Flooded Muck Fields as a Source of Phosphorus to Mona Lake

Final Project Report

February 2010

Alan D. Steinman
Mary E. Ogdahl

Grand Valley State University
Annis Water Resources Institute

Project Background and Description

The Mona Lake Watershed faces some of the most serious water quality challenges in west Michigan. The two major tributaries to Mona Lake, Black Creek and Little Black Creek, have both been placed on the State of Michigan's section 303(d) list for impaired biotic diversity and abundance resulting from excessive sand bedload. In addition to sedimentation, the tributaries to Mona Lake are impacted by heavy metals, PCBs, and polycyclic aromatic hydrocarbons . Total phosphorus (TP) levels in Mona Lake are in the eutrophic to hypereutrophic range (Steinman et al. 2006). Annual mean surface TP concentrations range from 60 to 80 µg/L with near-bottom concentrations reaching up to 400 µg/L (Steinman et al. 2006), due largely to internal loading from phosphorus-rich sediments (Steinman et al. 2009). These high phosphorus (P) concentrations have been implicated in cyanobacterial blooms in Mona Lake, including potentially toxic species (Hong et al. 2006).

Phosphorus source control is one of the priorities identified in the Mona Lake Watershed Management Plan developed by the Mona Lake Watershed Council (MLWC). Nearly 80% of the external TP load entering Mona Lake comes from Black Creek (Steinman et al. 2006). As a consequence, source control from this tributary provides the greatest benefit for Mona Lake. Abandoned muck farms used for celery production, but now converted into shallow lakes, abut Black Creek just before it enters Mona Lake. Given their location and past land use, these muck fields are likely to be major contributors of P to Mona Lake. This project is in direct response to a priority objective outlined in the Mona Lake Watershed Management Plan: *Determine exact contributions of nutrients from the old celery fields along Black Creek*.

Flooded fields, such as these, can be converted to wetlands, which can then serve as efficient retention areas for phosphorus via sorption to sediments and biotic uptake by periphyton, macrophytes, and microbial communities (Reddy et al. 1999). Physical retention of P-rich sediments provides an additional mechanism by which wetlands can reduce P loads in surface waters (Dosskey 2001). However, wetlands also have the potential to release P to the overlying water depending on the relationship between sediment P sorption and water column P concentrations (Richardson 1985, Pant and Reddy 2001). The equilibrium P concentration (EPC_0) of sediment represents the aqueous P concentration at which no net sorption or desorption occurs when sediment is suspended in a water sample. When the water column P concentration is less than the sediment EPC_0, sediments can potentially release P and vice versa.

We monitored conditions in Black Creek upstream and downstream of the muck fields from April to November, 2009. In the muck fields, we measured sediment EPC_0 and pore water phosphorus. Data collected as part of this study allow us to identify the importance of these historic muck flats as a P source to Mona Lake, the mechanisms responsible for the release, and possible mitigation strategies to treat the phosphorus which, ultimately, will help limit the development of potentially toxic cyanobacterial blooms in Mona Lake. In addition to providing data for local decision making, this project utilizes a methodology that can serve as a model for examining water bodies impacted by P loading from muck farming (e.g., the White Lake and Kalamazoo River AOCs, Lake Macatawa watershed, and Bear Lake watershed) and for assessing the importance of riparian or fringing wetlands as a source of phosphorus to connecting surface water bodies.

Methodology

Instream Monitoring

Two monitoring stations were established in Black Creek to measure conditions upstream (43.18843˚N, 86.20629˚W) and downstream (43.18478˚N, 86.22222˚W) of the muck fields (Figure 1). Instream monitoring was conducted weekly from April 6 to November 4, 2009.

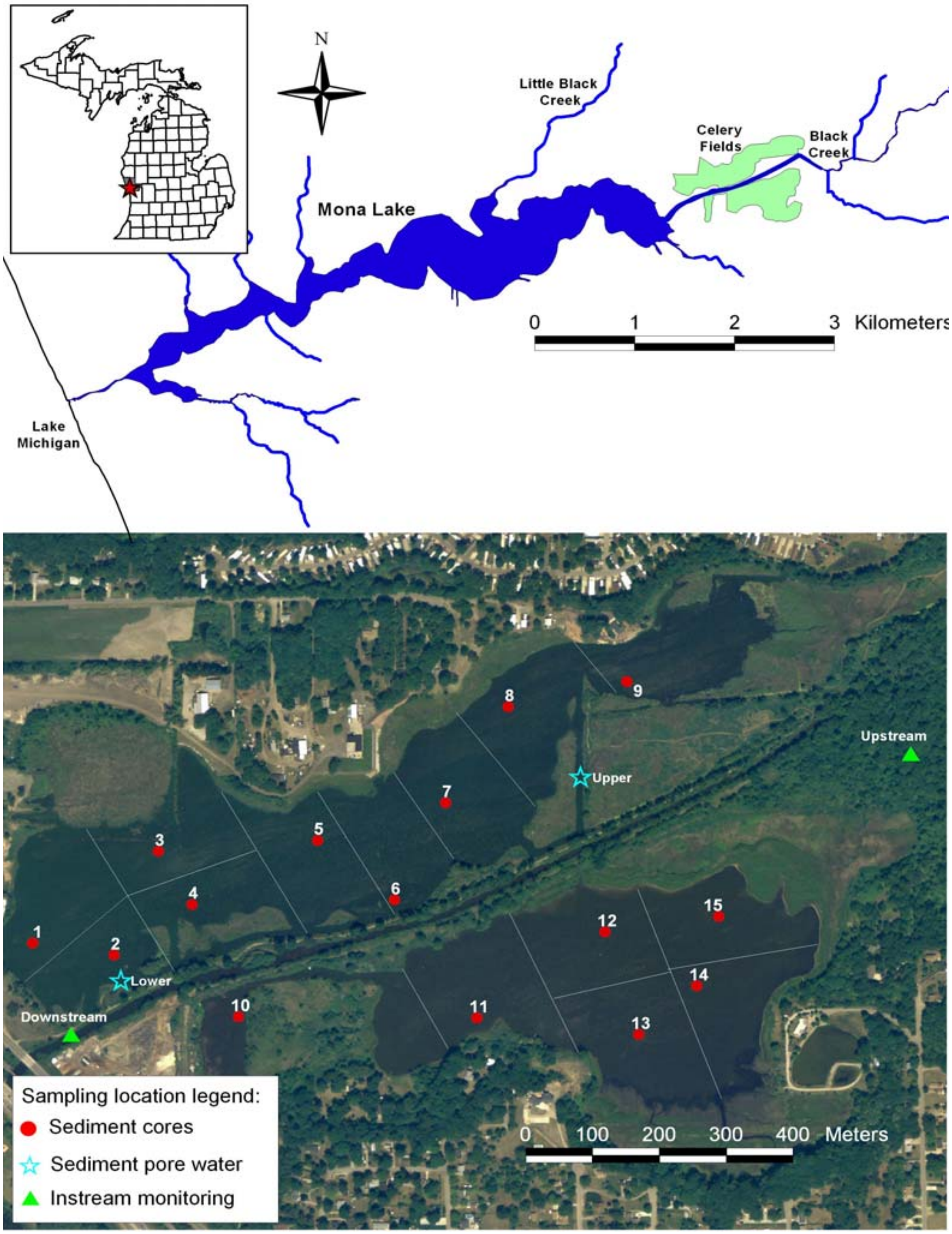


Figure 1: Study site map showing instream and muck fields sampling locations (bottom panel). The top panel shows the location of the celery fields in relation to Mona Lake. Inset: location of project area within the state of Michigan.

During each visit, grab samples were collected for analysis of TP and soluble reactive phosphorus (SRP). A YSI 6600 multi-probe sonde was used to measure dissolved oxygen, turbidity, pH, temperature, redox potential, specific conductance, total dissolved solids, chlorophyll *a,* and blue-green algae. Water depth and velocity were measured at twelve equally-spaced points along the transects. Water velocity was measured at 0.6 x depth according to USGS protocols using a Marsh-McBirney Flow Mate 2000 flow meter attached to a top-setting wading rod. The Windows-based hydrologic software, HYDROL-INF (Chu 2006) was used to calculate stream discharge.

Water samples were stored on ice until transported to the laboratory, always within 4 h of collection. Water for SRP was syringe-filtered through 0.45-µm membrane filters into scintillation vials and frozen until analysis. SRP and TP were analyzed on a BRAN+LUEBBE Autoanalyzer (APHA 1992).

Sediment Pore Water Sampling

Sediment pore water was sampled in 2 locations (upper: 43.18834˚N, 86.21296˚W; lower: 43.18551˚N, 86.22145˚W; Figure 1) within the north muck field in June and August, 2009, using modified Hesslein in-situ pore water samplers (i.e., peepers). Site selection was based on water depth and sediment type. The peepers were prepared in the laboratory by filling the compartments with deoxygenated deionized (DI) water and covering them with a 5 µm dialysis membrane and slotted cover. The peepers were submerged in deoxygenated DI water overnight before deployment in the field. Duplicate sets of peepers were deployed at both locations. Although our goal was to place the peepers so that at least 5 compartments extended above the sediment and into the water column, the flocculent nature of the sediment required us to place the peepers deeper for stability. As a result, only 0-2 compartments were exposed to the water column. Peepers were sampled after 14 days using a syringe needle to pierce the membrane and evacuate the liquid in 10 compartments from each peeper. These samples represented a SRP profile from the water column (or sediment surface) to approximately 35 cm sediment depth. Samples were processed and analyzed for SRP as described above.

Sediment Collection and Phosphorus Sorption Measurements

Sediment cores for phosphorus isotherm measurements and TP analysis were collected from 15 locations in the muck fields on July 7, 2009. The coring locations were chosen by dividing the muck fields into 15 sections, each of approximately equal size, and randomly selecting one location from each section. One sediment core (ca. 4-cm diameter) was collected using a modified core sampler (Davis and Steinman 1998) from each the 15 sites and transported to the laboratory for P isotherm measurements and TP analysis. Water column samples also were collected at each location for analysis of SRP and TP, according to the methods described above. General water quality parameters were measured at the time of collection using a YSI 6600 sonde (see above).

In the laboratory, the top 15 cm of sediment were extruded from each core, placed in plastic bags, and homogenized by hand. Phosphorus isotherms were determined in triplicate for each sediment core, according to a modified version of the procedure in Mozaffari and Sims (1994) and Novak et al. (2004). Briefly, 50-mL centrifuge tubes containing 5 g of wet sediment with 10 mL of inorganic P solutions (KH_2PO_4 dissolved in 0.01 M KCl) containing 0, 0.01, 0.1, 1, 5, 10, 50, 100, and 500 mg P/L were shaken for 24 h at 250 RPM on an orbital shaker table. At the beginning of the incubation, one of the three triplicate samples from each core was sterilized with 3 drops of chloroform to inhibit microbial activity. We chose KCl as the background electrolyte based on pH data that revealed alkaline conditions in the muck fields (Table 2), which can cause precipitation of Ca when $CaCl_2$ is used (Graetz and Nair 2000). After centrifugation and filtration (0.45 µm), the SRP in the supernatant was analyzed as described above. EPC_0 was calculated using the equations in Pant and Reddy (2001).

In addition to the isotherm samples, a 5-g subsample was used for determination of ash-free dry mass (AFDM) and sediment organic matter (OM). Sediment subsamples were dried for 24 hr at 105˚C, weighed, ashed at 550˚C for 4 hr, and re-weighed. The resultant ashed material was used for analysis of sediment TP, according to the method described above.

Data Analysis

Data were analyzed for differences in 1) water quality between upstream and downstream stations in Black Creek, 2) water quality and sediment characteristics between north and south muck fields, and 3) EPC_0 between sterilized and non-sterilized sediment cores using either t-test (normally-distributed data) or Mann-Whitney Rank Sum test (non-normally-distributed data). Normality was tested using the Kolmogorov-Smirnov test. All statistical analyses were conducted using SigmaStat (version 3.1; Systat).

Results

Instream Monitoring

Concentrations of several key water quality parameters indicative of eutrophication were significantly higher at the downstream sampling location than at the sampling site located upstream of the muck fields (Table 1). For example, average TP concentrations were extremely high (>0.050 mg/L) downstream of the muck fields. Two exceedingly high TP concentrations, greater than 0.20 mg/L, were measured on consecutive weeks during spring (Figure 2). TP concentration also was more variable downstream (SE 0.010 mg/L) than upstream of the muck fields (SE 0.002 mg/L; Table 1), but the variability did not appear to be related to precipitation events (Figure 2). Like TP, SRP was significantly higher downstream than upstream of the muck fields, but was an order of magnitude lower and less variable than TP (Table 1, Figure 2). In addition to phosphorus, turbidity, chlorophyll *a*, and blue-green algae were all higher downstream of the muck fields. Collectively, these data indicate that the muck fields are a source of phosphorus to Black Creek (and ultimately, to Mona Lake), and are contributing to ecological impairments in these aquatic systems.

Over the sampling period, TP load was significantly increased in Black Creek due to inputs from the muck fields. Average TP load downstream of the fields was 2.6 times higher than upstream (Table 1), and 2 weekly measurements during spring were 10-15 times higher (Figure 3). Although SRP load was higher (1.6 times) downstream than upstream of the muck fields, the difference was not statistically significant (Table 1). Similar to P concentration, variability in P load does not appear to be related to precipitation events (Figure 3).

Table 1: Summary of instream parameters averaged over the sampling period. P-values represent significance of upstream vs. downstream comparisons using Mann-Whitney Rank Sum Test. Sample number (*n*) is shown in the last column. Bold font indicates statistically significant differences (P<0.05).

	Upstream		Downstream			
	Mean	SE	Mean	SE	p	*n*
Temp (C)	13.36	0.73	13.89	0.79	0.409	24
DO (mg/L)	9.74	0.21	10.09	0.26	0.257	
DO% (%)	92.8	1.50	97.5	2.4	0.103	
pH	8.07	0.04	8.08	0.05	0.916	
SpCond (mS/cm)	0.429	0.01	0.439	0.009	0.422	
ORP (mV)	373	5.42	368	6	0.607	
TDS (g/L)	0.279	0.00	0.285	0.006	0.422	
Turbidity (NTU)	**4.8**	**0.48**	**7.3**	**0.8**	**0.018**	
Chlorophyll (µg/L)	**6.2**	**0.60**	**12.8**	**1.9**	**0.003**	
BGA (cells/mL)	**388**	**156**	**3956**	**956**	**0.004**	
Discharge (cms)	1.66	0.21	1.74	0.23	0.975	29
SRP-P (mg/L)	**0.008**	**0.001**	**0.012**	**0.001**	**0.017**	30
TP-P (mg/L)	**0.03**	**0.002**	**0.07**	**0.010**	**<0.001**	
SRP Load (kg/day)	1.00	0.12	1.61	0.25	0.096	29
TP Load (kg/day)	**4.10**	**0.63**	**10.86**	**2.47**	**0.011**	

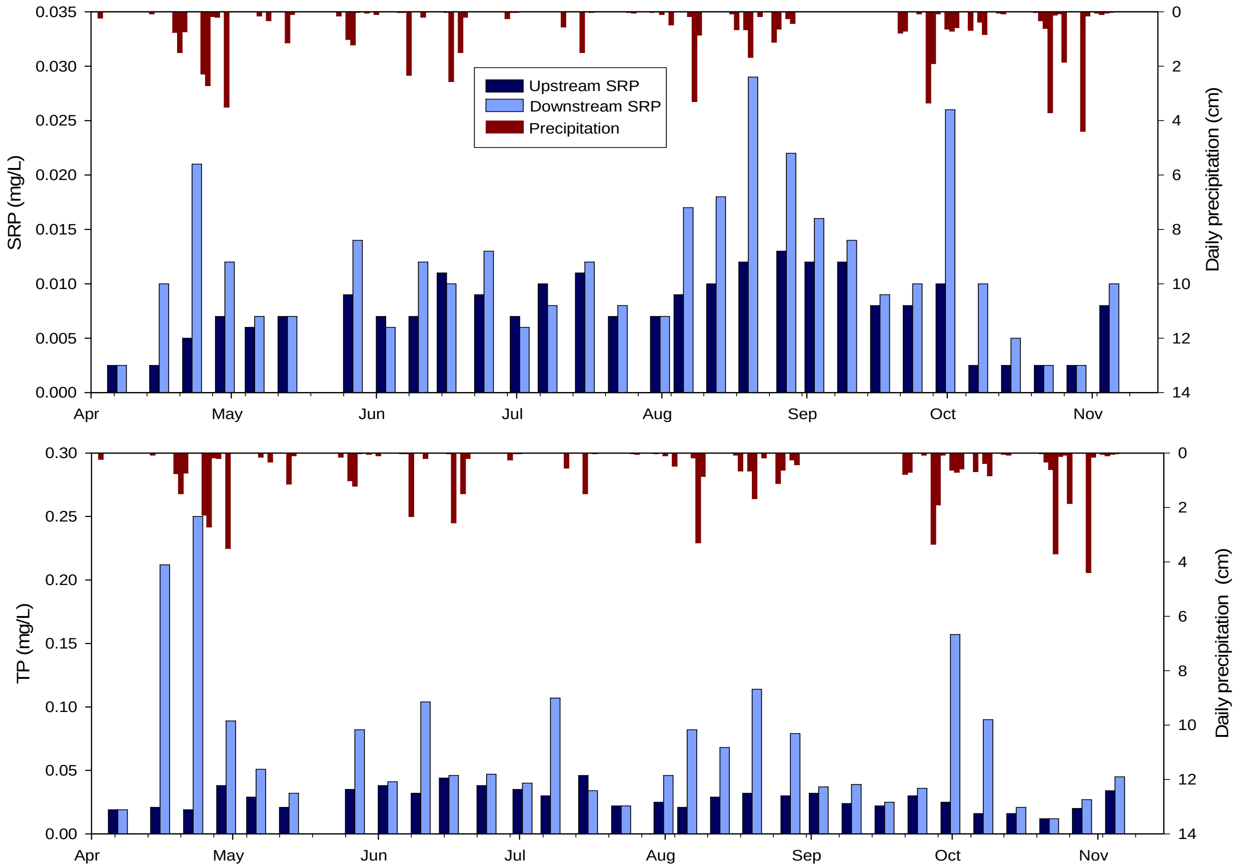


Figure 2: Instream SRP and TP concentrations (mg/L) upstream and downstream of the muck fields. Daily precipitation data (www.wunderground.com) are shown in red.

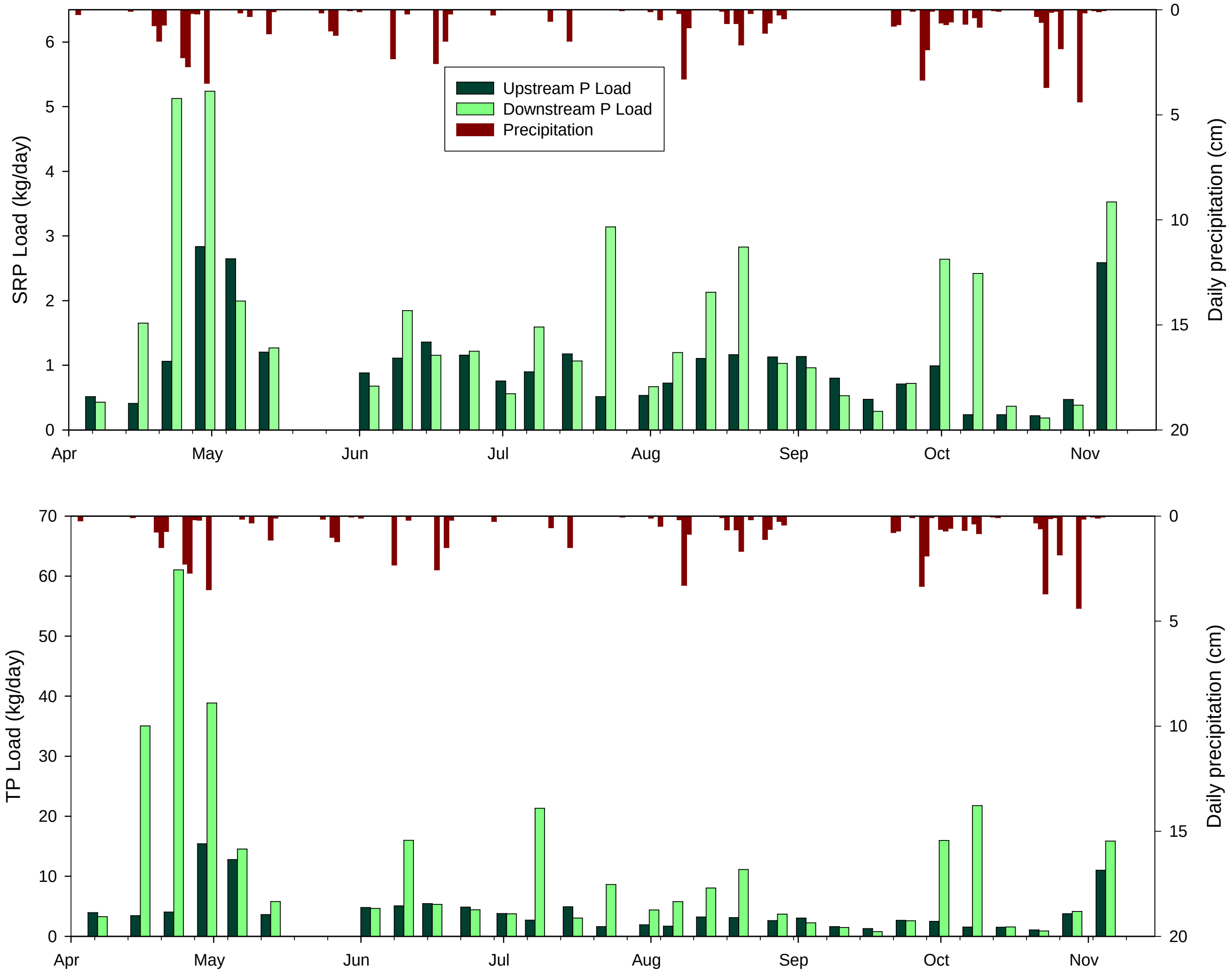


Figure 3: TP and SRP load (kg/day) measured upstream and downstream of the muck fields. Daily precipitation data (www.wunderground.com) are shown in red.

Importance of Muck Fields as Source of Phosphorus to Mona Lake

The difference in TP load between the upstream and downstream Black Creek sites gives an indication of how much total phosphorus is leaving the muck fields and entering Black Creek, just above where it flows into Mona Lake. There is a net influx of 6.76 kg/d of TP from the muck fields to Black Creek (Table 1). If this load is extrapolated to the entire year, the muck fields account for a TP load of 2,467 kg/yr, accounting for ~62% of Black Creek's total load when measured at the downstream site.

Based on the estimates from the current study, Black Creek's TP load entering Mona Lake is ~3,964 kg/yr (10.86 kg/d x 365 d). This value is about 40% greater than estimate from prior studies: 2,221 kg/yr (Steinman et al. 2006) and 2,258 kg/yr (Steinman et al. 2009). Differences among studies are likely attributable to variable precipitation amounts and sampling frequencies in the studies. Regardless of the value used to estimate total phosphorus load from Black Creek, it is clear that the TP load leaving the muck fields accounts for a substantial, if not overwhelming, amount of the TP reaching Mona Lake through Black Creek. Nonetheless, the TP concentration and load upstream of the muck fields is still substantial, suggesting that phosphorus control actions are needed not only in the muck fields but also in the upper reaches of the watershed.

Muck Fields Sediment and Water Quality

Water Quality and Sediment Characterization

Data from our mid-summer sampling of water quality in the muck fields indicate extreme hypereutrophic conditions that are more severe in the south than the north field (Figure 3, Table 2). Average TP concentrations in the north and south muck fields were 0.37 mg/L and 0.71 mg/L, respectively, which are 10 times greater than the average instream TP concentrations (Figure 4, Table 1). Furthermore, these concentrations are similar to, or greater than, those measured near the bottom of Mona Lake during summer, when internal P loading was high (Steinman et al. 2009). SRP concentrations were much lower than TP, suggesting rapid biotic uptake of phosphorus (Figure 4). In addition to phosphorus, concentrations of chlorophyll, turbidity, and dissolved oxygen were all extremely high in the muck fields (Table 2). Average water temperature was 23.8˚C, which was 8˚C warmer than Black Creek on the same day. The very high DO concentrations and elevated pH are indicative of very active photosynthesis, which is consistent with the high chlorophyll concentrations and cyanobacteria cell densities (Table 2).

Both sediment TP and organic matter (OM) were significantly higher in the north field than the south field (Figure 5), which is in contrast to the relationship we observed in water column TP (Figure 4). TP in ash-free sediment ranged from 2,030-9,250 mg/kg in the north field and 131-2,130 mg/kg in the south field. In comparison, sediment TP in Mona Lake ranged from 1,500-5,000 mg/kg during a previous study (Steinman et al. 2009). Sediment in the north field was typically muck, while the south field generally had greater sand content.

Table 2: Summary of water quality parameters measured on July 7, 2009 in the north and south muck fields. P-values represent significance of north vs. south field comparisons using t-test (normally-distributed data) or Mann-Whitney Rank Sum Test (non-normally distributed data). Bold font indicates statistically significant differences (P<0.05).

	North (*n*=9)		South (*n*=6)		
	Mean	SE	Mean	SE	P
Temp (C)	**22.87**	**0.14**	**25.25**	**0.22**	**<0.001**
DO (mg/L)	13.37	0.37	15.85	1.33	0.052
DO% (%)	**155.6**	**4.1**	**193.1**	**16.7**	**0.039**
pH	**9.22**	**0.06**	**9.79**	**0.29**	**0.039**
SpCond (mS/cm)	**0.535**	**0.014**	**0.506**	**0.020**	**0.039**
ORP (mV)	**305**	**3**	**244**	**5**	**<0.001**
TDS (g/L)	**0.348**	**0.009**	**0.329**	**0.013**	**0.039**
Turbidity (NTU)	**29.4**	**2.4**	**100.0**	**16.0**	**0.008**
Chlorophyll (μg/L)	**105.8**	**7.1**	**205.8**	**32.6**	**0.003**
BGA (cells/mL)	**49742**	**3711**	**231694**	**43637**	**0.039**

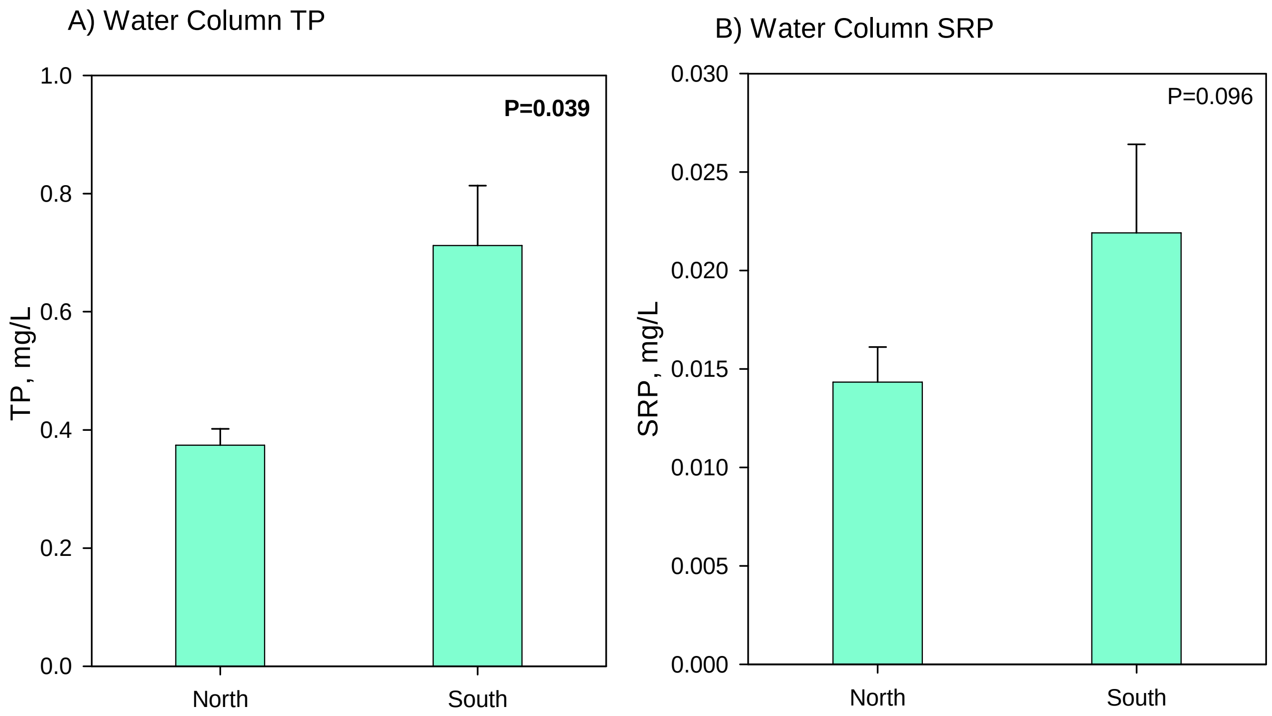


Figure 4. Average (±SE) TP (A) and SRP (B) in the water column of the north and south muck fields. P-values represent significance of north vs. south field comparisons using t-test (normally-distributed data) or Mann-Whitney Rank Sum Test (non-normally-distributed data). Bold font indicates a statistically significant difference (P<0.05).

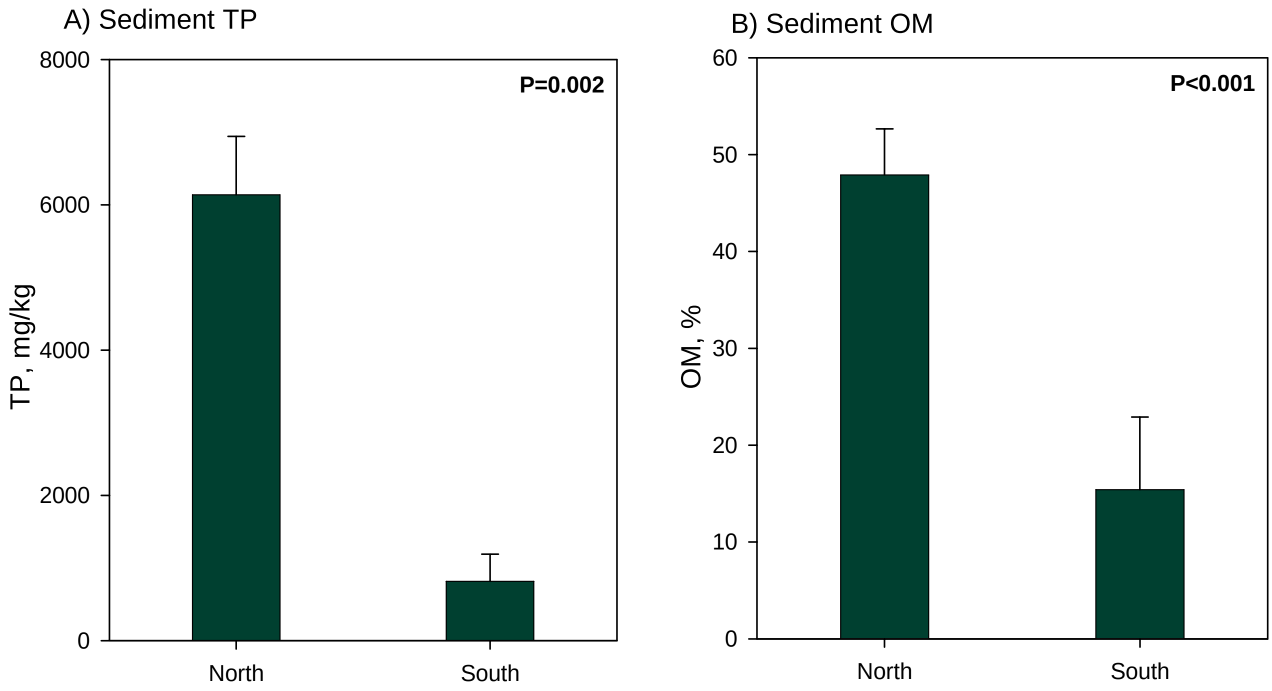


Figure 5. Average (±SE) TP in ashed sediment (A) and organic matter (B) in the sediments of the north and south muck fields. P-values represent significance of north vs. south field comparisons using t-test. Bold font indicates a statistically significant difference (P<0.05).

P Sorption

The equilibrium phosphorus concentration (EPC_0) of sediment represents the aqueous P concentration at which no net sorption or desorption occurs when sediment is suspended in a water sample. When the water column P concentration is less than the sediment EPC, sediments can potentially release P, and when the water column P concentration is greater than the sediment EPC, sediments can potentially serve as a sink for phosphorus.

In the current study, average EPC_0 ranged from 0.033-0.157 mg/L in non-sterilized sediment treatments and 0.047-0.327 mg/L in sterilized treatments; we believe the negative value in the core 2 sterilized treatment is anomalous (Figure 6). Regardless of whether the sediments were sterilized or not, the water column SRP was well below the EPC_0 at every coring location (Figure 6). This suggests that the sediments in these muck fields can serve as a significant source of dissolved P to the water column. Given the high water column chlorophyll concentrations and very high photosynthetic activities (based on DO concentrations), it is likely that rapid uptake of SRP is occurring upon release from the sediments. Unlike water column and sediment TP (Figure 4, 5), EPC_0 was not significantly different between the north and south fields.

Average EPC_0 was significantly higher in the sterilized (0.150 mg/L) than non-sterilized (0.079 mg/L) treatments (Figure 7), reflecting a greater ability by sediments to sorb P when the microbial community was left intact. This suggests that P sorption in the muck fields is regulated not only by sediment geochemistry, but also by biotic processes (Meyer 1979, Klotz 1988, Lottig and Stanley 2007). However, it is important to recognize that biocides such as chloroform have been shown to increase dissolved P in isotherm determinations, presumably due to the lysis of microbial cells, leading to an over-estimation of the importance of biotic uptake (Graetz and Nair 2000, Lottig and Stanley 2007).

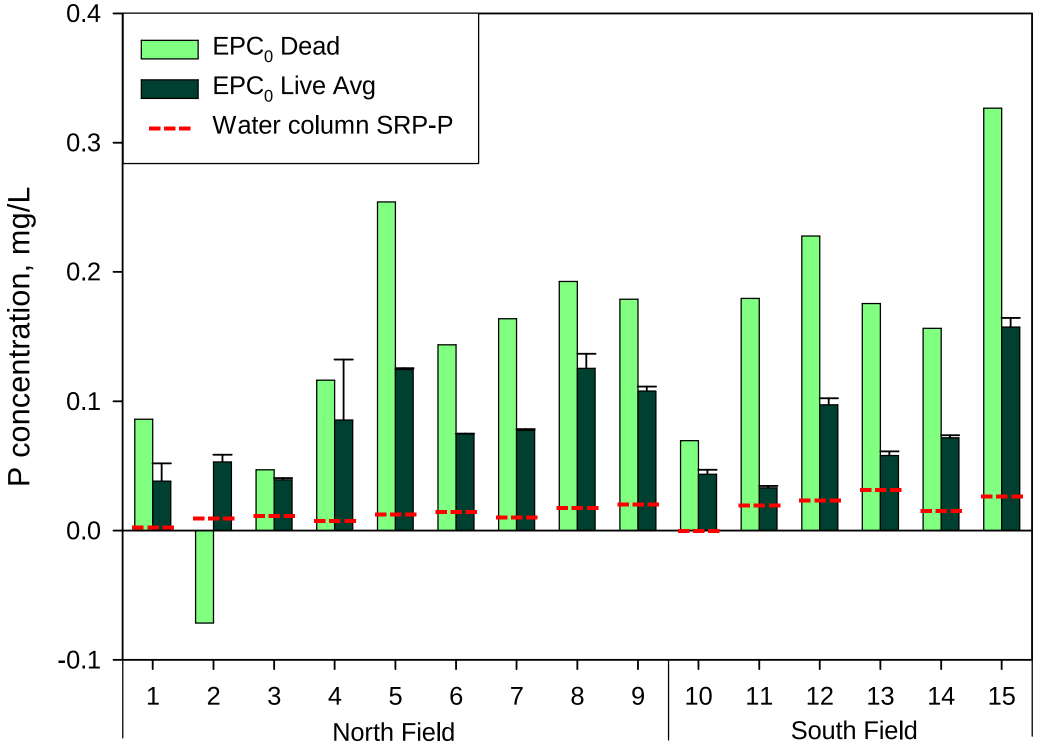


Figure 6: Equilibrium phosphorus concentration (EPC$_0$; mg/L) in sterilized (dead) and non-sterilized (live; average ±SE) cores and water column SRP-P concentration (mg/L) at each coring location in the muck fields.

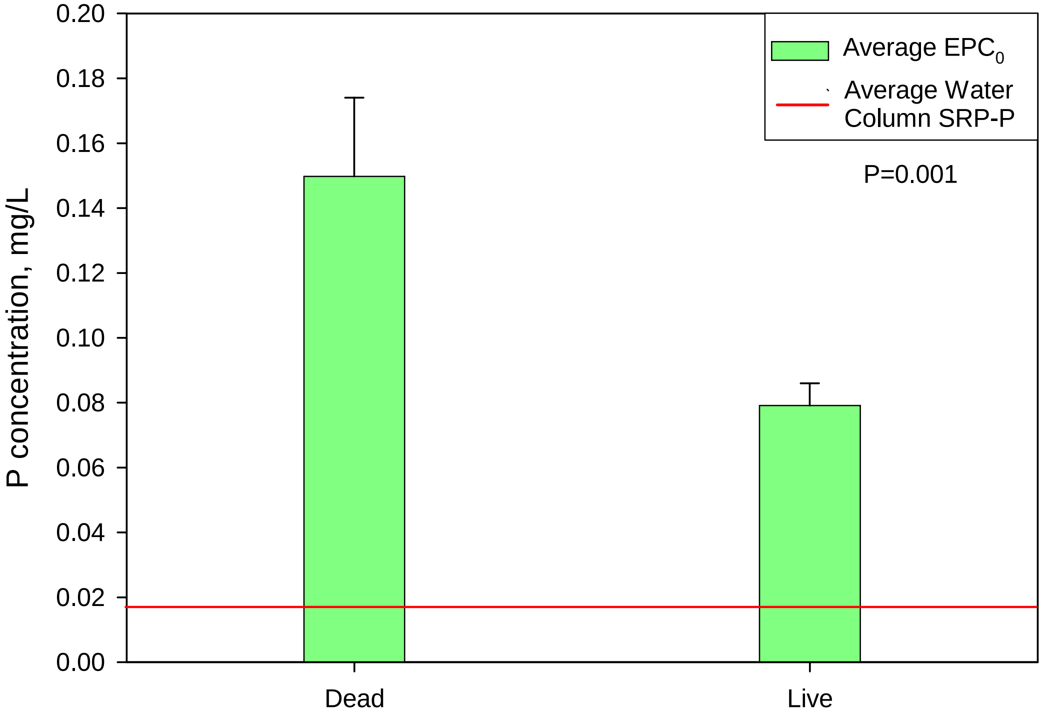


Figure 7: Average (±SE) equilibrium phosphorus concentration (EPC$_0$; mg/L) in sterilized (dead) and non-sterilized (live) treatments. Average water column SRP-P (mg/L) is represented by the red line. P-value indicates significance of dead vs. live treatments as determined using Mann Whitney Rank Sum Test.

Sediment Pore Water

Sediment pore water SRP in the north muck field was greater 1) at the lower site than the upper site (Figure 1) and 2) in August than in June (Figures 8, 9). In June, porewater SRP ranged from 0.024-0.130 mg/L at the upper site and from 0.225-1.650 mg/L at the lower site (Figure 8). In August, porewater SRP ranged from 0.003-1.550 mg/L at the upper site and from 0.026-2.950 mg/L at the lower site (Figure 9). These extremely high concentrations of porewater SRP are 2 orders of magnitude greater than what has been measured in Mona Lake (Steinman, unpublished data) and are a strong indication of the potential for these sediments to negatively impact the water quality of the muck fields, as well as Black Creek and Mona Lake. With the exception of the upper site in June, which had little variability with depth, the maximum porewater SRP was generally found at a depth between 13 and 22 cm (Figure 8, 9).

The flocculent nature of the sediments made it difficult to leave the upper cells of the peepers exposed to the water column. This was accomplished only in June, when one cell was 2 cm above the sediment surface (Figure 8). Water column SRP near the sediment-water interface was very high, 0.400 mg/L, which is similar to the water column TP concentration that was measured in the north field (Figure 4). This further suggests that SRP released from the sediments is cycled rapidly into TP through biotic uptake.

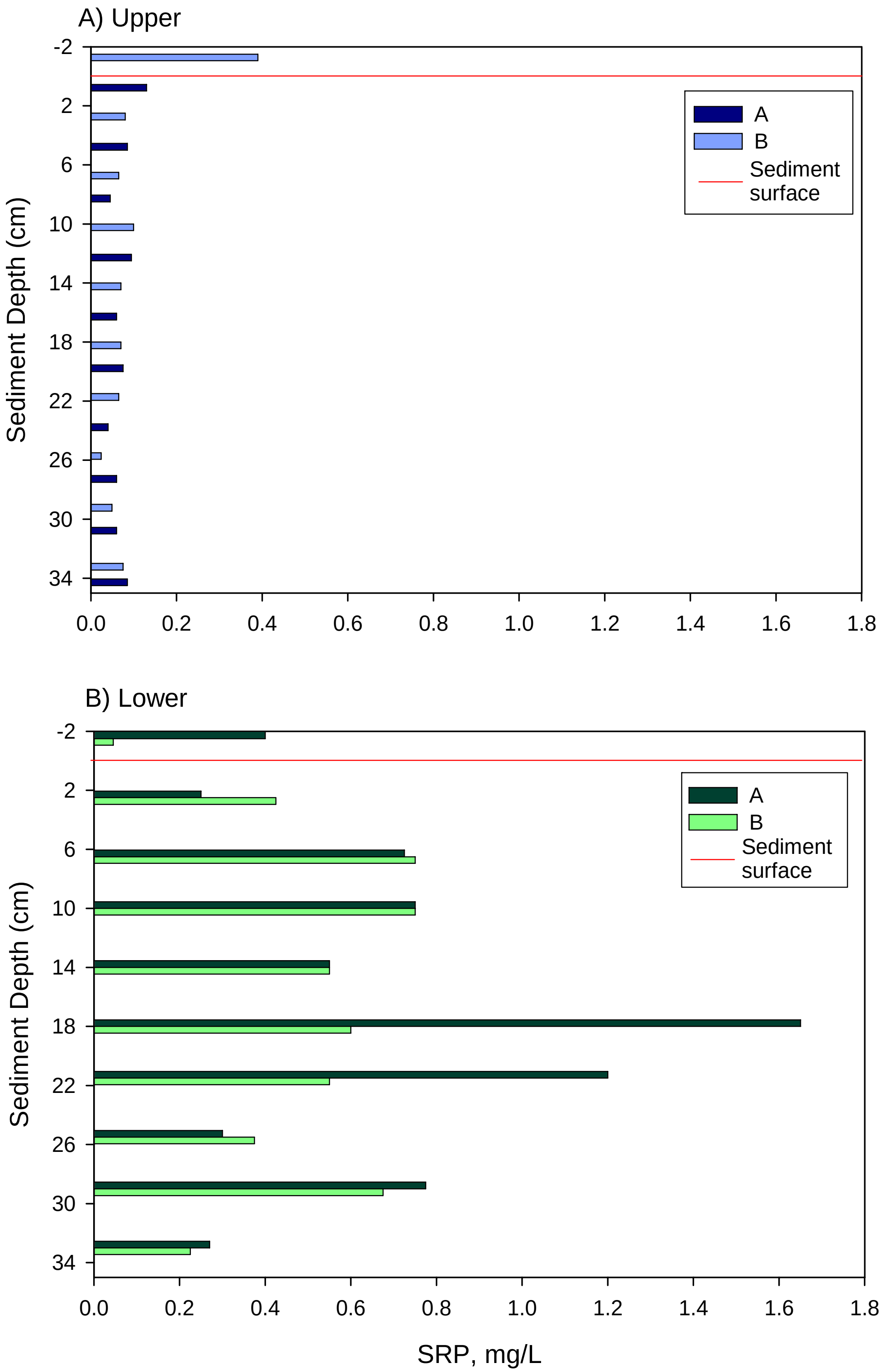
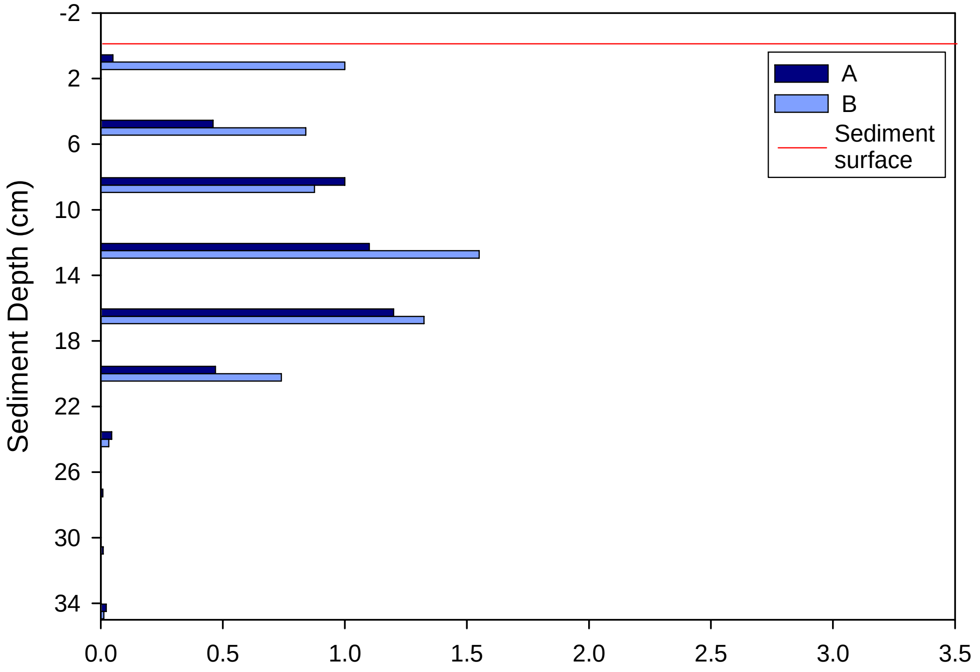


Figure 8: June 2009 sediment pore water SRP-P (mg/L) profiles from the upper and lower sites in the north muck field. Within each figure, the A and B bars represent samples from duplicate peepers.

A) Upper



B) Lower

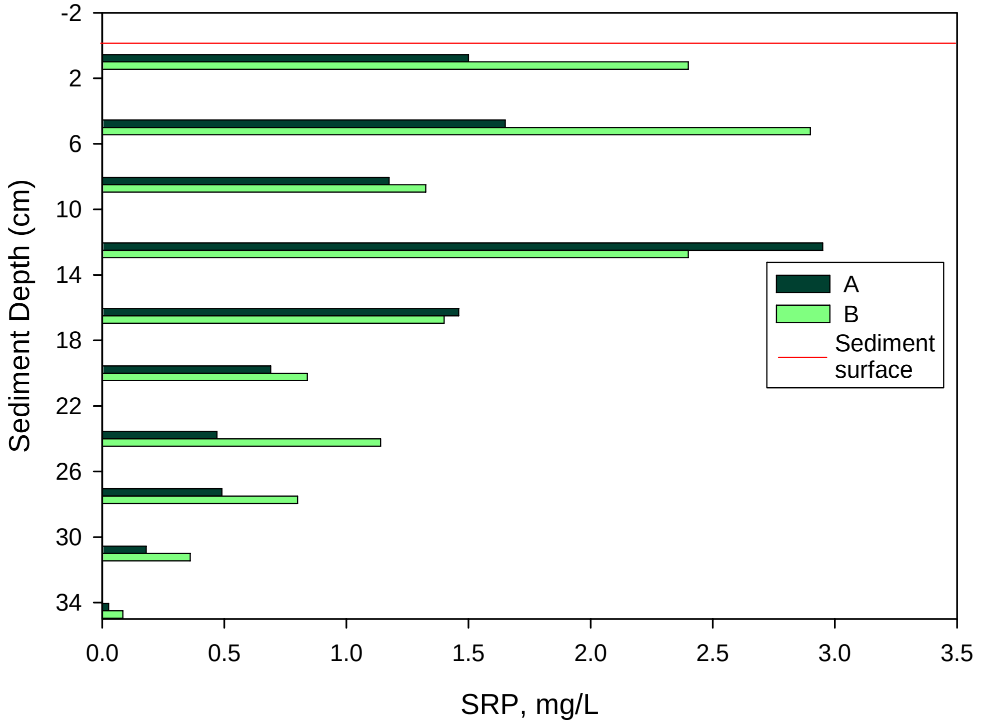


Figure 9: August 2009 sediment pore water SRP-P (mg/L) profiles from the upper and lower sites in the north muck field. Within each figure, the A and B bars represent samples from duplicate peepers.

Summary and Mitigation Strategies

Based on the results of this study, it is clear that the muck fields abutting Black Creek contain an extremely large amount of phosphorus in their sediments that is being released into the water column. Breeches in the levee between the fields and Black Creek allow the phosphorus-laden water from the fields to enter Black Creek and, after traveling only 300 m, Mona Lake. Given the identified need for phosphorus source control in Black Creek, the conclusion that the muck fields are indeed a significant source of phosphorus to the creek has important management implications. However, it is important also to recognize that average TP concentrations upstream of the muck fields were already in the eutrophic range. Therefore, any future mitigation actions in the muck fields should be implemented as part of a comprehensive P source control strategy for the Black Creek watershed that includes upstream sources. Without such a comprehensive approach, the success of mitigation actions in the muck fields to reduce P loading to Mona Lake will be offset by continued upstream P inputs.

The focus of this study was to determine whether the muck fields are a significant source of phosphorus to Black Creek and, consequently, Mona Lake. This was a critical first step to take in order to develop a specific mitigation strategy to address the issue. Although choosing which mitigation approach to implement was beyond the scope of this project, we evaluate four potential approaches (Table 3) that could be used to reduce the flux of P from the muck fields. When considering each option, it is important to keep in mind that costs for sediment P remediation are highly variable and project-specific. Although we included estimated costs where possible, it should be understood that these values are crude estimations and should only be used for relative comparisons among options. Preliminary investigations, as discussed below, are necessary to determine more exact costs.

- Phytoremediation – Aquatic plants are being increasingly used to clean up soil, sediment, and water that are contaminated with nutrients and/or toxic compounds. Phytoremediation is often preferred over physical or chemical engineering approaches because of its lower costs and fewer negative environmental effects (Lu et al. 2010). Vigorously-growing aquatic plants, such as cattail (*Typha* sp.), can be successful at removing excess nutrients from sediments by incorporating nitrogen and phosphorus into their biomass (Xiang et al. 2009).

  Planting costs for phytoremediation have been estimated at ~$10,000 per acre (Schnoor et al. 1995), which translates to $1 million for the 100-acre muck fields. However, this figure is likely an overestimate since this figure is geared more for organic contaminants.

  Several factors should be considered before implementing a phytoremediation plan in the muck fields:

    - Compared to other alternatives, it will likely take longer to see the benefits of phytoremediation (Schnoor et al. 1995). Chemical inactivation and dredging (see below) offer more immediate results.
    - There are recurring costs associated with phytoremediation. The plants must be harvested and disposed of in such a way to ensure the assimilated nutrients are not returned back to the impacted system (Brix 1997). Therefore, it is essential that a long-term source of funding for maintenance be secured.
    - Preliminary studies are necessary to determine which plant species are most appropriate for phytoremediation. Mapping water depth in the muck fields is an

important component of this decision, as deeper water depths will require careful consideration. Cattail (*Typha* sp.) is an aggressive grower in nutrient rich environments, but has depth limitations. *Typha latifolia* is a shallow water species, reaching a maximum density at a water depth of ~22 cm (Grace 1989) and having a maximum depth tolerance of ~68 cm (Grace and Wetzel 1998). *Typha angustifolia* has a greater depth tolerance and reaches maximum density at 60-90 cm (Grace and Wetzel 1998). One possible strategy for the muck fields would be to plant *T. angustifolia* in areas < 1 m deep and submergent vegetation (e.g., *Ceratophyllum;* see Pietro et al. 2006) in areas with deeper depths.

- Chemical inactivation – Chemical treatment with aluminum sulfate (alum), lime, or iron is one of the most common management techniques used to address sediment P release (Cooke et al. 1993). Alum is particularly effective due to its dual mode of action for P removal. Alum reacts with soluble P to form an insoluble precipitate (Stumm and Morgan 1996) and also forms an insoluble aluminum hydroxide floc at pH 6 to 8, which has a high capacity to adsorb large amounts of inorganic P (Kennedy and Cooke 1982). By these two mechanisms, an alum application can irreversibly bind P and inhibit diffusive flux from sediments. Costs of alum application generally range from $500-700 per acre (Holdren et al. 2001, Cooke et al. 2005), which translates to $50,000-$70,000 for the 100-acre muck fields.

  Longevity of alum treatment is dependent on several factors, including the degree to which external loading is controlled and presence of macrophytes. Welch and Cooke (1999) evaluated the effectiveness and longevity of alum applications in 21 lakes across the United States. Their analysis revealed that in polymictic (unstratified) lakes, such as these muck fields, internal loading rate was reduced in 6 out of 9 cases, with average reduction of about 67%, which lasted for 5-11 years. In dimictic (stratified) lakes, internal loading rate was reduced in 7 out of 7 cases, with average reduction of about 80%, and which lasted for 4 to 21 years. The major constraint in polymictic lake application was interference by macrophytes. They concluded that a reasonable expectation of longevity for alum treatments is 10 years in polymictic and 15 years in dimictic lakes.

  Several factors should be considered before implementing a chemical inactivation plan in the muck fields:

  - Bioturbation can limit the effectiveness of chemical treatments, as they are dependent on the floc layer remaining intact on the sediment surface (Cooke et al. 2005). During the project period, we observed a large amount of activity by what appeared to be carp, which are known to be very disruptive to sediments. Further investigation of the fish community, and the degree of bioturbation they may be causing in the fields, would be necessary before engaging in chemical treatment.
  - Because they may experience an increase in the dissolved and toxic aluminate ion during application, systems with high pH (> 9-10) are considered poor candidates for alum treatment (Cooke et al. 2005). Our one-time sampling of conditions in the muck fields revealed pH values ranging from 8.4-10.4, with a mean of 9.4. When pH is a concern, lime and/or alum buffered with lime is an alternative option for treatment (Cooke et al. 2005). There are drawbacks to using lime, however; it is generally not as effective as alum, usually requires multiple applications, and less is known about dose and application techniques than for alum (Cooke et al. 2005).

    - Preliminary studies to identify the pH characteristics of the fields, the appropriate chemical to apply, dosage amount, and P biogeochemistry would be necessary before a chemical inactivation plan could be implemented in the muck fields.

- Sediment removal – Dredging is a highly effective, but expensive, option for sediment P mitigation (Cooke et al. 2005). It is a long-term solution provided external nutrient loads are controlled. However, for many projects the benefits are not substantially greater than for chemical inactivation to justify the additional cost.

  There is wide variability in the estimated costs for dredging. Cooke et al. (2005) report a range from \$2.88/m^3 to \$7.23/m^3. If a continuous 50-cm depth of sediment was removed from the entire area of the muck fields, the cost would range from \$600,000 - \$1.5 million. Sediments that contain toxic contaminants are significantly more costly to remove, possibly exceeding \$52/m^3 (Cooke et al. 2005).

  Several factors should be considered before undertaking dredging in the muck fields:

    - A disposal area would have to be identified and associated costs figured into the costs of dredging. If a 50-cm depth of sediment was removed, the disposal area would need to accommodate ~212,000 m^3 of sediment.
    - Nutrient-rich sediment can have productive uses, such as soil amendments or potting soil. The sale of dredged sediment can significantly help to offset the overall costs of a dredging project (Cooke et al. 2005). Such options should be explored for the muck fields.
    - Preliminary studies are necessary to determine whether the muck field sediment is contaminated beyond nutrients. Such a condition would likely make dredging a cost-prohibitive option.
    - Additional studies are needed to ensure that dredging does not expose sediment that contains high P concentrations, which could be released into the water column, negating at least in part the benefits of dredging. Presumably, a low-P sand layer exists beneath the organic sediment; it would be necessary to determine the depth to reach that layer, and based on that information, calculate the appropriate depth to dredge, sediment volume involved in dredging, and the associated costs.

- Levee maintenance – Regardless of which strategy(ies) is deemed appropriate, one management practice that we believe deserves immediate attention is the closing of the breeches in the levee. We did not develop a cost-estimate for this action but it is likely to be low-cost relative to the other options outlined above.

  Several factors should be considered before securing the breeches in the levees:

    - It is likely that closing off the levees will not stop all P from reaching Black Creek, as some flow (and associated P) may still move from the muck fields to Black Creek through subsurface pathways.
    - It is important to better understand the water budget for the muck fields to ensure the closing of the breeches does not result in flooding of riparian lands.

Table 3. Summary table of mitigation approaches, estimated costs, and uncertainties.

Approach	Estimated Cost*	Uncertainty
Phytoremediation	$1,000,000**	- Requires ongoing harvest and disposal - Plantings depend on water depth
Chemical Inactivation	$50,000 – $70,000	- Length of treatment variable - Permitting required
Dredging	$600,000 -- $1.5 M	- Assumes nutrients are only contaminants - Permitting required
Levee Maintenance	Unknown	- Must not result in flooding

*Assumes mitigation involves 100 acres and includes cost of initial implementation only; does not include ongoing maintenance costs

****Based on single study; cost may be inflated

Acknowledgements

The authors gratefully acknowledge the financial support of the Michigan Department of Natural Resources and Environment, and the support of project manager Matthew Wesener. Field and laboratory support were provided by Maggie Weinert, Brian Scull, Betsy Shafer, Kelli Johnson, Kaitlyn Driza, Angela Defore, Alex Wieten, and Jordan Allison. We also thank Jeff Basch, Pam Babbitt, and Chuck Alstrom for access to the muck fields from their property.

References

APHA. 1992. Standard Methods for the Examination of Water and Wastewater. 18th Edition. American Public Health Association.

Brix, H. 1997. Do macrophytes play a role in constructed treatment wetlands? Water Science and Technology 35: 11-17.

Chu, X. 2006. Windows-Based Hydrol-Inf User's Manual, Version 2.03. Annis Water Resources Institute, Grand Valley State University.

Cooke, G.D., E.B. Welch, S.A. Peterson, and P.R. Newroth. 1993. Restoration and management of lakes and reservoirs. Lewis Publ., Boca Raton, FL.

Cooke, G.D., E.B. Welch, S.A. Peterson, and S.A. Nichols. 2005. Restoration and management of lakes and reservoirs, third edition. CRC Press, Boca Raton, FL.

Davis, W.P. and A.D. Steinman. 1998. A lightweight, inexpensive benthic core sampler for use in shallow water. Journal of Freshwater Ecology 13: 475-479.

Dosskey, M.G. 2001. Toward quantifying water pollution abatement in response to installing buffers on crop land. Environmental Management 28: 577-598.

Grace, J.B. 1989. Effects of water depth on *Typhya latifolia* and *Typha domingenisis*. American Journal of Botany 76: 762-768.

Grace, J.B. and R.G. Wetzel. 1998. Long-term dynamics of *Typha* populations. Aquatic Botany 61: 137-146.

Graetz, D.A., and V.D. Nair. 2000. Phosphorus sorption isotherm determination. *In* G.M. Pierzynski (ed.) Methods of phosphorus analysis for soils, sediments, residuals, and waters. Southern Coop. Ser. Bull. 396. Kansas State Univ., Manhattan.

Holdren, C., W. Jones, and J. Taggart. 2001. Managing lakes and reservoirs. North American Lake Management Society and Terrene Institute, in cooperation with the Office of Water, Assessment and Watershed Protection Division, U.S. Environmental Protection Agency, Madison, WI.

Hong, Y., Steinman, A.D., Biddanda, B., Rediske, R., and G. Fahnenstiel. 2006. Occurrence of the toxin-producing cyanobacterium *Cylindrospermopsis raciborskii* in Mona and Muskegon Lakes, Michigan. Journal of Great Lakes Research 32: 645-652.

Kennedy, R.H., and D.G. Cooke. 1982. Control of lake phosphorus with aluminum sulfate. Dose determination and application techniques. Water Resources Bulletin 18:389–395.

Klotz, R.L. 1988. Sediment control of soluble reactive phosphorus and factors in stream water and sediments in Hoxie Gorge Creek, New York. Canadian Journal of Fisheries and Aquatic Sciences 45: 2026-2034.

Lottig, N.R. and E.H. Stanley. 2007. Benthic sediment influence on dissolved phosphorus concentrations in a headwater stream. Biogeochemistry 84: 297-309.

Lu, Q., Z.L. He, D.A. Graetz, P.J. Stoffella, and X. Yang. 2010. Phytoremediation to remove nutrients and improve eutrophic stormwaters using water lettuce (*Pistia stratiotes* L.). Environmental Science and Pollution Research 17: 84-96.

Meyer, J.L. 1979. The role of sediments and bryophytes in phosphorus dynamics in a headwater stream ecosystem. Limnology and Oceanography 24: 365-375.

Mozaffari, M. and J.T. Sims. 1994. Phosphorus availability and sorption in an Atlantic Coastal Plain watershed dominated by animal-based agriculture. Soil Science 157: 97-107.

Novak, J.M., K.C. Stone, A.A. Szogi, D.W. Watts, and M.H. Johnson. 2004. Dissolved phosphorus retention and release from a coastal plain in-stream wetland. Journal of Environmental Quality 33: 394-401.

Pant, H.K. and K.R. Reddy. 2001. Phosphorus sorption characteristics of estuarine sediments under different redox conditions. Journal of Environmental Quality 30: 1474-1480.

Pietro, K.C., M.J. Chimney, and A.D. Steinman. 2006. Phosphorus uptake by the *Ceratophyllum*/periphyton complex in a south Florida freshwater marsh. Ecological Engineering 27: 290-300.

Reddy, K.R., R.H. Kadlec, E. Flaig, and P.M. Gale. 1999. Phosphorus retention in streams and wetlands: A review. Critical Reviews in Environmental Science and Technology 29: 83-146.

Richardson, C.J. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. Science 228: 1424-1427.

Schnoor J.L., L.A. Licht, S.C. McChutcheon, N.L. Wolfe, and L.H. Carreira. 1995. Phytoremediation of organic and nutrient contaminants. Environmental Science and Technology 29: 318-323.

Steinman, A.D., X. Chu, and M. Ogdahl. 2009. Spatial and temporal variability of internal and external phosphorus loads in an urbanizing watershed. Aquatic Ecology 43: 1-18.

Steinman, A.D., R. Rediske, X. Chu, R. Denning, L. Nemeth, D. Uzarski, B. Biddanda, and M. Luttenton. 2006. An environmental assessment of an impacted, urbanized watershed: the Mona Lake Watershed, Michigan. Archiv für Hydrobiologie 166: 117-144.

Stumm, W., and J.J. Morgan. 1996. Aquatic Chemistry, 3rd ed. John Wiley & Sons, Inc., New York.

Welch, E.B. and G.D. Cooke. 1999. Effectiveness and longevity of phosphorus inactivation with alum. Journal of Lake and Reservoir Management 15: 5-27.

Xiang, W., Y. Xiao-E, Z. Rengel. 2009. Phytoremediation facilitates removal of nitrogen and phosphorus from eutrophicated water and release from sediment. Environmental Monitoring and Assessment 157: 277-285.