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LIMNOLOGICAL EFFECTS OF ARTIFICIAL CIRCULATION
ON A WARMWATER EUTROPHIC LAKE

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Ralph Stewart Beebe

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Clarence D. McNabb
Major professor

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LIMNOLOGICAL EFFECTS OF ARTIFICIAL CIRCULATION ON A WARMWATER EUTROPHIC LAKE

By

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ABSTRACT

LIMNOLOGICAL EFFECTS OF ARTIFICIAL CIRCULATION ON A WARMWATER EUTROPHIC LAKE

By

Ralph S. Beebe

A study was undertaken in 1987 and 1988 to determine the effects of entire lake destratification on various limnological parameters. Two lakes were chosen for the study. Mud and Carpenter Lakes are two small (surface area 5.2 ha and 7.8 ha respectively) eutrophic lakes located in northeast Oakland County, Michigan, which stratify thermally in the summer months with corresponding anoxia in the hypolimnions. The lakes share the same watershed, with Mud Lake being connected downstream of Carpenter Lake by a short, shallow channel. During the summers of the period of study, Mud Lake was destratified by using an air-lift device (LAKE PRESERVER) manufactured by Kobe Steel, Ltd. of Japan. Data from the second summer of the study indicate that the device was successful in completely destratifying Mud Lake with respect to many physical-chemical parameters. A depth-discrete benthic survey was conducted during the second year of the study. The predominate benthic taxa in Mud Lake at lower depths (12 m and 16 m) were dipteran midge larvae. These larva were present at lower depths throughout the summer, which is not common for lakes with anoxic hypolimnions.

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INTRODUCTION

Many of Michigan's lakes are undergoing eutrophication as a result of human activities. Both point and nonpoint sources of pollution have contributed to water quality problems including abundant weed growth and nuisance algal blooms which in many instances have severely impaired the potential of lakes for recreational or other uses. In recent years, many restoration techniques have been proposed and studied (see Cooke et al. 1986). One technique that has shown promise for a number of eutrophic lakes has been artificial circulation or destratification. Although the effects of artificial destratification have not always been predictable (Cooke et al. 1986), the principal and most reliable effect has been to raise the dissolved oxygen content throughout the water column. Other physical-chemical parameters that have been evaluated have shown the following trends:

- pH is lowered throughout the water column (Maleug et al. 1973; Weiss and Breedlove 1973),
- isochemical concentrations with depth for many chemical species (Toetz et al. 1972),

- decreased concentrations of reduced forms of iron and manganese in lower waters (Wirth et al. 1967; Brezonik et al. 1969),

while others (such as Secchi disk transparency), have shown highly variable results (Pastorok et al. 1980).

The response of biological parameters to artificial destratification have not been documented as thoroughly as physical-chemical responses and studies have shown contradictory results. Results have indicated that there is sometimes a shift from blue-green to green algae after destratification (Weiss and Breedlove 1973; Cowell et al. 1987) and an overall decrease in algal biomass (Maleug et al. 1973; Cowell et al. 1987). Macroinvertebrates were found to increase in densities by Wilhm and McClintoch (1978) while Cowell et al. (1987) found decreasing densities. One study reported an increase in the zooplankton (Brynildson and Serns 1977) but in another they decreased (Cowell et al. 1987).

Work that is reported here was initiated to ascertain the chemical, biological, and physical effectiveness of one particular destratification device, the LAKE PRESERVER, an air-lift water convector marketed by Kobe Steel America, Incorporated. Mud and Carpenter Lakes (two small, eutrophic lakes in southeastern Michigan) were chosen for this study. Previous studies had been conducted on these lakes. Data on lake morphology, water quality, sediment chemistry, aquatic plants, benthic invertebrates, and fish populations were collected prior to the construction of a General Motors assembly plant on the eastern part of the watershed (Liston et al. 1980). Plant construction was completed in 1981 and the potential impacts of plant discharges were assessed in July 1985 (Liston et al. 1985). Data from these studies provide some baseline data for comparison of past lake conditions to those after installation and operation of the Lake Preserver. The Lake Preserver was installed in Mud Lake in the spring of 1987 (Batterson et al. 1987). Carpenter Lake, which is connected to Mud Lake by a short, narrow channel, was studied for comparative purposes. The two lakes were biologically similar prior to artificial destratification (Liston et al. 1980, 1985).

THE STUDY SITE

Mud and Carpenter Lakes are located in Orion Township, Oakland County, Michigan (T4N, R10E, Section 34). The topography of the watershed is flat to slightly rolling and is a sub-basin of the Clinton River. Carpenter Lake lies to the southeast of Mud Lake. The two lakes are immediately adjacent to one another, connected by a 2 m wide by 10 m long channel. Water flow within this channel is imperceptible. Water exits Mud Lake on the west side through a small ill-defined channel, eventually passing to Judah Lake over a distance of about 0.44 km.

Mud Lake is shaped like an irregular cone with a narrow littoral zone and steeply sloping sides (Figure 1). The lake has a maximum depth of 19.2 m, a mean depth of 7.1 m, a surface area of 5.2 ha and a total volume of approximately 371,000 m^3 (Liston et al. 1980). Hypsographic (depth-area) and depth-volume curves are presented in Figure 2. Mud Lake normally stratifies (Liston et al. 1980, 1985) and possesses a well-defined epilimnion that occupies the upper 2 m. Typically, the metalimnion extends to 6 m with an anoxic hypolimnion below (Liston et al. 1980, 1985).

Carpenter Lake is the shallower of the two lakes. The basin of Carpenter Lake is shaped like a bowl, with a

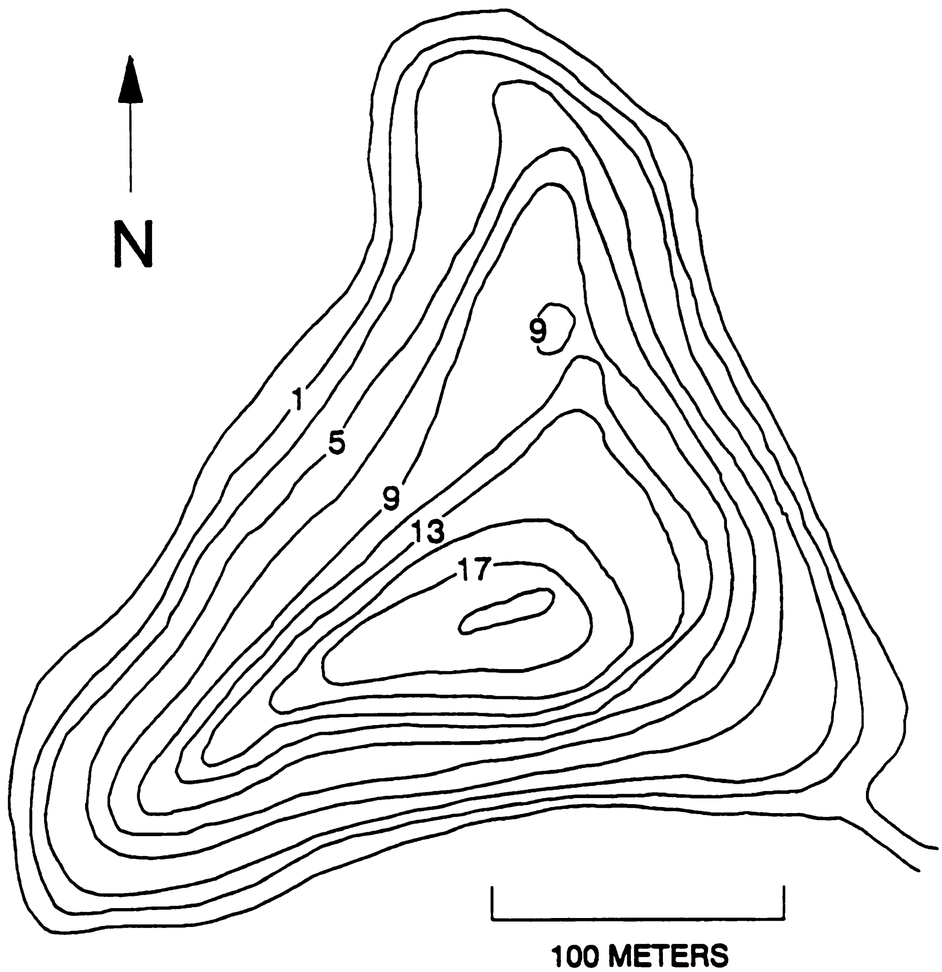
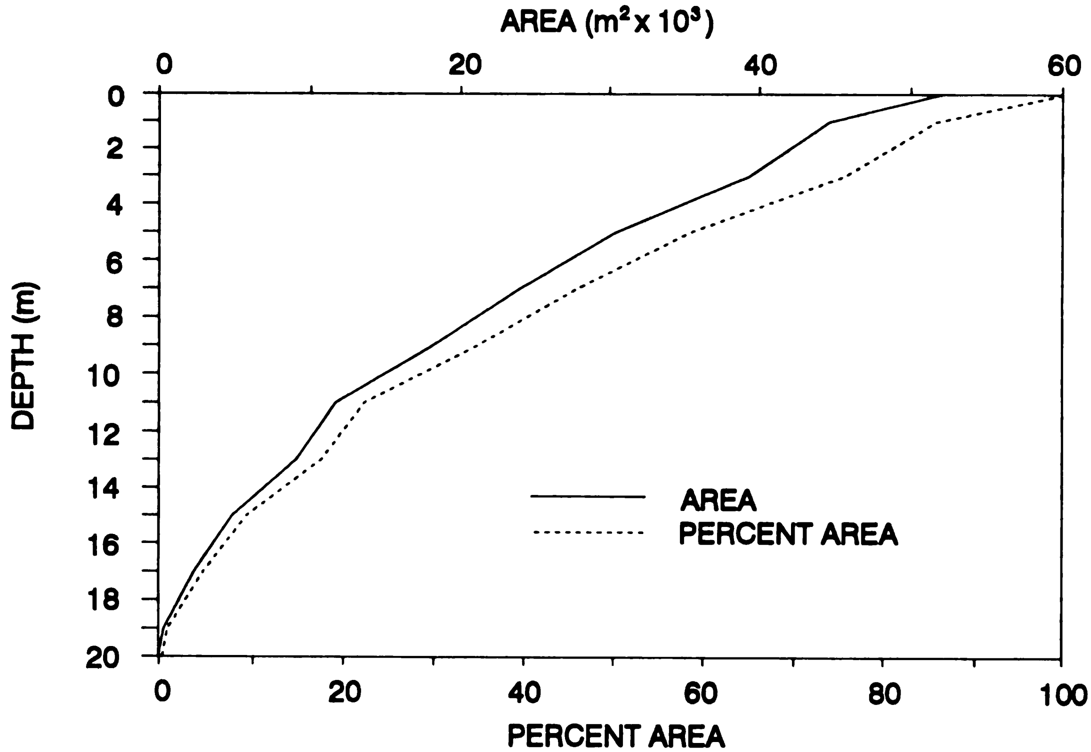


Figure 1. Bathymetric features of Mud Lake, Oakland County, Michigan.



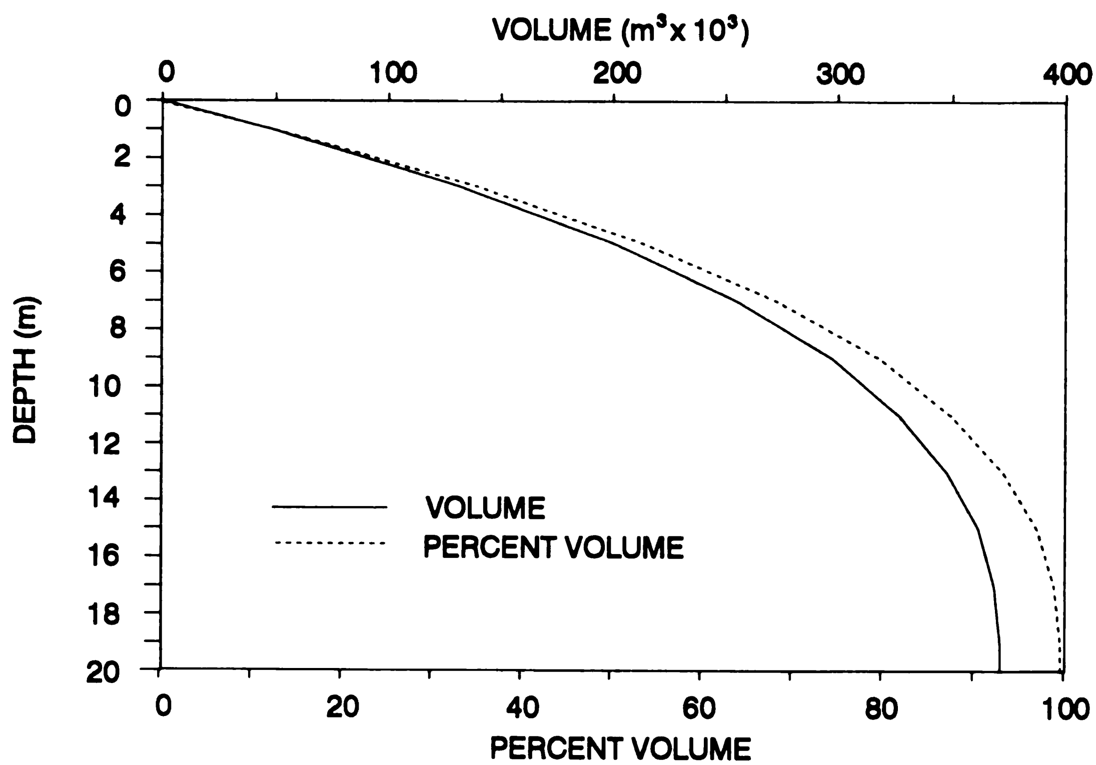


Figure 2. Hypsographic (depth-area) and depth-volume relationships for Mud Lake, Oakland County, Michigan.

gradually sloping littoral zone and then a sharp drop-off which extends down to a flat broad bottom (Figure 3). It has a maximum depth of 11.3 m, a mean depth of 4.4 m, a surface area of 7.8 ha, and a total volume of approximately 339,000 m^3. Hypsographic and depth-volume curves of Carpenter Lake are presented in Figure 4.

Mud and Carpenter Lakes both support fish populations characteristic of small, enriched, warm-water lakes (Liston et al. 1985). Both lakes are used by local residents for swimming, boating, and all-season fishing. Residents who live along the south side of Mud Lake informally manage that shoreline by removing aquatic weeds and spreading sand along that area. Both the southern and northern shores of Carpenter Lake have houses on them. The homeowners manage the lake shore in front of their homes in the same way as the residents of Mud Lake.

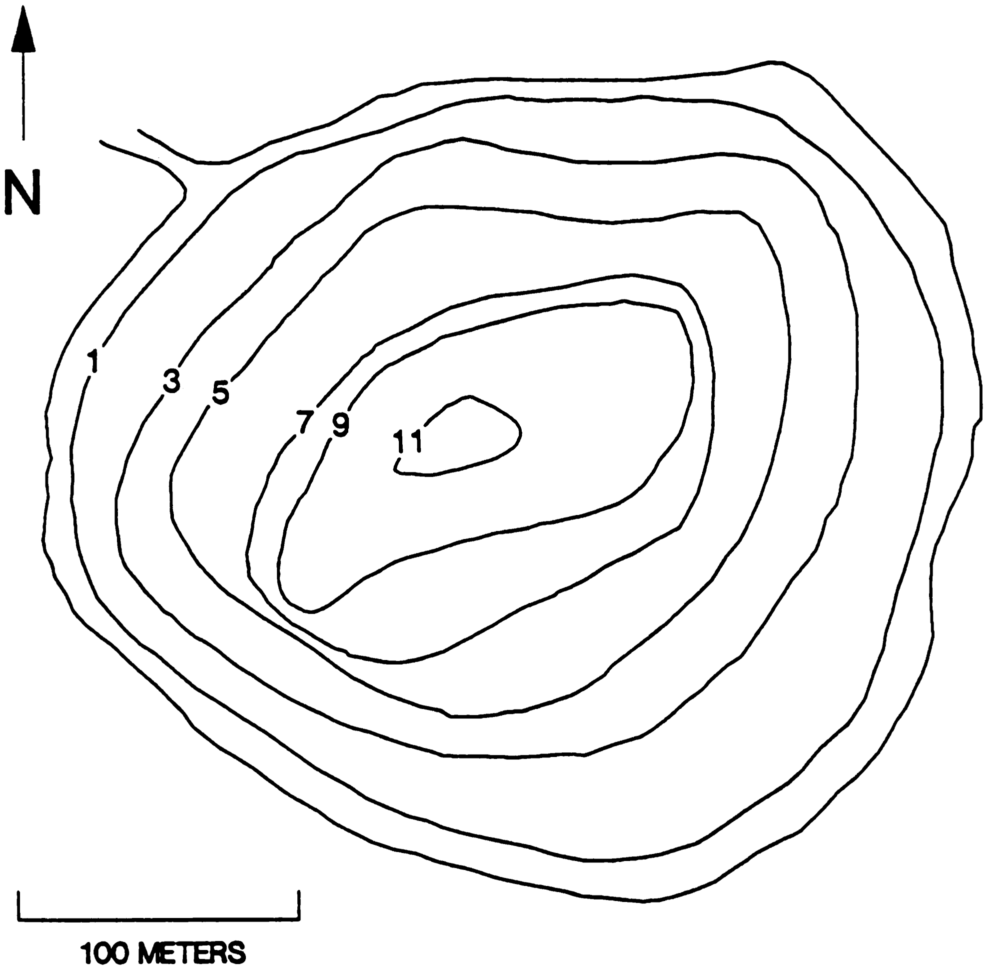


Figure 3. Bathymetric features of Carpenter Lake, Oakland County, Michigan.

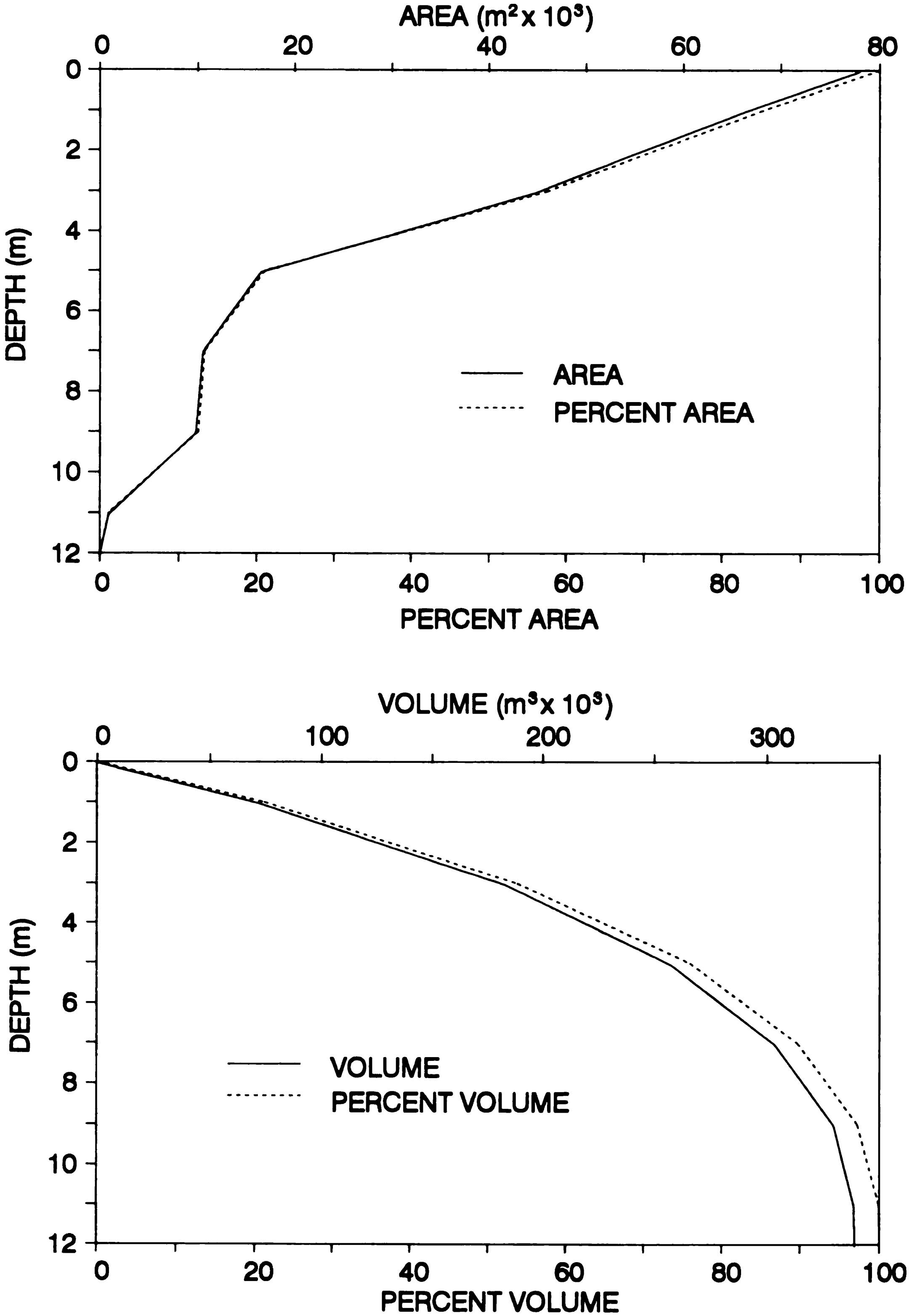


Figure 4. Hypsographic (depth-area) and depth-volume relationships for Carpenter Lake, Oakland County, Michigan.

THE LAKE PRESERVER

The Lake Preserver, an air-lift water convector marketed by Kobe Steel America, Inc., was installed on Mud Lake on April 16, 1987, approximately three weeks after ice-off. This device consisted of three components: a float, air chamber, and tube (Figure 5). It had an overall length of 9 m and a tube diameter of 40 cm. The Lake Preserver was anchored to the bottom of the lake by a sinker. A float around the upper portion kept the tube vertical. Air was pumped from two onshore electrical compressors (5 horsepower, 29 m^3/hr @ 175 psi) via hosing to the air chamber located at the bottom of the tube. The chamber was designed so that once it filled with air, a large bubble was released which traveled up the tube pushing water from the lower depths. When the large bubble exited the tube (at a depth of approximately 6 m) it would break up into smaller bubbles before reaching the surface.

The work that is reported here is for the second year of operation of the Lake Preserver. The results from the first year of the study were inconclusive due to a number of mechanical failures of the destratification device. During 1988, air was continuously delivered to the Lake Preserver from April 12 (after ice-off and during spring turnover) to

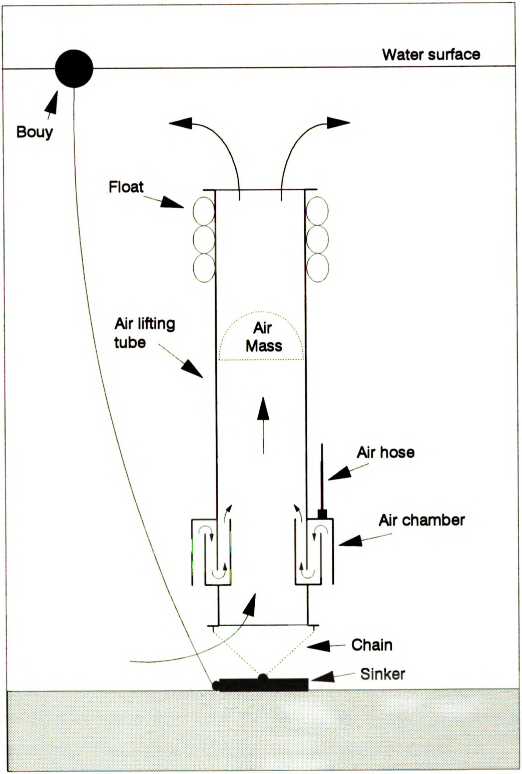


Figure 5. Schematic representation of the Lake Preserver destratification device installed in Mud Lake, Oakland County, Michigan.

July 28 at a rate of 27 m^3/hr. On July 28 the air flow rate was increased to 40 m^3/hr and continued at that rate until shut-down on November 14, 1988. At these flow rates there was a continuous stream of bubbles rising to the surface of the lake.

METHODS AND MATERIALS

PHYSICAL-CHEMICAL PARAMETERS

Weekly sampling for temperature, dissolved oxygen, conductivity, and water transparency was conducted from compressor start-up (April 12, 1988) to compressor shut-down (November 14, 1988). There were three sampling locations for each lake (Figure 6). A mid-lake station was located at the deepest point of each lake. Two other locations in each lake were also sampled for determination of horizontal homogeneity. At each location, all measurements were done at one meter depth intervals. Temperature and dissolved oxygen were measured with a Yellow Springs Instrument (YSI) Company Model 54A Dissolved Oxygen meter and probe with stirrer. The meter was air calibrated before each use. Conductivity was measured with a YSI Model 33 Salinity-Conductivity-Temperature meter. Conductivity values were temperature corrected and are presented as values at 25 °C using the method of APHA (1985). Water transparency was measured using a standard 20 cm Secchi disk. The limit of visibility was calculated by averaging the depth of disappearance and reappearance (Lind 1974).

Sampling to determine chemical conditions of Mud and Carpenter Lakes was conducted monthly from April to November

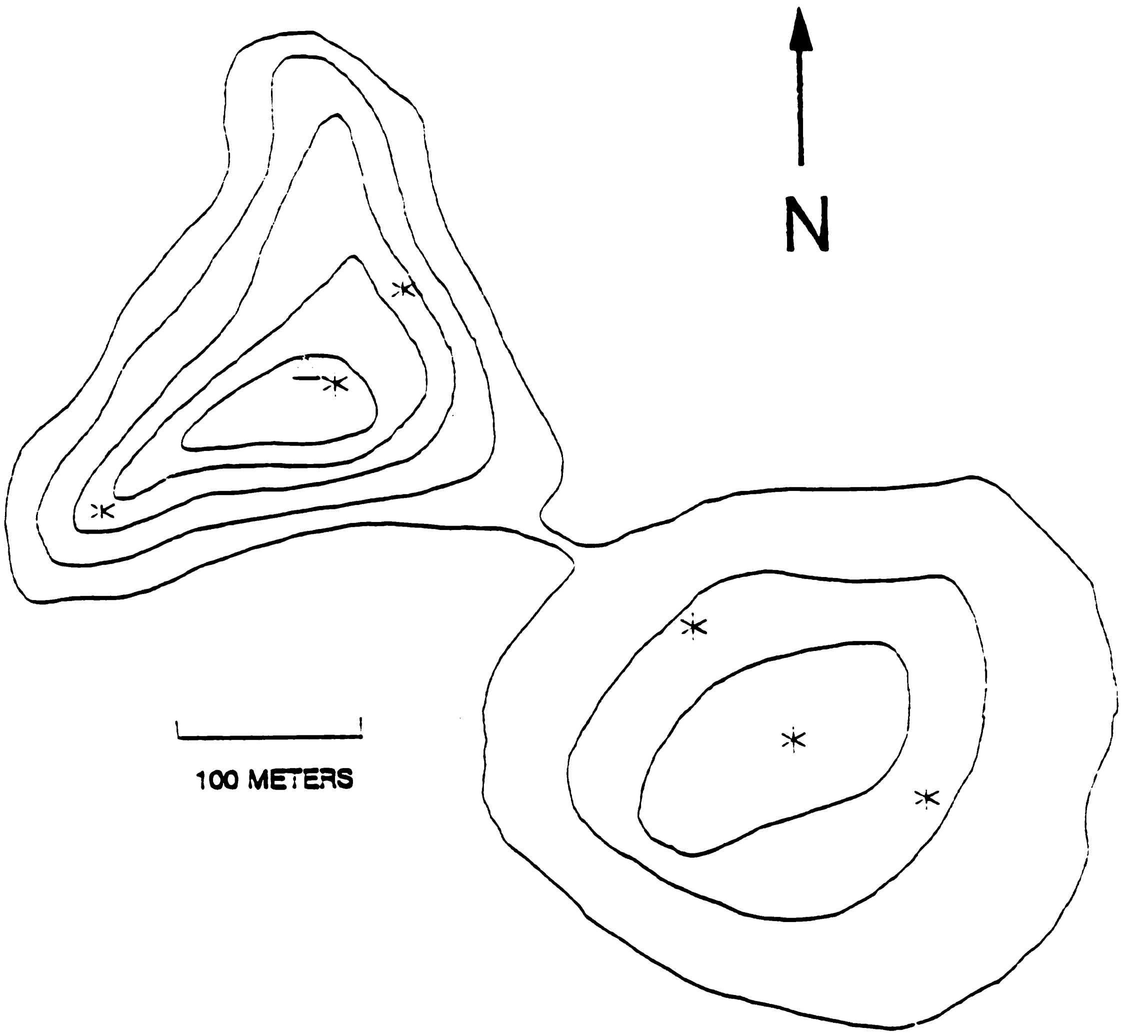


Figure 6. Water quality sampling sites (*), contours for collection of chironomid larvae, and the location of the Lake Preserver (△) in Mud and Carpenter Lakes, Oakland County, Michigan.

1988. For each sampling date, duplicate top (1 M below the surface) and bottom (1 M above the sediments), composite samples were collected in each lake. Each of the four composite samples was comprised of 15 individual, randomly collected, 500 ml grab samples.

After a composite sample was completed, it was immediately analyzed for hydrogen ion content with a Beckman Instruments Model 21 digital pH meter. Carboys were then tightly stoppered, stored on ice in coolers, and transported to the laboratory at Michigan State University (MSU). Within 24 hours of collection time, samples at room temperature were analyzed for alkalinity using a potentiometric pH end-point titration of 4.5 (APHA 1985).

Additional analyses were conducted on composite samples within 48 hours of sampling. These included total phosphorus (TP), nitrite+nitrate nitrogen (NO_2+NO_3-N), ammonia nitrogen (NH_3-N), Kjeldahl nitrogen (KN), elemental analyses (calcium, iron, magnesium, and manganese), total solids (TS), total volatile solids (TVS), and chlorophyll *a*. Additionally, October, analysis for chemical oxygen demand (COD) was conducted. Determinations of the forms of nitrogen, TP, TS, and TVS were done at MSU's Limnological Research Laboratory using "Standard Methods" (APHA 1985). This included the cadmium reduction method for NO_2+NO_3-N, distillation and Nesslerization for NH_3-N, semi-micro Kjeldahl digestion followed by distillation and Nesslerization for KN, persulfate digestion/ascorbic acid method for TP, total solids dried at 103-105 °C and total volatile solids ignited at 555 °C. Organic nitrogen, total inorganic nitrogen (TIN) and total nitrogen (TN) were determined by calculation. Organic nitrogen was determined by subtracting NH_3-N from KN, TIN equalled the sum of NO_2+NO_3-N and NH_3-N, and TN was the sum of TIN and organic nitrogen. Elemental analyses were conducted by the MSU Animal Health Diagnostic Laboratory using inductively coupled plasma-emission (ICPE) spectroscopy. Prior to analysis, samples were digested using nitric acid (APHA 1985). Chlorophyll *a* was determined spectrophotometrically using the method outlined in APHA (1985). COD analyses were conducted by SEG Enterprises, Incorporated of Lansing, Michigan.

BENTHOS

Benthic organisms were collected approximately every two weeks from late June through mid-October by a method of stratified random sampling using a petite PONAR grab sampler. Samples were randomly collected on depth contours of 4, 8, 12, and 16 m in Mud Lake, and 4 and 8 m in Carpenter Lake (Figure 6). To determine the location where samples should be collected, a map of each lake was overlain with a numeric grid, and numbers corresponding to x-y coordinates along depth contours were chosen from a random number table. The grab samples were placed in plastic bags, stored on ice in the dark, and transported to the laboratory. Within 24 hours after collection samples were passed through a 250 μm sieve. All retained materials were washed into labeled glass jars and preserved with 95% ethanol. All chironomids and *Chaoborus* were removed from preserved samples by hand-picking under 10x magnification and were counted and measured for total length. All sample counts were transformed to density estimates of number of organisms per m^2. Individuals from a random sub-sample of all chironomids collected during the sampling season were dried to constant weight at 105 °C. An analysis was preformed to determine an equation that represented the best fit for dry weight versus total length. Using that equation, estimates of chironomid biomass were calculated.

To determine the appropriate sample size required for estimating chironomid larvae density, a preliminary survey was conducted on June 8, 1988. On that date, ten random samples were collected from selected depth strata in Mud and Carpenter Lakes. The number of samples to be collected from each strata during subsequent studies was determined using the procedures of Elliott (1977).

PHYTOPLANKTON AND ZOOPLANKTON

Phytoplankton samples were collected monthly from April to November from a depth of 1m in both lakes at the mid-lake station. The samples were collected with a 1.2 L Kemmerer water sampler and preserved in the field with an algal killing and fixing solution described in Prescott (1978).

In the laboratory, 50 mL of each sample was passed through a 0.45 μm type HA Millipore membrane filter under as low a vacuum as possible. Filters were dried in an oven at 50 °C, then cleared with Type A non-drying microscopic immersion oil prior to examination. Filters were viewed at 450x and relative abundance determined by scoring the first 100 individuals (cells, clusters, or colonies) encountered in randomly selected fields. They were identified according to Prescott (1962, 1978).

Zooplankton samples were collected from the water remaining in the composite samples, generally 3-4 L, after the water necessary for the other analysis had been removed. This water volume was measured and passed through a 80 μm Wisconsin plankton net and then washed into a collection bottle. The samples were brought to a constant volume (100 mL), and then three 10 mL subsamples were counted. Densities in organisms per liter were obtained by dividing the average count of the subsamples by the volume of water originally sampled. Zooplankton were identified under 100x magnification using taxonomic keys of Edmondson (1959) and Pennak (1978).

RESULTS

PHYSICAL-CHEMICAL PARAMETERS

Temperature, dissolved oxygen, and conductivity data collected at the three stations within each lake were similar. Therefore, only data from the middle station of each lake is presented.

Temperature data for Mud and Carpenter Lakes during 1988 are presented in the upper panels of Figures 7 and 8, respectively. Mud Lake did not thermally stratify during the course of operation of the Lake Preserver (Figure 7). Top to bottom (0 to 15 m) temperature differentials were often less than 0.3 °C. Temperatures ranged from 2-3 °C in January and February to the mid-20s during the summer. A maximum surface temperature of 26.9 °C occurred on August 15, 1988. Conversely, Carpenter Lake stratified from early May through November with a metalimnion occurring at a depth of approximately 2-3 m (Figure 9). Temperature differentials between the top and bottom (0 to 9 m) waters often exceeded 20 °C during mid-summer. Surface waters of Carpenter Lake warmed more rapidly than those of Mud Lake and reached a higher maximum. The warmest surface temperature attained in Carpenter Lake was 29.0 °C, which

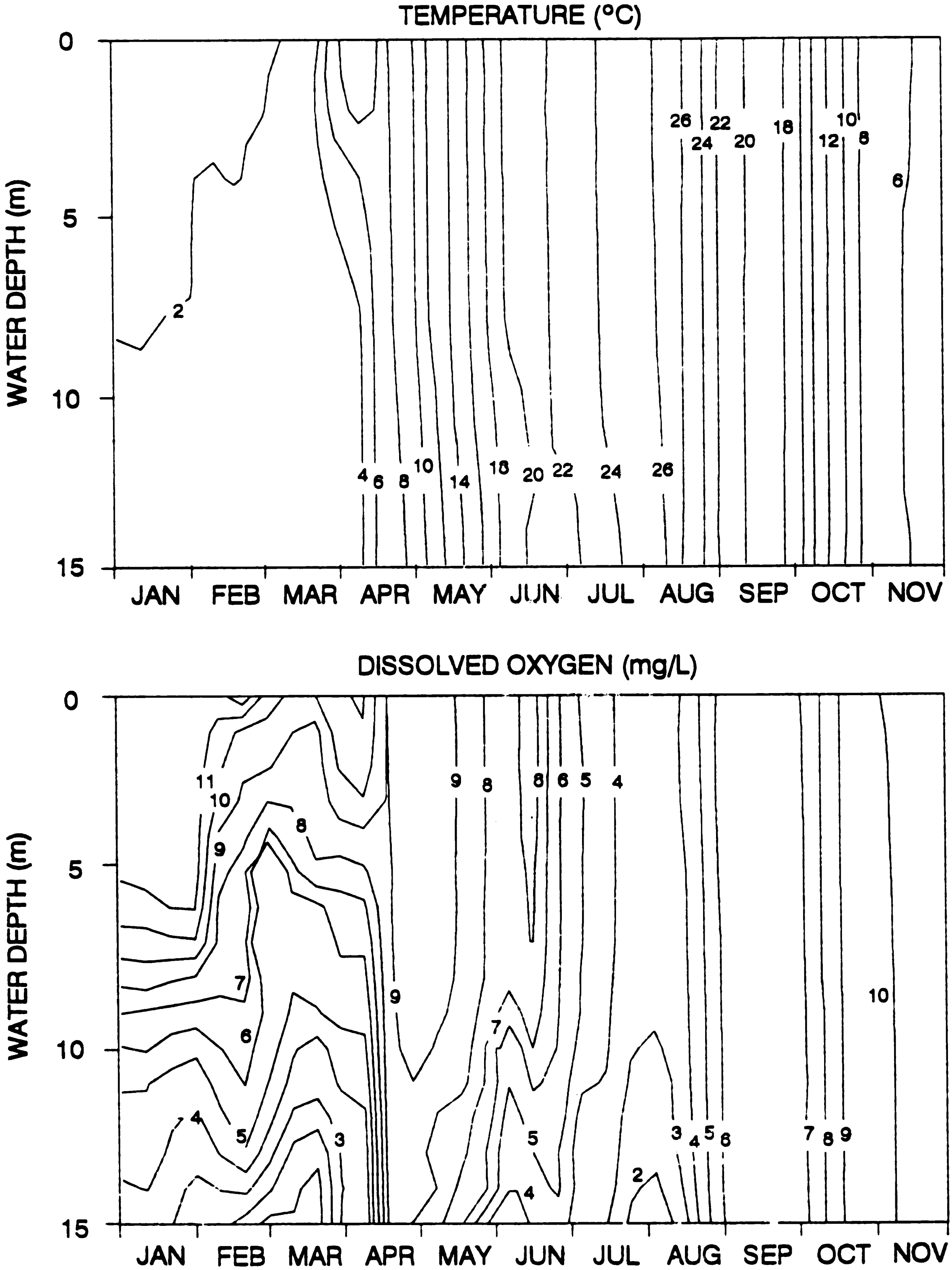


Figure 7. Temperature and dissolved oxygen iscpleths from the middle station of Mud Lake, Oakland County, Michigan during 1988.

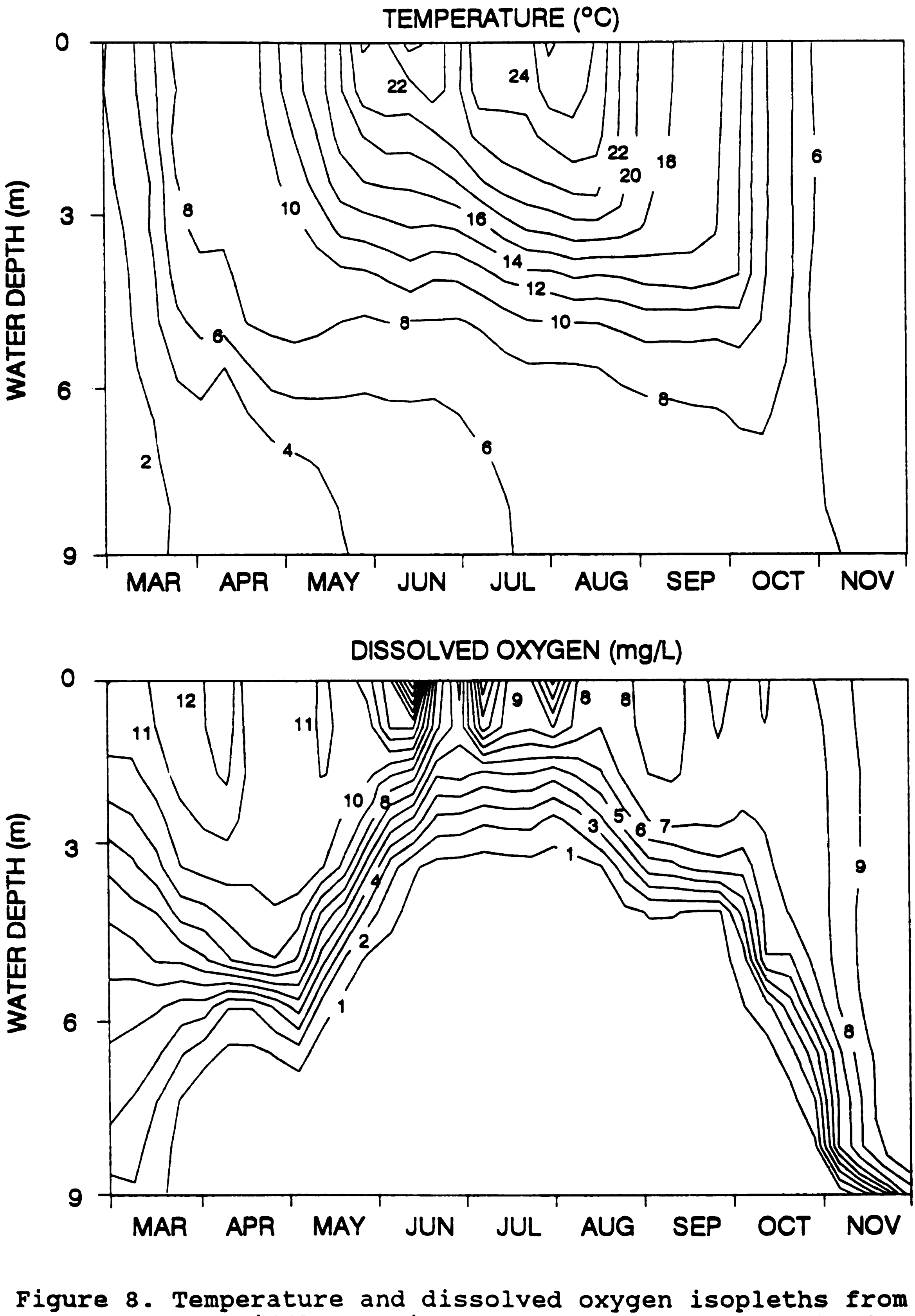


Figure 8. Temperature and dissolved oxygen isopleths from the middle station of Carpenter Lake, Oakland County, Michigan during 1988.

occurred on August 3, 1988, or 12 days earlier than Mud Lake's maximum.

Heat budget calculations showed that each lake gained approximately the same amount of heat from an assumed winter average of 4 °C to the time of air compressor operation on April 12, 1988. During that period, heat gain in Mud Lake was 2001 cal/cm^2, while in Carpenter Lake it was 1768 cal/cm^2. After start-up of the Lake Preserver, the Mud Lake water column gained heat more rapidly than Carpenter Lake. From April 12 to the time of temperature maxima in summer, Mud Lake gained heat at a rate of 86 cal/cm^2/day, while Carpenter Lake gained 41 cal/cm^2/day. Total heat gain from start-up to summer maxima was 10,787 cal/cm^2 and 4,590 cal/cm^2 for Mud and Carpenter Lakes, respectively. Summer heat income (Wetzel 1983) was calculated by summing the two different time periods for each lake. Summer heat incomes for Mud and Carpenter Lake during 1988 were 12,788 cal/cm^2 and 6,358 cal/cm^2, respectively.

Oxygen data for both Mud and Carpenter Lakes are presented in the lower panels of Figures 7 and 8, respectively. Prior to start-up of the air compressors, Mud Lake exhibited stratification with respect to oxygen content. Top to bottom differences often exceeded 7 mg/L. After start-up of the air compressors on April 12, 1988, oxygen differentials with depth were greatly reduced, although some stratification still persisted. Upper water values ranged from 4-9 mg D.O./L, while lower waters ranged from 2-4 mg D.O./L. As water temperatures continued to increase during June and July, dissolved oxygen concentrations decreased. Concomitantly, percent saturation of dissolved oxygen went from 80-90% after start-up to 40% or less by late July. Due to these deteriorating oxygen conditions, the flow rate of the compressors was increased from 27 m^3/hr to 40 m^3/hr on July 28, 1988. This new flow rate resulted in increased oxygen levels in the lower waters and eventually to conditions of increased, uniform oxygen concentrations throughout the water column by mid-August (Figure 7).

Similar to temperature, Carpenter Lake was strongly stratified with respect to dissolved oxygen content (Figure 8). Levels of oxygen below 1 mg/L occurred consistently at or below a depth of 3 m during mid-summer. Conversely, oxygen concentrations in the upper waters of Carpenter Lake during this period were often supersaturated (110-130%), values often exceeding 10 mg D.O./L. The maximum oxygen concentration occurred on June 14, 1988 when surface waters had 19.1 mg D.O./L and a corresponding saturation level of 235%.

Conductivity data during 1988 for both lakes are presented in Figure 9. Mud Lake showed little evidence of stratification with respect to conductivity. Values ranged from 1000 μmhos/cm in April and November to 1200 μmhos/cm in July. Carpenter Lake exhibited strong stratification with respect to conductivity. Values ranged from 800 to 2800

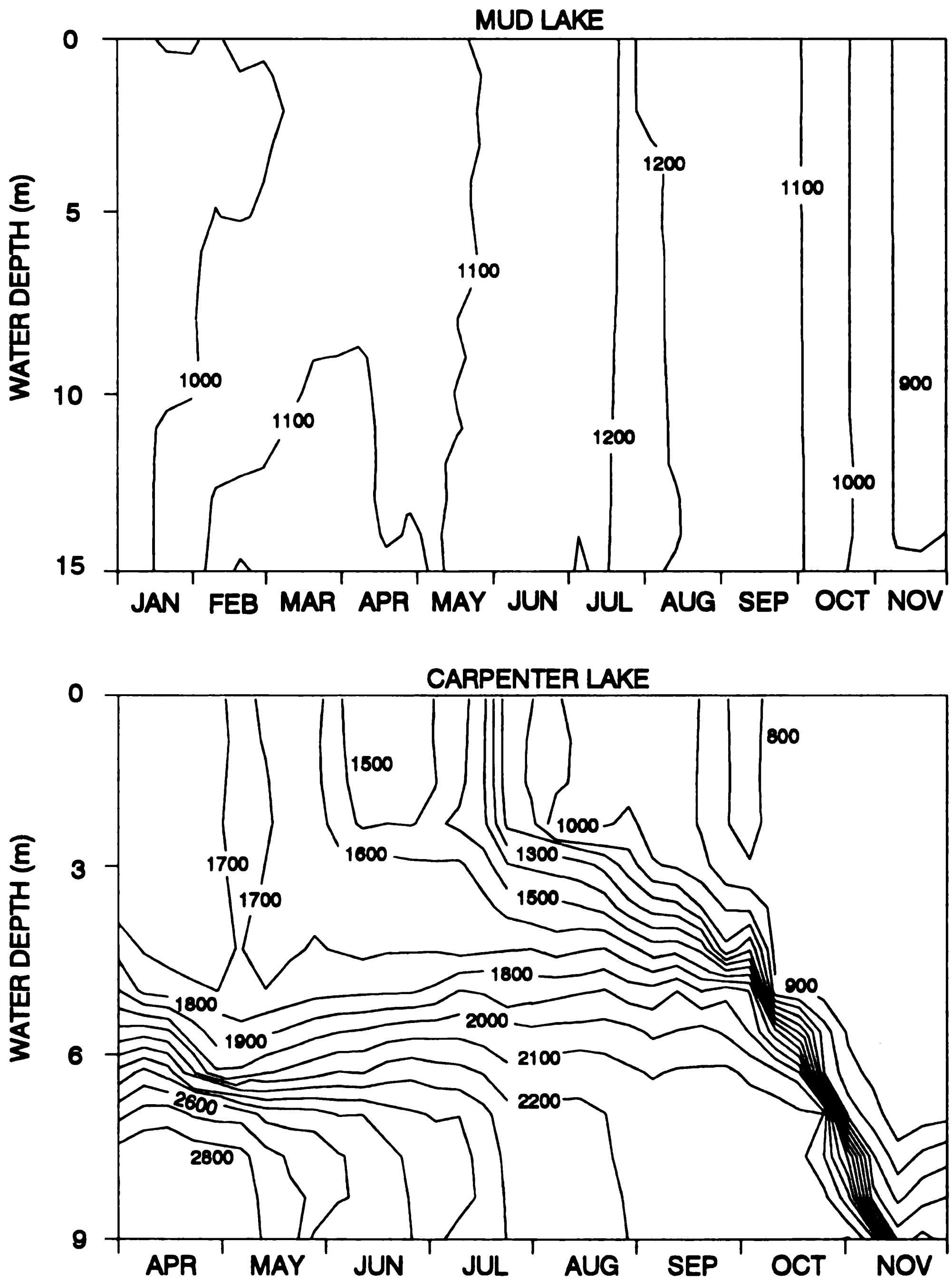


Figure 9. Conductivity isopleths from the middle stations of both Mud (upper) and Carpenter (lower) Lakes, Oakland County, Michigan during 1988.

μmhos/cm. In many instances values differed by as much as 1000 μmhos/cm between the surface and bottom waters of Carpenter Lake.

Values for chemical parameters, measured monthly during 1988 in Mud and Carpenter Lakes, are presented in Tables 1A and 1B. There was virtually no vertical and little temporal variation in concentrations of total phosphorus (TP) in Mud Lake during 1988. Concentrations ranged from 0.05 to 0.08 mg P/L. Except for April 12 values, nitrogen concentrations were similar between upper and lower waters. However, they did vary over time. Total nitrogen (TN) ranged from 1.03 mg N/L to a maximum of 1.39 mg N/L, which was measured in the surface waters on October 3, 1988. Organic compounds were the predominate form of nitrogen throughout the sampling season, accounting for 82 to 100% of the total. The small inorganic fraction was exclusively ammonia until November when nitrite+nitrate nitrogen accounted for approximately 40% of total inorganic nitrogen. Ammonia concentrations ranged from below levels of detection (<0.02 mg/L) to 0.20 mg NH_3-N/L, which was present in upper waters of Mud Lake on July 12.

The pH values in Mud Lake ranged from 7.11 to 8.42 and usually varied by less than 0.23 units vertically (Table 1A). However, on June 14 top to bottom variation was 0.46 units. Surface waters on this date had a pH of 8.42, the highest level recorded. Alkalinity values of Mud Lake showed little variation with depth, but did decrease over

Table 1A. Data for various chemical parameters for Mud Lake, Oakland County, Michigan during 1988. All values are means of duplicate composite samples expressed in mg/L, except for pH which is expressed in standard units and chlorophyll *a* which is in mg/m^3. "Top" is for water collected at a depth of 1 m below the water surface and "Bot" is for water collected 1 m above the sediment surface.

Date:		Apr 12	May 10	Jun 14	Jul 12	Aug 8	Sep 7	Oct 3	Nov 14
Total Phosphorus	Top	0.06	0.06	0.05	0.05	0.07	0.08	0.07	0.07
	Bot	0.06	0.06	0.05	0.05	0.07	0.08	0.07	0.07
Nitrate-Nitrite Nitrogen	Top	NDA	NDA	NDA	NDA	NDA	NDA	NDA	0.07
	Bot	NDA	NDA	NDA	NDA	NDA	NDA	NDA	0.08
Ammonia Nitrogen	Top	NDA	NDA	0.14	0.20	0.04	0.07	NDA	0.12
	Bot	0.13	NDA	0.08	0.19	0.11	0.10	0.07	0.11
Kjeldahl Nitrogen	Top	1.05	1.16	1.12	1.03	1.36	1.17	1.39	1.03
	Bot	1.16	1.10	1.08	1.15	1.35	1.16	1.29	1.06
Organic Nitrogen	Top	1.05	1.16	0.98	0.83	1.32	1.10	1.39	0.91
	Bot	1.03	1.10	1.00	0.96	1.24	1.06	1.22	0.95
Total Inorganic Nitrogen	Top	NDA	NDA	0.14	0.20	0.04	0.07	NDA	0.19
	Bot	0.13	NDA	0.08	0.19	0.11	0.10	0.07	0.19
Total Nitrogen	Top	1.05	1.16	1.12	1.03	1.36	1.17	1.39	1.10
	Bot	1.16	1.10	1.08	1.15	1.35	1.16	1.29	1.14
pH	Top	7.34	7.92	8.42	7.16	7.84	8.11	7.35	7.35
	Bot	7.11	7.73	7.96	7.60	7.66	7.98	7.13	7.40
Alkalinity (as $CaCO_3$)	Top	147	155	147	148	132	131	127	114
	Bot	154	153	149	147	134	130	128	115
Calcium (Ca)	Top	71.6	71.3	70.4	69.1	60.5	59.9	57.0	54.3
	Bot	72.3	71.1	71.2	67.7	60.4	60.2	57.3	54.5
Iron (Fe)	Top	0.13	0.07	0.10	0.05	0.14	0.09	0.08	0.16
	Bot	0.19	0.08	0.11	0.07	0.13	0.06	0.09	0.09
Magnesium (Mg)	Top	12.9	12.6	12.9	13.0	11.1	10.8	10.0	9.2
	Bot	12.8	12.5	13.0	12.7	11.0	10.9	10.0	9.3
Manganese (Mn)	Top	0.05	0.08	0.10	0.11	0.14	0.09	0.06	0.04
	Bot	0.20	0.09	0.22	0.14	0.17	0.09	0.06	0.04
Total Solids	Top	502.5	637.5	678.0	690.0	658.5	679.0	638.0	549.5
	Bot	518.0	632.0	685.0	690.0	684.5	703.0	634.0	534.0
Total Volatile Solids	Top	124.5	122.5	111.0	157.0	126.5	225.0	195.0	78.5
	Bot	108.0	98.0	116.5	153.0	128.5	221.0	250.0	124.5
Chlorophyll *a*	Top	20.0	20.0	42.7	10.7	6.7	24.0	13.3	5.3
	Bot	22.7	28.0	18.7	NDA	NDA	21.4	40.1	2.7

NDA = No Detectable Amount. Limits of detection in mg/L for those parameters for which there was no detectable amount were: Nitrate-Nitrite Nitrogen <0.01; Ammonia Nitrogen <0.02; Total Inorganic Nitrogen <0.02; and Chlorophyll *a* <0.001.

Table 1B. Data for various chemical parameters for Carpenter Lake, Oakland County, Michigan during 1988. All values are means of duplicate composite samples expressed in mg/L, except for pH which is expressed in standard units and chlorophyll *a* which is in mg/m^3. "Top" is for water collected at a depth of 1 m below the water surface and "Bot" is for water collected 1 m above the sediment surface.

Date:		Apr 12	May 10	Jun 14	Jul 12	Aug 8	Sep 7	Oct 3	Nov 14
Total Phosphorus	Top	0.07	0.06	0.13	0.07	0.08	0.04	0.07	0.10
	Bot	0.11	0.12	0.21	0.33	0.21	0.39	0.46	0.11
Nitrate-Nitrite Nitrogen	Top	NDA	NDA	NDA	NDA	NDA	NDA	NDA	0.22
	Bot	NDA	NDA	NDA	NDA	NDA	NDA	NDA	0.23
Ammonia Nitrogen	Top	NDA	NDA	0.05	0.08	0.26	0.10	0.06	0.32
	Bot	0.08	0.25	1.04	1.08	0.74	1.44	2.45	0.34
Kjeldahl Nitrogen	Top	1.01	1.09	2.77	1.93	1.57	1.26	1.07	1.09
	Bot	1.81	2.58	1.57	2.42	2.17	2.51	3.29	1.11
Organic Nitrogen	Top	1.01	1.09	2.72	1.85	1.31	1.16	1.01	0.77
	Bot	1.73	2.33	0.53	1.34	1.43	1.07	0.84	0.77
Total Inorganic Nitrogen	Top	NDA	NDA	0.05	0.08	0.26	0.10	0.06	0.54
	Bot	0.08	0.25	1.04	1.08	0.74	1.44	2.45	0.57
Total Nitrogen	Top	1.01	1.09	2.77	1.93	1.57	1.26	1.07	1.31
	Bot	1.81	2.58	1.57	2.42	2.17	2.51	3.29	1.34
pH	Top	7.88	8.22	9.58	9.06	8.56	8.21	7.42	7.34
	Bot	6.98	7.30	7.91	6.98	7.74	7.42	6.93	7.44
Alkalinity (as $CaCO_3$)	Top	149	141	103	107	103	113	101	127
	Bot	147	144	144	151	137	150	154	127
Calcium (Ca)	Top	78.8	73.4	53.4	53.4	48.0	49.4	43.0	46.7
	Bot	80.8	77.9	73.7	71.6	60.5	63.2	65.6	50.5
Iron (Fe)	Top	0.21	0.09	0.22	0.10	0.29	0.10	0.41	0.24
	Bot	0.22	0.10	0.15	0.18	0.38	0.16	0.30	0.25
Magnesium (Mg)	Top	12.8	12.7	11.4	11.7	8.1	7.9	6.3	7.2
	Bot	12.9	12.7	11.5	12.3	10.1	9.8	9.9	7.8
Manganese (Mn)	Top	0.09	0.13	0.07	0.07	0.08	0.16	0.03	0.11
	Bot	0.27	0.50	0.95	1.01	0.95	1.05	1.23	0.12
Total Solids	Top	876.0	939.0	854.5	888.0	592.5	589.0	440.0	423.5
	Bot	930.0	1029.0	1039.0	1071.0	856.5	903.0	866.0	455.0
Total Volatile Solids	Top	119.0	155.5	139.5	143.0	136.0	230.0	249.0	80.5
	Bot	114.0	131.5	148.5	166.0	125.5	225.0	190.0	99.0
Chlorophyll *a*	Top	25.4	5.3	126.8	37.4	34.7	17.4	8.0	NDA
	Bot	8.0	13.4	6.7	29.4	60.1	28.0	40.0	NDA

NDA = No Detectable Amount. Limits of detection in mg/L for those parameters for which there was no detectable amount were: Nitrate-Nitrite Nitrogen <0.01; Ammonia Nitrogen <0.02; Total Inorganic Nitrogen <0.02; and Chlorophyll *a* <0.001.

time from 147-155 mg $CaCO_3$/L in April-July to a low of 114 mg $CaCO_3$/L in November. Calcium and magnesium followed a similar trend (Table 1A), with calcium declining from a high value of 72.3 mg/L in April to a low of 54.3 mg/L in November. Magnesium concentrations declined from a high of 13.0 mg/L in June to 9.2 mg/L in November. Iron and manganese showed some variation with depth and time, concentrations ranging from 0.09-0.19 for iron and from 0.04-0.20 for manganese.

Concentrations of total solids in Mud Lake varied little with depth. The greatest difference between top and bottom waters equalled 26 mg/L on August 8. However, there was some temporal variation, with lowest concentrations occurring at the beginning and end of the sampling period during 1988. Total solid concentrations ranged from 502.5 mg/L in April to a high of 703.0 mg/L in lower waters in September. By November, values had declined to 534.0 mg/L. Total volatile solids followed a similar trend. Typically, other elements detected in the waters of Mud Lake were fairly consistent with both depth and time (Table 1A).

Conversely, Carpenter Lake displayed fairly large variations, both vertically and temporally, for most chemical parameters during 1988 (Table 1B). Concentrations of epilimnetic total phosphorus (TP) decreased from a high of 0.13 mg P/L in June to a low value of 0.04 mg P/L in September, while hypolimnetic values increased over time (April to October) from 0.11 to 0.46 mg P/L. In October, the greatest top-to-bottom total phosphorus difference of 0.39 mg P/L occurred. By turnover in November, there was very little vertical difference in TP concentrations or other chemical parameters. Except for June, total nitrogen (TN) concentrations in lower waters were always greater than in upper layers. Epilimnetic TN values ranged from 1.01 mg N/L in April to a maximum of 2.77 mg N/L in June. These waters were dominated by the organic fraction, typically representing greater than 90% of the total nitrogen. Hypolimnetic values ranged from 1.34 to 3.29 mg N/L. In these lower waters, 34-74% of the total nitrogen was in the inorganic form. In turn, with the exception of the November sampling period, ammonia was the only form of inorganic nitrogen that was detected. Upper water concentrations of ammonia ranged from below levels of detection (<0.02 mg/L) to 0.32 mg NH_3-N/L, while in lower waters, concentrations ranged from 0.08 to 2.45 mg NH_3-N/L. The percent of total unionized ammonia in the hypolimnion of Carpenter Lake was always less than 2%. However, in the epilimnion of Carpenter Lake during June, July, and August, a fairly significant fraction of total ammonia was in the unionized form, 63%, 38%, and 19% respectively. This was due to the relatively warm waters (22-26 °C) and high pH (8.56 to 9.58).

The pH values in Carpenter Lake (Table 1B) varied widely over the course of the summer with respect to time and depth. Epilimnetic pH varied from 7.34 on November 14 to a high of 9.58 on June 14, while hypolimnetic pH varied from a low of 6.93 on October 3 to a high of 7.91, which also occurred on June 14. The largest top to bottom variation in pH occurred on July 12 when the difference was 2.08 units. After turnover in November, vertical variation in pH was only 0.10 units. There was a decreasing trend in alkalinity in the epilimnion of Carpenter Lake during 1988 from a high of 149 mg $CaCO_3$/L in April to a low of 101 mg $CaCO_3$/L by October. In the hypolimnion, alkalinity tended to increase slightly from an April value of 147 mg $CaCO_3$/L to an October value of 154 mg $CaCO_3$/L. The largest vertical difference in alkalinity occurred in October, when the hypolimnion was 53 mg $CaCO_3$/L greater than that of the epilimnion. Calcium and magnesium followed the same trend as eplimnetic alkalinity in which there was an overall decrease over time. However, for these species the trend was the same in both the epilimnion and hypolimnion. Iron varied with both depth and time but not in any consistent pattern. Values ranged from a low of 0.09 mg/L in the epilimnion on May 10 to a high of 0.41 on October 3. Manganese varied dramatically with depth and showed increasing concentrations in the lower waters up until turnover in the fall. Upper water values ranged from 0.03-0.16 mg/L, while in lower waters concentrations ranged from a low of 0.12 mg/L to a high of 1.05 mg/L.

Total solids (Table 1B) were consistently higher in the lower waters of Carpenter Lake, as compared to the values found in the upper layers. Concentrations increased in both the upper and lower waters from April to July maxima, which were 888.0 mg/L and 1071.0, respectively. After the July maxima, the values decreased to minima of 423.5 mg/L in upper waters and 455.0 mg/L in lower waters which were observed in November. Total volatile solids followed a similar temporal pattern but concentrations were nearly the same for top and bottom water, except for samples collected on October 3.

The limit of visibility, as measured by Secchi disk is presented in Figure 10. Values in Mud Lake ranged from 1.0 to 3.5 m and in Carpenter Lake from 0.5 to 2.9 m. In general, the depth of light penetration was greater in Mud Lake than in Carpenter Lake. The overall average Secchi disk depth in Mud Lake was 1.9 m and in Carpenter Lake 1.3 m. The Mud Lake value was significantly greater ($p = 0.0008$, Wilcoxin Signed-Ranks Matched-Pairs Test) than that of Carpenter Lake.

Values for chemical oxygen demand in October were 31.5 mg/L for the surface of Mud Lake and 32.5 mg/L for the bottom waters. The values for the upper and lower waters of Carpenter Lake were 10.1 mg/L and 31.0 mg/L, respectively.

BENTHOS

Table 2 summarizes the results of the preliminary survey conducted to determine the sample size to be used for the collection of the benthic organisms.

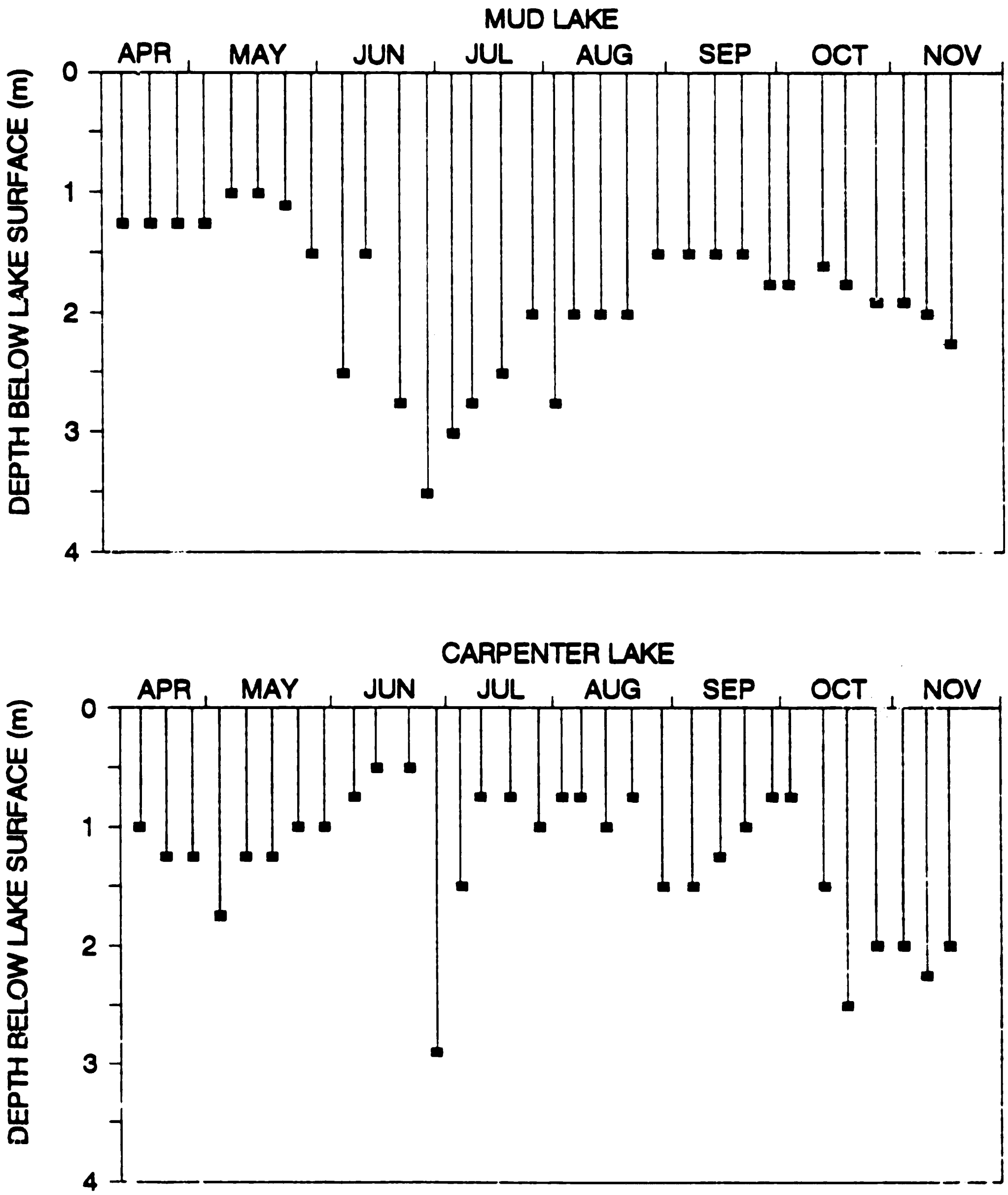


Figure 10. Secchi disk reading at the middle stations of both Mud (upper) and Carpenter (lower) Lakes, Oakland County, Michigan during 1988.

Table 2. Results of the preliminary survey on Mud and Carpenter Lakes, Oakland County, Michigan for the determination of sample size for collection of chironomid larvae.

Lake	Depth strata	Precision	Number of Samples
Carpenter	4 m	20%	9
Carpenter	8 m	30%	7
Mud	4 m	40%	7
Mud	8 m	20%	6
Mud	12 m	20%	6
Mud	16 m	20%	4

An error in the estimate of the mean value equal to 20% of the true value was desired. However, this was not always possible due to constraints of time and personnel for processing samples in the laboratory. Due to this fact estimates for two strata, Carpenter 8 m and Mud 4 m, were not as precise as the others.

Almost all benthic organisms collected were either chironomids or *Chaoborus*. No attempt was made to identify all chironomid larvae beyond family. However, random individuals were identified to species, and in every case they were identified as *Chironomus plumosus*.

The equation which best describes the relationship between chironomid dry weight and total length is the exponential equation presented in Figure 11. Using this equation estimates of chironomid biomass were obtained.

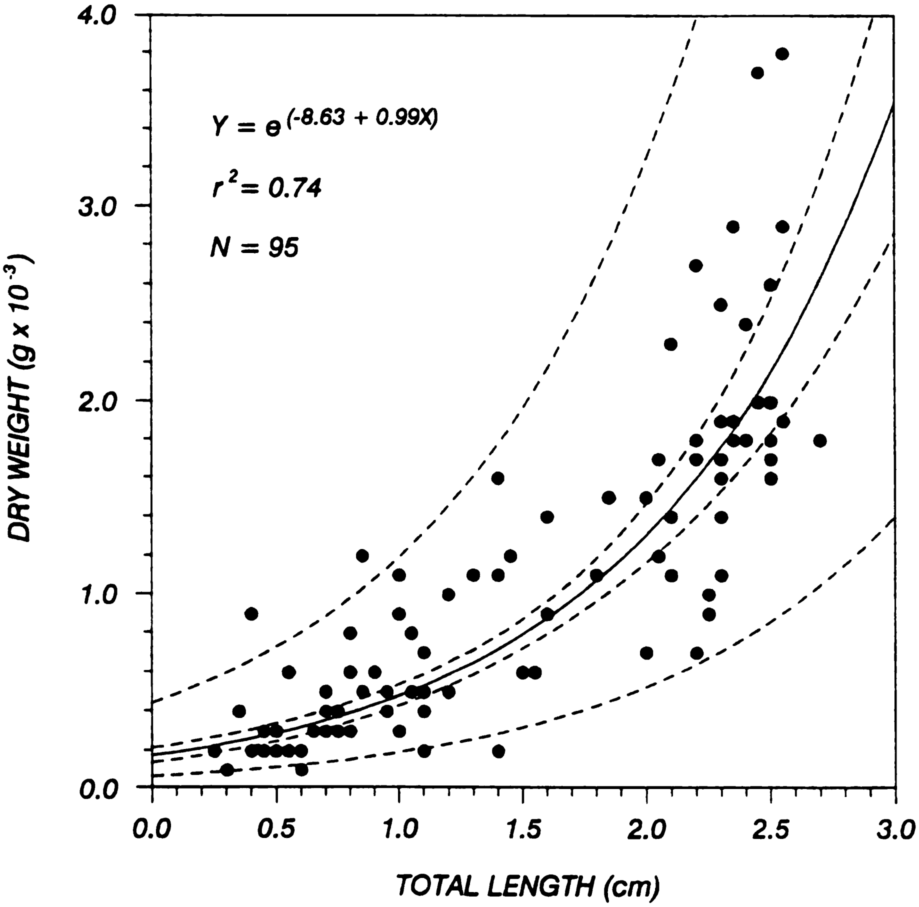


Figure 11. Length-weight relationship of chironomid larvae collected from Mud and Carpenter Lakes, Oakland County, Michigan during 1988.

Density and biomass of the chironomids are presented in Figures 12 and 13 respectively. Chironomids were always greater in Mud Lake in both density and biomass than in Carpenter Lake during 1988. In general, abundance of this organism in Mud Lake was greatest in June and then declined throughout the remainder of the season with a slight increase in biomass on the last sampling date in October. Initially chironomid density and biomass varied with depth in Mud Lake with the largest numbers occurring in the lowest strata. By late July, there were no significant differences between strata. In Carpenter Lake, the chironomid abundance was very sparse and by late August none were collected from the deepest strata.

Very few other species of macroinvertebrates were noted in samples collected from Mud Lake and when present were in low abundance. Those noted included Oligochaeta (Annelida) and *Chaoborus* (Diptera).

Chaoborus was the predominate organism in Carpenter Lake, although its numbers were very reduced. Oligochaetes were found occasionally in the samples.

PHYTOPLANKTON AND ZOOPLANKTON

Percent occurrence of the various algal divisions collected from Mud and Carpenter Lakes during 1988 are shown in Figure 14. There were no data for May from Carpenter Lake because of sample loss. Mud Lake was dominated by diatoms (Bacillariophyta) during the spring (April-June) and green

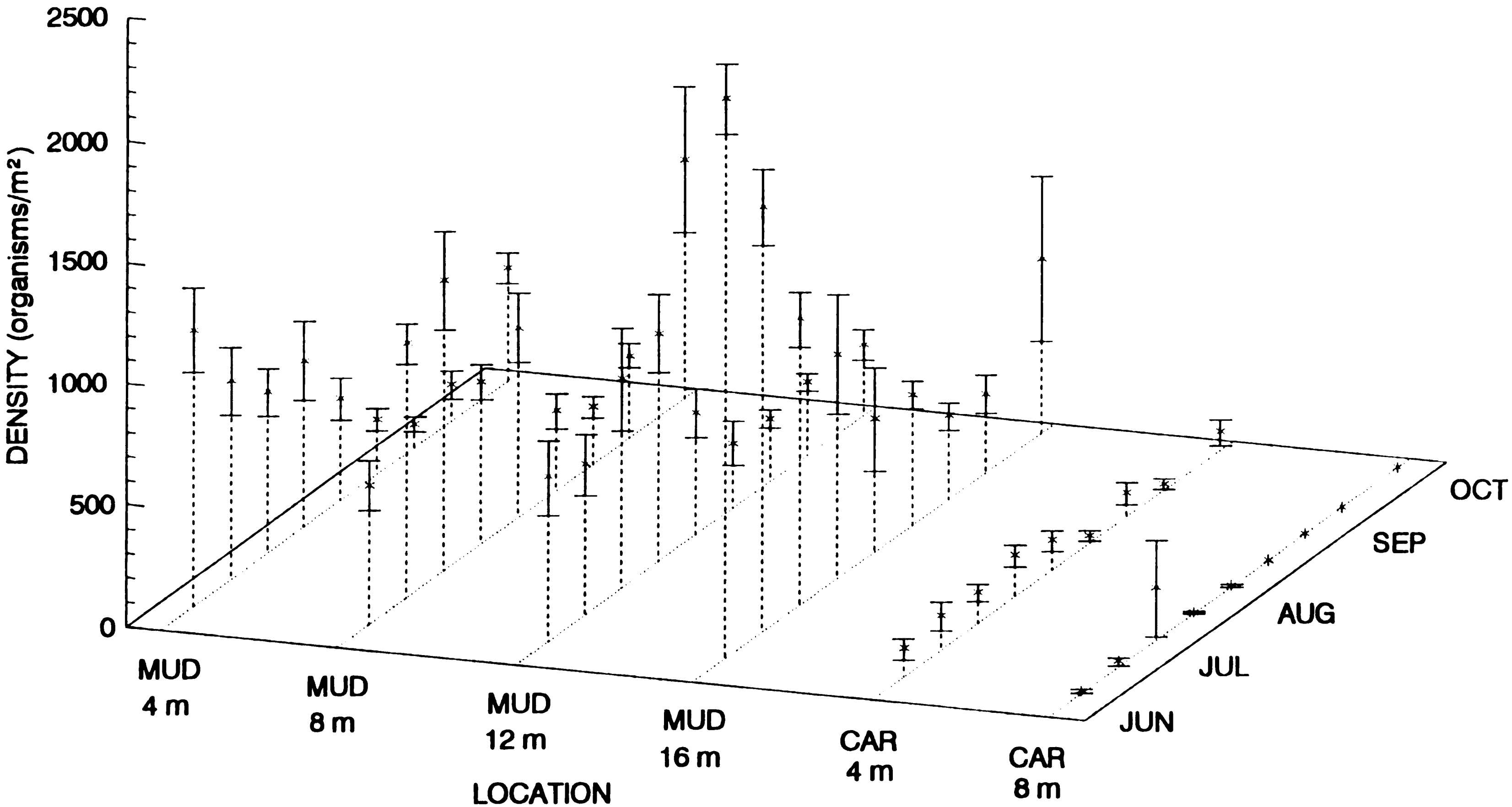


Figure 12. Changes in chironomid density measured over time from different depth strata of both Mud (MUD) and Carpenter (CAR) Lakes, Oakland County, Michigan during 1988.

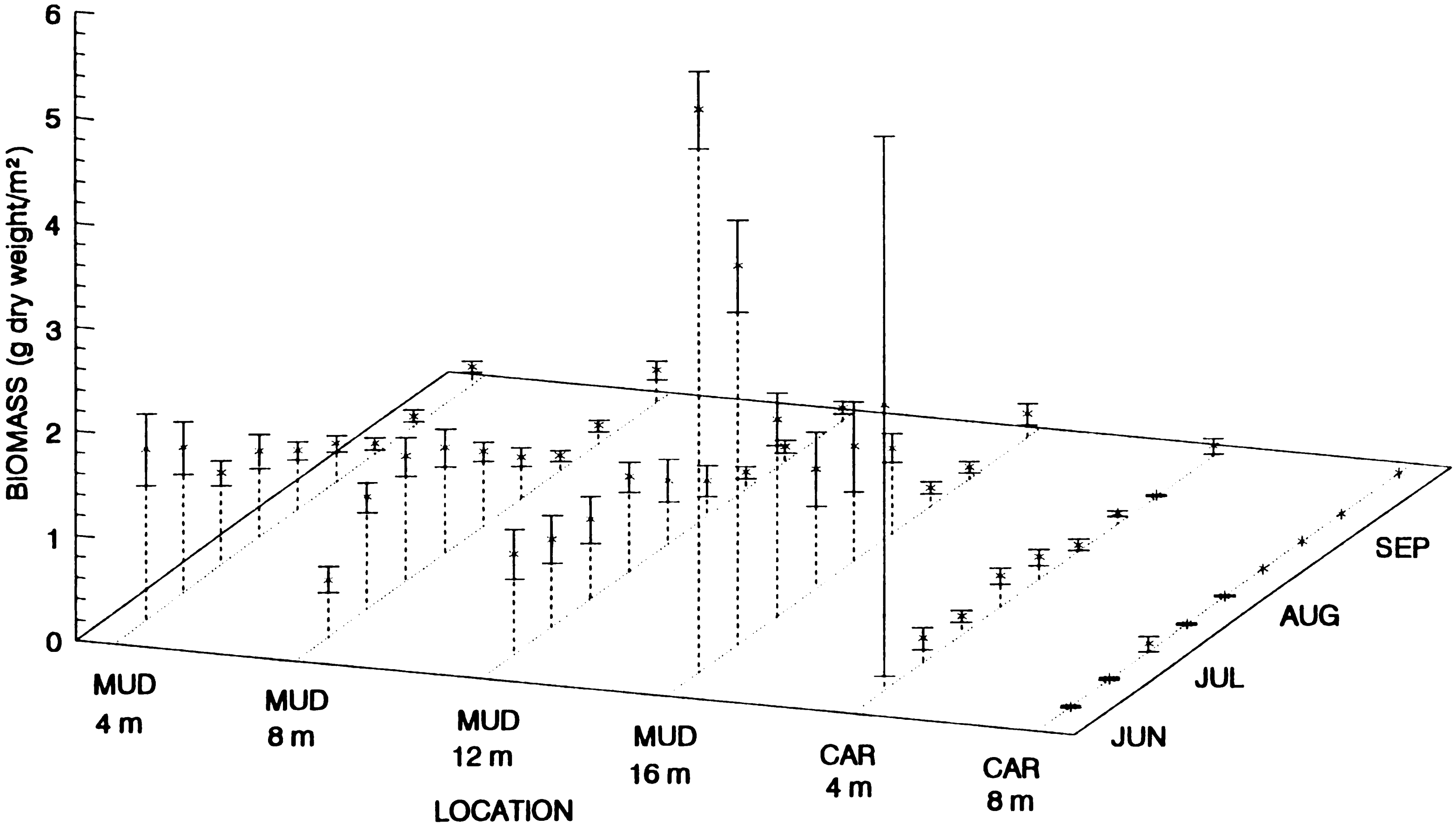


Figure 13. Changes in chironomid biomass measured over time from different depth strata of both Mud (MUD) and Carpenter (CAR) Lakes, Oakland County, Michigan during 1988.

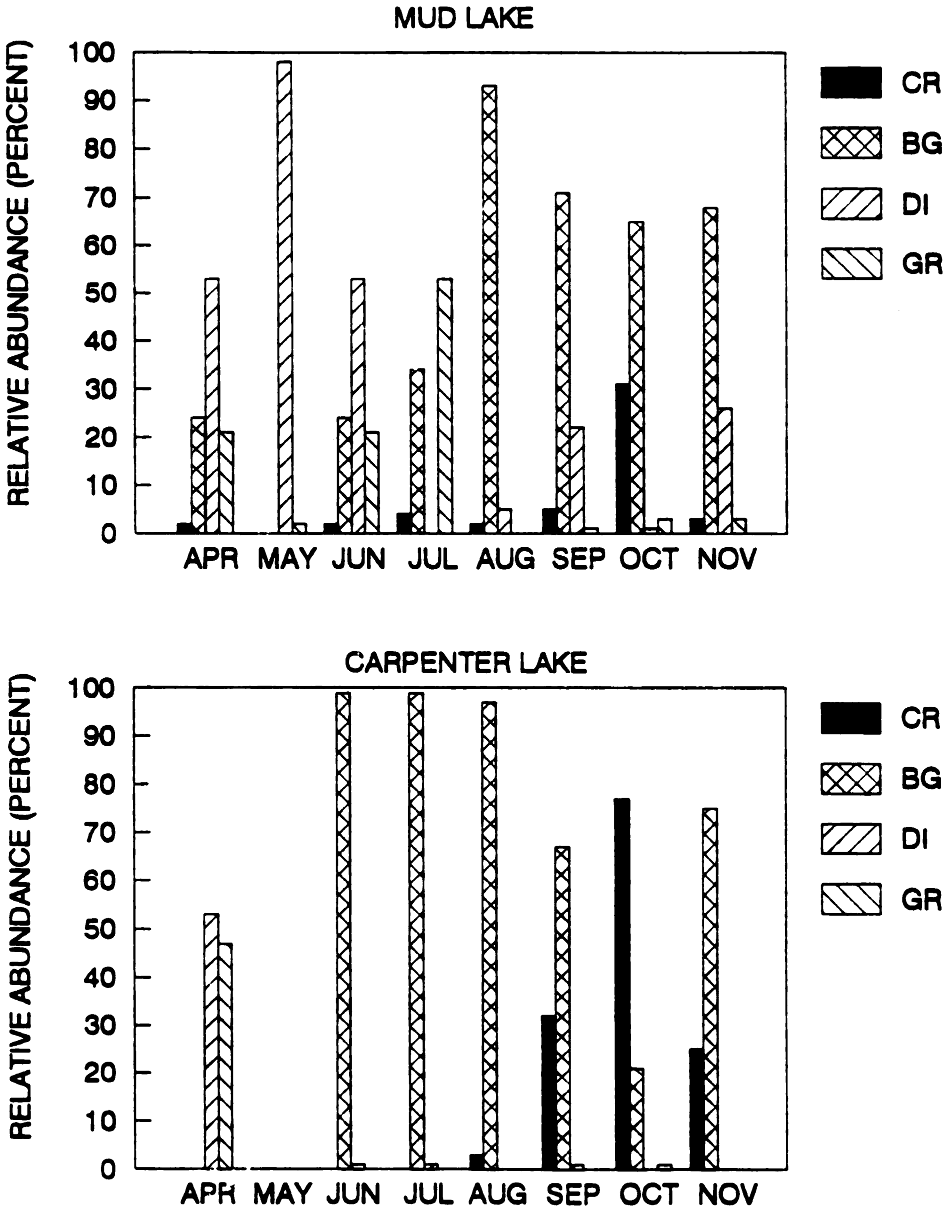


Figure 14. Relative abundance of phytoplankton in Mud (upper) and Carpenter (lower) Lakes, Oakland County, Michigan during 1988. Key to symbols: Cryptomonads (CR), blue-green algae (BG), diatoms (DI), and green algae (GR).

algae (Chlorophyta) during July. *Asterionella, Fragilaria, Cyclotella, Melosira, Cocconeis,* and *Tabellaria* were diatoms that were commonly observed. *Ankistrodesmus*, a small unicellular organism, was the most common green alga during July. From August through November, blue-green algae (Cyanophyta) dominated the phytoplankton. The most common blue-green algae were *Aphanizomenon flos-aquae*, *Anabaena*, *Microcystis*, and *Oscillatoria*.

There was little algal diversity in Carpenter Lake. April was dominated by only two taxa, diatoms and green algae, which occurred in equal abundance. Blue-greens almost exclusively dominated the phytoplankton during June, July and August. They remained in high abundance throughout the fall, only being exceeded by cryptomonads during October. The genera of the various algal divisions that were observed in Carpenter Lake were the same as those of Mud Lake, which are listed above.

Carpenter Lake had greater numbers of zooplankton than Mud Lake (Figure 15), except for the months of June and September. Figure 16 shows the distribution of some major types of zooplankton in both Mud and Carpenter Lakes. Rotifers were dominant in both lakes in May and June. Rotifers that were observed included *Branchionus*, *Kellicottia*, *Keratella*, and *Filinia*. In the month of July, the rotifer populations in Mud Lake declined dramatically while in Carpenter Lake they were still present in large numbers. Cladoceran populations reached their peak in September in Mud Lake while in Carpenter

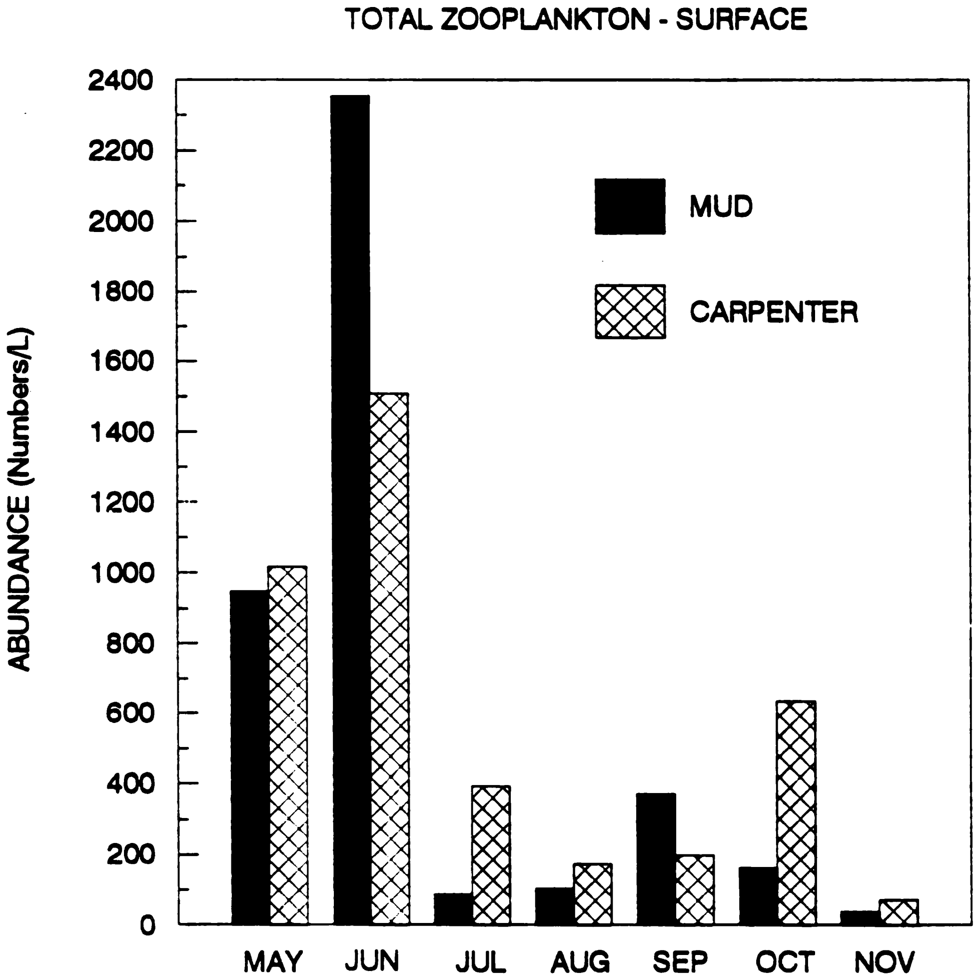
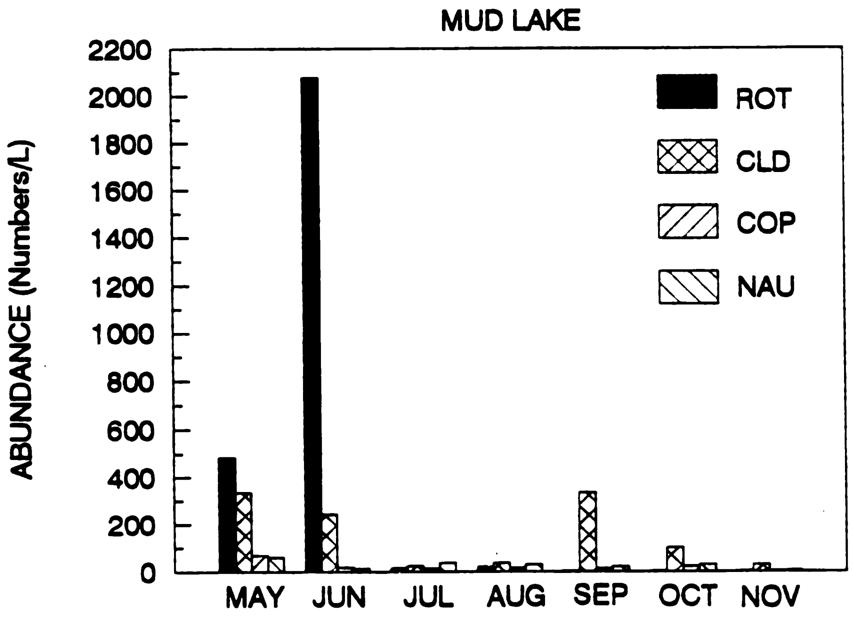


Figure 15. Comparisons of total zooplankton abundance collected from surface waters in both Mud and Carpenter Lakes, Oakland County, Michigan during 1988.



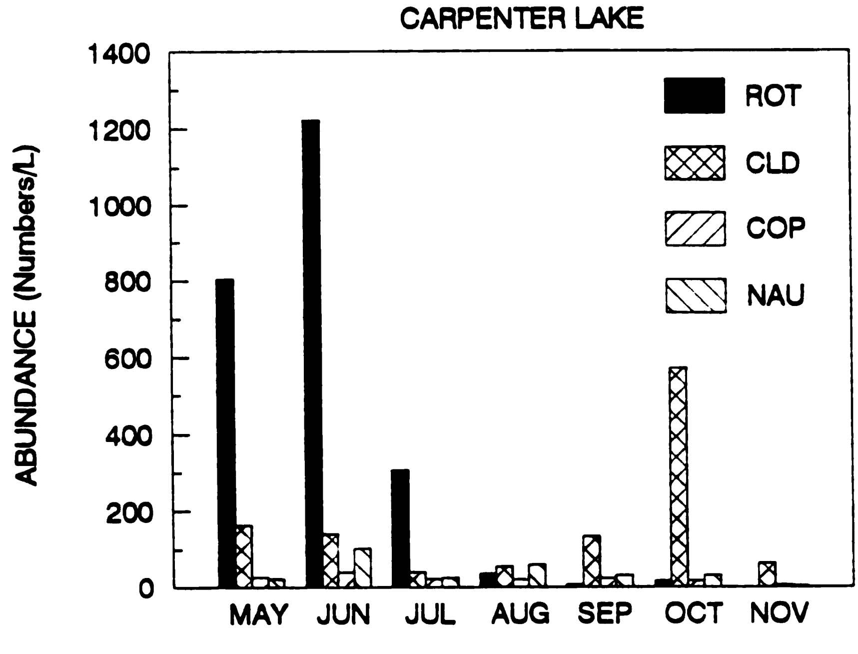


Figure 16. Abundance of zooplankton in Mud (upper) and Carpenter (lower) Lakes, Oakland County, Michigan during 1988. Key to symbols: Rotifers (ROT), cladocera (CLD), adult copepods (COP), and copepod naupulii (NAU).

Lake they did not peak until October. The most common cladocerans were *Daphnia* and *Bosmina*. Copepods (*Diacyclops* and *Limnocalanus*) and copepod naupulii numbers were generally small in both Mud and Carpenter Lakes.

DISCUSSION

PHYSICAL-CHEMICAL PARAMETERS

During summer in temperate regions of the world, lakes typically develop temperature differences in relation to depth, which prevent mixing of lower and upper waters due to density differences (Wetzel 1983). The Lake Preserver was designed to prevent such thermal stratification, allowing for continuous mixing of the water column, which in turn would maintain adequate levels of dissolved oxygen and create isochemical conditions throughout the lake.

Pastorok and Grieb (1984) summarized an extensive amount of information about the effects of destratification. They concluded that physical-chemical parameters that were most influenced by lake circulation were temperature, dissolved oxygen, iron, manganese, ammonia, pH and Secchi disk depth.

Lack of in-depth studies of conditions of Mud Lake prior to initiation of destratification preclude quantitative before and after comparisons of the parameters studied. However, many trends were apparent.

Mud Lake did not thermally stratify during 1988 as indicated by isothermal conditions from April through November (Figure 7). This had not been the situation in previous years (Liston et al. 1980, 1985), nor was it the case for Carpenter Lake during 1988 (Figure 8). Pastorok and Grieb (1984) used a temperature differential of less than 3 °C from top to bottom waters as an indication of successful destratification. Using this criterion, it can be concluded that Mud Lake was destratified by the operation of the Lake Preserver. Due to destratification, the entire water column continued to warm from spring to summer, and by August was 26 °C (Figure 7). In previous years temperatures of the lower waters of Mud Lake were approximately 5-6 °C during mid-summer (Liston et al. 1980, 1985), almost 20 °C cooler than in 1988. These elevated temperatures of lower water are significant for many interactions that take place within the sediments and across the sediment-water interface.

Temperature is a measure of heat intensity and is an important regulator of many physiological functions, including lake metabolism and productivity (Wetzel 1983). The temperature of a water body is used in calculating the relative heat content of a lake. Wetzel (1983) defines the heat budget, or content, of a lake as a measure of the heat storage capacity under existing conditions of morphometry and climate. Using data presented by Liston et al. (1985) and values for 1988, calculations were made of summer heat income, where summer heat income is defined as the amount of heat necessary to raise the temperature of the lake from an isothermal condition of 4 °C to the maximum observed summer temperature (Wetzel 1983). In 1988 the summer heat income of Mud Lake was 12,788 cal/cm^2, over twice that of the 1985 summer heat income of 5,870 cal/cm^2. However, the summer heat income of Carpenter Lake did not vary significantly between 1985 and 1988, which were 6,379 and 6,358 cal/cm^2 respectively. This indicates that there was little year to year variation and that it was the operation of the Lake Preserver that increased the heat content of Mud Lake.

Historically, both Mud and Carpenter Lakes had anoxic hypolimnions during the summer (Liston et al. 1980, 1985). This lack of oxygenated water in the lower depths restricts the distribution of fishes to upper waters, excludes aerobic benthic organisms from these regions, and allows for release of nutrients from the sediments via oxidation-reduction pathways (Mortimer 1941, 1942; Wetzel 1983). Similar to previous years, Carpenter Lake developed an anaerobic hypolimnion during 1988 (Figure 8). However, actions of the Lake Preserver resulted in dissolved oxygen being maintained throughout the water column of Mud Lake for 1988 (Figure 7).

When the Lake Preserver was put into operation on April 12, 1988, Mud Lake was stratified with respect to dissolved oxygen concentration. Upper water values were 10-11 mg D.O./L, whereas at lower depths concentrations were 3 mg/L or less. Within one week, dissolved oxygen differentials were virtually non-existent (Figure 8). However, by late May, stratification was evident with respect to oxygen, and the overall oxygen content of the lake was beginning to decrease, partially as a function of increasing water temperature, but more likely as a result of high oxygen demand. This trend continued throughout June and July. By early August, dissolved oxygen concentrations were 4 mg/L at the surface and less than 2 mg/L in lower waters. These conditions necessitated that the mixing rate of the Lake Preserver be increased. Previous work (Fast 1981) indicated that there is little enhancement of oxygen concentration directly from air bubbles of destratification devices and that increased dissolved oxygen is the result of increased surface diffusion due to turbulent mixing. It was apparent that the flow rate of the compressors (27 m^3/hr) was insufficient to maintain adequate dissolved oxygen concentrations for the conditions which were present in Mud Lake during May and June 1988. Therefore, the flow rate from the compressors was increased to 40 m^3/hr on July 28. This flow rate finally eliminated vertical dissolved oxygen differentials, and increased the overall concentration of oxygen within Mud Lake thereafter.

Oxygen values in the upper waters of Mud Lake were often lower than those of Carpenter Lake. Oxygen values in Mud Lake were frequently undersaturated with respect to atmospheric equilibria, while in Carpenter Lake supersaturated values were common. Although the actions of the Lake Preserver may have had an effect on the lack of supersaturation in Mud Lake due to turbulent mixing, previous studies indicated that Mud Lake did not exhibit supersaturation to the same degree as Carpenter Lake (Liston et al. 1980, 1985; Batterson et al. 1987).

Both Mud and Carpenter Lakes are characterized by thick layers of detrital sediments throughout their basins. Decomposition of these organically rich materials creates a high oxygen demand, which has resulted in anoxic hypolimnions (Liston et al. 1980, 1985) or as in the case of Mud Lake, low levels of dissolved oxygen concentrations in lower waters even after thermal destratification. For Mud Lake the oxygen demand was exacerbated by elevated water temperatures during summer months (mentioned above), as well as by the mixing process.

Constant circulation resulted in uniform levels of chemical oxygen demand (COD) in the upper and lower waters of Mud Lake during 1988 and were as high as those found in the hypolimnion of Carpenter Lake. The epilimnion of Carpenter lake had a COD which was only one-third the level of the COD in the hypolimnion. This high level of COD in Mud Lake's surface is significant in terms of the potential adverse lake reaction to failed operation of the Lake Preserver device. If the high levels of COD are being maintained due to the operation of the Lake Preserver device, and that device suddenly fails to operate, the lake will quickly become devoid of oxygen. This contention is supported by the data from the first year of the Lake Preserver's operation (Batterson et al. 1987). After a failure of the air compressors, the oxygen content declined dramatically, and recovery of the lake to pre-failure conditions took a prolonged period of time (over 30 days).

Eutrophic lakes, which stratify and develop anoxic hypolimnions, commonly show marked changes in concentrations of many chemical parameters with respect to depth and time. Nutrients, which are taken up by the algae and other trophic levels, rain down on lower waters as the organisms senesce and die, thus lowering concentrations in the epilimnion and increasing them in the hypolimnion. In addition, as the sediment-water interface becomes anaerobic, many chemical species, especially phosphorus, are released from the sediments into the water column (Wetzel 1983). This continues until the lake turns over. During 1988, Carpenter Lake showed these changes as indicated by the data in Table 1B. Concentrations of phosphorus, nitrogen, alkalinity, calcium, magnesium, and total solids followed these trends. Mud Lake, on the other hand, did not display these characteristics. Concentrations of most chemical parameters varied little with depth or time (Table 1A). The isochemical conditions of Mud Lake are attributed to the operation of the Lake Preserver. Although it appears that the Lake Preserver's actions in Mud Lake decreased the release of phosphorus from the sediments, the phosphorus concentrations remaining in the upper waters were still sufficient for high levels of productivity. The seasonal average for total phosphorus was 0.06 mg/L, while total nitrogen averaged 1.17 mg/L. Using criteria that Wetzel presents in Table 15-10 (Wetzel 1983, page 402), these seasonal averages would classify Mud Lake as being hypereutrophic. It is highly unlikely that continued operation of the Lake Preserver would improve the lake's trophic status due to the rich reserve of nutrients that are contained within the basin of Mud Lake.

Water transparency (Secchi disk) was similar to values in previous years (Liston et al. 1980, 1985; Batterson et al. 1987) and no definitive conclusions regarding the effects of the Lake Preserver on water clarity can be made.

BENTHOS

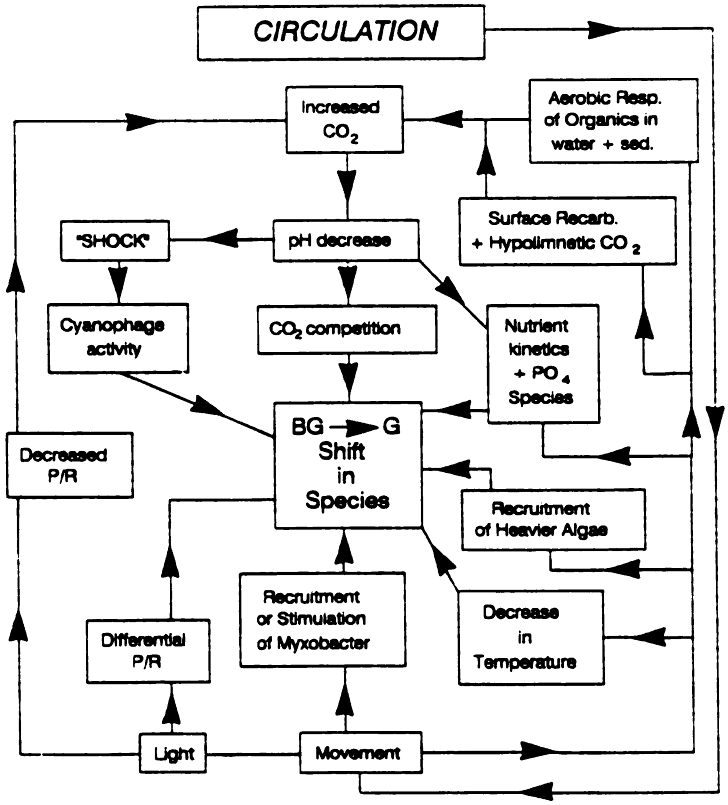
Previous sampling indicated that Chironomidae were the most common taxa in Mud Lake, although they were absent from the profundal zone during mid-summer (Liston et al. 1980, 1985). However, the previous sampling methods were inadequate for making valid quantitative estimates of density and biomass. The sampling scheme that was implemented in 1988 overcame these limitations and allowed for reliable estimates of chironomid populations.

Data presented in Figures 12 and 13 clearly show that during 1988, Mud Lake was more suitable for chironomids than was Carpenter Lake. Density and biomass were always low within Carpenter Lake, and by late August they had completely disappeared from depths 8 m or greater. A number of reasons could explain their absence: They may have been preyed upon by fish (Hayne and Ball 1955); they may have preferentially migrated to more suitable habitats within the lake such as littoral areas (Eggleton 1931, 1934; Davies 1976); they may have emerged; or they may have died due to intolerable conditions such as anoxia. But no matter what the reason, chironomids were absent from some portions of Carpenter lake during 1988. However, in Mud Lake, populations of chironomids were present at all times and from all depths, including deepwater regions during 1988. This was a dramatic change from 1980 and 1985, when no benthic organisms were collected from lower depths during the summer (Liston et al. 1980, 1985). In the 4 m and 16 m strata of Mud Lake, populations started off at high levels of density and biomass, declined through early-September, and then began to increase. The increase in late-September in total biomass was due to increasing numbers of small chironomid larvae. Appearance of these small midges indicated that emergence, mating, and recolonization had occurred. Within the 8 m and 12 m strata, density and biomass increased from June through July, and decreased thereafter. The mid-season arrivals, which were small in size, were most likely due to recolonization after adults had emerged and mated. That emergence could explain the decline in the populations at the 4 m and 16 m strata.

The data for chironomids in Mud Lake during 1988 indicate that the Lake Preserver was instrumental in increasing their abundance and distribution. This is attributed to oxygenated conditions throughout the water column due to operation of the Lake Preserver. In addition, the Lake Preserver also provided conditions suitable for emergence and recolonization due to elevated temperatures. Evidence suggests that there were at least two emergence and possible recolonization periods during the five months of sampling. The increased numbers of chironomids in the samples could have come from newly laid eggs. Hilsenhoff (1966) found that *Chironomus plumosus* eggs hatch in only 3 days when held at 24 °C. This temperature was reached or exceeded in Mud Lake for most of July and August 1988 (Figure 7), thus providing conditions conducive for rapid growth and maturation.

PHYTOPLANKTON AND ZOOPLANKTON

Artificial destratification and circulation has been proposed as an approach to lake restoration for problems caused by both nuisance algal types (primarily blue-green algae), and excess algal abundance. A number of theories have been proposed to explain how circulation might control algae. Shapiro (1981) has presented two models, which are shown in Figure 17. The top panel diagrammatically represents some of the results of whole lake circulation and their role in causing a shift from blue-green (identified as BG in the center box of that panel), to green algae (identified as G), which typically are a more desirous species assemblage. It is thought that this algal shift is mainly a result of competition for inorganic carbon. Based on uptake kinetics, King (1970) has hypothesized that when more free CO_2 is available, green algae will dominate, but when there is less free CO_2 available, blue-greens dominate. The amount of free CO_2 available at any time is a function of the pH, temperature, and alkalinity of the system. If the pH of a



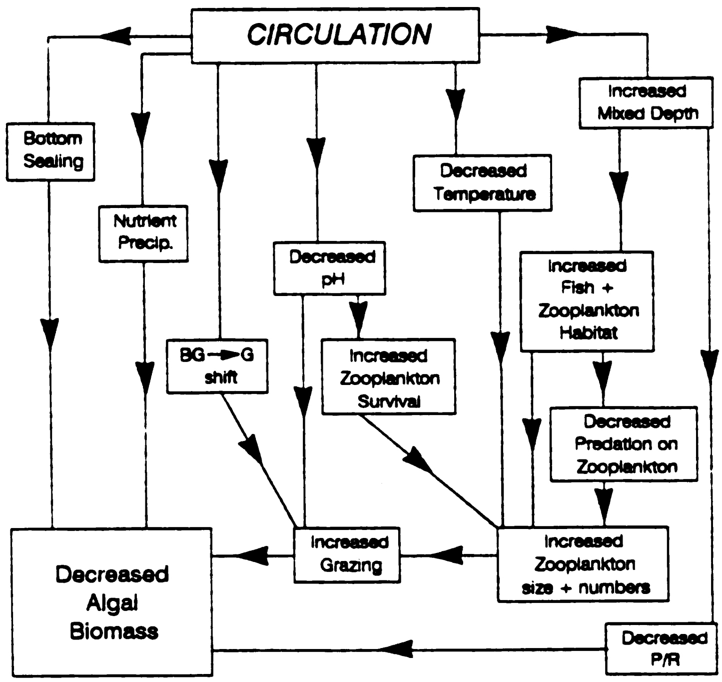


Figure 17. Proposed mechanisms controlling algal types (upper portion) and algal biomass (lower portion) in artificially mixed lakes (after Shapiro 1981).

system is lowered (which frequently occurs during circulation), the amount of available free CO_2 increases. Circulation can lower the pH of a lake in a number of ways. This includes increased respiration of organics in the water and sediments, increased CO_2 entry from the atmosphere due to increased turbulence, CO_2 movement from the hypolimnion, and a change in the photosynthesis/respiration (P/R) ratios due to changed light regimes (Shapiro 1981). Other factors which could play a role in causing an algal shift are presented in the top panel of Figure 17.

The phytoplankton species assemblage of Mud Lake during 1988 was not what would have been predicted from the models of King (1970) or Shapiro (1981). Calculations of free CO_2, based on the work of Harvey (1955) and Park (1969) using alkalinity, temperature and Ph presented in Table 1A, indicated that at no time were concentrations low enough for blue-green algae to dominate. Values ranged from 25.8-463.8 μmoles CO_2-C/L. However, Mud Lake continued to have problems with nuisance, floating blooms of blue-green algae in 1988, as was the case in 1987.

Shapiro (1981) also presents a model for how total algal biomass might decrease with whole lake circulation. This is depicted in the lower panel of Figure 17. Control of algal biomass is primarily the result of increased grazing pressure by zooplankton. Shapiro contends that zooplankton grazing pressure increases because there is an increase in the size and number of these organisms. This is a result of a greater mixed depth which allows for zooplankton and fish to inhabitat a larger volume of water and thus lessen size-selective predation, i.e. the tendency for zooplankton-eating fish to select the larger forms (Brooks and Dodson 1965). It is important to note that it assumes a species shift to green algae.

It appears that the zooplankton of Mud and Carpenter Lakes had little impact on algal biomass during 1988. It was observed, on a qualitative basis, that algal biomass was relatively constant throughout the year. However, zooplankton abundance was quite variable. Initially, there were 900-2300 individuals/L, but by July, these numbers had declined to less than 400/L (Figure 15). During the time of peak abundance, filter-feeding rotifers dominated the zooplankton (Figure 16). These organisms typically consume bacteria or detrital material as opposed to algae (Wetzel 1983). By July, when zooplankton populations declined, unpalatable, colonial blue-green algae dominated the plankton of the lakes.

CONCLUSION

Mechanical problems which plagued the study in 1987 were overcome in 1988. The Lake Preserver functioned continuously during the 1988 season and proved to be an effective means for destratifying the lake. However, the rate of air delivery that was necessary to maintain uniform dissolved oxygen concentrations caused the Lake Preserver to operate as an air diffusion device, rather than in an intermittent fashion as it had been designed. As found in this study, artificial destratification raised the temperature of the entire water column significantly during the summer months, which in turn increased the oxygen demand. Even short durations of inoperation led to rapid loss of dissolved oxygen as was the case in 1987. Therefore, constant operation of the Lake Preserver was imperative. This proved to be an expensive undertaking. Electrical costs necessary to operate the compressors to maintain isothermal and uniform oxygen conditions in Mud Lake during 1988 averaged over $500/month.

LITERATURE CITED

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American Public Health Association (APHA). 1985. Standard Methods for the Examination of Water and Wastewater, Sixteenth Edition. American Public Health Association, Washington, D.C. 1268pp.

Batterson, T.R., S.L. Chubb, R.S. Beebe, and R.M. Day. 1987. The effects of artificial destratification on the biota and selected water quality parameters of Mud Lake, Oakland County, Michigan. Research report to Kobe Steel, Ltd. New York. 59 pp.

Brezonik, P., J. Delfino, and G.F. Lee. 1969. Chemistry of N and Mn in Cox Hollow Lake, Wisconsin, following destratification. J. Sanit. Eng. Div., Am. Soc. Civ. Eng. 95:929-940.

Brooks, J.L. and S.I. Dodson. 1965. Predation, body size, and composition of plankton. Science 150:28-35.

Brynildson, O.M. and S.L. Serns. 1977. Effects of destratification and aeration of a lake on the distribution of planktonic crustacea, yellow perch and trout. Wisc. Dept. Nat. Resour., Tech. Bull. No. 99. 22pp.

Cooke, G.D., E.B. Welch, S.A. Peterson, and P.R. Newroth. 1986. Lake and Reservoir Restoration. Butterworth Publishers, Boston. 393pp.

Cowell, B.C., C.J. Dawes, W.E. Gardiner, and S.M. Scheda. 1987. The influence of whole lake aeration on the limnology of a hypereutrophic lake in central Florida. Hydrobiologia 148:3-24.

Davies, B.R. 1976. The dispersal of Chironomidae larvae: A review. J. Ent. Soc. S. Afr. 1:39-62.

Edmondson, W.T. (Ed.). 1959. Freshwater Biology, Second Edition. John Wiley and Sons, Inc., New York. 1248pp.

Eggleton, F.E. 1931. A limnological study of the profundal bottom fauna of certian freshwater lakes. Ecol. Monogr. 1:231-332.

Eggleton, F.E. 1934. A comparative study of the benthic fauna of four northern Michigan lakes. Mich. Acad. Sci. Arts Letters 20:609-644.

Elliott, J.M. 1977. Some methods for the statistical analysis of benthic invertebrates. Freshwater Biological Association, Sci. Pub. No. 25, Cumbria, England. 160 pp.

Fast, A. 1981. Hypolimnetic Aeration. In: Destratification of Lakes and Reservoirs to Improve Water Quality, Proceedings of Joint United States/Australia Seminar and Workshop, Melbourne, Australia, February 19-24, 1979. Australian Goverment Publishing Service, Canberra, Australia. pp.201-226.

Harvey, H.W. 1955. The Chemistry and Fertility of Sea Waters. Cambridge at the University Press, New York. 234pp.

Hayne, D.W. and R.C. Ball. 1955. Benthic productivity as influenced by fish production. Limno. Ocean. 1:162-175.

Hilsenhoff, W.L. 1966. The biology of *Chironomus plumosus* (Diptera: Chironomidae) in Lake Winnebago, Wisconsin. Ann. Ent. Soc. Am. 59:465-473.

King, D.L. 1970. The role of carbon in eutrophication. J. Water Pollut. Control Fed. 42:2035-2051.

Lind, O.T. 1974. Handbook of Common Methods in Limnology. C.V. Mosby Co., St. Louis, Missouri. 154pp.

Liston, C.R., C.D. McNabb, W.W. Taylor, and T.R. Batterson. 1980. Environmental baseline of Carpenter lake and Mud Lake, Orion Township, Oakland County, Michigan. Research Report to General Motors Assembly Division, Plant Replacement Task Force, Troy, Michigan. 70pp.

Liston, C.R., T.R. Batterson, and M.V. Thomas. 1985. Environmental studies during 1985 of Carpenter and Mud Lakes, Oakland County, Michigan with comparisons to 1980 data. Research Report to General Motors Buick, Oldsmobile and Cadillac Group, Orion Assembly Plant, Orion, Michigan. 64pp.

Maleug, K., J. Tilstra, D. Schults, and C.F. Powers. 1973. Effect of induced aeration upon stratification and eutrophication processes in an Oregon farm pond. Geophys. Monogr. Ser. 17:578-587.

Mortimer, C.H. 1941. The exchange of dissolved substances between water and mud in lakes. (Parts I and II). J. Ecol. 29:280-329.

Mortimer, C.H. 1942. The exchange of dissolved substances between water and mud in lakes. (Parts III, IV, summary, and references) J. Ecol. 30:147-201.

Park, P.K. 1969. Oceanic CO_2 system: an evaluation of ten methods of investigation. Limnol. Oceanogr. 14:179-186.

Pastorok, R.A. and T.M. Grieb. 1984. Prediction of lake response to induced circulation. In: Lake and Reservoir Management, Proceedings of the Third Annual Conference. North American Lake Management Society. EPA 440/5-84-001. U.S. EPA Washington, D.C. pp.531-541.

Pastorok, R.A., T.C. Ginn, M.W. Lorenzen, and T.C. Ginn. 1980. Review of aeration/circulation for lake management. In: Restoration of Lakes and Inland Waters. EPA 440/5-81-010. U.S. EPA, Washington, D.C. pp.124-133.

Pennak, R.W. 1978. Fresh-water Invertebrates of the United States, Second Edition. John Wiley and Sons, Inc., New York. 803pp.

Prescott, G.W. 1962. Algae of the Western Great Lakes Area, Second Edition. Wm. C. Brown Company Publishers, Dubuque, Iowa. 977pp.

Prescott, G.W. 1978. How to Know the Freshwater Algae, Third Edition. W.C. Brown Co. Publishers, Dubuque, Iowa. 293pp.

Shapiro, J. 1981. Effects of artificial circulation in reservoirs on algae. In: Destratification of Lakes and Reservoirs to Improve Water Quality, Proceedings of Joint United States/Australia Seminar and Workshop, Melbourne, Australia, February 19-24, 1979. Australian Goverment Publishing Service, Canberra, Australia. pp.376-399.

Toetz, D.W., J. Wilhm, and R. Summerfelt. 1972. Biological effects of artificial destratification and aeration of lakes and reservoirs - analysis and bibliography. Bureau of Reclamation Rept. No. REC-ERC-72-33. U.S. Dept. Int., Denver, Colorado. 117pp.

Weiss, C.M. and B.W. Breedlove. 1973. Water Quality Changes in an Impoundment as a Consequence of Artificial Destratification. N. Carolina Water Resour. Res. Inst., Report No. 80.

Wetzel, R.G. 1983. Limnology, Second Edition. CBS College Publishing, New York. 762pp + appendices.

Wilhm, J. and N. McClintock. 1978. Dissolved oxygen concentration and diversity of benthic macroinvertebrates in an artificially destratified lake. Hydrobiologia 57:163-166.

Wirth, T.L., R.C. Dunst, P.D. Uttormark, and W. Hilsenhoff. 1967. Manipulation of Reservoir Waters for Improved Quality and Fish Population Response. Wisc. Dept. Nat. Resour., Rep. No. 22, Madison, Wisconsin.