12-2017

Phosphorus Retention in West Michigan Two-stage Agricultural Ditches

Emily Kindervater

Grand Valley State University

ScholarWorks Citation

Kindervater, Emily, "Phosphorus Retention in West Michigan Two-stage Agricultural Ditches" (2017). *Masters Theses*. 867.
https://scholarworks.gvsu.edu/theses/867

Phosphorus Retention in West Michigan Two-stage Agricultural Ditches

Emily Kindervater

A Thesis Submitted to the Graduate Faculty of

GRAND VALLEY STATE UNIVERSITY

In

Partial Fulfillment of the Requirements

For the Degree of

Master of Science

Biology- Aquatic Science

December 2017

Dedication

This master's thesis is dedicated to all those who have supported me through life, especially my parents who always nurtured my curiosity.

And to Rodger, for his infallible positivity and dedication to AWRI.

Acknowledgements

I would like to acknowledge and thank everyone who made this thesis possible.

Dr. Alan Steinman

Committee Members- Dr. Mark Luttenton, Dr. Rick Rediske, Dr. Jennifer Tank

Lab Members- Maggie Oudsema, Brian Scull, Mike Hassett, Kurt Thompson, Delilah Clement, Nicole Hahn, Kim Oldenborg Paige Kleindl, Lidia Iavorivska

Dan Callam and Rob Vink

Anonymous producers of the Macatawa

Dr. Michael Hupfer

Fellow graduate students

Those that provided funding:

Donner Family

Grand Valley State University- Graduate Assistantship and Presidential Research Grant

Michigan Space Grant Consortium

Michigan Chapter, North American Lake Management Society

Society for Freshwater Science

Abstract

Input of excess nutrients into a water body can negatively impact ecological structure and function, as well as the economic vitality of surrounding communities, by contributing to eutrophication. For example, phosphorus (P) and sediment inputs from agricultural drainage have facilitated the development of hypereutrophic conditions in Lake Macatawa, a drowned river mouth lake located in Holland, Michigan. Two-stage ditches, an agricultural best management practice (BMP), are used in some areas of the Midwest to reduce N export downstream via denitrification. This BMP simulates a mini-floodplain by replacing a traditional, trapezoidal ditch with a channel that has excavated benches on each side to help capture nutrients and sediment when the ditches flood. However, less is known about the ability of two-stage reaches to reduce P export.

This project assesses the effectiveness of two- stage ditches within the Macatawa watershed at retaining P. Both biotic and abiotic factors were analyzed as potential P sinks. Initial, baseline results showed that total P varied between 0.1 to 1 mg P/g dry sediment and tended to be higher in the upstream, traditional reach compared to the downstream, two-stage reach. Equilibrium P concentration values suggest retention of P within the two-stage. P was bound within stable fractions in both two-stage and traditional reaches. Sediment held over 96% of TP within each reach compared to < 4% in bench vegetation and algae combined. Turbidity but not P was reduced in one study ditch while P but not turbidity was reduced in the other study ditch. Ability to retain P appears to be impacted by physical as well as biogeochemical characteristics. Results will be used to inform management decisions within the watershed.

Table of Contents

List of Tables

Table **Page**

List of Figures

Figure **Page**

Abbreviations

ADS – Agricultural Drainage Systems

AFDM – Ash-free Dry Mass

BMP – Best Management Practice

DO – Dissolved Oxygen

EPC – Equilibrium Phosphorus Concentration

MAC – Macatawa

N – Nitrogen

%OM – percent organic matter

P – Phosphorus

Ref – Reference reach

SRP – Soluble Reactive Phosphorus

T – “T” ditch, designated to leave producers anonymous

TMDL – Total Maximum Daily Load

TP – Total Phosphorus

2-S – Two-stage reach

Z – “Z” ditch, designated to leave producers anonymous

Chapter 1

Introduction

Phosphorus as a Pollutant – the Macatawa Watershed

Excess nutrients accumulating in water bodies have been implicated as a trigger to harmful algal blooms and associated hypoxic conditions, negatively affecting the ecological health of those water bodies (Scavia et al. 2014) and the economic vitality of surrounding communities (Dodds et al. 2009). Cultural eutrophication, caused by human-related nutrient accumulation, is a problem throughout the United States, but has received considerable media attention recently in the Michigan/Great Lakes area due to toxin-producing cyanobacteria blooms in the western basin of Lake Erie.

The majority of nutrient input into large water bodies originates from non-point sources such as agricultural drainage (Maccoux et al. 2016). Production of row crops is strongly influenced by nutrient and water availability. Growth-limiting nutrients such as phosphorus (P) and nitrogen (N) are used in fertilizer to increase agricultural yields and are also found in high concentrations in livestock manure (Jarvie et al. 2015). Agricultural ditches and subsurface tile drains provide routes for excess water to drain from fields, which if not removed can result in anaerobic soil conditions and reduced crop yield. However, in moving water from fields, these agricultural ditches provide a route to transport excess nutrients and eroded soils downstream.

In West Michigan, the Lake Macatawa watershed in Ottawa and Allegan Counties has a history of P loading that has resulted in eutrophication. The major land use in the watershed is row crop and livestock farming; agricultural runoff from these areas has been implicated as the major source of nutrient and sediment loading throughout the watershed (MACC 2012). The 2016 mean total P (TP) concentration within Lake Macatawa was 92µg/L (Hassett et al. 2017);

the watershed is in need of extensive restoration and management to reduce this concentration to the U.S. EPA-approved total maximum daily load (TMDL) goal of 50 µg/L (MACC 2014).

In an effort to reduce nutrient and soil export to Lake Macatawa and restore the watershed, a community-based, multidisciplinary restoration plan, Project Clarity, is helping to implement a series of best management practices (BMPs). Multiple projects, such as wetland restoration, two-stage ditches, and bank stabilization, have either been completed or are in planning stages since completion of the PC Comprehensive Restoration Plan in 2013 (Project Clarity, www.macatawaclarity.org).

Two-stage Ditches

The two-stage ditch is an agricultural BMP specifically designed to reduce P and sediment transport with minimal impact to crop yield (Fig 1.1). The two-stage form is excavated out of a traditional, steep-banked, trapezoidal ditch so that parallel benches run on both sides of the existing drainage channel. These mini-floodplains reduce flow velocity and allow for reduction in water P concentration due to P uptake by biota, particle settling, and soil/sediment adsorption during high flow conditions (Davis et al. 2015, Powell et al. 2007).

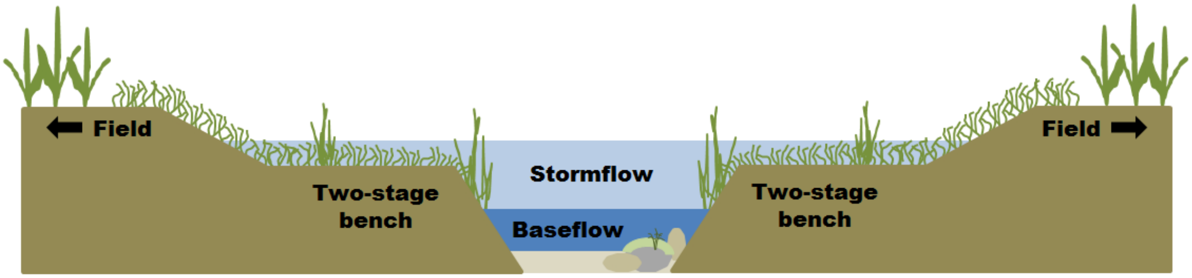


Fig. 1.1 – Representation of a two-stage ditch system and the connection between field and ditch channel. Dark blue represents water at baseflow. Lighter blue addition shows the water level increase and flow over the benches that occur during high flow events.

The floodplain benches also help decrease erosion into the channel during storm events, both reducing soil/sediment export and the need for maintenance. Two-stage ditches in the Midwest have been monitored over many years to determine the stability of the floodplain benches and potential erosion and deposition of sediments. Rates of geomorphic change of the two-stage ditches varied but all ditches were found to be stable relative to traditional, trapezoidal ditches and mimic natural fluvial changes (D'Ambrosio et al. 2015). Newly constructed two-stage ditch benches may experience erosion during high flow events until seeded and colonizing plant communities have been established (personal observations).

The removal of N in two-stage ditches has been well studied and is related primarily to denitrification in anaerobic zones that form on the benches (Davis et al. 2015, Mahl et al. 2015, Roley et al. 2012). Retention of P within two-stage ditches has received less attention. P can be exported through agricultural drainage in two different forms: aqueous dissolved P in the water and P that is bound to transported sediments (particulate P). While the intention of the two-stage ditches in the Macatawa watershed is to decrease the export of P and sediment, the mechanisms by which P is retained in agricultural ditch systems are largely unknown.

Very limited information is available on how and where P is bound in the soils and sediment of these agricultural drainage systems (Kallio et al. 2010). Reduction of P has been measured for two-stage systems but results are widely variable. Mahl et al. (2015) found that soluble reactive P (SRP- the more bioavaliable form of P in the water) was reduced anywhere between 3 and 53% compared to traditional ditches but they did not report results for total P (TP). Davis et al. (2015) found that SRP and TP were reduced only in two-stage systems with long water retention time. Watershed scale modeling to estimate possible P retention found two-stage systems could significantly reduce P export (Christopher et al. 2017). However, in some cases, reduction in TP was estimated with reductions in drainage turbidity instead of actual water total phosphorus concentrations.

Phosphorus Retention

Phosphorus can be retained within ecological systems chemically, physically, and biologically. P can be chemically sorbed to soil and sediment particles (Reddy et al. 1999). If the flow moving through the ditch is slow enough, suspended sediment can physically settle on the benches or bottom of the channel. P is also stored within pore water, the water between sediment grains. Vegetation and algae take up P from the soil and water for use in photosynthesis and tissue synthesis. The diagram Fig. 1.1 shows the abiotic (sediment and water flow) and biotic (crops, vegetation, and algae) factors in a two-stage system.

Inundation time (Sallade and Sims 1997, Steinman et al. 2014), degree of P saturation (Nair et al. 2015), and to what sediment fraction the P is bound (Dieter et al. 2015) all can influence retention in soils and sediment. P can bind to aluminum, calcium, iron, and other minor elements depending on mineral composition and redox conditions. If soil/sediment P is in mobile or loosely adsorbed forms, it may be quickly resuspended or re-released into the water column following high flow events.

The portion of P bound to oxidized iron (Fe^{3+}) forms iron oxyhydroxides (Dieter et al. 2015). Under prolonged inundation, anoxic conditions can result in Fe(III) reduction to Fe(II), thereby reducing the sorption potential of soil/sediments (Baldwin and Mitchell 2000) and allowing P to move back into the porewater and eventually into the water column. P bound to Al and Ca is more stable than P bound to Fe. Ca-P minerals are sensitive to low pH (Dieter et al. 2015, Reddy et al. 2000). As water levels decline during periods of summer baseflow, sediments can become exposed to the atmosphere. This oxidation process can decrease P sorption affinity and capacity of the sediment, and lead to increases in the labile and reductant-soluble forms of bound P (Dieter et al. 2015). Re-inundation of benches after the first stormflow, following prolonged desiccation, could result in release of mobile soil P to the water column.

In addition to abiotic uptake of P in soil/sediments, biotic uptake is also an important factor. Although likely only seasonally important sinks of P, variations in plant biomass or major plant types and their interactions with soil P could be an important factor of retention to consider. With increasing nutrient availability, plant biomass has been shown to increase in wetland buffers (Silvan et al. 2004). Root structure within plants can be a large portion of both biomass and P content within a plant (Moore and Kroger 2011; Teng et al. 2013). However, the stability of bench and bank soil was important for the continuation of the project and therefore roots could not be sampled.

Along with riparian plants, periphyton could be another possibly influential sink for P within two-stage ditches (Reddy et al. 2000). Periphyton is a compilation of benthic algae, bacteria, and other fungi, protozoan, and viruses (Larned 2010) and can impact movement of P between sediments and the water column by forming a boundary layer (Brennan et al. 2017, Drake et al. 2012). Despite the presence of periphyton communities in agricultural waterways (Brennan et al. 2017), their ecological roles are poorly understood in the context of agricultural two-stage drainage systems.

Factors such as flow, desiccation, light, and nutrient type and concentration can affect various algal factors such as abundance and growth form (Lange et al. 2015). These factors may differ between two-stage reaches and traditional trapezoidal shaped reaches. One of the ways two-stage ditches are intended to decrease erosion is through the reduction in shear stress (Powell et al. 2007). This reduction in flow during storm events could have an impact on the growth form of algae. For example, loosely attached and motile algae might be washed down stream during high strength high flow events. The reduction of particulate associated turbidity (Mahl et al. 2015) will increase light penetration in two-stage ditches compared to traditional ditches, thereby allowing for increased periphyton growth. Increases in deposited particulate matter may increase the relative abundance of motile taxa of algae (Wagenhoff et al. 2013). Filamentous algae such as *Cladophora* or *Oedogonium* composed the major portion of the periphyton community in high intensity agricultural streams in response to high concentrations of nutrients (Lange et al. 2015, Stevenson et al. 2012). The intended reduction in nutrients due to the two-stage form also could impact community structure by decreasing the abundance of high nutrient tolerant taxa.

Purpose

The purpose of this study is to understand the retention of P within the two-stage ditches of the Macatawa Watershed. By understanding how phosphorus is bound, where it is stored, and the likelihood of phosphorus release in these ditches, we can better understand the use of two-stage ditches as a best management practice and their ability to contribute to the restoration of the Macatawa Watershed.

Scope

This thesis focuses on the abiotic and biotic factors that influence phosphorus cycling within two separate two-stage ditches in the Macatawa watershed, specifically ditches constructed in the southern portion of the watershed through Project Clarity. The objectives of my thesis were to examine the retention of P in two-stage ditches compared to their corresponding upstream reference reaches that remained in the traditional trapezoidal form for reference purposes. Specifically, I examined the differences in: 1) water quality; 2) soil/sediment

P content, P fractionation, and the likelihood of its release back into the water column; 3) biotic P standing stock within channel periphyton and bench vegetation; and 4) the general impact of the two-stage construction on vegetation cover and periphyton community structure. Because only two systems were studied, the results discussed in this thesis should be considered preliminary in nature. The conclusions can be used to further research in other watersheds but should not be considered directly applicable to or representative of all two-stage systems.

Research Questions and Hypotheses

I anticipated that the effectiveness of the two-stage form at retaining P within soil/sediments compared to the traditional form would vary due to significant changes in P input, high flow events, desiccation, and plant growth. More specific questions and hypotheses are outlined below:

1) How does the two-stage ditch system affect the retention of P within agricultural drainage systems?

Hypothesis H1: The two-stage reach will retain more P in surface soil and sediments than the reference reach over the sampling year.

Rationale H1: The benches and associated vegetation will lead to decreased erosion, increased surface area for soil P sorption, and longer hydraulic residence times, thereby increasing sediment-bound P deposition within the two-stage reaches during stormflow (D’Ambrosio et al. 2015).

2) Are the P fractions different between the two-stage and reference reaches?

Hypothesis H2: There will be a larger labile P fraction within the two-stage reach. The reductant soluble P will be similar between two-stage and reference reaches. The P in the Al-bound fraction will also be higher in the two-stage.

Rationale H2: Due to increased plant biomass because of the larger bench surface area, mineralization of organic material will result in a larger labile P fraction within the two-stage reach and more organic P. Both reaches will remain oxic for the whole year and therefore reductant soluble P will remain similar (Dieter et al. 2015). Al-bound and organic P will be higher in the two-stage due to presence of more clay particles and higher organic P.

3) How will the equilibrium P concentration (EPC) differ between two-stage and reference reaches?

Hypothesis 3: The EPC values will be lower for the two-stage reach than the reference reach.

Rationale 3: Due to the increased vegetation growth and reduced height of the benches within the two-stage reach, less soil drying will occur during summer months compared to the reference reach. Dessication has been shown to significantly reduce sorption potential and thereby increase the EPC measured for sediments (Dieter et al. 2015).

4) How will the two-stage form affect periphyton ash-free dry mass and P content?

Hypothesis 4: The two-stage reach will have lower algal periphyton growth but higher AFDM per area compared to the reference reach.

Rationale 4: The two-stage reach will have lower turbidity compared to the reference (Mahl et al. 2015) and as the sediments deposit out of the water column and settle on the bottom, there will be a reduction in algal periphyton as cells will get shaded or smothered (Wang, 1974). Even at average small stream flows, deposited particles account for at least 50% of the dry mass of periphyton samples (Graham 1990). As the flow is slowed through the two-stage reach, even more suspended sediment will be able to settle out onto periphytic surfaces, potentially smothering the algae growing there and increasing the inorganic fraction of the total periphyton sample mass.

References

Baldwin, D.S., Mitchell, A.M. 2000. The effects of drying and re-flooding on the sediment and soil nutrient dynamics of lowland river-floodplain systems: a synthesis. Regul. River. 16: 457-567.

Brennan, R.B., Scott, J.T., Sharpley, A.N., Lally, H.T., Jarvie, H.P., Bowes, M.J., Haggard, B.E., Gbur, E. 2017. Linking soil erosion to instream dissolved phosphorus cycling and periphyton growth. J. Am. Water Res. Assoc. DOI: 10.1111/1752-1688.12534

Christopher, S.F., Tank, J.L., Mahl, U.H., Yen, H., Arnold, J.G., Trentman, M.T., Sowa, S.P., Herbert, M.E., Ross, J.A., White, M.J., Royer, T.V. 2017. Modeling nutrient removal using watershed-scale implementation of the two-stage ditch. Ecol. Eng. (In Press, http://dx.doi.org/10.1016/j.ecoleng.2017.03.015)

D'Ambrosio, J.L, Ward, A.D., Witter, J.D. 2015. Evaluating geomorphic change in constructed two-stage ditches. J. Gt. Lakes Res. 51:910-922.

Davis, R.T., Tank, J.L., Mahl, U.H., Winikoff, S.G., Roley, S.S. 2015. The influence of two-stage ditches with constructed floodplains on water column nutrients and sediments in agricultural streams. J. Am. Water. Res. Assoc. 51: 941-955.

Dieter, D., Herzog, C., Huper, M. 2015. Effects of drying on phosphorus uptake in re-flooded lake sediments. Environ. Sci. Pollut. R. 22: 17065-17081.

Dodds, W.K., Bouska, W.W., Eitzmann, J.L., Pilger, T.J., Pitts, K.L., Riley, A.J., Schloesser, J.T., Thornbrugh, D.J. 2009. Eutrophication of U.S. freshwaters: analysis of potential economic damages. Environ. Sci. Technol. 43: 12-19.

Drake, W.M., Scott, J.T., Evans-White, M. Haggard, B., Sharpley, A., Rogers, C.W., Grantz, E.M. 2012. The effecto of periphyton stoichiometry and light on biological phosphorus immobilization and release in streams. Limnology 13:97-106.

Graham, A.A. 1990, Siltation of stone-surface periphyton in rivers by clay-sized particles from low concentrations in suspension. Hydrobiologia 199: 107-115.

Hassett, M., Oudsema, M., Steinman, A.D. 2017. Project Clarity 2016 Annual Monitoring Report. www.gvsu.edu/cms4/asset/DFC9A03B-95B4-19D5-

F96AB46C60F3F345/final_report_2016_water_year.pdf

Jarvie, H.P., Sharpley, A.N., Flaten, D., Kleinman, P.J., Jenkins, A., Simmons, T. 2015. The pivotal role of phosphorus in a resilient water-energy-food security nexus. J. Environ. Qual. 44: 1049-1062.

Kallio, S., Ward, A., D'Ambrosio, J.L., Witter, J.D. 2010. Beyond the field: a look at agricultural ditch floodplains as a water quality BMP. CIGR XVII World Congress

Larned, S.T. 2010. A prospectus for periphyton: recent and future ecological research. J. North Am. Benthol. Soc. 29: 182-206.

Lange, K., Townsend, C.R., Matthaei, C.D. 2015. A trait-based framework for stream algal communities. Ecol. Evol. DOI: 10.1002/ece3.1822.

MACC. 2012. Macatawa Watershed Management Plan Appendix H www.the-macc.org/watershed/overview/

MACC. 2014. Macatawa Watershed Annual Progress Report. www.the-macc.org/watershed/overview/

Maccoux, M.J., Dove, A., Backus, S.B., Dolan, D.M. 2016. Total and soluble reactive phosphorus loadings to Lake Erie: A detailed accounting by year, basin, country and tributary. J Great Lakes Res. 42:1151-1165.

Mahl, U.H., Tank, J.L., S.S. Roley, Davis, R.T. 2015. Two-stage ditch floodplains enhance N-removal capacity and reduce turbidity and dissolved P in agricultural streams. J. Am. Water Res. Assoc. 51: 923-940.

Nair, V.D., Clark, M.W., Reddy, K.R., 2015. Evaluation of legacy phosphorus storage and release from wetland soils. J. Environ. Qual. 44: 1956-1964.

Powell, G.E., Ward, A.D., Mecklenburg, D.E., Draper, J., Word, W. 2007. Two-stage channel systems: part 2, case studies. J. Soil Water Conserv. 62:286-296.

Reddy, K.R., D'Angelo, E.M., Harris, W.G. 2000. Biogeochemistry of Wetlands. Handbook of Soil Science. M.E. Sumner, ed. G89-119.

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

Roley, S.S., Tank, J.L., Stephen, M.L., Johnson, L.T., Beaulieu, J.J., Witter, J.D. 2012. Floodplain restoration enhances denitrification and reach-scale nitrogen removal in an agricultural stream. Ecol. Appl. 22: 281-297.

Sallade, Y.E., Sims, J.T. 1997. Phosphorus transformations in the sediments of Delaware's agricultural drainageways: II. Effect of reducing conditions on phosphorus release. J. Environ. Qual. 26:1579-1588.

Scavia, D., Allan, J.D., Arend, K.K., Bartell, S., Beletsky, D., Bosch, N.S., Brandt, S.B., Briland, R.D., Daloglu, I., DePinto, J.V., Dolan, D.M., Evans, M.A., Farmer, T.M., Goto, D., Han, H., Hook, T.O., Knight, R., Ludsin, S.A., Mason, D., Michalak, A.M., Richards, R.O., Roberts, J.J., Rucinski, D.K., Rutherford, E., Schwab, D.J., Sesterhenn, T.M., Zhang, H., Zhou, Y. 2014. Assessing and addressing the re-eutrophication of Lake Erie: central basin hypoxia. J. Great Lakes Res. 40: 226-246.

Silvan, N., Vasander, H., Laine, J. 2004. Vegetation is the main factor in nutrient retention in a constructed wetland buffer. Plant Soil 258: 179-187.

Steinman, A.D., Ogdahl, M.E., Weinert, M., Uzarski, D.G. 2014. Influence on water level fluctuation duration and magnitude on sediment-water nutrient exchange in coastal wetlands. Aquat. Ecol. 48: 143-159.

Stevenson, R.J., Bennett, B.J., Jordon, D.N., French, R.D. 2012. Phosphorus regulates stream injury by filamentous green algae, DO, and pH with thresholds in responses. Hydrobiologia 695: 25-42.

Wagenhoff, A., Lange, K., Townsend, C.R., Matthaei, C.D. 2013. Patterns of benthic algae and cyanobacteria along twin-stressor gradients of nutrients and fine sediment: a stream mesocosm experiment. Freshwater Biol. 58: 1894-1863.

Wang, W-C. 1974. Effect of turbidity on algal growth. Illinois Dept. Regist. Ed. Circular 121.

Chapter 2

Phosphorus Retention within the Soil and Sediment of West Michigan Two-stage Agricultural Ditches

Abstract

Excess nutrients can contribute to eutrophication of surface waters, negatively impacting ecosystems and the economy of surrounding communities. Phosphorus (P) and eroded soil inputs from agricultural drainage have facilitated the development of hypereutrophic conditions in Lake Macatawa in western Michigan. Two-stage ditches, an agricultural best management practice, were installed in the watershed in an attempt to trap nutrients and sediment before they enter downstream water bodies. Two-stage ditches can effectively remove nitrogen through denitrification but less is known about their ability to retain P. This project assessed the effectiveness of soil and sediment at retaining P by comparing soil total P (TP), P fractionation of soil/sediment, and the equilibrium phosphorus concentration (EPC) of two separate two-stage reaches and corresponding upstream, traditional reaches within this watershed. Among the various P stocks measured for these systems (soil/sediment, periphyton, and vascular vegetation), soil/sediment made up over 96% of the P present. Mean soil TP was significantly higher in the traditional reaches compared to the two-stage reaches. The most abundant P fractions within the soil and sediment were stable Al- and Ca-bound fractions. Equilibrium P concentration values suggest the soil and sediment was more likely to retain P within the two-stage than the traditional reaches. The two-stage systems within the Macatawa Watershed can retain P within the soil and sediment.

Keywords: Two-stage, Nutrients, Phosphorus, Best management practices, Agriculture, Nonpoint source impacts

Introduction

Balancing water quality with crop yield is a major natural resource management challenge in agricultural watersheds (Jarvie et al. 2015). Availability of nutrients such as P and nitrogen (N) is necessary for the growth and development of autotrophic organisms (e.g., vegetation and algae); hence, P and N are often applied in fertilizer to increase agricultural yields. Agricultural ditches and subsurface tile drains (collectively, agricultural drainage systems or ADS) provide routes for excess water to drain from fields, which if not removed can result in anaerobic soil conditions and reduced crop yield. However, in draining water from fields, ADS provide an expedited route for excess nutrients and eroded soils to move downstream from agricultural lands to receiving water bodies (Blann et al. 2009). Nutrients and soil in field runoff and drainage ultimately accumulate in downstream water bodies, where they can facilitate significant algal blooms and associated hypoxic conditions, affecting the ecological health of those water bodies (Scavia et al. 2014) and the economic vitality of the surrounding community (Dodds et al. 2009).

Nonpoint source pollution (e.g., agricultural drainage) is a problem throughout the United States, but has received considerable attention recently in the Great Lakes region due to toxin producing cyanobacterial blooms in the western basin of Lake Erie. In West Michigan, the Lake Macatawa watershed (Ottawa and Allegan counties) has a history of P loading that has resulted in hypereutrophic lake conditions and associated algal blooms. In particular, agricultural runoff has been implicated as the major source of nutrient and sediment loading throughout the 450 km^2 watershed (MACC 2012). The 2016 mean surface total P (TP) concentration within Lake Macatawa was 92 µg/L (Hassett et al. 2017). The watershed is in need of extensive restoration and management to reduce this concentration to the U.S. EPA-approved total maximum daily load (TMDL) goal of 50 μg/L (MACC 2014). A community-based Comprehensive Restoration Plan aimed at reducing this P and sediment input was completed in 2013 (http://www.macatawaclarity.org/). This restoration initiative, called Project Clarity, has facilitated the implementation of best management practices (BMPs) such as buffer strips, bank stabilization, and two-stage ditches, as well as wetland restoration projects throughout the greater Macatawa watershed. My project focuses on two of the two-stage ditches constructed through Project Clarity.

Two-stage ditches are designed to reduce nutrient and sediment transport with minimal impact to crop yield. Parallel floodplain benches are excavated on each side of the channel of an existing traditional, steep-banked, trapezoidal ditch (Fig. 2.1). The small-scale floodplains increase stability, thereby decreasing channel erosion during storm events, reducing sediment export and the need for maintenance (D'Ambrosio et al. 2015, Powell et al. 2007). The benches become inundated during high flow events such as storms or spring snowmelt (Powell et al. 2007), allowing for reduction in water nutrient concentrations due to P and N uptake by biota, particle settling, and sediment P sorption (Davis et al. 2015, McDowell et al. 2017).

P can be exported through ADS in two different forms: aqueous dissolved P and P that is bound to transported sediments or in organisms. Water inundation time (Sallade and Sims 1997; Steinman et al. 2014), degree of P saturation (Nair et al. 2015), and to what sediment fraction the P is bound (Dieter et al. 2015) all can influence retention. P can bind to aluminum, iron, calcium, manganese, and other minor elements within the sediment. As water levels decline during periods of summer baseflow, sediments become exposed to the atmosphere. This exposure results in sediment oxidation, which can decrease sediment P sorption affinity and capacity, and lead to increases in the labile and reductant-soluble forms of bound P (Dieter et al. 2015). Re-inundation of benches during the first stormflow following prolonged desiccation could result in release of that loosely-bound sediment P to the water column. Also, under prolonged stagnant inundation, anoxic conditions can result in the reduction of Fe(III) to Fe(II), thereby reducing the P sorption potential of sediments (Baldwin and Mitchell 2000).

To determine what compounds P is bound to within a system, a series of chemical analyses can separate one P fraction at a time for measurement. P fractionation can be conducted with different numbers of chemical extractions depending on the detail desired (Wang et al. 2013). P will only stay bound within a system under certain conditions and as long as the concentration within the sediment is below the equilibrium P concentration (EPC) (Hongthanat et al. 2016, Pant and Reddy 2001). In order to calculate the EPC for a certain soil or system, an isotherm analysis testing the P retention at differing aqueous dissolved P concentrations is conducted within a laboratory.

In addition to abiotic retention of P in sediments, biotic uptake also may be important. The more bioavailable form of P, soluble reactive P (SRP), is readily available for uptake by vegetation and algae. Two-stage benches are typically seeded after excavation and/or naturally colonized by vegetation. With increasing nutrient availability, plant biomass has been shown to increase in wetland buffers (Silvan et al. 2004) and vegetation has been shown to be correlated to major P fractions and P concentrations found in the soil (Wang et al. 2012). Large variations in plant biomass or major plant types between two-stage and reference reaches within each ditch system need to be taken into account when comparing P in sediments.

The retention and removal of one common fertilizer nutrient, N, in two-stage ditches has been well studied and is related primarily to denitrification in anaerobic zones that form on benches (Roley et al. 2012a; Roley et al. 2012b; Mahl et al. 2015). The ability of two-stage ditches to retain P has received less attention. While the intention of the two-stage ditches in the Macatawa watershed is to decrease export of P and sediment, the exact mechanisms by which P is retained in these types of systems are largely unknown (Kallio et al. 2010). If two-stage ditches are to serve as an agricultural BMP for P retention, it is critical to know how they work so their design can be optimized and replicated in other systems.

In this study, my goal was to assess P retention in the Macatawa watershed. I compared P retention in two-stage ditches (two-stage, 2-S) versus upstream reaches (reference, Ref) that remain in the traditional ditch form within the Macatawa watershed. My objectives were to determine potential differences in: 1) water quality, EPC, and major P fractions; 2) vegetation cover, biomass, and vegetation TP between two-stage and reference reaches; 3) periphyton TP between two-stage and reference reaches as well as compare relative standing stocks (sediment, vegetation, and periphyton) of TP in each reach; and 4) sediment P across seasons. I hypothesized that the two-stage reach would have improved water quality, higher sediment P retention, and increased vegetation cover. I also anticipated that the sediment P retention would increase over season due to the growth of vegetation.

Methods

Study Sites

We examined two two-stage ditches and their corresponding upstream reference reaches that remained in the traditional, trapezoidal shape. These ditches are within the Allegan County portion of the Macatawa watershed in West Michigan (Fig. 2.2). These ditches are referred to as the “Z” and the “T” ditch systems to retain farmer anonymity. The Z two-stage reach was constructed in February 2015 and the T two-stage reach was constructed in November 2015 (Table 2.1).

Water Quality

Surface water was sampled at the downstream end of both treatment and reference sections in both ditches (cf. Fig. 2.3) monthly from March to December 2016. Water samples were stored on ice during transportation back to the laboratory. Samples for TP were stored at 4 °C and samples for SRP were filtered through 0.45 µm filters and then stored at 4°C until analysis with a SEAL Autoanalyzer (SEAL Analytical, Mequon, Wisconsin). Temperature (°C), dissolved oxygen (mg/L), and pH were measured using a YSI 6600 at time of surface water sampling. Turbidity sensors (Cyclops-7F Turner Designs) were installed at the downstream end of each reach and collected continuous (every 10 minutes) in-stream measurements from late April to December 2016. Sensors were checked and cleaned every two weeks and calibrated once per month.

Sediment and Soil

All sediment and soil samples described below were collected in April and October of 2016. Cores (top 3 cm) were collected for analysis using a modified clear PVC system. The clear PVC tube was pounded into the sediment, capped with a threaded cap for suction, and then removed. The sediment was extruded from the corer and the top 3 cm of sediment was cut and stored in a zip-seal bag and then stored on ice or at 4 °C until analysis. All samples were analyzed for ash-free dry mass (AFDM, 550 °C, 1 hour, Steinman and Ogdahl 2016) and TP (persulfate digestion, USEPA 1983). All P analyses were conducted on a SEAL autoanalyzer at AWRI (SEAL Analytical, Mequon, Wisconsin).

Both the two-stage and reference reaches were separated into five sections of equal length for the P survey, hereafter called “intervals” and numbered 1-5 (cf. Fig. 2.3). To determine the variability of soil TP throughout the two-stage and reference reaches, soil cores were collected within each of the five two-stage and five reference intervals (Fig. 2.3).

A separate set of sediment and soil cores (top 3 cm) was collected at the downstream end of each reach for fractionation and isotherm analyses in spring and fall as follows. Two-stage: one core in the channel and one core in the middle of each bench; Reference: one core in the channel and one core 1 m above the water level in the channel on each ditch bank. Samples were analyzed within one week of collection. Sediment and soil samples (~2 g) were sequentially fractionated at room temperature using modified methods from Hupfer et al. (2009). 25 mL of the following extractions were used: 1M NH_4Cl shaken for 30 minutes extracting loosely-bound and pore water P; 0.11 M $NaHCO_3$ $Na_2S_2O_4$ (buffered dithionite, BD) shaken for one hour extracting Fe- bound and redox sensitive P; 1M NaOH shaken for 16 hours extracting Al-bound P and organics; and 0.5 M HCl shaken for 16 hours extracting Ca-bound P. A separate pore water extraction was not possible due to limited water content and samples were subjected to only one cycle of reagent and shaking per step. Isotherms were used to calculate the EPC_0, which is the concentration at which there is no net exchange of P between the sediment/soil and the water column. Isotherm procedures were modified from Steinman and Ogdahl (2013) and used eight different P concentrations: 0, 0.01, 0.1, 1, 10, 50, 100, 500 mg P/L made from KH_2PO_4 in 0.01M KCl. Testing was conducted in triplicate: two “live” samples and one “dead” sample with three drops of chloroform added to halt microbial respiration. Total sample number equaled 296 including eight blanks. After addition of 20 mL P solution to 3g sediment or soil, samples were shaken for 24 hours and then centrifuged. The supernatant was removed, filtered (0.45 µm GF), and then analyzed for SRP. The P concentrations were graphed against the starting concentration of solution added. The slope of the exchange and the y-intercept was used to calculate EPC_0

(EPC_0 = -1* y intercept *$slope^{-1}$). The EPC_0 values are compared with the SRP values from surface water sampled in the ditch at the time of coring to determine if the sediments/soils are retaining P ($SRP>EPC_0$), releasing P ($SRP< EPC_0$) or if the system is at equilibrium ($SRP\approx EPC_0$).

Particle- size analysis was conducted using standard methods (ASTM 2007) on sediment and soil sampled from sites for isotherm and fractionation analysis for the following fractions: >2 mm = gravel/cobble, 1-2 mm = very coarse sand, 0.5-1 mm = coarse sand, 250-500 µm = medium sand, 125-250 µm = fine sand, 63-125 µm = very fine sand, <63 µm = silt/clay.

Vegetation

Using the same reach intervals that were applied for soil TP analysis, vegetation cover was visually estimated on a 0-5 Braun-Blanquet scale in both ditches and reaches. Aboveground biomass samples were collected only in the Z ditch due to low cover T ditch and limits on sample. Aboveground biomass was sampled within a PVC square of fixed area (0.25 m^2) at intervals 1, 3, and 5 in the two-stage and reference reaches of the Z ditch. For the benches and banks, the PVC square was lightly tossed perpendicular to the ditch channel at the distance mark to identify the first 0.25 m^2 sampling area. After the vegetation was cleared and stored in a large plastic bag, the square was flipped adjacent to the first sampling area, and another 0.25 m^2 aboveground vegetation was sampled for a total area of 0.5 m^2 at each interval, keeping left and right bench/bank separate. For the single channel site at each interval (1, 3, 5), 0.25 m^2 area of aboveground vegetation on each side of the channel (for a total of 0.5 m^2) was cut and stored in a large plastic bag. All samples were dried in the laboratory at ~60 °C until constant weight. Samples were then ground on a Wiley Mill and re-combined per site. Samples were re-dried and ashed in triplicate for analysis of TP (Berthold et al. 2015).

Periphyton

Periphyton was grown on eight 2.54 cm^2 pre-ashed, unglazed ceramic tiles adhered to a brick that was buried flush with the channel bottom at the downstream end of each reach. The tiles were collected monthly; at each collection, every other tile was removed from the brick for filtering and TP analysis. The brick was discarded and replaced with a new brick in the same position within the ditch with uncolonized tiles after each sampling monthly from April to December.

Periphyton was removed from the tiles by careful scraping. Samples were prepared for ash-free dry mass (AFDM) and P content analyses by filtering onto pre-combusted and weighed GF/F filters (Steinman et al. 2006). For AFDM (μg/cm^2), filtered samples were placed in pre-combusted and weighed aluminum weigh boats then dried at 70 °C until constant weight, then oxidized at 550 °C for 1 hour. After oxidation, the samples were digested with persulfate and then analyzed for TP (see *Sediment* above).

Phosphorus Stock

Mean P concentrations from each stock (combined sediment and soil, vegetation, and periphyton) for each reach was compared visually in mg P per gram of dry material and as a relative percent of the total area found within a reach. Area was calculated based on length of reach and mean width data for benches and channel.

Data Analysis

Z and T ditches were not compared statistically with each other as they are separate systems and represent different ages since construction of the two-stage reach. However, reference vs. two-stage reaches were compared within each separate system using paired samples t-tests (or Wilcoxon tests, depending on data normality). Water TP and SRP, as well as soil TP, general vegetation cover, and periphyton TP were compared with paired samples t-tests or Wilcoxon tests. In-stream turbidity readings at 10-minute intervals were averaged per hour and then compared with paired samples t-tests. For fractionation data, benches (or banks) and channel sites were compiled per site in order to compare differences in each fraction between two-stage and reference with ANOVA. Sediment and soil measurements were combined in order to focus the comparison on the potential differences between two-stage and reference. Due to small sample sizes, inferential analyses of isotherm and fractionation measurements among specific positions (left, channel, right) within each ditch were not possible; graphical representation allowed for comparisons described in the results below. In order to compare potential fraction changes between season, two-stage and reference data were combined but paired to retain treatment differences. T-test or Wilcoxon tests were used to compare spring and fall for each fraction for each ditch. All analyses were conducted in R (v99.447) and significance was set at $p < 0.05$.

Results

Water Quality

West Michigan experienced higher than average precipitation and temperatures in 2016. Mean annual precipitation for 2016 was 104 cm compared to the 1895-2015 average of 88 cm and mean annual temperature for 2016 was 10.4 °C compared to the 1895-2015 average of 8.7 °C (NOAA, www.ncdc.noaa.gov/sotc/national/201613). One snowmelt event and three storm events were sampled for a total of four stormflow samplings per ditch during 2016. Baseflow and stormflow TP and SRP in the Z ditch were not significantly different between two-stage and reference reaches (all $p > 0.05$, Table 2.2), although stormflow P concentrations were more than double those of baseflow (Table 2.2) in both reach types. Baseflow TP and SRP in the T ditch were significantly lower in the two-stage than in the reference reach (TP: $p = 0.009$; SRP: $p = 0.011$, Table 2.2). Stormflow TP and SRP in the T ditch were not significantly different (both $p > 0.05$, Table 2.2). In T, the increase in P concentration after storm events was even greater than in Z, increasing by more than an order of magnitude from baseflow in both reach types (Table 2.2).

Turbidity measurements from installed sensors were averaged across the entire year and include baseflow and stormflow values. Turbidity was correlated with rainfall for all sites ($p < 0.0001$). Z ditch turbidity was significantly lower in the two-stage compared to the reference ($p < 0.0001$, Table 2.2). Due to several issues with equipment, approximately half of the turbidity data for T two-stage were not usable. To compare paired data between T ditch two-stage and reference, the corresponding data from the reference also was not used, although there were still more than 6200 observations. T ditch turbidity was significantly higher in the two-stage compared to the reference ($p < 0.0001$, Table 2.2).

DO, pH, and temperature were not significantly different between two-stage and reference for baseflow or stormflow in the Z ditch (all $p > 0.1$, Table 2.2). In the T ditch, DO was not significantly different between two-stage and reference for baseflow or stormflow ($p = 0.443$, $p = 0.598$, Table 2.2). T ditch pH and temperature were significantly higher in the two-stage compared to the reference for baseflow ($p=0.016$, $p=0.011$ respectively) but not for stormflow ($p = 0.120$, $p = 0.561$ respectively, Table 2.2).

Sediment and Soil

Mean combined sediment and soil %OM was not significantly different between two-stage and reference for both Z and T (Table 1). TP concentrations in the bench and bank soil varied between 99 and 1000 mg/kg (dry weight) for both Z and T ditch systems. Mean combined sediment and soil TP was significantly lower in the two-stage compared to the reference in both Z and T ditch systems (Table 2.1, Fig. 2.4, Fig. 2.5).

In both the Z and T ditches, particle size distributions revealed that the major size class by weight was sand both on the benches and in the channel for both two-stage and reference (Fig. 2.6). Major P fractions during both spring and fall were Al-bound and organic P in Z and both Ca-bound and Al-bound and organic P in T (Figs. 2.7 and 2.8). Tests for significance between ditch reaches for each fraction in each season with left, channel, and right positions combined (i.e. spring Z 2-S NH_4Cl-P compared to spring Z Ref NH_4Cl –P) did not reveal any significant differences ($p > 0.05$). Single position fraction comparisons between reaches (i.e. spring Z 2-S channel NH_4Cl -P compared to spring Z Ref channel NH_4Cl -P) were not possible due to small sample size. Tests for significance between spring and fall for each fraction for both Z and T did not reveal any differences ($p >0.05$).

In the Z ditch at the time of spring sampling, SRP concentrations in the water column were greater than the EPC_0 values, suggesting that P was retained at all sites and that the sediment was serving as a sink (Fig. 2.9). At the time of fall sampling, EPC_0 values suggest again that P was retained in the sediment at all sites with the exception of both the left and right banks within the reference reach (Fig. 2.9). EPC_0 results were different in the T ditch; at the time of spring sampling, results suggest that P was retained at only the two-stage left bank and the reference channel (Fig. 2.10). In the fall, results indicate that left and right banks in the reference reach remained potential sources of P to the water column, as was seen in the spring, but the channel and right bank locations in the two-stage ditch had switched from potential P sources in the spring to P sinks in the fall (Fig. 2.10).

Vegetation

Mean vegetative P concentration was slightly lower, but not significantly so, in the two-stage compared to the reference reach of Z (Table 2.3). Again, P tissue content was not measured in T. In the Z ditch, mean vegetation cover was not significantly different between the two-stage and reference reaches ($p = 0.550$, Table 2.3). In the T ditch, the two-stage reach had lower mean total vegetation cover than the reference reach ($p = 0.003$, Table 2.3), which was due to differences in mean bench cover ($p < 0.0001$). The majority of the vegetation consisted of grasses as well as some minor clover, goldenrod, and willow plants in both the two-stage and reference in both Z and T ditches.

Periphyton

Seven sets of periphyton samples were sampled and analyzed for the Z ditch. Average periphyton P concentration was 7.0 ± 10.0 μg P/ cm^2 in the two-stage reach and 11.0 ± 6.0 μg P/ cm^2 in the reference reach (Table 2.3). The reference periphyton P was barely significantly greater than the two-stage P ($p=0.049$). Only six complete sets of periphyton samples were sampled and analyzed for the T ditch due to loss of artificial substrates in the reference during a storm in September. Two-stage average P was 16.0 ± 7.0 μg P/ cm^2 and mean P in the reference reach was 23.0 ± 8.0 μg P/ cm^2 (Table 2.3). There was no difference between the reference and two-stage μg P/ cm^2 ($p=0.273$).

Phosphorus Stock

The relative importance of each phosphorus stock was similar for two-stage and reference for both ditches. In the Z ditch, vegetation held the most P per gram of dry material, followed by periphyton and then combined sediment and soil (Fig. 2.11). For the T ditch, the vegetative normalized P was almost three times the amount from periphyton and combined sediment and soil (Fig. 2.11). When P was extrapolated to the entire reach, including either the area (for combined sediment and soil) or total biomass (for vegetation and for periphyton), the importance of each P stock changed (Fig. 2.12). For both Z and T and both two-stage and reference, the sediment held ≥ 96% of the total P for the reach. The periphyton holding was <1% and the vegetation holding made up the other ~3%.

Discussion

The web of P interactions between terrestrial and aquatic systems and between organic and inorganic forms makes it difficult to track P movement through ecological systems. N can be tracked via stable isotope analyses and has been monitored in two-stage agricultural systems around the Midwest. However, there is no stable isotope analog for P, so relatively little is known about P movement and standing stocks of P within two-stage systems (Kallio et al. 2010). Christopher et al. (2017) applied what is known about P movement in two-stage systems to a model to estimate the potential retention power of the two-stage within a watershed; if 25% of all agricultural ditches in the River Raisin watershed of Michigan were converted to two-stage, the model estimated a 12% increase in P retention. However, there is some uncertainty in this estimate, given that most of P was not measured directly, but derived from turbidity-TP relationships and then applied to a watershed currently without two-stage ditches. By increasing the information known about how two-stage systems function and how P is stored, both modeling and management can be improved.

Water TP and SRP concentrations were not significantly different between the two-stage and reference reaches in Z. This finding is consistent with other studies (Davis et al. 2015, Mahl et al. 2015), where a reduction in P via two-stage ditches was observed in only a small fraction of streams. This lack of change could be due to the limited length of the two-stage reach. In a recent study, Collins et al. (2016) found that ADS in general have the potential to retain P but generally lack sufficient hydrologic residence time for P retention processes to occur. While determining residence time was not within the scope of my thesis, it is possible that these two-stage reaches are not long enough to sufficiently increase the residence time for optimal P retention.

Another limitation was the presence of tile drains and very shallow runoff tributaries that flowed off the field into the middle of the ditch reaches. The drainage entering these ways would not experience the same flow or interactions with soil, sediment, and vegetation as water moving through the reference into the two-stage reach. In the Z ditch, a buried tile drain in the middle of the two-stage and an off-field tributary at the downstream end of the two-stage were the only inputs identified during storm sampling. In the T ditch, two large off-field tributaries that could not be successfully hardened would have had an impact on the two-stage. Any potential inputs that existed in the reference were not large enough to be identified. Due to limits on time and funding for sample analyses, the P load from these ephemeral but potentially important inputs (e.g., Clement and Steinman 2017) could not be measured.

Water TP and SRP concentrations were lower in the T ditch two-stage reach compared to the reference at baseflow. The difference in P concentrations between the reference and two-stage reaches could be due to the more recent excavation of the benches in the 2-stage reach. The T two-stage was constructed in November 2015 and little to no vegetation had grown on the benches before winter snowmelt and spring rainfall caused significant erosion. This erosion also changed bench structure and location (Fig. 2.13). At the end of the 2016, with higher than average rainfall, approximately 60% of the channel ran through what was originally constructed as the right bench. Some water did flow and was stored within the historic channel. Due to these morphological changes, baseflow in certain portions of the reach could have been more similar to sheetflow, thereby increasing the water to soil contact surface area and allowing for more retention of P within the two-stage reach.

Most of the eroded sediment and soil moving through the two-stage reach occurred during high flow events and any impact to TP measurements would have been masked by high TP throughout the ditch system. This erosion did increase the turbidity in the two-stage and resulted in a significantly higher full-year mean turbidity compared to the reference. The more stable and slightly older Z ditch two-stage system did not experience significant erosion and had significantly lower turbidity compared to the reference as expected. Both TP and turbidity were linked with rainfall and were orders of magnitude higher during stormflow compared to baseflow.

Along with significant differences in TP and SRP concentrations between two-stage and reference within the T ditch, pH and temperature were significantly lower in the reference compared to the two-stage. While these water quality parameters were statistically different, the relatively small differences may not be seen in older, highly vegetated two-stage systems. The T ditch two-stage was more open and had less vegetation due to the timing of construction and erosion of the benches. Therefore, the two-stage had less shading from banks or vegetation, which would have allowed more solar warming. Large mats of unidentified filamentous algae were present in the shallow bench-turned-channel flow for most of the growing season also due to the lack of shading compared to the reference (personal observation). This would have increased oxygen production from photosynthesis in the water column but the warmer temperatures likely offset the water's ability to hold more DO. Also, the increased photosynthesis could account for the higher pH values in the 2-stage reach (Welch and Jacoby 2004).

Soil TP was significantly lower in the two-stage compared to the reference in both Z and T ditches. This is likely due to construction and age of the two-stage reach. When the two-stage benches are excavated, the soil that had been exposed to runoff, drainage flow, and associated vegetation growth is removed. This leaves behind soil that is typically lower in P (Steinman and Ogdahl 2016). Other studies have seen varying TP trends with increasing depth (Wang et al. 2011, Wang and Liang 2015) but only tested the soil up to 35 cm deep. When bench construction occurred at both sites in my study, at least 0.5 m of soil was excavated. As P adsorption, sediment deposition, and vegetation growth and decay occur over time, P concentrations in the two-stage soil are likely to increase. Future work should test changes in total suspended solids and measure sediment deposition within two-stage ditches so that P transported in suspended sediment particles can be measured. Knowing the major form of transported P in these ditch systems would increase the efficacy of any future management decisions.

The major P fraction at all sites in the Z two-stage and reference sediment and soil was the Al- and organics fraction. The P bound in this fraction is not as readily available to be released and taken up by biota compared to the loosely bound and redox sensitive fractions. The major P fractions at sites within the T two-stage and reference sediment and soil were the Al- and organics fraction or the Ca- fraction; each site had the same major fraction in both spring and fall. Both of these fractions are relatively stable and less likely to be bioavailable compared to the other fractions. This suggests P spends very little time within the loosely bound fraction in the soil of these systems in the spring and fall. The source of the Al binding the P is found within clays and the Ca is more commonly found within gravel and sand (Reddy et al. 2000). However, the particle size distribution showed a larger amount of sand present compared to clay by weight. Since less P is absorbed by sandy soils, the relative amount of P held within the ditch could still be within the Al- and organics fraction. In the T ditch, again the major soil/sediment particle type was sand. The relative amounts of sand to clay does match the trend between the major fractions Al- and Ca-bound P.

The loosely bound and pore water fraction was consistently the smallest fraction found in our study followed by redox sensitive Fe-bound P. Little loosely bound P could suggest low cyclic exchange of the more bio-available sediment/soil P between the water and periphyton interface and sediments (Samanta et al. 2015) or that P spends very little time within the loosely bound fraction. Drying and rewetting of soils (e.g. two-stage benches between stormflow events) has been shown to increase the movement of loosely bound and pore water P compared to other fractions including redox sensitive P (Kinsman-Costello et al. 2016). This suggests that drying cycles of the two-stage benches throughout the summer might not significantly increase the release of P during stormflow inundation.

The calculated EPC_0 values in this study are similar to EPC_0 values from soil sampled in other agricultural areas (EPC_0: 0.022 to 0.284 mg/L, McDowell et al. 2017; EPC_0: 0.02 to 0.12 mg/L, Hongthanat et al. 2016). The Z ditch reference EPC_0 bank values increased from spring to fall and suggested that the banks were a possible source of P later in the year, possibly due to mineralization of vegetation. The Z two-stage EPC_0 values suggested combined sediment and soil served as a P sink in the two-stage throughout both seasons despite an increase in EPC_0 in the left bench. T ditch reference EPC_0 values increased from spring to fall suggesting a slight decrease in P retention in the fall. While the two-stage left bench EPC_0 value increased slightly, the channel and right bench EPC_0 values decreased, suggesting an increase in P retention. My initial thought was that this decrease in EPC values in the two-stage was due to the reduction in loosely bound P. However, there were no significant changes in the loosely bound fractions between spring and fall that would support this. It is possible that more mobile smaller particles were lost from the two-stage exposing more clay and increasing retention (Smith et al. 2005) but particle size distribution data was not collected for both spring and fall. These EPC_0 values were compared with the SRP present in the water column at the time of sampling. During floodplain and bank inundation, water column SRP was often an order of magnitude higher. Hence, some of these sites still have the potential to release P under high flow conditions.

In this study, only above ground vegetation biomass was sampled to avoid disrupting the existing soil integrity within the ditches. Root structure can be a significant portion of plant biomass and some studies suggest a major portion of the P is held within the root structure of plants (Moore and Kroger 2011; Teng et al. 2013). Hence our analysis of P content in vegetation is an underestimate and therefore conservative. There was significantly less vegetation cover on the two-stage benches of T and therefore this sparse vegetation would not likely have been a significant factor in the reduction of water column TP and SRP in the T ditch two-stage reach.

The slight significant difference in normalized periphyton P in the Z ditch is likely due to the extreme shading from channel and bench vegetation that occurred directly over the artificial substrate in the two-stage for most of the late summer months (personal observations). It is unclear if this reduction comes from a reduction in the cellular TP or if it is a reduction in total biomass that reduces the area-specific TP overall. Mixed results from studies on the effects of light and nutrients on periphyton growth and nutrient content have shown other unknown important factors impact algal nutrient uptake and retention. Hill et al. (2009) found that as light availability increase, the biomass-specific concentration of P in algae does not decrease but the biovolume of algal cells tends to increase while Fanta et al. (2010) found that algal P content decreased with increasing light. Drake et al. (2012) found that light availability and carbon to P ratios within cells were inversely related to each other with regards to how they impacted algal growth.

The vegetation and periphyton within these ditches have more P per dry mass compared to the combined sediment and soil, so it is possible that riparian plants and periphyton are seasonally important sinks of P in two-stage ditches. This important function may by temporally variable because of the cycle of growth and decay throughout the year. During the non-growing season when vegetation is not taking up as much water and nutrients, P loads entering these agricultural ditches through tile drains tend to be higher and can move more easily downstream (Clement and Steinman 2017). Bench vegetation can stabilize the soil, decrease erosion, and decrease storm transported nutrients during the growing season but the root system left behind can also stabilize the benches during the non-growing season. In-stream vegetation and periphyton can affect the pH of the surrounding water column when photosynthesizing (Welch and Jacoby 2004), thereby affecting the Al- and organics as well as the Ca-bound P fractions within the surface layer of sediment.

Combined sediment and soil held the most P of the measured stocks at the reach scale even though the vegetation and periphyton had the most P per dry mass. While examining the role of riparian plants in P retention and ditch stability should be considered for future study, the importance of this P sink is minimal compared to the sediment and soil. Any P held within the vegetation and periphyton will eventually be released through decomposition and mineralization of organic matter while nonreactive P and stably bound P such as Ca-P accumulated within the sediment is likely to stay bound across season and become legacy P. Attention should be paid to the interactions between sediment and the water column as P inputs into the drainage system are decreased through management as this may change the balance and release legacy P (Reddy et al. 2011). It is important to include sediment and soil analyses, not just water analyses in management of agricultural ditch systems. Future management and research should focus on increasing the P adsorption capacity of the sediment and soil in order to increase the effectiveness of two-stage systems.

Conclusions

The two-stage ditches within the Macatawa watershed have the potential to retain P within their sediment and soil. The majority of the P stored within the sediment and soil is bound within the more stable Al- and Ca- bound fractions. There was less TP within the combined sediment and soil of the two-stage reaches compared to the reference reaches suggesting this ability to retain P will continue until equilibrium is reached between the sediment/soil P and water column P. While short reach length and erosion in recently constructed two-stage systems may impair the effectiveness of the two-stage, the reduced P release potential and the continued retention of P within sediment and soil of Macatawa two-stage ditches suggest that their effectiveness will increase as they age and stabilize. Further monitoring and research will be required to determine if reductions in water column P will increase over time.

Acknowledgements

The author would like to thank Al Steinman, Maggie Oudsema, Brian Scull, Mike Hassett, Nicole Hahn, Kim Oldenborg, and Mark Luttenton for assistance in the field and laboratory. I would also like to thank all the sources of funding for this project: the Donner Family, Grand Valley State University Graduate Assistantship and Presidential Grant, the Michigan Space Grant Consortium, the Michigan Chapter of the North American Lake Management Society, and the Society for Freshwater Research. I would also like to thank Michael Hupfer of the Leibniz-

Institute of Freshwater Ecology and Inland Fisheries for assistance with the phosphorus fractionation scheme.

References

Baldwin, D.S., Mitchell, A.M. 2000. The effects of drying and re-flooding on the sediment and soil nutrient dynamics of lowland river-floodplain systems: a synthesis. Regul. River. 16: 457-567.

Berthold, M., Zimmer, D., Schumann, R. 2015. A simplified method for total phosphorus digestion with potassium persulphate at sub-boiling temperatures in different environmental samples. Rostock. Meeresbiolog. Beitr. 25: 7-25.

Blann, K.L., Anderson, J.L., Sands, G.R., Vondracek, B. 2009. Effects of agricultural drainage on aquatic ecosystems: a review. Critical reviews in environmental science and technology 39: 909-1001

Clement, D.R., Steinman, A.D. 2017. Phosphorus loading and ecological impacts from agricultural tile drains in a west Michigan watershed. J. Gt. Lakes Res. 43: 50-58.

Collins, S.D., Shukla, S., Shrestha, N.K. 2016. Drainage ditches have sufficient adsorption capacity but inadequate residence time for phosphorus retention in the Everglades. Ecol. Eng. 92: 218-228.

D'Ambrosio, J.L, Ward, A.D., Witter, J.D. 2015. Evaluating geomorphic change in constructed two-stage ditches. J. Gt. Lakes Res. 51:910-922.

Davis, R.T., Tank, J.L., Mahl, U.H., Winikoff, S.G., Roley, S.S. 2015. The influence of two-stage ditches with constructed floodplains on water column nutrients and sediments in agricultural streams. J. Am. Water. Res. Assoc. 51: 941-955.

Dieter, D., Herzog, C., Huper, M. 2015. Effects of drying on phosphorus uptake in re-flooded lake sediments. Environ. Sci. Pollut. R. 22: 17065-17081.

Dodds, W.K., Bouska, W.W., Eitzmann, J.L., Pilger, T.J., Pitts, K.L., Riley, A.J., Schloesser, J.T., Thornbrugh, D.J. 2009. Eutrophication of U.S. freshwaters: analysis of potential economic damages. Environ. Sci. Technol. 43: 12-19.

Drake, W.M., Scott, J.T., Evans-White, M., Haggard, B., Sharpley, A., Rogers, C.W., Grantz, E.M. 2012. The effect of periphyton stoichiometry and light on biological phosphorus immobilization and release in streams. Limnology 13: 97-106.

Fanta, S.E., Hill, W.R., Smith, T.B., Roberts, B.J. 2010. Applying the light: nutrient hypothesis to stream periphyton. Freshwater Biol. 55: 931-940

Hassett, M., Oudsema, M., Steinman, A.D. 2017. Project Clarity 2016 Annual Monitoring Report. www.gvsu.edu/cms4/asset/DFC9A03B-95B4-19D5-F96AB46C60F3F345/final_report_2016_water_year.pdf

Hill, W.R., Fanta, S.E., Roberts, B.J. 2009. Quantifying phosphorus and light effects in stream algae. Limnol. Oceanogr. 54: 368-380.

Hongthanat, N., Kovar, J.L., Thompson, M.L., Russell, J.R., Isenhart, T.M. 2016. Phosphorus source-sink relationships of stream sediments in the Rathbun Lake watershed in southern Iowa, USA. Environ. Monit. Assess. 188: 453-467.

Hupfer, M., Zak, D., Rofberg, R., Herzog, C., Pothig, P. 2009. Evaluation of a well-established sequential phosphorus fractionation technique for use in calcite-rich lake sediments: identification and prevention of artifacts due to apatite formation. Limnol. Oceanogr: Methods 7, 2009: 399-410

Jarvie, H.P., Sharpley, A.N., Flaten, D., Kleinman, P.J., Jenkins, A., Simmons, T. 2015. The pivotal role of phosphorus in a resilient water-energy-food security nexus. J. Environ. Qual. 44: 1049-1062.

Kallio, S., Ward, A., D'Ambrosio, J.L., Witter, J.D. 2010. Beyond the field: a look at agricultural ditch floodplains as a water quality BMP. CIGR XVII World Congress

Kinsman- Costello, L.E., Hamilton, S.K., O'Brien, J.M., Lennon, J.T. 2016. Phosphorus release from the drying and reflooding of divers shallow sediments. Biogeochemistry. DOI: 10.1007/s10533-016-0250-4.

MACC. 2012. Macatawa Watershed Management Plan Appendix H www.the-macc.org/watershed/overview/

MACC. 2014. Macatawa Watershed Annual Progress Report. www.the-macc.org/watershed/overview/

Mahl, U.H., Tank, J.L., S.S. Roley, Davis, R.T. 2015. Two-stage ditch floodplains enhance N-removal capacity and reduce turbidity and dissolved P in agricultural streams. J. Am. Water Res. Assoc. 51: 923-940.

McDowell, R.W., Elkin, K.R., Kleinman, P.J.A. 2017. Temperature and nitrogen effects on phosphorus uptake by agricultural stream-bed sediments. J. Eviron. Qual. Technical Reports: 1-7.

Moore, M.T., Kroger, R. 2011. Evaluating plant species-specific contributions of nutrient mitigation in drainage ditch mesocosms. Water Air Soil Pollut. 217: 445-454.

Nair, V.D., Clark, M.W., Reddy, K.R., 2015. Evaluation of legacy phosphorus storage and release from wetland soils. J. Environ. Qual. 44: 1956-1964.

NOAA. National Overview- Annual 2016. “State of the Climate”. www.ncdc.noaa.gov/sotc/national/201613.

Pant, H.K., Reddy K.R. 2001. Phosphorus sorption characteristics of estuarine sediments under different redox conditions. J. Environ. Qual. 30: 1474-1480.

Powell, G.E., Ward, A.D., Mecklenburg, D.E., Jayakaran, A.D. 2007 Two-stage channel systems: part 1, a practical approach for sizing agricultural ditches. J Soil Water Cons. 62: 277-286.

Project Clarity. Outdoor Discovery Center Macatawa Greenway. www.macatawaclarity.org

Psenner, R., Pucsko, R. 1988. Phosphorus fractionation: advantages and limits of the method for the study of sediment P origins and interactions. Arch. Hydrobiol. Beih. Ergebn. Limnol. 30: 43-59.

Reddy, K.R., D’Angelo, E.M., Harris, W.G. 2000. Biogeochemistry of Wetlands. Handbook of Soil Science. M.E. Sumner, ed. G89-119.

Reddy, K.R., Newman, S., Osborne, T.Z., White, J.R., Fitz, H.C. 2011. Phosphorus cycling in the Everglades ecosystem: Legacy phosphorus implications for management and restoration. Crit. Rev. Environ. Sci. Technol. 41: 149-186.

Roley, S.S., Tank, J.L., Stephen, M.L., Johnson, L.T., Beaulieu, J.J., Witter, J.D. 2012a. Floodplain restoration enhances denitrification and reach-scale nitrogen removal in an agricultural stream. Ecol. Appl. 22: 281-297.

Roley, S.S., Tank, J.L., and Williams, M.A. 2012b. Hydrological connectivity increases denitrification in the hyporheic zone and restored floodplains of an agricultural stream. J. Geophys. Res. 117:1-16.

Sallade, Y.E., and Sims, J.T. 1997. Phosphorus transformation sin the sediments of Delaware's agricultural drainageways: II. Effect of reducing conditions on phosphorus release. J. Environ. Qual. 26: 1579-1588.

Samanta, S., Debnath, D., Maitra, N., Banerjee, M., Chowdhury, A.N., Sharma, A.P., Manna, S.K. 2015. Sediment phosphorus forms and levels in two tropical floodplain wetlands. Aquat. Ecosyst. Health Manag. 18:467-474.

Scavia, D., Allan, J.D., Arend, K.K., Bartell, S., Beletsky, D., Bosch, N.S., Brandt, S.B., Briland, R.D., Daloglu, I., DePinto, J.V., Dolan, D.M., Evans, M.A., Farmer, T.M., Goto, D., Han, H., Hook, T.O., Knight, R., Ludsin, S.A., Mason, D., Michalak, A.M., Richards, R.O., Roberts, J.J., Rucinski, D.K., Rutherford, E., Schwab, D.J., Sesterhenn, T.M., Zhang, H., Zhou, Y. 2014. Assessing and addressing the re-eutrophication of Lake Erie: central basin hypoxia. J. Great Lakes Res. 40: 226-246.

Silvan, N., Vasander, H., Laine., J. 2004. Vegetation is the main factor in nutrient retention in a constructed wetland buffer. Plant Soil 258: 179-187.

Smith, D.R., Haggard, B.E., Warnemuende, E.A., Huang, C. 2005. Sediment phosphorus dynamics for three tile fed drainage ditches in northeast Indiana. Ag. Water Manag. 71: 19-32.

Steinman, A.D., Lamberti, G.A., Levitt, P. 2006. Biomass and pigments of benthic algae. Pgs 357-379. In: Methods in Stream Ecology. Hauer, R. and G. Lamberti (ed). Academic Press, NY.

Steinman, A.D., Ogdahl, M.E. 2013. Muskegon lake AOC habitat restoration design: Bear Lake hydrologic reconnection/wetland restoration. www.gvsu.edu/cms4/asset/DFC9A03B-95B4-19D5-F96AB6C60F3F345/bear_muck_final_report_final.pdf

Steinman, A.D., Ogdahl, M.E. 2016. From wetland to farm and back again: phosphorus dynamics of a proposed restoration project. Environ. Sci. Pollut. Res. 23:22596-22605.

Steinman, A.D., Ogdahl, M.E., Weinert, M., Uzarski, D.G. 2014. Influence of water level fluctuation duration and magnitude on sediment-water nutrient exchange in coastal wetlands. Aquat. Ecol. 48: 143-159.

Teng, W., Deng, Y., Chen, X.P., Xu, X.F., Chen, R.Y., Lv, Y., Zhao, Y.Y., Zhao, X.Q., He, X., Li, B., Tong, Y.P., Zhang, F.S., Li, Z.S. 2013. Characterization of root response to phosphorus supply from morphology to gene analysis in field-grown wheat. J. Exper. Bot. 64: 1403-1411.

USEPA. 1983. Methods for chemical analysis of water and wastes. USEPA, Washington, D.C.

Wang, C., Zhang, Y., Li, H., Morrison, R.J. 2013. Sequential extraction procedures for the determination of phosphorus forms in sediment. Limnology 14: 147-157.

Wang, L., Liang, T. 2015. Distribution characteristics of phosphorus in the sediments and overlying water of Poyang lake. PLOSone 10(5): e0125859.

Wang, L., Wang, G., Ge, X., Yu, Z., Gu, X., Cheng, Z., Xu, W., Zhang, F. 2012. Influence of submerged plants on phosphorus fractions and profiles of sediments in Gucheng Lake. Soil Sed. Contam. 21: 640-654.

Wang, Q., Li, Y., Ouyang, Y. 2011. Phosphorus fractionation and distribution in sediments from wetlands and canals of a water conservation area in the Florida Everglades. Water Res. Res. 47 W05550: 1-11.

Welch, E.B., Jacoby, J.M. 2004. Chapter 4 Nutrient Cycles, in: Pollutant Effects in Freshwater-Applied Limnology 3rd Ed. Taylor & Francis, New York, pp 56-85.

Figure Captions

Fig. 2.1 Comparison of traditional (A) and two-stage (B) cross-sectional shape. * denotes two-stage floodplain benches post construction and growth of vegetation.

Fig. 2.2 Location of ditches within Macatawa River watershed. Inset: location of watershed in lower peninsula of Michigan.

Fig. 2.3 Sampling locations for surface water sampling, soil TP survey, sediment and soil sampling for isotherms and fractionation, and vegetation sampling for bank/bench and channel edge samples within sample ditch system. Each reach divided into 5 equal length intervals. For lengths of reference and two-stage reaches see Table 1. Diagram, intervals, and symbols not to scale.

Fig. 2.4 Z Ditch variability of total phosphorus content in soil, April 2016 ($p < 0.05$). The dashed vertical line corresponds with connection between downstream end of the reference reach and the two-stage reach. Red lines correspond to mean TP in each reach. Two-stage TP was significantly lower than reference ($t=2.40$, $p=0.029$). Arrow indicates direction of drainage water flow. Soil cored to 8cm depth.

Fig. 2.5 T Ditch variability total phosphorus content in soil, October 2016 ($p<0.05$). The dashed vertical line corresponds with connection between downstream end of the reference reach and the two-stage reach. Red lines correspond to mean TP in each reach. Two-stage TP was significantly lower than reference ($t=4.24$, $p=0.001$). Arrow indicates direction of drainage water flow. Soil cored to 3cm depth.

Fig. 2.6 Particle size distributions for each site in % gravel (> 2 mm), % sand (2 mm- 63 μm), and % silt/clay (< 63 μm).

Fig. 2.7 Z Ditch fractionation, April and October 2016. Bars organized in reference and two-stage pairs based on position in ditch (left bank or bench, channel, and right bank or bench). Each layer in the bars corresponds with a respective fraction of P, from least stable to most stable. BD= buffered dithionite.

Fig. 2.8 T Ditch fractionation, April and October 2016. Bars organized in reference and two-stage pairs based on position in ditch (left bank or bench, channel, and right bank or bench).

Each layer in the bars corresponds with a respective fraction of P, from least stable to most stable. BD= buffered dithionite.

Fig. 2.9 Z Ditch Equilibrium Phosphorus Concentrations (EPC_0), April and October 2016. Red diamonds = soluble reactive phosphorus (SRP), Bars = EPC_0. Bars organized in reference and two-stage pairs based on position in ditch (left bank or bench, channel, and right bank or bench).

Fig. 2.10 T Ditch Equilibrium Phosphorus Concentrations (EPC_0), April and October 2016. Red diamonds = soluble reactive phosphorus (SRP), Bars = EPC_0. Bars organized in reference and two-stage pairs based on position in ditch (left bank or bench, channel, and right bank or bench).

Fig. 2.11 Standing stock comparisons in mg P/ g dry material.

Fig. 2.12 Standing stock comparisons in mg P/ m^2 in relative percentages. Total reach calculations (kg P/ reach) are overestimations and assume sediment/soil at 3cm depth for whole reach area, vegetation consistent for the whole bench area, and periphyton consistent for the whole channel area.

Fig. 2.13 Morphology of two-stage reaches right after construction (A, C) and after one year of weathering (B, D). T ditch (A, B, looking downstream) experienced erosion and channel changes; after one year 60% of the effective channel was running through the right bench area. Z ditch (C, D, looking upstream) is very stable and experienced very little erosion or morphologic changes.

Tables

Table 2.1 Ditch and soil characteristics and general agricultural practices in adjacent fields (mean ± one standard deviation): "2-S"= Two-stage reach, "Ref"= Reference reach, "TP"= total phosphorus (mg/kg dry soil), "%OM"= % organic matter. TP and OM values were tested for statistical differences between the 2-S and Ref reaches for each individual stream: Statistically significant difference ($p < 0.05$) shown in bold, ❖ from Web Soil Survey, USDA Natural Resources Conservation Service.

Ditch and Treatment	**Z 2-S**	**Z Ref**	**T 2-S**	**T Ref**
Reach Length (m)	373	207	357	254
Soil Type ❖	Capac loam		Colwood silt loam and Capac loam	
Mean Soil TP	**471.9 ± 223.3**	**676.2 ± 150.5**	**236.4 ± 79.1**	**495.4 ± 176.1**
Mean Soil %OM	2.3 ± 0.6	6.0 ± 1.3	1.8 ± 0.7	4.1 ± 1.7
Adjacent Fields' Main Crops	Winter wheat, soybeans	Corn	Soybeans	Corn

Table 2.2 Water Quality means ± one standard deviation: "2-S"= Two-stage reach, "Ref"= Reference reach, "SRP"= soluble reactive phosphorus (mg/L), "TP"= total phosphorus (mg/L), "In-stream Turb"= all turbidity values for 2016 sampling duration including baseflow and stormflow (NTU), "DO"= Dissolved Oxygen, "Temp"= Temperature, water quality variables were tested for statistical differences between the 2-S and Ref reaches for each individual stream: * statistically significant difference ($p < 0.05$) shown in bold.

Ditch and Treatment	# observations	Z 2-S	Z Ref	T 2-S	T Ref
Mean TP Baseflow	10	0.32 ± 0.2	0.35 ± 0.2	**0.06 ± 0.1**	**0.09 ± 0.1**
Mean SRP Baseflow	10	0.24 ± 0.2	0.26 ± 0.2	**0.02 ± 0.04**	**0.05 ± 0.05**
Mean TP Stormflow	4	1.21 ± 0.3	1.12 ± 0.4	0.89 ± 0.2	0.98 ± 0.4
Mean SRP Stormflow	4	0.65 ± 0.2	0.67 ± 0.2	0.81 ± 0.4	0.62 ± 0.5
Mean In-stream Turb	Z: 5407 T: 3101	**55.8 ± 187.1**	**66.3 ± 186.3**	**67.9 ± 252.4**	**43.2 ± 136.4**
Mean DO mg/L Baseflow	9	12.0 ± 2.7	11.5 ± 3.6	13.3 ± 1.5	13.1 ± 3.4
Mean DO mg/L Stormflow	4	9.3 ± 3.4	9.5 ± 2.9	10.1 ± 2.4	9.7 ± 2.8
Mean pH Baseflow	7	7.9 ± 0.4	8.1 ± 0.3	**8.5 ± 0.1**	**8.3 ± 0.2**
Mean pH Stormflow	4	7.6 ± 0.2	7.5 ± 0.2	7.8 ± 0.3	7.7 ± 0.2
Mean Temp °C Baseflow	9	12.6 ± 6.1	13.0 ± 5.9	**16.8 ± 8.2**	**15.4 ± 7.4**
Mean Temp °C Stormflow	4	11.8 ± 7.35	11.6 ± 7.4	8.8 ± 7.6	8.8 ± 7.5

Table 2.3 Vegetation and periphyton results ± one standard deviation: "2-S"= Two-stage reach, "Ref"= Reference reach, "Veg" = vegetation, all biomass values in kg dry wgt/m^2, "P" = Phosphorus, values in g/m^2 dry biomass. There are no biomass or vegetation P content values for the T ditch due to limits on sample analysis. Data were tested for statistical differences between the 2-S and Ref reaches for each individual stream. Statistically significant difference ($p < 0.05$) shown in bold.

Ditch and Treatment	**Z 2-S**	**Z Ref**	**T 2-S**	**T Ref**
Major Veg-Bench	Grasses, clover	Grasses, goldenrod	Miscellaneous, grasses	Grasses, willow
Major Veg-Channel	Grasses, bulrush	Grasses, willow	Grasses	Grasses, willow
Mean Cover (0-5) Bench	4.1	4.6	**1.4**	**5.0**
Mean Cover (0-5) Channel	3.8	3.4	4.0	3.0
Mean biomass Bench	0.22 ± 0.10	0.27 ± 0.19	NA	NA
Mean biomass Channel	1.24 ± 0.15	0.43 ± 0.36	NA	NA
Mean Veg P content	1.57 ± 1.78	1.02 ± 0.88	NA	NA
Periphyton P content	0.07 ± 0.10	0.11 ± 0.06	0.16 ± 0.07	0.23 ± 0.08

Figures

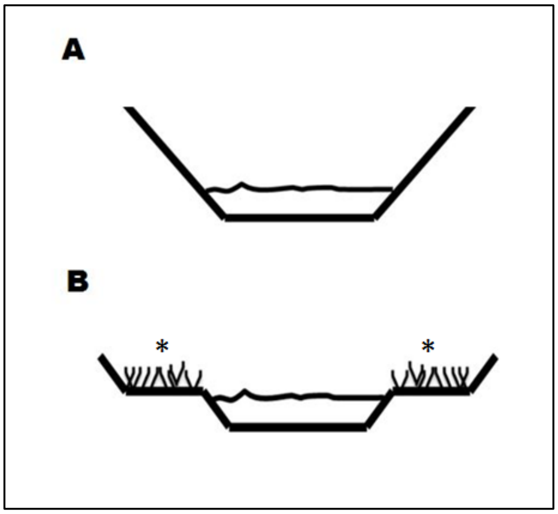


Fig. 2.1

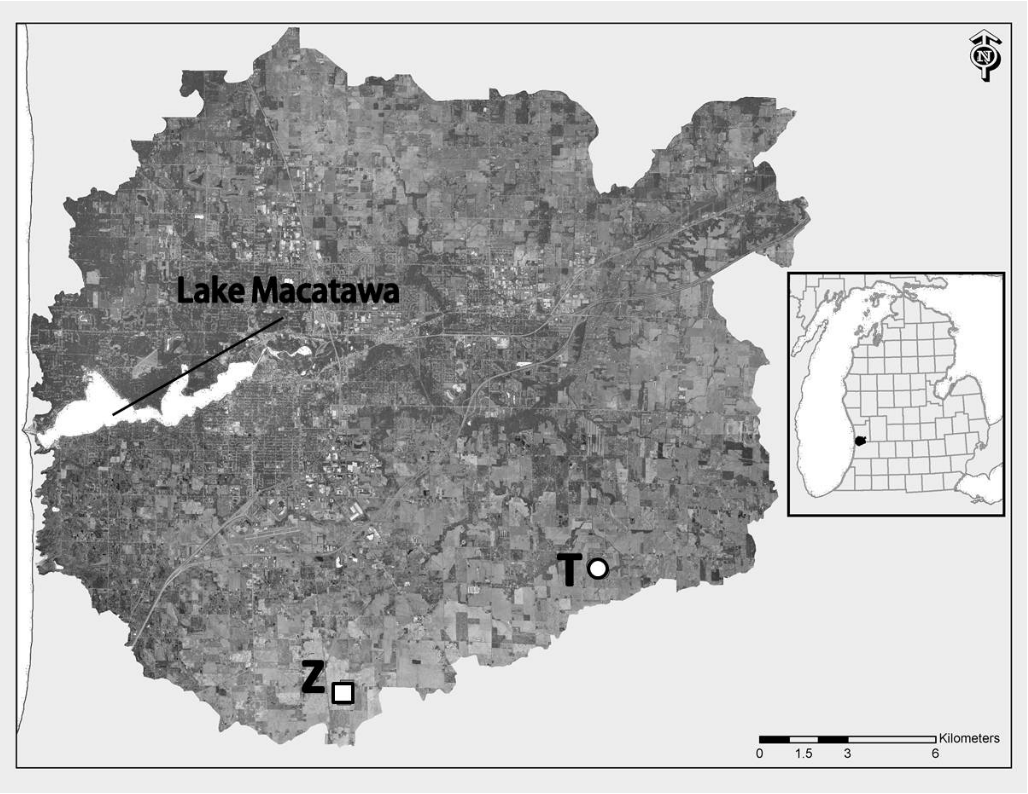


Fig. 2.2

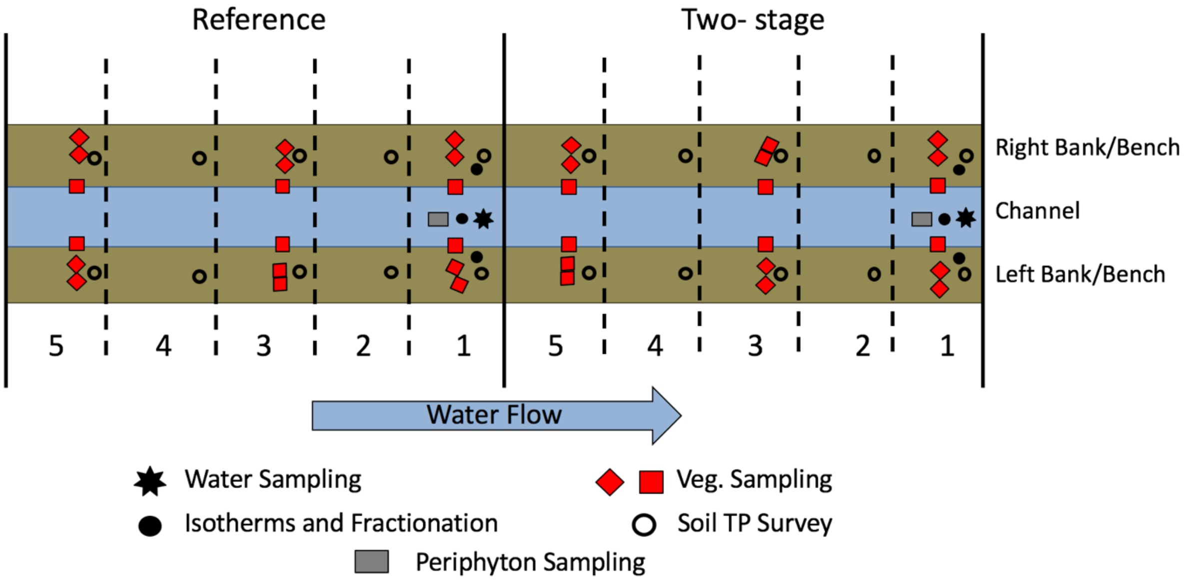

Reference
Two- stage
Right Bank/Bench
Channel
Left Bank/Bench
5
4
3
2
1
5
4
3
2
1
Water Flow
Water Sampling
Veg. Sampling
Isotherms and Fractionation
Soil TP Survey
Periphyton Sampling

Fig. 2.3

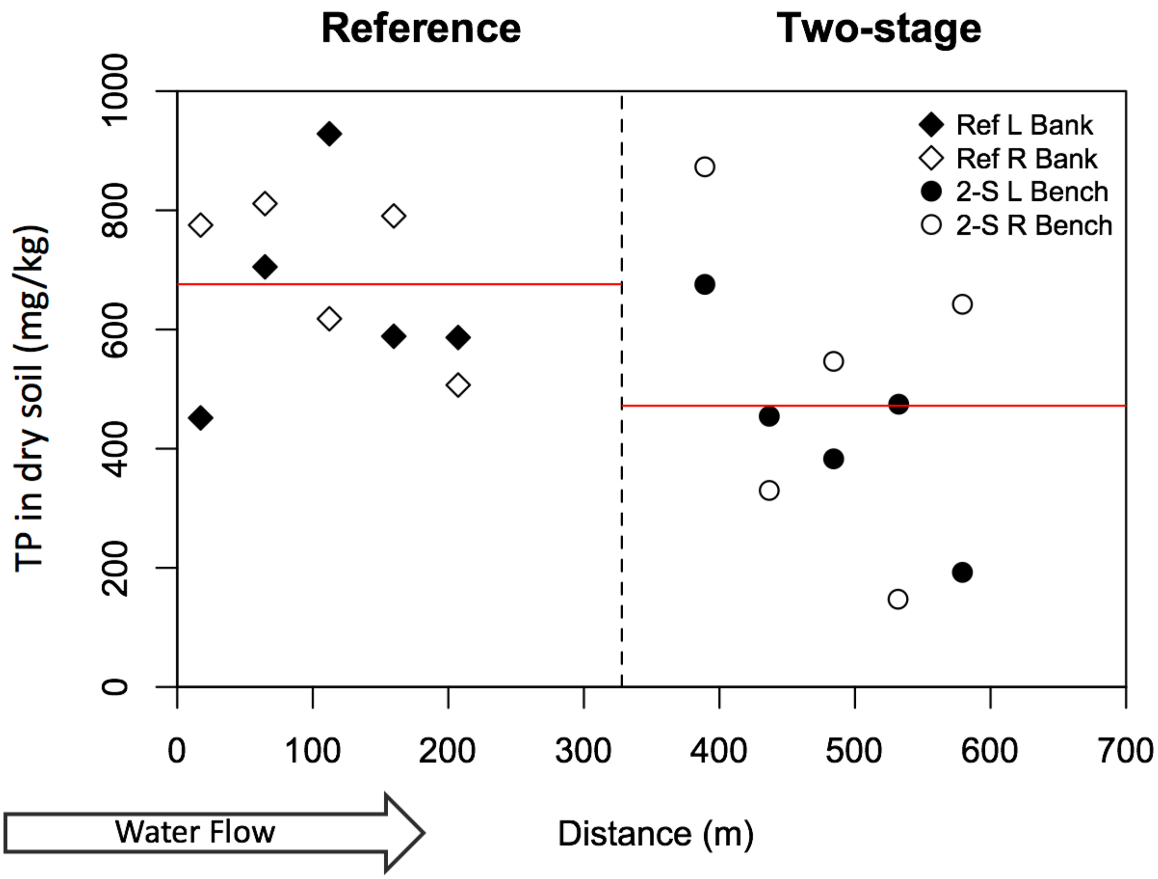
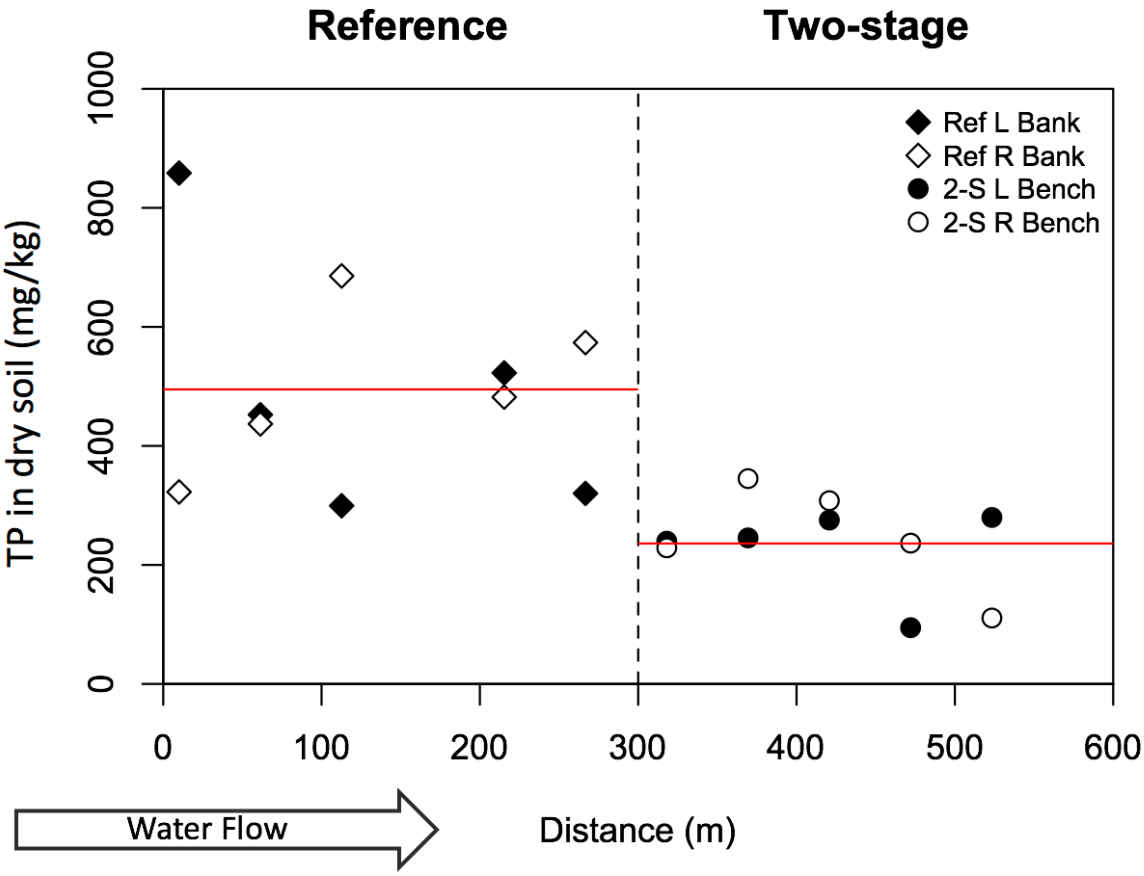

Reference
Two-stage
TP in dry soil (mg/kg)
0
200
400
600
800
1000
0
100
200
300
400
500
600
700
Distance (m)
Water Flow
Ref L Bank
Ref R Bank
2-S L Bench
2-S R Bench

Fig. 2.4



Reference
Two-stage
TP in dry soil (mg/kg)
0
200
400
600
800
1000
0
100
200
300
400
500
600
◆ Ref L Bank
◇ Ref R Bank
● 2-S L Bench
○ 2-S R Bench
Water Flow
Distance (m)

Fig. 2.5

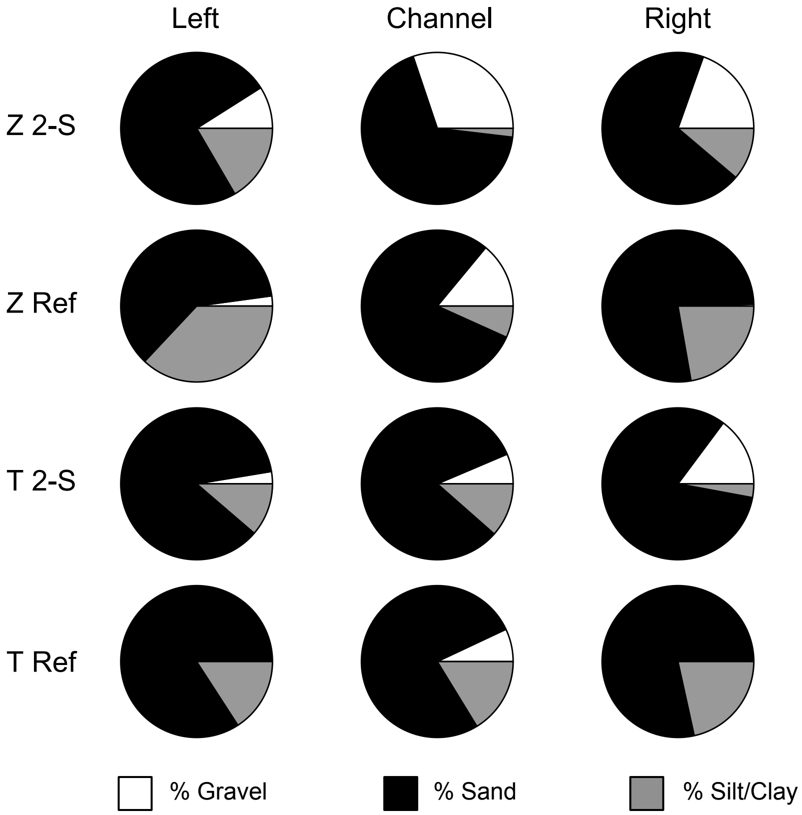

Left
Channel
Right
Z 2-S
Z Ref
T 2-S
T Ref
% Gravel
% Sand
% Silt/Clay

Fig. 2.6

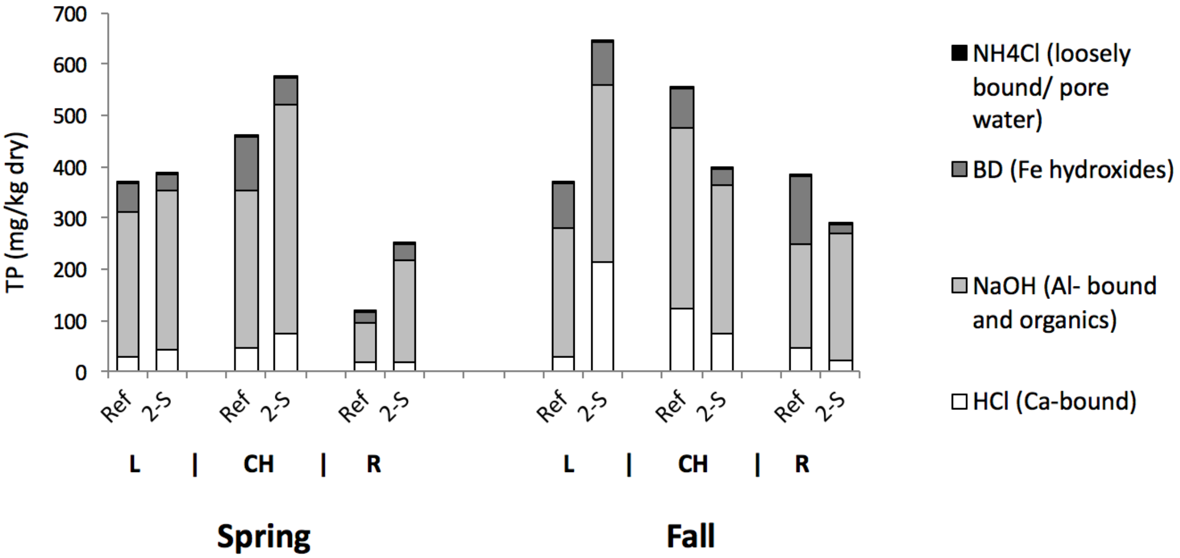
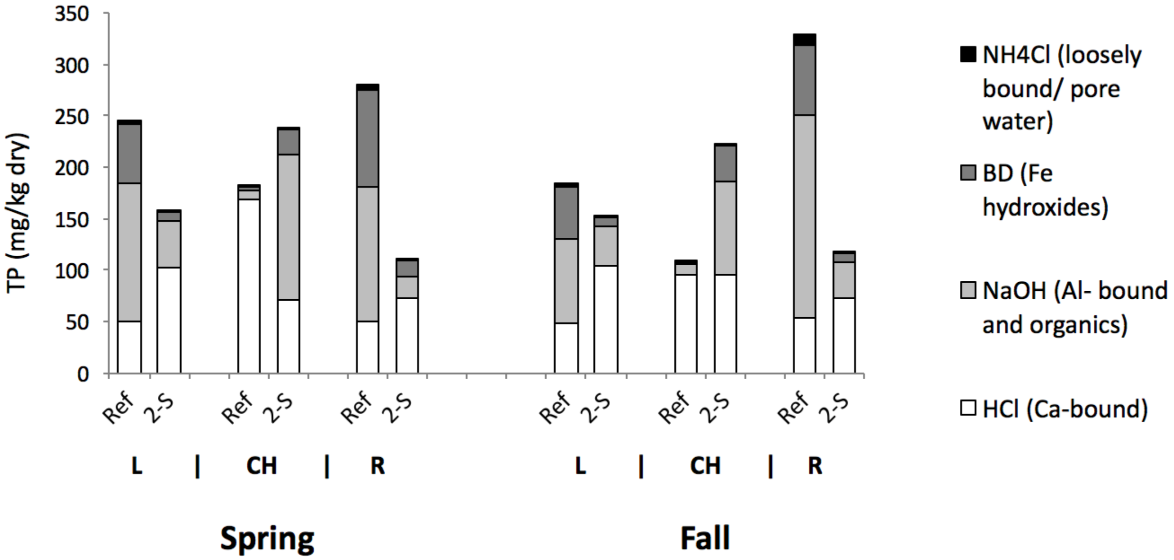
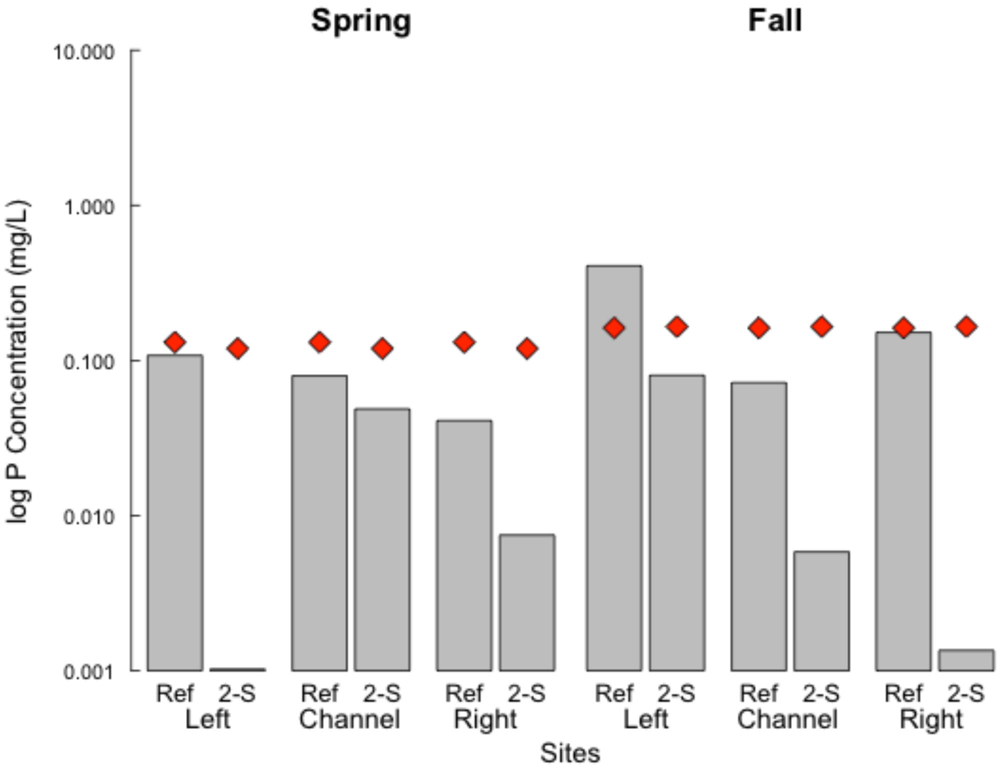

TP (mg/kg dry)
700
600
500
400
300
200
100
0
Ref
2-S
L
CH
R
Spring
Fall
NH4Cl (loosely bound/ pore water)
BD (Fe hydroxides)
NaOH (Al- bound and organics)
HCl (Ca-bound)

Fig. 2.7



350
300
250
200
150
100
50
0
TP (mg/kg dry)
Ref
2-S
L
CH
R
Spring
Fall
NH4Cl (loosely bound/ pore water)
BD (Fe hydroxides)
NaOH (Al- bound and organics)
HCl (Ca-bound)

Fig. 2.8



Spring
Fall
10.000
1.000
0.100
0.010
0.001
log P Concentration (mg/L)
Ref
2-S
Left
Channel
Right
Sites

Fig. 2.9

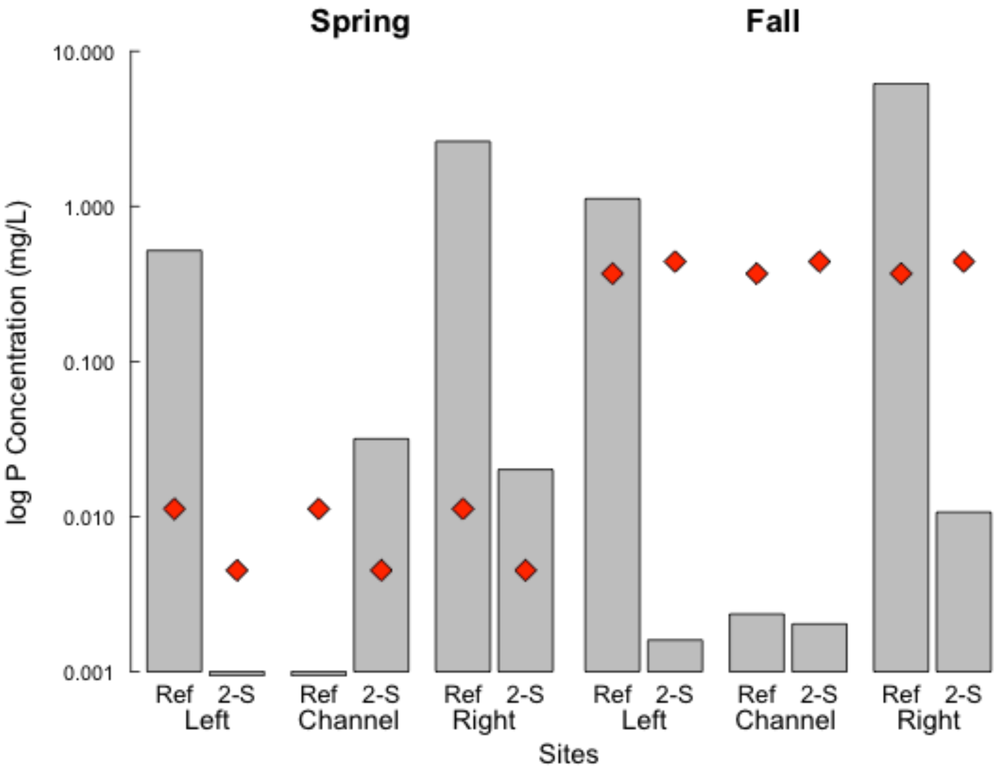

Spring
Fall
log P Concentration (mg/L)
10.000
1.000
0.100
0.010
0.001
Ref
2-S
Left
Ref
2-S
Channel
Ref
2-S
Right
Ref
2-S
Left
Ref
2-S
Channel
Ref
2-S
Right
Sites

Fig. 2.10

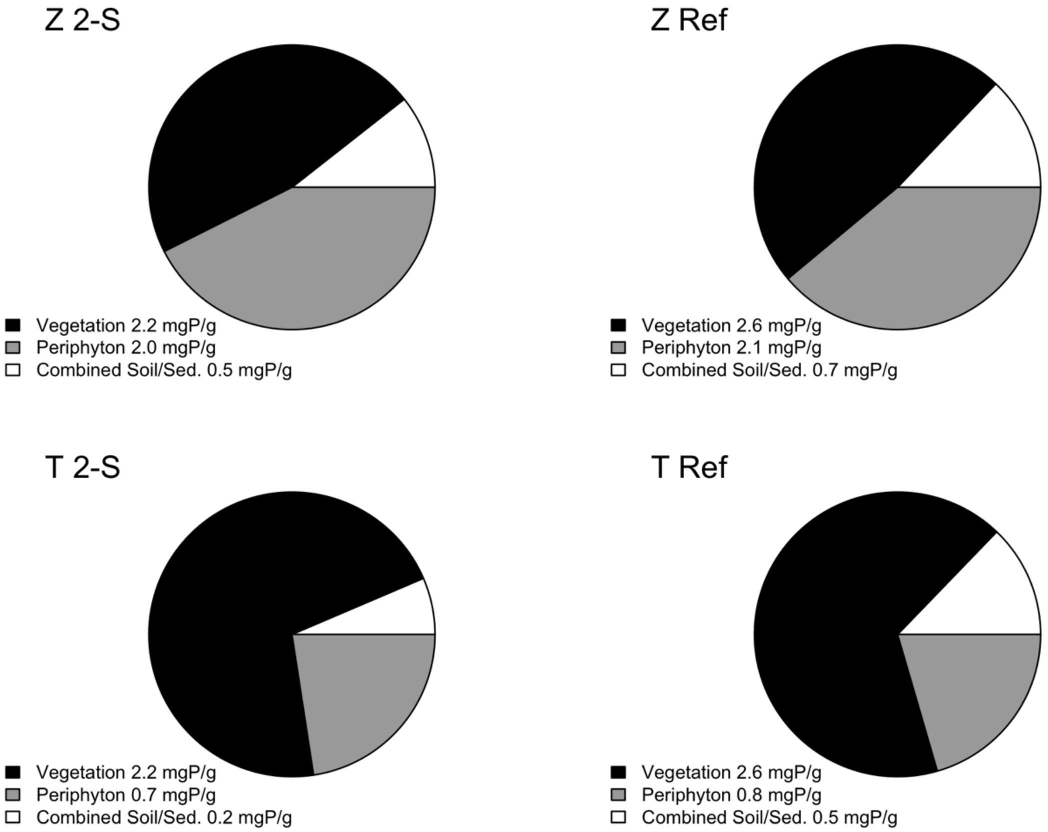

Z 2-S
■ Vegetation 2.2 mgP/g
▨ Periphyton 2.0 mgP/g
☐ Combined Soil/Sed. 0.5 mgP/g
Z Ref
■ Vegetation 2.6 mgP/g
▨ Periphyton 2.1 mgP/g
☐ Combined Soil/Sed. 0.7 mgP/g
T 2-S
■ Vegetation 2.2 mgP/g
▨ Periphyton 0.7 mgP/g
☐ Combined Soil/Sed. 0.2 mgP/g
T Ref
■ Vegetation 2.6 mgP/g
▨ Periphyton 0.8 mgP/g
☐ Combined Soil/Sed. 0.5 mgP/g

Fig. 2.11

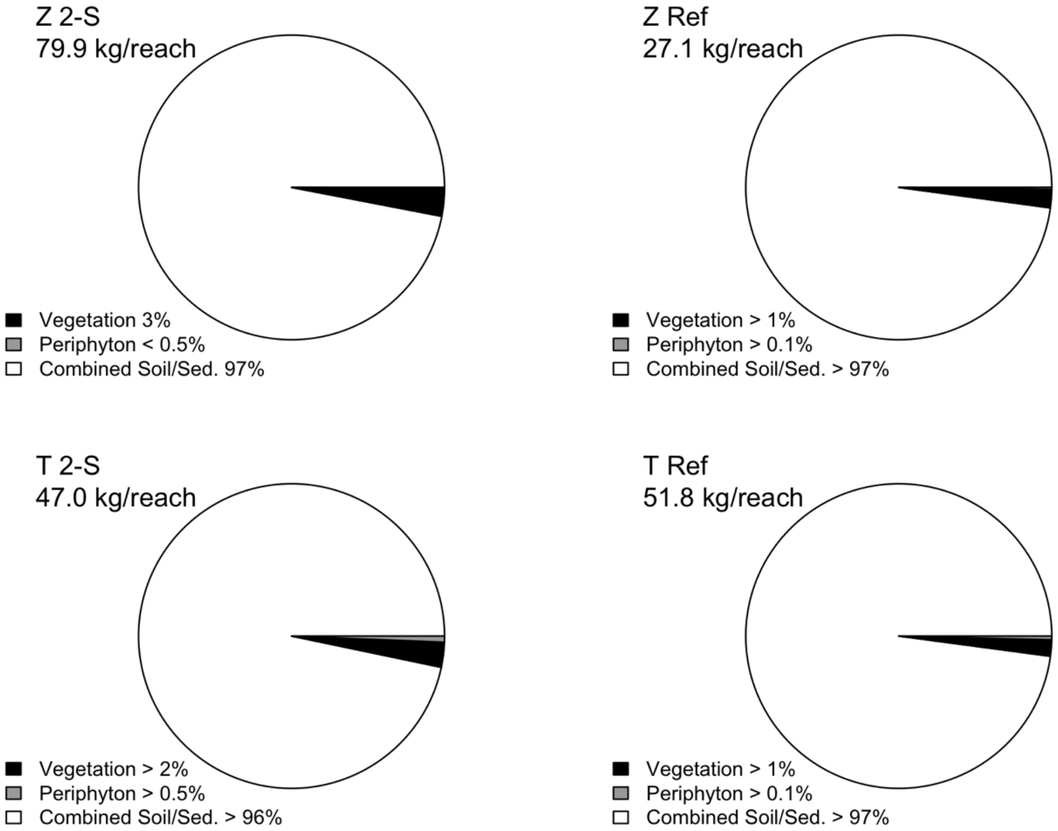

Z 2-S
79.9 kg/reach
■ Vegetation 3%
▨ Periphyton < 0.5%
☐ Combined Soil/Sed. 97%
Z Ref
27.1 kg/reach
■ Vegetation > 1%
▨ Periphyton > 0.1%
☐ Combined Soil/Sed. > 97%
T 2-S
47.0 kg/reach
■ Vegetation > 2%
▨ Periphyton > 0.5%
☐ Combined Soil/Sed. > 96%
T Ref
51.8 kg/reach
■ Vegetation > 1%
▨ Periphyton > 0.1%
☐ Combined Soil/Sed. > 97%

Fig. 2.12

Fig. 2.13

Chapter 3

Impacts of Two-stage Ditch Form on Periphyton Community Structure and Biovolume

Introduction

Agricultural drainage has been implicated as one of the causes of eutrophication in large water bodies throughout the world (Jarvie et al. 2015). Excess nutrients such as nitrogen (N) and phosphorus (P) that are not used by crops or stored in the field soil can make their way to drainage systems via tile drains and surface runoff, and then enter downstream waterbodies. These excess nutrients fuel algal blooms and can increase the potential for ecological impairments, such as beach closings, fish kills, and hypoxic water conditions. Agricultural systems also expedite the movement of fine soils and particulate matter eroded from fields and ditches, resulting in increased turbidity, and deposition of particulate matter that can shade and choke out benthic communities (Hill et al. 2009).

In an effort to reduce excess nutrients and sediment making their way from agricultural lands downstream, farmers can implement prevention measures called best management practices. These best management practices can be operational, such as no-till and planting cover crops, or structural, such as buffer strips, water and sediment control basins, and drainage outlet control boxes. A best management practice new to western Michigan is the two-stage ditch. The two-stage refers to two parallel floodplain benches excavated out of the sides of an existing traditional, trapezoidal shaped drainage ditch. These benches increase area for drainage during high flow events thereby decreasing flow velocity and turbidity, and increasing denitrification (Mahl et al. 2015, Roley et al. 2012). Little is known about the ability of two-stage ditches to retain phosphorus (P) (Kallio et al. 2010) and even less is known about the interactions between nutrient retention and communities of algae growing in the periphyton complex within agricultural waterways (Brennan et al. 2017).

Periphyton is an aggregation of algae, fungi, and bacteria as well as settled particulate matter than accumulates on benthic surfaces in aquatic systems (Larned 2010). In this study, the use of "periphyton" refers explicitly to the algal component of the periphyton complex. Many studies have experimented with the interactions between periphyton, phosphorus concentrations, and light in stream systems due to algae's need for P in photosynthesis and tissue growth (Drake et al. 2012, Fanta et al. 2010, Hill et al. 2009, Qin et al. 2007, Steinman and Duhamel 2017). Under high light and nutrient conditions, periphyton has been shown to be a P sink during the growing season (Drake et al. 2012). Periphyton can take up the more bioavailable form of P, soluble reactive P (SRP), from both the water column directly and after it is released from sediments (Brennan et al. 2017, Fanta et al. 2010).

Brennan et al. (2017) and Drake et al. (2012) also observed potential impacts of sediment-attached periphyton (epipelon) on the movement of nutrients between the water column and the sediment. Due to flow dynamics, a thin layer of anoxic water can form right at the water-sediment boundary as oxygen is used up in microbial respiration (Palmer-Felgate et al. 2010). If this anoxic layer forms, Fe^{3+} can be reduced to Fe^{2+} and release previously bound P back into the water column. Epipelic algal cells can prevent that thin layer of anoxic water from forming during the day by producing oxygen via photosynthesis, thereby reducing P release from sediments to the water column (Brennan et al. 2017). It is yet unclear if certain algal taxa are more effective than others at removing P in the water column and reducing P movement from the sediment to the water column by reducing the anoxic layer.

While particulate matter can transport bound nutrients to the benthos, particulate matter can also change periphyton biofilm structure and community (Lange et al. 2015). Graham (1990) found that particulate matter such as sediment can make up at least 50% of the dry mass of periphyton sampled in average small streams after the particles settled out of the water column or became trapped in algal mucilage. In high depositional areas, motile algae such as *Surirella spp.* make up a larger portion of the algal community as they can move out from under settled particulate matter (Hill et al. 2009, Wagenhoff et al. 2013). During storm events, the two-stage ditch can reduce current velocity, resulting in increased deposition of particulate matter (and thereby increasing motile taxa); these ditches also can reduce scouring power, thereby increasing relative biomass of filamentous taxa and retaining more algal cells in the ditches (Lamb and Lowe 1987, Powell et al. 2007).

Periphyton has the potential to significantly impact both nutrient dynamics and trophic level interactions (Qin et al. 2007); therefore, differences in periphyton community structure and biomass growing in a two-stage ditch form vs. the traditional, trapezoidal shaped ditch should be investigated. My objective was to determine the impacts of the two-stage form on channel periphyton by comparing abundance and biovolume as well as community richness, diversity, and evenness between the two-stage and corresponding upstream reference reaches. I expected algal biomass, abundance, and community richness to increase in the two-stage reaches compared to the reference due to reductions in current speed and an increase in settled particle matter.

Methods

Study Sites

I examined two two-stage (2-S) ditches and their adjacent upstream reference (Ref) reaches that remained in the traditional trapezoidal shape. Both ditches were within the southern portion of the Macatawa watershed in West Michigan (Fig. 3.1), although located in different subbasins. These ditches are referred to as the “Z” and the “T” ditch systems to retain farmer anonymity. The Z two-stage reach was constructed in February 2015 and the T two-stage reach was constructed in November 2015. Hence, the Z ditch was ~1 yr old and the T ditch was only 4 mo old when my sampling began. Paired reaches were similar in length: Z 2-S 373m vs. T 2-S 357 m; Z Ref 207 m vs. T Ref 254m.

Water Quality

Surface water was sampled at the downstream end of both treatment and reference sections in both ditches (cf. Fig. 3.2) monthly from March to December 2016. Water samples were stored on ice during transportation back to the laboratory. Samples for total P (TP) were stored at 4 °C and samples for soluble reactive P (SRP) were filtered through 0.45 μm filters and then stored at 4°C until analysis with a SEAL Autoanalyzer (SEAL Analytical, Mequon, Wisconsin, P detection limit = 0.005 mg/L). Temperature (°C), dissolved oxygen (DO, mg/L), and pH were measured using a YSI 6600 at the time of surface water sampling. Turbidity sensors (Cyclops-7F Turner Designs) were installed at the downstream end of each reach and recorded continuous (every 10 minutes) in-stream measurements from late April to December 2016. Sensors were checked and cleaned every two weeks and calibrated once per month. Stormflow current velocity was not measured due to safety concerns. Due to this and very low water depth and discontinuous flow along the reaches during most of the growing months, currently velocity was not measured during baseflow either. General observations about flow, water depth, and tile exposure were recorded.

Artificial Periphyton Substrate

Unglazed ceramic tiles with a surface area of 2.54 cm^2 were used as artificial substrate for periphyton growth. Prior to use, all tiles were soaked in DI water overnight, dried, and then ashed at 550 °C for one hour to clear them of dust and oils. After cooling, eight tiles were adhered with epoxy putty to a cement brick. One brick and associated substrate tiles were buried flush with the channel bottom at the downstream end of each reach. The tiles were collected monthly; every other tile was removed from the brick and combined for filtering, ash-free dry mass (AFDM), and TP analysis (Fig. 3.3, A). The other four tiles were removed and combined for community structure and biovolume analysis (Fig. 3.3, B). The tiles were returned to the laboratory and the periphyton was removed with DI water and careful brushing. The brick was discarded and replaced with a new brick and tiles in the same position within the ditch after each monthly (approximate) sampling from May to December 2016. Twenty-seven samples in total were successfully collected from the ditches. August T reference was not analyzed due to loss of tiles during a storm. At the time of substrate installation, the depth of the water column above the tiles was recorded as well as general information on the shading from bench and channel edge vegetation.

In- laboratory Periphyton Analysis

Samples for AFDM (μg/cm^2, Steinman and Ogdahl 2016) and P content analysis were prepared by filtering onto pre-combusted and weighed GF/F filters. Filtered samples were then placed in pre-combusted and weighed aluminum weigh boats and dried at 70 °C until constant weight. Samples were then oxidized at 550 °C for 1 hour. After oxidation for the determination of AFDM, the samples were digested with persulfate and then analyzed for TP (Qin et al. 2007, USEPA 1983). All P analyses were conducted on a SEAL autoanalyzer (SEAL Analytical, Mequon, Wisconsin).

Samples for algal identification were preserved in 1% Lugols. At least 300 algal cells were identified to genus in a known volume for each sample. Diatoms were grouped into centric or pennate, and identified to genus whenever possible. If possible, at least 10 cells of each taxon were measured for biovolume using cell volume equations from Hillebrand et al. (1999) and Sun and Liu (2003). A portion of each sample was cleaned for identification of diatoms to genus at 1000x magnification. First the sample was washed of Lugols and then digested for one hour in a 1:1 solution of sample and 5.7% sodium hypochlorite (standard household bleach) to remove the organics (Carr 1986). After the digestion, three rinses of DI water were used to remove the sodium hypochlorite from the sample. Each sample was dried on a cover slip and then inverted onto thin layer of a karo syrup and DI water solution on a slide and set flat to dry.

Riparian Vegetation

The two-stage and reference reaches for both Z and T were separated into five sections of equal length for the vegetation survey, hereafter called “intervals” (Fig. 3.2). Vegetation cover was visually estimated on a 0-5 Braun-Blanquet scale for both benches and both channel edges. The values for the bench were averaged and the values for the channel were averaged per reach. Observations on major vegetation types were recorded.

Data Analysis

Potential differences in water quality parameters, bench and channel edge vegetation cover, and periphyton phosphorus content between two-stage and reference were tested separately for each site (subbasin) using paired samples t-tests or Wilcoxon test depending on normality. When applicable, differences among months were tested for with Kruskal-Wallis tests.

Cells/mm^2 and total biovolume (µm^3/mm^2) were calculated from algal counts and volume measurements. To assess community structure, % diatom taxa of total (1), Shannon diversity index (2), and evenness (3) were calculated as follows.

1) (# diatom taxa/ total # taxa) *100

2) $H' = - \Sigma\, p_i \, {*}\ln p_i$

3) Evenness = $H'/ H_{max} = H'/ \ln S$ where S = total # of taxa

Differences in cells/mm^2, total biovolume, % diatom taxa, H', and evenness between two-stage and reference at each site were tested with paired-samples t-test or Wilcoxon test depending on normality. The variables that were not significantly different were combined for each ditch (2-S and Ref together) and differences among months were tested with the Kruskal-Wallis test. Canonical Correspondence Analysis (CCA) was used to determine correspondence between cells abundance and the following factors: treatment, shading, water level, periphyton P content, TP baseflow, SRP baseflow, turbidity, water temperature, and DO concentration. A scattermatrix was used to identify highly correlated environmental variables and variance inflation factor (VIF) values above 10 were removed. The final model included only treatment, turbidity, temperature, and DO. Partial CCA was used to determine significance for each factor in the model. All analyses were conducted in R (v99.447) and significance was set at $p < 0.05$.

Results

Water Quality

Overall, phosphorus concentrations were higher in the Z ditch than the T ditch (Table 3.1). Mean TP and SRP concentrations during baseflow in the Z two-stage ditch were not significantly different from those in the reference reach (Table 3.1). In contrast, mean TP and SRP concentrations during baseflow in the T two-stage ditch were significantly lower than those in the reference reach (TP: $p = 0.009$; SRP: $p = 0.011$, Table 3.1).

Z ditch turbidity was significantly higher in the reference compared to the two-stage ($p < 0.0001$, Table 3.1). Due to equipment malfunction, approximately half of the turbidity data for the two-stage in T were not usable. To compare data between the T ditch two-stage and reference reaches, the corresponding data from the reference reach also was discarded; however, there were still more than 6200 turbidity observations for the comparison. In contrast to Z ditch, T ditch turbidity was significantly lower in the reference compared to the two-stage ($p < 0.0001$, Table 3.1). The turbidity data encompass both baseflow and stormflow measurements for 2016 and there was high variability. The turbidity ranged from approximately 0.05 to 3,000 NTU in the Z ditch and from approximately 0.30 to 3,200 NTU in the T ditch. However, the paired nature of the sampling and the sample size (Z: 5,407 and T: 3,101 for each reach) allowed for significant differences between reaches to be observed.

Vegetation

Assessment of vegetation cover was divided into bench and channel edge regions. Both mean bench and channel edge vegetation cover in the Z ditch were not significantly different between two-stage and reference reaches (Table 3.1). In contrast, mean bench vegetation cover in the T ditch was significantly greater in the reference reach compared to the two-stage reach ($p<0.0001$, Table 3.1); however, T ditch mean channel edge vegetation cover between reaches was not significantly different.

Major types of bench and channel edge vegetation at the periphyton sampling sites were grasses, goldenrod, willow and bulrush. As these plants grew tall, the channel edge cover values in the Z two-stage and reference reaches and T two-stage reach likely underrepresented the shading caused by the vegetation at those specific sites. Unfortunately, irradiance values were not measured so quantitative comparisons cannot be made.

Periphyton

Periphyton P content was not significantly different between the two-stage and reference reaches for either ditch (Table 3.1). Water levels receded during the summer months; as a consequence, the tiles were found above the water line during certain tile checks and sampling months. All tiles collected were sampled for periphyton analysis no matter the degree of desiccation at the time of collection (Table 3.2). Observations on tile water cover and dryness are as follows: Z ditch two-stage and reference tiles were damp but out of the water in July and completely dry in August at the time of sampling. T ditch two-stage and reference tiles were barely under the surface of the water in July and dry in August at the time of sampling. At the time of sampling during the other months the water surface was at least 3 cm above the tiles. Overhanging vegetation also created variation in environmental conditions throughout the period of sampling. Z ditch two-stage tiles were partially covered with overhanging vegetation starting in June and completely covered from July to November. Z ditch reference tiles were partially shaded September through October. T ditch two-stage tiles were partially shaded from

September through October and shaded by floating filamentous algal mats in November. T ditch reference tiles were not shaded by bench or channel edge vegetation but were shaded by floating filamentous algal mats in November. Despite desiccation and/or shading in certain months, there were no significant differences in periphyton P content among months in all reaches (Z: $\chi^2 = 8.6$, df = 6, p = 0.2; T: $\chi^2 = 1.2$, df = 5, p = 0.9).

Z ditch mean algal density in the reference reach was considerably greater than in the two-stage reach (1,016 vs 170 cells/mm^2) but this was driven largely by one month (May), so the difference was only marginally significant (p=0.07, Table 3.2). Similar to Z, T ditch mean algal density was significantly greater in the reference reach than in the two-stage reach (p= 0.004, Table 3.2). Total biovolume (μm^3/mm^2) was not significantly different between two-stage and reference for either Z or T ditch; however, the Z ditch two-stage biovolume tended to be lower than the reference (Z: p= 0.06, Fig. 3.4 and T: p= 1, Fig. 3.5). Number of taxa, diatom % of taxa, H', and evenness were all not significantly different between two-stage and reference for either the Z or T ditches (Table 3.2). There was no significant difference in biovolume of motile taxa between the two-stage and the reference (Z: p= 0.40, T: p > 0.99, Table 3.3). Time had no significant effect (all p>0.1) on combined two-stage and reference values for total biovolume, #taxa, diatom % of taxa, H', and evenness.

The periphyton community structure was very similar between two-stage and reference for both Z and T ditches. Genera that were present in at least four out of the seven sampling months made up a large portion of the biovolume on average: Z 2-S 80.6%, Z Ref 75.5%, T 2-S 74.5%, T Ref 98.2% (Table 3.3). The only Chlorophyta taxa present in 4 out of 7 months were the *Stigeoclonium* and *Oedogonium*. *Stigeoclonium* was present in the Z reference reach and the T two-stage reach while *Oedogonium* was present in both the T two-stage and reference reaches.

The remaining dominant taxa were all diatoms. The three major genera by biovolume in the Z two-stage were *Rhoicosphenia, Gomphonema,* and *Navicula.* The three major taxa by biovolume in the Z reference were *Gomphonema, Cocconeis,* and *Navicula.* The four major taxa by biovolume in the T two-stage were *Navicula, Synedra, Gomphonema,* and *Gyrosigma.* The four major taxa by biovolume in the T reference were *Cocconeis, Synedra, Surirella,* and *Navicula.*

The initial CCA model, which included the highly correlated variables, explained approximately 76% of the variability in abundance data. After the highly correlated environmental variables were removed, the model included only treatment, turbidity, temperature, and DO. This model still explained 48.7% of the variability in the abundance data. The CCA plot revealed that the Z two-stage and reference reaches were more similar to each other than the T two-stage and reference reaches were to each other (Fig. 3.6). Algal community structure in the T two-stage reach was associated with temperature whereas in the T reference reach, community structure was influenced by DO concentration (Fig. 3.6). Earlier months of the sampling year (May, June, and July) were more scattered in the biplot than the later months, which tended to cluster closer together. All four factors were significant descriptors when tested with partial CCA, with DO being the most descriptive (14.6%, p= 0.004) followed by turbidity (11.3%, p=0.017); treatment and temperature described a similar percentage of the variability (Treatment 9.3%, p=0.023; Temperature 9.2%, p=0.027) (cf. Fig. 3.6).

Discussion

Periphyton can be an important component of the autotrophic base in aquatic systems (Liess et al. 2012, Qin et al. 2007). Light availability, nutrient concentrations, turbidity, flood disturbance, and grazing have been shown to impact the growth of periphyton (Biggs et al. 1998, Brennan et al. 2017, Hill and Knight 1988) as well as interactions between these factors (Lange et al. 2011). Light and nutrient concentrations have the largest impact on the growth of algae in many freshwater streams (Fanta et al. 2010, Qin et al. 2007). Brennan et al. (2017) found that P transferred by eroded soils and then released into the water column can also have a significant impact on benthic algal growth.

When algal growth negatively impacts the ecosystem services of a water body, it is termed nuisance algal growth. Nuisance algae are typically nutrient tolerant filamentous taxa such as *Cladophora* and *Oedogonium*, and can increase pH and diurnal DO fluctuations, as well as reduce algal community biodiversity (Stevenson et al. 2012, Dodds et al. 2002). While agricultural drainage systems are not known for their biodiversity and human recreational use, it is still important to maintain a level of biodiversity that allows the system to function effectively. If nuisance algae significantly increase pH and/or reduce DO at night, it may impact the retention of P within the top layers of sediment. By increasing pH, the potential for release of Al-bound P increases. If DO concentrations decrease significantly at night, there is potential of hypoxic/anoxic conditions that may reduce Fe^{+3} and release P back into the water column. Measurements of the interactions between large nuisance algae growth, pH, DO, and P dynamics are still not fully understood (Stevenson et al. 2012) and should be empirically studied and modeled at the field level in future research.

Temporary immobilization of P may be an important factor in the timing of P release from upstream ditches in agricultural watersheds; P taken up by algae and vegetation may be released during scouring and mineralization during the non-growing seasons. Drake et al. (2012) found that high light and high nutrient concentrations increased the P immobilization by periphyton and may also increase the length of time that that P stays immobilized in algae and vegetation tissue. Vegetation growth on the benches and channel sides (and therefore shading) was limited during the spring, and this period of time corresponded with the period of the growing season with the largest biovolume and density of algal growth in the Z and T ditches. Therefore, late spring may be the optimal time for unshaded algal growth and immobilization of nutrients by algae until the end of the growing season when cell death and P mineralization increases. In the North-central region of the United States, typically late April-early May corresponds with the final snowmelt and the beginning of spring storms which moves a significant load of soil and nutrients (Clement and Steinman 2017). It is still unclear how significant the impact of this temporary P immobilization is compared to the high loads in the ditches and when the P would be released back into the water column. This release of P from periphyton or vegetation would likely occur in the late summer- fall, during low nutrient load times for the Macatawa watershed (Clement and Steinman 2017) and have a greater chance of being adsorbed to the sediment. However, in certain subbasins this release of P may also align with fertilizer application and fall storms and have an undetectable impact on the system.

Given the concerns over eutrophication, harmful algal blooms, and hypoxia, it is important that any best management practice applied to farming in the Midwest is effective at retaining nutrients and particulate matter. In order to optimize the ability of BMPs to process and retain nutrients, we have to understand the movement of nutrients within two-stage and traditional ditches. One function of an effective two-stage should provide a balance between reducing nuisance algae and increasing the temporary retention of nutrients through seasonal algal growth.

Cell density for each reach was quite low, especially for slower streams (Lamb and Lowe, 1987). Cell densities (for all but the Z two-stage) during May were more similar to other studies. It is possible that on the artificial substrate used for this study and the deposition of particulate matter slowed the colonization rate. Also, the lack of irradiance and desiccation of tiles during the summer months reduced the growth of algae on the tiles. However, the total biovolumes were within range for agricultural streams and artificial substrates (Hill and Knight, 1988).

The Z ditch had high mean TP and SRP concentrations compared to the T ditch. I expected this to be the opposite as the fields within the Z system had winter crops and were interseeded, whereas the T ditch was tilled and left unplanted during the non-growing season. The Z ditch may have higher mean TP and SRP concentrations because of inputs farther up in the watershed but the reason for the high TP and SRP concentrations in the Z ditch is unclear. The T ditch also had lower TP within the sediment of both the two-stage and reference (Ch. 2) suggesting the P load may have been lower in this series of drainage ditches for some time.

The reduction in turbidity in the Z two-stage was not large enough to result in a significant reduction in TP and would have less direct impact on SRP concentrations within the two-stage reach. This reduction in turbidity at the downstream sampling site suggests that some of the particulate matter traveling out of the reference reach settled within the two-stage reach and less was present to settle out at the downstream algal sampling site. Increased deposition of particulate matter tends to increase motile algal taxa (Wagenhoff et al. 2013). None of the main taxa in Z two-stage or reference reaches was highly mobile taxa; all were attached or weakly motile taxa (Table 3.3). Even though the turbidity was higher within the reference reach, the only filamentous Chlorophyte present in four out of seven sampling months in Z (*Stigeoclonium spp.)* was found in the reference reach. *Stigeoclonium* is commonly found in nutrient rich, slow flowing waters growing on rocks (Bellinger and Sigee 2015). The substrate surrounding the brick and tile placement was primarily cobble and gravel in the reference while it was gravel and sand in the two-stage. The *Stigeoclonium* found growing on the reference tiles most likely originated from these stones and colonized the new substrate.

Decreased turbidity at the downstream end of the Z two-stage reach also could have increased irradiance to the benthic zone (Hill et al. 2009). However, shading by bench and channel vegetation most likely impacted the irradiance at the two-stage sampling site much more than changes in turbidity. It is unlikely that the reduction in turbidity would have an impact on the light and therefore growth of algae in the two-stage compared to the reference because there was no significant difference in the vegetation cover. With no significant differences in nutrients or light availability, it is not surprising that there were no significant differences between two-stage and reference algal biovolume, cells/ mm^2, or periphyton TP content.

In the T ditch, turbidity was significantly higher in the two-stage reach while TP and SRP were significantly lower within the two-stage reach. This increase in turbidity was due to erosion during high flow events. As the surface area of the functional channel increased and moved onto the benches during 2016, uptake of P via algae and adsorption of P to sediment could have increased and significantly reduced P concentrations in the drainage water. If particulate bound P increased within the water column, bioavailable P would have had to decrease significantly for the TP to also decrease within the two-stage.

The algae responsible for part of the reduction in P were most likely the mat algae found on the wet benches and in the channel since there was little difference in the algal biomass between the T ditch two-stage and reference reaches sampled on the artificial substrate. However, the widespread mat algae were not sampled or analyzed for this thesis. Increased turbidity as well as shading due to channel edge vegetation and the algal mats may have significantly reduced the density of the algae on the tiles in the two-stage compared to the reference reach but not significantly impact either biovolume or overall community diversity. Both T two-stage and reference communities were a mixture of non-motile, nutrient tolerant taxa and very motile taxa found on fine substrates. Two additional genera (*Gyrosigma* and *Stigeoclonium*) were present in at least four out of the seven sampling months on the T two-stage tiles compared to the reference (11genera total vs. 9 genera total, respectively). *Gyrosigma* and *Stigeoclonium* are more common in slower moving aquatic environments and on rocky surfaces (Bellinger and Sigee 2015). As the two-stage channel substrate was comprised of sand and clay, the presence of these taxa, especially *Stigeoclonium,* may be due to the use of tile artificial substrate. The T reference had high variability in algal density and biovolume. This is mirrored by the variability of T reference sampling months along the DO axes in the CCA plot (Fig. 3.6). The lack of difference in periphyton TP content may reflect the large amounts of deposited particulate matter that could not be separated from the algae during filtering and associated TP analysis.

There were no statistical differences in the environmental factors among months. However, there are trends that appear when all four reaches are combined. DO seems to explain a large portion of the algal variability, perhaps just by mirroring the growth and photosynthesis of the periphyton. Temperature was highly correlated with the shading factor that was removed. The T ditch experienced large changes in temperature due to less shading along the reach. The high multicollinearity of many factors could be because PAR (photosynthetically active radiation), water flow, and water depth were not measured quantitatively for this project. Turbidity data also were not available for the early months of algal sampling due to sensor malfunction.

In this study, ceramic tiles were used as artificial substrate to collect algae. Communities from more mobile substrates, such as sand, may not be well represented in this study. There is variable benthic substrate throughout each reach, ranging from fine sandy clay to sand and gravel to small cobble. By using tiles, we were able to remove variation that would have been present had the samples been taken from rocks or sediment that was present in the ditch, and also allowed for equal timing for particle settling onto the tiles for each reach, as suspended solids were not directly measured for this research.

There were significant growths of algae floating in the channel or in wet puddles on the benches. These algae, all characterized predominantly by filamentous taxa, formed large mats within both the two-stage and the reference of both ditches but appeared to be composed of different major taxa. They also played a part in shading the tiles during the fall. To understand the entire suite of algal communities within drainage ditches, future sampling should also include these algal mats.

Conclusions

Despite trends that suggest algal density and biovolume were lower in the two-stage compared to the reference reaches of both Z and T, only T ditch algal density was significantly different between two-stage and reference reaches. There were no significant differences in number of taxa, % diatom taxa, or community diversity between two-stage and the reference or among months. It is possible that finer taxonomic resolution down to the species level may have resulted in significant differences but it is clear that these relatively new two-stage ditches are not currently having a distinct impact on periphyton community structure. As the ditches mature and more vegetation growth occurs, these systems will have reduced periphyton growth in the summer months due to shading.

While vegetation shading may keep water temperatures down, allowing for the growth of photosynthesizing algae may prevent standing water from becoming anoxic (thereby reducing Fe^{+3}) and releasing P back into the water column during low baseflow conditions when the channel is disconnected and the water stagnant. In ditches with low water TP and SRP, the release of P from sediments during low flow conditions may significantly increase the P concentrations and potential for P export during the next high flow event. For shallow ditches like the Z and the T where late summer baseflow is low and there are few in-channel vegetation taxa (i.e., when algae are the main biotic sinks for P), keeping the vegetation growth trimmed on the benches to increase the irradiance into the channel allowing for increased algal photosynthesis may be a best management practice to retain P. Elevated periphyton growth in the spring when shading is low and nutrient load is high or in the summer with trimmed bench vegetation may reduce some P export during those months and immobilize it until potential mineralization later in the year. Then as water levels recede and P loads decline in late summer, and water in the channel becomes disconnected, the P released from mineralized periphyton will have a greater chance of being adsorbed to the sediments and not transported out of the reach. A more thorough look at sediment P properties, algal P cycling, and water column P and quality parameters throughout the year is necessary in order to quantify the actual impacts of algal P immobilization, respiration, and photosynthesis in two-stage systems.

Typically, the influence of turbidity and nutrients on algal growth is the focus of periphyton research in agricultural streams. While nutrient concentrations likely had some influence on algal growth in these ditches, multicollinearity among TP, SRP, and other measured environmental factors prevented a clear estimation of nutrient significance. It is likely that the significant determining factor of temperature was an artifact from reduced shading of the channel along the T two-stage reach and that differences in DO are likely a response to the growth of mat algae in the T two-stage reach. As the morphology of ditches becomes a larger focus of management, these less-studied factors such as temperature and DO production should be included in research to better understand algal growth and its impacts on nutrient cycling within agricultural drainage systems.

References

Bellinger, E.G., Sigee, D.C. 2015. A key to the more frequently occurring freshwater algae. Pages 141-251. In: Freshwater Algae Identification, Enumeration and Use as Bioindicators. (2nd Ed.) John Wiley and Sons, Ltd.

Biggs, B.J.F., Stevenson, R.J., Lowe, R.L. 1998. A habitat matrix conceptual model for stream periphyton. Arch. Hydrobiol. 143: 21-56.

Brennan, R.B., Scott, J.T., Sharpley, A.N., Lally, H.T., Jarvie, H.P., Bowes, M.J., Haggard, B.E., Gbur, E. 2017. Linking soil erosion to instream dissolved phosphorus cycling and periphyton growth. J. Am. Water Res. Assoc. 1-13. DOI: 10.1111/1752-1688.12534

Carr, J.M., Hergenrader, G.L., Troelstrup Jr., N.H. 1986. A simple, inexpensive method for cleaning diatoms. Trans. Am. Microsc. Soc. 105: 152-157.

Clement, D.R., Steinman, A.D. 2017. Phosphorus loading and ecological impacts from agricultural tile drains in a west Michigan watershed. J. Great Lakes Res. 43: 50-58.

Dodds, W.K., Smith, V.H., Lohman, K. 2002. Nitrogen and phosphorus relationships to benthic algal biomass in temperate streams. Can. J. Fish. Aquat. Sci. 59: 865-874.

Drake, W.M., Scott, J.T., Evans-White, M., Haggard, B., Sharpley, A., Rogers, C.W., Grantz, E.M. 2012. The effect of periphyton stoichiometry and light on biological phosphorus immobilization and release in streams. Limnology 13: 97-106.

Fanta, S.E., Hill, W.R., Smith, T.B., Roberts, B.J. 2010. Applying the light: nutrient hypothesis to stream periphyton. Freshwater Biol. 55: 931-940

Graham, A.A. 1990. Siltation of stone-surface periphyton in rivers by clay-sized particles from low concentrations in suspension. Hydroliologia 199: 107-115.

Hill, W.R., Fanta, S.E., Roberts, B.J. 2009. Quantifying phosphorus and light effects in stream algae. Limnol. Oceanogr. 54: 368-380.

Hill, W.R., Knight, A.W. 1988. Nutrient and light limitation of algae in two northern California streams. J. Phycol. 24: 125-132.

Hillebrand, H., Durselen, C.D., Kirschtel, D., Pollingher, U., Zohary, T. 1999. Biovolume calculation for pelagic and benthic microalgae. J. Phycol. 35: 403-424.

Jarvie, H.P., Sharpley, A.N., Flaten, D., Kleinman, P.J., Jenkins, A., Simmons, T. 2015. The pivotal role of phosphorus in a resilient water-energy-food security nexus. J. Environ. Qual. 44: 1049-1062.

Kallio, S., Ward, A., D'Ambrosio, J.L., Witter, J.D. 2010. Beyond the field: a look at agricultural ditch floodplains as a water quality BMP. CIGR XVII World Congress

Lamb, M.A., Lowe, R.L. 1987. Effects of current velocity on the physical structuring of diatom (Bacillariophyceae) communities. Ohio J. Sci. 87: 72-78.

Lange, K., Liess, A., Piggot, J.J., Townsend, C.R., Matthaei, C.D. 2011. Light, nutrients and grazing interact to determine stream diatom community composition and functional group structure. Freshwater Biol. 56: 264-278.

Lange, K., Townsend, C.R., Matthaei, C.D. 2015. A trait-based framework from stream algal communities. Ecol. and Evol. DOI: 10.1002/ece3.1822

Larned, S.T. 2010. A prospectus for periphyton: recent and future ecological research. J. North Am. Benthol. Soc. 29: 182-206.

Liess, A., Le Gros, A., Wagenhoff, A., Townsend, C.R., Matthaei, C.D. 2012. Landuse intensity in stream catchments affects the benthic food web: consequences for nutrient supply, periphyton C:nutrient rations, and invertebrate richness and abundance. Freshwater Sci. 31: 813-824.

Mahl, U.H., Tank, J.L., S.S. Roley, Davis, R.T. 2015. Two-stage ditch floodplains enhance N-removal capacity and reduce turbidity and dissolved P in agricultural streams. J. Am. Water Res. Assoc. 51:923-940.

Palmer-Felgate, E.J., Mortimer, R.J.G., Krom, M.D., Jarvie, H.P. 2010. Impact of point-source pollution on phosphorus and nitrogen cycling in stream-bed sediments. Environ. Sci. Technol. 44: 908- 914.

Powell, G.E., Ward, A.D., Mecklenburg, D.E., Jayakaran, A.D. 2007 Two-stage channel systems: part 1, a practical approach for sizing agricultural ditches. J Soil Water Cons. 62: 277-286.

Qin, P. Mayer, C.M., Schulz, K.L., Ritchie, M.E. 2007. Ecological stoichiometry in benthic food webs: effect of light and nutrients on periphyton food quantity and quality in lakes. Limnol. Oceanogr. 52: 1728-1734.

Roley, S.S., Tank, J.L., Stephen, M.L., Johnson, L.T., Beaulieu, J.J., Witter, J.D. 2012. Floodplain restoration enhances denitrification and reach-scale nitrogen removal in an agricultural stream. Ecol. Appl. 22: 281-297.

Steinman, A.D., Duhamel, S. 2017. Phosphorus limitation, uptake, and turnover in stream algae. Pages 197-218. In: Methods in Stream Ecology. Vol II (3rd Ed.) R. Hauer and G. Lamberti (Ed.). Elsevier Press.

Steinman, A.D., Ogdahl, M.E. 2016. From wetland to farm and back again: phosphorus dynamics of a proposed restoration project. Environ. Sci. Pollut. Res. 23:22596-22605.

Stevenson, R.J., Bennett, B.J., Jordan, D.N., French, R.D. 2012. Phosphorus regulates stream injury by filamentous green algae, DO, and pH with thresholds in responses. Hydrobiologia 695: 25-42.

Sun, J., Liu, D. 2003. Geometric models for calculating cell biovolume and surface area for phytoplankton. J. Plankton Res. 25: 1331-1346.

USEPA. 1983. Methods for chemical analysis of water and wastes. USEPA, Washington, D.C.

Wagenhoff, A., Lange, K., Townsend, C.R., Matthaei, C.D. 2013. Patterns of benthic algae and cyanobacteria along twin-stressor gradients of nutrients and fine sediment: a stream mesocosm experiment. Freshwater Biol. 58: 1849-1863.

Figure Captions

Figure 3.1 Location of the two study ditches (Z and T) within Macatawa River watershed. Inset: location of watershed in lower peninsula of Michigan.

Figure 3.2 Sampling locations for surface water sampling and vegetation sampling for bank (reference)/bench (two-stage) and channel edge samples within sample ditch system. Periphyton tile placement was adjacent to the surface water sampling site. Each reach was divided into 5 equal length intervals. Diagram, intervals, and symbols not to scale.

Figure 3.3 Diagram of artificial substrate tiles adhered to brick. One brick was installed per downstream site and removed every month. A: Tiles for ash-free dry mass and total phosphorus analysis; B: Tiles for community structure and biovolume analysis. Circles represent rebar installed on downstream end and sides to prevent brick movement and consistent placement within stream. Arrow indicates direction of water flow.

Figure 3.4 Z ditch total biovolume ($\mu m^3/mm^2$) of periphyton taxa over sampling months.

Figure 3.5 T ditch total biovolume ($\mu m^3/mm^2$) of periphyton taxa over sampling months. August reference data are not available due to loss of tiles.

Figure 3.6 CCA plot of algal abundance with the following environmental factors: Turbidity, dissolved oxygen concentration (DO.Conc), Temperature (Temp), Treatment (Treatment.D). M= May, J= June, Y= July, A= August, S= September, N= November, D= December; Red= Z two-stage, Blue= Z reference, Black= T two-stage, Orange= T reference. May T 2-S is not represented due to missing turbidity data and August T Ref is not represented due to loss of tiles during a storm.

Tables

Table 3.1 Z and T ditch characteristics. Water quality parameters (± one standard deviation) were tested for statistical differences between the 2-S and Ref reaches for each site, statistically significant difference ($p < 0.05$) shown in bold. "2-S"= two-stage reach, "Ref"= reference reach, "SRP"= soluble reactive phosphorus, "TP"= total phosphorus, "Veg."= vegetation.

Ditch and Treatment	Z 2-S	Z Ref	T 2-S	T Ref
Mean TP (mg/L) Baseflow	0.32 ± 0.2	0.35 ± 0.2	**0.06 ± 0.1**	**0.09 ± 0.1**
Mean SRP (mg/L) Baseflow	0.24 ± 0.2	0.26 ± 0.2	**0.02 ± 0.04**	**0.05 ± 0.05**
Mean In-stream Turbidity (NTU)	**55.8 ± 187.1**	**66.3 ± 186.3**	**67.9 ± 252.4**	**43.2 ± 136.4**
Mean Veg. Cover (0-5) Bench	4.1	4.6	**1.4**	**5.0**
Mean Veg. Cover (0-5) Channel	3.8	3.4	4.0	3.0
Periphyton P content (g/m^2 dry)	0.07 ± 0.10	0.11 ± 0.06	0.16 ± 0.07	0.23 ± 0.08

Table 3.2 Z and T ditch periphyton parameters across sampling months. Submerged vs. Emerged condition refers to condition of substrate tiles at time of sampling. Total cells/mm^2 include Bacillariophyta, Chlorophyta, and Cyanophyta. H'= Shannon- Weiner diversity index. August reference data are not available due to loss of tiles. Statistically significant differences between two-stage (2-S) and reference (REF) shown in bold.

SITE	MONTH	Submerged vs. Emerged	Cells/mm^2	# taxa	Diatom % of taxa	H'	Evenness
Z 2-S	MAY	Submerged	921.89	13	62	1.54	0.60
	JUNE	Submerged	19.61	11	73	1.41	0.59
	JULY	Emerged	27.05	7	100	1.44	0.74
	AUG	Emerged	6.83	4	100	1.35	0.98
	SEPT	Submerged	24.76	11	91	1.84	0.77
	NOV	Submerged	120.04	9	89	1.51	0.69
	DEC	Submerged	66.53	12	92	1.82	0.73
Mean			169.53	10	87	1.54	0.73
Z REF	MAY	Submerged	4725.86	11	73	1.38	0.58
	JUNE	Submerged	68.99	11	91	1.64	0.68
	JULY	Emerged	224.91	8	88	1.45	0.70
	AUG	Emerged	1717.62	11	64	1.60	0.67
	SEPT	Submerged	85.87	9	78	1.59	0.72
	NOV	Submerged	265.11	11	73	1.56	0.65
	DEC	Submerged	22.29	11	91	2.01	0.84
Mean			1015.81	10	79	1.60	0.69

SITE	MONTH	Submerged vs. Emerged	Cells/mm^2	# taxa	Diatom % of taxa	H'	Evenness
T 2-S	MAY	Submerged	1416.20	15	67	1.83	0.68
	JUNE	Submerged	359.00	11	64	1.01	0.42
	JULY	Sub./ Emer.	615.93	10	100	1.54	0.67
	AUG	Emerged	434.88	12	83	1.80	0.72
	SEPT	Submerged	827.78	11	73	1.70	0.71
	NOV	Submerged	370.29	12	75	1.84	0.74
	DEC	Submerged	308.68	13	100	2.35	0.91
Mean			**618.97**	12	80	1.72	0.69
T REF	MAY	Submerged	3110.96	14	71	1.49	0.56
	JUNE	Submerged	1434.20	17	65	2.05	0.72
	JULY	Sub./ Emer.	385.87	9	100	1.73	0.79
	AUG	Submerged	NA	NA	NA	NA	NA
	SEPT	Submerged	403.51	13	62	1.68	0.66
	NOV	Submerged	392.73	13	85	2.07	0.81
	DEC	Submerged	70.24	12	92	2.14	0.86
Mean			**966.25**	13	79	1.86	0.73

Table 3.3 Percent of total biovolume on average for those taxa present in at least 4 out of 7 months (Z) or 3 out 6 months (T). Δ marked genus are motile.

Z 2-S		Z REF	
Taxa	% of Total Biovolume	Taxa	% of Total Biovolume
Rhoicosphenia	31.1	*Gomphonema*	24.8
Gomphonema	19.7	*Cocconeis*	15.9
Navicula $^{\Delta}$	11.3	*Navicula* $^{\Delta}$	13.0
Planothidium $^{\Delta}$	6.4	*Planothidium* $^{\Delta}$	6.8
Nitzschia $^{\Delta}$	5.9	*Rhoicosphenia*	5.8
Surirella $^{\Delta}$	3.5	*Stigeoclonium*	5.5
Cocconeis	2.7	*Nitzschia* $^{\Delta}$	3.9

T 2-S		T REF	
Taxa	% of Total Biovolume	Taxa	% of Total Biovolume
Navicula $^{\Delta}$	16.2	*Cocconeis*	42.6
Synedra	11.4	*Synedra*	18.0
Gomphonema	10.6	*Surirella* $^{\Delta}$	15.1
Gyrosigma	10.2	*Navicula* $^{\Delta}$	14.9
Rhoicosphenia	5.9	*Gomphonema*	2.9
Nitzschia $^{\Delta}$	5.2	*Rhoicosphenia*	2.4
Surirella $^{\Delta}$	4.5	*Oedogonium*	1.3
Oedogonium	3.5	*Nitzschia* $^{\Delta}$	0.7
Cocconeis	2.7	*Planothidium* $^{\Delta}$	0.3
Stigeoclonium	2.3	--	--
Planothidium $^{\Delta}$	2.0	--	--

Figures

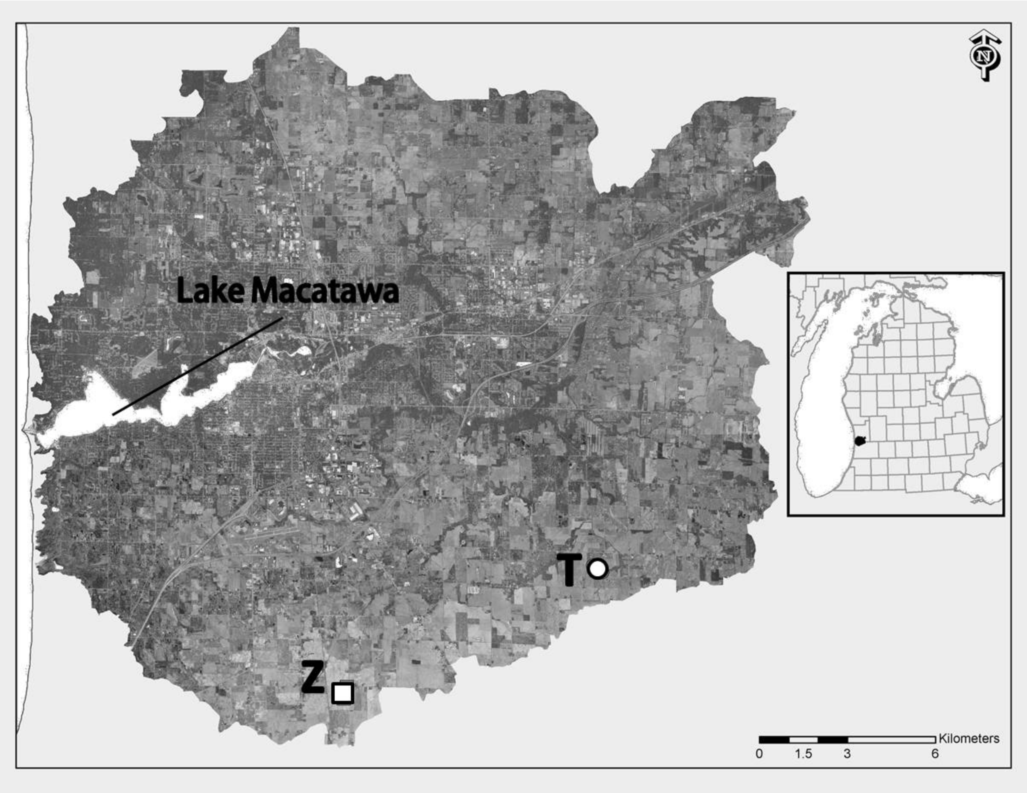


Figure 3.1

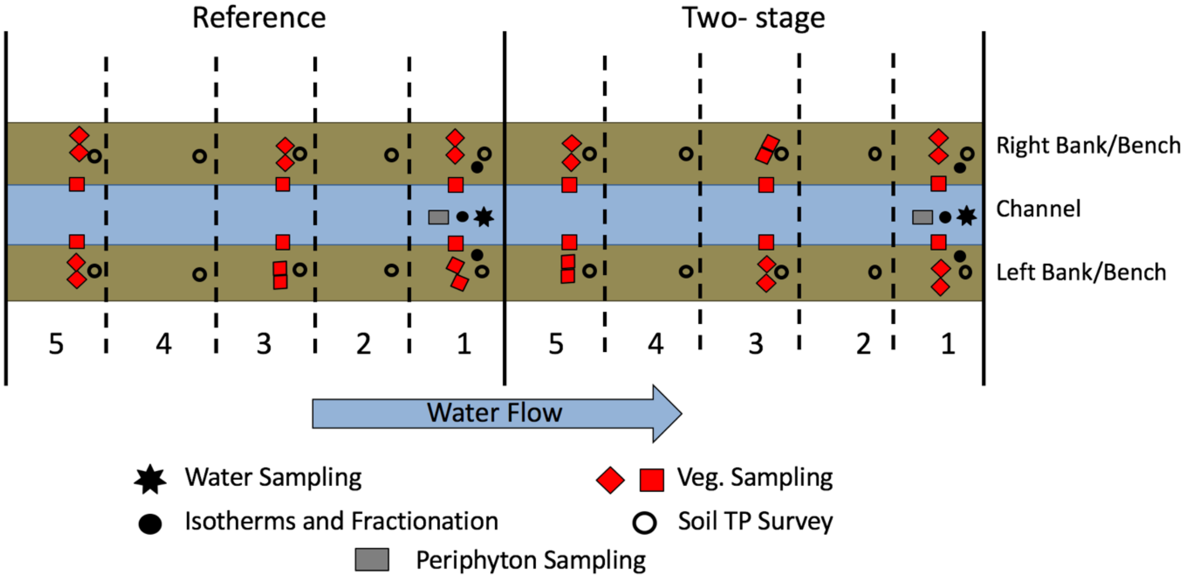


Figure 3.2

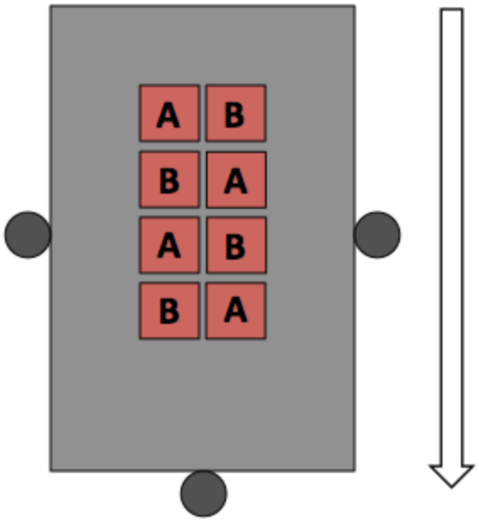


Figure 3.3

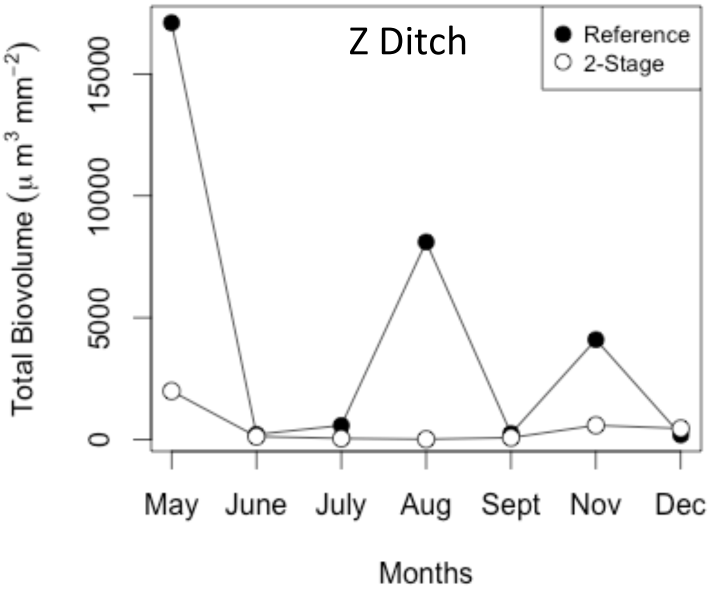


Figure 3.4

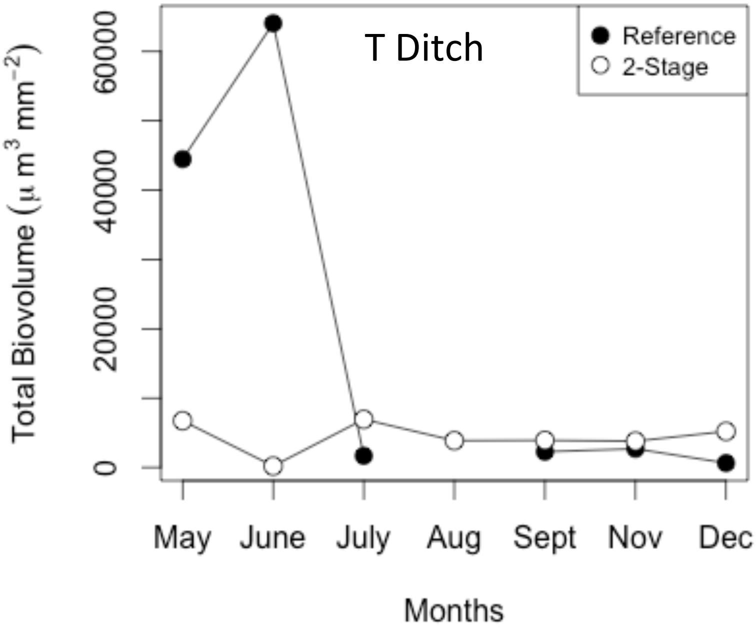
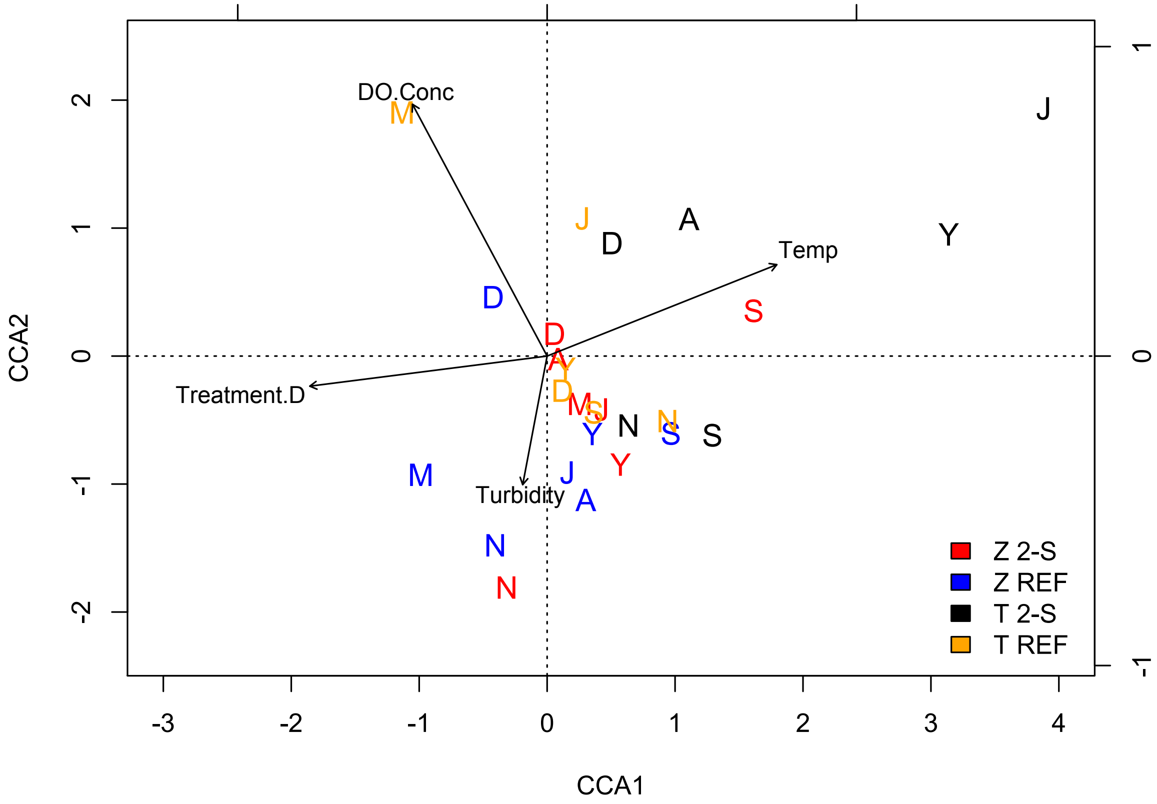

T Ditch
● Reference
○ 2-Stage
Total Biovolume ($\mu m^3 mm^{-2}$)
0
20000
40000
60000
May
June
July
Aug
Sept
Nov
Dec
Months

Figure 3.5



DO.Conc
Temp
Treatment.D
Turbidity
CCA1
CCA2
-3
-2
-1
0
1
2
3
4
Z 2-S
Z REF
T 2-S
T REF
M
J
D
A
Y
S
N

Figure 3.6

Chapter 4

Synthesis and Conclusions

Production of row crops for food and energy represents the major land use in many watersheds across the country. As of 2014, approximately 45% of the land in the United States was being used for agriculture, whether for row crops, livestock, or rangeland (The World Bank 2017). Row crop agriculture typically applies nutrients through fertilizer for improved crop yield. Agriculture associated with livestock can impact the stability of vegetative communities and produces bioavailable nutrient-rich manure as a byproduct. The movement of nutrients through water, erosion, transport, and improper management can disrupt the balance within ecosystems. While most nitrogen (N) for fertilizer is industrially fixed from the atmosphere by the Haber-Bosch process (Gruber and Galloway 2008), most phosphorus (P) used in fertilizer is mined from rock (Jarvie et al. 2015). As more N is fixed industrially and more P is mined, the amount of bioavailable nutrients moving through global cycles increases. These excess nutrients can have negative impacts on ecosystems by increasing eutrophication and the likelihood for harmful algal blooms.

Two-stage systems have been shown to significantly reduce N export through drainage ditches while having variable potential to retain P (Roley et al. 2012a; Roley et al. 2012b; Mahl et al. 2015). While the mechanism for N removal is well-studied, it is less clear how two-stage ditches influence P fate and transport (Kallio et al. 2010). The two two-stage ditches studied for this thesis represented two very different systems and two different surprising outcomes. While the Z two-stage reach functioned as a stable vegetated ditch and the morphology appeared to be "ideal" (Powell et al. 2007), it did not significantly reduce P export in drainage water. The T two-stage reach suffered from erosion, unstable banks, little vegetation growth on the benches, and significant algal mat growth, but did significantly reduce the TP and SRP in baseflow drainage.

Soils and Retention

In the Z ditch, only turbidity and soil TP were significantly lower within the two-stage reach compared to the reference reach (Fig. 4.1). The main reduction in turbidity occurred at stormflow when water was actually flowing over the benches. During stormflow, TP concentrations were an order of magnitude higher than at baseflow. Only one snowmelt event and 3 storm events were sampled during 2016. Any potential reduction in TP due to settled particulate matter was either very small in comparison to the overall TP moving through the reach at high flow or this difference was not detectable with only a total of four high flow events.

In the T ditch, water TP and SRP as well as soil TP were significantly lower within the two-stage reach compared to the reference reach; conversely, turbidity, pH, and temperature were significantly higher within the two-stage reach compared to the reference reach (Fig. 4.1). TP and SRP were reduced within the two-stage reach presumably due to increased mat algal growth and increased surface area contact between baseflow water and the soil. Turbidity increased in the two-stage reach likely due to erosion that occurred mostly during high flow events. The pH increase in the two-stage could be due to the increased photosynthesis by the filamentous mat algae (Welch and Jacoby 2004). The temperature of the water increased due to low vegetation cover and reduced shading as well as the very shallow depth at baseflow.

Soil TP was significantly lower in both Z and T two-stage reaches compared to their corresponding reference reach (Fig. 4.1). In both reaches of Z, the majority of the P stored within the sediment and soil was bound within the more stable Al-bound fraction. In both reaches of T, the majority of the P stored within the sediment and soil was within both the Al- and Ca- bound fractions. As Al- and Ca-bound P is more stable in sediment and soil than loosely-bound or Fe-bound P, the majority of P retained within will be relatively stable in all reaches at both sites.

Z equilibrium P concentration (EPC) values suggested that P was more likely to stay bound within the two-stage compared to the reference. T EPC values suggested that there was release of P from the sediment and soil of both two-stage and reference in the spring and but only the reference soil in the fall. The sites within the reference that were more likely to release P were the banks. These banks may be more likely to dry out compared to the benches or are inundated less often and therefore have more mineralized or loosely-bound P to release upon the next storm event. %OM tended to be lower in the two-stage compared to the reference in both Z and T ditches but there was no significant difference. All %OM was low, approximately 4% in the Z ditch and approximately 3% in the T ditch.

Vegetation and periphyton P accounted for less than 4% of the total P in each reach. In the Z ditch, vegetation cover, vegetation biomass, vegetation P content, and periphyton P content were not significantly different between two-stage and reference. In the T ditch, only bench vegetation cover was significantly lower in the two-stage compared to the reference. Due to the lack of vegetation on the two-stage, no vegetation biomass and vegetation P content analyses were conducted.

Impacts on Periphyton

In the Z ditch, algal density and biovolume tended to be lower in the two-stage compared to the reference reaches but were not significantly different. Also, there were no significant differences in number of taxa, % diatom taxa, or community diversity between two-stage and the reference reaches or among months. The Z ditch reference reach periphyton was dominated by more motile diatom taxa compared to the two-stage periphyton, which was composed of attached and weakly-motile diatom taxa. The lack of differences in periphyton community and biomass is not surprising considering the environmental factors such as vegetation cover (a proxy for irradiance) of the reference and two-stage were very similar in 2016. Also, during construction of the two-stage ditch one and a half years prior to the start of this project, the original wetted channel was not dredged or changed in any way and so the foundation of the algal community before construction still existed at the beginning of this project. The two-stage reach was not long enough to prevent migration of some cells through baseflow and stormflow from the downstream end of the reference to the downstream end of the two-stage. However, as the two-stage matures, the reach will most likely continue to reduce turbidity and collect more suspended sediments, thereby changing the available substrate and potentially increasing the relative number or biovolume of mobile taxa (Wagenhoff et al. 2013).

In the T ditch, algal density was significantly lower in the two-stage reach compared to the reference reach but the total biovolume was not significantly different. This suggests that while the number of cells decreased in the two-stage, the size of the cells was larger. The T ditch reference and two-stage reaches were both dominated by motile algae. The open, unshaded two-stage benches were consistently wet and facilitated the growth of large algal mats. While these large filamentous algal mats may have had a part in the reduction of TP and SRP within the two-stage reach, the mats were not analyzed for this study. As the two-stage matures, the benches become more stable, and vegetation grows on the benches, the algal communities will probably become even more similar between two-stage and reference.

The variable P retention results do not mean that two-stage ditches cannot significantly reduce P in the Macatawa watershed; rather, there are many interactions between the driving factors of P retention and release. Figure 4.2 is a conceptual model of water movement, erosion, and P interactions with sediment/soil, vegetation, and periphyton. Both two-stage and reference reaches are shown at baseflow and stormflow conditions. The solid arrows represent the movement of water; the thick arrows represent the major water movement from land into the ditch for each scenario. The dashed lines represent the movement of sediment (and associated sediment bound P). The arrow circles represent the cycling of P through uptake into autotrophs and then release through cell death and decomposition. The brackets represent adsorption of P to sediment and soil and potential release back into the water column.

During baseflow conditions, some tile drains will transport water from fields into the ditch. Tile drainage has been shown to account for around 50% of the annual P load in certain watersheds (King et al. 2015, Smith et al. 2015) and in the Macatawa, percent SRP from tile drains often exceeded 50% of the total P being exported (Clement and Steinman 2017). In the two-stage reach, that water is deposited onto the bench where adsorption and uptake by vegetation can occur. In the reference, the drainage water moves almost directly from the pipe to the wetted channel with very little interaction with the ditch banks. High drainage flow may cause erosion to occur on the bank sides, releasing soil and bound P into the channel. During stormflow, the tile drains (not shown in the diagram) may still move a large amount of water depending on the intensity and duration of the rain event, as well as soil conditions and tile drain construction; however, surface runoff from the field tends to be the main source of discharge during storms (King et al. 2015, Smith et al. 2015). The two-stage form slows water flowing through the ditch during high flow conditions, increasing the time for uptake by vegetation and periphyton, adsorption to sediment/soil, and sediment particle deposition. Intense flows typical in flashy traditional ditches can scour vegetation, algae, and sediment/soil reducing both the stability of the ditch and the ability to retain P and sediment.

The Z two-stage reach functioned much like this conceptual model. Sediment was deposited and reduced turbidity. The presence of algae and thick vegetation growth allowed for uptake of nutrients within the reach. However, these interactions were not enough to significantly reduce the TP or SRP moving through baseflow and stormflow water. Likely this is due to the reach length and the residence time within the reach. Collins et al. (2016) found that residence time had to exceed 96 hours (four days) in systems with sediment with low organic matter to result in net retention. Even extreme high flow events in the Z ditch would raise and lower below the benches within 24 hours of a storm event.

The T two-stage functioned differently than the conceptual model. Throughout the spring and during any storm event, there was significant erosion and morphological change. This was due to a lack of vegetation on the benches and very sandy soil. The benches lowered and the active channel migrated to one or the other bench for the majority of the reach. Water was still stored within the original channel as well and this created wet conditions across much of the surface area of the ditch (except during late summer when water levels were significantly lower). The surface area of soil that water flowed over was much larger by the end of the sampling season. While some portions of the benches remained as benches, other portions experienced wide and shallow sheetflow even at baseflow conditions. Large mats of algae grew in the original channel and on the wetted channel taking up P along with the channel edge vegetation. While there was enough erosion to elevate the turbidity levels, I speculate that the temporary storage of P within algae as well as increased adsorption to the soil, due to larger surface area, significantly reduced the TP and SRP in baseflow. While the residence time within the ditch may not have surpassed the 96-hour mark as described above (Collins et al. 2016), more of the flow in T had actual contact with the soil allowing for transfer of P compared to the Z ditch.

At the beginning of this project, it was determined that there were no open tile drains actively draining from the fields into either two-stage reach. After vegetation died back in November 2016, it became apparent that there was an old tile drain pipe that had become exposed in the Z two-stage. This old drain was running under the bench and into the channel. It was not possible to determine whether or not the pipe was actively flowing and draining water from the field. This drain was not taken into consideration during construction because the producer was unaware of its presence. While exact acreage of tile drained land within the Macatawa watershed is unknown, Clement (2016) estimates approximately 25-35% of the fields are tiled drained whether the current producer is aware or not. Before construction of a two-stage system, it is important to understand what other management practices are in place in order to optimize the effectiveness of the management.

Conclusions

Overall, P can be retained in periphyton, vegetation, and soils, and two-stage ditches are capable of retaining P under certain conditions. P will be bound in vegetation and periphyton temporarily but can be released as organisms die seasonally and decompose. Plant matter and algal cells also can be scoured during intense flows and be transported downstream along with their organically bound nutrients. Unfortunately, factors potentially impacting the effectiveness of this temporary sink such as shading by channel vegetation and algal mats were not measured quantitatively for this project. While vegetation and periphyton may be able to utilize nutrients during high load times during the growth season, future research should continue to focus on combined sediment and soil as the largest sink of P within these systems.

Legacy P is P that is bound within ecological systems and is retained for longer periods of time compared to traditional cycling (Sharpley et al. 2014), such as with Ca-bound P within sediment and soil. Legacy P can make it difficult to assess the effectiveness of management practices (Marsden 1989). For example, P adsorbed to sediment and soil may stay bound at low EPCs. If water P concentrations are lowered past the EPC point through management and restoration, P can be released back into the water column. When monitoring management practices and remediation, it is important to understand that factors such as legacy P may decrease or postpone the visible effects of restoration at a large scale. In this project, the adjacent two-stage and reference reaches sampled at the downstream ends allowed for estimation of the two-stage impact only on a short time scale and a short distance scale. While the soil within the two-stage was more likely to retain P than the soil within the reference, this comparison was conducted at current SRP concentrations. Reduction in P export from fields could increase the release of legacy P held in the sediment and soil and postpone the appearance of positive remediation.

Two-stage ditches should continue to be considered as a best management practice option within the Macatawa watershed. Lake Macatawa suffers from both P and sediment input from the watershed. Two-stage ditches, once stable with vegetation growing on the benches may be able to significantly reduce the turbidity in drainage water. These ditches can retain the P stored within their sediment and soil and also have the potential to reduce P export. Hence, we recommend longitudinal studies to assess the efficacy of these BMPs over time, as well as include a more detailed look at how the traditional reaches are impacting TP, SRP, and turbidity in comparison.

It is imperative that season, position and length of reach, cost, and stability of soil should all be considered before constructing a two-stage reach. The constructed two-stage should be stable enough to reduce the initial erosion from the system after construction. Allowing for vegetation growth time on the benches before seasonal rainfall or snowmelt can significantly impact the stability of the construction. It is hypothesized that as two-stage reaches mature, the greater the impact of P retention the ditch will have. Managers should consider length and position of proposed systems compared to the cost of construction. Two-stage ditches cannot reduce the P export from the Macatawa watershed alone. Other operational and structural BMPs can complement two-stage ditches to reduce P export within the Macatawa watershed.

References

Clement, D.R. 2016. Does Phosphorus from Agricultural Tile Drains Fuel Algal Blooms?. Unpublished Thesis. Grand Valley State University.

Clement, D.R., Steinman, A.D. 2017. Phosphorus loading and ecological impacts from agricultural tile drains in a west Michigan watershed. J. Gt. Lakes Res. 43: 50-58.

Collins, S.D., Shukla, S., Shrestha, N.J. 2016. Drainage ditches have sufficient adsorption capacity but inadequate residence time for phosphorus retention in the Everglades. Ecol. Eng. 92: 218-228.

Gruber, N. Galloway, J.N. 2008. An earth-system perspective of the global nitrogen cycle. Nature 451: 293-296.

Jarvie, H.P., Sharpley, A.N., Flaten, D., Kleinman, P.J., Jenkins, A., Simmons, T. 2015. The pivotal role of phosphorus in a resilient water-energy-food security nexus. J. Environ. Qual. 44: 1049-1062.

Kallio, S., Ward, A., D'Ambrosio, J.L., Witter, J.D. 2010. Beyond the field: a look at agricultural ditch floodplains as a water quality BMP. CIGR XVII World Congress.

King, K.W., Williams, M.R., Macrae, M.L., Fausey, N.R., Frankenberger, J., Smith, D.R., Kleinman, P.J.A., Brown, L.C. 2015. Phosphorus transport in agricultural subsurface drainage: a review. J. Environ. Qual. 44: 467-485.

Mahl, U.H., Tank, J.L., S.S. Roley, Davis, R.T. 2015. Two-stage ditch floodplains enhance N-removal capacity and reduce turbidity and dissolved P in agricultural streams. J. Am. Water Res. Assoc. 51: 923-940.

Marsden, M.W. 1989. Lake restoration by reducing external phosphorus loading: the influence of sediment phosphoru release. Freshwater Biol. 21: 139-162.

Powell, G.E., Ward, A.D., Mecklenburg, D.E., Jayakaran, A.D. 2007. Two-stage channel systems: Part 1, a practical approach for sizing agricultural ditches. J. Soil Water Cons. Soc. 62: 277-285.

Roley, S.S., Tank, J.L., Stephen, M.L., Johnson, L.T., Beaulieu, J.J., Witter, J.D. 2012a. Floodplain restoration enhances denitrification and reach-scale nitrogen removal in an agricultural stream. Ecol. Appl. 22: 281-297.

Roley, S.S., Tank, J.L., and Williams, M.A. 2012b. Hydrological connectivity increases denitrification in the hyporheic zone and restored floodplains of an agricultural stream. J. Geophys. Res. 117: 1-16.

Sharpley, A., Jarvie, H.P., Buda, A., May, L., Spears, B., Kleinman, P. 2014. Phosphorus legacy: overcoming the effects of past management practices to mitigate future water quality impairment. J. Environ. Qual. 42: 1308-1326.

Smith, D.R., King, K.W., Johnson, L., Francesconi, W., Richards, P., Baker, D., Sharpley, A.N. 2015. Surface runoff and tile drainage transport of phosphorus in the Midwestern United States. J. Environ. Qual. 44: 495-502.

The World Bank. “Agricultural land (% of land area)”. 2017. http://data.worldbank.org/indicator/AG.LND.AGRI.ZS?name_desc=false).

Wagenhoff, A., Lange, K., Townsend, C.R., Matthaei, C.D. 2013. Patterns of benthic algae and cyanobacteria along twin-stressor gradients of nutrients and fine sediment: a stream mesocosm experiment. Freshwater Biol. 58: 1849-1863.

Welch, E.B., Jacoby, J.M. 2004. Chapter 4 Nutrient Cycles, in: Pollutant Effects in Freshwater-Applied Limnology 3rd Ed. Taylor & Francis, New York, pp. 56-85.

Figures

Figure 4.1 A map of important results for the Z and T ditches with the point of view from reference to two-stage. Note that slope of lines is not to scale and do not signify increase or decrease. Decrease in algal biomass in Z not a trend but not quite significant (p= 0.06). “Veg.” = Vegetation, “Temp.” = Temperature.

Figure 4.2 A conceptual model for movement of water, soil, sediment, and P within agricultural drainage ditches. Solid arrows = movement of water (thick arrows represent the major water movement from land into the ditch); Dashed arrows = movement of sediment and associated bound P; Arrow circles = cycling of P through uptake into autotrophs and then release through cell death and decomposition; Brackets = absorption of P to sediment and soil and potential release back into the water column. Stormflow conditions include any high flow event such as thunderstorms or snowmelt.

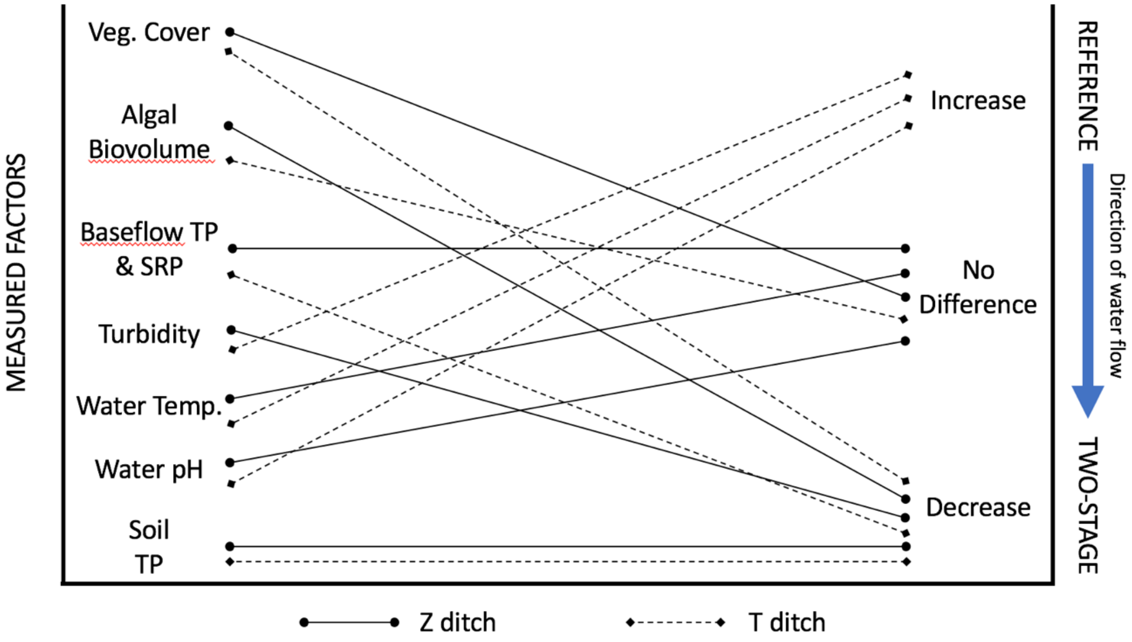

MEASURED FACTORS
Veg. Cover
Algal Biovolume
Baseflow TP & SRP
Turbidity
Water Temp.
Water pH
Soil TP
Increase
No Difference
Decrease
REFERENCE
Direction of water flow
TWO-STAGE
Z ditch
T ditch

Figure 4.1

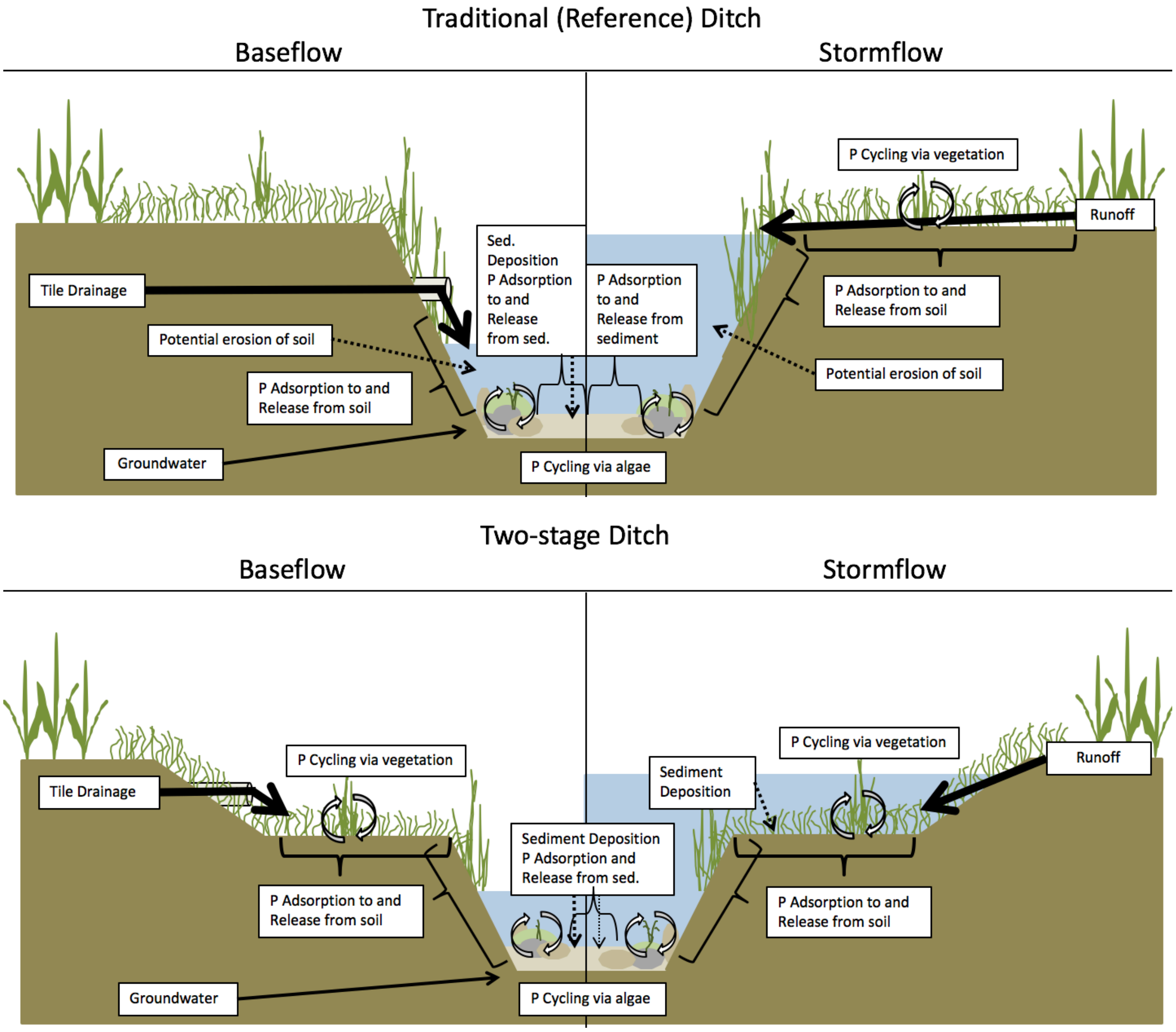

Traditional (Reference) Ditch
Baseflow
Stormflow
Tile Drainage
Potential erosion of soil
P Adsorption to and Release from soil
Groundwater
Sed. Deposition P Adsorption to and Release from sed.
P Adsorption to and Release from sediment
P Cycling via algae
P Cycling via vegetation
Runoff
P Adsorption to and Release from soil
Potential erosion of soil
Two-stage Ditch
Baseflow
Stormflow
Tile Drainage
P Cycling via vegetation
P Adsorption to and Release from soil
Groundwater
Sediment Deposition P Adsorption and Release from sed.
P Cycling via algae
Sediment Deposition
P Cycling via vegetation
Runoff
P Adsorption to and Release from soil

Figure 4.2

References

Baldwin, D.S., Mitchell, A.M. 2000. The effects of drying and re-flooding on the sediment and soil nutrient dynamics of lowland river-floodplain systems: a synthesis. Regul. River. 16: 457-567.

Bellinger, E.G., Sigee, D.C. 2015. A key to the more frequently occurring freshwater algae. Pages 141-251. In: Freshwater Algae Identification, Enumeration and Use as Bioindicators. (2nd Ed.) John Wiley and Sons, Ltd.

Berthold, M., Zimmer, D., Schumann, R. 2015. A simplified method for total phosphorus digestion with potassium persulphate at sub-boiling temperatures in different environmental samples. Rostock. Meeresbiolog. Beitr. 25: 7-25.

Biggs, B.J.F., Stevenson, R.J., Lowe, R.L. 1998. A habitat matrix conceptual model for stream periphyton. Arch. Hydrobiol. 143: 21-56.

Blann, K.L., Anderson, J.L., Sands, G.R., Vondracek, B. 2009. Effects of agricultural drainage on aquatic ecosystems: a review. Critical reviews in environmental science and technology 39: 909-1001

Brennan, R.B., Scott, J.T., Sharpley, A.N., Lally, H.T., Jarvie, H.P., Bowes, M.J., Haggard, B.E., Gbur, E. 2017. Linking soil erosion to instream dissolved phosphorus cycling and periphyton growth. J. Am. Water Res. Assoc. 1-13. DOI: 10.1111/1752-1688.12534

Carr, J.M., Hergenrader, G.L., Troelstrup Jr., N.H. 1986. A simple, inexpensive method for cleaning diatoms. Trans. Am. Microsc. Soc. 105: 152-157.

Christopher, S.F., Tank, J.L., Mahl, U.H., Yen, H., Arnold, J.G., Trentman, M.T., Sowa, S.P., Herbert, M.E., Ross, J.A., White, M.J., Royer, T.V. 2017. Modeling nutrient removal using watershed-scale implementation of the two-stage ditch. Ecol. Eng. (In Press, http://dx.doi.org/10.1016/j.ecoleng.2017.03.015)

Clement, D.R. 2016. Does Phosphorus from Agricultural Tile Drains Fuel Algal Blooms?. Unpublished Thesis. Grand Valley State University.

Clement, D.R., Steinman, A.D. 2017. Phosphorus loading and ecological impacts from agricultural tile drains in a west Michigan watershed. J. Gt. Lakes Res. 43: 50-58.

Collins, S.D., Shukla, S., Shrestha, N.J. 2016. Drainage ditches have sufficient adsorption capacity but inadequate residence time for phosphorus retention in the Everglades. Ecol. Eng. 92: 218-228.

D'Ambrosio, J.L, Ward, A.D., Witter, J.D. 2015. Evaluating geomorphic change in constructed two-stage ditches. J. Gt. Lakes Res. 51:910-922.

Davis, R.T., Tank, J.L., Mahl, U.H., Winikoff, S.G., Roley, S.S. 2015. The influence of two-stage ditches with constructed floodplains on water column nutrients and sediments in agricultural streams. J. Am. Water. Res. Assoc. 51: 941-955.

Dieter, D., Herzog, C., Huper, M. 2015. Effects of drying on phosphorus uptake in re-flooded lake sediments. Environ. Sci. Pollut. R. 22: 17065-17081.

Dodds, W.K., Bouska, W.W., Eitzmann, J.L., Pilger, T.J., Pitts, K.L., Riley, A.J., Schloesser, J.T., Thornbrugh, D.J. 2009. Eutrophication of U.S. freshwaters: analysis of potential economic damages. Environ. Sci. Technol. 43: 12-19.

Dodds, W.K., Smith, V.H., Lohman, K. 2002. Nitrogen and phosphorus relationships to benthic algal biomass in temperate streams. Can. J. Fish. Aquat. Sci. 59: 865-874.

Drake, W.M., Scott, J.T., Evans-White, M., Haggard, B., Sharpley, A., Rogers, C.W., Grantz, E.M. 2012. The effect of periphyton stoichiometry and light on biological phosphorus immobilization and release in streams. Limnology 13: 97-106.

Fanta, S.E., Hill, W.R., Smith, T.B., Roberts, B.J. 2010. Applying the light: nutrient hypothesis to stream periphyton. Freshwater Biol. 55: 931-940

Graham, A.A. 1990. Siltation of stone-surface periphyton in rivers by clay-sized particles from low concentrations in suspension. Hydroliologia 199: 107-115.

Gruber, N. Galloway, J.N. 2008. An earth-system perspective of the global nitrogen cycle. Nature 451: 293-296.

Hassett, M., Oudsema, M., Steinman, A.D. 2017. Project Clarity 2016 Annual Monitoring Report. www.gvsu.edu/cms4/asset/DFC9A03B-95B4-19D5-F96AB46C60F3F345/final_report_2016_water_year.pdf

Hill, W.R., Fanta, S.E., Roberts, B.J. 2009. Quantifying phosphorus and light effects in stream algae. Limnol. Oceanogr. 54: 368-380.

Hill, W.R., Knight, A.W. 1988. Nutrient and light limitation of algae in two northern California streams. J. Phycol. 24: 125-132.

Hillebrand, H., Durselen, C.D., Kirschtel, D., Pollingher, U., Zohary, T. 1999. Biovolume calculation for pelagic and benthic microalgae. J. Phycol. 35: 403-424.

Hongthanat, N., Kovar, J.L., Thompson, M.L., Russell, J.R., Isenhart, T.M. 2016. Phosphorus source-sink relationships of stream sediments in the Rathbun Lake watershed in southern Iowa, USA. Environ. Monit. Assess. 188: 453-467.

Hupfer, M., Zak, D., Rofberg, R., Herzog, C., Pothig, P. 2009. Evaluation of a well-established sequential phosphorus fractionation technique for use in calcite-rich lake sediments: identification and prevention of artifacts due to apatite formation. Limnol. Oceanogr: Methods 7, 2009: 399-410

Jarvie, H.P., Sharpley, A.N., Flaten, D., Kleinman, P.J., Jenkins, A., Simmons, T. 2015. The pivotal role of phosphorus in a resilient water-energy-food security nexus. J. Environ. Qual. 44: 1049-1062.

Kallio, S., Ward, A., D'Ambrosio, J.L., Witter, J.D. 2010. Beyond the field: a look at agricultural ditch floodplains as a water quality BMP. CIGR XVII World Congress.

King, K.W., Williams, M.R., Macrae, M.L., Fausey, N.R., Frankenberger, J., Smith, D.R., Kleinman, P.J.A., Brown, L.C. 2015. Phosphorus transport in agricultural subsurface drainage: a review. J. Environ. Qual. 44: 467-485.

Kinsman- Costello, L.E., Hamilton, S.K., O'Brien, J.M., Lennon, J.T. 2016. Phosphorus release from the drying and reflooding of divers shallow sediments. Biogeochemistry. DOI: 10.1007/s10533-016-0250-4.

Lamb, M.A., Lowe, R.L. 1987. Effects of current velocity on the physical structuring of diatom (Bacillariophyceae) communities. Ohio J. Sci. 87: 72-78.

Lange, K., Liess, A., Piggot, J.J., Townsend, C.R., Matthaei, C.D. 2011. Light, nutrients and grazing interact to determine stream diatom community composition and functional group structure. Freshwater Biol. 56: 264-278.

Lange, K., Townsend, C.R., Matthaei, C.D. 2015. A trait-based framework from stream algal communities. Ecol. and Evol. DOI: 10.1002/ece3.1822

Larned, S.T. 2010. A prospectus for periphyton: recent and future ecological research. J. North Am. Benthol. Soc. 29: 182-206.

Liess, A., Le Gros, A., Wagenhoff, A., Townsend, C.R., Matthaei, C.D. 2012. Landuse intensity in stream catchments affects the benthic food web: consequences for nutrient supply, periphyton C:nutrient rations, and invertebrate richness and abundance. Freshwater Sci. 31: 813-824.

MACC. 2012. Macatawa Watershed Management Plan Appendix H www.the-macc.org/watershed/overview/

MACC. 2014. Macatawa Watershed Annual Progress Report. www.the-macc.org/watershed/overview/

Maccoux, M.J., Dove, A., Backus, S.B., Dolan, D.M. 2016. Total and soluble reactive phosphorus loadings to Lake Erie: A detailed accounting by year, basin, country and tributary. J Great Lakes Res. 42:1151-1165.

Mahl, U.H., Tank, J.L., S.S. Roley, Davis, R.T. 2015. Two-stage ditch floodplains enhance N-removal capacity and reduce turbidity and dissolved P in agricultural streams. J. Am. Water Res. Assoc. 51: 923-940.

Marsden, M.W. 1989. Lake restoration by reducing external phosphorus loading: the influence of sediment phosphoru release. Freshwater Biol. 21: 139-162.

McDowell, R.W., Elkin, K.R., Kleinman, P.J.A. 2017. Temperature and nitrogen effects on phosphorus uptake by agricultural stream-bed sediments. J. Eviron. Qual. Technical Reports: 1-7.

Moore, M.T., Kroger, R. 2011. Evaluating plant species-specific contributions of nutrient mitigation in drainage ditch mesocosms. Water Air Soil Pollut. 217: 445-454.

Nair, V.D., Clark, M.W., Reddy, K.R., 2015. Evaluation of legacy phosphorus storage and release from wetland soils. J. Environ. Qual. 44: 1956-1964.

NOAA. National Overview- Annual 2016. "State of the Climate". www.ncdc.noaa.gov/sotc/national/201613.

Palmer-Felgate, E.J., Mortimer, R.J.G., Krom, M.D., Jarvie, H.P. 2010. Impact of point-source pollution on phosphorus and nitrogen cycling in stream-bed sediments. Environ. Sci. Technol. 44: 908- 914.

Pant, H.K., Reddy K.R. 2001. Phosphorus sorption characteristics of estuarine sediments under different redox conditions. J. Environ. Qual. 30: 1474-1480.

Powell, G.E., Ward, A.D., Mecklenburg, D.E., Jayakaran, A.D. 2007. Two-stage channel systems: Part 1, a practical approach for sizing agricultural ditches. J. Soil Water Cons. Soc. 62: 277-285.

Project Clarity. Outdoor Discovery Center Macatawa Greenway. www.macatawaclarity.org

Psenner, R., Pucsko, R. 1988. Phosphorus fractionation: advantages and limits of the method for the study of sediment P origins and interactions. Arch. Hydrobiol. Beih. Ergebn. Limnol. 30: 43-59.

Qin, P. Mayer, C.M., Schulz, K.L., Ritchie, M.E. 2007. Ecological stoichiometry in benthic food webs: effect of light and nutrients on periphyton food quantity and quality in lakes. Limnol. Oceanogr. 52: 1728-1734.

Reddy, K.R., D'Angelo, E.M., Harris, W.G. 2000. Biogeochemistry of Wetlands. Handbook of

Soil Science. M.E. Sumner, ed. G89-119.

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

Reddy, K.R., Newman, S., Osborne, T.Z., White, J.R., Fitz, H.C. 2011. Phosphorus cycling in the Everglades ecosystem: Legacy phosphorus implications for management and restoration. Crit. Rev. Environ. Sci. Technol. 41: 149-186.

Roley, S.S., Tank, J.L., Stephen, M.L., Johnson, L.T., Beaulieu, J.J., Witter, J.D. 2012a. Floodplain restoration enhances denitrification and reach-scale nitrogen removal in an agricultural stream. Ecol. Appl. 22: 281-297.

Roley, S.S., Tank, J.L., and Williams, M.A. 2012b. Hydrological connectivity increases denitrification in the hyporheic zone and restored floodplains of an agricultural stream. J. Geophys. Res. 117: 1-16.

Sallade, Y.E., and Sims, J.T. 1997. Phosphorus transformation sin the sediments of Delaware's agricultural drainageways: II. Effect of reducing conditions on phosphorus release. J. Environ. Qual. 26: 1579-1588.

Samanta, S., Debnath, D., Maitra, N., Banerjee, M., Chowdhury, A.N., Sharma, A.P., Manna, S.K. 2015. Sediment phosphorus forms and levels in two tropical floodplain wetlands. Aquat. Ecosyst. Health Manag. 18:467-474.

Scavia, D., Allan, J.D., Arend, K.K., Bartell, S., Beletsky, D., Bosch, N.S., Brandt, S.B., Briland, R.D., Daloglu, I., DePinto, J.V., Dolan, D.M., Evans, M.A., Farmer, T.M., Goto, D., Han, H., Hook, T.O., Knight, R., Ludsin, S.A., Mason, D., Michalak, A.M., Richards, R.O., Roberts, J.J., Rucinski, D.K., Rutherford, E., Schwab, D.J., Sesterhenn, T.M., Zhang, H., Zhou, Y. 2014. Assessing and addressing the re-eutrophication of Lake Erie: central basin hypoxia. J. Great Lakes Res. 40: 226-246.

Sharpley, A., Jarvie, H.P., Buda, A., May, L., Spears, B., Kleinman, P. 2014. Phosphorus legacy: overcoming the effects of past management practices to mitigate future water quality impairment. J. Environ. Qual. 42: 1308-1326.

Silvan, N., Vasander, H., Laine., J. 2004. Vegetation is the main factor in nutrient retention in a constructed wetland buffer. Plant Soil 258: 179-187.

Smith, D.R., Haggard, B.E., Warnemuende, E.A., Huang, C. 2005. Sediment phosphorus dynamics for three tile fed drainage ditches in northeast Indiana. Ag. Water Manag. 71: 19-32.

Smith, D.R., King, K.W., Johnson, L., Francesconi, W., Richards, P., Baker, D., Sharpley, A.N. 2015. Surface runoff and tile drainage transport of phosphorus in the Midwestern United States. J. Environ. Qual. 44: 495-502.

Steinman, A.D., Lamberti, G.A., Levitt, P. 2006. Biomass and pigments of benthic algae. Pgs 357-379. In: Methods in Stream Ecology. Hauer, R. and G. Lamberti (ed). Academic Press, NY.

Steinman, A.D., Ogdahl, M.E. 2013. Muskegon lake AOC habitat restoration design: Bear Lake hydrologic reconnection/wetland restoration. www.gvsu.edu/cms4/asset/DFC9A03B-95B4-19D5-F96AB6C60F3F345/bear_muck_final_report_final.pdf

Steinman, A.D., Ogdahl, M.E. 2016. From wetland to farm and back again: phosphorus dynamics of a proposed restoration project. Environ. Sci. Pollut. Res. 23:22596-22605.

Steinman, A.D., Ogdahl, M.E., Weinert, M., Uzarski, D.G. 2014. Influence of water level fluctuation duration and magnitude on sediment-water nutrient exchange in coastal wetlands. Aquat. Ecol. 48: 143-159.

Steinman, A.D., Duhamel, S. 2017. Phosphorus limitation, uptake, and turnover in stream algae. Pages 197-218. In: Methods in Stream Ecology. Vol II (3rd Ed.) R. Hauer and G. Lamberti (Ed.). Elsevier Press.

Stevenson, R.J., Bennett, B.J., Jordan, D.N., French, R.D. 2012. Phosphorus regulates stream injury by filamentous green algae, DO, and pH with thresholds in responses. Hydrobiologia 695: 25-42.

Sun, J., Liu, D. 2003. Geometric models for calculating cell biovolume and surface area for phytoplankton. J. Plankton Res. 25: 1331-1346.

Teng, W., Deng, Y., Chen, X.P., Xu, X.F., Chen, R.Y., Lv, Y., Zhao, Y.Y., Zhao, X.Q., He, X., Li, B., Tong, Y.P., Zhang, F.S., Li, Z.S. 2013. Characterization of root response to phosphorus supply from morphology to gene analysis in field-grown wheat. J. Exper. Bot. 64: 1403-1411.

The World Bank. “Agricultural land (% of land area)”. 2017. http://data.worldbank.org/indicator/AG.LND.AGRI.ZS?name_desc=false).

USEPA. 1983. Methods for chemical analysis of water and wastes. USEPA, Washington, D.C.

Wagenhoff, A., Lange, K., Townsend, C.R., Matthaei, C.D. 2013. Patterns of benthic algae and cyanobacteria along twin-stressor gradients of nutrients and fine sediment: a stream mesocosm experiment. Freshwater Biol. 58: 1849-1863.

Wang, C., Zhang, Y., Li, H., Morrison, R.J. 2013. Sequential extraction procedures for the determination of phosphorus forms in sediment. Limnology 14: 147-157.

Wang, L., Liang, T. 2015. Distribution characteristics of phosphorus in the sediments and overlying water of Poyang lake. PLOSone 10(5): e0125859.

Wang, L., Wang, G., Ge, X., Yu, Z., Gu, X., Cheng, Z., Xu, W., Zhang, F. 2012. Influence of submerged plants on phosphorus fractions and profiles of sediments in Gucheng Lake. Soil Sed. Contam. 21: 640-654.

Wang, Q., Li, Y., Ouyang, Y. 2011. Phosphorus fractionation and distribution in sediments from wetlands and canals of a water conservation area in the Florida Everglades. Water Res. Res. 47 W05550: 1-11.

Wang, W-C. 1974. Effect of turbidity on algal growth. Illinois Dept. Regist. Ed. Circular 121.

Welch, E.B., Jacoby, J.M. 2004. Chapter 4 Nutrient Cycles, in: Pollutant Effects in Freshwater-Applied Limnology 3rd Ed. Taylor & Francis, New York, pp 56-85.