Internal Phosphorus Loading and Alum Dosage Considerations for Long Lake, WI

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OBJECTIVES

The objectives of this investigation were to determine rates of phosphorus (P) release from sediments under laboratory-controlled aerobic and anaerobic conditions, quantify biologically-labile (i.e., subject to recycling via Eh, pH, and bacterially-mediated reactions in the sediment; loosely-bound, iron-bound, and labile organic P) P fractions in the sediment, and estimate aluminum sulfate (alum) dosage scenarios for Long Lake, Wisconsin. The specific outcomes and deliverables of this research were to,

1. examine vertical variations in biologically-labile phosphorus fractions from a centrally-located station in the profundal region of the lake to estimate the thickness of the active sediment layer the needs to be treated with alum,
2. estimate alum (as aluminum; Al) dosage scenarios for binding redox-sensitive P (i.e., the loosely-bound and iron-bound P fractions) in the upper active sediment layer, and,
3. provide cost estimates for alum dosage scenarios based on treatment areas in the lake.

APPROACH

Laboratory-derived rates of P release from sediment under aerobic and anaerobic conditions: Sediment cores were collected in late May, 2014, from centrally-located station 30 for determination of rates of P release from sediment under aerobic and anaerobic conditions (Figure 1 and Table 1). Cores were drained of overlying water and the upper 10 cm of sediment was transferred intact to a smaller acrylic core liner (6.5-cm dia and 20-cm ht) using a core remover tool. Surface water collected from the lake was filtered through a glass fiber filter (Gelman A-E), with 300 mL then siphoned onto the sediment contained in the small acrylic core liner without causing sediment resuspension. Sediment incubation systems consisted of the upper 10-cm of sediment and filtered overlying water contained in acrylic core liners that were sealed with rubber stoppers. They were placed in a darkened environmental chamber and incubated at a constant temperature (20 °C). The oxidation-reduction environment in the overlying water was controlled by gently bubbling nitrogen (anaerobic conditions, 3 replicates) or air (aerobic conditions, 3 replicates) through an air stone placed just above the sediment surface in each system. Bubbling action insured complete mixing of the water column but did not disrupt the sediment.

Water samples for soluble reactive P were collected from the center of each system using an acid-washed syringe and filtered through a 0.45 µm membrane syringe filter (Nalge). The water volume removed from each system during sampling was replaced by addition of filtered lake water preadjusted to the proper oxidation-reduction condition. These volumes were accurately measured for determination of dilution effects. Soluble reactive P was measured colorimetrically using the ascorbic acid method (APHA 2005). Rates of P release from the sediment (mg/m^2 d) were calculated as the linear change in mass in the overlying water divided by time (days) and the area (m^2) of the incubation core liner. Regression analysis was used to estimate rates over the linear portion of the data.

Phosphorus profiles and sediment chemistry: Undisturbed sediment cores were collected at four stations (Figure 1) for examination of spatial and vertical variations in sediment P fractions as a function of various depth contours in the lake (i.e., station 10 and 40 ~ 12 ft, station 30 ~ 15 ft, station 40 ~ 17 ft) for alum dosage purposes. The sediment core collected at station 30 was sectioned at 1-cm intervals over the upper 6 cm, at 2-cm intervals between 6 and 10 cm, and at 2.5-cm intervals below the 10-cm depth to determine the thickness of the excess P layer (i.e., concentrations of P are typically greatest in the upper 5 to 10 cm layer, declining to background concentrations below this depth). The upper 5-cm section from additional sediment cores collected at other stations (Figure 1) were analyzed for characterization of spatial variations in sediment chemistry.

All sections were analyzed for moisture content (%), density (g/cm^3), loss-on-ignition (i.e., organic matter content, %), loosely-bound P, iron-bound P, labile organic P, and aluminum-bound P (all expressed at mg/g). A known volume of sediment was dried at

105 ^{o}C for determination of moisture content, sediment, and bulk density and burned at 500 ^{o}C for determination of loss-on-ignition organic matter content (Håkanson and Jansson 2002). Phosphorus fractionation was conducted according to Hieltjes and Lijklema (1980), Psenner and Puckso (1988), and Nürnberg (1988) for the determination of ammonium-chloride-extractable P (loosely-bound P), bicarbonate-dithionite-extractable P (i.e., iron-bound P), and sodium hydroxide-extractable P (i.e., aluminum-bound P). A subsample of the sodium hydroxide extract was digested with potassium persulfate to determine nonreactive sodium hydroxide-extractable P (Psenner and Puckso 1988). Labile organic P was calculated as the difference between reactive and nonreactive sodium hydroxide-extractable P.

The loosely-bound and iron-bound P fractions are readily mobilized at the sediment-water interface as a result of anaerobic conditions that lead to desorption of P from sediment and diffusion into the overlying water column (Mortimer 1971, Boström 1984, Nürnberg 1988). The sum of the loosely-bound and iron-bound P fraction represents redox-sensitive P (i.e., the P fraction that is active in P release under anaerobic and reducing conditions; redox-P). In addition, labile organic P can be converted to soluble P via bacterial mineralization (Jensen and Andersen 1992) or hydrolysis of bacterial polyphosphates to soluble phosphate under anaerobic conditions (Gächter et al. 1988, Gächter and Meyer 1993, Hupfer et al. 1995). The sum of redox-sensitive P and labile organic P collectively represent biologically-labile P. This fraction is active in recycling pathways that result in exchanges of phosphate from the sediment to the overlying water column and potential assimilation by algae. In contrast, aluminum-bound P is more chemically inert and subject to burial rather than recycling.

Al dosage determination: The upper 5-cm section of sediment collected at station 30 (Figure 1) was subjected to a range of aluminum sulfate (as Al) concentrations to determine the Al dosage required to inactivate the redox-P fraction (Rydin and Welch 1999). Alum (as aluminum sulfate; $Al_2(SO_4)_3 \cdot 18\ H_2O$) was combined with 0.1 M sodium bicarbonate ($NaHCO_3$) to a concentration of 1.25 g Al/L to form an aluminum hydroxide ($Al(OH)_3$) floc. Aliquots of this solution, diluted to a final volume of 10 mL with distilled water, were added to centrifuge tubes containing the equivalent of 0.025 g dry weight (DW) of fresh sediment to obtain Al concentrations ranging from 0 (i.e., control) to ~ 130 mg Al/g DW sediment. The assay tubes were shaken for a minimum of 2 hours at 20 °C in a darkened environmental chamber, centrifuged at 500 g to concentrate the sediment, and decanted for redox-P determination (see method description above).

Al dosage was estimated as the concentration (g/m^2) required to bind at least 90% of the redox-P in the excess P layer. The dry mass concentration of redox-P (mg/g) was converted to an areal concentration (g/m^2) as,

$$\text{Redox-P } (g/m^2) = \text{Redox-P } (mg/g) \cdot \rho\ (g/cm^3) \cdot \theta \cdot h\ (m) \cdot 1{,}000{,}000\ (cm^3/m^3) \cdot 0.001\ (g/mg) \qquad 1)$$

where, ρ is sediment bulk density (g/cm^3), θ is the percentage of sediment solids (100 – percent moisture content; dimensionless), and h is the thickness of the excess P layer determined from the sediment vertical profile at station 30 (m). The Al concentration (g/m^2) was estimated as,

$$\text{Al } (g/m^2) = \text{Redox-P } (g/m^2) \cdot \text{Al:P}_{90\%} \qquad 2)$$

where, $\text{Al:P}_{90\%}$ is the binding ratio required to adsorb at least 90% of the redox- P in the sediment. Gallons of alum required to treat profundal sediment and generic cost scenarios were estimated by determining the sediment area (m^2) that was probably contributing to internal P loading. Sediment area to be treated was estimated by evaluating the seasonal depth and area of anoxia during the summer.

Maximum allowable Al dosage based on alkalinity and pH in the lake: Addition of aluminum sulfate to a lake leads to hydrolysis and the liberation of hydrogen ions which lowers the pH of the water column. Since Al toxicity to the biota can occur if the pH falls below ~4, maintaining a pH ≥ 6.0 as a margin of safety should also be considered in dose determination (Cooke et al. 2005). For situations where alkalinity is low or the required dosage exceeds the maximum allowable dosage to maintain pH ≥ 6.0, a buffered aluminum sulfate-sodium aluminate treatment will be needed to maintain pH near neutrality. Surface water collected from the lake was analyzed for total alkalinity and pH according to APHA (2005). A titration procedure was used to determine the maximum allowable dosage of aluminum sulfate that can be added and yet maintain pH above 6.0 (Cooke et al. 2005). A 1.25 g Al/L solution of $Al_2(SO_4)_3 \cdot 18\ H_2O$ was used as the titrant and 1.0 mL additions to 500 mL of lake water were each equivalent to 2.5 mg Al/L. Lake water was titrated with the Al solution until an endpoint of pH 6 was reached. A 1.0 mL aliquot of this solution added to 500 mL of lake water is equivalent to 2.5 mg Al/L. The total volume of Al solution needed to titrate lake water to pH 6 was multiplied by 2.5 mg Al/L to estimate the maximum allowable concentration. This calculation was then compared with estimates based on sediment redox-P to ensure that the latter was at or below the maximum allowable dosage. Caution needs to be used because a vertical alkalinity and pH profile over the entire vertical water column needs to be estimated in order to more accurately evaluate the maximum allowable dosage.

RESULTS AND INTERPRETATION

Diffusive phosphorus flux and sediment spatial and vertical characteristics

P mass and concentration increased approximately linearly in the overlying water column of station 30 sediment systems maintained under anaerobic conditions (Figure 2). The mean P concentration maximum in the overlying water at the end of the incubation period (i.e., day 8) was high at 1.155 mg/L (± 0.056 standard error; SE; Table 3). The mean rate of P release under anaerobic conditions was also high at 12.9 mg/m^2 d (± 0.8 SE; Table 3) and indicative of eutrophic conditions (Nürnberg 1988).

Although P accumulation in the overlying water column was much less under aerobic conditions, substantial P diffusion from sediment nevertheless occurred, resulting in a moderately high mean P concentration maximum of 0.089 mg/L (± 0.012 SE) in the overlying water column at the end of the incubation period. The mean rate of P release under aerobic conditions was also relatively high at 1.06 mg/m^2 d (± 0.01 SE; Table 3) and, thus, represented a potentially important internal source of P loading to the lake.

Typically, rates of P release are higher under anaerobic versus aerobic conditions, due to binding of P onto iron-oxyhydroxides (Fe~(OOH)) in the sediment oxidized microzone under the latter condition and suppression of diffusive flux into the overlying water column. Indeed, diffusional P flux from sediment can be negligible under aerobic conditions when Fe concentrations are sufficiently high in relation to P (i.e., Fe:P binding ratio > 10:1; Jensen et al. 1992). In contrast, Long Lake sediments appeared to act as a significant net P source even under aerobic conditions, a finding that needs to be considered in algal bloom management.

Moisture content was relatively high in the upper 5-cm sediment section at all stations, often exceeding 93% (Table 4). This pattern was indicative of fine-grained and very flocculent surface sediment. Vertically through the sediment column at station 30, moisture content was greatest at the sediment-water interface and declined slightly with sediment depth, a pattern attributed to compaction (Figure 3). Sediment wet bulk density (i.e., the mass of dry sediment and associated interstitial water per cm^3) exhibited the opposite pattern (Figure 3). The density was lowest within the surface sediment section at 1.02 g/cm^3, reflecting high interstitial water content (i.e., the density of water is ~ 1.0 g/cm^3), and increased gradually over the upper 6-cm sediment column (Figure 3). Ideally, sediment wet bulk density should be lower in relation to the density of the Al floc in order to promote its rapid sinking through the upper excess P layer for maximum binding efficiency. Stronger gradients of increasing density were observed between the 6- and 14-cm depths. The organic matter content of sediment sections collected at station 30 was moderately high in the upper 6-cm layer, exceeding 30% (Table 4 and Figure 3). Organic matter content generally declined as a function of increasing sediment depth, suggesting breakdown and mineralization of older deposits (Figure 3).

Loosely-bound P concentrations were relatively low in the upper 5-cm sediment layer at all stations and only represented ~ 6% to 14% of the redox-sensitive P concentration (i.e., the sum of loosely-bound and iron-bound P), due to much high concentrations of iron-bound and labile organic P (Figure 4). Loosely-bound P ranged between 0.018 mg/g and ~0.035 mg/g in the upper 5-cm section over all stations. Concentrations also tended to be higher at the centrally-located stations 20 and 30 versus the other stations (Table 4). Iron-bound P accounted for > 90% of the redox-sensitive P in the upper 5-cm sediment layer at all sediment sampling locations (Table 4 and Figure 4). In addition, concentrations were relatively high at stations 20 and 30 (~0.50 mg/g), versus much lower iron-bound P concentrations in the surface sediment located at stations 10 and 40, similar to spatial patterns observed for the loosely-bound P fraction. Overall, loosely-bound P accounted for only ~ 4% of the biologically-labile P fraction (i.e., the sum of the loosely-bound, iron-bound, and labile organic P fractions; Figure 4), By contrast, iron-bound P represented ~ 24% to 60%, and labile organic P ranged between 56% and 72%, of the biologically-labile P fraction. Iron-bound P was also the dominant biologically-labile P fraction at stations 20 and 30. Although concentrations were similar across all stations, the labile organic P fraction represented most of the biologically-labile P at stations 10 and 40 (Figure 4 and 5). Overall, the biologically-labile P concentration was greatest at station 20 and 30, approaching ~ 0.90 mg/g, and much lower at stations 10 and 40 (0.5 mg/g and 0.47 mg/g, respectively; Figure 5).

Vertically in the sediment column for the core collected at station 30, redox-sensitive and biologically-labile sediment P fractions exhibited concentration peaks over the upper 6 cm (Figure 6). In particular, iron-bound P was the dominant fraction within this sediment stratum and concentrations were high near the sediment surface, exceeding 1.0 mg/g (Figure 6). Concentrations declined to background levels below the 6-cm depth and were relatively constant. This vertical concentration profile indicated the buildup of biologically-labile P and redox-sensitive P near the sediment surface in excess of breakdown and burial (i.e., excess P), a pattern typically observed in eutrophic lake sediments (Carey and Rydin 2011). Others have shown that the mass of excess P in the surface sediment layer closely approximated the annual gross internal P loading rate (Rydin et al. 2011; Malmaeus et al. 2012). Overall, the vertical profile for redox-P suggested that inactivation of redox-sensitive P in the upper 6-cm sediment layer needed to be considered in aluminum sulfate dosage estimation to control internal P loading (Figure 6).

Alum dosage estimation and cost analysis

Alum dosage was based on the Al concentration required to inactivate the redox-sensitive P fraction in the upper 6-cm sediment layer (Table 5). For station 30 (i.e., the deepest area of the lake), the estimated required Al concentration was ~ 105 g/m^2 (Table 5). Because redox-sensitive P concentrations were lower for station 10 and 40 sediment, the estimated Al concentration was less. However, since average summer bottom water anoxia typically extends to the 4.0- to 4.5-m depth (Clemens 2013), anaerobic sediments located below the 15-ft depth contour are probably contributing to internal P loading and should be considered for Al treatment. Since results from stations 20 and 30 were most representative of sediment chemical characteristics below the 15-ft contour, an Al dosage concentration of 105 g/m^2 was recommended. The estimated cost to inactivate redox-sensitive P in the upper 6-cm sediment layer within the 15-ft contour of the lake was ~$259,000, including generic setup fee (Table 6). Costs for treating sediment below the 10-ft contour are also included in Table 6.

Recent lake Al dosage estimates have ranged between ~ 95 g Al/m^2 and ~140 g Al/m^2 (Table 7). These more recent Al dosage ranges are generally higher compared to historical ranges (Huser 2012) because they were targeted toward inactivation of the excess P pool in the sediment. The proposed Al dosage of ~ 105 g/m^2 to treat the upper 6-cm sediment layer in the profundal, anoxic zone of Long Lake falls within these recent ranges.

The total alkalinity at the time of sediment core sampling was moderately low at ~70 mg $CaCO_3$/L, suggesting low buffering capacity for regulating pH during alum application. Al binding of P is most efficient within a pH range of 6 to 8. As pH declines below 6, Al becomes increasingly soluble (as Al^{3+}) and toxic to biota. The maximum allowable Al dosage that could be applied and yet maintain pH at or above 6, determined via jar tests (Cooke et al. 2005), was low at only 10 mg Al/L (Table 8). Cooke et al. (2005) reported that treatment longevity (i.e., years of successful P control) generally coincided with Al dosages greater than ~ 12 to 18 g/m^3 for stratified lakes (range = 11.7 to 30 g/m^3; Table 8). The overall estimated volume-based Al dosage of 22 mg/L was higher than the maximum allowable Al dosage. This pattern was due to the relatively low alkalinity and, thus, low buffering capacity of lake water. Thus, there would be potential concerns regarding low pH during application if the entire dosage were applied in one treatment. This concern would be alleviated by splitting the application into smaller doses over several years (see below). An additional alkalinity-pH vertical profile would need to be examined during the spring to early summer period to verify and refine the maximum allowable Al dose.

The overall desired objective of an alum application in Long Lake is to have the $Al(OH)_3$ floc sink through the upper 6-cm sediment layer and bind the redox-sensitive P that is contributing to internal P loading and algal bloom development. In order to meet these objectives, the Al floc needs to be denser than the upper sediment layer and sink through that layer relatively quickly (i.e., within 3 months or less). Recent research has suggested that $Al(OH)_3$ binding efficiency for P decreases significantly (i.e., > 75% decrease) if it has not been exposed to and reacted with sediment redox-sensitive P within 90 days, due to changes in crystalline structure in the absence of adsorbed P (de Vicente et al. 2008a). Furthermore, as binding sites on the $Al(OH)_3$ floc become saturated with redox-sensitive P, additional P diffusing into the alum layer from deeper sediments over time can become re-adsorbed to Fe~(OOH) (i.e., redox-sensitive P; Lewandowski et al. 2003), eventually diffuse out of the sediment under anaerobic and reducing conditions, and again become an important internal P loading source years after alum treatment.

One current unknown is the exact $Al(OH)_3$ floc density after application, reaction with water, and deposition to the sediment. However, preliminary indications are that Al floc density is very low during the first 6 to 12 months after application. Thus, surface sediment wet bulk density should ideally be very low and moisture content high, on the order of 95% or greater, in order to promote rapid sinking and exposure of the Al floc to redox-sensitive P. Even though Long Lake surface sediments exhibited a relatively high moisture content and low wet bulk density, the Al floc may only sink through the upper 2-cm during the first year of application (i.e., Figure 3).

To date, there is no universally accepted and proven alum application strategy to maximize P binding effectiveness and longevity. Although there have been instances of multiple applications over a period of years (Lewandowski et al. 2003), generally, lake Al treatments in the upper midwestern United States have been one-time applications. In addition to the density and sinking concerns identified above, input of new sediment from the watershed can accrete over the $Al(OH)_3$ floc over time, reducing treatment effectiveness and longevity (Lewandowski et al. 2003, Cooke et al. 2005). For shallow lakes such as Long Lake, resuspension of sediments from shallower regions and focusing to deeper treated areas would also represent a significant input of new sediment that could bury the $Al(OH)_3$ floc (Huser et al. 2011). As mentioned earlier, upward diffusion of P through the alum layer from deeper sediments can eventually lead to P flux into the overlying water column, depending on the extent of binding site saturation by P on the $Al(OH)_3$ floc. Although the $Al(OH)_3$ floc continues to adsorb P for years (Lewandowski et al. 2003), its P binding efficiency apparently decreases over time (de Vicente et al. 2008a). If the $Al(OH)_3$ floc does not entirely sink through the excess P layer and, instead, stabilizes on top of sediments with high redox-sensitive P concentration, upward P diffusion from deeper sediment layers could eventually overwhelm the capacity of the $Al(OH)_3$ floc to bind this additional sediment P source. De Vicente et al. (2008b) suggested that smaller doses spread out over several years might maintain higher P binding efficiencies. More research is clearly needed to develop effective application strategies to maximize internal P loading reduction and extend Al treatment success and longevity.

To alleviate low pH and toxicity issues during treatment and promote maximum P binding efficiency on the $Al(OH)_3$ floc, the Long Lake Protection and Rehabilitation

District could consider multiple alum applications over a period of years and an adaptive management approach to assess P binding effectiveness, longevity of internal P loading control, and Al dosage needs for future treatments. For instance, half the dose (~50 g/m^2) could be applied over the 15-ft depth contour and allowed to react and bind with redox-sensitive P for several years before application of a second dose. A future second treatment should occur after P from deeper sediment layers has clearly diffused through the $Al(OH)_3$ floc and become an important internal P loading source during bottom water anoxia. Since this transition may take several years, laboratory sediment P flux and sediment vertical profiles should be monitored annually to more accurately determine the optimal timing of application and Al dosage for future treatment. An example Al application and monitoring scenario is shown in Table 9.

Finally, aerobic sediments in Long Lake also potentially contribute P to the water column, although at much lower rates. Overall internal P loading contributions from sediment determined via laboratory incubations (this study) need to be compared with budgetary and empirical modeling estimates in order to more accurately evaluate potential P contributions from aerobic sediments that may be driving algal blooms in the lake. Treatment of shallower sediments would be much more costly. In addition, because Long Lake is relatively shallow with a long fetch along the NW-SE wind rose, effects of sediment resuspension and focusing (i.e., movement and transport of sediment during wind-generated turbulence) processes on the distribution of the $Al(OH)_3$ floc would need to be considered (Egemose et al. 2009, 2013). Although shallow lakes can be treated successfully with alum, application should be timed to occur during extended calm periods to allow for sedimentation of the $Al(OH)_3$ floc and sinking into sediment. In addition, significant resuspension of the $Al(OH)_3$ floc into the water column during periods of elevated pH (i.e., > 8.5-9.0) caused by high algal productivity during the could lead to some Al dissolution and desorption of P (Reitzel et al. 2013). Thus, a late Fall application prior to ice formation may best address both concerns.

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Table 1. Station identification labels and numbers of sediment cores collected for determination of rates of phosphorus (P) flux under aerobic or anaerobic conditions, biologically-labile P fractions, and the dosage of aluminum (Al) required to bind redox-sensitive P.

Station	P Flux		P fractions		Al dosage
	Aerobic	Anaerobic	upper 5 cm	Vertical profile	
10			1		
20			1		
30	3	3	1	1	1
40			1		

Table 2. Sediment physical-textural characteristics, phosphorus species, and metals variable list.

Category	Variable
Physical-textural	Moisture content
	Wet and dry sediment bulk density
	Organic matter content
Phosphorus species	Loosely-bound P
	Iron-bound P
	Labile organic P
	Aluminum-bound P

Table 3. Mean (1 standard error in parentheses; n = 3) rates of phosphorus (P) release under aerobic and anaerobic conditions and mean P concentration (n = 3) in the overlying water column near the end of the incubation period for intact sediment cores collected at Station 30 in Long Lake, WI.

Station	Diffusive P flux			
	Aerobic		Anaerobic	
	(mg/m^2 d)	(mg/L)	(mg/m^2 d)	(mg/L)
30	1.06 (0.01)	0.089 (0.012)	12.9 (0.8)	1.155 (0.056)

Table 4. Vertical variations in physical-textural characteristics, loosely-bound phosphorus (Loose-P), iron-bound P (Fe-P), labile organic P, aluminum-bound P (Al-P), redox-sensitive P (i.e., the sum of the loosely-bound and iron-bound P fractions; redox-P, and biologically-labile P (i.e., redox-P plus labile organic P) at sediment sampling stations in Long Lake.

Station	Depth	Section		Moisture	Sediment Density		Organic	Loose-P	Fe-P	Labile org P	Al-P	Redox-P	Biol-labile P
		Top	Bottom	content	Dry	Wet	matter						
	(m)	(cm)	(cm)	(%)	(g/cm^3)	(g/cm^3)	(%)	(mg/g)	(mg/g)	(mg/g)	(mg/g)	(mg/g)	(mg/g)
10	4.2	0	5	92.8	0.076	1.033	27.5	0.018	0.200	0.283	0.076	0.218	0.501
20	5.0	0	5	93.1	0.071	1.031	27.7	0.035	0.488	0.393	0.090	0.523	0.916
30	5.7	0	1	95.0	0.051	1.021	34.0	0.128	1.050	0.532	0.224	1.178	1.710
		1	2	94.2	0.060	1.024	33.6	0.081	1.061	0.405	0.167	1.142	1.547
		2	3	93.6	0.067	1.027	33.3	0.086	1.147	0.372	0.183	1.233	1.605
		3	4	93.1	0.072	1.030	32.3	0.053	0.675	0.389	0.137	0.728	1.117
		4	5	92.5	0.078	1.032	31.6	0.031	0.333	0.308	0.097	0.364	0.672
		5	6	92.0	0.084	1.035	30.9	0.023	0.226	0.263	0.122	0.249	0.512
		6	8	90.7	0.098	1.043	27.8	0.019	0.204	0.227	0.089	0.223	0.450
		8	10	89.5	0.111	1.050	26.4	0.016	0.185	0.189	0.077	0.201	0.390
		10	12.5	89.0	0.118	1.053	25.6	0.017	0.147	0.160	0.082	0.164	0.324
		12.5	15	88.2	0.127	1.057	26.1	0.032	0.340	0.182	0.185	0.372	0.554
		15	17.5	88.1	0.128	1.056	27.9	0.062	0.523	0.178	0.368	0.585	0.763
		1.5	20	88.1	0.128	1.056	28.3	0.017	0.130	0.181	0.113	0.147	0.328
40	5.3	0	5	93.5	0.068	1.027	35.1	0.018	0.113	0.335	0.071	0.131	0.466

Table 5. Estimated alum (as Al) dosage to inactivate redox-sensitive P in the upper 6-cm sediment layer.

Station	Redox-P		Al dose
	(mg/g)[1]	(g/m^2)	(g/m^2)
10	0.218	0.973	77
20	0.523	2.232	102
30	0.746	2.870	105
40	0.492	0.492	54
Recommended			105

[1] average concentration over the upper 6 cm.

Table 6. Approximate cost scenario to treat the upper 6-cm sediment layer in the profundal anoxic zone located below the 10-ft and 15-ft depth contours in Long Lake with aluminum sulfate.

Variable	Depth contour	
	10-ft	15-ft
Treatment area (acres)	167	84
Al dosage (g/m^2)	105	
Alum ($)	$496,000	$249,000
Setup ($)	$10,000	
Total ($)	$506,000	$259,000

Table 7. Recent and proposed alum (as Al) dosages for various lakes. An asterisk denotes a future treatment.

Lake	Al Dose ($g\ Al\ m^{-2}$)	Reference
Long Lake, WI	105	Present study
Recent treatments		
Black Hawk, MN	145	(unpubl. data)
Tiefwarensee, Germany	137	Wauer et al. (2009)
East Alaska, WI	132	Hoyman (2012)
Half Moon, WI[1]	115	James (2011)
Susser See, Germany	100	Lewandowski et al. (2003)
Green, WA	94	Dugopolski et al. (2008)
Proposed future treatments		
Big Chetac, WI	135	(unpubl. data)
Squaw, WI*	120	(unpubl. data)
Cedar, WI*[2]	116	(unpubl. data)
Halsted's Bay, Minnetonka, MN*[3]	105	(unpubl. data)
Bald Eagle, MN*	100	(unpubl. data)

[1] West and east arm dosages were 150 and 75 g/m^2, respectively

[2] Average of a stratified treatment at 130 and 100 g/m^2

[3] Average of a stratified treatment at 140 and 80 g/m^2

Table 8. A comparison of the maximum allowable Al dose, based on a titration assay and nomograph estimate presented in (Cooke et al. 2005) and the the areal sediment redox-P based Al dosage converted to a concentration for the profundal anoxic region of Long Lake. Al dosages and longevity for other unstratified and stratified lakes are from Cooke et al (2005). Numbers on parentheses denote percent reductions in in total phosphorus. Longevity = as of publication of Cooke et al. (2005).

Lake		Al Dose ($g\ Al/m^3$)	Observed Longevity (years)
Long Lake	Maximum allowable	10	
	105 $g\ Al/m^2$ over the 10-ft contour	27	
	105 $g\ Al/m^2$ over the 15-ft contour	22	
Unstratified lakes	Long Kitsap County	5.5	11(30%)
	Pickerel	7.3	<1
	Long Thurston County North	7.7	>8 (56%)
	Pattison North	7.7	7 (29%)
	Wapato	7.8	<1
	Erie	10.9	>8 (75%)
	Campbell	10.9	>8 (46%)
Stratified lakes	Eau Galle	4.5	<2
	Morey	11.7	8 (60%)
	Cochnewagon	18	6 (not reported)
	Dollar	20.9	18 (68%)
	Annabessacook	25	13 (41%)
	West Twin	26	18 (66%)
	Irondoquoit Bay	28.7	5 (24%)
	Kezar	30	9 (37%)

Table 9. An example adaptive management scenerio approach in which 2 smaller Al doses are applied to Long Lake profundal sediments. The second Al application and dose is determined via annual sediment profile monitoring. In this example, a smaller Al dose was applied during year 1. Annual sediment core vertical profiling indicated that a second application should occur in year 4 at a specified dose.

Variable	Year 1	Year 2	Year 3	Year 4	Year 5
Al application	▨			▨	
Assessment[1]	▨	▨	▨	▨	▨

[1] Sediment core collection and vertical profile monitoring

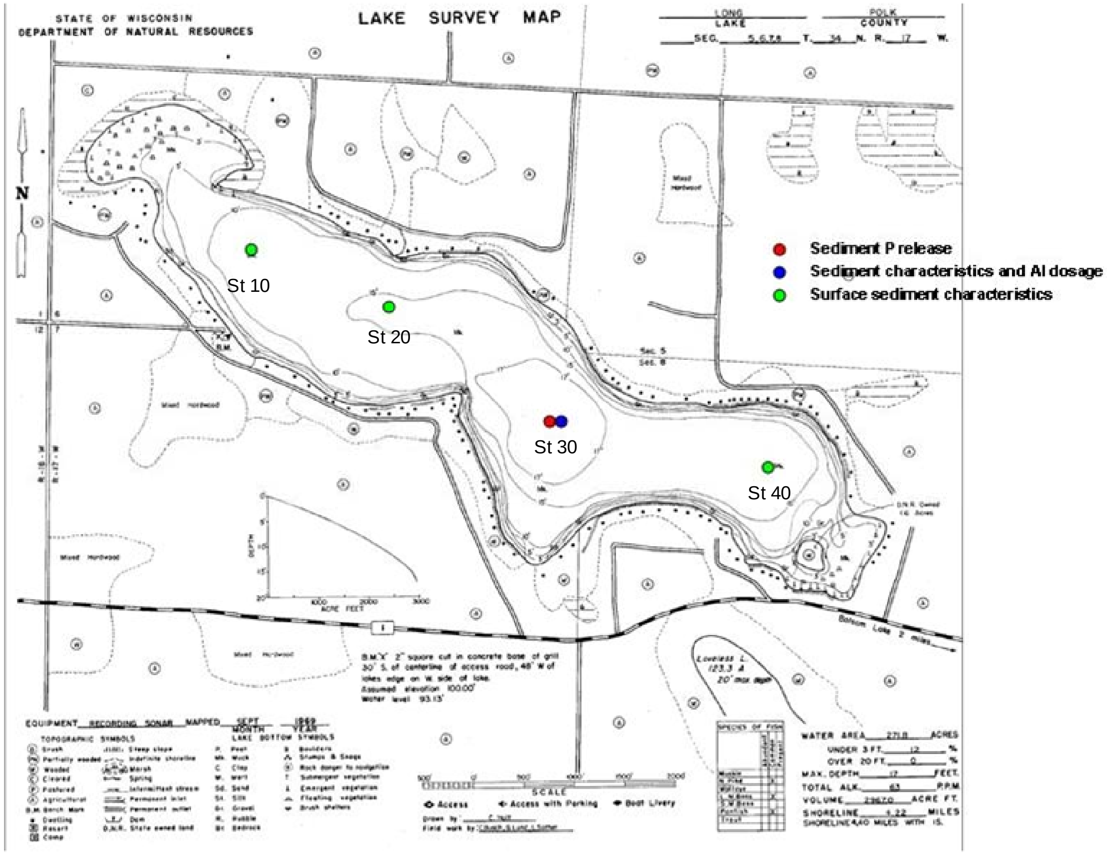

Figure 1. Station locations in Long Lake.

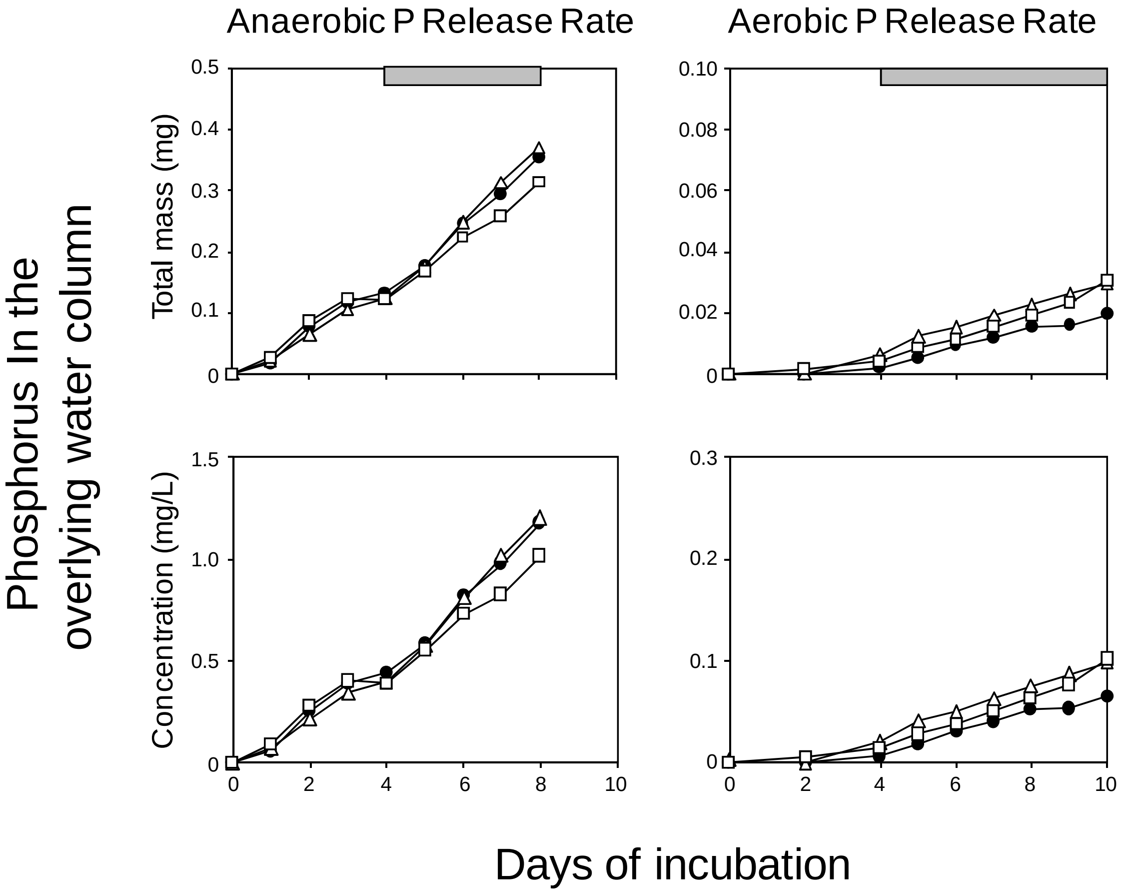

Figure 2. Changes in soluble reactive phosphorus mass (upper panels) and concentration (lower panels) in the overlying water column under anaerobic and aerobic conditions versus time for sediment cores collected at station 30 in Long Lake.

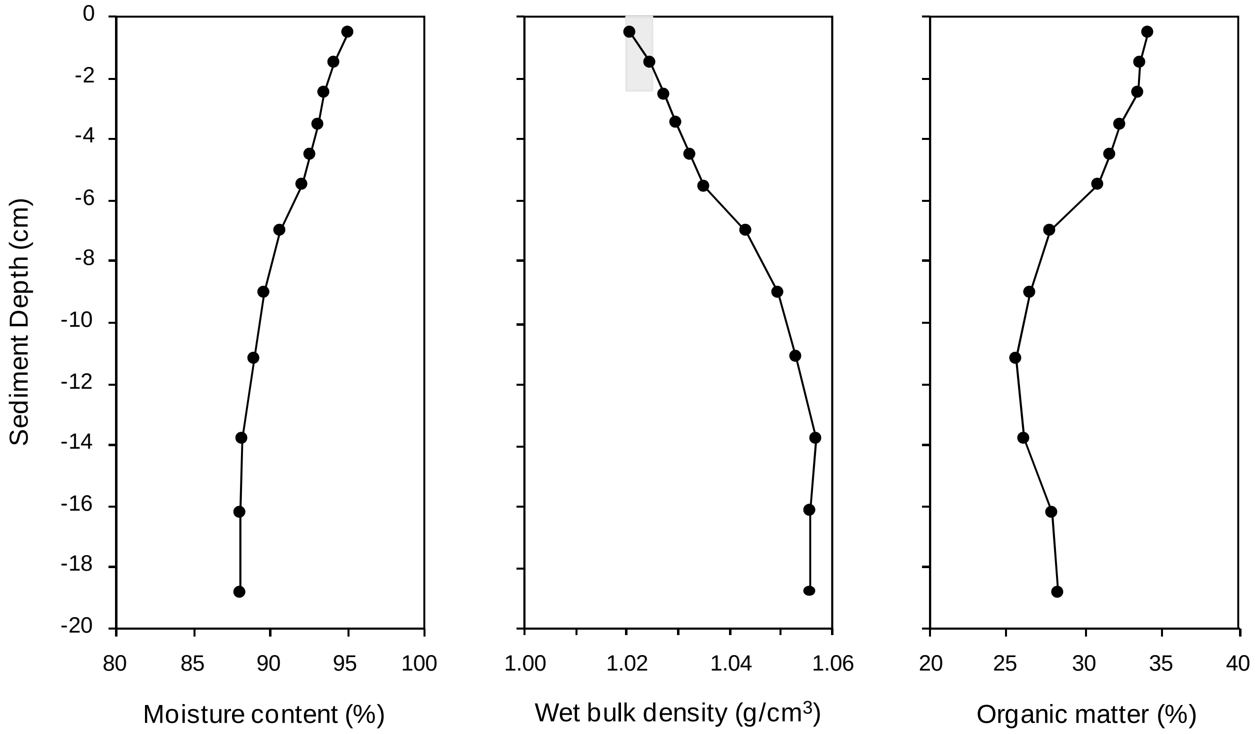

Figure 3. Variations in sediment moisture content (left panel), wet bulk density (middle panel), and organic matter content (right panel) as a function of depth below the sediment surface for a sediment core collected at station 30 in Long Lake. The shade box in panel 2 denotes the estimated wet bulk density range of the $Al(OH)_3$ floc layer (from research on Half Moon Lake, WI; WF James, unpublished data).

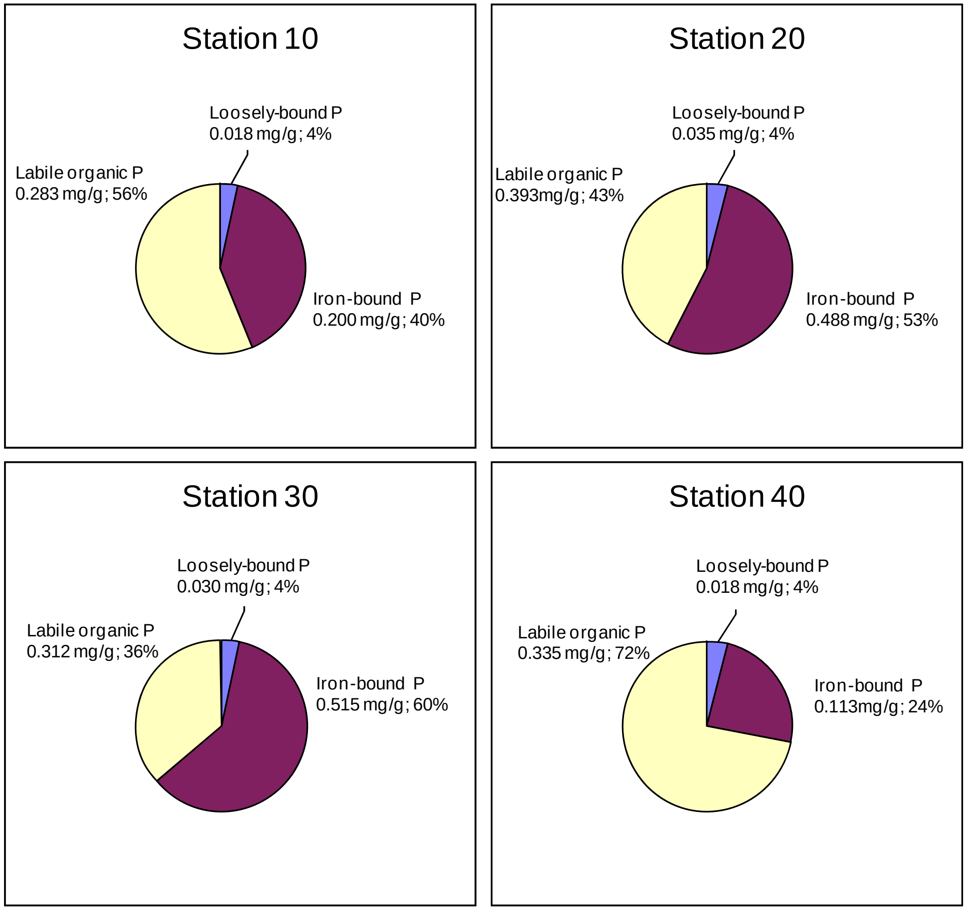

Figure 4. Variations in the composition of biologically-labile phosphorus (P; i.e., subject to recycling with the overlying water column; sum of the loosely-bound, iron-bound, and labile organic P) in the upper 5-cm sediment layer for cores collected in Long Lake.

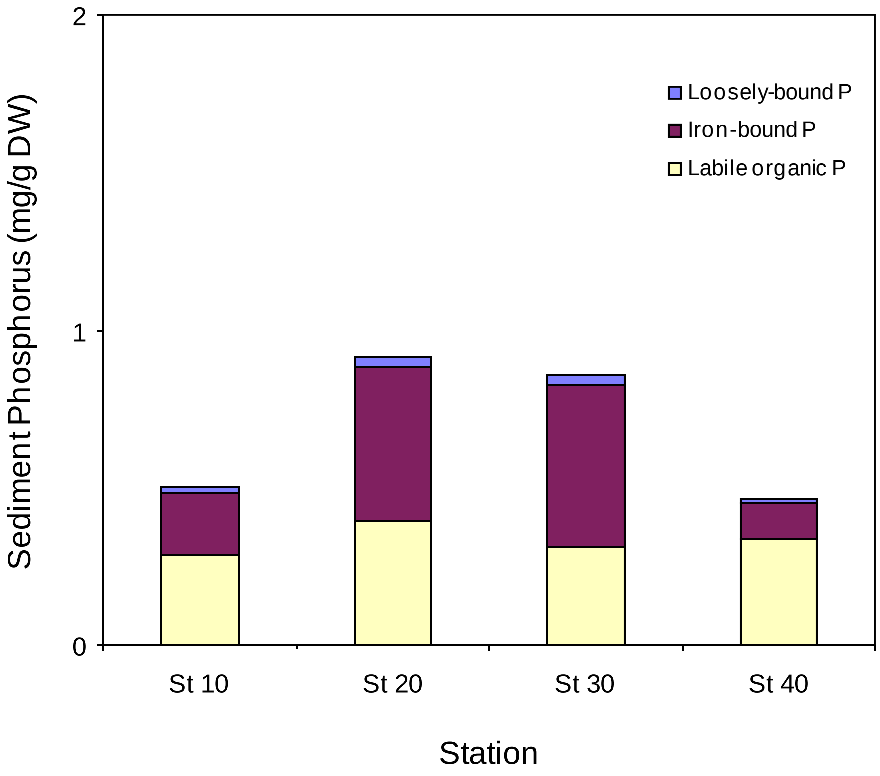

Figure 5. Variations in the concentration of biologically-labile phosphorus (P; i.e., subject to recycling with the overlying water column; sum of the loosely-bound, iron-bound, and labile organic P) in the upper 5-cm sediment layer for cores collected in Long Lake.

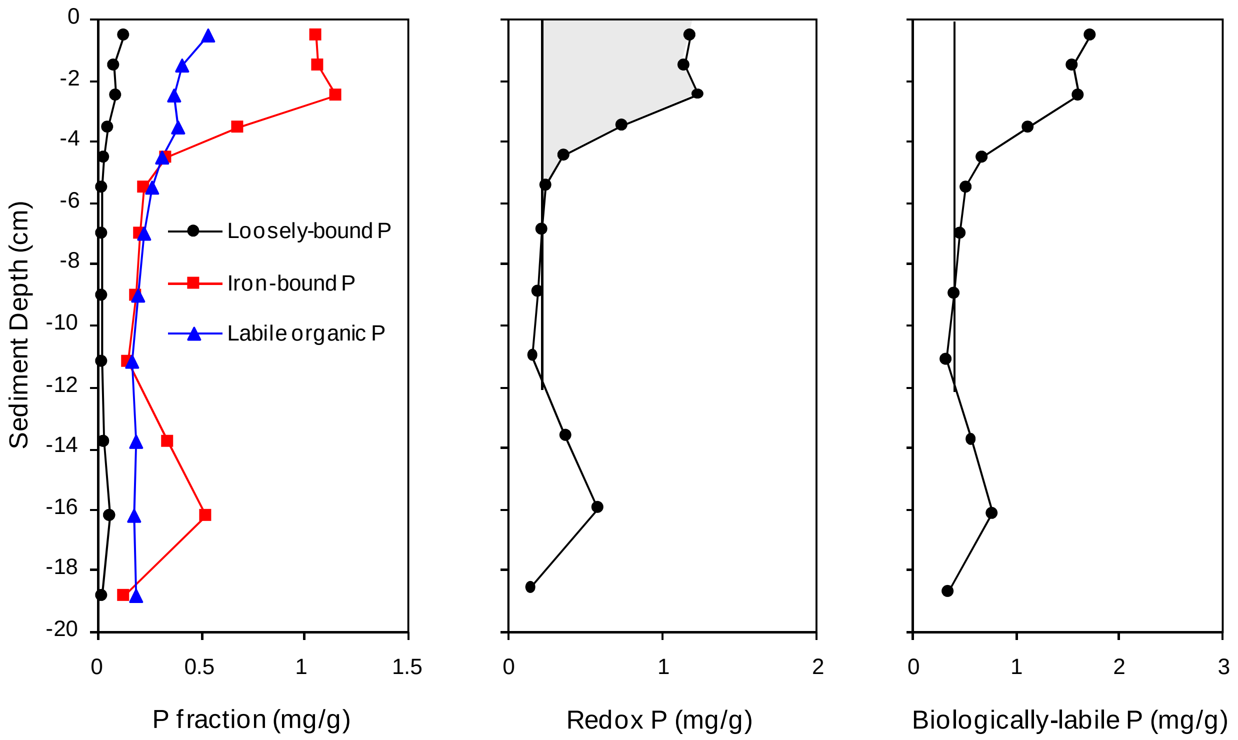

Figure 6. Variations in sediment loosely-bound phosphorus (P), iron-bound P and, labile organic P (left panel), redox-sensitive P (i.e., the sum of the loosely-bound and iron-bound P fractions; middle panel), and biologically-labile P (i.e., the sum of redox-sensitive P and labile organic P) as a function of depth below the sediment surface for sediment cores collected at station 30 in Long Lake. The gray shaded area delineates the upper 6-cm sediment layer of redox-sensitive P in excess of background concentrations. Background concentration was calculated as the average between the 8- and 20-cm sediment depth.