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Long-term effects of successive $\text{Ca}(\text{OH})_2$
and CaCO_3 treatments on the water
quality of two eutrophic hardwater lakes

By
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Effets à long terme de traitements successifs au $\text{Ca}(\text{OH})_2$ et au CaCO_3 sur la qualité de l'eau de deux lac eutrophes à eau dure

Prepas, E. E., J. Babin, T. P. Murphy, P. A. Chambers, G. J. Sandland, A. Ghadouani et M. Serediak

Sommaire à l'intention de la direction

De nombreux lacs et approvisionnements en eau potable des Prairies canadiennes sont fortement eutrophes. Afin d'atténuer les problèmes associés à l'eutrophisation, nous avons développé une méthode pour la réduction de la croissance des algues, basée sur la précipitation du phosphore par l'application de chaux. Ce projet dirigé par l'Université de l'Alberta a reçu de l'aide d'Environnement Canada, de TPSGC et d'un grand nombre d'organismes gouvernementaux de l'Alberta. Pour le traitement de lac entiers en Alberta, l'efficacité des applications répétées de chaux [$\text{Ca}(\text{OH})_2$ et/ou CaCO_3 , à 5 - 78 mg L^{-1}] a duré jusqu'à sept ans et on a noté une suppression de la croissance du phytoplancton et des macrophytes, ainsi que de la biomasse; techniquement, ce traitement est efficace. On peut considérer que sa commercialisation est un succès, puisqu'on a développé les équipements et les compétences nécessaires pour traiter des lacs de plus de 40 hectares. Alors que plusieurs petites entreprises poursuivent le traitement de petits approvisionnements en eau de boisson, on a dû vendre les gros équipements de chaulage et affecter à d'autres tâches les équipes de services commerciaux de traitement des grands lacs parce qu'il n'y avait pas assez de clients. En effet, la plupart des collectivités rurales ne peuvent défrayer les coûts du traitement des lacs, et les gouvernements, pour leur part, ne veulent pas accepter la responsabilité à long terme du traitement des lacs.

Long-term effects of successive $\text{Ca}(\text{OH})_2$ and CaCO_3 treatments on the water quality of two eutrophic hardwater lakes

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SUMMARY

1. Whole-lake experiments were conducted in two hardwater lakes (Halfmoon and Figure Eight) in Alberta, Canada, to investigate the effectiveness of repeated lime (slaked lime: $\text{Ca}(\text{OH})_2$ and/or calcite: CaCO_3) treatments ($5\text{--}78 \text{ mg L}^{-1}$) for up to 7 years.
2. Randomized intervention analysis of intersystem differences between the experimental and three reference lakes demonstrated a decline in euphotic total phosphorus and chlorophyll *a* concentrations in the experimental lakes after repeated lime treatments.
3. After the second lime application to Halfmoon Lake, mean winter total phosphorus release rates (TPRR) decreased to $< 1 \text{ mg m}^{-2} \text{ day}^{-1}$ compared with $3.6 \text{ mg m}^{-2} \text{ day}^{-1}$ during the winter after initial treatment. In the final year of lime application, mean summer TPRR decreased to $4.5 \text{ mg m}^{-2} \text{ day}^{-1}$ compared with $7.6 \text{ mg m}^{-2} \text{ day}^{-1}$ in the pre-treatment year.
4. Mean macrophyte biomass declined and species composition was altered at 1 and 2 m depths in Figure Eight Lake during lime application. Over the first 6 years of treatment, macrophyte biomass at 2 m declined by 95% compared with concentrations recorded during the initial treatment year. In the last year of the study, macrophyte biomass at 2 m reached initial treatment concentrations, which coincided with the greatest water transparency. Over the treatment period, macrophyte species shifted from floating to rooted plants.
5. Multiple lime applications can improve water quality in eutrophic hardwater lakes for periods of up to 7 years.

Keywords: chlorophyll *a*, lime, long-term study, macrophyte biomass, phosphorus

Introduction

Eutrophic hardwater lakes on western Canada's Boreal Plain are characterized by late-summer cyanobacterial blooms, hepato- and neurotoxin release during blooms, high macrophyte standing crop, and dissolved oxygen (DO) depletion that can kill aquatic organisms (Mitchell & Prepas, 1990; Kotak *et al.*, 1993). These conditions have been at least partially induced by human activities such as land clearing for

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agriculture and recreational development (Allan *et al.*, 1980; Mitchell & Prepas, 1990; Hall *et al.*, 1999; Manning, Prepas & Serediak, 1999). Numerous strategies have been employed to remedy such water quality problems, particularly cyanobacterial blooms, in shallow, eutrophic lakes (Cooke *et al.*, 1993). Although these strategies have met with various degrees of success, their effectiveness is often limited in duration.

Water column total phosphorus (TP) and algal biomass (often estimated by chlorophyll *a* (chl *a*) concentrations) are positively related in many freshwater systems (Dillon & Rigler, 1974; Schindler, 1977; Prepas & Trew, 1983). A classic approach to reduce annual algal production and biomass has therefore been to focus on reduction of external phosphorus (P) loading from drainage basins. However, this approach is unlikely to provide sustained improvement in water quality in lakes on the Boreal Plain, where much of the euphotic-zone TP for any given year is derived from internal sources. That is, periodic anoxia in the water column intensifies sediment P release, resulting in increased P loading to the surface waters (Mitchell & Prepas, 1990). Successful control of water column P concentrations in these lakes requires regulation of both internal and external P sources (Mitchell & Prepas, 1990; Cooke *et al.*, 1993; Welch & Cooke, 1995).

Lime (CaO , CaCO_3 , Ca(OH)_2) addition has been used to mitigate the effects of internal P loading. Lime treatment mimics natural calcite (CaCO_3) precipitation in hardwater lakes during summer, which strips P out of the water column (Vanderploeg *et al.*, 1987; Koschel, 1997), making it unavailable for algal uptake (Otsuki & Wetzel, 1972; Koschel *et al.*, 1983; Murphy, Hall & Yesaki, 1983). To increase water column Ca^{2+} concentration and promote P precipitation, lime has been applied to eutrophic lakes, stormwater retention ponds, reservoirs and man-made ponds (locally termed 'dugouts') on the Boreal Plain (e.g. Murphy, Hall & Northcote, 1988; Babin *et al.*, 1989; Prepas *et al.*, 1990; Babin, Prepas & Zhang, 1992). These studies reported the effects of multiple lime additions on water quality, including phytoplankton and macrophyte biomass, for up to 2 years. However, the effectiveness of multiple lime additions over several years has not been examined. In this study, we report the long-term effects of multiple lime (slaked lime: Ca(OH)_2 , and/or calcite: CaCO_3) applications on

euphotic-zone TP and chl *a* concentrations and macrophyte biomass in two hardwater lakes, in comparison with reference lakes.

Study lakes

The two treatment (Halfmoon, Figure Eight) and three reference (Nakamuñ, North and South Baptiste) lakes are hardwater eutrophic to hypereutrophic systems (mean summer TP: $46\text{--}247\ \mu\text{g L}^{-1}$), located on the Boreal Plain of central Alberta (Mitchell & Prepas, 1990). These headwater lakes have long-term water residence times ranging from 4 to > 50 years and are usually ice-covered from November to April. Under-ice DO depletion has been observed in the study lakes and has been associated with enhanced internal P loading (Mitchell & Prepas, 1990). Both treatment lakes were subject to lake and fisheries management interventions prior to lime treatment.

Halfmoon Lake is deeper than Figure Eight Lake; the maximum depths are 8.5 and 6 m, respectively. In combination with a location sheltered from prevailing winds, this results in thermal stratification and the formation of a well-defined hypolimnion in Halfmoon Lake each summer. Conversely, Figure Eight Lake mixes frequently and does not tend to thermally stratify.

Both Halfmoon and Figure Eight lakes were treated with multiple doses of lime (slaked lime: Ca(OH)_2 , and/or calcite: CaCO_3) at target concentrations of $5\text{--}78\ \text{mg L}^{-1}$. Initial treatments to both lakes involved mixtures of both Ca(OH)_2 and CaCO_3 (Table 1). Subsequent applications employed Ca(OH)_2 (which appeared more effective) in the more southerly Halfmoon Lake and a mixture of Ca(OH)_2 and the less expensive CaCO_3 at the remote Figure Eight Lake. Lime was applied to Halfmoon Lake in 1988, 1989, 1991 and 1993, and to Figure Eight Lake in 1986, 1987, 1990 and 1992 (see Prepas *et al.*, 2001 for details).

Pre-treatment monitoring for Halfmoon Lake included 27 May to 29 October 1982 and 6–19 July 1988. Post-treatment monitoring began 25 July 1988 (treatment initiation) and continued until October 1993 (final treatment year), with sampling occurring weekly from May to August and every 2–3 weeks over the winter from September to April. Pre-treatment winter samples were collected from Halfmoon Lake in 1982–83 and post-treatment sampling continued from 1988–89 to 1993–94.

Table 1 Mean concentrations of calcium (Ca^{2+}), total phosphorus (TP), chlorophyll *a* (chl *a*) and pH before (Pre), immediately after (< 7 days) lime application [CaCO_3 and/or $\text{Ca}(\text{OH})_2$], and during the remainder of the open-water season (> 7 days) for two hardwater lakes from 1986 to 1993

Lake	Application dates	Lime type	Dosage* (mg L^{-1})	Whole-lake pH			Whole-lake Ca^{2+} (mg L^{-1})			Euphotic TP ($\mu\text{g L}^{-1}$)			Euphotic chl <i>a</i> ($\mu\text{g L}^{-1}$)		
				Pre	< 7	> 7	Pre	< 7	> 7	Pre	< 7	> 7	Pre	< 7	> 7
Halfmoon	19–25 July 1988	$\text{Ca}(\text{OH})_2/\text{CaCO}_3$	25/30	8.2	8.1	8.1	26	26	22	135	122	107	113	35	66
	8–17 August 1989	$\text{Ca}(\text{OH})_2$	72	7.7	–	8.6	27	14	16	70	56	72	31	17	13
	27–30 May 1991	$\text{Ca}(\text{OH})_2$	75	8.1	9.1	8.4	30	–	17	77	41	81	20	2	32
	25–31 May 1993	$\text{Ca}(\text{OH})_2$	78	8.1	8.8	8.7	21	23	15	87	58	7	22	16	24
Figure Eight	18 June 1986	CaCO_3	14	7.5	9.3	7.9	26	28	29	266	266	229	94	89	40
	30 July–8 August 1986	$\text{Ca}(\text{OH})_2$	13	8.6	7.6	8.6	29	28	29	239	76	171	49	163	117
	14 July 1987	$\text{Ca}(\text{OH})_2$	18	7.6	8.2	8.2	38	37	37	43	78	82	6	16	13
	27 June 1990	$\text{Ca}(\text{OH})_2/\text{CaCO}_3$	14/24	7.7	8.3	8.5	38	37	24	59	36	127	6	3	45
	2 July 1992	$\text{Ca}(\text{OH})_2/\text{CaCO}_3$	15/15	7.8	7.7	7.7	41	35	30	45	48	84	18	6	42

*Target concentration of lime.

For Figure Eight Lake, initial treatment occurred over 2 months as a result of logistical and financial issues associated with transporting lime to this northern site (950 km from the industrial source). Pre-treatment monitoring was conducted from 1 to 17 June 1986. Post-treatment monitoring was conducted from June 1986 to 1987 and from 1989 to 1993. Intervals between sampling dates ranged from days (during summer) to months (during winter). Following the second summer of treatment in Figure Eight Lake, beginning in January 1987, an aerator was used in the deepest part of the south basin. Winter data are therefore not presented for Figure Eight Lake. In addition, Figure Eight Lake was not sampled in 1988, with the exception of macrophyte sampling, because of both financial and logistical constraints.

The reference lakes, Nakamun and Baptiste, were sampled in 1982 (Riley, 1983; Riley & Prepas, 1984). For 1986–93, data for Nakamun and Baptiste lakes were obtained from Alberta Environment. Further details regarding the lakes, liming procedures and sampling regimes can be found in Prepas *et al.* (2001).

Methods

Water sample collection and analysis

Water temperature was measured at 1-m intervals from the surface to the deepest point in each lake with a Flett (Winnipeg, MB, Canada) thermistor. Discrete water samples were collected with a 1.5-L aluminium,

drop-sleeve water bottle at 2-m intervals from 1 m below the surface to a depth of 5 m. Water transparency was estimated with a Secchi disk. During the open-water season, integrated euphotic zone samples were collected from up to five locations within each lake with a weighted Tygon tube fitted with a one-way valve. During the winter, samples were collected from the deepest part of each lake. All water samples were chilled until analysis.

Pooled integrated samples from each collection date were analysed for TP concentration following the potassium persulphate method (Menzel & Corwin, 1965) modified by Prepas & Rigler (1982). Chlorophyll *a* concentration was determined by the ethanol extraction spectrophotometric method (Bergmann & Peters, 1980). Concentrations of calcium (Ca^{2+}) in filtered water samples (Whatman (Maidstone, U.K.) GF/C filters) were measured with atomic absorption spectrophotometry (Perkin-Elmer (Norwalk, CT, U.S.A.) Model 3030 spectrophotometer) (APHA, AWWA & WPCF, 1989). Sample pH was measured with a Metrohm (Herisau, Switzerland) E588 pH meter within 1 h of collection. The DO concentration was determined by the modified Winkler titration method (Carpenter, 1965). The TP and DO concentrations in the hypolimnion were only determined for Halfmoon Lake, as Figure Eight Lake mixes frequently and does not stratify.

Winter sediment total phosphorous release rates (TPRR) were calculated as the slope of the regression line of whole-lake TP mass (g m^{-2}) against time, for the period from approximately 1 month after freeze-up

until mid-February. Summer TPRR for Halfmoon Lake were calculated for the period of distinct thermal stratification using the same method. As Figure Eight Lake does not thermally stratify, summer TPRR were not calculated. Mean winter values are from November to February, and mean summer values are from June to August. When sampling frequency was inconsistent, averages were calculated by weighing mass by time interval.

Core collection

Four sediment cores were obtained from Halfmoon Lake with a Kajak-Brinkhurst corer or a four-barrel corer on 13 July 1988, 19 September 1989, 19 September 1990 and 11 April 1993. Cores were sliced in the field at 1-cm intervals from the top of the core to a depth of 10 cm. Samples were placed in individual polyethylene bags and frozen for subsequent analysis. In the laboratory, water content was estimated by loss of mass after oven drying at 70 °C to constant weight. Dry sediments were pulverized and 1-g subsamples were extracted with aqua regia (HNO₃ and HCl); the extracts were then analysed for TP, Ca²⁺ and Fe (Prepas & Murphy, 1988; Reedyk, Prepas & Chambers, 2001).

Macrophyte collection

Macrophytes were only collected from Figure Eight Lake. Six transects were established and sampled by SCUBA divers at depths of 1 and 2 m (maximum colonization depth). Macrophytes were collected on one date only between late July and August of every sampling year (1985–93). During years of lime treatment (1986, 1987, 1990 and 1992), macrophytes were collected shortly after lime addition. Triplicate samples of above-ground standing crop were harvested from 0.25 m²-quadrats with mesh towers (to collect floating macrophytes) at each depth. Plants were identified to species, dried at 80 °C to constant weight, and then weighed. Samples were assumed to represent peak seasonal macrophyte biomass.

Statistical analysis

Water chemistry variables were analysed both qualitatively and quantitatively. Because of variations in liming and sampling dates, short-term (< 7 days)

patterns in TP, chl *a*, pH and Ca²⁺ were restricted to descriptive analyses. For the treatment lakes, annual post-treatment TP and chl *a* concentrations were compared with pre-treatment means using Dunnett's *t*-tests under analysis of variance (ANOVA). For the reference lakes, annual TP and chl *a* concentrations were compared with concentrations in 1982 (pre-treatment year for Halfmoon Lake) and 1986 (year with pre-treatment data for Figure Eight Lake).

Changes in TP and chl *a* concentrations over the treatment period were also evaluated with randomized intervention analysis (RIA) (Box & Tiao, 1975; Carpenter *et al.*, 1989). RIA is based on a comparison of a paired time-series of observations in experimental and reference systems (Carpenter *et al.*, 1989). Means for the pre- and post-treatment (after initial lime treatment) periods were calculated, and the difference between the pre (Pre) and post-treatment means (Post) was used as the test statistic, referred to as the actual difference (AD). We used the actual rather than the absolute difference, which permitted the construction of a two-tailed distribution of error. The experimental lake time-series (Halfmoon and Figure Eight lakes) were compared with the mean of the three reference lakes (North Baptiste, South Baptiste and Nakamun lakes). The probability *P* is calculated based on the rank of AD compared with the randomly generated distribution of the error (Pre) – (Post), after 999 permutations.

Winter and summer hypolimnetic TP concentrations were compared with *t*-tests. Differences in annual macrophyte biomass in Figure Eight Lake were also assessed with *t*-tests. Species composition was analysed qualitatively. Statistics were carried out with SPSS (Chicago, IL, U.S.A.) 8.0® for Windows. Results were considered significant at *P* < 0.05.

Results

Open water chemistry

In Halfmoon Lake, mean pre-liming concentrations of TP and chl *a* in 1988 were 135 and 113 µg L⁻¹, respectively (Table 1). Initial lime treatment resulted in an immediate (< 7 days) decrease in TP and chl *a* concentrations to 90 and 31% of mean pre-treatment values, respectively (Table 1). In the subsequent year of lime application (1989), TP and chl *a* concentrations were 52 and 27% of pre-liming values in 1988. In the

following two treatment years, TP and chl *a* concentrations were at times higher than in the previous year, and TP and chl *a* concentrations usually increased following lime addition. However, all post-treatment years (1989–93) had lower TP and chl *a* concentrations compared with pre-treatment values (all $P \leq 0.001$) (Fig. 1).

In Figure Eight Lake, initial pre-treatment TP and chl *a* concentrations in 1986 were 266 and 94 $\mu\text{g L}^{-1}$, respectively (Table 1). Immediately after initial application with CaCO_3 , TP concentration did not change and chl *a* concentration declined slightly to 95% of

pre-treatment values in 1986. After a second treatment with Ca(OH)_2 during the same year, TP concentration immediately decreased to 32% of pre-liming values immediately before lime addition, whereas chl *a* concentration increased by approximately threefold. In the second treatment year (1987), pre-liming TP and chl *a* concentrations were 16 and 6%, respectively, of pre-treatment values in 1986. Both variables increased immediately after lime application. Consistent with results from Halfmoon Lake, in subsequent years of lime addition, both TP and chl *a* concentrations in Figure Eight Lake typically increased

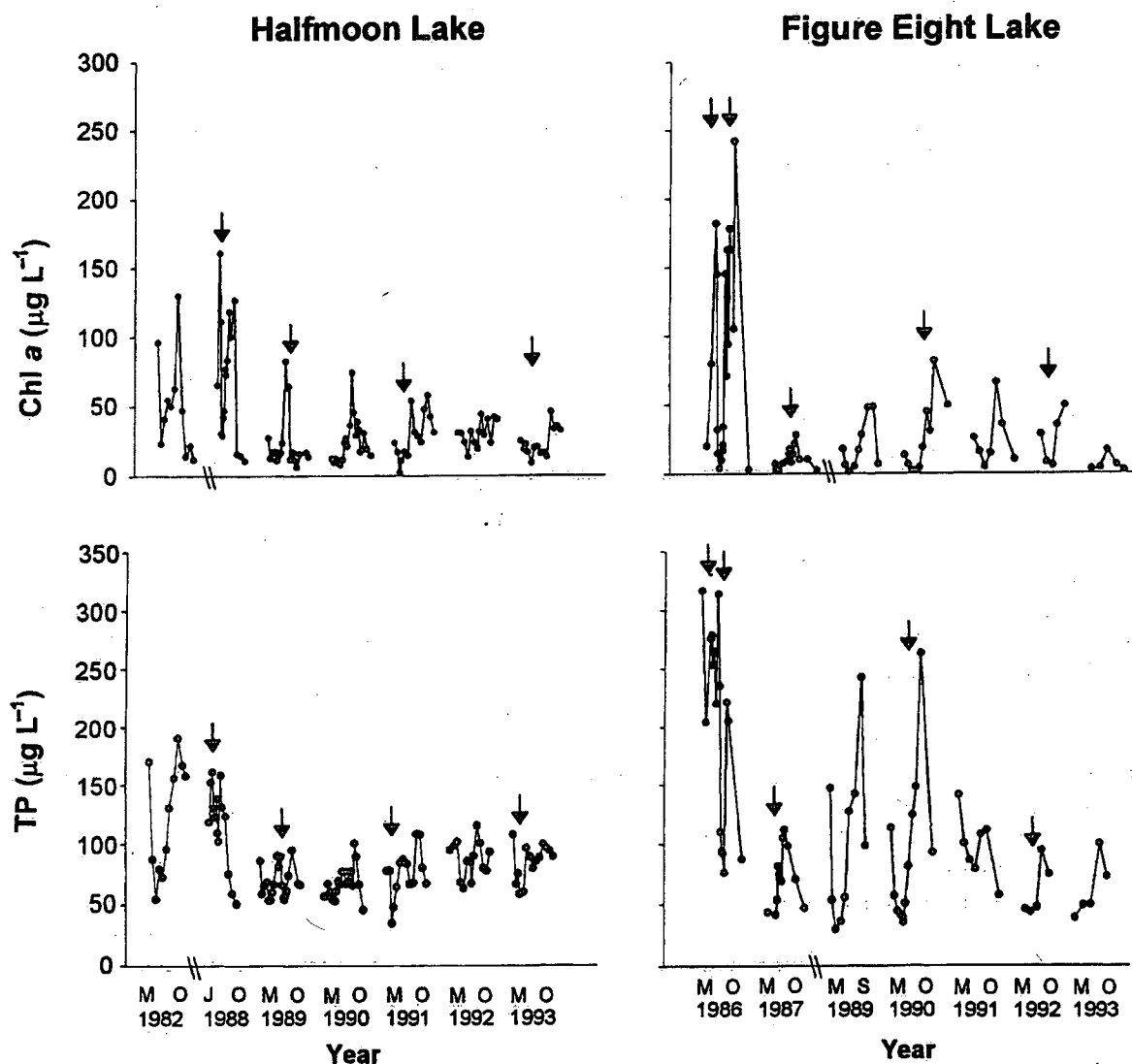


Fig. 1 Chlorophyll *a* (chl *a*) and total phosphorus (TP) concentrations in Halfmoon (1982–93) and Figure Eight (1986–93) lakes. M O, May to October; J O, July to October; M S, May to September. Arrows indicate lime application.

following the initial decline that occurred immediately after lime treatment. Again, TP and chl *a* concentrations were lower in all post-treatment years (1987, 1989–93) compared with pre-treatment values ($P < 0.001$) (Fig. 1).

There were significant declines in TP and chl *a* concentrations after lime addition in both Figure Eight and Halfmoon lakes as shown by RIA. In Figure Eight Lake, post-treatment TP concentrations declined by 91% compared with pre-treatment concentrations (AD = 142.1, $P = 0.002$), and chl *a* concentrations declined by 70% (AD = 68.9, $P = 0.009$). Similarly, in Halfmoon Lake, post-treatment TP and chl *a* concentrations declined by 77% (TP: AD = 31.3; chl *a*: AD = 24.6; $P < 0.002$) compared with pre-treatment concentrations.

Patterns in reference systems did not reflect those in treated lakes (Fig. 2). In all three reference lakes, annual TP and chl *a* concentrations did not differ (all $P > 0.05$) from concentrations in 1982, nor did they differ (all $P > 0.05$) when compared with concentrations in 1986. In the north basin of Baptiste Lake, TP and chl *a* concentrations reached peaks in 1991 of 210 and $91 \mu\text{g L}^{-1}$, respectively. In the south basin of Baptiste Lake, TP concentration reached a peak in of $193 \mu\text{g L}^{-1}$ in 1991 and chl *a* concentration reached a peak of $63 \mu\text{g L}^{-1}$ in 1987. In Nakamun Lake, both TP and chl *a* concentrations reached relatively high concentrations from 1989 to 1991. Conversely, TP and chl *a* concentrations were low between 1989 and 1991 in both treated lakes.

Overall, Ca^{2+} concentrations did not fluctuate immediately after lime addition in either of the treated lakes (Table 1). The exception was in Halfmoon Lake in 1989, when Ca^{2+} immediately declined to 52% of pre-liming concentrations that year. In general, in both treated lakes, Ca^{2+} concentrations decreased during the remainder of each application year (Table 1). Generally, pH either remained constant or increased immediately after lime application relative to pre-treatment values (Table 1). In both Halfmoon and Figure Eight lakes, pH remained at or below 9.5.

Phosphorus release rates

In Halfmoon Lake, under-ice TPRR was $3.6 \text{ mg m}^{-2} \text{ day}^{-1}$ following the summer of initial treatment (Fig. 3a). During the 1989–90 winter, under-ice TPRR

declined to $< 1 \text{ mg m}^{-2} \text{ day}^{-1}$ or 23% of 1988–89 TPRR, but steadily increased from there to 1993–94. Mean under-ice TP concentrations in 1989–90 did not change from those in 1988–89 (Fig. 3b). Similar to euphotic TP, under-ice TP increased steadily from 1990–91 to 1993–94. Between 1988–89 and 1993–94, mean DO concentration declined to 20% of the value recorded during initial treatment (Fig. 3c). In 1991–92, the average winter DO concentration was the lowest recorded in this lake during the study (0.14 mg L^{-1}), yet TPRR remained at 90% of TPRR during the winter after initial treatment, when DO averaged 1.3 mg L^{-1} . In 1993–94, DO concentration reached a second low, which corresponded to the highest winter TPRR in Halfmoon Lake.

Halfmoon Lake pre-treatment (1982) summer TPRR and hypolimnetic TP were $7.6 \text{ mg m}^{-2} \text{ day}^{-1}$ and $709 \mu\text{g L}^{-1}$, respectively (Fig. 4a,b). Between 1988 and 1993, summer TPRR decreased to $4.5 \text{ mg m}^{-2} \text{ day}^{-1}$ (59%) and hypolimnetic TP decreased to $207 \mu\text{g L}^{-1}$ (35%) of pre-treatment concentrations. Pre-treatment DO concentrations were not detectable (Fig. 4c), and this corresponded to the highest recorded TPRR and hypolimnetic TP of the study.

Sediment cores

Profiles of TP concentration in sediment cores from Halfmoon Lake showed that the lake sediments acted as a P sink. In 1988, after initial lime addition, TP concentrations were highest near the sediment–water interface, then markedly decreased with depth (Fig. 5a). Progressive burial and downward diffusion of this P was seen in the 1989 and 1990 profiles (Fig. 5a). By 1993, TP concentrations in the upper 10 cm of sediment were lower than in previous years. As open-water TP concentrations were lower in 1993 than in 1988, there was a net flux of TP into the sediments after lime addition. Sediment Fe concentrations did not demonstrate a clear pattern with time after lime addition, and thus, did not appear to be related to temporal patterns in the TP profile (Fig. 5b).

Calcium concentrations progressively increased in sediment cores in Halfmoon Lake over the period (Fig. 5c). Maximum Ca^{2+} concentrations were observed near the sediment–water interface in 1993, the year of final lime addition. Calcium was clearly accumulating in the sediments, being confined to the upper 2 cm until 1990, but being present at high concentrations

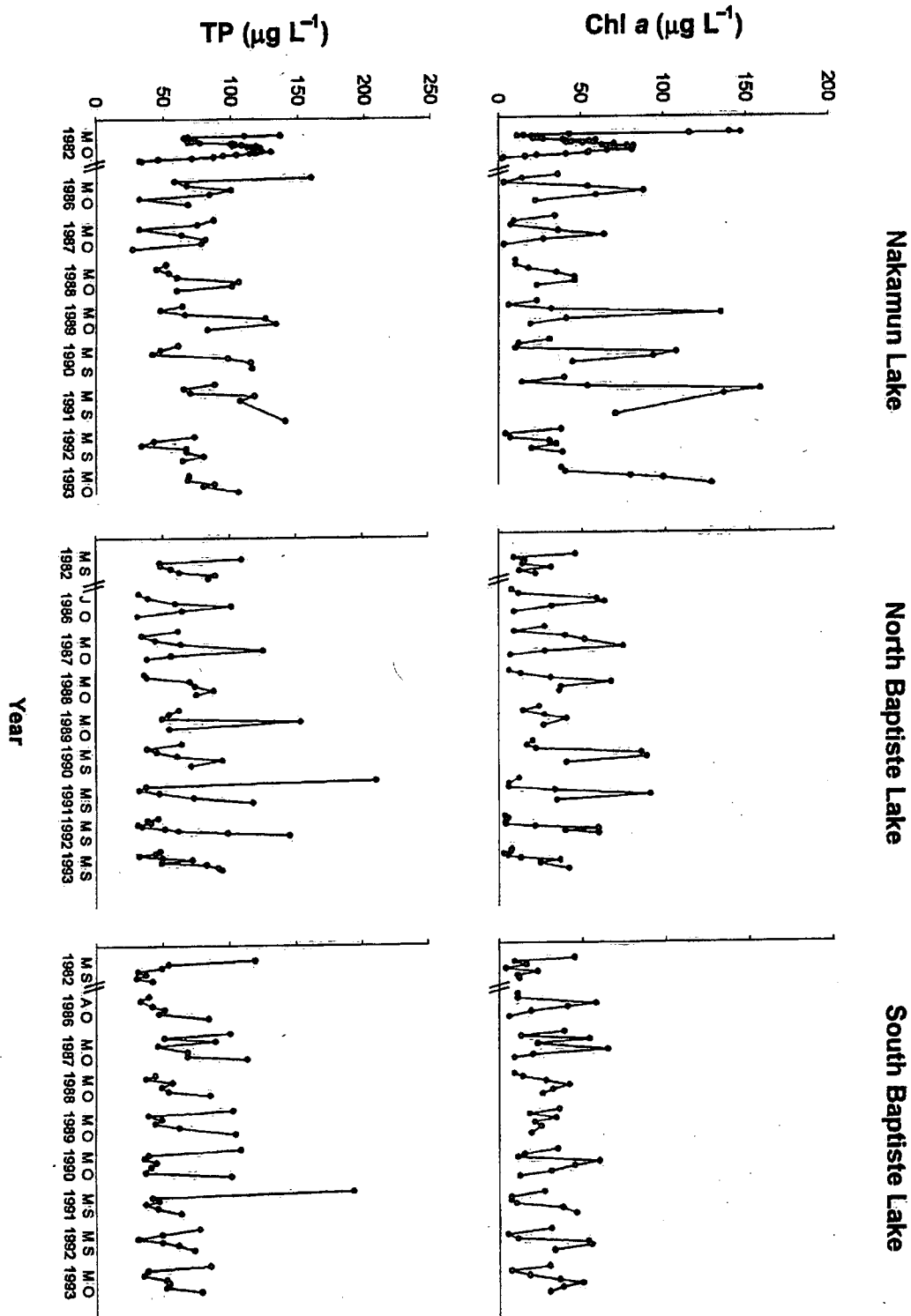


Fig. 2 Chlorophyll *a* (chl *a*) and total phosphorus (TP) concentrations in Nakamun, South Baptiste and North Baptiste lakes from 1982 to 1993. M O, May to October; M S, May to September; J O, June to October; A O, August to October.

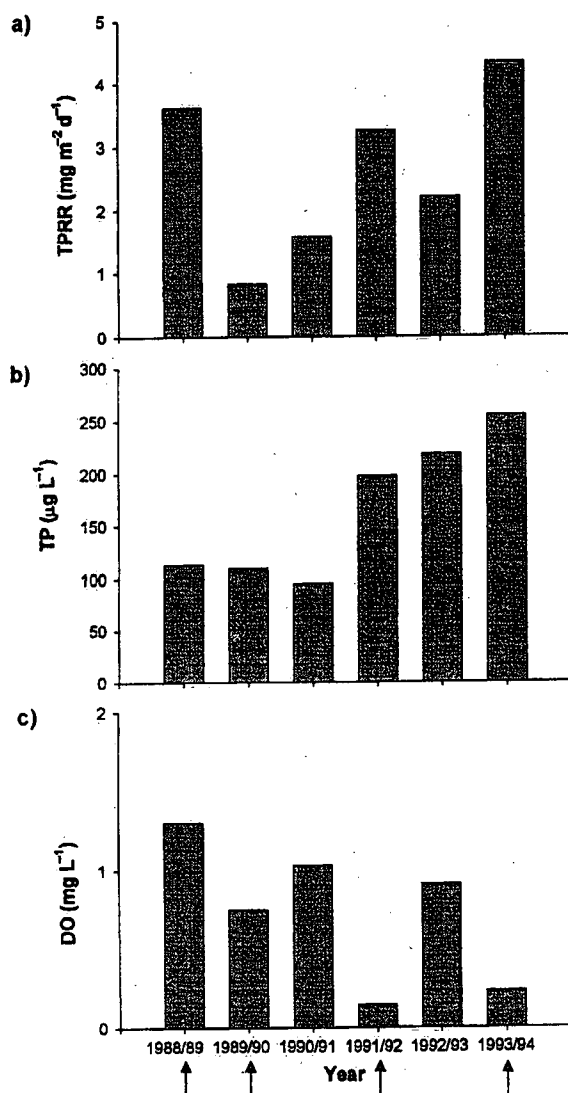


Fig. 3 Mean winter (November–February): (a) total phosphorus release rate (TPRR), (b) volume-weighted, under-ice phosphorus concentration (TP), and (c) dissolved oxygen concentration (DO) in Halfmoon Lake (1988–93). Arrows indicate years of lime application.

down to 3.5 cm in 1993. Ratios of Ca^{2+} : TP increased from 19 : 1 in 1988 to 110 : 1 in 1993.

Macrophytes

Macrophyte biomass in Figure Eight Lake declined at 2 m in the year after initial lime application and remained low until the year of final treatment (1992), when biomass had declined by 95% compared with

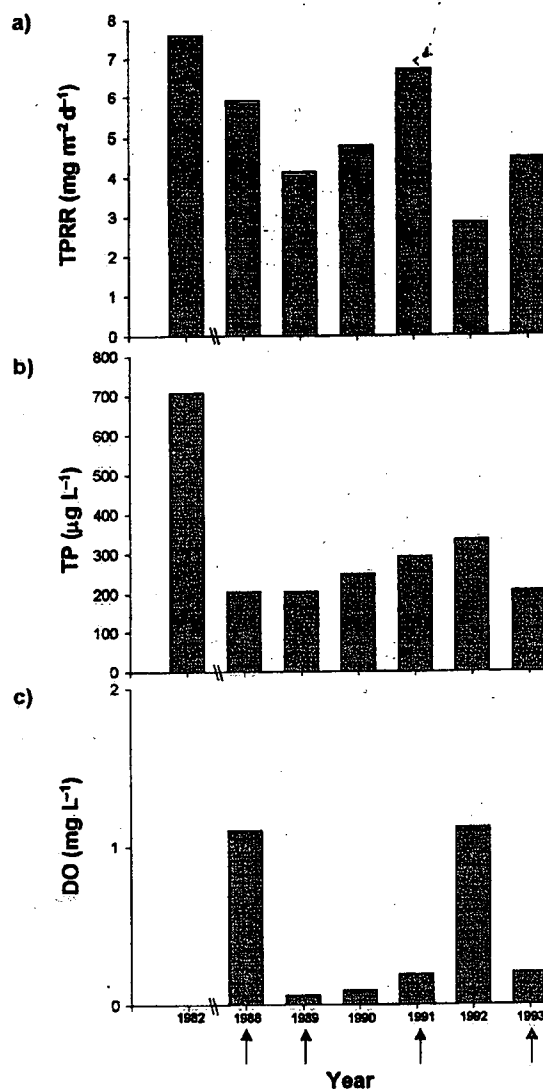


Fig. 4 Mean summer (June–August): (a) total phosphorus release rate (TPRR), (b) volume-weighted hypolimnetic phosphorus concentration (TP), and (c) hypolimnetic dissolved oxygen concentration (DO) in Halfmoon Lake (1982–93). Arrows indicate years of lime application. Dissolved oxygen was not detectable in 1982.

the initial treatment year (Fig. 6a). At a depth of 1 m, macrophyte biomass decreased after the second lime treatment. By 1989, biomass at 1 m had declined by 88% compared with the initial year of treatment, but concentrations increased in the following year (Fig. 6a). A peak in macrophyte biomass at 2 m in 1993 corresponded to the lowest average summer chl *a* concentration (Fig. 6b) and the greatest water transparency (Secchi depth of 2 m) in Figure Eight Lake

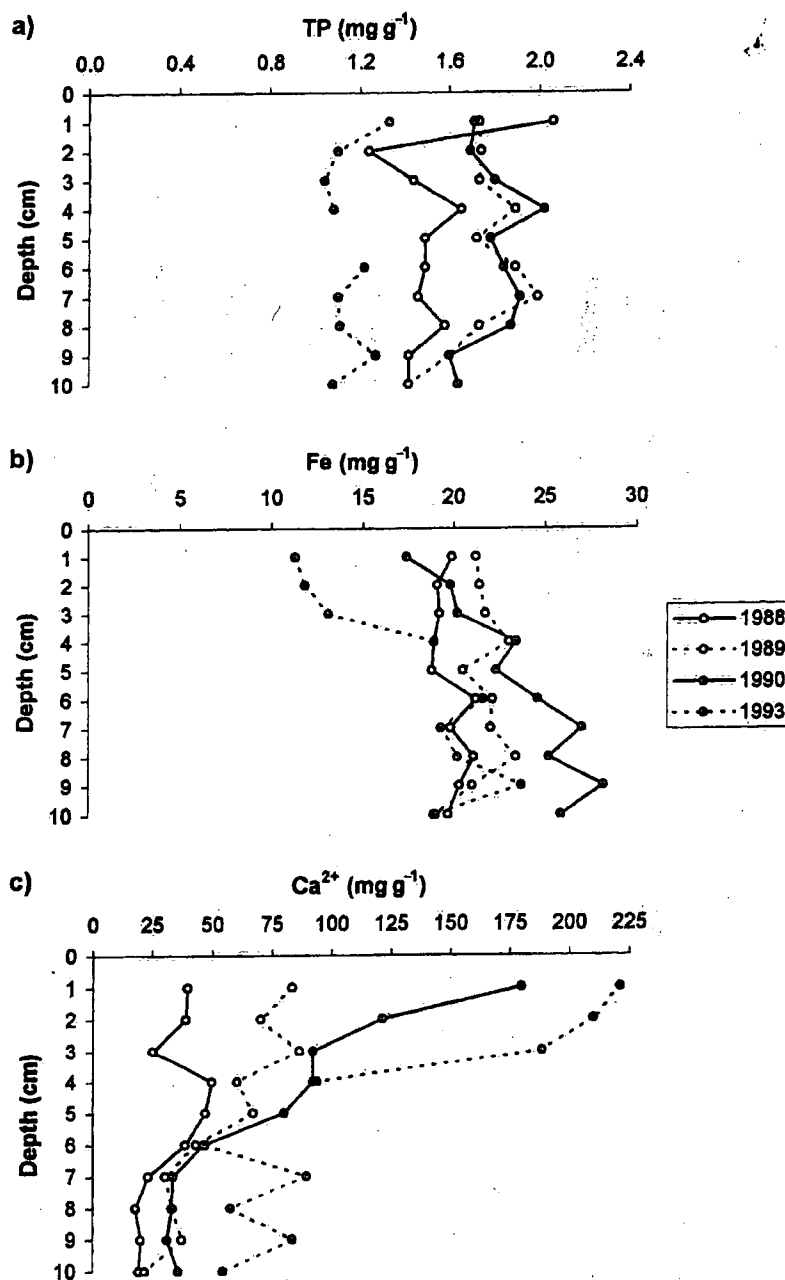


Fig. 5 Concentrations of sediment: (a) phosphorus (TP), (b) iron (Fe), and (c) calcium (Ca^{2+}) in cores from Halfmoon Lake (1988–90, 1993).

(Fig. 6c). Contrary to expectation, the deep Secchi depth reading in 1986 corresponded to a high mean chl *a* concentration. This was probably an effect of averaging, as there were a number of very high chl *a* values (up to $242 \mu\text{g L}^{-1}$) for several sampling dates, which may have skewed the mean.

Prior to 1989, *Lemna trisulca* comprised a large proportion of the total macrophyte biomass at 1 m depth; in 1986 and 1987, it comprised more than 65%

of total macrophyte biomass (Fig. 7a). At 2 m, *L. trisulca* comprised more than 85% of macrophyte biomass (Fig. 7b). At 1 m, species composition fluctuated after 1987 with a complete loss of *L. trisulca* after 1989 and increases in *Potamogeton* spp. (Fig. 7a). At 2 m, plant biomass decreased to undetectable concentrations in 1989 and 1990 (Fig. 7b). From 1991 to 1993, *Myriophyllum exalbescens* was the dominant species.

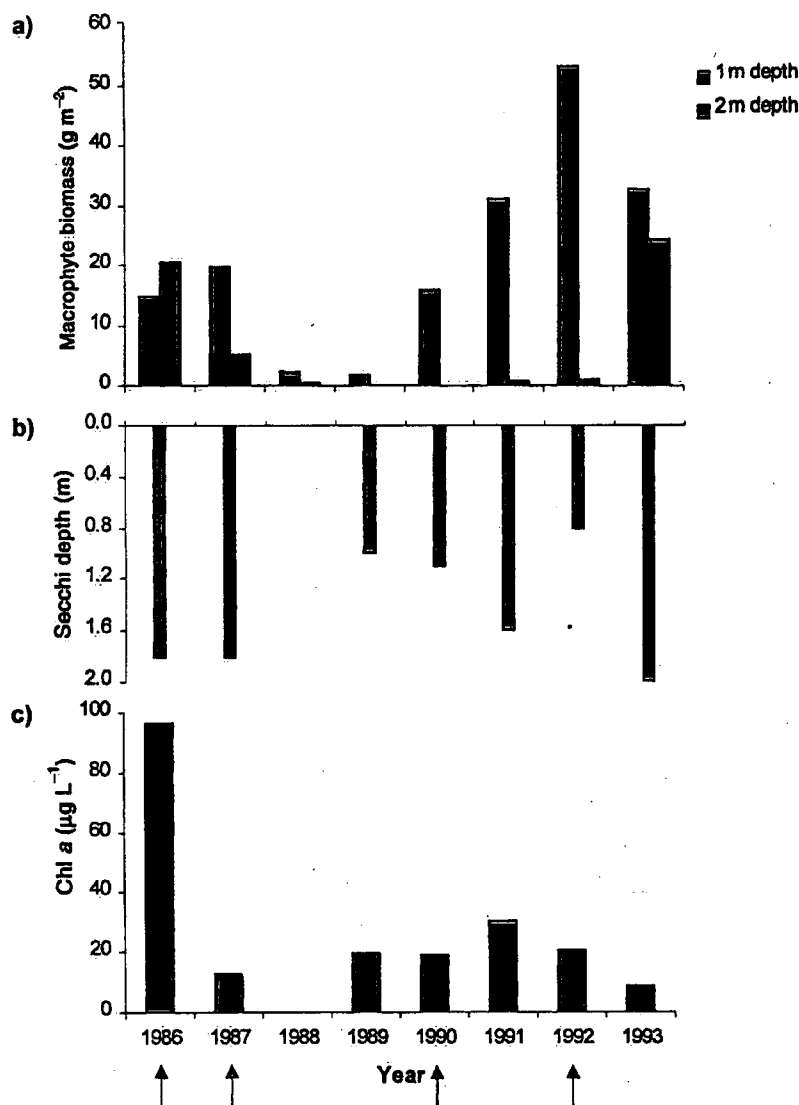


Fig. 6 Meansummer (July–August): (a) macrophyte biomass at 1 and 2-m depths, (b) Secchi depth, and (c) chlorophyll *a* (chl *a*) concentration in Figure Eight Lake from 1986 to 1993. Arrows indicate years of lime application. Macrophyte biomass was not detectable at the 2-m depth in either 1989 or 1990. Data were not available for chl *a* concentration and Secchi depth for 1988.

Discussion

After lime addition to Halfmoon Lake, euphotic TP and chl *a* concentrations both declined within 1 week of treatment. Immediate reductions in TP (Murphy *et al.*, 1988; Murphy *et al.*, 1990) and chl *a* (Babin *et al.*, 1989; Prepas *et al.*, 1990; Zhang & Prepas, 1996) have been reported for lime-treated ponds and hardwater lakes in western Canada, as the result of Ca²⁺-induced precipitation of TP and flocculation of algal cells. This process appears to be responsible for the immediate reductions in both chl *a* and TP in Halfmoon Lake. Another possibility is that Ca²⁺-precipitation reduced the concentrations of essential

growth nutrients, such as Fe, limiting phytoplankton biomass over the immediate term (Prepas *et al.*, 1990). Increased pH has been implicated in short-term chl *a* reductions. However, in this study overall whole-lake pH remained < 9.5, which is within the natural range for these lakes (Zhang & Prepas, 1996), although it could have been higher in pockets of water for brief periods of time. A lack of immediate reductions in TP and chl *a* concentrations in Figure Eight Lake following lime application may have been because of mixing events (masking TP reductions) (Prepas *et al.*, 1990), relatively low lime concentrations (13–38 mg L⁻¹), or the type of lime used (CaCO₃ alone or mixed with Ca(OH)₂) (Murphy *et al.*, 1990).

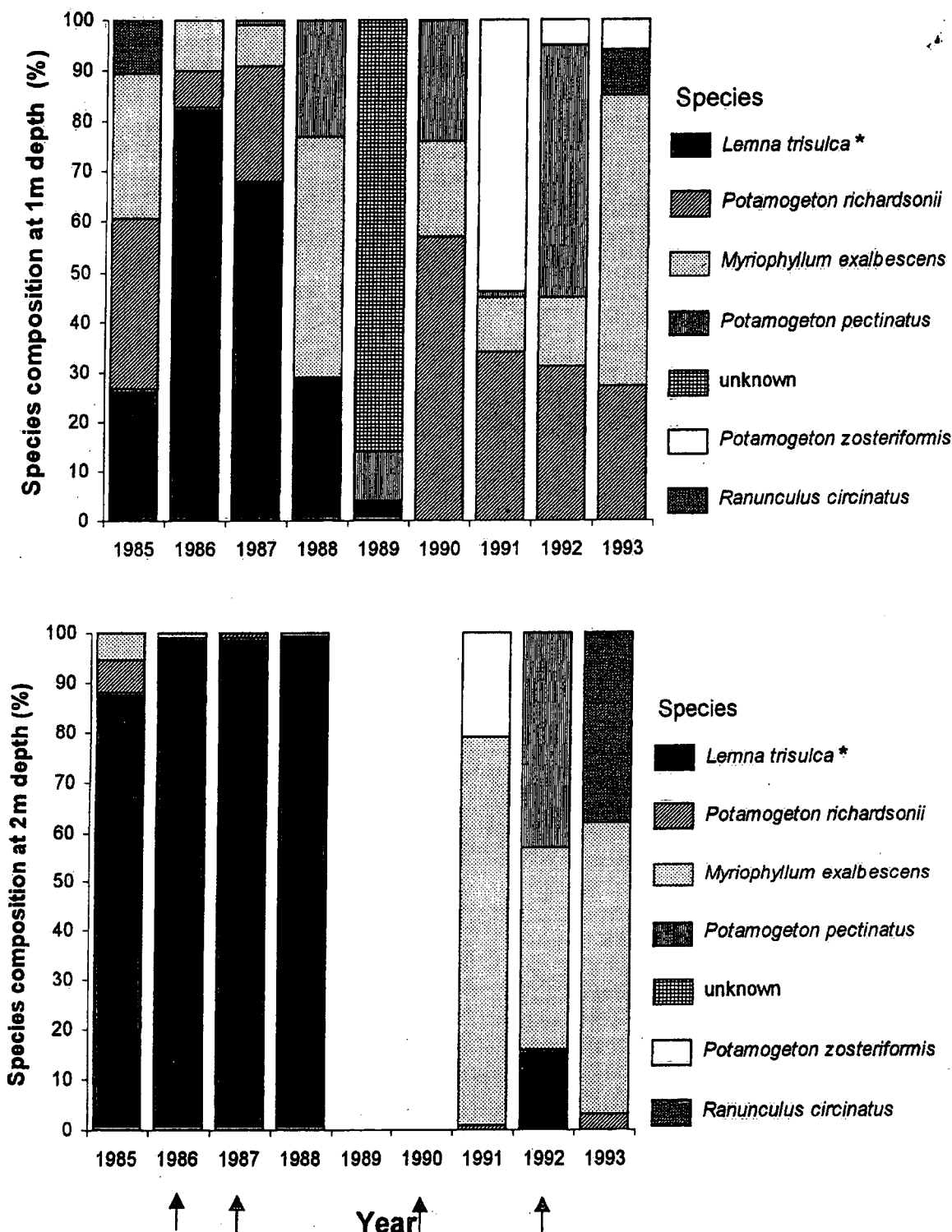


Fig. 7 Macrophyte community composition at (a) 1 m and (b) 2 m depths in Figure Eight Lake from 1986 to 1993. Macrophytes were not detected at the 2 m depth in either 1989 or 1990. *Lemna trisulca** is a floating macrophyte. Arrows indicate lime application.

Long-term decreases in mean annual TP and chl *a* concentrations were observed for both Figure Eight and Halfmoon lakes. Because these decreases were generally maintained during years when lime was not applied, mechanisms other than Ca^{2+} -induced precipitation of TP and phytoplankton cells were probably responsible for these reductions. TP is probably determining the size of the algal crop (Schindler, 1977; Prepas & Trimbee, 1988). Annual reductions in euphotic zone TP concentrations during the period of lime application probably explain long-term reductions in phytoplankton biomass in this study.

Variations in water quality in response to lime applications may be as a result of differences in lime type and concentration or, in the case of Figure Eight Lake, winter aeration and mixing events. Figure Eight Lake received, on average, lower concentrations of lime than Halfmoon Lake. Additionally, Halfmoon Lake was treated with $\text{Ca}(\text{OH})_2$ in all but the first year, whereas CaCO_3 and $\text{Ca}(\text{OH})_2$ were applied singly, and in combination, to Figure Eight Lake. Dissociation of $\text{Ca}(\text{OH})_2$ into small, newly formed CaCO_3 crystals provides a relatively large surface area for P adsorption. Thus, $\text{Ca}(\text{OH})_2$ may be more effective for removing both P and algae through calcite precipitation than direct applications with CaCO_3 (Babin *et al.*, 1989; Murphy & Prepas, 1990). Applications of CaCO_3 to Figure Eight Lake may have reduced the effectiveness of immediate treatment relative to Halfmoon Lake which received only one dose of CaCO_3 in combination with $\text{Ca}(\text{OH})_2$ in 1988. However, the generally lower and more frequent dosages of CaCO_3 may have been compensated for by winter aeration in Figure Eight Lake, which began in January 1987.

The specific mechanisms responsible for long-term reductions of TP and, consequently, chl *a* in these lakes are not completely clear. In Halfmoon Lake, overwinter TPRR was initially reduced to 23% of the value recorded in the year of initial lime application (1988), and summer TPRR was on average 59% of pre-liming concentrations. Redox mechanisms can regulate the release of sediment P (Mortimer, 1974; Burley, Prepas & Chambers, 2001), with P release dependent on both dissolved oxygen and sediment Fe concentrations. Under aerobic conditions, Fe is oxidized to Fe(III) increasing its capacity to absorb P, whereas under anoxic conditions Fe is reduced to Fe(II) thereby preventing P binding. The similar patterns of P and Fe in sediment cores suggest that

Fe is important for P regulation in Halfmoon Lake. In 1989, relatively high sediment Fe concentrations coincided with relatively low concentrations of hypolimnetic DO, yet TPRR remained relatively low. This suggests that other mechanisms were probably responsible for TPRR reductions. Hydroxyapatite (Ca-PO_4) is generated through lime addition and can bind P and reduce its release from sediments (e.g. Reedyk *et al.*, 2001). Increases in sediment Ca^{2+} concentrations in Halfmoon Lake suggest that hydroxyapatite might have reduced sediment P release, particularly during winters, when TPRR remained low despite hypoxic concentrations. Although cores were not taken from Figure Eight Lake, fluctuations in water-column Ca^{2+} , TP and chl *a* concentrations suggest that hydroxyapatite is important for regulating TPRR in this system as well (Murphy & Prepas, 1990).

Immediate and long-term reductions in TP and chl *a* concentrations following lime application did not reflect the natural patterns observed in reference systems. In both basins of Baptiste Lake and in Nakamun Lake, TP and chl *a* concentrations remained unchanged over the study period, while these parameters decreased in Halfmoon and Figure Eight lakes after lime addition. Thus, the water quality patterns observed in the treated lakes appeared unrelated to general weather-driven conditions. This suggests that reductions in phytoplankton biomass in the treated lakes were the result of lime-induced flocculation of phytoplankton cells immediately after liming and long-term reductions in nutrients, such as P. To date, long-term reductions in TP and chl *a* concentrations have been documented in lakes treated with only one dose of alum (Welch & Cooke, 1999). Our study is the first to demonstrate that successive lime applications [in particular $\text{Ca}(\text{OH})_2$] can be repeatedly effective for reducing both TP and chl *a* concentrations over periods up to 7 years. Like most chemical treatments, lime effectiveness appeared to be dependent on dosage and application frequency.

Similar to results for Figure Eight Lake, Reedyk *et al.* (2001) found that macrophyte biomass was also reduced in lakes that received single applications of lime. In contrast to treated lakes, lakes that served as reference systems showed no consistent changes in macrophyte biomass (Reedyk *et al.* 2001). Contrary to expectations, macrophyte biomass at 1 and 2 m decreased during periods of improved water clarity

and light penetration, after the first 2 years of lime treatment in Figure Eight Lake. Lime applications also appeared to affect macrophyte community composition at both depths. Reductions in *L. trisulca* biomass and increases in biomass of *Potamogeton* spp. and *M. exalbens* suggest that lime influenced plant composition in Figure Eight Lake during the treatment periods. *L. trisulca* is a floating macrophyte species that acquires nutrients from the surrounding water (Gangstad, 1986). Therefore, this species may have been more adversely affected by changes in water column nutrients than rooted species.

Three mechanisms may account for the reduction in macrophyte biomass and changes in species composition. First, lime-induced increases in pH may have impaired macrophyte photosynthesis by reducing inorganic carbon availability. Titus & Stone (1982) reported that HCO_3^- availability could limit the growth of submerged plant species such as *M. spicatum* and *Vallisneria spiralis*. However, the maximum pH values observed in both Figure Eight and Halfmoon lakes were 9.3 and 9.1, respectively, which is well within the natural range of productive hardwater lakes, although pH could have been higher in some areas for short periods of time where lime was directly applied. Other mechanisms were probably responsible for reductions in macrophyte biomass and alterations in species composition. Another possible explanation is that photosynthesis was reduced as a result of calcite settling on the plants causing shading and/or reducing gas exchange (e.g. Babin *et al.*, 1992). In the year after the last lime addition, macrophyte biomass rose sharply, exceeding pre-liming concentrations at both sampling depths, further suggesting that lime additions were responsible for plant biomass reduction. A third possibility is that lime-induced changes in sediment chemistry resulted in the observed changes in biomass and species composition. Brandrud & Roelofs (1995) found that increases in certain chemical species (CO_2 and NH_4^+) resulted in an increase in the biomass of *Juncus bulbosus* and a decrease in the abundance of other macrophytes such as *Lobelia dortmanna* and *Littorella uniflora*.

Prior to this study, the effects of lime on hardwater lakes had been restricted to short-term projects. A review of these studies strongly indicated that longer-term projects were required. Our present study demonstrates that repeated additions of lime, in particular $\text{Ca}(\text{OH})_2$, are effective for

controlling both phytoplankton and macrophyte biomass. Although this study demonstrated the effectiveness of lime types and doses on water quality, the mechanisms by which the chemicals act to reduce TP and macrophyte biomass are uncertain. Future studies should focus on these aspects, as elucidation of the mechanisms will lead to more efficient ways of improving water quality in lakes on the Boreal Plain.

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