Curlyleaf Pondweed and Internal Phosphorus Loading Management to Improve Water Quality and Increase Underwater Photosynthetically-Active Radiation Habitat:

Native Macrophyte Community Response in Shallow Half Moon Lake, Wisconsin

Proposal

20 December, 2014

BACKGROUND, PROBLEM STATEMENT, AND OBJECTIVES:

Native submersed macrophytes play a critical structuring role in biological community dynamics and water quality conditions of aquatic ecosystems (Scheffer 1990, Moss et al. 1996, Jeppesen et al. 1998). These communities promote increased light penetration and water clarity by dampening wave shear stress and stabilizing flocculent sediment from resuspension (Barko and James 1998). They also provide refugia for young fish and habitat for a diversity of invertebrates, including microcrustacea that play an important role in grazer control of phytoplankton (Lauridsen et al. 1996, Burks et al. 2003). These interactions and feedbacks foster a fishery dominated by piscivores, increased grazing pressure on pelagic phytoplankton, and high transparency; features that are desirable both from an ecosystem perspective and for aesthetic and recreational reasons.

Eutrophication can negatively impact submersed macrophyte growth and community diversity by stimulating excessive algal productivity and decreasing underwater light radiation penetration (Scheffer et al. 1993, Scheffer et al. 1997, Gulati and van Donk 2002). In addition, invasion by rapidly growing non-native species such as curlyleaf pondweed (*Potamogeton crispus;* CLP) and Eurasian watermilfoil (*Myriophyllum* *spicatum*; EWM) can exacerbate declines in native submersed macrophyte populations in north-temperate climates by forming a dense surface canopy in early summer that attenuates light radiation (Boylen et al. 1999). These perturbations can lead to a regime shift from clear water submersed macrophyte dominance to turbid cyanobacterial dominance in shallow lakes (Scheffer et al. 1993). These alternate ecosystem states are relatively stable due to positive feedbacks, making shallow lake management difficult, particularly when the rehabilitation trajectory requires substantial reduction in P loading and invasive macrophyte control to reach light attenuation levels sufficient for native macrophyte re-establishement.

Half Moon Lake is a small (area = 0.5 km^2; volume = 12.9 x 10^5 m^3), shallow (mean depth = 1.6 m; maximum depth = 4 m) wind-sheltered urban oxbow lake located in Eau Claire, Wisconsin. Excessive CLP biomass has impaired recreational and aesthetic uses of the lake for decades and early efforts to control it via harvesting were not been successful. In addition, EWM pioneer populations were detected in the southwestern portion of the lake in 2007. Historically, the lake has been classified as eutrophic to hypereutrophic (Carlson Trophic State Index = 74), exhibiting high algal biomass (> 100 mg/m^3 viable chlorophyll primarily cyanobacteria) and frequent blooms, resulting in very poor water clarity and Secchi transparency depths of well less than 1 m that restrict the maximum depth and colonizable area for native submersed macrophyte growth.

Prior to 2009, the macrophyte community in Half Moon Lake was dominated by CLP, with biomass levels exceeding 100 g dry $mass/m^2$ in unharvested areas of the lake. In 2008, CLP was present at ~90% of the sites. Senescence of CLP in early summer and high rates of internal phosphorus (P) loading primarily from sediment, have accounted for > 60% of the P inputs to the lake and contributed to blooms of undesirable cyanobacterial species (*Anabaena sp., Aphanizomenon sp., Aphanocapsa sp., Cylindrospermopsis raciborskii Microcystis aeruginosa, Planktolyngbia sp.*). These impacts have resulted in high photosynthestically-active radiation (PAR) attenuation (i.e. low water clarity) and impairment to the native macrophyte community in the lake (James et al. 2002; James 2010a).

The overall management goal for Half Moon Lake was to improve water quality and PAR penetration to promote the re-establishment of healthy native submersed macrophyte communities. Strategies to accomplish this goal have been twofold: 1) reduce infestations of canopy-forming CLP and EWM and 2) control internal P loading from sediment to limit cyanobacterial productivity and improve underwater PAR condition. First, motor boat activity was restricted on the lake to reduce P resuspension and internal loading. Second, as part of a previous Aquatic Invasive Species Control Grant (AIS), shading and P recycling caused by CLP decomposition were addressed by annual early spring herbicide (Endothall) treatments during the years 2009-2014 to selectively target this species with minimal impact to native plants (Netherland et al. 2000, Poovey et al. 2002, Skogerboe et al. 2008; Johnson et al. 2012). Similar selective control of EWM was also accomplished in 2009 under the grant. Finally, P release from sediments was managed in June, 2011, via application of buffered alum-aluminate, to drive algal productivity toward P-limited growth and increase light penetration for native macrophyte communities.

To assess and evaluate water quality and native macrophyte community response to these management measures, extensive limnological monitoring was conducted between 2008 and 2014. In situ (i.e., temperature, dissolved oxygen, pH, and conductivity) measurements, PAR penetration, P, and chlorophyll were monitored at six stations in the lake between 2008 and 2014 to assess improvement in underwater light habitat for comparison with the native macrophyte community response (Figure 1). Urban watershed P inputs from selected storm sewers draining portions of the City of Eau Claire were monitored to assess the impacts of BMPs on P loading to the lake. Sediment cores were collected at 2 stations (station 10 and 30) annually between 2008 and 2014 to evaluate the impacts of the Al treatment on sediment mobile P pools (i.e., P fractions related to recycling and diffusive flux) and rates of diffusive P flux from anaerobic sediment. Finally, macrophyte species biomass and frequency of occurrence were monitored in June and August of each year using point-intercept sampling techniques to evaluate the effectiveness of the herbicide treatments in controlling CLP and EWM and to quantify native macrophyte community response to changes in water quality and management of the targeted invasive species. Description of methods and results of these monitoring efforts can be found in James (2014).

Al treatment had an immediate impact on mean summer limnological conditions in 2011-13 (Figure 2). The period July-September was chosen because it represented the typical period of maximum nuisance algal blooms and did not include the late May-June clearing phase caused by zooplankton grazing. During the first 3 years (2011-2013) post-treatment, mean total P and chlorophyll decreased by > 60% over pretreatment means, while Secchi transparency increased by > 50% (Figure 2). The mean PAR attenuation coefficient (K_d) declined from > 2 m^{-1} prior to Al treatment to ~1 m^{-1} post-treatment (Figure 3). Estimated maximum colonizable depth for stem-forming and rosette-forming submersed macrophyte communities (Middelboe and Markager 1997) increased to ~ 2.8 m and 1.2 m, respectively, resulting in an increase in area for potential growth to 90% and 50% of the lake area, respectively.

The spring of 2013 represented the fifth consecutive year of endothall treatment to reduce the CLP turion bank in the sediment. April surveys quantifying germinated CLP turions suggested that both frequency and numbers were down tremendously compared to pre-treatment conditions (Figure 4). Based on these findings, a sixth consecutive early spring herbicide treatment was not conducted in 2014. Surprisingly, CLP growth rebounded after herbicide treatment cessation and it was present at over 40% of the point-intercept locations by June, 2014 (Figure 5). June CLP biomass also rebounded substantially from other herbicide treatment years to a lakewide average of 20 g/m^2 (Figure 5). CLP occurrence and biomass declined, predicatively, in August due to mid-summer dieback. These trends suggested that the turion bank in Half Moon Lake is still viable and has not been substantially depleted after 5 years of treatment. If left untreated, CLP will likely continue to propagate and compete with the native submersed macrophyte community for PAR and habitat. Johnson et al. (2012), similarly, found reoccurrence of CLP even after 5 to 7 consecutive years of treatment. Indeed, CLP management may require years to control and deplete the turion bank in the sediment. Because CLP potentially produces thousands of turions per square meter (Woolf and Madsen 2003) and has dominated the macrophyte community in Half Moon Lake for decades, the turion bank in the sediment is probably large and can be viable for many years (Johnson et al. 2012). More information is needed on the impacts of long-term selective endothall treatment (i.e., >5 y to 10 y) on turion bank reduction in order to develop sound and effective management approaches for CLP control.

Discrepancies in CLP frequency between April and June, 2014 (i.e., the year of treatment cessation), suggested that significant turion germination occurred in May that was not captured by the late April sampling event. Current conceptual models suggest that turion germination occurs primarily during fall and under winter ice cover (Woolf and Madsen 2003; Figure 6). However, CLP population dynamics in Half Moon Lake did not appear to fit this general germination pattern. Rather, germination timing may be much more complex and regulated by factors such as snow and ice cover, winter PAR penetration, the timing of ice-out, and spring temperature patterns. Indeed, significant turion germination may occur in May regardless of annual variations in winter and early spring conditions and needs to be considered in CLP control.

Quantifying turion germination has become more problematic in light of these recent findings. Since it appears that turion germination is very significant in late April and early May in Half Moon Lake, seasonal germination patterns should be examined in order to better understand and predict the optimal timing for treatment. A caveat has been that treatments have generally occurred in late April in conjunction with a water temperature range of 15 to 18 C to maximize selective control of CLP. While April point-intercept sampling has generally shown a declining germination trend over time, this approach has failed to account for projected later germination. Additionally, herbicide applications have probably inadvertently controlled these later germinations, as indicated by frequency and biomass trends in June. Germinated turion assessment should probably continue although perhaps in later spring (i.e., early May) to more accurately quantify frequency of germinated turions. Ultimately, depletion of the turion bank can best be assessed by evaluating frequency and biomass during a nontreatment year.

Native submersed macrophyte occurrence and biomass rebounded from the initial early spring endothall and 2,4-D treatment of 2009. Prior to herbicide management (2008), the native submersed macrophyte community was overwhelmingly dominated by coontail (Figure 7). Treatment with 2,4-D to eradicate EWM in 2009 also severely impacted coontail, resulting in nearly complete loss of the native community in the lake (Figure 7 - see 2009). However, both native macrophyte occurrence and biomass rebounded substantially in conjunction with reduced K_d and increased PAR penetration after the Al treatment (Figure 7). The community has largely been dominated by elodea post Al treatment; some other native species have also been observed (Stargrass, wild celery, coontail).

Although native biomass frequency of occurrence was greater than 80% in both June and August 2014, biomass declined substantially in August over previous years (Figure 7). Biomass declines were greatest in the deeper west arm of the lake (not shown). This pattern was probably related, in large part, to lower Secchi transparency and higher k_d in 2014 due to high chlorophyll concentrations. This pattern was largely related to the re-emergence of *Cylindrospermopsis sp* (see below). As a result, the maximum inhabitable depth for stem- and rosette-forming submersed macrophytes declined in 2014 (Figure 3). Underwater PAR habitat impairment was greatest in the west arm (not shown), reflecting lower native biomass in 2014 compared to other years (Figure 7). Thus, the native submersed macrophyte community appeared to be very susceptible to degradation of underwater light habitat, driven algal blooms in 2014.

Detailed inspection of sediment moisture content and wet bulk density vertical profile characteristics suggested that the Al floc was largely positioned on top of the original sediment interface and had not penetrated appreciably into the sediments three years post-treatment (Figures 8 and 9). For cores collected in the western arm of the lake (i.e., station 10) in 2014, sediment wet bulk density was very low, while moisture content approached 95%, in the upper 3-cm layer, contrasting substantially with original surface sediment physical-textural characteristics before the Al treatment (Figure 8 upper panels). East arm sediments exhibited a similar pattern of lower wet bulk density and higher moisture content over the upper 2-cm layer compared to 2010 conditions (Figure 10 upper panels). Differences in the apparent thickness of these layers were attributed to application of 2X more Al (i.e., 150 g/m^2) in the west versus the east (i.e., 75 g/m^2) arm of the lake. Overall, wet bulk density in the upper 1 cm at both stations declined substantially from ~ 1.04 g/cm^3 in 2010 a mean 1.02 g/cm^3 (±0.003 SE; n = 8), while moisture content increased from 91.5% to a mean 95.6% (± 0.7 SE; n = 8), after Al treatment (Figure 10). These patterns suggested that the bulk density of the consolidated Al floc layer was, in fact, much lower than that of the original sediment surface layer 3 years post treatment.

Prior to Al treatment in 2010, iron-bound P concentrations were greatest in the upper 5-cm sediment layer at both stations (Figures 8 and 9). Concentrations within this layer were greatest in west arm, exceeding 1.5 mg/g (Figure 8 lower left panel), versus much lower iron-bound P concentrations in east arm sediments (0.32 mg/g; Figure 10 lower left panel). In 2014, iron-bound P concentrations in west arm sediments were relatively low in the upper 3 cm Al layer and increased substantially at depths below the original 2010 sediment interface, reflecting 2010 concentrations. Iron-bound P was also lower within the upper 2-cm of the original sediment interface in 2014, relative to 2010 concentrations, indicating probable sequestration of some of the iron-bound P immediately below the original sediment interface. In contrast, aluminum-bound P concentrations were maximal in the upper 3-cm layer in conjunction with high Al concentrations (Figure 8 lower right panel). Similar vertical patterns in iron-bound and aluminum-bound P were observed in east arm sediments (Figure 10 lower panels).

Annual trends in surface sediment iron-bound P (i.e., upper 1-cm sediment layer) suggested that concentrations have been increasing over time since Al treatment primarily for west arm sediment, but not east arm sediments (Figure 11). Iron-bound P in west arm sediments was very low in the Al floc layer shortly after treatment in 2011, but steadily increased over time to 0.33 mg/g in 2014 (Figure 11). This buildup has been approximately linear over time (Figure 12) and is related to upward diffusion of porewater iron and P into the Al floc from sediments located below the original interface. Between-arm differences in iron-bound P concentrations in the upper 1-cm layer in 2014 were probably related to concentration differences in the original sediment surface layer prior to Al application (Figure 8 and 9). The establishment of steep vertical P (and Fe) gradients between original surface sediment and the overlying Al floc in west arm sediments probably drove enhanced upward P diffusion, relative to the east arm where vertical concentration gradients were much lower. While some of this upward P flux was probably sequestered by the Al floc, a portion clearly became re-adsorbed to Fe in west arm sediments with the potential for diffusive flux into the overlying water column under reducing conditions.

Laboratory-derived rates of P release from sediment for west arm sediments under anaerobic conditions increased from undetectable during the first 2 years post-treatment to a mean 1.0 mg/m^2 d (± 0.3 SE; n = 5) by 2014 (Figure 13 upper panel). While the 2014 anaerobic P release rate represented an ~ 90% decrease from the pretreatment mean of 11.8 mg/m^2 d (±1.7 SE) it, nevertheless, would be a potential source of P to the water column under anoxic conditions at the sediment-water interface. For east arm sediments, the anaerobic P release rate was essentially undetectable during the 2011-13 post-treatment years and very low at 0.18 mg/m^2 d (± 0.05 SE) in 2014 (Figure 13 lower panel). Despite slightly elevated laboratory-derived anaerobic P release rates in 2014, anoxic conditions were detected above the sediment interface at west arm stations 10 and 20 in only late June and mid-August, 2014. East arm stations 30 and 40 exhibited bottom-water anoxia primarily in late August, 2014, only. Thus, potentially reducing conditions conducive for anaerobic P release from sediments were very limited in 2014, suggesting a low potential for internal P loading.

Even though Al treatment has been effective in controlling diffusive P flux under anaerobic conditions, the re-emergence of *Cylindrospermopsis sp.* in 2014 may still be related to changes in iron-bound and porewater P concentration in the sediment after Al treatment. While preliminary, excystment of resting spores residing in the sediment, luxury uptake of P in the sediment porewater, and inoculation of the water column currently provides the most plausible explanation for the high 2014 mean summer K_d and chlorophyll concentration that impacted the native submersed macrophyte community. The current working hypothesis suggests that porewater Fe and P have slowly diffused from deeper sediment layers into the surface Al floc layer over time. While the Al floc may be sequestering some of this P, another portion is now coupled with Fe and can diffuse into the overlying water column under reducing conditions. More importantly, this P is directly available for uptake by *Cylindrospermopsis* spores residing in the sediment of Half Moon Lake. Even though an adequate Al dosage was applied to inactivate the high sediment mobile P concentrations, the Al floc did not sink into the upper sediment due to higher bulk density characteristics of the original sediment interface. Thus, Al binding efficiency has probably decreased over time (de Vicente et al. 2008a and b), allowing for upward P diffusion through the Al floc and re-adsorption onto Fe~(OOH) (Lewendowski et al. 2003). This outcome is most apparent in west arm sediments, where mobile P concentrations were extraordinarily high in the original surface sediment layer. Although similar diffusional processes are occurring in east arm sediments, much less mobile P has accumulated in the surface Al floc layer due to lower mobile P concentrations in the original surface sediment layer.

Interestingly, sediment P availability for algal bloom development is most likely via direct assimilation by *Cylindrospermopsis* spores residing in the sediment rather than indirect diffusive P flux into the overlying water column. Reducing conditions that would drive anaerobic P flux were rarely observed above the sediment interface in 2014. Nevertheless, sediment mobile P that has accumulated in the Al floc layer is apparently stimulating *Cylindrospermopsis* blooms through directly availability in the sediment.

Although P mass balance was difficult to quantify, comparisons provided some important insights into possible causes for the extensive algal blooms in the lake in 2014. Although P inputs from the Owen Park pumps were ~ 24 kg during the 41 day period, net P mass accumulation in the south arm of the lake, which receives these inputs, was only 3 kg (Table 1). This P input was probably largely short-circuiting into the Becca Brook outlet, located in the embayment south of the storm sewer, rather than moving into the main arm. Because bottom water anoxia was not detected in the south arm of the lake, I estimated zero anaerobic P release from sediments in this region. Lateral transport of P from the south to the east arm could not be estimated.

Net P mass accumulation was greatest in the west arm at ~20 kg versus only 7 kg in the east arm (Table 8). Although the anaerobic P release rate was ~ 1 mg/m^2 d for sediments located in the west arm, bottom water anoxic conditions were documented only in late June in both east and west arms. Assuming an anoxic period of 7 days, west and east arm sediments could have released ~ 2 and 0.2 kg, respectively, during the 41-day period. Thus, potential anaerobic P release from sediments would not explain the much greater net P mass accumulation rates in the water column of the west and east basins.

Instead, evidence points to direct assimilation of sediment P by *Cylindrospermopsis* resting spores residing in the sediment and inoculation of the water column. Similarly, an algal bloom composed primarily of *Cylindrospermopsis* developed in the west arm of the lake in August, 2011, shortly after the Al application. This pattern was attributable to a couple of factors that resulted in its unexpected success. This species probably assimilated luxury P (i.e., P stored in cells as polyphosphates well in excess of growth requirements) from the sediment well before the Al treatment occurred. These reserves were used in bloom development after excystment, providing the population with a competitive advantage over other species that did not scavenge P before the Al treatment.

Bloom suppression in 2012 and 2013 was likely linked to low mobile P concentrations (i.e., iron-bound P) in the sediment as well as negligible anaerobic diffusive P flux from sediment into the overlying water column due to the Al treatment. *Cylindrospermopsis* sp were detected in August and September of both years (not shown), but in low concentration. Higher mobile P concentrations in sediment in 2014, particularly in west arm sediments, probably played a role in subsidizing direct P uptake and assimilation by

Cylindrospermopsis. Thus, although sediment mobile P concentrations and anaerobic P release rates were still low in 2014 relative to pretreatment conditions, direct uptake of this new P source, that diffused upward from deeper sediment layers, by *Cylindrospermopsis* spores, excystment, and water column inoculation would provide a plausible mechanism explaining chlorophyll bloom formation in 2014. Additional, lower dose, Al treatments will be probably be needed in the west basin to sequester P that has diffused into the Al floc in order to limit and suppress future *Cylindrospermopsis* growth.

Combining selective endothall control of invasive CLP with Al treatments to improve underwater PAR habitat for native macrophyte growth represents a unique and innovative management approach to shallow lake restoration. A comprehensive and extensive program to monitor macrophyte and lake water quality response has provided important information that can be used to develop and improve management plans for lakes with similar problems throughout the State of Wisconsin. However, more information is needed to assess the effectiveness and longevity of this multifaceted management program on the suppression of targeted CLP, P-limition of cyanobacterial blooms, and vigor of the native macrophyte community. Specifically,

1. Since the turion bank has not been substantially depleted after 5 years of selective control, CLP growth and dominance is likely to rebound and impact efforts to improve water quality and native submersed macrophyte re-establishment in Half Moon Lake. The number of annual treatments required to suppress CLP growth and propagation is currently not known. Control efforts published in the literature have extended over 3 to 5 years; one study reported that CLP growth rebounded after 7 treatment years (Johnson et al. 2012). However, this information is critical to the State of Wisconsin in order to develop sound and effective management plans and strategies to control CLP.

2. Another unknown is the resiliency of CLP after long-term control. This information will be important in setting realistic goals and expectations. CLP frequency of occurrence was reduced from nearly 90% to 40% after five years of

3. What is the longer-term response of the native macrophyte community to CLP and P management? Previous research on Half Mon Lake has suggested that native macrophyte community re-establishment and growth is very susceptible to underwater PAR habitat. More information is needed on native macrophyte community response to long-term CLP management and sediment P control in order to better predict forward and reverse variable thresholds and hysteretic trajectories between turbid cyanobacteria and clear-water macrophyte stable equilibria in shallow lakes.

The objectives of this proposal are several-fold.

1. Continue the early spring endothall application for 5 more years (2016-2019) to further reduce the CLP turion bank in Half Moon Lake,

2. Monitor and quantify CLP biomass and frequency of occurrence in May (prior to treatment), June (peak biomass period), and August (new germination),

3. Monitor and quantify native submersed macrophyte biomass response in June and August,

4. Monitor water quality, PAR attenuation, and the predicted maximum colonizable depth for stem- and rosette-forming submersed macrophytes for comparision with native biomass and occurrence,

5. Evaluate the effectiveness of the 2011 Al treatment on concentrations and availability of mobile sediment phosphorus. Was the alum dosage effective in inactivating the iron-bound phosphorus in the upper layer of sediment? Was it sufficient enough to inactivate additional phosphorus migrating upward from deeper sediment?

6. Evaluate the position of the Al floc over time. Is it sinking?

7. Examine the binding strength of the alum floc after application. What is the actual Al:P binding ratio in the sediment (i.e., the mass of phosphorus adsorbed by 1 unit of Al) compared to predicted ratios based on prior laboratory assays?

8. Assess changes in the rate of diffusive phosphorus flux from sediment as a result of alum treatment. Was treatment sufficient enough to inhibit diffusive phosphorus flux over a 4 year period or longer?

9. Examine the effectiveness of the alum treatment in relation to other management strategies on improving water quality and the aquatic macrophyte community.

APPROACH:

Task 1. Estimation of treatment areas, time of treatment, herbicide requirements, dosage, and cost for 2016 through 2019

Endothall (as Aquathol K) will be applied in early spring to the entire lake (including Braun's Bay) at a rate of 0.8 mg/L active ingredient (ai) to control CLP. The application should be conducted when water temperatures are > 12 ^{o}C with a maximum temperature of 17 ^{o}C. Existing data is unclear on endothall efficacy at temperatures lower than 12. Existing small scale data has indicated that long term efficacy on CLP may be reduced as water temperatures exceed 20 ^{o}C, and susceptible natives may begin to grow at these warmer water temperatures.

Pretreatment CLP abundance surveys will be conducted to determine areas in the lake that require treatment. Time of treatment will be based on water temperature in the spring to ensure selectivity for target plants with minimal impact to native species. Actual herbicide applications will be contracted through the grant applicant or WI Department of Natural Resources (Table 2). The treatment cost is projected to be $50,000 per year.

Task 2. Limnological Monitoring (2016-2019)

Six stations will be established in the lake for limnological monitoring purposes (Figure 1). Monitoring will be conducted biweekly at each station between May and September (8 to 9 sampling events). Water temperature, dissolved oxygen, conductivity, and pH will be measured in situ at 0.5-m intervals using a sonde unit (Hydrolab Quanta, Hach Inc., Loveland, CO) that is precalibrated against known standards and Winkler titrations (APHA 2005). Secchi disk transparency will be measured at each station by lowering a 20-cm diameter alternating black and white disk into the water column until disappearance, then slowly pulling it back up until visible, and recording the depth of visibility. Underwater photosynthetically-active radiation will be measured at 10- to 25-cm intervals using a cosine quantum radiometer (Model LI1000, Li-Cor, Inc., Lincoln, NE). The light attenuation coefficient is calculated as,

$$k_d = \frac{\ln(I_o) - \ln(I_z)}{z}$$

where I_o is the surface radiation ($\mu E \cdot s^2$) and I_z is the radiation at depth z (m). In general, k_d is inversely related to Secchi disk transparency. Thus, higher k_d reflects lower light penetration into the water column and a lower Secchi disk transparency. Regression models developed by Middelboe and Markager (1997) will be used to estimate maximum colonizable depth limits for stem-forming (i.e., *Elodea canadensis*) and rosette-forming (i.e., *Vallisnaria americana*) aquatic macrophytes.

Water samples will be collected biweekly for total phosphorus, viable chlorophyll, and total alkalinity. An integrated water sample over the upper 1-m water column will be collected at each station. Total phosphorus samples will be predigested with potassium persulfate according to Ameel et al. (1993) before colorimetric analysis using the ascorbic acid method (APHA 2005). Samples for viable chlorophyll (i.e., a surrogate measure of algal biomass) will be filtered onto glass fiber filters (Gelman A/E; 2.0 μ nominal pore size) and extracted in 50:50 dimethyl sulfoxide:acetone before fluorometric determination (Welchmeyer 1994). Total alkalinity will be determined via titration of unfiltered water samples with 0.02 N sulfuric acid according to APHA (2005). Algal enumeration will be conducted on integrated water samples collected monthly at station 10.

Task 3. Evaluation of Phosphorus Loading from Selected Storm Sewers (2016-2019)

The objectives of this task are to re-evaluate phosphorus inputs to the system from selected storm sewers after management of internal phosphorus sources from CLP, sediment diffusive flux, and motor boat phosphorus resuspension. In particular, phosphorus loading information from various storm water culverts draining into the lake will be needed to evaluate the effectiveness of watershed best management practices implemented by the City of Eau Claire on lake water quality and the phosphorus budget.

Changes in pool elevation will be monitored using a continuously logging pressure transducer (ISCO model 4120). Flow will be measured at up to 4 storm sewers (Figure 1) between April and September using area-velocity and stage height sensors (ISCO model 4120 or 4150 loggers). Water samples will be collected at storm sewer inputs during periods of rainfall runoff using automated sampling techniques (ISCO model 6700). All samples will be analyzed for total and soluble reactive phosphorus. Samples for total phosphorus will be digested with potassium persulfate (Ameel et al. 1993) and analyzed colorimetrically using the ascorbic acid method (APHA 2005). Samples for soluble reactive phosphorus will be filtered through a 0.45 μm filter prior to colorimetric analysis (APHA 2005).

Task 4. CLP biomass and turion bank (2016-2019)

In June and August, submersed aquatic macrophyte biomass will be quantified at ~140 stations in the lake using the point-intercept method (Madsen, 1993). In May, numbers of germinated CLP per square meter will be quantified at each station. Since germination appears to occur between ice-out through May, this survey will only quantify CLP near the time of herbicide treatment and caution should be used in using this information to interpret herbicide treatment effectiveness. A rake-pull method will be used to collect samples. The rake is lowered to the sediment and raised to the lake surface at a constant, slow rate while twisting the handle to snag macrophyte stems within a ~ 0.13 m^2 area. The samples will be sorted by species and dried to a constant mass at 65 ^oC in a forced-air drying oven. Biomass ($g \cdot m^{-2}$) at each station is estimated as dry mass divided by the circular area covered by a 360 degree twist of the rake. The rake-pull method provides a reasonably accurate biomass estimate for species such as curly-leaf pondweed and Eurasian watermilfoil that is comparable to diver quadrat sampling (Johnson 2010). However, it overestimates biomass in areas dominated by coontail, elodea, and flatstemmed pondweed because these species tend to inter-tangle with plants outside the quadrat area, resulting in unintended sampling from an area wider than that of the rake diameter (Johnson 2010). Thus, caution needs to be used when interpreting biomass data dominated by these species.

Task 5. Evaluation of vertical profiles in aluminum and mobile phosphorus fractions in the sediment (2016-2019)

The objectives of this task are to determine the thickness and location of the Al floc in the sediment, the mass of mobile sediment P that was bound by alum, the Al:P binding ratio and changes in these variables over time. In particular, information generated from this task will be important in determining the extent of sinking of the Al floc over time and changes, if any, in the binding strength of the floc for mobile phosphorus.

A sediment core will be collected at station 10 and 30 (Figure 1) during the summer of each year for vertical sectioning and analysis. These stations are chosen to examine vertical sediment profiles in a region of the lake treated with 150 g Al/m^2 (i.e., station 10) and one treated with 75 g Al/m^2 (i.e., station 30). The cores will be sectioned at 1-cm intervals over the upper 6 cm, 2-cm intervals between 6 and 10 cm, and at 2.5-cm intervals below the 10-cm depth (total sections per station = 10). The close-interval sectioning will provide detail on the location and thickness of the alum floc and the extent of mobile P inactivation. All sections will be analyzed for moisture content (%), density (g/mL), and loss on ignition organic matter (%). This information will be used to estimate the mass of sediment, aluminum, and various mobile P fractions. A known volume of sediment is dried at 105 ^oC for determination of moisture content and sediment density and ashed at 500 ^oC for determination of loss-on-ignition organic matter content (Håkanson and Jansson 2002). Phosphorus fractionation (Table 2) will be conducted according to Hieltjes and Lijklema (1980), Psenner and Puckso (1988), and Nürnberg (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 P). A subsample of the sodium hydroxide extract is digested with potassium persulfate to determine nonreactive sodium hydroxide-extractable P (Psenner and Puckso 1988). Labile organic P is calculated as the difference between reactive and nonreactive sodium hydroxide-extractable P. Aluminum concentration will be determined to identify the location and thickness of the Al floc.

Task 6. Determination of rates of diffusive phosphorus flux from sediment under anoxic conditions (2016 – 2019)

The objectives of this task are to measure rates of diffusive phosphorus flux to evaluate the effectiveness of the Al application in reducing sediment internal phosphorus loading in the lake. Five replicate sediment cores will be collected in the summer of each year at stations 10 and 30 (Figure 1) for determination of rates of phosphorus release under anaerobic conditions according to James et al. (2002). All cores will be drained of overlying water and the upper 10 cm of sediment will be 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 will be 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 consist of the upper 10-cm of sediment and filtered overlying water contained in acrylic core liners that are sealed with rubber stoppers. They are placed in the darkened environmental chamber and incubated at a constant temperature (20 ^{o}C). The oxidation-reduction environment in the overlying water is controlled by gently bubbling nitrogen (anoxic) through an air stone placed just above the sediment surface in each system.

Water samples for soluble reactive phosphorus will be collected from the center of each system using an acid-washed syringe and filtered through a 0.45 µm membrane syringe filter (Nalge). The water volume removed from each system during sampling is replaced by addition of filtered lake water preadjusted to the proper oxidation-reduction condition. These volumes are accurately measured for determination of dilution effects. Soluble reactive phosphorus will be measured colorimetrically using the ascorbic acid method (APHA 2005). Rates of phosphorus release from the sediment (mg m^{-2} d^{-1}) are 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 is used to estimate rates over the linear portion of the data.

Task 7. Reporting

An annual report that describes the results will be provided by 31 December of each year. The reports will describe trends and responses in lake limnology, watershed P loading, sediment P, and macrophyte biomass to the Al and herbicide treatments.

Cost analysis.													
	Task	2016			2017			2018			2019		
		Grant	In-kind	City	Grant	In-kind	City	Grant	In-kind	City	Grant	In-kind	City
1	Herbicide Treatment	$25,000		$25,000	$25,000		$25,000	$25,000		$25,000	$25,000		$25,000
2	Limnological Monitoring	$4,193		$4,193	$4,193		$4,193	$4,193		$4,193	$4,193		$4,193
3	Storm Sewer Loading	$6,520		$6,520	$6,520		$6,520	$6,520		$6,520	$6,520		$6,520
4	Macropytes	$1,875		$1,875	$1,875		$1,875	$1,875		$1,875	$1,875		$1,875
5	Sediment profiling	$1,970		$1,970	$1,970		$1,970	$1,970		$1,970	$1,970		$1,970
6	Sediment P flux	$2,500		$2,500	$2,500		$2,500	$2,500		$2,500	$2,500		$2,500
7	Reporting	$1,525	$3,660	$1,525	$1,525	$3,660	$1,525	$1,525	$3,660	$1,525	$1,525	$3,660	$1,525
	Subtotal	$43,583	$3,660	$43,583	$43,583	$3,660	$43,583	$43,583	$3,660	$43,583	$43,583	$3,660	$43,583
			$87,166			$87,166			$87,166			$87,166	
	Grand Totals												
	City of Eau Claire	$174,332											
	WDNR-AIS Grant	$174,332											
	Total	$348,664											

UW-Stout cost breakdown							
Task	Analysis		Unit price ($)	Sampling Events	No.	Total	Total
Lake Limnology	Field Sampling		$500	9	1	9	$4,500
	Chemistry	Total phosphorus	$22	9	6	54	$1,188
		Alkalinity	$9	9	6	54	$486
		Chlorophyll	$28	9	6	54	$1,512
		Phytoplankton enumeration	$175	4	1	4	$700
		Subtotal					$8,386
Storm Inflow P	Field sampling		$500	20	1	20	$10,000
	Chemistry	Total phosphorus	$22	20	4	80	$1,760
		Soluble reactive phosphorus	$16	20	4	80	$1,280
		Subtotal					$13,040
4. Macropytes	Field Sampling		$500	6	1	6	$3,000
	Processing	Turion and macrophytes	$250	3	1	3	$750
		Subtotal					$3,750
5. Sediment	Field sampling		$500	1	1	1	$500
	Chemistry	Textural	$30	1	20	20	$600
		P fractions	$100	1	20	20	$2,000
		Al metals	$42	1	20	20	$840
	P flux		$500	1	10	10	$5,000
		Subtotal					$8,940
5. Reporting		Data reduction and interpretation	$61		60	50	$3,050
Total							$37,166

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Table 1. A comparison of net changes in phosphorus (P) mass in various regions of the lake between mid-June and late July (see Figure 31) versus estimated P fluxes from the Owen Park pumps and anaerobic sediment. Anaerobic sediment P flux, determined from sediment core incubations, was weighted with respect to an anoxic period of ~ 7 days for the west and east arms. P flux in the south arm was assumed to be negligible due to the occurrence of oxic conditions above the sediment interface throughout the summer.

Variable		(mg/m^2 d)	(kg)
Lake net change in P mass	Lake-wide	1.28	30
	West arm	1.68	20
	East arm	1.04	7
	South arm	0.65	3
Owen park pumps[1]		1.04	24
Anaerobic sediment P flux[2]	Lake-wide	0.72	2.3
	West arm	1.04	2.1
	East arm	0.16	0.2
	South arm	0.00	0

[1]JUN-SEP average

[2]Assumes that bottom waters were anoxic for a 7 days during the period of net lake P mass increase

Table 2. Treatment area and projected herbicide requirements.

Application Specifications:

Herbicide calculations

curly-leaf pondweed, 0.8 mg/L ai

Area (acres)	Depth (ft)	Volume (acre-ft)	Herbicide (gal/acre ft)	Total herbicide (gal)
134	7.64	1024	0.6	614.4

Table 3. Variable list	
In situ	Temperature
	Dissolved oxygen
	pH
	Conductivity
	Secchi Transparency
	PAR
Chemistry	Total phosphorus
	Chlorophyll
	Total alkalinity
	Phytoplankton enumeration[1]

[1]JUN, JUL, AUG, SEP (Station 10 see Figure 1)

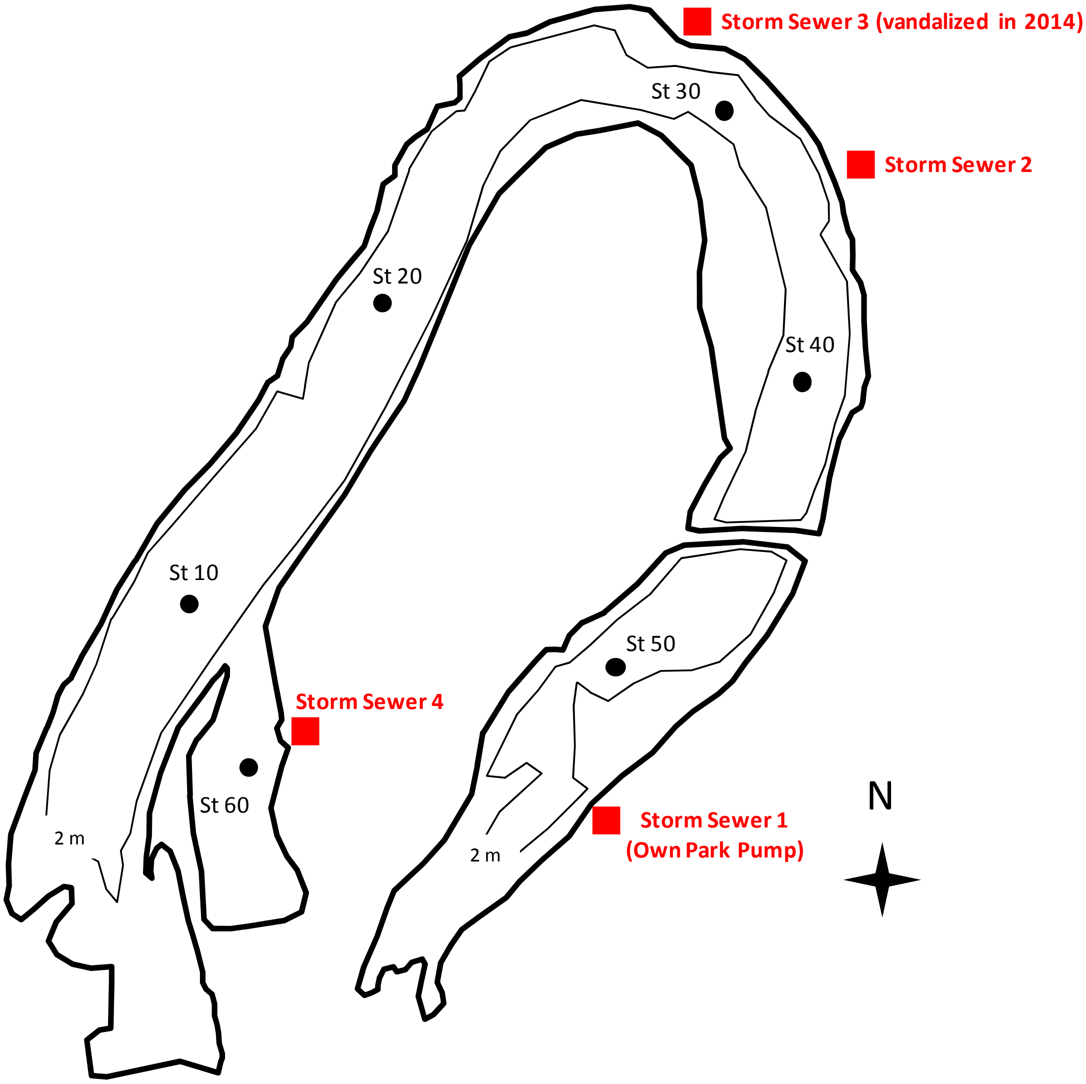

Figure 1. *Sampling station locations in Half Moon Lake.*

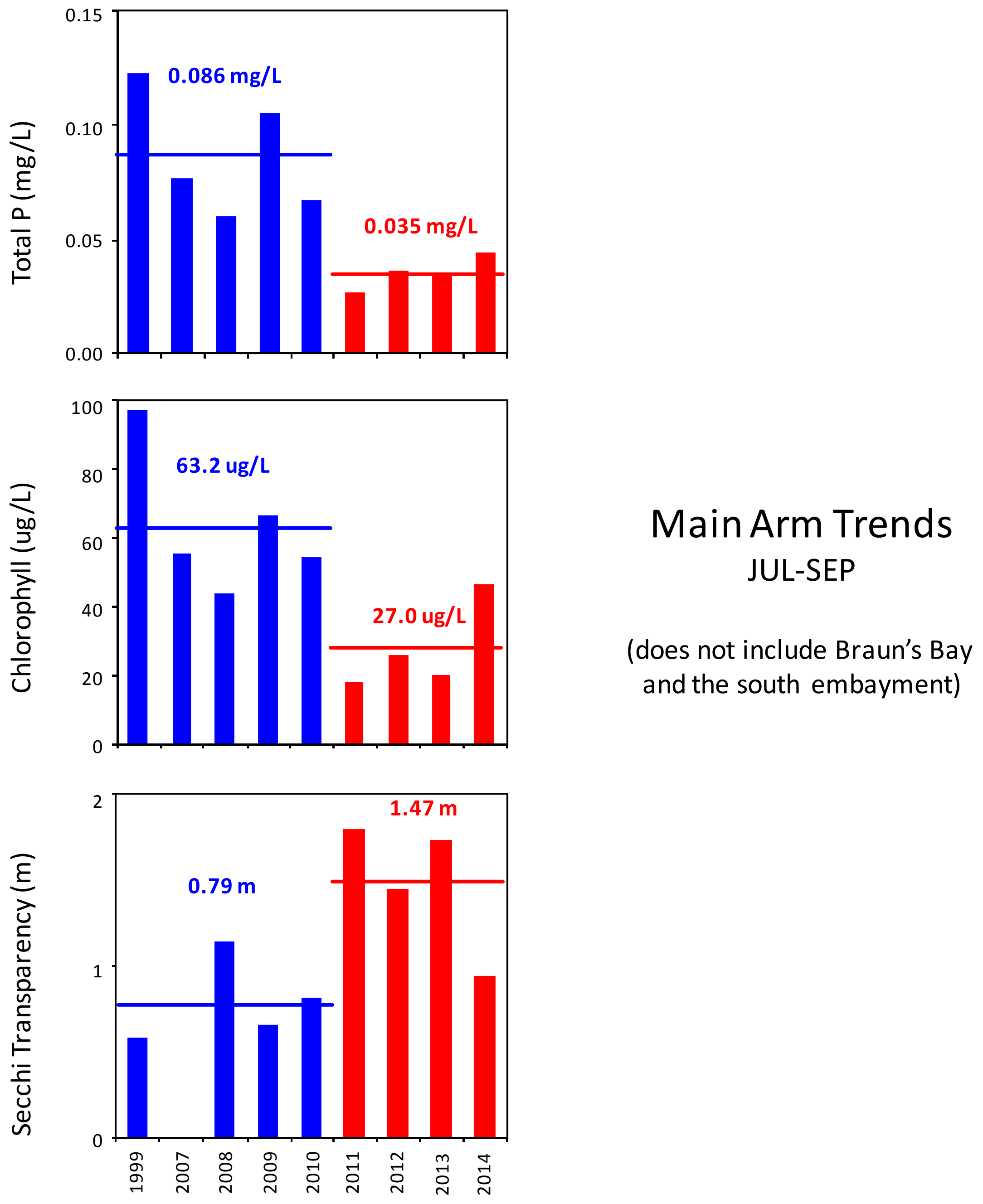

Figure 2. *Summer (JUL-SEP) mean total phosphorus (P), chlorophyll, and Secchi transparency in the main arm (i.e. west and east arms) of Half Moon Lake before (blue bars) and after (red bars) Al treatment (June, 2011). Horizontal lines represent overall pre- and post-treatment means.*

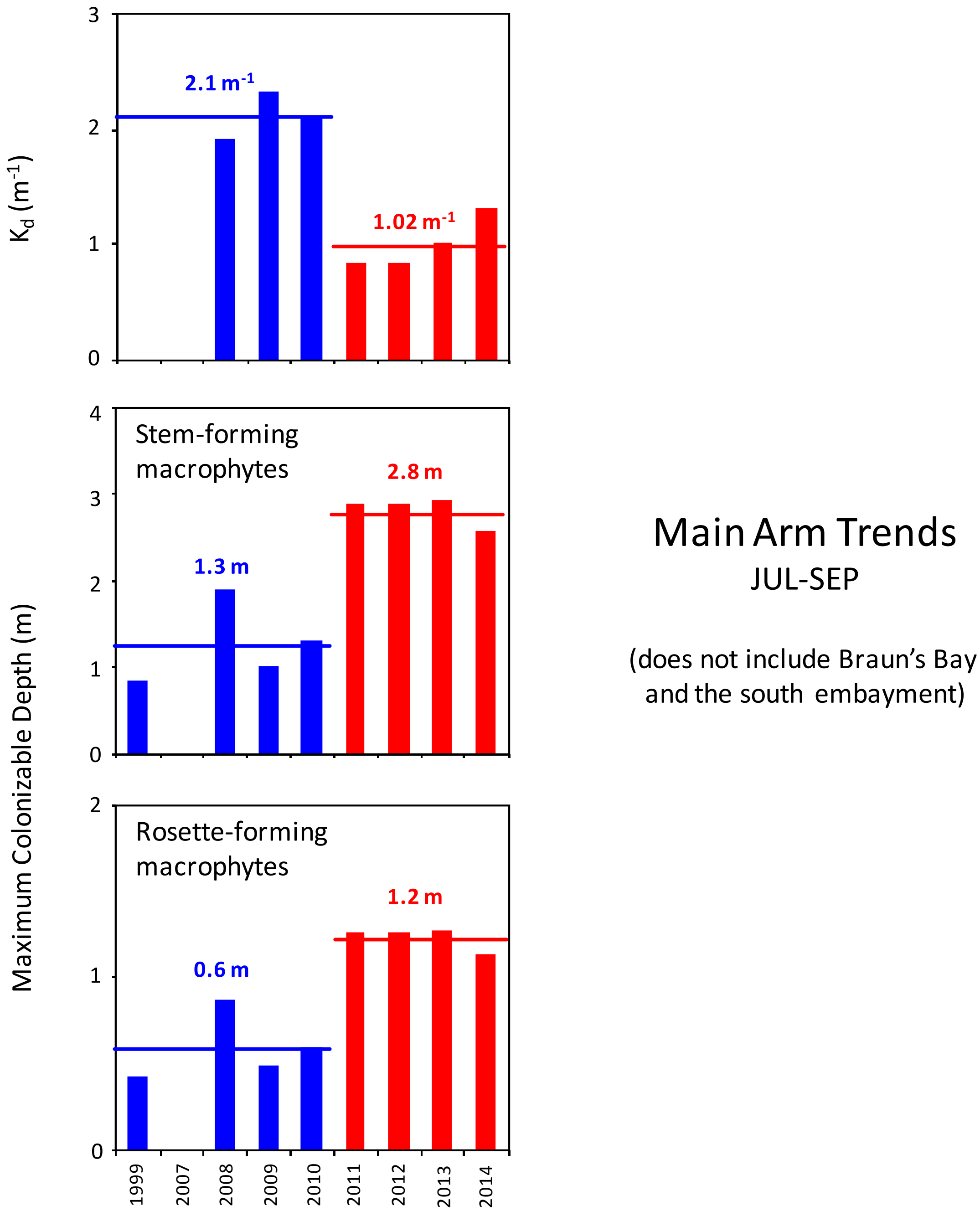

Figure 3. *Summer (JUL-SEP) mean maximum inhabitable depth for stem- and rosette-forming submersed aquatic macrophytes based on Middelboe and Markager (1997) in the main arm (i.e. west and east arms) of Half Moon Lake before (blue bars) and after (red bars) Al treatment (June, 2011). Horizontal lines represent overall pre- and post-treatment means.*

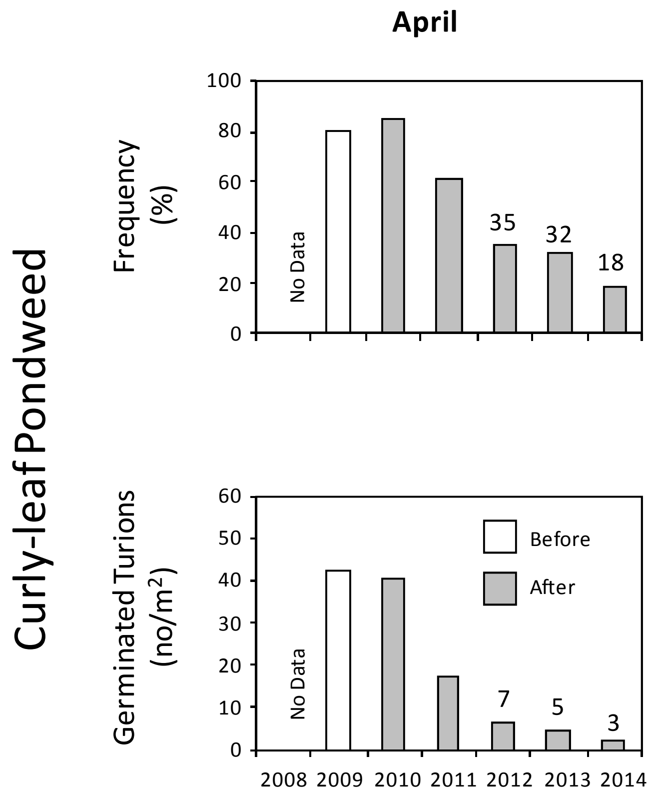

Figure 4. *Percent occurrence (n ~ 140-160; upper panel) and numbers (lower panel) of germinated curly-leaf pondweed tuions in April of various years. White columns represent means before the start of early spring Endothall treatments.*

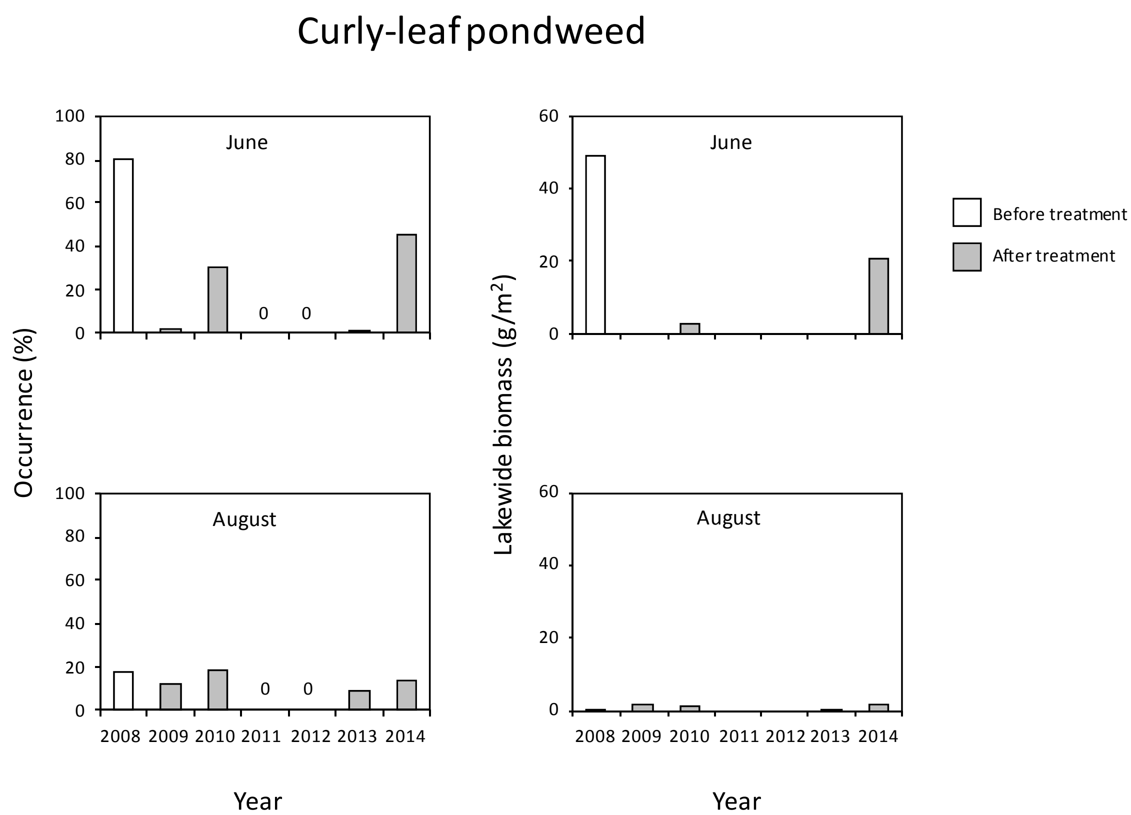

Figure 5. *Curly-leaf pondweed lake-wide mean biomass (n ~ 140-160) in June and August of various years. White columns represent means before the start of early spring Endothall treatments.*

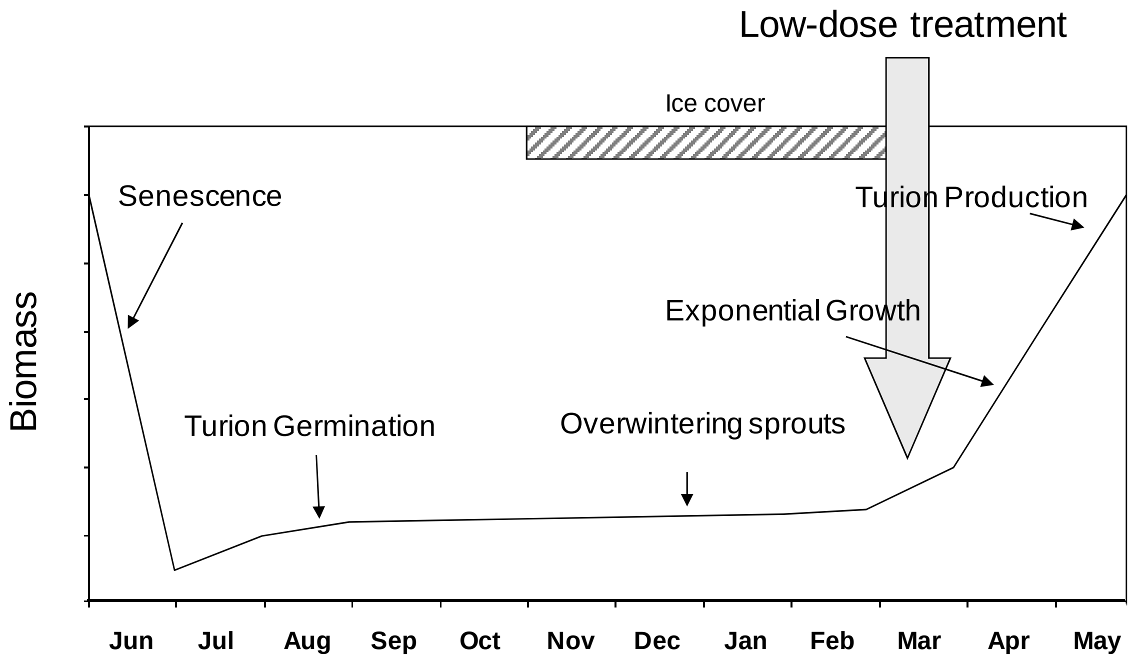

Figure 6. *Conceptual diagram depicting the annual life cycle of curly-leaf pondweed in the upper mid-western United States (Woolf and Madsen 2003).*

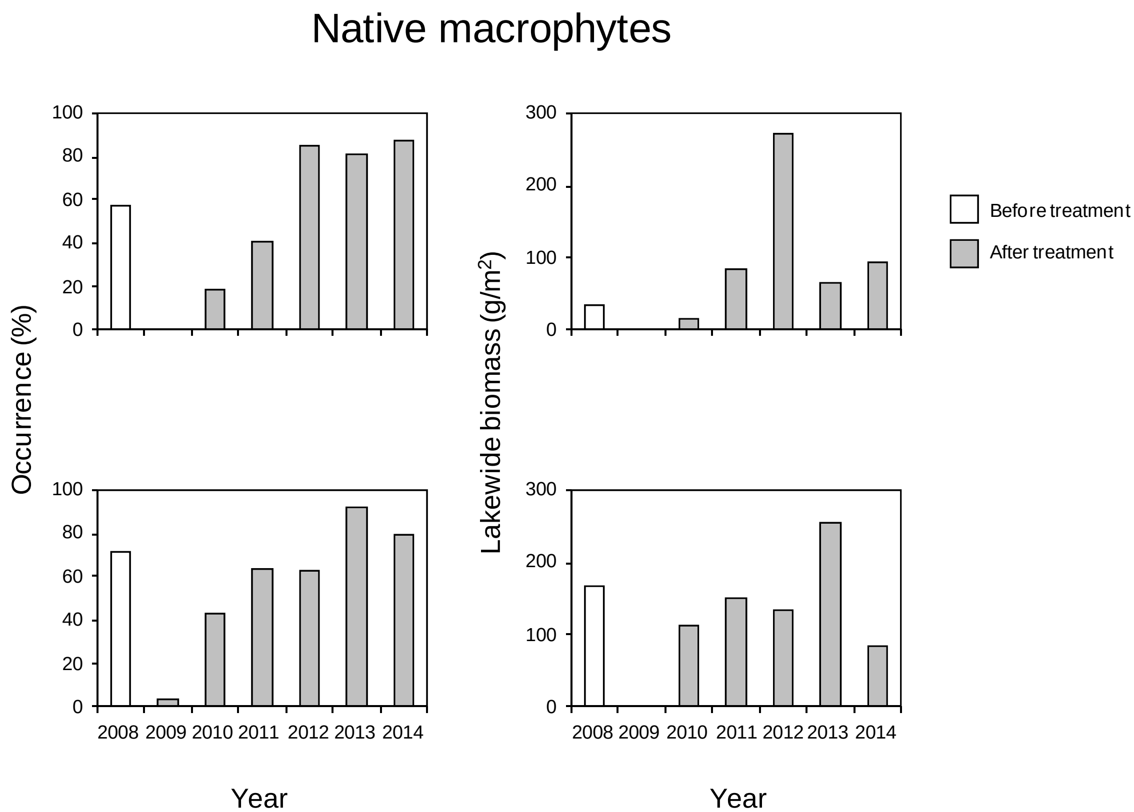

Figure 7. *Native submersed macrophyte lake-wide mean biomass (n ~ 140-160) in June and August of various years. 2008 was the pretreatment year. The pretreatment native macrophyte community in 2008 was dominated by coontail while elodea has dominated the post-treatment assemblage.*

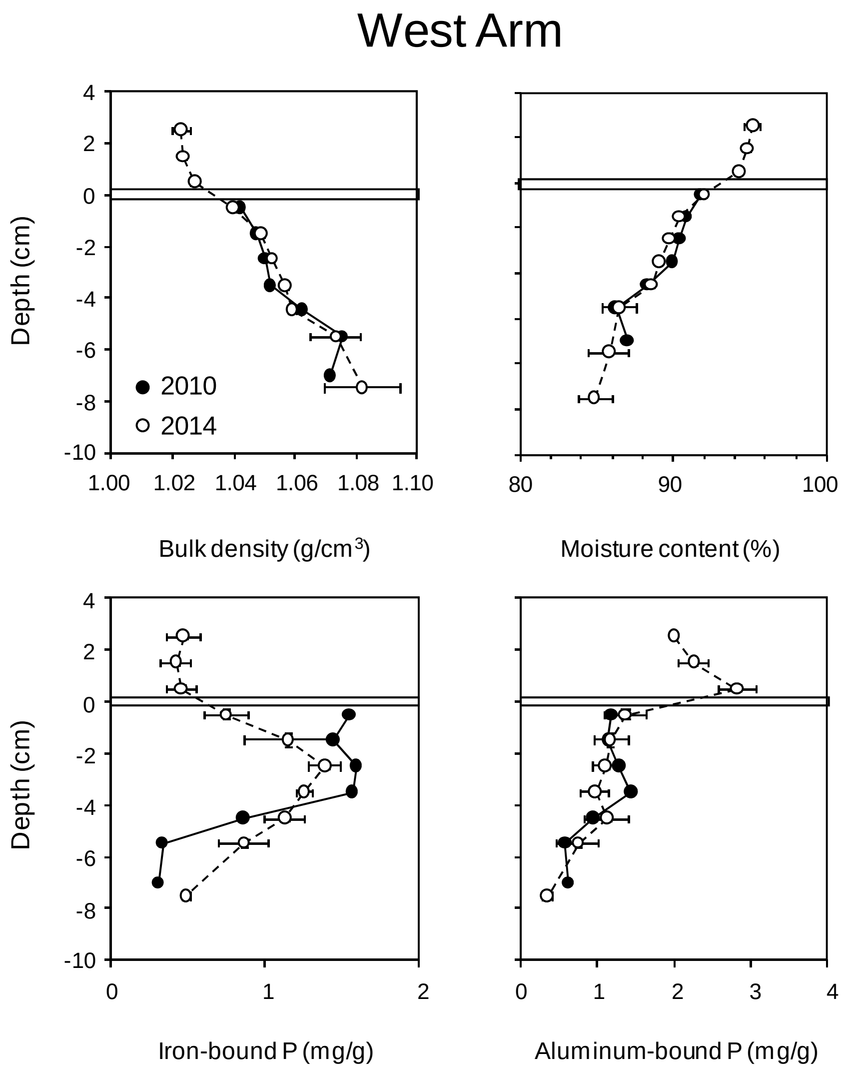

Figure 8. *Variations in sediment wet bulk density (upper left), moisture content (upper right), sediment iron-bound phosphorus (lower left), and aluminum-bound P (lower right) as a function of depth below the sediment-water interface for sediment cores collected at station 10 located in the west arm of Half Moon Lake. The year 2010 represents concentrations before alum treatment and the year 2014 represents conditions in July, three years after Al application. The horizontal gray line denotes the location of the original pretreatment surface sediment-water interface.*

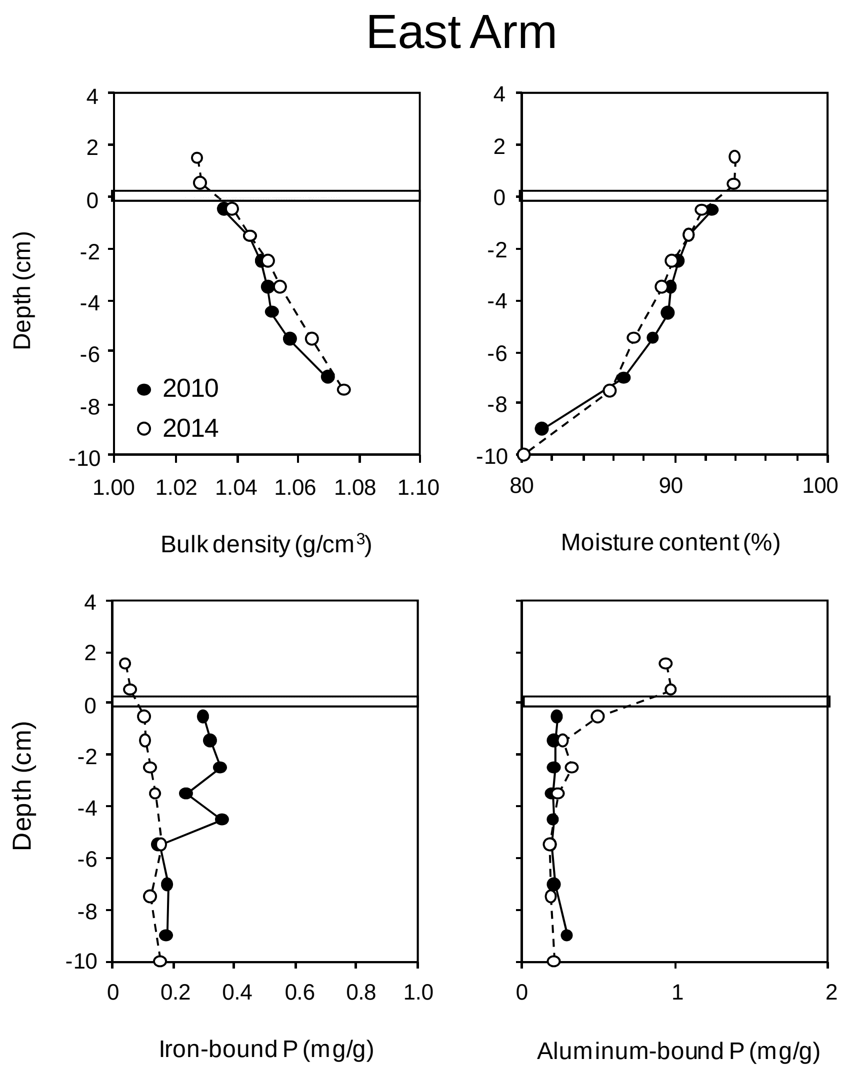

Figure 9. *Variations in wet bulk density (upper left), moisture content (upper right), iron-bound phosphorus (lower left), and aluminum-bound P (lower right) as a function of depth below the sediment-water interface for cores collected at station 30 located in the east arm of Half Moon Lake. The year 2010 represents concentrations before alum treatment and the year 2014 represents conditions in July, three years after Al application. The horizontal gray line denotes the location of the original pretreatment surface sediment-water interface.*

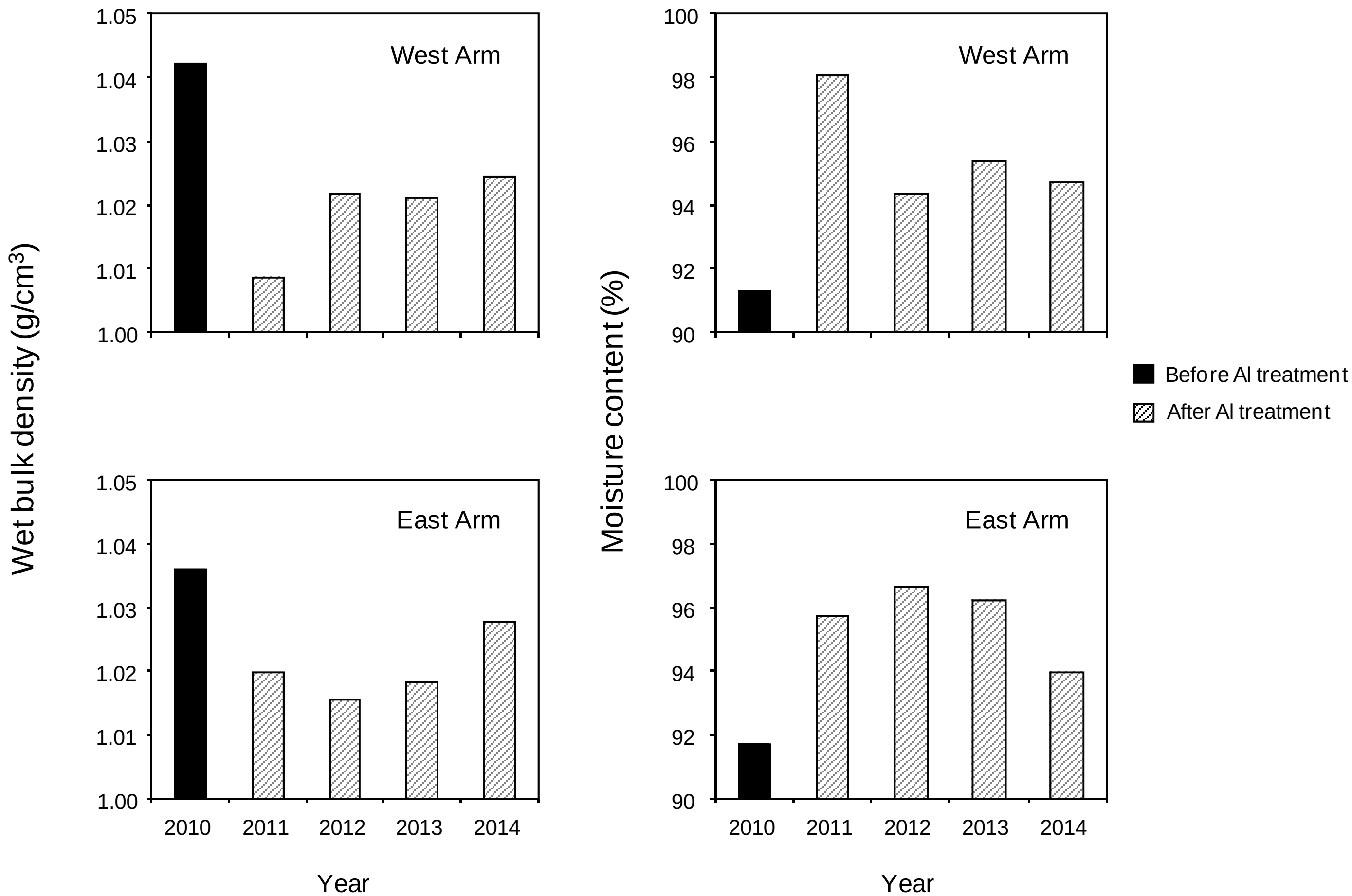

Figure 10. *Annual variations in wet bulk density (left panels) and moisture content (right panels) in the surface (upper 1 cm) sediment layer before and after Al treatment.*

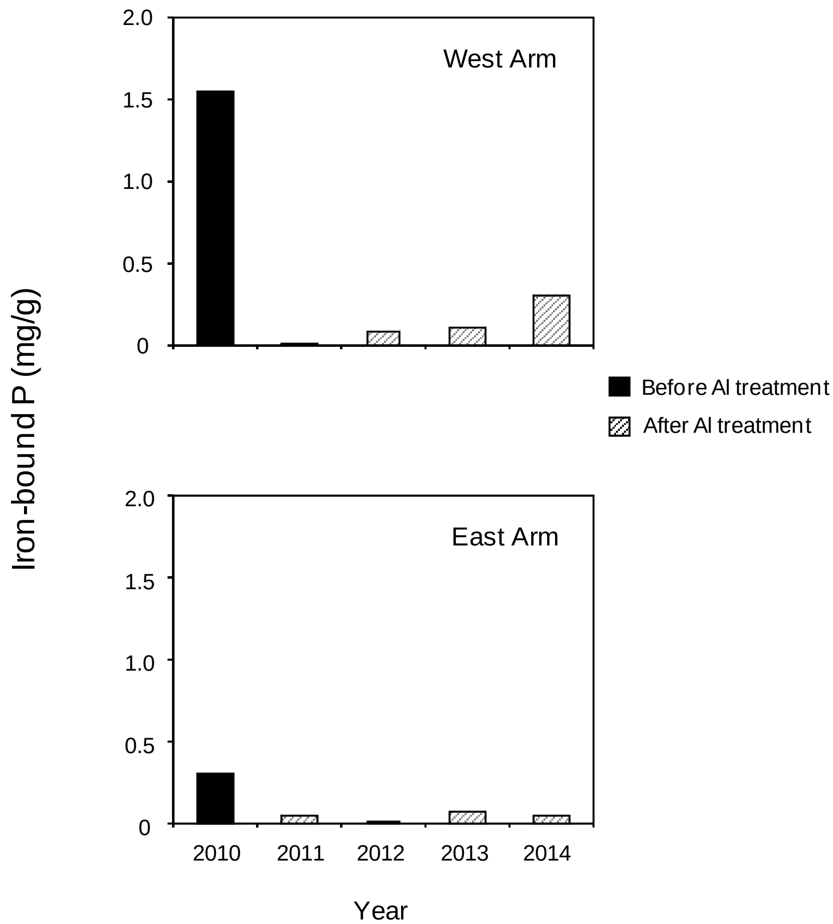

Figure 11. *Annual variations in iron-bound phosphorus (P) in the surface (upper 1 cm) sediment layer before and after Al treatment.*

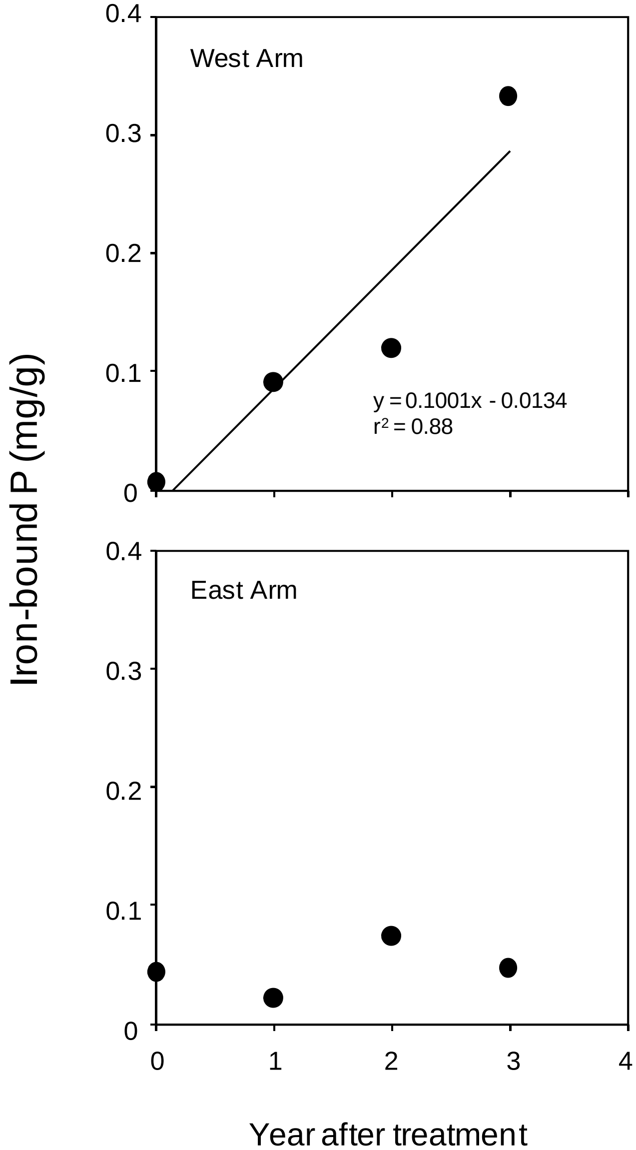

Figure 12. *Relationships between year after Al treatment and the concentration of iron-bound phosphorus (P) in the surface (upper 1 cm) sediment layer.*

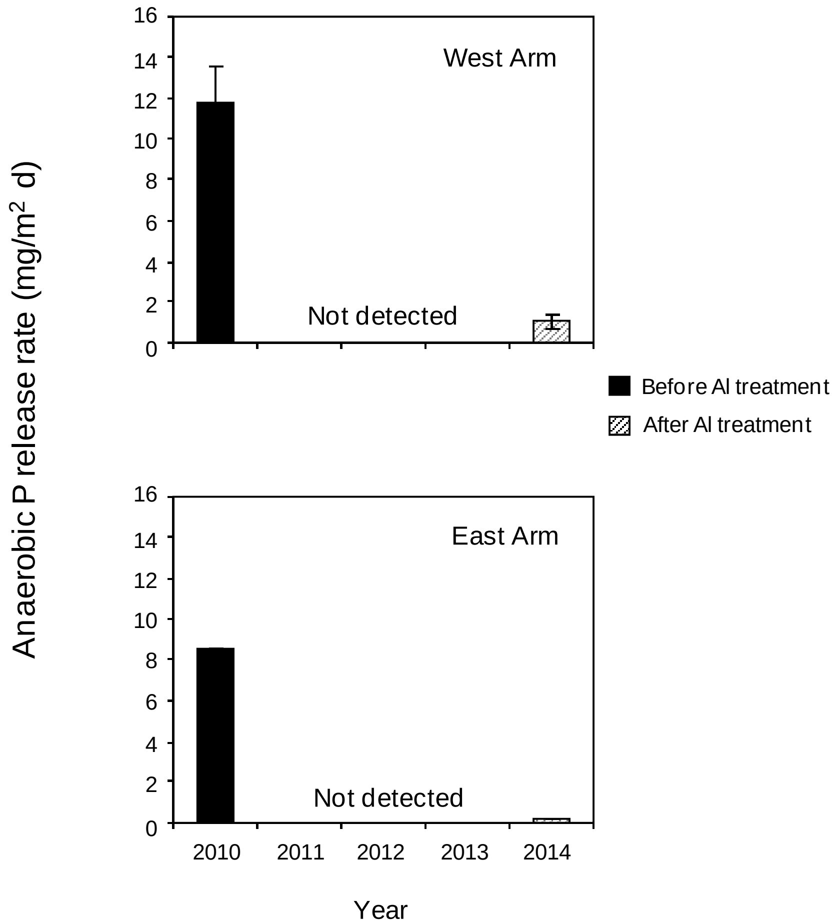

Figure 13. *Annual variations in the mean (n = 5) anaerobic phosphorus (P) release rate before and after Al treatment. Horizontal lines represent 1 standard error.*