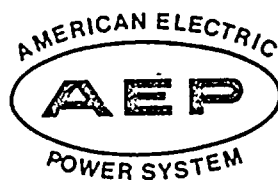


Annual Environmental Operating Report

January 1, Through December 31, 1986

Indiana & Michigan Electric Company
Bridgman, Michigan

Docket Nos. 50-315 & 50-316
License Nos. DPR-58 & DPR-74



Addendum

title p

GREAT LAKES RESEARCH DIVISION / PUBLICATION 22

SOUTHEASTERN NEARSHORE LAKE MICHIGAN:

IMPACT OF THE DONALD C. COOK NUCLEAR PLANT

Edited by

Ronald Rossmann

GREAT LAKES RESEARCH DIVISION
The University of Michigan
Ann Arbor, Michigan 48109

A FINAL REPORT TO INDIANA & MICHIGAN ELECTRIC COMPANY

1986



DEDICATION

This book is dedicated to the memory of Dr. John C. Ayers, oceanographer of the Great Lakes, who served as Project Director from the project's inception until his retirement. He expertly guided the project using his patience, understanding, and sensitivity to his fellow scientists. He not only guided the project but he also was an active participant in sample collection, data interpretation, and reporting of results. His wide-ranging expertise was utilized for all subtopics of the project. All who participated in the project received a special gift by their association with Dr. Ayers.

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ACKNOWLEDGMENTS

The authors thank all those who participated in this long-term study of a coastal region of the Great Lakes. Thanks to R. Stephen Schneider for his technical editing of this book and to Marion Luckhardt and Beverly McClellan for word processing services. Special thanks to all those not associated with Great Lakes Research Division who served as external reviewers of one or more chapters of the book. All provided constructive comments that improved the quality and presentation of materials within the book. Thanks to Indiana & Michigan Electric Company for supporting this work.

INTRODUCTION

Ronald Rossmann

The Laurentian Great Lakes serve as a source of cooling water for both fossil- and nuclear-fueled power plants. In the early 1960s, projected heavy utilization of this freshwater resource by power plants raised concerns about the environmental impact of power plants on the lakes. Such public, governmental, and industrial concerns led to the federal requirement that environmental impact studies be conducted at all proposed or newly constructed power plants, especially nuclear power plants. The Great Lakes Research Division of The University of Michigan was contracted by Indiana & Michigan Electric Company to conduct an environmental impact study of the Donald C. Cook Nuclear Plant, investigating the area through the pre-operational years of construction and into the period of plant operation.

DONALD C. COOK NUCLEAR PLANT

The Donald C. Cook Nuclear Plant is a 2,200 MW steam electric generating station situated in a belt of sand dunes on the southeastern shore of Lake Michigan about 18 km southwest of St. Joseph, Michigan (Indiana & Michigan Electric Company and Indiana & Michigan Power Company 1973) (Fig. 1). Preparation of the site for construction of the plant began in 1968. Unit 1 was operational in 1975 and Unit 2 in 1978. The plant was built at a time when public concern for the environment reached its zenith.

At full operation, the plant uses roughly 6,300 m³ of lake water per minute in once-through cooling of its condensers; the water returned to the lake is 9-120° above the intake temperature. Water is withdrawn from the lake through three intake cribs located at a water depth of 7.3 m approximately 686 m offshore (Fig. 2). The intake structures extend 2 m above the lake bottom. Maximum water velocity is 1.8 m/s within the three 4.9-m diameter intake pipes. Condenser cooling water is returned to the lake through two discharge pipes located at a water depth of 5.5 m approximately 366 m offshore. Riprap beds of crushed limestone were deposited around the intake and discharge structures to prevent erosion and scour of the lake bottom. During winter, water is only drawn into the plant through the two outer intake cribs. Heated water is discharged through the center intake, which is turned into a discharge to prevent ice formation around the other intake structures.

Within the screenhouse, trash bars 6.6 cm apart prevent large pieces of debris from entering the intake forebay. Vertical traveling 9.5-mm bar mesh screens within the forebay impinge trash and fish too large to pass through the condensers.

HISTORY OF THE ENVIRONMENTAL IMPACT STUDY

Though the Great Lakes Research Division's environmental impact study included physical and chemical studies of nearshore southeastern Lake Michigan, it focused

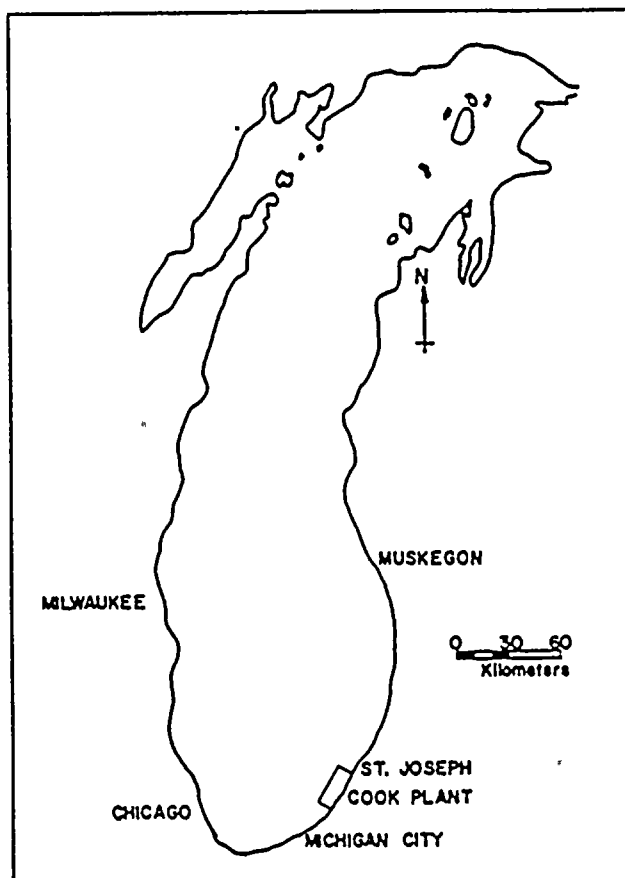


FIGURE 1. Location of the Donald C. Cook Nuclear Plant on the southeastern shoreline of Lake Michigan.

on major biological components of the ecosystem (benthos, phytoplankton, zooplankton, fish) and shoreline erosion. Preoperational surveys began in 1966. Until 1970, they were sporadic and consisted mainly of phytoplankton enumerations and some benthos and zooplankton enumerations. Between 1970 and 1972, seasonal surveys, primarily of phytoplankton and secondarily of zooplankton and benthos, were conducted. Beginning in 1972 and continuing into 1982, sampling frequency and intensity increased and equal effort was given to studies of phytoplankton, zooplankton, and benthos. Fishery studies began in 1972. They represented roughly 50% of the study effort between 1973 and 1982.

Until 1974, all studies were carried out on nearshore southeastern Lake Michigan. In 1974, preliminary studies of the entrainment of organisms through the plant and their impingement on the plant's traveling screens began. When unit 1 of the plant became operational in 1975, full-scale entrainment studies of phytoplankton, zoo-

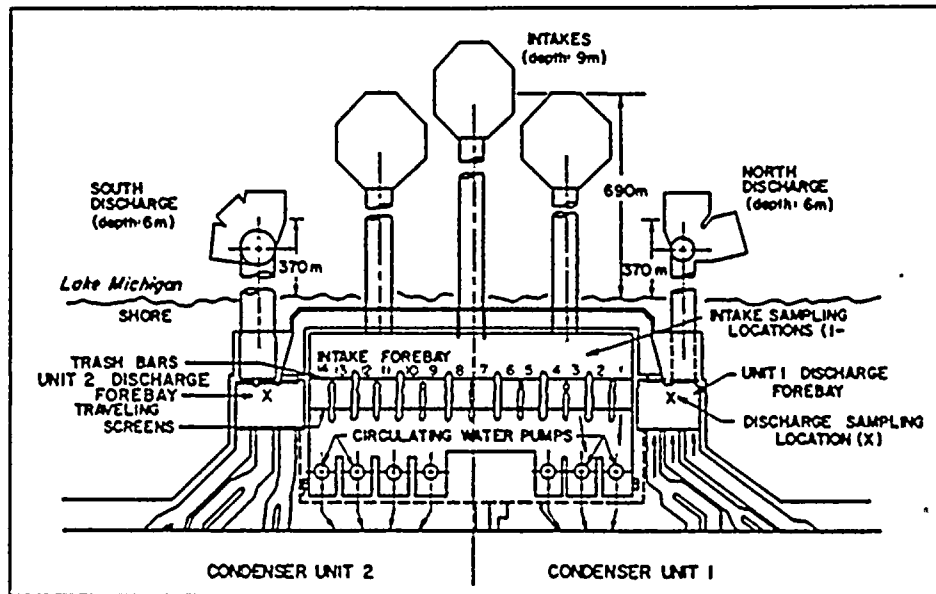


FIGURE 2. Schematic representation of the Donald C. Cook Nuclear Plant circulating water system.

plankton, benthos, fish eggs, and fish larvae began. At the same time, impingement of fish and benthos on the traveling screens was monitored. All field studies ceased in 1982. This book represents the highlights of the findings from the impact study. For details of each facet of the study, the reader is referred to the list of all project reports (Table 1). These are available upon request.

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This book has been structured so that each chapter dealing with a specific facet of the study will be complete in itself, and can be read as a separate entity by those who wish. To familiarize the reader with the region of southeastern Lake Michigan where the plant is situated, a description of the area's characteristics follows.

DESCRIPTION OF THE SOUTHEASTERN LAKE MICHIGAN REGION

LAKE BOTTOM TOPOGRAPHY

The topography of the lake bottom within the region of study slopes toward the northwest with greatest depths in excess of 38 m (Fig. 3). At depths greater than 20 m, a series of ridges and runnels perpendicular to shore occurs (Rossmann 1975).

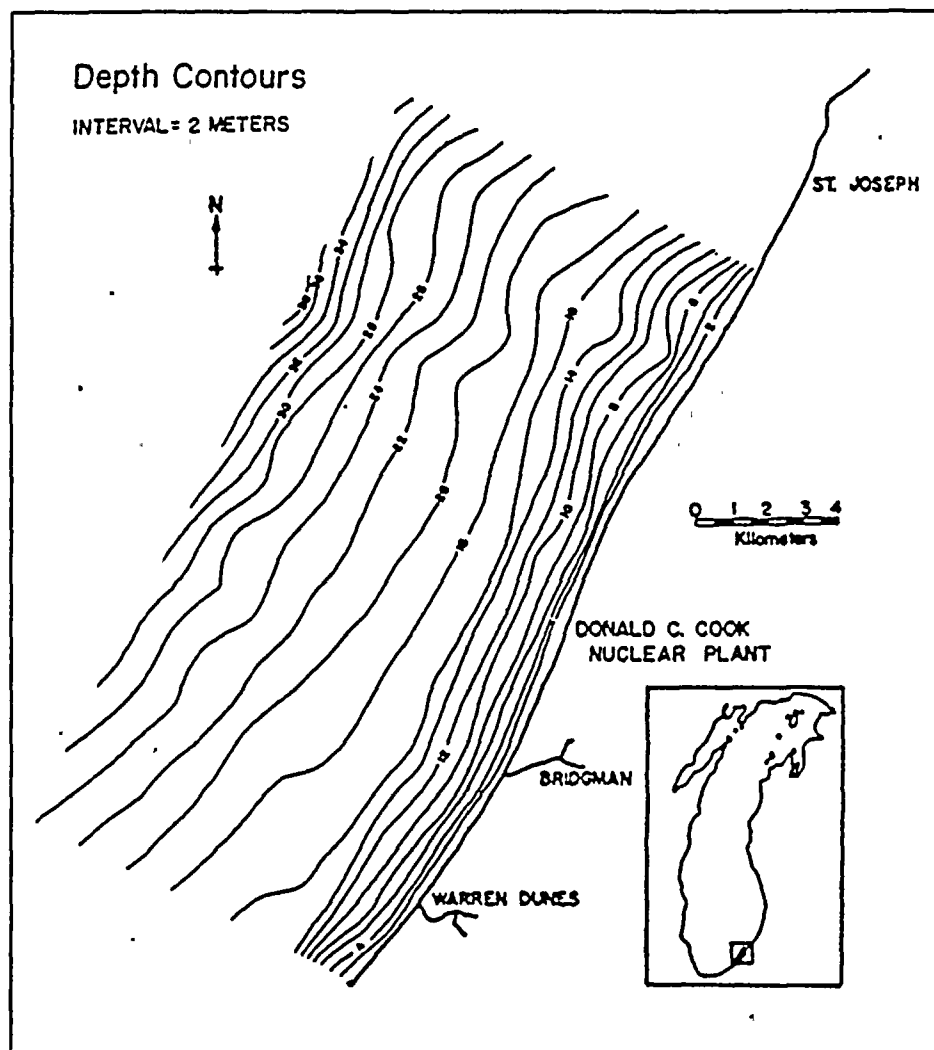


FIGURE 3. Bathymetry of nearshore southeastern Lake Michigan.

The ridges consist of coarser sediments than the runnels (Rossmann and Seibel 1977).

Near the shoreline, one ephemeral and two permanent sand bars exist (Seibel et al. 1976). The ephemeral bar is located approximately 47 m from the shoreline. The inner and outer permanent bars are located 116 and 216 m from the shoreline, respectively. Water depths between the inner bar and shoreline and between the two permanent bars are 1.5 to 1.8 m and 3.7 to 4.0 m, respectively (Ayers and Huang

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1967). The depth of water over the crest of the inner bar is 1.2 m and that over the outer bar is 2.6 m.

SEDIMENTS

Sediments vary from coarse to fine quartz sands at the shoreline to carbonate-rich very fine sands and silts offshore (Rossmann and Seibel 1977). The sediments are derived from sand dunes and unconsolidated bluffs along the shoreline (Hands 1970). The bluffs are clay-rich and contain carbonates (Rossmann 1975). During the early to mid-1970s, severe shoreline erosion occurred within the study area (Davis et al. 1973, Tanner 1975). Material derived from this erosion as well as from rain runoff and stream transport is resuspended and sorted by waves and currents (Sly and Thomas 1974). Materials are finally deposited offshore where waves and currents are incapable of transporting them farther.

CURRENTS

The major lake current within the region of Lake Michigan is called the Benton Harbor eddy. This eddy moves in a clockwise direction with currents moving southwestward along the shoreline (Ayers et al. 1958, Bellaire 1964). Inshore of this eddy, a counterclockwise eddy with currents moving northeastward adjacent to the shoreline was tentatively identified by Ayers et al. (1958).

Between May and November, current direction between the inner and outer sand bars is predominantly to the northwest. Direction was always to the north, northwest, or west between May and July regardless of wind direction (Ayers et al. 1967). Beginning in August when the wind was brisk from northerly directions, currents moving in a southward direction began to occur with increased frequency. Smaller velocity and frequency of northerly winds were required to produce southward currents later in the fall. For the months of May through October in 1967, over 86% of the time current direction between the two bars was northward. Monthly mean current velocities ranged between 31.7 and 69.8 cm/sec.

Inshore of the inner sand bar, current direction is controlled by wind direction. Water between the beach and inner bar is postulated to be water spilled over the inner bar during surf action. It is cut off from the main-lake circulation by the inner bar and is moved in a downwind direction (Ayers et al. 1967).

WAVES

Wave observations have been reported by Liu and Housley (1969) for St. Joseph, Michigan (Fig. 1). For the periods of 28 September 1966 to 16 December 1966 and 21 September 1967 to 30 December 1967, maximum-observed wave height was between 3.6 and 3.0 m. Maximum wave period was between 7 and 8 seconds. The most frequent wave height and period were 0.6 to 1.3 m and 5 to 6 seconds, respectively. Waves most often approached the shoreline from the northwest or southwest.

LOCAL WINDS

Based upon observations during 1967, surface winds adjacent to the shoreline are greatly influenced by a bluff of sand dunes 61 m from the shoreline that parallel the shore and run in a north-northeast to south-southwest direction (Ayers et al. 1967). These dunes serve to protect nearshore waters from winds from north-northeast, around through east and south, to south-southwest. They also channel winds from northerly and southerly directions into a NNE-SSW direction. Between May and mid-November the most often recorded winds were from the NNE (14.2% of the time) and SSW (22.6% of the time). Average wind velocity ranged from 9 km/h in June to 16 km/h in November. General prevailing winds for the region are from westerly directions throughout the year.

THERMAL CYCLE OF WATER COLUMN

The exact time of occurrence of various events in the thermal cycle is dependent upon the rate at which the lake warms as well as the severity of a winter. Warming of nearshore waters begins in spring with snowmelt and spring rains. As lake waters warm, a nearshore thermal bar develops which is delineated by the 4°C isotherm. As the lake continues to warm, the thermal bar progresses farther offshore. The thermal bar was usually found within the study area during April.

Eventually, the thermal bar moves out of the study area and a thermally stratified water column develops. This usually occurs in May or very early June. Thermal stratification usually continues into and occasionally through November. For the calendar year, surface water temperatures range between 1.1 and 26.9°C, while those of the hypolimnion range between 5.0 and 6.5°C. During stratification, upwellings of hypolimnion water with temperatures as low as 6°C are common along the shoreline. Except during upwellings, water depths (<45 m) in the study area were only great enough to permit sampling hypolimnetic water at those stations farthest offshore.

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SEDIMENTS OF SOUTHEASTERN NEARSHORE LAKE MICHIGAN

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INTRODUCTION

During 1973 and 1975, 158 surficial sediment samples (0-3 cm) and seven cores were collected from a 22- by 10-km region of nearshore southeastern Lake Michigan (Fig. 1). In general, concentrations of metals, carbon, total phosphorus, silt, and clay increased in an offshore direction. Conversely, insoluble fraction of sediment decreased in an offshore direction. These changes were primarily controlled by the physical limnology of this dynamic region (Rossmann and Seibel 1977).

METHODS

COLLECTION

Samples were collected using a Ponar grab sampler and a gravity corer. Subsamples for chemical analysis and grain size analysis were removed from the top 2-3 cm of the grab samples. Cored sediments were extruded from the plastic core liner while in a vertical position and sectioned in 1-cm intervals to a depth of 10 cm, 2-cm intervals to 20 cm, and 5-cm intervals to the bottom of the core. The sections were subsampled for textural and chemical analysis, placed in plastic bags, and stored at ambient temperature until processed.

SEDIMENT TEXTURE

Wet samples weighing 15 grams were treated with 30% hydrogen peroxide while heated on a hot plate. This treatment was continued until addition of hydrogen peroxide resulted in little or no reaction. Treated samples were thoroughly stirred in a mechanical mixer and then washed through a 0.064-mm sieve to separate the silts and clays from the coarser fraction. The coarser fraction was oven dried at 95°C and sieved at 1-phi intervals using a mechanical sieve shaker. The finer fraction was placed in 1-L settling tubes and analyzed at 1-phi intervals using the pipette method. Sodium hexametaphosphate solution (5.5 g/L) was added to the settling tubes with the samples to prevent flocculation. When the particle size fractions were calculated, they were corrected for the weight of sodium hexametaphosphate used. The method followed was that of Royse (1970).

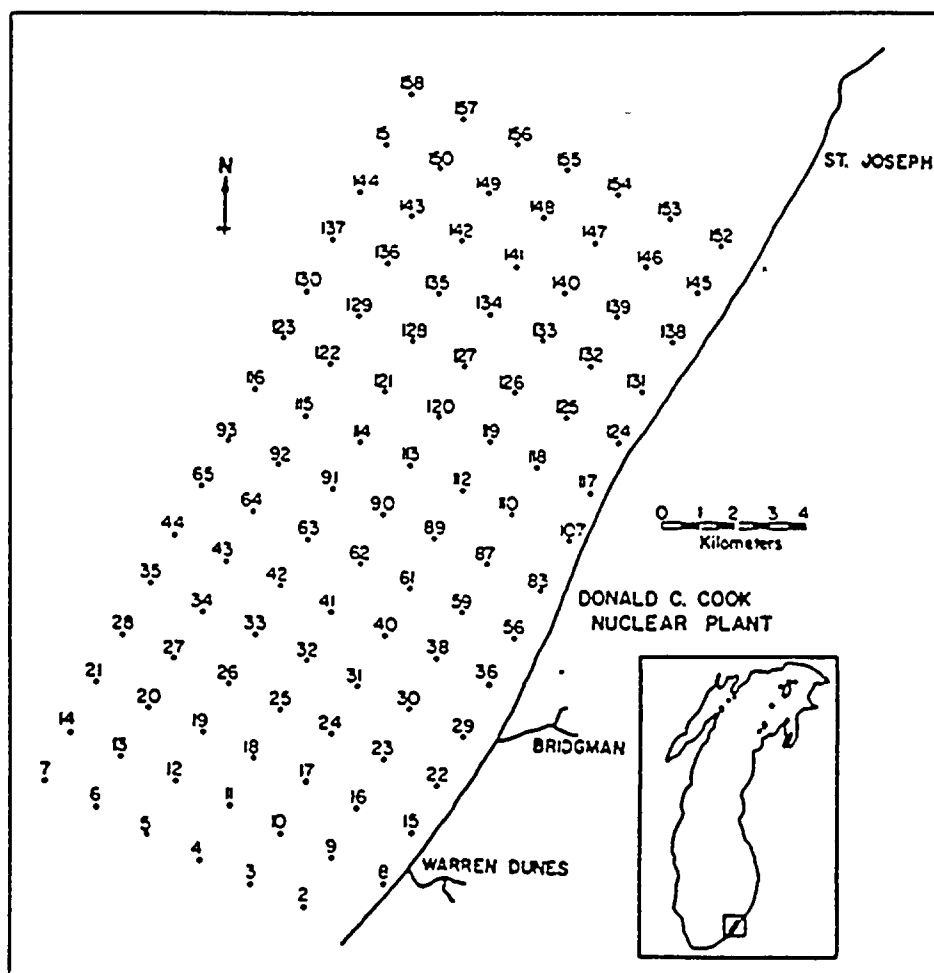


FIGURE 1. Southeastern nearshore Lake Michigan sediment sampling stations during 1973 and 1975.

SEDIMENT CHEMISTRY

Subsamples for chemical analysis were oven dried at 110°C for 24 hours, ground in a tungsten carbide mixer mill, and extracted in a 10% V/V HCl, with additions of 30% H_2O_2 , for 40 hours at approximately 95°C. The extracts were separated from the insoluble residue by filtration through pre-weighed fritted glass funnels. The residue was then dried and weighed to obtain weight percent insoluble. The filtered extract was brought to volume in a 50-mL volumetric flask and stored in polyethylene bottles. All elemental analyses were done by atomic absorption spectrophotometry (Perkin Elmer 1968). Phosphorus was done by extraction as molybdenum

heteropoly acids and measurement for molybdenum by atomic absorption spectrophotometry (Ramakrishna et al. 1969). Samples for loss on ignition were ignited in a muffle furnace at 1,000°C for a period of 1 hour. Total carbon analyses were done on oven-dried (110°C) ground samples gasometrically by hydroxide absorption using a LECO carbon analyzer. Inorganic carbon was measured using a modification of the LECO carbon analyzer system whereby the sample is reacted with hot 2N hydrochloric acid (Kolpack and Bell 1968). Organic carbon is considered to be equal to total carbon minus inorganic carbon.

TEXTURE

INTRODUCTION

The first comprehensive examination of Lake Michigan sediment texture was that done by Hough (1935). Subsequently, descriptive analyses of southern Lake Michigan sediments were provided by Hulsey (1962), McGeary (1964), Ayers and Hough (1964), Ayers (1967), Ayers and Huang (1967), Cote (1967), Gross et al. (1970), Lineback et al. (1970), Lineback et al. (1971), Lineback et al. (1972), and Lineback and Gross (1972). Results of full textural analyses of the sediment were presented by Somers and Josephson (1968), Callender (1969), Davis and McGeary (1965), Seibel et al. (1974), Rossmann and Seibel (1977), and Cahill (1981).

The sediments of southeastern nearshore Lake Michigan belong to the Waukegan Member of the Lake Michigan Formation described by Lineback et al. (1970). As described by them, the sediments vary from sand and gravel to soft, sandy silt or silty clay. Color ranges from brown or yellowish brown to its more characteristic dark gray. Its sediments become more sandy near shore. It is this Member in which trace elements are concentrated (Lineback et al. 1970).

AREAL TEXTURAL VARIATION OF SEDIMENTS

The textural variation of southeastern nearshore sediments is controlled by lake bottom topography, waves, and point sources of coarser sediments. The lake bottom slopes gently in a northwesterly offshore direction (Fig. 2). The maximum depth within the study area is slightly in excess of 38 m. Superimposed on this topography is the sorting action of waves and inputs of coarser materials from streams.

Variation in an Offshore Direction

The action of waves sorts and removes the fine-grained fraction from the nearshore sediments. This produces a textural pattern which is best illustrated by the distribution of mean grain size diameters (Fig. 3). Mean grain size decreases in an offshore direction. In general, fine sands grade into very fine sands which grade into silts furthest offshore.

Davis and McGeary (1965) described the sediments within this region of the lake within 7 km of shore to be coarse-to-medium sands with a grain size of approximately 2 phi. Cahill (1981) described sediments along the shore of southern Lake

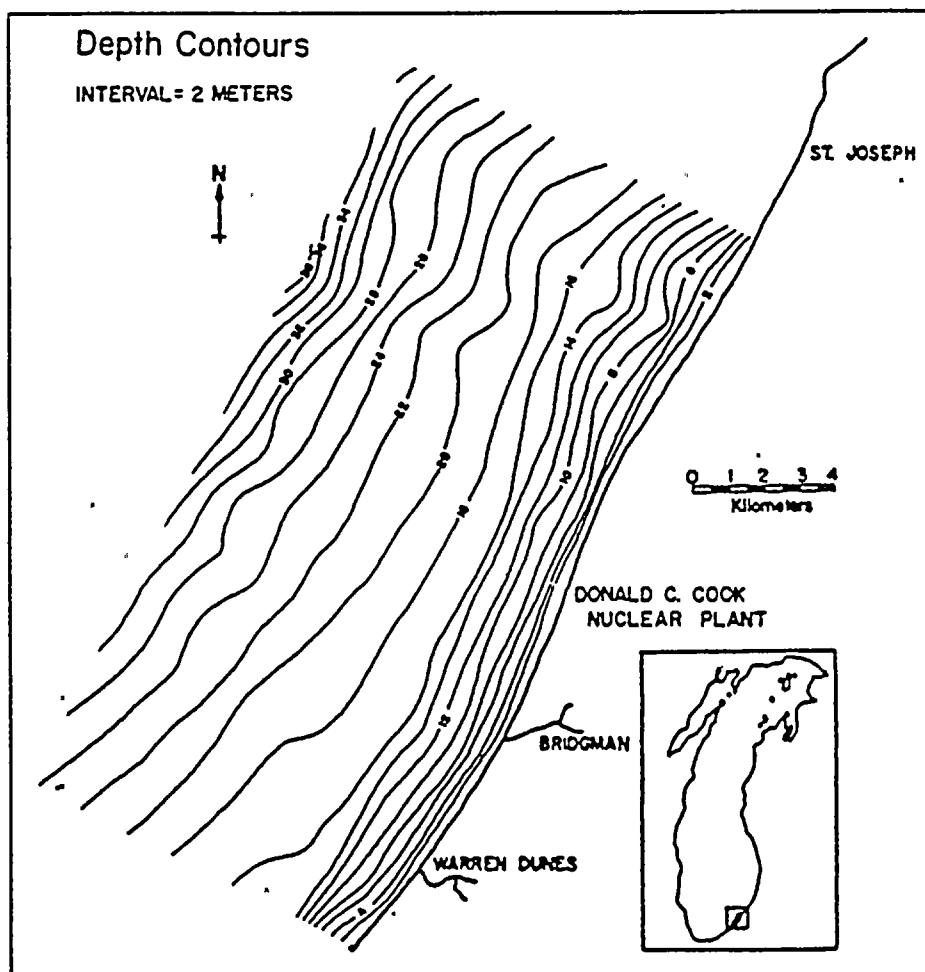


FIGURE 2. Bathymetry of nearshore southeastern Lake Michigan.

Michigan as coarse-to-fine sands with a mean size of 3 phi. The sediments beyond the outer bounds of this study (> 38 m water depth) have been described by Callender (1969) as silty clays having a median diameter of 6.2 phi, a standard deviation of the diameter of 2.44 phi, and a content of 25% sand, 37% silt, and 38% clay. Based upon a histogram of the mean grain size distribution, Cahill (1981) described the offshore sediments as very fine silts and clay-sized sediments with a mean size of 8 phi. Thus for southeastern Lake Michigan, sediments grade from fine sands (2-3 phi) nearest shore to very fine sands (3-4 phi) at depths of approximately 24-26 m to silts (4-8 phi) at depths greater than 28 m and finally to clays (> 8 phi) furthest offshore.

Based on cores collected from southeastern Lake Michigan, the sand content of

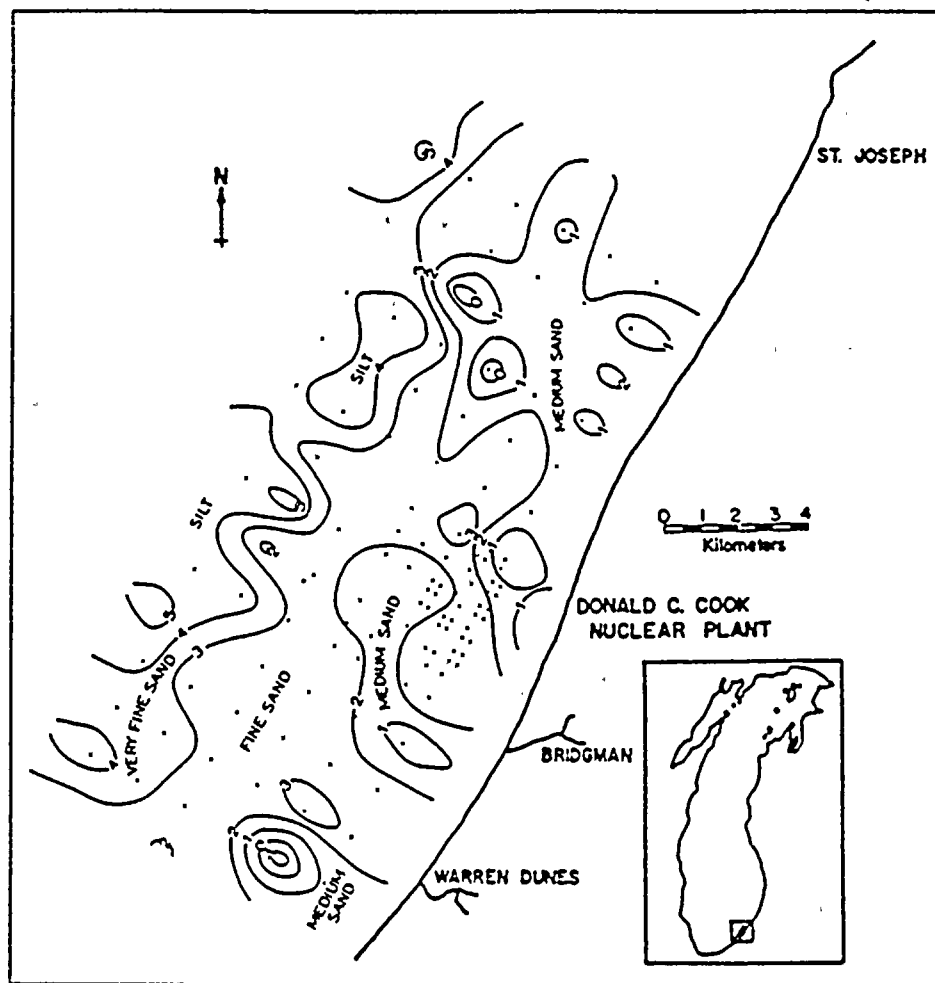


FIGURE 3. Mean grain size (phi units) of nearshore southeastern Lake Michigan surficial (0-3 cm) sediments.

the surficial sediments decreases with increasing distance offshore. Except for station 92 (Fig. 1), the surficial sediment sand content decreases in an offshore distance ($r = -0.948$, $p < 0.05$). This inverse relationship is not as strong when sand content is correlated with water depth ($r = -0.891$, $p < 0.05$). These results are in agreement with general lakeward decreasing grain size trends observed by others (Thomas et al. 1973, Rossmann and Seibel 1977, Thomas and Dell 1978, Cahill 1981) on a lake-wide basis in the Great Lakes and reflect the sorting of sediment particles during the transition from a high to a progressively lower energy environment.

Rossmann and Seibel (1977) concluded that the observed pattern of sediment texture is related to the dissipation of wave energy as waves interact with the lake

bottom. They found that for depths less than 18 m, the sediments were fine sands (mean $\phi = 2.08$) and were moderately sorted (standard deviation of mean $\phi = 0.848$). For depths ranging from 18 to 27 m, the sediments were fine sands (mean $\phi = 2.92$) and were poorly sorted (standard deviation of mean $\phi = 1.28$). For depths greater than 27 m, the sediments were coarse silts (mean $\phi = 4.14$) and were poorly sorted (standard deviation of mean $\phi = 1.66$). Thus they found that for sediments of this region, mean ϕ increased in an offshore direction and the degree of sorting decreased. These changes were not uniform with distance from shore. The observed changes were used to define two depth regimes having a boundary at a water depth of 24 m (Seibel et al. 1974). The predicted sediment size distribution calculated by Rossmann and Seibel (1977) was based upon the velocity for initiation of particle movement and wave data for southeastern Lake Michigan and agreed favorably with the observed sediment size distribution.

Perturbations in the Observed Pattern of Sediment Texture

Within the study region, the sedimentation pattern is interrupted by unexpected coarse sediments (Fig. 3). The coarser sediments directly offshore of the stream near and the streams southwest of the Donald C. Cook Nuclear Plant have been attributed to increased water velocities associated with the streams' interaction with the lake (Rossmann and Seibel 1977). The coarse sediments to the northeast have been postulated to represent a lag deposit derived from the interaction of waves with the Lake Border Morainic System (Rossmann and Seibel 1977). To the south of this region, the shoreline is characterized by sand dunes. Regions of coarse sediments adjacent to streams define the region of the lake impacted by each stream's input.

VERTICAL TEXTURAL VARIATION OF SEDIMENTS

Based on seven cores collected from nearshore southeastern Lake Michigan in 1975 and excluding core 92 (station 92, Fig. 1), surficial sediments have a higher sand content than deeper sediments (Figs. 4-10). Three of the cores (stations 7, 156, and 157) had a distinct subsurface sand maximum. Cores from stations 5 and 90 had rather uniform sand-rich (>95%) layers which extended to 6 and 8 cm, respectively.

For the six of seven cores having a higher sand content in surficial sediments than in deeper sediments, the smallest increase in sand content occurred in core 93 (Fig. 8) from station 93 (Fig. 1), an offshore core. Although the difference was small, increasing from an average sand content of 4.6 percent in the bottom 47 cm to an average of 9.4 percent over the top 18 cm, the difference was significant ($\alpha = 0.05$, Student's t-test). The largest increase, about 50 percent, occurred in core 90 (Fig. 6) from station 90 (Fig. 1), the core closest to shore. Cores from stations 5, 7, 156, and 157 displayed degrees of absolute sand enrichment in surficial sediments relative to deeper sediments between the extremes noted for cores 90 and 93.

The change in the sand content from 2 percent at the bottom of the core (40-50 cm) to 7 percent at the top (0-9 cm) of core 93 resulted from an increase in the amount of very fine sand (3 to 4 ϕ size fraction). In cores 7, 156, and 157 (Figs. 5, 9, 10), the higher sand content also reflected an increase in the amount of very fine sand. In the two most shoreward cores (stations 90 and 5, Fig. 1), increases in the amount of medium sand (1 to 2 ϕ fraction) in core 90 (Fig. 6) and the amount of fine sand (2 to 3 ϕ fraction) in core 5 (Fig. 4) accounted for the increased sand

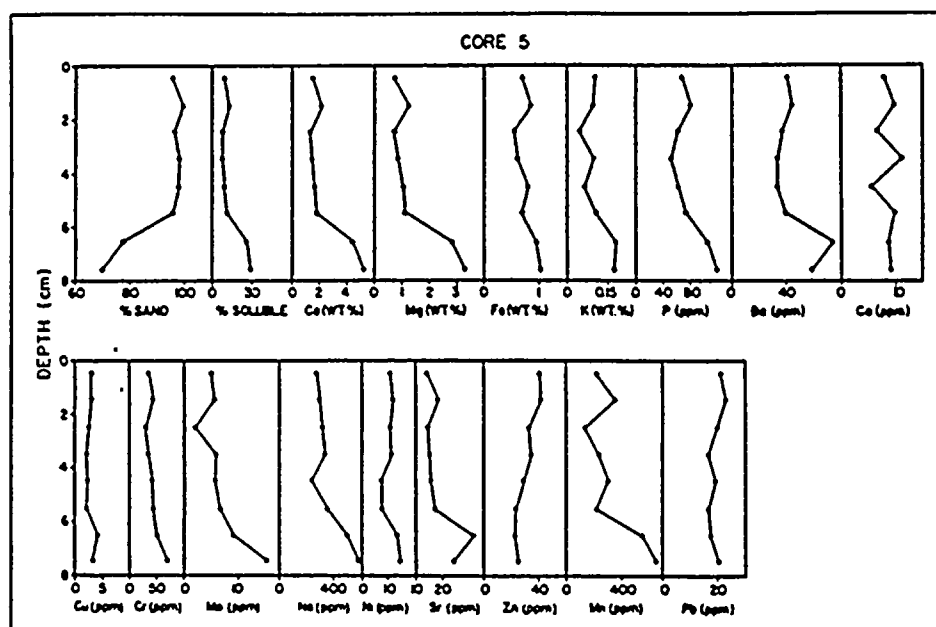


FIGURE 4. Vertical profiles of various parameters in nearshore southeastern Lake Michigan sediments collected in 1975 from station 5 (Fig. 1).

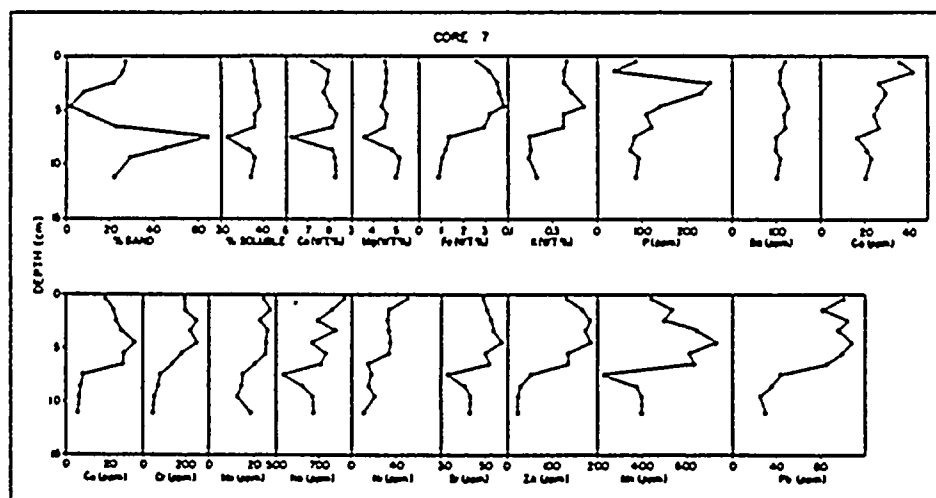


FIGURE 5. Vertical profiles of various parameters in nearshore southeastern Lake Michigan sediments collected in 1975 from station 7 (Fig. 1).

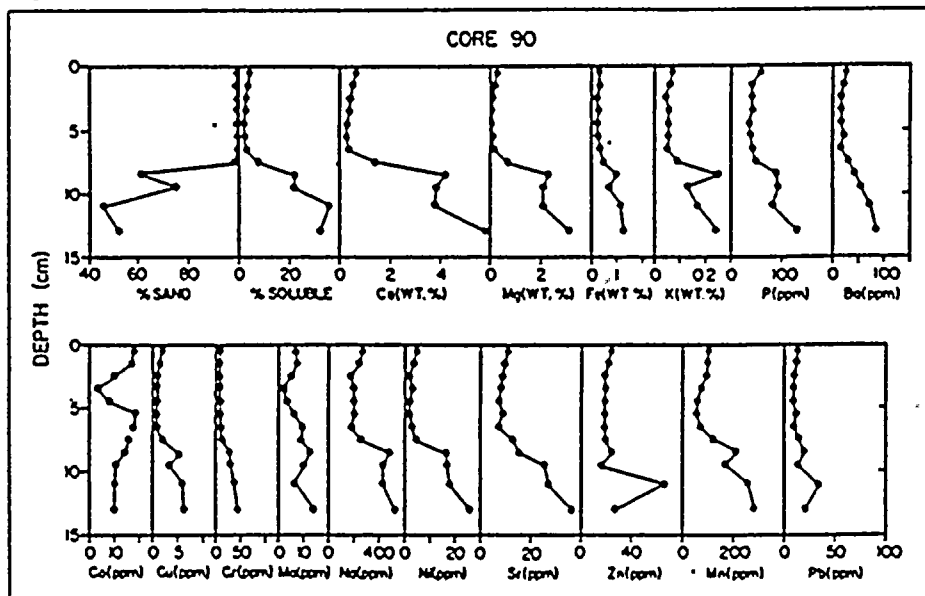


FIGURE 6. Vertical profiles of various parameters in nearshore southeastern Lake Michigan sediments collected in 1975 from station 90 (Fig. 1).

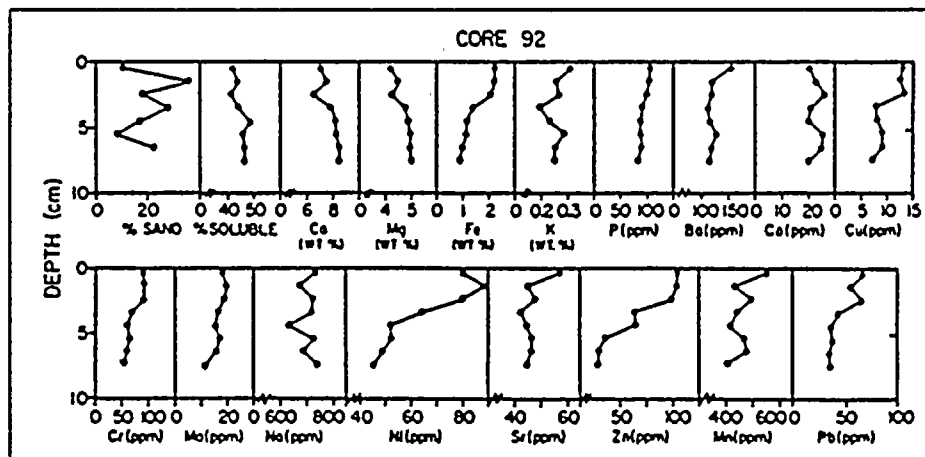


FIGURE 7. Vertical profiles of various parameters in nearshore southeastern Lake Michigan sediments collected in 1975 from station 92 (Fig. 1).

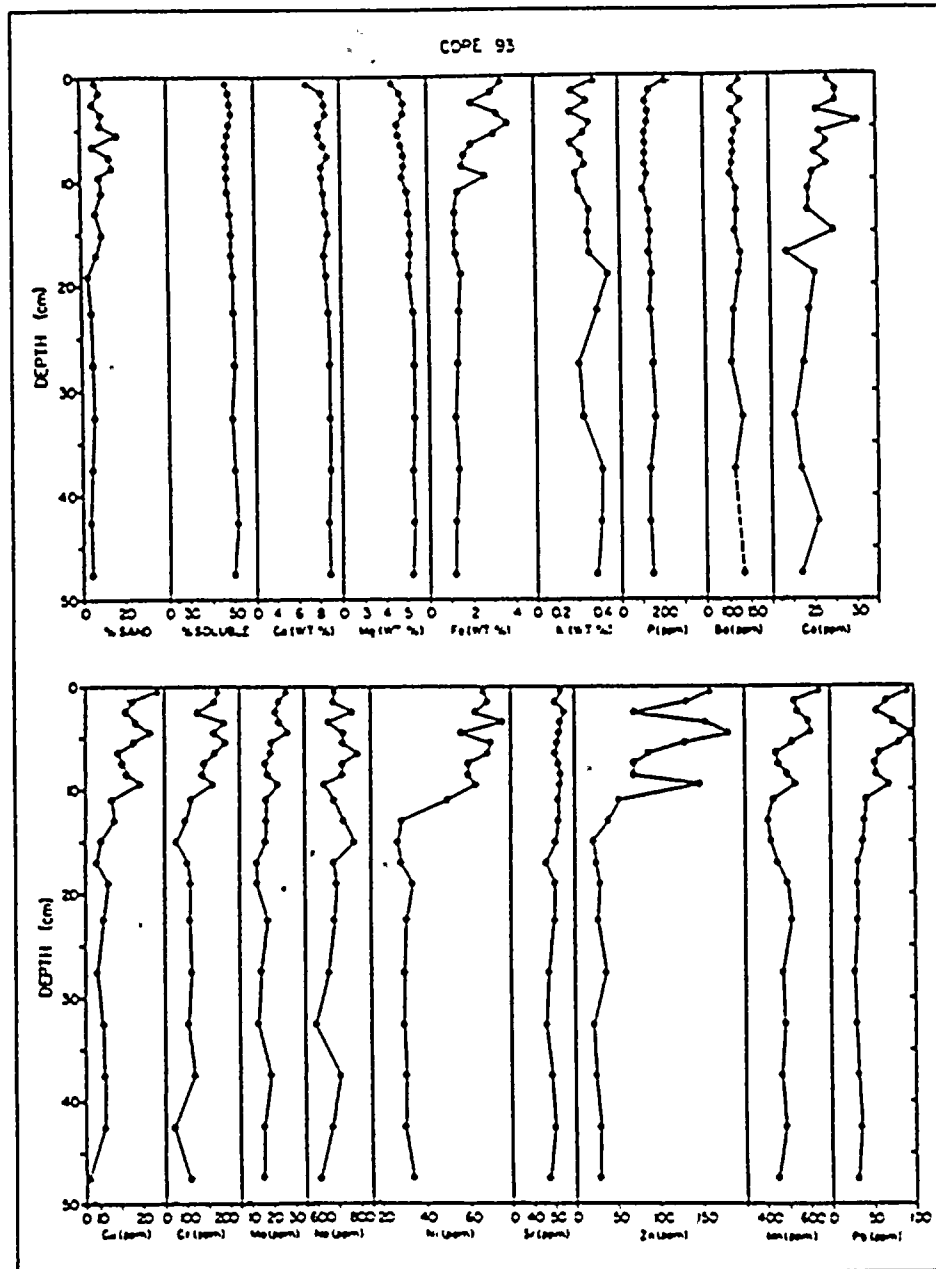


FIGURE 8. Vertical profiles of various parameters in nearshore southeastern Lake Michigan sediments collected in 1975 from station 93 (Fig. 1).

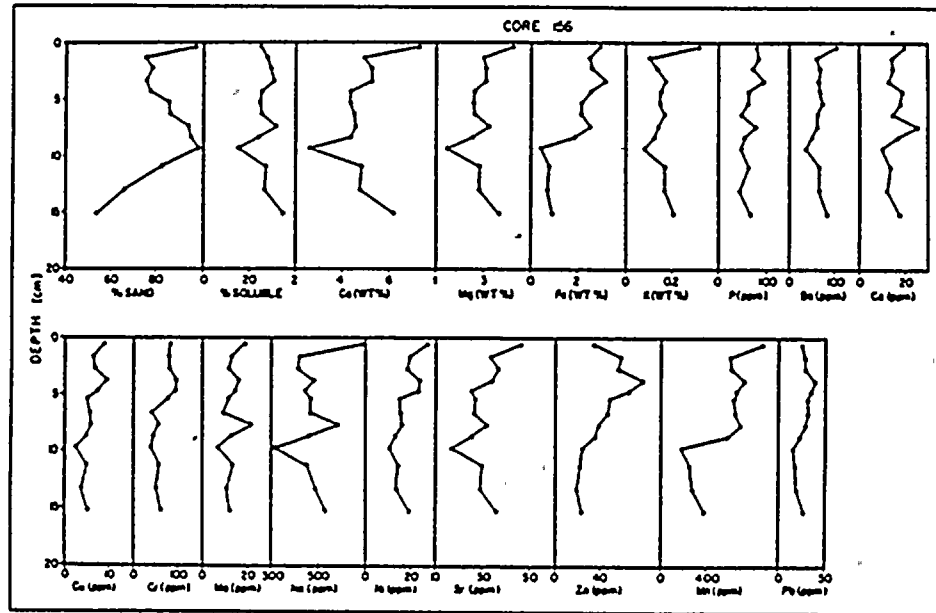


FIGURE 9. Vertical profiles of various parameters in nearshore southeastern Lake Michigan sediments collected in 1975 from station 156 (Fig. 1).

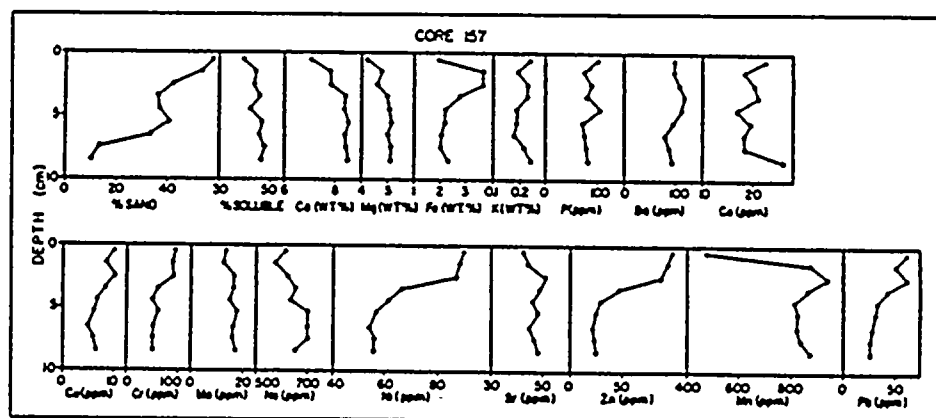


FIGURE 10. Vertical profiles of various parameters in nearshore southeastern Lake Michigan sediments collected in 1975 from station 157 (Fig. 1).

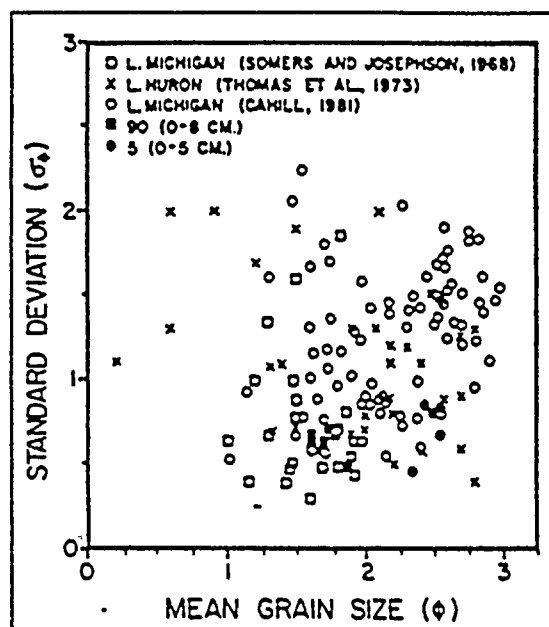


FIGURE 11. Relationship between standard deviation and mean grain size for Lake Michigan and Lake Huron sediments.

content of surficial sediments. The increasing sand content from the bottom to the top of the cores may reflect a change to a higher energy environment. If this assessment is correct, the observed sand enrichment could result from either a sorting of fines from an already deposited sediment or a change to conditions favoring the deposition of coarser sediments. A less likely explanation for the change in sand content of the cores is a decrease in the supply of fine-grained sediments due to decreased shoreline erosion. This erosion was observed to be very active during the period of sample collection.

The sand-rich sediments of cores 5 and 90 are not unique. Similar sand-rich surficial sediments occur in all of the Great Lakes. A comparison of the standard deviation—grain size relationship for the sand-rich surficial sediments of cores 5 and 90 to that for surficial sediments obtained by grab samplers from nearshore areas of Lakes Michigan and Huron illustrates that the medium-to-fine sand of cores 5 and 90 is better sorted than most of the samples of comparable grain size (Fig. 11). However, because the results of Cahill (1981) and Thomas et al. (1973) were based upon only the top 3 cm of sediment and because Somers and Josephson (1968) did not report the thickness of their samples, we do not know if the thickness of well-sorted sands reported in those studies is comparable to the 6- to 8-cm-thick layers found in our cores.

In Lake Ontario, sands up to 6 cm deep overlying glaciolacustrine clays have been attributed to the sorting of the fine-grained component of the sediment by currents and waves to form a coarse-grained lag deposit (Thomas et al. 1972). This interpre-

tation was based on the decrease in sorting and on the increasing clay content from the top to the bottom of the sand layer.

The sediments of cores 5 and 90 did not display the gradual decrease in sand content and increase in standard deviation with depth in the sand-rich layer as described by Thomas et al. (1972). Instead, the mean grain size and the standard deviation remained fairly constant with depth within the sand-rich surface layer and increased only after the sand content dropped below 95% (Table 1). Therefore, if the condition described by Thomas et al. (1972) is indicative of sorting, the sediments in the sand-rich sections of cores 5 and 90 were not the result of sorting.

A detailed examination of the individual phi classes with depth in cores 5, 90, and 93 reveals a pattern common to all three cores. The pattern in the sand-rich cores, 5 and 90, (Fig 12) consists of transition zone, characterized by a fairly uniform grain size distribution over the 1 to 4 phi range, separating a sand-rich upper layer with one dominant phi size from a lower layer where the dominant phi size is higher. In core 93, the sand-rich (9% sand) upper layer is separated from deeper sand-poor (4% sand) sediments by a transition zone of 18 to 20 cm depth (Table 2). Compared to sediments above and below it, the transition zone contains a greater amount of silts and clays (> 5 phi size fraction) which replace fine sand to a coarse silt (2 to 5 phi size fraction). Unlike the bottoms of cores 5 and 90, there was no dominant sand-size fraction in the bottom 45 cm of core 93. The occurrence of coarser sediments in surficial sediments separated from slightly finer deeper sediments by a transition zone is common to all three cores. This pattern is indicative of a similar origin for the observed textural variations within the three cores.

The transition to a more sand-rich sediment is not uniform throughout the region, as reflected in the subsurface sand maxima in cores 7, 156, and 157 (Fig. 4). The main factor contributing to the subsurface maximum in each core and to the increasing sand content toward the top of the cores is the increase in the 3- to 4-phi size fraction. In all three cores, both the depth of the subsurface maximum within the core and the percent sand of the subsurface sand maximum correlate inversely to the

TABLE 1. Variation of textural parameters with depth in cores 5 and 90 collected from southeastern Lake Michigan in 1975.

Depth (cm)	Core 5			Core 90		
	% Sand	Phi ¹	SD ²	% Sand	Phi ¹	SD ²
0-1	95.10	2.46	0.73	98.99	1.79	0.68
1-2	99.52	2.56	0.67	99.27	1.74	0.70
2-3	96.10	2.56	0.84	99.74	1.71	0.61
3-4	98.31	2.51	0.80	99.86	1.70	0.60
4-5	98.10	2.44	0.85	99.94	1.62	0.63
5-6	95.76	2.62	0.91	99.92	1.61	0.63
6-7	77.81	3.57	1.26	99.89	1.69	0.61
7-8	70.24	3.69	1.43	98.65	1.79	0.71
8-9				61.72	3.43	2.12
9-10				75.16	3.21	1.72
10-12				46.64	4.45	2.00
12-14				52.54	4.28	2.25

¹Mean grain size in phi units.

²Standard deviation.

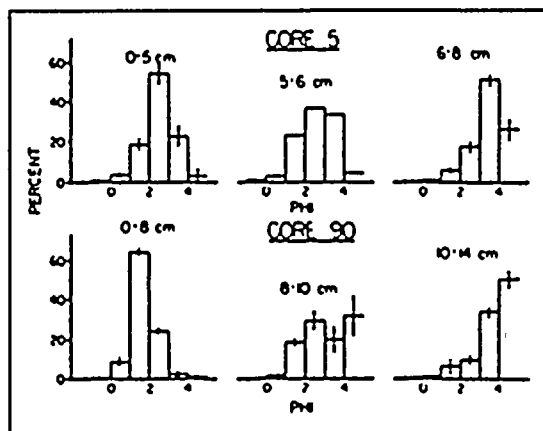


FIGURE 12. Histograms of percent sediment per phi interval in sand-rich sections of cores from stations 5 (0-5, 5-6, 6-8 cm depths) and 90 (0-8, 8-10, 10-14 cm depths). The open ended interval in each histogram includes all phi sizes greater than 4.

water depth ($r = -0.982$ and -0.997 , respectively, $p \leq 0.05$). Because of the small number of cores, it is difficult to determine whether or not the subsurface sand maximum is a general feature of water depths within the 26- to 29-meter range. Based on the inverse relationship between the depth of the sand maximum and water depth, it is possible that the subsurface maximum is a general feature of inshore cores. This feature would not have been observed in cores 5 and 90 because they may not have penetrated deeply enough into the sediment.

The sand content in six of the seven collected cores displays regular patterns. These patterns are a lakeward decrease in sand content, an increase in sand content from the bottom to the top of the cores, and the location and depth of occurrence of a subsurface sand maxima. Each of these correlates negatively to distance offshore or to water depth. Increasing water depth and increasing offshore distance are indicative of a transition from a high to a lower energy environment. The increased

TABLE 2. Distribution of phi sizes with depth in core 93 collected from southeastern Lake Michigan in 1975.

Depth (cm)	Percent of total weight for various phi size intervals					
	1-2	2-3	3-4	4-5	5-6	> 6
0-18 ¹	1.16	1.28	7.36 ²	53.04	18.76	18.30
18-20 ¹	1.20	0.85	1.04	40.43	28.84	27.52
20-65 ⁴	1.22	1.17	2.14	47.84	26.39	21.08

¹Average of 14 intervals.

²Without the extreme values in this phi size, 2.29, 10.61, 12.51, and 15.33, the average is 6.23.

³One 2-cm interval.

⁴Average of nine intervals.

sand content in the tops of the cores reflects a transition to a higher energy depositional environment which now favors the deposition of coarser sediment particles in nearshore Lake Michigan. This transition is postulated to result from long-term lake level fluctuations.

CHEMISTRY

INTRODUCTION

Surficial sediment chemistry can be used to describe the time-integrated impact of sources on the nearshore zone of lakes. Vertical profiles can be used to describe the time variation of contaminants, provided the contaminants are not mobilized by sedimentary diagenesis. Within the nearshore zone of southeastern Lake Michigan, changes in sediment chemistry are often related to changes in the environment of deposition.

The earliest reported major study of southern Lake Michigan sediment chemistry is that of Callender (1969) for sediments collected in 1968. Sediments were analyzed for concentrations of calcium, magnesium, iron, manganese, carbonate, organic carbon, and total nitrogen. Sediments collected in 1969 by the Illinois State Geological Survey were analyzed for aluminum, silicon, iron, calcium, magnesium, potassium, titanium, manganese, phosphorus, sodium, bromine, lanthanum, scandium, cobalt, beryllium, chromium, nickel, lead, vanadium, cadmium, copper, zinc, boron, total carbon, inorganic carbon, organic carbon, arsenic, and mercury (Shimp et al. 1970, Shimp et al. 1971, Ruch et al. 1970, Schleicher and Kuhn 1970, Kennedy et al. 1971). Sediment cores collected in the early 1970s were analyzed for arsenic, barium, bromine, calcium, cadmium, cerium, cobalt, chromium, cesium, copper, europium, iron, hafnium, potassium, lanthanum, lutetium, magnesium, manganese, sodium, nickel, phosphorus, lead, rubidium, antimony, scandium, samarium, terbium, thorium, uranium, zinc, organic carbon, and inorganic carbon (Edgington and Robbins 1976, Robbins and Callender 1975, Robbins and Edgington 1977). Sediment samples were collected from river mouths around southern Lake Michigan in 1972 (Fitchko and Hutchinson 1975). Lead, silver, cadmium, cobalt, copper, chromium, nickel, zinc, manganese, and mercury concentrations were determined for sediments from the Milwaukee and Root rivers of Wisconsin, the Calumet River of Illinois, Trail Creek of Indiana, and the St. Joseph, Black, Kalamazoo, and Grand rivers of Michigan. The most recent major survey of southern Lake Michigan sediments occurred in 1975 as a cooperative effort between the Illinois State Geological Survey and the Canada Centre for Inland Waters (Cahill 1981). Surficial sediments were analyzed for concentrations of barium, bromine, cerium, cesium, europium, gallium, hafnium, lanthanum, rubidium, antimony, scandium, samarium, strontium, tantalum, terbium, thorium, uranium, tungsten, yttrium, iron, potassium, sodium, manganese, silver, cobalt, molybdenum, chromium, nickel, zinc, aluminum, calcium, chlorine, phosphorus, silicon, titanium, magnesium, sulfur, lead, copper, beryllium, vanadium, zirconium, arsenic, selenium, mercury, cadmium, and organic carbon.

SURFICIAL SEDIMENTS

Metals concentrations in surficial sediments are dependent upon the relative abundance of major components of the sediment with which the metals are bound, the environment of deposition, and the proximity of metal sources. Components of nearshore (≤ 24 -m water depth) southeastern Lake Michigan surficial sediments include silicates (81%) dolomite (12.6%), ferric hydroxide (2.4%), calcite (1.1%), and organic carbon (0.4% C). Of the extractable fraction, the carbonates account for 83% of the total. The observed metal distributions and concentrations within this nearshore region are influenced by the carbonates, calcite and dolomite.

The environment in which the sediments exist in the nearshore southeastern Lake Michigan region is extremely dynamic. Sediments covering most of the region to depths as great as 27 m are routinely subjected to resuspension by waves (Rossmann and Seibel 1977), providing little opportunity for observable sedimentary diagenesis to occur.

The major inputs of metals and other constituents occur at the mouths of streams and rivers (Fig. 1). Metals and nutrients are supplied to nearshore southeastern Lake Michigan at concentrations above those of the ambient lake water (Rossmann 1986). Dissolved metals and nutrients are incorporated in sediment through adsorption or precipitation processes. Plumes of enriched sediments often mark the point of river input to the lake. These river plumes are usually discordant to the bathymetry of the lake.

Areal Variation

For nearshore southeastern Lake Michigan, areal variations of metals and phosphorus in surficial sediments are dependent primarily upon the redistribution of organic carbon and carbonates by waves. Detrital carbonates and organic carbon occur at highest concentrations in the silt- and clay-sized fractions of the sediment (Rossmann and Seibel 1977). Thus metal concentrations in these nearshore sediments are expected to increase in an offshore direction as a result of the preferential transport of fine-grained sediments offshore.

Stream inputs of sediment-associated materials to the nearshore zone produce a time-integrated plume of materials in the sediments. These plumes exist because of the constant supply of raw materials. The extent of a plume is dependent upon the rate of supply of materials from a stream or river relative to the rate of removal of materials by physical forces within the lake (waves and currents). In the absence of stream inputs, concentrations of metals and phosphorus in surficial sediments can be expected to increase in an offshore direction in a regular and predictable manner as a result of their association with fine-grained sediments which are sorted by waves and transported offshore. The expected metal and phosphorus concentrations increase in an offshore direction and are distorted by stream inputs.

Changes in Physical and Chemical Properties in an Offshore Direction

Rossmann and Seibel (1977) have observed for these nearshore sediments that areal variations in concentrations of metals and other parameters associated with the fine-grained fraction of sediments are controlled by the dissipation of wave energy on the lake bottom. Within the region of study, the lake bottom slopes gently offshore with only a few minor perturbations in the regularity of the bottom con-

tours (Fig. 2). Contours of physical and chemical properties of the surficial sediments are concordant with the bathymetry except for regions of changing nearshore lithology or of stream and river inputs.

Because carbonates are the major non-silicate component of the surficial sediments in the region of study and because the carbonates, organic carbon, phosphorus, and trace metals are all associated with the fine-grained fraction of the sediment, many parameters have a distribution pattern similar to that of the carbonates. Inorganic carbon (carbonates) concentrations increase in an offshore direction (Fig. 13). Contours become more regular at depths greater than 27 m which is where waves begin to interact with the lake bottom to resuspend sediments (Rossmann and Seibel 1977). The alternating higher and lower concentrations along lines parallel to shore are the result of sediment textural changes consisting of alternating regions of fine sand to very fine sand and silt (Fig. 3). Other areal distributions which mimic that of the carbonates include loss on ignition, total carbon, organic carbon, calcium, cobalt, chromium, copper, iron, potassium, magnesium, manganese, molybdenum, sodium, nickel, phosphorus, strontium, and zinc.

Perturbations of Concentration Gradients Due to Inputs

Shoreline erosion, groundwater, and streams impact the nearshore zone of southeastern Lake Michigan. Shoreline erosion is most evident in the southern one-third of the region (Warren Dunes) because the shoreline being eroded contains large amounts of carbonates. Inorganic carbon, loss on ignition, total carbon, calcium, and magnesium concentrations, representing the carbonates in sediments, are elevated offshore of this carbonate-rich shoreline (2.5 km south of Warren Dunes) (Figs. 13-17).

Stream inputs of particulate or sediment-associated metals and phosphorus form regions of surficial sediments having anomalously high concentrations of organic carbon, barium, cobalt, manganese, potassium, strontium, and phosphorus. A stream near Bridgman, Michigan, has the largest impact upon the nearshore sediments adjacent to the beach. The areal distributions of phosphorus and manganese reflect the stream's impact on this nearshore region (Figs. 18 and 19). Total phosphorus and manganese concentrations were extremely high at a location between Bridgman and Warren Dunes. A stream tributary to the lake in this region receives the effluent of a wastewater treatment plant. Measured concentrations of orthophosphate and total phosphorus were as high as 1,430 ppb P and 3,225 ppb P, respectively. Barium, cobalt, potassium, and strontium concentrations were elevated also in sediments adjacent to this stream (Figs. 20-23). Similarly, organic carbon concentrations were elevated within this region (Fig. 24). Three other regions adjacent to the shoreline with elevated organic carbon concentrations were a region 2.5 km southwest of Warren Dunes, a region 1.5 km northeast of the Donald C. Cook Power Plant, and a region 7 km north of the power plant.

Along the shoreline, the study area is impacted by groundwater inputs. The region most dramatically impacted is approximately 1 km southwest of the Donald C. Cook Power Plant. Iron concentrations are elevated adjacent to the shoreline (Fig. 25). A reconnaissance of the shoreline southwest of the power plant revealed groundwater upwelling along sheet pilings. Ferric hydroxides were observed to be precipitating due to oxidation of the groundwater. Groundwater is hypothesized to be the source of elevated iron concentrations in sediments adjacent to the shoreline. Likewise, elevated concentrations of chromium and strontium south of the power

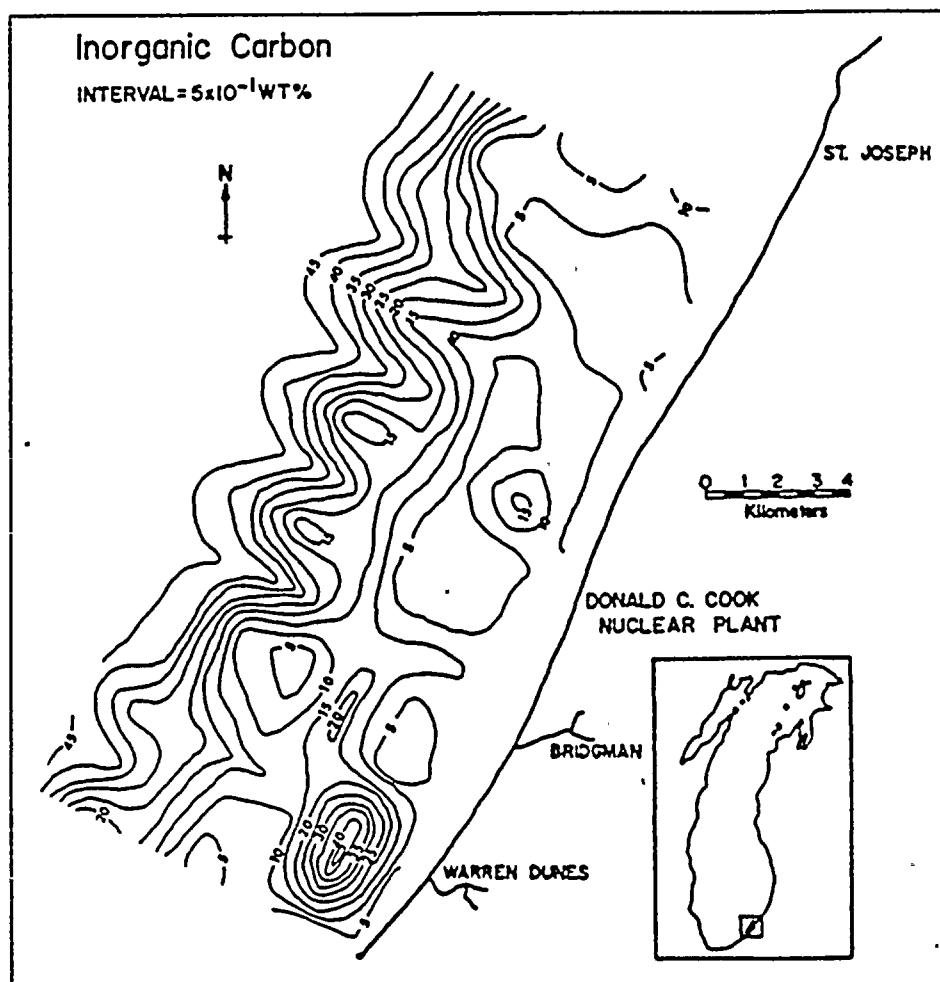


FIGURE 13. Distribution of inorganic carbon in nearshore southeastern Lake Michigan surficial sediments. To provide clarity, the numbers within the contour lines are an abbreviation of the given contour interval.

plant are probably due to their transport to the lake by groundwater (Figs. 23 and 26).

Inter-comparison of Stream, Nearshore, and Offshore Sediments

With only one exception, elemental concentrations in nearshore sediments of southeastern Lake Michigan are less than those observed for offshore stations (Table 3). The only exception is manganese which is diagenetically mobile and tends

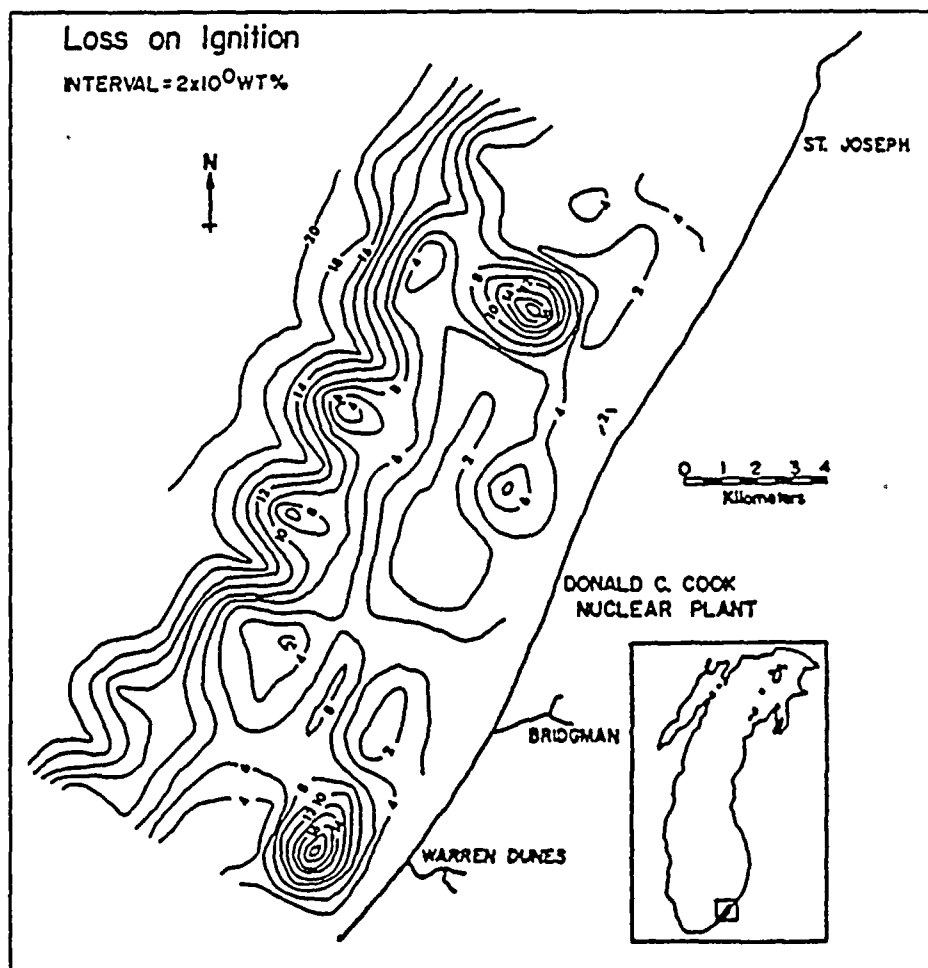


FIGURE 14. Distribution of loss on ignition in nearshore southeastern Lake Michigan surficial sediments. To provide clarity, the numbers within the contour lines are an abbreviation of the given contour interval.

to be most concentrated in the solid phases of oxidized sediments. St. Joseph River sediments have lower concentrations of cobalt, chromium, manganese, nickel, and zinc than nearshore sediments. Copper is higher in St. Joseph River upstream sediments than in nearshore sediments.

To investigate further the inverse relationship between grain size and elemental concentrations, nearshore and offshore sediments were intercompared (Table 4). In an offshore direction, grain size decreased and elemental concentrations increased. Major components of the sediment which are associated with fine-grained sediments

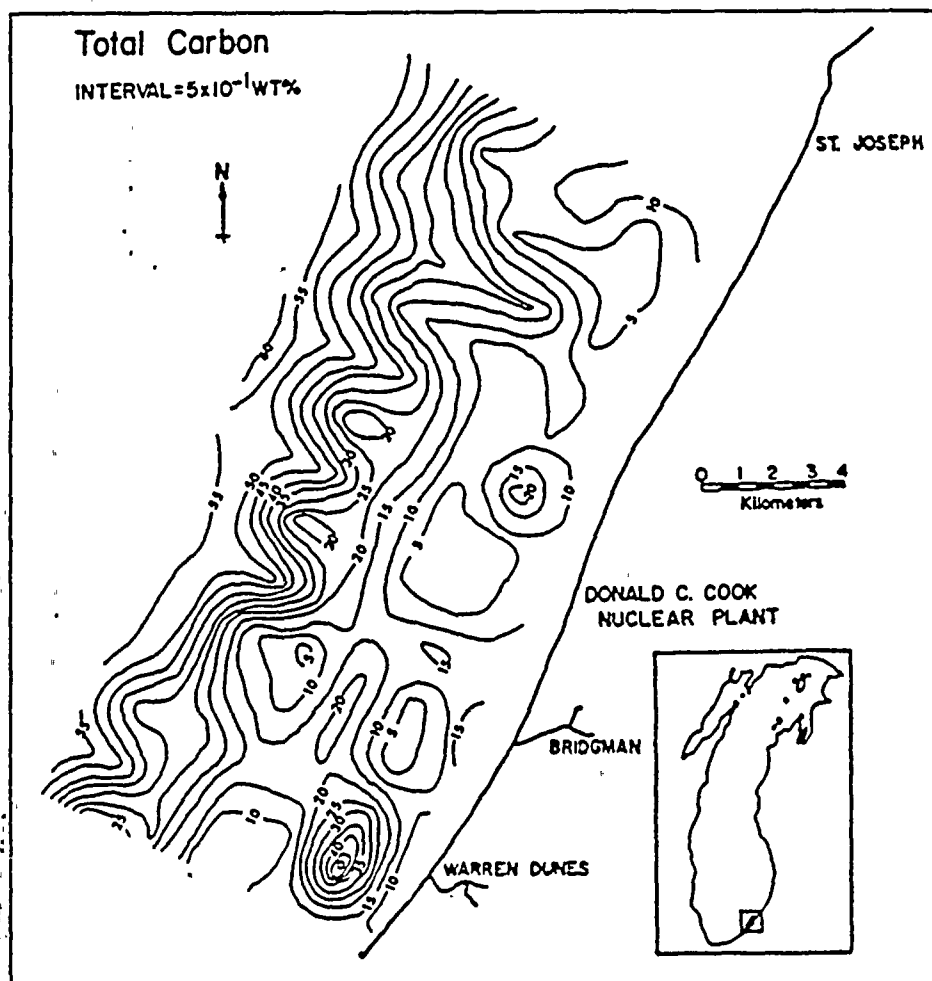


FIGURE 15. Distribution of total carbon in nearshore southeastern Lake Michigan surficial sediments. To provide clarity, the numbers within the contour lines are an abbreviation of the given contour interval.

and increase in an offshore direction include carbonates, organic carbon, and iron compounds. Ratios of nearshore to offshore concentrations illustrate that elements associated with carbonates (inorganic carbon), iron compounds, and organic carbon would increase a maximum of 1.7, 2.6, and 9.1 times, respectively. Loss on ignition, calcium, and magnesium which are known to be associated with the carbonates increase a maximum of 1.7, 1.7, and 1.6 times, respectively. Other metals which may fall into this group or be associated with the silicate fraction of sediment, not extracted by the method used, include manganese and sodium. The manganese is

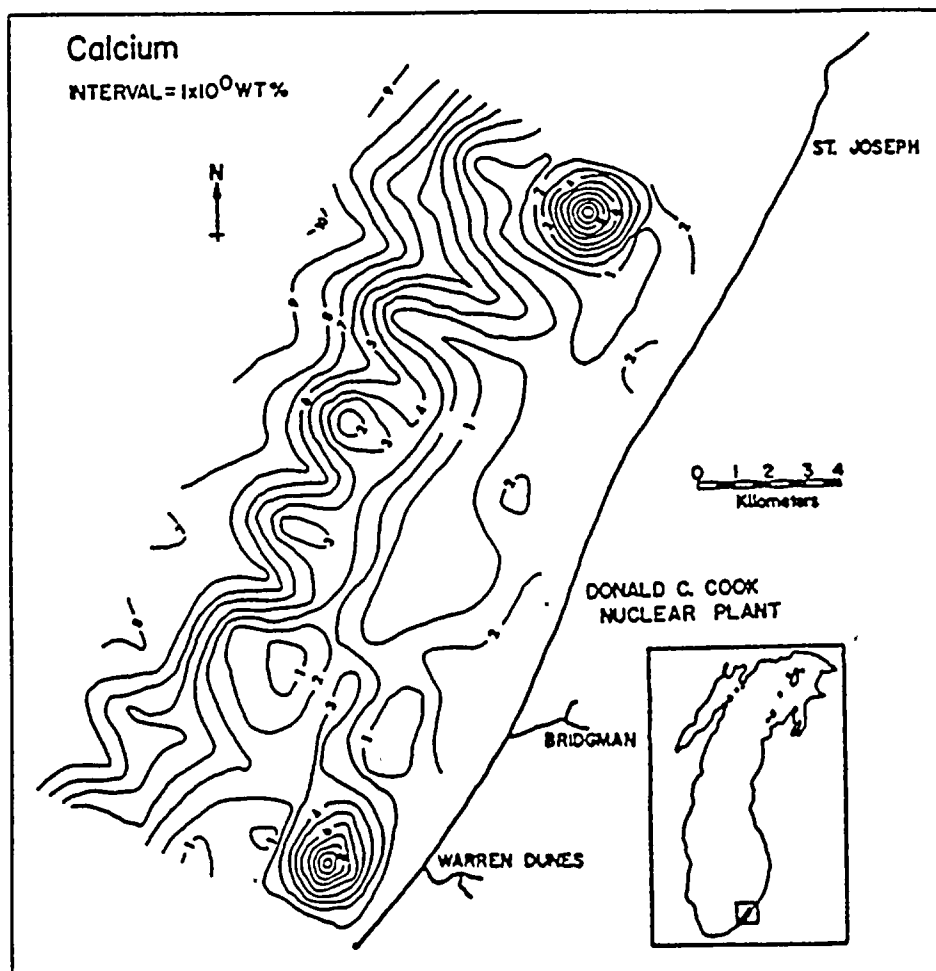
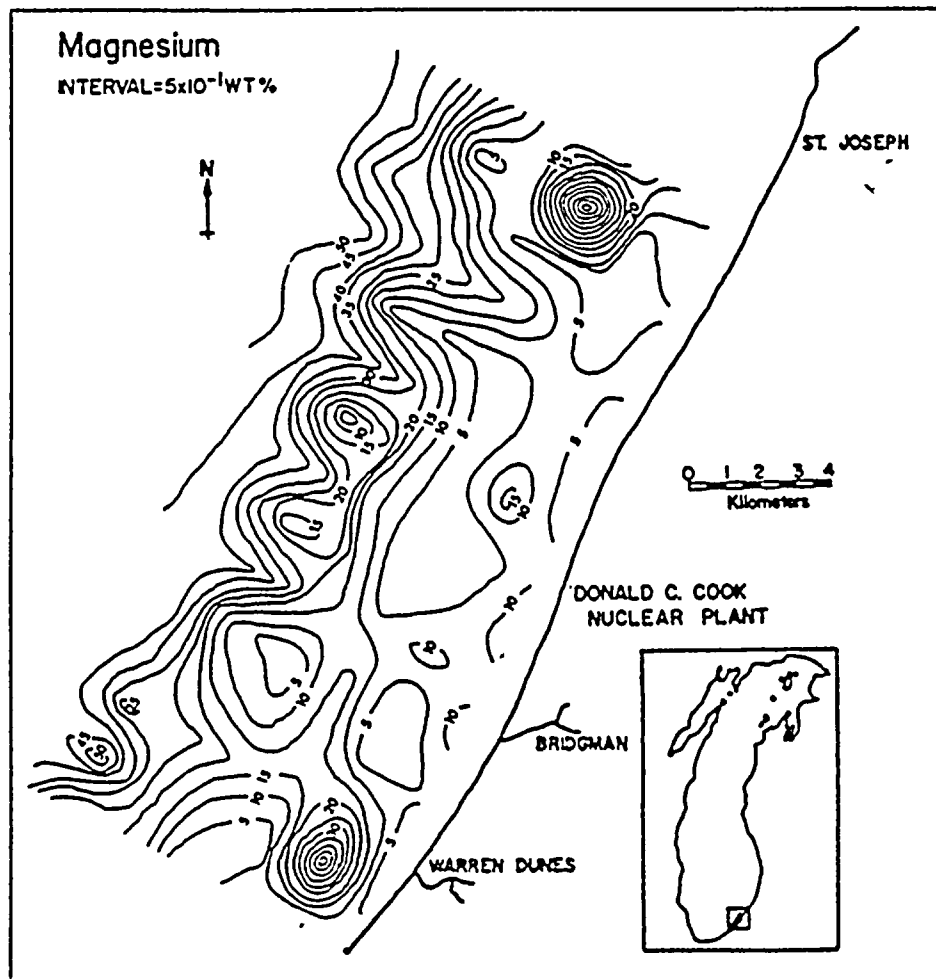


FIGURE 16. Distribution of calcium in nearshore southeastern Lake Michigan surficial sediments. To provide clarity, the numbers within the contour lines are an abbreviation of the given contour interval.

most likely depleted in offshore sediments due to diagenetic losses. Metals which could be associated with iron compounds include cobalt, molybdenum, and potassium. Increases in concentrations of barium, chromium, copper, nickel, phosphorus, strontium, and zinc in an offshore direction can be accounted for only by increases in organic carbon or some as-yet-unidentified component of the fine-grained sediments. Such an unidentified component could be clay minerals which are capable of adsorbing these elements.



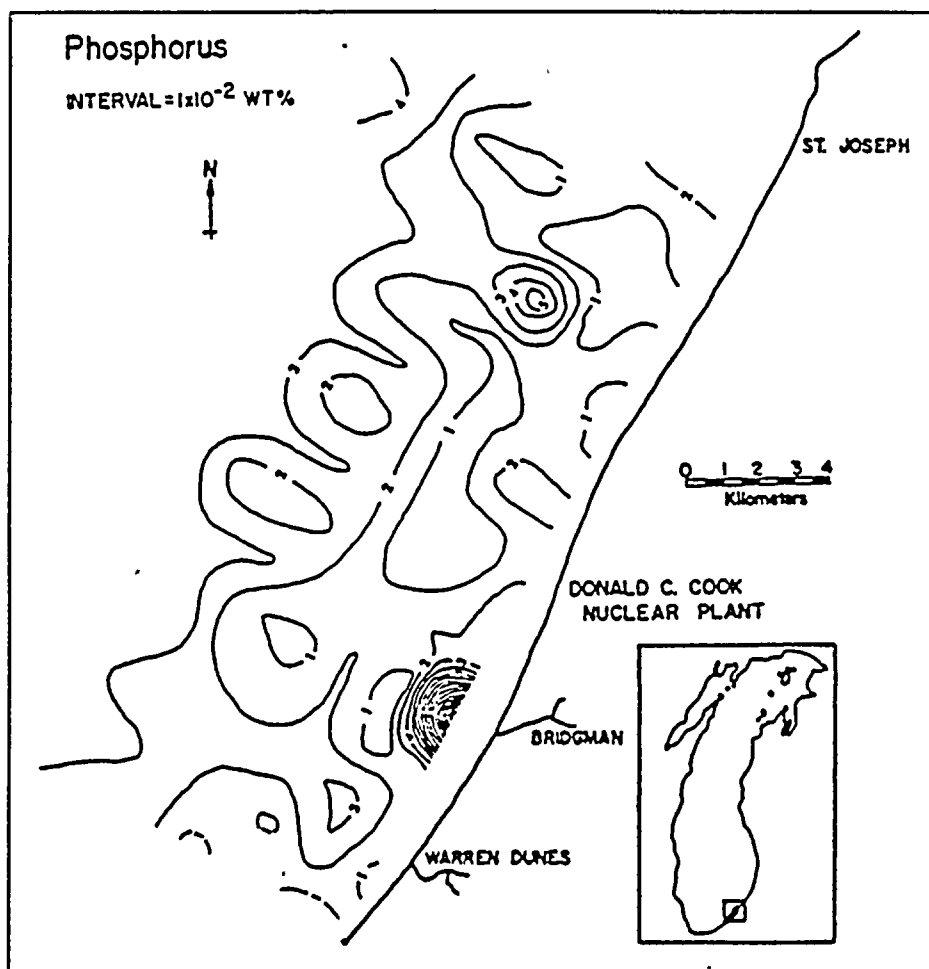


FIGURE 18. Distribution of phosphorus in nearshore southeastern Lake Michigan surficial sediments. To provide clarity, the numbers within the contour lines are an abbreviation of the given contour interval.

ferric hydroxides. These are derived from the erosion of unconsolidated bluffs and sand dunes (Hands 1970) and the precipitation of ferric hydroxide. The carbonates are a mixture of detrital dolomite and calcite. Because the epilimnetic waters are undersaturated with respect to calcite and supersaturated with respect to dolomite, dolomite is preferentially preserved in sediments (Rossmann 1980). All of these components are subject to the sorting action of waves, especially for water depths of less than 27 m (Rossmann and Seibel 1977). The action of waves serves to sort and remove the fine-grained sediments containing calcite, dolomite, and ferric hydrox-

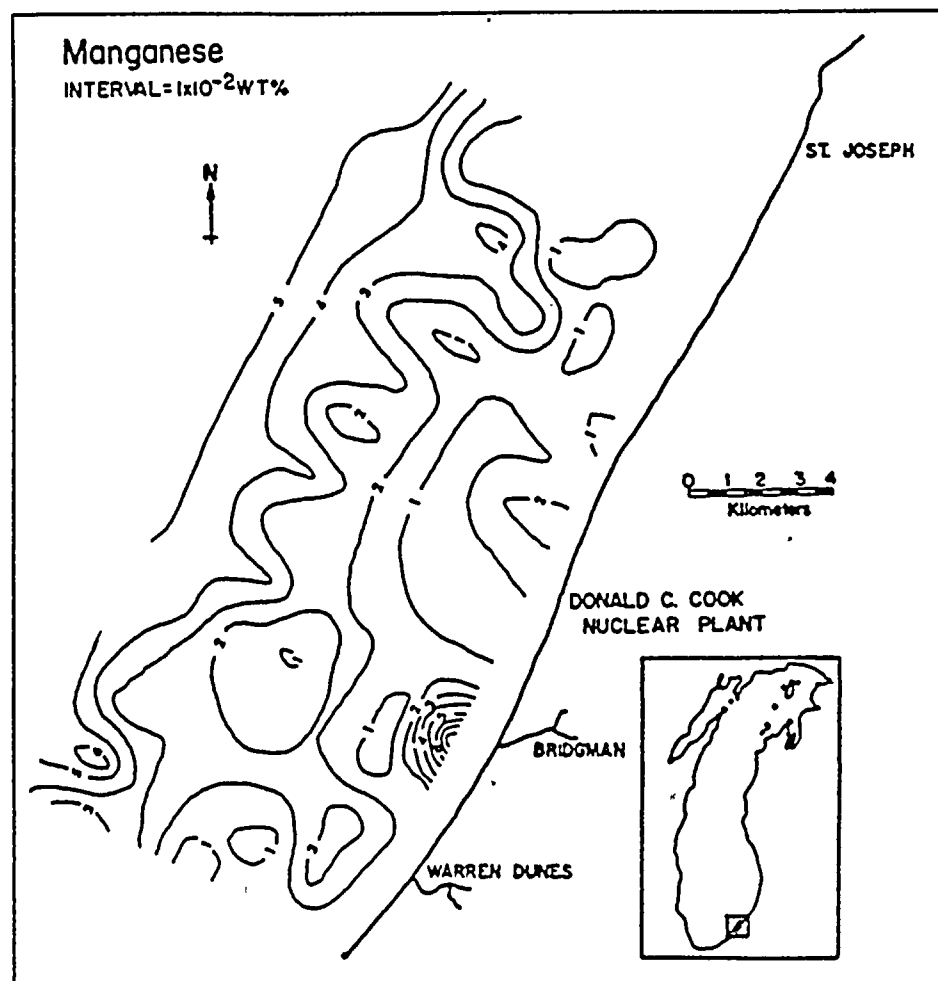


FIGURE 19. Distribution of manganese in nearshore southeastern Lake Michigan surficial sediments. To provide clarity, the numbers within the contour lines are an abbreviation of the given contour interval.

ides as well as organic matter from the coarser quartz-rich sediments. The chemistry and texture of vertical profiles reflect this process.

Vertical Variation of Major Components

Calcite and Dolomite

The carbonate mineral present at the bottom of each core is dolomite. The major carbonate mineral in Lake Michigan sediments is detrital dolomite, reflecting the

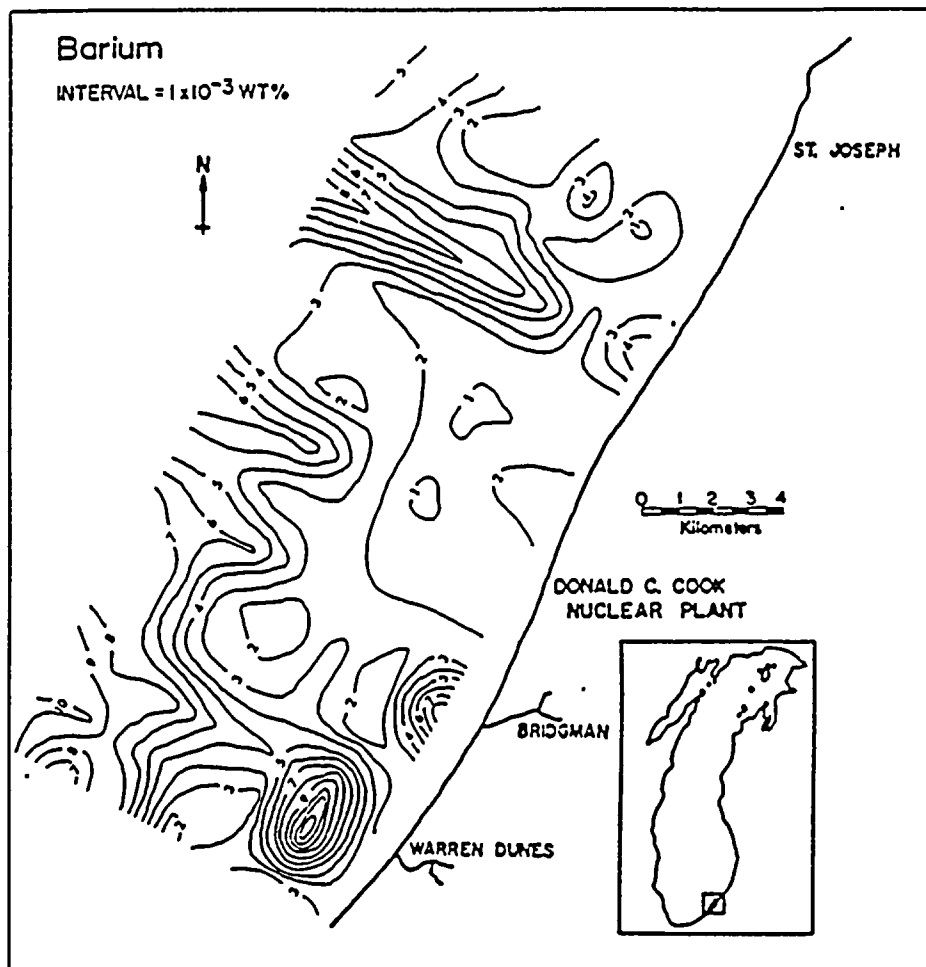


FIGURE 20. Distribution of barium in nearshore southeastern Lake Michigan surficial sediments. To provide clarity, the numbers within the contour lines are an abbreviation of the given contour interval.

preponderance of dolomite in the till and bedrock of the lake (Moore 1961, Callender 1969). The stoichiometric ratio of calcium to magnesium in dolomite is 1.67. Based upon this ratio, the carbonates in sediments in all cores are dolomite (Table 5). The divergence from the ideal ratio in cores from stations 5, 7, and 90 (Fig. 1) results from analytical uncertainties associated with measuring low concentrations of calcium and magnesium or from the presence of mineral phases such as clays that contain calcium and magnesium. Samples from stations 5, 7, and 90 would have an average of 0.3 to 0.7% calcite, representing calcium contents of 0.12% to 0.28%.

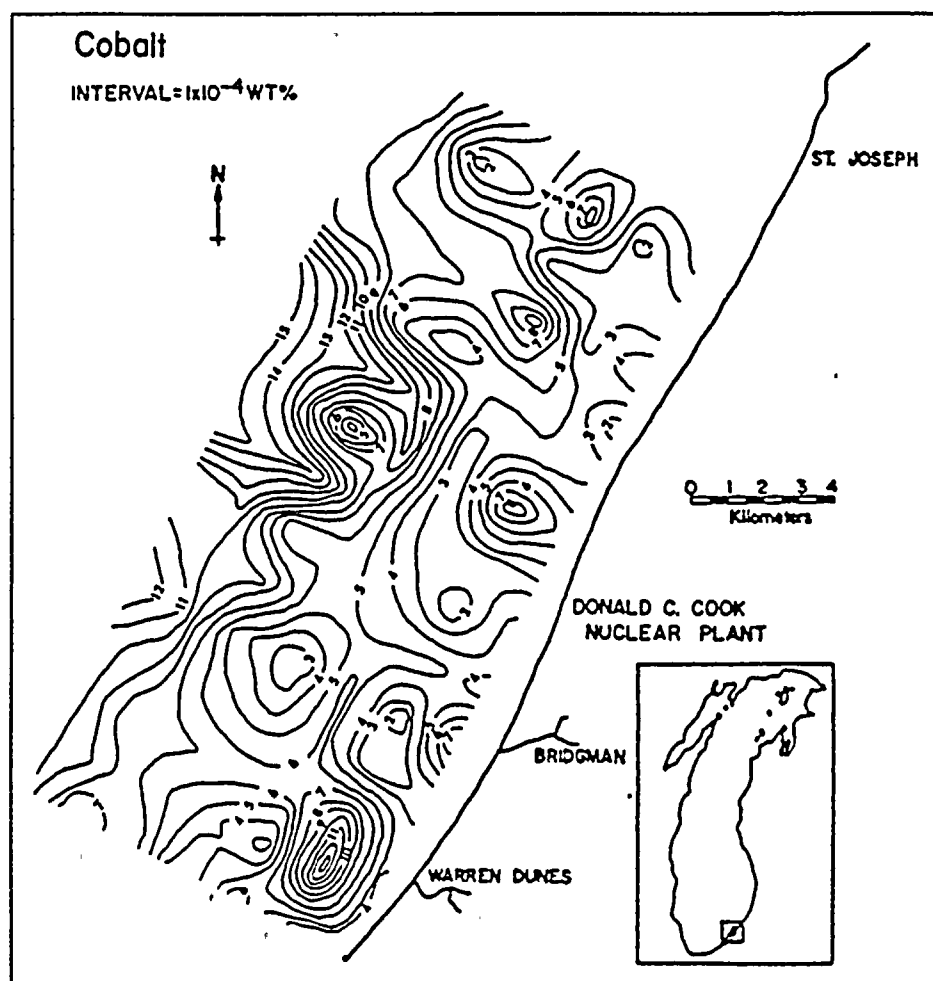


FIGURE 21. Distribution of cobalt in nearshore southeastern Lake Michigan surficial sediments. To provide clarity, the numbers within the contour lines are an abbreviation of the given contour interval.

These low calcium concentrations fall within the analytical uncertainties of the analyses.

Carbonates constitute 80 percent of the soluble material in all cores except core 90 and up to 40 percent of the total sediment (Table 6). The amount of carbonate present in cores increases with increasing water depth and increasing depth within the core. The increase in dolomite content with increasing sediment depth was not observed in core 156. This is attributed to the continuous input of dolomite to that location by the St. Joseph River.

The concentration of dolomite in the sediment is hypothesized to be controlled by

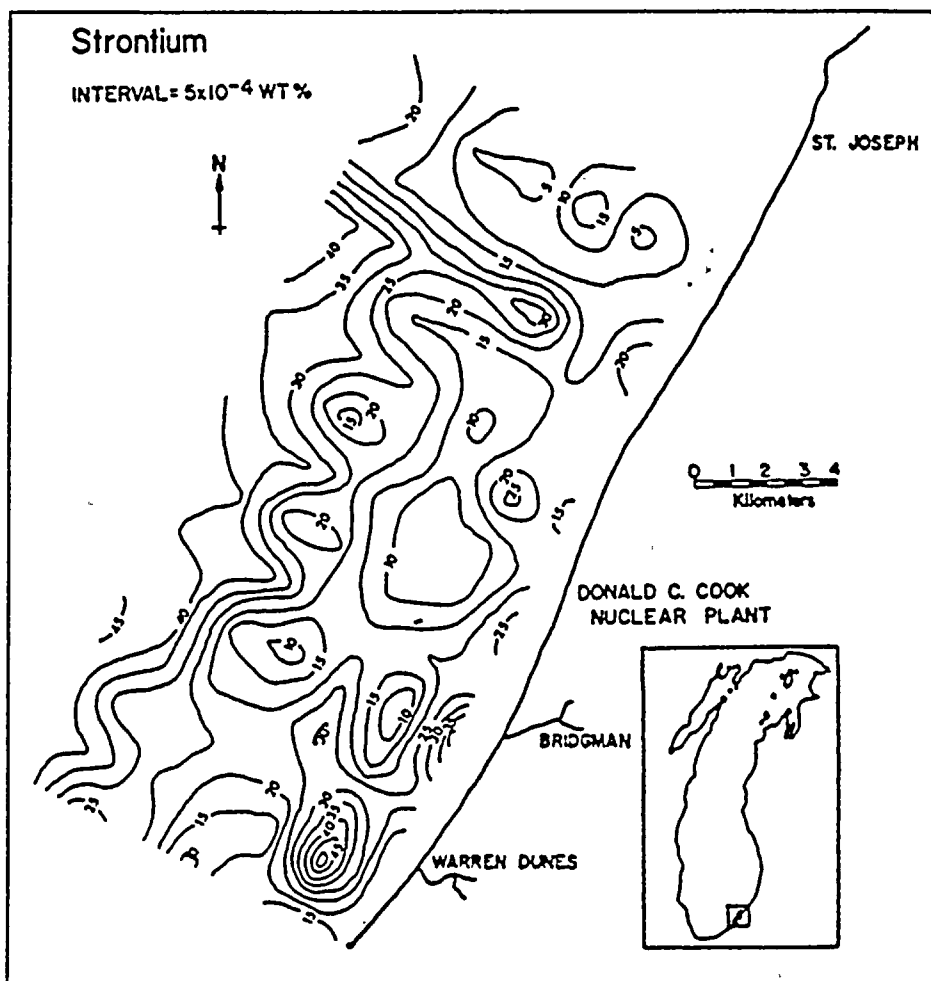


FIGURE 23. Distribution of strontium in nearshore southeastern Lake Michigan surficial sediments. To provide clarity, the numbers within the contour lines are an abbreviation of the given contour interval.

Iron

Like dolomite, iron is associated with fine-grained sediments. Within those cores collected from depths less than 27 m, the iron compounds are sorted from the surficial sediments (Figs. 4 and 6). Those cores collected from greater depths have surficial sediments noticeably enriched with iron relative to sediments from deeper within the cores. Surficial sediments within cores from stations 7, 92, 93, and 157

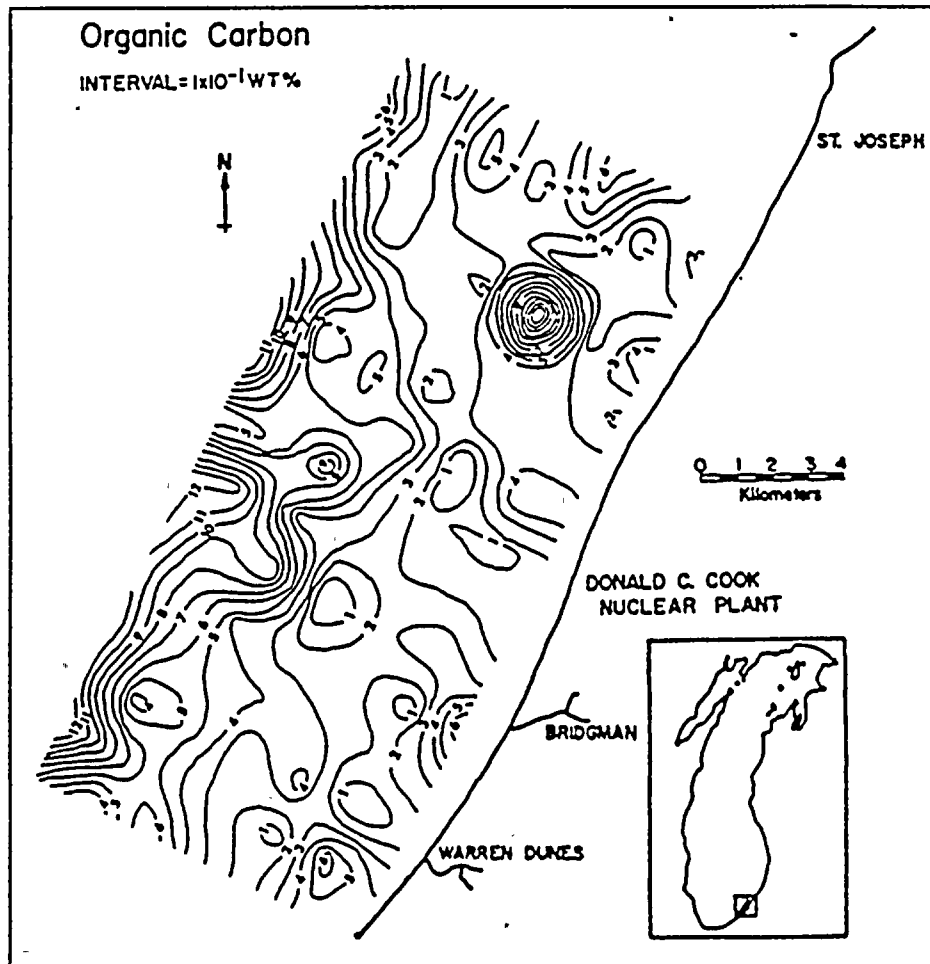


FIGURE 24. Distribution of organic carbon in nearshore southeastern Lake Michigan surficial sediments. To provide clarity, the numbers within the contour lines are an abbreviation of the given contour interval.

(Fig. 1) contain 1.4 to 4.0 times more iron than sediments from deeper within the cores (Figs. 5, 7, 8 and 10). Iron is enriched in the surficial sediments relative to deeper sediments of core 156 due to consistent inputs of fine-grained sediments by the St. Joseph River (Fig. 9). This degree of enrichment is markedly different from results obtained in the less oxidizing, fine-grained sediments of depositional basins of the Great Lakes where iron profiles display little or no enrichment (Robbins 1980).

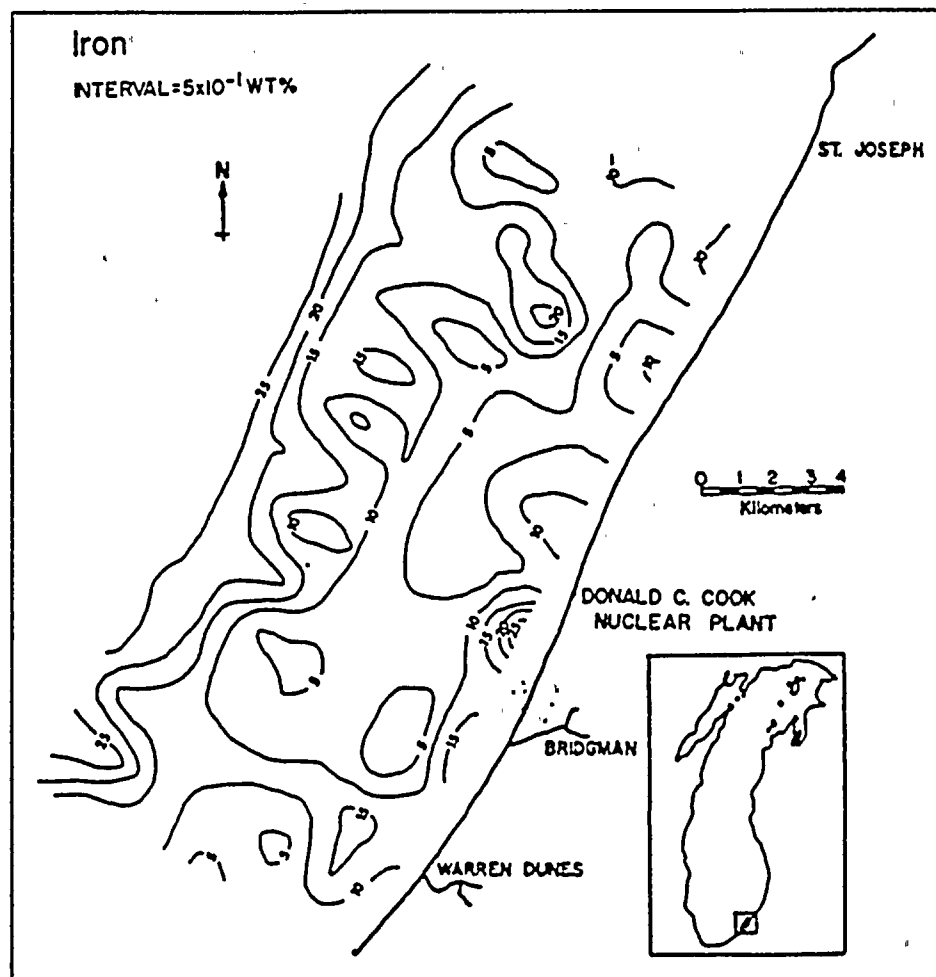


FIGURE 25. Distribution of iron in nearshore southeastern Lake Michigan surficial sediments. To provide clarity, the numbers within the contour lines are an abbreviation of the given contour interval.

Anthropogenic Enrichment of Surficial Sediments

Fine-grained surface sediments from the deep-water depositional basins of Lake Michigan and of the other Great Lakes are characterized by anthropogenically enriched concentrations of radioactive isotopes (Robbins and Edgington 1976) and heavy metals (Leland et al. 1973, Kemp and Thomas 1976, Farmer 1978, Robbins 1980). High metal concentrations in shallow (20 to 30 m), nearshore (10 to 12 km

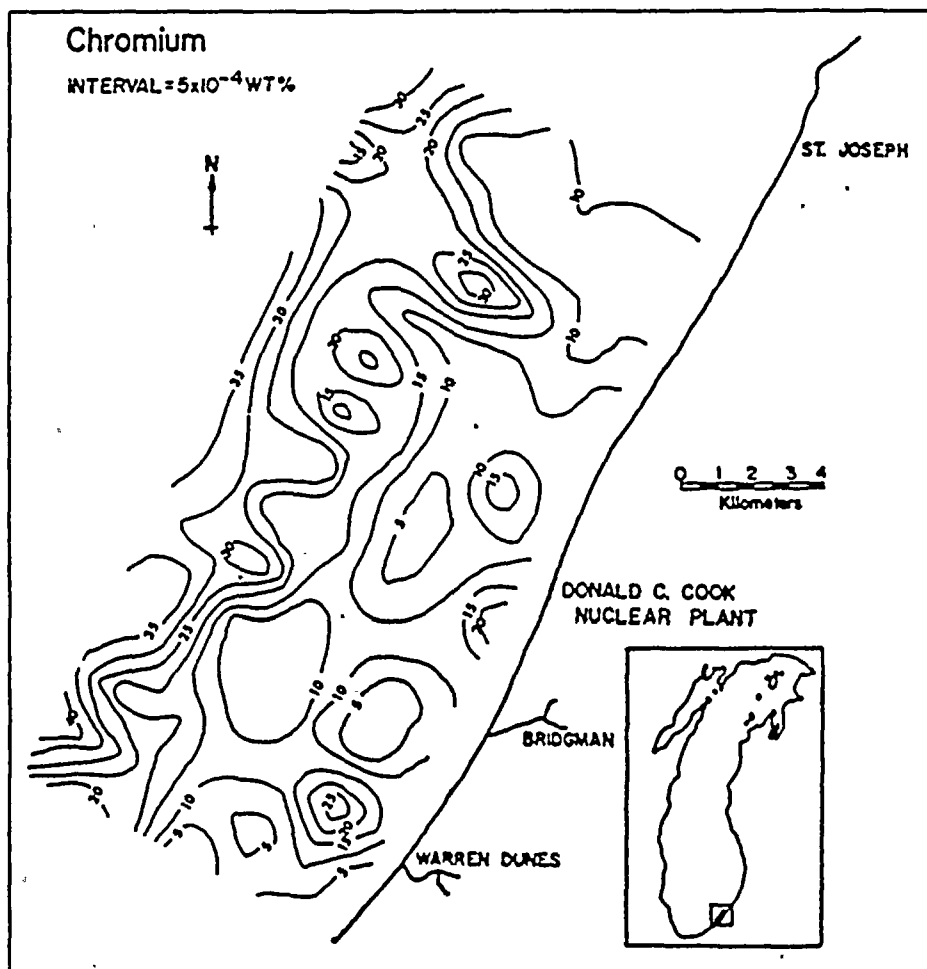


FIGURE 26. Distribution of chromium in nearshore southeastern Lake Michigan surficial sediments. To provide clarity, the numbers within the contour lines are an abbreviation of the given contour interval.

offshore) sediments of Lake Huron are also attributed to anthropogenic enrichment (Robbins 1980).

Surficial sediments collected from locations 7, 92, 93, 156, and 157 are enriched with iron, copper, chromium, zinc, and lead. Four of these five cores exhibit surficial enrichment with cobalt, nickel, strontium, and manganese (Figs. 5, 7, 8, 9, and 10). Elevated concentrations of Cu, Zn, and Pb in the surface sediments of depositional basins in Lake Michigan (Edgington and Robbins 1976, Leland et al. 1973).

and in the other Great Lakes (Robbins 1980, Farmer 1978, Kemp et al. 1976) as well as in many smaller North American (Iskander and Keeney 1974, Crecelius and Piper 1973) and European (Hamilton-Taylor 1979, Michler et al. 1980) lakes are attributable to cultural sources. All of the enriched elements in nearshore southeastern Lake Michigan sediments occur in significant amounts in atmospheric particulates originating in the Chicago-Gary area (Winchester and Nifong 1971, Gatz 1975, Fingleton and Robbins 1980).

The vertical profiles of the concentrations of iron, lead, manganese, zinc, molybdenum, copper, and chromium in core 93 display two subsurface maxima in addition to a surface maximum within the culturally enriched layer (0-14 cm) (Fig. 8). Radiometric dating of numerous cores from southern Lake Michigan places the date of the sediment horizon for anthropogenic lead at 1850 to 1900 depending upon location (Edgington and Robbins 1976). For core 93, the horizon for anthropogenic lead is 14 cm. Two minima above the initial increase in heavy metal profiles in core 93 may indicate temporary changes in the sedimentation pattern at this location during the last 75 to 150 years. It is likely that these minima resulted from resuspension and transport of older sediments to station 93. It is unlikely the variations observed are related to changes in the flux of anthropogenic lead. The source function developed by Edgington and Robbins (1976) does not account for the observed lead fluctuations in core 93. The heavy metal minima coincide with small but definite increases in the calcium and magnesium concentrations and decreases in sand content. Concentrations of calcium, magnesium, and sand are elevated in older sediments at site 93; all three increase with increasing sediment depth. As a result of

TABLE 3. *Inter-comparison of mean concentrations (ppm) of various parameters in the inshore and offshore sediments of southeastern Lake Michigan.*

Parameter	St. Joseph River		Nearshore	Offshore		
	St. Joseph River	upstream		Cahill (1981)	Callender (1969)	Frye and Shimp (1973)
	Fitchko and Hutchinson (1975)	Rossmann (1975)				
Organic Carbon	—	—	4,270.	39,000.	18,200.	25,100.
Barium	—	—	36.2	536.	—	—
Calcium	—	—	32,200.	52,290.	36,700.	55,900.
Loss on Ignition	—	—	71,600.	—	—	124,000.
Inorganic Carbon	—	—	16,100.	—	11,100.	29,500.
Cobalt	4.6	4.3	5.80	12.	—	13.
Chromium	0.5	6.3	16.4	83.	—	77.
Copper	1.2	6.3	5.34	39.	—	37.
Iron	—	—	12,400.	32,900.	20,900.	28,700.
Potassium	—	—	1,620.	2,080.	—	—
Magnesium	—	—	16,600.	26,500.	17,500.	—
Manganese	38.	44.	205.	124.	114.	89.2
Molybdenum	—	—	3.06	7.	—	—
Sodium	—	—	300.	375.	—	—
Nickel	2.8	6.0	10.3	34.	—	34.
Phosphorus	—	—	221.	1,320.	—	762.
Strontium	—	—	23.6	168.	—	—
Zinc	5.3	40.0	52.1	198.	—	206.

TABLE 4. Ratios of concentrations of various parameters in offshore sediments to those in-nearshore sediments.

Parameter	Offshore/Nearshore
Organic Carbon	4.3-9.1
Inorganic Carbon	0.7-1.8
Iron	1.7-2.6
Calcium	1.1-1.7
Magnesium	1.0-1.6
Loss on Ignition	1.7
Barium	14.8
Cobalt	2.1-2.2
Chromium	4.7-5.1
Copper	6.9-7.3
Potassium	2.6
Manganese	0.6-0.6
Molybdenum	2.3
Sodium	1.2
Nickel	3.3
Phosphorus	3.4-6.0
Strontium	7.1
Zinc	3.8-4.0

TABLE 5. Vertical distribution of Ca/Mg ratios in southeastern Lake Michigan sediments collected in 1975 from various water depths given in parentheses.

DEPTH (cm)	Core 5 (23 m)	Core 7 (28 m)	Core 90 (24 m)	Core 92 (32 m)	Core 93 (38 m)	Core 156 (26 m)	Core 157 (29 m)
0-1	1.90	1.58	2.31	1.67	1.58	1.70	1.71
1-2	1.82	1.72	2.36	1.67	1.74	1.63	1.67
2-3	1.89	1.70	3.05	1.54	1.71	1.69	1.71
3-4	1.80	1.71	3.10	1.64	1.77	1.74	1.70
4-5	1.77	1.79	4.89	1.68	1.75	1.72	1.67
5-6	1.72	1.80	2.94	1.66	1.71	1.69	1.68
6-7	1.68	1.80	2.58	1.71	1.71	1.69	1.67
7-8	1.69	1.69	2.00	1.69	1.74	1.69	1.66
8-9	—	1.67	1.82	—	1.68	1.72	1.69
9-10	—	1.62	1.83	—	1.70	1.72	—
10-12	—	1.65	1.84	—	1.65	1.72	—
12-14	—	—	1.88	—	1.68	1.70	—
14-16	—	—	—	—	1.69	1.68	—
16-18	—	—	—	—	1.63	—	—
18-20	—	—	—	—	1.66	—	—
20-25	—	—	—	—	1.66	—	—
25-30	—	—	—	—	1.66	—	—
30-35	—	—	—	—	1.66	—	—
35-40	—	—	—	—	1.69	—	—
40-45	—	—	—	—	1.64	—	—
45-50	—	—	—	—	1.68	—	—
50-55	—	—	—	—	1.62	—	—
55-60	—	—	—	—	1.59	—	—
60-65	—	—	—	—	1.61	—	—

resuspension and transport, modern sediments were diluted by pulses of fine sediments which are older and therefore not anthropogenically enriched with heavy metals. Although their duration is not known, these pulses are hypothesized to be the result of large, infrequent storms which resuspend sediment and which during periods of high lake water level may accelerate shoreline erosion (Davis et al. 1973). Evidence of these events has been observed in the sediments of northern Lake Michigan (Cline and Chambers 1977) and in sediments of the eastern basin of Lake Erie (Thomas et al. 1976).

SUMMARY

The sediments of nearshore southeastern Lake Michigan provide a highly variable substrate for benthic organisms. At depths less than 27 m, sediments are subject to resuspension by waves. This removes dolomite, organic matter, and other fine-grained components from the surficial sediments. Adjacent to shore, inputs of streams provide a constant source of organic matter, dolomite, and other fine-grained materials to the sediments, creating a substrate atypical of the nearshore region adjacent to the beach.

At water depths greater than 27 m, there is no evidence of sorting, and the fine-grained materials accumulate. Concentrations of organic matter, dolomite, iron compounds, and trace metals are higher in these surficial sediments than in those sediments from depths less than 27 m.

TABLE 6. *Calculated dolomite content in surficial and deeper nearshore southeastern Lake Michigan sediments.*

Core Location	Water Depth (m)	Percent Dolomite in Surficial Sediment	Percent Dolomite in Subsurface Sediments
5	23	8	34
7	28	34	38
90	24	1	15
92	32	34	38
93	38	35	40
156	26	23	23
157	29	34	38

TABLE 7. *Water depths and depth of sorting for nearshore southeastern Lake Michigan sediments.*

Core Location	Water Depth (m)	Depth of Sorting (cm)
5	23	6
7	28	0
90	24	8
92	32	0
93	38	0
156	26	10
157	29	0

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WATERS OF SOUTHEASTERN NEARSHORE LAKE MICHIGAN

Ronald Rossmann

INTRODUCTION

Nearshore waters of lakes are often complex. They have a chemical heterogeneity which is the result of redistribution of various inputs by physical processes such as currents and waves. These waters have higher nutrient, anion, and metal concentrations and Secchi disk transparencies than adjacent offshore waters. Each year a seasonal cycle occurs, beginning in the early spring with snow melt, spring rains, and onset of the warming of nearshore waters. As the waters warm, a nearshore thermal bar develops which is delineated by the 4°C isotherm. As the lake continues to warm, the thermal bar moves further offshore. While the bar is nearshore, in April, it serves as a barrier to the exchange of inshore waters with offshore waters. During this time, runoff water enriched in anions, nutrients, and trace metals enters the lake and is confined to the nearshore region by the thermal bar. This entrapment intensifies the impact of these inputs on the nearshore ecosystem. With continued warming and addition of nutrients, the spring phytoplankton bloom moves offshore along with the thermal bar.

The thermal bar dissipates in May or early June and a thermally stratified water column develops (Rossmann 1986). At this time, nearshore and offshore epilimnetic waters readily mix. Once thermal stratification is complete, nutrient depletion by algae may occur. Continuous inputs from streams usually maintain adequate supplies for phytoplankton growth in nearshore waters of phosphorus and nitrate but not of dissolved silica. During stratification, upwelling of hypolimnetic waters may occur within the nearshore zone, serving to infuse that area with nutrient-rich waters which include dissolved silica. These events change plankton assemblages and tend to increase the variability of all measured parameters. Therefore, these events must be documented and understood in order to interpret the chemical and biological parameters and processes in this very dynamic system.

The region of southeastern Lake Michigan which was studied intensively is located within 10 km of shore and is bounded by the St. Joseph River at St. Joseph, Michigan, and the Galien River and New Buffalo, Michigan (Fig. 1). Streams were occasionally sampled between these two points. Sampling of the lake occurred within a region 11 km to each side of the Donald C. Cook Nuclear Plant (Rossmann et al. 1986). Streams directly impacting this region are not gauged, which prevented making estimates of total amounts of elements entering the study area. Maximum water depths within the study area were slightly in excess of 38 m (Rossmann 1986). Only at the outermost stations (7, 21, 44, 116, 137, 158) could the hypolimnion be sampled during periods of thermal stratification (Fig. 2).

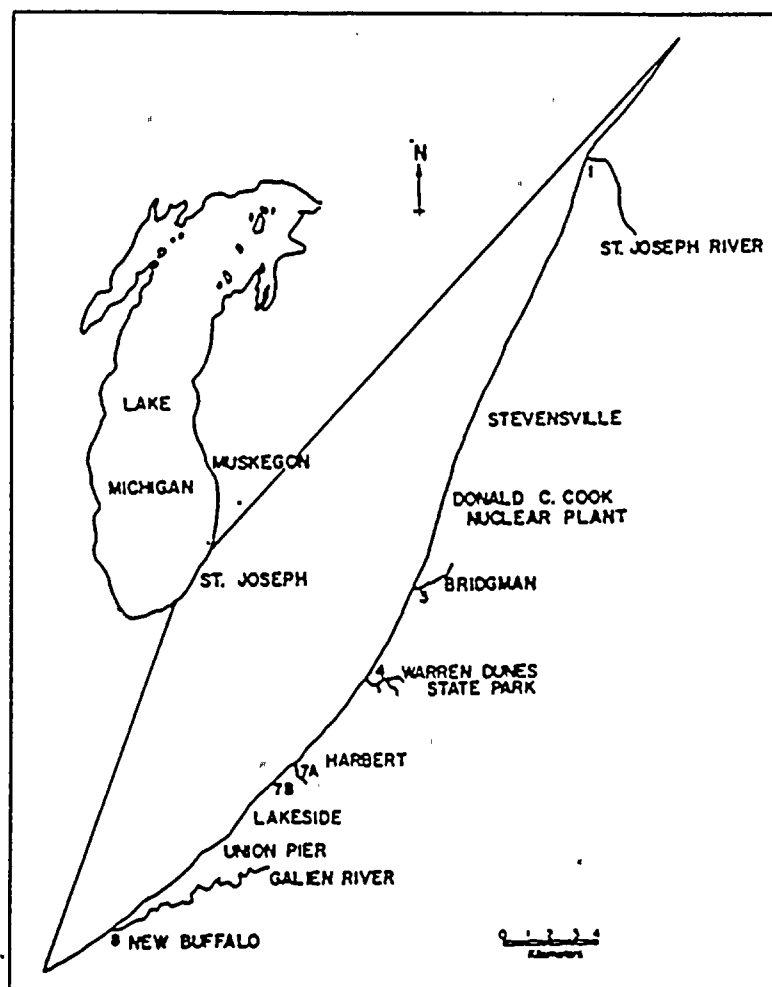


FIGURE 1. Sites (1, 3, 4, 7A, 7B, 8) on tributaries to southeastern Lake Michigan which were sampled from 1974 through 1976.

METHODS

COLLECTION

Samples were collected from a depth of 1 m using a Niskin[®] bottle. Immediately after collection, 100-mL subsamples for total phosphorus were stored in polyethylene bottles and frozen. Two additional 500-mL samples were filtered through 0.45- μ m pore-size membrane filters immediately after collection. Each of these was

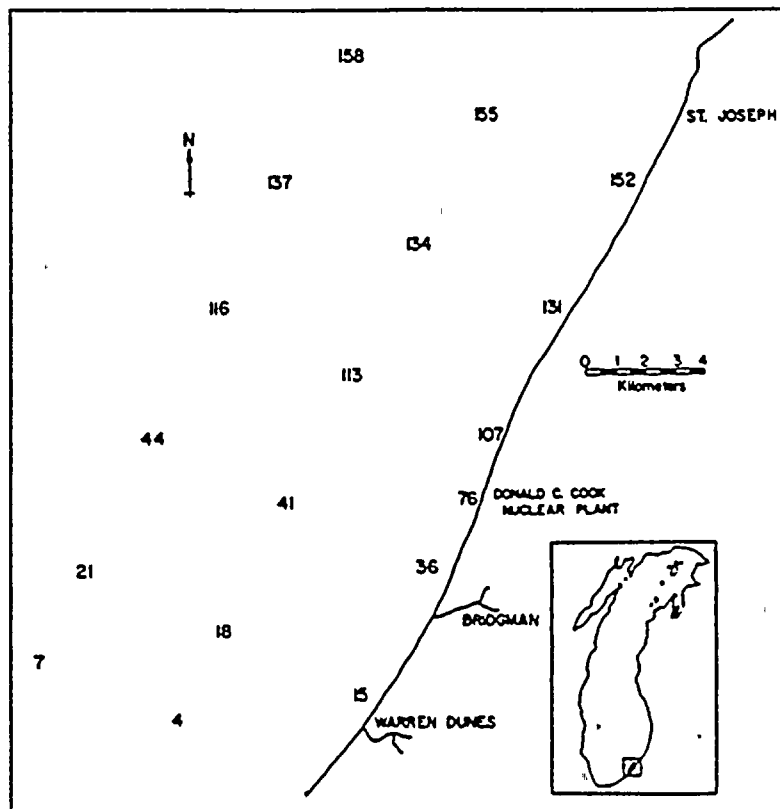


FIGURE 2. Station locations in southeastern Lake Michigan occupied from 1974 through 1976.

stored in a polyethylene bottle. One sample was refrigerated and the other was frozen. After defrosting, the frozen sample was analyzed for nitrate and nitrite. The refrigerated sample was analyzed for dissolved orthophosphate and dissolved silica within 24 to 48 hours of collection and for chloride and sulfate at a later date.

ANALYSES

Field measurements of Eh, pH, and temperature were made on water samples immediately after collection. All pH measurements were made using a rugged pH electrode and a calomel fiber junction saturated KCl reference electrode. Standardization for the measurements were made using commercially available pH buffer solutions. Eh measurements used the same reference electrode and a platinum inlay electrode. Standardization was against Zobell's solution (Zobell 1946). Temperature measurements were made with a standard glass laboratory thermometer. Oxygen and temperature measurements were obtained with a combination oxygen/

temperature meter. Alkalinity was done by adding 1-5 mL of 0.01N HCl to 20 mL of sample and reading the pH, which must fall between 3 and 4 (Kramer 1968). Conductivity was restored with a conductivity meter.

Using the ascorbic acid method, orthophosphate was complexed as phosphomolybdate and extracted in isobutanol using the methodology described by Sutherland et al. (1966). Dissolved silica was reacted with ammonium molybdate in an acid medium following the methodology given by Sutherland et al. (1966). Analysis for nitrate and nitrite followed that described by Strickland and Parsons (1972). By their method, samples for nitrite analysis were colorimetrically measured after reaction with sulfanilamide and N-(1-naphthyl) ethylene diamine dihydrochloride. Samples for nitrate analysis were first passed through a cadmium column to reduce the nitrate and nitrite. These were then treated in the same manner as the nitrite samples. Chloride was measured with a specific ion electrode using the method of standard additions. Sulfate was measured turbidimetrically by reaction with barium chloride (APHA 1965).

Samples for major, minor, and trace element analyses were filtered through 0.45- μ m filters, acidified with nitric acid (2 mL), and stored in polyethylene bottles. The filters for particulate matter analysis were stored in glass vials. Particulate matter was analyzed using the same techniques as those used for sediment analysis (Rossmann et al. 1986). Calcium, magnesium, sodium, and potassium in the water were measured using standard atomic absorption spectrophotometry techniques (Perkin-Elmer 1968). Copper, iron, manganese, chromium, zinc, strontium, molybdenum, nickel, barium, and cobalt were done by the method of standard additions using a graphite furnace attachment on the atomic absorption spectrophotometer.

STREAM INPUTS

Within the region of study, there are approximately eight tributary streams. A number of these are ephemeral. Six of the streams (sites 1, 3, 4, 7A, 7B, and 8; Fig. 1) were sampled during May and October of 1974, during April and August of 1975, and during July of 1976. In 1974, site 1 (St. Joseph River) was sampled monthly between April and October.

NUTRIENTS

Compared to nearshore southeastern Lake Michigan, the streams have high total phosphorus and much higher dissolved orthophosphate, dissolved silica, and nitrite concentrations (Table 1). Receiving effluent from a waste water treatment plant, the stream at site 3 had high mean concentrations of total phosphorus (980 μ g/L) and dissolved orthophosphate (570 μ g/L). Highest total phosphorus (3,200 μ g/L) and orthophosphate (1,400 μ g/L) concentrations were observed in October of 1974. Site 3 is expected to have direct impact upon the sediments, water, and biota of the intensively studied region of the lake.

Dissolved silica concentrations in this nearshore region often were limiting to phytoplankton growth even though stream concentrations were not limiting (Table 1). Parker et al. (1975) reported dissolved silica to be limiting below a concentration of 0.4 mg/L of SiO_2 . Tilman et al. (1982) reported dissolved silica limitations to be

species dependent. They found dissolved silica to be limiting to diatoms from concentrations as high as 0.5 ppm mg/L SiO_2 to concentrations as low as 0.01 mg/L SiO_2 .

In contrast to dissolved silica, total phosphorus and orthophosphate were seldom limiting to diatoms. Tilman et al. (1982) reported limiting total phosphorus concentrations ranges for diatoms, blue-green algae, and green algae to be 0.015–1.8 $\mu\text{g/L}$, 0.040–3.7 $\mu\text{g/L}$, and 1.8–35.6 $\mu\text{g/L}$, respectively. About one-half of the time, total phosphorus in the streams was limiting to green algae growth. It was never limiting to diatoms of blue-green algae. Within the lake, total phosphorus was limiting to green algae, rarely limiting to blue-green algae, and never limiting to diatoms.

ANIONS AND OTHER PARAMETERS

Compared to nearshore Lake Michigan, all streams had elevated sulfate and chloride concentrations (Table 2). Mean sulfate concentrations were highest in streams at sites 1 (42 mg/L) and 8 (48 mg/L) and lowest at sites 4, 7a, and 7b (27, 29, and 24 mg/L, respectively).

Concentrations of chloride at each tributary were at least twice those in the lake (Table 2). At site 4, mean chloride concentration (68 mg/L) was highest. Sites 2 and 7b also had high mean concentrations (51 and 48 mg/L, respectively).

Excluding the St. Joseph River (site 1), all streams exhibited some dissolved oxygen depletion. Dissolved oxygen saturations averaged below 90% at sites 3, 7b,

TABLE 1. Comparison of 1974 through 1976 nutrient concentrations in nearshore southeastern Lake Michigan epilimnetic waters with those of tributary streams.

Parameter	Units	Streams			Lake		
		N	Mean	Std. Dev.	N	Mean	Std. Dev.
Total Phosphorus	$\mu\text{g P/L}$	33	206.	574.	170	9.4	12.4
Orthophosphate	$\mu\text{g P/L}$	34	104.	271.	187	1.2	1.7
Dissolved Silica	mg SiO_2/L	34	7.1	3.0	186	0.39	0.34
Nitrate	$\mu\text{g N/L}$	6	245.	69.3	16	211.	40.2
Nitrite	$\mu\text{g N/L}$	6	34.5	23.1	18	1.3	2.5

TABLE 2. Comparison of 1974 through 1976 anion concentrations and other parameters in nearshore southeastern Lake Michigan epilimnetic waters with those of tributary streams.

Parameter	Units	Streams			Lake		
		N	Mean	Std. Dev.	N	Mean	Std. Dev.
Chloride	mg/L	33	38.0	22.4	170	11.7	1.6
Sulfate	mg/L	33	35.1	11.0	187	19.1	2.6
Oxygen Saturation	%	33	95.2	19.2	187	108.	9.
Alkalinity	meq/L	6	3.27	0.680	142	1.94	0.233
pH	.	6	8.1	0.24	160	8.4	0.21
Conductivity	$\mu\text{mhos cm}$	12	497.	78.	169	266.	21.

and 8, with mean saturations of 82, 78, and 88%, respectively. The lowest saturation was 54% at site 76 in August 1975.

The mean pH of the streams was slightly lower than that of the lake (Table 2). The pHs are typical of Lake Michigan waters in equilibrium or quasiequilibrium with detrital dolomite (Rossmann 1980). Mean alkalinities were considerably higher in the streams than in the lake (Table 2). Schelske and Roth (1973) described the lake to be well-buffered with calcareous waters.

All streams carried high concentrations of dissolved materials. The mean conductivity of the streams was nearly twice that of the lake (Table 2). For the St. Joseph River (site 1), conductivities ranged from 378 to 508 $\mu\text{mhos/cm}$ between April 1974 and July 1976. The values were lower than the range (465–605 $\mu\text{mhos/cm}$) reported for February through September 1971 by Robbins et al. (1972).

METALS

Within the region of study, tributary streams transported high concentrations of dissolved metals to the lake. Excluding molybdenum and strontium, all metals were higher in the streams than in the lake (Table 3). This is similar to the findings of Robbins et al. (1972) for samples collected from the St. Joseph River (site 1) during 1971. Metal reported by them are the same as those reported here. The major difference between the two data sets occurred for barium. The mean barium concentration reported by Robbins et al. (1972) was 28 $\mu\text{g/L}$. This is much less than the mean of 77 $\mu\text{g/L}$ obtained for the period of May 1974 to April 1975 for the St. Joseph River. For the streams reported here, iron and manganese were 19 and 72 times more concentrated in the streams than in the lake, respectively. The remaining dissolved metals were two to four times higher in the streams than in the lake. Thus the streams were transporting higher concentrations of dissolved metals to the

TABLE 3. Comparison of 1974 through 1976 dissolved metal concentrations in nearshore southeastern Lake Michigan epilimnetic waters with those of streams tributary to the region.

Parameter	Units	Streams			Lake		
		N	Mean	Std. Dev.	N	Mean	Std. Dev.
Barium	$\mu\text{g/L}$	21	68.	28.	88	43.	18.
Calcium	mg/L	28	53.	13.	186	35.	3.3
Cobalt	$\mu\text{g/L}$	16	2.4	1.1	88	1.2	0.56
Chromium	$\mu\text{g/L}$	22	2.1	0.73	88	1.7	0.63
Copper	$\mu\text{g/L}$	22	5.9	1.9	88	2.3	1.1
Iron	$\mu\text{g/L}$	16	130.	118.	88	6.7	3.3
Potassium	mg/L	22	2.3	1.4	187	1.2	0.13
Magnesium	mg/L	28	19.	5.3	187	11.7	0.50
Manganese	$\mu\text{g/L}$	16	37.	50.	92	0.51	0.50
Molybdenum	$\mu\text{g/L}$	16	13.	9.6	88	13.	7.6
Sodium	mg/L	28	19.	17.	186	5.4	0.67
Nickel	$\mu\text{g/L}$	16	26.	14.	88	7.5	3.7
Strontium	$\mu\text{g/L}$	16	110.	26.	88	108.	25.
Zinc	$\mu\text{g/L}$	16	13.	23.	92	4.5	2.5

nearshore zone of southeastern Lake Michigan. This can be expected to impact the nearshore ecosystem, especially during periods in which the thermal bar confines these inputs to a narrow band along the shoreline.

IMPACT OF THE THERMAL BAR

As the waters of southeastern Lake Michigan warm in the spring, a thermal bar develops. Its location is delineated by the 4°C isotherm. For the region of study, the bar was present during April (Rossmann 1986). By definition, inshore zone waters were always warmer than offshore zone waters in the early spring. Temperature differences between the two zones were usually within the range of 2.6 to 2.9°C (Table 4). Years which were notably above the range of difference were 1977 and 1981 with 6.0 and 3.7°C, respectively. Years for which the difference was below normal were 1975 and 1979 with 1.9 and 2.1°C, respectively. During periods when the thermal bar was within 2 km of the shoreline, stream inputs to the nearshore zone were confined between the shoreline and the thermal bar (inshore zone). If the inputs were nutrients, the concentration in the inshore zone was determined by the rate of input relative to the rate of assimilation by phytoplankton. The rate of assimilation was in turn dependent upon water temperature.

Based on mean temperature, the inshore zone was warmest in 1977 and coldest in 1975 (Table 4). Thus differences in the inshore and offshore phytoplankton assemblages would be expected to be accentuated most in 1977 and least in 1975 and 1979.

A second confounding difference between the two zones is that the inshore zone includes the entire zone of breaking waves, whereas the offshore zone is less physically active. Thus sediment resuspension is most pronounced in the inshore zone, resulting in elevated concentrations of sediment-associated elements.

Resuspension and increased phytoplankton numbers in the inshore zone significantly decrease water transparency relative to that in the offshore zone (Table 5) (Bowers et al. 1986). Division of the nearshore zone into inshore and offshore zones by the thermal bar was most pronounced in 1977.

Because of the inter-relationship between nutrients, temperature, and phytoplankton densities, differences in nutrient concentrations between the inshore and offshore zones are inconsistent. At times, nutrient concentrations are highest in the

TABLE 4. Inshore and offshore mean temperatures (°C) during spring (April) thermal bar formation.

Year	Inshore Zone			Offshore Zone		
	N	Mean	Std. Dev.	N	Mean	Std. Dev.
1974	11	6.2	1.2	6	3.3	0.3
1975	2	4.4	0.6	16	2.5	0.6
1976	14	6.1	1.2	4	3.4	0.3
1977	15	7.9	1.0	6	1.9	0.7
1978	20	5.4	0.7	10	2.8	0.9
1979	13	5.0	0.8	17	2.9	1.0
1980	12	5.4	1.2	18	2.8	0.7
1981	24	6.4	1.1	6	2.7	0.7
1982	18	5.6	1.4	12	2.8	1.1

inshore zone and at other times they are lowest in the inshore zone. On the other hand, non-nutrient materials or those far in excess of the nutritional requirements of the phytoplankton are expected to be elevated in the inshore zone relative to the offshore zone for those elements supplied to the lake at concentrations above those of ambient lake water.

NUTRIENTS

Phosphorus and dissolved silica may be limiting to phytoplankton growth in both the inshore and offshore of the thermal bar. Dissolved orthophosphate concentrations were lowest in the inshore zone during the springs of 1974 and 1980 (Table 6). Though streams supplied high concentrations of orthophosphate to the inshore zone, its concentration in the inshore zone was only one-third to one-half that in the offshore zone.

Although total phosphorus is recognized to be a nutrient, it is not as readily available to phytoplankton except after conversion to a more utilizable form. Some of the total phosphorus is contained within the phytoplankton. Though not statisti-

TABLE 5. Inshore and offshore 20-cm diameter Secchi disk depths (m) during spring (April) thermal bar formation.

Year	Inshore Zone			Offshore Zone		
	N	Mean	Std. Dev.	N	Mean	Std. Dev.
1974	11	2.4	0.4	6	4.5	0.6
1975	1	2.0		14	2.7	1.0
1976	14	1.8	0.6	4	3.8	0.7
1977	15	2.4	0.7	6	5.4	0.3
1978	20	1.2	0.6	8	3.3	0.7
1979	13	0.8	0.1	15	2.5	2.4
1980	11	1.1	0.3	17	2.3	1.5
1981	24	2.5	0.6	3	4.1	0.1
1982	17	1.3	0.2	12	2.9	1.6

TABLE 6. Inshore and offshore orthophosphate concentrations ($\mu\text{g/L}$) during spring (April) thermal bar formation.

Year	Inshore Zone			Offshore Zone		
	N	Mean	Std. Dev.	N	Mean	Std. Dev.
1974	10	0.58	0.42	6	0.92	0.58
1975	2	0.86	0.21	15	0.87	0.63
1976	13	1.6	0.63	4	1.2	0.28
1977	15	1.1	0.65	6	1.3	0.50
1978	17	2.9	0.57	9	2.3	0.74
1979	13	1.0	0.50	17	1.1	0.68
1980	12	0.51	0.052	18	1.4	0.75
1981	24	1.1	0.53	6	0.61	0.22
1982	18	1.1	0.53	12	0.87	0.35

cally significant, total phosphorus was higher inshore of the thermal bar during all years (Table 7). Within both the inshore and offshore zones, it was never limiting to diatom and green algae growth (Tilman et al. 1982).

Dissolved silica concentrations were highest in the offshore zone during the springs 1974-1977 and 1981 (Table 8). Concentrations in the offshore zone were not limiting to diatoms between 1974 and 1982. During the springs of 1974 and 1977, it may have been limiting to some species of diatom in the inshore zone (Tilman et al. 1982). In general, nutrients were not limiting to green algae or diatoms during the existence of the spring thermal bar.

ANIONS AND OTHER PARAMETERS

Differences between the inshore and offshore zones of nearshore southeastern Lake Michigan were evident for chloride, sulfate, and conductivity. Differences were not found for dissolved oxygen saturation with a range of 98 to 120%, pH with a range of 7.98 to 8.64, and alkalinity with a range of 1.66 to 2.22 meq/L. Mean conductivity was always highest in the inshore zone (Table 9). Differences were similar for all 3 years of data; however, mean conductivities increased each year.

TABLE 7. Inshore and offshore total phosphorus concentrations ($\mu\text{g/L}$) during spring (April) thermal bar formation.

Year	Inshore Zone			Offshore Zone		
	N	Mean	Std. Dev.	N	Mean	Std. Dev.
1975	2	23.	2.1	16	15.	4.3
1976	14	11.	4.3	4	5.6	1.5
1977	15	11.	4.0	6	6.1	2.4
1978	20	14.	5.7	10	4.3	3.3
1979	13	22.	8.1	17	16.	4.3
1980	12	15.	2.1	18	8.2	3.8
1981	24	8.5	3.4	6	8.4	2.7
1982	18	13.	7.3	12	12.	6.1

TABLE 8. Inshore and offshore dissolved silica concentrations (mg/L) during spring (April) thermal bar formation.

Year	Inshore Zone			Offshore Zone		
	N	Mean	Std. Dev.	N	Mean	Std. Dev.
1974	11	0.15	0.13	6	1.1	0.041
1975	2	0.50	0.13	16	0.66	0.17
1976	14	0.96	0.24	4	1.3	0.13
1977	15	0.38	0.13	6	0.75	0.088
1978	20	1.2	0.26	10	0.94	0.15
1979	13	1.8	0.41	17	1.4	0.46
1980	12	1.2	0.12	18	1.2	0.15
1981	24	1.0	0.15	6	1.2	0.068
1982	18	0.60	0.27	12	0.49	0.21

Except for 1981, mean sulfate concentrations was always higher in the inshore region than in the offshore region (Table 10). Differences were usually within the range of 2.2 to 2.8 mg/L. The exceptions occurred in 1975 and 1981. Within the inshore zone, concentrations were highest during 1975, a year the thermal bar had just begun to form during the period of sampling, and 1980-1982, years in which sulfate concentrations were also relatively high in the offshore zone.

For April 1974 through 1982, inshore zone mean chloride concentrations were always higher than those in the offshore zone (Table 11). For most years, the difference between the two zones ranged from 0.3 to 0.6 mg/L. The exceptions were 1974, 1977, 1978, and 1982 when differences were 1.8, 1.1, 1.5, and 2.8 mg/L, respectively.

METALS

As for total phosphorus, chloride, sulfate, and conductivity, both particulate and dissolved metal concentrations were elevated in the inshore zone relative to the offshore zone. Stream inputs were responsible for elevated particulate and dissolved metal concentrations, and sediment resuspension is postulated to have contributed to elevated particulate metal concentrations. Metals were measured during 1974, 1975, and 1976. Only the April 1974 and 1976 data were useful for comparing the inshore and offshore zones. Because the 1974 particulate and dissolved metal data set was the largest, it will be used to contrast the two zones.

TABLE 9. Inshore and offshore conductivities ($\mu\text{mohms/cm}$) during spring (April) thermal bar formation.

Year	Inshore Zone			Offshore Zone		
	N	Mean	Std. Dev.	N	Mean	Std. Dev.
1974	11	236.	10.	6	224.	8.
1975	2	265.	8.	15	250.	14.
1976	14	308.	20.	4	289.	2.

TABLE 10. Inshore and offshore sulfate concentrations (mg/L) during spring (April) thermal bar formation.

Year	Inshore Zone			Offshore Zone		
	N	Mean	Std. Dev.	N	Mean	Std. Dev.
1974	10	20.7	1.1	5	18.5	1.0
1975	2	24.1	2.1	16	20.0	2.1
1976	14	20.5	2.6	4	18.2	0.6
1977	15	19.2	1.0	6	17.0	0.3
1978	20	21.3	1.6	10	19.6	1.0
1980	12	23.4	2.6	18	21.2	2.7
1981	24	22.3	3.0	6	22.7	2.1
1982	18	22.8	1.5	12	20.0	1.7

Dissolved calcium and sodium concentrations were distinctly higher in the inshore zone than in the offshore zone; however, only the sodium difference was statistically significant ($\alpha \leq 0.05$) (Table 12). Other metals whose concentrations were greater, though not statistically significant, in the inshore zone were barium, copper, molybdenum, nickel, and zinc. Chromium and cobalt concentrations were lower in the inshore zone.

Some particulate metal concentrations were statistically significantly higher in the inshore zone (Table 13). These included calcium, iron, manganese, and potassium. Copper, sodium, and strontium concentrations were also higher in the inshore zone. Though not statistically significant, the remaining metal concentrations were higher

TABLE 11. Inshore and offshore chloride concentrations (mg/L) during spring (April) thermal bar formation.

Year	Inshore Zone			Offshore Zone		
	N	Mean	Std. Dev.	N	Mean	Std. Dev.
1974	11	11.7	0.7	6	9.9	0.6
1975	2	12.0	0.8	16	11.4	1.4
1976	14	12.1	1.3	4	11.8	1.6
1977	15	13.6	0.9	6	12.5	1.8
1978	20	12.4	0.8	10	10.9	0.9
1979	13	11.9	3.3	17	11.3	0.9
1980	12	13.0	2.4	18	12.4	2.1
1981	24	10.8	0.9	6	10.4	0.2
1982	18	17.9	0.9	12	15.1	1.3

TABLE 12. Inshore and offshore dissolved metal concentrations during April 1974 thermal bar formation.

Metal	Units	Inshore Zone N = 11		Offshore Zone N = 6	
		Mean	Std. Dev.	Mean	Std. Dev.
Ba	µg/L	40.	12.	33.	15.
Ca	mg/L	37.2	2.2	34.5	1.4
Co	µg/L	0.93	0.34	1.7	0.84
Cr	µg/L	1.8	0.61	2.3	0.86
Cu	µg/L	3.6	2.0	3.0	1.2
Fe	µg/L	7.7	2.2	7.6	2.3
K	mg/L	1.2	0.080	1.1	0.011
Mg	mg/L	10.4	0.44	10.0	0.22
Mn	µg/L	0.45	0.16	0.40	0.12
Mo	µg/L	10.	3.6	8.7	2.2
Na	µg/L	5.7	0.46	4.6	0.066
Ni	µg/L	11.	5.3	8.1	4.9
Sr	µg/L	106.	18.	105.	13.
Zn	µg/L	6.7	2.0	6.2	2.9

offshore. It is doubtful that the noted metal concentration differences could create any observable differences in numbers or species between the inshore and offshore ecosystems. Most likely, any possible impact would be due to the higher concentration of suspended particulates in the inshore zones.

THERMAL STRATIFICATION

During periods of thermal stratification, large differences in nutrients, anions, metals, and other parameters' concentrations exist between the epilimnion and hypolimnion. Within the southeastern nearshore region of Lake Michigan, stratification usually begins in May and continues into October. In November, cooling and intensified physical mixing of the lake water destroy the thermal structure and return the region to an isothermal, well-mixed condition. The months of most intense stratification, July-September, are expected to have distinctive vertical variations for both lake water chemistry and the biological components of the ecosystem, especially the phytoplankton which are strongly coupled to circulation patterns. During the upwelling of hypolimnetic waters along the shoreline, the contrast between the epilimnetic and hypolimnetic ecosystems is the most distinct.

EPILIMNION AND HYPOLIMNION CHARACTERISTICS

During periods of thermal stratification, the exchange of materials between the hypolimnion and epilimnion is restricted. The resuspension and exchange within the water column of materials associated with organic-rich, fine-grained sediments are restricted to the hypolimnion. Inputs from the atmosphere and streams impact the epilimnion. Within the epilimnion, productivity and the demand upon available nutrients are higher than in other parts of the water column. For the nearshore zone, the rate of assimilation of nutrients by phytoplankton is compensated in part by the rate of stream inputs. The difference between these two rates controls the observed epilimnion nutrient concentrations.

TABLE 13. Inshore and offshore particulate metal concentrations ($\mu\text{g/L}$) during April 1974 thermal bar formation.

Metal	Inshore Zone N = 11		Offshore Zone N = 6	
	Mean	Std. Dev.	Mean	Std. Dev.
Ca	271.	31.	211.	19.
Cr	1.5	0.87	1.6	1.6
Cu	1.8	0.23	1.3	0.28
Fe	105.	26.	54.	17.
K	51.	10.	24.	8.6
Mg	92.	10.	121.	152.
Mn	3.1	0.93	1.1	0.35
Na	75.	18.	65.	13.
Sr	0.49	0.18	0.35	0.17
Zn	18.	2.	21.	11.

Using July 1974 as representative of what typically occurs during thermal stratification, statistically significant physical and chemical differences ($p \leq 0.05$) existed between epilimnetic and hypolimnetic waters. The differences observed were similar to those observed during other periods of thermal stratification.

Nutrients

Of the three nutrients for which data are available, only dissolved silica was significantly ($\alpha \leq 0.05$) higher in the hypolimnion (Table 14). Total phosphorus was not limiting to diatoms or green algae while dissolved silica was limiting (< 0.05 mg/L) to diatoms in the epilimnion but not in the hypolimnion (Tilman et al. 1982). Thus upwelling of hypolimnetic waters would serve to replenish the supply of dissolved silica.

Anions and Other Parameters

Except for conductivity, all anions and other parameters were significantly ($\alpha \leq 0.05$) different in the epilimnion relative to the hypolimnion (Table 15). Mean temperature was 15.2°C higher in the epilimnion than in the hypolimnion. Because of stream and possibly atmospheric inputs, chloride and sulfate were higher in the epilimnion than in the thermally isolated hypolimnion. In response to the balance of production and utilization, dissolved oxygen was higher in the epilimnion. The Eh was higher in the hypolimnion than in the epilimnion due to differences in anion/cation balances. As expected of waters in contact with carbonate containing sedi-

TABLE 14. Comparison of epilimnion with hypolimnion nutrient concentrations in Lake Michigan nearshore waters, July 1974.

Parameter	Units	Epilimnion			Hypolimnion		
		N	Mean	Std. Dev.	N	Mean	Std. Dev.
Total Phosphorus	$\mu\text{g P/L}$	18	10.1	5.8	6	10.8	4.2
Orthophosphate	$\mu\text{g P/L}$	18	1.3	2.5	6	0.6	0.7
Dissolved Silica	$\text{mg SiO}_2/\text{L}$	17	0.1	0.1	6	1.4	0.3

TABLE 15. Comparison of epilimnion with hypolimnion anion concentrations and other parameters in Lake Michigan nearshore waters, July 1974.

Parameter	Units	Epilimnion			Hypolimnion		
		N	Mean	Std. Dev.	N	Mean	Std. Dev.
Temperature	$^\circ\text{C}$	18	21.2	0.8	6	6.0	0.2
Chloride	mg/L	18	11.6	0.6	6	10.7	0.5
Sulfate	mg/L	18	18.4	0.8	6	17.6	0.4
Oxygen Saturation	%	18	121.	7.	6	89.	1.
Alkalinity	meq/L	18	1.71	0.16	6	2.06	0.05
Eh	mv	18	417.	12.	6	449.	11.
Conductivity	$\mu\text{mhos/cm}$	18	287.	5.	6	284.	3.
pH		18	8.56	0.08	6	8.02	0.05

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ments, alkalinities were higher in the hypolimnion. The pH of the hypolimnion was lower due primarily to contact with the sediments and, in part, lower production.

Metals

There was only one statistically significant difference in dissolved metal concentrations between the epilimnion and hypolimnion (Table 16). The epilimnion had a mean manganese concentration of nearly twice that of the hypolimnion. This is postulated to be a result of stream inputs to this nearshore zone.

Unlike dissolved metal concentration differences, those for particulate metal concentrations were significantly different in the two strata (Table 17). Excluding chromium and zinc, differences were statistically significant ($\alpha \leq 0.05$) for calcium,

TABLE 16. Comparison of epilimnion with hypolimnion dissolved metal concentrations in Lake Michigan nearshore waters, July 1974.

Parameter	Units	Epilimnion			Hypolimnion		
		N	Mean	Std. Dev.	N	Mean	Std. Dev.
Barium	µg/L	18	37.	17.	6	45.	25.
Calcium	mg/L	18	31.8	4.2	6	33.4	0.9
Cobalt	µg/L	18	1.4	0.4	6	1.7	1.0
Chromium	µg/L	18	1.6	0.4	6	1.5	0.7
Copper	µg/L	18	2.0	0.9	6	2.3	1.0
Iron	µg/L	18	9.5	4.2	6	13.	4.
Potassium	mg/L	18	1.2	0.2	6	1.2	0.04
Magnesium	mg/L	18	11.0	1.4	6	11.6	0.3
Manganese	µg/L	18	0.47	0.20	6	0.27	0.03
Molybdenum	µg/L	18	9.1	4.9	6	12.	6.
Sodium	mg/L	18	4.7	0.7	6	4.8	0.5
Nickel	µg/L	18	5.5	2.1	6	7.7	3.7
Strontium	µg/L	18	102.	28.	6	107.	32.
Zinc	µg/L	18	5.9	2.3	6	6.6	2.0

TABLE 17. Comparison of epilimnion with hypolimnion particulate metal concentrations (µg/L) Lake Michigan nearshore waters, July 1974.

Parameter	Epilimnion			Hypolimnion		
	N	Mean	Std. Dev.	N	Mean	Std. Dev.
Calcium	18	81.	47.	6	262.	45.
Chromium	18	0.71	0.90	6	1.5	0.9
Copper	18	11.	1.	6	2.7	3.2
Iron	18	23.	8.	6	147.	22.
Potassium	18	23.	8.	6	90.	13.
Magnesium	18	20.	6.	6	132.	18.
Manganese	18	2.1	1.4	6	4.2	0.5
Sodium	18	61.	8.	6	70.	9.
Strontium	18	0.10	0.11	6	0.48	0.16
Zinc	18	7.8	3.7	6	9.4	3.8

copper, iron, potassium, magnesium, manganese, sodium, and strontium. Resuspension of sediments appears to be a major source of particulate calcium, iron, potassium, magnesium, sodium, and strontium in the hypolimnion. Stream and atmospheric inputs are the most likely sources of higher particulate copper (Table 3) concentrations in the epilimnion. Another possible source is the mass of plankton which bioconcentrates copper in the epilimnion (Greenslate et al. 1973). Copeland and Ayers (1972) found copper to be concentrated 1,000- to 1,200-fold in zooplankton and phytoplankton relative to Lake Michigan water.

The contrast in chemical and physical characteristics between the epilimnion and hypolimnion is dramatic. Greater plankton biomass and phytoplankton productivity in the epilimnion are postulated to be responsible for the observed depletion of dissolved silica, elevation of oxygen saturation and pH, and high concentration of particulate copper in the epilimnion. Stream and atmospheric inputs contribute to high concentrations of particulate copper, dissolved manganese, sulfate, and chloride. Resuspension of sediments or the occurrence and sampling of a near-bottom nepheloid layer (Eadie and Robbins in press) are responsible for elevated Eh and high concentrations of particulate calcium, iron, potassium, magnesium, manganese, sodium, and strontium.

IMPACT OF UPWELLING

The differences which exist between the epilimnion and hypolimnion become particularly important when upwelling events occur. Upwelling occurs within the nearshore zone of southeastern Lake Michigan. For the period of 1974 to 1982, upwelling events during sampling were recorded for July 1974, June 1975, June 1976, July 1976, August 1976, September 1976, June 1977, September 1977, October 1977, June 1978, July 1978, September 1979, October 1979, June 1980, September 1980, and July 1981. Upwelling events transport water which is characteristically different from that of the epilimnion to the nearshore zone.

An example of the differences between epilimnetic and upwelled hypolimnetic waters is seen in an upwelling in July 1978 which coincided with our sampling. For the purpose of subdividing the region, the data were divided into three subsets based on water temperature. The upwelled hypolimnion consisted of those stations with a surface water temperature less than 10°C, the mixed zone was those with temperatures between 10 and 15°C, and the epilimnion was those with a temperature greater than 15°C. The mean values of various epilimnetic parameters were then contrasted to those of various hypolimnetic parameters (Table 18). For the two sets of parameters, the difference in mean temperature was 11.1°C. This difference and those for dissolved silica, nitrate, chloride, oxygen saturation, and Secchi disk were significant at the 0.05 level of significance. As for the earlier comparison of the epilimnion to hypolimnion (Tables 14 and 15), dissolved silica was lower and chloride and oxygen saturation were higher in the epilimnion. The lower dissolved silica concentration and higher oxygen saturation in the epilimnion are postulated to result from high diatom utilization of available nutrients. Nitrate concentrations were lower in the epilimnion in response to phytoplankton utilization. The mean Secchi disk reading for upwelled hypolimnetic water was significantly lower than for the epilimnion. This is consistent with higher particulate metal concentrations in the hypolimnion (Table 17) which are related to a near-bottom nepheloid layer or resuspended sediment. The particulates would be transported toward the shore and upwelled in the

nearshore zone. These contrasts between epilimnetic and hypolimnetic waters, the constant inputs of streams, and the development of a thermal bar in the spring all serve to produce semi-permanent water masses within this nearshore region.

WATER MASSES

Within the nearshore region of southeastern Lake Michigan, water masses are identifiable from April through October. The most definitive water masses occur during the existence of the thermal bar in April. The second-most definitive water masses occur in the summer during periods of thermal stratification when hypolimnetic waters upwell along the coastline. During periods when the thermal bar or upwelling are absent, less definitive water masses are present. Their occurrence is dependent upon the processes which mix lake water with stream and river inputs.

Differences between water masses can be expected to be a factor responsible for observed variations in phytoplankton community structure. Those variables most impacted by the phytoplankton include the nutrients total phosphorus, orthophosphate, dissolved silica, and nitrate. Nutrients concentrations were used as input for deriving station groupings by cluster analysis. Each cluster was then designated a water mass, and each water mass was described using descriptive statistics (minimum, maximum, mean, standard deviation) for nutrients and temperature.

Stations occupied from 1974 through 1976 were the most suitable for the purpose of defining water masses. The stations were arranged in a grid pattern and did not provide a regional sampling bias (Fig. 2). Stations occupied from 1977 through 1981 provided intensive coverage of the inshore region but inadequate coverage of the offshore region.

WATER MASSES DERIVED FROM THERMAL BARS

From 1974 through 1982, a thermal bar was identified within the study area during most Aprils. Its location was based on surface water temperature. The water masses defined for 1974 and 1976 were inshore and offshore of the thermal bar

TABLE 18. Comparison of epilimnion with hypolimnion anion concentrations and other parameters in Lake Michigan nearshore waters during July 1974.

Parameter	Units	Epilimnion			Hypolimnion		
		N	Mean	Std. Dev.	N	Mean	Std. Dev.
Temperature	°C	6	17.6	0.8	12	8.5	0.6
Total Phosphorus	µg P/L	6	8.6	3.4	12	10.4	4.2
Orthophosphate	µg P/L	6	0.9	0.4	12	0.7	0.4
Dissolved Silica	mg SiO ₂	6	0.4	0.2	12	1.0	0.1
Nitrate	µg N/L	6	166.	55.	12	243.	58.
Chloride	mg/L	6	9.4	0.1	12	9.2	0.2
Sulfate	mg/L	6	24.2	0.5	12	24.5	0.5
Oxygen Saturation	%	6	105.	4.	12	95.	7.
Secchi Disk	m	6	4.8	1.0	12	2.1	0.2

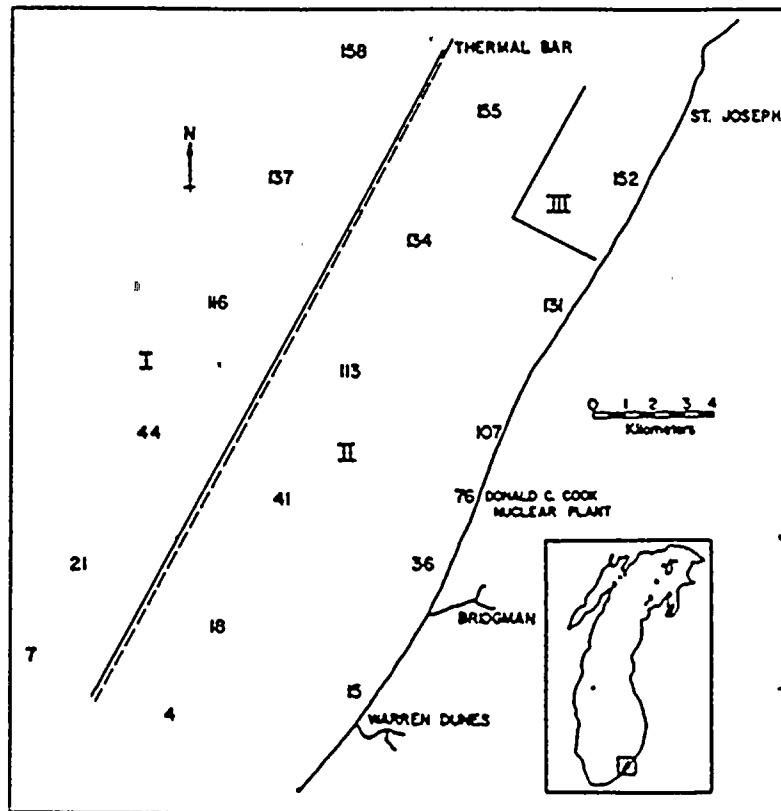


FIGURE 3. Water masses in southeastern Lake Michigan during April 1974. Masses were defined from cluster analysis of nutrient concentrations. Dashed line is the approximate location of the observed thermal bar.

(Figs. 3 and 4). Their definition was dependent upon the bar acting as a barrier to the free exchange of waters inshore and offshore of the bar.

Differences noted for the chemistry of the water masses in any one year were dependent upon the period of time during which the bar had existed prior to sampling. A measure of this time is the distance the thermal bar is offshore (Table 19). No bar existed in 1975, while its greatest development occurred during sampling in 1974 and 1976.

The 3 years considered in detail (1974-1976) encompass the entire range of thermal bar development. Based upon the significance or non-significance of temperature differences between inshore and offshore water masses, the resolving power of the cluster analysis to define the water masses was judged to be either "good" or "bad" (Table 19). For the sampling design of 1974-1976, the ratio of "good" to "bad" is 3:0; whereas it is only 1:2 for the design used from 1977 through 1982. The uneven distribution of stations in the latter design is believed to be the source of poorer resolution.

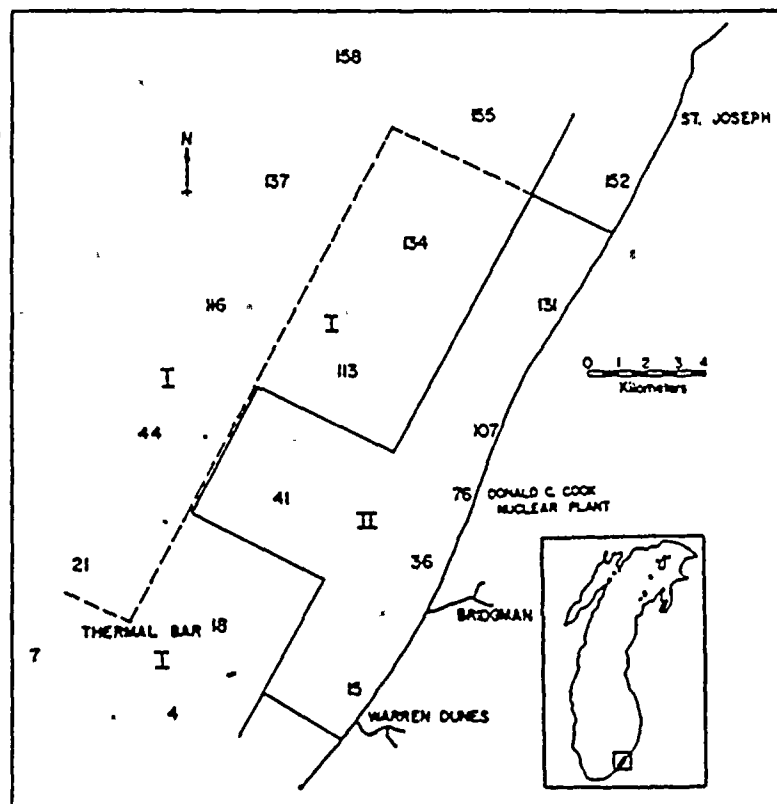


FIGURE 4. Water masses in southeastern Lake Michigan during April 1976. Masses were defined from cluster analysis of nutrient concentrations. Dashed line is the approximate location of the observed thermal bar.

The Aprils of 1974 and 1976 had thermal bars, while the April of 1975 had none. For 1974 and 1976, dissolved silica was a significant factor in defining the water masses. During both years, mean dissolved silica concentrations were higher in the offshore zones (I) than in the inshore zone (II) (Tables 20 and 21). Warmer waters in the inshore zone increased phytoplankton utilization of nutrients. Utilization of dissolved silica by diatoms depressed the inshore dissolved silica concentration despite the high flux of dissolved silica to the region from streams and rivers. Diatom concentrations were higher inshore of the thermal bar during 1974 and 1976 (Bowers et al. 1986). For 1976, total phosphorus was a major factor for defining the water masses. It was considerably higher in the inshore water mass (II) than in the offshore one (I) (Table 21). Its high level inshore is attributable to stream and river inputs.

For April 1974, a third water mass (III) was identified (Fig. 3). A high orthophosphate concentration distinguished this water mass, consisting of one station, from the other two (Table 20). Though its orthophosphate concentration was similar to

that of the St. Joseph River (19 $\mu\text{g P/L}$), its dissolved silica concentration was depressed to an undetectable level postulated to be caused by intense diatom utilization in a very phosphorus-rich region. This occurred despite high concentrations of dissolved silica (4.35 mg SiO_2) being supplied to the region by the St. Joseph River.

TABLE 19. Distance the thermal bar was offshore in southeastern Lake Michigan during April, 1974-1982. The resolving power of the cluster analysis to define water masses is estimated.

Year	Distance Offshore, km	Resolution Obtained from Cluster Analysis
1974	8	Good
1975	0	Good ¹
1976	8	Good
1977	6	Bad
1978	2	Good
1979	2	Bad
1980	2	Good
1981	6	Bad
1982	2	Bad

¹No thermal bar.

TABLE 20. Statistics for water masses present in nearshore southeastern Lake Michigan during April 1974. Areal distributions of water masses are found in Figure 3.

Variable	Water Mass I N = 6	Water Mass II N = 11	Water Mass III N = 1
Dissolved Silica (mg SiO_2/L)			
Minimum	1.0	0.0	0.0
Maximum	1.1	0.30	0.0
Mean	1.1	0.14	0.0
Standard Deviation	0.041	0.13	—
Orthophosphate ($\mu\text{g P/L}$)			
Minimum	0.10	0.0	18.
Maximum	1.60	1.1	18.
Mean	0.92	0.58	18.
Standard Deviation	0.58	0.42	—
Temperature ($^{\circ}\text{C}$)			
Minimum	3.0	4.8	6.8
Maximum	3.7	7.8	6.8
Mean	3.3	6.2	6.8
Standard Deviation	0.28	1.2	—

TABLE 21. Statistics for water masses present in nearshore southeastern Lake Michigan during April 1976. Areal distributions of water masses are found in Figure 4.

Variable	Water Mass I N = 11	Water Mass II N = 6
Dissolved Silica (mg SiO ₂ /L)		
Minimum	0.82	0.59
Maximum	1.5	0.95
Mean	1.2	0.78
Standard Deviation	0.19	0.13
Total Phosphorus (μg P/L)		
Minimum	3.5	7.9
Maximum	7.9 16.0	
Mean	6.6	14.0
Standard Deviation	1.2	1.8
Temperature (°C)		
Minimum	3.2	6.0
Maximum	6.5	7.6
Mean	4.5	7.0
Standard Deviation	1.2	0.56

WATER MASSES DERIVED FROM STREAMS AND RIVERS

April 1975 was a very unique sampling situation. It was the only year during which sampling preceded formation of the thermal bar. Even though the thermal bar was absent, two distinct water masses existed (Fig. 5). The variable which defined the two water masses was total phosphorus. It was higher in the inshore water mass (II) than in the offshore one (I) (Table 22). Depressed dissolved silica concentrations in the inshore water mass (II) had not yet developed due to lack of increased diatom productivity at the low water temperatures recorded for the inshore zone.

In July 1975, four water masses with their borders having a north-south orientation were found (Fig. 6). This occurrence is one of the best examples of the influence of the St. Joseph River on this nearshore region. The parameter responsible for defining the four water masses again was total phosphorus. It progressively decreased from its highest concentration in the water mass nearest the St. Joseph River (IV) to its lowest concentration in the one to the southwest (III) (Table 23). Inputs of total phosphorus-rich river water were transported by currents toward the southwest. Varying degrees of intermixing with nearshore lake water produced the observed water masses with their total phosphorus gradient.

WATER MASSES DERIVED FROM THE OCCURRENCE OF AN UPWELLING

For the years of appropriate sampling design (1974-1976), upwelling occurred only in 1976 during the time of sampling. Two water masses were identified (Fig. 7).

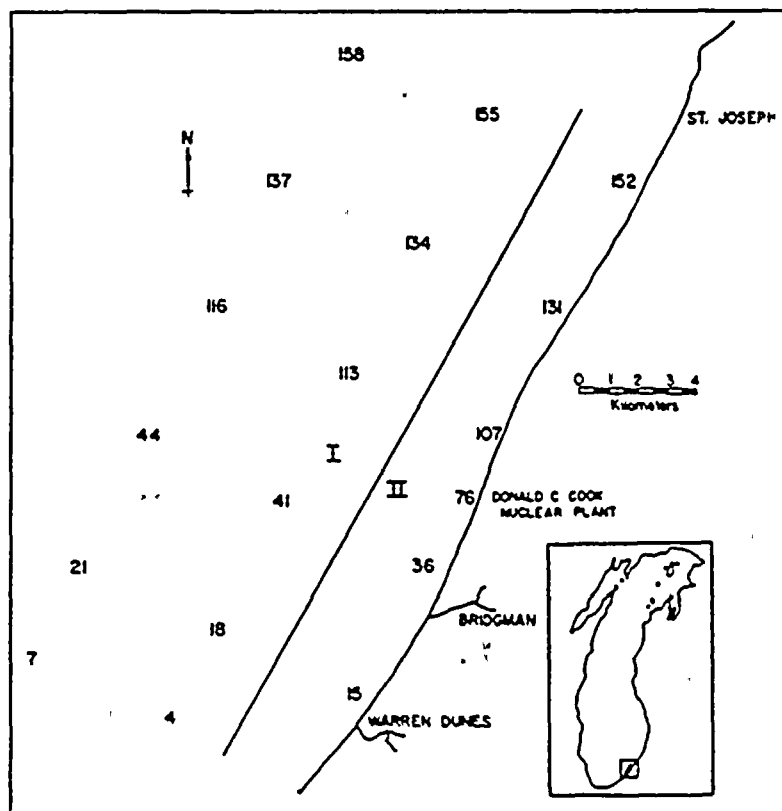


FIGURE 5. Water masses in southeastern Lake Michigan during April 1975. Masses were defined from cluster analysis of nutrient concentrations.

The offshore water mass (I) and the upwelled water mass (II) differed from one another in dissolved silica, total phosphorus, and nitrate concentrations. The slightly cooler upwelled mixture of hypolimnion and epilimnion waters (II) had higher total phosphorus and dissolved silica concentrations and a lower nitrate concentration (Table 24). The mean water temperatures were significantly different and differed by 2.0 C°. This upwelling transported waters enriched with total phosphorus and dissolved silica to the photic zone where they could be utilized by phytoplankton.

Thus distinct water masses occur in the nearshore region of southeastern Lake Michigan. They are derived from the presence of the thermal bar, upwelling events, and stream and river inputs to the region. Differences in the nutrient chemistry of the water masses are expected to create spatial differences in phytoplankton assemblages (Bowers et al. 1986).

TABLE 22. Statistics for water masses present in nearshore southeastern Lake Michigan during April 1975. Areal distributions of water masses are found in Figure 5.

Variable	Water Mass I N = 12	Water Mass II N = 6
Total Phosphorus ($\mu\text{g P/L}$)		
Minimum	8.7	21.
Maximum	16.0	24.0
Mean	13.0	22.0
Standard Deviation	2.2	1.3
Temperature ($^{\circ}\text{C}$)		
Minimum	1.9	3.0
Maximum	2.7	4.8
Mean	2.2	3.7
Standard Deviation	0.27	0.66

TABLE 23. Statistics for water masses present in nearshore southeastern Lake Michigan during July 1975. Areal distributions of water masses are found in Figure 6. Some stations did not cluster with any of the water masses (18, 21, 152).

Variable	Water Mass I N = 4	Water Mass II N = 5	Water Mass III N = 2	Water Mass IV N = 3
Dissolved Silica ($\text{mg SiO}_2/\text{L}$)				
Minimum	0.0	0.0	0.0	0.0
Maximum	0.17	0.29	0.040	0.10
Mean	0.034	0.12	0.020	0.067
Standard Deviation	0.076	0.12	0.028	0.058
Orthophosphate ($\mu\text{g P/L}$)				
Minimum	0.38	0.86	0.38	1.2
Maximum	1.5	1.8	1.2	2.0
Mean	1.0	1.2	0.76	1.4
Standard Deviation	0.43	0.37	0.54	0.50
Total Phosphorus ($\mu\text{g P/L}$)				
Minimum	2.1	4.1	0.87	0.0
Maximum	3.3	5.5	0.87	7.4
Mean	2.7	4.7	0.87	7.2
Standard Deviation	0.46	0.56	0.0	0.38
Temperature ($^{\circ}\text{C}$)				
Minimum	21.2	20.8	21.2	21.0
Maximum	23.0	23.0	21.3	22.5
Mean	22.0	21.7	21.2	21.8
Standard Deviation	0.82	1.1	0.071	0.75

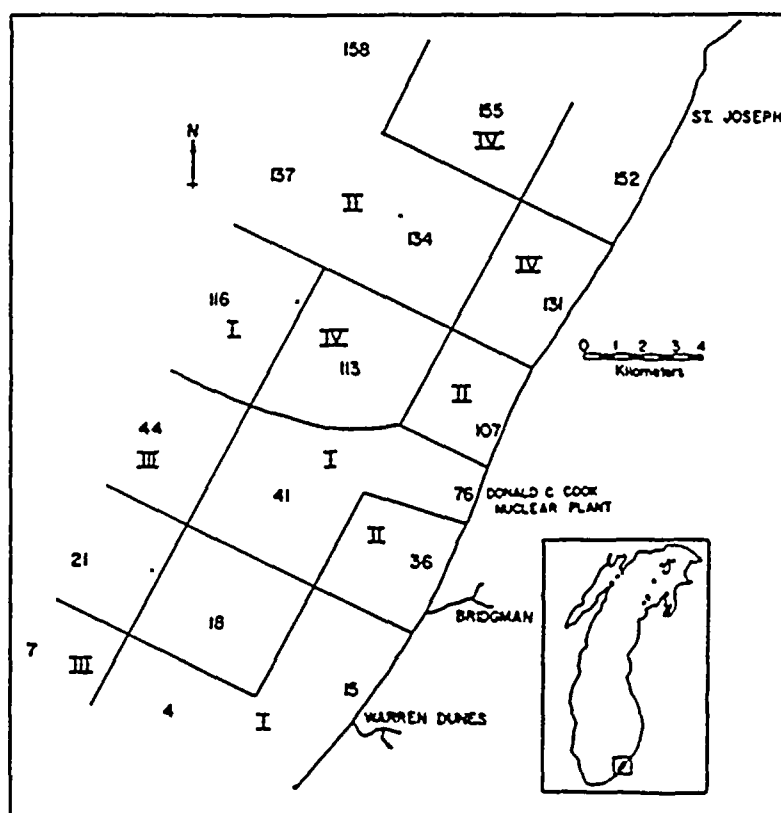


FIGURE 6. Water masses in southeastern Lake Michigan during July 1975. Masses were defined from cluster analysis of nutrient concentrations.

NUTRIENT AND ANION VARIATIONS BETWEEN 1974 and 1982

Just as water masses can create spatial differences in phytoplankton assemblages, month-to-month and year-to-year changes in nutrient chemistry can cause longer-term changes in the assemblages. To describe the changes in both nutrients and anions that have occurred in southeastern nearshore Lake Michigan between 1974 and 1982, regression and higher order polynomial fits were calculated.

GENERAL OBSERVATIONS

Nutrients

The data used represented 27 different periods of observation (Table 25) for nutrient and anion concentrations in southeastern Lake Michigan. Observations spanned the months April-October for the years 1974-1982. Median dissolved silica

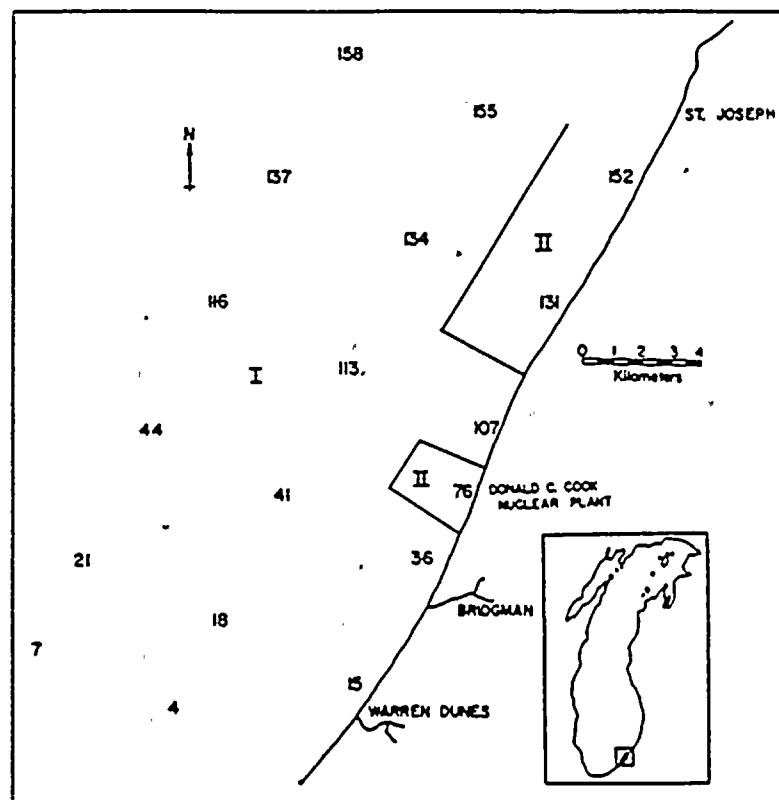


FIGURE 7. Water masses in southeastern Lake Michigan during July 1976. Masses were defined from cluster analysis of nutrient concentrations.

concentrations varied from undetectable in July 1975 to 1.6 mg SiO_2/L in April 1979. Based on the work of Parker et al. (1977) and Tilman et al. (1982), dissolved silica was limiting (<0.5 mg SiO_2/L) to some diatom species during April-September 1974, July and October 1975, July 1976, April-October 1977, July 1980, and July 1981. During July 1978, a very strong upwelling of dissolved silica-rich hypolimnetic water occurred. Surface water temperatures as low as 7.7°C were observed. The relatively high dissolved silica concentrations in July 1979 are unexplained.

Of the remaining nutrients, total phosphorus appears not to have been limiting to diatoms or green algae during the periods of observation. Median total phosphorus concentrations ranged from $2 \mu\text{g P/L}$ in July 1977 to $19 \mu\text{g P/L}$ in April 1979. Median nitrite-corrected nitrate concentrations varied from $160 \mu\text{g N/L}$ in April 1978 to $450 \mu\text{g N/L}$ in April 1980. Median orthophosphate concentrations ranged from $0.41 \mu\text{g P/L}$ in July 1981 to $2.8 \mu\text{g/L}$ in April 1978.

Anions

Anion concentrations varied considerably between 1974 and 1982. Analytical problems necessitated discarding the June 1974 and October 1977 chloride data and the April 1979 sulfate data (Table 26). The July and October 1978 sulfate data may be biased high. Median chloride concentrations ranged from 9.1 mg Cl/L in July 1975 to 17.0 mg Cl/L in April 1982. Median sulfate concentrations ranged from 14.0 mg SO₄/L in October 1977 to 30.0 mg SO₄/L in October 1978. In general, nutrients and anion concentrations were relatively high during April of each year. This is postulated to result from snowmelt and spring runoff.

TABLE 24. Statistics for water masses present in nearshore southeastern Lake Michigan during July 1976. Areal distributions of water masses are found in Figure 7.

Variable	Water Mass I N = 15	Water Mass II N = 3
Dissolved Silica (mg SiO ₂ /L)		
Minimum	0.16	0.61
Maximum	0.49	0.61
Mean	0.30	0.61
Standard Deviation	0.094	0.0
Orthophosphate (µg P/L)		
Minimum	1.3	1.8
Maximum	3.0	2.9
Mean	1.9	2.4
Standard Deviation	0.45	0.78
Total Phosphorus (µg P/L)		
Minimum	4.1	11.
Maximum	9.7	13.
Mean	7.1	12.
Standard Deviation	1.8	0.99
Temperature (°C)		
Minimum	18.5	16.8
Maximum	20.8	18.5
Mean	19.6	17.6
Standard Deviation	0.67	1.2
Nitrate (µg N/L)		
Minimum	200.	77.
Maximum	260.	77.
Mean	220.	77.
Standard Deviation	17.	4.2
Nitrite (µg N/L)		
Minimum	0.0	0.65
Maximum	6.8	6.5
Mean	0.97	3.6
Standard Deviation	2.4	4.2

TABLE 25. Nutrient concentrations in the nearshore (<38 m water depth) of southeastern Lake Michigan during selected months, 1974-1982. The number of observations varied between from 14 to 18 for 1974-1976 and from 25 to 30 for 1977-1982.

Date	Dissolved Silica (mg SiO ₂ /L.)			Nitrate (mg N/L.)			Orthophosphate (μg P/L.)			Total Phosphorus (μg P/L.)		
	Mean	Std. Dev.	Median	Mean	Std. Dev.	Median	Mean	Std. Dev.	Median	Mean	Std. Dev.	Median
4/74	0.47	0.47	0.30	—	—	—	1.7	4.2	0.85	—	—	—
5/74	0.26	0.22	0.15	—	—	—	0.73	1.0	0.42	22.	33.	7.5
6/74	0.29	0.16	0.29	—	—	—	1.0	1.8	0.50	4.1	2.2	5.0
7/74	0.11	0.11	0.11	—	—	—	1.3	2.5	0.45	10.	5.8	10.
8/74	0.28	0.21	0.30	—	—	—	0.65	0.50	0.45	7.5	10.	3.1
*9/74	0.34	0.12	0.34	—	—	—	1.7	0.86	1.4	3.6	2.7	3.8
4/75	0.64	0.17	0.62	—	—	—	1.1	1.0	0.71	16.	4.7	14.
7/75	0.056	0.085	0.0	—	—	—	1.3	0.60	1.1	4.6	2.6	4.2
10/75	0.43	0.21	0.46	—	—	—	0.68	0.29	0.67	7.6	4.3	6.6
4/76	1.0	0.26	1.0	—	—	—	1.8	1.2	1.5	9.8	4.5	7.4
7/76	0.35	0.14	0.32	0.21	0.040	0.22	2.0	0.48	1.9	7.9	2.9	7.3
4/77	0.44	0.19	0.41	0.33	0.067	0.30	1.2	0.60	0.94	9.3	3.6	9.7
7/77	0.19	0.11	0.21	0.22	0.18	0.17	1.4	0.26	1.3	3.0	3.2	2.0
10/77	0.47	0.14	0.49	0.31	0.11	0.31	1.7	0.82	1.5	5.4	2.1	5.3
4/78	1.1	0.26	1.1	0.14	0.072	0.16	3.0	0.89	2.8	11.	6.7	10.
*7/78	0.84	0.28	0.90	0.22	0.064	0.21	0.85	0.50	0.71	9.5	3.5	9.3
10/78	0.82	0.14	0.86	0.024	0.0079	0.25	1.4	0.56	1.6	10.	5.1	9.8
4/79	1.6	0.47	1.6	0.31	0.12	0.36	1.1	0.60	0.95	19.	6.8	19.
7/79	0.76	0.14	0.74	0.18	0.064	0.18	1.8	1.4	1.5	8.5	7.8	6.6
10/79	0.76	0.15	0.78	0.17	0.026	0.17	0.56	0.39	0.47	77.	30.	81.
4/80	1.2	0.14	1.2	0.45	0.072	0.45	2.2	2.8	1.3	11.	4.7	12.
7/80	0.24	0.096	0.19	0.24	0.047	0.25	0.84	0.76	0.65	27.	29.	16.
10/80	0.79	0.28	0.70	0.22	0.033	0.22	0.96	0.76	0.62	8.7	11.	6.2
4/81	1.1	0.15	1.0	0.34	0.062	0.35	1.0	0.52	0.96	8.5	3.2	8.3
7/81	0.38	0.10	0.34	0.23	0.029	0.23	0.47	0.40	0.41	8.4	4.4	7.6
10/81	1.2	0.26	1.2	0.20	0.028	0.19	0.52	0.28	0.47	5.6	3.2	5.4
4/82	0.56	0.25	0.52	0.32	0.077	0.32	0.98	0.46	0.95	13.	6.8	11.

*Upwelling event.

NUTRIENT AND ANION VARIATIONS WITH TIME

Polynomial Regression

Nutrients

For the determination of nutrient trends for 1974-1982, polynomial regression of nutrient concentration against time of observation was used. Any resulting regression lines or polynomial fits to the data having a level of significance less than 0.05 were accepted as being statistically significant. Significant first- (regression) and second-order fits to the data were found for total phosphorus (Fig. 8). Its concentration gently increased between 1974 and 1980, peaked in 1980-1981, and decreased from 1981 to 1982. Significant first- and fourth-order fits to the orthophosphate data were found (Fig. 9). Its concentration increased between 1974 and 1977, peaked in late 1977, decreased into 1981, and increased slightly in spring 1982. For dissolved silica, significant first- and third-order polynomial fits to the data were found (Fig.

TABLE 26. Anion concentrations in the nearshore waters of southeastern Lake Michigan during selected months, 1974-1982. The number of observations varied from 17 to 18 for 1974-1976 and from 29-30 for 1977-1982.

Date	Chloride (mg Cl/L)			Sulfate (mg SO ₄ /L)		
	Mean	Std. Dev.	Median	Mean	Std. Dev.	Median
4/74	11.	1.1	11.	22.	4.8	20.
5/74	12.	0.62	12.	19.	1.1	18.
6/74	—	—	—	18.	0.93	19.
7/74	12.	0.61	12.	18.	0.78	18.
8/74	14.	1.1	14.	18.	1.3	17.
9/74	12.	0.87	12.	17.	0.66	17.
4/75	11.	1.3	11.	20.	2.5	19.
7/75	9.1	0.73	9.1	21.	3.4	20.
10/75	11.	0.58	11.	18.	0.85	18.
4/76	12.	1.3	12.	20.	2.5	19.
7/76	12.	1.0	12.	18.	0.90	18.
4/77	13.	1.2	13.	19.	1.3	19.
7/77	13.	0.33	13.	19.	0.99	19.
10/77	—	—	—	14.	5.1	14.
4/78	12.	1.1	12.	21.	1.6	20.
7/78	9.3	0.24	9.3	25.	0.49	25.
10/78	9.3	0.32	9.2	30.	0.73	30.
4/79	12.	2.2	12.	—	—	—
7/79	11.	0.57	11.	21.	0.66	21.
10/79	9.8	0.79	9.7	20.	2.1	20.
4/80	13.	2.2	13.	22.	2.8	22.
7/80	12.	2.4	12.	19.	2.3	18.
10/80	11.	3.4	10.	19.	3.7	20.
4/81	11.	0.82	11.	22.	2.8	22.
7/81	15.	1.4	15.	19.	1.4	19.
10/81	14.	0.97	14.	19.	0.90	20.
4/82	17.	1.7	17.	22.	2.1	22.

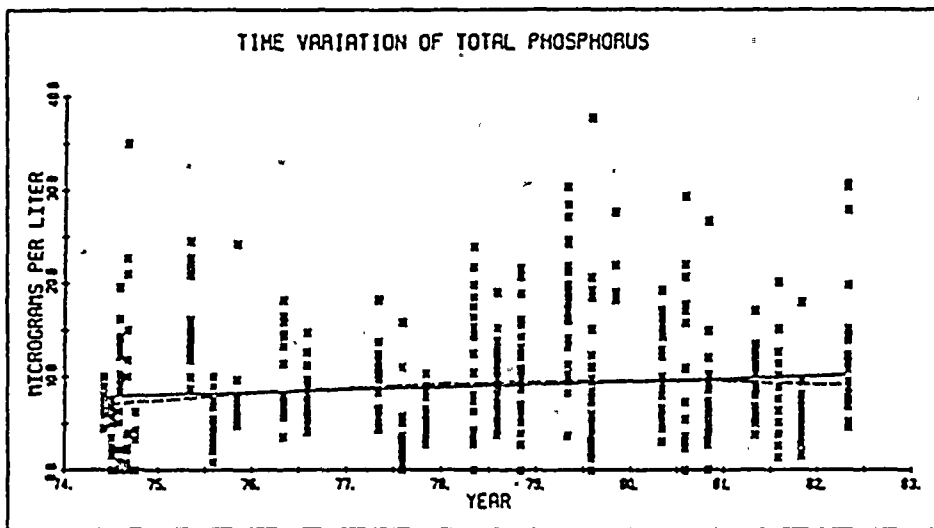


FIGURE 8. Time variation of total phosphorus (P) in southeastern Lake Michigan. The solid line is the regression fit to the data, and the dashed line is the second-order polynomial fit to the data. The symbols (*) are individual data points.

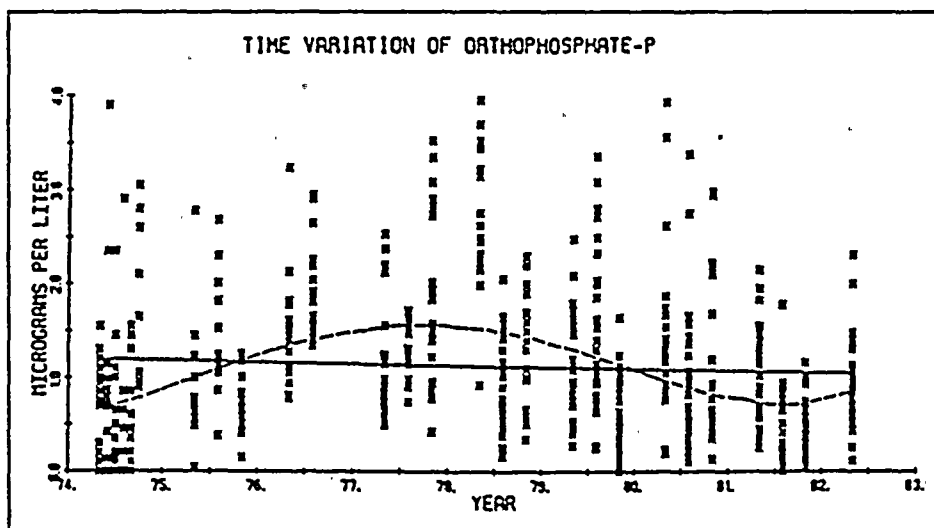


FIGURE 9. Time variation of orthophosphate (P) in southeastern lake Michigan. The solid line is the regression fit to the data, and the dashed line is the fourth-order polynomial fit to the data. The symbols (*) are individual data points.

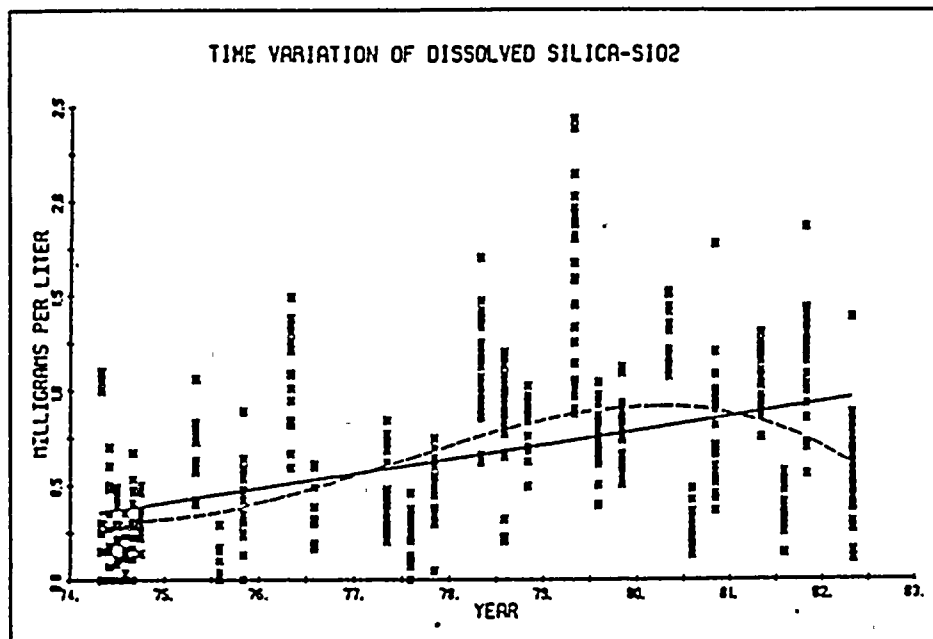


FIGURE 10. Time variation of dissolved silica in southeastern Lake Michigan. The solid line is the regression fit to the data points, and the dashed line is the third-order polynomial fit to the data. The symbols (*) are individual data points.

10). Dissolved silica concentrations increased between 1974 and 1980 and decreased slightly between 1980 and spring 1982. Significant first- and fourth-order polynomial fits to the nitrate data were found (Fig. 11). Its concentration decreased between 1976 and 1979, increased between 1979 and 1982, and decreased in spring 1982.

Anions

For sulfate, significant first- and fourth-order polynomial fits to the data were found (Fig. 12). Sulfate concentration decreased between 1974 and 1975, increased between 1976 and 1979, decreased between 1979 and 1982, and increased in spring 1982. Significant first- and fourth-order polynomial fits to the chloride data were found (Fig. 13). Chloride decreased slightly between 1974 and 1980 and increased dramatically between 1980 and spring 1982. For both nutrients and anions, higher-order polynomial fits to the data are representative of the year-to-year variation that can be expected in a dynamic nearshore zone.

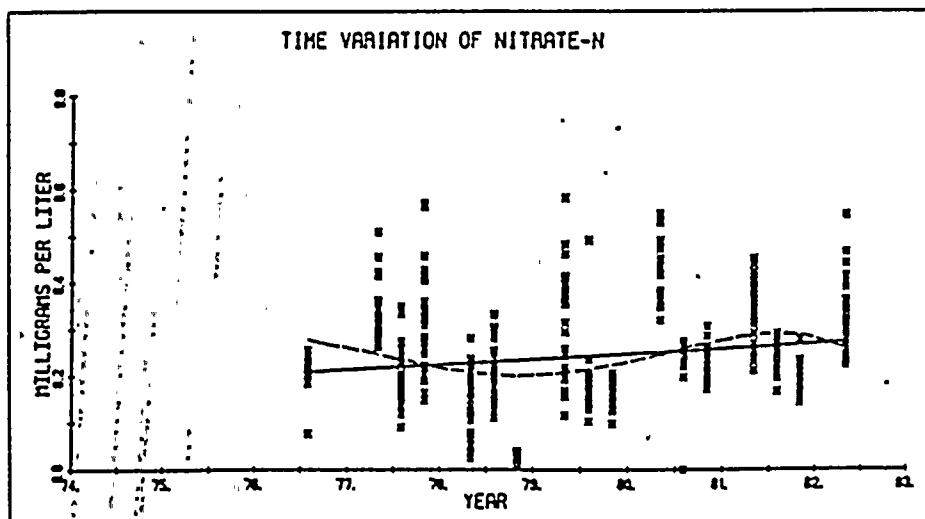


FIGURE 11. Time variation of nitrate (N) in southeastern Lake Michigan. The solid line is the regression fit to the data, and the dashed line is the fourth-order polynomial-fit to the data. The symbols (*) are individual data points.

Temporal Trends of Nutrients and Anions, 1974–Spring 1982

Nutrients

Because significant first-order polynomial fits were found for each of the parameters, the slope of the regression line represents the time-rate of change. For the period of observation (1974–1982), total phosphorus increased at a rate of 0.00029 mg P/L/yr (Table 27). Though total phosphorus concentration exhibited a significant linear trend, it appears that its peak concentration occurred in late 1979 and early 1980. For the same period, orthophosphate decreased at the rate of 0.000019 mg P/L/yr. At 0.00023 mg P/L/yr, the orthophosphate concentration rate of decrease between 1979 and 1981 was even more rapid. Orthophosphate has decreased since the State of Michigan ban on phosphorus in detergents was put into effect in October 1977. For the period of observation, dissolved silica concentrations increased at a rate of 0.074 mg SiO₂/L/yr. This is a reversal of the trend given by others for 1948–1962 (Table 27). Nitrate concentration increased at a rate of 0.012 mg N/L/yr. Powers and Ayers (1967) obtained mixed results for different sites on Lake Michigan. Their results ranged from a rate of increase of 0.0019 mg N/L/yr to a rate of decrease of 0.0083 mg N/L/yr.

Anions

From 1974 to 1982, sulfate concentrations increased at a rate of 0.31 mg So₄/L/yr (Table 27). This compares favorably with the rates presented by or derived from the data of Ackermann et al. (1970) for 1926 to 1969, Beeton (1965) for 1877 to 1961,

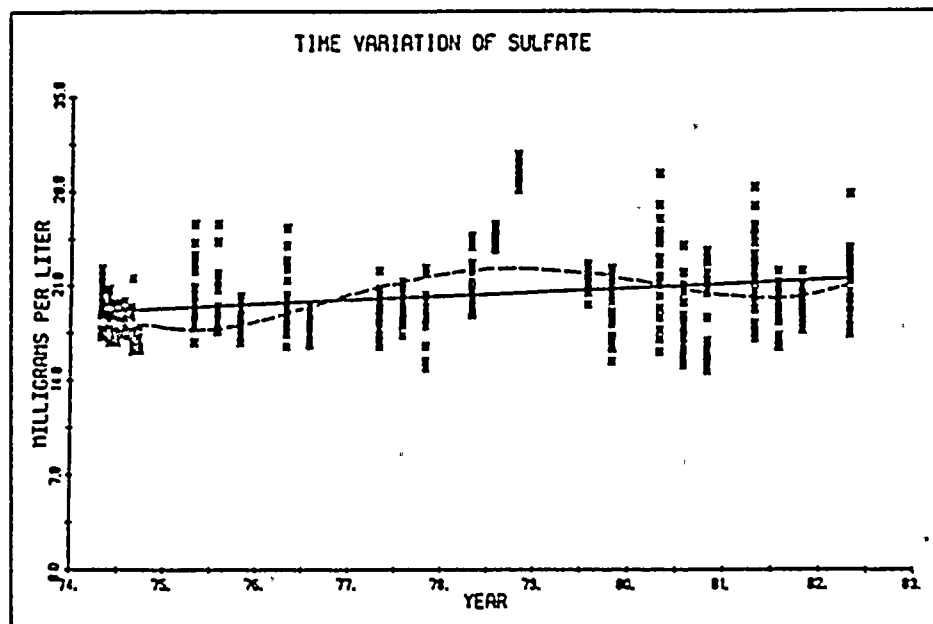


FIGURE 12. Time variation of sulfate in southeastern Lake Michigan. The solid line is the regression fit to the data, and the dashed line is the fourth-order polynomial fit to the data. The symbols (*) are individual data points.

and Powers and Ayers (1967) for 1948 to 1962. The rate of change appears to be increasing. Between 1974 and 1982, chloride concentration increased at the rate of 0.22 mg Cl/L/yr which is higher than rates given by or derived from Ackermann et al. (1970) for 1861 to 1969, Beeton (1965) for 1877 to 1961, and Powers and Ayers (1967) for 1948 to 1962. Powers and Ayers (1967) stated that the rate of increase of chloride concentration was increasing.

IMPLICATIONS OF CHANGES

Changes and fluctuation in nutrient chemistry may trigger changes in the phytoplankton community structure. For 1976-spring 1982, nitrate concentration increased. For 1974-spring 1982, orthophosphate concentrations decreased and chloride, sulfate, nitrate, dissolved silica, and total phosphorus concentrations increased. Increasing dissolved silica concentrations are attributed to decreased utilization by diatoms as a result of decreased orthophosphate concentrations. The apparent increase in nitrate concentrations will provide additional nutrients for green and blue-green algae growth if phosphorus concentrations increase concurrently.

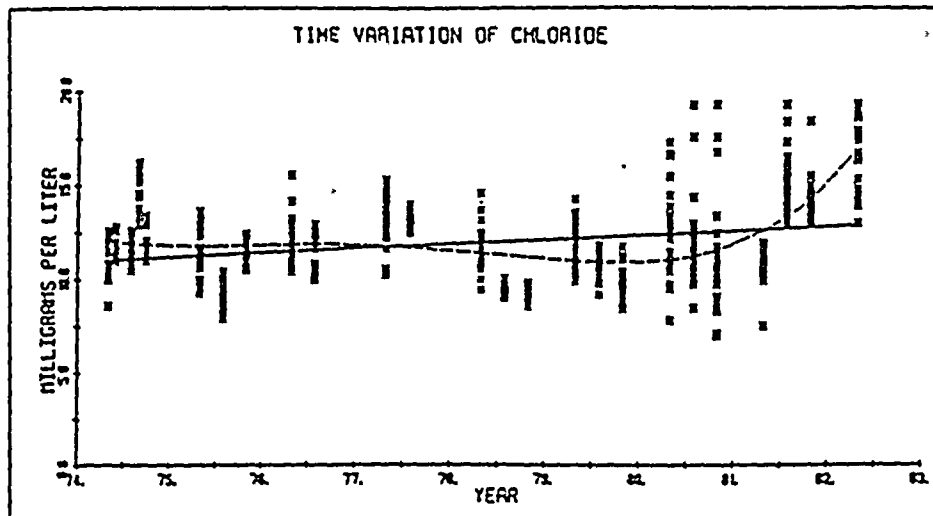


FIGURE 13. Time variation of chloride in southeastern Lake Michigan. The solid line is the regression fit to the data, and the dashed line is the fourth-order polynomial fit to the data. The symbols (*) are individual data points.

TABLE 27. Nutrient and anion trends in Lake Michigan waters.

Variable	Years Used For Calculation	Rate of Change (mg/L/yr)	Source of Information (see footnotes)
Chloride	1861-1969	+ .058	1
Chloride	1877-1961	+ .048	2
Chloride	1948-1962	+ .12 to + .16	3
Chloride	1974-1982	+ .22	4
Sulfate	1877-1961	+ .14	2
Sulfate	1926-1969	+ .26	1
Sulfate	1948-1962	+ .19 to + .74	3
Sulfate	1974-1982	+ .31	4
Nitrate-N	1926-1962	+ .0019	3
Nitrate-N	1939-1962	- .0083	3
Nitrate-N	1974-1982	+ .012	4
Dissolved SiO ₂	1948-1962	- .064 to - .34	3
Dissolved SiO ₂	1974-1982	+ .074	4
Total Phosphorus	1974-1982	+ .00029	4
Orthophosphate-P	1974-1982	- .000019	4
Orthophosphate-P	1979-1981	- .00023	4

¹Ackerman et al. (1970).

²Beeton (1965).

³Powers and Ayers (1967).

⁴This study.

EQUILIBRIUM CONSIDERATIONS

The availability of metals and anions to biota and for chemical reaction with one another is dependent upon each metal and anion concentration, complexation, and competition between metals for various anions. The availability of a metal or anion is dependent upon what fraction remains as a free ion and the kinetics of chemical equilibrium and complexation. Free-ion activities for nearshore southeastern Lake Michigan epilimnion waters were given by Rossmann (1980). Because of the unavailability of thermodynamic data for organic complexes, few data on organic ligand concentrations, and the lack of enthalpy data, only inorganic complexation was considered for July 1974 when the mean temperature was 25°C.

COMPLEXING

For southeastern Lake Michigan, Rossmann (1980) calculated the fraction of each metal complexed by carbonate, bicarbonate, sulfate, chloride, hydroxide, hydrogen phosphate, and dihydrogen phosphate. Metals for which complexation exceeded 5% of their free ion activities included divalent cobalt (39%), copper (68%), manganese (11%), nickel (30%), zinc (8%), and trivalent iron (53%). The major complexes of cobalt and nickel were with carbonate, those with copper were hydroxide, those with iron were sulfate, and those with manganese and zinc were bicarbonate.

POSTULATED OCCURRENCE OF AUTHIGENIC MINERALS AND COMPOUNDS

Rossmann (1980) calculated solubility products for various minerals and compounds found or postulated to be found in Great Lakes sediments. He found the waters of nearshore southeastern Lake Michigan to be supersaturated with respect to dolomite, malachite, hydroxylapatite, fluorapatite, hydroxoapatite, amorphous $\text{Fe}(\text{OH})_3$, and goethite. The removal of other divalent and trivalent metals to the sediments would have to occur by way of coprecipitation or adsorption onto the surfaces of the minerals and compounds found to be precipitating.

SUMMARY

The observed variation in chemistry of the nearshore waters of southeastern Lake Michigan is complex and derived from a variety of sources. Observed areal variations in chemistry are controlled by the way in which stream and river inputs are incorporated within the nearshore water mass. The mixing of these inputs with nearshore waters gives rise to water masses which trace their origin to stream inputs. Other water masses are derived from the occurrence of the thermal bar in spring and upwelling of hypolimnetic water during summer. Each water mass is chemically different from others, especially with respect to nutrients and temperature. These differences must be considered in any interpretation of nearshore phytoplankton distributions within any one sampling period or in month-to-month and year-to-year changes in phytoplankton assemblages.

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IMPACT OF THE DONALD C. COOK NUCLEAR PLANT ON PHYTOPLANKTON

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INTRODUCTION

PHYTOPLANKTON OF SOUTHEASTERN LAKE MICHIGAN

Historically, diatoms have dominated the Lake Michigan phytoplankton. Early studies (Ehrenberg 1854-1856, Briggs 1872, Thomas and Chase 1887, Ahlstrom 1936) reported that samples from the entire lake were dominated by diatoms at all seasons. Damann (1945) and Daily (1938) tabulated phytoplankton collections of the Chicago, Milwaukee, and Michigan City water works as well as the collection in Evanston, Illinois, which described the classic bimodal successional pattern found in temperate lakes.

Griffith (1955) also studied the nearshore area at Evanston, Illinois, and discovered higher total cells/mL than did either Daily or Damann. Until then, diatoms were dominant for all seasons, but Griffith (1955) found large numbers of *Anacystis*, indicating that blue-green algae were dominant. Diatoms were the next highest in number. Twenty-five diatom genera were identified throughout the year, with a slight decrease in number during summer months.

Several studies have been published on southern Lake Michigan since 1960 (see Tarapchak and Stoermer 1976). In 1963, at stations extending from Chicago to near Milwaukee, at all stations and all months diatoms constituted nearly one-half of the phytoplankton population that was dominant (Stoermer and Kopczynska 1967). *Stephanodiscus hantzschii* was dominant at inshore stations. Stoermer (1974) collected samples in southern Lake Michigan on two east-west transects and one north-south transect, sampling in both inshore and offshore water. Diatoms were dominant in the spring and early summer, but were replaced by the green and blue-green algae *Anacystis* spp. and *Anabaena flos-aquae* as silica concentrations reached limiting levels in late summer. Some diatom growth occurred in the fall, but blue-green remained dominant, indicative of the cultural eutrophication of the offshore water in the southern basin.

Limnological studies by Ayers and Feldt (1983) at the Donald C. Cook Nuclear Plant near Benton Harbor, Michigan, resulted in several publications on the phytoplankton in that vicinity. Diatoms represented about one half of the flora, with greens, blue-greens, chrysophycean flagellates, cryptomonads, and desmids making up the remainder. The diatoms have declined over the last 12 years. Blue-green algae increased from spring to fall; green algae increased in late summer; and flagellates were present for all seasons but have been increasing in numbers for the past several years (Ayers and Feldt 1983).

Stoermer and Yang (1970) categorized the trophic status of Lake Michigan using

diatom species. The entire lake is oligotrophic, with nearshore areas being mesotrophic to eutrophic. Southeastern Lake Michigan is oligotrophic offshore with a mesotrophic nearshore area which becomes eutrophic near Gary, Indiana. The various trophic levels and phytoplankton species associated with each are discussed in Holland (1968, 1969), Stoermer and Yang (1969, 1970), Holland and Beeton (1972), Tarapchak and Stoermer (1976), and Rockwell et al. (1980).

The Donald C. Cook Nuclear Plant is located near the St. Joseph River. Data show that the river adds large quantities of nutrients to the lake near the power plant (Van Landingham 1976, Schelske et al. 1980). Riverine phytoplankton species for this area are *Amphora ovalis*, *Amphora* sp., *Cyclotella meneghiniana*, *Melosira granulata*, *M. granulata* var. *angustissima*, *Navicula capitata*, *N. costulata*, *N. decussis*, *N. gastrum*, *Navicula* spp., *Nitzschia acicularis*, *Nitzschia* sp., and *Synedra ulna* (Ayers et al. 1972, Schelske et al. 1980). Other riverine species found that can survive in this nearshore region are *Cyclotella atomus*, *C. cryptica*, *C. meneghiniana* var. *plana*, *C. pseudostelligera*, *Fragilaria capucina*, *Stephanodiscus subtilis*, *S. tenuis*, and *Skeletonema potamus*. These species are able to survive nearshore in nutrient-rich water where the temperature is 6 to 10C° warmer than offshore water.

Stoermer and Ladewski (1976) have reported the optimal temperatures for some of the common phytoplankton in southern Lake Michigan. They conclude that this region of the lake is in a transition state, going from an oligotrophic to a disturbed, mesotrophic habitat.

Stoermer and Tuchman (1979) reported on phytoplankton along the southern shoreline of Lake Michigan in 1977. The species found are indicative of eutrophic and moderately disturbed areas (Stoermer and Yang 1969). Large abundances of *Nitzschia* spp. were found, an indication of poor water quality conditions.

Rockwell et al. (1980) reported in their survey of 1976-1977 that the phytoflagellates were the most abundant component of the phytoplankton. Phytoplankton populations were highest nearshore and decreased toward the open lake. Diatoms were abundant through June and decreased with stratification, while blue-greens increased during mid-summer.

The Great Lakes have undergone long-term changes as a result of a cultural eutrophication, including more industry, agricultural run-off, sewage effluent, and demand for fish. Phosphorus is the primary limiting nutrient. An increase in phosphorus loading has caused an increase in phytoplankton production, followed by reduced diatom production caused by silica limitation (Schelske and Stoermer 1971, 1972; Schelske et al. 1983; Stoermer 1978). Schelske et al. (1983) have shown that biogenic silica storage increased tenfold from 1940 to 1970 when phosphorus loading from detergents was increasing. After 1970, lack of silica limited diatom production.

Phytoplankton populations exhibit a temporal seasonal succession which can vary with nutrient and thermal conditions. The basic successional pattern is a low abundance in winter followed by a spring diatom bloom; non-diatom species, flagellates, and green and blue-green algae dominate in the summer, and a small diatom bloom follows in the fall (Moll and Stoermer 1982). Thermal bars develop in the spring and impact phytoplankton succession for the remainder of the year (Mortimer 1974; Rodgers 1965; Holland 1968, 1969). Thermal bars differentiate the warmer nearshore waters from colder offshore waters (Stoermer et al. 1971).

As stated earlier, *Cyclotella* spp. are dominant in oligotrophic communities. Many of these species are now less abundant or rare in the phytoplankton of southern Lake Michigan (Stoermer 1978). *Cyclotella michiganiana*, once an abun-

dant species, has been replaced with *Cyclotella comensis*. Ayers and Wiley (1979) reported that the first occurrence of *C. comensis* at the Cook plant was in October 1975; it has since become a dominant in the fall months. Blue-greens and greens have always been present in the lake but in minimal numbers. Now, they frequently become abundant in the summer months as the water temperature increases. The microflagellates were reported (Ayers and Wiley 1979) to be more abundant in southern Lake Michigan than they were a few years ago. They are poorly preserved in samples and also have received poor taxonomic treatment, so it is possible they were previously abundant.

In summary, southern Lake Michigan is in a transition stage between pronounced seasonal successions with low assemblage densities in the summer, and pronounced seasonal successions with a transient population maxima during the summer (Stoermer 1978).

IMPACT OF POWER PLANTS ON PHYTOPLANKTON

Electric power plants using once-through cooling for their condensers require large quantities of water to dissipate large amounts of heat. In some instances, the water utilized is chlorinated to prevent the growth of organisms on condenser walls which would decrease the cooling efficiency of the condensers. Passage of water from an ocean, lake, or stream through the condenser system of a plant has the potential to destroy or shock the entrained biota. Within the body of water receiving discharges from the plant, biota can be shocked or destroyed or their community structure can be altered. The impact on phytoplankton productivity ranges from stimulation to cessation. The impact is produced by mechanical pumping, heating, and chlorination of the circulating water.

Studies of the impact of power plants on phytoplankton have utilized a number of methods for quantifying the impact. The method selected is critical to the ability to detect an impact. Methods utilized have included measurement of chlorophylls, assessment of phytoplankton assemblage structure, and measurement of primary productivity.

It is well established that temperature plays an important role in determining phytoplankton species diversity and abundance (Patrick 1969). A healthy stream containing a mixed algal population was sampled by Cairns (1956); the phytoplankton culture obtained was subjected to gradually increased and then gradually decreased temperature. At 20°C, diatoms were predominant. As the temperature increased to 30–35°C green algae became dominant, and blue-green algae became dominant at 35–40°C. Diatom species once again were dominant 3 weeks after the temperature was reduced to 22°C, indicating that not all individuals of a species were killed at unfavorable temperatures, and that they could not successfully compete with better adapted species at the higher temperatures.

Patrick's (1969) review of temperature effects on freshwater algae concluded that a species will tend to experience increased growth and photosynthesis if it is provided with sufficient light and an optimal temperature range. As the temperature is artificially increased to the tolerance limits for a species, cell division, photosynthesis, and formation of reproductive cells may be repressed. Diatoms have relatively low temperature tolerances (30°C or less). Green algae are tolerant of higher temperatures, and blue-green algae are tolerant of even higher temperatures. Patrick (1971)

suggested that species go into resting phases under unfavorable temperature conditions.

Estimation of phytoplankton population changes that result from temperature increases due to thermal discharges or condenser passage are complicated by many factors. In the natural realm, phytoplankton undergo seasonal species succession, diurnal fluctuation of photosynthesis, local nutrient enrichment or limitation, and patchiness of population distribution in the water mass.

Gurtz and Weiss (1972), in their studies at the Allen steam-electric generating plant, Lake Wylie, North Carolina, sought to learn what effect short-term thermal stress had on a discrete phytoplankton population. Three condensers were regulated so that the water underwent 5.6, 11.1, and 16.7°C rises in temperature during condenser passage. Samples were simultaneously collected prior to and just after passing through each condenser and allowed to cool at a controlled rate for up to 26 hours over which time aliquots were removed for primary productivity measurements. This experiment was carried out on six different dates between July 1971 and June 1972 to include seasonal effects. These same water samples were spiked with nutrients following the controlled cooling period and were assayed for chlorophyll *a* and total carbon as indicators of algal recovery and growth following thermal shock. It was found that phytoplankton primary productivity in condenser cooling water decreased, with this depression related to initial temperature and the degree of temperature rise. Temperature increases of 5.6 and 11.1°C resulted in relatively constant inhibitions except above a 28.3°C intake temperature where greater inhibitions occurred. Inhibitions for the 11.1°C rise, however, showed seasonal effects with greater inhibitions of productivity being recorded for increased intake water temperature. Phytoplankton growth studies revealed that the samples which had received the greatest thermal stresses produced the largest final yields. This may have been due to altered species composition in the waters following condenser passage. The authors concluded that summertime temperature rise should be limited to 11.1°C for this plant, but greater increases could probably be tolerated in the winter.

Species composition changes in attached algae have been shown to occur due to thermal effluent from Sundance Power Station, Lake Wabamun, Alberta, Canada (Hickman and Klarer 1975). In studying the epiphyton on *Scirpus validus* between May and October, 1972, it was discovered that the algae went from a diatom-dominated community in unheated waters near the intake to one in which members of Chlorophyta (green algae) dominated at the heated site. Segments of *Scirpus validus* were collected at the heated and non-heated water sites and the attached algae were removed from the stems. Samples for primary productivity measurements were incubated 3 hours at the heated and non-heated sites with a ¹⁴C source. Mean primary productivities of both groups of samples increased when incubated at the heated site. Likewise, the mean standing crop as estimated by chlorophyll *a* concentration was larger in samples collected at the heated site than at the non-heated site. This was primarily due to the large spring and summer maxima of two green algae in the heated waters. The mean photosynthetic index, mg C/h/mg chlorophyll *a*, was also calculated for the same four cases. The index for non-heated samples incubated at the non-heated site was somewhat greater than that of heated samples incubated at the heated site. The index for non-heated samples incubated at the heated site was approximately twice that of non-heated samples at the non-heated site. The index of heated samples incubated at the non-heated site was half that of heated samples incubated at the heated site. The maximum heated water

temperature was 28°C. The authors concluded that water temperature in non-heated areas was probably not optimal for photosynthesis by the algal population and that the photosynthetic efficiency of the algae in the heated area was decreased by the continuous flow of heated water.

Studies in 1971 and 1972 at Wabamun Power Station, a plant on Lake Wabamun, revealed increased standing crops of epipelon (algae free-living on sediments) due to the thermal effluent (Hickman 1974). This was especially true in the discharge canal itself. A decrease in the number of diatom species was also discovered in the discharge canal. The thermal effluent had no effect on epipsammon (algae living among or attached to sand grains). The maximum temperature recorded in the discharge canal during this study was 31°C, 7°C° greater than in the unaffected portion of the lake (Gallup and Hickman 1975).

From May 1965 to April 1966, Poltoracka (1968) studied the species composition of net phytoplankton in three interconnected lakes near Konin, Poland. A thermal power plant drew its cooling water from Lake Patnow and discharged it into Lake Lichen whose annual temperature ranged from 7.4 to 27.5°C. From Lake Lichen the water passed into Lake Mikorzyn which exhibited slightly elevated temperatures and then into Lake Slesin whose yearly temperature, ranging from 0.8 to 20.7°C, did not reflect an increase due to the discharged heat. Lake Lichen differed from the other two lakes in that it contained a markedly higher number of species, especially from the class Chlorophyceae (green algae). It did not exhibit pronounced seasonal fluctuations in total number of algal species as did Lakes Mikorzyn and Slesin. Members of Chlorophyceae increased in number in the three lakes with respect to increased temperature while numbers of diatoms decreased.

Productivity studies were undertaken at an electric power generating station on the Patuxent River Estuary, Chalk Point, Maryland, by Morgan and Stross (1969) in August 1966 and continued through August 1967. The intake canal is located in a small bay at Chalk Point; the intake samples were collected from the mouth of the canal. The discharge canal joins the river approximately 3.2 km upstream; the discharge samples were collected 0.4 km from the mouth of the discharge canal. The time required for the water from the intake sampling point to reach the discharge sampling point was approximately 3 hours, dependent, of course, on the intake rate. Three single productivity measurements were made of intake and discharge samples in August and September 1966. For an approximately 8°C° rise in temperature, photosynthesis was stimulated when intake temperature was 16°C or cooler. When intake temperature was 23°C or warmer, the photosynthesis rate in the discharge was lowered to 0.06-0.31 of the intake rate. Further experiments were carried out in October 1966 and March and August 1967 which involved incubating repetitive intake samples taken at intervals of a few hours at the discharge temperature. At temperature differentials of 16°C° to 24°C° in October and 7.6°C° to 11.8°C° in March, the mean rates of carbon uptake increased from 16.0 to 41.6 mg C/m³/hr. Non-averaged repetitive samples showed considerable variation in March and October. Productivities of intake and effluent samples incubated at the effluent temperature also were compared for these months. A significantly lower rate of photosynthesis occurred for the effluent sample of the March experiment. In August, both the intake and effluent samples showed large decreases in productivities as compared with intake samples incubated at the intake temperature. The October experiment clearly showed the effects of chlorination when productivities were reduced to nearly zero at the times of chlorine addition. Chlorophyll *a* concentrations were greatly reduced, suggesting cell destruction. Recovery of photosynthetic rate did not

occur when effluent samples were returned to intake temperatures. The authors concluded that carbon uptake rates were inhibited at temperatures of 23°C or greater and an 8°C rise in temperature. At 16°C or lower, an 8°C rise in temperature stimulated carbon intake. When heat inhibited carbon uptake, condenser passage increased this inhibition. When heat stimulated the carbon uptake, chlorination and condenser passage may have negated this stimulation.

Fox and Moyer (1973) examined both the effects of thermal shock resulting from condenser passage on a phytoplankton population and changes experienced by the population as it gradually cooled in the discharge canal at the Crystal River plant-site, Florida. Two canals were dug for the cooling system at the plant, located on the Gulf of Mexico. The south canal serves as the intake and the north canal as the discharge. Of the sampling points chosen in the study, station 1 was located at the center of the intake canal. Station 2 was located in the center of the discharge canal at a point thought representative of thoroughly-mixed water coming from the two fossil-fueled units. Stations 3, 4, and 5 were located at 0.8-km intervals farther down the discharge canal, and station 6 was located in the Gulf, 0.8 km northwest of the discharge canal. Station 6 represented shallow, estuarine water that received water from the discharge canal only during ebb tide. Experiments were carried out on 28 April and 4 June 1971. Chlorination did not occur at the plant during the period of this study. Because Fox and Moyer desired to follow the changes occurring in the particular water mass sampled at the intake, they added uranine dye at station 1 and measured the time required for it to reach station 2. It took 8 minutes. Two drogues were placed in the water as the dye reached station 2 and samples were collected at stations 3, 4, and 5 as the drogues passed them. Water flow rates down the canal varied with the tides. Water from station 5 did not necessarily pass station 6. Parameters examined in this study were temperature, dissolved oxygen, total bacterial population, chlorophyll *a*, primary productivity, total and suspended solids, and adenosine triphosphate (ATP). Both power generating units were designed to have a maximum temperature rise across the condensers of 6.1°C. The temperature differentials recorded in three experiments on 28 April between stations 1 and 2 were 6.7, 5.0, and 6.0°C, respectively. Dissolved oxygen levels were inversely proportional to temperature though levels were never severely depressed between any of the stations. Weight of total solids did not correlate with any parameters. Samples for primary productivity measurements were incubated at the stations where they were collected. Primary productivities varied with respect to intake water temperature. Productivities decreased when intake temperature was 27°C or greater and the temperature differential between stations 1 and 2 was 5°C. Productivities continued to decrease downstream in the canal until the water temperature was lowered to 32°C or less. Chlorophyll *a* results seemed to indicate that the amount present was dependent on the time of day. Values decreased from stations 1 and 2 in the morning experiments and increased in the afternoon. Bacterial populations increased 45.5 to 550 percent between stations 1 and 2 following 48-hour incubation or with a greater increase when the temperature change was lowest, 5 to 5.5°C. Adenosine triphosphate (ATP) measurements were included as an indicator of viability and biomass of the phytoplankton population because ATP degrades rapidly following the death of an organism. The values increased from station 1 to station 2 in all experiments except one where a slight drop was encountered. In this case, the ATP level continued to drop for all stations down the canal. The highest temperature recorded in the experiments, 34.5°C, occurred at station 5 during that run. This study provided evidence

that phytoplankton were hindered in their ability to photosynthesize but that they were not necessarily being killed by the plant. Generally the organisms made some recovery while traversing the discharge canal as parameters measured at the end of the canal were not significantly different from values obtained at the intake.

Briand (1975) studied the effects of condenser passage at the Alamitos and Haynes generating stations on the San Gabriel River near Long Beach, California, which generated 3,575 MW, the most powerful generating complex in America at the time of the study. One sample was collected near the intake pipe at each plant and one in the middle of the river 300 m downstream from each discharge pipe, a place considered representative of thoroughly-mixed discharge waters. Water was sampled when the tide was receding so that immediate effects of condenser passage could be examined. It was calculated that 4 to 9 seconds were required for the water to traverse the condensers and that the water spent approximately 10 minutes going from either intake sampling site through the plant to the discharge station. Chlorine was added sequentially for all 21 units so that its concentration in the cooling water ranged from 0.2 to 1.0 mg/L and was considered a constant factor. Water temperatures ranged from 14 to 23°C at the intake stations (less than 0.2°C variation between the two intakes at any time). The usual discharge temperature was between 24 and 26°C, but it reached 31°C in August. The average temperature increase across the condensers was 9.3°C over the year, with the minimum rise of 6°C in June and the maximum rise of 11°C in December and January. Results of phytoplankton counts revealed a reduced species diversity following condenser passage, with diatoms being reduced in greater proportions than dinoflagellates. Primary productivity following condenser passage in February and June increased by 230% while it was reduced by 40% in September. In February and June, growth rates of survivors in the discharge, as estimated by the production: biomass ratio, increased three-fold, but it remained the same as intake populations in September. Thus it appears that the surviving stock would be capable of fast recovery during some times of the year. It was suggested that the reason for a poor growth rate in September could be due to an overly elevated discharge temperature, 31°C. The author reasoned that despite the fact that phytoplankton seemed to make a rapid recovery in replacement of the standing stock, the power plant could be looked upon as a serious disturbance to the ecosystem because of the selective reduction in numbers of certain species, while the relative abundance of the others, especially *Asterionella japonica* and *Gonyaulax polyedra*, was enhanced. Briand (1975) found a relationship between intake temperature, amount of heating, and phytoplankton mortality. When intake water was cooler than 15°C, the phytoplankton stocks seemed unaffected by temperature increases up to 11°C; however, when intake water was 16°C, an effect was seen when temperatures rose 8 to 9°C. On this basis, Briand (1975) advocated the use of cold deep-sea water for cooling coastal power stations.

Brook and Baker (1972), while attempting to study productivity in the vicinity of the King Plant, St. Croix River, Minnesota, could find no correlation between photosynthesis and respiration rates of phytoplankton and temperatures of the river, condenser cooling water, or discharge canal. When it was discovered that chlorine was added in 1-hour-long, four-times-daily doses to a subsidiary cooling system, they then were able to correlate the severe depression of photosynthesis and respiration with the chlorine additions. Their incubation studies revealed a 5 to 10% depression of photosynthesis along with up to a 50% stimulation of respiration when discharge samples and the control sample taken from a site upstream of the

plant were collected during a non-chlorinating period and compared. Photosynthesis was depressed 50 to 90% and respiration was often reduced to an unmeasurable rate when the same experiment was carried out during a period of chlorination. A water sample collected within the plant during chlorination contained $2,700 \pm 100$ $\mu\text{g/L}$ chlorine. This sample was serially diluted. A plot of concentration versus percent depression with respect to the control indicated that both respiration and photosynthesis were reduced to 50% of the control at a concentration of 320 $\mu\text{g/L}$ chlorine, photosynthesis was reduced to zero at 1,600 $\mu\text{g/L}$, and respiration was reduced to zero at 2,700 $\mu\text{g/L}$, and respiration was reduced to zero at 2,700 $\mu\text{g/L}$.

Brooks (1976) obtained Lake Michigan water from a Milwaukee filtration plant prior to any treatment and, following 30 minutes exposure to chlorine concentrations ranging from 0 to 1,400 $\mu\text{g/L}$ as total residual chlorine, incubated the sample for 24 hours at a constant temperature approximating that of the lake. The phaeophytin *a*/chlorophyll *a* ratio, as obtained by fluorescence, and net primary productivity were measured. Initial reduction in photosynthesis of 5 to 18% occurred at 3 $\mu\text{g/L}$ chlorine. The effect lasted 6 hours with nearly full recovery after 12 to 24 hours. At 616 $\mu\text{g/L}$, net productivity returned to only 52% of the control value after 24 hours. At 1,218 $\mu\text{g/L}$, there was no recovery of production after 24 hours. Results of pigment analyses showed similar trends and also indicated that no significant recovery occurred after 24 hours at any concentration where chlorophyll was destroyed through chlorine addition. It was noted that the concentrations of chlorine needed for 50% inhibition of primary productivity varied with season, the amount being two to three times lower in summer months than in the fall.

Mechanical effects due to turbulence and pumping were studied by Gurtz and Weiss (1972) who pumped water through a cooled, unused condenser at the Allen Steam Plant in June 1972. The water was pumped at three different rates. They collected samples prior to and after condenser passage and measured productivities after a 3-hour incubation at the intake temperature of 24.7°C. Results indicated a mild stimulation of primary productivity due to condenser passage. Effects encountered prior to passage due to pumping were also considered. It appeared that stimulation occurred but they did not place much confidence in their sampling system.

Thus studies have shown that phytoplankton may become inhibited or die due to entrainment and condenser passage. In addition, changes in community structure have been noted. Various authors have concluded that temperature rises which can be tolerated range from 8°C to 11°C. The actual delta-T permissible is related to the intake water temperature. The lower the intake water temperature the greater the tolerable temperature rise. If chlorination is also taking place, the phytoplankton may be killed outright or undergo varying degrees of inhibition. Communities have been observed to exhibit a decreased diversity promoted by a shift from a diatom-dominated community to one dominated by either green algae or blue-green algae in heated waters. Finally, some evidence exists which suggests that the phytoplankton may be mildly stimulated by mechanical pumping (Gurtz and Weiss 1972).

At the time of its construction, the Donald C. Cook Nuclear Plant was one of the largest nuclear power plants in the country (2,200 MW). Located in southwestern Michigan, it utilizes water from nearshore, southeastern Lake Michigan. Because of the plant's size and use of large quantities of lake water for once-through condenser cooling, concern was expressed that thermal discharges from the plant would alter the phytoplankton assemblage structure and promote large blooms of nuisance algae.

STUDY DESIGN

ENTRAINMENT SAMPLING

To assess the impact of the plant on entrained phytoplankton, sites for sample collection were selected in the plant's intake and discharge forebays. Because of the plant's design, the discharge forebays were always turbulently mixed, and the collection of representative samples was assured. This was not the case for the intake forebay. Water flow was uneven and dependent upon the number of circulating water pumps in operation.

Selection of representative sampling point in the intake forebay was based upon results of a heterogeneity study conducted during operation of all seven of the plant's circulating water pumps. In 1975, samples were collected from intake sampling locations 1, 3, and 5 in front of traveling screens 1, 3, and 5 (Fig. 1) (Rossmann et al. 1977). At each location, samples were collected in triplicate from depths of 0.5, 5.5, and 8.5 m. Phytoplankton were enumerated to the species level, and chlorophylls and phaeophytin *a* was measured for each sample.

The phytoplankton were subdivided into the nine major groups (coccoid blue-green, filamentous blue-green, coccoid green, filamentous green, flagellates, pennate diatoms, centric diatoms, desmids, other algae) plus total cells for determining the location of a representative sampling location. Other algae were primarily coccoid cells that could not be identified. At the 0.05 level of significance, there were no significant differences among the three sampling locations for the major groupings of phytoplankton (Table 1). For the major grouping, there were no significant differences among the three sampling depths (Table 2). Groups that were close to having significantly different concentrations among the three depths included filamentous green algae, centric diatoms, and total algae. The factor creating this situation was an unusually high abundance at the 8.5-m sampling depth. Based upon the May 1975 study, sampling location 5 with a sampling depth of 5.5 m was selected as the representative sampling point (Rossmann et al. 1977).

For the chlorophylls and phaeophytin *a*, there were no significant differences ($\alpha = 0.05$) among the three locations and among the three depths (Tables 3-6). The low concentrations of each contributed to the lack of significant differences among the locations and depths.

A second heterogeneity study was conducted in September 1978 using only chlorophylls and phaeophytin *a* (Change et al. 1981). By this time, both units of the plant were operational. Sampling locations compared were 2, 5, 10, and 14 (Fig. 1). Significant differences among sampling locations at 5.5-m depth were found (Tables 7-10). Chlorophyll *a* was significantly lower at locations 10 and 14 relative to locations 2 and 5. Phaeophytin *a* was significantly higher at location 14 than at the other locations. There were significant differences among the three sampling depths at locations 2, 5, and 10 (Table 11). At locations 2 and 5, chlorophyll *a* concentrations were significantly lower at 0.6 m relative to 5.5 and 8.5 m. At location 10, chlorophyll *a* was lowest at 5.5 m and highest at 8.5 m. Chlorophyll *b* was relatively low at 0.6 m at location 14 and was relatively high at 8.5 m at location 10 (Table 12). Chlorophyll *c* at location 10 had no distinct pattern (Table 13). Each depth had a concentration significantly different from the other two depths. There were no significant differences among the three depths at each location for phaeophytin *a* (Table 14). Because these results did not negate our previous choice of location 5 at a

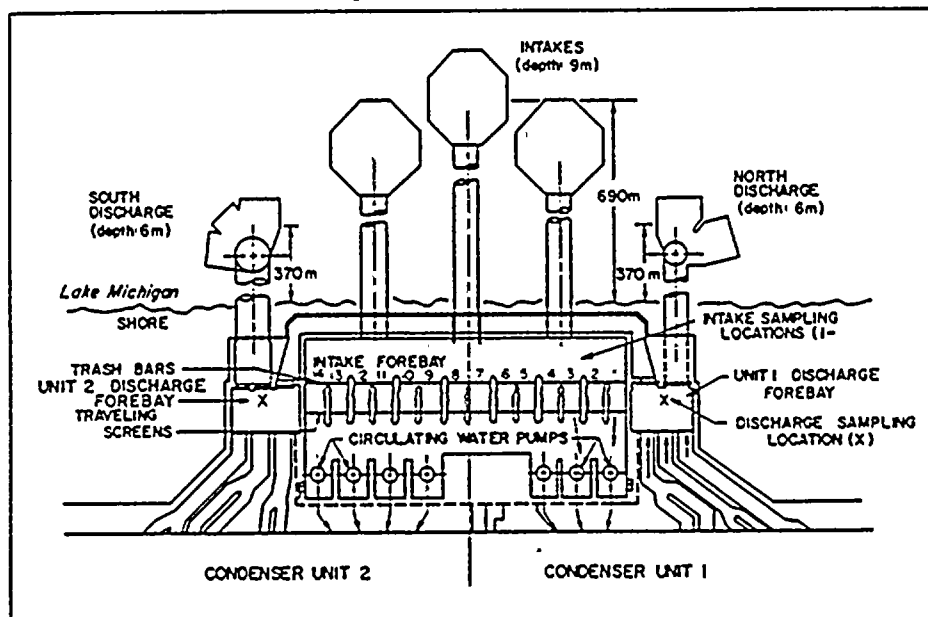


FIGURE 1. Sampling locations in the Donald C. Cook Nuclear Plant screen-house.

depth of 5.5 m as a representative sampling location and because it was preferable to continue to sample at the same location throughout the study, location 5 at 5.5 m continued in use as the representative sampling site.

Throughout the study, samples were collected from sampling location 5 and the discharge forebays (Fig. 1). Impact was assessed by comparing chlorophylls *a*, *b*, and *c*, phaeophytin *a*, the chlorophyll *a*/phaeophytin *a* ratio, and ^{14}C productivity results for the discharges with those for the intake (sampling location 5). Samples were collected in replicate (3-5) from the intake and discharges before morning twilight, at noon, and after evening twilight once per month to assess impact during periods of differing phytoplankton photosynthetic rates. During one of the sampling periods, an additional set of replicates was collected and incubated at the intake temperature. Incubation times varied between 24 and 48 hours. These samples were analyzed to determine if there was any delayed degradation of the chlorophylls. During the noon collection period, phytoplankton were collected for enumeration to the species level.

NEARSHORE SAMPLING

To detect any impact of the plant on the phytoplankton assemblages of nearshore southeastern Lake Michigan, the region was divided into a potentially-impacted inner region and a control outer region (Ayers 1978). The impacted region was described as a semi-circle centered on the plant's discharges and having a 3.2-km radius. The inner and outer regions were further divided into zones on the basis of water depth; Zone 0 was 0 to 8 m depth, Zone 1 was 8 to 16 m depth, and Zone 2 was 16 to 24 m depth (Fig. 2). Thus, for every zone, there was a potentially-

TABLE 1. Horizontal homogeneity of samples collected from the intake forebay during May 1975 (three replicates for each location). See Figure 1 for sampling locations.

Major Group	Location	Mean (cells/mL)	Std. Deviation (cells/mL)	One-way Analysis of Variance	
				F-statistic	Significance
Cocccoid Blue-Green	1	100.37	47.876	0.11917	0.8897
	3	136.63	98.428		
	5	115.40	114.43		
Filamentous Blue Green	1	41.467	14.450	0.55023	0.6034
	3	34.133	33.761		
	5	23.333	3.5218		
Cocccoid Green	1	53.700	27.084	1.9582	0.2215
	3	101.83	47.191		
	5	57.700	17.843		
Filamentous Green	1	4.6000	4.0150	0.80416	0.4904
	3	1.7667	1.6743		
	5	3.3667	1.9140		
Flagellates	1	381.20	104.86	1.2190	0.3595
	3	213.37	214.14		
	5	219.50	100.14		
Pennate Diatoms	1	92.367	52.979	0.87565	0.4638
	3	44.667	32.755		
	5	63.833	45.188		
Centric Diatoms	1	397.77	146.69	0.86047	0.4693
	3	205.90	164.71		
	5	361.00	244.58		
Desmids	1	0.9000	0.9000	0.75602	0.5095
	3	2.4000	2.8160		
	5	2.7333	1.6166		
Other	1	29.767	13.860	0.10907	0.8984
	3	24.000	20.736		
	5	24.567	14.545		
Total	1	1,102.2	279.79	0.49214	0.6340
	3	764.50	432.76		
	5	871.47	527.73		

TABLE 2. Vertical homogeneity of samples collected from the intake forebay during May 1975 (three replicates for each depth).

Major Group	Depth (m)	Mean (cells/mL)	Std. Deviation (cells/mL)	One-way Analysis of Variance	
				F-statistic	Significance
Cocccoid Blue-Green	0.6	257.83	94.921	0.74456	0.5142
	5.5	125.37	146.22		
	8.5	552.90	740.68		
Filamentous Blue Green	0.6	16.200	6.1098	0.30352	0.7489
	5.5	25.767	15.191		
	8.5	23.533	21.785		
Cocccoid Green	0.6	201.07	74.385	0.18826	0.8331
	5.5	220.97	222.83		
	8.5	274.57	117.90		
Filamentous Green	0.6	1.2000	1.0392	2.8865	0.1324
	5.5	1.2000	1.0392		
	8.5	3.0333	1.1547		
Flagellates	0.6	171.30	24.904	0.71739	0.5256
	5.5	160.90	117.26		
	8.5	256.13	140.95		
Pennate Diatoms	0.6	62.133	4.6608	0.74583	0.5137
	5.5	58.600	34.158		
	8.5	87.500	42.550		
Centric Diatoms	0.6	286.47	102.72	3.6509	0.0918
	5.5	255.27	113.00		
	8.5	514.10	160.92		
Desmids	0.6	2.1000	1.4731	0.22972	0.8014
	5.5	2.1000	1.4731		
	8.5	3.0667	2.8042		
Other	0.6	33.567	5.3948	0.75115	0.5115
	5.5	43.167	25.792		
	8.5	25.600	15.254		
Total	0.6	1,031.9	135.84	2.4142	0.1701
	5.5	893.47	362.10		
	8.5	1,740.5	787.42		

TABLE 3. Mean chlorophyll a concentrations (mg/m³) with standard errors and comparison of means using one-way analysis of variance for May 1975. See Figure 1 for sampling locations.

Location	Depth (m)	Replicates	Mean	Standard Error	F-statistic	Attained Significance Level
1	5.5	3	1.14	0.496	0.331	0.730
3	5.5	3	0.963	0.113		
5	5.5	3	0.839	0.201		
5	0.6	3	0.713	0.297	1.68	0.265
5	5.5	3	0.868	0.436		
5	8.5	3	1.69	0.462		

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TABLE 4. Mean chlorophyll b concentrations (mg/m³) with standard errors and comparison of means using one-way analysis of variance for May 1975. See Figure 1 for sampling locations.

Location	Depth (m)	Replicates	Mean	Standard Error	F-statistic	Attained Significance Level
1	5.5	3	0.147	0.0617	0.412	0.608
3	5.5	3	0.214	0.598		
5	5.5	3	0.178	0.0268		
5	0.6	3	0.174	0.0474	0.441	0.663
5	5.5	3	0.108	0.0480		
5	8.5	3	0.206	0.111		

TABLE 5. Mean chlorophyll c concentrations (mg/m³) with standard errors and comparison of means using one-way analysis of variance for May 1975. See Figure 1 for sampling locations.

Location	Depth (m)	Replicates	Mean	Standard Error	F-statistic	Attained Significance Level
1	5.5	3	0.0477	0.0477	0.591	0.584
3	5.5	3	0.173	0.173		
5	5.5	3	0.322	0.252		
5	0.6	3	0.572	0.224	1.07	0.403
5	5.5	3	0.107	0.0542		
5	8.5	3	0.316	0.316		

TABLE 6. Mean phaeophytin a concentrations (mg/m³) with standard errors and comparison of means using one-way analysis of variance for May 1975. See Figure 1 for sampling locations.

Location	Depth (m)	Replicates	Mean	Standard Error	F-statistic	Attained Significance Level
1	5.5	3	0.533	0.165	0.676	0.545
3	5.5	3	0.963	0.358		
5	5.5	3	0.502	0.373		
5	0.6	3	0.623	0.447	0.963	0.564
5	5.5	3	0.552	0.278		
5	8.5	3	0.0733	0.0369		

TABLE 7. Mean chlorophyll a concentrations (mg/m') with standard errors and comparison of means for four locations at each depth sampled using one-way analysis of variance. See Figure 1 for sampling locations.

Location	Depth (m)	Replicates	Mean	Standard Error	F-statistic	Attained Significance Level
2	0.6	5	4.10	0.0850	1.29	0.314
5	0.6	5	4.01	0.0547		
14	0.6	4	3.86	0.0874		
10	0.6	5	4.05	0.111		
2	5.5	5	4.88	0.205	6.58	0.00693
5	5.5	4	4.70	0.121		
14	5.5	3	4.34	0.259		
10	5.5	5	3.71	0.242		
2	8.5	4	4.50	0.316	2.32	0.125
5	8.5	4	4.93	0.161		
14	8.5	4	4.19	0.212		
10	8.5	5	4.42	0.0627		

TABLE 8. Mean chlorophyll b concentrations (mg/m') with standard errors and comparison of means for four locations at each depth sampled using one-way analysis of variance. See Figure 1 for sampling locations.

Location	Depth (m)	Replicates	Mean	Standard Error	F-statistic	Attained Significance Level
2	0.6	5	0.107	0.00386	0.244	0.861
5	0.6	5	0.106	0.00253		
14	0.6	4	0.118	0.00569		
10	0.6	5	0.0077	0.00196		
2	5.5	5	0.0037	0.00171	1.29	0.319
5	5.5	4	0.00919	0.00371		
14	5.5	3	0.00268	0.00268		
10	5.5	5	0.00347	0.00205		
2	8.5	4	0.158	0.00550	0.217	0.879
5	8.5	4	0.191	0.00722		
14	8.5	4	0.207	0.00363		
10	8.5	5	0.166	0.00241		

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TABLE 9. Mean chlorophyll *c* concentrations (mg/m³) with standard errors and comparison of means for four locations at each depth sampled using one-way analysis of variance. See Figure 1 for sampling locations.

Location	Depth (m)	Replicates	Mean	Standard Error	F-statistic	Attained Significance Level
2	0.6	5	0.527	0.0419	1.45	0.269
5	0.6	5	0.532	0.0268		
14	0.6	4	0.441	0.0441		
10	0.6	5	0.483	0.0223		
2	5.5	5	0.484	0.0676	0.629	0.610
5	5.5	4	0.511	0.0786		
14	5.5	3	0.456	0.134		
10	5.5	5	0.380	0.0432		
2	8.5	4	0.405	0.0510	2.86	0.0794
5	8.5	4	0.484	0.109		
14	8.5	4	0.661	0.0428		
10	8.5	5	0.582	0.0429		

TABLE 10. Mean phaeophytin *a* concentrations (mg/m³) with standard errors and comparison of means for four locations at each depth sampled using one-way analysis of variance. See Figure 1 for sampling locations.

Location	Depth (m)	Replicates	Mean	Standard Error	F-statistic	Attained Significance Level
2	0.6	5	0.347	0.0950	3.85	0.0329
5	0.6	5	0.358	0.0408		
14	0.6	4	0.646	0.0776		
10	0.6	5	0.236	0.107		
2	5.5	5	0.236	0.0680	0.306	0.818
5	5.5	4	0.300	0.157		
14	5.5	3	0.429	0.266		
10	5.5	5	0.346	0.114		
2	8.5	4	0.163	0.0769	2.20	0.138
5	8.5	4	0.0815	0.0815		
14	8.5	4	0.340	0.152		
10	8.5	5	0.363	0.0465		

TABLE 11. Mean chlorophyll a concentrations (mg/m³) with standard errors and comparison of means for three depths at each location using one-way analysis of variance. See Figure 1 for sampling locations.

Location	Depth (m)	Replicates	Mean	Standard Error	F-statistic	Attained Significance Level
2	0.6	5	4.10	0.0850	3.84	0.0555
2	5.5	5	4.88	0.205		
2	8.5	4	4.50	0.316		
5	0.6	5	4.01	0.0547	19.0	0.000752
5	5.5	5	4.70	0.121		
5	8.5	4	4.93	0.161		
14	0.6	5	3.86	0.0874	1.72	0.240
14	5.5	5	4.34	0.259		
14	8.5	4	4.19	0.212		
10	0.6	5	4.05	0.111	5.13	0.0255
10	5.5	5	3.71	0.242		
10	8.5	4	4.42	0.0627		

TABLE 12. Mean chlorophyll b concentrations (mg/m³) with standard errors and comparison of means for three depths at each location using one-way analysis of variance. See Figure 1 for sampling locations.

Location	Depth (m)	Replicates	Mean	Standard Error	F-statistic	Attained Significance Level
2	0.6	5	0.107	0.0386	2.55	0.124
2	5.5	5	0.0374	0.0171		
2	8.5	4	0.158	0.0550		
5	0.6	5	0.106	0.253	1.27	0.325
5	5.5	5	0.0919	0.0371		
5	8.5	4	0.191	0.0722		
14	0.6	5	0.118	0.0569	3.32	0.0700
14	5.5	5	0.0268	0.0268		
14	8.5	4	0.207	0.0363		
10	0.6	5	0.0770	0.0196	9.67	0.0387
10	5.5	5	0.0347	0.0205		
10	8.5	4	0.166	0.0241		

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TABLE 13. Mean chlorophyll *c* concentrations (mg/m') with standard errors and comparison of means for three depths at each location using one-way analysis of variance. See Figure 1 for sampling locations.

Location	Depth (m)	Replicates	Mean	Standard Error	F-statistic	Attained Significance Level
2	0.6	5	0.527	0.0419	1.17	0.348
2	5.5	5	0.484	0.0676		
2	8.5	4	0.405	0.0510		
5	0.6	5	0.532	0.0268	0.111	0.891
5	5.5	5	0.511	0.0786		
5	8.5	4	0.484	0.109		
14	0.6	5	0.441	0.0441	3.06	0.105
14	5.5	5	0.456	0.134		
14	8.5	4	0.661	0.0428		
10	0.6	5	0.483	0.0223	7.27	0.00943
10	5.5	5	0.380	0.0432		
10	8.5	4	0.582	0.0429		

TABLE 14. Mean phaeophytin *a* concentrations (mg/m') with standard errors and comparison of means for three depths at each location using one-way analysis of variance. See Figure 1 for sampling locations.

Location	Depth (m)	Replicates	Mean	Standard Error	F-statistic	Attained Significance Level
2	0.6	5	0.347	0.0950	1.25	0.325
2	5.5	5	0.236	0.0680		
2	8.5	4	0.163	0.0769		
5	0.6	5	0.358	0.0408	2.22	0.160
5	5.5	5	0.300	0.157		
5	8.5	4	0.0815	0.0815		
14	0.6	5	0.646	0.0776	1.01	0.407
14	5.5	5	0.429	0.266		
14	8.5	4	0.340	0.152		
10	0.6	5	0.236	0.107	0.537	0.600
10	5.5	5	0.346	0.114		
10	8.5	4	0.363	0.0465		

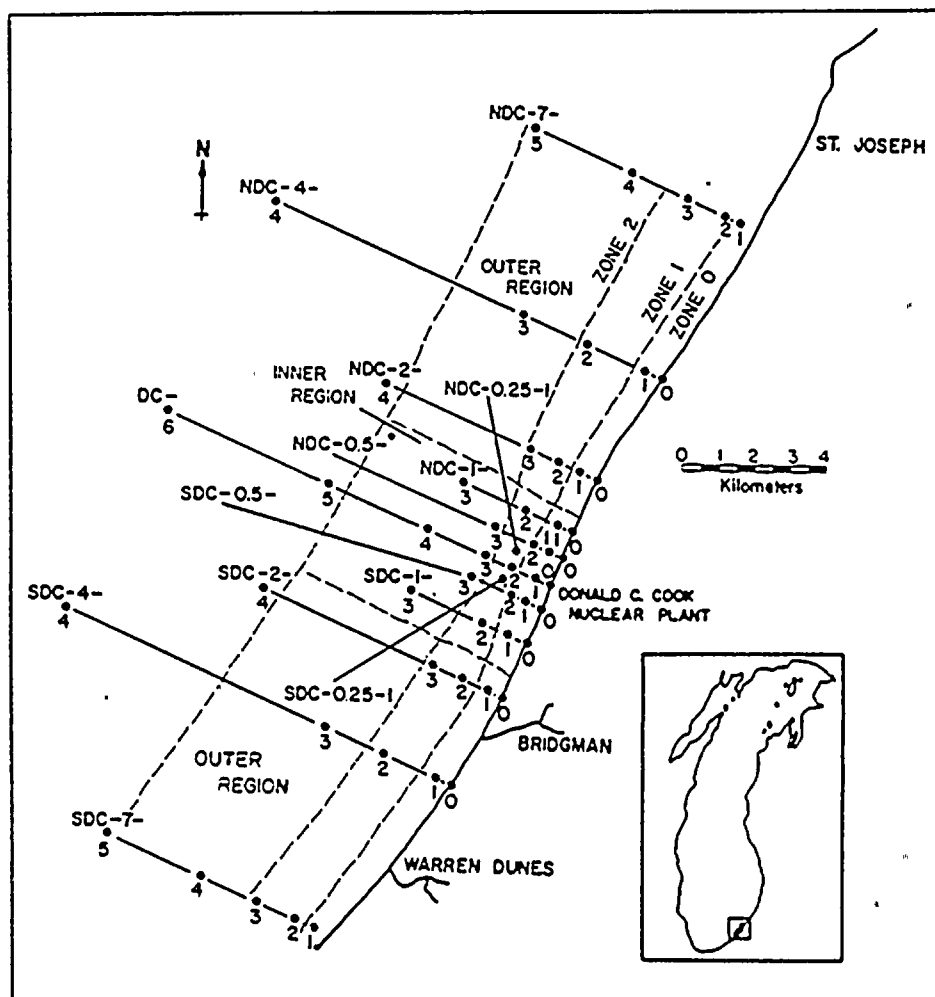


FIGURE 2. Phytoplankton sampling locations in Lake Michigan for 1970 to 1982.

impacted inner region and a control outer region. For each zone, phytoplankton assemblages of the inner and outer regions were compared using total phytoplankton cell concentrations and densities of nine major algal groups. In addition, redundancy and diversity, measures of assemblage structure, were compared for the inner and outer regions of each of the three zones (Wilhm and Dorris 1968).

Samples were collected monthly between April and November from a depth of 1 m (Ayers and Seibel 1973). Major collections were made during April, July, and October to permit collections representative of the spring, summer, and fall phytoplankton assemblages. Collections during intervening months were made to provide temporal continuity between major sampling periods. Methods used to enumerate the samples follow.

METHODS

Studies of the entrained phytoplankton at the Donald C. Cook Nuclear Plant Unit 1 began in January 1975 and continued until the termination of sampling in May 1982. Sampling was performed on a monthly basis with three approximately one-half-hour sampling periods in a 24-hour span: after twilight, morning twilight, and noon. During each sampling period, fourteen samples were collected: seven from the intake forebay and seven from the discharge forebay (Fig. 1). Two of the seven samples were preserved immediately with 6 mL of Lugols' iodine solution for microscopic determination of phytoplankton abundance and species composition. The remaining five samples were used for spectrophotometric determination of chlorophylls *a*, *b*, and *c* and phaeophytin *a*. During the evening sampling period, five additional samples were collected from both the intake and discharge forebays. These samples were incubated at the intake temperature for approximately 36 hours and treated in the same manner as nonincubated samples for analysis of the chlorophylls and phaeophytin *a*. During the noon sampling period, six additional samples were collected from the intake forebay for nutrient analysis.

Water was collected from a depth of 5.5 meters by diaphragm pumps through 3-inch hoses at an approximate rate of 227 L/min. As the water was pumped, the intake and discharge temperatures were measured, and samples were collected in 1-L polyethylene bottles. Rossmann et al. (1977) and Change et al. (1981) established the uniformity of the sampling locations across the intake forebay. Unit 1 uses 2.7×10^6 liters of cooling water per minute. Therefore, the 5-L chlorophyll sample and the 2-L phytoplankton sample represent approximately $6.2 \times 10^{-6}\%$ and $2.5 \times 10^{-6}\%$, respectively, of the water passing through the plant during a half-hour sampling period. Unit 2, which went into operation in early 1978, uses 3.5×10^6 liters per minute for cooling; thus the chlorophyll and phytoplankton samples represent about $4.8 \times 10^{-6}\%$ and $1.9 \times 10^{-6}\%$, respectively, of the water passing through the plant during a sampling period. With both units operating, the percentages are $2.7 \times 10^{-6}\%$ and $1.9 \times 10^{-6}\%$, respectively, of the water passing through the plant during a sampling period. With both units operating, the percentages are $2.7 \times 10^{-6}\%$ for the chlorophyll sample and $1.1 \times 10^{-6}\%$ for the phytoplankton sample.

OPEN-LAKE COLLECTION

Open-lake and nearshore phytoplankton samples for determining the long-term effect of the plant on phytoplankton were collected on a monthly basis from April through November, with 36-39 stations sampled during the seasonal surveys in

April, July, and October and 9-13 stations on the short surveys in May, June, August, September, and November from 1974 through May 1982 (Fig. 2). All open lake samples were collected from a depth of 1 meter with a Niskin[®] bottle. Near-shore collections (serial number zero stations) were made by submerging an open 1-L brown polyethylene bottle rinsed with lake water 10 cm below the water surface. For each sample, surface temperature was taken at the time of collection. All samples were 1L and were immediately fixed with 6 mL of Lugol's iodine solution. Both entrained and lake water samples were returned to the Ann Arbor laboratory for processing.

PHYTOPLANKTON ANALYSES

The samples of 1970, 1971, and April 1972 were prepared and counted by the Utermohl technique: placing an aliquot of the concentrated sample in a tubular combination settling and counting chamber and allowing the aliquot to settle overnight. The counting chamber containing the settled cells was then separated from the settling chamber, covered with a cover-slip, and placed on the microscope. The samples were counted on a binocular inverted microscope at 1,000X magnification.

Beginning in July 1972 and continuing to the end of the project, the method of concentration for species identification and enumeration was the settlefreeze method (Sanford et al. 1969). Permanent slides were prepared in the Ann Arbor laboratory where the 1-L samples were transferred to graduated cylinders and left undisturbed for 2 days to allow the algae to sink to the bottom. After the settling period, 900 mL of supernatant was siphoned away, leaving a 10-mL concentrated stock sample. Each concentrated sample was then mixed and a portion pipetted into a settling chamber consisting of a plexiglass cylinder clamped to a standard microscope slide. A thin layer of stopcock grease was used to form a seal between cylinder and slide. In most cases, the subsample to be settled was diluted so the algae on the resulting slide would be of countable density. Samples were left in the chambers for 2 days.

The settled algae were frozen onto the slide by carefully placing the entire chamber apparatus on a block of dry ice until the bottom 2-3 mm of sample had been frozen, about 25 seconds. The supernatant was poured off when the ice at the bottom of the chamber had melted sufficiently; the chamber was removed from the slide, and the slide with its thin wafer of ice and water was dehydrated in an anhydrous ethanol chamber for 2 days. The slide was then placed in a toluene vapor chamber for another 2 days to displace ethanol. Finally, the algae were mounted under a cover-slip using PermOUNT[®], a toluene-soluble mounting medium. The PermOUNT[®] required from 2 weeks to 1 month to harden before the slides could be counted.

All counting was done on microscopes at 1,250X and 1,000X magnifications. The scopes were fitted with oil objectives having a numerical aperture of 1.32. To help offset uneven distribution of cells on the slide, two complete 100- μ m-wide transects were made across each slide, one horizontal and one vertical. A minimum of 500 cells was counted for each slide to ensure reasonable group percentages. Additional transects were counted on slides with low density and those having a large propor-

tion of cells in dense colonial formations (e.g., *Anacystis*, *Gomphosphaeria*, *Fragilaria*, *Tabellaria*). Individual cells were counted in all species except blue-green filaments with cylindrical trichomes (*Oscillatoria* and *Schizothrix*) for which whole filaments were counted. Prior to 1974, all colonial blue-greens were counted as single organisms; the change in counting resulted in an apparent increase of blue-greens in 1974. Identification of specimens was carried to species and variety when possible. The process was limited by the condition of the cell, the condition of the sample, and the state of the taxonomy for any given group. In many cases, the cells were identifiable only to genera or major group. Species and forms are presented in the way in which they are recognized and counted. Examples are: the flagellate *Cryptomonas* is recognized and counted separately from unidentified "Flagellates"; *Anacystis* and *Chroococcus* are no longer recognized as separate entities, but counted together as *Anacystis* in accordance with Drouet's (1968) revision of blue-green taxonomy.

CHLOROPHYLLS AND PHAEOPHYTIN *a* ANALYSES

Immediately after collection, each 1-L chlorophyll sample was passed through a 4.25-cm diameter Whatman® GF/C glass fiber filter. After most of the water had passed through the filter, 1 mL of saturated $MgCO_3$ was added (1 g $MgCO_3 \cdot 4H_2O$ /100 g distilled water). The measuring flask and filtration apparatus were rinsed with distilled water. Following filtration, the filters were rolled with forceps, placed in amber vials, frozen, and transported to Ann Arbor. The samples selected for incubation were not filtered at the time of collection but were immediately placed in an incubator with the bottle caps removed and allowed to incubate in the dark for 24 to 48 hours at the intake temperature. Following this, they were filtered and treated in the same manner as the non-incubated samples, a modification of the method described by Strickland and Parsons (1972).

In the laboratory, the frozen samples were prepared for analysis by grinding with a tissue grinder and extracting with 90% acetone. The 90% acetone was prepared by swirling reagent grade acetone with anhydrous Na_2CO_3 and passing it through a Whatman® #4 filter (containing some additional Na_2CO_3) into a volumetric flask having the appropriate volume of distilled water for a 90% solution (v/v). Sample vials were removed from the freezer in groups of five and placed on ice in a dark ice chest next to the grinding apparatus. Sample vials were removed one at a time from the ice chest, and the frozen filters were transferred with forceps to a tissue grinding tube immersed in an ice bath. The filter was ground at approximately 100 rpm for 4 minutes in 1.5 to 2 mL of 90% acetone in a tissue grinding tube; the grinding tube was held firmly against the rotating pestle, lowered briefly, and raised back against the pestle approximately every 15 seconds. If the filter and 90% acetone were not reduced to a homogeneous slurry after 4 minutes, grinding was continued until this was accomplished, generally within 1 minute. The contents of the grinding tube were then poured into a 12-mL screw cap centrifuge tube. The tissue grinder was rinsed three times with 90% acetone into the centrifuge tube to adjust the final volume of 90% acetone to 10 mL. The centrifuge tube was then capped and returned to the ice chest. After all five samples were ground, they were placed in a dark refrigerator and allowed to extract for 24 to 36 hours. Following extraction, each sample was inverted three times, packed in ice, and centrifuged for 4 minutes at 2,000 rpm to

separate the filter fibers and MgCO_3 from the extract. The centrifuged samples were then refrigerated until shortly before analysis.

For analysis, individual samples were warmed to room temperature in a light-tight container. The extract was transferred using a pasteur pipette to two 5-cm long cuvettes. Two drops of 50% v/v HCl were added to the sample in one cuvette, which was shaken and then held for 4 minutes. The other cuvette was placed in a Beckman® Model 25 scanning spectrophotometer where sample absorbances were measured between 600 and 750 nm. The absorbance of the acidified sample was then measured over the same range.

PRODUCTIVITY ANALYSES

Entrained water for primary productivity estimates was collected on a monthly basis from March 1980 through May 1982. Water was pumped from the intake forebay and one of the two discharge forebays. Collections were made to coincide with the regular monthly chlorophyll sampling. When this was not possible, separate chlorophyll samples were taken along with the productivity samples. Discharge sample collection followed that of the intake by 5-10 minutes, the approximate length of time required for cooling water to pass through the condenser system.

In the laboratory, a light and dark bottle ^{14}C assimilation technique (Steeman-Nielsen 1952) was employed. The intake water was mixed and filled into 250-mL glass stoppered bottles, one dark bottle and two to five light bottles. The same preparation was made for the discharge water. Each sample bottle was then inoculated with 2 mL of ampulated ^{14}C bicarbonate solution having a specific activity of 1 $\mu\text{Ci/mL}$ and an adjusted pH of 8.8. The contents of an ampule was transferred to a sample using a 3-mL syringe and 18-gauge cannula. The ampule was then rinsed twice with approximately 2 mL of sample, the rinse being returned to the sample. The bottles were stoppered, inverted several times to mix, and placed in an incubator. The light level was 45 (edges) to 80 (center) $\mu\text{E/m}^2/\text{sec}$ (photosynthetically available radiation) at the point where the BOD bottles were placed (approximately 6 inches from the light source). The incubator temperature was set as closely as possible to the intake temperature.

After 24-hr incubation, each sample was vacuum filtered through an HA Millipore® filter (0.45 μm) at 10-20" Hg. The filters were rinsed each time with 20-25 mL of distilled water and were then transferred to glass scintillation vials to which 10 mL of aqueous counting solution was added, using an automatic pipetter. The vials were counted twice on a Beckman® LS 7500 Liquid Scintillation System. The counts were quench corrected and converted to g C m^{-3} . The available CO_2 for this conversion was estimated from the table in Saunders et al. (1962).

STATISTICAL TREATMENT OF DATA

Statistics is one of the major quantitative methods used throughout this study to ascertain the extent of impact due to the operation of the Donald C. Cook Nuclear Plant. The two main types of statistics used are descriptive and confirmatory statistics.

The descriptive statistics include the location of the observations and the extent of

dispersion of the observations. The former shows the representative value of observations and is usually indicated by the arithmetic mean (\bar{x}); whereas, the latter presents the range and distribution of the measurements and is frequently represented by the standard deviation (S. D.). In this study, both the arithmetic mean and the standard deviation are used extensively to ascertain the condition of the phytoplankton population in the water close to the plant.

Confirmatory statistics used in this study are t-statistics and ANOVA (Analysis of Variance). Both statistics are used to test the difference between the arithmetic means derived from groups of observations. The principle of these tests is to determine the probability at which the observations could be different from simple random selection or to examine whether the observations are different from simple random selection for a specific probability level. Both methods are a part of the general linear model, where t-tests are used to examine the difference between the means using the table t of the student-t distribution while ANOVA tests the observations for more than two groups based on F-distribution.

For t-statistics, if a value is greater or smaller than the student-t value for a particular level of probability in a two-tailed test as shown below (equation 1), then it is said

$$\frac{X_i - \bar{x}}{S/\sqrt{N}} \geq t_{n-1}, \quad \text{equation 1}$$

that the observed mean is significantly different from a simple random selection of the population (in other words, if the context of our study, a possible change may result).

ANOVA is based on the same principle but applies to those cases that contain more than two groups. It uses the F-ratio statistics where the test statistics are a ratio of between-group error (SSW) and within-group error (SSE), as shown in equation 2.

$$\frac{SSW}{SSE} \geq F \quad \text{equation 2}$$

If the above is greater than the expected F value, it is thought that the observations from a particular group are different from the random selection of the population.

Whether the observations differ from those of random selection is dependent upon the desired level of probability, which is largely the choice of the individual investigator. Statistical reference books, however, generally define a probability of 0.05 as "significant" and frequently represent such values with a star (*); as probability of 0.01 is considered "very significant" (**); a probability of 0.001 is "extremely significant" (***).

METHODS USED TO MEASURE IMPACT

Methods chosen to provide answers to concerns expressed about plant impact on phytoplankton were measurements of changes in chlorophylls and phaeophytin *a* concentrations, phytoplankton assemblage structure, and ^{14}C productivity. Chlorophylls and phaeophytin *a* samples were prepared using the spectrophotometric

method of Strickland and Parsons (1972). Assemblage structure was utilized in a number of ways for assessing plant impact. Samples collected from impacted and control stations were compared for differences in diversity, redundancy, dominant species, and numbers of cells in each of nine major groupings plus total cells.

IMPACT OF THE POWER PLANT ON ENTRAINED PHYTOPLANKTON

When phytoplankton pass through the condensers of an electric generating station, algae momentarily experience a rapid rise in temperature and turbulence in the surrounding water which is potentially damaging to algal cells. Three indicators of changes in cell viability in response to these stresses were chosen to assess the health of Lake Michigan phytoplankton after passing through the Donald C. Cook Nuclear Plant. Phytoplankton samples were collected from the intake and discharges to detect possible immediate and delayed algal damage using measured pigment concentrations, ^{14}C primary productivity, and samples which were enumerated to the species level to provide a visual assessment of cell damage. This last indicator of cell viability provided little information. Cell damage was never observed.

PIGMENT CONCENTRATIONS AS A MEASURE OF PLANT IMPACT

Measured concentrations of phaeophytin *a* and chlorophylls *a*, *b*, and *c* at the intake and discharges of the plant were compared using one-way analysis of variance. Utilizing three replicates and sonification for preparation of phytoplankton cells prior to extraction of the pigments during 1975 and 1976 proved to be unsatisfactory for detection of pigment changes. There were few recorded instances of passage through the plant being either stimulating or inhibiting to phytoplankton. The ability to detect chlorophyll changes with a 0.95 probability at the 0.05 level of significance was poor. Chlorophylls *a*, *b*, and *c* and phaeophytin *a* would have to change 48-79%, 104-434%, 96-150%, and 147-295%, respectively, to detect significant differences between the intake and discharges (Rossmann et al. 1977, Rossmann et al. 1979).

In order to decrease the percent change necessary to statistically detect ($\alpha = 0.05$) chlorophyll and phaeophytin *a* changes, two studies were made to compare methods of cell disruption (Change and Rossmann 1982). In the first study (September 1976), samples receiving no treatment were compared to those sonified and those ground (Table 15). For both ground and sonified prepared samples, chlorophylls *a* and *c* were higher than in the samples receiving no treatment. In addition, the ground samples always had significantly higher concentrations of chlorophylls *a* and *c*, 22% and 62%, respectively.

The second study (November 1976) was similar to the first one. The major differences were an increase in the number of replicates and a greater variety of sample treatment methods (Table 16). All sample treatments yielded higher chlorophyll *a* concentrations than the control or the sonified samples (45 seconds) (Change and Rossmann 1982). Though ground samples yielded a slightly higher concentration of chlorophyll *a* than the sonified and shaken (45 seconds) and sonified (3 minutes) samples, the difference was not statistically significant. The difference in concentration of chlorophyll *c* between ground and sonified samples was significant. Ground

samples yielded chlorophyll *c* concentrations 53% to 102% higher than sonified samples.

Beginning in 1977, five replicates were analyzed, and samples were ground prior to pigment extraction. The changes in chlorophylls *a*, *b*, and *c* and phaeophytin *a* necessary to detect significant differences between the intake and discharges with a 0.95 probability at the 0.05 level of significance became 25-40%, 400-800%, 70-80%, and 160-190%, respectively (Rossmann et al. 1980, Change et al. 1981). Except for chlorophyll *b*, the ability to detect changes in all improved. Chlorophyll *b* concentrations were considerably lower after 1976 than prior to 1976, accounting for the inability to detect significant changes.

With the improved ability to detect changes in pigments between the intake and discharges, occurrences of significant differences increased. Because the smallest, statistically significant changes could be detected for chlorophyll *a*, it was used as the primary measure of plant impact (Rossmann et al. 1982). For the period of 1975 through May 1982, chlorophyll *a* decreased significantly ($\alpha = 0.05$) in discharge samples 2.5 times more often than it increased (Table 17) (Barres et al. 1984). Increased occurrences of significant chlorophyll *a* changes after 1976 were directly related to improved analytical methodology discussed previously and in Rossmann et al. (1977). The occurrence of chlorophyll *a* decreases was almost always higher than that for increases. When chlorophyll *a* increased, indicative of phytoplankton stimulation, the average monthly increase ranged between 7 and 50%. Average decreases ranged from 7 to 46% of the intake concentration (Table 18).

There were no statistically significant differences between the mean number of occurrences of increases and decreases of chlorophyll *a* for entrained waters circulated through condensers of the Donald C. Cook Nuclear Plant for the period of 1975-1982. During 1977 and 1979, an abnormal number of chlorophyll *a* decreases and increases, respectively, occurred between the intake and discharge of the plant (Table 17). To illustrate and discuss these occurrences, the period of 1977 through 1980 is discussed. The intervening year 1978 is included for continuity, and 1980 is included as a year somewhat representative of general results for the entire study.

During 1977 and 1979, something occurred within the lake or plant that made the phytoplankton assemblages more susceptible to damage or enhancement by passage

TABLE 15. A comparison of sonification and grinding techniques for preparation of samples for pigment analysis (September 1976).

Variable	Control ¹		Ground ²		Sonified ³		Student's <i>t</i> Ground vs. Sonified
	Mean	Standard Error	Mean	Standard Error	Mean	Standard Error	
Chlorophyll <i>a</i>	6.14	0.214	7.37	0.269	6.05	0.160	4.22*
Chlorophyll <i>b</i>	0.398	0.0716	0.519	0.0486	0.426	0.0856	0.927
Chlorophyll <i>c</i>	1.07	0.144	1.80	0.0876	1.11	0.0393	7.18**
Phaeophytin <i>a</i>	2.20	0.460	1.92	0.134	2.10	0.139	-0.932
Phaeophytin <i>a</i> / Chlorophyll <i>a</i>	0.364	0.0892	0.263	0.0271	0.349	0.0321	-2.05

¹Sample extracted in 90% acetone, three replicates

²Sample ground and extracted in 90% acetone, three replicates

³Sample sonified and extracted in 90% acetone, three replicates

One star () = 0.05 level of significance. Two stars (**) = 0.01 level of significance.

TABLE 16. A comparison of sonification and grinding techniques for preparation of samples for pigment analysis (November 1976).

Variable	Control ¹		Sonified (45 sec) ²		Sonified & Shaken (45 sec) ¹		Sonified (3 min) ⁴		Ground (3 min) ⁵		Ground in Ice Bath (3 min) ⁶		Student's <i>t</i> ⁷
	Mean	Standard Error	Mean	Standard Error	Mean	Standard Error	Mean	Standard Error	Mean	Standard Error	Mean	Standard Error	Ground (3 min) vs. Sonified (45 sec)
Chlorophyll <i>a</i>	2.88	0.143	2.62	0.154	3.45	0.0900	3.07	0.151	3.69	0.102	3.71	0.0100	-5.51**
Chlorophyll <i>b</i>	0.0376	0.0242	0.	0.	0.	0.	0.	0.	0.0206	0.0123	0.	0.	—
Chlorophyll <i>c</i>	0.494	0.0977	0.484	0.0709	0.590	0.0685	0.455	0.128	0.902	0.100	0.920	0.0385	-3.49**
Phaeophytin <i>a</i>	0.977	0.168	1.20	0.241	0.408	0.139	0.406	0.0883	0.632	0.159	0.356	0.0960	1.89
Phaeophytin <i>a</i> / Chlorophyll <i>a</i>	0.352	0.0684	0.489	0.122	0.120	0.0435	0.130	0.0235	0.176	0.0469	0.0530	0.0141	2.22

¹Six replicates²Six replicates³Two replicates⁴Three replicates⁵Five replicates⁶Two replicates⁷One star (*) = 0.05 level of significance. Two stars (**) = 0.01 level of significance.

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through the plant's condensers. Possible sources for the anomalous results of 1977 and 1979 that could be investigated included changes in phytoplankton assemblage, changes in nutrient concentration, the occurrence of upwelling, and a change from one-unit to two-unit operation for the power plant. The last of these was quickly excluded. The change from one-unit operation to two-unit operation in 1978 did not produce results in 1978 that were distinctly different from those of other years (Tables 17 and 18).

TABLE 17. Percent occurrence of statistically significant changes ($\alpha = 0.05$) in chlorophyll *a* concentration between intake and discharge samples.

Year	Increase		Decrease	
	Non-Incubated	Incubated	Non-Incubated	Incubated
1975	0	0	0	5
1976	6	0	8	8
1977	0	0	30	70
1978	5	17	22	17
1979	16	30	15	12
1980	3	8	17	8
1981	3	17	14	25
1982 (through May)	13	0	13	0
	$\bar{x} = 5.8$	9.0	14.9	16.9
	$\alpha = 5.9$	11.3	8.8	19.0

TABLE 18. Mean changes in chlorophyll *a* concentration at times when statistically significant ($\alpha = 0.05$) increases or decreases in its concentration were detected between the intake and discharge of the plant.

Year	Hours Incubated	Percent Increase in Chlorophyll <i>a</i>	Percent Decrease in Chlorophyll <i>a</i>
1975	0	0	0
	39	0	46
1976	0	23	16
	36	0	28
1977	0	0	22
	33-39	0	25
1978	0	50	16
	33-36	14	17
1979	0	14	24
	28-36	27	20
1980	0	-13	14
	36-38	7	7
1981	0	13	10
	35-38	23	11
1982	0	23	11
	35-37	0	0

Changes in phytoplankton assemblage may have been at least partly responsible for the anomaly of 1977 (Table 19). Flagellate concentrations were relatively high and centric diatom concentrations were relatively low in 1977 compared to those between 1978-1980. Species or forms assigned to these groups accounted for slightly more than one-third of the total phytoplankton density for each of the years. Based on the work of Holland (1968, 1969), Stoermer and Yang (1969, 1970), and Holland and Beeton (1972), Tarapchak and Stoermer (1976) assigned species of diatoms to an apparent trophic level preference for Lake Michigan. For those diatom species, identified during the current study, for which a preferred trophic level is known, a summary of the annual occurrence of dominant diatom forms (> 10% of the total phytoplankton population) was tabulated (Table 20). During 1977, the last occurrence of *Cyclotella stelligera* as a dominant diatom was noted. During the study, it was the only dominant diatom associated with mesotrophic waters but intolerant of nutrient enrichment. During the same year, dominant eutrophic diatoms disappeared. These were replaced by mesotrophic forms tolerant of moderate nutrient enrichment. In 1975, *Cyclotella stelligera* accounted for approximately 10% of the entire phytoplankton population (Barres et al., 1984). By 1977, it averaged less than 3% of the phytoplankton population.

Between 1977 and 1980, there were no significant year-to-year differences in orthophosphate and dissolved silica concentrations (Table 21). However, the highest mean concentration of each nutrient, especially dissolved silica, occurred in 1979.

TABLE 19. Mean yearly concentrations (cells/mL) of the major groups of algae at the Donald C. Cook Nuclear Plant intake, 1977-1980.¹

Group	1977 ¹	1978 ¹	1979 ¹	1980 ¹
Cocoid blue-green	599 (159)	850 (134)	513 (144)	745 (199)
Filamentous blue-green	35 (10)	48 (13)	42 (19)	69 (28)
Cocoid green	80 (22)	128 (23)	89 (13)	170 (46)
Flagellates	540 (114)	454 (69)	375 (40)	389 (63)
Centric diatoms	366 (94)	623 (64)	479 (214)	810 (217)
Pennate diatoms	685 (155)	731 (75)	639 (236)	891 (255)
Other	110 (13)	161 (20)	93 (12)	157 (32)
Total	2,380 (228)	3,000 (272)	2,240 (387)	3,240 (488)

¹Desmids always less than 1 cell/mL. Numbers in parentheses are standard deviations.

TABLE 20. The annual number of occurrences of dominant diatom forms (> 10% of total population) for each trophic level for 1975-1981 and 5 months of 1982.

Trophic Level	1975	1976	1977	1978	1979	1980	1091	1982
Mesotrophic, intolerant of nutrient enrichment	34	11	1	0	0	0	0	0
Mesotrophic, tolerant of moderate nutrient enrichment	47	70	92	57	62	125	74	79
Eutrophic	8	13	0	28	30	56	58	11

Because of the statistical insignificance of differences, we can only hypothesize that a relatively high supply of nutrients to the region of the lake sampled by the plant may have in some way been partly responsible for the higher chlorophyll *a* concentrations observed in water passing through the condenser.

Upwelled phytoplankton may have been more susceptible to degradation of chlorophyll *a* during condenser passage than those not upwelled. For the period of 1977 to 1980, occurrences of upwellings of hypolimnion water were most frequent in 1977 (Table 22). Upwellings occurred either the week prior to sampling or the week of sampling. Roughly one-half of the statistically significant occurrences of chlorophyll *a* decreases between the intake and discharges of the plant in 1977 occurred during periods of upwelling.

Thus, it is hypothesized that changes in the phytoplankton assemblage structure combined with upwelling events produced a situation where the phytoplankton were more susceptible to chlorophyll *a* degradation during passage through the condensers of the plant. For the period of 1976 through spring 1982, the number of phytoplankton passing through the plant average 7.4×10^{19} cells/year (Table 23). For these cells, phytoplankton viability decreased 15.5% of the time during condenser passage with chlorophyll *a* decreasing 7 to 46% at the discharge. Viability increased 5.6% of the time with chlorophyll *a* increasing 7 to 50% at the discharge.

TABLE 21. Nutrient concentrations in intake waters of the Donald C. Cook Nuclear Plant

Year	Orthophosphate ($\mu\text{g/L}$)		Dissolved Silica (mg/L)	
	\bar{x}	σ	\bar{x}	σ
1977	1.2	0.40	0.52	0.33
1978	1.4	1.1	0.73	0.23
1979	1.5	1.3	0.97	0.66
1980	0.55	0.32	0.57	0.41

TABLE 22. Number of occurrences of upwelling prior and during entrainment sampling at the Donald C. Cook Nuclear Plant.

Year	Occurrences of Upwelling During Sampling	Occurrences of Upwelling Prior to Sampling	Total Upwellings
1977	1	4	5
1978	2	0	2
1979	1	1	2
1980	1	1	2

TABLE 23. Number of phytoplankton entrained by the plant during 1976-1981 and the first 5 months of 1982. — indicates no data.

Month	Numbers Entrained						
	1976	1977	1978	1979	1980	1981	1982
January	4.25×10^{11}	—	1.79×10^{11}	3.03×10^{11}	5.41×10^{11}	3.91×10^{11}	3.34×10^{11}
February	1.59×10^{11}	—	5.06×10^{11}	1.85×10^{11}	2.16×10^{11}	2.78×10^{11}	4.02×10^{11}
March	2.22×10^{11}	2.87×10^{11}	8.21×10^{11}	3.68×10^{11}	3.06×10^{11}	7.04×10^{11}	7.19×10^{11}
April	3.49×10^{11}	4.32×10^{11}	2.27×10^{11}	5.82×10^{11}	4.53×10^{11}	1.04×10^{12}	1.20×10^{12}
May	5.45×10^{11}	2.13×10^{11}	7.58×10^{11}	6.21×10^{11}	1.65×10^{12}	3.05×10^{12}	6.54×10^{11}
June	1.81×10^{11}	1.83×10^{11}	1.09×10^{12}	—	1.35×10^{12}	7.47×10^{11}	
July	9.57×10^{11}	2.53×10^{11}	1.08×10^{12}	1.85×10^{11}	6.19×10^{11}	5.16×10^{11}	
August	3.79×10^{11}	2.48×10^{11}	4.05×10^{11}	3.48×10^{11}	9.49×10^{11}	8.85×10^{11}	
September	5.89×10^{11}	2.48×10^{11}	4.05×10^{11}	3.48×10^{11}	9.49×10^{11}	8.85×10^{11}	
October	3.28×10^{11}	3.07×10^{11}	1.26×10^{12}	7.70×10^{11}	1.28×10^{12}	8.58×10^{11}	
November	3.60×10^{11}	3.07×10^{11}	1.26×10^{12}	7.70×10^{11}	1.28×10^{12}	8.58×10^{11}	
December	3.46×10^{11}	2.11×10^{11}	1.36×10^{12}	1.15×10^{12}	8.09×10^{11}	9.70×10^{11}	
Total	4.84×10^{12}	2.68×10^{12}	7.85×10^{12}	5.66×10^{12}	1.06×10^{13}	1.26×10^{13}	3.31×10^{12}

¹⁴C MEASURED PRIMARY PRODUCTION AS A MEASURE OF PLANT IMPACT

Algal primary production rate estimates are a proven measure of cell metabolism, and primary production has been observed to decrease after algae circulate through power plants (Hickman and Klarer 1975). Decreases in production were observed for the Allen steam-electric generating plant at Lake Wylie, North Carolina (Gurtz and Weiss 1972). The Allen plant was one of the only a few case studies where samples were collected similarly to those at the Donald C. Cook Nuclear Plant, just prior to and just after passing through the condensers. Studies at the Palisades Plant on Lake Michigan, conducted somewhat similarly to those at the Donald C. Cook Nuclear Plant, showed primary production decreased an average of 37.4%; however, the results were not consistent (Consumers Power Company 1973).

Because of the rather large changes in chlorophyll *a* concentration required to detect plant impact on phytoplankton, ¹⁴C primary production rate measurements were made from March 1980 through May 1982. Excluding April 1981, measurements were made each month.

Light and dark bottles were incubated at 45 (edges) to 80 (center) $\mu\text{E}/\text{m}^2/\text{sec}$ (photosynthetically available radiation) at the point where the biological oxygen demand (BOD) bottles were placed (approximately 15 cm from the light source). Incubation occurred for 24 hours at the temperature of intake waters at the time of collection. Each 250-mL BOD bottle was spiked with 2 mL of ¹⁴C bicarbonate solution having a specific activity of 1 $\mu\text{Ci}/\text{mL}$ and an adjusted pH of 8.8.

At the Donald C. Cook Nuclear Plant, primary productivity decreased 16 to 76% for each month between March 1980 and May 1982, excluding April 1981 (Table 24) (Change and Rossmann 1985). This was statistically significant ($\alpha = 0.05$). Mean decreases were 53, 50, and 39% for 1980, 1981, and 1982, respectively. The smaller mean decrease in 1982 was not significantly different from those of 1980 and 1981,

TABLE 24. *Decreases in phytoplankton primary productivity for the period of March 1980 through May 1982 at the Donald C. Cook Nuclear Plant (— = no sample)*

Month	Percent Decrease in ¹⁴ C Measured Primary Productivity Between the Intake and Discharge		
	1980	1981	1982
January	—	40	25
February	—	48	68
March	58	32	16
April	32	—	29
May	43	67	56
June	48	37	—
July	46	48	—
August	57	67	—
September	61	48	—
October	76	62	—
November	73	57	—
December	37	41	—

and no significant differences between seasons were found with succession of phytoplankton assemblages.

These results are in sharp contrast to those obtained using chlorophyll *a*. Decreases in chlorophyll *a* between the intake and discharge ranged between 0 and 215% for 1980 to 1982 (Table 17). The differences between chlorophyll *a* and ¹⁴C productivity results are postulated to be derived from the response of phytoplankton to passage through the plant's condensers. It is postulated that condenser passage was enough to greatly diminish phytoplankton photosynthesis; however, the impact of passage was not severe enough of or a long enough duration to degrade chlorophyll *a* to phaeophytin *a*. Cells which passed through the condensers showed little or no evidence of physical damage. Attempts to incubate the samples collected for chlorophyll *a* analyses for periods greater than 36 hours always met with failure. After 36 hours, the phytoplankton assemblage changed and was no longer representative of the indigenous collected assemblage. The permanence of productivity decreases after passage through the plant's condensers is also unknown. Longer incubation times met with the same failures as those for chlorophyll *a*. Thus the ability of algal cells to photosynthesize greatly diminished after passage through the plant's condensers; however, the permanence of the diminished photosynthetic capability is unknown.

Because chlorination ceased at the Donald C. Cook Nuclear Plant prior to 1980, the observed primary productivity reductions were believed to be related to thermal shock, hydromechanical damage to cells, or to changes in the availability of nutrients being related to corrosion of power plant parts (Change and Rossmann 1985). Experiments were conducted to determine the temperature at which significant decreases in primary productivity began to occur. Intake samples heated to the temperature of the discharge water experienced no significant reduction in primary productivity. However, intake samples heated to temperatures exceeding 35°C had a marked reduction in primary production, and intake samples subjected to temperatures greater than 50°C had very low primary productivities. In order to reduce primary productivity to levels found in the discharges of the plant, numerical analysis has shown the discharge water must consist of a mixture of waters consisting of parcels differentially heated during the 6-second passage time through the plant's condensers. This mixture of waters is postulated to contain 35 to 55% water heated to temperatures above 35°C, 20 to 30% water heated to temperatures below 35°C, and 25 to 35% water heated to temperatures between those of the discharge and intake (Chang and Rossmann 1985). Thus, simple heating of water to the temperature of the discharge cannot account for the observed decreases in primary production.

Microscopic examination of phytoplankton cells passing through the plant revealed no evidence of hydromechanical destruction of the cells (Change and Rossmann 1985). Fragments of phytoplankton did not significantly increase in discharge samples relative to intake samples. Similarly, the lack of consistent decreases in chlorophyll *a* concentrations between the intake and discharge was not observed and supports the microscopic observations of no significant increase in hydromechanical damage to the cells.

Investigations by Change and Rossmann (1985) led them to conclude that the presence of microelements could be a factor important to the observed decrease in primary production for phytoplankton passing through the plant. The addition to discharge water of ethylenediamine tetra acetic acid (EDTA), which renders metals inaccessible to phytoplankton, significantly enhanced the primary production.

Thus, some metal toxicity to phytoplankton was postulated to be occurring in the plant's circulating system due to corrosion of plant components (Change and Rossmann 1985).

IMPLICATIONS OF OBSERVATIONS

The Donald C. Cook Nuclear Plant has an impact upon entrained phytoplankton. The impact is not apparent from pigment analyses but is very apparent from primary productivity studies. A possible explanation for the apparent inability to detect plant impact on phytoplankton using pigment analyses is that algal cells may experience significant stress without pigment degradation. The observed reduction in photosynthetic rate may or may not be permanent. All attempts to determine whether or not the photosynthetic rate returned to normal after a period of recovery failed. Incubation of the samples for any period of time beyond that required for carbon fixation altered the assemblage structure severely and negated all results.

The impact of phytoplankton with a reduced photosynthetic rate on the nearshore phytoplankton community was negligible. Primary productivity measurements of samples collected from the thermal plume revealed no significant measurable decrease in productivity relative to that of the intake (Fig. 3). The lack of a measurable decrease may be in part a function of the discharge diffuser design. The diffusers were designed to entrain ambient water rapidly for the purpose of limiting the size of the region impacted and the severity of impact by the thermal plume. Samples collected near the plant's discharge in the lake represented only 30% of the water that actually passed through the plant (Evans et al. 1978). This dilution was enough to have placed the expected productivity within the error bars for the measured productivity. Thus, no statistically significant impact by the plant on phytoplankton production in the lake was found. However, it is important to note that there appeared to be no additional impact by the thermal plume on phytoplankton production within the lake. Any such impact should have depressed productivity enough for statistically significant differences to have been evident.

IMPACT OF THE DONALD C. COOK NUCLEAR PLANT ON SOUTHEASTERN LAKE MICHIGAN PHYTOPLANKTON

Lake phytoplankton surveys were performed to provide a description of phytoplankton abundances in spring, summer, and fall. They also provided species and group compositions under preoperational conditions against which the same parameters from surveys similarly conducted under operational conditions could be contrasted to determine long-term changes that might be attributable to Cook Plant operation.

To assess the impact of the Cook plant on nearshore southeastern Lake Michigan, the occurrences of differences in phytoplankton assemblage structure between plant-impacted stations and control stations were sought. One method used to identify changes in assemblage structure was the measurement of numbers of cells in each of the major phytoplankton groups. In addition, diversity and redundancy of the phytoplankton assemblage were calculated.

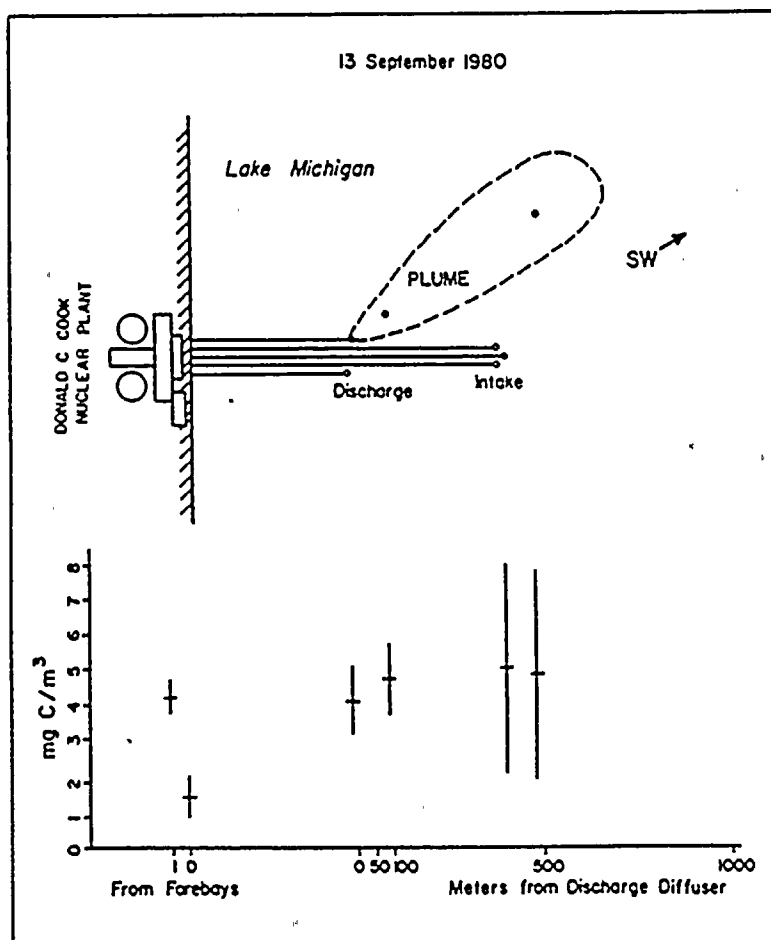


FIGURE 3. Phytoplankton primary productivity (mg C/m³) in Lake Michigan versus distance from a discharge diffuser of the Donald C. Cook Nuclear Plant.

CHANGES IN MAJOR PHYTOPLANKTON GROUPS

The period of comparison extended from July 1970 through April of 1982. The plant became operational in 1975. Only statistically significant ($\alpha \leq 0.05$) differences between inner and outer regions of each zone (Fig. 2) are noted in the sections which follow.

Coccoid Blue-green Algae

For the years 1970-1982, coccoid blue-green algae concentrations increased between 1970 and 1982 in all zones (Fig. 4). Concentrations were always highest in October (fall). Zone 2 concentrations of coccoid blue-green algae were statistically ($\alpha \leq 0.05$) greater during fall of 1975 and summer of 1978 in the inner region (impacted) than in the outer region (control stations, see Fig. 2) (Ayers and Wiley 1979, Ayers and Feldt 1982). In the spring of 1979, their concentrations were higher in the outer region than in the inner region in zone 0 (Ayers and Feldt 1982). Differences in water masses sampled did not account for the elevated concentrations in the inner region during fall of 1975 or spring of 1979. However, a large upwelling occurred during July 1978 sampling (Evans et al. 1982). This may account for the differences between the outer region and the inner region. In general, impacted (inner region) and control (outer region) abundances within each zone were similar to one another.

Filamentous Blue-green Algae

Filamentous blue-green algae concentrations increased in all zones throughout the duration of the study (Fig. 5). Peak concentrations occurred during the summers (July) of 1977, 1979, and 1981. On three occasions, filamentous blue-green algae concentrations were highest in the inner region. These occurred in zone 2 during the fall of 1975 and the summers of 1976 and 1977 (Ayers and Wiley 1979). Algal concentrations were higher in the outer region of zone 1 during the summer of 1975 (Ayers and Wiley 1979). In the fall of 1975, the inner region of zone 2 was within an area of elevated water temperature relative to lower water temperatures in the outer region (Evans et al. 1978). This difference appears to be distinctly related to plant operation.

Coccoid Green Algae

Between 1970 and 1982, coccoid green algae concentrations were generally quite stable (Fig. 6). Coccoid green algae concentrations were higher in the inner region than in the outer region during the summers of 1970, 1971, and 1977 and during fall of 1976 in zone 2 (Ayers and Wiley 1979). Their concentrations were higher in the outer region than in the inner region of zone 0 during fall of 1975 (Ayers and Wiley 1979).

Filamentous Green Algae

Except for April 1976, abundances of filamentous green algae were less than 75 cells/mL for the entire study period (Fig. 7). During April 1976, concentrations peaked at 145 cells/mL in the inner region of zone 1. Peak concentrations at this time were caused by the occurrence of *Ulothrix* sp. For the entire period of observation, there were no occurrences of significant differences between impacted (inner region) and control (outer region) stations (Ayers and Feldt 1983).

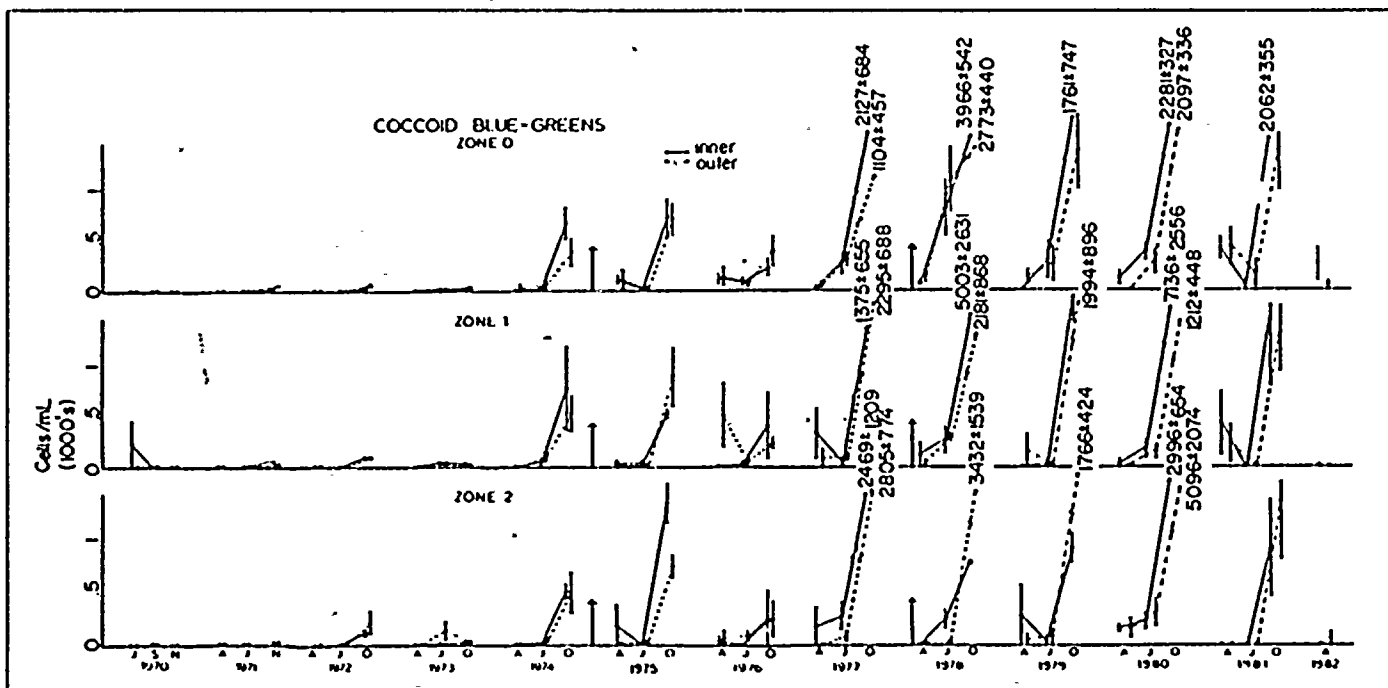


FIGURE 4. Mean abundances of coccoid blue-green algae in zones 0-2 in the spring, summer, and fall seasonal surveys of 1970 through April 1982. The vertical bars show standard errors. The vertical arrows represent the startup of units 1 and 2 in 1975 and 1978, respectively.

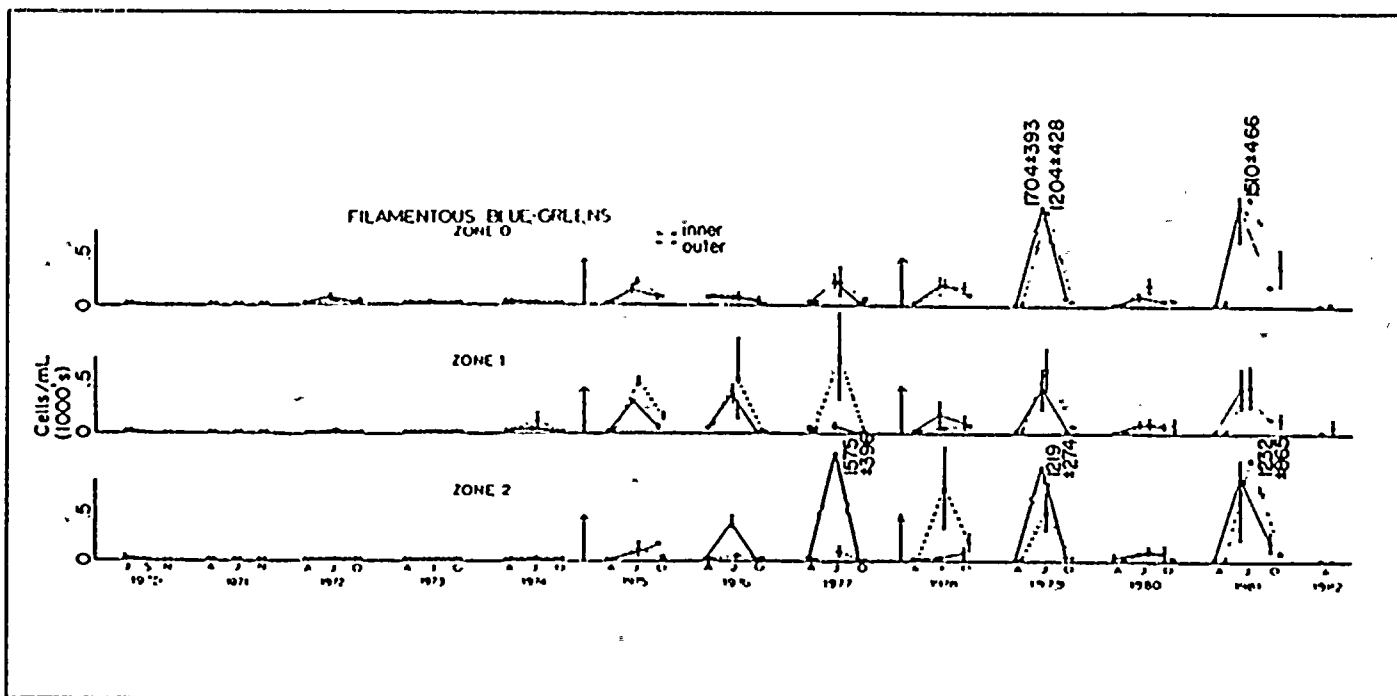


FIGURE 5. Mean abundances of filamentous blue-green algae in zones 0-2 in the spring, summer, and fall seasonal surveys of 1970 through April 1982. The vertical bars show standard errors. The vertical arrows represent the startup of units 1 and 2 in 1975 and 1978, respectively.

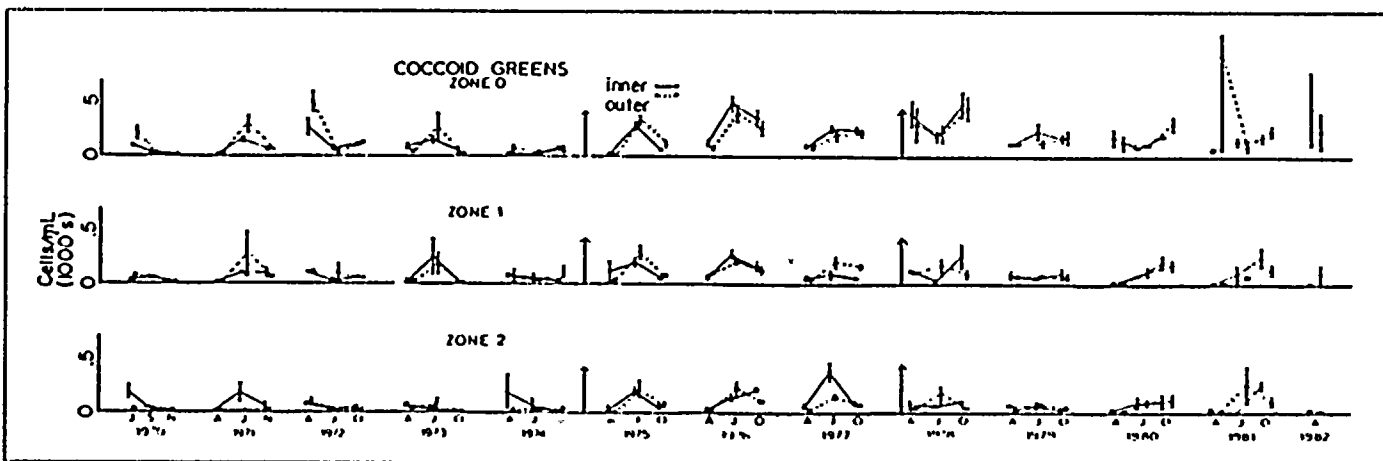


FIGURE 6. Mean abundances of coccoid green algae in zones 0-2 in the spring, summer, and fall seasonal surveys of 1970 through April 1982. The vertical bars show standard errors. The vertical arrows represent the startup of units 1 and 2 in 1975 and 1978, respectively.

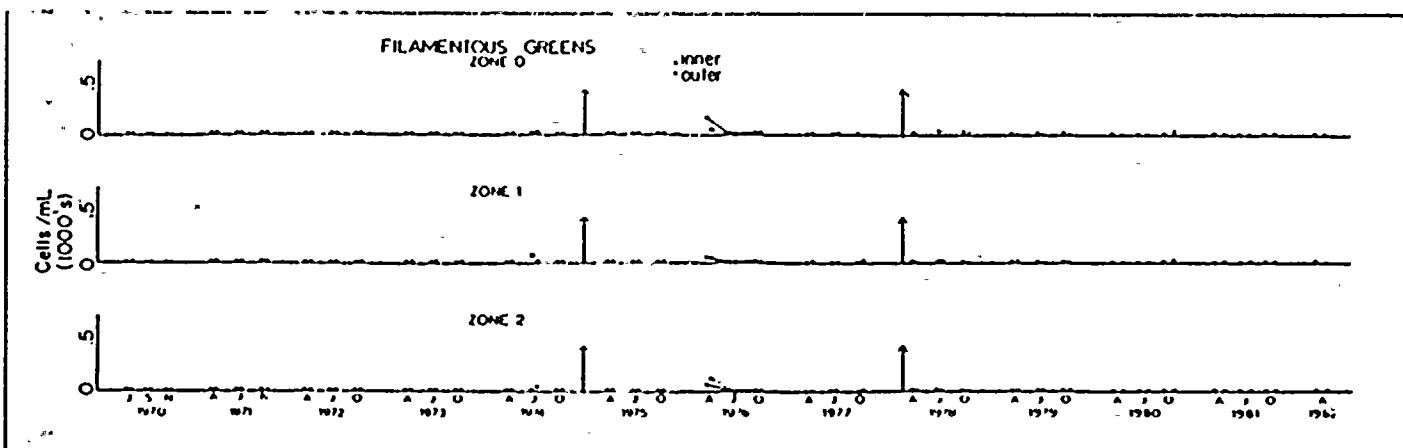


FIGURE 7. Mean abundances of filamentous green algae in zones 0-2 in the spring, summer, and fall seasonal surveys of 1970 through April 1982. The vertical bars show standard errors. The vertical arrows represent the startup of units 1 and 2 in 1975 and 1978, respectively.

Flagellates

Flagellate concentrations increased throughout the 13 years of study (Fig. 8). Their concentrations were higher at the control stations (inner region) than at the impacted stations (outer region) during the springs of 1972 and 1980 for zone 0, the summers of 1971 and 1977 in zone 1, the falls of 1973 and 1977 in zone 1, and the falls of 1974 and 1979 in zone 2 (Ayers and Wiley 1979, Ayers and Feldt 1982, Ayers and Feldt 1983). Flagellate concentrations were higher in the inner region during the fall of 1976 in zone 2 and during the summer of 1980 in zone 1 (Ayers and Wiley 1979, Ayers and Feldt 1983). Upwelling events in the fall of 1979 and the occurrence of a thermal plume from the plant in the fall of 1977 may have been responsible for lower concentrations of flagellates in the inner regions (Evans et al. 1982).

Centric Diatoms

For the period of observation, centric diatom concentrations were relatively high during 1972-1973, 1975-1976, and 1978-1979 in zone 0; 1973, 1975, and 1979 in zone 1; and 1975 and 1979 in zone 2 (Fig. 9). Centric diatom densities were higher in the outer region of zone 1 during spring of 1972 and of zone 2 during the spring of 1980 (Ayers and Wiley 1979, Ayers and Feldt 1983). Diatom densities were higher in the inner region during fall of 1972 in zone 1 and during fall of 1975 in zones 0 and 2. A thermal plume originating at the plant was evident during the fall of 1975 (Evans et al. 1978) and may have been responsible for elevated centric diatom densities in the inner regions of zones 0 and 2.

Pennate Diatoms

Pennate diatom densities increased between 1970 and 1978 in zone 0 and between 1970 and 1977 in zones 1 and 2 (Fig. 10). Since 1977-1978, pennate diatom densities have been decreasing. Pennate diatom densities were higher in the outer region during the summer of 1970 in zone 1, the summer of 1971 in zone 2, and the spring of 1973 in zone 1 (Ayers and Wiley 1979). Concentrations of these diatoms were highest in the inner region during the fall of 1975 in zone 2 and the fall of 1979 in zone 1 (Ayers and Wiley 1979, Ayers and Feldt 1982). A thermal plume originating from the plant was identified in the fall of 1975 (Evans et al. 1978). As it was for the centric diatoms, it may have also been responsible for elevated pennate diatom densities in zone 2 during 1975.

Desmids

The desmids showed almost no variation for the entire study period. Desmid concentrations were low throughout the duration of the study. Concentrations were higher in the outer region during summer of 1971 in zone 1 and were higher in the inner region during summer of 1971 in zone 2 (Ayers and Wiley 1979).

Other Algae

Other algae densities increased between 1970 and 1978 and then decreased slightly, but remained relatively constant after 1978 (Fig. 11). Concentrations of other algae were higher in the outer region than in the inner region during the springs of 1971

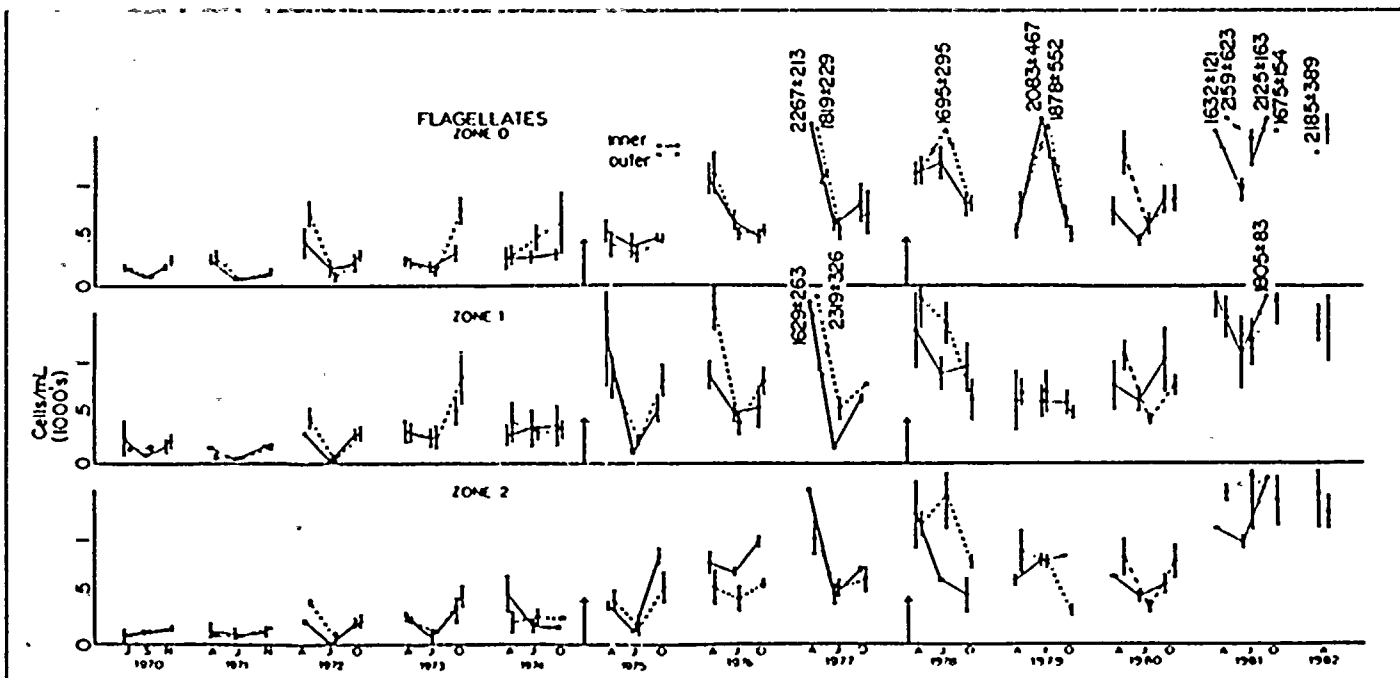


FIGURE 8. Mean abundances of flagellates in zones 0-2 in the spring, summer, and fall seasonal surveys of 1970 through April 1982. The vertical bars show standard errors. The vertical arrows represent the startup of units 1 and 2 in 1975 and 1978, respectively.

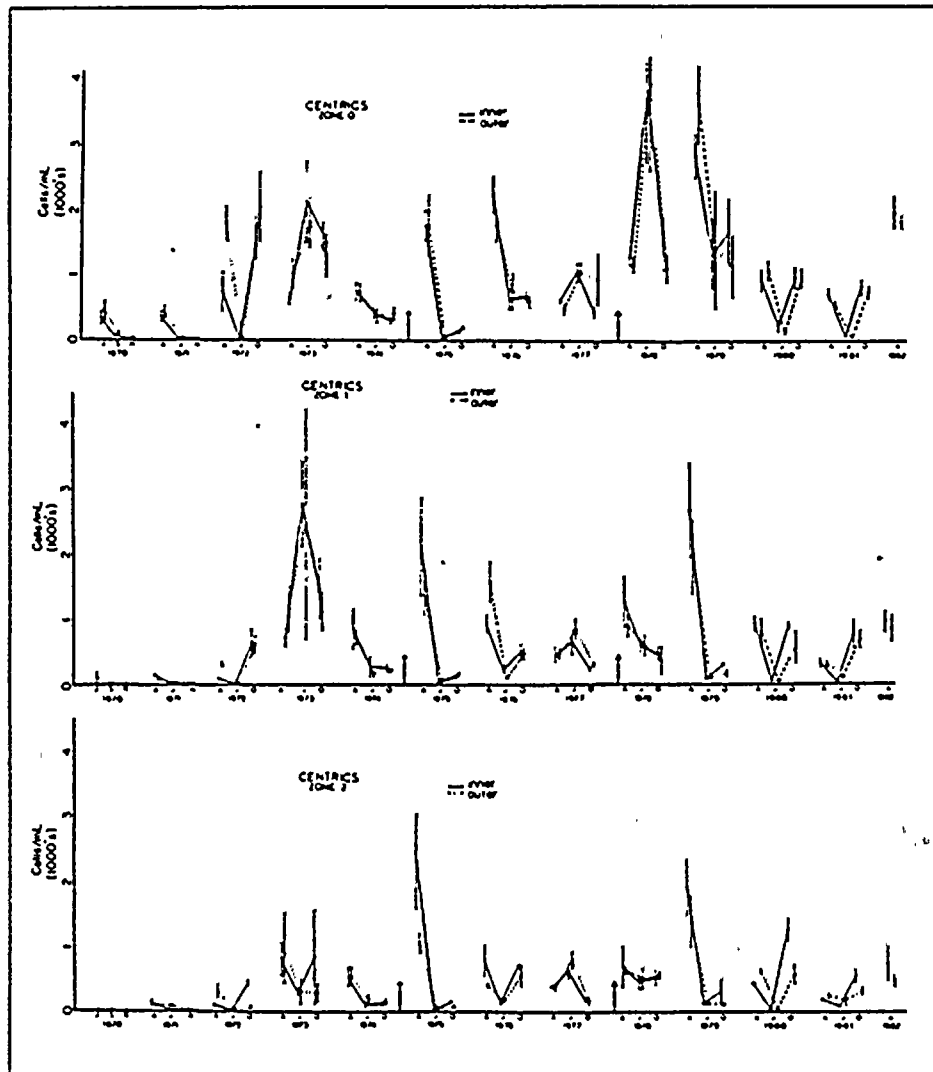


FIGURE 9. Mean abundances of centric diatoms in zones 0-2 in the spring, summer, and fall seasonal surveys of 1970 through April 1982. The vertical bars show standard errors. The vertical arrows represent the startup of units 1 and 2 in 1975 and 1978, respectively.

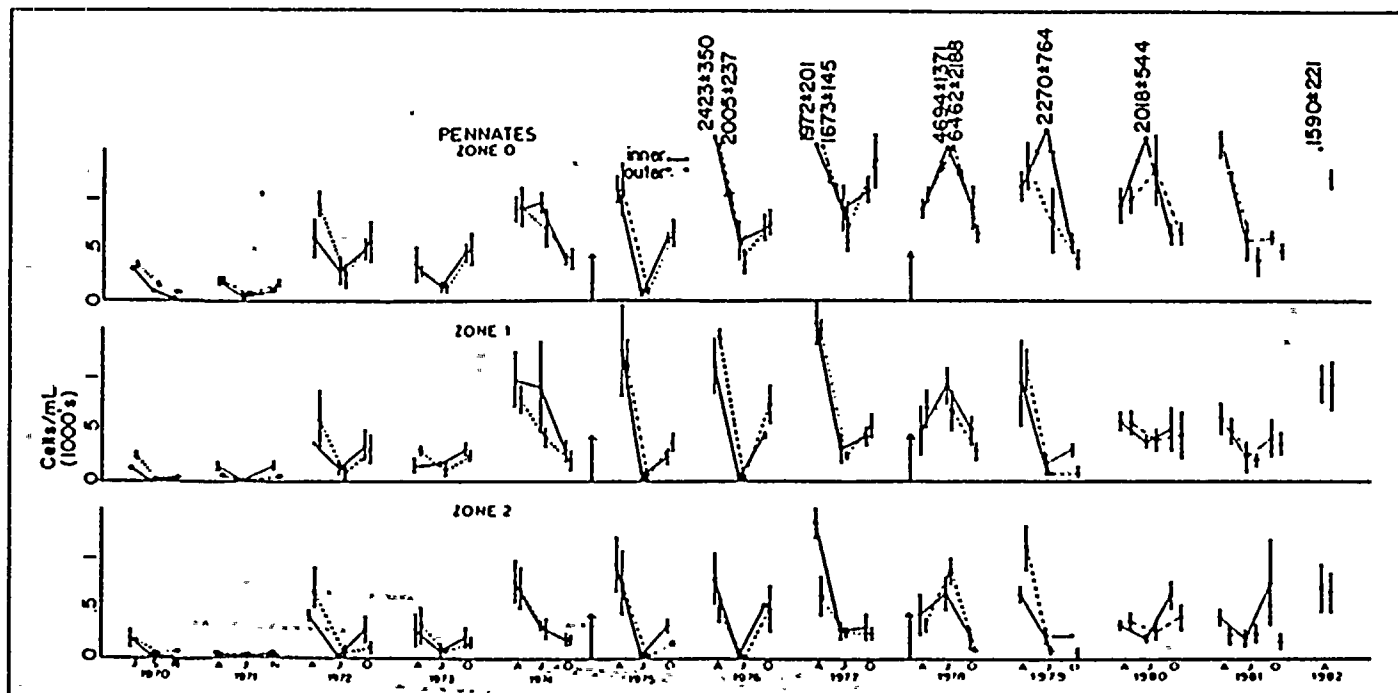


FIGURE 10. Mean abundances of pennate diatoms in zones 0-2 in the spring, summer, and fall seasonal surveys of 1970 through April 1982. The vertical bars show standard errors. The vertical arrows represent the startup of units 1 and 2 in 1975 and 1978, respectively.

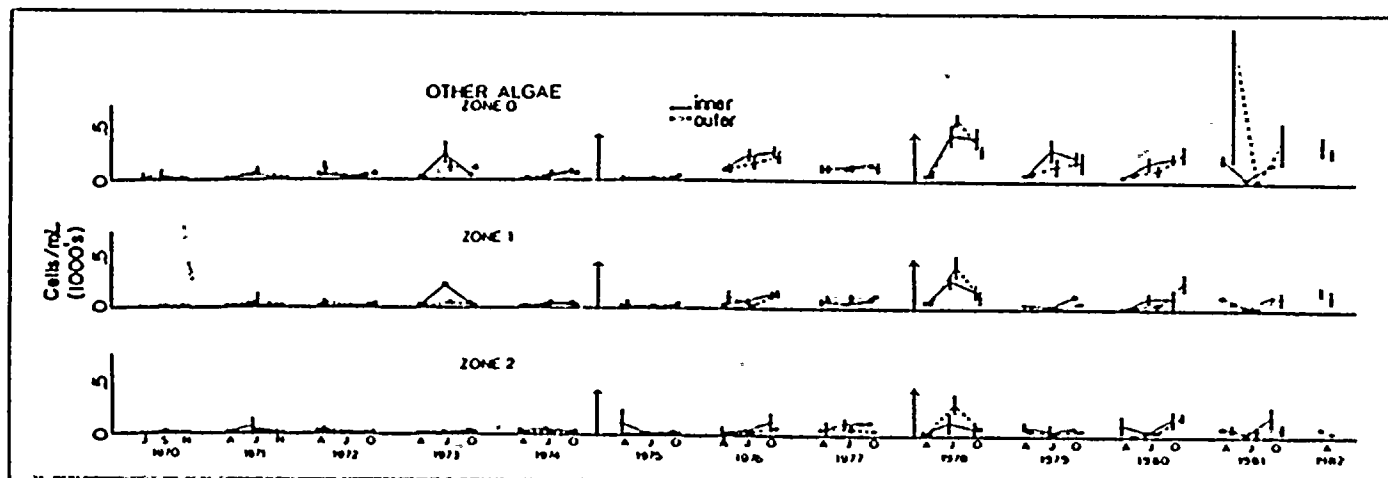


FIGURE 11. Mean abundances of "other algae" in zones 0-2 in the spring, summer, and fall seasonal surveys of 1970 through April 1982. The vertical bars show standard errors. The vertical arrows represent the startup of units 1 and 2 in 1975 and 1978, respectively.

and 1980 in zone 1 (Ayers and Wiley 1979, Ayers and Feldt 1983). Concentrations were higher in the inner region than in the outer region during the springs of 1973 in zone 0, 1974 in zone 2, and 1982 in zone 2, the summers of 1973 in zone 1 and 1981 in zone 0, and the falls of 1973 and 1977 in zone 2 (Ayers and Wiley 1979, Ayers and Feldt 1982, Ayers and Feldt 1983). A thermal plume attributable to the plant was observed in the fall of 1977 (Evans et al. 1982). Elevated concentrations of other algae may have been related to the plant's thermal plume in the inner region of zone 2 during the fall of 1977.

Total Algae

Total algae cell densities increased between 1970 and 1978 (Fig. 12). After 1978, concentrations were relatively high compared to the early years of the study. Total algae concentrations were higher in the outer region than in the inner region during the springs of 1972 in zones 0 and 2 and 1976 in zone 1 and the summer of 1978 in zone 2 (Ayers and Wiley 1979, Ayers and Feldt 1982). Concentrations were higher in the inner region than in the outer region during the summers of 1977 and 1979 in zone 2 (Ayers and Wiley 1979, Ayers and Feldt 1982).

CHANGES IN DIVERSITY AND REDUNDANCY

As for the changes in major phytoplankton groups, changes in diversity and redundancy (Wilhm and Dorris 1968) revealed no plant impact (Ayers and Feldt 1983). Statistically significant differences between diversities and redundancies at control and impacted stations for each zone were very few in number (Figs. 13 and 14). Those few differences found are believed to result from sampling a heterogeneous spatial distribution of phytoplankton. Water masses have been described as resulting from stream inputs, thermal bar development, and upwelling of hypolimnion water (Rossmann 1986). These water masses of differing physical and chemical character are hypothesized to be responsible for observed variations in the spatial distribution of phytoplankton in nearshore regions of the Great Lakes.

IMPLICATIONS OF OBSERVED VARIATIONS

For the period of 1970 to 1982, phytoplankton numbers and assemblage structure varied in response to changes within the lake itself and not to plant activities. Changes occurring at impacted stations (inner region) varied in unison with changes occurring at control stations (outer region) for both preoperational years (1970-1974) and operational years (1975-1982). For the few occasions on which a plume was readily detectable in the spring and fall, the plant may have accelerated the natural process of warming the waters adjacent to the shoreline in the spring and retarded their cooling in the fall. During the fall of 1975, springs of 1977 and 1978, and falls of 1977 and 1978, a thermal plume was evident which could account for the observed differences and represent a definitive recording of plant impact on the nearshore region of southeastern Lake Michigan. However, the number of significant differences found is less than 10% of the comparisons made for the period of observation. The differences identified for pre- and post-operational data are similar. Thus, the number of significant differences between control (outer region) and

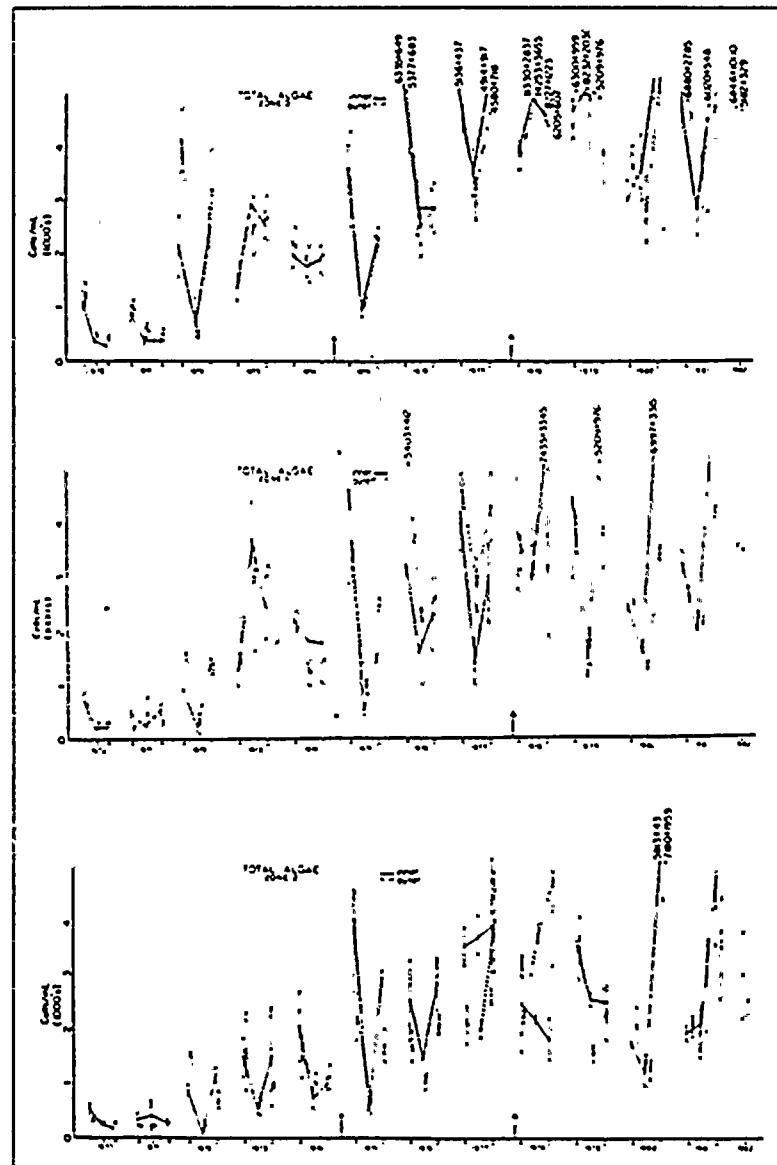


FIGURE 12. Mean abundances of total algae in zones 0-2 in the spring, summer, and fall seasonal surveys of 1970 through April 1982. The vertical bars show standard errors. The vertical arrows represent the startup of units 1 and 2 in 1975 and 1978, respectively.

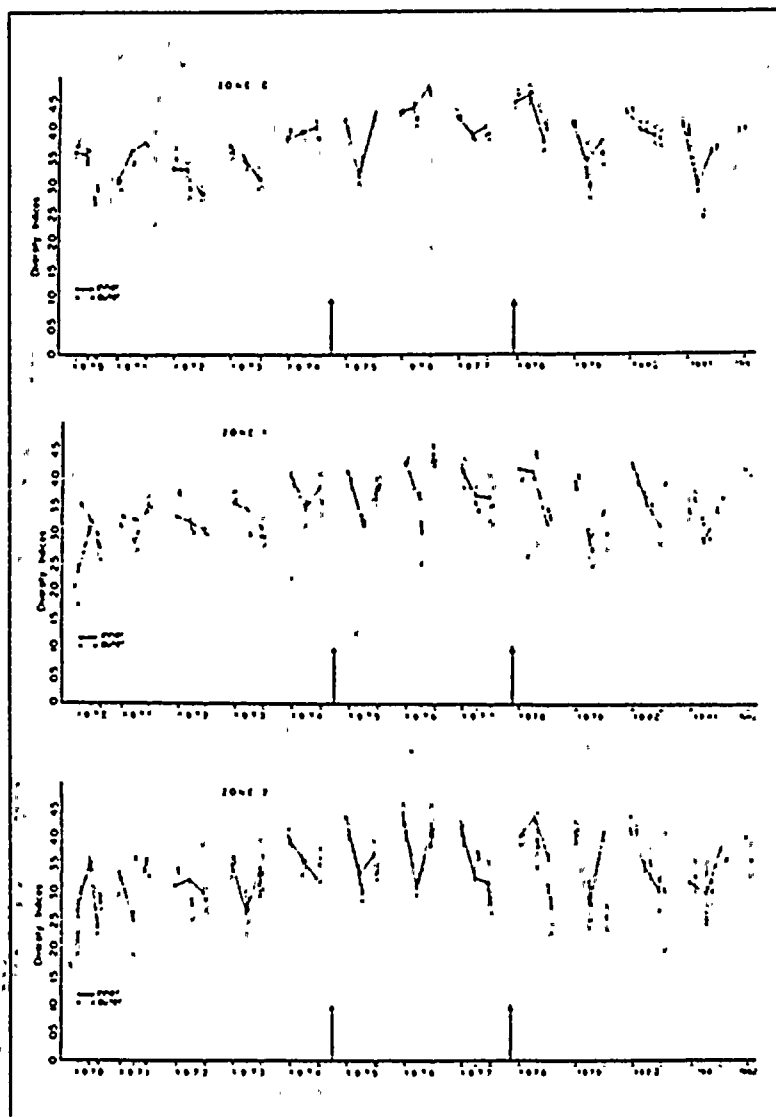


FIGURE 13. Mean diversities of phytoplankton in zones 0-2 in the spring, summer, and fall seasonal surveys of 1970 through April 1982. The vertical bars show standard errors. The vertical arrows represent the startup of units 1 and 2 in 1975 and 1978, respectively.

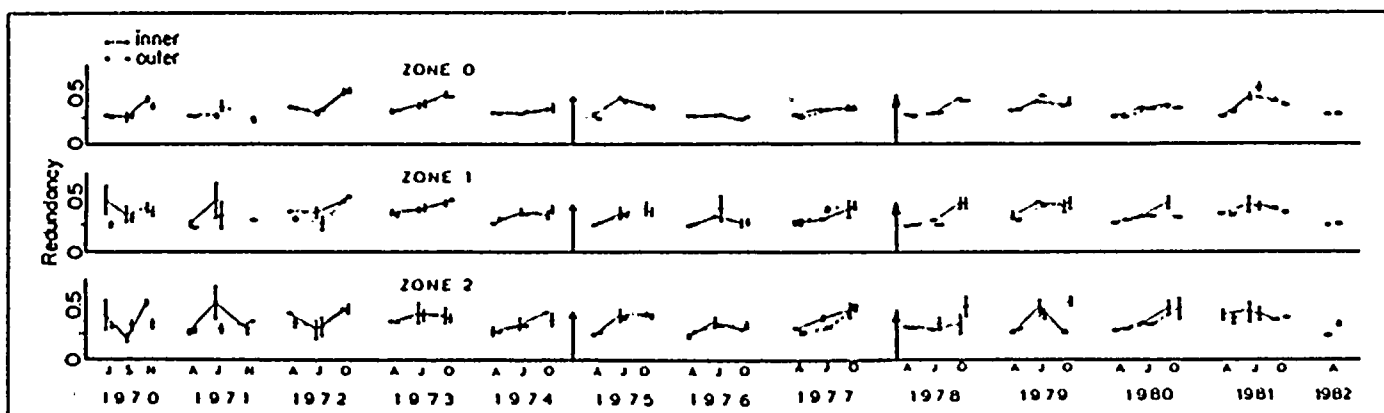


FIGURE 14. Mean abundances of phytoplankton in zones 0-2 in the spring, summer, and fall seasonal surveys of 1970 through April 1982. The vertical bars show standard errors. The vertical arrows represent the startup of units 1 and 2 in 1975 and 1978, respectively.

impacted (inner region) stations is within the range of natural variation (Ayers and Feldt 1983) and does not necessarily represent a consistently measured plant impact on the lake.

SUMMARY

Impact of the Donald C. Cook Nuclear Plant was undetectable within the lake despite the fact that phytoplankton productivities were significantly reduced during circulation through the plant. Chlorophyll *a* concentrations in cooling waters circulated through the plant were seldom altered during plant passage even though phytoplankton productivities decreased between 16% and 76% between the intake and discharge. There is no evidence from which to draw any conclusions about permanent damage to the phytoplankton. Though productivities decreased, chlorophylls did not. Thus the potential for recovery existed.

Because of the design of the plant's discharge diffusers, detection of an immediate plant impact upon the lake was impossible with the sampling plan and methods of measurement utilized for the study. The three-fold or more rapid dilution of discharge water with ambient lake water precluded measuring any immediate impact by the thermal discharge on the phytoplankton assemblage. The presence of this heat source in nearshore southeastern Lake Michigan does not appear to have had any detectable impact on the lake's phytoplankton for the 7 years of study during which the plant was operational. Changes in phytoplankton numbers and assemblage structure which occurred at impacted stations (inner region) paralleled similar changes at the control stations (outer region). Changes occurring during the years of plant operation were similar in magnitude to those occurring during pre-operational years. Therefore, the plant has had no statistically demonstrable impact upon the nearshore southeastern Lake Michigan phytoplankton.

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PHYTOPLANKTON POPULATIONS OF SOUTHEAST LAKE MICHIGAN 1974-1982

James A. Bowers, Ronald Rossmann, James Barres, and William Y. B. Chang

INTRODUCTION

To do science is to search for repeated patterns in nature (MacArthur 1972). Phytoplankton populations, which provide the energy base for limnetic food webs, have a dynamic community structure with temporal-spatial patterns cycling on diel and annual time scales. These rhythms are accompanied by short- and long-term random interruptions. Space scales in this dynamic structure range from microns to hundreds of kilometers. Both abiotic and biotic processes controlling algal species composition and algal interactions with the rest of the pelagic community also fluctuate in this continuous space-time plane. For example, at micron-millisecond coordinates, micronutrient patches created by excreting zooplankton partially define the uptake and competition for phosphorus and nitrogen (Lehman and Scavia 1982). At diel-decimeter scales, algal reproduction and mortality track the daily solar radiation cycle and diel vertical migration patterns of zooplankton herbivores (Bowers and Grossnickle 1978). Phytoplankton seasonal succession cycles predominate at annual-kilometer (basin) coordinates. Here, water temperature and photoperiod provide a rhythmic physical framework for cell division and death.

Stochastic interruptions to these cyclical patterns occur at any point on the space-time plane. Langmuir cells occur at hour-meter points, while up(down)welling events are observed at hour-kilometer coordinates. At the far corner of the plane are the effects of cultural eutrophication which stimulates algal productivity. Changes in phytoplankton community structure are best evaluated within this framework of short-term spatial-temporal dynamics and the long-term effects of eutrophication (Reynolds 1984).

PHYTOPLANKTON OF LAKE MICHIGAN

Beeton (1965, 1969) first awakened the scientific community to the accelerated eutrophication of the Great Lakes. He pointed out the striking correlation between demographic trends in Great Lakes basins with significant deteriorations in water quality. One of the more important changes noted was the increased phosphorus loading to the lakes which stimulated algal productivity. Although phosphorus limits phytoplankton biomass in Lake Michigan (Schelske and Stoermer 1972), increased phosphorus loading also stimulates diatoms enough to deplete dissolved silica levels in the lake; at this point a dominance of the diatoms ceases with a subsequent increase in green and blue-green species (Schelske and Stoermer 1971). In fact, silica depletion in Lake Michigan appears to be a reasonable measure of cultural eutrophication (Schelske et al. 1983).

Most long-term studies of algal changes have focused on the Bacillariophyta

because their silica frustules provide excellent taxonomic characters and are relatively immune from destruction in the sediments, providing an historical record. Long-term comparisons remain difficult. Stoermer and Yang (1968) noted that many earlier qualitative observations in Lake Michigan were performed by amateur diatomists. To collect algae, many early studies used fine plankton nets instead of water bottles (Stoermer 1978), thereby selectively omitting many small unicellular species. Even recently, samples were collected solely from the surface layers. During thermal stratification, some large filamentous diatoms reside primarily in the thermocline stratum and are underrepresented in collections (Brooks and Torke 1978). Enumeration techniques are sometimes selective. Often, diatom preparation techniques destroy delicate phytoflagellates. With these restrictions, diatom composition can be used to detect some obvious long-term structural trends in the eutrophication in Lake Michigan. Studies by Stoermer and Yang (1968) and Stoermer (1978) have correlated diatom species changes with water quality and serve as excellent examples of this approach.

Species composition temporally fluctuates in temperate lakes, forming a seasonal succession pattern. This, too, has been influenced by eutrophication (Stoermer 1978). Two trends were discovered. First, in oligotrophic waters the succession pattern possesses only one mode. When Damann (1966) plotted total plankton counts from the Milwaukee and Chicago municipal water "crib" intakes from 1940 to 1963, a unimodal pulse was observed in July at the Milwaukee intake where more oligotrophic conditions existed. At Chicago, a bimodal curve was observed with peak abundances during April-May and October. A second trend has been increased amplitudes of seasonal variability. Spring and fall blooms are much more pronounced and bisected by an obvious midsummer low. Light limitation of the continual intense production limits primary production. During isothermal periods, water movements were the least complex and almost entirely wind driven, but brief and episodic up(down)welling events with subsequent generation of internal waves masked any detectable net circulation pattern. Wind stresses, especially from storms along the main axis of the lake, alter the preceding current patterns. Shallower regions respond more quickly than deeper areas, because wind stress is inversely proportional to the depth of any layer. The generally observed current patterns are rotary motions offshore at near-interstitial frequencies and complex shore-parallel currents with frequent reversals in the nearshore regions (Sato and Mortimer 1975). This construct translates into a relatively isolated offshore component and a well mixed inshore area separated by a shearing zone. These inshore-offshore physical differences are reflected in the phytoplankton. Algal generation times and diatom densities are respectively shorter and higher inshore (Holland 1969, Holland and Beeton 1972) with the eutrophic species found inshore (Stoermer et al. 1971).

PHYSICAL LIMNOLOGY OF LAKE MICHIGAN

The current structure of Lake Michigan is a major contributor to the inshore-offshore differences seen in the phytoplankton community. While many physical studies focused on surface or whole basin dynamics (Ayers et al. 1958, Johnson 1960, *U.S. Department of Commerce* 1967, Noble and Anderson 1968, Noble and Ewing 1968, Johnson and Mortimer 1971, Huang 1971, Mortimer 1971), these efforts failed to predict net circulation patterns. Within the nearshore zone, defined by Mortimer (1971) to be less than 17 km from shore, two special types of current

structures dominate water movements, the springtime thermal bar and summertime up(down)wellings. Both significantly affect phytoplankton growth and reproduction. In April, when Lake Michigan has a temperature of approximately 3°C, progressive spring warming begins and leads to a convective current regime called a thermal bar (Mortimer 1971). This shoreparallel and very abrupt horizontal temperature gradient restrict water mass exchange between the offshore (<4°C) and the nearshore (>4°C) regions. At the bar itself, Forel (1895) postulated that the 4°C water, the most dense, sinks, thereby initiating sinking convergences. As warming continues, the thermal bar moves offshore, eventually dissipating upon itself in the central basin (Rogers 1966).

Although Forel's model has not been proven, the impact of thermal bars on phytoplankton communities in the trapped warmer waters behind the bar has been well documented in two studies. In Lake Ontario during April 1965, Nalewajko (1966, 1971) observed algal densities two to three times greater behind the bar. Interestingly, the population was dominated by a single diatom, *Stephanodiscus tenuis*, which in some samples comprised half of the assemblage. When the warm water moved offshore, this population followed the bar. Algal densities in areas beyond and behind the bar were positively correlated with nutrients and not temperature. Behind the thermal bar offshore from Grand Haven harbor, Lake Michigan, phytoplankton densities were fivefold higher than on the offshore side (Stoermer 1967). At the temperature interface, densities were an order of magnitude greater than in the offshore community. Species composition fell into four distinct floral groups: Grand Haven harbor, the region behind the bar, the region at the bar, and finally the much colder offshore waters. As in Nalewajko's (1966) study, species composition correlated only with their respective nutrient regimes.

After thermal bar disappearance and resulting stratification, strong persistent winds from the prevailing south-southwest or storms generated from the northeast lift cold hypolimnetic water toward the surface from equilibrium levels, creating large up(down)welling areas (Mortimer 1971). Up(down)wellings are a source of mass vertical water transport. These episodes are frequent in Lake Michigan and the other Great Lakes due to their large surface area and their size-dependent response to Coriolis forces. Up(down)wellings have been observed up to 20 km from the shoreline with intensities dependent upon wind speed and duration. They have been extensively studied in large lakes due to their capacity to generate long internal waves (seiches), large scale geostrophic flows, and coastal jets (Csanaday 1970). They are the mechanisms that advectively transport water masses along the shoreline.

Up(down)wellings are the most important driving force behind large-scale short-term fluxes, both vertical and horizontal, in the nearshore phytoplankton. Using water intake records from Milwaukee, Wisconsin, and Muskegon, Michigan, Bowers (1980) demonstrated the effect of up(down)wellings on the deep chlorophyll layer which "bounced" with the vertical displacements of water from internal seicheing. A further and much more important vertical effect results from the lifting of deep nutrient-rich water above the compensation depth for algal production. In Lake Michigan, highest phytoplankton densities and production rates are observed in upwelling areas during the summer where silica and nitrogen concentrations are much higher than in the offshore epilimnion (Schelske et al. 1971). Silica influxes also maintain diatom species within the upwelling area, while green and blue-green species dominate the silica-depleted regions farther offshore. Large filamentous forms are especially successful (Bothwell 1975). Stoermer et al. (1971) noted sub-

stantial changes within 24 hours, and emphasized the transient nature of upwelling communities.

In general, thermal bars and up(down)welling events create spatial heterogeneities among water masses which have different algal nutrient compositions. Theoretical considerations (Tilman 1977) and laboratory experiments (Tilman and Kilham 1976) have demonstrated that these limiting nutrient gradients affect species competition for phosphorus and silica. When this approach was applied to Lake Michigan, silica to phosphorus ratios accounted for 70% of the total variance in the test diatom species. More recently, Kilham and Kilham (1980) have proposed that changes in limiting nutrient resource gradients are the principal controlling factor of phytoplankton.

APPROACH

An intensive sampling program in the nearshore area adjacent to the Donald C. Cook Nuclear Plant was performed from 1972 to 1982 to assess the phytoplankton community of Lake Michigan. Analyses of these samples will be couched in the following approaches: long-term changes in algal group composition, changes in seasonal succession patterns, inshore-offshore differences, and the effects of temperature through thermal bars and up(down)welling events.

LONG-TERM CHANGES IN PHYTOPLANKTON COMMUNITY

CHANGES IN OFFSHORE STATIONS

Evaluating long-term changes in phytoplankton abundance, group composition, and seasonal succession required the use of the three farthest offshore sampling stations, SDC-4-4, DC-6, and NDC-4-4, and all of the monthly sampled stations. The offshore locations, positioned 11.2 km from shore, are the least likely locations to be affected by the transient nearshore current structure (Fig. 1). Only sampling during April, July, and October included all three offshore stations from 1974-1982. During 1982, samples were collected only in April. Means and standard errors were calculated for the three stations to minimize the large station-to-station variance due to the phytoplankton's horizontal patchiness. Long-term changes from 1974 to 1982 were also analyzed separately for each of the 3 months to negate successional effects on algal composition.

During April, total cell abundance at the offshore stations appeared to increase slightly (Fig. 2), but regression analysis indicated the increase was not significant ($P > 0.10$). Likewise, there were no significant trends in nutrients (Table 1). Algal group composition (Fig. 3) represented typical Lake Michigan springtime conditions throughout the 9-year period, with flagellates and diatoms dominating the offshore community. Between the Aprils of 1975 and 1976, there were significant decreases in centric and pennate diatoms with concurrent increases in flagellates ($P < 0.01$). Nutrient levels were not correlated with these changes, with the exception of 1979 when Si:SRP (Table 1) were significantly greater than other years correlated with the diatom increases from 1978 to 1979.

Total cell densities in July were much more variable than in April (Fig. 4) with no

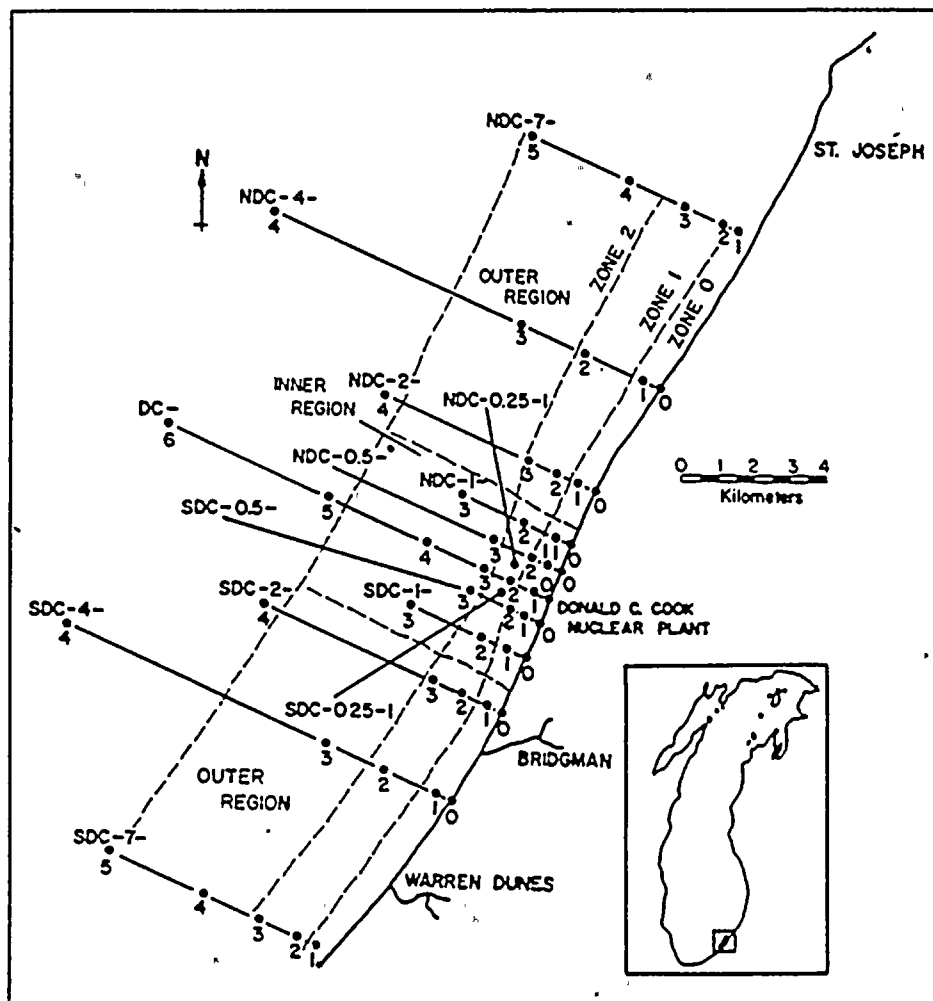


FIGURE 1. The sampling grid for algae at the Donald C. Cook Nuclear Plant.

detectable trends. In 1978, there was a significant increase in total cell density, which corresponded with a large-scale upwelling event centered approximately 0.5 km offshore. Surface temperature was much lower than in other years (Table 1). Accompanying this increase were increases in the flagellate and pennate diatom groups (Fig. 5). Other than this upwelling event, no noticeable trends were detected in the July sampling series. Nutrient data, limited to only 1977-1981 at the three offshore stations, also had no detectable trends.

The October data set was even more variable than that of July (Fig. 6). Rough weather prevented sampling during 1975 and 1976. Though total cell counts were variable, the relative proportions of each of the groups remained constant through

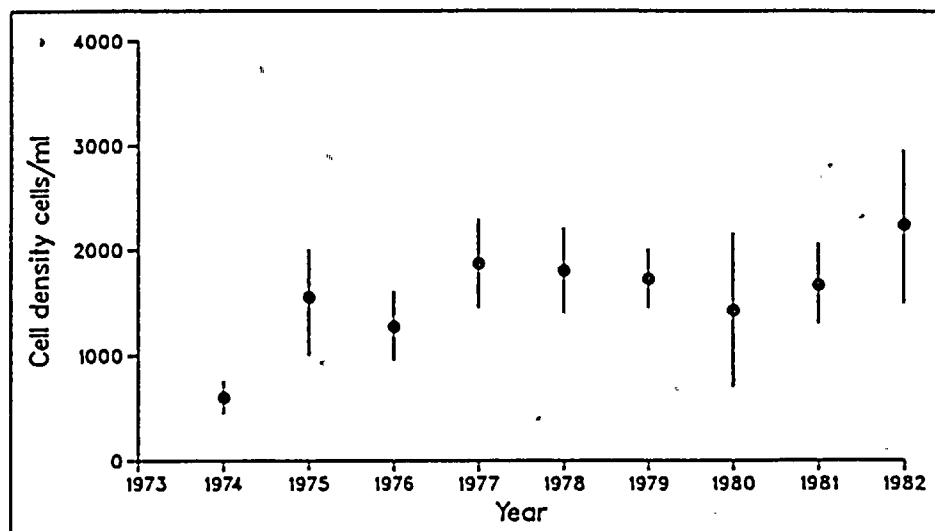


FIGURE 2. Total cell densities during April 1974-1982 at the three offshore stations of the sampling grid.

TABLE 1. Nutrient concentrations at the three offshore Donald C. Cook Nuclear Plant sampling stations during April, July, and October 1977-1982. SRP = soluble reactive phosphorus; TP = total phosphorus.

	Year	P-SRP ¹ ($\mu\text{g}\cdot\text{L}^{-1}$)	Si-SiO ₂ ² ($\mu\text{g}\cdot\text{L}^{-1}$)	P-TP ¹ ($\mu\text{g}\cdot\text{L}^{-1}$)	Temp. °C	Si:SRP Atomic Ratio
April	1977	0.74	0.42	5.00	6.1	292
	1978	2.29	0.86	12.50	3.9	195
	1979	0.27	0.87	17.00	3.1	1,611
	1980	1.17	1.13	5.90	2.4	497
	1981	1.38	1.22	5.60	4.3	453
	1982	0.95	0.56	15.10	3.2	302
July	1977	0.74	0.10	1.60	16.1	70
	1978	1.70	0.90	7.10	11.8	273
	1979	1.30	0.84	NS	18.9	333
	1980	1.40	0.20	NS	17.1	74
	1981	0.20	0.25	3.80	22.4	697
October	1977	0.96	0.65	2.80	10.8	352
	1978	1.40	0.89	8.90	16.0	331
	1979	0.82	0.74	NS	14.6	477
	1980	0.50	0.60	9.7	11.8	625
	1981	0.65	1.20	9.4	13.9	952

¹ $\mu\text{g}\cdot\text{L}^{-1}$ SRP or TP = 0.032 μMIP

² $\text{mg}\cdot\text{L}^{-1}$ SiO₂ = 16.7 μMSi

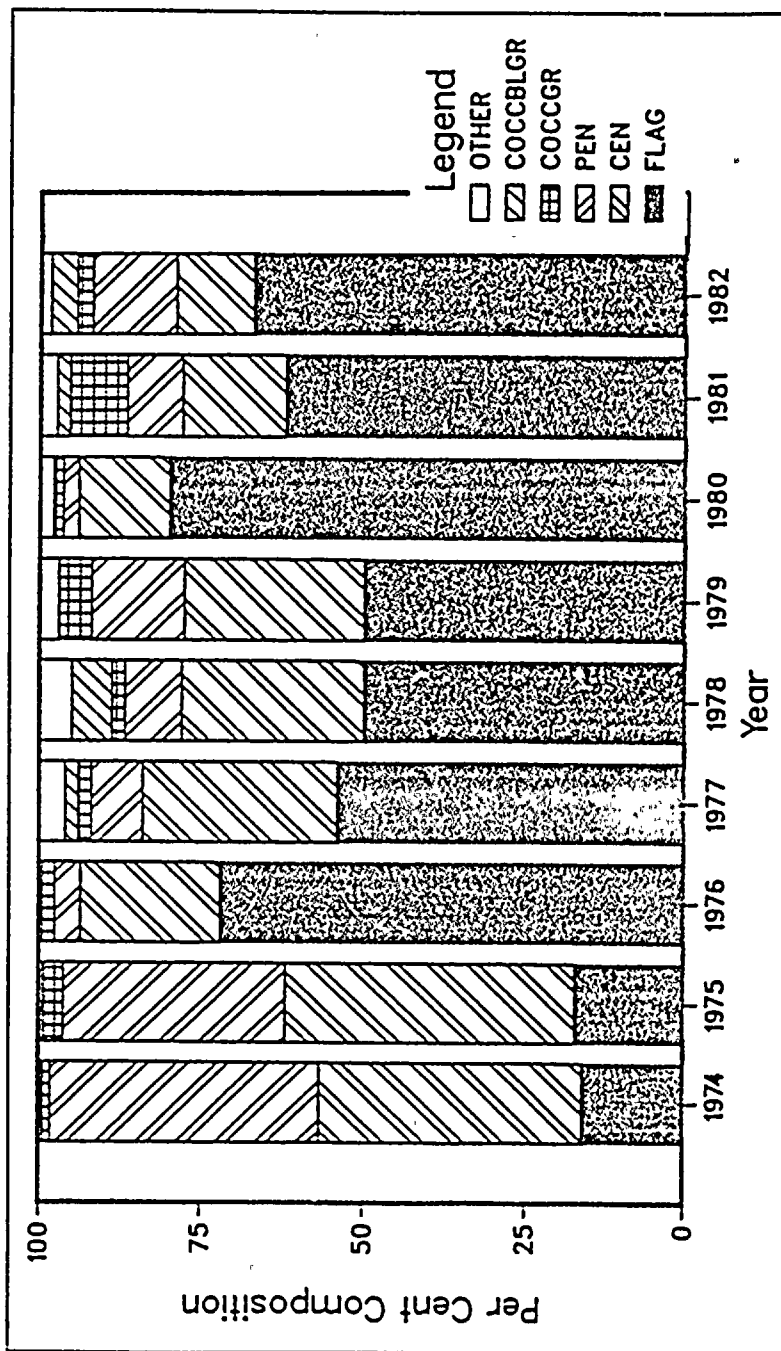


FIGURE 3. Phytoplankton group composition (%) during April 1974-1982 at the three offshore stations of the sampling grid.

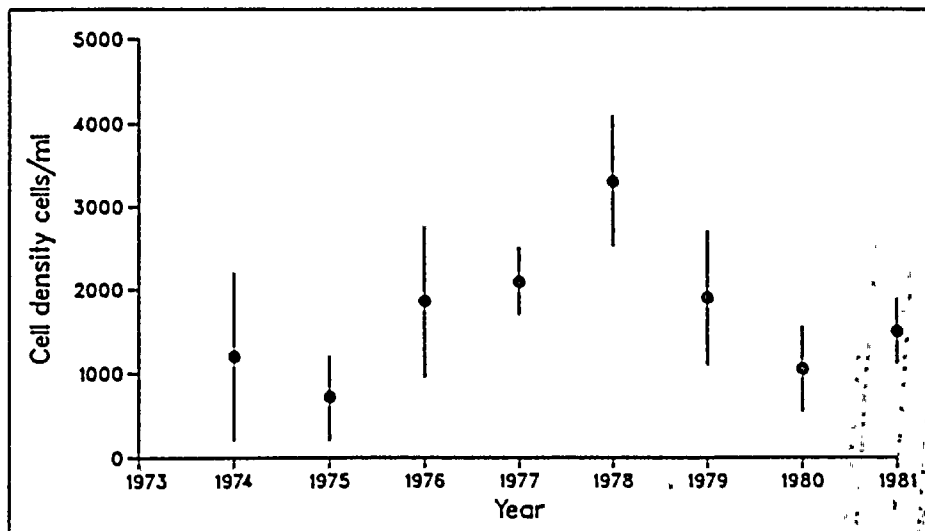


FIGURE 4. Total cell densities during July 1974-1982 at the three offshore stations of the sampling grid.

the 9-year period (Fig. 7). Coccoid blue-greens increased sometime after 1974. The flagellate group consistently comprised approximately 20% of the total counts. Again, there were no significant trends in nutrient concentrations with the exception of Si:SPP ratios which increased after 1978 (Table 1).

Seasonal Succession

Assessing long-term successional changes in southeast Lake Michigan required a conservative approach to the choice of sampling stations. Therefore, the mean of the algal densities from all the monthly stations was chosen to eliminate both station-to-station and inshore-offshore variances. Because the samples were collected from only a 1-m epilimnetic depth, deep-living phytoplankton were not represented during the thermal stratification. Successional patterns were analyzed as total counts and by taxonomic groupings. Nutrient data were also determined for the months of April, July, and October also at a 1-m depth.

Mean total cell counts at those stations sampled monthly indicated the often observed bimodal successional pattern for Lake Michigan (Fig. 8). During April, May, and sometimes through June, the spring bloom of mainly flagellates and diatoms occurs when stratification is developing. Following this isolation of the warmer mixed layer is a midsummer minimum beginning in June, usually reaching a maximum low in July, and continuing into August. As thermal stratification gradually breaks down because of autumn cooling, nutrient-rich deeper and cooler waters stimulate algal growth once again, resulting in a fall bloom during late September or October, and into October. Although no samples were collected from December

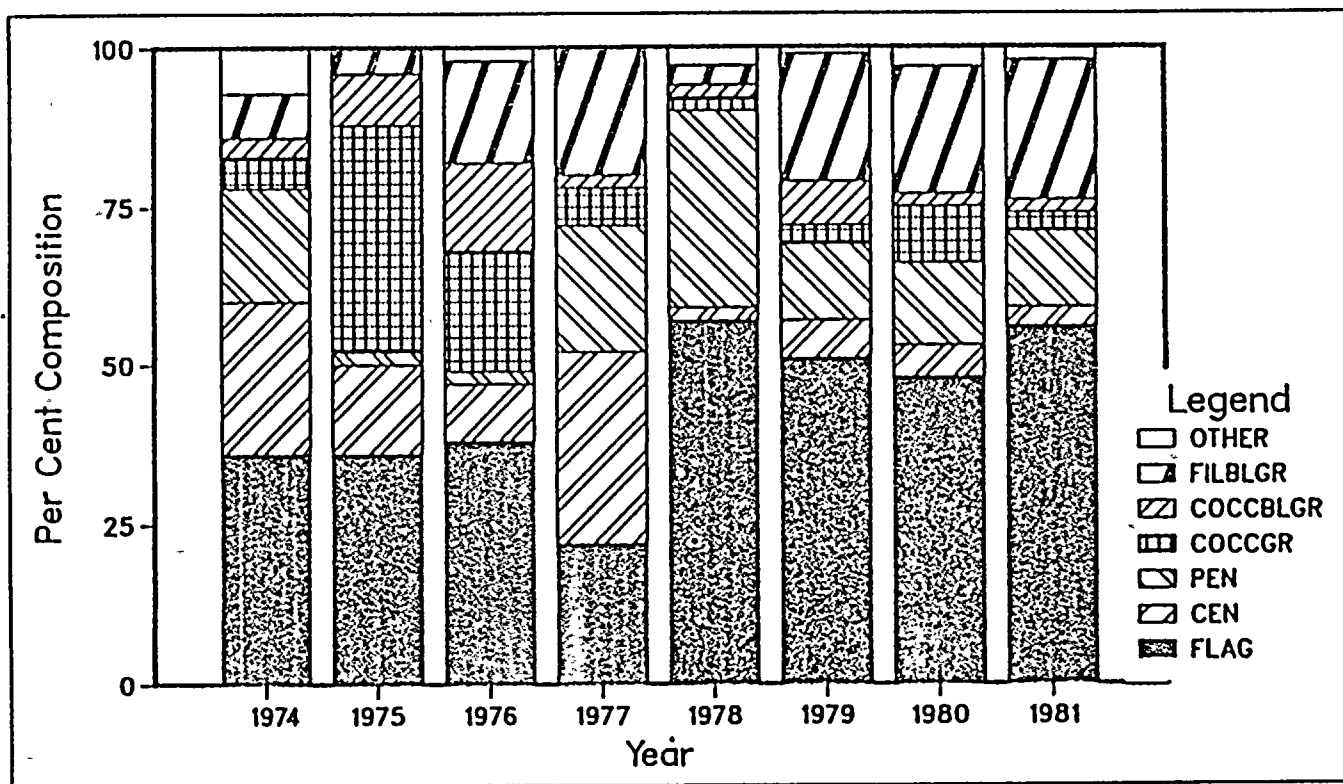


FIGURE 5. Phytoplankton group composition (%) during July 1974-1982 at the three offshore stations of the sampling grid.

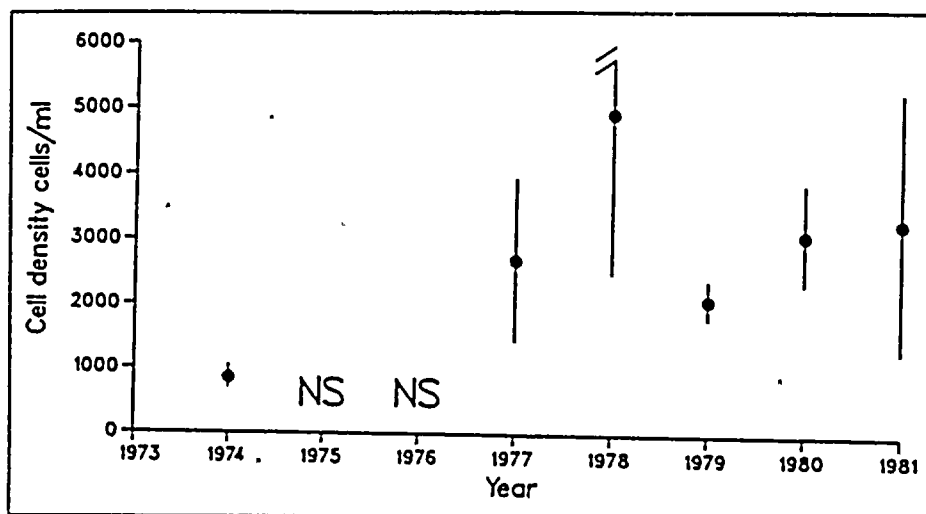


FIGURE 6. Total cell densities during October 1974-1982 at the three offshore stations of the sampling grid.

through March, a winter minimum follows this autumn bloom. There were no year-to-year trends in this pattern except those probably resulting from year-to-year differences in the weather-driven thermal structure of the lake (Fig. 8).

When phytoplankton succession is observed as individual taxonomic groupings, some groups follow the general bimodal pattern heavily contributing to the overall pattern, while others appeared as summer groupings.

Flagellates (Fig. 9), along with pennate diatoms, were the groups contributing the most to the bimodal pattern. From April through May, and often into June, flagellates formed an important component of the spring bloom. After a summer low in density during July and August, a smaller pulse during September, October, and November followed. The timing of this fall bloom is no doubt responsive to autumn cooling. There were no consistent changes to this pattern.

Equally abundant with the flagellates, pennate diatoms were also bimodal (Fig. 10). Unlike the flagellates, there occurred a time shift delay of 1 month in their seasonal succession. From 1974 to 1977, the spring pennate bloom occurred in April, while from 1978 to 1981, the bloom peaked in May. Likewise the fall bloom was delayed a month from October until November within the same years. Midsummer minimums still occurred consistently in August and September. The reasons for this time change are unknown.

Centric diatoms were less abundant than the pennates, but had a similar bimodal succession pattern. This pattern was, however, more variable (Fig. 11). During 1974, 1975, 1979, and 1982, the spring pulse peaked in April, while in the other years the bloom's mode was in May. October was the month most often associated with the fall bloom, although sometimes it occurred in September. Midsummer minimums from 1974 to 1979 were in August. In 1980 and 1981, they occurred in July.

The least abundant of all of the nine groups were the desmids (Fig. 12). Bimodally

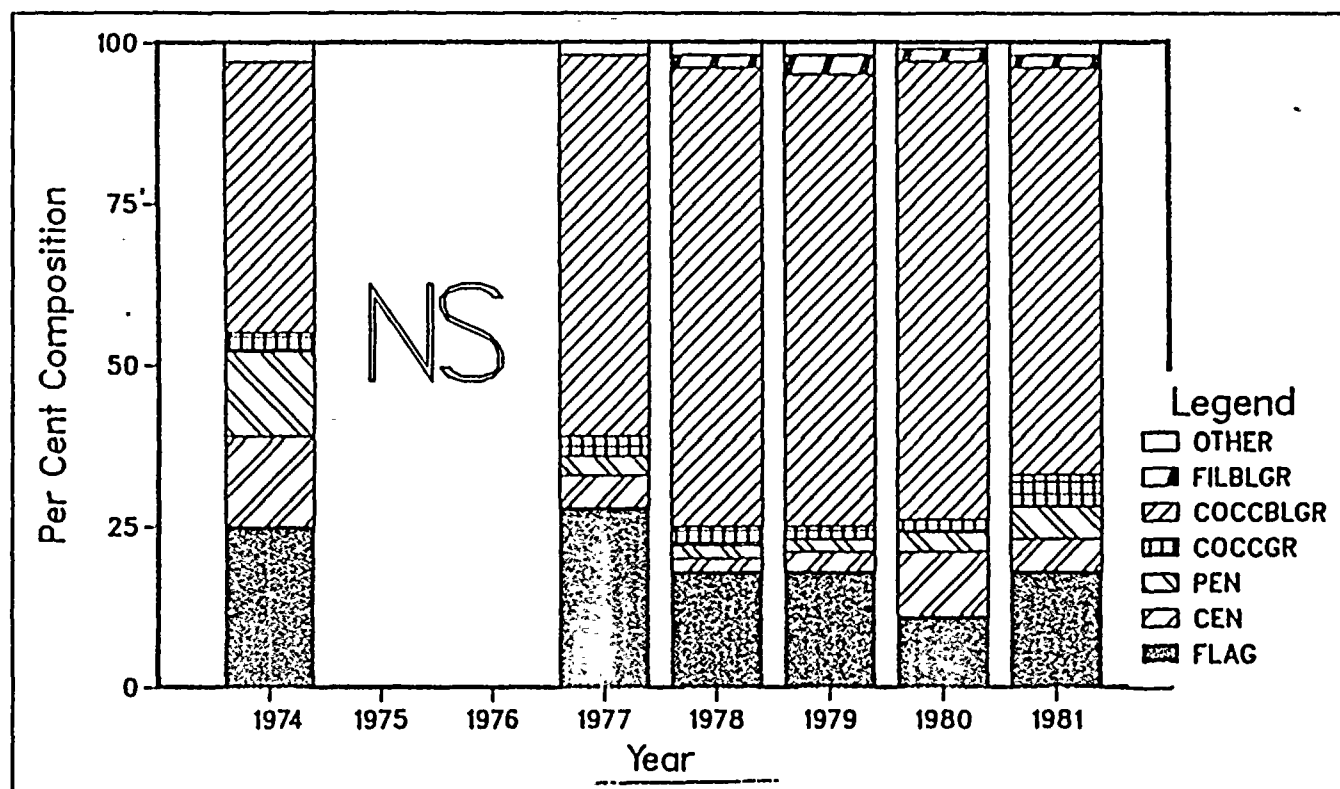


FIGURE 7. Phytoplankton group composition (%) during October 1974-1982 at the three offshore stations of the sampling grid.

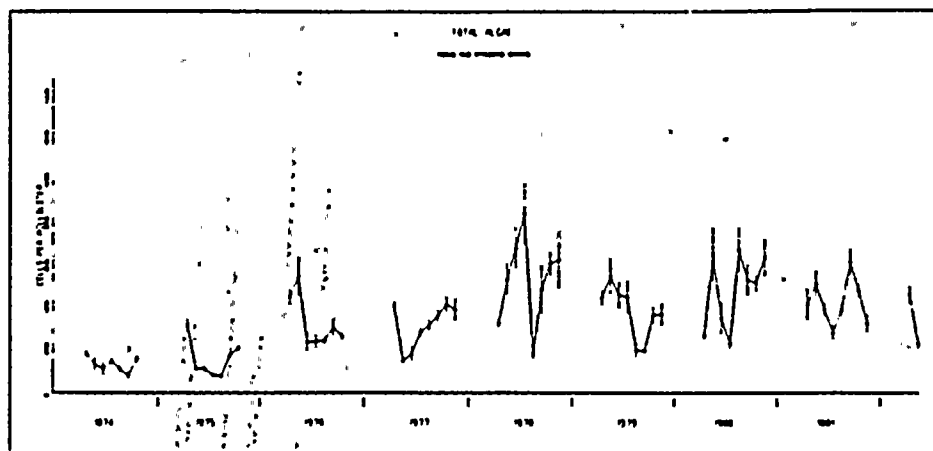


FIGURE 8. Mean monthly total densities of algae in the Donald C. Cook Nuclear Plant algal sampling grid 1974-1982.

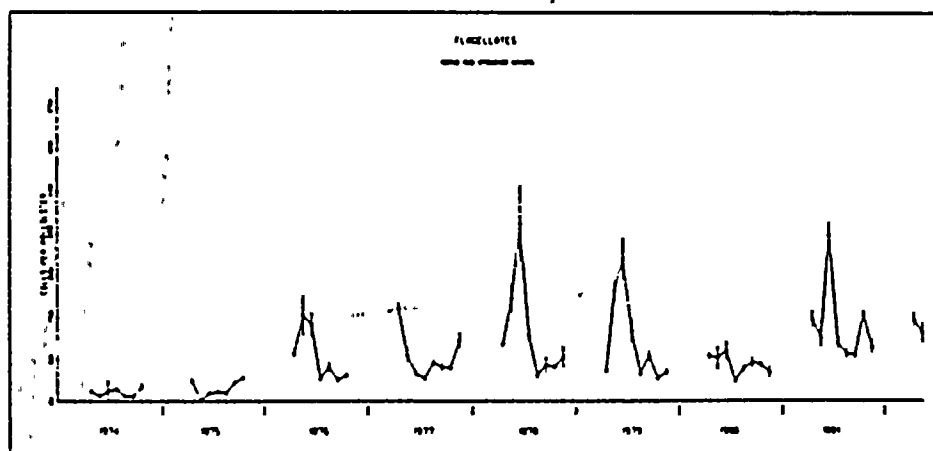


FIGURE 9. Mean monthly flagellate densities in the Donald C. Cook Nuclear Plant algal sampling grid 1974-1982.

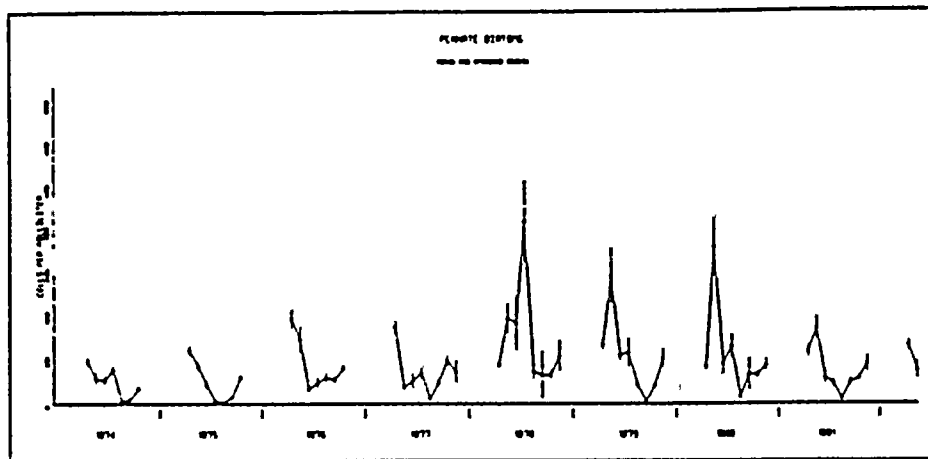


FIGURE 10. Mean monthly pennate diatom densities in the Donald C. Cook Nuclear Plant algal sampling grid 1974-1982.

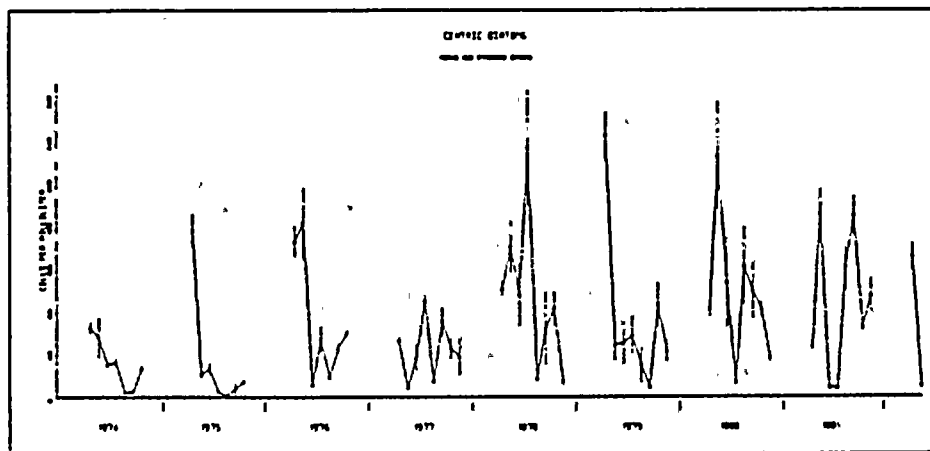


FIGURE 11. Mean monthly centric diatom densities in the Donald C. Cook Nuclear Plant algal sampling grid 1974-1982.

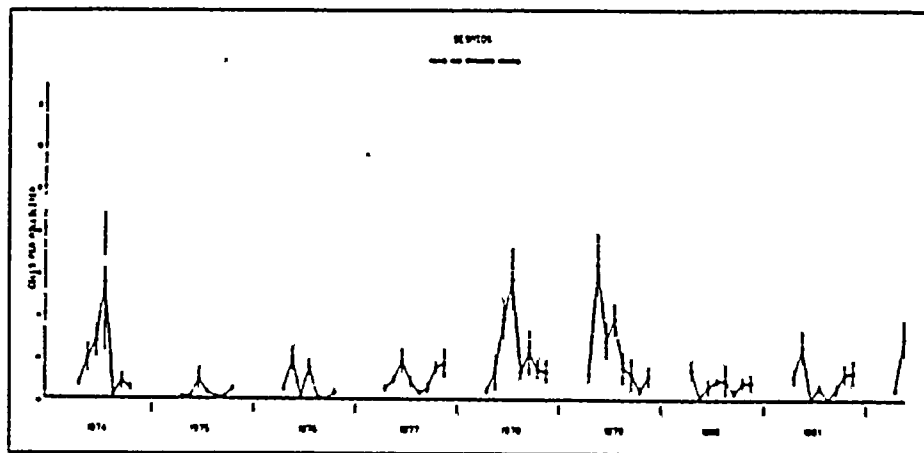


FIGURE 12. Mean monthly desmid densities in the Donald C. Cook Nuclear Plant algal sampling grid 1974-1982.

patterned, they had an early summer bloom during June and July followed by an autumn pulse in the late fall, October, and November.

All of the above groups give Lake Michigan its typical bimodal successional pattern, but other groups, filamentous blue-greens, coccoid blue-greens, and coccoid greens, are unimodal summer forms which coincide with the total cell count midsummer minimum. The filamentous blue-greens most often had their summer maximum in July (Fig. 13), while the coccoid blue-greens' peak was a distinct late summer group with maxima anytime from August, through September and October, and into the beginning of November (Fig. 14). Coccoid greens were most often a summer group with maxima from late July, August, and into early September (Fig. 15). However, in 1978, 1979, and 1981 they exhibited bimodal patterns. Not an abundant group, filamentous green algae were also unimodal with the maximum occurring during any month from May until October (Fig. 16).

The "other" group, composed of an assortment of several taxonomic genera, had an understandably variable pattern (Fig. 17). In 1974 and 1975, the pattern was unimodal having maxima in August and June. During 1976 and 1977, the spring bloom was less obvious, with dominant fall maxima in September or October. The pattern was bimodal with the spring bloom occurring from April until June and the autumn pulse following during some period from September through October for 1978-1981.

Single species or dominant groups varied throughout the sampling period at the three offshore sampling stations. Table 2 illustrates the dominant group or species for each month during the sampling period. During April, the flagellate group made up the largest proportion of the population. This was somewhat artificial because the flagellate group represents many species, especially the *Rhodomonas* and *Cryptomonas* genera. During July, flagellates again were dominant. *Anabaena flow-aquae* was the single most consistent species during July. This blue-green alga is indicative of eutrophic conditions and often nitrogen limitation. A mesotrophic

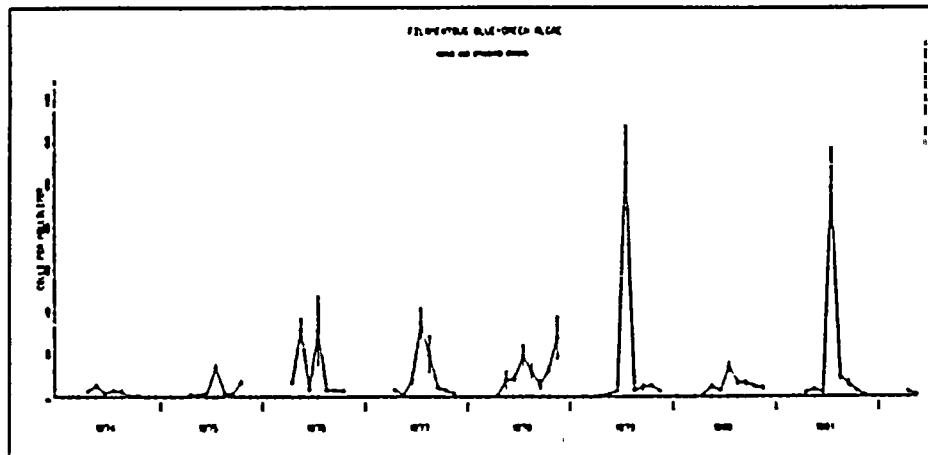


FIGURE 13. Mean monthly filamentous blue-green densities in the Donald C. Cook Nuclear Plant algal sampling grid 1974-1982.

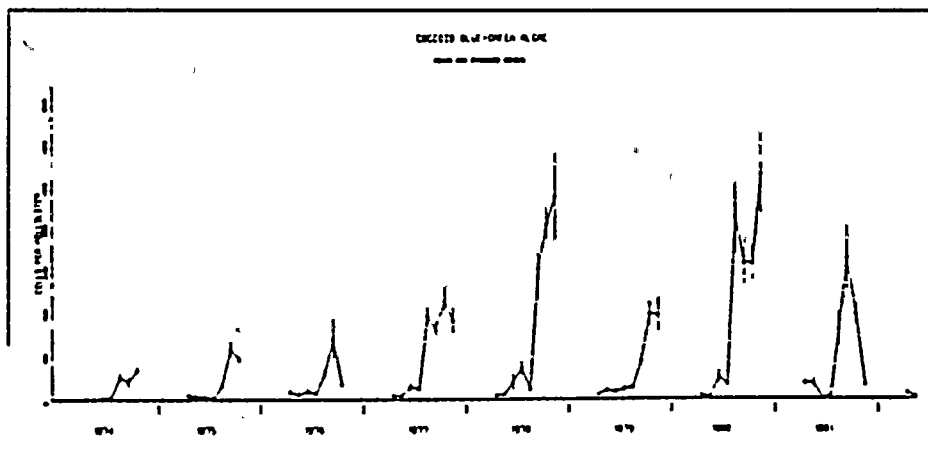


FIGURE 14. Mean monthly coccoid blue-green densities in the Donald C. Cook Nuclear Plant algal sampling grid 1974-1982.

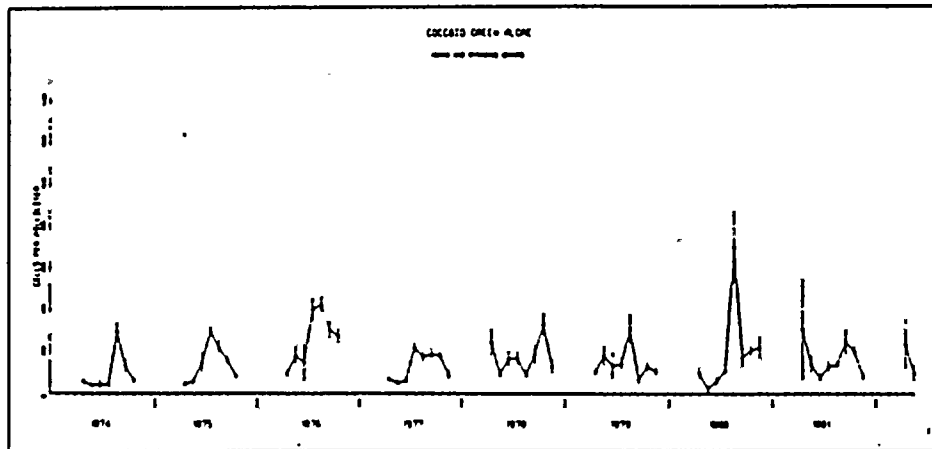


FIGURE 15. Mean monthly coccoid green densities in the Donald C. Cook Nuclear Plant algal sampling grid 1974-1982.

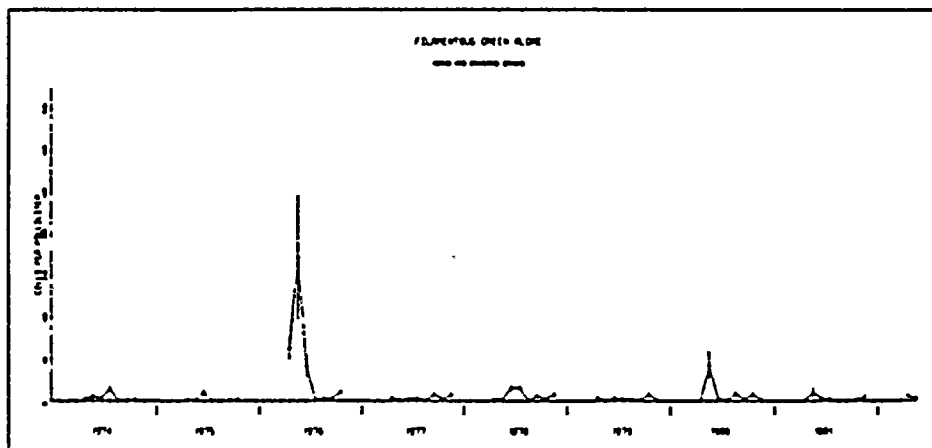


FIGURE 16. Mean monthly filamentous green densities in the Donald C. Cook Nuclear Plant algal sampling grid 1974-1982.

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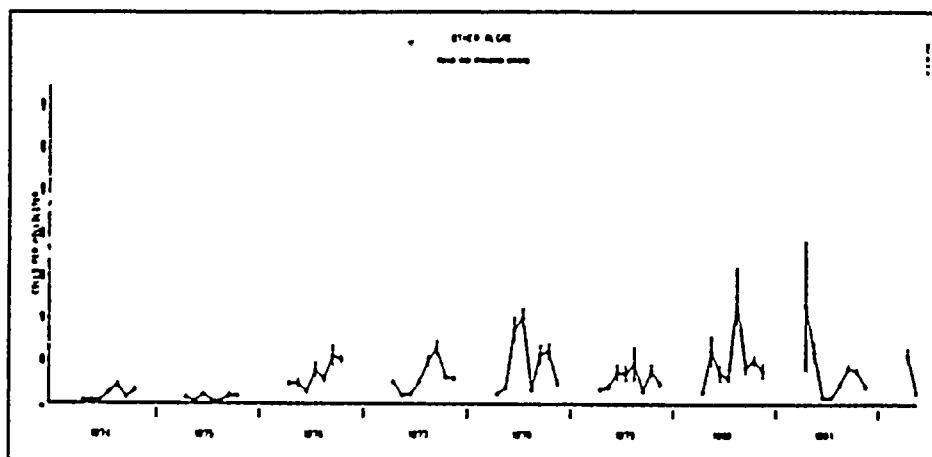


FIGURE 17. Mean monthly "other" densities in the Donald C. Cook Nuclear Plant algal sampling grid 1974-1982.

TABLE 2. Dominant phytoplankton groups and species for April, July, and October 1974-1982 in the three offshore Donald C. Cook Nuclear Plant sampling grids.

Year	Month	Groups and Species
1974	April	- <i>Synedra filiformis</i> , <i>Melosira italica</i> , <i>Fragilaria crotonensis</i> , Flagellates
	July	- <i>Fragilaria crotonensis</i> , <i>Thalassiosira pseudomona</i> , Flagellates
	October	- <i>Gomphosphaeria lacustris</i> , <i>Anacystis incerta</i> , Flagellates
1975	April	-Flagellates
	July	-Flagellates, <i>Cyclotella stelligera</i> , <i>Gloeocystis planktonica</i>
	October	- <i>Anacystis incerta</i> , <i>Gomphosphaeria lacustris</i>
1976	April	-Flagellates
	July	- <i>Gomphosphaeria lacustris</i> , <i>Anabaena flos-aquae</i> , Flagellates
	October	-none reported
1977	April	-Flagellates
	July	- <i>Anabaena flos-aquae</i> , <i>Fragilaria crotonensis</i> , <i>Cyclotella comensis</i> , Flagellates
	October	- <i>Anacystis incerta</i> , <i>Gomphosphaeria lacustris</i>
1978	April	-Flagellates
	July	- <i>Chryptophytic</i> spp., Flagellates, <i>Fragilaria crotonensis</i>
	October	- <i>Anacystis incerta</i>
1979	April	-Flagellates, <i>Ochromonas</i> spp.
	July	- <i>Fragilaria crotonensis</i> , <i>Anabaena flos-aquae</i> , Flagellates
	October	- <i>Anacystis incerta</i>
1980	April	-Flagellates, <i>Rhodomonas minuta</i>
	July	-Flagellates, <i>Anabaena flos-aquae</i>
	October	- <i>anacystis incerta</i> , <i>Gomphosphaeria lacustris</i>
1981	April	-Flagellates
	July	-Flagellates, <i>Anabaena flos-aquae</i>
	October	- <i>Gomphosphaeria lacustris</i> , <i>anacystis incerta</i> , <i>Rhodomonas minuta</i>
1982	April	-Flagellates

chainforming diatom, *Fragilaria crotonensis*, was the only other species to frequently be dominant. During fall overturn in October, *Gomphosphaeria lacustris* and *Anacystis incerta* were consistently dominant. However, neither species exhibited any tendency to change in abundance through the years.

HORIZONTAL PATCHINESS OF NEARSHORE PHYTOPLANKTON: THE EFFECT OF TEMPERATURE

Phytoplankton populations form patchy distribution patterns in pelagic waters on the horizontal axis. This horizontal patchiness is observed over a two-dimensional space-time continuum covering virtually all scales of time and distance. During April (1974-1982) and July (1974-1981), 36 to 38 stations were sampled for algal densities at a 1-m depth in the southeastern nearshore region of Lake Michigan adjacent to the Donald C. Cook Nuclear Plant where horizontal patterns were detected.

APRIL 1974-1982

April is characterized by the presence of springtime thermal bars ranging from 0.5 km to 4-5 km offshore in the 9-year period. To recognize algal distribution patterns, phytoplankton and nutrient data sets were assessed using three different techniques: multivariate linear correlation, principal components analysis, and cluster analysis. Relationships among various nutrients, nutrients and algal groups, and various algal groups were sought that would link algal distribution patterns to temperature in the sampling grid. Although every relationship was examined, no consistent pattern emerged. The reason for this lack of correlation patterns is the scale of the sampling grid and mesoscale patchiness of the phytoplankton. At a scale of hundreds of meters between sampling stations, algal densities for any algal group or even single species could change by an order of magnitude. Multivariate methods do not filter out this fine scale patchiness allowing correlations across the whole sampling grid. Thus in order to detect an April thermal bar event statistically, the data were grouped on a coarser scale.

Thermal bars create a gradient of temperature, with warmer temperatures in an inner region encompassed by the thermal bar itself at approximately 4°C followed by a colder outer region. April data were grouped into three zones according to temperature. An inshore zone, behind the bar, is defined by stations having a temperature >4.5°C. A middle or thermal bar zone is that region 3.5-4.5°C. The outer offshore zone is <3.5°C. This protocol was possible only during the years 1975, 1978, 1979, 1980, and 1982. In April 1974, 1976, 1977, and 1981, there were not enough stations (1-2) in the 3.5-4.5°C temperature range. All of the offshore stations were <4.0°C. In these cases, the inner and middle zones were separated at a temperature halfway between the inner and outer station temperatures. This method proved to be simple and effective in demonstrating how temperature alone is the single most important determinant of algal densities along this temperature gradient.

Table 3 lists temperature and phytoplankton group means. During April 1974, the thermal bar was located 5 km from the shoreline (Table 3). All algal groups decreased in density from the inner to outer zones. Not surprisingly, the largest

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TABLE 3. Mean values of water temperature (C) and phytoplankton groups (cells·mL⁻¹) during April 1974-1982 at three zones defined by temperature. I = inner zone, M = middle zone, O = outer zone. T = temperature, TOT = total, CEN = centrics, PEN = pennates, FLG = flagellates, FIL/BLGR = filamentous blue-greens, COCC/BLGR = coccoid blue-greens, COCC/GR = coccoid greens.

Year	T	TOT	CEN	PEN	FLG	FIL BLGR	COCC BLGR	COCC GR
1974								
I	7.6	2,106.1	735.3	915.8	287.6	23.8	*	71.6
M	6.1	1,575.7	573.5	662.4	277.5	12.0	*	25.7
O	3.5	723.5	291.4	295.3	120.0	5.3	*	4.8
1975								
I	5.6	3,540.1	1,839.6	1,015.1	454.5	8.9	159.1	34.1
M	3.8	3,379.2	1,582.8	1,135.4	547.0	10.7	21.1	66.5
O	3.1	2,325.4	1,130.6	709.7	360.3	6.3	48.4	21.4
1976								
I	8.5	5,087.4	1,719.4	1,828.7	1,045.4	76.3	126.4	*
M	6.0	2,573.4	691.4	760.4	805.7	34.7	58.5	*
O	2.9	2,426.0	683.7	648.0	891.7	29.3	5.7	*
1977								
I	9.0	4,520.9	533.8	1,734.8	1,926.9	36.3	79.9	104.9
M	7.1	4,346.0	442.7	1,514.4	2,078.4	34.3	92.4	49.6
O	2.6	1,714.0	461.8	216.5	903.8	7.3	45.6	27.5
1978								
I	6.1	3,549.6	1,079.8	862.8	1,296.5	*	91.9	155.3
M	4.2	3,718.6	1,193.9	810.7	1,075.6	*	34.9	556.9
O	2.9	2,300.0	652.5	360.8	1,102.7	*	68.1	70.2
1979								
I	5.5	7,676.2	2,602.3	974.7	538.8	*	84.5	77.0
M	3.8	5,414.8	3,075.0	1,352.2	697.1	*	83.8	141.7
O	2.3	3,002.5	1,211.8	802.3	808.4	*	80.3	43.7
1980								
I	7.5	3,443.6	1,029.5	1,041.3	1,062.7	9.3	43.4	*
M	3.8	2,470.9	927.8	529.5	819.8	9.5	121.1	*
O	2.4	1,776.7	435.8	240.0	965.3	7.0	79.3	*
1981								
I	8.5	5,112.8	553.8	1,234.1	1,805.8	20.7	415.9	431.3
M	5.7	2,450.0	295.9	473.6	1,508.3	8.0	75.0	25.7
O	3.4	1,943.5	291.8	215.2	1,195.0	6.5	25.8	124.0
1982								
I	6.2	5,905.1	1,876.6	1,392.4	1,798.9	25.2	41.6	416.8
M	4.2	4,142.7	988.7	1,070.6	1,575.9	31.6	242.4	81.7
O	2.2	2,169.3	316.1	341.7	1,321.1	5.0	101.7	41.9

* - Group not present in sample.

declines were observed in the transition from the middle to outer zone. Centric and pennate diatoms which comprised the largest proportion of the total decreased by a factor of two.

With cooler temperatures prevailing, the thermal bar in 1975 was much closer to the shoreline, 0.3 km, than in 1974. Only an average of 2.5°C separated the middle and outer zones. Consequently, variation was greater than in 1974. While total algae and centric diatom densities declined monotonically, pennate diatoms, flagellates, filamentous blue-greens, and coccoid greens were most dense in the middle region.

Farther offshore in 1976, the thermal bar region was nearly 1.5 km offshore. Total algae and the diatom groups decreased most significantly in the middle zone. Flagellate densities were higher in the outer zone compared with the middle region. Coccoid blue-green algae were almost undetectable in the outer offshore region.

With the thermal bar located approximately 2.5 km offshore, a large difference in temperature existed between the inner and outer zones during April 1977. The greatest difference in temperature was between the middle and outer zones, which was reflected in the proportional declines of total, pennate, flagellate, and filamentous blue-green groups. Coccoid blue-greens were the least responsive group.

With the bar residing 1.5–2.0 km offshore in 1978, diatom declines between the middle and outer zones were most dramatic. Coccoid blue-greens were lowest in density in the middle zone, while the coccoid greens decreased dramatically between the middle and outer regions.

In April 1979, the bar was positioned very near the shoreline (<0.5 km) with just six stations having temperatures $>4.5^{\circ}\text{C}$. The total temperature gradient averaged 3.2°C . Similar to 1975, total density changes in this small cline were proportionally small. Group means were not consistent. Coccoid blue-greens did not change significantly. Coccoid greens decreased appreciably in the offshore regions. Flagellates and diatoms were more dense in the middle zone than in the inner zone.

In 1980, the thermal bar was located approximately 0.5–1.0 km from the shoreline. Centric and pennate diatoms decreased substantially in the transition between the middle and outer zones. Although flagellate densities were greatest in the inner zone, densities in the outer zone exceeded those in the middle region. Having very large standard deviations, blue-green group means were low.

Located 4.0 km offshore, the thermal bar regime had a mean temperature gradient spanning 5.1°C in 1981. Offshore of the inner zone, coccoid blue-greens and coccoid green species decreased dramatically. While flagellates declined only moderately, diatoms also responded most significantly to the transition into the middle zone.

The bar was positioned very close (<1.0 km) to the shoreline in 1982, with a concurrent smaller temperature differential. Coccoid blue-green algae were most dense in the middle zone; whereas, coccoid green species declined in the thermal bar. Flagellate species decreased by only a small proportion from the inner to outer region. Diatom abundances were much lower in the outer region.

The consistent direct effect of temperature was most easily observed when the total and group algal densities were ranked within the three temperature zones (Table 4). The method gave a rank score of 1 if that zone had the highest density of the three zones. When a zone had the second highest cell density for a particular algal group, that zone was given a score of 2. The lowest density of the three zones was given a score of 3. When averaged from 1974 to 1982, trends are apparent. Mean rankings for total cell densities are very near the theoretical rankings for the continual decline of abundance going from the region behind the thermal bar,

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TABLE 4. Mean zone rankings, 1974-1982 (+SD), of each phytoplankton group. A value of 1 indicates that group had the highest cell densities in that zone, while a value of 3 indicates that zone always had the lowest cell density of the three zones. N = sample size.

	Inner (N = 9)	Middle (N = 9)	Outer (N = 9)
Total	1.11 (0.33)	1.89 (0.33)	3.00 (0.00)
Pennates	1.22 (0.44)	1.78 (0.44)	3.00 (0.00)
Centrics	1.22 (0.44)	1.89 (0.60)	2.89 (0.33)
Filamentous Blue-Greens	1.43 (0.53)	1.57 (0.53)	3.00 (0.00)
Coccoloid Greens	1.43 (0.53)	1.71 (0.76)	2.86 (0.38)
Coccoloid Blue-Greens	1.63 (0.92)	1.88 (0.83)	2.50 (0.53)
Flagellates	1.44 (0.73)	2.11 (0.78)	2.44 (0.73)

through it, and offshore from it. Note the outer zone had the perfect ranking of 3 and the lowest mean score (1.11) for the inner zone. Diatoms were the most temperature-sensitive groups in this ranking hierarchy. Pennate species had a score of 3 in the outer zone. Both pennates and centric forms had the lowest mean rankings in the inner zone (1.22). The filamentous blue-green and coccoloid green groups had larger deviations from perfect rankings in the inner and middle zones but retained high mean rankings for the coldest outer zone similar to those of the diatom groups. Coccoloid blue-greens and the flagellate species had the largest deviations from perfect rankings which indicated they were the least sensitive to the bar-induced temperature clines.

Thermal bars and their resultant temperature gradients regulated the April horizontal distributions of phytoplankton on a spatial scale of thousands of meters. On a scale of hundreds of meters, algal patchiness existed to a high degree but could not be correlated with physical or chemical conditions in the habitat. No trends were detectable on this scale.

JULY 1974, 1978

For the Lake Michigan epilimnion, July is a period when diatoms and flagellates dominate the phytoplankton community, silica and phosphorus limit algal productivity, and a deep chlorophyll layer composed of the genera *Fragilaria*, *Melosira*, and *Tabellaria* exists (Brooks and Torke 1978). Physically, the epilimnion is a warm turbulently mixed layer (17-25°C) that is isolated from the colder, deeper strata by the thermocline. Horizontal phytoplankton patchiness is random and small scale, reflective sometimes of surface currents. Multivariate analysis (multiple correlation, principal components, and cluster analysis) of the July data sets from 1974-1981 offered no consistent patterns because of the small scale randomness in the mixed layer for the years 1975-1979 and 1978-1981.

However, this July habitat is dramatically restructured in Lake Michigan by stochastic wind stresses from storms over the lake. In the far offshore waters, this frictional drag creates sheer forces in the thermocline which result in sudden and localized destructions of the thermocline (Poincare' waves) that generate areal "windows" of turbulent mixing of epilimnetic and hypolimnetic waters. These areas then

bounce on a vertical axis from the effects of multinodal internal waves. In nearshore regions of the lake, this episodic wind stress from summer storms is observed as large-scale up(down)welling events. North-northwest winds along the north-south axis of the lake with concurrent Coriolis Effect create westerly surface currents along the eastern Lake Michigan shoreline. Warm surface water is blown offshore and cold hypolimnetic water is brought to the surface in an elongated pattern that is shore-parallel. During July 1974 and 1978, upwelling events were observed in the intensive survey grid tangent to the Donald C. Cook Nuclear Plant.

During the July 1974 period, the upwelling region was confined to an area 2 km south to approximately 4 km north of the plant and extended almost 1 km offshore. It included 14 sampling stations (Fig. 18). Mean surface temperature was 12.7°C, indicating that the upwelled water was well mixed with surface water. The absence of 4°C water indicated that there was no contact of hypolimnetic water with the surface. Yet this region was substantially cooler than the 18°C offshore epilimnion. Based on temperature, the sampling grid was again segregated like that of the thermal bar regions in the April surveys.

Significant shifts in the phytoplankton community occurred during the July 1974 upwelling (Table 5). Total densities of algae increased by approximately 70% in the

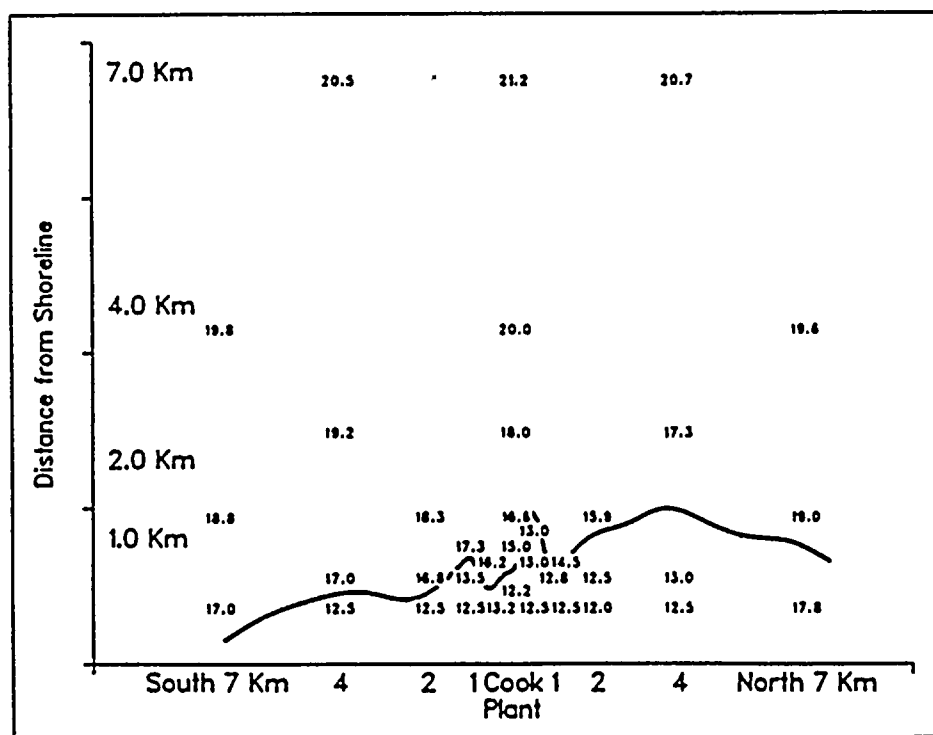


FIGURE 18. Temperature (C) in the Donald C. Cook Nuclear Plant sampling grid during the April 1974 upwelling event.

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TABLE 5. Mean values (\pm SD) for temperature ($^{\circ}$ C) and algal group cell densities (cells mL^{-1}) in the inshore upwelling and offshore waters of southeastern Lake Michigan, July 1974. N = sample size.

Parameter	Upwelled Region ($N = 14$)	Offshore Region ($N = 22$)
Temperature	12.7 (.4)	18.0 (1.9)
Total	2,078.9 (769.6)	1,224.5 (726.4)
Centric Diatoms	495.9 (214.3)	247.8 (231.6)
Pennate Diatoms	1,021.9 (386.8)	430.8 (350.4)
Flagellates	16.8 (18.2)	280.8 (146.5)
Filamentous Blue-Greens	13.4 (17.7)	41.0 (78.9)
Cocoid Blue-Greens	25.1 (38.6)	66.1 (90.0)
Filamentous Greens	16.9 (18.2)	29.2 (34.8)
Cocoid Greens	22.8 (23.8)	59.4 (58.6)

upwelled zone. This increase was due to 100% increases in both diatom groups. Flagellates, however, decreased by an order of magnitude. In the inshore zone, blue-green and green species declined to one-half of their offshore densities.

The large increases in diatom densities were due to the influx of hypolimnetic water rich in dissolved silica. Although epilimnetic waters had only $0.1 \text{ mg}\cdot\text{L}^{-1}$ dissolved $\text{Si}\cdot\text{SiO}_2$, hypolimnetic water had $1.4 \text{ mg}\cdot\text{L}^{-1}$ dissolved $\text{Si}\cdot\text{SiO}_2$ (Rossmann 1986).

The upwelling event of 1978 was different in form from that of 1974 (Fig. 19). Although the upwelled region was restricted to being close to the shoreline from 7 km south of the plant to 7 km north of the plant, it extended approximately 2 km offshore, making the total upwelling area much larger than in 1974. Interestingly, a small region of warmer water was trapped behind the cooler water and was directly tangent to the shoreline. This water was probably comprised of heated discharge water from the power plant and some of the remnant epilimnetic water normally associated with the beach area. Because of this horizontal distribution pattern, the sampling grid was divided into three distinct zones: the warm water behind the upwelled zone; the upwelled zone itself; and, beginning 2 km offshore, the July epilimnetic waters of the lake (Table 6). Temperatures in the inshore zone averaged 15°C . The upwelled region had a mean temperature of 9.7°C , which was significantly cooler than the 1974 upwelled surface temperature. These two zones were contained against the shoreline by an epilimnetic background temperature of 17°C .

Shifts in the relative abundance of the phytoplankton groups from the offshore community to the upwelled community were somewhat similar to the 1974 upwelling event. Total algal densities almost doubled in the upwelled region. Again, both diatom groups were most responsible for this increase. Flagellates also decreased as in 1974, but to a lesser degree. While both green phytoplankton groups did not change significantly, filamentous blue-greens in the upwelled zone had densities half those of offshore waters. In contrast, cocoid blue-greens increased in the upwelled zone more than an order of magnitude over offshore densities.

The transition from the upwelled zone to the six stations adjacent to the shoreline resulted in striking changes in the phytoplankton community structure. Total algal densities increased threefold, having some of the highest cell densities observed during the entire study. Most of this increase was in the diatom groups, although

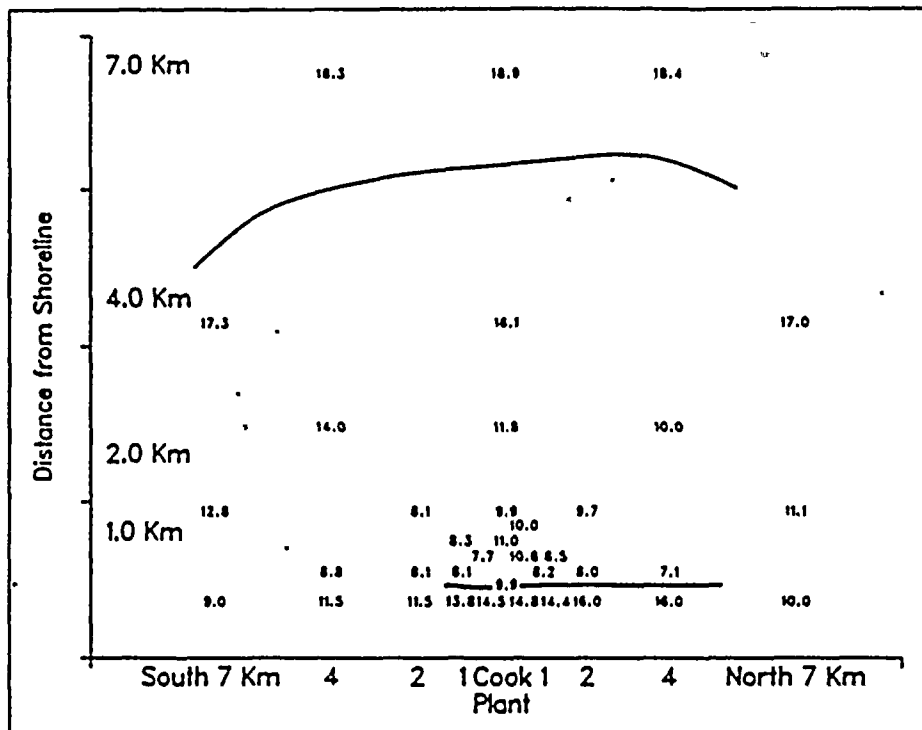


FIGURE 19. Temperature (C) in the Donald C. Cook Nuclear Plant sampling grid during the July 1978 upwelling event.

TABLE 6. Mean Values (+ SD) for temperature (C) and phytoplankton group cell concentrations (cells mL⁻¹) in the inshore region behind the upwelling zone, the upwelling region, and the offshore epilimnion of southeastern Lake Michigan, July 1978. N = sample size.

Parameter	Inshore Zone (N = 6)	Upwelled Zone (N = 25)	Offshore Zone (N = 7)
Temperature	15.0 (0.9)	9.7 (1.6)	17.1 (1.7)
Total	23,217.2 (5952.6)	6,015.2 (7181.5)	3,401.6 (791.2)
Centric Diatoms	8,582.1 (2418.3)	1,257.8 (2158.9)	270.6 (250.8)
Pennate Diatoms	10,893.2 (2684.8)	1,449.8 (2142.3)	906.9 (441.6)
Flagellates	738.2 (522.7)	1,415.1 (689.3)	1,612.3 (521.0)
Filamentous Blue-Greens	249.3 (447.5)	133.1 (157.1)	309.7 (632.7)
Cocoid Blue-Greens	1,747.7 (1270.9)	493.7 (552.7)	19.9 (49.9)
Filamentous Greens	26.7 (13.5)	25.6 (25.4)	30.7 (16.4)
Cocoid Greens	327.3 (179.1)	130.1 (190.4)	112.1 (131.1)

coccoid blue-green forms tripled in density. Coccoid greens contributed to the bloom by doubling in density, while flagellates declined. Filamentous greens did not significantly change in this warm and shallow region.

Persistent upwelling events have been studied for half a century in marine communities, because of their importance in neritic fishery production. In freshwater lakes, persistent upwelling phenomena occur only in those basins large enough to have sustained current patterns that generate them (Mortimer 1971). Yet, there is a commonality of the phytoplankton response to both freshwater and oceanic upwelling events. Diatoms are responsible for almost all of the phytoplankton density increases observed in the upwelled region. When swept upward into strata having higher light intensities and thermal radiation, algae residing in nutrient-rich cold water are suddenly stimulated by these ideal conditions for growth. This scenario seemed most evident in the six stations along the shoreline during 1978. Stoermer et al. (1971) observed an almost identical response in this same region of Lake Michigan. The cooler hypolimnetic water was high in phosphate and silica which are normally depleted in the mixed layer waters of the lake during July. Add the light for photosynthesis and the heat for temperature-dependent cell growth, and expected increases in density occur.

Since the Stoermer et al. (1971) study, there has been another discovery which could further enhance diatom increases in the upwelling zone. A characteristic feature of the vertical distribution of phytoplankton in Lake Michigan during July is a deep chlorophyll layer (Brooks and Torke 1978). Residing principally at the metalimnetic-hypolimnetic interface, this layer is composed of chain-forming diatoms that settle out of the mixed layer and, when encountering the more dense and much less turbulent interface, accumulate at this depth. During an upwelling event, this stratum is part of the water drawn upward, carrying with it high densities of diatom chains which add to *in situ* growth.

SUMMARY

Long-term changes in the phytoplankton community in southeastern Lake Michigan in the vicinity of the Donald C. Cook Nuclear Plant were assessed at the three farthest offshore sampling stations located 11.2 km from the shoreline during April, July, and October 1974-1982. Most phytoplankton groups, though variable, did not have any noticeable trends during April. However, flagellates and both diatom groups significantly decreased ($P < .05$) during 1974 and 1982. Nutrient concentrations also did not have any significant trends during the study period. Neither total cell densities nor nutrient concentrations had any trends during the July cruises, though the flagellate group increased after 1977 and remained at that proportion through 1981. Total cell densities during October were more variable than April or July, with coccoid blue-green algae increasing after 1975 ($P < .05$). Again, nutrients did not change significantly during the October samplings.

Seasonal succession patterns during the 1974-1982 period followed the typical Lake Michigan bimodal pattern. April, May, and June most often encompassed the spring diatom bloom, which led to a variable midsummer minimum during June, July, August, and part of September. A fall bloom began during September and continued through November. Flagellates and the diatoms followed this pattern closely, while the blue-green and green groups were most abundant during summer.

Anabaena flos-aquae, an eutrophic blue-green species, was particularly dominant during summer.

Spatial distributions of algal populations in the nearshore region were patchy during most of July and all of October. Temperature-controlled distributions during all of the April cruises and two of the July cruises were due to the temperature structures of thermal bars and upwelling episodes, respectively. Total cell densities were directly proportional to temperature in the thermal bars. Diatoms were most sensitive to temperature, while blue-greens and greens were less sensitive. Flagellates were the least sensitive to temperature. The 1974 and 1978 July sampling coincided with upwelling events along the shoreline adjacent to the power plant. Lowered temperatures in the upwelling zone were accompanied by increases in diatom concentrations. This probably resulted from the advection of incoming net phytoplankton from the deep chlorophyll layer and infusions of silica-rich hypolimnetic water.

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IMPACT OF THE DONALD C. COOK NUCLEAR PLANT ON ZOOPLANKTON IN SOUTHEASTERN LAKE MICHIGAN

Marlene S. Evans and Glenn J. Warren

INTRODUCTION

In the late 1960s and early 1970s there was concern that increased power plant construction on Lake Michigan shores, with the ensuing substantial usage of lake water for condenser cooling, would have detrimental effects on lake biota, including zooplankton. Monitoring programs were designed to determine these effects, with the size of the program proportional to power plant size. The Donald C. Cook Nuclear Plant, one of the largest plants on Lake Michigan, had a particularly large monitoring program. The lake sampling program, which included a large survey grid and several years of preoperational (1970-1974) and operational (1975-1982) monitoring, was designed to investigate the effects of power plant operation on zooplankton community structure over a 132 km² area of the lake. (Fig. 1).

Freshwater zooplankton are small animals, usually less than 2 mm in length. They are a major constituent of the aquatic food web, feeding as herbivores, omnivores, or carnivores, and are themselves food for fish and other vertebrate and invertebrate planktivores. Our sampling methods (156- μ m-mesh nets) result in zooplankton collections dominated by crustaceans, mainly copepods and cladocerans, which generally range from 0.2 to 2.0 mm in length. Rotifers, the third major component of Great Lakes zooplankton, are smaller than most crustacean zooplankton. Consequently, they are most efficiently collected by nets of 76- μ m mesh and smaller.

Life history strategies vary among the crustacean zooplankton. Except in rare instances, Cladocera reproduce parthenogenetically. The all-female populations are ovoviviparous. Eggs are retained in the brood pouch where they develop into the first immature stage. Immature Cladocera, which are morphologically similar to adults, develop through two or more instars before the adult stage is reached. Generation times are several days to weeks. Copepods reproduce sexually. After hatching from the egg, an individual develops through six nauplius stages and six copepodite stages, with the sixth being the adult. Generation times range from several days for some species, to a year or more for others (Wetzel 1975). In Lake Michigan, most copepods species produce one to three generations per year (Torke 1975).

MATERIALS AND METHODS

LAKE SURVEY SAMPLING

The lake survey program was designed to provide information on potential spatial and temporal alterations in zooplankton community structure as a result of plant

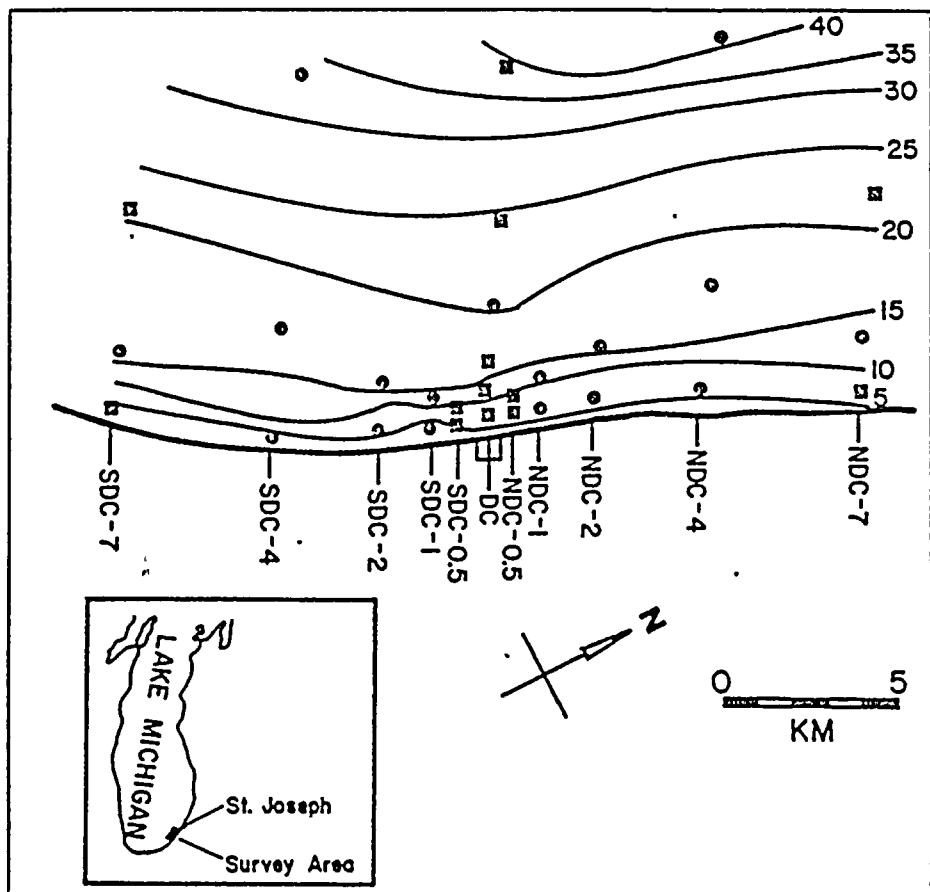


FIGURE 1. Station locations for the major surveys. Squares indicate the subset of short survey stations. Depth contours are in meters.

operation. Zooplankton preoperational sampling cruises began in April 1969 and terminated in October 1974. Operational cruises began in April 1975, the first cruise after Unit 1 went into operation in February 1975. Unit 2 went on line 3 years later in April 1978. The survey cruise program was terminated in May 1982.

The survey grid (Fig. 1) extended 11 km north and south of the plant site and 11 km offshore. Survey cruises were of two types. Major surveys (generally consisting of 30 stations) were conducted in April, July, and October of each year. Short surveys were initiated in 1972 and generally consisted of 14 stations. Short surveys were initiated to provide supplemental information on zooplankton population dynamics. Station depths ranged from 4 m to over 40 m and increased with distance from shore (Fig. 1). The three intake pipes utilized by the power plant are located

approximately 690 m offshore (between the first and second stations along the DC-transect) in 7.3 m of water at the apexes of a 75-m equilateral triangle. Two discharge pipes return heated water to the lake, with the northern pipe servicing Unit 1 and the southern pipe Unit 2. The two pipes are located approximately 380 m offshore (the most inshore station of the DC transect) in 5.5 m of water and are 100 m apart.

ZOOPLANKTON SAMPLING METHODS

Zooplankton were collected at each station with a 50-cm diameter, No. 10, 156- μ -mesh net equipped with a calibrated flowmeter (Evans and Sell 1985). Three replicate hauls generally were made at each station.

At each station, the net was hauled from as close to the bottom as possible (about 1 m) to the surface. The flowmeter was read, the outside of the net washed down, and the contents of the plankton bucket transferred to a labelled jar and preserved with a sugar-formaldehyde solution (Haney and Hall 1970). Additional information on the history of the lake sampling program is provided in Evans et al. (1986B).

COUNTING TECHNIQUES

In the laboratory, zooplankton in the first two replicate samples from each station generally were examined. Each sample was subdivided as many times as necessary to give two subsamples of 300 to 600 organisms each. Sources of variance associated with subsampling and replicate sampling are discussed in Sell and Evans (1982).

Zooplankton were enumerated in a circular counting dish using a microscope at a magnification of 20 to 140X. A compound microscope was used to verify certain species identifications and to identify species whose occurrences in the survey area had not been noted previously. In these instances, past collections were re-examined for the presence of these species.

TAXONOMIC IDENTIFICATIONS

The level to which zooplankton were identified varied with the taxon, the year, and the station (Evans et al. 1986B). Methods used for identifying zooplankton varied over the course of the study. Beginning in 1975, adult copepods and cladocerans were identified to genus at all stations and to species at most. Immature copepodites were identified to genus while nauplii were combined as a group. *Asplanchna* was the only rotifer genus routinely enumerated. Taxonomic keys included Pennak (1963, 1968), Deevey and Deevey (1971), Brooks (1957), and Brooks, Wilson, Tressler, and Yeatman in Edmondson (1959). Additional information on laboratory procedures is provided in Evans et al. (1980, 1986B).

DRY WEIGHT DETERMINATIONS

Beginning in 1975, zooplankton dry weights were determined to estimate biomass at each station. Each month, triplicate weight measurements were made with a

Cahn electrobalance for groups of 3 to 30 preserved animals from the numerically dominant taxa. Additional information on laboratory procedures is provided in Hawkins and Evans (1979), Warren et al. (1986), and Evans et al. (1986B).

PREOPERATIONAL-OPERATIONAL DIFFERENCES

Design of the Survey Grid

Prior to conducting preoperational and operational analyses of differences in zooplankton abundance, it was necessary to divide the survey grid into blocks. Examinations of the preoperational and operational data, by survey cruise, had shown significant spatial variations in zooplankton abundance. The greatest spatial variations observed during most cruises were associated with depth or distance from shore (Evans et al. 1980, 1982, 1986B). Subdivision of the survey grid into four depth-related regions (Fig. 2) was supported by the results of principal component analyses and graphical analyses of zooplankton density-depth trends. These regions are designated the inshore (5- to 10-m depth contour), the middle (10- to 20-m depth contour), the inner offshore (20- to 30-m depth contour), and the outer offshore (30- to approximately 45-m depth contour) regions.

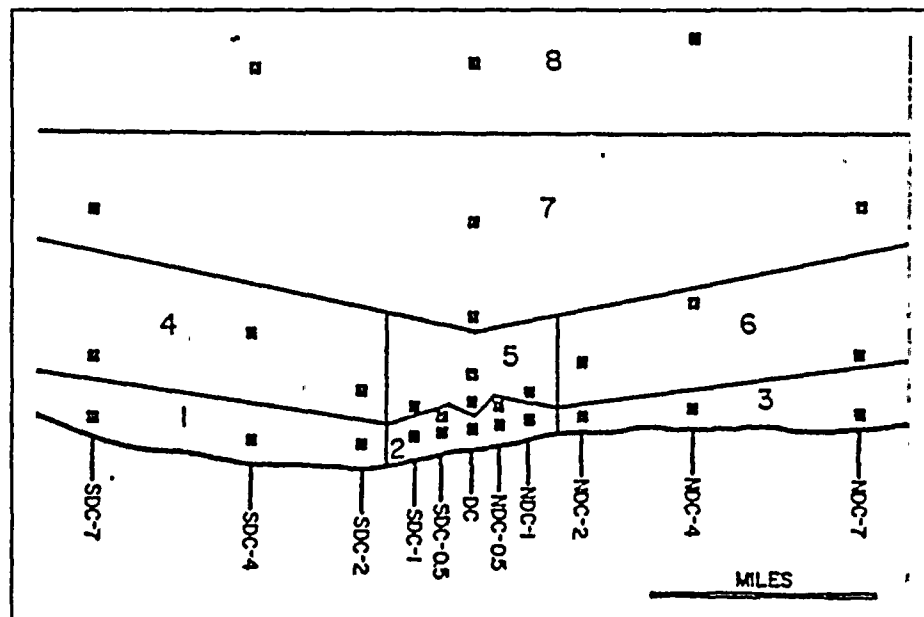


FIGURE 2. Thirty station major survey grid divided into the eight zones used in the preoperational and operational comparisons.

Although zooplankton varied in abundance along transects parallel to shore, this variation was not consistent from month to month and was probably associated with transient zooplankton patchiness. Further subdivisions were based on the location of the thermally detectable plume. Generally, the plume was detected only within 1.6 km of the discharge jets (DC-1) and, while it generally flowed parallel to shore, it sometimes had a strong offshore component (Indiana & Michigan Electric Company 1976, Evans et al. 1986B). The inshore and middle depth regions were subdivided into plume zones extending 1.6 km north and south of the discharge site, and into northern and southern control zones (Fig. 2). The small number of stations in the two offshore zones precluded further subdivisions of these areas (Fig. 2).

Total zooplankton in the inshore plume zone (zone 2), the middle plume zone (zone 5), and the inner and outer offshore zones (Zones 7 and 8) was examined in time series graphs. These graphs facilitate examination of long-term abundance trends. These graphs also provide information on the magnitude of temporal variations in zooplankton abundance both from month to month and from year to year. Data from both major surveys and short surveys were used to delineate long-term trends in the plume region. However, too few stations were sampled in the northern and southern control areas during the short surveys to justify similar preoperational and operational comparisons by month.

Preoperational and operational taxa abundances were compared to test whether significant differences in density had occurred between the two periods. Statistical analyses comparing zooplankton abundances between the preoperational and operational periods were the major tests used in evaluating plant effects. These analyses utilized the major survey data and compared zooplankton abundances by zone and by month between the preoperational and operational periods. Only data collected during the April, July, and October major survey cruises (27 to 30 stations) were used in preoperational-operational statistical comparisons. Too few stations were sampled during the short survey cruises to justify statistical comparisons between the preoperational and operational periods.

Statistical Test Design

The data sets were stratified into preoperational and operational blocks and station density estimates were compared by using the Mann-Whitney U test (Siegel 1956, Conover 1971). These comparisons were made separately for each month and zone in order to reduce spatial and temporal variability which was irrelevant to our investigation of power plant impact. The Mann-Whitney U test is a non-parametric procedure based on the ordered ranks of the data. A two-sided test was used to evaluate the zone densities before and during plant operation. Because not all data were normally distributed, even after transformation, parametric statistical tests were not used.

Calculations were performed on the AMDAHL 470V/8 computer at the University of Michigan using the TWOSAMPLE program incorporated into MIDAS. Zone densities differing at the 95% confidence level were considered statistically significant.

Zooplankton Taxa Tested

Although over 40 species of zooplankton taxa were identified in the survey area, it was neither practical nor necessary to analyze the distributions of all taxa. To

minimize statistical variability due to small population size, only those taxa which were numerically abundant during at least one season were analyzed. The numerically dominant species in the spring, summer, autumn, and winter were: *Cyclops* spp., *Diaptomus* spp., *Bosmina longirostris*, *Eubosmina coregoni*, and *Daphnia* spp. (Evans et al. 1980, 1982, 1986B).

Of the temperature-sensitive zooplankton, statistical analyses were performed only for *Limnocalanus macrurus* in April: this taxon was not sufficiently abundant over the survey grid in July and October to merit further statistical analyses. The preoperational data base was not adequate for *Daphnia longiremis* and *Diaptomus sicilis* since these taxa were identified to species at only a limited number of stations. Furthermore, *D. longiremis* was only rarely collected in the study area (Evans et al. 1980, 1982, 1986B).

Comparisons were made at several taxonomic levels. Order and suborder classifications (i.e., Cladocera, Cyclopoida, Calanoida) were used in order to utilize the largest possible preoperational data set (1971-1974) for making preoperational and operational comparisons. Comparisons at the genus or species level and for immature and adult copepodites could be made only with a 2- or 3-year subset of the preoperational data base.

MORTALITY STUDIES

Studies were conducted to determine the extent of zooplankton mortality attributable to plant passage. Mortality studies were conducted monthly from January 1975 to May 1982. Discharge units were sampled only when that unit was in operation.

Samples were collected from the intake and discharge forebays with the Zaggot Trap sampler and diaphragm pump (Yocum et al. 1978). Mode of operation is described in Evans et al. (1986B). Sampling was conducted for 2 minutes with approximately 0.2 m³ (40 gallons) of water filtered each minute. After July 1977, four samples were collected from each intake and discharge location. One subsample from each replicate was immediately examined (0 hour). Another subsample from each sample was examined 6 hours later and a third was examined 24 hours later. All samples were incubated at intake temperatures.

In the laboratory, each subsample was examined in a circular counting dish under a stereo dissecting microscope. Organisms which exhibited no visceral or appendicular movements even after gentle prodding were classified as "dead." Dead organisms were identified and placed in a separate vial and preserved with a sugar-formaldehyde solution (Haney and Hall 1970). After complete examination of the sample, the remaining live zooplankton were preserved for later examination.

The percentage of dead zooplankton in each intake and discharge sample was calculated for all zooplankton taxa observed. The average taxa mortality for each sampling location/incubation series was calculated as the weighted mean taxa mortality of the appropriate four replicate values.

To test whether discharge mortalities of the taxa were significantly greater than intake mortalities over the entire (1975-1982) operational period, a Wilcoxon sign-rank test (Conover 1971) was used. Ten animals per taxon was the lower limit for inclusion of an observation into the 8-year data set. Additional information on laboratory procedures is provided in Evans et al. (1986A, 1986B).

ENTRAINMENT ABUNDANCE ESTIMATES

Entrainment sampling was performed to determine the numbers and biomass of zooplankton that passed through the plant every month. These estimates were used in conjunction with estimated mortalities due to plant passage to determine the numbers and biomass lost due to power plant operation. Entrainment samples were collected once a month from February 1975 to May 1982. Samples in the intake forebay were collected from a depth of 5.5 m from a location determined to be statistically representative of the forebay (Evans et al. 1982).

Two replicate 5-minute samples (approximately 1 m³ for each sample) were collected at sunset, midnight, sunrise, and noon. A Hale diaphragm pump withdrew water from the forebay and discharged it into a 20-cm (February 1975 to July 1977) or 30-cm diameter (after July 1977), 156- μ m-mesh net that was suspended in a barrel of water. Two subsamples, obtained with a Folsom plankton splitter, were examined from each sample. Counting methods were as previously described. Biomass of zooplankton passing through the plant was determined as described previously. Estimates of individual biomass used in the calculations were derived from zooplankton collected during survey cruises. Estimates of numbers of biomass lost were calculated using 0-hour mortality data from Units 1 and 2 discharges. Additional information on field and laboratory methods is provided in Evans and Flath (1984) and Evans et al. (1986B).

RESULTS

General Seasonal Features of Zooplankton Community Structures

Zooplankton community structure varied both seasonally and spatially. Copepods dominated the spring zooplankton community while cladocerans were dominant during summer, especially in the nearshore area; autumn was a transition period. More than 40 species of zooplankton were observed in the study area (Table 1).

The abundance of each zooplankton taxon varied on a seasonal basis. During April, nauplii, adult *Diaptomus* spp., adult *Cyclops bicuspidatus thomasi*, and *Limnocalanus macrurus* generally were most abundant (Evans 1975, Evans et al. 1978, 1982, 1986B). Cladocera were rare. Distribution of total zooplankton in April 1977 (Fig. 3a) was representative of a pattern common to April in other years. Lower total zooplankton densities occurred generally at shallow stations, and densities decreased with distance offshore (Evans et al. 1978, 1982, 1986B). Biomass patterns were more complex, but also illustrate a trend of a decrease in standing stock (as mg dry wt/m³) with increasing depth (Fig. 3b). There was no evidence of gross alterations in zooplankton abundance in the vicinity of the thermal plume.

During May, nauplii were again numerically the most abundant group. Immature *Diaptomus* spp. and *Cyclops* spp. copepodites were also abundant (Evans et al. 1978, 1982, 1986B). Total zooplankton densities in May 1977 displayed a pattern associated with late spring and summer: densities increased with depth to 20 or 30 m and then remained constant or declined with greater station depth (Fig. 4a). Biomass mimicked this trend, increasing depth contour from 30 mg/m³ at the shallowest stations, to 60 mg/m³ at approximately 30 m, and decreasing to 30 mg/m³ at the deepest station (Fig. 4b).

TABLE 1. Zooplankton taxa found in the D. C. Cook Nuclear Plant area (southeastern Lake Michigan) from 1972 to 1982.

CYCLOPOIDA	CLADOCERA
<i>Cyclops bicuspidatus thomasi</i>	<i>Alona affinis</i>
<i>Cyclops vernalis</i>	<i>Alona guttata</i>
<i>Ergasilus</i> sp.	<i>Alona quadrangularis</i>
<i>Eucyclops agilis</i>	<i>Alona rectangula</i>
<i>Eucyclops prionophorus</i>	<i>Bosmina longirostris</i>
<i>Eucyclops speratus</i>	<i>Camptocercus rectirostris</i>
<i>Mesocyclops edax</i>	<i>Ceriodaphnia quadrangula</i>
<i>Paracyclops fimbriatus poppei</i>	<i>Chydorus sphaericus</i>
<i>Tropocyclops prasinus mexicanus</i>	<i>Daphnia galeata mendotae</i>
	<i>Daphnia longiremus</i>
	<i>Daphnia parvula</i>
	<i>Daphnia pulicaria</i>
	<i>Daphnia retrocurva</i>
CALANOIDA	<i>Diaphanosoma</i> spp.
<i>Diaptomus ashlandi</i>	<i>Disparalona rostrata</i>
<i>Diaptomus minutus</i>	<i>Eubosmina coregoni</i>
<i>Diaptomus oregonensis</i>	<i>Eurycercus lamellatus</i>
<i>Diaptomus reighardi</i> *	<i>Holopedium gibberum</i>
<i>Diaptomus sicilis</i>	<i>Iloycryptus acutifrons</i>
<i>Epischura lacustris</i>	<i>Iloycryptus sordidus</i>
<i>Eurytemora affinis</i>	<i>Iloycryptus spinifer</i>
<i>Limnocalanus macrurus</i>	<i>Latona setifera</i>
<i>Senecella calanoides</i>	<i>Leptodora kindtii</i>
	<i>Leydigia quadrangularis</i>
	<i>Macrothrix laticornis</i>
HARPATICOIDA	<i>Pleuroxus denticulatus</i>
<i>Bryocamptus</i> spp.	<i>Pleuroxus procurvis</i>
<i>Canthocamptus</i> spp.	<i>Polyphemus pediculus</i>
	<i>Scapholeberis kingi</i>
	<i>Sida crystallina</i>
	<i>Simocephalus serratulae</i>
	<i>Simocephalus vetulus</i>
ROTIFERA**	MALACOSTRACA
<i>Asplanchna herrichi</i>	<i>Mysis relicta</i>
<i>Asplanchna priodonta</i>	

*Only one specimen observed.

***Asplanchna* is the only rotifer genus routinely enumerated in these collections.

During June, *Bosmina longirostris* was often the most abundant taxon, followed by nauplii and immature *Cyclops* spp. and *Diaptomus* spp. copepodites (Evans et al. 1978, 1982, 1986B). Differences in inshore and offshore taxa were often evident, with *B. longirostris* more abundant inshore, and immature copepodites abundant offshore. During some years the rotifer, *Asplanchna* spp., was a major component of the June plankton (Evans et al. 1978, 1982, 1986B). Total zooplankton occurred in unusually high concentrations at the discharge (90,000/m³) in June 1977 (Fig. 5a). If this value is disregarded, there was a slight trend of increased density with depth.

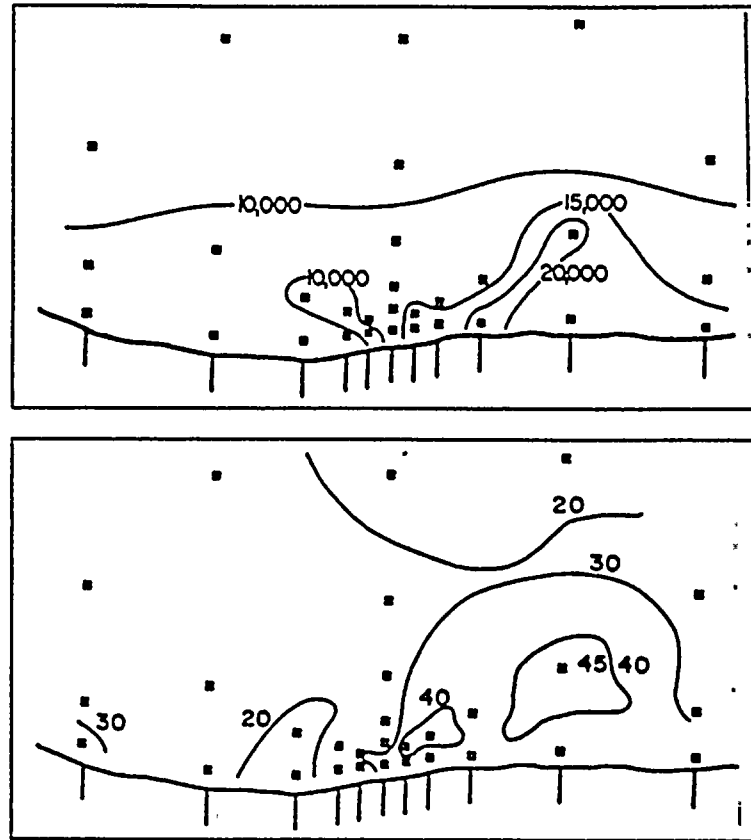


FIGURE 3. Horizontal distribution of (a) total zooplankton (number/m³) and (b) the standing stock of zooplankton (mg dry weight/m³) on 14 April 1977.

This trend is more evident in the biomass data (Fig. 5b). The pattern of increasing biomass with depth is one that often persisted for most of the summer.

Inshore-offshore gradients were usually well developed by July. *Bosmina longirostris* was the dominant zooplankter inshore, and immature *Cyclops* spp. and *Diaptomus* spp. copepodites and *Daphnia* spp. increased in dominance at offshore stations (Evans et al. 1978, 1982, 1986B). The most frequently observed summer pattern, an increased decrease farther offshore, was evident during July 1977 (Fig. 6a). However, biomass increased to the deepest station (Fig. 6b).

August zooplankton communities were generally dominated by *B. longirostris* in shallow waters, and by nauplii and immature *Cyclops* spp. and *Diaptomus* spp. copepodites offshore (Evans et al. 1978, 1982). The community gradient was not greatly different from that observed in July. Total zooplankton density (Fig. 7a) and

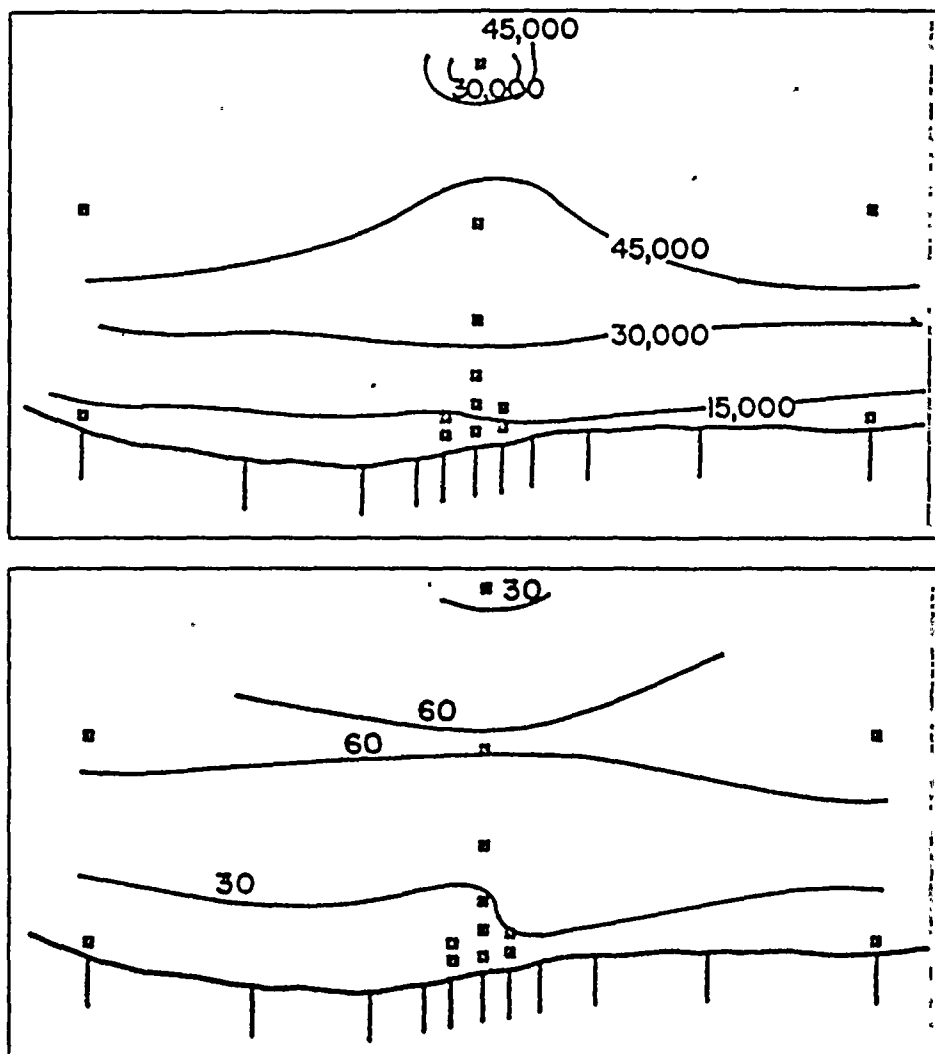


FIGURE 4. Horizontal distribution of (a) total zooplankton (number/m³) and (b) the standing stock of zooplankton (mg dry weight/m³) on 18 May 1977.

biomass (Fig. 7b) increased with depth in August 1977, similar to patterns observed during other August surveys.

Immature *Cyclops* spp. and *Diaptomus* spp. copepodites and nauplii often dominated the September zooplankton community. However, during several years, *B. longirostris*, and less frequently *Daphnia retrocurva*, contributed significantly to the

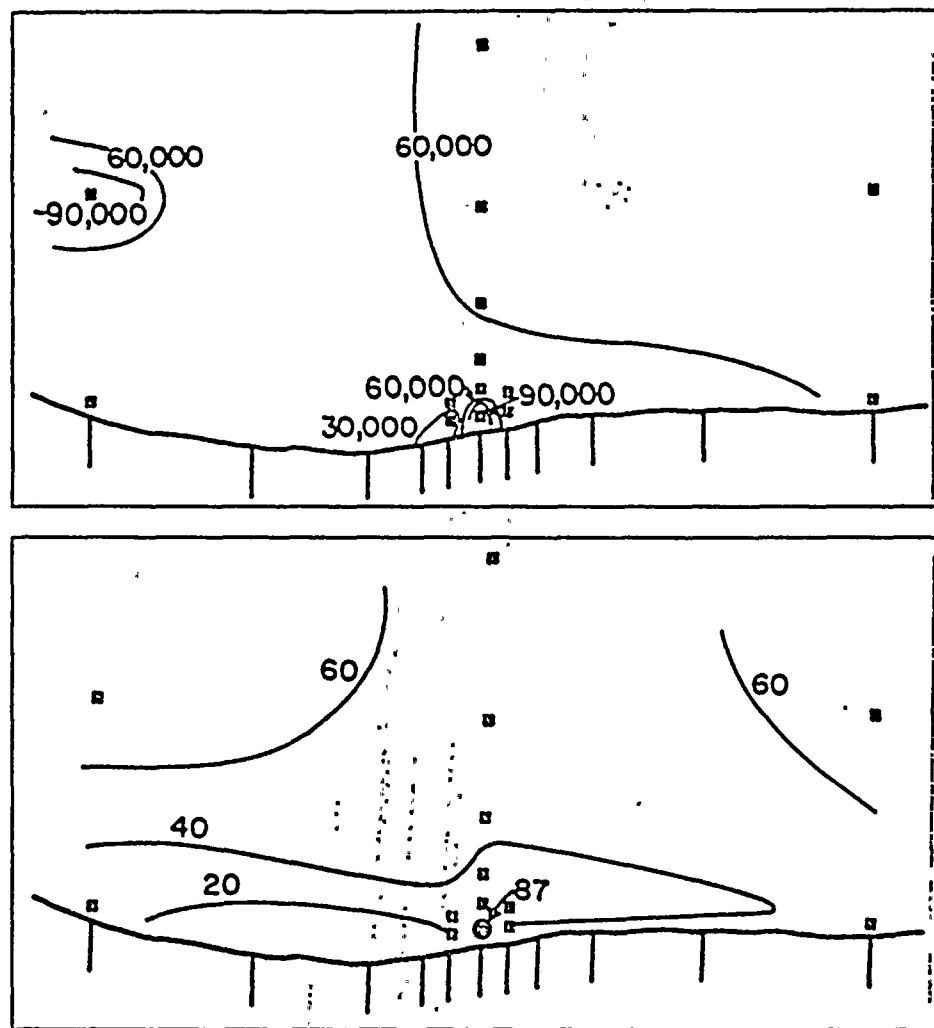


FIGURE 5. Horizontal distribution of (a) total zooplankton (number/m³) and (b) the standing stock of zooplankton (mg dry weight/m³) on 16 June 1977.

zooplankton communities at shallow, nearshore stations (Evans et al. 1978, 1982, 1986B). Total zooplankton numbers were highest nearshore adjacent to the plant in September 1977, numbers decreased with increasing water depth to depth contours of 20 or 30 m, and then remained fairly constant at deeper stations (Fig. 8a). The distribution of biomass over the survey grid had virtually no pattern in 1977 (Fig. 8b). This is contrary to the more usual pattern of an increase in biomass with depth.

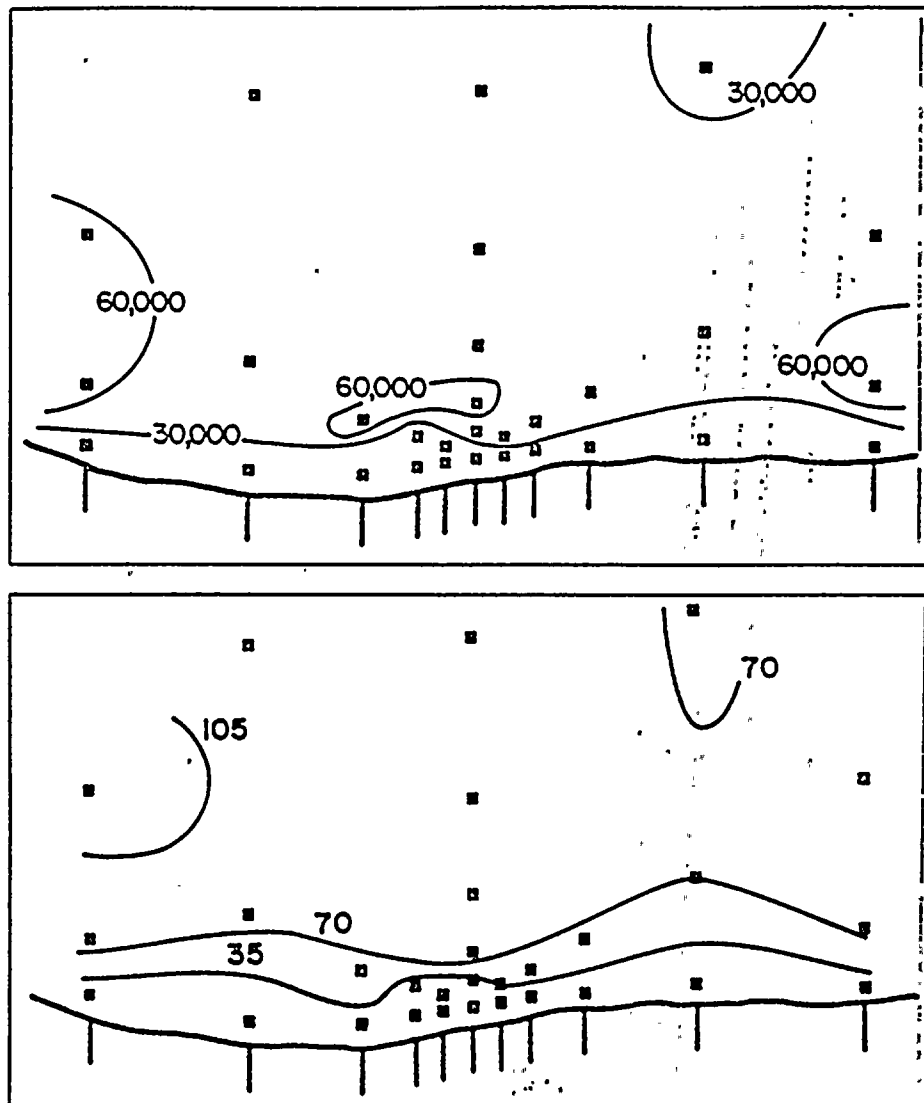


FIGURE 6. Horizontal distribution of (a) total zooplankton (number/m³) and (b) the standing stock of zooplankton (mg dry weight/m³m) on 13 July 1977.

The upwelling that occurred during this month may account for the lack of pattern (Evans et al. 1982).

The zooplankton community was again dominated by nauplii and immature *Cyclops* spp. and *Diaptomus* spp. copepodites in October. *Daphnia* spp., especially

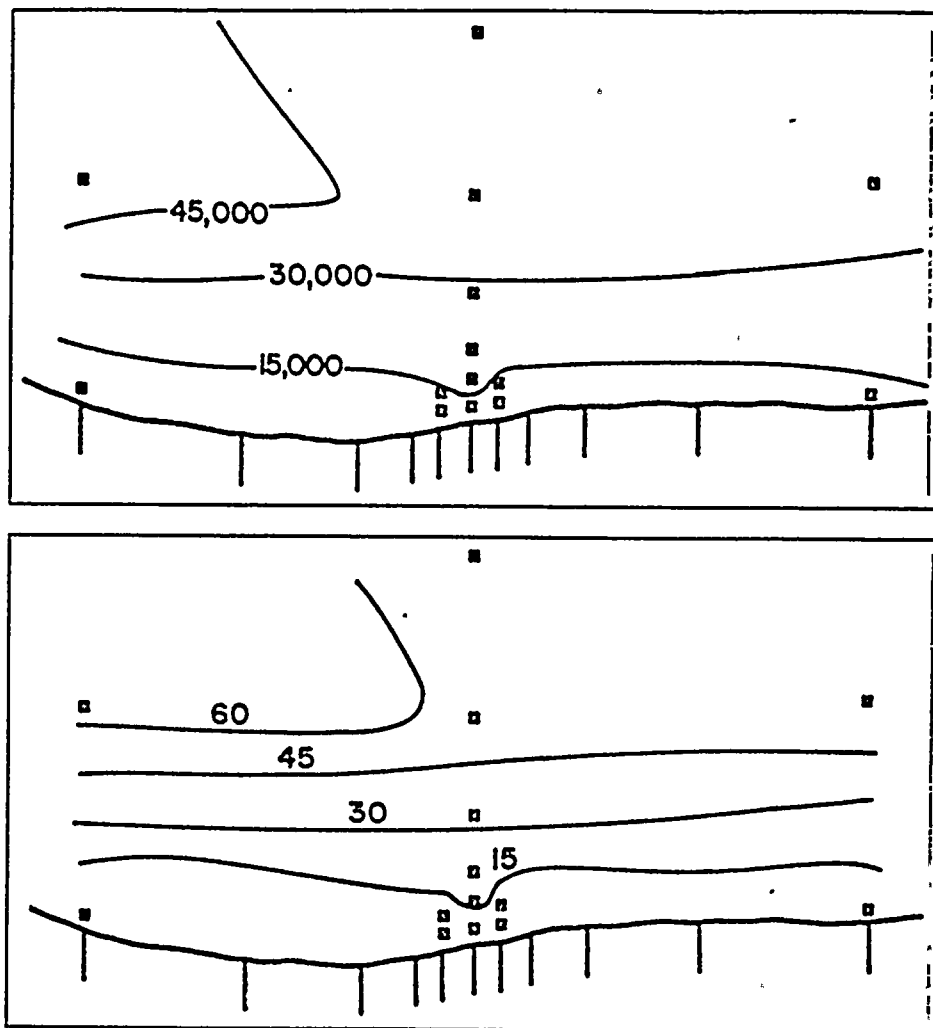


FIGURE 7. Horizontal distribution of (a) total zooplankton (number/m³) and (b) the standing stock of zooplankton (mg dry weight/m³m) on 10 August 1977.

D. galeata mendotae, were often an abundant component of the zooplankton at deeper stations, while *B. longirostris* was more abundant inshore (Evans et al. 1978, 1982). During October 1977, total zooplankton number varied both alongshore and offshore (Fig. 9a). However, there was a slight trend toward a decrease in numbers with increasing water depth. Biomass increased with depth to a depth contour of 20–30 m and then remained constant (Fig. 9b).

Immature *Diaptomus* spp. and *Cyclops* spp. copepodites and nauplii were the

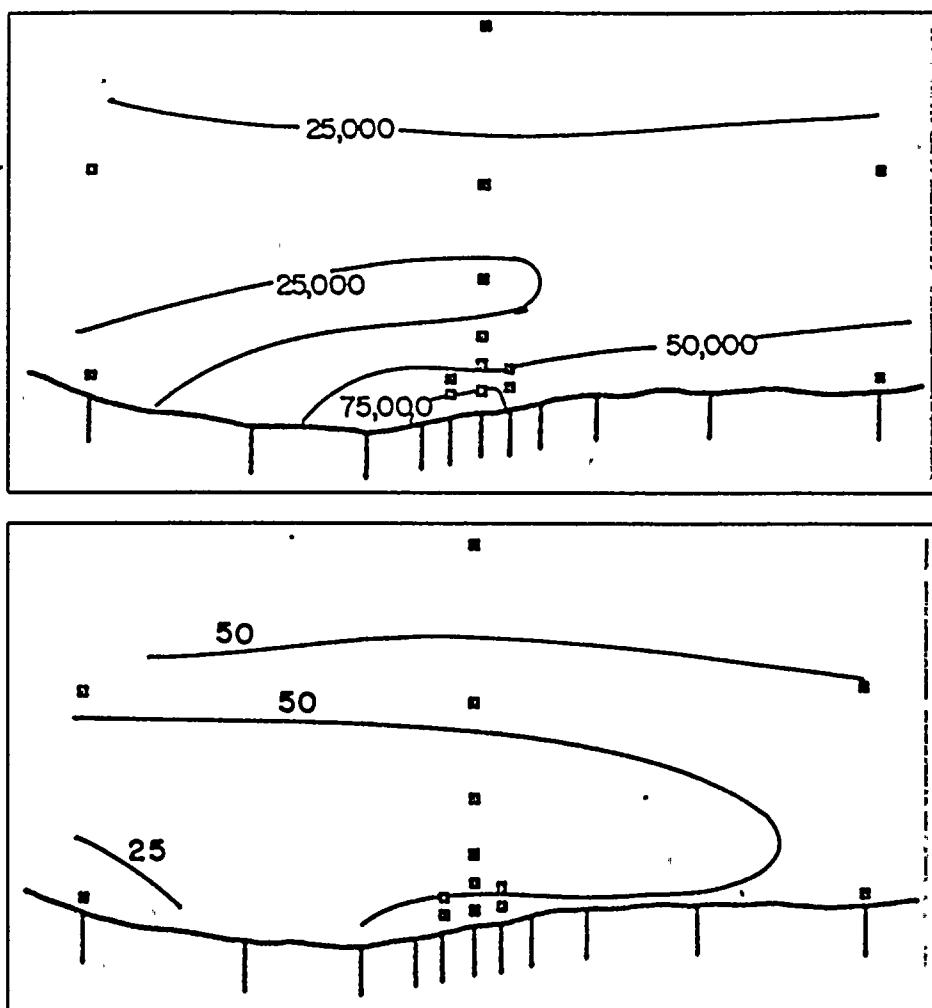


FIGURE 8. Horizontal distribution of (a) total zooplankton (number/m³) and (b) the standing stock of zooplankton (mg dry weight/m³m) on 14 September 1977.

major components of the zooplankton community in November. *Bosmina longirostris* or *Eubosmina coregoni* were occasionally abundant at shallow stations, but never as abundant as *B. longirostris* in summer (Evans et al. 1978, 1982, 1986B). Inshore-offshore community gradients were weak or nonexistent by November. In November 1977 total zooplankton abundances decreased with increasing water

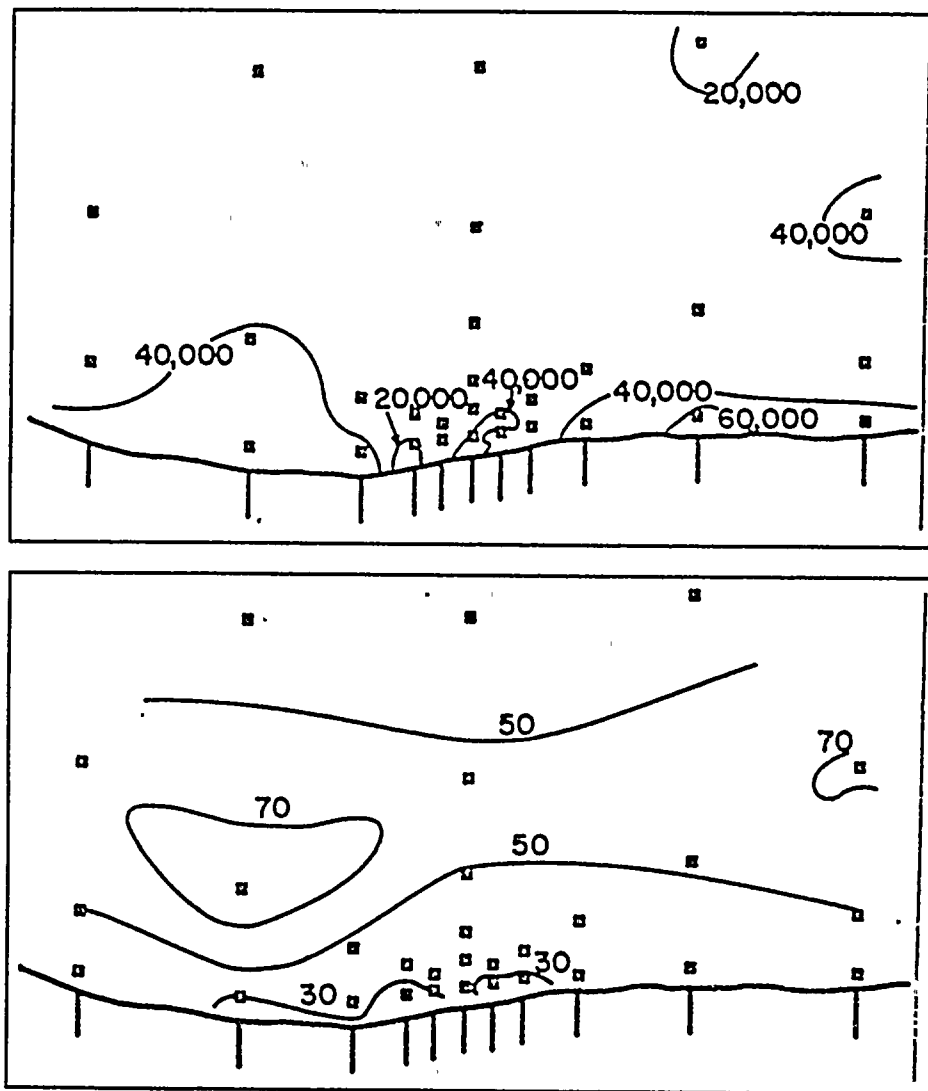


FIGURE 9. Horizontal distribution of (a) total zooplankton (number/m³) and (b) the standing stock of zooplankton (mg dry weight/m³m) on 14 October 1977.

depth (Fig. 10a). Biomass first increased, then decreased with increasing water depth in 1977 (Fig. 10b).

Winter conditions prevailed in the December zooplankton community. Adult *Diaptomus ashlandi* and immature *Cyclops* spp. copepodites dominated. The overwintering zooplankton were composed of adults and late instar immatures of most

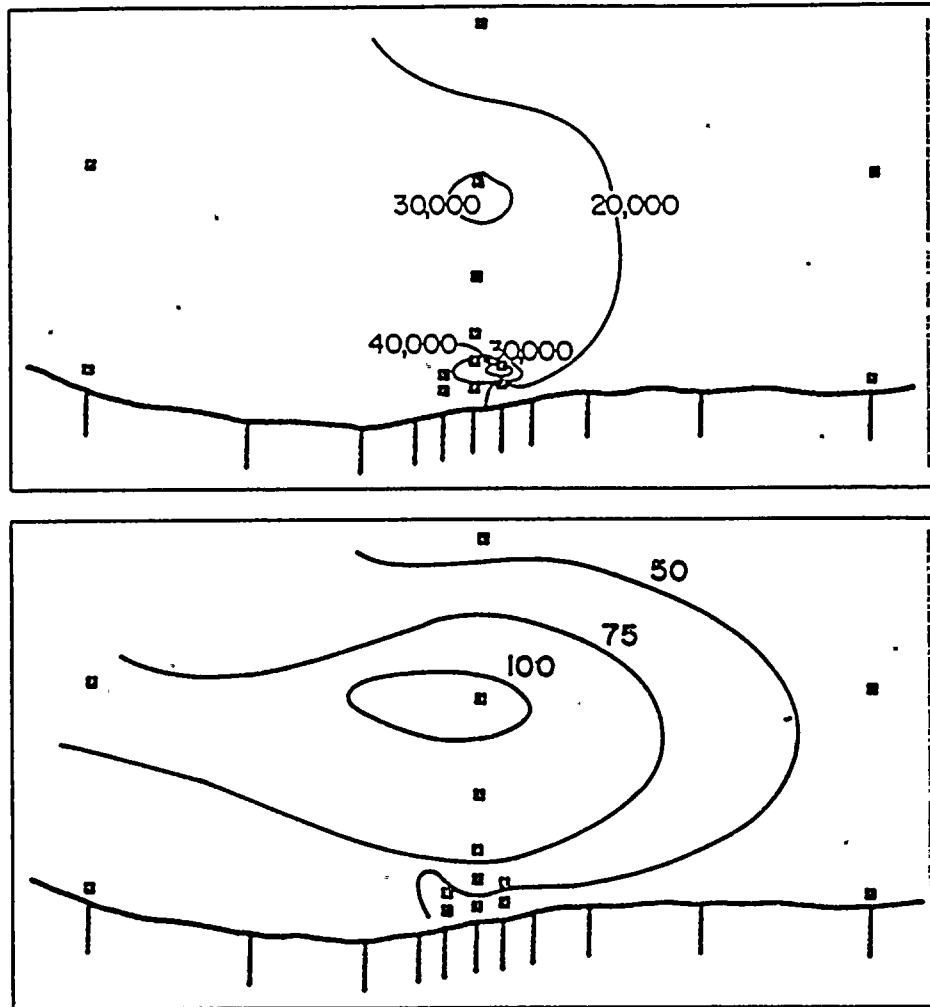


FIGURE 10. Horizontal distribution of (a) total zooplankton (number/m³) and (b) the standing stock of zooplankton (mg dry weight/m³) on 9 November 1977.

species (Evans et al. 1978, 1982, 1986B). Very little pattern was observed in either numbers (Fig. 11a) or biomass (Fig. 11b) of total zooplankton in December 1977.

There were no obvious disruptions in zooplankton distributional patterns in the vicinity of the thermal plume (Evans et al. 1978, 1982, 1986B) as determined from the lake survey cruises. In general, the thermal plume was detectable at only one to three stations. Zooplankton abundances at these stations were similar to those observed in north and south control regions.

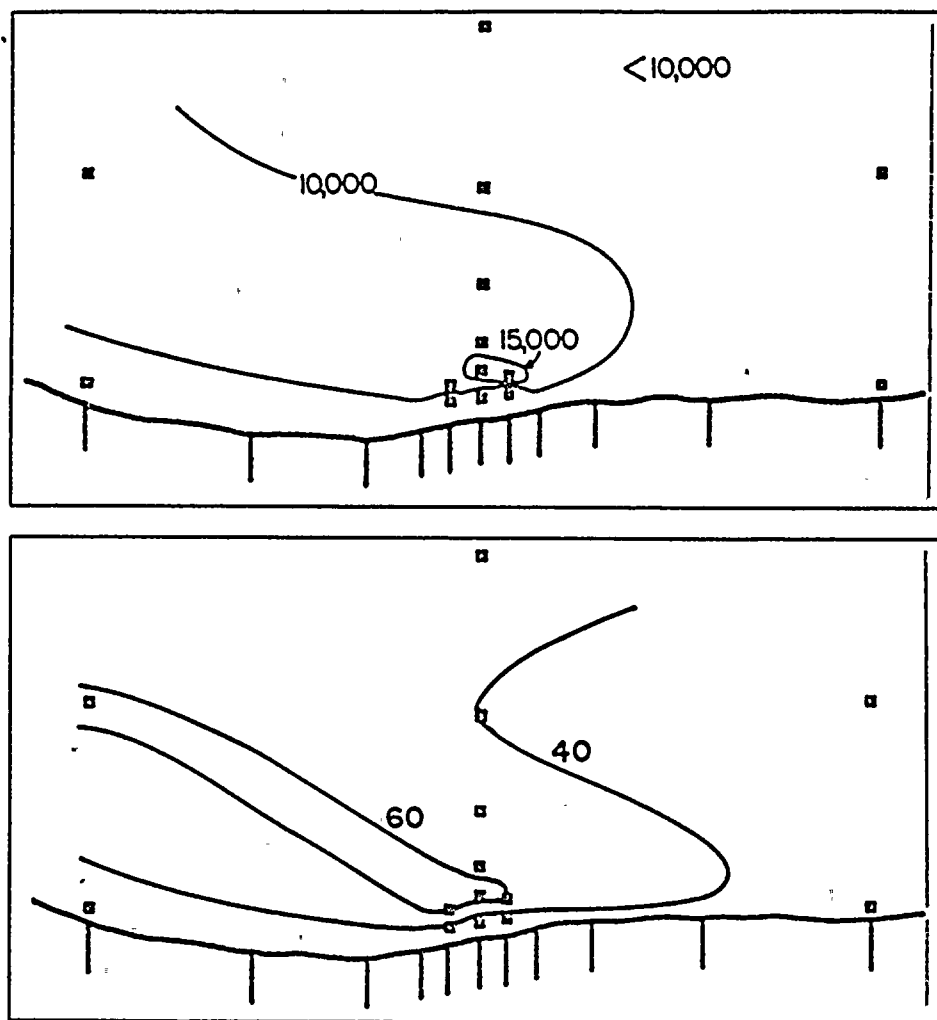


FIGURE 11. Horizontal distribution of (a) total zooplankton (number/m³) and (b) the standing stock of zooplankton (mg dry weight/m³) on 15 December 1977.

TEMPORAL ABUNDANCE PATTERNS OF TOTAL ZOOPLANKTON IN ZONES 2, 5, 7, AND 8 (1971-1982)

Although zooplankton differed in abundance with respect to depth, seasonality of total zooplankton densities was similar to zones 2, 5, 7, and 8 (Figs. 12-15). Highest densities occurred from July to September, with yearly maxima during one of these months. Ranges of total zooplankton densities during the preoperational and opera-

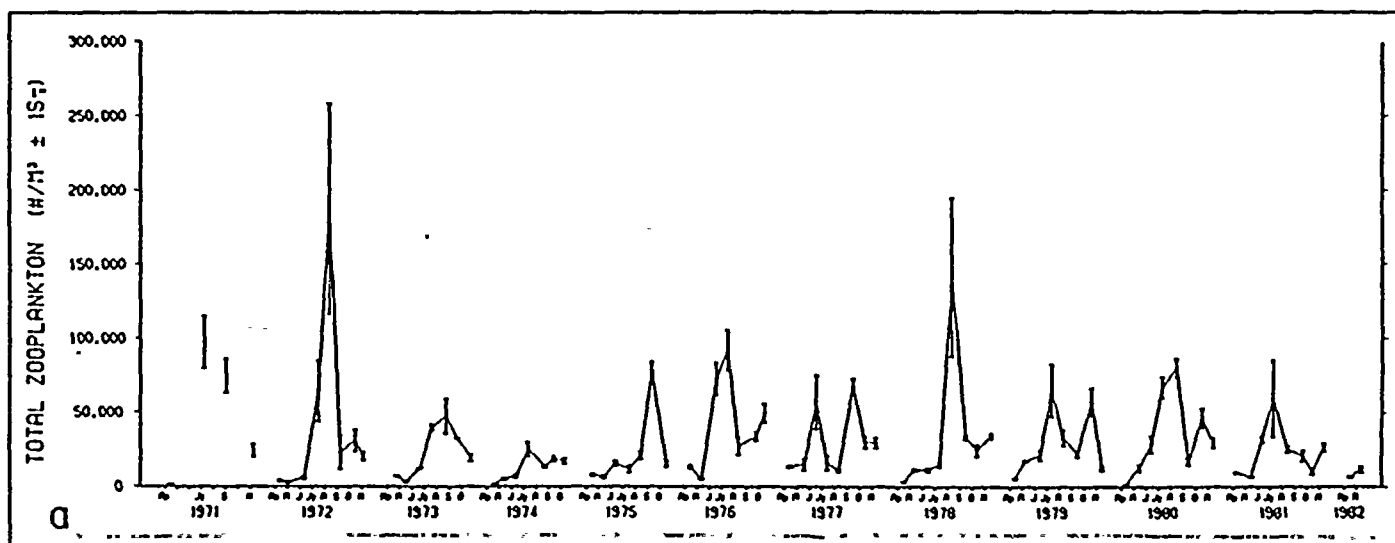


FIGURE 12. The monthly abundance of total zooplankton in the inshore plume zone (zone 2) between 1970 and 1982.

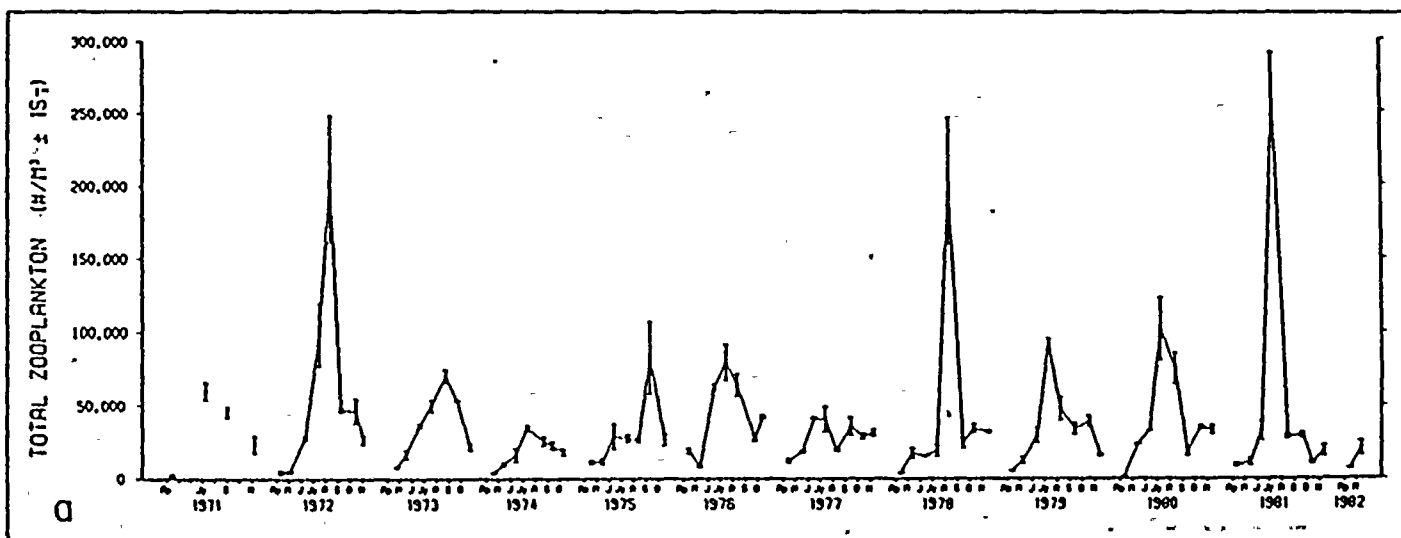


FIGURE 13. The monthly abundance of total zooplankton in the middle shore zone (zone 5) between 1970 and 1982.

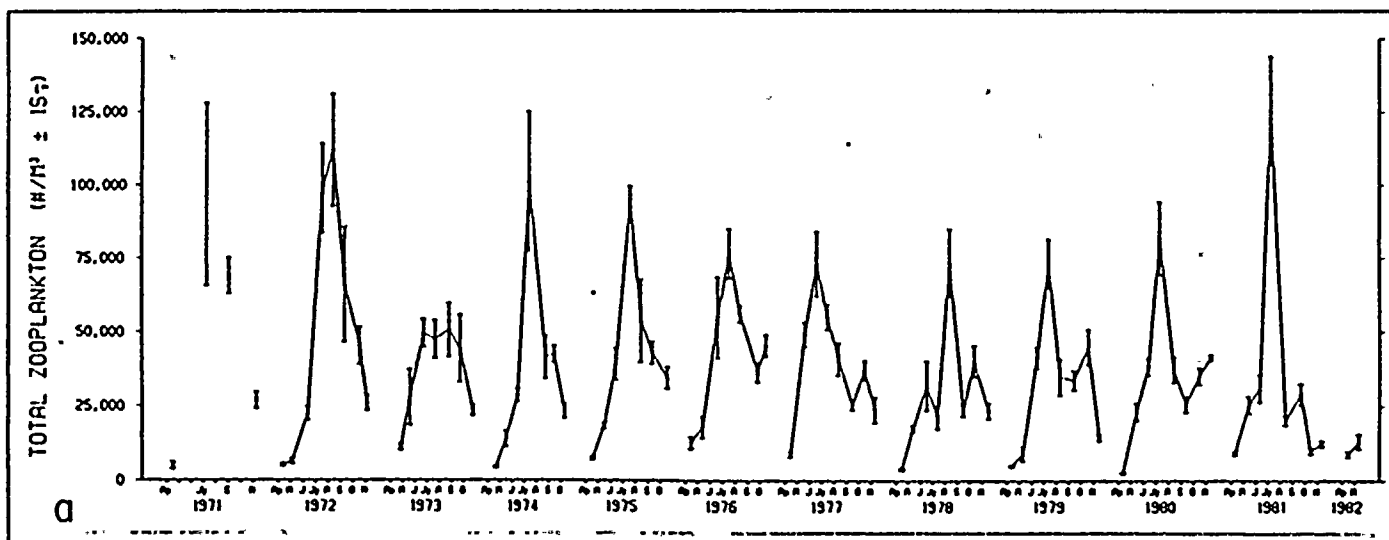


FIGURE 14. The monthly abundance of total zooplankton in the inner offshore zone (zone 7) between 1970 and 1982.

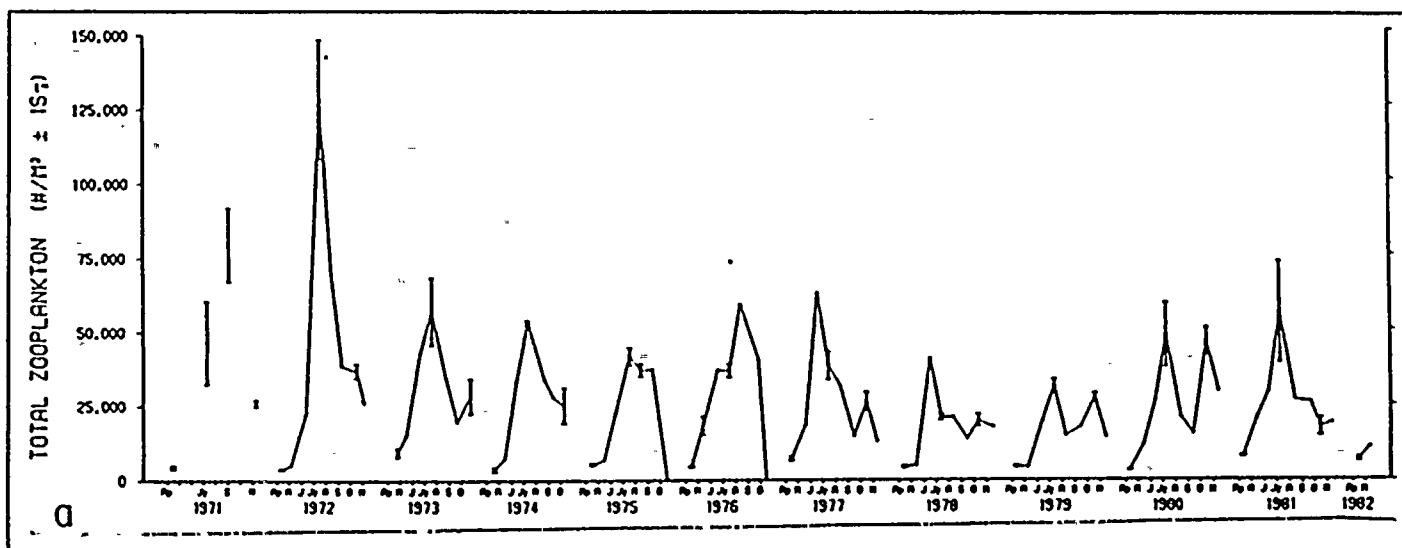


FIGURE 15. The monthly abundance of total zooplankton in the outer offshore zone (zone 8) between 1970 and 1982.

tional periods were generally similar in all depth zones. The variations in numbers during the summer were due mainly to *B. longirostris* in the more inshore zones, and to a combination of *B. longirostris*, *Cyclops* spp. C1-66, and *Diaptomus* spp. C1-C5 in more offshore zones (Evans et al. 1986B).

COMPARISONS OF APRIL, JULY, AND OCTOBER PREOPERATIONAL:OPERATIONAL TAXA ABUNDANCES

Statistical comparisons of zooplankton abundances in the preoperational and operational time periods were based on non-parametric statistical analyses of the eight zones comprising the survey grid. Results of Mann-Whitney U tests of taxa abundances by major survey cruise and for each of the eight zones are summarized. Detailed results of these analyses appear in Evans et al. (1986B).

With the exception of immature *Diaptomus* spp. copepodites, taxa analyzed exhibited statistically significant ($\alpha = 0.05$) differences between preoperational and operational abundances in at least one of the eight zones during April (Table 2). Several taxa exhibited no consistent pattern of preoperational-operational differences. Evans et al. (1986B) reported that total zooplankton densities were similar in the two periods, with the exception of zone 2 where operational densities were higher than preoperational densities. Adult *Cyclops* spp. were more abundant inshore during the operational period, and more abundant offshore during the preoperational period. Immature *Diaptomus* spp. were more abundant offshore during the preoperational period, but densities were generally similar between monitoring periods in other zones. Nauplii, adult *Diaptomus* spp., and *Limnocalanus macrurus* were generally more abundant during the operational and preoperational period. Of

TABLE 2. Results of the Mann-Whitney U tests comparing April preoperational and operational densities of nine zooplankton taxa in each of eight zones. The preoperational period is 1971-74 or a subset ending in 1974, and the operational period is 1975-82.

Taxon Order and Suborder Level	Zone								Period
	1	2	3	4	5	6	7	8	
Copepod nauplii	NS	*	NS	NS	NS	NS	NS	NS	72-82
Cyclopoids (C1-C6)	NS	NS	NS	*	*	*	*	*	71-82
Calanoids (C1-C6)	*	*	*	NS	*	*	NS	NS	71-82
<i>Genus and developmental stage</i>									
Cyclopoids (C1-C5)	NS	*	*	*	*	*	*	*	73-82
<i>Cyclops</i> spp. C6	NS	NS	NS	NS	NS	NS	*	*	73-82
<i>Diaptomus</i> spp. (C1-C5)	NS	NS	NS	NS	NS	NS	NS	NS	73-82
<i>Diaptomus</i> spp. C6	NS	*	NS	NS	*	*	NS	NS	73-82
<i>Limnocalanus macrurus</i> (C1-C6)	*	*	*	*	*	*	NS	NS	71-82
Total zooplankton	NS	*	NS	NS	NS	NS	NS	NS	72-83

*significant difference, $\alpha = 0.05$

NS not significant.

the taxa analyzed, only immature *Cyclops* spp. were generally less abundant during the operational than preoperational period.

Total zooplankton densities were again similar between operational and preoperational periods during July (Table 3). Immature and adult *Cyclops* spp. copepodites showed no consistent trends. Evans et al. (1986B) reported that immature *Cyclops* spp. operational densities were lower than preoperational densities at about half the zones, but this difference was significant only in zone 2. Adult *Cyclops* spp. densities were similar between periods with the exception of zones 2 and 8, where operational densities were significantly less than preoperational densities. Immature and adult *Diaptomus* spp. copepodites were generally less abundant during operational than preoperational periods, with significant differences occurring in several zones. Nauplii and *Asplanchna* spp. were generally less abundant during the operational period. However nauplii were more abundant during the operational period in zone 1. The opposite pattern was exhibited by *Bosmina longirostris* and *Daphnia* spp. which were generally more abundant during the operational period. While these increases were significant for *B. longirostris* only in zone 8, they were significant in all zones for *Daphnia* spp., which increased from four- to twenty-fold. The copepod *Eurytemora affinis* was also more abundant during the operational period although none of the differences were statistically significant.

During October, several taxa were more abundant during the operational than preoperational period (Table 4). Evans et al. (1986B) reported that these included total zooplankton, nauplii, immature *Cyclops* spp., *B. longirostris*, and *Eubosmina*

TABLE 3. Results of the Mann-Whitney U tests comparing July preoperational and operational densities of thirteen zooplankton taxa in each of eight zones. The preoperational period is 1971-74 or a subset ending in 1974, and the operational period is 1975-81. In column 1, results shown are from student's t-test analysis ($p < 0.05$) of the three inshore zones combined.

Taxon	Zone									
Order and Suborder Level	1-3	1	2	3	4	5	6	7	8	Period
Cladocerans	NS	NS	NS	NS	NS	NS	NS	NS	•	71-81
Copepod nauplii	NS	•	NS	NS	NS	NS	NS	NS	NS	72-81
Cyclopoids (C1-C6)	NS	NS	•	NS	NS	NS	NS	NS	•	71-81
Calanoids (C1-C6)	•	NS	•	NS	NS	NS	NS	NS	NS	71-81
<i>Genus, species, or developmental stage</i>										
Bosmina longirostris	NS	•	NS	NS	NS	NS	NS	NS	•	72-81
Daphnia spp.	•	•	•	•	•	•	•	•	•	71-81
Cyclopoids (C1-C5)	NS	NS	•	NS	NS	NS	NS	NS	NS	73-81
Cyclops spp. C6	•	NS	•	NS	NS	NS	NS	NS	•	73-81
Diaptomus spp. (C1-C5)	•	NS	•	•	NS	NS	NS	NS	NS	73-81
Diaptomus spp. C6	•	NS	•	•	•	•	NS	NS	NS	73-81
Eurytemora affinis (C1-C6)	NS	NS	NS	NS	NS	NS	NS	NS	NS	73-81
Asplanchna spp.	•	NS	•	•	NS	•	•	•	NS	71-81
Total zooplankton	NS	NS	NS	NS	NS	NS	NS	NS	•	72-81

*significant difference, $\alpha = 0.05$

NS not significant.

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TABLE 4. Results of the Mann-Whitney U tests comparing October preoperational and operational densities of twelve zooplankton taxa in each of eight zones. The preoperational period is 1972-74 or a subset ending in 1974, and the operational period is 1975-81. Stations in zone 8 were not sampled in 1975 or 1976 (see text). In column 1, results shown are from student's t-test analysis ($p < 0.05$) of the three inshore zones combined.

Taxon	Zone								Period
	1-3	1	2	3	4	5	6	7	8
Order and Suborder Level									
Cladocerans	NS	NS	NS	NS	NS	NS	NS	NS	NS
Copepod nauplii	•	NS	•	NS	NS	•	•	•	NS
Cyclopoids (C1-C6)	NS	NS	NS	NS	NS	•	NS	NS	NS
Calanoids (C1-C6)	•	•	NS	NS	NS	NS	NS	NS	NS
Genus, species, or developmental stage									
<i>Bosmina longirostris</i>	•	NS	•	NS	NS	•	•	•	NS
<i>Eubosmina coregoni</i>	•	NS	•	NS	NS	NS	NS	NS	•
<i>Daphnia</i> spp.	•	NS	•	•	NS	•	NS	NS	NS
Cyclopoids (C1-C5)	•	NS	NS	NS	•	•	NS	NS	NS
<i>Cyclops</i> spp. C6	•	NS	•	NS	NS	NS	NS	NS	NS
<i>Diaptomus</i> spp. (C1-C5)	NS	NS	NS	NS	NS	NS	NS	NS	NS
<i>Diaptomus</i> spp. C6	NS	NS	NS	NS	NS	NS	NS	NS	NS
Total zooplankton	•	NS	NS	NS	NS	NS	•	NS	NS

*significant difference, $\alpha = 0.05$

NS not significant.

coregoni. *Eubosmina coregoni* was less abundant, however, during the operational period in zones 3, 4, and 8. Total zooplankton concentrations were significantly higher during the operational period only in zone 6. Nauplii were up to twice as abundant in the operational period as in the preoperational period, and differences were significant in several zones. Immature *Cyclops* spp. were up to twice as dense in the operational versus preoperational periods. *Bosmina longirostris* was significantly more abundant during the operational period in several zones. Numbers of *E. coregoni* were significantly higher during the operational period in inshore zones 1 and 2 as well as in zone 5. Immature *Diaptomus* spp. densities were not significantly different between periods in any zone, but were generally more abundant during the operational period. *Daphnia* spp. and *Cyclops bicuspidatus thomasi* adults were generally less abundant during the operational period. Densities of *Daphnia* spp. were significantly lower during the operational period only in zones 2 and 3. *Cyclops bicuspidatus thomasi* was significantly less abundant during the operational period only in zone 2; however, operational densities were higher than preoperational densities (but not significantly so) in zone 3.

ZOOPLANKTON MORTALITIES

Since the study was conducted over 8 years, the data set is extensive. Practicality limits our graphic presentations of zooplankton mortality to a subset of these data.

The 1979-1982 period was chosen because it provides an extensive period of two-unit operation; 1977-1978 data appear in Evans et al. (1986A).

Intake water temperatures varied seasonally from less than 1°C to over 24°C (Fig. 16). Discharge water temperatures generally exceeded 30°C in late summer. Over the 8 years of study (1975-1982), the ΔT for Unit 1 averaged 9.9°C while the ΔT for Unit 2 averaged 8.9°C. The plant pumped water at a rate varying from 0.55 to 0.85 $\times 10^6$ gpm (2.1 to 3.2 $\times 10^3$ m³/min.) for Unit 1 and 0.75 to 1.18 $\times 10^6$ gpm (2.9 to 4.5 $\times 10^3$ m³/min.) for Unit 2 (Fig. 17). Over the 8 years of study, Unit 1 pumped at an average rate of 2,964 m³/min. while Unit 2 pumped at an average rate of 3,979 m³/min. Thus, Unit 1 operated at a lower pumping rate but higher ΔT than Unit 2.

Total zooplankton mortalities varied seasonally (Fig. 18). In general, discharge mortalities followed intake mortalities. Differences between intake and discharge mortalities generally were small, differing by only a few percent.

Intake mortalities for total zooplankton at 0-hour averaged 10.2% while mean mortalities in Unit 1 and Unit 2 discharge waters averaged 11.4% and 13.4%, respectively (Evans et al. 1986A, 1986B). For the 6-hour incubations, mean mortali-

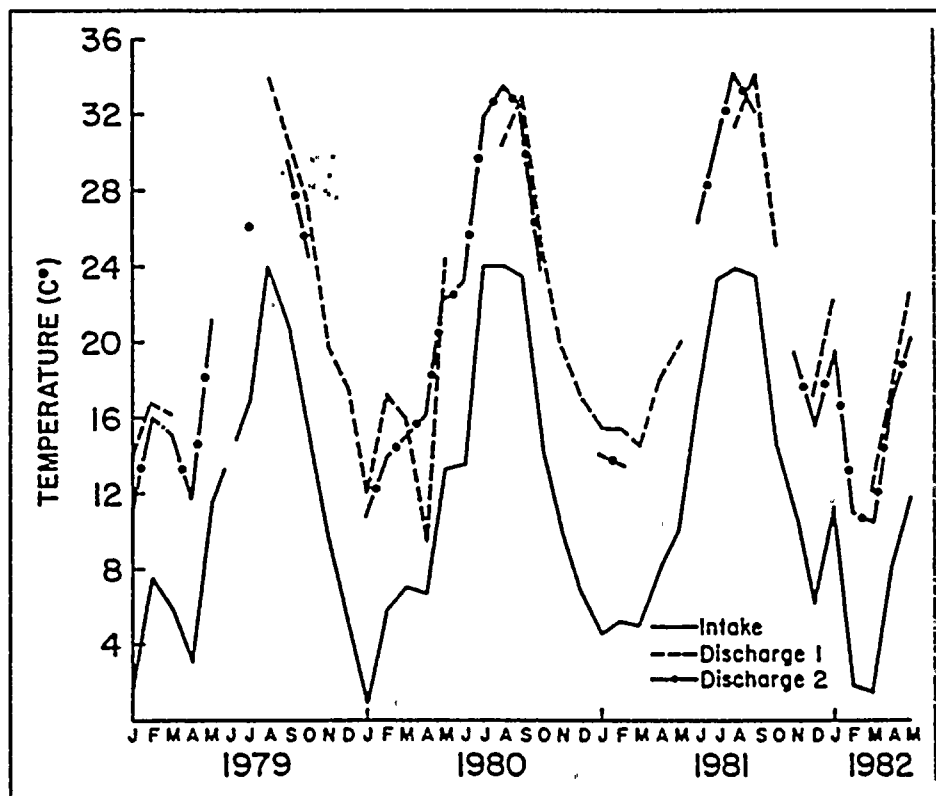


FIGURE 16. Intake and discharge water temperatures for January 1979 to May 1982.

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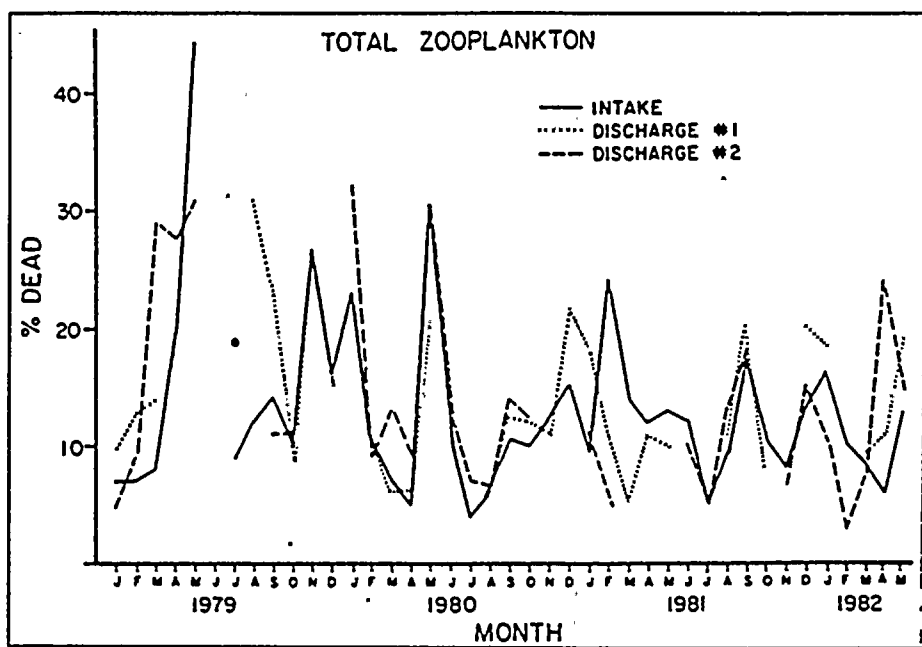


FIGURE 18. Monthly mean mortality (0-hour) of total zooplankton.

ences did not increase with incubation period for either discharge location. Furthermore, there was no evidence that zooplankton mortalities were substantially different between the two units (Evans et al. 1986A).

NUMBERS AND BIOMASS OF ZOOPLANKTON PASSING THROUGH THE POWER PLANT AND THE ESTIMATED MAXIMUM LOSSES

Because of the large size of the data set, we will again limit our graphical presentation to 1979-1982 data (Evans et al. 1986B). These years include data when both units were operating. Billions of zooplankton passed through the plant each month (Fig. 19). The number entrained ranged from 250×10^9 in March 1982 to $20,709 \times 10^9$ in July 1979 and averaged $5,081 \times 10^9$. The number of zooplankton entrained was related to zooplankton concentrations and plant pumping rates. More zooplankton passed through the plant during the summer and autumn months when zooplankton concentrations in the lake nearest the intakes were highest (Fig. 20). Average zooplankton concentration in the cooling waters was $23,915/\text{m}^3$ and ranged from $1,614/\text{m}^3$ in March 1982 to $96,730/\text{m}^3$ in July 1979.

Maximum estimated numerical losses generally followed the numbers entrained curve (Fig. 19). Estimates of maximum loss assumed that the immediate (0-hour) discharge mortality represented maximum losses in the vicinity of the discharge jets. The actual loss probably was lower.

TABLE 5. Zooplankton taxa for which discharge mortalities were significantly higher than intake mortalities over the 1975-1982 period as determined by the Wilcoxon sign-rank test. Intake and discharge values from a month were used only if both values were based on the observation of at least 10 animals. *n* is the number of monthly pairs of values examined, *p* is the attained level of significance, and *d* is the mean monthly difference in mortalities, discharge minus intake. Modified from Evans et al. (1986A).

Incubation	Taxon	n	d
DISCHARGE UNIT 1			
0-hr	<i>Diaptomus</i> spp. C1-C5	53	9.56%
	<i>Diaptomus minutus</i> C6	27	4.61%
	<i>Eurytemora affinis</i> C1-C5	17	1.74%
	<i>Limnocalanus macrurus</i> C6	11	8.57%
	<i>Daphnia retrocurva</i>	20	5.16%
6-hr	<i>Diaptomus</i> spp. C1-C5	53	6.27%
	<i>Diaptomus minutus</i> C6	21	6.53%
	<i>Diaptomus oregonensis</i> C6	14	3.03%
	<i>Limnocalanus macrurus</i> C6	11	3.82%
	<i>Daphnia galeata</i>	12	6.82%
	<i>Eubosmina coregoni</i>	19	4.51%
24-hr	<i>Diaptomus</i> spp. C1-C5	50	6.41%
	<i>Diaptomus ashlandi</i> C6	32	1.96%
	<i>Diaptomus minutus</i> C6	21	3.96%
DISCHARGE UNIT 2			
0-hr	<i>Diaptomus</i> spp. C1-C5	23	8.99%
	Total Zooplankton	26	2.49%
24-hr	<i>Diaptomus</i> spp. C1-C5	19	5.36%
	Total Zooplankton	26	2.06%

The monthly biomass of zooplankton entrained (Fig. 21) ranged from a low of 1,117 kg dry wt/month in March 1982 to a high of 21,991 kg dry wt/month in November 1981 and averaged 8,191 kg dry wt/month. Maximum estimated biomass loss ranged from 70 kg dry wt/month in October 1981 to 4,482 kg dry wt/month in July 1982 and averaged 1,135 kg dry wt/month. The seasonal pattern of biomass entrained did not match that of the numbers entrained. Highest values of biomass entrained occurred during winter months while greatest numbers entrained occurred in summer months. This is because winter zooplankton, while less abundant, were dominated by relatively large animals (mean dry weight of 3 to 7 μ g) while the more numerous summer zooplankton were dominated by relatively small animals (mean dry weight 1 to 2 μ g).

DISCUSSION

There was no evidence from the lake survey cruises that plant operation produced gross spatial or temporal alterations in zooplankton community structure. Seasonal-

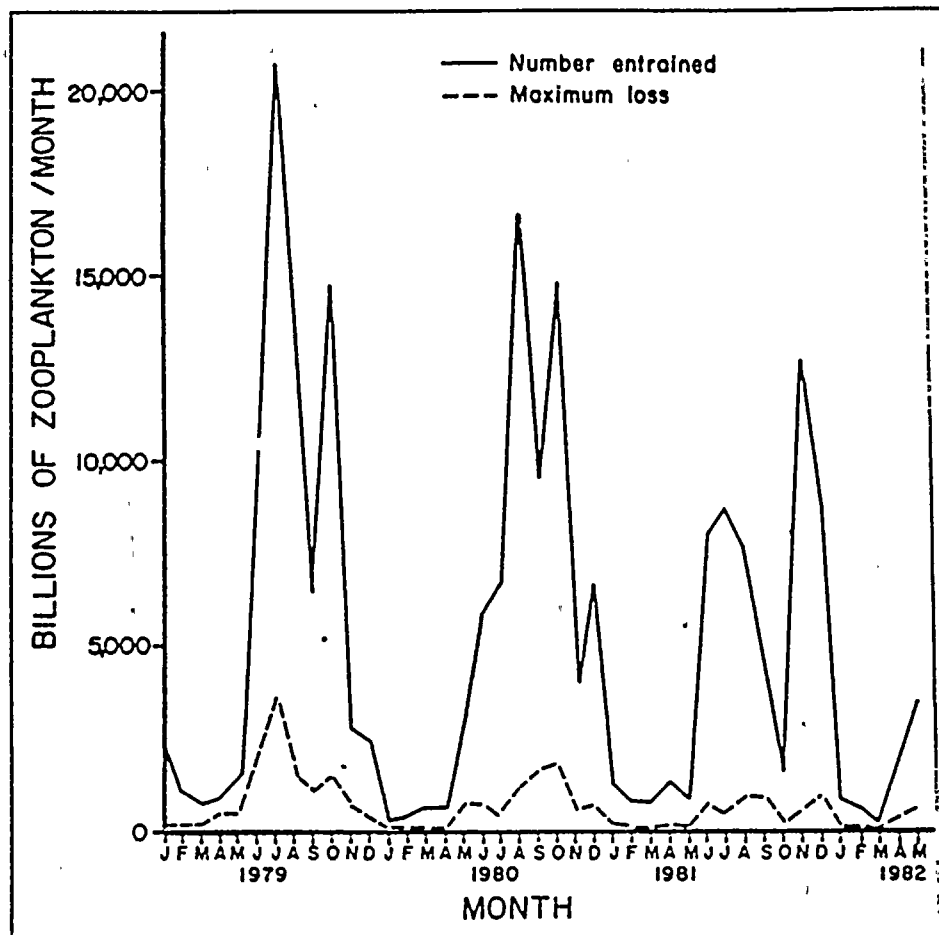


FIGURE 19. Estimates of numbers of entrained zooplankton and maximum losses from January 1979 to May 1982.

ity in zooplankton community structure generally was similar during the preoperational and operational years (Evans 1975, and Evans et al. 1978, 1980, 1982, 1986B). Zooplankton community structure ranged seasonally from a spring community dominated by calanoid and cyclopoid copepods to a summer and autumn community dominated by cladocerans and copepods. Spatial patterns in community structure also remained similar during the preoperational and operational years; water depth was the major factor related to spatial gradients in zooplankton community structure through much of the field season. (Evans et al. 1978, 1980, 1982, 1986B).

Although there was little evidence of gross spatial alterations in zooplankton community structure in the vicinity of the thermal plume based on the lake survey study design (Evans et al. 1986B), alterations were detected in specially-designed

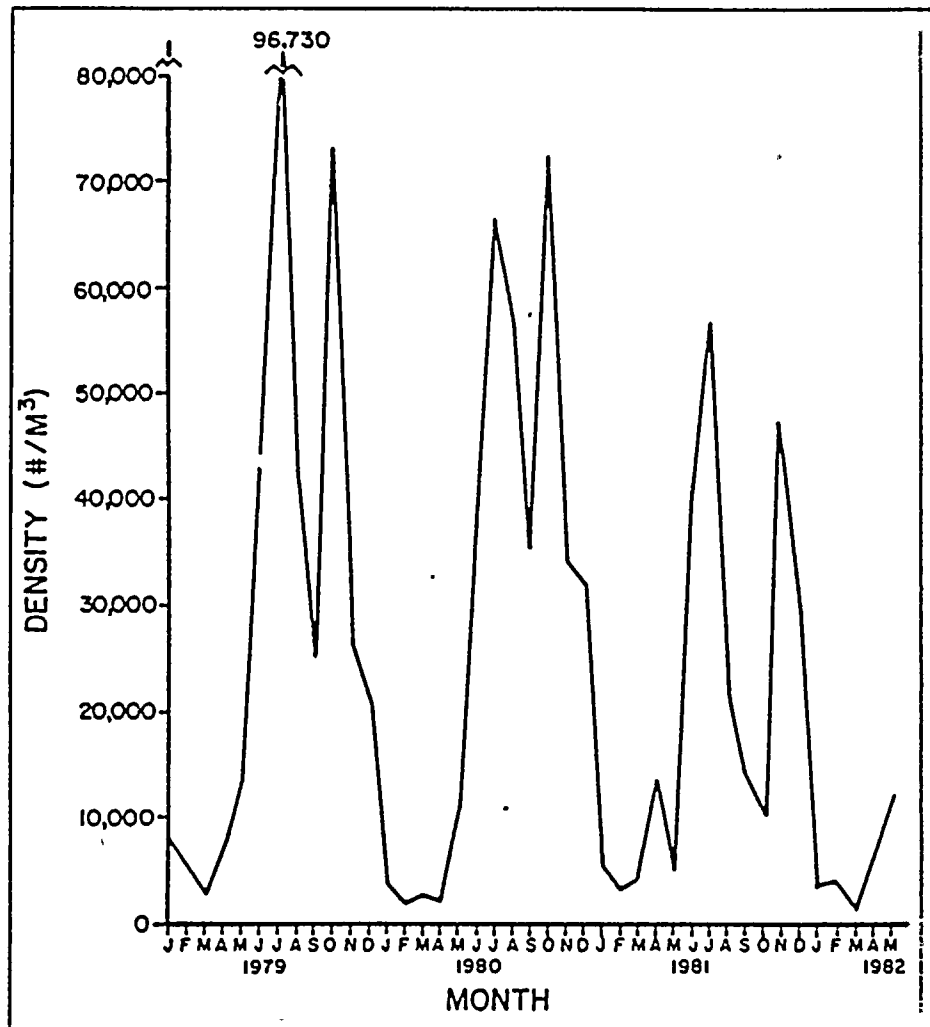


FIGURE 20. Total zooplankton densities (number/m³) in monthly entrainment samples taken from the intake forebay. There are no data for June 1979.

studies. Detailed thermal plume mapping did provide some evidence of elevated zooplankton abundances at the 1-m depth stratum in the immediate vicinity of the plant (Evans 1981). Such increased densities appeared to be due to the vertical displacement of deep-living zooplankton over the discharge jets. In addition, some localized enrichment of zooplankton could have occurred as the plant intakes withdrew deep water from the 9-m depth region. Later studies showed that the intake tends to withdraw water from the sediment-water interface (Evans and Flath 1984).

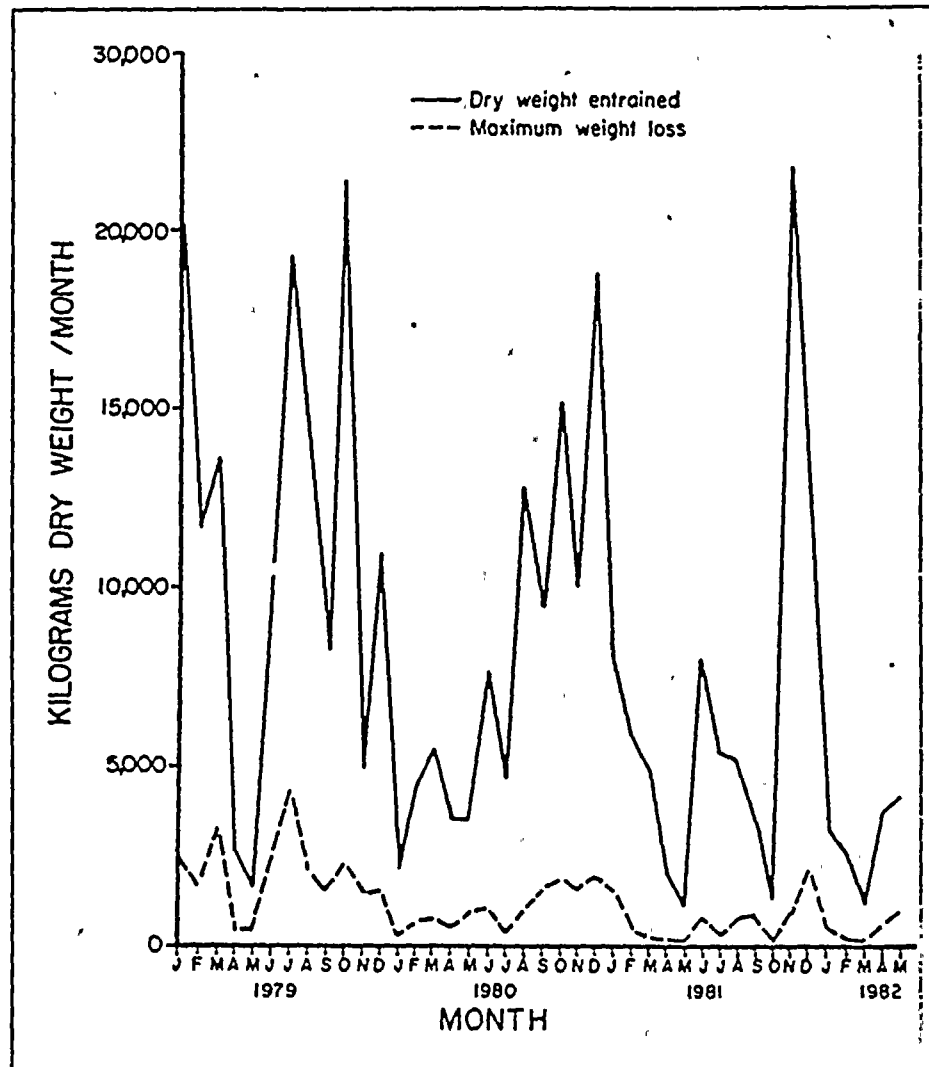


FIGURE 21. Monthly estimates of the dry weight of zooplankton entrained and the maximum weight lost from January 1979 to May 1982.

One important feature of the power plant's operating characteristics is the rapid dilution of condenser-passed water which occurs over the discharge jets. This not only dissipates thermal energy, but results in the dilution of those zooplankton killed during plant passage. Our best estimate of average zooplankton mortality of zooplankton as a consequence of plant passage is approximately 2%. Heated discharge water with its small percentage of dead zooplankton is rapidly diluted in the lake to

approximately 30% of its original concentration (assuming an in-plant temperature increase of 10C° and a thermal elevation of 3C° over the discharge jets: Evans et al. 1982, 1986B). Intense vertical mixing over the discharge jets prevents this small percentage ($30\% \times 2\% = 0.6\%$) of zooplankton from settling from the water column. Even if these zooplankton were to settle immediately from the water, a 0.6% loss over the discharge jets could not be detected. We have determined that the average coefficient of variation between stations in the inshore region is 39% (Evans and Sell 1983).

Comparison of the 4 years (1971-1974) of preoperational data with the 8 years (1975-1982) of operational data indicated that there were many statistically significant differences in zooplankton abundances between the two time periods (Evans et al. 1986B). These differences were not readily apparent by examining the time series plots of taxa abundances but were evident when comparisons were made across months.

During April, total zooplankton abundances in the inshore plume zone were significantly higher in the operational period (1975-1982) than the preoperational period (1971-1974). Nauplii were the major taxon accounting for this increase. However, *Limnocalanus macrurus* and adult *Diaptomus* spp. also were significantly more abundant in the inshore plume zone in the operational period. Conversely, immature cyclopoid copepods were less abundant in the operational period (Evans et al. 1986B). There was no evidence that these long-term changes in zooplankton abundances were related to any adverse effect of power plant operation. Similar magnitudes of change were observed in the inshore and middle plume zones as in the north and south control zones.

Total zooplankton standing stocks in the inshore plume zone were similar during the July preoperational (1971-1974) and operational (1975-1981) cruises. The greatest changes were associated with *Diaptomus* spp. copepodites and *Asplanchna* spp. which were less abundant in the operational period and with *Daphnia* spp. which were more abundant. These operational differences did not follow a consistent time trend. Rather, there were times in the operational period when zooplankton abundances were much higher (or lower) than observed during the preoperational period (Evans et al. 1986B). There was no evidence that these preoperational and operational differences were related to power plant operation. Similar magnitudes of change were observed in the inshore and middle plume zones as in the north and south control zones.

During October, total zooplankton occurred in similar preoperational (1972-1974) and operational (1975-1981) abundances in the inshore plume zone. Nauplii and *Bosmina longirostris* were more abundant in the operational period while adult *Cyclops* spp. and *Daphnia* spp. were less abundant (Evans et al. 1986B). These preoperational-operational differences did not appear to be due to plant operation as similar magnitudes of change were observed in the plume zones as in the north and south control zones.

Overall, there was no strong evidence that power plant operation had any adverse impact (spatial or temporal) on zooplankton community structure in the thermal plume region (Evans et al. 1986B). Preoperational-operational differences in abundance which did occur apparently were associated with lakewide changes in fish and zooplankton communities which first became evident in 1978 (Evans et al. 1980) but did not become pronounced until 1982-1984 (Evans and Jude 1986, Evans 1986 Scavia et al. 1986).

A large body of literature accumulated over the 1960s and early 1970s document

the effects of thermal discharges on zooplankton communities in a number of fresh, brackish, and marine habitats. In general, few effects were detected and thus thermal pollution no longer is an active area of research although site-specific monitoring studies continue where required.

Power plant operation can adversely affect zooplankton communities in a limited number of situations. The most obvious occurs when a large fraction of zooplankton is killed by plant passage and then discharged into a relatively small volume of water. Such adverse effects have been detected in rivers and cooling ponds (Zhitenjowa and Nikanorow 1972, Davies and Jensen 1974, Carpenter et al. 1974).

Power plant operation may affect zooplankton communities if ambient waters are significantly heated over long time periods (days at least). Such effects have been detected in relatively small water bodies including experimental enclosures, an open area created in an ice-covered lake by a thermal discharge, and small ponds (McMahon and Docherty 1975, Lanner and Pejler 1973, Patalas 1970). However, many researchers have failed to detect any obvious changes in zooplankton community structure in cooling ponds and small lakes (Heinle 1969, Whitehouse 1971, Mathur et al. 1980). No significant effects have been detected on large water bodies such as Lake Michigan (Industrial Bio-Test Laboratories, Inc. 1974a, b, 1975; Texas Instruments Inc. 1975; Evans et al. 1978, 1982, 1986B).

Our results suggest that plant passage is lethal to small percentage of zooplankton which pass through the condenser cooling system of the power plant. The mean 0-hour mortality difference for total zooplankton in Unit 1 and Unit 2 discharges averaged less than 3% (Evans et al. 1986A, 1986B). These relatively small mean mortality differences are similar to those observed at other power plants on Lake Michigan (Industrial Bio-Test Laboratories, Inc. 1974a; Wetzel 1975; Limnetrics, Inc. 1975, 1976). Furthermore, there was no evidence that delayed mortality effects were significant, either over a 6- or 24-hour period (Evans et al. 1986A, 1986B). Similarly, Davies et al. (1976) did not observe a significant long-term (2 weeks) increase in zooplankton mortalities.

There was strong evidence that calanoid copepods were most sensitive to plant passage. *Diaptomus*, *Eurytemora*, and *Limnocalanus* had significantly higher discharge water than intake water mortalities. Cladocerans were intermediate in sensitivity to plant passage, with *Eubosmina coregoni* and *Daphnia* the most sensitive taxa within this group. In contrast, cyclopoid copepods were relatively resistant to damage inflicted as a result of plant passage (Evans et al. 1986A, 1986B). Similar differential sensitivity among zooplankton taxa to plant passage has been observed in other studies (Davies and Jensen 1974, Wetzel 1975).

There was little evidence that thermal stresses were the major cause of zooplankton mortality. Mortalities did not increase in summer when discharge water temperatures were highest and zooplankton most likely to experience thermal regimes at (or above) their upper critical thermal limit. Nor did zooplankton mortalities increase in winter when cold-adapted species were prevalent (Evans et al. 1986A, 1986B).

Mortalities in discharge waters were not significantly correlated with ΔT (Evans et al. 1982). Nor were differences between discharge and intake mortalities significantly correlated with discharge water temperature (Evans et al. 1982). We concluded that the moderate thermal elevations ($\Delta T < 12^\circ\text{C}$) and short-term (minutes) exposures to discharge water temperatures below 35°C were not the major source of stress to zooplankton which pass through the condenser cooling system of the power plant.

Mechanical stresses probably were the major source of zooplankton mortality at

the power plant (Evans et al. 1986A, 1986B). Studies conducted at the Nanticoke Generating Station showed clear quantitative evidence that zooplankton were physically damaged by plant passage, although mortalities were not determined (Standke and Monroe 1981). Other studies have implicated mechanical stresses as the significant source of mortality both to zooplankton (Carpenter et al. 1974) and fish (Marcy 1973) at power plants where organisms are exposed to moderate thermal elevations for short periods of time.

It is unlikely that losses of zooplankton due to plant passage would have an adverse effect on water or sediment quality in the vicinity of the plant. This hypothesis is based on a comparison of the estimated depositional rate of plant-killed zooplankton to the natural depositional rate (Evans et al. 1978). Assuming a maximum two-unit operational loss of 4,482 kg dry wt. zooplankton/month over a depositional area of 2.2 km² gives an estimated maximum depositional rate of 67.5 mg/m²/day. This compares with an estimated natural depositional rate of 2,800 to 4,000 mg/m²/day or 0.07 to 0.10 mg/cm²/year (Evans et al. 1978), i.e., 1.7 to 3.4% of the estimated natural rate. Thus, deposition of zooplankton killed by plant passage appears to be small, adding little to the natural deposition which occurs in the area.

The results of an epibenthic and benthic study conducted in July 1980 also suggest that the Donald C. Cook Nuclear Plant had a moderate effect on sediment-water interface communities (Evans 1984). Macrobenthic taxa abundances were only slightly different (amphipods, chironomids) or the same (oligochaetes) in plume and control regions. Although microcrustacean taxa abundances exhibited greater magnitudes of difference between plume and control regions, the affected area was relatively small and located within a few hundred meters of the intake and discharge structures. Scuba divers reported that sediment troughs were clean or contained only small amounts of detritus. Thus, detrital matter probably does not persist in the shallow (<10 m) nearshore area for long periods of time. Copepods and cladocerans (microcrustaceans) with their high rate of population increase (Allen 1976) relative to that of the macrobenthos, may be especially well adapted to utilizing short-term accumulations of detrital material and, for this reason, were most strongly affected by plant operation.

Condenser-passed water is discharged at a high velocity through subsurface discharge jets resulting in a rapid loss of temperature and the formation of a floating plume. Thus, epibenthic and benthic taxa living within a few hundred meters of the discharge jets experienced a thermal regime similar to that of organisms living more than 1,000 m to north and south of the plant. It is probably only within a few tens of meters of the discharge jets where plume temperatures are more than 2C° above ambient and where the sediment is scoured by the high current velocities that epibenthic and benthic community structure is strongly altered by plant operation.

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EFFECTS OF THE DONALD C. COOK NUCLEAR PLANT ON MACROZOOBENTHOS IN NEARSHORE SOUTHEASTERN LAKE MICHIGAN

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INTRODUCTION

There have been numerous studies of the macrozoobenthos that populate sediments near Lake Michigan power plants (e.g., Bailly Power Plant 1976; Consumers Power Company 1975; Geers 1977; Limnetics Inc. 1974; Rains and Clevenger 1975; Winnell and Jude 1979, 1980, 1981, 1982; Wisconsin Electrical Power Co. and Wisconsin-Michigan EPC 1974). Most reports, however, are primarily descriptive of the lake bottom components prior to or following power plant construction, and few studies have actually addressed ecological relationships to power plants. In this chapter, we present data on the relationships between power plant operations and macrozoobenthos populations. The first relationship is impingement, or the actual capture of macrozoobenthos on the traveling screens of the power plant water intake. Second is entrainment, the capture of meroplankton, of normally-bottom-dwelling organisms, which actively enter the water column and which are passively pulled through the power plant's cooling system. Third is the relationship of the natural patchiness of the infauna, compared with potential changes brought on by power plant construction-operation. Fourth is the fauna of the riprap, an artificial reef constructed to protect the power plant intake/discharge structures. Where possible, we compare results at the Donald C. Cook Nuclear Plant with other power plant studies.

SITE DESCRIPTION AND GENERAL METHODS

Donald C. Cook is a nuclear power plant located on the southeastern shore of Lake Michigan (40°03' west, 86°40' north), 23 km south of Benton Harbor, Michigan. Plant construction began in the early 1970s with the first of two units becoming operational in 1975. The second unit began operation in early 1978. Cooling water is drawn through intake structures which extend lakeward to a depth of 9 m. Water is discharged at a point 6 m deep in the lake. Combined, both units draw a maximum of 6,200 m³ water per minute. Intake and discharge structures are protected by a covering of 0.1 to 1.0 m riprap.

Lake Michigan in the vicinity of the power plant has been described in several reports (see Seibel and Ayers 1974). In summary, the lake bottom gently falls away from the shoreline at a slope of approximately 3 m per km. Summer thermoclines are in the general range of 20 to 30 m deep. Down to 30 m, bottom sediments are composed of well sorted coarse to fine sands with only patchy silt and organic accumulations. Prevailing currents are south to north at a speed up to 30 cm sec.

The southern basin of Lake Michigan has a more or less counterclockwise current which potentially can carry pollutants toward D. C. Cook Nuclear Plant from the Milwaukee, Chicago, and northwestern Indiana regions, and it has been noted by Rains (1971), Mozley and Garcia (1972), and Mozley and Howmiller (1977) that the nearshore areas of southern and southeastern Lake Michigan may be regarded as slightly enriched or mesotrophic. This region of the lake has been described as featureless with no major rivers entering and little to no consolidated substrates.

SAMPLING SITES

Ecological studies near the D. C. Cook Nuclear Plant began in the mid-1960s, but systematic sampling for zoobenthos did not occur until 1970. With knowledge of zoobenthic distributions and densities gained from 1970 and 1971 seasons, several modifications were made in sampling procedures. Table 1 gives a summary of sampling design changes and changes in numbers of stations, sampling gear, and numbers of samples per station. For a variety of reasons, the number of samples actually examined often was lower than the maximum that could be collected, particularly in the early years of the survey. The two primary sampling grids (major and minor—see below) for radial systematic design are depicted in Figures 1 and 2. A standard Ponar grab sampler was used on surveys from July 1970 through April 1972 and then replaced by the triplex Ponar (Mozley and Chapelsky 1973) on all further surveys.

From July 1970 through April 1972, a fixed station, radial systematic survey design was conducted in April, July, and November using 45–46 stations. A single Ponar grab was taken at each station. Beginning in July 1972 the months of April, July, and October were designated as "major survey" months, and a systematic random sampling design was initiated which lasted through April 1974. This design was based on rerandomizing locations within depth and regional strata such that 126 samples were collected from 36 stations using a combination of full and one-third Ponar grabs. The major surveys reverted to a fixed station, radial systematic design in July 1974, and the number of stations was reduced to 30 (Fig. 1). Stations <8 m deep generally were sampled by four full Ponar grabs, stations >8 m deep by two one-third Ponar grabs. Because benthic density was much lower, more samples and greater volume were required in the nearshore sandy habitat. The latter design was continued throughout the remainder of the study period for the major survey months.

All other months samples beginning in May 1972 (May through December) were termed "minor surveys" and had reduced sampling effort. Nine to 13 stations were sampled each month (Fig. 2). All minor surveys, including 1972 and 1973, used the radial systematic design with fixed stations. The number of stations varied with power plant construction and the addition of four stations prior to 1975. Though variable, three or five full Ponar grabs were taken at stations <8 m deep and three one-third grabs at stations >8 m deep.

Sampling in all months, but particularly April, November, and December, was subject to weather conditions. Collections at a few shallow or deep stations were omitted due either to weather conditions or plant construction.

For purposes of analysis, stations in Figures 1, 2, and the systematic random surveys have been grouped by regions and by depth ranges. The regions were designed primarily to give areal trends which would aid in detection of effects of

TABLE 1. Monthly benthic sampling effort near the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1970-1978.

Year	Month	Survey Type	Survey Design ¹	Number of Stations Sampled (Replicates)			Ponar Size Conversion Factor		Total Number of Samples Monthly	
				Maximum	Actual		≤ 8 m	> 8 m	Maximum	Actual
					≤ 8 m	> 8 m				
1970	Jul	—	RS	46	9(1)	24(1)	9.07	9.07	46	31
	Nov	—	RS	46	6(1)	30(1)	9.07	9.07	46	36
1971	Apr	—	RS	46	12(1)	29(1)	18.13	18.13	46	41
	Jul	—	RS	46	10(1)	29(1)	18.13	18.13	46	39
	Nov	—	RS	46	9(1)	27(1)	18.13	18.13	46	36
1972	Apr	—	RS	45	11(1)	34(1)	18.13	18.13	45	45
	May	Minor	RS	9	4(3)	5(3)	20.4	60.6	27	27
	Jun	Minor	RS	9	4(3)	5(3)	20.4	60.6	27	27
	Jul	Major	SR	36	9(5)	27(3)	20.4	60.6	126	126
	Aug ²	Minor	RS	9	3(3)	5(3)	20.4	60.6	27	24
	Sep	Minor	RS	9	3(3)	5(3)	20.4	60.6	27	24
	Oct	Major	SR	36	9(5)	27(3)	20.4	60.6	126	126
	Nov	Minor	RS	9	3(3)	5(3)	20.4	60.6	27	24
	Apr	Major	SR	36	9(5)	26(3)	20.4	60.6	126	123
	May ³	Minor	RS	9	3(3)	7(3)	60.6	60.6	35	32
1973	Jun ⁴	Minor	RS	11	3(3)	7(3)	60.6	60.6	35	32
	Jul	Major	SR	36	9(5)	27(3)	20.4	60.6	126	126
	Aug	Minor	RS	11	3(3)	7(3)	60.6	60.6	35	32
	Sep	Minor	RS	11	3(5)	7(3)	20.4	60.6	41	36
	Oct	Major	SR	36	8(5)	27(3)	20.4	60.6	126	119
	Apr	Major	SR	36	9(5)	27(3)	20.4	60.6	126	126
	May ⁵	Minor	RS	13	6(5)	7(3)	20.4	60.6	51	50
1974	Jun	Minor	RS	13	6(5)	7(3)	20.4	60.6	51	51
	Jul	Major	RS	30	10(5)	20(2)	60.6	60.6	80	80
	Aug	Minor	RS	13	6(5)	7(3)	20.4	60.6	51	51
	Sep	Minor	RS	13	5(5)	7(3)	20.4	60.6	51	46
	Oct	Major	RS	30	10(4)	20(2)	60.6	60.6	80	80
	Apr	Major	RS	30	10(4)	20(2)	60.6	60.6	80	80
	May	Minor	RS	13	5(5)	7(3)	20.4	20.4	51	46
1975	Jun	Minor	RS	13	6(5)	7(3)	20.4	60.6	51	51
	Jul	Major	RS	30	10(4)	20(2)	60.6	60.6	80	80
	Aug	Minor	RS	13	6(5)	7(3)	20.4	60.6	51	50

TABLE 1. Continued

Year	Month	Survey Type	Survey Design ¹	Number of Stations Sampled (Replicates)			Ponar Size Conversion Factor		Total Number of Samples Monthly	
				Maximum	Actual		≤ 8 m	> 8 m	Maximum	Actual
					≤ 8 m	> 8 m				
1976	Sep	Minor	RS	13	6(5)	7(3)	20.4	60.6	51	51
	Oct	Major	RS	30	10(4)	20(2)	60.6	60.6	80	80
	Dec	Minor	RS	12	5(5)	6(3)	20.4	60.6	46	46
	Apr	Major	RS	30*	10(4)	20(2)	60.6	60.6	85	85
	May	Minor	RS	12	5(5)	7(3)	20.4	60.4	46	46
	Jun	Minor	RS	12	5(5)	7(3)	20.4	60.6	46	46
	Jul	Major	RS	30	10(4)	19(2)	60.6	60.6	80	78
	Aug	Minor	RS	12	5(5)	7(3)	20.4	60.6	46	46
	Sep	Minor	RS	12	5(5)	7(3)	20.4	60.6	46	42
	Oct	Major	RS	30	10(4)	19(2)	60.6	60.6	80	78
1977	Apr	Major	RS	30	10(4)	20(2)	60.6	60.6	80	80
	May	Minor	RS	12	5(5)	7(3)	20.4	60.6	46	46
	Jun	Minor	RS	12	5(5)	7(3)	20.4	60.6	46	45
	Jul	Major	RS	30	10(4)	20(2)	60.6	60.6	80	80
	Aug	Minor	RS	12	5(5)	7(3)	20.4	60.6	46	46
	Sep	Minor	RS	12	5(5)	7(3)	20.4	60.6	46	46
	Oct	Major	RS	30	10(4)	20(2)	60.6	60.6	80	79
	Nov	Minor	RS	12	5(5)	7(3)	20.4	60.6	46	46
	Dec	Minor	RS	12	5(5)	7(3)	20.4	60.6	46	46
1978	Apr	Major	RS	30	10(4)	20(2)	60.6	60.6	80	80
	May	Minor	RS	12	5(5)	7(3)	20.4	60.6	46	46
	Jun	Minor	RS	12	5(5)	7(3)	20.4	60.6	46	46
	Jul	Major	RS	30	10(4)	20(2)	20.4	60.6	80	80
	Aug	Minor	RS	12	5(5)	7(3)	20.4	60.6	46	46
	Sep	Minor	RS	12	5(5)	7(3)	20.4	60.6	46	46
	Oct	Major	RS	30	10(4)	20(2)	60.6	60.6	80	80
	Nov	Minor	RS	12	5(5)	7(3)	20.4	60.6	46	46
Total									3575	3473

¹Survey design: RS = radial systematic, SR = systematic random.²Station DC-1 deleted due to construction activities during Aug, Sep, and Nov 1973 and May, Jun, Aug, Sep 1974. DC-1 sampled regularly beginning in May 1974.³Five replicates collected at DC-0 (conversion factor = 20.4) during May, Jun, Aug, Sep 1973 through 1975 and 1976. Station deleted beginning May 1976.⁴Stations NDC 7-5 and SDC 7-5 added to minor surveys beginning June 1973.⁵Stations NDC 7-1 and SDC 7-1 added to minor surveys beginning May 1974.⁶The addition of DC-0 (5 replicates) increased station total from 30 to 31 and total number of samples from 80 to 85 in April 1976.

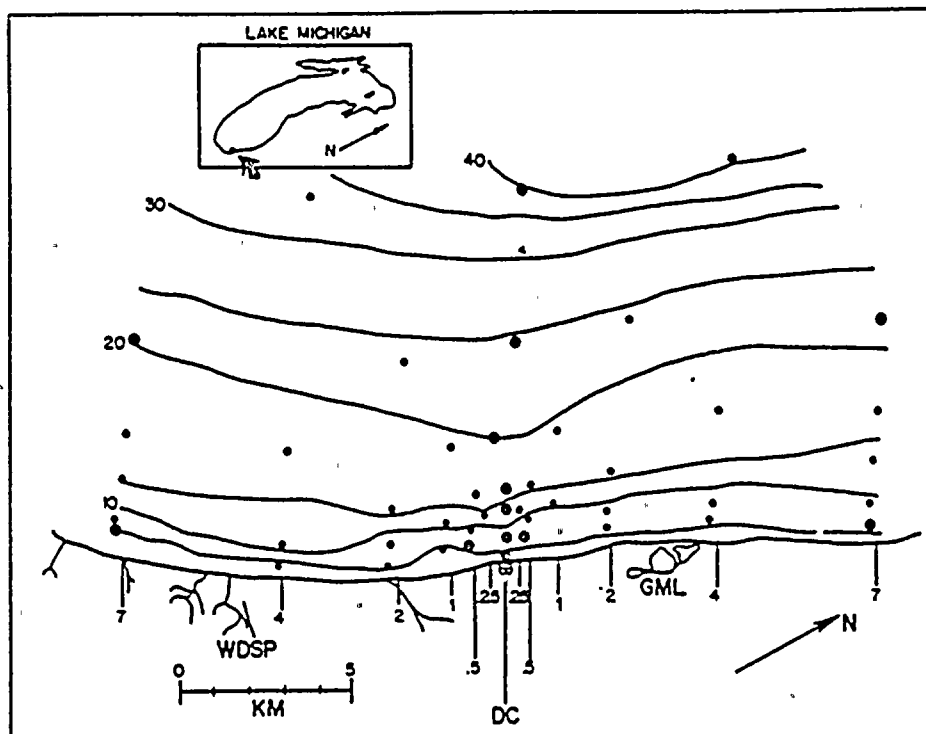


FIGURE 1. Grid of stations used in benthic sampling near the D. C. Cook power plant, southeastern Lake Michigan, for major survey months and zones. Solid large circles represent the Inner or Central region, open circles represent the North and South Outer regions. Locations of Grand Marais Lakes (GML) and Warren Dunes State Park (WDSP) are indicated.

power plant operations on the zoobenthos. The inner, or central, region was centered on the intake-discharge structures and extended 1.6 m to either side of the power plant and 11.3 km offshore. Two outer regions, one north and one south, extended from 1.6 to 11.3 km from the center of the plant and 11.3 km offshore. Thus, the entire sampling grid covered 22.5 km along the southeastern shoreline and outward 11.3 km, an area of 254.25 km².

Originally, all stations were combined into four depth ranges: 0–8 m, 8–16 m, 16–24 m, and >24 m (represented by zones in Fig. 1). The ranges were based on density and population trends discovered in early years of sampling. To examine distributions in greater detail here, the total range of depths has been redivided into 4-m depth intervals. The shallowest interval is 0–4 m; however, very few samples were collected <3 m deep. Also, very few stations in the grid were >32 m deep, and all greater than this depth have been combined.

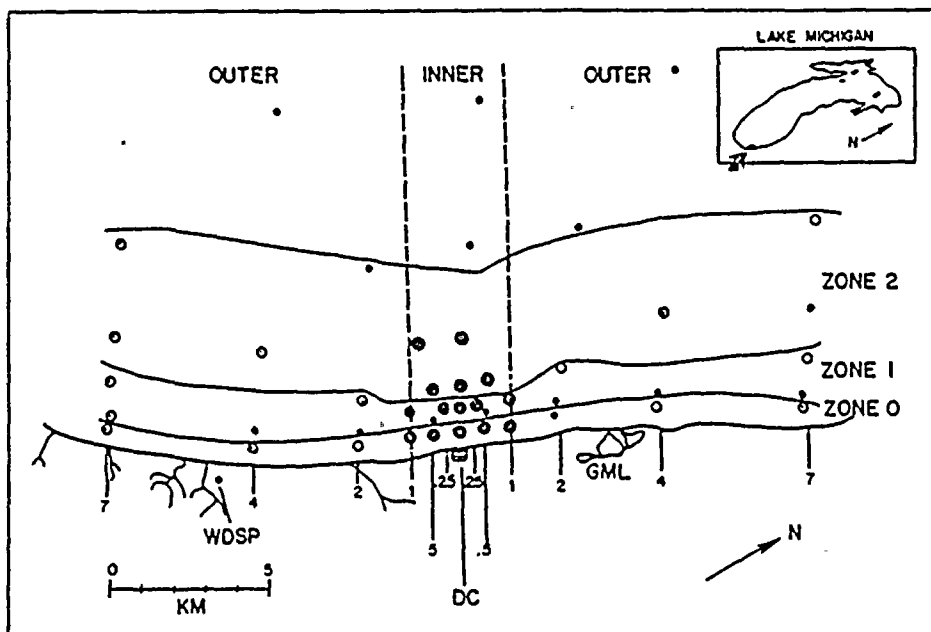


FIGURE 2. Grid of stations used in benthic sampling near the D. C. Cook power plant, southeastern Lake Michigan, for minor survey months with 5-m contour depth intervals. Solid large circles represent minor survey sites. Locations of Grand Marais Lakes (GML) and Warren Dunes State Park (WDSP) are indicated.

SAMPLE COLLECTION

Lake samples were collected using one of the Great Lakes Research Division research vessels, primarily the R/V *Mysis*. Sampling with the standard or triplex Ponar was aided by a power winch. Once on board, grab contents were washed into library trays where substrate types were visually determined and recorded. Samples were elutriated on board (Fig. 3), separating organisms and finer organic debris from gravels, sands, silts, and clays. Except in adjunct studies, sieve screen mesh used with the elutriator was 0.500 mm. Samples were concentrated in 0.5-liter Mason jars and preserved with 4% formalin buffered with calcium carbonate.

Samples were sorted to major taxa in the laboratory. All survey data were entered, checked, and stored on magnetic tape. The Michigan Interactive Data Analysis System (MIDAS) package programs were used to perform analyses on the AMDAHL 470V/8 computer at The University of Michigan.

SUBSTRATE ANALYSIS

Substrate or sediment types were determined visually for each grab. Size classes and terminology basically follow the Krumbein scale (see Cummins 1962) with some

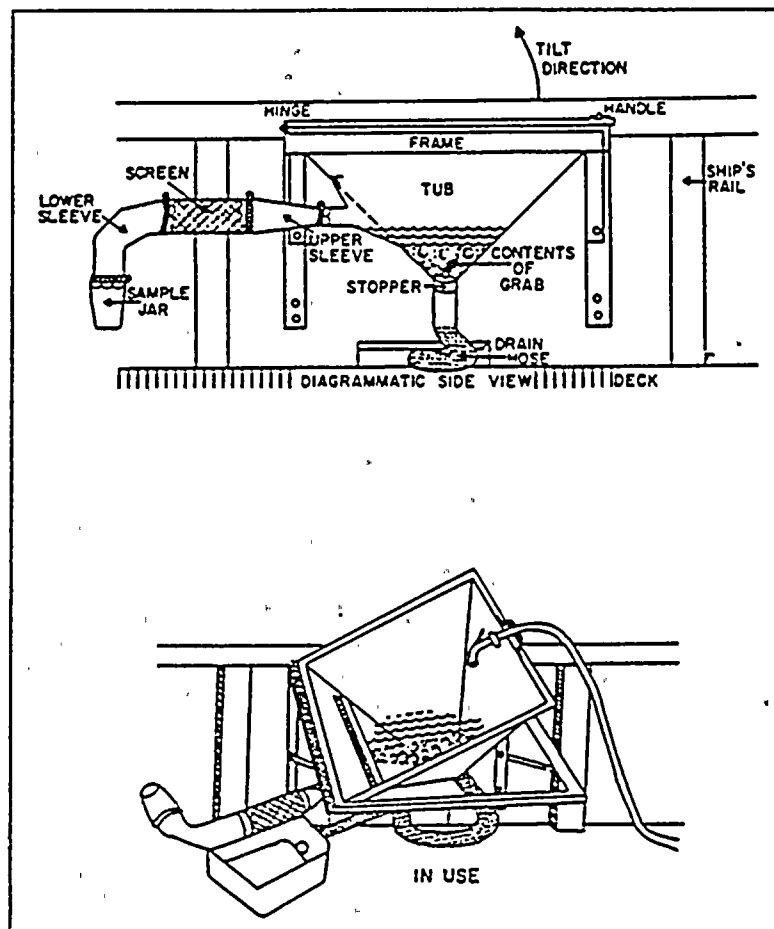


FIGURE 3. Elutriator used aboard ship for processing benthic samples at D. C. Cook power plant, southeastern Lake Michigan (redrawn from Mozley 1975).

modifications which were appropriate for the types of sediments found in the survey area. Table 2 is a rough approximation between the substrate categories used here and the Krumbein scale. Because the bottom gradually slopes away from the shoreline and the prevailing wind is from the northwest, most grabs from <30 deep contained a high portion of sands. Pure gravels, silts, and clays were uncommon in the area except in highly localized, often transitory patches.

TABLE 2. *Substrate categories used in this study and comparisons with the Krumbein scale and terminology. Table presents rough comparisons only as D. C. Cook Nuclear Plant sediments were determined visually.*

Substrate Categories Used at D. C. Cook Nuclear Plant	Krumbein Scale	
	Grade	φ
Clean substrates		
Gravel	Gravel	-1.0 to -2.0
Coarse sand	Very coarse to coarse sand*	1.0 to -1.0
Medium sand	Medium sand	2.0 to 1.0
Fine sand	Fine sand	3.0 to 2.0
Very fine sand	Very fine sand	4.0 to 3.0
Mixed fine substrates		
Silty sand		
Clayey sand	Silts	8.0 to 5.0
Sandy silt	Clays	14.0 to 8.0
Sandy clay		
Loam		

ADJUNCT STUDIES

Much of the contents of this report was derived from extensive lake surveys. To add to these data, results of several adjunct studies have been relied on. With exception of the Whole Lake Survey described below, the Adjunct or Special studies were conducted as part of zoobenthic studies of the D. C. Cook Nuclear Plant. Entrainment and impingement studies were a part of the regular sampling scheme, particularly once the power plant began operations. Riprap colonization, mero-plankton, mesh size retention, and the RSW studies were designed to answer specific questions and were of short duration. Nonetheless, each adjunct study has added significantly to understanding principals governing zoobenthos life histories and population dynamics. These studies are summarized below and expanded, where appropriate, in the body of the chapter.

ENTRAINMENT

Although actual operations of the D. C. Cook Nuclear Plant did not begin until January 1975, testing of the intake and discharge system began in mid 1974. First entrainment samples collected in 1974 from the intake forebay were to examine heterogeneity of organisms in the intake waters. The regular program of entrainment sampling began in January 1975. Samples were taken in the intake (two replicates) and discharge (one sample) forebays weekly from June through August and twice monthly from September through May. Each "sample" was taken over a 24-h span divided into four consecutive periods corresponding to midnight to sunrise, sunrise to noon, noon to sunset, and sunset to midnight.

In years 1975 through 1978, 1,147 entrainment collections were made, 733 from the intake forebay and 414 from the discharge forebay. In sample collection, water was continuously pumped from 5.5 m deep in the forebay and passed through a 0.35-mm mesh net. The total volume of water passing through the net was deter-

mined by an attached flow meter (see Jude et al. 1979 for system details). Once collected, samples were preserved in 4% buffered formalin. In this chapter only the intake forebay was considered.

IMPINGEMENT

Traveling screens with a 1.9-cm openings are used at the Cook Plant to remove large objects, preventing clogging of condenser tubes. While the screens are much too large to retain most zoobenthic invertebrates, crayfish were regularly impinged. All objects greater than 1.9 cm were washed into a large basket where crayfish and fish were sorted from debris. Crayfish were placed in bags, frozen, then returned to Ann Arbor for identification, sexing, and length and weight measurements. Collection methods were not entirely reliable. First, specimens with a diameter less than 2.0 mm were not usually collected, and second, many of the specimens were partially decomposed before freezing.

From 1975 through 1978 (power plant operational years), samples were available for 614 dates. Monthly and annual impingement were calculated for density and biomass. Numbers and weights of crayfish in semi-monthly periods were multiplied by the number of days in that period, then divided by the number of 24-h samples.

RIPRAP COLONIZATION

For protection, intake and discharge structures of the D. C. Cook Nuclear Plant were covered with a limestone riprap shield. Individual stones in the riprap ranged from 0.1 to 1.0 m in diameter. In this, an artificial reef was constructed in a region of the lake where there was little to no other consolidated substrate. Effects on zoobenthos were examined in a comparative study of colonization on the riprap and on a natural rocky shoal at Waugoshance Point (northern Lake Michigan) (Lauritsen 1979).

Concrete artificial substrates were placed at both the Cook Plant riprap and Waugoshance Point sites and allowed to colonize for a period of several weeks. Though there were similarities in fauna, the substrates at the D. C. Cook Nuclear Plant were dominated by predators and at Waugoshance Point by filter feeders. The extent of *Cladophora* on the Cook Plant riprap was assumed to be the most important factor controlling the types and numbers of species. Though *Cladophora* created favorable habitat in summer, numbers of organisms decreased sharply with its disappearance in late fall. Artificial substrates at the Cook Plant were colonized by a species assemblage different from that collected in the Ponar grabs from adjacent unconsolidated substrates. These data reveal the significance of riprap in allowing colonization of species not normally associated with this part of the lake.

MEROPLANKTON

Many more mobile zoobenthic invertebrates are known to leave the sediments and enter the water column either on a diurnal basis or at specific points in their life cycles. To examine the extent at which zoobenthos in the vicinity of the Cook Plant become pelagic and thereby are subject to entrainment, plankton tows were made

monthly, April through October 1973 (Wiley and Mozley 1978). A flowmeter equipped, 0.50-m diameter, 0.36-mm mesh vertically from just above the bottom to the surface. A fourth tow was made vertically from just above the bottom to the surface. Two 6-m and two 9-m deep stations were sampled each month. On each date at each station, four sets of tows were made conforming to afternoon, dusk, night, and morning. Net tows were supplemented by series of 7-cm ID hand cores. Over the 7-month period, 71 zoobenthic species were collected in the tows. Pelagic species or meroplankton at the 6-m stations reflected the composition of the infauna. At the 9-m stations the dominant meroplankton differed somewhat from the zoobenthos. All data from this study was summarized in Wiley and Mozley (1978) who proposed that meroplankton serve to maintain zoobenthic populations in more frequently disturbed shallow sediments.

MESH SIZE RETENTION

Life histories and distributions presented in this study are based on organisms retained by 0.50-mm mesh sieve. Because many small species and early fall history stages will pass through the 0.50-mm mesh net used in the regular field surveys near the D. C. Cook Nuclear Plant, two studies of mesh size retention were conducted in 1974. The first was to determine the significance of switching from the 0.50-mm mesh to 0.35-mm mesh for regular lake surveys. The second was a comparison between zoobenthos and meroplankton using a 0.15-mm mesh sieve.

In the 0.35-mm mesh test, materials passing through the 0.50-mm net were resieved through a 0.35-mm net. At shallower stations dominated by Chironomidae, 80% of the total animals passed through the larger mesh, and many rarer and smaller species were not retained at all. At deeper stations characterized by a *Pontoporeia* assemblage, 80% of the organisms were collected on a 0.50-mm screen, and at least a portion of all species was collected (see Mozley 1975 for specific details).

To compare zoobenthic abundance with numbers of organisms collected in a special drift survey, 23-34 July 1974, diver-collected cores were taken at 6-m and 9-m deep stations. Entire contents of the cores were preserved and decanted through a 0.15-mm mesh screen. Total densities of organisms collected by the 0.15-mm sieve were several fold greater than estimates normally obtained from Ponar grabs sieved through a 0.55-mm mesh. Additionally, greater numbers of several smaller species were observed in comparison with the meroplankton studies (0.35-mm mesh screens). Several species (particularly Naididae) were collected for the first time including some which appeared in relatively high numbers (see Mozley 1975 for specific data).

RSW SURVEY

The single sediment parameter measured in the regular lake survey was a visual categorization of primary sediment types. The RSW survey was designed to assess several additional sediment conditions with a goal of relating both physical and chemical parameters to zoobenthos densities and distributions. In September 1977, 112 stations were sampled along seven transects perpendicular to the shoreline the D. C. Cook Nuclear Plant. Zoobenthos were sorted from sediments using a 0.20-mm mesh sieve. Corresponding sediment samples were analyzed for precise particle

size distribution, dissolved and sediment bound heavy metals, organic carbon, and other selected parameters. As with the 0.15-mm sieve experiments, smaller zoobenthic species were captured in numbers very different from the regular lake surveys. RSW data also provided information on species occurring deeper in the lake, as the transect stations extended to 57.6 m deep.

WHOLE LAKE SURVEYS

The Whole Lake Surveys were sponsored by the U.S. Environmental Protection Agency (USEPA) and included two major studies of zoobenthos distributions throughout Lake Michigan. During August 1975, 273 single Shipek grabs were collected by the Canada Centre for Inland Waters aboard the C. S. S. *Limnos*. Stations were the intersections of a 14 by 14 km Universal Transverse Mercator grid over most of the lake. A 7 by 7 km grid was used in Green Bay and in the northeastern corner of the lake. Zoobenthos were field sorted using a 0.250-mm mesh sieve and preserved with 4% buffered formalin.

As the Canadian survey's grid precluded many stations less than 30 m deep along the eastern and western shores of Lake Michigan, a second survey (funded by the USEPA) was conducted by the Great Lakes Research Division during July and August 1977 aboard the R/V *Roger Simons*. Samples were collected by the methods above along 21 transects approximately 60 km apart around the shoreline of the lake (with the exception of Green Bay). At each transect, replicate samples were taken at depths of 9, 18, 36, and 54 m, a total of 83 sites in addition to the 273 above.

Distribution and density data from the Whole Lake Surveys (listed in the text as CCIW, EPA, or GLRD unpubl.) are used as comparative materials for Cook Plant data.

IMPINGEMENT OF ZOOBENTHOS

Crayfish (Decapoda) were the only benthic animals in the vicinity of the Donald C. Cook Nuclear Plant large enough to be impinged on the 1.9-cm-square-mesh traveling screens. As a part of the fish-impingement studies at the Donald C. Cook Nuclear Plant, impinged crayfish were routinely saved and frozen in plastic bags. All available specimens were identified, sexed, measured to the nearest 1.0 cm, and weighed to the nearest 0.1 g. Because individuals < 2 cm in some dimension were not retained on the screens, collection methods were not wholly reliable, and because some specimens were partially decomposed, impingement estimates are, at best, rough estimates. The annual number of crayfish impinged was determined by multiplying the number of crayfish actually counted by the ratio of 365 days divided by the number of days in the year for which crayfish were collected. Biomass was based on the length/weight (wet-weight) relationship from Winnell (1984a):

$$Wt = 0.015L^{3.26},$$

where Wt is wet-weight biomass in milligrams and L is total length in centimeters. Estimation of biomass from this relationship was based on 1-cm length classes with no distinction between sexes or reproductive status (Table 3).

Very little qualitative and no quantitative crayfish sampling was conducted in the

TABLE 3. Estimates of the number, biomass, and mean weight of crayfish impinged at the Donald C. Cook Nuclear Plant each year from 1975 to 1978.

Year	Number Impinged	Biomass (kg) Impinged	Mean Weight per Individual (g)
1975	16,151	111	6.9
1976	15,495	95	6.2
1977	10,985	77	7.0
1978	7,625	43	5.9

open lake or at the power plant's riprap structures. Some specimens were reported by divers to move about on the open sand bottom but not at any great distance from the riprap (Dorr and Jude 1986). Therefore, it is assumed that the crayfish population occurred primarily on the riprap and that impingement reflected abundance and seasonal variation in the riprap population. With a few exceptions, all crayfish impinged were *Orconectes propinquus* (Winnell 1984a).

Impingement density and biomass were highest in 1975 ($n = 16,151$; $wt = 111$ kg) and lowest in 1978 ($n = 7,265$; $wt = 43$ kg). A consistent decrease in both annual density and biomass of crayfish impinged from 1975 to 1978 may be the result of the population decreasing (Dorr and Jude 1986) but becoming more stable over time. At present there is no reason to assume that Donald C. Cook Nuclear Plant operations have any detrimental effects on crayfish populations because the plant provides the only suitable habitat in the survey area.

ENTRAINED ZOOBENTHOS

Entrainment of zoobenthos is of concern for a variety of reasons. First is the well-being of the organisms entrained. It is generally assumed, but rarely tested, that exposure to the mechanical forces or to the elevated water temperatures either directly kills the animals or affects their physiology (e.g., reproductive success). Second, if a significant portion of the Lake population is entrained, it may directly affect the population structure. For example, if the taxon is particularly important in lake processes or the food chain, entrainment may directly or indirectly affect distribution and abundance of other organisms (e.g., a fish which might primarily feed upon that organism).

Although a considerable number of taxa were irregularly collected in entrainment samples (most of the species listed in the Macrozoobenthos Ecology section), only macrocrustaceans present in significant numbers and important enough in the food chain were analyzed further. They included *Pontoporeia hoyi*, *Gammarus* spp., *Hyalella azteca*, *Mysis relicta*, and *Asellus* spp.

The average daily entrainment (ADE) rates for a specific month were estimated from forebay intake samples by

$$ADE = (X_d)(P_d) + (X_n)(P_n)$$

where X_d and X_n are the mean numbers of individuals per m^3 in daylight and nighttime periods, respectively, and P_d and P_n are the maximum number of m^3 of water pumped by the power plant in daylight and nighttime periods, respectively.

The maximum volume of water pumped during daytime and nighttime hours was the product of the percentage of light or dark hours attributable to the 15th day of each specific month times the maximum pumping rate at the Donald C. Cook Nuclear Plant ($8.928 \times 10^5 \text{ m}^3 \text{ day}^{-1}$). Daily entrainment rates for each specific month were weighted by the percentage of light and dark periods specific to each respective month and then adjusted to account for significant diel activity. Subsequent monthly entrainment rates were determined by multiplying the weighted daily entrainment specific to each month by the number of days comprising respective months. Annual entrainment estimates were the sum of individual monthly entrainment rates for each taxon. The biomass entrained was calculated for the daily, monthly, and annual totals as above by substituting average biomass m^{-3} in place of average no. m^{-3} . Annual entrainment density and biomass values were calculated for January through December and April through December, then the data were used to calculate entrainment from suspected areas of maximum impact.

Pontoporeia hoyi

Entrainment of *P. hoyi* averaged 0.0484 individuals m^{-3} . Diel activity was pronounced. Nightly entrainment (0.0770 individuals m^{-3}) was four times that during daylight hours (0.0191 individuals m^{-3}), with the greatest levels from sunset to midnight (0.0891 individuals m^{-3}) and the least from noon to sunset (0.0146 individuals m^{-3}) (Table 4).

Monthly distribution of *P. hoyi* among size classes and reproductive states for entrained specimens paralleled similar observations in lake bottom populations (Table 5). Regardless of the close agreement between entrainment and lake samples, one must be cautious when making comparisons. Alongshore, offshore, or onshore lake currents, as well as seasonal migration, can carry organisms to the intake, and transient organisms may be different from those residing on the lake bottom in the vicinity of the power plant (Winnell and White 1984). An additional factor is the warm-water discharge which may attract freely swimming *Pontoporeia* and other crustaceans. At the Donald C. Cook Nuclear Plant, the 4-year population structure, based on entrainment samples, was in close agreement with that derived from lake samples, and it suggested that the majority of entrained individuals originated from

TABLE 4. Average number of entrained malacostracans and total benthos (no. m^{-3}) during selected time periods within a generalized 24-hr time period and over all samples from the intake forebay of the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1975-1978.

Taxon	Time Period						All Samples
	Midnight to Sunrise	Sunrise to Noon	Noon to Sunset	Sunset to Midnight	Day	Night	
<i>Pontoporeia hoyi</i>	0.0654	0.0238	0.0146	0.0891	0.0191	0.0770	0.0484
<i>Gammarus</i> spp.	0.0101	0.0213	0.0117	0.0129	0.0164	0.0114	0.0139
<i>Hyalella azteca</i>	0.0006	0.0009	0.0004	0.0002	0.0007	0.0004	0.0005
<i>Mysis relicta</i>	0.0445	0.0089	0.0027	0.0444	0.0058	0.0443	0.0253
<i>Asellus</i> spp.	0.0011	0.0010	0.0017	0.0031	0.0015	0.0021	0.0018
Total benthos	0.6643	0.4514	0.3216	0.7920	0.3865	0.7261	0.5591
n	184	179	181	185	361	371	733

TABLE 5. Average monthly entrainment densities (no. m^{-3}) for malacostracans, for *Pontoporeia hoyi*, of different size classes and reproductive stages, and for total benthos in the intake forebay of the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1975-1978 (n = number of samples).

Taxon	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
<i>Pontoporeia hoyi</i>	0.2955	0.0077	0.0028	0.0636	0.0135	0.0212	0.0239	0.0363	0.0195	0.0093	0.0065	0.1897
< 3 mm	0.0004	0.000	0.0000	0.0592	0.0126	0.0102	0.0042	0.0040	0.0012	0.0013	0.0000	0.0000
3-5 mm	0.0006	0.0010	0.0000	0.0003	0.0009	0.0103	0.0164	0.0286	0.0167	0.0055	0.0017	0.0093
5-7 mm	0.0006	0.0009	0.0000	0.0010	0.0000	0.0002	0.0024	0.0036	0.0016	0.0023	0.0048	0.0116
≥ 7 mm	0.0000	0.0000	0.0000	0.0014	0.0000	0.0000	0.0004	0.0000	0.0000	0.0003	0.0000	0.0004
Gravid	0.0120	0.0017	0.0004	0.0007	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0004
Spent	0.0038	0.0000	0.0024	0.0010	0.0000	0.0000	0.0000	0.0001	0.0000	0.0000	0.0000	0.0000
Males	0.2779	0.0041	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.1577
<i>Gammarus</i> spp.	0.0033	0.0082	0.0024	0.0025	0.0062	0.0058	0.0098	0.0117	0.0176	0.0824	0.0098	0.0152
<i>Hyalella azteca</i>	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0006	0.0004	0.0007	0.0022	0.0006	0.0017
<i>Mysis relicta</i>	0.2286	0.0272	0.0303	0.0060	0.0089	0.0212	0.0103	0.0124	0.0359	0.0116	0.0084	0.0205
<i>Asellus</i> spp.	0.0043	0.0070	0.0020	0.0000	0.0000	0.0020	0.0026	0.0014	0.0018	0.0015	0.0000	0.0011
Total benthos	0.6077	0.4593	0.1349	0.2443	0.2833	0.6748	0.6504	0.4780	0.7145	1.6657	0.2555	0.4178
n	29	34	52	55	42	96	102	117	55	49	39	63

the 8-20-m depth contour. Thus, the total entrained density and biomass of *P. hoyi*, compared with that derived from lake samples, is assumed to be a best estimate of the effect of entrainment in the immediate vicinity of the power plant.

Based on daytime and nighttime entrainment rates for each month (Table 6), annual entrainment was 1.97×10^8 individuals or 11% of all entrained zoobenthos. However, nearly 75% (1.46×10^8) of the total *Pontoporeia* entrainment occurred in December and January during the peak nearshore reproductive period (Table 7) (Winnell and White 1984); the majority (90%) of the December-January specimens was males. Minor monthly peaks occurred in April (recently released young) and in August (primarily females).

The only data available for direct comparisons were from the Bailly Power Plant (1976) at the southern tip of Lake Michigan. The entrainment estimate reported by the Bailly Power Plant (3.6×10^6 individuals) was calculated incorrectly and should have been 1.18×10^7 *P. hoyi* yr⁻¹ (see Winnell 1984a for further explanations). Given that entrainment estimates from the Donald C. Cook Nuclear Plant were more detailed than those from Bailly, direct comparisons were tenuous; however, the Donald C. Cook Nuclear Plant was estimated to entrain 17 times more *P. hoyi* than the Bailly Power Plant. Because the Cook plant draws 1,700 times more water per year (based on maximum flow with no recirculation) than the Bailly Power Plant, the relative Donald C. Cook Nuclear Plant entrainment rate was approximately 0.01% that of Bailly. The Cook plant entrained 0.06 *P. hoyi* m⁻³ [based on $(1.97 \times 10^8 \text{ yr}^{-1}) / (3.26 \times 10^9 \text{ m}^3 \text{ yr}^{-1})$] to be comparable with the Bailly Plant estimates; the Bailly Power Plant entrained 6.05 *P. hoyi* m⁻³ [$(1.18 \times 10^7 \text{ P. hoyi yr}^{-1}) / (1.95 \times 10^6 \text{ m}^3 \text{ yr}^{-1})$]. As *P. hoyi* entrainment at the Cook Plant rarely exceeded the two to three organisms m⁻³ level, it seems likely that entrainment at the Bailly Plant either was severely overestimated or a large difference exists between power plant designs which influenced the entrainment rates. We do not know enough about the lake densities at the Bailly Plant to make further comparisons.

Mid-point, minimum, and maximum *P. hoyi* biomass entrained were estimated for an average day in each month by applying the length/ash-free dry weight equation of Johnson and Brinkhurst (1971) to mid-point, minimum, and maximum body lengths in each size class. Based on mid-point biomass, 297 kg of *P. hoyi* were entrained annually at the Cook Plant (Table 7). Biomass entrained during December and January constituted 92% of the annual entrainment. When minimum and maximum *P. hoyi* size classes were used in the calculations, annually entrained biomass ranged from a low of 285 kg yr⁻¹ to a high of 315 kg yr⁻¹, respectively. The small range of difference was largely due to predominance of males 7 mm long.

Application of annually entrained *P. hoyi* density and biomass to lake bottom density and biomass estimates required two assumptions. The first assumption pertained to the area considered to be most severely impacted by entrainment. Clearly, this area is not definitive in an absolute sense. For a worst-case analysis, a relatively small area in the vicinity of the intake structures may be hypothesized to supply all the entrainment. Here, two alternative ideas are considered in determining the maximum impact of entrainment:

1. The DC1-DC2 area is in the central portion of the survey areas defined by the shallowest recorded depth of Station DC1 (3. m) and the deepest recorded depth of Station DC2 (14.6 m); DC1-DC2 comprises an area of 2.56 km² (1.6 km × 1.6 km) (Figs. 1, 2).

2. The DC1-DC3 area is in the central portion of the survey area defined by the

TABLE 6. Average diel density (no. m^{-3}) for *Pontoporeia hoyi* entrained during each month in the intake forebay of the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1975-1978 (n = number of samples, D = Day, N = Night).

Parameter	D	N	D	N	D	N	D	N	D	N	D	N
	Jan		Feb		Mar		Apr		May		Jun	
Density	0.1996	0.3734	0.0000	0.0155	0.0055	0.0000	0.0662	0.0612	0.0085	0.0171	0.0038	0.0387
Minimum Weight	0.3023	0.7302	0.0000	0.0252	0.0063	0.0000	0.0033	0.0104	0.0001	0.0006	0.0005	0.0047
Mid-point Weight	0.3023	0.7311	0.0000	0.0265	0.0063	0.0000	0.0081	0.0170	0.0007	0.0020	0.0011	0.0107
Maximum Weight	0.3023	0.7324	0.0000	0.0284	0.0063	0.0000	0.0081	0.0285	0.0019	0.0048	0.0020	0.0205
n	13	16	17	17	26	26	27	28	18	24	48	48
	Jul		Aug		Sep		Oct		Nov		Dec	
Density	0.0015	0.0454	0.0042	0.0689	0.0024	0.0372	0.0052	0.0137	0.0000	0.0128	0.0492	0.3259
Minimum Weight	0.0001	0.0128	0.0011	0.0185	0.0003	0.0101	0.0011	0.0064	0.0000	0.0090	0.0910	0.5533
Mid-point Weight	0.0003	0.0240	0.0020	0.0360	0.0008	0.0199	0.0021	0.0110	0.0000	0.0146	0.0934	0.5676
Maximum Weight	0.0007	0.0404	0.0032	0.0615	0.0014	0.0340	0.0037	0.0174	0.0000	0.0221	0.0965	0.5867
n	50	52	59	58	28	27	25	24	19	19	31	32

shallowest recorded depth of Station DC1 (3.7 m) and the deepest recorded depth of Station DC3 (18.7 m); DC1-DC3 comprises an area of 5.12 km² (1.6 km × 3.2 km) (Figs. 1, 2).

A second implicit assumption was that *P. hoyi* density and biomass estimates based on lake bottom samples from April through December would not have been substantially altered if January through March lake bottom samples had been collected. Because an absence of alteration in annual lake bottom estimates cannot be confidently assumed, entrainment-impact analyses were performed both on a January through December and an April through December basis.

Average lake bottom density and biomass of *P. hoyi* in the DC1-DC2 area were 413 organisms m⁻² and 0.00015 kg m⁻² (mid-point biomass estimate) (Table 8). The lake bottom area of DC1-DC2 required to supply 1.97×10^6 individuals yr⁻¹ based on the January through December entrainment density estimate was 0.48 km² or

TABLE 7. Average daily density (no. m⁻² × 10³) and biomass (kg) of *Pontoporeia hoyi* entrained during each month, and average monthly density and biomass entrained in the intake forebay of the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1975-1978.

Month	Density (× 10 ³)		Daily			Monthly		
	Daily	Monthly	Min	Mid	Max	Min	Mid	Max
January	2.73	84.59	5.34	5.35	5.35	165.62	165.78	165.99
February	0.08	2.17	0.13	0.13	0.14	3.53	3.71	3.98
March	0.02	0.75	0.03	0.03	0.03	0.86	0.86	0.86
April	0.57	17.14	0.06	0.11	0.20	1.72	3.22	6.08
May	0.11	3.28	<0.01	0.01	0.03	0.08	0.33	0.84
June	0.15	4.38	0.02	0.04	0.08	0.54	1.22	2.32
July	0.16	4.92	0.04	0.08	0.14	1.33	2.52	4.27
August	0.28	8.65	0.07	0.14	0.25	2.32	4.49	7.63
September	0.17	5.10	0.04	0.09	0.15	1.34	2.66	4.55
October	0.09	2.70	0.04	0.06	0.10	1.09	1.91	3.06
November	0.07	2.04	0.05	0.08	0.12	1.43	2.32	3.52
December	1.98	61.48	3.39	3.48	3.60	105.15	107.88	111.50
Total	—	197.20	—	—	—	285.02	296.89	314.60

TABLE 8. Mean density (no. m⁻²) and biomass based on minimum, mid-point, and maximum size class intervals (gm m⁻²) for *Pontoporeia hoyi* in the DC1-DC2 and DC1-DC3 areas located directly in front of the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1975-1978.

Parameter	Area	
	DC1-DC2	DC1-DC3
Density	413	757
Km ²	2.56	5.12
Minimum biomass	0.0788	0.1513
Mid-point biomass	0.1525	0.2860
Maximum biomass	0.2675	0.4083

19% of the available area at DC1-DC2 (Table 9). Utilizing the April through December entrainment estimate (1.10×10^4 individuals yr^{-1}), 0.27 km^2 or 11% of the DC1-DC2 area would be required. The percentage of DC1-DC2 required to supply the annually entrained biomass of *P. hoyi* (January-December) ranged from as low as 1.07 km^2 (42% of available area) to as high as 3.99 km^2 (156% of available area) with a mid-point estimate of 1.95 km^2 (76% of available area). Similarly, using the April-December estimate (123 kg yr^{-1}), equivalent lake bottom areas required ranged from a low of 0.43 km^2 (17% of available area) to a high of 1.892 km^2 (71% of available area) with a mid-point estimate of 0.81 km^2 (32% of available area) (Table 9).

The obvious differences between the DC1-DC2 and DC1-DC3 alternatives are the greater *P. hoyi* density ($757 \text{ organisms m}^{-2}$), biomass ($0.00029 \text{ kg m}^{-2}$), and area (5.12 km^2) of the latter. Consequently, the total percent area required would be considerably smaller when compared with DC1-DC2. Using the January through December annual entrainment density estimate, the area of DC1-DC3 required was 0.26 km^2 or 5%. Using the April through December estimate, only 0.14 km^2 or 3% of the area was required. Similarly, the benthic biomass of *P. hoyi* required for a January-December estimate ranged from 0.57 to 2.08 km^2 (11% to 41% of available area), with a mid-point estimate of 1.03 km^2 or 20% of the available area. Using April through December data, 0.23 to 0.95 km^2 were required (5% to 19% of available area), but the mid-point was 0.43 km^2 or 8% (Table 9).

From the two examples above, it is evident that the area of the lake bottom impacted by entrainment is dependent upon the bottom area considered and the organism density. We posed the DC1-DC2 areas as a worst-case example. Clearly, decreasing the area believed to be affected to a smaller and more narrowly defined portion of just the DC1-DC2 areas, particularly in an on-shore direction, would suggest a much greater impact. However, as one proceeds toward shore, *P. hoyi* densities are more variable and subject to the influence of migrational activities (Winnell and White 1984). Consequently, considerable latitude should be given when scaling the potentially impacted area. Conversely, expanding the potential area to too large a base would underestimate effects of entrainment. For example, if the total survey area (254 km^2) and average *P. hoyi* abundance ($2,209 \text{ organisms m}^{-2}$)

TABLE 9. Potential impact of entrainment on *Pontoporeia hoyi* populations occurring in the DC1-DC2 and DC1-DC3 areas. Impact expressed in terms of the number of km^2 required to supply the densities and biomasses entrained on a January through December basis and on an April through December basis at the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1975-1978.

Parameter	January-December				April-December			
	DC1-DC2		DC1-DC3		DC1-DC2		DC1-DC3	
	Km^2	Areal %	Km^2	Areal %	Km^2	Areal %	Km^2	Areal %
Density	0.48	18.8	0.26	5.1	0.27	10.5	0.14	2.7
Minimum biomass	1.07	41.8	0.57	11.1	0.43	16.8	0.23	4.5
Mid-point biomass	1.05	76.2	1.03	20.1	0.81	31.6	0.43	8.4
Maximum biomass	3.00	155.0	2.08	40.6	1.82	71.1	0.05	18.6

were used, the area of Lake Michigan needed to supply 1.97×10^4 individuals would be only 0.09 km² or 0.04% of the *P. hoyi* population occurring in the entire survey area. Considering either just the DC1-DC2 area or the whole survey area, we have yet to demonstrate that entrainment has affected lake bottom populations in the vicinity of the Cook Plant, at least during the period of operational years studied.

Gammarus, Hyallella, and Asellus

As with the crayfish, populations of these taxa may be directly linked to the riprap structures; therefore, it was difficult to ascribe an effect on any lake-bottom population. Entrainment of *Gammarus* averaged 0.0139 organisms m⁻³ (Table 4), which was 2.7% of the total benthos entrained. Average entrainment was less than 0.0100 organisms m⁻³ during most months. Increased abundance was observed for late summer and early fall with a peak in October (Table 5). Daytime entrainment (0.0164 organisms m⁻³) was slightly greater than nighttime (0.0114 organisms m⁻³) with the greatest proportion occurring between sunrise and noon (0.0213 organisms m⁻³) (Table 4).

Hyalella azteca accounted for only 0.1% of total entrained benthos (Table 10), averaging 0.0005 organisms m⁻³ (Table 4). Greatest monthly abundance was in October and December with no specimens being collected between January and June (Table 5). Diel activity was similar to that of *Gammarus*, with greatest densities occurring between sunrise and noon (0.0009 organisms m⁻³).

Of the two *Asellus* present in the survey area, only *A. intermedius* was identified from entrainment. Entrained *Asellus* averaged only 0.0018 organisms m⁻³ (Table 4) or 0.4% of total entrainment (Table 10). Diel patterns were not as evident as with the amphipods. Seasonally, greatest densities were in January and February with remaining months either zero or slightly above.

TABLE 10. Average daily entrainment rate (no. m⁻³ × 10⁴) during each month, and average monthly entrainment rate (no. m⁻³ × 10⁴) for malacostracans (other than *Pontoporeia hoyi*) and for total benthos in the intake forebay of the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1975-1978.

Month	<i>Mysis relicta</i>		<i>Gammarus</i> spp.		<i>Hyalella azteca</i>		<i>Asellus</i> spp.		Total Benthos	
	Daily	Monthly	Daily	Monthly	Daily	Monthly	Daily	Monthly	Daily	Monthly
January	2.17	67.32	0.03	0.96	0.00	0.00	0.04	1.23	5.63	174.50
February	0.27	7.50	0.08	2.13	0.00	0.00	0.06	1.78	4.42	123.77
March	0.05	1.62	0.02	0.65	0.00	0.00	0.02	0.57	1.21	37.49
April	0.05	1.47	0.02	0.68	0.00	0.00	0.00	0.00	2.10	62.90
May	0.05	1.67	0.04	1.34	0.00	0.00	0.00	0.00	2.26	70.17
June	0.15	4.44	0.05	1.42	0.00	0.00	0.02	0.48	5.08	152.41
July	0.07	2.12	0.09	2.70	<0.01	0.23	0.02	0.57	5.23	162.01
August	0.09	2.89	0.10	3.25	<0.01	0.12	0.01	0.37	4.01	124.40
September	0.31	9.36	0.16	4.67	<0.01	0.20	0.02	0.48	6.29	188.65
October	0.11	3.46	0.69	21.36	0.02	0.60	0.01	0.42	14.72	456.44
November	0.09	2.71	0.10	2.98	<0.01	0.19	0.00	0.00	2.09	62.61
December	0.21	6.42	0.14	4.21	<0.01	0.44	0.01	0.32	4.14	128.25
Total	—	110.98	—	46.35	—	1.77	—	6.23	—	1,743.60

Mysis relicta

The average density of mysids entrained ($0.0253 \text{ organisms m}^{-3}$) (Table 2) accounted for 6.4% of the total annual entrainment. By far, the month of largest densities was January ($0.2286 \text{ organisms m}^{-3}$) (Table 3). Annually, 1.11×10^8 mysids were entrained at the Cook plant. The January estimate accounted for 61% of the average annual total (Table 10). A secondary peak occurred in September ($0.0359 \text{ organisms m}^{-3}$). High densities of *Mysis* in the vicinity of the water intake during September may be related to changing water temperature during upwelling and to decreasing light periods.

Grossnickle and Morgan (1979) observed greatest abundances of *Mysis relicta* ($1,019 \text{ organisms m}^{-3}$) at 30–50 m during September in Lake Michigan and postulated that rapidly declining mysid densities between October and December were due to onshore migration. While no difference in entrained mysid densities was found between October and December, high entrainment in January was believed to reflect active onshore migration. This hypothesis is further supported by size-dependent horizontal migrations of mysids shown for Lake Tahoe (Morgan 1982).

Mysis relicta displayed the most striking diel activity of all malacostracans encountered. Average daytime entrainment density was $0.0058 \text{ organisms m}^{-3}$, while during the nighttime it was $0.0443 \text{ organisms m}^{-3}$ (Table 4).

If Morgan and Beeton's (1976) average of $188 \text{ organisms m}^{-2}$ is used as an approximate estimate of the lake bottom mysid density in the offshore area near the survey site, an area equivalent to 0.59 km^2 would be required to supply 1.11×10^8 individuals yr^{-1} . Because the profundal region of southeastern Lake Michigan is enormous by comparison with 0.59 km^2 , it was concluded that there was no apparent effect on the *Mysis* population due to entrainment losses. One factor not measured which should be considered in any future study is lost reproductive potential due to entrainment of males during January and its subsequent effect on the total population structure.

When compared with the Bailly Power Plant (1976) average yearly mysid estimate (1.27×10^8 organisms), the plant entrained considerably more mysids (1.11×10^8 organisms year^{-1}). Given the rate of maximum yearly flow of water through both power plants and basing comparisons on the quotient of number entrained per year divided by number of cubic meters utilized per year by each plant, the Cook Plant entrainment rate ($0.034 \text{ organisms m}^{-3}$) was approximately half that of the Bailly Plant ($0.065 \text{ organisms m}^{-3}$). The significance of these differences is unknown, although it is clear that they are the same order of magnitude.

REGIONAL ZOOBENTHIC COMMUNITY STRUCTURE AND CHANGES IN POPULATIONS FOLLOWING POWER PLANT OPERATION

South, Central, and North Regional Distribution Patterns

For these analyses, stations were grouped by region and by depth range. Regions were designed primarily to give areal trends which would aid in detection of power plant effects on macrozoobenthos. The central (or inner) region was centered 1.6 km on either side of the intake structures. The south and north regions (outer) extended

11.3 km to each side of the intakes and 11.3 km offshore, a total area of 254 km². Individual stations were assigned to 4-m depth intervals beginning at the shoreline.

To understand changes that may have been brought about by power plant effects (i.e., the heated effluent), populations present and factors affecting their natural distributions must first be understood. In addition to the information derived from the substratum descriptions recorded for each Ponar grab in this study, Rossman and Seibel (1977) have detailed sediment patterns at Cook Plant. In general, at depths <27 m sediments are continually subject to wave generated resuspension (this depth approximates the normal limits of the summer thermocline, Mozley and Winnell 1975) and to along-shore currents. Nearshore substrata south of the power plant are patchier, finer grained, and have a higher level of total carbon than either the central or north regions. Greater patchiness in the south region may result from small stream input and in part from greater diversity in local current and wave patterns (also see Mozley 1975). The central region contains the artificial reef created by the riprap covering the intake/discharge structures of the central region. The reef may alter wave and sedimentation patterns but the extent of actual alterations is not known. Further discussion on the biology of the reef follows in this chapter.

Our data are sufficient at the species level to discuss regional distributions of the dominant oligochaetes, amphipods, chironomids, and pisidiids. Total oligochaete density decreased slightly from the south (4,325 individuals m⁻²) to the north (3,268 individuals m⁻²). Much of this variation was in the family Tubificidae which decreased from 3,002 individuals m⁻² to 1,986 individuals m⁻² (Table 11) (LaDronka 1984). Regional differences were most evident in the 12-32-m depth zones and were associated with slightly greater amounts of carbon, silts, and clays (see chapter on sediments this book). Species having significantly greater densities in the south or central and south regions were *Aulodrilus plurisetus*, *Isochaetides freyi*, *Limnodrilus cervix*, *Limnodrilus hoffmeisteri*, *Limnodrilus spiralis*, *Potamothenix vejovskyi*, and *Quistadrilus multisetosus multisetosus*. Following the classification of Mozley and Howmiller (1977, also see Lauritsen et al. 1985), *A. plurisetus*, *I. freyi*, *P. moldaviensis*, and *P. vejovskyi* are associated with environmental Type II (mesotrophic) conditions; *L. hoffmeisteri* is Type III (saprophilic); and *L. cervix*, and *Q. m. multisetosus* are Type IV (saprobiontic). No species had increased abundance in a northerly direction, and only one (*Tubifex superiorenensis*, a Type I (oligotrophic species) was greater in the Central Region.

Relative densities of Naididae (Table 12) were clearly lower in the south region than in the north or central regions (LaDronka 1984). Naidids appeared to be unaffected by sediment or sedimentation parameters which may have produced higher abundances of other animals in the south, possibly because of the shallower distribution of naids. Five rare naidid species were unique to the central region (Table 12), which may have been a result of greater sampling effort. Further, the riprap structures may harbor species different from those found in sands and silts.

Densities of the lumbriculid, *Stylodrilus heringianus*, were greater in the south and north regions than in the central region (Table 11). This distribution may reflect assumed lower sedimentation rates in the middle of the study area (see sediments chapter, this book) which favor *Stylodrilus* over tubificids (White unpubl data).

Ten of the 20 most common chironomid taxa showed regional abundance patterns (Table 13). Four taxa, *Heterotrissocladius changi*, *Heterotrissocladius oliveri*, *Micropsectra* sp., and *Paracladopelma winnelli* were most abundant in the north region and least abundant in the central region. Each of these species has been classified as Type 0 (representative of oligotrophic conditions) by Winnell and

TABLE 11. Mean density (\bar{x}) (no. m⁻²), standard error (SE), and frequency of occurrence (%FO) of oligochaetes (excluding naidids) and total benthos at 1-48 m within each area near the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1970 through 1978. The number of observations for tubificid species (Ns) and for totals and *S. heringianus* (Nt) is given at the bottom.

Taxon	North ¹			Central ²			South ¹		
	x	SE	%FO	x	SE	%FO	x	SE	%FO
Total Tubificidae	1,986.2	177.6	75.4	2,588.4	136.9	76.1	3,002.3	267.6	78.6
<i>Aulodrilus americanus</i>	4.9	1.9	4.0	5.3	1.2	3.5	4.0	1.4	2.1
<i>Aulodrilus pigueti</i>	<0.1	<0.1	0.2	0.0	0.0	0.0	0.0	0.0	0.0
<i>Aulodrilus plurisetus</i>	36.3	20.7	6.9	117.0	22.3	11.1	204.9	76.2	10.7
<i>Ilyodrilus templetoni</i>	0.2	0.2	0.4	0.5	0.3	0.5	1.3	0.8	0.6
<i>Ilyodrilus freyi</i>	13.2	3.0	9.0	15.7	2.1	10.9	17.1	2.2	14.7
<i>Limnodrilus angustipenis</i>	6.2	1.6	5.0	7.7	1.2	5.9	7.5	1.4	8.2
<i>Limnodrilus cervix</i>	4.2	1.8	2.3	8.7	2.0	4.0	9.9	5.0	3.2
<i>Limnodrilus claparedeianus</i>	0.6	0.2	1.1	1.5	0.5	1.5	1.6	0.8	0.9
<i>Limnodrilus hoffmeisteri</i>	116.4	13.1	33.5	136.8	12.7	32.5	146.0	21.6	34.7
<i>Limnodrilus profundicola</i>	3.0	0.7	5.0	4.5	0.8	5.6	4.6	0.9	7.6
<i>Limnodrilus spiralis</i>	4.1	1.1	4.8	8.9	1.7	7.0	10.0	3.4	6.6
<i>Limnodrilus udekemianus</i>	1.7	0.6	2.1	0.8	0.3	1.0	0.4	0.3	0.3
<i>Potamothenix bedoti</i>	0.2	0.1	0.4	0.3	0.1	0.5	0.0	0.0	0.0
<i>Potamothenix hammoniensis</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.2
<i>Potamothenix moldaviensis</i>	39.4	5.4	23.7	31.5	3.6	22.5	42.2	5.1	27.1
<i>Potamothenix vejdoskyi</i>	262.6	37.3	32.3	204.9	23.9	26.2	381.8	53.0	33.0
<i>Quistadrilus multisetosis multisetosis</i>	2.4	1.1	2.3	13.5	3.4	4.4	13.3	3.8	5.4
<i>Quistadrilus multisetosis longidentus</i>	0.0	0.0	0.0	0.3	0.2	0.2	0.1	0.1	0.2
<i>Rhyacodrilus coccineus</i>	2.7	1.3	1.7	2.1	0.8	1.3	0.4	0.3	0.5
<i>Spirosperma nikolskyi</i>	0.0	0.0	0.0	<0.1	<0.1	0.1	<0.1	<0.1	<0.1
<i>Tubifex kessleri americanus</i>	0.0	0.0	0.0	<0.1	<0.1	0.1	0.0	0.0	0.0
<i>Tubifex superiorenensis</i>	1.3	0.7	1.5	0.9	0.4	0.8	0.0	0.0	0.0
<i>Tubifex tubifex</i>	5.3	1.5	4.6	14.1	5.0	5.3	7.6	2.6	4.1
Other tubificids	0.0	0.0	0.0	0.1	0.1	0.2	0.3	0.3	0.2
Immatures without hair chaetae	1,283.9	169.5	67.7	1,420.0	103.6	66.9	1,475.6	148.0	74.4
Immatures with hair chaetae	113.4	20.1	30.0	125.4	18.2	26.1	192.0	54.2	26.5
Total Etehytracidae	2.5	0.8	2.1	2.0	0.6	1.4	1.4	0.6	1.2
<i>Stylodrilus heringianus</i>	1,060.8	75.0	47.9	839.4	40.7	42.9	1,101.4	69.3	50.2
Total Oligochaeta	3,268.0	211.2	83.7	3,750.8	153.2	84.2	4,325.2	293.9	87.1
Total benthos	7,939.1	394.4	96.0	7,746.7	236.7	96.0	8,359.0	409.5	98.0

¹Ns = 523 Nt = 728

²Ns = 1,274 Nt = 1,900

³Ns = 634 Nt = 845

TABLE 12. Mean density (\bar{x}) (no. m⁻²), standard error (SE), and frequency of occurrence (%FO) of naidids at 1-24 m within each area studied near the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1972 through 1978. The number of observations for tubificid species (Ns) and for total naidids (Nt) is given at the bottom.

Taxon	North ¹			Central ¹			South ¹		
	\bar{x}	SE	%FO	\bar{x}	SE	%FO	\bar{x}	SE	%FO
<i>Amplichaeta leydigii</i>	0.3	0.2	0.5	0.2	0.1	0.3	0.1	0.1	0.2
<i>Arctonais lomondi</i>	1.4	0.7	1.2	1.2	0.5	1.4	1.0	0.9	0.4
<i>Chaetogaster diaphanus</i>	59.9	10.6	17.6	90.9	13.2	16.4	53.6	8.4	17.9
<i>Chaetogaster diastrophus</i>	0.3	0.2	0.5	0.1	0.1	0.1	0.0	0.0	0.0
<i>Dero digitata</i>	<0.1	<0.1	0.2	0.1	0.1	0.1	0.0	0.0	0.0
<i>Dero behningi</i>	<0.1	<0.1	0.2	0.0	0.0	0.0	0.0	0.0	0.0
<i>Nais bretscheri</i>	0.0	0.0	0.0	<0.1	<0.1	0.1	0.0	0.0	0.0
<i>Nais pseudobtusata</i>	0.0	0.0	0.0	0.1	0.1	0.1	0.0	0.0	0.0
<i>Nais simplex</i>	0.0	0.0	0.0	0.1	0.1	0.1	0.0	0.0	0.0
<i>Nais variabilis</i>	46.4	11.4	17.6	36.0	6.3	15.2	18.1	2.7	13.9
<i>Ophidonais serpentina</i>	0.1	0.1	0.2	4.2	3.3	0.8	0.6	0.2	1.3
<i>Piguetiella michiganensis</i>	127.8	18.1	36.3	106.5	10.6	34.4	110.6	20.1	31.1
<i>Pristina foreli</i>	1.4	0.8	1.2	3.0	1.9	0.8	0.0	0.0	0.0
<i>Pristina longiseta</i>	0.0	0.0	0.0	0.1	0.1	0.1	0.0	0.0	0.0
<i>Pristina osborni</i>	0.0	0.0	0.0	<0.1	<0.1	0.1	0.1	0.1	0.2
<i>Serpentina appendiculata</i>	0.0	0.0	0.0	0.1	0.1	0.2	<0.1	<0.1	0.2
<i>Specarina josinae</i>	0.0	0.0	0.0	0.5	0.2	0.6	0.0	0.0	0.0
<i>Stylaris lacustris</i>	30.6	6.7	16.2	34.5	5.1	13.6	29.2	5.5	13.6
<i>Uncinaxis uncinata</i>	32.7	6.0	5.8	42.6	3.7	24.4	52.3	6.3	26.8
<i>Vejdovskyella intermedia</i>	8.0	2.4	5.8	6.1	1.0	4.8	10.3	3.3	5.9
<i>Wapsa mobilis</i>	<0.1	<0.1	0.2	0.0	0.0	0.0	0.0	0.0	0.0
Other naidids	1.6	1.1	1.4	0.5	0.2	0.7	0.3	0.3	0.2
Total Naididae	314.5	32.6	49.2	381.3	22.3	55.3	248.2	23.0	49.9

¹Ns = 433 Nt = 591

²Ns = 1,107 Nt = 1,585

³Ns = 560 Nt = 733

TABLE 13. Mean density (no. m^{-2}), standard error (SE), relative density as a percentage of total identified Chironomidae, and frequency of occurrence (%) in samples collected within each area (N = north, C = central, S = south) for the 20 most common Chironomidae from southeastern Lake Michigan, 1970-1978.

Taxon	Mean Density \pm SE						Relative Density			Frequency of Occurrence		
	N		C		S		N	C	S	N	C	S
<i>Chironomus fluviatilis</i> -gr.	81.2	8.4	191.0	12.8	157.1	12.7	22.7	32.3	26.7	35.1	50.0	49.2
<i>Chironomus anthracinus</i> -gr.	20.5	5.7	62.6	13.8	29.4	7.4	5.7	10.6	5.0	11.3	10.5	8.8
<i>Cladotanytarsus</i> sp.	10.7	2.1	22.2	2.7	82.1	15.6	3.0	3.8	14.0	8.2	13.9	15.0
<i>Cryptochironomus rolli</i>	0.8	0.4	1.7	0.3	0.5	0.2	0.2	0.3	0.1	1.5	3.7	1.0
<i>Cryptochironomus</i> sp. 1	2.1	0.5	2.5	0.4	2.3	0.6	0.6	0.4	0.4	4.2	4.0	3.0
<i>Cryptochironomus</i> sp. 2	25.6	2.6	36.2	2.2	25.2	2.0	7.1	6.1	4.3	27.2	33.2	27.7
<i>Cryptochironomus</i> sp. 3	2.3	0.7	7.7	0.9	9.0	1.7	0.6	1.3	1.5	2.9	9.4	9.3
<i>Heterotrissocladius changi</i>	14.3	2.3	11.6	1.4	14.0	2.2	4.0	2.0	2.4	13.4	25.4	11.6
<i>Heterotrissocladius oliveri</i>	11.1	2.3	3.5	0.7	3.6	1.0	3.1	0.6	0.6	7.9	2.5	2.8
<i>Micropsectra</i> sp.	34.2	10.0	11.6	2.1	21.0	4.8	9.6	2.0	3.6	16.6	8.5	11.6
<i>Monodiamesa tuberculata</i>	12.1	1.5	12.0	0.9	15.7	1.8	3.4	2.0	2.7	15.8	16.9	17.5
<i>Paracladopelma</i> cf. <i>undine</i>	5.4	1.6	7.0	1.2	7.1	1.6	1.5	1.2	1.2	6.1	6.0	8.4
<i>Paracladopelma camptolabis</i> -gr.	5.6	1.0	12.2	1.2	12.4	1.5	1.6	2.1	2.1	8.4	12.7	13.2
<i>Paracladopelma winnelli</i>	16.6	2.0	6.4	0.7	9.9	1.1	4.6	1.1	1.7	17.7	8.8	13.3
<i>Polypedilum scalaenum</i>	13.8	3.2	30.4	5.0	37.0	10.4	3.9	5.1	6.3	10.4	13.9	12.7
<i>Polypedilum</i> cf. <i>tuberculum</i>	2.0	0.5	2.4	0.5	2.3	0.5	0.6	0.4	0.4	2.9	2.7	3.0
<i>Potthastia longimanus</i>	3.6	0.8	3.9	0.5	3.1	0.7	1.0	0.7	0.5	6.1	7.3	5.1
<i>Procladius</i> sp.	29.0	5.3	33.6	3.6	56.9	8.8	8.1	5.7	9.7	20.5	18.8	25.9
<i>Psectrocladius simulans</i>	2.8	1.1	2.5	0.4	1.4	0.4	0.8	0.4	0.2	3.0	4.0	3.0
<i>Robackia demeijerei</i>	9.3	1.8	33.5	6.2	26.1	4.3	2.6	5.7	4.4	8.1	14.8	13.0
<i>Saetheria tylus</i>	46.0	7.1	77.7	6.1	60.1	7.6	12.8	13.1	10.2	17.0	27.5	22.2
Total Chironomidae	377.4	18.7	622.2	22.9	609.1	28.7	—	—	—	87.4	88.1	91.1

White (1985a) and are generally more numerous at depths > 20 m. Further, each of these four taxa primarily inhabits the sand substrata more widely distributed in the north region (see sediments chapter, this book). The three taxa numerically greater in the south region and somewhat greater in the central region are Type 1 mesotrophic: *Cladotanytarsus* sp., *Polypedilum scalaenum*, and *Procladius* sp. The remaining three, *Chironomus fluviatilis*-gr., *Chironomus anthracinus*-gr., and *Cryptochironomus* sp. 2, were classified as Type 2 eutrophic taxa and were most abundant in the central region. When Saether's (1975) chironomid key to trophic condition was tested with data from the > 28-m depth zone, all three regions were found to be δ -oligotrophic. The northernmost area was more oligotrophic (γ) than were either the central (ϵ) or south (δ) regions. In general, these analyses support a regional difference also noted by chironomid trophic index values (Winnell and White 1985a), and it is concluded that generally oligotrophic conditions prevail in the profundal zone and north region while slightly mesotrophic conditions occur in the nearshore and the south and central regions.

As a group, the Chironomidae have a much greater trophic diversity than either tubificids or psidiids, with some species feeding on particulates, some grazing on algae and other plant matter, and some preying on smaller invertebrates. Moreover, they range from sand-burrowing species to distinct tube builders (Winnell 1984b, Winnell and White 1985b).

Based on ecology and feeding habits, the fingernail clams (Psidiidae) can be roughly grouped into species which filter-feed on particles in the water column (*Sphaerium*) and into species which live within the substratum and filter deposited sediments (*Pisidium*) (Mackie et al. 1980, Zdeba and White 1985). We expected that distribution and abundance of the two genera might reflect regional differences and abundance of the two genera might reflect regional differences in substratum parameters. Further, Clarke (1979) has shown that psidiid distributions and abundance are associated with trophic status. Under a gradient of enriched conditions, he found that densities of most species increase to a point where species that are rarer, and seemingly more sensitive to organics, began to drop out.

Of the 25 Psidiidae taxa collected in the vicinity of the Donald C. Cook Nuclear Plant, only five were collected frequently enough or identified routinely enough to speculate on regional trends. *Pisidium conventus* is a common component of the sandy profundal zone along with *Heterotrissocladius oliveri*. As expected, its abundance followed that of *H. oliveri* by being most abundant in the north region and least abundant in the south region. *Pisidium casertanum* is a shallower water species and appeared to have an affinity for substrata composed of silt and clay and, to a lesser extent, fine sand (Zdeba and White 1985); it was most abundant in the south and central regions. A third species, *P. nitidum*, was more than twice as abundant in the north and south regions as in the central region. Reasons for this distribution remain unknown; however, it appears that the north region, while having fewer deposits of fine-grained organics, may be more stable than the central region. In the genus *Sphaerium*, *S. nitidum*, a deeper water species, was least abundant in the south region and most abundant in the north region.

S. striatinum is found in a wide variety of aquatic habitats from ponds to small streams to the Great Lakes (Mackie et al. 1980). Species with wide distributions often are less competitive when co-occurring with siblings, thus, it was not surprising that *S. striatinum* was most abundant in the central region where other species were less dense, particularly at depths less than 12 m in the vicinity of the riprap structures.

Areal distribution of the amphipod *Pontoporeia hoyi* (a profundal species often associated with *Stylodrilus*, *Heterotrissocladius oliveri*, and *Pisidium conventus*) displayed a predictable trend, increasing in a northerly direction. Average abundance was lowest in the south region ($1,871 \text{ m}^{-2}$) and highest in the north region ($2,731 \text{ m}^{-2}$).

Following the sedimentation patterns derived from bottom sample observations and from sediment structure analysis in this book, some conclusions can be drawn about macrozoobenthic distribution in the vicinity of the Donald C. Cook Nuclear Plant. In the south region, increased accumulation rates of organic material occur nearer the shoreline. Species discussed above which favor these more mesotrophic conditions (as particle feeders, predators on particle feeders, etc.) are generally more abundant at all depths and specifically more abundant in shallower depth zones. Populations of the mesotrophic taxa are generally less abundant in the central and north regions and reach maximum population sizes farther offshore. Species favoring the north region are more oligotrophic in nature but, as with the mesotrophic group, reach maximum densities farther offshore in the north than in the south regions. Associations in the central region are not so clear. Most often, community structure in the central region resembles the south region; however, the central region seems more patchy in organism distributions. Though we have insufficient data for statistically insignificant conclusions, it appears that patchiness in the central region may result from extremely localized current patterns established by the riprap and intake-discharge effects.

CHANGES IN ZOOBENTHOS DISTRIBUTIONS

A goal of power plant monitoring studies is to document the impact of power generation on organism populations in the vicinity of power plants. If an impact does occur, it should be statistically measurable as the result of some alteration in the biological processes of the organisms (distribution, abundance, feeding, reproduction, etc.) which influences the population structure. The primary difficulty in determining an impact is separation of plant effects from the natural population fluctuations in combination with small-scale patchiness. Natural populations at the Donald C. Cook Nuclear Plant are addressed in the Macrozoobenthos Ecology chapter in this book; patchiness is addressed above. As Brinkhurst (1974) pointed out, these difficulties can only be addressed in long-term studies but, "...there has never been a study of the benthos of a lake in which ... the sampling methodology and schedule have been properly evaluated, most of the major species identified, and which extended over all seasons for a consecutive number of years." Although there remain some problems in the methodologies used, the extensive sampling program, level of identifications, and knowledge of the biology of the species in the study area (LaDronka 1984; Winnell 1984a, 1984b, Zawacki 1985; Zdeba and White 1985) over 8 consecutive years (4 preoperational and 4 operational) provide the basis for a reasonable assessment of population trends. The trends have been examined by analysis of variance (ANOVA) of plant impact under the statistical framework originally created by Johnston (1973, 1974).

The ANOVA design was a nested, mixed model composed of five main effect factors: fixed factors included Month (M), Region (R), water Depth (D), and power plant Construction Time (C); the single random factor was Year(Y), which was

nested within Construction Time. The Month factor (M) included the major survey months of April, July, and October for all years, except 1971 for which November was substituted for October (Table 1). The Region factor (R) had two levels: (1) the inner region (or central region discussed above), into which discharge water was released, was considered the treatment area, and (2) the outer region (north and south regions combined) which served as the control area where no effects from heated discharge or plant operation were expected. The Depth factor (D) had the three levels of 0-8 m, 8-16 m, and 16-24 m (Zones 0, 1, and 2). The number of levels of the Depth factor varied from 1 to 3, based upon the taxon considered, and upon ecological data derived from the 8-m interval analyses. The Construction Time factor had the two levels of before and after plant operation based on the year 1975 that the Unit I generator began service. The Year factor was composed of eight levels (four in Before and four in After) consisting of the years 1971-1978. For the balanced design, 1970 data were not included. Years nested in the Before level included 1971-1974 and those in the After level included 1975-1978.

The statistical test used to estimate plant effect was the F-value derived from the ratio of the variance attributed to the Construction Time-Region (CR) and Construction Time (Year nested)-Region interactions (Johnston 1974):

$$F_{(a)(b)} = MS_{CR} / MS_{C(Y)R} \quad (1)$$

The null hypothesis tested stated that "the startup of the plant had no effect on the difference between the mean log-transformed zoobenthic density at the inner stations and the same quantity at the outer stations" (Johnston 1974). The alternative hypothesis was that there was some difference measurable at the $\alpha = 0.05$ level of significance following power plant operation. All analyses were based on $\log_{10}(x + 1)$ transformed average station densities (Elliot 1971, Johnston 1974, Chang and Winnell 1981). The most important considerations regarding sampling methodology (summarized in Table 1) were (1) that collections were made by various Ponars which ranged from 0.0165 to 0.1103 m² in grab area, (2) that replication per station varied from one to five grabs, and (3) that the number of stations (observations) per cell in the ANOVA was unequal. Sampling methods were least uniform between April 1971 and April 1974; after April 1974 a consistent pattern was followed.

The potential relationships to the power plant's operation were assessed for the five major macrozoobenthic components of the study area: *Pontoporeia hoyi* (Amphipoda), *Pisidium* spp. (Pelecypoda), *Stylodrilus heringianus* (Lumbriculidae, Oligochaeta), Tubificidae (Oligochaeta), and Chironomidae (Diptera, Insecta) and for total macrozoobenthos. Plant effect was based on density estimates for each zoobenthic component derived from lake sampling. Because the distributions of these taxa were shown to be depth dependent (Winnell 1984a, 1984b; LaDronka 1984; Zawacki 1985; Zdeba and White 1985), analyses considered only those depths where the respective taxa were common: *S. heringianus* in water between 16 and 24 m deep and *P. hoyi* and *Pisidium* spp. in water between 8 and 16 m and 16 and 24 m deep. Chironomidae and Tubificidae occurred commonly in all three depth categories (water 0 to 24 m deep) as did the "total benthos" category.

Johnston (1974) developed the least detectable true ratio (R) for the purpose of determining the degree to which "a relative increase or a relative decrease of the inner population" must change to be detected by the test. The logic underscoring this approach was based on collapsing the initial ANOVA to a simple two-factor design with Construction Time (two levels) and Region (two levels). From these, Johnston

(1974) calculated, using anti-logarithms, the "true mean number of animals per square meter" for any of the resulting four cells. The justification and derivation of R is lengthy (see Johnston 1974). R is based on the least detectable true difference of Sokal and Rohlf (1969) and extraction from and manipulation of components of the ANOVA. In our application, we used the following to determine R :

$$\delta = \sigma (2/n) (t_{(a-1)} + t_{2n-P(1)}) \quad (2)$$

where; δ = least detectable true change

σ = MS_{CIVR}

n = average sample size per cell*

t = Student's t

γ = degrees of freedom

α = 0.05 (Type I error)

P = 0.95 (Type II error)

* = since cell size was unequal, calculation of n followed the procedure of Sokal and Rohlf (1969):

$$n = [1/(a-1)] [\Sigma n_i - (\Sigma n_i^2 / \Sigma n_i)] \quad (3)$$

where; n = average sample size per cell

a = number of cells

n_i = number of observations in each cell.

Derivation of δ from these formulae was substituted into Johnston's (1974) equation for the least detectable true ratio:

$$R \geq 10^{(-23)} \text{ and } R \leq 10^{(+23)} \quad (4)$$

R values calculated as such for each zoobenthic component establish a range within which plant effects were either non-detectable at $\alpha = 0.05$ and $P = 0.95$ or non-existent. R -values exceeding this range indicate a detectable true population change significant at $\alpha = 0.05$ and $P = 0.95$.

A quantity not considered by Johnston (1974) was the "actual population change true ratio" which we designate as R' . The quantity is easily derived from the four cells of the collapsed two-way ANOVA by

$$R' = (I_a + 1/O_a + 1)/(I_b + 1/O_b + 1) \quad (5)$$

where; I_a = anti-logarithm of the average mean log density in the inner region during After operational years.

O_a = anti-logarithm of the average mean log density in the outer region during After operational years.

I_b = anti-logarithm of the average mean log density in the inner region during Before operational years.

O_b = anti-logarithm of the average mean log density in the outer region during Before operational years.

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The quantity estimates the actual true ratio change in the population densities for each zoobenthic component. R' -values for taxa will fall within the R -value range previously determined when the effect of plant operation is not significant. Conversely, R' values fall outside the R -value range for taxa experiencing significant plant effects. Only in cases where probability values are close to $\alpha = 0.05$ might there be contradictory conclusions drawn from R - and R' -values. The contradiction arises from the fact that the ANOVA considers only Type I errors whereas R considers both Type I and Type II errors. Elimination of the Type II error in the estimate of δ will result in agreement of conclusions derived from R - and R' -values. Estimation of R' -values provides a final check that the methodology is inherently correct. In addition, R' gives the degree of change actually occurring and the direction of the change.

For the taxa with a significant plant effect test, additional data from the final 3 years were analyzed to assess effects over time. The additional years were 1979, 1980, and 1981. Analysis of these years with respect to the 1971-1978 data set was accomplished by sequential inclusion and comparison of these years in the Inner/Outer-Before/After, collapsed two-way design using Scheffé simultaneous-confidence-intervals (Scheffé 1959, Winer 1971). All analyses were based on the $\log_{10}(x+1)$ average of station densities. Because Scheffé's test is very conservative (Winer 1971) with respect to the Type I error, comparisons were judged significant at $\alpha = 0.10$ (Statistical Research Laboratory, University of Michigan, pers comm.).

We found no significant plant effects between 1971-1978 using log-transformed average station densities for *Pontoporeia hoyi* (8-24 m), *Pisidium* spp. (8-24 m), *Stylodrilus heringianus* (16-24 m), Tubificidae (0-24 m), and total zoobenthos (0-24 m) (Tables 14-18). Among these five taxonomic categories, the least detectable true ration (R) ranged from a low of 2.14 for total zoobenthos to a high of 23.5 for *S. heringianus* (Table 19). In general, R , ranged from 2 to 5. The average actual density change (R') among these taxa was 1.10 and ranged from 0.62 to 1.61. In each case, R' values fell within the range of R -values where plant effects were non-detectable in accordance with the non-significance derived from the ANOVA.

Based on the test for plant effects, significant ($p = 0.0327$) density differences were associated with plant operation for Chironomidae (0-24 m) (Table 20). Although the R' -value (1.47) fell within the R -value range (0.57-1.75), this was an artifact of methodological differences. When correcting for error estimation differences, the R -range became 0.73-1.36, and the R' -value of 1.47 fell outside the range. Similar corrections for the other taxa maintained the logical coincidence of the non-significant plant effect conclusion from the ANOVA and R' -values falling within the non-detectable range of R -values.

Simultaneous, pairwise comparisons (Scheffé 1959) of cell means from the collapsed two-way ANOVA (Johnston 1974) indicated that Before Inner/Outer regional and After Inner/Outer regional comparisons of chironomid densities were not significantly different, respectively, for any sequential yearly combinations. However, Inner and Outer Before/After density comparisons were significantly different for all sequential yearly combinations, except the 1971-1978 combination. Nonetheless, for all sequential yearly comparisons, chironomid densities in the inner region after the plant was in operation were always significantly greater than those in the outer region. In addition, After-operational-year densities were greater than Before-operational-year densities for both regions. Finally, while both regions experienced significant density increases between the two time periods, the density ratio

TABLE 14. Analysis of variance for *Pontoporeia hoyi* based on $\log(x+1)$ mean station densities near the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1971-1978 (Const time = Construction time). Brackets [] denote nesting of Year [Y] within Construction time.

Source	Sum of Squares	DF	Mean Square	F-ratio	Denom. of F-ratio	Prob.	Signif.
Depth (D)	46.45	1	46.45	100.09	MSE	0.0000	***
Month (M)	48.55	2	24.28	52.31	MSE	0.0000	***
Region (R)	5.04	1	5.04	10.86	MSE	0.0011	**
Const Time (C)	8.13	1	8.13	1.39	C[Y]	0.2839	NS
C[Y]	35.21	6	5.87	12.64	MSE	0.0000	***
DM	3.38	2	1.69	3.64	MSE	0.0271	*
DR	0.18	1	0.18	0.38	MSE	0.5379	NS
DC	1.66	1	1.66	0.88	DC[Y]	0.3844	NS
DC[Y]	11.26	6	1.88	4.04	MSE	0.0006	***
MR	2.24	2	1.12	2.42	MSE	0.0902	NS
MC	0.39	2	0.20	0.16	MC[Y]	0.8539	NS
MC[Y]	14.71	12	1.23	2.64	MSE	0.0020	**
CR	0.0083	1	0.0083	0.0166	C[Y]R	0.9694	NS
C[Y]R	3.02	6	0.50	1.09	MSE	0.3675	NS
DMR	0.78	2	0.39	0.84	MSE	0.4324	NS
DMC	0.51	2	0.26	0.55	MSE	0.5774	NS
DRC	3.02	1	3.02	8.39	DRC[Y]	0.4111	NS
DCR[Y]	2.18	6	0.36	0.78	MSE	0.5860	NS
MRC	0.55	2	0.28	1.87	MRC[Y]	0.1964	NS
MRC[Y]	1.79	12	0.15	0.32	MSE	0.9857	NS
DMRC	0.28	2	0.14	0.31	MSE	0.7736	NS
ERROR	194.00	418	0.46	—	—	—	—

TABLE 15. Analysis of variance for total *Pisidium* spp. based on log (x+1) mean station densities near the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1971-1978 (Const time = Construction time). Brackets [] denote nesting of Year [Y] within Construction time.

Source	Sum of Squares	DF	Mean Square	F-ratio	Denom. of F-ratio	Prob.	Signif.
Depth (D)	16.00	1	16.00	22.46	MSE	0.0001	****
Month (M)	1.32	2	5.16	7.25	MSE	0.0008	***
Region (R)	0.016	1	0.016	0.023	MSE	0.8795	NS
Const Time (C)	4.02	1	4.02	1.03	C[Y]	0.3493	NS
C[Y]	12.26	6	2.89	4.37	MSE	0.0001	****
DM	4.66	2	2.33	3.27	MSE	0.0390	.
DR	1.07	1	1.07	1.51	MSE	0.2198	NS
DC	0.021	1	0.021	0.091	DC[Y]	0.7731	NS
DC[Y]	1.40	6	0.23	0.33	MSE	0.9211	NS
MR	1.23	2	0.61	0.86	MSE	0.4239	NS
MC	0.93	2	0.47	0.51	MC[Y]	0.6129	NS
MC[Y]	11.05	12	0.92	1.29	MSE	0.2213	NS
CR	1.05	1	1.05	1.98	C[Y]R	0.2090	NS
C[Y]R	3.17	6	0.53	0.74	MSE	0.6177	NS
DMR	0.64	2	0.32	0.45	MSE	0.6379	NS
DMC	1.60	2	0.81	1.12	MSE	0.3273	NS
DRC	0.41	1	0.41	1.08	DRC[Y]	0.3388	NS
DRC[Y]	2.30	6	0.38	0.54	MSE	0.7778	NS
MRC	0.85	2	0.42	0.84	MRC[Y]	0.4637	NS
MRC[Y]	6.08	12	0.51	0.71	MSE	0.7421	NS
DMRC	1.29	2	0.65	0.91	MSE	0.4033	NS
ERROR	297.71	418	0.71	-	-	-	1

TABLE 16. Analysis of variance for *Stylodrilus heringianus* based on $\log(x+1)$ mean station densities near the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1971-1978 (Const time = Construction time). Brackets [] denote nesting of Year [Y] within Construction time.

Source	Sum of Squares	DF	Mean Square	F-ratio	Denom. of F-ratio	Prob.	Signif.
Month (M)	9.86	2	4.93	6.63	MSE	0.0017	**
Region (R)	10.95	1	10.95	14.72	MSE	0.0002	***
Const Time (C)	0.081	1	0.081	0.063	C[Y]	0.8102	NS
C[Y]	7.74	6	1.29	1.73	MSE	0.1161	NS
MR	0.50	2	0.25	0.33	MSE	0.7107	NS
MC	0.77	2	0.39	0.41	MC[Y]	0.6726	NS
MC[Y]	11.47	12	0.96	1.29	MSE	0.2272	NS
CR	0.0075	1	0.0075	0.0052	C[Y]R	0.9338	NS
C[Y]R	8.61	6	1.43	1.93	MSE	0.0780	NS
MRC	0.088	2	0.044	0.058	MRC[Y]	0.9461	NS
MRC[Y]	9.13	12	0.76	1.02	MSE	0.4321	NS
ERROR	139.81	188	0.74	—	—	—	—

TABLE 17. Analysis of variance for Tubificidae based on $\log(x+1)$ mean station densities near the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1971-1978 (Const time = Construction time). Brackets [] denote nesting of Year [Y] within Construction time.

Source	Sum of Squares	DF	Mean Square	F-ratio	Denom. of F-ratio	Prob.	Signif.
Depth (D)	425.08	2	212.54	310.43	MSE	0.0000	***
Month (M)	32.58	2	16.29	23.79	MSE	0.0000	***
Region (R)	0.73	1	0.73	1.06	MSE	0.3036	NS
Const Time (C)	0.058	1	0.058	0.020	C[Y]	0.8922	NS
C[Y]	17.46	6	2.91	4.25	MSE	0.0003	***
DM	2.39	4	0.40	0.87	MSE	0.4815	NS
DR	0.98	2	0.49	0.71	MSE	0.4920	NS
DC	0.65	2	0.33	0.69	DC[Y]	0.5204	NS
DC[Y]	5.74	12	0.48	0.70	MSE	0.7523	NS
MR	2.26	2	1.13	1.65	MSE	0.1929	NS
MC	2.58	2	1.29	0.78	MC[Y]	0.4803	NS
MC[Y]	19.80	12	1.65	2.41	MSE	0.0047	**
CR	0.089	1	0.089	0.074	C[Y]R	0.7947	NS
C[Y]R	7.27	6	1.21	1.77	MSE	0.1027	NS
DMR	0.28	4	0.071	0.104	MSE	0.9811	NS
DMC	0.80	4	0.20	0.29	MSE	0.8845	NS
DRC	2.73	2	1.36	1.89	DRC[Y]	0.1934	NS
DRC[Y]	8.64	12	0.72	1.99	MSE	0.0228	*
ERROR	442.29	646	0.68	—	—	—	—

TABLE 18. Analysis of variance for total benthos based on $\log(x+1)$ mean station densities near the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1971-1978 (Const time = Construction time). Brackets [] denote nesting of Year [Y] within Construction time.

Source	Sum of Squares	DF	Mean Square	F-ratio	Denom. of F-ratio	Prob.	Signif.
Depth (D)	197.83	2	98.91	354.06	MSE	0.0000	***
Month (M)	61.75	2	30.88	110.52	MSE	0.0000	***
Region (R)	0.25	1	0.25	0.88	MSE	0.3486	NS
Const Time (C)	10.11	1	10.11	4.75	C[Y]	0.0721	NS
C[Y]	12.76	6	2.13	7.61	MSE	0.0001	***
DM	20.76	4	5.19	18.58	MSE	0.0000	***
DR	0.077	2	0.039	0.138	MSE	0.8711	NS
DC	5.40	2	2.70	5.87	DC[Y]	0.0167	*
DC[Y]	5.49	12	0.46	1.64	MSE	0.0764	NS
MR	0.52	2	0.26	0.93	MSE	0.3951	NS
MC	0.59	2	0.30	0.48	MC[Y]	0.6302	NS
MC[Y]	7.43	12	0.62	2.22	MSE	0.0097	**
CR	0.68	1	0.68	2.62	C[Y]R	0.1567	NS
C[Y]R	1.54	6	0.26	0.92	MSE	0.4798	NS
DMR	0.10	4	0.026	0.93	MSE	0.4459	NS
DMC	0.59	4	0.15	0.53	MSE	0.7137	NS
DRC	3.12	2	1.56	5.03	DRC[Y]	0.0259	*
DRC[Y]	3.77	12	0.31	1.12	MSE	0.3403	NS
ERROR	180.48	646	0.28	-	-	-	-

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TABLE 19. Comparison of the least detectable true ratio (R) and actual density ratios (R') calculated from mean square terms derived from analysis of variance methods for *Pontoporeia hoyi* (Ponto), *Pisidium* spp. (Pisid), *Stylodrilus heringianus* (Stylo), *Tubificidae* (Tubif), *Chironomidae* (Chir), and total benthos (TBenth). Limits of non-detectable plant effect are all R' -values between the maximum [$R(\max)$] and minimum [$R(\min)$] range of R -values. R -values as determined by the δ -term of Johnson (1974). p = probability of significant plant effect extracted from ANOVAs.

Taxon		$R(\max)$	$R(\min)$	R'	p
Ponto	0.3988	3.66	0.273	1.23	0.9694
Pisid	0.4084	3.78	0.265	0.62	0.2090
Stylo	0.9695	23.50	0.043	1.19	0.9338
Tubif	0.5094	5.25	0.190	1.10	0.7947
Chir	0.1716	1.75	0.572	1.47	0.0327
TBenth	0.5094	5.25	0.190	1.10	0.1567

of After/Before in the inner region averaged 2.09 (2.07-2.14). The same ratio in the outer region was only 1.46 (1.40-1.50).

Regardless of time period or region, *Chironomus fluviatilis*-gr. (presently composed of two indistinguishable species, Winnell 1984b) dominated the Chironomidae (Table 20). While an abundance of several taxa regularly comprised more than 1% of the total Chironomidae, and varied regionally among Before/After comparisons, taxa having significantly different annual percent changes were *Cryptochironomus* sp. 3, *Chironomus anthracinus*-gr. (two species), *Paracladopelma* (three species), and *Robackia demeijerei*. These four taxa comprised 15-24% of the total Chironomidae. Percent occurrence of *Cryptochironomus* sp. 3 was significantly greater during After operational years than during Before operational years in both regions. Percent occurrences of *Paracladopelma* spp. and *R. demeijerei* were significantly greater during After operational years than during Before operational years in the Inner Region only; all other comparisons were non-significant. For *C. anthracinus*-gr., there was an opposite pattern with significantly greater percent occurrences present during Before operational years in the Inner Region.

A basic ecological premise inherent in the ANOVA design assumes that, under normal lake conditions, macroinvertebrate population densities within the survey area are similarly affected by naturally changing factors (e.g., food availability, predation, reproductive success, weather, pollution). Highly variable annual and monthly abundances were expected, as well as variation due to depth preferences. Regional abundance differences indicated inherent regional disparities unless they were associated with Construction Time.

When examining changes among chironomid species, two trends were notable. First, although Before/After preoperational year chironomid densities increased significantly in both regions, they increased considerably more in the inner region relative to the outer region. This trend suggested that temporally inherent density increases were, to some degree, spatially dependent. The disproportionate density increase in the inner region suggested that a factor or set of factors affected the observed difference. One factor which we suspect contributed to the density difference was plant operation.

Second, significant regional changes in Before/After operational year percentages

TABLE 20. Analysis of variance for Chironomidae based on $\log(x+1)$ mean station densities near the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1971-1978 (Const time = Construction time). Brackets [] denote nesting of Year [Y] within Construction time.

Source	Sum of Squares	DF	Mean Square	F-ratio	Denom. of F-ratio	Prob.	Signif.
Depth (D)	13.93	2	6.96	20.15	MSE	0.0001	***
Month (M)	49.07	2	24.54	71.00	MSE	0.0000	***
Region (R)	0.46	1	0.46	1.32	MSE	0.2510	NS
Const Time (C)	8.65	1	8.65	2.04	C[Y]	0.2031	NS
C[Y]	25.52	6	4.25	12.14	MSE	0.0000	***
DM	52.41	4	13.10	37.92	MSE	0.0000	***
DR	0.013	2	0.0063	0.0183	MSE	0.9819	NS
DC	4.92	2	2.46	4.56	DC[Y]	0.0336	*
DC[Y]	6.46	12	0.54	1.56	MSE	0.0988	NS
MR	0.23	2	0.12	0.33	MSE	0.7190	NS
MC	0.48	2	0.24	0.75	MC[Y]	0.4933	NS
MC[Y]	3.83	12	0.32	0.92	MSE	0.5262	NS
CR	1.07	1	1.07	7.64	C[Y]R	0.0327	*
C[Y]R	0.82	6	0.14	0.40	MSE	0.8792	NS
DMR	2.74	4	0.68	1.98	MSE	0.0959	NS
DMC	3.09	4	0.77	2.24	MSE	0.0633	NS
DRC	4.27	2	2.13	17.75	DRC[Y]	0.0003	***
DRC[Y]	1.39	12	0.12	0.33	MSE	0.9838	NS
ERROR	223.23	646	0.35	—	—	—	—

among certain species were suspected to be related to plant operation. Before/After operational year regional community structure changes were similar, but they were more intensive in the inner region. Increased homogeneity in the inner region after operation of the plant might have reflected a plant effect on community structure, i.e., producing less variable conditions whereby inherent, annual variability was dampened and a more predictable community was established.

While plant operation affected the chironomid community nearest the plant, there remains the question of whether these effects are negative or positive. Clearly, no firm resolution can be drawn at this point, but analysis of several trends leads to a tentative resolution. Based on density decreases for *Chironomus* spp., species typically thought to be more numerous in more eutrophic conditions, coupled with increases in more mesotrophic taxa, such as *Saetheria tylus*, *Paracladopelma* spp., and *Robackia demeijerei*, we project that operation of the power plant may have helped stimulate a naturally occurring mesotrophic trend in the inner region, as evidenced by similar, though less intense, changes in the outer region. Localized changes near the plant may not be related to a heated effluent but to an alteration of current and wave activity as they affected the distributions of substrate. As currents and waves pass through the inner region, the riprap can disrupt normal sedimentation processes and increase scour, thereby creating a less stable environment which would favor particular mesotrophic taxa more able to utilize the habitat (Winnell and Jude 1982). It cannot be ruled out that discharge operations may have increased food supply to the inner area. Certain ephemeral food supplies could be continually introduced via discharge operations, consequently increasing abundances of particular chironomid species. Based on results from other components of the zoobenthos, only the Chironomidae seem to have responded to plant operation. While food supply or temperature might have influenced present trophic conditions, in the longterm, the massive effects of current, wave activity, and ice scour would purge the nearshore areas. Purging would occur particularly during spring, autumn, and occasional summer storms, the times when major surveys were made. The influence of potential food supply or temperature effects would need to be studied at sites farther offshore, at sites a considerable distance from the plant, and at sites where substrates have not been altered. Further, there is the obvious difficulty separating plant impacts from general trends in lakewide or regional conditions.

We conclude that some factor(s) of plant operations (1) resulted in an increased number of chironomids in the inner region disproportionate to that in the outer region, (2) aided in establishing conditions favoring a more homogeneous chironomid population structure in the inner region, and (3) aided in decreasing eutrophy in the inner region. Increases in the number of chironomids most likely reflect alteration of substrates, but increased (or at least more constant) food supply and temperature effects cannot be entirely ruled out. In generally, we interpret changes in the macrozoobenthic population structure attributable to the power plant as essentially benign, but nevertheless real in terms of ecological changes.

THE RIPRAP COMMUNITY

Perhaps the most dramatic change in the macrozoobenthic community caused by the Donald C. Cook Nuclear Plant was the creation of an artificial reef when the riprap protecting the intake-discharge structures was constructed. Lauritsen (1979) compared the riprap community with the community normally found on the open

lake bottom adjacent to the riprap and with the community of a natural shoal at Waugoshance Point (northeastern Lake Michigan). Fauna distributions recorded from both the riprap and the shoal compared favorably with distributions of similar habitats sampled in the Canadian nearshore areas by Barton and Hynes (1978a, 1978b) (Table 21). Species present on the riprap were distinctly different from the lake bottom but many were similar to those found on a natural shoal by Lauritsen (1979). With a few exceptions, the fauna of consolidated substrata at the Cook Plant was not found on the unconsolidated open lake bottom. The exceptions found in both habitats were primarily predators and species feeding on algae.

The similarities between Waugoshance Point and the Donald C. Cook Nuclear Plant riprap lie more in the number of shared species than in the similarity in relative abundances of shared species. Differences between the riprap and the shoal can, in part, be explained by differences in depth and by proximity to the potential colonizing populations. Other differences may lie in geographic location. Obviously, one would expect both an abundance of consolidated substrate species and a lack of infaunal species at the shoal and on the riprap. This was seen particularly in the number of rheophilic aquatic insects (e.g., Ephemeroptera, Trichoptera, and *Rheotanytarsus*) which are rarely found on the open lake bottom (Barton and Hynes 1978a, 1978b).

Waugoshance point is an ideal consolidated substrate, wave zone habitat, and this was evidenced by the more lotic benthic fauna found there. Filter-feeders were the dominant functional group, as they are in most streams. At the Donald C. Cook Nuclear Plant riprap site, the kinds and numbers of organisms found depended on the presence or absence of *Cladophora*. During summer months, when dense growths of *Cladophora* formed on the rocks, the zoobenthos were dominated by attached and clinging predatory forms which use algae for support. With the disappearance of *Cladophora* in late fall, the number of invertebrates found dropped sharply. The predator, *Hydra*, dominated the fauna during most of the year.

SUMMARY

1. Crayfish are the only macrozoobenthos in the vicinity of the Donald C. Cook Nuclear Plant large enough to be impinged on the travelling screens. Though a large number are impinged each year, the population probably is becoming stable. The riprap provides the primary habitat.

2. Water intake entrains a wide variety of zoobenthos. The effects of entrainment on lake populations of *Pontoporeia hoyi* and *Mysis relicta* are unknown but probably are small in comparison with lake populations present in the vicinity of the power plant.

3. Dynamics of the lake bottom zoobenthos populations in the vicinity of the Donald C. Cook Nuclear Plant show the general lake-wide trend toward mesotrophy. Populations in the south region are slightly more mesotrophic due to greater accumulations of organic particulates while populations in the north tend toward oligotrophy, again due to structure of the substrate.

4. While populations of Chironomidae (and other major taxa) show general density increases over all years of the study in all regions, the central or inner region densities have increased more rapidly and toward a more homogeneous species composition. Reasons for increases in the inner region are unknown but may reflect altered current patterns near the riprap or increased food supplies stimulated by the

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TABLE 21. Zoobenthic species collected from sites in Lake Michigan, July-October 1978.

Taxon	Waugo. Point	Cook Riprap	Cook Ponar
Coelenterata			
Hydroida			
Hydridae			
<i>Hydra</i> sp.	x	x	x
Clavidae			
<i>Corydyllophora lacustris</i> Allman		x	
Platyhelminthes			
Turbellaria		x	x
Annelida			
Hirudinea			
Glossiphoniidae			
<i>Helobdella stagnalis</i> (Linnaeus)			x
Oligochaeta			x
Aeolosomatidae		x	
Enchytraidae		x	
Limbriculidae			
<i>Stylodrilus heringianus</i> Claparede		x	
Naididae			
<i>Chaetogaster diaphanus</i> (Gruithuisen)	x	x	x
<i>Chaetogaster diastrophus</i> (Gruithuisen)	x	x	
<i>Dero</i> sp.	x	x	
<i>Nais alpena</i> Sperber		x	
<i>Nais behningi</i> Michaelsen	x	x	
<i>Nais bretscheri</i> Michaelsen		x	
<i>Nais elunguis</i> Müller		x	
<i>Nais simplex</i> Piguet	x	x	
<i>Nais variabilis</i> (= <i>pardalis</i>) (Piguet)	x	x	x
undetermined <i>Nais</i> sp.		x	
<i>Ophidonais serpentina</i> (Müller)		x	
<i>Piguetiella michiganensis</i> Hiltunen			x
<i>Pristina foreli</i> (Piguet)	x		x
<i>Pristina osborni</i> (Walton)		x	
<i>Specarina josinae</i> (Vejdovsky)		x	x
<i>Stylaria lacustris</i> (Linnaeus)	x	x	x
<i>Uncinaiis uncinata</i> (Orsted)			x
<i>Vejdovskyella intermedia</i> (Bretscher)	x	x	
Tubificidae			
<i>Aulodrilus pluriseta</i> (Piguet)			x
<i>Limnodrilus augustipenis</i> Brinkhurst & Cook			x
<i>Limnodrilus cervix</i> Brinkhurst			x
<i>Limnodrilus hoffmeisteri</i> (Walton)			x
<i>Ilyodrilus freyi</i> Brinkhurst			x
<i>Potamotheix moldaviensis</i> Vejdovsky and Mrazek			x
<i>Potamotheix vejdoskyi</i> (Hrabe)			x
Immature with hair chaetae		x	x
Immature without hair chaetae		x	x

TABLE 21. Continued

Taxon	Waugo. Point	Cook Riprap	Cook Ponar
Crustacea			
Amphipoda			
Gammaridae			
<i>Gammarus pseudolimnaeus</i> Blousfield		x	
<i>Gammarus troglophilus</i> Hubricht and Mackin	x		
Haustoriidae			
<i>Pontoporeia hoyi</i> Smith			x
Talitridae			
<i>Hyaella azteca</i> (Saussure)	x	x	x
Isopoda			
Asellidae			
<i>Asellus</i> sp.		x	
Hydracarina	x	x	x
Insecta			
Collembola	x		
Ephemeroptera			
Baetidae			
<i>Baetis</i> spp.	x		
Caenidae			
<i>Caenis</i> spp.	x		
Leptophlebiidae			
<i>Paraleptophlebia</i> sp.	x		
Heptageniidae			
<i>Heptagenia junio</i> McDunnough	x		
<i>Heptagenia maculipennis</i> Walshe	x		
<i>Heptagenia pull</i> (Clemens)		x	
undetermined <i>Heptagenia</i>	x	x	
<i>Stenonoema puchellum</i> (Walshe)		x	
<i>Stenonoema tripunctatum</i> (Banks)	x		
undetermined <i>Stenonema</i>	x	x	
Plecoptera			
Perlodidae			
<i>Isoperla</i> sp.	x		
Trichoptera			
Hydropsychiidae			
<i>Cheumatopsyche</i> sp.	x	x	
<i>Symphitopsyche recurvata</i> (Banks)	x	x	
Hydroptilidae			
<i>Agraylea</i> sp.	x	x	
<i>Ithytrichia clavata</i>	x		
Leptoceridae			
<i>Ceraclea ancylus</i> (Vorhies)	x		
<i>Ceraclea</i> sp. 2	x	x	
<i>Mystacides</i> spp.	x	x	
<i>Triaenodes</i> sp.		x	
Coleoptera			
Elmidae			
<i>Dubiraphia vittata</i> (Melsheimer)		x	
Dytiscidae			
<i>Hydroporus</i> sp.		x	

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TABLE 21. Continued

Taxon	Waugo. Point	Cook Riprap	Cook Ponar
Hemiptera			
Salidae	x		
Diptera			
Empididae			
<i>Hemerodromia</i> sp.	x		
Chironomidae			
<i>Chironomus anthracinus</i> -group			x
<i>Chironomus fluviatilis</i> -group			x
undetermined <i>Chironomus</i> sp.		x	x
<i>Cryptochironomus</i> cfr. <i>rolli</i> (Kirpitshenko)			x
<i>Cryptochironomus</i> sp. 2			x
<i>Cryptochironomus</i> sp. 3			x
<i>Demicryptochironomus</i> sp.			x
<i>Dicrotendipes</i> sp.	x	x	
<i>Endochironomus</i> sp.	x		
<i>Glyptotendipes</i> sp.		x	
<i>Parachironomus</i> sp.	x	x	
<i>Paracladopelma camptolabis</i> -group			x
<i>Paracladopelma</i> cfr. <i>nereis</i> (Townes)		x	
<i>Paracladopelma</i> cfr. <i>undine</i> (Townes)			x
<i>Phaenopsectra</i> sp.	x		
<i>Polypedilum</i> cfr. <i>fallax</i> (Johannsen)	x		
<i>Polypedilum</i> cfr. <i>scalaenum</i> (Schrunk)	x	x	x
<i>Robackia</i> cfr. <i>demeijeri</i> (Kruseman)			x
<i>Saetheria</i> cfr. <i>tylus</i> (Townes)		x	x
<i>Stictochironomus</i> sp. 1	x		
<i>Stictochironomus</i> sp. 2	x		
<i>Cladotanytarsus</i> sp.			x
<i>Micropsectra</i> sp.	x	x	
<i>Rheotanytarsus</i> sp.	x	x	
<i>Corynoneura</i> sp.	x		
<i>Cricotopus</i> cfr. <i>bicinctus</i> (Meigen)	x	x	
<i>Cricotopus cylindraceus</i> -group	x		
<i>Cricotopus festivellus</i> -group	x		
undetermined <i>Cricotopus</i> (<i>Cricotopus</i>) sp.	x		
<i>Cricotopus intersectus</i> -group	x		
<i>Heterotrissocladius</i> cfr. <i>changii</i> Saether	x		x
<i>Nanocladius</i> sp.	x	x	
<i>Orthocladius</i> cfr. <i>robaccki</i> Söponis	x		
undetermined <i>Orthocladius</i> (<i>Ortho.</i>) sp.	x		
<i>Parakiefferiella</i> sp.	x	x	
<i>Psectrocladius</i> cfr. <i>simulans</i> (Johansen)	x	x	x
<i>Synorthocladius</i> sp.	x		
<i>Thienemanniella</i> sp.	x		
<i>Monodiamesa</i> cfr. <i>tuberculata</i> Saether		x	
<i>Porthastia</i> cfr. <i>longimanus</i> Kieffer			x
<i>Nilotanytus</i> sp.	x		
<i>Thienemannimyia</i> -group	x	x	
Nematoda	x	x	

TABLE 21. Continued

Taxon	Waugo. Point	Cook Riprap	Cook Ponar
Bryozoa	x	x	
Porifera			
Haplosclerina			
Spongillidae	x		
Tartigrada		x	
Mollusca			
Gastropoda			
Physidae			
<i>Physella integra</i> (Haldeman)	x		
<i>Physella vinosa</i> (Gould)		x	
Hydrobiidae			
<i>Amnicola</i> sp.			x
<i>Somatogyrus</i> sp.			x
Valvatidae			
<i>Valvata sincera</i> (say)			x
Pelecypoda			
Sphaeriidae	x		
<i>Pisidium</i> sp.			x

heated effluent. Most dramatic effects of the zoobenthos are direct and indirect influences of the riprap.

5. A wide variety of benthic species have colonized the riprap (crayfish, amphipods, mayflies, caddisflies) which would not normally have been present on the open lake bottom.

6. The riprap may have altered current and sedimentation patterns in the inner region influencing the distribution and abundance of sediment-dwelling taxa, particularly the Chironomidae.

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ECOLOGY OF THE MACROZOOBENTHOS IN THE VICINITY OF THE DONALD C. COOK NUCLEAR PLANT

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INTRODUCTION

There have been few attempts to define the structure and function of nearshore macrozoobenthic communities in the Great Lakes or any of the of the world's larger bodies of fresh water (Brinkhurst 1974, Mozley and Howmiller 1977) because so little has been published on life cycles, feeding, and habitat requirements of even the most common species. The discussion of the ecology of macrozoobenthos in southeastern Lake Michigan primarily is a review and summary of 13 years of data from the Donald C. Cook Nuclear Plant monitoring program coupled with reviews of available literature (in particular Mozley and Howmiller 1977) and observations from a variety of studies on the Great Lakes. The ecology of many of the taxa listed below is given in much greater detail in Great Lakes Research Division Reports on minor taxa (Zawacki 1985), Oligochaeta (LaDronka 1984), Pisidiidae (Zdeba and White 1985), Malacostraca (Winnell 1984a), and Chironomidae (Winnell 1984 b). These reports include the special methodology, tabulation, and statistical analysis of the data presented here.

Presented first is a brief discussion of the macrozoobenthic community by water depth zones from the shoreline outward. This is followed by specific qutecology summaries of minor taxa (Hydrozoa, Turbellaria, Hirudinea, Hydracarina, non-dipteran insects, Gastropoda), Oligochaeta (Tubificidae, Naididae, Lumbriculidae), Pisidiidae, Malacostraca (Amphipoda, Isopoda, Mysidacea, Decapoda), and Chironomidae.

MACROZOOBENTHIC COMMUNITIES

THE NEARSHORE <4 M DEEP

The zone of bottom sediments from the shoreline out to water depths of 3 to 4 m is subject to considerable wave action during most of the year and to ice shear during winter months. Thus, the substrata consist of coarse shifting sands with little buildup of silts or organic debris. This habitat historically has been thought of as one devoid of most benthic life (Usinger 1968). While it appears that densities of macrozoobenthos are very low in this zone in comparison with deeper zones, certain species have become adapted to this habitat. The nearshore community primarily is composed of Chironomidae, Naididae, and both *Sphaerium* and *Pisidium* in the Pisidiidae. These animals may be more lotic in their evolutionary origin (many of the same species are found in streams and rivers) and are adapted for burrowing

within the sand grains (Wiley 1980). The feeding habits of most of the chironomids and some of the naids are predaceous, although many of the naids consume benthic algae, and the psidiids filter bacteria and limited nutrients from the water column and from particles trapped in the sands. Most of the nearshore, macrozoobenthic community is probably eliminated by winter ice and re-colonizes in spring (Barton and Hynes 1976). Our knowledge of the biology of the very shallow areas is limited primarily because the ships generally used for sampling macrozoobenthos in the Great Lakes are not small enough to enter shallow waters.

THE 4 TO 32 M DEPTH ZONE

This region of the benthic environment represents a gradient from the high wave energy areas at the shoreline to the beginnings of the more stable profundal areas. Only for convenience do we put depth boundaries on this zone. The actual depth range of the zone is dependent on the slope of the bottom, substratum characters, current and wave patterns, etc. At the Cook Plant the following changes are noted along this lakeward gradient. There are changes in sediments from shifting sands to finer sands and from only patchy accumulations of clays and silts to a regular pattern of deposition. There is an increase in sediment patchiness with a maximum patchiness occurring between about 12 and 24 m. There is a decrease in the frequency with which wind- and storm-generated waves resuspend bottom particles. There is a decrease in mean annual temperature fluctuation approaching nearly year-around temperatures of 4°C beyond 32 m deep (Mozley and Winnell 1975).

Accompanying the physical gradient is a pattern of increasing macrozoobenthic density and diversity toward a more stable community in deeper waters (an idealized community is depicted in Fig. 1). The numbers and kinds of predators, herbivores, and particle feeders reach a maximum in association with the maximum patchiness or diversity of bottom types which, in turn, occurs in the middle of the gradient. Along the gradient, psidiids are the first group to show distinct increases in diversity. The fingernail clams are generally associated with firmer substrata which allow them some physical support while filtering overlying waters. Most of the rarer psidiids collected at the Cook Plant (as well as along most of the Lake Michigan shoreline, GLRD unpubl. data) occur in less than 12 m of water (Zdeba and White 1985). A similar shallow-water pattern is seen for several of the minor taxa including snails and leeches (Zawacki 1985).

Chironomidae and Oligochaeta (particularly the Tubificidae) are most diverse and abundant in substrata between 12 and 24 m deep (LaDronka 1984, Winnell 1984b). Unlike the mollusks and some minor taxa, these two taxa represent many trophic levels which have evolved in a variety of preferred substrates. While the chironomids of the extreme nearshore are constantly moving among the shifting sand grains, a larger portion of the deeper water species are tube builders, constructing more permanent dwellings. The tube-dwelling species actively pump water into the substrata and are presumed to be responsible for increasing sediment oxygen levels. In the Oligochaeta, the density and diversity of predatory and algal-feeding Naididae decrease beyond 12 m deep while the deposit-feeding Tubificidae and Lumbriculidae (*Stylodrilus*) increase in diversity and density. Another dominant taxon, the burrowing amphipod *Pontoporeia hoyi*, follows a similar pattern of increasing density with increasing deposition of fine organic and inorganic particles. The feeding actions of *Stylodrilus*, the tubificids, and *Pontoporeia* combine to homogenize

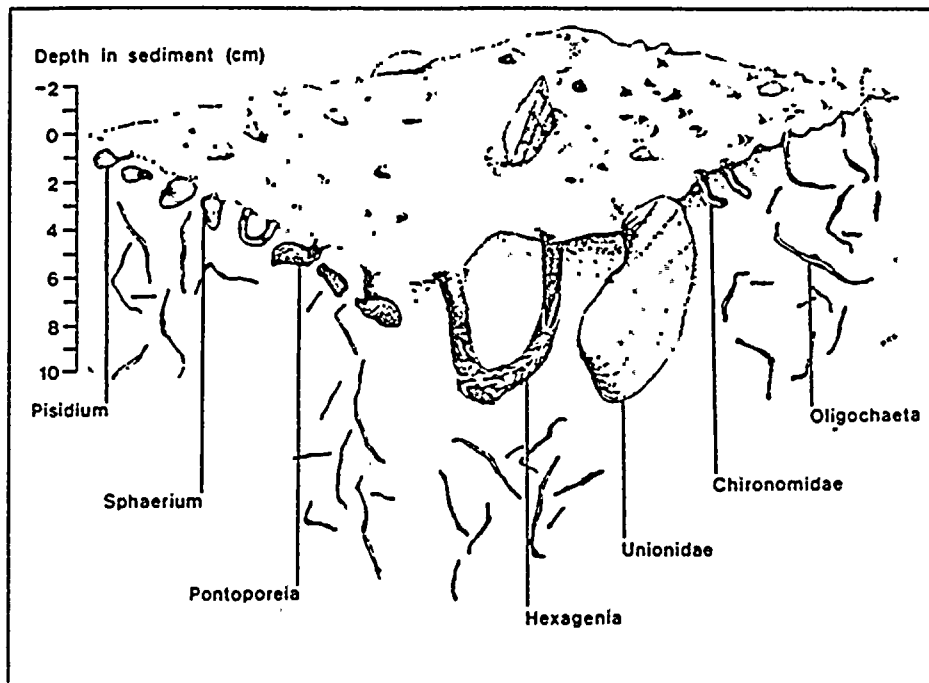


FIGURE 1. Idealized macrobenthic community.

(rework) surface sediments under two modes of bioturbation (Robbins 1982). The depth to which surface sediments are reworked is dependent on the rate of sedimentation which generally increases along the gradient. Bioturbation by *Pontoporeia* is eddy-diffusion, similar to the effect of small waves, while oligochaetes feed in a conveyor belt mode, bringing more deeply buried sediments to the surface. It is suspected at this time that sedimentation rates can affect depositional feeding by oligochaetes in that high sedimentation rates are favored by the more "pollution tolerant" species (Lauritsen et al. 1985) such as *Limnodrilus* and *Tubifex*, while the more sensitive species (e.g., *Stylodrilus heringianus*) function more effectively under low sedimentation rates. The intermediate species (listed as mesotrophic) probably function best in intermediate sedimentation rates. Further, it has been shown that, unlike many other closely related taxa, tubificid species may interact with each other, one species feeding upon the fecal remains of another, creating their own microcommunities (McCall and Fisher 1980).

THE 32 M AND DEEPER ZONE

Beyond 32 m, organism density begins to decrease but not as rapidly as does diversity. Most minor taxa disappear, as do Naididae, leaving a community composed primarily of Tubificidae, Pisidiidae, Chironomidae, and the amphipod *Pon-*

toporeia hoyi. These are taxa which utilize the increasing accumulations of silts and organic materials which reach a maximum deposition rate at depths of about 60 m in this portion of Lake Michigan (Edington and Robbins 1975). Species reaching maximum densities in the 32 to 60-m deep zone are *Tubifex tubifex*, *Potamothrix moldaviensis*, *Limnodrilus hoffmeisteri*, *Stylodrilus heringianus*, *Heterotrissocladius oliveri*, *Pisidium conventus*, and *Pontoporeia hoyi*. Beyond 60 m deep, when sedimentation rates decline, macrozoobenthos density and diversity decreases sharply, and the community consists primarily of four species: *Stylodrilus heringianus*, *Heterotrissocladius hoffmeisteri*, *Pisidium conventus*, and *Pontoporeia hoyi*.

AUTECOLOGY

Table 1 is a compilation of the taxa collected from the numerous surveys conducted at the Cook Plant and includes several species not listed in earlier reports. The systematics, taxonomys and identification of Great Lakes macrozoobenthos have greatly improved in recent years. At the onset of this study, many species were overlooked, misidentified, or lumped into higher taxonomic categories. The list of species in Table 1 reflects the degree of taxonomic detail now available. By example, we are now to identify virtually all Pisidiidae, including immatures, where only mature specimens were identified before the mid-1970s. The taxonomy of the Naididae is greatly improved, though we remain unable to identify most immature Tubificids. All Chironomidae larvae are now assignable to at least relatively narrow groups, although many cannot be separated with certainty in the early instars. Within all the major benthic taxa, there have been numerous synonyms, revisions, and name changes. The changes are given in more standard works on the identification of the various major taxa (e.g., Pennak 1978, Merritt and Cummins 1984).

MINOR TAXA

Hydrozoa (hydras), Turbellaria (flatworms), Hirudinea (leeches), Arachnoidea or Hydracarina (water mites), non-dipteran insects, and gastropods (snails) occur regularly along the nearshore areas of the Great Lakes, but usually in much lower densities or in much more patchy distributions than do the dominant, open-water Oligochaeta, Amphipoda, Chironomidae, and Pisidiidae (Table 1) (Zawacki 1985).

HYDROZOA

Hydra americana, the principal hydrozoan in the benthos at the Cook Plant, occurred in 11% of the lake samples with a mean density of approximately 15 m⁻² (Table 1). Maximum densities occurred in water between 20 and 24 m deep and usually were associated with some proportion of firm substratum (e.g., gravel, coarse sand). This sessile species is quite abundant on the riprap protecting the intake/discharge structures (Lauritsen 1979, Lauritsen and White 1981) and forms dense mats on the inside walls of the intake/discharge pipes where it feeds on small, entrained zooplankton.

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TABLE 1. Comprehensive species list for the 266 benthic macroinvertebrate taxa collected near the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1970-1982, including mean density (no. m⁻²), frequency of occurrence as a percentage of the total number of samples collected, and maximum observed density from a single Ponar grab sample. Taxa without data were collected between 1970 and 1982 but either were extremely rare (one or two occurrences) or were collected by means other than the Ponar grab.

Taxon	Mean Density	Frequency of Occurrence	Maximum Observed Density
Coelenterata			
Hydrozoa			
Hydroida			
Clavidae			
<i>Cordylophora lacustris</i>			
Hydridae			
<i>Hydra</i> nr. <i>americana</i>	14.8	11.1	2448
Platyhelminthes			
Turbellaria	29.9	21.2	1273
Tricladida			
Planariidae			
Neorhabdocoela			
Gyratricidae			
<i>Gyratrix hermaphroditus</i>			
Other Turbellaria			
Rotatoria			
Gastrotricha			
Nematoda			
Tardigrada			
Ectoprocta			
Annelida			
Branchiobdellida			
Hirudinea			
Pharyngobdellida			
Erpobdellidae			
<i>Dina parva</i>	0.1	0.2	61
<i>Mooreobdella microstoma</i>			
<i>Nephelopsis obscura</i>	0.4	1.0	182
Rhyncobdellida			
Glossiphoniidae			
<i>Alboglossiphonia heteroclita</i>			
<i>Glossiphonia complanata</i>	0.5	1.0	121
<i>Helobdella elongata</i>			
<i>Helobdella stagnalis</i>	18.0	16.0	1212
Piscicolidae			
<i>Myzobdella lugubris</i>			
Other Hirudinea			

TABLE I. Continued

Taxon	Mean Density	Frequency of Occurrence	Maximum Observed Density
Oligochaeta			
Haplotaxida			
Acleosomatidae			
<i>Acleosoma</i> sp.			
Enchytraeidae	2.0	1.5	788
Naididae			
<i>Amphichaeta leydigii</i>	0.2	0.3	121
<i>Arceonais lomondi</i>	1.2	1.0	545
<i>Chaetogaster diaphanus</i>	74.5	17.0	7333
<i>Chaetogaster diastrophus</i>	0.1	0.1	61
<i>Chaetogaster limnaei</i>			
<i>Chaetogaster setosus</i>			
<i>Dero digitata</i>	<0.1	0.1	61
<i>Nais alpina</i>			
<i>Nais barbata</i>			
<i>Nais behningi</i>	<0.1	<0.1	20
<i>Nais bretscheri</i>	<0.1	<0.1	20
<i>Nais communis</i>			
<i>Nais elinquis</i>			
<i>Nais pardalis</i>			
<i>Nais pseudobtusa</i>	<0.1	<0.1	61
<i>Nais simplex</i>	<0.1	<0.1	61
<i>Nais variabilis</i>	33.3	15.3	4606
<i>Ophidonais serpentina</i>	2.4	0.8	3631
<i>Piquetiella michiganensis</i>	112.0	33.9	6666
<i>Pristina foreli</i>	1.9	0.7	2000
<i>Pristina longiseta</i>	<0.1	<0.1	61
<i>Pristina osborni</i>	<0.1	<0.1	61
<i>Pristina sima</i>			
<i>Slavina appendiculata</i>	<0.1	<0.1	61
<i>Specaria josinae</i>	0.3	0.3	182
<i>Stylaria lacustris</i>	32.3	14.1	3212
<i>Uncinais uncinata</i>	43.2	24.3	1697
<i>Vejdovskyella comata</i>			
<i>Vejdovskyella intermedia</i>	7.6	5.3	1394
<i>Wapsa mobilis</i>	<0.1	<0.1	20
Tubificidae			
<i>Aulodrilus americanus</i>	4.9	3.3	909
<i>Aulodrilus piqueti</i>	<0.1	<0.1	20
<i>Aulodrilus pluriseta</i>	122.6	10.1	30724
<i>Ilyodrilus templetoni</i>	0.7	0.5	364
<i>Isochaetides freyi</i>	15.5	11.5	1091
<i>Limnodrilus angustipenis</i>	7.3	6.3	788
<i>Limnodrilus cervix</i>	8.1	3.4	2969
<i>Limnodrilus claparedeianus</i>	1.4	1.3	424
<i>Limnodrilus hoffmeisteri</i>	134.8	33.3	8811
<i>Limnodrilus profundicola</i>	4.2	6.0	606
<i>Limnodrilus spiralis</i>	8.2	6.4	1757
<i>Limnodrilus udekemianus</i>	0.9	1.1	326

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TABLE 1. Continued

Taxon	Mean Density	Frequency of Occurrence	Maximum Observed Density
<i>Potamothenix bedoti</i>	0.2	0.3	121
<i>Potamothenix hammoniensis</i>	<0.1	<0.1	61
<i>Potamothenix moldaviensis</i>	36.0	24.0	1818
<i>Potamothenix vejdevskyi</i>	263.5	29.3	21210
<i>Quistadrilus multisetosis multisetosis</i>	11.1	4.2	3575
<i>Quistadrilus multisetosis longidentus</i>	0.2	0.2	182
<i>Rhyacodrilus coccineus</i>	1.8	1.2	848
<i>Spirosperma nikolskyi</i>	<0.1	<0.1	18
<i>Tubifex kessleri americanus</i>	<0.1	<0.1	61
<i>Tubifex superiorenensis</i>	0.8	0.7	303
<i>Tubifex tubifex</i>	10.5	4.9	5818
Lumbriculida			
Lumbriculidae			
<i>Stylodrilus heringianus</i>	949.6	45.7	20363
Mollusca			
Gastropoda			
Pulmonata			
Basommatophora			
Lymnaeidae	6.0	7.0	970
<i>Lymnaea stagnalis appressa</i>			
Other Lymnaeidae			
Physidae	0.6	1.0	182
<i>Physella integra</i>			
<i>Physella</i> spp.			
Prosobranchia			
Mesopastropoda			
Hydrobiidae			
<i>Amnicola limosa</i>			
<i>Amnicola walkeri</i>			
<i>Amnicola</i> (above 2 spp.)	1.8	2.5	303
<i>Bythinia tentaculata</i>	0.5	0.6	182
<i>Somatogyrus subglobosus</i>	2.0	2.0	545
Valvatidae			
<i>Valvata sincera</i>	32.8	23.2	1333
<i>Valvata tricarinata</i>	<0.1	<0.1	61
Other Gastropoda			
Pelecypoda			
Heterodonta			
Pisidiidae			
<i>Pisidium (Cyclocalyx) adamsi</i>	0.4	0.1	121
<i>Pisidium (Cyclocalyx) casertanum</i>	172.9	39.1	4787
<i>Pisidium (Cyclocalyx) compressum</i>	6.7	6.3	545
<i>Pisidium (Cyclocalyx) fallax</i>	22.2	15.1	2000
<i>Pisidium (Cyclocalyx) ferrugineum</i>	2.6	1.7	364
<i>Pisidium (Cyclocalyx) henslowanum</i>	40.3	19.9	1879
<i>Pisidium (Cyclocalyx) lilljeborgi</i>	72.3	15.5	3333
<i>Pisidium (Cyclocalyx) milium</i>	11.0	5.3	667
<i>Pisidium (Cyclocalyx) nitidum</i> f. <i>nitidum</i>	147.2	24.8	5272

TABLE 1. Continued

Taxon	Mean Density	Frequency of Occurrence	Maximum of Observed Density
<i>Pisidium (Cyclocalyx) nitidum</i> f. <i>pauperculum</i>	4.9	6.3	424
<i>Pisidium (Cyclocalyx) subtruncatum</i>	32.0	6.0	2545
<i>Pisidium (Cyclocalyx) variabile</i>	14.3	10.4	970
<i>Pisidium (Cyclocalyx) ventricosum</i>	0.7	0.3	364
<i>Pisidium (Cyclocalyx) walkeri</i>	1.7	1.3	545
<i>Pisidium (Neopisidium) conventus</i>	297.2	11.1	9393
<i>Pisidium (Pisidium) amnicum</i>	0.9	2.1	182
<i>Pisidium (Pisidium) dubium</i>			
<i>Pisidium (Pisidium) idahoense</i>	0.4	0.7	121
<i>Musculium securis</i>	<0.1	<0.1	18
<i>Musculium transversum</i>	9.5	0.8	182
<i>Sphaerium (Sphaerium) corneum</i>	2.3	1.3	781
<i>Sphaerium (Sphaerium) nitidum</i>	95.0	26.2	4484
<i>Sphaerium (Sphaerium) rhomboidium</i>	0.2	0.3	121
<i>Sphaerium (Sphaerium) striatinum</i>	12.2	14.2	788
Arthropoda			
Arachnoidea	4.8	7.8	606
Acari			
Trombidiformes			
Hygrobatidae			
<i>Hygrobatas longipalpis</i>			
Lebertiidae			
<i>Lebertia porosa</i>			
Pionidae			
<i>Forelia</i> sp.			
<i>Piona rotunda</i>			
<i>Piona</i> sp.			
Sperchonidae			
<i>Sperchon</i> sp.			
Sarcoptiformes			
Eremaeidae			
<i>Hydrozetes</i> sp.			
Crustacea			
Branchiopoda			
Diplostraca			
Cladocera			
Copepoda			
Calanoida			
Cyclopoida			
Harpacticoida			
Ostracoda			
Podocopa			
Malacostraca			
Amphipoda			
Gammaridae	0.9	2.0	449
<i>Gammarus fasciatur</i>			
<i>Gammarus pseudolimnaeus</i>			
<i>Gammarus troglolophilus</i>			

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TABLE 1. Continued.

Taxon	Mean Density	Frequency of Occurrence	Maximum Observed Density
Haustoriidae			
<i>Pontoporeia hoyi</i>	2208.8	67.8	30664
Hyalellidae			
<i>Hyalella azteca</i>	<0.1	<0.1	61
Decapoda			
Astacidae			
<i>Cambarus diogenes</i>			
<i>Orconectes propinquus</i>			
<i>Orconectes virilis</i>			
Isopoda			
Asellidae	0.1	0.2	61
<i>Asellus intermedius</i>			
<i>Asellus racovitzae racovitzae</i>			
Mysidaceae			
Mysidae			
<i>Mysis relicta</i>	9.3	9.1	667
Insecta			
Collembola			
Entomobryidae			
<i>Orchesella</i> sp.			
Onychiuridae			
<i>Hypogastrura</i> sp.			
Ephemeroptera			
Baetidae			
<i>Baetis</i>			
Baetiscidae			
<i>Baetisca</i> sp.			
Caenidae			
<i>Caenis</i> sp.			
Ephemerellidae			
<i>Ephemerella</i> sp.			
Ephemeridae			
<i>Hexagenia</i> sp.			
Heptageniidae			
<i>Heptagenia</i> sp.			
<i>Stenacron</i> sp.			
<i>Stenonoma</i> sp.			
Leptophlebiidae			
<i>Leptophlebia</i> sp.			
Tricorythidae			
<i>Tricorythodes</i> sp.			
Odonata			
Coenagrionidae			
<i>Ischnura/Anomalagrion</i> sp.			
<i>Nehalennia</i> sp.			
Gomphidae			
<i>Gomphus</i> sp.			

TABLE 1. Continued

Taxon	Mean Frequency of Observed		
	Density	Occurrence	Maximum Density
Hemiptera			
Belostomatidae			
<i>Lethocerus</i> sp.			
Corixidae			
Notonectidae			
<i>Notonecta</i> sp.			
Pleidae			
<i>Plea striola</i>			
Trichoptera			
Hydropsychidae			
<i>Cheumatopsyche</i> sp.			
<i>Hydropsyche</i> sp.			
* <i>Potamyia flava</i>			
<i>Symphitopsyche</i> sp.			
Hydroptilidae			
<i>Hydroptila</i> sp.			
*nr. <i>Nectopsyche</i> sp.			
Phryganeidae			
<i>Banksiola</i> sp.			
Polycentropodidae			
<i>Polycentropus</i> sp.			
Coleoptera			
Curculionidae			
Dytiscidae			
<i>Deronectes</i> sp.			
Elmidae			
<i>Dubiraphia</i> sp.			
<i>Optioservus</i> sp.			
*nr. <i>Stenelmus</i>			
Diptera			
Ceratopogonidae			
<i>Bezzia/Palpomyia</i> sp.	<0.1	<0.1	20
<i>Ceratopogon</i> sp.	<0.1	<0.1	61
<i>Culicoides</i> sp.	<0.1	<0.1	41
<i>Probezzia</i> sp.	<0.1	<0.1	41
* <i>Sphaeremais</i>			
Chaoboridae			
<i>Chaoborus puntipennis</i>			
Chironomidae			
Chironominae			
Chironomini			
<i>Chironomus anthracinus</i> -gr. (2 spp.)	44.3	10.2	11150
<i>Chironomus fluviatilis</i> -gr. (2 spp.)	157.3	46.5	5393
<i>Chironomus halophilus</i> -gr.	<0.1	0.2	45
<i>Cladopelma</i> sp.			
<i>Cryptochironomus</i> cf. <i>digitatus</i>			
<i>Cryptochironomus</i> cf. <i>fulvus</i>			
<i>Cryptochironomus</i> (above 2 spp.)	30.9	30.4	970

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TABLE 1. Continued

Taxon	Mean Density	Frequency of Occurrence	Maximum Observed Density
<i>Cryptochironomus rolli</i>	1.2	2.5	184
<i>Cryptochironomus</i> sp. 3	6.8	7.9	848
nr. <i>Cryptochironomus</i> (sp.1)	2.4	3.8	242
* <i>Cyphomella</i> sp:			
<i>Demicryptochironomus vulneratus</i>	0.4	0.7	121
<i>Dicrotendipes modestus</i>	<0.1	0.2	61
<i>Endochironomus</i> cf. <i>nigricans</i>			
<i>Glyptotendipes</i> sp. 1			
<i>Glyptotendipes</i> sp. 2			
<i>Glyptotendipes</i> (above 2 spp.)	<0.1	0.1	61
<i>Harnischia</i> cf. <i>amachaerus</i>			
<i>Harnischia</i> <i>grisea</i>			
<i>Harnischia</i> sp.	0.2	0.3	182
<i>Kiefferulus</i> sp.	<0.1	0.3	55
<i>Microtendipes</i> cf. <i>tarsalis</i>			
<i>Parachironomus abortivus</i>	<0.1	<0.1	61
<i>Parachironomus frequens</i>			
<i>Parachironomus monochromus</i>			
<i>Parachironomus</i> cf. <i>pectinatellae</i>			
<i>Paracladopelma</i> cf. <i>nereis</i>	0.5	0.4	121
<i>Paracladopelma undine</i>	6.7	6.5	424
<i>Paracladopelma winnelli</i>	9.6	12.0	485
<i>Phaenopsectra</i> (<i>Phaenopsectra</i>) cf. <i>flavipes</i>			
* <i>Phaenopsectra</i> (<i>Tribelos</i>) sp.			
<i>Phaenopsectra</i> sp.	<0.1	0.2	61
<i>Polypedilum</i> (<i>Polypedilum</i>) cf. <i>convictum</i>			
<i>Polypedilum</i> (<i>Polypedilum</i>) cf. <i>tuberculum</i>	2.3	2.8	242
<i>Polypedilum</i> (<i>Tripodura</i>) <i>scalaenum</i>	28.4	12.8	5636
<i>Polypedilum</i> (<i>Tripodura</i>) cf. <i>simulans/digitifer</i>			
<i>Polypedilum</i> sp.	<0.1	<0.1	61
<i>Robackia demeijerei</i>	26.1	12.8	5515
<i>Robackia pilicauda</i>			
<i>Saetheria tylus</i>	65.9	23.7	2606
<i>Stictochironomus</i> sp.	<0.1	<0.1	20
Pseudochironomini			
<i>Pseudochironomus</i> cf. <i>fulviventris</i>			
Tanytarsini			
<i>Cladotanytarsus</i> sp.	35.6	12.9	6242
<i>Micropsectra</i> sp.	19.2	11.1	5575
<i>Paratanytarsus</i> sp.	<0.1	<0.1	20
<i>Rheotanytarsus</i> sp.	0.1	0.2	61
<i>Stempellina</i> cf. <i>baisei</i>			
<i>Tanytarsus</i> sp.	0.2	0.3	121
Orthocladiinae			
<i>Brillia</i> cf. <i>parva</i>			
<i>Cricotopus</i> (<i>Cricotopus</i>) cf. <i>annulator</i>			
<i>Cricotopus</i> (<i>Cricotopus</i>) cf. <i>bicinctus</i>			
<i>Cricotopus</i> (<i>Cricotopus</i>) nr. <i>bicinctus</i> -gr.			

TABLE 1. Continued

Taxon	Mean Frequency of		Maximum
	Density	Observed Occurrence	Density
<i>Cricotopus (Cricotopus) festivellus</i> -gr.			
<i>Cricotopus (Cricotopus) fuscus</i> -gr.			
<i>Cricotopus (Cricotopus) tibialis</i> -gr.			
<i>Cricotopus (Cricotopus) cf. tremulus</i>			
<i>Cricotopus (Cricotopus) tremulus</i> -gr.			
<i>Cricotopus (Cricotopus) cf. triannulatus</i>			
<i>Cricotopus (Cricotopus) trifascia</i> -gr.			
<i>Cricotopus (Cricotopus) cf. vierriensis</i>			
<i>Cricotopus (Isocladius) cf. suspiciosus</i>			
<i>Cricotopus (Isocladius) cf. sylvestris</i>			
<i>Cricotopus (Isocladius) sylvestris</i> -gr.			
<i>Doplocadius</i> sp.			
<i>Heterotrissocladius changi</i>	12.8	11.8	909
<i>Heterotrissocladius oliveri</i>	5.3	3.8	667
<i>Hydrobaenus cf. pilipes</i>	<0.1	0.1	61
<i>Limnophyes</i> sp.			
<i>Nanocladius (Nanocladius) cf. anderseni</i>			
<i>Nanocladius (Nanocladius) cf. crassicornus</i>			
<i>Nanocladius (Nanocladius) cf. distinctus</i>			
<i>Nanocladius (Nanocladius) cf. spinipennis</i>			
<i>Orthocladius (Orthocladius) cf. obumbratus</i>			
<i>Paracricotopus</i> sp.			
<i>Parakiefferiella</i> sp.			
<i>Psectrocladius (Psectrocladius) simulans</i>	2.3	3.5	545
<i>Psectrocladius</i> sp. 1			
<i>Rheocricotopus</i> sp. 1			
<i>Rheocricotopus</i> sp. 2			
<i>Smittia</i> sp.			
<i>Orthoclaudiini</i> spp.			
Diamesinae			
<i>Potthastia longimanus</i>	3.6	6.4	242
Prodiamesinae			
<i>Monodiamesa tuberculata</i>	13.0	16.8	606
Tanypodinae			
Macropelopiini			
* <i>Macropilopia</i> sp.			
<i>Psectrotanypus (Psectrotanypus)</i> sp.			
Natarsiini			
nr. <i>Natarsia</i> sp.			
Pentaneurini			
<i>Larsiasp.</i>			
<i>Nilotanypus</i> sp.			
<i>Thienemannimyia</i> -gr. (prob. <i>Conchapelopia</i>)	0.2	0.4	61
Procladiini			
<i>Procladius (Holotanypus) denticulatus</i>			
<i>Procladius (Holotanypus)</i> sp.			
<i>Procladius</i> (above 2 spp.)	38.8	21.1	3575
Culicidae			
nr. <i>Aedis excrucians</i>			

TABLE 1. Continued

Taxon	Mean Frequency of Density Occurrence	Maximum Observed Density
Scatopsidae		
nr. <i>Rexoza</i> sp.		
Simuliidae		
<i>Simulium</i> sp. 1		
<i>Simulium</i> sp. 2		
Stratiomyidae		
<i>Odontomyia (Odontomyiina)</i> sp.		
Tipulidae		
<i>Tipula</i> sp.		

TURBELLARIA

The flatworms occurred in 21% of the lake samples with a mean density of about 30 m⁻² (Table 1). Although several species were present, only *Gyratrix hermaphrodita* was identified with the shipboard preservative techniques used. As a taxon, turbellarians were most abundant in water between 16 and 20 m deep in regions of soft substrata with high proportions of silt and clay. Turbellaria reached peak densities in July and other "summer" months when nearshore waters were warmest.

HIRUDINEA

Eight species of leeches were identified from the survey area, but only the predatory, *Helobdella stagnalis* was particularly abundant (mean density of 18 m⁻², Table 1). Unlike many leech species, *H. stagnalis* was not confined to shallow waters but had highest densities (24 to 40 m⁻²) in the 12- to 24-m depth range. There seemed to be at least two generations per year in this portion of Lake Michigan, with maximum recruitment in spring and early summer. *Helobdella stagnalis* has been described in the literature as primarily inhabiting macrophytes, but they have been collected from every other type of substratum from boulders to silt (Pennak 1978). We recorded highest abundances in areas of fine particulates, which corresponded with highest densities of potential prey items such as oligochaetes and larval chironomids.

ARACHNOIDEA (HYDRACARINA)

Water mites, lumped for convenience under "Hydracarina," were not particularly abundant in the vicinity of the Cook Plant (4.8 m⁻²), and few specimens were identified beyond "Acari." cursory identification indicated that at least seven genera were present (Table 1); all are predaceous, feeding on benthic and planktonic ostracods, copepods, and cladocerans (Pennak 1978). Maximum densities occurred in

samples taken in coarse substrata in water less than 16 m deep. No specimens were present in water deeper than 20 m, or in silts or clays.

NON-DIPTERAN INSECTS

Many genera of non-dipteran insects have been reported from the open Great Lakes, and several are inhabitants of rocky shoals or artificial riprap. Few of the non-dipteran taxa, however, occur as a regular part of the open-lake zoobenthos. In the area around the Cook Plant, the frequency of occurrence in all samples was about 1% (Table 1). The Mayfly genera *Caenis*, *Leptophlebia*, *Stenonema*, and *Tricorythodes* and the caddisfly *Oecetis* were present in silty substrata in water between 4 and 16 m deep, a habitat that would be expected to support at least small numbers of these silt-dwelling taxa (Merritt and Cummins 1984). Emergence to the adult stage for most of the above taxa is from the water surface. Because lake water depths are much greater than the normal stream's water depths they are adapted to, it is questionable if any individuals ever emerge and mate; thus, are they truly a part of the lake fauna?

Several other taxa of non-dipteran insects which most likely originated from the riprap were found in entrainment samples (e.g., odonates and hemipterans, Table 1). As with mayfly and caddisfly larvae, these taxa might best be considered as erratics occurring sporadically but with no established lake populations.

GASTROPODA

The gastropod families Valvatidae, Hydrobiidae, Lymnaeidae, and Physidae were a constant but minor portion of the benthic fauna accounting for 0.6% of the total macroinvertebrates with a mean abundance of 46 m⁻². Overall, greatest densities occurred in water between 16 and 20 m deep where they were associated with the finer sediment types ranging from silty sand to sandy clay.

Of the nine gastropods identified, the lymnaeid snail *Lymnaea stagnalis appressa*, the second-most abundant species, was present in 7% of the lake samples and accounted for 13% of the total gastropod fauna (Table 1). Maximum densities occurred in water between 16 and 24 m deep in soft sediments. Little was learned about the life cycle because of high variability in density among samples; however, it was present year around, with highest densities in September through November. Hydrobiids (four species, Table 1) accounted for 9% of the total gastropod density and occurred in 5% of all samples. Maximum densities were in water between 8 and 12 m deep in sandy substrata. Again, none occurred deeper than 24 m in the survey area.

In the family Valvatidae, *Valvata sincera* is the most widely distributed of all snails in the Great Lakes (Mackie et al. 1980). In the survey area, this species accounted for 72% of the total gastropods and occurred in 23% of the lake Ponar samples (Table 1). *Valvata tricarinata* was extremely rare. As with *Lymnaea*, *V. sincera* was most abundant in deeper waters (20 to 24 m) in fine substrata, primarily sandy silt. This species was present in all months, but little about the life cycle can be inferred because of low monthly abundances.

All the snail species found in the Cook Plant area are known to occur in the Great

Lakes (including *Physella*: Physidae), inhabiting the softer silts where they feed on organic debris and benthic algae (Mackie et al. 1980).

OLIGOCHAETA

The oligochaeta or aquatic earthworms are the dominant taxa of the Great Lakes (Mozley and Howmiller 1977, LaDronka 1984) and the Cook Plant survey area. Further, they are a very diverse group of macrozoobenthos, second only to the dipterans. Mean density was $3,789 \text{ m}^{-2}$ which was equal to 48% of all organisms collected in the lake surveys. Mean abundance increased with depth, from a minimum of 101 m^{-2} shallower than 4 m to a maximum of $11,765 \text{ m}^{-2}$ between 32 and 60 m deep. At water depths greater than 12 m, frequency of occurrence in samples was nearly 100%. Overall mean densities were highest in October and November and lowest in August and December. Abundance was lowest in coarse sediments (gravel and coarse sand) and highest in fine-grained sediments (silts to loam). Most taxa (with the exception of some Naididae) are particle feeders consuming organic detritus and associated bacteria. Thus, they are associated with depositional zones (McCall and Fisher 1980).

Five oligochaete families were present, three of which were abundant. Aeolosomatidae (*Aeolosoma*) and Enchytraeidae were rare in samples, and the use of a large, 500 μm sieve may have greatly contributed to the underestimation of both of these very small worms. Enchytraeids (at least two undetermined taxa) averaged 2.0 m^{-2} and were collected in only 1.5% of the lake samples (Table 1).

TUBIFICIDAE

Twenty-three taxa of Tubificidae occurred in the study (Table 1), all of which previously had been recorded from Lake Michigan (Spencer 1980). The overall mean abundance of tubificids was $2,563 \text{ m}^{-2}$, with a maximum single abundance of $112,960 \text{ m}^{-2}$ from a "loamy" substratum in water in the 12-16 m depth range. Unidentified immatures lacking hair chaetae accounted for the largest proportion (64.5%) of the overall mean. Of the identified specimens, only four species comprised >1% of the mean tubificid density (*P. vej dovski* 12.1%, *L. hoffmeisteri* 6.2%, *A. pluriseta* 5.6%, and *P. moldaviensis* 1.7%, see Table 1). When immatures without hair chaetae were apportioned, *L. hoffmeisteri* became the numerically dominant taxon, accounting for 63% of the total Tubificidae.

The mean density of mature *L. hoffmeisteri* increased with water depth; the lowest densities occurred in the 12- to 48-m depth zone ($218\text{--}319 \text{ m}^{-2}$). The increase in density was related to increases in fine-grained sediment, which also reached a maximum in water at 28 to 32 m deep ($2,832 \text{ m}^{-2}$), with continuing high densities down to a depth of 58 m. The mean abundance of mature *L. hoffmeisteri* was highest (161 to 265 m^{-2}) from May through July, decreased sharply from July to August, and then reached a minimum in December (3 m^{-2}). Because the majority of unidentified immatures most likely were *L. hoffmeisteri*, it followed that the population density of this species was highest in October, when immatures reached maximum abundance. Spring-summer maxima and autumn-winter minima of mature worms has been observed in other Lake Michigan studies (Winnell and Jude 1979, 1980, 1981, 1982), but their actual life history remains unclear. Other *Limnodrilus*

occurring in the study area were *L. udekemianus*, *L. spiralis*, *L. cervex*, *L. profundicola*, and *L. claparedeianus*, each of which averaged less than 10 m⁻². Except for *L. udekemianus*, these species could be identified only in the mature state which probably resulted in underestimates of their densities.

Potamothrix vejovskyi is common in the Great Lakes and is characteristic of mesotrophic habitats (Mozley and Howmiller 1977, Lauritsen et al. 1985). The mean density of adults was the highest for any tubificid species in the study area (263 m⁻², Table 1). The density generally increased from low numbers in water less than 12 m deep (1-6 m⁻²) to a maximum in water between 28 to 32 m deep (1,493 m⁻²). Some seasonal trends were evident, with peak abundances occurring in May, July, and October. The greatest frequencies of occurrence and densities were in the finest substrata (combinations of silts and clays), but a number of specimens also occurred in patches of gravel.

Potamothrix moldaviensis often co-occurs with *P. vejovskyi* but has a much more littoral distribution (Hiltunen 1969). The mean abundance of mature worms was 36 m⁻², and the frequency of occurrence was 29.3%. Maximum densities were reached in water between 28-32 m deep, with significant numbers also occurring between 8 and 24 m deep. On an annual cycle, *P. moldaviensis* reached maximum densities in May and July, but little can be said about life cycles because the immatures cannot be positively separated from other tubificids. This species tends to prefer finer substrata with a sand component (i.e., very fine sand, clayey sand, and sandy clay), as has also been shown by Stimpson et al. (1975). Other *Potamothrix* species were relatively uncommon.

All species of *Aulodrilus* are surface dwellers, rarely burrowing deeply into substrate (Milbrink 1973). *Aulodrilus pleuriseta* is the most common *Aulodrilus* in the Great Lakes and was the third most abundant tubificid at the Cook Plant (122.6 m⁻², Table 1). The maximum single abundance in a sample was 30,724 m⁻² from loam in water in the 12- to 16-m depth range. Three distinct peaks of abundance occurred in April, July, and October. The highest mean densities and frequencies of occurrence were in the finest substrata (sandy silt, sandy clay, and loam) which occurred patchily in water between 12 and 24 m deep. *Aulodrilus americana* and *A. piqueti* were infrequently found in lake samples, and most occurrences were in loam or very fine particles, similar to *A. pleuriseta*.

Isochaetides freyi is uncommon in the Great Lakes (Mozley and Howmiller 1977), and the densities found at the Cook Plant appear to be the highest recorded (16 m⁻², Table 1). The maximum abundance was in water in the 12- to 16-m depth range in gravel or sand which contained high percentages of silt and clay. There were two peaks of seasonal abundance of matures with a distinct maximum in July and a minor peak in November.

Quistadrilus multisetosus multisetosus was not found in water less than 12 m deep in the survey area but was relatively abundant between 20 and 24 m deep (84 m⁻², Table 1). Like the majority of tubificids collected at the Cook Plant, *Q. m. multisetosus* attained highest mean densities in the finer sediments which were common in water depths greater than 20 m. No seasonal pattern was evident, although the frequency of occurrence did appear to increase slightly in autumn. A second form, *A. m. longidentus*, was quite rare, occurring only four times in the 13 years of study.

One of the most cosmopolitan and widespread species, *T. tubifex*, is common in nearshore as well as deeper waters in a wide variety of trophic conditions (Birnkurst 1969). At the Cook Plant, *T. tubifex* (matures only) attained an overall mean density of 10 m⁻² (Table 1) and comprised less than 1% of the overall mean

tubificid density. The maximum observed density was $5,818 \text{ m}^{-2}$ from a sample in silty sand in water in the 16- to 20-m depth range. Overall, *T. tubifex* was calculated to account for approximately 82% of unidentified immature tubificids with hair chaetae. When these immatures were apportioned, density estimates rose to 126 m^{-2} or 6% of the total tubificids. *Tubifex tubifex* was sparse at depths less than 12 m but reached peak densities in water between 16 and 20 m deep (33 m^{-2}). Mature specimens were most common in June but present in low numbers in all other months. The preferred habitat types for this particle feeder were sandy silt and loam.

The species *tubifex kessleri americanus*, *Tubifex superiorensis* (A Great Lakes endemic?), *Spirosperma nikolskyi*, *Ilyodrilus templetoni*, and *Rhyacodrilus coccini* were present in the survey area, primarily in silty substrata, but all were found in very low densities and with low frequencies of occurrence.

NAIDIDAE

Thirty species of Naididae have been identified from the various types of collections made in this study, 14 of which had not been listed previously for Lake Michigan (Spencer 1980) and one (*Pristina sima*) which was new to the Great Lakes. This number of species is high in comparison with other Great Lakes studies and reflects the amount of sampling effort and a variety of sampling methods in the nearshore region. Twenty-one species were collected in the regular lake Ponar samples, of which only six were common. An additional nine species were collected by gear other than the Ponar (e.g., rock baskets, entrainment). Most species occurred in water depths, naids averaged 3.44 m^{-2} and occurred in 53% of all Ponar grab samples. These numbers equate to approximately 5% of the total benthos and 10% of the total oligochaetes between 0 and 24 m deep.

Piquetiella michiganensis is one of the most common naids found in the nearshore areas of Lake Michigan (Hiltunen 1969) and was dominant at the Cook Plant (112 m^{-2} , Table 1), occurring in 34% of all samples taken in water less than 24 m deep. Guts of specimens were usually packed with fine, sandy sediments mixed with detritus and diatoms. This species appears to be a burrowing detritus-feeder, living more deeply within the substratum than many other naids which exist only in surficial layers. Relatively few specimens have been found in the water column (Wiley and Mozley 1978) or in entrainment samples. Reproduction for many naids is through fragmentation as well as sexual means. Seasonality data for *P. michiganensis* was unique among species collected at the Cook Plant with two peak periods of abundance. The early summer peak (June-July) may represent asexual reproduction, and the late summer and autumn (August-October) peak may result from sexual reproduction. The mean density tended to decrease with increasing fineness of the sediment type, which again was reflected in gut contents.

The second-most abundant naidid, *Chaetogaster diaphanus* (75 m^{-2} , Table 1), is a predator common in a variety of habitats from sewage filter beds to the Great Lakes (Learner et al. 1978). Like *P. michiganensis*, this species rarely occurs in pelagic samples (Wiley and Mozley 1978); however, it does occur frequently in entrainment samples, indicating that a large population may exist on the riprap structures around the water intake structures (Lauritsen 1979, Lauritsen and White 1981). The maximum densities were in water in the 12 to 16-m depth range, without a clear preference for a specific sediment type. The greatest abundance of this species was in July, with few specimens being found either in early spring or late autumn.

Uncinaiis uncinata was the third-most abundant species (43 m^{-2} , Table 1). This large naidid is often collected with *P. michiganensis* in the Great Lakes, but usually in very shallow, wave-swept areas in water less than 2 m deep (Hiltunen and Klemm 1980). Guts were packed with sand grains, and it is probably that *Uncinaiis* feeds on epipsammic flora and fauna (Timm 1970, Stevenson and Stoermer 1981). The greatest abundance and frequency of occurrence were in silty and clayey sands, which may be related to the possible nutritional benefit associated with these particle sizes. Peak seasonal abundance was in July, a pattern similar to most common naidids.

Ten species of *Nais* were collected of which only *N. variabilis* was common. The remaining nine species were low in density and frequency (Table 1). *Nais variabilis* is commonly found in a variety of habitats from shallow streams to moderately deep water of the Great Lakes, but its small size causes often severe underestimation of densities (Hiltunen and Klemm 1980). This species ranked fourth among naidids in overall mean density (33 m^{-2}) and frequency (15%) in lake samples. The maximum density was in clean, coarse sand in water between 4 and 8 m deep ($4,606 \text{ m}^{-2}$). *Nais variabilis* was one of the most abundant benthic animals collected in pelagic net-tows at depths of 6 m (0.14 m^{-3}) and 9 m (0.05 m^{-3}) (Wiley and Mozley 1978) and was very frequently seen in entrainment samples collected from the forebay in 1975. These data reflect the tendency of *N. variabilis* to enter the water column as well as its abundance on the surrounding riprap (Lauritsen 1979). The distribution of *N. variabilis* was limited to nearshore sandy areas, reaching a maximum density of 67 m^{-2} in water between 4 and 8 m deep. Specimens were encountered infrequently at depths greater than 16 m. As with other naidids, maximum abundance was in July, with very few specimens taken in early spring or autumn. Its substratum preference was sand (including coarse sand) where *N. variabilis* ingests deposited organic particles.

Stylaria lacustris is an omnivore (Timm 1970) and herbivore (Streit 1978) common to all the Great Lakes (Brinkhurst 1967). In this study, it ranked fifth among naidids in lake samples (32 m^{-2} , Table 1). Like *N. variabilis*, *S. lacustris* is a swimmer and was abundant in pelagic net tow samples (Wiley and Mozley 1978) and in entrainment samples. The maximum density was in water between 12 and 16 m deep, and none was found in water greater than 24 m deep. There were no distinct substratum preferences; however, seemingly greater densities occurred in coarse sand.

Of the two *Vejdovskyella* species, *V. comata* was extremely rare in the survey area. The second species, *V. intermedia*, was common and is widely distributed throughout the Great Lakes (Spencer 1980, Hiltunen and Klemm 1980), but its very small size makes population estimates inaccurate when large mesh sieves are used. *Vejdovskyella intermedia* ranked sixth among naidids (7.6 m^{-2} , Table 1) in lake samples, although the numbers are probably greatly underestimated. This species does not occur frequently in the water column (Wiley and Mozley 1978) and is not common on the riprap (Lauritsen 1979). Maximum densities were in water between 12 and 20 m deep ($15\text{--}16 \text{ m}^{-2}$) in finer substrata where it most likely is a particle feeder.

Other rare naidid species (Table 1) taken in Ponar grabs, pelagic tows, entrainment, and the riprap colonization studies included *Amphichaeta leydigii*, *Architeonais lomondi*, *Cheatoaster diaphanus*, *Chaetogaster distrophus*, *Chaetogaster limnaea*, *Cheatoaster setosus*, *Dero digitata*, *Ophiodonais serpentina*, *Pristina foreli*, *Pristina longiseta*, *Pristina osborni*, *Pristina sima* (new to Great Lakes), *Slavina appendiculata*, *Specaria josinae*, and *Wapse mobilis*.

LUMBRICULIDAE (*Stylodrilus heringianus*)

Stylodrilus heringianus is the only abundant lumbriculid in the Great Lakes and the only one found in the study area, but *L. variegatus variegatus*, *L. variegatus incostans*, and *Eclipidrilus lacustris* have been reported (Howmiller and Beeton 1970, Spencer 1980). According to Saether (1970), *S. heringianus* is the most common oligochaete in the profundal zone of deep, oligotrophic lakes, but populations have been found in a variety of habitats from peaty pools (Pickavance 1971) to the Aswan reservoir (Timm 1970). This species is a subsurface deposit feeder most likely deriving most of its nutrients from organic deposits and associated bacteria. Because a maximum of only 50% of the population was sexually mature in any month, Cook (1969) believed that the life cycle took 2 years.

Stylodrilus heringianus probably is the most abundant oligochaete in Lake Michigan, making up more than 90% of the oligochaete fauna at depths greater than 70 m (Mozley and Howmiller 1977). Along with the amphipod, *Pontoporeia hoyi*, it is one of the two dominant macrozoobenthic species and probably plays a significant role in the total food web of Lake Michigan. At the Donald C. Cook Nuclear Plant, *S. heringianus* occurred in 46% of all lake Ponar grabs and had the highest density of any oligochaete (949.6 m⁻², Table 1); this constituted 12% of all macrozoobenthos and 25% of all oligochaetes. Few specimens occurred in water shallower than 16 m deep, but beyond 16 m, density increased sharply to 4,376 m⁻² where *S. heringianus* comprised more than 50% of the total worms and 20% of the total macrozoobenthos. The overall maximum densities and frequencies of occurrence were lowest in coarse sandy substrata and highest in fine silts and clays; however, there seemed to be some preference for substrata with at least some portion of sand. *Stylodrilus heringianus* has been negatively associated with mean grain size less than ϕ (Dermott 1978), which may be related to tube construction (Mozley 1975), or to its preference for lower sedimentation rates occurring in the sandy substrata in water between 40 and 90 m deep (Nalepa and Thomas 1976, T. Keilty, GLRP, unpubl. data).

PISIDIIDAE

Three bivalve families occur in the Great Lakes: Unionidae, the freshwater mussels, Corbiculidae, the Asiatic clam, and Pisidiidae, the fingernail clams (Mackie et al. 1980, Whit et al. 1984). Unionids in the Great Lakes are confined to bays, to interconnecting channels, and to shallow areas with firm substrates and with wave action or current (Mackie et al. 1980). Although isolated empty shells of *Lampsilis radiata* have been collected along the southeastern shoreline of Lake Michigan, to date there are no known established unionid populations. The biofouling Asiatic clam, *Corbicula fluminea* has become established in Lake Erie (Scott-Wasilk et al. 1983) but not in the other Great Lakes. In Lake Michigan, live *Corbicula* were collected at the J. C. Campbell Power Plant north of the Donald C. Cook Nuclear Plant (White et al. 1984). One empty shell has been identified from the intake at the Donald C. Cook Nuclear Plant; however, it does not appear that there are any established populations, at least along the southeastern shoreline.

The pisidiids are the common bivalves of open lake bottoms, and the fauna of the Great Lakes is quite diverse. Of the three common genera (*Musculium*, *Pisidium*, and *Aphaerium*) with 44 species worldwide, 36 species are known from North

America and 24 from the vicinity of the Cook Plant (Table 1). The life history and ecology are known for only a few of the species in the Great Lakes. The life span varies from 1 to 3 years with either synchronous or continuous reproduction (Hornbach 1980, 1982). Their distributions depend on a variety of factors, including water temperature, depth, substrate type, and food availability (Mackie et al. 1980). Species of *Musculium* are more common in small lakes and vernal ponds where they are associated with aquatic macrophytes and soft, silty substrata. *Sphaerium* generally exists on the surface of more firm substrata where they filter the overlying lake water. The species of *Pisidium* are burrowers living within softer substrate where they filter water through the sediments. Only *Pisidium conventus* exists in the deepest portions of the lakes as part of the profundal assemblage containing *Pontoporeia hoyi*, *Heterotrissocladius oliveri*, and *Stylodrilus heringianus*. The remaining *Pisidium* species are confined to shallower habitats.

PISIDIUM

As a genus, *Pisidium* made up 18% of all macrozoobenthos and occurred in 62% of the Ponar grabs. In water between 6 and 32 m deep, the frequency of occurrence was highest (86 to 100%), as was density (670 to 6,390 m⁻²), indicating an ubiquitous distribution. Beyond water 32 m deep, the frequency of occurrence remained high (98%) but densities fell sharply. Three species were present at all depth strata (*P. casertanum*, *P. lilljeborgi*, and *P. nitidum*), and two species were dominant in water shallower than 16 m deep (*P. fallax* and *P. henslowanum*). These five species, along with *P. conventus*, were abundant enough (Table 1) to provide some data on ecology.

Pisidium casertanum was the dominant *Pisidium* of the nearshore area in water at depths less than 20 m and was a primary constituent at all depths greater than 20 m. The maximum abundance occurred in water between 24 and 28 m deep in substrata composed of silty clay, and to a lesser degree, very fine sand. Densities were greatest in October, with a minor peak in July indicating a bi-annual mode of reproduction (Mackie 1979).

Pisidium conventus is a cold-water stenotherm found in shallow areas of subarctic and high altitude waters and in the profundal of deep lakes farther south (Herrington 1962). In the vicinity of the Cook Plant, no *P. conventus* occurred in water shallower than 16 m deep, but peak abundance was reached in water between 28 and 32 m deep which is below the extent of the summer thermocline (Mozley and Winnell 1975). Henson and Herrington (1965) felt that coarse substrates limited the distribution of *P. conventus*, further supported by our data. Peak abundances were in substrata with varying amounts of silt and clay. These substrata do not occur with any regularity at the Cook Plant until water depths of 28 to 32 m deep are reached. We were not able to determine reproductive periods of this species, in part because of the depth limits of the survey; however, it would appear that new generations occurred in summer and in late fall to early winter, similar to the findings of Heard (1963).

Pisidium henslowanum, along with the majority of the less abundant species of *Pisidium* (Table 1), is part of a distinct nearshore group living in mixtures of sand and silt. The shallowest occurrence was in water between 4 and 8 m deep, and maximum abundance was in water between 12 and 16 m deep (208 m⁻²). Although some specimens occurred in water deeper than 32 m, it appeared that this taxon was

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confined to the region of the bottom shallower than the extent of the summer hypolimnion (about 20 to 25 m deep in the survey area, Mozley and Winnell 1975). The maximum abundance was in October.

The maximum density and frequency of abundance of *P. lilljeborgi* were in the vicinity of the summer thermocline (20 to 28 m deep), although many specimens occurred at all depths, particularly where the substratum was very fine sand mixed with silt and clay. This substratum preference was quite different from the clean, coarse sand habitat noted by Henson and Herrington (1965). The maximum abundance of *P. lilljeborgi* occurred in July, with apparently a single reproductive period, unlike most other *Pisidium* which release young both in spring to late summer and in late fall.

Two forms of *Pisidium nitidum* occurred in the vicinity of the Cook Plant (*P. nitidum* f. *nitidum* and *P. nitidum* f. *pauperulum*) of which only *P. f. nitidum* was common. The peak abundance and frequency of occurrence of the latter form were in water between 28 and 32 m deep, in fine sands mixed with silt and clay; however, significant numbers occurred in a variety of other substrates at all depths.

SPHAERIUM

Sphaerium were present in 35% of all Ponar samples but accounted for only 1.4% of the total macrozoobenthos. Four species were present (Table 1), but only *S. nitidum* and *S. striatinum* occurred regularly. Two peaks of abundance occurred with depth of water, the first at 8 m deep primarily containing *S. striatinum* and the second at 20 m containing *S. nitidum*. Distributions of both taxa were related to substrata containing silty, fine sands. Minor peak densities also occurred in coarser sands mixed with some silts and clays. The observed peak abundances were in July and November, suggesting a bi-annual reproductive cycle.

MUSCULIUM

The genus *Musculium* in the survey area was represented by both *M. securis* and *M. transversum*, but together they accounted for only 0.07% of all macrozoobenthos. *Musculium transversum* occurred most frequently in water between 12 and 16 m deep. *Musculium securis* occurred only twice.

MALACOSTRACA

Common crustaceans in the Great Lakes include the subclasses Ostracoda, Copepoda, and Malacostraca. Although Ostracoda and Copepoda (meiozoobenthos) are very abundant in and directly above the surficial sediments, only the macrozoobenthic Malacostraca are considered. The malacostracans in Lake Michigan near the Cook Plant are comprised of the orders Amphipoda (scuds or sideswimmers), Mysidacea (opossum shrimp), Isopoda (pill bugs and sow bugs), and Decapoda (crayfish).

AMPHIPODA

Three families in the suborder Gammaroidea are known to occur in the Great Lakes. In the survey area, all three families occurred, two of which (Haustoriidae and Hyalellidae) are monotypic. In the family Gammaridae, the genus *Gammarus* was represented by three species. Thus, of the 90 known species of North American amphipods (Pennak 1978), Lake Michigan, particularly along the southeastern shoreline, has a rather sparse diversity. Any lack of diversity is compensated for by the sheer abundance of the haustoriid, *Pontoporeia hoyi*. The gammarids *Gammarus fasciatus*, *G. pseudolimnaeus*, and *G. troglophilus* occurred in very low combined abundance (0.9 m^{-2}), as did the hyalellid, *Hyallela azteca* (0.02 m^{-2} , Table 1). While *P. hoyi* occurred throughout the entire survey area, the other four species were associated primarily with the central portion of the study site in the vicinity of the riprap surrounding the intake and discharge structures of the power plant.

Haustoriidae (*Pontoporeia hoyi*)

Pontoporeia hoyi was numerically second, as a taxon, to the total tubificids, contributing 28% of all macrozoobenthos or $2,209 \text{ m}^{-2}$. It was the third-most frequently collected taxon, occurring in nearly 68% of all samples. At both the genus and the species levels, *P. hoyi* was the dominant single taxon. These findings are similar to those of Alley and Mozley (1975) who estimated that *P. hoyi* comprised 73% of all macrozoobenthos abundance in Lake Michigan in water between 9 and 270 m deep. In the survey area, less than 30% of the *P. hoyi* population occurred at water depths less than 20 m deep. The mean density increased approximately ten-fold between the 4- to 8- and 8- to 12-m water depth intervals and two-fold between 10 and 36 m deep.

The primary food source for *P. hoyi* is bacteria associated with organic carbon deposits, and the abundance and distribution have been positively correlated with bacterial density (Marzlof 1965). Only weak or variable correlations have been made with depth, sediment type, or percent carbon (Alley 1968, Dermott 1978). A lack of correlations in part may have been due to the irregular topography of their study sites. The relatively flat, featureless topography of the bottom near the Cook Plant gave positive correlations between density and sample water depth (3 to 48 m deep) and between density and the finer grained sediments.

Two distinct life cycle patterns were exhibited by *P. hoyi*. The patterns appeared to be directly related to the differences in annual temperature cycles between near-shore and offshore waters (Winnell and White 1984). A 1-year life cycle was evident for the population existing in water between 8 and 20 m deep, and a 2-year cycle was evident at water depths greater than 32 m (Winnell and White 1984). The 1-year cycle was composed of an 8-month growth and development cycle (April through November) and a 4-month breeding and brooding period (December through March), both controlled by seasonal light fluctuations. The 2-year cycle differed in having both summer and winter reproductive periods with two cohorts produced annually. The end result for the survey area was six cohorts co-occurring within any 2-year period.

ISOPODA

Of the nearly 1,600 North American species, only 80 occur in freshwater aquatic habitats, and the vast majority of these are from springs or subterranean systems. In the family Asellidae, two genera (*Lirceus* and *Asellus*) and six species have been collected in the Great Lakes (Barton and Hynes 1976), but only *A. racovitzai* and *A. intermedius* occurred near the Cook Plant. While these species most likely are associated with the riprap, some individuals appeared in Ponar grabs (Table 1). Although lake bottom densities were low, *A. intermedius* was more abundant nearshore, a trend previously noted by Barton and Hynes (1976).

MYSIDACEA (*Mysis relicta*)

Mysis relicta is considered a glacial relict with close affinities to the primarily marine opossum shrimps (Ricker 1959). One mode of feeding is active predation on *Pontoporeia* as the amphipod leaves the sediment (Parker 1980). In the diurnal feeding mode when *M. relicta* is near the bottom, it often is captured by the Ponar grab. However, because the Ponar is an ineffective method of sampling mysids occurring in the water column, we could not relate bottom densities to the entire population. At best, the deeper the Ponar sample, the greater the occurrence of *Mysis* in grabs, particularly over sediments that contained greater numbers of *Pontoporeia* and other potential prey.

DECAPODA

Of the more than 150 species and subspecies of crayfish in North America, only a few occur in the open waters of the Great Lakes, and most of these are in association with rocky substrata. No specimens were actually taken in Ponar grabs (Table 1), but divers have reported seeing numerous specimens walking about the open sand bottom in the vicinity of the riprap (Dorr and Jude 1986). Of the three species collected in impingement studies (Table 1), only *Orconectes propinquus* was common, and the bulk of its population undoubtedly is in association with the riprap structures.

CHIRONOMIDAE (AND OTHER DIPTERANS)

The larvae of virtually all flies are aquatic to some degree; thus, it was not surprising to find a variety of taxa occurring in the vicinity of the Cook Plant (Table 1). With the exception of the Chironomidae, however, all other taxa were considered as waifs from streams and ponds or as having some association with the riprap structures. The minor exception to this was four genera of Ceratopogonidae (Table 1) which are true, large-lake taxa. Because densities of ceratopogonids were so low, little could be learned about their ecology.

The chironomids are the major benthic insect component of open waters of large freshwater lakes throughout the world. For North America north of Mexico, about 2,500 species have been described. Of these, at least 200 occur in the Great Lakes (Winnell 1984b; Winnell and White 1985a, 1985b, 1986). At least 90 taxa have been

collected at the Cook Plant (Table 1), making this the most diverse of the major groups.

Although chironomids are both abundant and diverse, surprisingly little is known about their life cycles or ecology. In general, egg, larvae (four instars), and pupae are aquatic. Prior to emergence, the pupae actively swim to the lake surface where eclosion to the adult stage is quite rapid. Adults live but a few days to weeks, depositing egg masses directly at the water surface. It is thought that the required aerial stage has limited the distribution of larval chironomids to shallower depths where the swimming pupae have a more reasonable chance of reaching the water surface (Hynes 1984). In the Great Lakes, very few species occur greater than 25 m deep and only *Heterotrissocladius oliveri* occurs commonly at depths greater than 50 m (Mozley and Howmiller 1977, Winnell and White 1986).

Chironomids were the most frequently collected macrozoobenthos, occurring in 89% of all Ponar samples. In part, this high frequency could be attributed to a variety of species occurring in the very-nearshore, sandy substrata. The total mean abundance was only 568 m⁻², however, which ranked them fifth among major taxa. Most of the 90+ species were collected on only a few occasions, but 36 taxa did occur regularly enough for some density estimates to be made (Table 1). Of these 36, two taxa were dominant (*Chironomus fluviatilis*-gr. and *Satheria tylus*) and an additional 10 were widespread. Several taxa were numerically important only in particular years of the study. *Procladius* sp., *Chironomus anthracinus*-gr., *Cryptochironomus* sp. 2, and *Chironomus fluviatilis*-gr. were dominant in years prior to 1975; *Satheria tylus*, *Robackia demeijerei*, and *Cladotanytarsus* sp. along with *C. fluviatilis*-gr. were dominant after 1974. The pattern seen here reflects the effects of the riprap, not as a habitat but as a probably physical force changing substratum stability in the Central Region of the survey area. The autecology given below is for those taxa which either were consistently common throughout the years of the study period or for those where sufficient numbers appeared, but were sporadic or appeared only over a few years.

Chironomus fluviatilis, as with several other chironomid taxa, was composed of at least two species not separable in the larval stage. From rearings, it appeared that most larvae in the group could be identified as *C. decorus*. *Chironomus fluviatilis*-gr. was the most numerous (157 m⁻²) and frequently collected (47%) chironomid near the Cook Plant. The single sample maximum density was 5,393 m⁻². The overall maximum densities were found in water between 4 and 12 m deep, associated with sandy nearshore substrata. Emergence peaks occurred in late May to early June, with peak abundance of 4th instar larvae in September. Only 4th instar larvae were present from October through May.

Chironomus anthracinus-gr. also was composed of two species, *C. nr anthracinus* and *C. nr decorus*. The maximum single sample density was the highest for any chironomid (11,150 m⁻², Table 1), but high densities were quite sporadic as frequency of occurrence was only 10.2%. With an overall mean density of 44 m⁻², *C. anthracinus* ranked third among the chironomids. This taxon was most numerous in water between 12 and 20 m deep, with a secondary peak of occurrence between 4 and 8 m deep. *Chironomus anthracinus* preferred finer sands mixed with some silt. The life history was difficult to follow. Beginning in August, very few specimens occurred in water less than 12 m deep throughout the remainder of the year, while larvae were always present in water greater than 12 m deep. Adults from both the shallow and deep components emerged from July to August, but the shallower population either remained as undetectable early instars until the following summer

or moved offshore, possibly in response to increased water temperatures. These data also could reflect differences in the life histories of more than one species. In any case, the lack of nearshore specimens from November to December remains puzzling.

Cladotanytarsus sp. is present in waters ranging from eutrophic to oligotrophic (Beck 1977). Population densities of this species were highly variable from year to year and month to month. The maximum single density was 6,242 m⁻², but the overall mean was only 35.6 m⁻² with a frequency of occurrence less than 13% (Table 1). Fine sands, very fine sands, and silty sands provided the best habitats in water between 4 and 12 m deep. No specimens occurred in water deeper than 24 m. Emergence peaks occurred between June and July and again in August to September, based on instar data. Early instars overwintered and did show further development until waters warmed in June.

Several species in *Cryptochironomus* occurred with some regularity at the Cook Plant (Table 1). *Cryptochironomus rolli* was the 20th-most abundant chironomid taxon (1 m⁻²), limited primarily to clean sands in water less than 8 m deep. Emergence was in May to June with overwintering most likely in the 4th instar. *Cryptochironomus* sp. 1 (which actually may not be within the "complex") was probably underestimated because of its small size. The mean density was 2.4⁻², and maximum density was only 242 m⁻² (Table 1). Using finer sieves, densities have ranged up to nearly 1,000 m⁻² for this portion of Lake Michigan (Winnell, unpubl. data, GLRD). The species was most abundant in fine sands in water between 4 and 8 m deep, where it emerged in mid-summer. The overwintering stage was as early instars. Because of difficulties in separating larvae, *C. cf. fulvus* and *C. digitatus* remain under the taxon *Cryptochironomus* sp. 2. As such it ranked second among the most frequently collected chironomids (30% of all samples), with a single sample maximum of 970 m⁻² and a mean density of nearly 31 m⁻². The abundance and frequency of occurrence were greatest in water 8 to 12 m deep with fine sand, with a secondary abundance in silty sand. Little can be said about life cycles until the two species can be separated in all stages. *Cryptochironomus* sp. 3 occurred in nearly 8% of all the samples, with a mean density of about 7 m⁻². The maximum abundance was in water between 4 and 8 m deep, although a large portion of the population may exist in water shallower than 4 m deep. There was a strong association with fine sands at these shallow depths. Data suggest both spring and fall emergences.

Of the two *Heterotrissocladius* species present in the vicinity of the Cook Plant, *H. changi* occurs in the lower littoral to upper profundal, while *H. oliveri* is a profundal species (Saether 1975). *Heterotrissocladius changi* occurred in 12% of the samples with a mean abundance of 13 m⁻² (Table 1) and reached a maximum density in water between 20 and 28 m deep in substrata with sandy silt/clay and gravel mixed with fine particulates. The maximum monthly abundance of *H. changi* was in April to May and again in August to September, suggesting emergence in May to June and September to October with overwintering in the 3rd instar. The deepwater species, *H. oliveri*, is distributed in oligotrophic to ultraoligotrophic lakes (Saether 1975). Little can be said about this species in the Cook Plant survey area as much of the population was beyond the depth limits that were examined; however, none were collected in water shallower than 16 to 20 m deep, and sediment types were limited to the characteristic sandy silt/clay and silt/clay of deeper waters.

Species of *Micropsectra* are listed as deposit feeders of the littoral zones of lakes and streams (Coffman and Ferrington 1984). The unassociated species in southeastern Lake Michigan builds tubes of sands and organic debris which, when numerous,

can cover most of the surface of the substratum. Large densities, up to 5,575 m^{-2} , were occasionally encountered, but the overall density was only slightly more than 19 m^{-2} , with a frequency of occurrence of 11%. *Micropsectra* was most abundant in water between 12 and 28 m deep, in finer sediments of silty/clayey sands, sandy silt/clays, and silt/clays, most notably in clayey sands from which its tubes were built. The data suggest that emergence periods were in May to June and again in October to December, with the 3rd instar overwintering.

The general distribution of *Monodiamesa tuberculata* is similar to that of *H. oliveri* in being common in deeper water, under oligotrophic conditions (Saether 1973). It was the 5th-most frequent chironomid (17%) with an overall mean density of 13 m^{-2} (Table 1). Although recorded from water 3 to 167 m deep in the Great Lakes (Saether 1973), the maximum abundance at the Cook Plant was in water between 12 and 24 m deep, with silty/clayey sands and sandy silt/clay. With only slight fluctuations in June and July, average monthly densities were nearly identical. The data suggest that emergence periods were in May and June and then again in late fall. Overwintering began in the 2nd and 3rd instars which completed development by April.

In the Cook Plant survey area, *Paracladopelma* consisted of the *P. camptolabius*-gr. (composed of *P. cf. undine* and *P. nereis* not separable in early instars) and *P. winnelli* (Table 1). The *camptolabius*-gr. occurred in 12% of all samples with a mean density of 12 m^{-2} . Its abundance was greatest in water between 4 and 12 m deep in fine to very fine sands, particularly in substrata free of silts and clays. Little can be said of the life cycle as the two species could not be separated in all stages. *P. winnelli* occurs in the sublittoral zone of large, oligotrophic lakes and in streams and seems to have very wide thermal and nutrient ranges (Jackson 1977). The species occurred in 12% of all samples with a mean density of 9.6 m^{-2} . Specimens were present in a wide range of water depths, with maximums between 12 and 32 m deep, primarily with silty/clayey sands and sandy silt. There appeared to be a single emergence period in July to August, with overwintering occurring in the 2nd and 3rd instars which reached 3rd and 4th instars by April.

Several *Polypedilum* occurred in the samples, but only *P. scalaenum* was particularly abundant (Table 1). *Polypedilum scalaenum* has been collected from a wide variety of habitats, in and on wood, as leaf miners, and on open sandy littoral zones of lakes (Coffman and Ferrington 1984). Larvae occurred in 12.8% of all Ponar grabs, with a mean density of 28.4 m^{-2} . The highest frequency of occurrence was in water between 4 and 16 m deep, with a peak between 8 and 16 m (16%) in gravel and clayey sand substrata. The peak emergence period was probably in July and August, with a possible emergence period prior to the April sampling events. Most December specimens were 4th instar, while only 1st, 2nd, and 3rd instars occurred in April.

Densities of *Potthastia longimanus* were never large but remained fairly consistent over the study years. The frequency of occurrence was 5.4% in all samples, and the mean density was 3.6 m^{-2} (Table 1). The primary depth of occurrence was in water between 12 and 20 m deep, with highest densities in gravel, very fine sand, and silty sand substrata. The emergence periods could not be adequately determined; however, they appeared to be in early or mid-summer. Overwintering appeared to be in the 4th instar.

The larvar of *Procladius* are primarily carnivores feeding on oligochaetes, meiozoobenthos, and other chironomids (Mozley and Howmiller 1977); early instars may feed on benthic algae (Chernovskii 1949). In the Cook Plant area, two species

were present, but these were not separated until very late in the study; thus, data on *Procladius* refer to both taxa (Table 1). In water between 3 and 18 m deep, *Procladius* was one of the two most numerous chironomid taxa and was the 4th most numerous both in overall mean density (39 m^{-2}) and frequency of collection (21%, Table 1). The maximum abundance was in water between 16 and 24 m deep in sandy silt, clay, and silt/clay substrata. The maximum monthly abundances were in spring and fall months, suggesting a late spring emergence with overwintering in the 4th instar.

Psectrocladius simulans was common in the vicinity of the Cook Plant (Table 1). From Ponar samples, *P. simulans* occurred in very low density (2 m^{-2}) and frequency (4%); however, it was the dominant chironomid on the riprap and artificial substrate samples (80-90%, Lauritsen 1979). In the open lake, maximum densities were in water between 4 and 8 m deep, with gravel and coarse sand. The maximum single sample density was 545 m^{-2} , far lower than has been observed at other sites in southeastern Lake Michigan (e.g., $34,542 \text{ m}^{-2}$, Winnell and Jude 1981). Emergence was probably in late summer with overwintering in early instars.

Robackia demeijerei is the common open-lake species of this genus in southeastern Lake Michigan (Table 1), with a frequency of occurrence of 12.8% and a mean density of 26.1 m^{-2} . *R. demeijerei* was ranked as the 9th-most abundant chironomid. Its small size, combined with the large sieve sizes used, may have led to a greatly underestimated population. The greatest abundances were in water from 4 to 8 m deep, with coarser substratum types within which it wanders freely (Mozley and Garcia 1972). The absence of *R. demeijerei* in November and December suggested a fall emergence with overwintering in the early instars.

Satheria tylus is common in the sublittoral sand of oligotrophic lakes and in streams but may exist under mesotrophic conditions (Jackson 1977). At the Cook Plant, it was the second-most abundant species (65.9 m^{-2}) and third-most frequently collected (23.7% of all samples, Table 1). The primary depth of occurrence was in water between 4 and 8 m deep, with clean fine, medium, and coarse sands. Based on instar data, there appeared to be at least two generations per year, and possibly a third, with emergence in late spring, mid-summer and early to mid-fall. Overwintering was in the 2nd and 3rd instars.

SUMMARY AND CONCLUSIONS

In the more than 13 years of benthic collections in southeastern Lake Michigan in the vicinity of the D. C. Cook Nuclear Plant, nearly 300 taxa have been collected. This data set is the most extensive ever collected on any of the Laurentian Great Lakes or any of the World's other temperate, deep-water lakes and is one of the longest continuous studies of any body of fresh water. In total, the results pose many more questions than they answer; however, we have been able to detail aspects of life history and ecology, to examine seasonal, yearly, and long-term (7+ years) cycles, and to look at community structure for about 100 of the more common species.

The lake bottom in the vicinity of the Cook Plant is featureless and not heavily influenced by input from rivers or by bays and rocky shorelines. As such, the macrozoobenthic fauna there seems representative of communities and processes occurring within the lake. The distributions of most taxa fit along a gradient related to depth, where the sediments are influenced by the extent of wave action (resuspen-

sion and deposition) and by the extent of the summer thermocline. Depth can roughly be divided into a nearshore region with a sparse wave zone community, a shallow water region with a diverse but patchy community, and a profundal region with a reduced but less patchy community.

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IMPACT OF THE DONALD C. COOK NUCLEAR PLANT ON FISH

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INTRODUCTION

The Donald C. Cook Nuclear Plant, located on the southeastern shore of Lake Michigan, was under construction during the late 1960s and early 1970s, a time of great concern about environmental impacts of nuclear power plants. Scientists feared that warm water discharged by the plants into the water body would have a detrimental effect on the zooplankton and larval and adult fish. In the early 1970s, very little was known about the taxonomy of larval fish; very few studies on larval fish had ever been done in Lake Michigan with the exception of Wells' (1966) study on larval bloaters. In addition, most researchers at the time concentrated on the pelagic deep waters, ignoring the nearshore zone (0-10 m) of Lake Michigan. Studies of fish focused on the deeper waters of the lake, beyond the 7.3-m water depth where the Cook Plant intakes are located. Therefore, data were lacking from which to make good predictions on impact of these power plants.

One of the goals in studying fish in the vicinity of the Cook Plant was to document which species utilize the nearshore zone, which ones spawn there, and which species use the inshore zone for nursery grounds. Another goal was to gather data on the kinds of possible impacts that might be expected from operation of the Cook Plant. Preoperational data were gathered during 1973 and 1974, and they were also used in comparisons with operational data to determine changes in species diversity or abundance between the two time periods. In addition to these two goals, the choice of sampling stations was made to establish a reference station south of the Cook Plant at Warren Dunes State Park and one at the Cook Plant. This experimental design allowed monitoring of any changes in fish populations as a result of the existence of the intake and discharge structures and the associated thermal plume, riprap, and currents. Comparison of catches between the reference and Cook stations documented whether fish were attracted, repelled, or unaffected by the different aspects of the physical structure and operation of the Cook Plant.

The Donald C. Cook Nuclear Plant is one of the largest nuclear power plants (2,200 megawatts) on Lake Michigan, and it takes in $104 \text{ m}^3/\text{s}$ (1.86 million gallons per minute) of lake water. The water is sent through the intake pipes, past the traveling screens, through the condensers, and then is discharged back into the lake after being heated about 10 C° over ambient. With these high flows and large ΔT s, the potential for affecting large numbers of organisms is high. The thrust of our Cook Plant fish studies concentrated on three possible areas of impact of the nuclear plant.

The first area of impact was the thermal discharge. In order to study this impact we organized our field sampling around reference and plant impact stations so that we could examine catch indices for larval, juvenile, and adult fish and determine

whether a particular species was repelled, attracted, or not affected by plant operation. We used analysis of variance and intervention analyses (Madenjian et al. 1986) to examine the data and determine if any of these effects were occurring. We did not look at the impact of the local and ephemeral thermal plume on larval or adult fish that became entrained in the water discharged immediately into Lake Michigan because we felt that the impact was negligible and this area was difficult to study. The warm water that is discharged from the Cook Plant goes through slot-type, jet diffusers which rapidly mix the heated water with Lake Michigan ambient water. The thermal plume that is created is usually small (less than 117 hectares) (Hoglund 1980). Our own temperature monitoring confirmed that the plume was small in area, difficult to distinguish from ambient water, particularly during days of wind or wave activity, and it rapidly dispersed.

The second area of study focused on impact of entrained larval fish and fish eggs. Entrainment was defined as any fish larvae or fish eggs that were brought into the Cook Plant in cooling water, passed through the 9.5-mm bar mesh traveling screens and the condensers, and were then discharged back into the lake. Entrainment was measured by sampling the Cook Plant forebay using diaphragm pumps. Concentrated sampling occurred during June, July, and August, the major period of spawning in the vicinity of the plant. Studies on the mortality of larval fish were not done because of the many difficulties involved with conducting such studies, particularly the high mortality experienced by larval fish in control samples. Therefore, mortality of larval fish was assumed to be 100% even though many studies (such as Marcy et al. 1978) have indicated that larval fish mortality is probably less.

The third area of concern was impingement of juvenile and adult fish. The fish that were impinged on the traveling screens at the plant were enumerated. In 1975, the first year of full operation of Unit 1, we weighed and measured all fish that came off the screens. A subsampling technique (Thurber and Jude 1984, 1985) was used when large numbers or a large biomass of fish were encountered. After 1975, impingement sampling was done for one continuous 24-hour period every fourth day and extrapolated for the entire year. These estimates were based on total weight of fish which was measured every day; species composition and length-frequency distribution data were obtained from every fourth-day sample and expanded to the other non-sampled fish. These data were used to document the total number, species, and size of fish lost on the Cook Plant traveling screens.

The sampling program design enabled us to generate a large and extensive dataset on the larval, juvenile, and adult fish which inhabit Lake Michigan. Besides the regular larval, juvenile, and adult fish sampling with plankton nets, seines, trawls, and gill nets that occurred during April-November, many supplemental samples were collected during winter months. These data were used to establish the distribution, spawning times, and behavior of common and less abundant Lake Michigan fishes. In addition to the field sampling we also had entrainment sampling, which occurred during all months of the year, and impingement sampling, which occurred every fourth day. These data were used to supplement the knowledge gained from the field sampling program and thus rounded out our descriptions of the distributions and spawning behavior of the fish occurring in the vicinity of the plant.

In addition to all of these studies, we also had a scuba diving program (Dorr and Miller 1975, Dorr and Jude 1980, Dorr and Jude 1986) which examined the distribution of fish in the vicinity of the intake and discharge structures. These observations gave us a better idea of the fish that were congregating there and what kinds of colonization and spawning activities were occurring. Observations of fish around

the structure also helped us understand the impingement process since we never saw fish pulled into the intakes as a result of the intake velocity. Rather fish seemed to be attracted to the structure and were displaced to the forebay fortuitously. There is a section of the intake pipe in which velocity increases substantially and, once a fish passes through this area, chances of it return to Lake Michigan are small.

Besides diving studies, we were able to use data gathered by the phytoplankton, benthos, and zooplankton researchers to augment our discussions of the feeding behavior, distribution, and abundance of fish in the Cook Plant vicinity. Determining the impact of the plant on the resident fish population was our goal. However, we were also able to make contributions to the body of ecological knowledge of fish species in the Cook Plant environs (see the Ecology of Fish chapter).

Objectives are to: 1) document the species of fish which inhabit the Cook Plant area, and their distribution, spawning behavior, and nursery grounds, 2) determine the impact of the thermal plume on fish by comparing catch indices between a control and reference area, 3) establish the numbers of larval fish and fish eggs that were entrained at the Cook Plant, 4) record the number of juvenile and adult fish impinged on Cook Plant traveling screens, 5) describe the ecology of the major fish species for which there were adequate data (see the Ecology of Fish chapter), 6) integrate the various datasets from the phytoplankton, benthos, zooplankton, and fish sections of the project to better understand their interactions in the nearshore zone of Lake Michigan, and 7) attempt to establish the significance of the entrainment of impingement losses through production forgone calculations. We have tried to condense the many reports and publications that have been produced during the Cook Plant project studies into this document. For additional detail and for all raw data, please consult the references that will be cited with each of the subject areas discussed.

THE DONALD C. COOK NUCLEAR PLANT

The Donald C. Cook Nuclear Plant is a two unit, 2,200 megawatt nuclear power plant located on the southeast shore of Lake Michigan in Berrien County, Michigan. The total heat rejection rate for two units operating at full power is 3.9×10^4 kcal/h (15.5×10^3 BTU/h) (AEC 1973). Condenser and service cooling water used to dissipate this heat is withdrawn from and returned to Lake Michigan via a once-through cooling system. The condenser cooling water flow rate for both units combined is 6,202 m³/min (AEC 1973). Differences in condenser design have resulted in different flow rates at the two units (2,700 m³/min Unit 1; 3,500 m³/min Unit 2). At maximum generating capacity, increases in cooling water temperature (ΔT) above ambient lake temperature are 12.1 C° at Unit 1 and 9.3 C° at Unit 2 (AEC 1973).

Cooling water for both units is drawn in through three intake structures (see Figure 2 in Introduction) located approximately 686 m offshore in 7.3 m of water at a lake level of 176.5 m (Mean Sea Level). Intake pipelines (4.9-m diameter) were laid beneath the lake bottom. Where they emerge, each intake pipe is surrounded by concrete and riprap which rises about 2 m above lake bottom. Each intake structure is protected by an octagonal steel frame and protrudes an additional 2.5 m above the riprap. Therefore, most cooling water is drawn from the 3- to 5-m strata of the water column. The steel frame surrounding each intake opening is fitted with bar racks and guides which form a 20- x 20-cm grill. Estimated water velocity at the intake grills is 0.4 m/s during normal conditions. Velocity in the intake pipes is about

1.8 m/s (AEC 1973). Cooling water travels through the intake pipes into the screen-house forebay where it first passes through trash racks (6- to 7-cm openings, 0.3 m/s water velocity) and then traveling screens (0.95-cm openings, 0.6 m/s water velocity) before reaching the circulating water pumps which send it to the steam condensers.

Heated water is discharged to the lake via two buried pipes, one 4.9 m in diameter (Unit 1) and one 5.5 m in diameter (Unit 2). Discharge water enters the lake approximately 366 m offshore in 5.5 m of water. Water is discharged through slot-type jet diffusers in an offshore direction. Exit velocity is approximately 4 m/s (AEC 1973).

Cooling water transit time from the intake, at the 7.3-m contour in Lake Michigan, to discharge, at the 5.5-m contour, is about 10 minutes. Duration of condenser passage is about 6 seconds. It is estimated that an entrained organism experiences the maximum temperature differential (12.1 C° at Unit 1, 9.3 C° at Unit 2) for approximately 3 minutes (AEC 1973). Mortality of entrained organisms is the result of a combination of physical, chemical, and thermal stresses. Physical stresses include shear, turbulent acceleration, rapid changes in pressure, and collision and abrasion with particles in the cooling water and with the internal surfaces of the system (e.g., screens, impellers, condenser tubes, jet diffusers). It is believed that the most severe physical stress occurs at the pumps (Marcy et al. 1978).

At the Cook Plant, chemical stresses include exposure to effluent from steam generator blowdown which contains chloride, copper, iron, phosphate, morpholine (a corrosion inhibitor), and hydrazine (a reducing agent) among other things (AEC 1973).

Thermal stress begins when the entrained organism reaches the steam condensers where an immediate temperature increase of up to 12 C° occurs. At the Cook Plant, temperature does not decrease appreciably until the point of discharge (AEC 1973).

Use of jet diffusers, combined with the offshore placement of the discharge structures, prevented us from sampling discharged organisms effectively. As a result, we could not obtain mortality estimates for larval fish entrained at the Cook Plant. Therefore, we have assumed 100% mortality and our entrainment loss estimates consequently represent a "worst case" situation.

METHODS

STATISTICAL ANALYSES OF LARVAL FISH DATA

ANOVA was applied to larval fish density data (no./1,000 m³) for the most abundant species (see Noguchi et al. 1985, Sokal and Rohlf 1969, and Scheffe 1959 for details). All ANOVA designs were Model I, full factorial, balanced designs calculated with the statistical package BMD8V (Statistical Research Laboratory 1975). To approach the assumptions of the model more closely, larval fish densities were transformed using $\log(\text{density} + 1)$. Data from two zones, beach and open water, were analyzed separately. Factors deemed appropriate in ANOVA applied to larval fish density data in the beach zone included Year (1973 through 1982), Month (June through August), Station (A, north Cook; B, south Cook; and F, Warren Dunes), and Diel Period (day and night) for alewife and spottail shiner. Factors used in the open water zone included Year, Month, Area (Cook and Warren Dunes), Depth (6- and 9-m contour), and Diel Period for alewife, and Year, Area, Depth,

and Diel Period for yellow perch. Only density data from June were used in yellow perch ANOVA.

Because preliminary tests showed no significant trend in larval fish densities among depth strata (surface to near bottom) for a given sampling site, diel period, and date, such samples were used as replicates in the ANOVAs of open water stations. Since larval fish samples were taken at 2-m intervals in open water, stations at 6 m, (C, south Cook, and G, Warren Dunes) had one less replicate than 9-m stations (D, south Cook, and H, Warren Dunes). To balance the design, the mean of densities from the four strata at 6-m stations replaced the missing 8-m value. The unweighted means method for balancing designs (Fox 1973) was then applied to the open water results. Treatment sums of squares were multiplied by the ratio of harmonic mean cell size to maximum cell size to adjust for substitutions, and the number of missing values was subtracted from degrees of freedom of the error term to adjust mean square error.

ENTRAINMENT SAMPLES

Sampling

Species and numbers of larvae and eggs entrained at the Cook Plant were monitored by standardized sampling from 1973 to 1982. However, sampling in 1973 and 1974 was limited because of sporadic testing of condenser cooling systems. These data are presented in detail by Jude et al. (1979), Bimber et al. (1984) and Noguchi et al. (1985) analyzed the data collected from 1973 to 1982.

An entrainment sampling unit included a Hale (type 30LC-1750) diaphragm pump (maximum capacity, 300 liters per minute; mean capacity, 208 liters per minute) with a 7.6-cm diameter hose extending into the intake forebay to a depth of 5 m (Fig. 1). The 5-m depth (maximum depth in the forebay is 9 m) was chosen because of results of our vertical and horizontal stratification testing in 1975 (Jude 1976, Bimber et al. 1984). Water was pumped through a 0.5-m diameter, 363- μ m mesh plankton net suspended in a 208-liter drum. A flowmeter installed in the drum's effluent pipe measured volume of water filtered. Entrainment sampling units (three) were located at grates 2 and 3 (see Introduction) and one at the Unit 1 discharge. Seven grates span the length of the screenhouse forebay floor.

A standardized group of entrainment samples was collected from 1975 to 1982, twice per month, except for June, July, and August, when sampling was done once or twice per week to coincide with peak abundance of fish larvae. Samples were collected over a 24-hour period. Each 24-hour period was divided into four diel sampling divisions which varied from 4 to 8 hours, depending on division and day length. The four divisions were sunrise-noon, noon-sunset, sunset-midnight, and midnight-sunrise. Sixteen samples, four replicates (three intake, one discharge) per division, were collected for each 24-hour period.

Statistical Analyses

Mean densities of entrained fish larvae and eggs were expanded to the volume of water circulated by the plant during the time represented by that diel period. The total number of fish larvae and eggs entrained over 24 hours was computed by totaling estimates from each of the four diel sample divisions during a sample

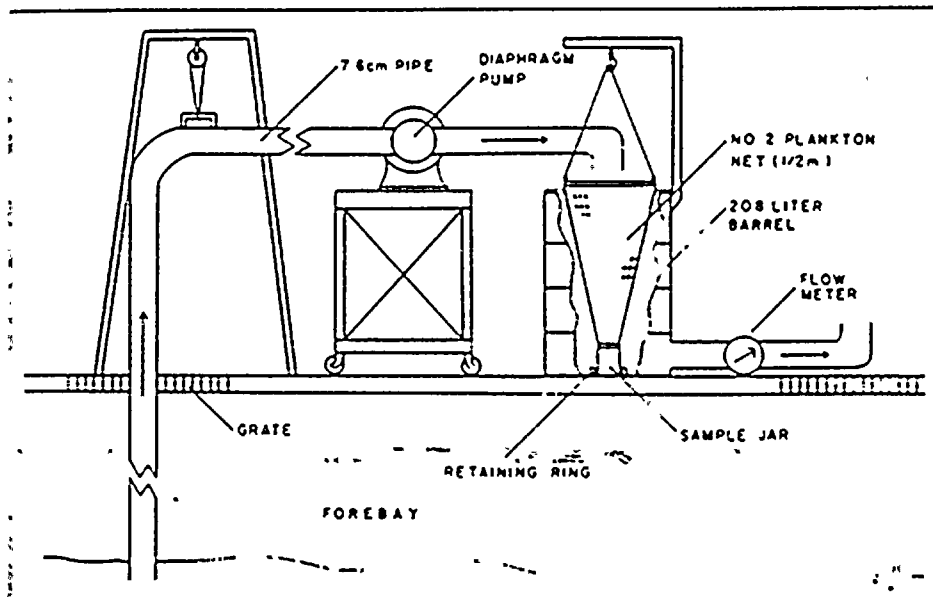


FIGURE 1. Schematic diagram of an entrainment sampling unit, showing the forebay, sampling pipes, diaphragm pump, plankton net, and flowmeter in the discharge pipe.

period. Each of these four estimates was derived by multiplying the mean density ($N = \text{four}$) times the total volume of water pumped through the plant during the time represented by that particular division. For yearly estimates, non-overlapping, contiguous time intervals (usually 1-2 weeks) were established such that the sampling date was the approximate midpoint of the interval. Estimated entrainment during a sampling period was assumed to be representative of fish larvae and egg abundance per unit volume of circulating water during the 1- to 2-week sample interval. These data were totaled for each month and then yearly estimates computed.

Analysis of entrainment data for alewife larvae, total larvae, and fish eggs ($\log_{10} - 1$ transformed data) was carried out by use of the Analysis of Variance (ANOVA) subprogram of the Statistical Package of the Social Sciences (SPSS) (Nie et al. 1975). A three-factor, fixed-effects, non-additive model was considered most appropriate, with three factors: Year, Grate, and Diel Period. To obtain the most balanced model (least amount of missing data) only data from 1976 through 1982 were examined and data from both discharge grates were combined (since they were not sampled simultaneously). Thus the analysis was across 7 years, 4 grates, and 4 diel periods and their associated interactions. Sample size was 1,699. Similar results were obtained for each ANOVA in that the only factors which exhibited attained significance levels less than 0.01 were Year and Diel Period.

Statistical Procedures for Adult Fish Data

Catch-per-unit-effort data of alewife, rainbow smelt, spottail shiner, trout-perch, and yellow perch were log-transformed using $\log(\text{catch} + 1)$, then analyzed using MIDAS (Fox and Guire 1973) and a Model 1, full factorial, balanced analysis of variance (ANOVA). Catch-per-unit-effort, an index of fish abundance, is a standard measure of catch for each type of fishing gear. One unit of effort was defined arbitrarily as a 12-hour gill net set, a 10-minute trawl tow, or a 61-m seine haul. Because each gear has its own unique biases, data from different gear were not combined. When assumptions of ANOVA were seriously violated, the Kruskal-Wallis test was employed. Details on these procedures are presented in Jude et al. (1979) and Tesar et al. (1985).

Given that ANOVA meets parametric assumptions (Conover 1971), power analysis estimates the ability of ANOVA to detect true changes in fish abundance. Values of the least detectable true change (LDTC) in log-transformed data were calculated (for details see Jude et al. 1979). Values of LDTC were transformed to least detectable true ratios (LDTR) to express power calculations in terms of numbers of fish (geometric means of catch plus one). The LDTR is the minimum ratio between geometric mean levels that can be detected by ANOVA and is a measure of the sensitivity of the design. An example of one set of values is presented as functions of alpha and power for two distinct sets of comparisons (Table 1). Note that Mean Square Error (MSE) and degrees of freedom refer to the full ANOVA using data from all 7 years (1973-1979), thus comprising the best estimate of true population variance of sampling. Values presented in earlier reports (Jude et al. 1975, 1979) used MSE from preoperational data alone, assuming it would be appropriate for operational data as well. In all cases, MSE values have not changed markedly from preoperational to operational years, and power results indicate that power of ANOVA for all 7 years is close to that previously estimated using preoperational data alone. Values of LDTR ranged from 1.34 for spottail shiner in trawls (1973-1979 data) (Table 1) to 2.73 for rainbow smelt in seines (Jude et al. 1979) for comparisons of 2-year preoperational data with 2-year operational data at $\alpha = 0.01$ and power = 0.95.

IMPINGEMENT SAMPLES

Fish and debris collected from the traveling screens were separated by Cook Plant personnel. All fish were bagged, labeled with date and time, and then frozen. University of Michigan personnel collected and weighed all frozen fish; a 24-hour sample was saved every fourth day and sorted by species and size. When many fish of the same size were collected in fourth-day samples, a subsample of up to 30 fish was randomly selected and the remaining fish were weighed and discarded. All saved fish were measured to the nearest mm (total length), weighed to the nearest g, sexed, and examined for presence of food, condition of gonads, presence of disease, or physical damage.

Fourth-day samples (number and weight of fish) and weight of fish impinged on interim days were both used to estimate total monthly impingement by species. Percent species composition by weight of fourth-day samples was used to partition the actual monthly weight of fish impinged into weight estimates by species, according to the formula:

$$Ew = (Sw/Pw)Tw$$

where: Ew = Estimated monthly weight of fish impinged for a given species.
 Sw = Monthly weight of fourth-day impingement samples for a given species.
 Pw = Monthly weight of fourth-day impingement samples, all fish combined.
 Tw = Total monthly weight of all fish impinged (includes fourth-day and interim samples).

TABLE 1. Least detectable true ratios (LDTR) in geometric mean abundance of the five most abundant fish species caught in standard series trawls at Cook Plant study areas, southeastern Lake Michigan, 1973-1979. Values are given as functions of type I error ($\alpha = a$) and power. Comparisons are of 1-year preoperational sampling to 1-year operational sampling and 2-year preoperational sampling to 2-year operational sampling at one area. Each LDTR is expressed as the ratio of the operational value to the preoperational value of the quantity "mean number per trawl tow plus one." n = sample size; MSE = mean square error of ANOVA; df = degrees of freedom of MSE .

			2-year sampling			4-year sampling		
MSE	df	a	n	Power		n	Power	
				0.90	0.95		0.90	0.95
Alewife								
0.1502	390	0.01	56	1.92	2.04	112	1.58	1.65
		0.02		1.84	1.95		1.54	1.61
		0.05		1.73	1.84		1.47	1.54
		0.10		1.64	1.74		1.42	1.48
Rainbow smelt								
0.0660	390	0.01	56	1.54	1.60	112	1.36	1.40
		0.02		1.50	1.56		1.33	1.37
		0.05		1.44	1.50		1.29	1.33
		0.10		1.39	1.44		1.26	1.30
Spottail shiner								
0.0520	390	0.01	56	1.47	1.52	112	1.31	1.34
		0.02		1.43	1.48		1.29	1.32
		0.05		1.38	1.43		1.26	1.29
		0.10		1.34	1.39		1.23	1.26
Trout-perch								
0.0576	279	0.01	40	1.62	1.68	80	1.40	1.45
		0.02		1.56	1.63		1.37	1.41
		0.05		1.49	1.56		1.33	1.37
		0.10		1.44	1.50		1.29	1.33
Yellow perch								
0.1067	279	0.01	40	1.91	2.03	80	1.58	1.65
		0.02		1.83	1.95		1.54	1.60
		0.05		1.72	1.83		1.47	1.54
		0.10		1.64	1.74		1.42	1.48

Number of fish impinged per month was then estimated using:

$$E_n = E_w/W$$

where: E_n = Estimated total number of fish impinged each month for a given species,

W = Mean weight per fish of a given species (calculated for each species from number and weight of fish of each species impinged in fourth-day samples for a given month).

PRODUCTION FORGONE

The production of a population over a finite interval is defined as the total elaboration of biomass irrespective of its fate (Ivlev 1966). Production includes not only the biomass accumulated by those individuals alive at the end of the time interval but also the biomass of those individuals which died before the end of the time interval. Thus, production results from the interaction of two fundamental processes: growth and mortality. The actual mechanisms underlying the interactions of growth and mortality are very complicated and poorly understood. A number of methods have been devised which permit easy estimation of production by assuming that growth and mortality are independent processes which act in a linear or exponential fashion (see Chapman 1973). Production loss due to Cook Plant operation was calculated using the procedure given by Ricker (1975). In this procedure, growth and mortality are assumed proportional to the weight of an average individual and number of individuals at time t , respectively.

Production forgone due to entrainment and impingement of fishes at the Cook Plant was estimated for alewife, spottail shiner, rainbow smelt, and yellow perch. Estimations were made using the following steps. (1) Length frequencies of impinged fish were transformed into age frequencies by using an age-length key. (2) Relative proportions of entrained pro- and post-larvae were estimated based on a critical length (5 mm for alewife) at which the larva changes from pro-larva to post-larva. (3) Mean weight of a given age group of fish was estimated from length-weight relations and length frequencies within the age group. (4) Mean weight of pro- and post-larvae was estimated by measuring preserved specimens and averaging the value found over the distribution of larval fish lengths within an age group. (5) Growth rate was estimated by assuming that the change in mean weight of an age group is proportional to the mean weight. (6) Age-specific survival rates were estimated by combining literature estimates with values obtained using a modified Horst model (Horst 1975). (7) The estimates obtained in steps 1-6 were used to calculate production forgone. The total production forgone (P_T) due to entrainment and impingement of fish larvae, juveniles, and adults was calculated using the formula:

$$P_T = \sum_{i=1}^n \left(\sum_{j=1}^m \frac{W_{ij}}{W_{ij}} \right) G \bar{B}$$

where: G_i = instantaneous rate of increase in weight; B_i = average biomass of the age group over the interval of production; and t_{min} = age of newly hatched larvae = 0 (t_{min} used for the sake of generality). An elaboration of the production forgone model is presented in detail by Rago (1984). Assumptions regarding the actual estimates of forgone production include: (1) The mean weight of an age group as determined from the entrainment and impingement samples is time invariant over the production interval. Thus, a larva weighing x gm at time t would have ultimately weighed y gm at time $t + n$ where y is the average weight of an individual of age n at time t . (2) Survival rates are also time invariant. This is one of the most controversial areas of fisheries research. (3) There are no positive or negative feedback effects. Thus, the production forgone calculation implicitly assumes that reduction in production in one generation does not reduce the number of adults recruited to following generations (see Goodyear 1977). (4) Each species exists separately and may be considered independently of its community and environment. (5) System of community productivity decreases in response to removal of fish via power plant operation. Hence, the loss of x kg of alewives (as estimated by P_f) is equivalent to a decrease of x kg in the system production. (6) Larval fish mortality due to entrainment was assumed to be 100%.

Sensitivity of the production model to changes in parameters and initial conditions was determined by examining the percent change in estimated production as a function of changes in survival and mean weight estimates ($\pm 25\%$) and in numbers entrained and impinged ($\pm 50\%$). Parameter and initial condition estimates were derived from a variety of sources including: our data base, literature, unpublished data from the Great Lakes Fishery Laboratory, U.S. Fish and Wildlife Service, Ann Arbor, Michigan, and personal communications with fishery scientists.

Findings were related to the real world by comparing production forgone with commercial catch data. Spottail shiners, which have no commercial value, were not included in this comparison. It was assumed that all production forgone occurred in the year under consideration and that its value was determined by the average market price in that year.

ALEWIFE SURVIVAL

Because alewife spawning and hatching are usually continuous over a 2- or 3-month period, it was impossible to identify a cohort and follow it through the season. Thus data for alewife larvae and young-of-the-year were pooled through the season and treated as one cohort. Entrained larvae were separated into the following size groups: 2-5 mm, 5.5-10 mm, 10.5-15 mm, 15.5-20 mm, 20.5-25 mm, and 5.5-25 mm pooled. For each size group, mean densities from each sampling period (all diel periods) were summed over the season, usually June through August. Length partitions were chosen based on growth rates of alewife larvae (Heinrich 1981, Kellogg 1982) and frequency of entrainment sampling (usually weekly), to prevent total density from including the same age-group more than once. Entrainment densities from 1981 and 1982 were halved because sample collection was twice a week.

Field larvae densities were partitioned based on size of larvae into 2- to 5-mm and 5.5- to 25-mm groups because monthly field sampling encompassed a long enough interval for larvae to grow from 5 to 20 mm and because very few 20- to 25-mm larvae were collected. Samples from the diel period showing greatest abundance of

alewives (usually night) were used. Mean field densities for each length interval, averaged over all depth strata at 6- and 9-m stations, were summed over the season, as with entrainment data (see Mansfield and Jude 1986 for details).

Our day trawl catches of alewives were nearly always greater than night trawl catches which was probably due to nocturnal vertical migrations. Similar results were obtained by Janssen and Brandt (1980) and Wells (1983). To express young-of-the-year catch as density, volume of water filtered by our trawl was estimated using published studies of trawls in motion (Amos et al. 1981, hatch et al. 1981) and measurements of our trawl out of the water. Densities of young-of-the-year alewives were calculated separately for each 10-mm length interval and summed over months when young-of-the-year were present at trawl depths and weather conditions were calm enough to permit sampling, usually September, October, and November. Because alewife young-of-the-year are believed to concentrate near bottom during the day (Janssen and Brandt (1980), densities were multiplied by the mean height of the trawl divided by the depth of the water column (6 or 9 m), then summed over the two bottom depths. This adjustment made trawl data comparable to field larvae data, which represented the whole water column.

Survival rates, the ratios of young-of-the-year densities to fish larvae densities, were calculated separately for each year using field and entrainment data for each length interval of larvae and the density of fish from the length interval containing the most young-of-the-year. The length interval with the highest young-of-the-year density was believed to be the best representation of young-of-the-year density, because it represented the time when most young-of-the-year were actually at trawl depths rather than inshore (late summer, smaller fish) or offshore (late fall and winter, larger fish). Densities for each length interval of fish larvae and modal-length young-of-the-year densities were also averaged over all years and then used to calculate survival rates for all years combined.

Daily mortality rates were calculated from entrainment data using mean densities over all years for the various length intervals available. Dates for each length interval were derived from time of peak catch of yolk-sac larvae, laboratory-derived alewife growth rates (Heinrich 1981), and time of peak young-of-the-year catch.

RESULTS

FISH LARVAE

Field

There were few indications of Cook Plant effects on distributions of fish larvae (Bimber et al. 1984, Noguchi et al. 1985). Statistical tests showed few significant differences among densities of larval fish species. Mean alewife abundance did not differ significantly between Cook and Warren Dunes stations over the 10 yr of data collection. Densities of spottail shiner larvae in night beach station samples did not differ significantly between areas. Yellow perch were at depressed levels during all but the last year of our study, due to the impact of the alewife (Christie 1974, Jude et al. 1979, Crowder 1980, Jude and Tesar 1985). Yellow perch larvae abundance followed no patterns attributable to plant operations. Yellow perch larvae were abundant in both Cook Plant and reference areas in 1974, 1977, and 1982, but not in

other years. During operational years 1977 to 1982, open-water yellow perch densities were not significantly different between Cook and Warren Dunes stations. Additionally, no significant difference existed between preoperational and operational years for June open water yellow perch densities.

Beach catches of rainbow smelt larvae during 1974, 1975, and 1980-1982 did not differ significantly between Cook and Warren Dunes stations. In open water, densities at the Cook Plant were significantly higher than at Warren Dunes (ANOVA, $p = 0.0015$) when preoperational and operational years were combined (1974, 1975, and 1980-1982). This significance was due to the unusually high catches at 6 and 9 m during May 1974. During operational years (1975 and 1980-1982), however, no significant difference between Cook and Warren Dunes open-water stations occurred. No plant impact on larval rainbow smelt populations was observed.

For less abundant species, the most striking abundance pattern was that for common carp larvae. Common carp larvae were never collected in the study area during preoperational years. During operational years, they were found mostly at Cook stations. Of the 23 samples containing common carp larvae, only two were collected at Warren Dunes (31-83 larvae/1,000 m³). These larvae probably drifted from the Cook Plant area. Apparently, common carp spawning took place at Cook Plant stations during operational years in response to the plant's warm water plume and currents. Thus common carp spawning at the Cook Plant was a clear plant effect.

Burbot, deepwater sculpin, and ninespine stickleback larvae showed no apparent differences in densities between Cook and reference areas. Quillback, unidentified Coregoninae, gizzard shad, emerald shiner, and others, identified only to genus or family (minnows, darters, sculpins), were collected so seldom that statistical analyses were not appropriate. Several other species, not abundant enough for statistical testing, were more abundant at the Cook Plant. During the 10-year study, 10 of 14 samples containing trout-perch larvae were from Cook Plant stations. More johnny darter and slimy sculpin larvae were collected at Cook Plant than reference stations, probably because riprap around the intake attracts these species for spawning.

Entrainment

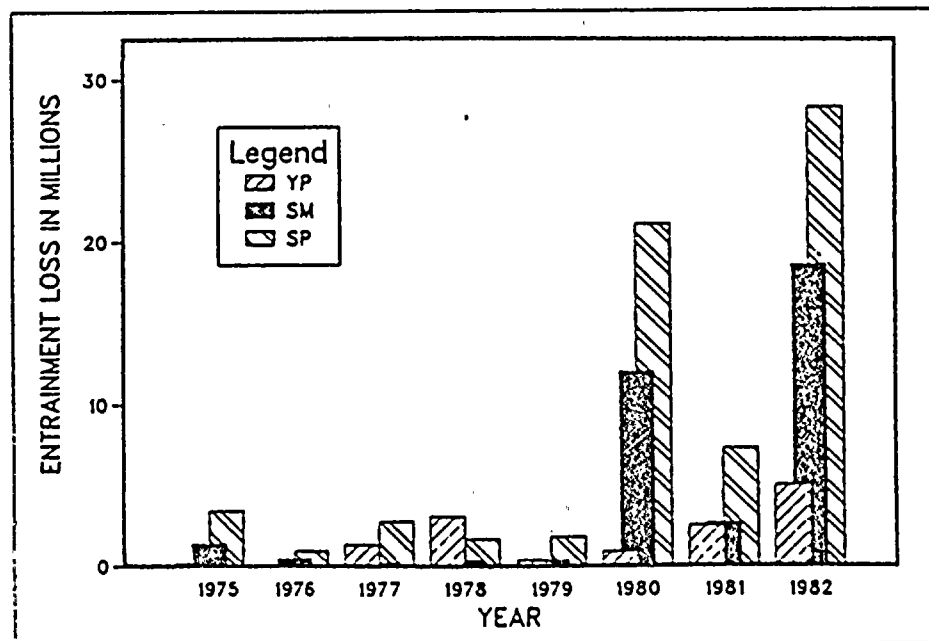
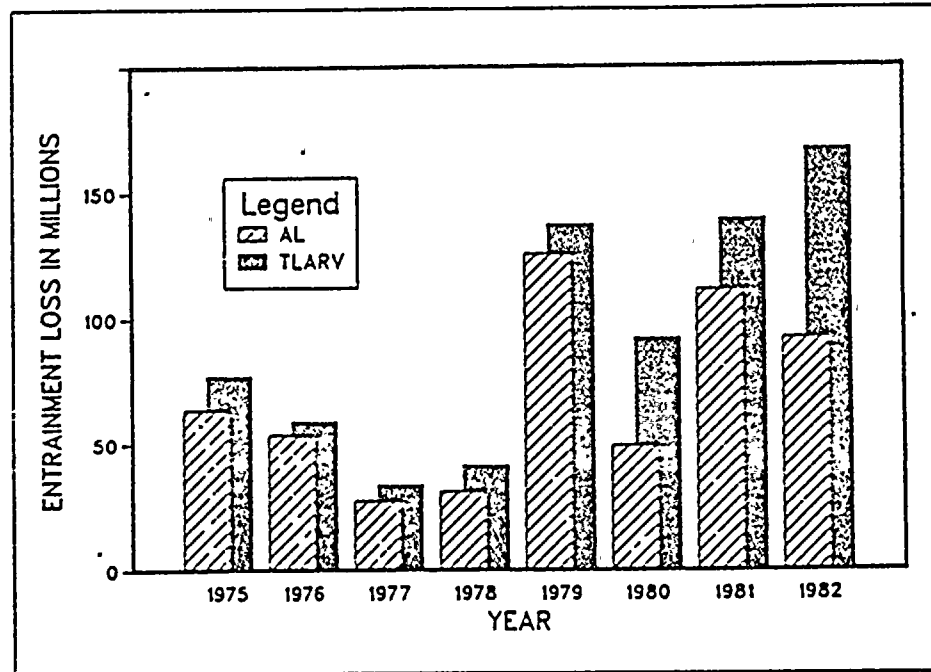
Nearly 750 million fish larvae and 23 billion eggs were entrained at the Cook Plant during 1975-1982 (Table 2). Estimated annual losses ranged from 33.5 million larvae in 1977 to 167 million larvae in 1982. Variations in annual losses were generally caused by a combination of biological and non-biological factors including fluctuations in year-class strength and differences in plant operation.

Thirteen species of fish larvae were found in entrainment samples during our 8-year study: alewife, burbot, common carp, deepwater sculpin, johnny darter, mottled sculpin, ninespine stickleback, quillback, rainbow smelt, slimy sculpin, spottail shiner, trout-perch, and yellow perch. In addition, there were four groups that could not be identified to species: coregonines (*Coregonus* spp.), darters (*Etheostoma* spp.), minnows (Cyprinidae), and sculpins (*Cottus* spp.). Approximately 8% of all fish larvae collected during entrainment sampling were damaged beyond recognition and <0.1% could not be identified to species at our current level of taxonomic sophistication.

Alewives were by far the most abundant species, accounting for between 54 and 92% of the total number of fish larvae entrained in every year (Fig. 2, Table 1) and 74% of the overall 8-year entrainment loss. Spottail shiners represented 9% of the

TABLE 2. Estimates (in millions) of annual entrainment losses of fish larvae and fish eggs at the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1975 to 1982. Calculations use actual reported flow rates of the circulating water system.

Taxon	Year of Estimate								Total	% Total
	1975	1976	1977	1978	1979	1980	1981	1982		
Alewife	63.708	53.7550	27.3888	31.098	125.6180	49.35	111.54	92.425	554.8828	74.34
Spottail shiner	3.41	0.9361	2.760	1.681	1.8228	21.06	7.257	28.2297	67.1566	9.00
Rainbow smelt	1.3608	0.4145	0.1795	0.3496	0.3726	11.954	2.6265	18.5233	35.7808	4.79
Yellow perch	0.17554	0.03807	1.3224	3.0655	0.3840	0.8971	2.506	4.9700	13.3586	1.79
Trout-perch	1.079	0.2509	0.1456	0.0194	0.6288	0.4858	0.5394	1.3749	4.5238	0.61
Johnny darter	0.0440	0.210	0.707	0.772	0.8105		0.153	0.7046	3.4011	0.46
Shiny sculpin	0.2431	0.06092	0.0256	0.130		0.553	1.002	0.4887	2.5033	0.34
Mottled sculpin	0.152	0.146	0.0483		0.131		0.143	0.4870	1.1073	0.15
Common carp		0.0912	0.0235	0.175	0.3603	0.0513	0.187		0.8883	0.12
Ninespine stickleback				0.124		0.379	0.156	0.0112	0.6702	0.09
Quillback			0.0628				0.534		0.5968	0.08
Burbot		0.0202		0.102				0.3428	0.4650	0.06
Deepwater sculpin				0.178	0.0141				0.1921	0.03
Unidentified sculpins	0.1899	0.0892	0.0918	0.175	0.0905	0.667	0.5953	0.5744	2.4731	0.33
Unidentified minnows			0.1248		0.8138	0.2846	0.1714	1.0280	2.4226	0.32
Unidentified coregonids			0.0850						0.0850	0.01
Unidentified darters			0.0276						0.0276	<0.01
Poor condition	6.555	2.8642	0.4274	3.352	5.9935	6.4765	11.859	17.9458	55.4734	7.43
Unidentified larvae	0.1693	0.0349	0.0887	0.100					0.3929	0.05
Total larvae	77.08664	58.91119	33.5088	41.3215	137.0399	92.1583	139.2696	167.1054	746.4013	
Total eggs	743.1879	2,269.4543	1,320.301	5,840.8138	1,392.5408	3,334.692	995.94	7,005.26	22,902.1898	
Total Cook Plant Flow (millions of m ³)	1,298.	1,292.	1,138.	2,370.	2,476.	2,830.	2,753.	2,749.		



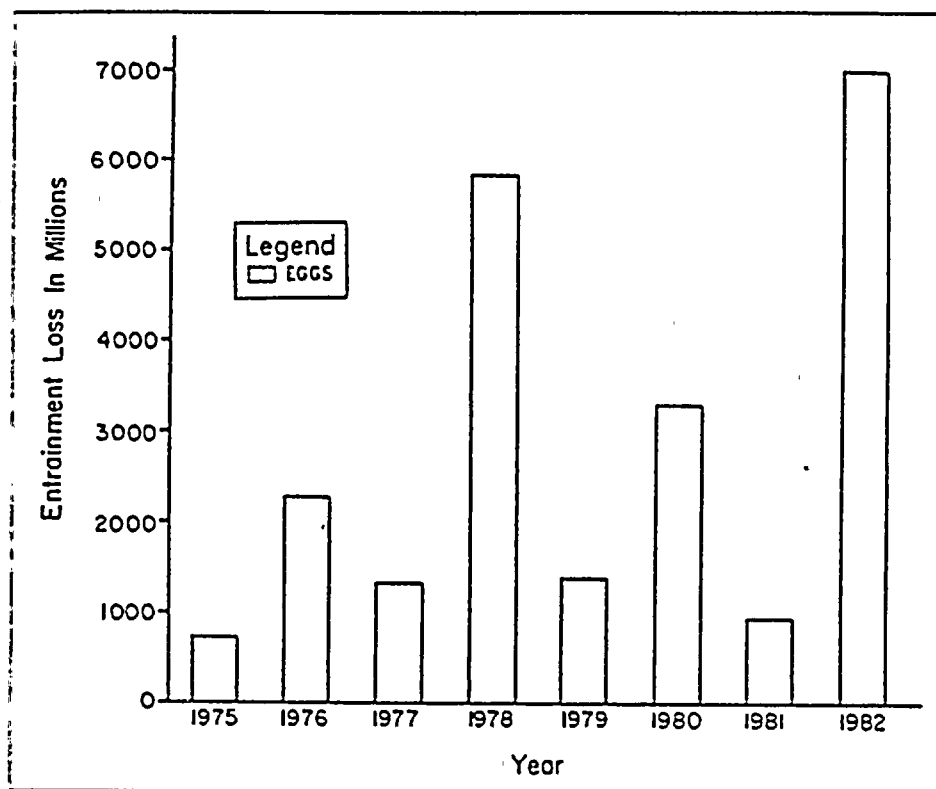


FIGURE 2. Entrainment losses at the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1975-1982, for alewife, yellow perch, rainbow smelt, spottail shiner, total larvae, and fish eggs.

1975-1982 entrainment loss, rainbow smelt represented 5%, and yellow perch 2%. Remaining taxa each accounted for <1% of the total 8-year loss.

Entrainment of fish larvae generally began for the year in April, peaked in June or July (when alewife spawning and hatching peaked), and terminated in October or November as fish larvae and young-of-the-year migrated to deeper offshore zones.

Entrainment rates were strongly influenced by diel period. Use of ANOVA showed that significantly ($p < 0.001$) more fish larvae were entrained at night (dusk-midnight and midnight-dawn sampling periods) than during the day (dawn-noon and noon-dusk) in most years. This suggests that fish larvae are actively avoiding the intake structures during the day and this avoidance behavior might be used to minimize entrainment if an intake structure could be designed appropriately.

Most Abundant Species—

Alewife—Alewife larvae were entrained in greater numbers and over a longer period of time in each year than any other fish species. A combination of factors contributed to the alewife's extraordinary susceptibility to entrainment; perhaps the most important were: 1) high abundance of adults and larvae, 2) mode of reproduction—randomly broadcasting large numbers of eggs, 3) time and location of spawning—from June through August in the nearshore (area of the intakes) zone, and 4) behavior, distribution, and form of larvae.

Alewife was the most abundant fish species in the vicinity of the Cook Plant (Jude et al. 1979, Tesar et al. 1985) during 1973–1981 and in Lake Michigan as a whole in the 1960s (Smith 1968). In addition, alewives are pelagic broadcast spawners, which places them in a reproductive guild (Balon 1975) that represents those fish that exhibit the least amount of parental care. As is typical of species in this guild, alewives produce large numbers of eggs (10,000–22,407/female, Auer 1982) and fish larvae compared to species which provide some form of protection for developing embryos and larvae.

Temporal and spatial characteristics of alewife spawning also contributed to their susceptibility to entrainment. In spring and early summer, adults moved inshore and spawned within the influence of the intakes. Alewives have an extended spawning season in comparison with most other species. As a result, newly hatched alewife larvae (the stage most vulnerable to entrainment) were often present in inshore waters throughout summer. Peak hatches generally occurred in July when demand for electricity, and therefore cooling water, was usually also at a peak.

Finally, the behavior and form of alewife larvae increases their chance for entrainment. Larval alewives are pelagic, so they are likely to be present in the strata (3–5 m) from which cooling water is drawn, and they are frail and poorly developed at hatching (2.5–5 mm, essentially planktonic), increasing their vulnerability to intakes.

Larval alewives were found in entrainment samples from April to November. Peak abundance occurred in June (2 year) or July (6 year). Newly-hatched larvae were entrained over an 8–12-week period. Highest mean densities recorded over a 24-hour period ($N = 16$) in each year ranged from 140 larvae/1,000 m³ (1978) to 1,831/1,000 m³ (1975). The magnitude of the abundance peaks appeared to be at least partially affected by the lake's temperature regime in the vicinity of the plant. In 1978, the year with the lowest abundance peak, mean June and July temperatures (13.7 and 14.1°C, respectively) were 9% and 20% below the overall 1973–1982 mean temperatures for those months. Conversely, in 1975, the year with the greatest abundance peak for larval alewives, mean June and July temperatures (16.2 and 19.5°C) were approximately 7% and 10% higher than the 10-year means. Field sampling at 6- and 9-m Cook Plant stations revealed the same pattern. The lowest annual abundance peak (mean density at stations C, D, and R combined, $N = 26$) occurred in 1978 and the highest in 1975.

Entrained alewife larvae ranged from 2 to 25 mm. Alewife between 25 mm and about 50 mm also passed through the plant, based on the few we recovered in entrainment and impingement samples. It appeared that fish in this size range were avoiding our entrainment sampling gear and passing through the 9.5-mm mesh vertical traveling screens. Thus this size group was inadequately sampled by our methods, a problem that extended to other species as well. The vast majority of alewife, between 61% (1979) and 94% (1975, 1980) annually, were newly hatched

(≤ 5 mm). Predominance of yolk-sac larvae was not unexpected as increasing age brings increased natural mortality and improved avoidance capabilities. Proportionally fewer newly hatched alewife larvae were collected in field samples compared with entrainment samples however.

The rate of entrainment for alewife larvae was strongly influenced by diel period. June-August samples from 1976 to 1982 contained significantly ($p < 0.001$) more alewife at night than during the day. Explanations for this nighttime increase include a possible diel vertical migration that brings more larvae within the influence of the intakes at night, and a reduction in avoidance capabilities of larval fish with darkness (Schumann 1963, Wong 1972).

Rainbow smelt—Rainbow smelt was the third-most commonly entrained fish species at the Cook Plant, accounting for not quite 5% of the total 8-year loss. Rainbow smelt larvae were found in entrainment samples from April to August, with peak abundance occurring in April (1 year), May (4 year), June (1 year), or July (2 year). Maximum 24-hour mean densities ($N = 16$) recorded each year ranged from 3 (1979) to 158 larvae/1,000 m³.

Entrained rainbow smelt larvae ranged in size from 3 to 25 mm and, like the larvae of most fish species, were most susceptible to entrainment soon after hatching. Fish larvae ≤ 8.0 mm usually represented over half the total rainbow smelt entrainment loss during 1975–1982. Recently hatched rainbow smelt larvae were entrained over a relatively short period of time, i.e., 1–2 weeks (5 year) or 3–4 weeks (3 year), mostly during May. In contrast, alewife yolk-sac larvae were commonly entrained over an 8–12-week period. Large rainbow smelt (> 17 mm to 25.4 mm) were also entrained in substantial numbers, accounting for 14–78% of annual rainbow smelt larvae losses during 1975–1982. In contrast, larvae 8–17 mm only represented 0–25% of annual losses. Rainbow smelt larvae disperse rapidly from the hatching site (Tin and Jude 1983) and increased susceptibility of the larger larvae may be a result of their concentrating at the 6–9-m depth contours, near the intakes. Rainbow smelt larvae were entrained more commonly at night than during the day.

Spottail shiner—The second-most commonly entrained fish species, spottail shiners, accounted for 9% of the total 8-year loss. Three factors contribute to their reduced vulnerability to entrainment compared with alewife. First, the adult population of spottail shiners in the vicinity of the Cook Plant is smaller than that of alewife. Spottail shiners were the second-most abundant species collected during our adult fish sampling program (Tesar et al. 1985, Tesar and Jude 1985). Second, and perhaps more important, spottail shiner spawning is concentrated in nearshore waters, ≤ 23 m, (Wells and House 1974, Tesar et al. 1985), outside the influence of intake currents. Although some spottail shiner spawning was observed by divers on the intake riprap (Jude et al. 1979), we believe that most entrained spottail shiner larvae drifted from nearshore nursery grounds. Finally, the behavior and form of spottail shiner larvae reduce their susceptibility to entrainment. Spottail shiner larvae are demersal (Jude et al. 1979, Jude et al. 1980), which makes them much less likely to be drawn into intakes than pelagic alewife larvae. Spottail shiners are also more well developed and robust at hatching than alewife and as a result are probably better able to avoid the intakes at an earlier age.

Spottail shiner larvae were found in entrainment samples from May to October. Peak abundance occurred in June (3 year), July (4 year), or August (1 year). Maximum 24-hour mean densities ($N = 16$) recorded in each year ranged from 8–1,000 m³ in 1976 to 32–1,000 m³ in 1980 (Noguchi et al. 1985).

Entrained spottail shiner larvae ranged in length from 3 to 14 mm. Nearly 75% of

the spottail shiner larvae in entrainment samples were ≤ 5.0 mm (newly hatched); larger larvae were relatively rare.

Entrainment of spottail shiner larvae was strongly affected by diel period. Annual percentages of spottail larvae entrained during darkness ranged from 76% (1981) to 97% (1980). Spottail shiner larvae may migrate off the bottom at night, exposing them to offshore currents and hence increasing their vulnerability to entrainment. Avoidance capabilities are probably somewhat impaired during nighttime as well. In addition, Dorr and Miller (1975) documented limited spawning by spottail shiners on the top of the intake structures. These larvae would also be vulnerable to entrainment when they hatched.

Yellow perch—Yellow perch larvae accounted for slightly less than 2% of the 1975–1982 total entrainment loss (Table 2). The number of yellow perch larvae entrained generally appeared to be related to larval yellow perch abundance in the inshore zone in the vicinity of the plant. Fewer yellow perch were entrained in years when larval fish densities observed in field samples were also low (1975, 1976, 1979, 1980), whereas higher entrainment occurred during periods of relatively high larval yellow perch abundance (1977, 1982). However, substantial numbers of yellow perch larvae were entrained in 1978 and 1981 when low densities of larval yellow perch were observed in inshore areas. It is possible that our once-per-month field sampling effort missed the peak hatch in those years.

Yellow perch larvae were found in entrainment samples from April to August with peak abundance occurring in June (7 year) or July (1 year). Annual maximum 24-h mean densities ($N = 16$) ranged from 0.1 to 74 larvae/1,000 m³.

Yellow perch larvae in entrainment samples ranged in size from 3 to 11 mm but 98% of all yellow perch entrained were ≤ 8.0 mm. Yellow perch larvae are usually 5.0–6.0 mm at hatching (Auer 1982) and generally grow about 0.5 mm/day (Mansueti 1964). Therefore even the largest yellow perch larvae entrained were probably less than 2 weeks old. This rapid decline in entrainment with increasing age is probably the result of a combination of factors including natural mortality, increased avoidance capabilities (Noble 1970), and a possible shift in distribution away from the influence of the intakes. More yellow perch larvae were entrained at night than during the day in all years, except 1976 and 1981.

Less Abundant Species—

Larval burbot (3.5–6 mm), entrained in 1976, 1978, and 1982, were found during March, April, and June. Common carp larvae (4.1–7.6 mm) occurred from June to August during 1976–1981 in entrainment samples; densities ranged from 11 to 66/1,000 m³. Although common carp larvae were abundant in the lake during 1975 and 1982, they were completely absent from entrainment samples. Conversely, in 1977 and 1981 common carp were entrained but none were caught at field stations.

Johnny darter larvae (4–18 mm) were entrained from June through August during every year except 1980, but they represented 0.46% of the 8-year total. Ninespine stickleback larvae were entrained from June through September in 1978 and 1980–1982 with a total projected loss of 0.7 million larvae (0.09% of the total).

Slimy sculpins, mottled sculpin, unidentified sculpin, and deepwater sculpin larvae were entrained during 1975–1982. Together they comprised 0.85% of all fish larvae lost to entrainment. Slimy sculpins were entrained in all years except 1979, mottled sculpins in all years except 1978 and 1980, and two deepwater sculpins (12.8 and 7 mm) were found in March 1978 and June 1979.

Slimy sculpins (6-18 mm) were the most frequently entrained sculpin species, with a total estimated loss over the 8 years of 2.5 million larvae (0.34% of total). They were entrained from late May to August but were most common in June. Mottled sculpins (6-9.2 mm) were generally entrained over a 1-2-week period during June. Mottled sculpin larvae were never captured in field collections. Slimy and mottled sculpins are very similar in appearance as larvae, and an accurate fin-ray count (Heufelder and Auer 1980) is essential to separate the two species correctly. Unidentified sculpins were either slimy or mottled sculpins but, due to their deteriorated physical condition or extremely early stage of development, they could not be identified with certainty. Unidentified sculpin larvae (5-10 mm) were entrained in all years (1975-1982), mostly during June.

Trout-perch larvae were entrained in every year during 1975-1982 with a projected loss of 4.5 million larvae for the entire period. Although juvenile and adult trout-perch were common in our study area (Tesar et al. 1985, Tesar and Jude 1985), larvae were relatively rare both in field and entrainment samples, probably due to their demersal habit. Larval trout-perch (4-19.3 mm) were normally entrained from June to October but in 1976 they were found in February and November samples as well; most occurred during August-October. Of the 45 entrainment samples containing trout-perch, 40 were collected during darkness.

Unidentified minnows—Unidentified minnow larvae (2.4 million over 8 years) were entrained from April to August and occurred during 1977 and 1979-1982. Highest yearly entrainment (1 million larvae) occurred in 1982.

Miscellaneous species—Several other fish species were entrained at the Cook Plant on isolated occasions. One unidentified coregonine (8.5 mm) was collected in April 1977 and one unidentified darter (5 mm) in June 1977. Two quillback larvae were found in entrainment samples, one in April 1977 (9.1 mm) and one in May 1981 (8.8 mm).

Fish eggs—A large number of fish eggs, thought to be mostly alewife, were entrained during the 8-year entrainment study. The number of eggs passing through the plant varied from 0.7 billion in 1975 (the first year of operation) to 7 billion in 1982 (our last year of monitoring) (Table 2). Alewife lay pelagic, semi-buoyant eggs (Auer 1982) at night, making them particularly susceptible to being entrained. However, studies by Marcy et al. (1978) have shown that marine alewife eggs can withstand a 10 C° temperature shock without effect. Therefore, we feel that the only potentially detrimental effect by the plant on fish eggs is the movement of these eggs from intake depths (around 7 m) to discharge depths (about 5 m). These types of movements are undoubtedly common in the nearshore zone of southeastern Lake Michigan, and are expected to have a negligible impact on alewife populations.

RECOMMENDATIONS FOR REDUCTION OF ENTRAINMENT LOSSES

Nearly 750 million fish larvae were entrained at the Cook Plant from 1975 to 1982. Data were summarized on 16 Lake Michigan power plants in 1975 (Spigarelli et al. 1981) and the Cook Plant ranked first in number of alewife larvae entrained, seventh in rainbow smelt entrainment, and second (of three reporting) in yellow perch entrainment. Alewife and rainbow smelt are important forage fish for salmonids. In recent years the Cook Plant has contributed substantially to Lake Michigan entrainment losses (Table 3). Potentially important sport or commercial fish entrained at the Cook Plant included yellow perch (13 million larvae), rainbow smelt

TABLE 3. Entrainment losses (estimated numbers of larvae) of alewife, rainbow smelt, and yellow perch at a number of Lake Michigan power plants. NR = not reporting.

Power Plant	Alewife	Rainbow Smelt	Yellow Perch
Bailly, 1975 (Texas Inst., Inc. 1976)	4.08×10^4	3.13×10^4	1.5×10^4
Zion, 1975 (Cima et al. 1976)	1.13×10^4	4.33×10^4	NR
Waukegan, 1975 (Cima et al. 1976)	6.98×10^4	6.31×10^4	NR
Ludington, 1978 (Liston et al. 1980)	3.05×10^4	1.05×10^4	3.68×10^4
Ludington, 1979 (Liston et al. 1981)	3.95×10^4	1.08×10^4	4.27×10^4
Campbell, 1978 (Jude et al. 1979)	5.1×10^4	1.7×10^4	1.8×10^4
Campbell, 1979 (Jude et al. 1980)	2.3×10^4	1.6×10^4	1.5×10^4
Cook Plant, 1975	6.4×10^4	1.4×10^4	1.8×10^4
1976	5.4×10^4	4.5×10^4	3.8×10^4
1977	2.7×10^4	1.8×10^4	1.3×10^4
1978	3.1×10^4	3.5×10^4	3.1×10^4
1979	1.3×10^4	3.7×10^4	3.8×10^4
1980	4.9×10^4	1.2×10^4	9.0×10^4
1981	1.1×10^4	2.6×10^4	2.5×10^4
1982	9.2×10^4	1.8×10^4	5.0×10^4

(36 million larvae), and coregonids (85,000 larvae). No other major gamefish species (salmonids, centrarchids, esocids, etc.) were found in entrainment samples.

It is far easier to estimate numbers of fish larvae lost to entrainment each year than it is to determine the impact of those losses on Lake Michigan fish populations. Compensatory responses (see Goodyear 1980, Mansfield et al. 1985) are poorly understood, particularly in a system as large as Lake Michigan.

Possible methods for reducing entrainment losses fall into three categories: changes in plant operation, screening and exclusion devices, and type and location of intake structures. Changes in plant operation would provide the least disruptive and most easily implemented alternatives. Cooling water flow rates have a direct effect on the number of larval fish entrained. Reducing flow rates during the peak season for larval fish (June–August) could substantially reduce entrainment losses. Refueling and periodic maintenance shutdowns should be (and usually are) scheduled during periods of maximum larval fish abundance. The potential for reduction in annual entrainment losses approaches 16 million larvae for every 5% reduction in flow (Bimber et al. 1984). Such reductions could have been attained by rescheduling maintenance and refueling or by diel variation in the cooling water pumping schedules. It should be noted that any alteration in cooling water flow rates from design

specifications results in violation of Nuclear Regulatory Commission (NRC) environmental technical specifications calling for a ΔT not exceeding 12.1 C° at Unit 1 and 9.3 C° at Unit 2.

Operational changes might accrue substantial cost to the power company. Indiana & Michigan Power Company (IMPC) (1979) evaluated the biological and economic impacts of reducing the flow rate of the circulating water system at the Cook Plant. Their analysis covered a 183-day period from 1 April to 30 September. They considered shutting down one circulating pump from Unit 1 and one or two pumps from Unit 2.

Operation of the Cook Plant at temperatures below the specified inlet temperature of 21°C would result in reduced power generation and an estimated loss of approximately \$833,600 (1978 dollars). Reduction in entrainment loss would be worth approximately \$144,600 based on production forgone analysis available at the time (Rago 1983). It was therefore concluded by IMPC that the reduced-flow schedule was not cost effective. A similar schedule, with the time period under consideration reduced to 1 June-30 August (91 days), should be considered. Our data indicate that in most years over 90% of the annual entrainment loss occurred during that period. Attention could also be directed toward reducing nighttime flow rates as most fish larvae (approximately 60-80%) were entrained at night.

Exclusion of fish larvae from power plant intakes by the use of screening devices has received considerable attention in recent years. Most screening and exclusion systems are designed to withdraw water at a low rate over a large surface area, thus decreasing the approach velocity and presumably increasing the opportunity for avoidance by larval fish. Intake openings can be covered with fine-mesh screens to exclude many of the larvae that still do not avoid the intakes.

The Unit 3 intakes at the J. H. Campbell Plant (north of the Cook Plant on Lake Michigan near Port Sheldon) were designed to reduce both impingement and entrainment losses. Intakes are screened (9.5-mm square, wedge-wire slots) and located 1,070 m offshore at the 11-m contour; through-slot water velocity is relatively low (15.2 cm/s maximum). Based on small-scale testing in Lake Michigan, Zeitoun et al. (1981) predicted that use of this system would result in an 11-fold difference between entrainment and field densities of fish larvae. However, in full-scale field operation, Unit 3 entrainment densities at Campbell were roughly equal to field densities (Jude et al. 1982). Entrained larvae at the Campbell Plant were generally smaller (<9 mm) than field-caught larvae (2.5-25.0 mm). The necessarily extensive riprap surrounding the intake area was found to attract fish, most notably yellow perch, substantial numbers of which utilized the limestone riprap as spawning substrate (Jude et al. 1982).

At the Cook Plant (without screens) densities of larval fish in field and entrainment samples were similar, with field densities occasionally greater. Sizes of entrained and field-caught larvae were also similar. During 1975-1982 61-94% of entrained larvae were yolk-sac larvae, and therefore would probably have been entrained even if screens similar to those employed at the Campbell Plant had been used (Noguchi et al. 1985). In 1981, the same year covered by the Campbell study, over 91% of the fish larvae entrained at the Cook Plant were \leq mm.

Results of Campbell Plant studies suggest that while the 9.5-mm wedge-wire screen is effective in eliminating impingement and reducing the entrainment of larvae larger than 9 mm, it probably would not substantially reduce entrainment or the ecological penalties associated with production forgone analyses at the Cook

Plant (Rago 1983, 1984). Schneeberger and Jude (1981) evaluated the effectiveness of different screen slot sizes in limiting entrainment based on larval fish morphology only. They indicated that even 2.0-mm mesh screen would not substantially reduce entrainment of small (\leq mm) larvae which are numerous during summer months.

It was found that biofouling and maintenance requirements were greater problems than had been anticipated for wedge-wire screens at Campbell. But the biggest problem associated with this system was that entrainment of some fish species, especially yellow perch, increased because adults were attracted to the riprapped area for spawning. Yellow perch, a gamefish and commercial species, was the dominant species entrained at Campbell's Unit 3 while at the other Campbell intakes which take water from about 4-6 m (and at the Cook Plant) alewives dominated entrainment losses. Reduced entrainment of large fish larvae (> 9 mm) by the use of stationary screens at Campbell may therefore be overshadowed by the fish-attracting qualities of a large, riprapped screenfield in a previously featureless area of Lake Michigan. Screenfields are effective artificial reefs which provide spawning substrate and habitat for additional fish species and shelter for potential fish-food organisms (Rutecki et al. 1985). Installation of screens at the Cook Plant would eliminate impingement but there is no real evidence that entrainment losses would be meaningfully reduced. There most likely would be a reduction in the number of alewives entrained but yellow perch entrainment might increase, particularly in view of the dramatic decline of alewife and concurrent increases in yellow perch populations in the early 1980s (Jude and Tesar 1985).

Location of intake structures can have a substantial influence on the magnitude of entrainment losses. The placement of the intakes at the Cook Plant appears to have successfully balanced engineering and biological considerations, although the latter may have been mostly accidental because the intakes were not specifically designed to reduce or prevent entrainment losses. Examination of biological data collected in the vicinity of the Cook Plant indicates that placement of the intakes at the 7.3-m contour (3- to 5-m depth strata) avoided nearshore nursery and spawning areas and as a result, current entrainment losses are probably lower than they would be if the intakes had been located in shallower water. Although distributions of fish larvae in Lake Michigan are often variable in relation to water depth, time of day, season, and year, larval alewife densities generally decline with increasing depth and distance offshore. Near the Cook Plant, larval fish densities were maximal in the beach zone and decreased rapidly with depth (Fig. 3) (Bimber et al. 1984; Noguchi et al. 1985). Similarly, data collected at the J. H. Campbell Plant indicated that during most years (1977-1981) alewife larvae were primarily concentrated in water less than 6 m (Jude et al. 1982).

Distribution of important fish-food organisms, specifically *Pontoporeia hoyi* and *Mysis relicta*, also indicates a direct relationships with depth (Fig. 4). Unlike larval fish however, densities of *Pontoporeia* and *Mysis* increase as depth increases. Thus, moving the Cook Plant intakes farther offshore would result in substantial entrainment increases for both species. Densities of *P. hoyi* in intake waters (currently $0.05-0.1/\text{m}^3$) would increase to more than $1.0/\text{m}^3$ at 12 m and $1.8-3.5/\text{m}^3$ at 15 m. *Mysis relicta* densities would increase approximately 10 times at 15 m and 100 times at 22 m. Zooplankton might better compensate for entrainment losses because their regeneration rates are much higher than those of fish, but relocation of the intakes to greater depths to reduce larval fish entrainment might impact local zooplankton populations, since they occur in somewhat higher densities at depths > 9 m (Evans et al. 1980).

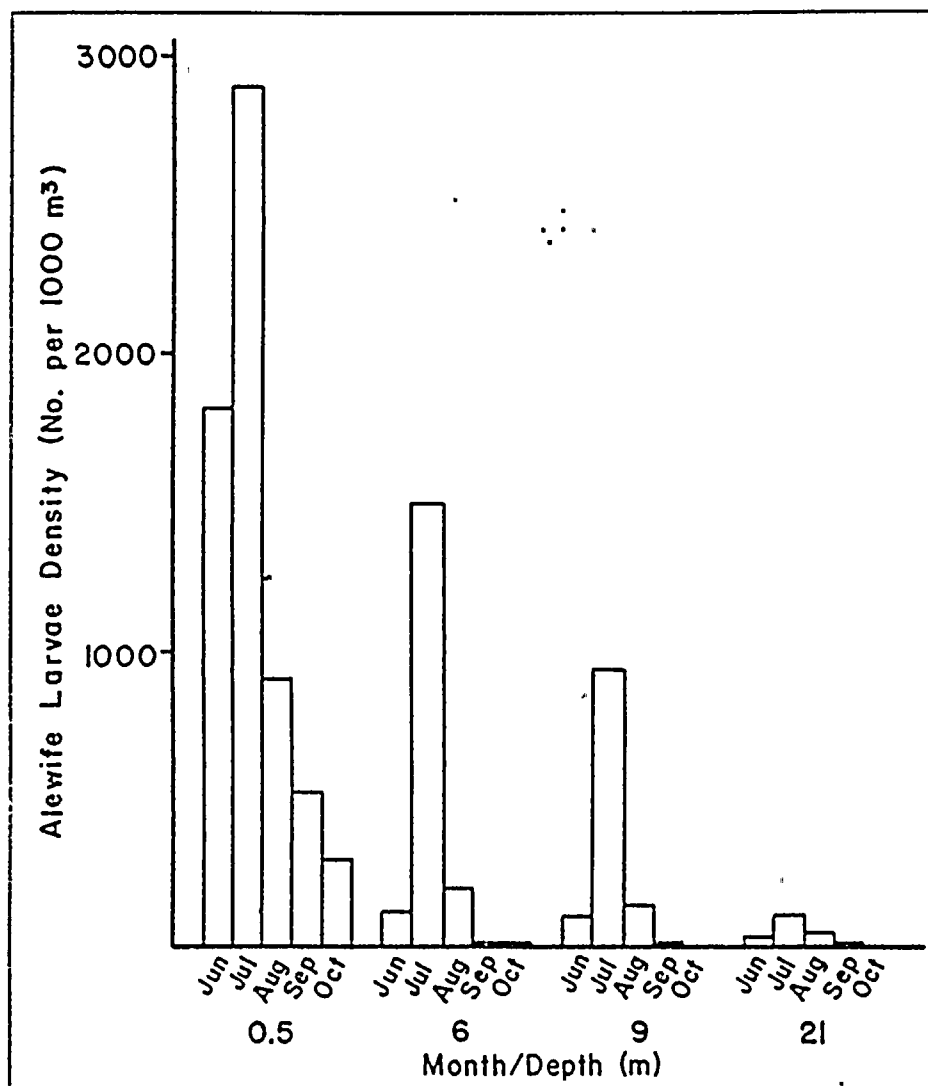


FIGURE 3. Depth distribution of alewife larvae in the vicinity of the D. C. Cook Nuclear Plant, southeastern Lake Michigan, from June to October, 1975-1979. All samples from beach (A, B, F), 6-m (C, G, R), 9-m (D, H), and 21-m (E, W) stations were included in this analysis.

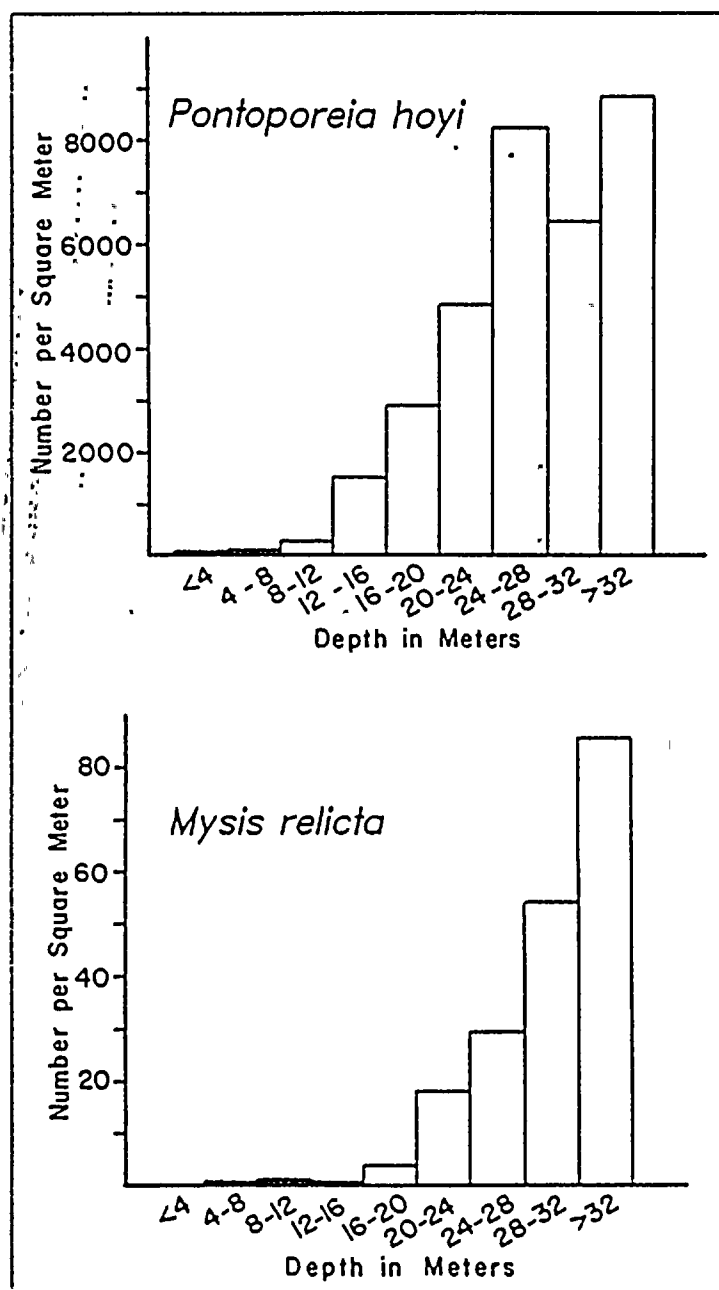


FIGURE 4. Depth distribution of benthic *Pontoporeia hoyi* and *Mysis relicta* in the vicinity of the D. C. Cook Nuclear Plant, southeastern Lake Michigan (Winnell 1984).

ADULT AND JUVENILE FISH

Field

Impact as Measured by Area Differences—

From 1973 to 1982, fish were surveyed by gillnetting, seining, and trawling at the plant site and at a reference site, Warren Dunes. The primary purpose of this surveillance was to determine the distribution and abundance of fish at the two areas before and during plant operation. For fish species caught in sufficient numbers, statistical tests were employed to establish whether differences between area catches were significant. Data on other species were examined for consistency or trends in annual abundances in the two areas. Details on statistical methods are presented in Jude et al. (1975, 1979) and Tesar et al. (1985). Madenjian et al. (1986) performed extensive statistical analyses on these data and compared ANOVA results with intervention analysis.

Similar annual abundances or distributions of a species at both sites over the 10 years established that the plant or its operation did not measurably affect juvenile and adult fishes. Changes in annual abundance which simultaneously occurred at both sites again established a lack of plant effect. On the other hand, consistently smaller or larger annual catches in the Cook Plant area compared with the Warren Dunes area were assumed to have resulted from the power plant. In general, four categories of effects in the Cook Plant area were noted. In general, four categories of effects in the Cook Plant area were noted. (1) There was greater abundance during preoperational and operational years resulting from fish being attracted to the plant's riprap. Diver observations helped to confirm this. We have no catch data before riprap placement, but because the topography and bottom sediments were similar at the Cook Plant and Warren Dunes areas, we believe that the preconstruction distributions of fish were similar. (2) There was greater abundance during operational years resulting from an attraction to the riprap, structures, and currents. (3) There was lesser abundance during preoperational and operational years resulting from fish avoiding the alterations caused by construction dredging and discharges. (4) There was lesser abundance during operational years resulting from either avoidance of the discharges and their effects or mortality caused by substantial impingement or entrainment. The objective of this section is to summarize these results. Details have been presented by Jude et al. (1975, 1979), Tesar et al. (1985), Jude and Tesar (1985), Tesar and Jude (1985), and Madenjian et al. (1986).

Over the 10 years, just over 1,100,000 fish of 59 species were caught (Table 4). Six of these species were abundant in the study areas. Alewife predominated, constituting 61% of the total catch over the 10 years. Spottail shiner contributed 21% of the total, rainbow smelt and yellow perch each contributed 7%, and trout-perch and bloaters each contributed just under 2%. All other species combined made up 1% of the total. Common species were defined as those with an average catch > 20 but < 1,000 fish/year.

Abundant species—Based on field catch data, alewife, bloater, and rainbow smelt were not affected by the power plant (Figs. 5, 6, 7). All three species were, at times, impinged in substantial numbers; entrainment of alewife larvae was also substantial. It appears therefore, that the field sampling design was either not precise enough to detect population declines from this plant-caused mortality given natural population fluctuations, or that populations of these species were able to compensate for the

TABLE 4. Total number of fish collected in standard series sampling at the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1973-1982. Common names assigned according to Robins et al. (1980).

Species	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982
Alewife	148,450	76,099	41,738	136,743	55,979	39,453	142,173	16,450	26,486	8,237
Spottail shiner	20,583	24,413	19,814	14,116	22,568	36,601	27,820	32,839	22,631	12,845
Rainbow smelt	16,294	5,754	3,109	1,265	1,455	9,521	5,276	12,443	14,250	7,837
Yellow perch	3,735	4,536	4,334	2,498	3,379	1,576	4,659	12,770	25,850	17,635
Trot-perch	3,558	1,578	905	1,955	3,239	3,088	1,730	3,122	1,608	280
Bloater	126	225	49	107	227	1,392	2,988	2,861	9,912	1,461
Banded killifish	-	-	-	-	-	-	-	-	-	1
Black bullhead	2	2	-	-	-	-	-	3	1	-
Blackchin shiner	-	-	-	-	-	1	-	-	-	-
Black crappie	-	-	-	-	-	-	1	-	-	-
Blacknose dace	-	-	-	-	-	-	-	-	1	-
Blacknose shiner	-	-	-	-	-	-	-	-	1	-
Bluegill	10	46	2	3	2	-	1	-	11	1
Bluntnose minnow	-	1	-	-	1	-	1	-	1	-
Brook silverside	-	-	-	1	-	1	-	-	1	-
Brown trout	78	51	26	90	61	162	60	40	3	24
Burbot	6	15	15	6	8	5	5	7	6	3
Central mudminnow	-	-	-	-	-	-	1	-	-	2
Channel catfish	10	17	9	13	9	5	8	-	20	9
Chinook salmon	29	47	59	35	160	337	332	182	47	25
Coho salmon	23	147	54	25	12	70	65	13	8	3
Common carp	27	27	50	32	92	36	71	28	48	84
Common shiner	-	-	-	-	-	-	-	-	-	1
Creek chub	-	-	-	-	1	-	-	-	-	-
Emerald shiner	49	13	1	-	27	10	12	-	19	17
Fathead minnow	2	-	-	-	-	1	2	1	1	-
Freshwater drum	-	-	-	-	1	-	-	-	-	18
Gizzard shad	23	84	193	51	104	106	159	53	82	389
Golden redhorse	-	-	-	-	9	-	3	-	-	1
Golden shiner	2	1	-	2	1	2	-	-	-	1

TABLE 4. Continued

Species	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982
Grass pickerel	—	—	—	—	—	—	—	—	1	—
Green sunfish	—	6	—	—	—	—	1	—	—	—
Johnny darter	207	293	142	304	423	401	233	198	151	92
Lake chub	—	—	—	—	—	—	1	—	—	—
Lake herring	—	1	1	—	—	1	—	—	—	—
Lake sturgeon	—	—	1	1	2	1	—	—	—	—
Lake trout	162	125	85	37	187	286	164	121	105	176
Lake whitefish	2	1	2	6	—	9	7	14	7	5
Largemouth bass	1	1	1	1	—	—	—	—	2	2
Logperch	—	—	2	—	—	—	—	—	—	—
Longnose dace	41	43	18	27	60	26	6	34	3	2
Longnose sucker	73	99	94	40	99	71	98	47	70	50
Mottled sculpin	16	—	—	—	3	—	6	15	7	10
Ninespine stickleback	19	24	26	9	7	5	8	8	2	2
Northern pike	30	16	6	—	—	2	4	—	1	1
Pumpkinseed	—	—	1	—	—	—	—	—	—	—
Quillback	—	—	1	2	3	2	—	—	—	6
Rainbow trout	86	8	15	14	12	21	14	22	18	4
Rock bass	2	—	—	—	1	—	—	—	1	—
Round whitefish	—	—	—	—	—	—	2	—	3	3
Sand shiner	—	4	34	39	23	12	7	10	27	12
Shorthead redhorse	—	—	4	—	1	—	4	—	1	8
Silver redhorse	—	—	1	3	1	1	10	1	3	5
Slimy sculpin	79	272	111	85	30	14	128	55	74	199
Smallmouth bass	—	—	—	1	—	—	—	—	1	—
Spotfin shiner	—	—	—	—	—	2	3	—	—	—
Walleye	—	—	—	—	—	—	—	—	1	4
White crappie	—	—	—	—	—	—	—	1	—	—
White sucker	174	126	89	89	173	118	188	113	131	169
Total number	193,899	144,075	70,992	157,600	88,360	93,339	186,241	81,451	101,596	49,624

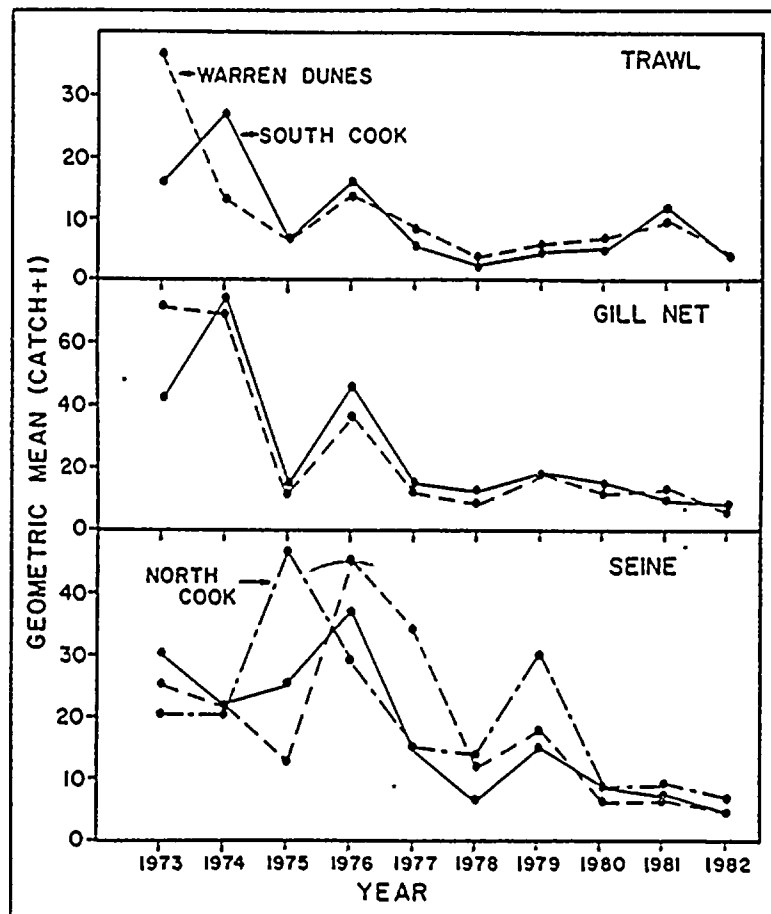


FIGURE 5. Yearly geometric mean number of alewives caught by standard series trawling, gillnetting, and seining in the study areas of southeastern Lake Michigan. The Cook Plant began operation in 1975.

mortality (see Mansfield et al. 1985). These species are three of the most abundant and mobile forage species in the lake. Thus, immigrations from other areas could obscure any depletion in numbers caused by local mortality.

Spottail shiner, trout-perch, and yellow perch abundance (Figs. 8, 9, 10) changed significantly; therefore, we assumed they were affected by the power plant. None of these effects was massive because for each species, only one of the three fishing gear detected a significant or consistent difference. Trawl catches of spottail shiner and trout-perch were significantly smaller in the Cook Plant area than at Warren Dunes in operational years; gill net catches were also smaller but the differences were not

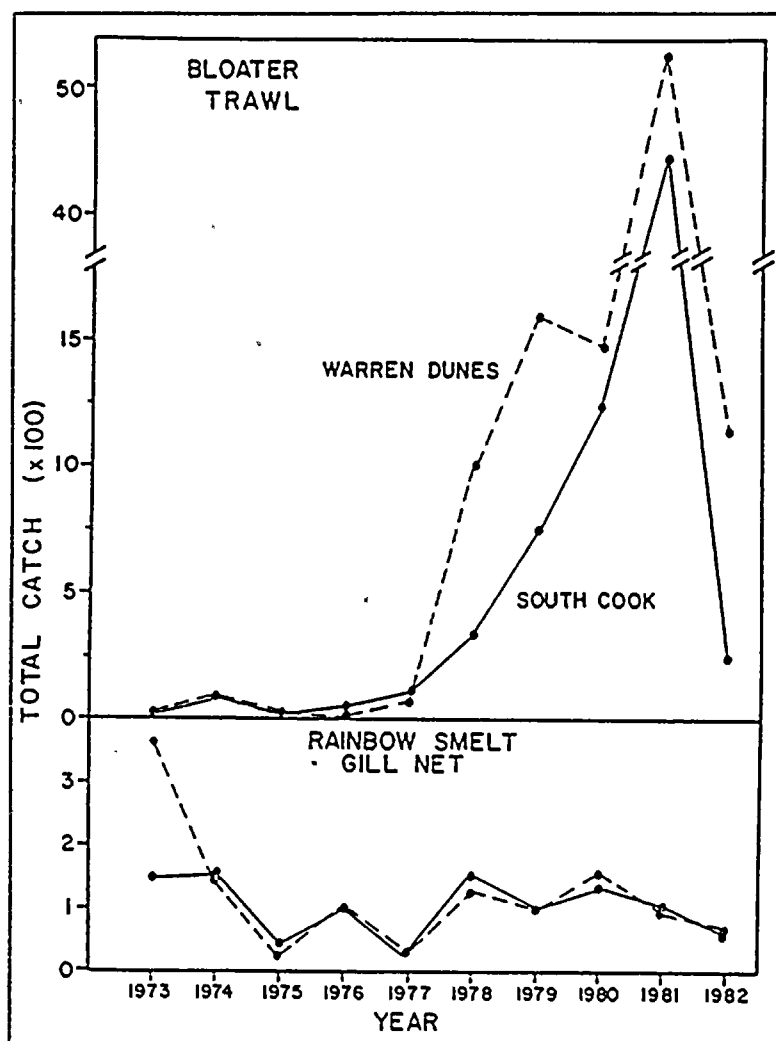


FIGURE 6. Annual catches of bloater and rainbow smelt in the study areas of southeastern Lake Michigan. The Cook Plant began operation in 1975.

significant. Because considerable numbers of both species were impinged, it is presumed that plant-induced mortality depressed their abundance in the Cook Plant area.

Results for yellow perch were ambiguous regarding a plant effect. During 3 of the last 4 years of the study, trawl catches were significantly larger at Warren Dunes than at the Cook Plant. Gill net catches were larger at Warren Dunes than at the

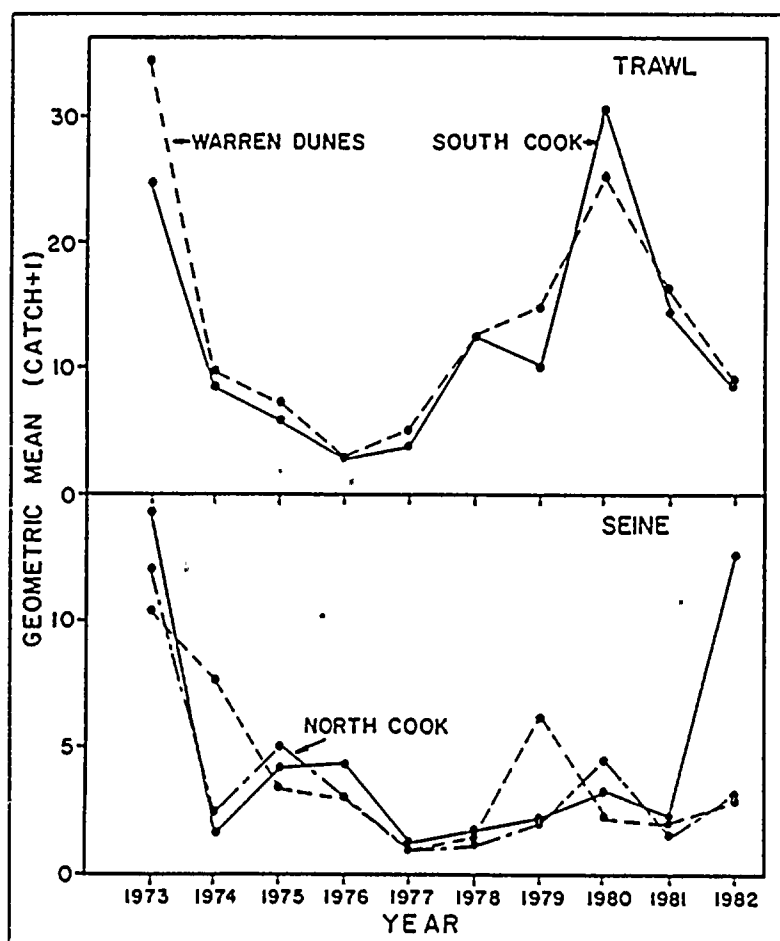


FIGURE 7. Yearly geometric mean number of rainbow smelt caught by standard series trawling and seining in the study areas of southeastern Lake Michigan. The Cook Plant began operation in 1975.

Cook Plant. Gill net catches were larger at Warren Dunes during 1981 and 1982, but larger at the Cook Plant during the other operational years. Because many yellow perch were impinged, especially in 1981-1982, these findings suggest that impingement mortality is depressing the abundance of this species, but only in recent years. Attraction of predominantly large yellow perch (> age 1) to the plant's riprap and underwater structures was indicated from diver observations (Dorr and Jude 1986, Dorr and Miller 1975, Dorr and Jude 1980) and from personal observations of fishermen at the plant site. It was presumed that yellow perch concentrated in the area because it provided food (crayfish, other invertebrates, and forage fish).

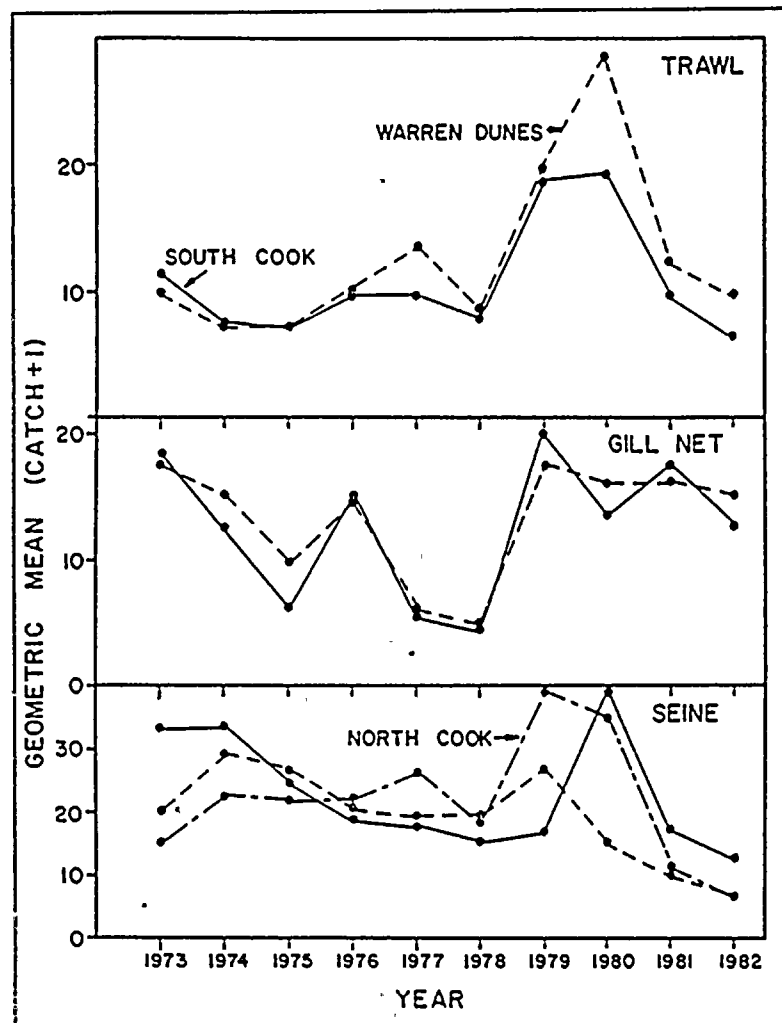


FIGURE 8. Yearly geometric mean number of spottail shiners caught by standard series trawling, gillnetting, and seining in the study areas of southeastern Lake Michigan. The Cook Plant began operation in 1975.

spawning substrate, and shelter from storm-driven currents and waves. The lack of consistent trends in field catches confounded the substantiation of these effects. Further, the combination of yellow perch being attracted to the plant site, thus increasing their abundance, with impingement mortality decreasing their abundance, may prevent determination of plant impact strictly using field catches.

Madenjian et al. (1986) performed extensive statistical tests on Cook Plant catch

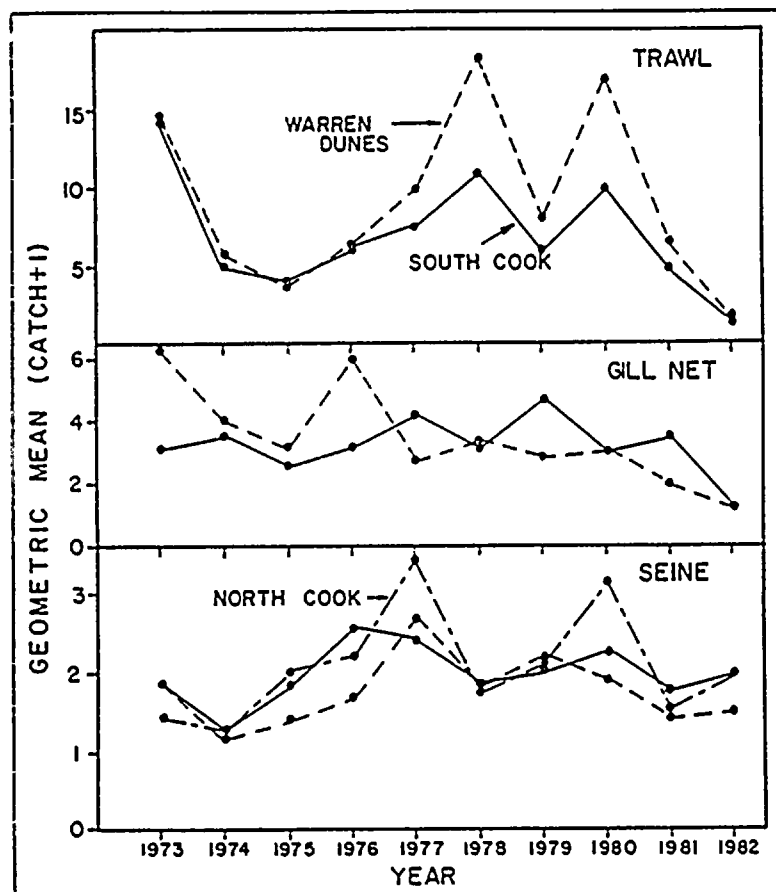


FIGURE 9. Yearly geometric mean number of trout-perch caught by standard series trawling, gillnetting, and seining in the study areas of southeastern Lake Michigan. The Cook Plant began operation in 1975.

data for yellow perch and alewife, examining trends in the catches for evidence of power plant impact. They also compared the utility of ANOVA versus intervention analysis to detect significant catch differences between study areas for these two species. No significant power plant impacts were found except for gillnetted yellow perch adults. The ANOVA indicated that there was a significant decrease in abundance of yellow perch at the plant-discharge transect relative to the reference transect as plant operation began. This established a plant effect where intervention analysis showed no change. However, when April-May catches (months of low abundance) were deleted, this plant effect was insignificant. Madenjian et al. concluded that impact assessment based only on ANOVA can result in detection of

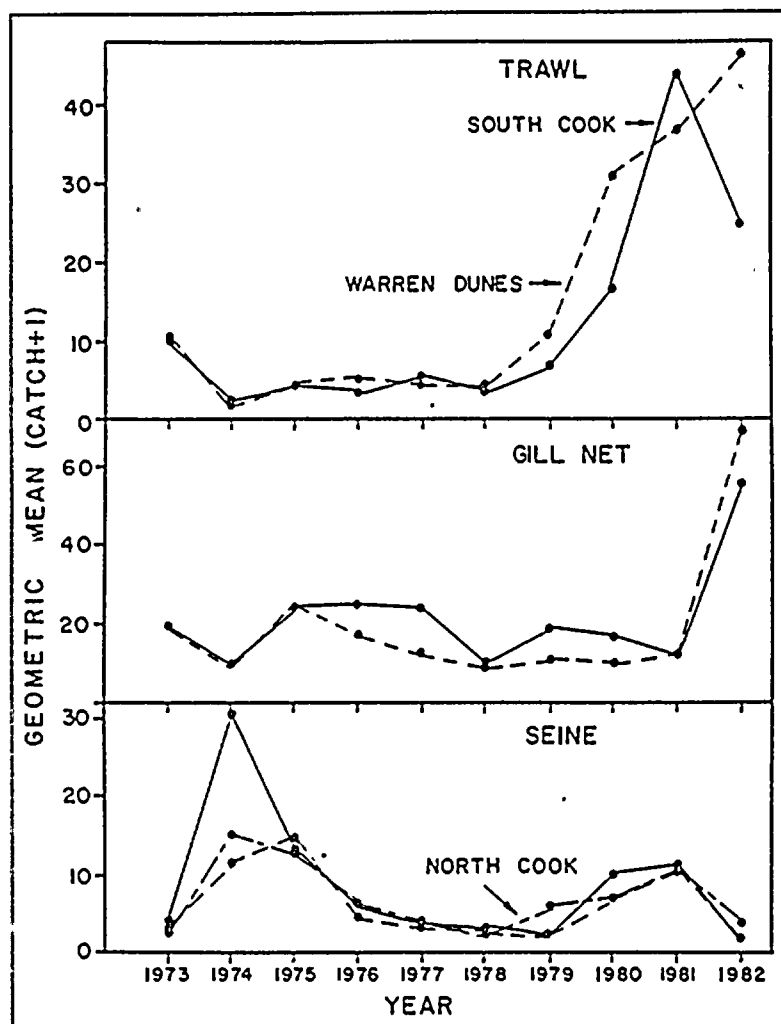


FIGURE 10. Yearly geometric mean number of yellow perch caught by standard series trawling, gillnetting, and seining in the study areas of southeastern Lake Michigan. The Cook Plant began operation in 1975.

impact when actually there was no effect (type 1 error) when observations are serially correlated (lack independence).

Common and rare species—Of 12 common species, catches of common carp, gizzard shad, johnny darter, lake trout, longnose sucker, slimy sculpin, and white sucker were significantly different between the study areas, presumably because of the power plant. In addition, brown trout and longnose dace showed trends in catch

attributable to plant effects. Brown trout, common carp, and gizzard shad were attracted to the discharge plumes presumably because of the warm water and entrained food organisms. Brown trout may have been attracted to crayfish and forage fish which were abundant on the riprap. Johnny darters, lake trout, and slimy sculpins were attracted to the plant's riprap because it provides shelter, spawning substrate, and food organisms. Longnose suckers and white suckers were significantly more abundant at Warren Dunes during all 10 study years. Studies at other Lake Michigan power plants (WEPC and WMPC 1975, Brazo and Liston 1979, Jude et al. 1981) also indicated lesser abundance of these species at plant sites compared to reference sites. Suckers may avoid plant sites because of dredging during construction and discharges during operation. Longnose dace prefer gravel-rock substrates rather than sandy substrates (Brazo et al. 1978) and presumably were attracted to the gravel and construction debris left at the south end of the Cook Plant site.

Forty-one species were categorized as rare in the study area. Ten of these species showed distinct trends in catch which we attributed to plant effects. Larger catches of channel catfish, emerald shiner, mottled sculpin, and northern pike at the Cook Plant were attributed to an attraction to the plant's riprap and underwater structures for shelter and food. Larger catches of golden redhorse, quillback, shorthead redhorse, and silver redhorse during operational years at the Cook Plant were attributed to an attraction to entrained food organisms in the discharges. Lake whitefish and sand shiner catches were greater at Warren Dunes, possibly because these species avoid the discharge plumes and their effects.

In summary, the abundance or distribution of 22 out of a total of 59 fish species was altered by the physical structure of the Cook Plant or its operation. These alterations varied from a minor change in distribution (for example, the apparent attraction of a few redhorse suckers to the plant's discharges) to decreases in spottail shiner, trout-perch, and yellow perch abundances at the plant site. The consequences of these changes to the lake's fish populations were not resolved and may be impossible to determine. For example, the plant's riprap attracted fish, thereby increasing their vulnerability to entrapment mortality, consequently diminishing abundance of some species. On the other hand, the riprap provided spawning substrate and food organisms which enhanced growth, reproduction, and survival of some species. Community structure may be only locally altered, with no important effect on the lake as a whole. In addition, the ability of the lake's fish populations to compensate for local abundance declines is unknown, and depends on the species, geographical area, and population density. For more information on compensation, the available literature is discussed and synthesized by Mansfield et al. (1985).

Impingement

During the first 8 years of plant operation, 1975-1982, the number of fish impinged annually at the Cook Plant ranged from 53,190 fish in 1977 to 2,307,654 fish in 1980 (Table 5; Thurber and Jude 1984, 1985). Both reactor units were operational by 1978, though maximum cooling water flow was not sustained for extended periods until 1980. From 1980 through 1982, the period of consistent two-unit operation, the minimum number of fish impinged annually (913,768) was four times the maximum number of fish impinged annually (224,736) during 3 years of one unit operation.

During the same 8-year period, annual total weight of impinged fish ranged from

TABLE 5. Common names (according to Robins et al. 1980) and total estimated number of each species impinged during 1975-1982 at the D. C. Cook Nuclear Plant, southeastern Lake Michigan. Weight is in kg.

Species	1975	1976	1977	1978	1979	1980	1981	1982
Alewife	174,341	114,958	31,498	238,133	330,709	1,815,490	1,415,821	831,051
Black bullhead	35	45	16	12	4	9	35	68
Black crappie	11	4	7	2	5	6	5	9
Bloater	49	63	302	23,085	2,456	21,448	3,144	212
Bluegill	48	23	10	11	—	12	73	37
Brown bullhead	—	—	—	11	4	—	7	6
Brown trout	—	37	24	61	95	120	166	176
Burbot	37	75	51	108	575	1,248	876	1,018
Central mudminnow	9	9	—	—	5	24	43	66
Channel catfish	50	70	27	26	50	87	175	87
Chestnut lamprey	4	—	—	5	—	—	—	—
Chinook salmon	7	16	—	59	729	875	22	34
Coho salmon	8	22	22	78	165	63	44	530
Common carp	2	6	—	5	34	33	18	12
Deepwater sculpin	1	5	—	—	—	27	80	33
Emerald shiner	1	—	—	5	—	—	—	—
Flathead catfish	—	—	—	—	—	31	—	—
Freshwater drum	—	—	—	18	2	4	3	8
Gizzard shiner	278	1,780	35	692	252	669	1,682	1,925
Golden shiner	5	—	—	—	—	—	—	9
Goldfish	2	—	—	—	5	4	—	—
Grass pickerel	—	1	—	—	—	—	—	—
Green sunfish	13	6	4	6	—	6	14	—
Johnny darter	180	346	103	108	59	107	682	13
Lake chub	—	5	6	6	—	13	—	32
Lake chubsucker	—	—	4	—	4	—	—	—
Lake herring	—	—	—	—	—	—	—	5
Lake sturgeon	—	—	—	—	—	8	—	—
Lake trout	101	115	115	243	282	320	517	342
Lake whitefish	1	—	—	—	10	15	7	8
Largemouth bass	13	4	8	—	11	5	—	—
Logperch	1	—	—	—	—	—	—	—

TABLE 5. Continued

Species	1975	1976	1977	1978	1979	1980	1981	1982
Longnose dace	6	8	19	43	—	5	8	8
Longnose gar	—	—	—	—	—	—	—	3
Longnose sucker	23	43	20	165	210	490	266	629
Mottled sculpin	—	—	14	392	532	1,078	1,364	373
Ninespine stickleback	194	107	95	288	65	429	111	71
Northern pike	3	17	—	5	—	—	17	7
Pirate perch	1	—	—	—	—	—	—	—
Pumpkinseed	23	32	2	15	—	—	5	9
Quillback	2	—	—	—	—	—	—	—
Rainbow smelt	3,746	2,772	1,488	51,013	35,398	149,085	112,837	13,863
Rainbow trout	4	17	—	6	14	20	37	24
Rock bass	3	1	4	8	5	3	14	3
Round whitefish	—	—	—	—	—	—	39	—
Sea lamprey	—	—	—	—	5	9	8	30
Shorthead redhorse	—	—	5	30	68	—	14	5
Silver redhorse	—	—	—	5	11	—	—	—
Slimy sculpin	8,136	7,402	2,232	1,034	2,622	8,371	6,974	5,820
Smallmouth bass	5	21	10	3	5	15	8	—
Spottail shiner	9,985	24,104	5,032	178,009	52,761	106,009	86,260	33,842
Spotted sucker	1	—	—	—	—	—	3	—
Stonecat	—	—	—	—	—	11	—	—
Tadpole madtom	—	5	—	—	—	—	—	6
Trout-perch	15,373	10,357	4,826	88,692	15,002	31,063	23,711	1,998
Walleye	—	—	—	—	—	—	—	6
Warmouth	—	—	—	—	—	2	—	—
White crappie	6	—	—	11	2	5	18	—
White sucker	16	27	14	186	271	173	141	584
Yellow bullhead	5	1	2	—	—	—	3	6
Yellow perch	12,006	21,309	7,195	32,811	38,349	170,262	391,983	38,811
Total number	224,735	183,813	53,190	615,390	480,776	2,307,654	1,947,235	913,768
Total weight	6,131	4,927	1,833	10,475	9,480	71,209	17,395	25,173

1,833 kg in 1977 to 71,209 kg in 1980 (Table 5). Generally, yearly differences in total weight of impinged fish paralleled yearly differences in total number of impinged fish, except in 1981 when weight was low compared to number of fish impinged. This discrepancy occurred because exceptionally low numbers of adult and large numbers of yearling alewives were impinged in 1981.

Alewife was the most abundant species in impingement samples, making up from 39% (1978) to 89% (1982) of the total number of fish impinged annually. Percent contribution, by number, that each species made to each year's total was averaged for 1975-1982; alewife comprised 68% of the total. Following in order of abundance were spottail shiner (10%), yellow perch (9%), trout-perch (5%), rainbow smelt (4%), and slimy sculpin (2%). This distribution was slightly different when contribution by weight to total biomass impinged annually was averaged for each species. Alewife still contributed the most, averaging 64% of total biomass. Following in order of their contribution were yellow perch (14%), lake trout (7%), spottail shiner (4%), trout-perch (2%), and longnose sucker (1.5%). The disproportionate contribution of lake trout and longnose sucker to total biomass, compared with the relatively low numbers of those species impinged, was due to generally large (primarily adult) fish being impinged.

Less abundant species made up the remaining 2% by number and 7.5% by weight of impinged fish (Table 5). Over 8 years, 61 species were impinged at the Cook Plant. Of these, 19 species were impinged during only 1 or 2 years and can be considered rare in the area. Fourteen species were impinged (Table 5) but were never collected in field samples (Table 4), and 12 species were collected in the field but were never impinged.

In 1975, 1980, and 1982, alewives dominated impingement collections (over 78% by number and over 81% by weight of fish) to a larger extent than in other years. During these years exceptionally large numbers of adult alewives were impinged in April and May, when the plant was operating continuously at full capacity (Table 6). Coincidental with full capacity operation was a pattern of nearshore warming (thermal bar) which resulted in a sharply defined and relatively narrow band of water close to shore being several degrees warmer than offshore water. Under these conditions, alewives concentrated inshore and were impinged in large numbers, while other species evidently did not concentrate as densely as alewives within this narrow band of warm water.

Species Composition—

Species composition of impinged fish remained similar from year-to-year, but most species showed a large annual variation in numbers impinged and in percent contribution to total number or weight. There was no clear trends, and most variation could be attributed to annual differences in year class strength, differences in weather phenomena that might contribute to impingement, and differences in cooling water flow during periods when each species was most abundant in the inshore area.

There were minor exceptions to the "no trend" pattern. Trout-perch comprised a lesser proportion of the fish impinged after both units became operational in 1978, dropping from an average 2.4% by number and 7.3% by weight (1975-1977), to 0.5% by number and 1.3% by weight (1980-1982). Divers did not observe large aggregations of trout-perch around the intake structures (Dorr and Jude 1986), and impingement patterns suggested that trout-perch were not attracted to the intakes

TABLE 6. Monthly water volume (in millions of cubic meters) pumped through the condenser circulating water system of the D. C. Cook Nuclear Plant, southeastern Lake Michigan, from 1975 to 1982. Unit 1 has been operational since January 1975. Unit 2 since February 1978.

Month	1975	1976	1977	1978	1979	1980	1981	1982
January	64.9	85.7	24.9	114.4	273.2	142.5	270.8	275.7
February	75.6	88.5	54.5	121.6	275.2	280.9	282.5	177.8
March	117.7	103.6	118.7	207.1	281.9	314.4	213.6	195.5
April	121.0	76.2	114.5	115.9	173.7	304.5	128.8	291.8
May	125.8	86.0	97.4	90.4	100.5	318.4	196.5	308.6
June	122.8	122.7	93.5	194.4	33.3	167.6	165.0	309.2
July	81.7	120.5	103.6	224.5	227.7	101.0	142.4	190.3
August	128.7	130.5	123.3	249.6	324.6	297.0	290.6	87.4
September	125.2	109.0	97.7	277.6	314.3	303.1	305.0	206.4
October	132.2	137.9	112.4	298.8	245.9	247.7	182.4	296.1
November	90.6	126.2	76.3	202.8	107.3	125.8	265.3	265.2
December	111.6	105.1	120.9	272.5	118.0	227.1	310.2	145.4
Annual total	1,298	1,292	1,138	2,370	2,476	2,830	2,753	2,749

(Thurber and Jude 1984). In addition, field catches (Jude and Tesar 1985) showed a decline occurred in catches of trout-perch in 1982 when collections were down by 13% compared with the 1973-1982 mean. A concomitant increase in yellow perch populations, a known predator of trout-perch, may be responsible for the decline we observed in both impingement and field collections.

Slimy sculpin averaged 4% by number and 0.8% by weight of total fish impinged annually during 1975-1977, and averaged 0.4% by number and 0.2% by weight during 1980 through 1982. At the depth of the intake structure (9 m), sculpins reside almost exclusively on the riprap and are seldom observed on the surrounding sandy plain (Dorr and Jude 1980). Thus, compared to other species, there is a very limited population susceptible to impingement.

During 1975-1979 (one unit operation), an average of 160 salmonids was impinged annually. An average of over 1,000 salmonids was impinged annually during 1980-1982 (Table 5). In 1975-1979 and 1981, most salmonids impinged were adult and juvenile lake trout (517 fish in 1981), but in 1980 and 1982, the most abundant salmonids were juvenile chinook salmon (probably newly planted—875 fish in 1980) and adult coho (530 fish in 1982).

Seasonal Changes in Impingement Rates—

Number of fish impinged per month at the Cook Plant varied seasonally (Thurber and Jude 1985). Most fish were impinged during April through October; few were impinged during winter. Each year was characterized by a month of peak impingement during June or July, and often another (usually secondary) peak in spring (April-May) or fall (September-November). During spring, many fish may be impinged as they move shoreward seeking warm water. High fish impingement rates during this time may result from increased activity and movement of fish and high densities of fish inshore, particularly when a well defined and persistent thermal bar occurs which may effectively confine large numbers of fish to a narrow band of warm water close to shore.

In June and July, impingement losses may be quite high as fish move shoreward to spawn. Among the more abundant species, alewife, spottail shiner, yellow perch, and trout-perch spawn during these months. Autumn impingement losses may be attributed to increased movement of fish as the water becomes isothermal and fish move offshore toward deeper water. In addition, impingement of large numbers of young-of-the-year fish occurs when they attain a size which can be retained by traveling screens.

Species composition of impingement losses also changed seasonally (Figs. 11-16). This was particularly notable for alewives, which comprised over 80% of spring losses, nearly 60% of summer losses, about 25% of fall losses, and about 8% of winter losses. The largest component of autumn impingement losses was young-of-the-year and yearling yellow perch; trout-perch also contributed to losses. Though winter impingement losses were low, spottail shiner, yellow perch, and rainbow smelt were all impinged frequently; evidently these species remain or forage near-shore during winter.

The biology of individual species is an important factor determining seasonal patterns in impingement rates. Alewife preference for warm water accounts for their concentration inshore. Peak impingement of alewives occurred during June or July, when alewives concentrated inshore to spawn. In late summer or autumn there was an increase in impingement of young-of-the-year alewives, which by this time were

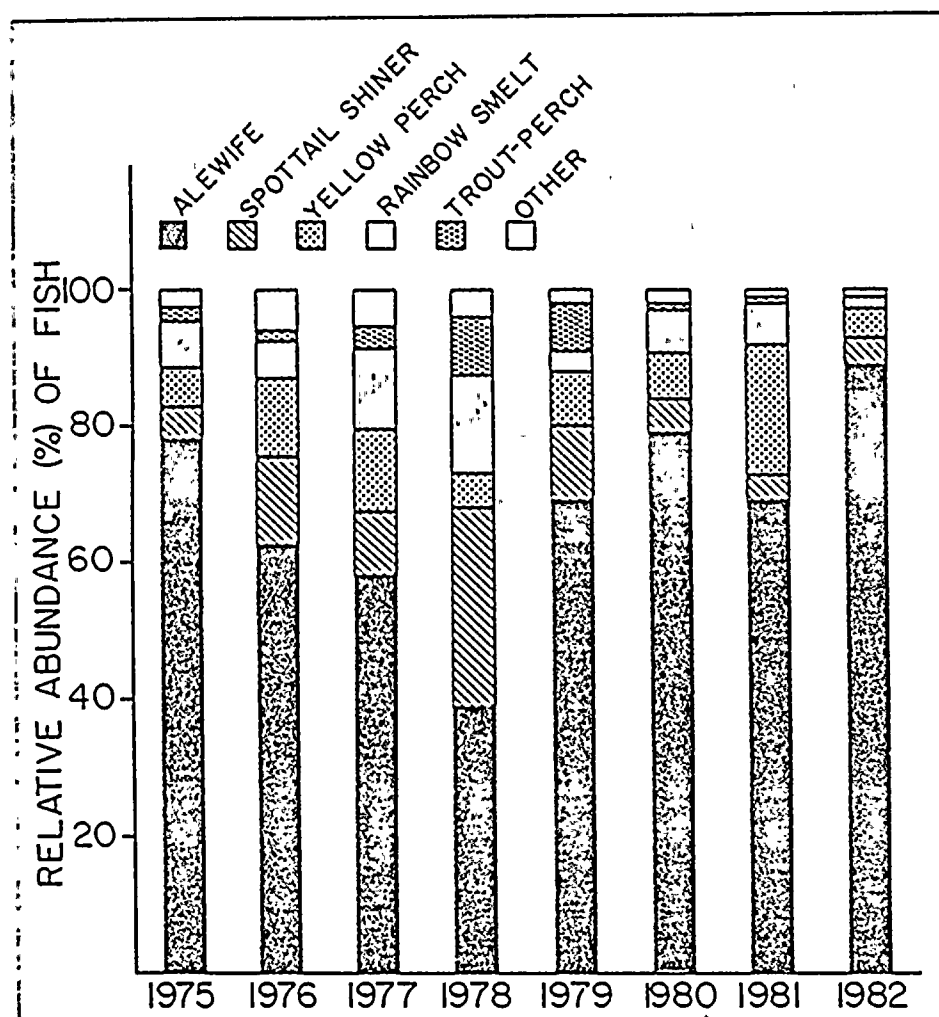


FIGURE 11. Species composition of the total number of fish impinged each year during 1975-1982 at the D. C. Cook Nuclear Plant, southeastern Lake Michigan.

moving offshore from their nearshore nurseries and were large enough (50 mm) to be retained by the traveling screens.

Spottail shiner preference for shallow depths and warm water (Wells 1968, Jude et al. 1979) affected spottail shiner impingement rates. Impingement of spottail shiners increased in March and April as spottail shiners moved shoreward seeking warmer, nearshore water. Except for 1980 and 1981, impingement of spottail shiners during May and June was low; field data indicated spottail shiners were mostly inshore of the intake depth (9 m) (Tesar et al. 1985). Peak spawning for spottail shiners occurs

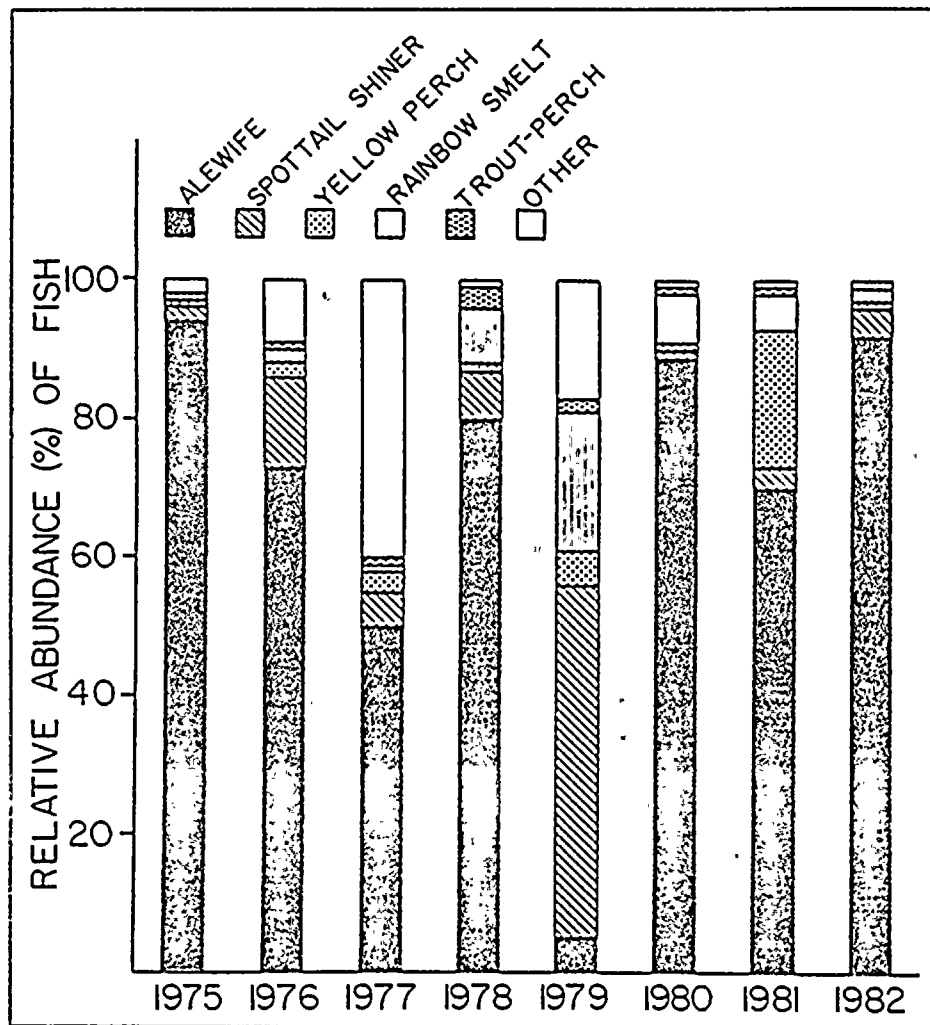


FIGURE 12. Species composition of the total number of fish impinged during spring (March, April, May) 1975-1982 at the D. C. Cook Nuclear Plant, southeastern Lake Michigan. Spring was defined as months of steadily rising water temperature.

in July (Jude et al. 1979), and they also begin their post-spawning migration from the area this month. Large late summer impingement of spottail shiners during 1978-1980 was probably due to increased activity and offshore movement during these months. Impingement of spottail shiners in October was attributed to offshore migration of adults and, in 1975-1977, of young-of-the-year, which by this time were large enough to be retained on the traveling screens. A few spottail shiners

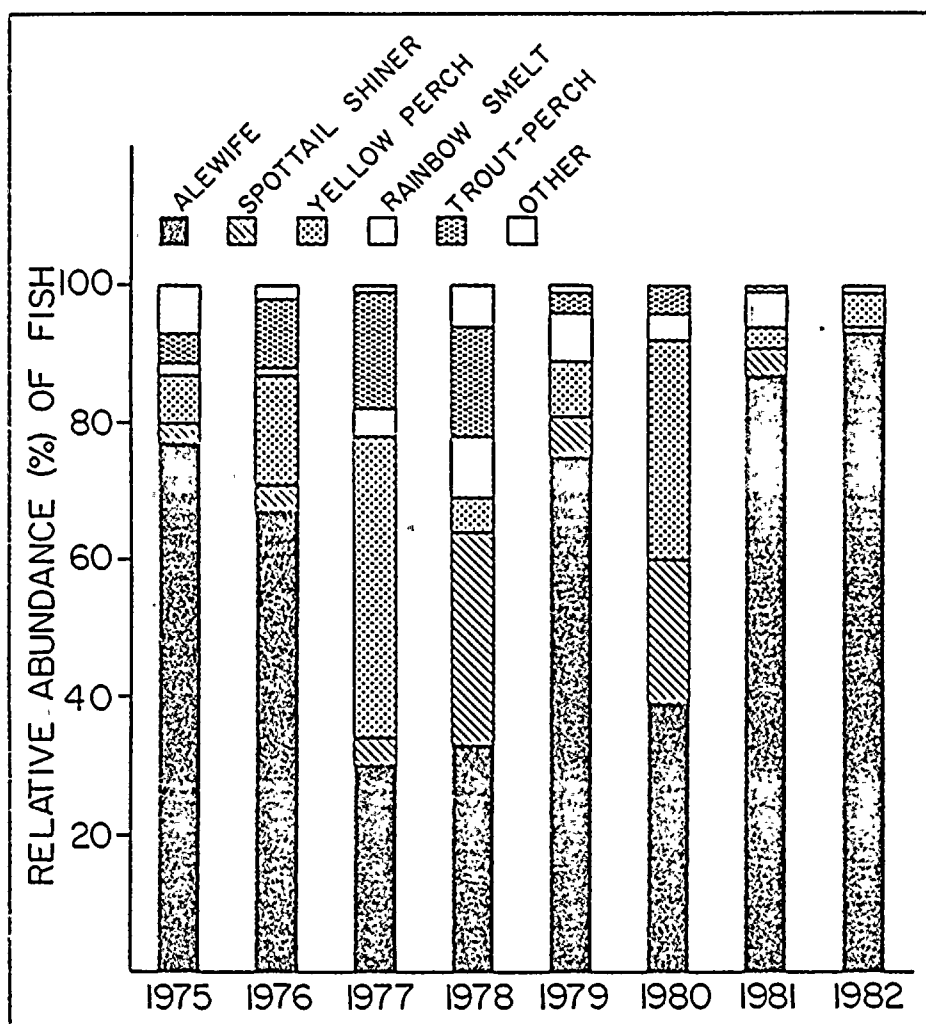


FIGURE 13. Species composition of the total number of fish impinged during summer (July, August, September) 1975-1982 at the D. C. Cook Nuclear Plant, southeastern Lake Michigan. Summer was defined as months of maximum average temperature.

remained inshore all winter. With a few exceptions, winter impingement rates seemed dependent on water temperature, increasing when temperature in the area rose to 3°C or above. Besides water temperature, winter impingement of spottail shiners was probably affected by attraction to the plume, winter storms, and ice cover.

Impingement of trout-perch began to increase in April and peaked in June and

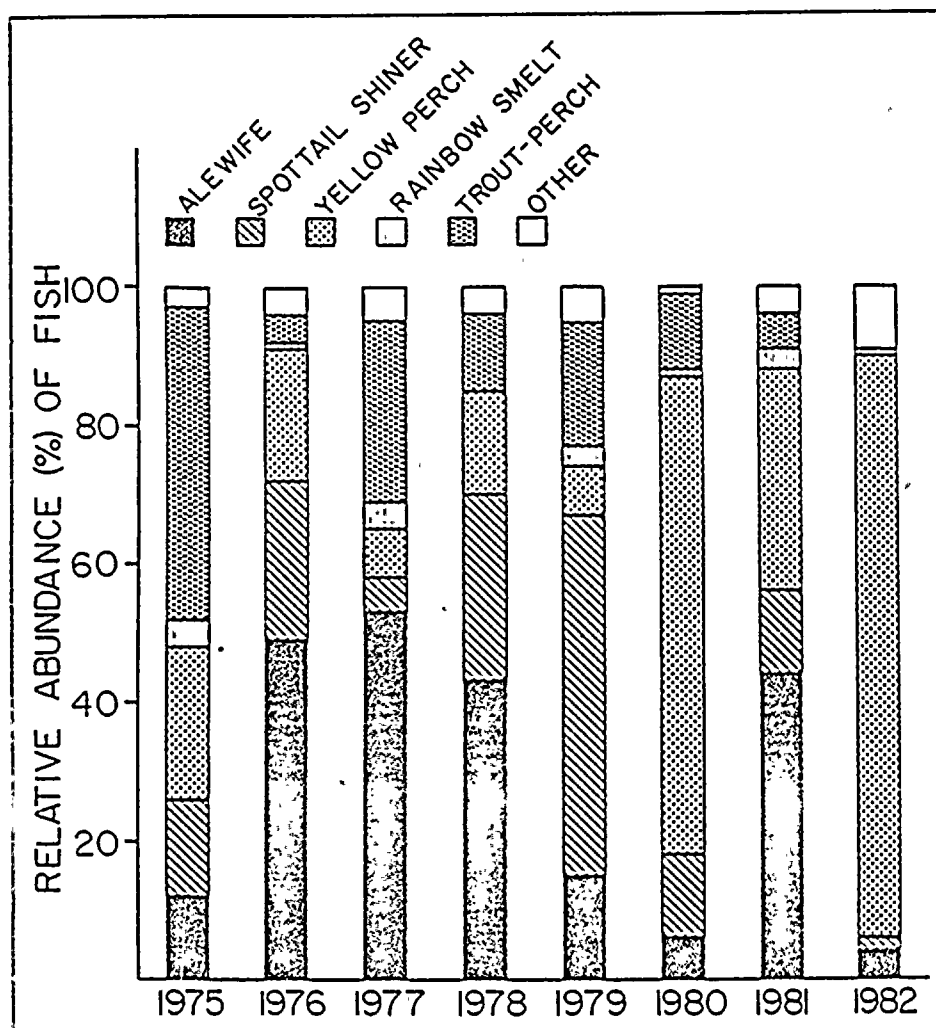


FIGURE 14. Species composition of the total number of fish impinged during fall (October, November) 1975-1982 at the D. C. Cook Nuclear Plant, southeastern Lake Michigan. Fall was defined as months of steadily falling water temperature.

July. In June, trout-perch move inshore for spawning, which continues throughout summer. Impingement of trout-perch during summer was spotty, and appeared to coincide with upwellings or storms. During most years, impingement losses tapered off during late summer and rose again in September or October as trout-perch migrated offshore. A few trout-perch were impinged during winter.

Impingement of yellow perch was usually low in March and April, as inshore water temperatures warmed in spring. Peak impingement of yellow perch occurred

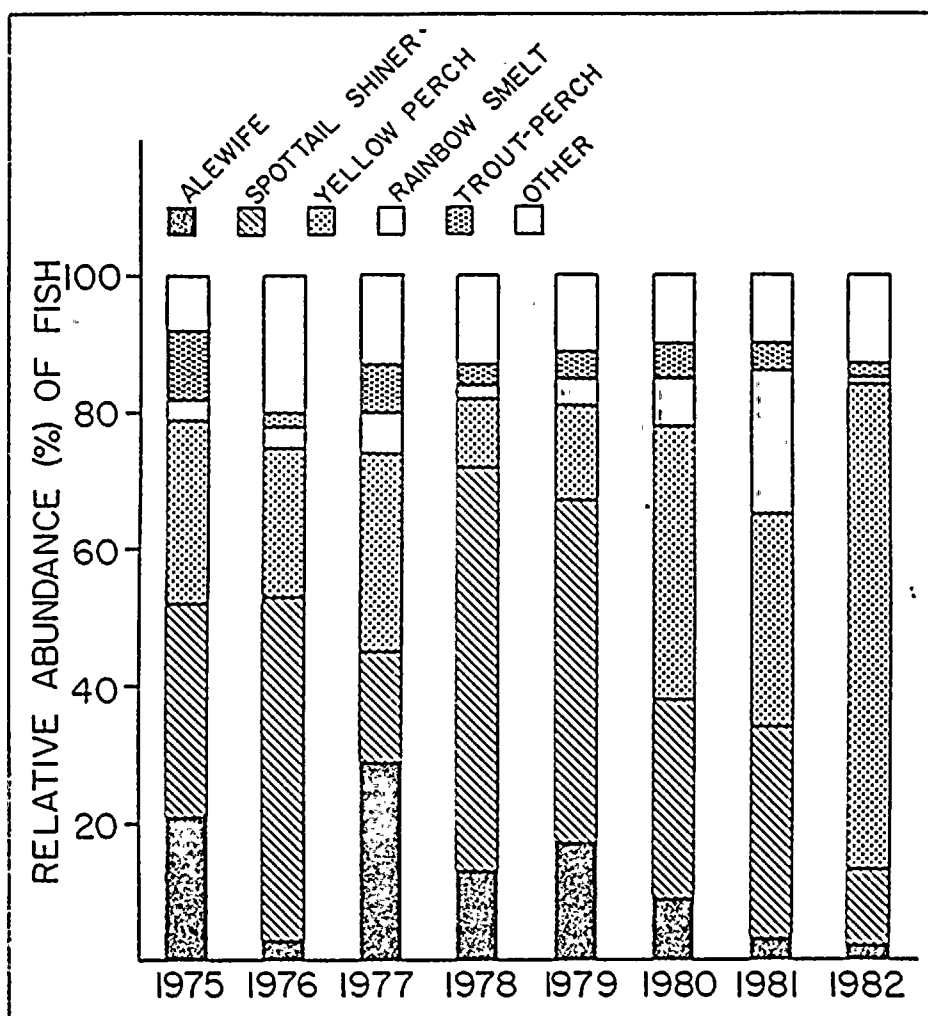


FIGURE 15. Species composition of the total number of fish impinged during winter (December, January, February) 1975-1982 at the D. C. Cook Nuclear Plant, southeastern Lake Michigan. Winter was defined as months of minimum average water temperature.

during June through September, except for 1975. Mid-to-late summer abundance of yellow perch in impingement samples evidently reflected post-spawning migration into the Cook Plant area from other areas of the lake (Jude et al. 1979, Dorr 1982). There was some evidence that impingement of yellow perch during summer increased sharply after storms. Young-of-the-year yellow perch were first impinged

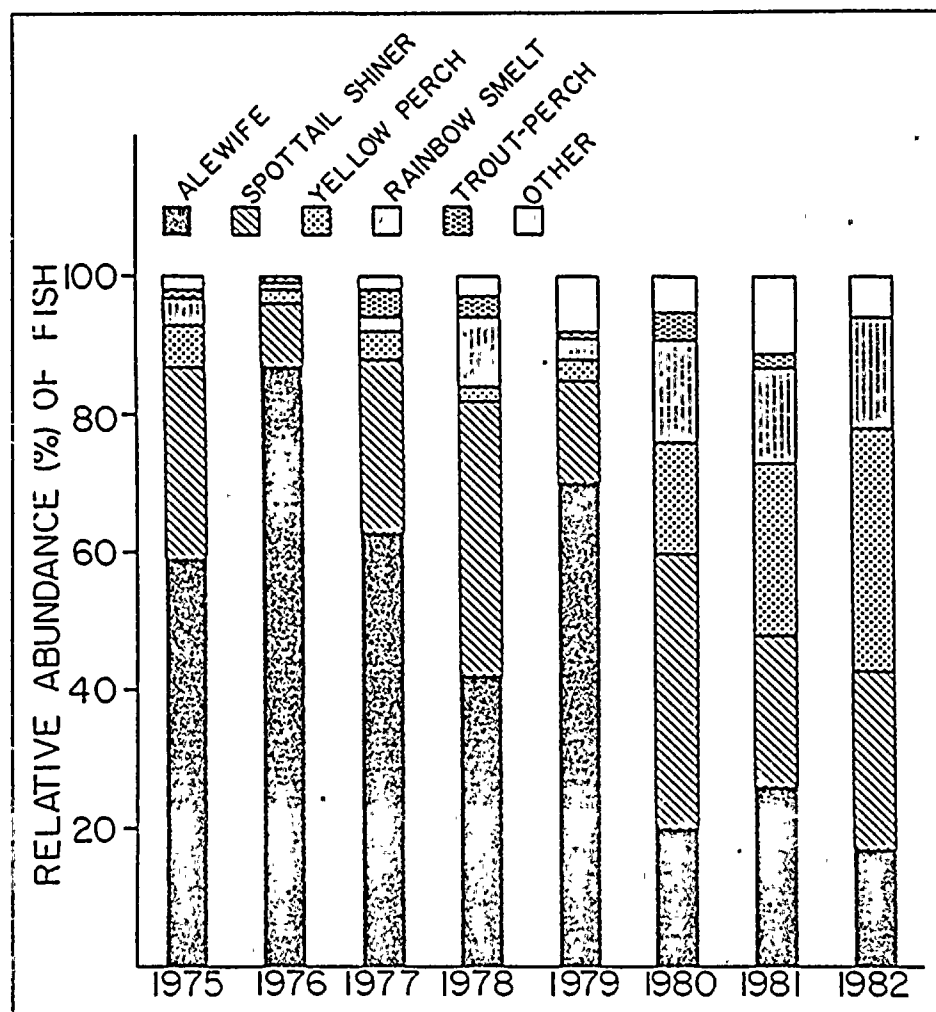


FIGURE 16. Species composition of the total number of fish collected during field sampling each year during 1975-1982 at the D. C. Cook Nuclear Plant, southeastern Lake Michigan.

in September and continued to be impinged during fall and winter, some apparently remaining inshore through the cold months.

Impingement of rainbow smelt usually peaked during spring, most often during April and May. During these months, rainbow smelt migrate inshore to spawn (Jude et al. 1979). Because adult rainbow smelt prefer cold water (Wells 1968, Jude et al. 1979), they move offshore during months when the inshore waters warm. Impingement of rainbow smelt during summer usually occurred when cold, upwelled water

moved shoreward, as in 1978 through 1981 when large numbers of rainbow smelt were impinged during summer. Young-of-the-year rainbow smelt were impinged in September and October.

Year-class strength, which may be related to yearly differences in water temperature during spawning and egg incubation, is another factor determining the magnitude of impingement loss for certain fish species. A strong year-class of alewives was reflected in large numbers of young-of-the-year impinged in fall 1979 and yearlings in spring 1980. Large impingement losses of yearling spottail shiners in 1978, 1980, and 1981 reflected strong year classes from each preceding year. Young-of-the-year and yearling yellow perch produced in 1979 and 1980 were impinged in large numbers in 1980. And a strong 1980 year-class of rainbow smelt contributed to impingement losses in 1981 and 1982.

While plant operation clearly affects impingement rates, number of fish impinged may be equally determined by abundance of fish in the nearshore zone and their rate of movement past the intake structure. Abundance and movement of fish, in turn, are often determined by changes, either gradual or abrupt, in water temperature.

One of the most dramatic consequences of changing water temperature can occur in the spring as inshore water begins to warm, usually at a faster rate than offshore water. Fish, especially alewives, tend to concentrate inshore, seeking the warmest water available. Where warm inshore water $>4^{\circ}\text{C}$ and cold offshore water $<4^{\circ}\text{C}$ meet, they mix until they reach 4°C (temperature of maximum density), then sink. This boundary zone of mixing and sinking water is called a "thermal bar." Inshore water is usually stratified, and offshore water homogeneous. Color differences (increased turbidity) and increased phytoplankton density separate inshore and offshore zones (Huang 1969). The thermal bar which usually forms in April and disperses by May, may persist 4-6 weeks. The thermal bar may be as close as 2 km to shore, and very large concentrations of fish may be "trapped" within a very narrow band of warm, nutrient-rich water. If the plant is operating at full capacity during this time, extremely large numbers of fish may be impinged (Fig. 17).

Though several species may be affected, alewives seem the most vulnerable, and comprise about 80% of April impingement losses during years when a persistent thermal bar forms relatively close to shore. Water temperature profiles recorded at the Cook Plant indicate this condition existed in 1975 (Evans et al. 1978) and 1980, and possibly 1982. During years when the thermal bar was less persistent or formed farther from shore, alewife impingement losses were less in April and usually peaked in late May or June, when alewives moved inshore to spawn.

Number of fish impinged during summer months may be strongly influenced by upwellings which occur when a strong offshore wind blows warm surface water offshore, allowing upwelling of cold, hypolimnetic water into the inshore zone. Upwelling increases fish activity as species seek their preferred water temperature (Wells 1968, Emery 1970, Jude et al. 1979). Rainbow smelt and bloaters, which prefer cold water (Wells 1968; Jude et al. 1979), may accompany cold, upwelled water inshore. Exceptionally high impingement losses during July 1978 coincided with three periods of strong upwelling (Thurber and Jude 1985). In contrast during summer 1982, a low frequency of upwelling (Table 7) may partially explain low impingement losses.

Storms may also increase impingement of fish, though they are not necessarily accompanied by changing water temperature. Storms increase fish movement which seems to increase the vulnerability of fish to entrapment. Water turbidity and wave action may disorient fish making them less able to avoid entering the intake struc-

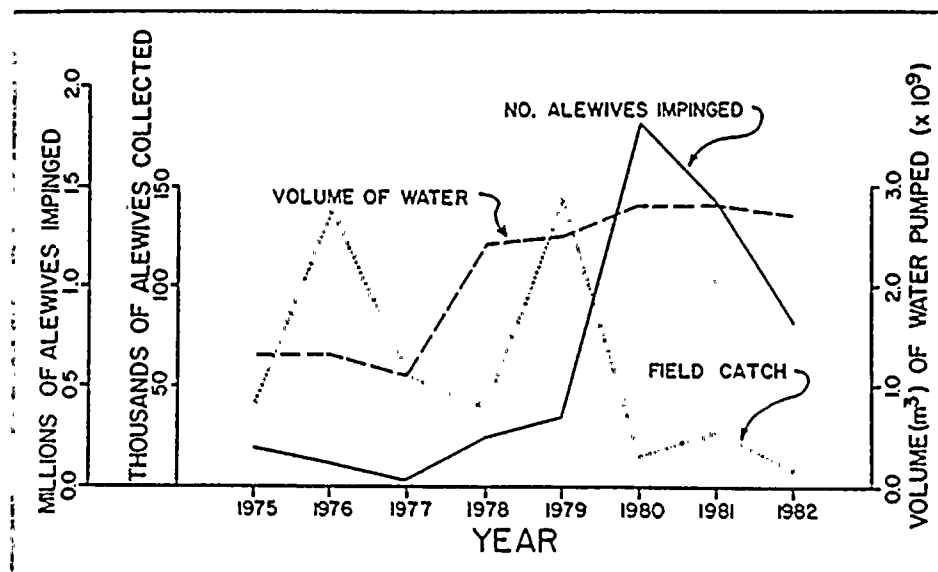


FIGURE 17. A plot of annual impingement losses of alewife, volume of water pumped by the plant, and field abundance of alewife at the D. C. Cook Nuclear Plant, 1975-1982. Unit 1 went on line in 1975, Unit 2 in 1978.

TABLE 7. Lake Michigan water temperatures ($^{\circ}\text{C}$) measured at the St. Joseph Municipal Water Plant, 16 km north of the Cook Plant, 1973-1982; intake depth—6 m. Data are monthly means of the daily average maximum and minimum temperatures.

Year	Month											
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1973	0.5	0.6	4.2	7.1	10.6	17.9	18.8	19.7	16.4	15.7	9.7	4.1
1974	1.2	1.1	3.7	7.5	11.3	14.9	17.2	16.5	16.2	13.3	9.2	3.0
1975	1.3	1.1	2.1	5.4	10.9	16.2	19.5	15.5	17.3	14.5	10.9	4.1
1976	1.1	2.1	5.4	9.6	11.0	16.7	19.2	20.5	18.0	14.6	8.3	2.0
1977	1.1	1.1	3.5	8.7	12.4	14.7	18.6	18.5	15.9	12.1	8.5	2.5
1978	1.7	1.4	1.5	5.6	10.2	13.7	14.1	17.9	18.4	14.0	10.1	3.1
1979	1.3	1.8	2.6	6.6	10.6	14.1	18.0	19.5	17.3	14.5	10.3	5.4
1980	3.0	2.4	2.4	5.9	9.8	14.3	16.7	16.4	15.2	12.4	7.2	2.6
1981	0.9	1.2	3.2	8.2	9.8	15.4	14.8	19.2	18.2	12.0	8.5	3.6
1982	1.2	0.7	1.8	6.3	10.7	13.7	18.5	20.1	18.5	14.4	8.9	5.8
1973-1982	1.3	1.4	3.0	7.1	10.7	15.1	17.6	18.4	17.2	13.8	9.2	3.6

ture, and fish have been observed to seek shelter from storms in the lee of an intake structure (Lifton and Storr 1978). Dorr and Jude (1986), reporting on their diving observations, often saw fish swimming in and out of the intake grills during their fair weather dives. Velocity at the intake grills is about 0.4 m/s which would allow any healthy juvenile or adult fish to resist this relatively weak current. However, once fish get deeper into the intake, the pipe constricts, velocity increases dramatically (up to 1.8 m/s—AEC 1973) and all fish, with the exception of a few robust salmonids, are essentially trapped within the Cook Plant intake system. Additional observations made at the Cook Plant suggested that increased impingement often followed storms, sometimes with a 2- to 4-day lag. In 1980, a storm on 14–15 April preceded the large impingement catch beginning 16 April. Though high impingement of fish persisted for 3 weeks, as many as 90% of the fish impinged during the week following this 3-week period were dead. We suspect they died from the journey into and the stress of overcrowding in the forebay, where they had probably resided for 2 or 3 weeks prior to being impinged.

The Effect of Local Fish Abundance and Plant Operation on Impingement Rates—

In general, species of fish most often impinged (Table 5) were also most abundant in field catches (Table 4). However, spottail shiners comprised a higher percentage of field catches (23%) than of impingement losses (10%). Though spottail shiners were locally abundant and were most abundant inshore (Tesar et al. 1985), they apparently were not attracted to the intake structure and even may have avoided the area. Mottled and slimy sculpins were impinged in much higher numbers than would be predicted from field catches. Benda and Gulvas (1976) and our own divers (Dorr and Jude 1986) observed that sculpins preferentially inhabit the riprap around intake structures; cryptozoic behavior and nocturnal activity patterns probably increased their susceptibility to impingement. The inside of the intake structures was probably colonized as attractive habitat, and eventually these fish became impinged.

Impingement at any given time was related to the instantaneous abundance of fish in the vicinity of the intake but did not necessarily reflect seasonal or yearly changes in the inshore abundance of certain species, particularly alewives. Annual impingement losses (Table 5) among other abundant species more closely followed field abundance (Table 4) but often there were 1 or 2 years when the two were not closely related. Annual impingement losses (Table 5) of trout-perch during one-unit operation (1975–1977) and rainbow smelt during two-unit operation (1978–1982) were inversely related to annual field abundance (Table 4). Differences may have been partially due to limitations of field sampling, because each gear type only sampled one 24-hour period each month, but this explanation alone is inadequate. Young-of-the-year fish were often more abundant in field catches than they were in impingement samples because they were most likely to be inshore of the intakes and because they were generally too small to be impinged.

Sharp contrasts between impinged losses and field catches occurred during 1980–1982, particularly for alewives. This species declined dramatically in field abundance during this time (Table 4; Tesar and Jude 1985, Jude and Tesar 1985) but continued to be impinged in extremely large numbers (Table 5). Several other species (yellow perch, slimy sculpin, and rainbow smelt) were more abundant during these years in both field catches and impingement losses. Bloaters were relatively abundant in field catches during 1978–1982, following a ban on commercial gillnetting of

this species in 1976. Impingement of bloaters was exceptionally high from 1978 to 1981, but both impingement and field abundance declined in 1982 (Tables 5, 4). Field abundance of all species is discussed in detail in Tesar et al. (1985).

Two-unit operation increased cooling water flow rate by more than 200% over one-unit operation (Table 8); fish impingement also increased substantially. When examining impingement rates in terms of one- and two-unit operation, 1977 was dropped from the comparison because so few fish were impinged compared with the first 2 years (1975-1976) of one-unit operation. This exclusion should contribute to a more conservative estimate of the differences between one-unit and two-unit operation. Data from 1978 and 1979 were compared separately from data collected from 1980 to 1982 because Unit 2 pumping during the initial 2 years was erratic, and the full effect of sustained two-unit pumping was not observed until 1980.

The largest percent increase in impingement losses for spottail shiner, trout-perch, and rainbow smelt occurred in 1978-1979 when compared with losses in 1975-1976 (Table 8). The plant pumping schedule was erratic and seldom involved more than three of the seven pumps for any extended period of time during spring and early summer 1978-1979. However, volume of water pumped during July through December 1978-1979 was high (Table 6) and impingement losses among species abundant inshore during these months were substantial. Both units were operating at nearly full capacity during most of spring and early summer 1980-1982, with correspondingly high impingement losses among species normally abundant during these months, particularly alewife and yellow perch. When mean values from 1975 to 1976 were compared with values from 1980 to 1982, there was a 114% increase in volume of water pumped and increased impingement losses among major species ranging from 342% to 2,721% (Table 8). Trout-perch were an exception; impingement losses declined during 1980-1982 (Table 5) as did field catches (Table 4; Tesar et al. 1985, Tesar and Jude 1985, Jude and Tesar 1985).

In summary, the variability among years in impingement losses was a function of abundance of various age-groups of fish in the lake and their behavior during major

TABLE 8. A comparison (in %) between two periods of the change in volume of water pumped and the change in the number of fish impinged annually at the D. C. Cook Nuclear Plant. Comparisons are among years of one-unit operation (1975-1976), years when two units operated sporadically (1978-1979), and years of full two-unit operation (1980-1982). The year 1977 was not included because volume pumped was exceptionally low and not considered characteristic of full one-unit operation. See Thurber and Jude (1985) for full explanation.

Parameter Compared	Years Compared		
	1975-1976 to 1978-1979	1978-1979 to 1980-1982	1975-1976 to 1980-1982
Volume of water pumped	87	15	114
Major fish species impinged			
Alewife	97	374	832
Spottail shiner	577	-35	342
Yellow perch	114	463	1,103
Trout-perch	303	-64	12
Rainbow smelt	1,226	112	2,721

physical events (thermal bar, storms, strong current). A second important factor was pumping rate, and whether maximum pumping rates occurred during certain key months when fish were particularly susceptible to impingement. In addition, during the early 1980s alewives declined dramatically in Lake Michigan, with several species (yellow perch, rainbow smelt, bloaters) increasing in response to the decline (Jude and Tesar 1985). However, alewives still comprised a high percentage of the losses in 1980-1982, which is partially related to high pumping rates in the spring and persistence of the thermal bar. More recent data from 1983-1986 which were collected by power plant personnel suggest that alewives now comprise a substantially lower percentage of the loss. As yellow perch, rainbow smelt, and bloaters have increased in abundance in Lake Michigan, they have also been impinged in higher numbers. Biology and behavior were important determinants of the susceptibility of individual species to impingement. Seasonal and diel movements because of spawning and feeding, and in response to temperature, brought different species at different times into the influence of the intakes. Any factors which decreased the avoidance capabilities of fish, such as darkness, storms, or high currents, caused increased impingement of species in the vicinity of the intakes. Alewives were particularly susceptible as they moved shoreward in spring and concentrated within the thermal bar seeking warm water. Persistence of the bar led to impingement of large numbers of alewives and other species each spring. The preference of spottail shiners for nearshore areas reduced their impingement susceptibility, while other species, such as sculpins and johnny darters, preferred the riprap around the intakes and were impinged in large numbers as a consequence. Upwellings resulted in the movement nearshore of cold water species such as lake trout, bloaters, and rainbow smelt. Their impingement losses during summer appeared to be directly related to upwelling frequency.

Impingement Impact and Mitigation —

Data from 1980-1982 indicated that full two-unit operation can result in extremely large numbers of fish being impinged in a short period of time. In 1980, over 1 million fish (mostly alewives) were impinged during a 3-week period of two-unit operation in April and May. The Zion Nuclear Power Plant in Illinois (Lake Michigan) also impinged large numbers of alewives during May 1975 (Kitchel 1975). Large impingement losses may impact local abundance of affected species, especially in combination with total impingement losses within the southern basin of Lake Michigan. Based on 1975 data, Jensen et al. (1982) estimated that water withdrawal through all intakes on Lake Michigan reduced alewife biomass by nearly 3%. Many fish impinged at the Cook Plant during peak periods in 1978 through 1982 were young-of-the-year or yearlings which had not spawned. Losses among young fish might be particularly detrimental to populations (Rago 1979).

Calculation of production forgone due to entrainment and impingement of larval and adult fish at the Cook Plant (Rago 1979) illustrated that most production forgone resulted from losses among post-larvae through yearlings. Through 1981, over 80% of the alewife production forgone was attributable to entrainment (Rago 1983); however, impingement of young-of-the-year and yearlings contributed significantly to rainbow smelt losses and dominated yellow perch production forgone (Rago 1979). If alewives continue to decline in abundance and are replaced by other species such as yellow perch (Jude and Tesar 1985), impingement may become increasingly important to total production forgone of yellow perch. There is some evidence that this population trend is occurring, as few alewives but many rainbow

smelt were impinged during 1983 and 1984, and yellow perch were very abundant in gill nets set at Cook Plant stations (E. Mallen, personal communication, D. C. Cook Nuclear Plant, Bridgman, Michigan) and in beach seine catches in September 1984 and July-August 1986. Jensen et al. (1982) felt that fish biomass lost due to impingement on Lake Michigan had more impact on Lake Michigan stocks of abundant species than did entrainment of fish larvae.

Several structural or functional modifications of the intake system might reduce or eliminate impingement and even entrainment of fish. Engineering and economic constraints of most of these alternatives were discussed at length in IMPC (1979). Bimber et al. (1984) discussed the effectiveness of several alternatives in reducing entrainment.

Perhaps the most favored system of those examined was use of fine-mesh, wedge-wire screens. Intakes of this type currently installed at the J. H. Campbell Plant (Lake Michigan, near Grand Haven, Michigan) eliminated impingement but had only a small effect on reducing entrainment (Jude et al. 1982). The suitability of similar technology for a plant with cooling water requirements as large as those of the Cook Plant is still unproven.

A barrier net which surrounds the intake structure and prevents fish entrapment was used with apparent success at Zion Station, Lake Michigan (Kitchel 1975). The 51-mm stretch mesh net was installed during April-November and appeared to have been effective at reducing impingement of adult alewives. A fine-mesh net placed around the intake at Ontario Hydro helped reduce impingement losses 64-84% at their Nanticoke plant on Lake Erie (Foster 1981). The net was not size selective but was species selective, deterring emerald shiner, rock bass, spottail shiner, trout-perch, white bass (*Morone chrysops*), and yellow perch more effectively than alewife and some other species. Barrier nets may be vulnerable to wave action and ice scour, but this would not be a problem if nets were deployed seasonally. Use of a barrier net may be a possibility for the Cook Plant, but a feasibility study would be necessary.

Other possibilities for alleviating impingement losses include operational strategies which would require no structural changes to the intake. The most effective approach would be to curtail volume of water pumped through the plant or schedule refueling of the plant during months of maximum impingement. Over 80% of all fish were impinged from April through July, though most young-of-the-year were impinged from August through October. During two-unit operation (1978-1982), an average 54% of annual impingement losses occurred during a 1-month period. Maximum losses usually occurred in June, while the second-largest losses usually occurred in May. Reduction in the volume of water pumped through the plant during May-July should result in impingement of less biomass, while reductions during September and October should result in fewer young-of-the-year impinged. Reduced cooling water use could be achieved by scheduling refueling and maintenance during these months. Reducing cooling water use without reducing plant operation would result in a Δ exceeding the maximum of 12.1C° for Unit 1 or 9.3C° for Unit 2 specified by Nuclear Regulatory Commission environmental technical specifications. However, analyses of plume effects (Kelso and Minns 1975, IMPC 1977) indicate that from a biological viewpoint, some kind of compromise might be possible regarding this issue.

Reducing flow at night might also reduce impingement. Though we made no diel studies, there is some evidence from other plants that more fish are impinged at night than during the day (Benda and Houtcooper 1976). We have also documented

that significantly more fish larvae are entrained at night than during daylight (Jude et al. 1982, Noguchi et al. 1985).

Production Forgone

Survival rates for pro- and post-larvae derived from our production forgone analyses were far higher than those given in the literature and those recently generated by Mansfield and Jude (1986). These survival rates were used so that logical consistency was maintained and arbitrary decisions were avoided. This resulted in an overestimation of production forgone. The severe upward bias caused by using these survival rates should be recognized. Many assumptions about the state of nature are required for the estimation of production forgone. The most critical and questionable assumptions are that system productivity decreases in response to removal of fish by the power plant and no feedback mechanisms operate (Rago 1984).

Estimated production forgone for alewives ranged from 0.4 kg in 1973 to 327,964 kg in 1976. Production loss estimates for 1974 and 1975 were 322,406 and 186,024 kg, respectively. Over 95% of the production lost was attributable to entrainment of pro- and post-larvae.

Estimates of production forgone for spottail shiner for 1973 through 1976 were 0.77, 165.5, 6,011, and 1,736 kg, respectively. Most production losses were attributable to entrainment of larval fish. From a low of 0.2 kg in 1973, production losses for rainbow smelt peaked in 1974 at 17,358 kg and then decreased to 4,617 kg in 1975 and 78.3 kg in 1976. The main reason for the 1974-1976 decline was the absence of extensive spawning by rainbow smelt in the vicinity of the Cook Plant. Most losses were due to entrainment in 1974-75 and impingement in 1976.

Yellow perch production losses were largely due to impingement of young-of-the-year and yearling fishes. Very few larval yellow perch were found in the vicinity of the Cook Plant. Estimated production losses for years 1973 through 1976 were 18, 64, 1,647, and 1,812 kg, respectively.

Sensitivity analyses were used to identify critical parameters and initial conditions in the production forgone model (Rago 1984). In general, the model was most sensitive to parameters and initial conditions which were the most difficult to estimate and about which we had the least confidence (mortality rates for pro-larvae, post-larvae, and young-of-the-year).

Most responses of estimated production lost were linear with respect to the parameter varied. As a result, empirical sensitivity coefficients, defined as the slope of the estimated production lost with respect to the parameter, were used to characterize the system. The apparent linearity of the production function seemed to hold over a broad range of most parameters examined.

Comparisons of estimated production forgone with respect to commercial catch statistics indicated that these losses constituted a negligible fraction of the annual yield from Lake Michigan for each species. Alewife losses ranged from 1.2 to 1.8% of the annual yield during 1974-1976. Rainbow smelt losses were estimated to be from a negligible fraction up to 2.2% in the same period. Yellow perch losses were negligible compared to the annual commercial yields. Average total value of losses was about \$10,500/year. Due to limited access to the fishery and exogenous factors, the annual yields were not considered to be representative of actual stock biomasses which were unknown.

Several factors should be considered to put these results in perspective. Model results are subject to considerable error due to possible model specification errors.

parameter estimation errors, and natural variability. The richness and complexity of nature result in considerable difficulty in the development of predictive models. A generally accepted paradigm regarding population response to exploitation does not exist, and even processes acting within simple microcosms have been difficult to identify. Fishery population dynamics cannot be dissociated from ecosystem dynamics because population stability may be due to community dynamics rather than single species characteristics.

Ability of fish populations to respond to changes in density is universally accepted, but the mechanisms of these responses are difficult to identify (Goodyear 1977). Many different processes tend to interact to produce compensatory responses in fish populations. While it is dangerous to assume that compensation acts over all ranges of population densities, the overwhelming body of evidence suggests that natural populations can withstand exploitation rates of about 20% (Mansfield et al. 1985). Numerous other factors cause changes in populations of indigenous species and must be considered. These are broadly classified as environmental stressors, natural abiotic factors, species fluxes, and commercial and sport harvests.

Special Studies

Alewife Survival During the First Growth Season—

Early survival of fishes may determine year-class strength and can be used to predict recruitment. The importance of alewife in the Lake Michigan ecosystem and its ability to recover quickly from catastrophic mortalities lend importance to estimating its early survival. An empirical measure of alewife survival would also be useful for production forgone calculations.

We used densities of various sizes of alewife larvae and young-of-the-year alewives collected in trawls to calculate survival during their first season of life (Mansfield and Jude 1986). Because of the widespread inshore, pelagic distribution of alewife larvae, and the offshore movement along the bottom by young-of-the-year during fall (Wells 1968, Brown 1972), data representative of the true abundance of alewife were obtained from plankton net tows, entrainment samples, and bottom trawls at 6- and 9-m stations.

Calculated seasonal survival rates fluctuated among years (Table 9). During 1974-1976 and 1979-1982, using either entrainment or field data, calculated survival from yolk-sac larvae (2-5 mm) to young-of-the-year was always below 2%. During 2 years, 1977 and 1978, fish larvae densities in both field and entrainment collections were low and young-of-the-year densities were high relative to other years, resulting in improbably high survival rates (Table 9). Mean densities of 2-5-mm larvae over all years were similar between field and entrainment samples, 1,720 and 1,810 larvae/1,000 m³, respectively, resulting in similar calculated survival rates over the entire study, about 1% from yolk-sac larvae to young-of-the-year (Table 9).

Highest daily mortality (27%) was found for entrained larvae which passed from the 2- to 5-mm group to the 5.5- to 10-mm group (Table 10). Mortality declined over time; subsequent daily mortalities were 8, 4, 2, and 2% between size groups of 5.5-10, 10.5-15, 15.5-20, and 20.5-25 mm to young-of-the-year.

Yearly mean densities and survival of alewife were not good predictors of year-class strength. Spring trawl catches of yearling alewife bore no relationship (Spearman rank correlations < 0.5) with larvae or young-of-the-year densities the previous year. Yearling alewives tend to inhabit midwater more than young-of-the-year

TABLE 9. Seasonal survival rates of alewife from each size group of larvae to young-of-the-year (YOY). Survival rates are ratios of peak density of YOY alewives taken in trawls to densities of yolk-sac larvae and post-yolk-sac larvae taken from entrainment and field collections each year. ND = no data Values > 1 indicate survival > 100%.

Year	Entrained Larvae (Length Interval - mm)						Field Larvae (Length Interval - mm)	
	2-5	5.5-10	10.5-15	15.5-20	20.5-25	5.5-25	2-5	5.5-25
1974	ND	ND	ND	ND	ND	ND	0.002	0.005
1975	0.003	0.186	0.775	0.358	> 1	0.097	0.005	0.018
1976	0.008	0.055	0.068	0.149	0.347	0.024	0.013	0.008
1977	0.023	0.103	0.247	0.541	> 1	0.063	0.096	0.059
1978	0.137	> 1	> 1	> 1	0.709	0.320	0.250	0.806
1979	0.009	0.031	0.079	0.079	0.076	0.014	0.004	0.040
1980	0.001	0.063	0.083	0.081	0.176	0.022	0.006	0.004
1981	0.001	0.019	0.045	0.038	0.026	0.007	0.004	0.008
1982	0.010	0.057	0.167	0.368	0.440	0.035	0.011	0.016
Means (from density means)	0.010	0.098	0.214	0.285	0.319	0.046	0.011	0.022

TABLE 10. Percent daily mortality calculated from mean alewife densities over all year, 1975-1982 (entrained larvae) and 1974-1982 (field-caught larvae). Design I used time intervals corresponding to peak occurrence of fish larvae in each length interval in our samples. Design II time intervals were calculated from laboratory-derived alewife growth rates (Heinrich 1981). YOY = young-of-the-year. NC = not calculated.

Length Interval (mm)	Entrained Larvae		Field-caught Larvae	
	Design I	Design II	Design I	Design II
2-5 to 5.5-10	11.7	27.3	NC	NC
5.5-10 to 10.5-15	19.5	8.3	NC	NC
10.5-15 to 15.5-20	3.5	4.0	NC	NC
15.5-20 to 20.5-25	1.2	1.7	NC	NC
25.5-25 to YOY	2.5	2.1	NC	NC
2-5 to 5.5-25	5.7	12.7	3.1	6.4
5.5-25 to YOY	5.1	4.1	5.7	4.8
2-5 to YOY	5.3	5.3	5.0	5.0

(Brown 1972), making bottom trawls unreliable estimators of yearling abundance and probably partly accounting for lack of correlation.

Variation in yearly survival estimates may be due to either true survival differences or incorrect assumptions. Alewife survival is certainly not the same each year, as temperature, food availability, abundance of predators, and other conditions fluctuate irregularly and affect alewife survival. The high survivals calculated for 1977 and 1978 may be attributed to the sampling schedule missing times of peak larval fish abundance, or spawning and hatching taking place outside the study area and juveniles moving into the area later (young-of-the-year collected by trawl).

During 1979, a year of frequent and prolonged upwelling of cold water in eastern Lake Michigan, relatively few alewife larvae > 5 mm were collected from 1- to 15-m depths, compared to fish larvae sampled during years when upwelling seldom occurred (Heufelder et al. 1982). Upwelling may result in increased mortality to larvae or may displace larvae from the inshore zone (Heufelder et al. 1982). Thus differences in calculated survivals of post-yolk-sac larvae from year to year may be due to actual mortality differences or to changes in distribution of older larvae from water mass movement. Yolk-sac larvae densities also may be depressed by direct mortality from upwellings or by transport from the area through water mass movements. However, occurrence of newly hatched larvae was prolonged into September during 1979, a year of frequent and intense upwellings (Heufelder et al. 1982), possibly compensating for decreased densities (greater mortality or transport) by increased duration of occurrence (longer spawning season). Thus, yolk-sac larvae to young-of-the-year survival rates are apt to be less variable than post-yolk-sac larvae to young-of-the-year survival rates.

The "critical period" hypothesis, that the transition from yolk-sac larva to exogenous feeding is a time of high mortality (Hjort 1914, May 1974), is supported by our data. Survival from yolk-sac larva to post-yolk-sac larva was much lower than post-yolk-sac to young-of-the-year. Survival curves for various herring differ regarding

the presence and timing of a critical period (Dahlberg 1979). The daily mortality rates we calculated for alewife were similar to those of some other Clupeidae. Our daily mortality rates (Table 10) were similar to those found by Crecco et al. (1983) for American shad (*Alosa sapidissima*): 19.8–25.6% mortality per day for first-feeding American shad larvae, 4.3–8.7% for larvae approaching metamorphosis, and 1.8–2.0% for juveniles. Other species' daily mortalities were often 20–30% per day during early stages or just after yolk absorption (Stevenson 1962; Houde 1977a, 1977b, 1977c; Schnack 1981). These mortality rates provide valid data for use in evaluating the production forgone calculations which used estimates for mortality since none were available. As indicated above, they agreed well with mortality estimates for other Clupeidae.

Comparative Efficiency of Nets and Pumps —

Pumps and nets often differ in efficiency inside power plants (Leithiser et al. 1979, King et al. 1980, Cada and Loar 1982). Our estimates of relative abundance of field-caught and entrained larvae would be in error if efficiencies of the pump used to sample entrained larvae and the net used to catch field larvae were dissimilar.

To investigate possible differences, we fished nets and pumps simultaneously in the Cook Plant forebay over a 24-hour period on 16–17 June 1982. Two nets, identical to those used in field and entrainment sampling, were mounted in a frame and suspended at 5-m depth alongside hoses used in entrainment sampling. To approximate the continuous sampling performed with pumps, nets were fished in a running series of 10-minute samples which continued throughout a 24-hour entrainment sampling interval. For analysis, we combined the 10-minute samples taken in a single time period (sunrise, day, sunset, or midnight) and computed mean density (total number of larvae divided by total volume of water filtered), for each replicate in each period. Net samples captured 214 alewife larvae and filtered 427 m³ of water, while pump samples took 339 alewife larvae and filtered 1,041 m³ in the 24-hour period.

Mean density (no./1,000 m³) was greater in net samples in all four divisions, with 484 vs. 204 in sunrise-noon samples; 228 vs. 168, noon-sunset; 754 vs. 533, sunset-midnight; and 691 vs. 545, midnight-dawn. However, density differences between gear were not significant in either day or night samples (Mann-Whitney tests, two-tailed, $P > 0.10$). Although efficiency of the two devices did not differ significantly, the result suggested that nets were about 50% more efficient.

In another study conducted at the Monroe fossil fuel power plant on Lake Erie (Jude et al. 1986), larval fish were found to be uniformly distributed in the vertical water column in the discharge canal of the plant. Water here was well mixed and flowing at a high velocity. Preliminary studies we did at the Cook Plant (Bimber et al. 1984, Noguchi et al. 1985) suggested a similar pattern existed in the Cook Plant forebay.

As a consequence of these studies, our entrainment losses at the Cook Plant may be a minimal estimate of the number of fish larvae passing through the plant. Higher production forgone estimates would also occur because of our underestimating larval fish passage through the plant. These studies also suggest that plankton nets may be a better entrainment sampling device than the low-volume diaphragm pumps we used.

CONCLUSIONS AND RECOMMENDATIONS FOR DESIGN OF IMPACT STUDIES

Since the passage of the National Environmental Policy Act in 1969 and the Federal Water Pollution Control Act Amendments in 1972, hundreds of studies have been undertaken to assess the impact of power plant operation on aquatic ecosystems. Many of the early assessment programs were harshly criticized (Schindler 1976) as they exhibited a variety of flaws generally brought about by unrealistic time constraints, scarcity of background information, lack of experience with large, highly variable systems, and often a lack of experience with large, highly variable systems, and often a lack of understanding of the essential problems to be addressed. The science of environmental impact assessment has become much more sophisticated over the last decade. Researchers have gained valuable experience and a great deal of information has been gathered so that today a large body of literature exists, which was not available in the early 1970s, to aid in the design of impact assessment study programs.

In recent years, a number of papers have been published outlining the strengths and shortcomings of the environmental impact assessment process in its present state (Rosenberg et al. 1981) and we will not attempt a thorough analysis here. However, several recommendations for the design of future studies, born of our own experience as well as the experiences of others, are presented below.

Perhaps the most critical phase of any impact assessment program is the background preparation, the work that is done before any samples are taken or monitoring is begun. This is also the phase that has most often been neglected or misdirected.

First, and foremost, objectives must be clearly stated and well understood. It is essential to define the problem if conclusions that are meaningful and helpful to decision makers and planners are to be obtained from the research program. The next step should be a case history and literature review. The intent is to develop a preliminary description of the ecosystem and its components, learn about the physical and operating peculiarities of the plant, and identify possible impacts.

Information obtained from the case history and literature review should be used to formulate a qualitative or conceptual model of the system. Qualitative modeling can be an important tool in environmental impact assessment because it forces a precise description of the system (Ward 1978) and therefore is useful in the generation of hypotheses (Rago et al. 1983). The importance of defining testable hypotheses can not be overemphasized. Failure to apply the scientific method to the design of study plans has been a major flaw in impact assessment programs in the past. The result has been the formulation of dubious conclusions and a loss of credibility for the entire field of environmental impact assessment (Schindler 1976, Rosenberg et al. 1981). Monitoring programs must be designed to test for impacts, not merely to describe the ecosystem in question (Van Winkle 1977).

Once hypotheses have been generated, data needed to test the hypotheses can be identified and a pilot study can be designed and carried out. Information obtained from a pilot study can be invaluable when developing the design of the actual monitoring program. The purpose of a pilot study is to increase understanding of the ecosystem and to examine proposed methods for gathering and analyzing the required data. Biota should be identified and diel, seasonal, and spatial distribution and abundance patterns should be determined. This will allow selection of target species for further study, help define geographical limits of the study site, and aid in

determining when and where to sample in order to obtain the data needed in the most efficient way possible. Instead of trying to study all species during the entire year, it may be more prudent to select several key species (Ward 1978, Gore et al. 1979) and concentrate sampling effort during periods when returns can be maximized (Thomas 1977, Gore et al. 1979). Collecting gear should be evaluated during the pilot study and estimates of the natural variability in the system obtained. It is also important during this phase to ensure that the final field design will be consistent with the assumptions of the statistical analyses selected (see Thomas et al. 1978).

Information and new insights gleaned from the pilot study should be used to reevaluate the model. Rago et al. (1983) stated that, "The iterative refinement of the conceptual model is the focal point of the environmental assessment and monitoring program." At this point a sampling program can be designed that will effectively test hypotheses. It is assumed that the sampling scheme and techniques used in the monitoring program will not simply represent an extension or continuation of the pilot study. The pilot study should be a learning experience which can be used to arrive at a final, better product.

Finally, it is important that once the monitoring program is initiated, data are analyzed promptly so that feedback can take place. This will allow evaluation of the quality of the results, i.e., to determine whether the information needed to test the hypotheses and meet objectives is being obtained. Feedback may suggest additional data needs, or improvements or modifications in study design or models (Thomas 1977, Rago et al. 1983). Once a sampling program has begun however, modification of study design should be done with care, allowing only those changes that can be expected to improve the quality of the data set (Gore et al. 1979, Rago et al. 1983). No change should be implemented until an interrelationship between the old and new methods can be established (Gore et al. 1979).

Although our execution may not yet be perfect, monitoring programs continue to provide the experience and data necessary to advance the science of environmental impact assessment (Rosenberg et al. 1981). Most past failures have been attributed to poor research design. This is not something that environmental scientists are unaware of and we, as well as others, have made suggestions on how to improve impact assessment study designs.

We have not discussed the optimal time span for an impact assessment study, largely because it can vary greatly depending on where the plant is located, what objectives are, etc. However, some guidelines should be mentioned. Rago et al. (1983) have suggested that the pilot study may require at least 1 year and most researchers agree that a minimum of 2 years of preoperational data are needed. Many studies have been criticized in the past for failure to continue monitoring for a long enough period of time after the plant became operational to be able to assess possible long-term impacts. A long-term data set can be very valuable, particularly for a highly variable system. However, we must guard against collecting data for the sake of collecting data and perhaps utilize a "stopping rule" as proposed by Thomas (1977) so that a study can be terminated if an impact cannot be detected within an agreed upon number of years.

Specific changes in the Cook Plant fish monitoring design we would recommend based on our experience, would be to increase field sampling to at least twice per month during the major spawning period, May-August. Once-per-month sampling sometimes missed peak spawning and led to poor correlations between larval fish densities and strength of year class as provided by fall trawl and seine index catches. There should also be a series of depth contours sampled at the reference station (e.g.

1, 3, 6, 9, 12, and 15 m) to document spatial distributions and relate these to the specific location of the intake. The impact transect should only have stations at depths where impact is expected, e.g., at the Cook Plant at the 1-, 6-, and 9-m depth contours. Data from these specific stations can then be used for statistical testing.

To evaluate the selectivity of the intake and aid in future designs, we compared the species, densities, and lengths of fish larvae collected in the field and those entrained. These comparisons were somewhat confounded by samples being collected on different days. Therefore we recommend that entrainment and field larval fish surveys be conducted on the same day to ensure sound conclusions are made.

Studies of the vertical and horizontal distributions of larval fish and eggs in power plant forebays and discharge canals should be made to ensure the sampling site picked is representative of the ichthyoplankton passing through the plant. A discussion of horizontal and vertical distribution of larval fish in power plant forebays is provided by Jude et al. (1986) and Leslie (1983). If different gear are used for each type of sampling, gear comparisons should be done to gain some idea of how efficient and comparable each gear type is and establish the magnitude of the bias, if any.

Removing and counting fish eggs in entrainment samples was costly and time consuming, considering the value of the data that emerged. We were only able to identify eggs based on size and season of capture. Rainbow smelt eggs had an adhesive stalk which made them unique. We were able to obtain estimates of fish egg abundance in the water column, on the bottom, and for those that passed through the plant. Egg abundance data from the field assisted in our evaluation of the magnitude, timing, and depths at which spawning was occurring. We felt that most entrained fish eggs that were entrained would survive plant passage unharmed, based on Schubel (1975) who exposed marine alewife, blueback herring, and striped bass eggs to a wide range of different temperature increases and times typical of most power plants. He found no significant differences in hatching success of controls or treated eggs at all development stages. Therefore, enumerating the abundance of entrained fish eggs quantitatively is unnecessary. We recommend, therefore, that future studies in the field note qualitatively the abundance of fish eggs to aid in determining spawning times and success. Entrained eggs should also be recorded qualitatively, unless the eggs of some important species are entrained, in which case quantitative enumeration is recommended.

We found that several of the most abundant species were those that were most impacted by the plant. Along with sport and commercial species, these targeted species should be selected and gear and fishing effort directed at them to maximize the information gained. Replication of sampling, including gill nets, is necessary to ensure robust analysis of the data.

Another important point is preoperational sampling — *before* any construction or other potentially disruptive activities occur in the area. We were unable to establish an unbiased preoperational distribution for certain species which appeared to shun the plant site. Sampling before any in-lake activity would eliminate this problem and give a reliable reference period.

Another problem was the use of static, passive gear like gill nets, which can introduce considerable bias into the data, depending on weather, water temperature, size, species-specific spawning behavior, and time of the year. Some of this variability evens out over replicates and years. We have had innumerable cases where an upwelling would affect only the reference station gill net or gill nets, while nets at the Cook Plant fished in warm epilimnetic water. Catches between sites were dramati-

cally different due to temperature differences alone. These incidences do lead to insight into fish behavior but raise havoc with statistical treatment of the data. Use of active gear, such as trawls and seines deployed only during consistent water conditions, is one way to cut down on this source of variability.

We used three gear types: seines, trawls, and gill nets. They gave an excellent balanced estimate of all species and sizes of fish which frequented the Cook Plant area. Seines were large (38 m long with a 1.8-m-long bag) so they collected representative samples of especially young of the year, but they also collected larger species and sizes of fish, giving us insight into which species used the nearshore zone for spawning, feeding, or migrations. Gill nets provided needed information on larger adults of many species. Trawls provided data on the more abundant forage species, such as alewife, rainbow smelt, troutperch, and bloaters. These estimates were instantaneous and provided the best data among the three gear types for statistical treatment. Trawls and seines were deployed twice at each station for replication while only one gill net was set. In our studies at the Campbell Plant, we corrected this deficiency by keeping the same length of net for each mesh size, but cutting them in half and setting two gill nets side by side. This provided a measure of replication for this gear type which was lacking in Cook Plant studies.

Another problem was that abundance of some species varied monthly. To avoid the statistical problems with zero or low abundance data during certain months, these months were deleted from the data set. Some preliminary examination of the literature and pilot studies would eliminate the need to sample during months of low abundance for target species.

We often noticed large accumulations of fish in and around the riprap area and plume. We were able to document relative abundance and the species in the vicinity of the riprap with our diving programs, but the plume area was not surveyed. Future studies should consider adding some type of acoustic sampling in the vicinity of the plume to obtain some relative estimates of density for comparisons among months and years. Field sampling with fishing gear should provide a good indication of what species was being attracted to the plume.

Lastly, some tracking of lakewide population changes should be included in the design. We found that our sampling adequately reflected these changes (Jude and Tesar 1985). However, massive, generally weather-related phenomena usually had more impact on local population movements, catches, and entrainment and impingement rates than any effects which could be attributed directly to the plant.

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ECOLOGY OF FISH IN NEARSHORE SOUTHEASTERN LAKE MICHIGAN, 1973-1982

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INTRODUCTION

An understanding of the history of the fish populations in Lake Michigan is necessary to put into perspective our fishery studies at the D. C. Cook Nuclear Plant. The Lake Michigan fish population has changed dramatically since the early 1900s, when lake trout and burbot were the only top predators and there was a seven-species complex of deepwater ciscoes in the lake (Christie 1974, Smith 1968). A long series of man-induced changes, including eutrophication, sea lamprey entry, and overfishing, has wreaked havoc with this original fish community. Overfishing of lake trout and the premium whitefish species diminished their resilience and ability to respond to environmental stress or invading species (Walters et al. 1980). The sea lamprey, given access to the upper Great Lakes with the building of the Welland and Erie barge canals, was one of the first, and probably the most pernicious exotic fish to enter Lake Michigan. Lampreys systematically fed on the largest species of fish available; lake trout were extirpated first, while burbot, lake whitefish, and many other species such as white sucker, were severely depleted. Deepwater ciscoes were decimated to the point that only one species, the bloater *Coregonus hoyi*, remained from the original seven species (Smith 1968). As a result, a severely disturbed fish community, essentially devoid of predators, was present, and many of the planktivores and benthivores were severely reduced in numbers or became extinct.

The alewife was first recorded in Lake Michigan in 1949 (Miller 1957) and, in the absence of predators or any substantial competitors, proliferated in the Lake. Rainbow smelt gained entry by a different mechanism; they were stocked in Crystal Lake in 1912 and subsequently migrated into Lake Michigan (Van Oosten 1937). As populations expanded unchecked, both species experienced massive die-offs, the rainbow smelt in 1943 (Van Oosten 1947) and the alewife in 1966 (Brown 1972). In response to the massive die-offs of alewives which accumulated on the beaches, and with an intent to revitalize the sport fishery, the Michigan Department of Natural Resources stocked five species of salmonids into the lake beginning in 1966. A spectacular sport fishery has subsequently developed.

With the success of salmonid stocking by the Michigan Department of Natural Resources, other states have initiated similar stocking programs in Lake Michigan, to the point that Stewart et al. (1981) cautioned against excessive stocking of predators. Through the 1970s, the period of our Cook Plant studies, salmonid stocking rates continued to increase. Starting in 1979, and continuing into the 1980s, signs of the effect of this increased predation were manifested in the decline of alewife populations (Jude and Tesar 1985). Concomitant with the alewife decline was a dramatic increase in yellow perch populations. Rainbow smelt, bloater, and deepwater sculpin populations also increased (Jude and Tesar 1985).

As the alewife population declined, species such as the yellow perch and emerald shiner, whose larvae are hypothesized to be consumed by alewife (Kohler and Ney 1980, Crowder 1980), were expected to increase. In 1986, we began to collect large numbers of emerald shiners in our seine hauls in eastern Lake Michigan (unpublished data, Great Lakes Research Division).

Our studies at the D. C. Cook Nuclear Plant were conducted from 1973 to 1982, during the period of relatively stable alewife populations, with a dramatic decline beginning in the early 1980s. The Lake Michigan fish community is still comprised of mostly marine exotic species, the most prominent of which are the sea lamprey, alewife, rainbow smelt, and several Pacific salmon. Because the predators are not linked functionally with their prey and the forage species are marine exotics, not well adapted to fresh water, we can expect fluctuations in their populations to continue.

Our studies of fish populations in the vicinity of the Cook Plant were conducted using three gear types for adult and juvenile fish: seines, trawls, and gillnets. Plankton nets were used for collection of larval fish. Seines produced 68% of the just over 1.1 million fish collected during 1973-1982, followed by trawls (24%), and gill nets (8%). We focused on the nearshore zone (<9 m deep) where the intake and discharge structures are located. The nearshore zone, especially the beach zone, has been generally ignored by most Great Lakes researchers. Our objective was to gather data on the abundance, distribution, and spawning migrations of fish in the vicinity of the plant. We use common names of fish throughout the report as established by Robins et al. (1980). Fish catch data were used to evaluate plant impact by comparing catch indices between the plant and a reference station at Warren Dunes.

We also used our field abundance and distribution data to assist in interpretation of entrainment and impingement results (see Plant Impact on Fish chapter). Field catches of larval, young-of-the-year, and adult fish as well as entrainment and impingement data were used to describe the amount, location, and importance of spawning in the nearshore zone. We were also able to evaluate the importance of that area as a nursery zone for juveniles and describe the seasonal and diel migrations for the common species inhabiting this area. These data and interpretations were integrated with our knowledge of the distribution and behavior of these species from studies done in offshore waters to provide a coherent picture of the distribution, population dynamics, and biology of the common species inhabiting the nearshore zone.

METHODS

FISH LARVAE

Field Collections

Sampling—

Fish larvae were collected with a conical, 0.5-m diameter, nylon plankton net with 363- μ m mesh. A Rigosha flowmeter attached to the center opening of the net measured volume of water sampled. Flowmeter readings were converted to volume filtered and total numbers of larvae and eggs captured in all subsurface tows were adjusted to compensate for upper strata contamination (Jude et al. 1979). Numbers

of eggs and larvae were converted to densities, i.e., number/1,000 m³, for all analyses. About 35 m³ of water were filtered in most tows.

Duplicate surface tow samples were collected at beach seining stations A (north Cook), B (south Cook), and F (Warren Dunes) (Fig. 1). Nets were towed by hand, just below the water surface, against the current for a distance of about 61 m. Beach tows were performed during both day and night, once a month, April through November.

At stations where depth was 6 m, we performed horizontal, 5-minute tows from the R/V Mysis at speeds of 3–6 km/h at discrete depth strata parallel to shore along three transects in Lake Michigan. The transects were at Warren Dunes, including stations F (1 m, i.e., beach), G (6 m), H (9 m), and W (21 m); south Cook, with stations B (1 m), C (6 m), D (9 m), and E (21 m); and north Cook, with stations A (1 m) and R (6 m) (Fig. 1). Open water tows were performed both day and night, once per month, April through September. For 6-m stations a tow was done at 0.5-, 2-, 4-, and 5.5-m strata; for 9-m stations depths were 0.5, 2.5, 4.5, 6.5, and 8.5 m; and for 21-m stations tows were done at 0.5, 7.5, 13.5, and 20 m.

Laboratory Procedures for Larval Fish Samples

All entrainment and field samples of fish larvae and eggs were preserved with a 10% formaldehyde solution immediately after collection and then transported to the Great Lakes Research Division's Fishery Laboratory for analysis. For our purposes, fish larvae were defined as any fish 25.4 mm or less in total length (TL). In the laboratory, larvae were sorted, identified, counted, and measured. Larvae were identified to species, when possible, otherwise to the lowest taxonomic group (see Table 1). Alewife, spottail shiner, and rainbow smelt were measured to the nearest 0.5 mm TL, while all others were measured to the nearest 0.1 mm. Fish eggs were counted but not identified to species. When large quantities were found, egg numbers were estimated by volumetric subsampling (Jude et al. 1975). All larvae and a subsample of eggs from each entrainment sample were catalogued and saved for future reference. Data were recorded directly on standard coding forms, key-punched, and transferred to computer tapes for analysis.

Larval fish identification was based on knowledge of species abundance and spawning times in southeastern Lake Michigan, comparison of specimens with those in the Great Lakes Regional Fish Larvae Collection (Dorr and Jude 1981), and reference to taxonomic works (Lippson and Moran 1974, Nelson and Cole 1975, Dorr et al. 1976, Hogue et al. 1976, Jude et al. 1979, Auer 1982).

ADULT FISH

Field Collections

Sampling—

Details on methods were presented by Jude et al. (1979), Tesar et al. (1985), and Tesar and Jude (1985) and only a brief description is given here. Seven sampling stations (A, B, C, D, F, G, and H) were established in southeastern Lake Michigan off the Cook Plant (experimental area) and off Warren Dunes State Park (reference area) where adult and juvenile fish were sampled (Fig. 1). Fish at 1-m deep beach

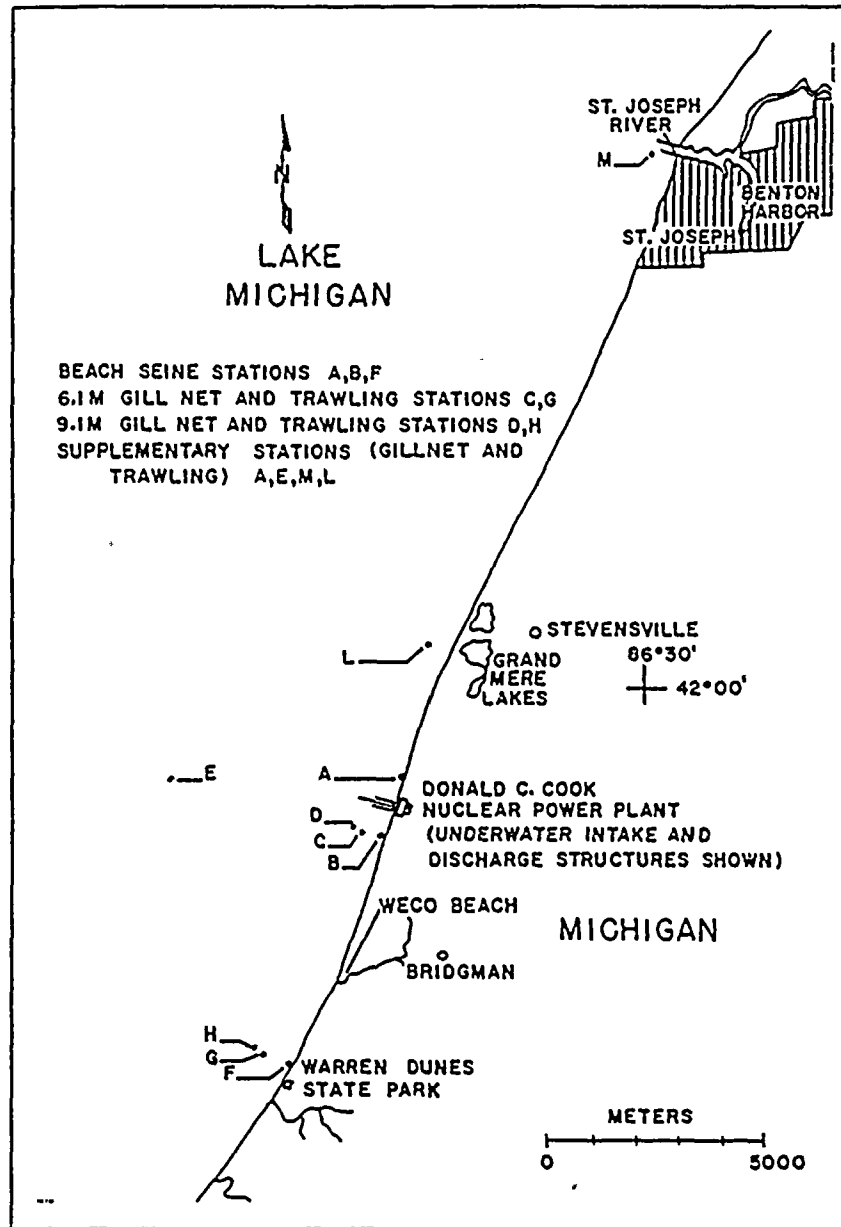


FIGURE 1. Map of southeastern Lake Michigan, showing locations of the D. C. Cook Nuclear Plant and our field fish sampling stations.

TABLE 1. A list of larval fish collected in the vicinity of the D. C. Cook Nuclear Plant from 1973 to 1982.

Common Name or Category	Common Name or Category
Alewife	Unidentified sculpins
Spottail shiner	Unidentified minnows
Rainbow smelt	Unidentified coregonids
Yellow perch	Unidentified darters
Trout-perch	Unidentified suckers
Johnny darter	Unidentified clupeids
Slimy sculpin	Unidentified fish larvae as a
Common carp	result of a poor condition
Ninespine stickleback	Unidentified fish larvae
Mottled sculpin	
Deepwater sculpin	Fish eggs
Burbot	
Quillback	
Emerald shiner	
Gizzard shad	

stations A, B, and F were sampled with a 38-m \times 1.8-m bag seine of 0.64-cm square mesh. Fish at 6-m-deep stations C and G and 9-m-deep stations D and H were sampled with a 4.9-m bottom trawl of 3.8-cm square mesh and 160-m \times 1.8-m bottom gill nets consisting of 12 mesh sizes from 1.3 cm to 10-cm square mesh. Two stations (R and Q, respectively, at 6- and 9-m depths) were also established in 1975, north of the plant's underwater structures, and fish were sampled by trawl at station R and by gill net at both stations. Fish were sampled during the day and at night approximately once per month at each station.

Yearly fishing effort was approximately equal between April and October. Some sampling also occurred in colder months, especially November; however, yearly effort was too sporadic and catches too small in these months to be used in statistical analyses.

Physical and limnological measurements were taken when sampling occurred in 1973-1982 and are presented in Jude et al. (1975), Jude et al. (1979), Tesar et al. (1985), and Tesar and Jude (1985). References to fish are by common name (Table 2). Common and scientific names are according to Robins et al. (1980).

RESULTS

OVERVIEW OF BIOLOGY AND POPULATION CHANGES OF ABUNDANT SPECIES

The nearshore (<9 m deep) zone of Lake Michigan is a dynamic place for many of the species of fish that inhabit the lake. Fish move in and out of this area in response to currents and upwellings, for spawning, to escape predation, and to search for food. Most of the earlier work on fish in Lake Michigan (Wells 1968, Brown 1972, Christie 1974) focused on the deeper offshore areas and little was

TABLE 2. Common and scientific names of fish species collected from D. C. Cook Nuclear Plant study areas, southeastern Lake Michigan, 1973-1982.

Common Name	Scientific Name	Common Name	Scientific Name
Alewife	<i>Alosa pseudoharengus</i>	Lake herring	<i>Coregonus artedii</i>
Banded killifish	<i>Fundulus diaphanus</i>	Lake sturgeon	<i>Acipenser fulvescens</i>
Black bullhead	<i>Ictalurus melas</i>	Lake trout	<i>Salvelinus namaycush</i>
Blackchin shiner	<i>Notropis heterodon</i>	Lake whitefish ¹	<i>Coregonus clupeaformis</i>
Black crappie	<i>Pomoxis nigromaculatus</i>	Largemouth bass	<i>Micropterus salmoides</i>
Blacknose dace	<i>Rhinichthys atratulus</i>	Logperch	<i>Percina caprodes</i>
Blacknose shiner	<i>Notropis heterolepis</i>	Longnose dace	<i>Rhinichthys cataractae</i>
Bloater	<i>Coregonus hoyi</i>	Longnose sucker	<i>Catostomus catostomus</i>
Bluegill	<i>Lepomis macrochirus</i>	Mottled sculpin	<i>Cottus bairdi</i>
Bluntnose minnow	<i>Pimephales notatus</i>	Ninespine stickleback	<i>Pungitius pungitius</i>
Brook silverside	<i>Labidesthes sicculus</i>	Northern pike	<i>Esox pungitius</i>
Brown trout	<i>Salmo trutta</i>	Pumpkinseed	<i>Lepomis gibbosus</i>
Burbot	<i>Lota lota</i>	Quillback	<i>Carpiodes cyprinus</i>
Central Mudminnow	<i>Umbra limi</i>	Rainbow smelt	<i>Osmerus mordax</i>
Channel catfish	<i>Ictalurus punctatus</i>	Rainbow trout	<i>Salmo gairdneri</i>
Chinook salmon	<i>Oncorhynchus tshawytschu</i>	Rock bass	<i>Ambloplites rupestris</i>
Coho salmon	<i>Oncorhynchus kisutch</i>	Round whitefish	<i>Prosopium cylindraceum</i>
Common carp	<i>Cyprinus carpio</i>	Sand shiner	<i>Notropis stramineus</i>
Common shiner	<i>Notropis cornutus</i>	Shorthead redhorse	<i>Moxostoma macrolepidotum</i>
Creek chub	<i>Semotilus atromaculatus</i>	Silver redhorse	<i>Moxostoma anisurum</i>
Emerald shiner	<i>Notropis atherinoides</i>	Slimy sculpin	<i>Cottus cognatus</i>
Fathead minnow	<i>Pimephales promelas</i>	Smallmouth bass	<i>Micropterus dolomieu</i>
Freshwater drum	<i>Aplokinotus grunniens</i>	Spotfin shiner	<i>Notropis spilopterus</i>
Gizzard shad	<i>Dorosoma cepedianum</i>	Spottail shiner	<i>Notropis hudsonius</i>
Golden redhorse	<i>Moxostoma erythrurum</i>	Trout-perch	<i>Percopsis omiscomaycus</i>
Golden shiner	<i>Notemigonus crysoleucas</i>	Walleye	<i>Stizostedion vitreum vitreum</i>
Grass pickerel	<i>Esox americanus vermiculatus</i>	White crappie	<i>Pomoxis annularis</i>
Green sunfish	<i>Lepomis cyanellus</i>	White sucker	<i>Catostomus commersoni</i>
Johnny darter	<i>Etheostoma nigrum</i>	Yellow perch	<i>Perca flavescens</i>
Lake chub	<i>Couesius plumbeus</i>		

known about how fish use the nearshore areas as spawning sites, nursery areas, feeding grounds, or escape habitat.

During the 10 years of sampling for this project, 59 species of fish were collected (Table 3), which testifies to the extensive use of the inshore zone by a diverse group of fishes. Of these, about six were considered abundant and included: alewife, rainbow smelt, spottail shiner, trout-perch, yellow perch, and bloater. Adults of five of these six species (all except bloater) began moving shoreward from offshore overwintering areas in March or April. Bloaters remained offshore and only entered the nearshore zone during upwellings. The other five species moved offshore after spawning, although some spottail shiners, trout-perch, and yellow perch remained inshore until fall. Winter abundance of all species was very low, with only longnose sucker, spottail shiner, white sucker, and yellow perch consistently present inshore.

Rainbow smelt spawned first in late April-May, followed by alewife and spottail shiner which spawned from June to August. Trout-perch had a prolonged spawning season, April-August. Yellow perch spawned over rocky substrates (Dorr 1982) outside the study area in late May-early June. Burbot, johnny darter, lake trout, ninespine stickleback, and slimy sculpin also spawned inshore, their eggs being deposited at various times in winter, summer, or fall.

Yearlings of the five most abundant species moved inshore 1 month later than adults and remained until early fall when they moved offshore. Young-of-the-year of these five species used the inshore waters as a nursery from summer to mid-fall; alewife and spottail shiner were extremely abundant at 1 m during the day but moved to deeper water at night.

Spring-planted brown trout, chinook and coho salmon, and rainbow trout juveniles were occasionally common inshore and moved alongshore in shallow (1 m) of water. Salmon moved offshore in mid-summer, while some trout remained in 25°C inshore water throughout summer, apparently to feed on terrestrial insects.

During upwellings, all five abundant species generally moved shoreward. Bloater, lake trout, northern pike, rainbow pike, rainbow smelt, and trout-perch closely followed the inshore movement of the thermocline. A general diel pattern of movement inshore at night, presumably starting at dusk, and offshore during daylight by most species, especially yellow perch and trout-perch, suggested that prey species sought protection while predators sought food.

Over the 10-year study period, some dramatic changes in species abundances occurred (see Jude and Tesar 1985 for detailed discussion). Alewife populations attained peak abundance in Lake Michigan during 1966 when fish in large numbers died and littered beaches (Brown 1972). The alewife was still the dominant species in our catches when we began sampling in 1973. However, by 1980-82, alewife catches declined by 86% from their 1973-1979 averages (Jude and Tesar 1985). Bloater catches increased 24-fold in 1980-82 from mean catches in 1973-79, since commercial fishing was prohibited in 1976. During the 1980s, rainbow smelt and yellow perch field catches increased three- and five-fold, respectively, presumably as a result of decreased alewife predation on the pelagic larvae of these two species. Such predation by alewife on a variety of species has been reported by Kohler and Ney (1980) and suggested by Crowder (1980) and Smith (1970). Spottail shiner and trout-perch populations showed no long-term changes, but abundances in catches declined noticeably in 1982. Alewife did not adversely impact spottail shiner or trout-perch, presumably because their demersal larvae escaped alewife predation. However, it appears that the increased yellow perch population has put more predatory pressure on spottail shiner and trout-perch, since their catches declined in 1982. Lastly, the

TABLE 3. Total number of fish collected in standard series sampling at the D. C. Cook Nuclear Plant, southeastern Lake Michigan, 1973-1982. Common names assigned according to Robins, et al. (1980).

Species	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982
Alewife	148,450	76,099	41,738	136,743	55,979	39,453	142,173	16,450	26,486	8,237
Spottail shiner	20,583	24,413	19,814	14,116	22,568	36,601	27,820	32,839	22,631	12,845
Rainbow smelt	16,294	5,754	3,109	1,265	1,455	9,521	5,276	12,443	14,250	7,837
Yellow perch	3,735	4,536	4,334	2,498	3,379	1,576	4,659	12,770	25,850	17,635
Trout-perch	3,558	1,578	905	1,955	3,239	3,088	1,730	3,122	1,608	280
Bloater	126	225	49	107	227	1,392	2,988	2,861	9,912	1,461
Banded killifish	—	—	—	—	—	—	—	—	—	1
Black bullhead	2	2	—	—	—	—	—	3	1	—
Blackchin shiner	—	—	—	—	—	1	—	—	—	—
Black crappie	—	—	—	—	—	—	1	—	—	—
Blacknose dace	—	—	—	—	—	—	—	—	1	—
Blacknose shiner	—	—	—	—	—	—	—	—	1	—
Bluegill	10	46	2	3	2	—	1	—	11	1
Bluntnose minnow	—	1	—	—	1	—	1	—	1	—
Brook silverside	—	—	—	1	—	1	—	—	1	—
Brown trout	78	51	26	90	61	162	60	40	3	24
Burbot	6	15	15	6	8	5	5	7	6	3
Central mudminnow	—	—	—	—	—	—	1	—	—	2
Channel catfish	10	17	9	13	9	5	8	—	20	9
Chinook salmon	29	47	59	35	160	337	332	182	47	25
Coho salmon	23	147	54	25	12	70	65	13	8	3
Common carp	27	27	50	32	92	36	71	28	48	84
Common shiner	—	—	—	—	—	—	—	—	—	1
Creek chub	—	—	—	—	1	—	—	—	—	—
Emerald shiner	49	13	1	—	27	10	12	—	19	17
Fathead minnow	2	—	—	—	—	1	2	1	1	—
Freshwater drum	—	—	—	—	1	—	—	—	—	18
Gizzard shad	23	84	193	51	104	106	159	53	82	389

Golden redhorse	-	-	-	-	9	-	3	-	-	1
Golden shiner	2	1	-	2	1	2	-	-	-	1
Grass pickerel	-	-	-	-	-	-	-	-	1	-
Green sunfish	-	6	-	-	-	-	1	-	-	-
Johnny darter	207	293	142	304	423	401	233	198	151	92
Lake chub	-	-	-	-	-	-	1	-	-	-
Lake herring	-	1	1	-	-	1	-	-	-	-
Lake sturgeon	-	-	1	1	2	1	-	-	-	-
Lake trout	162	125	85	37	187	286	164	121	105	176
Lake whitefish	2	1	2	6	-	9	7	14	7	5
Largemouth bass	1	1	1	1	-	-	-	-	2	2
Logperch	-	-	2	-	-	-	-	-	-	-
Longnose dace	41	43	18	27	60	26	6	34	3	2
Longnose sucker	73	99	94	40	99	71	98	47	70	50
Mottled sculpin	16	-	-	-	3	-	6	15	7	10
Ninespine stickleback	19	24	26	9	7	5	8	8	2	2
Northern pike	30	16	6	-	-	2	4	-	1	1
Pumpkinseed	-	-	1	-	-	-	-	-	-	-
Quillback	-	-	1	2	3	2	-	-	-	6
Rainbow trout	86	8	15	14	12	21	14	22	18	4
Rock bass	2	-	-	-	1	-	-	-	1	-
Round whitefish	-	-	-	-	-	-	2	-	3	3
Sand shiner	-	4	34	39	23	12	7	10	27	12
Shorthead redhorse	-	-	4	-	1	-	4	-	1	8
Silver redhorse	-	-	1	3	1	1	10	1	3	5
Slimy sculpin	79	272	111	85	30	14	128	55	74	199
Smallmouth bass	-	-	-	1	-	-	-	-	1	-
Spotfin shiner	-	-	-	-	-	2	3	-	-	-
Walleye	-	-	-	-	-	-	-	-	1	4
White crappie	-	-	-	-	-	-	-	1	-	-
White sucker	174	126	89	89	173	118	188	113	131	169
Total number	193,899	144,075	70,992	157,600	88,360	93,339	186,241	81,451	101,496	49,624

bloater appears to have been released from overfishing pressure and the adverse impact of the alewife since its population numbers increased dramatically. Evidence of this surge was documented both in field (Tesar and Jude 1985) and impingement (Thurber and Jude 1985) collections in the late 1970s and early 1980s.

These changes in the fish populations have also caused some substantial changes in the zooplankton and water transparency of Lake Michigan (Scavia et al. 1986, Evans and Jude 1986). With the decline of the zooplanktivorous alewife in the lake, there was a shift to larger species of zooplankton (*Daphnia pulicaria*) which presumably more efficiently removed algae from the epilimnion and caused increased water transparency. Inshore, the zooplankton species composition did not change, although there was a substantial decline in total biomass. Evans and Jude (1986) attributed this to the ever-increasing predatory pressure provided by the yellow perch which replaced the alewife inshore.

Data collected in 1986 in nearshore eastern Lake Michigan indicate that the yellow perch populations are still increasing, but showing signs of density-dependent growth. Yearlings we collected in August 1986 were 25 mm shorter than fish collected during August 1977-1981. Larval fish samples showed that alewife are considerably reduced from their abundances in 1977-1981, while larval yellow perch are present in densities never before recorded in Lake Michigan. The most interesting discovery may be the return of emerald shiner which was predicted by Jude and Tesar (1985) to return to its former abundance with the decline of the alewife. This minnow was collected in substantial numbers in seine hauls in August 1986. The restoration of this formerly abundant planktivore to the Lake Michigan fish fauna is yet another sign of the declining abundance and decreasing impact of the alewife on fish dynamics in the Lake Michigan ecosystem.

EFFECT OF PHYSICAL FACTORS ON FISH

Throughout our studies, we were constantly grappling with distinguishing the impact of plant operation from those caused by physical factors. Water temperature was a major environmental variable that could influence year-class strength of most species. Temperature controls the timing of spawning and, depending on the thermal history thereafter, strongly influences egg incubation and survival of newly hatched larvae. Storms can have an adverse impact on freshly deposited eggs. Correlations between years of elevated temperature with abundance of young-of-the-year were statistically significant for only spottail shiner, a species which is a resident of the nearshore zone, and not as mobile as the other abundant species with which we worked. There were strong indications that correlations would have been found for other species, if we would have collected data over a shorter interval than a month.

The timing of temperature stratification and warming in the nearshore waters where we sampled was directly related to the species complex we could expect to catch in our gear. In the spring, if temperatures were still cold during sampling, rainbow smelt, lake trout, and some bloaters were sure to be captured. If waters were warm, then our catches were dominated by alewife, spottail shiner, trout-perch, and fewer rainbow smelt, while bloaters and trout moved offshore to remain in cooler water. Year-to-year comparisons were thus confounded by the strong influence water temperature had on fish movements.

Upwellings had a dramatic impact on the movement and wellbeing of fish. In a related study north of the Cook Plant in eastern Lake Michigan, Heufelder et al.

(1982) determined that a year of frequent upwellings (1979) caused larval fish abundance to be reduced drastically during months of highest densities compared to a year of infrequent upwellings (1980). Larval alewives were transported out of their normal nearshore nursery areas to offshore waters where their fate was undetermined. These periods of upwellings in 1979 caused spawning season to be abnormally prolonged as well.

Upwellings also changed the results of our field sampling. During the summer when water was normally warm, we collected species of fish characteristic of warm water in nearshore Lake Michigan, e.g., yellow perch, spottail shiner, trout-perch. During upwellings, the cold-water species moved inshore with the moving water mass, and included lake trout, bloaters, rainbow smelt, and some other salmonids. We had cases when an upwelling affected only trawling stations at 6-9 m and not beach stations, and cases when the reference site was affected but not the Cook Plant stations. These catches confounded data interpretation and care had to be exercised in determining plant effects.

The thermal bar is another physical phenomenon which develops in the nearshore each spring. The 4°C water acts as a barrier isolating warmer enriched nearshore water from cold offshore water. Fish, particularly alewives, concentrated in the warmer nearshore zone, since it was the warmest water available at that time of year. Sampling in this zone sometimes resulted in large catches of fish, and spectacular impingement events occurred especially in conjunction with storms, which are also common at this time of year.

MOST ABUNDANT SPECIES

Alewife

Distribution and Growth by Age-Group—

Larvae—Alewife larvae occurred in the vicinity of the Cook Plant from June to October (Bimber et al. 1984, Noguchi et al. 1985). Peak monthly densities (250 to 3,300 larvae/1,000 m³) were generally observed in July when inshore water was relatively warm. Low densities of alewife larvae in July 1978 (250 larvae/1,000 m³) and July 1981 (110 larvae/1,000 m³) occurred at low water temperatures (6 to 14.8°C at most strata) during sampling. Upwelling of cold water in summer was hypothesized to inhibit alewife spawning and cause high larval fish mortality or displacement of newly hatched larvae to other areas of the lake (Heufelder et al. 1982). High densities of larval alewives in June 1973, 1974, and 1976 (1,300 to 2,900 larvae/1,000 m³) were associated with high water temperatures (17 to 24°C at most stations) during early June. Catches of alewife larvae substantially declined in August and September.

Abundance of alewife larvae generally decreased with increasing depths during summer. In June, July, and August, mean densities (no./1,000 m³) were 1,910 in the beach zone, 620 at 6-m stations, and 360 at 9-m stations. Densities were still lower at 21-m stations. These data indicated higher spawning activity in shallow than deep water. Otto et al. (1976) and Jude et al. (1979) reported alewives spawn in shallow water.

Alewife larvae were most abundant in the upper strata of the water column. During the day, highest densities were observed at the 2-m stratum. Abundance

declined at lower strata and at the surface. At night, alewife larvae were found most commonly in the 0-, 2-, and 4-m strata, indicating that larvae migrated toward the surface at night but returned to the 2-m stratum during the day.

More fish larvae were caught at night than during the day at beach stations and in the open water presumably because larvae were able to avoid nets more effectively in daylight. During August, however, catches in the beach zone were higher during the day than at night as a higher number of large larvae (>7 mm) was found in day catches. Larger alewife larvae migrated offshore at night and returned to the beach zone during the day.

Alewife larvae ranged from 2.5 to 12 mm during early June and from 2.5 to 25 mm in July and August. Newly hatched larvae (<7 mm) were scarce after August. Occurrence of substantial numbers of large larvae (7 to 25 mm) in July and August samples indicated that alewives remained in the inshore area (beach to 9 m) throughout their early life history. Length-frequency histograms showed one cohort of larvae with a modal length of 4 or 5 mm in early June. Another cohort with a modal length of 8 to 10 mm was occasionally observed during early June. Typically, three length modes occurred in July, corresponding to these major hatching peaks in June, early and late July. Because alewife spawning and hatching are influenced by water temperature, the number of length modes of larvae may be in part related to the frequency of upwellings of cold water (see Heufelder et al. 1982).

Young-of-the-year—Young-of-the-year represented 62–97% of annual catches of alewives (Fig. 2, Tesar et al. 1985, Tesar and Jude 1985). Annual catches showed strong year-classes of young-of-the-year in 1973, 1976, and 1979 with respective catches of 122,000, 120,000, and 138,000. During 1974, 1975, 1977, and 1978, young-of-the-year catches ranged from 30,000 to 52,000. Young-of-the-year populations declined sharply in 1980, 1981, and 1982, as indicated by annual catches of 3,200, 17,000, and 5,100, respectively. Some of this decline is related to salmonid predation on adult and juvenile alewives, which apparently adversely affected production of young-of-the-year (Jude and Tesar 1985).

Young-of-the-year inhabited inshore water in the vicinity of the 9-m contour throughout summer. They were first caught in standard-series gear in July and August. Peak catches generally occurred in August when most young-of-the-year were large enough to be retained in trawls. August catches ranged from 2,200 (1980) to 79,400 (1973). Fluctuations of monthly catches during summer and fall during the 10-year period may be related to changes in water temperatures in inshore areas. Young-of-the-year alewives migrate offshore in the fall (Graham 1956), and our catches of young-of-the-year began to decline in September. Newly hatched alewife larvae averaged 4 mm in June. By mid-July, mean lengths of the first cohort of young-of-the-year were 23 to 27 mm; growth rates during the first month of life were from 0.64 to 0.79 mm per day. Monthly growth of young-of-the-year during the rest of the summer and fall cannot be determined due to the mixing of several cohorts in samples. Yearling alewives measured 77 to 98 mm in April. Because yearling growth does not start until late June (Brown 1972), this size range represented the length attained by young-of-the-year at the end of the first growing season.

Yearlings—Yearlings were generally less abundant than adults or young-of-the-year in the study area (Fig. 2). In Lake Michigan, Wells (1968) and Brown (1972) reported most yearlings live offshore, in mid-water, until the third summer of life. Some yearlings migrated inshore during spring and were caught in gill nets and

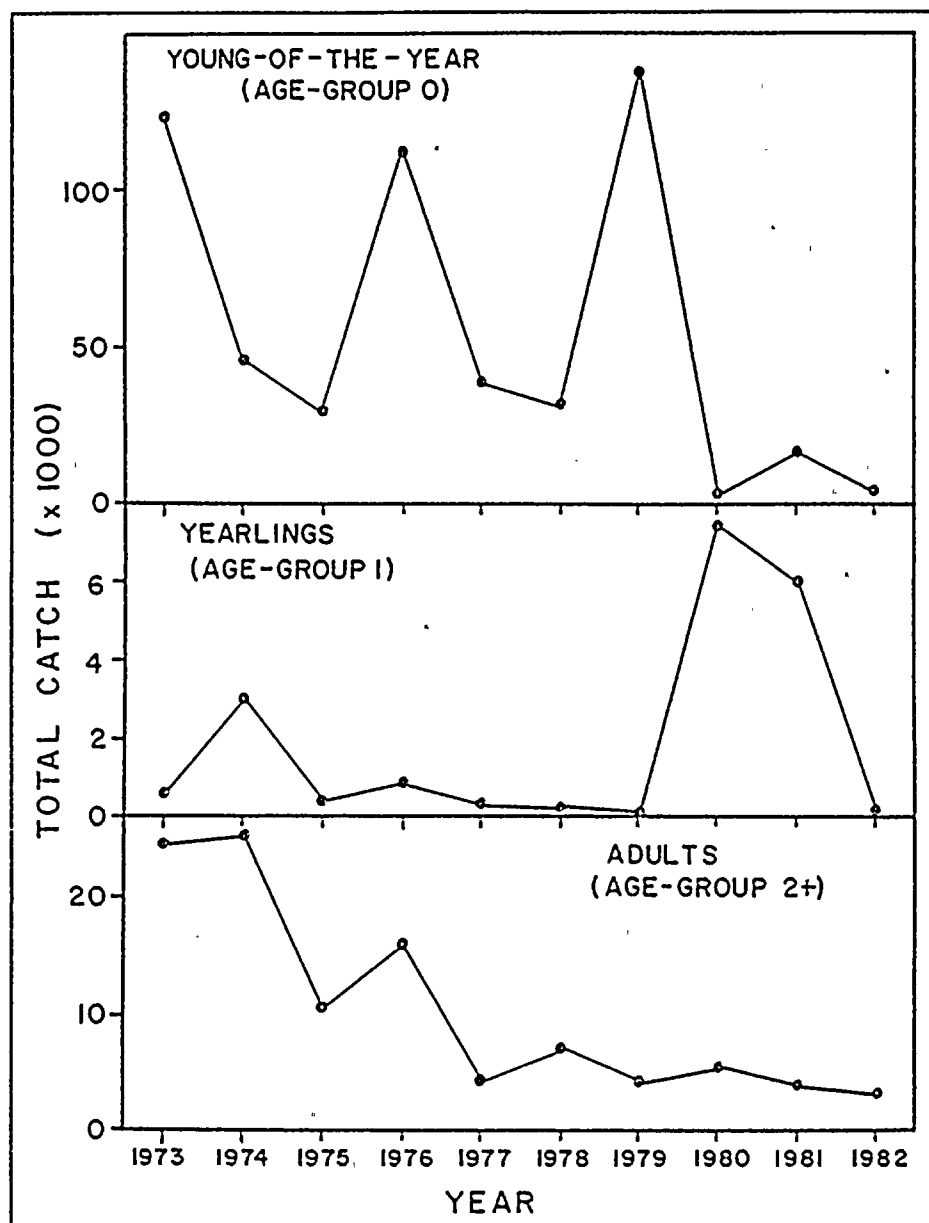


FIGURE 2. Yearly total catch of three age-groups of alewives. Fish were caught from April to October by standard series netting in Cook Plant study areas, south-eastern Lake Michigan.

trawls along with adults. Yearlings were collected from April to July, with most being found in May and June. Annual catches of yearlings ranged from 30 to 7,500.

During spring 1973, 1975, and 1978, mean lengths of yearlings ranged from 90 to 101 mm. Lower mean lengths in spring 1974 (86 mm), 1977 (84 mm), and 1980 (75 mm) may have resulted from intraspecific competition for food during the first summer of life, because as has been discussed, 1973, 1976, and 1979 year classes were very strong. Despite low abundance of young-of-the-year in 1980, however, yearlings had a mean length of only 84 mm in spring 1981.

Adults—Adult alewives (135 to 250 mm) migrated inshore during spring and returned to deep water in late summer. Adults occurred from the beach zone to 9 m in spring. Most were caught during May and June. Monthly catches began to decline in July, and adults were scarce in the study area in October and November.

Adult populations declined steadily over the 10-year period (Fig. 2, Jude and Tesar 1985). Annual catches were 26,000 fish in 1973 and 3,000 in 1982. Strong year classes in 1973, 1976, and 1979 did not produce an abundance of age-2 or older fish, suggesting heavy mortality of yearling and age-2 alewives occurred. Salmonid predation (Jude and Tesar 1985, Stewart et al. 1981) and cold winters (Eck and Brown 1985) have been forwarded as causes of the alewife decline.

In spring, most adult alewives were 160 to 180 mm. Based on age-length data of Lake Michigan alewives (Brown 1972), fish in this size range were approximately 3 to 5 years old. Most age-2 alewives were 110–130 mm and were immature in spring and summer. They remained in mid-water (Brown 1972) and were, therefore, not vulnerable to our sampling gear. Modal lengths of alewives were approximately the same from April to July, suggesting little growth took place during spring and early summer, a finding substantiated by Flath and Diana (1985).

Temperature-Catch Relationship—

Most alewives were collected from water having temperatures from 11 to 23°C. Wells (1968) reported alewives preferred similar temperatures (8 to 22°C) in eastern Lake Michigan. Alewives displayed an age-temperature relationship in which younger fish occurred in warmer water than older fish. Young-of-the-year, 25 to 67.5 mm, were collected from water having temperatures of 19 to 26°C, and adults, 140 to 240 mm, were found in water having temperatures 10 to 17°C. Brandt et al. (1980) reported catching young-of-the-year in 17 to 20°C water and adults in 11 to 14°C water.

Inshore migration of adults during spring is related to warming trends of the lake (Wells 1968). Low catches of adults in April 1975, 1978, and 1979 may in part be related to lower than normal water temperatures (1 to 8.7°C). Catches of young-of-the-year may also be influenced by water temperatures. Low catches of young-of-the-year in 1978 were probably due to upwellings of cold water (9 to 12.5°C), whereas high catches during October 1978 and 1979 may have resulted from relatively high water temperatures (14 to 18°C).

Other Considerations—

In Lake Michigan, alewives spawn from May to August (Norden 1967, Edsall 1970, Brown 1972). Gonad and fish larvae data indicated that most spawning took place in June and July in nearshore southeastern Lake Michigan. Spent adults were occasionally found in April, early May, and October. Dead alewives, apparently in

too poor condition to withstand harsh winters and spawning stress, were observed on the beach from May to July.

Adult alewives migrate to the surface at night and return to lower strata during the day (Graham 1956, Brandt et al. 1980). We caught more alewives during the day than at night because gill nets, our primary sampling gear, fished near bottom. Alewives were the most important forage species in Lake Michigan during 1973-1982. In our study, they were found in the stomachs of lake, brown, and rainbow trout, coho and chinook salmon, yellow perch, rainbow smelt, burbot, channel catfish, northern pike, and other alewives.

Bloater

Distribution and Growth by Age-Group—

Larvae—Bloater larvae were not collected in our study area during 1973-1982. However, unidentified coregonine larvae which could have been bloomers were collected once in field samples in August 1978 and in entrainment samples in May 1977. For southeastern Lake Michigan, Wells (1966) found most bloater larvae near bottom at 70- to 100-m depths. Using otolith analyses, Rice et al. (1985) evaluated larval bloater growth by collecting and fertilizing eggs in February; the eggs hatched in mid-May. At the J. H. Campbell Plant north of the Cook Plant, Jude et al. (1981b) collected larval bloomers in low numbers in June-August, 1977-1980.

Young-of-the-year—Bloater young-of-the-year were caught in the Cook Plant study area at 6 and 9 m during September, October, and November. A few young-of-the-year also occurred in the beach zone in September and October. At the Campbell Plant, a considerable number of young-of-the-year bloomers were collected in September to December with mean length ranges of 67 to 86 mm. In fall at the Cook Plant, young-of-the-year bloomers ranged from 35 to 117 mm (modal length 80 mm). At the end of their first summer of life, bloomers reach approximately 90 to 100 mm in Lake Michigan (Jobes 1949, Wells 1966, Brown 1970), Lake Ontario (Stone 1944), and Lake Superior (Dryer and Beil 1968).

Yearlings—Yearling bloomers were more commonly caught than adults and young-of-the-year (Fig. 3). Most yearlings occurred at 6 and 9 m in June and July; low numbers were observed in inshore areas during May and August. Yearlings migrated offshore in fall. In the summer, yearlings followed the thermocline as it moved shoreward during upwelling. Wells (1968) and Crowder and Magnuson (1982) reported yearlings live in the thermocline.

Yearlings collected in spring were usually at modal lengths of 90 to 100 mm. In July and August, yearlings reached modal lengths of 120 to 130 mm. At the end of their second summer of life, Great Lakes bloomers are approximately 145 to 155 mm (Stone 1944, Jobes 1949, Dryer and Beil 1968, Brown 1970).

Adults—Adult bloomers live in deep water and move inshore with upwelling of cold water in summer. Dryer (1966) and Wells (1968) reported adult bloomers migrated inshore to 18 m and occasionally to shallower water during summer. Near the Cook Plant, adults were found at 6 and 9 m during upwellings. Most were caught during June and July. Adult bloomers were scarce in the study area during April, May, and after July. Adult bloomers ranged from 145 to 314 mm, with a modal length of 155 to 165 mm. Starting in 1977, we saw a dramatic increase in the catch of young-of-the-year bloomers, a response attributed to the banning of gill nets in 1976 and the ongoing alewife decline (Fig. 3, Jude and Tesar 1985). Yearlings

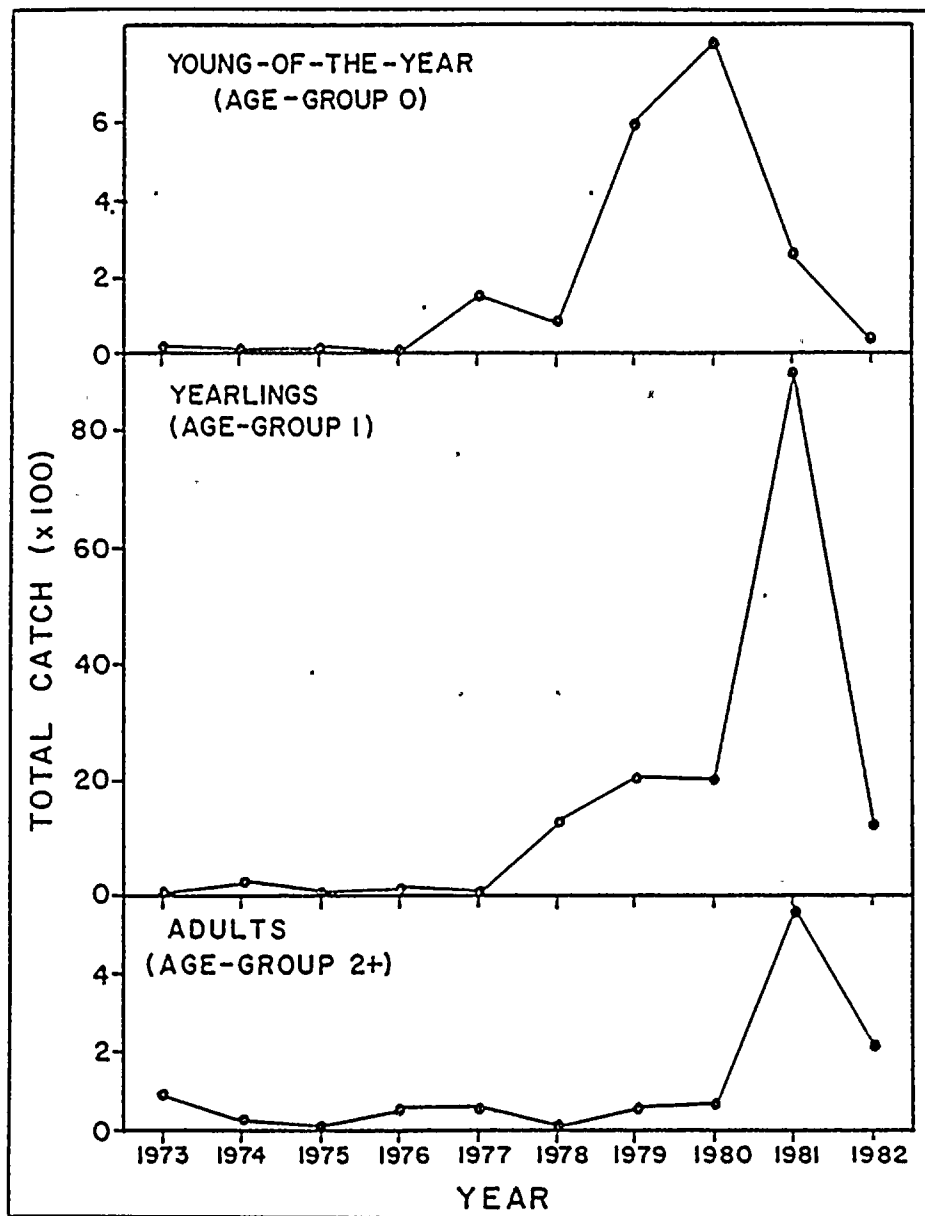


FIGURE 3. Yearly total catch of three age-groups of bloaters. Fish were caught from April to October by standard series netting in Cook Plant study areas, south-eastern Lake Michigan.

increased in 1978 and adults in 1981. We also noted increased impingement rates for this species.

Temperature-Catch Relationship—

Most bloaters were caught in water temperatures of 6 to 20°C. Wells (1968) reported that during summer adults were most abundant in water from 6 to 10°C, while Crowder and Magnuson (1982) found yearlings and adults most abundant at 5 to 15°C in southeastern Lake Michigan. The fact that we collected bloaters at somewhat higher temperatures than other investigators may have been related to the preponderance of juveniles in our samples, including young-of-the-year, which Wells and Crowder and Magnuson did not sample. Another possibility is that perhaps fish caught at higher temperatures only represented fringes of the bloater population while the major portion of the population may have been in cooler waters.

Other Considerations—

In southeastern Lake Michigan, bloaters spawn from mid-January to mid-March and larvae hatch from mid-May to mid-July (Wells 1966). Adult bloaters collected in early summer showed slight to moderate gonad development.

Rainbow Smelt

Distribution and Growth by Age-Group—

Larvae—Rainbow smelt larvae first appeared near the Cook Plant during late April or early May. Abundance of rainbow smelt larvae peaked in May and began to decline in June. Mean densities in May were from 0 to 420 larvae/1,000 m³. Rainbow smelt larvae were scarce after June. During May beach densities were generally higher than densities in the open water because most rainbow smelt spawning took place in the beach zone (Jude et al. 1979). Beach samples taken in May sometimes contained low densities of rainbow smelt larvae because sampling was conducted after most rainbow smelt larvae had dispersed to the open water. During June and July, rainbow smelt larvae were more common in the open water (6 and 9 m) than in the beach zone but were scarce at 21-m stations. Rainbow smelt larvae occurred almost uniformly throughout the water column, but they were generally absent from the surface in the day time.

Rainbow smelt larvae ranged from 3.5 to 8 mm in early May and from 4.5 to 22 mm during early June. Most hatching took place during early May. However, Tin and Jude (1983) reported substantial hatching of rainbow smelt larvae in eastern Lake Michigan during late June. Scarcity of newly hatched larvae (<7 mm) during June or early July suggested little late hatching of rainbow smelt near the Cook Plant.

Young-of-the-year—Annual catches of young-of-the-year fluctuated considerably during 1973–1982 (Fig. 4). Strong year classes were produced in 1973, 1978, and 1980. Distribution was influenced by bottom temperatures. Young-of-the-year moved to deeper water when water temperatures in the inshore zone were high and

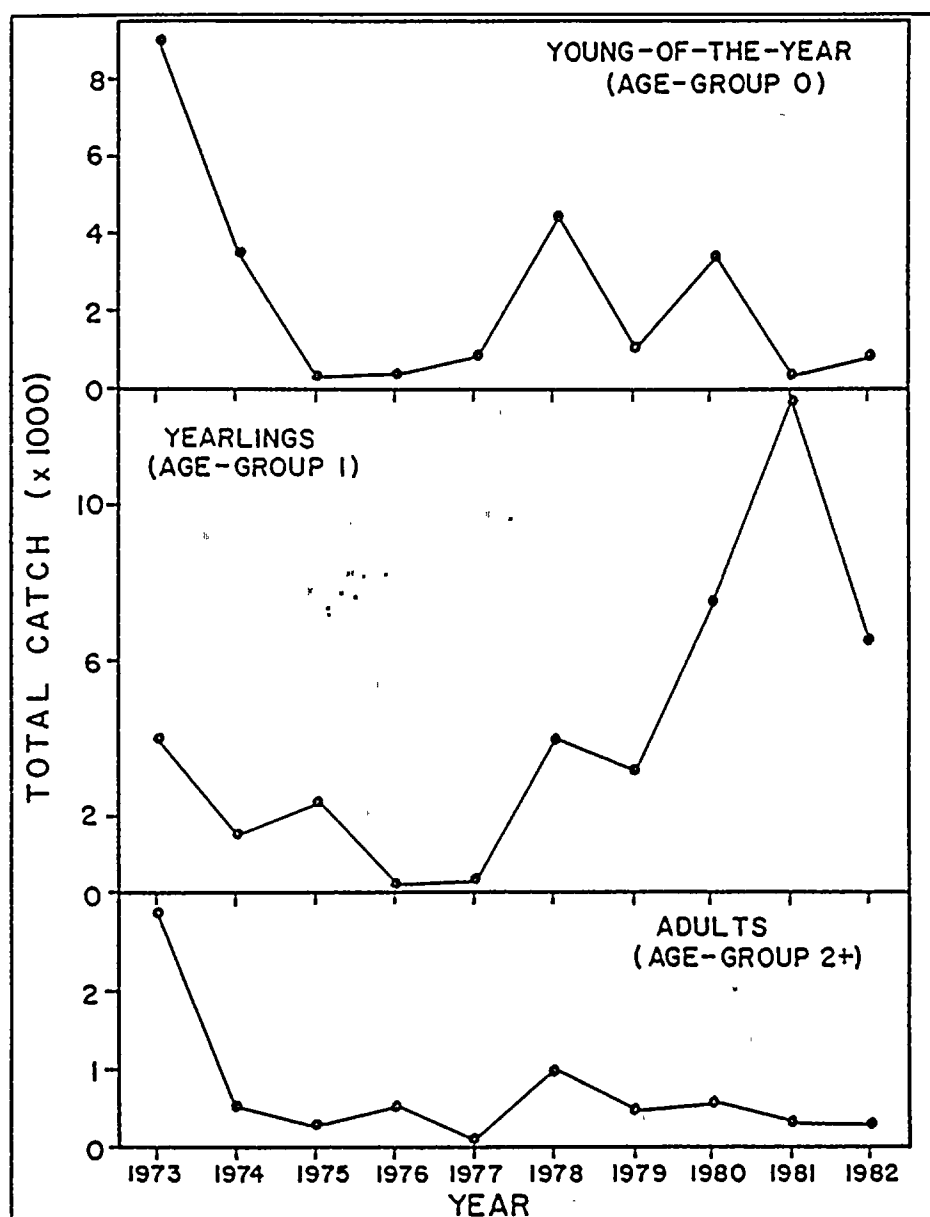


FIGURE 4. Yearly total catch of three age-groups of rainbow smelt. Fish were caught from April to October by standard series netting in Cook Plant study areas, southeastern Lake Michigan.

returned to within the 9-m contour with upwellings of cold water. High catches were observed in August 1973, 1974, and 1978 when bottom temperatures were from 8 to 18°C. Occasionally, young-of-the-year were collected in the beach zone during summer. Low young-of-the-year catches in summer were usually associated with high bottom temperatures (>18°C). In September, young-of-the-year began to migrate offshore; a substantial portion of their population remained inshore until November.

Young-of-the-year had mean lengths of 32 to 44 mm in August. They grew approximately 0.28 mm per day during the summer, and at the end of the first growing season they attained mean lengths from 46 to 64 mm.

Yearlings—Yearlings accounted for 54 percent of the total catch of rainbow smelt in standard-series gear. They were most abundant during 1980, 1981, and 1982 (Fig. 4). Yearlings migrated inshore during spring. They concentrated from the beach zone to 9 m in May. Most yearlings moved to water deeper than 9 m in June in response to warming in the inshore area. Yearling catches peaked in May and began to decline in June. In summer, yearlings sometimes followed upwelled cold water to the 6- and 9-m contours. Offshore migration of yearlings took place in September.

Yearlings reached mean lengths of 63 to 69 mm in April. Growth was slow (0.10 mm/day) from April to June, but it became more rapid (0.39 mm/day) during July and August. This increased growth rate may be related to increased inshore temperatures or food during summer. Mean lengths of yearlings were 122 to 142 mm in November.

Adults—Annual catch of rainbow smelt was highest in 1973 (Fig. 4). Rainbow smelt populations were at the lowest levels during 1975, 1976, and 1977. An increase in abundance was observed during 1978–1982. Adult rainbow smelt migrated to shallow water during spring spawning. Most were caught in April and May. They returned to deep water in June. During summer, substantial numbers of adults returned to the 6- and 9-m contours during upwelling of cold water. Rainbow smelt spawning season lasts only 2 weeks (Scott and Crossman 1973). Annual catches of adult rainbow smelt near the Cook Plant were generally low because our sampling period rarely coincided with major spawning runs. High catches were observed in 1973 when sampling coincided with peak spawning activity.

Rainbow smelt mature at age 2 (Baldwin 1950, Bailey 1964). Modal lengths of April-caught adults were 140 to 180 mm. Because age-1 individuals attained lengths of 122 to 142 mm by fall, most rainbow smelt in the 140- to 180-mm size range in April were age-2 and older fish. Rainbow smelt showed an increase in our catches during 1980–82, presumably in response to the decline in alewife populations.

Temperature-Catch Relationship—

The majority of rainbow smelt were collected in water temperatures from 6 to 16°C. Young-of-the-year most often occurred in water temperatures of 15 to 17°C, yearlings predominated in water with temperatures of 10 to 13°C, and adults were generally caught in water 10°C or less. These data agreed with those of Wells (1968) who found rainbow smelt at temperatures of 6 to 14°C. Ferguson (1965) reported a temperature preference of 6.1°C for rainbow smelt in Lake Erie. MacCallum and Regier (1970) pointed out that rainbow smelt can be found in a wide temperature range.

Other Considerations—

Of forage fish consumed by the five salmonid species inhabiting Lake Michigan, rainbow smelt ranked second. As the alewife population continues to decline, diet diversity of salmonids has increased, with rainbow smelt, yellow perch, bloater, and sculpins making up a larger proportion of the fish eaten (Hagar 1984). Stomach fullness has decreased and percentage of empty stomachs has increased, suggesting the species now being fed on are less available, particularly to salmon. However, growth data compiled by Hansen (1986) shows that among the salmonids, only large chinook salmon are showing a decline in growth, and only in the southern basin of Lake Michigan. The artificiality of the present Lake Michigan ecosystem, with its marine exotic forage fish and almost totally stocked salmonid predators, will fluctuate and continue to be unstable as the present fish fauna reaches some equilibrium level.

Rainbow smelt were susceptible to a fungal infection on parts of their bodies. This species was the only one which was consistently affected by this malady.

Spottail Shiner

Distribution and Growth by Age-Group—

Larvae—Spottail shiner spawning is concentrated in nearshore waters (Jude et al. 1979, Wells and House 1974), thus distribution of larvae is primarily nearshore. Few samples from open water stations contained spottail shiner larvae, whereas, spottail shiner larvae were consistently found in beach station samples during summer. More spottail shiner larvae were found at 6-m stations than 9-m stations, and very few were collected at 21 m (Noguchi et al. 1985). Spottail shiner larvae tend to be demersal, as demonstrated by their abundance in bottom sled tow samples at the Cook plant in 1974 (Jude et al. 1979) and at the J. H. Campbell Plant near Grand Haven, Michigan, during 1977–1981 (Jude et al. 1982, Madenjian and Jude 1985). Spottail shiner larvae were usually present from June through August. Newly hatched larvae were present in nearly all August samples, showing that some hatching continued into August of most years. Mean lengths of larvae for each month also gave an indication of continued hatching; some years mean length remained about the same or decreased during June to August. Growth from larvae to young-of-the-year was calculated using mean lengths of larvae in beach samples during months of peak abundance (generally June or July) and maximum mean lengths attained by young-of-the-year during fall. Calculated growth rates ranged from 0.31 mm/day in 1974 to 0.58 mm/day in 1982, and they averaged 0.40 mm/day for 1973–1982.

Young-of-the-year—Distribution of young-of-the-year spottail shiners was not as strongly nearshore as for larvae. Young-of-the-year at modal length intervals of 30 to 50 mm were usually recruited to our sampling gear in July or August and were more abundant in shallow than deep water during July to September (Fig. 5). Despite the fact that four or five stations were trawled from April through November while only three were seined, and each trawl haul fished a larger area than a seine haul, seine catches of young-of-the-year exceeded trawl catches in all but 2 years. In addition, total seine catches for the year were often more than double total trawl catches of young-of-the-year. More young-of-the-year were trawled at 6-m stations than at 9-m stations. During fall, young-of-the-year dispersed offshore, as shown by

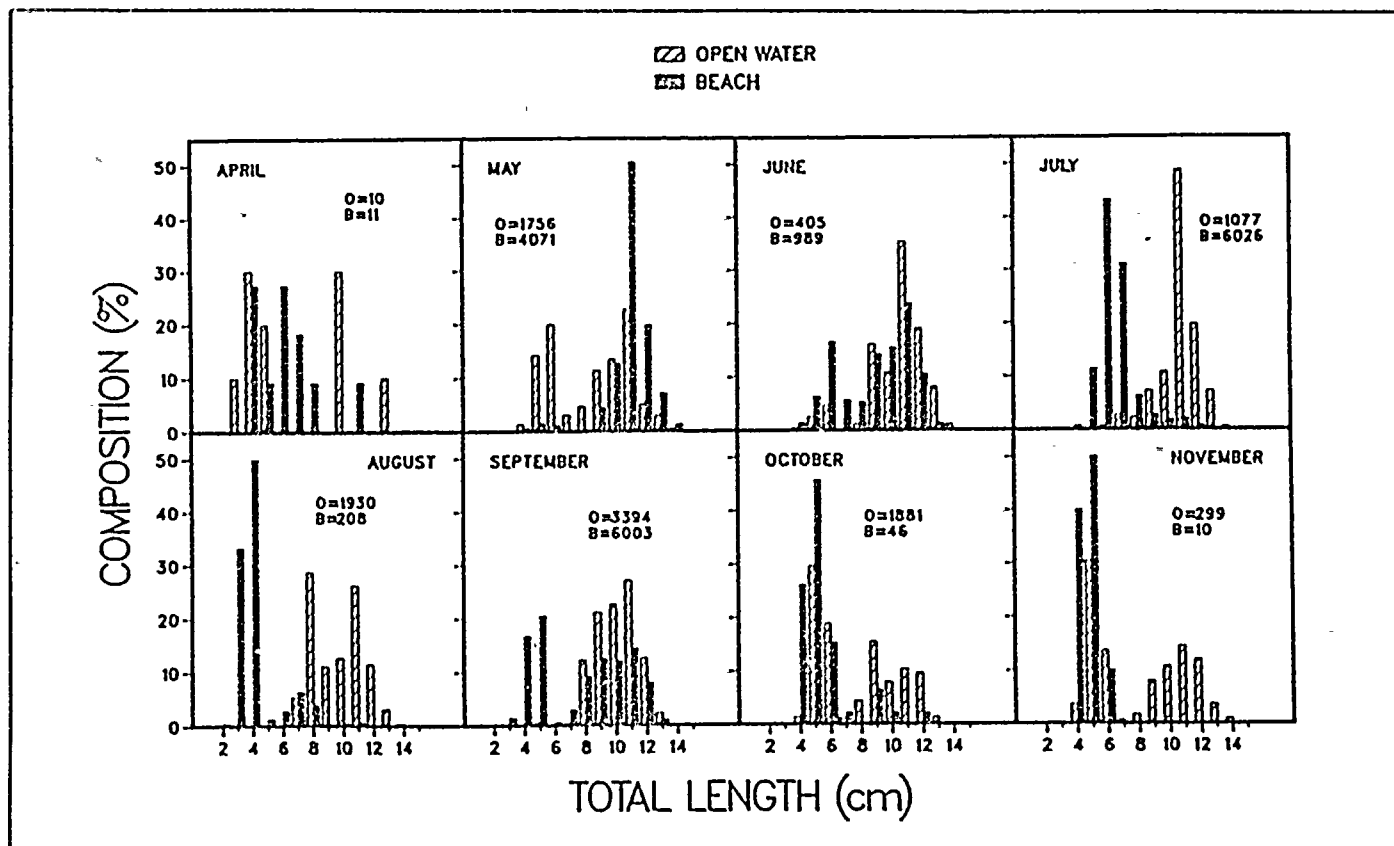


FIGURE 5. An example of monthly length-frequency plots (% composition) for spottail shiners collected with seines (beach) and trawls (open water) at all stations at the D. C. Cook Nuclear Plant, 1980. Sample size given for each month.

sharp decreases in seine and trawl catches and increases in impingement at the 7.3-m deep plant intake (Tesar et al. 1985; Thurber and Jude 1984, 1985). Young-of-the-year generally grew to lengths of 45 and 64 mm before moving offshore beyond sampling stations. Wells and House (1974) found spottail shiners attained a mean length of 62 mm during their first year in southeastern Lake Michigan. Growth rates of young-of-the-year from samples collected during late summer to early fall were between 0.15 and 0.35 mm/day. Calculated young-of-the-year growth was slower than larvae to young-of-the-year growth, and it did not follow the same yearly pattern. Growth rates of young-of-the-year from samples collected during late summer to early fall were between 0.15 and 0.35 mm/day. Calculated young-of-the-year growth was slower than larvae to young-of-the-year growth, and it did not follow the same yearly pattern. Growth rates of young-of-the-year appeared to be related to water temperature. The 2 years of slowest young-of-the-year growth, 1974 and 1978, were associated with below average water temperatures, while most rapid growth in 1973 occurred during above average temperatures (Tesar et al. 1985).

Yearlings—Spottail shiner yearlings, like young-of-the-year, were more abundant in shallow water than open water. Seine catches often exceeded trawl catches by orders of magnitude, particularly during summer. Numbers of yearlings in trawls (6 and 9 m) increased during spring, and then decreased as seine catches from the beach zone increased during June or July. The reverse migration pattern occurred during fall, indicating yearlings moved during spring from deeper water through the 6- to 9-m zone to the beach zone for summer, then back offshore in fall. Yearling spottail shiners collected in April ranged in length from 28 to 84 mm and grew to approximately 100 mm by the end of their second year. Similarly, Wells and House (1974) found spottail shiners attained a mean length of 96 mm by the end of their second year. Yearlings from the Cook Plant area during June to August grew 0.2 to 0.5 mm/day. Growth rates of young-of-the-year were a poor predictor of yearling growth; faster growing young-of-the-year cohorts (1973, 1977) tended to show slower growth rates as yearlings. The 1973 and 1977 year classes were more abundant as yearlings than any other year class; density-dependent factors, e.g., increased competition, may have inhibited growth. The 1978 year class exhibited slow growth as both young-of-the-year and yearlings, probably because of cool temperatures and competition from the larger 1977 year class (Tesar et al. 1985).

Adults (Age 2+)—Lake Michigan spottail shiners overwinter in moderately deep water (9–37 m) (Wells 1968). Spottail shiner adults were sometimes impinged during winter, particularly when water temperatures were above 3°C, but these occasions were somewhat exceptional (Thurber and Jude 1984, 1985). Shoreward migration began in March. As for yearlings, catches at 6- and 9-m stations increased during spring, then seine catches increased and peaked from May to July. After spawning, spottail shiner adults dispersed to deeper waters as shown by trawl and gill net catches in August and fall months (Tesar et al. 1985). Most spottail shiners we collected did not exceed 125 mm; however, a few in 1982 were 165–174 mm. Most (>90%) female spottail shiners matured by 87 mm, and most (>90%) males matured by 89 mm. Both sexes probably first spawned at age 2, but yearlings were sometimes collected along with adults during spawning. Wells and House (1974) found that spottail shiners matured between 65 and 84 mm, and about half the yearlings were mature.

Gonad Conditions and Spawning Times—

Spottail shiners spawned during June through July or August. Fish with ripe gonads were most abundant during June but also were abundant some years in May or July. Spent fish usually first appeared in June, but were more abundant in July. Peak spawning time varies from year to year (Wells and House 1974). Generally, rapidly increasing water temperatures induce early spawning, which may then occur over an extended period. Newly hatched larvae occurred from June through August, coinciding with the spawning season demonstrated by observed gonad condition of adults.

Temperature-Catch Relationship—

Spottail shiners were collected during mid-winter in the beach zone at water temperatures near freezing and during late summer at temperatures over 28°C. However, most spottail shiners were taken at water temperatures between 8 and 24°C (Tesar et al. 1985). Mean temperature at which spottail shiners were collected decreased with increasing size, from 23–25°C for 15–44-mm fish to 15–18°C for adult fish 95–144 mm. This relationship was a result of differential use of nearshore areas by different age-groups. Adult fish usually composed a large percentage of the catch in spring and early summer when inshore water temperatures were in the mid-teens and optimal for spawning. Young-of-the-year were most abundant and susceptible to our gear in late summer to early fall when water temperatures were at annual maxima. Wells and House (1974) also found larger spottail shiners in deeper water, whether or not temperatures were substantially different.

Other Considerations—

Myxosporidian parasitism affected a small number of spottail shiners, usually less than 4% of the catch. Parasitized females outnumbered males by more than a 2:1 ratio. Numbers of parasitized fish usually peaked in June. Parasitism may be spread by the aggregation of spottail shiners for spawning and exacerbated by spawning stress and increasing water temperatures in June.

Trophic relationships of spottail shiners were unclear from our data. Although they can be important forage for sport fish (Wells and House 1974), few spottail shiners were eaten by salmonids and yellow perch. However, with the increased yellow perch population, brought about presumably by the alewife decline, more predation on spottail shiner can be expected (Jude and Tesar 1985). Total combined spottail shiner catches did decline noticeably in 1982 (Fig. 6) because of a large decrease in yearlings. This could be due to increased predation by the expanding yellow perch population in the lake. Spottail shiners, unlike emerald shiners, appeared to be unaffected by changes in the alewife populations. Spottail shiners have demersal larvae which are not vulnerable to alewife predation, while the pelagic larvae of emerald shiners are.

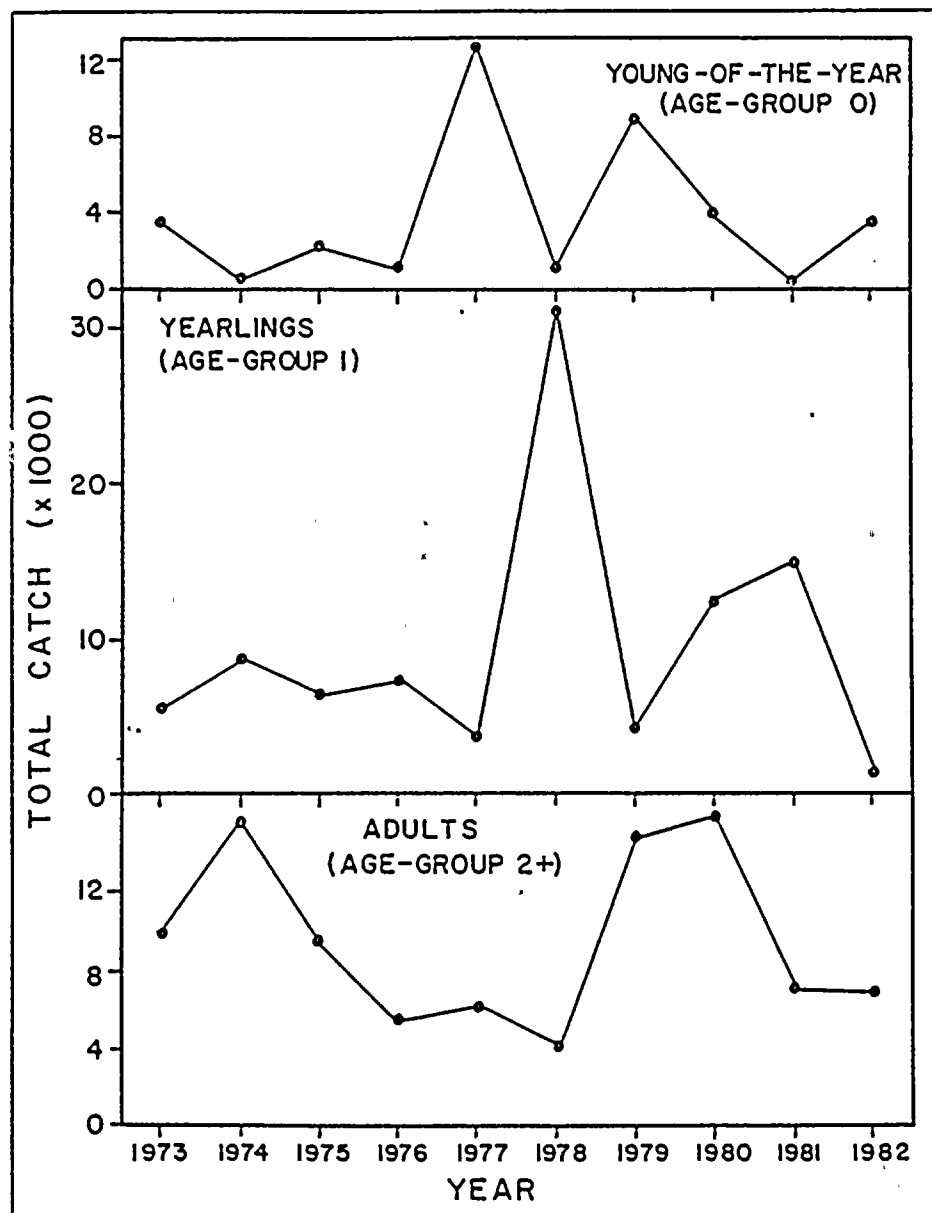


FIGURE 6. Yearly total catch of three age-groups of spottail shiners. Fish were caught from April to October by standard series netting in Cook Plant study areas, southeastern Lake Michigan.

Trout-Perch

Distribution and Growth by Age-Group—

Larvae—Trout-perch larvae (4 to 8 mm) occurred from May to October, suggesting that trout-perch had the longest spawning season of any species collected in the study area. Although adult and juvenile trout-perch were common near the Cook Plant, we caught only a small number of larvae. Larval trout-perch were collected in only a few samples each year. Sample densities ranged from 15 to 149 larvae/1,000 m³. Trout-perch larvae were found from the beach zone to 9 m. Most were caught at night.

Young-of-the-year—Young-of-the-year were caught sporadically by standard-series fishing from August to November. Scarcity of young-of-the-year may have been due to their preference for water deeper than 9 m. Magnuson and Smith (1963) reported young-of-the-year moved offshore as summer progressed. Many young-of-the-year were probably too small to be retained in trawls during summer and fall. However, a few young-of-the-year from 17 to 33 mm were caught during August and September. Shorter young-of-the-year mean lengths (14 to 32 mm) were observed in October due to the presence of a younger cohort that hatched during the latter part of the spawning season. House and Wells (1973) reported young-of-the-year trout-perch reached a mean length of 49 mm at the end of the first summer of life.

Yearlings—Yearlings occurred in the study area from April to October, with peak abundance in July. High catches of yearlings were sometimes observed in June or August. Yearlings were scarce in the beach zone; most were caught at 6 and 9 m. Yearling catches were low in September and October due to offshore migration. At the end of the second summer of life, yearlings attained a mean length of 83 mm both in southeastern Lake Michigan (House and Wells 1973) and in Lower Red Lake, Minnesota (Magnuson and Smith 1963). Trout-perch appeared to grow more slowly in our study area. Yearling trout-perch we collected averaged 29 to 49 mm in May and 61 to 83 mm in October. Growth rates of yearlings during summer and fall were 0.17 to 0.32 mm/day (Tesar and Jude 1985).

Adults—Adults were generally more abundant than young-of-the-year and yearlings (Fig. 7), accounting for 67 percent of trout-perch catches during the study. Adults migrated inshore in April. They were common at 6 and 9 m from June to August. As was found with yearlings, adult trout-perch were seldom caught in the beach zone. Catches substantially declined in November due to offshore migration. Modal lengths of adults ranged from 80 to 120 mm in April. Because yearlings were 61 to 83 mm in mean length at the end of the growing season, fish 80 mm to 120 mm in April were age 2 and older.

Temperature-Catch Relationship—

Trout-perch prefer water temperatures from 10 to 16°C in southeastern Lake Michigan (Wells 1968). In our study area, trout-perch were caught most frequently in 14.6 to 20.5°C water. Jude et al. (1982) reported finding trout-perch in a wider temperature range (4 to 24°C) near the Campbell Plant, eastern Lake Michigan. All age-groups of trout-perch exhibited similar temperature preference.

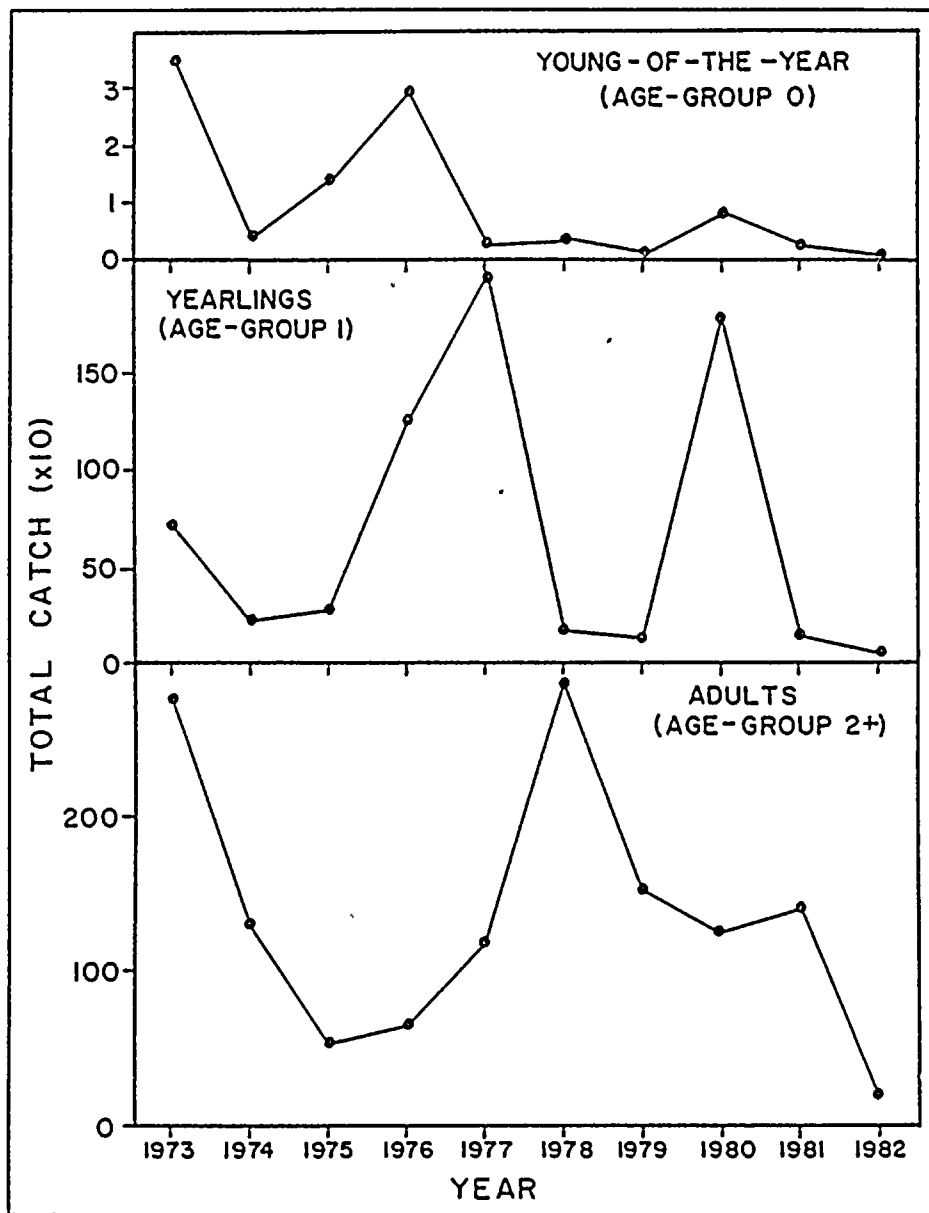


FIGURE 7. Yearly total catch of three age-groups of trout-perch. Fish were caught from April to October by standard series netting in Cook Plant study areas, southeastern Lake Michigan.

Other Considerations—

Ripe adults were found from April to October. Higher numbers of trout-perch in spawning condition were observed in June, July, and August than during April, May, September, and October, suggesting that spawning took place mainly during summer.

Trout-perch were reported to move shoreward at night and retreat to deep water during the day (Magnuson and Smith 1963, Emery 1973). Similar trout-perch behavior was observed in our study area. More trout-perch were caught at night than during the day, indicating that the majority of trout-perch remained in water deeper than 9 m during daytime. However, lower day catches also may have been due to net avoidance during daylight. Inshore-offshore migration may be affected by water temperature. During the day trout-perch appeared to reside farther offshore when the water was warm. In cooler water, trout-perch were less inclined to make diel inshore or offshore movements.

Trout-perch populations, like spottail shiners, appeared to be unaffected by changes in the alewife populations as numbers of trout-perch collected were stable over the 10-year study period (Fig. 7). However, there was a noticeable decline in 1982, which may be increased predation on trout-perch by the burgeoning yellow perch population (Jude and Tesar 1985).

Yellow Perch

Distribution and Growth by Age-Group—

Larvae—Yellow perch larvae were collected from April to July. Because adult yellow perch in the study area were not ready to spawn until May, larvae collected in April and early May probably entered Lake Michigan from inland lakes where spawning begins sooner than in Lake Michigan (see Dorr 1982, Perrone et al. 1983, Jude et al. 1981a). Yellow perch larvae abundance was generally low during April and May, suggesting that only a small portion of yellow perch populations in the study area originated from tributary waters. Peak densities (0.9 to 205 larvae/1,000 m³) occurred during June. Yellow perch larvae were scarce in July and August, presumably due to net avoidance (Noble 1970, Wong 1972).

Most yellow perch larvae were found at 6- and 9-m stations. Densities were generally low in the beach zone or at 21-m stations. Yellow perch larvae became pelagic shortly after hatching. They were distributed throughout the water column, except they were absent from the deepest strata. Night catches were generally higher than day catches due to more effective net avoidance during daylight.

Yellow perch larvae collected ranged from 3.5 to 10.5 mm, most being 7.5 mm or less. Peak hatching took place in early June. Newly hatched larvae were generally scarce after June. Occasionally, substantial hatching occurred in early July as observed in 1979 and 1980. Increased net avoidance by large larvae (Noble 1970) contributed to the complete absence of larvae larger than 10.5 mm.

Young-of-the-year—Young-of-the-year were collected in standard-series gear from July to November. Peak catches sometimes occurred from August to November but were observed most commonly in September. During August and September, young-of-the-year were distributed from the beach zone to 9 m during both day and night. Beach catches generally decreased at night due to movement of young-of-the-year to deep water at dusk. Young-of-the-year migrated offshore in October.

occasionally returning to the 6- and 9-m stations at night in fall. Large catches of young-of-the-year were observed in 1977 and 1982 (Fig. 8).

At the end of the first growing season, young-of-the-year reached lengths between approximately 65 and 105 mm and averaged 81 mm. From June to August, growth rate varied between 0.6 and 0.9 mm/day. These rates were about average for growth of young-of-the-year yellow perch in North America (Jude et al. 1979). From August to October, a slower growth rate (0.5 mm/day) was observed. Yearly growth of young-of-the-year was completed between October and November.

Yearlings—During winter and spring, most yearlings inhabited water deeper than 9 m. Shoreward migration began in June. Yearlings were concentrated in inshore water in July and large catches were obtained in the beach zone. Yearlings began to move away from the beach zone in August, and by December most were in water deeper than 9 m.

During April, yearlings had a mean length of 82 mm. Growth was most rapid in June, July, and August, and appeared to be complete by October or November. Mean lengths of yearlings during fall ranged from 123 to 171 mm, and overall average was 152 mm. Growth of yearlings in the study areas was similar to yearling growth in southern Lake Michigan (Koch 1972), Oneida Lake, New York (Vashro 1975), and Lake Erie (Jobes 1952), but it was less than in northern Lake Michigan (Brazo et al. 1975).

Adults—During fall, winter, and spring, most adult yellow perch were in water deeper than 9 m. Adult catches were low during March, April, and May. Wells (1968) found a similar distribution for adult perch in southeastern Lake Michigan. Adults migrated to inshore areas in June and remained there until September. Large catches of adults in the beach zone were observed in June and July. Migration to deep water took place in October.

Yellow perch spawning occurred from late May to early June over rough substrate along the eastern and southeastern shore of Lake Michigan (Wells 1977, Dorr 1982). Gonad and fish larvae data indicated that yellow perch spawning took place during late May in the vicinity of the Cook Plant. Dorr (1982) indicated that yellow perch spawning was concentrated in an area of rough substrate approximately 3 km north of the Cook Plant. Rock riprap around the intake structure was probably utilized by some yellow perch for spawning. Yellow perch were reported to spawn on similar rock riprap near the Campbell Plant (Jude et al. 1982).

Yellow perch populations, as indicated by annual catches of young-of-the-year, yearlings, and adults from 1973 to 1979 (1,600 to 4,600 fish), remained steady (Fig. 8). Yellow perch abundance substantially increased during 1980, 1981, and 1982, with respective annual catches of 13,000, 26,000, and 18,500 (Fig. 8). There was a concomitant decline in the alewife population, hypothesized to have occurred because of increased salmonine predation (Jude and Tesar 1985). We believe yellow perch larvae, which were preyed on less heavily by alewife, experienced higher survival with the decline in the alewife population.

Temperature-Catch Relationship—

Most (60%) yellow perch were caught at water temperatures between 18 and 24°C. Ferguson (1958) reported yellow perch in Lake Erie preferred water temperatures from 19 to 21°C. The temperature preferences for yellow perch determined from laboratory studies were 21 to 24°C (Scott and Crossman 1973). Laboratory experiments showed that young yellow perch select warmer temperatures than adults

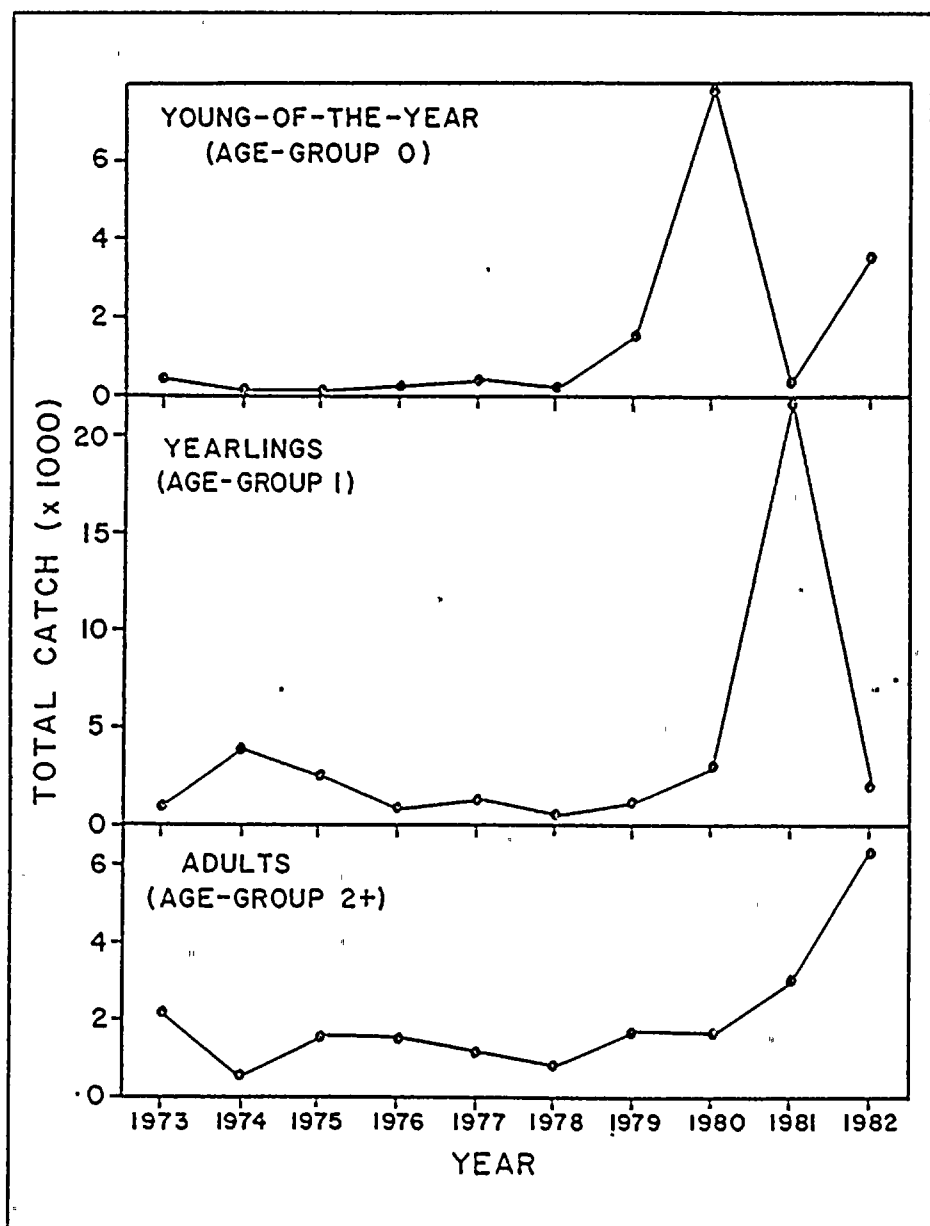


FIGURE 8. Yearly total catch of three age-groups of yellow perch. Fish were caught from April to October by standard series netting in Cook Plant study areas, southeastern Lake Michigan.

(Barans and Tubb 1973, McCauley and Read 1973). In our study areas, larger yellow perch tended to be found in cooler water than smaller individuals.

LESS ABUNDANT SPECIES

Less abundant species were arbitrarily defined as any fish whose average catch was >20 but $<1,000$ fish/yr. In this section, noteworthy species in this category are discussed alphabetically; the pertinent data can be found in Table 3.

Brown Trout

Brown trout were collected from March to November; most were found in spring and early summer. Individuals 300 mm and larger migrated inshore during spring, probably following the shoreward migration of alewives. They were most common in inshore water from March to June. Juvenile brown trout 100 to 280 mm inhabited water from the beach to 9 m during May, June, and July. Brown trout moved to deep water during late summer due to high temperatures in shallow areas. In fall, brown trout catches were usually low because adults were migrating upstream for spawning, and juveniles remained in deep water.

Brown trout collected were between the 120- to 770-mm length intervals. Based on age-length data reported by Merron (1982), brown trout 100 to 270 mm caught in spring and early summer were yearlings; whereas, those 300 mm and larger were 2-years old or older. Yearling brown trout in the study area may have originated from planting or natural reproduction. Brown trout smaller than 100 mm have never been caught in the study area. Frost and Brown (1967) reported that offspring of lake resident brown trout migrated downstream at an age of about 1 year. Yearling abundance fluctuated widely, with largest catches observed in 1973, 1976, 1978, and 1980. Year class strength of brown trout is dependent upon success of natural reproduction and level of plantings in tributary streams. Brown trout 300 mm and larger were most common in the study area during 1978 and 1979.

Brown trout spawn in streams in the fall (Brynildson et al. 1973); in our study area, ripe and spent adults were caught from August to November. Most brown trout were caught in water temperatures of 7 to 17°C. These data agreed with the temperature preferenda of 12.4 to 17.6°C reported for brown trout by Ferguson (1958). Several brown trout were also collected near the Cook Plant at water temperatures from 19 to 24°C, indicating that this species can tolerate relatively warm water.

Coho Salmon

Coho salmon catches were relatively high during spring and declined during summer and fall. Adult coho salmon migrated inshore during April, May, and June. During summer as inshore water temperatures increased, they resided in deeper water outside the study area. Low catches in fall may have been related to upstream migration for spawning. Juvenile coho salmon from planting and natural reproduction migrated downstream to Lake Michigan in spring. In the study area, they were found from the beach zone to 9 m during April, May, and June. Juvenile coho were offshore during summer and fall.

Coho salmon collected were in the 60- to 880-mm length intervals. Juveniles

250 mm and smaller represented 65% of the total coho salmon catch. Coho salmon grow rapidly in Lake Michigan, and most growth takes place during the second summer of life (Stephenson 1968). At planting time, yearlings were 100 to 152 mm (Parsons 1973). During spring of the third year of life, coho salmon collected were 400 to 550 mm. During summer and fall, most 3-year-old coho salmon were 500 mm and larger. This size range was comparable to that of adults from other areas of Lake Michigan where coho salmon over 506 mm were collected during the fall spawning runs of 1978 and 1979 (Patriarche 1980).

The majority of coho salmon were caught in water temperatures of 8 to 17°C. Engel and Magnuson (1971) found coho salmon in comparable water temperatures (8 to 16°C) during summer in a small lake. Final temperature preferendum for coho salmon was 11.4°C during spring (Reutter and Herdendorf 1974). Adults were caught over a wider temperature range than juveniles. Juvenile coho salmon 60 to 240 mm occurred in water temperatures of 12 to 20°C, whereas adults were caught in 5 to 25°C water.

Common Carp

Entrainment and field data indicated that common carp larvae occurred in the study area every year during 1975 to 1982 but were found in only a few samples each year. Larval common carp were collected from June to August, but were most common in July. Larval fish abundance varied greatly from 17 to 11,800/1,000 m³. Common carp larvae were more common in the beach zone than in open water, indicating that spawning took place in shallow water. Larval common carp in the vicinity of the Cook Plant.

Adult common carp occurred in the study area from April to November and showed no definite patterns in seasonal abundance. Peak monthly catches occurred during April to October, except July. Low July catches may have been related to increased spawning activities by common carp in areas not sampled by standard-series fishing. Common carp were found from the beach zone to 9 m. More common carp were, however, caught at 6 m than at 9 m or in the beach zone. Common carp generally preferred warm water, and larger catches were made at the Cook Plant than at Warren Dunes (Fig. 9). These larger catches were probably due to the presence of the thermal discharge and currents due to plant operation. Common carp occurred over a range of temperatures, but greatest catches were observed in water temperatures of 15 to 23°C.

Ripe adult common carp were caught from May through October, indicating an extended spawning season for this species. Larval fish data revealed that peak spawning in the study area took place in July. Very few immature individuals were collected, and the majority of common carp caught every year were between 500 and 750 mm.

Chinook Salmon

Chinook salmon were collected from April to November, with monthly catches generally higher during spring than summer or fall. Larger chinook salmon (> 300 mm) moved inshore during April, May, and June; they were caught mostly at 6 and 9 m. Planted and naturally produced chinook smolt migrated downstream to Lake Michigan in spring. Juveniles occurred at all depths sampled, most being found in the beach zone. Catches were low during July and August because juveniles

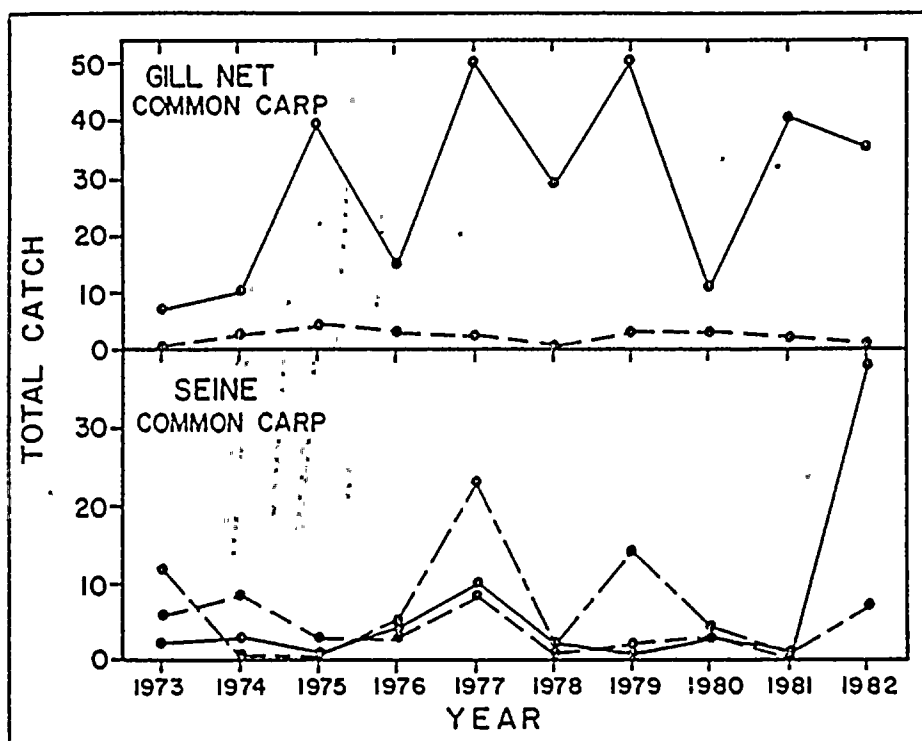


FIGURE 9. Annual catches of common carp in gill nets and seines in the study areas of southeastern Lake Michigan, 1973-1982. The Cook Plant began operation in 1975.

and adults avoided warmer inshore water. During fall, few adults were caught; most were probably migrating upstream for spawning. Juveniles appeared to remain in deep water outside the study area in fall.

Chinook salmon collected were between the 60- and 980-mm length intervals; the majority were juveniles 60 to 300 mm. Young-of-the-year were 51 to 76 mm when planted in spring (Parsons 1973). Juveniles collected in our study area grew to 60 to 170 mm during April to July and to 150 to 300 mm during August to November. Growth was very rapid during the second summer of life. Age-1 fish were 220 to 330 mm during February, March, and April, and 410 to 500 mm in fall. In Lake Superior, chinook salmon attained smaller size at the end of the second year of life (Berg 1978).

Chinook salmon in the Great Lakes spawn in the fall (Berg 1978). The majority of adult chinook collected in September were ripe-running, indicating that spawning took place during this month in southeastern Lake Michigan. Most adults were caught at water temperatures of 5 to 17°C; whereas juveniles 70 to 280 mm were found at water temperatures of 9 to 20°C. Preferred temperatures of young-of-the-year were 12 to 13°C (Brett 1952).

Gizzard Shad

Adult and yearling gizzard shad were found from April to November. Highest catches were observed from August to November. Scarcity of adults during spring and early summer may have been related to spawning activities outside the study area. Young-of-the-year were caught from September to November. Nearly all adults collected were taken at 6 and 9 m. Yearlings and young-of-the-year occurred from the beach zone to 9 m but were most common near the beach.

Most adults were 320 to 470 mm. Based on age-length data reported by Bodola (1966), fish this size were 2 to 4 years old. The largest adult collected was 520 mm. Yearlings were 85 to 164 mm during March, April, and May, and 115 to 195 mm during July and August. Young-of-the-year were 55 to 104 mm during September, October, and November.

Gizzard shad spawn in spring and summer (Scott and Crossman 1973) in lakes and rivers (Miller 1960). No fish in spawning condition were observed in our study area. Only one larva was collected during the 10-year period. These data suggested no gizzard shad spawning took place in the vicinity of the Cook Plant. However, the Grand River (unpublished data, Great Lakes Research Division) and, presumably, the St. Joseph River were utilized for spawning and a nursery by gizzard shad.

Gizzard shad prefer warm water. Reutter and Herdendorf (1974) reported a temperature preferendum of 20.5°C for Lake Erie fish. In our study area, most fish occurred in relatively cool water, 13 to 21°C. Gizzard shad less than 300 mm were generally found in cooler water than larger individuals.

Johnny Darter

Larval johnny darters (4 to 8 mm) were occasionally collected in the study area during June, July, and August. They occurred mainly in open water, suggesting that spawning took place outside the beach zone. Adults and yearlings moved inshore in April, and abundance peaked in May or June. Smaller catches were observed in summer due to offshore migration. Some adults and yearlings returned to inshore water in the fall. Relatively large johnny darter catches in fall resulted in the trawl. Johnny darters were caught mostly at 6 and 9 m; some were seined in the beach zone.

Ripe adults were collected from May to September. The spawning peak, however, took place during June and July. Eggs were found on the plant's riprap during May and June (Dorr and Jude 1980). A similar spawning season was reported near the Campbell Plant (Jude et al. 1981b). Most fish collected were within the 40- to 60-mm length intervals. Published data on length-at-age (Raney and Lachner 1943, Karr 1963, Brazo and Liston 1979) indicate most darters we collected were 1-, 2-, and 3-year-olds. Larger fish in the 70- and 80-mm intervals were possibly 4 years old. Young-of-the-year attained a modal length of 20 mm in July and August, and grew to modal lengths of 40 or 50 mm in October and November. Brazo and Liston (1979) collected young-of-the-year 20 to 25 mm near Ludington in August. Johnny darters were collected in a wide temperature range, with most found in 9 to 21°C water. Young fish were collected in warmer water than older fish.

Lake Trout

During spring and summer, lake trout resided in water deeper than Cook Plant sampling stations; they migrated inshore only during upwelling. Catches were generally low from April to August. Adult fish moved inshore as early as September if water was cool. Peak catches generally occurred in November. Most fish were caught at 6 and 9 m, but a few were also taken in the beach zone.

Lake trout collected ranged up to 874 mm; most were 605 to 755 mm. Length-frequency distributions did not change noticeably from year to year. Immature fish apparently did not move inshore with spawners, as we collected very few young individuals. Yearlings, 122 to 180 mm, were occasionally trawled during summer.

Spawning took place in early November, as indicated by the number of spent fish observed in November catches. At more northerly Lake Michigan sites, Ludington (Brazo and Liston 1979) and the Campbell Plant (Jude et al. 1982), lake trout spawned up to a month earlier, probably because water cooled sooner. Larvae and juveniles were collected near the riprap at the Campbell Plant (Jude et al. 1981a, 1982), yet we found no evidence of successful reproduction in the Cook Plant vicinity. However, in November 1976, large numbers of eggs were washed up on the beaches at the Cook Plant. No larvae have been collected in our study area. About 2% of lake trout collected at the Cook Plant had no fin clips. All non-clipped fish were adults > 550 mm.

Most lake trout were collected at water temperatures 6 to 16°C. This is similar to temperature preferenda of 10 to 11.8°C reported by McCauley and Tait (1970), Scott and Crossman (1973), and Spigarelli (1975). About 14.4% of the lake trout caught near the Cook Plant had lamprey scars, a lower rate than found by McComish and Miller (1975) or Jude et al. (1982), 25% and 17-36%, respectively. In our study area, lake trout with sea lamprey scars decreased in frequency from 1973 to 1978, then increased again in 1979. Most scars were healed, and few fish had multiple scars. *Acanthocephala* were rarely noted in the intestines of either adult or immature fish.

Longnose Dace

Longnose dace caught in standard sampling ranged from 20 to 120 mm. Young-of-the-year collected in October and November were in the 20- to 40-mm length intervals. Yearlings (50 to 70 mm) and adults (> 75 mm) occurred from April to November but were most common in fall. Longnose dace occurred at all depths sampled; highest catches were observed in the beach zone.

Longnose dace spawn on rocky substrate in shallow water (Gee and Machniak 1972). In eastern Lake Michigan, spawning takes place from May to late July (Brazo et al. 1978). There was a lack of both ripe adults and larvae during the spawning season which suggests that spawning took place outside the study area. Longnose dace were caught in water temperatures from 2 to 27°C, with over 50% being collected in water of 11 to 15°C. These data agree with those of Brazo et al. (1978) who found a substantial increase in catches when spring water temperatures reached 8 to 14°C.

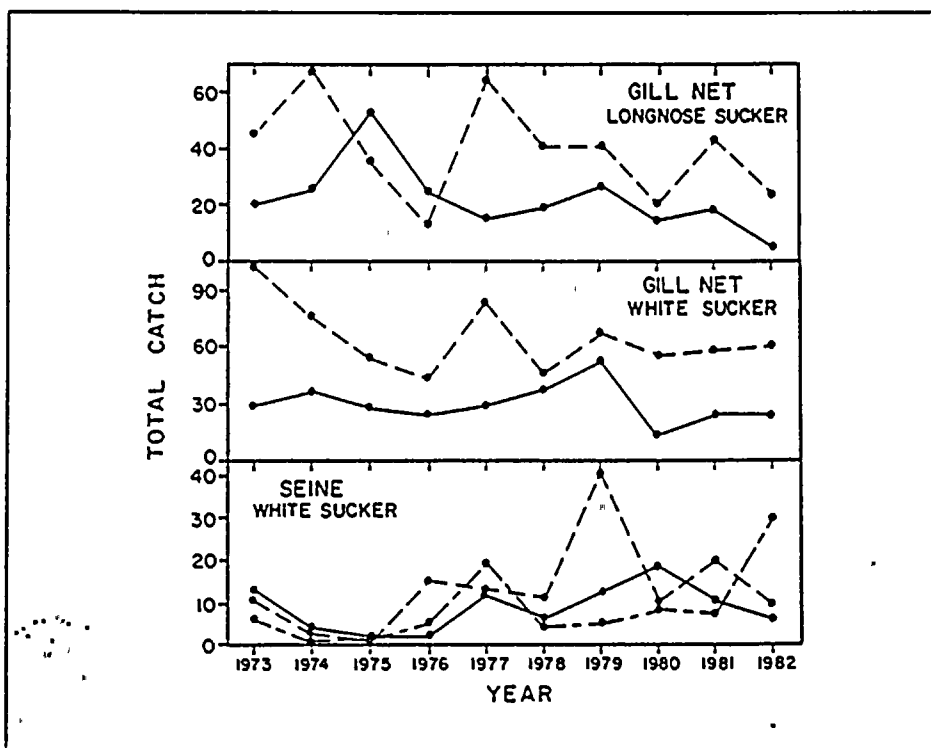


FIGURE 10. Annual catches of longnose suckers in gill nets and seines in the study areas of southeastern Lake Michigan, 1973-1982. The Cook Plant began operation in 1975.

Longnose Sucker

During most years, longnose suckers were slightly more abundant in spring and early summer than late summer and fall. This species may avoid inshore water during the warmest months. Most longnose suckers were caught in gill nets at 6 and 9 m. More were caught at Warren Dunes than at the Cook Plant (Fig. 10), a pattern of avoidance also observed for white suckers. Catches were higher at night than during the day, due to nocturnal inshore movements. During daylight, adults may be able to avoid nets or they remain in water deeper than 9 m.

Most longnose suckers collected were adults 400 to 570 mm; very few juveniles were caught. Young-of-the-year reached modal lengths of 90 to 110 mm in August. Yearlings were 170 to 240 mm in the fall. They spawned in April, probably in tributary streams rather than in the study area. Four sucker larvae (species undetermined) were collected in the study area during May 1978, but we presume that they were river-spawned fish which drifted into the lake from the St. Joseph River which is about 17 km north of the plant.

The majority of longnose suckers were caught in water from 7 to 15°C. Smaller

fish usually occurred in warmer water than larger individuals. Longnose suckers with neoplastic lesions (unnatural tissue growths, possibly cancerous) on lips and heads were occasionally found in the study area. However, there were no trends observed over the 10 years of the study.

Rainbow Trout

Rainbow trout were collected from April to November. They showed no distinct pattern of seasonal abundance, except that catches tended to be lower in summer than fall or spring. More rainbow trout were caught in the beach zone than in the open water. Rainbow trout collected ranged from 50 to 770 mm, most being from 120 to 270 mm. Great Lakes young-of-the-year rainbow trout grew to approximately 82 mm at the end of the first year of life (MacCrimmon and Gots 1972). In our study area, young-of-the-year reached lengths of 50 to 110 mm in October and November; yearlings ranged from 130 to 280 mm in spring. Rainbow trout larger than 300 mm were relatively uncommon in the study area. In the Great Lakes region, rainbow trout spawn from late December to late April (Dodge and MacCrimmon 1970). Ripe and spent adults were collected during April and May, suggesting that spawning took place during early spring in the vicinity of the Cook Plant.

Most rainbow trout were caught in water from 7 to 17°C. Juveniles were found in water of 10 to 20°C, whereas, individuals larger than 250 mm were caught in water of 3 to 18°C. Spigarelli and Thommes (1979) also found an inverse relationship between preferred temperatures and size of rainbow trout.

Slimy Sculpin

Slimy sculpins migrated inshore in spring and were most common in the study area during April and May. They were distributed at all depths sampled, but greatest catches occurred at 6 and 9 m. After June, catches became more sporadic as most adults returned to deeper waters. Wells (1968) also observed that sculpins abandoned shallow areas during summer and fall. Slimy sculpins collected were within the 20- to 120-mm length intervals; most were 60 to 90 mm. Spawning took place during April and May, probably in the rock riprap near the intake structures. Divers observed sculpin eggs on riprap in May (Dorr and Jude 1980). Larval slimy sculpins (6 to 9.3 mm) were commonly found in June during 1973-1982. They occurred only at open water stations.

Slimy sculpins are typically inhabitants of cooler areas in lakes and streams with rocky substrate (Scott and Crossman 1973). In southeastern Lake Michigan, Wells (1968) caught most sculpins in water from 4 to 6°C. The highest catches in our study area were observed in water from 4 to 7.9°C. Migration of sculpins to deeper water in the summer may be related to increasing inshore temperature. Sculpins were occasionally found in water 14 to 17.9°C.

Slimy sculpins were infected with acanthocephalan parasites during all operational years. Incidence of parasitism at Cook Plant stations ranged from 17% in 1975 to 63% in 1979. Higher frequency of occurrence of acanthocephalan parasites in slimy sculpins was observed near the Campbell Plant (Heufelder and Schneeberger 1980).

White Sucker

White suckers live in deep water in winter and migrate up streams to spawn in spring. In the study area, catches were generally lower in spring than in summer or early fall. Fish occurred at all depths sampled. Most juveniles were caught in the beach zone, whereas most adults were found at 6 and 9 m. More white suckers were collected at Warren Dunes than at the Cook Plant (Fig. 10), apparently due to avoidance of the Cook Plant area.

Spawning usually took place in April and May but extended into March or June during some years. No white sucker spawning was observed near the Cook Plant. Four sucker larvae (species undetermined) were collected in the study area during May 1978, but we presume that they were river-spawned larvae that moved into the lake. Most fish collected were adults 400 to 550 mm. Young-of-the-year, usually collected in summer and fall, were typically at modal lengths of 50 to 60 mm in July, but occasionally they were 70 to 80 mm. By October–December, young-of-the-year had grown to modal lengths of 120 to 130 mm. Yearlings were rarely collected.

White suckers were collected over a wide temperature range. Most occurred in water temperatures between 13 and 23°C. Smaller fish tended to be caught in warmer water than larger fish. Neoplastic lesions on the lips and heads of white suckers, first observed in 1974, were also found during operational years. No trends were discernible in the dataset. White sucker and longnose sucker were the only two species which were afflicted with this condition.

RARE SPECIES

Forty-two species of fish were classified as rare in the study area (Table 4). Of these, only four species, burbot, channel catfish, lake whitefish, and ninespine stickleback, were caught during every year of the 10-year period. Mean annual catches ranged from 0.1 to 17 fish per year. For 10 species, only one specimen was collected during the 10-year period.

SUMMARY

Fish populations were monitored in the vicinity of the D. C. Cook Nuclear Plant, southeastern Lake Michigan, during 1973–1982 using seines, trawls, and gill nets. Nets were deployed from April to November, during the day and night, once per month. Larval fish were collected on a similar schedule using 363- μ m-mesh nets. These data, along with ancillary data from entrainment and impingement monitoring, were compiled to describe the spatial and temporal distribution of all life stages of fish frequenting the inshore (<9 m deep) zone of Lake Michigan. We also documented which species of fish spawned inshore and which used the plant vicinity as a nursery area. Results were also used to determine if plant operation had any impact on fish populations by contrasting catch indices between the Cook Plant and a reference site (see Plant Impact on Fish chapter). We also drew on the data set generated here to assist in interpreting the entrainment and impingement trends that we observed.

A brief discussion of historical changes in Lake Michigan fish populations will help put our results in historical perspective. The Lake Michigan fish community

TABLE 4. Total catch of rare species and number of years during 1973-1982 in which these species were collected at least once in Cook Plant study areas, southeastern Lake Michigan.

Species	Total Catch	Number of Years	Species	Total Catch	Number of Years
Burbot	77	10	Lake herring	3	3
Ninespine stickleback	113	10	Rock bass	4	3
Channel catfish	100	9	Round whitefish	8	3
Lake whitefish	53	9	Central mudminnow	3	2
Sand shiner	168	9	Common shiner	3	2
Bluegill	76	8	Freshwater drum	19	2
Emerald shiner	148	8	Green sunfish	7	2
Silver redhorse	25	8	Smallmouth bass	2	2
Northern pike	60	7	Spotfin shiner	5	2
Golden shiner	9	6	White crappie	6	2
Largemouth bass	8	6	Banded killifish	1	1
Mottled sculpin	57	6	Blackchin shiner	1	1
Fathead minnow	7	5	Black crappie	1	1
Quillback	14	5	Blacknose dace	1	1
Shorthead redhorse	18	5	Blacknose shiner	1	1
Black bullhead	8	4	Creek chub	1	1
Bluntnose minnow	4	4	Grass pickerel	1	1
Lake sturgeon	5	4	Lake chub	1	1
Brook silverside	3	3	Logperch	2	1
Golden redhorse	13	3	Pumpkinseed	1	1
			Walleye	1	1

has undergone some of the most dramatic upheavals of any large body of water in the world. It is important to be aware of these changes to understand our results. Prior to the 1900s, the Lake Michigan fish community was dominated by coregonine fish, which occupied all parts of the lake from the deep abyss to the nearshore shallows. Lake trout and burbot were the only top predators. Deepwater sculpin were abundant in the deepest parts of the lake while emerald shiner, yellow perch, and trout-perch were common near shore. Overfishing began to impact the large species, lake whitefish, blackfin cisco, and lake sturgeon, in the 1900s. By the 1920s, some stocks were severely depleted which reduced their capacity to resist any further stresses. Thus, when sea lampreys entered the lake, they also preyed on the largest species, and along with overfishing caused the decimation of six of the seven deep-water ciscoes plus lake trout.

This major loss of predators and coregonines set up the lake for invasion by marine species, such as the alewife and rainbow smelt, which do not proliferate in the presence of an established predator population (Christie 1974). Alewife populations peaked in 1966, when massive quantities of these fish died and littered the beaches. This was the same year the Michigan Department of Natural Resources first stocked salmonids, which have produced a spectacular sport fishery. With the proliferation of alewife populations, we have seen the once abundant emerald shiner populations all but disappear. Yellow perch populations also declined from pre-alewife days. Predation by alewife on eggs and larvae on these and other species is the suspected mechanism causing the declines. In the late 1970s and early 1980s, we

have seen the reversal of these trends as alewife populations declined due to salmonid predation and cold winters. Yellow perch, bloaters, and rainbow smelt populations rebounded as a result. Most recently, emerald shiner populations also appear to be making a comeback.

In summary, our studies during 1973-1982 covered the period in Lake Michigan when alewives were very abundant, however they began to decline as cold winters and excessive salmonid predation began to impact their numbers substantially. Rainbow smelt were very common, along with trout-perch, spottail shiner, and yellow perch. Stocked salmonids were common to abundant in the study area during spring, fall, and upwellings when inshore temperatures were optimal. Toward the end of the study, as the alewife population declined, yellow perch populations, rainbow smelt, and bloater populations rebounded in our catches and these species were also impinged in larger quantities.

Three physical factors, all related to temperature, appeared to have substantial impacts on year-class strength or distribution of fish, and hence our catch rates for those species. The duration and intensity of inshore heating in a given year, especially during the spring spawning period, had a positive effect on survival of eggs and larval fish of several species, especially spottail shiner. Growth of most species that resided nearshore, such as spottail shiner, alewife, rainbow smelt, and yellow perch, was also generally positively correlated, but not always statistically, with years of consistently higher than average water temperatures. Timing of warming and cooling of Lake Michigan in spring and fall also had an impact on the numbers and species of fish that were collected during these volatile periods of the year. Mixed catches confounded statistical comparisons during these times (see Plant Impact on Fish chapter).

The second major factor influencing our catches was upwellings during periods of stratification. Our catches during upwellings would be mostly cold water species, such as lake trout and bloaters, while areas not affected would yield catches of predominantly warm water species. Several years of data and statistical manipulations were necessary to overcome the variability introduced into data sets because of these occurrences.

The third factor influencing the congregation of fish was the thermal bar which formed in the spring. Alewife and many other species were collected in unusually large numbers in this warmer, nutrient-enriched water along shore; large quantities of fish were also impinged.

Over the 10 years of the study, we collected 59 species of fish; six were considered abundant and included alewife, spottail shiner, rainbow smelt, trout-perch, yellow perch, and bloater. From 1973 to 1982, over 1,100,000 fish were collected; the largest catch occurred in 1973 (almost 200,000 fish) while the lowest occurred in 1982 (just under 50,000 fish). Seines produced 68% of all fish collected, while trawls (24%) and gill nets (8%) contributed the remainder.

Alewife adults first appeared inshore in spring, seeking the warmest water available. They spawned nearshore at night from June through August; larvae were abundant during these months, with highest densities generally in the 3-m depth zone.

Larval alewives were common from surface to bottom out to the 9-m depth contour and, depending on epilimnion depth, sometimes were present in substantial numbers out to 21 m. Because they were so widely distributed, especially in the vicinity of the plant intakes, they were entrained in large numbers. Young-of-the-year fish remained inshore until they reached a threshold size, around 50 mm, then

began migrating offshore. All young-of-the-year departed by late November or earlier depending on the thermal regime. Salmonids that we examined over the 10-year study were eating from 62-90% by weight alewife, an indication of how essential this forage fish is to the current Lake Michigan predator-prey system. The alewife population began to decline in the early 1980s and predators are now adjusting to changing abundances of prey species in the lake.

Bloaters spawn in deep water offshore in winter; unidentified coregonine larvae were only collected once in our sampling year during the study period. Young-of-the-year (35-117 mm) were collected in fall. Of all life stages, yearlings were caught most often in our study area; most were taken during upwellings in summer. We collected a few adults from 145 to 314 mm in summer and also during upwellings. Populations increased substantially from 1979 to 1982, which was linked to banning of commercial gillnetting and to the alewife decline in Lake Michigan.

Rainbow smelt spawn in April or May in the beach zone when water temperature there reaches about 10°C. Peak abundance of fish larvae was in May when densities to 420/1,000 m³ were reached. They move offshore soon thereafter. Young-of-the-year stayed within the 9-m contour when bottom temperatures were 8-18°C, but moved offshore when temperatures warmed. They began migrating offshore in September and completed their migration by November. Yearlings were most abundant in our adult fish catches, particularly in 1980-1982. They migrated inshore in spring, then moved offshore when water temperatures exceeded about 18°C. Many were collected during upwellings.

Catches of adult rainbow smelt were highest in 1973 and low during the middle 1970s with an increase in 1978-1982 concomitant with the alewife decline. Most adults were caught during the spring inshore spawning migration; they moved offshore soon after and were only collected during summer upwellings. Rainbow smelt was the second-most important fish by weight consumed by salmonids. They were also one of the few species we found which was infected with fungus.

Spottail shiner spawning occurs in ≤ m water usually during June but some continue until August. Larvae are demersal like the eggs and, because both life stages are less vulnerable to being eaten by the alewife, spottail shiner populations have not responded to increases or decreases in alewife abundance. Larvae were seldom entrained because of their nearshore distribution. Growth rates of young-of-the-year ranged from 0.31 to 0.58 mm/day during 1973-1982. Young-of-the-year moved offshore during fall and attained lengths of 45-64 mm. The year of highest growth was associated with highest water temperatures.

Yearling spottail shiners moved to nearshore areas in spring, then offshore in fall. Adults overwintered in 9-37-m water, and migrated inshore in March. They spawned, then dispersed to deeper waters starting in August. Myxosporidian parasitism was found consistently in this species (<4% incidence), usually in June, presumably associated with spawning and increasing water temperatures. Few spottail shiners were eaten by salmonids. We did observe a slight decline in spottail shiner populations in 1982 which may have been related to increased yellow perch predation.

Trout-perch spawned over a prolonged period, May to October; few larvae were ever collected. We did not catch many young-of-the-year in the inshore zone. Peak abundance of yearlings was in July in 6-9-m depths. Adults made up the highest proportion of our catches (67%). They moved inshore in April, stayed at 6-9 m during June-August, and migrated offshore by November.

Yellow perch spawn in June, with maximum densities of larvae in late June-July.

Another cohort of yellow perch was collected in April, which we believe entered from connecting water bodies. Young-of-the-year were recruited to our gear in August–November; they were abundant from the beach zone to 9 m. Yearlings migrated inshore in June with large catches in the beach zone in July; they moved offshore starting in August. Adult yellow perch were at depths >9 m during fall, winter, and spring. They moved inshore in June and remained steady, although repressed by the alewife, from 1973 to 1979. Abundance increased dramatically in 1980–1982, which we attributed to slackened predation by alewife on larval yellow perch. Hagar (1984) has seen increased predation on yellow perch by salmonids.

Twelve species of fish were designated as common and included the five salmonids, white and longnose suckers, common carp, gizzard shad, johnny darter, longnose dace, and slimy sculpin. The small salmonids we collected were generally stocked and remained in beach zone water throughout the summer months. Adults were captured nearshore in spring, fall, and during upwellings. Large numbers of lake trout were captured inshore during their spawning season in November. Common carp were abundant throughout the summer in the vicinity of the heated discharge which had apparently attracted them. Spawning was also documented for this species in the plant vicinity, but not at the reference site. Gizzard shad were usually collected in fall, sometimes in large numbers. They were suspected to have come from the St. Joseph River north of the plant where they were abundant. Johnny darters and longnose dace were collected in the study area in limited numbers. We also collected the larvae of johnny darters, documenting spawning by this species in the area. The two sucker species were absent from the study area during April and May, probably spawning in nearby rivers. Their catches were stable over the study period and these fish were year-long residents of area. We did establish that these species had an aversion to the plant site since we collected fewer at the Cook Plant site than at the reference site. Slimy sculpins spawned near shore during April and May, then moved offshore with increasing water temperatures. This species was attracted to the plant's riprap where spawning occurred. They also resided there during most of the year, living among the crevices of the riprap covering the intake and discharge structures.

An additional 41 species were considered rare in the study area. Some of these species included burbot, lake sturgeon, lake herring, and round whitefish which historically were important members of the fish community. Other species appeared to be strays from rivers and lakes connected to Lake Michigan, and included such species as bluegill, fathead minnow, northern pike, rock bass, and central mudminnow. We also collected one walleye.

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LAKE AND SHORE ICE CONDITIONS ON SOUTHEASTERN LAKE MICHIGAN

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INTRODUCTION

As part of the environmental studies relative to the Donald C. Cook Nuclear Plant located on the southeastern shore of Lake Michigan (Fig. 1), the ice conditions on the lake were monitored to understand the effects of the Cook Plant on the nearshore ice complex. The monitoring consisted of: regular observation of the ice using a high oblique time lapse photographic system, regular field observation of the ice to correlate ground truth with the photographic observations, and regular overflights during the ice period on southeastern Lake Michigan.

Ice on the Great Lakes has been a popular topic for investigation. This particular study attempted to ascertain the mechanics of the ice complex using a regular systematic monitoring program over a sufficiently long period of time to observe the ice. Earlier investigators were often hampered by the short term or irregular nature of the observations and hence were usually restricted to qualitative investigations of morphology or process. Some investigations (Heap 1963, Heap and Noble 1966) were dependent on the expertise of various shore-based observers at numerous stations over a large geographical area, while others (Marshall 1966, 1978) based their observations on aerial photographic reconnaissance. Such data were gathered for the gross evaluation of Great Lakes ice distribution and provided a significantly better understanding of the distribution and duration of the Great Lakes ice field.

The relationship between nearshore ice and nearshore morphology has been examined by Evenson (1973), Evenson and Cohn (1979), Seibel et al. (1975, 1976), Marsh et al. (1973), Fahnestock et al. (1973), and Ayers and Yocum (1972), while the genesis of shore ice in response to various climatological, geomorphological, and hydraulic parameters has been addressed by Evenson and Cohn (1979), Seibel et al. (1975, 1976), O'Hara and Ayers (1972), and Bryan and Marcus (1972). These last three research teams proposed that there was a correlation between the location of the ice ridges and the offshore sand bars. Seibel et al. (1975, 1976) demonstrated that relationship with measurements from the shoreline to the position of the breaker zone and that of the ice ridges and concluded that the position of the breaker zone and that of the ice ridges was coincidental.

In this paper, the author presents a summary of the information gathered over a 10-year period about the ice complex: the meteorological conditions that produce the nearshore ice complex as well as some observations about the interrelationship of the ice complex with the offshore breaker zone and off-shore bars. An analysis of the relationship between the climatological conditions causative to the formation and breakup of the nearshore ice complex is made. The ice complex study at the Cook Plant site also intended to ascertain whether the melthole produced by discharge water in the vicinity of the nuclear power plant impacted the nearshore ice complex. Whether the melting of the ice would be of a magnitude sufficient to

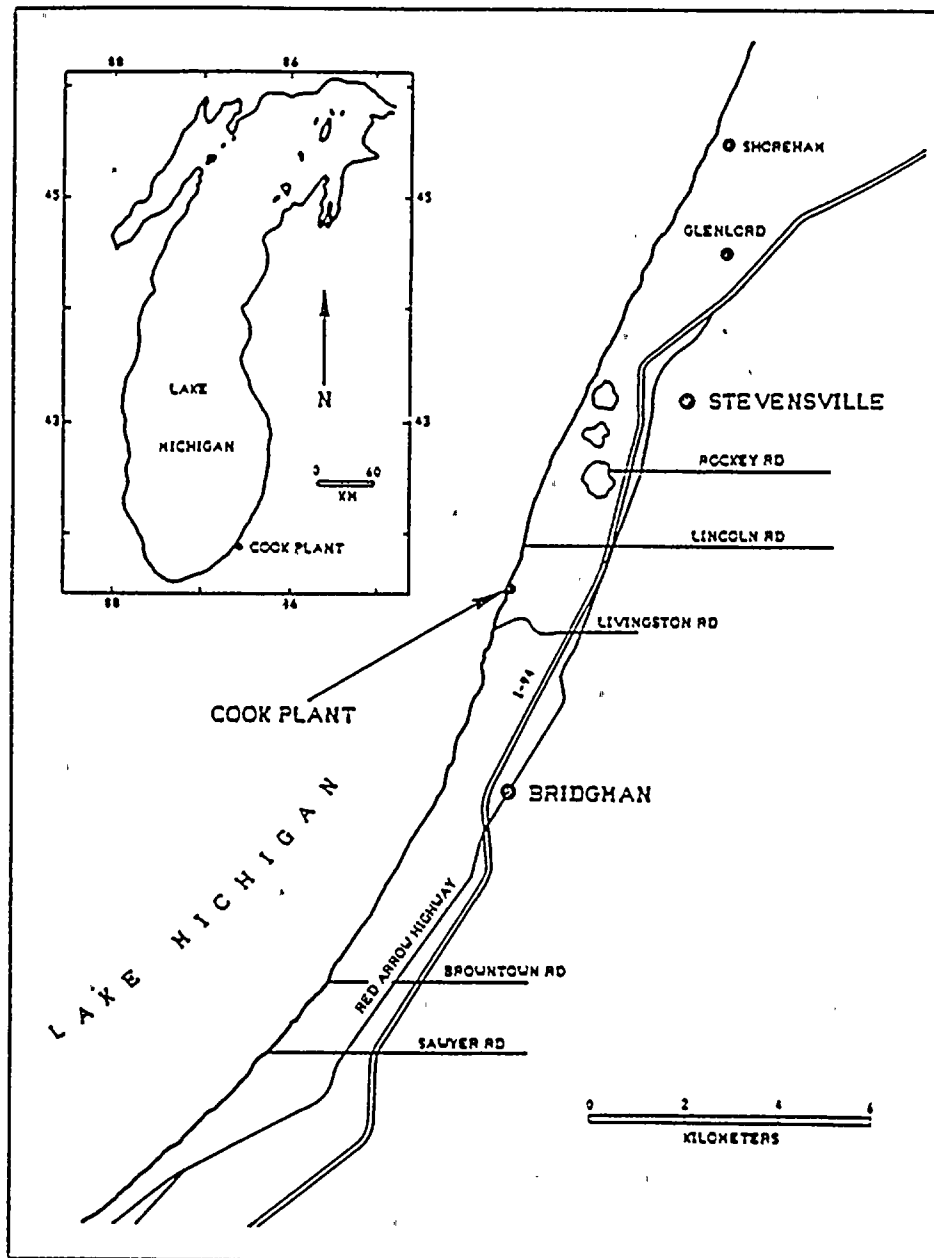


FIGURE 1. Location of the study site on southeastern Lake Michigan.

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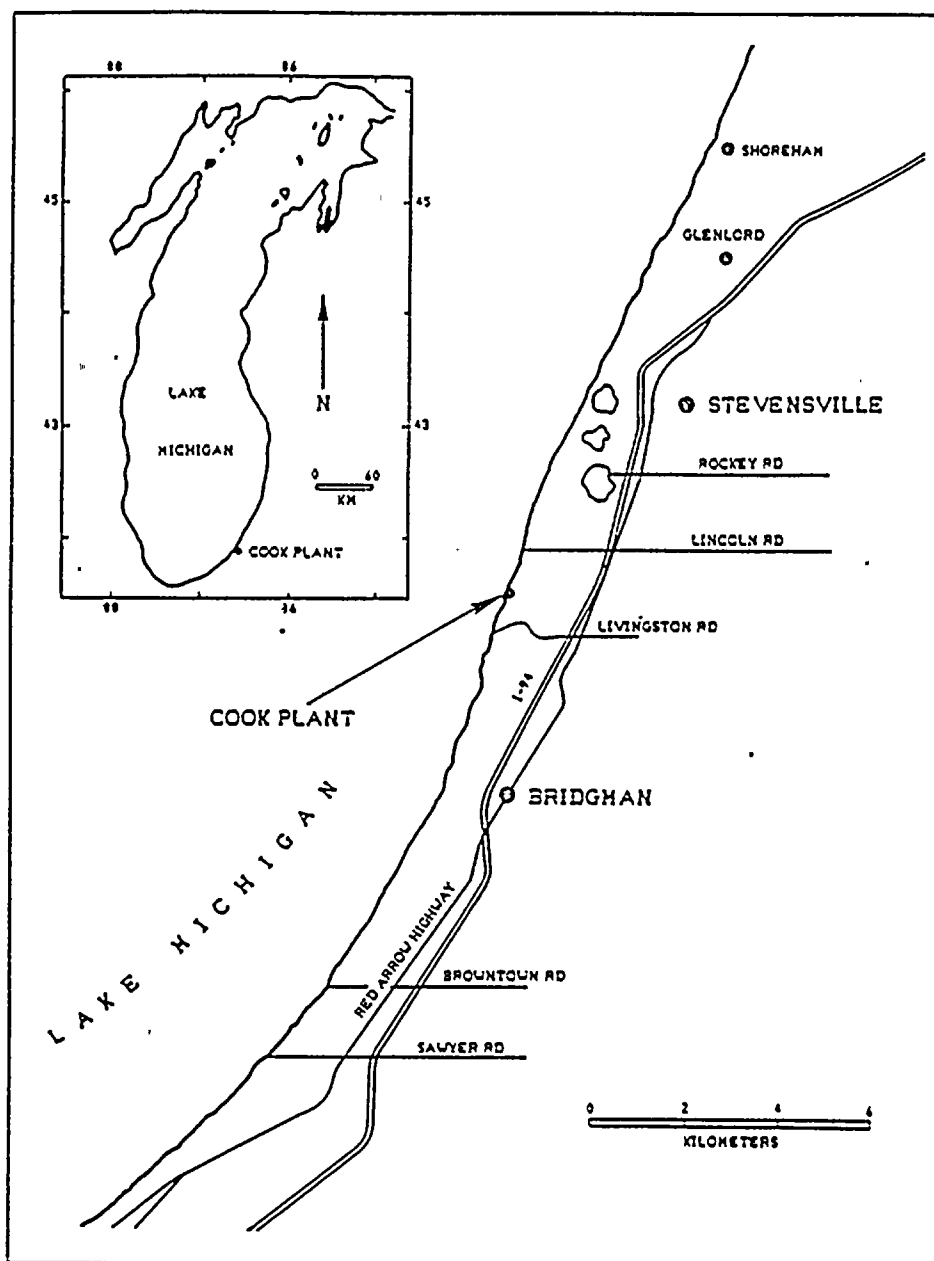


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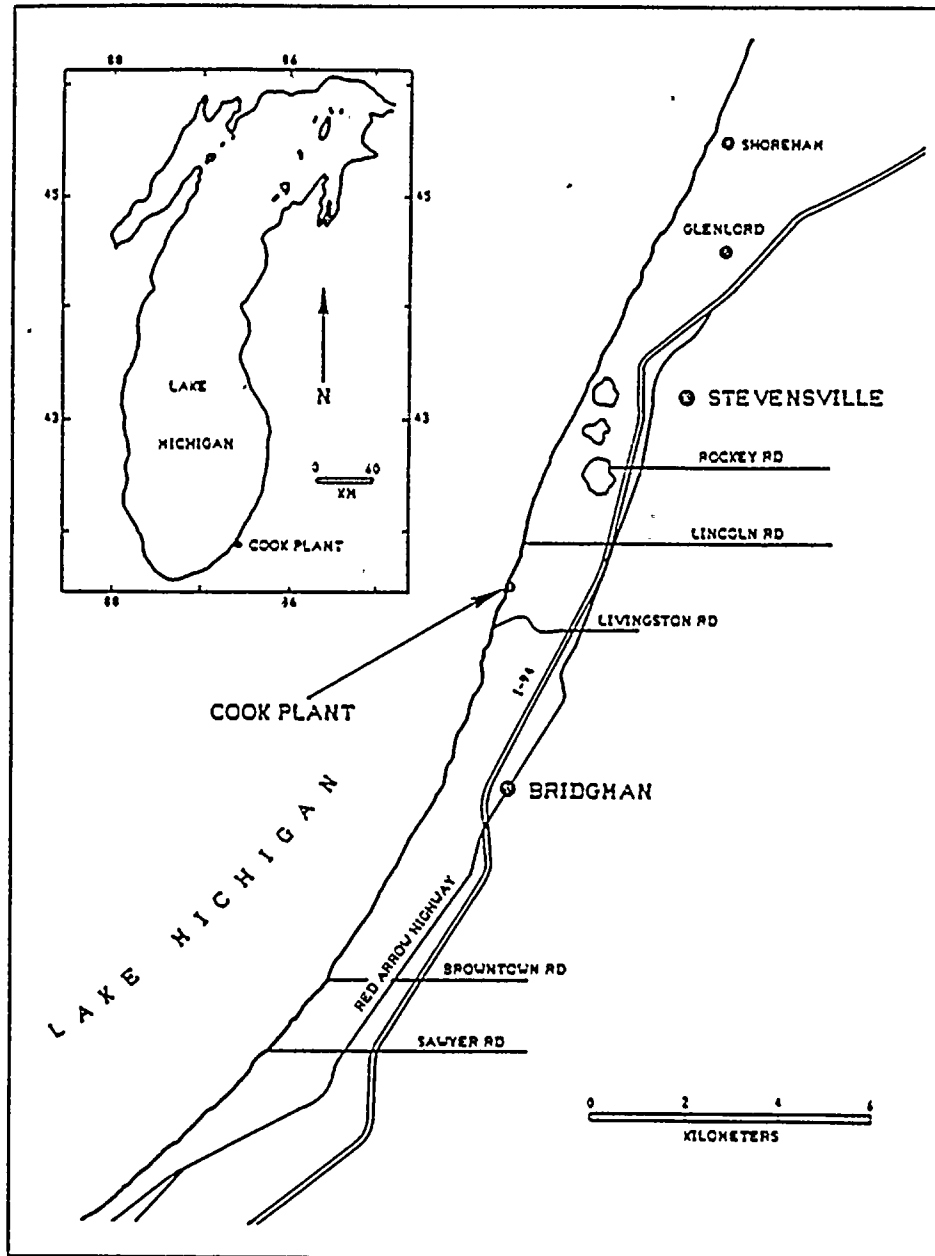


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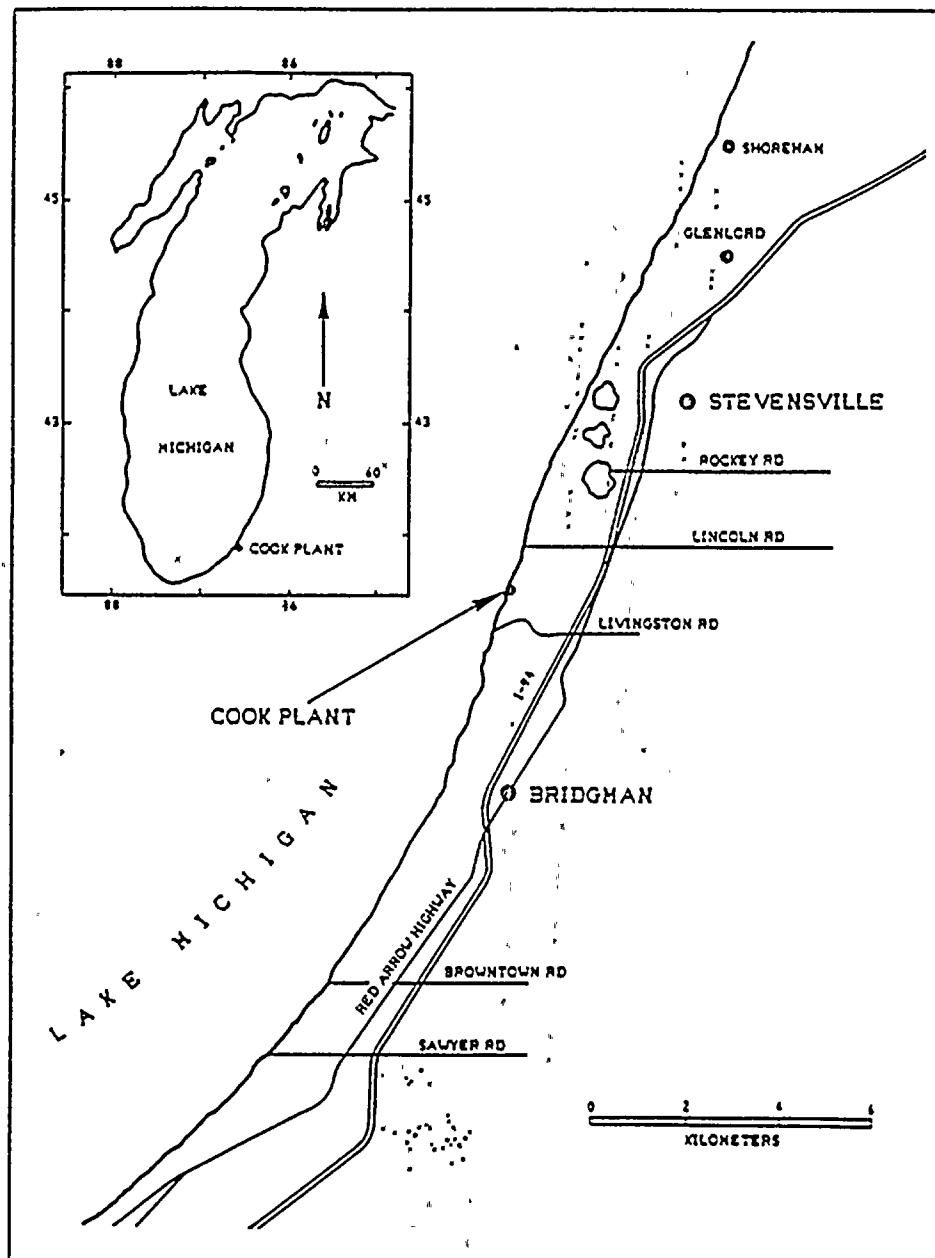


FIGURE 1. Location of the study site on southeastern Lake Michigan.

remove ice in front of the plant from the shoreline on outward was also of interest and is reported upon in this paper. Both quantitative and qualitative data are presented.

METHODOLOGY

Ice conditions were studied for a period of 10 years using field observations, airplane overflights, and a time-lapse photographic system installed at the site. The field observations were made on a regular basis and provided field truth for the time-lapse photographic system. The overflights provided a larger geographical overview of the shoreline ice so that comparisons between the ice conditions at the study site and other locations along southeastern Lake Michigan could be made. The main source of data on the formation and breakup of the ice at the study site was the time-lapse photographic system developed specifically for this study.

High oblique photographs were obtained for analysis with Canon F-1 single lens reflex 35 mm cameras with Canon 50 mm f1.8 lens. Two cameras with Canon EE Servo Finder and timing units were mounted indoors facing in a NNW direction. They automatically provided an average of five photographs daily from early morning to late afternoon. Because it was not possible to determine the exact time of exposure for each slide, a relative time was assigned to each photo: early am, late am, noon, early pm, and late pm. The positions of a sheet pile seawall and a range pole, evident in each photograph, were determined by standard field survey methods and were used as reference points.

The camera location permitted observation of isolated ice blocks as well as cross-sectional ice ridge profiles visible through wave-breached portions of the ridge system. In each slide the horizon was evident. Reflected glare from the lake surface, ice, or window glass did not interfere significantly with the quality of the photographs.

Slides were selected, instead of prints, for use in the analysis for two reasons. First, slides are less expensive to process than prints; and secondly, analysis required that the images of variables in the oblique photographs be marked on a tracing paper overlay. Slides were easily enlarged with a slide projector to minimize human error in selecting points from the image.

Analysis of the slides was divided into three steps. First, points were marked locating the positions of variable features in the projected oblique representation, and their coordinates were determined relative to an (x, y) coordinate system with origin at the principal point of the oblique photograph. Secondly, the positions of these points were calculated for the equivalent vertical photograph with coordinate system origin at its isocenter. Finally, the real ground coordinate positions of the points were determined relative to a coordinate system with origin at the camera. The geometry basic to this method of oblique photo analysis is discussed under *Geometry of oblique photographs*.

A fixed reference system was requisite for slide-to-slide comparisons. Establishment of such a coordinate system with origin at the principal point of the projected oblique photograph necessitated the removal of the color transparency from the developer's cardboard mount and remounting on a 2.0-inch plexiglass square as suggested by Maresca (1975). Inscribed on the plastic mount was a series of lines parallel to its borders to aid in placement of the positive. Also etched on the mount

were the fiducial axes whose origins would be located at the principal point of the remounted color transparency.

The remounted slide was projected by a Kodak Ektagraphic slide projector onto a wall covered with a tracing paper overlay. The projected image of the fiducial coordinate axes, now superimposed on the projected photograph, was located on the overlay. To minimize error due to remounting, the location of the projected principal point was checked and adjusted when necessary.

The perpendicular distance was measured between two of the inscribed horizontal lines on the plastic mount, one at the top and one at the bottom of the mount. The distance between the projected images of these same two lines was measured on the screen, and these values were used in the computer program for calculation of the focal length of the projector. The positions of the reference points common to all slides were marked on the overlay, and subsequently the projected positions of the variables unique in each slide were located. These variables included the horizon, breaker zones, ice ridges and interr ridge ice lagoons, zones of brash ice accumulations, heights of the icefoot, ice blocks, and ice ridges and the waterline when visible.

The computer program utilizes the mean still water level (MSWL) as the datum for ground coordinate calculations, thus requiring that the selected data points be located at this elevation. Points defining variable positions were therefore marked on the tracing paper overlay where their intersection with the water surface was evident in the projected picture. The points marking the location of breaker zones were placed at the base of the breaking waves. Points marked along the crest of an ice ridge, above the MSWL datum, would represent only an estimate of the ridge's position. Consequently, when determining ice ridge location, it was frequently necessary to use a later slide which depicted the breaching of or breakup of that ice ridge when its intersection with the water surface at midridge was visible. Points defining the outline of brash ice areas on the edges of the interr ridge ice lagoons were in most cases considered to be at the MSWL. The calculation of the height of a variable required two points of definition: one at the object's base where intersection with the water was apparent and the other marking its vertical extent. For the reference points, which were not located at the still water level, surveyed heights were specified in the program for determination of their ground coordinate location.

The distance from the principal point of the oblique photograph to the apparent horizon is utilized in the program for calculation of distances on the principal plane diagram; hence, the position of a poorly defined horizon may introduce errors. The quality of the horizon was evaluated as excellent, good, fair, or poor, thus offering one possible explanation of error in poor computer output values.

The determination of the (x, y) coordinates for the points locating the positions of variables on the projected oblique photograph required a coordinate grid. Coordinate values were read to the third decimal place from a cartesian coordinate grid with a scale of twenty divisions to the inch placed under the tracing paper overlay.

The computer program converts the oblique (x, y) coordinate points to coordinates for an equivalent vertical photograph and then calculates the real ground coordinates for each ordered pair. Printer plot subroutines, called in the main program, construct a ground coordinate grid with origin at the camera. The computed ground coordinates of each data point were scaled and plotted within the grid. A unique plotted character was used for each monitored variable. Characters representing the location of each variable were positioned in linear arrays on the printer plots and were approximated by straight lines to facilitate distance measurements.

The distances to all variable positions were calculated perpendicular to a baseline along the shore. The sheet pile seawall was chosen as the baseline because the waterline is subject to frequent changes in position and is usually not visible during periods of ice cover. The computer program does not calculate the distances from the baseline to the various variable positions. A reference point along the baseline was established and all distance measurements were made perpendicular to the baseline from that point. A normal to the baseline was constructed of sufficient length to intersect the lines defining the variable positions. This normal formed the hypotenuse of a right triangle. Determination of the x and y components of the baseline normal and application of the plotting scale factor and the Pythagorean Theorem yielded distance to the location of the variable. This technique was used to calculate the distances from the baseline to monitored variables.

GEOMETRY OF OBLIQUE PHOTOGRAPHS

The analysis of an oblique representation can be obtained in any standard reference or text but is presented here briefly so that the correction for the projection of the image through a slide projector can be included. The geometry of the principal plane diagram (Fig. 2) was used to derive the fundamental equations required to solve the oblique photograph problem. The distance, PT, is measured on the screen from the apparent horizon to the principal point of the photograph. Because this distance is projected onto a screen from a slider projector, the image distance must be corrected for the focal length of the slide projector. A constant, C, is used to make this correction. The constant, C, is defined as

$$C = I/O \quad (1)$$

where I is the image distance and O is the object distance. The image distance, I, is calculated from the lens maker's formula

$$I = O \times f_{proj} / O - f_{proj} \quad (2)$$

where f_{proj} is the focal length of the projector. The distances on the principal plane positive can now be defined. The angle of depression, Θ' , is defined as

$$\tan \Theta' = PT / f_{cam} \quad (3)$$

where

$$PT = PT_{screen} \times C \quad (4)$$

and f_{cam} is the focal length of the camera. The distance PT and the angle Θ' was calculated to the apparent horizon. A small correction can be made to determine Θ' to the true horizon by adding

$$\Delta\Theta' = 0.98 \sqrt{H} \quad (5)$$

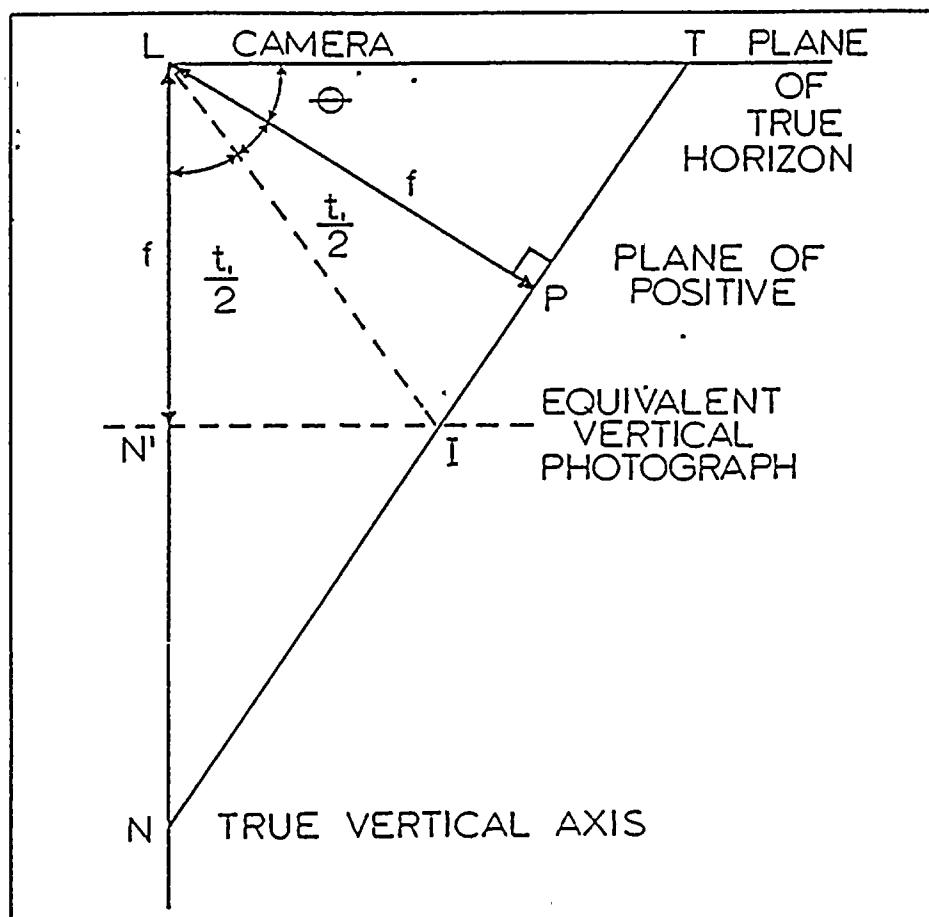


FIGURE 2. *Principal plane diagram.*

in minutes to Θ' where H is the difference in elevation between the water surface and the camera elevation. Because $\Theta = \Theta' + \Delta\Theta$, the distance PT must be recalculated. The geometry of the principal plane diagram (Fig. 2) may be used to derive the distance PI using

$$PI = (\tan t/2) \times f_{i,m} \quad (6)$$

where t equals $90 - \Theta$. The remaining distances in the principal plane diagram are given in equations 7, 8, and 9.

$$PN = f_{cm} \times (\tan t) \quad (7)$$

$$TI = PT + PI \quad (8)$$

$$TN = PT + PN \quad (9)$$

Using this information, all points in the oblique representation can be rotated into an equivalent vertical photograph and real ground distances can be obtained.

Individual ordered pairs, (X_{scr}, Y_{scr}) , are picked off the oblique photographic projection on the screen relative to a rectangular coordinate system with an origin at the principal point. The coordinates $(X_{v.p.}, Y_{v.p.})$ of the equivalent vertical photographs are calculated as

$$X_{v.p.} = X_{scr} \times C \times (TI / (TI - K)) \quad (10)$$

$$Y_{v.p.} = K \times (TI / (TI - K)) \quad (11)$$

where

$$K = (Y_{scr} \times C) + PI \quad (12)$$

The real ground coordinates (X_{grd}, Y_{grd}) can be calculated from a simple proportion between the equivalent-vertical photograph and the ground relative to a rectangular coordinate with the origin at the camera. The (X_{grd}, Y_{grd}) is calculated as

$$X_{grd} = H \times X_{v.p.} / f_{cm} \quad (13)$$

$$Y_{grd} = H \times (PI + Y_{v.p.}) / f_{cm} \quad (14)$$

The fundamental assumption basic to the analysis is that all points lie within the datum plane. If a point lies outside the datum, as for example a point on the beach, then a different elevation, H , must be used.

Vertical elevations or vertical heights were calculated from the formula

$$h = H \times \frac{d'}{r'} \times \frac{TN}{TS} \quad (15)$$

where d' is the height of the object on the screen, r' is the distance from the principal point to the top of the object in the oblique view, TN is the distance from point T to point N , and TS is the distance from the point T to the point S which is the base of the object.

Using the above formulae, a comprehensive computer program, ICESTUDY, was written to convert from the oblique to the real ground distances. The information in regard to the relationship between the nearshore ice ridges, the offshore breaker zone, and the offshore bars was determined using the above-outlined procedure.

A third camera used in this study was mounted looking in a westerly direction. The west-looking camera was established to monitor conditions in the vicinity of the melthole resulting from the plant's discharges of waste heat. Because of its orientation, some of the late afternoon photographs were of a reduced quality due to sunlight entering the lens.

STAGES OF ICE AT THE STUDY SITE

The terminology that has been applied throughout this study is that of O'Hara and Ayers (1972), which follows that recommended by Kivisild (1970). The initial stage of the shore ice development is the formation of a frozen beach. This is usually followed by the buildup of a small ridge, or icefoot, at or near the water's edge. The icefoot is the result of spray produced by small amplitude waves lapping against the frozen beach face and the accumulation of brash ice fragments. Subsequent to the development of the icefoot is the formation of a tabular ice lagoon which terminates at the first ice ridge. Numerous small, irregularly spaced undulations may be observed on the ice lagoon and are normally oriented roughly parallel to the lakeward edge of the accreting ice mass. Each undulation marks a short hiatus in the lakeward ice advance when freezing spray and minor amounts of wave-tossed brash ice accumulate on the advancing icefront. The first ice ridge is generally the smallest of the lake ice ridges. Like its more lakeward counterparts it has a steep, nearly vertical lakeward face and a more gently sloping landward surface. When the ridges deteriorate during the period of ice breakup it is not uncommon to observe grounded remnant ice blocks linearly oriented at the position previously occupied by the ice ridge. The ice complex at the study site can be defined by the patterned combination of these three features: the icefoot, the ice lagoon, and the ice ridge. A schematic representation of these features is shown in Figure 3.

Seibel et al. (1975) categorized the ice formation and breakup into distinct stages

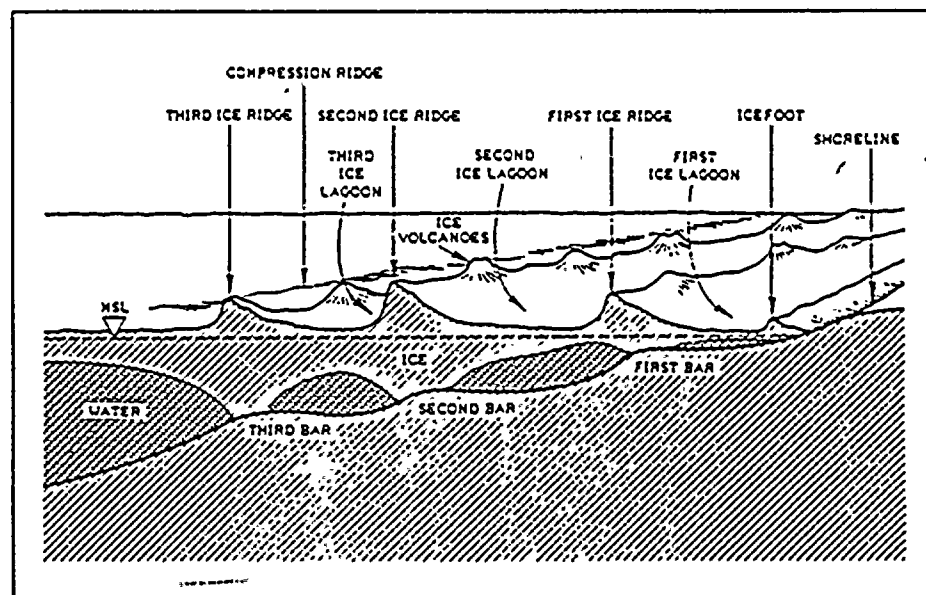


FIGURE 3. Schematic representation of the typical nearshore ice complex at the study site.

of accretion, stasis, deterioration, and breakup. Accretion is a visible addition of ice from the previous observation while deterioration and breakup are a reduction in the ice mass. Deterioration is a gradual reduction of the ice mass while breakup is a rapid destruction of the nearshore ice complex and is usually exemplified by the landward displacement of the ice complex under severe weather conditions. Static ice conditions imply that no changes were visible from the previous observation.

SUMMARY OF ICE AND ASSOCIATED METEOROLOGICAL CONDITIONS

One aspect of this study from its onset was to attempt to establish relationships between the ice stages and existing meteorological conditions. The stages of ice development were related to wind direction, wind speed, and air temperature. The meteorological parameters were collected at the St. Joseph harbor entrance. The data used in the plots were an average condition for the day. It was recognized that an average condition may introduce some error in the correlations but it was considered the best approach to take in the analysis intended. Seibel et al. (1975) looked at the surface water temperature and concluded that this parameter had the "least influence on the stages of ice development." To provide insights into the relationships between the wind direction, wind speed, and air temperature and stages of ice development at the study site, 2 years of data are presented and interpreted. The winters of 1978-79 and 1979-80 are used in the discussion that follows. Figures 4-14 are plots of the meteorological parameters against the ice development stages for December 1978 through March 1979, while Figures 15-26 are the plots for December 1979 through March 1980.

AIR TEMPERATURE AND ICE DEVELOPMENT

Seibel et al. (1975) found no relationship between ice development and surface water temperature, but the same was not true of air temperature. It was clear in their limited sample that accretion occurred when the air temperature dropped and the wind shifted to the west. Figure 4 provides an interesting example of how the ice complex at the study site on southeastern Lake Michigan appears to be initiated. The air temperature prior to 24 December was hovering around 0°C when it began to drop below 0°C and remained there for 4 days. Before 24 December there was no ice present, but an accretionary sequence occurred starting on 24 December and continued for 4 days. As the air temperature rose, there was a deterioration of the ice complex. Perhaps an even more explicit example of the relationship between ice development and air temperature is that of January 1979 (Fig. 6). Air temperature during January was extremely low and remained below 0°C for 20 consecutive days. During that period the ice either accreted or remained static, with the exception of 1 day when the ice showed deterioration even though the air temperature decreased, a peculiarity that will be explained below. After 20 January, the air temperature rise was accompanied by a parallel deterioration of the ice. It was observed that the ice response generally lagged behind the increasing air temperature by about 1 day. The trend just described appears to hold true for February 1979 (Fig. 9) as well.

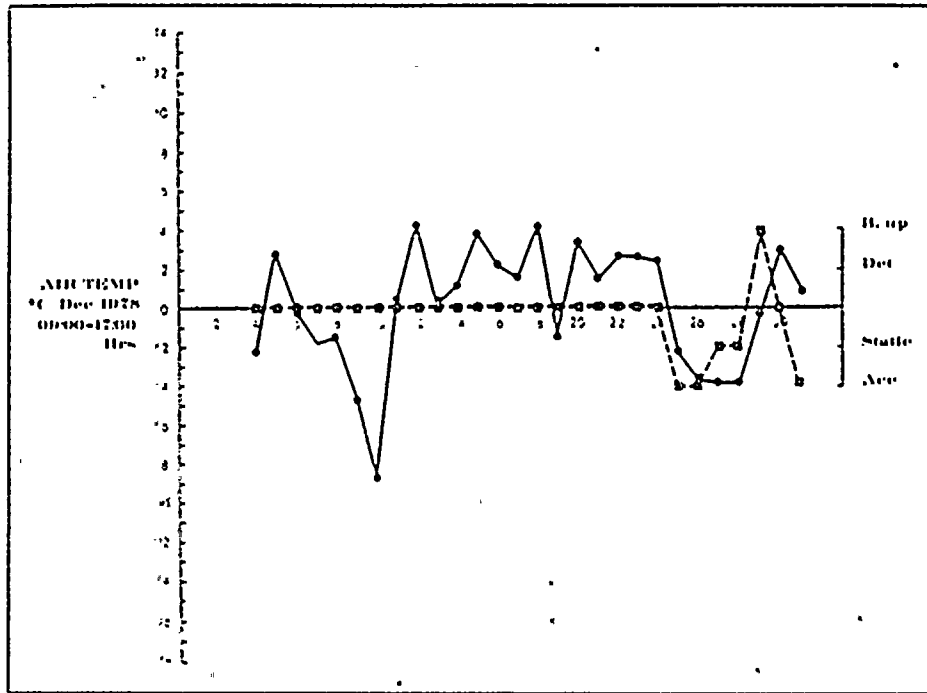


FIGURE 4. Plot of ice conditions and air temperature: December 1978.

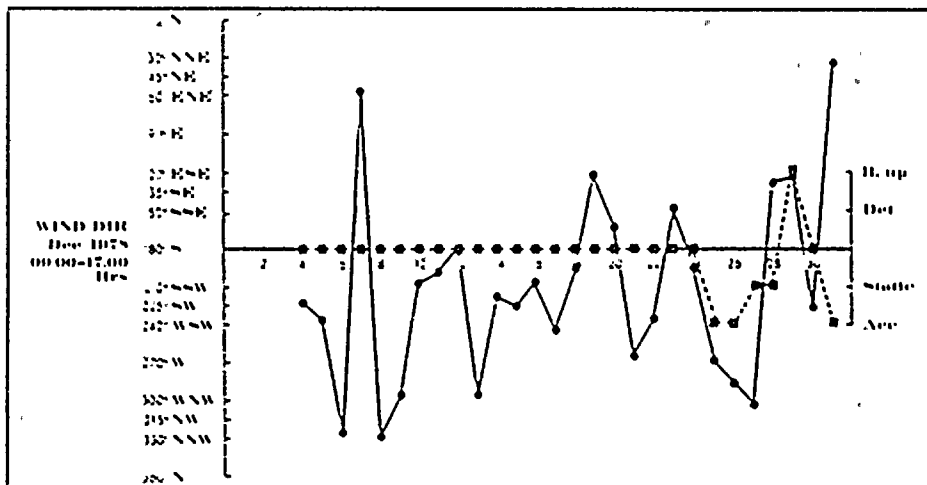


FIGURE 5. Plot of ice conditions and wind direction: December 1978.

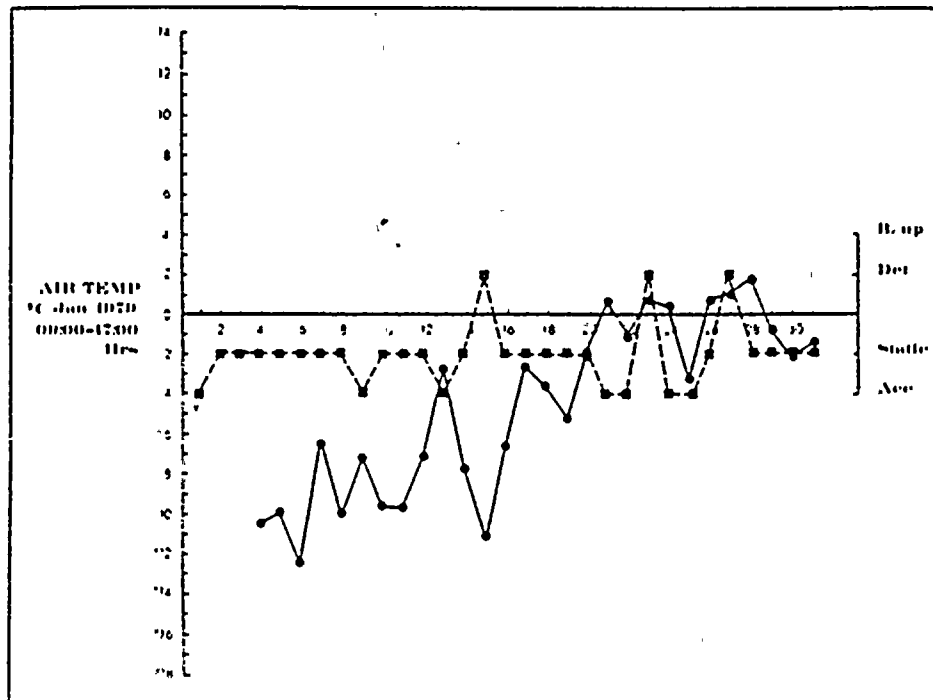


FIGURE 6. Plot of ice conditions and air temperature: January 1979.

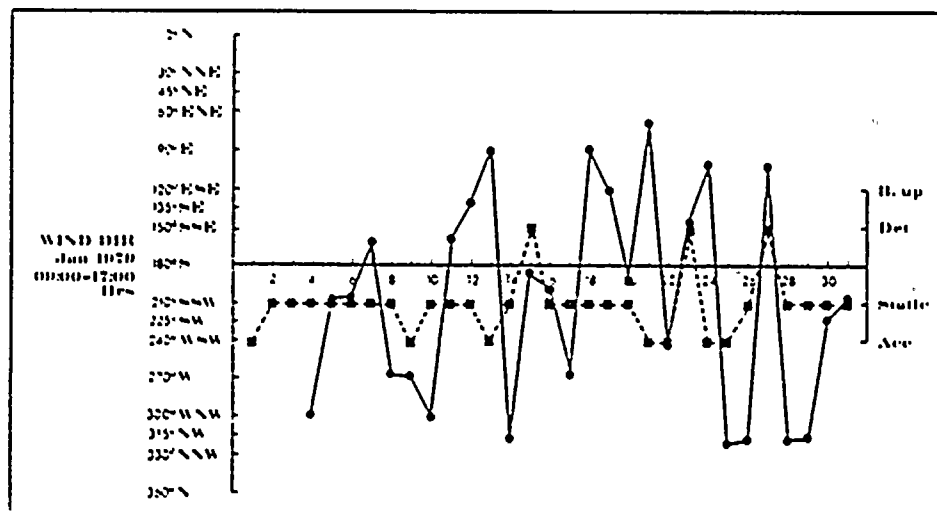


FIGURE 7. Plot of ice conditions and wind direction: January 1979.

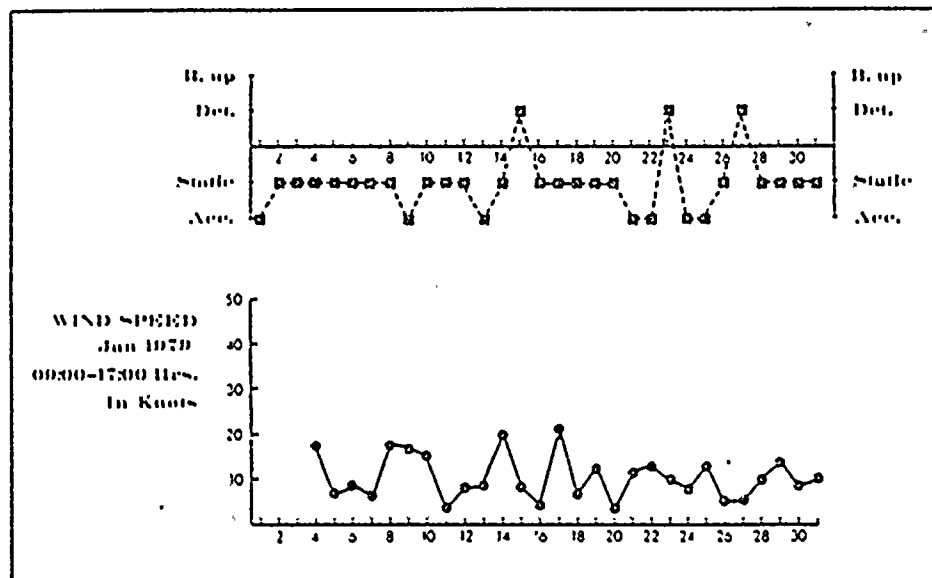


FIGURE 8. Plot of ice conditions and wind speed: January 1979.

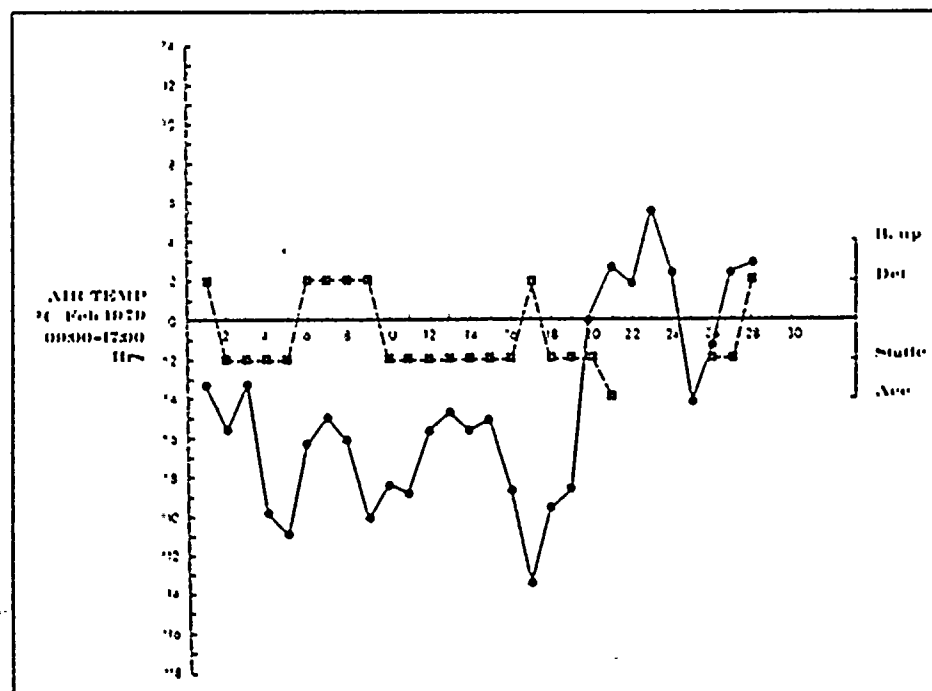


FIGURE 9. Plot of ice conditions and air temperature: February 1979.

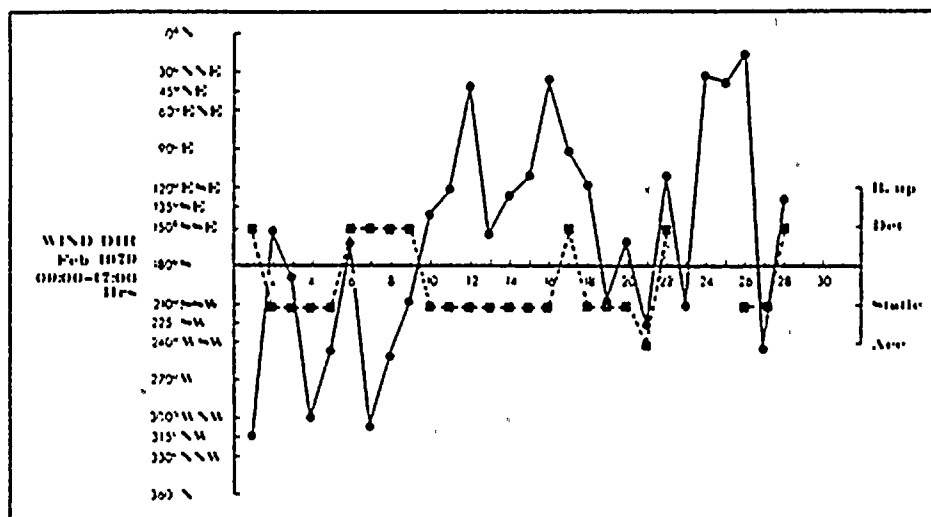


FIGURE 10. Plot of ice conditions and wind direction: February 1979.

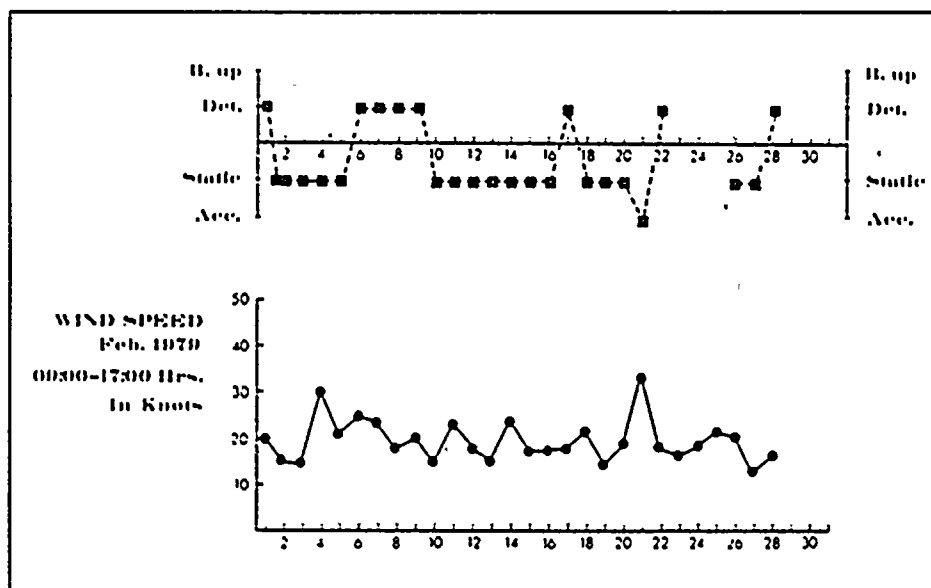


FIGURE 11. Plot of ice conditions and wind speed: February 1979.

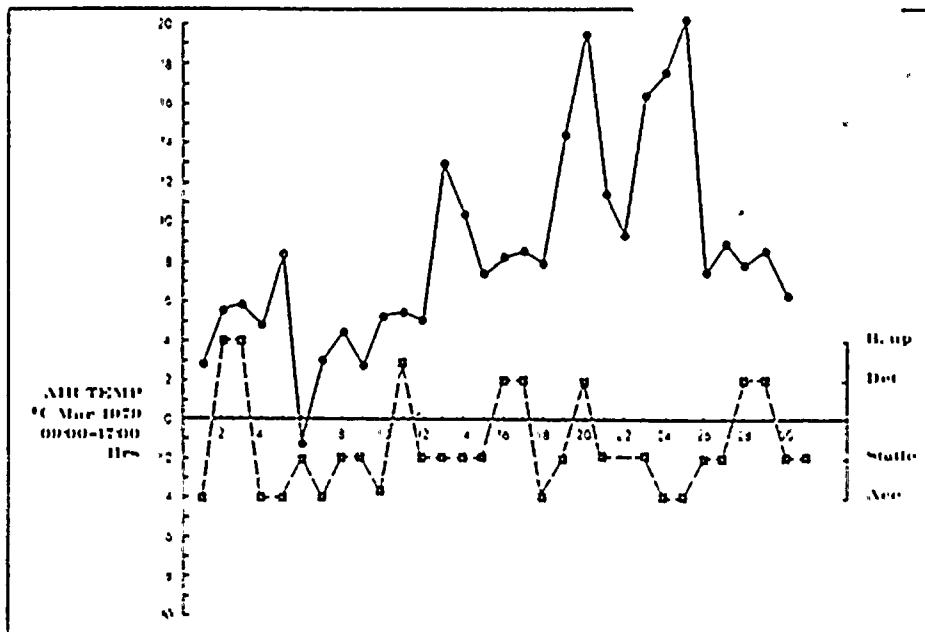


FIGURE 12. Plot of ice conditions and air temperature: March 1979.

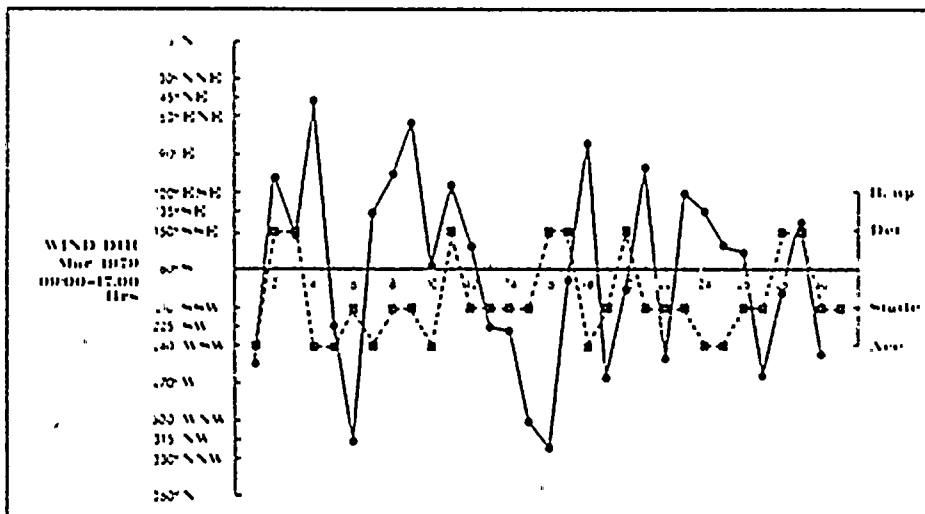


FIGURE 13. Plot of ice conditions and wind direction: March 1979.

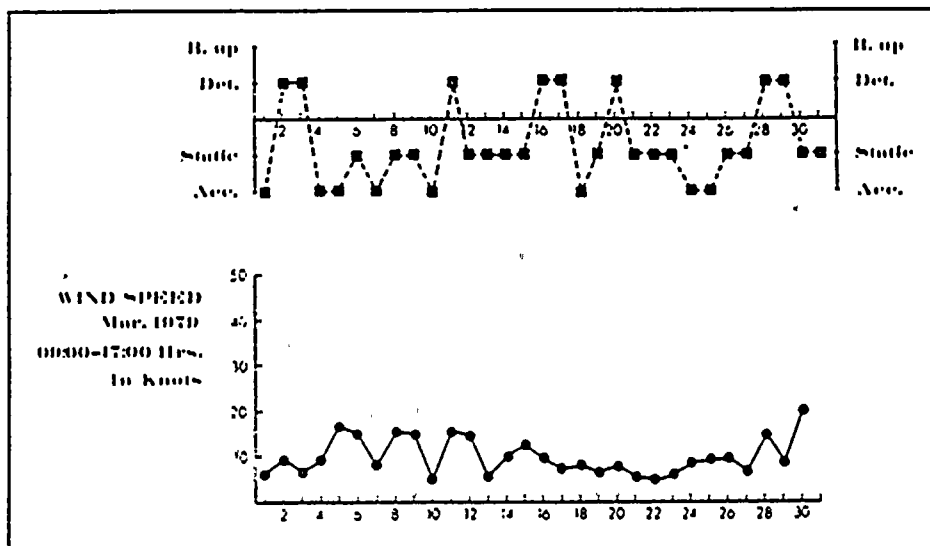


FIGURE 14. Plot of ice conditions and wind speed: March 1979.

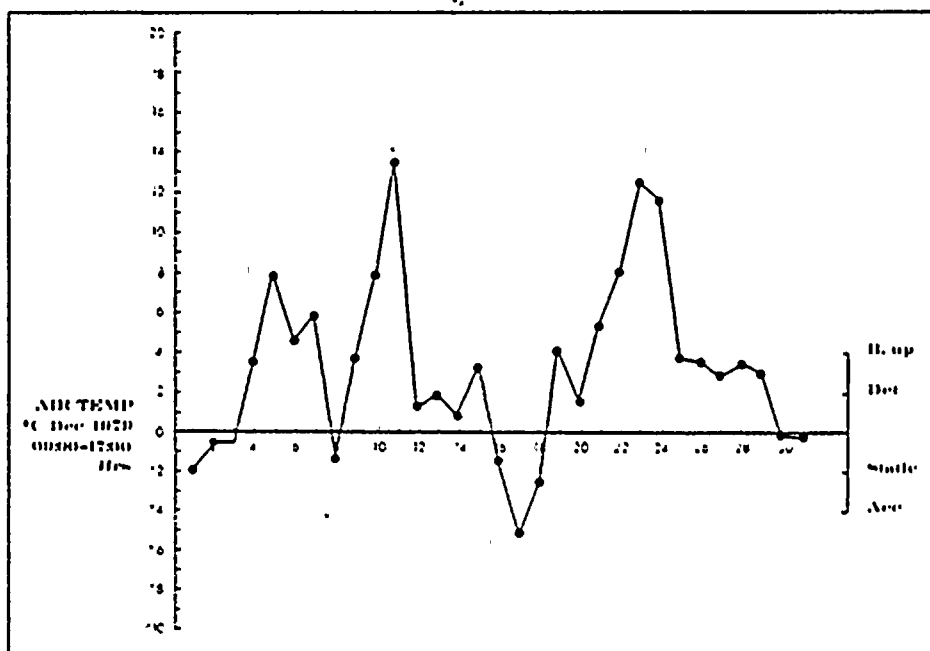


FIGURE 15. Plot of air temperature: December 1979.

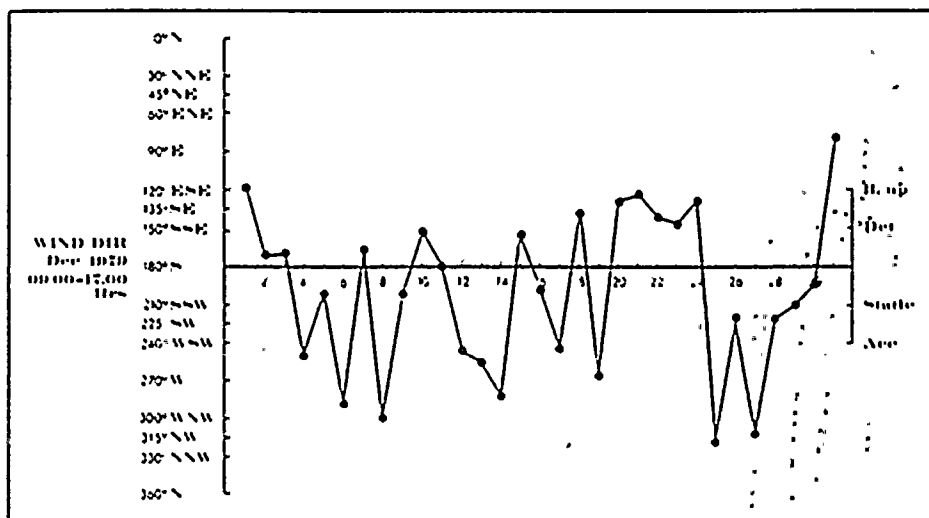


FIGURE 16. Plot of wind direction: December 1979.

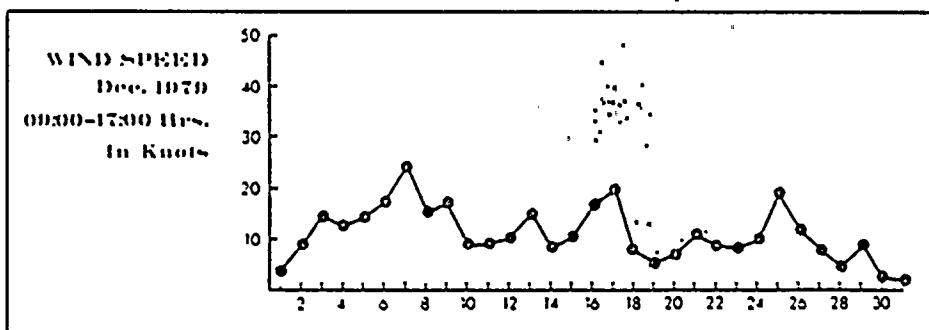


FIGURE 17. Plot of wind speed: December 1979.

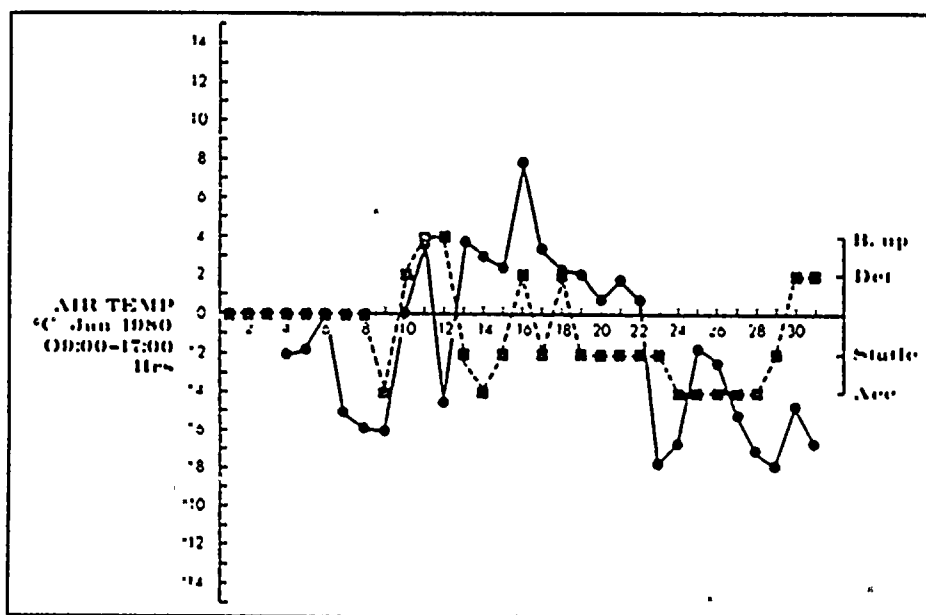


FIGURE 18. Plot of ice conditions and air temperature: January 1980.

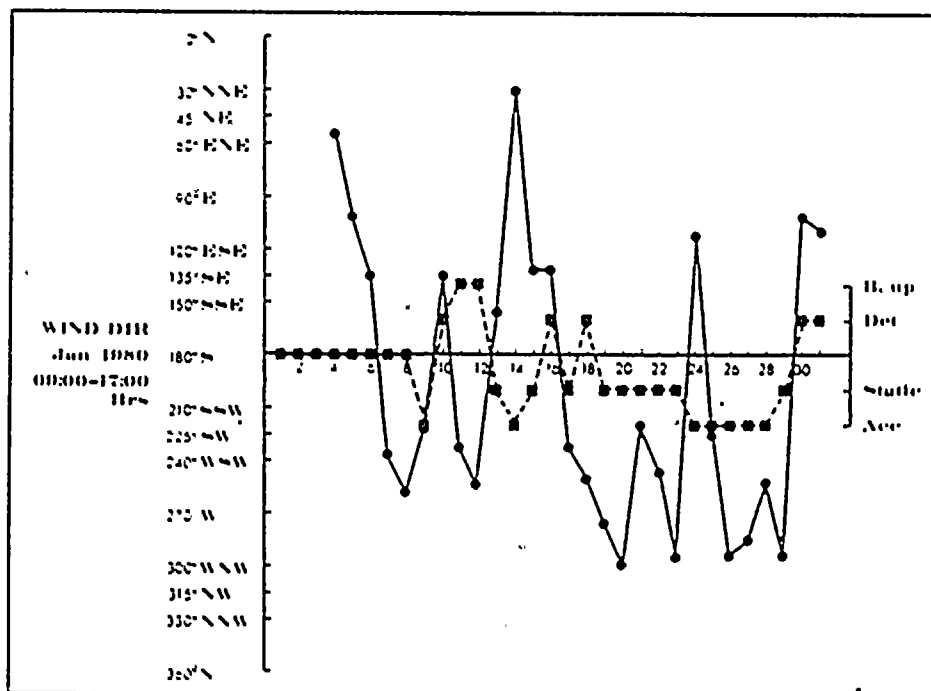


FIGURE 19. Plot of ice conditions and wind direction: January 1980.

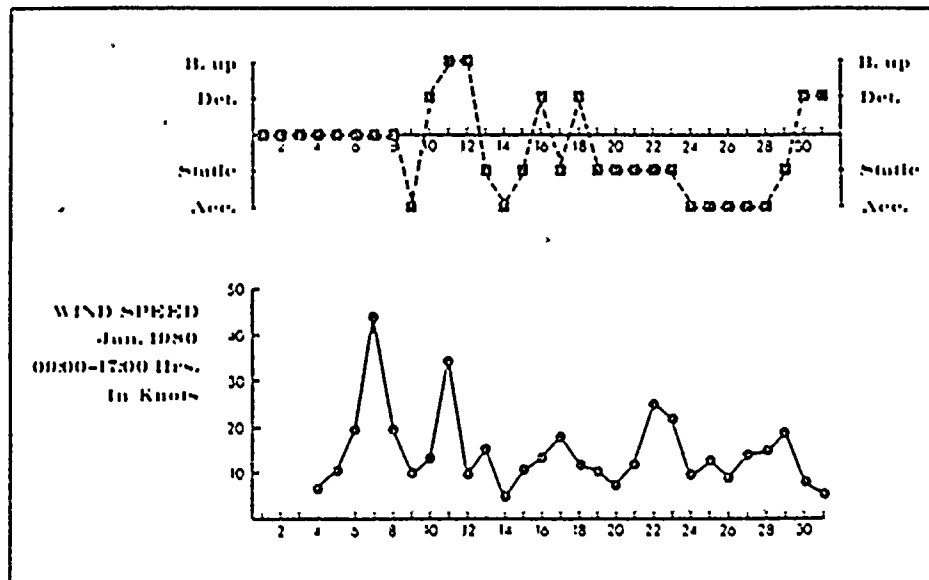


FIGURE 20. Plot of ice conditions and wind speed: January 1980.

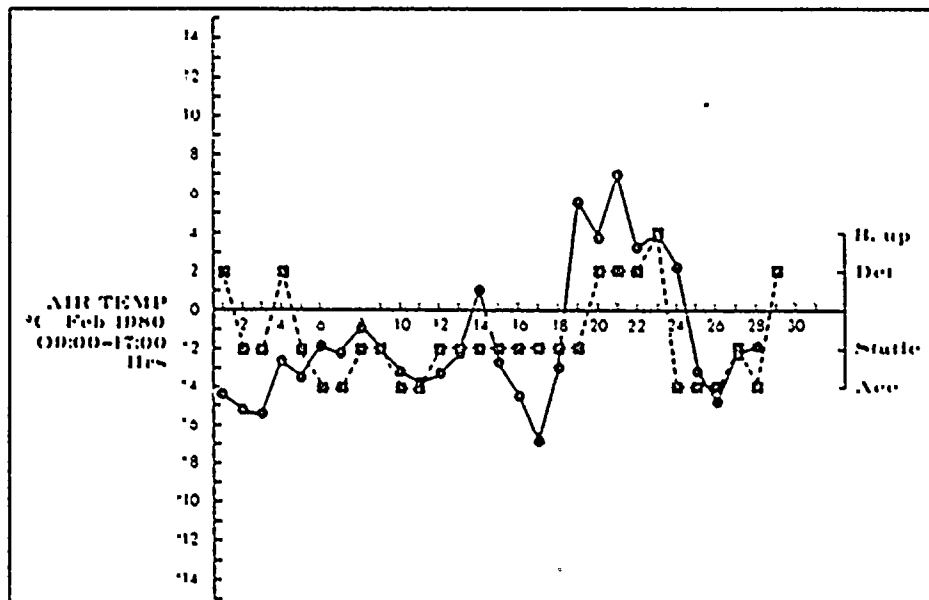
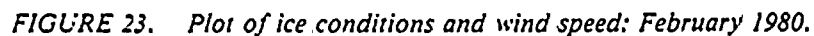
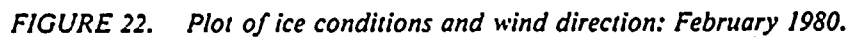


FIGURE 21. Plot of ice conditions and air temperature: February 1980.



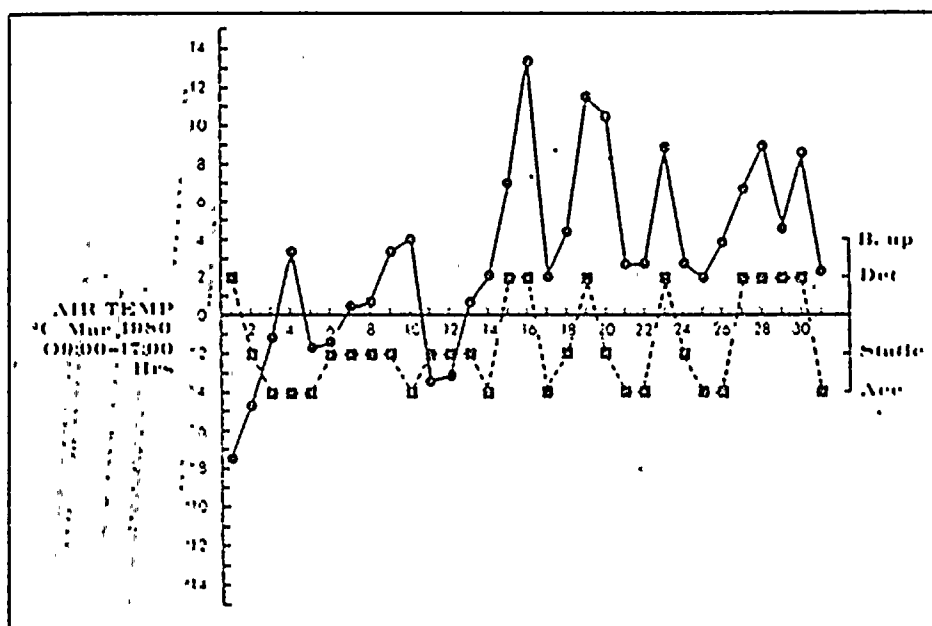


FIGURE 24. Plot of ice conditions and air temperature: March 1980.

The period of ice deterioration and breakup at the end of the winter is illustrated by March 1979 data (Fig. 12). It is evident, as one would expect, that as the air temperature increased the nearshore ice complex began to deteriorate. A variable that was qualitatively assessed but not quantified, and perhaps should be in future studies, is the impact of ice-bound sediment in relation to solar heat absorbed by the ice as the air temperature increases.

In general, the nearshore ice complex is initiated by a sustained lowering of air temperature below 0°C over the Great Lakes area. When air temperatures increase markedly, the ice complex responds accordingly with the onset of deterioration; if the increase in the air temperature is sustained, breakup of the ice complex occurs.

Applying this notion to the winter of 1979-1980, one observes that indeed the general pattern repeats itself. Nearshore ice is not present during the month of December (Fig. 15) when air temperature remained above 0°C , but as the temperatures at the end of December began to decrease, and those in January (Fig. 18) fell below 0°C , the ice complex quickly formed and established itself. As the air temperature fluctuated during January the ice complex responded either by deterioration or additional accretion. A comparison of March 1980 (Fig. 24) with March 1979 (Fig. 12) again shows that as air temperatures rises, deterioration or breakup was initiated. The relationship between ice conditions and air temperature which was observed throughout the study is considered a significant outcome. It identifies a necessary condition for the initiation and breakup of the nearshore ice complex.

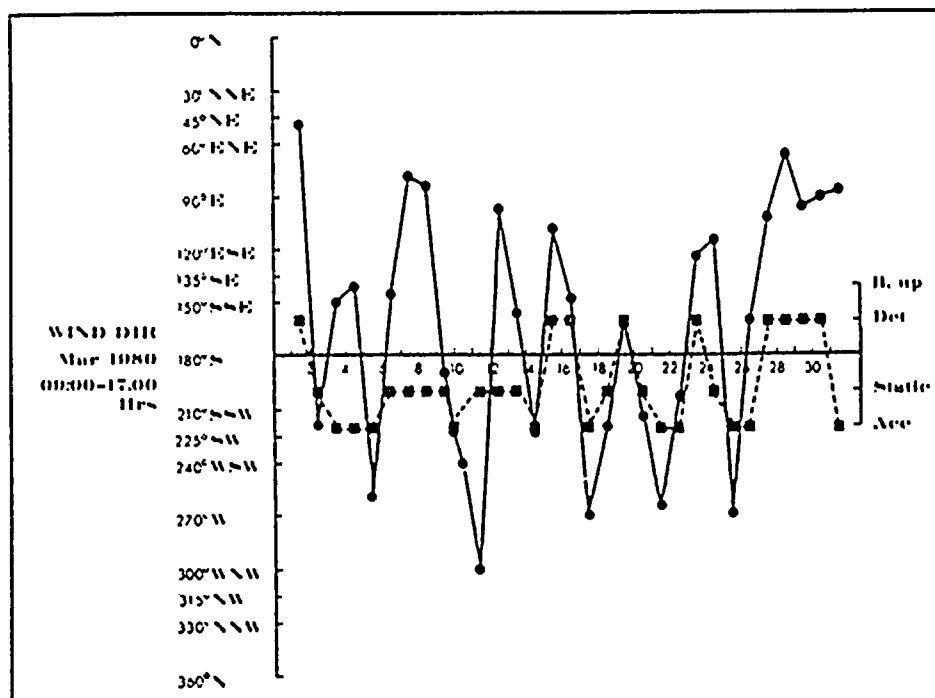


FIGURE 25. Plot of ice conditions and wind direction: March 1980.

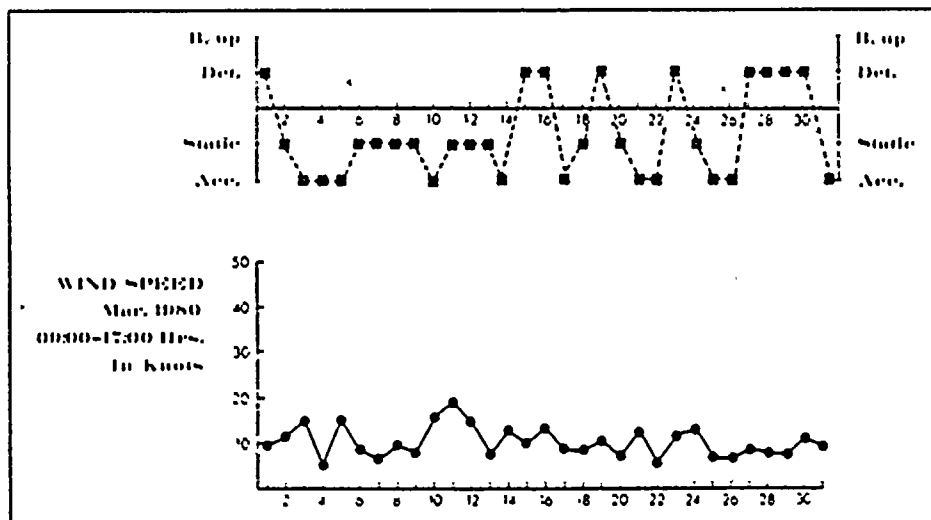


FIGURE 26. Plot of ice conditions and wind speed: March 1980.

WIND DIRECTION, WIND SPEED, AND ICE DEVELOPMENT

The subzero air temperature and the resulting heat exchange that occurs at the air-water interface are a necessary condition for changes in the ice complex at the study site. The data for the study site further substantiated the observations reported by Seibel et al. (1975) that the ice conditions were markedly influenced by the change in wind direction and impacted to a lesser degree by the wind speed. To illustrate how wind direction played a role in ice complex changes, some typical sequences of accretion and deterioration that occurred during two winters will be used. The wind direction is plotted by day and against the ice complex change over that same day (Fig. 5). Wind directions above the horizontal axis are easterly, and those below the axis are westerly. Deterioration and breakup conditions are plotted above the horizontal axis, and the static and accretional conditions are plotted below the axis. A point plotted directly on the axis indicates that no ice was observed.

It was previously suggested that the initiation of the nearshore ice complex on 24 December 1978 (see Figs. 4 and 5) was directly related to the sudden change in air temperature and resulting reduction in the surface water temperature, which in turn allowed ice to form. It is revealing, however, to observe that, coincidental to the air temperature falling below 0°C, the wind direction shifted from easterly to westerly. Three days later there was a gradual increase in the air temperature and accompanying shift of the wind from westerly to easterly, which were followed by breakup of the ice complex. It should be noted that the breakup of the nearshore ice complex does not necessarily imply that the lake proper in the vicinity of the study site was devoid of ice. During the study period, we observed that when the winds were easterly, lake ice moved lakeward (offshore), while when the winds were westerly it moved in the shoreward direction and impacted the nearshore ice complex by attaching itself to it. From these observations it was concluded that the lake ice complex, as well as the nearshore ice complex, was linked to the predominant wind directions at the study site.

The above scenario was repeated throughout our observations of the nearshore ice complex at this study site. The final breakup of the ice complex, as is illustrated in the March 1979 and 1980 plots (Figs. 12-14 and 24-26), is more complex than its initiation phase. There was a relationship among the stage of development of the ice, the air temperature, and the wind direction. It was observed that easterly flow of wind was more often associated with deterioration and breakup than was westerly flow, which was predominantly associated with accreting ice conditions. Air temperatures for most of March 1979 were above zero, and changes in the ice complex were in part linked to the changes in air temperature. An examination of Figure 13 provides information on the wind direction for that same time period. There is a good correlation of wind offshore and deterioration. It is clear when one examines the data that no one meteorological variable controls the process of ice complex formation and breakup.

THE NEARSHORE ICE RIDGE

The nearshore ice complex is composed of the icefoot, lagoons, and ice ridges (Fig. 3). Ayers et al. (1973) suggested that the location of the ice ridges at offshore bars was coincidental. O'Hara and Ayers (1972) observed from diving operations that remnant blocks of a deteriorating ice ridge were embedded in the lake bottom,

and Bryan and Marcus (1972) observed during an aerial reconnaissance that remnant ice blocks were grounded along offshore bars.

These qualitative observations were taken by Seibel et al. (1975) and tested. The time lapse high oblique photographs provided the basis for quantitative analysis of the hypothesis of coincidence of location of the offshore ice ridges, offshore breaker zone, and offshore bars. The data presented by Seibel et al. (1975, 1976) showed that the three characteristic shore features were indeed coincident. Additional data are provided in Figure 27, wherein are shown the ice ridge and breaker zone locations for the winter of 1974-75 at the study location. As was illustrated by Seibel et al. (1975) in their later report at the same study site, the data presented in Figure 27 clearly show the coincident nature of the offshore breaker zone and the offshore ice ridges. Seibel et al. (1975) found the mean location of the three breaker zones and three ice ridges to be at about 50, 115, and 215 meters from the shoreline. Figure 27 reveals close concordance with the above findings. Total concordance should not be expected because the position of the breaker zones are a function of the deep water wave heights, which in turn are a function of the storms generating them. Hence, the features produced by the breaking waves, namely the offshore bars and the nearshore ice ridge, will vary with changing storm conditions and deep water wave heights.

The significance of the nearshore ice ridge lies in its role in the modification of offshore topography and in shoreline erosion. Seibel (1972) noted that the ice ridge played a role in both offshore topography modification and shoreline erosion. Seibel et al. (1975) speculated that a storm immediately following breakup could cause greater shoreline change than a storm of equal magnitude several weeks after the breakup, when offshore bars have had sufficient time to reform. During the winter months, a time when storm activity on the Great Lakes can be intense, the presence of the offshore and nearshore ice complexes plays a role in reducing the impact of wave attack on the shoreline. Thus the presence of the nearshore ice ridges prevents oncoming waves from direct access to the beaches and coastal bluffs.

THE MELTHOLD AND THE NEARSHORE ICE COMPLEX

The nearshore ice complex was impacted by the plant's discharge of waste heat into the nearshore waters of Lake Michigan. In order to evaluate the role of this discharge on the nearshore ice complex a west-facing camera installed at the plant site was used to monitor conditions in the melthole (the point of discharge, under the ice, of the warm effluent) and the adjacent nearshore ice complex. Because of the camera's orientation some of the late afternoon photographs were of a reduced quality due to sunlight entering its lens. This camera, like those used in the high oblique photographic work, was connected to a timer that provided five shots on a daily basis: early morning, mid morning, noon, early afternoon, late afternoon, and midnight. The midnight photograph provided for daily separation of photographs as a check of the system. Photographing the site during periodic overflights provided panoramic vistas of ice conditions north and south of the plant as well as westward in the direction of the lake. The configuration of the melthole during the two comparison winters is shown in Figures 28 through 33.

The meltholes were generally longer than they were wide. The major axis of the meltholes was in the offshore direction, with the minor axis being in the longshore direction. With the exception of the ice conditions on 12 March 1980 (Figs. 32 and

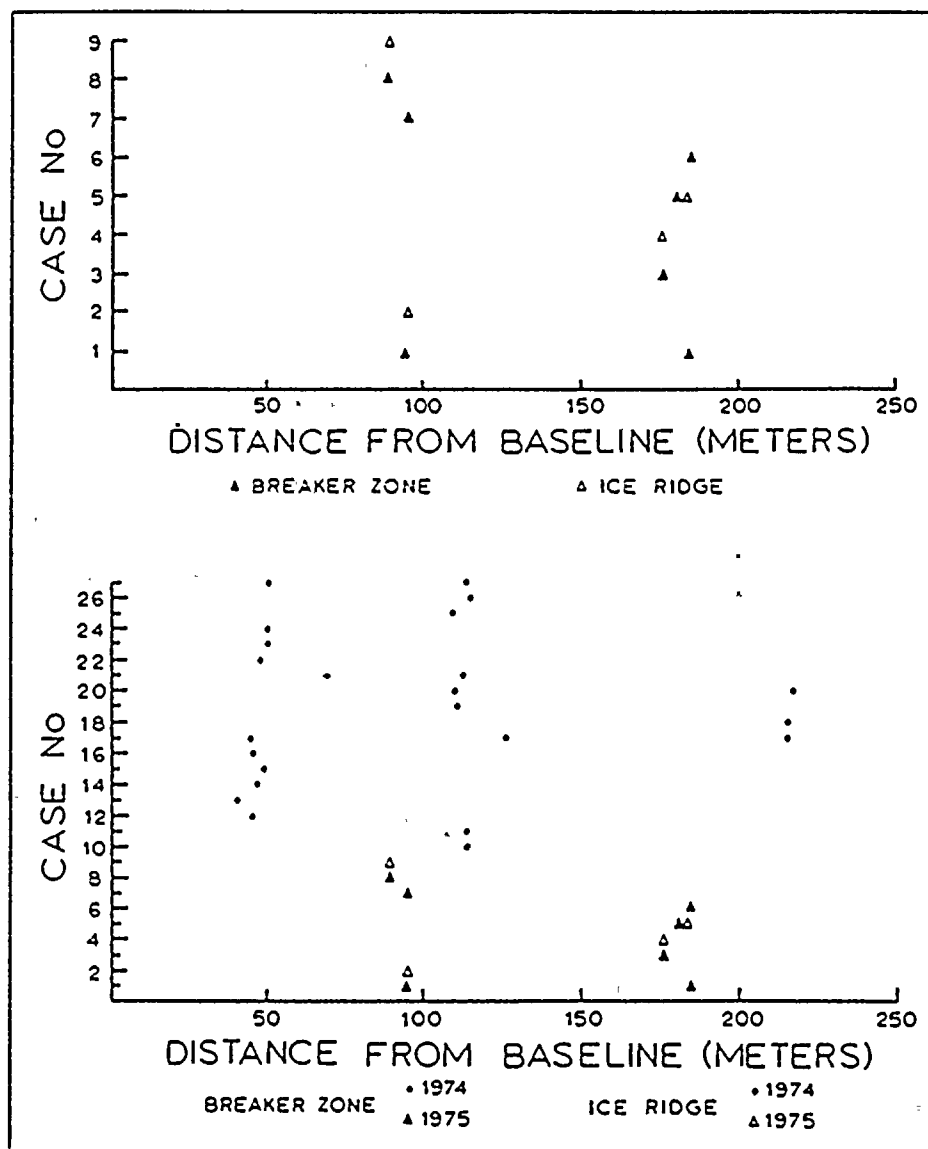


FIGURE 27. Plot of the locations of breaker zones and ice ridges for the winters of 1974 and 1975.

33), the melthole was surrounded by lake ice. This is an important observation, because in the early debates concerning the possible effects of the heated discharge on lake ice it was suggested that the size of the melthole would be larger and would

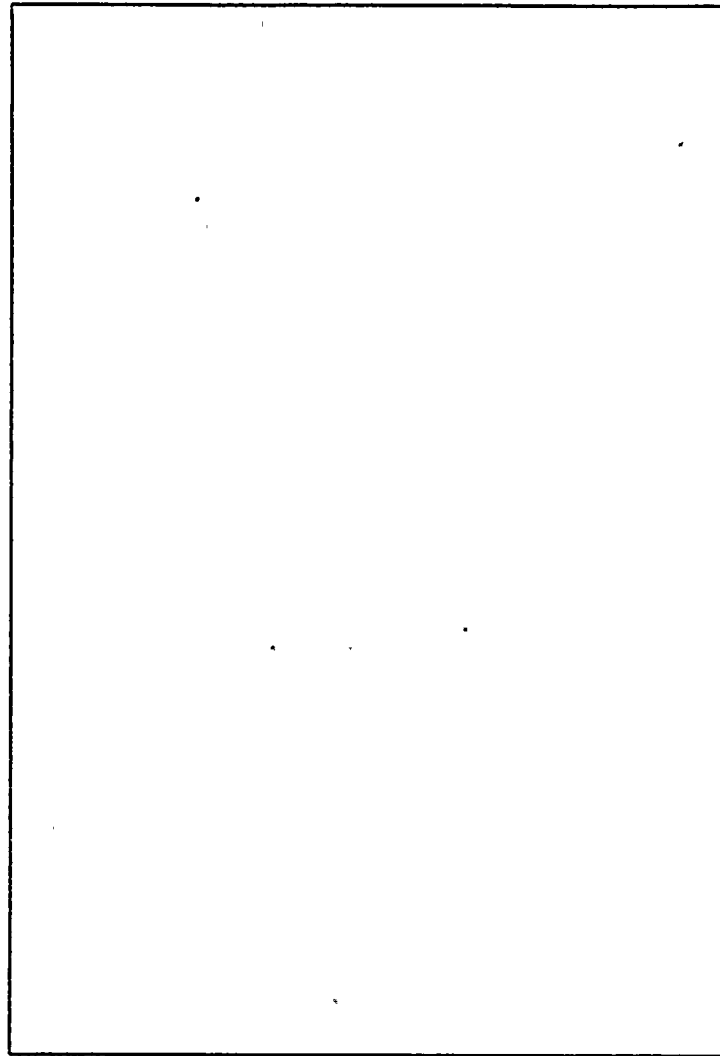
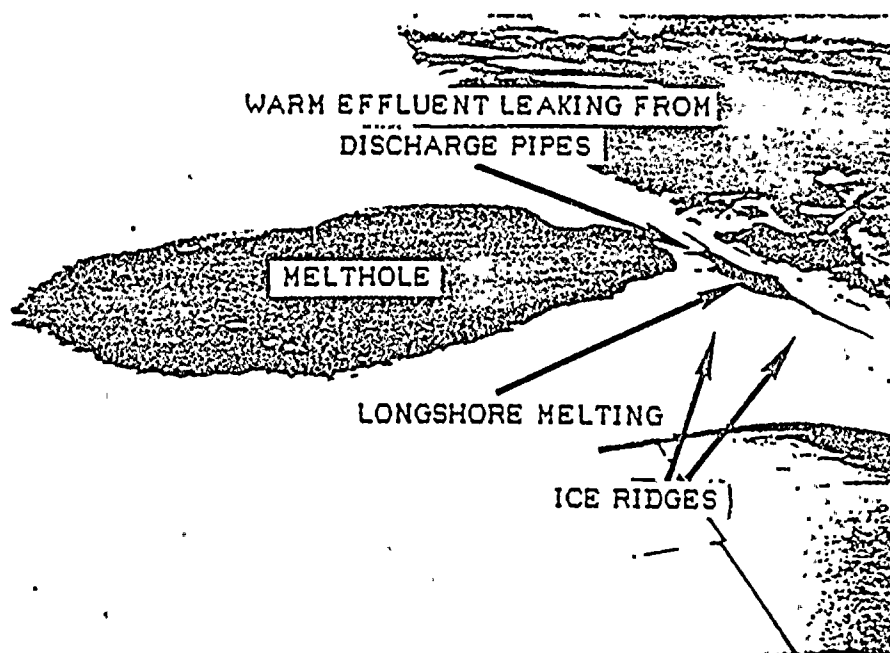
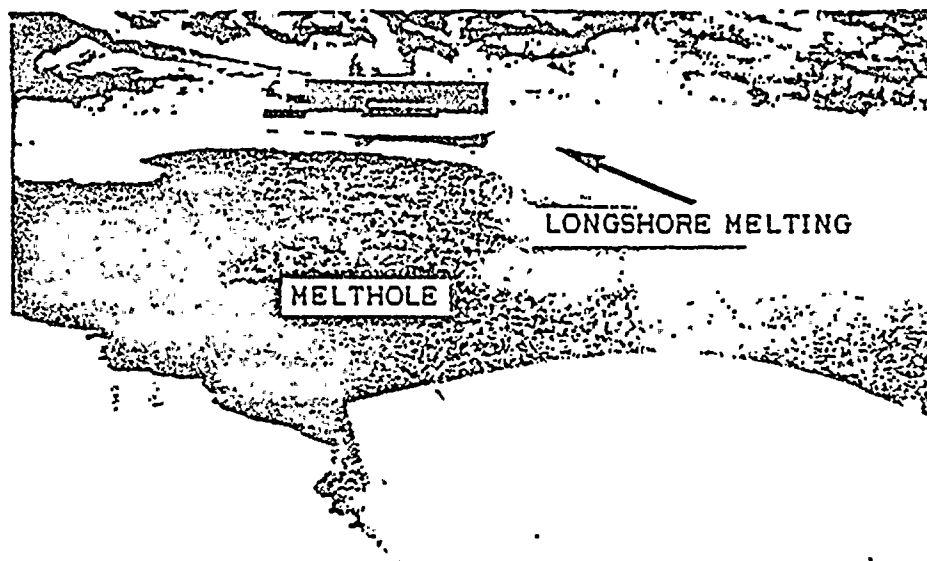


FIGURE 28. Two photographs of melthole, 1 February 1979. Top photograph looks east and bottom photographs looks north.

impact a greater portion of the lake ice in the vicinity of the plant than was subsequently shown to be true. It was anticipated that the melthole would break through all of the ice ridges and effectively result in the shoreline being exposed to wave action from winter storms for a significant distance north and south of the plant site. This condition was not observed. Instead, the offshore ice ridges nearest the



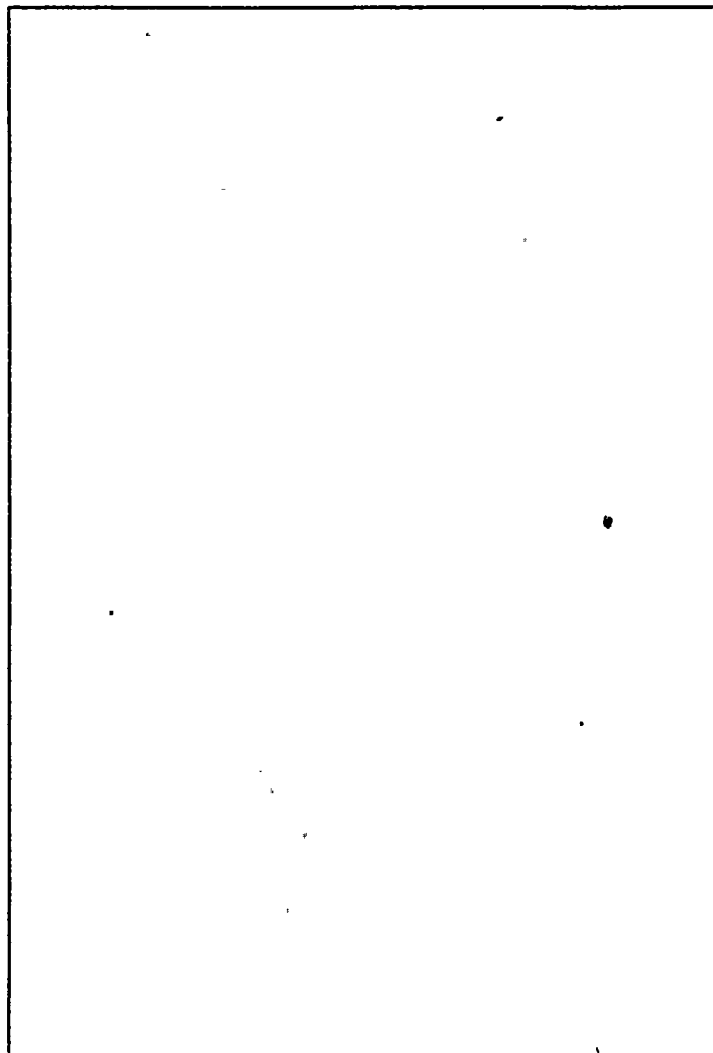
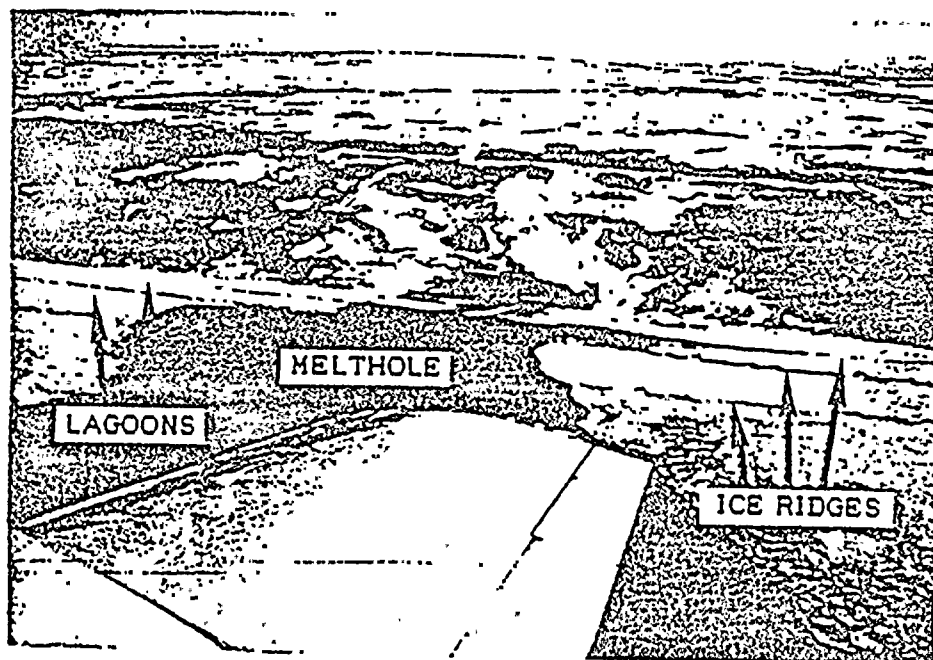
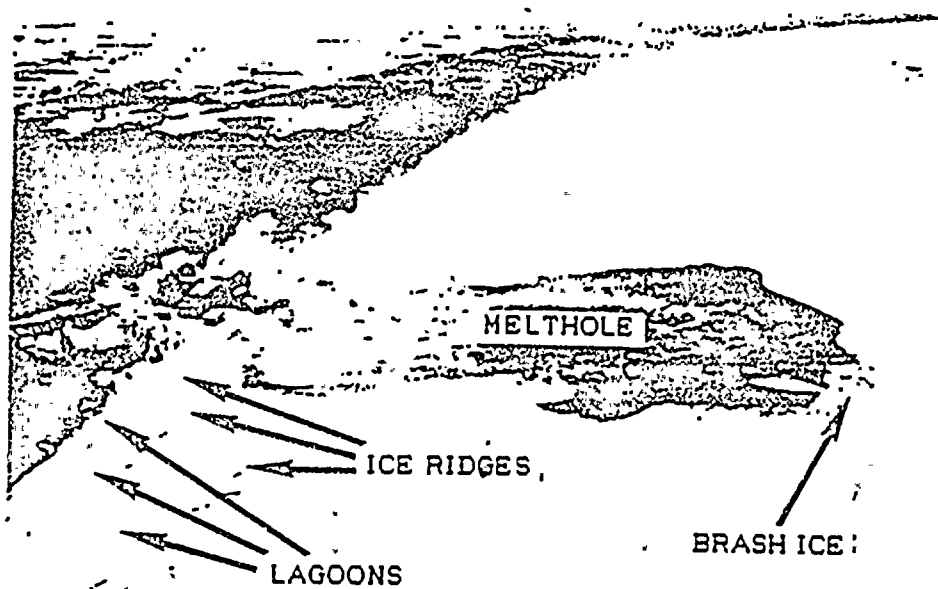


FIGURE 29. Two photographs of melthole, 17 February 1979. Top photograph looks south and bottom photograph looks east.

discharges were shown to be impacted, but those closer to the shoreline were not normally breached by the plume of the discharge waters.

Figures 28, 30, 31, and 32 reveal that there was open water present along the shoreline in front of the plant. Figure 28 reveals two linear features oriented in the offshore direction connecting the melthole proper with the water in front of the plant. These two linear features, which indicate water leaking from the discharge



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FIG. NO. 27

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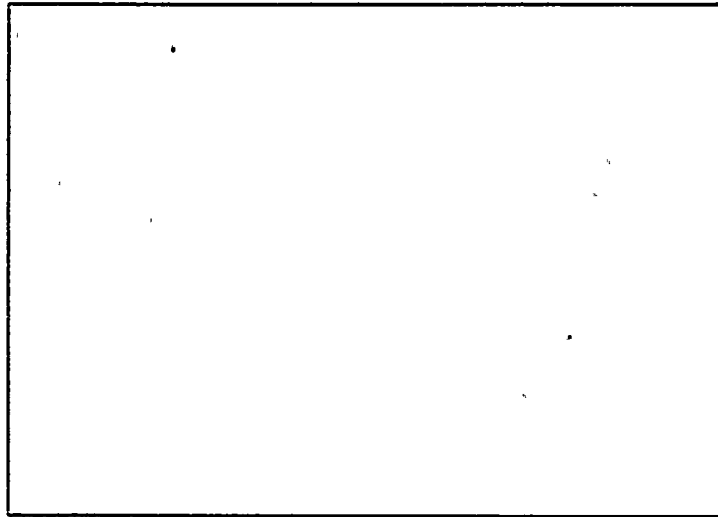
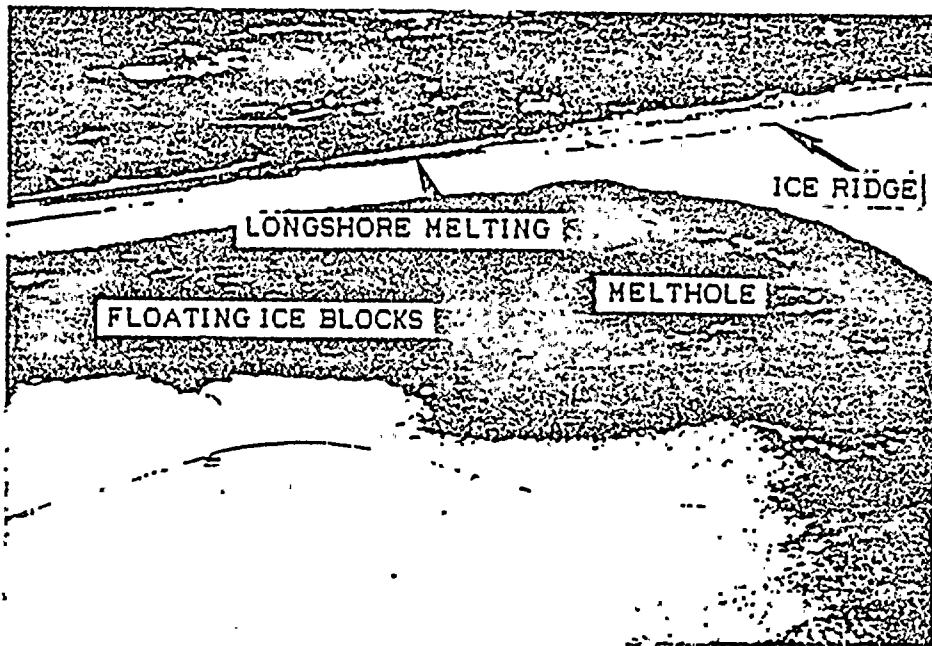


FIGURE 30. Photograph of melthole, 12 March 1979, looking southeast.

pipes, resulted in the melting of the nearshore ice complex including the ice ridges closest to the shoreline along the path of the pipes. Once that warm water discharge along the pipes was corrected, the effect was no longer observed (Fig. 30). The open water along the shoreline was observed almost from the commencement of the study, and prior to the discharge of heated water from the plant. Figure 28 provides a clue to the presence of this open water. A small channel of water was observed flowing across the beach face into the lake. This water was coming from land runoff, in particular a nearby parking lot. Because the nearshore ice complex closest to the shoreline is relatively thin, this land runoff was sufficient to cause the melting of ice closest to the source of that water.

Figures 28 through 33 show the extent of the meltholes as ascertained through aerial photography. Melthole size estimates determined by scaling the aerial photographs indicate that the average melthole is approximately 0.25 square miles (Table 1).

Visual observation of the ice ridges during deterioration revealed noticeable amounts of sediment incorporated in them. The presence of sediment in the nearshore ice structure may be an additional factor influencing the deterioration and breakup of the nearshore ice complex. Figures 28 and 31 illustrate that the ice ridges are darker in contrast to the adjacent lake ice. This contrast is a result of the sediment incorporated at the ridge crest. Although this phenomenon was observed visually throughout the study, no attempt was made to quantify the amount of sediment that was incorporated along the ridges. It is known that when waves break, nearshore bottom sediment is moved either as bed load or suspended load. That the sediment was present predominantly in the ice ridges suggests that the mechanism of breaking waves responsible for the formation of ice ridges is also the likely reason for sediment accumulation along their crests.



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FIG. NO. 30

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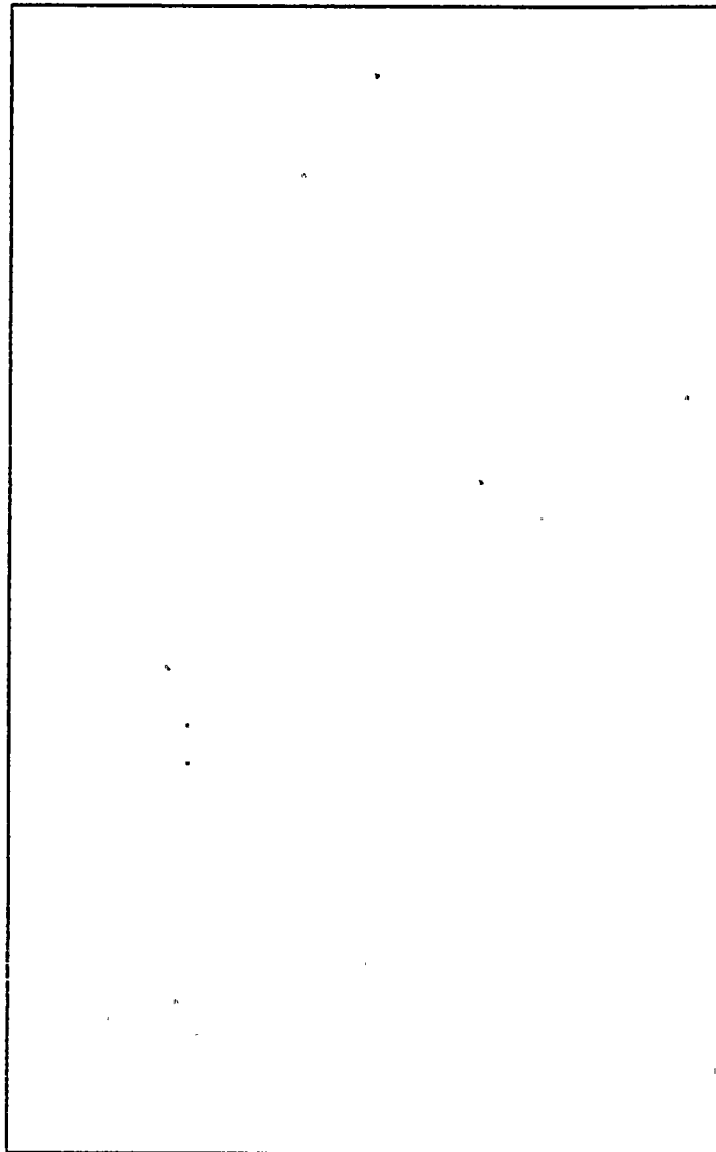
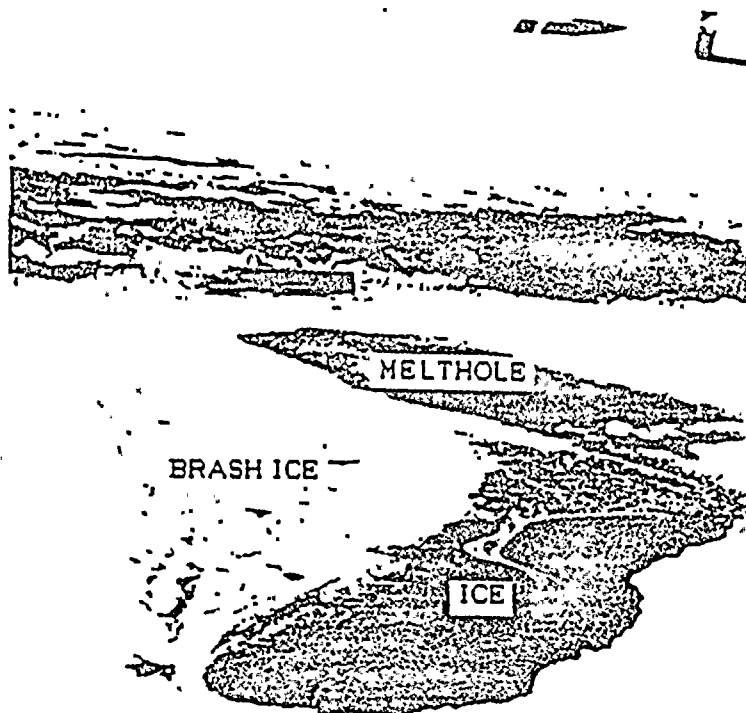


FIGURE 31. Two photographs of melthole, 18 February 1980. Top photograph looks southeast. Bottom photograph looks north.

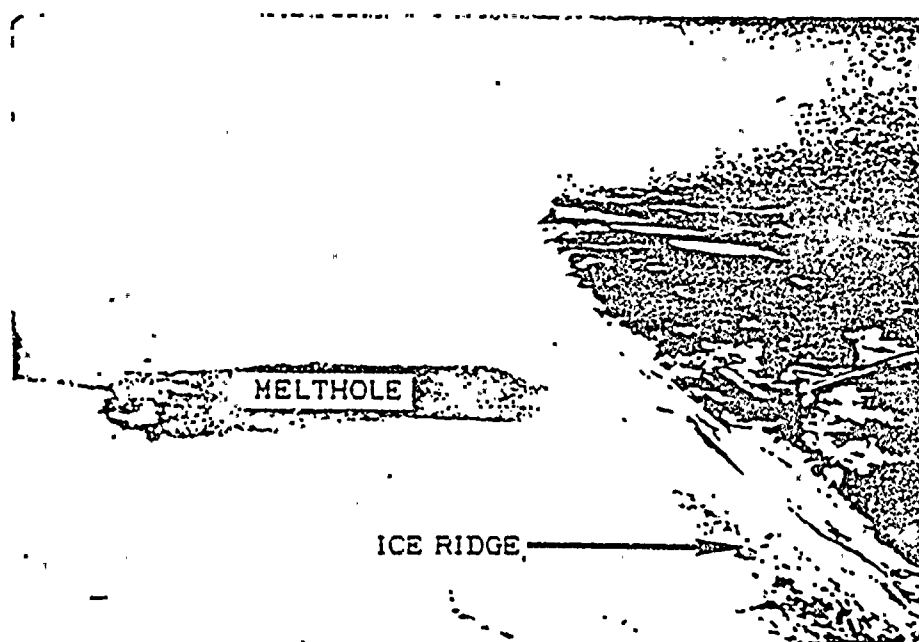


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FIG. NO. 31

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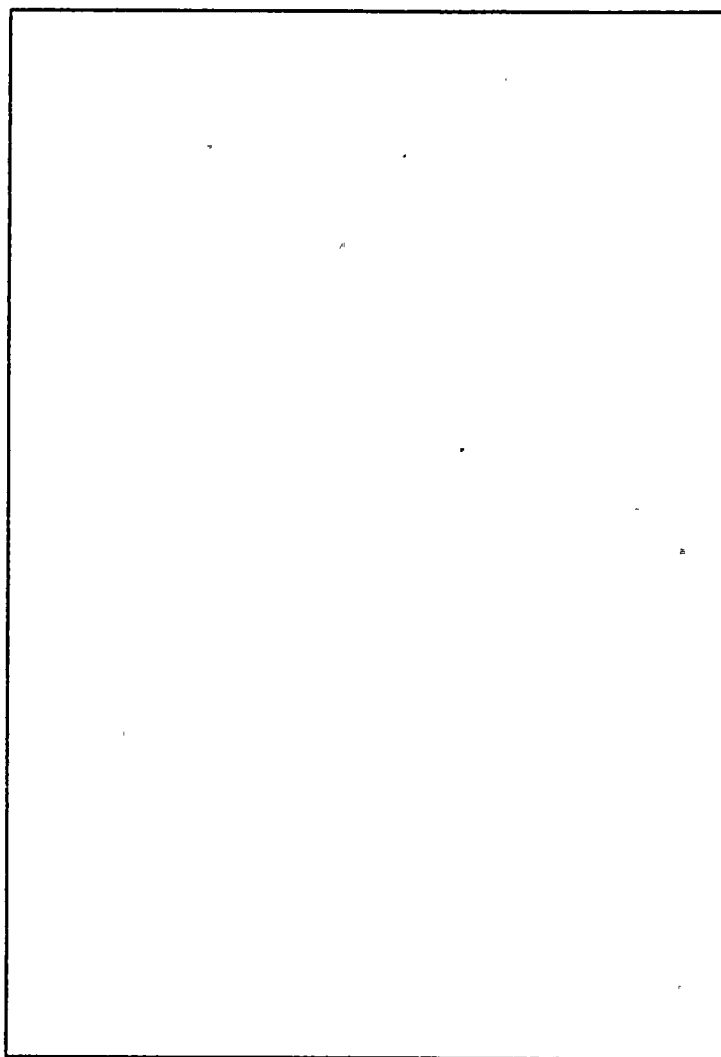
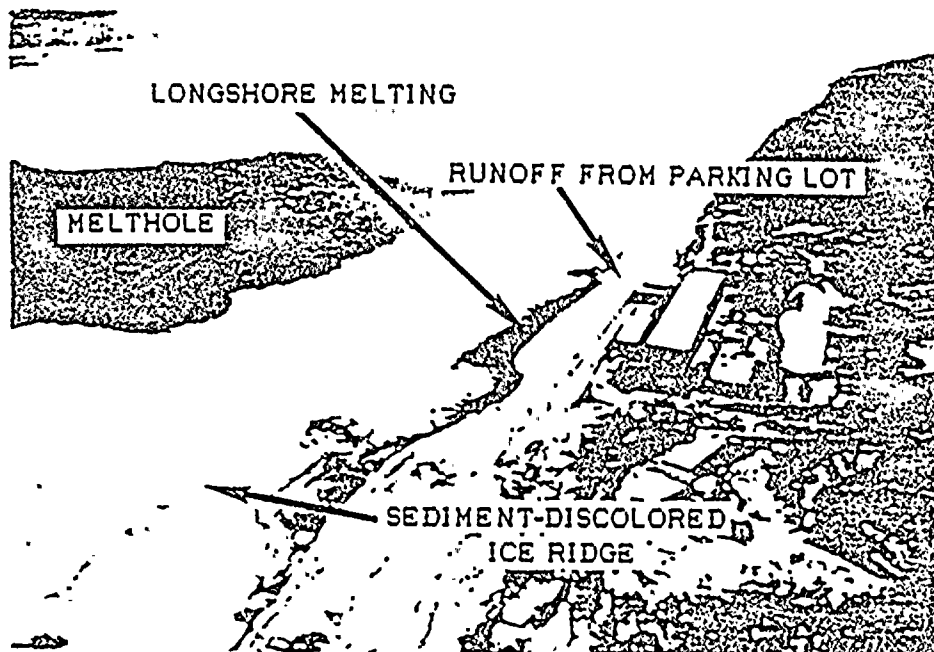
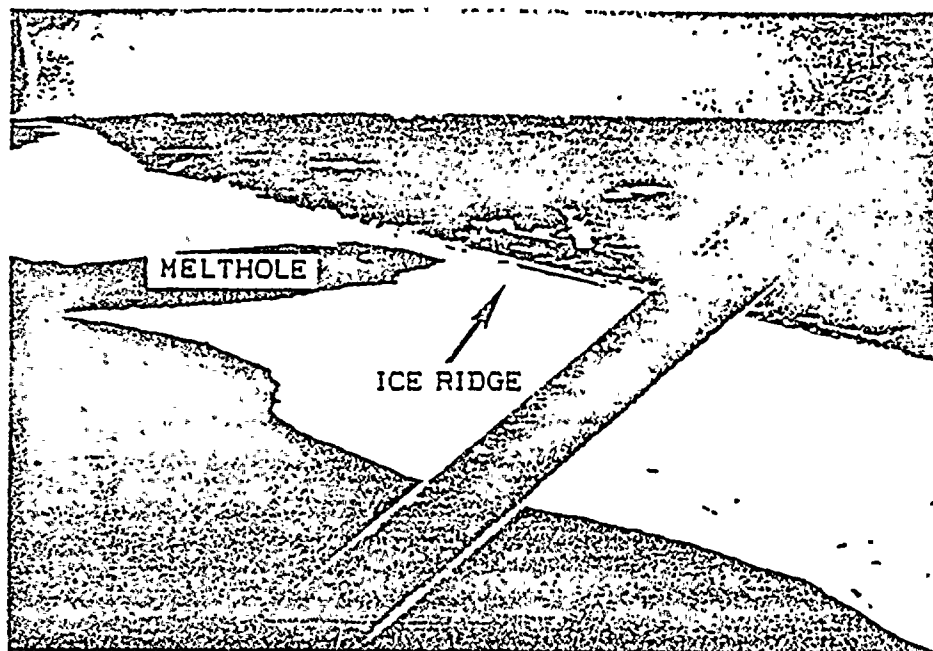


FIGURE 32. Two photographs of melthole, 12 March 1980. Top photograph looks northeast. Bottom photograph looks north and shows closeup of ice complex near shore. Discolored ice ridge is clearly visible in bottom left portion of photograph.

Although the plant's waste heat produces a large melthole in the ice complex during the winter months, the extent of the melthole is limited for the most part to the area directly in front of the plant in the proximity of the discharge pipes. Melting of the first lagoon was observed and attributed to runoff, containing road salt, from the plant's parking lots. Except for the area directly in front of the plant, the melted



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FIG. NO. 32

_____ %

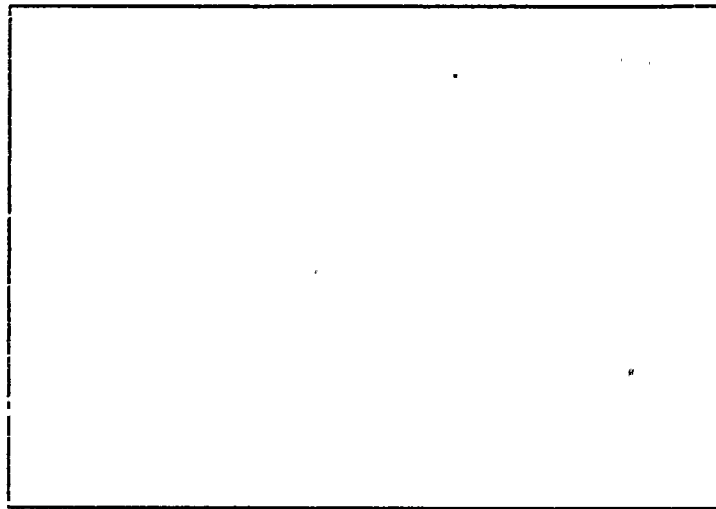
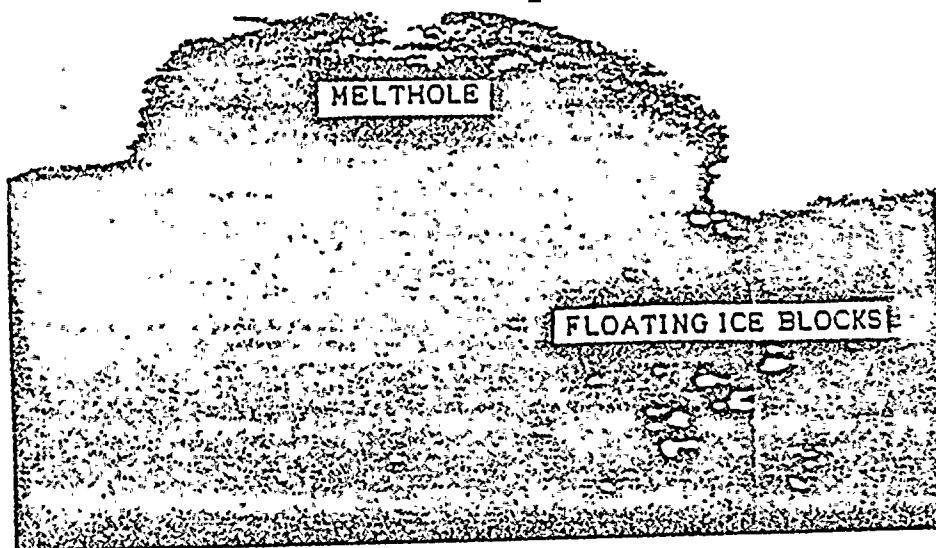
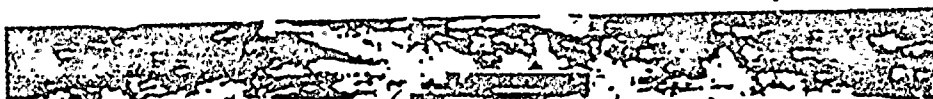


FIGURE 33. Photograph of melthole, 12 March 1980, looking east.

TABLE 10. Percent daily mortality calculated from mean alewife densities over all year, 1975-1982 (entrained larvae) and 1974-1982 (field-caught larvae). Design I used time intervals corresponding to peak occurrence of fish larvae in each length interval in our samples. Design II time intervals were calculated from laboratory-derived alewife growth rates (Heinrich 1981). YOY = young-of-the-year; NC = not calculated.

Length Interval (mm)	Entrained Larvae		Field-caught Larvae	
	Design I	Design II	Design I	Design II
2-5	11.7	27.3	NC	NC
5.5-10 to 10.5-15	19.5	8.3	NC	NC
10.5-15 to 15.5-20	3.5	4.0	NC	NC
15.5-20 to 20.5-25	1.2	1.7	NC	NC
25.5-25 to YOY	2.5	2.1	NC	NC
2-5 to 5.5-25	5.7	12.7	3.1	6.4
5.5-25 to YOY	5.1	4.1	5.7	4.8
2-5 to YOY	5.3	5.3	5.0	5.0



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FIG. NO. 33

11 / 13.3 %

first lagoon and bared beach were protected from shoreline erosion by presence of the ice ridges. The intact condition of the adjacent ice ridges protected the shoreline from erosion by direct wave action.

CONCLUSIONS

Several pertinent conclusions appear to be evident from the monitoring of the nearshore ice complex on this portion of Lake Michigan:

1. The data presented here and by others reveal that the offshore ice ridges, offshore bars, and breaker zones, three characteristic features found along this shoreline, are indeed coincident.

Ice ridges appear to be grounded features of the nearshore ice complex and they serve a dual role. They protect the beaches from incoming wave energy when present, and during the breakup of the ice complex may modify the topography in the vicinity of offshore bars.

2. The stages of ice development appear not to be controlled by any single meteorological variable but by a complex interrelationship between ice development and meteorologic conditions. Air temperatures below freezing were found to be a necessary condition for initiation of the ice foot. Accretion of the ice complex was associated with a westerly wind, and deterioration with an easterly wind.

3. The plant's waste heat produced a melthole that ranged in size from 0.10 to 0.50 square miles in the ice complex. The melthole was restricted to the vicinity of the discharge area. The ice ridges closest to the shoreline were minimally affected by the melthole, and the effectiveness of the "ice ridge" complex as a wave energy dissipator to protect the coast was not significantly altered.

4. North and south of the melthole there was no noticeable change in the normal ice complex of ridges and lagoons and the nearshore ice complex was not discernibly altered due to the presence of the waste heat produced by the plant in that area.

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