LOW ENERGY MECHANICAL DESTRATIFICATION
AND REAERATION OF RESERVOIRS

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ABSTRACT

A cluster of 16 axial flow units was used to destratify a lake having a surface area of 951 hectares. Complete destratification was not achieved. However, a significant reduction of the anoxic hypolimnion was accomplished. The penetration of a low velocity jet was limited by density differences and was accurately predicted.

Attempts were made to correlate seasonal changes in water near the bottom, the sediments, and physiological changes in benthic invertebrates with the dissolved oxygen content and temperature of the lake water. Examination of the vertical variation of physicochemical conditions indicated that mechanical pumping did not improve water quality of the water 1 m above the bottom at either the 15 m stations in the arms or in the central pool. Particle size decreased with water depth. Organic matter did not vary significantly among stations and seasonal variation was slight. Inorganic phosphorus in the sediment was generally lower in summer than in spring and winter. Phosphorus in the sediments was generally greater at the deeper stations. Minimum sedimentation rates occurred in the central pool. Iron did not show variation with depth. Seasonal trends were observed in sorbed manganese of the sediment. Sediment caloric content decreased significantly throughout the summer. Higher hemolymph ion (Na^+, K^+, Cl^-) concentrations for _Chironomus tentans_ and _Chaoborus punctipennis_ were observed at the deep water stations. Oxygen consumption rates of _Chaoborus_ were highest from those collected at the deep water station. Oxygen consumption increased in the sediment column as the water temperature increased, and the oxygen concentration decreased to 0.0 ppm.

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Objectives and Extent of Achievement of the Objectives

The objectives of the study were:

(1) To determine the relationship between pumping rate and destratification time of a reservoir,
(2) To determine the relationship between temperature stability and oxygen concentration in a reservoir,
(3) To test the hypothesis that lakes can be destratified to a preselected depth using the Garton pump,
(4) To determine certain biological and physicochemical effects of lake destratification.

Total destratification was not achieved until late summer in 1977 and 1978, and was largely due to normal cooling. Because the principal cause of destratification was not due to the pumping rate used, no relation could be obtained to satisfy the first objective.

A graphical presentation of the temperature stability index versus the oxygen distribution index reveals a relationship satisfying the second objective.

Achievement of the third was possible even though complete destratification was not achieved. The depth to which a reservoir may be destratified was shown to be a major limiting factor when using the Garton pump at low flow velocities.

Data collected by the Agricultural Engineering Department and the School of Biological Sciences were used to satisfy the fourth objective.

Part I of this report discusses the first three objectives and the physiochemical effects of the fourth objective. Part II presents additional physiochemical as well as the biological findings in the water near the bottom, the sediments, and the macroinvertebrates.

PART I

PART I

Introduction

Although lakes normally provide good quality water, annual water quality deterioration occurs within the lake due to thermal stratification and eutrophic conditions.

By late spring, thermal stratification is usually well defined by three distinct layers within the lake. The upper stratum, termed epilimnion, is characterized by warm, well-mixed, well-oxygenated, good-quality water. The lower stratum, hypolimnion, characteristically is cold and unmixed. The interface zone between these layers is termed metalimnion. It is characterized by a rapidly decreasing temperature gradient. Because of the density differences of these three layers, they do not intermix. Only the upper layer receives the benefit of atmospheric aeration and significant mixing due to wind action.

Eutrophication from the Greek word eutrophos is used to denote a high degree of nutrients available for algal growth. A eutrophic lake is characterized by visible algal growth, commonly resulting in a green color. Although algae produce oxygen in the daylight, their death and subsequent decay results in a high biological demand for oxygen. In a stratified lake, this high demand for oxygen usually depletes the supply in the hypolimnion. It is this anoxic condition which indirectly results in water quality deterioration.

With depletion of oxygen in the hypolimnion, high concentrations of hydrogen sulfide, ammonia, iron, manganese, phosphates, nitrates, and other toxic substances can occur. Fish are confined to the epilimnion and possibly greater than 50 percent of the lake's volume is unfit as a municipal water supply, for fish growth, or for stream releases. Additionally, a decreased aerobic area on the lake bottom is available for invertebrates and benthic organisms.

As fall cooling occurs, the epilimnion cools and sinks. This is often referred to as a lake "turnover." Upon turnover, all the undesirable substances, formerly present in the hypolimnion, are distributed throughout the lake. The overall oxygen concentration is lowered to the point that massive fish kills may occur. Taste and odor problems in municipal water supplies are very common.

Artificial destratification may be mechanically accomplished to eliminate thermal stratification so that the entire body of water is circulated and becomes oxygenated. An American Water Works Association committee (1976) surveyed various water-utility managers to determine the water quality problems and quality control methods. The committee recommended destratification as a relatively inexpensive technique to help water suppliers who experience water quality deterioration due to an anoxic hypolimnion.

An inexpensive, effective low energy device has been developed by Quintero and Garton (1973) to pump the oxygen rich surface waters downward into the hypolimnion.

Strecker (1976) used a single axial flow pump to destratify a 40 ha (100 ac) lake. The pump used 0.645 kW to pump 1.58 m^3/sec. The study showed that overall water quality in the lake improved, and that the pump proved to be a simple, economical and effective destratifying device. In an earlier study on the same lake, the effect of destratification on water quality parameters was presented (Steichen et al. 1974).

Garton and Rice (1976) used a 5.03 m diameter axial flow pump to destratify Lake Arbuckle in 1975. The pump operated at a capacity of 13.06 m^3/sec.

Although the lake was fairly well destratified thermally, it remained chemically stratified.

Experimental Design and Equipment

Use of an 8.53 cooling tower fan was studied. Because of the questionable ability of the fan to stand the operating stresses, the manufacturer would not guarantee the device based upon their own engineering study. Additionally, a single large pump would require complex construction, and have handling and anchoring problems beyond our capabilities.

The most attractive alternative was to use a cluster of 16 axial flow pumps, each 1.829 m diameter, arranged in an open-centered square. The units were easy to fabricate, transport, and put into operation. Each unit was powered by a 460V, 1.12 kW motor. The cluster had a estimated pumping capability of 26.4 m^3/sec (416,000 gpm) with an energy input of 9.5 kW (12.8 horsepower). Figure 1 is a picture of a pump impeller. A 100:1 gear reduction was used to rotate the impellers at 17.35 rpm. An assembled pump unit, Figure 2, cost about $1,600.00 for the major items and materials.

Location of Equipment

Destratification studies were conducted at Lake Arbuckle, located near Sulphur, Oklahoma, latitude 34° 26' 30" N, longitude 97° 01' 30" W. The lake was constructed by The Bureau of Reclamation and was filled by April, 1968, to provide municipal water supplies as well as a recreational area. At conservation pool level, the lake has a surface area of 951.0 ha (2350 ac), a volume of 89300 megalitres (72,475 ac-ft.) a mean depth of 9.4 m (31 ft.), and a maximum depth of 27.4 m (90 ft.). A map of the lake's profile is given in Figure 3.

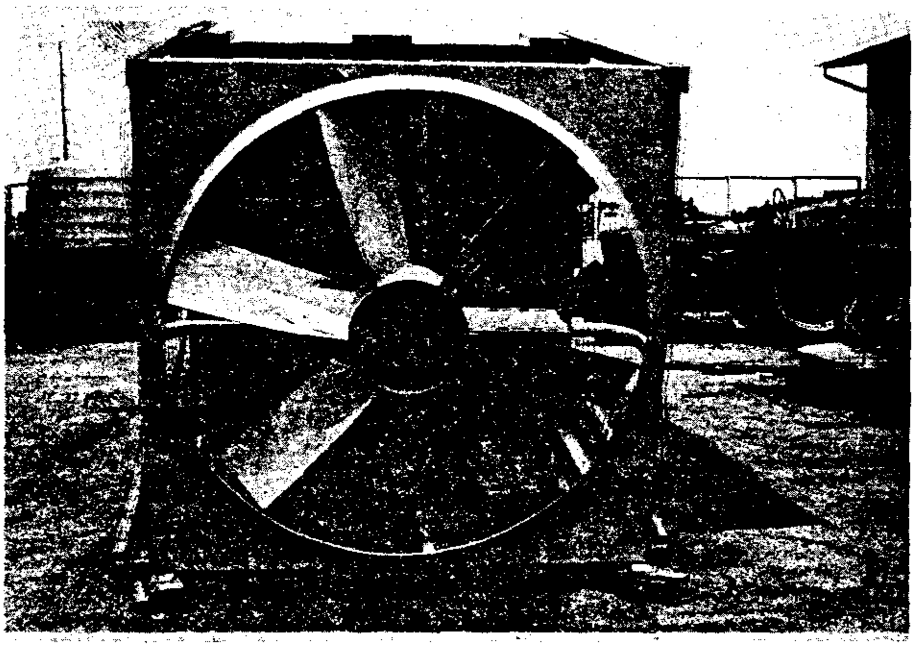

Figure 1. Bottom View of Pump Impeller

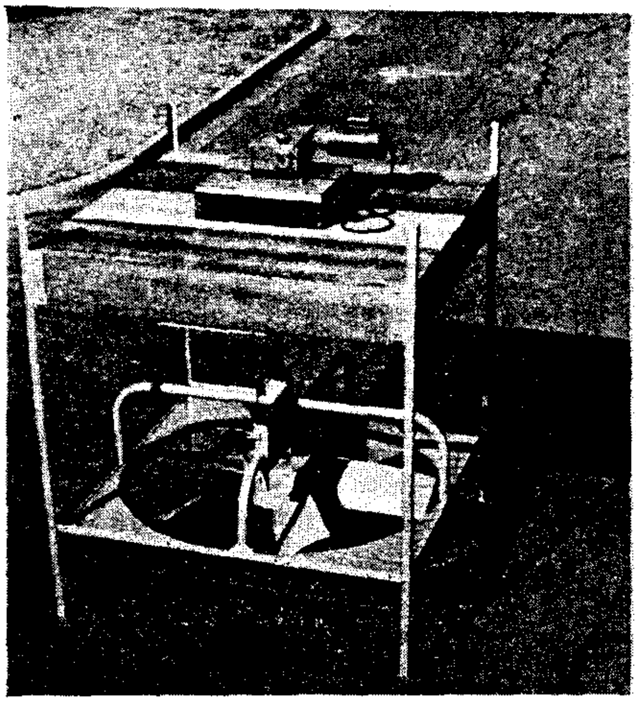

Figure 2. Single Pump Unit

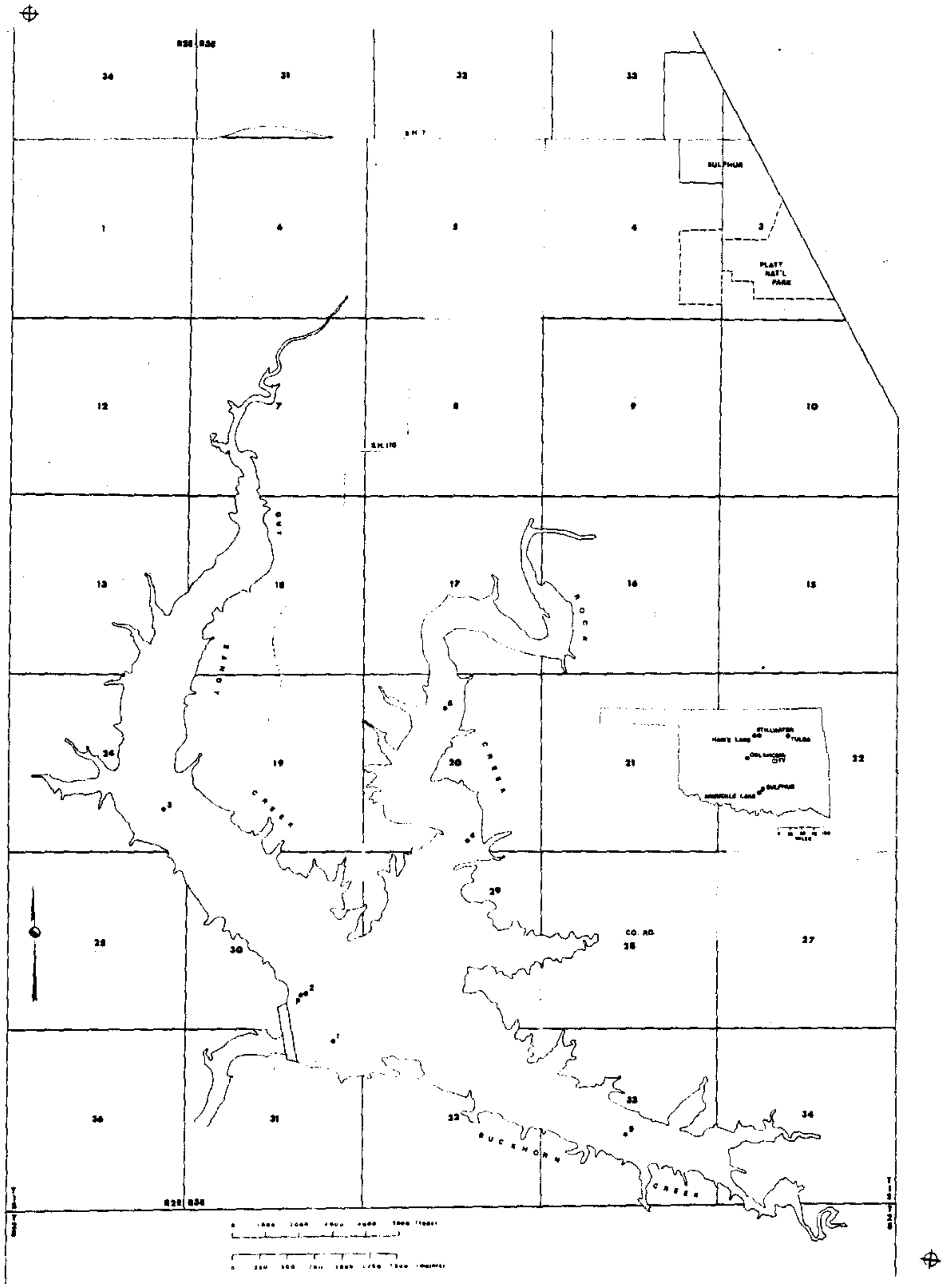

Figure 3. Map of Lake Arbuckle

Destratification studies were conducted during the summers of 1977 and 1978. The cluster of pumps was located near the dam (Figure 4) and six observation stations were established (Figure 3). To determine the destratifying influence of the pumping cluster, the temperature and dissolved oxygen (DO) profiles were monitored at each station. The readings from each station were averaged before analysis and graphic aids were prepared.

Analysis of Data

Temperature and DO Data from Previous Years

Data collected by Garton and Rice (1976) were available for the summers of 1975 and 1976. A destratifier was operated in 1975 but not operated in 1976. The temperature and DO profile for 1975 (Figure 5) show the results of pumping 13.06 m^3/sec for the period shown. A uniform warming occurred at all depths in the lake until September 14, when the lake turned over. For most of the pumping period a weak metalimnion existed between three to six m deep and the lake was anoxic below nine m.

The temperature and DO profiles for late June through September, 1976 (Figure 6), was typical for a stratified eutrophic lake. Temperature profiles were stable below nine m and anoxia was prevalent below six m.

Temperature and DO Data from 1977

Thermal stratification began in April and was fully developed by early May. By mid-June, the lake was more strongly stratified than in previous years. Pumping operations began on July 1, 1977 (Figure 7), and a uniform warming of the lower depths did not occur as in 1975. Warming occurred only down to a specific depth which gradually lowered throughout the summer. This indicated that the penetration of the pump jet was limited.

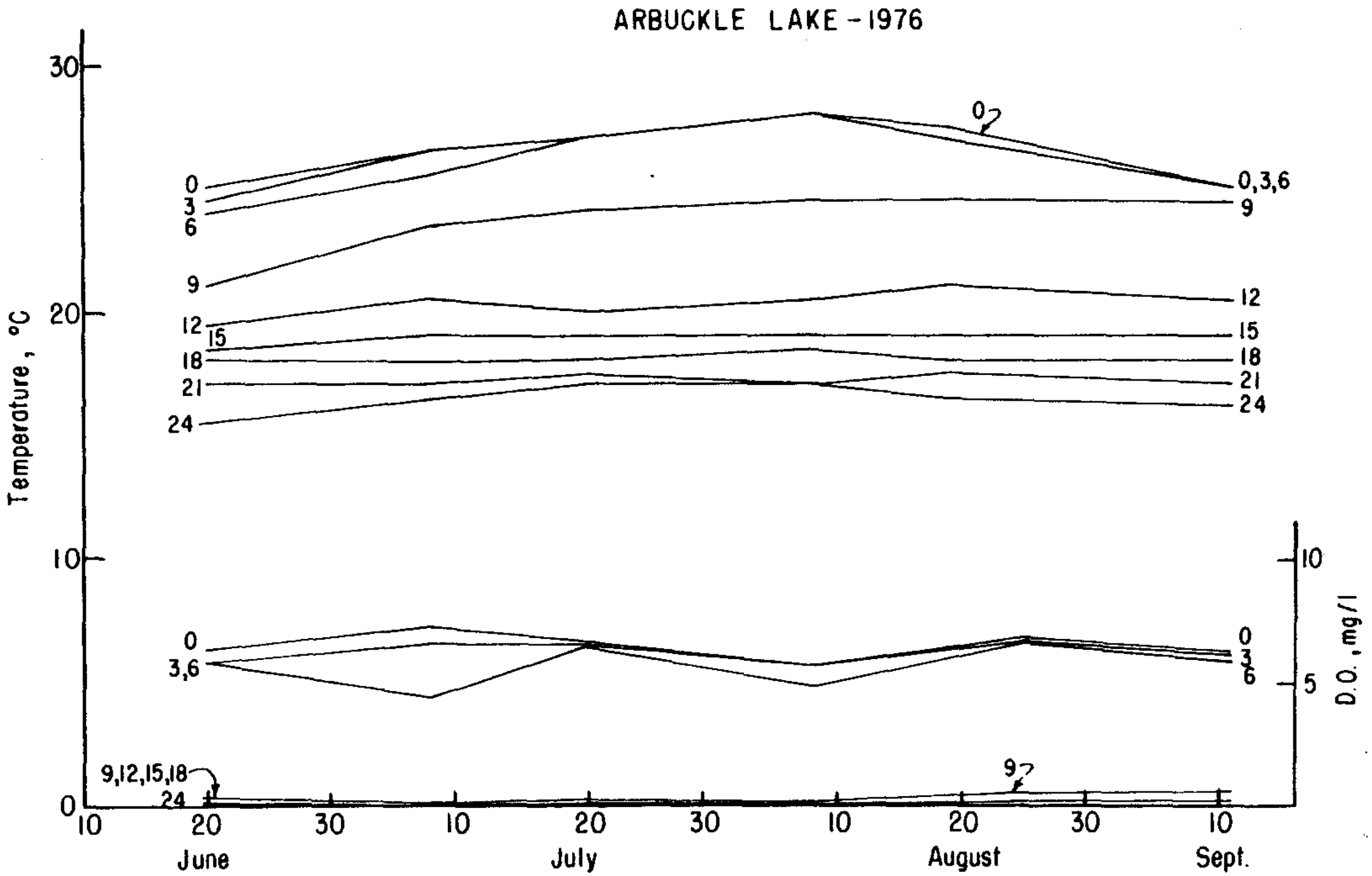

Figure 6. Temperature and DO for 3m Interval Versus Time for Non-pumping Year 1976.

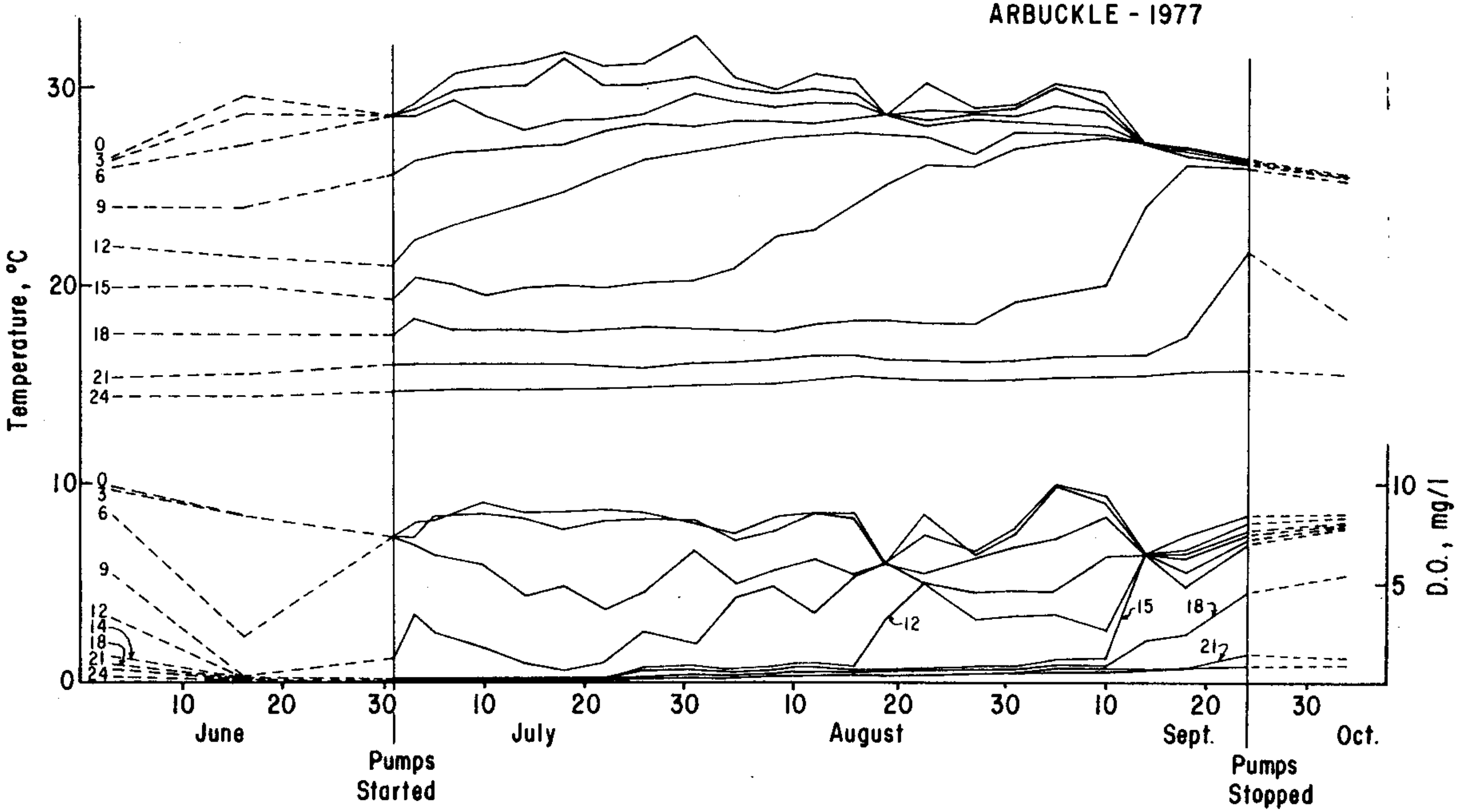

Figure 7. Temperature and DO for 3 m Intervals Versus Time for 1977

On July 22 as the 9 m depth warmed to within 3°C of the surface temperature, a significant DO increase began at 9 m. On August 16, as the 12 m depth warmed to within 3°C of the surface temperature, there was a corresponding DO increase. The same pattern can be seen for the 15 m depth on September 10. This pattern indicated that DO increases were a result of natural lake mixing when the temperature difference between the surface and some depth is less than 3°C in the afternoon. In 1977 increases were observed at the 18 m depth prior to fall turnover on September 14.

Temperature and DO Data from 1978

Fifteen pumps began operation on April 8, prior to strong thermal stratification. A storm in mid-May rendered eight pumps temporarily inoperable due to broken impeller shafts. The lake was then able to strongly stratify with little hindrance. The number of pumps in operation for the 1978 summer is shown in Figure 8. The results of the pumping was similar to that of 1977. As warming of the lower depths occurred, resulting DO increases were noted. The lake was more thermally destratified and had a deeper oxygen profile in August of 1978 than in previous years during August.

Temperatures and DO Comparisons

The average temperatures of the lake in 1975, 1976 and 1978 are shown in Figure 9. Although destratification does increase the overall heat content of a lake, average daily radiant input has a major influence on average water temperatures. Prior to pumping in 1977, the lake was warmer than the corresponding period in other years. The average DO content of the lake was higher in 1977 and most of 1978 than in 1976 and 1975.

To insure good water quality for municipal supplies and to provide suitable fish habitat, the DO should be maintained at concentrations no less than

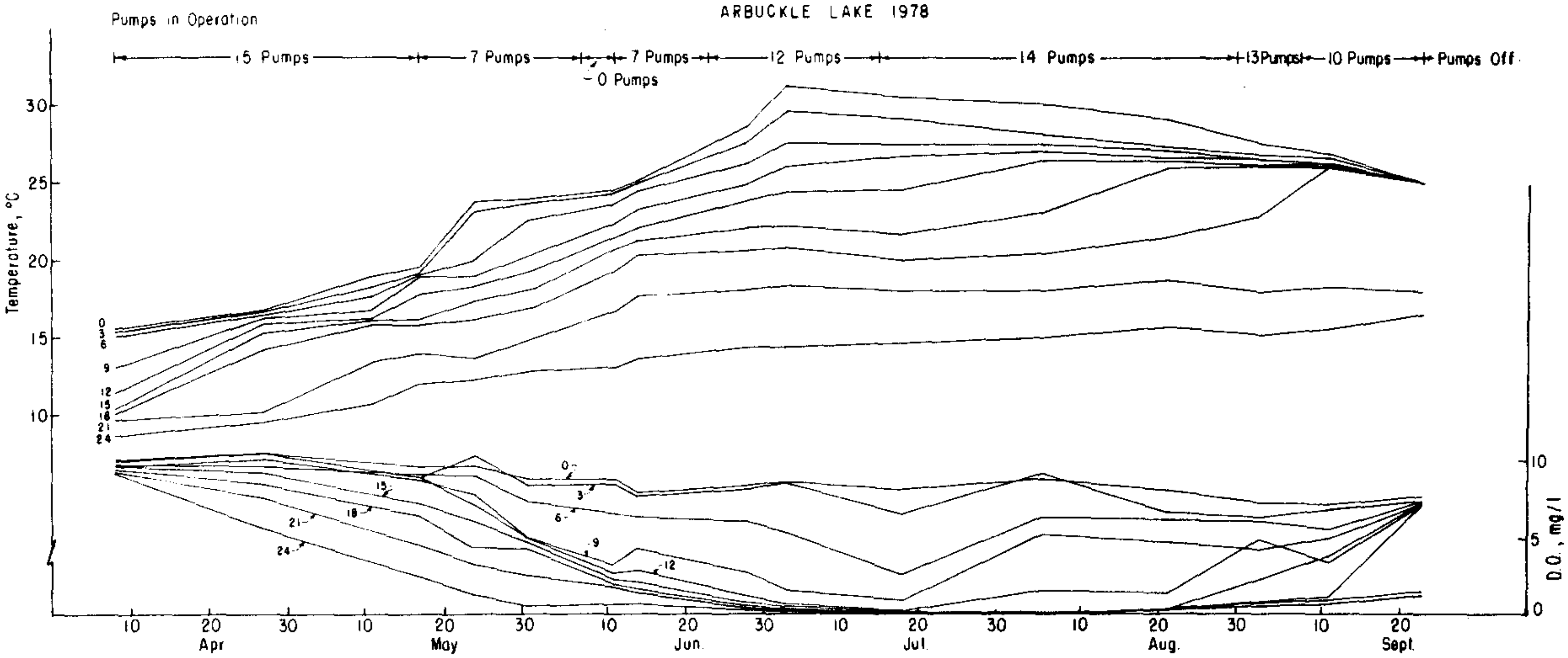

Figure 8. Temperature and DO for 3 m Intervals Versus Time for 1978.

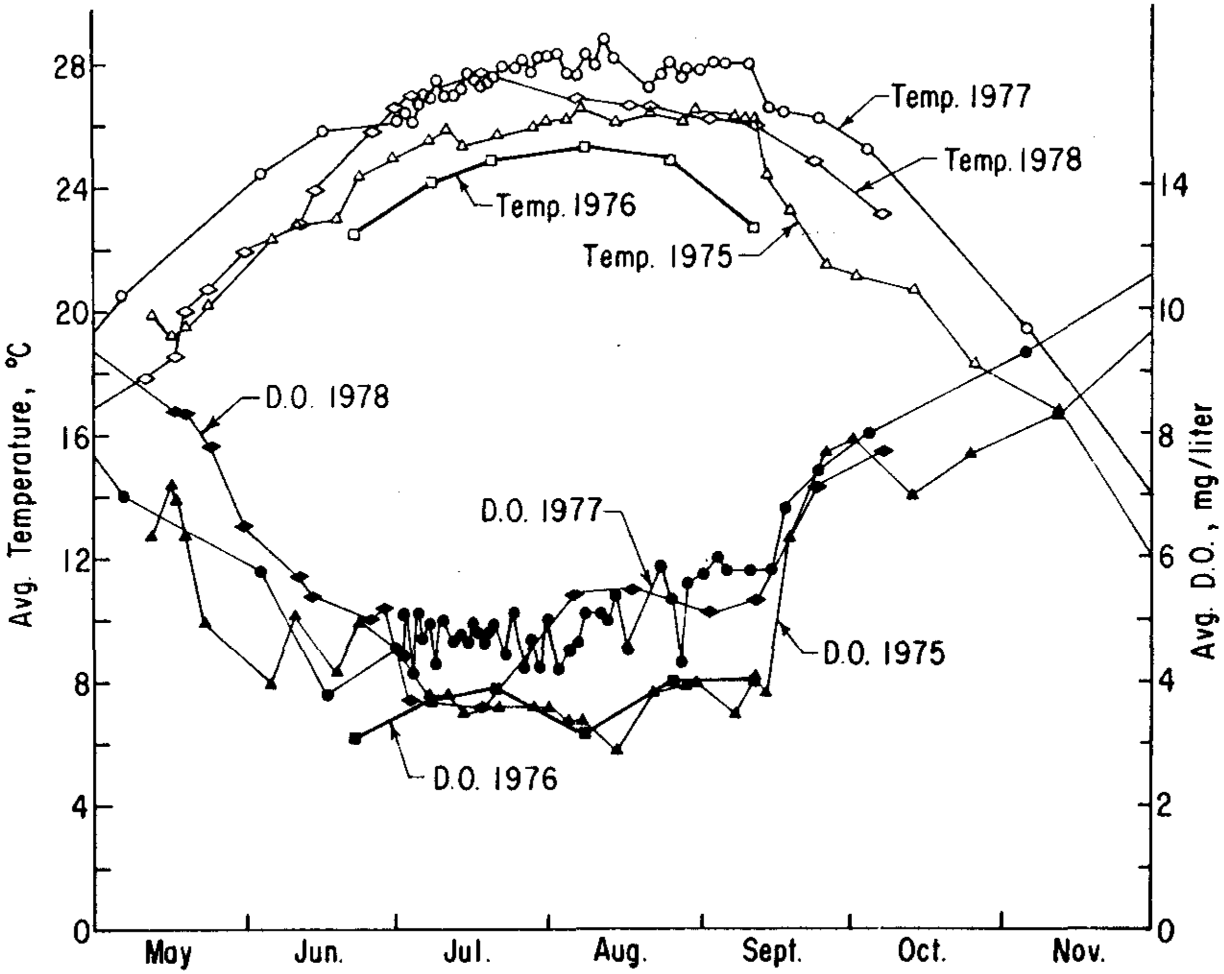

Figure 9. Average Temperature and Average DO Versus Time.

2 mg/l (Bernhardt, 1967, Summerfelt and Gebhart, 1975). The percentage of lake volume above 2mg/l DO in August was about 85 percent in 1977 and 1978 compared to less than 65 percent in 1975 and 1976 (Figure 10). Thus the epilimnion was increased about 20 percent.

Stability Index and Oxygen Distribution Index

The magnitude of thermal-density stratification may be indicated by the stability index (SI). The stability index represents the amount of work required to raise the centroid of a stratified lake to the centroid of the lake having a uniform temperature profile. When pumping began in 1975, the SI was 685 W-h, Figure 11. During 1976, the SI peaked about 110 kW-h. When pumping began in 1977, the SI was 1250 kW-h, and after reaching a high value of 1550 kW-h was lowered to a value less than that of 1976 by early August. The SI for 1978 peaked at 1200 kW-h and was considerably less than that for 1976 by late July.

Although the SI for 1975 was less than the other years, the lake remained more strongly chemically stratified. The shallow metalimnion is believed to have caused the strong chemical stratification.

The oxygen distribution index (ODI) is calculated in a similar manner to the SI, except the weight of oxygen present in the lake is used (Garton and Rice, 1976). A zero value indicates uniformity of oxygen distribution.

A plot of ODI versus SI (Figure 12) shows the initial increase of ODI to be coincident to rising values of SI. At the higher SI values, the ODI exhibits little or no dependence upon the SI. At Lake Arbuckle, the SI increased from zero in early March to maximum values by mid-July. After stratification began, a corresponding decrease in DO was evident in the hypolimnion resulting in an increased ODI. Alghough the ODI and SI increased concurrently, the rates of increases were confounded with solar flux variations,

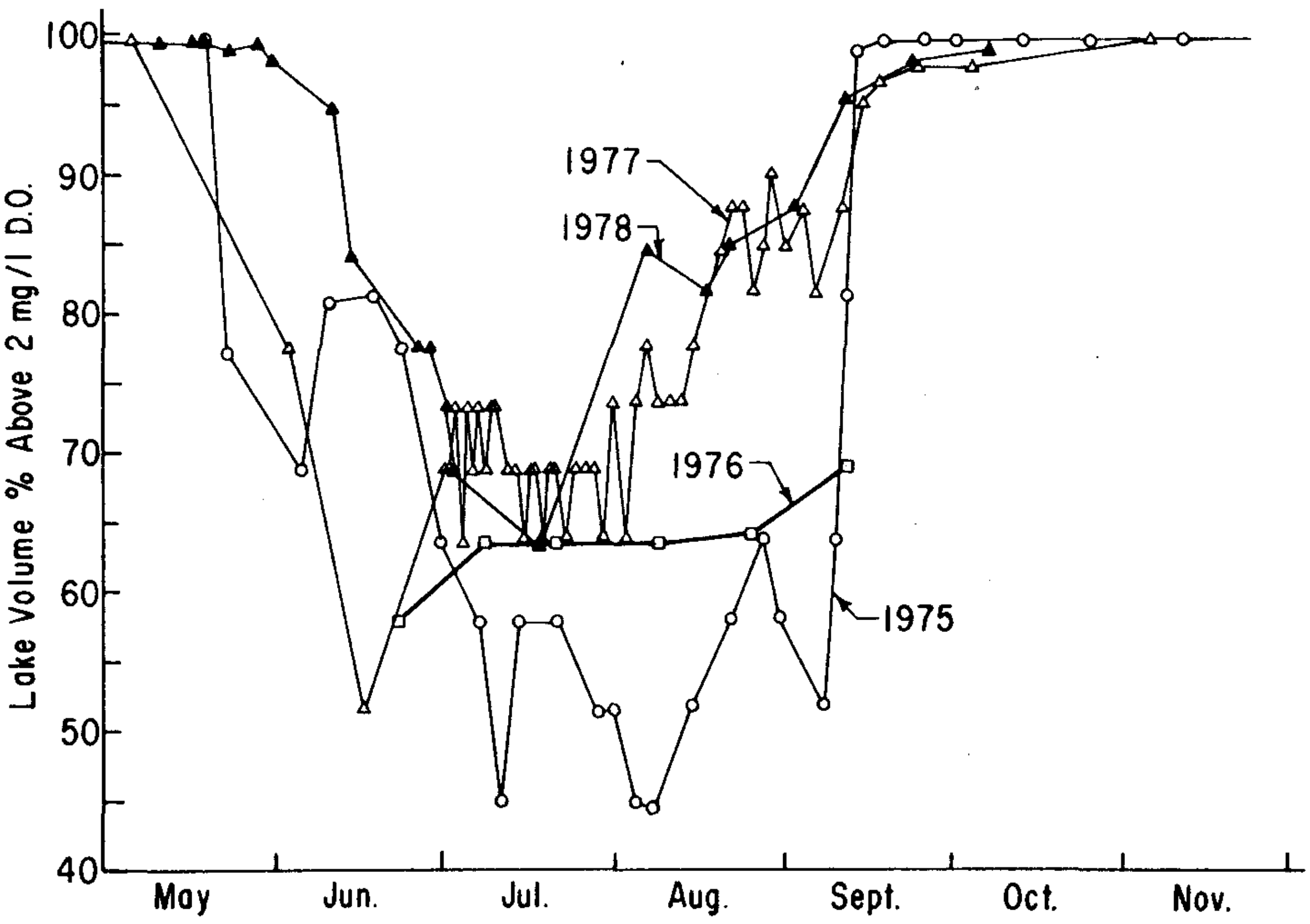

Figure 10. Lake Volume Percent Above 2.0 mg/1 DO Isopleth Versus Time.

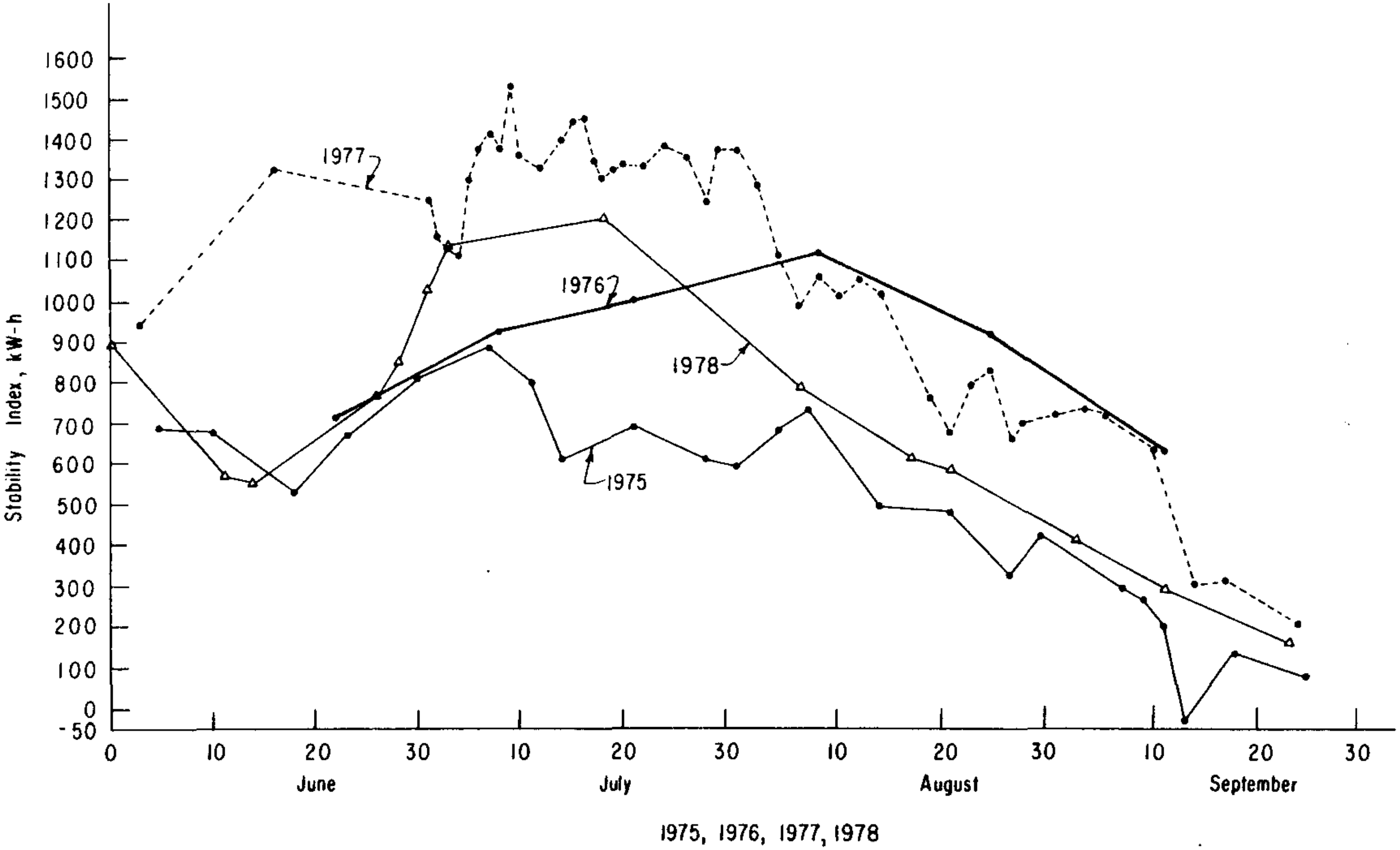

Figure 11. Stability Index Versus Time

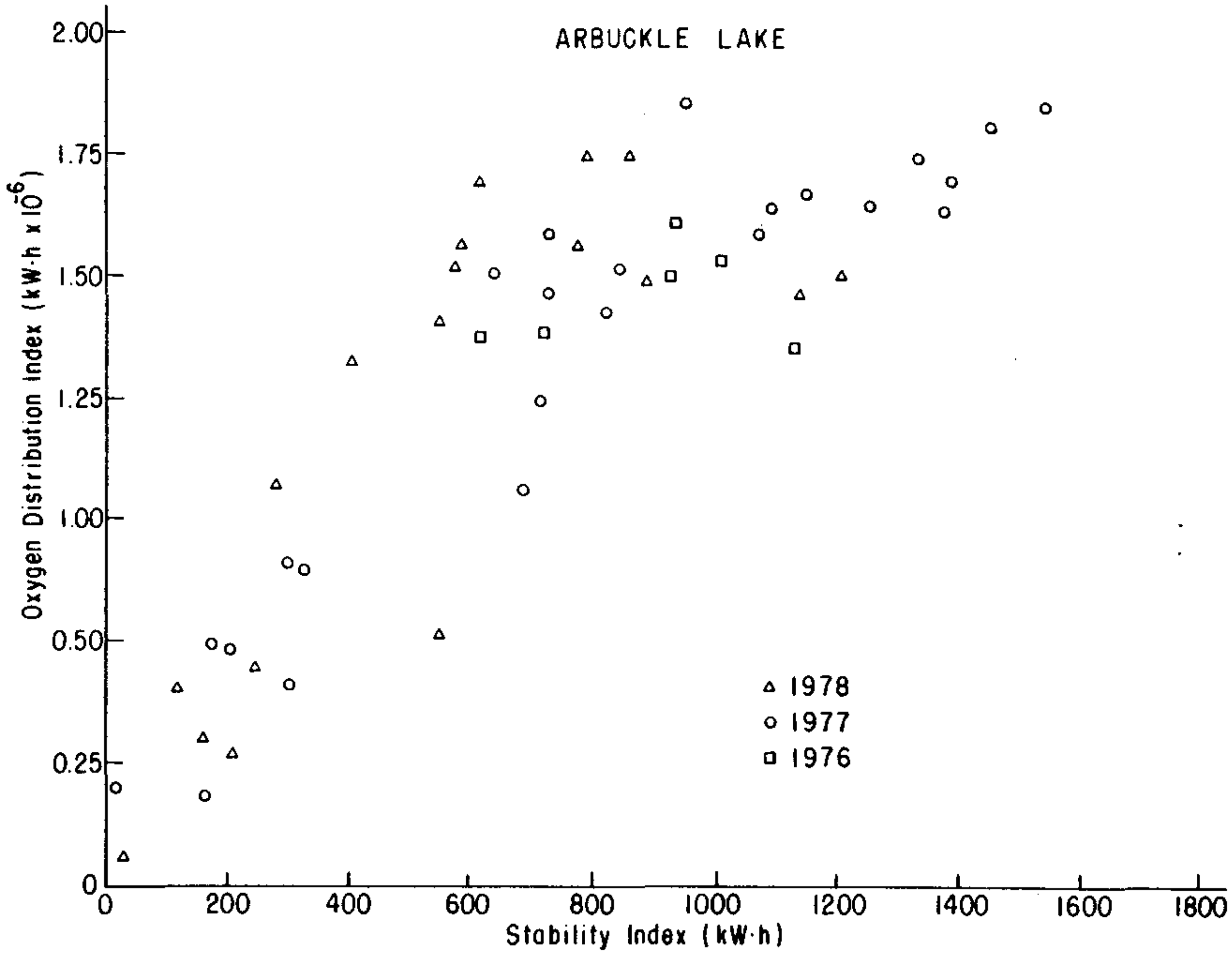

Figure 12. Oxygen Distribution Index Versus Stability Index.

wind and wave action, rates of biotic activity, decreases of DO saturation levels, and other factors. From late June to mid-July, the hypolimentic volume varied only slightly and remained near the maximum value. Surface temperature increases caused SI increase with little or no related ODI changes.

Depth of Penetration

When pumping warm surface waters downward to mix with the cold lower stratum, the depth of penetration was limited by the thermal-density layers within the lake. The limit to which a low velocity jet may be thrust into a stratified lake is of great importance in determining the mixing effectiveness for destratification.

The depth to which the surface waters were pumped on July 1, 1977, was about 13 m. This depth varied slightly until mid-July when the depth began to lower about 1 m per week.

A method was developed which would predict the depth of penetration of the pump plume based upon the thermal profile of the lake, geometry of the plume, velocity of the water through the impeller, depth of the impeller, angle of entrainment and the number of pumps used. Geometry of a plume may be determined by using a temperature probe since the temperature within a plume remains nearly constant. For a single pump unit the plume geometry was that of a 40° truncated cone. For an open-centered cluster of pumps, the plume geometry was that of a truncated pyramid having an apex angle of 20°. Since the plume displaced a volume of the lake that had a higher specific weight, a bouyant force was exerted on the plume according to Archimedes' Principle. The momentum flux of the pump provided an equal and opposite force required to maintain the plume. To predict the depth of the plume, it was necessary to calculate the bouyant force for each 1 m slice of the plume. When the sum of the bouyant forces equal the momentum flux of the pump, the number of slices required is the predicted depth of the plume from the depth of the impeller. This method of prediction has been proven accurate for shallow hypolimnetic penetrations but it is not known if the method is valid for hypolimnetic pentrations greater than 5 m.

The lower face of the plume was characterized by highly variable readings in DO, temperature, and conductivity. By lowering a water quality probe into the plume, the depth of active mixing could be measured. Figure 13 shows good agreement between both the predicted depth and measured range of the mixing of the plume for 1977.

A computer program developed to predict the depth of penetration was presented by Punnett (1978).

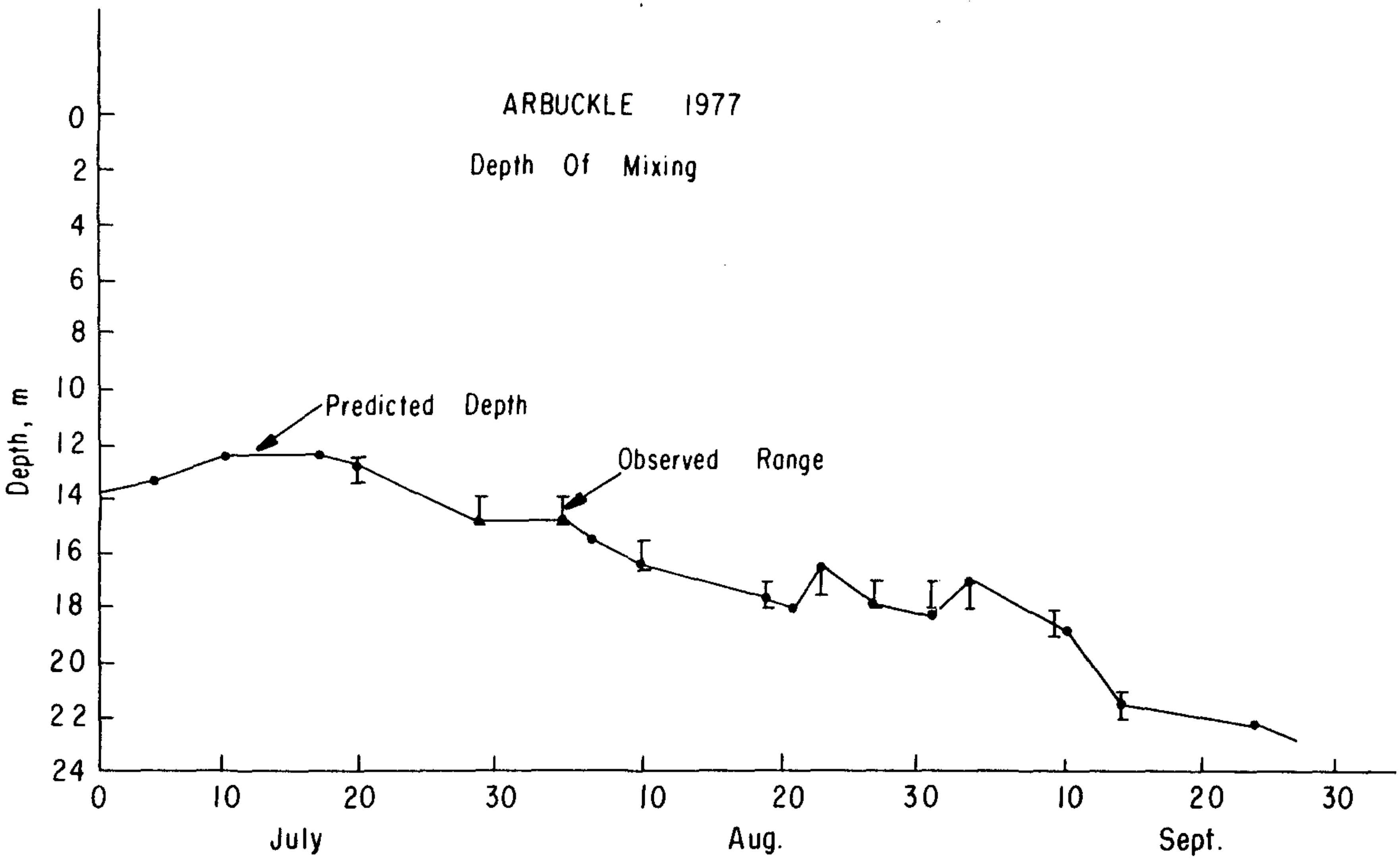

Figure 13. Depth of Mixing Versus Time for 1977.

Discussion

A mechanical destratification device having a large capacity, low velocity, and low energy requirement can be easily made by clustering axial flow pumps. The total pumping rate of 26.4 m^3/sec requiring 17.1 kW, was not sufficient to effectively destratify Lake Arbuckle which has a volume of 89300 megaliters and a surface area of 951 ha. However, a decreased hypolimnion resulted in overall water quality improvement. About a 20 percent increase in epilimnion was noted for August during 1977 and 1978. More than 85 percent of the lake volume was oxygenated prior to fall turnover as compared about 63 percent in non-pumping years.

Effectiveness of the destratification method depends upon being able to warm the bottom waters to near surface temperatures. The elimination of strong thermal-density layers within the lake will allow natural lake mixing which is necessary for oxygenation of the bottom waters. For significant oxygen transport to occur to a specific depth, the temperature at depth must be within 3°C of the surface water temperature. This trend was evident in all pumping years.

The success of the destratification with the mechanical pump will depend on whether the plume will initially penetrate to the bottom and on the ratio of the pumping rate to the volume of the normal hypolimnion. The method presented in this report for predicting the depth of plume penetration in a stratified body of water has been found to be accurate to ± 0.5 m for shallow hypolimnetic penetrations ($\leq$ 5.0 m). More research on plume characteristics should be conducted to be able predict deep hypolimnetic penetrations since this is an important design consideration.

The most difficult period of time to destratify a lake is in the early summer when the surface water is warming rapidly. During this period a high ratio of pumping rate to normal hypolimnetic volume is required. Garton and Punnett (1978) suggest pumping a volume equal to the normal hypolimnion every two days.

Fall turnover in Lake Arbuckle is normally a step-wise process which is not complete until mid-October. In 1977 and 1978 the lake experienced a turnover to 20 m on the first turnover. Thus greater than 97 percent of the lake volume was well-mixed by early September without a significant DO depression.

The deepening of the epilimnion would produce substantial increases in release water quality from reservoirs where the outlet is located at or below the depth of the metalimnion. In large stratified reservoirs where total destratification is not desirable, "point" destratification could immediately improve release water quality be placing the destratifier over the outlet. Design of a point destratifier would largely depend upon the ability to accurately predict the depth of penetration.

Conclusions

1. The pumping rate of the device tested was not sufficient to completely destratify Lake Arbuckle. Based on studies on a smaller lake, four to six times the pumping rate would be required.

2. As indicated by the relationship between the Stability Index and the Oxygen Distribution Index, the Stability Index must be greatly reduced before oxidation of the hypolimnion will occur. When the daytime temperature at a given depth became less than 3°C below the surface temperature, a significant increase in dissolved oxygen at the depth was observed.

3. The mixing depth of a downward flow of low density water was limited by the high density water of the hypolimnion and was accurately predicted for shallow hypolimnetic penetrations. The initial depth of mixing in 1977 was about 13 m. The depth of penetration increased about one m per week beginning in mid-July and continued until the autumn turnover. Any future devices should be designed with sufficient velocity to penetrate to the bottom of a stratified lake.

4. At the autumn turnover, the pump plume was penetrating to the bottom of the lake and the column was a uniform temperature.

5. Since the oxygen content of the water column was nearly uniform, very little depression in oxygen was observed at the autumn turnover. The higher than normal temperatures of the water maintained in the depths of the lake resulted in the turnover occurring 4 to 6 weeks earlier than in normal years.

6. The improved oxygen levels in the lake can be observed by comparing the percent of the volume maintained above 2 mg/1 with the percent in normal

years. During the last two weeks in August an average of about 85% of the lake volume was above 2 mg/l during the two mixing years compared to about 63% during the 1976 control summer.

7. In-lake benefits of lake destratification resulted even with incomplete mixing. Improvement of the quality of water released from lakes, by localized mixing at hypolimentic outlets, would have a beneficial effect. In order to utilize this approach, improved knowledge of the depth of penetration of buoyant jets in stratified lakes is needed.

PART II

PART II

Introduction

Bottom sediments regulate the freshwater ecosystem (Odum 1971). Metabolism of detrital and particulate organic carbon, which occurs largely in the sediments, provides stability to ecosystems (Wetzel 1975). Banin et al. (1974) stated that the upper layer of lake sediments affects the cycles and balance of nutritional elements in the lake water. Sediments can play a critical role in determining the trophic status of lakes (Golterman 1966). Hargrave (1973) found that bottom sediments yield valuable information about production and mineralization in the water column. McLachlan (1969) concluded that the nature and distribution of the bottom fauna are strongly influenced by substrate characteristics.

Parameters which have been studied most in characterizing the sediment are: 1) sedimentation (Pennington 1974, Kirchner 1975); 2) exchange mechanisms at the mud-water interface (Mortimer 1941, Hayes et al. 1958, Kamp-Nielsen 1973); 3) nutrient chemistry (Frink 1969, Serruya 1971, Wildung et al. 1974); 4) organic matter (Bordovisky 1965, Kemp 1971, Hargrave 1973); 5) DO (Hayes and McAulay 1959, Edwards and Rolley 1965, Pamatmat 1971, Edberg and Hofsten 1973); and 6) particle size (Mortland 1954, Hargrave 1972, Johnson 1974, Banin et al. 1974). These processes are influenced by lake morphology and seasonal variations which change the overall thermal structure and dynamics of a lake.

Iron and manganese have great effects on other parts of aquatic ecosystems. A large seasonal variation exists in the dissolved oxygen (DO) concentration of the water in many lakes. Mortimer (1941) showed that oxygen depletion is accompanied by a release of iron, manganese, and other substances to the water. When oxygen is returned to the bottom waters during spring overturn, iron and manganese are reprecipitated onto the bottom sediments. This precipitation and dissolution of iron and manganese hydroxides helps to regulate the levels of phosphorus and many other trace elements (Mortimer 1941, 1942, Hutchinson 1957, Jenne 1969). The seasonal cycle of copper and zinc in natural systems is less well known.

Metals are not evenly distributed in the sediments of lakes. A high positive correlation exists between iron and manganese concentrations in the sediments and water depth (Delfino, Bortleson, and Lee 1969). This is probably the result of the metals being carried to the center of the lake bound to small and finely divided particles. Sediment metals may also be in a physical or chemical form which renders them unavailable (Cross et al. 1970).

Some metals may be so tightly bound to the sediment or to dissolved organic compounds that environmental changes may not release them. Thus, it is important to consider the portion of these metals that is available for cycling in the system. Although various researchers have estimated this portion (Nriagu 1967, Reimer and Toth 1970, John, Chauah, and Van Laerharon 1972), a scarcity of literature exists concerning the forms and changes of the metals in the sediments (i.e. water-soluble forms, exchangeable forms) and in the water (i.e. dissolved or particulate forms) of lakes.

Considerable research has been conducted on the heavy metal concentrations in organisms resulting from pollution. The concentration factor of copper, chromium, lead, and zinc was determined to be higher in chironomids than sediments (Namminga and Wilhm 1977). Heavy metal concentrations in the sediment were influenced by percent loss on ignition and particle size. Mathis and Cummings (1973), however, reported highest metal concentrations in the sediment, lowest in the water, and intermediate in the biota. No studies have looked at the seasonal changes in the concentration factors in the biota.

Chironomus sp. and Chaoborus sp. larvae are benthic macroinvertebrates often used as indicators of water quality. They are abundant in the bottom areas of lakes and vital links in the food chain. Consequently, they are of interest to investigators in productivity studies as well as water quality.

Chironomus sp. and Chaoborus sp. are found in greatest numbers in the hypolimnion of eutrophic lakes. The hypolimnion has lower temperatures, less DO, and more reduced materials such as hydrogen sulfide and ferrous iron than the epilimnion. During summer and early fall, a typical thermocline is present. During this period of stratification, some benthic invertebrates migrate from the hypolimnion to the epilimnion (Hilsenhoff 1968). Organisms that remain in the hypolimnion survive in the soft bottom mud or in the water above the sediments.

Reish and Richards (1966) states that perhaps the single most important environmental factor for the growth, reproduction, and survival of aquatic animals is the DO in the water. Organisms have tolerance ranges and exhibit adaptations to seasonal changes in temperature and DO. Little information exists on the seasonal changes in oxygen uptake of the organism in lakes.

Caloric content varies between 5400-6100 cal/g for many species (Slobodkin and Richman 1961). It varies with season (Comita and Schindler 1963, Wissing and Hasler 1971), environmental conditions (Spoehr and Milner 1949), and the state in the life history (Wiegart 1956). Golley (1961) stated that caloric constants cannot be used in intensive surveys because of the response of caloric content to fluctuating environmental factors. No studies have been undertaken to determine the effect of anoxia in the hypolimnion on the caloric content of the benthos, especially chironomids and chaoborids.

The maintenance of a normal metabolic rate at low oxygen pressures and the capacity to withstand fluctuations in temperatures are features in the physiology of Chironomus sp. and Chaoborus sp. which enable them to live under such conditions. These physiological characteristics are adaptive and it is reasonable to suppose that they have evolved both respiratory and osmoregulatory specializations to be able to colonize the anoxic, colder hypolimnetic waters.

We assume that maintenance of a hyperosmotic hemolymph state involves metabolic work. The association of a higher osmotic concentration with higher temperatures and lower dissolved oxygen in the medium, which demands the most osmotic work, could be related to an increase in the metabolic rate, as observed by oxygen consumption rates, and to an increase in cell permeability. Thus, our primary concern in relation to destratified or stratified lake conditions is to see if the energy required by osmoregulatory activities is being supplied at the cost of other physiological processes (oxygen consumption) at various temperatures, salinity, and DO concentrations.

The objectives of this part of the study are to observe spatial and temporal changes in the following variables in Arbuckle Lake:

1) Temperature; dissolved oxygen; conductivity; pH; alkalinity; concentration of iron, manganese, copper, and zinc; and sodium, potassium, and chloride of the hypolimnetic water.
2) Particle size; organic carbon; phosphorus; clay mineral composition; pH; concentration of iron, manganese, copper, and zinc; caloric content; and oxygen uptake of the sediments.
3) Concentration of iron, manganese, and zinc; caloric content; hemolymph ion concentration; osmotic pressure; and oxygen uptake of Chaoborus (phantom midge) and Chironomus (bloodworm).

Studies were initiated in 1974 to measure vertical distribution in physicochemical variables species composition and diversity of zooplankton, algae and benthic macroinvertebrates and distribution and growth of fish (McClintock and Wilhm 1977, Parrish and Wilhm 1978).

Two stations were established at 2m and 15m depths in each of the three arms of Arbuckle Lake and two in the central pool, at 24m and 26m. Numbers in parentheses specify water depth in meters. Station numbers correspond to those used in concomitant studies. Following is a list of the stations established on the Lake, as indicated in Figure 3.

3(2)
3(15)
4(2)
4(15)

3(2)
3(15)
4(2)
4(15)

5(2)
5(15)
2(24)
1(26)

Methods

Sampling was conducted from May, 1977, through December, 1978. Most variables were not measure during this entire period. The sampling dates for the different variables and the stations involved are given on the tables in the "Results" section.

Physicochemical Condition of the Water

Physicochemical conditions of the water about 1m above the bottom were measured during the entire study. Temperature and dissolved oxygen (DO) measurements were taken with a Yellow Springs Instrument (YSI) meter (Model 54A) and conductivity with a YSI conductivity meter (Model 33). Alkalinity was determined by titration with 0.02 N sulfuric acid to a methyl purple endpoint (APHA 1977). The pH of the water and sediments were recorded using a Corning Model 610A pH meter. Transparency was measured with a Secchi disk.

Particle Size, Organic Carbon, Phosphorus, Minerals

Particle size of the sediments was determined by hydrometric analysis during 1977. Four replicate samples were collected at each station by scooping the top 2 cm from an Ekman dredge sample. In the laboratory, the samples were oven dried at 60°C, ground to size 60 mesh, and sieved to remove leaf litter. A mixture of 25 g of sediment and 50 ml distilled water was brought to a pH of 9 by adding a 2% solution of sodium carbonate-sodium bicarbonate ($Na_2CO_3 \cdot NaHCO_3$). After dispersal with a Hamilton Beach mixer, samples were transferred to a 1000 ml cylinder and placed in a constant temperature room. The samples were stirred and hydrometer readings followed at intervals of 0.5, 1, 3, 10, 20, 120, and 720 min. Percentages of soil in suspension were calculated by the formula described in ASTM (1977), while the diameter of the soil particles were determined by Stokes Law.

Organic carbon content was determined for four replicate sediment samples during 1977 and 1978. Sediment was obtained by pushing a polyethylene tube, 2.54 cm in diameter, into an Ekman grab sampler. The cores were sealed with cork stoppers, placed on ice, and frozen at the laboratory until analyzed. A 0.25 g sample from the top 2 cm of the core, which had been oven dried at 60°C, was analyzed for carbon by the Schollenberger method, an oxidation procedure (Black 1968).

Phosphorus content of the sediment was determined for four replicate samples during the entire study. Sediment from 1, 5, and 10 cm sections of each sample was muffled at 550°C for 1 h and analyzed for particulate phosphorus by the method of Stainton et al. (1974).

An identification of the clay minerals present in the sediment at each lake station was made by means of X-ray diffraction. A slurry was made with the fraction of the sediment less than 2 μ in particle size. The slurry was distributed on a ceramic tile slide and allowed to dry at room temperature. A diffractogram was obtained by scanning at 2° 2θ per minute.

Rate of sedimentation was estimated for sediment collected in traps suspended 1 m from the bottom until spring, 1978. The traps were constructed of two 3.7 x 8.8 cm cedar boards, 60 cm long, bolted together at right angles. Eight 500 ml nalgene collecting bottles with a 43 mm mouth opening were attached to grooves in the boards on each trap. Four were placed right side up and four upside down. After exposure, samples were oven dried at 100°C for 1 h. The weight of the aufwuchs in the upside down bottle was subtracted from the weight of the corresponding upright bottle to obtain the sediment weight free of biomass contributed by attached organisms. Samples were ashed at 550°C for 1 h and reweighed.

For both lakes, comparisons of particle size, organic matter, and phosphorus among stations and sampling dates were made with an analysis of variance (AOV). An AOV was also used for comparisons of sedimentation rate.

Iron, Manganese, Copper, and Zinc

The concentration of iron and manganese in the sediments and water were made during the entire study. Copper was analyzed in 1977 and zinc in 1978. For determining dissolved and total concentrations of heavy metals in the water, six water samples were collected with a Van Dorn water bottle about 1 m above the sediment. Three acid-rinsed 250 ml bottles were filled and returned to the laboratory for determining turbidity and dissolved metals. The other three water samples were acidified with 3 ml of concentrated HNO_3 which keeps the pH below 2 and prevents change in metal content during storage.

For determining concentration of metals in the sediment, three (six in later studies) replicate sediment samples were obtained by inserting 2.5 cm polyethylene tubes into a 15 x 15 cm Ekman grab sampler. The ends of the tubes were sealed with corks and they were stored in ice until they could be frozen in the laboratory until analyzed.

Chaoborus and Chironomus were collected with an Ekman grab sampler and placed on ice until they could be refrigerated.

In the laboratory, a small portion of three of the ice water samples was used to determine turbidity using a Spectronic 20. The remainder of each sample was filtered through 0.45 μ millipore filter apparatus and acidified with 3 ml of concentrated HNO_3. The 200 ml of the filtered water samples were placed in acid-rinsed beakers and evaporated on a 100° hot plate. After being brought to 25 ml volume with 0.2N HNO_3, the samples were analyzed on a Varian Techtron Atomic Absorption Spectrometer (AA) for dissolved iron and managese and copper or zinc.

To obtain total concentrations of heavy metals, 200 ml of each of the other three water samples were placed in acid-rinsed beakers, 3 ml of concentrated HNO_3 added, and the samples evaporated on a 100°C hot plate. The residue was digested with 3 ml of concentrated HNO_3 until light colored, 3 ml of 1:1 HCl was added to the water to dissolve the residue, and the volume brought up to 25 ml with 0.02N HNO_3. The samples were then analyzed on the AA.

The surface 3 cm of sediment was removed from the core samples, placed in acid-rinsed 200 ml plastic bottles, and dried at 60°C. Dry weight of the sediment was determined and half of a 1:10 ration of ammonium acetate to sediment weight was added (some of the sediment in each tube was withdrawn for analysis of caloric content). After shaking, the flasks were left 12 h. The samples were filtered in a Buchner funnel and the remaining ammonium acetate added to leach out remaining sorbed metals (Jackson 1958). The filtrate was then analyzed for sorbed. During 1977, soluble metals were also determined; however, since values were negligible this was discontinued in 1978.

During 1978, the concentration of heavy metals in Chaoborus and Chironomus were determined. After being refrigerated overnight, the organisms were separated from the debris and placed in clean water for 12 h to allow evacuation of gut contents. After drying at 60°C for 12 h, the sample was divided into aliquots, one for measuring concentration of heavy metals and one for measuring caloric content (see next section).

Tared duplicate samples of about 0.1 g dry weight of Chaoborus or Chironomus were fired at 550°C for 12 h in a muffle furnace. After cooling in a dessicator, the samples were reweighed to determine ash-free weight. The muffled chaoborid samples were washed into 25 ml acid-rinsed flasks using 10 ml of 0.1N HCl. The muffled chironomid samples were washed into 25 ml flasks using 25 ml of 0.1N HCl. These solutions were analyzed on the AA. The final concentrations were calculated using the following equation:

$$\text{mg/l} \times \frac{\text{volume (ml)}}{\text{sample wt. (g)}} = \mu\text{g/g organism}$$

Caloric Content

During 1978, some of the sediment in the tubes and aliquots of Chironomus and Chaoborus described in the previous section were analyzed for caloric content. A Phillipson microbomb calorimeter, standarized with 10 samples of benzoic acid ranging from 5-10 mg, was used. Separate samples of Chaoborus and Chironomus, ranging in weight from 1-10 mg, were formed into a pellet with a press, tared, charged with oxygen, and allowed to equilibrate. After firing, a Honeywell Electronik 19 Recorder recorded the curve resulting from the heat liberated. From this the calories liberated was analyzed and calculated as cal/g dry weight. Concentrations were determined using the following equation:

$$\text{mg/l} \times \frac{\text{volume (ml)}}{\text{weight (g)}} = \mu\text{g/g sediment}$$

Caloric content of the sediment was determined for three samples of sediment from each station. For each sample, an amount of sediment and an amount of benzoic acid approximately equal to the weight of sediment was weighed. Both the sediment and the benzoic acid were mixed in a mortar and pestle. Three samples, weighing from 7-16 mg, were analyzed for caloric content. Possible calories due to the amount of benzoic acid present were subtracted to determine the calories liberated by the sediment. The value was divided by sediment weight to give cal/g dry weight sediment.

Hemolymph and Lake Water Ion Analysis

Chironomus and Chaoborus were collected with an Ekman grab sampler and placed in bottom water collected with a Van Dorn water sampler. The jars were then placed in ice chests, returned to the laboratory, and maintained at temperatures measured in the hypolimnion.

For collection of hemolymph samples, the dipteran larvae were surface dried on Kimwipes and secured to masking tape on a microscope slide. Body weights of the larvae were determined to the nearest 0.01 mg on a Mettler balance (Model H20). Under a dissecting microscope, a dorsal puncture was made with a #2 insect pin on the larvae's mesothorax. The hemolymph issuing from the wound was collected in disposable glass micro-pipettes (Microcaps, Drummond Scientific Co.). Samples were discarded if they became contaminated with other body tissues. All samples were analyzed immediately after collection, since freezing changes the hemolymph (Stephen and Johnson 1962).

Measurements of sodium (Na^+) and potassium (K^+) were made on individual insects. Hemolymph Na^+ and K^+ samples (0.5 μl) were transferred into 500 μl of deionized water for analysis with a Beckman 440 atomic absorption spectrophotometer. The chloride (Cl^-) analysis was made directly without dilution and measured with a Fiske Marius microchlor-o-counter. Because of limited amount of hemolymph obtained from Chaoborus sp., several larvae of approximately the same weight were used to obtain sufficient quantities for one chloride analysis.

Lake water from each sampling station was analyzed for Na^+ and K^+ using and AA. Chloride in the lake water was measured with a chlor-o-counter.

Freezing point depressions of the hemolymph and lake water were measured with a Clifton Technical Physics Nanoliter Osmometer, sensitive to the nearest ± 0.001°C (Frick and Sauer 1973). Because of the minute quantity of fluid required, less then 0.1 µl was obtained from an individual organism for each assay.

Hemolymph ion analyses and osmolarities were determined in controlled laboratory experiments after the organisms were subjected to various temperatures, dissolved oxygen, and salinity levels. The temperatures selected for the controlled ion and oxygen consumption experiments were 3, 8, 12, 17, 22, 28, 32, and 38°C. The DO levels were 0, 3, 6, and 10 ppm, while the salinity concentrations were 0.1, 0.3, 0.5, and 0.8% NaCl.

Oxygen Consumption

Organisms were collected from the lakes as described in the previous section. Oxygen consumption measurements were taken on a Gilson respirometer. Larvae were selected on the basis of uniform size.

Larvae were placed into 15 ml reaction flasks with 5 ml of lake water. Ten chironomid (McFarland and McLusky 1972) and 20 chaoborid larvae were used per flask for the oxygen tests. For carbon dioxide absorption, 0.2 ml of a 10% KOH solution was added to Whatman No. 1 filter paper in the center well of each reaction flask. The flasks were attached to the respirometer with the water bath at the temperature of the lake water from which the organisms were collected.

During the experiments, the flasks were shaken at a constant, moderate speed. This allowed the larvae to respire without making the characteristic wriggling movements and therefore reduce the oxygen consumed by the animals due to activity (McFarland and McLusky 1972). Oxygen consumption was measured between 1200 and 1800 h to minimize the influence of any circadian respiratory patterns (Buffington 1968). The respirometer was kept in darkness so that variations in lighting would not affect the larvae. Five milliliters of lake water were placed in control flasks to correct for oxygen uptake by factors other than the larvae (Umbreit et al. 1972, Cole and Wilhm 1973).

After oxygen uptake measurements were made, the larvae were removed, dried with Kimwipes, and weighed. They were then oven-dried at 105°C for 24 h and reweighed. Results were expressed in μl O_2 consumed/mg dry wt/hr.

Oxygen consumption was measured at eight temperatures (3, 8, 12, 17, 22, 28, 32, and 38°C). Respiratory rate determinations were made five times each at all experimental temperatures. All tests lasted 4 to 6 h with readings taken every 15 min. Oxygen uptake measurements were also made on larvae subjected to various dissolved oxygen and salinity levels.

Oxygen consumption of the sediments was measured at several stations in both lakes during 1978. An Ekman grab sampler was used to collect the sediment. Core tubes were placed into the sediment within the dredge. Corks were inserted at both ends with the overlying water being captured to delineate the top of the core. All sediment samples were subsectioned into 0.0 - 1.0, 4.0 - 5.0, and 9.0 - 10.0 cm fractions and used within 24 h for the respiration studies. To each reaction flask, 3 g of sediment (wet wt) and 2 ml of bottom lake water were added. One reaction flask contained 2 ml of lake water as a control. All sediment consumption tests ran 3 h with an initial incubation period of 15 min (Liu 1973).

Results

Physicochemical Conditions of the Water

Since little variation existed among physicochemical variables measured at the same depths in the arms or between the 20 and 24 m depths in the central pool, data were averaged in Table 1. Mean temperature varied from 5°C on March 15 to 31°C at 2 m on July 20 and August 22. Lower temperatures undoubtedly existed in January and February; however, an extensive ice cover precluded sampling. Little variation existed between the 15 and 22 m depths in fall and winter. The maximum temperature measured in the central pool was 17°C on several dates and 26°C at the 15 m stations.

Dissolved oxygen (DO) varied from 0.1 - 12.0 mg/l. DO remained relatively high at the 2 m station during 1977. Values were similar at 15 m and in the central pool except the deeper stations reached minimum levels earlier in the spring and remained lower longer in the autumn. It appears that DO is severely limiting at the bottom for approximately 5 months in the central pool and less than 4 months at 15 m.

Conductivity varied from 298-538 μmhos. Values tended to decrease with depth; however, differences were not significant. Values tended to be highest in warm-water conditions, while minimum values were recorded in December.

Alkalinity did not exhibit large fluctuations. Values ranged from 127-165 mg/l and were generally slightly greater in the central pool than at the 15 m depths; however, differences were not significant. Values tended to increase during summer and then decrease during during autumn.

pH of the bottom water was generally similar in the arms and the central pool. Values ranged from 7.4-8.4. On June 9 the low mean in the central pool was caused by a pH of 6.4 at Station 1 (24). At other times pH values were

TABLE 1

MEAN TEMPERATURE, DISSOLVED OXYGEN, CONDUCTIVITY, ALKALINITY, pH, AND TURBIDITY OF THE BOTTOM WATER IN THE ARMS (2, 15 m) AND THE CENTRAL POOL AND SECCHI DISK TRANSPARENCY IN 1977 AND 1978*

Variable	DEPTH (m)	MAY 16	JUNE 13	JULY 20	AUG. 22	OCT. 21	DEC. 30	MAR. 15	APR. 29	JUNE 9	JULY 11	AUG. 15	SEP. 16	OCT. 22	NOV. 24	DEC. 16
Temperature	2	-	29	31	31	20	-	-	-	-	-	-	-	-	-	-
(°C)	15	15	19	21	25	20	8	5	17	19	24	26	26	20	14	8
	22	13	15	15	16	16	8	5	11	14	16	17	17	17	14	8
Dis. Oxygen	2	8.4	6.5	6.2	7.9	7.9	-	-	-	-	-	-	-	-	-	-
(mg/1)	15	2.2	0.1	0.1	0.1	7.5	12.0	8.8	5.4	1.0	0.1	0.1	2.5	4.9	-	8.4
	22	0.4	0.1	0.1	0.1	5.8	12.0	8.6	4.7	0.5	0.1	0.1	0.2	2.4	-	7.7
Conductivity	2	-	499	538	480	425	-	-	-	-	-	-	-	-	-	-
(μmhos)	15	388	396	428	497	418	274	310	417	385	425	460	457	425	352	298
	22	380	380	410	485	410	273	315	390	410	422	455	410	422	348	305
Alkalinity	2	-	-	-	-	-	-								-	
(mg/1 $CaCO_2$)	15	150	127	139	141	136	-	150	151	128	140	137	129	136	131	-
	22	145	129	134	160	136	-	150	151	138	141	165	161	138	132	-
pH	2	-	7.8	8.2	7.4	7.4	-	-	-	-	-	-	-	-	-	-
	15	7.9	8.0	7.6	7.4	7.6	7.9	7.6	8.1	8.1	8.0	8.2	8.1	8.1	8.3	8.4
	22	7.6	8.0	7.7	7.4	7.5	7.9	7.6	8.2	7.1	8.2	8.0	8.3	8.0	8.3	8.4
Turbidity	2	-	12	14	8	4	-	-	-	-	-	-	-	-	-	-
(JTU)	15	6	13	13	9	10	-	11	10	13	32	26	31	10	7	-
	22	13	7	12	10	9	-	8	9	4	12	17	23	20	4	
Secchi Disk (m)	SURF.	-	-	-	-	-	-	-	2.0	1.1	1.7	1.8	1.8	1.3	1.3	1.8

\- Data not measured

* Values are the means of the three 2 m stations and three 15 m stations and two stations (20 and 24 m) in the central pool in 1977.

in the central pool. Although no consistent trends were observed, values in 1978 were generally greater than values in 1977.

Turbidity of the bottom water was generally low. Values were generally slightly higher at the 15 m depths than in the central pool. A maximum value of 32 JTU was reached on June 11 at 15 m, while the maximum in the central pool was 23 JTU on September 16. Secchi disc transparency ranged from 1.1 m on June 9 to 2.0 m on April 29.

Particle Size, Organic Carbon, Phosphorus, Minerals

The percent and diameter of sediment remaining in suspension at progressive time intervals after mixing are shown in Table 2. Particle size of the sediment decreased with station depth in Arbuckle Lake. After 12 h, over 60% of the particles remained in suspension at stations in the central pool, while 51-56% remained at the 15 m depths. The 2 m depths had less than 25% of the particles <0.002 mm in suspension after 12 h. Examination of the trends shows that the 2 m stations had a more even distribution of silt-sized particles than the deeper stations. Seasonal variation was also statistically significant (AOV, p = 0.05). Particle size generally decreased from May 12 to June 13 changed little over the summer, and increased from August 18 to October 20.

Values of most sediment variables have also combined by depth (Table 3). The variation in percent organic matter was slight, 3.8-6.8%. Neither seasonal nor station variation was significant. From October 20 to the end of the study, values in the central pool exceeded those at 15 m. Percent organic matter tended to be higher in 1978 than in 1977 and increased between April 29 and June 9, 1978, accompanying anoxic conditions. Values were generally less at the 4-5 cm depth than in the surface layer.

TABLE 2

GRAIN SIZE ANALYSIS OF THE SEDIMENTS*,†

Station	Time (min.)					
(Depth - m)	0.5	1	5	20	120	720
3(2)	70	62	48	38	29	24
3(15)	90	88	86	82	69	56
4(2)	78	72	54	39	28	22
4(15)	89	88	87	81	69	55
5(2)	79	72	53	40	31	25
5(15)	80	78	77	73	62	51
2(20)	92	91	90	87	77	62
1(24)	90	89	88	85	75	61
Diameter (mm).	(.070)	(.050)	(.022)	(.011)	(.005)	(.002)

* Values are percent of soil remaining in suspension at specified times; values in parentheses at the bottom of each column are the maximum diameter of particles remaining in suspension.

† Values are the means of four replicate cores each collected on May 12, June 15, July 18, September 20, October 18, 1977.

Loss on ignition as determined gravimetrically was compared with the organic matter as determined by oxidation. Although the oxidation procedure has been questioned as a technique when clay content is high, similar trends were observed between values generated by the two techniques. Loss on ignition was greater in the central pool than in the arms,and values were less at the 4-5 cm depth than in the surface layer. However, loss on ignition exhibited a seasonal trend. Values increased in summer during anoxic conditions and then decreased in the fall. Loss on ignition values varied from 8.5-14.2% in the central pool and 7.9-12.5% in the arms, considerably higher values than observed for organic carbon.

pH of the sediments varied from 6.6-8.4. No consistent nor significant trend existed between depths. The lake was more alkaline in 1978 than in 1977.

Orthophosphate varied from 16-37 μg/g. Values in the central pool were significantly greater than values in the arms ranging from 21-37 in the central pool and 16-28 μg/g at 15 m. In both areas, surface sediments generally contained more orthophosphate than the 4-5 cm layer. In the arms and central pool values of both layers decreased during summer and then increased during fall.

The depth of sediments deposited varied from 0.4-39.4 mm/month, while oven-dry weight varied from 41-5150 g/m^2 per month (Table 4). Variation among sampling areas and time were significant. Generally, less sediment was deposited in the central pool than in the arms. The amount deposited in the central pool ranged from 0.4-17.8 mm/month. Values were high in the period from September 16 - October 20, 1977 and extremely low from December 30 - March 15, 1978 when the lake was ice-covered.

TABLE 3

ORGANIC MATTER, LOSS ON IGNITION, pH, AND
ORTHOPHOSPHATE OF THE SEDIMENTS IN THE ARMS
(2, 15 m) AND THE CENTRAL POOL IN 1977 AND 1978*

Variable	Water Depth (m)	Sediment Depth (cm)	MAY 5	JUNE 15	JULY 18	AUG. 18	OCT. 20	DEC. 30	MAR. 15	APR. 29	JUNE 9	JULY 11	AUG. 15	SEP. 16	OCT. 22	NOV. 24	DEC. 16
Organic	2	0-1	4.4	6.4	6.0	6.8	3.8	-	-	-	-	-	-	-	-	-	-
Matter	15	0-1	4.8	4.6	4.7	4.8	6.0	4.4	6.7	4.7	5.2	5.1	5.3	5.8	5.9	5.9	5.2
(%)		4-5	-	-	-	-	-	-	-	-	5.4	5.0	5.0	5.6	5.3	5.6	5.2
	22	0-1	4.3	4.7	4.0	4.5	6.6	6.5	6.8	5.8	6.0	6.4	6.8	6.2	6.7	6.3	6.3
		4-5	-	-	-	-	-	-	-	-	5.6	6.0	6.2	6.4	6.4	5.5	6.2
Loss on	15	0-1	-	-	-	-	-	-	-	10.9	12.0	11.5	12.5	9.6	8.7	9.1	8.4
ignition		4-5	-	-	-	-	-	-	-	9.4	9.6	8.7	11.5	9.2	8.1	9.0	7.9
(%)	22	0-1	-	-	-	-	-	-	-	12.6	11.5	14.1	14.2	10.9	10.8	10.1	8.4
		4-5	-	-	-	-	-	-	-	12.1	12.1	13.8	14.1	10.9	10.2	8.5	9.6
pH	15	0-1	7.8	7.9	6.6	6.8	7.4	7.4	7.5	8.0	8.2	8.0	8.2	8.2	8.1	8.4	8.4
	22	0-1	7.6	7.7	6.7	7.2	7.2	7.6	7.5	8.3	7.9	7.9	8.0	8.3	7.9	8.3	8.4
Phosphorus	15	0-1	-	-	-	-	-	-	-	26	17	19	21	21	19	21	26
(μg/g)		4-5	-	-	-	-	-	-	-	28	16	17	19	20	17	21	25
	22	0-1	-	-	-	-	-	-	35	37	26	23	27	26	25	25	37
		4-5	-	-	-	-	-	-	34	37	21	22	26	25	24	24	34

\- Data not measured

* Values are the means of the four samples each at the three 2 m, and three 15 m stations and two stations (20 and 24 m) in the central pool.

The types and relative amount of quartz and clay minerals in the sediments were identified by X-ray diffraction. Quartz was generally dominant in the sediments of Arbuckle Lake, while kaolinite and mica were relatively weak (Table 5). Montmorillonite was prevalent at Stations 5(2) and 3(15) only, while a significant amount of monmorillonite-vermiculite mixture was present at Stations 3(2), 4(2), and 4(15).

Iron, Manganese, Copper, and Zinc

During 1977 and 1978, total iron in the water ranged from nondetectable to 0.79 mg/l (Table 6). Soluble iron averaged less than 18% total iron (Table 7). Little variation existed among depths in 1977; however, values were slightly greater at 15 m than at the central pool in 1978. Although seasonal variation was slight at 15 m, values were considerably lower in 1978 than in 1977 in the central pool. Changes did not accompany stratification.

Sorbed iron exhibited considerably more fluctuation in the sediments than in the bottom water with values ranging from nondetectable to 14.44 μg/g sediment (Table 6). Values generally increased with depth. A decrease occurred during late spring in both years. The concentration of sorbed iron increased abruptly at the 15 m depths and in the central pool between the December 1977 and March 1978 sampling dates (Table 8).

Total iron of the sediment ranged from 14,400 to 39,672 μg/g during 1977. The values increased with depth from the 2 m stations to the 15 m stations.

Total manganese in the water ranged from nondetectable to 1.16 mg/l (Table 6). Of the manganese present in the water column, soluble manganese accounted for an average of 59% (Table 9). Concentrations averaged less at the 2 m stations than the 15 m stations in 1977. In 1978 maximum concentrations were found at 2(24). Concentrations were generally higher in 1978

TABLE 4

DEPTH AND OVER-DRY WEIGHT OF
DEPOSITED SEDIMENTS IN 1977 AND 1978*

Station	JUNE 29-JULY 27		JULY 27-SEP. 16		SEP. 16-OCT. 20		OCT. 20-DEC. 30		DEC. 30-MAR. 15		MAR. 15-JULY 11		JULY 11-SEP. 16	
(Depth in m)	mm/mo	g/m²/mo	mm/mo	g/m²/mo	mm/mo	g/m²/mo	mm/mo	g/m²/mo	mm/mo	g/m²/mo	mm/mo	g/m²/mo	mm/mo	g/m²/mo
3(8)	4.2	364	18.0	3114	-	-	-	-	-	-	-	-	-	-
3(15)	7.1	767	15.0	1625	39.4	5000	-	-	1.2	178	-	-	-	-
4(8)	13.3	1826	13.5	1835	17.8	314	7.9	1271	1.4	211	12.4	2372	-	-
4(15)	7.8	818	12.0	1528	37.8	3150	16.4	2615	1.1	144	12.2	1912	-	-
5(8)	18.2	2706	-	-	-	-	4.4	886	1.6	380	-	-	-	-
5(15)	11.5	1255	18.0	2651	18.0	2486	4.1	634	1.6	159	13.1	1970	-	-
2(20)	5.0	559	8.8	505	15.6	1587	16.2	2083	0.4	41	-	-	7.3	484
1(24)	5.8	396	7.9	527	12.4	1373	17.8	2346	-	-	-	-	5.8	364

\- Data not measured.

* Values are means of four replicate samples.

TABLE 5

RELATIVE AMOUNTS OF QUARTZ AND CLAY MINERALS IN 1977*

Station (Depth in m)	Quartz	Kaolinte	Mica	Montmor-illonite	V-M**
3(2)	W-M	W-M	W	O	S
3(15)	S	M	W-M	S	O
4(2)	S	W-M	W	O	M
4(15)	M	S	W	O	M
5(2)	S	W	W	S	O
5(15)	S	W	W	S	O
1(24)	S	W-M	W	O	W-M

* S-strong, M-moderate, W-weak, O-none detected.

** V-M-vermiculite-montmorillonite.

TABLE 6

TOTAL IRON AND MANGANESE (mg/l) OF THE BOTTOM WATER AND SORBED METALS (µg/g) OF THE SEDIMENTS IN 1977 AND 1978

Metal	Depth (m)	Water / Sediment	MAY 16	JUNE 14	JULY 19	AUG. 18	OCT. 29	DEC. 3	MAR. 15	APR. 22	MAY [illegible]	JUNE 11	[illegible] 13	SEPT. 16	OCT. 22	NOV. 24
Iron	2	W	-	.51	.56	.54	-	-	-	-	-	-	-	-	-	-
		S	ND	.45	.08	.54	ND	-	-	-	-	-	-	-	-	-
	15	W	.62	.78	.79	.44	.65	-	.65	.54	.65	.41	.68	.24	[illegible]	.51
		S	.50	1.16	.42	.63	.13	ND	10.13	.99	1.13	.63	.41	.21	1.35	.42
	22	W	.58	.43	.06	.58	-	-	.29	.30	.53	.20	.51	.26	.29	.54
		S	2.79	1.25	.97	.51	.69	1.17	14.44	4.80	1.11	1.16	.59	.51	1.26	1.58
Manganese	2	W	-	.09	.07	.09	.02	-	-	-	-	-	-	-	-	-
		S	68	89	50	118	104	-	-	-	-	-	-	-	-	-
	15	W	.13	.53	.67	.55	.07	.04	.03	.86	.14	.25	.08	.05	.05	.05
		S	-	-	-	-	-	-	200	214	269	167	108	255	152	159
	22	W	.33	.17	1.08	1.16	.14	.02	.04	.05	.46	1.11	1.88	.16	.20	.07
		S	317	378	477	591	579	498	576	384	407	457	469	407	444	554

TABLE 7

DISSOLVED AND TOTAL IRON (mg/l) OF THE WATER IN THE HYPOLIMNION

Station (Depth in m)	Dissolved / Total	MAY 16	JUNE 14	JULY 19	AUG. 18	OCT. 20	DEC. 30	MAR. 15	APR. 29	JUNE 9	JULY 11	AUG. 15	SEP. 16	OCT. 22	NOV. 24
3(2)	D	-	-	.07	.09	.07	-	-	-	-	-	-	-	-	-
	T	-	.75	-	.65		-	-	-	-	-	-	-	-	-
3(15)	D	.30	.02	.13	.17	.03	-	-	-	-	-	-	-	-	-
	T	.69	.67	.62	.26		-	-	-	-	-	-	-	-	-
4(2)	D	-	.02	.05	.01	.08	-	-	-	-	-	-	-	-	-
	T	-	.66	.45	.54		-	-	-	-	-	-	-	-	-
4(15)	D	.06	.26	.06	.17	.04	-	-	-	-	-	-	-	-	-
	T	.54	.95	.82	.60		-	-	-	-	-	-	-	-	-
5(2)	D	-	.03	.32	.01	.05	-	-	-	-	-	-	-	-	-
	T	-	.11	.66	.42		-	-	-	-	-	-	-	-	-
5(15)	D	-	.16	.07	.01	.05	-	.06	.08	.03	.03	.03	.04	.04	.03
	T	.50	.73	.94	.45		-	.65	.34	.63	.41	.68	.24	.20	.51
2(20)	D	.12	.18	.07	.03	.11	-	.02	.07	.07	.03	.11	.12	.05	.03
	T	.58	.43	.66	.58		-	.26	.27	.33	.20	.47	.22	.27	.42
1(24)	D	-	-	-	-	-	-	.13	.06	.03	.03	.08	.06	.02	-
	T	-	-	-	-	-	-	.32	.34	.33	.20	.56	.29	.31	.26

\- Data not measured.

TABLE 8

SORBED IRON (µg/g) IN THE SURFACE SEDIMENTS IN 1977 AND 1978

Station (Depth in m)	MAY 16	JUNE 14	JULY 19	AUG. 18	OCT. 20	DEC. 30	MAR. 15	APR. 29	JUNE 9	JULY 11	AUG. 15	SEP. 16	OCT. 22	NOV. 24
5(2)	-	0.52	ND	0.07	ND	-								
5(15)	0.85	1.18	0.35	0.54	ND	ND								
4(2)	-	0.30	0.13	0.30	ND	-								
4(15)	0.66	1.68	0.66	0.75	0.38	ND								
3(2)	ND	0.53	0.13	0.66	ND	-								
3(15)	ND	0.62	0.26	0.67	ND	ND	10.13	0.99	1.15	0.68	0.41	0.22	1.03	.42
2(20)	2.79	1.03	0.97	0.31	0.69	1.17	11.00	8.32	0.91	1.15	0.55	0.33	1.17	.47
1(24)							17.89	1.29	1.31	1.18	0.48	0.68	1.34	2.30

than 1977 at the deep water stations. Manganese concentrations increased in the hypolimnial waters as summer progressed, reaching a high in August.

Sorbed manganese concentrations in the sediments were always larger than hypolimnial water concentrations. Concentrations of sorbed sediment manganese ranges from 50 μg/g to 591 μg/g sediment (Table 10). The 2 m stations averaged lower concentrations of manganese than the 15 m stations in 1977. The sorbed manganese concentrations increased as the summer progressed at the deep water stations, reaching its highest concentration in August 1977 and 1978.

Total copper concentration of the hypolimnial waters fluctuated throughout the summer of 1977, ranging from nondetectable to 16.7 mg/l (Table 11). Soluble copper averaged 70% of the total copper concentration in the water.

The 1977 concentrations of sorbed in the sediment fluctuated from 11.1 to 55.3 μg/g (Table 12). Concentrations were greater at the 15 m stations than at the 2 m stations. There were no obvious seasonal trends in 1977 exhibited by concentrations of copper in the hypolimnial waters or sorbed in the sediments.

Total zinc concentrations of the hypolimnial waters ranged from 0.001 mg/l to 0.032 mg/l in 1978 (Table 13). Soluble zinc averaged 89% of the total zinc present in the water. The concentrations fluctuated throughout the summer; however, no seasonal changes in concentration were evident.

Sorbed zinc concentrations in the sediment were generally low, ranging from 0.05 μg/g to 1.66 μg/g sediment throughout 1978 (Table 14). Considerable fluctuation occurred in the sorbed zinc concentration throughout 1978 but no obvious seasonal trends were observed.

Caloric Content

The caloric content of the sediment at Stations 1, 2 and 5 decreased from March to an average low of 50 cal/g sediment in August. The values then

TABLE 9

DISSOLVED AND TOTAL MANGANESE (mg/1) OF THE WATER IN THE HYPOLIMNION

Station (Depth in m)	Dissolved / Total	MAY 16	JUNE 14	JULY 19	AUG. 18	OCT. 20	MAR. 15	APR. 29	JUNE 9	JULY 11	AUG. 15	SEP. 16	OCT. 22	NOV. 24
3(2)	D	-	-	.01	.02	.01	-	-	-	-	-	-	-	-
	T	-	-	.05	.08	.03	-	-	-	-	-	-	-	-
3(15)	D	.005	.52	.68	.33	.01	-	-	-	-	-	-	-	-
	T	.12	.53	.77	.20	.09	-	-	-	-	-	-	-	-
4(2)	D	-	.003	.01	.03	.01	-	-	-	-	-	-	-	-
	T	-	.12	.07	.14	.02	-	-	-	-	-	-	-	-
4(15)	D	.12	.62	.52	.53	.02	-	-	-	-	-	-	-	-
	T	.19	.68	.64	.73	.09	-	-	-	-	-	-	-	-
5(2)	D	-	.002	.02	.02	.01	-	-	-	-	-	-	-	-
	T	-	.06	.08	.05	.01	-	-	-	-	-	-	-	-
5(15)	D	-	.36	.66	.54	.01	.02	.02	.20	.13	.19	.05	.01	.01
	T	.08	.42	.58	.68	.02	.04	.03	.86	.14	.25	.08	.05	.05
2(20)	D	.20	.13	1.0	1.32	.04	.01	.02	.03	.45	1.11	1.23	.08	.04
	T	.33	.17	1.08	1.16	.14	.02	.04	.05	.46	1.11	1.88	.16	.08
1(24)	D	-	-	-	-	-	.02	.06	1.46	.31	1.75	1.68	.13	.03
	T	-	-	-	-	-	.02	.14	1.48	.32	2.36	1.87	.24	.07

- Data not measured.

TABLE 10

SORBED MANGANESE (μg/g) IN THE
SURFACE SEDIMENTS IN 1977 AND 1978

Station (Depth in m)	MAY 16	JUNE 14	JULY 19	AUG. 18	OCT. 20	DEC. 30	MAR. 15	APR. 29	JUNE 9	JULY 11	AUG. 15	SEP. 16	OCT. 22	NOV. 24
3(2)	-	97	114	164	39	-								
3(15)	332	428	347	212	217	249								
4(2)	-	120	84	121	216	-								
4(15)	274	390	185	457	152	279								
5(2)	68	52	54	69	57	-								
5(15)	122	177	150	326	164	206	200	214	269	167	108	233	132	159
2(20)	317	378	477	391	379	498	344	424	461	535	517	432	514	393
1(24)							408	344	353	379	420	383	375	315

TABLE 11

DISSOLVED AND TOTAL COPPER (mg/l) OF THE WATER IN THE HYPOLIMNION

Station (Depth in m)	Dissolved / Total	MAY 16	JUNE 14	JULY 19	AUG. 18	OCT. 20
3(2)	D	-	-	7.0	6.0	.6
	T	-	5.4	7.5	3.8	7.2
3(15)	D	4.8	4.2	5.2	8.2	2.7
	T	5.1	4.1	5.7	6.0	4.8
4(2)	D	-	7.2	11.6	3.8	1.0
	T	-	4.8	7.2	9.2	5.9
4(15)	D	3.1	9.7	5.3	5.2	3.9
	T	5.2	6.4	9.3	5.3	8.0
5(2)	D	-	9.3	8.0	3.1	.7
	T	-	4.1	6.9	16.7	6.2
5(15)	D	2.4	6.3	6.0	3.2	.8
	T	7.9	6.4	2.2	5.8	3.8
2(20)	D	.0018	6.5	5.3	3.8	2.9
	T	6.0	6.1	6.4	4.2	1.0

Average soluble copper concentration = 70.03%

TABLE 12

TOTAL COPPER (μg/g) IN THE SURFACE SEDIMENTS

Station (Depth in m)	MAY 16	JUNE 14	JULY 19	AUG. 18	OCT. 20	DEC. 30
3(2)	-	19.6	16.3	16.3	12.3	-
3(15)	27.4	24.4	39.8	55.3	24.0	23.4
4(2)	-	21.7	16.1	39.7	22.4	-
4(15)	25.7	26.1	21.2	-	20.2	23.4
5(2)	11.0	16.1	17.2	-	14.9	-
5(15)	15.7	22.2	24.2	-	24.2	17.8
2(20)	29.6	23.1	26.3	33.1	25.1	26.1

TABLE 13

DISSOLVED AND TOTAL ZINC (mg/l) OF THE WATER IN THE HYPOLIMNION

Station (Depth in m)	Dissolved / Total	MAR. 15	APR. 29	JUNE 9	JULY 11	AUG. 15	SEP. 16	OCT. 22	NOV. 24
5(15)	D	.018	.020	.014	.009	.023	.012	.008	.005
	T	.024	.019	.019	.012	.016	.008	.009	.007
2(20)	D	.032	.024	.026	.025	.012	.004	.016	.003
	T	.030	.015	.011	.012	.018	.002	.021	.003
1(24)	D	.025	.029	.015	.012	.016	.003	.016	.001
	T	.022	.026	.013	.014	.013	.003	.018	.003

Average of soluble zinc in water = 88.96%

TABLE 14

TOTAL ZINC (μg/g) IN THE SURFACE SEDIMENTS

Station (Depth in m)	MAR. 15	APR. 29	JUNE 9	JULY 11	AUG. 15	SEP. 16	OCT. 22	NOV. 24
5(15)	1.08	.69	1.20	.89	1.03	.61	.54	1.58
2(20)	1.13	.69	.72	.87	.82	.59	.51	1.28
1(24)	1.05	.78	1.94	.86	1.66	.74	.50	1.11

increased slightly throughout the fall months at Stations 2 and 5. Station 1 maintained its nondetectable caloric content throughout November. Sediment caloric values from Station 5 fluctuated throughout the year. All stations showed a high variability in the individual calorie values.

Due to the sporadic collection of the benthic organisisms, no trends in caloric content could be observed.

Hemolymph and Lake Water Ion Analysis

The hemolymph sodium (Na^+) concentration in *Chaoborus punctipennis* was maintained at a constant mean level at all stations of 120 mM for the months of May, July, and September, 1977 (Figure 14). In June, August, and December, 1977, hemolymph Na^+ concentrations dropped to a mean value of 70 mM. Hemolymph Na^+ levels recorded in March, 1978 were comparable with those observed at all stations in 1977. the same trends were observed in fluctuations of hemolymph ion concentrations between the deeper and shallower stations. Hemolymph Na^+ decreased steadily throughout the 1978 sampling period at all stations. Average Na^+ values for March, July, and December were 116.0, 78.0 and 90 mM, respectively.

The hemolymph potassium (K^+) and chloride (Cl^-) showed little fluctuation in *Chaoborus* during the entire sampling period (Figure 2). Potassium values were maintained between 3.5 - 9.0 mM at all stations. Chloride concentrations were maintained at a mean value of 50 mM at all stations in 1977. Only slight decreases in concentration to 38 mM were observed in those months with low Na^+ concentrations. A gradual decline in Cl^- also accompanied the declining Na^+ in 1978.

The hemolymph Na^+ concentration in *Chironomus tentans* was maintained at a constant mean level of 150 mM at all stations from May through October, 1977. In June, August and December the Na^+ concentration was maintained throughout

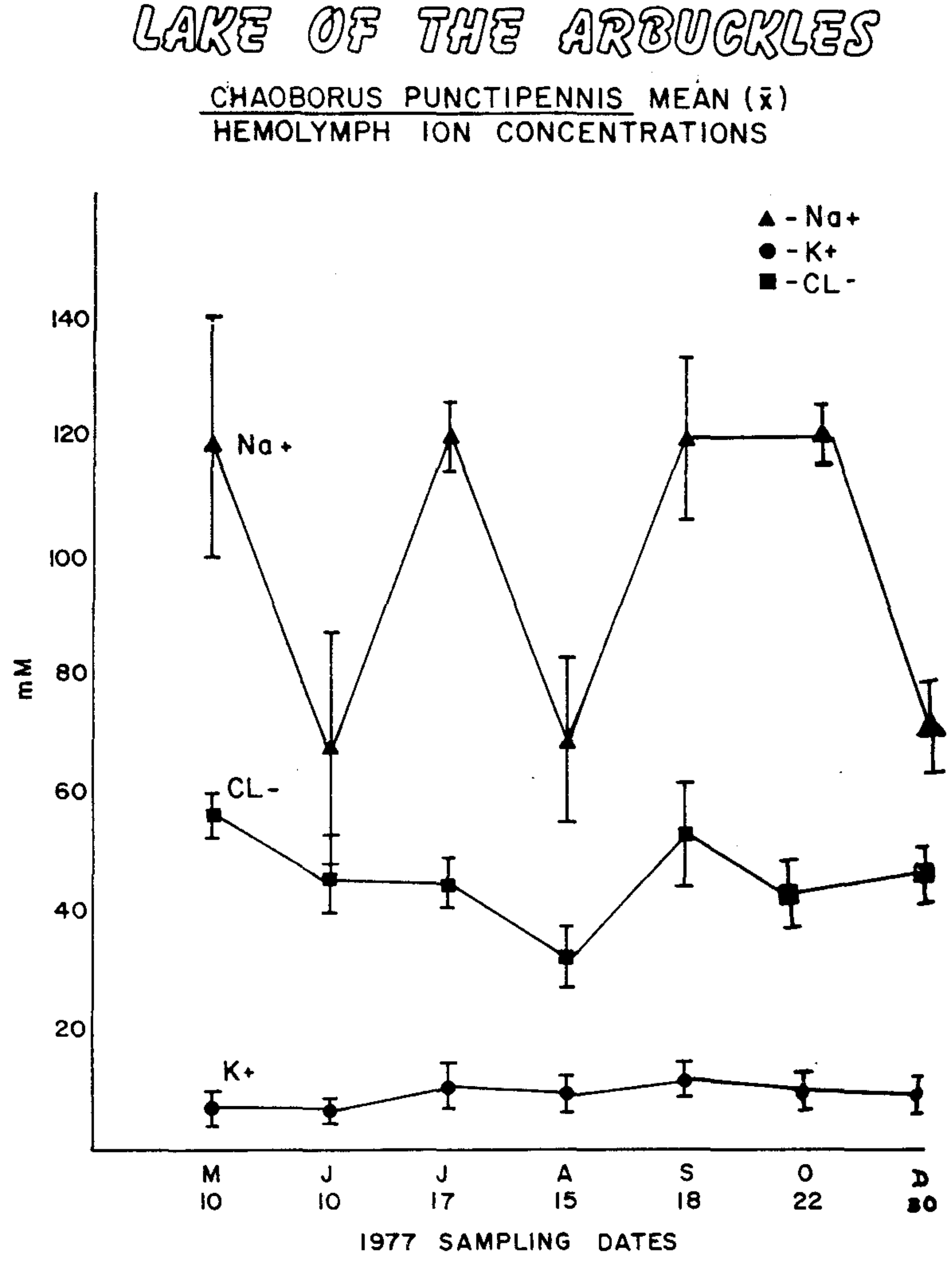

Figure 14. *Chaoborus punctipennis* Mean ($\bar{X}$) Hemolymph Ion Concentrations.

the winter until April, 1978. Average Na^+ values increased in April, 1978 to 110 mM and then steadily declined with stratification. A return to higher Na^+ concentrations was observed from October through December.

The hemolymph K^+ concentration in Chironomus was maintained in the range of 3.0 - 9.0 mM from April, 1977 through December, 1978. Chloride concentrations remained fairly constant at an average value of 54 mM with lower values observed in those months of 1977 with lower Na^+ values. Chloride concentrations exhibited a gradual decline throughout the 1978 sampling period. Average chloride values for March, July and November, 1978 were 41.0, 38.0, and 56.0 mM repectively.

Sodium was present only in the lake water at Station 2 in April, 1977 (Figure 15). In April, 1978, Na^+ was present at al stations with an average concentration of 45 mM. Potassium and Cl^- concentrations were relatively non-detectable in the water.

In both species, the fluctuations in hemolymph osmotic pressure corresponded to the changes observed in the hemolymph Na^+ levels. Chaoborus maintained an average osmotic pressure of .430 -Δ°C from October, 1977 through April, 1978 (Fig. 16). Chironomus maintained a lower average osmotic pressure of .250 -Δ°C for this same period. A gradual decline in the osmotic pressure in both species accompanied the declining Na^+ values from .460, .300 and .440 -Δ°C in March, August, and November, 1978 respectively.

Due to the very dilute concentrations of Na^+ and Cl^- in the water, the lake water osmotic pressure values were generally in the range of .014 - 0.17 -Δ°C throughout the sampling period (Figure 16).

Differences were observed between the shallow and deeper stations in concentration values for the hemolymph ions and its osmotic pressure for both species. Higher values or concentrations of the ion(s) in the organisms were generally observed at Stations 1 and 2.

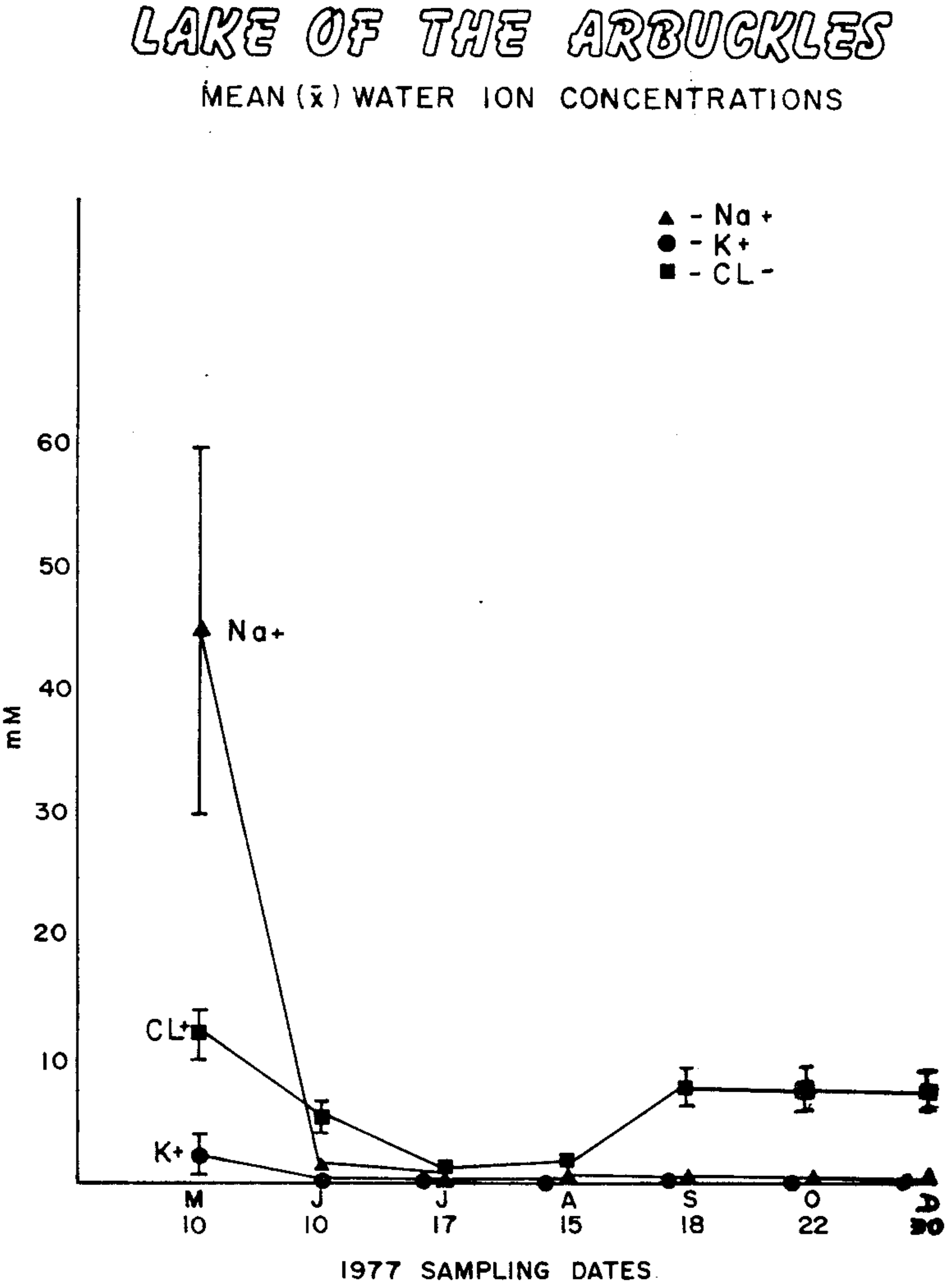

Figure 15. Mean Water Ion Concentrations.

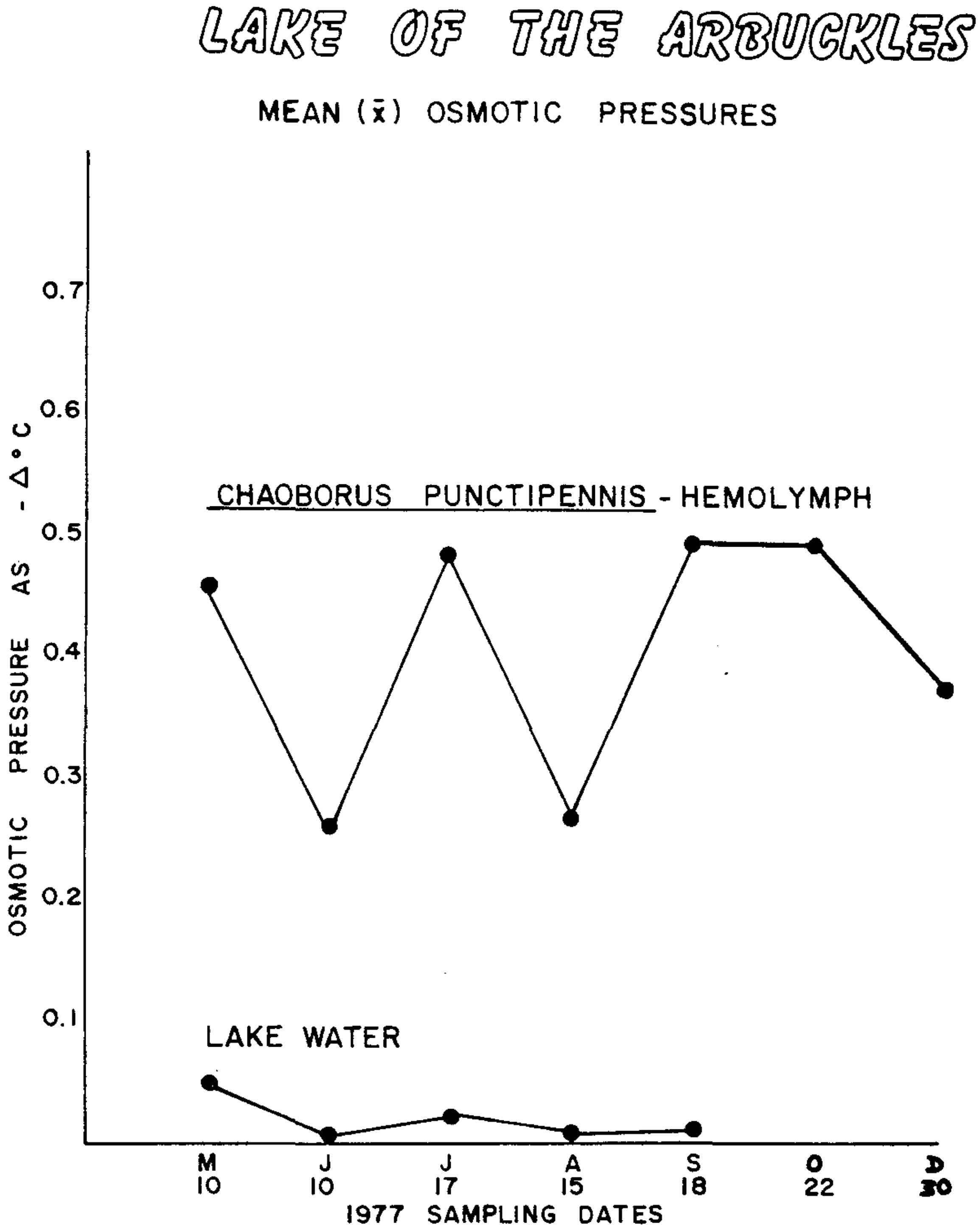

Figure 16. Mean Osmotic Pressures.

Oxygen Consumption

Oxygen consumption by the larva Chaoborus punctipennis was fairly uniform throughout the lake in March, 1978 (Table 15). Consumption uptake dropped by the organism to 1.1 µl O_2/mg d wt/h at the deeper stations (1 and 2) while increasing at the 15 m stations (3, 4 and 5) in April. During the summer anoxic months, a reversal of the spring uptake rates was observed in the organism in relation to depth. Uptake values remained relatively high at the deeper central pool stations from August through December with a mean consumption rate of 8.4 µl O_2/mg d wt/h. During this same period, from August through October, uptake rates were on the average lower for all the 15 m stations with a mean value of 2.7 µl O_2/mg d wt/h. Consumption rates did increase at the 15 m stations in December and their consumption values were comparable with those recorded at Stations 1 and 2.

Oxygen consumption of the sediment column was also measured at intervals of 1, 5 and 10 cm. At Stations 1 and 2, consumption values were lower uniformly in the sediment column from March through June (Table 16). An average sediment column uptake value of 7.1 µl O_2/g d wt/h was recorded for this period. Sediment consumption values remained relatively high for the rest of the sampling period at Stations 1 and 2. The highest oxygen uptake occurred in the 1 cm layer of the sediment column of which 15% was due to chemical uptake.

Sediment uptake values progressively declined from March through November at all the 15 m stations (Table 16). Consumption rates recorded during the oxic spring months had an average value of .48 µl O_2/g d wt/h. A progressive increase in consumption values was recorded during the fall sampling period.

TABLE 15

MEAN* ($\bar{x}$) OXYGEN CONSUMPTION RATES (μl O_2/mg d wt/h) OF *CHAOBORUS PUNCTIPENNIS* IN 1978

Station (Depth in m)	MAR. 15	APR. 29	JULY 11	AUG. 15	SEP. 16	OCT. 22	DEC. 16
1(24)		1.1		9.4	4.0	9.2	9.3
2(20)	2.6	1.2		4.3	7.8	12.6	10.8
3(15)	1.2	8.1		1.9	1.4	4.9	9.1
4(15)	1.2	8.2	3.4	1.7	1.3	5.3	11.1
5(15)	.98		8.0	2.3	2.6	2.8	7.1

* Mean value obtained obtained from five replicates

TABLE 16

MEAN ($\bar{x}$)* SEDIMENT OXYGEN CONSUMPTION RATES (µl O_2/g d wt/h) in 1978

Station (Depth in m)	Core Depth (cm)	MAR. 15	APR. 29	JUNE 9	JULY 11	AUG. 15	SEP. 16	OCT. 22	NOV. 24	DEC. 16
1(24)	1	.44	.76	.77	1.36	.89	.82	.73	.84	.95
	5	.51	.77	.72	1.03	1.18	.86	.57	.74	.61
	10			.75	1.20	.89	.98	.63	.70	.90
2(20)	1	.45	.62	.81	.99	.97	.77	.90	.80	1.07
	5	.53	.82	.86	.54	1.03	.87	1.10	.65	.64
	10			1.20	1.09	.43	.59	.94	.63	.63
3(15)	1		.37	.70	.46	.25	.07	.44	.49	.70
	5		.80	.64	.47	.15	.11	.45	.45	.70
	10			.66	.40	.13	.09	.27	.48	.58
4(15)	1		.38	.62	.32	.38	.10	.61	.70	.41
	5		.58	.75	.28	.12	.09	.42	.69	.34
	10			.65	.22	.11	.09	.70	.62	
5(15)	1	.49	.41	.40	.19	.17	.09	.36	.44	.83
	5	.62	.65	.40	.16	.13	.07	.21	.44	.85
	10			.46	.15	.07	.07	.35	.37	.76

* Mean value obtained from three replicates.

Discussion

The temperature and dissolved oxygen data suggested that the lake stratified in early May in the central pool and was not completely mixed until October. Stratification developed later and complete mixing earlier in the arms. Conductivity was greatest in summer suggesting the build-up of inorganic ions in the anoxic hypolimnion. Conductivity tended to decrease with depth; however, the values were not significantly different. Alkalinity and pH did not exhibit significant trends or fluctuations. Examination of the vertical variation of physicochemical conditions indicated that the mechanical pumping did not improve water quality of the water 1 m above the bottom at either the 15 m stations in the arms or in the central pool.

Particle size decreased with water depth in both lakes. A longitudinal gradation resulted as sediments were carried by currents from shallow to deep water. Larger particles settle first and smaller particles remain in suspension until reaching the quieter, deeper waters. Wilson (1936) found that particulate materials migrated towards the deepest part of the lake and lodged permanently in the depressions in Winona Lake, Wisconsin. Wind-induced currents and wave action are the principal mechanisms for transporting sediments from littoral waters to deeper waters where currents are reduced (Scott & Mainter 1936). Frink (1969) established good correlations between particle size and water depth. Currents and wave action were sufficient to carry the fine material into deeper water without the drastic mixing occurring in more open bodies of water. In the present study, particle size tended to decrease over the summer at the deeper stations in both lakes. During thermal stratification, strong wind-induced currents are confined to epilimnetic waters (McKee et al. 1970) and small particles may settle in the less turbulent areas. The lack of a seasonal change in particle size at the shallow stations may be attributed to the proximity to the terrestrial environment and the greater susceptibility to natural and man-made turbulence.

Organic matter did not vary significantly among stations and seasonal variation was slight. Kemp (1971) found that the quantity of organic matter in the top cm of sediments in the Great Lakes was directly proportional to the clay size fraction of sediment and independent of depth, rate of sedimentation, and degree of eutrophication. A linear relation between organic matter and sediment particle size has been observed by many workers (Trask 1932, Bordovisky 1965, Thomas 1969, and Longbottom 1970). No correlation existed between organic matter, depth, and particle size in the present study. Variation in particle size among stations may not be sufficient to influence significantly variation in the amounts of organic matter which accumulate. The actual amount of organic material reaching the sediments may be small. Kajak et al. (1970) found that approximately one-half of the amount of organic carbon reaching the hypolimnion is oxidized before it reaches the sediments. At shallower stations decomposing macrophytes and leaf litter may contribute significant quantities of organic matter.

Percent loss on ignition values as determined gravimetrically were generally greater than percent organic carbon as determined by oxidation. It is likely that the oxidation procedure is biased in lake sediments with a high clay content. Inorganic phosphorus in the sediments was generally lower in summer than in spring and winter. Although summer stratification and the concurrent breakdown of the microzone at the sediment surface are related to phosphorus release by the sediment (Mortimer 1941), phosphorus may be preferentially associated with the fine particles characteristic of deeper waters (Seyers et al. 1973). Particle size decreased during summer at the deeper stations in the present study. Phosphorus in the sediments was generally greater at the deeper stations than at the shallower stations. Inorganic sediment phosphorus generally increases with increasing depth of the overlying water column (Delfino et al. 1969, Williams et al. 1970).

Minerals comprising the sediment in both lakes were quartz, kaolinite, mica, montmorillonite, and a montmorillonite-vermiculite mixture. Kaolinite was the dominant clay particle. Kaolinite has a relative low surface area and a low capacity to adsorb cations. The specific surface area of montmorillonite greatly exceeds the total area of kaolinite and has a high cation adsorption capacity, perhaps 10-15 times that of kaolinite (Brady 1974).

Factors which may influence sedimentation rate include rate of stream flow; watershed size and use; topographic and soil characterictics; vegetational cover; and the kind, amount, and distribution of precipitation (McKee et al. 1970, Oschwald 1972). In Arbuckle Lake minimum sedimentation occurred in the central pool. The inverse relation of depth to sedimentation rate may be explained by size sorting of particles. Smaller, less dense particles which are carried to deeper waters tend to remain in suspension, while heavier particles settle rapidly upon entering the shallower waters (Wilson 1936).

Iron did not show variation with depth from the shallow to the deep water stations. Most of the iron present in the water column was particulate, probably present as ferric hydroxide. Concentrations of iron in the water did not accompany changes in stratification.

Sorbed iron concentrations in the sediment were very small. The concentrations fluctuated throughout the summer, not responding to changes in stratification. Most of the iron present in the sediment was iron sulfide, an insoluble black precipitate.

Total manganese concentrations in the water were larger at the deep water stations. Most of the manganese present was in the dissolved from. This follows the expected trend of manganese solubilizing and leaving the sediments when the redox potential of hypolimnial waters is reduced (Mortimer 1941, Wetzel 1975).

A definite increase in the concentration of sorbed manganese of the sediment was observed as stratification progressed throughout the summer.

Low concentrations of copper and zinc were observed in the hypolimnial waters. Concentrations of these ions fluctuated throughout the sampling period showing no significant seasonal trends. Most of these metals were in dissolved form. Sorbed concentrations of these metals in sediment were also very low, with no apparent seasonal changes accompanying stratification.

Sediment calorie content did exhibit some seasonal trends at Stations 1, 2 and 5. Sediment caloric content decreased significantly throughout the summer at these stations. This decrease could be due to the increased respiration rate of sediment microbial populations. Increasing water temperatures throughout the summer can also increase metabolic rates. Colder temperatures will have an inhibitory effect on respiration thus allowing organic material to increase.

Consistently higher concentrations of hemolymph ions were observed at the deep water stations throughout the sampling periods in both species. It appears that the interaction at a certain temperature and DO level will cause the organisms to maintain high concentrations of hemolymph ions. When subjected to warmer temperatures and constant DO levels of approximately 1-2 ppm, the organisms are able to maintain ion concentrations at a lower level for a longer period of time. This was observed at Stations 3, 4 and 5.

A similar condition was observed in Ham's Lake during 1978. Chaoborus and Chironomus riparius were collected from a very anoxic, warm station (#4) and from two destratified cooler, oxic stations (#5 and 6). A linear increase in concentration of ions was observed in both species at station 4 as stratification progressed. At the destratified stations, ion concentrations increased through July but were maintained at lower levels through December, 1978.

Wright (1975) observed the hemolymph Na^+ level in *C. tentans* from 2-14°C to be constant at 102 mM. A slight decrease was observed from 21-24°C to 97 mM with a subsequent increase in Na^+ around 26°C. Hemolymph Na^+ in *C. tentans* usually has a range of 92-120 mM (Boné 1945, Firling 1970, Wright 1975).

The significant decrease in concentrations of ions observed in June and August of 1977 may be due to physiological changes associated with molting (Firling 1970, Schin and Moore 1977).

Oxygen consumption at Stations 1 and 2 increased in the sediment column progressively for the entire sampling period. Similar increases in sediment consumption under anoxic conditions have been observed by Brewer et al. (1977). These maximum uptake trends observed at the central pool stations during the anoxic period might be explained best through the effects of temperature and dissolved oxygen on first-order enzyme saturation kinetics. Further studies need to be conducted into the kinds of metabolic substrate(s) available to substantiate any correlations with temperature and DO variation.

Sediment consumption at Stations 3, 4 and 5 which were oxic for the entire sampling period exhibited significant decreases in their consumption values. The trends observed at these stations may be related to the particle size and/or organic matter present in the sediment. Hargrave (1972) related oxygen consumption directly to the amount of organic matter present. Consumption was also found to be inversely related to particle size. The larger the particle size, the lower the consumption value will be due to a smaller quantity of organic matter present in sand. Mechanical properties such as the diffusion rate of oxygen in sand must be greater than that in mud and must also be considered as a factor influencing the respiration rate at different temperatures (Hargrave 1969).

Two trends are therefore evident from the sediment consumption studies. Oxygen consumption increased in the sediment column as the water temperature increased and the oxygen concentration decreased to 0.0 ppm; and an inverse consumption rate was also observed with increasing temperatures and DO levels of not less than 2.0 ppm.

A 10% formalin solution did inhibit the biological uptake activity in the sediment column. Chemical uptake accounted for only 15% of the respiration rate at Stations 1 and 2 during the anoxic period. No chemical uptake was observed at the 15 m stations. Difficulties also arise in interpreting the consumption values due to the difficulty in measuring anaerobic benthic metabolism and the diversity of metabolic types of heterotrophic bacteria.

The sediment study does present evidence that oxygen removal from the water by the sediments is essentially a biological process. This biological process is probably mediated by the standing crop of aerobic, heterotrophic bacteria. Significant differences in consumption rates do exist between the central pool and the 15 m stations.

In general, most aquatic insects show an oxygen consumption that is independent of the oxygen content of the surrounding media.

The consumption rates were highest in Chaoborus collected from Stations 1 and 2 throughout the sampling period. Slower oxygen consumption rates were recorded consistently from March through December at the 15 m stations. From these results and controlled laboratory experiments, it appears that oxygen uptake by Chaoborus is more dependent upon prolonged exposure to different temperatures than to the oxygen concentration.

Conclusions

1. Examination of the vertical variation of physicochemical conditions indicated that the mechanical pumping did not improve water quality of the water 1 m above the bottom at either the 15 m stations in the arms or in the central pool.

2. Particle size tended to decrease over the summer at the deeper stations in the lake. Particle size at the shallow stations did not exhibit seasonal changes.

3. Organic matter did not vary significantly among stations and seasonal variation was light.

4. Inorganic phosphorus in the sediments was generally lower in summer than in spring and winter.

5. Kaolinite is the dominant mineral in the sedimant.

6. Sedimentation rates where highest at the shallow water stations.

7. Low concentrations of copper and zinc were observed in the hypolimnial water and sediment. No significant seasonal trends were observed.

8. Total manganese concentrations in the water were larger at the central pool stations. Sediment sorbed manganese concentrations increased with stratification.

9. Iron concentrations of the water and sediment fluctuated throughout the summer, not responding to changes in stratification.

10. Sediment calorie content decreased significantly throughout the summer at Stations 1, 2 and 5.

11. Higher concentrations of hemolymph ions were observed at the deep water stations in Chironomus and Chaoborus. Seasonal fluctuations were observed.

12. Seasonal changes in oxygen consumption rates in Chironomus and Chaoborus were observed at all stations. Consumption rates were highest from those collected in the central pool.

13. Oxygen consumption at Stations 1 and 2 increased in the sediment column during stratification. Sediment consumption rates at Stations 3, 4, and 5 decreased with stratification. The oxygen removal from the water by the sediments is essentially a biological process probably mediated by aerobic, heterotrophic bacteria.

SUMMARY

This research was designed to test the feasibility of destratifying and reaerating eutrophic reservoirs using large diameter axial flow pumps with modest energy consumption. Additionally, it was desired to quantify the physical laws involved in the process and to determine physicochemical and biological response of destratification and reaeration.

The specific objectives were: 1) to determine the relationship between pumping rate and destratification time of a reservoir, 2) to determine the relationship between temperature stability and oxygen concentration in a reservoir, 3) to test the hypothesis that lakes can be destratified to a preselected depth using the Garton pump, and 4) to determine certain biological and physicochemical effects of lake destratification.

The lake selected as the intensive research site was Lake Arbuckle in south central Oklahoma. Lake Arbuckle has a surface area of approximately 951 ha (2350 acre), a maximum depth of 27.4 m (90 ft.) mean depth of 9.4 m (31 ft.), and a volume of 89300 megaliters (72,425 acre ft.). Results of previous closely related research at Arbuckle is contained in OWRT technical completion reports C-5228.

A cluster of sixteen axial flow pumps were arranged in a hollow square and anchored near the dam. Each pump was 1.829 m (6 ft.) in diameter, was powered by an electric motor of 1.12 kW (1.5 hp), and driven at about 17.35 rpm through a right angle reduction gear. The combined pumped volume of the cluster was estimated to be 26.4 m^3/sec (416,000 gpm) with initial downard velocity of 0.648 m/sec (2.1 ft./sec). Total energy input into the cluster was approximately 17.1 kW (12.8 hp).

The low initial velocity was not sufficient to penetrate to the bottom of Lake Arbuckle when stratified. The downward flow of surface water was limited by thermal-density layers in the lake. During pump operation, the lake was destratified to the depth to which the pump plume was penetrating. This depth gradually lowered as pumping continued. The pumping rate was not sufficient to effectively destratify Lake Arbuckle. However the overall water quality improved as the epilimnion was increased due to the pumping activity.

Effects of the destratification effort upon temperature and DO profiles were monitored. Because the lake was not destratified to 15 m until late in the summer, studies of the water near the bottom, the sediments, and physiological changes in benthic invertebrates could only be correlated to seasonal changes in temperature and DO.

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