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Nutrient Loading Reduction in a Tile Drained Agricultural Watershed through Watershed-Scale Cover Cropping: a High Resolution Analysis

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Nutrient pollution originating from agricultural regions in the Midwest is a serious issue, leading to pollution of drinking water sources as well as large hypoxic zones in the Gulf of Mexico. The source of much of this contamination has been shown to be runoff from agricultural fields in the Upper Mississippi River Basin. One method that has been shown to reduce this pollution from the Upper Mississippi River Basin is the planting of winter cover crops. Winter cover crops such as rye and tillage radish have been shown to significantly reduce nitrate exported from agricultural fields, even in tile-drained watersheds that are resistant to other nitrate management methods such as riparian zones. However, most studies take place in small agricultural study fields, where planting and fertilization is tightly controlled. There is also a lack of studies looking at the effectiveness of cover cropping in reducing phosphorus export.

In this study, we looked at the effectiveness of winter cover crops in reducing nitrate and total phosphorus (TP) loading from tile- drained agricultural watersheds in Central Illinois. We compared nitrate loading from two agricultural watersheds that were 445 ha and 312 ha, one of which was 54% planted with cover crops in fall 2015 and fall 2016 while the other was left fallow. We used discharge probes and automatic sampling systems to capture high temporal resolution discharge and concentration data from tile drains draining each watershed, and using these measurements we compared nitrate and TP loading.

We found no noticeable pattern of nitrate or TP reduction in spring 2016, despite that period having the greatest total cover crop biomass. However, in the fall 2016 cover cropping period, there was a pattern of reduced nitrate loading at our treatment site relative to the loading at our reference site. This appears to be due to a statistically significant ($p=5.045x10^{-9}$) reduction in discharge at the treatment site relative to the reference site during the fall 2016 cover cropping period. Nitrate loading reduction was particularly strong during periods of storm flow. The effects of cover cropping in fall 2016 were more mixed when it came to TP loading. There was a lesser percentage of TP that occurred during storm events at our treatment site during this period, but there was not a significant change ($p=0.0522$) in TP loading relative to the reference site when incorporating baseflow.

KEYWORDS: Nitrate, Cover Crop, Phosphorus, Loading, Agriculture, Hydrology

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B. G. B.

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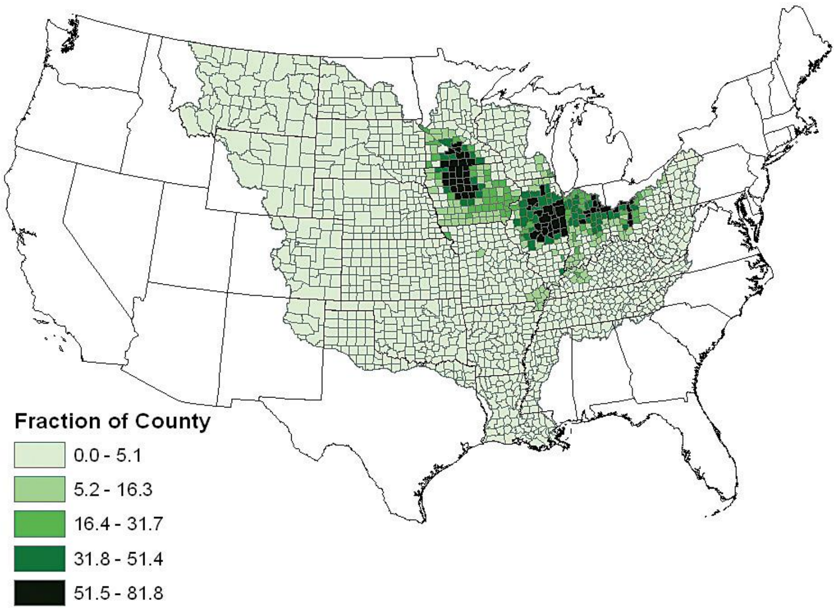
CHAPTER I: INTRODUCTION

Nitrate pollution is a growing global concern, increasing with the industrialization of agriculture. Leaching of excess nitrogen fertilizer carries nitrate into streams and groundwater, causing contamination of streams and drinking water. Drinking water sources near agricultural areas often exhibit high nitrate levels, which can lead to methemoglobinemia, also known as blue baby syndrome, a health condition where blood cannot carry adequate oxygen. In addition, nitrate laden runoff can be carried through streams and rivers to marine areas, where it has been shown to contribute to algal blooms (Rabalais et al 1996). These algal blooms cause huge areas of the ocean to become hypoxic. These hypoxic areas, known as dead zones, lead to mass death of fish and macroinvertebrates (Michelak et al. 2013, Rabalais et al. 2002, Beman et al. 2005).

The largest dead zone in the Atlantic is in the Gulf of Mexico and has been linked to nitrate runoff originating in the upper Mississippi River Basin, the Corn Belt (Howarth et al. 1996, Rabalais et al 1996, David et al 2010). Despite efforts to reduce nutrient runoff from the upper Mississippi River Basin, the dead zone in the Gulf of Mexico continues to reach massive size each year. In 2015, NOAA measured the dead zone at 6,474 square miles (NOAA 2015), which is about the size of Connecticut and Rhode Island combined. This is larger than the 5,052 square miles reached in 2014 and higher than the average size of the dead zone from 2009-2014 (NOAA 2015).

The Corn Belt runs from southern Minnesota through Iowa, Illinois, Indiana, and Ohio. This area is intensively farmed, in most cases exclusively on a corn-soybean rotation. To support this intensive agriculture, large amounts of nitrate fertilizer are applied yearly, leading to high riverine nitrate export from these areas (Fig 1b). This nitrate export is particularly severe in areas with a high amount of tile drainage (Fig. 1a), such as south-central Minnesota, central Ohio and most of Iowa, Illinois, and Indiana (David et al 2010).

A.



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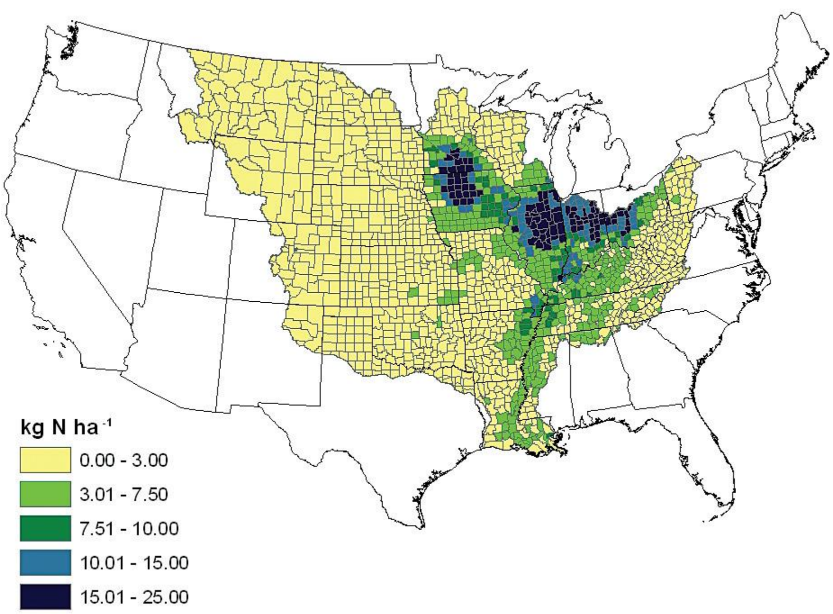


Fig. 1: (a) Fraction of each county in the Mississippi River Basin that is tile drained and (b) nitrate export per county in the Mississippi River Basin. Figures are from David et al. 2010.

Since the 1950's, nitrate export from the Upper Mississippi River Basin has increased at a rapid rate, corresponding with the increased use of chemical nitrogen fertilizer and the conversion of land to corn production over the same time period (Dinnes et al. 2002). Farmers typically apply nitrogen fertilizer to fields in the fall after harvest or in spring before planting (Bierman et al. 2012). The predominant forms of chemical nitrogen fertilizer applied are anhydrous ammonia and urea (Bierman et al 2012). These fertilizers, though not initially very water soluble, oxidize into nitrate, which is much more soluble, through nitrification. This fertilizer-derived nitrate is then easily dissolved and transported by water leaching through soil.

Much of this nitrate enters stream systems from tile drains (David et al. 1997, David et al 2010, Fenelon and Moore 1998), which are pipes placed beneath agricultural fields to drain water to prevent ponding on agricultural fields and the resulting drowning of crops. This is an essential function in many areas of the Midwest, especially areas with flat topography, where water does not drain naturally. While important for agricultural production, these tile drains make many forms of nitrate management ineffective. Tile drains allow nitrate to flow directly through soil into streams, bypassing riparian zones and making best management practices such as strip tillage less effective (Lemke et al. 2010).

Since nitrate is so water soluble, its transport is closely linked to storm events (David et al. 1997, Royer et al 2006). Nitrate export is thus not evenly distributed throughout the year, it fluctuates on a short term, often day to day basis (David et al. 1997, Royer et al. 2006), depending on soil nitrate content in combination with precipitation amount. Because nitrate export can vary so much in such a short period of time, frequent monitoring is needed to ensure that it is calculated accurately. The majority of nitrate export occurs during times of above average discharge (David et al. 1997, Royer et al. 2006). For this reason, it is beneficial to measure nitrate loading at increased rates during periods of high discharge.

In order to reduce nutrient loading to the Gulf of Mexico, there has been a push to find other methods to reduce nutrient runoff from tile-drained agricultural areas, methods that can capture nitrate and phosphorus before they leach through soil to tile drains. The state of Illinois has a plan in place to reduce nutrient export known as the Illinois Nutrient Loss Reduction strategy. The goal of this strategy is to reduce TP loading into the Mississippi by 25% and nitrate export by 15% by the year 2025 (Illinois EPA 2016). This strategy's ultimate goal is a 45% reduction in riverine TP loading and nitrate loading in order to meet a goal set by the EPA to reduce the size of the hypoxic zone and improve water quality in the Mississippi River Basin (Illinois EPA 2016). To do this, they encourage voluntary action by farmers to reduce nutrient export from their fields, including the implementation of practices such as no-till farming, reduced fertilizer application, and cover cropping (Illinois EPA 2016).

One particularly promising method of nitrate reduction is the use of fall-seeded cover crops. Cover cropping has been shown to be effective in removing nitrate in many locations, from tropical climates (Amado et al. 1998) to the much colder environment of the Corn Belt (Johnson et al. 1998, Strock et al. 2004). These cover crops, which include oat (avena sativa), rye (secale cereal), hairy vetch (Vicia villosa), and tillage radish (raphanus sativus), can remove nitrate and water from the soil, thus reducing leaching into streams and groundwater (Dean and Well 2009, Fraser et al. 2013). They can also reduce soil erosion (De Baets et al. 2011, Kaspar et al 2001), increase soil organic matter (Moore et al. 2014, Villami et al. 2006), reduce soil compaction (Williams and Weil 2004), suppress weed growth (Creamer et al. 1996, Teasdale and Daughtry 1993, Teasdale and Daughtry 1996, O'Reilly et al. 2011), and suppress crop parasites

(Koch et al. 2012, Koch et al. 2015). In addition, cover cropping does not require land to be set aside, unplanted, unlike other nutrient mitigation methods such as constructed wetlands and woodchip bioreactors.

While cover cropping has been shown to reduce nitrate loading, the effects of cover cropping on phosphorus loading from tile drained agricultural systems in the Midwest is limited. This is despite results indicating that phosphorus loading contributions from tile drainage can be greater than those from surface runoff (Algoanzy et al 2007, Van Esbroeck et al 2016) in heavily tile drained agricultural regions. There are some studies on the effects of cover cropping on phosphorus activity in non-tile drained agricultural areas, and their findings vary. Several Scandinavian studies found that winter plant cover actually increases soluble phosphorus losses from a system after freezing (Ulén 1997, Øgaard 2015). A synthesis of cover cropping studies in Scandinavia concluded that cover crops do not significantly reduce phosphorus leaching (Aronsson et al. 2016).

The cover crop most commonly used in the Midwest is cereal rye (Secale cereal). Cereal rye is highly resistant to cold and has the ability to overwinter. In a study of the effect of cover crops on erosion, De Baets et al (2011) found rye to remain more effective after frost than other cover crops such as forage radish and oats. Geiger and Miedaner (2009) classify rye as the most winter resistant grain. Rye has also been more effective at reducing nutrient leaching than other crops. Kaspar et al. (2012) found rye to be twice as effective as oats at reducing drainage water nitrate concentrations. In a study of the effectiveness of different crops in the Midwest, Appelgate and Lenssen (2016) found that planting rye and rye mixtures led to the greatest amount of plant biomass, carbon, and nitrogen accumulation and the lowest soil nitrate levels amongst their studied cover crops. Lacey and Armstrong et al. (2015) showed that cereal rye can significantly impact the distribution of soil inorganic nitrogen in the spring, causing more of nitrogen to be in soil closer to the surface rather than at deeper depths closer to tile drainage.

The effectiveness of cover crops is contingent on planting and weather conditions. For example, cover crops may grow better when planted with a grain drill rather than overseeding (Kaspar et al. 2007), and early planting of cover crops is essential for cover crops to reach their full growth and nutrient removal potential (McCarty et al. 2008, Teixeira et al. 2016). During a dry fall, rye growth can be reduced (Kaspar et al. 2007). Cover crops are more effective in warmer years than colder years (Teixeira et al. 2016, Strock et. al 2004).

There have been several studies performed in the Midwest on the effects of rye cover cropping on nutrient export reduction from tile-drained systems. In a 4-year Iowa study of tile-drained plots, Kaspar et al. (2007) observed an average nitrate load reduction of 61%. Malone et al (2014) found, using a model, that using a rye cover crop has the potential to reduce nitrogen loading by 42.5% across the Midwest. Strock et al. (2004) found more a more modest reduction of 13% using cereal rye in a three-year Minnesota study, which may be indicative of the reduced efficacy of cover crops in colder areas. Qi et al. (2008) in a two-year Iowa study, found a reduction of up to 21.7% in flow-weighted NO3-N concentrations one year, but no statistically significant difference was found in the other year of the study. David et al. (2015), in a paired field scale study of a tile-drained system in Illinois, found a reduction of 34% in tile nitrate yield. Kladivko et al (2004) in a 15-year study in Southeastern Indiana, found a 60% reduction in tile drain nitrate loading with a combination of cover cropping, changes in tillage, and reduction in nitrogen fertilization rates.

Most cover cropping studies have been performed on a plot scale (Kaspar et al. 2012, Strock et al. 2004, Rasse et al. 2000, Fraser et al. 2013, Drury et al. 2014, Zhu et al. 1989, O'Reilly et al. 2011, Dean and Well 2009, Kaspar et al 2001, Johnson et al. 1998, Moore et al. 2014, De Baets et al 2011, Kaspar et al 2007, Kaspar et al 2012, Williams and Weil 2004, Qi et al. 2008, Olson et al. 2014, Kladivko et al. 2004, Ball Coelho et al. 2005), where treatments are applied on small plots of land, usually in the range of 1-5 acres. At these plots, factors such as the timing and method of planting, harvesting, and fertilization are tightly controlled.

A few studies have been performed on a field scale. David et al. (2015) planted cereal rye and tillage on a 17 ha field in east-central Illinois and estimated that they reduced nitrate loading by 34% compared to a control field. Staver and Brinsfield (1998), in a long-term study, planted cereal rye on field-scale watersheds in the mid-Atlantic coastal plain and found a reduction in nitrate leaching of about 80% compared to a control field left fallow in the winter through the duration of their study.

A few studies do look at the effect of watershed scale cover cropping on nutrient export, but they often use models rather than empirical datasets from field studies. Yeo et al. (2014) used a SWAT model to simulate the effects of winter cover cropping on watersheds near the Chesapeake Bay, finding that they had the potential to reduce nitrate loading by 67%. A follow up study in the same location using an updated SWAT model found that a rye cover crop could reduce NO3-N loads by 49.3% (Lee et al. 2016). Qi et al. (2011) used a RZWQM2 model calibrated using data from a plot-scale experiment in north central Iowa to find that cover cropping would reduce nitrate loss by 22% compared to a control. Malone et al. (2014) found, using an RZWQM model to simulate the effects of cover cropping on nitrate loading at locations across the Midwest, that cover crops had the potential to reduce nitrate loss in tile drainage by 42.5% across the Midwest. Ferrant et al. (2015) used a model called as TNT2 to predict the effects that planting of a mustard cover crop would have on a watershed in France, finding a potential reduction in nitrogen flux of 20%. A potential nitrate reduction by cover cropping of 51% and a phosphorus reduction of about 50% was found using an APEX model of a watershed in Central Indiana (Francesconi et al. 2015).

Other watershed-scale field studies have observed the effectiveness of best management practices other than cover cropping on nutrient export from watersheds. Park et. al (1994) applied best management practices including no-till cultivation, grazing land protection, and sediment retention structures over a 1464 ha watershed in Virginia and found that nitrogen concentrations were reduced by about 42% and TP concentrations by 35% after the establishment of these BMP's, compared to levels before the establishment of these BMP's. Edwards et al. (1996) found that the implementation of best management practices such as nutrient management, waste utilization, pasture and hayland management, and waste storage structure construction on a 3240-ha basin in Northwest Arkansas significantly reduced nitrogen concentrations in runoff from the basin. Lemke et al. (2010) found no significant effect of the implementation of best management practices on nitrate and total phosphorus export in a paired watershed study of two 4000 ha tile-drained Illinois watersheds. None of these studies, however, include cover crops as one of the best management practice.

To my knowledge, there are no watershed scale studies looking at the effects of cover cropping on nutrient runoff from agricultural areas. This is potentially problematic because it does not allow for a look at on the effects that implementation of cover crops will have on the quality of water stemming from an entire watershed with farmers planting and harvesting independently. There is also potential for increased per unit area runoff in watersheds compared to plots due to the increased influence of impermeable subsurface horizons at that scale (Dabney 1998), which could skew results at the plot scale.

High temporal resolution sampling has been important in measuring nutrient loss from agriculture accurately. In a French study of the differences between high resolution data calculated using a nitrate probe and lower resolution data in calculating nitrate loading in a 350-ha agricultural watershed, an hourly water sampling interval was necessary for maximum accuracy during storm events (Ferrant et. al 2013). This same study found that over the two-year period of their study, a biweekly sampling strategy would have led to a 7.9% underestimation of nitrate loading compared to high resolution sampling. Most studies of nutrient loss from tile drained systems use some form of high-resolution water sampling and discharge measurement. David et al. (2015) installed v-notch weirs into their tiles with pressure transducers to measure discharge and took samples based on this discharge, taking one sample weekly or bi-weekly during baseflow and one or more samples per day during storm events. Kaspar et al. (2012) used pumped tile water from collection basins and measured the pumped water using a flow meter to measure discharge and took a number of samples proportional to this discharge.

In this study, we will examine the effects planting of fall cover crops over an agricultural watershed will have on nutrient export from that watershed, including nutrient export during storm events, using high temporal resolution concentration and discharge data.

Hypotheses:

1. The planting of these cover crops will reduce nitrate and TP runoff in the watershed.

2. These cover crops will reduce overall discharge from the watershed, especially during storm events.

CHAPTER II: METHODS

Study Site

This study includes two watersheds in Central Illinois, northeast of Normal, IL (Fig. 2). The southern watershed, designated as the treatment watershed, is 445 hectares while the northern watershed, designated as the reference watershed, is 312 hectares. The landcover in the area is dominated by tile drained corn-soybean agriculture. Both sub-watersheds consist of tile-drained, privately owned and farmed agricultural land. The tiles drain into drainage ditches which then flow into Money Creek, a tributary of Lake Bloomington. Lake Bloomington is a drinking water reservoir for the City of Bloomington which sometimes shows nitrate levels above the EPA standard of 10 mg/L. Lake Bloomington is a tributary of the Mackinaw River, which is a tributary of the Illinois River, which is a tributary of the Mississippi River. Our two water sampling sites are located where the tile drains emerge from the subsurface to empty into the drainage ditches (Fig. 2).

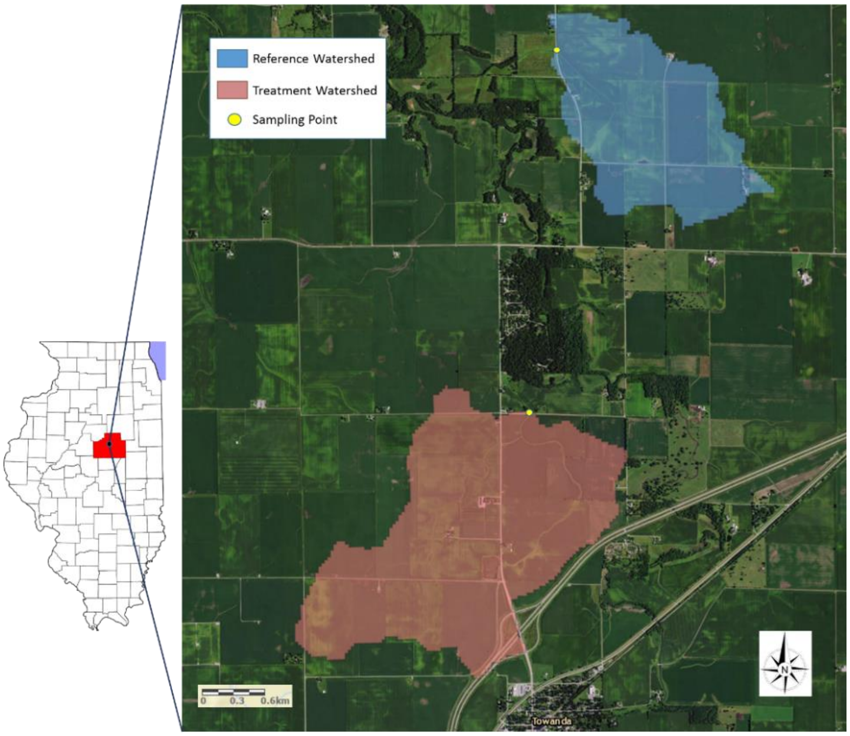


Fig. 2: Study watersheds and water sampling locations

Cover Crop Planting

No cover crops were planted at the reference watershed, while at the treatment watershed 54% of the watershed was planted with cover crops in fall 2015 and fall 2016 (Table 1). Of the 242 ha of land in the watershed planted with cover crops, 213 ha was planted with a mixture of cereal rye and tillage radish, while 30 acres was planted with a mixture of oats and tillage radish in 2015 (Fig. 3). Cover crops were planted via airplane overseeding in late August and early September 2015, before the harvest of the main crop. Cover crops were again planted on the treatment watershed in early September 2016. Rye cover crops were killed with glyphosate application before planting of the main crop in the first week of April 2015. Farmers of the study area chose tilling, planting, and fertilizer application processes independently.

Table 1: Cover crop planting statistics from fall 2015 and fall 2016

Cover Crop	2015 Area planted (ha)	2015 Percentage of watershed planted	2016 Area planted (ha)	2016 percentage of watershed planted
Oat/Radish	30	6.6	92	20.7
Cereal Rye/Radish	213	47.8	29	6.5
Annual Rye/Radish	0	0	121	27.2
Total	242	54.5	242	54.3

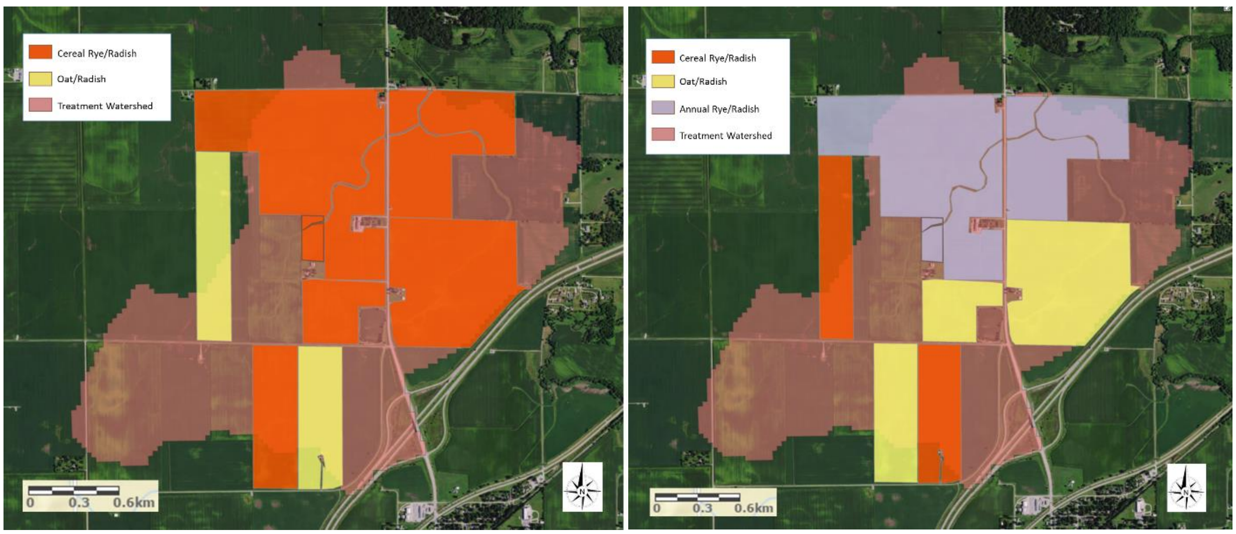


Figure 3: Cover crops planted on our treatment watershed in fall 2015 (left) and fall 2016 (right).

Cover Crop Nitrogen Uptake and Biomass

The cover crop biomass and nitrogen uptake was measured on December 15, 2015, April 2, 2016, and November 18, 2016 to capture both fall and spring cover crop growth each year. The cover cropped watershed was divided into 30-acre grids using GIS, and a sampling point was established at the center of each square in the grid. At these sampling points, 1m x 1m squares were placed on the ground and the surface vegetation within these squares was collected. These samples were then dried and weighed to calculate total biomass. The dried samples were then ground, sieved, and tested for total nitrogen using the dry combustion method.

The nitrogen uptake and biomass values through the fields planted with cover crops were interpolated from the 1m x 1m square values using inverse distance weighting in ArcGIS. Values for fields planted with different cover crops were not allowed to influence each other in these interpolations. By summing the nitrogen and biomass uptake values within the boundaries of our treatment watershed, the total biomass and nitrogen uptake within the treatment watershed through each period of our study was determined.

This study covers the time period between 2/18/16 to 12/13/16 and incorporates three time periods (Table 2). For simplicity, we refer to the periods used in this study as spring, summer, and fall. Spring refers to the early months of 2016, during which overwintering cover crops were growing in the treatment watershed (having been planted in fall 2015). Summer refers to the period of corn or soybean growth. Finally, fall refers to the time period after which cover crops were seeded in 2016. While in practice the cover crops are seeded into the corn or soybeans, at that point the corn and soybeans are not growing, but mostly senescing until ready for harvest.

Table 2: Dates and names of the cover cropping time periods in this study

Time Period	Time Period Name	Cover Cropped?
2/18/16-4/9/16	Spring	Yes
4/10/16- 9/9/16	Summer	No
9/10/16-12/13/16	Fall	Yes

Temperature and Precipitation data

Precipitation data was collected at both sites at 15-minute intervals using two FTS® RG-T tipping bucket rain gauges. Temperature data was collected from a weather station at the Illinois

State University Research Farm located 11.3 km from our reference site and 14.5 km from our treatment site.

Autosampler Discharge and Sample Collection

ISCO® 6712 autosamplers were installed at the reference and treatment sites to measure nutrient concentration and discharge at a greater frequency than our biweekly sampling. One autosampler was installed where the tile drained emerged into the drainage ditch at the cover cropped watershed. Because the reference watershed was drained by three separate tile drains, two autosamplers were installed there to sample the two tile drains with the highest amount of discharge. Due to installation difficulties and software issues, these autosamplers were activated at different times and their discharge measurement methods were changed.

The autosampler at the treatment watershed was activated January 22, 2016. It was initially programmed to take one 110 mL sample every 3 hours. Eight 110 mL samples were collected into each bottle, so that each bottle contained a composite of 8 sampling events throughout a 24-hour period.

On March 16, 2016, this autosampler was switched to discharge based sampling. It was programmed to take 8 110 mL composite samples every day during baseflow conditions at the site. Baseflow at this site was about 4,838,400 L day^{-1}, so the autosampler was programmed to collect a sample every time 604,800 L discharged through the tile drain. On September 8, to increase the resolution of samples during storm events, the programming was changed so that the autosampler collected a sample after every 449,850 L of discharge. From March 16th to August 9th, discharge was measured at 15-minute intervals using an ISCO® 2150 flow velocity meter mounted within the tile pipe. On August 9th, a new Sontek® IQ Pipe flow probe was installed within the tile pipe to improve data management and to improve discharge readings. Because of software issues involving the Sontek® IQ Pipe, the discharge measurements during the timeframe from August 9th to August 19th were sporadic. We used the sporadic velocity measurements from the Sontek® IQ Pipe and interpolated unknown flow velocities linearly. Due to the software issues, we reinstalled the ISCO® 2150 on August 19th. This method of discharge measurement continues to the present.

At the reference watershed, sampling with both autosamplers began on February 17, 2016. Both autosamplers, like the treatment site autosampler, were initially programmed to take one 110 mL sample every three hours. Discharge during this time was measured using two ISCO® 2150 flow velocity probes. Because of the high water clarity of the tile water, which interfered with the ability of the probe to measure flow velocity, the ISCO® 2150s only measured velocity measurements occasionally. Depth was measured by the ISCO® 2150 probes consistently at 15-minute intervals. We created depth-velocity curves by plotting the sporadic velocity measurements combined vs. depth measurements and creating lines of best fit (Fig. 4 a & b). We used these curves to calculate discharge at 15-minute intervals at both reference watershed tiles.

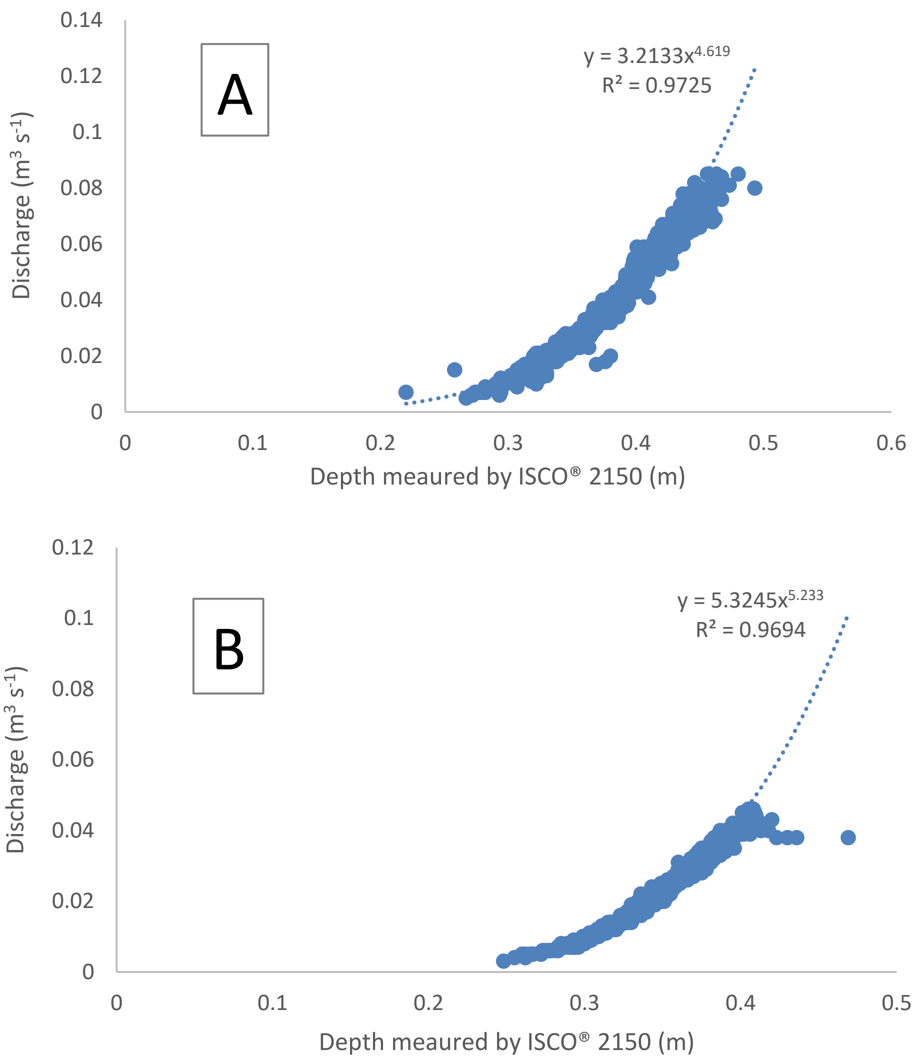


Fig. 4 a&b: Ratings curves used for discharge calculations at the eastern (a) and southern (b) tile drains using ISCO 2150 measurements before installation of pressure transducers.

The southern autosampler was switched to discharge-based sampling on May 27, 2016. A v-notch weir was installed so that it dammed the flow of the tile pipe and forced water to run through the 45° v-notch weir (Fig.5). The head of the water in the pipe relative to the v-notch was measured at 15-minute intervals using an OTT PLS pressure transducer. The following equation was used to calculate discharge at 15-minute intervals:

$$Q = \frac{8}{15} C_d \tan\left(\frac{\theta}{2}\right) \sqrt{2g} \times H^{2.5}$$

Where Q is discharge (cfs), C_d is the coefficient of discharge (set to .60), Θ is the angle of the v-notch weir in degrees, which was 45º at both sites, g is acceleration due to gravity (32.170 ft s^{-2}), and H is the head of the water in the pipe relative to the v-notch weir (ft).

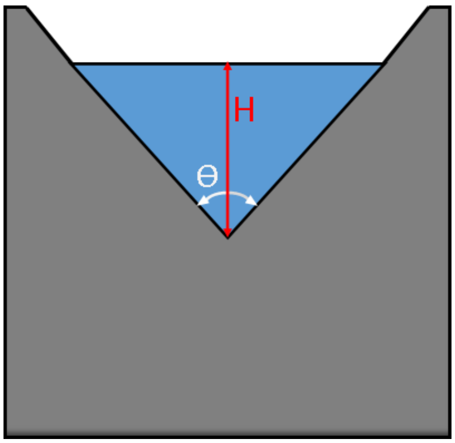


Fig. 5: Diagram of a v-notch weir system and the associated variables

During times of high discharge, water would often flow over the v-notch weir, invalidating the v-notch weir equations. To calculate discharge during these time periods, we created rating curves using discharge measurements from the respective tile drains determined using the same method as was used in our bi-weekly sampling mentioned above. By plotting manual discharge measurements against the depth measured in the v-notch weir systems at the times these measurements were taken, we created rating curves for discharge when water was above the v-notch weir (Fig. 6). The equation from this rating curve was used to calculate discharge whenever the water level rose above the v-notch weir.

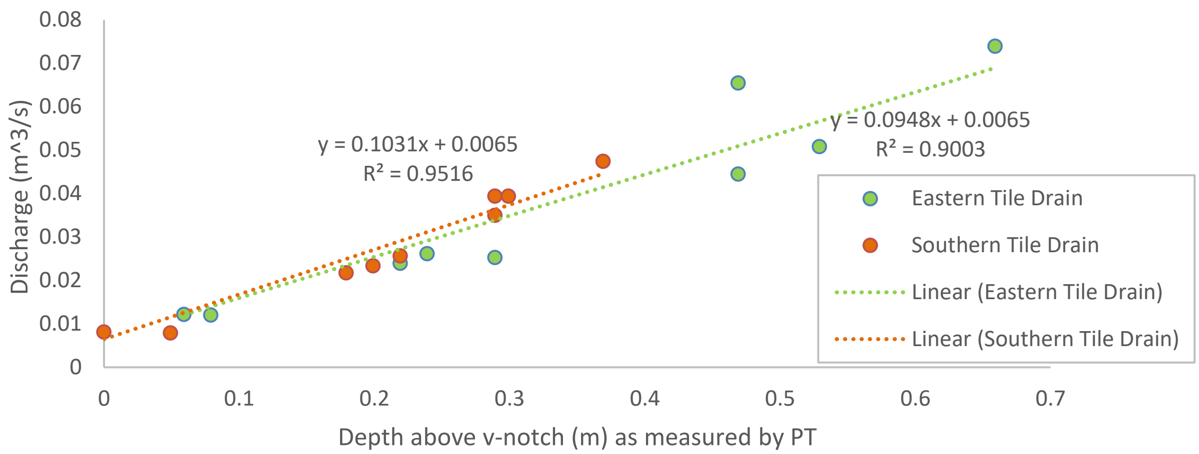


Fig. 6: Rating curves used to calculate discharge from the reference site tile drains when the water level was over the v-notch weirs.

These autosamplers at our reference site, like the one at our treatment site, were programmed in a manner to take approximately eight 110 mL samples per day during baseflow. Since baseflow from the southern tile drain was about 611,600 L day^{-1}, the autosampler was programmed to take sample every time 76,450 L flowed through the tile.

The eastern autosampler was switched to discharge based water sampling with the same v-notch weir/OTT pressure transducer setup as the southern tile drain on July 20th, 2016. From July 20th to September 8th, 2016, this autosampler was programmed to take one 110 mL sample every time 108,000 L flowed through the tile. To increase sample resolution during storm events, on September 8th the autosampler was reprogrammed to take one 110 mL sample every time 70,792 L flowed through the tile. The OTT pressure transducers were calibrated at both sites every 14 days to maintain the accuracy of depth measurements.

Samples were collected from all autosamplers every 14 days. Once collected and transported back to the lab, the samples were immediately refrigerated. Within 48 hours of collection, an unfiltered sample was placed into an acid-washed, pre-labeled 60ml Nalgene bottle, and a second sample was filtered, using either a syringe or Büchner filter, through a 0.45 µm glass fiber filter into an acid-washed, pre-acidified, pre-labeled 60 mL Nalgene bottle. Labeling information included location, collection date, and person sampling. Samples were then logged and refrigerated or frozen to keep the samples at 4°C or below until analysis.

Autosampler Loading Calculations

Loading values were first calculated for the time period of each bottle, which consisted of 8 composite samples. To do this, the concentration from each bottle was multiplied by the total discharge during the time frame the bottle received samples. These values were then divided the time frame during which the bottle received samples (in days) to give us a daily loading value for each period.

To determine daily loading values for the reference site, it was necessary to sum the loading values obtained from both autosamplers at the site. To sum these loading values, we needed to determine daily loading values for every day of our sampling. To do this, we found the time in the middle of the sampling period of each water sample bottle. The loading value associated with each sampling period was assigned to the date of the midpoint. For dates where multiple bottles were collected in one day, and thus there were two midpoints on one day, the loading values determined using each bottle were averaged together and assigned to the date of the midpoints. For dates with missing values, linear interpolation was used to determine daily loading values for each date, so that we had loading values for each day. This allowed us to add the loading values together from both autosamplers at the reference site to obtain the total nitrate and TP loading from our reference watershed.

A potential issue with this method is in the interpolation of loading values for days where there were no bottles because there was low discharge. Because the interpolation is linear, it calculates the loading values for these days of low discharge using the loading values from the days when the previous bottle is collected and the next bottle collected. If the next bottle collected was on the day of a storm event, the loading value was high, and thus when using this high value for interpolation of the missing days, the loading values for these days is overestimated. The peak of the storm event remains correct, but the loading on dry days prior to the storm event is overestimated.

Storm Event Delineation

To calculate statistics of nutrient loading during storm events, storm events were delineated in two ways. The first was by determining days at each site where discharge was above the 75th and 90th percentile daily discharge values. These percentiles were calculated for all three cover cropping periods. This allowed us to calculate differences between time periods of higher flow and time periods of lower flow, as well as differences at the different sites during the cover cropped and non-cover cropped periods.

The second method of defining storm events was by looking for changes in the hydrographs. Periods of high increase in discharge were defined as the beginnings of storm events, and the end of the storms were defined as when change in discharge slowed. This method allowed us to define 10 storm events at each site (Table 3). In this method, we separated the baseflow discharge and loading from the storm event discharge by defining a line of baseflow based on the discharge levels before and after each storm event (Fig. 7). The area of the curve above this line was considered storm event flow, the area below was considered baseflow. Because of an autosampler malfunction at our treatment site during storm event 6 in July 2016, we did not gather nitrate or TP concentration data during that time, and thus that storm event was excluded from our calculations.

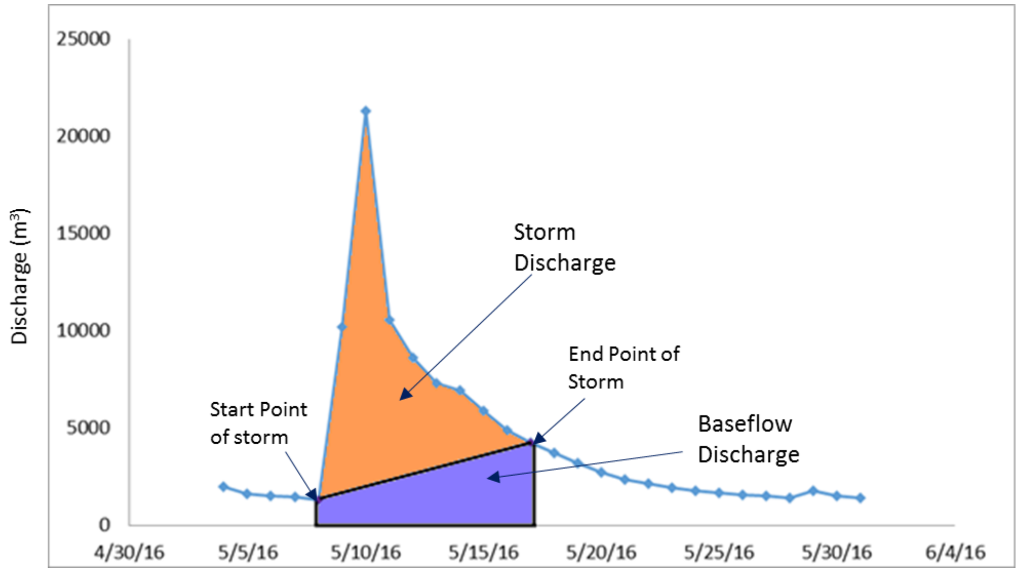


Fig. 7: Example of baseflow and stormflow calculation for a hydrograph-delineated storm event

Table 3: Storm event timing and lengths as delineated by our hydrographs

Storm Number	Storm Start	Storm End	Duration of Storm Hydrograph (days)
1	3/27/2016	4/12/2016	17
2	4/29/2016	5/6/2016	8
3	5/8/2016	5/17/2016	10
4	6/20/2016	6/27/2016	8
5	7/5/2016	7/16/2016	12
6	7/21/2016	8/2/2016	13
7	8/9/2016	8/20/2016	12
8	10/4/2016	10/19/2016	16
9	10/31/2016	11/15/2016	16
10	11/27/2016	12/13/2016	17

Statistical Analyses

Paired t-tests were used to compare the percentage of nitrogen and TP loading that occurred at each site during storm events. A one-sample t-test was used to determine if rain amounts varied between the two sites during the cover cropping periods. Two-sample t-tests were used to determine if the difference between the discharge, nitrate loading, and TP loading at the two sites was higher during the fall period than during the summer period.

Patterns in Temperature, Precipitation, and Discharge

Temperature trends followed the hot summer humid continental climate of Illinois, with hot summers and cool winters (Fig. 8). During the late fall to winter, temperatures fell to levels that were prohibitive to cover crop growth.

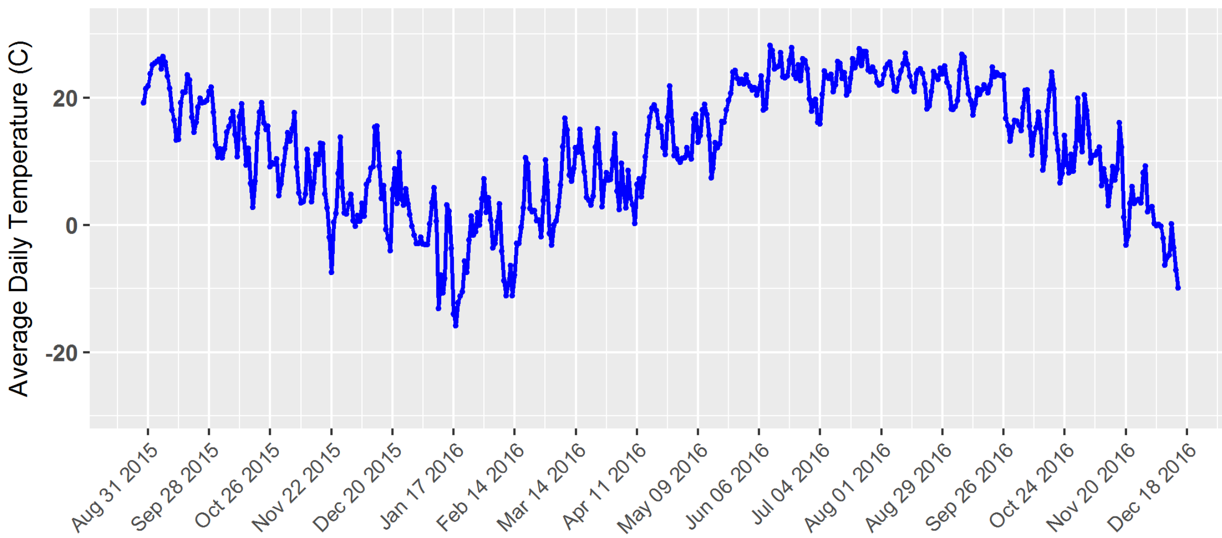


Figure 8: Average daily temperature for the duration of our study

Table 4: Comparison of average monthly temperatures in fall 2015 and fall 2016

	Average Monthly Temperature (^oC)		
Month	2015	2016	Difference (^oC)
September	20.2	20.7	0.39
October	14.5	14.5	2.3
November	7.7	7.7	0.72

Precipitation was highest during the summer months and lowest during the winter and fall at both sites (Fig. 9). There were slight differences in daily precipitation between the reference and treatment sites, but these were not statistically significant.

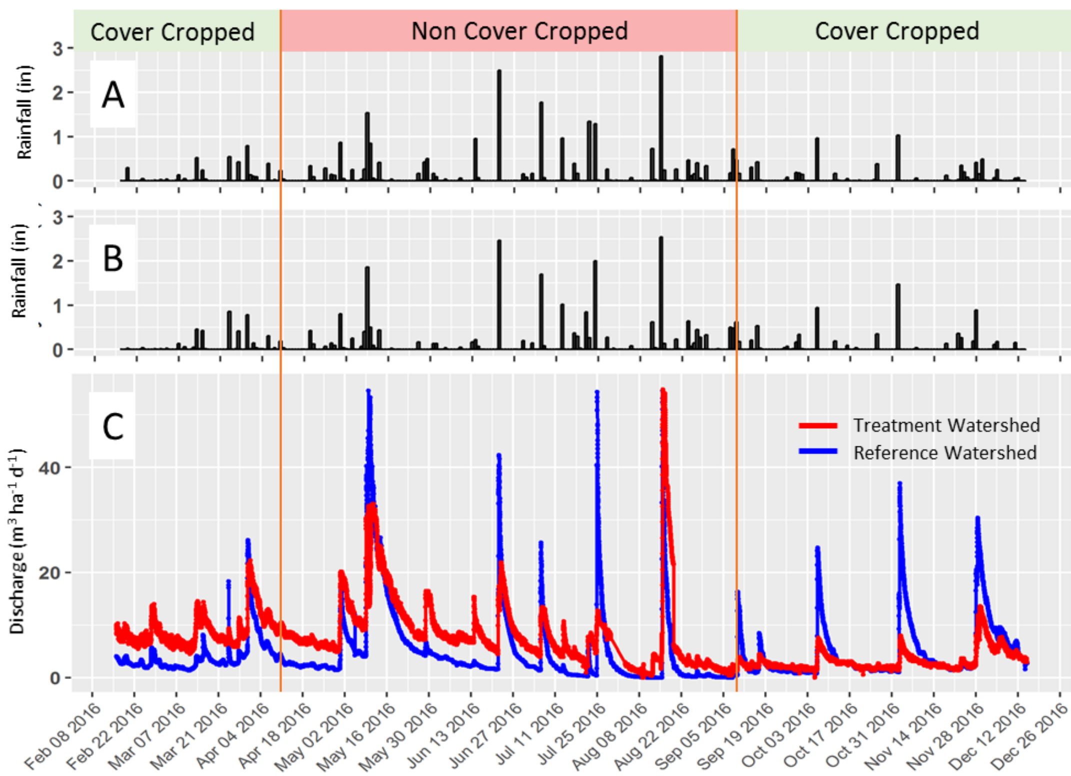


Fig. 9: Precipitation patterns at the A) treatment and B) reference watersheds from February through December 2016 and C) discharge patterns at the reference (blue) and treatment (red) watersheds.

From the beginning of high resolution data collection in February 2016 to early September 2016, discharge per ha at the treatment site was higher at all times except for some periods at the peak of storm events. Peak discharge during storm events was often higher at the reference site before quickly declining again to below the discharge at the treatment site. From early September until the end of our study on December 13, 2016 the pattern changed, discharge at our treatment site became much closer to the discharge at our reference site. This is reflected in a comparison of the mean discharge values from each site during the summer 2016 non-cover cropped period and the fall 2016 cover cropped period (Fig. 10), where mean discharge per ha was higher at our treatment site than our reference site in every period but the fall 2016 cover cropped period. The difference in discharge between the two watersheds was significantly greater during the non-cover cropped period than it was in the cover cropped period ($p < 0.001$)

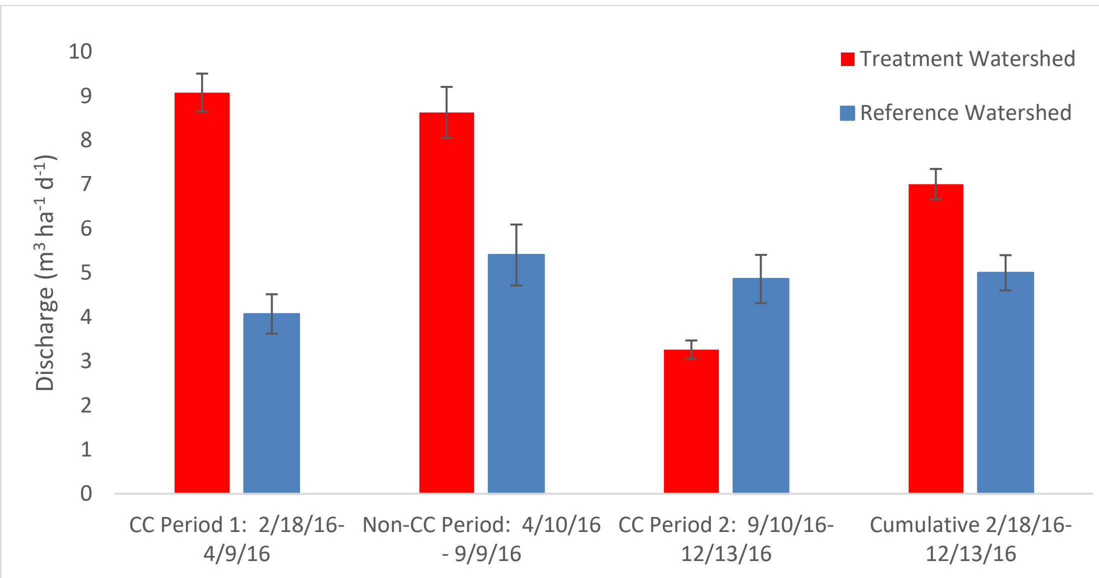


Fig. 10: Comparison of mean daily discharge per ha at our treatment and reference sites during the different cover cropping periods.

Nutrient Concentrations and Losses: Nitrate

Nitrate concentrations were highest at both sites during spring to mid-summer (Fig. 11). During this period (March 15 to July 15), mean nitrate concentrations at the treatment site were 10.13 mg L^{-1} and 11.19 mg L^{-1} at the reference site. Concentrations peaked during a period from early to mid-May, where a maximum concentration of 18.5 mg L^{-1} was reached at our treatment site on May 4th and 15.15 mg L^{-1} was reached at our reference site on May 5th.

Concentrations throughout the rest of the year were generally lower. The average nitrate concentration from July 16th 2016 to December 13th, 2016 was 7.99 mg L^{-1} at our treatment site and 8.02 mg L^{-1} at our reference site. Overall, nitrate concentrations were above the EPA nitrate standard of 10 mg L^{-1} for 42.5% of samples across the duration of the study at our treatment site and for 71.4% of samples the duration of the study at our reference site.

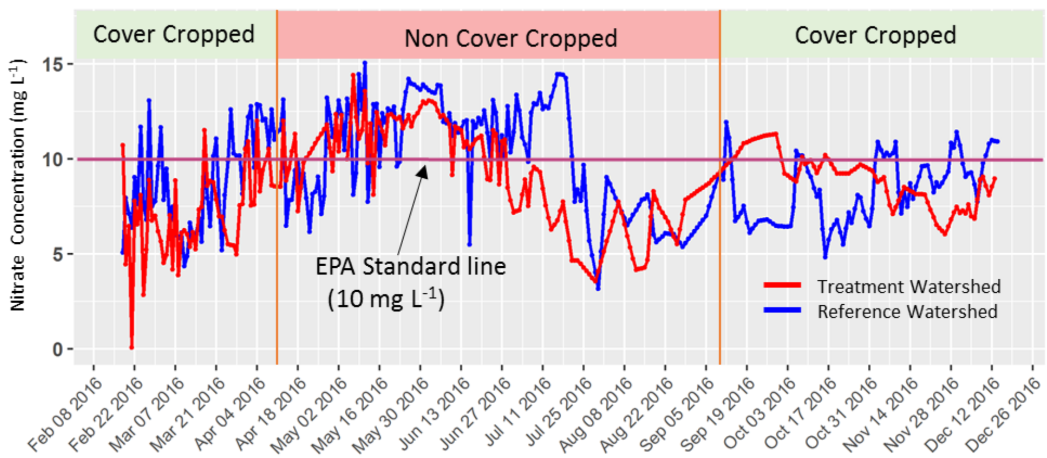


Fig. 11: Time series of nitrate concentration at our treatment watershed (red) and our reference watershed (blue). The EPA standard of 10 mg L^{-1} is shown by the purple line.

The nitrate concentrations did not differ from summer to fall at our treatment site ($p>0.05$). At the reference site, summer nitrate concentrations were significantly higher than those in fall ($p <0.0001$).

Nitrate loading per hectare followed the trend of discharge (Fig. 12). For most of the study period, nitrate loading per hectare was higher at the treatment side than at the reference site. However, during fall this trend changes and nitrate loading at our treatment site becomes much lower relative to the loading at our reference site. A comparison of the difference in daily loading between the two sites reflects this: the difference was significantly lower during fall than during summer ($p<0.001$).

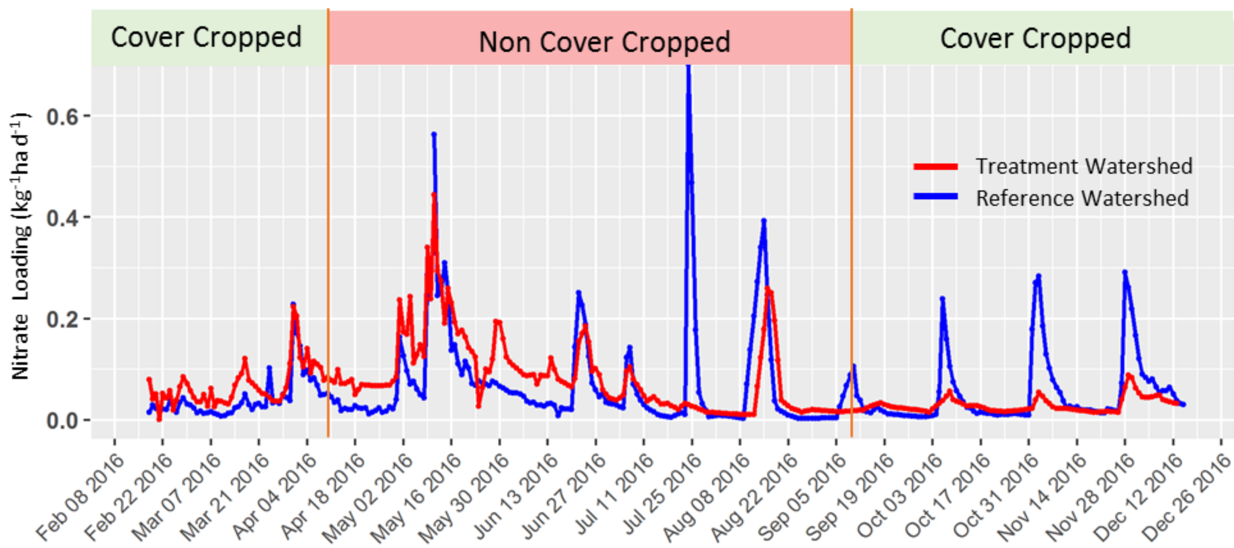


Figure 12: Time series of nitrate loading at our treatment (red) and reference (blue) watersheds.

This change in nitrate loading pattern was also seen when comparing mean nitrate loading of the two sites during the three time periods (Fig. 13). In spring, the average nitrate loading per ha day was 1.6 times higher at the treatment watershed than at the reference watershed. In summer, the average loading per ha was 1.1 times higher at the treatment watershed than at the reference watershed. In fall, the pattern was reversed; the average daily nitrate loading was 1.8 times higher at the reference watershed than the treatment watershed.

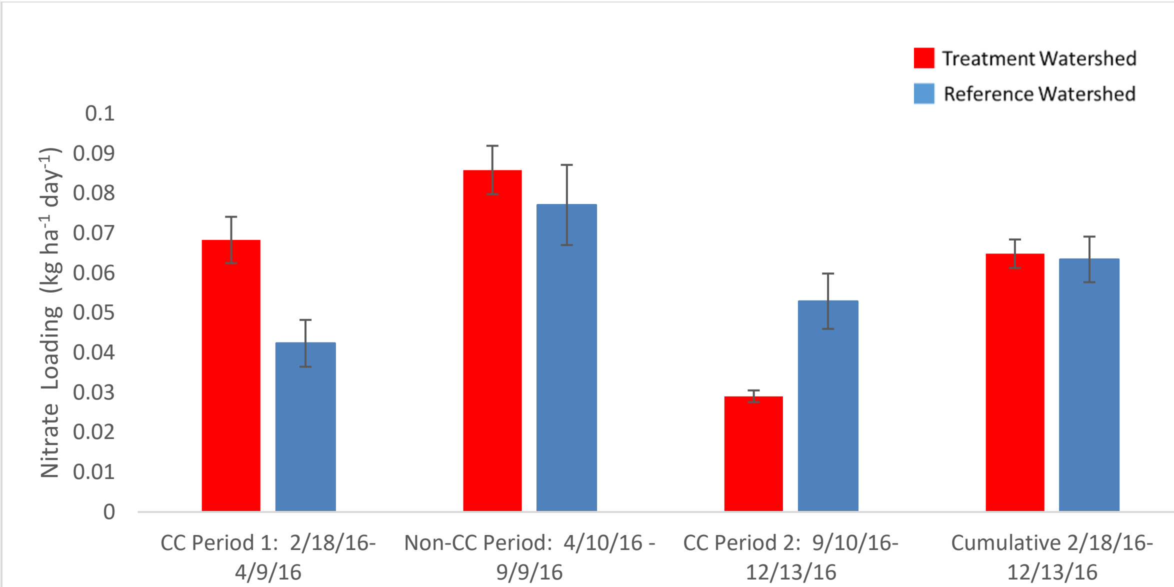


Fig. 13: Comparison of average nitrate loading between our treatment (red) and reference (blue) watersheds across different time periods.

During all periods of the study, average flow weighted nitrate concentration was higher at the reference site than at the control site. In spring, the flow-weighted concentration was 1.38 times higher at the reference site than at the treatment site. In summer, the flow-weighted concentration was 1.54 times higher at the reference site than the treatment site. In fall, the reference site had a flow-weighted nitrate concentration 1.67 times higher than that at the treatment site (Fig. 14).

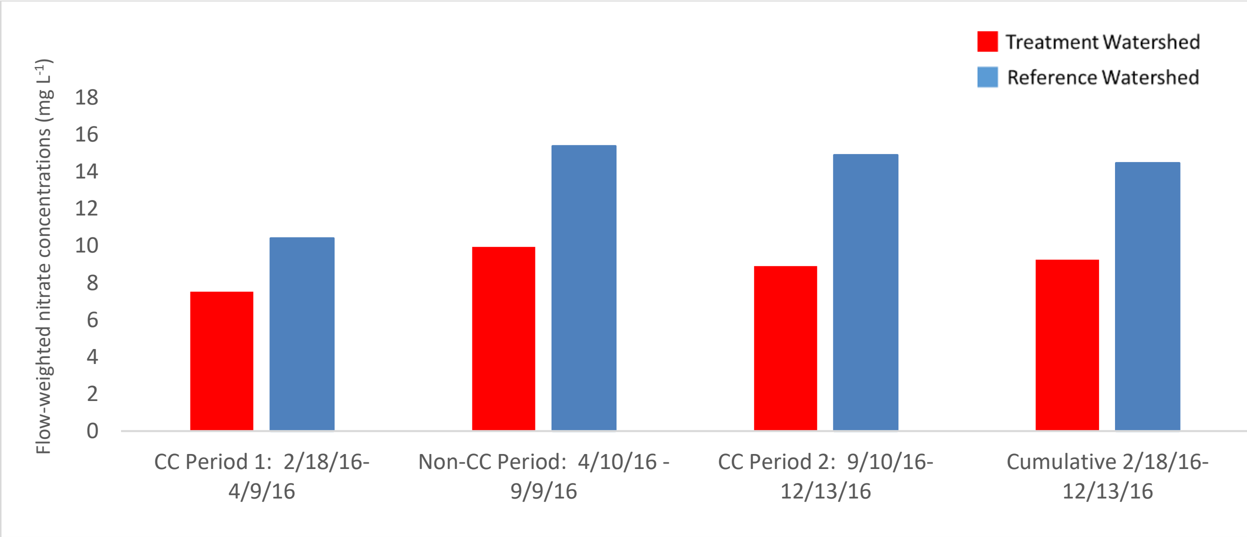


Fig 14: Flow weighted nitrate concentrations for our treatment (red) and reference (blue) sites across different time periods.

Influence of Storm Events on Nitrate Loading

Periods of high discharge had a disproportionate impact on the loading regimes of the sites, especially at the reference site (Figs. 15 a & b). Days with discharge greater than the 90th percentile were responsible for 38.0% of nitrate loading for the duration of the high resolution portion of the study at our reference site and 32.7% of the nitrate loading from our treatment site. The difference in this amount between the two sites was most pronounced during fall, where 38.5% of nitrate loading at the reference site happened on days with >90th percentile of discharge, compared to 19.96% at our treatment site.

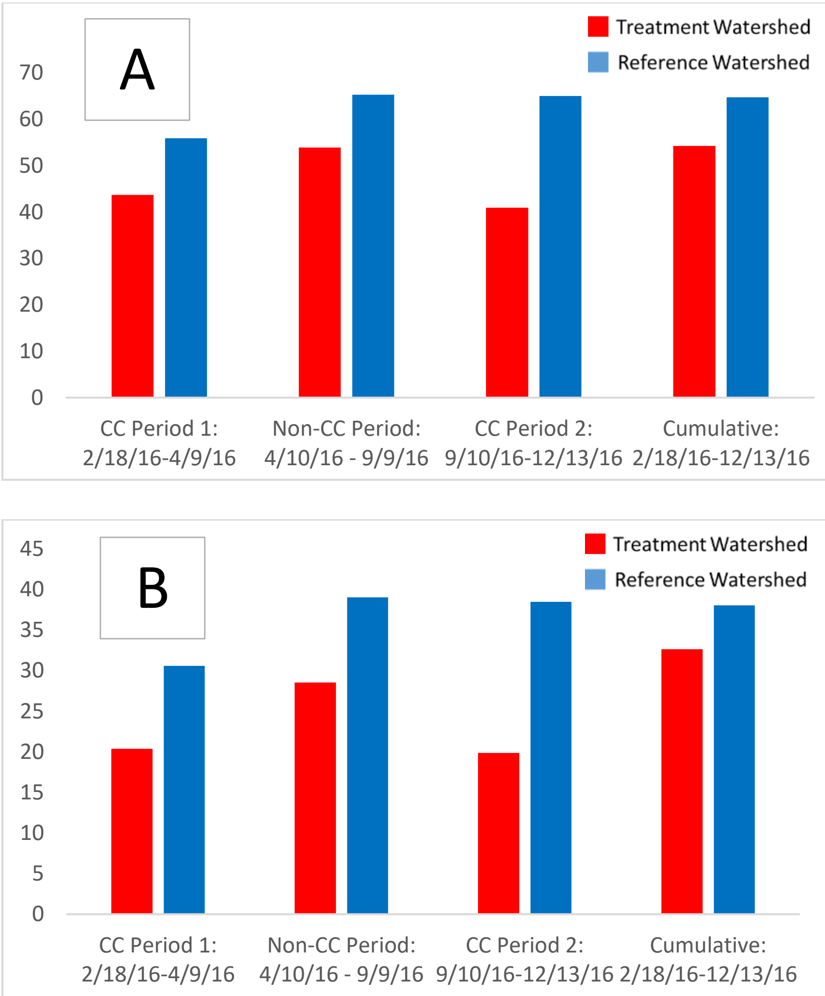


Fig 15 a & b: Comparison of percentages of nitrate loading that occurred during storm events where discharge was above the 75th (a) and 90th (b) percentiles across different time periods at our treatment (red) and reference (blue) watersheds.

The percent nitrate contribution that occurred during storm events as delineated through our hydrographs was statistically greater at T2 than at T1 ($p<0.001$). The reference-to-treatment nitrate loading ratio was much higher during storms in the second cover cropping period than it was during storms in the non-cover cropped and first cover-cropped period (Table 5).

Table 5: Nitrate loading at our treatment and reference sites during each hydrograph-delineated storm event. Note how the reference/treatment loading ratio was much higher during the second cover cropping period.

Cover Cropping Period	Storm Number	Storm Start	Storm End	Length of Storm (days)	Storm Nitrate Loading (kg ha^{-1} d)		Reference/Treatment Loading Ratio
					Treatment	Reference	
CC1	1	3/27/2016	4/12/2016	17	0.0502	0.0465	0.9250
NCC	2	4/29/2016	5/6/2016	8	0.0504	0.0408	0.8102
NCC	3	5/8/2016	5/17/2016	10	0.1002	0.2188	2.1837
NCC	4	6/20/2016	6/27/2016	8	0.0426	0.0956	2.2457
NCC	5	7/5/2016	7/16/2016	12	0.0184	0.0304	1.6525
NCC	7	8/9/2016	8/20/2016	12	0.0810	0.1450	1.7901
CC2	8	10/4/2016	10/19/2016	16	0.0090	0.0449	5.0129
CC2	9	10/31/2016	11/15/2016	16	0.0081	0.0799	9.8659
CC2	10	11/27/2016	12/13/2016	17	0.0270	0.0818	3.0289

Components of Nitrate Loading

Nitrate loading was linearly related to discharge (R^2 =0.77, p= <2.2x10^{-16} for treatment, R^2 =.64, p<2.2x10^{-16} for reference) (Fig. 16). The relationship between nitrate concentration and nitrate loading was less pronounced at both sites (R^2 =0.21, P <2.2x10^{-16} for treatment, R^2 =0 .074, p=1.613x10^{-6} for reference) (Fig. 17). The relationship between daily discharge and nitrate concentrations was weak at both sites (R^2 =0.101, p=1.854x10^{-6} for reference, R^2=0.0236, p=0.2065 for treatment) (Fig. 18).

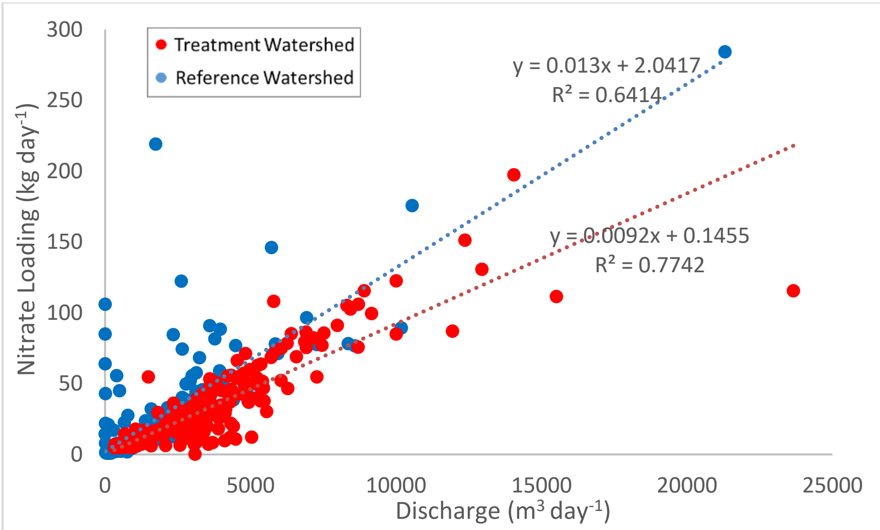


Fig 16: Relationship between discharge and nitrate loading at our reference (blue) and treatment (red) sites.

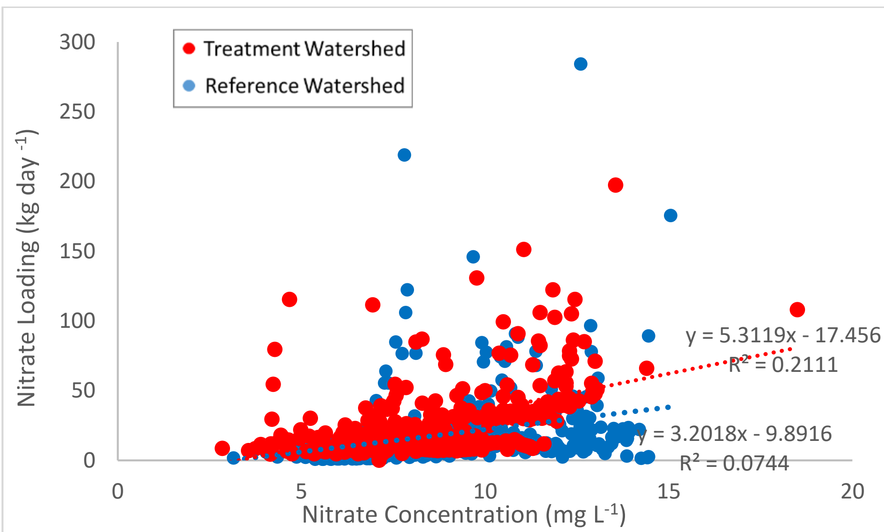


Fig 17: Relationship between nitrate concentration and nitrate loading at our treatment (red) and reference (blue) sites.

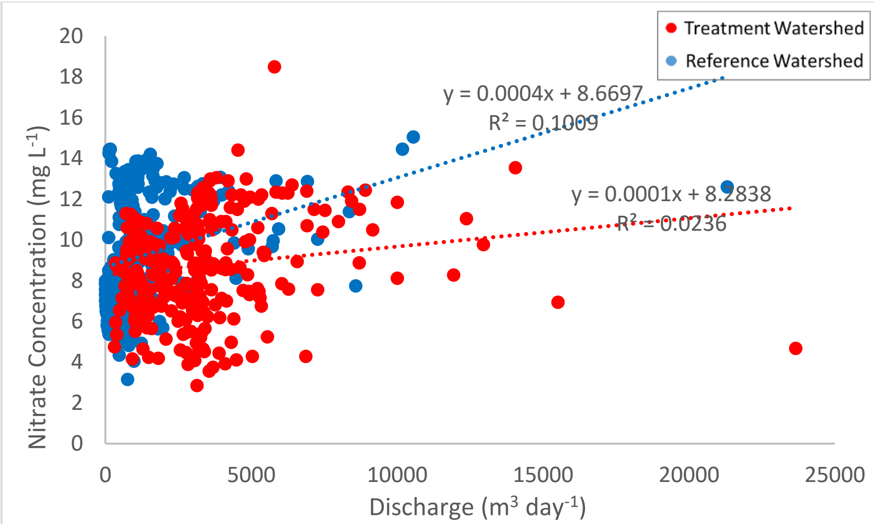


Fig 18: Relationship between discharge and nitrate concentration at our treatment (red) and reference (blue) sites.

Nutrient Concentrations and Losses: Total Phosphorus

TP concentrations tended to remain low and stable during baseflow but spiked severely and rapidly during storm events (Fig 19). Because these spikes started and ended in such a short timeframe, they were usually missed by our biweekly sampling. These spikes were more pronounced at our treatment site, as indicated by the higher concentration values at higher percentiles (Table 6).

Table 6: Percentiles of TP concentrations at our reference and treatment sites

	TP concentration (µg L-1)	
Percentile	Reference	Treatment
50th	28.9	56.1
75th	53.2	158.8
90th	84.9	393.4
95th	114.2	491.7

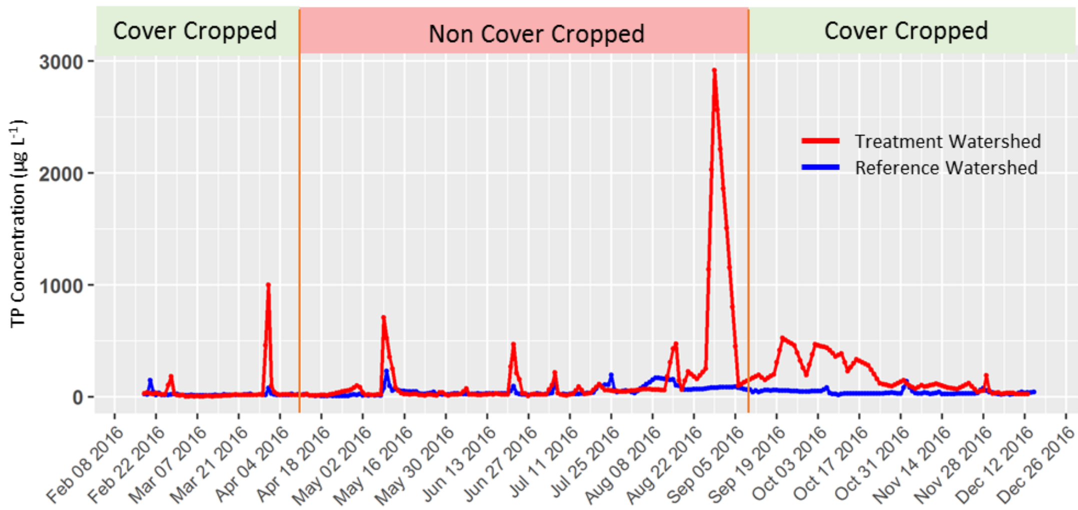


Fig. 19: TP concentration time series of our treatment (red) and reference (blue) sites.

TP loading, like TP concentration, remained low and stable during periods of baseflow and spiked during periods of storm flow (Fig. 20). This loading was often caused by an increase in discharge combined with a drastic increase in TP concentrations. The difference in TP loading per hectare between the reference and treatment sites during the summer 2016 non-cover cropped period was not significantly greater (p=0.05) than the difference in TP loading per hectare between the control and treatment sites during the fall 2016 cover cropped period.

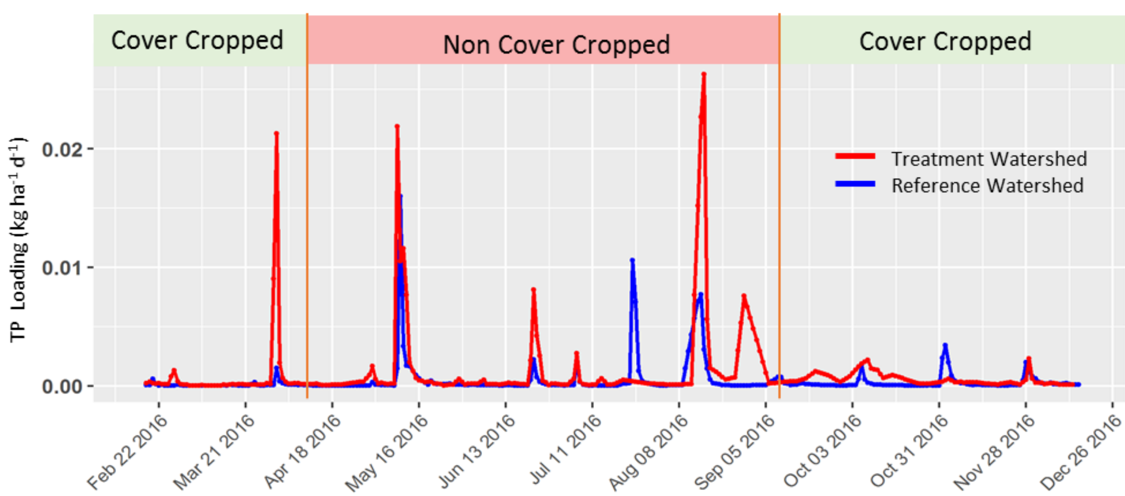


Fig. 20: Time series of TP loading form our treatment (red) and reference (blue) watersheds.

TP daily loading per hectare and flow weighted concentration were both consistently much higher at our treatment site than at our reference site (Figs 21 & 22). For TP daily loading per hectare, the difference was greatest during the first cover cropping period, with the daily TP loading per hectare being 3.1 times greater at the treatment site during this period compared to 1.36 times greater during both the non-cover cropped period and 2.15 times greater during the second cover cropped period.

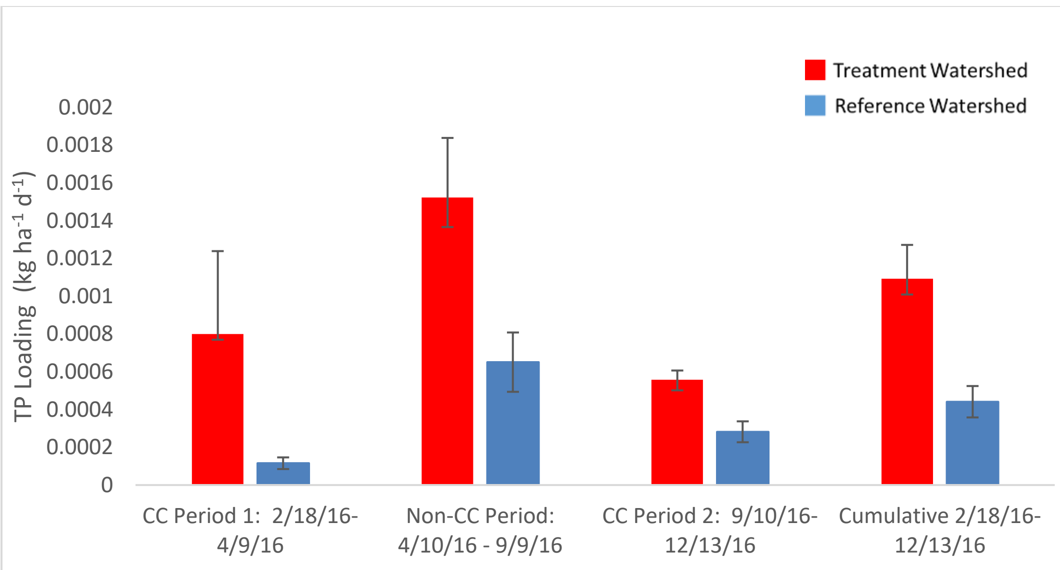


Fig. 21: Comparison of average TP loading between our treatment (red) and reference (blue) watersheds across different time periods.

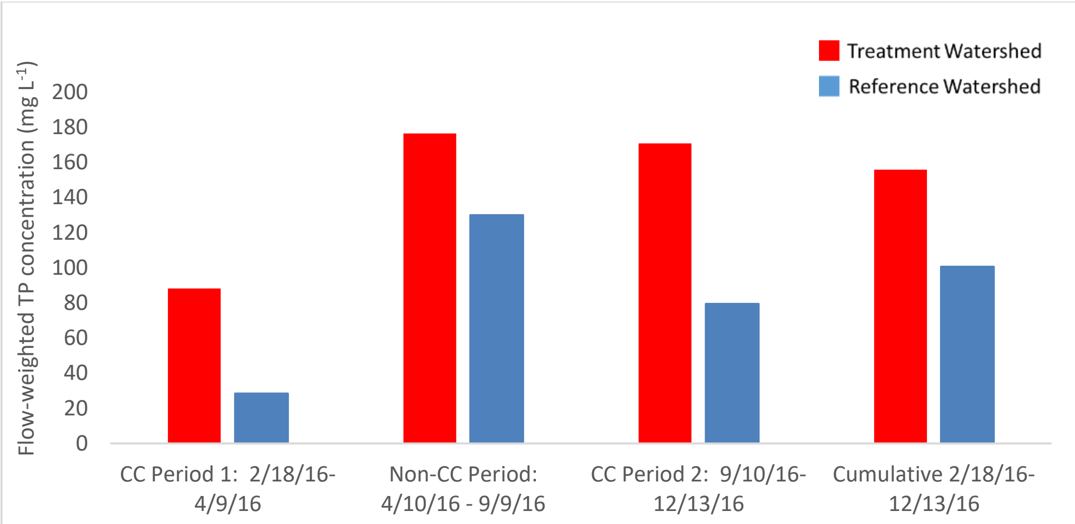
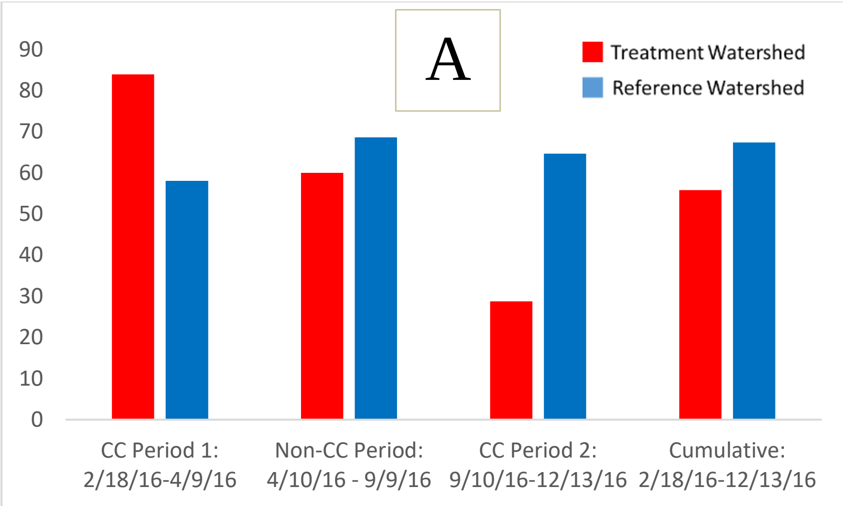


Fig. 22: Comparison of average TP loading between our treatment (red) and reference (blue) watersheds across different time periods.

Influence of Storm Events on TP Loading

The effect of high discharge on TP loading was even greater than its effect on nitrate (Figs. 23 a & b). Days with >90th percentile discharge were responsible for 41.4% of the TP loading at the reference site and for 48.2% of our loading at the treatment site during the duration of the high resolution portion of our study. Again, this percentage during the second cover cropped period was much higher at our reference site (38.5%) than at our treatment site (13.4%).



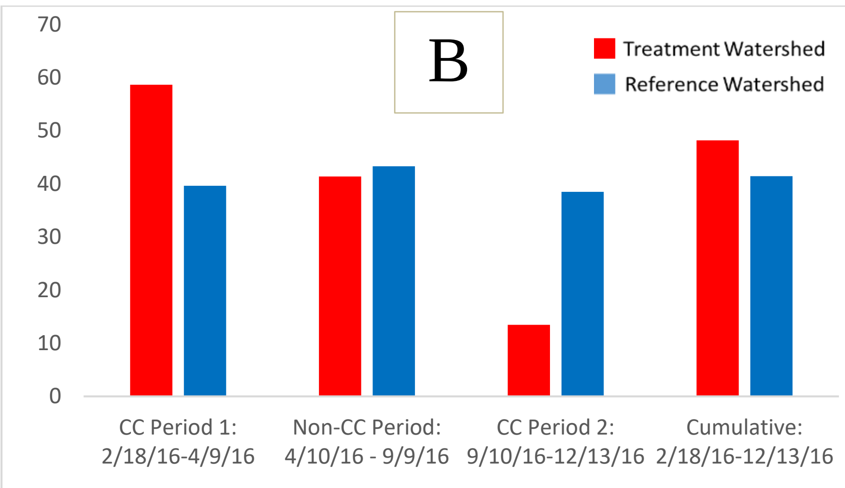


Fig 23 a&b: Comparison of percentages of TP loading that occurred during storm events where discharge was above the 75th (a) and 90th (b) percentile across different time periods at our treatment (red) and reference (blue) watersheds.

The percent of TP loading that occurred during storm events throughout the entire study period as delineated by our hydrographs was not statistically greater at our reference site than at our treatment site (p=0.0661). There was not a very strong pattern of the reference/treatment TP loading ratio being higher during the second cover cropping period than the first (Table 7).

Table 7: TP loading at our treatment and reference sites during each hydrograph-delineated storm event. Cover cropped periods are highlighted in green, non-cover cropped periods in yellow.

Cover Cropping Period	Storm Number	Storm Start	Storm End	Length of Storm (days)	Storm TP Loading (kg ha^{-1}d-1)		Reference/Treatment Loading Ratio
					Treatment	Reference	
CC1	1	3/27/2016	4/12/2016	17	0.001955	0.000150	0.08
NCC	2	4/29/2016	5/6/2016	8	0.000176	0.000060	0.34
NCC	3	5/8/2016	5/17/2016	10	0.005357	0.002513	0.47
NCC	4	6/20/2016	6/27/2016	8	0.002144	0.000483	0.23
NCC	5	7/5/2016	7/16/2016	12	0.000139	0.000205	1.48
NCC	7	8/9/2016	8/20/2016	12	0.006176	0.002767	0.45
CC2	8	10/4/2016	10/19/2016	16	0.000238	0.000202	0.85
CC2	9	10/31/2016	11/15/2016	16	0.000080	0.000614	7.69
CC2	10	11/27/2016	12/13/2016	17	0.000262	0.000359	1.37

Cover Crop Biomass and Nitrogen Uptake

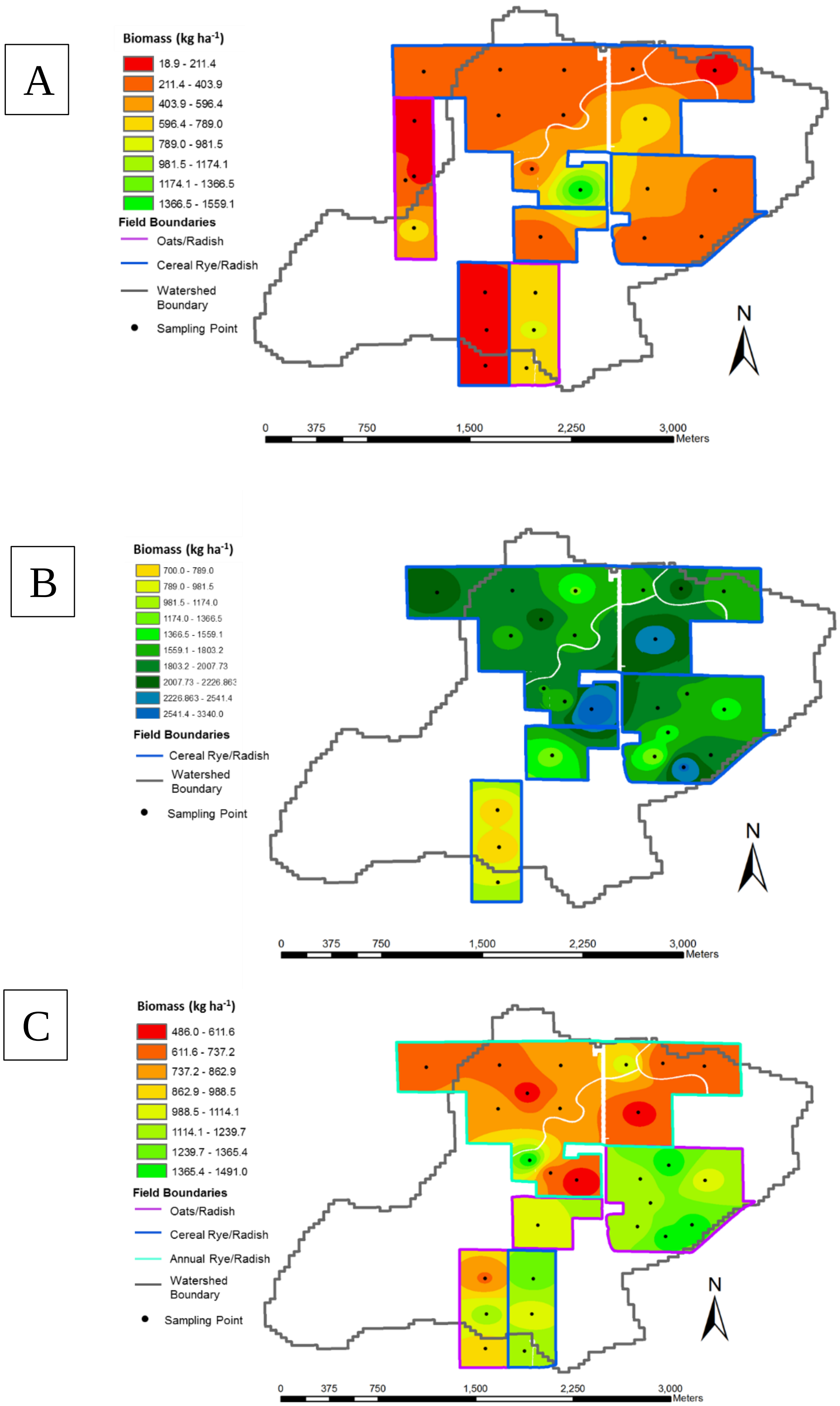


Fig. 24: Cover crop biomass in fall 2015 (a), spring 2016 (b), and fall 2016 (c)

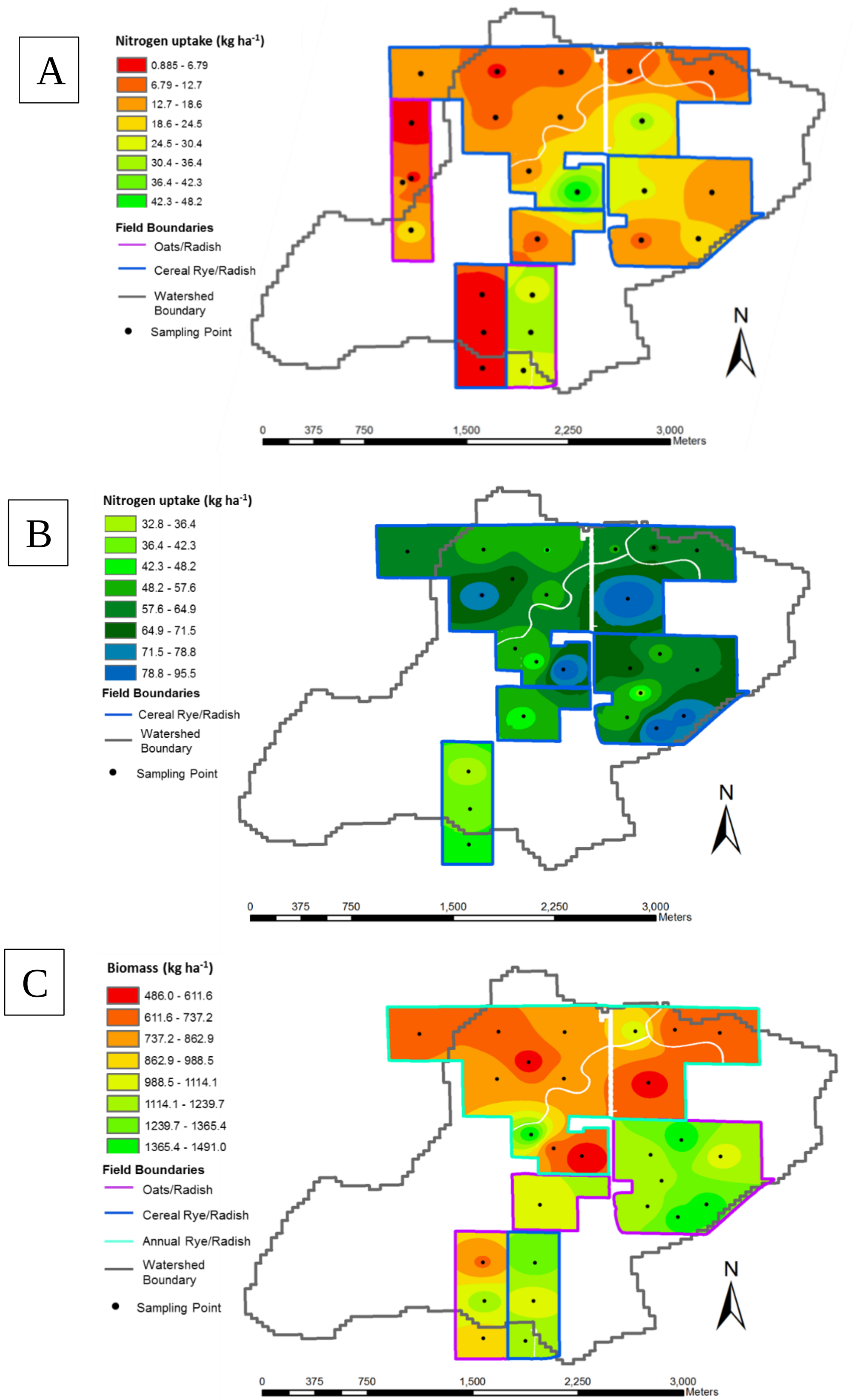


Fig. 25: Cover crop nitrogen uptake in (a) fall 2015, (b) spring 2016, and (c) fall 2016

Growth of each type of cover crop was greater in fall 2016 than in fall 2015 (Table 7). Nitrogen uptake was also greater in fall 2016 than in fall 2015 (Table 7). Cereal rye displayed higher biomass and nitrogen uptake than the other cover crops, especially when taking its spring growth into account (Table 8). Cereal rye is perennial, and thus regrew in spring, while oats lasted only for the fall season. The spring 2016 growth of rye (1762 kg/ha) was 4.4 times higher than the fall 2016 growth of rye (405 kg/ha). The spring biomass and nitrogen uptake of cereal rye were higher than any other biomass or nitrogen uptake. In fact, even though less total acreage of cover crops grew in spring 2016 than fall 2015 or fall 2016, total biomass was still higher in spring 2016 due to the strong regrowth of cereal rye.

Table 8: Biomass and nitrogen uptake statistics of the different varieties of cover crops planted within our treatment watershed.

	Biomass (kg ha^{-1})			Nitrogen Uptake (kg ha^{-1})		
	Fall 2015	Spring 2016	Fall 2016	Fall 2015	Spring 2016	Fall 2016
Oats/Radish	640	0	1159	23.9	0	40.5
Cereal Rye/Radish	405	1762	1182	14.9	54.2	43.1
Annual Rye/Radish	NA	NA	760	NA	NA	24.6

Biomass was almost four times greater in spring 2016 than it was in fall 2015 (Table 9). Spring 2016 had the greatest total biomass and nitrogen uptake despite having only 213 acres of growth.

Table 9: Total and per ha biomass and nitrogen uptake statistics from our three seasons of cover crop growth

Time Period	Total area (ha)	Total Biomass (kg)	Biomass per ha (kg ha^{-1})	Total Nitrogen Uptake (kg)	Nitrogen uptake per ha (kg ha^{-1})
Fall 2015	242	105080	433	4337	18
Spring 2016	213	374967	1762	12940	61
Fall 2016	233	222276	954	7565	32

CHAPTER IV: DISCUSSION

Hydrologic Patterns in Precipitation and Discharge

Discharge was greatest in May and June and reached low points in August through October. This was similar to the patterns observed in other Midwestern studies (Royer et al. 2006, Kaspar et al 2007), and corresponds with seasonal precipitation patterns found in the Midwest. The highest discharge peaks occur soon after precipitation with little delay. This rapid response to storm events, both at our sites and in other studies, suggests quick percolation of precipitation through the soil into tile drainage (Schilling et al. 2008, Smith et al. 2015). This quick link means that there is little time for TP to sorb to soil, which many contribute to the high TP concentrations we see during storm events.

Discharge appears to be reduced at our treatment site in the second cover cropping period, as shown by the reduced difference between loading our treatment site and loading at our reference site. This change was not likely due to differences in precipitation between the two sites, as the difference in precipitation between the two sites was insignificant for the second cover cropping period. Cover cropping is commonly found to reduce discharge through increased evapotranspiration (Dabney et al 1998, Logsdon et al. 2002, Strock et. al 2004) and this discharge reduction is one mechanism through which cover cropping decreases nutrient loading.

Link between temperature and cover crop growth

Cover crop growth was higher in fall 2016 than in fall 2015. The reason for this increased growth is likely the warmer weather from September-October in fall 2016. The pattern of better cover crop growth in warmer seasons has found in other studies. Strock et al. (2004) found that a combination of dry and cold weather one year caused cover crop biomass to drop to one-fifth of what it was in a year with better growing conditions. Lacey and Armstrong (2015) found that tillage radish growth was 1.8 times higher in a warmer fall season than the growth in a cooler year.

TP Loading

TP loading tended to be dominated by storm events at both sites, during which TP concentrations rose rapidly for a brief period before quickly returning to baseflow levels. This rapid rise in concentration combined with the increase in discharge leads to sharp spikes in TP loading. This pattern of storm-dominated phosphorus loading is commonly found in agricultural watersheds. Royer et al. (2006) found in a study of streams in east-central Illinois that >80% of the phosphorus loading in the studied watersheds occurred during extreme storm events when discharge was $\geq 90^{th}$ percentile. In the same area, Gentry et al. (2007) found that TP concentrations in tile drainage peaked during times of high discharge before returning to normal during baseflow. Dils and Heathwaite (1999) found, in an agricultural watershed in the UK, that TP was less than 100 µg/L during baseflow, but that TP sometimes reached concentrations above 1000 µg L^{-1} during storm flow periods. Because TP loading was dominated by these short-term storm events, high temporal resolution concentration and discharge monitoring was essential to properly calculate TP loading since lower resolution data is unlikely to capture these storm events (Royer 2006, Gentry et al. 2007).

While surface flow often contains greater concentrations of phosphorus, tile drainage can nevertheless cause a significant percentage of tile-drain loading in heavily tile drained areas.

Smith et al. 2015, Van Esbroeck et al 2016). In east-central Illinois, Algoazany et al. (2007) found that greater TP loads occurred via tile drainage than via surface runoff. In Indiana, Smith et al. (2015) found that 48% of TP loading in 4 monitored fields occurred through subsurface tile drainage. In Ontario, Van Esbroeck et al. (2016) found that tile drains exported 40-77% of TP loads from agricultural fields. While overland flow was not monitored at our sites, the results of these studies at similar sites as well as the high degree of tile drainage within our watersheds suggest that they transport a significant amount of the total TP loading from our watersheds.

The findings on the effects of cover cropping on phosphorus export have been inconsistent, from actually increasing phosphorus concentrations after freezing by a small amount (Ulén 1997) and creating a source of DRP during winter thaw (Øgaard 2015), to not having any effect (Arronson et al. 2016) to a reduction in phosphorus loading when cover crops were planted simultaneously with corn (Kleinman et al. 2005). Our site likewise showed inconsistencies. There was a lesser percentage of TP loading that occured during storm events during the fall 2016 cover cropped period at our treatment site relative to our reference site when classifying storm events by the percentile of discharge. However, this does not hold true when incorporating baseflow discharge, as the difference in TP loading per hectare between the reference and treatment sites during summer was not significantly greater ($p=0.0522$) than the difference in TP loading per hectare between the reference and treatment sites during fall. Also, when looking at the study as a whole, there was no significant difference in TP loading between our reference and our treatment sites ($p=.0661$), and there was no noticeable pattern of TP loading reduction in spring 2016. Overall, there may be potential for cover cropping to reduce TP loading by reducing discharge.

The reason for the lack of a difference between the two sites in spring 2016, when cover biomass and growth was greater than during fall 2016, may be due to differences in the type of cover crop planted in spring vs the fall, which I discuss in greater detail in the nitrate loading section. Another reason may be that differences between the soils at each site led to different sediment and TP transport dynamics. The difference in sediment transport was noticed when measuring the turbidity at each site. The turbidity at our reference site was much lower than the turbidity at our treatment site, which corresponds with the higher amount of TP export from our treatment site. Because there is such a difference in magnitude in TP transport between the two sites, due in part to the difference in sediment export between the two sites, the impact of cover crop planting on the TP export dynamics of our treatment watershed during spring is difficult to measure.

Nitrate Loading

Loading Reduction in Fall

In fall, there was a pattern of reduced nitrate loading due to decreased discharge. There was a noticeable change near the start of fall, where the nitrate loading at our treatment site became lower with relation to our reference site than it was during summer (Fig. 13). The difference between the loading at our treatment site and the loading at our reference site was significantly lower during the fall 2016 cover cropping period than during the summer 2016 non-cover cropped period.

This shift in the nitrate loading difference was most noticeable during storm events. In summer, the average daily nitrate loading during storm events was usually relatively similar between the two sites. However, during fall, the average daily nitrate loading during storm events became much higher at the reference site than the treatment site, as shown by the reference/treatment nitrate loading ratio (Table 5). This shift was also noticeable in percentages of nitrate loading that occurs during high discharges: in fall, nitrate loading per ha was less from the treatment watershed relative to the reference watershed than it was in the summer (Fig 13).

The main mechanism through which nitrate loading was reduced during fall appears to bed this reduction of discharge, rather than a reduction in nitrate concentration. If reduction in nitrate concentration was the major mechanism, we would expect to see a decrease in nitrate concentration at our treatment site relative to our reference site, but this did not occur. The nitrate concentrations at the treatment site did not change significantly from summer to fall, and the difference in nitrate concentrations between the treatment and the reference sites was actually larger during fall than it was during the summer, which suggest that a reduction in nitrate concentration was not responsible for the observed reduction in nitrate loading. Additionally, if nitrate concentration was responsible for loading reduction, we would expect to see a strong correlation between concentration and nitrate loading, but this relationship was weak (R^2=0.21) (Fig.17) while there was a relatively strong relationship between discharge and nitrate concentration (R^2=0.77) (Fig. 16).

A third indicator that discharge was the driver of the reduced nitrate loading is the lack of change in flow-weighted nitrate concentration between summer and fall. Flow-weighted concentration is the total amount of loading during a given period divided by the total amount of discharge during that same period. If reduced concentration was the driver of reduced nitrate loading, we would expect to see a corresponding reduction in flow-weighted concentration. The fact that we do not see a reduction in flow-weighted concentration suggests that the reduced nitrate loading is instead due to a reduction in discharge.

The mechanisms through which nitrate loading is reduced in other Midwestern cover cropping studies, whether via reduced concentration or reduced discharge, varies. Kaspar et al. (2007), in a plot scale study in Iowa using a rye cover crop, found that of the 61% average reduction in nitrate export they observed, only about 16% of it could be attributed to reduced discharge, while the rest was due to a reduction in nitrate concentration. In a 2012 follow up study at the same location, Kaspar et al. (2012) found a 48% reduction in flow-weighted concentration when comparing rye planted fields to control fields, but found no significant reduction in discharge.

However, in other cases nitrogen reduction is due to declines in discharge. Strock et al. (2004) performed a plot scale watershed study in Minnesota and found that a plot planted with a cereal rye cover crop experienced a 13% reduction in nitrate loss compared to a control plot. This was largely due to the 11% reduction in drainage in the cover cropped field. Armstrong and O'Reilly (2017), in an Illinois plot scale study nearby our study site, found a 51-67% reduction in nitrate loading in cereal rye/tillage radish planted fields vs. control fields during cover cropped periods. A good portion of that reduction was due to a 32%-42% reduction in discharge from the cover cropped fields during the cover cropped periods. In these studies, reduction of discharge has been a viable mechanism for reduction of nitrate loading.

There is evidence that cover crops may have an impact on discharge from cover cropped fields even when cover crop coverage is sparse. In one year of Strock et al (2004), cover crop growth was sparse due to poor weather conditions. In that year, 2001, they only had 0.5 Mg ha^{-1} of cover crop growth, compared to 1.0 Mg ha^{-1} and 2.7 Mg ha^{-1} in the 2000 and 1999, respectively. In 2001, they did not find a significant difference between flow-weighted nitrate concentration between their treatment and control sites. However, they did find that a soybean plot cover cropped with rye had significantly less discharge at 54mm than their control soybean field, which was not planted with rye and had 60mm of discharge (Strock et al. 2004).

Absence of nitrate loading reduction in spring 2016

One reason we see this pattern of reduced discharge in fall but not spring, even though spring had greater cover crop growth and biomass than fall, may be due to the higher proportion of the watershed planted with oats and radish growing instead of the cereal rye monoculture present in spring due to winter kill of oats and radish (Table 8). Logsdon et al. (2002), in a soil monolith study, found that both oats and rye cover crops reduced water drainage thorough the system compared to a control, but that oats led to a greater reduction in drainage than rye. The drainage in oat monoliths was less that from rye monoliths in all periods of the study, ranging from 4-85% less. This indicates that an oat cover crop can lead to a drastically higher reduction in discharge than a rye cover crop.

Another reason we may not have observed a decrease in discharge during the spring 2016 period was because of the higher overall discharge we see during that period compared to the fall 2016 period (Fig 10). The amount of water that can be removed from a system by cover cropping through transpiration has limits, it is not a flat percentage of the amount of discharge going through the system. For example, if soil is already saturated, as it likely was in our study area for much of the spring due to high precipitation, any additional water introduced through additional precipitation will flow through the system quickly through tile drains before it can be removed through transpiration. Because of this, the effect of the cover crops evapotranspiration is going to be more visible when the overall amounts of precipitation and discharge are smaller, such as in our fall period, as the amount of water removed through evapotranspiration was a greater percentage of the overall discharge. When there is a greater amount of precipitation and discharge, as in the spring period of our study, any reduction in discharge caused by evapotranspiration will be more difficult to observe, as the amount removed by evapotranspiration is a lesser percentage of the overall amount of discharge.

A third reason for this lack of a pattern may be due to the lack of data before the spring 2016 period. Without this data, it is difficult to fully understand the loading trends or the effects that cover cropping is happening. As the study continues and further data is gathered, this trend should become clearer.

Absence of nitrate concentration reduction

The reason that the flow-weighted nitrate concentration was not reduced very much by cover cropping in our study (Fig. 14) despite the nitrogen uptake of the cover crops may be due to the pool of N present as organic N in the soil. Sebilo et al. (2013), in a three-decade study in a fertilized agricultural region of France, found that 61-65% of applied fertilizer was taken up by plants while 12-15% of the fertilizer remained in the soil as organic matter. This pool of organic N accumulates from year after year of nitrogen fertilization. It provides a source of nitrate through mobilization, where the organic nitrogen transitions to ammonium, which then transitions to nitrate (Sebilo et al. 2013). This organic N pool has been found to leach nitrogen for years, even when the application of nitrogen fertilizer was stopped (Randall 2000, Sebilo et al. 2013). Randall (2000) performed a study of nitrate leaching from tile-drained agricultural fields in Minnesota that were left fallow from 1987 to 2000. In 1990, after 3 years of being left fallow, flow-weighted annual concentrations in the drainage water were 57 mg L^{-1} (Randal

2000). Flow weighted concentrations in the following years declined until 1994, where they reached a plateau of 20 mg L^{-1} (Randall 2000).

These previous studies suggest that mobilization of organic nitrogen from soil does provide a significant source of nitrogen in Midwestern agricultural fields. In our study, the amount of nitrogen uptake by cover crops was probably insignificant compared to the nitrate provided by the mobilization of this pool of organic nitrogen, as well as through fertilization. While the cover crops in our study do seem to reduce nitrate loading through reducing discharge, they may not take up enough nitrate from the organic nitrogen pool or the fertilizer to reduce nitrogen concentrations in runoff.

Plot scale studies tend to have a higher percentage of cover crop coverage than our study. Studies such as Kaspar et al. (2007), Kaspar et al (2012), and Armstrong and O'Reilly (2017), which do show a decrease in flow weighted nitrogen concentration, were performed at the plot scale. Because these plots cover a smaller area and have near 100% cover crop coverage, the cover crop nitrogen uptake relative to the nitrogen introduction through fertilization and through mobilization the organic nitrogen pool is greater than at our watershed scale, which has a much larger organic pool of nitrogen and a lesser percentage of cover crop coverage.

Improving the cover crop nitrogen uptake could lead to a decrease in flow-weighted nitrate concentration at our site. One way in which this nitrogen uptake could be increased is through increased participation in cover cropping within our watershed. Another way would be through planting a greater percentage of fields with a cereal rye/radish mixture rather than with oats/radish or annual/rye radish, due to the higher nitrogen uptake potential of cereal rye/radish. Both changes would increase cover crop nitrogen uptake, which would decrease nitrate leaching and perhaps lead to a decrease in nitrate concentration as well as discharge. However, even our limited cover crop coverage seems reducing nitrate loading through a reduction in discharge, similar to the effects found on a plot scale in a year with poor cover crop growth by Strock et al. (2004).

Challenges in loading interpolation

The error caused by our method of interpolation, explained in the methods section, should be limited in most cases. This is because the same interpolation method was used for both our reference and our treatment sites. The biggest exceptions to this may be the storm events that occurred in fall 2016, when baseflow discharge was very low and fewer samples were taken between storm events because of this low flow. During this period, flow at our reference site dried up almost completely while flow at our treatment site was slow but continuous, and thus more bottles were collected at our treatment site. This means that during these periods, this interpolation method may have led to an overestimation of loading at our reference site vs. our treatment site in the times before these storm events.

These overestimations do not apply to the discharge though, which during the fall storm events did decrease significantly at our treatment site relative to our reference site, especially during storm events. So while the magnitude of the difference in loading may be overestimated in the dry periods before storm events, the overall pattern remains: the discharge was reduced during the fall 2016 cover cropping period at our treatment site relative to our reference site, which led to a corresponding decrease in nitrate loading.

CHAPTER V: CONCLUSION

Planting cover crops on a watershed scale in a Midwestern tile drained agricultural watershed showed potential to reduce nitrate loading, even with only 54% of the watershed being planted with cover crops. This reduction in loading was primarily caused by a reduction in discharge rather than through a reduction in nitrate concentration. This pattern of nitrate loading reduction through the reduction of discharge from the system during fall 2016 was especially prevalent during storm events, where nitrate loading was almost six times greater at the reference site. There was not a reduction in flow-weighted concentration during the fall, and this may be due to the limited coverage of our cover crops. Increasing cover cropped acreage or increasing the amount of the watershed with cereal rye rather than oats or annual rye may lead to a reduction in nitrate concentration, which would lead to a further reduction in loading.

Cover cropping did not reduce TP loads in this study. Although the reduction in discharge observed in fall did seem to lead to a reduction of TP loading during storm events during that period, there was no significant change from summer to fall at our treatment site relative to our reference site. TP loading was closely related to sharp increases in discharge and concentration during storm events, so high temporal resolution monitoring of both TP concentration and discharge is essential for accurate tracking of TP loading.

This difference in nitrate loading and discharge was not observed during the spring 2016 cover cropped period, despite the total biomass and nutrient uptake by the rye being greatest in the spring. This may be due to the lower overall acreage of cover crop growth. It may also be due to the higher discharge observed in spring compared to fall. Finally, the lack of an observable pattern of reduction in nitrate loading or discharge could be because of the lack of data before spring 2016 to be used for comparison.

Our results also suggest a couple of interesting conclusions regarding the implementation of cover cropping. First of all, our study indicates that farmers will participate in large scale cover cropping if provided with incentive. We managed about 50% participation in cover cropping including multiple landowners in the first year of implementing this study. Our results also show that airplane overseeding is an effective method of cover crop planting over a large area. This is important because it is a quicker, easier method of seeding than the use of a grain drill.

This study currently only covers a limited timeframe of about 10 months from Feb. 18, 2016 to December 13th 2017. More data will be needed to better define the loading patterns in each watershed so that changes in these loading patterns because of differences in cover cropping area, crop, and cover crop biomass can more solidly defined.

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