Journal of Lake and Reservoir Management 15(3):173-184, 1999

Long-Term Water Quality and Biological Effects of Alum Treatment of Lake Morey, Vermont

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ABSTRACT

Smeltzer, E., R. A. Kirn and S. Fiske. 1999. Long-term water quality and biological effects of alum treatment of Lake Morey, Vermont. Lake and Reserv. Manage. 15(3):173-184.

Lake Morey received one hypolimnetic treatment with alum and sodium aluminate in 1986 to reduce the occurrence of algae blooms caused by internal P loading. Thirteen years of post-treatment monitoring have documented a sustained improvement in water quality in the lake. Comparison of pre-treatment (1977-1985) and post-treatment (1986-1998) data indicated that significant improvements occurred in all water quality variables measured, including spring TP (reduced 66%), summer photic zone TP (reduced 68%), summer photic zone Chl *a* (reduced 61%), summer Secchi disk transparency (increased 79%), late summer hypolimnetic TP (reduced 83%), and late summer hypolimnetic DO (increased 193%). The one-time alum/aluminate treatment of Lake Morey has thus far succeeded in breaking a cycle of sediment P release and re-precipitation that was initiated during an earlier period in the lake's history. Relative weight loss was observed in samples of large yellow perch during the 3 years immediately following the treatment. Yellow perch condition returned to pre-treatment levels by 1991. Toxicity from elevated dissolved aluminum concentrations in the water column during the summer of treatment may have caused the observed weight loss in fish. The benthic macroinvertebrate community in the upper hypolimnion experienced a 90% decline in density during the year immediately following the alum treatment. The benthic community recovered during subsequent years with density and taxa richness generally exceeding pre-treatment values, possibly in response to improved hypolimnetic DO conditions.

Key Words: alum, phosphorus inactivation, internal loading, lake restoration, lake monitoring, aluminum toxicity, fish, benthic macroinvertebrates.

Phosphorus (P) inactivation with aluminum compounds such as alum (aluminum sulfate) is an important lake restoration tool for controlling internal phosphorus loading from sediments (Cooke et al. 1993a). Welch and Cooke (1999) reviewed numerous case histories of lake alum treatments and found that in the majority of cases, including both stratified and unstratified lakes, alum treatment was at least partly responsible for controlling internal P loading and reducing water column total phosphorus (TP) and chlorophyll *a* (Chl *a*) concentrations over post-treatment monitoring periods ranging from 4 to 20 years. However, relatively few studies have investigated the impacts of alum treatments on lake biota, and Cooke et al. (1993a) noted the need for more long-term biological monitoring of alum-treated lakes.

Lake Morey (VT) provides an alum treatment case history where both water quality and biological monitoring have occurred over a 13-year post-treatment period. The rationale for the treatment and the initial post-treatment monitoring results were reported by Smeltzer (1990). This paper describes long-term changes in water quality, fish relative weight, and benthic macroinvertebrate community structure after alum treatment of Lake Morey.

Description of Site and Sampling Methods

Lake Morey is a 220-ha stratified, dimictic lake with a maximum depth of 13 m, a mean depth of 8.4 m, and a 92% forested watershed of 1,900 ha. P budget studies by Walker (1983) and Morgan et al. (1984) indicated that internal P loading from anoxic hypolimnetic sediments was the primary cause of excessive P levels and algae blooms in the lake. Paleolimnological studies (Smeltzer and Swain 1985) suggested that P loading from improper waste disposal practices and land clearing during the early 1900s caused eutrophic conditions that were perpetuated by internal recycling processes, even after the external loads declined in recent decades through better land management and wastewater disposal methods.

The hypolimnion of Lake Morey was treated with a pH-balanced mixture of alum and sodium aluminate in June 1986 to control the internal P loading. A buffered alum mixture was used to avoid pH changes in the moderately low alkalinity (35 to 45 $mg \cdot L^{-1}$ as $CaCO_3$) water of the lake. Chemical dose rates were determined using guidelines adapted from Cooke and Kennedy (1981). The entire 133-ha hypolimnion (≥8 m depth) was treated at a rate of 44 $g\ Al \cdot m^{-2}$ (12 $mg\ Al \cdot L^{-1}$).

Water Quality Sampling

Long-term water quality monitoring has been conducted on Lake Morey by the Vermont Department of Environmental Conservation (DEC) under a variety of programs. TP at spring turnover was measured in vertically integrated samples of the entire water column, obtained on one date annually during 1977-1998. The Vermont Lay Monitoring Program (Warren and Lohner 1991) provided summer monitoring data on Lake Morey from 1979-1998, supplemented with comparable data obtained by Vermont DEC staff in 1978. Vertically integrated samples of the photic zone (twice the Secchi disk depth) for TP and Chl *a* were obtained weekly during June to early September each year.

Vertical profiles of TP, dissolved oxygen (DO), and other variables were determined as part of a diagnostic-feasibility study and lake restoration project on Lake Morey (Morgan et al. 1984, Smeltzer 1990) and other Vermont DEC programs. Discrete-depth samples were obtained at 1-m intervals on at least one day during the late summer (August to early September) period of maximum hypolimnetic P accumulation and DO depletion, during 1979-1998. Vertical profiles of dissolved aluminum concentration were measured at bi-weekly intervals during April through November in 1986 and 1987.

Chemical analyses were conducted by the Vermont DEC Laboratory using standard methods (USEPA 1983, American Public Health Assoc. 1989). TP samples were acid-persulfate digested in the original glass sampling containers, followed by colorimetric analysis using the ascorbic acid method. Chl *a* samples were filtered in the field onto glass fiber filters and frozen for storage. The filters were ground in a tissue grinder with 90% acetone, and the extract was analyzed fluorometrically with a correction for pheophytin. DO was measured by the Winkler titration method. Dissolved aluminum was measured on 0.45-μm membrane-filtered samples using atomic absorption spectroscopy.

Fish Sampling

Yellow perch (*Perca flavescens*) were sampled with experimental gill nets (5 panels, 7.6 x 1.8 m each, with bar mesh increasing in size from 19, 25, 32, 38 to 44 mm) set perpendicular to the shoreline and fished overnight during October from 1984-1989 and 1991-1994. Individual yellow perch were weighed to the nearest g and total length was measured to the nearest mm, except for 1984-1987 when length was measured to the nearest 2.5 mm (0.1 inch).

The "condition" of yellow perch was described by the relative weight (Wr) index developed by Willis and Guy (1991). This index presents the actual weight of an individual fish as a percentage of a standard weight for fish of the same length (Wege and Anderson 1978). Standard weights for yellow perch were developed from length-weight relationships of 78 yellow perch populations from 20 states and 6 Canadian provinces (Willis and Guy 1991). Mean Wr values were compared among the 9 years of sampling for fish 202 to 278 mm (8.0 to 10.9 inches) in length. This length range was selected to maximize the use of pre-treatment data, and comprised 89% of the fish captured in the 2 years prior to the alum treatment. Comparisons restricted to this specific size range also account for the potential influence of fish size on feeding behaviors, such as piscivory and prey size selection (Knight et al. 1984, Paszkowski and Tonn 1994, Mittelbach and Persson 1998).

Benthic Macroinvertebrate Sampling

Benthic macroinvertebrate community monitoring in Lake Morey was conducted by the Vermont DEC in the late winter during one pre-treatment year (1986) and during eight post-treatment years (1987-1993 and 1997). Samples were collected on one date each winter from two depth contours (9 m and 12 m) in the profundal zone of the lake. Six replicate samples were collected at each depth using a standard 6-in^2 (0.02 m^2) Ekman dredge. Samples were rinsed through a #30 sieve and preserved in the field using 75% ethanol. In the laboratory, the samples were picked completely of all macroinvertebrates. The organisms were then identified to genus or species (except the Oligocheata), enumerated, and archived in 75% ethanol. Identifications were determined according to Johannsen (1970), Simpson and Bode (1980), Bode (1990), Peckarsky et al. (1990), and Strayer (1990). Median density and taxa richness were compared statistically between the pre-treatment (control) year and the post-treatment years using Dunnett's test (Steel and Torrie 1960, Jandel Scientific 1994). Taxa richness is defined as the number of different taxonomic categories (primarily genus or species) found in each sample.

Results

Water Quality

Spring TP and summer mean photic zone TP increased from 1975 to 1985, and then declined immediately after the alum treatment in 1986 (Fig. 1). Post-treatment levels remained below any of the annual values recorded during the pre-treatment period. Comparison of mean pre-treatment and post-treatment annual values (Table 1) indicated a spring TP reduction of 66% and a summer TP reduction of 68%.

Summer mean photic zone Chl *a* concentrations also increased during the years preceding the alum treatment (Fig. 1). This trend culminated in a severe cyanobacteria bloom *(Oscillatoria* sp.) and associated fish kill in August 1985, affecting primarily yellow perch. Chl *a* declined immediately after the alum treatment, and remained low and stable in recent years following some variability during the early post-treatment period. The alum treatment was followed by a 61% decline in summer mean Chl *a* concentration (Table 1).

Summer mean Secchi disk transparency increased immediately after the alum treatment, and water clarity continued to increase throughout the post-treatment period (Fig. 1). Unprecedented summer mean Secchi disk transparencies exceeding 8 m have been recorded in Lake Morey during recent years. A 79% increase in mean Secchi disk transparency was observed after the alum treatment (Table 1).

Results from the TP and DO depth profiles were used with lake morphometric information to compute volume-weighted mean hypolimnetic concentrations (8- to 12-m-depth range) for one late summer date during each year for which data were available. Hypolimnetic TP concentrations were high and variable (0.038- to 0.245-mg$\cdot$L^{-1} range) before the alum treatment (Fig. 1), and declined to moderate and more stable values (0.013- to 0.050-mg$\cdot$L^{-1} range) after the treatment. Vertical profile data were averaged by depth for the pre-treatment and post-treatment periods to produce the average TP and DO depth profiles shown in Fig. 2. TP concentrations were reduced throughout the water column following the treatment, and TP accumulation in the hypolimnion was sharply curtailed.

Late summer DO concentrations in the hypolimnion increased after the treatment, although considerable year-to-year variability was evident (Fig. 1). Summer DO profiles (Fig. 2) showed a downward shift in the depth of the DO maximum and a reduced extent of hypolimnetic anoxia following the treatment.

During and following alum treatment, dissolved aluminum concentrations exceeded the target safe level of 50 g$\cdot$L^{-1}, which was recommended by Cooke and Kennedy (1981) and used to determine the maximum aluminum dosage for this treatment (Smeltzer 1990). Fig. 3 shows that dissolved aluminum concentrations ranged from 100 to 200 μg$\cdot$L^{-1} for more than 30 days within the 5- to 7-m zone, and between 50 to 100 μg$\cdot$L^{-1} from late May until fall turnover from the surface to 7 m in depth. The elevated dissolved alum-inum concentrations may have been the consequence of unanticipated mixing of treatment chemicals into the upper waters where pH was higher (7.5 to 8.5 range, Fig. 3), and where aluminum solubility would be greater.

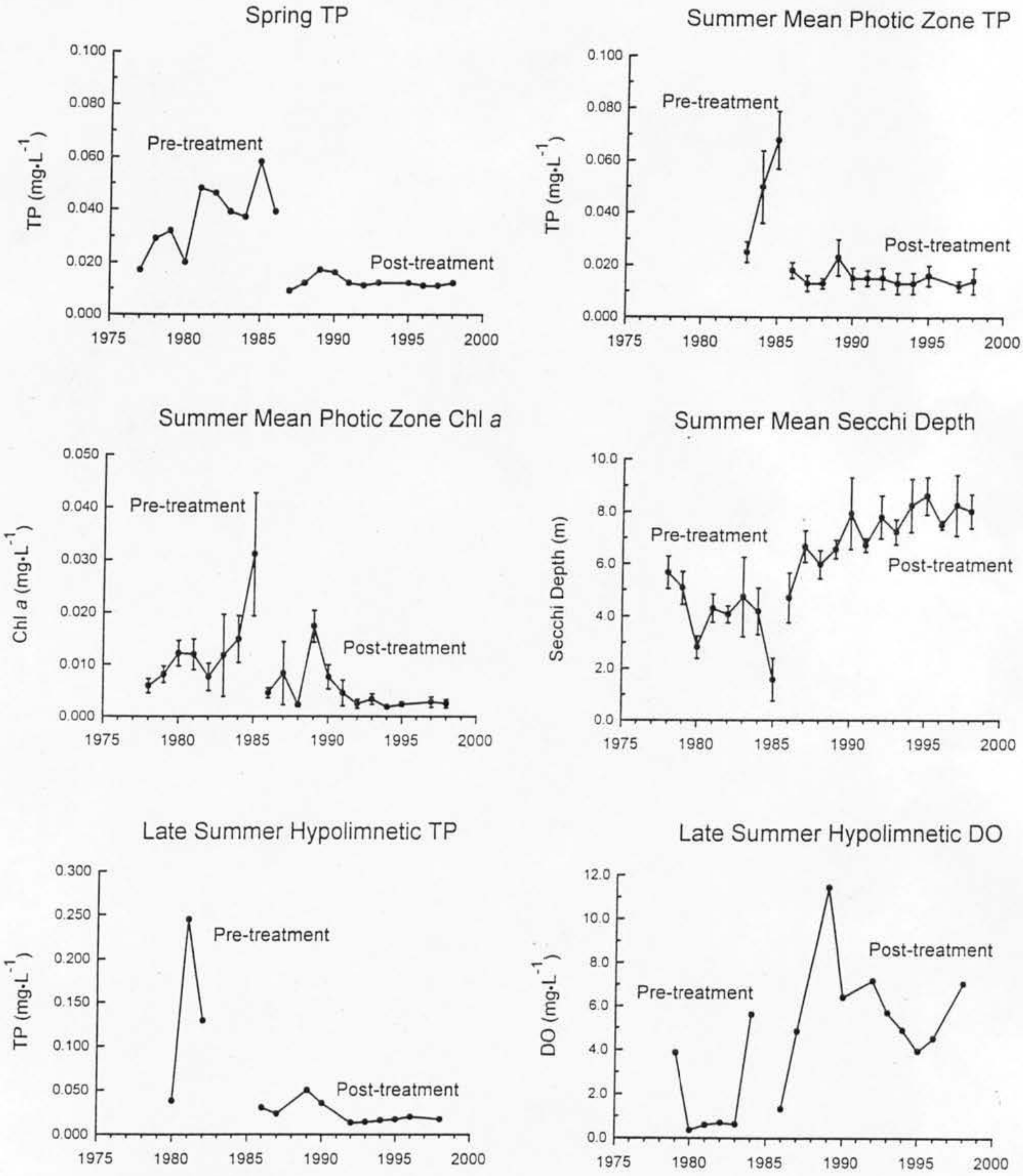

Figure 1.–Long-term water quality monitoring results for Lake Morey. Error bars are 95% confidence intervals for the annual mean values.

Fish

Considerable changes in gill net catches and Wr of large yellow perch were observed during the study period. Large yellow perch density, reflected in average catch per gill net, decreased from 53/net in 1984 to 11/net in 1985 (Fig. 4) following the partial fish kill in 1985. An increase in mean Wr, representing an average weight increase of 4%, was observed coincident with this decline in catch (Fig. 4).

Substantial declines in Wr values for large yellow perch were observed following the alum treatment in

Table 1.–Summary of water quality changes in Lake Morey following the alum/aluminate treatment. Values in parentheses are the number of years of monitoring data on which the means of the annual values are calculated. All differences between means are statistically significant at p<0.05, based on a two-tailed t-test for independent means.

Variable	Pre-Treatment Mean	Post-Treatment Mean	Percent Change
Spring TP ($mg \cdot L^{-1}$)	0.037 (10)	0.012 (11)	- 66%
Summer Photic Zone TP ($mg \cdot L^{-1}$)	0.048 (3)	0.015 (12)	- 68%
Summer Photic Zone Chl *a* ($mg \cdot L^{-1}$)	0.0129 (8)	0.0050 (12)	- 61%
Summer Secchi Disk Transparency (m)	4.0 (8)	7.2 (13)	79%
Late Summer Hypolimnetic TP ($mg \cdot L^{-1}$)	0.137 (3)	0.023 (10)	- 83%
Late Summer Hypolimnetic DO ($mg \cdot L^{-1}$)	1.9 (6)	5.7 (10)	193%

1986 (Fig. 4). The Wr values prior to the alum treatment averaged 77 and 81 for 1984 and 1985, respectively, and declined to 66 in 1986 and remained between 60 and 63 through 1989. This decline in Wr corresponded to average weight losses of 11 to 16% from 1984, and 14 to 20% from 1985 samples. Catches of large yellow perch during this period continued to decline to a low of 2/net in 1989.

Recovery of mean Wr to pre-treatment levels was observed in 1991 (mean Wr = 75) and continued throughout the remainder of the study, averaging 81 each year from 1992-1994. Gill net catch of large yellow perch also increased during this period, from 13/net in 1991 to 39/net in 1994.

The distribution of individual Wr values further describes the changes in yellow perch condition following the alum treatment (Table 2). Prior to the alum treatment, the majority of individual fish had Wr values between 70 and 90, while the proportion of fish observed with Wr values less than 70 ranged between 12 to 18%. No fish were observed with a Wr value less than 60 in 1984 or 1985. In 1986 the proportion of fish with Wr values less than 70 increased to 71%, and remained between 79 to 90% from 1987-1989. Sixteen to 47% of the yellow perch during this time period had Wr values less than 60, including individual fish with Wr values as low as 30. During the recovery period from 1991-1994, Wr values less than 70 accounted for only 2 to 11% of the fish measured. As in pre-treatment samples, no fish with Wr values less than 60 were observed from 1991-1994.

Benthic Macroinvertebrates

The density of animals at the 9 m depth declined significantly by over 90% during the first year following the alum treatment (Fig. 5). However, median density at the 9 m depth recovered fully to pre-treatment levels by 1988. The density of animals at the 12 m depth was highly variable during the monitoring period, and showed little change following the alum treatment.

In winter 1986, prior to the alum treatment, the benthic community was dominated by only a few taxa considered to be tolerant of oxygen-deficient environments, primarily *Chaoborus punctipennis* and *C. flavescens,* and *Chironomus decorus, C. riparius and C. tentans* grps. (Hynes 1971, Beck 1977, Wiederholm 1980, Bazzanti and Seminara 1987). At the 9 m depth, taxa richness began to increase a few years after the treatment (Table 3, Fig. 5). Median taxa richness at 9 m doubled by 1997, with a significantly higher number of species recorded in samples obtained during 1992 and 1997 than were found during the pre-treatment year (Fig. 5).

The taxonomic composition of the benthic community at the 12 m depth was less affected by the alum treatment. Two species of *Chaoborus* (*C. punctipennis* and *C. flavescens)* dominated the 12 m depth both before and after the treatment (Table 4). However, median taxa richness at the 12 m depth increased significantly during 1993 and 1997 (Fig. 5).

Discussion

Cooke et al. (1993a) proposed that the success of sediment P inactivation treatments be evaluated using three criteria: (1) Did the treatment reduce sediment P release for at least several years? (2) Did the treatment lower the P concentration in the lake's photic zone? (3) Was the treatment non-toxic? Payne et al. (1991) suggested that attainment of the lower 25^{th} percentile of the appropriate lake ecoregion TP

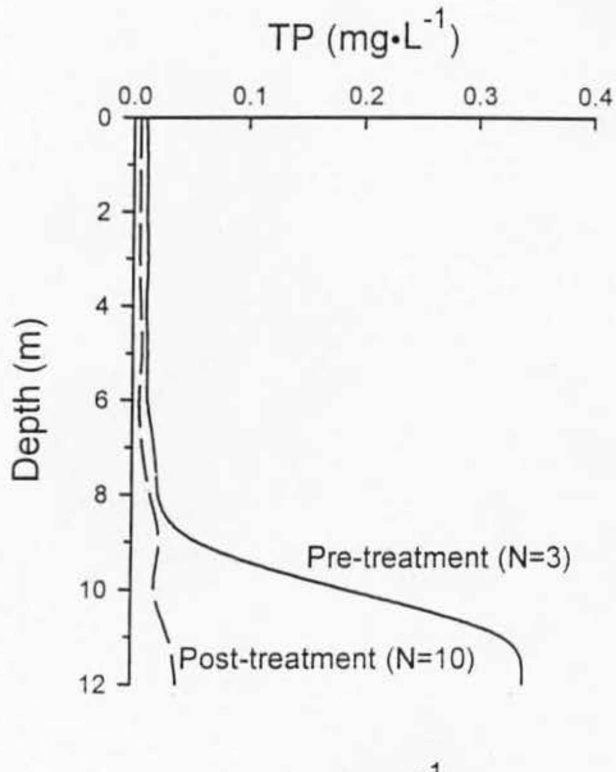

Figure 2.–Comparison of mean late-summer TP and DO depth profiles between pre-treatment and post-treatment monitoring years.

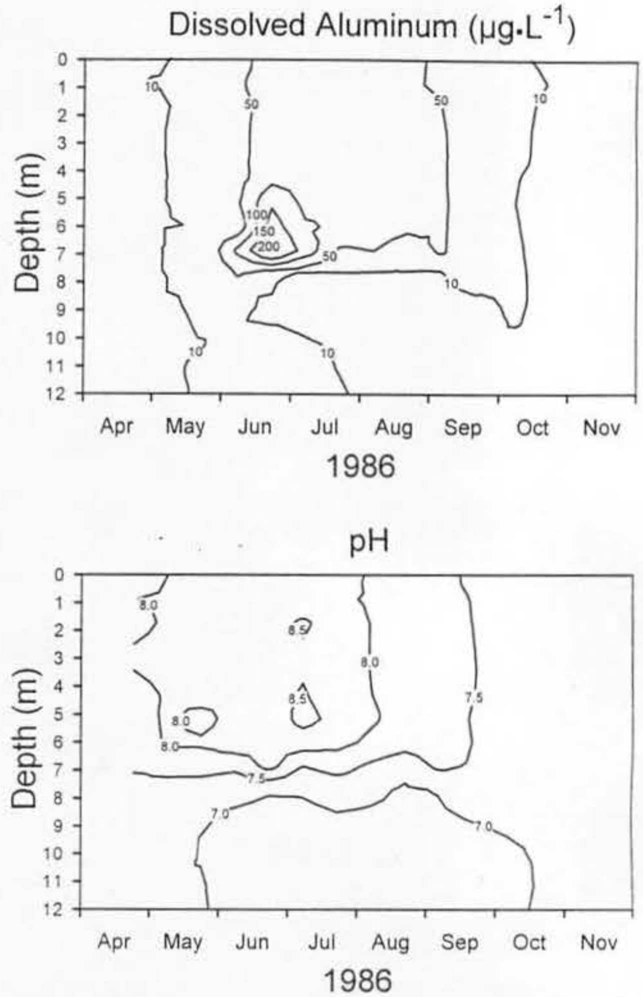

Figure 3.–Dissolved aluminum concentrations and pH in the water column of Lake Morey during the year of treatment.

distribution be used as a treatment effectiveness criterion.

In Lake Morey, the alum/aluminate treatment appears to have significantly reduced sediment P release (inferred from hypolimnetic TP accumulation) and photic zone TP concentrations over a 13-year post-treatment monitoring period. Lake Morey lies within a lake ecoregion with a lower 25th percentile TP value of about 0.015 mg · L^{-1} (Rohm et al. 1995). Post-treatment summer photic zone TP values have averaged 0.015 mg · L^{-1} (Table 1). Therefore, the Lake Morey treatment can be judged effective by the quantitative criterion of Payne et al. (1991).

The extent of P reduction observed in Lake Morey can be attributed to several factors. First, external P loads were under control prior to treatment (Smeltzer 1990). P export to the lake from the watershed was low (0.16 kg · ha^{-1} · yr^{-1}) and typical of a predominately forested watershed (Reckhow and Chapra 1983). Pre-treatment lake TP concentrations were much higher than the 0.009 mg · L^{-1} value predicted from the external loading rate using a simple lake model (Dillon and Rigler 1974, Smeltzer 1990) that did not take internal loading directly into account. After the treatment,

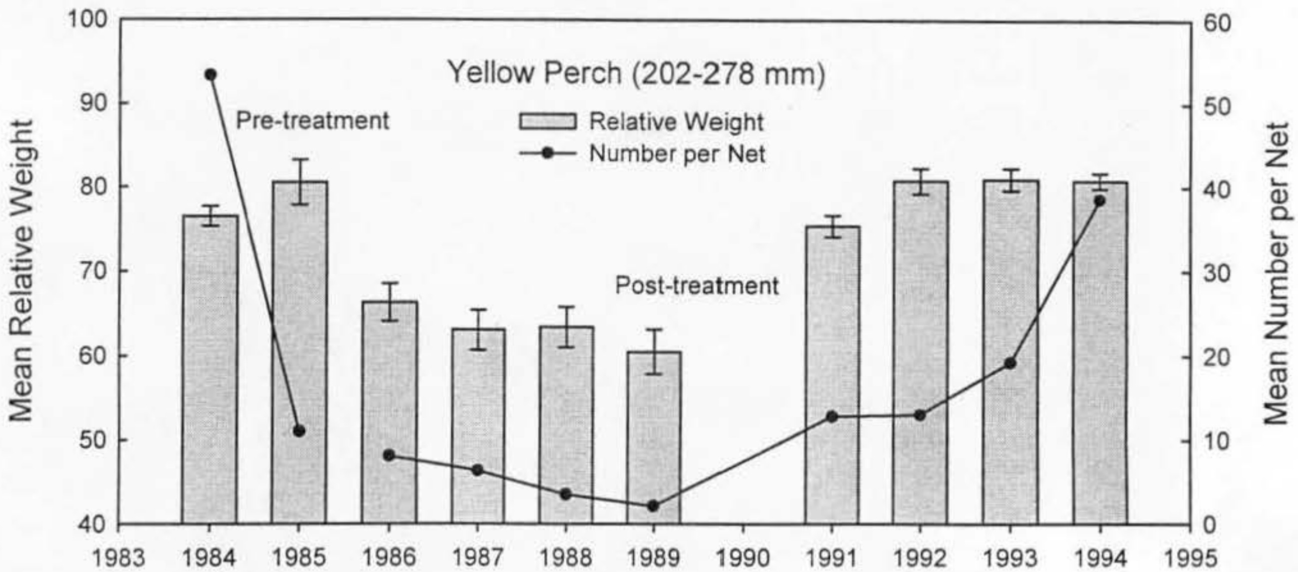

Figure 4.–Mean relative weight and number of fish per net for large yellow perch caught in experimental gill nets in Lake Morey during 1984-1994. Error bars are 95% confidence intervals for the mean relative weight values. No data were collected during 1990.

TP concentrations approached the model-predicted value.

Cooke et al. (1993b) suggested that lakes such as Lake Morey with a low ratio of mean depth to square root of surface area ("Osgood Index") would be good candidates for alum treatment because of the strong potential for vertical transport of hypolimnetic P into the photic zone during the summer. However, Mataraza and Cooke (1997) later found that substantial transport also could occur in deeper lakes with strong vertical P concentration gradients. Walker (1983) concluded that vertical mixing at fall turnover and conservative retention of P over the winter were more important factors in the delivery of P to the photic zone in Lake Morey than thermocline erosion during the summer. In any case, various mechanisms existed in Lake Morey for delivery to the photic zone of P released from hypolimnetic sediments.

The increase in hypolimnetic DO concentrations following the alum treatment was an unexpected occurrence. Variations in DO concentrations in the hypolimnion of Lake Morey appear to be strongly influenced by the photosynthetic activity of a mid-depth algal layer usually dominated by *Oscillatoria sp.* Greater

Table 2.–Distribution (percent of fish measured each year) of relative weight (Wr) values for 202- to 278-mm yellow perch collected in experimental gill nets from Lake Morey during 1984-1994.

Year	Sample Size	Relative Weight (Wr) 30	40	50	60	70	80	90	100
1984	160	0	0	0	18	53	24	4	1
1985	33	0	0	0	12	33	45	9	0
1986	65	2	0	14	55	23	6	0	0
1987	51	0	4	27	51	16	2	0	0
1988	60	2	7	23	47	20	2	0	0
1989	28	0	11	36	43	11	0	0	0
1991	64	0	0	0	11	69	20	0	0
1992	65	0	0	0	2	42	48	9	0
1993	96	0	0	0	5	38	51	6	0
1994	193	0	0	0	4	45	45	7	0

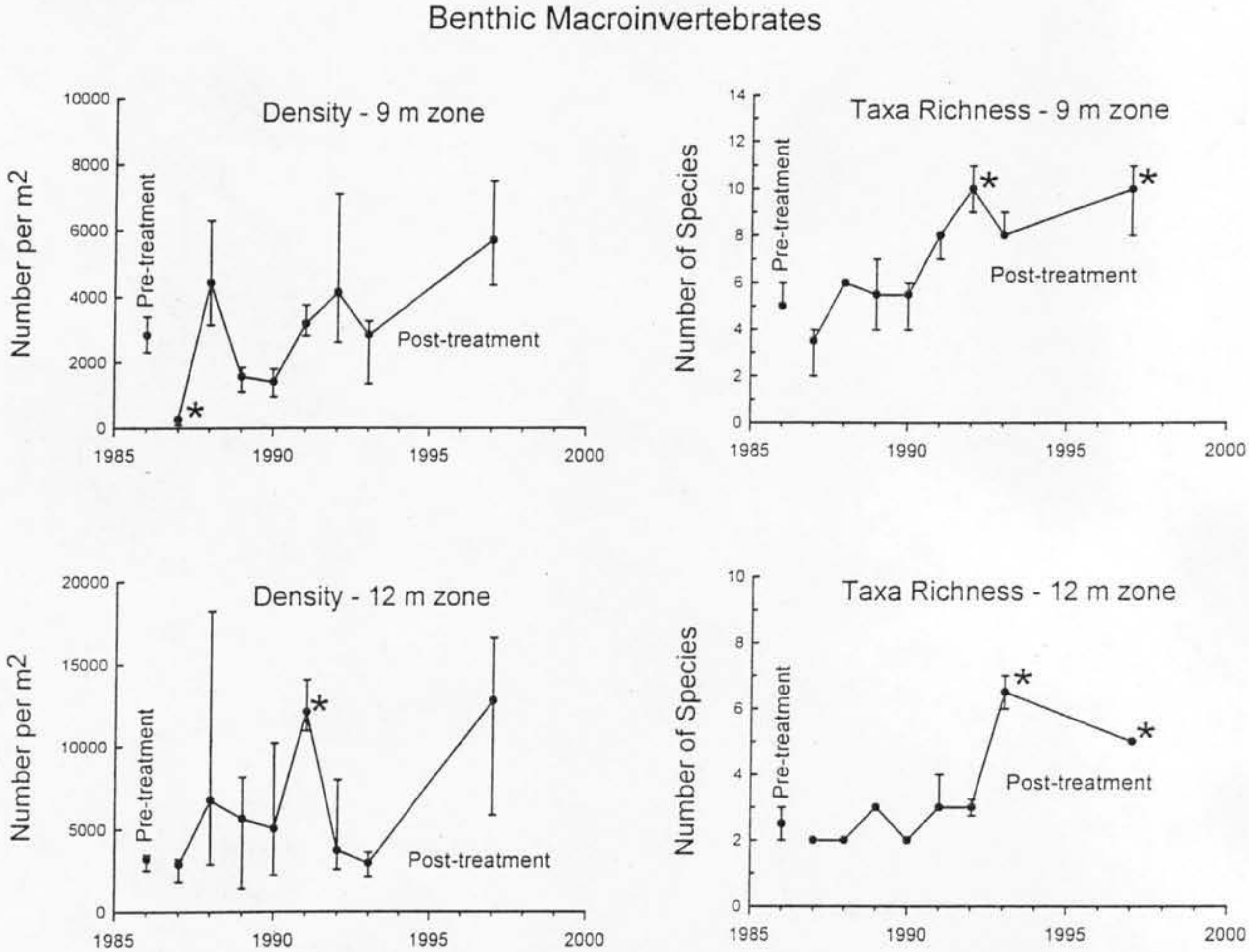

Figure 5.–Density and species richness of benthic invertebrates at the 9 m and 12 m depths in Lake Morey during 1986-1997. Median values and interquartile intervals among replicate samples are shown. Asterisks indicate medians that are significantly different from the median value for the pre-treatment year (1986), based on Dunnett's test at $p<0.05$.

water clarity after the treatment was associated with a shift in the vertical location of maximum Chl *a* and DO concentrations downward from the metalimnion into the hypolimnion (Smeltzer 1990). Possible explanations for the hypolimnetic DO increase include (1) deeper light penetration after the treatment allowing photosynthetic oxygen production in the hypolimnion, and (2) reduced P concentrations and lake productivity leading to lower community respiration rates in the hypolimnion. Better sediment oxidation may be contributing to treatment effectiveness and longevity by curtailing anoxic P release from the hypolimnetic sediments.

The Lake Morey alum/aluminate treatment was not as successful in achieving the non-toxic criterion of Cooke et al. (1993a), as indicated by the elevated dissolved aluminum concentrations in the 0- to 7-m-depth zone during the summer of treatment. The soluble forms of aluminum generally occur below a pH of 6.0 or above 8.0, and are regarded as the most acutely toxic forms to fish (Freeman and Everhart 1971, Decker and Menendez 1974, Hunter et al. 1980, Gundersen et al. 1994), although insoluble forms have also been shown to have lethal and sublethal effects (Freeman and Everhart 1971, Decker and Menendez 1974, Baker and Schofield 1982, Gundersen et al. 1994). Nearly anoxic conditions prevailed below 9 m, precluding this area as a possible refuge for fish from aluminum exposure.

The modest increase in Wr observed from 1984 to 1985 may have been a response to reduced densities of large yellow perch and diminished competition for food resources (Swingle 1956) resulting from the partial fish kill in 1985. Following the alum treatment in 1986 however, mean Wr declined from 1986-1989 despite relatively low densities of large yellow perch.

The decline in yellow perch Wr is consistent with symptoms of sublethal aluminum toxicity reported in

Table 3.–Taxonomic percent composition and total taxa richness of the late-winter benthic macroinvertebrate community at the 9 m depth in Lake Morey during 1986-1997.

	9 m Depth								
	1986	1987	1988	1989	1990	1991	1992	1993	1997
Chaoborus punctipennis	60	24	73	50	7	54	7	4	6
Chaoborus flavescens	4	21	10	2	<1	1	<1		
Chironomus decorus grp.	18		4	3		2	<1	13	<1
Chironomus riparius grp.	12	21	6	8		2	4	15	4
Chironomus tentans grp.	5	9	3	6	1	8	2	4	4
Trebelos sp.					<1		<1	<1	
Paratendipes sp.			<1						
Cladopelma sp.					<1	4	7	<1	
Cryptochironomous sp.								<1	
Dicrotendipes neomodestus								<1	
Tanytarsus sp.					2	<1	5	<1	4
Diplocadius sp.								1	
Zalutschia sp.			<1	1	3	1	9	8	20
Procladius sp.	<1	12	4	28	46	25	18	30	22
Culicoides sp.							<1		
Sphaeromias sp.					6		2	6	<1
Oecetis sp.	<1								
Sialis sp.		3		<1		1		2	1
Musculium sp.									<1
Pisidium sp.									<1
Hydrachnidia uid.								<1	<1
Oligocheata uid.		9		1	34	<1		16	36
Total Taxa Richness	**7**	**7**	**8**	**9**	**10**	**11**	**12**	**16**	**13**

studies of other alkaline waters. Freeman and Everhart (1971) exposed rainbow trout *(Oncorhyncus mykiss)* to 52, 520 and 5,200 $\mu g \cdot L^{-1}$ of aluminum for up to 45 days at pH levels between 7.0 and 9.0. Although no acute or chronic effects were observed at the lowest concentration, exposure to higher levels resulted in gill damage, darkened coloration, and reduced feeding. Recovery times for treatments were proportional to aluminum concentration and length of exposure. Fish exposed to the highest aluminum concentration for short periods of time (4.7 to 9.3 days) experienced weight differentials as much as 50% below control fish (Freeman and Everhart 1971) and remained 11 to 20% below control weights after a recovery period of over 5 months (Freeman 1973). Exposures for 45 days at pH 7.0 to aluminum concentrations of 520 and 5,200 $\mu g \cdot L^{-1}$ resulted in extremely emaciated fish (24 to 33% of control weights). After more than 9 months, the weight of these fish remained 47 to 53% below that of the control group despite the fact that normal growth rates generally resumed after 2 weeks of recovery.

Gundersen et al. (1994) reported similar results from 96-h (acute) and 16-d (subacute) exposures of rainbow trout to various combinations of aluminum/hardness and aluminum/humic acid concentrations between pH 7.14 and 8.58. Dissolved aluminum exposures varied widely with pH but were generally in the

Table 4.–Taxonomic percent composition and total taxa richness of the late-winter benthic macroinvertebrate community at the 12 m depth in Lake Morey during 1986-1997.

	12 m Depth								
	1986	1987	1988	1989	1990	1991	1992	1993	1997
Chaoborus punctipennis	91	64	73	95	94	96	89	41	84
Chaoborus flavenscens	6	35	27	4	6	2	10	3	2
Chironomus decorus grp.		<1				2	<1	9	6
Chironomus riparius grp.		<1		1			<1	18	<1
Chironomus tentans grp.								4	7
Cladopelma sp.								<1	
Tanytarsus sp.								<1	
Zalutschia sp.				<1				<1	
Procladius sp.						<1		23	
Oligocheata uid.	3			<1					<1
Total Taxa Richness	3	4	2	5	2	4	4	9	6

range of 20 to 500 $\mu g \cdot L^{-1}$. Increased mortality of rainbow trout was associated with concentrations of dissolved aluminum, while insoluble forms of aluminum appeared more potent in restricting growth. Decreased growth was at least partially due to appetite loss as evidenced by reduced consumption. Subacute toxicity (i.e., mortality) of rainbow trout was reduced by hardness and humic acid. However, hardness did not protect against aluminum-related growth impacts. Cooke et al. (1993a) suggested that low total organic carbon levels found in Lake Morey minimized aluminum complexation and may have explained the toxic effects observed in yellow perch in this study.

The sustained recovery in yellow perch condition observed from 1991-1994, despite increasing fish densities, further suggests the decline in Wr following the alum treatment in 1986 was a direct result of aluminum toxicity rather than an indirect effect of reduced lake productivity. This recovery in yellow perch condition may have been a result of gradual loss of aluminum-exposed perch from the population in conjunction with recruitment of perch produced in years following the alum treatment. The decline in gill net catches of large yellow perch from 53/net in 1984 to 2/net in 1989 reflects this loss of large yellow perch from the population. Likewise, the gradual improvement in gill net catches from 13/net in 1991 to 39/net in 1994 indicates recruitment of new year classes into this size range.

In contrast with the effects observed on fish, adverse changes in the benthic macroinvertebrate community of Lake Morey following the alum treatment were much more short-lived. The observed 90% decrease in density of benthic macroinvertebrates at the 9 m depth the year following the treatment may have been caused by chronic toxic effects of the alum treatment. Because the *Chaoboridae* are highly mobile, they may have been exposed to dissolved aluminum concentrations as high as 200 $g \cdot L^{-1}$ while feeding on zooplankton in the algal-rich 6-m to 8-m-depth zone of Lake Morey.

The observed decrease in benthic macroinvertebrate density in the 9-m-depth zone was not likely to have been caused by acute toxicity. Havas and Likens (1984) showed *Chaoborus punctipennis* and *Chironomus anthrocinus* to be tolerant of dissolved aluminum concentrations ranging up to 1,000 $\mu g \cdot L^{-1}$. In laboratory testing, Lamb and Bailey (1981) also showed the midge *Tanytarsus dissimilis* to be unaffected acutely by alum doses as high as 960,000 $\mu g \cdot L^{-1}$. However, Lamb and Bailey (1981) did observe subtle chronic effects at all the alum concentrations tested (80,000 to 960,000 $\mu g \cdot L^{-1}$). These effects included heavy floc impeding movements, and failure of the larvae to develop. The Lake Morey alum treatment may have caused the failure of the *Chaoboridae* population in the 9-m-depth zone to complete development in 1986, leading to a low density in 1987. However, it is unclear why the benthic community was not similarly affected at the 12 m depth. The benthic community in the 9-m zone recovered to pre-treatment densities within 2 years following the treatment.

The long-term response of the benthic community in Lake Morey to the alum treatment has been one of generally increased density and improved taxa richness. Prior to the alum treatment, the profundal zone benthic macroinvertebrate community of Lake Morey was dominated by the anoxia-tolerant *Chaoborus spp*, with only a minor presence of the almost equally tolerant *Chironomous spp.* and *Oligocheata*. This relatively simple benthic community was present in the deep profundal zone of Lake Morey since at least the summer of 1975 (Vermont Department of Water Resources 1975). The low diversity of the benthic community was probably a consequence of severe hypolimnetic anoxia which existed throughout most of the summer, and to a lesser extent during the winter as well (Walker 1983).

The changes in taxonomic composition of the benthic community following the alum treatment may have been the result of improved hypolimnetic DO conditions. Significant increases in taxa richness relative to pre-treatment conditions were observed by 1992 as less anoxia-tolerant species began to appear. During the most recent year of monitoring (1997), the benthic community at the 9 m depth was dominated by several less anoxia-tolerant taxa, including *Zalutschia sp.*, *Tanytarsus sp.*, and *Sialis sp.* Median taxa richness at the 9 m depth seems to have stabilized in recent years at 8 to 11 taxa, representing a two-fold increase over pre-treatment levels. Similar improvements in taxa richness and the appearance of less anoxia-tolerant species were observed at the 12 m depth, beginning in 1993. However, the changes here were less dramatic, reflecting the continued severity of anoxia present at the 12 m depth.

The benthic macroinvertebrate community changes observed in Lake Morey were similar to the findings of Narf (1990), who reported no long-term (6 years post-treatment) negative effects on either the density or species richness of the profundal zone benthic macroinvertebrate communities in five alum-treated lakes in Wisconsin. Like Lake Morey, all the Wisconsin lakes were poor in species richness and dominated by anoxia-tolerant *Chaoboridae* and *Chironomus spp.* Narf (1990) observed post-treatment increases in species richness in several of the Wisconsin lakes.

Conclusions

The Lake Morey project, as well as numerous other alum treatment case histories reviewed by Welch and Cooke (1999), have demonstrated that sediment P inactivation with aluminum salts can be an effective lake management tool. Long-term water quality improvement and enhancement of some biological communities are possible following alum treatment if external P loads are under control. However, alum treatments involve environmental risks to fish and other aquatic biota in both acidic and alkaline environments. Treatments should be designed carefully to minimize these risks.

ACKNOWLEDGMENTS: Water quality and biological monitoring on Lake Morey were supported by grants from the U.S. Environmental Protection Agency Clean Lakes Program, and by several program efforts sponsored by the Vermont Department of Environmental Conservation and the Vermont Department of Fish and Wildlife. The fisheries component of this paper was made possible by Vermont fishing license sales and matching funds made available through the Federal Sportfish Restoration Act. Local citizen volunteers made indispensable contributions to the water quality monitoring effort through participation in the Vermont Lay Monitoring Program. This paper benefited from the comments of four anonymous reviewers.

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