IMPACT OF PHOSPHORUS REDUCTION VIA METALIMNETIC ALUM INJECTION IN BULLHEAD LAKE, WISCONSIN

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Frontispiece: *Looking southwest over Bullhead Lake and a portion of its 256 ha (1 mile^2) drainage basin.*

ABSTRACT

Bullhead Lake, a 27 ha (67 acre) hard water dimictic lake with a maximum depth of 10.5 m (35 ft.), was treated with aluminum sulfate (alum) via metalimnetic injection on 23 August 1978. The treatment was designed to evaluate the use of metalimnetic alum injection for phosphorus control and to determine the effect of alum on plankton and benthos. The project began in June 1975 and continued through December 1982. Nutrient content, planktonic biota, and benthic community composition and response were determined.

The alum significantly reduced internal phosphorus recycling. Summer epilimnetic total phosphorus (TP) was reduced from a 1978 pretreatment mean concentration of 41 *ug* TP/L to 16 *ug* TP/L in 1979. Mean soluble reactive phosphorus (SRP) levels were reduced from 8 *ug* to <4 *ug* SRP/L in the same period. Post-treatment phosphorus levels remained low to the end of the study period.

The reduction in phosphorus had a direct effect on the biota. Green algae, flagellates, and diatoms increased and blue-green algae decreased. Generally, the zooplankton increased in density and diversity; the rotifers increased dramatically. The benthic community also increased in numbers and taxa. The increase in the biotic community indicates that the aluminum hydroxide flocculent had little or no toxic effect upon these organisms.

Metalimnetic injection is a practical application method. Injecting the alum below the epilimnion reduces possible toxic effects and places the alum on the anoxic sediment which is the primary source of recycled phosphorus.

KEY WORDS: Aluminum Sulfate, Benthos, Internal Nutrient Recycling, Metalimnetic Injection, Nutrients, Phosphorus, Total Phosphorus (TP), Soluble Reactive Phosphorus (SRP).

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INTRODUCTION

Phosphorus is often the major cause of excessive algae growth (Hutchinson 1957, Wetzel 1983), and phosphorus management is critical to lake rehabilitation schemes (Vollenweider 1968). In stratified lakes where external nutrient sources are limited and algae is the dominant primary producer, internal phosphorus reduction by chemical inactivation is most useful.

One way of inactivating the internal phosphorus cycle is the addition of an aluminum salt. Aluminum sulfate (alum) acts as a substitute for iron controlling cycles in productive calcareous lakes (Stauffer 1981). Anaerobic sediments in eutrophic lakes during summer stratification have reduced ferrous iron which is soluble and releases phosphate. Therefore, phosphate is allowed to diffuse to the overlying water (Gorham 1958, Holdren 1977, Kamp-Nielson 1974, Mortimer 1941 and 1942). In addition, sulfate concentrations are relatively high in this region of Wisconsin. Reduced forms of sulfur additionally remove iron from the lake system by forming stable and insoluble iron sulfide.

Alum, mixed with water, produces aluminum oxides. Soluble reactive phosphorus (SRP) adsorbs to the aluminum oxide floc due to the strong affinity of the Al^{+3} ion for both OH^- and PO_4^{-3} radicals. As a result the aluminum hydroxide flocculent in the lake sediment immobilizes the release of SRP. A mixture of aluminum sulfate and sodium aluminate is used in poorly buffered lakes.

Alum was initially used at Lake Langsjon, Sweden, and it effectively reduced phosphorus concentrations (Jernelov 1970). The first application in North America was made at Horseshoe Lake, Wisconsin, in 1970 and phosphorus levels were substantially reduced for several years (Peterson et al. 1973). Since the Horseshoe Lake treatment, 11 lakes in the United States have been treated with aluminum salts (Cooke and Kennedy 1978, Dominie 1980, Dunst et al. 1974, Funk et al. 1980, Garrison and Knauer 1983, Gasperino et al. 1980, Smith 1984, pers. comm.).

Bullhead Lake provided an excellent site to study injection of alum since it is a moderate sized stratified lake with abundant algae growth and minimal external nutrient impact. The objectives of the Bullhead Lake project were to: 1) determine the effect of metalimnetic alum injection on the phosphorus cycle, 2) determine the effect of alum on the lake's plankton and benthos, and 3) determine the ability of the aluminum floc to inactivate phosphorus and restrict phosphorus transport from the hypolimnetic sediments.

STUDY SITE

Bullhead Lake is a 27 ha (67 acre) hard water dimictic lake with a maximum depth of 10.5 m. It is a kettle lake of glacial origin located on the western edge of Manitowoc County (Table 1, Fig. 1). There is a well-defined summer stratification period and a strong internal nutrient recycling mechanism.

The lake's drainage area is small, consisting of only 256 ha (1 mile^2). There is no channelized inlet or outlet. Water level is apparently maintained by direct precipitation and some surface runoff. The clay subsoil reduces permeability and is important in minimizing subsurface water transport. The southeast shoreline is sandy which could al-

TABLE 1. *Bullhead Lake morphology.*

Area:	
Total (A)*	27 ha
Top of metalimnion (4.5 m)	5.6 ha (21% of total)
Maximum length (l)	765 m
Maximum width (b)	483 m
Shoreline	1,980 m
Shoreline development (D_L)	1.08
Volume:	
Total (V)	10.67×10^5 m^3
4.5-10.5 m depths	1.79×10^5 m^3 (17% of total)
Depth:	
Maximum (Zm)	10.5 m
Mean ($\bar{z}$ = V/A)	4.0 m

* Wetzel (1983).

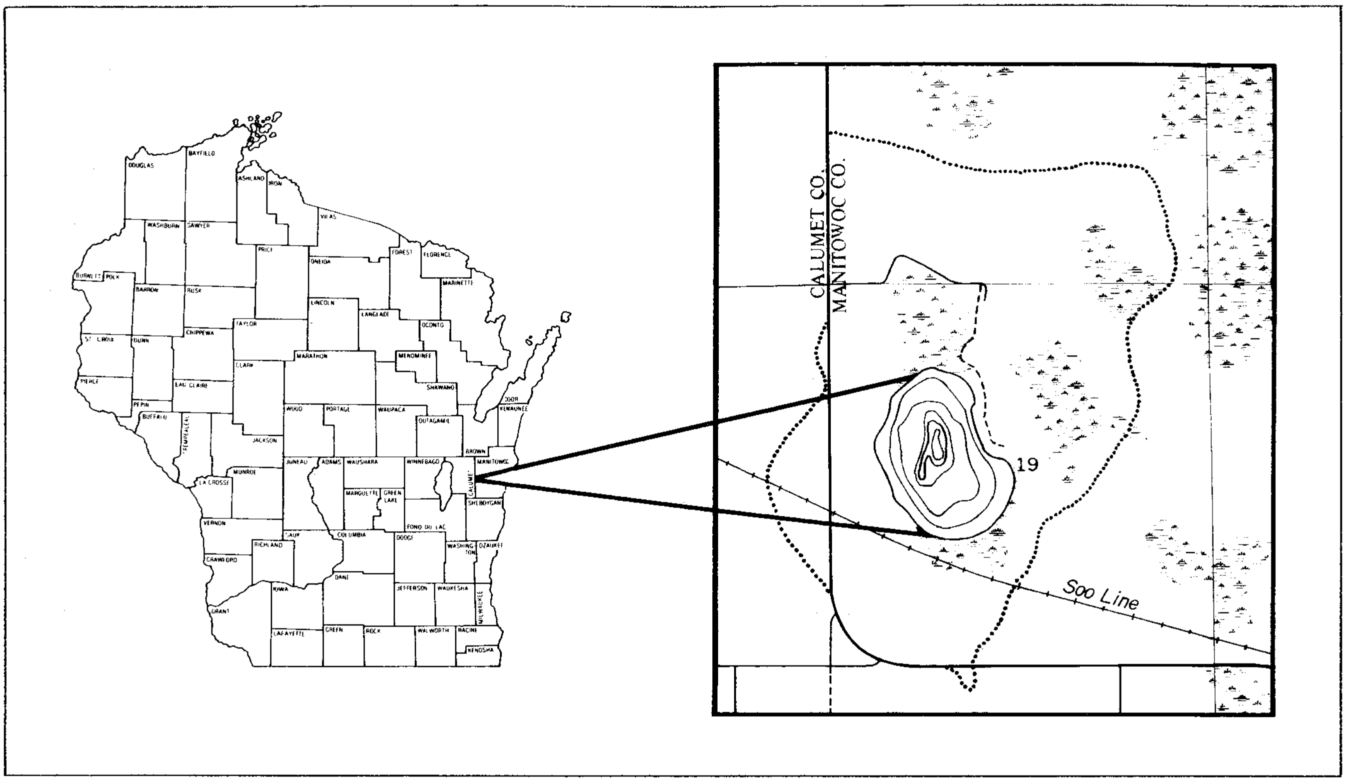


FIGURE 1. *Local topography and drainage basin outline of Bullhead Lake.*

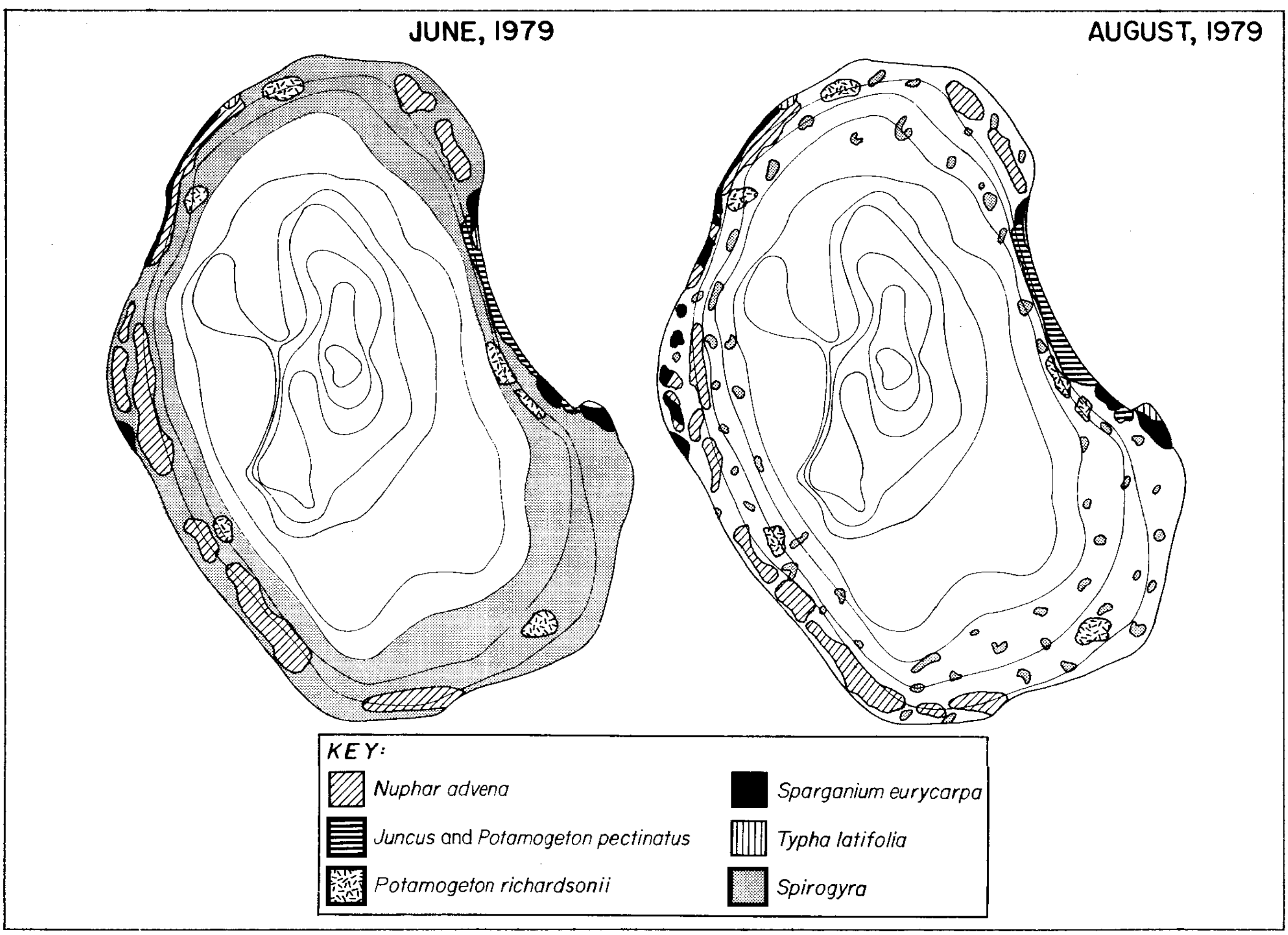


FIGURE 2. *Macrophyte distribution during the 1979 summer season.*

low water transfer between the lake and adjoining wetlands. Nearly all runoff filters through the wooded wetlands adjoining the lake. Direct surface runoff occurs only after long periods of heavy rain.

Submerged macrophytes are generally lacking. Where present they consist of scattered patches located in the littoral zone to a depth of 2 m (Fig. 2). The major macrophytes, in order of abundance, were: yellow water lily (*Nuphar advena* Ait.), Richardson pondweed (*Potamogeton richardsonii* (Benn.) Rydb.), bulrush (*Juncus* sp.), and burreed (*Sparganium eurycarpum* Engelm.).

Filamentous algae were routinely found in the littoral zone to a depth of 2-3 m. These algae usually remained from early spring through June, leaving small scattered patches during late summer and fall. The heavy growth often covered the macrophytes.

Taste and odor problems in fish were often mentioned by fishermen during the 1970s. At present the lake contains walleye (*Stizostedion vitreum vitreum* (Mitchill), largemouth bass (*Micropterus salmoides* Lacepède), and panfish.

METHODS

WATER CHEMISTRY

A sampling station was established and marked with a buoy at maximum depth (10.5 m) near the lake center (Fig. 3). Samples were taken with a clear lucite 2.2 L vertical Kemmerer water sampler at 1-m intervals starting at 0.5 m. An electronic temperature and BOD probe (Weston-Stack Model 330), calibrated in the field with a standardized thermometer and azide modified Winkler titration, was used for dissolved oxygen (DO) and temperature measurements prior to 1981. From 1981 to 1982, the BOD probe was discontinued, and the azide modification of the Winkler method and standardized thermometer were used. Water clarity was determined with a 20-cm Secchi disk.

Water samples were analyzed for phosphorus, nitrogen, pH, alkalinity, and chlorophyll-*a*. Methods of analysis (Table 2) were those of the American Public Health Association (1975), Eisenreich et al. (1975), Jirka et al. (1976), Lind (1974), Strickland and Parson (1968), U.S. Environmental Protection Agency (USEPA) (1974, 1979), and U.S. Geological Survey (1970).

BIOLOGICAL SAMPLING

Biological samples were taken to determine species and density of phytoplankton, zooplankton, and benthos. Composite epilimnetic plankton samples were obtained with a Kemmerer sampler from depths of 0.5 and 3.5 m. The phytoplankton samples were preserved with acidified Lugol's solution and the zooplankton samples with formaldehyde (USEPA 1973). Species composition of the phytoplankton was determined by the Utermohl technique (Lund et al. 1958) using an inverted microscope. Taxonomic keys for phytoplankton were from Patrick and Reimer (1966 and 1975), Prescott (1962), Skuja (1948), Smith (1920, 1924, and 1950), and Weber (1971). Biovolume was determined by estimating the various geometric forms of the individual plankton and applying the cell counts.

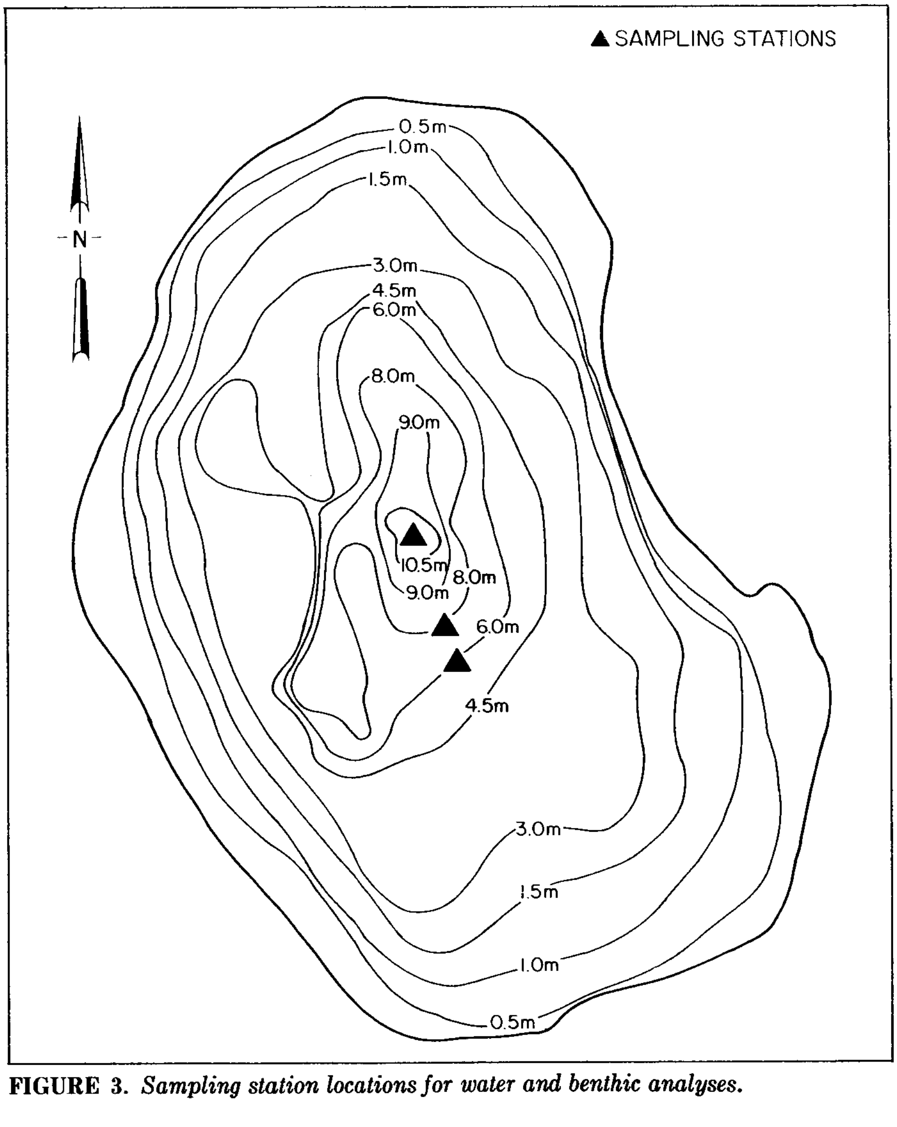


FIGURE 3. *Sampling station locations for water and benthic analyses.*

Zooplankton were counted in a Sedgewick-Rafter cell using a compound microscope (USEPA 1973). The zooplankton were identified using the keys of Brandlova et al. (1972), Brooks (1957), Chengalath et al. (1971), Deevy and Deevy (1971), Edmondson (1959), Smith and Fernando (1978), and Torke (1976).

Chlorophyll-*a* samples were taken at the same time and depths as the water chemistry samples. The water was filtered through a 0.45 *u* membrane filter (Gelman GA-6 Metricel) and extracted with 90% acetone. The filters were slurried with a tissue grinder and stored at -18 C for a minimum of 48 hours. The samples were centrifuged at 1000 g for at least 30 minutes. Absorption was measured on a Bausch and Lomb (B&L) Spectronic 20 before 1977 and thereafter on a B&L Spectronic 70. Chlorophyll-*a* was calculated by the trichromatic formula of USEPA (1973).

Benthic sampling began in 1977 at the 10.5-m buoy. Additional sampling locations at the 8 and 6 m contours south-southeast of the 10.5-m site were established in 1978 (Fig. 3). A 0.2 m^2 Eckman dredge was used to obtain four samples per location (Lind 1974, USEPA 1973). Each sample was sieved through a 188 x 412 *u* screen and preserved with 95% ethyl alcohol. They were then diluted with 70% ethanol to 400 ml and randomly mixed. Ten 2-ml subsamples were taken for determination of taxa smaller than 5 mm. The balance of each sample was counted to determine all other benthos greater than 5 mm. Any benthos larger than 5 mm found in the subsample was included in the total.

SEDIMENT ANALYSIS

Pretreatment sediment cores were taken near the 10.5-m buoy in February and August 1978 to determine the distribution of aluminum and phosphorus. Post-alum treatment cores were taken in August 1978 and May 1979 at the 10.5-, 8-, and 6-m buoys. The apparatus consisted of an 8.7 cm I.D. lucite piston corer fitted to a series of pushrods. The cores were field sectioned into 2.5 cm slices except the mud-water interface, which was 5 cm thick. The cores were processed by the University of Wisconsin Soils and Plant Analysis Laboratory for analysis by mass spectrophotometer at the Wisconsin Alumni Research Foundation.

ALUM APPLICATION

Jar tests with various concentrations of aluminum were performed using Bullhead Lake hypolimnetic water. Copious amounts of floc resulted when more than 7 mg Al^{+3}/Lwas used. Treatment level was established at an average concentration of 13 mg Al^{+3}/L of hypolimnetic water in order to provide a thicker flocculent layer and increase the project's longevity. The material selected for use was commercial aqueous aluminum sulfate.

The treatment zone was divided according to hypolimnetic volume and marked with buoys (Fig. 4). Centrifugal pumps mixed some lake water with the alum and force-fed the mixture to the distribution manifold (Figs. 5, 6, 7). A three man crew tended each barge and controlled the injection manifold depth of 5.0-5.5 m, water pump, water alum mixture, and barge speed and direction. The 32,500 L (approx. 42,500 kg) of commercial grade alum was applied in 5 hours on 23 August 1978. The water volume treated was 1.79×10^5 m^3 or 17% of the total lake volume (Table 3). The cost of alum delivered to the lake was \$2,630 or about \$0.08/L.

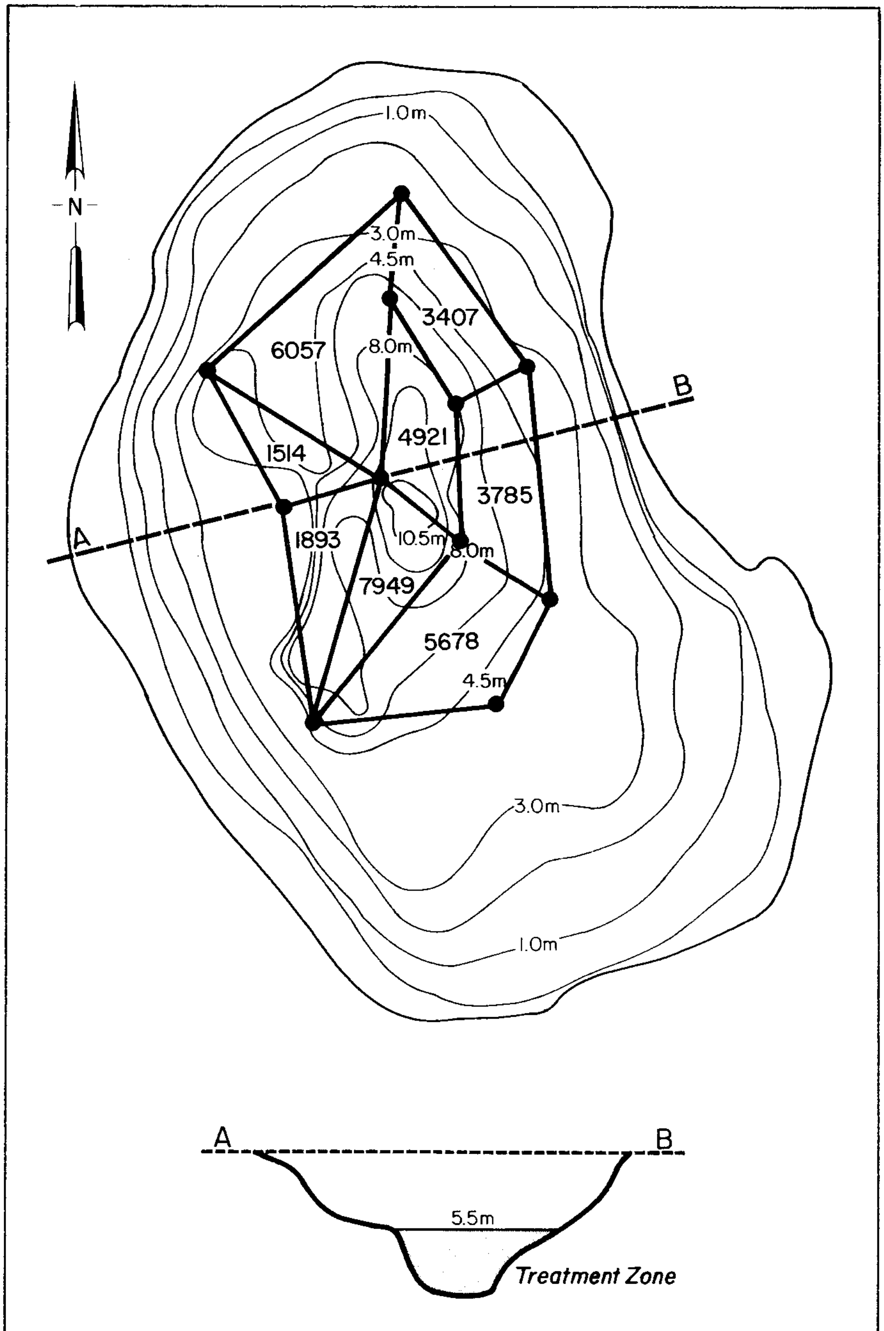


FIGURE 4. *Area of Bullhead Lake injected with alum, showing buoy placement and alum liters allocated to each treatment zone.*

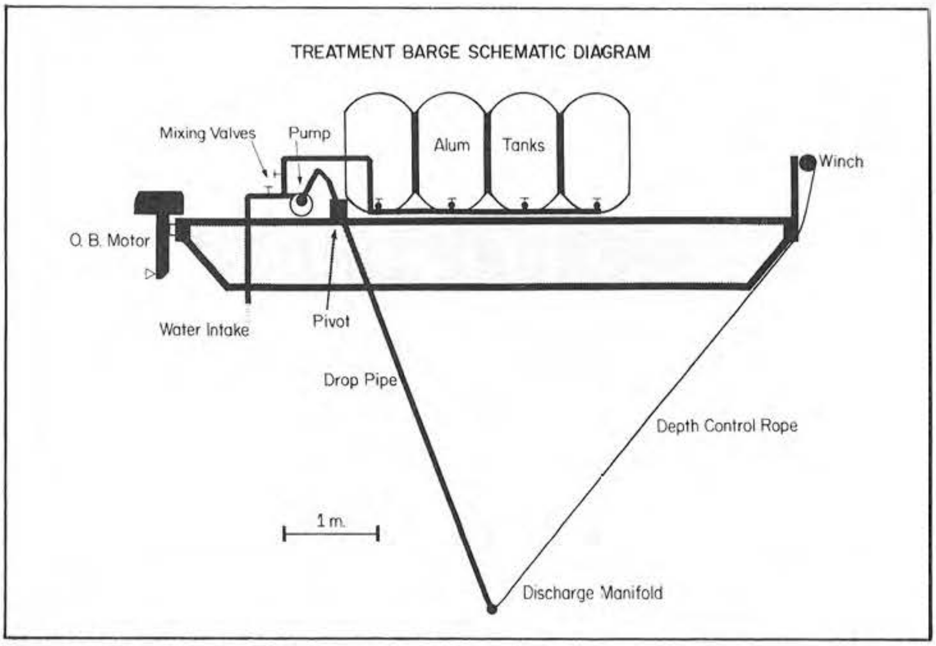


FIGURE 5. *Diagram of treatment barge.*

FIGURE 6. ***Loading of treatment barge with alum from tanker truck. Valve on the hose end facilitates flow control.***

FIGURE 7. ***Treatment barge showing injection manifold in raised position.***

TABLE 2. *Analytical methods used in the Bullhead Lake sample program.*

Parameter	Method	Reference* For Years: 1975-79	1980-82
Field			
Temperature	Electric thermister	6	6
	Standardized thermometer		6
Dissolved oxygen	Weston-Stack electronic BOD probe		
	Azide modified winkler		1
Laboratory			
pH	Glass electrode	6	6
Alkalinity	Acid titration and pH meter	6	6
Total phosphorus	Molybdate colorimetry (unfiltered)	2	4
Soluble reactive phosphorus	Molybdate colorimetry (filtered, 0.45*u*)	2	4
Total nitrogen	Summation	3	5
Ammonia nitrogen	Distilled-nesslerization	3	4
Nitrate nitrogen	Brucine sulfate	3	
	Cadmium reduction		4
Nitrite nitrogen	N-(1-naphthyl)-ethyl-enediamine	3	
	Cadmium reduction		4
Organic nitrogen	Digestion, distillation, and nesslerization	3	
	$TKN\text{-}NH_3N$		5
Chlorophyll-*a*	Filtration and extraction	6	6

* References: 1-American Public Health Association (1975)
2-Eisenreich et al. (1975)
3-USEPA (1974) 625/6-74-003
4-USEPA (1979) 600/4-79-020
5-Jirka et al. (1974)
6-Strickland and Parson (1968)

TABLE 3. *Water volume and water depth relationships in Bullhead Lake.*

Depth (m)		Water Volume
0.0-1.0		$2.47 x 10^5$ m^3
1.0-2.0		2.04
2.0-3.0		1.60
3.0-4.0		1.48
4.0-5.0		1.30
	treatment	$8.89 x 10^5$ m^3
	depth	
5.0-6.0		0.62
6.0-7.0		0.41
7.0-8.0		0.33
8.0-9.0		0.22
9.0-10.0		0.15
10.0-10.5		0.06
		$1.79 x 10^5$ m^3
	Total	$10.68 x 10^5$ m^3

RESULTS AND DISCUSSION

TRANSPARENCY

The Secchi disk (SD) transparency values prior to lake treatment fluctuated seasonally (Fig. 8). Water clarity during the spring was typically greater than during summer stratification, and fall transparency decreased with the density of fall algae blooms. Following treatment, these transparency fluctuations were reduced. The spring diatom bloom increased and, therefore, the transparency was decreased. During the fall the blue-green algal bloom declined thereby increasing fall transparency.

Summer algae peaks and crashes caused great variability in Secchi disk measurements prior to 1979. These fluctuations typified a population dominated by one or two species of blue-green algae, many of which reduced transparency by cell clumping and increases in chlorophyll abundance. The common algae in the summers prior to treatment were the blue-greens *Lyngbya Birgei, Aphanizomenon flos-aquae,* and *Anabaena* spp. in conjunction with *Fragilaria crotonensis* and *Ceratium hirundinella.*

After the alum treatment, blue-green algae decreased and green algae and flagellates increased. The increased diversity of algae reduced the effect of a single blue-green species peaking and crashing during the growing season. Except for the increased transparency in 1979 — a period of algal adjustment to new phosphorus concentrations — the large Secchi disk fluctuations commonly observed before alum addition were decreased (Fig. 8).

DISSOLVED OXYGEN

Bullhead Lake's hypolimnion typically became anoxic from mid-May until fall overturn which occurred in early October. The summer dissolved oxygen (DO) was indirectly affected by the alum treatment (Fig. 9). The volume of water containing more than the critical minimum (5 mg DO/L) for fish survival increased slightly after treatment. The post-treatment summer mean depth of the 5 mg/L concentration was 4.6 m, a 0.5-m increase in depth from the pretreatment mean of 4.1 m. The 0 mg/L DO concentration isopleth typically occurred 1 m deeper than the 5 mg/L isopleth.

Winter DO improved for the first two years after treatment. During the winters of 1979-80 and 1980-81, the 5 mg/L concentration reached a depth of 8 m during ice cover; in similar periods of other years, a DO of 5 mg/L seldom occurred below 5 m. This increase probably can be attributed partially to the lack of snowfall, which permitted additional light penetration and subsequent photosynthesis of algae during the winters of 1979-80 and 1980-81. In addition, the winter of 1980-81 was milder and the ice cover formed at a later date.

The moderate increase in dissolved oxygen (DO) present after the treatment provided a greater lake volume for biotic activity. The increased zooplankton population (described below) is, at least in part, a response to this additional area of habitation.

pH, ALKALINITY, AND TEMPERATURE

Epilimnetic pH did not change as a result of the treatment. Hypolimnetic pH was initially depressed 0.6-0.8 units after addition of the aqueous alum, but returned to pretreatment levels during fall overturn. The lake's alkalinity adequately buffered the alum addition.

Summer water temperature varied only slightly between years (Fig. 9). Highest temperatures occurred in the surface waters during 1979 and 1981 when 28 C was recorded during July of each year. Lower summer maximum temperatures were observed in 1978 when the surface waters reached only 22 C. These data correspond to atmospheric temperatures reported for the same periods.

SULFATE

High sulfate $SO_4^=$ concentrations in the lower metalimnion and upper hypolimnion were present for 2 months following treatment (Fig. 10). This sulfate, apparently disassociated from the aqueous aluminum sulfate, remained near the application depth. Additional sulfate spikes of 15 mg/L each were observed near the sediment. The presence of the sulfate indicates that reduction of the $SO_4^=$ was not completed for several weeks. The sulfate diffused within the lake until fall overturn.

PHOSPHORUS

Pretreatment

Before lake treatment, phosphorus concentrations routinely reached high levels shortly after the lake's stratification in mid-May. Maximum phosphorus concentrations usually occurred in the hypolimnion during late summer stratification. This phosphorus was transferred to the epilimnion during the downward movement of the thermocline prior to fall overturn. The general increase of chlorophyll during the late summer months of the years before treatment reflects the enrichment from this phosphorus.

Phosphorus levels were compared based upon mean concentrations (and volumetric mass) for shallow and deep waters (Tables 4 and 5). Summer deep water (7.5 m to bottom) mean concentrations typically exceeded 600 *ug*/L for total phosphorus (TP). Most of this phosphorus was soluble reactive phosphorus (SRP), which exceeded 450 *ug*/L in the deep waters.

Summer shallow water (0.5 m to 3.5 m) TP concentrations exceeded 30 *ug*/L in the years before treatment. SRP did not exceed 20 *ug*/L since it was readily utilized by phytoplankton. The availability of phosphorus in the euphotic zone, due to the downward movement of the thermocline and cellular decomposition, was the driving force behind the abundant algae blooms.

Spring and fall overturn samples show variability from year to year (Tables 4 and 5), indicating the difficulty of sampling a lake whose stratification is established rapidly, continues well into October, and has an ice cover by December. The fall TP values for the euphotic zone ranged from 60 to 210 *ug*/L with SRP concentrations ranging from 8 to 17 ug/L. The high organic phosphate component present represents the large fall algae bloom found in this lake prior to alum addition.

Post-treatment

Phosphorus hypolimnetic concentrations immediately after treatment were not significantly reduced. The settling floc had little apparent effect on the hypolimnetic TP and SRP which remained relatively unchanged through the balance of the 1978 statification period. The alum was added to the water column in a more concentrated form than typical past lake treatments. As a result, some of the alum did not produce a pinfloc, but settled rapidly to the sediment.

Phosphorus levels were reduced during the 1978 fall overturn period with

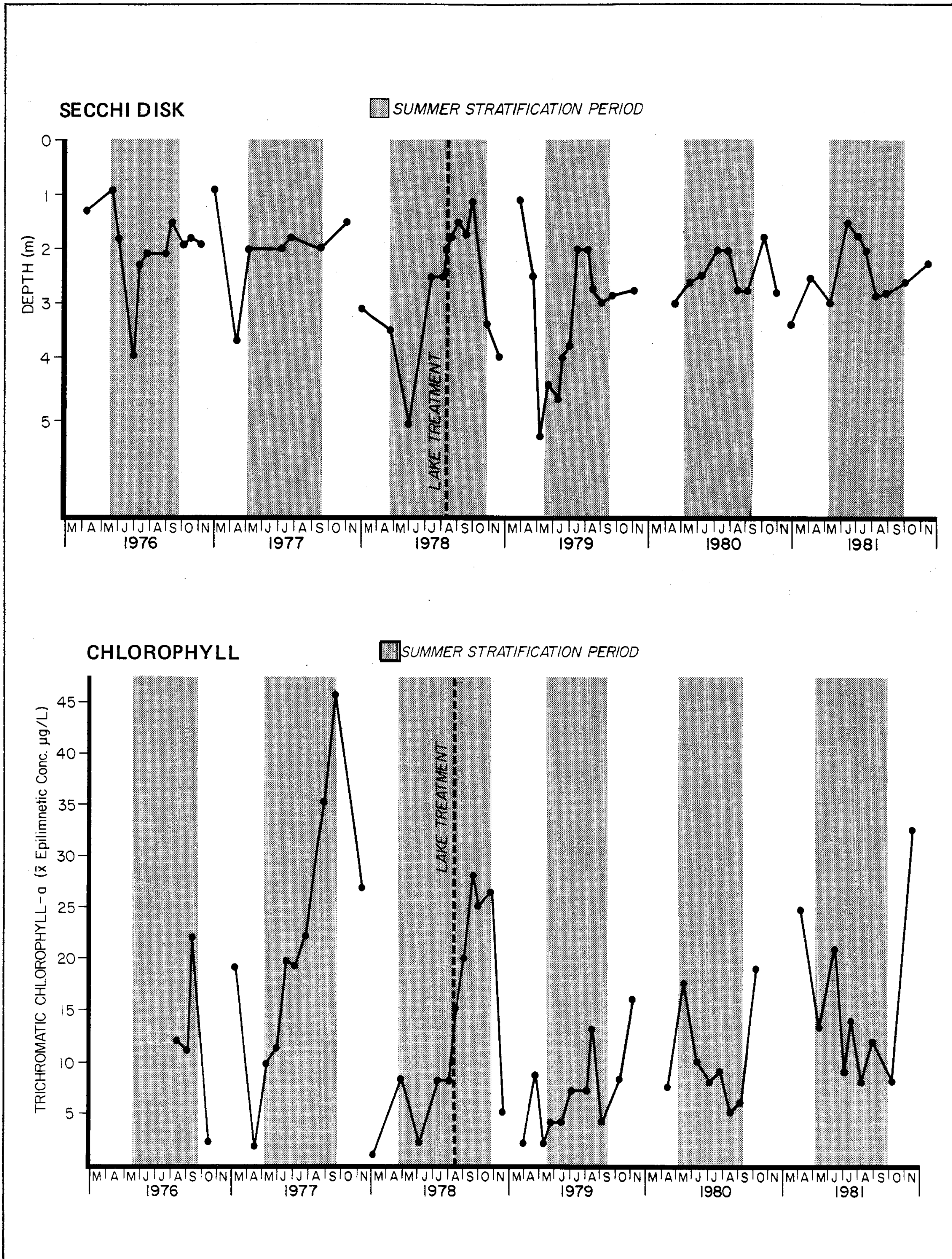


FIGURE 8. *Bullhead Lake seasonal Secchi disk levels and mean epilimnetic chlorophyll-*a *concentrations, 1976-82.*

TABLE 4. *Seasonal total phosphorus mean concentrations and volumetric mean mass for shallow and deep waters of Bullhead Lake.* *

Season**	Pretreatment 1975	Pretreatment 1976	Pretreatment 1977	Pretreatment 1978	Post-treatment 1978	Post-treatment 1979	Post-treatment 1980	Post-treatment 1981	Post-treatment 1982
Winter									
Shallow	—	—	88[a]	128	—	50	—	44	55
	—	—	3.00[b]	4.37	—	1.82	—	1.42	2.09
Deep	—	—	249	223	—	110	—	46	90
	—	—	0.13	0.09	—	0.04	—	0.02	0.03
Spring									
Shallow	—	—	74	135	—	42	42	40	—
	—	—	3.13	4.81	—	1.52	1.43	1.40	—
Deep	—	—	185	145	—	60	68	38	—
	—	—	0.11	0.07	—	0.02	0.03	0.02	—
Summer									
Shallow	30	51	53	41	—	16	35	26	32
	1.00	1.77	1.82	1.50	—	0.59	1.15	0.91	1.08
Deep	815	617	877	742	—	71	78	103	116
	0.34	0.30	0.48	0.31	—	0.02	0.03	0.04	0.04
Fall									
Shallow	60	210	102	—	43	25	33	48	—
	2.42	8.19	3.50	—	1.57	0.91	1.15	1.71	—
Deep	768	220	530	—	39	23	43	34	—
	0.63	0.12	0.07	—	0.02	0.01	0.02	0.01	—

* Shallow = 0.5-3.5 m depth; deep = 7.5-10.5 m depth.
** Winter = January to ice-out; spring = ice-out to mid-May; summer = June to mid-September; fall = fall turnover to ice cover.
[a] Mean concentration (ug/L).
[b] Mean volumetric mass (kg).

TABLE 5. *Seasonal soluble reactive phosphorus mean concentrations and volumetric mean mass for shallow and deep waters of Bullhead Lake.* *

Season**	Pretreatment 1975	Pretreatment 1976	Pretreatment 1977	Pretreatment 1978	Post-treatment 1978	Post-treatment 1979	Post-treatment 1980	Post-treatment 1981	Post-treatment 1982
Winter									
Shallow	—	—	19[a]	102	—	27	—	1	3
	—	—	0.61[b]	3.38	—	1.03	—	0.34	1.18
Deep	—	—	137	235	—	75	—	14	63
	—	—	0.06	0.10	—	0.03	—	0.01	0.03
Spring									
Shallow	—	—	23	76	—	13	tr[c]	tr	—
	—	—	0.95	2.59	—	0.41	0.07	0.04	—
Deep	—	—	173	135	—	13	tr	tr	—
	—	—	0.09	0.05	—	tr	tr	tr	—
Summer									
Shallow	8	19	7	8	—	1	tr	tr	tr
	0.27	0.66	0.26	0.28	—	0.25	0.07	0.07	0.08
Deep	644	491	712	662	—	25	tr	34	38
	0.27	0.25	0.38	0.27	—	0.01	tr	0.01	0.01
Fall									
Shallow	8	47	17	—	4	8	tr	tr	—
	0.32	1.64	0.52	—	0.14	0.37	tr	tr	—
Deep	325	70	26	—	11	tr	tr	tr	—
	0.45	0.03	0.01	—	tr	tr	tr	tr	—

* Shallow = 0.5-3.5 m depth; deep = 7.5-10.5 m depth.
** Winter = January to ice-out; spring = ice-out to mid-May; summer = June to mid-September; fall = fall turnover to ice cover.
[a] Mean concentration (ug/L).
[b] Mean volumetric mass (kg).
[c] tr = <1 ug/L concentration or <0.005 kg mean mass.

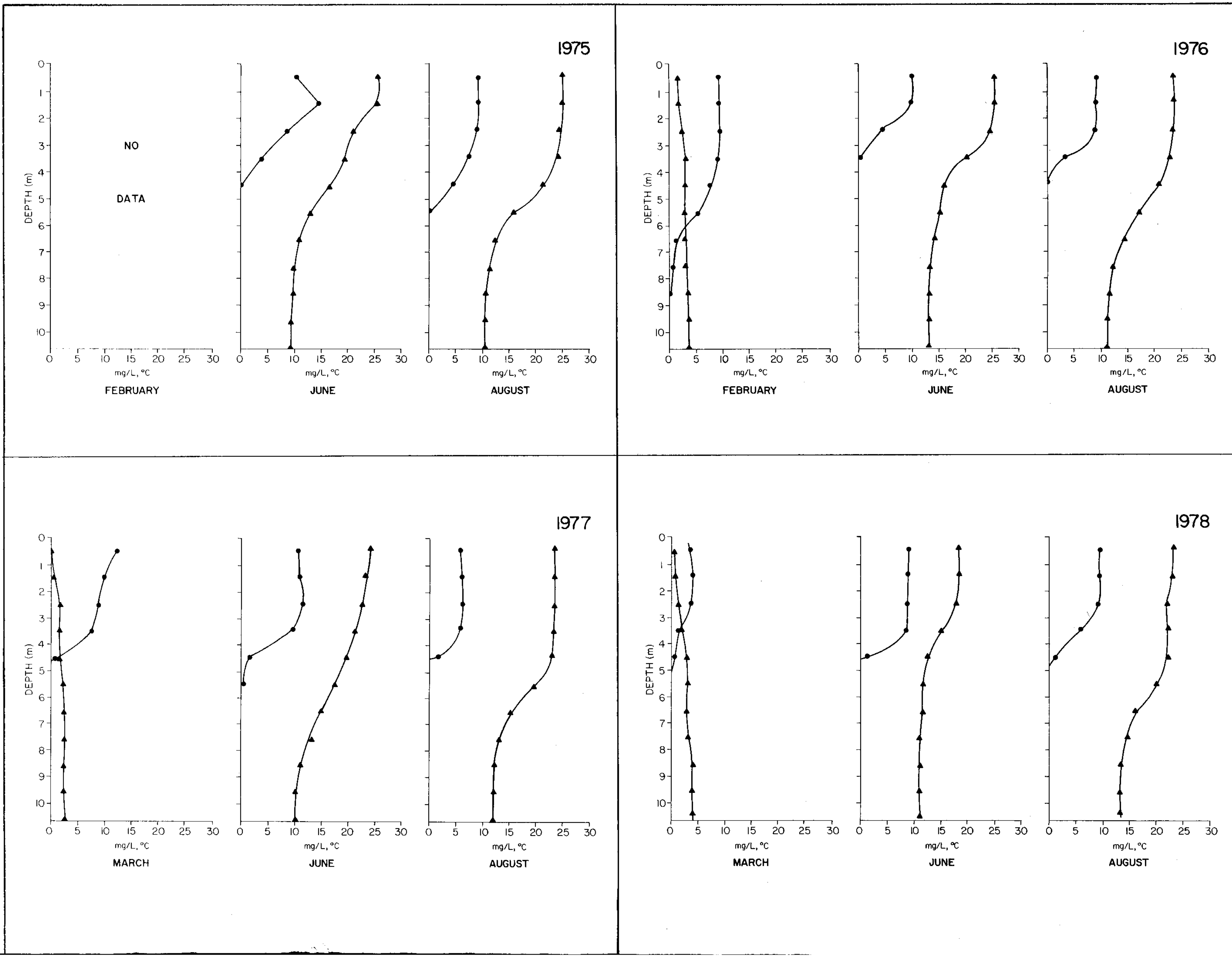
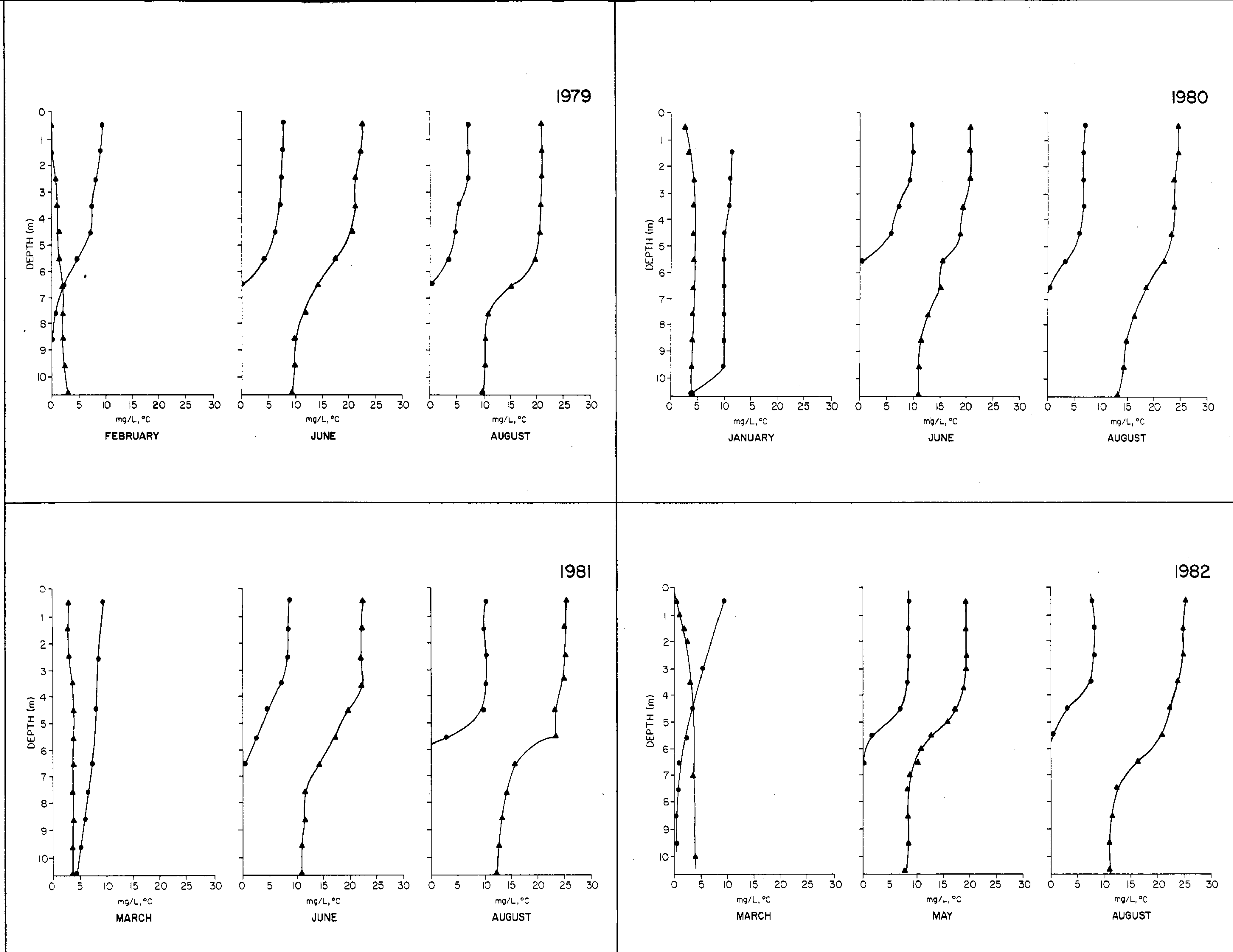


FIGURE 9. *Selected temperature and dissolved oxygen profiles for winter, spring, and summer, in Bullhead Lake, 1975-82.*


1979
DEPTH (m)
mg/L, °C
FEBRUARY
JUNE
AUGUST
1980
JANUARY
JUNE
AUGUST
1981
MARCH
JUNE
AUGUST
1982
MARCH
MAY
AUGUST

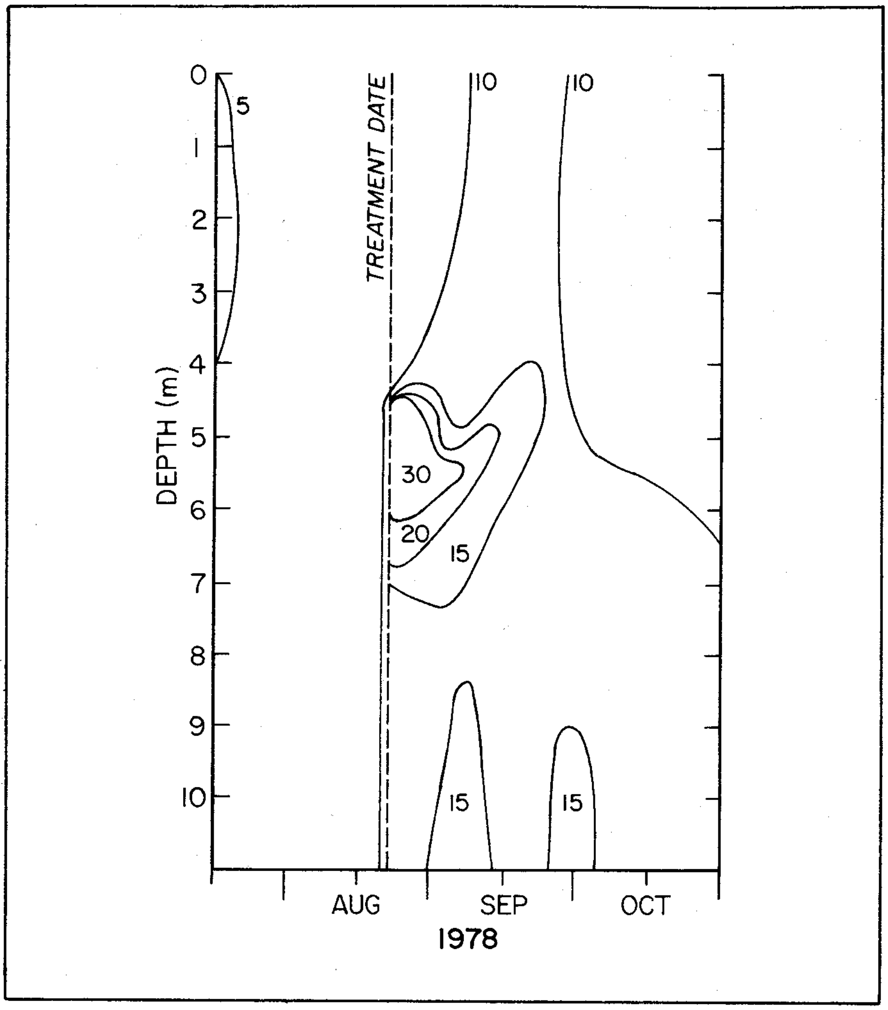


FIGURE 10. *Sulfate concentrations following the alum treatment.*

maximum reduction occurring during the 1979 spring mixing period. During mixing the waters were able to circulate near the alum flocculent where the phosphorus was removed by either direct SRP-floc contact or by adsorption of remineralized SRP from particulate organic phosphorus in the seston.

Only trace levels (< 2 *ug*/L) of SRP were observed in the euphotic zone during the open water seasons of 1980-82, and most bottom water samples contained only trace quantities of SRP. The absence of a readily available phosphorus supply altered the diversity and density of plankton and chlorophyll-*a* production.

Based on the mean concentrations shown in Tables 4 and 5, the alum treatment reduced epilimnetic summer TP and SRP average concentrations by 38% and 92%, respectively. The hypolimnetic summer mean concentrations were likewise reduced by 90% and 96%. The absence of a major reduction of TP in the epilimnion is the result of the continued presence of algae.

NITROGEN

Total nitrogen concentrations (mostly NH_4^+) in individual samples of the hypolimnion often exceeded 6 mg/L before alum treatment. In September 1977, the hypolimnion had the largest observed concentration (12.6 mg/L) at the 9.5-m depth. Following treatment, the total nitrogen sample levels were generally at or just below the 6 mg/L typical of pretreatment levels.

These variations in nitrogen concentration had little effect on the overall volumetric lake mass. A comparison of mean concentrations and seasonal mass shows only a small reduction in summer and fall quantities after treatment (Tables 6, 7, 8). The most significant reduction occurred in the inorganic nitrogen during the 1980 and 1981 summer seasons (Table 7). Although these reductions were less than half of the typical pretreatment levels, the significance of these changes could not be determined because of the greater impact of the phosphorus reduction.

CHLOROPHYLL

Before alum treatment, the chlorophyll values typically increased throughout the summer (Fig. 8). Chlorophyll concentrations were reduced at fall overturn (1978) and remained low into midsummer (1979). Following treatment chlorophyll values remained below pretreatment (1975-78) levels but increased from 1980 to 1981. Mean summer values for 1979, 1980, and 1981 were 6.5, 7.6, and 12.8 *ug* chlorophyll-*a*/L, respectively. By contrast summer values for 1976 through 1978 were 15.0, 20.5, and 13.7 *ug* chlorophyll-*a*/L. The treatment reversed the usual summer increase in chlorophyll. Summer epilimnetic chlorophyll levels now tended to decrease throughout the summer stratification period.

BIOLOGICAL CHANGES

Phytoplankton

The ratio of total nitrogen to total phosphorus (TN:TP) in the epilimnion influences the relative proportion of blue-green algae present in the lake. TP:TN ratios greater than 10-12:1 (Sakamoto 1966) and 29:1 (Smith 1983) have been suggested for reduction in the blue-green algae component. The application of alum reduces the availability of phosphorus creating a high TN:TP ratio and interrupted the established growth pattern of algae. This alteration reduced the blue-green algae and created a more diverse fauna. The algal diversity present after the treatment did not produce rapid increases and declines in summer chlorophyll production. Consequently, the lake's general appearance improved with the absence of clump-forming algae (typically blue-greens) found prior to treatment.

Nitrogen fixing blue-greens were common in pretreatment samples (Table 9), but decreased in the immediate years after treatment. Green algae were uncommon in the samples prior to treatment and more abundant after treatment. Surface TP and chlorophyll-*a* levels were not reduced nor stabilized until after lake treatment. In addition, the nitrogen components remained the same throughout the study. Post-treatment TN:TP mean epilimnetic ratios ranged from 140:1 (1979) to 33:1 (1982); the greatest range of values occurred in 1979 immediately after treatment (Fig. 11). Blue-greens increased slightly in 1980 and 1981, but the population maintained greater diversity than before treatment.

TABLE 6. *Seasonal total nitrogen mean concentrations and volumetric mean mass for shallow and deep waters of Bullhead Lake.* *

	Pretreatment				Post-treatment				
Season**	1975	1976	1977	1978	1978	1979	1980	1981	1982
Winter									
Shallow	—	—	2,586[a]	2,633	—	2,363	—	2,135	2,483
	—	—	90.16[b]	91.71	—	83.25	—	75.00	91.43
Deep	—	—	3,395	3,175	—	3,399	—	2,110	2,348
	—	—	2.04	1.29	—	1.31	—	0.91	0.99
Spring									
Shallow	—	—	1,809	2,168	—	1,660	1,697	1,730	—
Deep	—	—	64.98	77.90	—	59.15	59.43	60.78	—
	—	—	2,779	2,508	—	2,484	2,099	1,930	—
	—	—	1.43	1.03	—	0.97	0.87	0.86	—
Summer									
Shallow	1,117	1,519	1,590	1,604	—	1,237	1,402	1,308	1,373
	38.37	54.92	56.30	56.07	—	43.21	49.22	45.82	47.91
Deep	5,130	5,525	6,767	5,944	—	3,822	3,455	4,692	3,598
	2.05	2.63	3.73	2.54	—	1.61	1.38	1.88	1.48
Fall									
Shallow	1,884	2,589	2,753	—	2,811	1,588	1,599	2,324	—
	65.95	93.04	97.13	—	98.95	56.04	56.04	77.93	—
Deep	3,578	2,450	2,787	—	2,793	1,767	2,649	2,013	—
	1.17	1.32	1.56	—	1.15	0.97	1.05	0.87	—

* Shallow = 0.5-3.5 m depth; deep = 7.5-10.5 m depth.
** Winter = January to ice-out; spring = ice-out to mid-May; summer = June to mid-September; fall = fall turnover to ice cover.
[a] Mean concentration (*ug*/L).
[b] Mean volumetric mass (kg).

TABLE 7. *Seasonal inorganic nitrogen mean concentrations and volumetric mean mass for shallow and deep waters of Bullhead Lake.**

	Pretreatment				Post-treatment				
Season**	1975	1976	1977	1978	1978	1979	1980	1981	1982
Winter									
Shallow	—	—	887[a]	1,165	—	1,216	—	750	1,128
	—	—	30.42[b]	40.73	—	42.45	—	26.34	43.05
Deep	—	—	1,431	1,662	—	1,997	—	783	465
	—	—	0.92	0.68	—	0.73	—	0.33	0.24
Spring									
Shallow	—	—	396	1,072	—	690	340	437	—
	—	—	15.59	37.96	—	24.22	11.88	15.44	—
Deep	—	—	1,414	1,330	—	1,209	723	440	—
	—	—	0.69	0.55	—	0.44	0.27	0.19	—
Summer									
Shallow	153	140	139	259	—	181	55	69	138
	5.42	5.25	4.86	9.12	—	6.56	1.86	2.29	4.40
Deep	3,079	3,697	4,921	4,104	—	2,472	1,780	3,256	2,199
	1.19	1.79	2.89	1.73	—	1.04	0.63	1.25	0.85
Fall									
Shallow	322	834	820	—	996	528	291	1,034	—
	11.43	30.14	28.40	—	35.17	18.85	10.20	32.55	—
Deep	1,881	878	914	—	955	617	1,265	771	—
	0.39	0.44	0.76	—	0.41	0.34	0.45	0.33	—

* Shallow = 05-3.5 m depth; deep = 7.5-10.5 m depth.
** Winter = January to ice-out; spring = ice-out to mid-May; summer = June to mid-September; fall = fall turnover to ice cover.
[a] Mean concentration (*ug*/L).
[b] Mean volumetric mass (kg).

TABLE 8. *Seasonal organic nitrogen mean concentrations and volumetric mean mass for shallow and deep waters of Bullhead Lake.**

	Pretreatment				Post-treatment				
Season**	1975	1976	1977	1978	1978	1979	1980	1981	1982
Winter									
Shallow	—	—	1,699[a]	1,468	—	1,147	—	1,385	1,355
	—	—	59.74[b]	50.98	—	40.80	—	48.66	48.38
Deep	—	—	1,964	1,513	—	1,402	—	1,327	1,883
	—	—	1.12	0.61	—	0.58	—	0.58	0.75
Spring									
Shallow	—	—	1,413	1,096	—	970	1,357	1,293	—
	—	—	49.39	39.94	—	34.93	47.55	45.34	—
Deep	—	—	1,365	1,178	—	1,275	1,376	1,490	—
	—	—	0.74	0.48	—	0.53	0.60	0.67	—
Summer									
Shallow	964	1,379	1,451	1,345	—	1,056	1,347	1,239	1,235
	32.95	49.67	51.44	46.95	—	36.65	47.36	43.53	43.51
Deep	2,051	1,828	1,846	1,840	—	1,350	1,675	1,436	1,399
	0.86	0.84	0.84	0.81	—	0.57	0.75	0.63	0.63
Fall									
Shallow	1,562	1,755	1,933	—	1,815	1,060	1,308	1,290	
	54.52	62.90	68.73	—	63.78	37.21	45.84	45.38	—
Deep	1,697	1,572	1,873	—	1,838	1,150	1,384	1,242	—
	0.78	0.88	0.80	—	0.74	0.63	0.60	0.54	—

* Shallow = 0.5-3.5 m depth; deep = 7.5-10.5 m depth.
** Winter = January to ice-out; spring = ice-out to mid-May; summer = June to mid-September; fall = fall turnover to ice cover.
[a] Mean concentration (*ug*/L).
[b] Mean volumetric mass (kg).

Chlorophyll production was limited by inorganic nitrogen when TN:TP was <10:1 and by inorganic phosphorus when TN:TP was >17:1 (Sakamoto 1966). Therefore, blue-green algae that fix nitrogen (*Anabaena* spp. and *Aphanizomenon* spp.) would have an advantage when the TN:TP ratio was <10-12:1. Our results indicate that ratios higher than 10-12:1 will still support blue-green production, although that production will be reduced and modulated by other species of algae and chlorophyll production can remain unchanged. The 29:1 TN:TP ratio suggested by Smith (1983) may better suit blue-green algae reduction in this lake.

The phytoplankton community present before the alum treatment was characteristic of other mesotrophic/eutrophic lakes with low nutrient inputs during stratification (Findenegg 1965, Garrison and Knauer 1983). Although phosphorus concentrations were high in the hypolimnion, the typically low diffusivity coefficient across a thermocline restricted the phytoplankton from utilizing hypolimnetic phosphorus during stratification until the thermocline moved downward in late summer. Phytoplankton assimilated some phosphorus released by other decomposing plankton while the balance of the TP (mainly organic P) settled to the bottom as seston, building the organic layers in the sediments and remineralized.

Spring diatom pulses are often short and were not observed in most years due to infrequent sampling and the rapid onset of stratification, which often prevented total lake mixing. The settling diatom frustules and stratification inhibited the resupply of silicates and other nutrients necessary for diatom production. Likewise, the deposition of algal seston in the spring and early summer of 1979 created phosphorus-limited conditions in the epilimnion. The only strong pulse of algae observed during spring was found in 1981. This pulse was dominated by the diatom *Asterionella formosa*.

During the summer months of this study the Chlorophyceae (green algae) were numerically dominant followed by the Cyanophyceae (blue-green algae) (Table 9). The 1977 summer plankton assemblage was dominated by the diatom *Fragilaria crotonensis* and the pyrrhophyte *Ceratium hirundinella* and the blue-greens *Lyngbya Birgei, Aphanizomenon flos-aquae,* and *Anabaena* spp. Blue-green algae blooms were sometimes present in large enough quantities to be considered a problem by local residents.

Following the August 1978 treatment, changes occurred in the summer phytoplankton. While the summer biovolumes were similar to previous values, changes occurred in the dominant genera. The blue-green *Aphanizomenon flos-aquae* and diatom *F. crotonensis* abundance decreased and the pyrrhophyte *Cryptomonas* spp., blue-greens *Anabaena flos-aquae* and *A. planctonica* were more common. In 1979 the blue-green *L. Birgei* increased and continued to be an important component of the algal assemblage through the remainder of the study. Some taxa were represented by only a few individuals; the 1979 taxa list, for example, contained 15 taxa only found during that year. The high fecundity of algae could account for these ephemeral occurrences during this unstable period of lake chemistry.

The alum greatly reduced the fall algae bloom. The large population of blue-green algae typical after fall overturn decreased greatly starting in the fall of 1979 when the phosphorus levels were reduced during the fall overturn. A large fall bloom was not observed after alum treatment. In 1980 and 1981, blue-green algae were only a minor component of the fall assemblage, with the pyrrhophyte *Cryptomonas* spp. and the diatom *F. crotonensis* dominating.

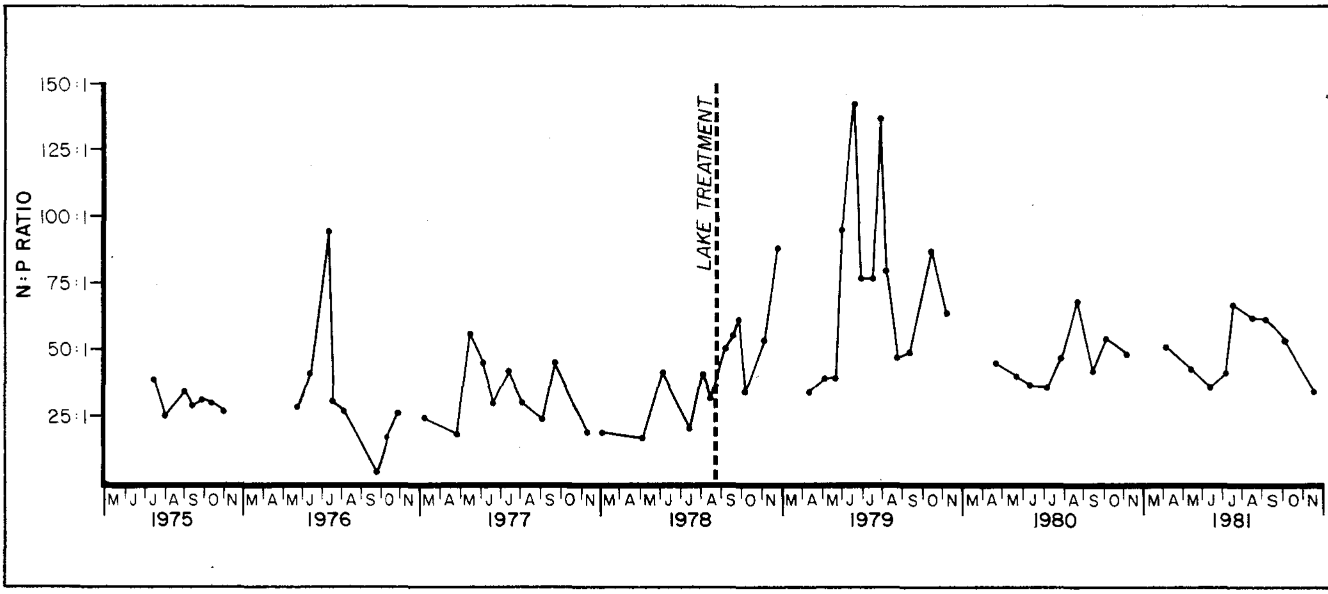


FIGURE 11. *Nitrogen to phosphorus ratio (TN:TP) for Bullhead Lake, 1975-81.*

Zooplankton

All zooplankton responded similarly following the alum treatment. Peak diversity and biomass occurred from late 1978 through 1980. These changes followed the shift in phytoplankton toward a more diverse community of diatoms, flagellates, and greens, which provided a more suitable diet for the zooplankton.

Copepods. The calanoids were limited to one taxa — *Skistodiaptomus oregonensis* (Lilljeborg). Copepod density was typically low but increased in late 1980 and early 1981, apparently in response to the greater numbers of cyclopoids and smaller forms of green algae present at that time. The cyclopoids and the smaller forms of green algae in particular are the desired food of the calanoid *S. oregonensis* (McQueen 1970).

Interaction between plankters was observed in several cases (Fig. 12). *Cyclops bicuspidata thomasi* Forbes is a known predator on its own nauplii and on the calanoid *Skistodiaptomus oregonensis* (McQueen 1969). Adults of *C. b. thomasi* were most abundant when large numbers of cyclopoid nauplii were present. The substantial 1980 increase in cyclopoid numbers coincided with the greatest abundance of green algae observed during this study, thus providing a desirable food source.

The major cyclopoid taxa were *Cyclops bicuspidata thomasi* Forbes and *Mesocyclops edax* Forbes. The dominant growth stadia were the nauplii, and the balance of the fauna included many young copepodids.

Cladocerans. Cladocerans maintained a small diverse population from 1975 until the fall of 1979 (Fig. 13). Before alum treatment, the dominant taxa were *Daphnia pulicaria* and *D. galeata mendota* Birge. Population density was variable and typically remained below 50/L. Following alum addition, the taxa composition became more complex with *Bosmina longirostris* (Muller) and *Ceriodaphnia lacustris* Birge becoming equally represented in the taxa. The 1979 September peak was nearly all *B. longirostris* and the 1980-81 samples combined varying numbers of *B. longirostris, C. lacustris, D. galeata mendota,* and *Daphnia* spp.

Rotifers. Rotifers had the greatest population shift of the zooplankton examined (Fig. 14). Population estimates were typically less than 100/L prior to lake treatment. The main pretreatment taxa were *Keratella cochlearis* (Grosse), *K. quadrata* Muller, and *Polyarthra vulgaris* Carlin. In October 1978, the population increased to about 500/L. The 1979 population remained below 100/L until July, when the density increased to more than 600/L.

Following treatment, the rotifers increased in both density and diversity. The population reached its maximum (ca. 1200/L) in October 1980. The increased diversity included several raptorial species. *Polyarthra vulgaris* Carlin, *Trichocerca cylindrica* (Imhof), *T. similis* (Wierzejski), and *T. multiformis* (Kellicota) routinely preyed on *Conochilus unicornis* Rousselet, *Keratella cochlearis* (Grosse) and *Gastropus stylifer* Imhof. Periodic declines throughout the season appear to result from the appearance of the raptorial species (Fig.14). The dichotomy of predator-prey density is most evident in the 1979-81 samples when *C. unicornis, G. stylifer, K. cochlearis,* and *K. bostoniensis* routinely crashed as the predator (mainly *P. vulgaris*) increased.

Benthos

Chaoborids. A portion of the meroplanktonic *Chaoborus punctipennis* (Say) (Fig. 15) population was sampled during the open water season using Eckman grabs. This summer population is underestimated since the I and II instar larvae are usually planktonic and not readily obtained by benthic grab; III and IV instar larvae were found in the sediments at all sample buoys.

The greatest density of *C. punctipennis* was found in the sediment at depths below 6 m. Maximum annual density was found one year after lake treatment when more than 43,000/m^2 occurred at both the 8- and 10-m depths during November 1979. At other times the population at 8 m did not exceed 8,000/m^2, and the 10-m population varied from 100/m^2 to 48,000/m^2. The largest population typically occurred at the 10-m depth during spring and fall.

Copepods. Copepods (Fig. 15) were abundant in the summer benthic samples. *Cyclops bicuspidata thomasi* and *Mesocyclops edax* were present in the bottom muds during the warm summer months. Throughout the study, their occurrence was proportional to the planktonic population. The greatest

TABLE 9. *Relative volume of epilimnetic phytoplankton in Bullhead Lake for June through September, 1977-81.*

Taxa	Relative Abundance *				
	1977	1978	1979	1980	1981
Chlorophyta					
Chlorophyceae					
Ankistrodesmus falcatus(West & West) G.S. West			P	P	
Carteria spp.				P	
Chlamydomonas gloeophila Skuja		R			
C. mucicola Schmidle			R		
Chlamydomonas spp.		R	R	R	R
Chlorella vulgaris Beyerinck				R	
Closterium spp.					R
Cosmarium spp.			R	R	R
Crucigenia rectangularis (A. Braun) Gay		R			
Euastrum spp.				R	
Eudorina elegans Ehrenberg				R	R
Geminella spp.			R		
Gloeocystis spp.	P			P	R
Golenkinia radiata (Chod.) Wille				R	
Golenkinia spp.			R		
Kirchneriella contorta (Schmidle) Bohlin				R	
Lagerheimia citriformis Tiffany & Ahlstrom				R	
L. *quadriseta* (Lemm.) G.M. Smith				R	R
Micractinium pusillum Fresenius			R		
Oocystis crassa Wittrock					R
O. pusilla Hansgirg				P	R
O. solitaria Wittrock				R	
O. submarina Lagerheim				R	C
Oocystis spp.	P	R	R	R	P
Pandorina spp.			R		
Pediastrum Boryanum (Turp.) Meneghini		P			
P. duplex Meyen			P	P	
Quadrigula lacustris (Chod.) G.M. Smith		R			R
Quadrigula spp.		R		R	
Scenedesmus abundans (Kirch) Chodat			R		
S. denticulatus Lagerheim			R		
S. dimorphus (Turp.) Kuetzing			R		
S. longus Meyen				R	
S. quadricauda (Turp.) de Bredisson			R	R	
Schroederia Juday G.M. Smith	R	R			R
S. setigera (Schrod) Lemmermann			R		
Selenastrum spp.			R	R	R
Sphaerocystis Schroeteria Chodat	R	P		P	P
Sphaerocystis spp.		R			
Staurastrum contortum G.M. Smith				R	
Staurastrum spp.	P		R	R	R
Tetraedron minimum (A. Braun) Hansgirg				R	R
Ulothrix subtilissima Kuetzing			R		
Cyanophyta					
Myxophyceae					
Anabaena flos-aquae (Lyngb.) de Brebisson				P	P
A. helicoidea Berharn	C				
A. planctonica Brunnthaler				R	C
A. spiroides Klebahn				P	P
Anabaena spp.	C		P		P
Aphanizomenon flos-aquae (L.) Ralfs	R	A			
Aphanocapsa elachista West & West	R				R
Aphanothece spp.	P		R	R	
Arthrospira Jenneri (Kuetz.) Stizenberger			R		
Chroococcus dispersus G.M. Smith	P				
C. limneticus Lemmermann	R	R	R	P	R
C. minimus (Keissl.) Lemmermann				R	
Chroococcus spp.	R	R			R
Coelosphaerium dubium Grunow				P	
C. Kuetzingianum Naegeli	R			R	
C. Naegelianum Unger	R	R	R	P	P
C. pallidum Lemmermann					R
Dactylococcopsis rhaphidioides Hansgirg			R		
Lyngbya Birgei G.M. Smith	C	R	C	P	A
Microcystis aeruginosa Kuetzing			C	P	R
Oscillatoria Agardhii Gomont		R			P
Spirulina spp.			R		

TABLE 9. (Cont.)

Taxa	Relative Abundance * 1977	1978	1979	1980	1981
Chrysophyta					
Bacillariophyceae					
Amphora ovalis (Kutz.) Kutz.			R		
Cyclotella comensis Gronow			R	C	
Fragilaria crotonensis Kitton	A	C	P	P	
Fragilaria spp.				R	
Melosira granulata (Ehr.) Ralfs		P			
Stephanodiscus tenuis Ehrenberg				R	
Synedra acus Kutz.			P		
S. amphicephala Kutz.			P		
S. nana Meist.			R		
Synedra spp.					R
Chrysophyceae					
Dinobryon spp.			R		
Erkena subaequiciliata Skuja				R	
Erkena spp.	R	R	R		R
Mallomonas tonsurata Teiling			P		
Mallomonas spp.		R	P		
Ochromonas spp.			P	P	
Pyrrhophyta					
Cryptophyceae					
Chroomonas acuta Utermohl	R	R	R	P	P
C. coerulea (Geitl.) Skuja	R	R		R	R
C. reflexa Kisselew				R	
Chroomonas spp.		R	P		
Cryptomonas ovata Ehrenberg				P	
C. spp.	R	P	P		C
Dinophyceae					
Ceratium hirundinella (O. F. Muell.) Dujardin	C	C	C	C	P
Glenodinium pulvisculus (Ehrenb.) Stein				P	
Glenodinium spp.					P
Gymnodinium spp.					R

SUMMARY:

Number of Taxa Represented:	1977	1978	1979	1980	1981
Chlorophyceae	5	10	18	25	16
Myxophyceae	11	6	9	11	12
Bacillariophyceae	1	2	6	4	1
Chrysophyceae	1	2	5	2	1
Cryptophyceae	3	5	3	4	3
Dinophyceeae	1	1	1	2	2

*A = Abundant (> 20% by biovolume)
C = Common (> 10% - < 20%)
P = Present (> 1% - < 10%)
R = Rare (< 1%)

numbers were observed at the 10-m depth in 1978, 1980, and 1981.

Chironomids. A moderate population of chironomids (*Chironomus* spp.) was present in the bottom mud throughout the study (Fig. 15). The maximum pretreatment population was 500/m^2 at the 8-m buoy in May 1978; however, typical populations did not exceed 100/m^2.

Chironomid I instar and some II instar larvae are primarily planktonic and were not collected in the benthic grabs. In addition, I and II instar larvae pass through the 188 x 412 um mesh sieve during sample processing. Even with these possible influences on chironomid data, the enumerated population increased substantially during 1980 and 1981 (Fig. 15). The greatest increase occurred in 1981 at the 6- and 8-m depths, where densities of 1,400 and 2,400/m^2 occurred. The maximum observed population at the 10.5-m depth was 600/m^2 in April 1981.

Palpomyia spp. were found in small numbers at all buoys throughout the study. Other benthic chironomidae appeared in Bullhead Lake after the treatment. These included the chironomids *Procladius* spp. and *Coelotanypus* spp., also water mites, snails, and leeches.

The enhanced zooplankton population present after lake treatment apparently provided an improved food source for the benthic inhabitants. The large chaoborid populations (Fig. 15) of May 1978 and November 1979 responded to this increased fecundity. *Chaoborus* spp. feed primarily on plankton (Pastorok 1980), and the choice of food is determined by catchability and size (Pastorok 1981). They use medium-sized prey due to their greater encounter and capture susceptibility (Pastorok 1981). Cyclopoid copepods were present in increased numbers in spring 1978 (Fig. 12), and *Bosmina longirostris* was observed in large numbers during August and September 1979 (Fig. 13). These small to moderate sized (ca. 0.6 mm) plankton provided an abundant food source for

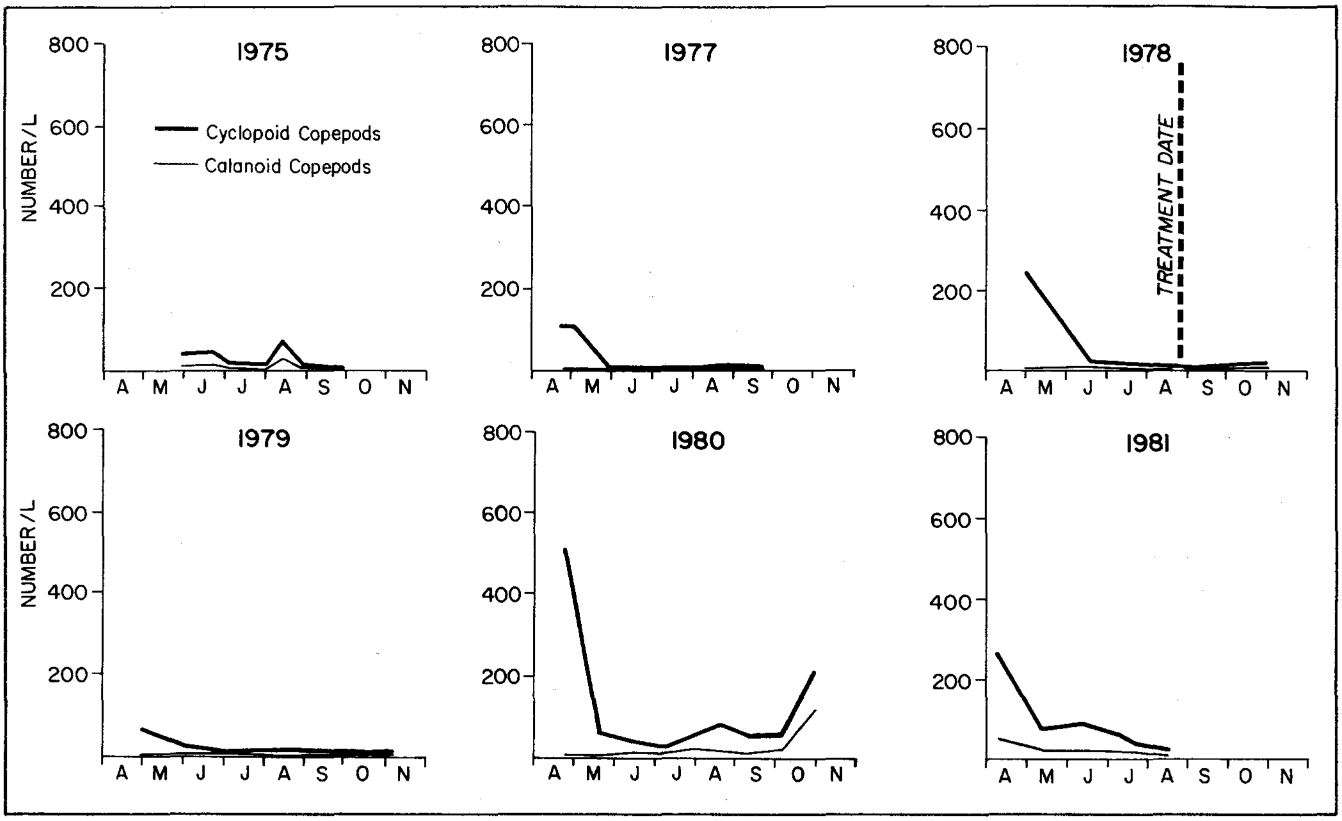


FIGURE 12. *Epilimnetic copepod population estimates (accumulated total) from mid-lake composite water samples, 1975-81.*

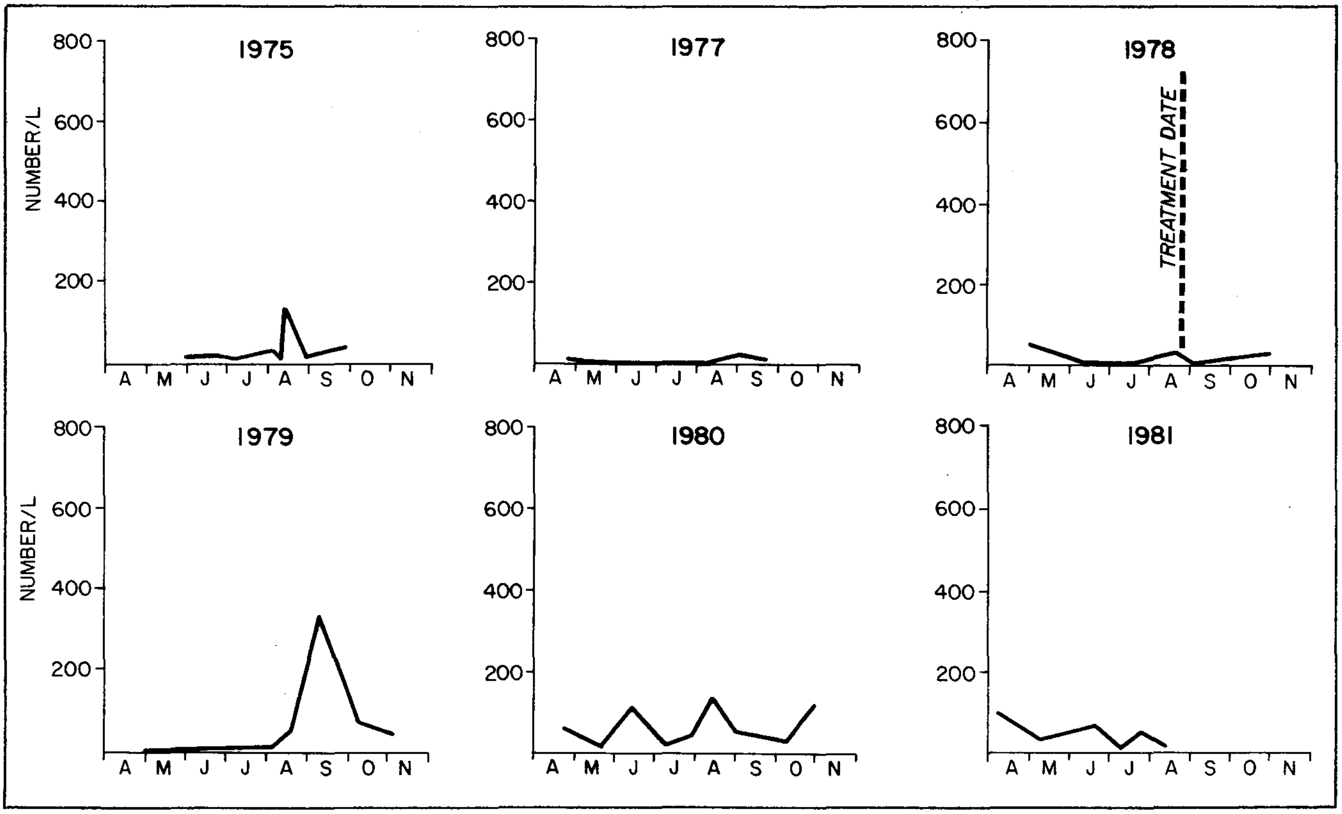


FIGURE 13. *Epilimnetic cladoceran population estimates from mid-lake composite water samples, 1975-81.*

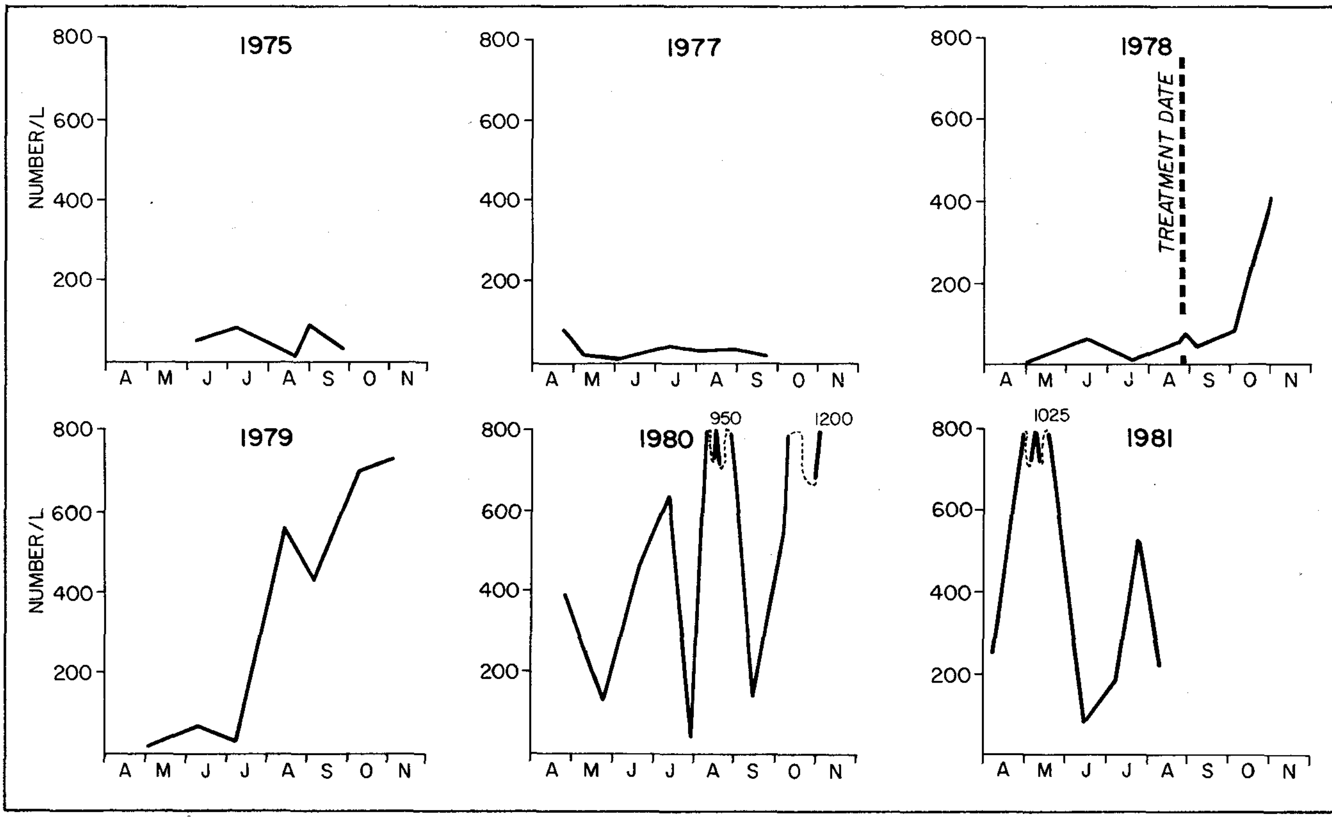


FIGURE 14. *Epilimnetic rotifer population estimates from mid-lake composite water samples, 1975-81.*

III and IV instar *Chaoborus punctipennis*. The I and II instar larvae of July and August 1979 were able to feed on the abundant rotifer *Keratella cochlearis* (size ca. 0.12-0.15 mm) as suggested by Pastorok (1980).

The sampling buoys were established to show lateral movement of chironomid larvae from the 6- and 8-m substrate to the 10-m depth. A reduction in plankton biomass was expected with subsequent decrease in benthic oxygen demand. A partial reduction in seston production did occur. Winter anoxia decreased below the 8-m depth, but there was no decrease in summer anoxia. Lacking a shorter summer anoxia at the 10-m depth, an increase in chironomid numbers could not be expected.

The 1981 chironomid increase was due partially to a dissolved oxygen increase in winter, but was most likely influenced by food supply. Evidence of this was the chironomid:copepod density relationships during 1980 and 1981 (Fig. 15). Planktonic copepods, namely *Cyclops bicuspidata thomasi* and *Mesocyclops edax* were present in the sediment during the summer. The benthic summer copepod count at the 10-m buoy remained high while the 8- and 6-m copepod count was significantly lower. Chironomid numbers were reversed with the greatest densities occurring at the 8- and 6-m buoys. Apparently, the copepods provided a food source for those chironomids occupying the muds where the oxygen content had improved. Chironomids scavenge food from the area around their burrows and routinely feed on organic matter near their burrows (Merritt and Cummins 1978, McCafferty 1981).

Procladius spp., *Coelotanypus* spp., and *Palpomyia* spp. are typically found in other Wisconsin lakes (Hilsenhoff 1966, Hilsenhoff and Narf 1968). These forms appeared in Bullhead Lake after the alum treatment, due to the general lake improvement. These three taxa are moderately (*Procladius* and *Coelotanypus*) to predominately (*Palpomyia*) predacious and responded to the presence of a food source, either young chironomids, crustaceans, or other forms accumulating on the bottom.

The addition of alum caused no toxic effects to any member of the benthic or meroplanktonic community during 1979-81. Greater benthic density and diversity occurred in the years following treatment. The benthos, previously consisting of only two taxa (*Chironomus* spp. and *Chaoborus* spp.), now contains the more diversified fauna typical of other Wisconsin lakes (Hilsenhoff 1966, Hilsenhoff and Narf 1968).

SEDIMENT PHOSPHORUS AND ALUMINUM

Lake sediment was tested for phosphorus and aluminum content in 1978 before treatment and again in May 1979 following treatment. Background levels of total phosphorus (TP) and total aluminum (TAl) in the sediment samples for 1978 ranged from 10-15 mg TP/g and 15-20 mg TAl/g (Fig. 16). Aluminum and phosphorus were significantly higher in the substrate at the 10- and 6-m water depths in May 1979 after alum addition. Phosphorus concentrations increased proportionately where the aluminum concentrations were present in the sediment column. The May 1979 cores showed the highest concentration of aluminum in the upper 5 cm of sediment with some increase in phosphorus down to 7.5 cm., indicating a "barrier" effect from the added alum.

Multiple sediment samples for visual observation were taken in August 1980. A SCUBA diver inserted and capped glass tubing (2 cm diameter) into the substrate. Sediment closest to the interface was found to have a reduced abundance of the aluminum floc. The largest concentration of floc in the sediment cores was observed at the 3-6 cm depth. Moderate floc occurred at 0-

3 cm and no floc was observed below 7 cm. Density equilibrium was apparently established at the 3-6 cm depth.

This movement has also been observed in other alum-treated lakes (Garrison and Knauer 1983, Narf 1978) and was indicated by Fink (1969) and Loring and Nota (1973), based on density differentials. In addition, other biotic and abiotic activities, such as chironomid burrowing and anaerobic gas release (Narf 1981), can cause rapid movement of sediment particles and a subsequent rearrangement of the upper 5 cm of sediment.

Although the larger amount of alum floc was below the mud surface, the alum did remain functional through 1980. Midsummer and fall phosphorus samples in 1980 and 1981 showed a continued significant reduction in phosphorus concentration.

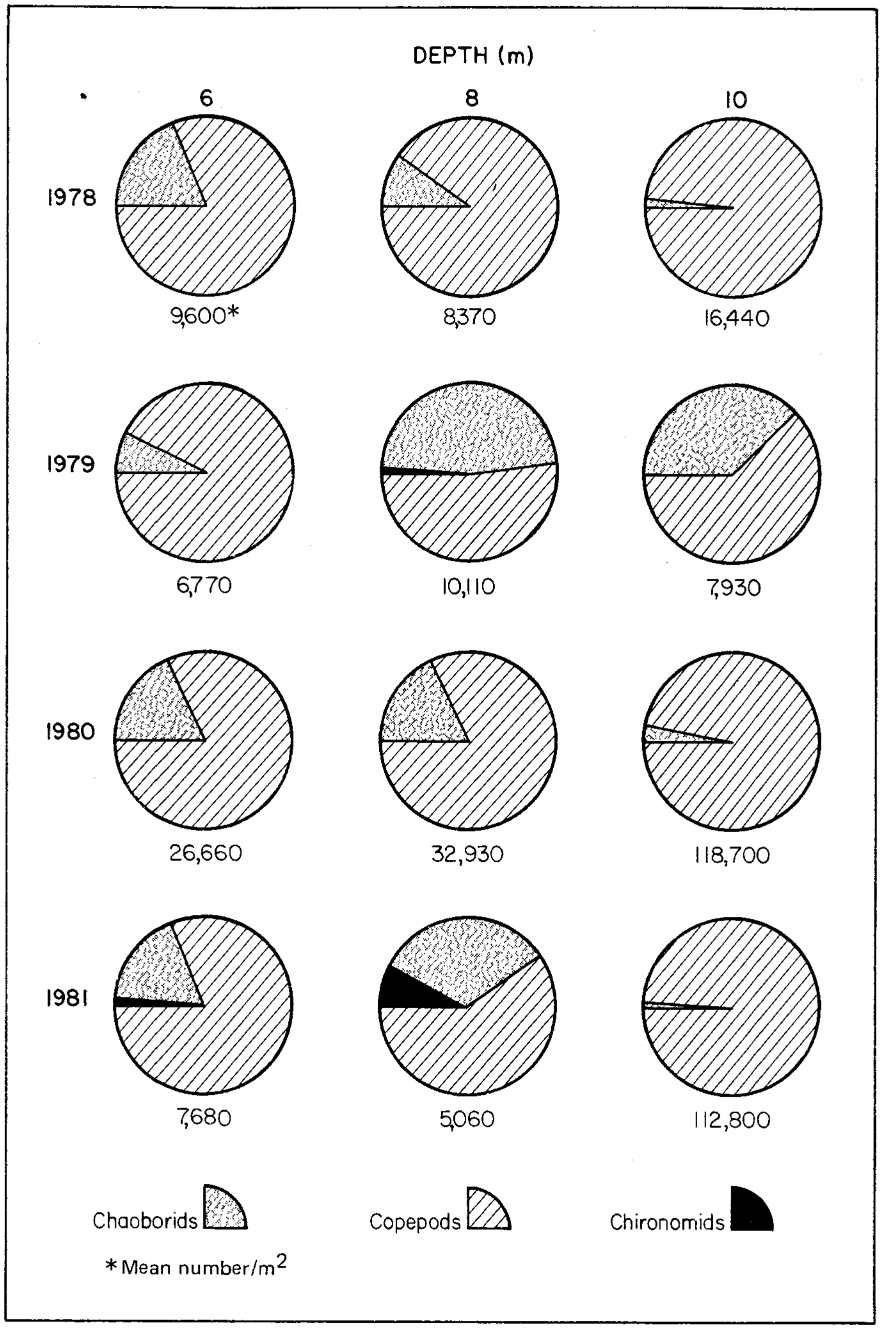


FIGURE 15. *Benthic chaoborid, chironomid, and copepodid population estimates from Eckman dredge samples, 1975-81.*

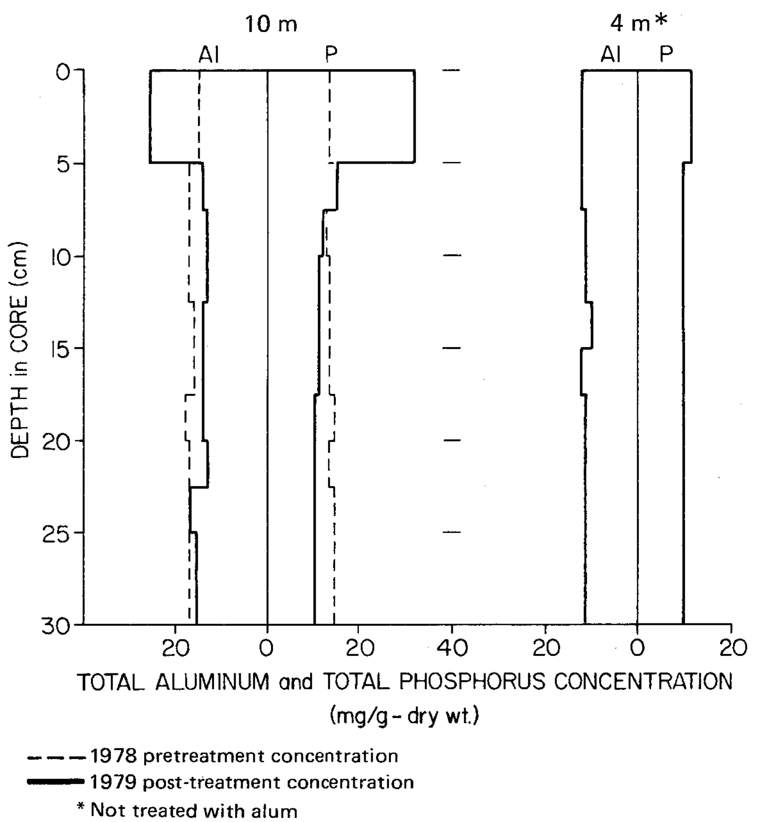


FIGURE 16. *Aluminum and phosphorus concentrations in sediment cores taken from Bullhead Lake at 10 m (1978), and at 10 and 4 m (1979).*

SUMMARY

Phosphorus control with alum is particularly helpful in stratified calcareous lakes where nutrient sources are limited and algae are the dominant primary producers. Alum could also be used in place of short-term controls (i.e., copper sulfate) to reduce undesired algae blooms in addition to its long-term effect on the internal recycling of phosphorus from the sediments.

This study showed alum injection to be an effective method for reducing in-lake phosphorus. Average epilimnetic summer total phosphorus (TP) and soluble reactive phosphorus (SRP) concentrations were reduced 38% and 92%, respectively, based on a comparison of summer samples in the four years before treatment and the four years after treatment. Likewise, the hypolimnetic summer TP and SRP average concentrations were reduced by 90% and 96%, respectively. The reduced phosphorus shifted the TN:TP ratio against the formation of a blue-green phytoplankton community. In its place the diatoms, green algae, flagellates, and rotifers flourished and provided a base for the food chain. The meroplanktonic *Chaoborus punctipennis* population increased dramatically, and the benthic chironomid community increased in both density and diversity. This increased production provided additional food organisms for fish and other higher life forms. No apparent acute or chronic detrimental effect was observed during the three-year project evaluation. The increased production of the invertebrates illustrates the safety of this technique. In addition, metalimnetic injection placed the concentrated alum below the euphotic zone and the immediate area of most biotic activity.

Phytoplankton biomass as expressed in the chlorophyll-*a* data was not significantly reduced. Some of the blue-green algae, dominant before treatment, were replaced by the more desirable green algae. However, the general lake appearance did improve with the absence of those blue-greens which have a tendency to clump.

The aluminum floc mixed into the sediment following treatment. Most of the floc was located in the 3-5 cm sediment depth and no distinctive "blanket" effect was observed in 1979 and 1980. Any SRP that might have been released into the hypolimnion from the overlying detritus was not significant. The floc still remained functional during 1982, four years after injections, and is anticipated to inactivate phosphorus for several additional years, as shown historically by the other alum-treated lakes.

LITERATURE CITED

AMERICAN PUBLIC HEALTH ASSOCIATION
1975. Standard methods for the examination of water and wastewater. 14th ed.

BRANDLOVA, J., Z. BRANDL, AND C. H. FERNANDO
1972. The cladocera of Ontario with remarks on some species and distribution. Can. J. Zool. 50:1373-1403.

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

CHENGALATH, R., C. H. FERNANDO, AND M. G. GEORGE
1971. The planktonic rotifera of Ontario with keys to genera and species. Univ. Waterloo, Ontario.

COOKE, G. D. AND R. H. KENNEDY
1978. Effects of a hypolimnion application of aluminum sulfate to a eutrophic lake. Verh. Int. Verin. Limnol. 20:486-89.

DEEVY, E. S. AND G. B. DEEVY
1971. The American species of *Eubosmina* Seligo. Limnol. and Oceanogr. 16:207-18.

DOMINIE, D. R.
1980. Hypolimnetic aluminum treatment of soft water Annabessacook Lake. pp. 417-23 *in* Restoration of lakes and inland waters. U. S. Environ. Prot. Agency 440/5-81-010.

DUNST, R. C., S. M. BORN, P. D. UTTORMARK, S. A. SMITH, S. A. NICHOLS, J. O. PETERSON, D. R. KNAUER, S. L. SERNS, D. R. WINTER, AND T. L. WIRTH
1974. Survey of lake rehabilitation techniques and experiences. Wis. Dep. Nat. Resour. Tech. Bull. No. 75. 179 pp.

EDMONDSON, W. T.
1959. Rotifers. pp. 420-94 *in* H. B. Ward and G. C. Whipple, eds. Fresh-water biology, 2nd ed. John Wiley & Sons, New York and London.

EISENREICH, S. J., R. T. BANNERMAN, AND D. E. ARMSTRONG
1975. Method for analysis of ortho and total phosphorus. Environ. Lett. 9:43-53.

FINK, C. R.
1969. Chemical and mineralogical characteristics of eutrophic lake sediments. Soil Sci. Soc. Am. Proc. 33:369-72.

FINDENEGG, I.
1965. Factors controlling primary productivity, especially with regard to water replenishment, stratification, and mixing. pp. 107-19 *in* Goldman, ed. Primary production in aquatic environments. Mem. Ist. Ital. Idrobiol. 18, Suppl. Univ. Calif. Press, Berkeley.

FUNK, W. H., H. L. GIBBONS, AND G. C. BAILEY
1980. Lake assessment in preparation for a multiphase restoration treatment. pp. 226-37 *in* Restoration of lakes and inland waters. Int. symp. on inland waters and lake restoration. September 8-12, 1980. U.S. Environ. Prot. Agency 440/5-81-010.

GARRISON, P. J. AND D. R. KNAUER
1983. Lake restoration: a five-year evaluation of the Mirror and Shadow lakes projects, Waupaca, Wisconsin. U. S. Environ. Prot. Agency R804687-01. Corvallis, Oregon. 100 pp.

GASPERINO, A. F., M. A. BECKWITH, G. R. KEIGUR, R. A. SOLTERO, D. G. NICHOLS, AND S. M. MIRES
1980. Medical Lake improvement project: a success story. pp.424-28. *in* Restoration of lakes and inland waters. U.S. Environ. Prot. Agency 440/5-81-010.

GORHAM, E.
1958. Observations on the formation and breakdown of the oxidized microzone at the mud surface of lakes. Limnol. and Oceanogr. 3:291-98.

HILSENHOFF, W. L.
1966. Ecology and population dynamics of *Chironomus plumosus* (Diptera: Chironomidae) in Lake Winnebago, Wisconsin. Ann. Entomol. Soc. Am. 60:1183-94.

HILSENHOFF, W. L. AND R. P. NARF
1968. Ecology of chironomidae, chaoboridae, and other benthos in fourteen Wisconsin lakes. Ann. Entomol. Soc. Am. 61:1173-81.

HOLDREN, G. C., JR.
1977. Factors affecting phosphorus release from lake sediments. Univ. Wis.-Madison. PhD Thesis.

HUTCHINSON, G. E.
1957. A treatise on limnology. pp. 727-53 *in* Chemistry of lakes. Vol. 1, Part 2. John Wiley and Sons, Inc.

JERNELOV, A.
1970. Phosphate reduction in lakes by precipitation with aluminum sulfate. Proc. Water Pollut. Res. Conf. Stockholm, Sweden.

JIRKA, A. M., M. J. CARTER, D. MAY, AND F. D. FULLER
1976. Ultramicro semiautomatic method for simultaneous determination of total kjeldahl nitrogen in wastewaters. Environ. Sci. and Tech. 10:1038-44.

KAMP-NIELSON, L.
1974. Mud-water exchange of phosphate and other ions in undisturbed sediment cores and factors affecting the exchange rates. Arch. Hydrobiol. 73:218-27.

LIND, O. T.
1974. Handbook of common methods in limnology. C. V. Mosby Co. 154 pp.

LORING, D. H. AND D. J. G. NOTA
1973. Morphology and sediments of the Gulf of St. Lawrence. J. Fish. Res. Board Can. Bull. 182.

LUND, J. W. G., C. KIPLING, AND E. D. LECREN
1958. The inverted microscope method of estimating algal numbers and statistical basis for estimations by counting. Hydrobiol. 11:143-70.

MCCAFFERTY, W. P.
1981. Aquatic entomology. Science Books Int. Boston. 448 pp.

MCQUEEN, D. J.
1969. Reduction of zooplankton standing stocks by predacious *Cyclops bicuspidata thomasi* in Marion Lake, British Columbia. J. Fish. Res. Board Can. 26:1605-18.
1970. Grazing rates and food selection in *Diaptomis oregonensis* (Copepoda) from Marion Lake, British Columbia. J. Fish. Res. Board Can. 27:13-20.

MERRITT, R. W. AND K. W. CUMMINS
1978. An introduction to the aquatic insects of North America. Kendall/Hunt, Dubuque. 441 pp.

MORTIMER, C. H.
1941. The exchange of dissolved substances between mud and waters in lakes. J. Ecol. 29:280-329.
1942. The exchange of dissolved substances between mud and waters in lakes. J. Ecol. 30:147-201.

NARF, R. P.
1978. An evaluation of past aluminum sulfate lake treatments: present sediment aluminum concentrations and benthic insect communities. Wis. Dep. Nat. Resour. Rep. to Off. Inland Lake Renewal. Memo. 12 pp.
1981. Influence of lake benthos on inorganic phosphate translocation from untreated and aluminum treated lake sediments. Wis. Dep. Nat. Resour. Rep. to Off. Inland Lake Renewal. Memo. 11 pp.

PASTOROK, R. A.
1980. Selection of prey by Chaoborus larvae: a review and new evidence of behavioral flexibility. pp. 538-54 *in* Kerfoot, W. C., ed. Evolution and ecology of zooplankton communities. Special Symp. Vol. 3. Am. Soc. Limnol. and Oceanogr. Univ. Press New England.
1981. Prey vulnerability and size selection by *Chaoborus* larvae. Ecol. 62:1311-24.

PATRICK, R. AND C. REIMER
1966. The diatoms of the United States. Vol. I. Acad. Nat. Sci. Phila. Monogr. No. 13. 688 pp.
1975. Diatoms of the United States. Vol. 2. Part 1. Acad. Nat. Sci. Phila. Monogr. No. 13. 213 pp.

PETERSON, J. O., J. P. WALL, T. L. WIRTH, AND S. M. BORN
1973. Nutrient inactivation by chemical precipitation at Horseshoe Lake, Wisconsin. Wis. Dep. Nat. Resour. Tech. Bull. No. 62. 18 pp.

PRESCOTT, G. W.
1962. Algae of the western great lakes area. Wm. C. Brown Co., Inc. Dubuque. 977 pp.

SAKAMOTO, M.
1966. Primary production by phytoplankton community in some Japanese lakes, and its dependence on lake depth. Arch. Hydrobiol. 62:1-28.

SKUJA, H.
1948. Taxonomie des phytoplanktons einiger seen in Uppland, Schweden. Symb. Bot. Upsal. 9:1-339.

SMITH, G. M.
1920. Phytoplankton of the inland lakes of Wisconsin. Part I. Wis. Geol. and Nat. Hist. Surv. 243 pp.
1924. Phytoplankton of the inland lakes of Wisconsin. Part II (Desmidiaceae). Wis. Geol. and Nat. Hist. Surv. 227 pp.
1950. The freshwater algae of the United States. McGraw-Hill. New York. 719 pp.

SMITH, K. AND C. H. FERNANDO
1978. A guide to the freshwater calonoid and cyclopoid copepod crustacea of Ontario. Univ. Waterloo. Biol. Ser. No. 18. Ontario.

SMITH, V. H.
1983. Low nitrogen to phosphorus ratios favor dominance by blue-green algae in lake phytoplankton. Science 221:669-71.

STAUFFER, R. E.
1981. Sampling strategies for estimating the magnitude and importance of internal phosphorus supplies in lakes. U. S. Environ. Prot. Agency EPA 600/3-81-015.

STRICKLAND, J. D. H. AND T. R. PARSONS
1968. A practical handbook of sea water analysis. J. Fish. Res. Board Can. Bull. No. 167. 311 pp.

TORKE, B.
1976. A key to the identification of the cyclopoid copepods of Wisconsin, with notes on their distribution and ecology. Wis. Dep. Nat. Resour. Res. Rep. No. 88. 16 pp.

U. S. ENVIRONMENTAL PROTECTION AGENCY
1973. Biological field and laboratory methods for measuring the quality of surface waters and effluents. U. S. Environ. Prot. Agency 670/4-73-001.
1974. Methods for chemical analysis of water and wastes. U. S. Environ. Prot. Agency 625/6-74-003.
1979. Methods for chemical analysis of water and wastes. U. S. Environ. Prot. Agency 600/4-79-020.

U. S. GEOLOGICAL SURVEY
1970. Techniques of water resources investigations. Book No. 5. Chap. A-1.

VOLLENWEIDER, R. A.
1968. Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in eutrophication. Organ. Econ. Coop. Dev. (Paris) Tech. Rep. DAS/DSI/68.2F P. 192.

WEBER, C. I.
1971. A guide to the common diatoms at water surveillance system stations. U. S. Eviron. Prot. Agency. Natl. Environ. Res. Cent. Cincinnati, Ohio.

WETZEL, R. G.
1983. Limnology. 2nd ed. CBS College Publ. 767 pp.

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