

Phosphorus Budget and Management Strategies for Cedar Lake, Wisconsin



Cyanobacterial bloom wind-blown against the north shoreline of Cedar Lake

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PREFACE

This research was conducted in response to a request from the Cedar Lake Protection and Rehabilitation District (CLPRD) and the State of Wisconsin Department of Natural Resources (WI-DNR) to critically evaluate phosphorus (P) dynamics, the P budget, and P loading reduction scenarios for improving water quality conditions in Cedar Lake. More specific objectives were to use annual steady-state empirical modeling approaches to establish feasible water quality targets based on reducing watershed runoff and tributary P loading from the Horse Creek watershed and internal P loading from anoxic profundal sediments in the lake. Since prior research has established that internal P loading is a dominant P source that fuels high levels of potentially toxic harmful algal blooms in Cedar Lake, the final objective of the report was to estimate dosage, treatment area, and cost scenarios for an aluminum sulfate treatment to control redox-sensitive P and anoxic P release from sediments in the lake. I gratefully acknowledge and thank the CLPRD, the WI-DNR, and the U.S. Army Engineer District (USAED), St. Paul, for providing funding for this research. The study coordinators were Mr. Patrick Sorge, Lake Management Specialist, WI-DNR, and Mr. Dennis Early, President, and Mr. Stuart Nelson, Treasurer, CLPRD. The Section 22 coordinator for the USAED, St. Paul, was Mr. Terry Zien. Mark Baumgartner, Nina Borchoweic, Aaron Carlson, Douglas Dahl, Jeff Jackson, Travis Jones, and Patrick Wears, ERDC Eau Galle Aquatic Ecology Laboratory and the University of Wisconsin - Stout, and Neil Trombly, WI-DNR, are gratefully acknowledged for participating in field sampling, flow gauging, sample processing and analysis, and execution of studies on phosphorus release from sediments. I also thank Jeremy Williamson, Polk County, WI, Land and Water Resources Department, for GIS information on the Horse Creek watershed.

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This report is dedicated to Stuart Nelson, who contributed tremendously to all aspects of the Cedar Lake research and rehabilitation project.

EXECUTIVE SUMMARY

Comprehensive limnological investigations were conducted on Cedar Lake and the Horse Creek watershed between 2009 and 2012 in order to examine seasonal lake nutrient (primarily phosphorus, P) and algal dynamics, develop detailed hydrological and P loading budgets, and identify important sources of P to the lake for management (James 2011a, 2012). The destratification system, operated during most summers since 1991 to control internal P loading, was suspended in 2009-2010 and turned on in 2011 in order to compare and evaluate limnological conditions and dynamics under stratified and destratified conditions. Annual steady-state empirical models were used to evaluate predicted lake trophic state response (i.e., total P, chlorophyll, and Secchi transparency) to external and internal P loading reduction scenarios and set water quality management goals. Additional research, conducted in 2012 by the Land and Water Resources Department of Polk County, examined sources of soil P, the P index for various agricultural fields, and runoff potential to develop future best management practices (BMPs) for controlling P loading in the watershed. Finally, aluminum sulfate dosage requirements and costs to control P release from anoxic profundal sediments were investigated in 2012.

Annual precipitation was slightly below but near the ten year mean over the three year period of research. The hydrological budget was dominated by Horse Creek tributary inflows (versus direct precipitation on the lake). The areal water load was moderate and varied between 3.8 and 4.9 m/y. Annual runoff (including snowmelt) represented 25 to 34% of the direct precipitation over the landscape. Unit area runoff was modest at ~ 0.20 to 0.25 m/y (~2000 to 2500 m³/ha y) over the entire watershed and much lower at ~ 0.13 to 0.15 m/y (~1300 to 1500 m³/ha y) for the portion of the watershed draining areas downstream of County K (i.e., the flow monitoring station located in the upper watershed below Horse Lake). Theoretical hydraulic residence times for Cedar Lake exceeded one year during the three monitored years (i.e., 1.5 y in 2009, 1.4 y in 2010, and 1.2 y in 2011).

The annual flow-weighted total P (TP) and soluble reactive P (SRP) concentration for Horse Creek at 10^{th} Avenue (i.e., the flow monitoring station located immediately upstream of Cedar Lake) over the three year period was moderate at 0.089 and 0.031 mg/L, respectively, and SRP accounted for ~ 35% of the tributary TP load to the lake. The annual flow-weighted SRP concentration was also highest at 10^{th} Avenue versus a much lower annual flow-weighted SRP concentration of 0.010 mg/L at County K, suggesting possible transformation of TP to SRP or net loading of SRP between the two sites.

The mean annual TP export coefficient for the Horse Creek catchment area above the County K gauging station (i.e., 2009-2011) was moderate to low at 0.34 kg/ha y (0.38 lbs/ac y). It was also higher at County K compared to the entire catchment area above 10th Avenue (i.e., 0.27 kg/ha y or 0.30 lbs/ac y). This pattern was related, in part, to carp and wind-induced resuspension of particulate P in shallow Horse Lake and discharge to Horse Creek. When corrected for annual estimated discharge of this internal TP load from Horse Lake (154 kg P/y; J. Williamson, personal communication), the annual TP export coefficient at County K and 10th Avenue was reduced to only 0.23 kg/ha y (0.25 lbs/ac y) and 0.22 kg/ha y (0.24 lbs/ac y), respectively. These estimates well fell below the "most likely" range for watersheds with greater than 50% agricultural land use practices in Wisconsin (0.56 kg/ha y; Panuska and Lillie 1995) and reflected implementation of BMPs in the watershed in the last two decades.

Summer discharge of TP from Cedar Lake (i.e., from the Cedar Lake outflow structure) was usually lower than the inflow concentration (i.e., Horse Creek at 10th Avenue) during periods of elevated flow, indicating P retention of watershed loads. However, there were notable periods when TP was higher in the discharge than the inflow, particularly during the mid-August to September lake turnover period, indicating net P export due to internal P loads that had accumulated in the bottom waters of the lake during periods of anoxia at the sediment-water interface. P export was greatest during the latter part of summer, 2011, in conjunction with operation of the destratification system. Overall, artificial

destratification exacerbated internal P loading due to poor Fe binding efficiency and a low Fe:P ratio as discussed below.

The lake stratified and developed bottom water anoxia that usually extended to the 6-m depth during the summers of 2009-2010 (i.e., no operation of the destratification system). Continual destratification and mixing in 2011 resulted in rapid warming of the entire water column throughout the summer. Hypolimnetic anoxia developed in the bottom waters between May and August of that year, although anoxic conditions extended from the lake bottom to only about the 7-m depth in 2011 versus the other summers. Overall, the anoxic factor (i.e., the number of days that an area equivalent to the lake surface area is anoxic) was similar and relatively high during the summers of 2009 and 2010 at 48 and 47 days, respectively, and lower in 2011 at 36 days in conjunction with operation of the destratification system.

Cedar Lake was very susceptible to wind-induced water column mixing during the summer and fall due to its shallow morphometry (maximum depth = ~ 10 m), long fetch (3,500 m along the N-S wind rose), and low Osgood Index (2.71). These features and the uncoupling of iron (Fe) from binding control over P at the sediment-water interface played an important role in driving vertical transport of internally-derived SRP to the epilimnion for algal assimilation and development of nuisance cyanobacterial blooms. TP (primarily as SRP) concentrations increased substantially in the hypolimnion during periods of anoxia (i.e., > 1.0 mg/L) as a result of internal P loading from sediment. However, diffusion and accumulation of dissolved Fe (DFe) from anoxic sediment into the hypolimnion was low relative to SRP, resulting in TFe:TP (mass:mass) and DFe:SRP ratios that were relatively low at 1.89 and 0.62, respectively. Ratios less than \sim 3.6:1 indicate poor binding efficiency for P during chemical oxidation to Fe^{+3} . These patterns suggested the possibility that bacterially-mediated SO_4^{2-} , or chelation with organic acids, were indirectly removing DFe from diffusion into the water column via formation of FeS_x and organic carbon-Fe complexes. During intermittent mixing periods in the summer, and particularly during the fall turnover period, significant SRP was entrained into the surface waters for algal assimilation rather than becoming bound and precipitated

to oxidized Fe³⁺ oxyhydroxides and removed via deposition. High internal P loading from sediment coupled with substantial transport to the surface water during mixing resulted in blooms in excess of 50 to 100 μ g/L chlorophyll that were often sustained from late August through October. These algal blooms were also largely dominated by potentially toxin-producing cyanobacteria (*Aphanocapsa incerta* or *Microcystis smithii*). Chlorophyll concentrations greater than 50 μ g/L occurred on average ~ 17% of the time in the summer (June-October), exceeding the World Health Organization threshold of risk associated with potential exposure to cyanotoxins (WiscCALM 2013).

Internal P loading from anoxic profundal sediment dominated both the annual and summer P budget of Cedar Lake, representing ~64% and 85% of the measured inputs, respectively. Summer net internal P loading, determined via mass balance, was very high at ~ 6.1 to 15.7 mg/m² d and indicative of hypereutrophic conditions. Rates were greatest in 2011 in conjunction with operation of the destratification system. Despite continual destratification and water column mixing throughout the summer of 2011, hypolimnetic anoxia developed and extended over the entire profundal area (i.e., depths greater than 25 ft) during operation. Continual water column mixing most likely increased horizontal water velocity above the sediment-water interface, promoting the development of a diffusive boundary layer that exacerbated dissolved oxygen demand and enhanced anoxic diffusive P flux from the sediment. Since the Fe:P ratio was low, internal P loads, primarily as SRP, were continually entrained into the surface waters by the destratification system, resulting in concentrations in excess of 0.10 mg/L throughout the summer. In all likelihood, this pattern of enhanced internal P loading coupled with high water column SRP concentrations occurred during the 18+ years of operation of the destratification system.

Annual steady-state empirical modeling indicated that immediate control of internal P loading by, for instance, an aluminum sulfate treatment would result in a predicted substantial improvement of the annual mean TP concentration to ~ 0.033 mg/L (from 0.062 mg/L under current P loading conditions). Mean summer (June-October) chlorophyll would decrease from ~ 33 μ g/L to ~ 13 μ g/L and the summer bloom

frequency (percent of days) of nuisance chlorophyll concentrations (i.e., > 20 µg/L; WisCALM threshold) would decline from 69% to 20%. These predicted future conditions would fall well below the WisCALM (2013) impairment thresholds. Additionally, bloom frequency exceeding 50 µg/L (World Health Organization threshold of risk associated with potential exposure to cyanotoxins, WiscCALM 2013) was predicted to decline from ~17% to only 1% following internal P loading control.

In contrast, future implementation of additional watershed BMPs to reduce watershed P loading by ~ 30% (i.e., without internal P loading control) would result in only minor declines in annual mean lake TP (to 0.054 mg/L), mean summer chlorophyll (to 27.5 μ g/L), and summer bloom frequency of nuisance (i.e., > 20 μ g/L) and potentially-toxic (i.e., > 50 μ g/L) chlorophyll concentrations (to ~61% and 12%, respectively). These future predicted conditions substantially exceeded WisCALM (2013) threshold limits. Thus, control of internal P loading at a minimum is required to meet water quality goals and reduce the human health risk to potentially toxin-forming cyanobacterial blooms in Cedar Lake.

Empirical modeling suggested that an additional 30% reduction in watershed TP loading to the lake plus internal P loading control via an aluminum sulfate treatment would result in further water quality improvement. The predicted annual mean TP concentration under this scenario would decline to ~ 0.025 mg/L. Mean summer chlorophyll concentrations would decrease to less than 10 μ g/L and the summer bloom frequency of nuisance chlorophyll concentrations greater than 20 μ g/L would decline to less than 8%. The frequency of blooms exceeding 50 μ g/L was essentially zero under this scenario. Predicted mean Secchi transparency would increase from 2 m to 5.1 m.

Management of internal P loading should be the top priority to meet water quality goals and reduce human health risks due to potentially toxin-forming cyanobacterial blooms in Cedar Lake. Internal P loading should be controlled via application of aluminum sulfate (alum or Al) to irreversibly bind P in the sediment versus artificial destratification. Major considerations and decisions regarding alum treatment of Cedar Lake included 1) the

dosage concentration needed to sequester the mobile P layer in the sediment, 2) treatment area, and 3) buffered versus unbuffered Al application. Most importantly, a higher Al dosage on the order of 100 to 130 g Al/m² needed to be considered based on actual dosage assays conducted on Cedar Lake sediment and recent lake alum application case studies published in the literature.

A stratified application approach is recommended. An Al dosage of 130 g/m^2 should be applied to lake sediments located at the 25-ft contour and deeper. An Al dosage of 100 g/m^2 should be targeted for lake sediments located between the 20- and 25-ft depth contours to treat areas located at depths that experience seasonal anoxia. Another consideration is the buffering capacity of lake water during alum application. The total alkalinity for Cedar Lake was moderate at ~100 mg CaCO₃/L, suggesting some buffering potential for moderating pH during alum application. However, the areal dosages suggested for the lake approach the maximum allowable dosage to safely maintain pH at or above 6.0 to prevent Al^{3+} toxicity. Thus, further alkalinity analysis will be needed near the time of actual treatment in order to refine application strategies. Two options are available that avoid the potential for exceeding the maximum allowable Al dose. One option is application of buffered alum (aluminum sulfate and sodium aluminate) to maintain pH near neutrality during reaction. This option is more expensive. Option two would entail multiple alum applications at a lower dosage to prevent the risk of low pH, but over a period of several years (i.e., 2-3 year intervals). This latter option has the added advantage of maximizing P binding efficiency to Al flocs. Overall estimated costs to treat sediment under option 2 is 2.16 million dollars.

Implementation of BMPs in recent decades and reduction in P runoff from the landscape has already been extensive in the Horse Creek watershed (Wojchik 2013). Overall, the P index values are very low for agricultural fields (< 2.0) and soil test P concentrations of at least ~ 46% of the fields examined fell within optimum ranges (15 to 35 ppm). Further opportunities for management and P loading reduction are presented in Wojchik (2013). In particular, a newly implemented farmer-led watershed council pilot project is being

conducted in the Horse Creek watershed that will involve the agricultural community in the future management of P runoff.

OBJECTIVES

The objectives of this report are severalfold:

- 1. summarize limnological research conducted on Cedar Lake between 2008 and 2012,
- 2. critically evaluate the phosphorus (P) budget and important P sources and sinks to the lake,
- 3. establish water quality goals for P, chlorophyll, and Secchi disk transparency using empirical modeling techniques,
- 4. determine the extent of external and internal P loading reduction required to meet those goals and immediately reduce the magnitude and frequency of cyanobacterial blooms and associated potential human health concerns,
- 5. evaluate Horse Creek watershed soil P and land use practices to identify critical source areas for BMP implementation to reduce soil P runoff, and
- 6. develop aluminum sulfate dosage and cost scenarios to control internal P loading.

This summary of research findings will be used to develop a comprehensive management plan for Cedar Lake (Clemens 2013).

STUDY SITE

The Horse Creek watershed, located in southern Polk County, Wisconsin, drains a 140 km² basin upstream of the Cedar Lake outflow (Fig. 1). Watershed topography comprised of gently rolling terrain formed by glaciation. Soils were derived from glacial and alluvial deposits and are primarily comprised of the Rosholt-Cramwell-Menahga and Alban-Carnpia-Cornstock associations. The upper reaches of the watershed contain numerous

small lakes that form headwater tributaries draining into Horse Creek. Because glaciation scoured the terrain and created local depressions, a significant portion of the watershed (57 km²; 42%) does not directly drain into Cedar Lake, but rather stores runoff internally (Table 1). Within the Cedar Lake direct drainage basin, forest, grasslands, open water, and wetlands comprise 58% of the land cover (Fig. 2). Row and forage crop production and pasture account for 34% of the land cover (WI-DNR 2001).

Cedar Lake is large (surface area = 4,522,767 m²; volume = 25,235,867 m³; mean depth = 5.78 m; maximum depth = 10.4 m; Table 1) and elongated along the north-south axis with a maximum fetch of ~ 3,500 m (Fig. 3). Lake morphometry is sinusoidal with a large and uniformly deep basin below the 6-m depth (~ 20 ft), a region of steep depth gradients between 2 and 6 m, and a relatively small shallow shoreline region (Fig. 4). The lake is susceptible to turbulent vertical entrainment of nutrients from the bottom waters to the euphotic zone due to its large surface area in relation to mean depth. The Osgood Index for the lake is 2.7 (i.e., mean depth divided by the square root of the surface area; $Z_{mean} \div \sqrt{A}$; A = surface area, km²). Lakes with values < 6 have been associated with higher surface total P than can be predicted by external P loading. Approximately 60% of the sediment area is greater than 25 ft (7.6 m).

Historically, the lake is classified as hypereutrophic (TSI_{TP}=73; TSI_{chla}=68; TSI_{SD}=54) and mean summer surface concentrations (averaged over a 20 year period between 1988 and the present) of total phosphorus (TP), chlorophyll, and Secchi disk transparency are 0.121 mg/L, 45.3 mg/m³, and 1.52 m, respectively (Fig. 5). Important components of the P budget include internal P loading from anoxic profundal sediment, carp activity and excretion of soluble P, and watershed loading (Sorge and Engel 1989, 2001). The submersed aquatic plant community, dominated by coontail and *Najas*, occupies ~ 40% of the area at depths less than ~ 2 m (Konkel 2002). Curly-leaf pondweed has also increased in abundance in recent years. Hydrologically, the lake has a relatively high theoretical residence time (> 1 y) and low flushing rate. Nominal pool elevation is maintained during late April through October and lowered by ~ 1 ft or more throughout the winter.

A destratification system was deployed in Cedar Lake in the early 1990s to control internal P loading by mixing and reaeration of the bottom waters to increase the redox potential. The system consisted of a series of four perforated pipes branching off the main supply line that provided a stream of fine bubbles in the approximate center of the lake (Fig. 3). The system was operated nearly continuously between 1991-92 and 1994-2008 (exact periods of operation between 1994 and 2008 are not precisely known; Fig. 5). For research purposes, it was turned off in 2009-2010 and operated in 2011 to examine P dynamics under ambient versus artificially destratified conditions. James (2012) found that artificial destratification actually exacerbated internal P loading because the Fe:P ratio was low in the hypolimnion and iron oxyhydroxide precipitation was not controlling P. Destratification promoted such high total and soluble reactive P in the surface waters that algal productivity was probably not limited by P availability during operation of the system.

RESEARCH PROGRAM

Tributary Hydrology and Constituent Loading

Gauging stations were maintained on Horse Creek at County K and at 10th Ave from late 2008 through 2011 for flow determination (Fig. 1). Horse Creek flows through two elliptical culverts each measuring 3.33 ft high by 4.75 ft wide at County K and one large 6.75 ft high by 11 ft wide elliptical culvert at 10th Avenue near County M. A data logger and area-velocity meter (Isco Model 4150; Hach Co., Loveland, CO) collected velocity (m/s) and stage height (m) at 15-min intervals in each culvert. Volumetric flow (m³/s) was estimated from this information using Isco Flowlink[®] 5 software. Daily pool elevation changes were recorded from a staff gauge deployed at the outflow structure of Cedar Lake by local volunteers. Daily discharge (Out_{lake}, m³/s or Hm³/y) at Cedar Creek (Fig. 3) was calculated using a hydrologic budget equation as,

where, Δ lake volume = daily change in lake volume, In_{trib} = daily tributary inflow from Horse Creek at 10th Ave, In_{precip} = daily precipitation over the lake surface (measured at Amery, WI, by the State of Wisconsin), and Out_{evap} = daily evaporation from the lake surface. Daily evaporation was estimated using a modified Blaney-Criddle equation (University of Minnesota pers. commu.). Chloride mass balance indicated that net groundwater inputs were minor so groundwater was not included in the budget.

1),

Water grab samples were collected at weekly to biweekly intervals during the ice-free season and monthly or less during the winter at the tributary and outflow stations for determination of total nitrogen (TN), nitrate nitrite-N (NO_x), ammonium-N (NH_x), total phosphorus (TP), soluble reactive phosphorus (SRP), total and dissolved iron (TFe and DFe), and chloride (in 2010-11 only). Annual and seasonal loadings (kg/y or kg/d) were estimated using the computer program FLUX (Walker 1996).

Annual TP export coefficients (kg/ha y or lb/ac y) were estimated for each gauging station by dividing the annual TP load by the upstream catchment area. The Big Lake subwatershed was not included in the calculation because discharge out of Big Lake did not occur during the study period (Fig. 1). Additionally, internal TP loading and discharge from Horse Lake represented a portion of the annual TP load at County K (~ 154 kg TP/y or 16% of the TP load at County K; J. Williamson, personal communication, Polk County Land and Water Resources Department, Balsam Lake, Polk County, WI) and was considered in the estimation of annual TP export at County K. Since the lake is very shallow and exposed, wind- and carp-induced sediment resuspension plays an important role in internal TP load at both County K and 10th Avenue to correct annual TP export coefficients for discharge of resuspended P loads.

Limnological Monitoring

Water samples and in situ profiles were collected weekly at a centrally-located station (Fig. 3) at 1-m intervals between the lake surface and within 0.2 m above the bottom (maximum average depth = 7.8 m). In situ vertical profiles of temperature, dissolved oxygen, specific conductance, and pH were measured using a Hydrolab Quanta[®] (Hach Co., Loveland, CO) that was precalibrated against known buffer solutions and Winkler dissolved oxygen analyses (APHA 2005). Water transparency was estimated using a 20-cm alternating black and white Secchi disk. Water samples were collected using a peristaltic pump and tubing and stored in a cooler on block ice in the field. For dissolved constituent analysis, samples were pumped directly into a 60-cc syringe without exposure to air and immediately filtered through a 0.45 μ m filter. Samples were analyzed for TN, NO_x, NH_x, TP, SRP, TFe, DFe, alkalinity, and viable chlorophyll *a*. An additional integrated (0-3 m) water sample was collected monthly for determination of the algal assemblage.

A continuously-recording thermistor (i.e., temperature probes; Hobo Water Temp Pro v2 model U22-001; Onset[®], Pocasset, MA) array was deployed at station 2 for determination of seasonal and vertical variations in stratification and mixing in the pelagic region of the lake. Probes were positioned at 1-m intervals between the lake surface and bottom to record temperature at 1-h intervals between May and October. From these data, Schmidt stability (S; g-cm/cm²) was calculated as:

$$S = 1/A \int_{0}^{z_{m}} (z - z_{g})(\rho_{z} - \rho_{g}) dz$$
 2),

where A = surface area (m²), z_m = maximum depth (m), z = depth at stratum z, z_g = depth of the center of mass or ρ_g , and ρ_z = the density of water (kg/m³) at depth z (Idso 1973). ρ_g was calculated as:

$$\rho_{\rm g} = 1/V \int_0^{z_{\rm m}} V_z \rho_z dz$$
(3),

where V is lake volume (m^3) and V_z is the volume at depth z. Schmidt stability represented the amount of work (in the form of wind power, motor boat activity, etc) required to completely mix a water body (without loss of gain in heat content) that is stratified due to vertical differences in water density. Higher stability values were indicative of strong stratification and greater work required to disrupt stratification. Conversely, lower stability values were indicative of weak stratification and less work required to disrupt stratification. The location and extent of the metalimnion was quantified using the Relative Thermal Resistance to Mixing (RTRM) statistic (Vallentyne 1957).

Sediment Phosphorus Diffusive Fluxes and Textural-Chemical Characteristics

Spatial variations in anoxic diffusive P flux and sediment textural and chemical characteristics were evaluated in 2010 (James 2011a). Twenty-six stations were established along a grid in the deep region of the lake (> 20 ft deep) for sediment sampling purposes (Fig. 3). Results from the 2009 study (James 2011a) indicated that fine-grained sediments were located primarily below the 20-ft depth contour and that bottom material was coarse-grained sand and pebble at shallower depths. Thus, sediment sampling was focused on collection of fine-grained material for analysis in 2010. In addition, 2009 results indicated that P release from sediment was negligible under oxic conditions; so sediment cores were collected for determination of rates of P release under anoxic conditions only and for sediment textural and chemical characteristics.

Intact sediment cores were collected using a gravity coring device (Aquatic Research Instruments, Hope, ID) in August. The 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 the 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 (anoxic) through an air stone placed just above the sediment surface in each system.

Water samples for SRP were collected from the center of each system using an acidwashed syringe and filtered through a 0.45 μ m membrane syringe filter. 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. Rates of P release from the sediment (mg/m² d) were calculated as the linear change in mass in the overlying water divided by time (days) and the area (m²) of the incubation core liner. Regression analysis was used to estimate rates over the linear portion of the data.

After incubation for anoxic diffusive P flux, sediments were homogenized for analysis of moisture content (%), sediment density (g/mL), loss-on-ignition (i.e., organic matter content, %), loosely-bound P, iron-bound P, aluminum-bound P, calcium-bound P, labile and refractory organic P, TP, TFe, TMn, and TCa (all expressed at mg/g). A known volume of sediment was dried at 105 °C for determination of moisture content and sediment density and ashed at 500 °C for determination of loss-on-ignition organic matter content (Håkanson and Jansson 2002). Additional sediment was dried to a constant weight, ground, and digested for analysis of TP, TFe, TMn, and TCa using standard methods (Plumb 1980; APHA 2005). Phosphorus fractionation (Table 2) was conducted according to Hieltjes and Lijklema (1980), Psenner and Puckso (1988), and Nürnberg (1988) for the determination of ammonium-chloride-extractable P (1 M NH₄Cl; loosely-bound P or loose-P), bicarbonate-dithionite-extractable P (0.11 M BD; iron-bound P or Fe-P), sodium hydroxide-extractable P (0.5 N HCl; calcium-bound P or Ca-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 hydroxideextractable P. Refractory organic P was estimated as the difference between total phosphorus and the sum of the other fractions.

The loose-P and Fe-P fractions (Table 2) are readily mobilized at the sediment-water interface as a result of Eh (i.e., oxidizing and reducing conditions) and pH reactions that result in desorption of P from sediment and diffusion into the overlying water column (Mortimer 1971, Boström 1984, Nürnberg 1988). The sum of the loose-P and Fe-P fraction represents redox-sensitive P (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 SRP under anaerobic conditions (Gächter et al. 1988; Gächter and Meyer 1993; Hupfer et al. 1995). The sum of redox-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, Al-P, Ca-P, and refractory organic P are more chemically inert and subject to burial rather than recycling (Table 2).

Vertical variations in sediment textural and chemical constituents were examined in 2012 for characterization of the thickness and mass of the mobile P pool. This information was used for estimating the concentration of aluminum sulfate required to control internal P loading (see below). A sediment core collected in the deep basin of the lake (near station 2) was sectioned at 1-cm intervals over the upper 10 cm and at 2-cm interval below that depth. Sediment sections were analyzed for moisture content, sediment density, organic matter content, loose-P, Fe-P, Al-P, Ca-P, labile and refractory organic P, TP, TFe, TMn, and TCa using procedures described above.

Lake Phosphorus Budget

Summer P budget approaches were used to assess the importance of various external and internal P sources to Cedar Lake. Net internal P loading ($P_{net internal load}$, kg) was defined as the flux of P from sediment stored in the lake in excess of P sedimentation. $P_{net internal load}$ was estimated by difference from the equation,

$$\Delta P_{\text{lake storage}} = (P_{\text{external load}} - P_{\text{outflow}}) \pm P_{\text{net internal load}} \pm P_{\text{groundwater}}$$

$$4),$$

where $\Delta P_{\text{lake storage}}$ = the change in lake P mass over a defined summer period (kg), $P_{\text{external load}}$ = the P mass input to the lake from Horse Creek (kg), P_{outflow} = the P mass that was discharged from the lake (kg), $P_{\text{external load}} - P_{\text{outflow}}$ = net retention of external P loads ($P_{\text{net retention}}$, kg), and $P_{\text{groundwater}}$ = net P flux into or out of Cedar Lake via groundwater. $P_{\text{groundwater}}$ was assumed to be negligible based on analysis of the hydrologic budget and the chloride budget. $P_{\text{lake storage}}$ was calculated as,

$$P_{lakestorage} = \sum_{i=0}^{n} P_{concentration} \cdot Volume$$
 5),

where $P_{\text{concentration}} =$ the TP concentration (mg/L or g/m³) and Volume = the water volume (m³) at depth *i* (m). The product of these variables (kg) for each depth layer was summed over the entire water column (n = maximum depth, m) to estimate $P_{\text{lake storage}}$. $P_{\text{external load}}$ and P_{outflow} were estimated over the defined summer period using the computer program FLUX. Mass estimates were converted to rates (mg/m² d) by dividing mass by the area of the lake and defined time period (~ 100 days).

Empirical Modeling and Phosphorus Loading Reduction Scenarios

Empirical modeling was conducted using *Bathtub* (Walker 1996) and equations developed by Nürnberg (1998 and 2009). *Bathtub* is a windows-based software program

that provides a suite of equations for predicting lake annual averages of P, chlorophyll, and Secchi transparency. The Nürnberg (1998 and 2009) P empirical model specifically accounts for both external and internal P loadings to predict lake annual average P concentration according to the following equation,

$$P_{lake} = \frac{(L_{ExtPLoad} + L_{GrosstIntP Load})}{q_s} \times (1 - R_{Pr ed})$$
6).

Variables defined for equation 6 are: $L_{GrossIntPLoad}$ = the gross rate of internal P loading based on total flux of P from anoxic sediment, including internal P loads that may have settled back to the sediment via chemical adsorption and biological uptake and deposition (mg/m² y) and R_{Pred} = the predicted fraction of external P loads retained in the lake. Since it is difficult to estimate retention of external P loads in lakes with significant internal P loading, such as Cedar Lake, R_{Pred} was estimated according to the following empirically derived equation developed by Nürnberg (1984) as,

$$R_{Pred} = 15/(18 + q_s)$$
 7).

The summer anoxic factor (AF, d/summer), days that a sediment area normalized with respect to the lake surface area is anoxic, was quantified for 2009-2011 according to the following equation developed by Nürnberg (1995) as,

$$AF = \frac{\sum_{i=1}^{n} t_i a_i}{A_{LakeSurface}}$$

8),

where, t_i = the time interval of anoxic conditions (d), a_i = anoxic sediment area over the interval (m²), and $A_{LakeSurface}$ = the lake surface area (m²). Anoxia was defined as dissolved oxygen concentration less than or equal to 1.0 mg/L.

Aluminum Sulfate Dosage Determination

Profundal surface sediments, collected from several locations in the lake, were subjected to various concentrations of Al stock to determine the dosage required to inactivate the redox-P fraction (Rydin and Welch 1999). Alum (as aluminum sulfate; $Al_2(SO_4)_3 \cdot 18$ H₂O) was mixed with 0.1 M sodium bicarbonate (NaHCO₃) to a concentration of 0.7 g Al/L (at pH 7.0). 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 mass of fresh sediment to obtain Al concentrations ranging from 0 (i.e., control) to 300 mg Al/g of dry sediment or higher. The assay tubes were shaken for a minimum of 2 hours at 20 °C in a darkened environmental chamber and centrifuged at 500 g to concentrate the sediment. The treated sediments were then extracted for loosely-bound and iron-bound P using procedures described above.

The Al concentration (g/m^2) was estimated as,

Al
$$(g/m^2) = \text{redox-P}(g/m^2) \cdot \text{Al:P}_{90\%}$$
 9),

where, Al:P_{90%} was the binding ratio required to adsorb 90% of the redox-P in the sediment (James 2011b). Redox-P was calculated as,

redox-P
$$(g/m^2)$$
 = redox-P $(mg/g) \cdot \rho (g/cm^3) \cdot \theta \cdot h (m) \cdot 1000000 (cm^3/m^3) \cdot 0.001 (g/mg)$ 10),

where, ρ is sediment bulk density (g/cm³), θ is the percent sediment solids (100 – percent moisture content; dimensionless), and *h* is sediment thickness (m).

A titration procedure was used to determine the maximum allowable dosage of Al to maintain pH above 6.0 (Cooke et al. 2005). The hydrolysis of Al leads to the production of hydrogen ions which lowers the pH of the water column. Al toxicity to the biota can occur if the pH falls below 6 and must be considered in dose determination. Lake water was titrated with 1.25 g Al/L to determine this maximum allowable dosage and compared with the dosage estimated from sediment assays. For situations where alkalinity is low or the required dosage exceeds maximum allowable dosage, buffered aluminum sulfate-sodium aluminate treatment will be needed to maintain pH above 6.0.

Lake bathymetry was compared with hypolimnetic anoxia patterns and the anoxic factor to determine the sediment area in the lake that was likely to be active in internal phosphorus loading and in need of alum treatment. The areal (g/m^2) and volumetric (mg/L) concentration of Al was converted to gallons of aluminum sulfate by considering the treatment area and volume of the lake. This information was then converted to a cost estimate to treat the lake based on generic applicator setup fees and the current cost of alum per gallon. A stratified (i.e., with respect to different sediment areas of the lake) treatment scenario was developed, based on the thickness of the sediment layer to be treated and the area of anoxic sediment in the hypolimnion.

Analytical Methods

Samples for TN and TP were predigested with potassium persulfate according to Ameel et al. (1993) before analysis. NH_x, NO_x, and SRP samples were filtered through a 0.45 µm filter prior to analysis (Millipore MF). N and P were determined colorimetrically on a Lachat QuikChem automated water chemistry system (Hach Company, Loveland, CO). Chlorophyll was determined via a fluorometric technique following extraction in a 1:1 solution of acetone and dimethyl sulfoxide (Welschmeyer 1994). Total metals were digested with a combination of nitric-hydrochloric acids and hydrogen peroxide according to EPA method 3050b. Total and soluble metals were analyzed using atomic absorption spectroscopy (APHA 2005; AAnalytist 100, PerkinElmer Corp, Waltham, MA USA).

RESULTS AND DISCUSSION

Hydrology and tributary loading

On an annual basis, precipitation in 2010 slightly exceeded the 10-year average of 31.09 inches recorded in Amery, WI (Fig. 6). It was slightly lower than the average in 2009 and 2011. In contrast, flow was greatest in 2011 as a result of contributions from snowmelt runoff (Table 3). Flows at County K, located below Horse Lake, represented ~ 51 to 57% of the flow at 10th Ave (Fig. 7). Unit area runoff (i.e., flow \div watershed area) was modest at ~ 0.20 to 0.25 m³/m² y (~ 2,000 to 2,500 m³/ha y) over the entire watershed and much lower at ~ 0.13 to 0.15 m³/m² y (~ 1,300 to 1,500 m³/ha y) for the portion of the watershed draining areas downstream of County K. The areal water load (i.e., flow + direct precipitation \div lake area) was moderate relative to the size of the lake (3.8 to 4.9 m/y; Table 3), resulting in rather long theoretical hydraulic residence times that exceeded one year during the three study years (i.e., 1.5 y in 2009, 1.4 y in 2010, and 1.2 y in 2011).

Over the summer period (May – Sep), water income approximately balanced losses with tributary inputs and discharge at the outflow structure dominating the hydrological budget (Table 4). Water income was greatest during the summer of 2011 in conjunction with more frequent precipitation (Fig. 7), resulting in a relatively low summer theoretical hydraulic residence time of ~ 0.5 y. In contrast, summer theoretical hydraulic residence times were much higher at 2.1 y in 2009 and 1.4 y in 2010.

Concentrations of TP for Horse Creek at 10th Avenue were generally highest during periods of snowmelt runoff and spring or fall storms and declined during late summer low flow periods (Fig. 8). SRP concentrations tended to be lower during the higher flow periods in April and May and increased in conjunction with mid- to late summer storms at the 10th Avenue site. Although TP patterns and concentrations were generally similar at County K, SRP was much lower at this site compared to SRP concentrations at 10th

Avenue. This pattern may be related to agricultural influences in the watershed below County K.

At County K, nitrogen species were composed primarily of TKN during the summer months (Fig. 9). Both NH_x and NO_x concentrations were elevated during the winter to early spring months and declined to low values by May of each year. At the 10th Avenue site, NO_x concentrations fluctuated with storm inflows. In particular, concentrations exhibited maxima in the late winter, early spring, and fall of each year and declined substantially during summer low flow periods. The occurrence of higher NO_x concentrations at the 10th Avenue site again suggested possible agricultural influences in the watershed downstream of County K. Ammonium concentrations rapidly declined to minima (near detection limits) by early to mid-April and remained low in concentration over the summer and fall of each year.

Annual constituent loading was greater in 2011 compared to other years in conjunction with higher flows driven in part by excessive snowmelt runoff and a high frequency of precipitation events in July (Table 5). The annual flow-weighted TP and SRP concentration for Horse Creek at 10th Avenue over the three year period was 0.089 and 0.031 mg/L, respectively (Table 6). SRP accounted for \sim 35% of the TP load to the lake (Fig. 10). The flow-weighted TP concentration was slightly higher at County K; however, the flow-weighted SRP concentration at this site was much lower compared to the concentration for Horse Creek at County K (Table 6). This pattern suggested possible transformation of TP to SRP or net loading of SRP and net loss of particulate P (i.e., via deposition onto the floodplain) between the two sites. Flow-weighted concentrations of NH_x and NO_x combined represented ~ 48%, while TKN accounted for ~ 52% of the flow-weighted TN concentration for Horse Creek at 10th Avenue (Fig. 10). NO_x was the dominant inorganic N species and the flow-weighted concentration was relatively high at ~ 0.7 mg/L. In contrast, flow-weighted concentrations of NO_x were much lower at the County K station and combined inorganic N species (i.e., NH_x and NO_x) represented only $\sim 20\%$ of the flow-weighted TN.

The mean annual TP export coefficient for the Horse Creek catchment area above the County K gauging station (i.e., 2009-2011) was moderate to low at 0.34 kg/ha y (Table 7). It was also higher at County K compared to the entire catchment area above 10th Avenue (i.e., 0.27 kg/ha y; Table 7). This pattern was related, in part, to carp and wind-induced resuspension of particulate P in shallow Horse Lake and discharge to Horse Creek. When corrected for annual estimated discharge of this internal TP load from Horse Lake (154 kg P/y; J. Williamson, personal communication), the annual TP export coefficient at County K and 10th Avenue was reduced to only 0.23 kg/ha y and 0.22 kg/ha y, respectively. These estimates fell well below the "most likely" range for watersheds with greater than 50% agricultural land use practices in Wisconsin (Panuska and Lillie 1995).

Summer discharge of TP from Cedar Lake (i.e., from the Cedar Lake outflow structure; see Fig. 3) was usually lower than the inflow concentration (i.e., Horse Creek at 10th Avenue) during periods of elevated flow, indicating P retention of watershed loads in the lake (Fig. 11). However, there were some notable periods when TP was higher in the discharge than the inflow, particularly during the mid-August to September turnover period, indicating some net P export that due to internal P loads accumulating in the bottom waters of the lake during periods of anoxia at the sediment-water interface. For instance, some net TP export from the lake was evident during the fall turnover period of late August through September, 2009, and early September, 2010. In particular, discharge TP was very high during August through October, 2011, as a result of operation of the destratification system, which greatly enhanced internal P loading (James 2012). Lake TP concentrations tended to be higher in Cedar Lake at station 2 than the discharge during that late summer and early fall period of 2011, suggesting that some of the internal P load, incorporated into algal biomass, settled back to the sediment instead discharging from the lake. Deposition of a portion of the internal P load was plausible because travel time and distance to the outflow (i.e., several km) was relatively long due to the high residence time of the lake.

Overall, mean *summer* (June-October) flow-weighted TP concentrations in the inflow were ~ 0.10 mg/L and SRP represented ~ 35% of the TP (mean = 0.036 mg/L; Table 8). Mean summer discharge TP concentrations were ~ 0.05 mg/L during 2009-10 and composed primarily of particulate P (probably as algal biomass) as indicated by very low mean SRP in the outflow that accounted for only ~ 10% of the TP. By comparison, SRP represented ~ 43% of the TP in the outflow during summer operation of the destratification system in 2011.

Limnological patterns in Cedar Lake

Stratification and dissolved oxygen dynamics. Temperature patterns indicated that Cedar Lake can exhibit strong stratification with relatively high thermal stability during warm summers (such as in 2010) and frequent polymixis during the passage of summer cold fronts (such as in 2009) due to shallow morphometry, a low Osgood Index, and a relatively long effective fetch along the north-south wind rose. For instance, stratification was frequently disrupted during the summer of 2009 (Fig. 12 and 13), due to the net loss of heat during the passage of cold fronts and strong winds that resulted in decreased Schmidt stability and either partial of complete water column mixing (Fig. 14). The summer of 2010 provided a marked contrast in that passages of disruptive climatological fronts over the region were less frequent, resulting in warmer, calmer days versus patterns observed in 2009. Stratification developed in mid-May of that year and the lake was strongly stratified throughout the summer (Fig. 13) with only one period of significant net heat loss and partial mixing that occurred in June (Fig. 14). During strongly stratified periods (as in 2010), the epilimnion was relatively thick at ~ 4.0 to 4.5 m. Due to the large surface area:volume ratio and sinusoidal shape of the basin, the mean depth (5.7 m) and center of gravity of density stratification (~5 m) was deep relative to the maximum depth of the lake, resulting in an expansive metalimnion with little to no defined hypolimnion. The bottom waters tended to heat significantly even during strongly stratified summers such as 2010 and were greater than 20 °C by August (Fig. 12 and 13).

This pattern probably played an important role in enhancing sediment oxygen demand and nutrient fluxes in the profundal sediments.

Stratification patterns were very different in 2011 in conjunction with operation of the destratification system, compared 2009 and 2010 when the system was turned off. Overall, the lake pelagic region was either destratified or very weakly stratified throughout the summer (Fig. 12 and 13). The lake was only briefly stratified in early and late July. In addition, destratification resulted in pronounced heating of the bottom water throughout the summer, as warmer surface water was continuously mixed throughout the water column (Fig. 13). Mean lake temperature increased to greater than 25 °C and overall heat content reached a peak by July, 2011 (Fig. 14). In contrast, Schmidt stability was very low and often approached zero during the summer, 2011, compared to other years.

Dissolved oxygen depletion and anoxic conditions developed rapidly in the bottom waters of Cedar Lake in conjunction with stratification (Fig. 12). During periods of stratification in 2009 and 2010, bottom water anoxia (i.e., dissolved oxygen < 1.0 mg/L) usually extended to at least the 6-m depth. Despite continual destratification and water column mixing throughout the summer period of 2011, hypolimnetic anoxia developed in the bottom waters between May and August (Fig. 12). Anoxic conditions extended from the lake bottom to only about the 7-m depth in 2011 versus the other summers when the vertical extent of anoxia was more expansive. Overall, the anoxic factor (i.e., the number of days that an area equivalent to the lake surface area is anoxic) was similar during the summers of 2009 and 2010 at 48 and 47 days, respectively, and lower in 2011 at 36 days in conjunction with operation of the destratification system. These values fell within ranges observed for some other eutrophic lakes in the United States (Mendota, Onondaga, Rawson, Shagawa, E. and W. Twin; Nürnberg 1995)

Pelagic chemistry and algal dynamics. Lake phosphorus and chlorophyll patterns were influenced by both tributary loading and diffusive P flux from sediment during periods of anoxia. TP concentrations exhibited a peak early July and late September,

2010, which coincided with elevated TP tributary loading. During periods of stratification and anoxia, TP increased in the bottom waters over the summer with marked gradients of increasing concentration toward the sediment surface as an apparent result of diffusive P flux from anoxic sediment (Fig. 15), a seasonal pattern that was a strong indicator of internal P loading (Nürnberg 2009). As discussed in more detail later, summer lake-wide TP buildup greatly exceeded tributary P loading by an order of magnitude, supporting a contention that P sources were probably derived from sediment previously stored in the lake (i.e., older deposited sediment from previous years or decades of P accumulation) versus newly deposited external P loads (e.g., Hupfer et al. 1995, Hoffman et al. 2013).

Seasonal TP concentration increases in the bottom waters were more pronounced in 2010 due to stable stratification and continuous anoxic conditions throughout the summer months (Fig. 15). In addition, nearly all of this TP was in a soluble form as SRP. More frequent partial and complete water column mixing in 2009 resulted in reintroduction of dissolved oxygen to the bottom waters and disturbance of anoxic conditions, which temporally disrupted sediment diffusive P flux and the accumulation of P in the bottom waters (James 2011a). On the other hand, periods of summer mixing and fall turnover were associated with the development of algal blooms, indicating upward entrainment of SRP derived from the anoxic sediment to the lake surface for assimilation and growth. In particular, algal blooms were most severe after periods of fall turnover in 2009 and 2010 (Fig. 15 and 16). P accumulation in the bottom waters was at a peak prior to fall turnover. Upward entrainment of P during fall turnover was followed by the development of high chlorophyll concentrations that often exceeded 50 to 100 μ g/L in September and October of those years, indicating linkages to internal P loading and vertical P transport via entrainment (Fig. 15 and 16).

Phosphorus patterns in 2011 were strongly influenced by operation of the destratification system (James 2012). Unlike other years where phosphorus concentrations increased in the anoxic hypolimnion during stratification, mixing and water column destratification resulted in the development of relatively high TP throughout the water column during most of the summer period (Fig. 15). Although hypolimnetic TP concentration gradients

were much lower in 2011, lake-wide mean TP concentrations increased linearly from late May through the end of August and the maximum lake-wide mean TP concentration exceeded 0.3 mg/L in 2011 versus the other summers (Fig. 16). In addition, a substantial portion of the TP was in dissolved form as SRP throughout the water column (Fig. 15 and 16). In particular, concentrations of SRP exceeded 0.1 mg/L in the surface waters from mid-July through September (Fig. 15) and the maximum lake-wide mean approached a peak near 0.2 mg/L in late August (Fig. 16).

In conjunction with destratification and higher concentrations of SRP in the euphotic zone in 2011, several peaks in chlorophyll were observed throughout the summer period (Fig. 15). This pattern of apparently greater summer bloom frequency throughout the summer of 2011 contrasted with the predominance of peak chlorophyll concentrations confined primarily to fall turnover periods in 2009 and 2010, when the destratification system was not in operation. These contrasting patterns indicated that summer destratification was actually enhancing internal P loading and vertical transport for algal uptake, rather than controlling P.

Secchi transparency exhibited a significant inverse pattern to that of chlorophyll, indicating that light extinction was due to algae versus inorganic turbidity (Fig. 17). Secchi transparency was generally greatest in May and June of 2009-10 and declined to minima, often less than 1 m, in late August through September in conjunction with fall turnover and the development of algal blooms in the lake (Fig. 18). In contrast, Secchi transparency was lowest during the July through September period of 2011 in association with operation of the destratification system, high chlorophyll, and high soluble P concentrations in the surface waters.

Total and dissolved fractions of Fe, as well as P concentrations, increased in the bottom waters immediately above the sediment interface in conjunction with the development of anoxia (Fig. 19). SRP accounted for most of the TP at this depth stratum. In contrast, DFe concentrations were lower relative to TFe, suggesting that some of the TFe was in particulate form, perhaps as an iron oxyhydroxide (Fe~(OOH)). Regression relationships

between bottom water concentrations Fe and P indicated that the TFe:TP (mass:mass) and DFe:SRP ratios were relatively low at 1.89 and 0.62, respectively. Molar ratios were 1.05 and 0.34 for TFe:TP and DFe:SRP, respectively. Low Fe relative to P in the bottom waters during the summer anoxic period indicated poor P binding efficiency onto precipitated Fe~(OOH) during periods of water column mixing and reintroduction of dissolved oxygen (Hoffman et al. 2013). Gunnars et al. (2002) found that P binding was complete at a mass Fe:P ratio of ~3.6:1 (2.0:1 molar ratio), much higher than observed in Cedar Lake. This finding suggested that much of the soluble P in the bottom waters remained in solution rather than becoming adsorbed or precipitated to Fe~(OOH), providing further evidence that algal bloom development in the fall was related to apparent vertical entrainment of soluble P into the surface water.

Mean summer epilimnetic and entire water column ecological indicator variables are shown in Table 9. The averaging period was defined as June through October to account for the usually high concentration of nuisance algal blooms that occurred after fall turnover (other averaging periods are shown in Table 9 for comparison with State of Wisconsin assessment methods). Generally, mean TP and chlorophyll concentrations were higher in 2010 versus 2009, due to the more prolonged period of stable stratification and bottom water anoxia in the latter summer which allowed for the accumulation of internal P loads in the lake prior to fall turnover. Mean TP concentrations were greatest during operation of the destratification system in 2011. TSI estimates from mean TP and chlorophyll concentrations were between ~ 63 and 72 in 2009 and 2010, indicating eutrophic to hypereutrophic conditions. TSI_{TP} and TSI_{CHLA} means were also similar in value, indicating that algal growth was P-limited in 2009 and 2010. As discussed in James (2012), continual vertical entrainment of P (primarily as SRP) derived from internal loading during operation of the destratification system in 2011, resulted in very high mean TP and SRP concentrations and mean TSI_{TP} that approached 80. However, TSI_{CHLA} was 11 to 14 units lower than TSI_{TP} in 2011 (Table 9) because algal growth was most likely limited by N rather than P (James 2012). Mean TSI_{SD} was typically lower than the other TSI indicators as a probable result of cyanobacterial populations which

exhibited high mean concentrations but allowed for greater light penetration into the water column due to flake and strand-forming colonies.

Cyanobacteria overwhelmingly dominated phytoplankton abundance, and biomass to a lesser extent, during the 2009-11 summer periods and were generally most abundant during the fall turnover period in mid-August through September (Fig. 20 and 21). In addition, potential toxin-producing cyanobacteria taxa dominated the assemblage during these periods (Fig. 20 and 21). Overall, many of the cyanobacteria taxa identified in Cedar Lake had the potential for producing toxins (Table 10). Aphanocapsa incerta dominated the abundance of potential toxin-producing cyanobacteria taxa during the fall turnover period of 2009 (Fig. 22). Aphanizomenon flos-aquae and Woronichinia naegeliana were also abundant during that period. In 2010, Aphanocapsa incerta was the overwhelming dominant taxon during fall turnover. In contrast, Microcystis smithii and *M. novacekii* dominated the fall assemblage in 2011. Variations in the fall cyanobacteria assemblage between 2009 and 2010 may have been due, in part, to differences in mixing patterns during both the summer and fall periods, which promoted differential disturbance and disruption of the phytoplankton community. For instance, the lake exhibited more frequent periods of partial and complete water column mixing in the summer of 2009 followed by a several intermittent periods of stratification after fall turnover. By contrast, the lake was strongly stratified throughout the summer of 2010, followed by complete mixing and cooling during the autumn. In 2011, operation of the destratification system may have played a role in influencing the assemblage of fall potential toxin-producing cyanobacteria taxa due to nearly complete mixing throughout the summer and fall.

Anoxic Sediment Phosphorus Release Rates and Sediment Characteristics

Flocculent (i.e., high moisture content and low bulk density) sediments located at depths greater than 25 ft generally exhibited highest anoxic P release rates and concentrations of

Fe-P (Fig. 23). The mean anoxic P release rate was high at 9.5 mg/m² d for all sediment sampling stations and 11.7 mg/m² d for stations located at depths greater than 25 ft (Table 11). Within the 25 ft depth contour, anoxic P release rates were greatest along the north transect and central to southern transects with rate maxima of ~ 18 mg/m² d in both the central and northern portion of the lake (Fig. 23).

For sediment sampling stations deeper than 25 ft, mean Fe-P, which is empirically related to anoxic P release rates (Nürnberg 1988), was modest at 0.230 mg/g (range = 0.169 to 0.307 mg/g; Table 11 and Fig. 23). Mean biologically-labile P (i.e., subject to recycling back to the overlying water column; loose-P, Fe-P, and labile organic P) accounted for ~ 52% of the mean sediment TP concentration and labile organic P, which can be recycled to the water column as a result of bacterial metabolic processes, was the dominant fraction (Fig. 24 and 25). Mean TP was moderate at 1.517 mg/g (Fig. 25) and the mean Fe:P ratio was high at greater than 13:1 (Table 11), suggesting iron control of P release under aerobic conditions (Jensen et al. 1992). James (2011a) found that rates of P release from intact sediment cores were negligible when incubated under aerobic conditions in the laboratory, supporting this hypothesis.

Vertical concentration profiles in the sediment indicated peak concentrations of redox-P (i.e., the sum of loosely-bound and iron-bound P; active in P release under anaerobic and reducing conditions) and biologically-labile P species in the upper 8- to 12-cm sediment layer (Fig. 26). Concentrations of these constituents declined and were relatively constant below the 15-cm sediment layer. This pattern suggested the buildup of mobile P in the upper sediment layer in excess of breakdown and burial, a pattern that is typical for eutrophic lake sediments (Carey and Rydin 2011). In particular, peak concentrations of loose-P and Fe-P of ~ 0.13 mg/g near the surface and ~ 0.60 mg/g at 2.5 cm, respectively, were relatively high and probably play an important role in explaining the high anoxic P release rates in the lake. Overall, vertical profile patterns for redox-P suggested that the upper 8-cm sediment layer at a minimum should be considered in aluminum sulfate dosage estimation to control internal P loading in the lake (see *Alum Dosage Estimation* below).

I used a similar approach to that of Rydin et al. (2011) to roughly estimate the turnover time of excess sediment TP, and the gross P deposition rate, P burial rate, gross P release rate, and P burial efficiency from sediment core vertical concentration patterns (Fig. 26f). These estimates represented only ballpark approximations intended to provide an indication of the magnitude of P burial in relation to gross P deposition in Cedar Lake. Given that the current thickness of the excess total P layer is ~ 12 cm (or 120 mm; Fig. 26f) and the mean sedimentation rate (reported in Garrison 2002) since ~ 1975 has been ~0.08 g/cm² y (range = 0.06 to 0.10 g/cm² y), a relatively long sediment TP turnover time for this layer was estimated at ~ 163 y (i.e., 120 mm \div 0.8 mm/y; Table 12). Assuming a mean seston sedimentation rate of ~ 800 g/m² y over this same period (Garrison 2002), the gross P deposition rate was ~ 6,427 kg/y (Table 12). This rate was 2X greater than the P burial rate, suggesting only ~ 50% P burial efficiency. Rydin et al. (2011) reported TP burial efficiencies ranging between 51 and 66% for coastal Baltic Sea regions. The difference between the gross P deposition and P burial rate represented the annual rate of internal P loading (Table 12). The calculations suggested that it would take on the order of decades to flush excess sediment TP from the lake in order to reduce internal P loading.

Summer and Annual Phosphorus Budget Analysis

Summer phosphorus budget and net internal phosphorus loading. The time period used for estimating $P_{net internal loading}$ was based on the net increase in $P_{lake storage}$ between an early summer minimum (i.e., near the onset of stratification) and a maximum P mass that occurred near the onset of fall turnover. Because P mass fluctuated over the summer, regression analysis was used to estimate the average change in $P_{lake storage}$ that occurred over the selected period. $P_{lake storage}$ mass exhibited a net increase of ~ 2700 kg and ~3900 kg in 2009 and 2010, respectively (Table 13). In contrast, $P_{lake storage}$ mass increased tremendously during the summer of 2011 (~ 6900 kg) in conjunction with operation of the destratification system (Table 13). Net retention of external P loads (i.e., R_{Sed}) was very low in 2009 and 2010 and negative in 2011 (i.e., $P_{outflow} > P_{external load})$, indicating that $P_{net internal load}$ (solved by difference using equation 4) was the overwhelmingly dominant P source to the lake. The area-based $P_{net internal load}$ was relatively high 6.1 and 8.8 mg/m² d in 2009 and 2010, respectively. It was very high in 2011 at ~16 mg/m² d.

Annual internal and external phosphorus loading. A range of annual internal P loading estimates derived from empirical equations (Nürnberg 1998, 2009; Walker 1996), changes in summer or fall lake P concentration, and from the anoxic factor and laboratory-derived anoxic P release rates are shown in Table 14. The calculation of gross versus net internal P loading is distinguished by inclusion of the portion that becomes deposited back to the sediment on an annual basis (Nürnberg 2009). For stratified lakes exhibiting hypolimnetic anoxia, gross internal P loading technically represents an approximation of diffusive P flux from anaerobic sediment and theoretical accumulation in the absence of biological uptake, recycling, precipitation, and deposition. Other mechanisms such as bioturabation; aquatic macrophyte-mediated root uptake, P translocation, senescence, and leaching; and diffusive P flux from sediment under aerobic conditions may also contribute to gross internal P loading. In addition, a portion of the gross internal P load can originate from newly deposited tributary P loads during the current year as well as from previously deposited sediment, but differentiation of these P inputs is very difficult. Porewater P derived from deeper sediment layers can also diffusive upward, contributing to the mobile P pool at the sediment-water interface.

In contrast, net internal P loading represents diffusion out of minus deposition back to the sediment. For example, the change in P mass over the summer stratified period approximates net internal P loading because the calculation includes deposition mostly of recycled internal (and newly deposited external P loads) during the same timeframe (Table 13). Overall, true gross and net internal P loadings are difficult to estimate and current calculation methods fall somewhere along a continuum between these two endpoints (Fig. 27).

For Cedar Lake, estimate 1 of Table 14 approximated an annual *net* internal P load while other rate estimates approximated or approached annual *gross* internal P load for 2009 and 2010. The year 2011 was not included because summer operation of the destratification system represented abnormal conditions that greatly enhanced internal P loading. Also included for comparison was internal P loading estimated from vertical total P profiles in the sediment (i.e., from Table 12). The mean (i.e., the average of estimate number 2-7; Table 14) annual internal P load estimate was 2,573 kg/y. By comparison, the gross internal P loading rate estimated from vertical sediment total P profiles (Table 14) was slightly higher at 3,281 kg/y. This was not surprising because this estimate approximated a gross internal P loading rate integrated over a nearly 40 year period (1975 to present) that included years of destratification system operation.

Like the summer period, internal P loading represented a very significant percentage of the measured P sources to the lake on an annual basis (Table 15 and Fig. 28). Even though the anoxic factor, and potential for P release from sediments, was only ~ 48 days during each year, mean internal P loading still represented 64% of the measured annual P input to the lake (Table 15 and Fig. 28). Potential bottom water anoxia and diffusive P flux from sediment may have also occurred in the winter, but was not measured or accounted for in the internal P loading estimate and annual P budget. During the summer period, internal P loading accounted for ~ 85% of the measured annual P input to the lake (Table 15 and Fig. 28).

Annual Steady State Empirical Modeling

Walker (1996) Bathtub lake P projections. Empirical models used in the Bathtub annual projections are shown in Table 16. All coefficients were set to default and the phosphorus model was calibrated by incorporation of an annual internal P loading rate (mg/m² d) as an additional P input to predict annual mean lake TP concentration (Table 17), versus adjusting (i.e., reducing) the value of the P sedimentation coefficient to account for internal P loading. Means over the years 2009-10 were used as input. Three P loading reduction scenarios were examined (Fig. 29). Scenario 1 represented a nearly

complete reduction (90% or greater) in internal P loading via, for instance, an alum treatment. Scenario 2 represented a 30% reduction in watershed tributary P loading and Scenario 3 simulated internal P loading control plus a 30% reduction in watershed tributary P loading. Implementation of Scenarios 1 or 3 would result in a \sim 50% to \sim 70% P load reduction over current conditions due to control of internal P loading. Scenario 2 represented an \sim 10% reduction over current conditions.

Lake water quality responses to future annual *tributary* P loading reductions or increases were minor (Fig. 30). For instance, a 30% reduction in annual tributary P loading through best management practices (BMP) implementation resulted in only a ~ 9% decrease in the mean annual lake TP concentration to ~ 55 μ g/L (Fig. 30). Under the same scenario, chlorophyll declined by 13% to ~ 28 μ g/L (Fig. 30). Secchi transparency increased to ~ 2.2 m, a 13% improvement over current conditions (Fig. 30).

In contrast, reducing P release from sediment via, for instance, an alum treatment resulted in substantial and immediate improvement in mean summer water quality variables (Fig. 30). In addition, the model predicted greater water quality improvement from additional tributary P loading reduction after inactivation of internal P loading. This effect was likely related to probable enhancement of the severity of P limitation to algal productivity and, thus, increased sensitivity to tributary P loading reduction. For instance, predicted mean TP and chlorophyll declined by ~ 43% (to 35 μ g/L) and 56% (to 14.5 μ g/L), respectively, while Secchi transparency increased to ~ 3.7 m as a result of internal P loading reduction. Furthermore, a 30% reduction in tributary P loading coupled with inactivation of internal P loading resulted in an additional ~20% decline in predicted TP (to 27 μ g/L) and chlorophyll (to 10 μ g/L).

Summer chlorophyll bloom frequency response to tributary and internal P loading changes followed a similar pattern (Fig. 31). Under current tributary P loading conditions, the model predicted that blooms exceeding chlorophyll concentrations of 30 μ g/L (i.e., considered a nuisance threshold level) occurred 44% of the time during the summer period. A 30% reduction in tributary P loading resulted in a decline in 30 μ g/L

chlorophyll bloom frequencies to ~35% of the time. Reducing sediment internal P loading under current tributary P loading conditions resulted in a predicted substantial decline in summer chlorophyll bloom frequency to only 7% of the time. A 30% reduction in tributary P loading coupled with inactivation of sediment internal P loading resulted in a predicted chlorophyll bloom frequency decline to less than 3% of the summer period. Thus, inactivation of sediment internal P loading coupled with a modest reduction in tributary P loading offered substantial improvement in predicted water quality characteristics.

Nürnberg (1998) lake P projections. The Nürnberg model predicted similar future lake TP trends following P loading reduction (Fig. 32). In particular, control of internal P loading alone (i.e., no watershed management) resulted in a predicted reduction in annual lake TP concentration of 49% to ~ 31 μ g/L. This pattern contrasted with tributary P loading control in that it would take a greater than 90% reduction in watershed P inputs to produce a similar lake TP concentration response (Fig. 32).

Lake Rehabilitation Goals and Management Scenarios

Current annual lake TP concentrations for Cedar Lake were high at 0.062 mg/L for 2009-10 (Table 18). The annual lake TP concentration was much higher at 0.104 mg/L in 2011 in conjunction with operation of the destratification system (not presented in Table 18). High annual lake TP coincided with high mean summer chlorophyll concentrations (33.1 μ g/L) and relatively low summer Secchi transparencies under current P loading conditions (Table 18).

Empirical modeling of conditions in 2009-2010 suggested that control of internal P loading alone via, for instance, an alum treatment, could result in a projected annual lake TP concentration of ~0.033 mg/L, depending on the empirical model (Table 18). Annual lake TP concentration reduction to this level is feasible and would well exceed TMDL requirements. Projected mean summer chlorophyll concentrations after management of internal P loading would fall to ~ 13 μ g/L while projected mean summer Secchi
transparency would increase to ~ 4.0 m under this management scenario. The bloom frequency of nuisance levels of summer chlorophyll (\geq 30 µg/L) was projected to decline to ~ 7% (Table 19).

In contrast, while a 30% reduction in watershed P loading would also result in a lower projected annual lake TP, concentration response would be less compared to internal P loading control due to the dominance of this latter P source to the overall P budget of the lake. For instance, projected annual lake TP would decline to ~ 0.054 mg/L under this watershed management scenario (Table 18). Projected mean summer chlorophyll concentrations of ~ 28 μ g/L would represent a slight improvement over current conditions (Table 18), but projected nuisance algal bloom frequencies would still be unacceptable at 35% of the summer (Table 19).

Combined watershed and internal P loading management would result in projected further declines in annual lake TP concentration to only ~ 0.025 mg/L (Table 18). Much lower annual lake TP concentrations would be associated with projected mean summer chlorophyll concentrations of less than ~ 9 μ g/L and very clear projected Secchi transparencies of greater than 5 m. Finally the projected frequency of occurrence of nuisance algal blooms would essentially be less than 5% under this management scenario (Table 19).

An unknown in the above scenarios is the projected long-term response of internal P loading if only tributary P loading is managed and reduced. Internal P loading will eventually reach a new lower equilibrium over time as excess sediment P flushes from the lake. However, the timeframe for lake TP recovery is difficult to predict and could take decades.

To explore the feasibility of reducing tributary P loading by \sim 30%, changes in the type of land use to reduce runoff and improvements in conservation by reducing P fertilization, were estimated using a P export coefficient approach (Table 20). Currently, agricultural land use practices accounted for \sim 33% of the Horse Creek watershed

draining directly into Cedar Lake. Lesser managed forested/woodlot and grass/prairie areas occupied ~ 30 % of this watershed. To approximate current (2009-10) annual P loading conditions, relatively modest P export coefficients were used to represent row crop and forage land use practices. However, these fell well within ranges reported in the literature (Panuska and Lillie 1995; Lin 2004).

Although there are many possible scenarios, two example scenarios were explored to reduce annual tributary P loading by $\sim 30\%$; 1) change land use by converting forage and row crop production to grass/prairie/CRP and 2) change the P export coefficient to reflect improved conservation and control of soluble P transport. Under scenario 1, conversion of ~ 1,300 ha from forage and row crop production to grass/prairie/CRP would represent $a \sim 30\%$ reduction. In this scenario, $\sim 55\%$ of the forage and row crop production would need to be converted to grass/prairie/CRP with concomitant reduction in the P export coefficient. For scenario 2, conservation practices that reduced soluble P runoff from forage and row crop production and residential area would be most practical for decreasing P export (Table 20). For example, conservation practices that resulted in a 38% and 33% decline in P export coefficients from forage and row crop production and residential land-use practices would reduce tributary P loading to target goals. A combination of land use conversion to grass/prairie/CRP and BMPs to existing forage and row crop production fields and residential area will be the most feasible approach to tributary P loading reduction. Scenarios will ultimately need to be generated based on specific P index and soil test P findings that identify critical source areas in the Horse Creek watershed (Wojchik 2013). In addition, runoff reduction will need to target soluble P.

Aluminum Sulfate Dosage Determination

Redox-P concentrations in profundal sediments (i.e., upper ~ 10-cm layer homogenized) collected from several stations in the lake ranged between ~ 0.24 and 0.32 mg/g (mean = $0.267 \text{ mg/g} \pm 0.017 \text{ S.E.}$; n = 4). The Al dosage required to bind at least 90% of the redox-sensitive P was ~ 17 mg Al/g DW sediment, while the measured Al:P ratio (i.e.,

parts of Al required to bind one part of redox-sensitive P) was ~ 66:1 (Fig. 33). By comparison, the measured Al:P ratio for Cedar Lake sediments was similar to overall regression relationships developed from several lakes in the region (Fig. 34).

The vertical redox-P profile collected near the center of the lake (Fig. 26d) was used to estimate Al dosage scenarios based on thickness of the sediment layer to be inactivated. For instance, redox-P concentrations were in excess of background levels in the upper 8 - cm layer and peak concentrations were observed in the upper 6-cm layer. The Al dosage estimated to control redox-P in the upper 6- and 8-cm sediment layer was ~ 100 g/m² and 130 g/m², respectively.

The cost estimate to treat profundal sediments with Al is shown in Table 21. Two treatment zones were chosen for Al application; the area encompassed by the 20-25 ft contour and areas greater than 25 ft (Fig. 35). The area between 20 and 25 ft represented the mean depth of summer anoxia and, thus, the extent of sediment exposure to anaerobic conditions. Since redox-P concentrations and rates of P release under anoxic conditions were lower in this region, versus sediments located at depths greater than 25 ft, the Al dosage was set at 100 g/m² (Fig. 35). Al dosage for sediment located at depths greater than 25 ft was set at 130 g/m² to account for excess mobile P in the upper 8-cm sediment layer (Fig. 35). Because the lake area is relatively large and profundal (i.e., sediments located in the deep central area of the lake), treatment cost was ~ \$2.16 million dollars (Table 21).

Importantly, the 100 to 130 g Al/m² dosage range estimated for Cedar Lake fell within area-based dosage ranges reported for either recently-treated lakes or proposed future treatments (Table 22 and 23). These more recent Al dosage ranges are generally higher compared to historical ranges (Huser 2012) because they were targeted toward inactivation of the excess mobile P pool in the sediment versus the rate of internal P loading (Kennedy et al. 1987). In general, these ranges suggested that an areal Al concentration of at least $\sim 100 \text{ g/m}^2$ will likely be linked to successful sediment P control.

However, more information on the long-term success of these area-based concentration ranges is needed in order to evaluate overall treatment longevity at these levels.

When converted to a volume-based Al dosage (i.e., g Al/m³), actual estimates fell near but slightly below the theoretical maximum allowable Al dosage for Cedar Lake, based on total alkalinity and pH (Table 23 and Fig. 36). Since aluminum sulfate reaction to an Al-hydroxide floc in lake water results in the production of H⁺, pH can decline rapidly to low and unsafe levels during application to poorly buffered systems, resulting in soluble Al³⁺ toxicity to biota as pH declines below 5. The theoretical maximum allowable dosage represents the estimated Al concentration that can be applied while maintaining pH levels of at least 6.0 or greater as a margin of safety. 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/m³ for stratified lakes (range = 11.7 to 30 g/m³; Table 23). The estimated volume-based Al dosage for Cedar Lake fell within the lower portion of that reported finding.

CONCLUSIONS AND MANAGEMENT RECOMMENDATIONS

The most significant finding of this research was the dominating influence of anoxic sediment internal P loading and vertical transport to the surface waters on algal productivity and biomass in Cedar Lake. Summer stratification and hypolimnetic anoxia in 2009 and 2010 coincided with the buildup of substantial soluble P gradients above the sediment interface and high concentrations by August of each year primarily as a result of diffusive P flux from profundal sediment under anoxic conditions (versus recycling of newly deposited tributary P loads). The relatively large fetch and high aspect ratio of the lake made it susceptible to periodic wind-induced mixing during the summer stratified period, resulting in metalimnetic migrations, entrainment of sediment-derived hypolimnetic P, and subsequent mixing to the surface waters for uptake by algae. This recycling mechanism played an important role in driving nuisance cyanobacterial blooms

in the lake. The greatest cyanobacterial blooms occurred in conjunction with the onset of fall turnover (late August-early September) during both years. P mass and concentrations in the hypolimnion were typically greatest at that time and in a soluble, directly available, form for assimilation. Mixing and vertical transport of P for algal uptake during fall turnover resulted in surface chlorophyll blooms in excess of 50 to 100 μ g/L that were often sustained from late August through the end of October.

A primary hypothesis explaining the prevailing vertical P transport in Cedar Lake is an uncoupling of Fe control over P dynamics at the sediment-water interface during summer hypolimnetic anoxia. In lakes receiving higher tributary loads and retention of Fe relative to sulfate (SO_4^{2-}) , Fe typically efficiently binds and retains P in the hypolimnion with minimal vertical transport to the epilimnion (Fig. 37a and b). Under these conditions, Mortimer (1941) suggested that phosphate (PO_4^{3-}) recycling was tightly coupled with the reduction of Fe^{3+} to Fe^{2+} at the sediment-water interface. According to this model, Fe in the aerobic sediment microzone is in an oxidized state (Fe^{3+}) as an Fe-oxyhydroxide (Fe~(OOH); solid precipitate) during spring overturn and has a strong adsorption capacity for PO_4^{3-} , often resulting in low to negligible P release from the sediments under oxygenated conditions (Fig. 37b). During summer stratification and hypolimnetic anoxia, bacteria use Fe⁺³ as an alternate electron acceptor during anaerobic metabolism, resulting in reduction of Fe~(OOH) to soluble Fe²⁺ and release of both SRP (which approximates the concentration of PO_4^{3-}) and DFe (which approximates the concentration of Fe²⁺) from the sediment and diffusion into the hypolimnion (Fig. 37a). Hypolimnetic Fe^{+2} becomes chemically oxidized to Fe~(OOH) via reaction with dissolved oxygen during metalimnetic migrations, adsorbing or precipitating PO_4^{3-} and removing it from entrainment into the surface waters (Fig. 37a). Fe~(OOH) precipitation and adsorption of PO₄³⁻ also occurs during fall turnover and reoxygenation (Fig. 37b). Under these scenarios, internal P recycling is confined primarily to the profundal sediment and hypolimnion, with little vertical entrainment to the surface waters for algal uptake.

However, patterns of hypolimnetic DFe and SRP accumulation in Cedar Lake deviated from the Mortimer hypothesis in that the ratio of the two substances (Fe:P w:w ratio)

was less than 2:1 before the onset of fall turnover. Gunnars et al. (2002) found that P adsorption onto precipitated Fe~(OOH) was constrained to a minimum stoichiometric Fe:P ratio of ~ 3.6:1 w:w (molar ratio ~ 2) and that considerable SRP remained in solution after formation of Fe~(OOH) if the initial molar DFe:SRP ratio was less than 2. The molar DFe:SRP ratio for Cedar Lake bottom waters was ~ 0.5, implying inefficient binding of SRP during periods of rexoxygenation (Fig. 37c and d). In contrast, the Fe:P ratio of the sediments was high. These patterns suggested the possibility that bacterially-mediated SO₄²⁻ reduction was indirectly removing DFe from diffusion into the water column via formation of FeS_{x(solid)}, which is insoluble and generally inert from further reaction (Fig. 36c; Golterman 1984, 2001; Miltenberg and Golterman 1988; Caraco et al. 1991, 1993; Amirbahman et al. 2003; Hupfer and Lewandowski 2008).

However, preliminary $SO_4^{2^2}$ determinations in 2010 indicated that sulfate concentrations were relatively low in Cedar Lake (James unpubl data), suggesting that sulfate reduction and FeS_x formation may not entirely explain the low Fe:P ratios. Chelation and sequestration of Fe by organic carbon acids can also play a role in burying Fe and removing it from recycling (Kleeberg et al. 2012, 2013; LaLonde et al. 2013). This mechanism of Fe sequestration in the sediment may be more important in reducing the hypolimnetic Fe:P ratio in Cedar Lake but more research is needed to clarify the role of organic carbon in sediment Fe dynamics.

The implications of uncoupled Fe-P interactions were that much of the hypolimnetic P remained soluble and available for algal uptake rather than becoming adsorbed to chemically-precipitated Fe~(OOH) during fall turnover and reoxygenation. Indeed, SRP concentrations were temporarily elevated during fall turnover events of both years (James 2011a), indicating availability for assimilation and growth by the algal community. SRP concentrations declined rapidly in conjunction with the development of algal blooms, indicating assimilation and conversion to a particulate organic form. In contrast, reaeration and oxidation of iron from Fe⁺² to Fe⁺³ would have been expected to result in adsorption/precipitation of SRP and deposition to the sediment surface if Fe:P ratios were higher in the hypolimnion at the onset of fall turnover.

Another significant finding and concern was the apparent linkage between internal P recycling and the development of nuisance algal blooms dominated by potential toxinproducing cyanobacteria during fall turnover periods. More research is needed to identify cyanobacterial toxins and concentrations in the lake to better understand the potential risks on human and animal health. However, the overwhelming dominance of the algal assemblage by potential toxin-producing cyanobacteria, coupled with very high chlorophyll concentrations, clearly pointed to a health concern and risk that needs to be addressed immediately via P control. Cyanotoxin exposure has been linked to liver damage, neurological impacts, illness, and even death in humans, domestic animals, and wildlife. In addition, cyanobacteria prevalence does not necessarily indicate the presence of cyanotoxins in the environment and our understanding of the cues that trigger the production of cyanotoxins is very incomplete. Orihel et al. (2012) suggested that the probability of cyanotoxicity and concentration maxima increased as the cellular mass N:P ratio declined below ~ 23 . Reasons for this correspondence are not readily apparent but may be attributed to a physiological cue to produce toxins. More likely, low N:P ratios may be an outcome of relative N deficiency that favors cyanobacterial blooms, high P loading and hypereutrophy, and luxury uptake of P by cyanobacteria (Orihel et al. 2012). James (2012) found that mass N:P ratios were \sim 20:1 or less (range = 10:1 to 21:1) during the three year limnological study on Cedar Lake and lowest during 2011, suggesting a strong potential for cyanotoxicity.

Steady-state empirical modeling indicated that management of internal P loading, primarily P release from sediment under anoxic conditions, would result in substantial improvement in mean water quality indicators. Indeed, internal P loading represented 85% of the P income during the summer and greater than 60% of the overall annual P income to the lake. In contrast, simulated reduction of tributary P loading alone resulted in less improvement of water quality indicators because it represented a smaller component of the P budget. However, the model predicted greater lake water quality response to tributary P loading reduction after control of internal P loading and lake TP to concentrations that limit algal growth. This pattern implied that watershed BMP's to

reduce P runoff will become much more effective in improving lake water quality and reducing cyanobacteria bloom frequency after control of internal P loading.

An assumption that had critical implications for the management of P in Cedar Lake was the direct incorporation of internal P loading as input into empirical models to predict current and future limnological conditions (i.e., conditions before and after internal P loading control). An alternate modeling approach was to adjust (i.e., lower) the P sedimentation coefficient to implicitly account for internal P loading input (Walker 1996). Implied in the latter approach was that reducing tributary P loading would eventually lead to reduction in internal P loading via flushing, diagenetic transformation, and burial of excess sediment P. Thus, the management goal under this scenario would be targeted toward reduction of tributary P loading to eventually reduce internal P loading. To achieve a water quality goal of ~0.031 mg TP/L entirely via watershed BMPs to eventually reduce internal P loading (versus an alum treatment; Table 18), tributary P loading would need to be reduced by ~ 50%. To reach a goal of ~ 0.025 mg TP/L (i.e., alum treatment and 30% reduction in tributary P loading; Table 18), watershed management would need to reduce tributary P loading by ~ 65% of current conditions.

However, this approach does not provide an estimate of the number of years it will take to achieve a lower internal P loading equilibrium to meet target limnological water quality goals. From a long-term study of deep and shallow lakes receiving external P loading reduction, Jeppesen et al. (2005) concluded that response time was on the order of 10 to 15 y. However, internal P loading contributions were generally not as substantial as for Cedar Lake (i.e., Lake Washington was part of the data set). In contrast, for shallow lakes with redox-P rich sediment and considerable internal P loading, negative P retention and delayed recovery continued for decades after tributary P loading reduction, as sediment P originating from depths approaching 20 cm was mobilized over time (Søndergaard et al. 2003). They also indicated that the duration and importance of internal P loading during lake recovery was related to the flushing rate (i.e., recovery was faster with increasing flushing rate increased). Importantly, however, the rapidly flushed shallow hypereutrophic Lake Søbygaard (Denmark) still exhibited negative P retention

(indicative of internal P loading) 20 years after reduction of external P loading by greater than 80% (Søndergaard et al 1999). Thus, lakes such as Cedar, with a relatively large accumulated sediment mobile P pool and low flushing rate may not respond to reduced watershed loading for many decades (Jeppesen et al. 1991).

Tributary P loads are currently modest in the Horse Creek watershed and efforts to further reduce P export from the landscape might be just as costly as an alum treatment with diminished returns on investment and a long and uncertain timeframe for lake response (Wagner 2013; Osgood 2013). Based on sediment core analysis, the moderate sedimentation rate (0.08 g/cm² y since 1975; Garrison 2002), and the relatively long residence time of the lake, reduction of excess sediment P could be on the order of decades. In addition, experimental flushing of internally-derived P has actually taken place over the last 18 years by operation of the destratification system. However, internal P loading remains high as of 2009-10, suggesting that substantial watershed P export reduction coupled with decades of flushing will be needed to reach target water quality goals. Since internal P loading and entrainment of available SRP during mixing and turnover is currently driving severe blooms of potentially toxin-producing cyanobacteria, there is an immediate human health-related need to control internal P loading via independent means.

Thus, management of internal P loading should consider application of aluminum sulfate to irreversibly bind P in the sediment. Alum is usually applied to the surface waters over the active sediment area (Fig. 38), where it disassociates to form an Al(OH)₃ precipitate that then settles to the sediment as a milky white floc. Dosages applied to inactivate mobile P fractions in the sediment can be effective in controlling internal P loading for decades if tributary P loading is low (Cooke et al. 2005), which is the case for Cedar Lake.

Major considerations and decisions regarding alum treatment of Cedar Lake included 1) the dosage concentration, 2) treatment area, and 3) buffered versus unbuffered application. Most importantly, a higher Al dosage on the order of 100 g Al/m² or greater

needs to be considered based on actual dosage assays conducted on Cedar Lake sediment (this report) and recent lake alum application case studies published in the literature (Table 22). Spatial patterns in laboratory-derived anoxic P release rates and sediment iron-bound P concentrations suggested that the potential for sediment internal P loading was greatest for areas in the lake that were deeper than 25 ft, because this depth contour encompassed the deep, expansive central basin of the lake and represented a depositional sink for P-enriched particles (Fig. 23). Thus, Al treatment should target this area of the lake at a minimum to control internal P loading. Although sediment Fe-P concentrations and anoxic P release rates were generally lower at depths less than 25 ft, seasonal anoxia extended to near the 20-ft depth contour, suggesting that shallower sediment will also represent a potential internal P source. The cost of applying a lower Al dosage of 100 g/m² to sediment located between the 20- and 25-ft contour and the higher 130 g/m² dose to sediment depths greater than 25 ft was approximately 2.16 million dollars.

Another consideration is the buffering capacity of lake water during alum application. The total alkalinity for Cedar Lake was moderate at ~100 mg CaCO₃/L, suggesting some buffering potential for moderating 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 for Cedar Lake, determined via jar tests (Cooke et al. 2005), was moderate at 20 mg Al/L. Treatment of sediment with a maximum dosage of 130 g Al/m² over the 25-ft depth contour would be equivalent to a volumetric dosage of 16 mg/L, which approaches the maximum allowable dosage. 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/m³ for stratified lakes (range = 11.7 to 30 g/m³; Table 23). The estimated volume-based Al dosage for Cedar Lake fell within the lower to median portion of that reported finding. 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.

There is currently uncertainty regarding the longevity of P control and no predictive models or empirical relationships between the strength of Al dosage and number of years

of internal P loading control. Although essential for cost-benefit analysis, this lack of predictive information is due to minimal long-term limnological monitoring of lakes treated with Al (monitoring costs are a factor). However, Cooke et al. (2005) reported that lakes receiving volumetric dosages of 10 mg Al/L or more exhibited long-term effective internal P loading control and reduction in water column total P concentration (Table 23). Effective reductions in total P concentrations reported in Cooke et al. (2005) ranged from 5 to 18 y and counting (i.e., long-term control in years may be greater but monitoring was stopped). Furthermore, Welch and Cooke (1999) found that experimental sediment P release rates under anaerobic conditions were lowest for lakes treated with Al doses of 10 mg/L or greater. Since inactivation of redox-P by Al is being targeted to control internal P loading in Cedar Lake, tributary P loading is relatively modest, and the sedimentation rate is moderate (Garrison 2002), effectiveness and longevity should be on the order of 20+ years or more. In particular, the sedimentation rate has already declined with implementation of the Horse Creek Priority Watershed Project. Further reductions in the sedimentation rate will be expected with the alum treatment and the farmer-led watershed council pilot project. Thus, any burial of the Al floc over time with new sediment will be minimal.

Two options are available that avoid the potential for exceeding the maximum allowable Al dose. One option is application of buffered alum (aluminum sulfate and sodium aluminate) to maintain pH near neutrality during reaction. Since sodium aluminate is more expensive than alum, this option would result in a substantial increase in overall cost. Option two would entail multiple alum applications at a lower dosage to prevent the risk of low pH, but over a period of several years (i.e., 2-3 year intervals). A multiple application approach has been successful in controlling internal P loading (Tiefwarensee, Germany) and has much merit as a viable treatment scenario. For instance, Al could be applied at a dosage of ~ 45 g/m² at two- to three-year intervals for a total of three treatments. This treatment scenario would be equivalent to a single application that targets the upper 8-cm sediment layer, but has several advantages. Costs are spread out over a period of several years and because each incremental dosage is low relative to the target, the Al floc has a greater chance of becoming saturated with sediment P

immediately after application. Other research has suggested that Al binding efficiency for P declines with time as the Al reacts to form more orderly Al~(OOH) polymer chains (Berkowitz et al. 2005, de Vicente et al. 2008). Sediment redox-P and aluminum-bound P could be monitored after each application for effectiveness in control of sediment P. Subsequent Al applications might ultimately be lower if previously applied alum flocs have efficiently inactivated most of the redox-P in the surface sediment layers, resulting in overall cost savings.

Alum dosage for Cedar Lake accounted for binding of the more rapidly mobilized redoxsensitive P but did not consider gradually released labile organic P and slower P diffusion upward from deeper sediments or downward from sediment freshly deposited on top of the Al floc. In particular, labile organic P represented over 50% of the biologically-labile P in the sediment and could represent a potentially important source of internal P loading. There is currently some uncertainty regarding whether simply increasing Al dosage to account for these future P sources will result in the desired longer-term control. de Vicente et al. (2008) showed that aging of the Al(OH)₃ floc without previously sorbed PO_4^{3-} could result in substantially reduced future binding efficiency (up to 75% reduction in adsorption capacity over 90 d) due to changes in crystalline structure of the floc (Berkowitz et al. 2005). They suggested that smaller doses spread out over several years (as discussed above), versus one large dose, might maintain higher binding efficiencies for these future P sources. For Cedar Lake, Al dosage could be adjusted to account for these potential additional sources of P, but more research is needed to clarify both dosage estimation and application strategies for longer-term control of labile organic P and P diffusion from adjacent sediment layers.

Management of runoff and tributary P loading has already been extensive in the Horse Creek watershed (Wojchik 2013), as indicated by the currently low annual TP export coefficients measured for the period 2009-2011. Evaluation of critical source areas (CSA's) for P runoff in the watershed using Snap-Plus (a P Index tool) has indicated that soil P fertility (mean soil test P = 34.5 ppm; mean ranges for various drainage areas = 26.6 to 52.4 ppm) and P runoff (mean P index = 1.6; mean ranges for various drainage

areas = 0.9 to 1.8) are currently modest. Future watershed management will involve the newly implemented farmer-led watershed council pilot project.

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TABLES

hed and lake characteristics.				
Characteristic		Valı	ue	
		. 2		
I otal area	140	km~	34,743	ac
Area draining directly to Cedar Lake	81	km ²	20,173	ac
Area storing runoff internally	59	km ²	14,569	ac
Surface Area	452	ha	1,118	ac
Volume	25,235,867	m ³	20,459	ac-ft
Mean depth	5.78	m	18.96	ft
Maximum depth	10.40	m	34.12	ft
Maximum fetch	3,500	m	2.17	mi
Osgood Index	2.71		2.71	
	hed and lake characteristics. Characteristic Total area Area draining directly to Cedar Lake Area storing runoff internally Surface Area Volume Mean depth Maximum depth Maximum fetch Osgood Index	hed and lake characteristics. Characteristic Total area 140 Area draining directly to Cedar Lake 81 Area storing runoff internally 59 Surface Area 452 Volume 25,235,867 Mean depth 5.78 Maximum depth 10.40 Maximum fetch 3,500 Osgood Index 2.71	hed and lake characteristics.CharacteristicValueCharacteristicValueTotal area140 km²Area draining directly to Cedar Lake81 km²Area storing runoff internally59 km²Surface Area452 haVolume25,235,867 m³Mean depth5.78 mMaximum depth10.40 mMaximum fetch3,500 mOsgood Index2.71	hed and lake characteristics.CharacteristicValueTotal area140 km²34,743¹Area draining directly to Cedar Lake81 km²20,173Area storing runoff internally59 km²14,569Surface Area452 ha1,118Volume25,235,867 m³20,459Mean depth5.78 m18,96Maximum depth10.40 m34.12Maximum fetch3,500 m2.17Osgood Index2.712.71

¹Wisconsin Department of Natural Resources (2001)

Table 2. Sediment s	Fable 2. Sediment sequential phosphorus (P) fractionation scheme, extractants used, and definitions of recycling potential.					
Variable	Extractant	Recycling Potential				
Loosely-bound P	1 M Ammonium Chloride	Biologically labile; Soluble P in interstitial water and adsorbed to CaCO ₃ ; Recycled via direct diffusion, eH and pH reactions, and equilibrium processes				
Iron-bound P	0.11 M Sodium Bicarbonate-dithionate	Biologically labile; P adsorbed to iron oxyhydroxides (Fe(OOH); Recycled via eH and pH reactions and equilibrium processes				
Labile organic P	Persulfate digestion of the NaOH extraction	Biologically labile; Recycled via bacterial mineralization of organic P and mobilization of polyphosphates stored in cells				
Aluminum-bound P	0.1 N Sodium Hydroxide	Biologically refractory; AI-P minerals with a low solubility product				
Calcium-bound P	0.5 N Hydrochloric Acid	Biologically refractory; Represents Ca-P minerals such as apatite with a low solubility product				
Refractory organic P	Determined by subtraction of other forms from total P	Biologically refractory; Organic P that isresistant to bacterial breakdown				

Table 3. Mean annual hydrological character	istics.		
Variable	2009	2010	2011
Precipitation (m)	0.655	0.899	0.735
Flow at 10th Ave (m ³ /s)	0.518	0.559	0.665
Flow at County K (m ³ /s)	0.265	0.305	0.378
Gross water income (Hm ³ /y)	19.20	21.56	24.18
Areal water load (m/y)	3.8	4.1	4.9
Unit area runoff above 10th Ave (m/y)	0.20	0.21	0.25
Unit area runoff below County K (m ^{\prime} y)	0.13	0.13	0.15
Precipitation (inches)	25.80	35.38	28.94
Flow at 10th Ave (ft ³ /s)	18.30	19.76	23.49
Flow at County K (ft ³ /s)	9.34	10.77	13.35
Gross water income (acre-ft x 10 ⁶ /y)	9.39	12.88	10.54
Areal water load (ft/y)	12.47	13.45	16.08
Unit area runoff above 10th Ave (ft/y)	0.65	0.70	0.83
Unit area runoff below County K (ft/y)	0.43	0.43	0.49

-			
Summer	Varia	ble	Rate
			(m ³ /s
2009	Lake Storage		-0.01
	Inputs	Tributary	0.416
		Precipitation	0.095
		Total	0.510
	Outputs	Evaporation	0.133
	·	Discharge	0.381
		Total	0.514
2010	Lake Storage		0.020
	Inputs	Tributary	0.517
		Precipitation	0.234
		Total	0.751
	Outputs	Evaporation	0.146
		Discharge	0.587
		Total	0.733
2011	Lake Storage		-0.002
	Inputs	Tributary	1.017
		Precipitation	0.547
		Total	1.564
	Outputs	Evaporation	0.115
		Discharge	1.545
		Total	1.660

Table 5. Mean daily flow and annual constituent loading for the Horse Creek watershed at 10th Avenue (i.e., immediately upstream of Cedar Lake) and County K (i.e., below Horse Lake).

	10th Avenue		
Variable	2009	2010	2011
Flow (m ³ /s)	0.518	0.559	0.665
Total nitrogen (kg/y)	27,131	27,983	33,240
Kjeldahl nitrogen (kg/y)	14,390	15,556	19,768
Nitrate-Nitrite (kg/y)	11,790	12,490	14,357
Ammonium (kg/y)	3,069	3,681	3,999
Total phosphorus (kg/y)	1,432	1,524	1,862
Particulate phosphorus (kg/y) ¹	951	1,007	1,182
Soluble reactive phosphorus (kg/y)	481	517	680
	County K		
Variable	2009	2010	2011
Flow (m ³ /s)	0.265	0.305	0.378
Total nitrogen (kg/y)	11,395	13,488	15,891
Kjeldahl nitrogen (kg/y)	10,083	11,935	13,863
Nitrate-Nitrite (kg/y)	1,467	1,736	2,337
Ammonium (kg/y)	880	1,042	1,247
Total phosphorus (kg/y)	784	905	1,121
Particulate phosphorus (kg/y) ¹	698	806	999
Soluble reactive phosphorus (kg/y)	86	99	122

¹Particulate phosphorus ~ total phosphorus minus soluble reactive phosphorus

Table 6. Flow-weighted constituent concentrations for the Horse Creek watershed at 10th Avenue (i.e., immediately upstream of Cedar Lake) and County K (i.e., below Horse Lake).

Variable	County K	10th Avenue
	(m	g/L)
Total Nitrogen	1.330	1.580
Kjeldahl Nitrogen	1.150	0.843
Nitrate-Nitrite	0.172	0.654
Ammonium	0.105	0.128
Total Phosphorus	0.094	0.089
Soluble Reactive Phosphorus	0.010	0.031

Table 7. Mean annual total phosphorus expor Creek watershed.	t coefficients	for the Horse
Total phosphoru	IS	
Watershed	2009	-2011
	(kg ha ⁻¹ y ⁻¹)	(lbs ac ⁻¹ y ⁻¹)
Direct drainage above County K	0.336	0.377
Direct drainage above 10th Ave ¹	0.271	0.304
Total phosphoru	IS	2 looding ²)
(corrected for export of horse Lake in		- Ioauing)
Watershed	2009-	-2011
	(kg ha ⁻¹ y ⁻¹)	(lbs ac ⁻¹ y ⁻¹)
Direct drainage above County K	0.226	0.253
Direct drainage above 10th Ave ¹	0.216	0.242

¹Includes direct drainage above County K

²Export of internal P loads from Horse Lake was estimated at 154 kg/y (J. Williamson, Polk County Land and Water Resources Division, Balsam Lake, Wisconsin)

Table 8. Mean summer (June-October) flowweighted total phosphorus (P) and soluble reactive P (SRP) in the inflow to and discharge from Cedar Lake.

Inflow	Total P	SRP		
	(mg/L)	(mg/L)	(%)	
2009	0.119	0.039	32.6	
2010	0.101	0.036	36.1	
2011	0.095	0.034	35.5	
Discharge	Total P	SR	P	
	(mg/L)	(mg/L)	(%)	
2009	0.052	0.007	12.9	
2010	0.055	0.006	10.2	
2011	0.129	0.055	42.5	

Table 9a. Mean summer (JUN-OCT) total phosphorus (P), soluble reactive P (SRP), chlorophyll, Secchi transparency and Carlson Trophic State Index (TSI) for Cedar Lake.

Year	Integrated Mean	Total P	SRP	Chlorophyll	Secchi	TSITP	TSI _{CHLa}	TSI _{SD}
	(JUN-OCT)	(mg/L)	(mg/L)	(mg/m ³)	Transparency (m)			
2009	Lake (0 - 8 m)	0.073	0.035	23.6	2.4	66.0	61.6	47.4
	Epilimnion (0 - 4 m)	0.060	0.018	27.5	2.4	63.2	63.1	47.4
2010	Lake (0 - 8 m)	0.108	0.050	36.2	1.6	71.7	65.8	53.2
	Epilimnion (0 - 4 m)	0.062	0.007	39.5	1.6	63.8	66.6	53.2
2011	Lake (0 - 8 m)	0.171	0.105	32.2	1.3	78.3	64.6	56.2
	Epilimnion (0 - 4 m)	0.165	0.097	39.0	1.3	77.8	66.5	56.2

Table 9b. Mean summer (MAY-SEP) total phosphorus (P), soluble reactive P (SRP), chlorophyll, Secchi transparency and Carlson Trophic State Index (TSI) for Cedar Lake.

Year	Integrated Mean	Total P	SRP	Chlorophyll	Secchi Transparency	TSI _{TP}	TSI _{CHLa}	TSI _{SD}
	(MAY-SEP)	(mg/L)	(mg/L)	(mg/m ³)	(m)			
2009	Lake (0 - 8 m)	0.065	0.031	17.9	2.8	64.4	58.9	45.1
	Epilimnion (0 - 4 m)	0.054	0.017	21.9	2.8	61.7	60.8	45.1
2010	Lake (0 - 8 m)	0.107	0.054	26.7	2.0	71.6	62.8	50.0
	Epilimnion (0 - 4 m)	0.059	0.008	30.6	2.0	63.0	64.1	50.0
2011	Lake (0 - 8 m)	0.157	0.094	32.4	1.3	77.1	64.7	56.2
	Epilimnion (0 - 4 m)	0.168	0.086	38.5	1.3	78.1	66.4	56.2

Table 9b. Mean summer (1 JUN - 15 SEP) total phosphorus (P), soluble reactive P (SRP), chlorophyll, Secchi transparency and Carlson Trophic State Index (TSI) for Cedar Lake.

Year	Integrated Mean	Total P	SRP	Chlorophyll	Secchi Transparency	TSI _{TP}	TSI _{CHLa}	TSI _{SD}	
	(1 JUN - 15 SEP)	(mg/L)	(mg/L)	(mg/m ³)	(m)				
2009	Lake (0 - 8 m)	0.065	0.029	19.8	2.6	64.4	59.9	46.2	
	Epilimnion (0 - 4 m)	0.051	0.013	24.2	2.6	60.9	61.8	46.2	
2010	Lake (0 - 8 m)	0.122	0.067	26.5	1.7	73.5	62.7	52.3	
	Epilimnion (0 - 4 m)	0.059	0.008	31.6	1.7	63.0	64.4	52.3	
2011	Lake (0 - 8 m)	0.173	0.109	31.7	1.3	78.5	64.5	56.2	
	Epilimnion (0 - 4 m)	0.166	0.099	39.3	1.3	77.9	66.6	56.2	

Cyanobacterial taxon	Potential producer of	Notes
Anabaena lemmermanni	anatoxin-a(s), microcystins	Sivonen et al. (1992), Belykh et al. (2011), Lepisto et
Anabaena sigmoidea		al. (2006)
Anabaena sp.		
Anabaena spiroides	anatoxin-a(s), microcystins	Greenwater
Anabaenopsis circularis	anatoxins, microcystins, saxitoxins	Greenwater; see taxonomic note below.
Aphanizomenon flos-aquae	anatoxin, saxitoxins, cylindrospermopsin	
Aphanocapsa holsatica	positive mouse bioassay	Greenwater
Aphanocapsa conferta		
Aphanocapsa delicatissima		
Aphanocapsa incerta	suspected	Greenwater
Aphanocapsa planctonica		
Aphanocapsa pulchra		synonym of Microcystis smithii
Aphanocapsa sp.		
Aphanothece clathrata		
Coelosphaerium dubium		
Gomphosphaeria natans		
Lyngbya birgei		
Microcystis aeruginosa	microcystins	Greenwater
Microcystis flos-aquae	microcystins	Baker et al. (2002), Codd et al. (2005)
Microcystis incerta		synonym of Aphanocapsa incerta
Microcystis novacekii	microcystins (-LR, -YR, -RR)	Li et al.(2009)
Microcystis smithii	microcystin (-RR)	Liu et al. (2011)
Microcystis wesenbergii	microcystins	Greenwater
Planktolyngbya limnetica		
Pseudanabaena catenata	anatoxin	Gorham et al. (1982)
Pseudanabaena limnetica		
Synechococcus sp.	marine species can produce hemolysins	Greenwater
Woronichinia compacta		
Woronichinia naegeliana	neuro/hepatotoxins; mouse bioassay; microcystin genes present	Greenwater; Oberholster et al. (2006)

Table 10. Cyanobacteria taxa in Cedar Lake and toxin production potential (G. Lalibertie, Wisconsin DNR, personal communication).

Station	Moisture content	Sediment density	Bulk density	Organic matter	Anoxic P rate	Total Fe	Total P	Fe:P	Loosely-bound P	Iron-bound P	Labile organic P	Aluminum- Bound P	Calcium- bound P	Refractory organic P
	(%)	(g/cm ³)	(g/cm ³)	(%)	$(mg m^{-2} d^{-1})$	(mg/g)	(mg/g)	(w:w)	(mg/g)	(mg/g)	(mg/g)	(mg/g)	(mg/g)	(mg/g)
SED-0	92.7	0.077	1.033	28.6	18.8	17.132	1.482	11.6	0.038	0.223	0.404	0.157	0.169	0.491
SED-1	93.5	0.067	1.028	30.9	15.7	21.894	1.742	12.6	0.046	0.244	0.439	0.201	0.169	0.643
SED-2	93.8	0.058	1.026	32.2	10.5	20.866	1.482	14.1	0.034	0.206	0.486	0.175	0.168	0.413
SED-3	93.8	0.059	1.026	32.7	8.4	22.118	1.528	14.5	0.037	0.207	0.425	0.146	0.152	0.561
SED-4	92.0	0.078	1.036	29.8	10.3	17.679	1.357	13.0	0.023	0.167	0.337	0.126	0.173	0.531
SED-7	94.2	0.059	1.024	33.7	8.7	20.358	1.59	12.8	0.051	0.227	0.504	0.185	0.188	0.435
SED-8	94.5	0.066	1.023	34.5	3.2	16.759	1.528	11.0	0.067	0.307	0.480	0.173	0.182	0.319
SED-9	94.2	0.067	1.024	34.3	8.7	21.115	1.526	13.8	0.058	0.230	0.640	0.177	0.164	0.257
SED-10	93.7	0.062	1.026	33.4	no data	22.676	1.527	14.9	0.037	0.173	0.520	0.145	0.144	0.508
SED-11	87.1	0.131	1.071	16.3	11.3	13.849	1.316	10.5	0.034	0.169	0.251	0.171	0.189	0.502
SED-12	93.3	0.074	1.030	30.1	12.8	18.766	1.567	12.0	0.052	0.212	0.580	0.158	0.186	0.379
SED-13	94.1	0.058	1.025	33.2	18.3	19.607	1.593	12.3	0.056	0.227	0.577	0.190	0.192	0.351
SED-14	94.1	0.058	1.025	33.6	9.0	21.099	1.666	12.7	0.052	0.220	0.579	0.190	0.163	0.462
SED-15	94.0	0.069	1.025	33.5	5.6	21.284	1.456	14.6	0.038	0.219	0.520	0.152	0.153	0.374
SED-16	92.6	0.087	1.034	27.2	12.7	16.892	1.687	10.0	0.083	0.289	0.530	0.318	0.153	0.314
SED-17	93.1	0.081	1.030	30.2	9.3	18.594	1.475	12.6	0.046	0.228	0.724	0.166	0.191	0.120
SED-18	93.0	0.069	1.031	30.3	13.7	20.469	1.468	13.9	0.037	0.209	0.461	0.170	0.183	0.408
SED-19	93.8	0.063	1.026	33.4	9.9	22.125	1.557	14.2	0.051	0.277	0.821	0.186	0.177	0.045
SED-20	93.5	0.068	1.027	33.9	6.0	21.552	1.49	14.5	0.048	0.201	0.527	0.152	0.162	0.400
SED-21	87.6	0.156	1.068	16.8	5.5	14.511	1.178	12.3	0.028	0.158	0.284	0.092	0.255	0.361
SED-22	92.9	0.073	1.033	26.8	11.0	17.508	1.522	11.5	0.051	0.212	0.481	0.140	0.263	0.375
SED-23	93.1	0.071	1.031	29.9	4.7	21.132	1.521	13.9	0.030	0.157	0.503	0.123	0.180	0.528
SED-24	93.6	0.068	1.028	31.1	7.0	21.791	1.583	13.8	0.032	0.167	0.530	0.128	0.233	0.493
SED-25	93.8	0.065	1.027	31.3	4.6	21.086	1.614	13.1	0.037	0.219	0.493	0.142	0.165	0.558
SED-26	93.1	0.081	1.031	29.1	2.6	20.570	1.458	14.1	0.029	0.153	0.410	0.122	0.242	0.502
MIN	87.1	0.058	1.023	16.3	2.6	13.849	1.178	10.0	0.023	0.153	0.251	0.092	0.144	0.045
MAX	94.5	0.156	1.071	34.5	18.8	22.676	1.742	14.9	0.083	0.307	0.821	0.318	0.263	0.643
CV	1.9	30.1	1.2	15.4	45.7	12.4	7.7	10.3	31.0	18.9	24.5	25.4	17.3	32.7
MEAN	93.0	0.075	1.032	30.3	9.5	19.657	1.517	13.0	0.044	0.212	0.500	0.163	0.184	0.413

Table 11. Textural and chemical characteristics of the upper 10-cm sediment section collected at variation stations in Cedar Lake. P = phosphorus, Fe = iron, CV = Coefficient of Variation.

Table 12. Estimated annual gross phosphorus (P) deposition, burial, and release in Cedar Lake (modified from Rydin et al. 2011). The mean total P concentration in the upper sediment layer (0 - 4 cm) was used to estimate the gross P deposition rate. The mean total P concentration in the lower sediment layers was used to estimate the P burial rate. See Figure 26f for mean total P concentrations in the various sediment layers.

Variable	Value		Calculations		
Gross P deposition rate	6,427	kg/y	1.778 mg/g x 800 g/m ² y x 4.52 km ² x 1 kg/1,000,000 mg		
P burial rate	3,146	kg/y	0.870 mg/g x 800 g/m ² y x 4.52 km ² x 1 kg/1,000,000 mg		
P release rate	3,281	kg/y	Gross P deposition rate - P burial rate		
P burial efficiency	49	%	P burial rate/Gross P deposition rate		

Table 13. Summer phosphorus (P) budget for estimation of internal P loading in Cedar Lake. See equation 4 and text for variable definitions.

P Flux	6/23/09 to 9/29/09			
	(kg)	(mg m ⁻² d ⁻¹)		
P _{lake storage}	2662	6.2		
P _{external load}	215	0.5		
Poutflow	163	0.4		
P _{net retention}	52	0.1		
P _{net internal load}	2610	6.1		
P Flux	5/12/10) to 8/17/10		
	(kg)	$(mg m^{-2} d^{-1})$		
Plake storage	3931	9.3		
P _{external load}	445	1.1		
Poutflow	238	0.6		
P _{net retention}	208	0.5		
Pnet internal load	3723	8.8		
P Flux	6/1/11	to 9/13/11		
	(kg)	$(mg m^{-2} d^{-1})$		
P _{lake storage}	6860	15.1		
P _{external load}	597	1.3		
Poutflow	841	1.9		
P _{net retention}	-244	-0.5		
Pnet internal load	7104	15.7		

2011 is not included due to operation of the destratification system.					
Estimate	Internal P load estimate	2009	2010	Average	
1	Net internal P loading from Nurnberg (1998) empirical model	429	713	571	
2	Gross internal P loading from Nurnberg (1998) empirical model	1,375	2,219	1,797	
3	Partially gross internal P loading from Bathtub (Table 17)			1,989	
4	Partially gross internal P loading from fall turnover P	1,869	2,630	2,250	
5	Partially gross internal P loading from summer P budget (Table 13)	2,610	3,723	3,167	
6	Gross internal P loading from P release rates and anoxic factor (Table 11)	2,897	3,011	2,954	
7	Gross internal P loading estimated from sediment core profiles (Table 12)			3,281	
	Mean of estimates 2 through 7 (~ gross internal P loading)		2,573		

 Table 14. Estimates of net, partially gross, and gross internal phosphorus (P) loading for Cedar Lake (kg/y). The year

 2011 is not included due to operation of the destratification system.

Table 15. Average summer (JUN-SEP) and annual external and internal phosphorus (P) loading and percent contribution to Cedar Lake during 2009-10. The year 2011 was not included in the average due to operation of the destratification system. Internal P loading represented the mean of several estimates shown in Table 14. The summer and annual internal P load represent summer inputs entirely and does not include inputs during the winter period (not measured).

	Summ	ner	Annual		
	(kg/summer)	(%)	(kg/y)	(%)	
External P load	449	15	1,478	36	
Internal P load	2,573	85	2,573	64	

Table 16. Algorithms used for Bathtub (Walker 1996)phosphorus loading reduction modeling.						
Variable	Model					
Phosphorus	Canfield & Bachmann Lakes					
Chlorophyll	Jones and Bachmann					
Secchi Transparency	versus Chlorophyll & Turbidity					

Table 17. Various mean (i.e., over 2009-2010) inputs to the Bathtub model.P = phosphorus, SRP = soluble reactive P.							
Tributary inputs	Flow	17.2	Hm³/y				
	Total P	89	µg/L				
	SRP	30	µg/L				
Lake variables	Total P ¹	62	µg/L				
	Chlorophyll ²	33	µg/L				
	Secchi transparency ²	2.0	m				
	Internal P loading	1.2	mg/m² d				

¹Repesents a mean annual concentration

²Represents a mean summer (June-October) concentration

Table 18. A comparison of current trophic state variable conditions and predicted future conditions as a function of three phosphorus (P) loading reduction scenarios;1) 30% reduction in external tributary P loading, 2) 90% or greater reduction in internal P loading, and 3) a combined 30% and 90% reduction in external tributary and internal P loading, respectively. Future predicted trophic state variable conditions were estimated using Bathtub (Walker 1996) and Nurnberg (1998) steady-state annual empirical models. Numbers in parentheses denote the predicted percent reduction from current conditions as a result of management.

Scenario	Annual total P	Summer ¹ chlorophyll	Summer ¹ Secchi		
	(mg/L)	(ug/L)	(m)		
Current conditions	0.062	33.1	2.0		
Walker (1996)					
30% reduction in external P loading	0.055	28.7	2.2		
90% reduction in internal P loading	0.035	14.5	3.7		
30% and 90% reduction in external and internal P loading, respectively	0.027	0.027 10.0		4.8	
Nurnberg (1998)					
30% reduction in external P loading	0.052	26.2	2.2		
90% reduction in internal P loading	0.031	12.2	4.0		
30% and 90% reduction in external and internal P loading, respectively	0.023	7.7	5.3		
Average					
30% reduction in external P loading	0.054	(14%) 27.5	(17%) 2.2	(110%)	
90% reduction in internal P loading	0.033	(47%) 13.4	(60%) 3.9	(193%)	
30% and 90% reduction in external and internal P loading, respectively	0.025	(60%) 8.9	(73%) 5.1	(253%)	

¹June through October

Table 19. Current and future projected algal bloom frequencies of chlorophyll concentrations greater than 20 ug/L (i.e., the WisCALM 2013 nuisance concentration threshold), 30 ug/L (i.e., perceived concentration threshold or when blooms become noticable), and 50 ug/L (World Health Organization threshold of risk associated with potential exposure to cyanotoxins; WiscCALM 2013).

	1			
Scenario	Bloom frequency (% of days)			
	20 ug/L	30 ug/L	50 ug/L	
Current conditions	69.3	44.0	16.5	
30% reduction in external P loading	60.7	35.1	11.5	
90% reduction in internal P loading	20.4	6.9	1.1	
30% and 90% reduction in external and internal P loading, respectively	7.9	2.0	0.2	
Table 20. Example changes in land use or management practices in the Horse Creek watershed to achieve a 30% reduction in tributary total phosphorus (P) loading.

Land use practice	Current			Land-use change			Management practice change		
	Area (ha)	P export (kg/ha y)	P load (kg/y)	Area (ha)	P export (kg/ha y)	P load (kg/y)	Area (ha)	P export (kg/ha y)	P load (kg/y)
Forage	1,660	0.25	415	860	0.25	215	1,660	0.15	249
Row Crop	1,162	0.50	581	362	0.50	181	1,162	0.30	349
Forest	2,158	0.10	216	2,158	0.10	216	2,158	0.10	216
Grass/Prairie/CRP	1,992	0.10	199	3,592	0.10	359	1,992	0.10	199
Open water	664	0.05	33	664	0.05	33	664	0.05	33
Wetlands	664	0.05	33	664	0.05	33	664	0.05	33
Total P load (kg/y)			1,477			1,037			1,079

Table 21. Approximate cost scenario totreat two sediment areas with differentconcentrations of aluminum sulfate.

Variable	Sediment area			
	20-25 ft contour	> 25 ft contour		
	(306 ac)	(370 ac)		
Al dosage (g/m ²)	100	130		
Alum (\$)	\$839,012	\$1,316,981		
Setup (\$)	\$7,000			
Total (\$)	\$2,162,993			

Lake	Al Dose (g Al m ⁻²)	Reference	
Cedar, WI ^{*1}	116	James (unpubl. data)	
Successful treatments			
Black Hawk, MN	145	James (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			
Squaw, WI*	120	James (unpubl. data)	
lalsted's Bay, Minnetonka, MN ^{*3}	105	James (unpubl. data)	
Bald Eagle, MN [*]	100	James (unpubl. data)	

Table 22. Recent and proposed alum (as AI) dosages for various lakes. An asterisk denotes a

¹Average of a stratified treatment at 130 and 100 g/m²

 $^{2}\text{West}$ and east arm dosages were 150 and 75 g/m², respectively

 $^{3}\mbox{Average}$ of a stratified treatment at 140 and 80 $\mbox{g/m}^{2}$

Table 23. Volume-based alum (as AI; g/m³) dosages and observed longevity of successful internal P loading control for various lakes (Cooke et al. 2005). Numbers in parentheses indicate the latest percentage reduction in internal P loading. The maximum allowable AI dose is compared with dosages determined via the Rydin and Welch (1999) assay technique for Cedar Lake.

	Lake	Al Dose	Observed Longevity
		(g Al m ⁻³)	(years)
Cedar Lake	Maximum allowable	15-16	
	100 g Al/m ² dosage	12.2 - 13.7	
	130 g Al/m ² dosage	15.9 - 17.9	
Unstratified			
	Long Kitsap Co.	5.5	11 (30%)
	Pickerel	7.3	<1
	Long Thurston Co. 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			
	Eau Galle	4.5	<2
	Morey	11.7	8 (60%)
	Cochnewagon	18	6 (0%)
	Dollar	20.9	18 (68%)
	Annabessacook	25	13 (41%)
	West Twin	26	18 (66%)
	Irondoquoit Bay	28.7	5 (24%)
	Kezar	30	9 (37%)

FIGURES



Figure 1. Watershed map showing tributary sampling locations and direct drainage areas. The Big Lake catchment area was not included in annual total phosphorus export coefficient estimation because discharge from Big Lake did not occur during the study period.



¹Wisconsin Department of Natural Resources (2001)

Figure 2. Land use practices in the Horse Creek watershed (Wisconsin Department of Natural Resources 2001).



Figure 3. Map of Cedar Lake showing the central water sampling location in 2009-2011. The red circle denotes the approximate location of the destratification system diffuser pipes. Depth contours are in feet.



Figure 4. Area and layer volume hypsograph for Cedar Lake.



Figure 5. Variations in mean summer (MAY-SEP) total phosphorus (upper panel), chlorophyll (middle panel), and Secchi transparency (lower panel). Horizontal gray bar and red circles denote years of destratification system operation. The horizontal stippled bar delineates probable periods of destratification system operation (records of actual operation during that period were not available).



Figure 6. Variations in annual precipitation between 2000 and 2011 measured at Amery, Wisconsin. Horizontal line represents the 10-year average.



Figure 7. Seasonal variations in daily precipitation and flow measured on Horse Creek at County K (i.e., below Horse Lake) and 10^{th} Avenue (i.e., immediately above Cedar Lake).



Figure 8. Seasonal time series of precipitation, Horse Creek flow at 10th *Avenue, and phosphorus (P) concentrations.*



Figure 9. Seasonal time series of precipitation, Horse Creek flow at 10^{th} Avenue, and nitrogen (N) concentrations. TKN = total Kjeldahl N (i.e., organic forms of N plus ammonium N).







Figure 11. Seasonal variations in (upper panels) total phosphorus (P) and (lower panels) soluble reactive P (SRP) concentrations for Horse Creek at 10th Avenue, Cedar Lake at Station 2 (see Fig. 3), and the Cedar Lake outflow.



Figure 12. Seasonal and vertical variations in (a) water temperature and (b) dissolved oxygen at station 2 between May and October. The light blue shaded area represents the location of the metalimnion, defined as regions where the Relative Thermal Resistance to Mixing (RTRM) was greater than 30 (Vallentyne 1957).



Figure 13. Seasonal variations in hourly water temperature for various depths at station 2.



Figure 14. Seasonal variations in (upper panels) mean, minimum, and maximum daily air temperature; (middle panels) mean lake temperature at station 2; and (lower panels) Schmidt stability at station 2.



Figure 15. Seasonal and vertical variations in (a) total phosphorus, (b) soluble reactive phosphorus, and (c) viable chlorophyll at station 2.



Figure 16. Seasonal variations the Horse Creek total phosphorus (P) load (upper panel) and Cedar Lake volume-weighted total P and soluble reactive P (middle panel) and chlorophyll (lower panel) concentrations.



Figure 17. Relationship between chlorophyll and Secchi transparency.



Figure 18. Seasonal variations in Secchi transparency.



Figure 19. Seasonal variations in dissolved oxygen (upper panel), total and dissolved iron (Fe) concentrations (middle panel), and total phosphorus (P) and soluble reactive P (SRP) concentrations (lower panel) in the bottom waters (i.e., < 0.25 m above the sediment interface) of Cedar Lake.



Figure 20. Major algal taxa (upper panel) and nontoxic versus potential toxin-producing cyanobacteria (lower panel) abundance.



Figure 21. Major algal taxa (upper panel) and nontoxic versus potential toxin-producing cyanobacteria (lower panel) biomass. Please note scale differences between upper and lower panels.





Figure 22. Relative abundance of potential toxin-producing cyanobacteria.



Figure 23. Distribution of laboratory-derived rates of diffusive phosphorus (P) flux (mg/m^2d) and concentrations of iron-bound P. Contour line represents the 25 ft depth.



Figure 24. Total phosphorus (P) composition for sediment (upper 10-cm layer) collected in Cedar Lake. Loosely-bound, iron-bound, and labile organic P are biologically reactive (i.e., subject to recycling) while aluminum-bound, calcium-bound, and refractory organic P are more inert to transformation (i.e., subject to burial). Values next to each label represent concentration (mg/g sediment dry mass) and percent of the total sediment P concentration, respectively.



Figure 25. Box and whisker plot comparing various sediment phosphorus (P) fractions measured at various stations in Cedar Lake (James 2011). Loosely-bound, iron-bound, and labile organic P are biologically-labile (i.e., subject to recycling) and aluminum-bound, calcium-bound, and refractory organic P are more are more inert to transformation (i.e., subject to burial).



Sediment P fraction





Figure 27. Conceptual diagram showing various approaches to estimate internal phosphorus (P) loading. Gross and net P loading represent endpoints along a continuum estimates. For instance, laboratory-derived sediment P release rates approximate gross internal P loading while changes in lake P mass during summer stratification or fall turnover represent an estimate somewhere in between the two endpoints.



Figure 28. Pie charts showing the summer (please note date range) and annual contributions of external and internal phosphorus (P) to the P economy of the lake.



Figure 29. Pie charts showing current and proposed future annual total phosphorus (P) loading composition to Cedar Lake. Scenario 1 = 90% reduction in internal P loading, Scenario 2 = 30% reduction in watershed tributary P loading, Scenario 3 = 90% and 30% reduction in internal and watershed P loading, respectively.



Walker (1996) Bathtub Projections for 2009-10

Tributary Phosphorus Loading (% of 2009-10 conditions)

Figure 30. Bathtub (Walker 1996) model output of predicted changes in total phosphorus, chlorophyll, and Secchi transparency as a function of increases or decreases in average 2009-10 tributary P loading conditions (black lines) and tributary plus 90% reduction in internal P loading (red lines). Dotted lines and arrows denote lake variable responses to a 30% reduction in tributary P loading.



Figure 31. Bathtub (Walker 1996) model output of predicted changes in algal bloom frequency of occurrence (as chlorophyll) as a function of increases or decreases in average 2009-10 tributary P loading conditions (upper panel) and tributary plus 90% reduction in internal P loading (lower panel).


Tributary Phosphorus Loading (% of 2009-10 conditions)

Figure 32. Nürnberg (1998) model projections of predicted changes in total phosphorus, chlorophyll, and Secchi transparency as a function of increases or decreases in average 2009-10 tributary P loading conditions (black lines) and tributary plus 90% reduction in internal P loading (red lines). Dotted lines and arrows denote lake variable responses to a 30% reduction in tributary P loading.



Figure 33. Variations in the concentration of redox-sensitive phosphorus (P; upper panel) and percent removed or adsorbed to the aluminum (Al) floc (lower panel) as a function of increasing Al concentration.



Figure 34. Relationships between the redox phosphorus (i.e., the sum of the looselybound and iron-bound phosphorus, P) concentration and the Al:P ratio (i.e., the mass of Al required to bind at least 90% of the redox P). The red circles represent Cedar Lake sediment.



Figure 35. Stratified aluminum sulfate (as Al) treatment scenarios described in Table 21. The light green area encompasses the 25-ft depth contour and receives an Al dosage of 130 g/m^2 . The light blue area represents sediment between the 20- and 25-ft depth contours and receives an Al dosage of 100 g/m^2 . The 20-ft depth is the summer average extent of hypolimnetic anoxia in the water column.



Figure 36. Nomographic representation of the relationship between total alkalinity, pH, and maximum volumetric Al dosage to maintain pH above 6.0 (after Cooke et al. 2005). Cedar Lake dosage and alkalinity ranges are shown for comparison.



Figure 37. Conceptual diagram of phosphorus (P) and iron (Fe) cycling in the hypolimnion. The left panels represent coupled iron-phosphorus dynamics while the right panels denote uncoupled dynamics. Uncoupling of Fe-P can occur when sulfate reduction is high relative to Fe, resulting in formation of iron sulfide (FeS_x). Uncoupling can also occur due to formation of organic carbon (OC)-Fe complexes due to high productivity in relation to Fe. The latter mechanism is probably responsible for low Fe:P ratios in Cedar Lake. After Hupfer and Lewandowski (2008) and Kleeberg et al. (2013).



Figure 38. Aluminum sulfate application at Half Moon Lake, Wisconsin.

APPENDICES



A1. Seasonal and vertical variations in (upper panel) total nitrogen, (middle panel) nitrate-nitrite nitrogen, and (lower panel) ammonium nitrogen at station 2.



A2. Seasonal variations in volume-weighted total nitrogen (N) and dissolved inorganic nitrogen species.



A3. Seasonal and vertical variations in (upper panel) in situ pH and (lower panel) total alkalinity at station 2 in 2010.