern shore of the lakes with an associated lowering of water level at the western shore. This phenomenon is termed *wind tide.*

In order to investigate the variations in water levels between different stages for strong winds from the southwest, a number of plots were prepared comparing stages at McHenry Lock and Dam with stages at Nippersink Lake, Columbia Bay in Fox Lake, and Channel Lake. Typical plots are included in this report. In general, there are some variations in the stages with or without wind.

Figures 14, 15, and 16 show the stage variations between McHenry Lock and Dam and Nippersink Lake, Columbia Bay, and Channel Lake, respectively. These figures show the variations for both light winds (2 to 5 mph) and strong (10 to 30 mph) southwest winds. Table 9 was prepared on the basis of these figures and shows the maximum differences between various stations for light and strong winds. Most of the time, a maximum of about 0.5 foot variation can be attributed to the wind effect. However, this is a maximum variation, not an average value.

Figure 17 shows the stage variation between Channel Lake and Nippersink Lake for strong SW winds. The average variation is about 0.3 foot with a maximum value of 0.55 foot. Similarly figure 18 shows the stage variation between Columbia Bay in Fox Lake and Nippersink Lake for strong W, WSW, and SW winds. The effect of a SW wind on Fox Lake should be predominant because it is located exactly in the SW direction. However, on the average the variation between these stations is in the order of 0.1 foot with a maximum variation of about 0.4 foot. Therefore, it appears that either the wind tide is not significant or the data on stages were not collected simultaneously from various gages when the wind was blowing hard from the SW.

A sample computation was made to estimate the amount of wind tide in Fox Lake for some hypothetical wind conditions blowing from the SW (table 10). Many researchers have studied this phenomenon as reported by Saville et al. (1962) and the following relation is generally accepted as valid to estimate wind tide

$$S = (K V_z^2 F_e \cos \alpha)/D$$

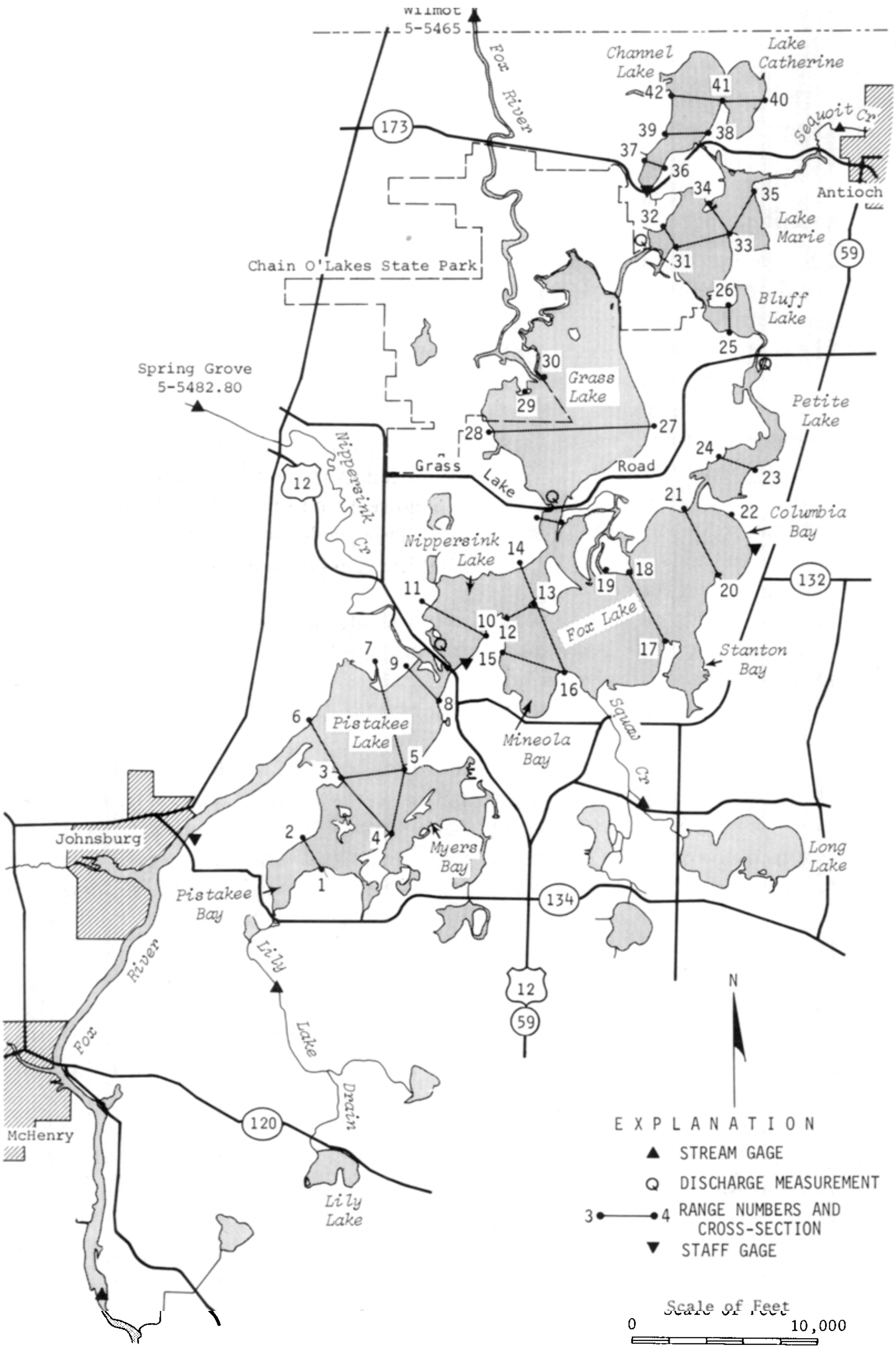


Figure 13. Fox Chain of Lakes

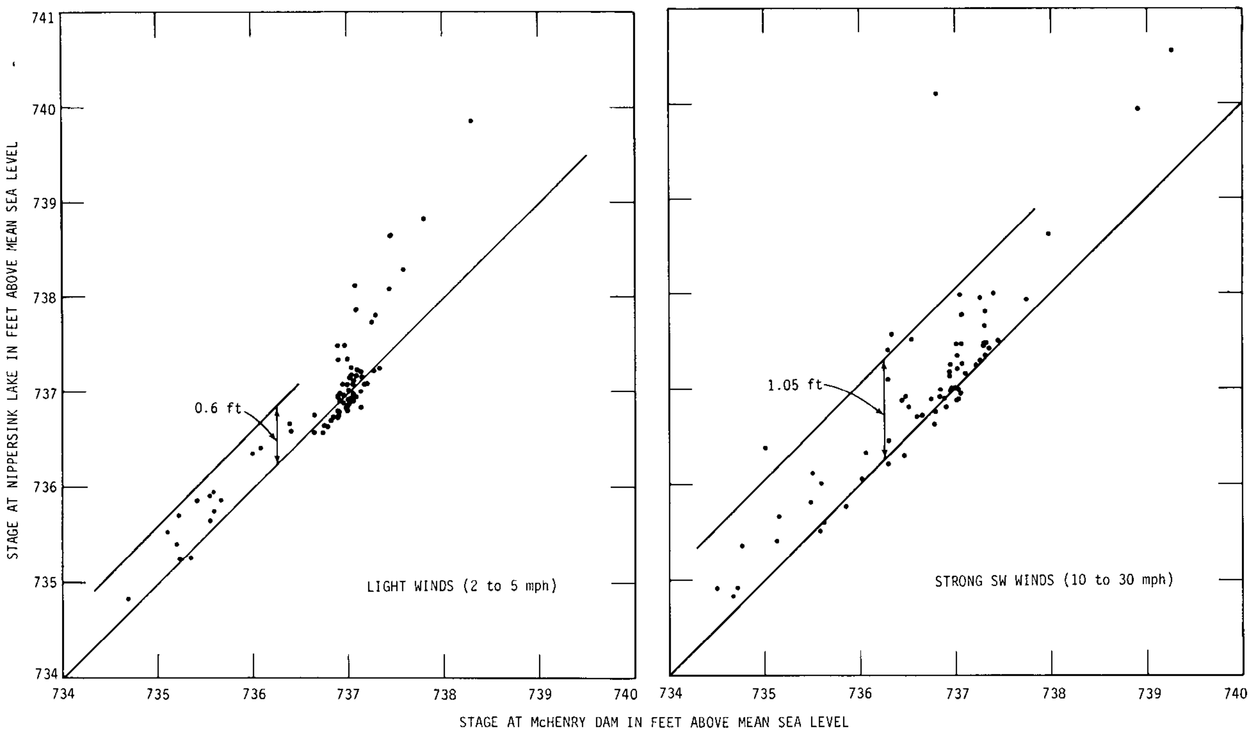


Figure 14. Stage variations between Nippersink Lake and McHenry Lock and Dam for different wind conditions

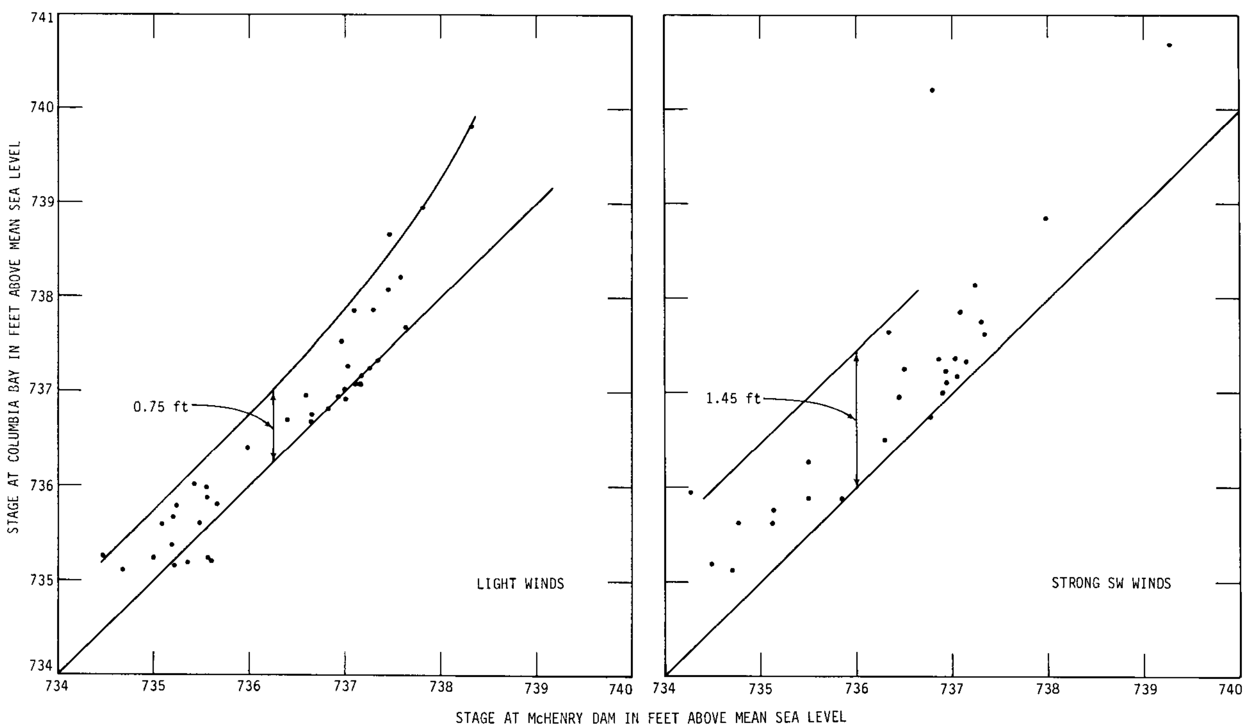


Figure 15. Stage variations between Columbia Bay and McHenry Lock and Dam for different wind conditions

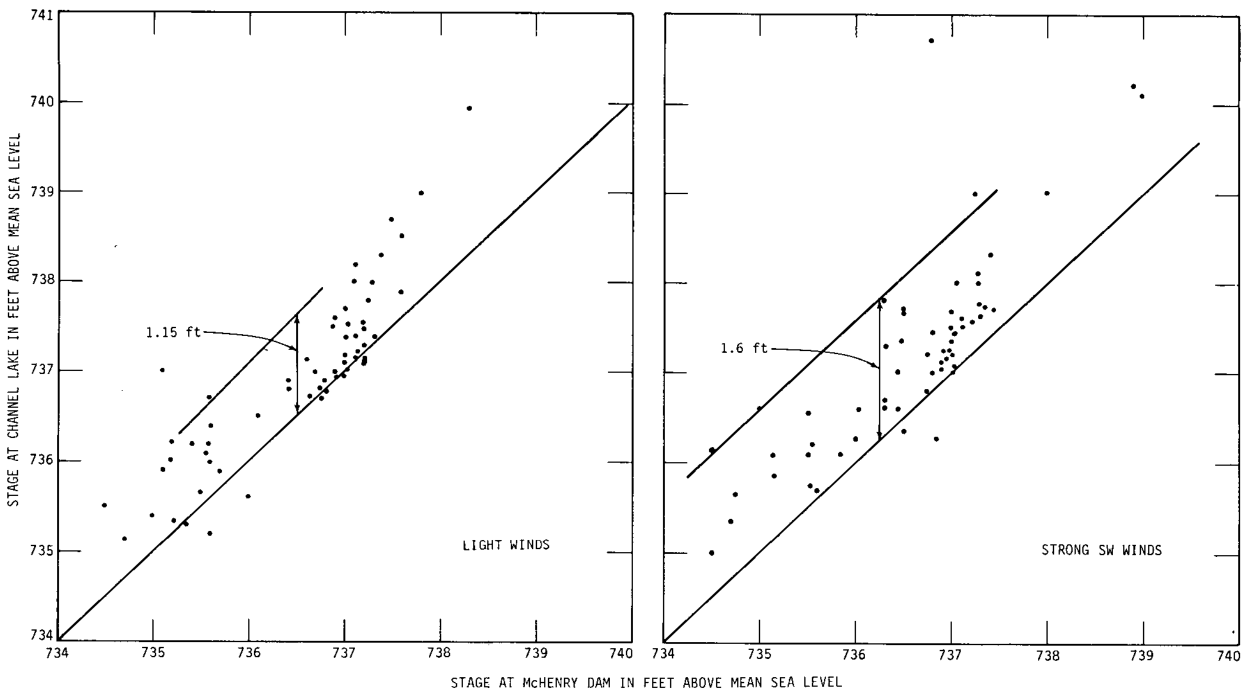


Figure 16. Stage variations between Channel Lake and McHenry Lock and Dam for different wind conditions

Table 9. Differences in Water Surface Elevations*

Wind condition	*Maximum difference in elevation (in feet) above McHenry Dam* to Nippersink	to Columbia Bay	to Channel Lake
Light or no wind	0.60	0.75	1.15
Strong SW wind	1.05	1.45	1.60

*Normal pool level 736.5 ft above msl at McHenry Lock and Dam; data for 1959-1975

Table 10. Wind Tide in Fox Lake with SW Wind

Wind speed, V_z (mph)	*Wind tide, S, in feet, Fox Lake only**	*Wind tide, S, in feet, Nippersink-Fox Lake***
5	0.009	0.010
10	0.037	0.044
15	0.082	0.098
20	0.148	0.176
30	0.330	0.393
40	0.600	0.715
50	0.900	1.07

*Fetch = 2.88 miles for Fox Lake

**Fetch = 3.43 miles as measured from a point near Riverside Island in Nippersink Lake to the shore of Columbia Bay in Fox Lake

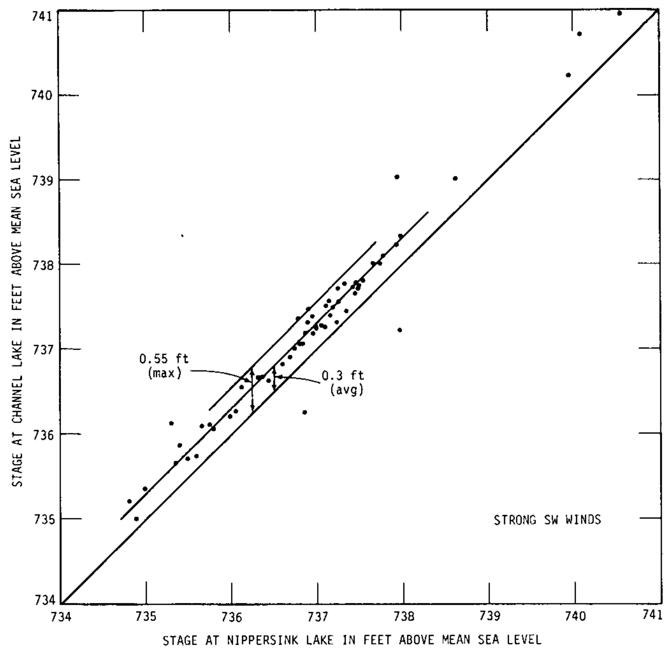


Figure 17. Stage variations between Channel Lake and Nippersink Lake for strong southwest winds

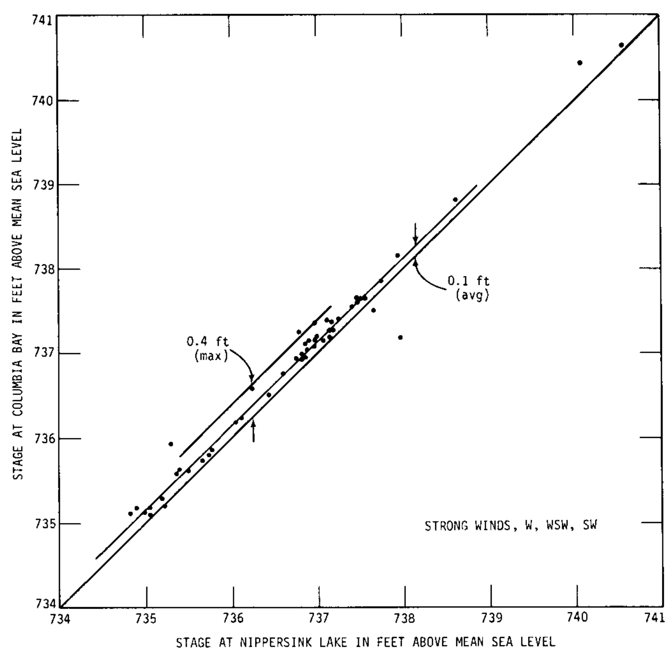


Figure 18. Stage variations between Columbia Bay and Nippersink Lake for strong winds

where S is the wind tide, K is a coefficient, V_z is the wind velocity, F_e is the fetch or length of water surface, a is the wind angle with the main fetch length, and D is the average depth of water in the lake. The value of K is taken to be 1/1400 (Saville et al., 1962) with F_e in miles, V_z in mph, cos a equal to 1.0, and D equal to 5.6 feet for Fox Lake (from figure 28).

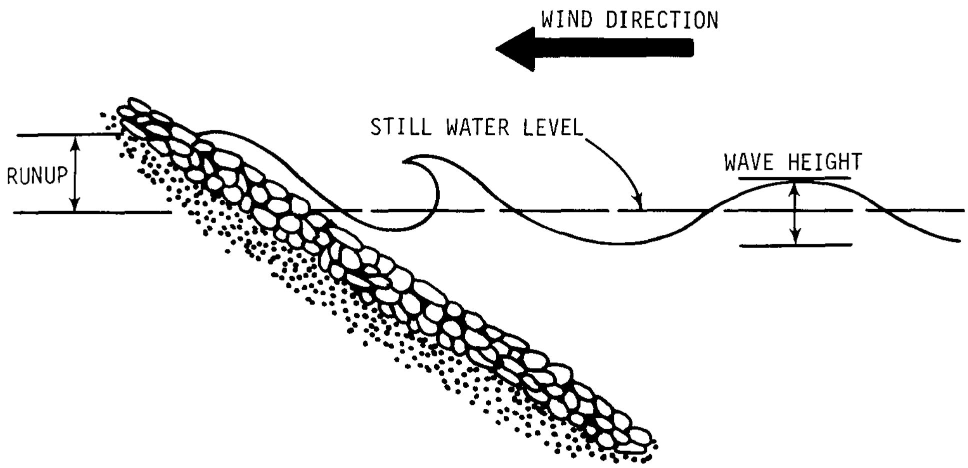


Figure 19. Sketch of a wave breaking on a lake shore and waves breaking at Eagle Point Park on a windy day

Thus is appears that, for a sustained wind of 50 mph from the SW, a maximum wind tide of 1.07 feet is possible in the Columbia Bay area of Fox Lake. However, in order for a wind tide of 1.07 feet to occur, the wind must be blowing continuously for some time from the SW in line with the maximum fetch of the lake.

Another phenomenon which might give the appearance of an increased water depth on the eastern shore of Fox Lake with wind blowing from the SW is termed *wave runup.* When a wave breaks up on a high shore or sea wall, it rushes up the shore and breaks above the normal water level. The water then travels back down the shore and returns to the lake. With a sustained wind, this breaking of waves is continuous, and at a glance will look like an increased depth of water on the leeward side of the lake. Figure 19 shows a sketch of a wave breaking up on a shore and a picture of breaking waves at Eagle Point Park on one of the many windy days.

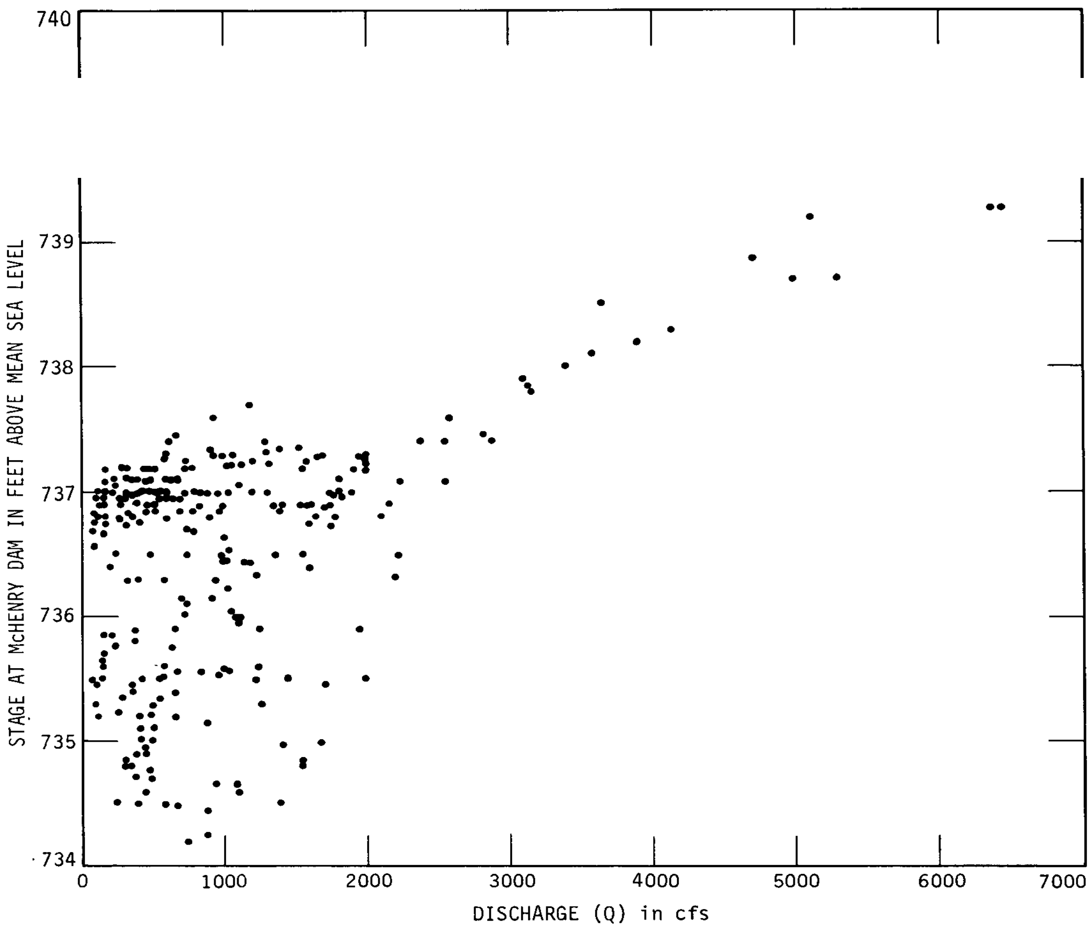


Figure 20. Stage versus discharge at McHenry Dam (1959-1974)

According to Hedar (1953) the *wave runup* can be as much as 1.25 times the wave height when the ratio of wave length to the wave height is about 20. Wave heights in any enclosed body of water, such as lakes and ponds, can be computed by the method presented by Bhowmik (1976).

Figure 20 shows the stage versus discharge relationship at McHenry Lock and Dam. Above a stage of about 737.5 feet above msl, the gates at McHenry Dam have little or no control of the total quantity of flow that will pass below the dam.

Long-term stages from all the Chain of Lakes gages were also statistically analyzed to determine cyclic variations in the stages. Auto-correlation analysis indicated cyclic variations of 12 months, which also contain 3-, 6-, and 9-month cyclic variations. This is natural, since all these lakes are basically fed by streams and their water levels fluctuate with inflows which have a basic cyclic variation of 1 year.

Field Data Collection

Field Surveys

Extensive field data collected in connection with this investigation required some precise surveying in the Chain of Lakes region. Forty-two permanent concrete monuments

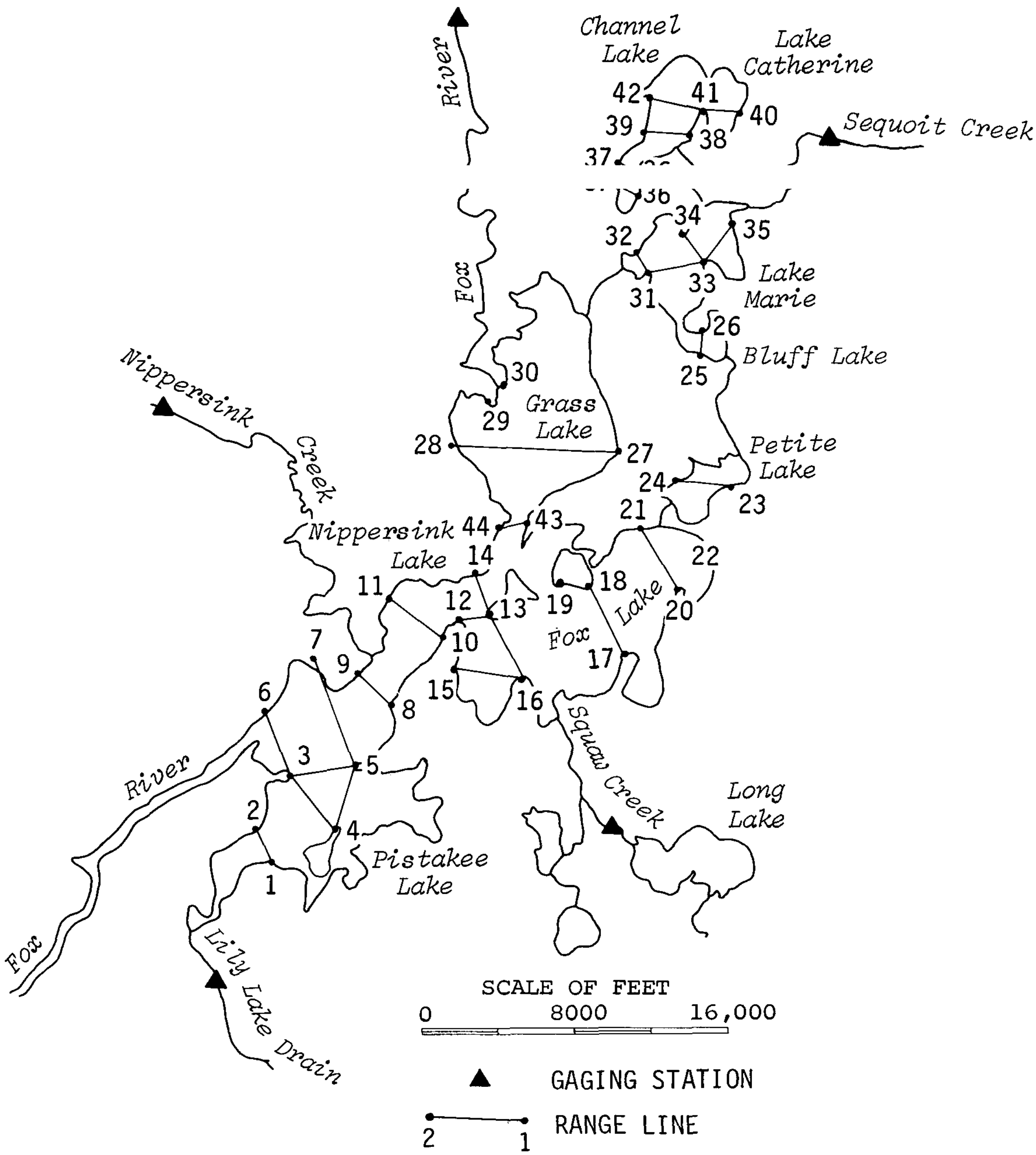


Figure 21. Locations of depth measurement monuments

and a number of semi-permanent monuments were installed around the lake shores (figure 21). A line joining any two monuments indicates that the 1975 lake bed elevations were determined at those locations. The station 29 monument could not be installed since this location is marsh.

Horizontal Control. The horizontal control consisted of an electronic traverse, run to second-order specifications, using T2 theodolites and a Laser Ranger distance-measuring device. The primary control stations for the project were USGS triangulation stations located in the Chain of Lakes area. The total control network consisted of 106 stations and 166 observed lines.

In cases where a number of stations surrounded a lake, all possible intervisible lines were observed and measured, and all measured lines were used in the final adjustment procedures.

The closures of the various lines and loops ranged from 1 part in 16,100 to 1 part in 569,400, with an average of 1 part in 78,600.

Whenever possible, at least one prominent object was pointed as an azimuthal mark at each station. In addition, coordinates were determined by intersection on 10 prominent structures (water tanks, radio masts, etc.) within and around the area.

Vertical Control. The vertical control consisted of lines of spirit leveling for which the '3 wire' method was used, with automatic and prism levels, and 'invar' precise rods. A number of water crossings were required, and these were accomplished by standard 'river crossing' techniques with multiple observations.

The project was run on mean sea level datum as determined from nine USGS benchmarks distributed about the area. The entire network was adjusted as a whole, holding fixed all of the undisturbed USGS benchmarks.

Although the USGS benchmarks were listed as being of third-order accuracy, nearly all of the benchmark to benchmark closures, as well as the loop closures, met second-order accuracy requirements.

In addition to determining elevations on all of the monuments, approximately 100 supplementary benchmarks were set at intervals along the level lines. Also, whenever possible, a reference benchmark was set near each monument, as a precaution against the monument being lost or inadvertently disturbed.

Depth Measurements

The 1975 lake bed elevations were determined along 26 cross sections as shown in figure 21. Depths were measured for a minimum of 5 points and a maximum of 10 points in each cross section. The end points of the cross section lines were the various concrete monuments set as part of the control survey. Individual soundings were located by lining the boat with a transit and reading the distance out with the Laser Ranger.

The lake bed measurements were made with a steel pipe rod, 20 feet in length, with a 4-inch diameter plate on the bottom and a sliding level target. Depths over 20 feet were measured on calm days, using a chain to extend the rod.

The three independent readings of the lake bed elevation taken at each sounding were: 1) depth from the water surface; 2) level reading from shore to the sounding rod target; and 3) vertical angles from shore to the top of the rod. The last two were corrected for curvature and refraction, and all three were averaged for a final lake bed elevation.

The lake bed character was tentatively determined by the feel of the rod on the bed, and by examination of the lake bed material on the boat anchors as they were pulled.

Inflows and Outflows

Basically the inflow and outflow data were gathered from established gaging station records. However, at the beginning of the project, it was observed that at least three small tributaries for which no inflow records were available enter the Chain of Lakes. Three temporary wire-weight gages were installed at the three tributaries, namely, Sequoit Creek, Squaw Creek, and Lily Lake Drain (figure 21). These three gages were read daily and the USGS developed rating curves at all three locations. In addition to these three gaging stations, records from permanent gaging stations of Wilmot, Wisconsin, on the Fox River and from Spring Grove on Nippersink Creek, as well as discharge records at McHenry Lock and Dam, were also utilized.

Interflow between Lakes

In order to understand the flow pattern and the general movement of water in all the lakes some qualitative and quantitative information regarding interflow between the lakes should be known. Interflow between Grass Lake and Lake Marie, flow from Bluff Lake to Petite Lake at Grass Lake Road Bridge, flow from Grass Lake to Fox and Nippersink Lakes at Grass Lake Road Bridge, and the flow from Nippersink Lake to Pistakee Lake under both bridge openings of Highway 12 were gaged in the months of December 1974 and May, July, and October of 1975. In addition to these gaging data, some qualitative information was obtained by using floats and vanes at some locations in November 1975.

During the latter part of the data collection period it was observed that a measurable amount of water was flowing under the east and west bridges of Highway 173 between Channel Lake and Lake Marie. On four occasions in October and November 1975, the flow under these bridges was estimated by the use of floats and cross-vanes.

Circulation Patterns

Interflow between lakes gives some indication of the general movement of water in all of the lakes. However, circulation patterns in some of the connecting areas were needed for a better understanding of the flow patterns of the lakes as a whole. Extensive data on circulation patterns were collected in 1975.

The average flow velocity within any one of the lakes is very small. To date, there is no commercial flow meter available that can measure the very small velocity in lakes. After considerable literature review, a method used by Shulman and Bryson (1961) to measure wind driven currents in Lake Mendota (in Wisconsin) was adopted. The method consists of dropping a cross-vane suspended by a nylon cord and a float at some location in the lake and then following the float by two theodolites from two fixed shore stations. Figure 22 shows a sketch of the float and cross vane arrangement. The depths of the vanes varied depending upon the flow depth in the lakes. Three or four drops were made at each location with vanes at different depths. This technique resulted in mapping the flow patterns at each selected location at all possible depths below the water surface. Simultaneous readings at some specified interval, usually 5 to 10 minutes, from two fixed shore stations were needed to map the direction and to compute the magnitude of the flow velocity for each drop of a float and vane. In order to test the cross-vane movement with the average velocity of the surrounding water, the cross-vanes with floats were calibrated at the Water Survey hydraulics laboratory in the 2-foot-wide and 60-foot-long flume. The correlation was excellent.

Figure 23 shows a sketch of the actual operation of determining the circulation patterns by this technique. At every predetermined interval of time, angles a and b were measured simultaneously. At the end of time t, angles a' and b' were measured and this procedure was repeated until sufficient data had been collected. Then the floats were moved to another location and the procedure was repeated. The base distance between 8 and 9 in figure 23 was determined by using an electronic distance-measuring instrument. A simple triangulation technique was needed to determine the actual location of the floats in the lake. Normally, three floats at different depths (12, 24, and 36 inches) were dropped and followed at the same time.

Figure 24a shows a float in the water. Sometimes a brightly painted table tennis ball was fixed on a thin rod on top of the float to make it easier to see against the reflections on the water surface. Figure 24b shows the location of Station 32 on the west shore of Lake Marie where the flow patterns from Grass Lake to Lake Marie were being tracked.

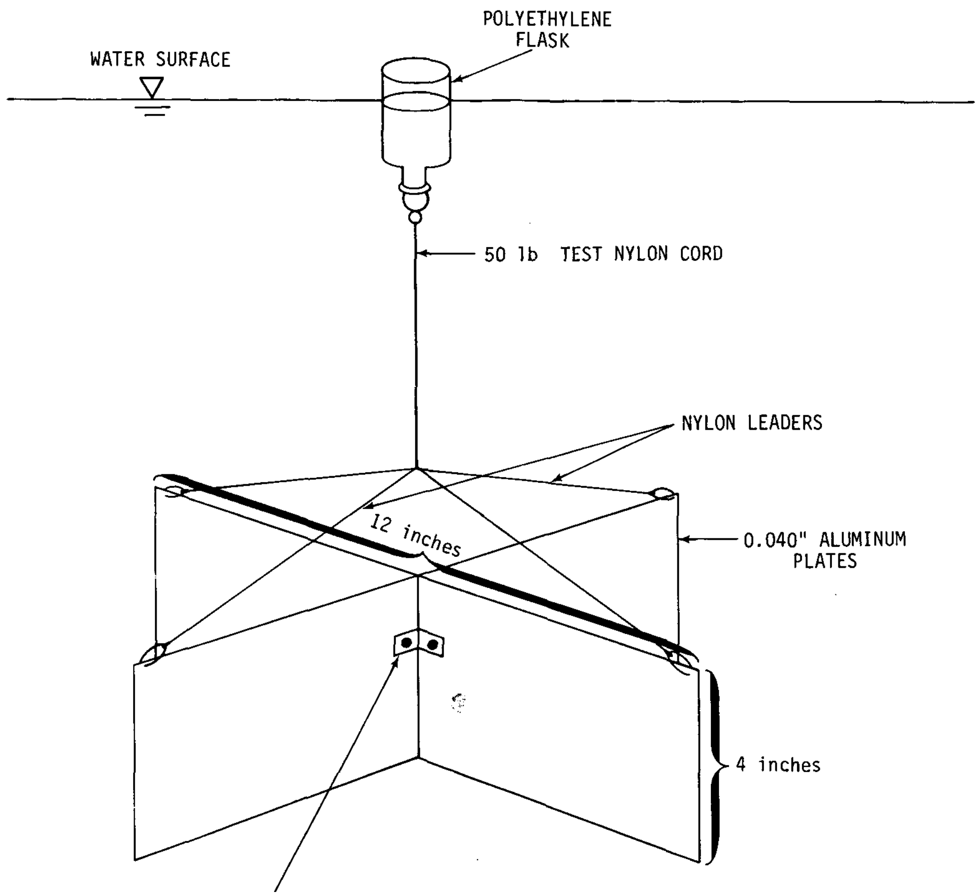


Figure 22. Float and cross-vane used to measure circulation patterns

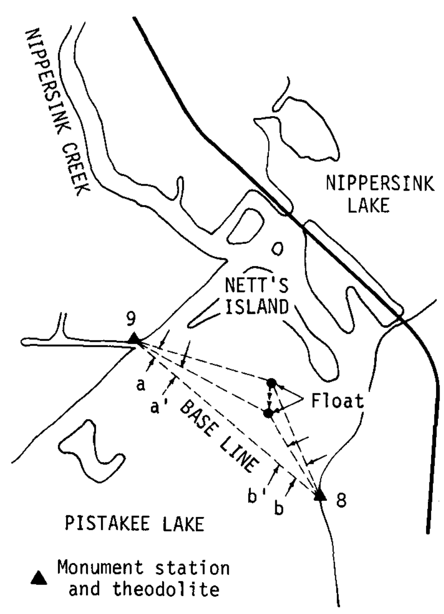


Figure 23. Schematic of circulation data collection

(a)

(b)

Figure 24. Float with suspended vanes in water (a) and tracking of the floats with theodolite at station 32 on the west shore of Lake Marie (b)

Sustained wind blowing from any direction on a lake or reservoir will generate some movement of water in the lake. This is termed wind current or wind-generated circulation. It was suspected that wind-generated currents in the Chain of Lakes would be significant and should be investigated for the present study. A recording wind set was thus installed in the Chain of Lakes, whenever field data on circulation patterns were collected. Figure 25 shows the installed wind set at Korpan's Landing on the shore of Fox Lake. These wind data were analyzed and subsequently used in conjunction with the circulation data.

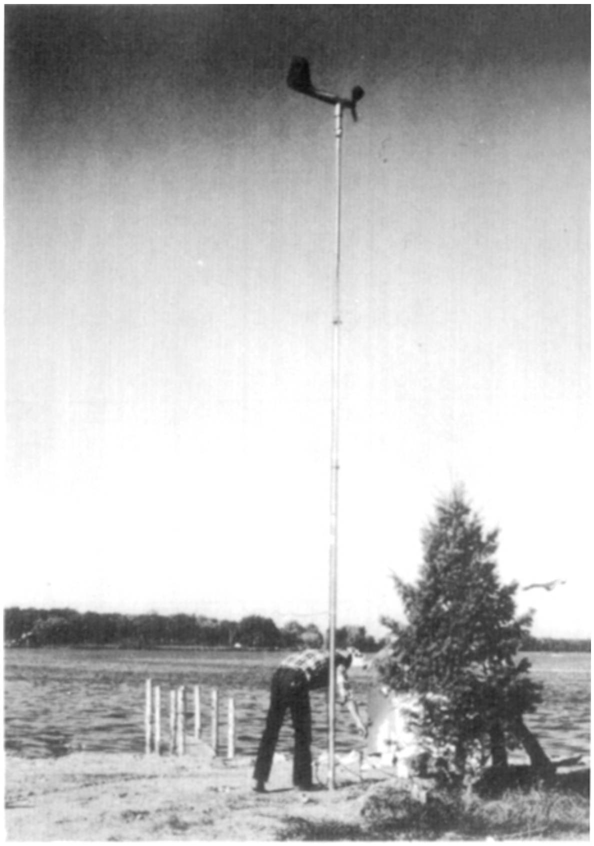


Figure 25. Wind set installed at Korpan's Landing on the shore of the Fox Lake

DATA ANALYSES

Depth-Area and Depth-Storage Relationships

To estimate the effective volume of any lake for hydraulic detention time, knowledge of the depth-area and depth-storage relationships is essential. Development of the depth-storage relationship requires up-to-date information related to bed topography of the lakes. Data on 1975 bed elevations were collected at only 26 cross sections in all of the lakes. This information was superimposed on the maps published by the Division of Water Resources to obtain an adjusted 1975 bed topography for the lakes. Because only one or two cross sections were measured in some of the lakes, the estimates required a considerable amount of judgment, though in most cases the 1975 bed topography determined in this manner should be satisfactory.

The modified maps with contour lines representing the 1975 lake bottoms were used in the development of the depth-area and depth-storage relationships. The surface areas of each lake at each selected contour interval were planimetered, the storage capacity was computed, and the individual relationships were then developed.

The depth-area and depth-storage relationships for all the lakes are shown in figures 26-30. Lake Catherine and Channel Lake are the deepest lakes in the Chain and Grass Lake is the shallowest. Pistakee Lake has the largest surface area and storage capacity among the lakes. The average depths are as follows on the next page:

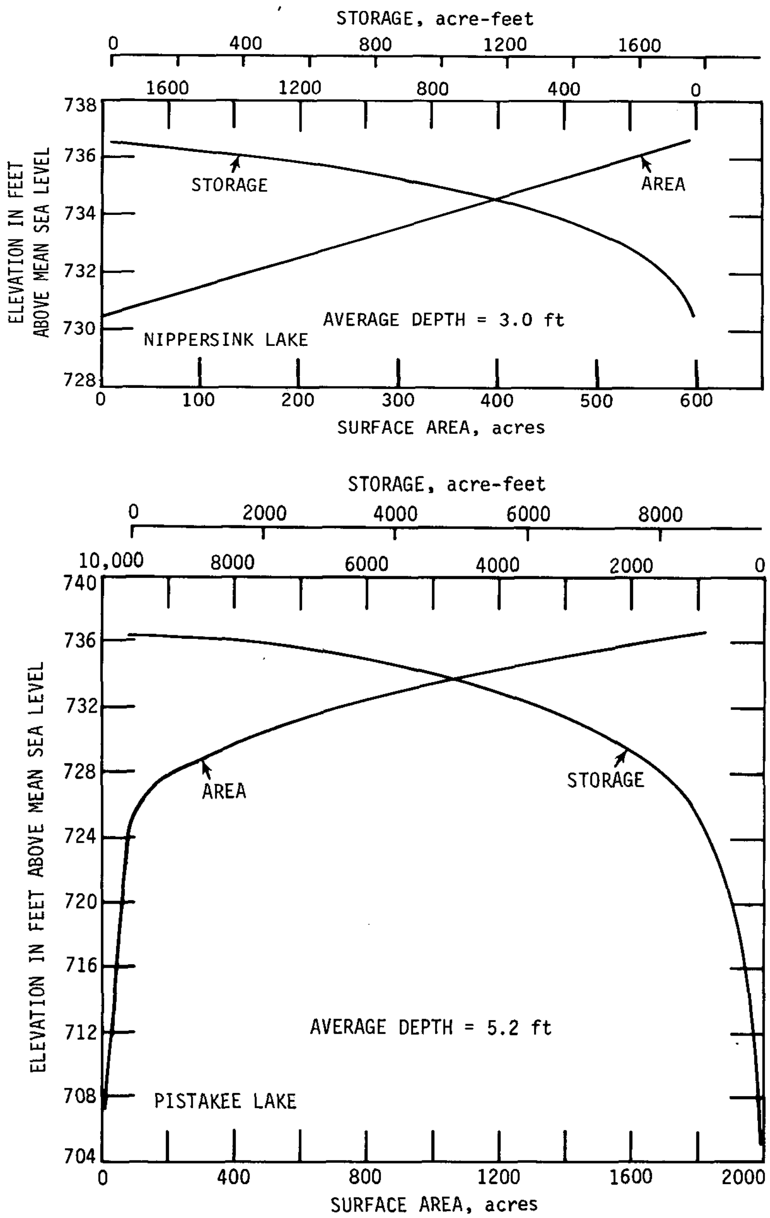


Figure 26. Depth-area and depth-storage relationships for Pistakee and Nippersink Lakes

	(feet)		*(feet)*
Pistakee Lake	5.2	Petite Lake	7.7
Nippersink Lake	3.0	Bluff Lake	10.5
Pistakee Bay	11.3	Grass Lake	2.7
Lake Catherine	16.7	Lake Marie	9.2
Fox Lake	5.6	Channel Lake	13.8

The total 1975 storage capacity of all 9 lakes below the elevation of 736.5 feet above msl is 38,718 acre-feet and the total water surface area at this elevation is 6844 acres. The total storage capacity of all the lakes below the 20-foot depth or 716.5 feet elevation

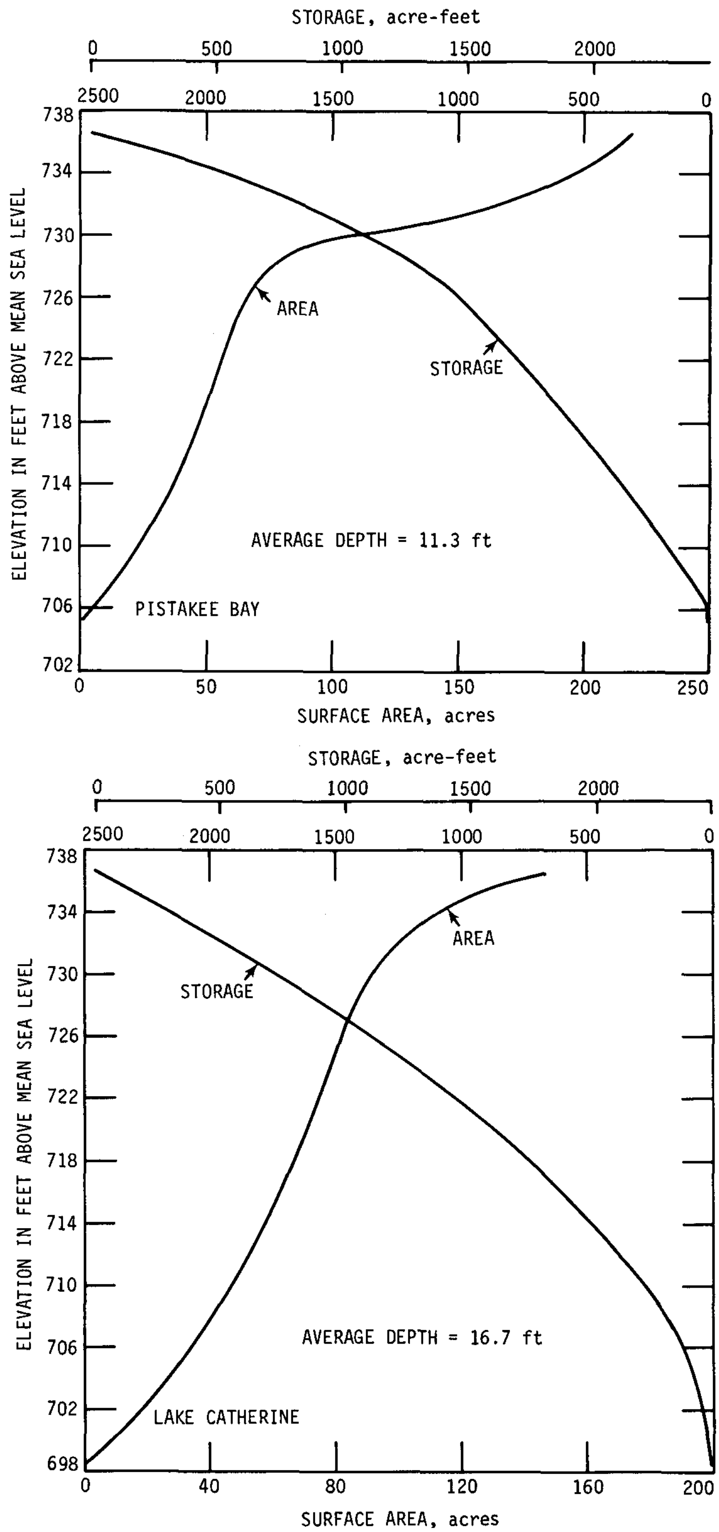


Figure 27. Depth-area and depth-storage relationships for Pistakee Bay and Lake Catherine

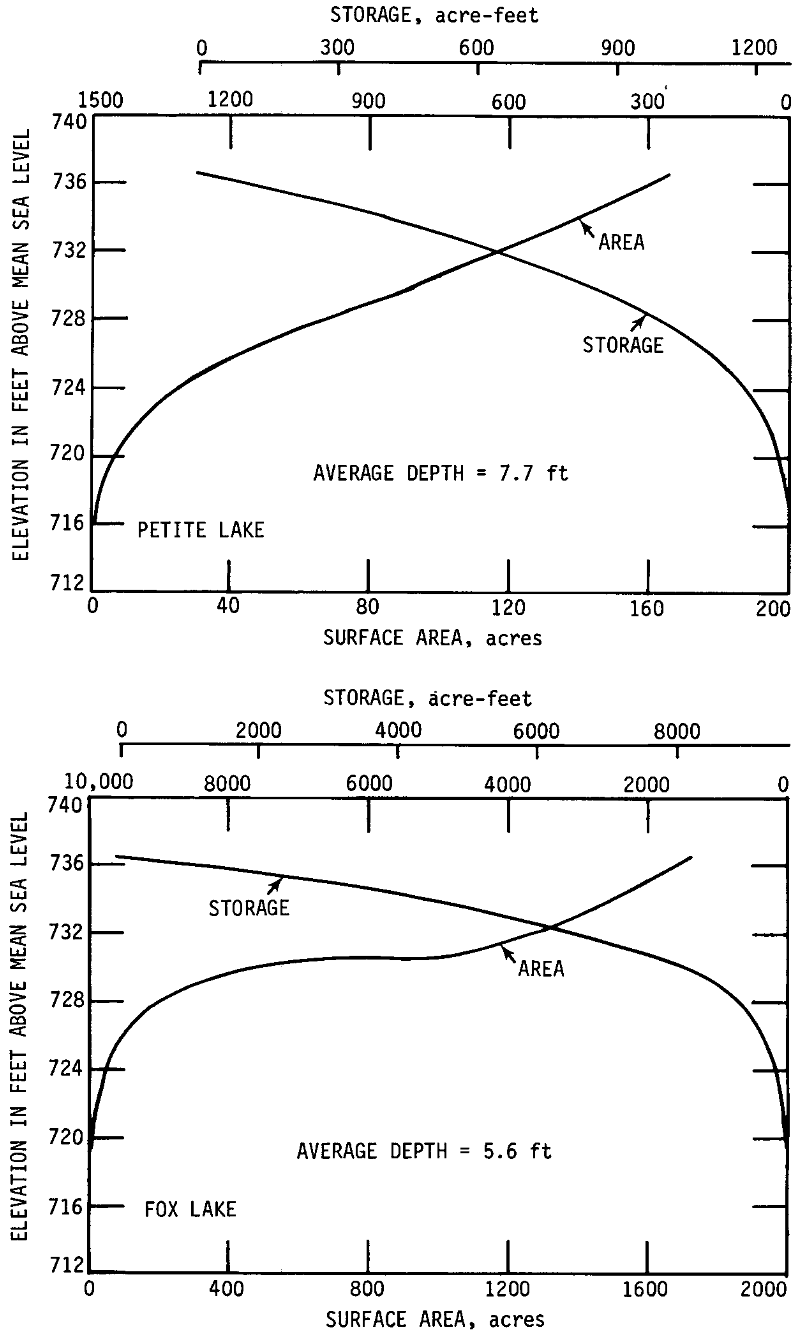


Figure 28. Depth-area and depth-storage relationships for Petite Lake and Fox Lake

is 2161 acre-feet or about 6 percent of the total storage capacity. However, most of this storage is in Channel Lake and Lake Catherine. For example, the storages below the 20-foot depth in Channel Lake and Lake Catherine are 19 and 25 percent, respectively, of the capacities of those lakes. For Pistakee Lake, Lake Marie, and Bluff Lake, the storage below that depth is 2.7, 7.4, and 10.9 percent, respectively, of their total capacities.

In general, lakes less than 20 feet deep are homogeneous in character. Stratification

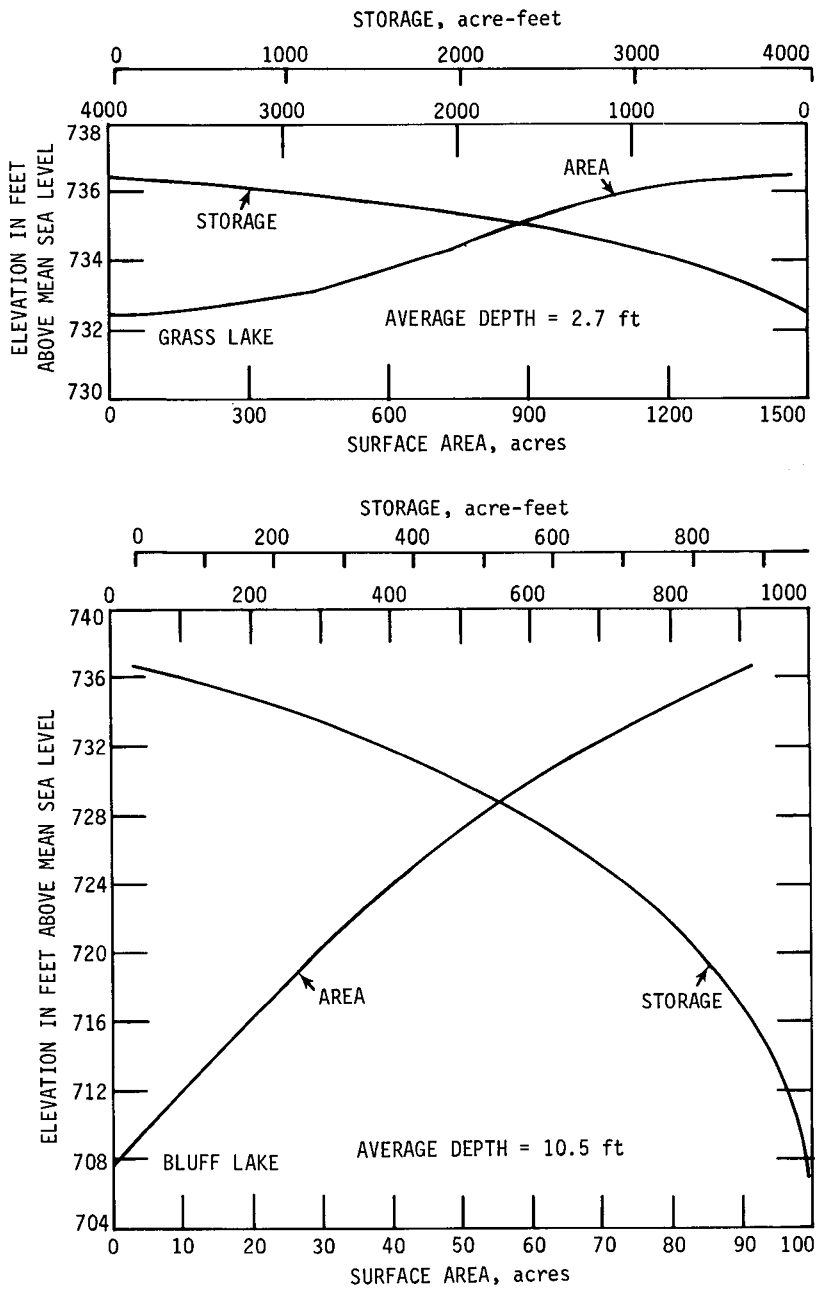


Figure 29. Depth-area and depth-storage relationships for Grass Lake and Bluff Lake

in lakes may occur when depths of the lakes exceed about 20 feet. Therefore, the depth-storage relationships are essential for the analysis of dead storage for water quality studies.

Depth Soundings

It has already been pointed out that depths were measured at 26 cross sections in the lakes. The sounding results were compared with the depths of the lakes published in a set of five maps by the Division of Water Resources (DOWR), from lake bed elevations measured in the years 1953, 1962, and 1964.

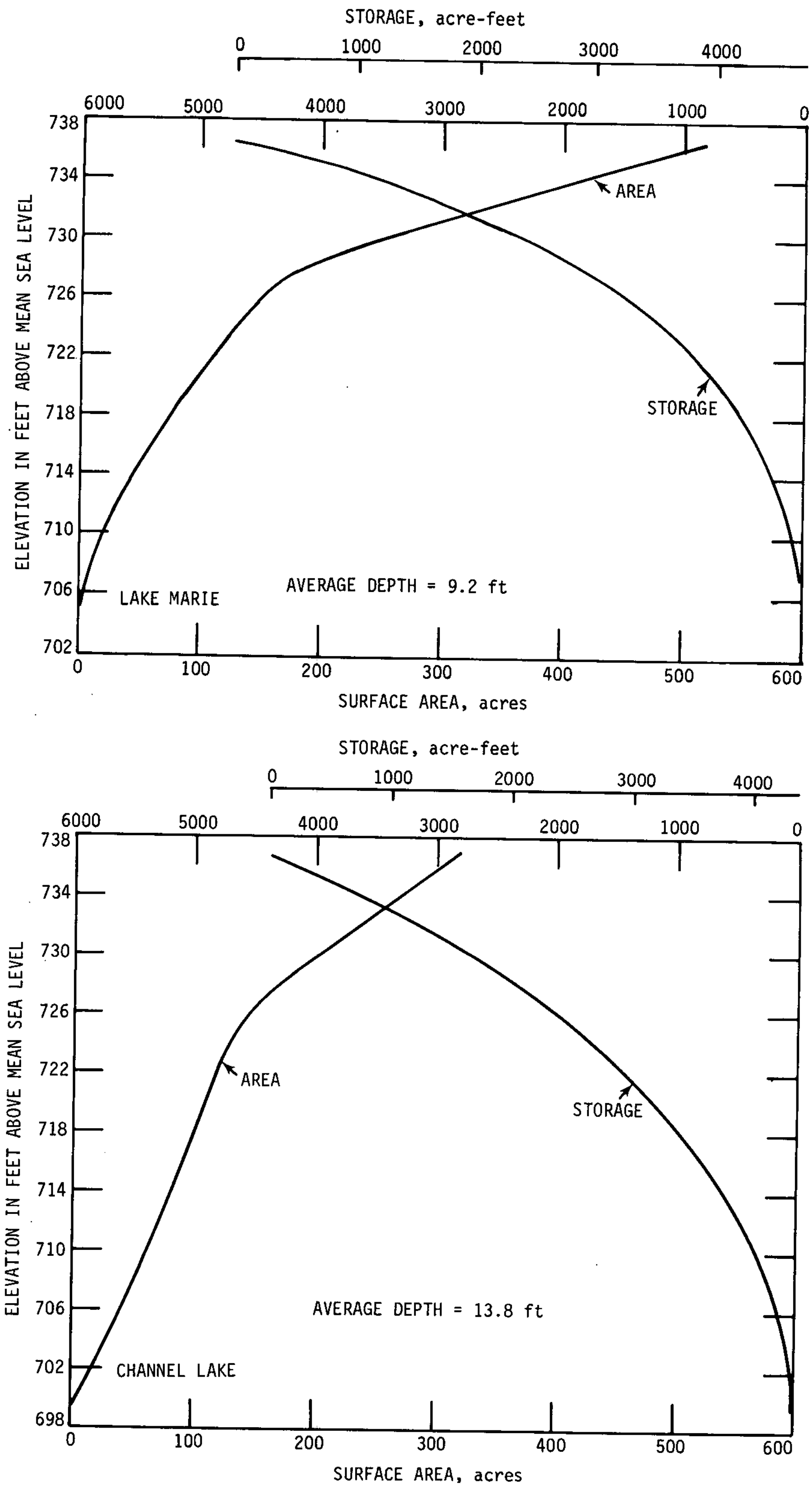


Figure 30. Depth-area and depth-storage relationships for Lake Marie and Channel Lake

The comparison of the two sets of depth measurements at the 26 cross sections indicates the variability of the lake bed and shows any sedimentation or scour pattern that might have occurred in the last 11 to 22 years. Readers must be warned that the data published by DOWR show the depth to the nearest 1 foot, and that the points where 1975 bed elevations were measured do not always coincide with those used for the DOWR maps. This necessitates some interpolation of the DOWR data to obtain an exact elevation at the points used in 1975.

The 1975 bed elevations at all the cross sections were plotted, as were the old bed profiles from the DOWR maps. Some general comments and conclusions may be drawn as to the areas and rates of sedimentation and/or erosion in the lakes. Locations of the lakes and cross sections are shown in figure 21.

Grass Lake. The 1975 bed elevations were determined only at cross section 27–28. DOWR mapped Grass Lake in 1962. A comparison of the two measurements showed that for all practical purposes bed elevations did not undergo any significant change in the last 13 years. It is probable that Grass Lake may be in a state of equilibrium for the present inflow and outflow conditions.

Lake Marie. Depths were measured at four cross sections in Lake Marie. DOWR mapped this lake in 1964. At cross section 33–35, some sedimentation did occur in the last 11 years with a maximum deposition in the order of 3 to 4 feet. Cross section 33–34 showed both deposition and some erosion. Cross section 31–33 showed some substantial amount of deposition near the middle of the lake. The maximum deposition was in the order of 5 to 6 feet. Cross section 31–32 did not show any significant variation. In general Lake Marie appeared to have lost some capacity.

Channel Lake and Lake Catherine. These are the two northernmost and deepest lakes in the Chain of Lakes system. Channel Lake is connected with Lake Marie at two locations (figure 13). Depths were measured at four cross sections. Cross section 36–37 indicated some deposition for almost the whole width of the lake except for a little erosion near the eastern shore. The maximum deposition was in the order of 2 to 3 feet. Cross section 38–39 appeared to have been eroded almost the full width of the lake. The maximum erosion was in the order of 2 to 3 feet.

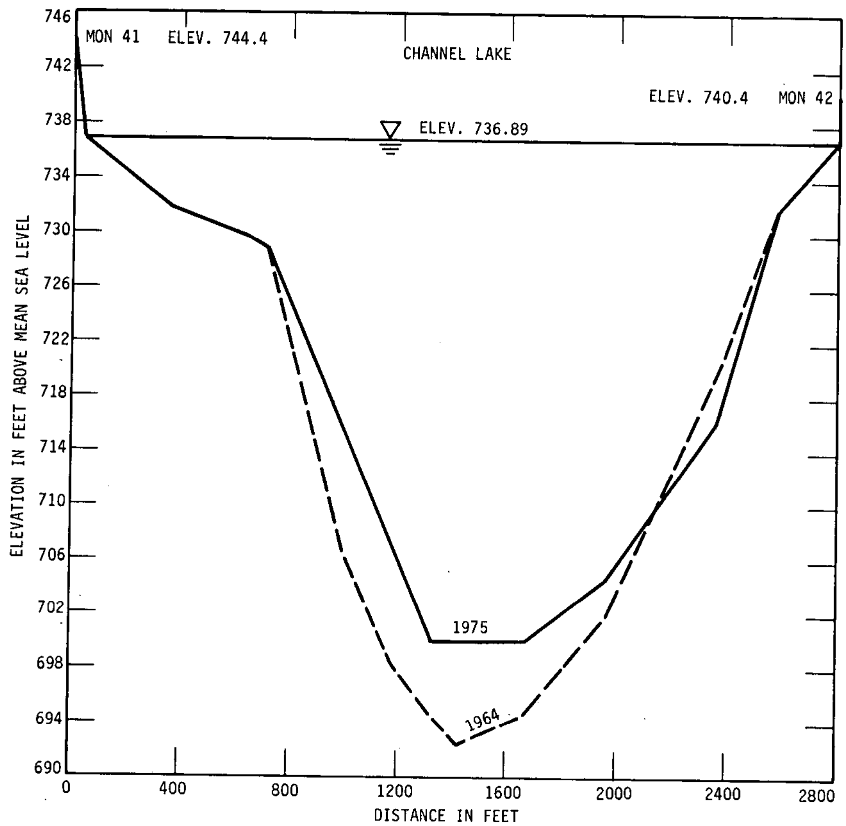
The 1975 bed elevations of the lake at cross section 41–42 were surprising, since they indicated the greatest sedimentation in any of the lakes. Figure 31 shows a comparison of the old and new cross-sectional profiles. A maximum of 7 to 8 feet of sedimentation since 1964 was evident. In contrast, cross section 40–41 in figure 31 showed some deposition near both shores and some erosion near the middle of Lake Catherine. The maximum erosion in this cross section is in the order of 3 feet.

Bluff Lake. Depths were measured at only one cross section in this lake. No measurable variations were observed. DOWR mapped this lake in 1964.

Petite Lake. For this lake also, depths were measured at only one cross section. Some sedimentation near both the east and west shores and some erosion near the middle of the lake were observed, but the net variation was negligible. DOWR mapped this lake in 1964.

Fox Lake. Depths were measured at four cross sections in this lake. DOWR mapped this lake in 1962. Cross sections 17–18 and 20–21 did not show any significant variations for this 13-year period. Figure 32 shows the old and new cross-sectional profiles between monuments 20 and 21.

The southwestern areas of Fox Lake showed some significant amount of sedimentation. Cross sections 13–16 and 15–16 showed similar variations. The maximum depth measured was in the order of 14 feet; a comparison with 1962 data indicated a maximum deposition of about 6 feet. Figure 32 shows the old and new cross-sectional profiles between monuments 15 and 16.



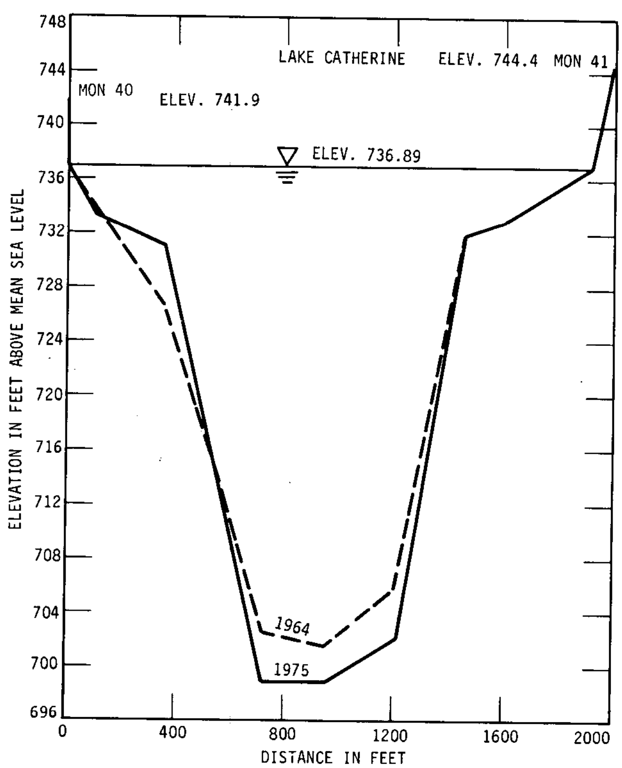


Figure 31. Typical old new cross-sectional profiles for Channel Lake and Lake Catherine

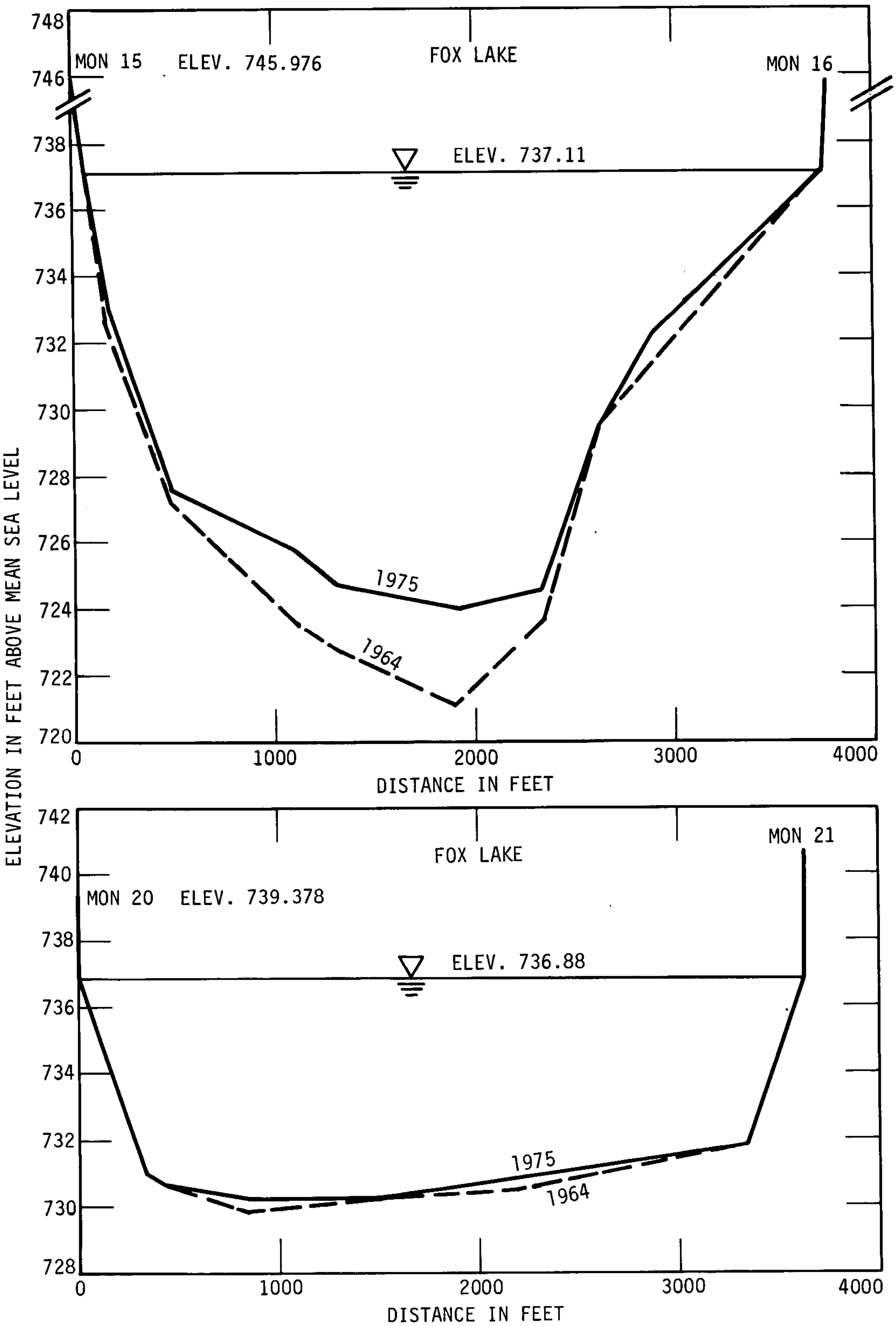


Figure 32. Old and new cross-sectional profiles for Fox Lake

Nippersink Lake. Depths were measured at three cross sections in this very shallow lake. No measurable variations in depths were observed.

Pistakee Lake. Depths were measured at seven cross sections in this lake. No significant variations were observed in most of them. However, at cross section 1–2 some sedimentation near the western half and some erosion near the eastern half were observed, though the net cross-sectional variation was negligible.

In general it can be concluded that the northernmost lakes such as Lake Marie and Channel Lake showed the maximum amount of sedimentation. The southwestern portion of Fox Lake also showed a significant amount of sedimentation. The rest of the lakes did not show much variation. Data related to depths at many more cross sections are needed to analyze precisely the extent of sedimentation and/or erosion rate in any one of the lakes.

Inflows and Outflows

One of the main objectives of this investigation was to analyze the inflows and outflows in the Fox Chain of Lakes and to quantify the water budget. As previously pointed out, data on inflows and outflows were either collected in the field or gathered from other available sources. Since in the water quality studies, the basic data for water year 1975 (September 1974 to October 1975) were utilized, it is of interest to investigate whether the 1975 water year in the Chain of Lakes region represents an average year when compared with long-term flow records.

Average Discharges

Mean monthly discharges for the Fox River gaging station at Wilmot, Wisconsin, for July 1974 through September 1975 are plotted as solid lines in figure 33. The average monthly flows for the year 1961 to 1974 are shown as dashed lines. As can be seen, the average flow of the Fox River at Wilmot for 1974–1975 was greater than the long-term average. Similar relationships for the Nippersink Creek near Spring Grove and the Fox River at McHenry Lock and Dam are shown in figures 34 and 35. These two figures also indicate that the 1974–1975 inflow and outflow at the Chain of Lakes were above average compared with long-term average flows.

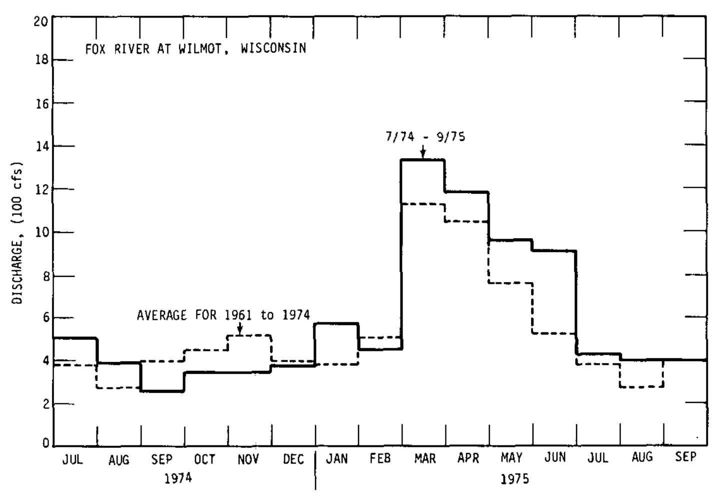


Figure 33. Monthly mean discharges for 1974–1975 and long-term mean discharges for the Fox River at Wilmot, Wisconsin

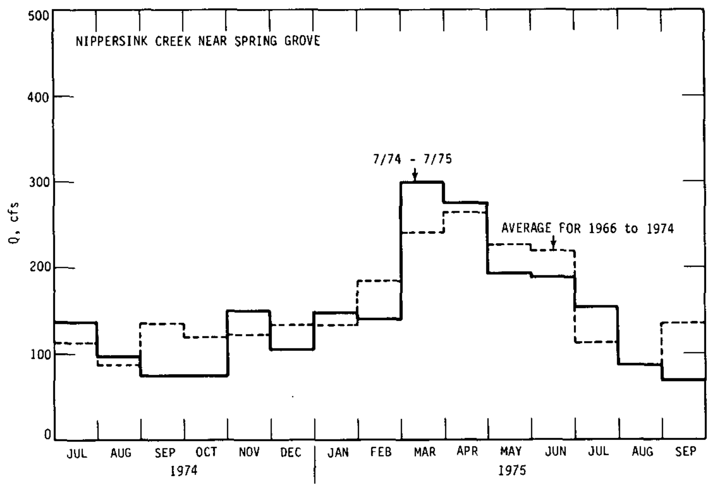


Figure 34. Monthly mean discharges for 1974-1975 and long-term mean discharges for Nippersink Creek near Spring Grove

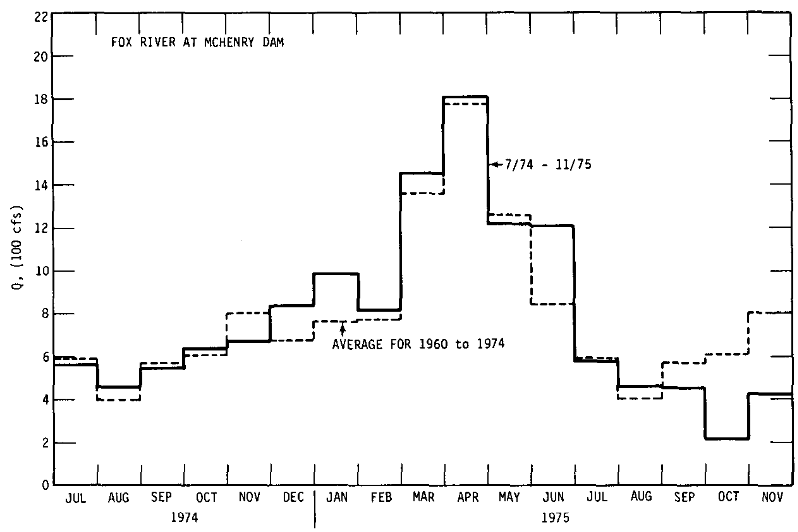


Figure 35. Monthly mean discharges for 1974-1975 and long-term discharges for Fox River at McHenry Dam

Figure 36 shows the flow duration curves for the three permanent gaging stations in the Chain of Lakes region: Fox River at Wilmot, Wisconsin; Nippersink Creek near Spring Grove; and Fox River at Algonquin. Figure 36 indicates that the frequency of occurrence of the mean flow for the 1975 water year for the Fox River at Wilmot is 22 percent; for Nippersink Creek at Spring Grove, 34 percent; and for the Fox River at Algonquin, 24 percent. The frequency of occurrence indicates the percent of time the flow either exceeded or equaled this value. The frequency of occurrence of the long-term average flows is also indicated on figure 36. For two stations, the 1975 average flow occurred less frequently than the long-term average discharges.

Water Budget

Figure 37 shows the inflows for the Fox River at Wilmot and for Nippersink Creek near Spring Grove for October 1974 through September 1975. The maximum flows occur in

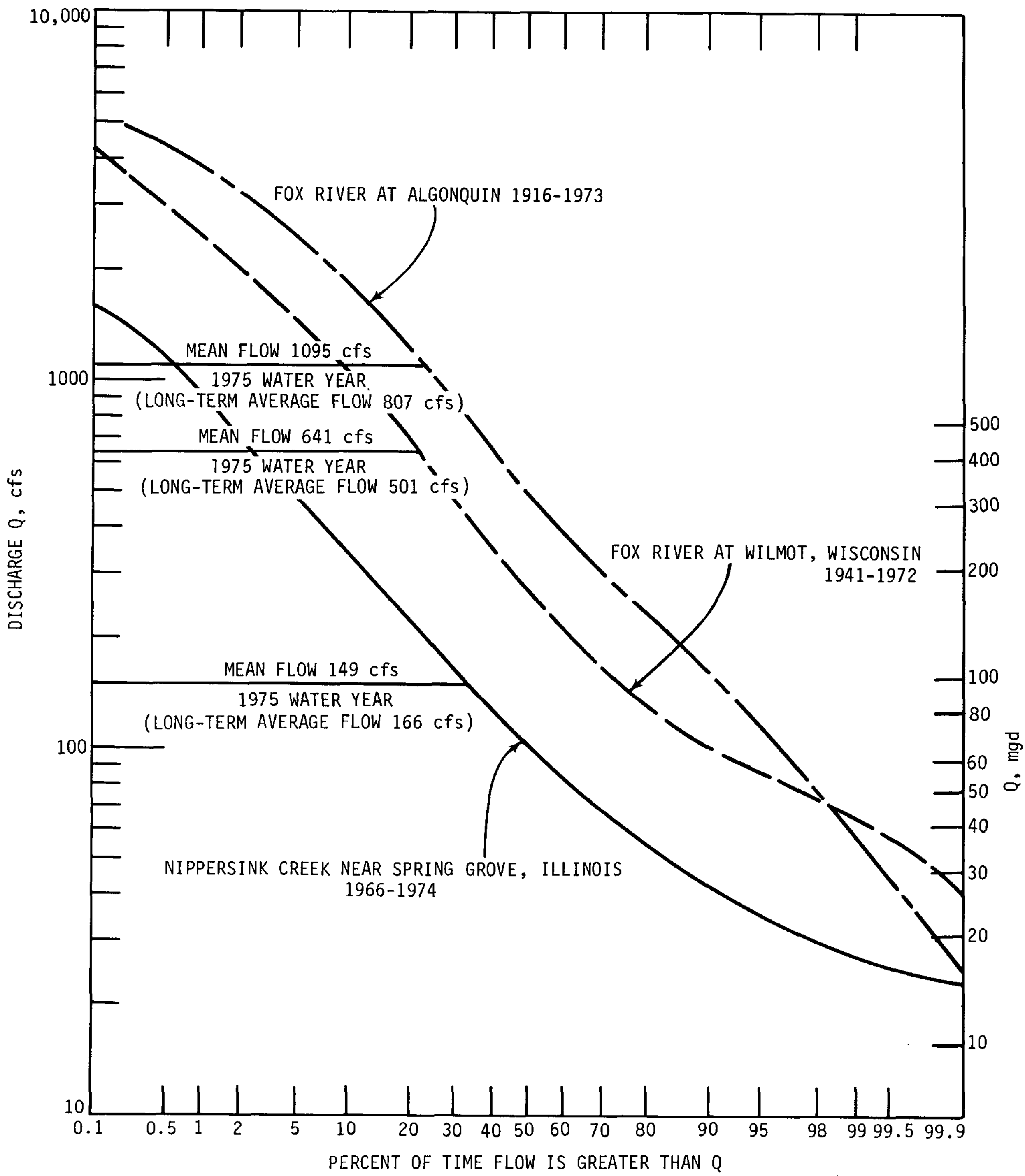


Figure 36. Flow duration curves for gaging stations in Chain of Lakes region

the months of March through June. Figure 38 shows the inflow record for Sequoit Creek near Antioch, Squaw Creek near Fox Lake, and Lily Lake Drain near Johnsburg for the months of January through November 1975, and the precipitation for July 1974 through

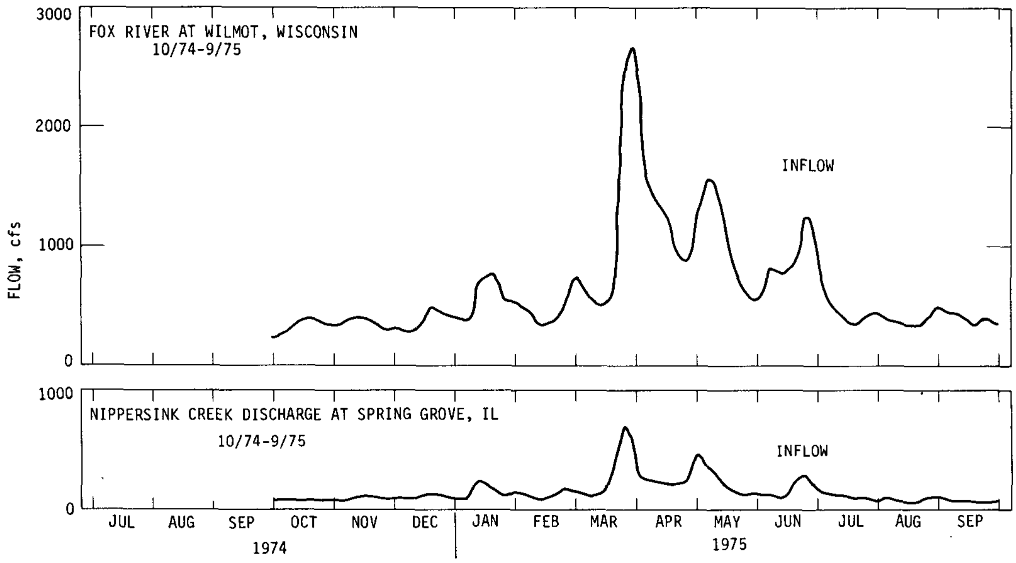


Figure 37. Major inflows to Chain of Lakes

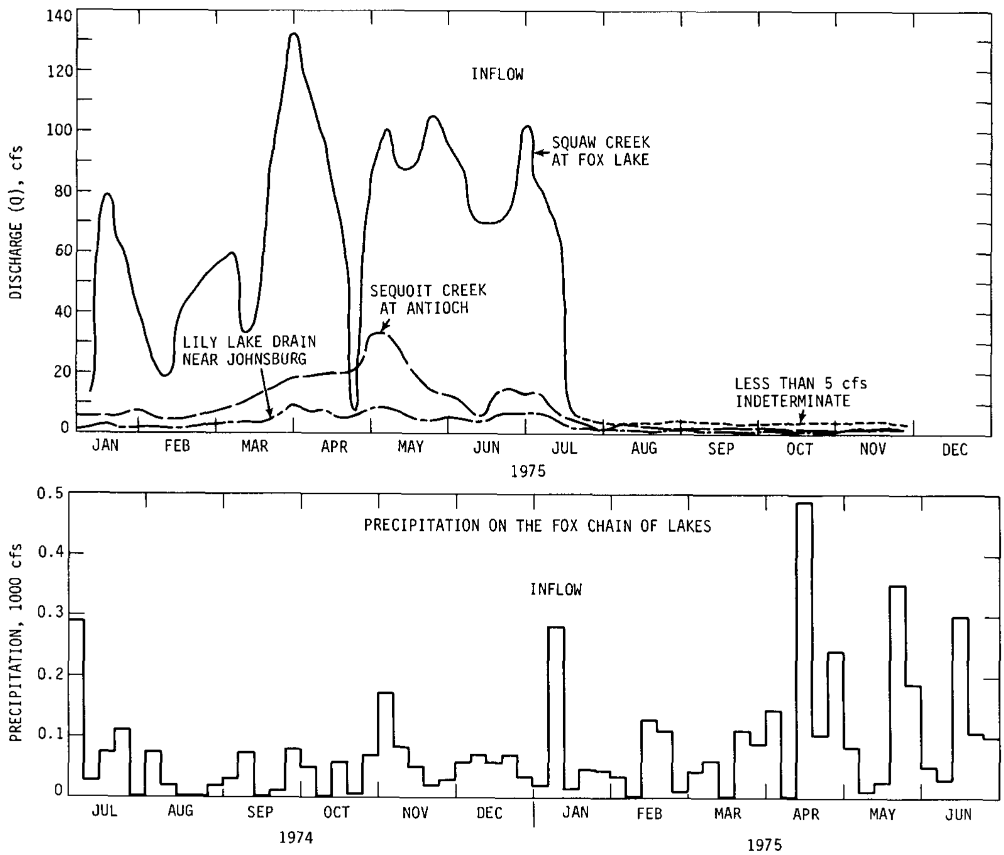


Figure 38. Minor tributary inflows and precipitation for Chain of Lakes

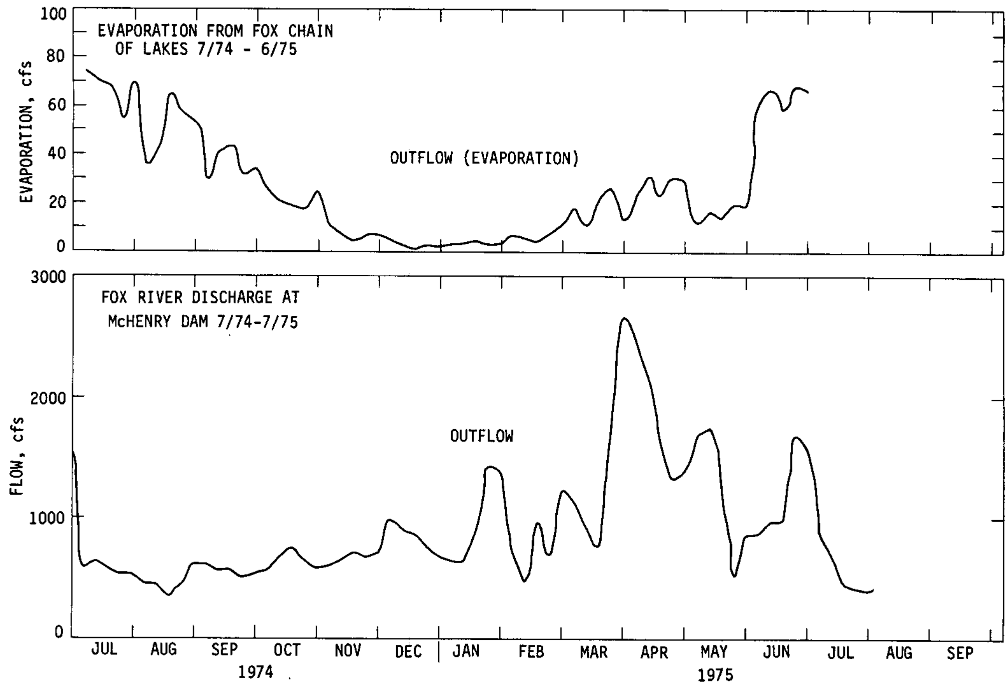


Figure 39. Evaporation losses for lakes and outflow at McHenry Dam

June 1975. The maximum inflow from Sequoit Creek occurs in April and May, and for Squaw Creek in March and April. Maximum precipitation occurs in April through June. Figure 39 shows the evaporation for the months of July 1974 through June 1975 and the outflow records at McHenry Lock and Dam from July 1974 through July 1975. The evaporation was computed following the procedure suggested by Roberts and Stall (1967). Maximum evaporation occurred in June through August and the maximum outflow occurred in April through June.

All of the inflow and outflow records shown in figures 37-39 were combined and tabulated, and a weekly water budget computation was made. Table 11 shows typical water budget computations for August 1974 and April 1975. For both of these months total outflow exceeded the inflow. However, it must be pointed out that the outflows and inflows considered here were the ones that could be measured or computed. There are some other minor inflows from the surrounding areas which could not be accounted for in this computation. This might explain the reasons for differences in inflow and outflow values in table 11. Table 12 shows the monthly values of all the inflows and outflows for July 1974 through June 1975. As can be seen, the difference between inflows and outflows fluctuates from positive to negative values.

Interflow between Lakes

Data on interflow between lakes at selected locations were collected six times from December 1974 to November 1975. On a few occasions data could not be collected from all selected locations because of ice cover or other unavoidable circumstances.

Figure 40 shows the inflow and interflow between the lakes for December 11 and 12, 1974, and May 15, 1975. On the December dates the general flow pattern was from north to

Table 11. Fox Chain of Lakes Inflow and Outflow for August 1974 and April 1975
(Flows in cfs)

Time period	Inflow: Wilmot	Inflow: Nippersink Creek	Inflow: Squaw Creek*	Inflow: Lily Lake Drain*	Inflow: Sequoit Creek*	Inflow: Precipitation	Inflow: Total	Outflow: McHenry Dam	Outflow: Evaporation	Outflow: Total	Inflow minus outflow
August 1974											
1-6	380	105				75	560	455	36	491	
7-12	330	85				20	425	445	43	488	
13-18	435	97				2	534	365	65	430	
19-24	425	90				1	516	435	57	492	
25-31	370	88				18	476	618	53	671	
Monthly average	388	93				23	502	464	50	514	-12
April 1975											
1-6	1530	241	114	7	19	146	2057	2440	24	2464	-407
7-12	1345	221	100	8	19	0	1693	2160	31	2191	-498
13-18	1040	213	73	5	19	489	1839	1715	24	1739	100
19-24	863	222	7	5	20	104	1221	1330	30	1360	-139
25-30	1249	461	85	8	32	243	2078	1380	30	1410	668
Monthly average	1205	272	76	7	22	196	1778	1805	28	1833	-55

*Inflow data unavailable for Squaw Creek, Sequoit Creek, and Lily Lake Drain for August 1974

south, i.e., water was moving from Grass Lake and Lake Marie into Nippersink, Fox, and Pistakee Lakes and finally to the Fox River in the downstream direction. On May 15 the basic flow pattern was also in the downstream direction with the flow from Lake Marie moving into Grass Lake. However, on May 21 when circulation data were collected between Stations 31 and 32 (figure 21) in Lake Marie, the water was flowing from Grass Lake into Lake Marie.

Figure 41 shows the inflow and interflow between lakes for July 14 and 15, 1975. Basically, the flow was moving in the downstream direction except on July 15, 1975, when water was moving from Grass Lake to Lake Marie. On July 18 the water was also moving from Grass Lake to Lake Marie.

Table 12. Water Budget for Fox Chain of Lakes, July 1974 through June 1975
(Flows in cfs)

Date	*Total inflows*	*Total outflows*	*Inflow minus outflow*
July 1974	772	660	112
August	502	514	-12
September	367	590	-223
October	461	661	-199
November	518	674	-156
December	540	839	-298
January 1975	849	988	-139
February	710	822	-112
March	1786	1468	318
April	1778	1833	-55
May	1431	1236	195
June	1316	1274	42
Average total for 1-year period	919	963	-44
	665,000 acre-feet	697,000 acre-feet	-32,000 acre-feet

Figure 42 shows the inflow and interflow between lakes for October 7 and 10, 1975. On both of these occasions, the flows under the Highway 173 bridges between Channel Lake and Lake Marie were measured with floats and vanes. Surprisingly, water was moving into Channel Lake under the east bridge on both of these days. However, on October 7 water was moving into Lake Marie under the west bridge. On October 10 water was moving into Channel Lake, and the wind was blowing

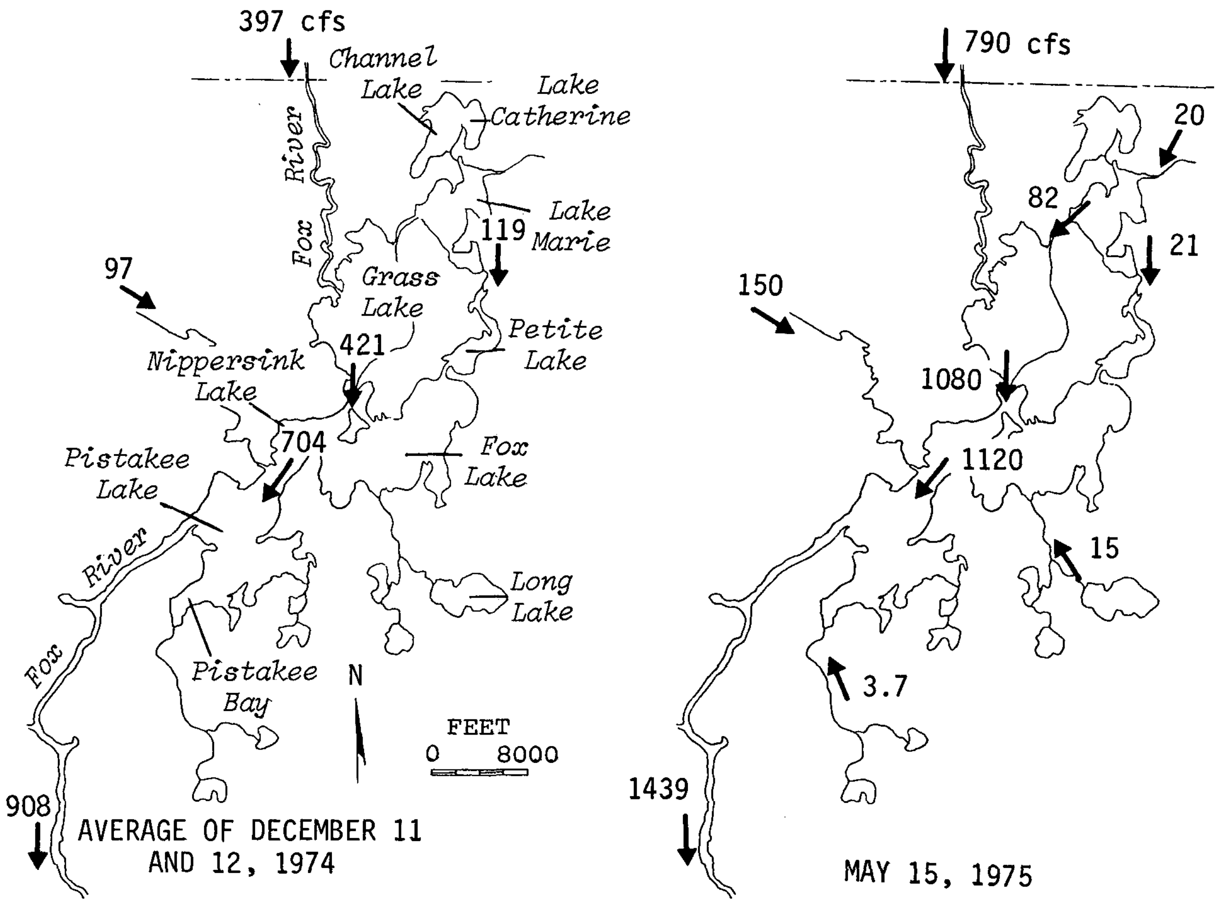


Figure 40. Inflow and interflow between lakes for December and May dates

moderately from the south or southeast at 7.5 mph. Discharge was measured between Grass Lake and Lake Marie on October 8. On this date part of the water in the cross section was flowing in one direction and part in the opposite direction, with a net flow of 1.9 cfs into Lake Marie.

The other surprising interflow data on October 7 was the 34 cfs flow into Nippersink Lake from Pistakee Lake. However, on October 9 the net outflow from Nippersink Lake was into Pistakee Lake. It appears that when the inflows are low (e.g., the 159 cfs flow at Wilmot on October 7 has a frequency of occurrence of 72 percent, figure 36) the water within the lakes wanders around and its movement may to some extent depend on wind velocity and direction. Circulation data indicated that on October 7 and 10 water was basically moving into Lake Marie from Grass Lake.

Figure 43 shows the inflow and interflow between lakes on October 23 and November 18, 1975. On both of these days, there was a net outflow from Lake Marie into Channel Lake. The data from the four fall days (October 7, 10, 23, and November 18) showing a net outflow from Lake Marie into Channel Lake came as a surprise. It appears that strong winds move water from Lake Marie toward the northeast, piling it up in Channel Lake and Lake Catherine. This excess water will of course return to Lake Marie and down to the Fox River when the wind subsides.

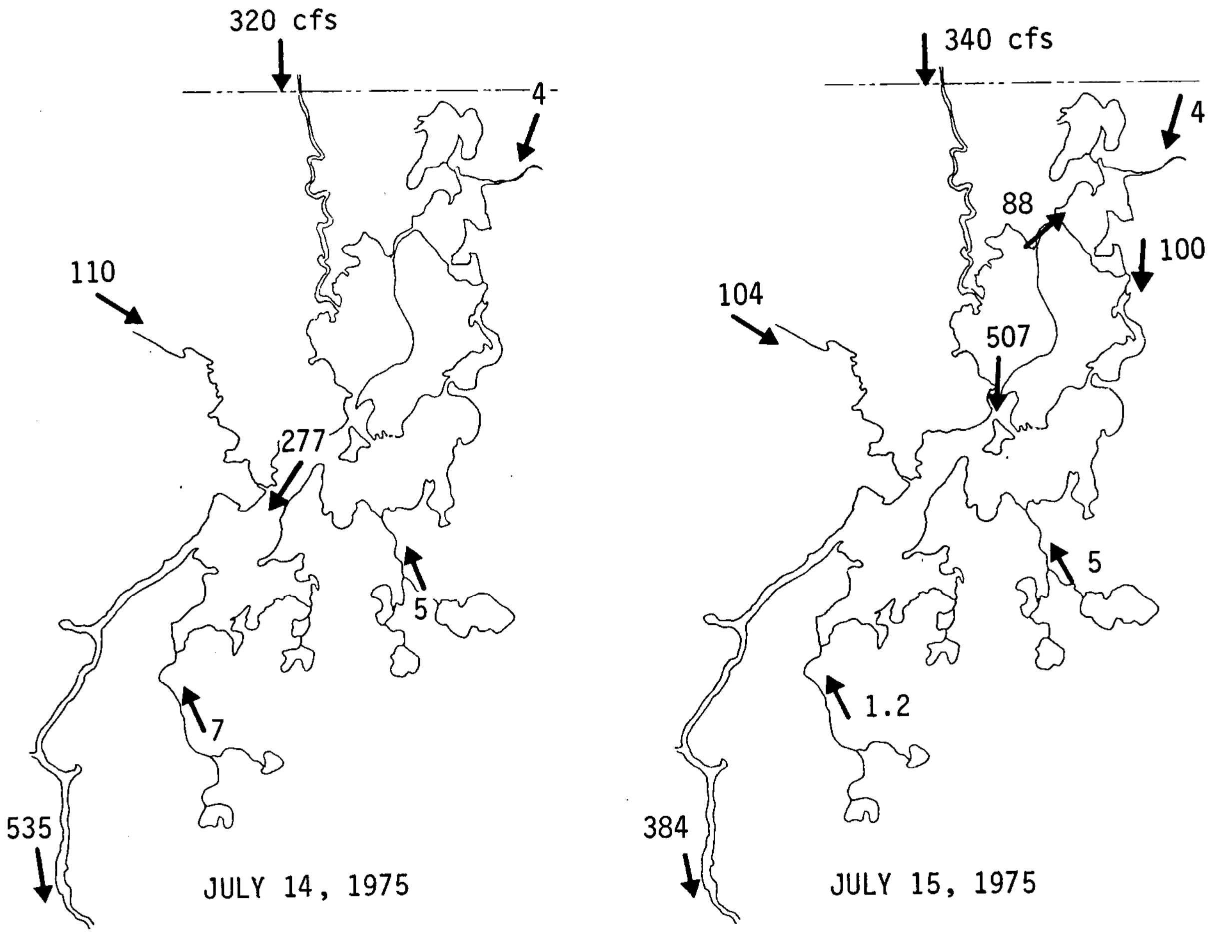


Figure 41. Inflow and interflow between lakes for July dates

Table 13. Flow Velocity under Highway 173 Bridges

Date	East bridge Flow (cfs)	East bridge Average velocity (fps)	East bridge Direction	West bridge Flow (cfs)	West bridge Average velocity (fps)	West bridge Direction
October 7, 1975	77	0.127	Into Channel Lake	9.2	0.111	Into Lake Marie
October 10, 1975	108	0.178	Into Channel Lake	2.0	0.033	Into Channel Lake
October 23, 1975	33	0.052	Into Channel Lake	7.0	0.117	Into Lake Marie
November 18, 1975	113	0.182	Into Channel Lake	8.0	0.125	Into Channel Lake

Some computations were made to estimate the flow velocity under the Highway 173 bridges at the time the discharge was estimated with floats and vanes, as shown in table 13. The maximum average velocity measured is on the order of 0.182 fps. Although this average velocity is low, it is capable of transporting fine materials in suspension. Moreover, these velocities were measured on only four days in a year. Flow may increase under these bridges at other times of the year with higher velocities.

Circulation data (to be presented in a later section) and the interflow data presented in figures 40–43 show that in general water moves from Grass Lake into Lake Marie, but that water also moves into Channel Lake from Lake Marie at least when the data were collected. The observed peculiar flow patterns may be a clue to the heavy sedimentation rate observed in

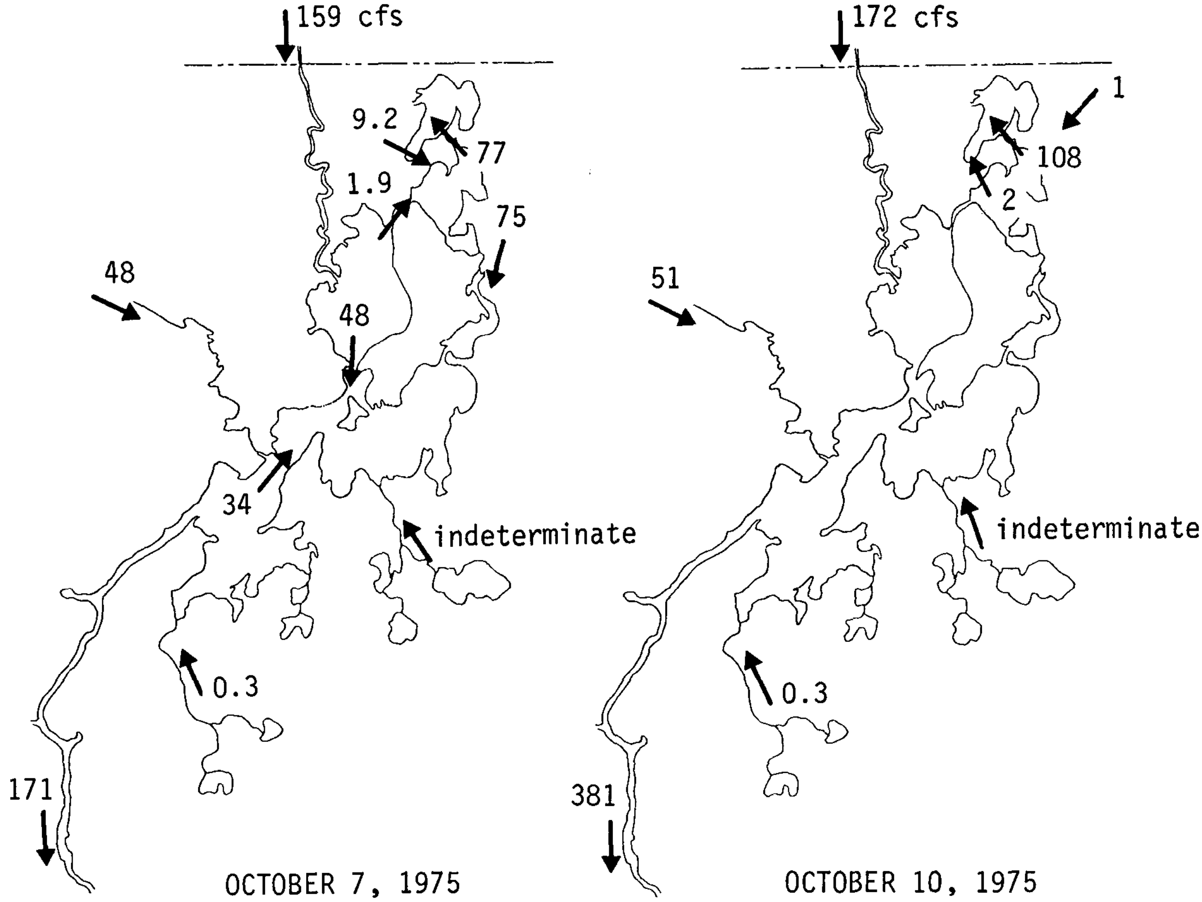


Figure 42. Inflow and interflow batween lakes for early October dates

Channel Lake, as was shown in figure 31. It appears that the water moving into Channel Lake carries some fine materials. These may have originated from Fox River water moving through Grass Lake to Lake Marie and ultimately to Channel Lake. Or, part of the load may come from bed erosion at the southern part of Channel Lake in the vicinity of cross section 38-39, the materials being finally deposited in the deeper part of Channel Lake near cross section 41-42 (figures 21 and 31).

In general, it appears that when inflow rates are high, water generally moves in a southwesterly direction and eventually flows down the Fox River. However, during low inflow rates, the general movement of water within the lakes is erratic, and the water wanders to and fro depending on wind velocity and direction until it finally flows downstream through the Fox River. This complex nature of the flow makes it difficult to establish a reliable hydraulic detention time in the Chain of Lakes.

Theoretical Hydraulic Detection Times

The theoretical detention times in the lakes were computed for various inflow and outflow characteristics. For this computation, lake levels were assumed to be at 736.5 feet above msl. Initially, it was assumed that the lakes were homogenous, that total storage capacities of the lakes were effective in retaining the inflows, and that a somewhat uniform

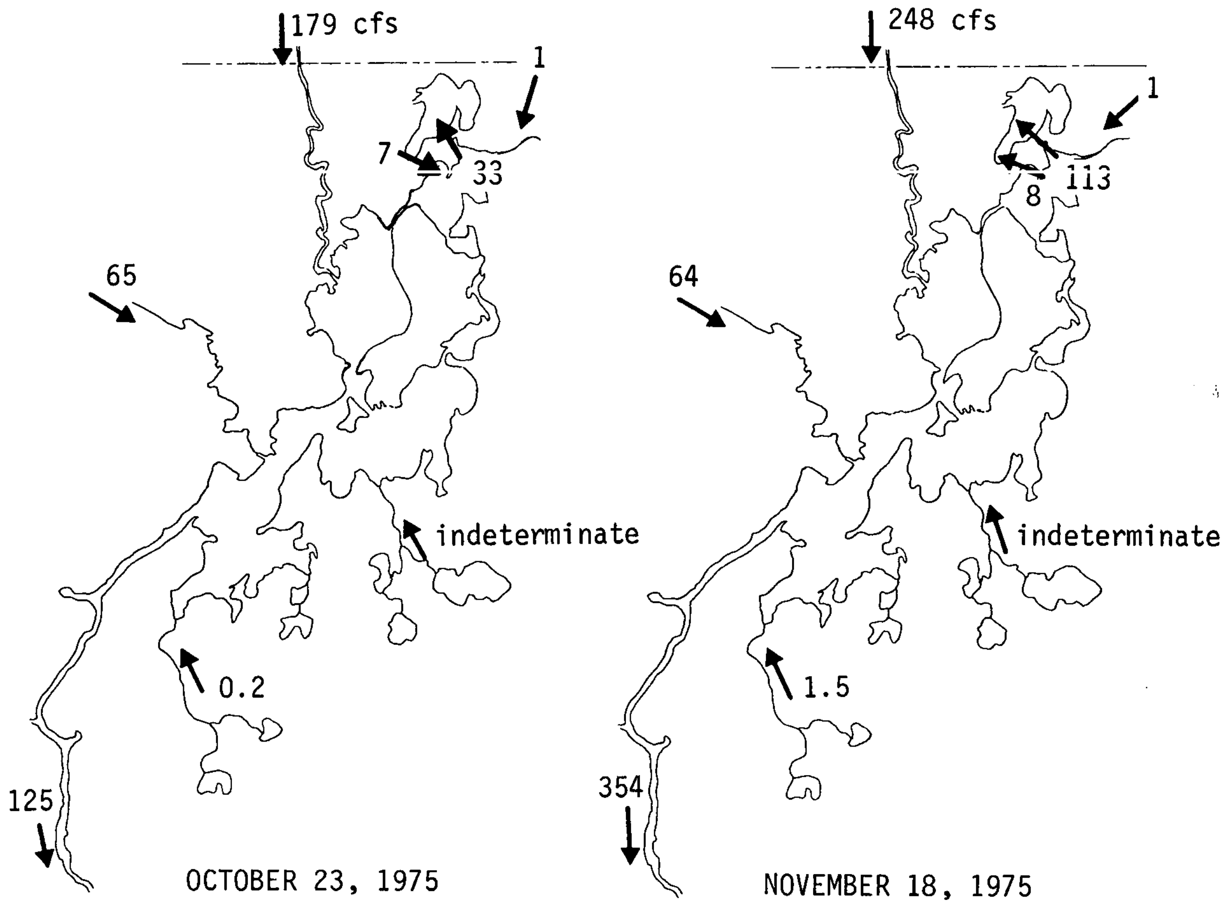


Figure 43. Inflow and interflow between lakes for October 23 and November 18

type of flow existed throughout the time it took the inflow to pass through the lakes. With these gross assumptions, computations were made to estimate the detention times in all the lakes.

Detention times for flows of July 14 and 15, 1975 (figure 41), and the associated assumptions that were used in the computations are given in table 14.

Inflow data into Channel Lake and Lake Catherine were collected on a few occasions (figures 42 and 43), and the detention times for these two lakes on October 7, 1975 (figure 42), also are given in table 14.

Average Detention Time in Chain of Lakes. The following computation was made to get some idea about an average detention time in all of the Chain of Lakes. This average discharge is obtained by multiplying with the proportional factor of the drainage areas at McHenry Lock and Dam (1253 square miles) and the Fox River near the outlet of Pistakee Lake (1207 square miles). The average outflow at McHenry Lock and Dam for the years 1960 to 1974 is 869 cfs. Therefore, the average outflow at the mouth of the Fox River near the outlet of Pistakee Lake is estimated to be (869 x 1207)/(1253) = 837 cfs. It is assumed that the end product of all the inflows, precipitation, evaporation, effluents, and influents is an average outflow of 837 cfs at Johnsburg. The total storage capacity of all the lakes below a water surface elevation of 736.5 feet above msl is 38,718 acre-feet. Therefore, the average detention time is 23.32 days. If we assume that neither all the lakes nor their full capacities are effective in retaining this flow, then the detention time becomes 10.79 days (table 14).

Table 14. Detention Times in the Chain of Lakes for Given Flow Periods

Lake	*Storage (ac-ft)*	*Q (cfs)*	*Special provision*	*Detention time (days)*	*Remarks*
July 14 and 15, 1975					
Grass	3,983	468.4	Full capacity	4.3	
Grass	2,390	468.4	60 percent capacity	2.6	
Marie	4,742	92	Full capacity	26	
Marie	3,685	88	Western part of lake only	21.1	
Bluff	965	100	Full capacity	4.9	
Bluff		100	Upper 20 ft of lake	4.3	
Petite	1,272	100	Full capacity	6.4	
Fox	9,622	105	Full capacity	48.5	Not reasonable
Fox			50 percent capacity	97	Not reasonable
Nippersink	1,772	607	Full capacity	1.5	
Pistakee	3,016	455	Northwest part of lake	3.7	
Pistakee			Full capacity	10.6	
October 7, 1975					
Channel	4,368	77	Full capacity	28.6	
Catherine	3,540	77	Upper 20 ft only	44.7	Not reasonable
Average flow of 1960-1974					
All lakes	38,718	837	Full capacity	23.3	
Assumed effective storage					
Grass	2,390		60 percent capacity		
Marie	3,685		Western part		
Bluff	965		Full capacity		
Petite	1,272		Full capacity		
Fox	4,811		50 percent capacity		
Nippersink	1,772		Full capacity		
Pistakee	3,016		Northwest part only		
Total effective storage	17,911	837		10.8	
July 14 and 15, 1975					
Effective storage	17,911	459	Same special provisions as used above	20.4	
October 23, 1975					
Effective storage	17,911	125		75	Not reasonable
7 day 10 year low flow					
Effective storage	17,911	39.4		229	Not reasonable

Detention Time for Low Streamflow Regimen. In order to estimate a detention time in all of the lakes, the low flow value of 39.4 cfs at the outlet of Pistakee Lake was taken as representative of 7-day 10-year low flows (Singh and Stall, 1973). With this flow, detention time was 229.19 days.

It appears that during low flows, a considerable amount of time is needed to pass the flow through all of the lakes. One must understand that the flow will not, of course, stay constant for 229 days, so this estimate of detention time is not reasonable (nor are other high numbers for the specific lakes as noted in table 14). However, an estimate of 229 days points out the fact that during low flows, the water is going to wander around in the lakes for a long time.

The computed detention times are summarized in table 14. Before utilizing any value from the table, the reader must understand the limitations and assumptions that were used in the computations.

Seasonal Variation of Detention Time. Seasonal variations of the detention times were computed using the average outflow at Johnsburg for the water year 1975 (October 1974

Table 15. Theoretical Detention Time in the Chain of Lakes with Four Capacity Assumptions

(Detention times in days)

Date	*Assumption 1*	*Assumption 2*	*Assumption 3*	*Assumption 4*	*Flow at Johnsburg (cfs)*
October 1974	31.8	30.0	14.7	14.4	613
November	30.4	28.7	14.1	13.7	642
December	24.4	23.0	11.3	11.0	800
January 1975	20.6	19.4	9.5	9.3	949
February	24.9	23.5	11.5	11.2	784
March	14.0	13.2	6.5	6.3	1397
April	11.2	10.6	5.2	5.1	1739
May	16.6	15.7	7.7	7.5	1175
June	16.7	15.8	7.7	7.5	1166
July	34.9	32.9	16.1	15.7	560
August	44.2	41.6	20.4	19.9	442
September	45.4	42.8	21.0	20.5	430

1 Full capacity for all lakes

2 All lakes (only upper 20 ft of water is effective)
Storage capacity = 36,512 ac-ft

3 Grass Lake — 60 percent capacity
Lake Marie — Western part
Bluff Lake — Full capacity
Petite Lake — Full capacity
Fox Lake — 50 percent capacity
Nippersink Lake — Full capacity
Pistakee Lake — Northwestern part
Storage capacity = 17,911 ac-ft

4 Grass Lake — 60 percent capacity
Lake Marie — Western part, upper 20 ft
Bluff Lake — Upper 20 ft
Petite Lake — Upper 20 ft
Fox Lake — 50 percent capacity
Nippersink Lake — Full capacity
Pistakee Lake — Northwestern part
Storage capacity = 17,456 ac-ft

Note: Water surface at 736.5 above msl for all assumptions

through September 1975). The computed detention times, in days, with different assumptions are listed in table 15 and shown in figure 44. As can be seen, detention time can vary from 11 days in April to 45 days in September for full capacity of all the lakes. With assumption number 4 (table 15), the detention time varies from 5 days in April to about 21 days in September.

Circulation Patterns

Movement of water in any closed or open-ended body of water is induced by a number of natural or man-made factors. Inflow or outflow characteristics, temporary detention and release of water, winds blowing over the water surface, density differential due to temperature variation, and a host of other factors can start and sustain a pattern of water movement inside any lake or reservoir. When the quantity of inflow into a lake is very small compared with the total effective storage of the reservoir, the measurable pattern of water movement inside the lake will in all probability be affected by external forces such as wind blowing over the water surface.

Basic inflow to the Chain is from the Fox River and Nippersink Creek, and water in the lakes moves in a south, southwesterly direction. However, in some of the connecting areas water moves in the northeast or northerly direction (figures 40 to 43). The localized flow in any one of the lakes is sometimes variable and the water may move in a completely opposite direction from the normal flow. Moreover, except for the two northernmost lakes, all of the lakes in the Chain are basically shallow. Wind blowing over the water surface will

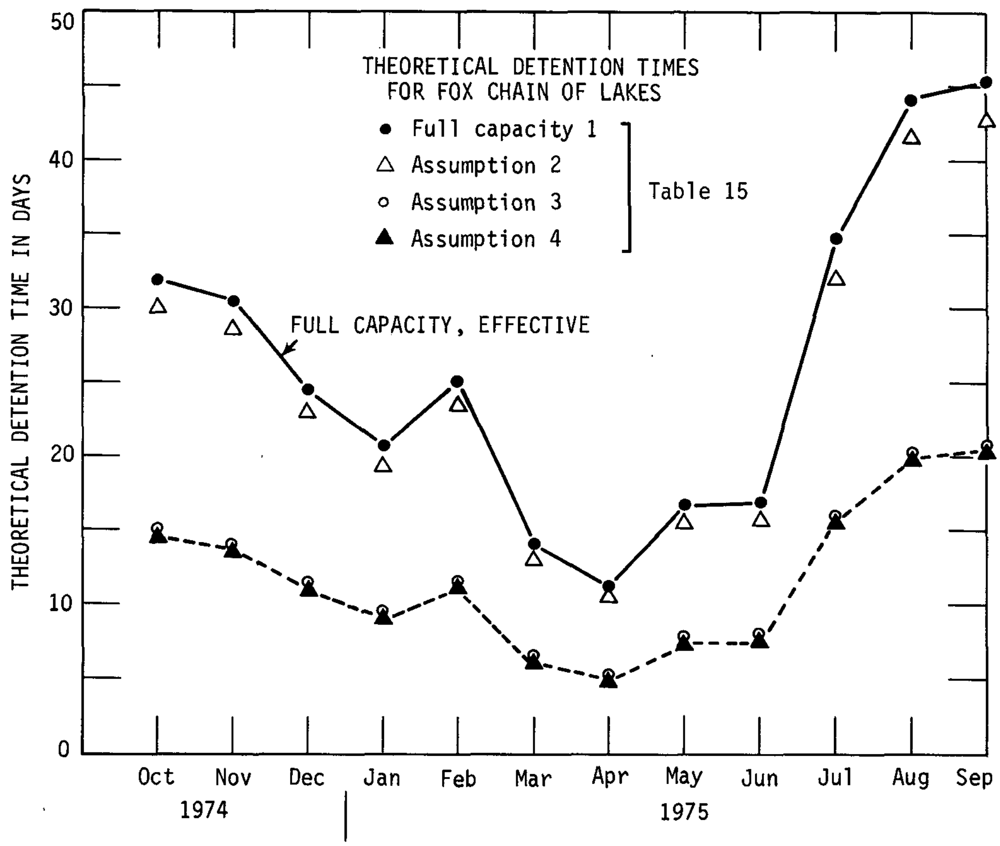


Figure 44. Theoretical detention times for Fox Chain of Lakes

transmit momentum to the water and the water will start to move with the wind. If the direction of the wind remains constant, this movement of water will attain a specific pattern and may continue for some time. Movement of water generated by the wind blowing over the water surface is called the *wind-generated circulation pattern.* The Chain of Lakes is affected by the wind, and wind-generated circulation patterns sometimes govern the interflow between some of the lakes at some time of the year.

Theory. Various investigators have studied the wind-generated circulation patterns in lakes. Most of these studies were confined to the theoretical or analytical analysis with some computer simulation to depict the water movement in the lakes. Ligget and Hadijtheodorou (1969) presented a mathematical simulation to study the wind-generated circulation patterns in shallow homogeneous lakes. They developed Calcomp plots of circulation patterns in an idealized rectangular lake. The effects of bottom topography and lake configuration were incorporated into their analysis. Ligget (1970) presented a cell method to compute three-dimensional wind-driven circulation in a homogeneous lake. This method allowed an arbitrary variation of surface wind stress and the eddy viscosities could be varied from cell to cell. A mathematical model to compute the circulation pattern in stratified lakes was also presented by Liggett and Lee (1971). Mathematical formulations of wind-generated circulation patterns were also studied by Murty and Rao (1970), Cheng and Tung (1970), Bonham-Carter and Thomas (1973), and a number of other investigators.

Laboratory investigations of circulation patterns in lakes were performed by Li et al. (1975) and Hoopes et al. (1973). These laboratory experiments shed some light on the basic mechanism of circulation patterns and enabled the researcher to study the effects of various input parameters on the development of the pattern of water movement.

Gedney and Lick (1971) computed the steady-state wind-generated circulation patterns in Lake Erie and compared their results with current meter velocities. Some current data were collected from Lake Michigan by the Federal Water Pollution Control Administration (1967). Plots of circulation patterns for varying wind conditions were shown. Drogues and current meters were used to collect the circulation data.

Wind-Velocity Distribution. The wind-velocity distribution above the ground or water surface was found to be logarithmic by some investigators. The general equation is a form of the logarithmic distribution given below:

$$V_z = (V_* / \kappa) \ln (z/z_0)$$

where V_* is the shear velocity, κ is the Von Karman's constant generally taken to be 0.4, z_0 is the air roughness height. Various researchers have postulated different values of roughness height. Shemdin (1973) found the value of z_o to be 0.1 mm. Both Ruggles (1969) and Lai and Shemdin (1971) indicated that a wind-velocity profile follows a logarithmic distribution, but De Leonibus (1971) indicated that in all his field measurements, he rarely came across a logarithmic wind-velocity profile.

Surface Drift. Surface drift can be defined as the movement of the uppermost layer of water as a result of wind shear or blowing wind. However, the surface layer of water can also move because of a seiche, boatgenerated disturbances, gradient currents, and differential in temperature between air and water. Many investigators have studied the surface drift phenomena and have indicated the relative magnitude of the surface current compared with wind velocity.

The ratio between surface current and wind velocity is termed the *wind factor.* Haines and Bryson (1961) have described most of the important contributions on this aspect of wind-generated current.

Ekman *(in* Haines and Bryson, 1961) has shown that

$$\text{wind factor} = V_s / V_w = (0.0127)/(\sin \phi)^{1/2}$$

where V_s is the surface drift, V_w is the wind velocity, and ϕ is the latitude. According to this relation, the surface drift is constant for any fixed latitude. For the Fox Chain of Lakes the surface drift is 0.0155 for a latitude of 42° 25 'N. Rossby and Montgomery *(in* Haines and Bryson, 1961) have indicated that wind factor decreases with high latitudes and wind velocity and ranged from 2.3 to 3.2 percent. A mean wind factor of 2 percent was found to be valid in Lake Erie by Olson *(in* Haines and Bryson, 1961).

Keulegan (1951) from experiments in a laboratory tank has proposed that

$$V_s / V_w = 0.033$$

for Reynolds number greater than about 1000. Van Doren (1953), using data from a pond, obtained similar results between surface drift and wind velocity. Both of these investigators also have shown that surface drift is independent of waves. Haines and Bryson (1961) analyzed wind and surface drift data from Lake Mendota and have shown that at a wind speed below about 19 fps, the average wind factor was about 1.3 percent. However, above this wind velocity, which they termed the critical wind velocity, the wind factor assumed a nonlinear distribution with some reduction in absolute magnitude. In a recent article, Shemdin (1973) observed that the surface drift can be taken as about 3.4 percent of the wind velocity measured at a height of 32.8 feet above the water surface.

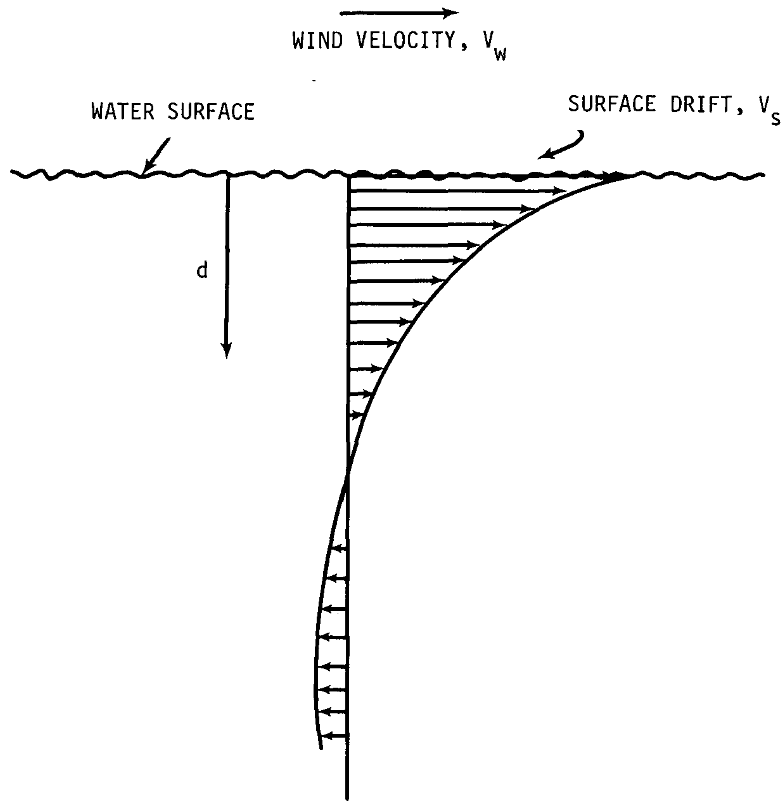


Figure 45. Schematic of surface drift and flow profile

Wind-Generated Current Profiles in Water. The surface drift that is produced by wind is generally accompanied by movement of water below the water surface. The magnitude of flow velocity near the water surface is high compared with the flow velocity at a given depth below the surface. This reduction of flow velocity is very sharp and may attain a value of zero at some depth after which a return flow may exist. Figure 45 shows a sketch of the surface drift and flow profile below the water surface for a hypothetical wind condition.

Bye (1965) collected field data on wind-generated current profiles from Lough Neagh in Northern Ireland. The total depth of water was 32.8 feet, wind velocity varied from about 23 to 26 fps, and the fetch was about 9 miles. He used weighted floats about 1-inch square in lengths varying from 1 inch to about 3 feet to measure the velocity of water at different depths. Using the concept of zero resultant drag on each float, he deduced the velocity profiles from a series of floats of different lengths. Bye's data indicated that the velocity distribution was logarithmic and could be expressed by the relationship given below:

$$V_s - V_d = [V_* / k] \ln (z/z_{ow})$$

where V_d is the velocity of water at a depth equal to d, V_* is the water shear velocity, and Z_{ow} is the water roughness parameter. Shemdin (1973) collected extensive data on current profiles from a laboratory facility and also concluded that the current profile can be approximated by a logarithmic distribution similar to this equation. Shemdin also observed some return flow in their wind-water tunnel facility similar to the sketch shown in figure 45.

General Circulation Patterns in the Chain of Lakes. As mentioned earlier, extensive localized circulation data were collected from different lakes in the Chain. These data are presented as a group for each site. Figure 46 shows the locations of all the areas where specific circulation data were collected.

Fox River Entrance to Grass Lake. The flow pattern data at this location were collected at various times during 1975. Since only one semipermanent station could be located here (station 30, figure 46), stadia distance and bearing angle were used to locate the float in the water. Figure 47 shows the circulation pattern for May 23, 1975. Here the flow pattern varies as the direction of the wind changes and a strong effect of wind is clearly visible. Figure 48 shows the flow patterns for July 15 and October 8, 1975. Because the morning of October 8 was almost calm, some flow pattern data were collected near the middle of the lake. It appears that during such a calm period, the Fox River water flows downstream toward the Grass Lake Bridge outlet. Figure 49 shows the flow pattern for November 18, 1975. The effect of wind on the flow pattern is clearly visible.

Flow Pattern from Grass Lake to Lake Marie. Stations 31 and 32 (figure 46) were used to collect the flow pattern data at this location. In general, water was found to be flowing from Grass Lake into Lake Marie. However, on a number of occasions, it was observed that the wind affected the pattern of flow, changed the direction of the floats, and the water followed a path somewhat resembling the resultant of the inflow and wind direction.

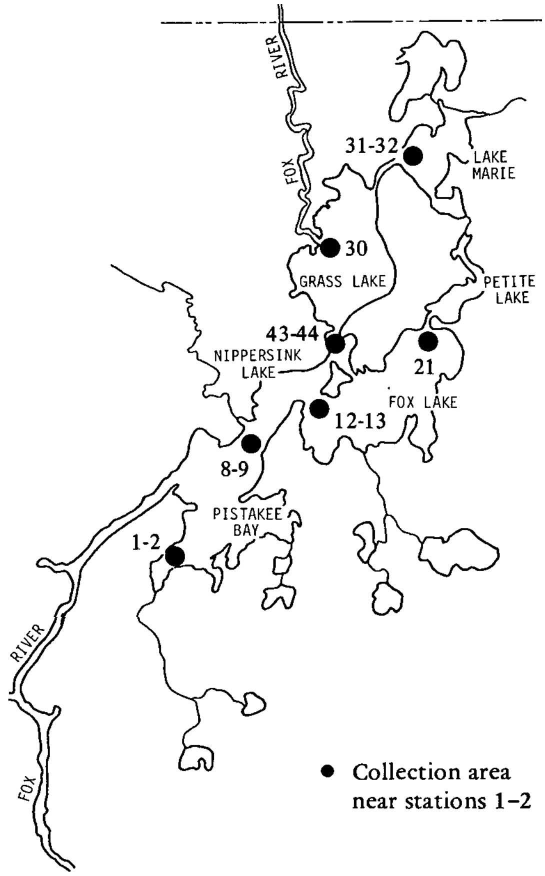


Figure 46. Circulation data collection areas

Figure 50 shows the flow pattern for May 21, 1975. The 12- and 24-inch deep vanes indicate the formation of a long shore current along the northern shore of Lake Marie. Figure 50 also shows the flow pattern on July 15, 1975, when the effect of the wind was also predominant. Figure 51 shows the flow pattern for October 7 and October 10, 1975. On October 10, the vanes were moved inside Lake Marie by a strong westerly wind.

The flow patterns on November 18 and 21, 1975, are shown in figure 52.

Entrance to Fox Lake from Petite Lake. Permanent stations 21 and 22 were used to collect the flow pattern data. Data were collected only on May 21, 1975. Extreme difficulty in the collection of the data at this location precluded any additional work for this area of the lake. Figure 53 shows the flow pattern for this date, and the effect of wind is definitely visible.

Flow Pattern below Grass Lake Road Bridge. In general, water flows downstream at this particular location. Once in a while a strong southerly wind would induce some local change in the flow pattern, but the overall downstream flow pattern remained the same.

Figure 54 shows the flow pattern on May 22 and July 16, 1975. The southerly wind on

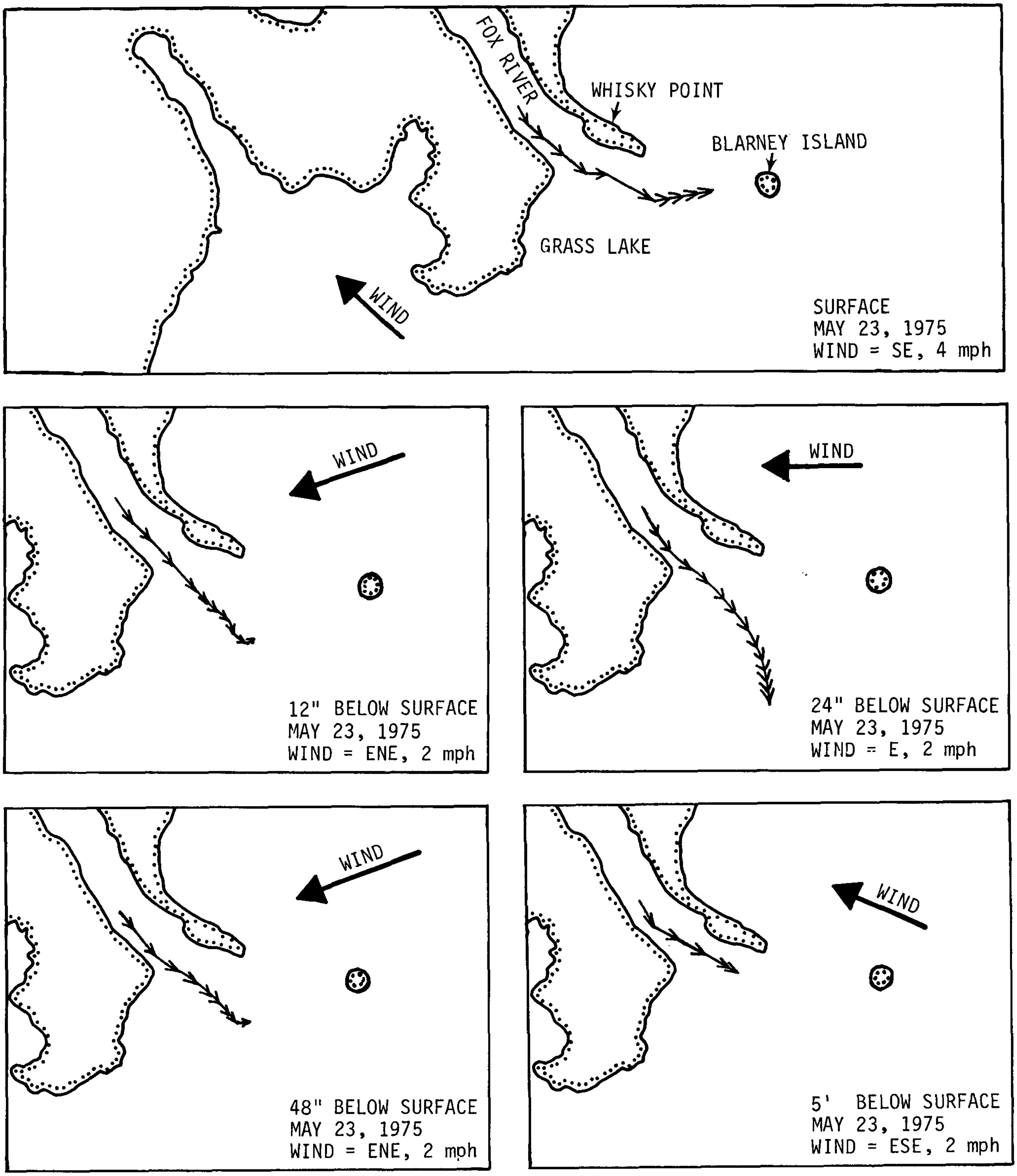


Figure 47. Flow pattern at the entrance of Fox River to Grass Lake for May 23, 1975

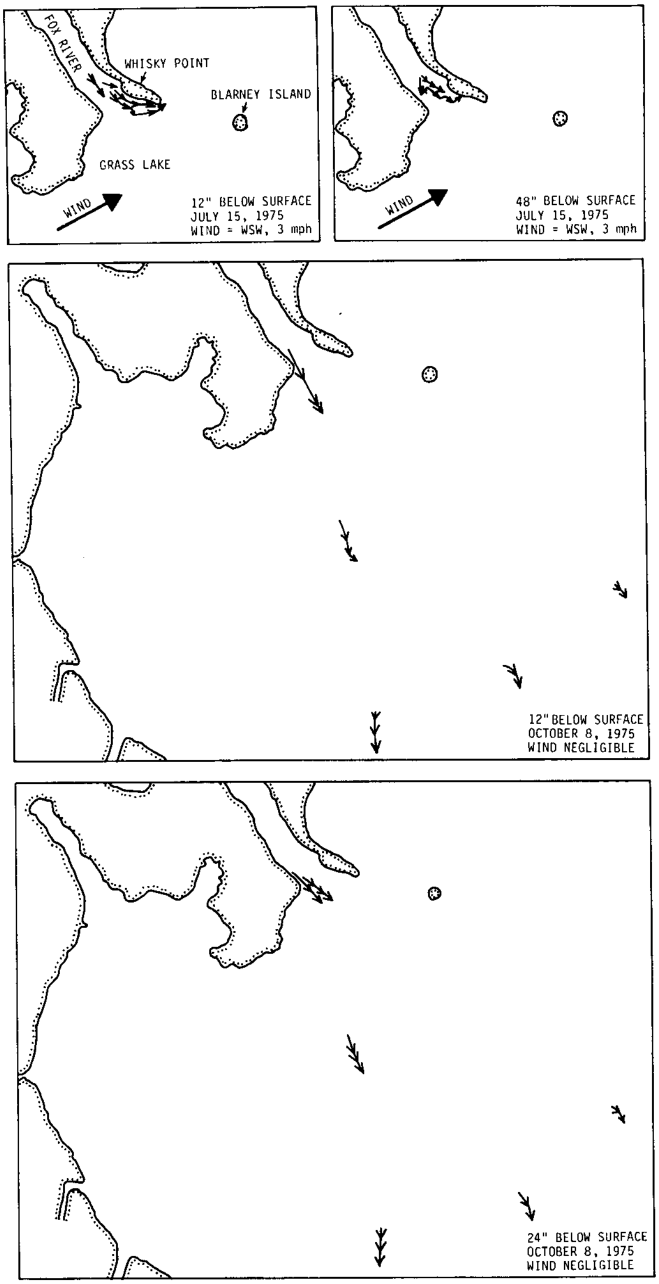


Figure 48. Flow pattern at the entrance of Fox River to Grass Lake for July 15, and October 8, 1975

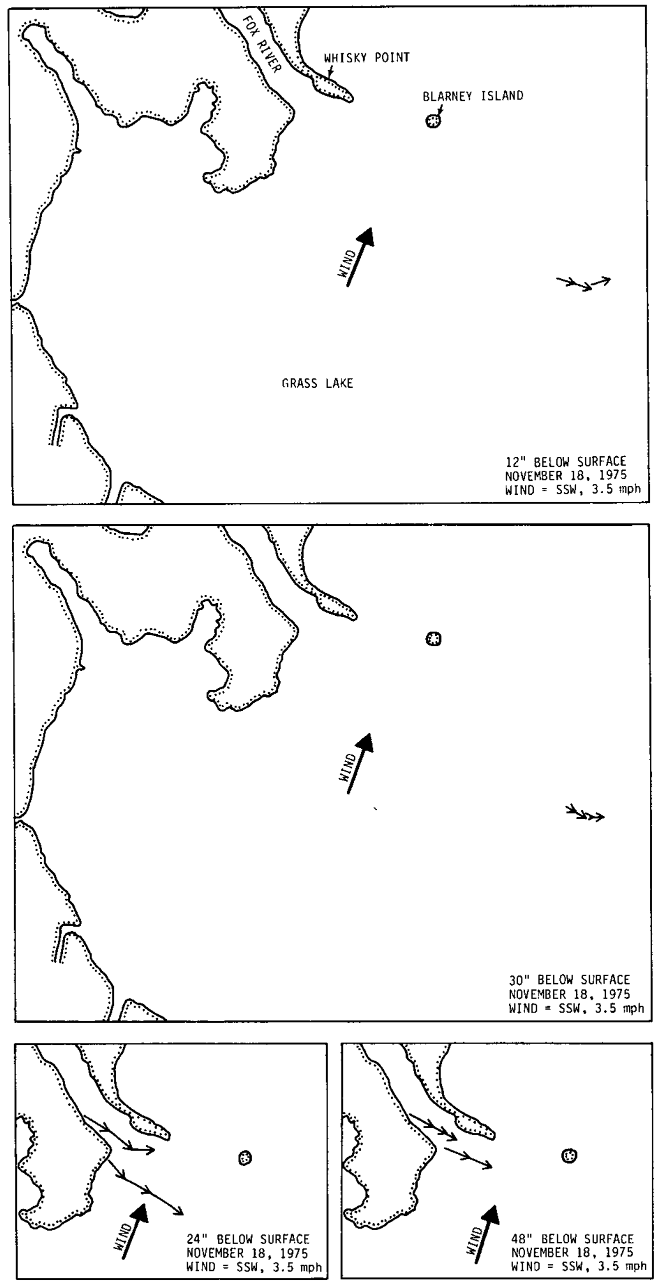


Figure 49. Flow pattern at the entrance of Fox River to Grass Lake for November 18, 1975

July 16 affected the 12-inch deep vanes by moving them toward the bridge, but the 24-inch deep vanes moved in the downstream direction. The flow patterns for October 7 and November 19 are shown in figure 55.

Flow Pattern between Fox Lake and Nippersink Lake. These data were collected between stations 12 and 13 (figure 46). Flow patterns on May 20 and 22, and on July 16, 1975, are shown in figure 56. In general, water flows from the Fox Lake into Nippersink Lake with some deviation of the flow path because of wind movement.

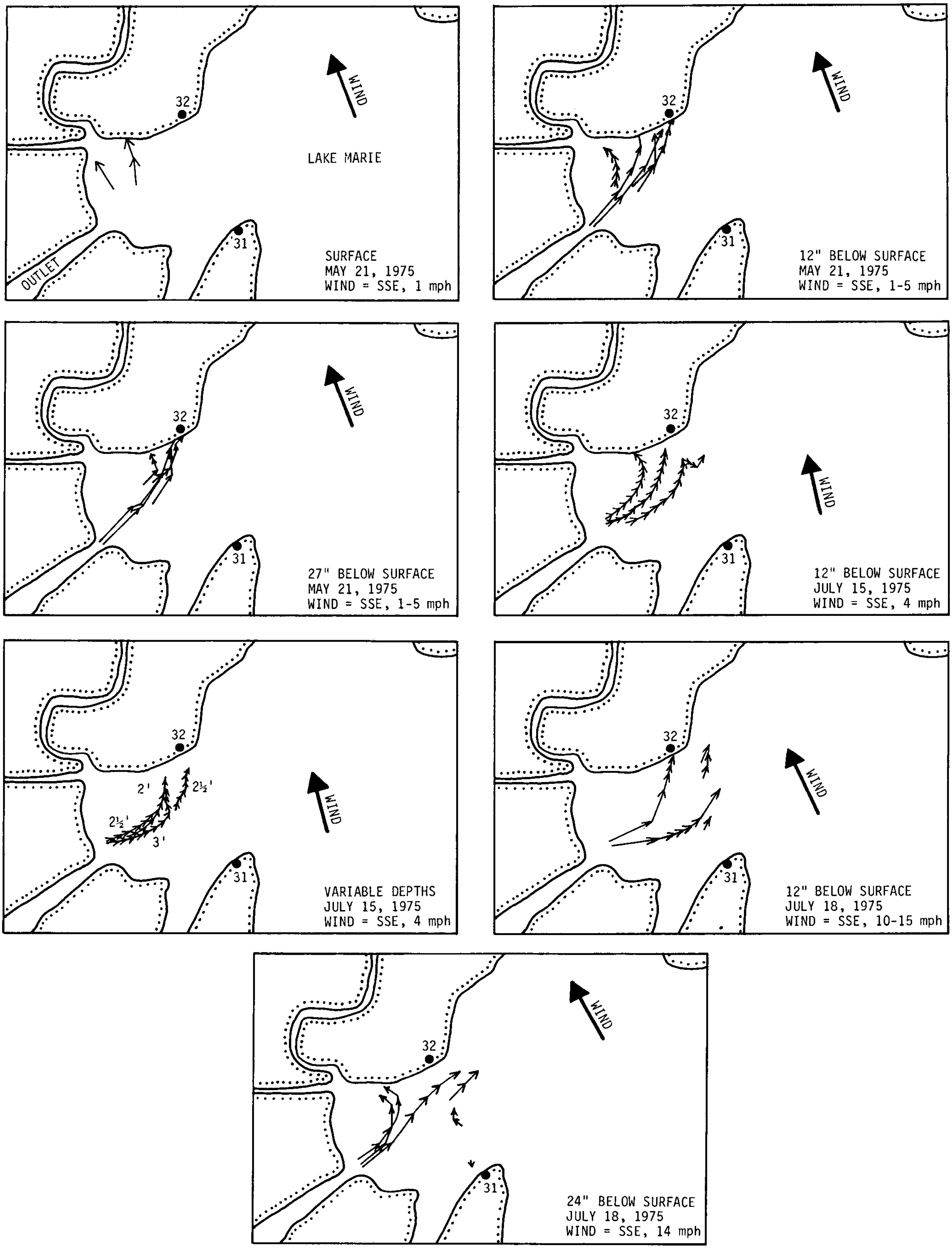


Figure 50. Flow pattern from Grass Lake to Lake Marie for May 21, July 15, and July 18, 1975

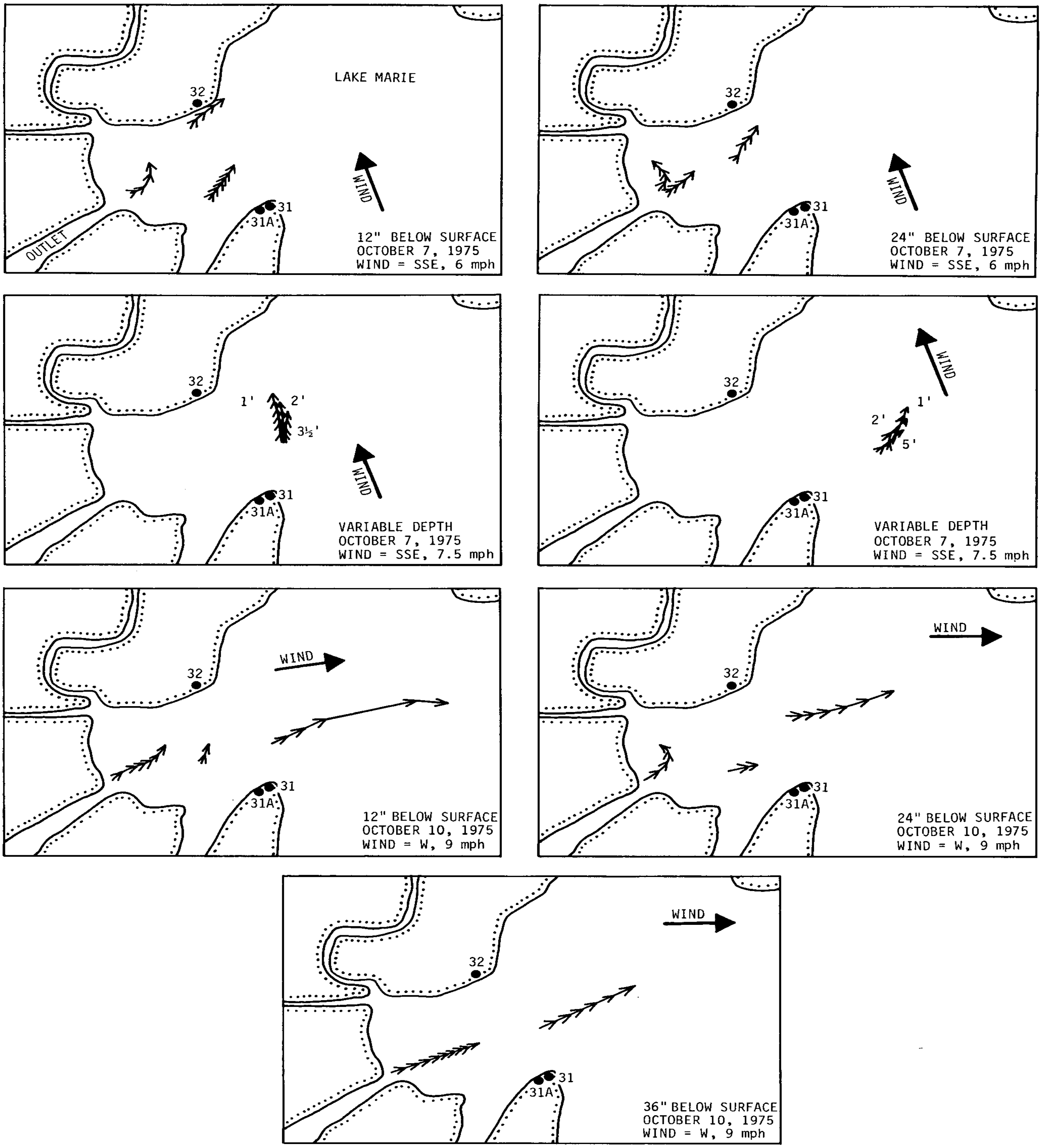


Figure 51. Flow pattern from Grass Lake to Lake Marie for October 7 and October 10, 1975

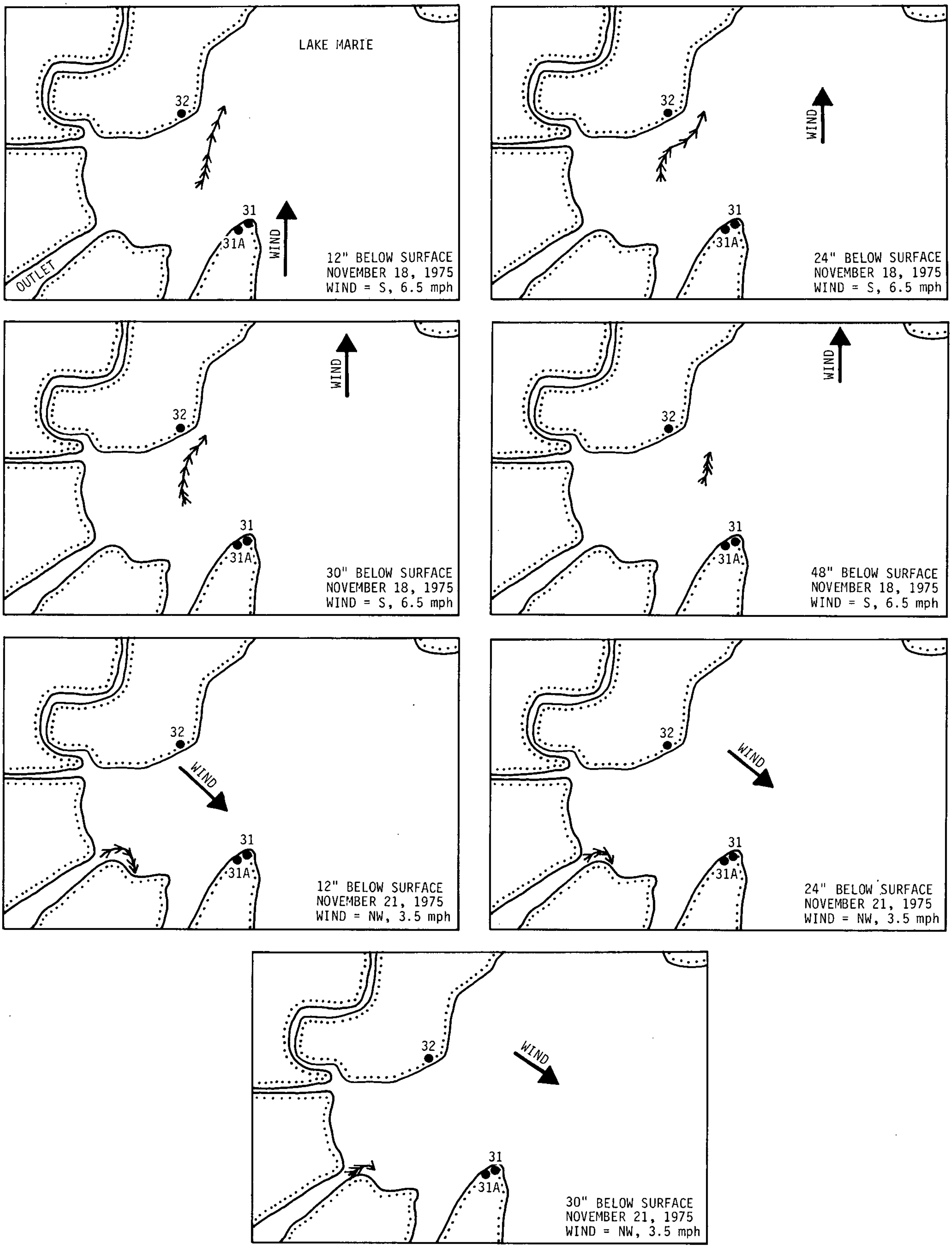


Figure 52. Flow pattern in Lake Marie near the entrance from Grass Lake for November 18 and November 21, 1975

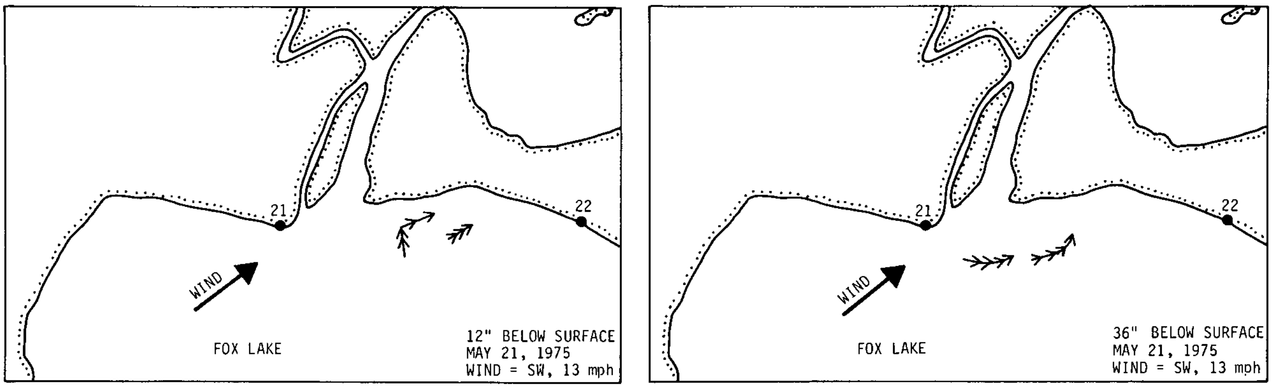


Figure 53. Flow pattern in Fox Lake at the entrance from Petite Lake for May 21, 1975

Figure 57 shows the flow pattern on October 8, 1975. Presence of a moderate wind from the east-southeast has modified the flow pattern with the development of a long shore current along the south shore of Fox Lake. Figure 57 shows a change in the flow pattern for November 19, caused by a change in wind direction.

Flow Pattern Downstream of Highway 12 Bridge. Stations 8 and 9 (figure 46) were used to collect data at this location. Water flows in the downstream direction below the highway bridge. When this flow reaches Pistakee Lake, the presence of southerly winds generally deflects the flow and moves it in a circular route before it finally moves on downstream. Nett's Island acting as a barrier also modified the flow pattern.

Figure 58 shows the measured and probable flow patterns for May 15, 1975. Effects of southerly wind and Nett's Island are clearly visible. Figure 58 also shows the flow pattern for July 17, 1975. The flow pattern for October 9, 1975, is shown in figure 59.

Flow Pattern in Pistakee Bay. Extensive flow data from Pistakee Bay were collected with the use of stations 1 and 2 (figure 46). The inflow to Pistakee Bay from Lily Lake Drain can be considered negligible.

Figure 60 shows the flow pattern for May 15, 1975. The surface floats were basically moving in the direction of the wind. The movement of the 12-inch deep floats close to the western shore definitely indicated the presence of a long shore current. Figure 60 also shows the flow pattern for May 16 at the surface and 12 inches below surface. Wind was blowing from the east-northeast and basically the upper layer of water was moving in the same direction. However, 5 feet below the water surface, water was moving in the opposite direction indicating the presence of some type of return flow (figure 61). It is interesting to note that 10 feet below the water surface the water was moving in the direction of the wind.

The flow pattern for July 17 in figure 61 shows the upper layer of the water moving in the direction of the wind, whereas 5 feet below the surface water was moving in the opposite direction. The presence of a long shore current can be seen in this figure.

Figures 62 and 63 show the circulation patterns for October 9, 1975. Some deviations of water movement are present.

General Observations Regarding Circulation Patterns. The circulation patterns in all of the lakes in areas of negligible inflows were generally governed by the wind. The direction and magnitude of the wind played a very important role in determining the flow patterns. In zones of significant inflow, such as the Fox River inflow (figures 47-49) or the inflow from Grass Lake to Lake Marie (figures 50-52), as soon as the diffusion of the inflow jets reduced the flow velocity, the wind-induced circulation patterns dominated the

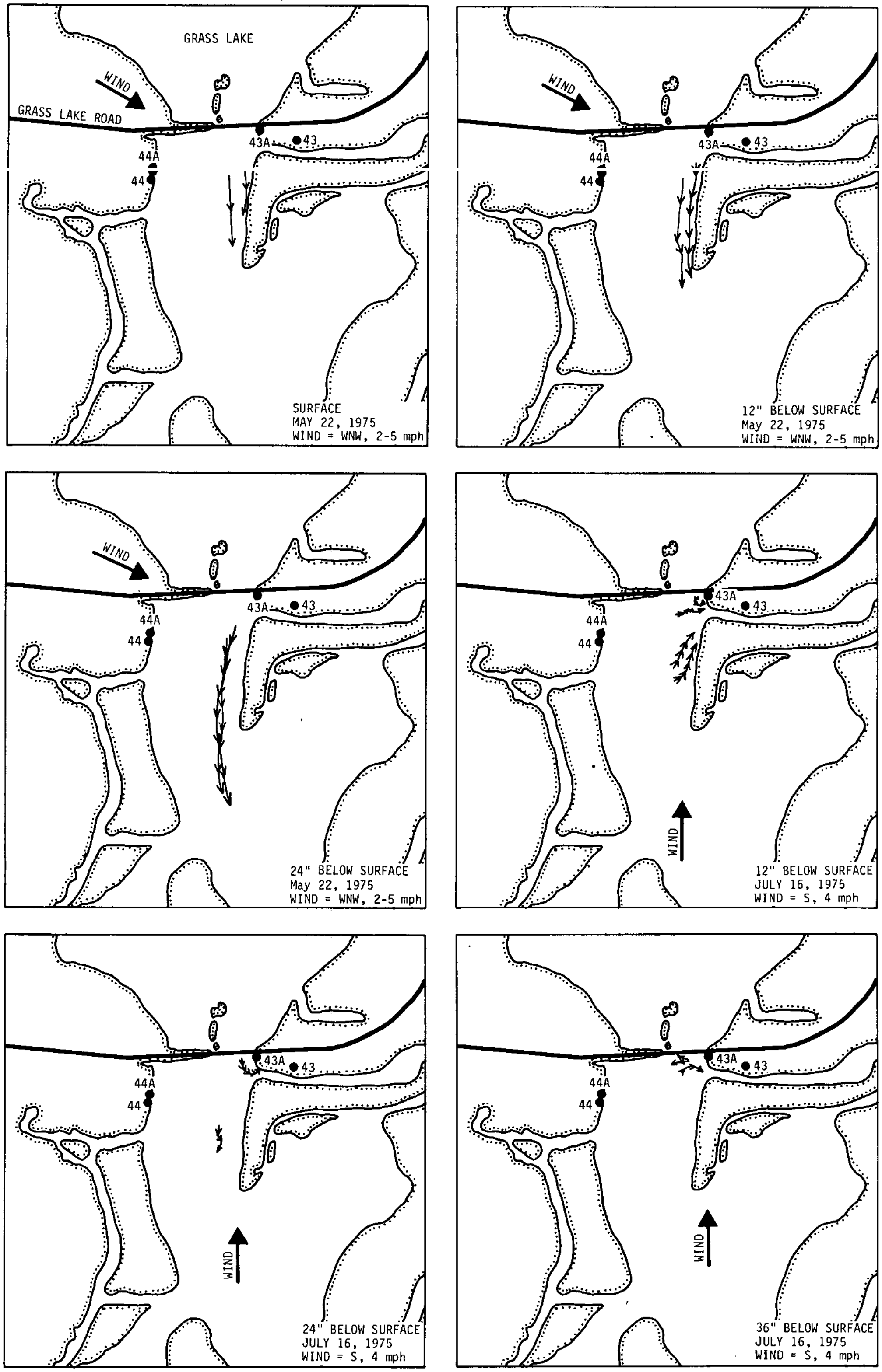


Figure 54. Flow pattern below Grass Lake Road for May 22 and July 16, 1975

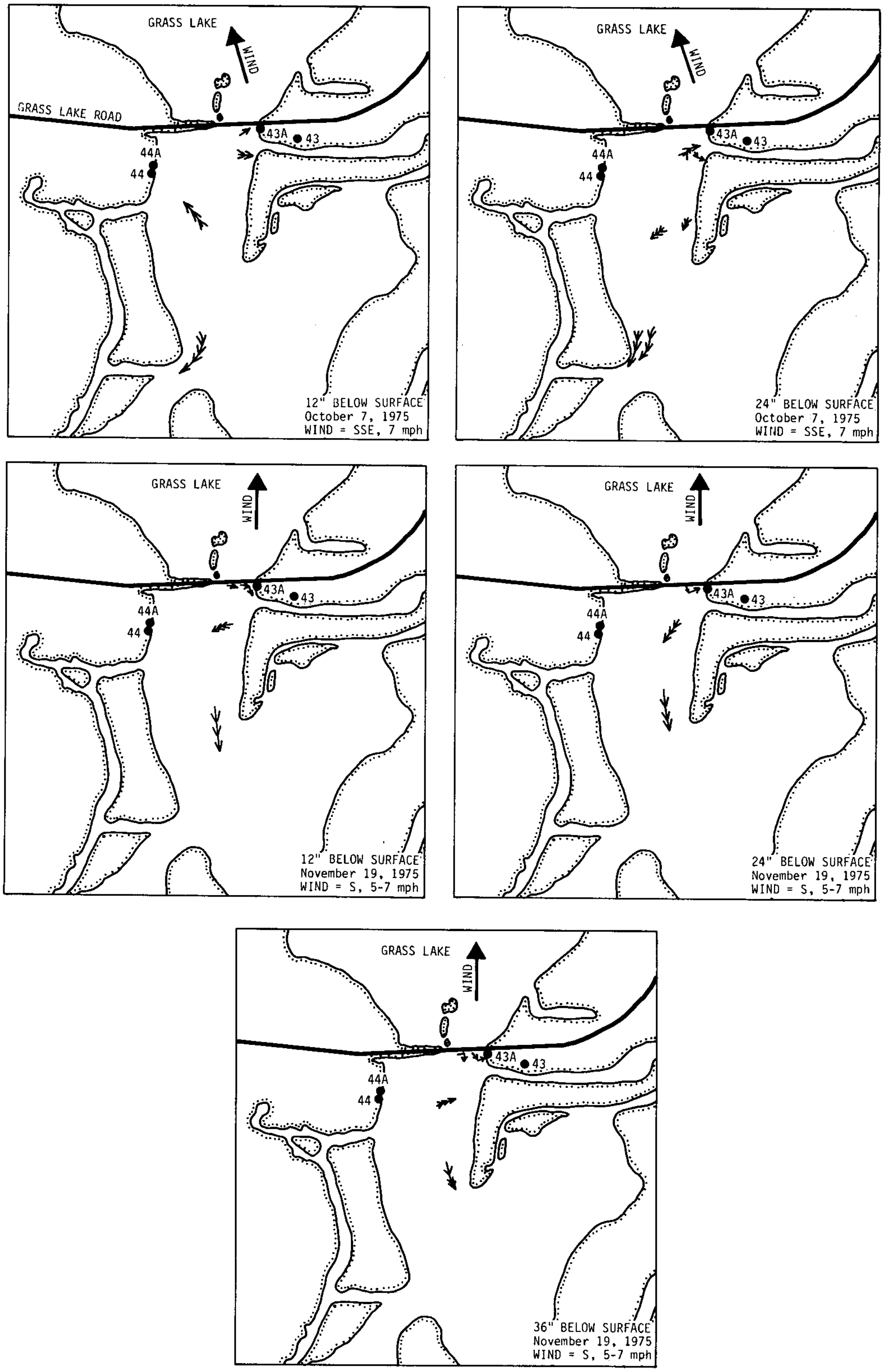


Figure 55. Flow pattern below Grass Lake Road for October 7 and November 19, 1975

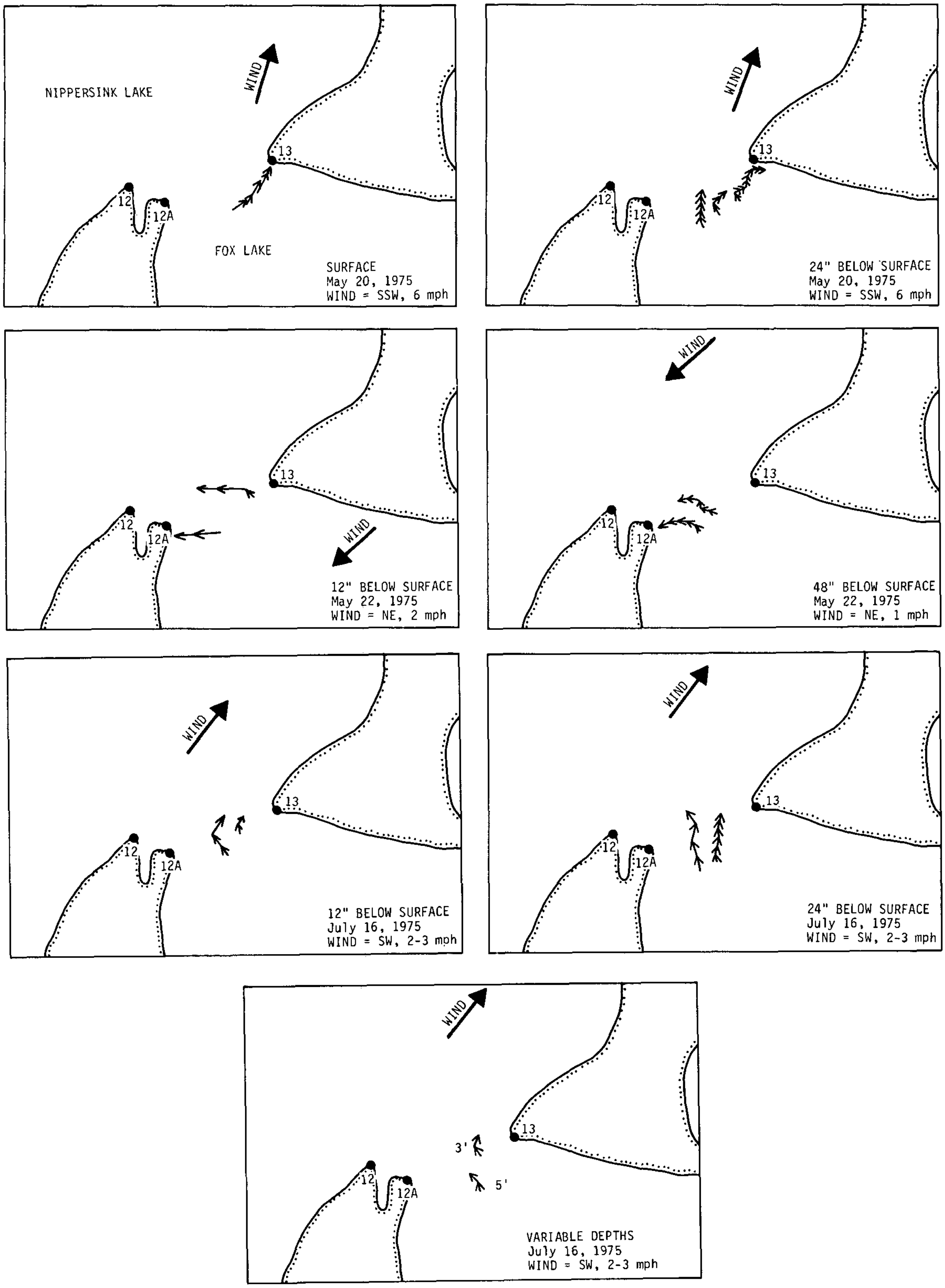


Figure 56. Flow pattern between Fox Lake and Nippersink Lake for May 20, May 22, and July 16, 1975

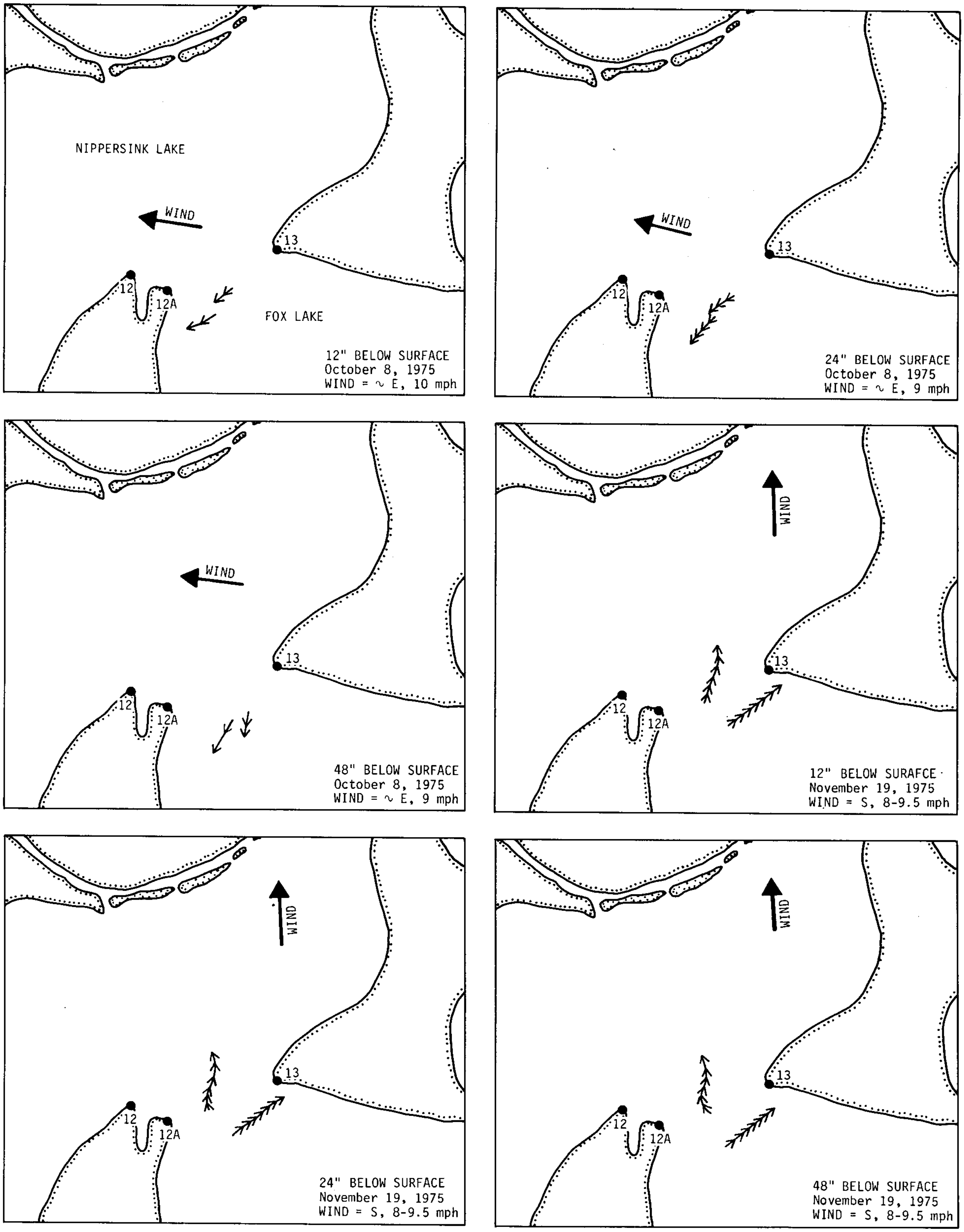


Figure 57. Flow pattern between Fox Lake and Nippersink Lake for October 8 and November 19, 1875

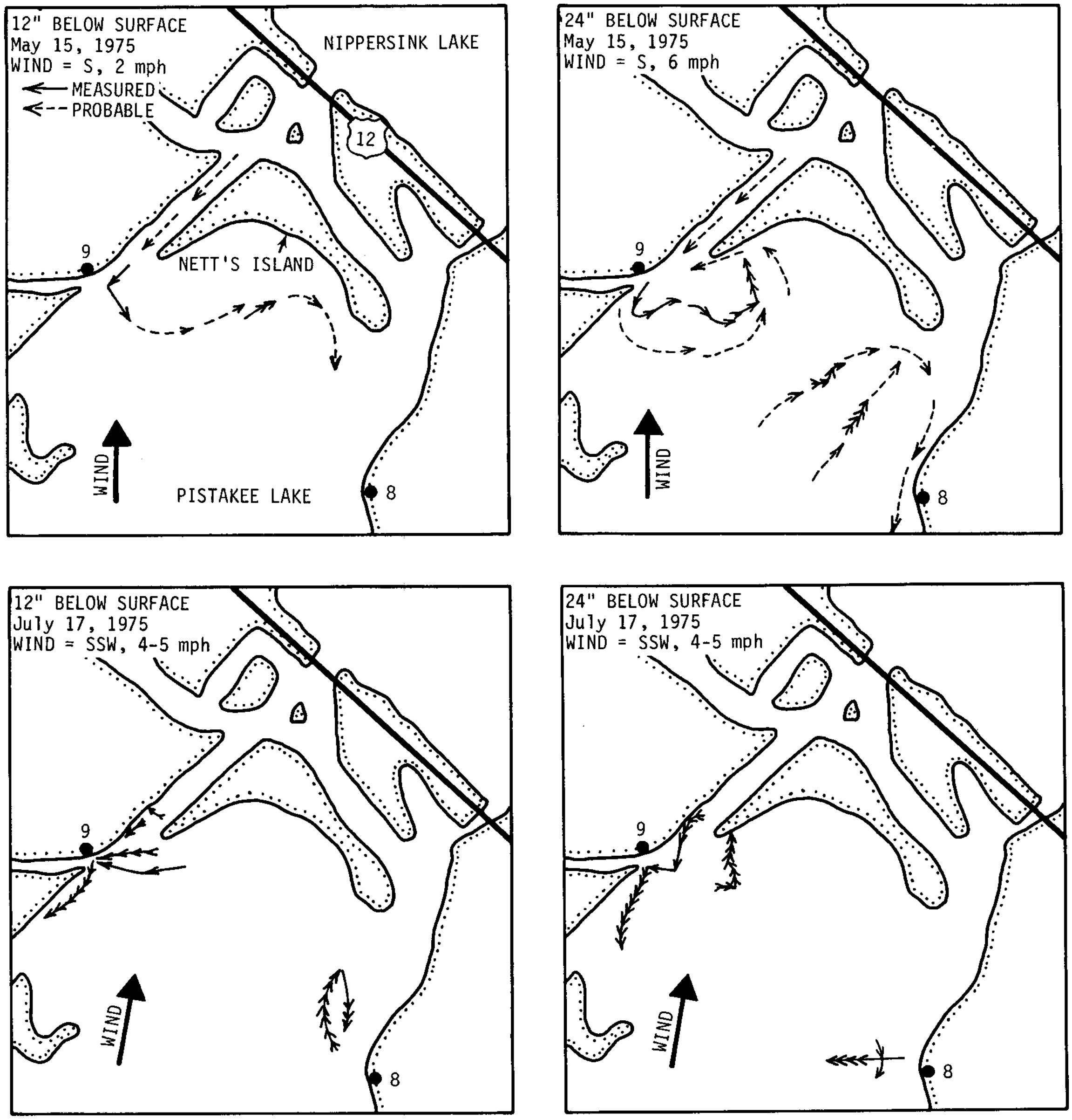


Figure 58. Flow pattern downstream of Highway 12 Bridge for May 15 and July 17, 1975

general movement of the water. Figures 50-52 also show that the velocity vector entering Lake Marie from Grass Lake was deflected as a result of the presence of a moderate southerly wind. During the whole process of data collection, it was repeatedly observed that the circulation pattern is a function of the wind velocity and direction.

Measured Current Profile. Generally, as described earlier, three sets of floats with cross vanes were dropped to various depths and followed simultaneously from two fixed shore stations, and these data were used to compute the flow velocity at different depths. Figure 64 shows some typical current profiles plotted on semilogarithmic paper. Data from three different locations are plotted in this figure. In general, these vertical current profiles follow logarithmic distributions as postulated in Bye's equation (*page 68*). Figure 65 shows

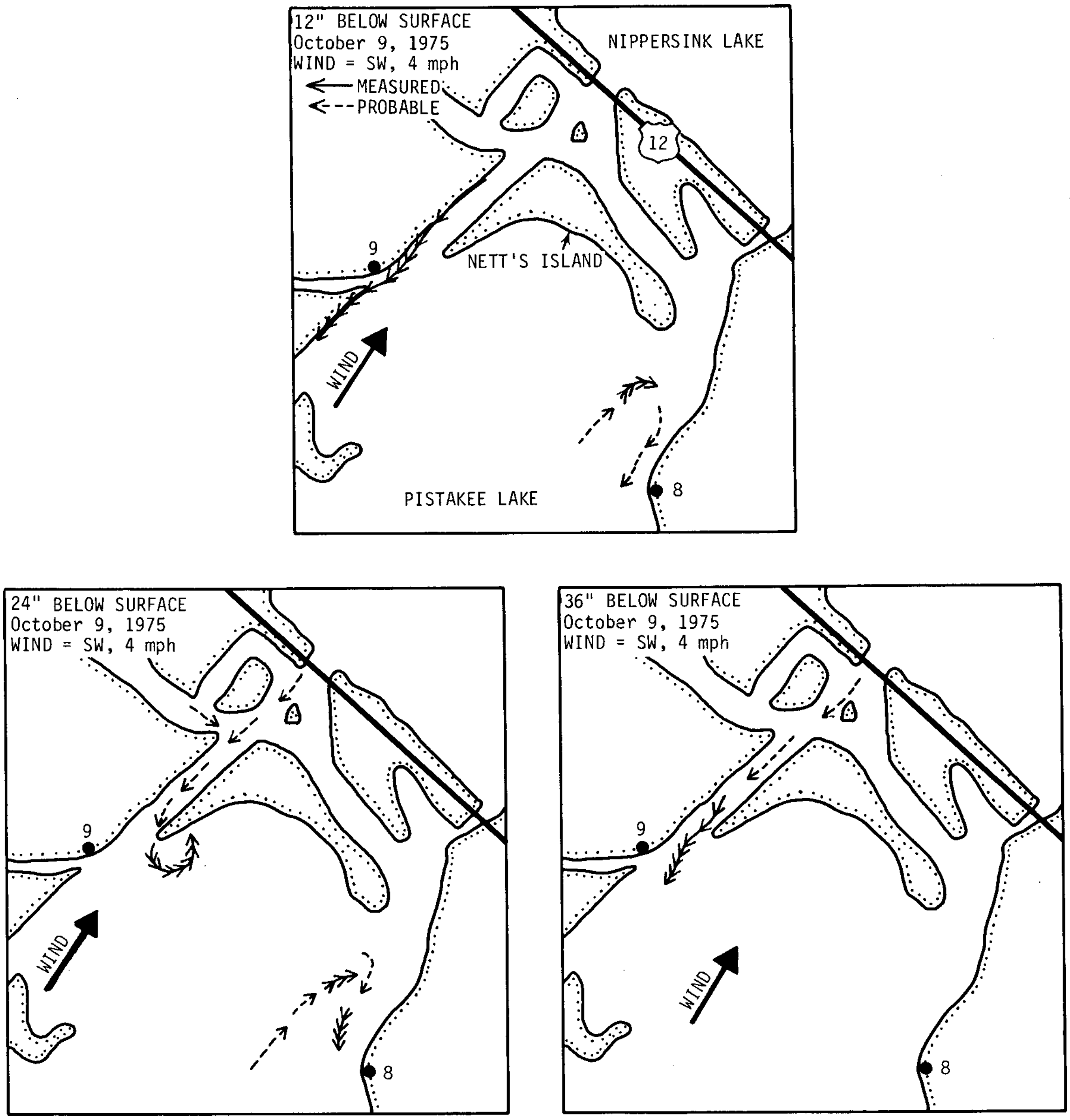


Figure 59. Flow pattern downstream of Highway 12 Bridge for October 9, 1975

the vertical current profile plotted on coordinate paper. Data were collected on July 16, 1975, between Fox and Nippersink Lakes. The shape of this current profile is similar to the one shown in figure 45. Basically, the flow velocity near the water surface is high and velocity decreases with depth.

Data collected from Pistakee Bay indicated the presence of return flow for some wind conditions, as exemplified in figures 60 and 61. The vertical current profile for May 16, 1975, is shown in figure 66. On this particular day water near the surface moved in the same direction as the wind, at about 5 feet below the surface the water was moving in the opposite direction, and somewhere below 10 feet the water was again moving in the direction of the wind. This indicated the very complex nature of the flow patterns. It must be pointed out

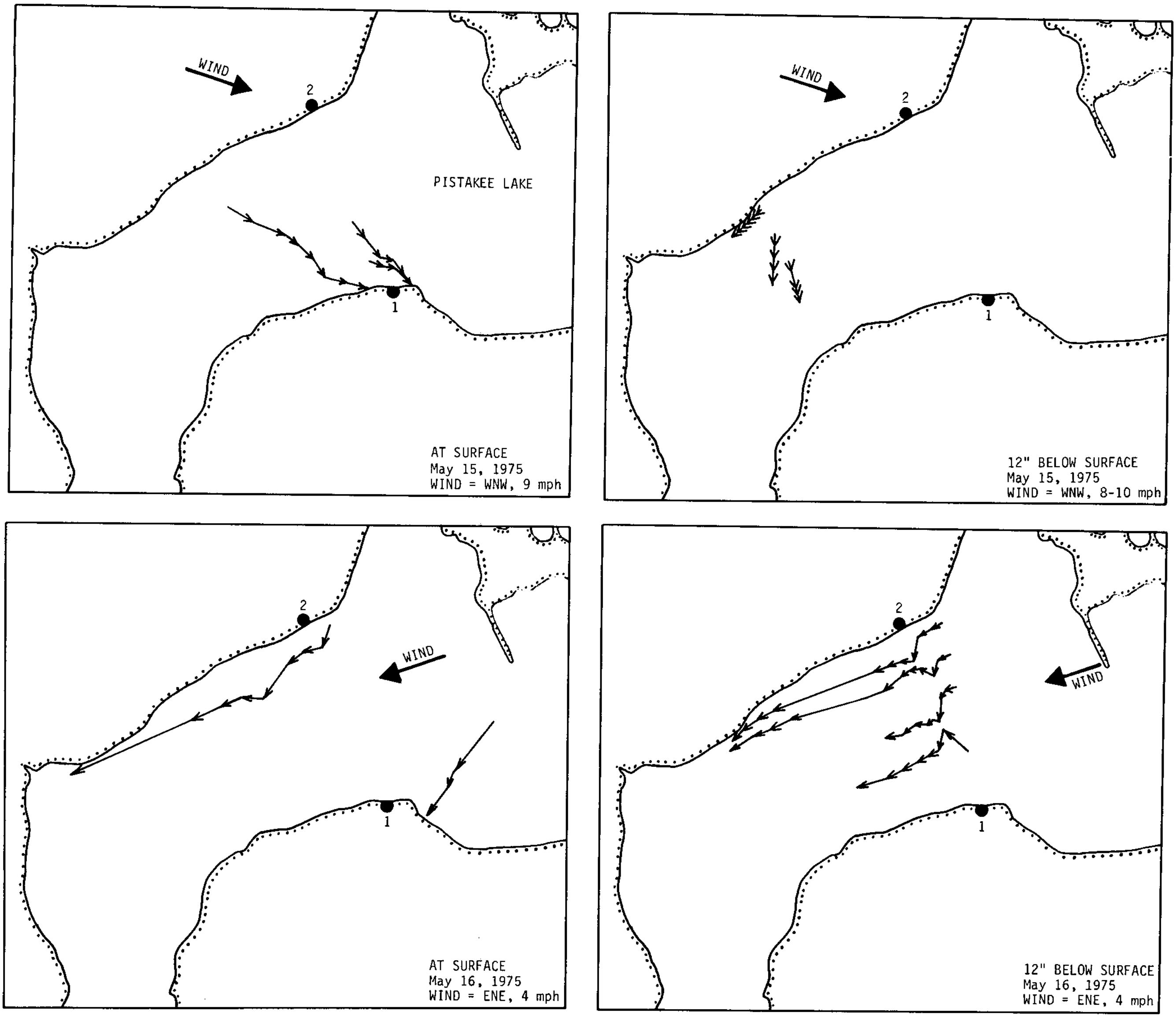


Figure 60. Flow pattern in Pistakee Bay for May 15 and May 16, 1975

that the magnitude of the velocities is very small and only long periods of observation confirmed these movements.

Therefore, it appears that in most cases the wind-generated current profile is logarithmic in nature and that a return flow may exist below the water surface, though the absolute magnitude of this flow may be very small.

Available published data related to surface drift and wind factor have already been discussed. Some of the data collected in the Chain of Lakes on surface drift are given in table 16. The wind factor, defined as the ratio of surface drift to wind velocity, varied from 2 to 11 percent. However, the higher values of wind factor at stations 8–9, 31–32, and 43–44 do not represent the effect of wind-induced surface current because the influence of the inflow is very strong at these locations. For the other stations, the wind factor varied from 2 to 7 percent.

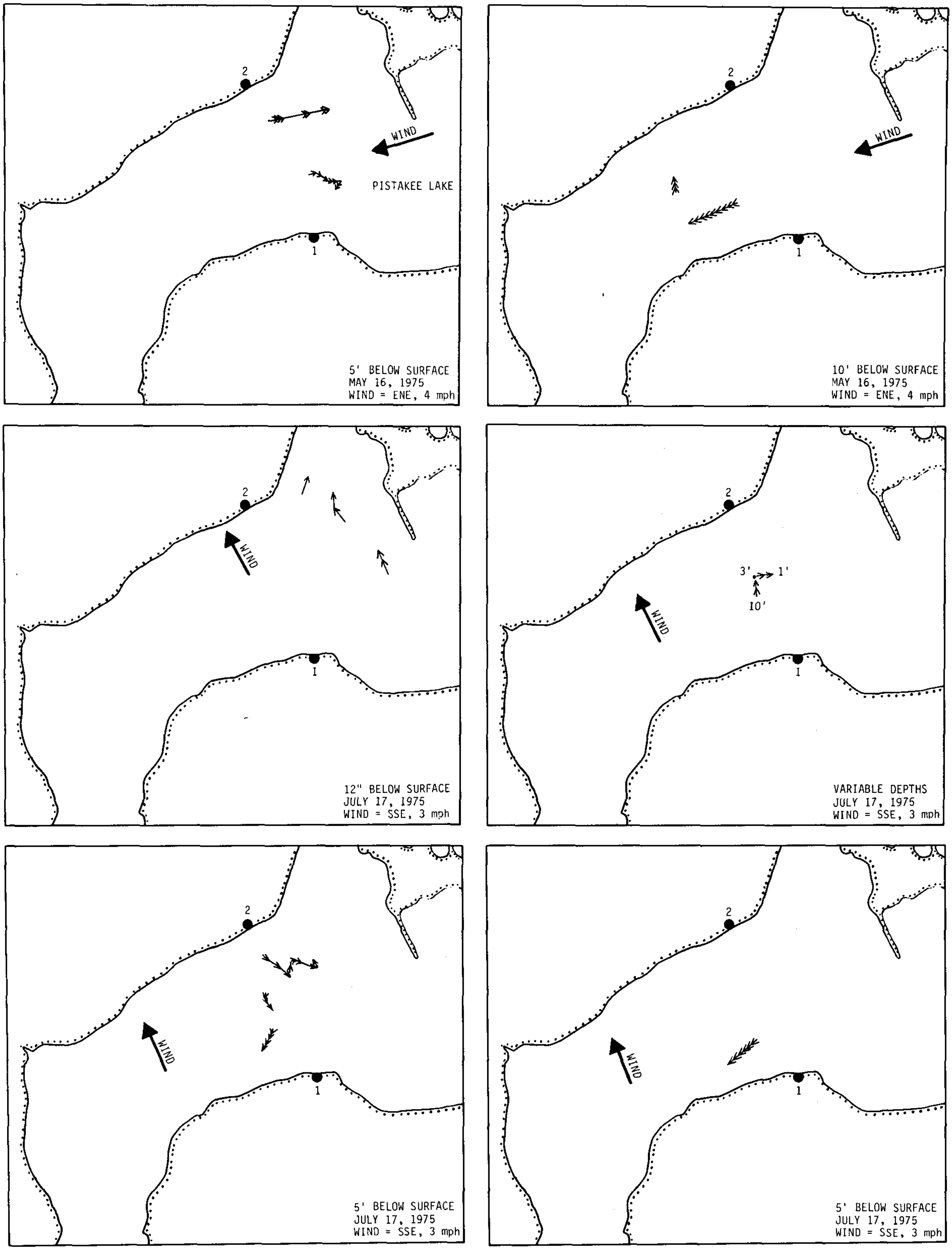


Figure 61. Flow pattern in Pistakee Bay for May 16 and July 17, 1975

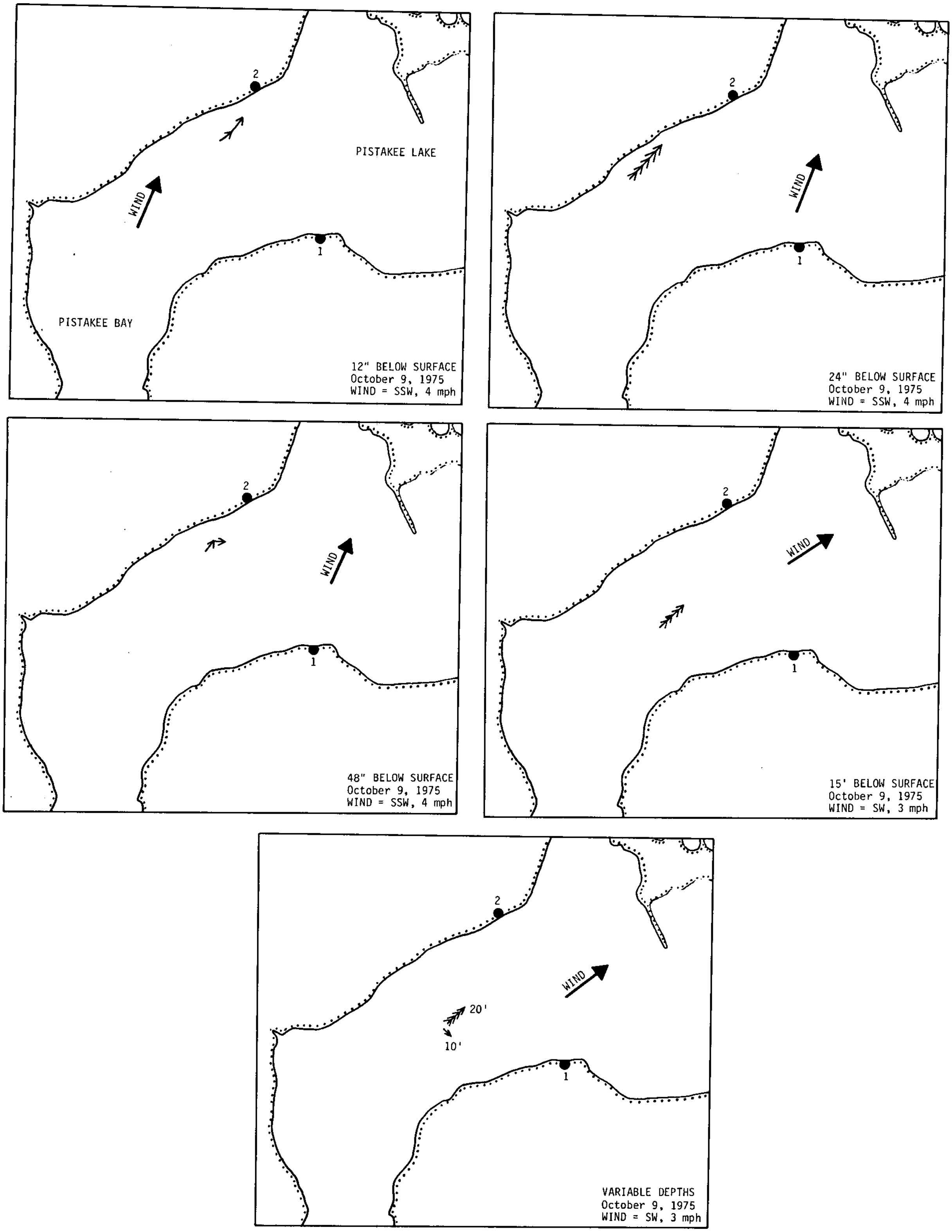


Figure 62. Flow pattern in Pistakee Bay for October 9, 1975

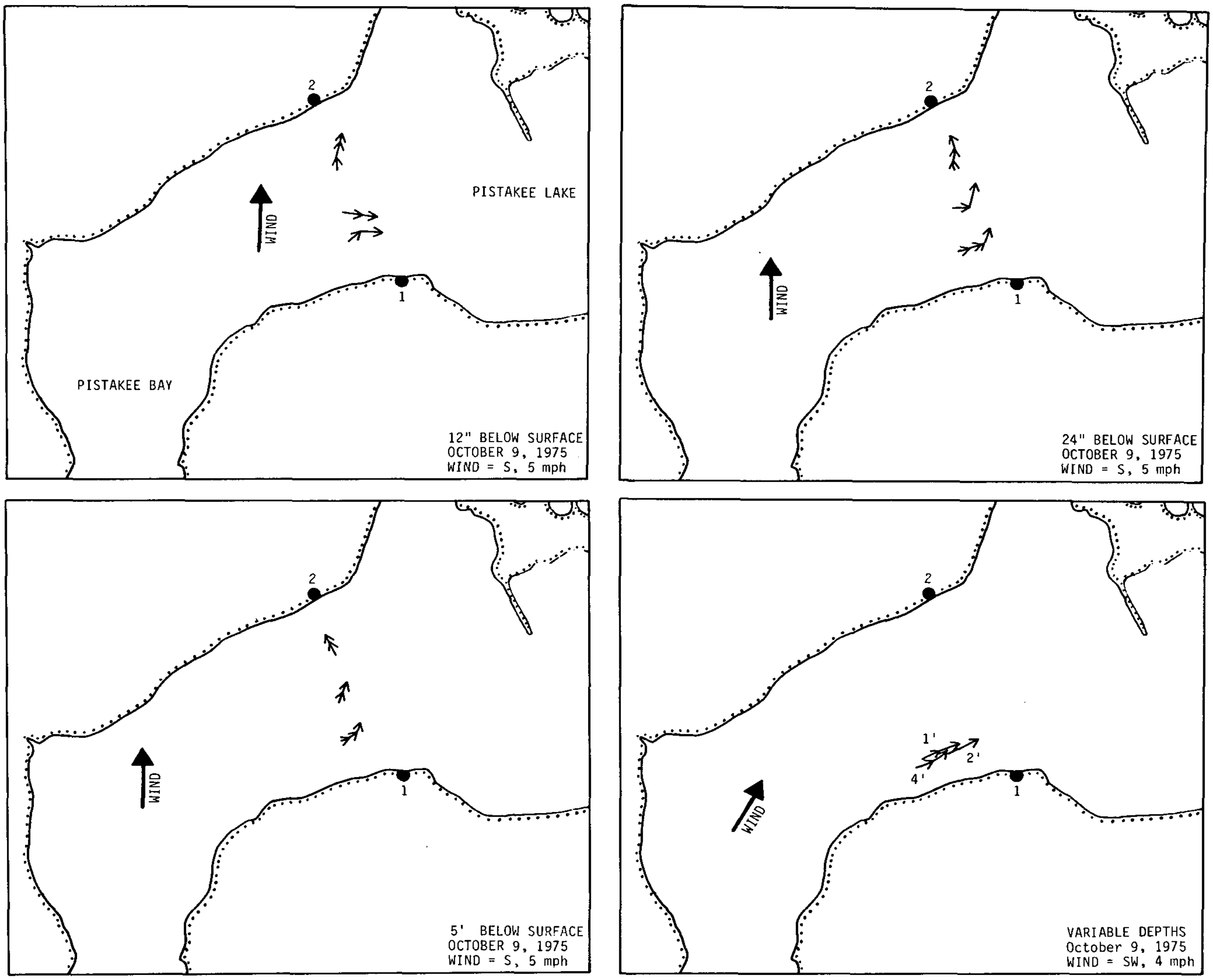


Figure 63. Flow pattern in Pistakee Bay for October 9, 1975

Comparison of Theoretical Circulation Patterns and Field Data. A considerable amount of theoretical and analytical work has been done by various investigators on circulation patterns in large lakes. Methods such as finite element, finite difference, method of characteristics, and others were utilized to solve the equation of momentum transport. Recently Song (1976) using the finite element method proposed by Gallagher et al. (1973) computed a theoretical circulation pattern for the Pistakee Bay area of the Chain of Lakes. The incompressible form of the momentum transport equation used is given in vector form (Liggett and Hadijtheodorou, 1969) as follows:

$$\frac{\partial \vec{v}}{\partial t} + (\vec{v} \cdot \vec{v}) + \vec{F} + \frac{1}{\rho}\vec{\nabla} p - \vec{g} - \eta \nabla^2 v = 0$$

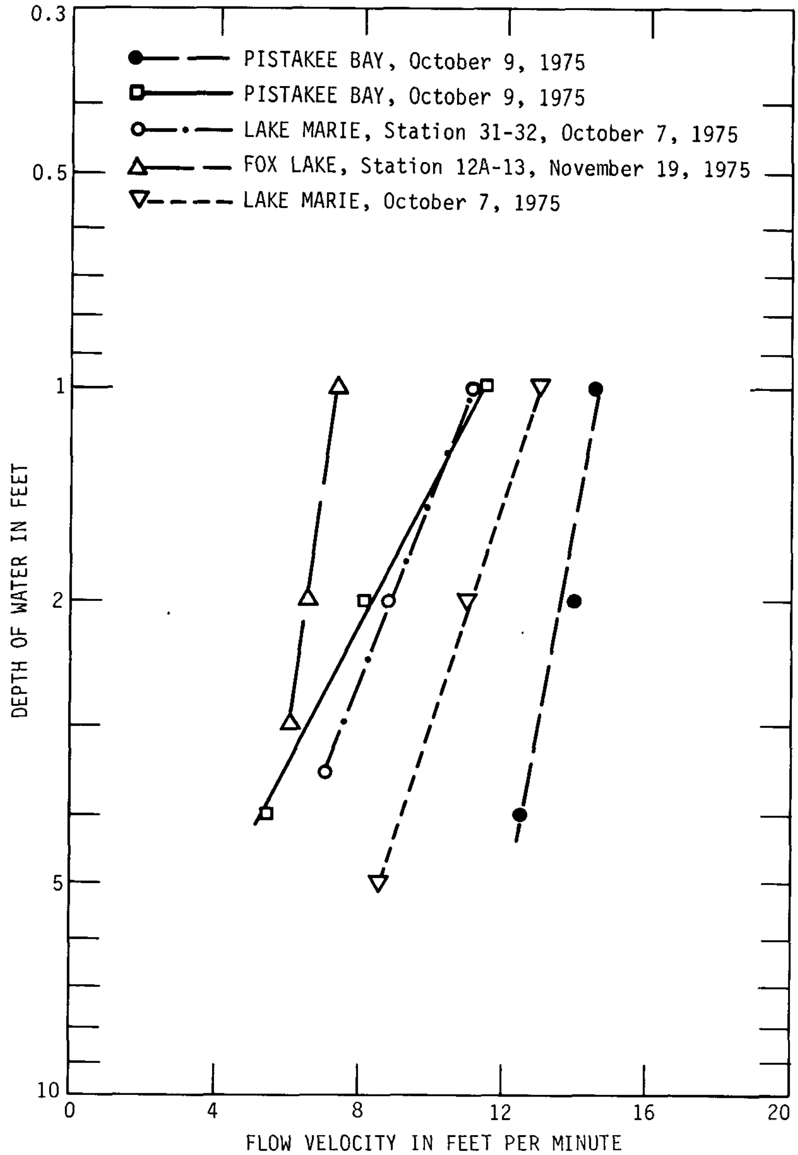


Figure 64. Typical vertical current profile (logarithmic)

where $\vec{v}$ is the water velocity, $\vec{F}$ is the Coriolis force which accounts for the earth's rotation on the flow, $\vec{g}$ is the gravity factor, ρ is the fluid density, η is the eddy viscosity, and p is the pressure. This equation was simplified for a shallow lake with hydrostatic pressure distribution under the influence of steady-state wind conditions. The Coriolis parameter was assumed to be constant and the Rossby number (ratio of inertial forces to rotational forces) was taken to be small. The above equation can thus be simplified to

$$\vec{F} - \vec{g} = -\frac{1}{\rho}\vec{\nabla}\rho + \eta\nabla^2 \vec{V}$$

This equation and the finite element method as proposed by Gallagher et al. (1973) were utilized by Song (1976) with appropriate boundary conditions to obtain circulation patterns

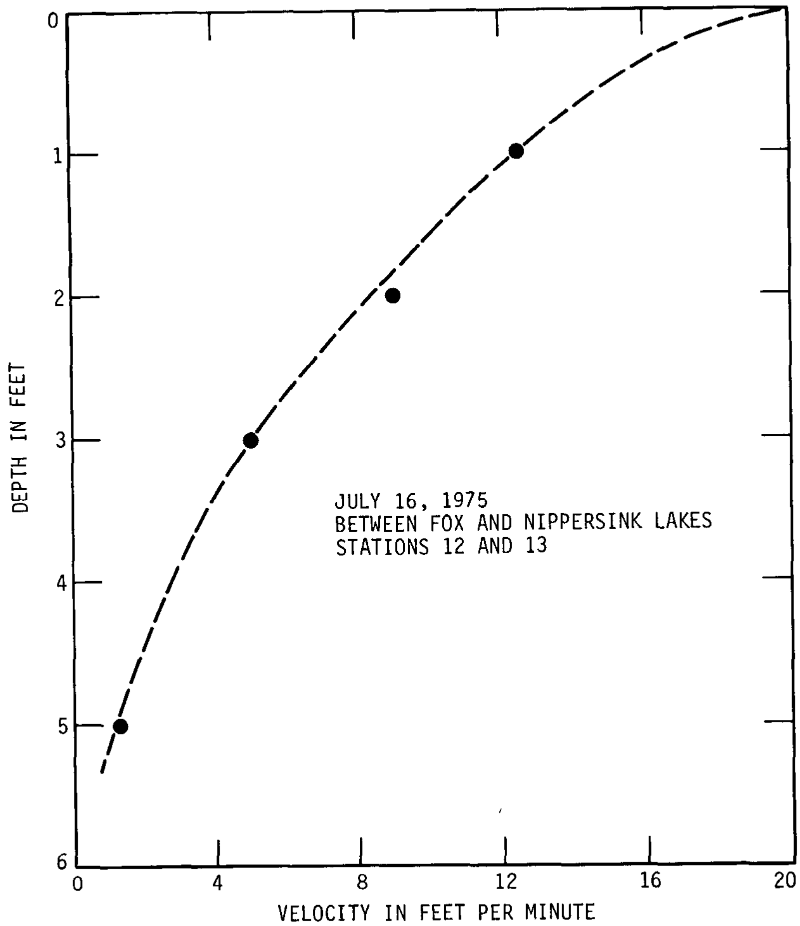


Figure 65. Current profile in Fox Lake on July 16, 1975

in the Pistakee Bay area for some known wind conditions. A number of Calcomp plots were obtained by varying the numerical value of the eddy viscosity, η. Figures 67-70 compare some of the typical computer plots with the observed field data. Correlations between theory and the field data appear to be very good. The magnitude and direction of the flow velocities obtained in the Calcomp plots follow very closely the flow pattern data obtained from Pistakee Bay.

This analysis indicates that the wind-generated circulation patterns in natural lakes can be predicted to some extent by using the analytical and mathematical formulations that have already been developed. One area where further work is needed is the proper determination of the numerical value of eddy viscosity η for actual field conditions.

Table 16. Wind Factor in the Chain of Lakes

Date of data collection	*Stations*	*Wind factor* (%)	*Lake*
May 15, 1975	8-9	10.2*	Pistakee
		9.1*	Pistakee
May 21, 1975	31-32	5.7	Marie
		7.2*	Marie
May 20, 1975	12-13	2.1	Between Fox
		3.3	and Nippersink
May 22, 1975	43-44	11.8*	Below Grass
		7.5*	Lake Road Bridge
May 15, 1975	1-2	2.7	Pistakee Bay
	1-2	2.1	Pistakee Bay
	1-2	2.0	Pistakee Bay
May 16, 1975	1-2	7.1	Pistakee Bay
		5.4	Pistakee Bay

* Affected by inflow

Note: Wind factor = (Surface drift/wind velocity) x 100

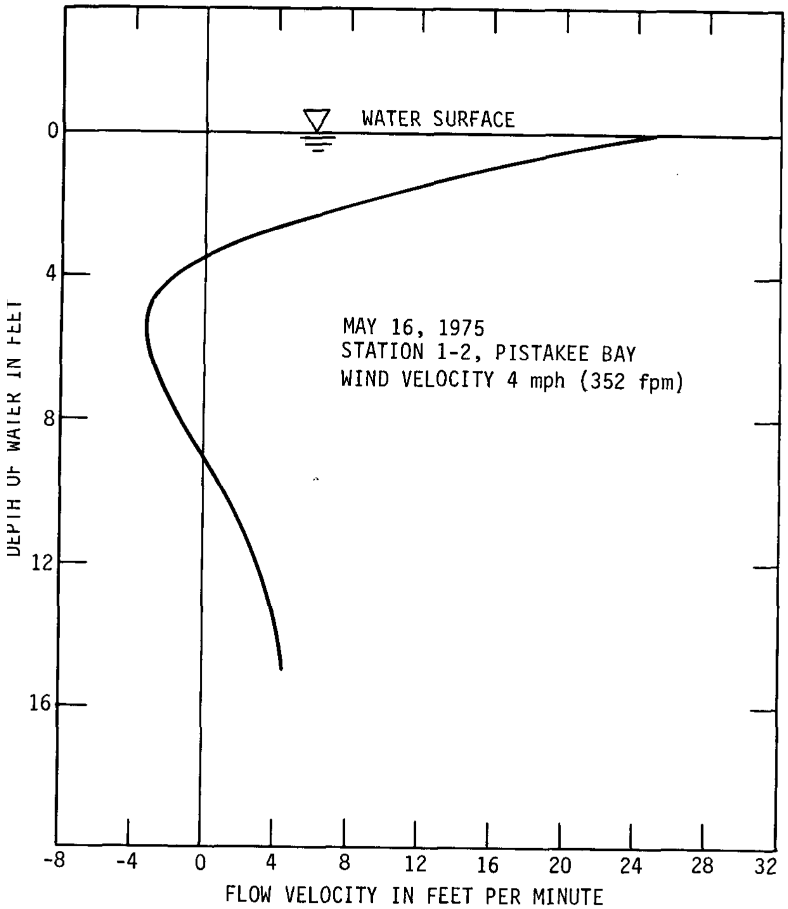


Figure 66. Vertical current profile in Pistakee Bay on May 16, 1975

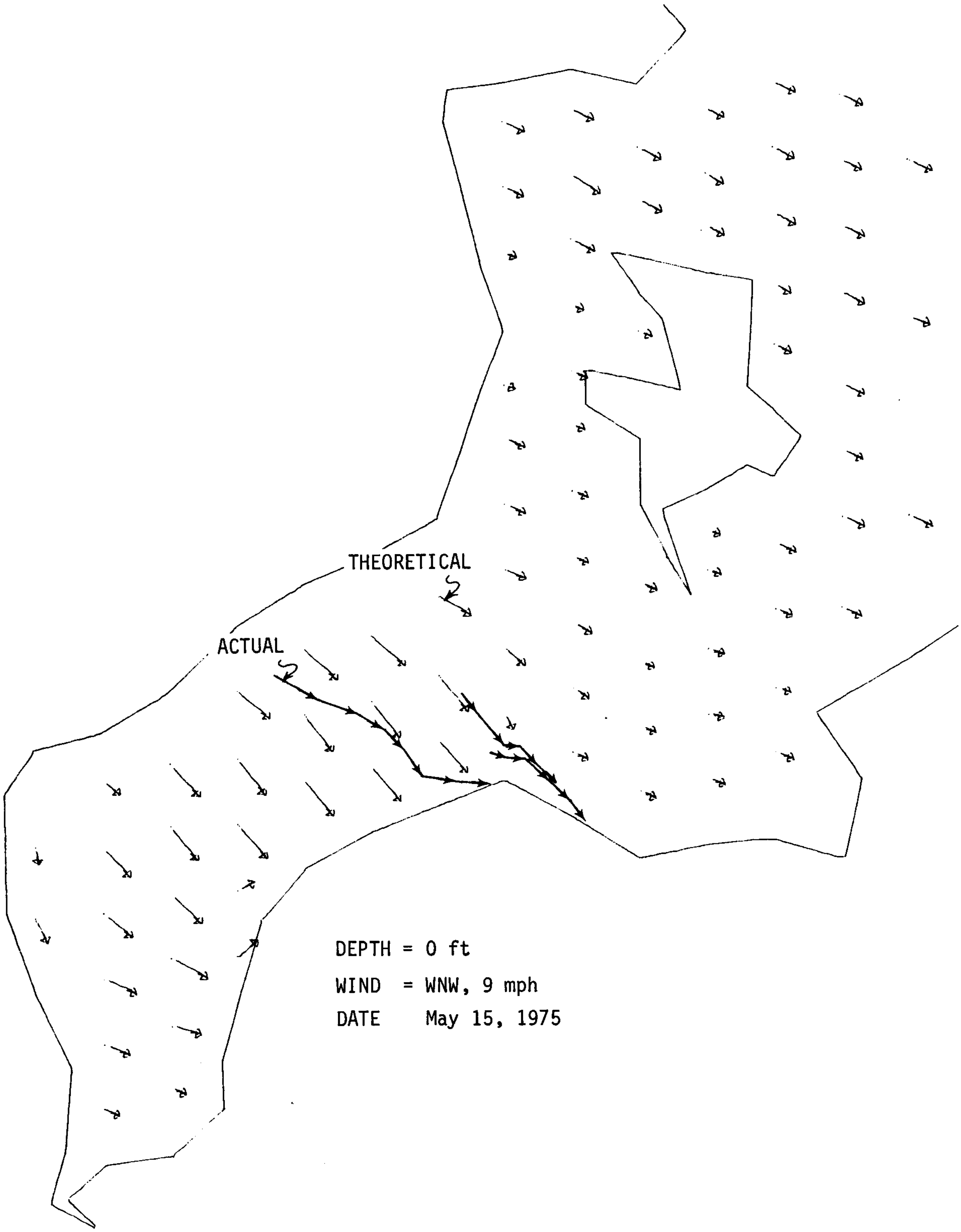


Figure 67. Theoretical and actual circulation patterns in Pistakee Bay

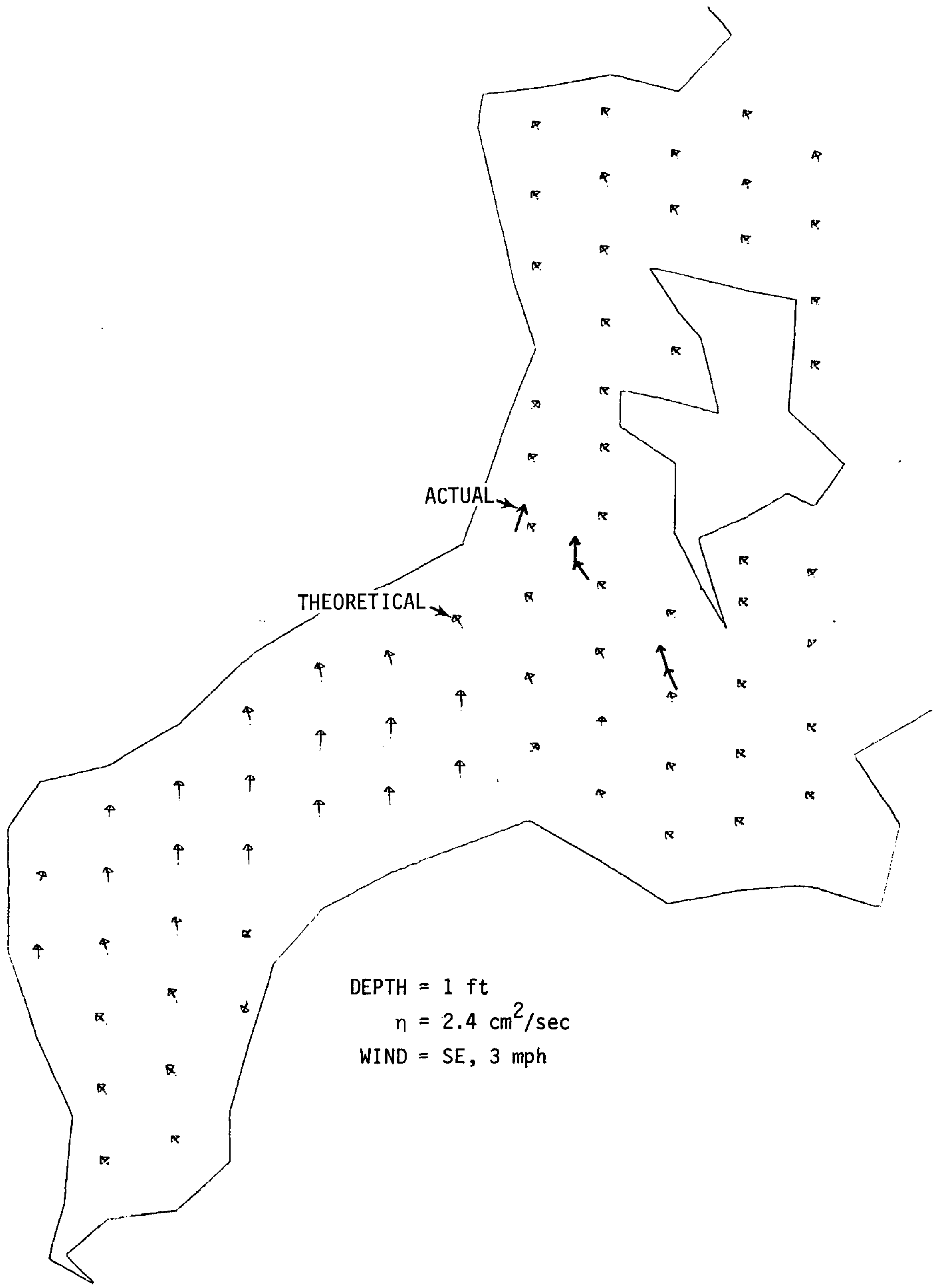


Figure 68. Theoretical and actual circulation patterns in Pistakee Bay, July 17, 1975

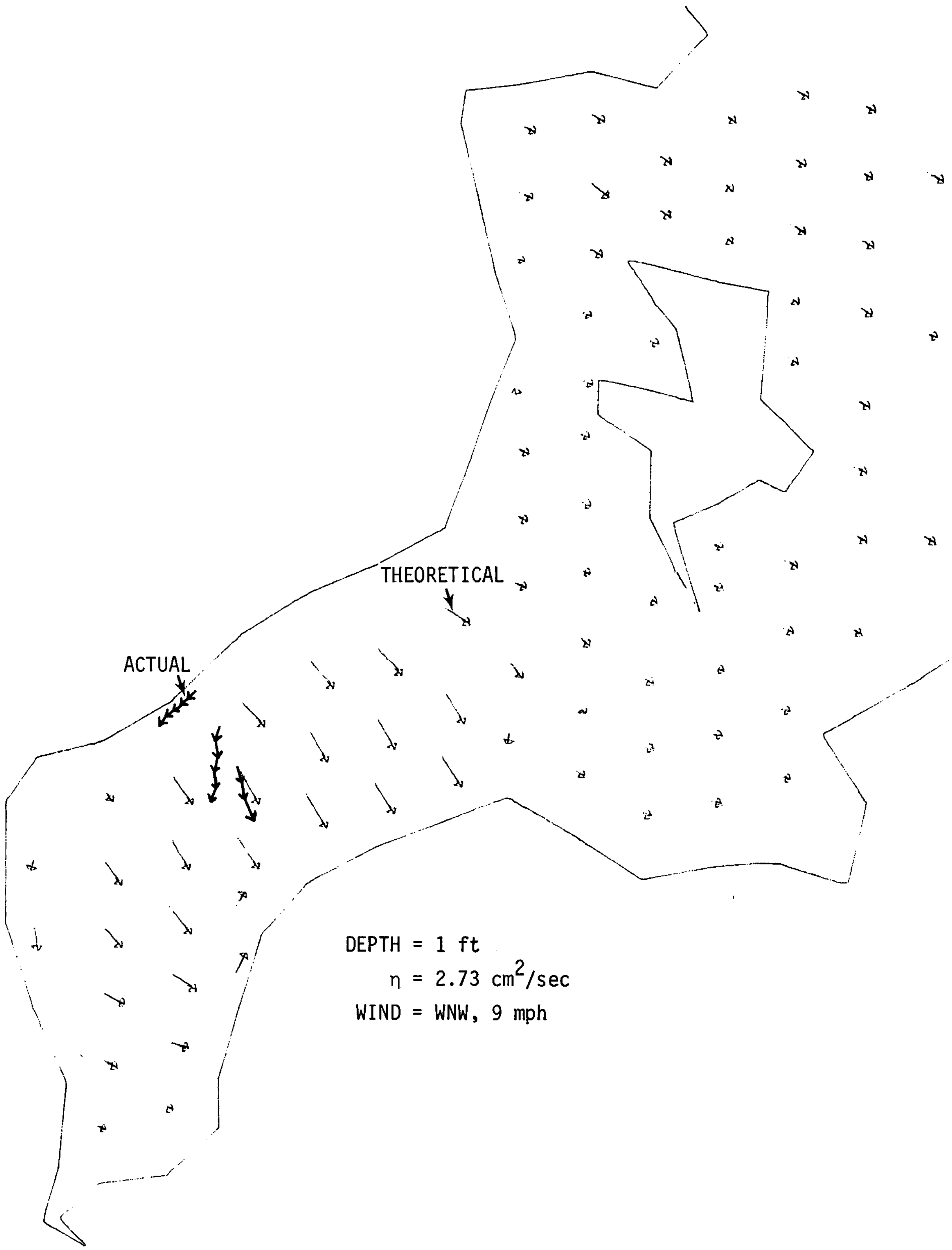


Figure 69. Theoretical and actual circulation patterns in Pistakee Bay, May 15, 1975

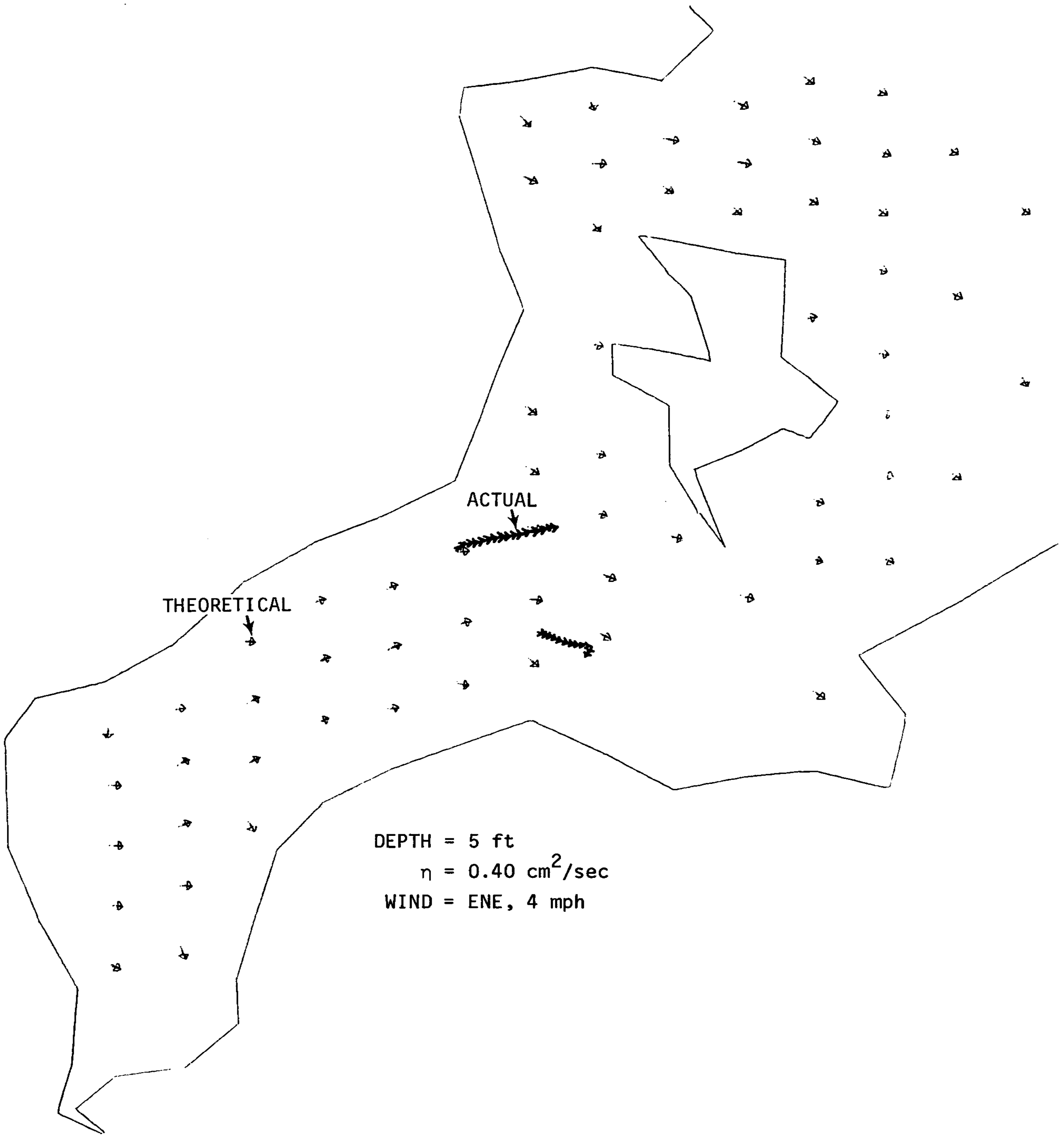


Figure 70. Theoretical and actual circulation patterns in Pistakee Bay, May 16, 1975

SUMMARY OF HYDRAULICS-HYDROLOGY OF LAKES

The hydraulics and hydrology of the Fox Chain of Lakes were developed from previously available data and from extensive new field data. The analyses provided water budgets including inflows and outflows, scour and deposition patterns, theoretical detention times in the various lakes, and circulation patterns in and between the lakes.

Depth-area and depth-storage relationships were developed for each of the lakes. The total storage of the lakes below a mean water surface elevation of 736.5 feet above msl is 38,718 acre-feet and the total surface area is 6844 acres. Pistakee Lake has the largest total surface area and storage capacity among the lakes.

Lake bed elevations in 1975 were determined from 26 cross sections. Comparisons with maps prepared about 20 years earlier showed that in general Grass, Bluff, Petite, Nippersink, and Pistakee Lakes had very little variation over that period. Lake Marie lost some capacity and in one cross section had deposition of 5 to 6 feet. The maximum observed deposition was about 8 feet in Channel Lake. Some erosion and deposition were observed in Lake Catherine. The southwestern part of Fox Lake showed deposition as much as 6 feet, though the northern part of that lake showed neither erosion nor deposition.

Most of the inflow to the lakes is from the Fox River and Nippersink Creek. Minor inflows come from Sequoit and Squaw Creeks and from Lily Lake Drain. A water budget analysis considering all inflows and outflows was made.

Measurements of interflow between the lakes indicated that when inflow rates are high, water generally moves in a southwesterly (downstream) direction toward the Fox River. However, when inflow rates are very low, the water within the lakes wanders about depending on wind velocity and direction before it finally moves downstream through the Fox River. As an example of apparent upstream movement, most of the time water moves from Grass Lake into Lake Marie, and on at least four occasions water was flowing into Channel Lake from Lake Marie.

Theoretical hydraulic detention times were computed for the lakes and combinations of lakes under various assumed conditions of lake capacity and flow. For one set of conditions, detention times varied from 5 days in April to about 21 days in September. During low flows, a considerable amount of time is needed to pass the flow through all of the lakes.

Extensive circulation data were collected at seven locations on four different occasions during 1975. In areas of negligible inflows, the circulation patterns in all of the lakes were governed by the wind. Even in areas of significant inflow, the wind dominated the movement of water as soon as diffusion of the inflow jets reduced the flow velocity. Generally, the surface water moves with the wind, and there is some return flow below certain depths. Measured wind-generated current profiles in the lakes indicated they generally follow a logarithmic distribution. The wind factor defined as the ratio of surface water velocity to wind velocity varied from 2 to 7 percent.

Theoretical circulation patterns in Pistakee Bay for a few selected wind conditions were computed by a finite element method. Calcomp plots were developed and the theoretical circulation patterns were compared with the measured circulation patterns. Correlation between the theoretical and measured circulation patterns was found to be good.

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Part 3. Water Quality Problems and Solutions

WATER QUALITY OF THE LAKES AND STREAMS

In order to assess the water quality characteristics, and to estimate the nutrient loads transported by the Fox River and other tributaries to the Fox Chain of Lakes, weekly water samples were obtained at six locations beginning December 4, 1974, and ending on November 26, 1975. Water samples collected from the Fox River at the Illinois Route 173 site, represent the water quality characteristics of the river entering the lake system, and the samples collected at Johnsburg represent the quality of water leaving the lake system. In addition, samples were collected from four other tributaries to the lakes, namely Nippersink Creek, Lily Lake Drain, Squaw Creek, and Sequoit Creek. The tributary sampling sites are shown in figure 71.

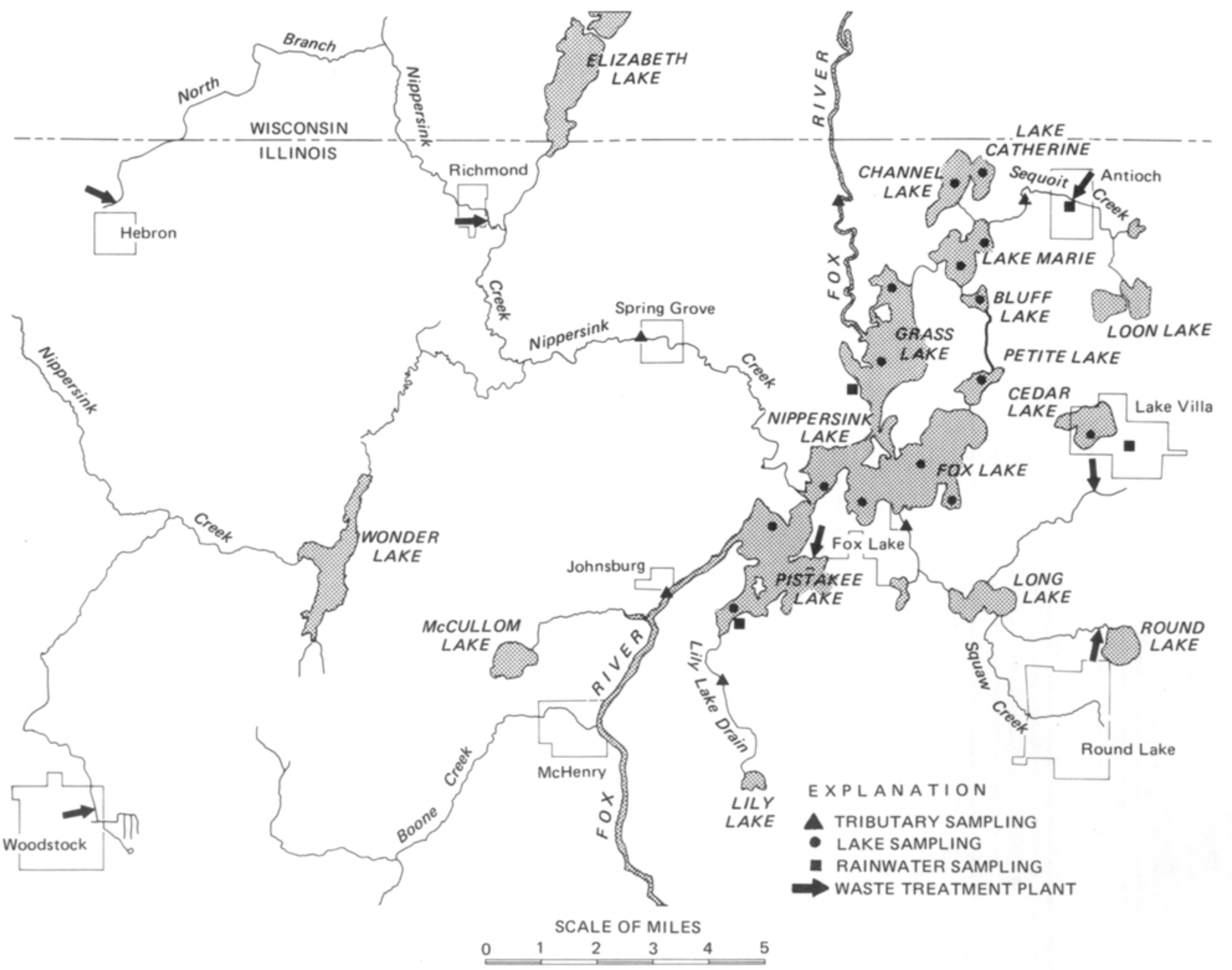

Figure 71. Water quality sampling locations in fox Chain of Lakes.

Water samples were collected in plastic lined bottles and transported to the laboratory through United Parcel Service. Upon receipt, they were kept refrigerated until all of the chemical analyses were completed. Observations were made for temperature and dissolved oxygen concentrations at the time of sample collection.

The daily flow values for the Fox River at Wilmot (about 2 miles upstream of the Illinois Route 173 sampling site) and for Nippersink Creek were obtained from U.S. Geological Survey (USGS) water resources data. Temporary wire-weight gages were installed on Sequoit Creek, Squaw Creek, and Lily Lake Drain. Daily stage observations were recorded for these creeks and daily average flows were estimated from stage-discharge relationships developed by the USGS for these streams. The average daily flow values for the Fox River at Johnsburg were estimated from the flow data developed for the Fox River at McHenry Dam by the Illinois Department of Transportation. The flow at Johnsburg was reduced in proportion to the watershed areas of the Fox River at Johnsburg and McHenry. Table 17 shows the watershed areas of the Fox River and other tributaries relevant to this investigation.

Table 17. Watershed Areas of Fox River and Other Tributaries

Location	*Watershed area (square miles)*
Fox River at Wilmot	868
Fox River at Johnsburg	1,184
Fox River at McHenry	1,253
Lily Lake Drain at Lincoln Road	4.75
Nippersink Creek at Spring Grove	193
Sequoit Creek at Highview Drive	10.74
Squaw Creek at Rollins Road	42.56

Chemical analyses on water samples were performed to determine turbidity, pH, alkalinity, hardness, nitrate–N, Kjeldahl–N, ammonia–N, total silica, total iron, chloride, sulfate, total solids, total dissolved solids, suspended solids, total phosphorus, dissolved orthophosphorus, and algal growth potential. The analyses were made according to the procedures outlined in table 18. Results of determinations are expressed as milligrams per liter (mg/l) except in the case of temperature, turbidity, and pH. Temperature is expressed in Celsius units, turbidity in Formazin turbidity units, and pH is dimensionless.

Algal growth potential is a laboratory procedure to evaluate the ability of filtered water samples to sustain algal growth, under standardized laboratory conditions, following inoculation with mixed algal culture (Wang et al., 1973). It is defined as the algal or organic mass resulting from a 7-day incubation of an algal culture grown on a natural substrate under standardized light and mixing conditions and expressed as milligrams of dry organic mass per liter of sample.

Temperature

Temperature plays a vital role in the rate of chemical reactions and the nature of biological activities in fresh water. Temperature affects the environment of the aquatic medium, e.g., viscosity, reaeration rates, and oxygen capacity. Composition of aquatic communities depends on temperature in addition to other characteristics of their environment.

Table 18. Analytical Procedures

Turbidity	Nephelometric method, using Turner Fluorometer Meter, Model 110. Formazin was used as a standard.
pH	Glass electrode method using Leeds and Northrup 7401 and later Beckman 4500
Alkalinity	Potentiometric method
Hardness	EDTA titrimetric method
Nitrate–N	Chromatropic method (West, 1966)
Kjeldahl–N	Kjeldahl digestion and ammonium determined by phenate method
Ammonium–N	Phenate method
Total silica	Molybdosilicate method
Total iron	Phenonthroline method
Chloride	Argentometric method
Sulfate	Turbidimetric method
Total dissolved solids	Residue on evaporation and filtration at 103 to 105°C
Total phosphorus	Sample was digested with sulfuric-nitric acids mixture and determined by ascorbic acid method
Dissolved orthophosphorus	Ascorbic acid method after filtration through 0.45μ filter paper
Algal growth potential	Method outlined by Wang et al. (1973)

Note: Unless otherwise stated all the methods used were according to Standard Methods (American Public Health Association, 1971)

Organisms have upper and lower thermal tolerance limits, optimum temperature for growth, and temperature limitations for migration and spawning. The Illinois Pollution Control Board (IPCB) stipulated that water temperatures in general should not exceed 15.5°C in the months of December through March and 32.2°C in the months of April through November (Illinois Pollution Control Board, 1972).

The means and ranges of values of temperature and other water quality parameters observed over a period of a year for the Fox River at Illinois Route 173, Lily Lake Drain, Nippersink Creek, Sequoit Creek, Squaw Creek, and the Fox River at Johnsburg are shown in tables 19 through 24. To facilitate comparison of the water quality characteristics for the tributaries to the Fox Chain of Lakes, the mean values are tabulated in table 25.

The observed temperatures in the streams ranged from 0°C in winter to 27.5°C in summer. Because there are no industrial thermal discharges to any of the tributaries, the variations in water temperature are solely due to natural causes. The IPCB regulations with respect to temperature were met by all the streams considered here.

Dissolved Oxygen

The dissolved oxygen (DO) content in water at equilibrium with a normal atmosphere is a function of the temperature and salinity of the water. The ability of water to hold oxygen decreases with increases in temperature or dissolved solids. Natural waters are not generally at equilibrium or at saturation point because temperatures are changing and the physical, chemical, and biological activities are utilizing or liberating oxygen. Inadequate DO in surface waters may contribute to an unfavorable environment for fish and other aquatic life. The absence of DO may give rise to odoriferous products of anaerobic decomposition. The Illinois Pollution Control Board (1972) mandates that the DO in the state's waters shall not be less than 6.0 mg/l during at least 16 hours of any 24-hour period, nor less than 5.0 mg/l at any time.

Table 19. Water Quality Characteristics, Fox River at Route 173

(Concentrations in mg/l)

Parameters	*Number of analyses*	*Mean*	*Range*
Temperature (°C)	51	13.8	0.0-27.0
Dissolved oxygen	49	10.9	5.0-15.0
Turbidity (FTU)	50	10.7	2.6-22.0
pH	52		7.75-8.59
Alkalinity	52	223	120-286
Hardness	53	305	136-386
Nitrate-N	52	1.17	0.07-4.51
Kjeldahl-N	52	1.27	0.21-4.08
Ammonia-N	52	0.22	0.00-0.88
Total silica	52	5.46	0.00-10.97
Total iron	52	0.92	0.08-4.35
Chloride	53	38	21-55
Sulfate	53	56	38-89
Total solids	52	447	294-527
Total dissolved solids	53	408	242-502
Suspended solids	52	38	0-126
Algal growth potential	53	56	1-142
Total phosphorus	53	0.24	0.11-0.79
Dissolved ortho-phosphorus	53	0.11	0.00-0.30

Table 20. Water Quality Characteristics, Lily Lake Drain

(Concentrations in mg/l)

Parameters	*Number of analyses*	*Mean*	*Range*
Temperature (°C)	52	12.8	0.0-26.0
Dissolved oxygen	51	8.9	2.0-15.0
Turbidity (FTU)	50	8.4	2.3-20.0
pH	53		7.93-8.62
Alkalinity	53	257	160-311
Hardness	53	336	136-447
Nitrate-N	52	0.63	0.06-2.70
Kjeldahl-N	52	1.29	0.19-2.64
Ammonia-N	52	0.43	0.03-13.00
Total silica	51	6.67	0.00-12.36
Total iron	52	0.78	0.18-5.96
Chloride	53	12	0-38
Sulfate	53	61	39-97
Total solids	52	443	354-584
Total dissolved solids	53	414	292-510
Suspended solids	52	29	0-182
Algal growth potential	53	25	5-141
Total phosphorus	53	0.10	0.0-0.63
Dissolved ortho-phosphorus	53	0.05	0.0-0.51

Table 21. Water Quality Characteristics,Nippersink Creek Fox River at Route 173

(Concentrations in mg/l)

Parameters	*Number of analyses*	*Mean*	*Range*
Temperature (°C)	52	12.9	0.0-26.0
Dissolved oxygen	49	10.1	5.0-15.0
Turbidity (FTU)	50	11.1	2.3-54.0
pH	52		7.89-8.70
Alkalinity	52	234	142-329
Hardness	53	313	136-457
Nitrate-N	52	1.54	0.16-2.91
Kjeldahl-N	52	1.14	0.31-3.59
Ammonia-N	52	0.29	0.02-2.46
Total silica	52	7.04	0.0-13.20
Total iron	51	1.61	0.22-23.33
Chloride	53	30	17-62
Sulfate	53	59	36-87
Total solids	51	451	342-626
Total dissolved solids	53	409	270-480
Suspended solids	51	43	0-246
Algal growth potential	53	60	11-117
Total phosphorus	53	0.25	0.0-1.78
Dissolved ortho-phosphorus	53	0.16	0.0-1.51

Table 20. Water Quality Characteristics, Sequoit Creek Lily Lake Drain

(Concentrations in mg/l)

Parameters	*Number of analyses*	*Mean*	*Range*
Temperature (°C)	51	12.5	0.0-25.0
Dissolved oxygen	50	8.2	3.0-15.0
Turbidity (FTU)	49	15.9	4.2-196.0
pH	53		7.82-8.47
Alkalinity	53	210	138-349
Hardness	53	261	115-386
Nitrate-N	52	1.03	0.32-4.68
Kjeldahl-N	51	3.46	0.42-16.70
Ammonia-N	52	2.16	0.05-11.10
Total silica	51	9.53	0.00-20.90
Total iron	51	1.64	0.07-27.08
Chloride	53	60	33-126
Sulfate	53	71	34-148
Total solids	51	488	308-1092
Total dissolved solids	52	442	290-678
Suspended solids	51	49	0-770
Algal growth potential	53	101	42-211
Total phosphorus	52	1.27	0.25-3.06
Dissolved ortho-phosphorus	51	1.09	0.0-2.76

The DO concentrations observed in the Fox River upstream of the Fox Chain of Lakes was never less than 5.0 mg/l. The DO concentration in the Fox River at Johnsburg was found to be less than 5.0 mg/l only 4 percent of the time. Lily Lake Drain, which drains a relatively small area of 4.75 square miles of marshland, showed the lowest oxygen concentration recorded. The lowest observed DO concentration in Sequoit and Squaw

Table 23. Water Quality Characteristics, Squaw Creek

(Concentrations in mg/l)

Parameters	*Number of analyses*	*Mean*	*Range*
Temperature (°C)	51	13.5	0.0-27.5
Dissolved oxygen	50	10.1	3.0-16.0
Turbidity (FTU)	50	10.4	3.4-31.0
pH	53		7.98-8.75
Alkalinity	53	201	133-262
Hardness	53	289	102-413
Nitrate-N	52	0.92	0.04-2.37
Kjeldahl-N	52	2.49	0.5-9.89
Ammonia-N	52	1.11	0.03-3.65
Total silica	52	3.77	0.0-9.09
Total iron	52	0.66	0.09-4.28
Chloride	53	36	27-46
Sulfate	53	81	28-114
Total solids	52	449	392-510
Total dissolved solids	53	423	348-490
Suspended solids	52	27	0.0-72.0
Algal growth potential	53	78	10-170
Total phosphorus	53	0.83	0.21-1.94
Dissolved ortho-phosphorus	53	0.67	0.0-1.46

Table 24. Water Quality Characteristics, Fox River at Johnsburg

(Concentrations in mg/l)

Parameters	*Number of analyses*	*Mean*	*Range*
Temperature (°C)	51	13.0	0.0-27.5
Dissolved oxygen	50	11.7	4.0-20.0
Turbidity (FTU)	50	12.9	2.4-28.0
pH	53		7.98-8.72
Alkalinity	53	211	138-302
Hardness	53	286	88-393
Nitrate-N	52	0.86	0.02 -2.39
Kjeldahl-N	52	1.54	0.11-3.44
Ammonia-N	52	0.29	0.01-2.50
Total silica	52	3.91	0.0-9.83
Total iron	52	0.79	0.08-5.16
Chloride	53	35	20-63
Sulfate	53	54	37-72
Total solids	52	418	302-493
Total dissolved solids	53	386	266-460
Suspended solids	52	33	0-88
Algal growth potential	53	48	2-120
Total phosphorus	53	0.27	0.07-1.77
Dissolved ortho-phosphorus	53	0.13	0.0-1.47

Table 25. Comparison of Mean Values of Water Quality Characteristics of Tributaries to the Fox Chain of Lakes

(Concentrations in mg/l)

Parameters	*Fox River at Route 173*	*Lily Lake Drain*	*Nippersink Creek*	*Sequoit Creek*	*Squaw Creek*	*Fox River at Johnsburg*
Temperature (°C)	13.8	12.8	12.9	12.5	13.5	13.0
Dissolved oxygen	10.9	8.9	10.1	8.2	10.1	11.7
Turbidity (FTU)	10.7	8.4	11.1	15.9	10.4	12.9
pH	7.75-8.59	7.93-8.62	7.89-8.70	7.82-8.47	7.98-8.75	7.98-8.72
Alkalinity	223	257	234	210	201	211
Hardness	305	336	313	261	289	286
Nitrate-N	1.17	0.63	1.54	1.03	0.92	0.86
Kjeldahl-N	1.27	1.29	1.14	3.46	2.49	1.54
Ammonia-N	0.22	0.43	0.29	2.16	1.11	0.29
Total silica	5.46	6.67	7.04	9.53	3.77	3.91
Total iron	0.92	0.78	1.61	1.64	0.66	0.79
Chloride	38	12	30	60	36	35
Sulfate	56	61	59	71	81	54
Total solids	447	443	451	488	449	418
Total dissolved solids	408	414	409	442	423	386
Suspended solids	38	29	43	49	27	33
Algal growth potential	56	25	60	101	78	48
Total phosphorus	0.24	0.10	0.25	1.27	0.83	0.27
Dissolved orthophosphorus	0.11	0.05	0.16	1.09	0.67	0.13

Creeks was 3.0 mg/l. The sampling stations on these two creeks were in close proximity to municipal waste treatment plants.

With the exception of the Fox River upstream of the lakes and Nippersink Creek, observed DO concentrations were on several occasions less than 5 mg/l. In Sequoit Creek and Lily Lake Drain, 10 percent of the observations were less than 5 mg/l. The average DO concentrations were high in all of the streams and each of them exhibited a high degree of supersaturation.

Turbidity

Turbidity in water is a measure of its ability to scatter and absorb light. Light scattering property of a liquid medium is dependent upon its suspended solids content as well as the particle size and shape. Wang and Brabec (1969) observed a high degree of correlation between turbidity and suspended solids concentration in the Illinois River. Silt and clay, derived from soil erosion, constitute the major components of turbidity causing particulate matter in streams.

Soil erosion and the concommitant sedimentation are dependent on factors such as intensity and duration of rainfall, soil characteristics, slope of the drainage basin, cropping practices, etc. Not only does erosion result in a loss of valuable top soil from agricultural lands, it also has deleterious effects on water quality, recreational enjoyment, fish propagation, and the management of impoundments.

The mean turbidity values measured in the tributaries to the Fox Chain of Lakes ranged from 8.4 to 15.9 FTU which are much less than the values determined by the Water Survey for the Spoon River in central Illinois. Turbidity in the Spoon River, measured at 5 different sites over a period of 2 years, ranged from 147 to 3470 FTU. The Spoon River is unimpounded along its entire length, whereas in the Fox River, Squaw Creek, and Nippersink Creek impoundments exist upstream of the sampling sites. Lily Lake Drain's small watershed is an undisturbed area and Sequoit Creek drains a small but mostly urban area.

pH, Alkalinity, and Hardness

The logarithm (base 10) of the reciprocal of the hydrogen ion concentration is commonly designated by the symbol pH. The concentration of weakly disassociated acids and bases markedly affects the pH value and the ease with which it can be altered. The presence of carbonates, phosphates, and similar ions give water a buffering capacity. Because pH values are reciprocals of logarithm of hydrogen ion concentrations, arithmetic means of pH values were not computed and only the ranges of values observed are shown in tables 19 through 25. The pH values for all the streams ranged between 7.75 and 8.75 which are within the range stipulated by IPCB.

The alkalinity of a water is its capacity to accept protons and is generally imparted by the bicarbonate, carbonate, and hydroxide components. The species composition of alkalinity is a function of pH and mineral composition. The carbonate equilibria, in which carbonate and bicarbonate ions and carbonic acid are in equilibrium, are the predominant chemical system present in natural waters. The IPCB does not stipulate any standard for alkalinity in natural waters. High alkalinity waters have been reported to have a distinctly unpleasant taste because they are generally associated with high values of pH, hardness, and dissolved solids (McKee and Wolf, 1963).

Any substance in water that will form an insoluble precipitate with soap causes hardness. Hardness is defined as the sum of the polyvalent cations expressed as the equivalent quantity of calcium carbonate. As iron, manganese, copper, barium, lead, zinc, and other trace elements are seldom present in appreciable concentrations in natural waters, hardness is attributable principally to calcium and magnesium.

Hard waters have no demonstrable harmful effects upon the health of consumers. The detrimental effects of hardness include excessive soap consumption, formation of scums and cruds in homes and laundries, and the formation of scales in boilers, hot water heaters, pipes, and utensils. The major detrimental effect of hardness is economic.

The alkalinity and hardness values observed in the Fox River and other tributaries are typical of surface waters in the Midwest. The mean alkalinity values observed in these streams varied from 201 to 257 mg/l which compare well with the mean values of 222 to 265 mg/l for the Spoon River at 5 locations. Hardness values ranged from 261 to 336 mg/l in the Fox River and other tributaries, whereas mean hardness values varied from 332 to 346 mg/l at 5 of the Spoon River sampling sites.

Nitrogen

Nitrogen in natural waters is generally found in the form of nitrate, organic nitrogen, and ammonia nitrogen. Nitrates are the end product of the aerobic stabilization of organic nitrogen, and as such they occur in polluted waters that have undergone self purification or aerobic treatment processes. Nitrates also occur in percolating groundwaters as a result of the application of fertilizer at surface sources. In surface waters and groundwaters, ammonia nitrogen results from the decomposition of nitrogenous organic matter, being a constituent of the complex nitrogen cycle. Ammonia nitrogen could also result from municipal and industrial waste discharges to streams and rivers.

The concerns for nitrogen as a contaminant in water bodies are twofold. First, because of adverse physiological effects on infants and because the traditional water treatment processes have no effect on the removal of nitrate, concentrations of nitrate plus nitrite as nitrogen are limited to 10 mg/l in public water supplies. Second, a concentration in excess of 0.3 mg/l is considered sufficient to stimulate nuisance algal blooms (Sawyer, 1974). The IPCB stipulates that ammonia nitrogen and nitrate plus nitrite as nitrogen should not exceed 1.5 and 10.0 mg/l, respectively.

The range of values of nitrate observed in the Fox River was 0.02 to 4.51 mg/l. In the streams, the overall range was 0.02 to 4.68 mg/l. The range of nitrate concentrations in other streams in Illinois was more extensive: 0.0 to 11.8 mg/l in the Spoon River, 0.05 to 16.08 mg/l in Six Mile Creek near Bloomington, and 3.70 to 10.17 mg/l in the Middle Fork Vermilion River. The observed magnitudes of nitrate were also 5 to 10 orders of magnitude smaller than those observed for central Illinois streams. The ammonia concentrations in Sequoit and Squaw Creeks were much higher than in the other streams in the area. This is mainly because of the closeness of municipal waste treatment discharges to the water sampling sites in these two creeks. The IPCB ammonia standard was found to be violated about 40 percent of the time in these two creeks.

The nitrate and ammonia loads transported by the Fox River and other streams are shown in table 26. The nitrogen load transported in terms of pounds per day was estimated by:

$$\text{Pounds per day} = \text{flow (cfs)} \times \text{concentration (mg/l)} \times 5.39$$

The means of average daily flows for a 7-day period were used in estimating the load. In the case of Lily Lake Drain, Sequoit Creek, and Squaw Creek, the overall means of flow observations were used on days when flow values could not be estimated for lack of stage

Table 26. Nitrate and Ammonia-N Transport by Different Streams

Streams	Nitrate lbs/day	Nitrate lbs/sqmi/day	Nitrate lbs/ac/yr	Ammonia-N lbs/day	Ammonia-N lbs/sqmi/day	Ammonia-N lbs/ac/yr
Fox River at Route 173	5354	6.17	3.50	1010	1.16	0.66
Lily Lake Drain	14	2.95	1.68	3	0.66	0.38
Nippersink Creek	1371	7.10	4.05	286	1.48	0.84
Sequoit Creek	52	4.87	2.78	54	5.06	2.89
Squaw Creek	168	3.95	2.25	211	4.96	2.83
Fox River at Johnsburg	4738	3.93	2.24	1391	1.15	0.66

observations. The stage observations for Sequoit Creek were not as regular as those for Lily Lake Drain or Squaw Creek. When a number of stage observations were missing in Sequoit Creek, the flows were estimated from Lily Lake Drain flows adjusted for drainage area and wastewater discharge from Antioch waste treatment plant. Information on pounds per square mile per day (lbs/sq mi/day) and pounds per acre per year (lbs/ac/yr) is also shown in table 26.

The mean nitrate concentration was highest in the Fox River. Presumably this is due to the fertilizer applied to agricultural lands and the self purification of municipal waste discharges in the upper reaches of the Fox River in Wisconsin. The nitrate load transported by the Fox River into the Chain of Lakes is about 5350 lbs/day. About 1370 lbs/day is transported by Nippersink Creek.

Sequoit and Squaw Creeks, as shown in table 25, have high mean concentrations of ammonia due mainly to the closeness of the stream sampling sites to waste treatment plants. The mean ammonia concentration in Lily Lake Drain, which drains mostly marshlands, is 0.43 mg/l. This is higher than the mean ammonia concentrations observed in Nippersink Creek and Fox River. However, mean nitrate concentrations in the latter two streams were significantly higher.

For purposes of comparison of nutrient transports by the different streams, unit load factors were considered. Nippersink Creek showed the highest rate of nitrate transported (4.05 lbs/ac/yr). This was followed by 3.50 lbs/ac/yr for Fox River at the Route 173 site. Unit ammonia-N loads transported by Sequoit and Squaw Creeks were several orders of magnitude higher than for the other streams.

The unit nitrate loads transported by all these streams are considerably less than the average nitrate contribution of 21 lbs/ac/yr observed on the 1030-square-mile drainage area for the Kaskaskia River upstream of Shelbyville (Harmeson and Larson, 1970).

Silica

The element silicon is not found free in nature. It occurs as silica or silicates, and is present in natural waters in soluble and colloidal forms. A silica cycle occurs in many bodies of water containing organisms such as diatoms that utilize silica in their skeletal structure. The silica removed from the waters may be slowly returned by re-solution of the dead organisms.

In concentrations found in natural or treated waters, silica does not appear to cause any adverse physiological effects. However, silica in high concentrations may cause difficulties arising from turbidity. Silica in boiler feed waters results in scaling.

Concentrations of silica in the Fox River and other tributaries do not appear to be excessive and are within the range cited in the literature (American Public Health Association, 1971). The mean and range of values observed for Sequoit Creek appear to have been influenced to a great extent by the Antioch waste treatment plant. Evans (1968), considering the addition of common ions from domestic use of water, documented that there is nearly a two- to three-fold increase in silica concentration in municipal wastewaters compared to the tap waters in their respective locations.

Total Iron

Iron ranks next to aluminum in abundance of metals in the earth's mantle. In spite of this, natural waters contain variable but minor amounts of iron. Under reducing conditions, iron exists in the ferrous state and is relatively soluble. In the absence of complex forming ions and upon exposure to oxygen, hydrated ferric oxide is formed which is insoluble. These precipitates tend to aggregate, flocculate, and settle or be adsorbed on surfaces. Hence the concentration of soluble iron in water with dissolved oxygen present is seldom high.

Generally accepted standards for iron in surface water supply sources is 0.3 mg/l or less. The limit is based on aesthetic and taste considerations rather than physiological reasons. The IPCB limits the concentration of total iron in streams to 1.0 mg/l.

The mean concentrations of total iron in Nippersink and Sequoit Creeks were found to be higher than the IPCB limits. All the other streams had mean total values less than the stipulated limits. On a few occasions, the concentration of iron observed in Nippersink and Sequoit Creeks exceeded 20 mg/l. In all the other creeks, the maximum observed was in the order of 5 mg/l. In Nippersink and Sequoit Creeks, observed iron concentrations exceeded the IPCB limit 41 and 31 percent of the time, respectively. In the other streams, the IPCB standard was violated about 19 to 30 percent of the time.

Chloride and Sulfate

Chlorides are found in practically all natural waters. They are conservative substances and have been used as a tracer element in engineering studies of surface waters and treatment unit processes. On the other hand, sulfate can occur as the final oxidized stage of sulfide, sulfites, and thiosulfites, and as the oxidized state of organic matter of the sulfur cycle. Sulfate can serve as a source of energy for sulfur bacteria. Both chlorides and sulfates in streams could be of mineral origin. They may emanate from municipal and industrial waste discharges as well.

Chloride and sulfate are the predominant anions causing what is known as 'permanent hardness.' Because of the salty taste caused by the chloride ion and the cathartic effect of sulfate, the IPCB limits each of these two constituents to 250 mg/l in public water supplies.

The mean chloride concentration of 12 mg/l in Lily Lake Drain was the lowest of all the observed mean values, reflecting the chloride contribution independent of cultural developments. The average concentration in Sequoit Creek, 60 mg/l, was the highest and was influenced by the effluent discharge from the Antioch waste treatment plant. All the other streams showed average values varying from 30 to 38 mg/l. The mean sulfate concentrations varied in these streams from 54 to 81 mg/l, as shown in table 25. The observed concentrations of chloride and sulfate in all the streams were well within the IPCB limits.

Total Solids, Total Dissolved Solids, and Suspended Solids

Total solids consist of dissolved solids and suspended solids. These could be further classified into the categories of inorganic and organic fractions, but only the two major forms of solids are dealt with here.

In natural waters, the total dissolved solids (TDS) consist mainly of bicarbonates, chlorides, sulfates, nitrates and phosphates of sodium, calicum, magnesium, and potassium with traces of iron, manganese, and other substances. The extent and composition of minerals occurring in natural waters is influenced by the geochemistry, morphometry, and land use patterns of the watershed contributing to the surface water and groundwater resources. The IPCB has stipulated a limit of 1000 mg/l for TDS in natural waters. High mineral content in water supplies render them unpalatable and such waters have a laxative action on new users.

Suspended solids consist normally of eroded silt, clay, organic detritus, and plankton. Man's activities alter and augment the suspended solids in surface waters by the discharge of liquid wastes from communities and industries, by increased erosion from cultivated areas and other results. A strong association between suspended solids and turbidity in natural waters has been reported (Wang and Brabec, 1969).

The mean values of total solids and total dissolved solids at all six sampling locations for stream quality characteristics are comparable. The temporal variations in these two parameters are moderate in all the streams except for one extremely high value observed in Sequoit Creek. The suspended solids concentrations exhibited a wider range of values, varying from 0 to about 250, with the exception of one unusually high value of 770 mg/l observed in Sequoit Creek.

Algal Growth Potential

Algal growth potential (AGP) is a bioassay procedure carried out under controlled laboratory conditions to evaluate the growth response of an algal inoculum in test waters. Traditionally, the primary productivity of a water body is predicted on the basis of the chemistry of the water such as alkalinity, dissolved solids, and particularly the concentrations of nutrients like nitrogen and phosphorus. The AGP procedure, developed by the Illinois State Water Survey (Wang et al., 1973), has been applied successfully to various surface water bodies in Illinois. It is a diagnostic tool, used to supplement the existing knowledge about the predictability of primary production in natural waters.

Evans and Schnepper (1974) reported that the mean AGP values observed for 14 streams in Illinois ranged from 14 to 77. The highest value had been observed for the Illinois River. The mean AGP values observed at the six sampling sites shown in table 25 indicate that Lily Lake Drain had the lowest value with 25 mg/l and Sequoit Creek the highest with 101 mg/l. Interestingly, the mean dissolved orthophosphorus, which is the readily available form of phosphorus, was the lowest at Lily Lake Drain and highest at Sequoit Creek. Inorganic nitrogen concentration, likewise, was the lowest and highest, respectively, at these two sites.

The range of values for AGP in all the streams, except Sequoit Creek, was considerable, varying from 1 to 170 mg/l. AGP values of less than 20 mg/l were generally associated with dissolved orthophosphorus values of 0.3 mg/l or less. All other values were equal to or greater than 0.14 mg/l, except for one observation of 0.0 mg/l of dissolved orthophosphorus in Sequoit Creek. The availability of unlimited quantities of inorganic nitrogen and phosphorus, due to the proximity of the Antioch municipal waste treatment plant discharges, resulted in high observed AGP values for Sequoit Creek.

It is difficult to relate the laboratory AGP values to naturally occurring algal growth conditions. Diurnal changes in factors such as turbidity, temperature, or sunlight could alter the growth potential in natural waters.

Phosphorus

Phosphorus as phosphate may occur in surface waters or groundwaters as a result of leaching from minerals or ores, in natural processes of degradation, or from agricultural drainage. Phosphorus is an essential nutrient for plant and animal growth and, like nitrogen, it passes through cycles of decomposition and photosynthesis.

Because phosphorus is essential to the plant growth processes, it has become the focus of attention in the entire eutrophication issue. With phosphorus being singled out as probably the most limiting nutrient and the one most easily controlled by removal techniques, various facets of phosphorus chemistry and biology have been extensively studied in natural environments. To prevent biological nuisances, IPCB (1972) stipulates, "Phosphorus as P shall not exceed 0.05 mg/l in any reservoir or lake, or in any stream at the point where it enters any reservoir or lake."

In any river system, the two aspects of interest for phosphorus dynamics are the phosphorus concentration and phosphorus flux (concentration times flow rate) as functions of time and distance. The concentration itself indicates the possible limitations that this nutrient can place on vegetative growth in the stream, and the phosphorus flux is a measure of phosphorus transport rate at any point in the river.

In the Illinois River, Wang and Evans (1970) found that the river flow rate was the principal variable that affected the soluble orthophosphorus concentration and that the relationship was an inverse one. A high flow rate was associated with a lower orthophosphorus concentration, i.e., a dilution effect. Unlike nitrate nitrogen, phosphorus applied as fertilizer is held tightly to the soil. Most of the phosphorus carried into streams and lakes from runoff over cropland will be in the particulate form. On the other hand, the major portion of phosphate-phosphorus emitted from municipal sewer systems is in a dissolved form. Consequently, the form of phosphorus, namely particulate or dissolved, is indicative of its source to a certain extent.

The means and range of values for total phosphorus and dissolved orthophosphorus for the Fox River and other tributaries are shown in tables 19 through 25. Sequoit Creek exhibited the highest mean, the widest range of values, and the highest of all the minimum values for total phosphorus and dissolved orthophosphorus. These high values were caused by the waste treatment plant at Antioch. The lowest phosphorus concentration was observed for Lily Lake Drain, which is the least affected by human cultural activities. The impact of different phosphorus concentrations in these streams on laboratory AGP values has been discussed earlier. The ratio of means of dissolved orthophosphorus to total phosphorus was the highest in Sequoit Creek at 0.86, followed by 0.81 for Squaw Creek. The ratio was the lowest for the Fox River at Illinois Route 173.

Table 27 shows the flux in pounds per day (lbs/day) and unit load factors expressed as pounds per square mile per day (lbs/sq mi/day) and pounds per acre per year (lbs/ac/yr) of total phosphorus and dissolved orthophosphorus at the six stream sampling sites. The unit load factor estimated for the Kaskaskia River at Shelbyville was 0.17 lbs/ac/yr (Engelbrecht and Morgan, 1959). The unit load factor for the Spoon River varied from 0:47 lbs/ac/yr near the headwaters to 0.26 lbs/ac/yr near its confluence with the Illinois River. It decreased

Table 27. Phosphorus Transport by Different Streams

Streams	Total phosphorus			Dissolved orthophosphorus		
	lbs/day	*lbs/sq mi/day*	*lbs/ac/yr*	*lbs/day*	*lbs/sq mi/day*	*lbs/ac/yr*
Fox River at Route 173	1187	1.37	0.78	428	0.49	0.28
Lily Lake Drain	1.7	0.36	0.20	0.45	0.95	0.05
Nippersink Creek	223	1.16	0.66	119	0.62	0.35
Sequoit Creek	61	5.68	3.24	33	3.07	1.75
Squaw Creek	131	3.08	1.76	107	2.51	1.43
Fox River at Johnsburg	1093	0.91	0.52	509	0.42	0.24

gradually in the downstream direction and thus exhibited the dilution effect. The phosphorus transport data for the Fox River and other tributaries to the Fox Chain of Lakes, with the exception of Lily Lake Drain, appear to be several orders of magnitude higher when compared with the Kaskaskia River or Spoon River data.

The phosphorus concentrations in the Fox River, Sequoit Creek, and Squaw Creek exceeded the IPCB limit all the time. Phosphorus concentrations were less than 0.05 mg/l for Nippersink Creek in only 3 out of the 53 observations over a period of 1 year. Phosphorus concentrations exceeded 0.05 mg/l for Lily Lake Drain in 32 of the 53 observations.

WASTEWATER TREATMENT FACILITIES AND EFFLUENT CHARACTERISTICS

The shoreline around the Fox Chain of Lakes has been intensely developed with permanent year-round homes, condominiums, hotels, restaurants, marinas, and service facilities ancillary to boating and other recreational activities. In addition, approximately 30 miles of dredged channels have been created to develop waterfront properties. It is estimated that 85 percent of the developed properties on the shores of the Chain are in unincorporated areas and remain on septic tank disposal systems. Detailed surveys conducted by the health departments of McHenry and Lake Counties indicate there are about 6900 residential units being served by household septic tank systems. In addition, some schools, churches, hotels, motels, and commercial enterprises are similarly served.

An estimated waste flow of 15 5,000 gallons per day (gpd) is applied to septic tank-tile field systems around the lakes and in the immediate vicinity (William Mellon and Richard A. Wissell, personal communication, 1976). Analyses of the effluent quality characteristics of the septic tank systems were not made. However, the impact of this method of waste disposal on the water quality of the lakes, with specific reference to the nutrients nitrogen and phosphorus, is discussed in the section *"Nutrient Budget."*

There are seven municipal waste treatment plants (WTP) located within the Chain's watershed that discharge directly or indirectly into the lake system. The WTPs are located at Hebron, Richmond, Woodstock Northside, Lake Villa, Round Lake Sanitary District, Antioch, and Fox Lake. Of these, only the Fox Lake WTP discharges directly to the lake system; all of the other treatment plants discharge to streams which are tributary to the lakes. Treated effluent from the Woodstock Die Casting Company and a Hebron meat packing industry are the only significant industrial discharges in the area. The relative locations of these waste treatment plants with respect to the Chain of Lakes along with the tributaries to the lakes are shown in figure 71. Information pertaining to the population presently served,

Table 28. Municipal Waste Treatment Plants Discharging Directly or Indirectly into the Chain of Lakes

Municipality served	Receiving stream	Population 1975	Design flow (mgd)	Type of treatment *	Estimated efficiency (%) **
Hebron	Nippersink	780	0.110	TF	80-85
Richmond	Nippersink	1,150	0.375	Biodisc	80-85
Woodstock (north-side	Nippersink	10,230	3.500	AS	85-90
Lake Villa	Eagle Creek-Squaw Creek	1,090	0.300	AS	85-90
Round Lake Sanitary District	Squaw Creek	16,350	1.600	TF, P	85-90
Antioch	Sequoit Creek	3,680	1.000	AS, P	85-90
Fox Lake	Myers Bay-Pistakee Lake	4,510	0.720	TF	80-85

* TF = trickling filter; AS = activated sludge; P = polishing ponds
**Estimated efficiency in 5-day BOD removal

design flow, type of treatment provided, and other details regarding the seven WTPs are provided in table 28. Details of plant improvements, modifications, future expansions, and the current effluent quality characteristics of each of these wastewater treatment facilities are reported individually here.

In order to assess the wastewater effluent characteristics, four separate visits were made to the plants to collect samples for chemical analyses. Grab samples were collected from treatment plants employing polishing ponds or oxidation lagoons. In all other cases composited samples were obtained. The samples, representative of plant flow weighted samples, were collected at 30-minute intervals during the period 9 a.m. to 3 p.m. The samples were stored in ice during collection and transportation and refrigerated until the necessary analyses were performed. Alkalinity, pH, nitrate-N, Kjeldahl-N, ammonia-N, chemical oxygen demand (COD), total solids, dissolved solids, total phosphorus, and dissolved orthophosphorus were determined. Methods of chemical analyses performed are reported in the section *"Water Quality of the Lakes and Streams."*

Hebron WTP

The municipal wastewater is treated in a 0.11 mgd Imhoff trickling filter plant. The treatment facility for the meat-packing industry consists of two anaerobic lagoons and two oxidation lagoons. The facility is designed to handle trade wastes from a meat-packing operation processing 200 head of cattle per day. The industrial waste treatment facility is owned and operated by the city of Hebron. Tables 29 and 30 show the chemical characteristics of the four samples obtained at these plants and also the means of the observed values. The trickling filter plant discharges higher concentrations of inorganic nitrogen and phosphorus than the lagoons. The ratios of dissolved orthophosphorus to total dissolved phosphorus are 0.83 and 0.63, respectively, for the trickling filter plant and the lagoons.

In the near future, waste flows from the municipal and industrial facilities are to be combined and treated by the contact stabilization activated sludge process. The final effluent, after chlorination, is to be disposed of by spray irrigation. The sludge generated at the treatment plant is also to be used in spray irrigation. This scheme has been approved by the Illinois Environmental Protection Agency.

Table 29. Hebron Trickling Filter Plant Effluent Characteristics

(Concentrations in mg/l)

Parameters	*Dates of observations*				
	12/12/74	*3/25/75*	*8/11/75*	*11/6/75*	*Mean*
pH	7.85	8.06	7.88	8.31	
Alkalinity	333	364	386	361	361
Nitrate-N	13.51	11.23	5.91	17.14	11.95
Kjeldahl-N	10.36	44.64	4.44	17.0	19.11
Ammonia-N	7.77	20.00	3.64	8.82	10.05
COD	52.23	86.08	72.44	71.18	70.48
Total solids	996	1165	1155	1262	1145
Dissolved solids	949	1105	1128	1192	1094
Total phosphorus	8.77	10.79	12.89	21.70	13.54
Dissolved orthophosphorus	8.49	9.62	11.32	12.58	10.50
Dissolved orthophosphorus/total dissolved phosphorus	0.97	0.89	0.88	0.58	0.83

Table 30. Hebron Lagoon Effluent Characteristics

(Concentrations in mg/l)

Parameters	*Dates of observations*				
	12/12/74	*3/25/75*	*8/11/75*	*11/6/75*	*Mean*
pH	8.38	7.59	9.53	8.75	
Alkalinity	400	413	164	274	313
Nitrate-N	1.60	1.37	0.47	0.66	1.03
Kjeldahl-N	20.81	17.68	13.80	25.00	19.32
Ammonia-N	16.27	10.70	0.06	0.62	6.91
COD	118.04	104.64	229.29	312.35	191.08
Total solids	1445	1057	1678	1894	1519
Dissolved solids	1320	1004	1366	1586	1319
Total phosphorus	3.59	6.31	2.44	2.96	3.83
Dissolved orthophosphorus	2.43	5.73	0.75	1.79	2.68
Dissolved orthophosphorus/total dissolved phosphorus	0.68	0.91	0.31	0.60	0.63

Table 31. Richmond WTP Effluent Characteristics

(Concentrations in mg/l)

Parameters	*Dates of observations*				
	12/12/74	*3/25/75*	*8/11/75*	*11/6/75*	*Mean*
PH	7.73	8.00	7.78	7.65	
Alkalinity	333	324	369	349	344
Nitrate-N	8.02	4.69	0.54	5.89	4.79
Kjeldahl-N	15.84	15.45	15.60	3.30	12.55
Ammonia-N	9.67	15.40	8.89	0.80	8.69
COD	59.70	43.88	59.21	47.06	52.33
Total solids	942	1079	793	1066	970
Dissolved solids	888	1044	788	1066	947
Total phosphorus	7.16	4.29	5.37	20.13	9.24
Dissolved orthophosphorus	5.62	3.51	3.55	9.59	5.57
Dissolved orthophosphorus/total dissolved phosphorus	0.78	0.82	0.66	0.48	0.69

With the proposed disposal system, nutrient emissions, particularly that of phosphorus, will be totally eliminated from Nippersink Creek.

Richmond WTP

The expansion and modification of this treatment plant with a design flow of 0.375 mgd was completed in 1975. Secondary treatment is carried out with 4-stage biodisc units following Imhoff tanks. The biodisc units are similar to trickling filters where a fixed biological mass is exposed alternately to the dissolved organic matter and air. The secondary solids are returned to the Imhoff tanks for anaerobic digestion. The treated effluent is chlorinated in an aerated chlorine contact chamber prior to discharge into Nippersink Creek. Digested solids are dried in sludge drying beds and disposed by landfill. Provision is made for adding alum between the third and fourth stages of the biodiscs for phosphorus precipitation and removal in the final clarifiers. Phosphorus reduction from the wastewater effluent is yet to be practiced. Table 31 shows the wastewater effluent characteristics for this plant.

Woodstock Northside WTP

This treatment plant is in the process of expansion and upgrading. Waste flow is given the conventional activated sludge treatment after grit and primary solids removal. Chemicals will be added for phosphorus removal prior to the secondary clarification stage. The waste flow will pass through two polishing ponds, then will be chlorinated and discharged. Solids generated in the treatment process will be aerobically digested and dried in sludge drying beds prior to ultimate disposal. At the time of sampling for wastewater effluent characterization, the phosphorus removal step was not practiced. Results of the chemical analyses of the effluent samples pertaining to this treatment plant are shown in table 32.

Woodstock Die Casting Company WTP

The company is primarily engaged in nonferrous die casting and plating operations. The municipal and industrial waste streams generated in the plant are collected separately with the municipal waste discharged to the city sewers. Different types of trade wastes are segregated initially and given pretreatment before being pumped to the central waste treatment plant. For example, chromium bearing rinse waters are treated by sulfonation, cyanide bearing waste waters are chlorinated to oxidize cyanide, and finally acid and alkaline rinse waters are diverted to a collection sump for neutralization, all prior to being pumped to the central collection tank at the treatment plant.

The collected waste streams are treated through a neutralizing tank using an automatic pH controlled feed system for the addition of lime or sulfuric acid. The neutralized water then flows through upflow clarifiers where alum and polyelectrolytes are added. The clarified waters are filtered through dual media filters before being discharged to the receiving stream. The sludge is initially thickened in a sludge thickener with the addition of polyelectrolytes. The sludge is then dewatered in a vacuum filter using diatomaceous earth as a filtering aid. The dewatered sludge is disposed of by landfill.

The company has plans to increase the number of upflow clarifiers by two, thus doubling the retention time in the clarifiers. The chemical characteristics of the final waste effluent from the treatment plant are shown in table 33.

Table 32. Woodstock Northside WTP Effluent Characteristics

(Concentrations in mg/l)

Parameters	*Dates of observations* 12/12/74	3/25/75	8/11/75	11/6/75	*Mean*
PH	7.57	7.85	7.72	8.18	
Alkalinity	404	337	369	353	366
Nitrate-N	0.31	0.65	0.12	1.12	0.55
Kjeldahl-N	13.22	7.67	11.65	35.9	17.11
Ammonia-N	12.25	5.97	10.04	23.94	13.05
COD	35.02	67.17	45.98	42.35	47.63
Total solids	1352	1134	1150	1556	1298
Dissolved solids	1308	1075	1136	1516	1259
Total phosphorus	7.29	4.94	7.95	10.06	7.56
Dissolved orthophosphorus	7.23	3.94	6.02	4.72	5.63
Dissolved orthophosphorus/total dissolved phosphorus	0.99	0.80	0.86	0.47	0.78

Table 33. Woodstock Diecasting Company Effluent Characteristics

(Concentrations in mg/l)

Parameters	*Dates of observations* 4/28/75	8/11/75	11/5/75	*Mean*
pH	9.69	9.42	9.10	
Alkalinity	231	324	298	284
Nitrate-N	1.93	1.12	2.04	1.70
Kjeldahl-N	2.43	2.61	1.05	2.03
Ammonia-N	0.13	0.38	0.07	0.19
COD	41.54	42.83	40.00	41.46
Total solids	558	1259	1236	1018
Dissolved solids	536	1258	1212	1002
Total phosphorus	3.35	3.23	1.87	2.82
Dissolved orthophosphorus	3.19	2.62	1.10	2.30
Dissolved orthophosphorus/total dissolved phosphorus	0.95	0.81	0.59	0.78

Table 34. Lake Villa WTP Effluent Characteristics

(Concentrations in mg/l)

Parameters	*Dates of observations* 11/21/74	4/8/75	7/8/75	11/5/75	*Mean*
pH	8.21	8.33	8.35	7.91	
Alkalinity	284	284	297	243	277
Nitrate-N	0.71	0.17	0.26	15.62	4.19
Kjeldahl-N	8.62	14.05	5.67	1.20	7.39
Ammonia-N	3.82	11.88	2.51	0.52	4.68
COD	43.01	72.93	35.81	24.12	43.97
Total solids	761	628	560	714	666
Dissolved solids	717	545	532	674	617
Total phosphorus	5.19	4.32	3.61	14.94	7.02
Dissolved orthophosphorus	4.44	2.98	3.02	7.08	4.38
Dissolved orthophosphorus/total dissolved phosphorus	0.86	0.69	0.84	0.47	0.72

Lake Villa WTP

Until the early part of 1975, the wastewater was treated in an Imhoff tank and two oxidation lagoons, one of which was aerated with diffused air. An entirely new waste treatment facility consisting of a packaged activated sludge treatment process with a design capacity of 0.3 mgd is now in operation. The effluent is chlorinated and discharged to Eagle Creek which empties into Long Lake. There is no provision for phosphorus removal in the new treatment facility.

The Northeastern Illinois Planning Commission (NIPC, 1974) has proposed a regional wastewater plan which recommends that the new plant be operated only until its bonded indebtedness has been satisfied. Anticipated excess flow could be diverted to the regional interceptor and ultimately the area could be served by a proposed regional plant. (Details of the proposed regional plan are discussed on page 114.) Wastewater effluent characteristics are shown in table 34.

Round Lake Sanitary District WTP

The present facility is a 1.6 mgd trickling filter plant with three lagoons. The first lagoon is aerated and the final effluent is chlorinated. The Sanitary District serves portions of the municipalities of Round Lake, Round Lake Beach, Round Lake Heights, and Round Lake Park.

In its 'interim expansion and upgrading' plans, the Sanitary District has proposed changes to the clariflocculator, anaerobic digesters, and aeration tanks; installation of tertiary filters; and modifications to the trickling filters, lagoons, and other facilities. The Sanitary District has also proposed the construction of four new interceptors in the communities it serves.

NIPC proposed that the area be integrated into the Lake County regional wastewater treatment system. The regional plan recommends the construction of the four new interceptors up to the Round Lake treatment plant. When full treatment at the plant is terminated, the plant would be kept on a standby basis to temporarily store and/or treat excess wet weather flows. The citizens of the Sanitary District area voted in 1973 against joining the County regional system by a 3 to 1 margin.

In the existing treatment facilities, phosphorus removal is not practiced. Table 35 shows the wastewater effluent chemical quality characteristics.

Antioch WTP

Until mid-1975, the wastewater at this plant was treated in an 0.5 mgd activated sludge plant with three tertiary ponds. The final effluent was chlorinated and discharged to Sequoit Creek. The treatment facilities have since been expanded and upgraded to a design capacity of 1.0 mgd. The wasteflow is given a tertiary treatment in mixed media filters for solids removal. The polishing ponds are bypassed prior to chlorination. Phosphorus removal facilities have been incorporated now, but were not commissioned at the time of sampling.

NIPC (1974) proposes that the plant be operated until its economic life expires. Following that, the plan calls for it to be used to either store and release excess flows for treatment at the regional plant or to treat the excess flows on site and discharge the effluent.

Effluent quality characteristics for this treatment plant are shown in table 36.

Table 35. Round Lake WTP Effluent Characteristics

(Concentrations in mg/l)

Parameters	*Dates of observations* 11/21/74	4/8/75	7/8/75	11/6/75	*Mean*
pH	7.85	8.22	7.98	8.28	
Alkalinity	306	293	311	278	297
Nitrate-N	0.68	0.52	0.25	0.80	0.56
Kjeldahl-N	27.17	34.12	11.30	25.30	24.47
Ammonia-N	16.62	23.45	5.23	15.30	15.15
COD	27.67	42.74	35.81	60.88	41.78
Total solids	683	629	600	696	652
Dissolved solids	668	604	578	658	627
Total phosphorus	9.02	4.63	3.89	2.83	5.09
Dissolved orthophosphorus	7.82	3.27	3.61	2.03	4.18
Dissolved orthophosphorus/total dissolved phosphorus	0.87	0.71	0.93	0.72	0.81

Table 36. Antioch WTP Effluent Characteristics

(Concentrations in mg/l)

Parameters	*Dates of observations* 11/21/74	4/8/75	7/8/75	11/5/75	*Mean*
pH	8.10	7.90	8.12	8.04	
Alkalinity	235	284	315	353	297
Nitrate-N	12.76	2.36	0.40		5.17
Kjeldahl-N	2.14	6.53	12.3	24.0	11.24
Ammonia-N	2.09	5.06	5.44	10.75	5.84
COD	18.94	29.85	51.29	46.76	36.71
Total solids	802	653	640	670	691
Dissolved solids	744	624	592	652	653
Total phosphorus	8.62	3.37	3.78	7.86	5.91
Dissolved orthophosphorus	7.16	3.26	3.44	3.62	4.37
Dissolved orthophosphorus/total dissolved phosphorus	0.83	0.97	0.91	0.46	0.79

Table 37. Fox Lake WTP Effluent Characteristics

(Concentrations in mg/l)

Parameters	*Dates of observations* 11/21/74	4/8/75	7/8/75	11/5/75	*Mean*
pH	8.15	8.38	8.08	8.21	
Alkalinity	413	319	395	443	393
Nitrate-N	1.43	0.47	3.00	1.51	1.60
Kjeldahl-N	21.93	19.14	15.8	24.0	20.22
Ammonia-N	18.62	19.12	6.77	11.79	14.08
COD	30.68	107.87	70.64	68.53	69.43
Total solids	1252	1035	1124	1244	1164
Dissolved solids	1236	953	1066	1202	1114
Total phosphorus	7.39	5.79	4.25	11.01	7.11
Dissolved orthophosphorus	6.95	3.29	3.72	5.35	4.83
Dissolved orthophosphorus/total dissolved phosphorus	0.94	0.57	0.88	0.49	0.72

Fox Lake WTP

The existing treatment designed for an average flow of 0.72 mgd, presently treats an annual average flow of 0.25 mgd. The waste flow is treated by trickling filters, chlorinated, and discharged. Lake County Department of Public Works and the village of Fox Lake have submitted a joint application for implementing the regional plan in this service area. Included is the construction of a new 6 mgd plant which is discussed below. The effluent quality characteristics for the Fox Lake WTP are shown in table 37.

Regional Plan – Lake County Northwest Service Area

The Northeastern Illinois Planning Commission believes the most critical water pollution problem in northeastern Illinois lies in the northwestern part of Lake County (NIPC, 1974). Here, in the Commission's judgment, exists deteriorating water quality in the Chain of Lakes, health hazards from malfunctioning septic tanks, and inadequate wastewater treatment plants.

The regional plan proposed by NIPC (1974) for the northwestern sector of Lake County calls for the construction of a single treatment plant located in the village of Fox Lake, together with a regional interceptor sewer system which will provide service to the major part of the sector. The proposed regional system will permit consolidation of Fox Lake, Round Lake Sanitary District, Antioch, and Lake Villa municipal treatment facilities. Sewer service will be available to existing developments now on septic tanks and to industrial and miscellaneous plants in the sector when and where it is found economical to make the necessary connections.

The plan envisages total waste flows of 6.0 mgd by 1980, 12.0 mgd by 1990, and 19.0 mgd by 2010. The initial construction cost would be $12.9 million. This includes construction of a 6.0 mgd waste treatment plant at Fox Lake together with the first phase of the interceptor system and related pumping stations. The plant design criteria include secondary treatment, phosphorus removal, nitrification-denitrification, and a polishing lagoon.

The existing Fox Lake plant is planned to be phased out of service with the implementation of the regional plan. The Round Lake, Antioch, and Lake Villa plants will continue in service until their economic lives expire. At that time they will be placed on standby service to handle excess wet weather flows. It is contemplated that waste treatment service could be provided to areas in the Nippersink Creek drainage basin in McHenry County at a later date upon completion of other interceptor sewers not included in the plan now.

Pollution and Abatement Plans for Fox River Watershed, Wisconsin

Because 868 square miles of the Fox River watershed (measured at Wilmot near the Wisconsin-Illinois border) lies within Wisconsin, the water quality of the Fox River as it enters the Fox Chain of Lakes has a significant impact on the water quality of the Chain of Lakes. Therefore, the magnitude of waste loads, relative importance of municipal, agricultural, and urban sources of pollutants, and the water quality management needs assessed by the Southeastern Wisconsin Regional Planning Commission (1969) are summarized here.

Twelve major municipal waste discharges exist within the Fox River watershed in Wisconsin. Information about population served, the type and efficiency of treatment used, approximate daily discharge and other pertinent information is given in table 38. Four of the

Table 38. Major Municipal Waste Discharges in Fox River Watershed within Wisconsin, 1966*

Municipality served	*Population served*	*Average daily discharge (mgd)*	*Type of treatment ** *	*Efficiency of BOD removal (%)*	*5-day BOD discharged (lbs/day)*
Sussex	1,400	0.19	TF	80	48
Brookfield	2,200	0.32	AS	78	82
Pewaukee	2,900	0.32	TF	80	99
Waukesha	37,500	7.50	TF	85	1,770
Mukwonago	1,900	0.19	TF	80	65
Waterford	1,600	0.19	P	25	204
East Troy	1,500	0.19	TF	80	51
Lake Geneva	4,500	0.52	TF	80	153
Burlington	6,200	1.10	TF	80	210
Twin Lakes	3,100	0.18	TF	80	105
Genoa City	1,050	0.10	TF	80	36
Silver Lake	200	0.02	EA	80	7

*Southeastern Wisconsin Regional Planning Commission
**TF = trickling filter; AS = activated sludge; P = primary; and EA = extended aeration

municipal plant discharges, including the Waukesha waste treatment plant with the maximum discharge, are located in the headwaters of the Fox River. Four other waste discharges, including the Burlington waste treatment plant, are distributed along the rest of the main stem of the Fox River. The other four, including that of Lake Geneva, are located on tributaries to the Fox River.

All of the municipal waste treatment plants in the watershed, except the one at Waterford, provide secondary treatment. Almost 75 percent of the total population load as measured by S-day BOD enters the river at and above the city of Waukesha. Approximately 75 percent of the low flow of the Fox River below Waukesha consists of effluent from the waste treatment plants at Waukesha, Brookfield, Pewaukee, and Sussex.

Nineteen major industrial waste discharges exist within the Fox River watershed. Eight are located in the vicinity of the city of Waukesha. Manufacturing, milk processing, canning, and cooling are the major uses for these wastewaters. No information is available on the quantity or quality of the waste effluents discharged.

While discussing the types of pollutants (silt, nutrients, pesticides, organic wastes, etc.) arising from agricultural practices, and their possible impact on receiving streams and lakes, the Southeastern Wisconsin Regional Planning Commission (1969) concludes that agricultural drainage and runoff generally constitute a relatively minor source of stream pollution in the basin in comparison with the pollution caused by municipal discharges. This judgment is based on the stream quality data for the Fox River and its watershed. Likewise the Commission has concluded that the effects of urban runoff and combined sewer overflows on stream water quality in the Fox River watershed is insignificant compared with the waste sources from the municipal sewage treatment plants.

The Commission's recommended plan for pollution abatement in the Fox River watershed proposes the following measures:

1) Provide advanced waste treatment for biochemical oxygen demand and nutrient removal and disinfection at all major waste discharge locations within the watershed. This would include:
   - A single large sewage treatment plant providing advanced waste treatment for the four municipalities within the entire upper watershed, along with a system of trunk sewers to convey the wastes from the upper watershed to this plant

- Advanced waste treatment facilities at the six sewage treatment plants in the lower reaches of the watershed, i.e., Mukwonago, Waterford-Rochester, East Troy, Lake Geneva, Burlington, and Twin Lakes
- Silver Lake and Genoa City would continue to be served by secondary treatment plants with post chlorination disinfection

2) Institute improved soil and water conservation practices on farmlands in the agricultural areas of the basin in order to minimize the effects of runoff from agricultural areas containing silt, fertilizers, herbicides, and pesticides.
3) Connect to the public sanitary sewerage systems 16 of the 19 major industrial waste sources. In addition the plan recommends that all other industrial and resort waste discharges not connected to centralized public sanitary systems be given a level of treatment equivalent to secondary treatment and disinfection.

The Commission reported that implementation of the recommended stream and lake water quality management plan would abate all of the existing major sources of stream pollution within the watershed. It is anticipated that the municipal waste loadings on the stream system would be reduced from 2800 pounds per day of BOD and 390 pounds per day of phosphorus to 900 pounds and 30 pounds, a reduction of 68 and 92 percent, respectively.

LIMNOLOGY OF THE LAKES

In order to assess the physical, chemical, and biological characteristics of the lake system, field trips were undertaken to make *in situ* observations of temperature, dissolved oxygen, and secchi disc readings and to obtain water samples for chemical and biological evaluations. Biweekly trips were made during the period May 19, 1975, to July 14, 1975, and thereafter weekly trips were made until October 20, 1975.

Fourteen lake stations were established for water sampling and *in situ* observations (figure 72). These sampling sites were selected at the deepest part of each of the lakes. In waters with depths greater than 30 feet, as in the case of Lakes Catherine, Channel, Marie (west basin), and Pistakee Bay, water samples were obtained at three different depths — at the surface, mid-depth, and 1 foot from bottom, here designated as the deep sample. In the lakes with maximum depths in the range of 15 to 30 feet, water samples were obtained from surface and deep locations. In all other cases, only water samples from the surface were collected.

For measuring the secchi disc transparencies, an 8-inch diameter secchi disc with black and white quadrant markings attached to a calibrated line was used. The disc was lowered until it disappeared from view and the depth of immersion of the disc was noted. The disc was lowered farther and then raised slowly until it reappeared. The depth of immersion was noted. The average of these two observations was recorded as the secchi disc reading.

In situ dissolved oxygen and temperature measurements were made with a galvanic cell oxygen analyzer equipped with a thermister. An oxygen meter, Yellow Spring Instrument Company model 54, with a 50-foot probe lead was used for this purpose. At the beginning of each day's field survey, the probe was standardized in lake surface water in which the dissolved oxygen content was determined by a modified Winkler Method as outlined by the American Public Health Association (1971). Temperature and dissolved oxygen

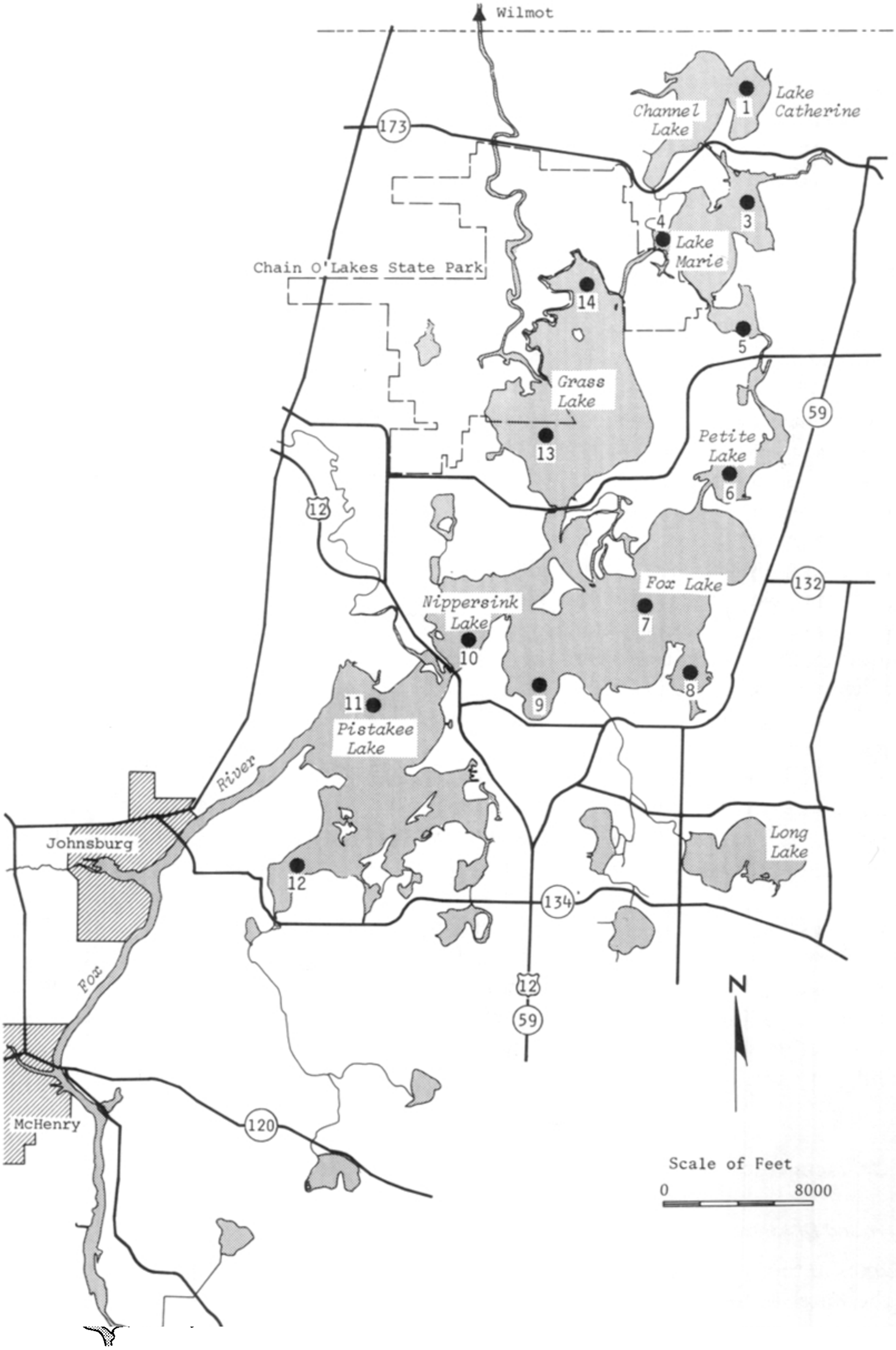

Figure 72. Locations of *in situ* sediment oxygen demand measurements

measurements were obtained at 2-foot intervals commencing from the surface of the lake. Observations were made at l-foot intervals in the zone of the thermocline of deep lakes and in the lakes less than 15 feet deep.

Water samples for chemical analyses were collected at required depths in plastic bottles, stored on ice until transported to the laboratory, and refrigerated until chemical analyses were performed. For ammonia determinations, 50 milliliters (ml) of water samples were filtered through type HA, 0.45 μ millipore filters 37 millimeters (mm) in diameter. The filters were placed over filter pads which were held between two-piece circular holders. A set of eight such holders were held in a wooden frame designed and fabricated at the Survey laboratory. Positive pressure for filtering the samples was provided by a syringe to force the sample through the filters. The filtrates were collected in small plastic bottles. Micropore filtration eliminates any bacterial activity which could alter the ammonia concentration in the collected samples. Laboratory tests extending over a period of 4 weeks indicated that the ammonia concentrations remained stable in the filtered samples. This method of sample preservation is considered superior to acidification or other chemical additives.

Water samples in a volume of 380 ml were collected for algal identification and enumeration. These samples were preserved with 20 ml of formalin at the time of collection and stored at room temperature until examined.

Water samples, for chemical and algal analyses representative of the desired depths of the water column, were obtained with a Kemmerer water sampler.

In addition to collecting water samples and making *in situ* measurements of temperatures and dissolved oxygen concentrations in the Fox Chain of Lakes, measurements and sample collections were made in Cedar Lake. This lake is located about 4 miles east of the Fox Chain. It has been classified as a mesotrophic lake in the National Eutrophication Survey (US Environmental Protection Agency, 1975a). A comparison of the limnological characteristics of Cedar Lake with those of Fox Chain of Lakes was thought desirable. Cedar Lake was visited on a biweekly basis from July 14 to October 20, 1975. The depth at the sampling site was 42 feet. *In situ* measurements of temperature, dissolved oxygen, and secchi disc readings were made, and water samples were collected from surface, mid-depth, and deep water strata.

The following physical and chemical determinations were made on water samples in the laboratory: turbidity, pH, alkalinity, hardness, nitrate-nitrogen, Kjeldahl nitrogen, ammonia-nitrogen, total silica, total iron, chloride, sulfate, total solids, dissolved solids, suspended solids, algal growth potential (AGP), total phosphorus, and dissolved orthophosphorus. The methods and procedures involved in these determinations were discussed at the beginning of Part 3 of this report.

Physical Characteristics

Temperature and Dissolved Oxygen

Lakes in the temperate zone generally undergo seasonal variations in temperature through the water column (Kothandaraman and Evans, 1970). These variations, with their accompanying phenomena, are perhaps the most influential controlling factors within the lakes.

The temperature of a deep lake in the temperate zone is about 4°C during early spring. As the air temperatures rise, the upper layers of water warm up and mix with the lower layers by wind action. By late spring, the differences in thermal resistance cause the mixing to cease and the lake approaches the thermal stratification of the summer season. Following closely the temperature variation in water is the physical phenomenon of increasing density with decreasing temperature up to a certain point. These two interrelated forces are capable of creating within the lake strata of water of vastly differing characteristics.

During thermal stratification the upper layer (the epilimnion) is isolated from the lower layer of water (the hypolimnion) by a temperature gradient (the thermocline). Temperatures in the epilimnion and hypolimnion are essentially uniform. The thermocline will typically have a sharp temperature drop per unit depth from the upper to the lower margin. When the thermal stratification is established, the lake enters the summer stagnation period, so named because the hypolimnion becomes stagnated.

With cooler air temperatures during the fall season, the temperature of the epilimnion decreases. This decrease in temperature continues until the epilimnion is the same temperature as the upper margin of the thermocline. Successive cooling through the thermocline to the hypolimnion results in a uniform temperature through the water column. The lake then enters the fall circulation period, called 'fall turnover' and is again subjected to a complete mixing by the wind.

Declining air temperatures and the formation of an ice cover during the winter produce a slight inverse thermal stratification (Fair et al., 1963). The water column is essentially uniform in temperature at about 3 to 4°C but slightly colder temperatures of 0 to 2°C prevail just below the ice. With the advent of spring and gradually rising air temperatures, the ice begins to disappear, and the temperature of the surface water rises. The lake again becomes uniform in temperature and the spring circulation occurs.

The most important phase of the thermal regime from the standpoint of eutrophication is the summer stagnation period. The hypolimnion, by virtue of its stagnation, traps sediment materials such as decaying plant and animal matter thus decreasing the availability of nutrients during the critical growing season. In an eutrophic lake, the hypolimnion becomes anaerobic or devoid of oxygen because of the increased content of highly oxidizable material and because of its isolation from the atmosphere. In the absence of oxygen, the conditions for chemical reduction become favorable and more nutrients are released from the bottom sediments to the overlying waters.

However, during the fall circulation period, the lake water becomes mixed, and the nutrient rich hypolimnetic waters are redistributed. The nutrients which remained trapped during the stagnation period become available during the following growing season. Therefore, a continual supply of plant nutrients from the drainage basin is not mandatory for sustained plant production. Fruh (1967) and Fillos and Swanson (1975) stated that after an initial stimulus, the recycling of nutrients within a lake might be sufficient to sustain highly productive conditions for several years.

Figures 73 through 80 show the isothermal plots for the lake system with depths of 17 feet or more. At the time of the first observation for temperature profiles, on May 19, 1975, the deep lakes in the Fox Chain had already begun to stratify. In the case of Lake Bloomington, in central Illinois, thermal stratification was found to set in during mid-June (Kothandaraman and Evans, 1970).

The lake surface water temperatures in the Fox Chain increased gradually reaching a maximum of about 29°C on August 4, 1975, and decreased gradually from then on. In all the lakes, the depth of epilimnetic zone at the time of peak stratification was found to be confined to the upper 15 feet.

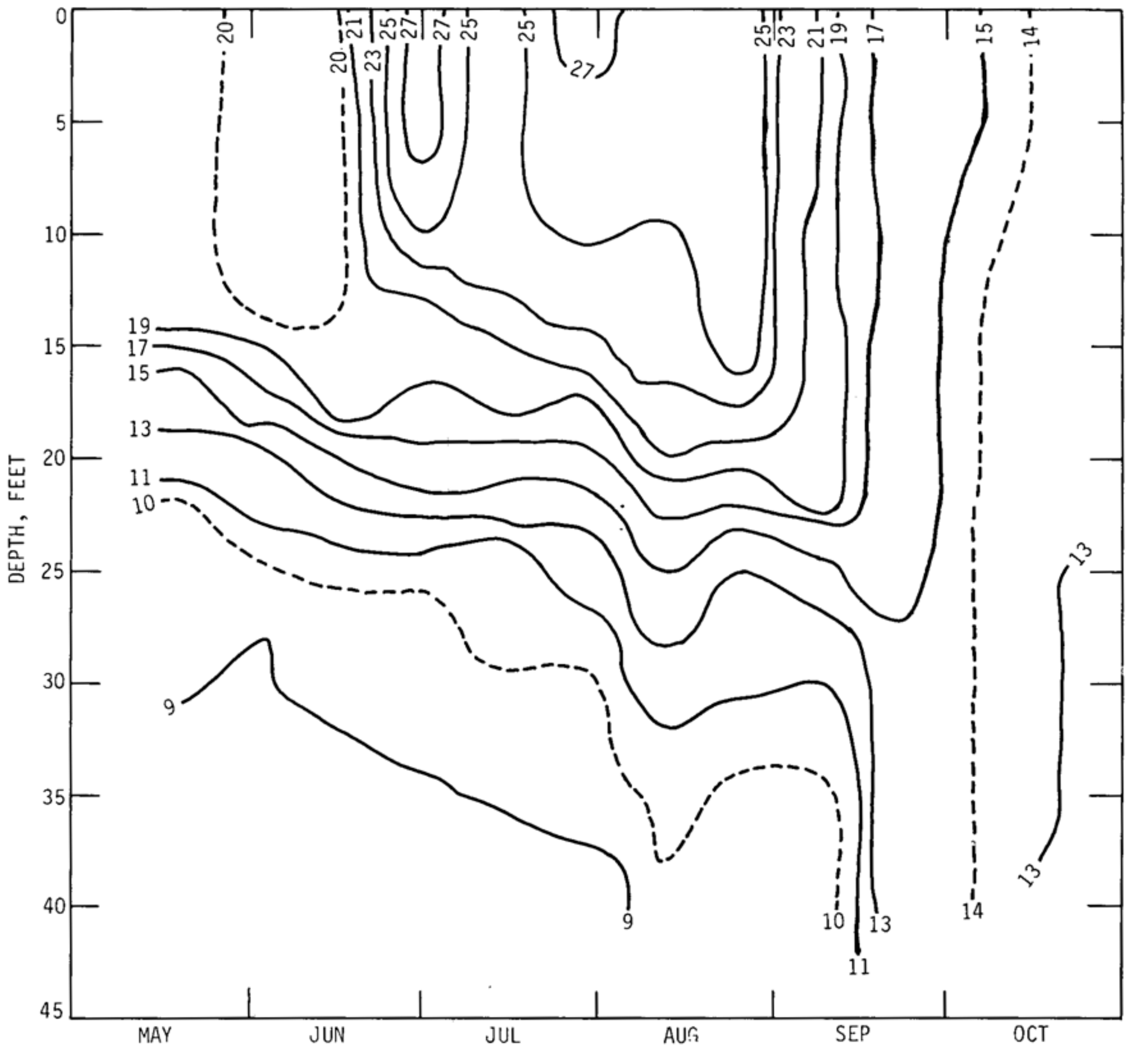

Figure 73. Isothermal plots for Lake Catherine

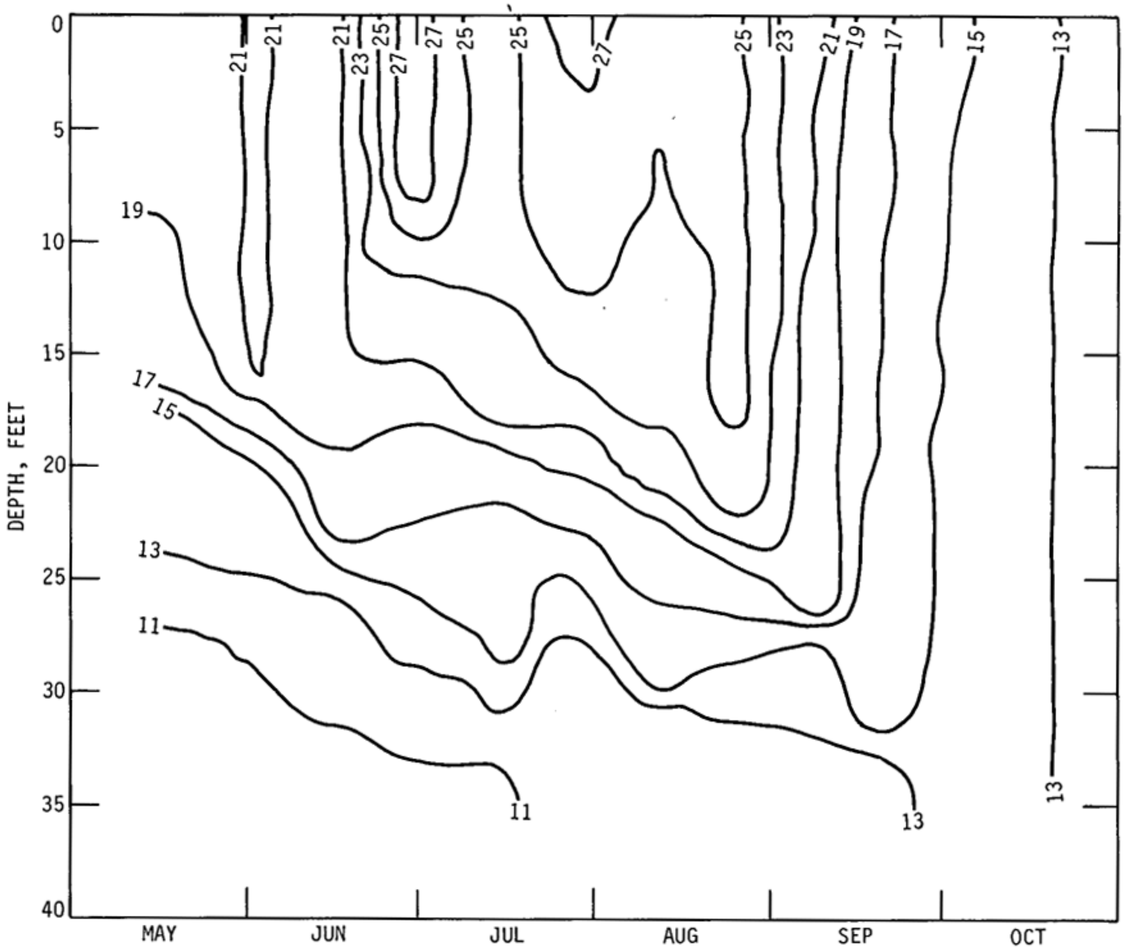

Figure 74. Isothermal plots for Channel Lake

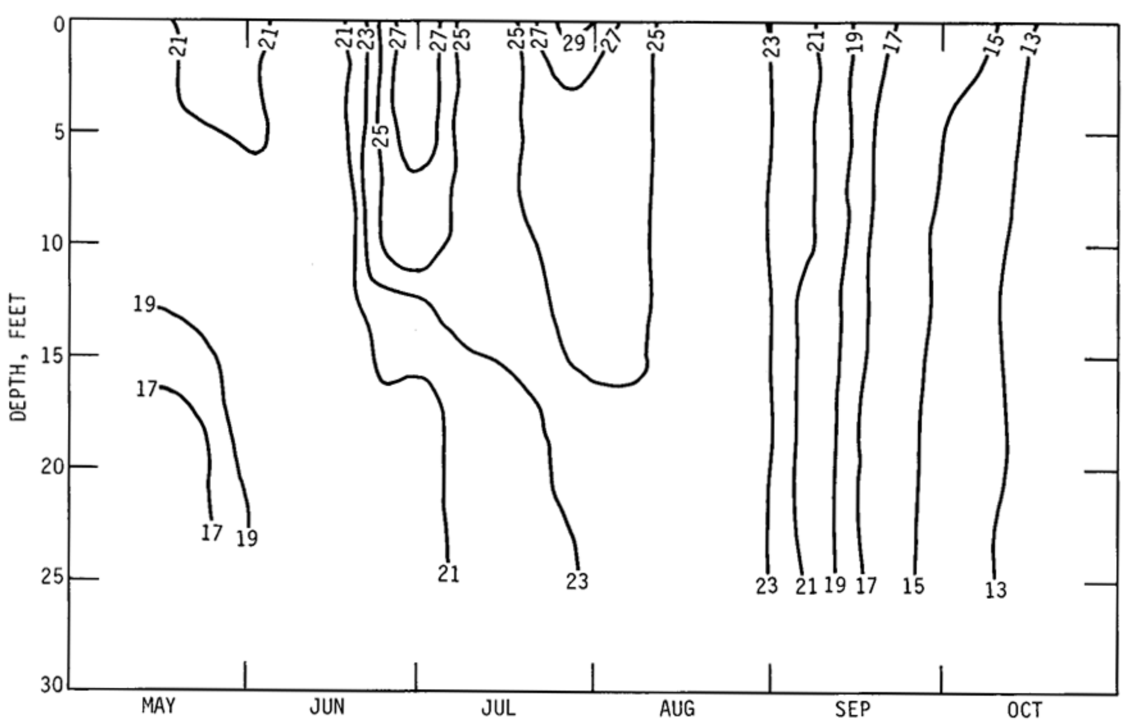

Figure 75. Isothermal plots for Lake Marie (east basin)

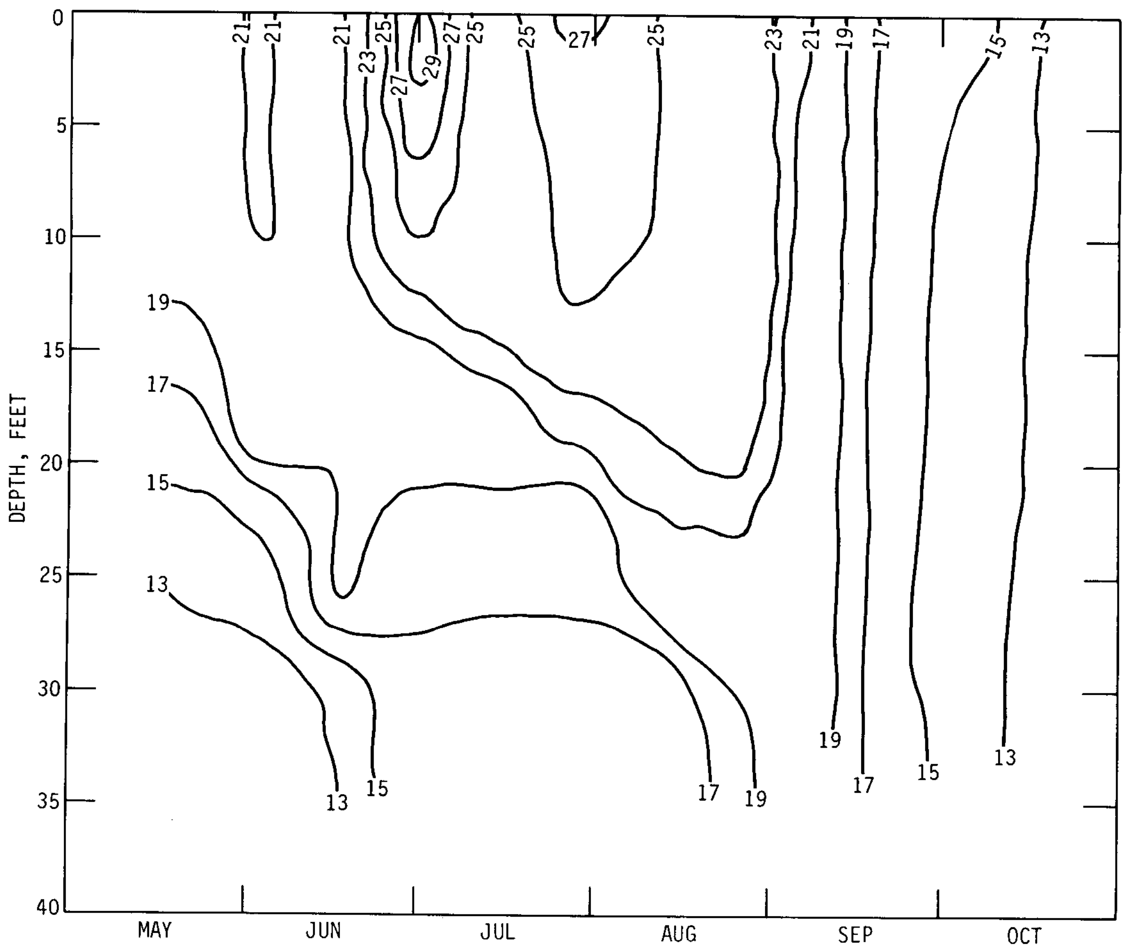

Figure 76. Isothermal plots for Lake Marie (west basin)

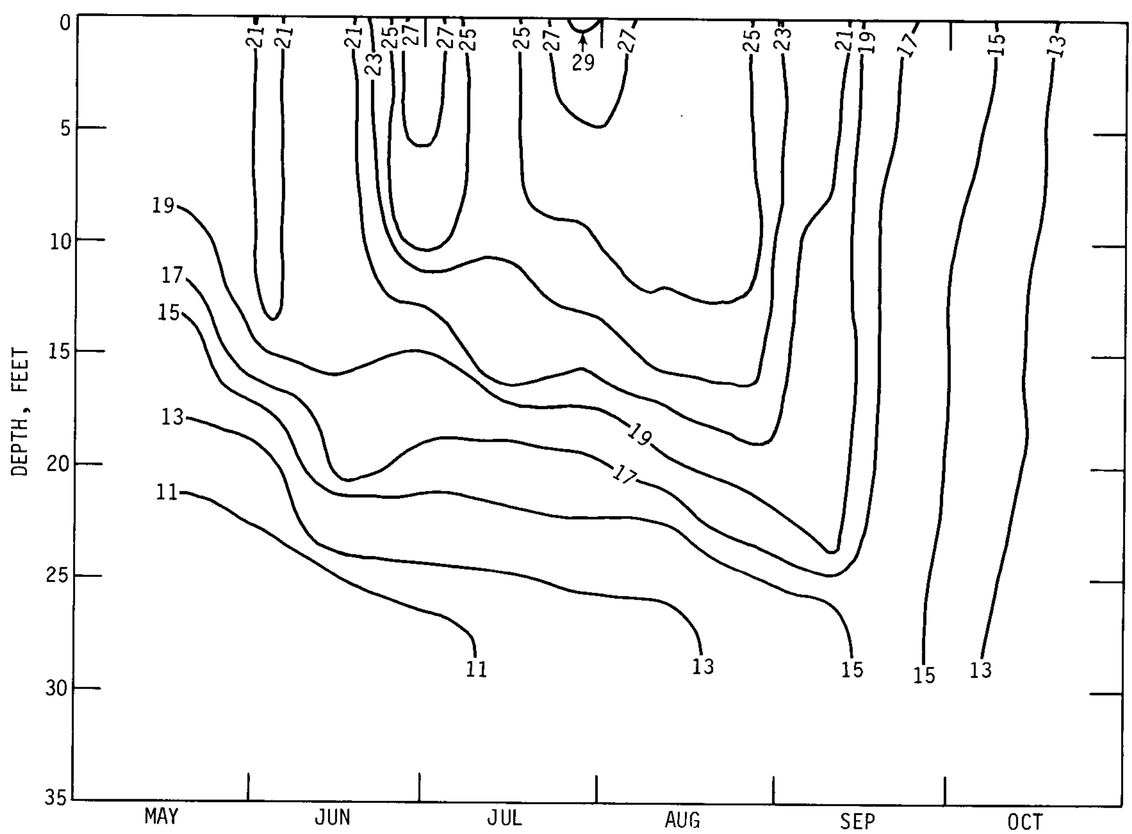

Figure 77. Isothermal plots for Bluff Lake

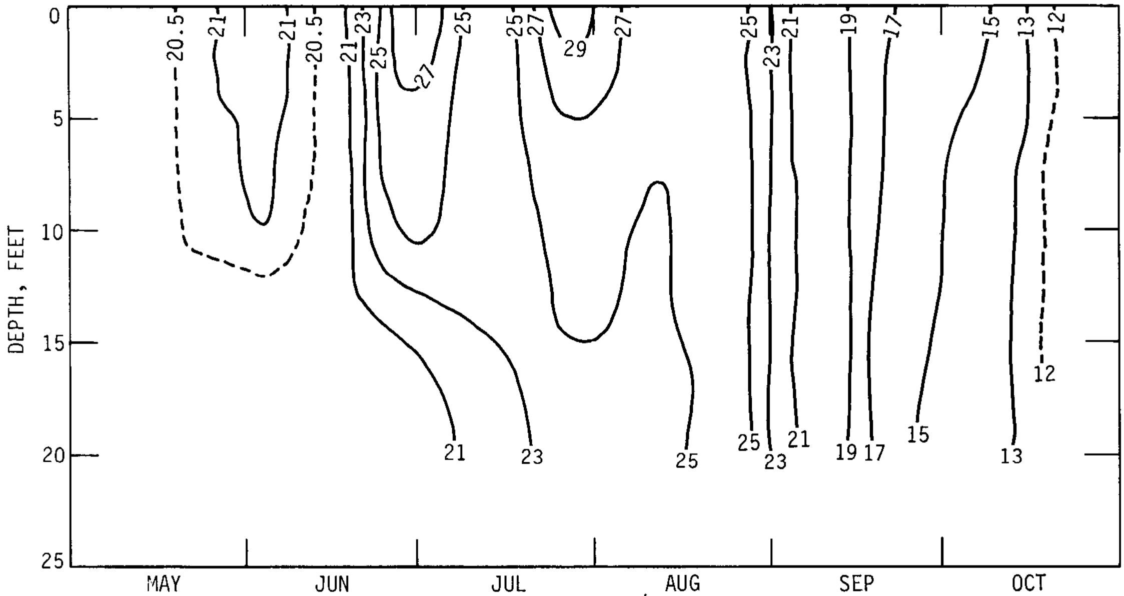

Figure 78. Isothermal plots for Petite Lake

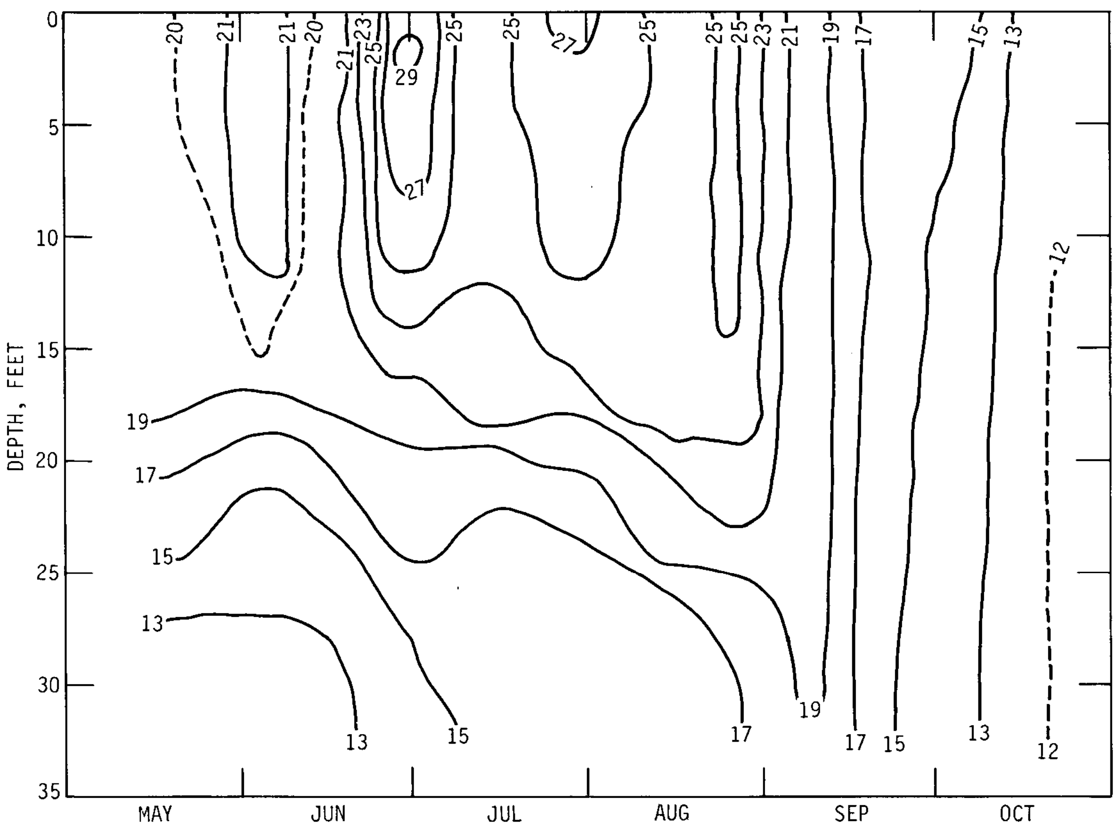

Figure 79. Isothermal plots for Pistakee Bay

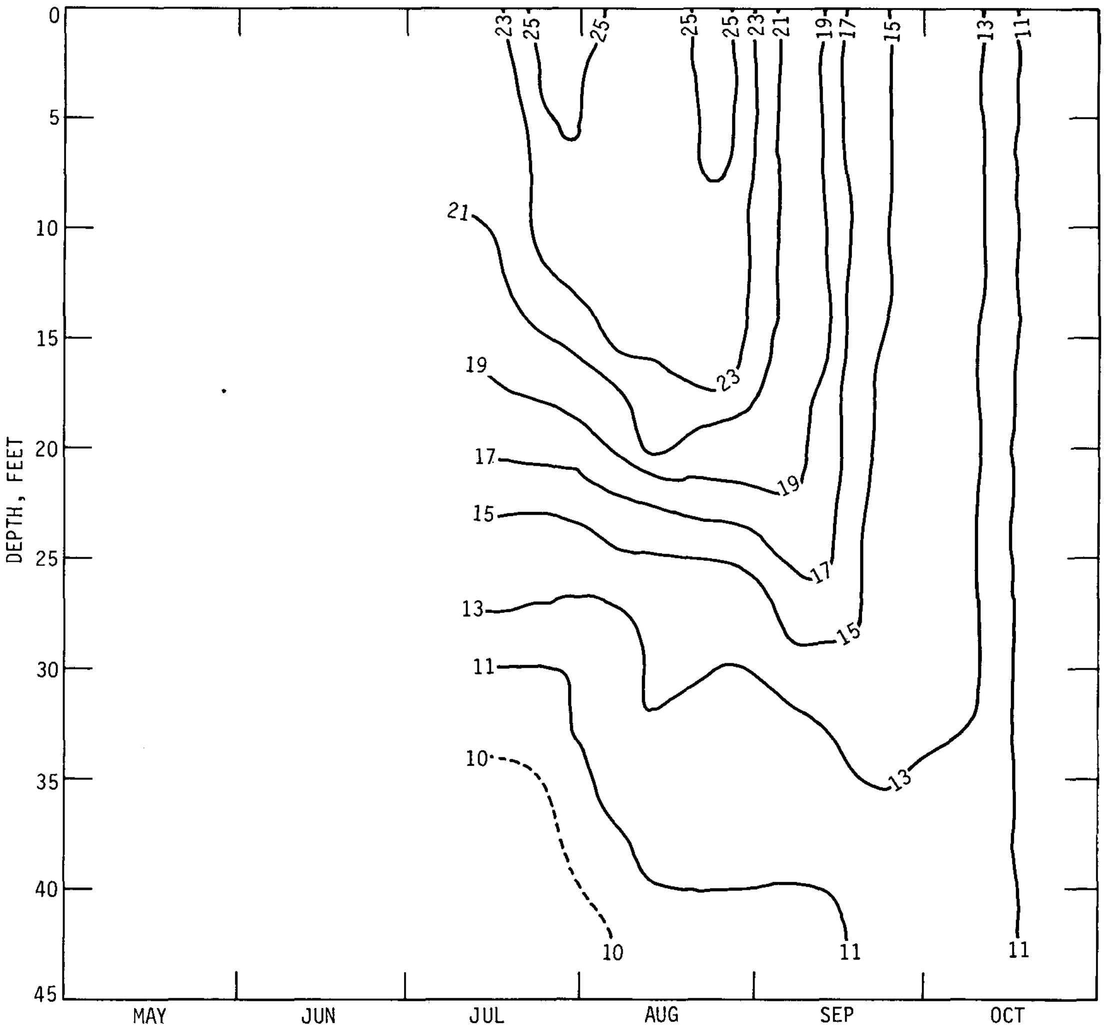

Figure 80. Isothermal plots for Cedar Lake

The deep water temperatures in the lakes varied depending on the depth of the lakes. For instance, in Lake Catherine and Cedar Lake (figures 73 and 80, respectively), which are the two deepest of all the lakes investigated, the water strata adjacent to the bottom ranged from 9 to 11°C during the period of summer stagnation. Deep strata water temperatures in shallower lakes were higher. Likewise, the deepest lakes destratified later than the shallow ones. Lake Catherine and Cedar Lake destratified in late September, whereas in Channel Lake, Lake Marie (west basin), and Bluff Lake (figures 73, 75, and 76, respectively), the fall circulation occurred in the early part of September. This group of lakes has a maximum depth of about 30 feet. The phenomenon of thermal stratification was not pronounced in Lake Marie (east basin) nor in Petite Lake where the maximum depths are only about 20 feet. In these two lakes only, a temperature gradient existed from top to bottom without any well-defined thermal stratification.

Figures 81 through 87 show the temperature and dissolved oxygen profiles in Fox Lake and its two bays and in Nippersink, Pistakee, and Grass Lakes. These lakes are relatively shallow with maximum observed depths of less than 15 feet. The temperature profiles are

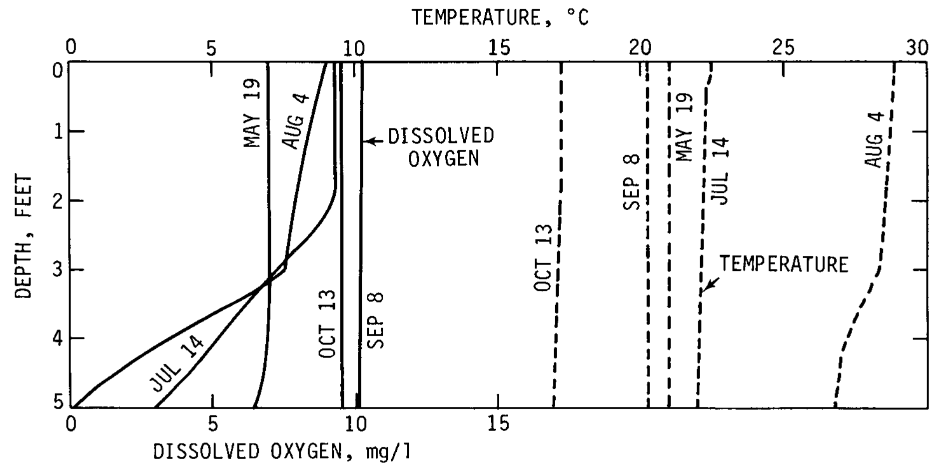

Figure 81. Temperature and dissolved oxygen profiles in Fox Lake (Stanton Bay)

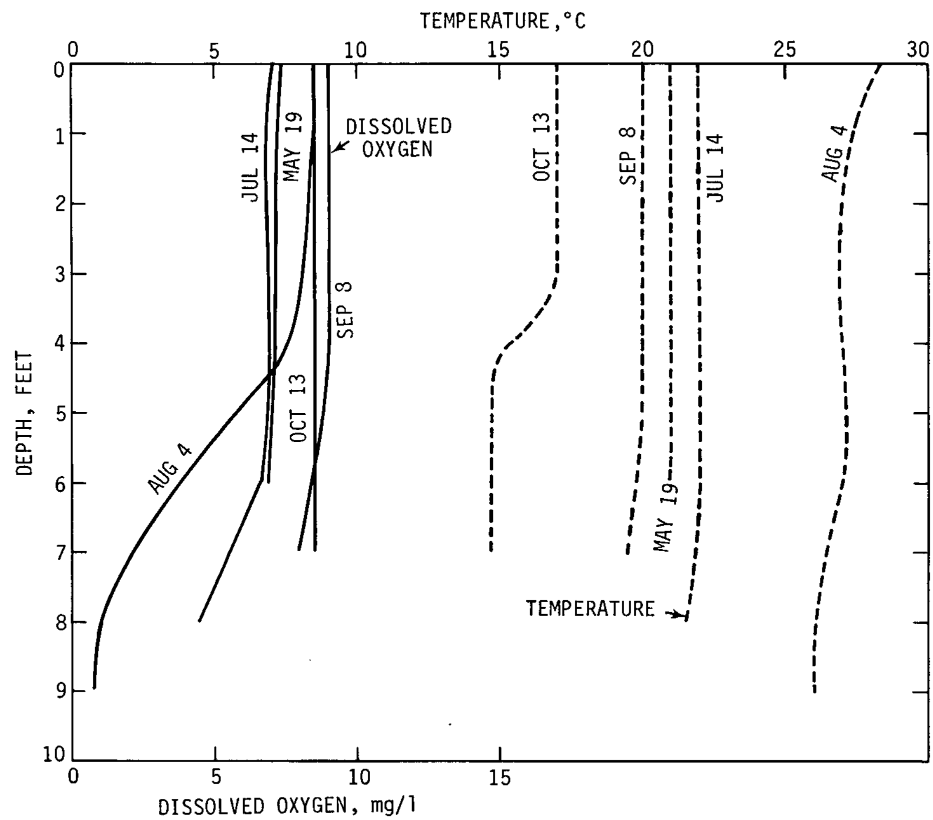

Figure 82. Temperature and dissolved oxygen profiles in Fox Lake (main body)

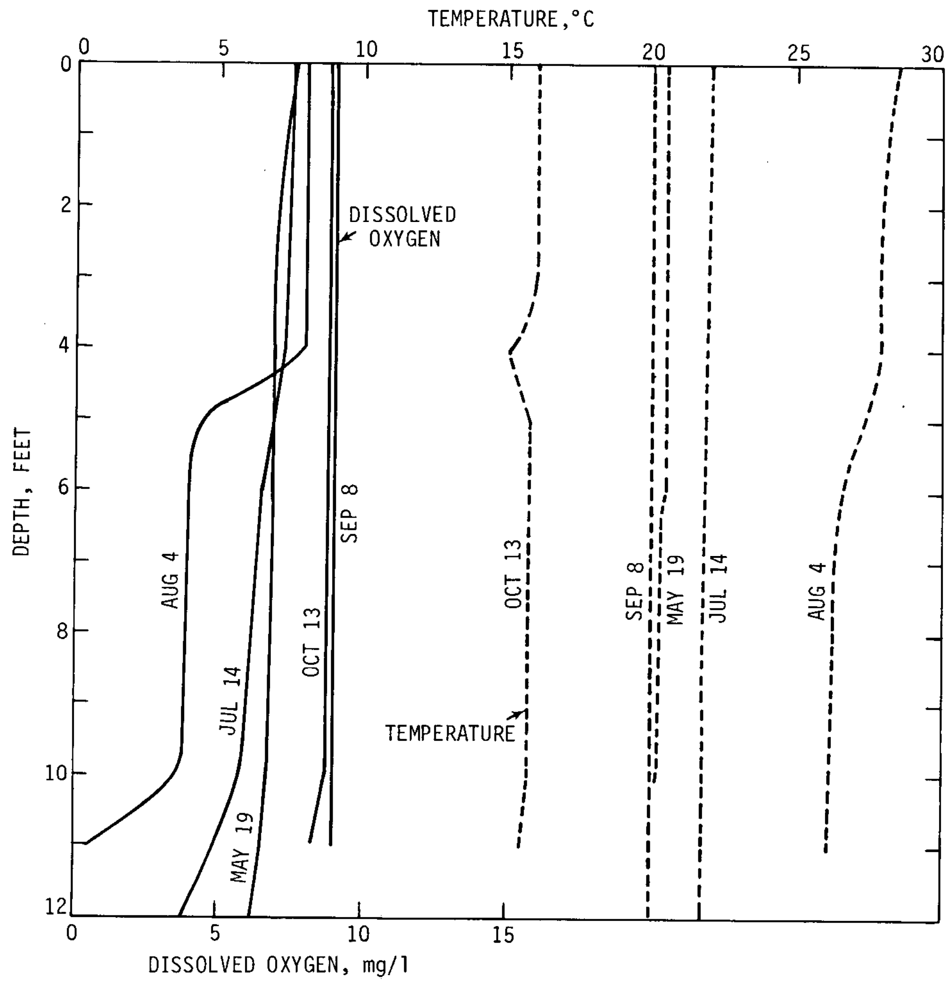

Figure 83. Temperature and dissolved oxygen profiles in Fox Lake (Mineola Bay)

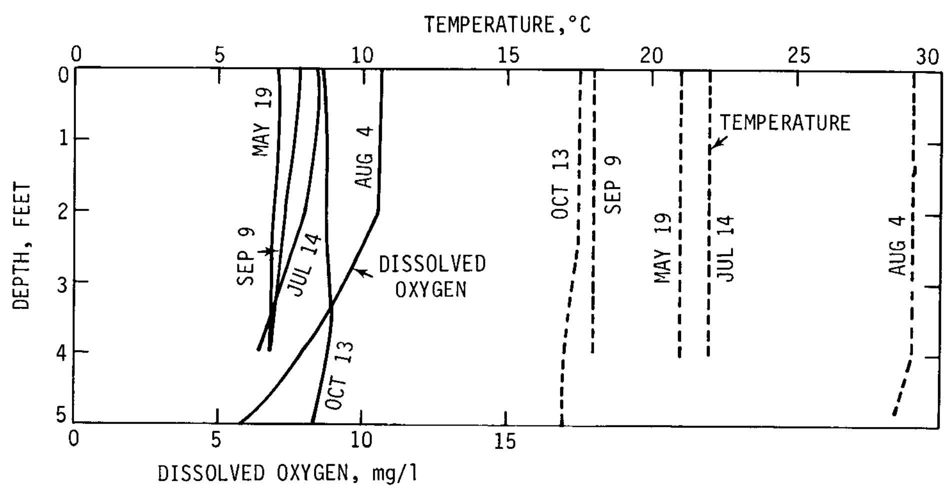

Figure 84. Temperature and dissolved oxygen profiles in Nippersink Lake

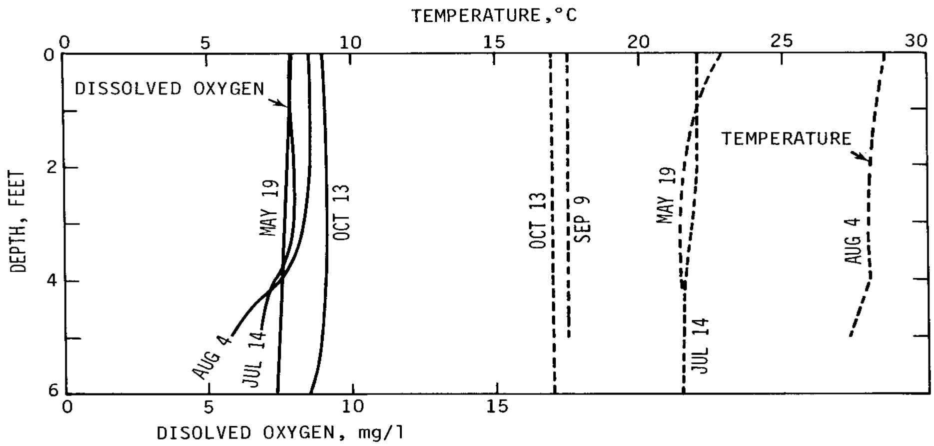

Figure 85. Temperature and dissolved oxygen profiles in Pistakee Lake

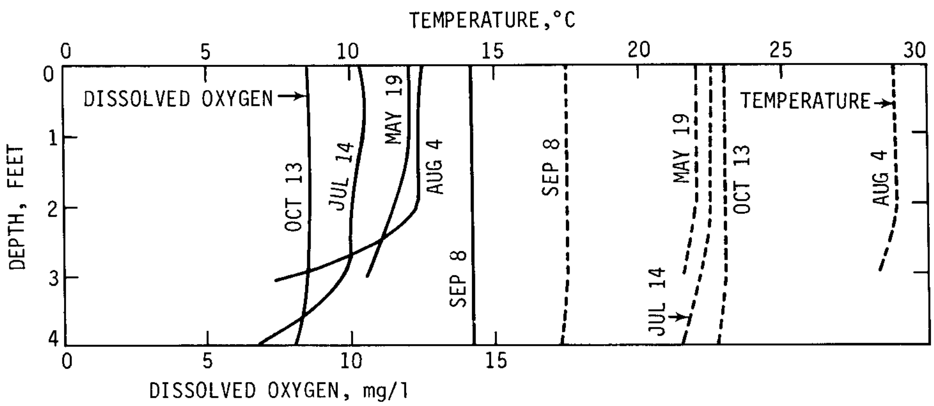

Figure 88. Temperature and dissolved oxygen profiles in Grass Lake (south)

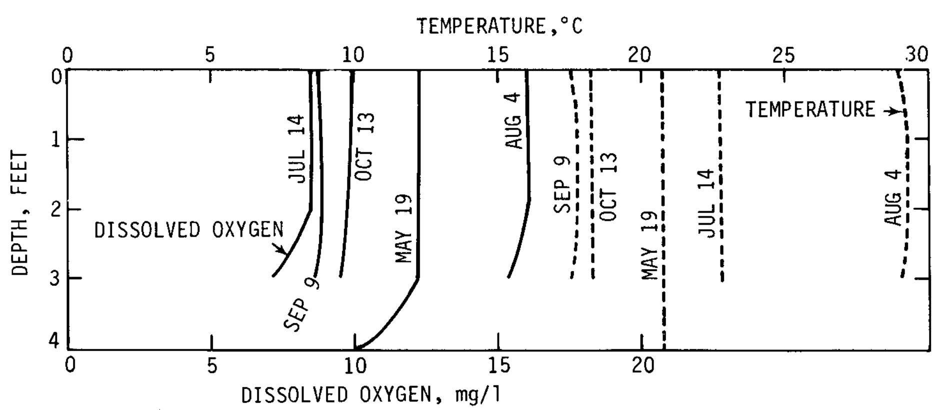

Figure 87. Temperature and dissolved oxygen profiles in Grass Lake (north)

plotted for five different dates. The surface water temperatures reached a maximum of about 29°C on August 4. Because the lakes are shallow, they are completely mixed, both by wind action and by the intense boating activities in these lakes. Thermal gradients in these lakes are very insignificant.

On the basis of the data collected in the Fox Chain of Lakes, it can be concluded that the upper deep lakes in the Chain stratify thermally and, at the period of peak stagnation, the epilimnetic zone is only 15 feet deep. In the lakes with maximum observed depths of about 20 feet, no well-defined thermal stratification existed. However, a perceptible degree of temperature gradient existed. In shallow lakes with depths less than 15 feet, the entire water mass was thermally uniform.

It is common knowledge that the impoundment of water, natural or man-made, alters its physical, chemical, and biological characteristics. The literature is replete with detailed reports on the effects of impoundments on various water quality parameters. The physical changes in the configuration of the water mass following impoundment reduce reaeration rates to a small fraction of those of free flowing streams. Where the depth of impoundment is considerable, the thermal stratification acts as an effective barrier for the wind-induced mixing of the hypolimnetic zone. The oxygen transfer to the deep waters is essentially confined to the molecular diffusion transport mechanism.

During the period of summer stagnation and increasing water temperatures, the bacterial decomposition of the bottom organic sediments exerts a high rate of oxygen demand on the overlying waters. When this rate of oxygen demand exceeds the oxygen replenishment by molecular diffusion, anaerobic conditions begin to prevail in the zones adjacent to the lake bottom. Hypolimnetic zones of man-made impoundments were also found to be anaerobic within a year of their formation (Kothandaraman and Evans, 1975).

The isopleth plots of dissolved oxygen for Lake Catherine are shown in figure 88. At the beginning of the field investigation, the lake waters were anoxic from depths of approximately 30 to 40 feet. As the summer stagnation intensified, the anoxic zone of hypolimnetic waters increased progressively reaching a maximum by the end of July. The extent of this anaerobic zone started diminishing thereafter and the DO concentration became uniform in the entire water column by mid-October. It must be pointed out that the progression of this anoxic zone coincided with the progression of the thermal stratification in the lake. At the time of peak thermal stratification, about 980 acre-feet or 40 percent of the lake volume was devoid of oxygen. At that time, only the epilimnetic zone, about 15 feet deep, had acceptable levels of dissolved oxygen.

Other deep lakes in the Fox Chain of Lakes system exhibited similar trends. The dissolved oxygen isopleths for these deep lakes are shown in figures 89 through 95. Even though no well-developed thermal stratification existed in Petite Lake, isopleth plots of dissolved oxygen for this lake shown in figure 93 indicate that there were periods when the water layers adjacent to the lake bottom were totally devoid of oxygen. Such episodes must have occurred during the absence of wind-induced mixing. Also in Cedar Lake (figure 95), which is classified as a mesotrophic lake, hypolimnetic waters from depths below 25 feet were anaerobic. All the deep water samples collected from lakes with depths of 30 feet or more, with the exception of Cedar Lake, had a strong septic odor predominantly that of hydrogen sulfide. The maximum extent of the anoxic zones in these lakes is shown in table 39. It is apparent that significant portions of the deep lakes are anoxic, particularly in Lake Catherine, Channel Lake, and Pistakee Bay.

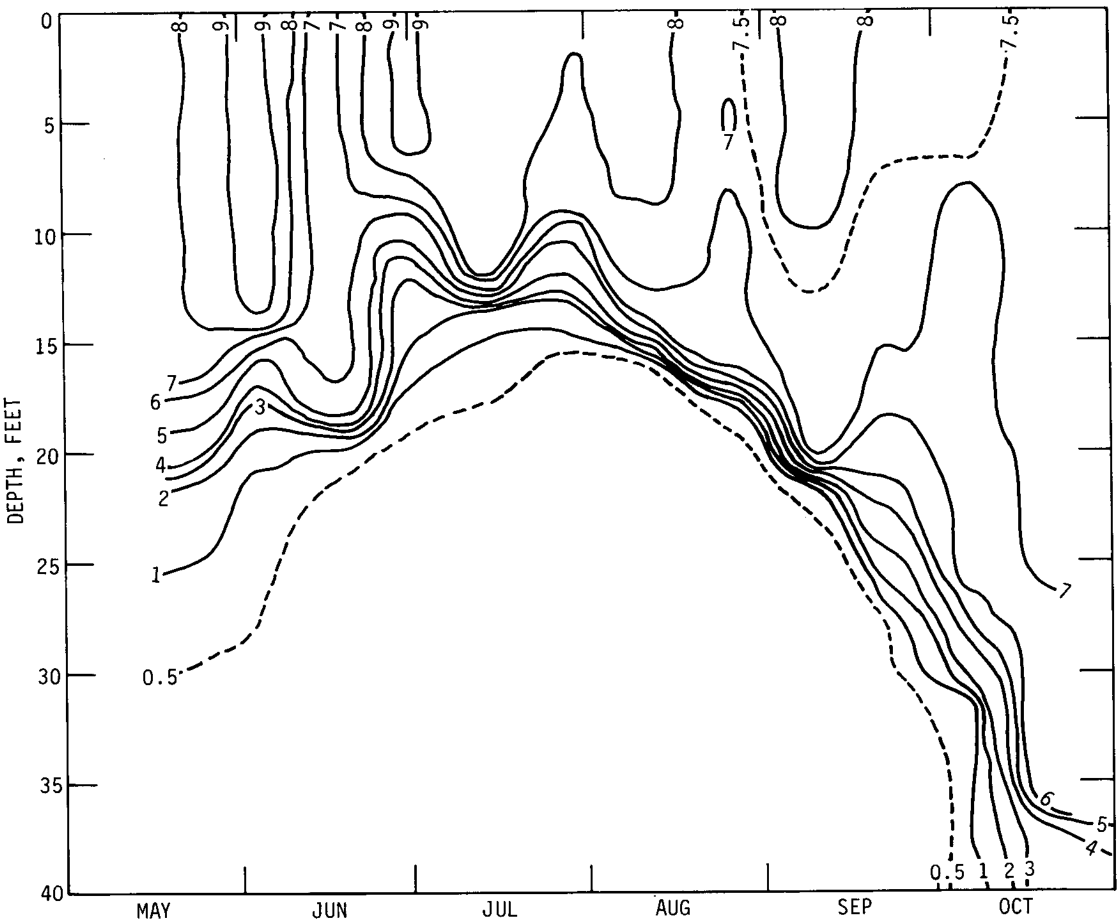

Figure 88. Isopleths of dissolved oxygen in Lake Catherine

Table 39. Extent of Anoxic Zones in Fox Chain of Lakes

Lake	*Depth to anoxic zone from surface (ft)*	*Volume of anoxic zone (acre-feet)*	*Percent of total volume*
Catherine	15	981	39.9
Channel	15	1368	31.3
Lake Marie (east basin)	15	823	17.4
Lake Marie (west basin)	15		
Bluff	15	165	17.1
Pistakee Bay	10	1025	41.6
Cedar	25		

Dissolved oxygen profiles for shallow lakes are shown in figures 81 to 87. Even though the lakes are thermally uniform, or nearly so, during the peak summer periods the oxygen demands exerted by the lake bottom sediments on the overlying waters appear to be much in excess of the oxygen replenishment from the atmosphere and algal activity. This phenomenon is much pronounced in all the shallow lakes from mid-July to mid-August, when the observed DO concentrations at 2 to 4 feet from the bottom were much less than DO concentrations in the upper layers. In order to evaluate the extent of the rate of sediment

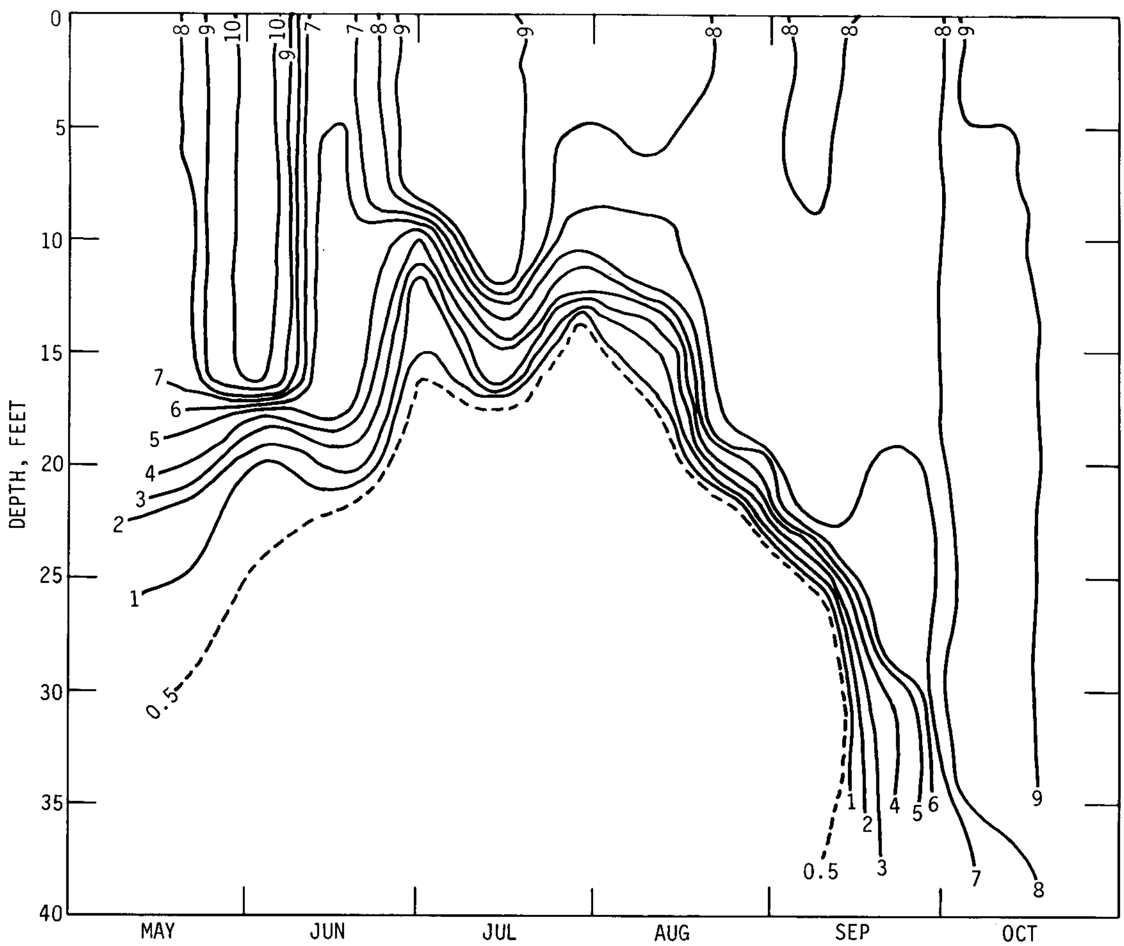

Figure 89. Isopleths of dissolved oxygen in Channel Lake

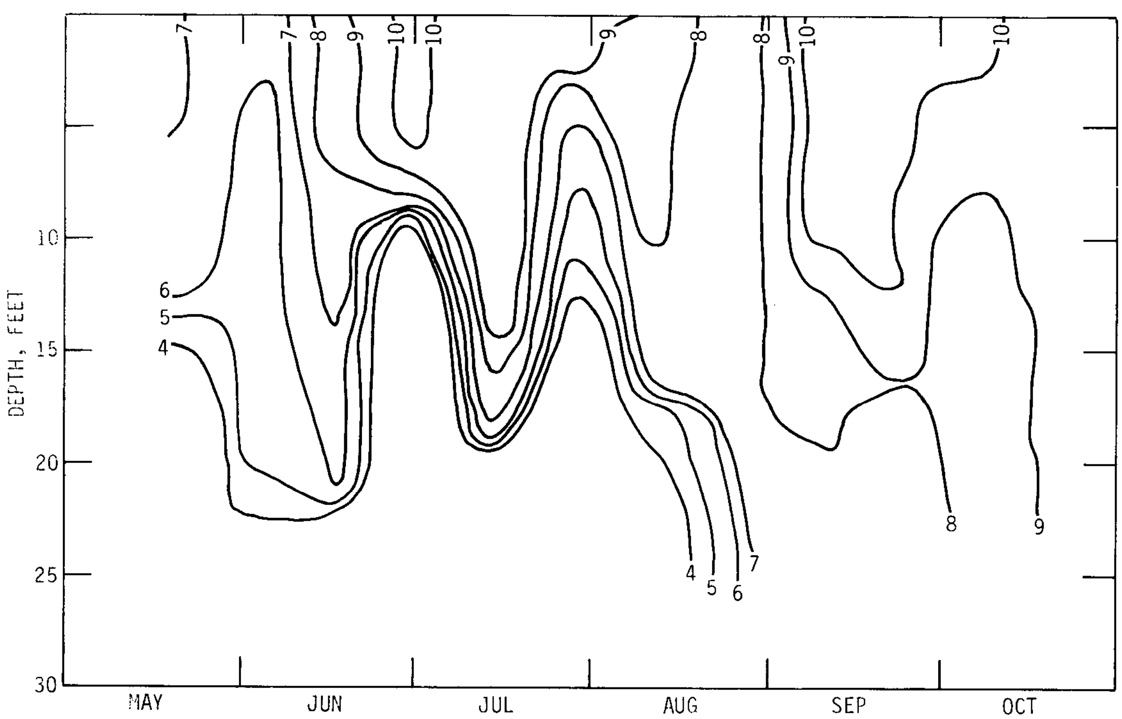

Figure 90. Isopleths of dissolved oxygen in Lake Marie (east)

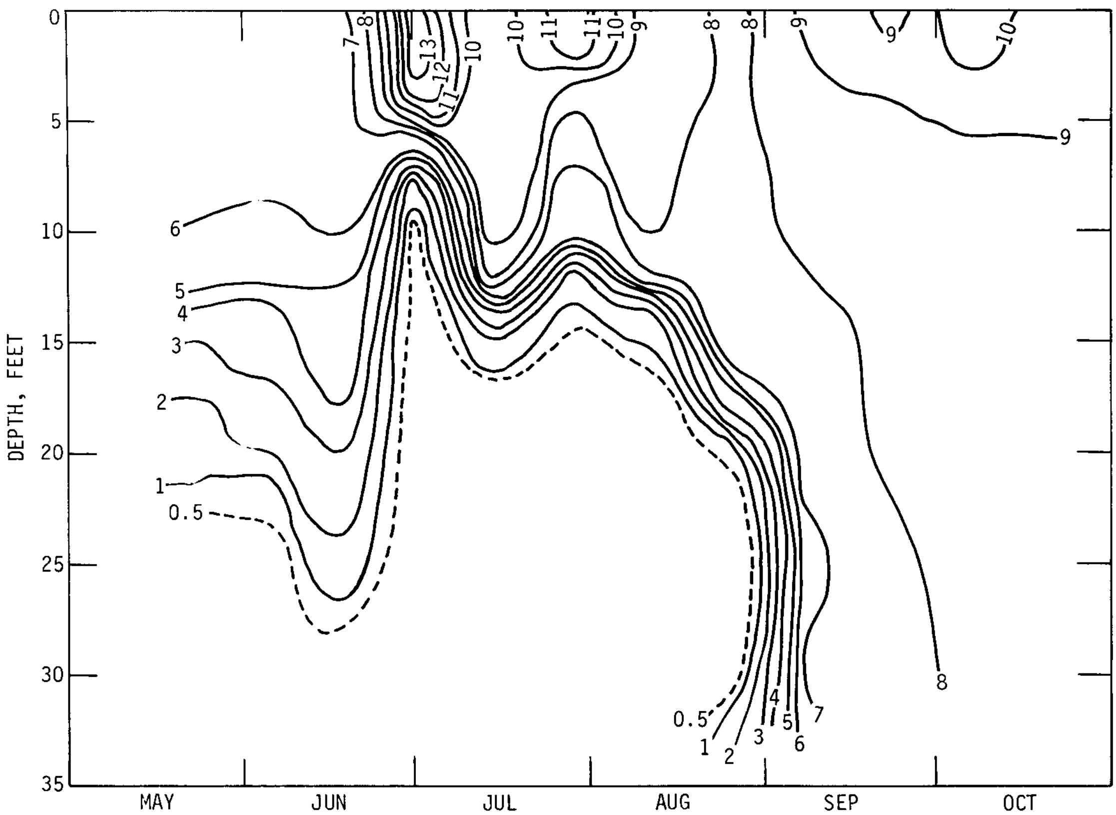

Figure 91. Isopleths of dissolved oxygen in Lake Marie (west)

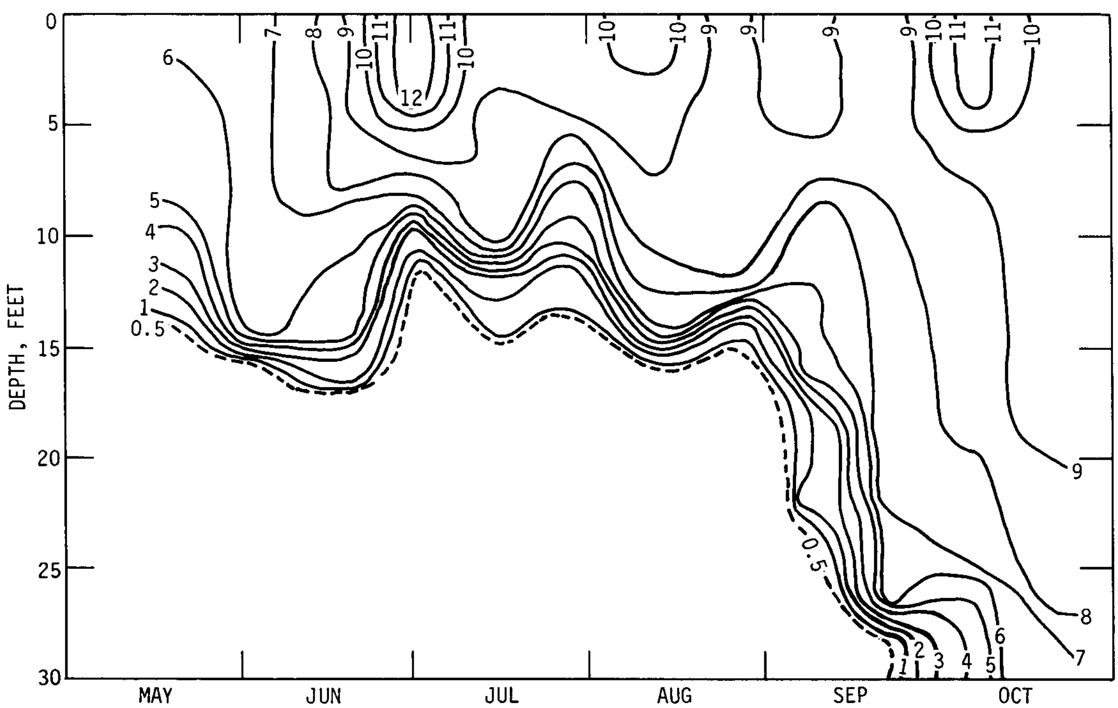

Figure 92. Isopleths of dissolved oxygen in Bluff Lake

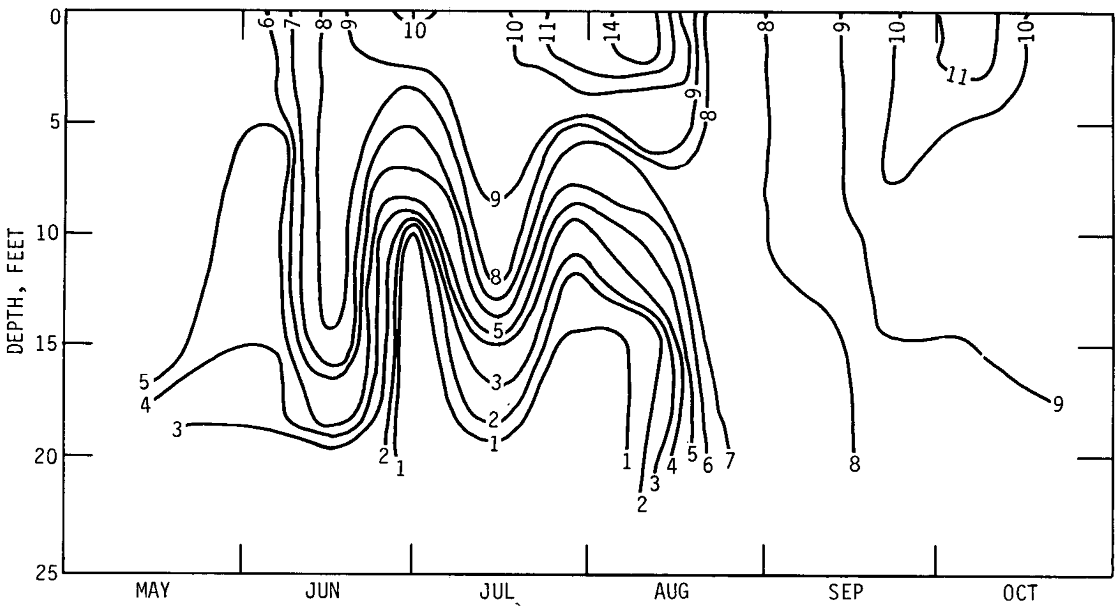

Figure 93. Isopleths of dissolved oxygen in Petite Lake

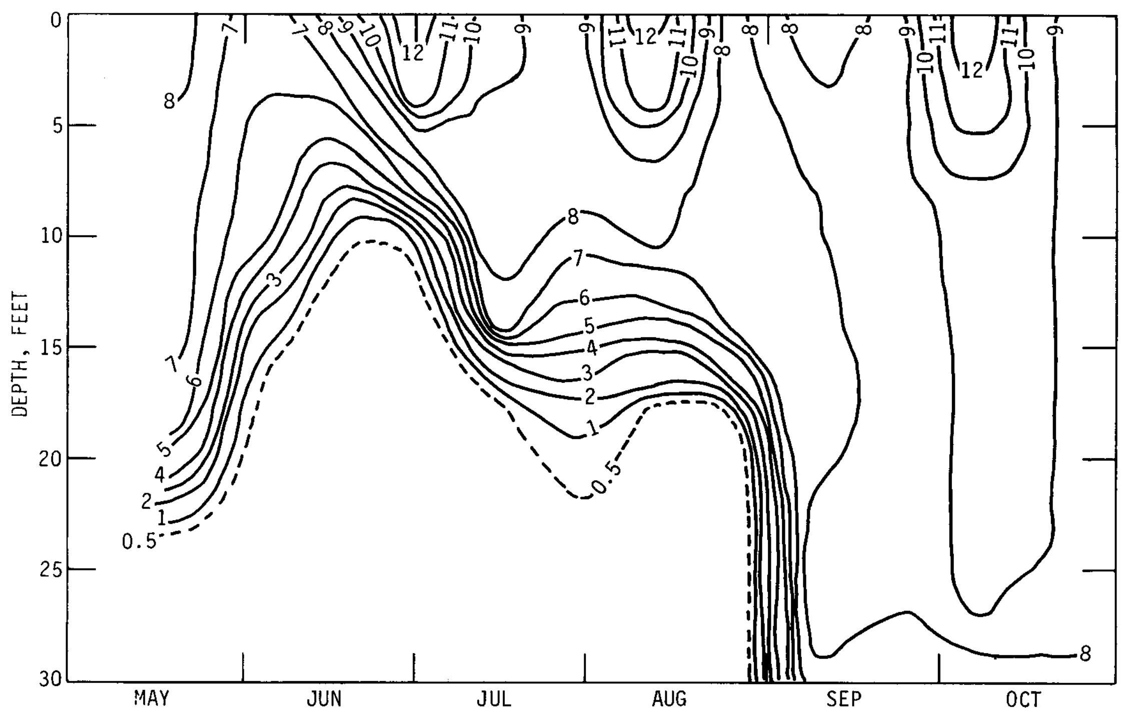

Figure 94. Isopleths of dissolved oxygen in Pistakee Bay

oxygen demand on the overlying waters, *in situ* measurements were made in these lakes. A detailed discussion on this aspect is included in the next section.

The temporal variations in the percent dissolved oxygen saturation in lake surface waters are shown in figure 96. The saturation concentrations for dissolved oxygen at the observed temperatures were computed from the following expression (Butts et al., 1973):

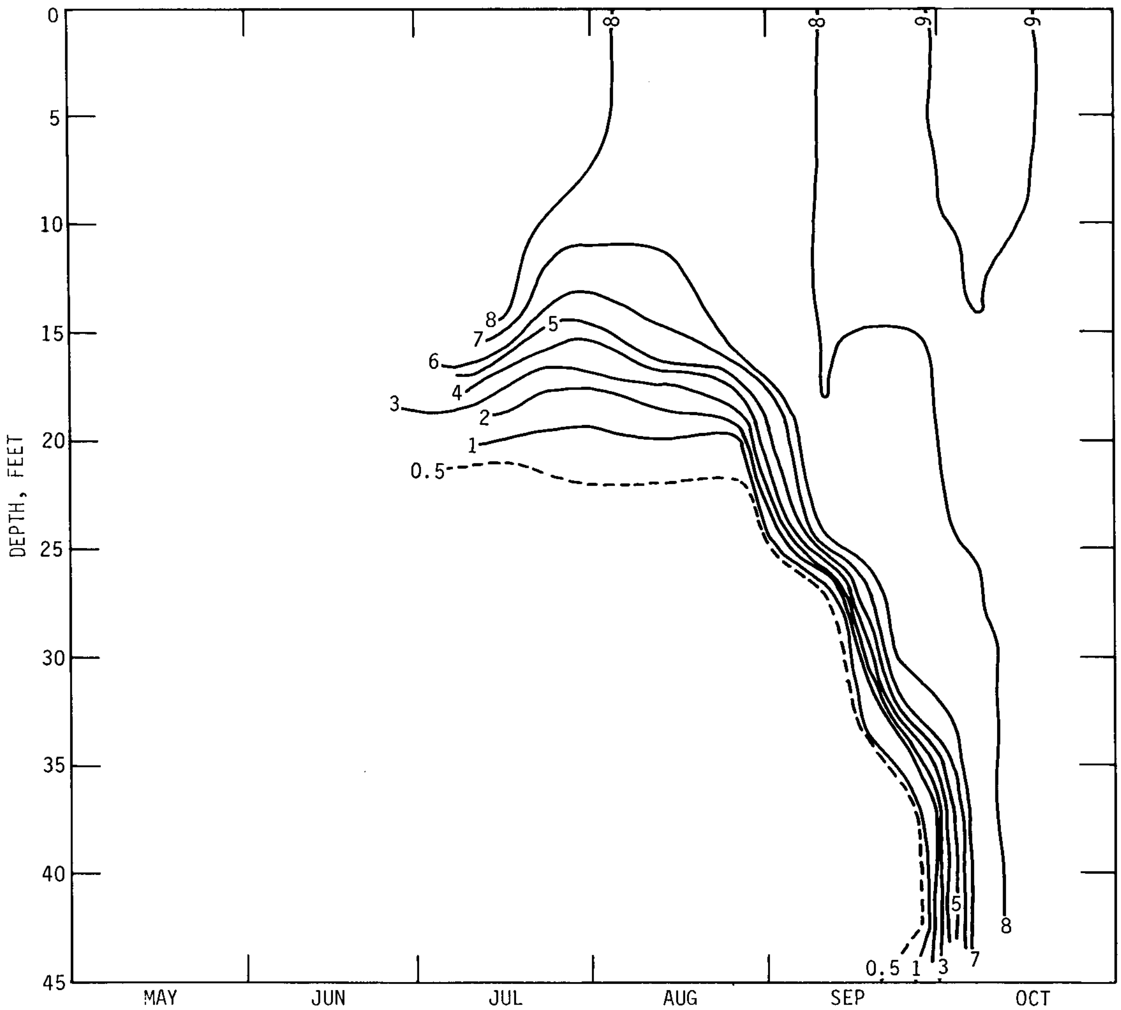

Figure 95. Isopleths of dissolved oxygen in Cedar Lake

$$DO = 14.652 - 0.41022\,T + 0.007991T^2 - 0.000077774\,T^3$$

where

DO = the dissolved oxygen, mg/l

T = water temperature, °C

At the beginning of the field investigation, the percent saturation of DO was significantly less than 100 in all the lakes with the exception of Grass Lake north and Grass Lake south (figure 96). However, supersaturated conditions prevailed during the summer months in all the lakes with the exception of Cedar Lake. The supersaturated conditions in the lakes paralleled the algal activities. Grass Lake had the highest geometric mean of algal counts, predominantly diatoms, and it remained supersaturated most of the time. The degree of supersaturation reached about 220 percent in Grass Lake and Pistakee Lake. The dissolved oxygen concentration in Cedar Lake remained near saturation or undersaturated for the entire period of field monitoring. The algal density in Cedar Lake was the lowest among the group of lakes investigated.

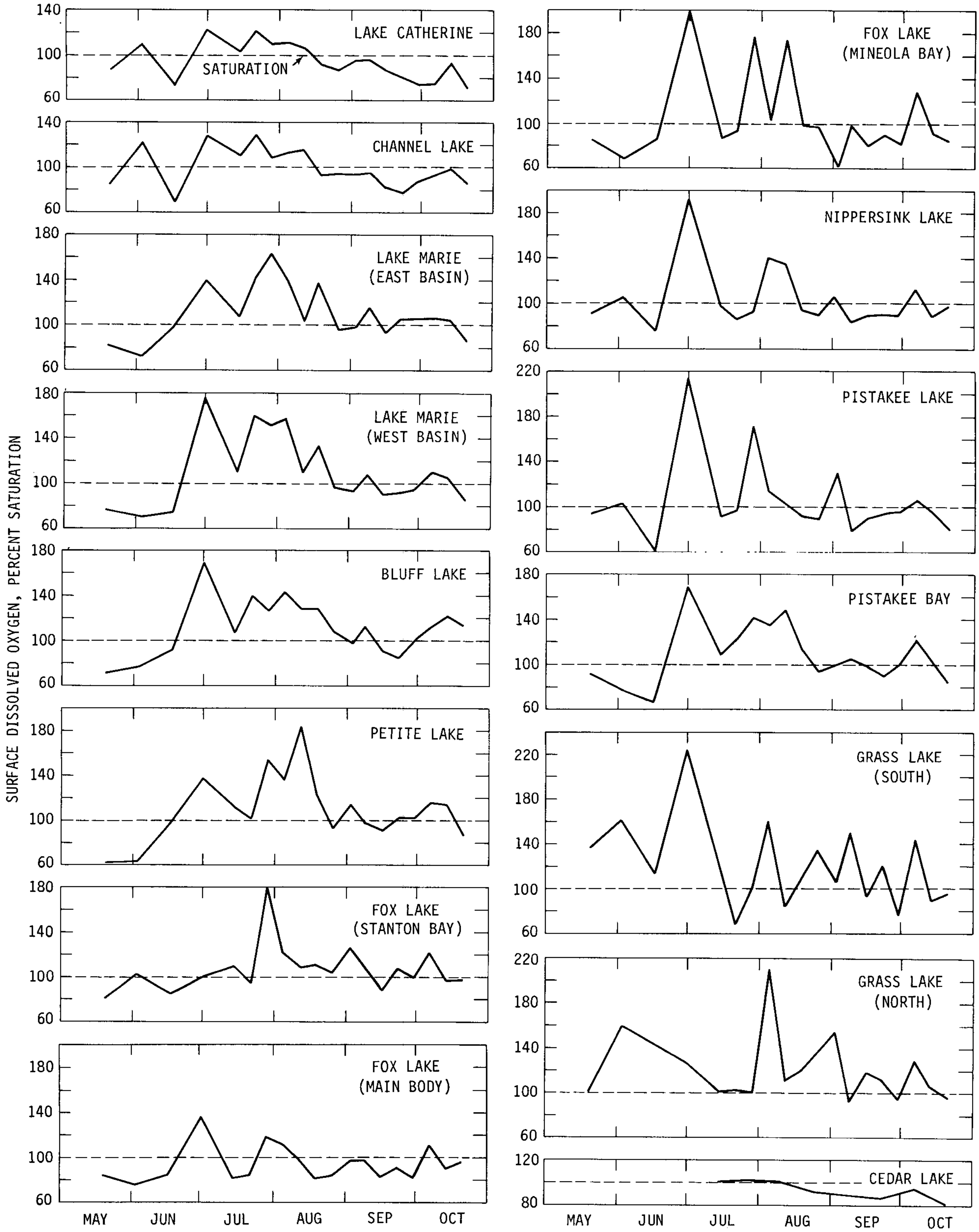

Figure 96. Lake surface dissolved oxygen, percent saturation

Sediment Oxygen Demand

In situ observations for sediment oxygen demand (SOD) rates were made with the basic techniques and equipment developed by Butts (1974). However, an improvement in the circulation equipment was implemented; the generator driven pump was replaced with a DC pump powered by a 12-volt automobile battery. A measurement was made in each lake, except for Channel Lake which was considered an extention of Lake Catherine as far as bottom sediments are concerned. Each measurement was made near the water sampling stations except for Marie (east and west) and Petite Lakes. For Lake Marie (east) measurement was made near the shallow east side; for Marie (west) and Petite Lakes measurements were made near the shallow west sides. These SOD sampling sites are shown in figure 72. Benthic sediment samples were collected in conjunction with the SOD measurements. Of interest were the consistency of the sediments (percent water), percent fixed and volatile solids, and the physical appearance and condition of the sediments before and after ashing. Results are summarized in tables 40 through 43.

The types of bottoms and the SOD rates varied within the lake system. A low SOD value of 3.73 grams per square meter per day ($g/m^2/day$) occurred in the open water area of Fox Lake. A high value of 31.58 $g/m^2/day$ occurred in Pistakee Bay. The latter value exceeds by over threefold the highest value observed for the polluted bottoms of the upper Illinois Waterway (see table 42). The highest value found in a literature search is 19.2 $g/m^2/day$. Six of the 13 lake measurements exceeded the maximum recorded for the polluted sediments of the Illinois Waterway.

Generally, the bottoms of most of the lakes were composed of silts or marls, rich in organic materials. All were in a highly reduced state; all but one of the samples emitted noticeable hydrogen sulfide smells (table 43). Many samples showed a relatively high volatile solids content reflecting the organic content of the sediments. On the basis of the SOD results alone, it is concluded that most of the bottoms throughout the lake system are in some state of degradation.

The degree to which these bottoms affect the overlying water is primarily dependent upon water depth. For example, a bottom having a significant SOD in a deep stratified lake will quickly deplete the DO in the lower depths. The rate of oxygen demand cannot be maintained by natural reaeration or by photosynthesis. Oxygen used by high SODs in shallow waters is usually naturally replenished so that the influence of the degraded bottom conditions is not quite so evident as in deeper waters. As an example, Grass Lake which is relatively shallow in most areas (less than 5 feet), had extremely high SOD rates of 14.83 and 12.65 $g/m^2/day$ in the two locations sampled. However, at the time of the SOD measurements the DO was above saturation even near the bottom. In contrast to this, the deep waters of Pistakee Bay and Lake Catherine were devoid of oxygen.

Even the lowest SOD recorded, 5.34 $g/m^2/day$ for Fox Lake, is a highly significant SOD value and would cause rapid DO depletion in deeper lakes. Without oxygen replenishment an SOD rate of 3.73 $g/m^2/day$ would deplete a 5-meter column of water saturated with dissolved oxygen at 20°C in approximately 12 days. For a severely degraded bottom such as that observed for Pistakee Bay, DO depletion would occur in less than 2 days. However, an inherent danger exists in having high SOD rates in shallow water. When the bottom sediments are resuspended physically, such as by boating activity, the SOD rate is momentarily increased several fold causing severe temporary oxygen depletion. As an example, the bottom of Grass Lake at station 14 remained disturbed for 20 minutes after lowering the sampler to the bottom; the disturbed SOD rate was 46.25 $g/m^2/day$ compared with the stabilized rate

Table 40. Sediment Oxygen Demand Rates for Fox Chain of Lakes

Lake	*Temperature T (ºC)*	*Timeframe (min)*	*SOD at TºC* (g/m²/day)	*SOD at 25ºC* (g/m²/day)
Catherine	13.8	0-2	91.63	153.27
	-	2-26	9.80	16.42
		24-53	7.76	*12.98**
Marie (east)	20.8	0-4	26.18	32.04
		4-66	5.70	*6.91**
Marie (west)	19.8	0-14	28.05	36.36
		14-49	12.72	*16.04**
Bluff	20.25	0-26	9.82	12.13
		26-60	5.58	*6.98**
Petite	20.3	0-10	13.75	16.98
		10-55	6.25	7.74
		55-103	5.59	*6.95**
Fox	20.4	0-20	4.58	5.70
		20-62	2.81	*3.44**
Stanton Bay	21.0	0-13	16.11	19.72
(Fox Lake)		13-64	5.78	*6.91**
Mineola Bay	19.0	0-6	30.54	40.24
(Fox Lake)		6-60	4.12	*5.43**
Nippersink	19.05	0-8	37.64	49.58
		8-68	6.33	*8.32**
Pistakee	19.0	0-7	14.97	19.57
		7-55	4.36	*5.76**
Pistakee Bay	20.5	0-2	144.00	177.06
		2-26	27.80	34.20
		26-41	42.76	52.58
		41-67	25.68	*31.57**
Grass (south)	18.75	0-21	21.19	28.44
		21-61	11.13	*14.76**
Grass (north)	17.9	0-20	33.38	47.11
		20-63	9.13	*12.54**

*Italicized values represent stabilized linear portion of SOD curve

Table 41. Solid-Liquid Composition of SOD Samples *(Compositions in percent)*

Lake	*Water*	*Fixed solids*	*Volatile solids*
Catherine	75.0	86.7	13.3
Marie (east)	40.0	90.7	9.4
Marie (west)	56.9	95.9	4.1
Bluff	25.5	74.4	25.6
Petite	17.5	85.3	14.7
Fox	62.2	90.8	9.2
Stanton Bay (Fox Lake)	52.3	92.8	7.2
Mineola Bay (Fox Lake)	80.4	74.2	25.8
Nippersink	79.2	79.4	20.6
Pistakee	73.8	83.3	16.7
Pistakee Bay	75.3	84.7	15.3
Grass (south)	55.8	96.7	3.3
Grass (north)	79.7	74.9	25.1
Cedar	23.7	84.2	15.8

Table 42. Comparison of SOD Values for Three Illinois Studies

Location	Number of samples	Range of values: Water (%)	Volatile solids (%)	SOD at 25°C ($g/m^2/day$)
Illinois Waterway	22	33.0-74.9	2.5-25.7	0.64-9.30*
Lake Meredosia	3	66.3-69.4	8.8-8.9	2.83-4.74
Chain of Lakes	13	17.5-80.4	4.1-25.8	3.73-31.58

*Values from Butts (1974) revised

Table 43. Descriptions of Lake Bottoms at SOD Sampling Stations

Lake	Deptb (ft)	Before incineration	After incineration
Catherine	39	Sulfide smell, thin watery gritty muck	Hard gray crust of silt and clay powderable
Marie (east)	10	Slight oily smell, muddy, gritty, slurry with fibrous material	Reddish-gray to brown with some very small shells, easily powdered
Marie (west)	10	Slight sulfide smell, small white snail shells and small-to-moderately large shell fragments, marl like	White snail shells and shell fragments, easily crushed to fine powder
Bluff	12	Sulfide smell, muddy, gritty, slurry	Clay-silt with some minute shell fragments, powderable
Petite	12	Strong sulfide smell, thin watery muddy grit	Gray hard crust, uniform silt-clay mixture
Fox	8	Slight sulfide smell, thin watery mud and small crushed shells	Very light fluffy mixture of snail and small clam shells, easily powdered
Fox (Stanton Bay)	4	Slight sulfide smell, muddy mixture of shells, fibers, and roots	Crusted mixture of calcium materials and shell fragments, easily powdered
Fox (Mineola Bay)	12	Sulfide smell, thin watery muddy grit	Moderately hard crust, light brown, powderable
Nippersink	6.5	Sulfide smell, thin watery muddy grit	Moderately hard crust light brown, powderable
Pistakee	5	Very strong sulfide smell, muddy, gritty slurry	Moderately hard crust light brown, powderable
Pistakee Bay	32	Sulfide smell, thin watery muddy grit	Moderately hard crust very light brown, powderable
Grass (south)	3.2	Slight sulfide smell, a little mud, mostly small white crushed shells	Dark snail shells and shell fragments, easily powdered
Grass (north)	3.2	Slight sulfide smell, thin watery grit	Soft reddish gray crust, easily powdered
Cedar	45	Sulfide smell, thin watery muddy grit	Fine reddish material with some leaf and wood ash, powderable

of 12.65 $g/m^2/day$. During this disturbed period the DO was lowered in the sampler by approximately 2.5 mg/l. In contrast, for a 43-minute period after restabilization occurred, the DO was lowered only 1.5 mg/l in the sampler. Table 40 lists the disturbed rates and the subsidence times for all the sampling locations. The consistency of the bottom sediments has a great influence on both the magnitude of the disturbed SOD rate and the time interval over which it is significant. SOD rates are highest in areas of algal silt type bottom sediment (figure 2).

Secchi Disc Observations

Secchi disc visibility is a measure of the lake water transparency or its ability to allow light transmission. Though the study of the light transmission by means of suitable photosensitive instruments lowered into the water has come into vogue, the very simple procedure of determining transparency, in a restricted sense, with the secchi disc still retains its value (Hutchinson, 1957).

Even though the secchi disc transparency is not an actual quantitative indication of light transmission, it serves as an index and a means of comparison of similar bodies of water or of the same body of water at different times. Since changes in water color and turbidity in a deep lake are generally caused by aquatic flora and fauna, transparency is often related to this entity. Greeson (1971) reported that in Oneida Lake in New York, the 1 percent penetration depth averaged 13.1 feet, and a concurrent secchi disc transparency reading averaged 5.6 feet which equaled the 10.5 percent penetration depth. The 1 percent penetration depth is generally defined as the lower limit of the euphotic zone.

The secchi disc observations in the lakes, and the maximum, minimum, and mean values, are shown in figure 97. The maximum of all the observations, 180 inches, was in Lake Catherine on June 3, 1975. However, Cedar Lake which is mesotrophic, exhibited the highest mean and highest minimum values. Among the Fox Chain of Lakes, Lake Catherine had the highest mean transparency value of 58 inches. The shallow lakes, namely, Fox, Nippersink, Pistakee, and Grass, showed an average value of about 10 inches.

Figure 97 reveals that the secchi disc values are somewhat related to the maximum depth in these lakes. This suggests that the boating activities and the wind and wave actions, which could stir up the bottom sediments in shallow lakes, affect significantly the depth of light penetration. Yousef (1974), assessing the effects of boating activities on water quality in shallow lakes of Florida, has documented that agitation and mixing caused by motorboats could increase the turbidity and average particle size of suspended material through the water column.

Turbidity

High turbidity affects the aesthetic quality of the water. Its origin may be municipal and industrial wastes, clastic materials derived from a drainage basin, soil erosion resulting from agricultural practices, urban and highway developments, sediments in lakes stirred by wind, wave, and high-speed boating activities in shallow lakes, detrital remains of aquatic and terrestrial plants and animals, and algae.

Lee (1974) suggests that if the turbidity in Green Bay (Lake Michigan) were reduced, more profuse algal growth might be encountered. Wang (1974) experimented with Illinois and Fox River waters to delineate the effects of turbidity on algal growth. He reported that unfiltered river water samples showed less algal growth than filtered samples under laboratory conditions. However, during periods of low nutrients in the water samples, unfiltered samples stimulated algal growth rather than retarded it. He concluded that the effect, inhibitory or stimulatory, may be related to the quantity of nutrients in solution.

Temporal variations of turbidity in three of the lakes investigated are shown in figures 98 through 104. These figures, in addition to turbidity, show the temporal variations of all of the chemical parameters evaluated. Figures 98, 99, and 100 pertain to Lake Catherine surface, mid-depth, and deep water samples, respectively. These are typical of the deep lakes in the Fox Chain. Figure 101 shows the temporal variations of the parameters in Fox Lake (main body) which is typical of the shallow lakes. Figures 102, 103, and 104 represent the water quality characteristics in the mesotrophic Cedar Lake. Table 44 summarizes the observed data for turbidity and the chemical parameters.

The turbidity of surface and mid-depth samples of deeper lakes had relatively low values ranging from 2 to 12 FTU. Turbidity in Cedar Lake was generally the lowest. This is mainly because the lake is entirely springfed and the horsepower of the motor boats used for recreational purposes is restricted to 5 hp. The deep water samples in the Chain of

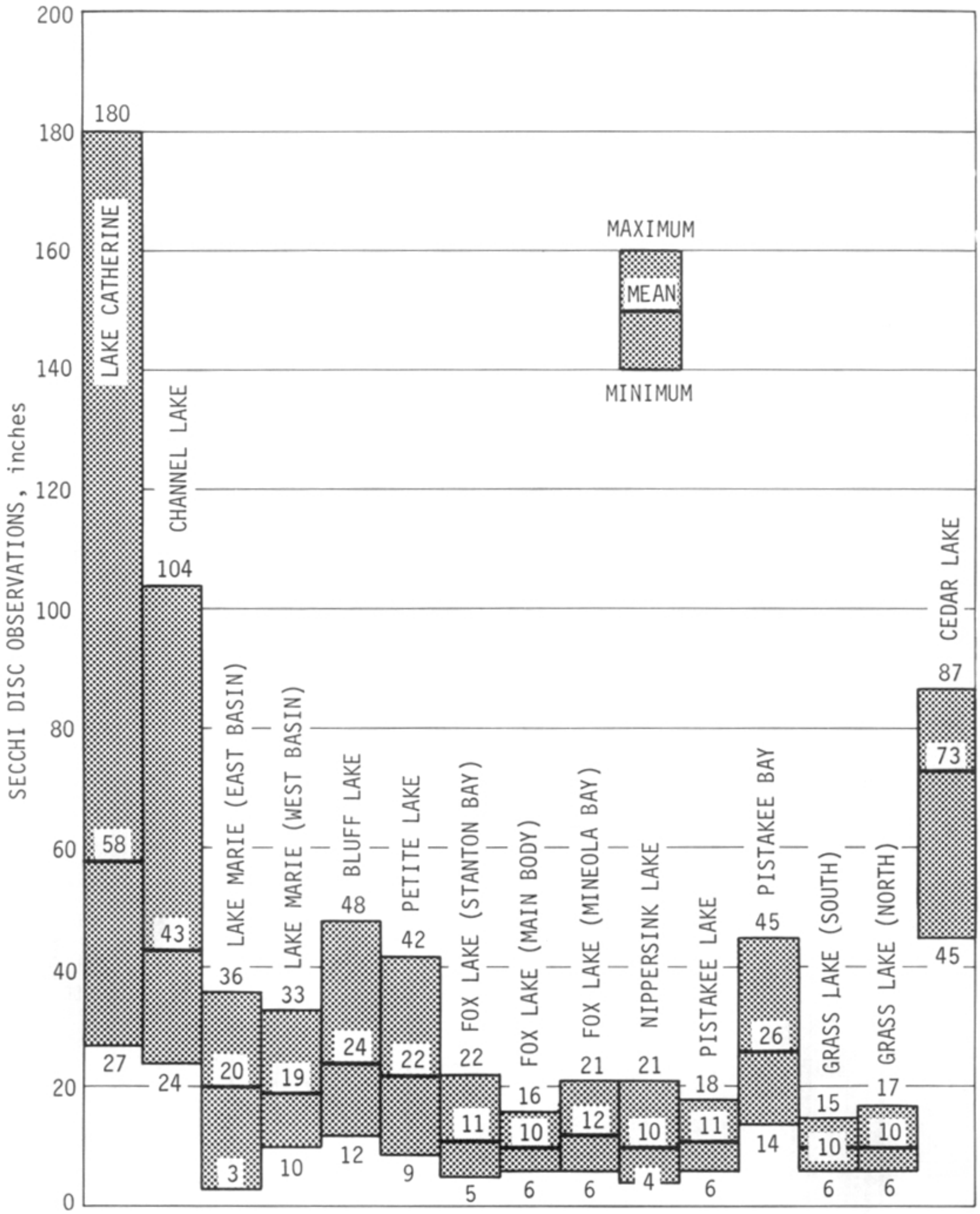

Figure 97. Secchi disc observations

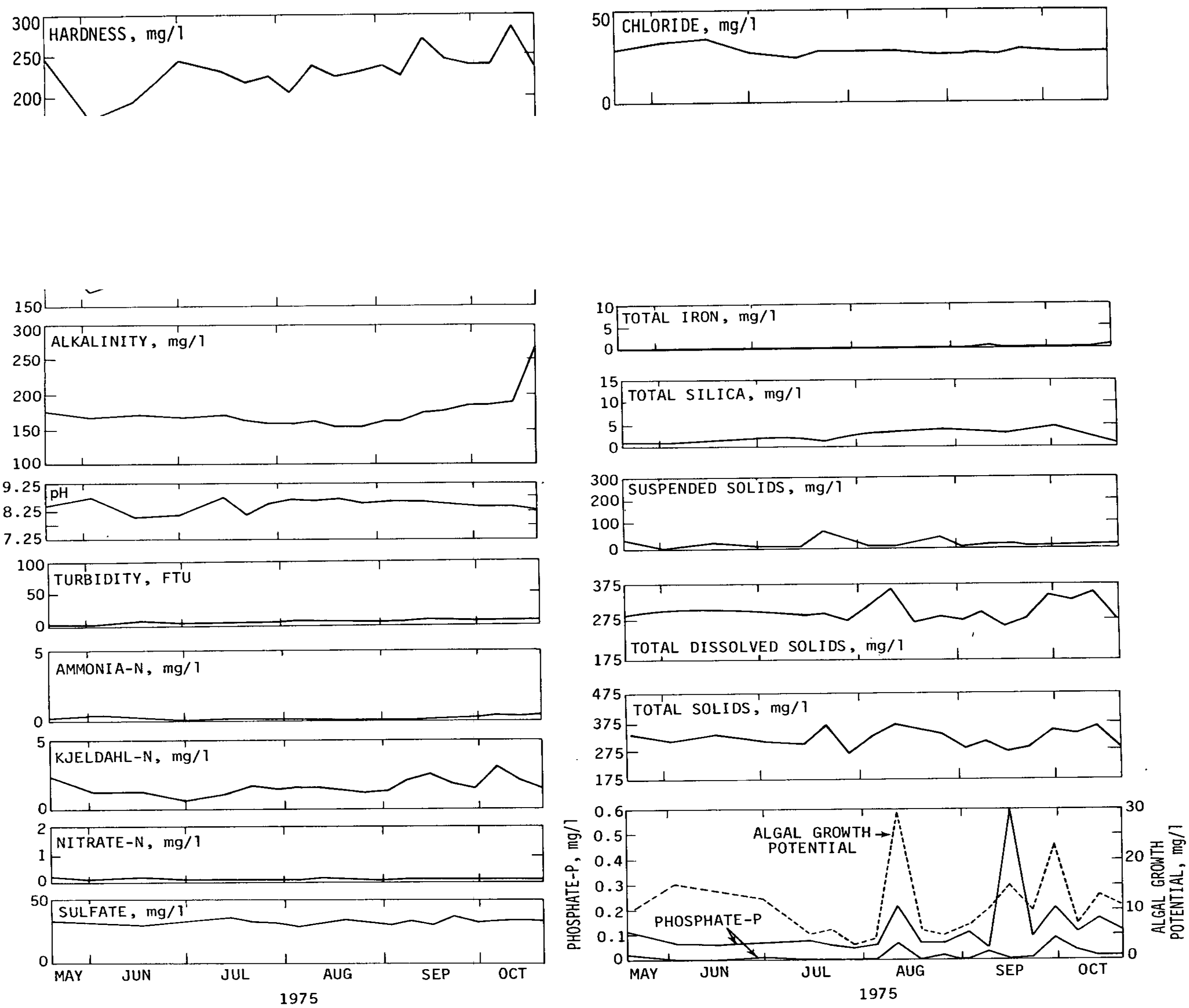

Figure 98. Temporal variations in water quality characteristics in Lake Catherine (surface)

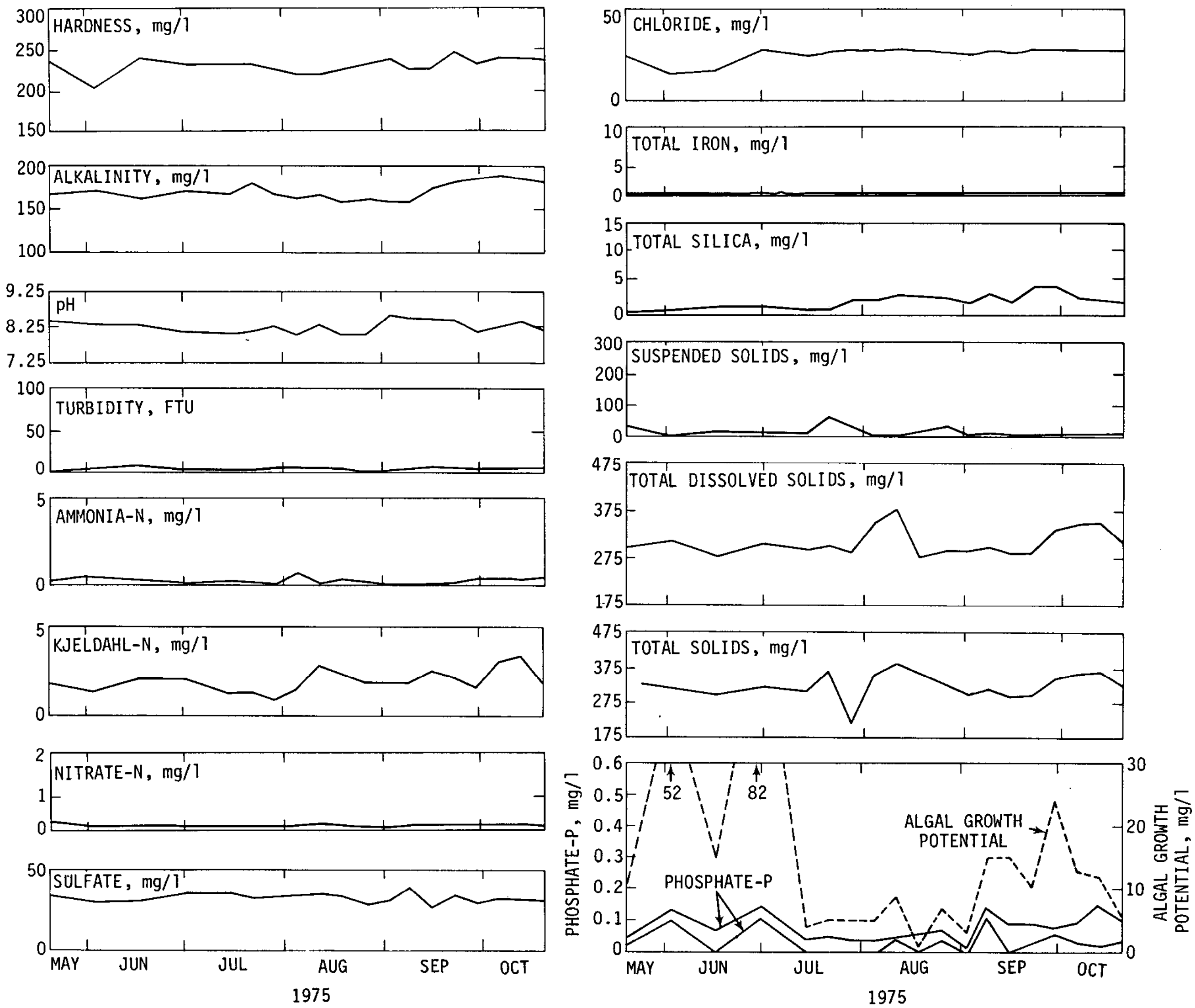

Figure 99. Temporal variations in water quality characteristics in Lake Catherine (mid-depth)

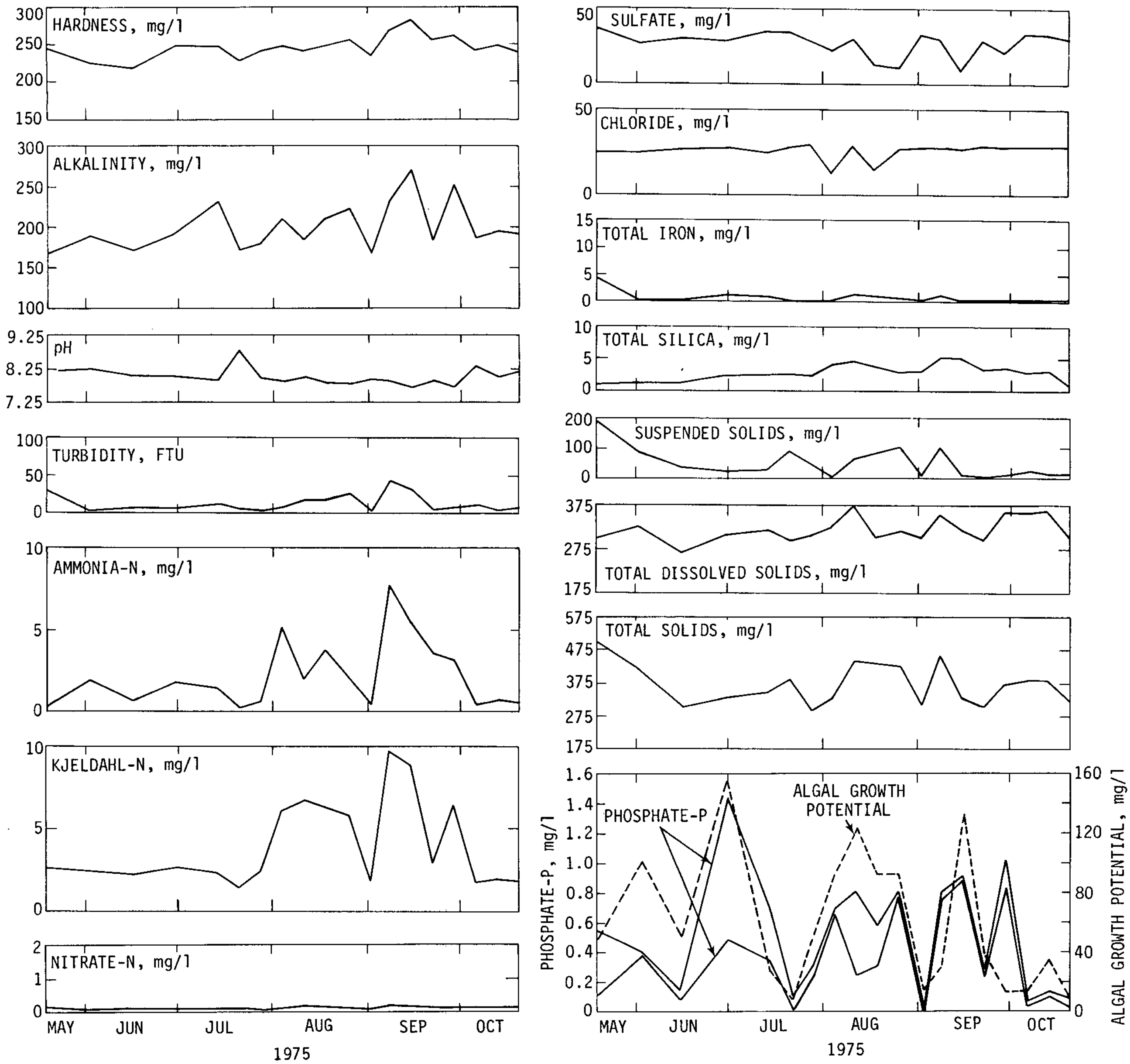

Figure 100. Temporal variations in water quality characteristics in Lake Catherine (deep)

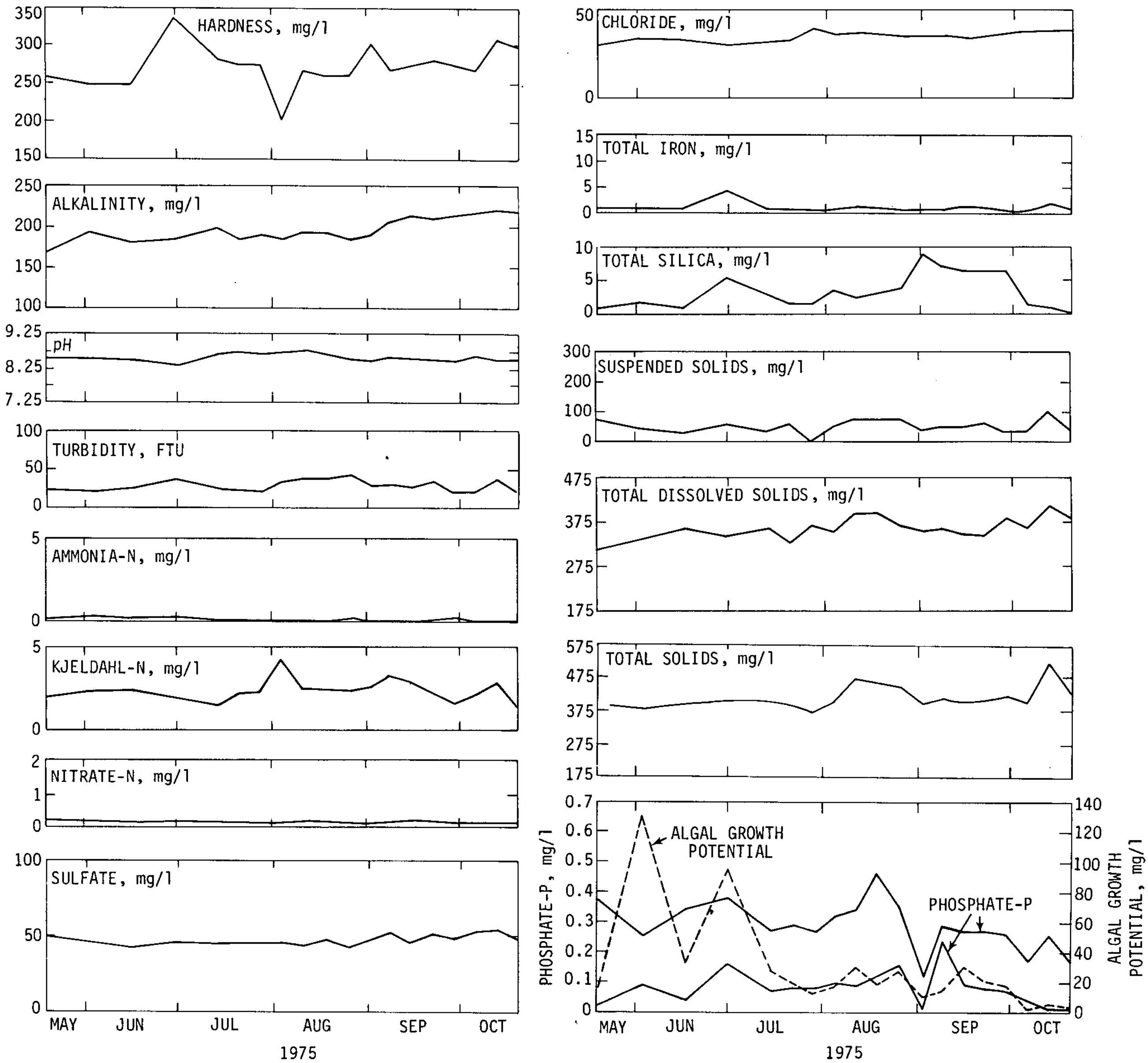

Figure 101. Temporal variations in water quality characteristics in Fox Lake (main body)

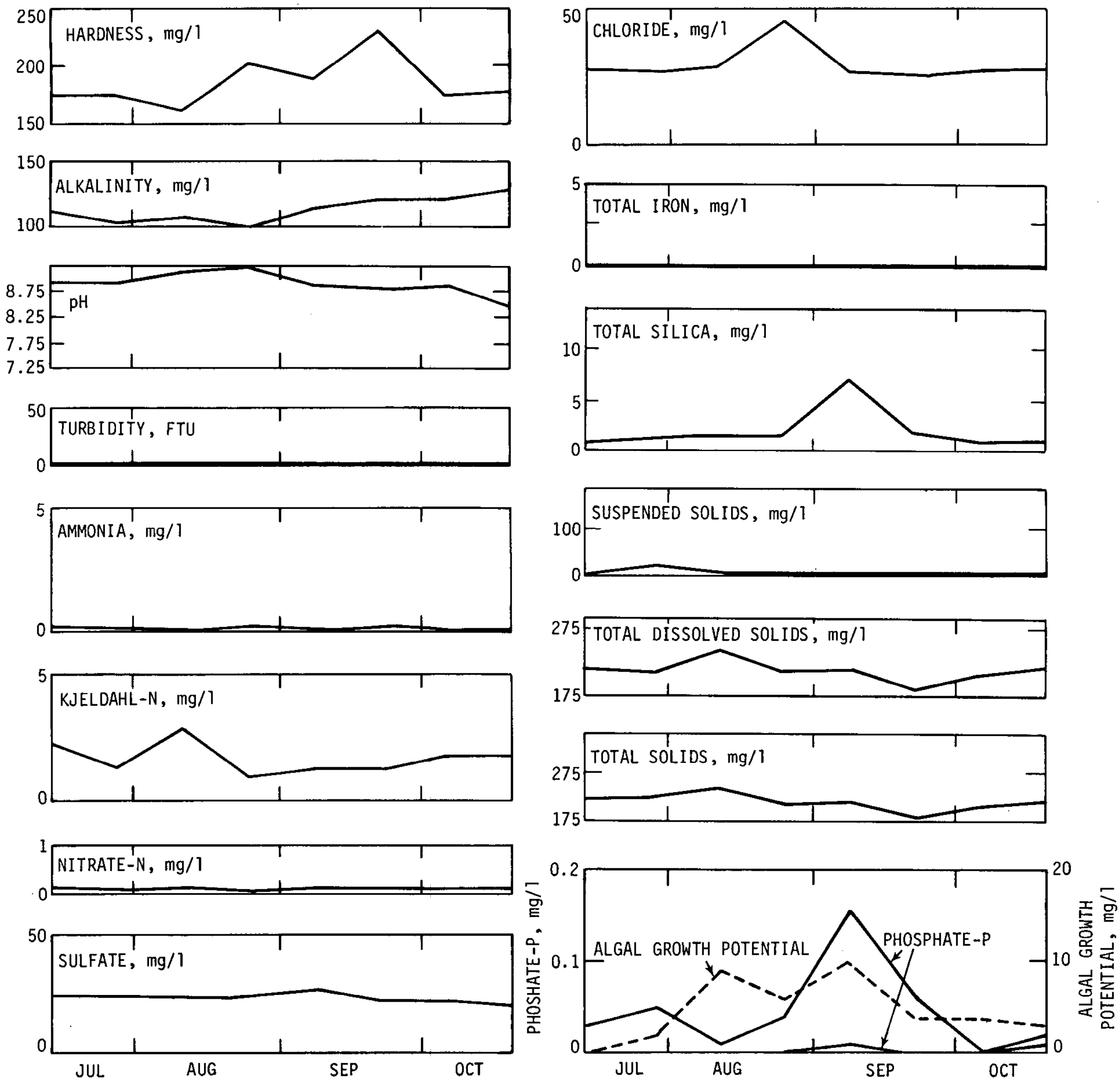

Figure 102. Temporal variation in water characteristics in Cedar Lake (surface)

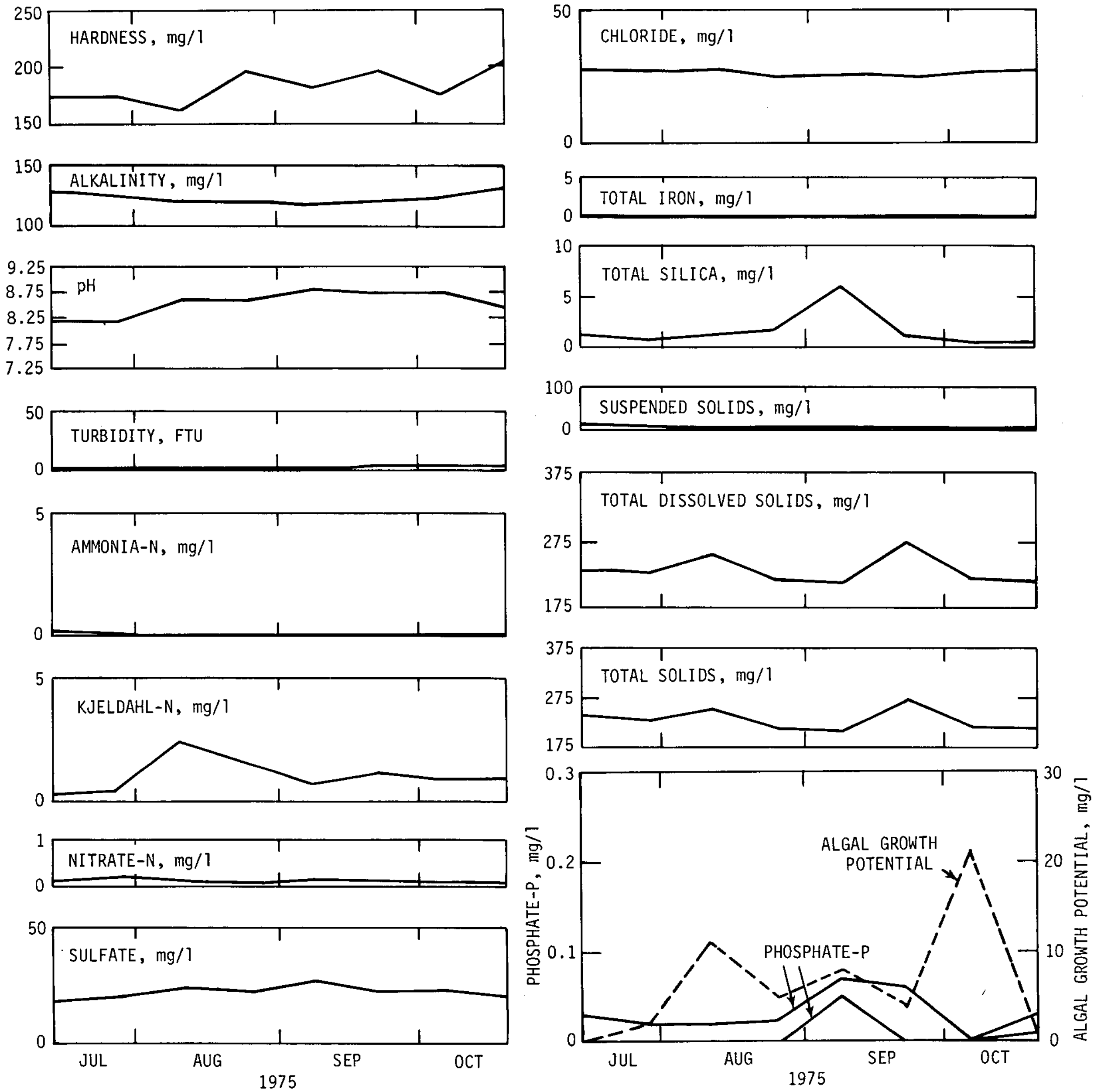

Figure 103. Temporal variations in water quality characteristics in Cedar Lake (mid-depth)

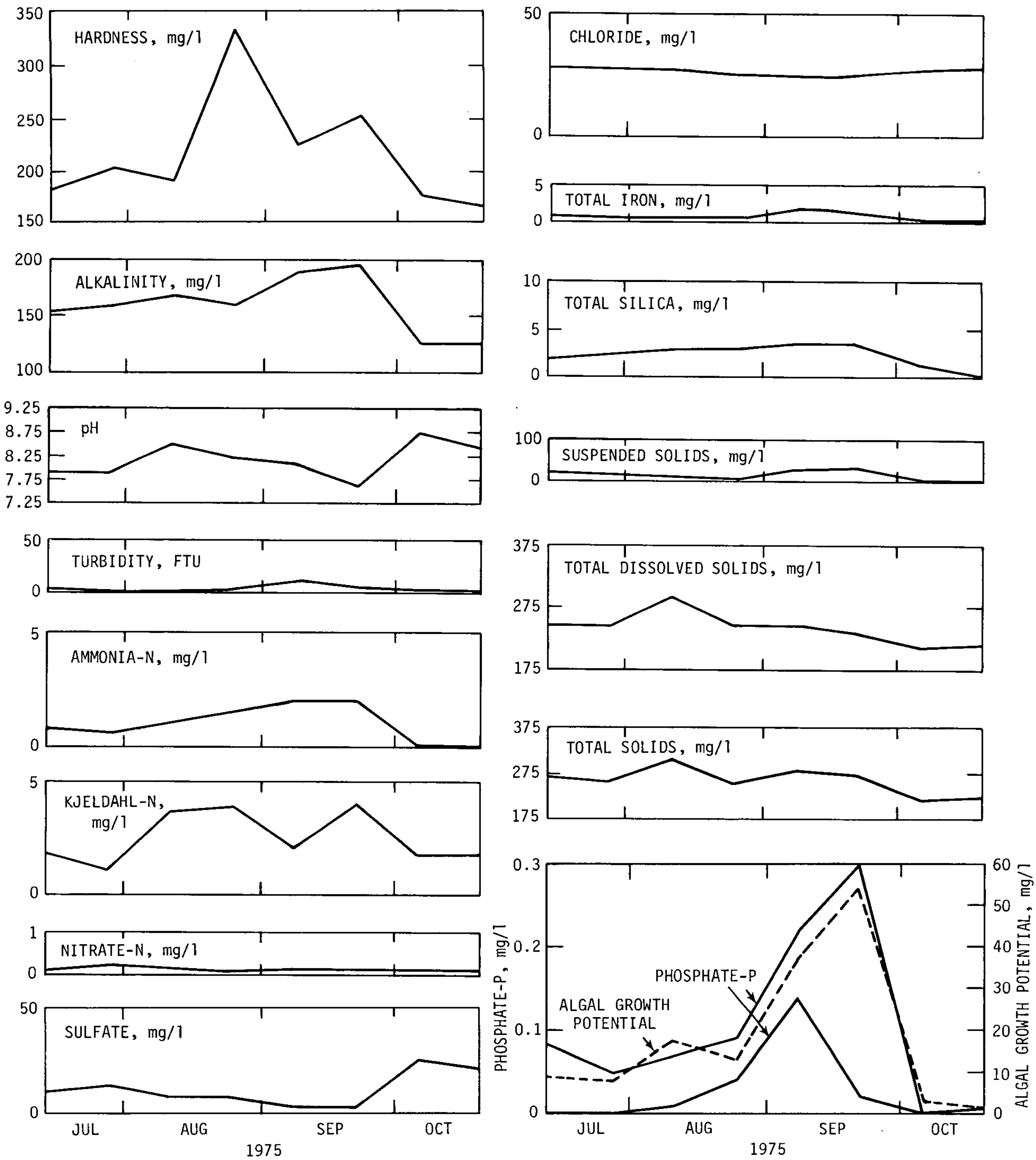

Figure 104. Temporal variations in water quality characteristics in Cedar Lake (deep)

Table 44. Summary of Water Quality Characteristics for Fox Chain of Lakes

(Concentrations in milligrams per liter)

Lake		*Depth at sampling point (ft)*	*Turbidity (FTU)* *Mean*	*Turbidity (FTU)* *Range*	*pH* *Range*	*Alkalinity* *Mean*	*Alkalinity* *Range*	*Hardness* *Mean*	*Hardness* *Range*	*Nitrate-N* *Mean*	*Nitrate-N* *Range*	*Kjeldahl-N* *Mean*	*Kjeldahl-N* *Range*
Catherine	Surface	39	5.4	2.2-7.7	8.01-8.72	176	155-270	231	170-285	0.13	0.07-0.26	1.62	0.61-2.97
	Mid-depth		4.3	1.4-7.2	8.04-8.59	173	160-192	228	200-244	0.13	0.06-0.25	1.91	0.85-3.37
	Deep		12.5	1.7-43.0	7.68-8.78	203	169-274	242	215-278	0.12	0.06-0.19	3.81	1.35-4.57
Channel	Surface	35	6.9	3.2-11.0	8.32-8.92	171	155-200	225	196-251	0.12	0.09-0.16	1.70	0.36-3.15
	Mid-depth		6.2	1.3-10.0	8.06-8.78	171	155-188	226	193-273	0.13	0.09-0.18	1.44	0.18-2.30
	Deep		14.0	2.1-77.0	7.58-8.80	200	164-278	240	185-285	0.13	0.07-0.19	4.21	0.90-13.10
Marie (east)	Surface	22	15.7	7.1-26.0	8.33-9.18	186	164-212	253	213-298	0.14	0.08-0.18	2.29	1.17-3.93
	Deep		17.7	6.8-52.0	8.20-8.92	187	160-212	259	226-278	0.14	0.08-0.19	2.43	1.05-4.25
Marie (west)	Surface	31	14.0	8.0-25.0	8.42-9.11	189	173-212	256	230-285	0.16	0.09-0.31	2.28	1.36-3.3.5
	Mid-depth		13.0	5.9-25.0	8.18-8.99	187	169-212	258	219-298	0.15	0.09-0.25	2.06	0.89-3.04
	Deep		21.9	3.6-72.0	7.68-8.78	207	155-271	266	200-291	0.15	0.08-0.23	4.13	1.18-12.50
Bluff	Surface	27	11.9	3.9-18.0	8.28-9.11	184	160-212	256	237-318	0.15	0.08-0.28	2.08	0.61-3.86
	Deep		19.3	3.5-84.0	7.45-8.80	215	173-275	267	207-305	0.14	0.09-0.22	5.31	1.31-12.90
Petite	Surface	17	14.5	6.4-31.0	7.97-9.18	186	160-216	253	222-305	0.15	0.10-0.23	2.49	1.14-4.21
	Deep		20.6	6.4-40.0	8.21-8.86	186	160-212	256	215-305	0.16	0.10-0.25	2.69	1.15-3.96
Fox (Stanton Bay)	Surface	4.4	29.0	11.0-63.0	8.30-8.91	194	160-223	271	222-332	0.17	0.11-0.25	2.46	1.03-5.84
Fox (main)	Surface	7.2	28.2	20.0-44.0	8.33-8.78	199	169-223	269	196-332	0.16	0.12-0.23	2.30	1.27-4.20
Fox (Mineola Bay)	Surface	12	22.7	16.0-50.0	8.44-8.91	196	173-227	272	196-325	0.17	0.12-0.29	2.55	1.45-4.08
	Deep		26.1	12.0-67.0	8.42-8.88	198	164-227	272	233-305	0.18	0.12-0.27	2.30	1.30-3.78
Nippersink	Surface	4.9	27.9	17.0-45.0	8.32-8.96	202	178-235	275	237-325	0.16	0.11-0.23	2.72	1.48-5.06
Pistakee (main)	Surface	5.4	23.5	15.0-32.0	8.32-8.88	211	182-255	288	244-330	0.26	0.11-0.47	2.52	1.26-3.55
Pistakee Bay	Surface	30	11.9	5.4-21.0	8.33-9.03	200	164-231	271	210-312	0.16	0.09-0.28	1.85	0.70-3.01
	Mid-depth		11.5	4.2-19.0	8.29-8.91	203	164-239	280	222-318	0.18	0.10-0.44	1.66	0.48-2.97
	Deep		22.1	2.7-82.0	7.54-8.62	235	178-320	290	219-393	0.16	0.08-0.23	4.75	0.97-15.50
Grass (south)	Surface	3.7	22.1	15.0-44.0	8.25-8.89	211	169-243	276	224-332	0.14	0.10-0.19	2.12	0.82-4.98
Grass (north)	Surface	3.3	20.7	10.0-38.0	8.36-8.96	207	164-235	278	237-305	0.15	0.10-0.24	2.49	1.28-3.91
Cedar	Surface	42	1.8	0.9 2.6	8.40-9.19	113	98-129	180	156-224	0.10	0.05-0.13	1.60	0.88-2.78
	Mid-depth		2.1	1.4-3.2	8.15-8.79	123	118-133	177	156-198	0.13	0.07-0.21	1.03	0.28-2.36
	Deep		4.1	1.4-12.0	7.57-8.69	160	125-196	209	158-325	0.12	0.07-0.21	2.46	0.96-3.81

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Table 44. (Continued)

(Concentrations in milligrams per liter)

Lake		*Depth at sampling point (ft)*	*Ammonia-N*		*Total silica*		*Total iron*		*Chloride*		*Sulfate*		*Total solids*	
			Mean	*Range*	*Mean*	*Range*	*Mean*	*Range*	*Mean*	*Range*	*Mean*	*Range*	*Mean*	*Range*
Catherine	Surface	39	0.15	0.01-0.39	2.23	0.80-3.76	0.11	0.03-0.30	28	24-35	32	29-36	322	268-374
	Mid-depth		0.25	0.01-0.67	1.76	0.40-3.66	0.15	0.04-0.37	25	14-28	33	27-40	321	214-387
	Deep		2.12	0.20-7.50	2.61	0.62-4.71	0.54	0.05-4.03	25	12-29	26	8-36	367	290-502
Channel	Surface	35	0.09	0.00-0.24	2.59	0.00-5.74	0.21	0.04-1.56	26	23-29	33	27-36	327	280-371
	Mid-depth		0.15	0.02-0.74	3.13	0.00.11.92	0.15	0.01-0.57	27	23-29	34	28-40	335	284-535
	Deep		2.17	0.07-8.48	3.99	0.00.13.85	0.79	0.08-5.58	25	10.30	27	2-39	386	264-818
Marie (east)	Surface	22	0.09	0.00-0.59	2.65	0.00-5.63	0.33	0.10-0.65	33	25-37	44	36-50	377	317-458
	Deep		0.29	0.02-1.69	3.01	0.18-6.97	0.58	0.18-2.48	33	28-38	45	39-58	388	336-542
Marie (west)	Surface	31	0.11	0.01-0.35	2.47	0.07-5.90	0.31	0.11-0.75	34	30-38	45	36-52	374	316-419
	Mid-depth		0.18	0.01-0.58	2.40	0.0-6.76	0.37	0.19-0.74	34	29-38	44	34-50	377	324-420
	Deep		1.22	0.06-3.85	3.43	0.18-7.40	1.00	0.15-3.75	34	29-38	38	13-51	420	342-576
Bluff	Surface	27	0.12	0.01-0.37	2.83	0.00-8.25	0.24	0.04-0.46	34	29-40	45	36-79	369	330-434
	Deep		2.76	0.05-9.15	3.76	0.04-7.20	0.90	0.06-4.92	31	14-37	36	7-56	404	324-650
Petite	Surface	17	0.09	0.00-0.56	3.57	0.00-6.64	0.29	0.10-0.59	33	24-38	43	35-52	367	324-418
	Deep		0.21	0.01-0.91	3.60	0.18-6.04	0.75	0.15-2.06	33	29-38	45	36-51	394	336-489
Fox (Stanton Bay)	Surface	4.4	0.10	0.00-0.30	3.76	0.00-9.24	1.01	0.17-5.88	34	30.39	46	40.55	417	367-478
Fox (main)	Surface	7.2	0.12	0.02-0.33	3.48	0.00-8.67	0.95	0.31-4.01	35	30-39	48	42-55	415	372-519
Fox (Mineola Bay)	Surface	12	0.10	0.00-0.27	3.86	0.08-10.55	0.58	0.16-1.36	34	28-42	49	44-59	398	574-433
	Deep		0.19	0.02-0.67	4.14	0.08-8.72	0.85	0.27-2.34	35	28-41	48	43-55	427	366-544
Nippersink	Surface	4.9	0.06	0.00-0.14	3.27	0.28-7.38	1.19	0.56-2.79	35	29-42	50	41-58	421	354-488
Pistakee (main)	Surface	5.4	0.10	0.01-0.27	3.66	0.00-9.66	1.11	0.39-3.17	34	29-38	49	40-56	426	376-504
Pistakee Bay	Surface	30	0.11	0.00-0.53	2.22	0.00-6.60	0.23	0.07-0.60	33	28-38	47	40-56	385	346-436
	Mid-depth		0.19	0.03-0.68	2.91	0.25-6.61	0.28	0.11-0.66	32	26-38	48	41-56	390	334-474
	Deep		3.55	0.05-16.38	4.98	0.18-9.31	0.77	0.14-3.36	31	14-39	38	8-54	423	328-624
Grass (south)	Surface	3.7	0.08	0.00-0.16	1.96	0.32-6.30	1.72	0.32-5.38	37	30-45	49	36-55	441	326-778
Grass (north)	Surface	3.3	0.05	0.00-0.11	1.11	0.00-3.12	1.17	0.30-3.14	35	33-44	49	33-64	418	550-526
Cedar	Surface	42	0.08	0.02-0.14	1.90	0.81-6.65	0.07	0.00-0.15	29	25-45	23	20-27	216	185-243
	Mid-depth		0.06	0.02-0.11	1.62	0.43-5.88	0.06	0.00-0.09	27	25-28	22	18-27	232	210-272
	Deep		0.96	0.01-1.94	2.59	0.00-3.47	0.60	0.06-1.70	26	24-28	11	2-24	257	212-303

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Table 44. (Concluded)

(Concentrations in milligrams per liter)

Lake		*Depth at sampling point (ft)*	*Dissolved solids*		*Suspended solids*		*AGP*		*Total phosphorus*		*Dissolved orthophosphorus*	
			Mean	*Range*	*Mean*	*Range*	*Mean*	*Range*	*Mean*	*Range*	*Mean*	*Range*
Catherine	Surface	39	307	264-370	16	0-70	10	3-23	0.12	0.05-0.60	0.03	0.00-0.09
	Mid-depth		311	278-384	14	0.62	15	1-82	0.08	0.01-0.15	0.05	0.00-0.11
	Deep		322	268-376	47	1-202	59	7-147	0.51	0.01-1.44	0.39	0.00-0.90
Channel	Surface	35	308	270.358	20	0-80	10	1-23	0.12	0.05-0.30	0.04	0.00-0.15
	Mid-depth		316	264-526	20	0-62	14	1-56	0.10	0.01-0.26	0.03	0.00-0.12
	Deep		318	276-380	77	0-448	67	5-180	0.50	0.05-1.63	0.42	0.00-1.24
Marie (east)	Surface	22	343	300-396	37	0-158	27	5-103	0.23	0.09-0.43	0.07	0.00-0.19
	Deep		349	304-396	43	2-162	33	7-145	0.27	0.05-0.83	0.12	0.01-0.68
Marie (west)	Surface	31	346	302-394	33	0-96	22	3-71	0.20	0.08-0.33	0.07	0.00-0.23
	Mid-depth		351	314-392	29	4-78	27	1-101	0.22	0.09-0.37	0.08	0.00-0.19
	Deep		361	318-408	66	6-246	62	3-173	0.63	0.14-1.78	0.50	0.00-1.74
Bluff	Surface	27	346	306-416	25	2-74	23	8-64	0.21	0.06-0.52	0.08	0.00-0.19
	Deep		352	306-404	57	6-320	76	2-182	0.81	0.11-1.90	0.65	0.00-1.81
Petite	Surface	17	344	302-384	25	0-49	24	1-92	0.22	0.10-0.32	0.07	0.00-0.16
	Deep		344	294-396	52	2-148	28	4-137	0.28	0.15-0.50	0.11	0.00-0.32
Fox (Stanton Bay)	Surface	4.4	361	322-398	55	14-124	20	1-72	0.25	0.07-0.39	0.06	0.00-0.11
Fox (main)	Surface	7.2	364	316-414	53	2-105	28	2-129	0.28	0.12-0.38	0.08	0.01-0.24
Fox (Mineola Bay)	Surface	12	359	316-400	39	12-82	29	3-95	0.27	0.14-0.41	0.09	0.01-0.14
	Deep		361	320-410	69	20-184	30	10.101	0.28	0.10-0.46	0.10	0.01-0.18
Nippersink	Surface	4.9	363	338-408	60	33-134	21	2-55	0.30	0.16-0.41	0.06	0.00-0.12
Pistakee (main)	Surface	5.4	368	324-408	56	15-138	23	9-57	0.27	0.12-0.40	0.06	0.00-24.00
Pistakee Bay	Surface	30	362	324-418	24	8-104	30	5-203	0.17	0.08-0.25	0.06	0.00-0.13
	Mid-depth		363	280-436	29	0-124	26	10.100	0.18	0.09-0.27	0.07	0.00-0.16
	Deep		372	252-484	57	4-254	80	8-216	0.87	0.08-2.61	0.73	0.00-2.47
Grass (south)	Surface	3.7	366	315-422	80	0-432	15	3-28	0.33	0.10-0.63	0.03	0.00-0.10
Grass (north)	Surface	3.3	360	316-402	64	22-148	17	1-64	0.27	0.14-0.39	0.03	0.00-0.06
Cedar	Surface	42	212	184-242	4	1-16	5	0-10	0.05	0.00-0.13	0.00	0.00-0.01
	Mid-depth		229	210-272	3	0-12	7	0-21	0.04	0.00-0.07	0.01	0.00-0.05
	Deep		242	210-294	15	2-34	17	1-54	0.12	0.00-0.30	0.03	0.00-0.14

Lakes exhibited high turbidity. It was most likely due to the active decomposition of the bottom sediments. The turbidity of deep water samples in Cedar Lake was relatively low, because of the low productivity of that lake.

The surface turbidities in the shallow lakes, Fox, Nippersink, Pistakee, and Grass, were much higher than surface turbidities of deeper lakes. This must be primarily due to the resuspension of bottom sediments in the shallow lakes. Grass Lake, throughout the period of this investigation, exhibited a brownish color akin to soil and humus. The other shallow lakes reflected a greenish hue during periods of algal bloom.

Chemical Characteristics.

pH

It is generally considered that pH values above 8.0 in natural waters are produced by a photosynthetic rate that demands more carbon dioxide than the quantities furnished by respiration and decomposition (Mackenthum, 1969). Values of pH below 8.0 indicate failure of photosynthesis to utilize completely the carbon dioxide so produced. Photosynthesis by aquatic plants utilizes carbon dioxide, removing it from bicarbonate and producing carbonate, when no free carbon dioxide exists in the water medium. Carbonates of calcium and magnesium, which are weakly soluble tend to precipitate out. Decomposition and respiration tend to reduce pH and increase bicarbonates, whereas the tendency of photosynthesis is to raise pH and reduce bicarbonates.

As seen from figures 98, 101, and 102 and table 44, the pH values of lake surface water samples were never less than 8.0 and they generally fluctuated, with very few exceptions, in the range of 8.0 to 9.0. This is indicative of active photosynthesis in which carbon dioxide is utilized from the bicarbonate alkalinity. Bottom sediments mixed with marl, of biological origin, were found extensively in the lake system.

During the periods of thermal stagnation in the deep lakes, deep water samples had pH values less than 8.0 generally when the bottom sediments were undergoing anaerobic decomposition (figures 100 and 104). The pH values of these waters appear to increase when the lakes experience fall overturn.

Alkalinity and Hardness

These factors are governed to a large extent by the geochemistry of the watershed of the lakes. The lakes of the Fox Chain are typical of Midwestern lakes, high in alkalinity and hardness. The interrelationship of alkalinity, pH, photosynthesis, and to a certain extent of hardness, was discussed in an earlier section, *"Water Quality of the Lakes and Streams."*

Alkalinity of the surface water samples in the Fox Chain of Lakes ranged from a low of 155 mg/l to a high of 278 mg/l. The mean alkalinity values for surface samples exhibit a spatial variation among the lakes. In the upper lakes of the Chain (Catherine, Channel, Marie, Bluff, and Petite), which are deeper than the rest, the mean alkalinity values are lower. Also in these lakes, the alkalinity in deep water samples are relatively higher than their corresponding surface sample values. This is true also in the case of Cedar Lake, which most of the time had the lowest of all the observed values. Differences between summer and winter alkalinity values have been used to define the trophic status of lakes in Europe (Vollenweider, 1968), oligotrophic lakes showing the least change. The temporal variations of alkalinity in the surface samples were moderate. Deep water sample variations, however, showed a noticeable peak-and-valley pattern.

The mean hardness values of surface water samples of the deep upper lakes are lower than those of the shallower lakes in the Chain. However, the differences between the surface, mid-depth, and deep values are insignificant. The hardness of Cedar Lake waters is considerably less than that of the lakes of the Fox Chain. Fluctuations of hardness values with time are much more pronounced when compared with the variations in alkalinity.

Nitrogen

Nitrogen is present in water either as dissolved organic nitrogen, or as inorganic nitrogen such as ammonium, nitrate or nitrite, or as elemental nitrogen. These various forms cannot be used to the same extent by different groups of aquatic plants and algae. Nitrogen is one of the principal elemental constituents of amino acids, peptides, proteins, urea, and other organic matters.

Vollenweider (1968) reports that in laboratory tests, the two inorganic forms of ammonia and nitrate are as a general rule used by planktonic algae to roughly the same extent. However, Wang et al. (1973) reported that during periods of maximum algal growth under laboratory conditions ammonium nitrogen was the source of nitrogen preferred by planktons. In the case of higher initial concentrations of ammonium salts, yields were noted to be lower than equivalent concentrations of nitrates (Vollenweider, 1968). This was attributed to the toxic effects of ammonium salts. The use of nitrogenous organic compounds has been noted by several investigators according to Hutchinson (1957). However, Vollenweider (1968) cautions that the direct use of organic nitrogen by planktons has not been definitely established, citing that not one of 12 amino acids tested with green algae and diatoms was a source of N when bacteria free cultures were used. But the amino acids were completely used up after a few days when the cultures were inoculated with a mixture of bacteria isolated from water. He has opined that in view of the fact that there is always bacterial fauna active in nature, the question of the use of organic nitrogen sources is of more interest to physiology than to ecology.

The mean nitrate concentrations in all the lakes, including Cedar Lake, and at all the depths examined were similar. The temporal variations were also moderate. With one or two exceptions the ranges of values for nitrate observed in all the lakes were comparable.

Kjeldahl nitrogen, as reported here, is the combination of ammonia nitrogen and organic nitrogen in the water samples. In the case of deep lakes, the Kjeldahl nitrogen concentrations at surface and mid-depth were comparable to those of shallow lake surface water samples. However, the concentrations in deep water samples were nearly two to three times those observed for the surface. Increase in ammonia concentrations in the deep water zones, because of anaerobic decomposition of settled organic matter, accounts for most of the increase in the Kjeldahl nitrogen concentrations reported here. The temporal variations of ammonia and Kjeldahl nitrogen shown in figures 98 through 104 indicate that these values are much higher in deeper zones of the lake. The ammonia and organic nitrogen concentrations in deep water samples increased during the period of lake stagnation but dropped significantly as soon as the fall overturn occurred. The values of ammonia and Kjeldahl nitrogen in the surface samples of Cedar Lake are comparable to those of the deep lakes, namely, Catherine, Channel, and Marie, and Pistakee Bay. However, the values for the deep water samples in Cedar Lake are significantly less when compared with the values for the deep lakes of the Chain. This is probably indicative of the extent of organic silt accumulation and subsequent decomposition in these lakes.

Silica

The element silicon is not found free in nature, but it occurs as silica (silicon dioxide) in the form of finely divided or colloidal matter. An abundance of silica in water, along with other necessary nutrients, favors the growth of diatoms. Lee (1974) postulates that when phosphate detergents are replaced by silicate based detergents, the silica budget for Lake Michigan could increase. This could result in reversing the current trend of the change from diatoms to green and blue-green algae associated with increasing phosphorus input. Since diatoms contain significant amounts of silica in the frustule, they can be expected to alter the silica concentrations in water bodies. Wang and Evans (1969) reported a high degree of inverse relationship between silica and diatom populations which constitute 60 to 90 percent of algal counts in the Illinois River.

The shallow lakes with the exception of Grass Lake showed higher silica concentrations than did the deeper lakes. The higher concentrations are probably associated with higher suspended solids and turbidity in these lakes. A high degree of association among these parameters has been reported in the past. The predominant algal species in Grass Lake were diatoms, constituting about 75 percent of the algal population. This could account for the lower values for silica in Grass Lake compared with the other shallow lakes in the system. The temporal variations of silica for deep waters, shown in figures 98 and 104 suggest that this nutrient is released from the mud-water interface to the overlying waters during the periods of lake stagnation.

Iron

Iron in amounts occurring in natural waters is not detrimental to biological life processes. Limits on iron concentrations are not based upon physiological considerations but rather on aesthetic and taste considerations. Iron in trace amounts is essential for nutrition. Certain bacteria, Crenothrix and Gallionella, and other iron bacteria utilize iron as a source of energy and store it in their protoplasm. The relatively insoluble ferric compounds associated with lake sediments are reduced to more readily soluble ferrous compounds under anaerobic conditions and released to the overlying waters.

The observations made for silica in the lakes also appear to hold true for iron. The surface sample iron concentrations in deep lakes are much lower than those in the shallow lakes. The higher concentration of iron in shallow lakes is probably associated with increased turbidity and suspended matter. Concentrations in Grass Lake are also high and comparable to other shallow lakes. Iron concentrations are significantly higher in the deep waters of the lakes. The mean and range of values observed for iron in the lakes are shown in table 44. The temporal variations in iron for three of the lakes investigated are shown in figures 98 through 104.

Chloride

Chlorides are found in practically all natural waters. They may be of mineral origin, or derived from human and animal wastes, industrial effluents, road salting during winter, or other sources. Chloride is a conservative element and is not directly involved in any of the physical or biological processes occurring in natural water bodies.

The mean and range of values for chloride are shown in table 44. Except for a few abrupt fluctuations in chloride values, temporal variations appear to be insignificant (figures 98 through 104). The chloride values observed in the Fox Chain of Lakes are comparable to the values found in Lake Evergreen (Kothandaraman and Evans, 1975) and in Lake Bloomington.

Sulfate

Sulfates occur naturally in water as a result of leaching gypsum, pyrite bearing strata, and other common minerals. Sulfates may also occur as the oxidized state of organic matter in the sulfur cycle, but they, in turn, may serve as an energy source for sulfur bacteria. Experiments indicate that waters containing less than 0.5 mg/l of sulfate will not support growth of algae (McKee and Wolf, 1963). Apart from the fact that sulfate exerts a cathartic effect on transient users, recommended limits on sulfate do not appear to be based on taste or physiological effects.

With the exception of Lake Catherine and Channel Lake, the mean sulfate concentrations of surface and mid-depth water samples are comparable and fall within the narrow range of 43 to 50 mg/l. Sulfate concentrations in these two lakes are much lower. Cedar Lake exhibited the lowest of all values observed. Sulfate concentrations in the deep water zones were much lower than in the surface or mid-depth water samples because of the anaerobic reduction of sulfate to hydrogen sulfide as evidenced by the strong septic odor noted during the sampling of the deep water zones. The temporal variations of sulfate, as shown in figures 98 through 104, are relatively small in surface and mid-depth waters. There is a reduction and greater fluctuation of sulfate in the deep waters.

Total Solids, Total Dissolved Solids, and Suspended Solids

Total solids, as presented here, include total dissolved solids and suspended solids. In natural waters, the dissolved solids consist mainly of carbonates, bicarbonates, sulfates, chlorides, phosphates and nitrates of calcium, magnesium, sodium, and potassium with traces of iron, manganese, and other substances. The constituent composition of these minerals are to a large extent dependent on the geochemistry of the area contributing to the surface or groundwater resource. The amount of suspended solids found in impounded waters is small compared with the amount found in streams because solids tend to settle to the bottom in lakes. However, this aspect is greatly modified in shallow lakes by wind and wave actions and by the type and intensity of use to which these lakes are subjected.

All salts in solution change the physical and chemical nature of the water and exert an osmotic pressure. Some have physiological as well as toxic effects. However, possible synergistic or antagonistic interreactions between mixed salts in solution may cause the effects of salts in combination to be different from those of salts occurring separately.

Greeson (1971) observed that high dissolved solids contents of Oneida Lake (New York) in 1967 and 1969 accompanied the high production of algae. Low dissolved solids content in 1968 accompanied lesser algal production. He concluded that these relationships indicate that the dissolved solids content is an important index of potential productivity conditions because no element, ion, or compound is likely to be a limiting factor on algal production when the dissolved solids content is high.

All of the lakes in the Fox Chain except Catherine and Channel exhibited similar values of solids concentrations. Catherine and Channel exhibited values next in order of magnitude. Cedar Lake had the lowest solids concentrations of all. The total dissolved solids and suspended solids concentrations in deep waters were relatively higher than the respective values for mid-depth and surface water samples. This is essentially due to the solubilization and release to overlying waters of settled decomposing organic matter. The increased suspended matter in the deep water samples is most likely caused by the dislodging of the sediments by the evolving gaseous by-products of anaerobic decomposition. The temporal variations of suspended, total dissolved, and total solids shown in figures 98 through 104 are typical of all the lakes investigated.

Algal Growth Potential (AGP) and Phosphorus

Because of the strong association of phosphorus and algal growth potential observed in this investigation, these two parameters are treated together. Moreover, phosphorus has been implicated as being primarily responsible for algal blooms in lakes. Gakstatter et al. (1975) reported that out of 623 lakes surveyed in the states east of the Rocky Mountains, under the National Eutrophication Survey Program, 67 percent were phosphorus limited.

Algal growth potential (AGP), as previously discussed, is a laboratory procedure carried out under standard and controlled conditions to determine the extent of algal growth sustained by the filtered water sample after being inoculated with a mixed algal culture obtained from a surface water source. Though it is impossible to duplicate in the laboratory all the conditions obtainable in nature, this test provides at least a relative estimate of the algal growth potentials of a water body over a period of time or between different bodies of water.

Phosphorus is an active element which does not occur free in nature. It is found in the form of phosphates in several minerals and it is a constituent of fertile soils, plants, protoplasm, and tissues and bones of animal life. It is an essential nutrient for plant and animal growth, and like nitrogen, it passes through cycles of photosynthesis and decomposition. Phosphorus plays a vital role in the energy transfers during cell metabolism. The most important form of phosphorus for plant nutrition is ionized phosphate.

The phosphorus compounds present in water are generally classified as orthophosphorus, polyphosphates, and organic phosphorus of both dissolved and particulate forms (Sullivan and Hullinger, 1969; Vollenweider, 1968). Only total phosphorus and dissolved orthophosphorus, which is the most readily available form for plant growth, were considered in this investigation.

Sawyer (1952), from his experimental work with Wisconsin lakes, concluded that aquatic blooms are likely to develop in lakes during summer months when concentrations of inorganic nitrogen and inorganic phosphorus are in excess of 0.3 mg/l and 0.01 mg/l, respectively. These critical levels for nitrogen and phosphorus concentrations have been accepted and widely quoted in scientific literature. Numerous published reports indicate that productivity is largely determined by two factors, i.e., phosphorus and nitrogen. Phosphorus is predominant over nitrogen as a limiting factor. Vollenweider (1968) concluded, after extensive analyses of data pertaining to the lakes of central Europe, that the critical levels for phosphorus and nitrogen concentrations suggested by Sawyer were also valid for those lakes. He further observed that phosphorus was the more critical of the two nutrients.

The mean and range of values for phosphorus concentrations and AGP in the Fox Chain of Lakes and Cedar Lake are shown in table 44. Temporal variations in these parameters for three of the Chain lakes typical of deep, shallow, and nutrient deficient lakes are shown in figures 98 through 104. From the data presented, it is seen that the mean dissolved orthophosphorus concentrations in Cedar Lake are much less than the values observed in the Fox Chain of Lakes. Characteristically, the dissolved orthophosphorus concentrations in the deep water zones were found to be several orders of magnitude higher than the values determined for surface and mid-depth water samples in these lakes. This was true in the case of Cedar Lake also. However, a comparison of the mean dissolved orthophosphorus concentrations in the deep water samples indicates that concentrations in the Fox Chain of Lakes are 10 to 25 times the value observed for Cedar Lake. The mean orthophosphorus concentrations in the surface and mid-depth samples were 0.00 and 0.01 mg/l, respectively.

The orthophosphorus concentrations in the surface and mid-depth samples in the Chain lakes were higher than the critical levels of phosphorus concentration suggested by Sawyer (1952). The mean orthophosphorus concentrations observed in deep zones of the Fox Lakes were about 40 to 75 times higher than the critical level.

The temporal variations in phosphorus concentrations and the AGP values shown in figures 98 through 104 suggest a strong association between the AGP of the lake water samples and the phosphorus concentrations. The AGP values for Cedar Lake were very low, and the lake showed the lowest algal densities of all the lakes investigated. The AGP values determined with deep water samples from the Chain lakes showed consistently high values. The phosphorus concentrations in these samples were very high compared with surface and mid-depth sample values.

Considering the phosphorus concentrations, AGP values, and the algal densities observed in the Fox Chain of Lakes and Cedar Lake, it can be concluded that phosphorus is the key element in the eutrophic condition of the Fox lakes. The nitrate-nitrogen concentrations in all the lakes including the deep water zones were comparable. Ammonia concentrations were higher in the hypolimnetic zones of the deep lakes. Phosphorus appears to be the most critical of the two nutrients.

Biological Characteristics — Phytoplankton

The frequency and locations for phytoplankton collections were the same as those previously described for chemical analyses. In the deeper bodies of water (Lake Catherine, Channel Lake, Lake Marie (west), Pistakee Bay, and Cedar Lake) samples were collected at the surface, mid-depth, and about 1 foot from the bottom. At the other 10 locations (see table 45) samples were collected at the water surface. Twenty-five samples were collected per trip for phytoplankton enumeration and identification.

Table 45. Sampling Station Location and Depth

Station number	*Location*	*Average depth (ft)*
1	Lake Catherine	39
2	Channel Lake	35
3	Lake Marie (east)	22
4	Lake Marie (west)	31
5	Bluff Lake	27
6	Petite Lake	17
7	Fox Lake (Stanton Bay)	4.4
8	Fox Lake (main)	7.2
9	Fox Lake (Mineola Bay)	12
10	Nippersink Lake	4.9
11	Pistakee Lake	5.4
12	Pistakee Bay	30
13	Grass Lake (south)	3.7
14	Grass Lake (north)	3.3
15	Cedar Lake	42

Methods

Surface samples were collected with a bucket. Samples below the surface were collected with a Kemmerer sampler. Experience has shown that collection in this manner, in contrast to the use of a plankton net, produces a better representation of the naturally dispersed aquatic organisms. A bias in organism shape and size is avoided. A water sample volume of 380 ml was put in a small-mouth glass bottle, preserved with formalin, and stored at room temperature until examined.

Examinations were preformed within a week, at which time the sample was thoroughly mixed and a 1-ml aliquot pipetted into a Sedgwick-Rafter cell. An inverted phase contrast-microscope equipped with 10X eyepieces, 20X objective, and a Whipple disc was used for identification and counting purposes. Five short strips (about 280 fields) were counted. Dilution or concentration was not required.

Phytoplankton were identified to species by employing several keys (Palmer, 1959; Patrick and Reimer, 1966; Prescott, 1962, 1970; Smith, 1950; and Tiffany and Britton, 1951). They were classified in five main groups, i.e., blue-greens, greens, diatoms, flagellates, and desmids.

Types of Algae

Algae are classified, in part, according to their color. The blue-green algae are so named because, in addition to chlorophylls, they contain phycocyanin which gives them a blue to dark green tint. A red pigment is sometimes also present. They contain about 1500 species and may be autotrophic or heterotrophic. Most blue-green algae grow in nonfilamentous colonies or in branched or unbranched filaments. They are widely distributed and occur in varied habitats, but when they occur in massive numbers (a bloom), they are found at the water surface. They are more frequently found in lakes and ponds than in the running waters of streams.

The green algae usually contain one major group of pigments, the chlorophylls, and most are autotrophic. This group includes about 700 species. Although a number live in salt water, the group as a whole is more characteristic of fresh water. They may be either free-floating or attached and are usually either single cells or filamentous colonies. If numerous, they display a green cast to the water.

Somewhat between blue-green and green algae is a group known as diatoms. Diatoms are characterized by the presence of silica in their cell walls, and by the presence of green, yellow, or brown pigment associated with the chlorophylls depending on the stage in their life cycle. They vary in color from brown to green. There are about 16,000 species. The cell wall is composed of two halves (valves) one overlapping the other like the top and bottom of a pill box. Generally the cell is oblong to circular, although there can be a variation in shape.

In several divisions of algae, including some greens, there are species that are unicellular and equipped with flagella. Flagella are a whiplike organ that make mobility possible. The organisms so equipped are flagellates. They are motile and may be either autotrophic or heterotrophic. The cells range from spherical to ovoid depending on the species. They are most commonly found in organically enriched waters.

Desmids belong to the subgroup *Desmidiaceae* of the green algae. They are characterized by cells of distinctive shapes, one half of which corresponds in shape, size, and contents to the other half. In many desmids the two 'semicells' are connected by a short narrow tube (isthmus). They are numerous species of desmids and they are usually associated with lakes and ponds.

For enumeration, in general, blue-green algae were counted by the number of trichomes. Green algae were counted by individual cells except *Actinastrum, Coelastrum,* and *Pediastrum,* which were recorded by each colony observed. *Scenedesmus* was counted by each cell packet. Diatoms were counted as one organism regardless of their grouping or connections. For instance, a unit was considered to be a filament of *Melosira,* a cluster of *Asterionella* or *Fragilaria* cells, or single cells of *Stephanodiscus* or *Surirella.* For flagellates, a colony of *Dinobryon* or a single cell of *Ceratium* was recorded as a unit.

Algal Composition

During the 5-month collection period, 64 algal species were recovered from 414 samples. The species included 6 blue-green algae, 18 green algae, 29 diatoms, 9 flagellates,

Table 46. Occurrence and Type of Algae at Sampling Locations

(Percentage of time present at sampling locations)

Algaltype	Station number* 1A	1B	1C	2A	2B	2C	3	4A	4B	4C	5	6	7	8	9	10	11	12A	12B	12C	13	14	15A	15B	15C	Number of stations
Blue-green algae																										
Anabaena spiroides	47	31	13	32	19		11		6		11	32	16	5	5	5		5	6		16	11	14	14		19
Anacystis cyanea	11																									1
A.flos aquae	26	13	6	11	19	6	37	31	31	13	37	58	37	37	26	16	32	31								21
A.therinalis	11	13	13	25	31	19	26	27	38	38	37	37	26	21	16	21	26	19								20
Aphanizomenon flos.aquae	95	81	56	74	88	63	68	58	50	35	58	68	53	47	58	47	47			63	47	57	43	29		24
Oscillatoria putrida																						14				1
Green algae																										
Actinastrum bantzscbii								5							5	5	11				21	21				6
Ankistrodesmus falcatus													5		5	11		5	6		16	11	29	14	14	10
A.convolutus							5				5										5					1
Cblorella ellipsoidea							5																			2
C.pyrenoidosa																										1
Coelastrum microporum	11	6									5	5	11	5	5			11			5	5				10
Crucigenia rectangularis											5		5								16					3
Oocystis borgei	16	13	6	11			21	26	6		11	16	32	11	11	11	11	11	6	6		16				18
Pediastrum duplex	11						16	11	6		16	21	32	11	5	26	21	16	13		11	26			14	16
P.simplex			13					11					5	11	5		11	5			11	11				9
Scenedesmus carinatus														5		11		5			5					4
S.dimorphus	5						16	5		6	5	5	11	16	5	16	16		6	13	74	47				15
S.quadricauda													5													1
Sphaerocystis scbroeteri					6							5														2
Tetraedron limneticum		13	6	11	6	6	21	32	19	6	21	32	16	11	11	11	5	5	6			5				19
T.sp.	5																									1
Ulotbrix variabilis	16	6			6	6	11	11	13		16	11	37	37	32	32	42	21	21	38	26	21		14	14	21
U.zonata														5												1
Diatoms																										
Asterionella formosa							11	11	6		11	11		5		5	5									8
Caloneis ampbisbaena		6	6							13	5								6	6	5					7
Cyclotella acellata																	5	5								2
C.glomerata																		5			5	5	14	14	14	6
C.meneghiniana	16	25	31	32	13	13	42	63	44	44	37	37	37	79	53	89	68	37	31	31	95	95				22
C.michiganiana	5						5		6												5				14	5
Cymbella mixicanum						6										5										2
Diploncis interrttpta													5													1
D.smitbii			6																							1
Gomphonema olivaceum																5					5					2
Gyrosigma kutzingii														5			5									2
G.wormleyi			6																							1
Melosira ambigua															5	5										2
M.granulata	11	19	13	16	13	13	26	47	25	31	26	47	63	74	47	58	74	74	81	50	58	37				22
Nauicula gastrum	5					6															16	5				4
N.gracilis	5							5	1			5	5													5
N.odiosa																	11				5	5	14	14	14	6
N.pcregnina														5												1
N.sp.													5													1
Neidiul.dubium																5										1
N.productum																	5									1
Nitzscbia acictdaris														5												1
N.angustata																		5					14			2
N.dentccula																					5					1
Stephanodiscus niagara'			6				11	5					11		16	5	5	5			5	5				10
Sutirella ovata								5	6									5	6		5					5
Synedra acus						6		11	6								5			6		11				6
S.tdna		6	6									5		5		5				6	5					7
Tabellaria fenestrata			6	2																	5		57	43	43	6
Flagcllates																										
Ceratlttm birttndinella	63	75	44	74	63	38	37	32	25	13	21	16	5	5	5	5					5	5	14			19
Cblamydonloas sp.				5																						1
Dinobryon sertularia																							14			1
Eudorina elegans																		5								1
Eugelna gracilis								5					5	5		5			6							5
E.oxyuris														5								5				2
E.viridis	5			5			5	11		6		11			11	11	5			6	37	21		14		13
Gymnodinium sp.	5													5			11	16	6							5
Phacus pleuroncctcs		6											11	5	11		5				26	5				7
Desmlds																										
Cosmariun ponanense								5																		1
Stattrastrttm cornutum														5												1
Number of species	19	14	16	12	10	11	18	21	16	10	17	18	23	26	20	24	22	21	16	11	26	25	10	8	8	
Number of samplcs	19	16	16	19	16	16	19	19	16	15	19	19	19	19	19	19	19	19	16	16	19	19	7	7	7	

*A = surface; B = mid-depth; C = Jeep; Unlettered station numbers are surface samples

and 2 desmids. The occurrences and types of algae observed are listed in table 46. The number of species per sampling location, excluding Cedar Lake, ranged from 10 at Channel Lake mid-depth and Lake Marie (west) deep stations to 26 at Fox Lake (main lake) and at Grass Lake (southside).

As shown in table 46,the occurrence of desmids was negligible. Blue-green algae were predominant in most of the lakes. The most frequently occurring species was *Aphanizomenon flos-aquae*, noted in 24 of the 25 sampling stations. It did not occur in the deep waters of Pistakee Bay. *Anacystis* and *Anabaena* occurred frequently at some locations.

For the two locations sampled on Grass Lake, diatoms were predominant. The most common were *Cyclotella meneghiniana and Melosira granulata.* These two species are commonly recovered in Illinois streams (Lin et al., 1972, 1973, and 1975). The green algae, Ulothrix variabilis, Tetraedron limneticum, and Oocystis borgei occurred at most of the stations. Scenedesmus dimorphus was one of the most common algae in Grass Lake. The only flagellate of importance was Ceratium hirundinella, and its bloom concentration was confined to Lake Catherine and Channel Lake.

The recovery rates of the 17 most predominant algae at 25 sampling locations are depicted on figure 105. This figure is a partial summary of data tabulated in table 46.

Algal Density

The average composition for each algal type at each station, based on algal density, is shown in table 47. Also included is the maximum percentage composition of the four major groups for each sampling location. As shown, the blue-green algae were the most common in the deeper northernmost lakes (Lake Catherine, Channel Lake, Lake Marie, Bluff Lake, and Petite Lake). Several samples taken from these stations consisted solely of blue-green algae.

Table 47. Percentage Composition of Algal Types Based upon Algal Density

		Average				Maximum			
	*Sampling station **	*Blue-green*	*Green*	*Diatom*	*Flagellate*	*Blue-green*	*Green*	*Diatom*	*Flagellate*
1A	Lake Catherine	67.0	8.6	8.9	15.5	100	77.1	47.8	64.8
1B	Lake Catherine	64.2	3.8	11.0	21.0	100	22.7	45.5	86.9
1C	Lake Catherine	55.2	8.1	16.3	20.4	100	63.6	81.3	100
2A	Channel Lake	59.3	4.4	15.7	20.6	100	67.4	89.7	71.0
2B	Channel Lake	70.9	4.1	8.4	16.6	100	64.4	57.1	78.9
2C	Channel Lake	62.2	5.8	13.9	18.1	100	80.0	100	100
3	Lake Marie (east)	55.3	10.6	29.2	4.9	100	46.1	100	32.1
4A	Lake Marie (west)	56.6	7.2	29.7	6.5	100	35.5	100	62.5
4B	Lake Marie (west)	57.0	7.8	33.5	1.7	100	38.7	100	12.8
4C	Lake Marie (west)	59.4	1.8	36.7	2.1	100	15.2	100	25.0
5	Bluff Lake	66.5	12.2	20.0	1.3	100	67.6	68.2	9.3
6	Petite Lake	71.8	10.7	16.8	0.7	100	58.3	50.0	9.7
7	Fox Lake (Stanton Bay)	42.5	20.5	36.0	0.7	100	92.9	92.8	4.2
8	Fox Lake (main)	31.9	14.4	52.9	0.8	89.9	94.2	97.8	9.0
9	Fox Lake (Mineola Bay)	40.8	22.5	35.4	1.3	96.3	100	100	8.8
10	Nippersink Lake	32.6	16.5	50.6	0.3	94.6	88.9	96.6	4.7
11	Pistakee Lake	24.1	18.2	56.5	1.2	89.0	93.5	100	12.5
12A	Pistakee Bay	43.9	13.9	41.1	1.1	100	90.5	100	16.7
12B	Pistakee Bay	42.4	13.4	43.4	0.8	100	92.6	100	10.0
12C	Pistakee Bay	29.4	20.8	48.2	1.6	100	100	100	25.0
13	Grass Lake (south)	4.9	16.0	76.6	2.5	23.8	59.2	97.3	15.4
14	Grass Lake (north)	10.5	18.0	70.3	1.2	78.7	80.9	98.5	8.9
15A	Cedar Lake	28.9	11.4	56.6	3.1	69.2	79.9	100	20.0
15B	Cedar Lake	37.4	17.1	43.1	2.4	100	100	100	16.7
15C	Cedar Lake	16.1	25.0	58.9	0	87.5	100	100	

*A = surface; B = mid-depth; and C = deep; Unlettered station numbers are surface samples

In contrast, diatoms dominated the southernmost lakes, particularly Grass Lake. Flagellates averaged about 20 percent of the total for Lake Catherine and Channel Lake, but were not significant in other lakes. Except for a bloom of *Ulothrix variabilis* on

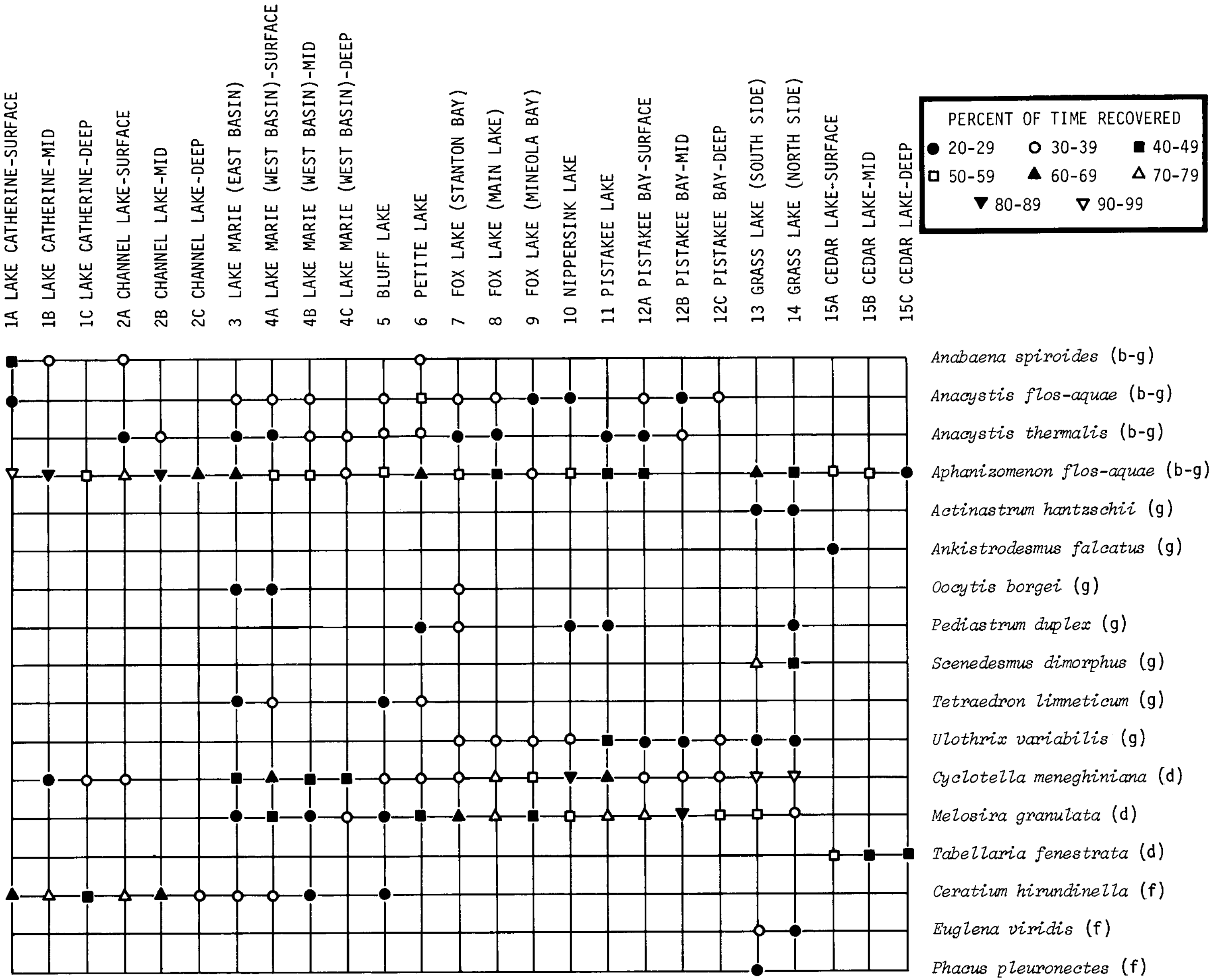

Figure 105. Occurrence of abundant algae in Fox Chain of Lakes

Pistakee Bay and Pistakee Lake on October 6, green algae were not an important factor in algal mass. The highest average percentage of green algae was 22.5 percent for Fox Lake at Mineola Bay.

The algal density data expressed as cell counts per milliliter (cts/ml), along with some other statistical results were calculated for each station. [These data are available upon request from the Illinois State Water Survey.] The observed total algal densities ranged from a low of 22 cts/ml for several locations to a high of 14,000 cts/ml in Fox Lake at Mineola Bay. Other stations having singularly high algal count were Channel Lake (surface) 13,000 cts/ml, Grass Lake (south) 11,000 cts/ml, and Fox Lake (main lake) 10,000 cts/ml.

Most algal counts for surface and mid-depth samples were in the range of 100 to 500 cts/ml except for Nippersink and Grass Lakes. In Grass Lake approximately 60 percent of the algal counts were more than 2000 cts/ml. As expected, the algal densities of the deep samples and of Cedar Lake were low, mostly below 100 cts/ml.

An examination of the algal density data for each station showed them to be generally distributed in a log-normal pattern. Therefore the central tendency and dispersion of the data were expressed in geometric terms. Excluding data of Cedar Lake, the geometric mean varied from a low of 65 cts/ml for Channel Lake (deep) to a high of 2200 cts/ml for Grass Lake (south). The geometric mean for Grass Lake (north) was also high (2000 cts/ml).

Diversity Index

There have been many methods suggested for defining the structure of a biological community. The most widely used procedure is the diversity index and the one most commonly used is Shannon's (1949) index. For this report the index for each sampling location on each day of collection was determined as follows (Hutchinson, 1967):

$$D = -\sum_{i=1}^{m} p_i \log_2 p_i$$

where $p_i = N_i/N_s$ and is the probability of the occurrence of the ith genera, N_i is the density of the ith genera, N_s is the total algal density of the sample, and m is the number of species per sample. For convenience $\log_2 p_i$ may be expressed as 1.44 ln p_i. The index, D, has a minimum value (zero) when m equals 1 and a maximum value when $m = N_s$.

The greatest species index developed was 2.62 for Grass Lake (south). Nine species of algae were observed in the sample. Diversity indexes of zero occurred for some samples. These were unispecies occurrences which were common in mid-depth and deep stations. The mean index ranged from 0.75 for Pistakee Bay (deep) to 1.32 for Petite and Pistakee Lakes excluding the control lake (Cedar Lake). In general, the index decreased with depth.

An attempt was made to correlate total algal density and species diversity index. Three models, i.e., linear, semi-log, and log-log, were evaluated for each sampling station. The value of 0.01 was used to replace zero in diversity index. Correlated coefficients are shown in table 48. Relatively high coefficients were found for only a few stations (1B, 1C, 2A, 2B, 10, and 14).

In a study of north-central Florida lakes, Brezonik et al. (1969) reported that species diversity is positively correlated (semi-log) with total plankton counts for Anderson-Cue Lake (an oligotrophic lake). This was not the case for lakes of the Fox Chain or for Cedar Lake.

Population Dynamics

This discussion relies mainly on the visual evidence of population succession depicted in figure 106a-1. As mentioned earlier the blue-green algae *Aphanizomenon flos-aquae* were the most dominant species in the waters of the Fox Chain. Their predominance is very well demonstrated in figure 106a-e. These figures are for the upper northern tier of lakes commencing with Lake Catherine and Channel Lake on down to and including Petite Lake.

In figure 106a and b (Lake Catherine and Channel Lake) there were 4 to 5 recorded major pulses of *Aphanizomenon flos-aquae.* These blooms commenced about June 6 and extended until about August 11. During this time the water temperature ranged from 20 to 26°C. Peculiar to these two lakes were two pulses of the flagellate *Ceratium hirundinella.* One of these pulses was recorded on July 21 at a water temperature of about 28°C and the other occurred on September 15 at a water temperature of about 18°C. This suggests the water temperature range for this flagellate is not a critical one. On these two lakes, also, a pulse or two of the blue-green *Anabaena spiroids* occurred. These occasions were the only ones where blooms of this particular algae were noted.

Table 48. Correlation Coefficient for Total Algal Density (×) versus Diversity Index (Y)

Sampling station	*Sample number*	*Linear* $Y=a+bX$	*Semi-log* $Y=a\ exp\ (bX)$	*Log-log* $Y=aX^b$
1A	19	-0.55	-0.67	-0.56
1B	16	-0.27	-0.11	0.99
1C	16	0.33	0.94	0.18
2A	19	-0.56	-0.76	-0.50
2B	16	-0.57	-0.77	-0.57
2C	16	0.27	0.34	0.38
3	19	-0.10	-0.12	-0.13
4A	19	-0.14	-0.51	-0.75
4B	16	0.13	0.11	0.13
4C	15	0.11	0.33	0.42
5	19	0.13	0.13	0.18
6	19	0.30	-0.27	-0.32
7	19	-0.13	-0.12	-0.21
8	19	-0.52	-0.44	0.38
9	19	-0.30	-0.57	0.21
10	19	-0.10	-0.73	-0.11
11	19	-0.55	-0.57	-0.59
12A	19	0.14	0.11	0.09
12B	16	-0.47	-0.53	-0.17
12C	16	-0.34	-0.56	-0.38
13	19	0.42	-0.66	-0.23
14	18	0.16	0.22	-0.85
15A	7	0.63	0.39	0.47
15B	7	-0.33	-0.37	-0.29
15C	7	0.30	0.29	0.32

On Lake Marie (west), Bluff, and Petite Lakes (figure 106c-e) *Anacystis flos-aquae* pulses occurred from about mid-June until about the end of August. Water temperatures ranged from 20 to 25°C. These pulses were generally coincidental with larger *Aphanizomenon flos-aquae* blooms.

Differing however from Lake Catherine and Channel Lake, the three lakes for which data are depicted in figure 106 c-e supported another blue-green algae on September 15, *Anacystis thermalis.* The water temperature was about 18°C. According to Prescott (1962) this algae is usually found in soft or semisoft waters. This is not the case in the Fox Chain. Of the northern tier lakes only Lake Marie (west) supported a diatom population of any consequence. Minor populations of *Cyclotella meneghiniana* occurred as shown in figure 106c. This is probably due to the channel connection between Lake Marie and the northern portion of Grass Lake.

The main part of Fox Lake (figure 106f) did support a substantial population of *Aphanizomenon flos-aquae* during June. Thereafter the diatoms *Melosira granulata* and *Cyclotella meneghiniana* prevailed. Mineola Bay of Fox Lake supported three major *Aphanizomenon flos-aquae* blooms from about June 1 to August 1 (figure 106g). *Cyclotella meneghiniana* was prevalent thereafter. The algal population of Mineola Bay was similar to that maintained by Channel and Catherine Lakes in terms of blue-greens. Nippersink Lake (figure 106h) was more like the main part of Fox Lake (figure 106f) in that *Aphanizomenon*

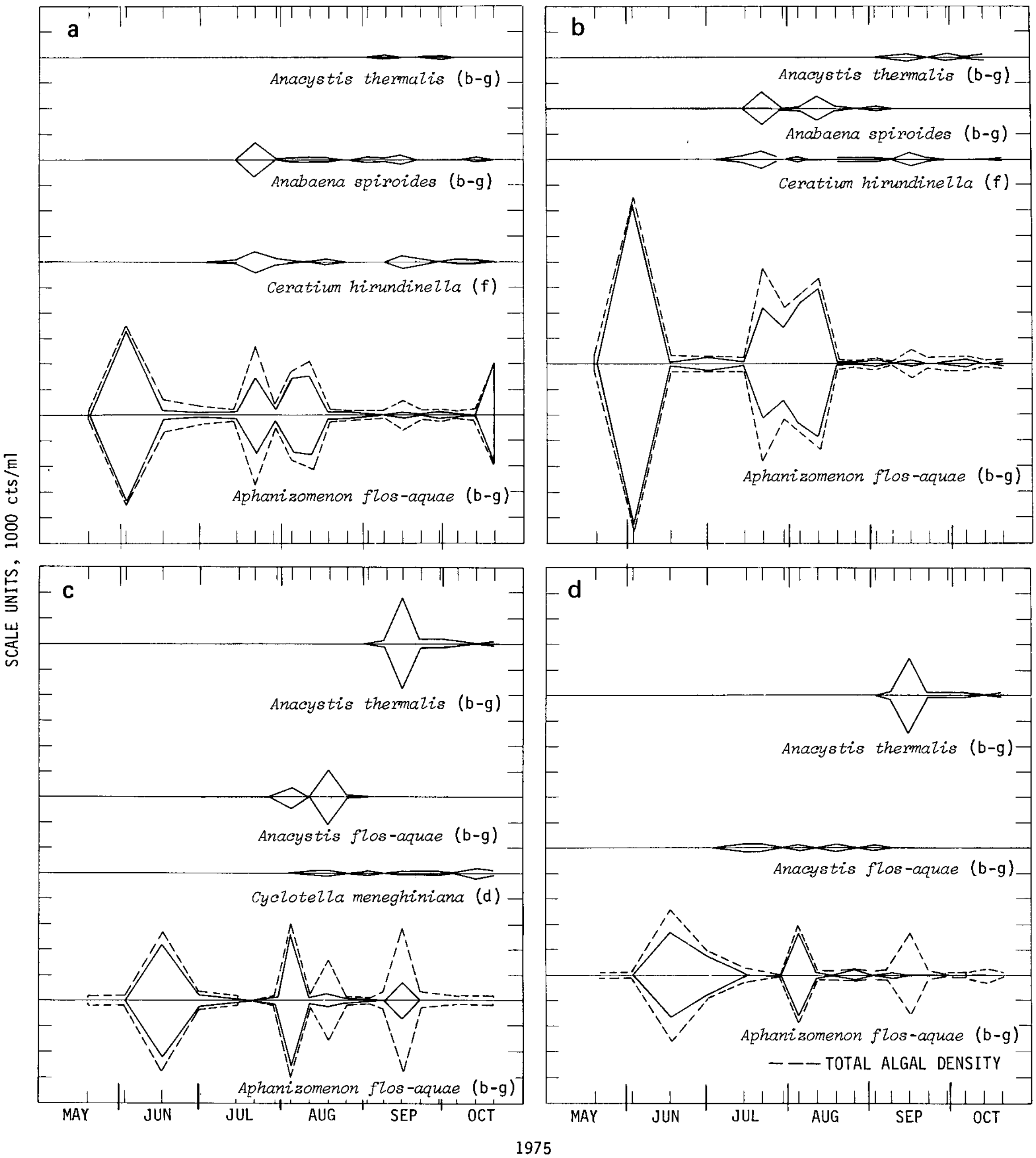

Figure 106a-d. Predominant algae at water surface in Lake Catherine (a), Channel Lake (b), Lake Marie-west (c), and Bluff Lake (d)

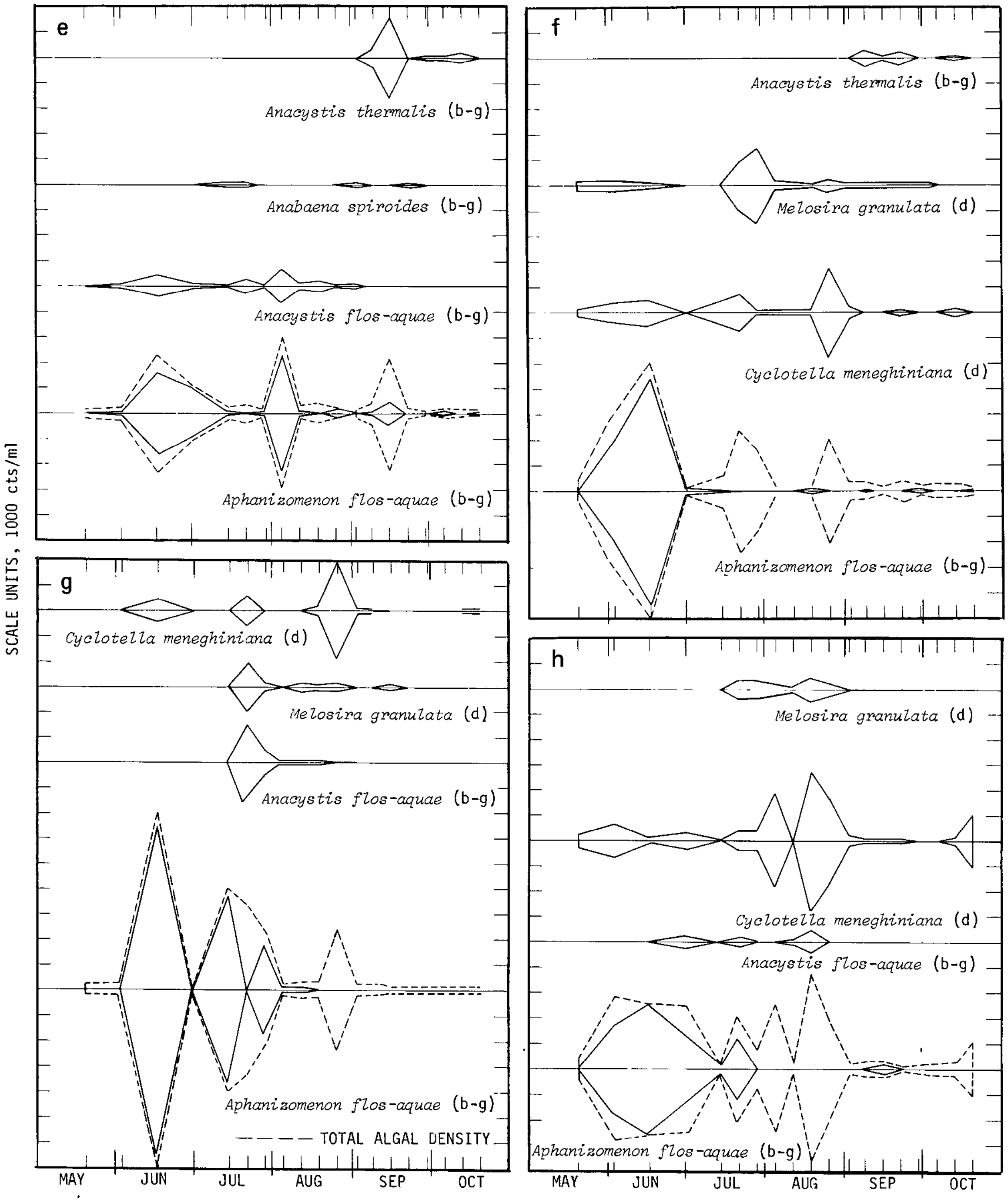

Figure 106e-h. Predominant algae at water surface in Petite Lake (e), Fox Lake-main body (f), Fox Lake–Mineola Bay (g), and Nippersink Lake (h)

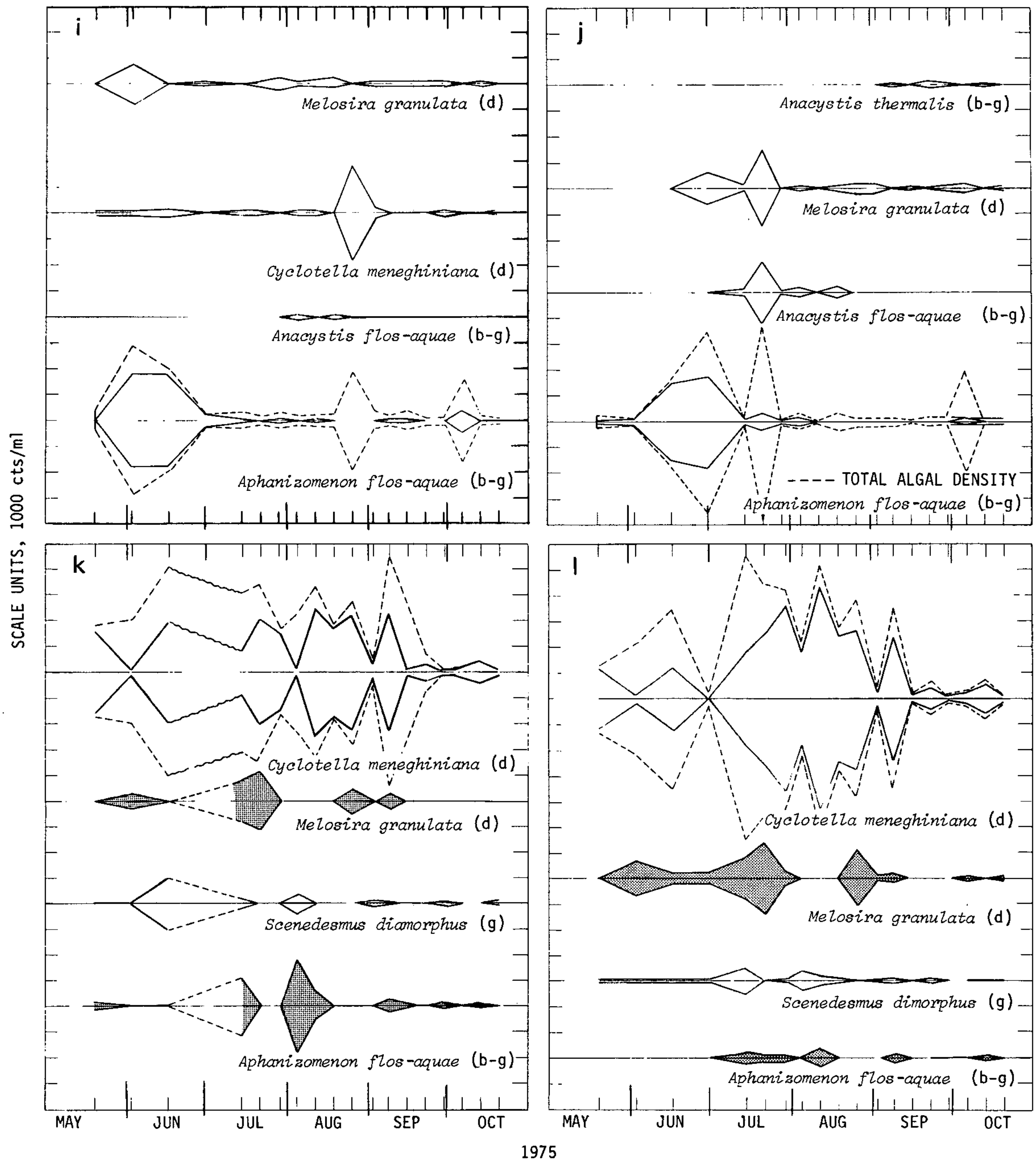

Figure 106i-l. Predominant algae at water surface in Pistakee Lake (i), Pistakee Bay (j), Grass Lake-north (k), and Grass Lake-south (l)

flos-aquae was more limited in June and *Cyclotella meneghiniana* was the prevailing diatom in August.

On Pistakee Lake blue-green algae were limited to June (figure 106i) with *Melosira granulata* and *Cyclotella meneghiniana* the important diatoms. Figure 106i suggests also that *Melosira granulata* is not incompatible with *Aphanizomenon flos-aquae* but *Cyclotella meneghiniana* may be. This is not clear-cut in figure 106f-h but the tendency leads toward such a conclusion. Pistakee Bay (figure 106j) supported a joint bloom of *Melosira granulata* and *Anacystis flos-aquae* on July 21 suggesting the environmental conditions for these differing types of organisms are similar.

It is obvious from figure 106k and 1 that the waters of Grass Lake are mainly diatom producers. *Aphanizomenon flos-aquae* did occur significantly during late July and early August, but *Cyclotella meneghiniana* was the predominant organism for June through October.

As mentioned earlier Cedar Lake has been considered a mesotrophic lake. Low algal densities as well as fewer species were recovered there. In fact, only two meaningful species were detected. The diatom *Tabellaria fenestrata was* recovered more than any other organism but was never found in significant numbers in any of the Chain lakes. The maximum at Cedar occurred at the water surface, at 270 cts/ml, on September 8. *Cyclotella meneghiniana* and *Melosira granulata,* the predominant diatoms in the Chain, were not recovered in significant numbers from Cedar Lake. The other important algae was *Aphanizomenon flos-aquae,* however, the density of this blue-green generally did not exceed 100 cts/ml.

Summary

Sixty-four algal species were recovered from 414 samples. The number of species per sampling location varied from 10 at Channel Lake and Lake Marie to 26 on Fox Lake (main) and Grass Lake. Blue-green algae were predominant and occurred at 24 of the 25 sampling stations. They consisted mainly of *Aphanizomenon flos-aquae.* The only flagellate of importance was *Ceratium hirundinella,* and significant concentrations of this organism were limited to Channel Lake and Lake Catherine. The only green algae bloom was created by *Ulothrix variabilis* on the waters of Pistakee Lake. Diatoms were quite significant on the shallower water bodies.

Algal densities ranged from about 25 to 14,000 cts/ml. The highest count occurred on Mineola Bay. Other high counts observed were 13,000/ml on Channel Lake; 11,000/ml on Grass Lake (south), and 10,000/ml on Fox Lake (main). In Grass Lake 60 percent of the collections had algal densities in excess of 2000 cts/ml. Most of these were the diatom *Cyclotella meneghiniana.*

The highest diversity index was developed from a collection on Grass Lake (south). Nine species of algae were in the collection. Mean diversity indexes ranged from 0.75 for Pistakee Bay to 1.32 for Petite Lake. A meaningful correlation between algal densities and diversity indexes did not exist.

The predominance of algal types, i.e., blue-green and diatoms, is related to the physical characteristics of the lakes. The lakes in the northernmost tier are relatively deep (20-40 feet), and their lesser expanse of areal water surface makes them less exposed to wind action than other lakes in the system. These lakes support similar algal types, mainly blue-greens. Likewise, other protected water bodies such as the bay areas of Pistakee Lake and Fox Lake mainly support blue-green algae.

The open expanse of the waters of Fox, Pistakee, and Nippersink Lakes, after an early June bloom of *Aphanizomenon flos-aquae,* mainly supported a diatom population. The shallow and expansive Grass Lake was principally a habitat for diatoms.

The importance of algal types and species for each body of water is shown in table 49. In the rating system, based upon concentration and frequency of occurrence, the lower the numerical rating, the greater its prominence on a corresponding body of water.

Table 49. Rating of the Most Important Algae for Each Water Body

	Blue-green				*Green*		*Diatom*			*Flagellate*
Location	Aphanizomenon flos-aquae	Anabaena spiroides	Anacystis flos-aquae	Anacystis thermalis	Scenedesmus dimorphus	Ulothrix variabilis	Cyclotella meneghiniana	Melosira granulata	Tabellaria fenestrata	Ceratium hirundinella
Lake Catherine	1	2								3
Channel Lake	1	2								3
Lake Marie (east)	1		2	3			4			
Lake Marie (west)	1		2	3			4	5		
Bluff Lake	1		3	2			4	5		
Petite Lake	1		2	3			5	4		
Fox Lake (Stanton Bay)	1		3	4			5	2		
Fox Lake (main)	3		4			5	1	2		
Fox Lake (Mineola Bay)	1		4			5	2	3		
Nippersink Lake	2		4			5	1	3		
Pistakee Lake (main)	3					4	1	2		
Pistakee Bay	2		4				3	1		
Grass Lake (south)	4				3		1	2		
Grass Lake (north)	2				2		1	4		
Cedar Lake	2								1	

Biological Characteristics — Benthic Organisms

Fourteen stations on the Fox Chain of Lakes were sampled on three separate dates (June 16, July 21, and September 8) in 1975. Three grabs with an Ekman dredge (6 x 6 inches) were taken at each station. The samples were then washed in a 30-mesh screen bucket, placed in quart jars, and preserved with 25 ml of formalin. In the laboratory, the samples were again washed in a 30-mesh sieve. The organisms were then picked from the bottom detritus, identified, counted, and preserved in 70 percent ethyl alcohol.

The benthos found at the sampling stations can be characterized as pollution tolerant organisms. The dipteran larvae *Chaoborus* is the predominant organism in most of the lakes. The *Chaoborus* (phantom midges) has air sacs which permit it to visit the surface waters to feed and renew its oxygen supplies at night. During the day, this organism burrows in the sediments and may survive there in great numbers, even in anaerobic conditions. *Chironomous* and *Procladius,* the other two principal components of the benthos, belong to the dipteran family Chironomidae (true midges). These larvae remain in the bottom sediments until just before emergence. Both of these genera have a hemoglobin-like blood pigment and special gills to extract sufficient oxygen from nearly anoxic waters. The predominence of the genera *Chironomous* and *Chaoborus* in the benthos has been associated with an advanced stage of eutrophication.

Community structure can be used to determine the health or balance of a given habitat. Community structure or diversity can be determined by relating the number of species and their populations. A diverse community has many species with relatively few individuals in each one. A simplified community, indicating harsh environmental conditions, has few species with high populations in each one. The community structure of the lakes sampled in this study exemplify the latter condition. Low dissolved oxygen and unstable benthic substrates contribute heavily to the harsh benthic environment.

Table 50. Benthic Organisms per Square Meter in the Fox Chain of Lakes

	Chaoborus				Chironomus				Procladius			
Lake	*6/16*	*7/21*	*9/8*	*Average*	*6/16*	*7/21*	*9/8*	*Average*	*6/16*	*7/21*	*9/8*	*Average*
Catherine	58	43	87	66	14			5				0
Channel		189	131	107				0	14			5
Marie (east)	14	29	218	87	14			5				0
Marie (west)		14	14	9	14			5				0
Bluff	43		116	53				0				0
Petite	29	72	594	232	102	14		39	131		43	58
Fox (Stanton Bay)	130	58		63	73	14		29	43	174		72
Fox (main)		43		14	87	14		34	14			5
Fox (Mineola Bay)			43	14	87	14	14	38	101	72	14	62
Nippersink				0				0				0
Pistakee (main)				0	275			92				0
Pistakee Bay			159	53				0				0
Grass (south)				0		44	29	24		29	87	39
Grass (north)				0			14	5				0

Chaoborus was found at all the sampling stations (see table 50) except Nippersink, Pistakee (main), and Grass (south and north). The absence of *Chaoborus* from these locations can be explained by the fact that these stations are located on the main dredged channel through the lake system. *Chaoborus* are not found in rivers or lakes with currents or high benthic turbulence.

Low dissolved oxygen concentrations limit the *Chironomous* and *Procladius* populations in the deeper lakes, i.e., Catherine, Channel, Marie (east and west) and Bluff. The intermediate depth, wind-mixed lakes (Petite and three stations on Fox) maintain a higher level of dissolved oxygen and exhibit a more constant and diverse community. Lake Nippersink showed the highest level of dissolved oxygen of any station and yet no benthic organisms were found there. It is a small, shallow lake and the sampling station was located on a heavily traveled, dredged channel. It is believed that the motor boat traffic churns the bottom sediments prohibiting the establishment of a benthic community. Pistakee (main) and Grass (north and south) show a reduced Chironomidae population. These stations have high dissolved oxygen concentrations, yet are still on dredged channels. They are able to maintain a limited benthic population mainly because of reduced boat traffic at these stations. The Pistakee Bay station is over a deep hole in an otherwise shallow lake. Its benthos are similar in character to the benthos of the deep lakes.

NUTRIENT BUDGET

Though nitrogen and phosphorus are not the only nutrients required for algal growth, they are generally considered to be the two main nutrients involved in the lake eutrophication process. In spite of the controversy over the role of carbon as a limiting nutrient, a vast majority of the researchers regard phosphorus as the most frequently limiting nutrient in lakes (Bartsch, 1972; Gakstatter et al., 1975; Malueg et al., 1973; U.S. Environmental Protection Agency, 1973; and Vollenweider, 1968).

Several factors have complicated attempts to quantify the relationship between lake trophic status and measured nutrient concentrations of the lake waters. For example, measured inorganic nutrient concentrations do not denote nutrient availability to the plant, but merely represent what is left over by the lake production process. A certain fraction of the nutrients (particularly phosphorus) while passing through successive biological cycles, become refractory. In addition, numerous morphometric and chemical factors affect the availability of nutrients in lakes. Factors such as mean depth, basin shape, and detention time affect the amount of nutrients a lake can absorb without developing nuisance conditions. Nutrient budget calculations represent the first step in quantifying the dependence of lake water quality on the nutrient supply. It is often essential to quantify the sources of nutrients from management and eutrophication control viewpoints. Thus Vollenweider (1968) and others have emphasized the importance of quantifying nutrient supply, particularly those of nitrogen and phosphorus, in determining lake trophic status.

A few nutrient budgets for lakes have been reported in the literature, e.g., Hetling et al. (1975) for Canadargo Lake in New York; Greeson (1971) for Oneida Lake in New York; McGauhey et al. (1963) for Lake Tahoe in California; and Sonzogni and Lee (1972) for Lake Mendota in Wisconsin. Vollenweider (1968) has summarized most of the budget calculations for European and North American lakes. Comprehensive evaluation of nutrient balance for a lake system requires measurement of all potential nutrient sources and sinks over an extended period to assess seasonal and other effects.

Potential sources of nitrogen and phosphorus for lakes are the watershed drainage which can include agricultural runoff, urban runoff, swamp and forest runoff, domestic and industrial waste discharges, septic tank discharges from lakeshore developments, precipitation on lake surface, dry fallouts (i.e., leaves, dust, seeds, and pollen), groundwater influxes, nitrogen fixation, sediment recycling, and aquatic bird and animal wastes. Potential sinks can include outlet losses, fish catches, aquatic plant removal, denitrification, groundwater recharge, and sediment losses. Some sources and sinks, e.g., groundwater, nitrogen fixation and denitrification, and nitrogen and phosphorus recycling from sediments, require elaborate sampling and experimental procedures for adequate evaluation. Consequently, manpower and time constraints have resulted in less than complete nutrient balance studies reported thus far.

A partial nitrogen and phosphorus budget is presented here. The budgets are referred to as ‘partial’ since no attempt was made to account for such sources of nitrogen and phosphorus as groundwater, leaves, pollen, seeds, or nitrogen fixation.

Nutrient emissions from municipal and industrial waste treatment plants in Illinois, contributing to the lake system either directly or through tributaries, were estimated by assessing the effluent quality characteristics. The nutrients transported by the Fox River and other tributaries to the Fox Chain of Lakes were measured by determining the weekly phosphorus and nitrogen concentrations in the streams in conjunction with the streamflow rates. Because the sampling sites were located on the tributaries close to the lakes, nutrient transport values estimated for the tributaries combine the contributions of point and nonpoint sources. Thus, knowing the total nutrient transport of the streams, and the municipal and industrial wasteflow contributions, the magnitude of nonpoint sources contribution could be estimated by difference. Nutrient contributions from precipitation were determined from phosphorus and nitrogen concentrations of rainwater samples collected periodically at three sampling sites in the area and from the total annual precipitation for the area.

The nutrient contribution from groundwater is considered to be insignificant compared with other sources. In Cedar Lake, which is entirely springfed, the nitrogen concentrations were found to be comparable to those found in the Fox Chain of Lakes. However, the phosphorus concentrations in Cedar Lake were much less. As discussed earlier, this was true even in the hypolimnetic zone of Cedar Lake which was anoxic during summer months.

Leaves have been reported as a source of phosphorus. Cowen and Lee (1973) reported that inorganic phosphorus was rapidly leached out of leaves collected in early autumn. Even leaves weathered by rainwater or lake water contained significant amounts of leachable phosphorus. A leaching rate of 54 to 230 micrograms (μg) of phosphorus per gram of leaves was reported by Cowen and Lee. The Fox Chain of Lakes and the interconnecting channels were observed on several occasions to be covered with leaves, cottonwood fuzz, seeds, and other dry fallouts. Nutrient contributions from these fallouts and from aquatic bird wastes could not be readily evaluated.

Contributions from Precipitation

The nitrogen and phosphorus concentrations measured in rainwater samples collected from three locations in the Fox Chain of Lakes area are shown in table 51. For comparison, analyses of rainwater samples collected at the site of the Water Quality Section's laboratory in Peoria is also shown in the table. Even though the analyses were made on rainwater samples only, these results are equally applicable to snow samples. Murphy (1974) investigated the phosphorus content in rain and snow in the Chicago area and found the concentrations of phosphorus were similar. The mean concentrations of nitrate nitrogen, ammonia nitrogen, Kjeldahl nitrogen, and total phosphorus in rainwater, based on all the observations made in the Fox Lakes area, are 0.93, 2.02, 2.71, and 0.41 mg/l, respectively. The long-term average annual precipitation at Antioch in the Fox Chain of Lakes area, as reported by the National Weather Service, is 33 inches. The annual rate of nitrate, ammonia, and total phosphorus addition, expressed as pounds per acre of lake surface per year (lbs/ac/yr), due to direct precipitation amounts to 6.61, 15.10, and 2.84, respectively. The Fox Chain of Lakes with a total surface area of about 6850 acres receive annually 44,600 lbs of nitrate, 103,400 lbs of ammonia, and 19,500 lbs of total phosphorus from direct precipitation.

Murphy (1974) reported a rate of 0.22 lbs/ac/yr of total phosphorus input from precipitation (29 inches) in the Chicago area. Sonzogni and Lee (1972) have reported

Table 51. Rainwater Quality Characteristics at Four Locations in Illinois
(in milligrams per liter)

Location	*Number of analyses*	*Range*	*Mean*	*Number of analyses*	*Range*	*Mean*
		Nitrate-N			**Kjeldahl-N**	
Antioch	11	0.26-2.90	0.77	9	0.65-5.01	2.88
Chain of Lakes State Park	8	0.32-1.58	0.81	3	1.81-11.10	5.45
Lake Villa	16	0.37-2.43	1.11	9	0.85-6.03	2.53
Peoria	6	0.30-0.85	0.53	3	0.46-1.22	0.86
		Ammonia-N			**Total phosphorus**	
Antioch	11	0.38-4.78	2.22	11	0.12-0.64	0.27
Chain of Lakes State Park	8	0.32-5.67	1.97	8	0.00-0.70	0.32
Lake Villa	14	0.40-4.82	1.90	14	0.03-2.09	0.56
Peoria	6	0.34-1.07	0.68	5	0.04-0.41	0.17

loading rates of 2.9 lbs/ac/yr for nitrate, 2.5 lbs/ac/yr for ammonia, and 0.21 lbs/ac/yr for total phosphorus to Lake Mendota (Wisconsin) from atmospheric precipitation (30 inches). The U.S. Environmental Protection Agency (1973) has reported rates of nitrogen contribution from precipitation comparable to rates found in this study for other midwestern cities. However, the phosphorus rates summarized in that report are much less than those obtained in this study.

Contributions from Septic Tanks

Septic tank field-tile systems are generally a problem only if they become clogged or water-bound which is likely to occur in non-sandy soils such as the types found in the Fox Chain of Lakes area. When this occurs, surface discharge results, and nutrients in the effluent can be transported via overland flow to the lakes. Local residents have reported such occurrences that are apparent in winter under snow-cover conditions.

In the absence of detailed field measurements to quantify the nutrient contribution to the lakes from septic systems, an indirect approach in estimating the nutrient contribution was taken here. It was reported earlier that there are about 6900 residential units around the lakes that use septic systems, and institutions like schools, churches, hotels, restaurants, and other commercial enterprises contribute an estimated daily wasteflow of 155,000 gallons. This wasteflow is equivalent to that flow from a population of 1550 persons at the rate of 100 gallons per capita per day. Further, assuming an average occupancy of three persons per residential unit, the total number of humans contributing to wasteflow could be 22,250. Therefore, the assumption here is that 22,250 persons are served by septic tanks.

Phosphorus and nitrogen contribution rates, in terms of pounds per capita per year (lbs/cap/yr), used in estimating the impact of septic tank systems by different investigators have differed significantly. Sonzogni and Lee (1972) used rates of 3.0 lbs/cap/yr for soluble orthophosphorus; 4.5 lbs/cap/yr for total phosphorus; 6 lbs/cap/yr for inorganic nitrogen; and 8 lbs/cap/yr for total nitrogen in estimating septic tank contributions to Lake Mendota in Wisconsin. The U.S. Environmental Protection Agency (1973) recommends rates of 3.3 lbs/cap/yr for total phosphorus and 14.3 lbs/cap/yr for total nitrogen with the suggestion to reduce these values to account for seasonal occupancy, lack of laundry facilities, and other factors. Shannon and Brezonik (1972) employed rates of 8.7 and 1.75 lbs/cap/yr for nitrogen and phosphorus, respectively, in estimating nutrient input from septic tank systems around 55 lakes in Florida. The rates used by Sonzogni and Lee for Lake Mendota were used in this investigation to evaluate the impact of septic tank disposal systems.

The McHenry County Public Health Administrator estimates that about 10 to 20 percent of the septic tank systems in the Pistakee Lake area could be malfunctioning (William Mellon and Richard Wissell, personal communication 1976). If we use a failure rate of 20 percent for the entire Fox Chain of Lakes area, the amount of nitrogen and phosphorus added to the Fox Chain of Lakes from malfunctioning units are estimated to be: inorganic nitrogen, 26,700 lbs/yr; total nitrogen, 35,600 lbs/yr; dissolved orthophosphorus 13,400 lbdyr; and total phosphorus 20,000 lbs/yr.

Nutrient Balance

An inventory of nutrient emissions to the Fox Chain from major point sources within Illinois is shown in table 52. The average emission rates in terms of pounds per day for ammonia, nitrate, organic nitrogen, dissolved orthophosphorus, and total phosphorus are shown in this table. Total inorganic nitrogen and dissolved orthophosphorus, which are the readily available forms of plant nutrients discharged by the Woodstock northside treatment plant, were the highest among the observed values. These were followed by the discharges from the Round Lake Sanitary District waste treatment plant.

Table 53 shows the nutrient loads transported by the Fox River and other tributaries to the Fox Chain of Lakes. The nutrient loads transported by the streams are further classified as to point or nonpoint sources. Because no detailed inventory of point nutrient sources in Wisconsin was made, the nutrients transported by the Fox River into the Fox Chain of Lakes are all shown as from nonpoint sources. The relative importance of point sources and nonpoint sources in nitrogen and phosphorus contributions is also shown in the table.

Nutrient flux from major sources to the Fox Chain of Lakes is shown in table 54. Rates of transport in tons per year for ammonia, nitrate, total nitrogen, dissolved orthophosphorus, and total phosphorus are listed. Tributaries to the Fox Chain, direct waste discharge, impact of septic tank disposal systems, and precipitation were the major sources of nutrients considered in this investigation. The sum of the first eight items in the list indicates the annual gross loading rates of nutrients. Data pertaining to the Fox River at Johnsburg indicate the magnitude of nutrients transported away from the Fox Chain. The

Table 52. Inventory of Point-Source Nutrient Emissions

Treatment plant	Mean flow (mgd)	Ammonia-N Mean concentration (mg/l)	Ammonia-N Load (lbs/day)	Nitrate-N Mean concentration (mg/l)	Nitrate-N Load (lbs/day)	Organic-N Mean concentration (mg/l)	Organic-N Load (lbs/day)	Total phosphorus Mean concentration (mg/l)	Total phosphorus Load (lbs/day)	Dissolved Orthophospho rus Mean concentration (mg/l)	Dissolved Orthophospho rus Load (lbs/day)
Hebron Trickling Filter Plant	0.10	10.05	8.38	11.95	9.97	9.06	7.56	13.54	11.29	10.50	8.76
Hebron Lagoons	0.09	6.91	5.19	1.03	0.77	12.41	9.32	3.83	2.87	2.68	2.01
Richmond	0.15	8.69	10.87	4.79	5.99	3.86	4.83	9.24	11.56	5.57	6.97
Woodstock, Northside Plant	2.00	13.05	217.67	0.55	9.17	4.06	67.76	7.56	126.10	5.68	94.74
Woodstock Die Casting Company	0.55	0.19	0.87	1.70	7.80	1.84	8.45	2.82	12.94	2.30	10.55
Lake Villa	0.15	4.68	5.85	4.19	5.24	2.71	3.39	7.02	8.78	4.38	5.48
Round Lake Sanitary District	1.60	15.15	202.16	0.56	7.47	9.32	124.44	5.09	67.92	4.18	55.78
Antioch	0.75	5.84	36.52	5.17	32.34	5.40	33.80	5.91	36.97	4.37	27.33
Fox Lake	0.25	14.08	29.36	1.60	3.34	6.14	12.81	7.11	14.82	4.83	10.07

Table 53. Nutrient Loads Transported by the Fox River and Other Tributaries

(Nutrient loads in pounds per day)

Stream	Ammonia-N Point source	Ammonia-N Nonpoint source	Nitrate-N Point source	Nitrate-N Nonpoint source	Total nitrogen Point source	Total nitrogen Nonpoint source	Dissolved orthophosphorus Point source	Dissolved orthophosphorus Nonpoint source	Total phosphorus Point source	Total phosphorus Nonpoint source
Inflow										
Fox River at Route 173	1010		5354			10,874		428		1187
Lily Lake Drain	3		14			45		0.45		1.7
Nippersink Creek	243	43	34	1337	375	2,021	123	4	165	58
Sequoit Creek	37	17	32	20	103	46	27	6	37	24
Squaw Creek	208	3	13	155	349	224	61	46	77	54
Total nutrient inflow	488	1076	79	6880	827	13,210	211	476	279	1325
Percentage*	3.5	7.7	0.6	49.0	5.9	94.1	13.2	29.7	17.4	82.6
Outflow										
Fox River at Johnsburg	1	139		4738		10,723		509		1093

*Computed with respect to total nitrogen or total phosphorus as the case may be

Table 54. Nutrient Balance for Fox Chain of Lakes

(Rates in tons per year)

Sources	*Ammonia-N*	*Nitrate-N*	*Total nitrogen*	*Dissolved orthophosphorus*	*Total phosphorus*
Inflow					
Fox River at Route 173	184.3	977.1	1984.5	78.1	216.6
Lily Lake Drain	0.6	2.6	8.2	0.1	0.3
Nippersink Creek	52.2	250.2	437.3	21.7	40.7
Sequoit Creek	9.9	9.5	26.8	6.0	11.1
Squaw Creek	38.5	30.7	104.6	19.5	23.9
Fox Lake WTP	5.4	0.6	8.3	1.8	2.7
Septic tank systems	6.7*	6.7*	17.8	6.7	10.0
Precipitation	51.7	22.3	113.2	4.9**	9.8
Gross loading rate	349.3	1299.7	2700.7	138.8	315.1
Outflow					
Fox River at Johnsburg	253.9	864.7	1956.9	92.9	199.5
Net loading rate	95.4	435.0	743.8	45.9	115.6
Percent retained	27.3	33.5	27.5	33.1	36.7

*Assumed inorganic nitrogen equally divided
**Assumed dissolved orthophosphorus 50 percent of total phosphorus

Table 55. Relative Importance of External Sources of Nutrients, in Percent

Sources	*Ammonia-N*	*Nitrate-N*	*Total Inorganic nitrogen*	*Tota nitrogen*	*Dissolved orthophosphorus*	*Total phosphorus*
Fox River at Route 173	52.8	75.2	70.4	73.5	56.3	68.7
Lily Lake Drain	0.0	0.0	0.2	0.0	0.0	0.1
Nippersink Creek	14.9	19.3	18.3	16.2	15.6	12.9
Sequoit Creek	2.8	0.7	1.2	1.0	4.3	3.5
Squaw Creek	11.0	2.4	4.2	3.9	14.0	7.6
Fox Lake WTP	1.5	0.0	0.4	0.3	1.3	0.9
Septic tank systems	1.9	0.5	0.8	0.7	4.8	3.2
Precipitation	14.8	1.7	4.5	4.2	3.5	3.1

difference between the gross loading rate and the nutrients carried away from the Chain of Lakes represents the amount of nutrients retained by the lakes or the net loading rate of nutrients to the lake system. It is seen that about 27 percent of total nitrogen and 37 percent of total phosphorus applied to the system are retained in the lake system.

The relative importance of the various sources of nutrient input is presented in table 55. In addition to the values for ammonia nitrogen and nitrate-nitrogen, percentage figures for total inorganic nitrogen are also given. These represent the readily available form of nitrogen for biological processes. The Fox River transports about 70 percent of inorganic nitrogen, and together with Nippersink and Squaw Creeks contributes about 93 percent of the total inorganic nitrogen. Also, these three streams contribute about 86 percent of the dissolved orthophosphorus with the Fox River applying about 56 percent. The impact of septic tank systems and precipitation appears to be equal as far as total phosphorus is concerned. Each contributes about 3 percent of the total.

The best available guidelines for relating the nutrient flux to water quality in lakes were first proposed by Vollenweider (1968). These guidelines were used subsequently by several investigators (Bartsch, 1972 and Gakstatter et al. 1975). According to Vollenweider, for lakes with mean depths of 5 meters (16.4 feet) or less, permissible loading levels of biochemically active nitrogen and phosphorus are, respectively, 1.0 and 0.07 g/m^2 /yr. For the same average depth, loading rates greater than 2.0 g/m^2 /yr for nitrogen and 0.13 g/m^2 /yr for phosphorus are considered excessive from a eutrophication point of view. The overall mean depth of the Fox Chain of Lakes is 5.7 feet; therefore, the guidelines cited by Vollenweider are applicable to the Chain of Lakes. The loading rates of total inorganic nitrogen and dissolved orthophosphorus for the Chain of Lakes were found to be 54.0 and 4.5 g/m^2 /yr, respectively.

The net loading rates, considering only the available nitrogen and available phosphorus retained in the lakes, were found to be 17.4 and 1.5 g/m^2 /yr, respectively, for nitrogen and phosphorus. It is seen that even the net loading rates of biologically active nitrogen and phosphorus are about ten times greater than the excessive levels of nutrient loading rates suggested by Vollenweider. The total nitrogen and total phosphorus loading rates for Lake Mendota were reported to be 3.10 and 0.17 g/m^2 /yr, respectively, and those for Lake Washington were 31.4 and 1.34 g/m^2 /yr (Vollenweider, 1968). Lake Waubesa in Wisconsin was found to have nutrient loading rates comparable to the rates observed for Fox Chain of Lakes. These three lakes were classified as eutrophic.

The total nitrogen and total phosphorus loading rates contributed to the lakes by the Fox River are 65.0 and 7.1 g/m^2 /yr, respectively. The Fox River inorganic nitrogen and dissolved orthophosphorus loading rates amounted to 38.0 and 2.6 g/m^2 /yr. These loading rates are several orders of magnitude larger than the excessive loading rate suggested by Vollenweider and others. The importance of controlling nutrient influx through the Fox River is apparent. *Efforts to minimize eutrophic conditions by controlling all the other sources without curbing nutrients transported by the Fox River would be futile.*

Nutrients entering a lake become incorporated in various chemical and biological cycling processes. Part of the nutrients are utilized in the metabolism of organisms, i.e., phytoplankton, zooplankton, vascular plants, and fish, and on death and decay of the organisms the nutrients are liberated for reuse. Part of the nutrients remain in solution, part become temporarily or permanently incorporated in the bottom sediments, and part flow out of the lakes. Part of the nutrients that accumulate in the lake, after a period of years, become available in concentrations sufficient to support algal blooms.

Organic matter, whether derived from external sources or produced internally, decomposes and oxidizes in the aerobic parts of the lakes. Organic decomposition, during periods of large algal blooms and in anaerobic zones, occurs when the biological production exceeds the oxidizing capacity of the waters. This results in higher concentrations of organic detritus in the bottom sediments of deeper water areas. Highly productive areas as in bays and dug channels, also exhibit accumulations of organic debris.

When anaerobic conditions develop at the sediment-water interface, nutrients contained in the organic debris are released back into the water. Such 'internal loading' varies from one water body to another and may be very high in certain specific cases. Vollenweider concluded, based on his extensive studies of European and North American lakes, that internal loading is a more serious threat to small lakes than to big lakes, whereas external loading plays a more important part in bigger lakes.

Vollenweider (1968) estimated sediment nutrient release rates of 1.2 and 0.01 g/m^2 /day for ammonia and phosphorus, respectively, under anaerobic conditions. Fillos and Swanson (1975) reported phosphorus release rates of 1.2 and 26.0 mg/m^2 /day under aerobic and anaerobic conditions, respectively, for the Lake Warner, Massachusetts, sediment samples.

Sridharan and Lee (1974) reported from laboratory leaching experiments with Green Bay (Lake Michigan) sediment samples that release rates of orthophosphorus under anaerobic conditions were much higher than the rates of release under aerobic conditions. They also found that the leaching rates were inversely affected by the presence of orthophosphorus in the leaching medium. They further postulated that the anoxic orthophosphorus release resulted from chemical reactions which are most likely abiotic. Bannerman et al. (1975) experimented with intact core samples from Lake Ontario and found the inorganic phosphorus release rates to be about 0.2 mg/m^2 /day, and estimated that this release rate constituted a sediment phosphorus contribution amounting to 10 percent of the external phosphorus loading to the lake.

Nutrient release, particularly that of phosphorus from sediment, appears to have a significant bearing on the productivity of lakes especially those that experience thermal stratification. Peterson et al. (1974) reported on a lake restoration experience with Snake Lake in Wisconsin. Dilution and flushing of the lake basin was tried as a restoration technique, by induction of low nutrient groundwater flow. They established that equilibration with lake bottom sediments could easily supply the dissolved nutrients observed in the lake and concluded that dilution coupled with other management techniques like nutrient inactivation, dredging, etc., may be necessary to achieve long-lasting satisfactory results. The remarks of Stumm (1975) are pertinent here:

> "The concentration of phosphorus in a surface water cannot be predicted merely on the basis of the supply of phosphorus and hydrographic conditions. For the assessment of the potential synthesis of biomass in an ecosystem, under conditions where phosphorus is a growth-limiting nutrient, two gross parameters are pertinent: (1) the ultimate capacity for photosynthetic production is given by the total reserve of phosphorus in a body of water i.e., the quantity of soluble particulate, sestonic and accessible sedimentary phosphorus, (2) the productivity measures the rate of biomass production. Both of these parameters are influenced by the extent and rate of regeneration of phosphorus and its exchange with sediments."

As indicated earlier, anoxic conditions in the Fox Chain of Lakes exist in the areas where the depth of water columns exceed 15 feet. From the depth-area relationships developed for the lakes, the areal extent of the lake system with water depths greater than 15 feet at normal pool level was estimated as 440 acres. Assuming a phosphorus release rate of 10 mg/m^2 /day (Vollenweider, 1968) under anaerobic conditions, the amount of phosphorus released from the sediments to the overlying waters during the 5-month period of thermal stratification is estimated to be 2.673×10^6 grams or approximately 5900 pounds. This alone results in a phosphorus loading rate of 0.1 g/m^2 /yr for the entire lake system. The loading rate approaches the excessive loading rate from eutrophication viewpoint suggested by Vollenweider (1968). Phosphorus recycled from the sediments in deep water areas of the Fox Chain of Lakes alone appears to be adequate for sustaining algal blooms for a long period of time.

REHABILITATION SCHEMES

Previous Investigations

Investigations designed to determine the interrelationship of the many factors contributing to the ecological system of the Fox Chain of Lakes did not exist until the present investigation was undertaken. However, in order to present a historical perspective of the problems associated with the lakes, a brief review of the previous studies is presented here.

It appears that in most instances past investigations were limited to a 2 to 3 day period and were predicated upon complaints, or had as an objective some other single facet associated with the waters such as fish population or bacterial densities. Most of the water chemistry data, temperature, and dissolved oxygen measurements have been incidental to the main purposes of the investigations. No examination of bottom deposits for nutrient material or organic content has been previously undertaken. With the exception of the Lake County Health Department 14-month study (1963-1964) regarding a nutrient budget for the Chain, there does not appear to have been a long-term study with the objective of determining specifically the principal factors which stimulate and sustain annually nuisance algal blooms.

Lake County 1962 Report. A report by the Lake County Health Department (1962) summarizes its observations in an effort to evaluate the quality of lake and stream waters in the county that might affect public health. Solely on the basis of bacterial densities, it was concluded that " . . . at present the streams rather than the inland lakes, are the source of concern . . . . While Lake County bathing beaches are reasonably free from bacteria, they are not free from algae blooms . . . . The Chain O'Lakes was not extensively evaluated [in 1959] because little data had been collected from these lakes at that time . . . ." The report also noted, "A special study was conducted on eight selected lakes, i.e., Cedar Lake, Long Lake, Third Lake, Loch Lomond, Bangs Lake, Slocum Lake, Lake Zurich, Diamond Lake, on a year round basis. The study concluded that although adding effluent which has been properly purified and chlorinated to lakes probably is not a danger to the health of the public, it often accelerates the natural aging process, eutrophication."

Public Health Service Report. The U.S. Public Health Service (1963) in its memorandum report summarized a 2-day sampling program and set forth some chemical and biological data. The probable causes of the algal problem existing at that time in the lake system was stated to be 1) shallow depths of the lakes, 2) influx of nutrients, 3) extensive canalization of low areas of the lakes, 4) septic tank drainage at the lakes' edges, 5) climatic conditions including low streamflows, and 6) the recirculation of nutrients from the bottom muds. The bloom was of *Aphanizomenon flos-aquae.* The conclusions reached were:

1) "The several lakes of the Fox Chain are biologically overproductive as manifested by algal blooms of such density and quality as to cause nuisance conditions and concomitant devaluation as aesthetic and recreational resources."

2) "The algal condition in 1963 was caused by a combination of low flow in the Fox River and fertilization from three principal sources: (a) treated sewage from the villages of Antioch, Fox Lake, and Round Lake; (b) infiltration of septic tank absorption field in unsewered residential areas; and (c) agricultural runoff in the drainage basin and from the nutrient rich waters of the Fox River."

3) "According to Dr. Baker, no serious health hazard existed. However, the quality was so poor as to be undesirable for bathing and aesthetic enjoyment."

4) "A program to remedy the problem should be initiated immediately. Such a program must involve the best informed technical talent available and will require the full cooperation of local residents as well as the assistance of federal and state agencies having capabilities in this field."

The remedial measures proposed consisted of a combination of actions, some of which could be taken immediately and others on a long-term basis. These, as stated, were as follows:

"A wide area master sewage plan should be developed as soon as possible. This master plan should provide for the proper collection, treatment, and ultimate disposal of domestic and industrial waste waters, and it should encompass all of the area of the Fox River watershed that is within the Chicago Metropolitan area, rapidly becoming urbanized. The plan should identify strategic points at which waste waters can be discharged with minimum detriment to the stream, and analysis should be made of the impact of waste loads on the system, for both present and projected future loads. This does not mean that a single sewerage system must be built for the whole area. It does mean, however, that coordinated planning should be done for the whole area. Piecemeal planning by individual municipalities, or even by single counties, will not produce an adequate plan, as the water does not respect political boundaries. In addition, piecemeal planning will not receive the widespread acceptance essential to promulgation.

"Reduction in present nutrient contributions will be required if the waters of the Chain of Lakes and the Fox River are to be improved. This can be accomplished by the elimination of septic tanks in built-up areas, especially those draining to the lakes; by construction of intercepting sewer systems; and by changing locations of wastewater outfall points to bypass the lakes and discharge to flowing streams. Concentrations of nutrients might also be reduced by placing restrictions on the operation of high-speed motor boats that may cause resuspension of nutrients from the bottom sediments. Bottom-feeding fish, such as carp and catfish, may also cause resuspension of bottom sediments. The possibility of removal of bottom-feeders, and restocking with more desirable species should be investigated. Future construction of long, narrow embayments where stagnant water conditions develop should be prohibited.

"Institution of the foregoing measures as soon as practicable should result in considerable alleviation of the presently unsatisfactory conditions. Further progress toward restoration of the subject waters will require several long-range approaches. If determined feasible from an engineering viewpoint, the organic sediments in the lakes should be dredged and the spoil used for filling or diking off the bordering marshlands. The possibilities for flow regulation by upstream storage that would augment low flow in the Fox River, and for further reduction in nutrient concentrations in wastewaters by additional treatment, should be investigated. The latter will require research to develop economically feasible techniques. The possibility should be explored of using biological controls such as stocking the lakes with plankton feeding fish or establishing rooted aquatic plants that will successfully compete with the algae for nutrients and light. No efficient methods of biological control are known at this time."

Illinois Conservation Department Report. The Illinois Department of Conservation (1964) report on fishery investigations primarily dealt with fish population in the Chain. Some limited data regarding water chemistry is presented in the report along with the observation concerning "The apparent depletion of abundance of submergent vegetation and the expanding problem of planktonic algae dramatizes the significance of the biological changes which are taking place." Conclusions other than those pertaining to the fish population consisted primarily of the following:

"The main cause for concern at this time would be the future water quality. The increasing severity of planktonic blue-green algae blooms during the warm months indicates environmental conditions which approach stagnation in some lakes of the Chain causing fish mortality in areas, as a result of algae drift and decomposition. Low water flow caused by prolonged drought was a contributing factor to algae abundance in 1963. Because blue-green algae blooms are usually related to the influence of extra nutrients such as from sewage treatment plants and septic systems, it is recommended that a central sewage collection be expanded in the residential areas around these lakes and that sewage treatment plants' effluents be diverted away from the watershed leading into these lakes.

"A regular program of water sample collections should be set up for the Chain, which would include chemical and biological analyses of the water. Permanent sampling stations and standardized sampling procedure should be a part of such a program. All public agencies concerned with the Fox Chain of Lakes area, be it conservation, health, boat safety, waterways, or recreational planning should coordinate and share their programs and information one with another."

Illinois Public Health Report. The Illinois Department of Public Health in its survey of the Fox River in Illinois (April, June, July 1964) presented an appraisal of bottom fauna existing in the Fox River basin. The report identifies sampling points and presents a list of those organisms collected. The investigator stated:

"It is possible from a study of the interrelationship of the species of bottom-dwelling organisms and their individual responses to a pollution load to devise a system of stream classifications. Such a system acknowledges the degree to which an environment is capable of maintaining a state of balanced aquatic production."

Nine locations on the lakes and their tributaries were sampled. The stations and their classifications are as follows:

Fox Lake	Polluted
Pistakee Bay	Semi-polluted
Squaw Creek	Unbalanced
Fox-Petite Channel	Semi-polluted
Fox-Grass Channel	Polluted
Sequoit Creek	Semi-polluted
Nippersink Creek	Unbalanced
Fox River at Illinois Route 173	Semi-polluted
Fox River at Wisconsin-Illinois line	Unbalanced

The following criteria were used in classifying the streams and lakes:

- A clean or balanced environment is one in which conditions are maintained capable of supporting a great variety of organisms, mostly intolerant species, from various taxonomic groups.
- An unbalanced environment is one in which the balance of life, as described for a clean water environment, has been disrupted but not destroyed. The population numbers of some of the intolerant forms are reduced and an increase becomes apparent in some of the more tolerant forms.
- A semi-polluted environment is one in which the balance of life found in a clean water environment is destroyed. Intolerant forms are completely absent or reduced to a minimum and the environment is composed predominately of tolerant forms.
- A polluted environment is one in which only the very tolerant forms are able to exist. These are usually present in great numbers unless excluded from the environment by severe conditions.

Lake County Recent Study. Report of the nutrients study undertaken by the Lake County Health Department (1963–1964) has not yet been prepared. Preliminary findings indicate that the phosphorus load to the lake system was 295 tons annually. Of this, approximately 75 percent originated from the drainage area in Wisconsin. These figures and the phosphorus load transport values for other tributaries are found to be comparable to the values obtained in this investigation.

Eutrophication

Because one of the primary objectives of this investigation is to develop strategies to restore and manage the water quality of the Chain of Lakes which involve reversing or retarding the aging process of the lakes, it seemed desirable to review some information pertaining to the eutrophic process in general, and the lake restroation techniques tried elsewhere.

Eutrophication is a natural aging process that affects every body of water from the time of its formation. Many interacting factors contribute to the overall process of eutrophication—a term more widely known to mean the nutrient enrichment of waters. The eutrophication of a lake system consists of the gradual progression from one life stage to another based upon changes in the degree of nutrient input or productivity. The youngest stage of the life cycle is characterized by low concentrations of plant nutrients and little biological productivity. Such lakes are called oligotrophic lakes. At a later stage in the succession, the lake becomes mesotrophic, and as the life cycle continues the lake becomes eutrophic or highly productive. The final life stage before extinction is a pond, marsh, or swamp.

As a lake ages, the degree of enrichment by nutrient materials increases. In general, the lake traps a portion of the nutrients originating in the surrounding drainage basin. In addition, precipitation, dry fallout, and in certain cases groundwater inflow are contributing sources. The shore vegetation and higher aquatic plants utilize part of the inflowing nutrients, grow abundantly, and in turn trap the sediments. The lake gradually fills in, becoming shallower by the accumulation of plants and sediments on the bottom and smaller by the invasion of shore vegetation, and eventually becomes a dry land. The extinction of a lake is the result of enrichment, productivity, decay, and sedimentation.

Human activities, such as altering lake drainage basins, agricultural practices, deforestation, and urban development, have hastened the nutrient addition to natural waters. When the pollutants are of a nutritional type, the enrichment of the recipient water is greatly accelerated and the rate of aging is consequently greatly increased. In this way, cultural eutrophication can significantly alter the rate of the natural process and shorten the life expectancy of the affected body of water.

Because eutrophic lakes contain an abundance of available nutrients, biological production is high and results in nuisance growths which adversely affect man's use of the water body. Plants, particularly algae, are of primary concern because they utilize dissolved inorganic nutrients from the water and thus become primary producers of new organic matter on which aquatic animal life depends. In oligotrophic lakes the phytoplankton are represented by large numbers of a few species. An overabundance of algae is generally called an algal bloom. Lackey (1949) and later Fruh (1967) arbitrarily defined an algal bloom as 500 cts/ml of raw water sample.

With the increased productivity associated with accelerated rates of eutrophication comes the filling of the lake basins with organic materials which subsequently exert an increased oxygen demand on the overlying waters. The increased oxygen demand may result in total depletion of oxygen in the cooler bottom waters during the summer, accompanied by an increase in the products of decomposition, e.g., carbon dioxide, ammonia, hydrogen sulfide, and methane. These developing anaerobic conditions result in replacement of benthic organisms with less desirable types.

In addition to restricting fish populations, highly eutrophic lakes are undesirable aesthetically and with respect to water use. Algal blooms produce taste and odor problems and create unsightly surface scums which discourage water contact recreational activities. Accumulation of algal mats and dense weed growths are most pronounced near shore. The accumulated algal masses decay resulting in extremely foul smelling conditions.

In addition to their deleterious effects on aesthetic and recreational aspects of the lake, excessive planktonic growths affect water supply resources. They create color, taste, and odor problems in water supplies and increase the rate of clogging of filters at water treatment plants.

Certain blue-green algae have been shown to have toxic effects on fish and animals. Evidence also exists that skin irritation and gastrointestinal disturbances may result in humans from contact and ingestion of lake water in which algae exist in bloom proportions.

Frequent and excessive algal blooms and the concomitant undesirable aesthetic effects have become a chronic problem in the Fox Chain of Lakes and have been the source of public complaints in the past few decades.

The algal blooms form a floating scum on the surface when the wind and wave actions are absent. Presence of gas vacuoles in the blue-green algae causes these algal species to rise to the surface forming a surface film. The algal masses, upon exposure to sunlight, become bleached giving the appearance and consistency of latex paint. Floating globs of decaying algae have been observed throughout the summer period in the Fox Chain of Lakes.

The dominance of cyanophyta in the deep lakes of the Chain, low algal diversity indices, high oxygen demand of the organic sediments in the bottom of the lakes with the consequent depletion of oxygen in the hypolimnetic zones of deep lakes, high concentrations of nitrogen and phosphorus in the lake waters, and other telltale signs point clearly to the fact that the lake system has reached the advanced stage of eutrophication. There is an abundance of macrophytes in the littoral zones of Channel Lake and Lake Catherine and in several of the man-made channels. Only a few isolated patches of macrophytes exist in the channel connecting Grass and Fox Lakes.

An algal assay was carried out with surface water samples from all of the lakes to determine the stimulatory effect of nitrogen and/or phosphorus on algal growth. This special examination was performed on three successive weekly surface water samples. The samples were spiked with either sodium nitrate at a concentration level of 2.5 mg/l as N, or monobasic potassium phosphate at 0.5 mg/l as P or both. A control was also run. The results are shown in table 56. The dissolved orthophosphorus concentrations in the water samples originally present are also listed. The means of all three experimental runs are shown in table 57.

The dissolved orthophosphorus concentrations and the biomass values in the control samples were low for all three test runs. This is mainly because the nutrients in solution have already been utilized by the active standing crop of algae leaving the lake surface waters basically deficient in nutrients. For the assays, the samples were filtered initially with 0.45 μm filter papers leaving the test media nutrient poor. The addition of either nitrogen or phosphorus resulted in only a very slight increase in some cases and a reduction of algal biomass in a few cases. The differences in mean values are so small that they could have arisen either from experimental procedures or from the marginal stimulatory or inhibitory effects of the added nitrogen and phosphorus (table 57). However, when both nitrogen and phosphorus were added to the water samples, the algal biomass increased several orders of magnitude.

The effort to delineate the effects of nitrogen and phosphorus clearly indicates that both these nutrients are essential to algal growth in the Chain. Lack of one of these nutrients

Table 56. Effects of Nitrogen and Phosphorus Additions on Algal Biomass

(in milligrams per liter)

Lake	*Control of dissolved orthophosphorus*	*Control*	*Nitrogen added*	*Phosphorus added*	*Nitrogen and phosphorus added*
October 6, 1975					
Catherine	0.04	8.6	15.2	12.1	59.6
Channel	0.15	8.6	49.5	6.8	58.6
Marie (east)	0.02	5.4	6.8	1.1	47.4
Marie (west)	0.02	3.2	6.1	1.1	47.5
Bluff	0.03	4.1	8.9	6.7	58.5
Petite	0.03	1.1	6.8	4.2	48.5
Fox Lake (Stanton Bay)	0.01	1.1	1.1	1.1	41.4
Fox Lake (main)	0.04	1.3	14.9	3.5	45.4
Fox Lake (Mineola Bay)	0.02	6.6	16.1	8.1	38.4
Nippersink	0.02	3.5	9.1	10.1	49.9
Pistakee (main)	0.01	7.4	8.7	9.6	57.5
Pistakee Bay	0.01	5.5	9.7	5.4	53.5
Grass Lake (south)	0.00	3.3	1.3	6.6	54.5
Grass Lake (north)	0.00	1.1	2.4	4.5	48.5
Cedar Lake	0.00	3.5	5.5	8.7	52.7
October 13, 1975					
Catherine	0.02	13.1	15.2	16.2	63.6
Channel	0.00	9.7	10.1	8.9	58.6
Marie (east)	0.00	6.8	15.1	14.1	64.6
Marie (west)	0.00	7.7	8.1	9.1	63.7
Bluff	0.00	9.7	7.1	10.1	61.5
Petite	0.00	7.6	7.6	9.1	53.6
Fox Lake (Stanton Bay)	0.00	6.6	12.1	7.6	54.6
Fox Lake (main)	0.01	5.6	21.2	8.1	56.4
Fox Lake (Mineola Bay)	0.03	9.5	27.2	7.1	54.6
Nippersink	0.03	4.6	13.2	9.8	58.6
Pistakee (main)	0.00	12.1	15.2	10.2	61.6
Pistakee Bay	0.00	9.1	11.1	10.1	51.6
Grass Lake (south)	0.01	12.2	11.2	10.9	56.6
Grass Lake (north)	0.01	13.1	14.1	12.1	55.5
October 20, 1975					
Catherine	0.02	6.2	16.1	23.2	55.7
Channel	0.01	9.1	9.1	10.1	54.6
Marie (east)	0.01	13.1	19.1	12.1	66.6
Marie (west)	0.01	8.8	14.1	6.9	49.6
Bluff	0.01	8.9	17.1	7.9	62.7
Petite	0.01	9.7	9.1	7.1	58.6
Fox Lake (Stanton Bay)	0.01	3.1	1.1	1.5	47.3
Fox Lake (main)	0.01	2.1	1.4	5.6	37.3
Fox Lake (Mineola Bay)	0.01	3.2	4.6	1.7	60.6
Nippersink	0.00	1.2	6.7	7.7	35.4
Pistakee (main)	0.01	9.1	17.2	16.8	69.7
Pistakee Bay	0.01	7.1	7.9	7.8	16.2
Grass (south)	0.01	11.7	12.1	4.5	17.7
Grass (north)	0.01	5.5	10.1	8.7	44.4
Cedar Lake	0.01	3.2	1.4	1.1	54.6

Note: Nitrogen added was in the form $NaNO_3$ (sodium nitrate) with 2.5 mg/l of nitrogen; phosphorus added was from solution K_2 HPO_4 with 0.5 mg/l of phosphorus.

Table 57. Effects of Nitrogen and Phosphorus Addition on Algal Biomass (Mean of Three Experimental Runs)

(in milligrams per liter)

Lake	*Control of dissolved orthophosphorus*	*Control*	*Nitrogen added*	*Phosphorus added*	*Nitrogen and phosphorus added*
Catherine	0.03	9.3	15.5	17.2	59.6
Channel	0.05	9.1	22.9	8.6	57.3
Marie (east)	0.01	8.4	13.7	9.1	59.5
Marie (west)	0.01	6.6	9.4	5.7	53.6
Bluff	0.01	7.6	11.0	8.3	60.9
Petite	0.01	6.1	7.8	6.8	53.6
Fox Lake (Stanton Bay)	0.01	3.6	4.8	3.4	47.8
Fox Lake (main)	0.02	3.0	12.5	5.7	46.4
Fox Lake (Mineola Bay)	0.02	6.4	16.0	5.6	51.2
Nippersink	0.02	3.1	9.7	9.2	48.0
Pistakee (main)	0.01	9.5	11.0	12.2	62.9
Pistakee Bay	0.01	7.2	9.6	7.8	40.4
Grass (south)	0.01	9.1	8.2	7.3	42.9
Grass (north)	0.01	6.3	8.9	8.4	49.5
Cedar	0.01	3.4	3.5	4.9	53.7

Note: Nitrogen added was in the form $NaNO_3$ (sodium nitrate) with 2.5 mg/l of nitrogen; phosphorus added was from solution K_2HPO_4 with 0.5 mg/l of phosphorus.

in the water effectively curbs algal growth. Blue-green algal species have been known to be capable of fixing nitrogen from the atmosphere and are capable of thriving at a much lower nitrogen concentration in water than is required by diatoms and greens. Consequently, controlling the availability of phosphorus in the waters at both external and internal sources appears to be an effective procedure for controlling algal blooms in the Chain of Lakes system. There is an abundance of evidence in the technical literature to indicate that algal blooms and phosphorus concentrations in lakes are directly related (Bartsch, 1972; Bachmann and Jones, 1974). Bartsch concluded that limiting phosphorus availability in lakes is the single most important and necessary step to be taken in eutrophication control.

Remedial Measures

Two recent publications (Dunst et al., 1974; U.S. Environmental Protection Agency, 1973) provide excellent summaries of remedial measures which have been applied in lake rehabilitation programs. The information given below has been taken mainly from these two sources.

Measures which may be effective in the restoration and enhancement of the quality of lakes can be considered under the following two major categories:

Limiting nutrient influx
- Point source nutrient removal and control
- Nutrient diversion
- Control of nonpoint sources of dnutrients

In-lake treatment and control measures
- Dredging
- Nutrient inactivation/precipitation
- Dilution and dispersion

- Lake bottom sealing
- Artificial destratification and hypolimnetic aeration
- Sediment exposure and desiccation
- Harvesting nuisance organisms
- Chemical control of nuisance organisms
- Biological control of nuisance organisms

The lake restoration techniques mentioned here have been employed either alone or in combination with one or more of the other techniques in lake restoration schemes. The U.S. Environmental Protection Agency (1973) report states:

> "The approach to the rehabilitation of degraded lakes is two-fold: (1) by restricting the input of undesirable materials and (2) by providing in-lake treatment for the removal or inactivation of undesirable materials. Obviously the only means of maintaining the quality of a lake once desired conditions are achieved, is by rigidly restricting the input of undesirable materials. In some lakes reducing or eliminating the primary sources of waste loading is the only restorative measure needed to achieve the desired level of improvement. Once the source of pollution is abated, natural flushing and dilution with uncontaminated water may result in substantial improvements in the quality of the lake. However, in many lakes, particularly in hypereutrophic lakes with slow flushing rates, in-lake treatment schemes may also be required before significant improvements will be realized. In-lake treatment alone, without controlling pollutional inflows, cannot be termed a restorative measure as only the symptoms or products of eutrophication and pollution are treated and no permanent improvements in quality are achieved. In any lake restoration program, controlling the input of undesirable materials is the initial step towards permanent lake rehabilitation; all other remedial measures are supplementary to this action."

Limiting Nutrient Influx

Limiting nutrient influx and diversion have been widely practiced as lake restoration schemes in the United States. Lake Tahoe is a striking example where advanced waste treatment for nutrients removal, removal of both nitrogen and phosphorus, and diversion of treated effluents were practiced even before any problems of eutrophication began to appear in the lake. Lake Washington near Seattle, prior to 1963, received heavy nutrient loading from 11 sewage treatment plants discharging directly into the lake. The lake deteriorated to a state of eutrophy. A series of steps were instituted by the municipality of metropolitan Seattle to divert sewage from Lake Washington. Complete diversion was achieved by 1968. Phosphorus was identified as the main element causing eutrophic conditions in the lake. With the reduction of phosphorus by about 80 percent of the initial levels, algal growth in the lakes decreased, secchi disc readings improved, and water quality improvement has been dramatic.

Madison lakes in Wisconsin have a long history of algal problems. Initially Lake Monona, the second of the series of four lakes on the Yahara River, received the sewage from the city of Madison. As a consequence, algal growths became prolific and a regular program of treatment with copper sulfate was established in 1925. In 1928, treated effluent was diverted to the Yahara River downstream from Lake Monona. Since the diversion, the amount of copper sulfate needed to prevent algal blooms decreased significantly. A change in species composition which did not form surface scums resulted in diminished copper sulfate application. Shortly after the effluent was moved downstream from Lake Monona, the symptoms of eutrophication in the lower two lakes began to intensify. Finally in 1958, the treated effluent from Madison was diverted to the Yahara River downstream from all the lakes. Since diversion, the conditions of the lakes have been reported to have improved (Sonzogni et al., 1975).

Nonpoint sources of pollution, which are incidental to land uses throughout the drainage basin of a lake, are a significant cause of lake degradation. Efforts to limit nutrients and sediment inputs from lands within drainage basins, for lake protection as well as rehabilitation, have followed two general lines: 1) structural and land treatment measures to intercept nutrients and sediments before they reach water bodies; and 2) regulatory approaches, particularly land use controls to restrict uses with direct or indirect pollution potential. Soil conservation practices like contour farming, grassed waterways, crop residue management, feedlot waste management, etc., fall into the first category. Regulations on fertilizer application shoreline protection statutes, etc., fall into the latter category.

In-Lake Treatment and Control Measures

Even though all of the in-lake treatment and control measures listed earlier are not directly applicable to the Fox Chain of Lakes, a brief description of these rehabilitation schemes is outlined here. Along with advantages and disadvantages, a few case studies where these techniques were employed are cited. This information should aid in developing a rational management scheme for the Fox Chain of Lakes.

Dredging. Dredging is considered a feasible method of nutrient control for preventing the recycling of nutrients from lake bottoms (U.S. Environmental Protection Agency, 1973; Pierce, 1970). As indicated earlier, highly eutrophic lakes receive large amounts of autochthonous materials resulting from massive algal growths. Much of the organic material entering the hypolimnetic zones will not degrade rapidly because of the lack of oxygen. Dredging of the accumulated organic matter has thus been proposed as a remedial technique.

Investigation of 49 lakes and ponds led Pierce (1970) to conclude that very little information is available as to whether dredging improves or damages a lake's aquatic environment. Most of the dredging reported in the literature is for maintaining navigational channels or for increasing the storage of water supply lakes. Churchill et al. (1975) reported that with limited dredging in Lake Herman, South Dakota, over a period of three years, significant amounts of nutrient were removed. Wilbur (1974) reported selective dredging in the littoral zones of two lakes in Florida where an organic muck bottom was converted to a sandy bottom which supported desirable benthic populations. The dredging effort was undertaken primarily to reverse the loss of a productive lake bottom and thus improve the gamefish population.

Lake bottom dredging experience in Lake Tummen in Sweden (143 acres, 6 feet mean depth), where the technique was adopted with the specific purpose of arresting the eutrophic trend in the lake, is worth mentioning here. The upper 2 feet of nutrient rich sediments were dredged. The phosphorus concentration of the lake water in the year following dredging was about 0.1 mg/l compared with values as high as 1.0 mg/l in the predredging period. Dissolved oxygen concentrations remained well above critical levels compared with total depletion in earlier years. Green algae replaced blue-green algae to a large extent, although phytoplankton production was still high. A general improvement in water quality has been observed (Dunst et al., 1974).

Dredging was also employed in Crystal Lake in Minnesota. This lake has a surface area of about 400 acres and a mean depth of about 15 feet. The effort was undertaken to minimize winter fish kills and reduce algal production. It is reported that no noticeable improvement in lake water quality occurred but further lake deterioration has been arrested (Dunst et al., 1974).

Among the several disadvantages cited for dredging, the significant ones are:

1) Dredging operations are expensive
2) The operation may release nutrients from the sediments into the overlying waters
3) The nutrient content of the sediments may remain high at considerable depth, thus making it impossible to reach a low nutrient level in sediment
4) Turbidity resulting from the process may persist for a considerable time during and following operations
5) Satisfactory disposal of the spoils may be very expensive

There are several methods available for dredging sediments from the lake bottom. These methods can be classified as either mechanical dredging or hydraulic dredging. Pierce (1970) has presented a detailed discussion on the various types of equipment and the methods employed in lake dredging operations. On the basis of his investigation of 49 lakes, Pierce reported that anticipated excavation costs for an average project (50,000 cubic yards of excavation or more) would be from 45 to 75 cents per cubic yard (¢/cu yd), if done on a contract basis. As the size of the project increases, the unit cost will decrease to the low side of the range. The U.S. Environmental Protection Agency (1973) has reported dredging costs of 38 and 47¢/cu yd in Wisconsin and Iowa lakes. Roberts (1975) reported a dredging cost of 78¢/cu yd for Lake Oakland in Illinois.

Nutrient Inactivation/Precipitation. This technique is viewed as a method of hastening the recovery of a lake from a eutrophic condition. The purposes of this in-lake treatment are:

1) To change the form of a nutrient to make it unavailable to plants
2) To remove the nutrient from the photic zone
3) To prevent release or recycling of potentially available nutrients within the lake

In-lake nutrient inactivation techniques have been primarily directed toward phosphorus. Inactivants which have received the most attention are aluminum, iron, and calcium salts. Of these three, aluminum appears to be the only one applicable to lakes in Illinois. Calcium is ineffective in removing phosphorus at pH values less than 9. Ferric iron is undesirable because of its tendency to be reduced to the soluble state under anaerobic conditions. Peterson et al. (1974a) reported that compounds of lanthanium, zirconium, tungsten, and titanium used in laboratory tests were capable of removing phosphorus from lake water. Other materials being used or considered as coagulants include ion exchange resins, polyelectrolytes, fly ash, powdered cement, and clay.

Horseshoe Lake (surface area 22 acres; maximum depth of 55 feet) in east-central Wisconsin was treated in May 1970 for nutrient inactivation. As reported by Peterson et al. (1973), slurried alum was applied to the top 2 feet of water at a concentration level of about 18 mg/l (200 mg/l alum). The results of the treatment were: 1) a decrease in total phosphorus in the lake water during the summer following treatment; 2) no large increase in total phosphorus in the hypolimnion during the following two summer stratifications; 3) some increase in the transparency of the water during the summer following treatment; 4) a short-term decrease in color; 5) an absence of the nuisance planktonic algal blooms that had been common in previous years; 6) marked improvement in dissolved oxygen conditions, particularly during the following winters; and 7) no observations of adverse ecological consequences.

Lake Langsjon (surface area of 86.5 acres; maximum depth of about 10 feet) located in Stockholm, Sweden, was treated with 33.5 tons of granulated aluminum sulfate (Jernelov, 1970). The immediate results of alum treatment were an increase in depth of secchi disc readings and a reduction in dissolved orthophosphorus from 60 to 5 μg/l. The flocculated aluminum hydroxide covered the bottom with a light grey layer of 0.5- to 1-inch thick. Better oxygen conditions prevailed during the winter of 1968–1969 and no release of phosphorus from the sediments was noted though conditions were anaerobic.

Shannon et al. (1974) reported the results of alum treatment (dosage 2.5 mg/l as Al) of the Fourth Welland Canal in Canada. Alum treatment was found to be effective in controlling algal mass. The zooplankton population was not adversely affected. The cost of treating the Fourth Welland Canal amounted to approximately $90 per acre of water surface.

The major drawbacks cited against this method of lake renovation are:

- The relatively high cost of treatment, particularly the manpower costs
- Possible toxic effects from the introduction of an excess of a metal
- Adverse biological effects from the formation of floc (The floc could conceivably suffocate aquatic organisms, as well as interfere with the benthic ecology.)
- Lack of information on the effective duration of the treatment (Continued inflow of nutrients and bacteriologic and benthic organism activity could influence the longevity of treatment effects.)

Dilution and Dispersion. This technique has been attempted to alleviate excessive algal growths and associated problems by reducing nutrient levels within the lake. This is accomplished by the replacement of nutrient rich waters with nutrient deficient waters and the washout of phytoplankton. Nutrient dilution has been attempted by two procedures: 1) pumping water out of the lake and permitting increased inflow of nutrient poor groundwater; and 2) routing additional quantities of nutrient poor surface waters into the lake.

The first procedure was used in Snake Lake in Wisconsin (Peterson et al., 1974b). Nutrient levels were initially reduced significantly and duckweed blooms were eliminated. Leaching from nutrient rich sediments limited the effectiveness in this particular case.

The second procedure has been tried in several places. Two of the most successful experiments were at Green Lake in Washington, and Buffalo Pound Lake in Canada (Dunst et al., 1974). After 5 years of flushing at rates of 3.5 times per year or less, and after some initial dredging in Green Lake, the blue-green algal standing crop was suppressed and there was a shift in dominance with the elimination of *Aphanizomenon.* Subnuisance levels of blue-green algae were attained after 4 years in Buffalo Pound Lake.

Lake Bottom Sealing. In lieu of physical removal of organic rich sediments, sediment sealing may provide control at less cost. Covering of bottom sediments with sheeting material (plastic, rubber, etc.) or particulate material (sand, clay, fly ash, etc.) can prevent the exchange of nutrients from the sediments to the overlying waters either by forming a physical barrier or by increasing the capacity of surface sediments to hold nutrients.

The problem encountered when covering sediments is the ballooning of the sheeting, or rupturing of the seal due to gas production in the underlying sediments. Sand and other materials of large size tend to sink below flocculant sediments. Clay, fly ash, bentonite, and other similar materials appear to be best suited for sediment covering.

Covering of sediment to improve lake conditions has been done at Marion Millpond, Wisconsin (University of Wisconsin, 1974). The 110-acre lake was treated in 1971 by the scraping of overburden to a sand substrate, providing a sand blanket, and covering a part of the lake sediments with black plastic sheeting anchored with sand and gravel.

Fly ash in combination with lime has been used for phosphorus removal and sediment covering in Lake Charles East in northeastern Indiana (Higgins et al., 1975) as a demonstration project. Treatment of a 7-acre portion of the lake required 2000 tons of fly ash and 24 tons of lime at an estimated cost of $22,000. To prevent phosphorus breakthrough for an extended period, a 2-inch layer of fly ash seal was found necessary.

Artificial Destratification and Hypolimnetic Aeration. Artificial destratification and hypolimnetic aeration is a process by which the lake waters are oxygenated and circulated. This is accomplished by either mechanical water pumps or by compressed air released at the lake bottom. In the case of compressed air mixing, vertical water currents are generated as the bubbles rise to the surface. The colder and denser bottom water mixes with warmer surface water, then sinks to a level of equal density and spreads horizontally. Oxygen is added to the water directly from the compressed air as well as by contact with the atmosphere. As the mixing process continues, complete circulation is achieved and the lake approaches uniform temperature and dissolved oxygen conditions from the surface to the bottom. The whole water mass becomes inhabitable by lake biota.

In contrast to total aeration, several types of aeration devices have been designed to oxygenate the hypolimnetic waters without disrupting thermal stratification. Typically, the aerator consists of a large diameter pipe which extends from the lake bottom to a few feet above the water surface. Water inlet ports are located near the bottom of the pipe and outlet ports are located below the thermocline. The bottom water is airlifted up the vertical tube. The rising bubbles are vented to the atmosphere and the water is returned to the hypolimnion.

The advantages of artificial destratification in eutrophic lakes are:

- With increased oxygen levels in the hypolimnion, there is a reduction in the anaerobic release of nutrients from the bottom sediments
- Oxidation of reduced organic and inorganic materials occurs in the water (This is particularly advantageous when the lakes serve as a raw water source, because taste, odor, and color problems caused by iron, manganese, and/or hydrogen sulfide are eliminated or at least minimized.)
- The range of benthic populations is extended to the profundal region which was once anaerobic (An increase in the number of fish and a shift to more favorable species can result from the greater availability of food organisms.)
- Favorable changes in algal populations occur with a decrease in undesirable blue-green species (This is a result of the lowering of water temperature of the algae between the euphotic and aphotic zones; however, there is no reduction in the productivity of the lake.)
- Evaporation rates are reduced in summer with the reduction in surface water temperatures
- Artificial destratification often results in increased water clarity
- Winter fish kills may be prevented by maintaining sufficient oxygen levels under ice

The disadvantages of artificial destratification include:

- Increased heat budget in the lake
- Aeration may temporarily increase water turbidity due to the resuspension of bottom sediments
- In most investigations artificial destratification resulted in a reduction in blue-green algae, but in other instances there had been no observable effect on blue-green algae
- The artificial destratification may induce foaming
- The oxygen demand of resuspended anaerobic mud may result in a decrease in oxygen concentrations, temporarily at least, that may kill fish

Kezar Lake in Sutton, New Hampshire, which is a recreational lake (surface area 182 acres; maximum depth 27 feet) began to experience severe and objectionable blue-green algal blooms in 1963 (New England Regional Commission 1971, 1973a, 1973b). Control of algal blooms with copper sulfate proved ineffective. Artificial destratification by diffused aeration was tried as a means of alleviating the aesthetic problems created by the algal blooms. Subsequently, a series of tertiary waste treatment steps, including phosphorus removal, were instituted and artificial destratification was continued. The lake restoration scheme was deemed successful as aeration achieved the stated objective of increased secchi disc transparencies of 4 feet or more during summer months. The lake was isothermal and the dissolved oxygen was homogeneous throughout the water column. There was a rapid decrease in the dominant blue-green algal species of *Aphanizomenon.*

Hypolimnetic aeration was applied continuously to Hemlock Lake in Michigan from June 14 to September 7, 1970 (Fast et al., 1973). Hemlock Lake is a eutrophic lake with a surface area of 6 acres and a maximum depth of 61 feet. Before artificial aeration, hypolimnion oxygen concentrations were typically zero below thermocline. Phytoplankton populations usually limited the secchi disc transparencies. After initiating aeration, there was an initial increase in phytoplankton cell numbers. However, the standing crop decreased subsequently from 30,000 cells/ml to less than 500 cells/ml. Concomitantly, secchi disc measurements increased to over 30 feet, the deepest ever recorded for the lake. Following aeration, zooplankton inhabited the deeper lake waters and their numbers increased until predation stress by fish caused zooplankton numbers to decline. The total number of zoobenthos increased although the biomass remained the same.

The U.S. Environmental Protection Agency (1973) compiled cost data for artificial destratification. The minimum, mean, and maximum initial costs (purchase and installation) were reported as \$0.05, \$4.39, and \$19.70 per acre-foot, respectively. It was reported that 90 percent of the respondents utilized compressed air devices for aeration, and 4 percent used mechanical pumps. One-third of the respondents employed continuous operation, one-third continuous during the summer, and one-third intermittent operation. The operating costs varied from a maximum of \$4.62 to a minimum of \$0.003/acre-foot/year, with a mean of \$0.94/acre-foot/year. The initial cost and the operating cost per unit volume declined as the volume of the reservoir increased. No clear trend emerged with regard to the costs of the type of equipment or the operating schedule employed.

Sediment Exposure and Desiccation. Water level manipulation has been employed as a mechanism for enhancing the quality of certain lakes and reservoirs. The exposure of lake bottom mud to the atmosphere reduces sediment oxygen demand and increases the oxidation state of the mud surface. This procedure may retard the movement of nutrients from the sediments to the overlying water when flooded once again. Sediment exposure can also curb sediment nutrient release by physically stabilizing the upper flocculant zone of the sediments. Lake drawdown has been investigated as a control measure for submerged rooted aquatic vegetation, and as a mechanism for lake deepening through sediment consolidation. Because this method does not hold any promise in the Fox Chain of Lakes, it is not discussed further.

Harvesting Nuisance Organisms. Harvesting of nuisance organisms is limited to macrophytes and some undesirable fish. Technical difficulties have precluded in-lake harvesting of algal cells. The technique has been advocated as a practical means of accelerating the nutrient outflow from lake systems. However, this technique alone is deemed inadequate for lowering nutrient supplies in lakes receiving cultural enrichment (Neel et al., 1973). Removal of weeds and fish may hasten nutrient depletion after elimination of extraneous nutrient influx.

Mechanical control deals with harvesting and removal of aquatic plants from the water.

Mechanical removal of plants allows immediate use of the harvested area and the plants removed from the water are not available to deplete dissolved oxygen resources and release nutrients for new plant growth. According to a survey of mechanical harvesting used in 32 locations in the upper Midwest, information on the acreage harvested, including the costs, was not reliable. The high initial investment in machinery was the most serious obstacle to implementing a management program. Costs for weed control by mechanical harvesting were reported to vary from about $15 to $75 per acre.

Chemical Control of Nuisance Organisms. Nuisance algal blooms, dense growth of macrophytes, and unbalanced fish populations often restrict various recreational and domestic uses of surface waters. Chemical treatment has been most widely used as a treatment method. Chemical treatment has the greatest utility and justification in highly eutrophic lakes in which the nutrient supply cannot be effectively controlled or in which nutrient input control measures are envisaged sometime in the future. Based on the intent, chemical controls can be divided into three categories: 1) algicides, 2) herbicides, and 3) piscicides. Because algal blooms are the most severe problem in the Fox Chain of Lakes, only algicides will be dealt with here.

Copper sulfate is probably the most widely used chemical for control of blue-green algae, taste and odor producing algae, and some filter clogging algae. Over 10,000 tons of copper sulfate are used for this purpose each year in this country at concentrations ranging from less than 0.5 mg/l to more than 10.0 mg/l (Fitzgerald, 1971). The amounts of oxygen, organic matter, and alkalinity in the water determine the dosages required for effective plankton control (Fitzgerald, 1971; Mackenthun, 1969; and Mulligan, 1969). For waters with alkalinity greater than 40 mg/l, copper sulfate at a rate of 1 mg/l for the upper 2 feet of water regardless of actual depth has been widely used (Mackenthun, 1969). On an acreage basis, the concentration would amount to 5.4 lbs/surface acre. For lakes with alkalinity less than 40 mg/l, a concentration of 0.3 mg/l of copper sulfate amounting to 0.9 lbs/acre has been suggested. The difference is mainly due to the fact that the effectiveness of copper sulfate is reduced in high alkalinity waters because of the formation of an insoluble precipitate of copper basic carbonate. Chelated copper sulfate (cutrine), which does not precipitate as readily as copper sulfate, has been employed successfully in controlling noxious algal blooms (Dunst et al., 1974).

Copper sulfate has a low mammalian toxicity, is inexpensive, and is effective in controlling a wide range of plankton algae. However, instances of fish kills have been reported soon after copper sulfate applications. These were generally traced to improper application and excessive dosage rates. The toxic effect of copper sulfate on plants is caused by the inactivation of enzymes and precipitation of proteins by the divalent ion Cu^{2+}.

Copper sulfate may be applied in a variety of ways: bag dragging, dry feeding behind power boats, liquid spray, or airplane application of either dry or wet material. Application by blowing the chemical rather than by slurry has also been employed (Mackenthun, 1969). The advantage of the blower-type machine is the ability to treat a large surface area rapidly with a light dosing of material. Use of blower-type machines is dependent upon the wind for distribution of the chemical. However, there is always some loss of copper sulfate dust that is carried by wind to the shore of the lake. Helicopters have also been used in chemical distribution. The East Bay Water Company, Oakland, California, found that a more efficient treatment could be attained with a helicopter (Mackenthun, 1969).

Chemical algal control measures should be undertaken before the maximum development of algal blooms. Mackenthun (1969) suggests that it is a good practice to subdivide the total area into sections and control the nuisance in one section followed by treatment of other sections at intervals of 7 to 10 days. This procedure will ensure that sufficient dissolved oxygen is present to satisfy the demands of the decomposing algae.

The frequency of copper sulfate application varies from a single annual application to monthly applications during spring and summer. The continuous feed of copper sulfate to the inlet of a reservoir has also been reported (Muchmore, 1973).

The toxicity of copper sulfate toward humans presents no problem at the concentration levels normally used in lakes and reservoirs. Of concern, in long-term treatment of water supplies with copper sulfate, is the potential of accumulating harmful amounts of copper in the bottom sediments (Muchmore, 1973). The copper added as copper sulfate will end up in bottom sediments. Muchmore reports that a study of a group of Wisconsin lakes where copper in bottom muds of reservoirs that had been routinely treated with copper sulfate was considerably lower in concentration than the 9000 ppm (dry basis) they found necessary to affect bottom dwelling organisms. No difference in the diversity of benthal populations could be attributed to the presence of copper.

Cost per acre for chemicals and application have been reported by the Water Survey as about $2.00 in 1966. Dunst et al. (1974) reported that, for Mascoma Lake (1110 acres) in New Hampshire, the cost of application of copper sulfate including chemical cost amounted to about $2.60 per acre. The Southeastern Wisconsin Regional Planning Commission (1969) used the following cost figures: cost of chemicals (copper sulfate) at $1 per acre treated; a boat or barge and spraying apparatus at an initial cost of $1250; and operation and maintenance costs of $50 per day.

Other algicides of some use are the rosin amines, triazine derivatives, mixture of copper sulfate and silver nitrate, quarternary ammonium compounds, organic acids, aldehydes, ketones, etc. Prows and McIlhenny (1973, 1974) reported after examining more than 10,000 compounds that p-chlorophenyl-2-thienyl iodonium chloride is an effective chemical for algal control. Based on laboratory tests and limited field evaluations, the authors concluded that the compound is safe to applicators, fish, and other higher aquatic plants and animals; it has a fairly rapid degradation pattern under open atmospheric conditions with a half life of 1 to 2 days; and it exhibits a high degree of specificity to nuisance algae, particularly *Anabaena, Microcystis, Aphanizomenon,* and *Oscillatoria.* It must be pointed out that none of these algicides has been used as extensively as copper sulfate.

Biological Control of Nuisance Organisms. This approach encompasses the introduction or promotion of organisms that are inimical to the target organisms. Dense growths of aquatic macrophytes were found to inhibit the growth of phytoplankton, both by direct competition for nutrients and by shading. One of the natural ways in which algal populations are kept under control is through predation by zooplankton and fish species. Effective grazing by *Daphnia* and related zooplankton on phytoplankton populations in a mesotrophic lake has been reported (U.S. Environmental Protection Agency, 1973). Dunst et al. (1974) reported that suitable plankton feeding fish species are *Tilapia mossambica,* and its allies, *Hypophthalmichthys molitriz* and *Mugil cephalus.*

Dunst et al. (1974) reported about the only deliberate in-lake treatment to control blue-green algae by the use of virus. Blue-green algal scums were apparently dissolved as a result of spraying cyanophages on the surface of a lake in the U.S.S.R. Evaluation of biological controls has been limited, with much of the testing conducted in laboratory and experimental ponds. In general, biological control measures have met with only very limited success.

Water Quality Management Plan

The lakes of the Fox Chain have a common glacial origin, though their shapes, sizes, depths, and volumes are quite different. Control techniques must be equally diverse. There is, however, a common thread inherent in all water quality restoration schemes for lakes. It is the fundamental need to minimize the input of critical nutrients to the lake system.

Carbon, nitrogen, phosphorus, sulfur, iron, silica, and other elements in trace amounts are essential for algal growth. The concensus among limnologists is that nitrogen and phosphorus are the critical elements. As reported here earlier, both of these elements are growth limiting for the waters of the Chain. To minimize one of the elements will have the effect of minimizing both. The techniques for removing phosphorus are more advanced, predictable, and effective than those thus far developed for removing nitrogen. It makes sense therefore to undertake a major effort to reduce the phosphorus burden from controllable sources now being imposed upon the lake system.

As goals for phosphorus reduction, the findings of Sawyer (1952) and Vollenweider (1968) are useful. Sawyer, as mentioned earlier, concluded that nuisance algal blooms are likely to develop in lakes during summer months when concentrations of inorganic nitrogen and inorganic phosphorus are in excess of 0.3 and 0.01 mg/l, respectively. Vollenweider concluded that loading rates of biochemically active nitrogen and phosphorus in excess of 2.0 and 0.13 $g/m^2/yr$, respectively, are critical. The attainment of these goals in the Fox Chain of Lakes is unlikely. To do so would require the estimated total phosphorus load of 10.4 $g/m^2/yr$ (1675 lbs/day) now applied to the lake system to be reduced to about 0.13 $g/m^2/yr$ (21 lbs/day).

About 1190 lbs/day of total phosphorus, an estimated 69 percent of the total, originates in Wisconsin. The Southeastern Wisconsin Regional Planning Commission envisages a water quality management plan for the Fox River basin in Wisconsin. Phosphorus removal facilities for municipal and industrial waste discharges are anticipated. Control measures applicable to soil conservation practices for the management of nonpoint sources are contemplated. Any long-term restoration scheme instituted in Illinois without coordinated activity in Wisconsin can have, at best, only temporary and marginal impact in remedying eutrophic conditions in the lakes.

Of the estimated 485 lbs/day total phosphorus emanating within Illinois (excluding precipitation as a source) about 295 lbs/day comes from point sources. Of the 190 lbs/day derived from nonpoint sources, about 55 lbs/day comes from malfunctioning septic tank systems.

The complete removal of phosphorus from the lake system now imposed by Antioch, Lake Villa, Round Lake Sanitary District, and Fox Lake would provide a reduction of 130 lbs/day. A collection system for the septic tank areas would remove an additional 55 lbs/day.

The upgrading of the treatment facilities at Hebron, Richmond, and Woodstock would eliminate phosphorus emission from the Hebron facility, and would provide about a 95 percent reduction at the other two sites. This would mean an overall reduction of an additional 145 lbs/day total phosphorus.

An effective wastewater management scheme founded on current technology could reduce the point sources of phosphorus in Illinois from 295 to 20 lbs/day, as well as reduce the nonpoint sources of 190 to 135 lbs/day. This is an overall reduction of about 93 percent for the point source load and about 29 percent for the nonpoint source load.

This demonstrates the importance of instituting a program for controlling nonpoint sources on the watershed within Illinois. About 115 of the 135 lbs/day of phosphorus comes into the lake system by way of Nippersink and Squaw Creeks. The remedial measures required for nonpoint source reductions on these two streams would differ. The watershed of Nippersink Creek is principally rural; the Squaw Creek watershed is principally urban.

Another complicating factor is that Wonder Lake and Long Lake intercept Nippersink and Squaw Creeks, respectively, before the streams discharge into the Fox Chain system. Both of the lakes have been classified by the U.S. Environmental Protection Agency (1975b and 1975c) as highly eutrophic. The influence of these lakes is significant to the phosphorus transport of their respective outlet streams.

Pistakee Bay and the upper lakes in the Fox Chain system (Catherine, Channel, Marie, Bluff and Petite) experience total oxygen depletion below the thermocline during summer months because of their greater depths. The dominant algal species in these lakes are the scum forming blue-greens. Also the benthic population in these lakes is sparse and tolerant of extremely low oxygen concentrations. Destratification and circulation in these lakes is desirable. Oxygenation of the hypolimnetic waters will likely produce a shift in the dominant algal species from the blue-greens to the less objectionable greens. The development of increased and desirable benthic populations, leading to increased fish population, can also be anticipated. The estimated initial and operating costs for destratification in these lakes by the use of air are shown in table 58. Average initial and operating costs of $4.39 and $0.94/acre-foot/yr were used in developing these costs. Destratification in these lakes should be undertaken as soon as practicable.

Table 58. Cost Estimates for Lake Destratification

(Cost estimates in dollars)

Lake	*Initial cost*	*Annual operating cost*
Catherine	10,800	2300
Channel	19,200	4100
Marie	20,800	4500
Bluff	4,200	900
Petite	5,600	1200
Pistakee Bay	10,800	2300

Note: Average initiai and operating costs of $4.39 and $0.94 per acre-foot per year, respectively, were used to estimate costs

As pointed out earlier, the recycling of phosphorus from the organic rich bottom sediments in the deep lakes is a significant source with which to reckon. The rates of phosphorus release from sediments under aerobic conditions are substantially less than release rates under anaerobic conditions. The creation of aerobic conditions by hypolimnetic aeration will lessen the impact of bottom sediments as an influential source of phosphorus. There are claims that aeration will also reduce the volume of organically enriched sediments through means of oxidation.

Anoxic conditions did not develop in the waters of the shallow lakes like Fox, Grass, Nippersink, and Pistakee Lakes, except at the mud-water interfaces. An appropriate restoration scheme for these lakes is a combination of nutrient inactivation and chemical control. The predominant species of algae in Grass Lake are diatoms. Nuisance algal blooms have not developed in this lake. Other than being turbid the water quality of Grass Lake is not degraded and a program for water quality restoration is not justified at this time.

The other three shallow lakes (Fox, Nippersink, and Pistakee) need remedial measures. Because of the uncertainties involved in the response of lakes to chemical treatment, it is proposed that demonstration schemes be initiated before embarking on full-scale measures for these three lakes. On the basis of the results of the pilot schemes, an appropriate large-scale program can be devised.

Mineola and Stanton Bays in the Fox Lake appear to be suitable for the pilot investigations. These waters are fairly well isolated from the main lake and are of such size that extrapolation of results obtained in them will be meaningful and readily applicable to the rest of the shallow lakes.

Chemical control of algae using cutrine (chelated copper sulfate) is recommended for Mineola Bay which has an area of about 205 acres. The chemicals can be applied in about 5 hours. For six applications during the period of May through October, the cost for equipment, chemicals, and application amounts to about $3000.

Nutrient inactivation by alum precipitation is proposed for Stanton Bay. Because of the high powered, high speed boating activites in the lakes and the shallowness of the bay (3 feet average depth) aluminum floe may remain in suspension. The effectiveness and longevity of this treatment will have to be carefully monitored. The cost of treating 95 acres of Stanton Bay is estimated to be $8500.

Whether or not the use of hypolimnetic aeration will minimize the influence of the bottom muds on the overlying waters will have to await trials. The alternative is to physically remove the sediments by dredging. The estimated costs of dredging the pertinent lakes are shown in table 59. These costs are predicated on dredging bottom areas of the lakes. The areal extent of sediment removal is to be confined to that portion of the bottom muds designated as 'algal silt' in figure 2. The depth of dredging is to be limited to the top 2 feet of the sediments. Costs were developed from an average rate of $0.50 per cubic yard.

Table 59. Cost Estimates of Dredging Organic Sediments

Lake	*Area of proposed dredging (acres)*	*Cost ($)*
Catherine	70	113,000
Channel	180	291,000
Marie	215	347,000
Bluff	40	65,000
Petite	95	153,000
Pistakee Bay	60	97,000

Note: Dredging costs per acre were developed by use of an average rate of $0.50 per cubic yard

SUMMARY

The results of an extensive technical program conducted jointly by the Illinois State Water Survey and Geological Survey on the Fox Chain of Lakes are presented in this report. This effort has been directed toward ascertaining the causes of lake eutrophication and the means by which these causes may be halted, reversed, or at least minimized. Geographic, geologic, hydrological, physical, chemical, and biological characteristics of the Fox Chain of Lakes have been employed in identifying the causes of eutrophic conditions of the lakes. The trophic condition of the lakes is discussed as related to their limnological characteristics and nutrient budget. The nutrients responsible for the accelerated aging of the lakes are identified, and the sources of the nutrients and proportional contribution of each source are assessed. The salient features of this investigation are summarized here.

Only about 27 percent of the Fox Chain of Lakes drainage area as measured at Johnsburg lies within Illinois. The Chain's water surface area is approximately 6850 acres and the lakes have a mean depth of about 5.7 feet at the normal pool elevation 736.5 feet above msl.

In Illinois there are seven municipal waste treatment plants serving about 37,800 persons located within the watershed. The population equivalent served by septic tank-field disposal systems in the vicinity of the Fox Chain of Lakes is estimated to be 22,250. In Wisconsin, there are 12 sewage treatment plants serving about 64,050 persons (1966 population) in the watershed. The Northeastern Illinois Planning Commission and the Southeastern Wisconsin Regional Planning Commission have developed plans to upgrade the waste treatment facilities including phosphorus removal in their respective regions.

The Fox River and other tributaries to the Fox Chain of Lakes are typical of midwestern streams — high in alkalinity, hardness, and total dissolved solids. The nitrate load transported by the tributaries ranged in value from 4.05 lbs/ac/yr for Nippersink Creek to 1.68 lbs/ac/yr for Lily Lake Drain. Total phosphorus load transported varied from 3.24 lbs/ac/yr for Sequoit Creek to 0.20 lbs/ac/yr for Lily Lake Drain.

The deep upper lakes of the Chain develop distinct summer stratification. These lakes become totally anoxic below the thermoclines. Marked increases of ammonia, phosphorus, silica, and other end results of anaerobic decomposition of bottom sediments were in evidence. The shallow lakes exhibited homogeneous thermal profiles. However, the dissolved oxygen concentrations near the lake bottoms showed a marked drop that presumably was caused by the high rate of oxygen demands of the bottom sediments.

Lakes with concentrations of 0.3 mg/l of inorganic nitrogen and 0.01 mg/l of available phosphorus at the time of spring overturn have been reported to experience algal blooms. The mean concentrations of total phosphorus varied from 0.08 to 0.30 mg/l in surface water samples and from 0.27 to 0.87 mg/l in deep water samples. Algal assay procedures were carried out to determine the effect of additional nitrogen and phosphorus in the surface water samples of all the lakes. Nitrogen or phosphorus added alone did not have any stimulatory effect on algal growth, but when added together to the water samples, the algal growth potential exhibited nearly 7- to 10-fold increase compared with the unspiked water samples.

Tributaries, direct discharge of waste treatment plant effluent, septic tank systems in the vicinity of the lakes, and precipitation are the major sources of nutrients for the Fox Chain of Lakes. The total phosphorus and total nitrogen loading rates were, respectively, 10.3 and 86.0 g/m^2/yr. The critical loading rates for lakes with mean depths less than 5 meters (16.4 feet) suggested in the literature, are 0.07 and 1.0 $g/m^2/yr$, respectively, for phosphorus and nitrogen. The total phosphorus and total nitrogen loading from the Fox River alone was estimated as 7.1 and 65.0 $g/m^2/yr$, respectively. About 26 percent of total nitrogen and 37 percent of total phosphorus transported to the lakes are retained within the lake system.

Approximately 69 percent of the external phosphorus input to the lakes is brought by the Fox River. Nippersink Creek accounts for 13 percent. Precipitation and the impact of malfunctioning septic tank disposal systems account for 3 percent, each, of the total phosphorus input.

The phosphorus reaching the lake waters from nonpoint sources within Illinois and the recycling of phosphorus from the organic rich bottom sediments to the overlying waters are each of sufficient magnitude to cause algal growth problems in the lakes.

Controlling the nutrient input to the lakes from all sources is of paramount importance for reversing the eutrophic trend in the Fox Chain. Several regional plans for nutrient reduction have been proposed for Illinois and Wisconsin. Counter-proposals have been offered. The time schedule for these plans are not clear cut and it is likely that several years will elapse before these schemes can be implemented. In the interim, it is proposed that Lake Catherine be destratified and results evaluated before this technique could be applied to other deep lakes in the system. Channel Lake could serve initially as a control. The total initial costs of equipment and accessories are estimated to be $10,800 and the annual operation costs are estimated to be $2300.

The control of phosphorus recycling from the organic rich sediments of deep lakes by dredging can await the results of destratification. The cost of dredging the top 2 feet of the sediments rich in organic content in the deep lakes is estimated to be $1,066,000.

Two pilot scale demonstration schemes are proposed in order to develop a reliable interim restoration scheme for the shallow lakes. The effects of the control of algal blooms by the application of chelated copper sulfate to the waters in Mineola Bay (Fox Lake) at a cost of about $3000 is recommended. Nutrient inactivation by the application of alum is recommended for Stanton Bay (Fox Lake).

Artificial destratification and the chemical controls proposed here are designed to offer some immediate relief from the consequences of eutrophication. The ultimate success in reversing the eutrophic trends in the Fox Chain of Lakes will depend on the drastic reduction of the nutrient loading to the lake system both from without and within the lake area.

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