FINAL REPORT DOCUMENTS

Aluminum sulfate dosage and application strategies for Big Doctor Lake, Wisconsin

15 March 2022

William F. James
Aquatic Restoration and Research, LLC
607 Pine Ave
Menomonie, WI 54751
715-338-4395
wfjames.arr@gmail.com

SUMMARY

1. The recommended Al dose is ~ 95 g/m^2. The application should be split into lower doses of 40 g/m^2 in year 1, 30 g/m^2 in year 3, and 25 g/m^2 in year 5.
2. The treatment area should encompass sediments within the 7-ft depth contour (89 ac) to account for sediments located in seasonally anoxic or hypoxic areas.
3. Application should occur in mid-June or earlier to avoid peaks in chlorophyll that might interfere with Al floc formation.
4. The Al doses are above the maximum allowable Al dose to maintain pH above 6.0. Thus, a buffered alum (aluminum sulfate-sodium aluminate) application would be needed.
5. Application costs for 95 g/m^2 are approximately $457,000 for buffered alum starting in 2023. The 40 g/m^2 application in 2023 would cost ~ $181,000, the 30 g/m^2 application in 2025 would cost ~ $ 146,000, and the 25 g/m^2 application in 2027 would cost ~ $ 130,000.
6. Because alum dose and application strategies are still evolving and becoming refined with new research world-wide, I recommend an adaptive management approach using post-treatment lake and sediment monitoring to assess lake response to alum treatment. Aquatic ecosystem management with alum is complicated particularly in a large shallow lake because alum may move or become redistributed by water currents or resuspension, necessitating adjustments in future application.

The suggested alum application strategy for Big Doctor Lake.

Parameter	Year 1	Year 2	Year 3	Year 4	Year 5
Application period					
Al dose	40 g/m^2		30 g/m2		25 g/m^2
Projected cost for buffered alum	$181,000		$146,000		$130,000
Application year	2023		2025		2027

BACKGROUND

Big Doctor Lake has historically (2012-2018) exhibited relatively high average summer (June – September) total P (0.084 mg/L), chlorophyll (57.7 µg/L), and low Secchi transparency (0.8 m or 2.8 ft, James and Clemens 2020). The average summer buildup of total P in the lake is 64 kg/summer (Fig. 1) and P flux from sediment under anaerobic conditions accounts for much of the lake increase, depending on the length of time that bottom waters are anoxic. The lake does not directly receive watershed P loads from a defined tributary input and hydrology is dominated by groundwater seepage, suggesting much of the P accumulation in the lake is derived from internal P loads originating from sediment. Although wastewater P effluent from the Village of Siren, WI has historically discharged high P concentrations into an adjacent wetland, effluent P concentrations have been reduced substantially since ~ 2017. Empirical modeling suggests that control of internal P loading from bottom sediment would lead to improved summer water quality conditions that meet State of Wisconsin WQ standards (Fig. 2). Average summer total P, chlorophyll, and Secchi transparency are predicted to improve to 0.031 mg/L, 13 µg/L and > 8 ft, respectively (Fig. 2). The percent of time in the summer that algal blooms exceed 20 µg/L is predicted to improve from 91% to only 17% if internal P loading is controlled (Fig. 2).

Aluminum (Al) sulfate has been successfully used to manage and control internal phosphorus (P) loading and improve water quality in lakes since the 1970's (Huser et al. 2016). Advances in Al dosage and application strategies (Berkowitz et al. 2006, de Vicente et al 2008, Huser 2012, James and Bischoff 2020) suggest that application of lower Al dosages over multiple years and use of an adaptive management approach to monitor the effectiveness of Al in binding sediment P and controlling internal P loading will lead to improved longevity and be more cost effective.

OBJECTIVES

The objectives of these investigations are to 1) estimate the dosage of alum (as aluminum or Al, g/m^2) required to bind and inactivate mobile P fractions in the sediment responsible for internal P loading, 2) determine the alum treatment area in the lake, 3) develop an alum application strategy, 4) and provide cost estimates for alum application to Big Doctor Lake.

APPROACH

Aluminum dosage

Al dosage was estimated based on the concentration of redox-P (i.e., loosely-bound P and iron-bound P that is associated with internal P loading) concentration in the sediment for stations sampled in 2018. This dry mass P concentration (mg/g) was converted to an area-based concentration (g/m^2) as,

$$P\ (g/m^2) = P\ (mg/g) \cdot \rho\ (g/cm^3) \cdot \theta \cdot h\ (m) \cdot 1000000\ (cm^3/m^3) \cdot 0.001\ (g/mg) \qquad 1)$$

where, ρ is sediment bulk density (g/cm^3), θ is sediment porosity (100 – percent moisture content; dimensionless), and h is sediment thickness (m). The Al concentration (g/m^2) was estimated as,

$$Al\ (g/m^2) = P\ (g/m^2) \times Al{:}P \qquad 2)$$

where, Al:P is the binding ratio estimated from James and Bischoff (2015).

Maximum allowable Al dose determination

Addition of aluminum sulfate to a lake leads to hydrolysis and the liberation of hydrogen ions which lowers the pH of the water column. Since Al toxicity to the biota can occur if the pH falls below ~4, maintaining a pH $\geq$ 6.0 as a margin of safety should also be considered in dose determination (Cooke et al. 2005). For situations where alkalinity is low or the required dosage exceeds the maximum allowable dosage to maintain pH $\geq$ 6.0, a buffered aluminum sulfate-sodium aluminate treatment will be needed to maintain pH near neutrality. Surface water collected from Big Doctor Lake was analyzed for total alkalinity and pH according to APHA (2011). A titration procedure was used to determine the maximum allowable dosage of aluminum sulfate that can be added and yet maintain pH above 6.0 (Cooke et al. 2005). A 1.25 g Al/L solution of $Al_2(SO_4)_3 \cdot 18\ H_2O$ was used as the titrant: 1.0 mL additions to 500 mL of lake water were each equivalent to 2.5 mg Al/L. Lake water was titrated with the Al solution until an endpoint of pH 6 was reached. The total volume (mL) of Al solution needed to titrate lake water to pH 6 was multiplied by 2.5 mg Al/L to estimate the maximum allowable concentration. This calculation was then compared with estimates based on sediment redox-P to ensure that the latter was at or below the maximum allowable dosage. Caution needs to be used because a vertical alkalinity and pH profile over the entire vertical water column needs to be estimated to more accurately evaluate the maximum allowable dosage.

Application strategy

The application strategy was based on publications by Lewandowski et al. (2003), Berkowitz et al. (2006), de Vicente et al (2008a and b), Jensen et al. (2015), Huser et al. (2016), and James (2017). Other research has suggested development of an adaptive management approach of applying lower Al concentrations spread out over a period of years (i.e., 1-3 year application intervals) and monitoring lake response for future Al maintenance applications. The goal of these approaches is to increase overall P binding efficiency and internal P loading control longevity by stabilizing $Al(OH)_3$ polymerization and enhancing P saturation of binding sites. Application of multiple Al concentrations spread out over a period of years may be more effective in saturating binding sites, lowering the Al:P binding ratio, and stabilizing polymerization for longer internal P loading control. Dose splitting can also be used as an adaptive management approach to address slower degradation of labile organic P into mobile forms as well as increased P binding efficiency onto the Al floc.

RESULTS

Table 1. Mean redox-sensitive phosphorus (redox-P) in the upper 6-cm sediment layer and the areal Al dosage for Big Doctor Lake.

Station	Redox-P (mg/g)	Al dose (g/m^2)
1	0.695	104
2	0.523	81
	Estimated Al dose	**~95**

Sediment phosphorus concentrations in 2018 exhibited a surface peak in the upper 6-cm layer (Fig. 3). This sediment thickness was used to estimate Al dosage (i.e., *h* in equation 1 *see Methods*). The Al dose needed to bind this redox-P in the upper 6-cm sediment layer is shown in Table 1. The Al dose varied between station 1 and 2, ranging between a low of 81 g/m^2 (station 2) to a high of 104 g/m^2 (station 1). The recommended Al dose was 95 g/m^2 (i.e., the ~ average of the 2 stations). Recent lake Al dosage estimates have ranged between ~ 94 g Al/m^2 and ~145 g Al/m^2 (Table 2).

Earlier research on Big Doctor Lake found that bottom anoxia can occur for weeks, depending on the development of stratification. The sediment area impacted by anoxia is not precisely known. However, sediment areas encompassing the 7 ft depth contour are, at a minimum, anoxic during periods of summer stratification. This area should be considered for alum application (Fig. 4).

Table 2. Recent alum (as Al) dosages for various lakes.

Lake	Al Dose ($g\ Al\ m^{-2}$)	Reference
Big Doctor Lake[1]	95	Present study
East Balsam Lake, WI	100	James (2018)
Long Lake, WI	105	James (2017)
Cedar Lake, WI	100-130	James (2013)
Bald Eagle, MN	100	(Bischoff et al. 2017)
Black Hawk, MN	145	(unpubl. data)
Tiefwarensee, Germany	137	Wauer et al. (2009)
East Alaska, WI	132	Hoyman (2012)
Half Moon, WI[2]	115	James (2011)
Susser See, Germany	100	Lewandowski et al. (2003)
Green, WA	94	Dugopolski et al. (2008)

[1]Over the upper 8-cm sediment layer

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

The Al dose should be split into 3 applications and can be applied according to Table 3 to improve P binding efficiency. Since the treatment area is large there is potential for the Al floc to move via winds and water mixing during and after application (Egemose et al. 2009, 2013; James & Bischoff 2020). Wind speed and direction thresholds should be built into the application specifications. Application should occur in mid-June or earlier to avoid peaks in chlorophyll that might interfere with Al floc formation.

Table 3. The suggested alum application strategy for Big Doctor Lake.

Parameter	Year 1	Year 2	Year 3	Year 4	Year 5
Application period					
Al dose	40 g/m^2		30 g/m2		25 g/m^2

Table 4. A comparison of the maximum allowable Al dose based on a titration assay and nomograph estimate presented in (Cooke et al. 2005) and the the areal sediment redox-P based Al dosage converted to a concentration. Al dosages and longevity for other unstratified and stratified lakes are from Cooke et al (2005). Numbers on parentheses denote percent reductions in lake total phosphorus. Longevity = as of publication of Cooke et al. (2005).

Lake		Al Dose ($g\ Al/m^3$)	Observed Longevity (years)
Big Doctor Lake	Maximum allowable	1.25	This study
	Equivalent sediment dose 40 g/m^2	18.7	
	Equivalent sediment dose 30 g/m^2	14.1	
	Equivalent sediment dose 25 g/m^2	11.7	
Unstratified lakes	Long Kitsap County	5.5	11 (30%)
	Pickerel	7.3	<1
	Long Thurston County North	7.7	>8 (56%)
	Pattison North	7.7	7 (29%)
	Wapato	7.8	<1
	Erie	10.9	>8 (75%)
	Campbell	10.9	>8 (46%)
Stratified lakes	Eau Galle	4.5	<2
	Morey	11.7	8 (60%)
	Cochnewagon	18	6 (not reported)
	Dollar	20.9	18 (68%)
	Annabessacook	25	13 (41%)
	West Twin	26	18 (66%)
	Irondoquoit Bay	28.7	5 (24%)
	Kezar	30	9 (37%)

Total alkalinity was moderately low at ~10.5 mg $CaCO_3$/L, suggesting low buffering capacity for regulating pH during alum application. Al binding of P is most efficient within a pH range of 6 to 8. As pH declines below 6, Al becomes increasingly soluble (as Al^{3+}) and toxic to biota. The maximum allowable Al dosage that could be applied and yet maintain pH at or above 6, determined via jar tests (Cooke et al. 2005), was low at 1.25 mg Al/L (Table 4). This concentration was lower than all partial sediment-based Al doses (i.e., 40, 30, and 25 g/m^2) indicating aluminum sulfate-only applications would lower the pH below 6. Thus, alum must be buffered with sodium aluminate to maintain pH at safe levels.

Approximate overall costs to treat areas of the lake > 7 ft with 95 g/m^2 over 5 years starting in 2023 are shown in Table 5. If this application scenario were started in 2023, the total cost is projected at $457,000 (i.e., buffered aluminum production costs are projected to increase over time). Because alum dose and application strategies are still evolving and becoming refined with new research world-wide, I recommend an adaptive management approach using post-treatment lake and sediment monitoring to assess lake response to alum treatment. Aquatic ecosystem management with alum is complicated particularly in a large lake because alum may move or become redistributed (for instance, by wind resuspension) necessitating adjustments in future application. For instance, the second partial Al dose and application areas may need to be adjusted based on lake response. Depending on the effectiveness of the first application, the second treatment could be delayed, or the dose may be adjusted. Lake District budgetary needs and finances also need to be incorporated into the overall management plan to control internal P loading. Thus, the suggested alum dose, application area, and application strategy represent a template rather than a concrete plan that can be further adjusted and modified based on District financial and budgetary needs.

Table 5. Projected buffered alum costs for Big Doctor Lake. Costs are based on treating 89 acres (i.e., sediments located at depths > 7 ft).

Year	Al dose	Alum (gal)	Sodium Aluminate (gal)	Cost
2023	40 g/m^2	28,268	14,134	$181,000
2025	30 g/m^2	21,201	10,600	$146,000
2027	25 g/m^2	17,667	8,834	$130,000
Total	95 g/m^2	67,136	33,568	$457,000

Al treatment can on average control internal P loading for up to 21 years (Huser et al. 2016). Addition of a maintenance Al application 10-20 years later may be needed if the Al floc eventually gets buried by wave activity and sediment redistribution (Egemose et al. 2013, James and Bischoff 2020). In contrast, a maintenance dose may not be needed but it is good for planning purposes if the Al floc gets buried over time by sediment movement and redistribution.

REFERENCES

APHA (American Public Health Association). 2011. Standard Methods for the Examination of Water and Wastewater. 22th ed. American Public Health Association, American Water Works Association, Water Environment Federation.

Bischoff J, Kocian M, Beck B, James W. 2017. Restoring water quality in Bald Eagle Lake, Minnesota. Lakeline Winter 2017 p 35-42.

Cooke GD, Welch EB, Peterson SA, Nichols SA. 2005. Restoration and management of lakes and reservoirs. 3rd ed. Boca Raton (FL): CRC Press.

de Vicente I, Huang P, Andersen FØ, Jensen HS. 2008a. Phosphate adsorption by fresh and aged aluminum hydroxide. Consequences for lake restoration. Environ Sci Technol 42:6650-6655.

de Vicente I, Jensen HS, Andersen FØ. 2008b. Factors affecting phosphate adsorption to aluminum in lake water: Implications for lake restoration. Sci Total Environ 389:29-36.

Huser BJ et al. 2016. Longevity and effectiveness of aluminum addition to reduce sediment phosphorus release and restore lake water quality. Wat Res 97:122-132.

James WF. 2017. Phosphorus binding dynamics in the aluminum floc layer of Half Moon Lake, Wisconsin. Lake Reserv Manage 33:130-142.

James WF, Bischoff JM. 2015. Relationships between redox-sensitive phosphorus concentrations in sediment and the aluminum:phosphorus binding ratio. Lake Reserv Manage 31:339-346.

James WF, Bischoff JM. 2020. Sediment aluminum:phosphorus binding ratios and internal phosphorus loading characteristics 12 years after aluminum sulfate application to Lake McCarrons, Minnesota. Lake Reserv Manage 36:1-13.

James WF, Clemens C. 2020. Internal phosphorus loading from deposited and resuspended sediments in Big Doctor Lake, Wisconsin. Report submitted to the Big Doctor Lake Improvement and Rehabilitation District, 1 May 2020.

Jensen HS, Reitzel K, Egemose S. 2015. Evaluation of aluminum treatment efficiency on water quality and internal phosphorus cycling in six Danish Lakes. Hydrobiologia 751:189-199.

Lewandowski J, Schauser I, Hupfer M. 2003. Long term effects of phosphorus precipitations with alum in hypereutrophic Lake Süsser See (Germany). Wat Res 37:3194-3204.

Wauer G, Gonsiorczyk T, Hupfer M, Koschel R. 2009. Phosphorus balance of Lake Tiefwarensee during and after restoration by hypolimnetic treatment with aluminum and calcium salts. Lake Reserv Manage 25:377-388.

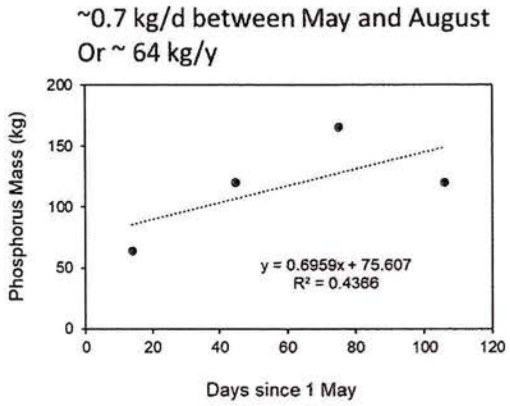


Figure 1. Increases in mean monthly phosphorus mass in Big Doctor Lake (From James and Clemens 2020).

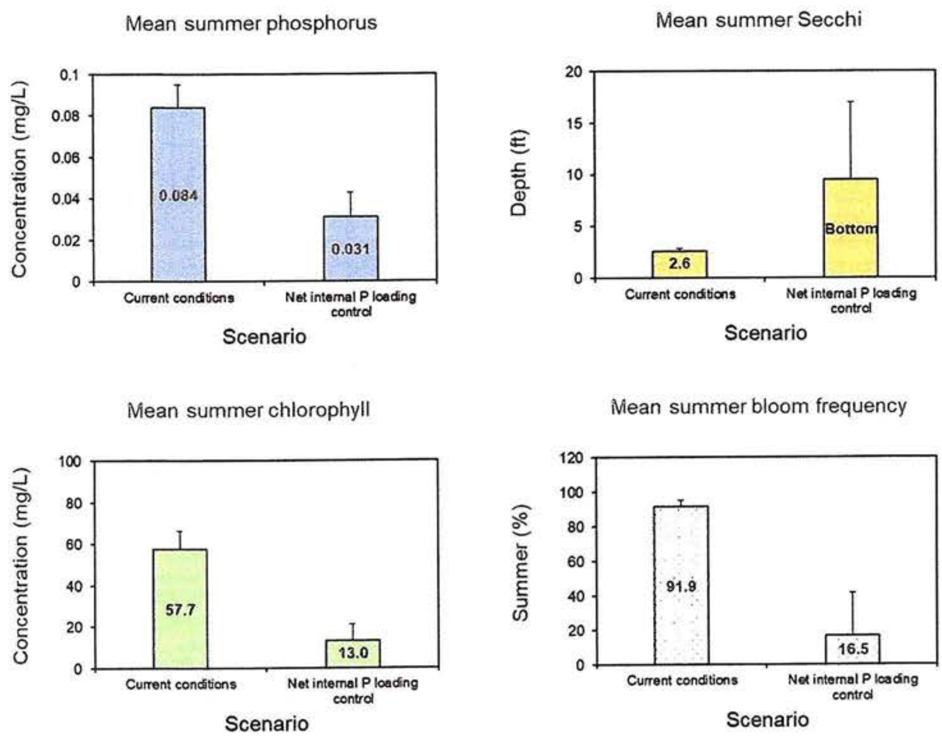
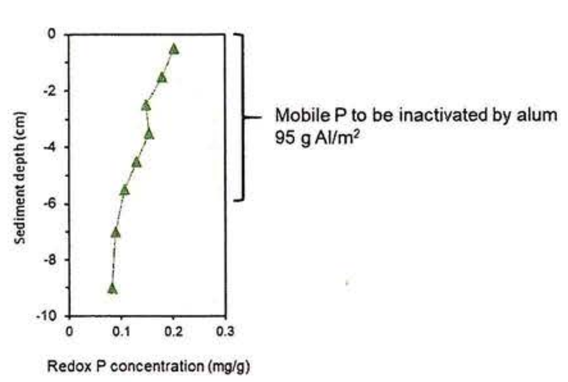


Figure 2. Current conditions and projected changes in mean summer total phosphorus, chlorophyll, Secchi transparency, and frequency of algal blooms as a result of internal phosphorus loading control (from James and Clemens 2020).

Figure 3. Vertical variations in sediment redox-P concentration (i.e., sediment P that contributes to internal P loading. The surface concentration peak is used to estimate the alum dosage.



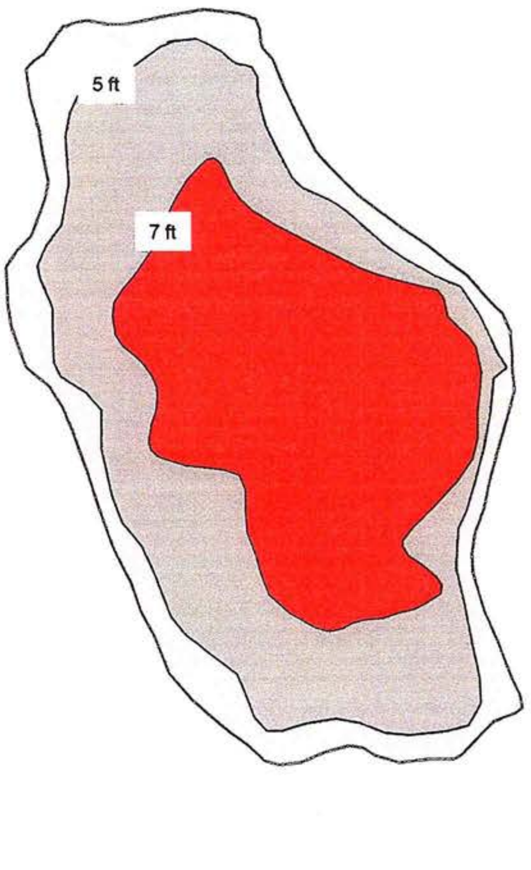


Figure 4. Recommended alum application area in Big Doctor Lake (shown in red).

Further investigations on Big Doctor Lake, Wisconsin 2021: Wetland soil phosphorus characteristics

15 March 2022

University of Wisconsin – Stout
Sustainability Sciences Institute - Center for Limnological Research and Rehabilitation
Department of Biology
260 Jarvis Hall
Menomonie, Wisconsin 54751
715-338-4395
jamesw@uwstout.edu

1.0 BACKGROUND.

A significant finding from the 2019 research (James and Clemens 2020) was that estimated diffusive P flux from sediment only accounted for ~ 55% of the lake P mass increase. Unmeasured or unaccounted for P sources could be due to longer periods of bottom anoxia and anaerobic P release than found in James and Clemens (2020). Another possible mechanism of internal P loading is direct uptake by algae of sediment P. Algal assemblage identification from samples collected in the summer would be useful in providing insight into species that vertically migrate and form resting stages in the sediment. These species (primarily cyanobacteria) can potentially take up P directly from the sediment and could thus inoculate the water column and grow with stored surplus P assimilated from sediment.

In addition, the adjacent wetland, although not directly connected to the lake via a discrete channel, may exchange hydrologically with Big Doctor Lake (Craig Roesler, personal communication). This interaction may be significant because the local village of Siren's WWTP discharged effluent into the wetland for decades. Although P in the effluent has been lowered significantly in recent years, the sediments in the wetland may have become saturated with P over time and have now become a source of P to the wetland. Exchanges of water and P between the wetland and Big Doctor Lake may provide a significant source of P for algal growth.

2.0 PURPOSE.

The objectives of these investigations are several-fold:

1. measure rates of P release from wetland sediment under anoxic conditions and wetland sediment mobile P fractions that are active in internal P loading to determine if wetland sediments are a source or sink of P,
2. identify algal species to determine those that can vertically migrate and form resting stages.

3.0 SCOPE OF WORK.

Task 1. Field sediment collection from the wetland complex

Sediment cores were collected at 5 stations in the wetland complex Big Doctor Lake in August 2021 for determination of anaerobic diffusive P flux and sediment chemistry. The stations were established along a longitudinal axis that extended from near the WWTP effluent to Big Doctor Lake (Fig. 1).

Task 2. Laboratory-derived rates of diffusive P from sediment under anaerobic conditions

Sediment accumulated in the wetland complex may represent a potentially significant internal source of P recycling to the overlying water column via diffusion from porewater to the overlying water column. Wetland soils can become anoxic, depending on the microbial dissolved oxygen demand at the sediment-water interface. While temporary flooding can provide a dissolved oxygen supply to the underlying wetland soils, rapid dissolved oxygen demand via microbial respiratory activities may rapidly reduce concentrations in the flooded soils resulting in P release into porewater. This mobilized P can diffuse into the overlying flood water if dissolved oxygen consumption exceeds supply at the sediment-water interface (Reddy and DeLaune 2008). Hydrologic exchange of this P source with Big Doctor Lake could contribute to cyanobacterial blooms. The objectives of this task were to evaluate rates of diffusive P flux from sediment under anaerobic conditions.

Wetland soil cores were carefully drained of overlying water in the laboratory and transferred intact to a smaller acrylic core liner (6.5-cm dia and 20-cm ht). Surface water collected from Big Doctor Lake were 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. They were placed in a darkened environmental chamber and incubated at a constant temperature of ~20 °C to reflect summer conditions. The Eh environment in the overlying water was controlled by gently bubbling nitrogen through an air stone placed just above the sediment surface in each system. Bubbling action ensured complete mixing of the water column but not disrupt the sediment. A total of 5 wetland soil cores were collected for assessment of anaerobic diffusive P flux.

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

Task 3. Wetland sediment P characteristics

The objectives of this task were to quantify sediment textural characteristics and mobile P fractions involved in sediment internal P loading to the overlying water column. For the wetland complex, potentially high sediment P may be related to diffuse WWTP inputs over a period of decades.

Wetland soil cores were sectioned over the upper 5 cm (i.e., 5 cm section) for analysis of P fractions. Soil sections were analyzed for the variables listed in Table 1. Subsamples were dried at 105 °C to a constant weight and burned at 550 °C for determination of moisture content, sediment density, and organic matter content (Håkanson and Jensson 2002). Phosphorus fractionation was conducted according to Psenner and Puckso (1988) for the determination of ammonium-chloride-extractable P (1 M NH_4Cl; loosely-bound P), bicarbonate-dithionite-extractable P (0.11 M BD; iron-bound P), and sodium hydroxide-extractable P (0.1 N NaOH; Aluminum-bound and labile organic P).

Table 1. Textural-physical variables and mobile sediment phosphorus pools.

Moisture content (%)
Sediment wet and dry bulk density (g/cm^3)
Organic matter content (%)
Loosely-bound P (mg/g)
Iron-bound P (mg/g)
Labile organic P (mg/g)

The loosely-bound and iron-bound P fractions are readily mobilized at the sediment-water interface under anaerobic conditions that result in desorption of P from bacterially-reduced iron compounds (i.e., Fe^{+3} to Fe^{+2}) in the sediment and diffusion into the overlying water column (Mortimer 1971, Boström 1984, Nürnberg 1988). The sum of these fractions is referred to as redox-sensitive P (i.e., redox-P; the P fraction that is active in P release under anaerobic and reducing conditions). In addition, labile organic P (LOP) can be converted to soluble P via bacterial mineralization (Jensen and Andersen 1992) or hydrolysis of bacterial polyphosphates to soluble phosphate under anaerobic conditions (Gächter et al. 1988; Gächter and Meyer 1993; Hupfer et al. 1995). The sum of redox-P and LOP is collectively referred to a biologically-labile P. This fraction is generally active in recycling pathways that result in exchanges of phosphate from the sediment to the overlying water column and potential assimilation by algae.

Task 4. Algal species ID

The lake was visited 19 August 2021 for collection of an integrated (1 m) sample for algal identification. The sample was analyzed for cyanophyta and well as other families. Cyanophyta were also be grouped into those that are potential toxin-forming species and those that form resting stages. This information will be useful in evaluating the both the potential health hazard to Big Doctor Lake and likelihood they may be obtaining P directly from sediment either as a resting stage or as a result of vertical migration to the sediment.

5.0 SUMMARY OF RESULTS

Wetland soil characteristics

In general, 3 wetland soil core sampling sites were located north of the highway 70 culvert while 2 were established south of the culvert (Fig. 1). Station 1 was established adjacent to Big Doctor Lake and station 5 was position near the Siren WWTP. During sampling. wetland soil cores wet but not covered with water: moisture content (i.e., the percentage of water in the sample) ranged from > 80% to < 50% (Fig. 2). Wetland soil moisture contents were highest for cores collected north of the highway 70 culvert and lowest south of the culvert (Fig. 2). In addition, wetland soil organic matter content was very high in station 1 and 2 cores at 78% and 89%, respectively (i.e., the soil was very organic in composition), a pattern attributable to their location in a boggy area of the wetland. These patterns suggested that the wetland soils were almost entirely organic with high interstitial porewater content north of the highway 70 culvert. In contrast, organic content was low at station 4 and moderate at station 5.

Labile organic P (i.e., the P fraction that is easily mineralized or broken down by microbial communities) concentration patterns were similar to organic matter trends (Fig 2). Labile organic P was highest at stations 1 and 2 and moderate at station 5, coinciding with organic matter concentration trends. With the exception of wetland station 3 and 4 soils, labile organic P concentrations were also high relative to those measured in ~ 50 lake sediments in the Minnesota-Wisconsin region (Fig. 3).

In contrast, iron-bound P concentration, which has been related to internal P loading from anaerobic sediments, was highest at station 5 near the Siren WWTP, suggesting potential sequestration of P effluent onto wetland soils in this area (Fig. 2). The iron-bound P concentration at station 5 was also high compared to concentrations in the lake surface sediment and other wetland soil stations, further suggesting linkages to P effluent from the WWTP (Fig. 3). Overall, iron-bound P at wetland station 5 fell near the median when compared to lake sediments in the Wisconsin-Minnesota region, indicating the concentration was not unusually high despite its proximal location to the WWTP. Other wetland soil stations exhibited moderately low iron-bound P concentrations compared to lake sediments in the region (Fig. 3).

Interestingly, rates of anaerobic diffusive P flux were not related to iron-bound P concentration (i.e., higher iron-bound P was not correlated with anaerobic diffusive P flux from wetland sediment, Fig. 4 and 5). Indeed, P fluxes were negatively correlated with organic matter content and labile organic P. For example, P fluxes were lowest for wetland soils that exhibited the highest organic matter content and they increased as a function of decreasing organic matter content. Anaerobic P flux was lowest in highly organic station 1 wetland soil, located immediately upstream of Big Doctor Lake (Fig. 5). Fluxes increased to a maximum at station 3 and were constant at near 10 mg/m^2 d at stations 4 and 5. These stations exhibited lower organic matter content that was < 40% versus station 1 and 2 (i.e., > 60% organic content). Overall, fluxes were highest at wetland stations located south of the highway 70 culvert and lower north of the culvert. Except for station 1, anaerobic P flux was relatively high, particularly at station 3 (Fig. 6).

Algal community in late August

The sample for algal identification was collected on 19 August 2021. Samples for total P and chlorophyll, collected by local residents in August, averaged ~ 105 µg/L and 141 µg/L, respectively (Lake Water Quality 2021 Annual Report (wisconsin.gov)). Cyanobacteria accounted for 89% of the algal assemblage in August (Fig. 7 and Table 1). Algal genera in the division Chlorophyta accounted for the remaining 11%. The dominant cyanobacteria were *Aphanizomenon gracile* (37%), *Dolichospermum flos-aquae* (24%, and *D. spirodes* (20% Fig. 8). *A. gracile* (Table 2) can form akinetes that overwinter in the sediments and germinate after iceout or as an overwintering vegetative population in the water column (i.e., low metabolism cellular stage; Cirés et al. 2013). Akinetes are dormant cells produced by cyanobacteria species that can overwinter in sediment in a vegetative state (Sukenik et al. 2019). They can germinate under optimal environmental conditions in the summer, become resuspended particularly in shallow lakes, and inoculate the water column for later growth and bloom formation. *A. gracile* is also potentially toxin-producing and forms cylindrospermopsin, which causes gastrointestinal inflammation in humans. *Dolichospermum sp.* also form overwintering akinetes that can incoluate the water column and produces anatoxin and microcystin, which can impact the nervous system and inflame the liver (Table 2). The rarer Microcystis can overwinter in a vegetative state in the sediments and forms microcystin (Table 2).

Conclusions and recommendations

- Although wetland soil chemistry suggests the potential for P recycling, hydrological connectivity to Big Doctor Lake is probably low, suggesting minimal impact on P loading. The wetland stations were not inundated at the time of sampling. In addition, anaerobic diffusive P flux potential was low for wetland soil stations located near the lake (i.e., stations 1 and 2). In addition, wetland soils north of highway 70 were very organic and peaty and may store P primarily as organic forms versus mobile forms.
- The algal assemblage in August 2021 was dominated by cyanobacteria. Most dominant genera potentially formed overwintering cells called akinetes that can become resuspended in shallow lakes to inoculate the water column. Germinated cells can potentially assimilate internal P loads that accumulate in the bottom waters of Big Doctor

Lake during periods of stratification and anoxia. These dominant cyanobacteria can also produce toxins that can impair human health.

- Alum treatments to control internal P loading and reduce cyanobacteria biomass appears to be a viable management tool for Big Doctor Lake.

4.0 REFERENCES.

APHA (American Public Health Association). 2005. Standard Methods for the Examination of Water and Wastewater. 21th ed. American Public Health Association, American Water Works Association, Water Environment Federation.

Boström B, Jansson M, Forsberg C. 1982. Phosphorus release from lake sediments. *Archiv Hydrobiol–Beih Erge Limnol 18:*5-59.

Boström B. 1984. Potential mobility of phosphorus in different types of lake sediment. *Internat Rev Hydrobiol 69:* 457–474.

Cirés S, Wörmer L, Agha R, Quesada A. 2013. Overwintering populations of *Anabaena*, *Aphanizomenon* and *Microcystis* as potential inocula for summer blooms. J Plankton Res 35:1254-1266.

Mortimer CH. 1971. Chemical exchanges between sediments and water in the Great Lakes – Speculations on probable regulatory mechanisms. *Limnol Oceanogr* 16:387-404.

Nürnberg GK. 1988. Prediction of phosphorus release rates from total and reductant-soluble phosphorus in anoxic lake sediments. *Can J Fish Aquat Sci* 45:453-462.

Psenner R, Puckso R. 1988. Phosphorus fractionation: Advantages and limits of the method for the study of sediment P origins and interactions. Arch Hydrobiol Biel Erg Limnol 30:43-59.

Reddy KR, DeLaune RD. 2008. Biogeochemistry of wetlands: Science and applications. CRC Press, Taylor & Francis Group, Boca Raton FL

Sukenik A, Rücker J, Maldener I. 2019. Chapter 4 - Dormant Cells (Akinetes) of Filamentous Cyanobacteria Demonstrate a Great Variability in Morphology, Physiology, and Ecological Function. Pg 65-77. In: (A.K. Mishra, D.N. Tiwari, A.N. Rai, eds), Cyanobacteria. Academic Press.

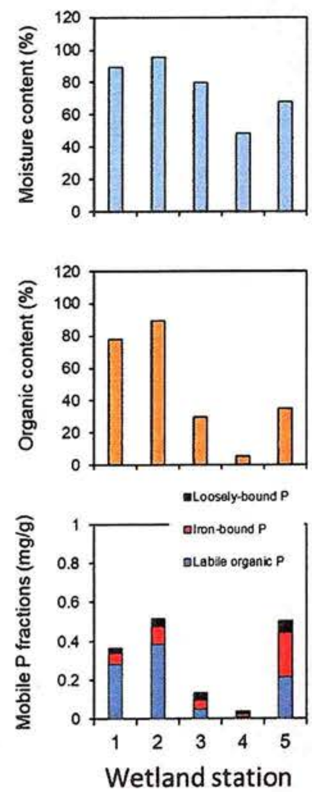
Table 1. Algal community biomass assemblage on 19 August 2021.		
	BIOMASS	
	mg/m^3	(%)
CHLOROPHYTA		
Chlamydomonas sp.	48.9	0.2
Coelastrum cambricum	659.6	2.3
Coelastrum reticulatum	195.4	0.7
Dictyosphaerium pulchellum	762.2	2.7
Elakatothrix gelatinosa	10.3	0.0
Pediastrum duplex	1062.0	3.8
Scenedesmus arcuatus	366.6	1.3
Scenedesmus brasiliensis	36.7	0.1
Scenedesmus quadricauda	175.6	0.6
Scenedesmus serratus	59.4	0.2
Staurastrum sp.	62.7	0.2
Tetrastrum triangulare	78.7	0.3
Total CHLOROPHYTA	3518.1	12.5
CYANOBACTERIA		
Aphanizomenon gracile	9,123	32.3
Dolichispermum spiroides	4,832	17.1
Dolichospermum flos-aquae	5,978	21.2
Microcystis novacekii	55	0.2
Microcystis smithii	1,220	4.3
Raphidiopsis mediterranea	3,505	12.4
Total CYANOBACTERIA	24,713	87.5
TOTAL	28231.1	100.0

Table 2. Cyanobactera genera, typical toxin production, and potential human health impacts.

Genera	Toxin	Symptoms
Aphanizomenon gracile	Cylindrospermopsin	Gastrointestinal inflamation
Dolichispermum spiroides	Anatoxin-a, Microcystin	Nervous system twitching, loss of coordination; Liver inflamation
Dolichospermum flos-aquae	Anatoxin-a	Nervous system twitching, loss of coordination
Microcystis novacekii		
Microcystis smithii	Microcystin	Liver inflamation
Raphidiopsis mediterranea	Anatoxin-a, Homoanatoxin-a, Cylindrospermopsin	Gastrointestinal inflammation; Nervous system twitching, loss of coordination

Fig. 1. Wetland soil station locations.

Fig. 2. Moisture content, organic matter content, and mobile (i.e., subject to recycling pathways) phosphorus (P) concentrations at various wetland soil sampling locations.



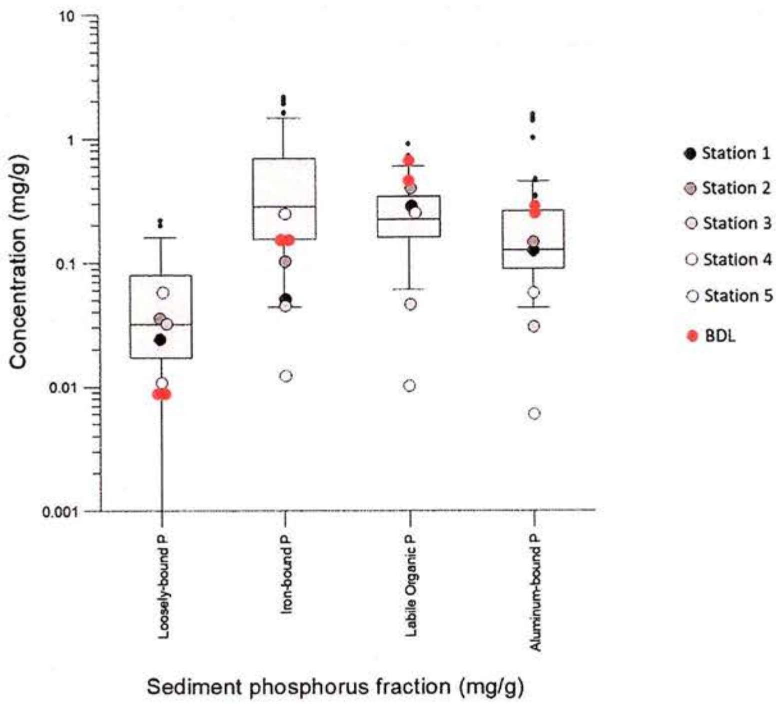


Fig. 3. Wetland and Big Doctor Lake soil phosphorus concentrations compared to lake sediments in the Minnesota-Wisconsin region.

Fig. 4. Changes in phosphorus (P) concentration in the overlying water column in anaerobic diffusive P flux incubation systems.

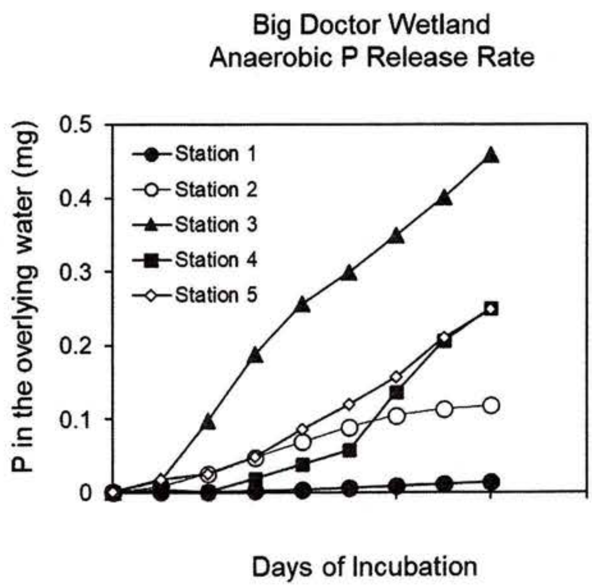
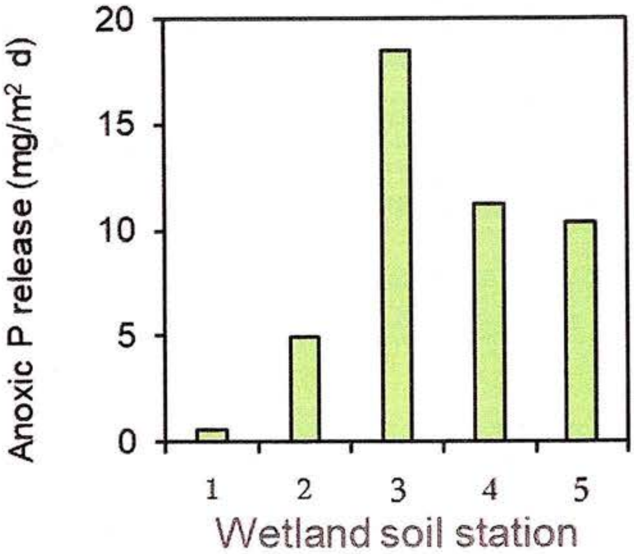


Fig. 5. Variations in laboratory-derived anaerobic diffusive phosphorus flux at wetland soil stations.



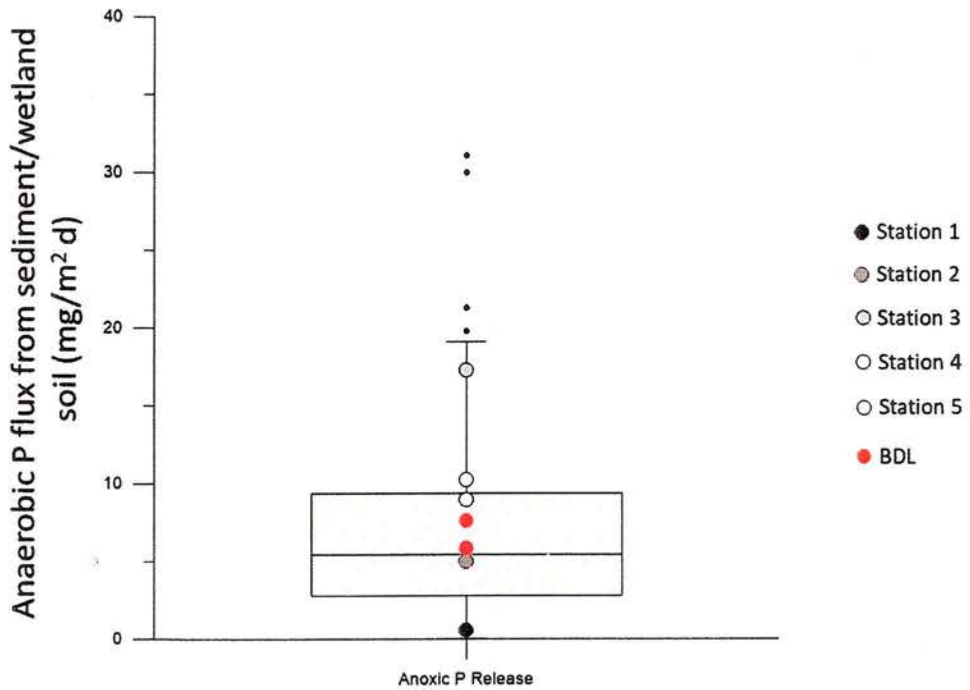


Fig. 6. Laboratory-derived anaerobic diffusive phosphorus flux compared to lake sediments in the Minnesota-Wisconsin region.

Fig. 7 Algal division biomass assemblage in Big Doctor Lake on 19 August 2021.

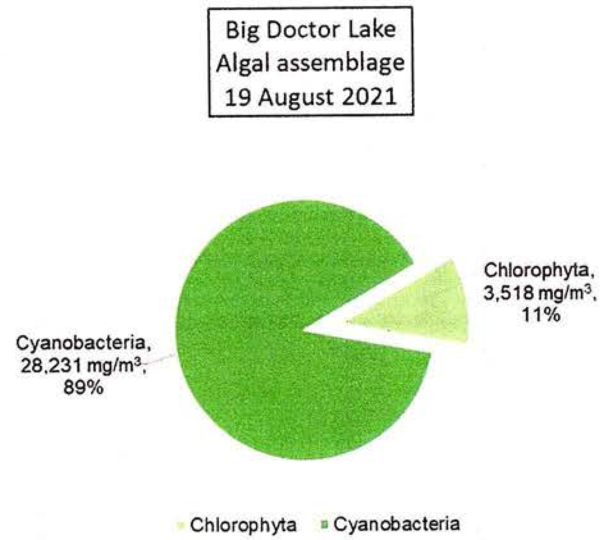


Fig. 8. Cyanobacteria biomass assemblage on 19 August 2021.