8-2018

Impacts of Sediment Dredging on Phosphorus Dynamics in a Restored Riparian Wetland

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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 in Biology

Department of Biology

August 2018

Thesis Approval Form

GRAND VALLEY
STATE UNIVERSITY

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Dedication

This master's thesis is dedicated to the staff and faculty at Northland College who mentored me as an undergraduate. Your encouragement and excitement for scientific research greatly influenced my pursuit of graduate education.

Acknowledgements

I sincerely thank my graduate advisor, Dr. Alan Steinman, for challenging me to think clearly and critically about my research and for his continual patience and guidance throughout the writing process. I owe much of the improvement in my scientific reasoning and writing skills to his thoughtful reviews of my proposals and thesis drafts. Additionally, I thank the members of my committee, Dr. Bopaiah Biddanda and Dr. Donald Uzarski, for their valuable comments and insights into my research. I thank the faculty, staff, and students of the Grand Valley State University Biology Department and the Annis Water Resources Institute who have supported, guided, and laughed with me throughout my time as a graduate student. Finally, I thank my family and partner for their support and reminders to get some sunshine and fresh air.

Funding for this study was provided by a Presidential Research Grant from GVSU, an AWRI Graduate Research Assistantship, and Dr. Alan Steinman. Field and laboratory support were provided by Maggie Oudsema, Mike Hassett, Emily Kindervater, Nicole Hahn, Lidiia Iavorivska, Brooke Ridenour, Kurt Thompson, and Paige Kleindl. I thank Brian Scull at AWRI for performing and teaching me how to conduct phosphorus analyses and also Brian Majka at GEI consultants for providing helpful images, details, and data regarding the project construction. The Willbrandt family graciously provided access to their property and helpful background information. Funding for the Bear Creek Hydrologic Reconnection and Aquatic Habitat Restoration Project came from NOAA's Great Lakes Habitat Restoration Program Grant Program and NOAA's Great Lakes Habitat Restoration Regional Partnership Grants. The restoration project would not have been completed without the collaborative support of NOAA, GEI Consultants, and the West Michigan Shoreline Regional Development Commission.

Abstract

Global reductions in biodiversity and water quality are having major consequences for ecosystem health and societal well-being. The restoration of riverine floodplains and wetlands provides an ideal opportunity to increase biodiversity and water quality because their hydrologic connectivity to adjacent streams and rivers promotes the formation of heterogeneous habitat while also facilitating their functioning as a nutrient sink, in general. However, many historic floodplains and riverine wetlands have been drained for the creation of agricultural land, resulting in an accumulation of nutrients in the soils. Therefore, restoration practices that hydrologically reconnect former agricultural land to an adjacent stream or river can stimulate the release of nutrients into downstream waters, at least in the short-term, which can result in the restoration of wildlife habitat at the expense of downstream water quality.

To avoid the high risk of a wetland habitat restoration project in the Muskegon Lake Area of Concern resulting in phosphorus (P) release to downstream waters, the former agricultural land was dredged prior to hydrologic reconnection. I evaluated restoration success by measuring sediment P release in the wetland after dredging and comparing those results to studies that measured P release before dredging. My results showed that maximum P release rates were reduced by 95-99 % after dredging, regardless of temperature or dissolved oxygen treatment. In turn, this avoided between ~25-250 kg of total phosphorus (TP) from entering a eutrophic lake downstream per year (depending on transport scenarios). While internal P loading was drastically reduced, P adsorption isotherm experiments suggested that the deep dredging depth (~1 m on average) exposed sediments with significantly reduced binding capacities, resulting in the wetland acting as a phosphate sink only when water column soluble reactive phosphorus

concentrations exceed 40 µg L^{-1}. This study showed that the ability of sediment dredging to reduce sediment P release largely depends on the underlying sediment characteristics. If pre-restoration monitoring indicates that deeper sediments have low TP and labile P concentrations, sediment dredging can be a useful technique for balancing the goals of both habitat restoration and water quality improvements in wetlands restored on former agricultural lands.

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Abbreviations

Al – Aluminum

AOC – Area of Concern

Ca – Calcium

DO – Dissolved Oxygen

EPC_0 – Equilibrium Phosphorus Concentration

Fe – Iron

Mg – Magnesium

N – Nitrogen

OM –Organic Matter

P – Phosphorus

RMP – Rotations per minute

S_{max} – Phosphorus Sorption Maximum

SRP – Soluble Reactive Phosphorus

TMDL – Total Maximum Daily Load

TP – Total Phosphorus

Chapter 1

Introduction

Riverine floodplains are among the most diverse ecosystems on earth (Tockner & Stanford 2002). This is largely due to the mosaic of habitats and successional stages within floodplains (Ward et al. 2002) that are created from the geomorphological processes of erosion, deposition, and channel migration that occur from the lateral exchange of river water during flood events (Junk et al. 1989). In addition to floodplains' vital role in biodiversity support, they also contribute to over 25 % of all terrestrial ecosystem services, despite covering only approximately 1.4 % of the planet's land surface (Mitsch & Gosselink 2000). The major ecosystem services of floodplains include flood mitigation, nutrient retention, and carbon storage (Tockner & Stanford 2002). The estimated value of the services provided by floodplains is \$19,580 ha yr^{-1} compared to \$969 ha yr^{-1} for forests and \$92 ha yr^{-1} for crops (Constanza et al. 1997).

Despite these benefits, floodplains are one of the most threatened ecosystems on the planet (Tockner & Stanford 2002). In the United States, there are over 2.5 million dams less than 8 meters high (NRC 1992), which has led to 90 % of the total discharge in U.S. rivers having strongly altered hydrology (Jackson et al. 2001). This is problematic for floodplains because modification of river flow with use of dams and levees subsequently reduces flooding frequency and flood magnitude, thereby altering the geomorphological processes needed to maintain floodplain habitat (Ward & Standford 1995).

In addition to alteration of river flow, floodplains are threatened by land-use change. The fertile soils of floodplain wetlands have long been recognized as valuable areas for agricultural production, which has resulted in 46 % of North American floodplains (excluding northern Canada and Alaska) and 80 % - 90 % of European floodplains being used for agricultural production (Tockner & Stanford 2002). The conversion of riverine floodplains into agricultural land greatly diminishes their ability to provide ecosystem services due to their reduction in size (Zedler & Kercher 2005). In addition to the direct anthropogenic threats of land-use change and river flow modification to floodplains, global climate change is expected to exacerbate the loss and degradation of many wetlands (Junk et al. 2003). Climate change is expected to have a pronounced effect on wetlands, such as floodplains, through alterations in hydrological regimes (Bates et al. 2008) and increased water temperatures (Burkett & Kusler 2000).

The recognition of global losses in biodiversity, especially in freshwater ecosystems (Dudgeon et al. 2006; Strayer & Dudgeon 2010), has motivated many wetland restoration projects (Hansson et al. 2005) because wetlands can play a key role in supporting biodiversity of entire regions through provisions of food and shelter for both terrestrial and aquatic species (Findlay & Houlahan 1997; Keddy 2000). As a result of biodiversity loss being closely linked to the degradation of aquatic habitats (Erwin 2009; Strayer & Dudgeon 2010), the restoration of floodplain systems provides an excellent opportunity to aid in increasing biodiversity (Tockner & Stanford 2002).

To restore floodplains for fish and wildlife habitat, it is critical to restore the hydrologic connectivity of these systems to their adjacent streams, as this drives the ability of aquatic organisms to access the floodplain and reestablishes the fluvial dynamics needed to form and maintain the floodplain habitat (Hughes 1997; Ward et al. 1999). However, due to the past agricultural land use of many former floodplains and riverine wetlands (Tockner & Stanford

2002), hydrologic reconnection of these areas to adjacent water bodies causes concern for downstream water quality (Jackson & Pringle 2010).

Even though wetlands are well known for their ability to transform nutrients (Kadlec & Knight 1996), at times they can be a net source rather than a sink of nutrients to downstream waters (Richardson 1985). For example, agricultural fertilization can leave soils saturated with phosphorus (P) and reflooding of this land can stimulate P release into the water column (Pant & Reddy 2003; Montgomery & Eames 2008; Duff et al. 2009; Ardón et al. 2010; Steinman & Ogdahl 2011; Smit & Steinman 2015). Due to sediments having a finite capacity to bind P and concentration gradients playing a driving role in sediment P release (Froelich 1988), sediment P can be released when overlying water column concentrations of dissolved P are less than sediment porewater concentrations (Reddy et al. 1999). Sediments with historically high external P loading (such as agricultural sediments) will release P until an equilibrium is reached between the sediment porewater and overlying water column (Ryding 1981; Marsden 1989). The time needed for equilibrium to occur is variable, as the desorption of P from surface sediments occurs within minutes to hours, while the diffusion of P out of sediment particles can take days to months (Froelich 1988). Indeed, it has been estimated that after reduction of external P loading in some small shallow lakes, sediment P release may occur for up to 30 years before it reaches equilibrium with overlying water column P concentrations (Søndergaard et al. 2001).

Sediment P release as a consequence of floodplain habitat restoration is problematic because in freshwater ecosystems, excess P is often the cause of eutrophication (Schindler 1977). Eutrophication of freshwater resources is a major stressor to water quality worldwide because it causes negative cascading effects throughout a water body, which results in consequences for both human use and environmental health (Smith & Schindler 2009). Some of these effects include harmful algal blooms (Anderson et al. 2002) and bottom water hypoxia with subsequent stress or death to fish (Smith 1988; Weinke & Biddanda 2018). Given that freshwater is necessary for life on earth and aquatic ecosystems are being severely altered and destroyed at a greater rate than at any other time in human history (NRC 1992; Baron et al. 2002), it is imperative that the restoration of floodplain and riverine habitat does not occur at the expense of downstream water quality.

In addition to the challenges that legacy P causes for wetland restoration and water quality management, climate change has been recognized as a major threat to the integrity of wetland ecosystems worldwide (Hulme 2005). This may result in making efforts to restore and manage wetlands more complex (Erwin 2009). The predicted shifts in temperature and precipitation due to climate change are anticipated to threaten floodplain systems primarily through alteration of the hydrologic regime (Poff et al. 2002). However, increased water temperatures driven by climate warming are also likely to increase rates of sediment P release (Jensen & Anderson 1992; Wu et al. 2014), because processes that drive sediment P release, such as microbial activity and rates of P desorption tend to increase with rising temperature (Boström et al. 1988; Froelich 1988). For example, iron redox cycling with subsequent mobilization of iron-bound P has been regarded as one of the primary mechanisms regulating sediment P release (Mortimer 1941; Rydin 2000; Amirbahman et al. 2003; Spears et al. 2007; Smith et al. 2011). However, due to the chemical reduction of Fe(III) to Fe(II) being governed in part by iron-reducing bacteria (Boström et al. 1988), climate warming may stimulate the release of iron-bound P (North et al. 2014) by creating more favorable conditions for iron-reducing bacteria through increased water temperature (Price & Sowers 2004) and enhanced water column stratification (Lovley & Phillips 1988).

The effects of past land use and future climate change combine to create many challenges for present-day floodplain restoration. Despite these challenges, it is increasingly important for the restoration of floodplains to continue, as their ecosystem services improve overall ecosystem health and societal well-being. For ecologically responsible restoration of floodplain habitats to occur, it is imperative for managers to understand the risks of wetland restoration and the ability of management techniques to reduce these risks. With this information, floodplain restoration can become a part of the solution to increasing both global water quality and biodiversity.

Purpose

The purpose of this study is to assess if sediment dredging, prior to the hydrologic reconnection of the west Bear Lake wetland to its adjacent stream, prevented P release to downstream Bear Lake. The findings from this study can contribute to wetland restoration and water quality management decisions by determining the degree to which sediment dredging, prior to hydrologic reconnection, is an effective method for returning agricultural land back to wetland habitat without causing harm to downstream water quality.

Scope

This thesis focuses on how sediment P dynamics in an area once used for agriculture respond to wetland restoration and climate warming. Specifically, I examined differences in water quality, sediment metals, P sorption gradients, sediment P binding capacity, and sediment P release rates before and after dredging. This was done to determine the degree to which sediment dredging, prior to hydrologic reconnection, can reduce sediment P release in a wetland restored on former agricultural land. Additionally, I compared after-dredging sediment P release rates among four treatments, which involved manipulations of water temperature (ambient;

ambient +2 °C) and dissolved oxygen concentration (oxic; hypoxic) to investigate if climate warming may affect dredging results in the short-term.

Assumptions

Sediment cores preserve stratigraphic integrity, physically, chemically, and biologically, which all play a role in determining P cycling. Therefore, it is assumed that results from all analyses and experiments conducted on sediment cores from this study, and referenced studies, are representative of the sediment P dynamics in the systems that they were collected from at that time. However, there is spatial heterogeneity in sediments that cannot be fully captured with sediment cores.

Research Questions (Q) and Hypotheses (H)

Q1) Will the west Bear Lake wetland's sediments act as a potential source or sink of dissolved P after sediment dredging and hydrologic reconnection?

Hypothesis H1: P adsorption isotherm experiments will reveal that the restored wetland acts as a sink for dissolved P by indicating that there is a downward P concentration gradient across the sediment-water interface and that the sediments have additional capacity to adsorb P.

Rationale H1: Before-dredging experiments found that the sediments 30-60 cm below the surface had the potential to sorb nearly double the amount of P that was already present (Steinman & Ogdahl 2016). Dredging will expose these sediments with high P binding capacities to the water column and stimulate a downward P concentration gradient that ultimately results in the restored wetland acting as a sink for dissolved P.

Q2) Will sediment dredging significantly reduce sediment P release in the restored wetland?

Hypothesis H2: Sediment core incubation experiments will show that sediment P release rates are significantly lower in after-dredging treatments compared to before-dredging treatments.

Rationale H2: Sediment dredging can reduce the pool of labile P in the sediments (Yu et al. 2017; Chen et al. 2018); therefore, there will be less mobile P in the sediment available for release after sediment dredging.

Q3) Will climate warming stimulate sediment P release in the restored wetland after dredging?

Hypothesis H3: Sediment core incubation experiments will indicate that climate warming stimulates P release in the after-dredging cores, albeit to a lower degree than before dredging. Hence, treatments with increased water temperature and/or low DO concentration will have higher sediment P release rates than treatments with ambient water temperature and/or high DO concentration

Hypothesis H3: Due to iron-based redox reactions likely driving P dynamics in this wetland, sediment P release rates will be higher in treatments with increased temperature and low DO concentrations, as these conditions facilitate the chemical reduction of Fe(III) to Fe(II) by iron-reducing bacteria and result in increased solubility of iron-bound P.

Significance

Global reductions in freshwater biodiversity (Barnosky et al. 2001; Dudgeon et al. 2006) have motivated many wetland restoration projects to be focused on the creation of fish and wildlife habitat (Hansson et al. 2005), with many of these projects focused on former agricultural lands (Zedler, 2003; USDA 2012). Hydrologic reconnection is often needed to achieve these restoration goals; however, in some highly impacted, human-modified systems, restoration activities that increase the hydrologic connectivity of aquatic systems can result in undesirable ecological effects by increasing the transport of sediment, nutrients, toxins, or invasive species to downstream waters (Jackson & Pringle 2010). Given that there is ongoing eutrophication of freshwater resources worldwide (Smith 2003), it is imperative to avoid such risks as part of wetland restoration. This study will help to determine if sediment dredging in conjunction with knowledge of the underlying sediment characteristics can be used as a technique to avoid situations where wetland restoration is done at the expense of downstream water quality.

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Chapter 2

Impacts of Sediment Dredging on Phosphorus Dynamics in a Restored Riparian Wetland

Abstract

A wetland restoration project within the Muskegon Lake watershed (MI) was initiated in 2012 to increase fish and wildlife habitat by hydrologically reconnecting former agricultural land to its adjacent stream. However, to reduce the risk of sediment phosphorus (P) releasing into downstream waters upon hydrologic reconnection, the wetland was drained and P-rich sediments were dredged. I evaluated restoration success by measuring sediment P release in the wetland after dredging and compared those results to studies that measured P release before dredging. Sediment cores were incubated under two water temperatures (ambient; +2° C) and two oxygen levels (oxic; hypoxic). My results showed that maximum P release rates in all four after-dredging treatments were reduced by 95-99 %. These results suggest that 1) sediment removal is an effective approach for reducing sediment P release in wetlands restored on agricultural land, and 2) the remaining substrate is not currently susceptible to increased P release as a result of increased temperature or decreased dissolved oxygen. While internal P loading was drastically reduced, P adsorption isotherm experiments suggested that the deep dredging depth (~1 m on average) exposed sediments with significantly reduced binding capacities, resulting in the wetland acting as a phosphate sink only when water column soluble reactive phosphorus concentrations are over 40 µg L^{-1}. Ultimately, this study shows that sediment dredging can be a useful technique to balance the goals of habitat restoration and water quality improvements in wetlands restored on former agricultural lands.

Introduction

The structure and function of riparian wetlands is largely driven by hydrologic connectivity (Junk et al. 1989), which helps to generate ecosystem services such as biodiversity support (Zedler 2003), flood mitigation (Blann et al. 2009), nutrient filtration (Lowrance et al. 1984; Fennessy & Cronk 1997; Reddy et al. 1999), and carbon sequestration (Kayranli et al. 2003). Despite these benefits, over 50 % of wetland area has been lost in the USA since European settlement (USDA 2012; Gibbs 2000), with many losses resulting from wetland drainage for the creation of agricultural land (Dahl 2000). However, the growing recognition of wetland ecosystem services has resulted in many wetland restoration projects around the world (Mitsch et al. 2005). Specifically, global reductions in biodiversity (Barnosky et al. 2001) have motivated many wetland restoration projects to be focused on the creation of fish and wildlife habitat (Hansson et al. 2005), as wetlands can play a key role in supporting biodiversity of entire regions through provisions of food and shelter for both terrestrial and aquatic species (Findlay & Houlahan 1997; Keddy 2000).

Hydrologic reconnection is often needed to achieve these restoration goals; however, in some highly impacted, human-modified systems, restoration activities that increase the hydrologic connectivity of aquatic systems can result in undesirable ecological effects by increasing the transport of sediment, nutrients, toxins, or invasive species to downstream waters (Pringle 2003; Jackson & Pringle 2010). For example, when wetland restoration includes hydrologic reconnection of former agricultural land to an adjacent waterbody, there can be a threat of nutrient movement from the sediments to downstream waters because of mobilization of legacy phosphorus (P) accumulated from agricultural fertilization (Pant & Reddy 2003; Aldous et al. 2005; Lindenberg & Wood, 2006; Smit & Steinman 2015). This scenario creates a dilemma where the restoration of fish and wildlife habitat is done at the expense of downstream water quality. Given that there is ongoing eutrophication of freshwater resources worldwide (Smith 2003), it is imperative to avoid such risks in wetland restoration.

Sediment dredging is an engineering practice that removes sediments from aquatic systems and has been used as a method for attempting to reduce internal P loading in shallow eutrophic lakes around the world (Van der Does et al. 1992; Fan et al. 2004; Björk et al. 2010; Liu et al. 2016). Experiments simulating sediment dredging in the lab have found that dredging should be an effective technique for reducing internal P loading by reducing labile sediment P (Yu et al. 2017; Chen et al. 2018), reducing sediment organic matter (Zhong 2008), and increasing the P binding capacity of the sediments by aerating the top sediment layer (Zhong 2008; Yu et al. 2017; Chen et al 2018). Despite encouraging results in the lab, some field experiments have reported a rise in the sediment P flux after dredging (Fan et al. 2004; Liu et al. 2016), while others have shown no obvious changes (Lohrer & Wetz 2003). The disconnect between success in the lab and failure of long-term success in the field has been linked to the continuation of high external P loads from surface waters after dredging, which contributes labile P in the sediments (Keelberg & Kohl 1999; Reddy et al. 2007; Jing et al. 2015; Liu et al. 2016).

Additionally, climate warming could also affect the ability of sediment dredging to reduce sediment P release, but it has been a less-investigated phenomenon. Given that water temperature and DO are closely linked to the ambient air temperature (Michalak 2016), climate warming may create more favorable conditions for iron-reducing bacteria (Lovley & Phillips 1988; Price & Sowers 2004) and subsequent mobilization of iron-bound P (Boström et al. 1988) by enhancing water column stratification and increasing water temperature (North et al. 2014).

Therefore, climate warming may stimulate the release of iron-bound P into the water column, even after sediment dredging.

The main objective of this study was to determine if sediment dredging was successful at reducing sediment P release in a recently dredged and hydrologically reconnected wetland that was formerly agricultural land. I assessed this objective by conducting two experiments. First, I compared results of P adsorption isotherm experiments conducted before dredging (Steinman & Ogdahl 2016) and after dredging (current study) to determine if the wetland was acting as a potential source or sink of dissolved P. Second, I conducted sediment core incubation experiments and determined if sediment P release was reduced by comparing release rates before dredging (Smit & Steinman 2015) and after dredging (current study). Additionally, to determine if climate warming may affect dredging results in the short-term, I compared after-dredging sediment P release rates among four treatments, which involved manipulations of water temperature (ambient; +2 °C) and DO concentration (oxic; hypoxic).

I hypothesized that the P adsorption isotherm experiments would suggest that the wetland was acting as a sink for dissolved P after sediment dredging because before-dredging experiments suggested that the deeper sediments (now exposed after dredging) had the potential to sorb nearly double the amount of P that they already contained (Steinman & Ogdahl 2016). I also hypothesized that sediment core incubation experiments would show significantly lower sediment P release rates in all after-dredging treatments compared to the before-dredging treatments, as sediment dredging can reduce the pool of labile P in the sediments (Yu et al. 2017; Chen et al. 2018). However, I hypothesized that climate warming would stimulate P release in the after-dredging cores, albeit to a lower degree than before dredging, resulting in treatments with increased water temperature and/or low DO concentration having higher sediment P release rates than treatments with ambient water temperature and/or high DO concentration, due to iron-based redox reactions likely driving P dynamics in this wetland.

Methods

Study Area

The study area is within the Muskegon Lake (MI) Great Lakes Area of Concern (AOC), which was so designated in 1985. One of the major beneficial use impairments still in need of remediation is the loss of fish and wildlife habitat due to industrialization along the shoreline in the 1900s (Steinman et al. 2008). The AOC geographic boundary also includes Bear Lake, which is connected to Muskegon Lake through a navigation channel. In an effort to increase fish and wildlife habitat within the AOC, a project was initiated in 2012 to restore wetland habitat just upstream of Bear Lake.

The restoration site consists of two historic wetlands that were hydrologically separated from their adjacent stream by earthen berms in the 1900s (Fig. 2.1). Between the 1930s and the late 1990s/early 2000s, these disconnected wetlands were pumped dry and used for agricultural celery production that involved P fertilization. After farming ended, the pumps were turned off and this area refilled with water to form two ponds. The ponds were separated from each other by a road and have been referred to as the east pond and the west pond. In the late 2000s, the east pond was partially dredged for topsoil, but the west pond was never dredged. Therefore, this study focuses only on the restoration of the west pond to avoid the potentially confounding effects of the partial dredging in the east pond.

Initial studies indicated that the west pond had very high total phosphorus (TP) concentrations both in the water column (~1000-2000 µg TP L^{-1}) and the sediment (~2500-4000 mg TP kg^{-1}), so reconnection risked the net movement of P from the restored wetland to downstream waters (Smit & Steinman 2015; Steinman & Ogdahl 2016). Introducing additional P to downstream waters would be problematic because ~200 m downstream of the wetland is a eutrophic waterbody, Bear Lake, which is a P limited system (Xie et al. 2011) under a federal mandate to reduce its P levels (MDEQ 2008). Therefore, due to the high risk of this hydrologic reconnection project releasing P to downstream waters, management action was needed before the hydrologic reconnection could occur. Sediment dredging was determined to be the best approach to prevent P release to downstream waters. In the spring of 2016, both ponds were drained by pumping the overlying water to a water treatment facility. The P-rich sediments were dredged to an average depth of 1 m with a backhoe excavator in the summer and fall of 2016. The pond was reflooded with Bear Creek water in the winter of 2016 via a pipe. Berm removal and hydrologic reconnection to the adjacent creek was completed in the spring of 2017.

Experimental Design

Hydrologic reconnection and dredging treatments

Before-dredging sediment core incubation experiments were conducted in the summer and fall of 2013 by Smit and Steinman (2015). After-dredging sediment core incubation experiments were conducted in the summer and fall of 2017 (this study). In the before-dredging sediment core incubation experiments the ambient water column was carefully removed to within 5 cm of the surface sediments using a siphon apparatus and then reflooded with Bear Creek water to represent how hydrologic reconnection would affect sediment P release if dredging did not occur (Smit & Steinman 2015). In after-dredging sediment core incubation experiments the overlying water column was left intact because hydrologic reconnection had already occurred.

Temperature treatment

To simulate the effects of climate warming, before-dredging cores were incubated at either the average ambient bottom water temperature at the time of sediment core collection or at 2 °C higher than the measured average ambient bottom water temperature. A plus 2 °C increase was used for the experimental warming as annual air temperatures in the Great Lakes region are predicted to increase by 1.4±0.6 °C in the near future and from 1 °C to 5 ±1.2 °C by the end of the century depending on different emissions scenarios (Hayhoe et al. 2010). It is assumed that these projected temperature increases would also occur in the water column of small, shallow water bodies, such as this restored wetland (Kadlec & Reddy 2001). In an effort to investigate the effect of warming on sediment P release rates after dredging, but not confound the effect of dredging, the average ambient bottom water temperature at the time of after-dredging core sampling was adjusted by a maximum of ±3 °C so that the before- and after-dredging incubation temperatures were either the same (summer experiments) or within 1 °C of each other (fall experiments) (Table 2.1).

Dissolved oxygen treatment

In before-dredging experiments water column DO concentrations remained well oxygenated in the all sediment cores because the wetland was relatively shallow (average depth ~1 m) before dredging, prompting the assumption that the overlying water column would remain relatively well mixed and oxygenated (Reddy and DeLaune 2004). After dredging, the average depth of the wetland increased to 2 m, which increased the likelihood that thermal stratification and associated bottom water hypoxia could occur; therefore, after-dredging experiments included hypoxic core treatments in addition to oxic core treatments.

In summary, the before-dredging experimental design was a 1×2 factorial, involving one level of DO treatment (oxic) and two levels of the temperature treatment (ambient and +2 °C). In each of the before-dredging experiments (summer and fall) the two treatments were measured once per site, at six sites, for a total of 12 sediment cores per experiment. The after-dredging experimental design was a 2×2 factorial, involving two levels of the DO treatment (oxic and hypoxic) and two levels of the temperature treatment (ambient and +2 °C). In the after-dredging experiments (summer and fall) the four treatments were measured twice per site, at six sites, for a total of 48 sediment cores per experiment.

Site selection

Coring sites for the before-dredging incubation experiments were determined with a stratified random selection process where the wetland was divided into six sections of equal area and one location was randomly chosen within each section (Fig. 2.1; Smit & Steinman 2015). Coring sites for the after-dredging incubation experiments were in slightly different locations than before-dredging sites (Fig. 2.1). This discrepancy occurred because I adjusted the location of before-dredging sediment core incubations sites 0-50 m to align them with the nearest P adsorption isotherm site. The effect of dredging likely overwhelms the slight spatial variation that occurred between the before- and after-dredging sediment core incubations sites.

Sediment cores used for the P adsorption isotherm experiments were collected at the same sites both before and after dredging (Fig. 2.1). These sites were initially selected with a stratified random selection process where the wetland was divided into five sections of equal area and one location was randomly selected within each area (Steinman & Ogdahl 2016).

Field Sampling and Procedure

Sediment sampling

Sediment cores used in the before-dredging incubation experiments were collected in the summer and fall of 2013, whereas sediment cores used in the after-dredging incubation experiments were collected in the summer and fall of 2017. All cores were obtained using a modified piston coring apparatus (Fisher et al. 1992; Steinman et al. 2004). The modified piston corer was constructed of a 0.6-m long, 7-cm inner diameter, 7.6-cm outer diameter polycarbonate tube that was marked in 1-cm increments. A polyvinyl chloride assembly coupled with a 3.81-cm in-line sump pump check valve was used to drive cores into the sediment, and provide suction within the tube when the core was retrieved. The modified corer was positioned vertically at the sediment-water interface, and core tubes were carefully driven into the sediment to minimize disruption of the sediment surface to a depth of at least 15 cm (before-dredging) or 10 cm (after-dredging). A 15-cm depth could not be achieved in the after-dredging sediment cores due to sediment compaction from the heavy machinery used in the dredging operations. After cores were sampled, the bottoms of the core tubes were sealed with a rubber stopper and duct tape, and the tops sealed with plastic caps. The resulting sediment cores consisted of at least 10-15 cm of sediment and an overlying water column of approximately 45 cm. Sediment cores were then stored in an upright position for transport back to the lab. An additional 5-cm deep core was sampled at each site for the analysis of sediment metals. After collection, the 5-cm core was extruded from the core tube in the field and stored in a plastic bag.

Sediment cores used for before-dredging P adsorption isotherm experiments were collected from 5 sites in the summer of 2012 (Steinman & Ogdahl 2016). Cores for after-dredging P adsorption isotherm experiments were collected from the same 5 sites in the summer of 2017 while collecting sediment cores for the incubation experiments. A 15-cm (before-dredging) or 10-cm (after-dredging) sediment core was obtained for the analysis of P adsorption isotherms, sediment TP, and sediment organic matter (OM) using the modified coring apparatus. Cores for these analyses were extruded from the core tube in the field and stored in a plastic bag. All intact sediment cores and extruded sediment samples were stored on ice after collection and transported to the lab within 6 hours.

Water

Prior to collecting sediment cores at each incubation site, water column readings of temperature, DO, pH, and specific conductance were collected at the water surface and near bottom with a YSI 6600 sonde (YSI Incorporated, Yellow Springs, OH). For before-dredging incubation experiments, water used to simulate hydrologic reconnection and refill sediment cores after sampling events was collected from one location in Bear Creek, and stored in acid-washed carboys. In after-dredging experiments, water used to refill sediment cores after sampling events was collected in one location in the center of the wetland with acid-washed carboys. Additionally, at each P adsorption isotherm site, grab samples for (soluble reactive phosphorus) SRP were taken just below the water surface. All water samples were placed on ice after collection and stored this way until transported to the lab within 6 hours.

Laboratory Procedure

Incubations

In the lab, sediment cores were adjusted to a sediment depth of 15 cm (for the before-dredging cores) or 10 cm (for the after-dredging cores) by removing excess sediment from the bottom of the core. In order to simulate hydrologic reconnection in the before-dredging cores, the overlying water column was carefully removed and reflooded to a depth of 25 cm with filtered

Bear Creek water using the methods outlined in Smit and Steinman (2015). In the after-dredging cores, the ambient overlying water also was adjusted to a depth of 25 cm using a peristaltic pump.

In all experiments, sediment cores were placed in one of two dark environmental growth chambers (Powers Scientific Inc, Pipersvill, PA) depending on the specific temperature treatment for that core. Temperature accuracy within the environmental growth chambers was checked daily using an additional thermometer. In before-dredging experiments, cores were maintained under oxic conditions by gently bubbling air into the cores using aquarium pumps that maintained water column oxygen concentrations at 75–100 % equilibrium with atmospheric oxygen throughout the incubation period. In the after-dredging experiments, cores were gently bubbled with either air (oxic core treatments) or with gas from a cylinder containing 95 % N_2 with 5 % CO_2 to buffer pH (hypoxic treatments). Regardless of treatment, gases were bubbled into core tubes at a uniform rate through tubing placed above the sediment surface; caution was taken to ensure bubbling did not disturb the sediment surface. DO concentrations in the water columns of each core were measured at the end of all experiments using a YSI Pro-DO sonde placed near the water-sediment interface.

Despite our methods, equipment, and bubbling rate being consistent with prior studies (Steinman et al. 2004), I was not able to achieve DO concentrations below 2 mg L^{-1} in the hypoxic core treatments by the end of either after-dredging experiment. At the end of both after-dredging experiments, DO concentrations in the water column of hypoxic cores averaged 3.5 ± 0.4 mg L^{-1}. To investigate if the higher than expected DO levels were a result of oxygen contamination within the gas cylinder, I bubbled the gas mixture into three core tubes containing only deionized water (no sediment). After 25 days, the average DO concentration in the water column was 4.03 ± 0.07 mg L^{-1}, suggesting that oxygen contamination in the gas cylinder caused the elevated DO levels in the hypoxic core treatments.

Sediment cores were incubated for either 24 or 25 days and water samples for TP and SRP were collected at various time intervals (Table 2.1). Water samples were collected from the middle of the water column by inserting a syringe into sampling ports that went through the top plastic cap on each core tube. A 60- or 20-mL unfiltered sample for TP analysis was collected for the before- and after-dredging experiments, respectively. Additionally, a 20-ml 0.45-μm filtered sample (ThermoFisher Nylon Syringe Filter, ThermoFisher Scientific, Waltham, MA) was collected for SRP analysis in all experiments. Filters used for SRP analysis in all experiments were acid rinsed prior to use. SRP samples were frozen immediately after collection, and TP samples were stored at 4 °C. Water collected from the creek (before-dredging) and wetland (after-dredging) was filtered through a 0.2-μm filter (Graver Technologies, Glasgow, DE) using a peristaltic pump then stored in acid-washed carboys at 4 °C. This water was used to refill core tubes after each water sampling event so a constant water column depth was maintained throughout the incubation period.

TP and SRP water samples were analyzed on a Bran + Luebbe Autoanalyzer (SEAL Analytical, Mequon, Wisconsin; APHA 1998, TP/SRP detection limit = 5 μg L^{-1}). Concentrations below the detection limit were assigned a value of one-half the detection limit (Smith 1991). The additional 5-cm deep sediment samples collected from each sampling location were analyzed for Ca, Fe, Mg, and Al according to EPA method 6010B using inductively coupled plasma-atomic emission spectrometry (ICP-AES) (U.S. EPA 1994).

P Sorption Isotherms

In both the before- and after-dredging experiments the field extruded cores were brought back to the lab and analyzed for TP and sediment organic matter (OM) and used for P sorption isotherm measurements. P sorption isotherms were determined in triplicate for each sediment core, according to a procedure modified from Mozaffari and Sims (1994) and Novak et al. (2004). The collected sediment was homogenized and 3 g of sediment was placed into a 50-ml centrifuge tube with 20 ml of inorganic P solutions (KH_2PO_4 dissolved in 0.01 M KCl) containing 0, 0.01, 0.1, 1, 5, 10, 50, 100, and 500 mg P L^{-1} for before-dredging experiments (Steinman & Ogdahl 2016) and 0, 0.01, 0.025, 0.05, 0.1, 0.25, 0.5, 1, and 5 mg P L^{-1} for after-dredging experiments. The centrifuge tubes containing the sediment and P solution were then shaken on an orbital shaker table for 24 hours at 250 RPM. After this, the tubes were centrifuged for 20 minutes at 2500 RMP and the supernatant was removed, filtered (0.45 µm), and analyzed for SRP using the same method described previously. Final SRP concentrations in the supernatant were used to calculate the equilibrium phosphorus concentration (EPC_0) and phosphorus sorption maximum (S_{max}) using equations in Pant et al. (2001).

The equilibrium P concentration (EPC_0) of the sediment represents the aqueous P concentration at which no net sorption or desorption occurs between water and sediment. Therefore, by comparing EPC_0 values with SRP concentrations in the water column at the time of sampling, it can be determined if the sediments act as a potential P source (SRP< EPC_0), P sink (SRP>EPC_0), or are at relative equilibrium (SRP≈EPC_0). The phosphorus sorption maximum (S_{max}) is an estimate of the sediment's capacity to adsorb P and is useful if compared with sediment TP concentrations at the time of sampling. If sediment TP is less than the S_{max}, the sediments have potential to adsorb additional phosphorus. Alternatively, if sediment TP is greater than the S_{max}, the sediments are potentially saturated with P and may have little binding capacity left; however, this comparison must be interpreted with caution as a high content of organic P in the sediments may cause the sediments to appear to be fully P saturated when they are not.

OM content was determined by placing 20 g of homogenized sediment in a pre-ashed crucible (550 °C for 1 hr). Sediment in the crucible was then dried for 48 hr at 105 °C and re-weighed before being ashed at 550 °C for 1 hour. Percent OM content was measured as the mass loss due to combustion. A subsample of the ashed material was used for analysis of sediment TP using the same method previously described for the water samples.

Analysis

To investigate the effect of sediment dredging on sediment P flux I compared maximum P release rates measured before dredging (Smit & Steinman 2015) and after dredging (current study). Additionally, to investigate if simulated climate warming had an effect on sediment P release after dredging, I compared both *maximum* and *average* P release rates among the four after-dredging treatments. Maximum P release rate calculations were based on the methodology used in Steinman et al. (2004). In brief, the maximum P release rates of TP and SRP were determined using the following equation:

$$P_{flux} = (C_t - C_0)*V/A$$

where, P_{flux} is the rate of TP or SRP release in mg m^{-2} d^{-1}; C_t is the concentration of TP or SRP at time t; C_0 is the TP or SRP concentration of the water at time 0; V is the volume of the overlying water column; and A is the planar surface area of the sediment. Flux calculations were based on the linear portions of the water column nutrient concentration curves measured through time in order to capture the maximum apparent release rate; however, C_0 and C_t could not be consecutive dates in order to avoid potential short-term bias. The average P release rates in the after-dredged cores were determined by using the methodology in Pant and Reddy (2003), which used the following equation:

$P_{flux} = slope*V/A$

where, P_{flux} is the rate of TP or SRP release in mg m^{-2} d^{-1}; *slope* is the linear best fit line of the TP or SRP water column concentrations throughout the entire incubation period (a linear relationship was used because it captured the general trend of a core's TP or SRP concentrations throughout the incubation period and explained 63 % of the variability on average); *V* is the volume of the overlying water column; and *A* is the planar surface area of the sediment.

Separate blocked two-way ANOVAs were used to determine if the maximum P release rates in oxic and hypoxic conditions were affected by sediment dredging or incubation temperature, with dredging (before or after) and temperature (ambient or +2ºC) as main effects, and sampling site as a blocking factor. Additionally, separate blocked two-way ANOVAs were used to determine if climate warming affected maximum or average release rates in the dredged wetland with DO (oxic or hypoxic) and temperature (ambient or +2ºC) as main effects, and sampling site as a blocking factor. In all ANOVAs, the two seasons were treated and analyzed separately. Duplicate cores in the after-dredging experiments were treated as independent samples. When necessary, data were transformed (ln, square root, power) to meet the assumptions of ANOVA. Normality was tested using the Shapiro-Wilk goodness of fit test, and equality of variance was tested using a Levene's test. Results from P adsorption isotherm experiments, sediment analysis, and water column characteristics were compared before and after dredging using a paired t-test, with sampling site as the pairing term. All statistical analyses were conducted using R software (Rstudio Team 2015; R Core Team 2016).

In the fall, the *average* TP and SRP release rates from after-dredging cores collected at site 3 and subjected to the hypoxic/+2 °C treatment were removed before analysis because TP release rates were on average 185× greater and SRP release rates 152× greater than the release rates at all other sites. While this created an unbalanced design in the analysis of the fall average release rates, it was necessary to remove these cores to achieve the normality assumption for ANOVA analysis. The high release rates from these cores were likely due to effects of bioturbation (Phillips et al. 1994), as turbidity measurements associated with these cores were approximately 50× greater than others , and their surface sediments had numerous invertebrate burrows.

Results

Water Quality and Sediment Metals

Dredging had variable impacts on water quality (Table 2.2). In both seasons, mean water column DO was significantly higher after dredging, whereas specific conductance was significantly lower after dredging. There was no significant change in pH after dredging in either season. Temperature was significantly lower after dredging in the summer samples, but significantly higher after dredging in the fall samples. Dredging had no significant effect on the mean values of sediment Al, Ca, Fe, and Mg in either season (Table 2.2).

P Sorption Isotherms

The mean EPC_0 and water column SRP concentrations were significantly reduced after dredging (Table 2.3). Prior to dredging, the mean SRP concentration in the overlying water column was ~5× greater than the mean EPC_0 value; however, after dredging and hydrologic reconnection, water column SRP concentrations were lower than EPC_0 concentrations at all sites. This result suggests that the dredged sediments now may serve as a potential source of SRP to the overlying water. Despite this result, the significant reduction of sediment TP after dredging suggests that there is not a large pool of P available for release into the water column (Table 2.3). The phosphorus sorption maximum (S_{max}) was also significantly reduced after dredging, possibly due in part to the large reduction in the soil organic matter content (Table 2.3; Syers et al. 1973; Reddy et al. 1995). The significant reduction in S_{max} after dredging resulted in sediment TP values being either near or over S_{max} concentrations at all sites, suggesting that these sediments may not have the capacity to adsorb additional P under current conditions.

P Flux

Maximum release rates- effect of dredging

Oxic and hypoxic maximum TP and SRP release rates were significantly reduced after dredging in both seasons (Fig. 2.2; Table 2.4, 2.5). Mean maximum TP release rates in the summer ranged from ~45-85 mg m^{-2} d^{-1} before dredging and 0-1 mg m^{-2} d^{-1} after dredging; similarly, mean maximum TP release rates in the fall ranged from ~40-60 mg m^{-2} d^{-1} before dredging and ~1-7 mg m^{-2} d^{-1} after dredging. This corresponded to a 99 % reduction in the mean maximum TP release rate in the summer and 95 % reduction in the fall after dredging. Mean maximum SRP release rates showed similar trends. Summer SRP rates ranged from ~36-60 mg m^{-2} d^{-1} before dredging and ~0.5-1 after dredging and fall mean maximum SRP rates ranged from ~33-36 mg m^{-2} d^{-1} before dredging and ~0-2 mg m^{-2} d^{-1} after dredging. These reductions corresponded to a 98 % reduction in both seasons.

The reductions in maximum TP and SRP release rates after dredging are reflected by the low TP and SRP concentrations measured in the water columns of all after-dredging treatments throughout the incubation period, compared to the dramatic increase in water column P concentrations in the before-dredging cores (Fig. 2.2). For example, TP concentrations in before-dredging cores of the summer experiment started at initially low concentrations (near 10 μg P L^{-1}), and then increased to maximum concentrations between ~1600–2300 μg P L^{-1} (Smit & Steinman 2015; Fig. 2.2). In contrast, TP concentrations in the after-dredging cores of the summer started slightly higher, between ~40-60 μg P L^{-1}, but concentrations stayed relatively stable and even decreased throughout the incubation period, regardless of the water column's DO concentration or temperature (Fig. 2.2). Similar trends also were observed with SRP concentrations (Fig. 2.2).

Temperature had no statistically significant effect on maximum TP or SRP release rate in either the before- or after-dredging cores, regardless of season or DO concentration (Table 2.4); however, higher maximum TP release rates were observed in after-dredging hypoxic cores incubated at a +2 °C temperature than at ambient temperature (Table 2.5). For SRP, this trend was observed only in the fall after-dredging hypoxic cores (Table 2.5). There was a significant but weak interaction between dredging and temperature in the maximum TP and SRP release rates of the oxic cores in the summer (Table 2.4). These interaction effects can be attributed to the before-dredged cores having higher release rates in warmed temperatures (though not significant), while the after-dredging cores did not (Table 2.5). Additionally, there was an interaction between dredging and temperature in summer TP release rates of the hypoxic cores because the before-dredging cores had higher maximum TP release rates in the warmed cores (though not significant), but the after-dredging cores did not show this trend as strongly (Table 2.4; Table 2.5).

Maximum release rates- effect of DO concentration and temperature after dredging

Maximum TP and SRP release rates within after-dredging treatments were relatively similar in both seasons (Table 2.5), as they were not affected by DO concentration ($p > 0.10$ in all ANOVAs) or temperature ($p > 0.10$ in all ANOVAs) in either season. The notably higher mean maximum TP and SRP release rates in the fall hypoxic/+2 ºC treatment can be attributed to high release rates from site 3, where maximum TP release rates were ~15-35 mg m^{-2} d^{-1} compared to < 1 mg m^{-2} d^{-1} at all other sites. Similarly, the maximum SRP release rates from site 3 were ~10-30 mg m^{-2} d^{-1}, whereas all other sites were < 0 mg m^{-2} d^{-1}.

Average Release Rates- effect of DO concentration and temperature after dredging

While there was no significant effect of DO concentration on the maximum release rates, there was an effect of DO concentration on the average release rates in the dredged cores (Table 2.6), with oxic cores unexpectedly having higher average TP and SRP release rates than the hypoxic cores in both the summer and fall (Fig. 2.3). This trend is reflected by the consistently higher average water column TP and SRP concentrations in the water columns of the oxic cores, regardless of incubation temperature (Fig. 2.2.). Similarly to the maximum P release rates, the average TP and SRP release rates were relatively unaffected by the +2 °C temperature increase, except in the fall when the average TP release rate was significantly higher in the oxic cores (Table 2.6). There was no significant interaction between temperature and DO concentration in the average release rates of TP or SRP in either season.

Discussion

While many studies have documented the use of sediment dredging to reduce internal P loading in shallow lakes (Van der Does et al. 1992; Fan et al. 2004; Björk et al. 2010; Liu et al. 2016), relatively little is known about sediment dredging, conducted prior to hydrologic reconnection, as a technique for reducing sediment P release in wetlands restored on agricultural land. Results from the sediment core incubation experiments in this study clearly indicated that sediment dredging done before hydrologic reconnection can significantly reduce sediment P release of a wetland restored on former agricultural land, regardless of season (but see below). Additionally, after-dredging results from this study indicated that in the wetland's current state, maximum P release rates may not be significantly affected by decreased DO levels or a 2 °C mean increase in water temperature (Table 2.4; Table 2.5), both of which are expected to occur in shallow waterbodies as the climate warms (Hayhoe et al. 2010; Michalak 2016).

While seasonality is recognized as a major influence on ecosystem functioning in wetlands (Kadlec & Reddy 2001) and also known to affect the ability of dredging to reduce sediment P release (Chen et al. 2018), I did not observe any strong seasonal variation in maximum or average P release rates (Table 2.5). This may be because the incubation temperatures both within and between summer and fall experiments were not dramatically different (Table 2.1). The low temperature variation may also explain why I was unable to detect a significant effect of temperature.

The 95-99 % reductions in P release rates observed in this study after dredging were similar to other sediment core incubation findings that simulated dredging. For example, simulated dredging from sediment cores collected in Lake Trehörningen (Sweden) resulted in a ~50 % reduction in P release, regardless of DO concentration (Ryding 1982). Peters & van Liere (1985) simulated dredging in two shallow eutrophic Dutch lakes by suctioning the uppermost 10 – 23 cm of sediment and P release rates were reduced by 90 %. Additionally, simulated dredging of 10 cm of sediment reduced hypoxic P release rates between 70 and 99 % in a shallow eutrophic lake in Germany (Kleeberg & Kohl 1999).

While the controlled conditions of these lab experiments do not take into account the variability of environmental conditions in the field, this study was unique compared to most other sediment core incubation studies investigating the use of sediment dredging as a method to reduce sediment P release. In particular, dredging was completed in-situ for this study, rather than simulated in the lab, providing a more representative view of the sediment P release rates in the wetland, given that both dredging depth (Kleeberg & Kohl 1999; Fan et al. 2004; Reddy et al. 2007; Liu et al. 2015) and dredging method (Fan et al. 2004) play a considerable role in the success of dredging.

It is likely that the relatively deep dredging depth (~1 m on average) at the study site played a key role in reducing sediment P release by removing a considerable amount of sediment organic matter and TP (Table 2.3). While sediment TP can be a poor predictor of P release due to the inability to discriminate between mobile and non-mobile fractions (Jensen et al. 1992), the drastic reduction in sediment TP after dredging likely reduced all P fractions as pore water SRP was ~2-5 mg L^{-1} before dredging (Steinman & Ogdahl 2016) and < 0.2 mg L^{-1} after dredging (Hassett & Steinman 2018). Additionally, I speculate that microbial respiration may have been reduced after dredging either due to lack of organic matter (Sinsabaugh et al. 2008) or reduced microbial abundance (Moore et al. 2017). This could explain, in part, why the temperature and DO treatments had little effect on the maximum P release rates, as the primary mechanism by which both of these treatments would increase P release is by stimulating microbial respiration (Boström et al. 1988).

Interestingly, while a significant effect of DO concentration was not detected in the *maximum* P release rates of the after-dredging treatments, a significant effect was found in the *average* P release rates, with oxic cores having significantly higher average P release rates than hypoxic cores (Table 2.6; 2.3). This result is contrary to what I hypothesized and contradicts a long-standing paradigm in limnology that suggests sediments with oxidized surfaces release less phosphate than sediments with chemically reduced surfaces because iron bound-P remains stable under oxidized conditions but increases in solubility under hypoxic conditions (Mortimer 1941). Our opposing results to the Mortimer model could be attributed to the sediment dredging changing the chemical, physical, and biological composition of the sediments in two ways.

First, geologically older sediments with more structured crystalline forms of Fe and Al minerals were likely revealed by the deep depth of sediment dredging (Lijklema 1980). The crystalline structure of Fe minerals is significant to sediment P dynamics because highly crystalline Fe(III) oxyhydroxides have less surface area for P sorption than poorly crystalline Fe(II) hydroxides (McLaughlin et al. 1981). Given that hypoxic conditions stimulate the production of poorly crystalline Fe(II), it is possible that the hypoxic treatments had more Fe minerals with poorly crystalline structures, which contributed to higher sediment P binding capacities in hypoxic conditions rather than oxic conditions (Davidson 1992). The sediment characteristics also support this speculation of Fe mineral structure playing a larger role in sediment P dynamics than redox state in the dredged wetland because there was no significant change in the mass of any sediment metals after dredging (Table 2.2) but the sediment's P binding capacity was greatly reduced at all sites (Table 2.3).

Second, it is possible that these release rate results were due in part to decreased microbial activity after dredging. In agricultural ditches, dredging can reduce microbial activity by nearly 70 % and microbial activity may take up to 1 year to recover to before-dredging levels (Moore et al. 2017). Due to the sediments at our study sites being exposed to air throughout the dredging process, aerobic bacteria may have recovered faster than anaerobic bacteria and continued to be more abundant after hydrologic reconnection, as microbes in dry soil can readily adapt to flooded conditions (Lundquist et al. 1999; Drenovsky et al. 2004). Therefore, there may have been more P mineralization due to higher microbial abundance in the oxic cores than in the hypoxic cores. Furthermore, slower recolonization of anaerobic bacteria could have contributed to lower average release rates in the hypoxic cores by keeping iron in an oxidized state (Mortimer 1941). It is possible that as the sediments of this recently dredged wetland accumulate organic matter and develop more abundant microbial communities, the classic Mortimer model describing sediment P dynamics will apply again. Until then, these unexpected results provide another example to support the growing evidence that wetland restoration does not fully restore biogeochemical functions, at least in the short-term (Peralta et al. 2010; Moreno-Mateos et al. 2012).

EPC_0 results were consistent with results of the sediment core incubation experiments and indicated that in oxic conditions, there will be a net release in this system when the overlying water column has SRP concentrations < 40 µg L^{-1}, on average. Seven monthly post-restoration water quality monitoring events found that SRP concentrations in the wetland averaged 5±2 µg L^{-1} with incoming Bear Creek water also averaging 5±3 µg L^{-1} (Hassett & Steinman 2018), suggesting that net phosphate release is likely to remain, albeit at low levels, until sediment P concentrations or binding capacity increases from sediment deposition and organic matter accumulation (Reddy et al. 1999). Additionally, even if overlying SRP concentrations were to increase above the EPC_0 in the near future and create a downward concentration gradient, the sediment currently has little P binding capacity (Table 2.4).

Prior studies have emphasized the need to reduce external P loading in order for dredging to successfully reduce internal P loading (Keelberg & Kohl 1999; Reddy et al. 2007; Jing et al. 2015; Liu et al. 2016). This is not currently a concern for this wetland as the incoming Bear Creek water has relatively low SRP concentrations under baseflow conditions (Hassett & Steinman 2018). However, I did observe that effects of bioturbation could become more significant under increased temperatures and hypoxic conditions, and P concentrations are usually higher under stormflow conditions, both of which may increase sediment P release in the restored wetland (Phillips et al. 1994).

While sediment dredging can be expensive and comes with many logistical challenges, I found it to be successful at reducing sediment P release from a restored wetland that was formerly P-rich agricultural land. Additionally, I found that dredging will likely continue to be successful into the near future even with the increased water temperatures and hypoxic conditions that may occur due to climate change. This research helps contribute to wetland restoration and water quality management decisions; the results indicate that sediment dredging prior to hydrologic reconnection is an effective method for converting agricultural land back into wetland for increased fish and wildlife habitat without causing harm to downstream water quality.

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Tables

Table 2.1 Experimental parameters for the summer and fall sediment core incubation experiments conducted both before dredging (2013; Smit & Steinman 2015) and after dredging (2017) in the west Bear Lake wetland. Bottom water temp refers to the average bottom water temperature measured at each site at the time of coring. Ambient incubation temp refers to the incubation temperature used for the ambient temperature treatment. +2 incubation temp refers to the incubation temperature used for the ambient+2 °C temperature treatment.

Parameter	Before dredging		After dredging	
Season	Summer	Fall	Summer	Fall
Coring date	July 1, 2013	Oct. 3, 2013	June 26, 2017	Sept. 25, 2017
Bottom water temp (°C)	23	17	20	21
Ambient incubation temp (°C)	23	17	23	18
+2 incubation temp (°C)	25	19	25	20
Sampling days	hr 0; 24; day 5, 10, 15, 20, 25	hr 0; day 3, 6, 12, 18, 24	hr 0, 12, 24; day 5, 10, 15, 20, 25	hr 0, 12, 24; day 5, 10, 15, 20, 25
Treatments	Oxic ambient; Oxic +2	Oxic ambient; Oxic +2	Oxic ambient; Oxic +2; Hypoxic ambient; Hypoxic +2	Oxic ambient; Oxic +2; Hypoxic ambient; Hypoxic +2

Table 2.2 Summary of mean (±1 SD, n=6) YSI readings of surface water column dissolved oxygen (DO), specific conductance (SpCond), pH, and temperature as well as mean (±1 SD, n=6) sediment Al, Ca, Fe, and Mg measured in the west Bear Lake wetland in the summer and fall both before dredging (2013; Smit & Steinman 2015) and after dredging (2017).

		Before dredging	After dredging	
Season	Parameter	mean±SD	mean±SD	*p*-value
Summer				
Water Column	DO (mg L^{-1})	3.34±1.64	8.63±0.61	**0.001**
	SpCond (μS cm^{-1})	638±17	376±0.89	<**0.001**
	pH	8.3±0.66	7.9±0.22	0.297
	Temperature (°C)	23.40±0.80	20.46±1.11	**0.008**
Sediment	Al (mg kg^{-1})	635±316	723±262	0.558
	Ca (mg kg^{-1})	2250±535	16755±29801	0.328
	Fe (mg kg^{-1})	1232±428	1445±1486	0.770
	Mg (mg kg^{-1})	252±102	576±669	0.310
Fall				
Water Column	DO (mg L^{-1})	5.78±2.91	9.06±0.08	**0.047**
	SpCond (μS cm^{-1})	802±7.33	437±0.55	**<0.001**
	pH	8.3±0.45	8.3±0.08	0.802
	Temperature (°C)	17.79±0.59	23.62±0.22	**<0.001**
Sediment	Al (mg kg^{-1})	2067±1096	1866±535	0.556
	Ca (mg kg^{-1})	5550±1199	3633±4778	0.087
	Fe (mg kg^{-1})	2517±679	2350±459	0.724
	Mg (mg kg^{-1})	442±109	475±238	0.779

Table 2.3 Mean (±1 SD, n = 5) water column soluble reactive phosphorus (SRP) concentrations and mean (±1 SD, n = 5) sediment characteristics from the west Bear Lake wetland before dredging (2012; Steinman & Ogdahl 2016) and after dredging (2017) including equilibrium phosphorus concentration (EPC_0), P sorption maxima (S_{max}), total phosphorus (TP), organic matter (OM). Asterisk (*) indicates values were determined in triplicate from each soil sample.

		Before dredging	After dredging	
	Parameter	mean±SD	mean±SD	*p*-value
Water Column	SRP (μg L^{-1})	1123±0177	4±1	<**0.001**
Sediment	*EPC_0 (μg L^{-1})	217±151	39±27	<**0.001**
	*S_{max} (mg kg^{-1})	3413±2868	58±80	<**0.001**
	TP (mg kg^{-1})	2771±1243	146±134	**0.006**
	Organic matter (%)	27±14	3±4	**0.016**

Table 2.4 Summary of statistical results of blocked two-way ANOVA analysis of total phosphorus (TP) and soluble reactive phosphorus (SRP) maximum release rates depending on dredging (Dredging) and temperature (Temp).

		Oxic				Hypoxic			
		TP		SRP		TP		SRP	
Season	Factor	*F*	*p-value*	*F*	*p-value*	*F*	*p-value*	*F*	*p-value*
Summer	Dredging	304.2	**<0.001**	385.9	**<0.001**	162.3	**<0.001**	500.6	**<0.001**
	Temp	0.21	0.152	1.94	0.175	2.96	0.097	3.49	0.073
	Dredging x Temp	4.41	**0.045**	4.55	**0.042**	6.46	**0.017**	3.34	0.079
Fall	Dredging	194.5	**<0.001**	235.2	**<0.001**	53.6	**<0.001**	256.3	**<0.001**
	Temp	0.51	0.481	0.20	0.656	1.82	0.189	1.67	0.207
	Dredging x Temp	2.18	0.15	0.09	0.763	2.70	0.112	0.64	0.430

Table 2.5 Mean (±1 SD) maximum total phosphorus (TP) and soluble reactive phosphorus (SRP) release rates from sediment cores collected in the west Bear Lake wetland before dredging (2013; Smit & Steinman 2015) and after dredging (2017) under different treatment combinations. Oxic refers to oxic treatment; Hypoxic refers to hypoxic treatment. Ambient refers to ambient water temperature; +2 °C refers to ambient+2 °C water temperature.

		TP Release Rate (mg P m^{-2} d^{-1})		SRP Release Rate (mg P m^{-2} d^{-1})	
Season	Treatment	Before dredging	After dredging	Before dredging	After dredging
Summer	Oxic ambient	46.8±17.7	0.8±3.61	35.9±14.1	0.98±0.53
	Oxic +2 °C	84.9±43.8	-0.04±0.64	59.5±32.3	1.09±1.33
	Hypoxic ambient	-	0.54±4.37	-	0.56±0.72
	Hypoxic +2 °C	-	0.97±7.37	-	0.53±0.62
Fall	Oxic ambient	57.1±26.7	0.95±1.82	36.0±22.8	0.21±0.20
	Oxic +2 °C	40.4±26.3	1.12±1.70	33.5±19.2	0.34±0.48
	Hypoxic ambient	-	1.30±4.12	-	0.20±0.42
	Hypoxic +2 °C	-	6.73±11.44	-	2.13±4.99

Table 2.6 Summary of statistical results of blocked two-way ANOVA analysis of total phosphorus (TP) and soluble reactive phosphorus (SRP) average release rates in the after-dredging cores depending on water column dissolved oxygen concentration (DO) and water temperature (Temp).

		TP		SRP	
Date	Factor	*F*	*p*-value	*F*	*p*-value
Summer	DO	24.63	**<0.001**	111.34	**<0.001**
	Temp	1.54	0.22	1.48	0.23
	DO × Temp	0.13	0.72	2.13	0.15
Fall	DO	86.43	**<0.001**	74.01	**<0.001**
	Temp	5.79	**0.02**	1.25	0.272
	DO × Temp	0.54	0.46	0.00	0.96

Figure captions

Fig. 2.1 Sampling schemes and locations within the study area. *Upper left inset* shows the study area (*star*) within the Laurentian Great Lakes Region. *Lower left inset* shows the study area (*box*) relative to, Muskegon Lake, and Lake Michigan. Bear Creek flows from NE to SW. *Right panel* shows a zoomed in view of the west Bear Lake wetland and locations of sampling. The east pond is not shown on this panel but is directly northeast of Witham Drive.

Fig. 2.2 Mean (±1 SD) total phosphorus (TP) and soluble reactive phosphorus (SRP) concentrations released into the surface water of sediment of cores collected from the west Bear Lake wetland in the (**A)** summer and (**B)** fall both before dredging (2013; Smit & Steinman 2015) and after dredging (2017) in four treatment combinations (DO concentration, temperature) over the incubation period. Oxic ambient: oxic condition ambient temperature; Oxic +2: oxic condition +2 °C temperature; Hypoxic ambient: hypoxic condition ambient temperature; Hypoxic +2: hypoxic condition +2 °C temperature. Light blue line represents mean concentrations in the hypoxic +2 treatment excluding data from cores collected from site 3. In the fall, the large error bars in the hypoxic +2 treatment were truncated in the right two panels so patterns could be observed among other treatments. Note varying scales on the y-axis due to "after dredging only" panels being a blow up of the "before & after dredging" panels

Fig. 2.3 Mean (±1 SD) average total phosphorus (TP) and soluble reactive phosphorus (SRP) release rates from the after-dredging sediment to the overlying water column measured from cores collected in the west Bear Lake wetland in the summer and fall of 2017. Results represent the four after-dredging treatment combinations simulating climate warming through manipulations of DO concentration and temperature from both the summer and fall experiments. Oxic refers to oxic treatment; Hypoxic refers to hypoxic treatment. Ambient refers to ambient water temperature; +2 °C refers to ambient+2 °C water temperature.

Figures

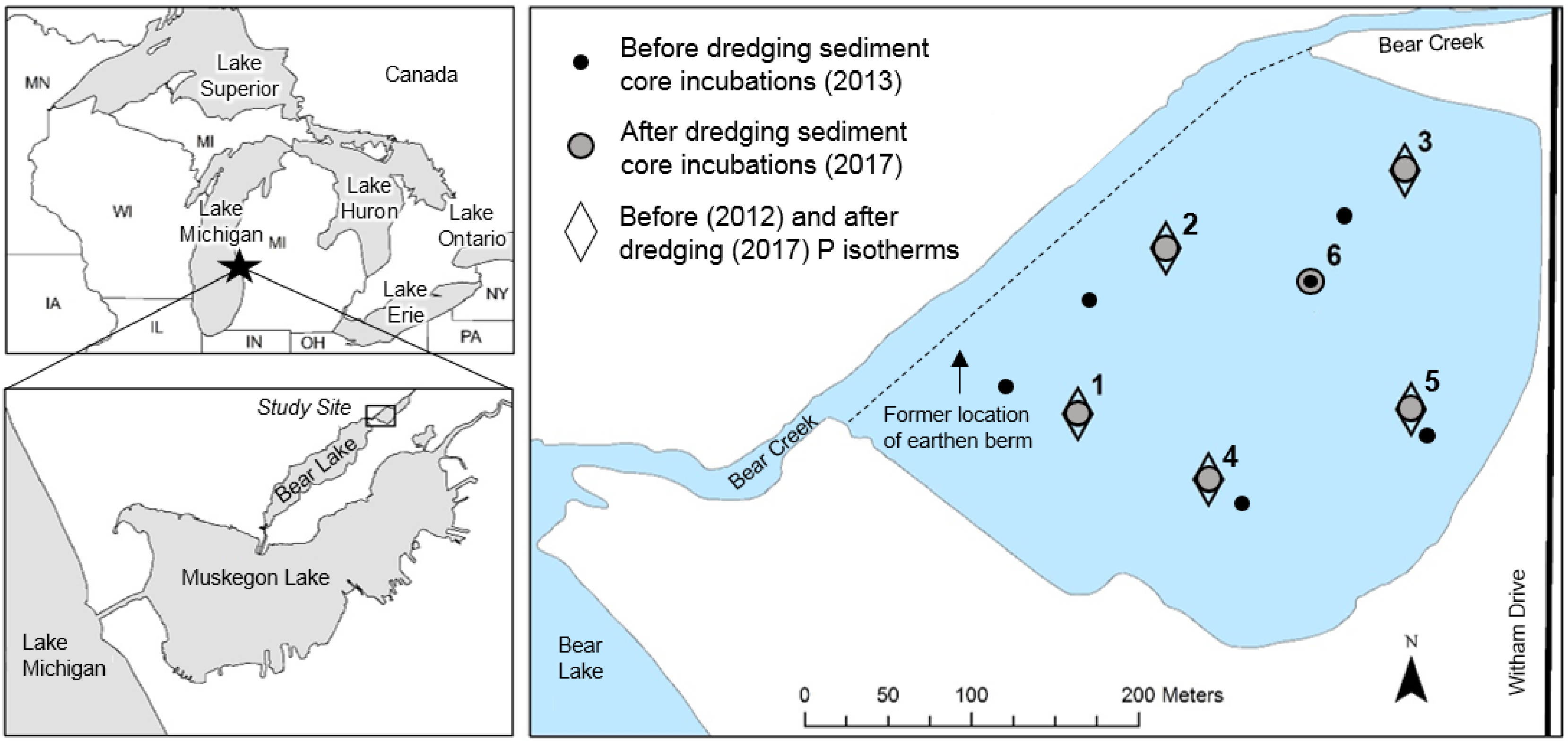

Fig. 2.1

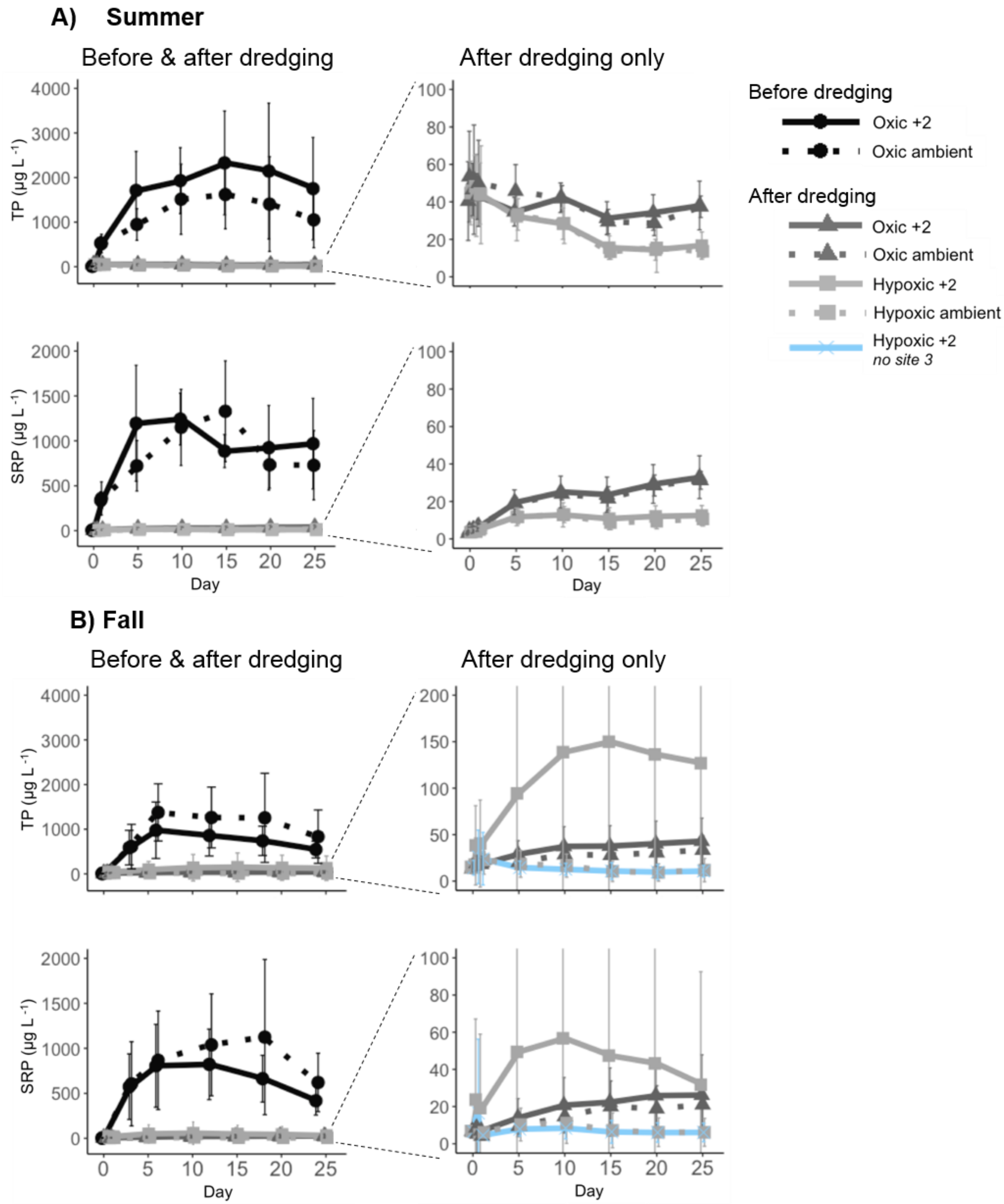
A) Summer
Before & after dredging
After dredging only
TP (µg L -1)
SRP (µg L -1)
Day
Before dredging
Oxic +2
Oxic ambient
After dredging
Oxic +2
Oxic ambient
Hypoxic +2
Hypoxic ambient
Hypoxic +2
no site 3
B) Fall
Before & after dredging
After dredging only
TP (µg L -1)
SRP (µg L -1)
Day

Fig. 2.2

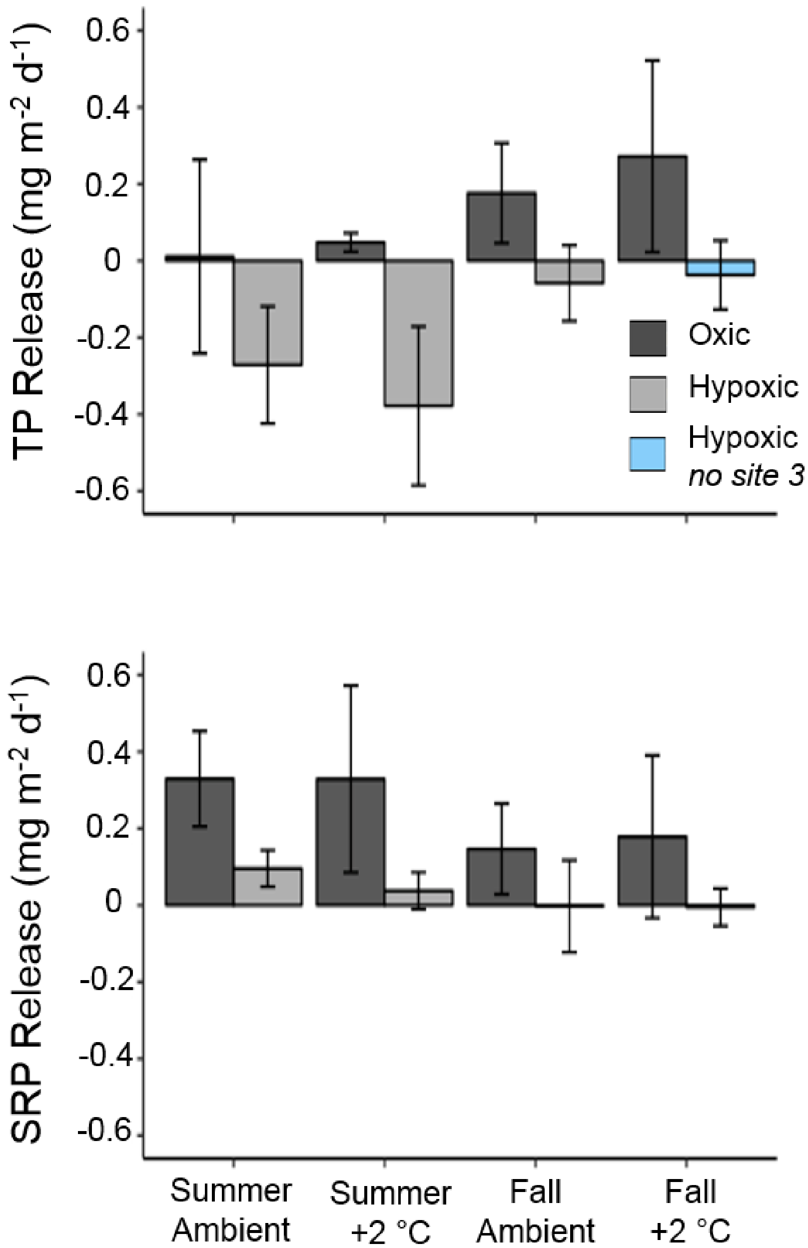
TP Release (mg m-2 d-1)
0.6
0.4
0.2
0
-0.2
-0.4
-0.6
Oxic
Hypoxic
Hypoxic
no site 3
SRP Release (mg m-2 d-1)
0.6
0.4
0.2
0
-0.2
-0.4
-0.6
Summer
Ambient
Summer
+2 °C
Fall
Ambient
Fall
+2 °C

Fig. 2.3

Chapter 3

Synthesis and Conclusions

Over 50 % of wetland area has been lost in the USA since European settlement (USDA 2012; Gibbs 2000), with many losses resulting from wetland drainage for the creation of agricultural land (Dahl 2000). Wetland losses are often concentrated around riverine wetlands, as these fertile systems have long been recognized as highly valuable agricultural lands (Tockner & Stanford 2002; Zedler 2003). The conversion of riparian wetlands and floodplains into agricultural land greatly reduces their ability to provide ecosystem services, especially for biodiversity support and water quality improvements (Zedler & Kercher 2005). However, the recognition of global reductions in freshwater biodiversity (Dudgeon et al. 2006; Strayer & Dudgeon 2010) and water quality (Smith 2003) has begun to motivate many wetland restoration projects around the world (Mitsch et al. 2005).

The restoration of floodplain and riparian wetlands provides an excellent opportunity for increasing both biodiversity and water quality because their hydrologic connectivity to adjacent lotic systems helps to create a mosaic of unique habitats for supporting biodiversity (Ward et al. 2002) while also facilitating the deposition and transformation of nutrients (Tockner & Stanford 2002), which contributes to improved downstream water quality. However, when restoration requires the hydrologic reconnection of former agricultural land to an adjacent waterbody, there can be a threat of nutrient movement from the sediments to downstream waters; mobilization of legacy phosphorus (P) from prior agricultural fertilization can lead to eutrophication (Pant & Reddy 2003; Aldous et al. 2005; Lindenberg & Wood 2006; Smit & Steinman 2015). This results in a scenario where the restoration of fish and wildlife habitat is done at the potential expense of downstream water quality.

To reduce the risk of sediment P release to downstream waters in a recent wetland restoration project in the Muskegon Lake Area of Concern (MI), sediment dredging was conducted prior to hydrologic reconnection of former agricultural land. To the best of my knowledge, there are no published studies investigating the ability of sediment dredging (conducted in situ) to reduce sediment P release in wetlands restored on former agricultural land. Therefore, to fill this knowledge gap and determine the ability of sediment dredging to reduce sediment P release in the restored west Bear Lake wetland and its practicality as a technique in future wetland restoration projects, I compared sediment P dynamics before and after dredging.

Impacts of Sediment Dredging on Sediment Phosphorus Dynamics

My results clearly show that sediment dredging prior to hydrologic reconnection was successful at reducing sediment P release rates regardless of dissolved oxygen (DO) or temperature treatment, by directly reducing the P content in the sediments (Chapter 2 of this thesis; Fig. 3.1). Specifically, my analyses showed that the high levels of sediment total phosphorus (TP) and sediment organic matter (OM), due in part from the past agricultural legacy of this system, were significantly reduced after sediment dredging (Chapter 2 of this thesis). Additionally, studies of this wetland conducted prior to and in parallel with this thesis found significant reductions in sediment pore water soluble reactive phosphorus (SRP) concentrations after dredging (Steinman & Ogdahl 2016; Hassett & Steinman 2018), indicating that sediment dredging can reduce sediment P release by directly decreasing labile P (Fig. 3.1).

The ability of sediment dredging to reduce sediment P release through reductions in sediment TP, sediment OM, and pore water SRP have also been observed in lacustrine sediments where dredging was simulated in sediment cores (Zhong et al. 2008; Yu et al. 2017) and conducted in the field (Chen et al. 2018). These prior studies have also attributed the ability of dredging to reduce sediment P release by increasing the P sorption capacity of the sediments, mainly by aerating the top sediment layer and promoting P sorption to Fe(III) oxyhydroxides (Yu et al. 2017; Chen et al. 2018). However, I found P sorption capacities to be significantly reduced after dredging in the west Bear Lake wetland (Chapter 2 of this thesis), a result also found in dredged agricultural ditches (Smith et al. 2006).

Despite the drastic reduction in sediment P release rates after dredging of the west Bear Lake wetland, the reduced sediment P sorption capacity after dredging has implications for sediment P release. The decreased sediment P sorption capacity was likely the mechanism responsible for the west Bear Lake wetland continuing to be a source (albeit minor) of dissolved P to the overlying water column in oxic conditions (Chapter 2 of this thesis). Sediment analyses indicated that the sediments were already approaching P saturation and an upward P concentration gradient was present in oxic conditions (Chapter 2 of this thesis). While wetlands are often net sinks of P due to accumulation and burial of particulate phosphorus (Caraco et al. 1991), the result of net dissolved P release after dredging in the west Bear Lake wetland was contrary to our hypothesis. Pre-restoration monitoring of the wetland found that sediment ~30-60 cm below the surface had a high potential to bind P and EPC_0 concentrations indicated the wetland would act as a P sink (Steinman & Ogdahl 2016). While a sediment's P binding capacity is just one of many characteristics that determines a wetland's function as a source or sink of dissolved P, the reduction in the sediment P binding capacity after dredging in the west Bear Lake wetland was likely to be disproportionately driving sediment P dynamics after restoration. Therefore, it would beneficial to understand why this reduction occurred so it can be avoided if dredging is used as a technique to reduce sediment P release in subsequent wetland restoration projects.

The reduction in sediment P binding capacity found in this study and Smith et al. (2006) can be partially attributed to the reduction in sediment OM content, as OM has been found to positively correlate with P sorption capacities (Syers et al. 1973; Reddy et al. 1995). However, it is unlikely that reductions sediment OM content alone account for the decrease in P sorption capacity found after the dredging in the west Bear Lake wetland. Clay content and Al, Fe, and Ca concentrations can also be primary drivers of a sediment's P sorption capacity (Reddy & DeLaune 2008), but these characteristics are unable to explain the observed reductions in P sorption capacity of the west Bear Lake wetland. Sediment clay content was not likely driving P dynamics in this system as the sediments were noted to be primarily organic muck before dredging (Steinman & Ogdahl 2016) and sand after dredging (Chapter 2 of this thesis). Additionally, the mass of total Al, Fe, and Ca in the sediments did not significantly change after dredging.

To account for the significant reduction in P sorption capacity after dredging, I speculate that the deep dredging depth exposed Al, Fe, and Ca minerals with more crystalline structures. While I did not measure P fractions or metal fractions in the sediment before and after dredging, minerals with more crystalline structures have less surface area for P sorption compared to minerals with poorly crystalline structures (McLaughlin et al. 1981; Axt & Walbridge 1999). Decreased sediment P binding capacity due to increased crystallinity of minerals after sediment dredging is also supported by, and helps to explain, the unexpectedly lower average P release rates in hypoxic core treatments than in oxic treatments (Chapter 2 of this thesis). Hypoxic core treatments likely stimulated the metabolism of iron-reducing bacteria, which can chemically reduce crystalline Fe(III) oxyhydroxides into soluble Fe(II), which can then form gel-like Fe(II) hydroxide complexes (Fig. 3.1; Davidson 1992). The potentially higher amount of soluble Fe(II) and Fe(II) hydroxide complexes in hypoxic treatments could have each differently contributed to higher sediment P sorption capacities in hypoxic treatments than in oxic treatments (Williams et al. 1971; Patrick & Khalid 1974).

Fe(II) hydroxide gel complexes have more surface area for P sorption than Fe(III) hydroxides; however, Fe(III) oxyhydroxides bind P more firmly, with less potential for P desorption (Patrick & Khalid 1974; Reddy et al. 1999; Chacon et al. 2006). Therefore, Fe(II) hydroxides usually release more P than they adsorb (Patrick & Khalid 1974; Holford & Patrick 1981; Chacon et al. 2006). These characteristics have led to a limnological paradigm that suggests dissolved oxygen controls P release from the sediments, with oxic sediments releasing less P than hypoxic sediments (Mortimer 1941). I speculate that this paradigm did not hold after sediment dredging of the west Bear Lake wetland due in part to the greatly reduced sediment TP concentrations (Chapter 2 of this thesis). The low sediment TP concentrations likely reduced the amount of P solubilized upon Fe reduction and resulted in the reduced Fe(II) hydroxide complexes acting as a net P sink rather than a P source.

Additionally, I speculate that lower P release rates in hypoxic core treatments were also due in part to a higher concentration of soluble Fe(II) hydroxides and their subsequent precipitation into amorphous Fe(III) oxyhydroxides. Soluble Fe(II) hydroxides produced by microbial iron reduction can precipitate as an orange amorphous floc on the sediment surface if they are re-oxygenated from the overlying water (Davidson 1992; Gunnars et al. 2001). Due to the dissolved oxygen concentrations in the overlying water column of the hypoxic core treatments reaching only ~3.5 mg L^{-1} (Chapter 2 of this thesis), this could have facilitated the precipitation of Fe(II) hydroxides into an amorphous Fe(III) oxyhydroxide floc which increased the sediment's P binding capacity and subsequently lowered P release rates in the hypoxic core treatments (Holford & Patrick 1981; Phillips & Greenway 1998). Therefore, the lower average P release rates observed in the hypoxic core treatments of this experiment could be an artifact of the relatively high water column DO concentrations. Nevertheless, results from these experiments still contribute to the growing notion that sediment P release is a complex process governed by more than a single paradigm (Welch & Cooke 2005; Hupfer & Lewandowski 2008).

Potential Use of Sediment Dredging in Other Systems

Sediment dredging significantly reduced sediment P release rates in the west Bear Lake wetland by reducing sediment TP, sediment OM, and pore water SRP. However, the result of sediment dredging decreasing P sorption capacities contradicts recent literature (Yu et al. 2017; Chen et al. 2018) and reveals that the effectiveness or need for sediment dredging to reduce sediment P release is due in part to the underlying geology and sediment characteristics. For example, while it has been well documented that the reflooding of former agricultural land can lead to sediment P release, at least in the short term (Pant & Reddy 2003; Aldous et al. 2005; Duff et al. 2009; Kinsman-Costello et al. 2014; Smit & Steinman 2015), active agricultural land that was reflooded near the Everglades, had substantially lower sediment P release rates than many other studies (Table 3.1; Newman & Pietro 2001). Interestingly, while Newman & Pietro (2001) found pore water SRP concentrations rising to 4 mg L^{-1} after 2-3 months of reflooding, the overlying water column SRP concentrations never exceeded 30 µg L^{-1} and decreased after the initial flooding. The low sediment P release rates, despite the high pore water SRP concentrations, were attributed to the sediment characteristics of the underlying limestone bedrock, which contributed to high concentrations of Ca and Mg in the water column and sediment. These characteristics likely reduced sediment P release into the overlying water column by promoting P co-precipitation (Newman & Pietro 2001).

The high sediment P release rates usually measured when reflooding former agricultural lands indicates that systems with legacy P are often potential candidates for restoration through sediment dredging (Table 3.1). However, the low P release rates found when reflooding some agricultural systems (Newman & Pietro 2001; Kinsman-Costello et al. 2014) and the decreased sediment P sorption capacities upon dredging of the west Bear Lake wetland suggests that sediment P dynamics in wetlands restored in agricultural systems can be highly variable across the landscape and with depth. This variability highlights the importance of pre-restoration monitoring to ensure the risks of wetland restoration are identified on a case-by-case basis, so appropriate restoration techniques are used. Specifically, pre-restoration monitoring in areas with legacy P should include analyses of sediment TP, P fractions, and P sorption capacities at various depths and spatial scales (Reddy et al. 2007; Steinman & Ogdahl 2016; Yu et al. 2017). These results can help inform 1) the risk of sediment P release; 2) if dredging could be successful; and 3) the dredging depth and spatial extent required. Combined, this information can help determine if sediment dredging is necessary, has high potential for success, and determine the most problematic sediments.

Additionally, if a wetland is proposed for restoration by dredging sediment prior to hydrologic reconnection or reflooding, pre-restoration monitoring should also include monitoring of overlying P concentrations (Steinman & Ogdahl 2016). Overlying water column P concentrations drive P sorption gradients across the sediment-water interface (Boström et al. 1988); therefore, drastic reductions in overlying water column P after hydrologic reconnection and sediment dredging can prevent the wetland from acting as a sink for dissolved P (Chapter 2 of this thesis). Alternatively, if high external P loads exist after restoration, the ability of dredging to reduce internal P loading in the long-term may be negated due to a replenishment of labile P into the sediments (Keelberg & Kohl 1999; Jing et al. 2015; Liu et al. 2016).

Total Phosphorus Load to Bear Lake

To more clearly examine the magnitude of TP loading that did not occur to downstream Bear Lake due to dredging of the west wetland prior to hydrologic reconnection, I used after-dredging P release rates from chapter 2 of this thesis and before-dredging P release rates from Smit & Steinman (2015) to calculate the "before- and after-dredging" internal TP load of the west wetland. Loading calculations were based on average TP release rates, as maximum TP release rates are unlikely to persist long-term (Smit & Steinman 2015). After-dredging average TP release rates were calculated by using the slope of a linear trend line fitted to each cores' P concentrations throughout the entire incubation period (see chapter 2 for more details). Comparatively, before-dredging average TP release rates were calculated using the change in initial to final water column TP concentrations in each sediment core because a linear trend line did not represent the general trend of the before-dredging data. Because there were generally no significant effects of incubation temperature on TP release rates before or after dredging (Chapter 2 of this thesis; Smit & Steinman 2015), release rates from cores incubated at ambient and ambient+2°C temperature treatments were combined to determine the mean average TP release rate for each season and redox treatment both before and after dredging.

Average release rates (mg TP m^{-2} d^{-1}) for each season and redox treatment were then converted into TP loads (lbs TP d^{-1}), in order to be consistent with the format of Bear Lake. These daily loads were then converted into yearly loads using three different transport scenarios, as it was unknown what percentage of the wetland's internal TP load would actually reach Bear

Lake due to biotic P uptake that may occur in either the wetland or Bear Creek. The three transport scenarios accounted for 100 % of the wetlands internal TP load reaching Bear Lake ($Load_{100}$), 50 % ($Load_{50}$), and 10 % ($Load_{10}$). Loads were calculated both before and after dredging according to an equation adapted from Steinman & Ogdahl (2015):

$$L_{transport} = ([RO_{summer} \times D_{summer} \times AO_{summer}] + [RO_{spring/fall} \times D_{spring/fall} \times AO_{spring/fall}] + [RX_{summer} \times D_{summer} \times AX_{summer}]) \times \mathit{transport}$$

where $L_{transport}$ is the annual internal TP load from the west Bear Lake wetland transported into Bear Lake, RO_{summer} is the summer oxic release rate, D_{summer} is the number of days in summer (91), AO_{summer} is the oxic wetland area in the summer (assumed to be 100 % before dredging; 80 % after dredging as indicated by in situ measurements- see details below). $RO_{spring/fall}$ is the oxic release rate in the spring and fall (while I only measured fall release rates, prior sediment core incubation experiments in Bear Lake found no significant difference between spring and fall TP release rates (Steinman & Ogdahl 2015); therefore, I assumed fall release rates would be similar to spring release rates in the west Bear Lake wetland), $D_{spring/fall}$ is the number of days in spring and fall (182), $AO_{spring/fall}$ is the oxic wetland area in the spring and fall (assumed to be 100 % before dredging; 100 % after dredging as indicated by in situ measurements- see details below). RX_{summer} is the summer hypoxic release rate, AX_{summer} is the hypoxic wetland area in summer (assumed to be 0 % before dredging; 20 % after dredging as indicated by in situ measurements- see details below). Hypoxic parameters apply only to after-dredging load calculations but were not applied to spring/fall load calculations, as fall diel dissolved oxygen (DO) monitoring did not indicate the presence of hypoxic conditions during this time. *Transport* is the proportion term (1, 0.5 or 0.01) representing the proportion of the west wetland's internal P load that actually enters Bear Lake. Winter release rates were assumed to be zero (Nürnberg 2009, Nürnberg et al. 2013).

While this assumption may be cause slight underestimates of the TP load in the west Bear Lake wetland (Orihel et al. 2017), the calculations are still useful for illustrating the impact of sediment dredging on the internal TP load of the west Bear Lake wetland.

To obtain information on the redox status of the dredged wetland and account for DO concentrations in the after-dredging internal TP load calculations, I measured diel DO concentrations once in the summer and once in the fall of 2017. Water column DO concentrations were measured overnight, from the afternoon until at least the next morning, to characterize diel fluctuations in DO. At the deepest site (site 4), a YSI 6600 sonde was suspended from both the near-surface and at the near-bottom of the water column with an anchored buoy. At the shallower sites (site 3 and site 5), a sonde was suspended at only the near-bottom. In the fall monitoring event, the sensor deployed at the surface of site 4 malfunctioned, so no data were collected at the surface of site 4. These sites were chosen because they represent a range of depths within the dredged wetland. Diel DO concentrations were measured in the summer (24-25 July 2017) and fall (19-20 October 2017) to correspond with the seasonal sediment core incubation experiments. Wind speed data for all diel DO events were downloaded from the Muskegon Lake Buoy Observatory (AWRI 2012), located ~0.5 km south of the sampling sites.

Results from the summer diel DO monitoring indicated that the bottom of site 4 (3.0 m depth) remained anoxic, regardless of time of day (Fig. 3.2). DO concentrations at site 5 (2.3 m depth) fluctuated between ~2 and 6 mg L^{-1} throughout the monitoring period. At site 2 (1.7 m depth) and the surface of site 4, the water was well oxygenated and DO concentrations ranged between ~6 and 9 mg L^{-1}. While generalizing bottom-water DO concentrations by depth may not be accurate in this wetland due to spatial heterogeneity caused by groundwater inputs, creek flow, biotic processes, wind events, or over time due to the short period of monitoring, I nevertheless used the results from the diel DO monitoring to categorize all depths 3.0 m or greater as being hypoxic during the summer. Bathymetric analysis after dredging indicated that ~20 % of the wetland was at least 3.0 meters or greater in depth. This percentage resulted in 18,000 of the 90,000 m^2 wetland area being classified as anoxic in the summer. The information from this monitoring was ultimately used for calculating the AO_{summer} and AX_{summer} parameters in the internal loading model described previously. Results from the fall diel DO monitoring indicated that bottom-water DO concentrations stayed above ~6 mg L^{-1} at all sites, regardless of time of day (Fig. 3.2). Therefore, I assumed that oxic conditions were present in 100 % of the wetland throughout the fall season, which was used to calculate the $RO_{spring/fall}$ parameter in the internal loading model. While I only measured diel concentrations in the summer and fall, I assumed that results from the fall diel monitoring were representative of the DO concentrations in the spring as well.

The results of the annual internal loading calculations show that dredging of the west wetland prior to hydrologic reconnection drastically reduced TP release to downstream Bear Lake, regardless of the transport scenario (Table 3.2). Overall, the calculations indicate that dredging avoided between 56 lbs (under the most conservative transport scenario) and 557 lbs (under the complete transport scenario) of sediment-derived TP entering Bear Lake per year. These reductions correlate to sediment dredging having reduced the internal TP loading from the west Bear Lake wetland into Bear Lake by over 99 %.

The ability of dredging to reduce unintended sediment P release is encouraging for the potential to restore wetlands on former agricultural land without harming downstream water quality. However, at this study location, the downstream Bear Lake still has TP concentrations above the TMDL goal of 0.03 mg L^{-1} (MDEQ 2008; Hassett & Steinman 2018). Additionally, the TMDL states that TP contributions from all wetlands in the watershed are 0 mg L^{-1}; therefore, avoiding this unintended TP release to downstream Bear Lake did not directly contribute toward the 50 % reduction in Bear Creek's TP load required by the TMDL (MDEQ 2008). However, Bear Creek contributes 87 % of the estimated external TP load (1,529 lbs TP yr^{-1}) to Bear Lake (MDEQ 2008), and monthly water quality monitoring in 2017 revealed that TP concentrations in Bear Creek still generally exceed the TMDL goal of 0.03 mg TP L^{-1} in Bear Lake (Hassett & Steinman 2018). Therefore, this hydrologically reconnected wetland now has a high potential for reducing Bear Creek's TP load by facilitating the deposition of particulate P and promoting biological uptake with subsequent burial of OM and associated P (Reddy et al. 1999). Additionally, given that this wetland is positioned near the mouth of Bear Creek, it is in an ideal hydrologic location for reducing the TP load into Bear Lake as nearly 100 % of Bear Creek's flow has the potential to exchange with the wetland.

In addition to this restored wetland reducing TP loads to downstream Bear Lake, it may also improve downstream water quality by reducing nitrogen (N) loads (Salk et al. 2017). Sediment core experiments conducted prior to restoration of the west Bear Lake wetland found that simulated sediment dredging and hydrologic reconnection promoted net retention of ammonium in the wetland, with the potential for removing up to 10 % of the ammonium in Bear Creek (Salk et al. 2017). While N to P molar ratios indicate that production in Bear Lake is limited by only P (Cadmus & AWRI 2007; MDEQ 2008; Xie et al. 2011), there has been an increased focus in the literature for the need to manage both N and P in order to successfully reduce harmful algal blooms (Conley et al. 2009; Paerl et al. 2016). Therefore, the potential of the west Bear Lake wetland to act as a net sink for both N and P may contribute to greater improvements in downstream water quality as compared to it reducing only one nutrient.

Additional Considerations

Due to sediment dredging being an engineered solution that physically removes sediments from aquatic systems, it is inherently a major disturbance to ecological systems. There is likely to be negative effects on the benthic invertebrate populations after dredging, but they often reestablished ~2 years after dredging (Crumpton & Wilbur 1974; Carline & Brynildson 1977). Additionally, dredging may negatively affect the plant community and subsequent habitat quality because highly disturbed wetlands (such after dredging) may promote the establishment of invasive species (Zedler & Kercher 2004). However, vegetative plantings can be done to reduce the risk of invasive species establishment after a wetland restoration (Streever & Zedler 2000). While effects to benthic biota should be considered prior to sediment dredging, the potentially long-term improvements in water quality after dredging usually offset the short-term impacts to the biota (Lewis et al. 2001; Cooke et al. 2005).

The major limitations for the use of sediment dredging as a restoration technique is the high cost, obtaining permits, and locating a site for disposal of the dredged material (Cooke et al. 2005). A review of ten restoration-driven sediment dredging projects in lakes and ponds of the U.S. indicated that the average cost of sediment dredging is ~$7 per m^2 (Cooke et al. 2005; all budgets adjusted for inflation). The cost to restore the east and west Bear Lake wetlands was much higher, averaging $43 per m^2. This high cost was due in part to the draining, pumping, and treating of the overlying water and transport of the dredged material to a landfill. Prior to the Bear Lake wetland restoration project, the most expensive per m^2 sediment dredging restoration project in the U.S. was done to restore a pond in New York City's Central Park, which amounted to \$28 per m^2 (Cooke et al. 2005; adjusted for inflation). This restoration project also drained the pond prior to dredging and transported the dredged material away from the site, suggesting that draining and transport of the dredged material adds significant costs to the use of sediment dredging as a technique to reduce sediment P release. Comparatively, using chemical inactivation, such as aluminum sulfate, to reduce sediment P release has an average cost of only \$0.13 per m^2 (Cooke et al. 2005). However, sediment dredging has a significant long-term advantage over the use of chemical inactivation because it can remove the nutrient source, rather than leaving it bound in the sediments with potential for release (Cooke et al. 2005). Therefore, sediment dredging is likely to be a longer-lasting solution than chemical inactivation (Cooke et al. 2005).

Finally, while pre-restoration monitoring can help to determine the feasibility of sediment dredging as technique for reducing sediment P release in wetlands restored in former agricultural land, it is difficult to determine how future climate change and anthropogenic disturbance may affect dredging success. Nevertheless, climate change and anthropogenic disturbances can have profound effects on wetland biogeochemistry (Burkett & Kusler 2000; Reddy & DeLaune 2004); therefore, consideration of future limitations to dredging success should also be considered.

Changes in hydrological regime due to altered precipitation patters or diversion/impoundment of upstream water may be the biggest threat for the ability of sediment dredging to reduce sediment P release. Fluctuations in wetland hydrology that result in the drying of wetland sediments can result in sediment P release upon reflooding (Steinman et al. 2014; Kinsman-Costello et al. 2016), likely by decreasing the sediment's P sorption capacity (Dieter et al. 2015). Additionally, increased water temperatures from climate warming or industrial effluent can alter P sediment biogeochemistry by stimulating benthic microbial respiration and increasing bottom water hypoxia (North et al. 2014). Decreased redox status in the benthos stimulates microbial iron reduction which can lead to phosphate desorption and potential release into the overlying water (Marsden 1989). While I did not find a 2º C increase in water temperature to currently have a significant effect on sediment P release rates in the west Bear Lake wetland, this result may be due to the relatively small changes in incubation temperatures within experiments and between seasons, so it is difficult to determine if the lack of temperature effect is an artifact of the experimental design or representative of sediment dredging's ability to lower P release regardless of water temperature.

Conclusions

This research helps contribute to the field of ecological restoration and water quality management by providing cautiously optimistic results that sediment dredging, informed with knowledge of the underlying sediment characteristics can be used as a technique to avoid situations where restoration of wetland habitat is done at the expense of downstream water quality. However, sediment dredging may be unnecessary or inappropriate for all wetlands restored on former agricultural lands because of underlying sediment characteristics or project costs. Therefore, pre-restoration monitoring should be conducted to ensure that the most appropriate technique is selected for restoration. At a local level, the results from this study are useful because it is now known that additional management for P release is not currently needed for in the west Bear Lake wetland, and downstream water quality was not degraded due to this wetland restoration. Continued monitoring will be needed in this wetland to more accurately determine how sediment P dynamics respond as the wetland revegetates and begins to accumulate organic matter. Additionally, continued water quality and habitat monitoring will help determine the relative importance of this restored wetland for reducing TP loads to downstream Bear Lake and the relative trade-off between increased water depth and wildlife habitat (Hansson et al. 2005). More broadly, the results from this study in conjunction with continued monitoring efforts will be useful for informing subsequent wetland restoration efforts in areas with legacy P and ultimately shed light on the use of this wetland restoration project as a success story for both water quality and habitat improvements.

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Tables

Table 3.1 Published sediment soluble reactive phosphorus (SRP) release rates after (simulated) reflooding of former agricultural lands.

Source	Location	Site/Study Description	Time in agriculture (years)	Agricultural use	SRP release ($mg\ m^{-2}\ d^{-1}$)
Newman & Pietro 2001	Everglades Nutrient Removal Project, FL	Reflooding of active agricultural land	~45 years	Vegetables & Sugarcane	0.3 - 1.5[a]
Pant & Reddy 2003	Okeechobee Drainage Basin, FL	Simulated reflooding of abandoned and active agricultural land	34-20 years	Dairy	15-93[b]
Aldous et al. 2005	3 restored wetlands near Upper Klamath Lake, OR	Reflooding of abandoned agricultural land	40-90 years	Crops & Beef Cattle	10-55[b,c]
Duff et al. 2009	Wood River Wetland, Upper Klamath Lake, OR	Reflooding of abandoned agricultural land	40-50 years	Crops & Beef Cattle	19-72[a]
Kinsman-Costello et al. 2014	Fort Custer Wetland, MI	Reflooding of abandoned agricultural land	Unknown	Unknown	0.34-11.8[b]
Smit & Steinman 2015	West Bear Lake wetland, Muskegon, MI	Simulated hydrologic reconnection of abandoned agricultural land	70 years	Celery	38-63[b]
This Study	West Bear Lake wetland, Muskegon, MI	Dredging prior to hydrologic reconnection of abandoned agricultural land	70 years	Celery	0.2-2.1[b]

[a] Rates determined by measuring nutrient profiles across the sediment-water interface
[b] Rates determined by incubating intact sediment cores
[c] Rates representative of a variety of hydrologic conditions (dry, moist, flooded)

Table 3.2 Summary of annual internal total phosphorus (TP) loads in the west Bear Lake wetland A) before dredging and B) after dredging that may reach downstream Bear Lake after hydrologic reconnection when considering scenarios where 100 % of the wetland's internal TP load is transported into Bear Lake ($Load_{100}$), 50 % ($Load_{50}$), and 10 % ($Load_{10}$).

A. Before Dredging					
Season	Summer		Spring & Fall		Annual
Redox	Oxic		Oxic		
Avg. RR (mg TP m^{-2} d^{-1})	13.9		8.5		
Area of wetland (%)	100 %		100 %		
Period of loading (days)	91		182		
$Load_{100}$ (lbs TP yr^{-1})	251		308		559
$Load_{50}$ (lbs TP yr^{-1})	125		154		279
$Load_{10}$ (lbs TP yr^{-1})	25		31		56
B. After Dredging					
Season	Summer		Spring & Fall		Annual
Redox	Oxic	Hypoxic	Oxic	Hypoxic	
Avg. RR (mg TP m^{-2} d^{-1})	-0.02	-0.32	0.11	-0.04	
Area of wetland (%)	80 %	20 %	100 %	0 %	
Period of loading (days)	91	91	182	-	
$Load_{100}$ (lbs TP yr^{-1})	-0.2	-1.1	3.8	-	2.5
$Load_{50}$ (lbs TP yr^{-1})	-0.1	-0.6	1.9	-	2.1
$Load_{10}$ (lbs TP yr^{-1})	0.0	-0.1	0.2	-	0.1

Figure Captions

Fig. 3.1(a,b) Generalized schematic of the sediment phosphorus dynamics in the west Bear Lake wetland (**A**) before dredging with simulated hydrologic reconnection and (**B**) after dredging and hydrologic reconnection.

Fig. 3.2 Diel dissolved oxygen data measured (**A**) 24-25 July 2017 and (**B)** 19-20 October 2017 at three locations in the west Bear Lake wetland. Wind speed shown in dark gray, nighttime shown in light gray

Figures

A) Before dredging with simulated hydrologic reconnection

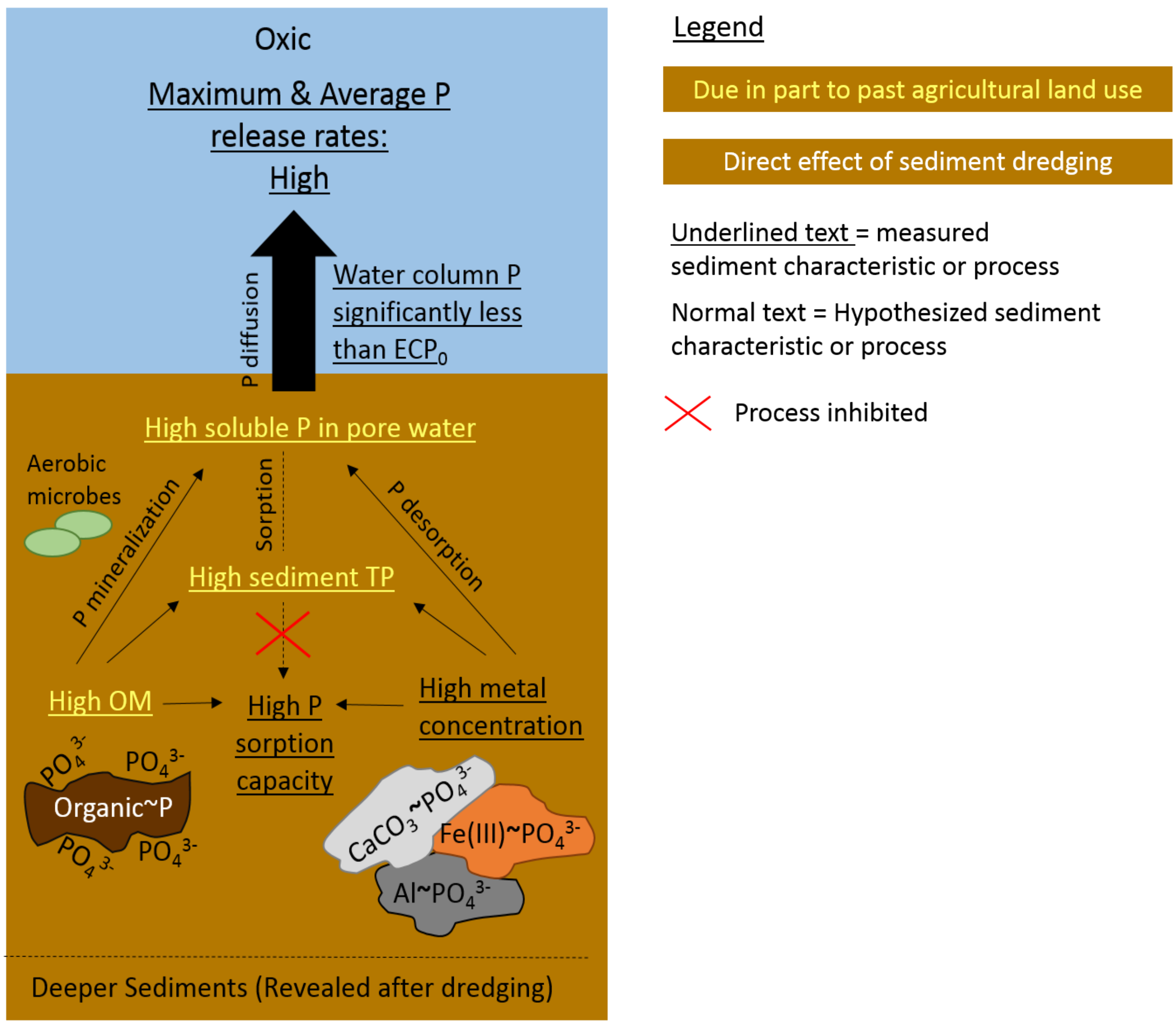

Fig. 3.1a

B) After dredging and hydrologic reconnection

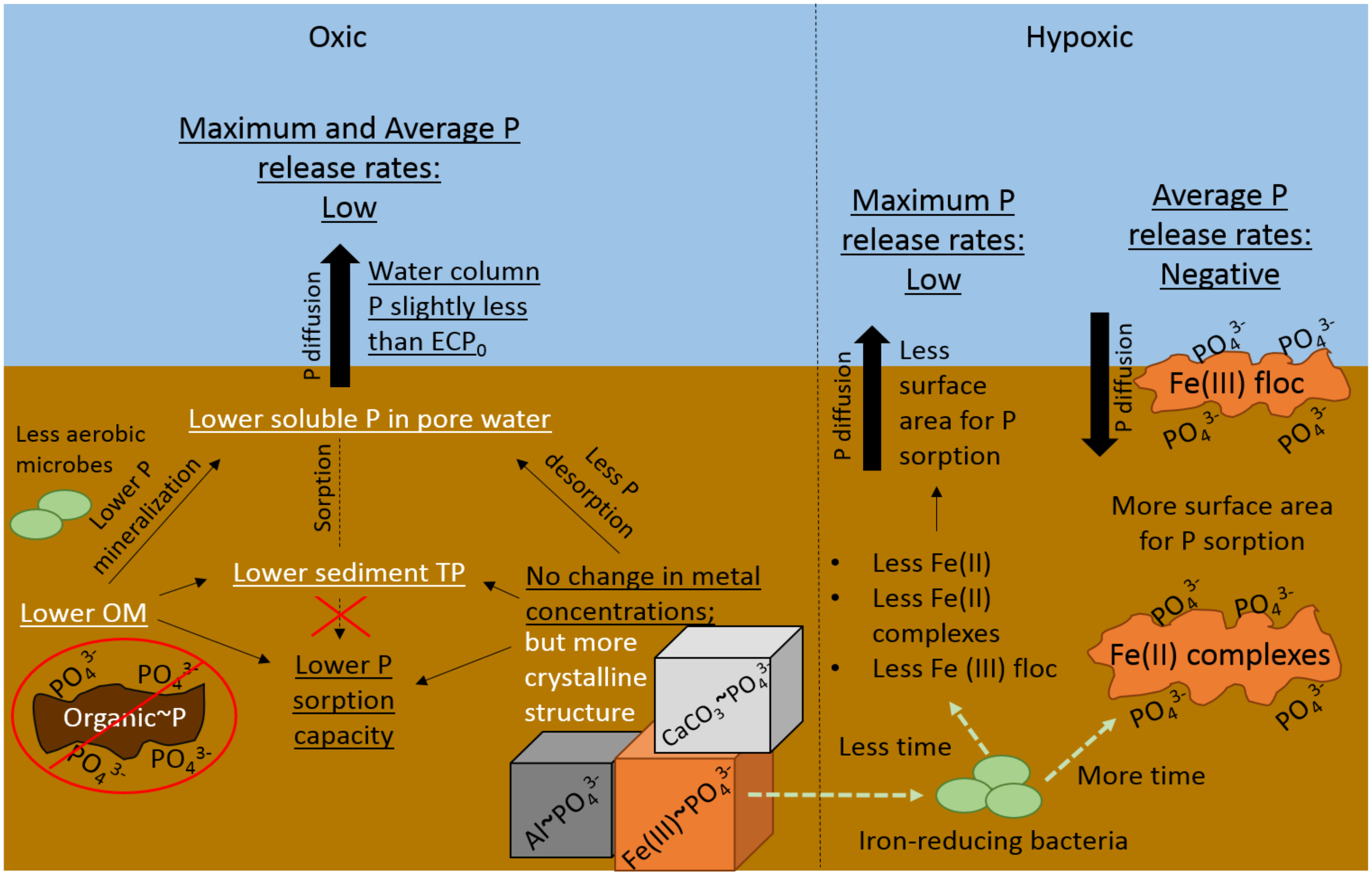

Fig. 3.1b

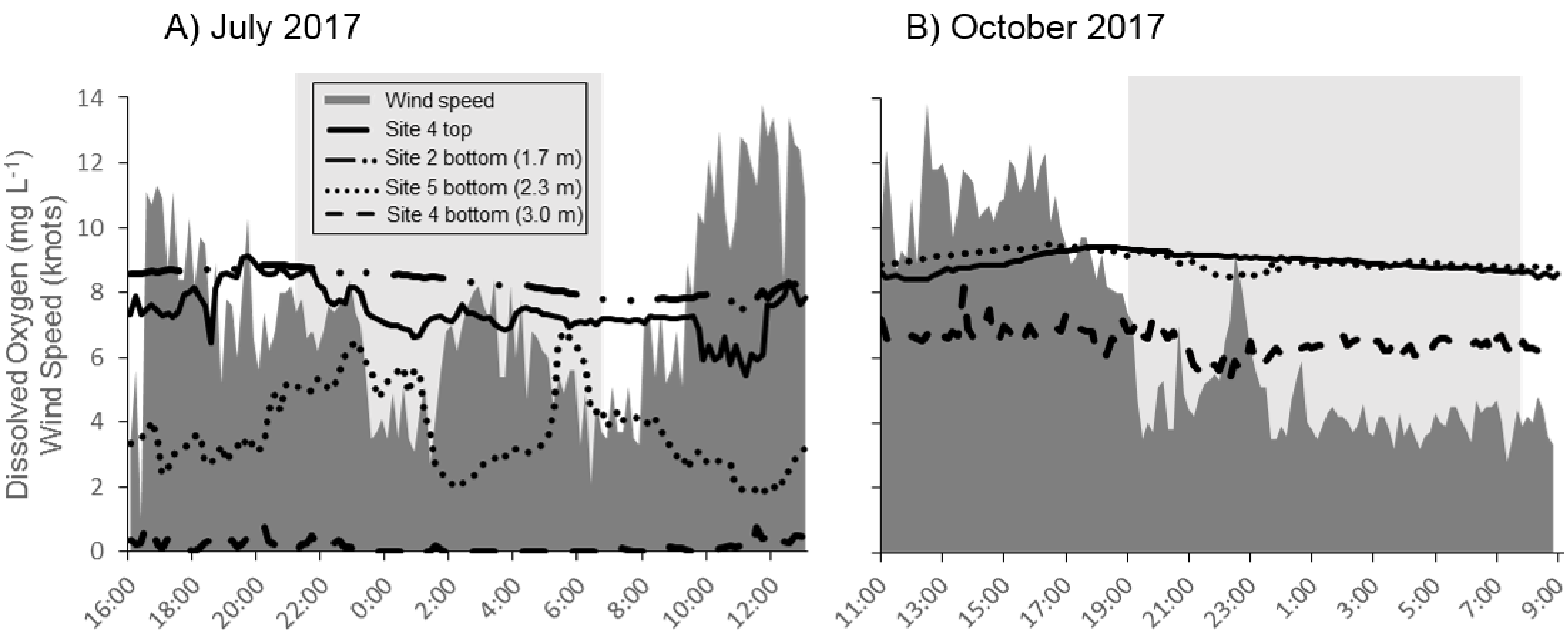
A) July 2017
B) October 2017
Wind speed
Site 4 top
Site 2 bottom (1.7 m)
Site 5 bottom (2.3 m)
Site 4 bottom (3.0 m)
Dissolved Oxygen (mg L-1)
Wind Speed (knots)
0
2
4
6
8
10
12
14
16:00
18:00
20:00
22:00
0:00
2:00
4:00
6:00
8:00
10:00
12:00
11:00
13:00
15:00
17:00
19:00
21:00
23:00
1:00
3:00
5:00
7:00
9:00

Fig. 3.2