Long Lake, Wisconsin - Limnological response to alum treatment: 2019 interim report

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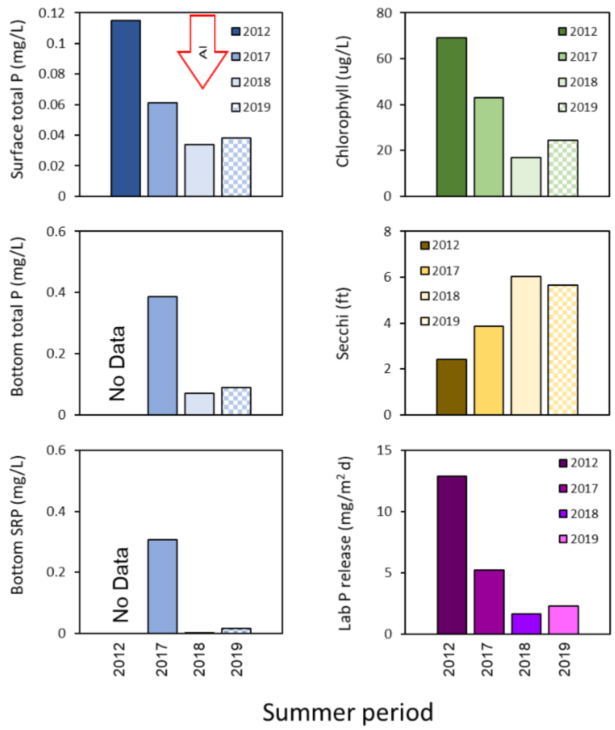
Executive summary



1. Approximately 1 year after a 60 g/m^2 alum treatment to sediments located within the 15 ft contour, mean (Jul-Sep) surface total P was 0.040 mg/L (34% reduction over 2017), mean bottom total P and SRP were 0.088 mg/L (77% reduction) and 0.018 mg/L (94% reduction), respectively, mean chlorophyll was 25 µg/L (43% reduction), and mean Secchi transparency was 5.7 ft (46% improvement) in 2019.
2. Seasonally, chlorophyll ranged between ~ 7 µg/L and 33 µg/L between July and August 2019. A peak concentration of 46 µg/L occurred in early September shortly after turnover. Secchi transparency declined to 3.9 ft during the bloom. The bloom was short-lived, and concentrations declined to 12 µg/L in early October. Excessive precipitation and shoreline erosion in late July and August may have subsidized these modest blooms since internal P loading was minor throughout the summer.
3. Laboratory-derived diffusive P flux from sediment under anaerobic conditions at station 30 was ~ 2.28 mg/m^2 d in 2019, representing a 56% decline over pretreatment means. In contrast, anaerobic diffusive P fluxes before the alum treatment were much higher at 12.9 mg/m^2 d in 2014 and 5.2 mg/m^2 d in June 2018.
4. Because lake limnological response and improvement continued to meet target expectations in 2019, the next alum treatment of ~ 25 g/m^2, scheduled for 2020, could either proceed or be delayed until 2021. The advantage of delaying the next application is potential exposure of the freshly deposited Al floc to hypolimnetic P. This exposure would stabilize $Al(OH)_3$ crystallization and increase overall longevity of internal P

loading control. A disadvantage of delaying Al application until 2021 is possible degradation of WQ conditions in 2020 and development of more severe cyanobacteria blooms. Stakeholder input and discussion is recommended to reach a sound decision.

Objective

Shallow Long Lake (273 ac surface area) has exhibited excessive summer cyanobacterial blooms and poor water quality (WQ) conditions (high phosphorus and chlorophyll concentrations and low water clarity) linked to internal phosphorus (P) recycling from sediments (James & Clemens 2017). A total aluminum sulfate (alum) dosage of ~ 105 g/m^2 split into lower concentrations and spread out over 2-3-year intervals was recommended to control internal P loading. The first alum dosage of 60 g/m^2 was applied to sediments located within the 15-ft depth contour of the lake on 11-13 June 2018.

Post-treatment monitoring of water and sediment chemistry was started in 2018 to document the trajectory of water quality improvement during rehabilitation as part of a comprehensive adaptive management program aimed toward making informed decisions regarding adjusting alum application and dosage to meet future water quality goals. Post-treatment monitoring included field and laboratory research to document changes in 1) lake limnological response variables (total P, soluble reactive P, chlorophyll, Secchi transparency), 2) diffusive P flux from sediment under anaerobic conditions for stations located within and outside the treatment area, and 3) binding of P by the alum floc. Overall, lake water quality was predicted to respond to internal phosphorus loading reduction with lower total phosphorus and chlorophyll concentrations throughout the summer of 2018 and 2019, lower bloom frequency of nuisance chlorophyll levels, and higher water transparency. The objectives of this interim report are to describe Long Lake limnological and sediment internal P loading response one year after the 2018 the alum application.

Methods

Lake monitoring

A station located in the central, deepest area of the lake was sampled biweekly between May and October 2019 (Fig. 1). An integrated sample was collected over the upper 2-m for analysis of total P, soluble reactive P (SRP), and chlorophyll. Additional discrete samples were collected at 1-m intervals from the lake surface to within 0.5 m of the sediment surface for analysis of these same variables. Total P samples were predigested with potassium persulfate according to APHA (2011). Total and soluble reactive P (i.e., P available for uptake by algae) were analyzed colorimetrically using the ascorbic acid method (APHA 2011). Samples for viable chlorophyll (i.e., a surrogate measure of algal biomass) were filtered onto glass fiber filters (Gelman A/E; 2.0 μ nominal pore size) and extracted in 90% acetone before fluorometric determination (EPA 445.0). Secchi transparency and in situ measurements (temperature, dissolved oxygen, pH, and conductivity) were collected on each date using a YSI 6600 sonde (Yellow Springs Instruments) that was calibrated against dissolved oxygen Winkler titrations (APHA 2011) and known buffer solutions. Vertical in situ profiles were collected at 0.5-m to 1-m intervals.

Sediment chemistry

Sediment sampling stations. Sediment cores were collected at 4 stations (10, 20, 30, and 40) in Long Lake in 2019 (Fig. 1). These station locations coincided with those visited in 2014 (James (2014). Station 20, 30, and 40 were located within the Al treatment zone (i.e., the 15-ft contour) while station 10 was located at a shallower depth outside the treatment zone.

Vertical variations. A sediment core was collected on late July 2019 at station 30 for determination of vertical profiles of various sediment characteristics and P fractions (see Analytical methods below). Sediment cores were sectioned at 1-cm intervals between 0 and 6 cm and at 2-cm intervals below the 10-cm depth for determination of moisture content, wet and dry bulk density, loss-on-ignition organic matter, loosely-bound P, iron-bound P, labile organic P, and aluminum-bound P.

Laboratory-derived diffusive phosphorus flux from sediments under anaerobic conditions. Anaerobic diffusive P fluxes were measured from intact sediment cores collected at all sediment sampling stations shown in Figure 1. Duplicate sediment cores were collected at stations 10, 20, and 40, while triplicate cores were collected at station 30 in late July 2019, to monitor alum treatment effectiveness after application. The sediment incubation systems were placed in a darkened environmental chamber and incubated at 20 C for up to 7 days. The incubation temperature was set to a standard temperature for all stations for comparative purposes. The oxidation-reduction environment in each system was controlled by gently bubbling nitrogen through an air stone placed just above the sediment surface to maintain anaerobic conditions.

Water samples for SRP were collected from the center of each system using a 60-cc syringe and filtered through a 0.45 µm membrane syringe filter (Nalge). The water volume removed from each system during sampling was replaced by addition of filtered lake water preadjusted to the proper oxidation-reduction condition. These volumes were accurately measured for determination of dilution effects. Rates of P release from the sediment (mg/m^2 d) were calculated as the linear change in mass in the overlying water divided by time (days) and the area (m^2) of the incubation core liner. Regression analysis was used to estimate rates over the linear portion of the data.

Analytical methods. A known volume of sediment was dried at 105 ^{o}C for determination of moisture content, wet and dry bulk density, and burned at 550 ^{o}C for determination of loss-on-ignition organic matter content (Avnimelech et al. 2001, Håkanson and Jansson 2002). Phosphorus fractionation was conducted according to Hieltjes and Lijklema (1980), Psenner and Puckso (1988), and Nürnberg (1988) for the determination of ammonium-chloride-extractable P (loosely-bound P), bicarbonate-dithionite-extractable P (i.e., iron-bound P), and sodium hydroxide-extractable P (i.e., aluminum-bound P). Additional sediment was dried to a constant weight, ground, and digested for analysis of total Al at the University of Minnesota, Research Analytical Laboratory.

The loosely-bound and iron-bound P fractions are readily mobilized at the sediment-water interface as a result of anaerobic conditions that lead to desorption of P from sediment and diffusion into the overlying water column (Mortimer 1971, Boström et al. 1982, Boström 1984, Nürnberg 1988). The sum of the loosely-bound and iron-bound P fraction represents redox-sensitive P (i.e., the P fraction that is active in P release under anaerobic and reducing conditions) and will be referred to as ***redox P***. Aluminum-bound P reflects P bound to the Al floc after aluminum sulfate application and its chemical transformation to aluminum hydroxide ($Al(OH)_3$).

Summary of Results

Lake limnological response

Annual precipitation measured at nearby Amery WI was higher than normal in 2019 versus the average 33 inches since 1980 (Fig. 2). In addition, severe storms and potentially high winds occurred near Long Lake in late July (~19 July), resulting in local precipitation at Amery and Luck WI in excess of 2 inches (Fig. 3). In addition, tornado touchdowns were reported in the vicinity on 19 July. Local precipitation (measured at Amery and Luck WI) in excess of 1 inch occurred in early and mid-May, mid- and late July, mid- and late August, and late September (Fig. 3). Monthly precipitation in August 2019 exceeded the long-term average by > 2 inches (Fig. 4).

Long Lake was stratified between mid-June and mid-August 2019 (Fig. 5). The water column completely mixed in late August 2019. Temperary stratification redeveloped in late September due to warmer air temperatures and relatively calm conditions. Hypolimnetic anoxia persisted between June and mid-August and redeveloped in late September in conjunction with stratified conditions. The maximum vertical extent of anoxia was ~ 5 m. The severe storms and winds in late July 2019 coincided with some mixing and deepening of the epilimnion and reintroduction of dissolved oxygen down to ~ 5 m. Bottom anoxia rapidly redeveloped thereafter and extended to the 5-m depth in mid-August.

In 2019, total P and SRP concentrations continued to remain very low in the bottom waters one year after the 2018 alum treatment despite hypolimnetic anoxia (Fig. 6). Bottom SRP concentrations were usually below detection limits throughout the summer but increased slightly to ~ 0.09 mg/L in early August 2019. Before alum treatment, they exceeded 0.80 mg/L as in early August 2017. In 2019 bottom total P ranged between only 0.03 and 0.17 mg/L. Similar to 2018, these modest increases in hypolimnetic total P concentration were confined to the > 5.5-m depth in Long Lake in 2019 (Fig. 7). Slight increases in hypolimnetic SRP were very short-lived, occurred only in early August, and were similarly confined to depths > 5.5 m (Fig. 8).

Surface total P concentrations ranged between 0.021 mg/L and 0.055 mg/L in 2019 (Fig. 9). The modest peak surface total P concentration of 0.054 mg/L in late May 2019 may have reflected Diatoms, a desirable algal species. Otherwise, surface total P concentrations were lowest in mid-July 2019 and gradually increased to 0.053 mg/L in late August 2019. These peak surface total P concentrations coincided with the late August mixing event and were associated with increases in chlorophyll (Fig. 10). Overall, seasonal dynamics in 2019 surface total P concentration were very similar to those observed in 2018, the year of alum application (Fig. 9), suggesting continued improved water quality. Vertically in the water column, total P concentrations were very low over the upper 4 m in July and increased slightly to a peak conjunction with the late August Fall turnover (Fig 7). In contrast, surface total P concentrations were much greater in the pretreatment summers of 2012 and 2017 (Fig. 9). Concentrations also increased over the summer period and were maximal in August of each pretreatment year, coinciding with peaks in chlorophyll in the surface waters.

Surface chlorophyll concentrations were usually low throughout the summer of 2019 after the alum application, ranging between 6.5 μg/L in July and 46-52 μg/L in early September (Fig. 10). Peak concentrations of surface chlorophyll in early September 2019 were short-lived and declined substantially by late September 2019. Surface chlorophyll was only 12 μg/L in early October 2019. In contrast, peak chlorophyll typically approached or exceeded 100 μg/L in August of 2012 and 2017 (Fig. 10).

Interestingly, peaks in chlorophyll were observed near the lake bottom in 2018 and 2019 (Fig. 7). These peaks may have reflected a profundal (i.e., deep water) algal population residing in and adapted to low light conditions. The possibility that these populations inoculated the surface waters to form blooms is uncertain. Although connections between profundal and surface chlorophyll seem to be weak, more research is needed to characterize this population.

Overall, Secchi transparency improvement continued in 2019 (Fig. 11). During the pretreatment summers of 2012 and 2017, Secchi transparency generally declined to < 1.0 m, and sometimes to < 0.5 m, during chlorophyll concentration peaks in August and September. In 2019, Secchi transparency exceeded 3 m in June. It declined to a minimum (1.2 m) in conjunction with the algal bloom in early September, then increased to nearly 3 m in early October 2019. shortly after the alum treatment and remained above 1 m in August and September.

Strong linear relationships existed between 2-m integrated total P and chlorophyll over the 2017-19 summer periods, indicating algal productivity was P-limited in Long Lake and responded to lower internal P loading in 2018-19 (Fig. 12). Secchi transparency was also inversely related to 2-m integrated chlorophyll concentrations over the 2-year summer period, suggesting that P-limited algal growth and reduction in biomass as a result of the 2018 alum application translated into greater summer water clarity in Long Lake.

A comparison of mean summer (July-September) limnological response variables before (i.e., 2012 and 2017) and after (i.e., 2018-19) alum treatment is shown in Figure 13. Although the pretreatment summer of 2012 provided a stark contrast to trends in 2018-19, the summer of 2017 was chosen for pre- versus post-treatment comparisons primarily because bottom total P and SRP data were available for the 2017 pretreatment year. Mean bottom concentrations of total P and SRP remained very low in 2019 as a result of alum treatment (Fig. 13), representing a 77% and 95% reduction over the 2017 pretreatment mean (Table 1). Mean summer surface total P and chlorophyll improvement continued in 2019 and mean concentrations were lower by 38% and 43%, respectively (Table 1). Mean Secchi transparency was 46% improved in 2019 versus the pretreatment mean in 2017 (Table 1).

Table 1. Summary of changes in lake water quality and lasboratory-derived phosphorus release from sediment in Long Lake after the initial alum treatment in June 2018. Overall goals after completion of the treatment schedule are shown in the last column.

		Variable	Historical (1993, 1996, 2000, 2014)	2012	2017	2018	2019	Percent improvement (2017 versus 2018)	Percent improvement (2017 versus 2019)	Goal after internal P loading control
Lake	Mean (Jul-Oct)	Mean surface TP (mg/L)	0.092	0.115	0.061	0.034	0.040	44% reduction	34% reduction	< 0.040
		Mean bottom TP (mg/L)			0.386	0.040	0.088	82% reduction	77% reduction	< 0.050
		Mean bottom SRP (mg/L)			0.307	0.003	0.018	99% reduction	94% reduction	< 0.050
		Mean chlorophyll (ug/L)	53.59	69.04	43.06	17.05	24.52	60% reduction	43% reduction	< 20
		Mean Secchi transparency (ft)	4.17	2.43	3.87	6.04	5.65	56% increase	46% increase	>10
Sediment	Station 30	Sediment diffusive P flux (mg/m^2 d)		12.9	5.24	1.68	2.28	68% reduction	56% reduction	< 1.5

Changes in anaerobic diffusive phosphorus flux and sediment chemistry

Laboratory-derived anaerobic diffusive P fluxes remained improved (i.e., lower than pretreatment fluxes) at stations located in the treatment area (i.e., station 20, 30, and 40) in late July 2019 (Fig. 14). However, rates were slightly elevated in 2019 (one year after alum application) compared to rates measured shortly after treatment in 2018 (Fig. 14). These patterns suggested that some of the sediment mobile P diffusing into the Al floc did not become bound to $Al(OH)_3$ and may eventually diffuse into the water column in the future. However, additional Al applications have been planned to address $Al(OH)_3$ crystallization and this future internal P load. Nevertheless, diffusive P flux measured in the laboratory was > 50% lower in 2019 compared to rates measured immediately before the June 2018 alum treatment.

The concentration of Al-bound P increased in the surface Al floc layer in 2019, indicating continued binding of P to in the Al floc layer (Fig. 15). The theoretical Al:P binding ratio in the Al floc layer was ~ 200:1 mass (180:1 molar) as a result. Ideally, a target Al:P binding ratio of 10:1 to 20:1 will be achieved in several years. Redox-P concentrations in the station 30 upper surface sediments remained lower in 2019 at < 0.40 mg/g compared to June 2018 pretreatment patterns (Fig. 15). The goal of the multiple Al applications over a period of 6+ years is to sequester and irreversibly bind redox-P to remove it from internal P loading pathways.

Summary and recommendations

The 2018 alum application continued to be successful in substantially reducing concentrations of total and soluble P in the hypolimnion of Long Lake in 2019. Mean summer (July-September) bottom total P (0.386 mg/L in 2017 versus 0.088 mg/L in 2019) and SRP (0.307 mg/L in 2017 versus 0.018 mg/L in 2019) declined by 77% and 94%, respectively, in 2019 compared to the pretreatment year 2017. Surface total P decreased from a mean 0.061 mg/L in 2017 to 0.040 mg/L in 2019, representing a 34% reduction in conjunction with alum treatment. The 2019 mean summer total P concentration was also at the WI state standard of 0.040 mg/L for shallow lakes (WIsCalm 2014). Mean summer chlorophyll was ~25 µg/L in 2019 compared to 43 µg/L in 2017, representing a 43% improvement. The 2019 mean concentration was higher than the mean in 2018 (17 µg/L), but still fell below the WIsCalm (2014) benchmark of 27 µg/L for aquatic habitat. Finally, mean summer Secchi transparency was ~ 5.7 ft in 2019, compared to 3.9 ft in 2017 and 2.4 ft in 2012. The 2019 mean represented a to 6.0 ft in 2018, a 56% improvement over the 2017 mean.

Currently, monitoring and adaptive management approaches are being used to assess water quality and sediment response in order to adjust future application timing and Al dosage, if necessary, to meet goals and expectations. A higher Al dose of 60 g/m^2 was applied during year 1 and lower doses of 25 g/m^2 and 20 g/m^2 are anticipated for 2020 and 2022, respectively.

Although 2019 summer mean limnological response variables declined slightly relative to 2018, means, they met target WQ goals established for Long Lake. Excessive precipitation and localized runoff in August 2019 may have played a role in subsidizing modest algal blooms. In addition, strong winds and wave activity likely caused shoreline erosion, potentially contributing unwanted P to the water column for algal uptake and growth. The 2020 alum application of 25 g/m^2 could be implemented as scheduled. Alternatively, it could be cancelled until 2021. The advantage of delaying the next application until 2021 would be increased exposure of the fresh Al floc to P, improving its longevity. The disadvantage of an application delay might be potential temporary degradation in WQ conditions if Al floc binding efficiency declines. Stakeholder discussion on the timing of the next alum application is recommended.

References

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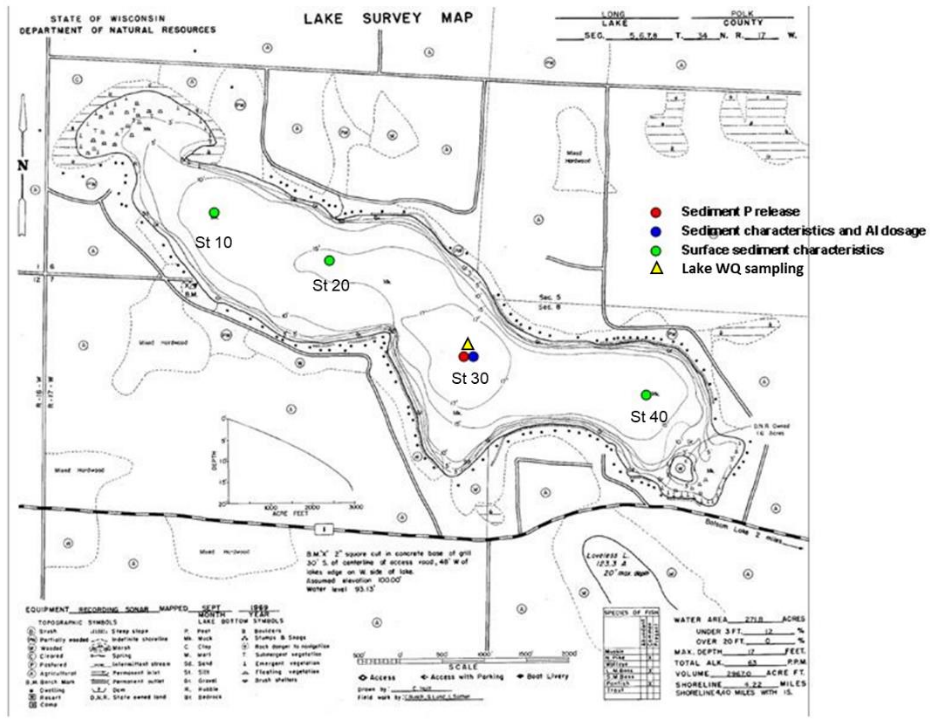

Figure 1. Sediment and water sampling stations.

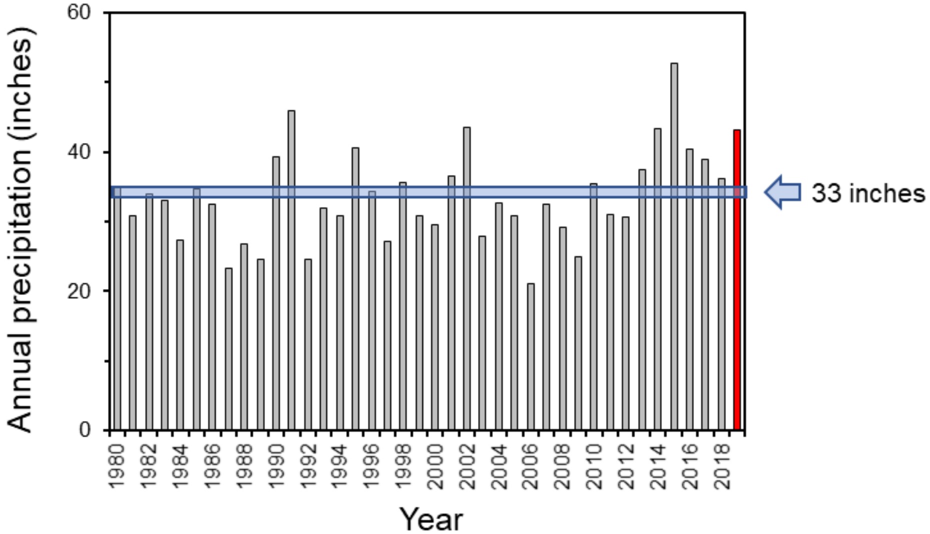

Figure 2. Variations in annual precipitation at Amery, WI. Blue horizontal line represents the average (33 inches). The year 2019 is highlighted in red.

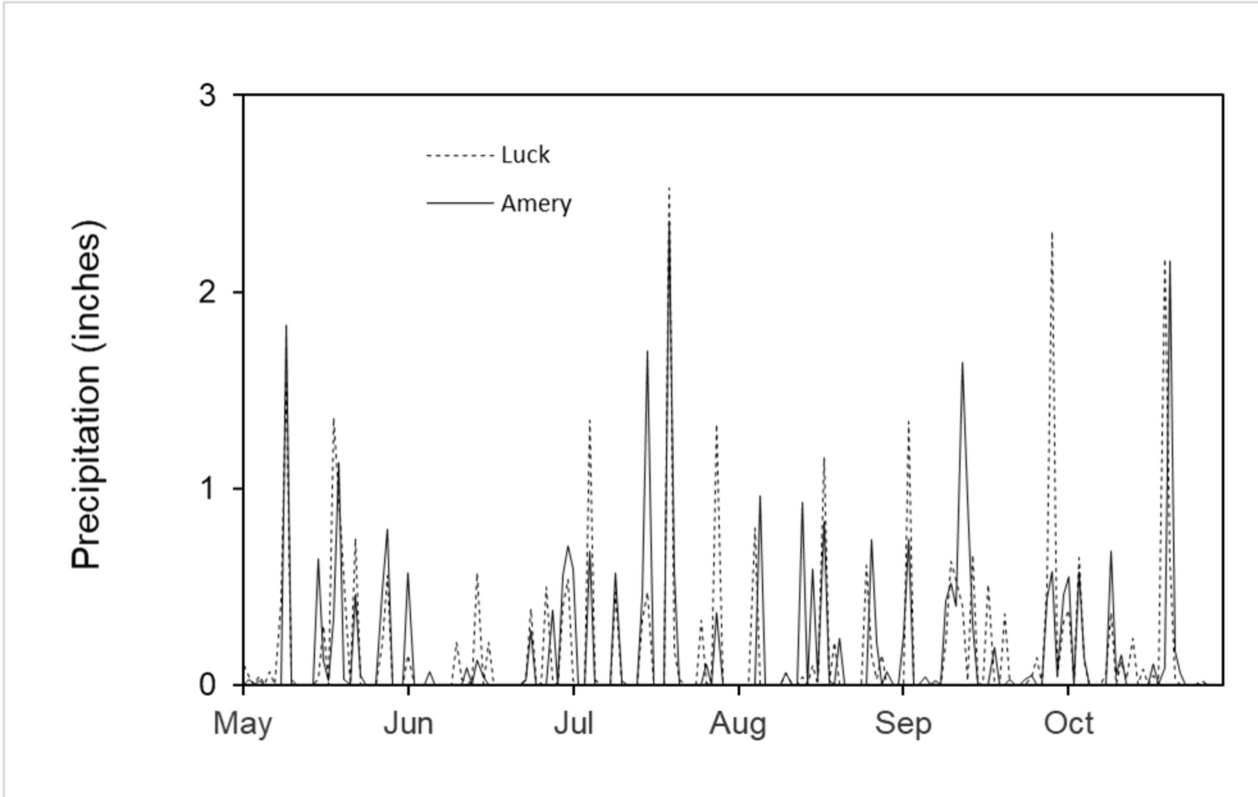

Figure 3. Variations in daily local precipitation measured at Amery and Luck WI in 2019.

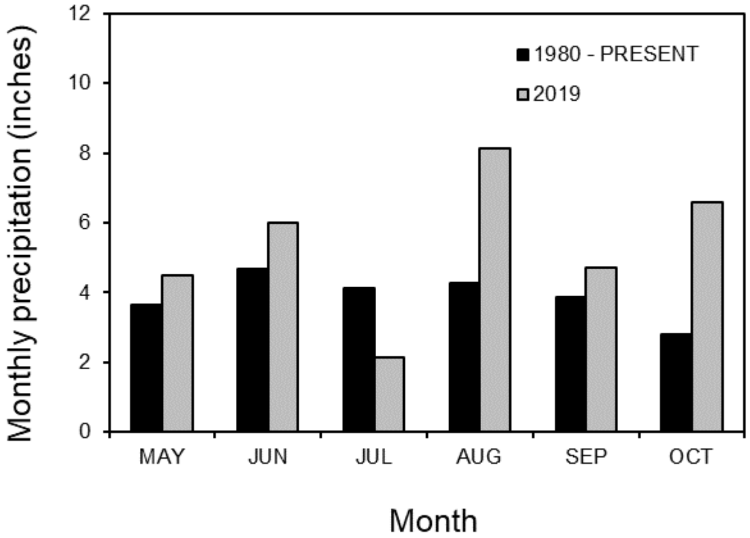

Figure 4. A comparison of average monthly precipitation (data from Amery WI).

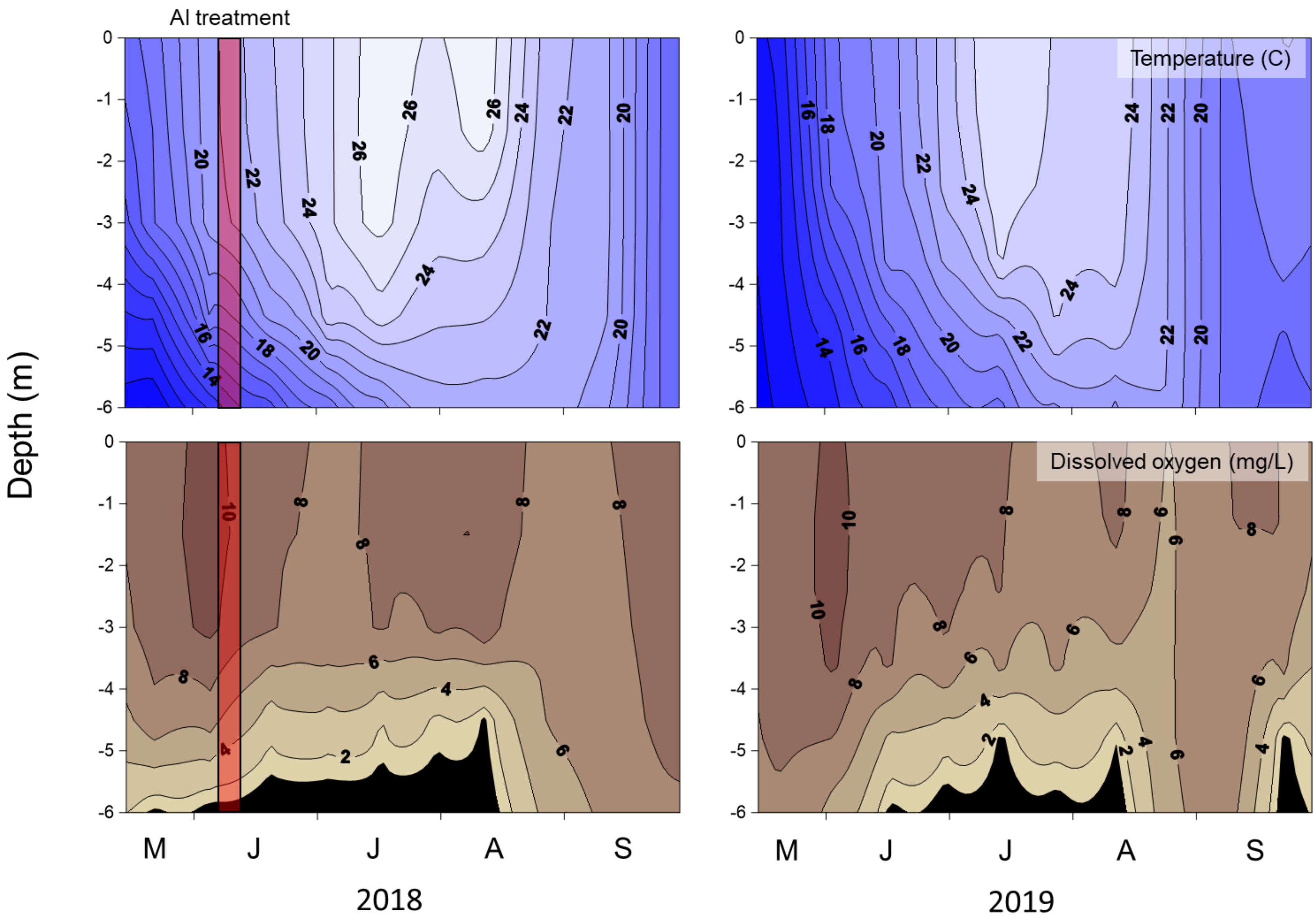

Figure 5. Seasonal and vertical variations in temperature (upper panel) and dissolved oxygen (lower panel) in 2018 and 2019.

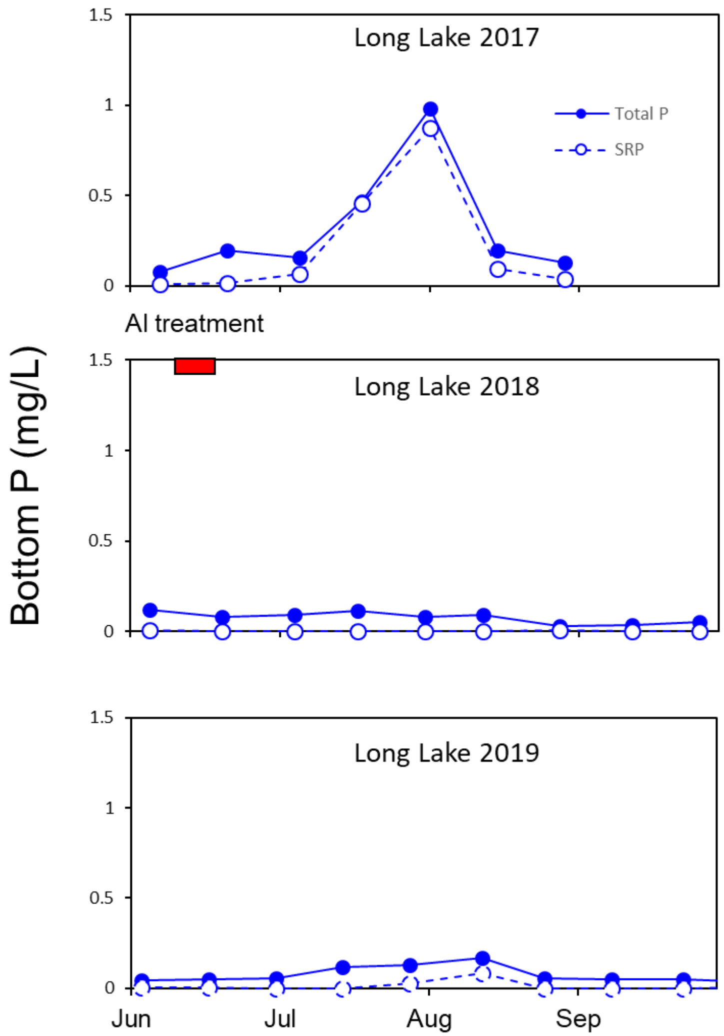

Figure 6. Seasonal variations in bottom (i.e., ~ 0.25 m above the sediment-water interface) total P, and bottom soluble reactive P (SRP) during a pretreatment year (2017), the year of alum treatment (2018), and 2019. Red horizontal bar denotes the period of alum treatment.

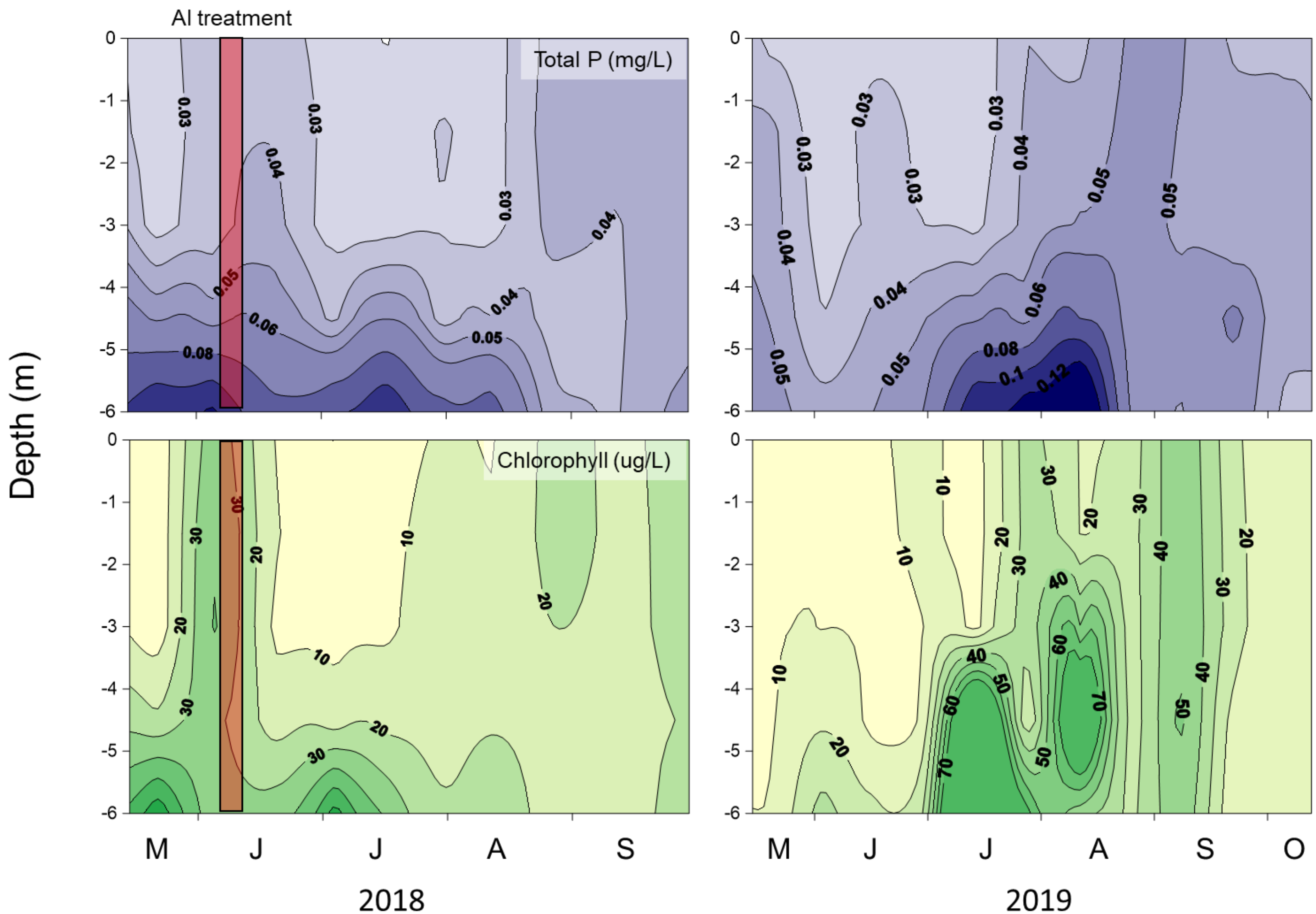

Figure 7. Seasonal and vertical variations in total phosphorus (P, upper panel) and chlorophyll (lower panel) in 2018 and 2019. Red horizontal bar denotes the period of alum treatment.

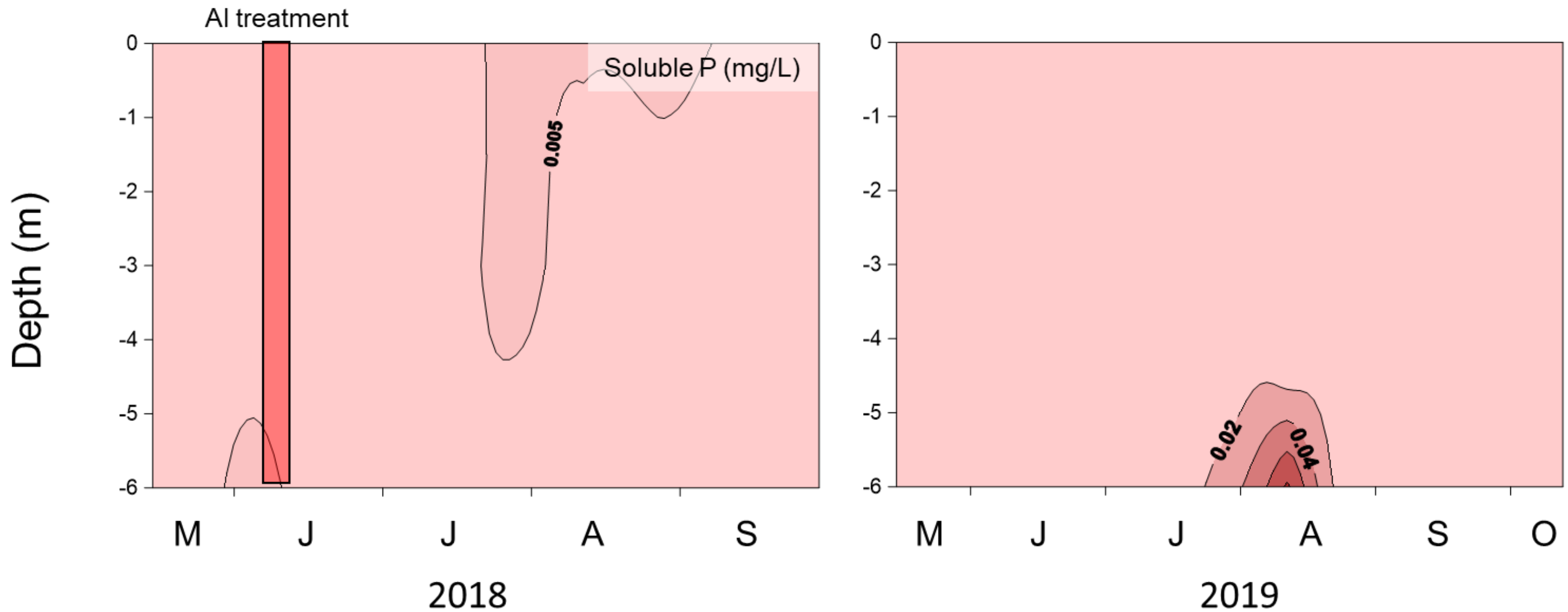

Figure 8. Seasonal and vertical variations in soluble phosphorus (SRP) in 2018 and 2019. Red horizontal bar denotes the period of alum treatment.

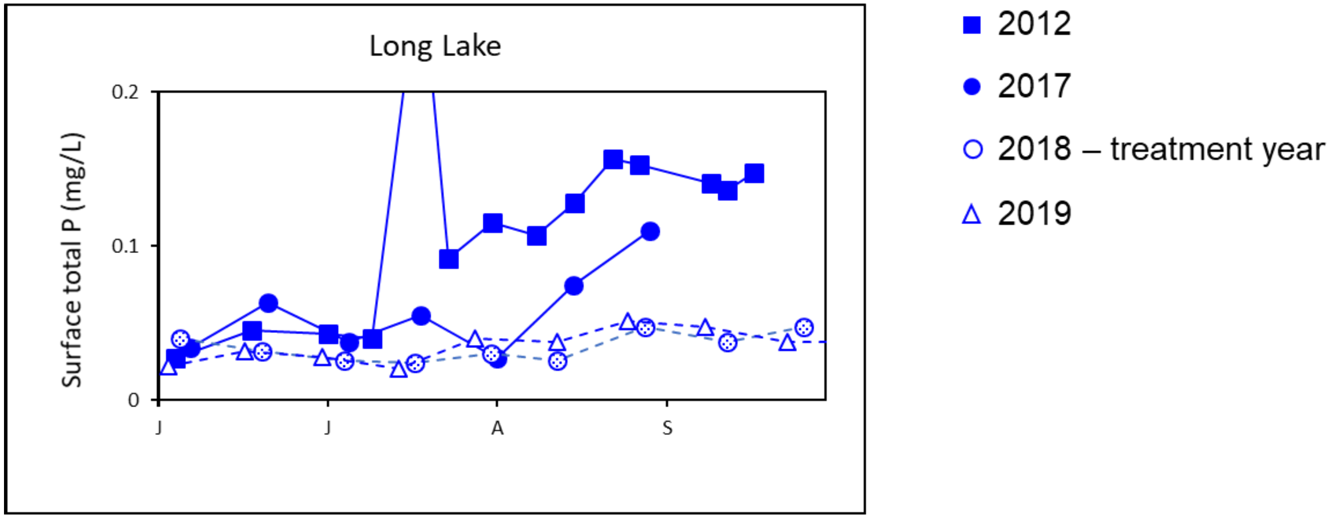

Figure 9. Seasonal variations in surface total P during pretreatment years (2012 and 2017), the year of alum treatment (2018), and 2019.

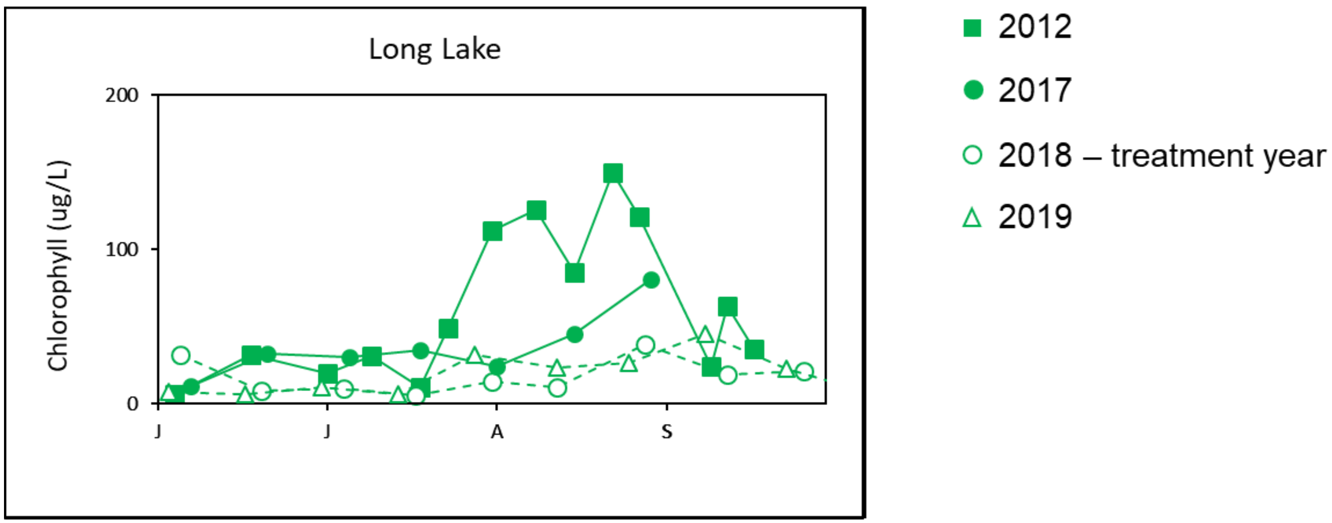

Figure 10. Seasonal variations in surface chlorophyll during pretreatment years (2012 and 2017), the year of alum treatment (2018), and 2019.

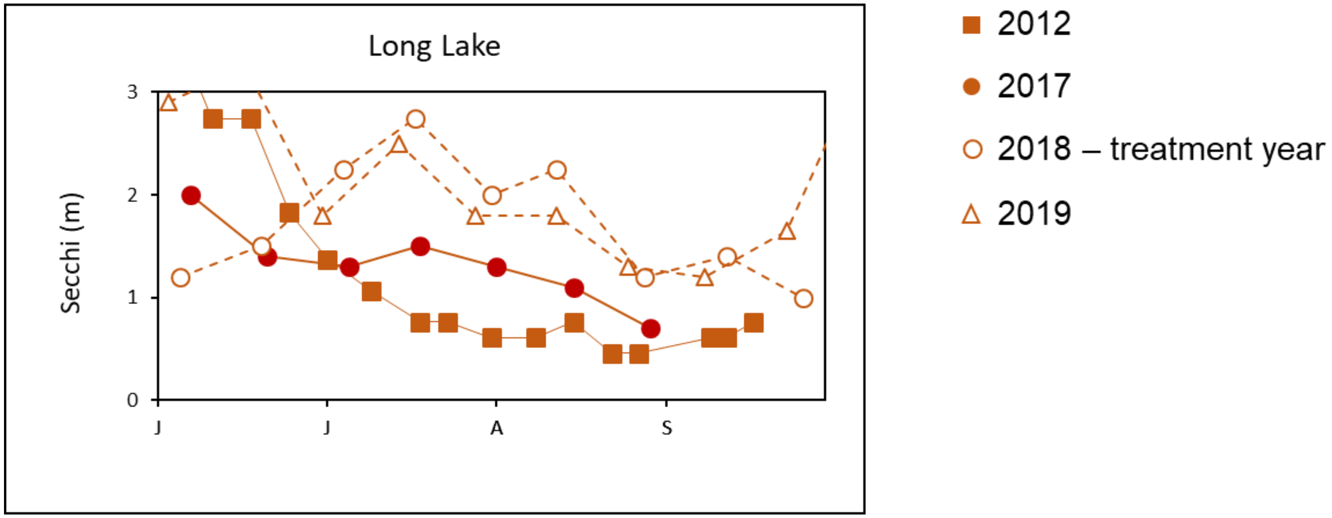

Figure 11. Seasonal variations in Secchi transparency during pretreatment years (2012 and 2017), the year of alum treatment (2018), and 2019.

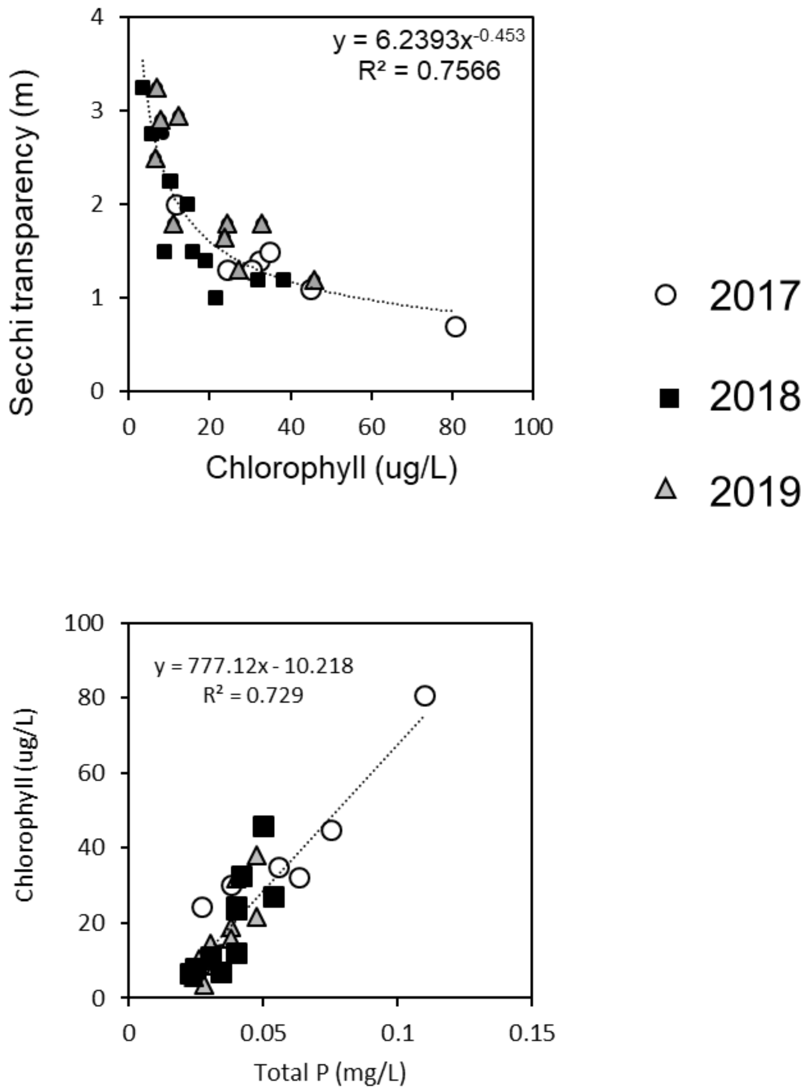

Figure 12. Relationships between Secchi transparency and chlorophyll (upper panel) and total phosphorus (P) versus chlorophyll (lower panel) during the summer 2017 (pretreatment) and 2018-19 (post-treatment).

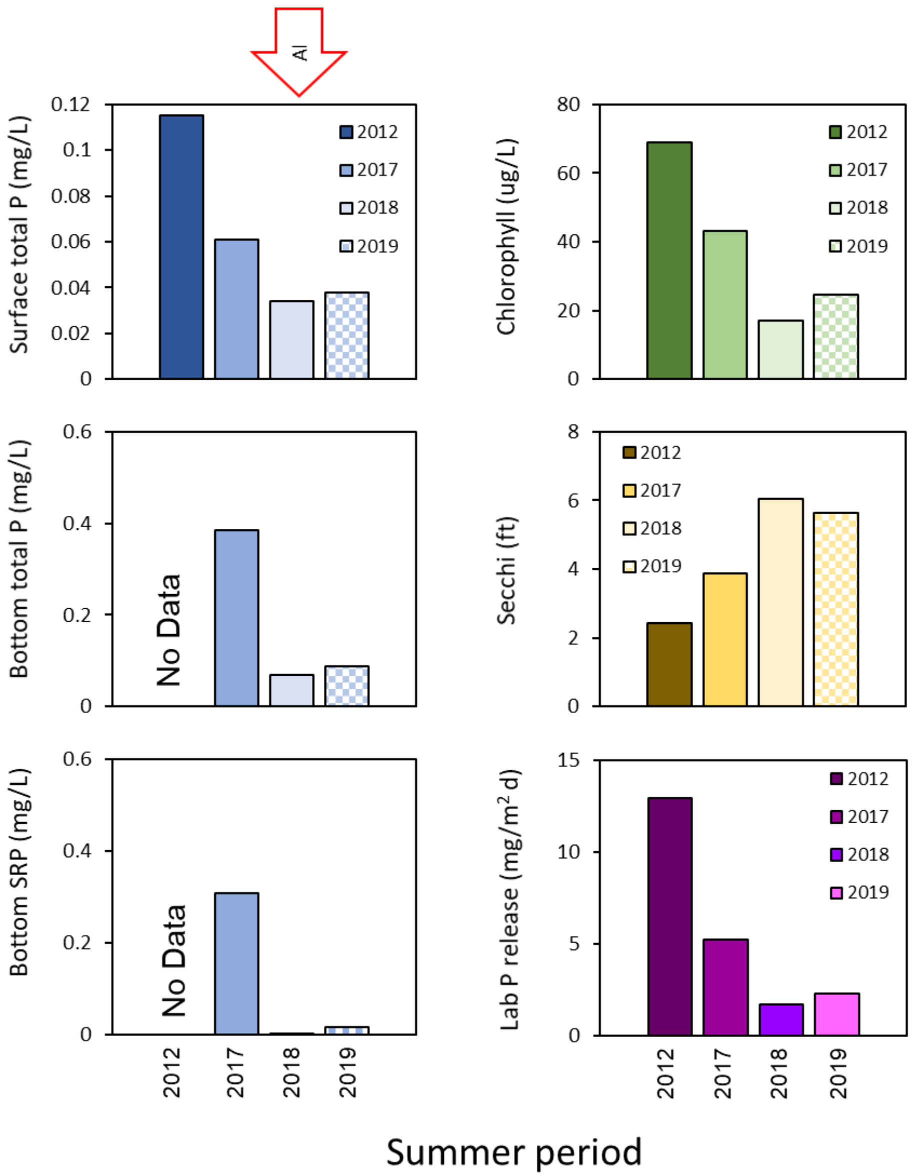

Figure 13. A comparison of mean summer (July-early October) summer concentrations of surface and bottom total phosphorus (P) and soluble reactive P (SRP), chlorophyll, Secchi transparency, and mean laboratory-derived diffusive P flux from sediment (station 30) during pretreatment (2012 and 2017) and post alum treatment (2018-19) years.

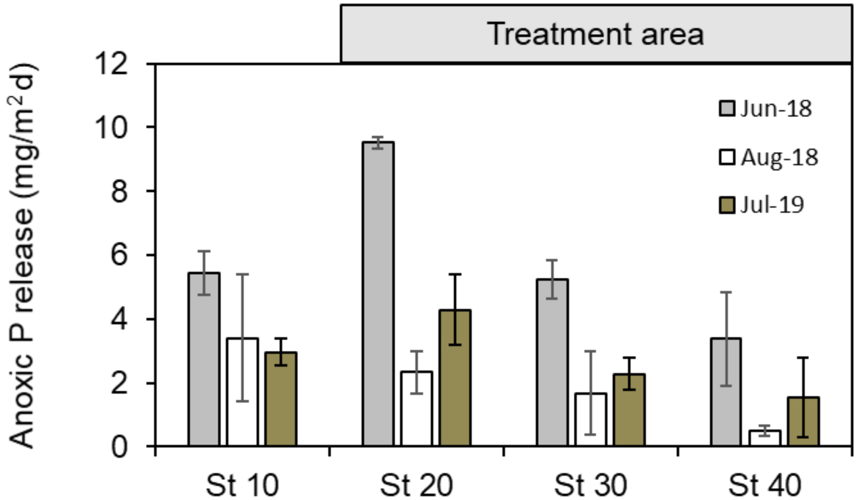

Figure 14. A comparison of mean anaerobic phosphorus (P) release rates for various stations in Long Lake. Horizontal lines represent ± 1 standard error.

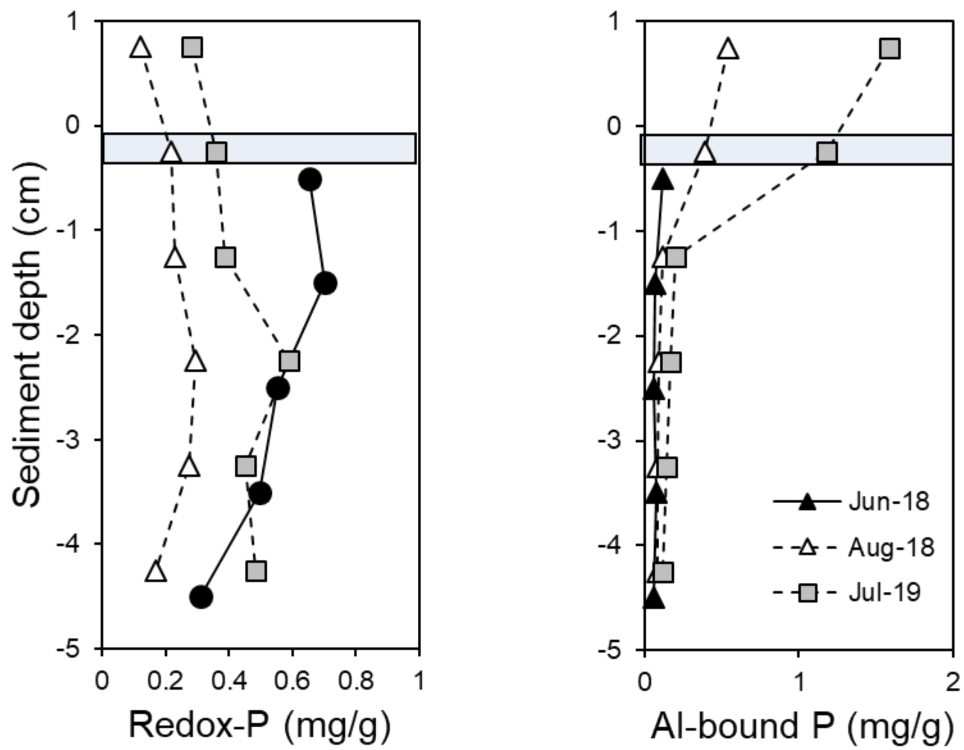

Figure 15. Vertical variations in sediment redox- (i.e., the sum of the loosely-bound P and iron-bound P sediment fractions) phosphorus (P) and aluminum (Al)-bound P concentrations for a sediment core collected from station 30 (Figure 1) in June and August 2018 and July 2019. The sediment profile in June represents pre-treatment conditions while August 2018 and July 2019 represents post-alum treatment conditions.