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Does Phosphorus from Agricultural Tile Drains Fuel Algal Blooms?

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Abstract

Phosphorus (P) is often implicated as a contributing factor to algal blooms. Attention has been focused on P in surface runoff, but agricultural tile drains also can be a source. Lake Macatawa is a hypereutrophic lake located in west Michigan, and the watershed is dominated by row crop agriculture. Further research is needed to understand the influence of bioavailable P originating from tile drains on water quality in Lake Macatawa. The objectives of this study were to 1) conduct a tile drain effluent sampling survey to assess their importance as a source of P in the Macatawa Watershed; 2) investigate the change in tile drain P concentrations spatially and temporally over a one-year period; and 3) use growth chamber algal bioassays and the ratio of soluble reactive phosphorus to total phosphorus (SRP:TP) to assess tile drain P bioavailability. During March 2015 – February 2016, P concentrations varied significantly among sample sites, and the highest P loads occurred during the non-growing season. The SRP:TP ratio measured at the tile drain outlets had a positive correlation with acreage drained by the tile system. Four of six bioassays resulted in a positive relationship between SRP and algal growth, but results from only one bioassay were statistically significant. There was a clear change in the algal community structure when incubated in tile drain water, and dominance was by diatoms, not cyanobacteria as expected. Based on these results, there is a need to quantify the tile drained area in the Macatawa watershed and manage for high P loads during the non-growing season.

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Abbreviations

SRP, soluble reactive phosphorus; TP, total phosphorus; TMDL, Total Maximum Daily Load; NPDES, National Pollutant Discharge Elimination System; chl-*a*, chlorophyll-*a*; HAB, harmful algal bloom; Fe, iron; Al, aluminum

Chapter I

Nonpoint vs. Point Source Pollution

Several well-known environmental disasters of the 1960's including Lake Erie nuisance algal blooms (Dolan 1993) and the Cuyahoga River fires (Tuckerman and Zawiski 2007) prompted stricter control on pollutant discharge to US waterways. Among growing public concern for surface water quality, the federal Clean Water Act was passed in 1972 to regulate pollution sources (USEPA 2016b). The Act targeted wastewater effluent from industrial processes, and it became illegal to discharge pollutants via discrete, point sources such as man-made ditches or pipes into navigable waterways without a permit. Overseen by the US Environmental Protection Agency (EPA), the National Pollutant Discharge Elimination System (NPDES) regulates the permits and has effectively reduced pollution from point sources (USEPA 2016b).

However, pollution also travels to water bodies via precipitation-induced runoff. In contrast to point sources, nonpoint pollution originates from non-specific, diffuse sources (USEPA 2016a). The quantity and type of non-point pollution is influenced by land use. For instance, agricultural operations are the primary nonpoint source of pollution impacting US streams, the second leading source to wetlands, and the third leading source to lakes (USEPA 2016a). When a water body is too degraded to meet water quality standards, it is listed under section 303(d) of the Clean Water Act (USEPA 2016c). After listing, specific pollutants are identified and a plan to limit pollutant loads to the waterway is developed. This type of plan, called a Total Maximum Daily Load (TMDL), identifies both point and nonpoint pollution sources (USEPA 2016c), and allocates load reductions to each of the sources. Relative to point sources, measuring and regulating nonpoint sources presents a significant challenge to addressing degraded waterways in the US. Unlike point sources of pollution, nonpoint sources are not permitted and the assumption is that once best management practices are implemented to address pollutants, load reductions will occur.

Phosphorus & Eutrophication

An often-limiting nutrient for plant growth, phosphorus (P) is frequently implicated as a contributing factor to algal blooms (Daniel et al. 1998). P occurs naturally in the environment in the form of phosphate rock deposits. Under most natural conditions, the amount of dissolved phosphorus in fresh water that comes from weathered rock formations is in low concentrations relative to the amount needed for growth by algae. However, industrial and agricultural activities have modified the global P cycle by mining phosphate rock and redistributing it as P-based fertilizer (Gilbert 2009). As autotrophs, algae require both P for DNA synthesis and nitrogen (N) for proteins, so excess supply of these nutrients can trigger blooms of algae (Conley et al. 2009; Elser et al. 2007). Classic empirical studies have demonstrated that P in particular controls algal growth in freshwater systems (Schindler et al. 1997). P and N transport via agricultural and urban runoff are major sources of nutrients to freshwater lakes and streams and cause degradation of water quality, including algal blooms (Carpenter et al. 1998).

Blooms of algae can impair aquatic ecosystems because they can: 1) be toxic; 2) decrease dissolved oxygen concentrations upon mineralization; and 3) disrupt food webs (Conley et al. 2009). Addressing point sources of P through the NPDES permit system initially reduced P loads to water bodies. For example, Lake Erie algal blooms decreased after establishing the Clean Water Act (Makarewicz 1993). The annual total phosphorus (TP) load to Lake Erie was reduced from 25,000 metric tons to the target load of 11,000 metric tons during the 1980s. However, the more bioavailable, dissolved form of P load to the lake has increased since the mid-1990s (Scavia et al. 2014). Nonpoint P inputs, especially from agriculture, are still a major source of nutrients to Lake Erie, and algal blooms are again a major water quality issue (Michalak et al. 2013). In fact, agricultural runoff is a key nonpoint source of nutrients in much of the Great Lakes basin (Danz et al. 2007).

The Great Lakes are located within the US Midwest where millions of hectares of swamplands were converted to highly productive agricultural lands during the initial installation of subsurface tile drains (King et al. 2015). Historically made of clay tile, these drainage systems are pipes installed in the soil column beneath farm fields to lower the water table and prevent crops from drowning (Fausey et al 1987). An estimated 18 to 28 million hectares of cropland in the Midwest region are managed with the use of tile drains (King et al. 2015). Tile drains change the hydrology of a field to increase infiltration of water and reduce the amount of overland runoff (Reid et al. 2012). Decreasing surface runoff by increasing infiltration is assumed to also reduce the loss of P via topsoil erosion. However, the tile drains represent a direct conduit from the field to the outlet, and from there directly into the bordering ditch, so nutrients that reach the tile can be carried from a much larger area of the landscape than would otherwise be possible (Smith et al. 2015; Reid et al. 2012). Traditionally, P leaching to tile drains in agricultural fields was not considered a contributor to total phosphorus (TP) export in a watershed (Eastman et al. 2010; Sims et al. 1998). Indeed, overland flow is generally the dominant transport mechanism for P, but there are situations when significant P transport has occurred through agricultural tile drainage (King et al. 2015). However, the amount of P that reaches the tiles cannot be predicted by a single parameter.

Phosphorus Transport to Tile Drains

Factors influencing P transport to tile drains include soil type, precipitation, time of year, and land management factors such as tillage or crop regime (Fig 1). Soil P saturation, chemical reduction or oxidation, and drain depth or spacing also can affect transport (King et al. 2015), but these additional factors were not a focus of this study. Soil type influences P transport to tile drains primarily by its tendency to form macropores or to promote matrix flow. Soil matrix flow is a relatively slow pathway by which solutes have time to interact with soil particles, minerals, and organic materials (Reid et al. 2012; Sharpley et al. 2001). Alternatively, P transport through soil macropores is a faster and more direct pathway via earthworm burrows, shrinkage fractures, or channels from plant roots. Macropores provide a significant route for both dissolved and particulate P to artificial drains (Tan and Zhang 2011). Macropores serve as a transport mechanism more frequently in clay soils than sandy soils, and conversely, sandy soils promote matrix flow.

Preferential transport through macropores is an important process during precipitation and snowmelt, as both processes cause increased flow through the soil column (Macrae et al. 2007). Numerous studies have demonstrated that periods of high flow result in increased P loss through tile drainage, as well (Algoazany et al. 2007; Ball Coelho et al. 2012; Gentry et al. 2007; Morrison et al. 2013). There is a pulse of P export from an agricultural watershed during a high flow event, and a review of the literature shows that tile drains are a contributing source to this pulse (King et al. 2015). Furthermore, P speciation is affected by rain storm characteristics. Vidon and Cuadra (2011) found soluble reactive phosphorus (SRP) to be significantly higher during larger storms with more bulk precipitation, yet increases in TP were not statistically significant. SRP is often used as a proxy to quantify the dissolved, bioavailable fraction of TP.

More importantly, although dissolved P generally composes less than 50% of the TP exported from tile drains, it may have a disproportionate influence on biological response because of its bioavailability.

Seasonality of tile flow is yet another factor to consider in P transport through drains. Several studies found the majority of nutrient leaching to occur during the winter months corresponding with increased flow from tile drains (Kladivko et al. 2004; Laubel et al. 1999; Royer et al. 2006). Even if fertilizer and manure are applied just before and during the growing season, snowmelt and storm events during the winter are still able to initiate P movement to tile drains (Macrae et al. 2007). Other studies have found that the greatest P loss through subsurface drains happens during the spring (Vidon and Cuadra 2011). When P fertilizer or manure is applied to bare ground lacking crops in the spring, there is a higher risk for P loss to artificial drainage (Eastman et al. 2010; Kinley et al. 2007). Typically, tile drains cease to flow during the summer months except following rain events (King et al. 2015).

A large variety of land management factors can affect P transport to tile drains including type of fertilizer, tillage, and crop regime. First, long-term application of manure or fertilizer increases the risk of leaching as water moves through the P-saturated soil (Laubel et al. 1999; King et al. 2015). Several studies agree that use of manure over inorganic fertilizer results in more P loss to tile drains, which can supersede the effect of soil texture and may be because organic P is less strongly sorbed to soil particles in comparison to inorganic P (Kinley et al. 2007; North 2013; Sims et al. 1998). Second, both tillage and crop cover influence movement of P through the soil column (Kinley et al. 2007). No-tillage management decreases evaporation, increases soil permeability, and most importantly decreases runoff and erosion. Conversely, tillage disturbs macropores in the topsoil, which reduces the connectivity of the surface to tile drains (Geohring et al. 2001; King et al. 2015). Finally, during the growing season, row crop agriculture is known for higher nutrient input to freshwater systems than low-intensity pastures for livestock (North 2013). Regardless of crop regime, the presence of plant roots creates channels in the soil promoting macropore flow, so the use of winter cover crops promotes macropore formation by root channels even during the non-growing season (Vidon and Cuadra 2011). In conclusion, the loss of P through tile drains in agricultural fields is both temporally and spatially variable, and depends heavily on local factors. All of the factors described above potentially influence P movement to drains. Because tile drains are a direct connection from field to stream, this study was designed to investigate their role in nutrient transport, especially as a source of bioavailable P leading to eutrophication in the Great Lakes region.

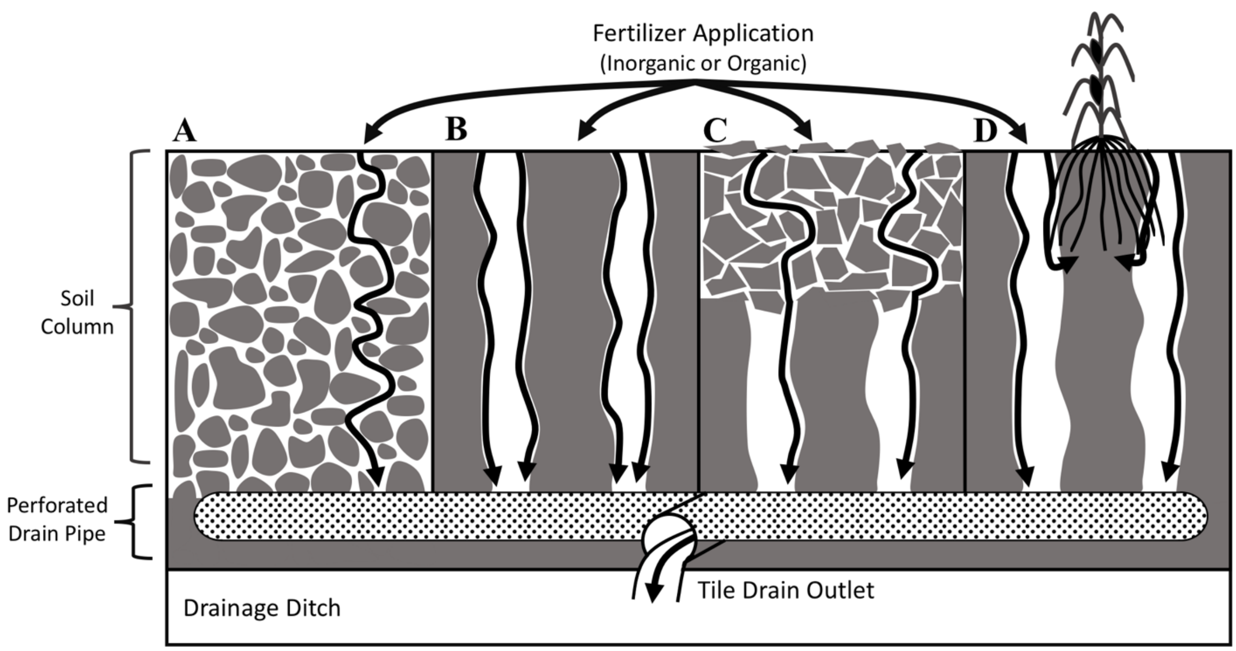

Figure 1. Vertical cross-section of P transport to agricultural tile drains with A) sandy soil promoting matrix flow; B) clay soil promoting preferential flow via macropores; C) tilled soil with disturbed macropores; and D) uptake of nutrients by crops. Diagram is not to scale.

Study Site

This study took place in the Macatawa Watershed, which is located in southwestern Michigan, and encompasses 450 km^2. Nearly 90% of the area's wetlands in historic Holland, MI were lost to development when European settlers drained the soil water and straightened tributaries (Hope College 2012). The resulting increased flow of runoff water has led to widespread sediment erosion into Lake Macatawa and associated nutrient-rich conditions throughout the watershed. Previous stream sediment and nutrient monitoring has shown that excess nutrients, including P, originate in the outer sub-basins of the watershed, which are dominated by agricultural fields of row crops (MWP 2012).

Average monthly TP concentrations in Lake Macatawa have exceeded 125µg/L in recent years and at times were more than 200µg/L (Holden 2014). The lake and its tributaries are on Michigan's 303(d) list for not attaining water quality standards for warm water fishery and other aquatic life, and the lake is currently under a TMDL calling for a reduction in TP concentration of 72% (Walterhouse 1999). The relationship between flow and P concentrations in the lake shows that the water quality of Lake Macatawa is influenced by nonpoint sources of pollution, including agricultural areas during periods of higher flow caused by rain events. Conversely, P concentrations in Lake Macatawa are lower after long periods of baseflow (Holden 2014). A current 10-year restoration project aims to reduce P loads and meet the TMDL target through remediation and implementation of best management practices (MWP 2012).

Objectives

Lake Macatawa is an important recreational and commercial port for the City of Holland, MI. Eutrophication, and associated harmful algal blooms (HABs), negatively impact the region's economy and cultural identity. An overarching goal of this study is to identify the role of tile drain runoff in stimulating algal blooms; this information is intended to improve management practices, which in turn will lead to improved water quality in both the Macatawa Watershed and Lake Macatawa. As a consequence, economic activity, recreation, and community pride will be enhanced.

Given the necessity to dramatically reduce P concentrations in the Macatawa Watershed, further research is needed at the individual field level to understand the influence of bioavailable forms of P originating from tile drains. The objectives of my study are to: 1) conduct a tile drain effluent sampling survey to assess the importance of tile drains as a source of P in the Macatawa Watershed; 2) use bioassays and the ratio of SRP:TP to measure the bioavailability of P found within the tile drains; and 3) investigate the change in tile drain P concentrations over a one-year period. The results will help inform the ongoing restoration project in the watershed. Although this study is focused on a particular watershed in Michigan, the results will contribute to the wider body of research regarding phosphorus as a cause of eutrophication.

In addition, as HABs continue to plague the Great Lakes region, it is critical that we better understand the mechanisms driving these blooms. Phosphorus loss via tile drains correlates with elevated flow. According to most general circulation models of future climate, there will be an increase in the intensity of storm events, as well as their frequency, in the Great Lakes region (Hayhoe et al. 2010). This may cause an increase in phosphorus loss via tile drains, especially in concentrated agricultural areas. Therefore, a complete understanding about the sources of phosphorus and the subsequent progression to eutrophication and HABs is crucial to protecting freshwater quality in the near future.

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Chapter II

Does Phosphorus from Agricultural Tile Drains Fuel Algal Blooms?

Introduction

Among growing public concern for surface water quality, the federal Clean Water Act was enacted in 1972 to regulate pollution sources (USEPA 2016). Regulation of discharge permits under the National Pollutant Discharge Elimination System (NPDES) has effectively reduced pollution from point sources (USEPA 2016). However, inputs from agriculture are still a major nonpoint source of nutrient pollution to surface waters (Daniel et al. 1998). An often-limiting nutrient in freshwater ecosystems, excess P is frequently implicated as a contributing factor to algal blooms (Elser et al. 2007; Schindler et al. 1977). Blooms of algae can impair aquatic ecosystems because they can: 1) be toxic (Carmichael 2001); 2) decrease dissolved oxygen concentrations upon mineralization (Scavia et al. 2014); and 3) disrupt food webs (Conley et al. 2009).

An estimated 18 to 28 million hectares of cropland in the Midwest region are managed with the use of tile drains (King et al. 2015a). When tile drains are installed, they change the hydrology of a field to increase infiltration of water and reduce the amount of overland runoff (Reid et al. 2012). While tile drains can reduce surface runoff, and thereby the loss of P via topsoil erosion, they also represent a direct conduit from the field to the outlet and bordering ditch. Hence, nutrients reaching the surface drainage system not only can derive from an expanded area of the landscape (Smith et al. 2015; Reid et al. 2012), they also reach the surface waters without the opportunity for assimilation or adsorption through the soil profile. Overland flow is generally the dominant transport mechanism for P, but there are situations when significant P transport has occurred through agricultural tile drainage (King et al. 2015a).

Factors influencing P transport to tile drains include soil type, precipitation, time of year, and land management practices such as tillage or crop regime (King et al. 2015a). Soil type influences P transport to tile drains primarily by its tendency to form macropores or to promote matrix flow. Soil matrix flow is a relatively slow pathway by which solutes have time to interact with soil particles, minerals, and organic materials (Reid et al. 2012; Sharpley et al. 2001). Alternatively, P transport through soil macropores is a relatively fast, more direct pathway via earthworm burrows, shrinkage fractures, or root channels (Laubel et al. 1999). Macropores provide a route for substantial amounts of both dissolved and particulate P to artificial drains (Tan and Zhang, 2011). Macropores serve as a transport pathway more frequently in clay soils than sandy soils, and conversely, sandy soils promote matrix flow (King et al. 2015a).

Preferential transport through macropores is an important process during precipitation and snowmelt, as both events cause increased flow through the soil column (Macrae et al. 2007; Smith et al. 2015). Numerous studies have demonstrated that periods of high flow result in increased P loss through tile drainage (Algoazany et al. 2007; Ball Coelho et al. 2012; Gentry et al. 2007; Morrison et al. 2013). There is a pulse of P export from an agricultural watershed during a high flow event, and tile drains can be a contributing source to this pulse (Lam et al. 2016; King et al. 2015a). Even if fertilizer and manure are applied to a field primarily during the growing season, snowmelt and storm events during the winter are still able to initiate P movement to tile drains (Macrae et al. 2007). Typically, tile drains cease to flow during the summer months except following rain events, and several studies found the majority of nutrient leaching to occur during the winter months with increased flow from tile drains (Kladivko et al. 2004; Laubel et al. 1999; Royer et al. 2006).

In addition to climatic variables, a variety of field-specific management practices can affect P transport to tile drains including type of fertilizer, tillage, and crop regime. First, long-term application of manure or fertilizer increases the risk of leaching as water moves through the P-saturated soil (Laubel et al. 1999; King et al. 2015a). Several studies agree that use of manure over inorganic fertilizer results in more P loss to tile drains, which may be because organic P is less strongly sorbed to soil particles in comparison to inorganic P (Kinley et al. 2007; North 2013; Sims et al. 1998). Second, both tillage and crop cover influence movement of P through the soil column (Kinley et al. 2007). No-tillage management decreases runoff and erosion, but promotes macropore formation. Conversely, tillage disturbs macropores in the topsoil, which reduces the connectivity of the surface to tile drains (Geohring et al. 2001; King et al. 2015a). Finally, during the growing season, row crop agriculture is known for higher nutrient input to freshwater systems than low-intensity pastures for livestock (North 2013). Regardless of crop regime, the presence of plant roots creates channels in the soil promoting macropore flow, so the use of winter cover crops promotes macropore formation even during the non-growing season (Vidon and Cuadra 2011). In conclusion, the loss of P through tile drains in agricultural fields is both temporally and spatially variable, and depends heavily on local factors.

All of the factors described above potentially influence P movement to drains. Because tile drains are a potentially important source of nutrients from the field to downstream receiving water bodies, this study was designed to investigate their role as a source of bioavailable P contributing to eutrophication in the Great Lakes region. We examined the spatial and temporal variability of P in tile drain effluent, and its ecological impacts in the Lake Macatawa watershed. This agriculturally-dominated watershed is located in west Michigan, and drains into Lake Macatawa, an important recreational and commercial port for Holland, MI. Excess sediment and phosphorus negatively impact the region's economy and cultural identity. Therefore, the objectives of this study were to: 1) conduct a tile drain sampling survey to assess the importance of effluent as a source of P in the Macatawa Watershed; 2) investigate the change in tile drain P concentrations spatially and temporally over a one-year period; and 3) use algal bioassays to measure the bioavailability of P found within the tile drains.

Methods

Study Site

The Macatawa Watershed encompasses 450 km^2 in southwestern Michigan; the outer regions of the watershed are dominated by row-crop agriculture. This type of land use has increased the flow of runoff water, led to widespread sediment erosion into Lake Macatawa, and caused nutrient-rich conditions throughout the watershed. Previous stream sediment and nutrient monitoring has shown that excess nutrients, including P, originate in the watershed's outer sub-basins (MWP 2012). Lake Macatawa is hyper-eutrophic, and average monthly TP concentrations have exceeded 125µg/L in recent years (Holden 2014). The lake and its tributaries are on Michigan's 303(d) list for not attaining water quality standards for warm water fishery and other aquatic life, and the lake is currently under a total maximum daily load (TMDL) calling for a 72% reduction in TP concentration (Walterhouse 1999). P loading to Lake Macatawa is heavily influenced by precipitation events, and lake concentrations of TP are lower after long periods of baseflow (Holden 2014). A current 10-year restoration project aims to reduce P loads and meet the TMDL target through remediation and implementation of best management practices (PC 2016).

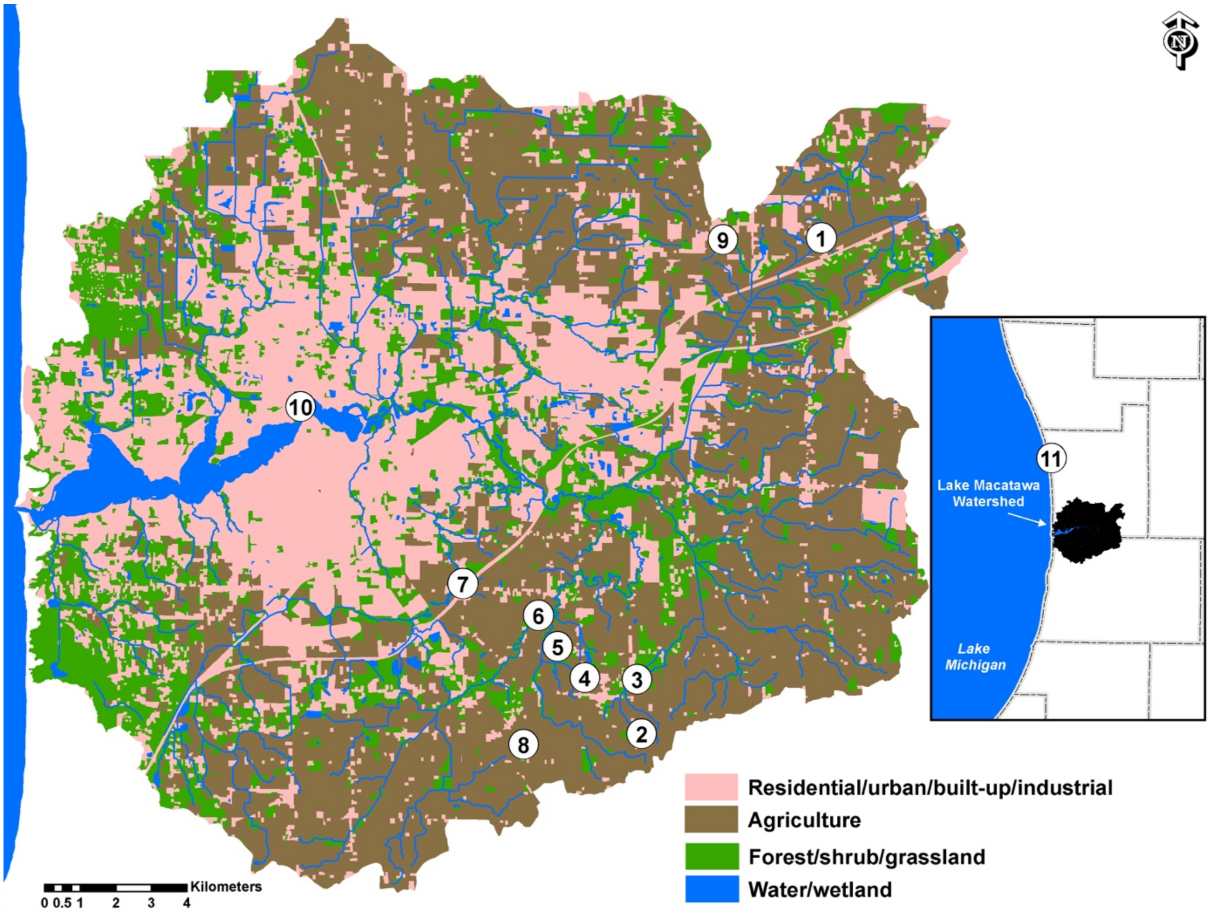

Figure 1. Lake Macatawa watershed boundary showing land use distribution, location of the tile drain sampling sites (circles; 1-9), as well as Lake Macatawa sample site (10). See inset for location of the Lake Michigan sample site (11). Inset: location of watershed in west Michigan.

Tile Drains

Our original intent was to sample tile drains distributed throughout the watershed, but limited cooperation from agricultural producers resulted in the identification of nine tile drain sampling sites (Figure 1). Because the property owners wished to remain anonymous, the exact coordinates of each site are not given. Eight of the sites were PVC outlets draining to a ditch, and at site 9, plastic tubing (acid-washed) was used to hand-pump water from a depth of ~3m to the surface from a tile drain vent. Site 9 was added to the study in June, 2015. Total phosphorus (TP) and soluble reactive phosphorus (SRP) were measured monthly for one year (March 2015-

February 2016) from each site; grab samples were obtained using acid-washed 250mL plastic bottles. A 20-mL subsample was filtered through a 0.45-µm nylon syringe filter (ThermoFisher Scientific, Waltham, MA) immediately after sampling for SRP analysis. All samples were transported to the laboratory on ice. TP samples were stored at 4°C, and SRP samples were frozen at -18°C before analysis with a SEAL Autoanalyzer (SEAL Analytical, Mequon, Wisconsin). The ratio of SRP (mg P/L) to TP (mg P/L) (SRP:TP) for each tile drain was calculated and is referred to as %SRP.

Discharge at each outlet was estimated using a 500mL cup and stopwatch during sample collection. Some tile outlets were partially submerged by ditch water on a few sample dates, but active flow through the outlet allowed for grab samples and P concentration analysis. In those cases, discharge could not be measured from partially submerged outlets, so flow was estimated based on other non-submerged visually similar sampling dates. SRP and TP load were calculated by multiplying concentration values by the corresponding discharge rate. Finally, the land management factors of each tile-drained field were recorded (Table 3). Factors included acres drained, summer and winter crop regime, fertilizer application, tillage equipment, and dominant hydrologic soil type (USDA SSURGO).

Bioassay Setup

Algal bioassays were used to evaluate the bioavailability of P in tile drain water. Three separate bioassays were conducted to assess seasonal effects: spring (April 23), summer (August 5), and fall (October 26) of 2015. Acid-washed 10L carboys were used to collect water for the experiments. Carboys of water also were collected from Lake Macatawa and from Lake Michigan (Figure 1) to serve as controls as explained below. Carboys were transported to the lab on ice and stored at 4°C. Within 24 hours, tile drain water and lake water samples were filtered through 0.1-μm Graver QMC™ Series Filter Cartridges using a peristaltic pump. Filtering by this method did not affect SRP concentrations of the tile or lake water. Prior to bioassay setup, water from each carboy was filtered into a 20mL vial using a 0.45-μm glass membrane for SRP analysis.

Two types of bioassays were run simultaneously each season in a Powers Scientific Growth Chamber. The first type of bioassay used commercially purchased *Selenastrum sp.* (Carolina Biological®)*,* a green alga, while the second type of bioassay used phytoplankton collected from the surface (0-0.5m depth) of Lake Macatawa with a 23-μm plankton net (Figure 1). The Lake Macatawa phytoplankton sample was transported in an opaque plastic bottle and filtered through a 200-μm sieve to prevent zooplankton grazing before use in the bioassay. For the first type of bioassay, 90mL of filtered water from each sampling site was placed in an acid-washed, autoclaved 125mL Erlenmeyer flask, in triplicate, and inoculated with 10mL of *Selenastrum*. The same design was used for the second type of bioassay, but flasks were inoculated with 10mL of Lake Macatawa phytoplankton. At the start of the incubation period, three initial subsamples from each bioassay were saved for chlorophyll-*a* analysis as described below. 20mL of the Lake Macatawa phytoplankton subsamples were preserved in 1% Lugol's for initial determination of the phytoplankton community. All *Selenastrum* flasks were incubated for seven days on a shaker table set to 175RPM. Three flasks of Lake Michigan water were included in this bioassay as a low-SRP control. The Lake Macatawa phytoplankton flasks were set up on a shaker table set to 175RPM on the other side of the chamber; this bioassay also included the three control flasks of Lake Macatawa phytoplankton in Lake Macatawa water. A black, opaque sheet divided the chamber vertically and separated the two treatments, and all flasks were stoppered with foam. No attempt was made to keep the conditions axenic.

Ambient conditions of Lake Macatawa were measured at the time of water sampling and the temperature and light irradiance of the growth chamber were set to mimic lake conditions depending on the season. During the spring, summer, and fall, the chamber temperature was set to 11.0°C, 26.6°C, and 13.9°C, respectively. The light:dark cycle was set to 13.5:10.5 in the spring, 14.5:8.5 in the summer, and 10.5:13.5 in the fall. Irradiance measured as photosynthetically active radiation (PAR) remained at 45 $\mu mol/m^2/s$ for all three *Selenastrum* bioassays while the Lake Macatawa phytoplankton side of the chamber was set at 190 in the spring, 355 in the summer, and 303 $\mu mol/m^2/s$ in the fall. The Lake Macatawa bioassay PARs were based on an average of measurements in Lake Macatawa at 0m, 0.5m, and 1.0m. All irradiance measurements were made with a LiCor Li-193SA spherical quantum sensor. Flasks were rearranged to random positions on the shaker tables each day to minimize variability in irradiance within the chamber.

Bioassay Analysis

At the end of the incubation period, all flasks were subsampled for chlorophyll-*a* and SRP, while Lake Macatawa phytoplankton flasks were additionally subsampled for taxonomic structure. An aliquot from each flask was filtered using a GF/F filter (Whatman®) and frozen at -18°C. Each filter was ground and then steeped in 90% buffered acetone in the dark for 24 hours. After centrifuging each sample, the chlorophyll-*a* concentration of the supernatant was analyzed using a Shimadzu UV-1601 spectrophotometer (Steinman et al. 2006). A second aliquot from each Lake Macatawa phytoplankton flask was preserved in 1% Lugol's in an opaque, plastic bottle for taxonomic analysis. At least 300 phytoplankton units from each sample were identified using a Nikon H550L inverted microscope (Utermöhl, 1958) to the division level, and whenever possible, to genus and species. Phytoplankton biovolumes were calculated based on the shape and appropriate measurements of 10 units of each taxon (Hillebrand et al. 1999).

Statistical Analysis

Shapiro-Wilk was used to test for normality of all data. SRP and TP concentrations were non-normally distributed and hence, violated the balanced design assumption of a repeated-measures ANOVA, so a non-parametric Kruskal-Wallis test was used for comparisons when grouped by sampling location or date. A Bonferroni post-hoc adjustment revealed individual significant differences. Relationships between chlorophyll-*a* and bioassay SRP concentrations were assessed using linear regression. The spring Lake Macatawa phytoplankton bioassay chlorophyll-*a* concentrations were non-normally distributed, so their relationship to SRP was based on Spearman's correlation analysis. Correlation also was used to relate SRP, TP, and %SRP concentrations per sampling site to acres drained by the tile system. Statistical analyses were conducted with R version 3.1.1 (R Core Team, 2016), and statistical significance was set at an alpha value less than or equal to 0.05.

Results

Table 1. Summary of lowest, highest, mean, and median concentrations of soluble reactive phosphorus (SRP), total phosphorus (TP), and percent SRP for nine tile drain sampling sites during March 2015 – February 2016.

	low	high	mean ±1SE	median
SRP ($mg \cdot L^{-1}$)	<0.005†	0.447	0.093 ± 0.011	0.064
TP ($mg \cdot L^{-1}$)	0.010	0.560	0.136 ± 0.013	0.102
%SRP	†	89%	60% ± 3%	69%

†SRP concentration is below the detection limit.

Tile Drain Phosphorus Concentrations

Tile drain SRP, TP, and %SRP varied both spatially and temporally in the Macatawa watershed. SRP concentrations ranged from below detection to 0.447 mg L^{-1}, whereas TP concentrations ranged from 10 to 560 mg L^{-1} (Table 1). Averaged across all sites and sampling months, SRP composed a relatively large portion of the TP concentration, as indicated by both the mean %SRP (60 ± 3%) and median %SRP (69%) (Table 1). There was no significant effect of time on any of the P parameters (Table 2). However, there was a highly significant effect of site location on all three P parameters (Table 2). There was a greater number of significant differences among sites for SRP and TP concentrations than for %SRP. The P concentrations varied the most at site 9 (SRP cv = 0.94; TP cv = 0.77) and the least at site 2 (SRP cv = 0.21; TP cv = 0.21).

Table 2. Results of Kruskal-Wallis tests comparing tile drain water soluble reactive phosphorus (SRP), total phosphorus (TP), and percent SRP concentrations when grouped by site location or sample date. Asterisks (*) indicate statistical significance.

Phosphorus Fraction	Grouping	Chi-Square	df	p-value
SRP	Location	59.834	8	<0.001***
	Time	3.4993	11	0.9823
TP	Location	54.53	8	<0.001***
	Time	4.737	11	0.9432
%SRP	Location	43.552	8	<0.001***
	Time	8.5723	11	0.6613

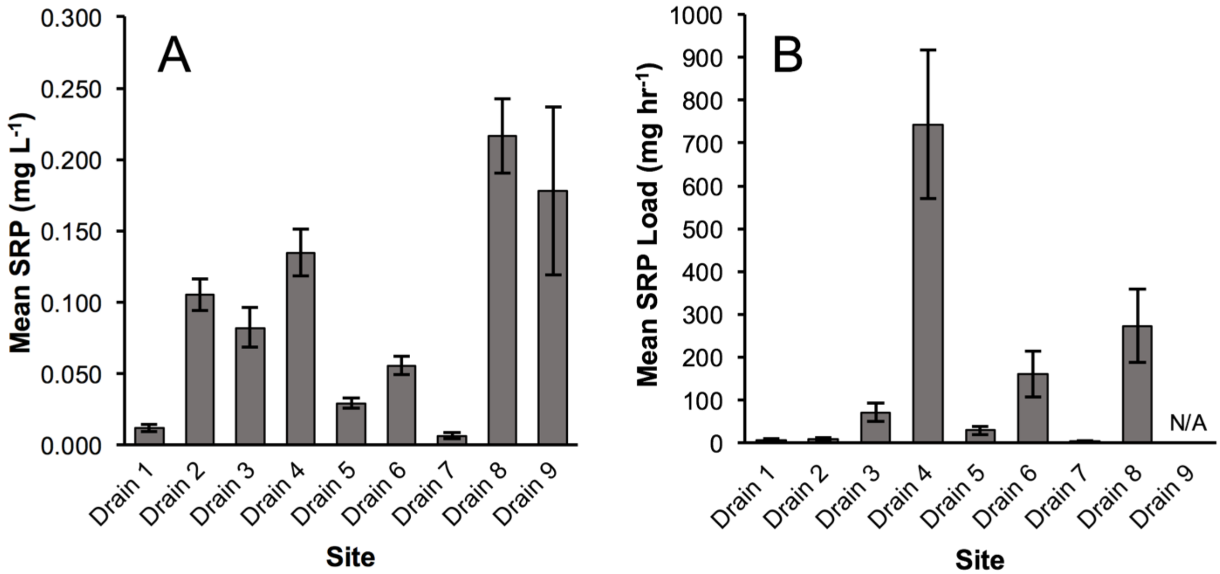

Figure 2. A) Mean soluble reactive phosphorus (SRP) concentration per sample site (X^2 = 59.834, $p < 0.001$); and B) mean SRP load per sample site (± SE).

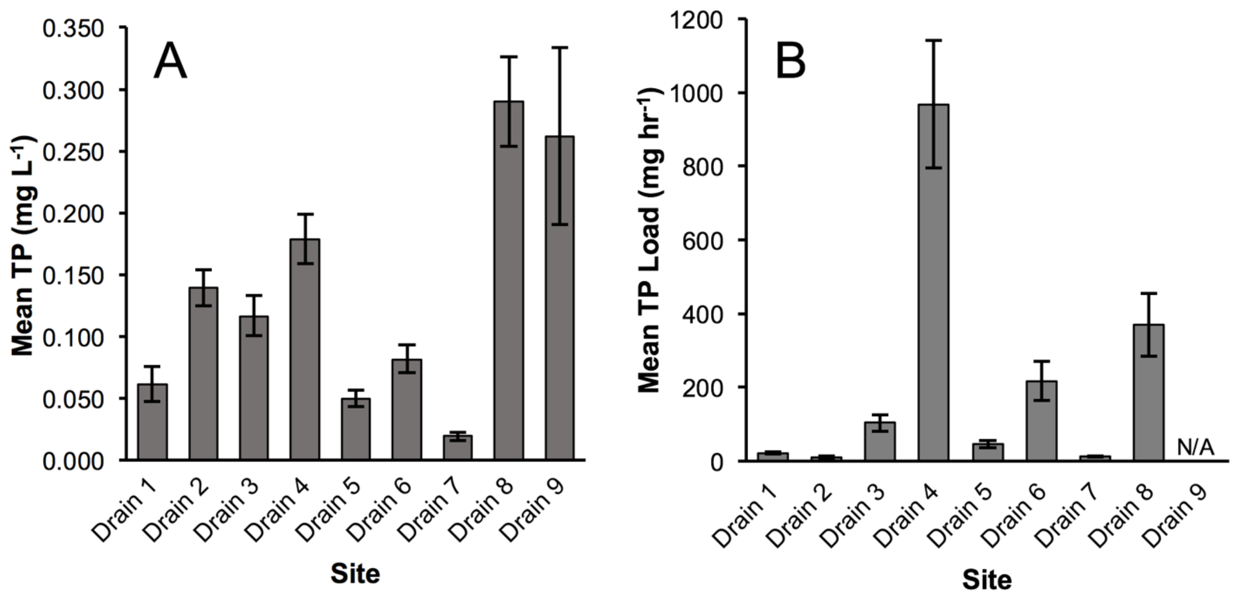

Figure 3. A) Mean total phosphorus (TP) concentration per sample site (X^2 = 54.53, $p < 0.001$); and B) load per sample site (± SE).

Tile Drain Phosphorus Load

Discharge from the tile drain systems varied by sample date and season. All sampling sites were actively flowing during March – June 2015, whereas the least number of sites were flowing in October 2015. SRP and TP loads were highest during February 2016 and lowest during October 2015 (Figure 4). Site 4 consistently flowed during all months and had the highest discharge rates while site 2 flowed the least with some of the lowest discharge rates. Overall, the highest P loads occurred during and post-snowmelt in both 2015 and 2016 (Figure 4).

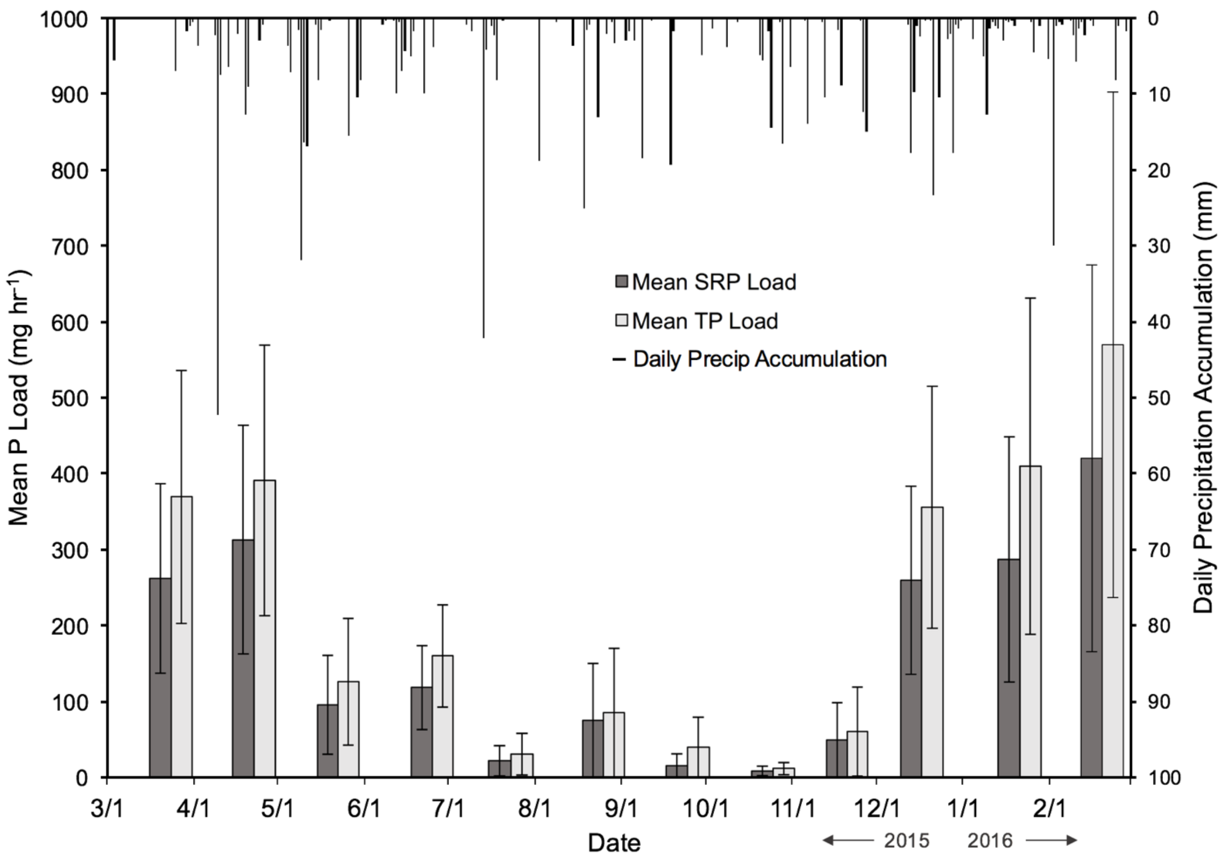

Figure 4. Mean soluble reactive phosphorus (SRP ± SE) load and mean total phosphorus (TP ± SE) load per sample date. Daily rain accumulation is also shown (National Climate Data Center – Tulip City Airport).

Land Management Factors

The majority of sampling sites drained fields containing corn, whereas a few contained soybeans (Table 7). In addition, inorganic P fertilizer was generally applied during planting at the beginning of the growing season with the exception of sites 1 and 7. The smallest field, site 1, drains only 7 acres and is unique because it is a community farm growing a wide variety of crops with the use of fish emulsion fertilizer. The majority of sites were classified as hydrologic type C soil, which has low-moderate infiltration rates (USDA SSURGO).

Percent SRP and acres drained by the tile system were significantly and positively correlated (Figure 5). Site SRP and TP concentrations also were positively related with acres drained, but were not statistically significant. No other relationship between tile drain P concentrations and a specific land management factor was apparent.

Table 3. Land management factors per sample site.

Site	Acres Drained	Crops	Winter Cover Crops	Fertilizer	Tillage	Dominant Soil Hydrologic Type††
1	7	Variety	No	Fish emulsion for some crops	No-tillage	A/D
2	65	Primarily corn with some soybeans	No	Inorganic fertilizer at spring planting	No-tillage	C
3	80	Corn	No	Inorganic fertilizer at spring planting; manure in the fall	Disc-tilled at planting	C
4	65	Corn	No	Inorganic fertilizer at spring planting; manure in the fall	Disc-tilled at planting	A/D, B
5	22	Corn	No	Inorganic fertilizer at spring planting; manure late summer	Disc-tilled at planting	C
6	36	Corn	No	Inorganic fertilizer at spring planting	No-tillage	C
7	30	Soybeans	No	No P-containing fertilizers	Vertical tillage before planting	B
8	50	Corn	No	Inorganic fertilizer at spring planting, manure in the fall	Disc-tilled at planting & fall tillage sub-soiler	C
9	39	Corn	Yes: radish, oats, clover	Inorganic fertilizer at spring planting	Vertical till twice during planting	C

††Obtained from USDA SSURGO database.

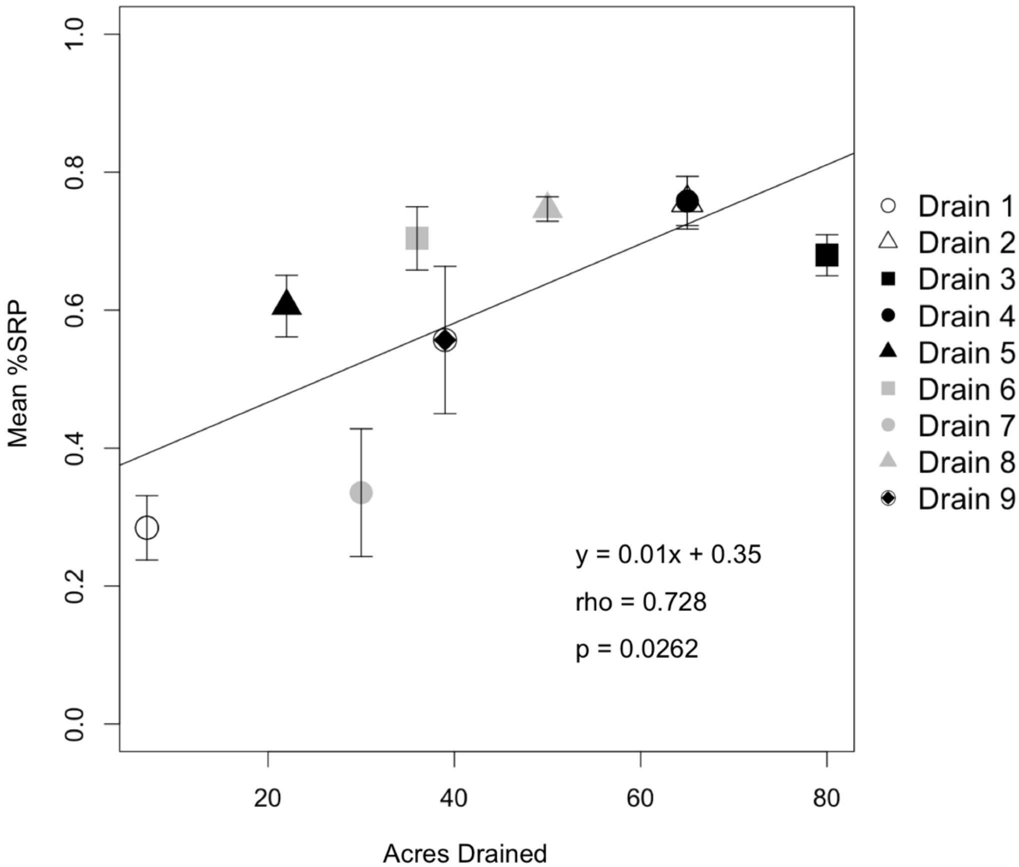

Figure 5. Spearman correlation comparing mean percent soluble reactive phosphorus (%SRP) at each tile drain outlet to acreage drained by the tile system (p = 0.0262).

Bioassay Chlorophyll-a Concentrations

Due to lack of flow from some tile outlets, not all sampling sites were used in each bioassay. Bioassays conducted in the spring used water from sites 1-8; those in the summer used sites 3, 4, 5, 7, 8, and 9, and bioassays in the fall used sites 1, 3, 4, 8, and 9. Contrary to expectations, only four of the six sets of bioassays revealed a positive relationship between tile water SRP concentration and mean change in chlorophyll-*a* concentration (Table 4). The only significant relationship was found during the spring *Selenastrum* bioassay (Table 4, Figure 6).

Table 4. Results of linear regression or Spearman correlation testing the relationship between soluble reactive phosphorus (SRP) concentration and mean (n = 3) change in chlorophyll-*a* for all seasonal bioassays. Asterisk (*) indicates statistical significance.

Season	Algae Inoculum	Statistical Test	Regression Equation	Test Statistic	p-value
Spring	*Selenastrum*	Linear Regression	y = 4217x + 316	$R^2_{adj} = 0.478$	0.0236*
	Lake Mac Phytoplankton	Spearman Correlation		rho = -0.267	0.4933
Summer	*Selenastrum*	Linear Regression	y = 12335x + 232	$R^2_{adj} = 0.289$	0.1227
	Lake Mac Phytoplankton	Linear Regression	y = 132x + 9	$R^2_{adj} = -0.145$	0.6446
Fall	*Selenastrum*	Linear Regression	y = -1510x + 385	$R^2_{adj} = -0.159$	0.6060
	Lake Mac Phytoplankton	Linear Regression	y = 46x - 2	$R^2_{adj} = -0.202$	0.7104

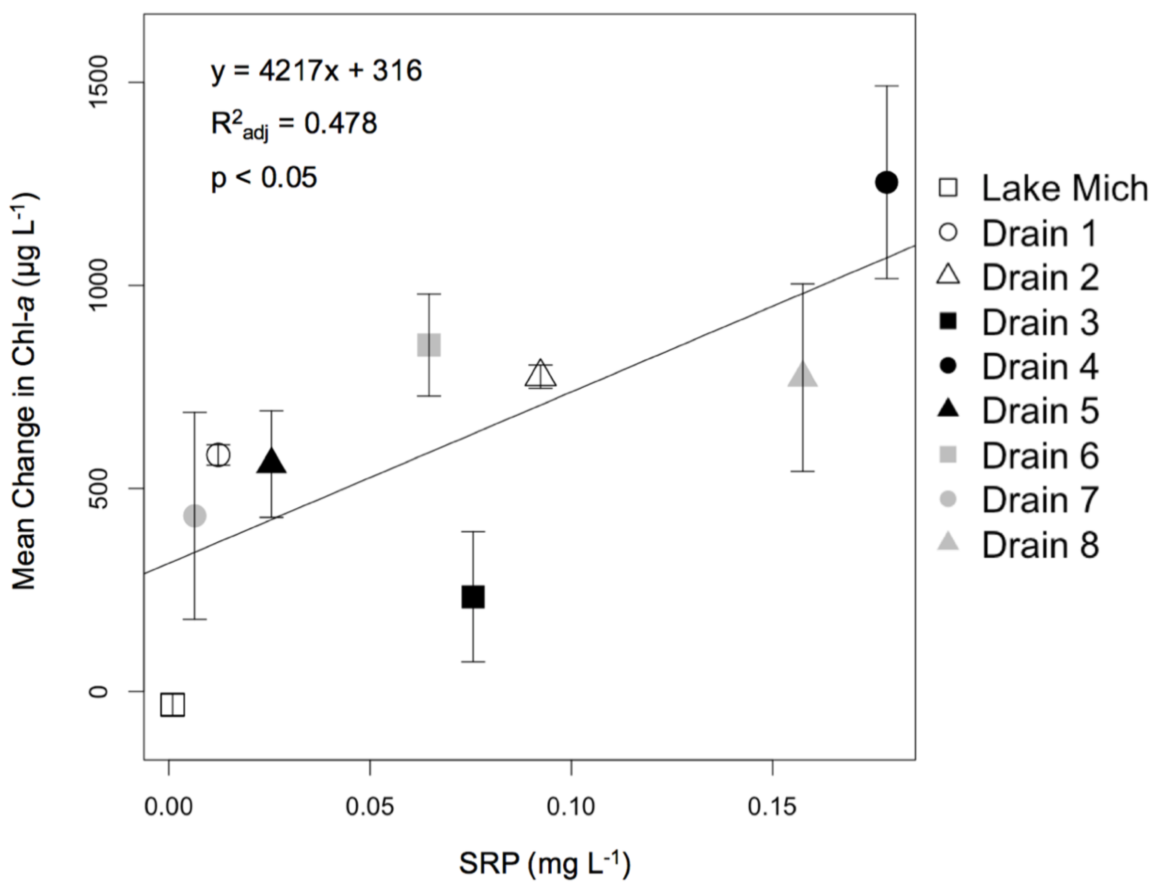

Figure 6. Linear regression comparing tile water soluble reactive phosphorus (SRP) to mean change in chlorophyll-*a* (± SE) during the spring *Selenastrum* bioassay.

Bioassay Algal Communities

The dominant algal taxa changed over the 7-day incubation during all three seasonal Lake Macatawa phytoplankton bioassays (Figure 7-9). The spring bioassay inoculum was dominated by the diatom (Bacillariophyta) genera *Asterionella* and *Aulocoseira*. *Asterionella* increased its dominance after incubation in tile drain water, but not to the same extent in the Lake Macatawa control flasks (Figure 7). During the summer bioassay, filamentous *Oscillatoria* (Cyanobacteria) was the dominant genus in the inoculum. However, after incubation in tile drain water dominance shifted to *Synedra* (Bacillariophyta). *Oscillatoria* remained dominant in the Lake Macatawa water controls along with larger populations of *Synedra* and *Pediastrum* (Chlorophyta) (Figure 8). Similarly, the initial community of the fall bioassay primarily consisted of *Oscillatoria*, and the filamentous cyanobacteria remained dominant after incubation in tile drain water with an increase in *Aulocoseira*. There was little change in the fall bioassay community when incubated in Lake Macatawa water (Figure 9). With the exception of *Oscillatoria* in the fall bioassay, there was little response from potential producers of cyanotoxins, such as *Microcystis* or *Anabaena*. *Microcystis* was present in some initial community samples, but its proportion in the community did not increase under any treatment (data not shown).

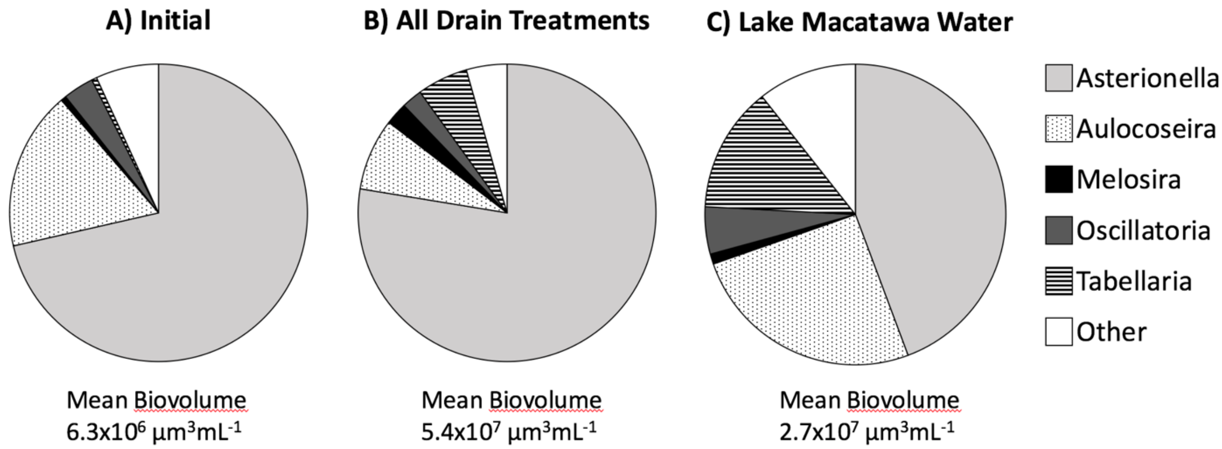

Figure 7. Lake Macatawa phytoplankton community based on calculated biovolume before and after the spring bioassay. A) Mean biovolumes of initial community at the start of incubation; B) mean biovolumes of phytoplankton incubated in all tile drain water flasks; C) mean biovolumes of the community after incubation in Lake Macatawa water (control flasks).

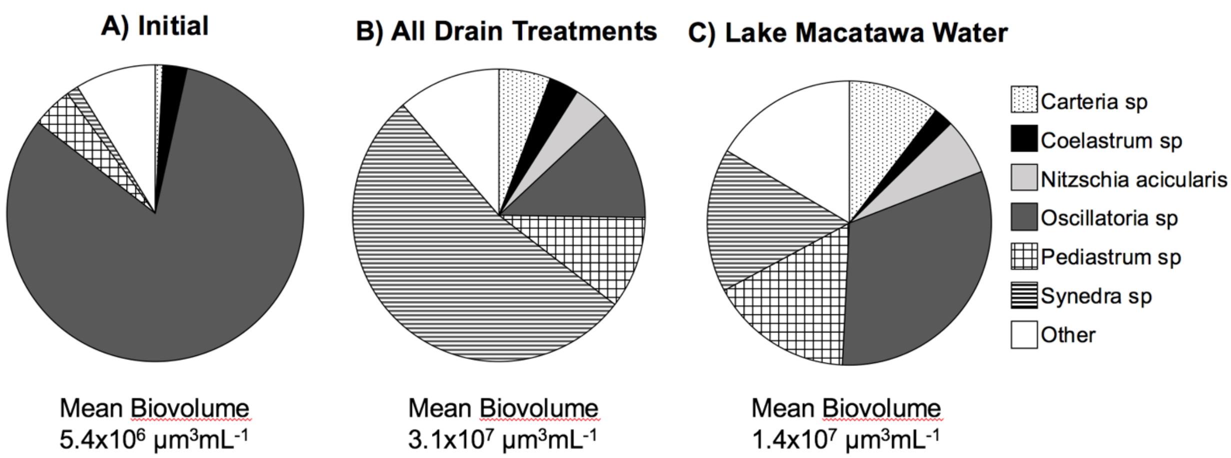

Figure 8. Lake Macatawa phytoplankton community based on calculated biovolume before and after the summer bioassay. A) Mean biovolumes of the initial community at the start of incubation; B) mean biovolumes of phytoplankton incubated in all tile drain water flasks; C) mean biovolumes of the community after incubation in Lake Macatawa water (control flasks).

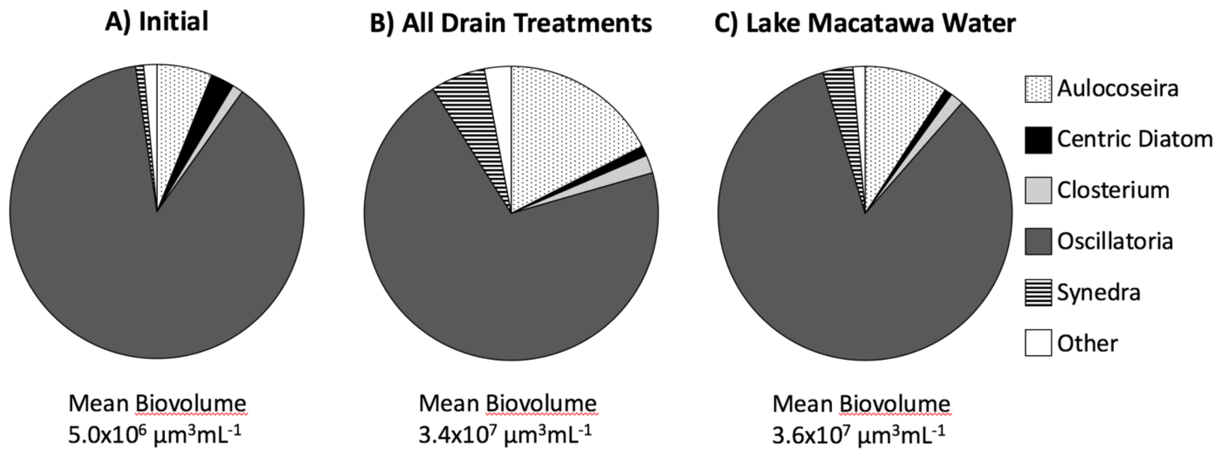

Figure 9. Lake Macatawa phytoplankton community based on calculated biovolume before and after the fall bioassay. A) Mean biovolumes of initial community at the start of incubation; B) mean biovolumes of phytoplankton incubated in all tile drain water flasks; C) mean biovolumes of the community after incubation in Lake Macatawa water (control flasks).

Discussion

Based on SRP and TP concentrations measured at the tile drain sampling sites, these subsurface drainage systems can discharge effluent with very high concentrations of bioavailable P to the drainage ditches in the Macatawa Watershed. Percent SRP measurements frequently exceeded 50% at the majority of sampling sites, with SRP concentrations 1-2 orders of magnitude greater than what is measured in Lake Macatawa (Holden 2014). The %SRP did not vary significantly over time, suggesting that bioavailable P was being transported from the tile drains to drainage ditches and likely further downstream throughout the year. Despite its year-round availability, it is unclear how much of this bioavailable P reaches Lake Macatawa given the opportunity for assimilation and/or adsorption; uptake lengths of P in these particular agricultural ditches have not yet been calculated. Retention in streams can occur via uptake by algae, sorption onto Fe or Al hydroxides, assimilation by microbes, and sedimentation of particulate matter on flood plains (Gelbrecht et al. 2005; Reddy et al. 1999). Deposition of particle-bound P on floodplains is an important aspect of ditch management. Using two-stage ditches in agricultural channels is an area of ongoing research and implementation of best management practices to increase P retention (Davis et al. 2015). However, because the factors influencing P retention and cycling vary spatially and temporally, extrapolating to a watershed scale is a complicated task.

In contrast to the absence of significant temporal variation in tile drain P concentrations, spatial variation of SRP and TP concentrations was significantly different in the Macatawa Watershed. Each tile system drained an agricultural field with varying land management practices, and there was a significant correlation between %SRP measured at the tile outlet and number of acres drained. Nine sampling sites, many of which are clustered in one area of the watershed, compose a relatively small representation of the watershed, and the total area of tile-drained land remains unquantified. Yet, it is estimated that 25-35% of the watershed's land area contains functional tile drains (Macatawa Area Coordinating Council, personal communication). Under this assumption, using only the lowest annual TP load measured at a sample site in this study (Drain 2), all tile drains would contribute 85-199 lb yr^{-1} to the watershed. In contrast, using the highest annual TP load (Drain 4), tile drains would contribute 8238-11,533 lb yr^{-1}. Given that the TMDL for Lake Macatawa set a goal of reducing nonpoint TP sources to 35,000 lb annually, tile drains could potentially contribute almost 33% of the allotted nonpoint source TP load, assuming none of it was retained before reaching Lake Macatawa. These load-per-acre calculations are coarse estimates based on instantaneous measurements, but the additional positive relationships found between both SRP and TP concentrations and acres drained provides a motive for quantifying the tile drains in the watershed, to better estimate the overall contribution of P by these systems.

Furthermore, the discharge rates and corresponding loads did differ by month. Tile drain discharge rates were higher during the winter and spring, resulting in higher SRP and TP loads during those seasons. These results correspond with other studies that measured tile drain P loading. For instance, Lam et al. (2016) observed the majority of P loss through tile drains in Ontario between October and May during snowmelt. In addition, the review by King et al. (2015a) lists several studies correlating tile drain P loss to elevated flow. The 2015-2016 study period in the Macatawa Watershed was a relatively dry summer with few precipitation events. The few rain storms in the summer or fall were relatively small and did not induce flow through the consistently dry tile drain sampling sites. Future summer rain events of greater intensity, as predicted by climate models (Hayhoe et al. 2010) may produce a different result and merit further investigation in this watershed. Indeed, Bettez et al. (2015) found that both climate variation and land use change have significant effects on N retention.

Management to address high P loads from tile drains during the non-growing season deserves more attention in the Macatawa Watershed. In other agricultural watersheds, construction of wetlands to receive tile drain outflow (Kynkäänniemi et al. 2013) or installation of drainage control structures have shown some success in limiting P transport (Frey et al. 2013). In contrast to passively flowing drains, controlled tile systems can limit nutrient loads by restricting flow when field drainage is not crucial. Also, in one study by Nash et al. (2015), soluble P concentrations were lower in controlled tile drains than passively flowing drains regardless of discharge rate. Moreover, time of sample collection in the Macatawa Watershed was not compared to stream hydrographs. P concentrations have been found to peak early during a precipitation event and slightly before hydrograph peaks (Tomer et al. 2010; Smith et al. 2015).

If this is true of the Macatawa Watershed, a top priority should be mitigating the “first flush” of P from tile drains at the onset of a precipitation event.

The seasonal bioassays allowed us to investigate the link between tile drain SRP and algal growth. A positive relationship was expected between SRP concentration and change in chorophyll-*a* in all bioassays given that P is often viewed as the limiting resource in freshwater systems (Dillon and Rigler 1974, Schindler 1977), although this paradigm has received considerable challenges of late (Conley et al. 2009, Harpole et al. 2011). However, the spring Lake Macatawa phytoplankton bioassay and the fall *Selenastrum* bioassay revealed a negative, albeit non-significant, relationship. It is possible the phytoplankton used in the bioassays were already P-saturated and therefore would not respond positively to additional SRP inputs. Xu et al. (2010) experienced a similar situation conducting bioassays in hypereutrophic Lake Taihu, China. During the summer and fall, excess available P stimulated phytoplankton growth only when N also was in excess, making P the secondarily limiting nutrient. Despite our unexpected results, the mean biovolume per bioassay flask did increase during incubation in tile drain and Lake Macatawa water, suggesting the algae were responding positively to something in the ambient water, possibly a micronutrient.

The most notable response during the Lake Macatawa bioassay algal community was by diatoms (Bacillariophyta). Because Lake Macatawa has a history of potentially harmful algal blooms (HABs), we anticipated potentially toxin-producing cyanobacteria, such as *Microcystis*, to grow in response to the tile effluent water, especially in the summer. *Microcystis* thrives on high SRP concentrations and warmer water temperatures in comparison to other phytoplankton taxa (Michalak et al. 2013). There are several reasons why this cyanobacterium may have been absent from the bioassays. The phytoplankton sample was collected relatively close to the mouth of the Macatawa River, so currents and wind may have pushed any concentrated HAB away from our lake sample site, resulting in low numbers in the inoculum. Moreover, filtering the phytoplankton sample through 200μm to remove zooplankton may have reduced the abundance of colonial phytoplankton forms, such as *Microcystis*. Lastly, the bioassays were conducted on shaker tables, and calm conditions with limited water column mixing has been known to facilitate *Microcystis* dominance in a phytoplankton bloom (Chen et al. 2003; Michalak et al. 2013).

While the results of this study do not rule out the possibility that SRP originating from agricultural tile drains helps fuel HABs in Lake Macatawa, it does suggest that other factors may also be involved. Small scale bioassays may not accurately reflect all processes of a lake ecosystem (Schindler et al. 2008). Future research should focus on the potential effects of nitrogen (N), which is commonly included in inorganic and organic fertilizers, the N:P ratio in the water column, internal loading of P from the lake sediments, micronutrients, and light penetration through the water column.

The results presented in this study are a snapshot of a one-year period, so further investigation is necessary to better our understanding of P transport to tile drains in the Macatawa Watershed. Even if effective nutrient management practices limit P application during the growing season, SRP and TP loads are highest during the non-growing season. P retention on agricultural fields during the winter and spring is crucial to reducing nutrient loss through tile drains. Because tile drains contribute bioavailable P to the Macatawa Watershed with the potential to fuel algal growth, limiting P transport by this route should be integrated into restoration efforts in the watershed.

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Chapter III

Conclusions and Synthesis

The federal Clean Water Act was passed to regulate pollution sources among growing public concern for surface water quality (USEPA 2016b). Under the Act, the National Pollutant Discharge Elimination System (NPDES) has effectively reduced pollution from point sources (USEPA 2016b); however, pollution also travels to water bodies via nonpoint sources originating from non-specific, diffuse sources (USEPA 2016a). Relative to point sources, measuring and regulating nonpoint sources presents a significant challenge to addressing degraded waterways in the US. Unlike point sources of pollution, nonpoint sources are not subject to permits, and measuring pollution transport relies on selective monitoring and extrapolation through modeling. The quantity and type of nonpoint pollution is influenced by land use, and in much of the Great Lakes basin, agricultural runoff is a key nonpoint source of nutrients (Danz et al. 2007). Phosphorus (P) transport via agricultural runoff can cause degradation of water quality, including algal blooms (Carpenter et al. 1998). Classic empirical studies have demonstrated that P in particular controls algal growth in freshwater systems (Schindler et al. 1997), although this paradigm has received considerable challenges of late (Conley et al. 2009, Harpole et al. 2011).

When a US water body is too degraded to meet water quality standards, it is listed under section 303(d) of the Clean Water Act (USEPA 2016c). After listing, a plan to limit pollutant loads to the waterway is developed. This type of plan, called a Total Maximum Daily Load (TMDL), allocates load reductions to point and nonpoint sources necessary to attain water quality standards (USEPA 2016c). Lake Macatawa and its tributaries, located in southwestern Michigan, are on Michigan's 303(d) list for not attaining water quality standards for warm water fishery and other aquatic life, and the lake is currently under a TMDL calling for a reduction in

TP concentration of 72% (Walterhouse 1999). My study took place in the Macatawa Watershed where nearly 90% of the area's wetlands were lost to development when European settlers drained the soil water and straightened tributaries (Holden 2014). The resulting increased flow of runoff water has led to nutrient-rich conditions throughout the watershed. Previous stream sediment and nutrient monitoring has shown that excess nutrients, including P, originate in the outer sub-basins of the watershed, which are dominated by agricultural fields of row crops (MWP 2012). Because subsurface tile drains are a potentially important source of nutrients from an agricultural field to receiving water bodies, this study was designed to investigate their role as a source of bioavailable P in the Macatawa Watershed.

Tile Drain Total Phosphorus Loads

Based on the measurements I conducted at the tile drain sampling sites, it is clear that these subsurface drainage systems can discharge significant concentrations of soluble reactive phosphorus (SRP) and total phosphorus (TP) in the Macatawa Watershed. However, it is not clear if they represent a significant load of P in the watershed relative to the other sources. To more thoroughly examine the P contribution of tile drains to the watershed, I performed a series of load calculations to compare my own findings to TP load models developed by the Michigan Department of Environmental Quality (MDEQ). In 2009, the MDEQ published two reports examining pollutant loads and the hydrology of the watershed; Lake Macatawa's tributaries were separated into 55 sub-basins. The Macatawa Watershed Hydrological Study evaluated the hydrologic characteristics of each sub-basin including land use and climate data (Fongers 2009a). A companion study, the Macatawa Watershed Modeled Pollutant Loads report, used TP concentrations and land use-dependent runoff estimates to calculate the TP load originating from each sub-basin (Fongers 2009b).

Table 1. Characteristics of Macatawa Watershed sub-basins where tile drain samples were collected in this study. Acres of agriculture calculated based on land use data†.

Tile Drain	Sub-basin Number*,†	Sub-basin Area† (acres)	Agriculture Land Use†	Acres of Agriculture
1	3	1715	65.4%	1122
2	18	2605	85.1%	2217
3	11	3424	84.8%	2904
4	18	2605	85.1%	2217
5	18	2605	85.1%	2217
6	17	1440	54.5%	785
7	25	3046	51.0%	1554
8	14	2867	95.4%	2735
9	4	2899	67.5%	1957

*Fongers (2009b)
†Fongers (2009a)

My tile drain sample sites are located in seven of the sub-basins determined by the MDEQ studies (Table 1). Sites 2, 4, and 5 are all located within the same sub-basin. The area of the sub-basins containing my sample sites ranged from 1440 to 3424 acres, and land use designated as agriculture exceeded 50% in all seven locations (Table 1). In the first comparison between my own findings and the DEQ load models, I compared mean TP load from each of my tile drain sites to TP load contributed by the corresponding sub-basin. In all tile drain TP load calculations, sample site 9 was not included because discharge could not be measured during the study period. To calculate TP load for each of the other tile drain sites, I multiplied mean TP concentration by mean discharge rate for the tile drain pipe. The lowest tile drain TP load was 0.2 lb yr^{-1} and the highest was 18.7 lb yr^{-1}, and overall the loads from tile drains were much less than the TP load from each corresponding sub-basin (Table 2). For instance, most sample sites supply less than 1% of the TP load to their corresponding sub-basin. This makes sense because one tile-drained field represents a small area of a sub-basin.

Next, I divided the mean tile drain TP load by the number of acres drained by that system to calculate yield from the sub-basins. The lowest tile drain TP yield was 0.003 lb yr^{-1} $acre^{-1}$ while the highest was 0.288 lb yr^{-1} $acre^{-1}$ (Table 2). Evaluating area-weighted TP load (yield) at each tile drain site and sub-basin is a more telling comparison than simply using the total TP load. Although all tile drain yields are below the values modeled for each entire sub-basin (Table 2), the tile drain site with the highest yield could be targeted as a large contributor of P to the corresponding sub-basin. For example, the yield at sample site 4 is 72% of the yield modeled in the sub-basin. Similarly, sub-basins with both high total TP loads and high yields relative to the entire Macatawa Watershed could be prioritized as regions needing the most effort toward limiting P transport to surface water (cf. Steinman et al. 2006).

Table 2. Mean total phosphorus (TP) loads and yields calculated from my study results compared to TP load from the corresponding sub-basin. Tile drain percent of total sub-basin load in parentheses. Drain 9 not included because discharge could not be measured during the study period.

Tile Drain	Acres Drained	Sub-basin Number*	Mean Tile Drain TP Load (lb yr^{-1})	Sub-basin TP Load* (lb yr^{-1})	Mean Tile Drain TP Load (lb yr^{-1} $acre^{-1}$)	Sub-basin TP Load* (lb yr^{-1} $acre^{-1}$)
1	7	3	0.4	635 (<1%)	0.060	0.371 (16%)
2	65	18	0.2	1040 (<1%)	0.003	0.398 (1%)
3	80	11	2.0	1270 (<1%)	0.025	0.370 (7%)
4	65	18	18.7	1040 (2%)	0.288	0.398 (72%)
5	22	18	0.9	1040 (<1%)	0.041	0.398 (10%)
6	36	17	4.2	509 (1%)	0.116	0.353 (33%)
7	30	25	0.3	1630 (<1%)	0.009	0.535 (2%)
8	50	14	7.1	1330 (<1%)	0.143	0.465 (31%)
9	39	4	N/A	1460	N/A	0.505

*Fongers (2009b)

The exact land area containing functional tile drains in the Macatawa Watershed is currently unquantified. This makes scaling TP loads up from the agricultural field level to the sub-basin level, or even to the watershed level, a complicated task. However, the acres of each sub-basin used for agricultural operations can be calculated with land use data (Table 1). Because the number of acres drained by tile drains is undetermined, I estimated their percent load contribution by creating four scenarios with differing percentages of agricultural land underlain by a functional drainage system (Table 3). I calculated the TP load from tile drains in each corresponding sub-basin assuming all tile drains in that particular basin supply the same TP yield as the sample site from my study. Therefore, I multiplied the agricultural acres by the tile drain yield. For example, the sub-basin containing tile drain sample site 8 is classified as 95.4% agricultural land use. Using the mean TP load from tile drain 8, and assuming 25% of the agricultural land in the sub-basin has functional tile drains, the drainage systems would contribute 98 lb yr^{-1} TP (0.143 lb yr^{-1} $acre^{-1}$ x 388 acres of agriculture). Conversely, if 100% of agricultural land in the same sub-basin contains functional tile drainage systems, they would contribute 391 lb yr^{-1} (Table 3). In comparison, the entire sub-basin TP load calculated by the DEQ was 1330 lb yr^{-1} suggesting that tile drains contribute a substantial amount to the modeled P load.

Table 3. Total phosphorus load (lb yr^{-1}) contributed to each sub-basin by tile drains under various percentages of agriculture area potentially tile drained. Load calculated based on land use data† (Table 1). Tile drain percent of total sub-basin load in parentheses. Drain 9 not included because discharge could not be measured during the study period.

Tile Drain	Sub-basin Number*,†	Mean Tile Drain TP Load (lb yr^{-1} $acre^{-1}$)	25% Tile-drained Area	50% Tile-drained Area	75% Tile-drained Area	100% Tile-drained Area
1	3	0.060	17 (3%)	34 (5%)	50 (8%)	67 (11%)
2	18	0.003	2 (0%)	3 (0%)	5 (0%)	7 (1%)
3	11	0.025	18 (1%)	36 (3%)	55 (4%)	73 (6%)
4	18	0.288	159 (15%)	319 (31%)	478 (46%)	638 (61%)
5	18	0.041	23 (2%)	45 (4%)	68 (7%)	91 (9%)
6	17	0.116	23 (4%)	46 (9%)	69 (13%)	91 (18%)
7	25	0.009	3 (0%)	7 (0%)	10 (1%)	14 (1%)
8	14	0.143	98 (7%)	195 (15%)	293 (29%)	391 (29%)
9	4	N/A	N/A	N/A	N/A	N/A

*Fongers (2009b)
†Fongers (2009a)

Because the Lake Macatawa TMDL addresses nutrient pollution from the entire watershed, it is important to examine tile drain TP loads on a watershed-scale as well. Although the exact land area containing tile drains is unquantified, it is estimated that 25-35% of the watershed's land area contains functional tile drains (Macatawa Area Coordinating Council, personal communication). To address TP load from tile drains on this larger scale, I applied the potentially tile-drained area scenarios to the whole watershed (Table 4). The mean TP yield at each sample site measured during my study (Table 2) was multiplied by a range of watershed area possibly containing functional tile drains (15-40%). Using the lowest tile drain TP yield (Drain 2) and the most conservative land area estimate (15%), tile drains would contribute 51 lb yr^{-1} (Table 4). However, under the highest land area scenario (40%) with the highest tile drain TP yield (Drain 4), these drainage systems would supply 13,180 lb yr^{-1} to the Macatawa Watershed (Table 4). Given that the TMDL for Lake Macatawa set a goal of reducing nonpoint TP sources to 35,000 lb annually, tile drains could potentially contribute over 37% of the allotted nonpoint source TP load, assuming none of it was retained before reaching Lake Macatawa.

Table 4. Tile drain total phosphorus load (lb yr^{-1}) based on potential tile drained area (%) of the entire Macatawa Watershed. Drain 9 not included because discharge could not be measured during the study period.

Tile Drain	15% Tile-drained	20% Tile-drained	25% Tile-drained	30% Tile-drained	35% Tile-drained	40% Tile-drained
1	1027	1370	1712	2054	2397	2739
2	51	68	85	102	119	137
3	431	574	718	861	1005	1148
4	4943	6590	8238	9885	11533	13180
5	704	938	1173	1407	1642	1876
6	2000	2667	3334	4001	4668	5334
7	152	202	253	303	354	404
8	2456	3274	4093	4911	5730	6548
9	N/A	N/A	N/A	N/A	N/A	N/A

The TP load calculations above are coarse estimates based on instantaneous measurements during my study, but they can shed light on potential areas for targeted efforts to reduce nutrient transport from agricultural areas in the Macatawa Watershed. Tile drains will not play as large of a role in TP load from sub-basins containing a relatively small percentage of agricultural land use, such as sub-basin 25, in comparison to sub-basins dominated by agriculture, such as sub-basin 14 (Table 1). Overall, the results of this analysis provide a rationale for quantifying the tile drained area in the watershed, to better estimate the overall contribution of P by these systems. Moreover, I found soluble reactive phosphorus (SRP) makes up a large portion of TP coming from tile drains in the watershed. Although load comparisons with the Lake Macatawa TMDL focus on TP, I can infer that P transported by tile drains includes considerable amounts of bioavailable SRP. Because tile drains contribute bioavailable P to the Macatawa

Watershed with the potential to fuel algal growth, limiting P transport by this route should be integrated into restoration efforts in the watershed.

Potential Solutions

Based on the results of my study, P load from tile drain systems should be included in efforts to the meet the Lake Macatawa TMDL goal. Tile drain discharge rates were higher during the winter and spring, resulting in higher SRP and TP loads during those seasons. These results correspond with other studies that measured seasonal tile drain P loading (King et al. 2015; Lam et al. 2016). Management to address high P loads from tile drains during the non-growing season is a potential area of research in the Macatawa Watershed. In other agricultural watersheds, installation of drainage control structures has shown some success in limiting P transport (Frey et al. 2013). In contrast to passively flowing drains, controlled tile systems can limit nutrient loads by restricting flow when field drainage is not crucial. As demonstrated by my study of the Macatawa Watershed, tile drain P loading is highly influenced by discharge rate from the outlet, so reducing discharge will simultaneously reduce P load. In addition, the study by Nash et al. (2015) found soluble P concentrations were lower in controlled tile drains than passively flowing drains regardless of discharge rate. If water flows through the soil column over a longer period of time, dissolved P can adsorb onto Fe(III) hydroxides and particulate matter instead of being flushed down to tile drains (Gelbrecht et al. 2005). Overall, P retention on agricultural fields during the winter and spring is crucial to reducing nutrient loss through tile drains.

In my study, the %SRP measured at tile drain outlets did not vary significantly over time, suggesting that bioavailable P was being transported from the tile drains to drainage ditches and further downstream throughout the year. Regardless of season, construction of wetlands designed to receive tile drain outflow is another management solution used to retain excess P (Kynkäänniemi et al. 2013). The main function of a constructed wetland receiving agricultural runoff is the settling of sediment-bound P and nutrient uptake by wetland plants (King et al. 2015). Therefore, most research in this area focuses on overland runoff instead of subsurface tile drain effluent (Tanner et al. 2005). Furthermore, there are mixed results from the few studies investigating a constructed wetland's ability to mitigate P load from tile drains (Kovacic et al. 2000; Mitsch et al. 1995). Soil characteristics and maturity of a wetland following construction could influence variable P retention (Tanner et al. 2005). Similar to the functions of a wetland, using two-stage ditches in agricultural channels is an area of ongoing research with the purpose of increasing P retention by allowing deposition of particle-bound P on constructed floodplains (Davis et al. 2015). Implementation of best management practices has the potential to limit P transport from tile drains to downstream freshwater systems.

Summary

Recognizing tile drains discharge potentially very high concentrations of P to connecting ditches in the Macatawa Watershed is an important first step in addressing their overall role as a source of P. Although the TP load calculations are coarse estimates, they indicate tile drains may be a significant source of P, and support the need to quantify the tile drained area in the Macatawa Watershed. The watershed is dominated by agriculture, and all of the sub-basins containing my sample sites exceeded 50% agricultural land use area. Best management practices designed to limit P transport from tile drains such as drainage control structures or constructed wetlands provide an opportunity to mitigate P load to the Macatawa Watershed. Moreover, management of P transport via tile drains can also be used to protect water quality in other agricultural watersheds. Nonpoint sources of P from agricultural drainage are not subject to permits. To restore surface water quality, tile drain outlets need to be viewed as more stringently controlled point sources rather than passive nonpoint transport routes for nutrient pollution.

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