DISSOLVED PHOSPHORUS DYNAMICS AND MANAGEMENT WITHIN THE AGRICULTURAL LANDSCAPE

A Dissertation
SUBMITTED TO THE FACULTY OF THE
UNIVERISTY OF MINNESOTA
BY

Laura J. Bender

IN PARTIAL FULFILLMENT OF THE REQUIREMENTS
FOR THE DEGREE OF
DOCTOR OF PHILOSPHORY

Christian Lenhart

May, 2022

Acknowledgements

This dissertation is dedicated to my partner Marika, my daughter Finley and my son Wyatt. This work is as much theirs as it is mine. I would also like to acknowledge my advisor Chris Lenhart for graciously providing professional, academic and financial support over the last several years. Also, to my committee members including Bruce Wilson, Jacques Finlay and John Nieber for providing both their time and professional guidance.

Financial support was provided by the Minnesota Department of Agriculture for work on Chapter 1 of this dissertation. Work on Chapter 2 of this dissertation was provided by the University of Minnesota MnDrive and Institute on the Environment programs as well as by the CFANS Graduate Student Board. The Minnesota Department of Agriculture Discovery Farms Program provided invaluable data for work on Chapter 3 of this dissertation.

I would also like to acknowledge the landowners who provided their farms as field sites for this research including David Legvold of Northfield Minnesota and Darwin Roberts of Granada Minnesota. Lastly, I would like to extend my gratitude to Professor Emeritus Fred Bergsrud and Marty Bergsrud for providing fellowship funding to support completion of my doctoral degree.

Abstract

Agricultural phosphorus loss was identified as a water quality priority within the Minnesota Nutrient Reduction strategy identifying a 45% yield reduction goal. Implementation plans call for total phosphorus (TP) reduction with traditional management strategies designed for erosion control and particulate nutrient removal, leaving the dissolved, or bioavailable, forms of phosphorus un-accounted for. Substantial yield increases in agricultural tributaries over recent decades highlight the need for dissolved reactive phosphorus (DRP) management, with some sources documenting over 50% DRP contributions to TP loads in Minnesota. DRP, hydrology, management and site-specific factors were investigated at two field research sites in southern Minnesota, with additional data harnessed from the Minnesota Discovery Farms Program. Data was used to assess the impacts of various site and management factors including cover crops, riparian buffers, edge-of-field wetlands, tillage category, fertilizer application and soil properties on phosphorus loads from farm fields and edge-of-field best management practice (BMPs) uptake.

Four project objectives were addressed; 1.) to quantify and characterize current and target DRP yields from Southern Minnesota agricultural fields, 2.) to quantify the influence of local field and management conditions on DRP yields, 3.) to assess the effectiveness or inefficacy of common management practices for phosphorus and nitrogen removal, and 4.) to explore novel management strategies for DRP yield reductions including treatment trains, microbial soil amendment and plant harvest. Methodology included hydrologic monitoring, soil assessment, edge of field nutrient concentration analysis and measurement of phosphorus in vegetation to track phosphorus movement through the soil, water and plant components of agroecosystems.

Current DRP loss rates from agricultural fields were quantified at 0.49 kg ha^{-1}, with a target DRP yield of 0.27 kg ha^{-1} to achieve a 45% phosphorus reduction. To meet target yields, project results demonstrated the importance of both surface and subsurface DRP loss pathways, legacy phosphorus monitoring and management and the need for coordinated edge of-field and in-field management strategies. Significant conditions driving drain tile DRP concentrations included manure application rate, number of tillage passes and soil test phosphorus (STP). Significant conditions driving surface DRP concentrations included cumulative manure and fertilizer application rate and STP. STP accumulation was driven most significantly by manure application rate, number of tillage passes, organic matter content, clay content, soil pH and cover crop implementation. Cover crops, which were placed into the context of an agricultural treatment train, were found to reduce subsurface DRP concentrations by 63% and annual yields by .07 kg ha^{-1} through reduced constituent mobilization at higher flows but also to contribute to increased STP. Crop use efficiency, fertilizer application and soil phosphorus draw down where also associated as part of a mass balance to further correlate management action to DRP yields. Research findings will help to inform agricultural management for DRP removal strategies necessary for setting and meeting realistic nutrient reduction and water quality goals.

Table of Content

List of Tables

List of Figures

General Introduction

Scope

In 2014, the Minnesota Nutrient Reduction Strategy called for a 45% reduction in phosphorus yields from baselines conditions prior to 2025 (MPCA, 2014). This reduction goal was set in place as part of a larger Gulf of Hypoxia Task Force nutrient reduction strategy for phosphorus (EPA, 2020). Excess phosphorus in Minnesota waterbodies is the main driver behind eutrophication, or an over-occurrence of toxic blue-green algal blooms, leading to diminished aquatic recreation, loss of habitat and harm to aquatic and human health (Anderson, Gilbert, & Burkholder, 2002; Ansari, Singh, Lanza, & Rast, 2010). Recently, phosphorus management initiatives have included installation of best management practices (BMPs) at the edge of agricultural fields designed to treat surface and subsurface runoff before nutrients reach nearby waterways (Lenhart, et al., 2017).

These practices, including cover crops, treatment wetlands and riparian buffers, among others; have traditionally been designed for particulate nutrient removal leaving dissolved phosphorus forms unaccounted for (Lenhart, Wilson, & Gordon, 2016). In addition, many water quality planning initiatives in Minnesota focus only on total phosphorus (TP) losses. Dissolved reactive phosphorus (DRP) is the proportion of TP remaining in the water column, after filtration and removal of particulate phosphorus, that is readily available for use by plants (Caduto, 1990). DRP is a particular water quality concern due to its high level of bioavailability, supporting rapid algae growth (Baker, et al., 2014). Increased agricultural DRP yield observed in recent decades is likely attributed to agricultural expansion, legacy phosphorus accumulation, BMP design standards and climate change; highlighting the need for phosphorus management (Boardman, Danesh-

Yazdi, Foufoula-Georgiou, Dolph, & Finlay, 2019; Maccoux, Dove, Backus, & Dolan, 2016; McDowells, Depree, & Stenger, 2020).

These increased yields may also be attributed to changes in agricultural management practices, particularly tillage category and fertilizer application. Conservation tillage, a recent and widespread practice, may enhanced macropore development and phosphorus mineralization, contributing to DRP loss (Baker, Johnson, Confesor, & Crumrine, 2017; Daryanto, Wang, & Jacinthe, 2017; Jarvie, et al., 2017). In addition, phosphorus is abundant in agricultural fertilizers and manure which concentrate in agricultural runoff, contributing to the majority of non-point source pollution. This non-point source pollution constitutes two-thirds of the TP yield in Minnesota surface waters under normal flow conditions (Barr Engineering Co., 2004).

Legacy phosphorus, or high concentrations of phosphorus remaining in crop and soil residue from decades of past management activity, may also contribute to observed increases in agricultural DRP yields. As surface runoff is a dominant source of DRP, this phenomenon is further exacerbated by climate change patterns in the form of more frequent and intense storms. Legacy phosphorus also contributes to observed BMP "lag time," where pollutant reductions from management initiatives may take years or even decades to observed (Sharpley, et al., 2013; Jarvie, et al., 2013).

Agricultural BMP monitoring efforts have also shown great variability in DRP removal efficacy with some, at times, found to contribute DRP to nearby waterways (Dodd & Sharpley, 2015; Dodd, Sharpley, & Berry, 2018; Jarvie, et al., 2017; Lenhart, Wilson, & Gordon, 2016; Smith, King, & Williams, 2015; Young E. O., Ross, Jaisi, & Vidon, 2021). Practice variability is the result of complex factors including local climate, landscape, geology, land use, and socioeconomics. These complexities are difficult to account for and make short-term quantification of progress towards water quality goals difficult (Minnesota Pollution Control Agency, 2019).

In addition to quantifying and characterizing DRP removal efficacy of BMPs and the influence of management strategies on DRP yields, a need exists to quantify current DRP loss rates and drivers for setting realistic and attainable phosphorus water quality goals. This work proposes to address these research needs through the completion of three independent field-based studies completed at multiple agricultural research sites located throughout Southern Minnesota.

Objectives

Identified research needs are addressed within the following four primary project objectives: 1.) to quantify current and target DRP yields from Southern Minnesota agricultural fields, 2.) to quantify the influence of local field and management conditions on DRP yields, 3.) to assess the effectiveness or inefficacy of common BMPs for DRP removal, and 4.) to explore novel management strategies for DRP yield reductions.

The first chapter entitled “Identifying Upper Mississippi River Basin target loads, loss mechanisms and management need for reduction of agricultural dissolved reactive phosphorus losses,” addresses project objective one and project objective two through the three following sub-objectives (a) to identify current and target DRP yield characteristics and rates from Upper Mississippi River Basin farmlands to meet Gulf of Mexico nutrient reduction goals, (b) identify dominant site and management mechanisms driving surface and drain tile DRP concentrations and soil test phosphorus (STP) accumulation and (c) to calculate a phosphorus mass balance to evaluate DRP losses relative to fertilizer application, crop use efficiency and soil phosphorus saturation.

Findings from objectives (b) and (c) may be synthesized to inform more realistic target for field loss and strategies on how to reach those targets. Site and management dominant mechanisms included within this analysis were those measured and reported at the field scale including STP, number of tillage passes, manure and fertilizer application rate, cover crop implementation, organic matter content, clay content, pH, rainfall depth, surface runoff and drain tile discharge.

The second chapter entitled "Cover crops reduce subsurface nutrient loads to downstream management practices improving system performance and cost-effectiveness," addresses project objective three and project objective four through addressing the five following research questions: (1). Do cover crops provide subsurface total nitrogen (TN), TP and DRP flow weighted mean concentration (FWMC) and load reductions? (2). What TN, TP and DRP removal cost efficiencies are associated with cover crop implementation? (3). How do subsurface TN, TP and DRP concentration discharge relationships vary in the presence and absence of cover crops? (4). How do drain tile flow volumes vary in the presence and absence of cover crops? (5). Do cover crops provide lower per-unit cost nutrient removals than the use of a stand-alone treatment wetland when implemented as part of an agricultural BMP treatment train?

In addition to exploring the use of cover crops as a DRP management strategy, a novel management practice explored within this chapter is that of an agricultural BMP treatment train. This is defined as a series of BMPs placed along a landscape gradient to treat portions of the same runoff and load with the purpose of improving system-wide nutrient removal and cost efficiency (Lenhart, et al., 2017; Lien & Magner, 2017).

The third chapter entitled “A case study of dissolved reactive phosphorus contribution, loss drivers and management techniques within a Southern Minnesota agricultural riparian buffer,” addresses project objective 2, project objective 3 and project objective 4 through the three following sub-objectives (1). to demonstrate the occurrence of DRP contributions from an agricultural riparian buffer (2.), to characterize dominant drivers of riparian buffer DRP loss, and (3.) to explore riparian zone DRP loss mitigation strategies. Drivers of DRP loss explored at the riparian buffer field site included the re-establishment of organic matter, flooding frequency and soil phosphorus saturation. DRP mitigation strategies explored included plant harvest and soil test phosphorus monitoring. An additional novel DRP mitigation strategy explored within this study in the use of a soil arbuscular mycorrhizal fungi amendment and soil microbial community structural assessment.

Chapter 1: Identifying Upper Mississippi River Basin target loads, loss mechanisms and management need for reduction of agricultural dissolved reactive phosphorus losses

Summary

Agricultural dissolved reactive phosphorus (DRP) losses contribute to algal bloom occurrence in water bodies and associated water quality impairments. While extensive research has been conducted in surrounding regions, this study quantifies current DRP losses from agricultural lands within the State of Minnesota in the Upper Mississippi River Basin and compares them to established targets, while also identifying dominant loss mechanisms and management needs. Farm management, hydrology and water quality data collected from seven agricultural research sites over thirty-seven study years were provided by the Discovery Farms Minnesota program. DRP monitoring data were compared against phosphorus reduction goals identified in the Minnesota Nutrient Reduction Strategy to set a regional target DRP yield of 0.27 kg ha^{-1} yr^{-1}. Development of a phosphorus mass balance identified the need for joint input reduction and edge-of-field management strategies for attaining DRP target loads. Multiple linear regression analysis of flow-weighted mean concentrations (FWMCs) of DRP were used to assess influence of soil test phosphorus (STP), number of tillage passes, manure and fertilizer application rate, cover crop implementation, organic matter content, clay content, pH, rainfall depth, surface runoff and drain tile discharge. Multiple linear regression analysis was also utilized to assess the impact of these factors on STP. Dominant drain tile DRP FWMC drivers included manure application rate, number of tillage passes and STP. Dominant surface DRP FWMC drivers included total phosphorus (TP) application rate and STP.

Dominate drivers of STP accumulation included manure application rate, number of tillage passes, organic matter content, clay content, soil pH, and cover crop implementation. Results highlight the need for input reduction, STP monitoring and edge-of-field practice implementation for meeting regional goals identified as part of the greater Hypoxia Task Force Nutrient Reduction Strategy. Results also highlight increased DRP loss under conservation tillage as well as STP contributions associated with cover crop implementation.

Introduction

Agricultural management practices were historically designed for sediment management with many in recent decades designed primarily for nitrogen removal as a result of national prioritization of the Gulf of Mexico hypoxia problem (Christianson, Frankenberger, Hay, Helmers, & Sands, 2016; Lenhart, Wilson, & Gordon, 2016). Re-evaluation of nutrient contributions, however, found that both nitrogen and phosphorus management strategies are required for successful Gulf of Mexico hypoxia mitigation (Adhikari, White, Maiti, & Nguyen, 2015; Liu & Shuting, 2016; Scavia & Donnelly, 2007). Phosphorus management is also of particular importance for management of eutrophication in Minnesota where the landscape dominated by lakes (Boardman, Danesh-Yazdi, Foufoula-Georgiou, Dolph, & Finlay, 2019). In addition, sediment management focuses on particle settling, while phosphorus has both particulate and dissolved forms (Choudhury, Robertson, & Finnigan, 2016; Van der Grift, et al., 2018).

The accumulation of sediment favors the accumulation of particulate phosphorus attached to soil particles. Sediment accumulation can thus contribute to high levels of

STP that may be indicative of easily mobilized phosphorus forms that are transported downstream as DRP (Da-Peng & Huang, 2010; Ellison & Michael, 2006). Of the various phosphorus forms contained within TP, DRP is the main contributor to downstream eutrophication and harmful algal bloom development (Baker, et al., 2014). Within the Lake Erie basin, for example, a decline in TP levels was observed within agricultural tributaries alongside steep increases in DRP (Jarvie, et al., 2017; Smith, King, & Williams, 2015). This trend may be due to the implementation of practices, such as conservation tillage or cover crops, that reduce soil erosion but contribute to macropore development and topsoil phosphorus accumulation (Daryanto, Wang, & Jacinthe, 2017; Rodriques, et al., 2012). These trends highlight the need to characterize agricultural DRP loss mechanisms, to develop effective management strategies, and to quantify current loss rates for appropriate water quality planning.

Much of this work has already been conducted within the Lake Erie Basin located in northern Ohio (Ni, Yuan, & Liu, 2020; Pease, et al., 2018). This previous work contributes to the goals of the Mississippi River/Gulf of Mexico Hypoxia Task Force through identification of target DRP yields and management need, which may serve as a reference for similar management planning in other regions. The Mississippi River/Gulf of Mexico Hypoxia Task Force is a multi-state Environmental Protection Agency initiative set forth to develop comprehensive nitrogen and phosphorus reduction strategies on a state-by-state basis (EPA, 2020). Similar research to that in Ohio is sparse within Minnesota, a region that may have varying agricultural practices, climate and field conditions, DRP yields and management needs to those identified in the Lake Erie Basin.

This study aims to fill this research need within the Upper Mississippi River Basin region of the Greater Mississippi River Basin through analysis of management, water quality and hydrology data collected by the Discovery Farms Minnesota program, a farmer-led program supported by the Minnesota Department of Agricultural. The goal of this program is to gather water quality information from different farming systems in various landscapes across the state (Discovery Farms MN, 2021). Discovery Farms Minnesota data collection includes management factors such as tillage practice, crop type and rotation, fertilizer and manure application and cover crop implementation; and site condition data including STP, soil characteristics, climate and hydrology. Farm management and site condition data collection are paired with water quality and quantity sampling in the form of drain tile and surface flow volume and DRP. Within this study, statistical analysis of these data served to identify baseline DRP yield relative to management and site conditions useful for informing initiatives of the Minnesota Nutrient Reduction Strategy as part of the larger Mississippi River/Gulf of Mexico Hypoxia Task Force (Anderson, Wall, & Olson, 2016).

The overarching project goal is to inform the Minnesota Department of Agriculture monitoring program with specific research objectives being (a) to identify current and target DRP yield characteristics and rates from Upper Mississippi River Basin farmlands to meet Gulf of Mexico nutrient reduction goals, (b) identify dominant site and management mechanisms driving surface and drain tile DRP concentrations and STP and (c) to calculate a phosphorus mass balance to evaluate DRP losses relative to fertilizer application, crop use efficiency and soil phosphorus saturation. Findings from objectives (b) and (c) may be synthesized to inform more realistic target for field loss and strategies on how to reach those targets.

Research questions addressed under objective (a) included: (1). What proportion of the cumulative DRP yield occurs in surface flow verse drain tile flow? (2). What proportion of the total phosphorus (TP) yield occurs as DRP? (3). What is the current mean annual DRP yield from agricultural fields and what is the target DRP yield based on a 45% reduction goal for management initiatives taken at the field scale? (4). What relationships exist between drain tile discharge depth and drain tile DRP yield at the annual scale? (5). What relationships exist between surface runoff depth and surface DRP yield at the annual scale? It was hypothesized that while greater mean DRP FWMC and yield would be observed in surface flow relative to drain tile flow quantitatively, the prominence of DRP drain tile loss would also be demonstrated. It was further hypothesized that weak associations would be observed between DRP yields and runoff depth on an annual scale, highlighting the prominence of storm events and management on DRP loss.

Research questions addressed under objective (b) included: (1). Of the field scale site and management factors including STP, number of tillage passes, manure and fertilizer application rate, cover crop implementation, organic matter content, clay content, pH, rainfall depth, surface runoff and drain tile discharge; which most strongly influenced drain tile DRP flow weighted mean concentration (FWMC)? (2). Of these same field scale site and management factors; which most strongly influenced surface DRP FWMC? (3). Of the field scale site and management factors including number of tillage passes, manure and fertilizer application rate, cover crop implementation, organic matter content, clay content, pH, rainfall depth, surface runoff and drain tile discharge; which most strongly influenced surface STP accumulation? It was hypothesized that STP and total phosphorus application rates would most strongly influence both drain tile and surface DRP FWMC. It was further hypothesized that conservation tillage, in the form of reduced tillage passes, would contribute to higher DRP FWMC and cover crop implementation to increased STP accumulation.

Lastly, it was hypothesized under objective (c.) that (1). evidence would exist supporting the historic accumulation of soil phosphors and its role in DRP loss, (2). That source management practices, such as fertilizer application, would be directly related to DRP loss, and (3). that source management alone, however, would be insufficient to mitigate agricultural DRP losses.

Methods

Study Sites

Farm management data, in addition to surface and subsurface hydrology and nutrient monitoring data was obtained from the Discovery Farms Minnesota program, from seven agricultural research sites across Minnesota (Figure 1, Table 1, Table 2) (Discovery Farms MN, 2021). Within this study we assessed field scale management and site condition factors including fertilizer and manure application rates, tillage, cover crop implementation, STP, organic matter content, clay content, pH, rainfall depth, surface runoff depth and drain tile discharge depth as related to DRP loss on an annual timescale. All sites included within the study were tile-drained over the study period. Edge-of-field farm monitoring stations measured water quantity and quality data automatically when surface or drain tile flow occurred.

We also utilized runoff volume, TP and DRP data, in addition to precipitation from on-site weather stations, to identify mean annual field scale agricultural DRP loss rates. For hydrologic analysis, we monitored total precipitation, drain tile and surface runoff depths by annual water year spanning October 1st through September 30th. We further calculated mean annual FWMCs and cumulative yield in kg ha^{-1} across each water year for use in statistical analysis from DRP concentration samples.

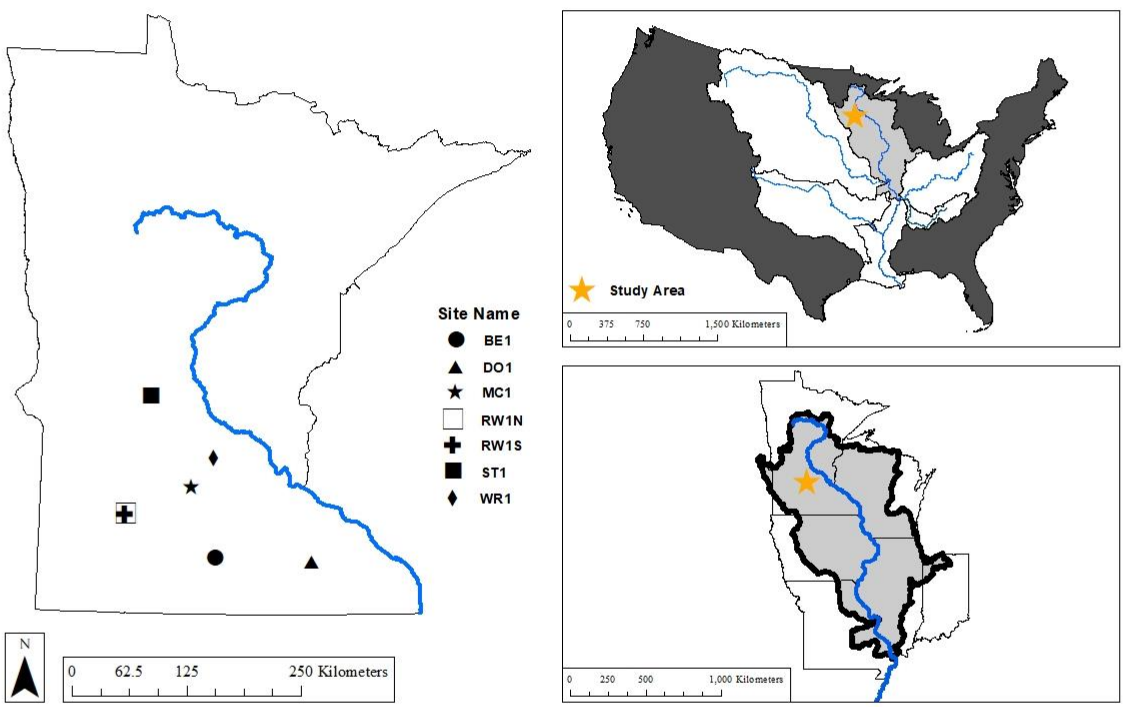

Figure 1. Minnesota field study sites within the context of the greater and Upper Mississippi River Basin drainage areas.

Table 1. Selected characteristics of study field monitoring sites.

Field ID	n	Drainage Area		Agri-Business	Crop Rotation	Soil Type	Cover Crop	Tillage Category
		Surface	Subsurface					
		ha						
BE1	6	5.79	10.6	SG	CS	Silty clay Loam	No	CV
DO1	8	5.63	5.63	SG	CS	Silt loam	No	CV
MC1	3	24.52	24.52	G	CS	Clay loam	No	CV
RW1N	3	5.06	5.06	G	CS	Loam	Yes	CN
RW1S	3	4.13	4.13	G	CS	Loam	Yes	CN
ST1	6	11.41	9.79	D	CA	Loam	No	CV
WR1	8	9.67	9.67	D	CA	Clay loam	Yes	CN

Note: n, water years monitored; SG, swine finishing and grain; G, grain; D, dairy; CS, corn-soybean; CA, corn-alfalfa; CV, conventional tillage; CN, conservation tillage.

Table 2. Mean study site phosphorus application and soil test phosphorus (STP) values.

Field ID	Mean Annual Fertilizer Application (kg ha^{-1})	Mean Annual Manure Application (kg ha-1)	Mean Soil Test Phosphorus (mg l-1)
BE1	2.93	6.04	24.60
DO1	8.02	23.58	63.20
MC1	14.25	0.00	15.00
RW1N	38.66	0.00	30.00
RW1S	38.66	0.00	44.00
ST1	0.00	12.08	49.50
WR1	0.00	33.54	49.83

Data Analysis

Hydrology and Phosphorus Loss

Paired surface and drain tile hydrology and nutrient monitoring data was available from seven field study sites spanning thirty-seven data years. These seven field sites included BE1, DO1, MC1, RW1N, RW1S, ST1 and WR1 as monitored by the Discovery Farms Minnesota program (Figure 1, Table 1, Table 2) (Discovery Farms MN, 2021). Data distribution required non-parametric statistical approaches based on the Shapiro-wilk test (Shaphiro & Wilk, 1965). As such, a Mann-Whitney U Test was utilized in order to test the hypothesis that greater mean DRP FWCM and yield would be observed in surface verse drain tile flow. Assumptions under this test include a continuous dependent variable, two categorical independent variables, independence of observations and variables with parametric distributions (McKnight & Najab, 2010).

In order to test the hypothesis that weak associations would be observed between DRP yields and runoff depth on an annual scale, we used Pearson's Correlation to determine the strength and significance of linear association between both surface and drain tile discharge depth and DRP yields. This test determines if the linear relationship within sample data accurately represents that of the population, under the assumptions that outliers are absent and that data is normally distributed and measured on an interval scale (Obilor & Amadi, 2018).

Linear Regression Modeling

We conducted multiple linear regression modeling for both DRP surface and drain tile FWMC as related to STP, number of tillage passes, manure and fertilizer application rate, cover crop implementation, organic matter content, clay content, pH, rainfall depth, surface runoff and drain tile discharge. We also conducted multiple linear regression modeling for STP accumulation as related to number of tillage passes, manure and fertilizer application rate, cover crop implementation, organic matter content, clay content, pH, rainfall depth, surface runoff and drain tile discharge. This methodology was utilized to test the hypotheses present under objective (b) of this work. It was hypothesized that STP and cumulative phosphorus application rates would most strongly influence both drain tile and surface DRP FWMC. It was further hypothesized that conservation tillage, in the form of reduced tillage passes, would contribute to higher DRP FWMC and cover crop implementation to increased STP accumulation.

The goal of this modeling was to mathematically evaluate the best fit model utilizing the Akaike information criterion (Chaurasia & Harel, 2012). Independent variables were defined as site-specific field conditions or management practices and dependent variables were defined as DRP FWMC. An individual record was characterized as one water year sample or as an annual field or management characteristics. Multiple linear regression equations reflect the form displayed in Equation 1:

$$(1).\ yi = \beta 0 + \beta 1x1i + \beta 2x2i + \cdots + \beta pxpi + e\mathrm{i}$$

where y_i is the dependent variable as DRP surface or subsurface losses in kg ha^{-1} yr^{-1}; β_0 is the intercept, β_{pxpi} is the product of the independent variable coefficient and value, e_{i} is the coefficient of the error term (Tranmer, Murphy, Elliot, & Pampaka, 2008).

Variables included within the analysis were selected based upon data availability and collinearity analysis (Table 3). Dummy variables were utilized in place of categorical independent variables with defined reference categories. We performed collinearity analyses to identify correlation between independent variables utilizing an absolute maximum value of 0.75 with the Spearman’s rank correlation coefficient (ρ) and the simple regression analysis coefficient of determination (R^2) for numerical to numerical and numerical to categorical variables, respectively (Ni, Yuan, & Liu, 2020; Spearman, 1904). We checked model assumptions including normality, homoscedasticity and independence utilizing residual diagnostics and the Shapiro-Wilk test (Shaphiro & Wilk, 1965).

Table 3. Site condition regression model variables.

Variable	Description	Reference Category
Rainfall Depth	Total water year rainfall depth	NA
Runoff Depth	Total water year surface or drain tile runoff	NA
Fertilizer Application	Annual rate of P_2O_5 application as fertilizer	NA
Manure Application	Annual rate of P_2O_5 application as manure	NA
Cover Crop Implementation	Annual rye cover crop presence or absence	Absent
Soil Test Phosphorus	Plant available soil phosphorus	NA
Soil Organic Matter	Fraction of soil organic component	NA
pH	Soil acidity or basicity	NA
Clay Content	Fraction of soil clay component	NA

Phosphorus Mass Balance

All 37 site-years from each of the seven field sites was utilized to develop a field-scale phosphorus mass balance. We utilized literature based atmospheric deposition values, land-owner provided fertilizer and manure application rates and crop yields; and field monitored surface and drain tile DRP yields as displayed in Equation 2:

(2). Annual net phosphorus balance (kg ha-1) = [atmospheric deposition + fertilizer application] – [plant phosphorus removal + surface DRP yield + drain tile DRP yield]

where all inputs and outputs are expressed in kg ha^{-1} yr^{-1}. Atmospheric deposition was defined at a rate of 0.14 ± 0.10 kg ha^{-1} yr^{-1}as utilized by Pease et al. (2018). Plant phosphorus removal of elemental soil phosphorus was calculated utilizing ratios of 0.07

kg bushel^{-1} and 0.16 kg bushel^{-1} for corn and soybean, respectively (Jacobson, David, & Drinkwater, 2011). Elemental phosphorus application was calculated based upon the amount of P_2O_5 applied in monoammonium phosphate fertilizer or manure with a ratio of 0.44 applied based upon phosphorus and oxygen atomic weights.

In order to test the hypothesis that source management practices, such as fertilizer application, would be directly related to DRP loss, we again utilized a Pearson's correlation coefficient to determine the strength and significance of linear association between annual net phosphorus balance and cumulative DRP water losses and between cumulative phosphorus inputs and total DRP water losses (Obilor & Amadi, 2018). A lack of significant correlation between these variables would serve as evidence for the role of legacy phosphorus saturation in DRP water loss.

Results

Dissolved Reactive Phosphorus Concentrations

Mean DRP FWMC across 37 study years at seven field research sites was higher in surface flow (M = 0.74 mg l^{-1}, SD = 0 .83 mg l^{-1}) relative to drain tile flow (M = 0.09 mg l^{-1}, SD = 0.09 mg l^{-1}), ($Z(36) = -6.60$, $p < .001$). Mean annual FWMC in surface flow was 0.42 ± 0.32 mg l^{-1}, 0.56 ± 0.33 mg l^{-1}, 0.34 ± 0.05 mg l^{-1}, 1.82 ± 1.67 mg l^{-1}, 1.73 ± 1.33 mg l^{-1}, 0.54 ± 0.08 mg l^{-1}, and 0.69 ± 0.49 mg l^{-1} at sites BE1, DO1, MC1, RW1N, RW1S, ST1 and WR1, respectively. Mean annual FWMC in drain tile flow was 0.03 ± 0.004 mg l^{-1}, 0.02 ± .01 mg l^{-1}, 0.07 mg l^{-1} ± 0.05, 0.05 mg l^{-1} ± 0.04, 0.07 ± 0.03 mg l^{-1}, 0.14 ± 0.05 mg l^{-1}, and 0.20 ± 0.10 mg l^{-1} at sites BE1, DO1, MC1, RW1N, RW1S, ST1 and WR1, respectively. Mean annual FWMC in surface flow at sites under a corn-soybean rotation (BE1, DO1, MC1, RW1N, RW1S) was 0.89 ± 1.00 mg l^{-1} and 0.04 ± 0.03 mg l^{-1} in drain tile flow. Mean annual FWMC in surface flow at sites under a corn-alfalfa rotation (ST1, WR1) was 0.63 ± 0.38 mg l^{-1} and 0.18 ± 0.09 mg l^{-1} in drain tile flow (Figure 2, Figure 3).

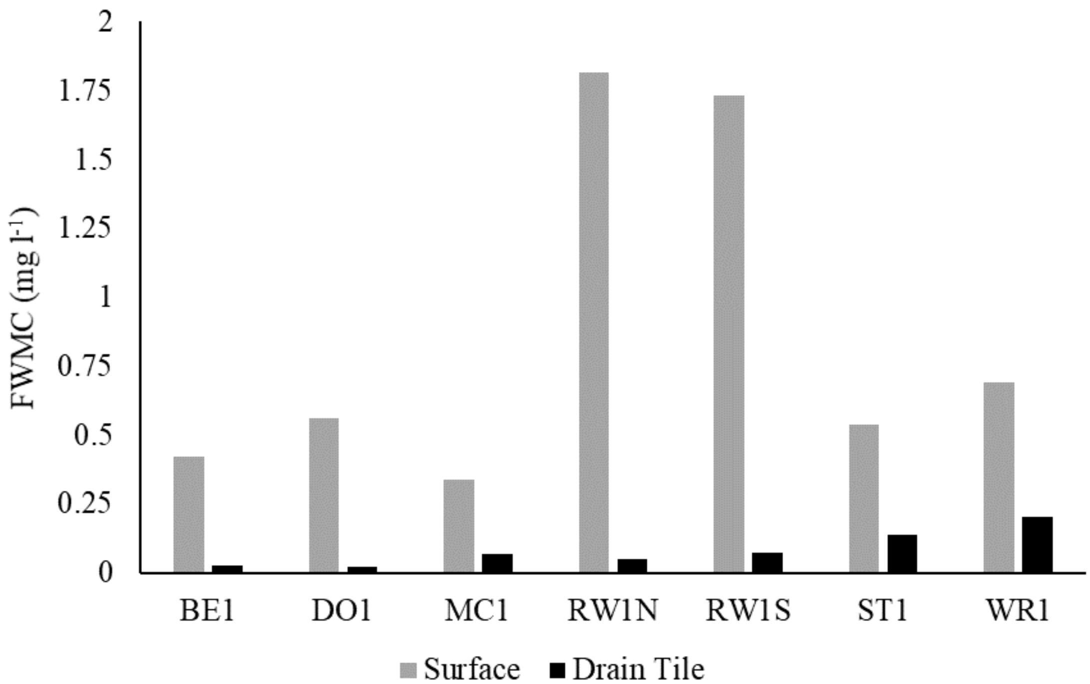

Figure 2. Mean edge-of-field dissolved reactive phosphorus (DRP) flow weighted mean (FWMC) concentrations.

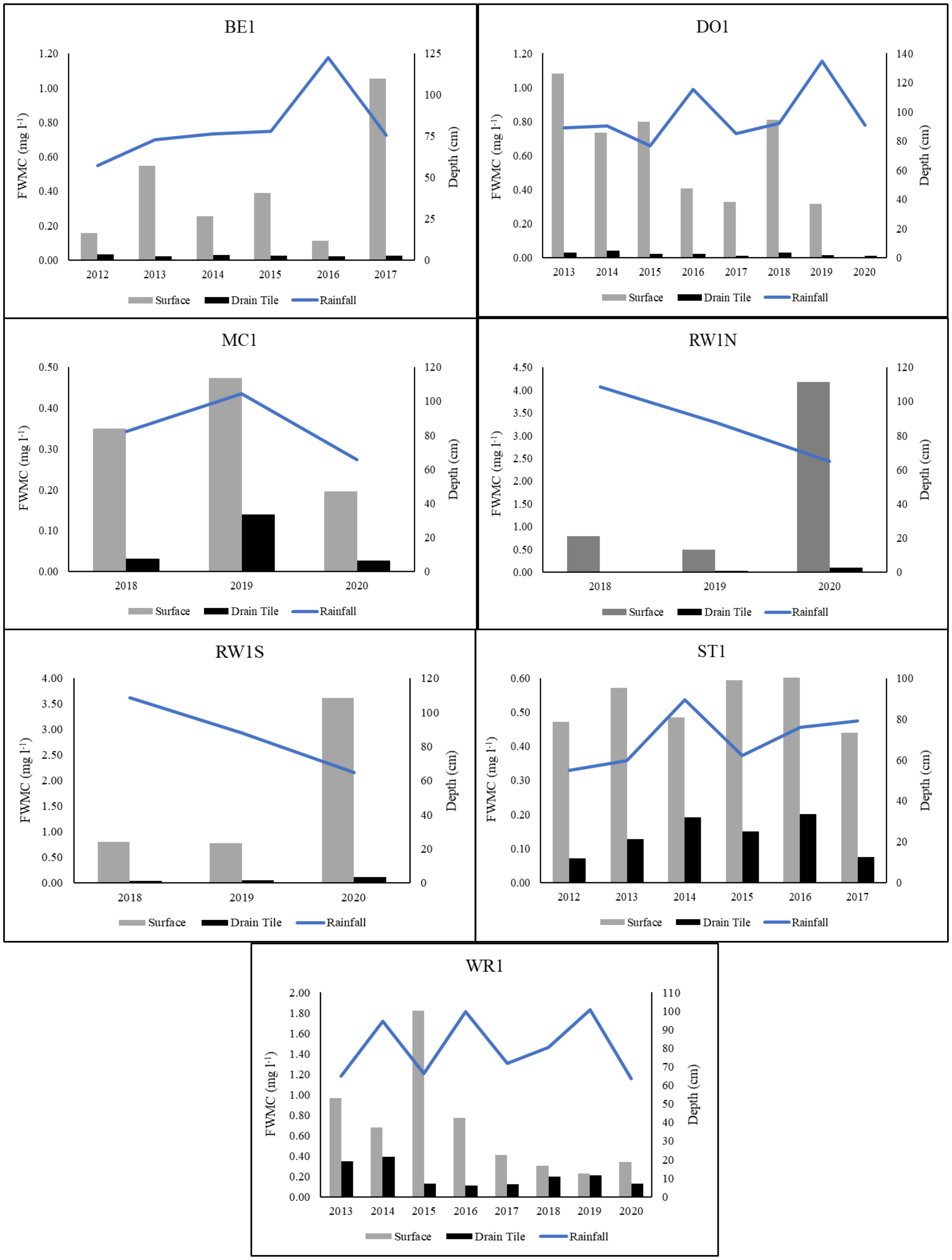

Figure 3. Annual edge-of-field DRP FWMC by field site, present with cumulative annual rainfall depth

Dissolved Reactive Phosphorus Yields

Cumulative mean DRP yield across 37 study years at seven field research sites was 0.48 ± 0.41 kg ha^{-1} yr^{-1}, with higher yields occurring in surface flow (M = 0.37 kg ha^{-1} yr^{-1}, SD = 0.33 kg ha^{-1} yr^{-1}) relative to drain tile flow (M = 0.11 kg ha^{-1} yr^{-1}, SD = 0.13 kg ha^{-1} yr^{-1}). ($Z(36) = -3.87$, $p < .001$). Forty-seven percent of the cumulative mean annual TP yield occurred in the dissolved form, with DRP comprising 67% of the mean annual drain tile TP yield and 43% of the mean annual surface flow TP load (Figure 4).

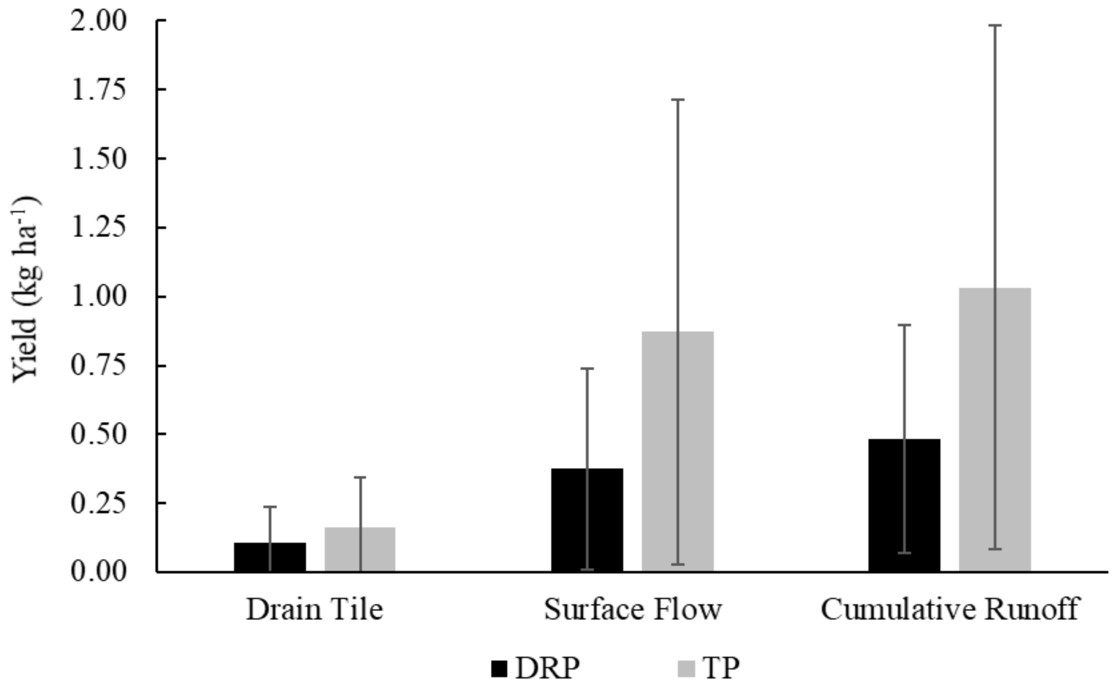

Figure 4. Mean fractions of the total phosphorus (TP) load occurring as DRP across all field sites.

Mean proportion of the cumulative annual DRP yield occurring in drain tile was 32 ± 24% (range: 3 to 100%) and 68 ± 24% (range: 0 to 97%) in surface flow (Figure 5). The greatest volume and proportion of DRP surface yields occurred at sites RW1N and RW1S which is present with management practices that differ from those common at other study field sites, including high fertilizer application rates, conservation tillage and cover crop implementation. The greatest volume and proportion of DRP in drain tile flow occurred at site MC1 which differed from other sites with similar crop rotation, tillage and fertilizer practices (DO1, BE1) in that it had a corn soybean rotation without the presence of swine finishing. Lastly, sites ST1 and WRI where those under a corn alfalfa rotation and in general saw higher relative proportions of DRP loss in drain tile relative to sites under a sorn-soybean rotation (Figure 5, Figure 6, Table 4).

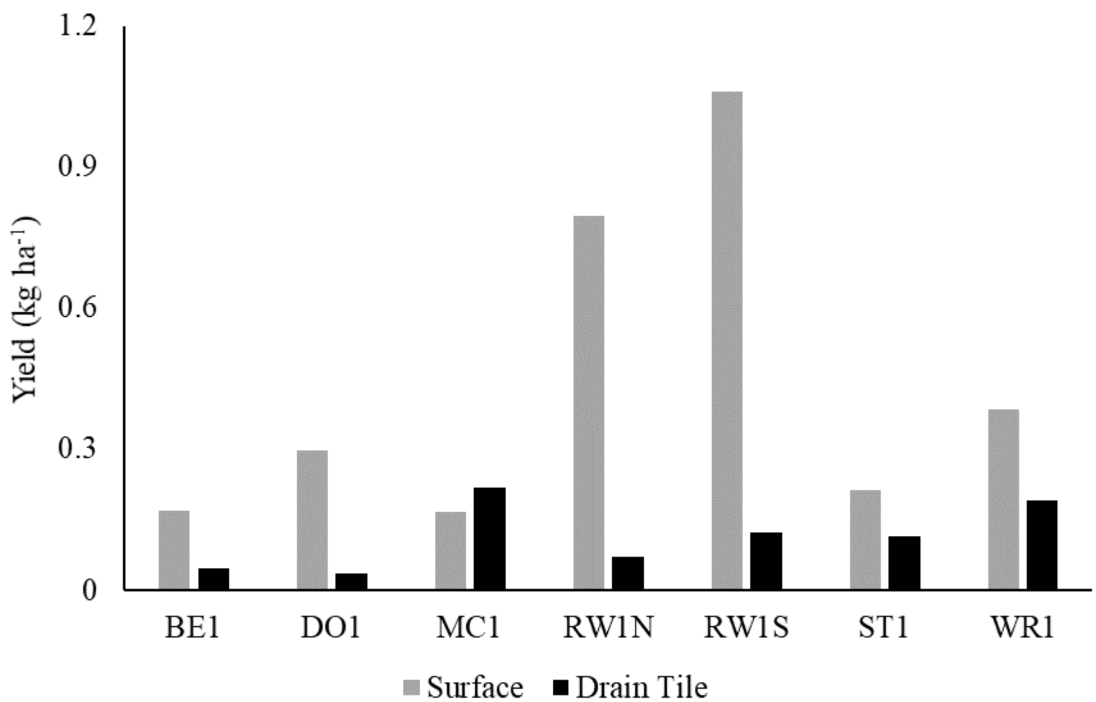

Figure 5. Mean annual edge-of-field DRP yields.

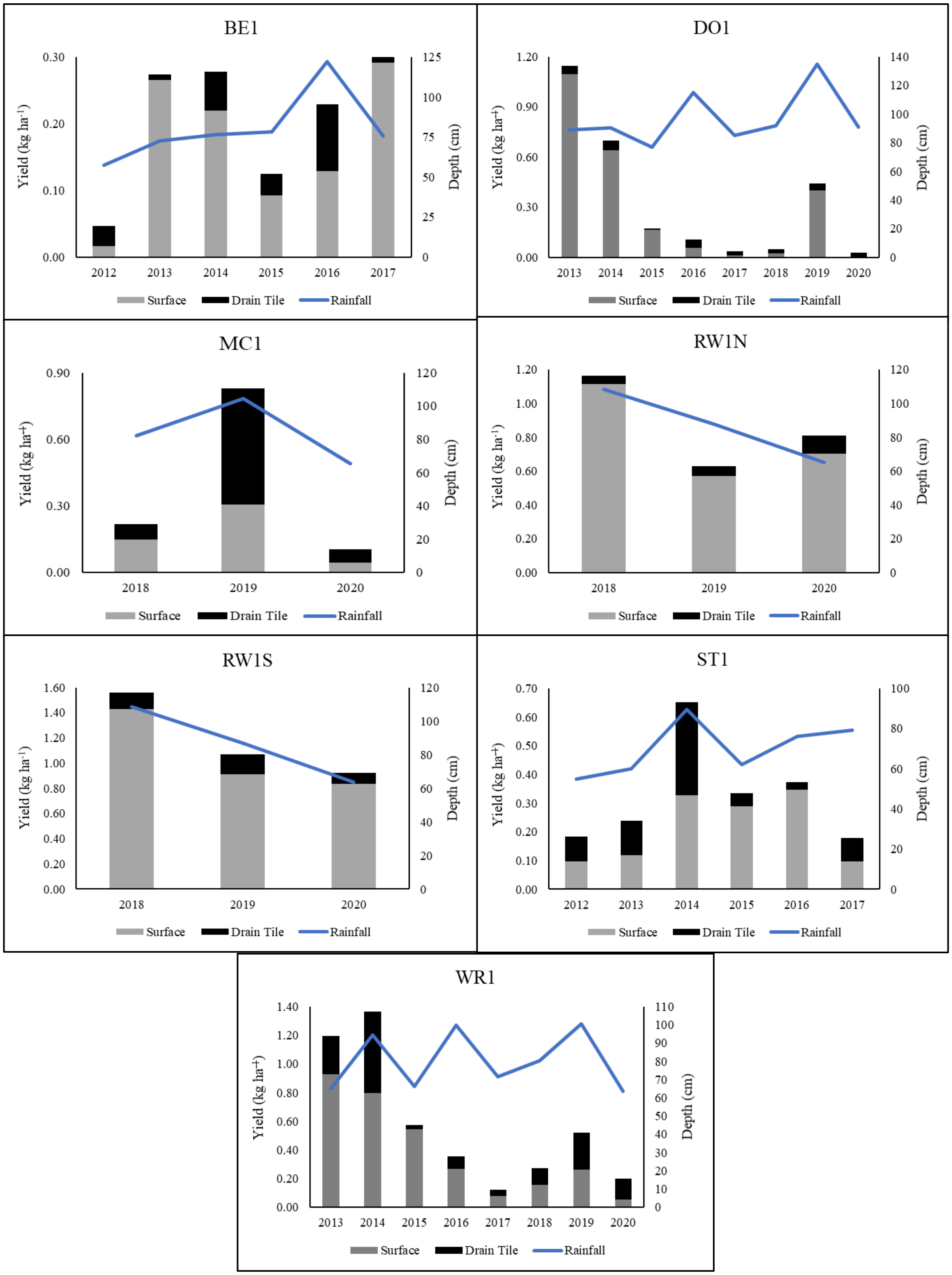

Figure 6. Annual DRP surface and drain tile yield proportions by field site, present with cumulative annual rainfall depth.

Table 4. Mean annual edge-of-field DRP surface and drain tile yields and proportions

Site	Mean Yields (kg ha^{-1} ha^{-1})		Yield Proportion (%)	
	Surface Flow	Drain Tile Flow	Surface Flow	Drain Tile Flow
BE1	0.17	0.05	72	28
DO1	0.30	0.04	62	38
MC1	0.17	0.22	50	50
RW1N	0.80	0.07	91	9
RW1S	1.18	0.12	89	11
ST1	0.21	0.11	64	36
WR1	0.39	0.19	63	37

Precipitation and Discharge Distribution

Across all study sties and study years, annual drain tile discharge (M = 15.67 cm, SD = 4.60 cm) was greater than surface runoff (M = 5.66 cm, SD = 10.36 cm), $Z(36) = 4.70$, $p < .001$. The mean proportion of annual total runoff occurring as drain tile flow was 73% (range: 21 to 100%) with a mean proportion of 27% (range: 0 to 79%) occurring as surface flow. Annually, the mean percentage of rainfall becoming tile flow was 18% (range: 2 to 38%) with a mean percentage of 3% (range: 2 to 5%) occurring as surface flow.

Runoff Yield Relationships

At site BE1 a significant positive association was identified between drain tile discharge depth and drain tile DRP yield at the 5% level ($r(6) = 0.98$, $p < .001$). At site DO1 a significant positive association was identified between surface runoff depth and surface DRP yield at the 5% level ($r(8) = 0.81$, $p = .015$). At site ST1 a significant positive association was identified between drain tile discharge depth and drain tile DRP yield at the 5% level ($r(6) = 0.83$, $p = .043$), and between surface runoff depth and surface

DRP yield at the 5% level (r(6) = 0.95, p = .004). Significant positive associations between both surface and drain tile discharge depths and surface and drain tile DRP yields were also identified at site WR1 at the 5% level (r(8) = 0.68, p = .065; r(8) = 0.82, p = .014). No significant associations were identified between drain tile runoff discharge and drain tile DRP yield or between surface runoff depth and surface DRP yield at sites RW1N and RW1S. Site MC1 was excluded from this analysis due to small sample size.

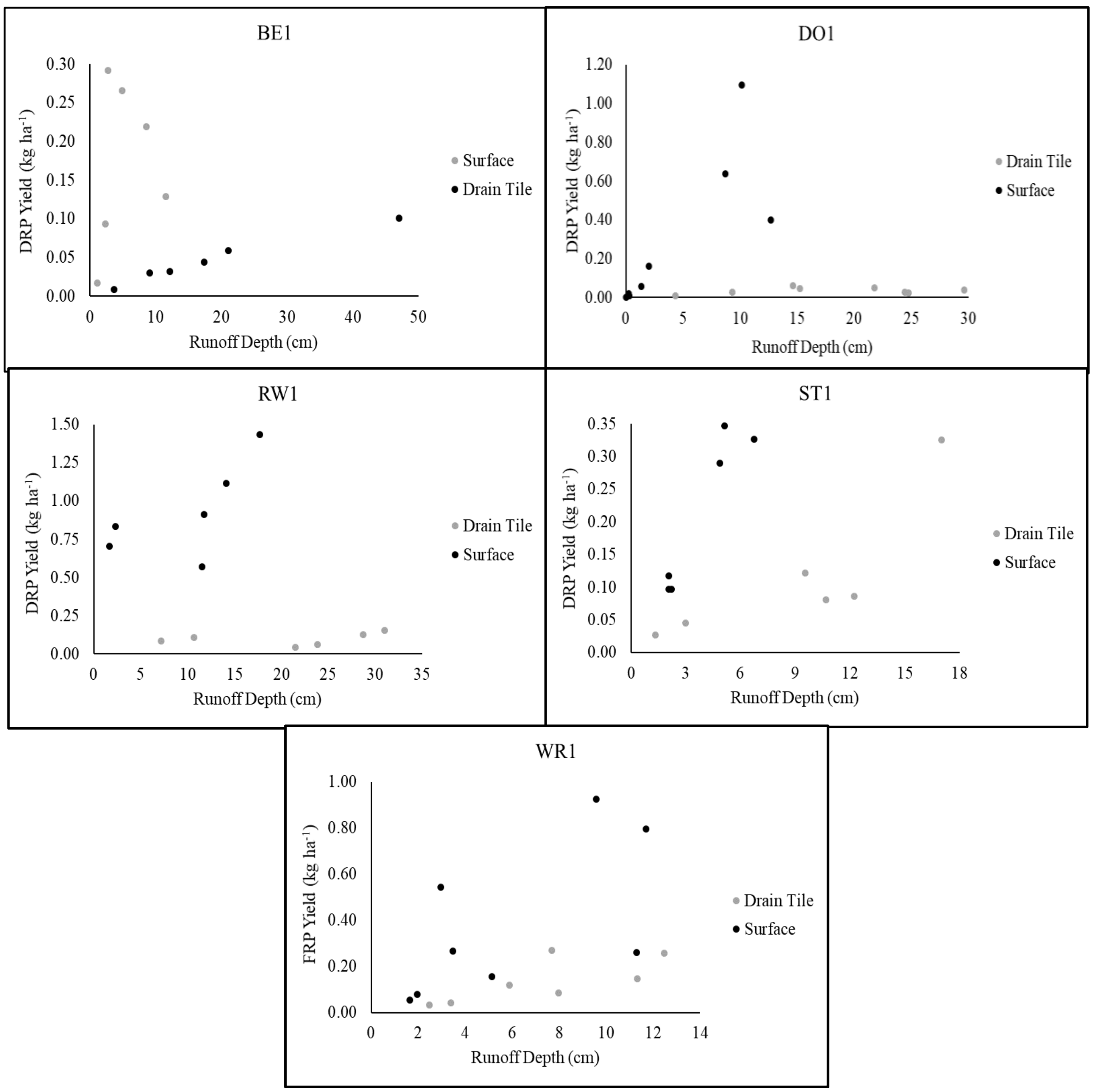

Figure 7. Annual DRP yield as a function of runoff depth.

Drivers of Drain Tile Dissolved Reactive Phosphorus Concentration

The model for drain tile DRP flow weighted mean concentration included parameters for manure application rate, number of tillage passes and STP, R^2 = .74, F(3, 34) = 32.66, p = < .001. Direct relationships were identified between manure application rate and increased drain tile DRP FWMC (Table 5). Alternately, increased DRP FWMC was observed in associated with decreased number of tillage passes. Model assumptions for the linear regression were verified for normality, homoscedasticity and independence (see diagnostic plots in Appendix 1, Figure 23).

Table 5. Multiple linear regression model of drain tile dissolved reactive phosphorus (DRP) concentration drivers.

Variables	Coefficient	Standard Error	p-value
Intercept	0.21	0.03	0.000
Manure Application Rate	0.00	0.00	0.000
Number of Tillage Passes	-0.06	0.01	0.000
Soil Test P ppm	0.00	0.00	0.000

Drivers of Surface Dissolved Reactive Phosphorus Concentration

The model for surface DRP flow weighted mean concentration included parameters for cumulative phosphorus application rate and STP, R^2 = .43, F(2, 34) = 13.15, p = < .001. A direct relationship was identified between total phosphorus application rate; defined as the cumulative manure and fertilizer application rate, and increased drain tile DRP FWMC (Table 6). Model assumptions for the linear regression were verified for normality, homoscedasticity and independence (see diagnostic plots in Appendix 1, Figure 24).

Table 6. Multiple linear regression model of surface DRP concentration drivers.

Variables	Coefficient	Standard Error	p-value
Intercept	0.30	0.15	0.052
Cumulative Phosphorus Application	0.00	0.00	0.002
Soil Test P ppm	0.01	0.00	0.098

Drivers of Soil Test Phosphorus Accumulation

The model for STP accumulation included manure application rate, number of tillage passes, organic matter content, clay content, soil pH and cover crop implementation, $R^2 = .65$, $F(7, 44) = 11.62$, $p = < .001$. Direct relationships were identified between STP accumulation and manure application rate, organic matter content and cover crop implementation. Alternately, STP accumulation was inversely related to the number of tillage passes, clay content and soil pH (Table 7). Model assumptions for the linear regression were verified for normality, homoscedasticity and independence (see diagnostic plots in Appendix 1, Figure 25).

Table 7. Multiple linear regression model for drivers of STP accumulation.

Variables	Coefficient	Standard Error	p-value
Intercept	105.19	34.13	0.004
Manure Application Rate	0.29	0.04	0.000
Number of Tillage Passes	-14.18	2.68	0.000
Organic Matter Content	7.33	2.51	0.005
Clay Content	-83.21	28.59	0.006
Soil pH	-9.40	4.05	0.025
Cover Crop Implementation	15.29	7.94	0.060

Phosphorus Mass Balance

Data analysis from 37 study years at seven agricultural field research sites, showed a mean annual net negative phosphorus balance of -15.44 ± 27.88 kg ha^{-1}, thus identifying greater phosphorus outputs relative to inputs. Annual phosphorus application rates averaged 9.63 ± 18.92 kg ha^{-1}, with mean crop removal rates exceeding this value at 40.02 ± 21.85 kg ha^{-1}. Mean phosphorus losses in site runoff were 0.37 ± 0.37 kg ha^{-1} and 0.11 ± 0.13 kg ha^{-1}, for surface flow and drain tile flow, respectively (Figure 8).

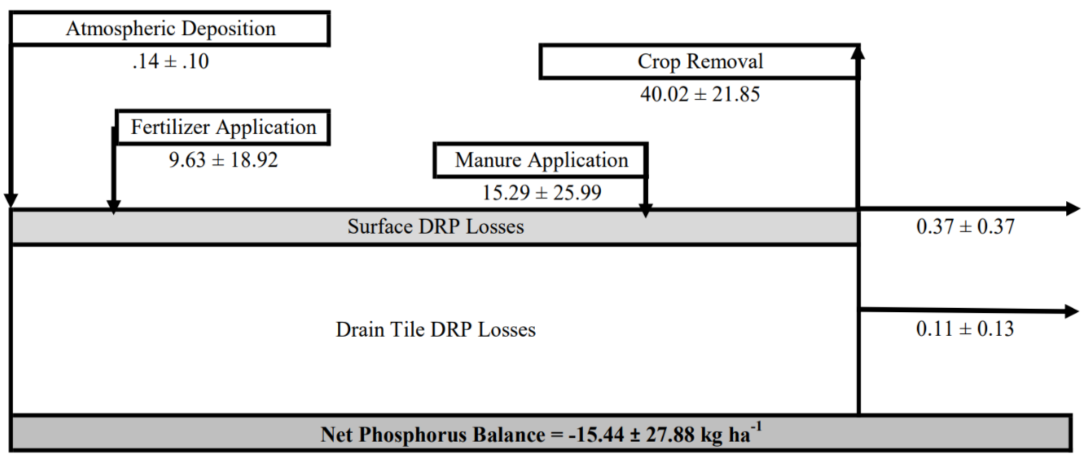

Figure 8. Elemental phosphorus balance across edge-of-field monitoring sites. All units are expressed in kg ha^{-1} yr^{-1}.

Mean STP values were 24.60, 63.20, 15.00, 37.00, 37.00, 49.50, 49.83 mg l^{-1} at sites BE1, DO1, MC1, RW1N, RW1S, ST1, and WR1, respectively (Table 2). STP value recommendations for corn and soybean in the region range from 16 to 20 mg l^{-1}, and 21 to 25 mg 1^{-1} (Mallarino, Sawyer, & Barnhart, 2013). As such, site MC1 is the only site with soil phosphorus saturation ratios below regional guidelines and the only site not exceeding associated fertilizer application rate recommendations. (Table 8).

Mean annual net positive phosphorus balances of 5.68 and 5.36 kg ha^{-1} were found at field sites RW1N and RW1S, respectively. RW1N and RW1S were characterized with the highest annual fertilizer application rates of 38.66 kg ha^{-1} as well as the highest occurrence of surface DRP loss, and were the only sites with fertilizer application in exceedance of crop yield.

Total edge-of-field phosphorus loss was not correlated with final mass balances at any field site, providing evidence for the role of legacy phosphorus saturation as outputs often exceeded inputs. The role of legacy phosphorus is further demonstrated as the majority of sites demonstrated no significant relationship between total phosphorus inputs and water losses. WR1 was the only site with a significant positive association identified at the 5% level, between total phosphorus inputs and total edge-of-field losses ($r(8) = 0.79$, $p = 0.021$).

Table 8. Annual mean phosphorus mass balance by field site.

Site	Inputs			Outputs			Mass Balance
	Atmospheric Deposition	Fertilizer	Manure	Surface	Drain Tile	Crop Yield	
BE1	0.14	2.93	6.04	0.17	0.05	27.30	-18.40
DO1	0.14	8.02	23.58	0.30	0.04	31.54	-0.14
MC1	0.14	14.25	0.00	0.17	0.22	24.83	-10.82
RW1N	0.14	38.66	0.00	0.80	0.07	32.26	5.68
RW1S	0.14	38.66	0.00	1.06	0.12	32.26	5.36
ST1	0.14	0.00	12.08	0.21	0.11	55.19	-43.29
WR1	0.14	0.00	33.54	0.39	0.19	58.16	-25.06
	0.14	9.63	15.29	0.37	0.11	40.02	-15.44

Note: all units expressed in kg ha^{-1}.

Discussion

Cumulative mean edge-of-field DRP yield across study sites and years was 0.49 kg ha^{-1} yr^{-1}, as consistent with finding from other studies in the region. Jacobson et al. (2011), noted annual DRP losses of 0.42 kg ha^{-1} within the Mississippi River Basin. The U.S. Geologic Survey reported annual TP losses of .88 kg ha^{-1} within the Minnesota region of the Upper Mississippi River Basin (Robertson & Saad, 2019). Within this study, the fraction of the cumulative surface and subsurface TP load occurring as DRP was identified as 47%, which would be equivalent to a DRP loading rate of 0.41 kg ha^{-1} yr^{-1} from U.S. Geologic Survey TP load estimates.

The Minnesota Nutrient Reduction Strategy, set in place as part of a larger Lake Winnipeg Basin Gulf of Mexico hypoxia and nutrient reduction initiative, calls for a 45% reduction in phosphorus loading prior to 2025 (Anderson, Wall, & Olson, 2016). Based on this reduction goal and current loading estimates, an annual DRP target load of 0.27 kg ha^{-1} yr^{-1} can be set for agricultural fields within the Minnesota region of the Upper Mississippi River Basin drainage area. This DRP loading target is comparable to a value of 0.29 kg ha^{-1} yr^{-1} set within the Maumee River Basin of Ohio for reducing phosphorus loading to Lake Erie (Pease, et al., 2018).

Up until the 1980s, agricultural phosphorus management initiatives in the United States north central region, focused primarily on surface flow, with subsurface losses often considered to be negligible (Baker, Campbell, Johnson, & Hanway, 1975; Logan, Randall, & Timmons, 1980). The importance of subsurface phosphorus loss pathways, however, began to gain prominence during the 1990s (King, et al., 2015; Sims, Simard, & Joern, 1998). Policy initiatives related to DRP subsurface loss began occurring in some states, including Ohio, during the 2010s and policy initiatives still lacking in many states, including Minnesota, at the present time. Within this study, 32% of the cumulative annual mean DRP load occurred through drain tile with target loads of 0.27 kg ha^{-1} yr^{-1} exceeded in four of 37 study years through drain tile loss alone. Additionally, cumulative surface and subsurface DRP losses exceeded target loads in 21 of 37 study years. These findings demonstrate the need for greater phosphorus loss management practices that address both surface and drain tile DRP losses. While quantification and characterization of agricultural phosphorus losses and target loads are necessary for water quality implementation, a working knowledge of field condition and management influence on phosphorus transport is also necessary.

Dominant factors driving drain tile DRP losses included manure application rate, soil test phosphorus and number of tillage passes. Decreased number of tillage practices was significantly associated with increased drain tile DRP concentrations. Although effective for soil erosion control, conservation tillage may promote enhanced macropore development and phosphorus mineralization from crop residue and topsoil phosphorus stratification, thus contributing to greater DRP losses (Baker, Johnson, Confesor, & Crumrine, 2017; Daryanto, Wang, & Jacinthe, 2017; Jarvie, et al., 2017). Further, practices that promote preferential flow path development, particularly conservation tillage, should be avoided in high clay content soils (Shipitalo, Dick, & Edwards, 2000). While clay content may increase phosphorus retention it also can promote macropore and preferential flow development through aggregation (Djodjic, Bergstrom, Ulen, & Shirmohammadi, 1999; King, et al., 2015). These flow paths serve to decrease soil water holding volume, thus increasing drain tile flow volume and phosphorus loading.

Dominant factors driving surface DRP FWMC included cumulative phosphorus application rate and STP. Within this study, greater fertilizer application rates were associated with higher surface DRP FWMC. Results of a phosphorus mass balance identified high crop use efficiency and the potential for reduced fertilizer application, as crop removal rates far exceeded phosphorus inputs. Fertilizer application across study sites and years was predominately in the liquid form. Results were consistent with those found by Peterson et al. (2017), who documented greater phosphorus exports relative to imports within an agricultural watershed located in south-central Minnesota. Within the study area, crop yields have increased significantly since the 1970's alongside more efficient phosphorus inputs (USDA, 2021). These historic and less efficient management activities likely contributed to modern soil phosphorus accumulation and leeching. Results from both studies, however, indicate that previously stored legacy phosphorus is now being mined from cropland soils (McCrackin, et al., 2018; Zhu, Li, & Whelan, 2018).

Within this study specifically, a significant correlation was not identified between annual net site DRP loss and net annual phosphorus balance, providing evidence for the role of legacy phosphorus (Pease, et al., 2018). In order to meet target DRP yields, however, both source and edge-of-field management practices, such as treatment wetlands or detention basins, will be required until soil phosphorus saturation is equivalent to crop yield requirements. This is demonstrated within the study mass balance equation, as both surface and subsurface DRP losses are still occurring despite greater crop uptake relative to fertilizer application.

STP, found as a driver of both drain tile and surface DRP FWMC within this study, may serve as a proxy for legacy phosphorus (McDowell, Sharpley, Brookes, & Poulton, 2001). The mean annual STP value across study sites and years was 39 mg l^{-1}, with average fertilizer application rates of 14.65 kg ha^{-1} greatly exceeding regional recommendations for corn and soybean grain production (Mallarino, Sawyer, & Barnhart, 2013). Monitoring of STP values, as such, may prove as a useful tool for identifying relative DRP loss risks in agricultural fields, for prioritizing management practice implementation and for reducing phosphorus inputs (Zheng & Zhang, 2014).

Dominant factors driving STP accumulation included manure application rate, number of tillage passes, organic matter content, clay content, soil pH and cover crop implementation. Cover crop use has also been found to increase DRP loss as a results of winter crop decay enhanced by winter freeze thaw cycles and release of DRP preserved in soil as a result of erosion reduction (Cober, Macrae, & Van Eerd, 2019; Sharpley, et al., 2013). Therefore, cover crop harvesting, rather than senescence or herbicide termination methodologies may further aid in obtaining DRP reductions from cover crops which may also serve as biofuel or livestock feed (Blanco-Canqui, et al., 2020).

Increased soil organic matter content was associated with increased STP accumulation. All sites included within this analysis were characterized by mineral soils. Conflicting soil organic matter content and DRP loss relationships have been documented, as soil organic matter content may expediate DRP loss through increased macropore occurrence but may also increase phosphorus adsorption capacity (Franzluebbers, 2011; Kang, Hesterberg, & Osmond, 2009). Soil organic matter content and soil phosphorus saturation relationships may also vary between mineral and organic soil types (Guppy, Menzies, Moody, & Blamey, 2005; Yang, Chen, & Yang, 2019) Greater research is required to further document the influence of soil organic matter content under site specific management conditions and soil types.

Alternately, STP accumulation was found to decrease in association with increased soil clay content. Few studies exist that document the relationship between clay content and DRP soil retention in the regions, but findings were consistent with those reported by Ni et al. (2020) within the Lake Erie Basin. Clay soils may contribute to phosphate adsorption due to greater specific surface area than other soil types, thus contributing to soil phosphorus bioavailability (Fang, Cui, He, Huang, & Chen, 2017).

Various cropping rotations have been found to carry influence on DRP loss in different ways. This study observed greater mean annual DRP FWMC in surface flow at sites under a corn-soybean rotation and greater mean annual DRP FWMC in drain tile flow at sites under a corn-alfalfa rotation. Greater drain tile DRP losses associated with corn-alfalfa silage systems may be their association with dairy and beef enterprises that contribute to additional system-wide phosphorus in the form of manure which is typically injected in liquid form (Ball, Murray, Lapen, Topp, & Bruin, 2012). Crop rotations involving soybean and corn may also be beneficial for DRP loss management as lower fertilizer application under this crop type are typically lower, thus reducing cumulative system inputs (Saadat, Bowling, Frankenberger, & Kladivko, 2018).

Fertilizer application rates in excess of crop requirements however or varying tillage categories, may contribute to higher observed DRP yields at sites under a corn-soybean rotation as observed within this study. Only one site each, under a corn-soybean rotation were observed to have statistically significant correlations between annual surface runoff depth and annual DRP yield and between annual drain tile depth and annual DRP yield. As runoff volume is a prominent factor in nutrient yield calculations, this statistically anomaly provides evidence for the prominence of management factors, notably reduced tillage or fertilizer application; and storm characteristics on DRP loss.

Conclusion

Study results identified a target DRP yield of 0.27 kg ha^{-1} yr^{-1} for meeting nutrient reduction goals outlined as part of the Mississippi River/Gulf of Mexico Hypoxia Task Force for the state of Minnesota in the Upper Mississippi River Basin. Through analysis of a multi-site phosphorus mass balance, findings demonstrate the need for joint source and edge-of-field management actions to meet target DRP loads. This includes balancing fertilizer application with crop uptake alongside phosphorus control structures or controlled drainage implementation. Baseline field monitoring data analysis also highlighted the need to address both surface and drain tile DRP losses.

Dominant drivers of drain tile DRP loss were identified as manure application rate, number of tillage passes and STP. Dominant drivers of surface DRP loss were identified as cumulative phosphorus application rate and STP. Dominant drivers of STP accumulation included manure application rate, number of tillage passes, organic matter content, clay content, soil pH and cover crop implementation. Additional resultant DRP management implications included STP monitoring, avoidance of conservation tillage under high clay conditions, use of cropping systems including soybean, and cover crops implementation with harvest termination methods. Identified DRP management strategies may aid meeting phosphorus reduction goals identified within the Minnesota Nutrient Reduction Strategy and greater Hypoxia Task Force Nutrient Reduction Strategy.

Although this study serves to characterize a number of DRP loss mechanisms and management strategies, DRP losses remain highly variable and dependent on numerous site and time specific factors. One unanswered question, that is unique to the study region, includes the impact of frozen soils and snowmelt runoff. In particular, practices that help reduce phosphorus during non-frozen time period, notably no-till and cover crops, may be making the problem worse during the snowmelt time period. In addition, the magnitude of losses occurring during storm events as compared to annual base flow was not addressed within this study, and may account for a large proportion of phosphorus losses. Lastly, greater research is required on site-specific factors including drainage design, soil phosphorus saturation, organic verses mineral soils, and fertilizer application rates or methods.

Chapter 2: Cover crops reduce subsurface nutrient loads to downstream management practices, improving system performance and cost-effectiveness

Summary

Cover crop adoption has expanded rapidly in the Midwestern United States in recent years and has been shown as one of the most cost-effective practices for reducing agricultural nutrient losses. In particular, cover crops may aid in the prevention of nutrient loss through drain tile, a common and necessary agricultural practice for removing excess soil moisture. When implemented as part of an agricultural best management practice (BMP) treatment train, a series of management practices placed across a landscape gradient, cover crops may also serve to improve system-wide cost and nutrient removal efficiencies. Through modeling techniques and field monitoring spanning 2013-2019 at a southern Minnesota agricultural demonstration field site, this study aims to characterize total nitrogen (TN), total phosphorus (TP), and dissolved reactive phosphorus (DRP) concentrations and load reductions provided by cover crops. Implementation costs and additive system-wide nutrient removal provided through the cover crop implementation upstream of a treatment wetland were also quantified. Analysis of drain tile water quality and quantity data showed annual concentration reductions of 48%, 75% and 63% TN, TP and DRP, respectively. Reduction rates were quantified at 120 g ha^{-1} and 70 g ha^{-1} for TP and DRP, respectively, with annual implement costs of \$.99 g^{-1} ha^{-1} and \$1.69 g^{-1} ha^{-1}. Reduction rates were quantified at 9.13, kg^{-1} ha^{-1} for TN with an annual implementation cost of \$12.96 kg^{-1} ha^{-1}. The addition of cover crops upstream of a treatment wetland were also found to improve

system-wide nutrient removal costs by \$18.47 kg^{-1} for TN and \$211.38 kg^{-1} for TP annually, compared to those of a larger standalone treatment wetland.

Introduction

The use of cover crops as an agricultural conservation management practice includes the planting of forbs, legumes or grasses that provide fall and spring coverage on otherwise bare agricultural lands (SARE/CTIC, 2016; Carlson & Stockwell, 2013). This vegetative cover increases agroecosystem water and nutrient demands, through plant uptake and growth requirements, while also improving soil quality (Blanco-Canqui, et al., 2015; Keesstra, et al., 2018). In addition to providing economic benefits including increased yield and reduced tillage costs, cover crops have the potential to reduce subsurface nutrient losses (Drury, et al., 2014; Sustainable Agriculture Research and Education, 2020; Zhang, Tan, Zheng, Welacky, & Wang, 2017). Rye winter cover crops have been shown to reduce drain tile TN loads by an average of 15.40 kg ha^{-1} and to reduce concentrations by up to 39% (Ruffatti, 2019; Yang W. , Feng, Adeli, & Qu, 2021). Erosion control provided by cover crops on otherwise bare soil may also reduce particle-bound phosphorus transport (Liu, et al., 2019; Maltais-Landry, Scow, Brennan, & Vitousek, 2015).

While cover crops reduce nutrient losses, subsurface drainage (i.e., drain tiles), a necessary agricultural management practice for crop yield maintenance and stress reduction, can contribute to downstream water quality impairments (Blann, Anderson, Sands, & Vondracek, 2009; Erik, et al., 2018; Zucker & Brown, 1998). Increases in drain tile acreage, have been associated with increased DRP loads in agricultural tributaries with variable phosphorus dynamics observed among different cropping systems and variable agroecosystems (Nuruzzaman, Lambers, Bolland, & Veneklaas, 2005; Smith, King, & Williams, 2015; Hanrahan, King, Duncan, & Shedekar, 2021; Jarvie, et al., 2017). Studies have documented a wide range of phosphorus drain tile responses to cover crop implementation, including both phosphorus concentration increases and decreases, as well as observed phosphorus reductions when implemented in association with additional management practices (Horst, Kamh, Jibrin, & Chude, 2001; Zhang, Tan, Zheng, Welacky, & Wang, 2017; Lenhart, et al., 2017; Ni, Yuan, & Liu, 2020).

Cover crops are planted in the fall after cash crop harvest, followed by manual post-emergent herbicide termination, senescence, or other removal methods in late spring (Hanrahan, et al., 2018; Rosario-Lebron, Leslie, Yurchak, Chen, & Hooks, 2019). Direct cover crop implementation cost factors, which are required to determine impacts to land-owner profitability; include seed mix, planting, and termination (Christianson, Tyndall, & Helmers, 2013; Lazarus & Keller, 2018). In addition to environmental benefits, cover crops have the potential to improve farm-wide profit through crop system management alterations, yield improvements, and reduced labor, time, and machinery wear under no-till or reduced tillage practices commonly associated with cover crops. (Seifert, Azzari, & Lobell, 2018; Singh, et al., 2021; Bergtold, Ramsey, Maddy , & Williams, 2017).

When placed in the context of an agricultural BMP treatment train, cover crops also serve to reduce treatment pressure on downstream edge-of-field practices including treatment wetlands or riparian buffers (Lien & Magner, 2017). An agricultural BMP treatment train is comprised of multiple management practices placed in series along a landscape gradient, designed to improve system-wide performance by treating portions of the same runoff and nutrient load (Apfelbaum, Eppich, Price, & Sands, 1995; Lenhart, et al., 2017). The treatment train framework may provide greater cumulative nutrient reduction and lower per-unit cost nutrient removal than the use of a single practice (Lenhart, et al., 2017; Magner, 2011). One framework may consist of an “avoiding” practice such as a cover crop, placed at higher elevation in the landscape to reduce flows and nutrient delivery to a downstream “trapping” practice, such as a pond or wetland (Figure 9) (Lenhart, et al., 2017).

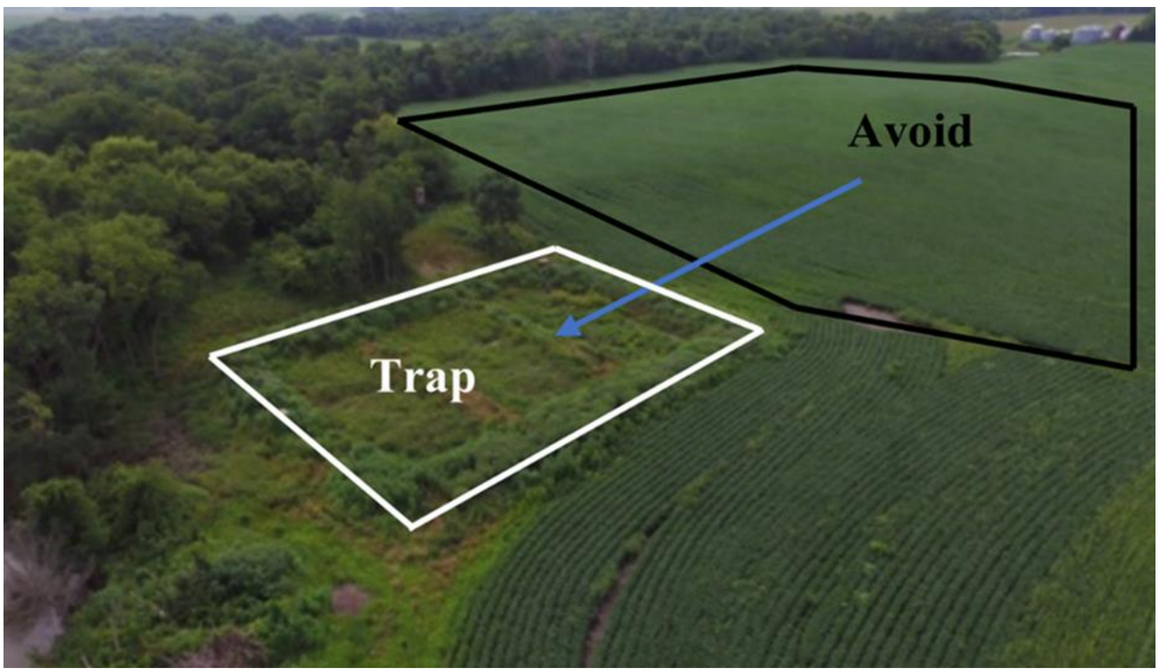

Figure 9. Management practice treatment train framework at the study site. Upstream practices that avoid losses of pollutants by retaining soil and nutrients, such as cover crops, such as the treatment wetland shown above, help to reduce the load to downstream practices. Arrow represents water flow. *Photo credit: David Hansen.*

Few studies have documented the economic and water quality benefits of agricultural management practice treatment trains (Barber, et al., 2016). This study fills this research gap through modeling methodologies and field data collection at an agricultural field research site present with a rye winter cover crop located upstream of a treatment wetland. At the research site, located in southern Minnesota, 10.12 hectares of upland cropland are present with a drain tile system that outlets at the treatment wetland, constructed in 2013. We conducted water quality and quantity monitoring at this outlet between 2013 and 2019 with rye winter cover crop use occurring between the fall of 2015 and the spring of 2019. Data spanning 2013-2015 allowed for the analysis of drain tile nutrient transport without cover crops and data spanning 2016-2019 for analysis with cover crops (Gordon, et al., 2021).

Data was utilized to address the five following research question: (1). Do cover crops provide subsurface TN, TP and DRP flow weighted mean concentration (FWMC) and load reductions? (2). What TN, TP and DRP removal cost efficiencies are associated with cover crop implementation? (3). How do subsurface TN, TP and DRP concentration discharge relationships vary in the presence and absence of cover crops? (4). How do drain tile flow volumes vary in the presence and absence of cover crops? (5). Do cover crops provide lower per-unit cost nutrient removals than the use of a stand-alone treatment wetland when implemented as part of an agricultural BMP treatment train?

Methods

Study Site

The research site is located in Martin County, Minnesota near the Iowa border within the Blue Earth River Watershed, draining to Elm Creek (Assessment Unit ID 07020009-502). The site has been extensively studied and serves as an agricultural BMP demonstration site (Lenhart, et al., 2016; Ross, 2014; Gordon, et al., 2021). Cropland, covering 10.12 ha, is present under a bi-annual corn and soybean rotation with conservation tillage. Crop yields averaged 187-bushel units for corn and 52-bushel units for soybean between 2013 and 2019. Average annual crop yields in Minnesota are 192-bushel units for corn and 49-bushel units for soybean (USDA, Minnesota Ag News 2020 Crop Production., 2021). Phosphorus fertilizer application rates were consistent, averaging 90 kg ha^{-1}, slightly passing regional rate recommendations (Kaiser, Lamb, & Eliason, 2011) (Kaiser et al., 2011). Nitrogen fertilizer application rates averaged 18 kg ha^{-1} between 2013 and 2017 increasing to an average rate of 170 kg ha^{-1} in 2018 and 2019, surpassing region rate recommendations (Kaiser, Lamb, & Eliason, 2011). A drain tile system is present with a spacing of 24 m, and a depth of 1.2 m, within loam and clay loam soils consisting of 22% clay, 42% sand and 36% silt. Drain tile flow outlets at an Agri Drain control structure located at the inlet of a subsurface treatment wetland covering .10 ha, where water quality and quantity sampling was conducted between 2013-2019 (43° 45’ 4’’N, 94° 20’ 51’’W; Figure 10).

The use of a rye winter cover crop began in the Fall of 2015 allowing for pre-cover crop analysis between the years of 2013-2015 and post-cover crop analysis between the years of 2016-2019. The cover crop mix consisted of 96.55% fall rye (*Secale cereale*), 1.16 % tillage radish (*Raphanus sativus*), 1.14% purple top turnips (*Brassica rapa*) and 1.15% trophy rapeseed (*Brassica napus*). Cover crops were aerial applied, via a plane, between late August and early September and terminated in early May.

Direct costs for establishment of cover crops on site was $118.24 ha^{-1}. Direct cost is defined as the cumulative cost for cover crop seed mix, installation and termination alone without the inclusion of other economic factors including changes to crop yield or machinery use. We obtained seed mix costs from the La Crosse Seed Company based out of La Crosse, WI, for the specific mix used on site at a rate of \$35.95 ha [1]. Costs for aerial seed application and spring herbicide application were provided by the University of Minnesota Extension Service at rates of \$42.01 and \$40.28 ha^{-1}, respectively (Lazarus & Keller, 2018). Direct cover crop cost at the field research sites were comparable to those report in other studies including \$205 ha^{-1} by Lenhart et al., (2017), \$151 ha^{-1} by Roley et al. (2016) and at \$115 ha^{-1} by Christianson et al. (2013)

Figure 10. Study site layout and location; drain tile monitoring location present at wetland inlet.

Nutrient Concentration and Load Analysis

Depending on study year, we collected weekly, bi-weekly, or monthly water quality grab samples at the drain tile outlet between April and November. Samples were analyzed at the Minnesota Valley Testing Laboratory in New Ulm, MN, for TN, TP and DRP, methods EPA 365.1 for P and EPA 300.0 for N. TN was measured as NO_3^{-1} as previous studies on site and in the region showed this to be the predominant form of N (Nustad, Rowland, & Wiederholt, 2015; Discovery Farms MN, 2021). A one-way analysis of variance was utilized to test for significant differences in drain tile constituent concentrations under the presence and absence of cover crops.

Within the FLUX32 software program, we calculated annual flow weighted mean concentrations (FWMCs) and loading volumes for each constituent (Minnesota Pollution Control Agency, 2021). FLUX32 software provides the coefficient of variation (CV) for each annual FWMC to compare annual concentration variation around the mean. A flow weighted mean concentration value was also calculated for years without cover crops and years with cover crops. FWMC values for each constituent both with and without cover crops were multiplied by the average annual site drain tile volume to determine annual loading rates. Loading rates with cover crops were subtracted from those without to estimate annual nutrient load reductions. Annual load reductions were further divided by the study site area to calculate cover crop nutrient reduction benefits in kilograms per hectare per year.

Concentration-Discharge Values

Utilizing individual water quality grab samples and associated instantaneous drain tile discharge data, we developed concentration-discharge relationships for comparing nutrient loss trends under no cover crop and cover crop conditions. We then used the power function equation to draw conclusions about the movement of constituent solutes under a range of discharge values, specifically mobilization and sources of constituents, following Equation 3:

$$(3).\ C = aQ^b$$

where a is the curve coefficient, C is the drain tile concentration in mg l^{-1}, and Q is the flow rate in cubic meters per second. The exponent "b" serves to quantify the per unit concentration increase relative to a per unit discharge increase. Values of $b < 0$ suggest concentrations follow a diluting pattern, values of $b > 0$ suggest constituent mobilization with flow increases, and values of $b = 0$ suggest chemostatic behavior, or no significant concentration changes in association with discharge (Dolph, et al., 2019; Godsey, Kirchner, & Clow, 2009).

Hydrology

We mounted Solinst Levelogger pressure transducers and paired Solinst Barologgers in an Agri Drain water control structure located at the treatment wetland inlet, to capture drain tile outlet flow. A known stage-discharge relationship calibrated by the Minnesota Department of Agriculture enabled for the calculation of flow rate based on water level over a v-notch weir in the control structure recorded in 10-minute time intervals. Flow rate was calculated as follows in Equation 4

$$(4).\ Q = 0.98x^{2.08}$$

where Q is the flow rate in cubic feet per second and X is water stage above the v-notch weir in feet. We then converted flow rate intervals to cubic meters per minute, multiplied by ten and summed to obtain annual and seasonal drain tile outlet volumes. Solinst Leveloggers have an estimated accuracy of ±0.05% from which the error ranges for field monitored tile outflow volumes were generated (Chun & Cooke, 2008).

We utilized annual flow volumes to calculate annual subsurface nutrient loading rates and cover crop load removals. Growing season flow volumes were then calculated between April 1st and September 30th and further divided into early and late period volumes. Early period flow volumes were compared to late period volumes to demonstrate the proportion of total drain tile flow occurring and available for treatment during the time frame of cover crop influence. Early flow volumes were calculated from April to June and late flow volumes from July to September.

Prioritize, Target and Measure Application Treatment Train Analysis

The Prioritize, Target and Measure Application (PTMapp) is a water quality model that leverages geospatial data and information systems to characterize nutrient, sediment, and hydrologic loading by field scale catchments. From this information locations are identified in the landscape that are feasible for various agricultural management and conservation practices. The application calculates hydrologic travel times, runoff volume and peak flow to characterize landscape TN and TP load and yield for a 10-year, 24-hour storm event; an event which is used as a proxy for annual values. While this application was designed primarily as a water quality planning tool for use by local government units, it does rely on known and accepted scientific methods and equations.

One such method is the stream power index (SPI) which utilizes physical landscape characteristics to determine erosion potential though identification of concentrated flowpaths. The Revised Universal Soil Loss Equation (RUSLE) is utilized to estimate annual sediment yield based on land cover, soil type, management and topography. TN and TP yields are based on empirical data obtained from literature values to determine export coefficients based on landcover. TN and TP delivery is based on the development of a travel time raster and first order loss equation. Runoff volume and peak discharge are determined based upon the Natural Resource Conservation Service (NRCS) runoff curve number method. Performance of particular BMPs are based upon reduction ratios determined by treatment type calculated from the volume of water that can be treated or how fast water moves through a BMP. This reduction ratio is then input into an empirical treatment decay function (Houston Engineering, 2016).

Nutrient reductions and implementation costs associated with feasible practices are calculated, both for single practice and treatment train scenarios. PTMApp cost estimates are determined from NRCS EQIP payment schedules. Costs for treatment wetland construction and annual cover crop implementation were taken from model outputs for use within this analysis. Cumulative treatment train load reductions are estimated through calculation of both the localized flow volume in addition to flow volumes and loads delivered from upstream best management practices to downstream best management practices (Houston Engineering, 2015; Houston Engineering, 2016).

We modeled the .10 ha treatment wetland at the field site within the Prioritize, Target and Measure Application to estimate TN and TP load reductions and implementation cost as a standalone practice. This practice was then modeled as part of a treatment train with 10.12 ha of upland cover crop. Wetland nutrient reduction efficacy, defined as kg ha^{-1} yr^{-1} of reduction, was utilized in association with the additive load reduction resulting from the treatment train scenario to calculate what size reduction could be applied to the wetland to achieve the same cumulative site nutrient reduction. Finally, we calculated the new cost associated with the reduced wetland size, in addition to the cost for cover crop implementation to determine the cost of nutrient reduction, defined as dollar kg^{-1} of nutrient reduction, associated with using cover crops as part of an agricultural best management practice treatment train.

Results

Annual Drain Tile Nutrient Concentrations

Annual TN FWMC ranged from 6.9 mg l^{-1} to 23.3 mg l^{-1}, with a flow weighted mean concentration of 11.3 mg l^{-1} across all study years (Figure 11). TN FWMC across study years without cover crops was 17.13 mg l^{-1} and 8.97 mg l^{-1} across study year with cover crops, resulting in a TN FWMC reduction of 48%. Results of a one-way analysis of variance test found a significant difference at the 99% level between TN concentrations in years without cover crops and years with cover crops ($F(1, 113) = 80.02$, $p < .00$).

Annual TP FWMC ranged from 0.02 mg l^{-1} to 0.15 mg l^{-1} , with a flow weighted mean concentration of 0.04 mg l^{-1} across all study years (Figure 12). TP FWMC across study years without cover crops was 0.12 mg l^{-1} and 0.03 mg l^{-1} across year with cover crops, resulting in a TP FWMC reduction of 75%. Results of a one-way analysis of variance test found a significant difference at the 99% level between TP concentrations in years without cover crops and years with cover crops ($F(1, 110) = 12.29$, $p < .00$).

Annual DRP FWMC ranged from 0.02 mg l^{-1} to 0.12 mg l^{-1} , with a mean FWMC of .04 mg l^{-1} across all study years (Figure 13). DRP FWMC across study years without cover crops was .08 mg l^{-1} and .03 mg l^{-1} across study years with cover crops, resulting in a DRP FWMC reduction of 63%. Results of a one-way analysis of variance test found a significant difference at the 99% between DRP concentrations in years without cover crops and years with cover crops ($F(1, 85) = 25.81$, $p < .00$).

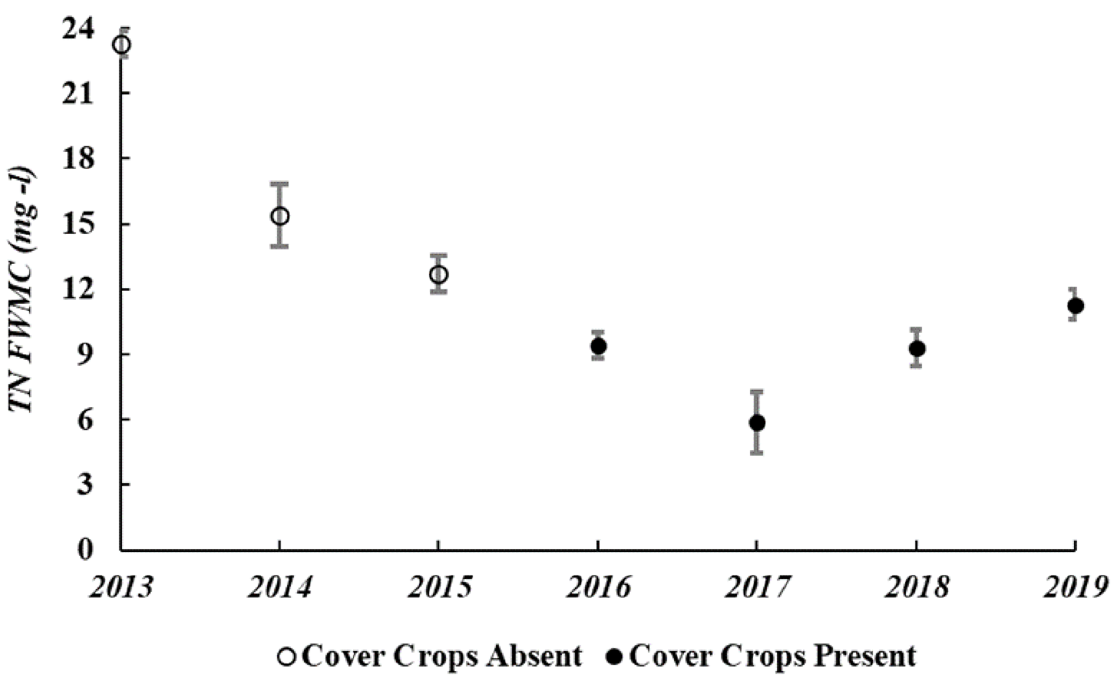

Figure 11. Average annual total nitrogen (TN) flow weighted mean concentrations (FWMC). Note. CV displayed in gray bars.

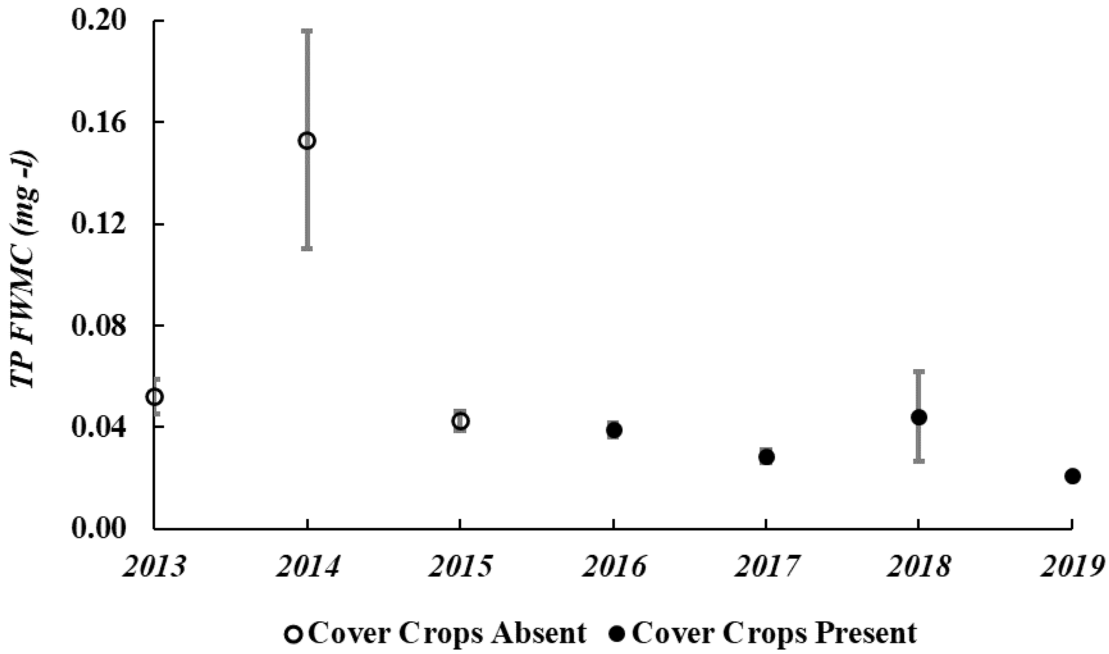

Figure 12. Average annual total phosphorus (TP) FWMC.

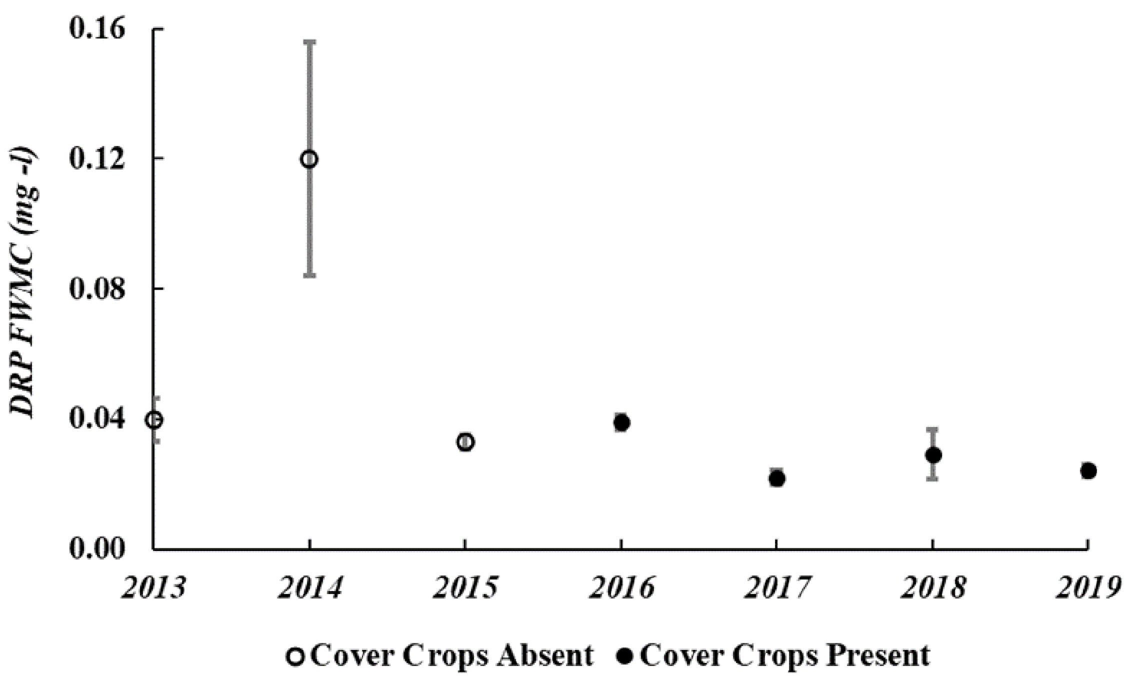

Figure 13. Average annual dissolved reactive phosphorus (DRP) FWMC.

Drain Tile Nutrient Load Analysis

Annual TN loads ranged from 8 kg ha^{-1} to 170 kg ha^{-1}, with a mean load of 64 kg ha^{-1} across all study years (Table 9). The average TN load in years with cover crops absent was 22.81 kg ha^{-1} and 13.69 kg ha^{-1} in years with cover crops present, accounting for a cover crop TN reduction benefit of 9.13 kg^{-1} ha^{-1} yr^{-1} (Table 10).

Annual TP loads ranged from 0.02 kg ha^{-1} to 0.24 kg ha^{-1} with a mean load of .09 kg ha^{-1} across all study years (Table 9). The average TP load in years with cover crops absent was 0.16 kg ha^{-1} and .04 kg ha^{-1} in years with cover crops present, accounting for a cover crop TP reduction benefit of 0.12 kg^{-1} ha^{-1} yr^{-1} (Table 10).

Annual DRP loads ranged from 0.02 kg ha^{-1} to 0.16 kg ha^{-1}, with a mean load of 0.07 kg ha^{-1} across all study years (Table 9). The average DRP load in years with cover crops absent was 0.11 kg ha^{-1} and 0.04 kg ha^{-1} in years with cover crops present, accounting for a cover crop DRP reduction benefit of 0.07 kg^{-1} ha^{-1} yr^{-1} (Table 10).

Table 9. Annual monitoring season drain tile nutrient loads.

Year	Cover Crops	Total Nitrogen (kg ha^{-1})	Total Phosphorus (kg ha^{-1})	Dissolved Reactive Phosphorus (kg ha^{-1})
2013	Absent	18	0.04	0.03
2014		21	0.21	0.16
2015		8	0.02	0.02
2016	Present	26	0.24	0.11
2017		53	0.03	0.02
2018		152	0.04	0.05
2019		170	0.06	0.07
Ave.		64	0.09	0.07

Table 10. Annual nutrient loads under cover crop conditions.

Cover Crop	Total Nitrogen (kg ha^{-1})	Total Phosphorus (kg ha^{-1})	Dissolved Reactive Phosphorus (kg ha^{-1})
Absent	22.81	0.16	0.11
Present	13.69	0.04	0.04
Benefit	9.13	0.12	0.07

Cost Benefit Analysis

Nutrient removal cost efficiency was defined as the annual dollar per unit of consistent removal over a unit area. The annual removal rate of TN provided by cover crops was determined to be 9.13 kg ha^{-1}; based on a direct cover crop implementation cost of $118.24 ha^{-1}, the annual TN cost efficiency would be $12.95 kg^{-1} ha^{-1} (Table 11). The annual removal rates of TP and DRP provided by cover crops were determined to be 120 g ha^{-1} and 70 g ha^{-1}; based on a direct cover crop implementation cost of $118.24 ha^{-1}, annual TP and DRP cost efficiencies would be $0.99 g^{-1} ha^{-1} and $1.69 g^{-1} ha^{-1}, respectively (Table 12).

Table 11. Cover crop nitrogen removal cost.

Constituent	Annual Removal Rate (kg ha^{-1})	Direct Cover Crop Implementation Cost ($ ha^{-1})	Annual Cost ($ kg^{-1} ha^{-1})
Total Nitrogen	9.13	118.24	12.95

Table 12. Cover crop phosphorus removal costs.

Constituent	Annual Removal Rate (g ha^{-1})	Direct Cover Crop Implementation Cost ($ ha^{-1})	Annual Cost ($ g^{-1} ha^{-1})
Total Phosphorus	120	118.24	0.99
Dissolved Reactive Phosphorus	70	118.24	1.69

Concentration-Discharge Relationships

Parameter “b” of the log-log concentration discharge relationship for TP with cover crops was -0.32, indicating source limitation and concentration dilution at high flows. Parameter “b” of the log-log concentration discharge relationship for TP with no cover crops was 0.23, indicating constituent mobilization at higher flows through erosional processes. Similarly, parameter “b” of the log-log concentration discharge for DRP was -0.46 with cover crop and 0.22 without cover crops (Figure 14). Parameter “b” of the log-log concentration discharge for TN was close to zero both with and without cover crops, indicating minimal concentration changes relative to variation in discharge (Dolph, et al., 2019).

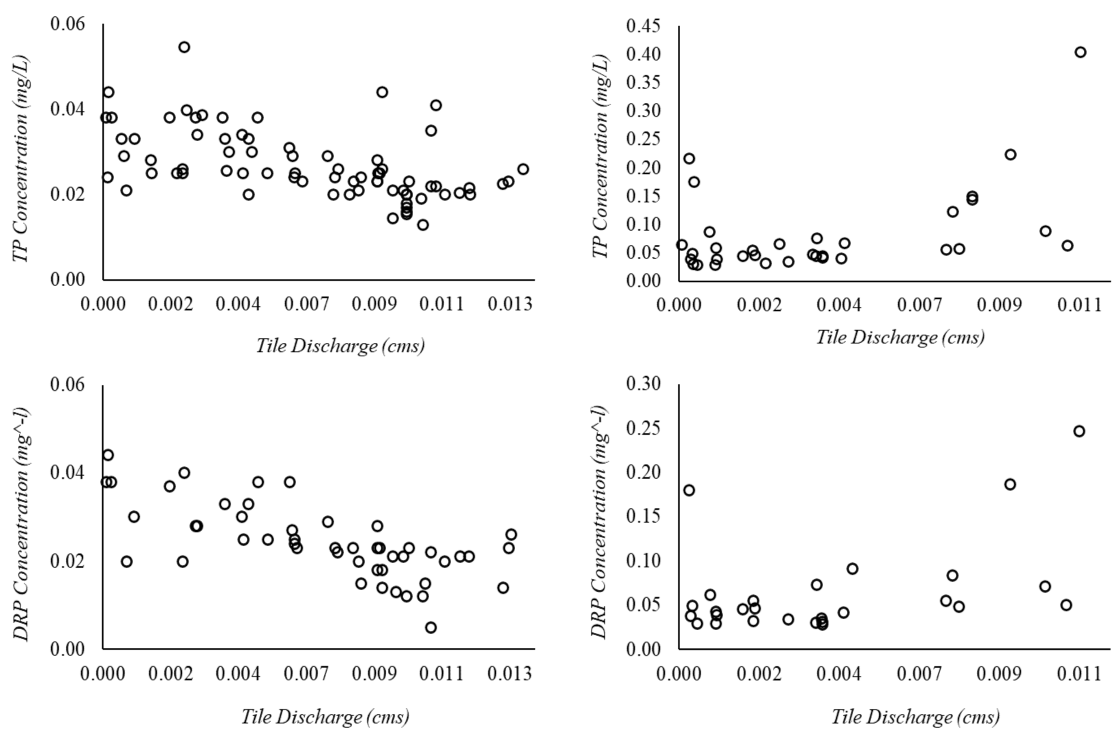

Figure 14. TP (top) and DRP (bottom) concentration discharge relationships with cover crops (left) and without cover crops (right).

Drain Tile Flow Volumes

Annual growing season drain tile volumes ranged from 5,583 m^3 to 19,289 m^3, averaging 12,173 m^3 across the study period. Proportions of the total growing season drain tile volume occurring during the early period of April 1 through June 30th ranged from 55% to 96%, averaging 80% across the total seven-year study period. Alternatively, drain tile volumes during the late period of July 1 through September 30th ranged from 4% to 45% of the total drain tile volume averaging 20% across the total seven-year study period. The average proportion of the total growing season drain tile volume during the early period was 84% in years without cover crops and 77% in years with cover crops (Figure 15). Rainfall depth during the growing season averaged 61.50 cm between 2013 and 2019, with 54% occurring during the early period and 46% occurring during the late period.

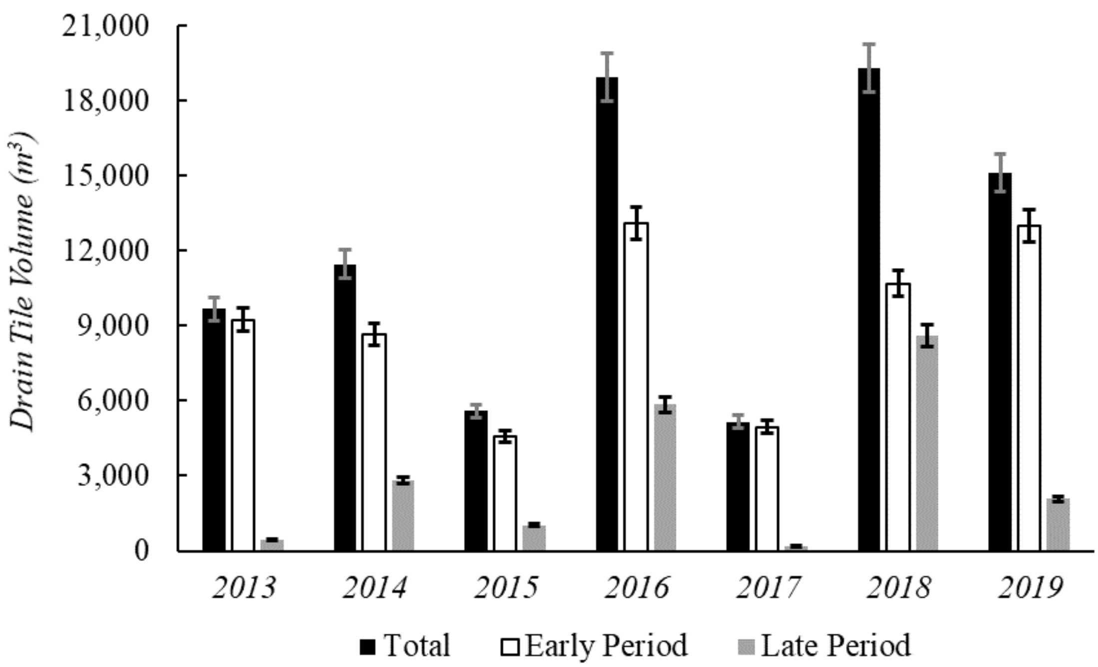

Figure 15. Growing season drain tile volumes by early and late periods.

Prioritize, Target and Measure Application Treatment Train Analysis

Prioritize, Target and Measure Application modeling results showed that the 0.10 ha wetland provides a TN load reduction of 141.49 kg and a TP load reduction of 10.56 kg, both to the catchment outlet, for a 24-hour, 10-year storm event at a construction cost of $5,110. Planting and termination of 10.12 ha of cover crops upstream of the wetland provides additional event load reductions to the catchment outlet of 106.69 kg of TN and 6.34 kg of TP, at a cost of $838. With these additive reductions, the wetland size could be reduced to 0.04 hectares at a new construction cost of $2,040 while providing the same annual cumulative nutrient reductions as the original standalone wetland (Figure 16). Construction of this smaller wetland area would also preserve .06 ha of cropland for production.

The construction cost for the 0.04 ha wetland and for planting and termination of 10.12 ha of cover crop would be $2,878 proving cumulative treatment train load reductions of 163.19 kg and 10.56 kg for TN and TP, respectively, to the catchment outlet annually (Figure 8). Nutrient removal cost of the standalone wetland would be $36.11 kg^{-1} for TN and $483.97 kg^{-1} for TP, while nutrient removal cost of the treatment train system would be $17.64 kg^{-1} and $272.59 kg^{-1} for TN and TP, respectively.

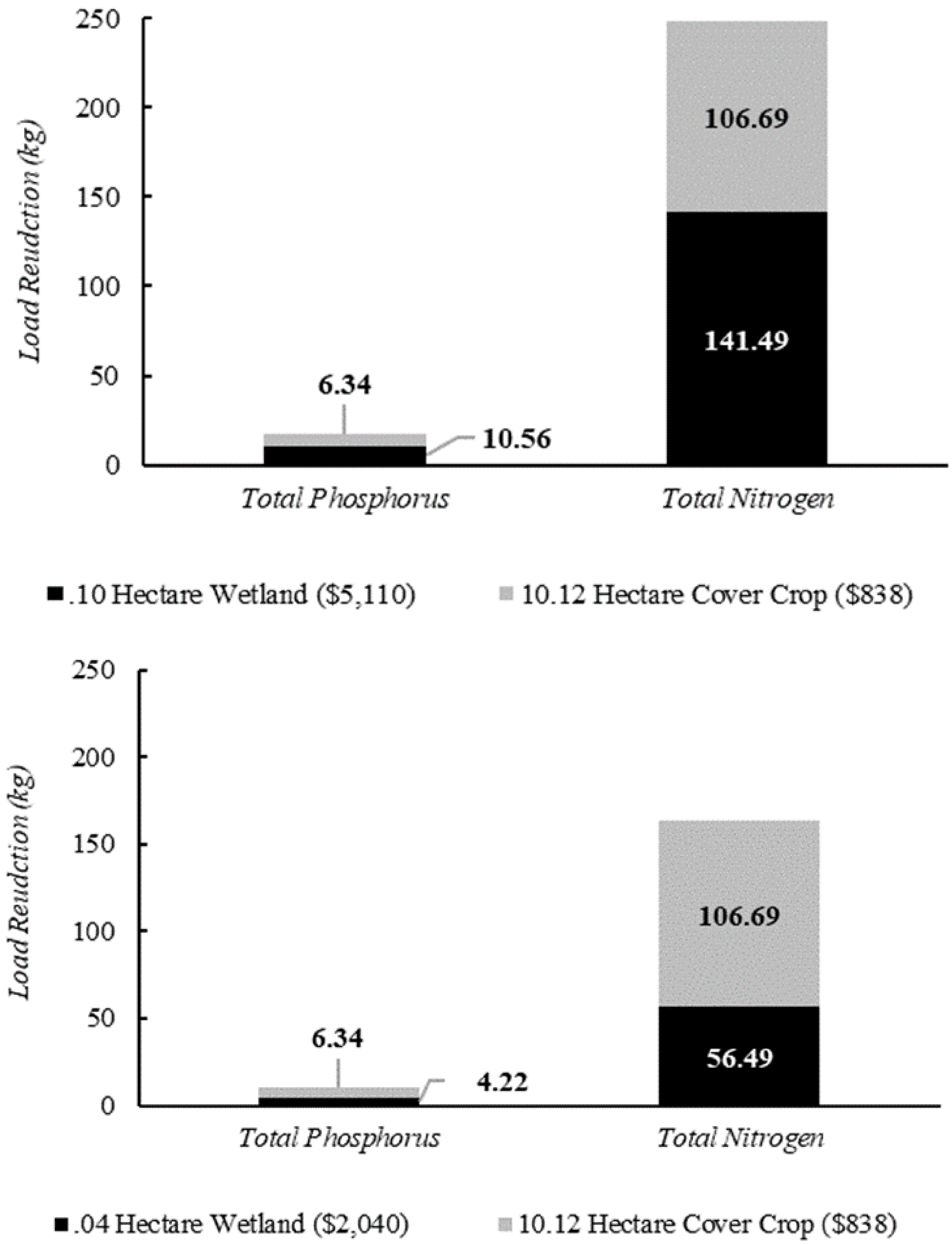

Figure 16. Prioritize, Target and Measure Application results for a 10 year, 24-hour storm event; standalone wetland reductions with additive cover crop benefits (top) and treatment train scenario with reduced wetland area and additive cover crop benefit (bottom). Dollar values represent wetland construction cost and cover crop planting and termination costs.

Discussion

Significant drain tile nutrient concentration reductions were observed with the implementation of cover crops at reduction rates of 48%, 75% 63% for TN, TP and DRP, respectively. These reductions were likely driven by the presence of cover crops on previously bare land between April and June, when the largest volume of drain tile flow occurred on site. In addition to nutrient uptake and soil stability, cover crops reduce deep drainage and improve soil water storage during the early crop growth period of April-June, serving to retain nutrients otherwise lost in drain tile (Yang W. , et al., 2019)**.** An average of 84% of the total annual drain tile volume occurred on-site between April and June of 2013-2015 with no cover crops, dropping to an average of 77% between April and June of 2016-2019 with cover crops. Although these differences may be attributed to weather variation, cover crops have been found to contribute to reduced soil moisture storage and increased evapotranspiration rates, serving to decrease drain tile volumes (Meyers, Bergez, Constantin, & Justes, 2019).

The greatest CVs around annual FWMCs for both TP and DRP were observed in 2014 and 2018, driven by high drain tile concentrations during storm events. A 3.8-inch rain event in June of 2014 produced TP and DRP concentrations of .40 mg l^{-1} and .25 mg l^{-1}, respectively. A 2.21-inch rain event in June of 2018 produced TP and DRP concentrations of .19 mg l^{-1} and .09 mg l^{-1}, respectively. Rain events in 2018 resulted in a lower CV around the annual mean flow weight concentration relative to 2014. While drain tile TP and DRP losses are associated with large storm events, this provides evidence for added cover crop soil stability and soil water storage. Phosphorus reductions are largely driven by erosion control during large storm events while nitrogen reduction is driven primarily by water retention (Trentman, et al., 2020; Pease, et al., 2018).

Variation in concentration-discharge relationships on site between cover crop and without cover crop provide additional evidence for cover crop reduction of drain tile phosphorus losses. TP and DRP relationships with no cover crops follow a concentrating pattern, indicating constituent mobilization at high flows through erosion or landscape connectivity. TP and DRP relationships with cover crops follow a diluting pattern, indicating concentration dilution occurring at high flows resulting from constituent source limitation (Godsey, Kirchner, & Clow, 2009).

No significant difference in nitrogen concentration-discharge relationships were observed with and without cover crops. Annual nitrogen flow weighted mean concentrations increased in 2018 and 2019 following reductions in 2016 and 2017 after cover crop implementation. Increased in nitrogen drain tile flow weighted mean concentration were associated with large fertilizer rate increases on site, consistent with nitrogen fertilizer observations in previous studies (Jaynes, Colvin, Karlen, Cambardella, & Meek, 2001). Concentrations, however, were still lower than those observed previously under no cover crops and with lower fertilizer application rates.

Results quantifying cover crop nutrient load reduction rates and direct implementation costs will benefit Midwestern cover crop implementation efforts. The Minnesota Nutrient Reduction Strategy is a multi-agency initiative begun in 2014 as part of a larger 12 state task force to working to reduce nutrient loading to the Gulf of Mexico. Initiative efforts included cover crop expansion and called for 12% phosphorus and 25% nitrogen reductions prior to 2025 (Anderson, Wall, & Olson, 2016). It was recognized within the strategy that there was no realistic way to demonstrate reduction achievement without the advancement of research on cover crop implementation and success (Wall, et al., 2020). The plan called for 1,900,000 additional acres of cover crops to be implemented; however only 200,000 acres were implemented in the first five years, with little of these acres implemented on cropland with corn and soybean rotations.

Results of the study found annual cover crop TN reduction benefits of 9.13 kg^{-1} ha^{-1} at a direct annual implementation cost of \$12.96 kg^{-1} ha^{-1}. Results also found annual cover crop TP and DRP reduction benefits of 120 g^{-1} ha^{-1} and 70 g^{-1} ha^{-1} at direct annual implementation costs of \$.99 g^{-1} ha^{-1} and \$1.69 g^{-1} ha^{-1}, for TP and DRP respectively. Christianson et al. (2021) noted annual reduction costs of both \$2.70 kg^{-1} and \$3.25 kg^{-1} for TN and \$0.04 g^{-1} and \$0.05 g^{-1} for TP. These cost estimates provided by Christianson et al. (2021) factor in cost savings associated with fertilizer, weed control, erosion repair and yield increases.

Direct implementation costs identified from a field-based study, that will be incurred by landowners, can help further refine cover crop implementation goals and inform initiative efforts. Additional cover crop implementation initiatives in Minnesota include the University of Minnesota Forever Green Initiative, the Conservation Reserve Enhancement Program, and the Working Lands Watershed Restoration Feasibility Study and Program Plan (Wall, et al., 2020). The Midwest Cover Crop Council (https://mccc.msu.edu/) is also leading joint state efforts by bringing together leaders from major universities.

While cost per kg reduced for TP and DRP are orders of magnitude larger than those for TN, less phosphorus contributions are required for water body impairment and algal bloom occurrence in freshwater ecosystems, which is driven by small contributions of highly bioavailable phosphorus (Baker, et al., 2014). Minnesota water quality standards are 0.065 mg/l and 10 mg/l for TP and TN, respectively (Minnesota Legislature, 2018). Cost for TP and DRP removals must also be placed into context of the economic implications of algal blooms relative to tourism and recreation, commercial fishing, properties values, drinking water treatment and human health (U.S. Environmental Protection Agency, 2021).

Identification of direct landowner costs and economic benefits will also aid in landowner cover crop implementation, including the potential for increased cash crop yield (Farzadfar, Knight, & Congreves, 2021). At the field site average soybean yield increased from 43 to 56-bushel units under cover crops and average corn yield increased from 182 to 191-bushel units under cover crops. Identifying beneficial uses of cover crops in a more systematic manner, such as within the treatment train approach may also encourage implementation. Modeling of cover crop implementation was shown to increase nutrient reduction rates within a downstream subsurface treatment wetland, demonstrating the potential for wetland size and cost reduction while obtaining the same cumulative nutrient reductions rates in association with cover crops.

Cover crop benefits to wetland performance could be substantial, as the main limiting factor in landowner wetland implementation in this region is cost and land area taken out of production. Landowners may not be financially able or willing to remove large areas of productive cropland for wetland implementation (Hyberg et al., 2015). As such, while treatment wetlands are effective, opportunities to place them are limited. Subsurface flow wetlands have been shown to be more effective at nutrient removal than surface flow wetlands, however, cannot treat large volumes of water (Kadlec & Wallace, 2009; Gordon, et al., 2021). By reducing flow with the implementation of cover crops, subsurface flow wetlands become more economically and hydrologically feasible.

A similar study conducted by Hanrahan et al. (2021) found a monthly average drain tile load reduction of 50% for TN, TP and DRP provided by cover crops from 40 agricultural field sites in northcentral Ohio. Southern Minnesota differs from Ohio in having less intensive tile drainage and a later spring drainage season, thus making cover crops less effective. When used in associated with drainage water management, cover crops were found to reduce TP flow weighted mean concentrations by 26% with no significant effect on DRP concentrations (Zhang, Tan, Zheng, Welacky, & Wang, 2017). Waring et al., (2020) noted consistent reductions in subsurface nitrate leaching through implementation of both cover crops and no-till practices in north-central Iowa. Few long-term field studies exist that document drain tile nutrient reduction potential of winter cover crops as a standalone practice for both nitrogen and phosphorus in the Midwest.

Rural areas in the United States are experiencing more intensive water management due to regulations and increasing demand for food and fuel, presenting a strong need to maximize agricultural best management practice performance and implementation (Thompson, Reeling, Michelle , Prokopy, & Armstrong, 2021). Relative to other areas of the nation, cover crop implementation is lowest in the Midwest, despite significant nutrient contributions to the Gulf of Mexico (Hamilton, Mortensen, & Kammerer Allen, 2017). Challenges to Midwestern cover crop implementation include a short growing season paired with primary cultivation of full season corn and soybean crop (Carlson & Stockwell, 2013). While landowner perceptions of cover crops in the

Midwest are generally positive, direct net returns for implementation have been negative in many cases, highlighting a need for the development of more state or region-specific implementation recommendations (Plastina, Liu, Miguez, & Carlson, 2018).

This study demonstrates the benefits of cover crop implementation in the Midwest while also quantifying associated nutrient reduction rates and direct implementation costs. The use of cover crops in a systematic manner for improved cumulative site nutrient reductions and cost efficiencies are also demonstrated. Research limitations include findings limited to only one field research site in addition to lack of continuous nutrient monitoring to capture the full influence of storm events across the study period. Results could also be improved through field monitoring data within the on-site treatment train to complement findings from modeling techniques. Limitations also include exploring the applicability of cover crops within treatment trains as related to DRP specifically, as cover crop may actually contribute DRP to downstream practices through increased growth and movement of biological material.

Next steps include research on site- and region-specific factors including work on multiple sites to account for variations in soil, landscape, local climate, or management practices and to document the influence of various seed mixes or seeding and termination dates and methodologies. Further knowledge on economic considerations including those related to cash crop yield, accessibility and benefit of governmental cost share programs and long-term landowner return on investments will also aid in increased implementation. Finally, expansion of the treatment train framework to include additional practice types, to improve landscape positioning and to document varying hydrologic conditions will contribute to even greater cumulative conservation practice performance.

Conclusion

While there is a need for increased cover crop implementation in the Midwest, a lack of research exists on implementation success, nutrient reduction benefits and economic implications. In addition, a growing demand for food and fuel production while also protecting water resources has called for conservation practice performance improvements. Through field data collection at an agricultural field demonstration site in southern Minnesota, this work serves to quantify subsurface nutrient reduction benefits provided by cover crops at rates of 120 g ha^{-1} and 70 g ha^{-1} for TP and DRP, respectively, with associated direct annual implement costs of \$0.99 g^{-1} ha^{-1} and \$1.69 g^{-1} ha^{-1}. An annual reduction rate of 9.13, kg^{-1} ha^{-1} was determined for TN with an associated direct annual implementation cost of \$12.96 kg^{-1} ha^{-1}.

The systematic use of cover crops for improved BMP cost and nutrient reduction efficiency is also demonstrated through desktop modeling of the treatment train framework. Through additive upstream flow and nutrient reduction, cover crops allow for size reductions to more costly downstream treatment wetlands while maintaining the same cumulative nutrient reduction as the original stand-alone wetland. In addition, this serves to minimize the area of land taken out of production to maintain landowner profitability. Future research building upon this work would aid in the development of field and state specific cover crop implementation guidelines, while also expanding on the treatment train framework for improved conservation practice performance.

Chapter 3: A case study of dissolved reactive phosphorus contribution, loss drivers and management techniques within a Southern Minnesota agricultural riparian buffer

Summary

Riparian saturated buffers, while known to be effective for nitrogen and particulate phosphorus removal, have been shown to lose dissolved reactive phosphorus (DRP). The drivers of these losses, including soil characteristics, flooding frequency, microbial community structure and plant phosphorus content are not well documented. In addition to demonstrating the occurrence of DRP loss at an agricultural saturated buffer demonstration site located in Southern, Minnesota, this study aims to assess dominant drivers of this loss through field data collection. Soil test phosphorus (STP) and organic matter, plant species inventory and phosphorus content, groundwater phosphorus concentrations, hydrology, and microbial community structure data was collected at the field research site between 2018 and 2021. Microbial community structure data was in the form of microbial phosphorus biomass and arbuscular mycorrhizal fungi (AMF) plant root colonization. AMF are known to form mutualistic relationships with plant roots, facilitating nutrient uptake and soil biogeochemical regulation. These communities, however, are often degraded in agricultural settings. The use of an AMF amendment for DRP loss mitigation, shown effective in terrestrial settings, is explored within this study. Plant harvest is also explored as a management technique to facilitate system wide removal of phosphorus following plant uptake.

Introduction

Edge of field practices such as riparian, or streamside, buffers play a critical role in nutrient processing for water quality. Vegetation in the riparian zone facilitates the settling of sediment and associated particulate pollutants while also reducing erosion through soil stabilization (Hoffman, Kjaergaard, Uusi-Kamppa, Christian Brunn Hansen, & Kronvang, 2009; Hook, 2003). Saturated riparian buffers are a relatively new management practice designed to capture and treat subsurface drainage water that would otherwise bypass the buffer (Lenhart, Wilson, & Gordon, Factors impacting the variability of effectiveness of agricultural best management practices (BMPs) in Minnesota., 2016). Tile drainage is diverted through the subsoil via a control structure and perforated pipe, or "distribution line", with the purpose of storing and slowing the movement of water for increased denitrification (Jaynes & Isenhart, 2014).

As lower concentrations of suspended sediment and total phosphorus occur in sub-surface drainage water relative to surface water, saturated buffers are primarily implemented as a nitrogen control measure (Lenhart, et al., 2017). While shown effective for nitrate removal, research quantifying DRP removal in saturated buffers is minimal and largely inconclusive (Jaynes & Isenhart, 2019; Utt, Jaynes, & Albertsen, 2015; Christianson, Frankenberger, Hay, Helmers, & Sands, 2016). Previous studies have documented DRP losses in similar vegetated riparian buffer areas with a wide range of phosphorus efficiencies spanning from a 36% load removal up to an 89% load contribution (Hoffman, Kjaergaard, Uusi-Kamppa, Christian Brunn Hansen, & Kronvang, 2009; Habibiandehkardi, Lobb, Sheppard, Flaten, & Owens, 2017; Kieta, Owens, Lobb, Vanrobaeys, & Flaten, 2018).

Although vegetated riparian buffers have been shown to contribute phosphorus to nearby waterways, the dominant drivers of these contributions are not well understood (Walton, et al., 2020). Particularly, the roles of soil microbial community structure, soil characteristics, occurrence of leaching in runoff or under flooded and anaerobic conditions, and the role of plant phosphorus uptake (Chandrasoma, Christianson, & Christianson, 2019; Singh, Kaur, Williard, Schoonover, & Nelson, 2020; Utt, Jaynes, & Albertsen, 2015). In addition to documenting DRP loss from a saturated buffer site located in Southern, Minnesota, this study aims to address phosphorus research needs through flood frequency analysis and soil loss quantification, plant community inventory and phosphorus content quantification, and microbial community characterization in the form of AMF plant root colonization and microbial phosphorus biomass volume.

Soil microbial communities, particularly fungi, which can re-establish slowly on the scale of decades, are typically altered, or degraded in agricultural settings following decades of tillage, soil compactions, and fertilizer application thus, reducing ecological function (Koziol & Bever, 2017; de Vries & Shade, 2013). Soil microbes, specifically AMF, form mutualistic symbiosis with plant roots and are an important component in plant nutrient uptake and soil biogeochemical environment regulations (Eckhard, Marschner, & Jakobsen, 1995; Jakobsen, Smith, Gronlund, & Smith, 2011). Similarly, soil microbial biomass represents the living component of soil which plays key roles in plant and soil nutrient availability (Singh & Gupta, 2018). Microbial biomass can store phosphorus in organic forms and further, convert to inorganic plant available forms (Xu, Thornton, & Post, 2013; Brookes, Powlson, & Jenkinson, 1982).

Use of AMF amendment in disturbed soils has proved successful in re-establishing soil microbial ecological functioning and associated nutrient storage in upland grassland restorations, however, is not well documented within wetland and riparian habitats (Koziol & Bever, 2015; Koziol & Bever, 2017; Bohrer, Friese, & Amon, 2004). As such, potential exists for the use of AMF soil amendment within vegetated riparian zones as a DRP loss management technique through greater plant phosphorus uptake (Rubin & Gorres, 2021). In addition, leaving previously disturbed land to fallow may also contribute to the re-establishment of organic matter and associated soil microbial communities (House & Bever, 2018).

While soil microbial management may contribute to increased soil phosphorus storage, greater phosphorus conversion to plant available forms and increased plant phosphorus uptake; microbial and plant phosphorus storage are considered to be transient pools (De Groot & Golterman, 1993; Hoffman, Kjaergaard, Uusi-Kamppa, Christian Brunn Hansen, & Kronvang, 2009). Therefore, management strategies are needed to reduce phosphorus loss. Plant harvest is an accepted management technique for achieving phosphorus system removal before the occurrence of plant senescence, nutrient translocation, and microbial decomposition (Hille, et al., 2019; Menon & Holland, 2014). As such, use of microbial management strategies in association with plant harvest will be explored as a vegetated riparian buffer DRP loss management techniques within this study.

The primary hypothesis of this study was that the saturated buffer site would be contributing to DRP yields entering the nearby waterway. It was further hypothesized that soil microbial community amendment would contribute to (1). perennial grass establishment success, (2). increased AFM root colonization and (3). increased plant phosphorus uptake; ultimately contributing to reductions in STP. Associated project objectives included (1). to demonstrate the occurrence of DRP contributions from an agricultural riparian buffer, (2). to characterize dominant drivers of riparian buffer DRP loss, and (3). explore riparian zone DRP loss mitigation strategies.

Methods

Study Site

The study site is located on a saturated buffer adjacent to a 130 meters of stream frontage in southern Dakota County, MN. Installation of the saturated buffer, adjacent to Mud Creek (AUID 07040002-558), was conducted in 2015 by Ecosystem Services Exchange consisting of the creation of an additional .10 ha of vegetation riparian buffer (Ecosystem Services Exchange, 2021). The .10 ha saturated buffer study area receives drainage from 19.9 acres of upland cropland under a bi-annual corn and soybean rotation and strip tillage. Soils on site are a drained muck characterized by 28% sand, 29% silt and 43% clay with an average pH of 7.7. Work for this project began on-site in the fall of 2018 with the collection of baseline soil, plant and microbial community structure data and the installation of four shallow groundwater wells (44° 30’ 8’’N, 93° 10’ 5’’W Figure 17).

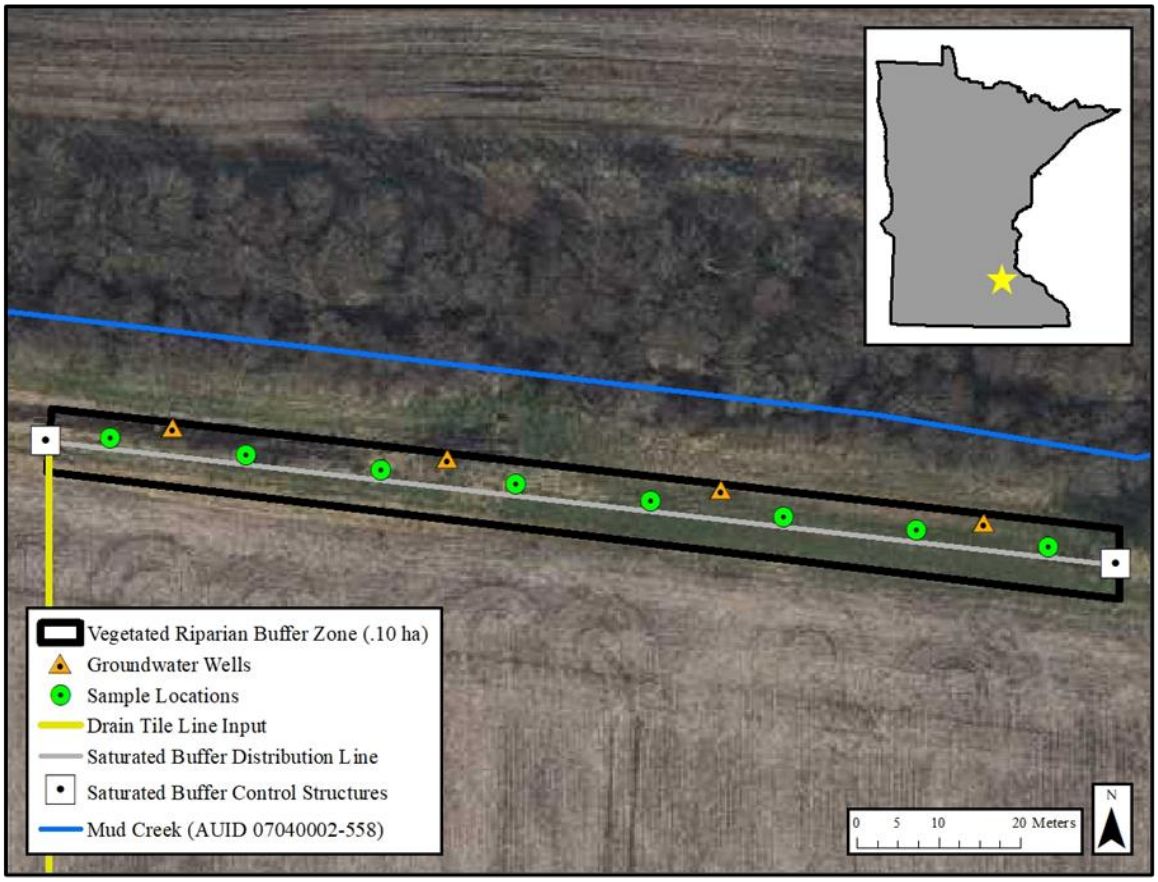

Figure 17. Saturated buffer field site location and layout.

In the Fall of 2018, the site was inoculated with an AFM amendment obtained from a company called MycoBloom 2018 and seeded with both warm and cool season perennial grass seed, both conducted through broadcast methods (MycoBloom, 2021). The warm season perennial grass mix included prairie cordgrass (*Spartina pectinata*), big bluestem (*Andropogon gerardii*), Indian grass (*Sorghastrum nutans)* and switchgrass (*Panicum virgatum*). The cool season perennial grass mix included bluejoint (*Calamagrostis canadensis*), wild rye (*Elymus virginicus*), American manna grass (*Glyceria grandis*), and bottlebrush sedge (*Carex comosa*). Baseline site vegetation included smooth brome grass (*Bromus inermis*), reed canary grass (*Phalaris* *arundinacea*) and quackgrass (*Elymus repens)* with several additional week species such as clover (*Trifolium sp.*), curly dock (*Rumex crispus*), timothy (*Phleum pratense*), foxtail barley (*Hordeum jubatum*), daisy fleabane (*Erigeron strigosus*), dandelion (*Taraxacum sp*), goldenrod (*Solidago sp*).

Field Data Collection

Field data collection spanned 2018-2021 and was completed in the fall at the culmination of each growing season. Data collected during each study year included Olsen STP, soil organic matter, plant phosphorus content and AMF root colonization rates. We conducted additional sampling for Olsen STP, soil organic matter and AMF spore count within the adjacent agricultural field in 2020 to compare rates against those in the riparian buffer zone. We collected orthophosphorus concentrations in shallow groundwater wells during the 2019 and 2020 sampling seasons and within the saturated buffer inlet control structure during the 2020 sampling season. We completed vegetation quadrat sampling during the 2020 and 2021 growing seasons following observed establishment of native perennial grasses in 2020 (Cox, 1990).

An Oregon based company called MycoRoots completed AMF plant root colonization assessments (MycoRoots, 2021). Additional microbial community assessment included the quantification of soil microbial phosphorus by the University of Toledo (Zhao, Li, & Lin, 2008). Microbial phosphorus samples were collected in during the 2020 study year and which we then paired with organic matter and Olsen STP samples.

Data Analysis

In order to confirm the occurrence of DRP loss from the field site, we calculated DRP groundwater yields from 2019 and 2020 orthophosphorus concentration samples and groundwater output flow volumes. Further, input DRP yields from the saturated buffer distribution line were determined from input structure orthophosphorus concentration samples and inflow volumes. Saturated buffer DRP inputs were deducted from DRP groundwater yields to determine DRP yield contributions from the newly established .10 ha vegetated riparian buffer zone.

We completed A Pearson's correlation coefficient to determine the strength of linear relationships between soil organic matter content and microbial phosphorus biomass and between organic matter and STP (Puth, Neuhauser, & Ruxton, 2014). These relationships were assessed to identify trends in microbial community establishment relative to the re-establishment of organic matter content and associated soil phosphorus storage and bioavailable phosphorus availability. In order to further assess soil phosphorus storage, we calculated a soil phosphorus saturation ratio (PSR) based on soil aluminum and iron content. This ratio allows us to determine site soil phosphorus storage capacity relative to a defined threshold value for the occurrence of water-soluble phosphorus increases, based on Equation 5.

$$(5).\ PSR = (P/31) / [(Fe/56) + (Al/27)]$$

where PSR is the phosphorus saturated ratio, P is phosphorus, Fe is iron, and Al is aluminum (Berkowitz, VanZomeren, Nia , & Sebastian, 2021).

We estimated total soil storage volume on site by applying an assumed root zone depth of 150cm to the site area and further applying a soil bulk density value obtained from the SSURGO Website Soil Survey (USDA, 2020). Olsen STP values were then applied to this soil storage volume to determine the amount of bioavailable phosphorus storage in soils and to compare annual storage volumes relative to flood frequency for a given water year. Similarly, we determined plant phosphorus storage through applying a grass species biomass value of 1,882 kg ha^{-1}, obtained from Gordon (2019), to the site area to calculate total plant biomass. This biomass was further applied to plant phosphorus samples, in g kg^{-1}, to determine total plant phosphorus storage across the site.

Hydrology

We developed a site mass water balance to estimate the volume of water mobilized from soil pores, for use in estimating annual subsurface riparian zone DRP yield contributions as shown in Equation 6. With this calculation it was assumed that all groundwater contributions to the site were from the saturated buffer control structure drain tile line input (Figure 17).

$$(6).\ SPmobile = (DLin + SFin + P) - (DLout + SFout + ET)$$

where SPmobile is mobilized soil pore water, DLin is distribution line input, DLout is distribution line output, SFin is surface inflow, SFout is surface flow out, P is precipitation and ET is evapotranspiration.

Saturated buffer distribution line input and output structure volumes were monitored by Ecosystem Services Exchange in 2018 (Ecosystem Services Exchange, 2021). We then calculated input and output volume to drainage area discharge ratios, based on 2018 inflow and outflow monitoring data and 2018 rainfall data obtained from the Minnesota State Climatology Office Gridded Precipitation Database (MN DNR, 2020). These ratios were applied to 2019 and 2020 drainage area discharges, to estimate inflow and outflow structure volumes for both years. Rainfall depths were applied to the 19.9 ha drainage area to determine drainage area discharge values in each study year.

Similarly, annual rainfall depths obtained from the Minnesota State Climatology Office Gridded Precipitation Database was utilized to calculate precipitation input based on athe.10 ha study site area (MN DNR, 2020). The nearest weather station was location at Fairbault, MN approximately 16 miles southwest of the study site. Evapotranspiration (ET) data was obtained from the USGS AppEEAR web platform, providing ET depths based remotely sensed data collected from the Moderate Resolution Imaging Spectroradiometer (MODIS) NASA Terra satellite and the Penman-Monteith equation (Howell & Evett, 2004; Running, Mu, Zhao, & Moreno, 2019). We applied evapotranspiration depths to the study site area to obtain output ET volumes.

We estimated surface flow inputs to the site from a ratio of surface flow volume to cumulative surface and subsurface runoff volume determined from similar field sites within the southern Minnesota agricultural region. This ratio, determined to be .29 (SD: .18), reflected similar literature values and was calculated as the mean ratio from field data collected at 81 sites over 40 study years, monitored as part of the Discovery Farms Minnesota Program (Discovery Farms MN, 2021; Pease, et al., 2018). Cumulative site runoff within this study was back calculated from drain tile inflow volumes, represented within as the saturated buffer inlet control structure volume, utilizing a value of .71 (SD:

.18). From this estimated cumulative runoff volume, 29% was then assumed to be the input surface flow contribution (Equation 7). We calculated surface flow outputs from the site utilizing the Soil Conservation Service curve number method (Mishra & Singh, 2013).

(7). Upland Surface Runoff Contribution (m3) = .29 * (Saturated Buffer Distribution Line Input Volume (m3) / .71)

We determined flooding frequency on site through use of a stream cross section obtained from the MnTOPO web application, from which a bankfull discharge was calculated (DNR & MNGeo, 2014). We then calculated the bankfull discharge utilizing the Spreadsheet Tools for River Evaluation, Assessment and Monitoring (Mecklenburg & Ward, 2004). We obtained stream gauge data from United States Geological Survey (USGS) gauge number 5355024 south of the study site and adjusted for drainage area based on equations provided by (Ziegeweid, Lorenz, Sanocki, & Czuba, 2015). The site was then assumed to be flooded on days when stream discharge measurements exceeded the calculated bankfull discharge. Although this methodology relies on assumptions, it serves to identify study years with low, medium and high flooding frequency relative to other years.

Results

Dissolved Reactive Phosphorus Contributions

Mean drain tile line DRP concentration measured at the saturated buffer inlet control structure was .07 mg l^{-1}, consistent with data collected at 13 field research sites over 81 study years monitored as part of the Discovery Farms Minnesota Program (Discovery Farms MN, 2021). Distribution line volume contributions to the buffer were 1,992 m^3 and 1,700 m^3 for 2019 and 2020, respectively, calculated as the difference between inlet and outlet control structure volumes (Figure 18, Figure 19). From these concentration and volume calculations, DRP inputs from the saturated buffer line were calculated at 1.28 kg ha^{-1} and 1.09 kg ha^{-1} for 2019 and 2020, respectively.

Mean shallow groundwater well DRP concentrations within the riparian zone were .03 mg l^{-1} and .07 mg l^{-1} with shallow groundwater flow volumes of 7,406 m^3 (range: 4,086-9,413 m^3), and 6,326 m^3 (range: 3,492-8,038 m^3), for 2019 and 2020, respectively (Figure 18, Figure 19). From these concentration and volume calculations, DRP yield contributions from the 0.10 ha riparian zone were calculated at 1.88 kg ha^{-1} and 4.65 kg ha^{-1} for 2019 and 2020, respectively. Accounting for the difference from saturated buffer line inputs, total DRP yield contributions from the .10 riparian buffer zone were calculated at .60 kg ha^{-1} and 3.56 kg ha^{-1} for 2019 and 2020, respectively.

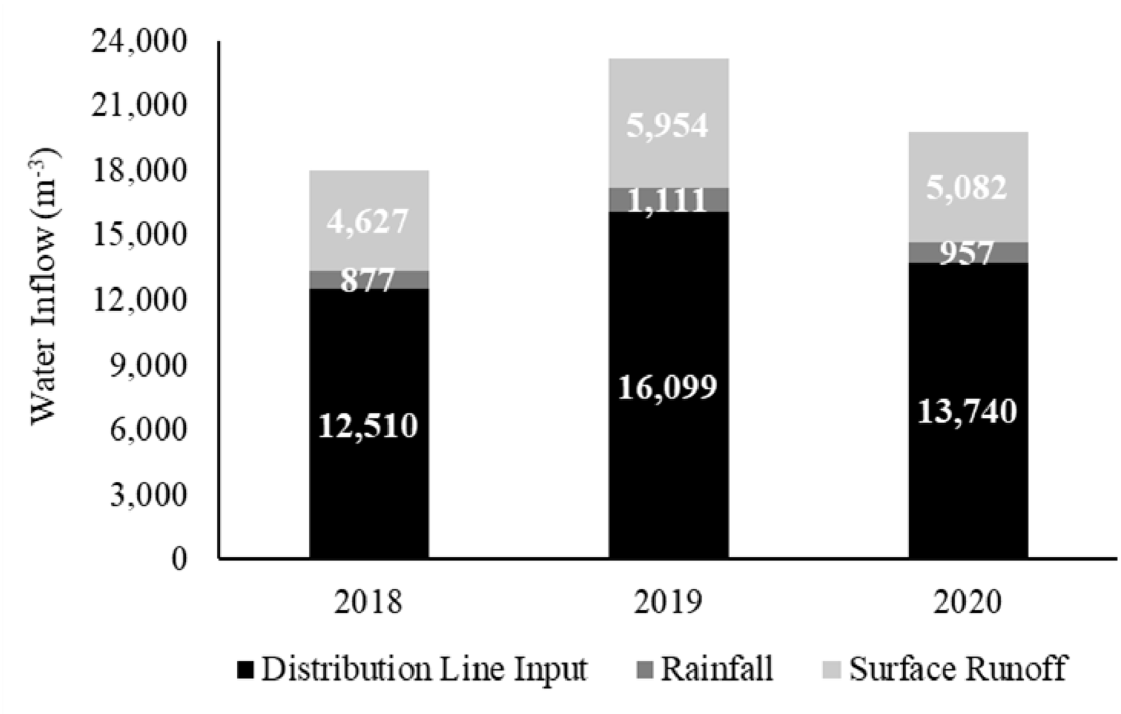

Figure 18. Distribution of saturated buffer inflow hydrology.

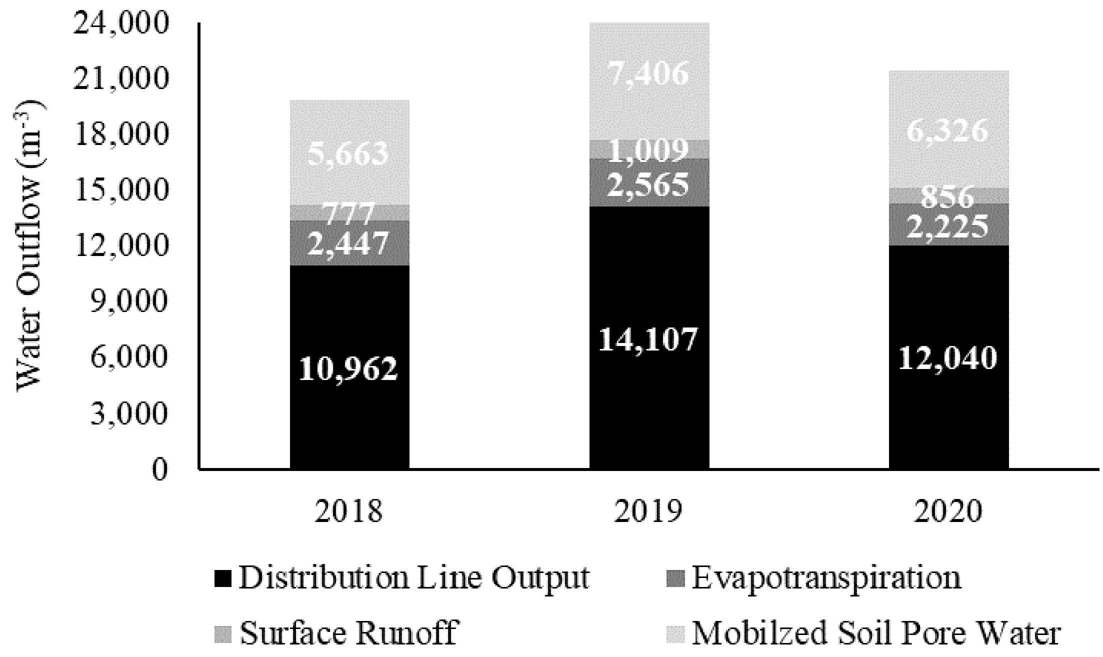

Figure 19. Distribution of saturated buffer outflow hydrology.

Organic Matter and Microbial Community Establishment

A strong positive correlation was identified between soil microbial phosphorus and soil organic matter content (*r* (30) = .74, *p* < 0.001) (Figure 20). In addition, regression analysis determined that soil microbial phosphorus biomass can be predicted from soil organic matter content as displayed in Equation 8 (F(1,28) = 35.68, p = < 0.00, R^2 = .54). As soil microbial phosphorus biomass was associated with greater organic matter content, correlations between organic matter content and STP were also observed (r (86) = 0.66, p < .001) (Figure 21).

(8). Soil Microbial Phosphorus (μg/g) = -.34 + (.26 * Organic Matter (%))

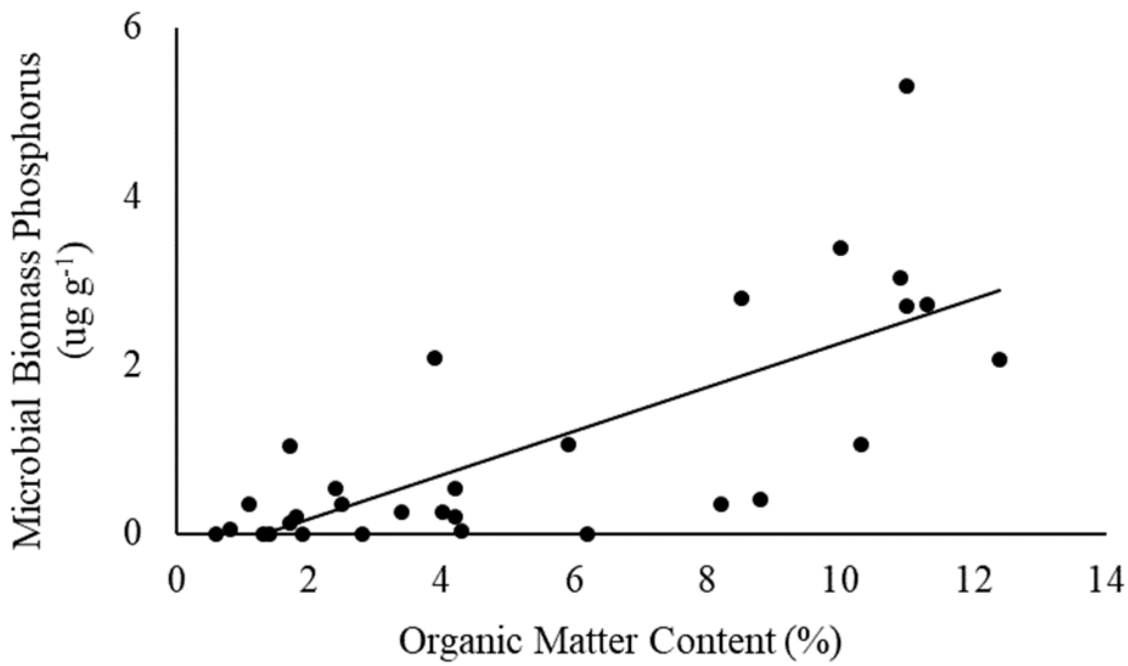

Figure 20. Microbial biomass phosphorus as a function of organic matter.

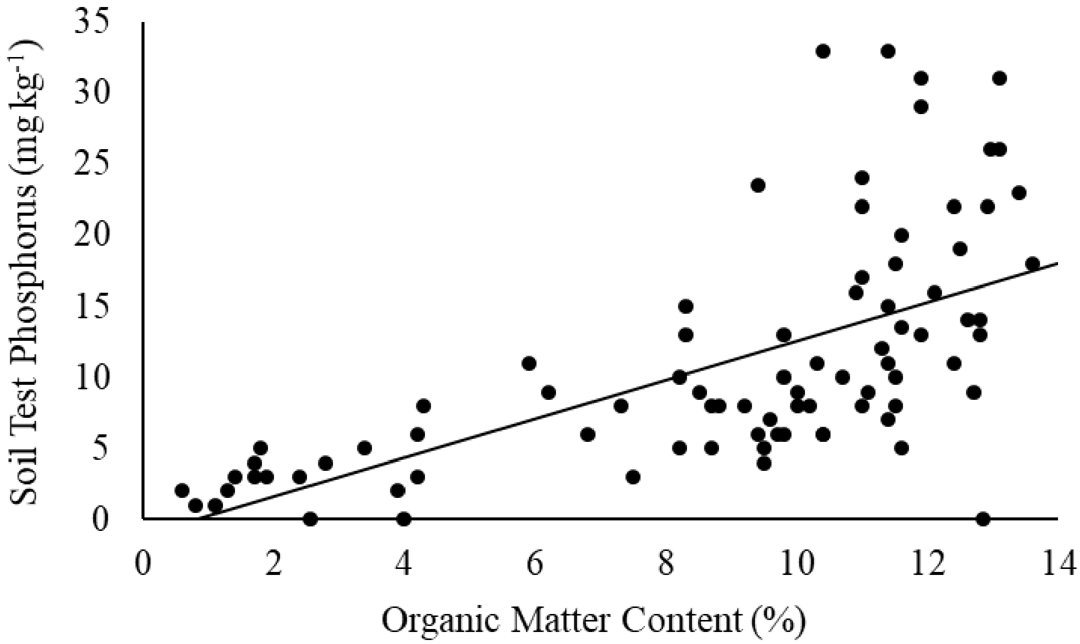

Figure 21. Soil test phosphorus as a function of organic matter.

AMF spore counts completed in 2020 found a mean of 42 spores per gram within the riparian buffer area, and a mean of 20 spores per gram within the adjacent agricultural field. In addition, agricultural field soil samples, those under strip tillage and row-cropping, had a mean organic matter content of 8% and a mean STP value of 7.58 mg kg^{-1} compared to buffer soil samples with a mean organic matter content of 12% and average STP values of 11.63 mg kg^{-1}. These findings suggest that leaving land fallow, or un-altered, contributes to the re-establishment of organic matter and associated soil microbial communities while also increasing phosphorus bioavailability (House & Bever, 2018).

Further, mean AMF root colonization rates across the study site increased in association with mean organic matter content between 2019 and 2021 study years. This increase in mean organic matter content and AMF occurrence was also associated with increased mean STP values (Table 13). Increases in AMF colonization, organic matter accumulation and STP in 2020 and 2021 were associated with low relative flooding frequency over the 2020 and 2021 water years. Alternately, decreases in mean AMF colonization, organic matter content and STP were observed between 2018 and 2019 in association with high flooding frequency during the 2019 water year.

Table 13. Soil and microbial community parameters by study year.

Factor	2018	2019	2020	2021
Arbuscular Mycorrhizal Root Colonization (%)	16.3	1.0	2.6	30.3
Organic Matter Content (%)	10.7	10.7	11.2	11.7
Soil Test Phosphorus (mg kg^{-1})	18.9	9.3	11.6	15.6

Flood Frequency Analysis: Soil Desorption and Microbial Plant Phosphorus Uptake

Occurrence of flooding within riparian zones may influence soil phosphorus desorption potential and subsequent loss through soil porewater (Young & Ross, 2018). Bankfull discharge for Mud Creek (AUID 07040002-558) was determined to be 2,500 m^3 sec^{-1}. This discharge was exceeded, resulting in site flooding on 17, 84, 37 and 2 days in 2018, 2019, 2020 and 2021, respectively. Flooding was assumed to occur on site through calculations, not through consistent field observation (Figure 22). Associated STP values in 2018, 2019 and 2020 were 18.9 mg kg^{-1}, 9.3 mg kg^{-1}, 11.6 mg kg^{-1}, 15.6 mg kg^{-1}, resulting in 19.9 kg of bioavailable phosphorus lost from soil during the 2019 water year, with 4.9 kg and 8.2 kg stored during the 2020 and 2021 water years (Table 13). The greatest bioavailable soil phosphorus losses were observed in 2019 in association with the greatest flooding frequency. Alternately, bioavailable phosphorus storage was observed in 2020 and 2021 with less occurrence of flooding.

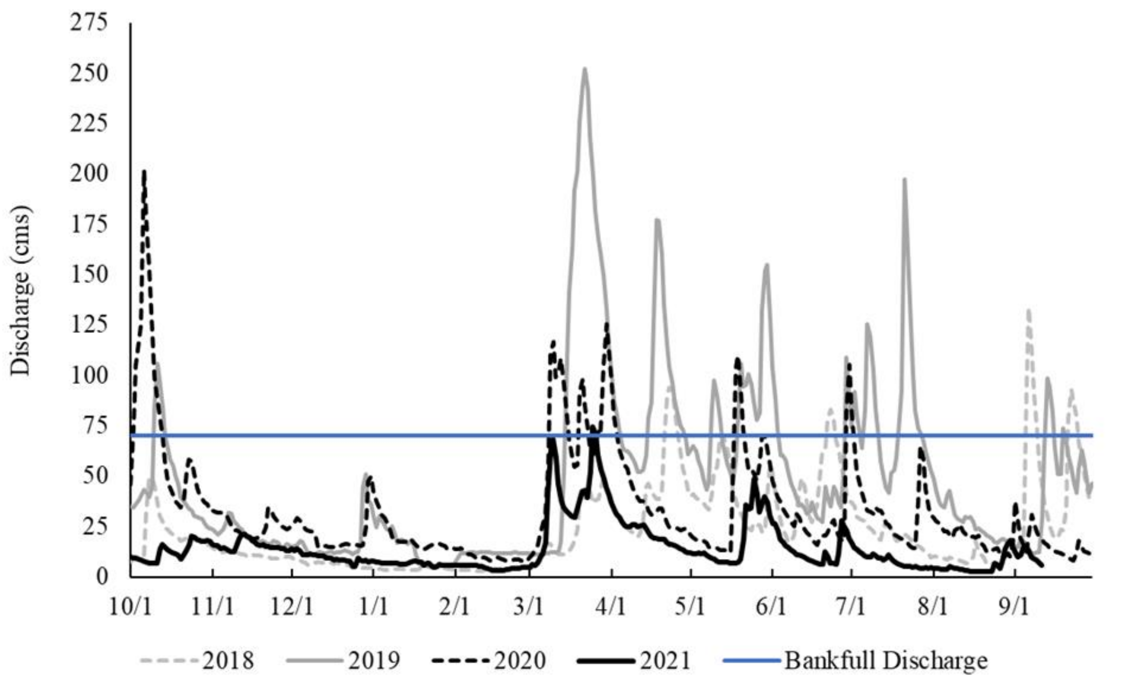

Figure 22. Site water year flood frequency analysis.

Phosphorus soil adsorption potential may also be diminished in the presence of phosphorus saturation, or “legacy phosphorus” accumulation, and the occurrence of high flows (Hoffman, Kjaergaard, Uusi-Kamppa, Christian Brunn Hansen, & Kronvang, 2009; Habibiandehkordi, Lobb, Owens, & Flaten, 2019; Lenhart, Wilson, & Gordon, Factors impacting the variability of effectiveness of agricultural best management practices (BMPs) in Minnesota., 2016). STP values in 2018 were classified as “high” for agricultural settings, and as “optimum” in 2019 and 2020 for agricultural settings by Mallarino, et al. (2013). As such, STP values dropped during the 2019 water year in association with soil phosphorus saturation and flooding and increased during the 2020 and 2021 water years in association with soil phosphorus binding capacity and minimal flooding.

The phosphorus saturation ratio is defined as the ratio between amount of soil phosphorus present in soil and the total capacity of that soil to retain phosphorus. This ratio was calculated as .10 and .06 in 2018 and 2020 respectively, based on the contents of aluminum and iron in available soil samples. Utilizing a water-soluble phosphorus threshold value of .1 for wetland soils in Minnesota, this indicates that soils in 2018 were at or near the change point at which phosphorus concentration in soil solution increases, whereas soils in 2020 were below this phosphorus saturation ratio (Nair, 2014).

Plant Phosphorus Uptake and Harvest

Over the 2019 water year, mean AMF root colonization dropped from 16.30% to .96% with an associated site-wide plant phosphorus content increase of .40 kg ha^{-1} and the greatest occurrence of flooding. Over the 2020 water year, mean AMF root colonization increased from .96% to 2.6% with an associated site-wide plant phosphorus decrease of 0.11 kg ha^{-1} and with lower flooding occurrence. Similarly, over the 2021 water year, mean AMF root colonization increased from 2.6% to 30.3% with an associated site-wide plant phosphorus content decrease of 1.17 kg ha^{-1} in addition to minimal water year flooding occurrence. These trends are consistent with similar studies documenting improved plant phosphorus uptake via AMF pathways under flooding conditions, despite plant root colonization decreases (Bao, Wang, & Olsson, 2019; Miller & Sharitz, 2000; Fougnies, et al., 2008).

Plant harvest may prove a useful strategy for complete system removal of bioavailable phosphorus stored in transient plant and microbial biomass pools (Habibiandehkordi, Lobb, Owens, & Flaten, 2019; Richardson & Marshall, 1986).

Bioavailable phosphorus stored in above ground biomass on site at the end of each growing season was calculated as 5.57 kg ha^{-1}, 5.97 kg ha^{-1}, 5.86 kg ha^{-1}, and 4.69 kg ha^{-1} for 2018, 2019, 2020, and 2021, respectively. Harvest of this material, as such would serve to remove a mean annual value of .56 kg of DRP from the site each year that may otherwise release back into the system by nutrient translocation and microbial decomposition during plant senescence (Kroger, Holland, Moore, & Cooper, 2007; Menon & Holland, 2014).

Arbuscular Mycorrhizal Fungi Soil Amendment

Data from this study brings into questions the applicability of AMF soil amendment within this setting as related to phosphorus management. A strong correlation was not identified between AMF plant root colonization and STP (r (28) = .29, p = .16) or between AMF plant root colonization and plant phosphorus content (r (20) = .08, p = .77); disproving the hypothesis that increased AMF plant root colonization would be associated with greater plant phosphorus uptake and lower STP. Occurrence of flooding likely also contributes to amendment wash-outs as well as frequent AMF root colonization destruction, as AMF establishment benefits under lower water regimes (Hu, Chen, Vosatka, & Vymazal, 2020).

Establishment of native perennial grasses was not successful on-site following three growing seasons post-seeding, disproving the hypothesis that AMF amendment would contribute to perennial grass establishment. During the 2020 and 2021 growing seasons, dominant grass cover was from quack grass (*Elymus repens)*; a non-native, noxious cool season grass season species, characterized on site by mean relative cover of

29% (n = 32, SD = 25%). The plant community was also characterized by minor establishment of switchgrass with a mean relative cover of 5% (n = 32, SD = 9%), and dominance of other weedy, noxious species.

Discussion

Contributions of DRP from the saturated buffer field site were demonstrated with an estimated annual loss rate of 2.08 kg ha^{-1}. Multiple studies have reported similar DRP losses from vegetated riparian buffers ranging from 1.00 to 8.30 kg ha^{-1} with documented DRP removal efficiencies of -71% to -36% (Hoffman, Kjaergaard, Uusi-Kamppa, Christian Brunn Hansen, & Kronvang, 2009; Stutter, Langan, & Lumsdon, 2009; Kieta, Owens, Lobb, Vanrobaeys, & Flaten, 2018). A similar 2015 saturated buffer study documented DRP groundwater contributions from multiple study sites across the Midwest as observed through higher DRP shallow groundwater well concentrations relative to inlet control structure concentrations (Utt, Jaynes, & Albertsen, 2015).

A dominant factor driving DRP loss from agricultural riparian buffers as demonstrated within this study, is the re-accumulation of organic matter and associated microbial phosphorus biomass storage (Dodd, Sharpley, & Berry, Organic phosphorus can make an important contribution to phosphorus loss from riparian buffers, 2018). As organic matter is re-established within previously disturbed systems, fungal colonization and phosphorus bio-availability increase, due to soil microorganism's role in transforming organic phosphorus into soluble and inorganic forms (Salas, Elliott, Westfall, Cole, & Six, 2003). Similar studies have documented greater microbial biomass in association with organic matter as well as less organic matter content in distributed fields (Bunemass, Marschner, Smernik, Conyers, & McNeill, 2008; Gottshall, Cooper , & Emery, 2017). Roberts et. al., (2012) documented greater soil phosphorus saturation and biological activity in riparian buffer soils relative to adjacent agricultural fields. While this re-establishment of microbial phosphorus biomass is beneficial for soil phosphorus retention and bioavailability, this stored phosphorus may be lost from riparian zones under flooding or soil phosphorus saturation (Roberts, et al., 2013).

An additional driver of DRP loss demonstrated within this study was that of flooding associated within riparian areas. Under anoxic conditions, those associated with flooding, microbial induced dissolution and release of soil adsorbed phosphorus particles is promoted, thus releasing DRP into solution (Pant & Reddy, 2001; Kumaragamage, et al., 2019). Phosphorus loss from microbial biomass storage is further exacerbated under drying and rewetting cycles as often is observed in riparian settings (Khan, Blackwell, & Busquets, 2019). These findings are consistent with trends on site were STP and organic matter increases were observed following years with minimal flooding and decreases observed following higher flood years.

Increased plant phosphorus uptake was observed on-site under flooded conditions, likely also contributing to decreased soil phosphorus levels observed between 2018-2019. Similarly, decreased plant uptake under minimal flooding likely contributed to increased soil phosphorus levels observed between 2019-2020 and between 2021-2021. These trends are consistent with similar studies documenting improved plant phosphorus uptake via AMF pathways under flooding conditions, despite plant root colonization decreases (Bao, Wang, & Olsson, 2019; Miller & Sharitz, 2000; Fougnies, et al., 2008). Stevens et al (2002) also documented minimal influence of AMF colonization on plant phosphorus supply under inundated conditions. These trends may provide evidence for the potential of water level management within saturated buffers as a technique to increase plant phosphorus supply.

Plant uptake, however, is a transient phosphorus pool requiring further mitigation for complete removal from the system (Richardson & Marshall, 1986). Within this study, a mean value of .56 kg ha^{-1} was calculated as the potential phosphorus removal through plant harvest practice on site. While this rate is in exceedance of estimated annual bioavailable phosphorus contributions from the site, continued plant harvest may serve to lower STP over time. Alsadi (2019) documented similar phosphorus removal rates of 3.2 kg ha^{-1} associated with STP of 20%. Gordon (2019) documented approximately 80% lowering in soil phosphorus test values through plant harvest over five years

A final driver of DRP loss is that of soil phosphorus accumulation and soil desorption and adsorption potential. Within this study, the positive linear relationship between soil organic matter content and STP becomes weaker as soils become more organic. As organic matter content on site reaches 10 percent, the approximate divide between mineral and organic soils, disproportionate increases in STP are observed (Huang, Patel, Santagata, & Bobet, 2009). These findings are consistent with similar studies documenting organic matter content thresholds for phosphorus adsorption (Guppy, Menzies, Moody, & Blamey, 2005; Yang, Chen, & Yang, 2019). Additionally, natural wetlands, particularly peatlands, tend to have high organic matter content with low nutrient availability; however, contain high STP when drained or present with altered hydrology (Daly, Jeffrey, & Tunney, 2006; Negassa, Michalik, Klysubun, & Leinweber, 2020).

The potential for phosphorus adsorption has also been shown to decrease in the presence of phosphorus saturation, or the point at which nutrient loading surpasses soil capacity for phosphorus retention (Hoffman, Kjaergaard, Uusi-Kamppa, Christian Brunn Hansen, & Kronvang, 2009; Habibiandehkordi, Lobb, Owens, & Flaten, 2019; Berkowitz, VanZomeren, Nia , & Sebastian, 2021). This was demonstrated on site through determination of the phosphorus saturation ratio or the potential for soil phosphorus retention based on aluminum and iron content (Nair, 2014). As this ratio neared threshold values in 2018 STP reductions were observed in association with flooding. As such, monitoring of STP values may prove a useful management technique to identify phosphorus loss risk, monitor plant harvest or water level management effectiveness, or to determine suitable or unsuitable locations for management practice placement (Sims, Edwards, Schoumans, & Simard, 2000; Duncan, 2017).

The final phosphorus loss management strategy explored within the study was the use of AMF amendment. Although shown effective for perennial grass establishment and increased plant nutrient uptake in upland settings, this management practice was not found to be applicable in the riparian setting (Koziol & Bever, 2017; Koziol & Bever, 2015). Strong correlations where not identified between AMF plant root colonization and STP or between AMF plant root colonization and plant phosphorus content. Flooding frequency on site also likely served to wash-out applied amendment and destroy establishing AMF plant root colonization.

The plant community within the riparian zone also provided evidence against the applicability of an AMF amendment in riparian settings. It has been shown that cool season grasses such as quack grass, the dominant grass present on-site, can establish successfully without AMF relationships (Koziol & Bever, 2017; Habibiandehkardi, Lobb, Sheppard, Flaten, & Owens, 2017). Additionally, a demonstrated benefit of AMF amendment in other studies is that of noxious weed suppression, which was not observed on-site following AMF amendment in 2018 (Lee, Tu, Chen, & Hu, 2014). The role of AMF in wetland settings, is still not well understood with further research required to understand responses to plant species, soil type, seasonality, and hydrologic regime (Calheiros, Pereira, Franco, & Paula, 2019).

Conclusion

The primary objective of this study was to demonstrate DRP loss from a saturated riparian buffer located adjacent to an agricultural field in Southern, Minnesota. Associated objectives included documentation of DRP loss drivers and exploration of DRP loss mitigation techniques. Organic matter and microbial community structure re-establishment were identified as a primary DRP loss driver, through increased phosphorus availability resulting from microbial dissolution, soil desorption and conversion to organic forms. Flooding frequency was also documented as a dominant DRP loss driver with the greatest soil losses observed under the greatest flooding frequency, resulting from increased soil desorption and microbial activity under anoxic conditions. Soil phosphorus accumulation, further contributing to losses under high flows and reducing soil phosphorus adsorption potential, was identified as the final DRP loss driver on site. This highlights the need for STP sampling as a phosphorus loss management technique.

Use of an AMF soil amendment to increase plant root colonization and associated perennial grass establishment success leading to greater plant phosphorus uptake and lower STP was explored as the primary study hypothesis. This management technique proved unapplicable within this study as strong correlations were not identified between AMF colonization rates, plant phosphorus uptake and STP values. Further, on-site flooding was likely to contribute to amendment washout and destruction of established AMF plant root colonization. Flooding conditions on-site, however, were observed in association with greater plant phosphorus uptake rates. As plant phosphorus storage is a transient pool, plant harvest was proposed as an effective management technique for system phosphorus removal.

Project Conclusions

Under the first project objective, to quantify current and target dissolved reactive phosphorus (DRP) yields from Southern Minnesota agricultural fields; DRP loss rates from agricultural fields were determined to be 0.49 kg ha^{-1}, as consistent with other monitoring efforts in the region. Based on this yield rate a target DRP yield of 0.29 kg ha^{-1} would need to be achieved to meet the 45% phosphorus reduction goals outlined in the Minnesota Nutrient Reduction Strategy. In order to reach these target yields, the mechanisms driving DRP losses, and the effectiveness of current management strategies must be well understood.

Project objective two called for the characterization of dominant mechanisms driving DRP losses from agricultural fields. In addition to demonstrating the importance of both pathways for DRP loss mitigation, this study identified dominant site condition and management mechanisms driving both drain tile and surface DRP losses. Dominant drivers of drain tile DRP loss were identified as manure application rate, number of tillage passes and soil test phosphorus (STP). Dominant drivers of surface DRP loss were identified as cumulative phosphorus application rate and STP. Dominant drivers of STP accumulation included manure application rate, number of tillage passes, organic matter content, clay content, soil pH and cover crop implementation. Conservation tillage as well as fertilizer and manure application rates were found to contribute to increased DRP yields, with cover crops contributed to increased STP accumulation DRP yields.

Cover crop use was further explored under objective three of this study, to demonstrate the effectiveness or inefficiency of common best management practices (BMPs) at DRP removal. The implementation of cover crops was found to contribute to

75% and 63% concentration reductions for total phosphorus (TP) and DRP drain tile concentration reductions, respectively; although found to contribute specifically to STP accumulation in a separate chapter. Drain tile nutrienAnnual yield reductions provided from cover crop implementation were quantified at 120 g ha^{-1} and 70 g ha^{-1} for TP and DRP drain tile load reductions, respectively, at per unit removal costs of \$0.99 g^{-1} ha^{-1} and \$1.69 g^{-1} ha^{-1}. Additional BMP types explored under project objective three included riparian buffers and treatment wetlands.

DRP contributions were identified from a southern Minnesota riparian buffer demonstration site at an estimated annual rate of 2.08 kg ha^{-1}. Further contributing to project objective two, the dominant conditions driving these riparian zone losses were identified as re-accumulation of organic matter, flooding frequency and soil phosphorus saturation. Strategies explored for mitigation of riparian DRP loss included plant harvest and soil test phosphorus monitoring. Although DRP loss rates were in exceedance of potential plant harvest DRP removal rates, this practice may have potential to lower soil test phosphorus and subsequent losses if implemented over a period of time. An additional management technique explored as part of project objective four, to explore novel management strategies for DRP yield reductions, was the use of an AMF soil amendment. Although this management strategy was not found to applicable for phosphorus management in riparian BMPs, it may merit research for use in other BMP types.

An additional novel management strategy explored under project objective four was the implementation of an agricultural BMP treatment train. Within this study, the implementation of cover crops was found to improve the performance of a downstream treatment wetland demonstrated through the a per unit TP cost removal reduction of \$211 kg^{-1}. Lastly, project findings demonstrate the need to implement both source and edge-of-field management actions to meet target DRP loads. This need was identified through the development of a phosphorus mass balance at seven field research sites, demonstrating greater system DRP outputs relative to inputs. As such, the implementation of source management practices alone, particularly that of fertilizer reductions, will not be sufficient to meet DRP yield goal alone.

As previously demonstrated, this work serves to quantify current and target DRP yields, to characterize dominant DRP loss mechanisms, to document the effectiveness of current agricultural BMPs and to explore novel agricultural management techniques. Results from this work will contribute to the development of realistic and obtainable water quality goals and implementation scenarios aimed at reducing DRP yields from the agricultural landscape. Specifically, these reduction scenarios may further contribute to the goals set forth within the Minnesota Nutrient Reduction strategy to account for both dissolved and total phosphorus losses.

References

Adhikari, P. L., White, J. R., Maiti, K., & Nguyen, N. (2015). Phosphorus speciation and sedimentary phosphorus release from the Gulf of Mexico sediments: Implications for hypoxia. *Estuarine, Coastal and Shelf Science, 164*(5), 77-85.

Alsadi, N. (2019). *Treatment Wetland Vegetation Harvesting for Phosphorus Removal in Upper Midwest Agricultural Watersheds.* Retrieved from https://hdl.handle.net/11299/211693

Anderson, D. M., Gilbert, P. M., & Burkholder, J. M. (2002). Harmful algal blooms and eutrophication: nutrient sources, composition, and consequences. *Estuaries, 25*(4), 704-726.

Anderson, W. P., Wall, D., & Olson, J. L. (2016). Minnesota nutrient reduction strategy. *10th International Drainage Symposium Conference* (pp. 1-9). Minneapolis, Minnesota: American Society of Agricultural and Biological Engineers.

Ansari, A. A., Singh, G. S., Lanza, G. R., & Rast, W. (2010). *Eutrophication: causes, consequences and control.* New York: Springer Science & Business Media.

Apfelbaum, S. I., Eppich, J. D., Price, T., & Sands, M. (1995). The priarie crossing project: Attaining water quality and stormwater management goals in a conservaiton development. *In proceedings of the national Symposium of Using Ecological Restoraion to Meet Clean Water Act Goals*, (pp. 33-38). Chicago, Illinois.

Baker, D. B., Confesor, R., Ewing, D. E., Johnson, L. T., Kramer, J. W., & Merryfield, B. J. (2014). Phosphorus loading to Lake Erie from the Maumee, Sandusky and Cuyahoga rivers: The importance of bioavailability. *Journal of Great Lakes Research, 40*(3), 502-517.

Baker, D. B., Johnson, L. T., Confesor, R. B., & Crumrine, J. P. (2017). Vertical stratification of soil phosphorus as a concern for dissolved phosphorus runoff in the Lake Erie Basin. *Journal of Environmental Quality, 46*(6), 1287-1295.

Baker, J. L., Campbell, K. L., Johnson, H. P., & Hanway, J. J. (1975). Nitrate, phosphorus, and sulfate in subsurface drainage water. *Journal of Envrionmental Quality, 4*, 406-412.

Ball, C. B., Murray, R., Lapen, D., Topp, E., & Bruin, A. (2012). Phosphorus and sediment loading to surface waters from liquid swine manure application under different drainage and tillage practices. *Agricultural Water Management, 104*, 51-61.

Bao, X., Wang, Y., & Olsson, P. (2019). Arbuscular mycorrhiza under water -- carbon-phosphorus exchange between rice and arbuscular mycorrhizal fungi under different flodding regimes. *Soil Biology and Biochemistry, 129*, 169-177.

Barber, N., Reaney, S. M., Barker, P. A., Benskin, C., Burke, S., Cleasby, W., . . . Quinn, P. F. (2016). The treatment train approach to reducing non-point pollution from agriculture. *In AGU Fall Meeting Abstracts,* (pp. H13D-1396). San Francisco, California.

Barr Engineering Co. (2004). *Detailed Assessment of Phosphorus Sources to Minnesota Watersheds.* Minnesota Pollution Control Agency.

Bergtold, J. S., Ramsey, S., Maddy , L., & Williams, J. R. (2017). A review of economic considerations for cover crops as a conservation practice. *Renewable Agriculture and Food Systems, 34*(1), 1-15.

Berkowitz, J. F., VanZomeren, C. M., Nia , H. R., & Sebastian, K. J. (2021). *An Evaluation of Soil Phosphorus Storage Capacity (SPSC) at Proposed Wetland Restoration Locations in the Western Lake Erie Basin.* Vicksburg, MS: U.S. Army Engineer Research and Development Center.

Blanco-Canqui, H., Ruis, S. J., Proctor, C. A., Creech, C. F., Drewnoski, M. E., & Redfearn, D. D. (2020). Harvesting cover crops for biofuel and livestock production: Another ecosystem service? *Agronomy Journal, 112*(4), 2373-2400.

Blanco-Canqui, H., Shaver, T. M., Lindquist, J. L., Shapiro, C. A., Elmore, R. W., Francis, C. A., & Herbert, G. W. (2015). Cover crops and ecosystem services: insights from studies in temperate soils. *Agronomy Journal, 107*(6), 2449-2474.

Blann, K. L., Anderson, J. L., Sands, G. R., & Vondracek, B. (2009). Effects of agricultural drainage on aquatic ecosystems: a review. *Critical reviews in environmental science and technology, 39*(11), 909-1001.

Boardman, E., Danesh-Yazdi, M., Foufoula-Georgiou, E., Dolph, C. L., & Finlay, J. C. (2019). Fertilizer, landscape features and cliamte regulate phosphorus retention and river export in diverse Midwestern watersheds. *Biogeochemistry, 146*(3), 293-309.

Bohrer, K. E., Friese, C. F., & Amon, J. P. (2004). Seasonal dynamics of arbuscular mycorrhizal fungi in differing wetland habitats. *Mycorrhiza, 14*(5), 329-337.

Brookes, P. C., Powlson, D. S., & Jenkinson, D. S. (1982). Measurement of microbial biomass phosphorus in soil. *Soil biology and biochemistry, 14*(4), 319-329. doi:https://doi.org/10.1016/0038-0717(82)90001-3

Bunemass, E. K., Marschner, P., Smernik, R. J., Conyers, M., & McNeill, A. M. (2008). Soil organic phosphorus and microbial community composition as affected by 26 years of different management strategies. *Biology and Fertilizer of Soils, 44*(5), 717-726. doi:https://doi.org/10.1007/s00374-007-0254-2

Caduto, M. J. (1990). *Pond and brook.* Hanover, NH: University Press of New England.

Calheiros, C. S., Pereira, S. I., Franco, A. R., & Paula, C. M. (2019). Diverse Arbuscular Myocrhizal Fungi (AMF) Communities Colonize Plants Inhabiting a Constructed Wetland for Wasterwater Treatment. *Water, 11*(8), 1535. doi:https://doi.org/10.3390/w11081535

Carlson, S., & Stockwell, R. (2013). Research Priorities for Advancing Adoption of Cover Crops in Agricultural-Intensive Regions. *Journal of Agriculture, Food Systems, and Community Develpment, 3*(4), 125-129.

Chandrasoma, J. M., Christianson, R. D., & Christianson, L. E. (2019). Saturated Buffers: What is Their Potential Impact across the US Midwest. *Agricultural and Environmental Letters, 4*(1), 180059. doi:https://doi.org/10.2134/ael2018.11.0059

Chaurasia, A., & Harel, O. (2012). Using AIC in Multiple Linear Regression framework with Multiply Imputed Data. *Health Services and Outcomes Research Methodology, 12*(2), 219-233.

Choudhury, T., Robertson, W. D., & Finnigan, D. S. (2016). Suspended Sediment and Phosphorus Removal in a Woodchip Filter System Treating Agricultural Wash Water. *Journal of Environmental Quality, 45*(3), 796-802.

Christianson, L. E., Frankenberger, J., Hay, C., Helmers, M. J., & Sands, G. (2016). *Ten Ways to Reduce Nitrogen Loads from Drained Cropland in the Midwest.* University of Illinois Extension.

Christianson, L., Tyndall, J., & Helmers, M. (2013). Financial comparison of seven nitrate reduction strategies for Midwestern agricultural drainage. *Water Resources and Economics, 2*, 30-56.

Christianson, R., Fox, J., Law, N., & Wong, C. (2021). Effectiveness of Cover Crops on Water Pollutant Reduction from Agricultural Areas. *Transactions of the ASABE, 64*, 1007-1017.

Chun, J. A., & Cooke, R. A. (2008). Technical Note: Calibrating Agridrain Water Level Control Structures Using Feneralized Weir and Orifice Equations. *24*(5), 595-602.

Cober, J. R., Macrae, M. L., & Van Eerd, L. L. (2019). Winter Phosphorus Release from Cover Crops and Linkages with Runoff Chemisty. *Journal of Environmental Quality, 48*(4), 907-914.

Cox, G. (1990). *Laboratory manual of genearal ecology 6th Ed.* Dubuque, Iowa: William C. Brown.

Daly, K., Jeffrey, D., & Tunney, H. (2006). The effect of soil type on phosphorus sorption capacity and desorption dynamics in Irish grassland soils. *Soil Use and Management, 17*(1), 12-20.

Da-Peng, L., & Huang, Y. (2010). Sedimentary phosphorus fractions and bioavailability as influenced by repeated sediment resuspension. *Ecological Engineering, 36*(7), 958-962.

Daryanto, S., Wang, L., & Jacinthe, P. A. (2017). Meta-analysis of phosphorus loss from no-till soils. *Journal of Environmental Quality, 46*(5), 1028-1037.

De Groot, C. J., & Golterman, H. L. (1993). On the presence of organic phosphate in some Camargue sediments: evidence for the importance of phytate. *Hydrobiologia, 252*(1), 117-126. doi:https://doi.org/10.1007/BF00000133

de Vries, F. T., & Shade, A. (2013). Controls on osil microbial community stability under climate change. *Frontiers in Microbiology, 4*, 265. doi:ttps://doi.org/10.3389/fmicb.2013.00265

Discovery Farms MN. (2021). *About Discovery Farms MN*. Retrieved August 19, 2021, from https://discoveryfarmsmn.org/about-us/

Djodjic, F., Bergstrom, L., Ulen, B., & Shirmohammadi, A. (1999). Mode of transport of surface-applied phosphorus-33 through a clay and sandy soil. *Journal of Environmental Quality, 28*(4), 1273-1282.

DNR, M., & MNGeo. (2014, January). *MnTOPO*. Retrieved January 3, 2020, from http://arcgis.dnr.state.mn.us/maps/mntopo/

Dodd, R. J., & Sharpley, A. N. (2015). Conservation practice effectiveness and adoption: unintended consequences and implicaitons for sustainable phosphorus management. *Nutrient Cycling in Agroecosystems, 104*, 373-392.

Dodd, R. J., Sharpley, A. N., & Berry, L. G. (2018). Organic phosphorus can make an important contribution to phosphorus loss from riparian buffers. *Agricultural and Environmental Letters, 3*(1), 18002. doi:https://doi.org/10.2134/ael2018.01.0002

Dolph, C. L., Boardmann, E., Danesh-Yazdi, M., Finlay, J. C., Hansen, A. T., Baker, A. C., & Dalzell, B. (2019). Phosphorus transport in intensively managed watersheds. *Water Resources Research, 55*(11), 9148-9172.

Drury, C. F., Tan, C. S., Welacky, T. W., Reynolds, W. D., Zhang, T. Q., Oloya, T. O., . . . Gaynor, J. D. (2014). Reducing Nitrate Loss in Tile Drainage Water with Cover Crops and Water-Table Management Systems. *Journal of Environmental Quality, 43*, 587-598.

Duncan, E. W. (2017). Linking soil phosphorus to dissolved phosphorus losses in the Midwest. *Agricultural & Environmental Letters, 2*(1), 170004.

Eckhard, G., Marschner, H., & Jakobsen, I. (1995). Role of arbuscular mycorrhizal fungi in uptake of phosphorus and mitrogen from soil. *Critical Reviews in Biotechnology, 15*(3-4), 257-270.

Ecosystem Services Exchange. (2021). *About Us*. Retrieved September 20, 2021, from http://www.ecoexch.com/about-us/

Ellison, M. E., & Michael , B. T. (2006). Particulate phosphorus bioavailability as a function of stream flow and land cover. *Water research, 40*(6), 1258-1268.

EPA. (2020, December 11). *Mississippi River/Gulf of Meixoc Hypoxia Task Force*. Retrieved September 4, 2021, from https://www.epa.gov/ms-htf/hypoxia-task-force-nutrient-reduction-strategies

Erik, S., Gillette, T., Blann, K., Coburn, M., Hoppie, B., & Rhees, S. (2018). *Drain TIles and Groudwater Resources: Understanding the Relations.* Minnesota Ground Water Association. Retrieved from https://www.mgwa.org/documents/whitepapers/Drain_Tiles_and_Groundwater_Resources.pdf

Fang, H., Cui, Z., He, G., Huang, L., & Chen, M. (2017). Phosphorus adsorption onto clay minerals and iron oxide with consideration of heterogeneous particle morphology. *Science of The Total Environment, 605*, 357-367.

Farzadfar, S. J., Knight, D. J., & Congreves, K. A. (2021). Rye cover crop improves vegetable crop nitrogen use efficiency and yields in a short-season growing region. *Canadian Journal of Plant Science*.

Fougnies, L., Renciot, S., Muller, F., Plenchette, C., Prin, Y., Miana De Faria, S., . . . Ba, A. (2008). Arbuscular mycorrhizal colonization and nodulation improve flooding tolerance in Pterocarpus officinalis Jacq. seedlings. *Biological Nitrogen Fixation: Towards Poverty Alleviateion through Sustainable Agriculture*, 133-134.

Franzluebbers, A. J. (2011). *Stratification of Soil Porosity and Organic Matter. In: Gliński J., Horabik J., Lipiec J. (eds) Encyclopedia of Agrophysics. Encyclopedia of Earth Sciences Series.* Dordrecht: Springer.

Godsey, S. E., Kirchner, J. W., & Clow, D. W. (2009). Concentration-discharge relationships reflect chemostatic characteristics of US catchments. *Hydrological Process: An International Journal, 23*(13), 1844-1864.

Gordon, B. A. (2019). *Hydrology, Nutrient Removal, and Cost Effectiveness of Small, Edge-of-Field Tile Drainage Treatment Wetlands.* doi:https://hdl.handle.net/11299/206271

Gordon, B. A., Lenhart, C., Peterson, H., Gamble, J., Nieber, J., Current, D., & Brenke, A. (2021). Reduction of nutrient loads from agricultural subsurface drainage water in a small, edge-of-field constructed treatment wetland. *Ecological Engineering, 160*(1), 106128.

Gottshall, C. B., Cooper , M., & Emery, S. M. (2017). Activity, diversity and function of arbuscular mycorrhizae vary with changes in agricultural management intensity. *Agriculture, Ecosystmes and Environment, 241*(1), 142-149. doi:https://doi.org/10.1016/j.agee.2017.03.011

Guppy, C. N., Menzies, N. W., Moody, P. W., & Blamey, F. P. (2005). Competitive sorption reactions between phosphorus and organic matter in soil: a review. *Soil Research, 43*(2), 189-202.

Habibiandehkardi, R., Lobb, D. A., Sheppard, S. C., Flaten, D. N., & Owens, P. N. (2017). Uncertainties in vegetated buffer strip function in controlling phosphorus export from agricultural land in the Canadian prairies. *Environmental Science and Pollution Research, 24*(22), 18372-18383. doi:10.1007/s11356-017-9406-6

Habibiandehkordi, R., Lobb, D., Owens, P., & Flaten, D. (2019). Effectiveness of vegetated buffer stips in controlling legacy phosphorus exports from agricultural land. *Journal of environmental quality, 48*(2), 314-321.

Hamilton, A. V., Mortensen, D. A., & Kammerer Allen, M. (2017). The state of the cover crop nation and how to set realistic future goals for popular conservation practice. *Journal of Soil and Water Conservation, 72*(5), 111A-115A.

Hanrahan, B. R., King, K. W., Duncan, E. W., & Shedekar, V. S. (2021). Cover crops differentially influenced nitrogen and phoshorus loss in tile drainage and surface runoff from agricultural fields in Ohio, USA. *Journal of Environmental Management, 293*, 112910.

Hanrahan, B., Tank, J., Christopher, S., Mahl, U. H., Trentman, M. T., & Royer, T. V. (2018). Winter crover crops reduce nitrate loss in an agricultural watershed in the central U.S. *Agriculture, Ecosystems and Environment, 265*(1), 513-523.

Hille, S., Graeber, D., Kronvang, B., Rubaek, G. H., Onnen, N., Molina-Navarro, E., . . . Stutter, M. I. (2019). Management options to reduce phosphorus leaching from vegtated buffer strips. *Journal of Environmental Quality, 48*(2), 322-329.

Hoffman, C. C., Kjaergaard, C., Uusi-Kamppa, J., Christian Brunn Hansen, H., & Kronvang, B. (2009). Phosphorus retention in riparian buffers: review of their efficiency. *Journal of environmental quality, 38*(5), 1942-1955.

Hook, P. B. (2003). Sediment Retention in Rangeland Riparian Buffers. *Journal of Environmental Quality, 32*(3), 1130-1137.

Horst, W. J., Kamh, M., Jibrin, J. M., & Chude, V. O. (2001). Agronomic measures for increasing P availability to crops. *Pland and Soil, 237*, 211-233.

House, G. L., & Bever, J. D. (2018). Distrubance reduces the differentiation of mycorrhizal funal communities in grasslands along a precipitation gradient. *Ecological Applications, 28*(3), 736-748. doi: https://doi.org/10.1002/eap.1681

Houston Engineering. (2015). *Treatment Train Memo.* Minnesota Board of Water and Soil Resources. Retrieved from https://ptmapp.bwsr.state.mn.us/files/BMPsandTreatment_Memo_FINAL_Approved.pdf

Houston Engineering. (2016). *(PTMAPP): Theory and Development Documentation.* Minnesota Board of Water and Soil Resources. Retrieved from https://ptmapp.bwsr.state.mn.us/files/04052016_PTMA_Theory_Report.pdf

Howell, T. A., & Evett, S. R. (2004). The Penman-Monteith Method. *Evapotranspiration: Determination of Consumptive Use in Water Rights Proceedings* (p. Section 3). Denver, CO.: Continuing Legal Education in Colorado, Inc.

Hu, S., Chen, Z., Vosatka, M., & Vymazal, J. (2020). Arbuscular mycorrhizal fungi colonization and physiological functions toward wetland plants under different water regimes. *Science of the Total Environment, 716*, 137040. doi:https://doi.org/10.1016/j.scitotenv.2020.137040

Huang, P.-T., Patel, M., Santagata, M. C., & Bobet, A. (2009). *Classificatin of Organic Soils.* West Lafayette, Indiana: Joint Transportation Research Program, Indiana Department of Transportation and Purdue University.

Jacobson, L. M., David, M. B., & Drinkwater, L. E. (2011). A Spatial Analysis of Phosphorus in the Mississippi River Basin. *Journal of Environmental Quality, 40*(3), 931-41.

Jakobsen, I., Smith, S. E., Gronlund, M., & Smith, A. F. (2011). Roles of arbuscular mycorrhizas in plant phosphorus nutrition: interactions between pathways of phosphorus uptake in arbuscular mycorrhizal roots have important implications for understanding and manipulating plant phosphorus acquisition. *Plant Physiology, 156*(3), 1050-1057.

Jarvie, H. P., Johnson, L. T., Sharpley, A. N., Smith, D. R., Baker, D. B., Bruulsema, T. W., & Confesor, R. (2017). Increased soluble phosphorus loads to Lake Erie: unintended consequences of conservation practices? *Journal of Environmental Quality, 46*(1), 123-132.

Jarvie, H. P., Sharpley, A. N., Spears, B., Buda, A. R., May, L., & Kleinman, P. J. (2013). Water quality remediation faces unprecedented challenges from "legacy phosphorus". *Environmental Science & Technology, 47*(16), 8997-8998.

Jaynes, D. B., & Isenhart, T. M. (2014). Reconnecting Tile Drainage to Riparian Buffer Hydrology for Enhanced Nitrate Removal. *Journal of environmental quality, 43*(2), 631.

Jaynes, D. B., & Isenhart, T. M. (2019). Performance of saturated riparian buffers in Iowa. *Journal of Environmental Quality, 48*(2), 289-296.

Jaynes, D. B., Colvin, D. L., Karlen, D. L., Cambardella, C. A., & Meek, D. W. (2001). Nitrate loss in subsurface drainage as affected by nitrogen fertilizer rate. *Journal of environmental quality, 30*(4), 1305-1314.

Kadlec, R. H., & Wallace, S. (2009). *Treatment Wetlands.* CRC Press.

Kaiser, D. E., Lamb, J. A., & Eliason, R. (2011). *Fertilizer Guidelines for Agronomic Crops in Minnesota.* University of Minnesota Extension.

Kang, J., Hesterberg, D., & Osmond, D. L. (2009). Soil Organic Matter Effects on Phosphorus Sorption: A Path Analysis. *Soil Science Society of American Journal, 73*(2), 360-366.

Keesstra, S., Nunes, J., Novara, A., Finger, D., Avelar, D., Kalantari, Z., & Cerda, A. (2018). THe superior effect of nature based soultions in land management for enhancing ecosystem services. *Science of the Total Environment, 610-611*, 997-1109.

Khan, S. U., Blackwell, M. S., & Busquets, R. (2019). Microbial Biomass Responses to Soil Drying-Rewetting and Phosphorus Leaching. *Frontiers in Environmental Science, 7*, 133. doi:https://doi.org/10.3389/fenvs.2019.00133

Kieta, K. A., Owens, P. N., Lobb, D. A., Vanrobaeys, J. A., & Flaten, D. N. (2018). Phosphorus dynamics in vegetated buffer stips in cold climates: a review. *Environmental Reviews, 26*(3), 255-272. doi:https://doi.org/10.1139/er-2017-0077

King, K. W., Williams, M. R., Macrae, M. L., Fausey, N. R., Frankenberger, J., Douglas, S. R., . . . Brown, L. C. (2015). Phosphorus Transport in Agricultural Subsurface Drainage: A Review. *Journal of environmental quality, 44*(2), 467-485.

Koziol, L., & Bever, J. D. (2015). Mycorrhizal response trades off with plant growth rate and increases with plant successional stutus. *Ecology, 96*(7), 1768-1774.

Koziol, L., & Bever, J. D. (2017). The missing link in grassland restoration: arbuscular mycorrhizal fungi inoculation increases plant diversity and accelerates succession. *Journal of Applied Ecology, 54*(5), 1301-1309.

Kroger, R., Holland, M. M., Moore, M. T., & Cooper, C. M. (2007). Plant senescence: a mechanism for nutrient release in temperate agricultural wetlands. *Environmental Pollution, 146*(1), 114-119. doi:https://doi.org/10.1016/j.envpol.2006.06.005

Kumaragamage, D., Amarawansha, G. S., Indraratne, S. P., Jayaranthne, K., Flaten, D. N., Zvomuya, F., & Akinremi, O. O. (2019). Degree of phosphorus saturation as a predictor of redox-induced phosphorus release from flooded soils to floodwater. *Journal of Environmental Quality, 48*(6), 1817-1825. doi:10.2134/jeq2019.04.0154

Lazarus, W., & Keller, A. (2018). *Economic Analysis of Cover Crops on Farms Participating in the Southeastern Minnesota Cover Crop and Soil Health Initiative.* University of Minnesota Extension. Environment and Natural Resources Trust Fund.

Lee, M. R., Tu, C., Chen, X., & Hu, S. (2014). Arbuscular mychorrhizal fungi enhance P uptake and alter plant morphology in the invasive plant Microstegium vimineum. *Biological invasions, 16*(5), 1083-1093.

Lenhart, C., Gordon, B., Gamble, J., Current, D., Ross, N., Herring, L., . . . Peterson, H. (2016). Design and hydrologic performance of a tile drainage treatment wetland in Minnesota, USA. *Water, 8*(12), 549.

Lenhart, C., Gordon, B., Peterson, J., Eshenaur, W., Gifford, L., Wilson, B., . . . Utt, N. (2017). *Agricultural BMP Handbook for Minnesota, 2nd Edition.* St. Paul, MN: Minnesota Department of Agriculture. Retrieved from https://www.mda.state.mn.us/protecting/cleanwaterfund/research/handbookupdate

Lenhart, C., Wilson, B., & Gordon, B. (2016). Factors impacting the variability of effectiveness of agricultural best management practices (BMPs) in Minnesota. *2016 10th International Drainage Symposium Conference, 6-9 September 2016* (pp. 1-8). Minneapolis, Minnesota: American Society of Agricultural and Biological Engineers.

Lien, E., & Magner, J. (2017). Engineered biosystem treatment trains: A review of agricultural nutrient sequestration. *Invention Journal of Reserach Technology in Engineering and Management, 1*(11), 1-1.

Liu, J., Macrae, M. L., Elliott, J. A., Baulch, H. M., Wilson, H. F., & Kleinman, P. J. (2019). Impacts of cover crops and crop residues on phosphorus losses in cold climates: A review. *Journal of Environmental Quality, 48*(4), 850-868.

Liu, Z., & Shuting, L. (2016). High Phosphate Concentrations Accelerate Bacterial Peptide Decomposition in Hypoxic Bottom Waters of the Northern Gulf of Mexico. *Environmental Science and Technology, 50*(2), 676-684.

Logan, T. J., Randall, G. W., & Timmons, D. R. (1980). *Nutrient content of tile drainage from cropland in the north central region. Research Bulletin 1119.* Wooster, OH: Ohio Agricultural.

Maccoux, M. J., Dove, A., Backus, S. M., & Dolan, D. M. (2016). Total and soluble reactive phosphorus loadings to Lake Erie: A detailed accounting by year, basin, country, and tributary. *Journal of Great Lake Research, 42*(6), 1151-1165.

Magner, J. (2011). Tailored watershed assessment and integrated management (TWAIM): A systems thinking approach. *Water, 3*, 590-603.

Mallarino, A. P., Sawyer, J. E., & Barnhart, S. K. (2013). *A General Guide for Crop Nutrient and Limestone Recommendations in Iowa.* Iowa State University, Agronomy. Ames, Iowa: Extension and Outreach Publications. Retrieved from https://lib.dr.iastate.edu/extension_pubs/82

Maltais-Landry, G., Scow, G., Brennan, K., & Vitousek, P. (2015). Long-Term Effects of Compost and Cover Crops on Soil Phosphorus in Two California Agroecosystems. *Soil Science Society of America Journal, 79*(2), 688-697.

McCrackin, M. L., Muller-Karulis, B., Gustafsson, B. G., Howarth, R. W., Humborg, C., Svanback, A., & Swaney, D. P. (2018). A century of legacy phosphorus dynamics in a large drainage basin. *Global Biogeochemical Cycles, 32*(7), 1107-1122.

McDowell, R., Sharpley, A., Brookes, P., & Poulton, P. (2001). Relationship between soi ltest phosphorus and phosphorus release to solution. *Soil Science, 166*(2), 137-149.

McDowells, R. W., Depree, C., & Stenger, R. (2020). Likely controls on dissolved reactive concentrations in baseflow of an agricultural stream. *Journal of Soils and Sediments, 20*(8), 3254-3265.

McKnight, P. E., & Najab, J. (2010). The Corsini Encyclopedia of Psychology. In *Mann-Whitney Test* (pp. 1-1). John WIley and Songs.

Mecklenburg, D. E., & Ward, A. (2004). In STREAM modules: spreadsheet tools for river evaluation, assessment and monitoring. *Self-Sustaining Solutions for Streams, Wetlands, and Watersheds, 12-15, September 2004, p. 56. 701P0904.* American Society of Agricultural and Biological Engineers.

Menon, R., & Holland, M. M. (2014). Phosphorus release due to decomposition of wetland plants. *Wetlands, 34*(6), 1191-1196. doi:10.1007/s13157-014-0578-2

Meyers, N., Bergez, J.-E., Constantin, J., & Justes, E. (2019). Cover crops reduce water drainage in temperate climates: A meta-analysis. *Agronomy for Sustainable Development, 39*(1), 1-11.

Miller, S. P., & Sharitz, R. R. (2000). Manipulation of flooding and arbuscular mycorrhiza formation influences growth and nutrition of two semiaquatic grass species. *Functional Ecology, 14*(6), 738-748.

Minnesota Legislature. (2018). *Minnesota administrative rules, 7050.0222*. Retrieved July 7, 2021, from https://www.revisor.mn.gov/rules/7050.0222/

Minnesota Pollution Control Agency. (2019). *Minnesot'a Nutrient Reduction Strategy, Developing a Progress Report.* Saint Paul, MN: Minnesota Board of Soil and Water Resources.

Mishra, S. K., & Singh, V. P. (2013). *Soil conservation service curve number (SCS-CN) methodology.* (Vol. 42). Springer Science and Business Media.

MN DNR. (2020). *Minnesota State Climatology Office*. (Division of Ecological and Water Resources, Producer) Retrieved from Monthly Gridded Precipitation Database: https://climateapps.dnr.state.mn.us/gridded_data/precip/monthly/monthly_gridded_precip.asp

MPCA. (2014). *Minnesota's Nutrient Reduction Strategy.* wq-s1-80. Retrieved from https://www.pca.state.mn.us/sites/default/files/wq-s1-80.pdf

MycoBloom. (2021). *MycoBloom Mycorrhizae*. Retrieved September 20, 2021, from About Us: http://mycobloom.com/about.html

MycoRoots. (2021). *MycoRoots*. Retrieved September 20, 2021, from https://mycoroots.com/

Nair, V. D. (2014). Soil phosphorus saturation ratio for risk assessment in land use systems. *Frontiers in Environmental Science*, *2*, 6.

Negassa, W., Michalik, D., Klysubun, W., & Leinweber, P. (2020). Phosphorus Speciation in Long-Term Drained and Rewetted Peatlands of Northern Germany. *Soil Systems, 4*(1), 11.

Ni, X., Yuan, Y., & Liu, W. (2020). Impact factors and mechanisms of dissolved reactive phosphorus (DRP) losses from agricultural fields: A review and synthesis study in the Lake Erie basin. *Science of the Total Environment, 714*, 136624.

Nuruzzaman, M., Lambers, H., Bolland, M. D., & Veneklaas, E. J. (2005). Phosphorus Benefits of Different Legume Crops to Subsequent Wheat Grown in Different Soils of Western Australia. *Plant and Soil, 271*(1-2), 175-187.

Nustad, R. A., Rowland, K. M., & Wiederholt, R. G. (2015). *Water-quality characteristics in runoff for three discovery farms in North Dakota.* US Department of the Interior, US Geological Survey.

Obilor, E. I., & Amadi, E. C. (2018). Test for significance of pearson's correlation coefficient. *International Journal of Innovative Mathematics, Statistics and Energy Policies, 6*(1), 11-23.

Pant, H. K., & Reddy, K. R. (2001). Phosphorus sorption characteristics of estuarine sediments under different redox conditions. *Journal of Environmental Quality, 30*(4), 1474-1480. doi:https://doi.org/10.2134/jeq2001.3041474x

Pease, L. A., Kevin, K. W., Williams, M. R., LaBarge, G. A., Duncan , E. W., & Fausey, N. R. (2018). Phosphorus export from artificially drained fields across the Eastern Corn Belt. *Journal of Great Lakes Research, 44*(1), 43-53.

Peterson, H. M., Baker, L. A., Bruening, D., Nieber, J. L., Ulrich, J. S., & Wilson, B. N. (2017). Agricultural Phosphorus Balance Calculator: A tool for watershed planning. *Journal of Soil and Water Conservation, 72*(4), 395-404.

Plastina, A., Liu, F., Miguez, F. E., & Carlson, S. (2018). Cover crops use in Midwestern US agricultural: perceived benefits and net returns. *Renewable Agriculture and Food Systems, 35*(1), 1-11.

Puth, M. T., Neuhauser, M., & Ruxton, G. D. (2014). Effective use of Pearson's product-moment correlation coefficient. *Animal Behaviour, 93*, 183-189.

Richardson, C. J., & Marshall, P. E. (1986). Processes controlling movement, storage, and export of phosphorus in a fen peatland. *Ecological Monographs, 56*(4), 279-302. doi:https://doi.org/10.2307/1942548

Roberts, W. M., Matthews, R. A., Blackwell, M. S., Peukert, S., Collins, A. L., Stutter, M. L., & Haygarth, P. M. (2013). Microbial biomass phosphorus contributions to phosphorus solubility in riparian vegetated buffer stip soils. *Biology and Fertility of Soils, 59*(8), 1237-1241. doi:10.1007/s00374-013-0802-x

Roberts, W. M., Stutter, M. I., & Haygarth, P. M. (2012). Phopshorus Retention and Remobilization in Vegetated Buffer Strips: A Review. *Journal of Environmental Quality, 41*(2), 389-399. doi:https://doi.org/10.2134/jeq2010.0543

Robertson, D. M., & Saad, D. A. (2019). *Spatially Referenced Models of Streamflow and Nitrogen, Phosphorus, and Suspended-Sediment Loads in Streams of the Midwestern United States.* Reston, Virginia: U.S. Geologic Survey.

Rodriques, M., Withers, P. J., Soltangheisi, A., Vargas, V., Holzschuh, M., & Pavinato, S. P. (2012). Rodrigues, M., Withers, P. J. A., Soltangheisi, A., Vargas, V., Holzschuh, M., & Pavinato, P. S. (2021). Tillage systems and cover crops affecting soil phosphorus bioavailability in Brazilian Cerrado Oxisols. *Soil and Tillage Research, 205*, 104770.

Roley, S. S., Tank, J. L., Tyndall, J., & Witter, J. D. (2016). How cost-effective are cover crops, wetlands, and two-stage ditches for nitrogen removal in the Mississippi River Basin? *Water Resources and Economics*, 43-56.

Rosario-Lebron, A., Leslie, A. W., Yurchak, V., Chen, G., & Hooks, C. R. (2019). Can winder cover crop termination practices impact weed suppression, soil moisture and yield in no-till soybean [Glycine max (L.) Merr.]? *Crop Protection, 116*, 132-141.

Ross, N. B. (2014). *Constructred wetland used to treat nitrate pollution generated from agricultural tile drainage waters in Southern Minnesota.* University of Minnesota-Twin Cities, M.S. thesis. https://conservancy.umn.edu/handle/11299/167310.

Rubin, J. A., & Gorres, J. H. (2021). Potential for Mycorrhizae-Assisted Phytoremediation of Phosphorus for Improved Water Quality. *Environmental Research and Public Health, 18*(1), 7. doi:https://doi.org/10.3390/ijerph18010007

Ruffatti, M. D. (2019). Impacts of nitrogen application timing and cover crop inclusion on subsurface drainage water quality. *Agricultural Water management, 211*, 81-88.

Running, S., Mu, Q., Zhao, M., & Moreno, A. (2019). *MOD16A3GF MODIS/Terra Net Evapotranspiration Gap-Filled Yearly L4 Global 500 m SIN Grid V006. NASA EOSDIS Land Processes DAAC.* Retrieved January 4, 2021, from https://doi.org/10.5067/MODIS/MOD16A3GF.006

Saadat, S., Bowling, L., Frankenberger, J., & Kladivko, E. (2018). Nitrate and phosphorus transport through subsurface drains under free and controlled drainage. *Water Research*, *142*(1), 196-207.

Salas, A. M., Elliott, E. T., Westfall, D. G., Cole, C. V., & Six, J. (2003). The Role of Particulate Organic Matter in Phosphorus Cycling. *Soil Science Society of America Journal, 67*(1), 181-189. doi: https://doi.org/10.2136/sssaj2003.1810a

SARE/CTIC. (2016). *Annual Report 2015-2016, Cover Crop Survey.* Retrieved from https://www.sare.org/wp-content/uploads/2015-2016-Cover-Crop-Survey-Report.pdf

Scavia, D., & Donnelly, K. (2007). Reassessing hypoxia forecasts for the Gulf of Mexico. *Environmental science and technology, 41*(23), 8111-8117.

Seifert, C. A., Azzari, G., & Lobell, D. B. (2018). Satellite deteciton of cover crops and their effects on crop yield in the Midwestern United States. *Environmental Research Letters, 13*(6), 064033.

Shaphiro, S. S., & Wilk, M. B. (1965). An analysis of variance test for normality. *Biometrika*, *52*(3), 591-611.

Sharpley, A. N., Jarvie, H. P., Buda, A., May, L., Spears, B., & Kleinman, P. (2013). Phosphorus legacy: overcoming the effects of past management practices to mitigate future water quality impairment. *Journal of Environmental Quality, 42*(5), 1308-1326.

Shipitalo, M. J., Dick, W. A., & Edwards, W. M. (2000). Conservation tillage and macropore factors that affect water movement and the fate of chemicals. *Soil and tillage research, 53*(3-4), 167-183.

Sims, J. T., Edwards, A. C., Schoumans, O. F., & Simard, R. R. (2000). Integrating soil phosphorus testing into environmentally based agricultural management practices. *Journal of Environmental Quality, 29*(1), 60-71.

Sims, J. T., Simard, R. R., & Joern, B. C. (1998). Phosphorus loss in agricultural drainage: Historical perspective and current research. *Journal of Environmental Quality, 27*, 277-293.

Singh, G., Kaur, G., Williard, K., Schoonover, J., & Nelson, K. A. (2020). Managing Phosphorus Loss from Agroecosystems of the Midwestern United States: A Review. *Agronomy, 10*(4). doi:https://doi.org/10.3390/agronomy10040561

Singh, J. S., & Gupta, V. K. (2018). Soil microbial biomass: A key soil driver in management of ecosystem functioning. *The Science of the Total Environment, 634*, 497.

Singh, J., Wang, T., Kumar, S., Xu, Z., Sexton, P., Davis, J., & Bly, A. (2021). Crop yield and economics of cropping systems involving different rotations, tillage and cover crops. *Journal of Soil and Water Conservation*.

Smith, D. R., King, K. W., & Williams, M. R. (2015). What is causing the harmful algal blooms in Lake Erie? *Journal of Soil and Water Conservation, 70*(2), 27A-29A.

Spearman, C. (1904). The proof and measurement of association between two things. *American Journal of Psychology, 15*(1), 72-101.

Stevens, K., Spender, S., & Peterson, R. (2002). Phosphorus, arbuscular mycorrhizal fungi and performance of the wetland plant Lythrum salicaria L. under inundated conditions. *Mycorrhiza, 12*(6), 277-283. doi:10.1007/s00572-002-0184-7

Stutter, M. I., Langan, S. J., & Lumsdon, D. G. (2009). Vegetated Buffer Strips Can Lead to Increased Release of Phosphorus to Waters: A Biogeochemical Assessment of the Mechanisms. *43*(6), 1858-1863. doi:https://doi.org/10.1021/es8030193

Sustainable Agriculture Research and Education. (2020). *National Cover Crop Survey Annual Report 2019-2020.* United States Department of Agriculture. Retrieved from https://www.sare.org/wp-content/uploads/2019-2020-National-Cover-Crop-Survey.pdf

Thompson, N. M., Reeling, C. J., Michelle , F. R., Prokopy, L. S., & Armstrong, D. S. (2021). Examining intensity of conservation practice adoption. *Food Policy*, 102054.

Tranmer, M., Murphy, J., Elliot, M., & Pampaka, M. (2008). Multiple Linear Regression (2nd Edition). *The Cathie Marsh Centre for Census and Survey Research, 5*(5), 1-5. Retrieved from http://hummedia.manchester.ac.uk/institutes/cmist/archive-publications/working-papers/2020/multiple-linear-regression.pdf

Trentman, M. T., Tank, J. L., Royer, T. V., Speir, S. L., Ursula, H. M., & Lienne, R. S. (2020). Cover crops and precipitation influence soluble reactive phosphorus losses via tile drain discharge in an agricultural watershed. *Hydrological Processes, 34*(23), 4446-4458.

U.S. Environmental Protection Agency. (2021). *A Compilation of Cost Data Associated with the Impacts and Control of Nutrient Pollution*. Retrieved July 7, 2021, from https://www.epa.gov/nutrient-policy-data/compilation-cost-data-associated-impacts-and- control-nutrient-pollution/

USDA. (2020). *Gridding Soil Survey Geographic Database (gSSURGO), User Guide.* USDA, NRCS. Washington, D.C.: National Soil Survey Center. Retrieved from https://www.nrcs.usda.gov/wps/PA_NRCSConsumption/download?cid=nrcs142p2_051847&ext=pdf

USDA. (2021). *Minnesota Ag News 2020 Crop Production.* Saint Paul, Minnesota: National Agriculture Statistics Service.

USDA. (2021). *Quick Stats*. Retrieved from Washington, DC: National Agricultural Statistic Service: http://quickstats.nass.usda.gov/

Utt, N., Jaynes, D., & Albertsen, J. (2015). Demonstrate and evaluate saturated buffers at field scale to reduce nitrates and phosphorus from subsurface field drainage systems. *Mississippi River Basin Water Management*, 1-74. Retrieved from http://www.saturatedbufferstrips.com/docs/final_report.pdf

Van der Grift, B., Oste, L., Schot, P., Kratz, A., van Popta, E., Wassen, M., & Griffioen, J. (2018). Forms of phosphorus in suspended particulate matter in agricultural-dominated lowland catchments: Iron as phosphorus carrier. *Science of the Total Environment, 631-632*(1), 115-129.

Wall, D., Kjaersgaard, J., Drewitz, M., Graziani, M., Scheirer, L., Olmanson, R., . . . Nustad, R. (2020). *5-year Progress Report on Minnesota's Nutrient Reduction Strategy.* Saint Paul, MN: Minnesota Pollution Control Agency.

Walton, C. R., Zak, D., Audet, J., Petersen, R. J., Lange, J., Oehmke, C., . . . Hoffmann, C. C. (2020). Wetland buffer zones for nitrogen and phosphorus retention: Impacts of soil type, hydrology and vegetation. *Science of the Total Environment, 727*, 138709. doi:https://doi.org/10.1016/j.scitotenv.2020.138709

Waring, E. R., Lagzdins, A., Pederson, C., & Helmers, M. J. (2020). Influence of no-till and a winter rye cover crop on nitrate losses from tile-drained row-crop agricultural in Iowa. *Journal of Environmental Quality, 9*(12), 852.

Xu, X., Thornton, P. E., & Post, W. M. (2013). A global analysis of soil microbial biomass carbon, nitrogen and phosphorus in terrestrial ecosystems. *Global Ecology and Biogeography, 22*(6), 737-749. doi:https://doi.org/10.1111/geb.12029

Yang, W., Feng, G., Adeli, A., & Qu, Z. (2021). Simulated long-term effect of wheat cover crop on soil nitrogen losses from no-till corn-soybean rotation under different rainfall patterns. *Journal of Cleaner Production, 280*, 124255.

Yang, W., Feng, G., Read, J. J., Ouyang, Y., Han, J., & Li, P. (2019). Impact of cover crop on corn-soybean productivity and soil water dynamics under different seasonal rainfall patterns. *Agronomy Journal*, *112*(2), 1201-1215.

Yang, X., Chen, X., & Yang, X. (2019). Effect of organic matter on phosphorus adsorption and desorption in a black soil from Northeast China. *Soil and Tillage Research, 187*, 85-91.

Young, E. O., & Ross, D. S. (2018). Phosphorus mobilization in Flooded Riparian Soils From the Lake Champlain Basin, VT, USA. *Frontiers in Environmental Science, 6*, 120.

Young, E. O., Ross, D. S., Jaisi, D. P., & Vidon, P. G. (2021). Phosphorus Transport along the Cropland-Riparian-Stream Continuum in Cold Climate Agroecosystems: A Review. *Soil Systems, 5*(1), 15.

Zhang, T. Q., Tan, C. S., Zheng, Z. M., Welacky, T., & Wang, Y. T. (2017). Drainage water management combined with cover crop enhances reduction of soil phosphorus loss. *Science of the Total Environment, 586*, 362-371.

Zhao, X., Li, G., & Lin, Q. (2008). Interference of soil-extractable phosphorus in measuring soil microbial biomass phosphorus. *Communications in soil science and plant analysis, 39*(9-10), 1367-1374.

Zheng, Z., & Zhang, T. Q. (2014). Soil Testing to Predict Dissolved Reactive Phosphorus Loss in Surface RUnoff from Organic Soils. *Soil Science Society of America Journal*, *78*(5), 1786-1796.

Zhu, J., Li, M., & Whelan, M. (2018). Phosphorus activators contribute to legacy phosphorus availability in agricultural soils: A review. *Science of the Total Environment, 612*, 522-537.

Ziegeweid, J. R., Lorenz, D. L., Sanocki, C. A., & Czuba, C. R. (2015). *Methods for estimating flow-duration curve and low-flow frequency statistics for ungaged locations on small streams in Minnesota.* U.S. Geological Survey Scientific Investigations Report 2015-5180. Retrieved from https://pubs.er.usgs.gov/publication/sir20155170

Zucker, L. A., & Brown, L. C. (1998). *Agricultural drainage: water quality impacts and subsurface drainage studies in the midwest.* Ohio State University Extension.

Appendix 1

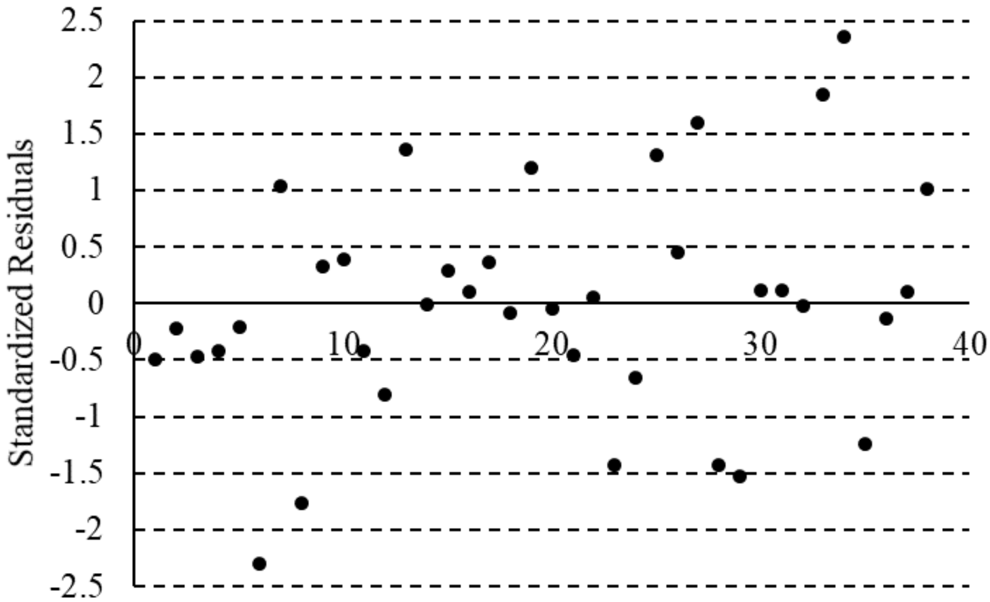

Figure 23. Standardized residuals for the best fit drain tile flow dissolved reactive phosphorus (DRP) flow weighted mean concentration (FWMC) regression model.

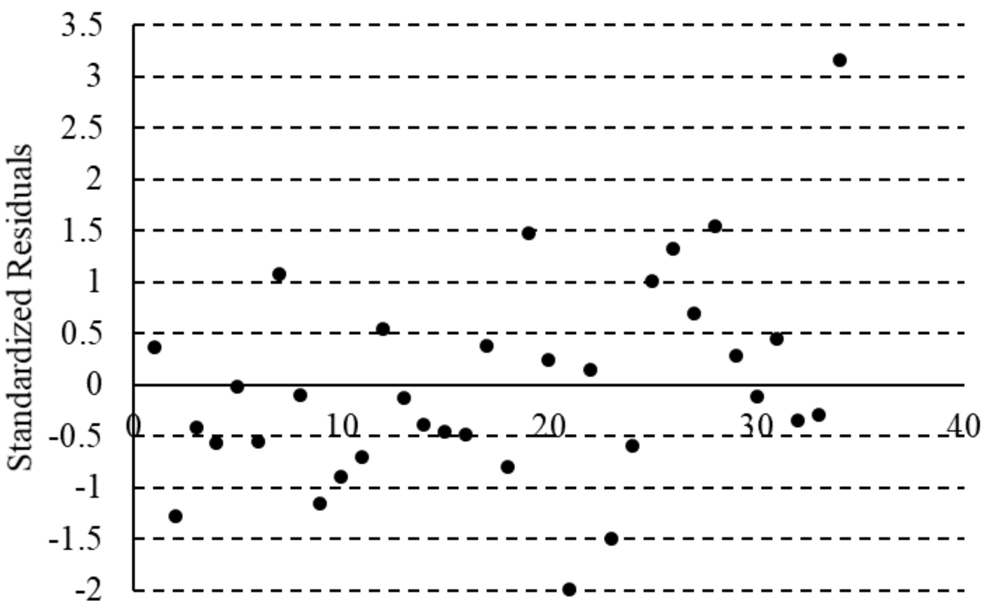

Figure 24. Standardized residuals for the best fit surface flow DRP FWMC regression model.

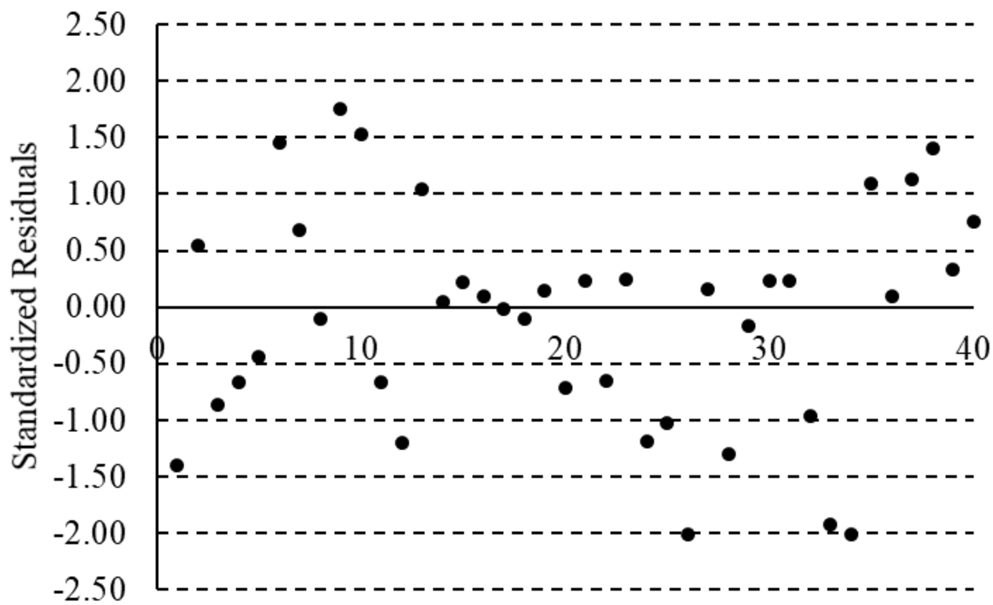

Figure 25. Standardized residuals for the best fit soil test phosphorus regression model