

# Effects of Phosphorus Reduction on Water Quality: Comparison of Alum-Treated and Untreated Portions of a Hypereutrophic Lake

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## ABSTRACT

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The effects of a reduction in total phosphorus concentration on the water quality and plankton community structure in a 86-ha hypereutrophic sandpit lake with high internal phosphorus loading were assessed by dosing an isolated 4.6-ha section of the lake with 34,065 L (dose = 10 mg Al · L<sup>-1</sup>) of liquid aluminum sulfate (alum). During the three summers following treatment, hypolimnetic total dissolved phosphorus, epilimnetic total phosphorus, and epilimnetic total nitrogen were decreased by 97%, 74%, and 36%, respectively, in the treated section. Secchi depth was 134% greater in the treated area. Alum treatment also increased the volume of usable fish habitat by 22%, as the depth of the 3.0-mg/l dissolved oxygen isocline was 52% deeper in the treated portion than in the untreated portion. Total phytoplankton biovolume decreased by 40% and chl *a* concentration by 65% in the treated area. Although cyanophytes continued to dominate in the treated area, there was a shift in relative abundance from cyanophytes to bacillariophytes and chlorophytes, especially during the second summer after treatment. By the second post-treatment summer, daphnid biomass increased substantially and there was a subtle shift in relative abundance from copepods to daphnids. Overall, alum was extremely effective in controlling sediment phosphorus release rates and lowering water column phosphorus concentrations and thus improving water clarity, reducing phytoplankton biomass, shifting phytoplankton species composition from cyanophyte dominance toward bacillariophytes and chlorophytes, increasing daphnid biomass, and increasing usable fish habitat.

**Key Words:** phosphorus inactivation, aluminum sulfate, lake restoration, phosphorus, phytoplankton, zooplankton.

Nutrient loading has accelerated the eutrophication of many lakes and reservoirs throughout the world. Water quality problems associated with eutrophication are well known, including impaired recreational use, nuisance algal blooms, low transparency, noxious odors, severe dissolved oxygen depletion, fish kills, and the degradation of potable water supplies. Phosphorus (P) is often the key aquatic biomass-limiting nutrient (Vollenweider 1968, Likens 1972, Schindler 1978) and thus is the primary macro-nutrient targeted for reduction by most lake restoration approaches. Lower water column P has been associated with improvements in water quality such as decreased phytoplankton biomass, phytoplankton species abundance shifts away from cyanophyte dominance, and greater transparency (e.g., Edmondson and Lehman 1981, Cooke et al. 1982, Welch and Schriever 1994, Holz and Hoagland 1996), as well as changes in zooplankton biomass and species composition (Holz and Hoagland 1996).

Sediment P release (internal loading) can be an important source of P and can maintain high water column P concentrations, even in the absence of significant external loading (Marsden 1989). Phosphorus inactivants such as aluminum sulfate (alum) have been used as potentially long-term methods of controlling sediment P release (Cooke et al. 1982, Cooke et al. 1993, Welch and Schriever 1994). Alum is added to the water column to form aluminum phosphate and a colloidal hydroxide floc which binds to certain P fractions. The floc settles through the water column (removing large amounts of particulates) and briefly forms a cover over the sediments. The floc then consolidates with the sediments where it can continue to sorb and retain P for over 20 years (Welch and Schriever 1994, Welch and Cooke 1995). P which would otherwise diffuse into the water column is sorbed, thus reducing internal loading and water column P concentration. Toxic species of aluminum remain extremely low within a pH range of 6 to 8, but increase substantially when

pH is below 6. A short-term pH reduction is the most obvious side effect of an alum treatment and therefore excludes alum addition as an restoration option in many soft water lakes. In some cases, the pH reduction can be mitigated by adding alum and sodium aluminate simultaneously in less pH-sensitive water bodies (Cooke et al. 1993).

Most whole lake restoration studies, including alum treatments, have been limited to before-and-after observational data. However, the design of the present study differed fundamentally from other P inactivation studies in that it compares the effects of alum addition in a completely isolated portion of a lake to an untreated portion of the same lake. This design allowed detailed, although unreplicated, treated versus untreated comparisons of the physiochemical conditions and planktonic community structure in a hypereutrophic lake over a 2.5-year period.

## Site Description

The experiment was conducted in Lake Leba (41°26'N, 96°31'W) near Fremont, NE, USA (Fig. 1). Lake Leba is a privately owned, 86-ha sandpit lake located along the Platte River, with a mean depth of 4.0 m and a maximum depth of 9.5 m. The sand bottom lake was created by gravel mining operations in the 1960s and is groundwater-fed with no point-source

surface water inputs. Initially, Lake Leba was pristine with the sandy sediments containing low amounts of organic matter and periods of notable water clarity (>4 m). In the 1980s, a dramatic decrease in the water quality coincided with high P flood waters entering the lake from the nearby Platte River.

During the two summers prior to alum addition, mean epilimnetic P concentration was  $159 \mu\text{g} \cdot \text{L}^{-1}$ , indicative of a hypereutrophic lake (Vollenweider 1968). Secchi depths were <50 cm and total alkalinity ranged between 105 and 200 mg as  $\text{CaCO}_3 \cdot \text{L}^{-1}$ . The lake is susceptible to nuisance cyanophyte blooms, with a predominance of *Oscillatoria* spp. and *Anabaena* spp. beginning in the early spring and persisting until fall turnover. Lake Leba is also typically anoxic in the lower two-thirds of the water column during summer stratification and the sandy sediments are now covered by an unconsolidated, 15-cm layer of organic material. The lake has an Osgood Index ( $\bar{z}/A_o^{0.5}$ ) of 4.3, which indicates that Lake Leba may be prone to deep mixing events during stratification (Osgood 1988, Cooke et al. 1993). The fish community is a mixed assemblage of largemouth bass, bluegill, walleye, catfish, white perch, and carp.

## Methods

A 4.6-ha portion of Lake Leba (Fig. 1) was isolated from the main portion of the lake with two sand embankments in April 1994 to allow alum treated versus untreated comparisons of water quality and plankton community structure. The isolated section ( $V = 1.96 \times 10^8 \text{ L}$ ,  $\bar{z} = 4.2 \text{ m}$ ,  $z_m = 9.0 \text{ m}$ , Osgood Index = 19.8) was dosed with 34,065 L (dose =  $10 \text{ mg Al} \cdot \text{L}^{-1}$ ) of liquid alum (8.3%  $\text{Al}_2\text{O}_3$ ; General Chemical, East St. Louis, IL) over a 3-day period in May 1994 (5/23-5/25) using a modified barge similar to that described by Kennedy and Cooke (1982). Alum dosage was determined by titrating water samples from various depths with liquid alum to a pH endpoint of 6.0. The dose for each depth interval was determined and the total dose to the lake was calculated by summing the individual depth interval doses (Cooke et al. 1993). Approximately 80% of the alum was injected 2 m below the surface through a PVC manifold, while 20% was applied to the surface using a fire hose to remove suspended P (e.g., algae, detritus) and to adequately dose the littoral zone.

Three treated and three untreated sites of similar depth ( $z \approx 5.25 \text{ m}$ ) were sampled each month during stratification and twice during the winter for 2.5 years following alum addition. Water samples for the analysis of total P (TP), total dissolved P (TDP), total nitrogen

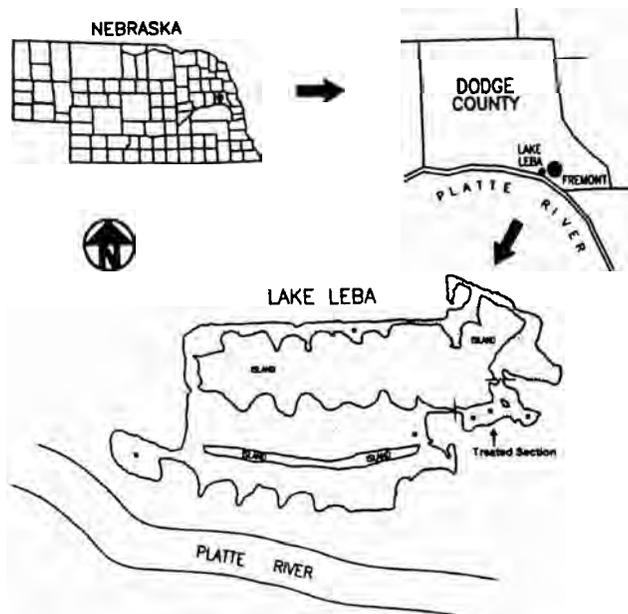


Figure 1.—Map of Lake Leba. Solid lines to the north and west of the treated portion indicate the location of two sand embankments used to isolate a 4.6-ha portion of Lake Leba. The solid square symbols indicate sampling locations.

(TN), and pH were collected in the epilimnion ( $z = 0.5$  m) and hypolimnion ( $z = 4$  m) with a vertical Van Dorn bottle. pH was measured in the field, and water chemistry samples were transported to the laboratory on ice in acid-cleaned, polyethylene amber bottles. Samples were analyzed using the following methods: (i) for TP (unfiltered sample) and TDP (sample filtered through 0.45- $\mu$ m Millipore membrane filter prior to digestion), the ascorbic acid method (Lind 1985), (ii) for TN, the alkaline potassium persulfate digestion (D'Elia et al. 1977) and UV absorption method (APHA 1985), and (iii) for pH, a Fisher Scientific Accumet pH meter (model 1003). In addition to Secchi depth, dissolved oxygen and temperature were measured at each site on each sampling date with a YSI (model 57) meter and probe. pH means were calculated based on the log inverse of each value.

Unfiltered phytoplankton samples (1 L) and chlorophyll *a* (chl *a*) samples were obtained from an integrated sample of the epilimnion. The integrated sample was collected by lowering a PVC sampling tube (5.08 cm i.d.) fitted with a one-way valve to the bottom of the epilimnion. The sample was preserved in 1% Lugol's solution and allowed to settle overnight in sedimentation chambers. Thirty random fields of view were counted with a Nikon Diaphot inverted microscope at 400x, and the algae were identified to the lowest practicable taxon based on preliminary examinations of unfixed samples. Twenty-five cells, chains, or colonies of each taxonomic group were measured to determine average lengths and widths. Biovolumes were then calculated using combinations of formulae of simple geometric shapes.

Zooplankton samples were collected at each site by vertical tows with a 35- $\mu$ m plankton net. The samples were preserved in 5% sucrose-formalin and 5 to 10% of each sample was counted in a Sedgwick-Rafter cell with a Nikon Labophot-2 compound microscope at 40 or 100x. Fifteen to 25 individuals of each taxonomic group were measured to determine average body lengths and widths. Zooplankton dry weight was then calculated using the length-weight relationships of Bottrell et al. (1976). Although physical and chemical sampling continued through October 1996, plankton sampling was discontinued in May 1996.

Important differences between the two sections can be identified by considering the two lake sections as separate populations (but not treatment replicates). Thus, a one-way ANOVA was performed on all 1994, 1995, and 1996 physical, chemical, and biological variables to test for differences between the two sections during summer stratification (i.e., May-October). The analysis was completed using SAS/STAT version 6 (SAS Institute Inc. 1989) and significance was inferred at  $P \leq 0.05$ .

## Results

### *Physical and chemical conditions*

The physical and chemical environments in the two lake portions differed greatly during the three summer stratification periods following alum addition. Mean hypolimnetic TDP concentration (an indicator of sediment-P release) was 97% lower in the treated portion, and the lower sediment-P release ultimately reduced P in the water column, as mean epilimnetic TP was 74% lower in the treated section over the summer stratification periods of 1994-96 (Table 1, Fig. 2). These reductions in epilimnetic P shifted Lake Leba from a highly hypereutrophic to a moderately eutrophic state (Vollenweider 1968). The greatest difference in epilimnetic TP occurred during the summer of 1996 when the treated mean was 82% lower than the untreated mean (Table 1, Fig. 2). Averaged over all three summers, epilimnetic TN was 36% lower in the treated section and water clarity improved as Secchi depth increased by 134% from an untreated mean of 69 cm to a treated mean depth of 162 cm (Table 1, Fig. 2). The most substantial difference in Secchi depth occurred in 1996 when the summer treated mean was 335% greater than the untreated mean (Table 1, Fig. 2).

Mean epilimnetic pH was 6.8 in the alum treated section and 8.9 in the untreated section 2 days after alum application, but there was essentially no difference between the sections within 2 weeks of the application (treated = 8.6, untreated = 8.8). Averaged over the three summers, epilimnetic pH in the treated and untreated sections did not differ (Table 1, Fig. 2). The volume of usable fish habitat increased by 22% as the mean depth of water containing  $\geq 3.0$  mg  $\cdot$  L<sup>-1</sup> of dissolved oxygen was 52% deeper in the treated portion than in the untreated portion (Table 1, Fig. 2).

### *Phytoplankton composition*

Alum addition reduced mean total phytoplankton biovolume by 40% and chl *a* concentration by 65% over the summer stratification periods of 1994-95 (phytoplankton and chl *a* samples were not collected in 1996) (Table 2, Fig. 3). Mean cyanophyte biovolume (mainly *Anabaena* spp., *Coelosphaerium* sp., and *Oscillatoria* spp.) was 58% lower in the treated section (Table 2, Fig. 3). However, mean bacillariophyte biovolume (mainly *Asterionella formosa* Hass., *Cocconeis* sp., *Cyclotella meneghiniana* Kütz., *Fragilaria* spp. *Synedra* spp.) was 361% higher and mean chlorophyte biovolume (mainly *Chlorella vulgaris* Beij., *Closterium*

**Table 1.**—Summer means and % change in limnological variables in the alum treated and untreated portions of Lake Leba. Treated portion means followed by an asterisk differ significantly from the untreated mean ( $\alpha = 0.05$ ).

	1994-96			1994		
	Untreated $\bar{x}$	Treated $\bar{x}$	% Change	Untreated $\bar{x}$	Treated $\bar{x}$	% Change
Hypolimnetic TDP ( $\mu\text{g}\cdot\text{L}^{-1}$ )	534	16*	-97	694	11*	-98
Epilimnetic TP ( $\mu\text{g}\cdot\text{L}^{-1}$ )	143	38*	-74	128	39*	-70
Epilimnetic TN ( $\mu\text{g}\cdot\text{L}^{-1}$ )	936	595*	-36	930	511*	-45
Secchi Depth (cm)	69	162*	+134	70	155*	+123
Depth of 3 $\text{mg}\cdot\text{L}^{-1}$ D.O. Threshold (m)	2.88	4.38*	+52	3.18	4.31*	+36
Epilimnetic pH	8.70	8.37	0	8.88	8.02*	-10

	1995			1996		
	Untreated $\bar{x}$	Treated $\bar{x}$	% Change	Untreated $\bar{x}$	Treated $\bar{x}$	% Change
Hypolimnetic TDP ( $\mu\text{g}\cdot\text{L}^{-1}$ )	370	28*	-93	379	7*	-98
Epilimnetic TP ( $\mu\text{g}\cdot\text{L}^{-1}$ )	148	42*	-71	167	30*	-82
Epilimnetic TN ( $\mu\text{g}\cdot\text{L}^{-1}$ )	916	614*	-33	973	676*	-31
Secchi Depth (cm)	92	167*	+82	40	173*	+335
Depth of 3 $\text{mg}\cdot\text{L}^{-1}$ D.O. Threshold (m)	2.42	4.50*	+86	2.75	4.38*	+59
Epilimnetic pH	8.53	8.67	0	8.53	8.67	0

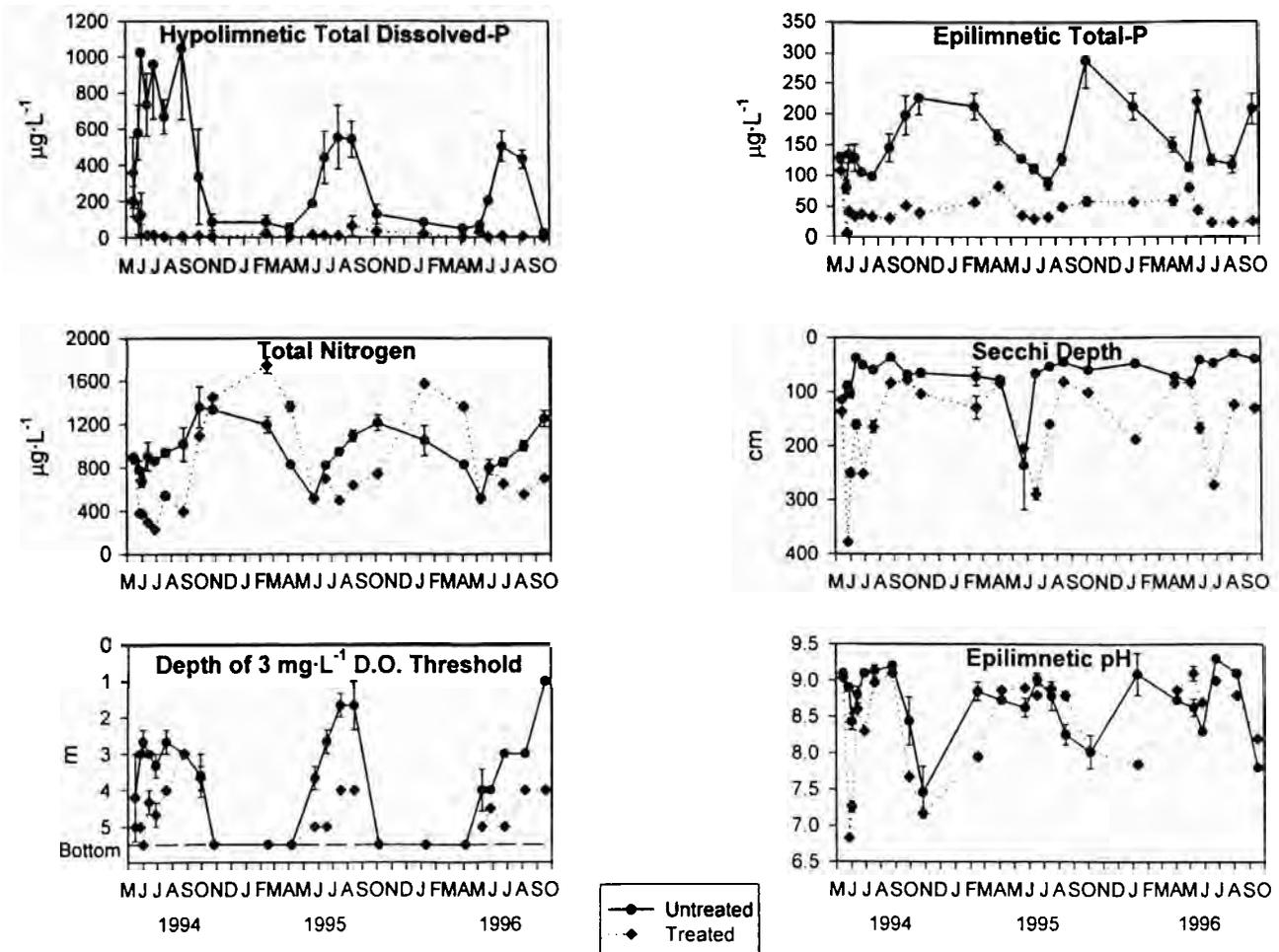
**Figure 2.**—Mean responses of chemical and physical variables in the alum treated and untreated portions of Lake Leba ( $\pm$ SE). Alum addition occurred 5/23-5/25/94. The first data point represents pre-treatment conditions and subsequent points represent post-treatment conditions.

Table 2.—Mean summer biovolume ( $\times 10^6 \mu\text{m}^3 \cdot \text{L}^{-1}$ ) and % change in biovolume for all phytoplankton and the major algal groups in the alum treated and untreated portions of Lake Leba. Mean relative abundance (%) and % change in relative abundance are shown in parentheses. Treated portion means followed by an asterisk differ significantly from the untreated mean ( $\alpha = 0.05$ ).

	1994-95			1994			1995		
	Untreated	Treated	% Change	Untreated	Treated	% Change	Untreated	Treated	% Change
Total Phytoplankton	126.8	75.8*	-40	129.5	66.2*	-49	122.7	94.2*	-23
Chl <i>a</i> ( $\mu\text{g} \cdot \text{L}^{-1}$ )	65.2	22.5*	-65	66.8	16.8*	-75	62.6	30.5*	-51
Cyanophytes	119.2 (89)	49.7* (69*)	-58 (-23)	123.2 (89)	51.7* (79*)	-58 (-12)	47.0* (55*)	-57 (-38)	
Bacillariophytes	3.8 (5)	17.5* (17*)	+361 (+215)	1.2 (3.6)	8.2* (7.5)	+583 (0)	8.0 (8)	30.5* (30*)	+281 (+276)
Cryptophytes	3.3 (4.6)	2.4 (7.2)	0 (0)	2.5 (5.2)	2.1 (10.0)	0 (0)	4.7 (3.7)	3.0 (3.2)	0 (0)
Chlorophytes	0.3 (0.9)	5.0* (5.8*)	+1567 (+544)	0.2 (1.1)	0.5* (3.2*)	+150 (+191)	0.3 (0.4)	11.4* (9.5*)	+3700 (+2275)

*parvulum* Näg., *Coelastrum microporum* Näg., *Gleocystis* sp., *Oocystis borgei* Snow, *Staurastrum* sp.) was 1567% higher in the treated area (Table 2, Fig. 3).

Although cyanophytes continued to dominate both sections of the lake, there was a shift away from cyanophytes to bacillariophytes and chlorophytes over the 1994-95 stratification periods as indicated by the lower mean cyanophyte relative abundance in the treated area (69%) compared to the untreated area (89%) (Table 2, Fig. 4). While the relative abundance of the cyanophytes decreased, the mean bacillariophyte and chlorophyte relative abundances were higher in the treated section by 215% and 544%, respectively.

The most pronounced shift in phytoplankton community composition occurred in the second summer after alum addition (i.e., 1995) when mean cyanophyte, bacillariophyte, and chlorophyte relative abundances in the treated section were 55%, 30%, and 10%, respectively. In the untreated section, mean cyanophyte, bacillariophyte, and chlorophyte relative abundances were 88%, 8%, and 0.4%, respectively. These means indicate that cyanophyte relative abundance was 38% lower, but bacillariophyte and chlorophyte relative abundances were higher by 276% and 2275% in the treated area during the summer of 1995 (Table 2, Fig. 4).

### Zooplankton composition

Mean total zooplankton biomass and mean rotifer biomass (predominantly species of the following genera: *Brachionus*, *Euchlanis*, *Filinia*, *Keratella*, *Lecane*, *Polyarthra*, *Trichocera*) did not differ between the two lake sections over the 1994-95 summer stratification periods (zooplankton samples were not collected in 1996) (Table 3, Fig. 5). The mean biomass of small cladocerans (mainly *Bosmina longirostris*, with some *Chydorus sphaericus*) and cyclopoid (mainly *Diacyclops thomasi* and *Mesocyclops edax*) were 526% and 116% higher in the treated section, but mean *Daphnia* spp. (*D. galeata mendotae* and *D. pulex*), calanoid (*Aglaodiaptomus clavipes*, *Leptodiaptomus siciloides*, and *Skistodiaptomus pygmaeus*), or juvenile copepod biomass (nauplii and copepodids) did not differ between the lake sections over 1994-95 (Table 3, Fig. 5).

The zooplankton community response averaged over both post-treatment summers is misleading because the 1994 community was substantially different from the 1995 community. During the summer of 1994, total zooplankton biomass was 55% lower, and rotifers comprised 73% of the total zooplankton biomass in the treated section compared to 89% in the untreated section (Table 3, Figs. 5 and 6). In addition to the 67% reduction in rotifer biomass, mean *Daphnia* spp. biomass

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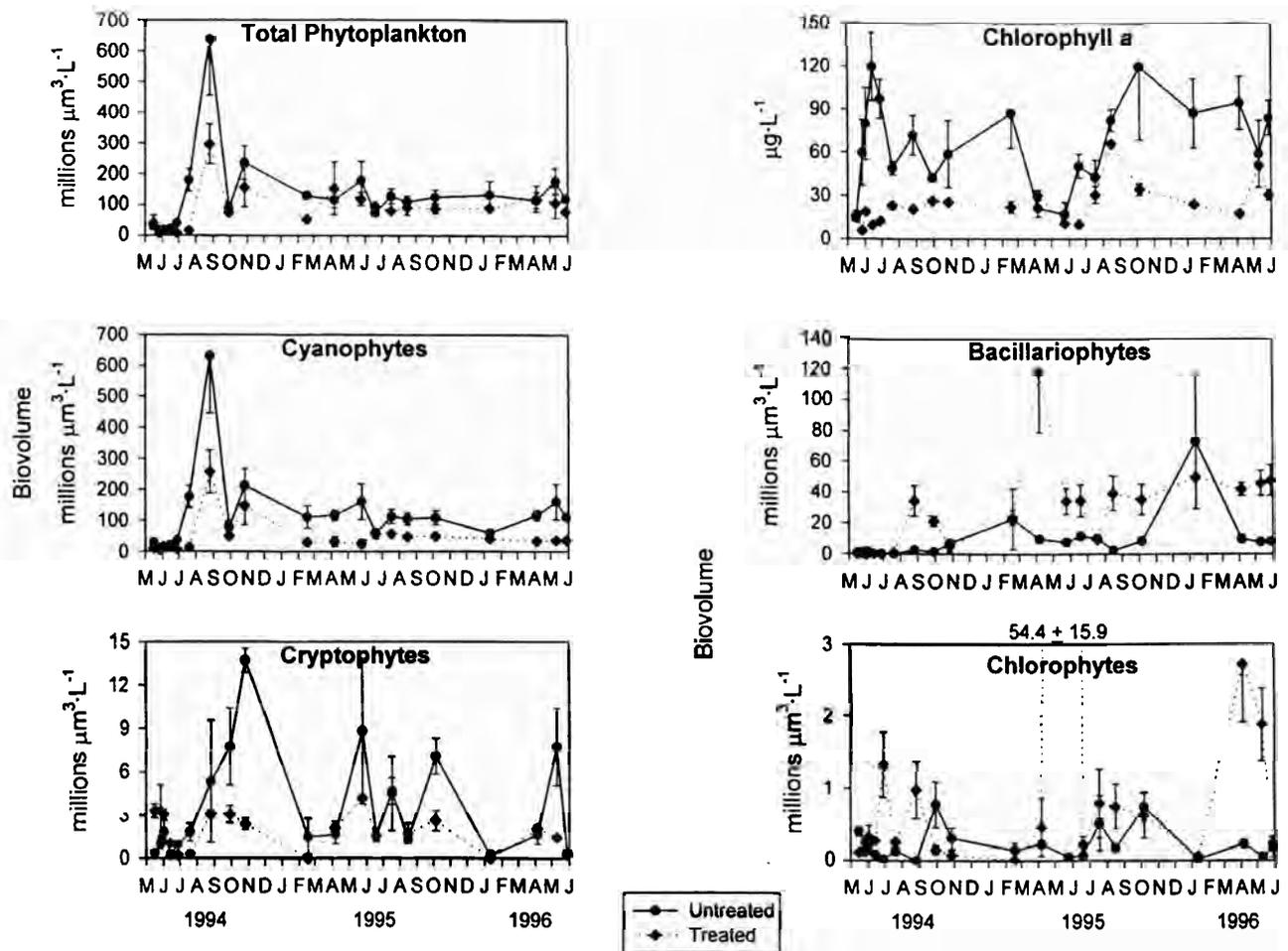


Figure 3.—Mean responses of phytoplankton biovolume and chlorophyll *a* concentrations in the alum treated and untreated portions of Lake Leba ( $\pm$  SE). Alum addition occurred 5/23-5/25/94. The first data point represents pre-treatment conditions and subsequent points represent post-treatment conditions.

was 85% lower, mean small-bodied daphnid biomass was 596% higher, and mean cyclopid biomass was 84% higher in the treated portion during the summer of 1994 (Table 3, Fig. 5).

Although rotifers continued to dominate both sections of the lake in 1994, there was a shift from rotifers and *Daphnia* spp. to smaller-bodied daphnids, cyclopid, calanoids, and juvenile copepods as indicated by the lower mean rotifer (74%) and mean *Daphnia* spp. (2%) relative abundances in the treated area versus untreated area (89% and 6%, respectively) (Fig. 6). As the relative abundance of the rotifers and *Daphnia* spp. decreased, the mean relative abundances of the small-bodied daphnids, cyclopid, and calanoids were higher in the treated section by 760%, 426%, and 129%, respectively. The mean small-bodied daphnid, cyclopid, and calanoid relative abundances were 9%, 10%, and 2% in the treated area compared to 1%, 2%, and 0.7% in the untreated area (Table 3, Fig. 6).

Zooplankton community composition was quite different in the summer of 1995. In the treated section, total zooplankton and rotifer biomass did not differ from the untreated area, but *Daphnia* spp. biomass was considerably higher than both the 1994 treated mean and the 1995 untreated mean. Small-bodied cladoceran biomass dropped between 1994 and 1995, but the 1995 treated mean was 164% higher than the 1995 untreated mean. Cyclopid biomass in the treated area increased from 1994 to 1995, and the treated mean was 147% greater than in the untreated section during 1995. Mean calanoid biomass decreased by 42% in the treated portion during the summer of 1995 (Table 3, Fig. 5).

During the summer of 1995, *Daphnia* spp. and small-bodied daphnid relative abundances were higher by 84% and 300%, and calanoid relative abundance was 86% lower in the treated area. Although cyclopid biomass was higher in the treated area, their relative abundance was 71% lower during 1995 (Table 3, Fig. 6).

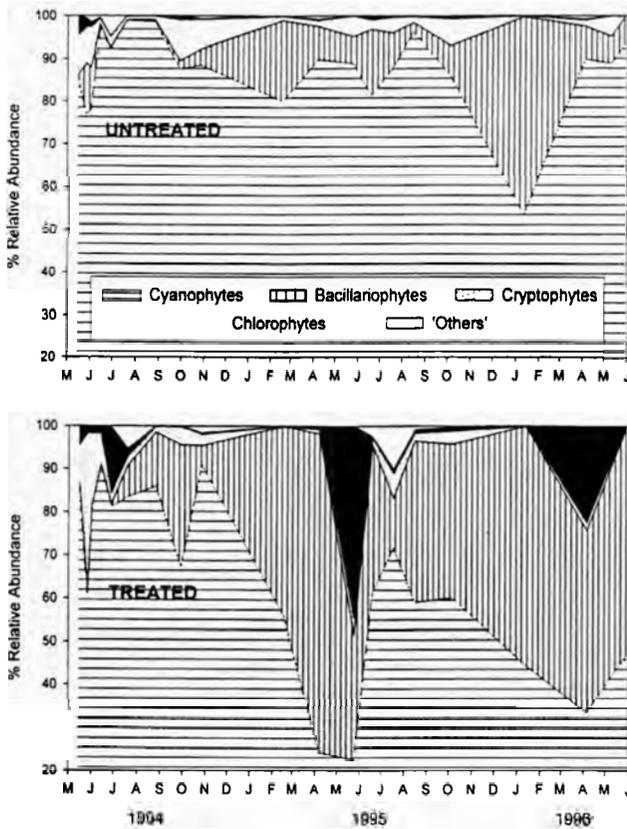


Figure 4.—Mean relative abundances for the major algal groups in the alum treated and untreated portions of Lake Leba. Alum addition occurred 5/23-5/25/94.

## Discussion

### *Physical and chemical conditions*

Alum addition was extremely effective in controlling sediment P release at Lake Leba as indicated by the remarkably low hypolimnetic TDP concentrations in the treated portion relative to the untreated portion (Fig. 2). Hypolimnetic TDP was 97% lower in the treated section over the three post-treatment summers, and did not increase through sustained periods of anoxia and multiple mixing events. Most other aluminum sulfate treatments have also successfully reduced sediment P release (Cooke et al. 1993). For example, alum additions to Mirror and Shadow Lakes, WI resulted in internal loading rates well below the pretreatment rates for at least 12 years, despite the deposition of 8 to 10 cm of sediments on the floc layer (Cooke et al. 1993). Anaerobic sediment P release rates from the intact sediment cores of West Twin Lake, OH, demonstrated that alum was still curtailing release rates 14 years after application (Cooke et al. 1993). Internal P loading was 50% lower than the pre-treatment loading

	1994-95			1994			1995		
	Untreated $\bar{x}$	Treated $\bar{x}$	% Change	Untreated $\bar{x}$	Treated $\bar{x}$	% Change	Untreated $\bar{x}$	Treated $\bar{x}$	% Change
Total Zooplankton	696.9	624.8	0	527.0	235.9*	-55	968.7	1169.1	0
Rotifers	653.4 (83)	546.3 (80)	0 (0)	487.5 (89)	159.3* (73*)	-67 (-17)	918.8 (75)	1088.1 (90)	0 (0)
<i>Daphnia</i> spp.	15.1 (4.1)	11.7 (2.5)	0 (0)	18.9 (5.5)	2.8* (1.8*)	-85 (-67)	9.1 (1.9)	24.0* (3.5*)	+164 (+84)
Small cladocerans	2.2 (0.6)	13.9* (5.2*)	+526 (+767)	3.1 (1.0)	21.9* (8.6*)	+596 (+760)	0.7 (0.1)	2.8* (0.4*)	+272 (+400)
Cyclopoids	7.3 (4.0)	15.8* (6.8)	+116 (0)	6.6 (1.9)	12.1* (10.0*)	+84 (+426)	8.5 (7.5)	21.0* (2.2*)	+147 (-71)
Calanoids	6.4 (2.9)	4.6 (1.3*)	0 (-55)	2.7 (0.7)	2.8 (1.6*)	0 (+129)	12.3 (6.4)	7.2* (0.9*)	-42 (-86)
Juvenile copepods	12.4 (5.1)	15.0 (4.3)	0 (0)	8.2 (2.5)	7.1 (4.9)	0 (0)	19.2 (9.3)	26.1 (3.4)	0 (0)

Table 3.—Mean summer biomass ( $\mu\text{g L}^{-1}$ ) and % change in biomass for all zooplankton and the major zooplankton groups in the alum treated and untreated portions of Lake Leba. Mean relative abundance (%) and % change in relative abundance are shown in parentheses. Treated portion means follow by an asterisk differ significantly from the untreated mean ( $\alpha = 1.05$ ).

rate 21 years after alum addition in Horseshoe Lake, WI, and 90% lower after 19 years in Snake Lake, WI (Welch and Cooke 1995). A sodium aluminate/alum treatment in a moderately alkaline lake (Lake Morey, VT) resulted in an 80% to 90% reduction in hypolimnetic P that has remained effective for at least 7 years (Cooke et al. 1993, Welch and Cooke 1995). Even in

shallow, well-mixed lakes, alum controlled internal loading by 40% to 82% for 6 to 10 years in Washington (Welch and Cooke 1995).

Despite the clear success in controlling internal P loading, alum has less consistently reduced epilimnetic P due to limited vertical entrainment of hypolimnetic P, inadequate control of external loading, substantial

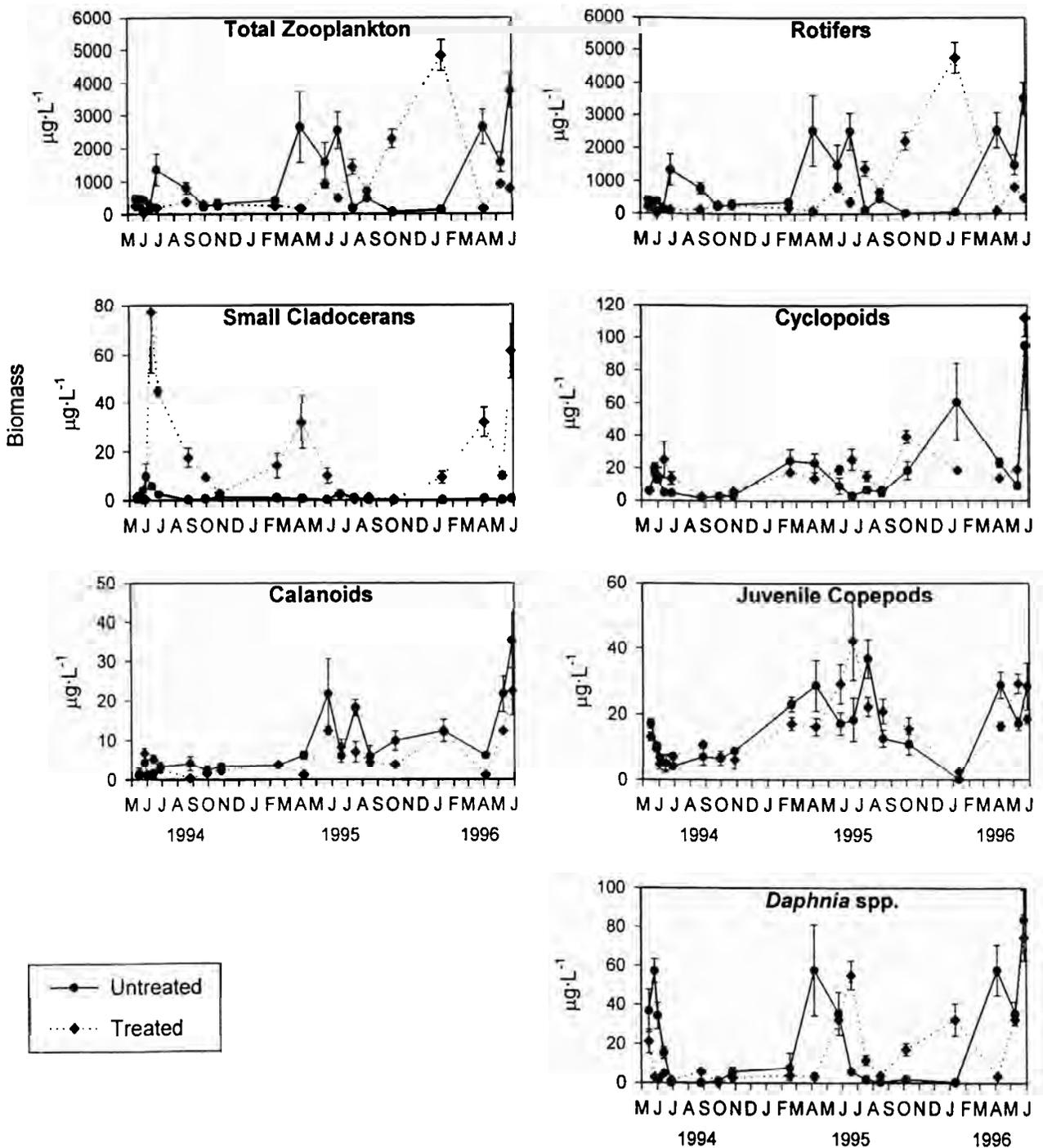


Figure 5.—Mean responses of zooplankton biomass in the alum treated and untreated sections of Lake Leba ( $\pm$ SE). Alum addition occurred 5/23-5/25/94. The first data point represents pre-treatment conditions and subsequent points represent post-treatment conditions.

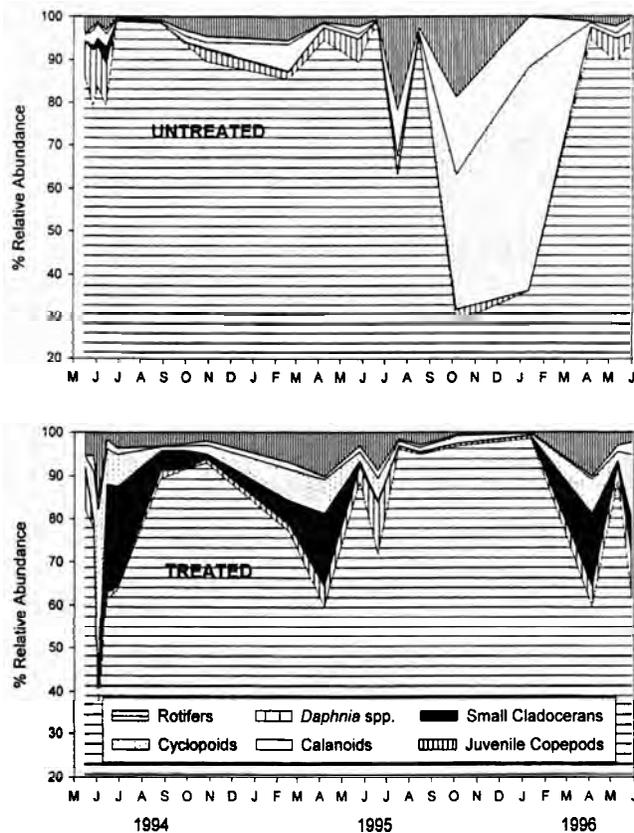


Figure 6.—Mean relative abundances for the major zooplankton groups in the alum treated and untreated portions of Lake Leba. Alum addition occurred 5/23-5/25/94.

P loading from oxic shallow water sediments, and significant P recycling by macrophytes (Welch et al. 1988, Cooke et al. 1993). For example, the control of sediment P release in West Twin Lake had only a minor impact on epilimnetic P, probably due to low vertical transport of hypolimnetic P to the upper waters (Cooke et al. 1993). In other alum-treated lakes (e.g., Mirror, Shadow, Horseshoe, and Snake Lakes), the effects of lower internal loading cannot be entirely separated from the effects of lower external loading (i.e., nutrient diversion) (Cooke et al. 1993, Welch and Cooke 1995). However, in Lake Morey the control of internal P loading reduced epilimnetic P from about  $13 \mu\text{g} \cdot \text{L}^{-1}$  to  $4\text{--}6 \mu\text{g} \cdot \text{L}^{-1}$  over the 7 post-treatment years (Welch and Cooke 1995). In shallow lakes, where internal loading reductions may have a more direct effect on epilimnetic P than in deeper lakes, Welch and Cooke (1995) reported that in six successful treatments, whole-lake TP was 29% to 75% lower than pre-treatment TP 5 to 8 years after alum addition to Washington lakes. Alum treatment in two other shallow Washington lakes (Wapato and South Pattison Lakes) were unsuccessful due to the effects of macrophytes and decreased dilution (Welch and Schriever 1994).

P loading from the shallow sediments and P recycling by macrophytes is presumably unimportant in Lake Leba, because the gravel mining operation which formed Lake Leba resulted in a narrow littoral zone and consequently low macrophyte densities. Groundwater flow in shallow areas of a lake may force the additional release of P from the sediments (Cooke et al. 1993); P content of the groundwater near Lake Leba is 5 to  $15 \mu\text{g} \cdot \text{L}^{-1}$  (Holz and Hoagland, unpub. data). Despite the potential impact of groundwater flow and an Osgood Index of 19.8 which predicts that deep water mixing events are uncommon (Osgood 1988, Cooke et al. 1993), internal loading from hypolimnetic sediments appears to be the major source of P in the treated area's euphotic zone, and the 97% reduction in hypolimnetic P ultimately accounted for much of the

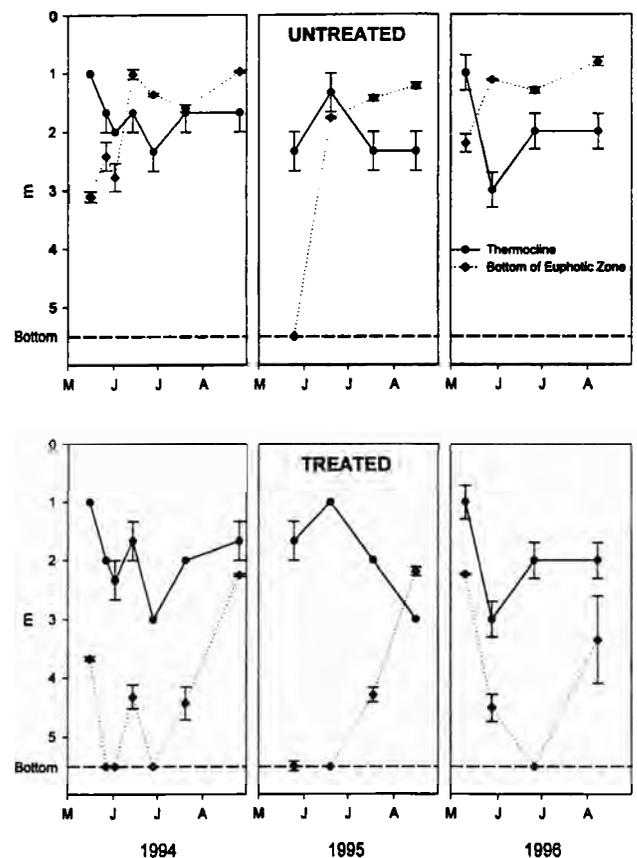


Figure 7.—Mean summer thermocline depths and bottom of the euphotic zone in the alum treated and untreated portions of Lake Leba during stratification ( $\pm$  SE). Alum addition occurred 5/23-5/25/94. The first data point represents pre-treatment conditions and subsequent points represent post-treatment conditions. The thermocline is defined as the plane of maximum temperature change (Wetzel 1983), and its location was estimated by depth where temperature changes were greater than  $1^\circ\text{C}$  per meter (Horne and Goldman 1994). The bottom of the euphotic zone ( $z_{0.01}$ ) is defined as the depth at which the downwelling irradiance ( $K_d$ ) of PAR falls to 1% of that just below the surface and was estimated by  $4.6/K_d$  (Kirk 1994), where  $K_d$  is given by  $1.7/z_{0.01}$  (Wetzel 1983).

74% reduction in epilimnetic P (Fig. 2). In addition to complete mixing events during spring and fall turnover, deep water P may also reach upper waters during stratification via hypolimnetic entrainment, thermocline descent, short-period internal waves, and vertical migration of phytoplankton, zooplankton and fish (Wetzel 1983, Horne and Goldman 1994). Increased water clarity in the treated portion further promoted water exchange between the hypolimnion and euphotic zone by lowering the bottom of the euphotic zone to depths well below the thermocline during the three post-treatment summer periods (Fig. 7). In contrast, the bottom of the euphotic zone remained at or above the thermocline during most of the stratification periods in the turbid, untreated section (Fig. 7). This suggests a decreased opportunity for water exchange between the hypolimnion and euphotic zone in the untreated portion, although the Osgood Index of 4.3 indicates an increased probability of deep mixing events (Osgood 1988, Cooke et al. 1993).

The post-treatment mean Secchi depth in the treated section of Lake Leba was 2.34 times greater than the untreated section due to reductions in algal biomass. Only a subset of other alum treatments exist in which sediment P releases were controlled, in turn reducing epilimnetic P, but were not confounded by the effects of nutrient diversion. In these cases, transparency increased substantially, also due to lower algal biomass. For example, Secchi depth increased from 2.4 m (pre-treatment) to 7.1 m 5 years after treatment in Lake Morey (Cooke et al. 1993), and Secchi depth was 13% to 69% higher than pre-treatment means in six shallow lakes during the last 2 years of effective TP reduction (Welch and Schriever 1994).

The amount of usable fish habitat in Lake Leba was estimated by measuring the depth of the  $3 \text{ mg} \cdot \text{L}^{-1}$  dissolved oxygen threshold. Large portions of non-salmonid freshwater fish populations are severely affected by concentrations below this level (Davis 1975), which makes this estimate of usable fish habitat rather conservative. The depth of the  $3 \text{ mg} \cdot \text{L}^{-1}$  threshold increased by a factor of 1.52, which represents a  $4.37 \times 10^7 \text{ L}$  (22%) increase in the volume of usable fish habitat for the treated area (Fig. 2). Alum application also increased the hypolimnetic dissolved oxygen content in Lake Morey, apparently the result of increased algal photosynthesis in deeper waters due to greater light penetration (Smeltzer 1990, Cooke et al. 1993).

### *Phytoplankton composition*

Phytoplankton community responses to alum addition are the opposite of responses generally reported for P enrichment. Numerous studies have convincingly

identified a strong correlation between phytoplankton biomass and TP (e.g., Sakamoto 1966, Vollenweider 1968, Dillion and Rigler 1974, Threlkeld 1988, Drenner et al. 1989, 1990, Lancaster and Drenner 1990, Holz and Hoagland 1996). Increases in cyanophytes and cryptophytes, as well as decreases in bacillariophytes have been common changes in the species composition and relative abundances of the phytoplankton in mesocosm enrichment manipulations (Threlkeld 1988, Vanni 1987, Drenner et al. 1989, 1990, Lancaster and Drenner 1990); these shifts are generally attributed to variation in the minimum P requirement of these algae (Reynolds 1984).

The results of the present study are the opposite of the above mesocosm studies. Alum addition to Lake Leba reduced chl *a* and total phytoplankton biovolume by 65% and 40%, respectively, while cyanophyte and cryptophyte biomass and relative abundances decreased and bacillariophyte biomass and relative abundance increased (Figs. 3 and 4). These changes in the phytoplankton community are similar to the changes observed in an experimental microcosm study of the effects of P reduction on the plankton community structure of Lake Leba (Holz and Hoagland 1996). When TP in fishless, 1000-L tanks was reduced to concentrations near those measured in the alum-treated section of Lake Leba, chl *a* was reduced by 62% to 87%, total phytoplankton biovolume was reduced by 90%, and species composition shifted from 95% cyanophyte dominance toward diatoms and dinoflagellates (Holz and Hoagland 1996).

In other successful alum treatments, chl *a* concentrations in shallow Washington lakes were 43% to 82% lower after treatment for a period of at least 5 to 9 years. In Lake Morey, chl *a* dropped from a pre-treatment concentration of  $31 \mu\text{g} \cdot \text{L}^{-1}$  in 1985 to about  $1 \mu\text{g} \cdot \text{L}^{-1}$  during 1990-93 (Welch and Cooke 1995). Although reports of post-treatment changes in community composition are less common, Welch and Cooke (1995) report that cyanophyte relative abundance at Long Lake (Kitsap) was reduced from 91% (pre-treatment) to an average of 57% during years 8 to 9 (post-treatment). They also observed reductions in cyanophyte relative abundance from 38% to an average of 24% during years 5 to 7 at Erie Lake, and from 70% to an average of 39% over years 5 to 7 at Campbell Lake. Despite the near 100% pre-treatment dominance of *Aphanizomenon* during the late summer in Erie and Campbell Lakes, it has not been detected since the alum additions (Welch and Cooke 1995). Results of studies conducted at Lake Leba and other alum addition studies suggest that P dynamics are driving changes in the phytoplankton community, and that the effects of enrichment are indeed reversible and predictable; the response is opposite to that of eutrophication.

Perhaps Lake Leba represents a worse case scenario in lake restoration, given its extremely hypereutrophic state and near 100% cyanophyte dominance from late April through fall turnover. Although nutrient reduction occurred on a short time scale, species replacement should occur on a longer time scale due to the algal immigration pool in the sediments. Changes in the phytoplankton relative abundances from 1994 to 1995 suggest that community composition is changing, and that cyanophytes will likely continue to decline as new competitors enter the community and cyanophyte dominance in the sediment immigration pool diminishes with time. Therefore, changes in the phytoplankton community to less nuisance conditions are expected to continue, even though the control of sediment P release and subsequent reductions in epilimnetic P are immediate.

### *Zooplankton composition*

Zooplankton community response to alum addition differed between 1994 and 1995 in Lake Leba. In 1994, the physical effect of the alum floc and increased water clarity in the early summer apparently reduced rotifer and *Daphnia* biomass and relative abundance in the treated section. Rotifers, owing to their small size and weaker swimming ability, were apparently unable to avoid precipitation by the alum floc, and their biomass remained low throughout 1994 (Fig. 5). *Daphnia* biomass was likely reduced by a combination of physical flocculation and increased predation during the clear-water conditions of the early summer. Similar *Daphnia* reductions were observed immediately after alum addition at Liberty and Newman Lakes, and were also attributed to the combined impacts of the physical effects of the floc and increased predation pressure (Gibbons et al. 1984, Schumaker et al. 1993). *Daphnia* populations were unable to recover from these reductions, and their low biomass persisted for the remainder of the summer at Lake Leba, Liberty Lake, and Newman Lake (Fig. 5, Gibbons et al. 1984, Schumaker et al. 1993). At Lake Lyngby Sø, Denmark, cladocerans were reduced immediately following an alum treatment, but recovered within 2 months. However, the initial reduction was primarily attributed to the toxic and physical effects of alum (due to a substantially higher dose), not increased fish predation (Sonnichsen 1977).

Assuming that *Daphnia*, small-bodied cladocerans (primarily *Bosmina longirostris*), and cyclopoids would have been similarly impacted by physical flocculation, increases in *Bosmina* and cyclopoid biomass and relative abundances suggest that fish predation during the early summer was driving zooplankton community

changes in 1994. Planktivorous fish are well known to selectively remove larger daphnids (Brooks and Dodson 1965), which apparently favored *Bosmina* and the more elusive cyclopoids. Cyclopoids, which have an inherently low susceptibility to planktivores and strong predator avoidance abilities (Allan 1976), also increased as *Daphnia* biomass decreased immediately after alum addition at Newman Lake (Schumaker et al. 1993).

In the treated section during 1995, *Daphnia* and *Bosmina* biomass and relative abundances were likely higher than in the untreated section due to shifts in the phytoplankton community from cyanophytes towards edible diatoms and greens, because cladoceran abundance is correlated positively with edible algal species and negatively with cyanophyte biomass (Arnold 1971, Porter 1973). In addition, a limiting food environment should favor *Daphnia* over *Bosmina* when predation pressure is low due to the higher filtering rates of *Daphnia* (Haney 1973). The success of the larger cladocerans (i.e., *D. galeata mendotae*) suggests that the overall predation pressure was not as great in 1995, owing to the absence of the clear-water state found in the early summer of 1994.

Longer-term (i.e.,  $\geq 1$  year) zooplankton community responses to alum addition have been varied. Gibbons et al. (1984) found that the overall effect of alum addition on the Liberty Lake zooplankton community was minimal, and there was no shift in dominance over the 4-year study. Rotifer abundance increased and community dominance shifted from a mixed community of cladocerans and copepods toward one primarily dominated by copepods during the 2.5 years following alum addition to Newman Lake (Schumaker et al. 1993). As in the present study, the longer-term compositional shifts at Newman Lake were mostly attributed to phytoplankton shifts from cyanophytes towards more edible species, although increased size-selective predation may have also contributed (Schumaker et al. 1993).

### *Conclusions*

The fields of restoration ecology and lake and reservoir management have emphasized the goal of returning polluted waters to pre-enrichment conditions, and phosphorus inactivation has been identified as a potentially important tool for lake managers working toward this objective. The results of the present study, combined with results from numerous other lake treatments, clearly show that alum is a highly effective and potentially long-term method of controlling sediment P release. However, reductions in epilimnetic TP and subsequent changes in chl *a* and water clarity depend on the degree of vertical entrainment of

hypolimnetic P, control of external loading, P loading from the littoral zone, and P recycling by macrophytes. Sandpit lakes may be ideal in this regard, primarily due to low external loading and narrow littoral zones. In the present study, substantially lower P release rates successfully improved the epilimnetic lake state from highly hypereutrophic to moderately eutrophic, suggesting that sediment P drives lake dynamics in Lake Leba. Specific improvements included greater water clarity, decreased chl *a* concentrations, reduced cyanophyte biomass and dominance, increased *Daphnia* biomass and relative abundance, and an increase in usable fish habitat; all of which are common lake management objectives.

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