Nutrient Removal in Constructed Wetlands Treating Agricultural Tile Drainage

by

Hyunjung Lee

Submitted to the graduate degree program in Civil, Environmental, and Architectural Engineering and the Graduate Faculty of the University of Kansas in partial fulfillment of the requirements for the degree of Master of Science.

Chairperson Edward F. Peltier, Ph.D.

C. Bryan Young, Ph.D., P.E.

Ray E. Carter Jr, Ph.D.

Date Defended: September 2, 2016

The Thesis Committee for Hyunjung Lee
certifies that this is the approved version of the following thesis:

Nutrient Removal in Constructed Wetlands Treating Agricultural Tile Drainage

____

Chairperson Edward F. Peltier, Ph.D.

Date approved: September 6, 2016

Abstract

Agricultural runoff can carry substantial loadings of nitrogen and phosphorus that can impact local surface water quality and contribute to impairment of water bodies further downstream. Subsurface tile drainage, a drainage water management practice commonly used in the Midwest, is known to contribute to elevated levels of these contaminants. Strategies to improve drainage water quality must be implemented in a way that minimally impacts land utilization and crop yield. In this study, three constructed wetlands were utilized to treat runoff from tile outlet terrace (TOT) agricultural fields managed under either a no-till corn-soybean rotation with wheat prior to soybean, or a no-till soybean crop. Nutrient and sediment removal efficiencies and runoff impact on receiving streams were determined during two growing seasons in 2014 and 2015. Water samples were collected with an auto-sampler at the wetland influent and effluent locations at Harvest Hills North (HHN/site 1), Harvest Hills Middle (HHM/site2), and Dan Cain site (Cain/site3). Using stream bottles, samples were also collected from two local streams that receive TOT runoff during and after storm events.

Over the two years, changes in nutrient and sediment loads to the wetlands were observed. Runoff quality was affected by changes in crop type, fertilizer application rate, and precipitation pattern, frequency and intensity. During the two growing seasons, TOT runoff was responsible for 99.5, 71.2 and 197.7 kg of TN entering the wetlands at sites 1, 2 and 3, respectively, of which 67.7, 59.3 and 93.8 kg exited the system in the wetland effluent (32, 17 and 53% load removal). For TP, approximately 16.54, 8.75 and 45.18 kg entered the wetlands, of which 10.24, 5.25 and 19.67 kg exited (38, 40 and 56% removal). For total suspended solids (TSS), roughly 14793, 4023 and 64624 kg entered, of which 4824, 1748 and 10876 kg exited (67, 57 and 83% removal). Compared to the year with soybean crop coverage (2014) at the sites with a no-till corn-soybean

rotation (sites 1 and 2), higher sediment concentration in TOT runoff was observed at the site with a no-till soybean crop both years (site3). The wetlands' performance was typically better with higher influent concentrations, although the wetland design and inflow volume also seemed to contribute as well. Variations in behavior between two similar wetlands (sites 1 and 2) were likely due to differences in seepage rates and flow distribution through the wetlands, which is believed to have changed as sediments built up near the influent discharge pipe at site 1.

Stream monitoring results showed that median concentrations of TN and TP were higher than the benchmark values for streams in U.S. EPA Region 7, with no measureable impact from either the treated (wetland effluent) or the untreated runoff. Potential reasons for why no significant impact to stream quality was observed are the relatively low volume of discharge relative to stream flow, and the relatively high stream levels of nutrients and sediments even upstream of the discharge location.

Acknowledgements

I would first like to express my sincere gratitude to my advisor Dr. Edward Peltier for his continuous guidance and support through my master's work. Dr. Peltier has been supportive not only academically in successfully completing tough coursework and thesis project, and emotionally with invaluable encouragements and patience, but also by providing a teaching assistantship and a research assistantship over the two years.

I would like to thank the committee members, Dr. Bryan Young and Dr. Ray Carter. I would like to thank Dr. Bryan Young for his patience and assistance by providing first insight into data analysis using R, feedback and edit suggestions of the thesis draft, and constructive comments on my work. I would also like to thank Dr. Ray Carter for providing analytical and technical assistance, and critical comments and edit suggestions of the thesis write up. I also appreciate LlynnAnn Luellen's time and patience as she assisted me with learning the laboratory tests and her work with collection of field samples.

I gratefully acknowledge the Ross E. McKinney Environmental Engineering Scholarship that allowed me to undertake my master's work.

I would also like to thank my parents for the endless support and belief, and for always being there for me even though we live almost 9000 miles away. And finally to Sanghee, who has been by my side along this journey, encouraging and supporting me to do my best.

Table of Contents

1 INTRODUCTION

1.1 BACKGROUND

According to U.S. EPA [1], agricultural runoff was responsible for almost 40 percent of the impairment in assessed rivers and lakes in the United States. Nutrients and sediment were the fifth and the seventh leading causes, along with pathogens, habitat alteration, oxygen depletion and metals. Agricultural runoff can contain significant loadings of suspended solids, nitrate, phosphorus, and agricultural chemicals, which are normally mobilized in association with precipitation events [2-4]. Studies suggest that such runoff can negatively impact local receiving surface water as well as water bodies further downstream such as the Gulf of Mexico [5,6]. This deterioration of receiving water bodies, especially those associated with high levels of nitrogen and phosphorus, are likely to contribute to an increase in algal growth, which can lead to variety of problems like oxygen depletion, turbidity and stream habitat degradation [1,5]. Effective treatment of pollutants from nonpoint sources can significantly reduce such impacts but must be achieved with minimal reduction to land utilization and crop yield [2].

1.2 TILE OUTLET TERRACE (TOT) DRAINAGE AND ITS POTENTIAL IMPACTS

The subsurface drainage pipes that make up a network system in tile outlet drainage systems were originally made of clay, not plastic as is used today [7]. French farmers are generally known to have found the modern type of tile drainage, although it may have first used far before that [7]. Today, a commonly used TOT system incorporates conventional terraces with perforated risers that drains runoff when water level is above a designated height, allowing longer residence time for removal of nutrients and sediments (Figure 1). In the United States, subsurface drainage systems are commonly used in the Midwest as a water management practice in soil with a water table near or above the soil surface due to poor drainage. Prolonged soil saturation interrupts plant growth and development, these systems improve productivity, which allows a rapid rate of return on the investment.

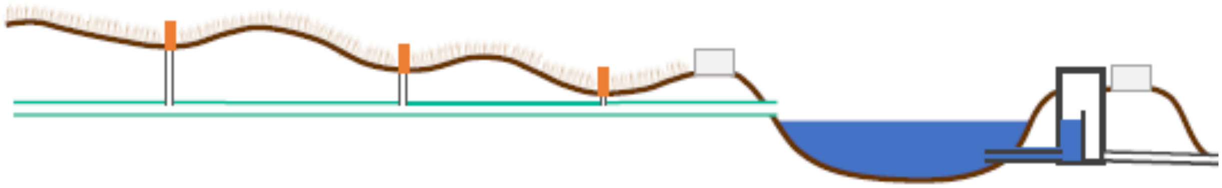


Figure 1. Tile outlet terrace (TOT) drainage system with a wetland receiving TOT discharge [8].

Tile outlet terrace (TOT) drainage systems minimize stormwater impact to land utilization by using subsurface tiles and reduce soil erosion by limiting surface flow of water. However, they shorten the residence time of agricultural runoff water containing various agrochemicals and nutrients and thus transport pollutants more rapidly to the point of discharge [9]. Even when recommended best management practices (BMPs) are followed, the use of TOT drainage systems that extensively modify the hydrology of the impacted area has the potential for large nutrient loads, especially nitrate-N [10-12]. Recent studies have shown that subsurface drainage systems may contribute to higher average soluble phosphorus than surface runoff drainage systems due to greater drainage flow volume [13-15]. The average volume of subsurface flow was observed to peak during the growing season (March to June) as a result of relatively lower transpiration from low vegetation cover and increasing precipitation [13,16].

Subsurface drainage with infiltration was responsible for drainage ratios of 13.2 to 40%, with generally higher subsurface flow volume for no-till plots as a result of higher infiltration rate and volume [17-19]. The no-till plots also may have matured in terms of drainage path in the soil profile, also allowing higher subsurface drainage flow volumes, particularly through infiltration [20,21]. The effect of crop on subsurface drainage was found to be non-significant compared to the impact from yearly and seasonal effects, partly due to changing rainfall pattern, intensity and amount between the years [17,19]. Early rainfall occurrence, for example, before crop coverage was shown to increase drainage volume even with very similar annual discharge volumes [16].

One way to indicate contamination potential, especially crucial when streams receiving tile runoff join a drinking water source, is studying flow-weighted average NO_3-N concentrations (FWANC) [17,22]. Typical NO_3-N FWANC values found ranged from 5 to 15.5 mg-N/L [17-19]. It was found that lower levels of NO_3-N in tile water can occur due to the prior year was very wet, resulting in excessive flushing from soil profile, dilution effects from high tile drainage volume, reduction in fertilizer application rate and/or coverage of plots in winter with a "trap crop" [17-19,23].

Despite the fact that subsurface drainage is subject to higher nutrient concentrations and loadings, there are few specific effective policies and plans that target this issue. For example, the Clean Water Rule, which aims to address issues related to water pollution, protects only waters that are already covered by the Clean Water Act (CWA), and fails to address problems of wastewater generated from crop land which includes tile drains. The Hypoxia Task Force Action Plan 2008 [24], a national program to reduce Gulf hypoxia, called for a significant reduction of nitrogen loading from the Mississippi River Basin to the Gulf through a combination of several proven techniques, which includes the creation and restoration of wetlands and riparian reservoirs [25,26]. Despite the efforts and investments to reduce inputs of nitrogen into the system, the most recent report[27] states that the levels of nitrogen have stayed the same or increased in 84 percent of streams in the United States.

1.3 AGRICULTURAL RUNOFF TREATMENT WETLANDS

Wetlands have demonstrated potential to address water quality problems associated with agricultural runoff and to provide an environmental buffer [3,4,28]. Wetlands can target a range of contaminants alone or in combination, such as suspended solids, nitrate, phosphorus and agrochemicals [4]. Compared to other treatment options, constructed wetlands provide passive, low maintenance systems that are capable of dealing with pulses of flow and contaminants associated with highly variable storm events [28]. Unfortunately, the effectiveness of these systems depends on many variables, including rainfall pattern, intensity and frequency, influent nutrient loading, and hydraulic retention time [29,30].

In previous studies, TN load removal in wetlands ranged between 33 to 55% and TP load removal from negative removal to 80% [28,31,32]. At all times, TN loads exiting the constructed wetlands were lower than those flowing into the wetlands [28]. The fraction of dissolved nitrogen, particularly nitrate nitrogen (NO_3-N) was found to decrease as water flows through the wetlands, and a possible explanation for this is that there was some production of organic N within the wetland [31]. The wide range of removal efficiency of wetlands can be due to the combination of complex processes and interactions. For treatment wetlands to be designed for the best performance, it is therefore important to understand the impact of local climate, farming practices and soil conditions.

1.4 KANSAS WATERS AND LANDS

In Kansas, a program called Watershed Restoration and Protection Strategy (WRAPS) addresses the issues regarding impaired water bodies affected by nonpoint sources, and aims to reduce contaminant loading from those to achieve Clean Water Act requirements [33]. The assessment of the Upper Wakarusa Watershed identified it as one of the watersheds that needs restoration, and thus a WRAPS program for this watershed area began in 2001 [33]. The Upper Wakarusa is 235,400 acres in area, of which roughly 83% is made up of grassland/rangeland (roughly 56%) and cropland (roughly 27%) [33]. Farmed land with steeper slopes is usually terraced in an effort to reduce soil erosion [33].

Serving as the primary drinking water source to most residents of Douglas County [33], Clinton Lake must maintain appropriate water quality standards. Nutrient and sediment load reductions within the watershed are believed to be sufficient to meet these requirements [33]. The Kansas Water Vision for the Kansas Water Regional Planning Area, which includes Douglas County, suggests learning the strengths and limitations of technologies and best management practices for better utilization [34]. In an effort to do this, the Kansas Water Office (KWO) and U.S. EPA, along with local landowners, invested in the construction of treatment wetlands in Douglas County as a pilot project. This paper focuses on the strength and weakness of wetlands as a strategy to treat agricultural tile drainage before discharging to receiving water bodies.

1.5 STUDY OBJECTIVES

To assess the effectiveness of constructed wetlands for treating runoff from tile outlet terrace (TOT) agricultural fields, we collected and tested influent and effluent water from more than 20 storm events over two growing seasons. To find the impact of direct TOT runoff and wetland effluent and to determine reference values, two intermittent streams located adjacent to farmlands were monitored. This paper presents the effectiveness of these wetlands on effluent water quality and downstream nitrogen and phosphorus loadings.

2 MATERIALS AND METHODS

2.1 SITE BACKGROUND

The three wetlands observed in this study were the Harvest Hills North (HHN), Harvest Hills Middle (HHM) and Dan Cain (Cain) wetlands, all located within the Upper Wakarusa River watershed in Douglas County, Kansas (Figure 3a and b). As can be seen from Figure 3, the two Harvest Hills wetlands are similar in shape and size with a length to width ratio close to 1:1, whereas the Cain wetland has a ratio of roughly 1:4. At the HHN and HHM wetland retention sites, the contributing drainage areas (CDA) are 14.8 and 17.4 acres of cropland, respectively. It should be noted that the drainage surface inlet on the second terrace at HHN is exposed without a riser, thus is vulnerable to significant erosion during heavy storm events. The crops planted were winter wheat in the last quarter of 2013 then soybeans in 2014 and corn in 2015. At the Cain wetland site, the CDA is roughly 29 acres through a tile drainage system, in addition to a small area below the last terrace that drains directly into the wetland. The crops planted were soybeans in both years.

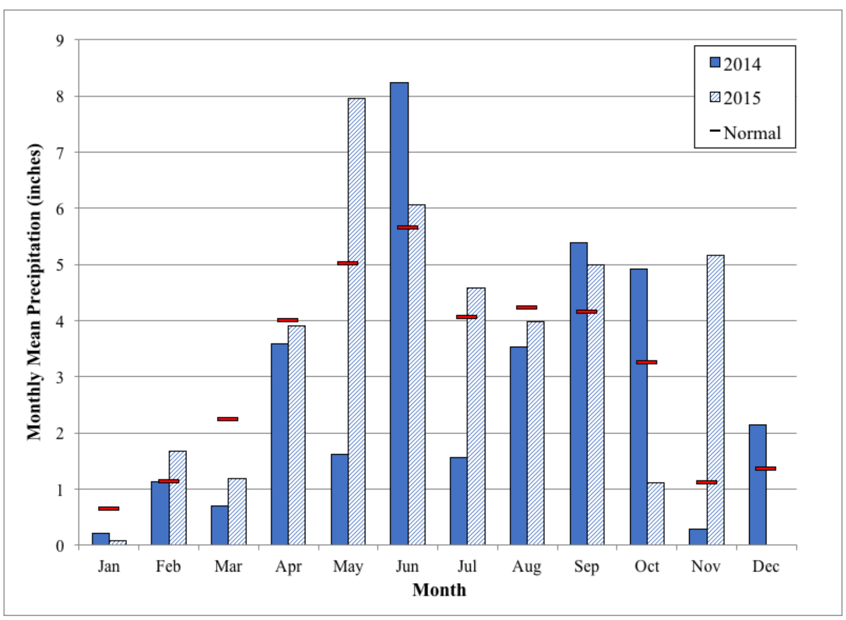


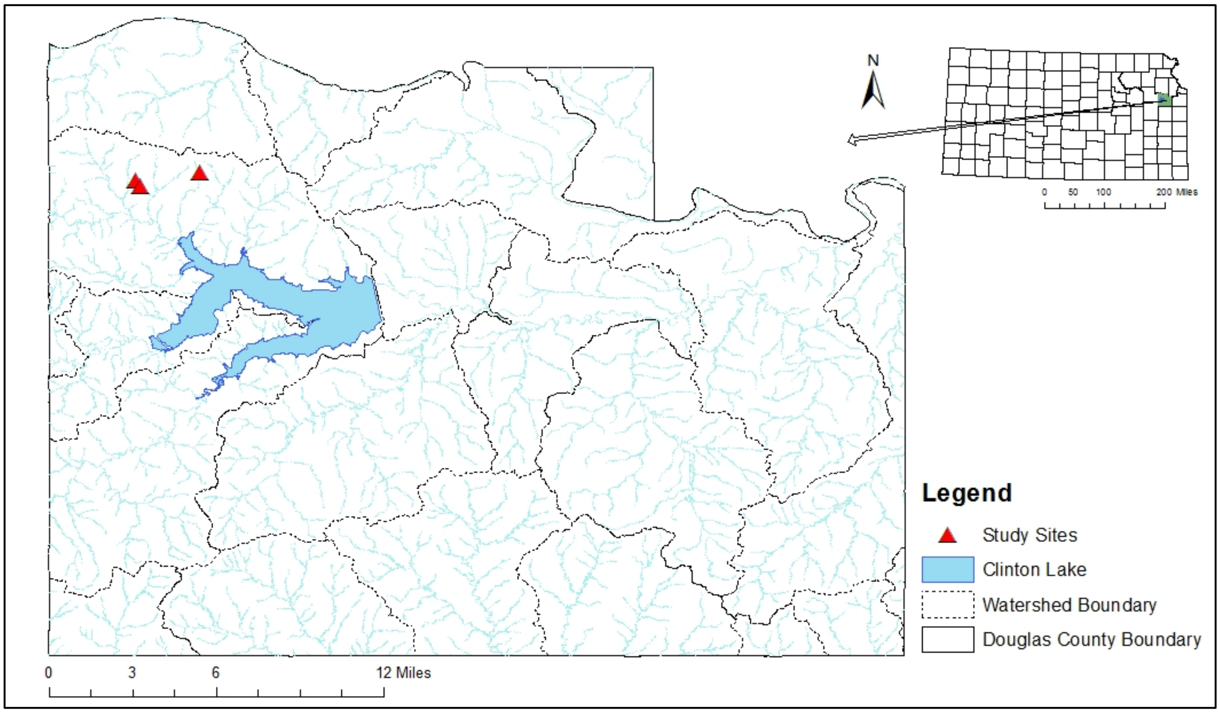
Figure 2. Comparison between normal monthly mean precipitation for Clinton Lake, KS, to 2014 and 2015 monthly mean precipitation [35].

Located in the Northeastern Kansas, the study region receives about 36 to 38 inches of average annual rainfall. Precipitation occurs particularly during the spring and early summer seasons. Figure 2 shows normal monthly mean precipitation and 2014 and 2015 monthly mean precipitation. Most precipitation events occurred from early Spring to June, with highest mean precipitation in June 2014 and in May 2015.

2.2 COLLECTION, PRESERVATION AND STORAGE OF WATER SAMPLES

The sampling setup consisted of an autosampler, area-velocity meter and rain gage at each sampling location. The autosamplers to sample inflow were installed at the outlet of the TOT system at the Harvest Hills sites and at the standpipe on third terrace at the Cain site. Outflow autosamplers sampled from the pond at the weir overflow box at the Harvest Hills sites and from the pipe outlet pond at the Cain site. During each rain event, flow is registered by the sensor, which turns on the autosamplers. The area-velocity meters (ISCO Model 750 Area Velocity Flow Module) were set up in the tile drain over the effluent weir at Harvest Hills sites and in the effluent pipe at the Cain site. In addition, an ISCO Model 674 Tipping Bucket Rain Gauge installed near each inflow sampling location measured rainfall amounts. The system was powered with an ISCO 12 VDC Battery with Solar Panel Charger (Figure 4).

Samples were collected at the wetland influent and effluent throughout the growing seasons in 2014 and 2015 (June-October in 2014, n = 5-17 per site; April-November in 2015, n= 9-12 per site). Due to discharges that were mostly event-driven, auto samplers (ISCO Model 6712 Full-Size Samplers) were programmed to collect 200 mL into a bottle (ISCO Single-Bottle 2.5 Gallon (9.46 L) Polyethylene Round Bottle) for each specified trigger volume along with grab samples, occasionally, at each monitoring location for entire runoff events. The details in calculating the magnitude of the specified trigger volume can be found in the standard operating



(a)

(b)

Figure 3. Location of (a) the study sites within the Douglas County, KS, and (b) close-up imagery of the Harvest Hills wetland sites and Dan Cain wetland sites.

procedures for the monitoring project [8]. Time sequence samples were collected for a few storm events but only data from composited samples were used for this paper.

Figure 4. Typical wetland auto-sampling equipment set up [8].

Stream samples were collected at the Cain site (site 3) and the Haase site to investigate the impacts of treated and untreated TOT runoff on surface water quality, and to provide reference values for sediment and nutrient values in TOT runoff. The Haase site used similar farming practices as the two Harvest Hills sites but had no wetland, which means it discharged untreated TOT runoff directly into an adjacent stream. TOT discharge samples were collected at both locations. At Haase, an outfall sample was collected immediately downstream of the TOT discharge pipe from this cropland. At the Cain site, it was collected just downstream of a submerged bubble-up pipe discharging direct TOT runoff from a portion of that site that does not drain to the wetland. Stream samples were also collected upstream and downstream of the TOT outfall (Haase site) and the wetland effluent discharge location (Cain site).

Collected water samples were sub sampled and preserved according to the specific analytical methods. The sub samples were kept in a cooler packed with ice to achieve a water temperature at or below 10 °C when brought back to the laboratory. For total phosphorus (TP), total nitrogen (TN), total dissolved phosphorus (TDP), total dissolved nitrogen (TDN) and total suspended solids (TSS), more than 300 mL, typically around 500 mL, were sub sampled in the laboratory into a plastic container. For whole sample and dissolved sample analyses, maximum holding times were 28 days and 2 days, respectively, at 4 °C. Dissolved water samples were obtained by filtering whole water samples through a microfiber filter (Fisher Scientific # 09-874-35 or equivalent) after removing any large chunks of plant material.

2.3 ANALYSIS OF WATER SAMPLES

Aliquots of each water sample were measured for TSS, TP, TDP, TN and TDN. If not stated otherwise, QA/QC procedures were adapted from Standard Method 1020, and sample collection and preservation guidelines followed Standard Method 1060 [36]. All water samples were analyzed following published SOPs approved for this project by USEPA, which can be found in the final project report [8]. A brief summary of each method is provided below.

2.3.1 TOTAL SUSPENDED SOLIDS (TSS) ANALYSIS

Each whole water sample of 200 mL (or less, if the sample volume was not sufficient) was analyzed for TSS following Standard Method 2540 D [36]. A pre-measured 47 mm diameter binder-free glass microfiber filter was used to vacuum filter water samples (Fisher Scientific # 09-874-35 or equivalent). The filter was then dried in an oven at 103 to 105 °C for at least one hour, cooled in a desiccator and weighed. The TSS was calculated as the following:

$$mg\ total\ suspended\ solids/L = \frac{(A - B)\times 1000}{sample\ volume, mL}$$

where, A = weight of filter + aluminum dish + dried residue, mg, and

B = weight of filter + aluminum dish, mg.

2.3.2 TOTAL PHOSPHORUS (TP) AND TOTAL DISSOLVED PHOSPHORUS (TDP) ANALYSIS

Standard Method 4500-P (E) was used in the analysis of water samples for TP and TDP [36]. In this method, persulfate digestion is used to convert all phosphorus in the sample to ortho-phosphate. Phosphate concentrations are then determined by colorimetric analysis. For each sample 30 mL of either whole water sample (for TP) or filtered water (for TDP) was mixed with 7.5 mL of a 40 g/L potassium persulfate solution in acid-cleaned 55 mL Pyrex tubes. The samples were then autoclaved for 45 minutes at 121 °C and 15 psi.

Digested samples were allowed to be stored at 4 °C for no more than one week before analysis. Immediately before analysis, 3.75 mL of the final mixed reagent was added in 1 mL increments at one-minute intervals, mixing tubes by inversion after each addition, for color development. The final reagent, stable was prepared from mixing the phosphorus premix reagent, which is a mixture of potassium antiomonyl tartrate, antimonyl tartrate, concentrated sulfuric acid, deionized (D.I.) water, and ascorbic acid. The tubes were then let to stand at room temperature for 30-35 minutes to allow for full color development. A Shimadzu 1650-PC UV/Visible light spectrophotometer was used at 885 nm to measure the concentration of ortho-phosphate in water samples in a 10-cm path length plastic cuvette with reagent water in the second cuvette.

2.3.3 TOTAL NITROGEN (TN) AND TOTAL DISSOLVED NITROGEN (TDN) ANALYSIS

Standard Methods 4500-NO_3^- (b). was used in the analysis of water samples for TN and TDN [36]. In this method, alkaline-persulfate digestion is used to convert all inorganic and organic nitrogenous compounds to nitrate. Nitrogen concentrations are then determined by spectrophotometric analysis with a Shimadzu 1650-PC UV/Visible light spectrophotometer at two wavelengths, 220 nm and 275 nm. For each sample, 30 mL of either whole water sample (for TN) or filtered water (for TDN) was mixed with 7.5 mL of a 20 g/L potassium persulfate solution and 0.75 mL of a 6 N sodium hydroxide solution in acid-cleaned 55 mL Pyrex tubes. The samples were then autoclaved for 45 minutes at 121 °C and 15 psi. Before spectrophotometer analysis, the autoclaved samples were acidified with 0.75 ml of 7N hydrochloric acid.

2.3.4 STATISTICAL ANALYSIS

Due to the presence of particularly high or low numbers, median values were used to represent central tendencies of the data rather than the mean. Statistical analyses were performed using R version 3.3.1 for Macintosh. To avoid the assumption of normally distributed data, non-parametric tests were used for statistical analyses. The two-sample Wilcoxon test, also known as the Mann-Whitney test, with a significance level of 95% ($p = 0.05$) was used to check for significant differences between median values in the influent and effluent water samples from the wetlands, and in the upstream and downstream, and upstream and outfall water samples from the stream samples. A one-sample Wilcoxon test with a significance level of 95% ($p = 0.05$) was used to estimate whether the median relative differences between paired upstream and downstream samples were significantly different from zero. Due to the presence of ties in most of datasets, which are not allowed by the R statistical package, adjustments were made to ties to process them for nonparametric statistical analysis. Tied values were adjusted by adding/subtracting a tenth of the significant digit (i.e. 0.13, 0.13 and 0.13 mg-N/L were adjusted to 0.131, 0.13 and 0.132 mg-N/L) without changing the overall distribution. Some samples demonstrated results that were below detection limits, especially for dissolved nitrogen and phosphorus in 2014, and those were assigned values that were half of the lowest calibration standard for nitrogen and phosphorus analysis, after a thorough review of all standard curves. For TSS, a value of 5 mg/L was assigned to those samples with below detection limit results.

3 RESULTS AND DISCUSSION

3.1 NITROGEN IN TOT RUNOFF

During the two-year study period, each site's tile outlet terrace was responsible for approximately 99.5, 71.2 and 197.7 kg of TN entering the wetlands for sites 1, 2 and 3, respectively, of which 67.7, 59.3, and 93.8 kg exited the system in the wetland effluent (Table 1). Overall, the majority of the load that drained into the wetlands occurred in 2015, rather than in 2014, due to delayed sampling and consequently fewer number of samples obtained in the first year of the study.

Concentrations of TN in TOT runoff, or wetland inflow, from 2014 and 2015 are shown in boxplots (Figure 5). The boxes indicate the middle 50 percent of the data, the line in the box marks the median, error bars show full data range, and the asterisks points out significant differences between 2014 and 2015 data at each site (Figure 5). For example, the median concentrations of TN at site 1 were 3.1 mg-N/L and 7.6 mg-N/L in 2014 and 2015, respectively, and the difference between the two values was significant ($p = 0.01$) (Figure 5). Likewise, significant increases in incoming TN were observed at site 2 and site 3, from 1.8 to 9.1 mg-N/L ($p < 0.001$) and 3.1 to 4.2 mg-N/L ($p = 0.046$), respectively. This significant increase in TN can be traced to application of fertilizer prior to planting in 2015 and a consequent increase in total dissolved nitrogen, which will be discussed further in the following paragraph.

In 2015, total dissolved nitrogen (TDN) was analyzed along with TN whereas, in 2014, only dissolved nitrite and dissolved nitrate were individually measured. Dissolved nitrite and nitrate concentrations were no longer measured in 2015 because they were found to be below detection limit in most samples in 2014. The median dissolved nitrogen percentage in the influent was 71% at site 1, 76% at site 2 and 25% at site 3, found by dividing TDN by TN.

Dissolved fraction values were not calculated for dissolved data from 2014. Higher fractions of influent dissolved nitrogen at the two Harvest Hills sites, compared to that at Cain site, are believed to be from the application of higher levels of nitrogen prior to planting corn than soybeans (Figure 6). By contrast, no nitrate fertilizer was applied to the Cain site in either year. This suggests that the major dissolved nitrogen source was fertilizer and that there was background dissolved nitrogen in the runoff possibly from erosion and soil mineralization [37].

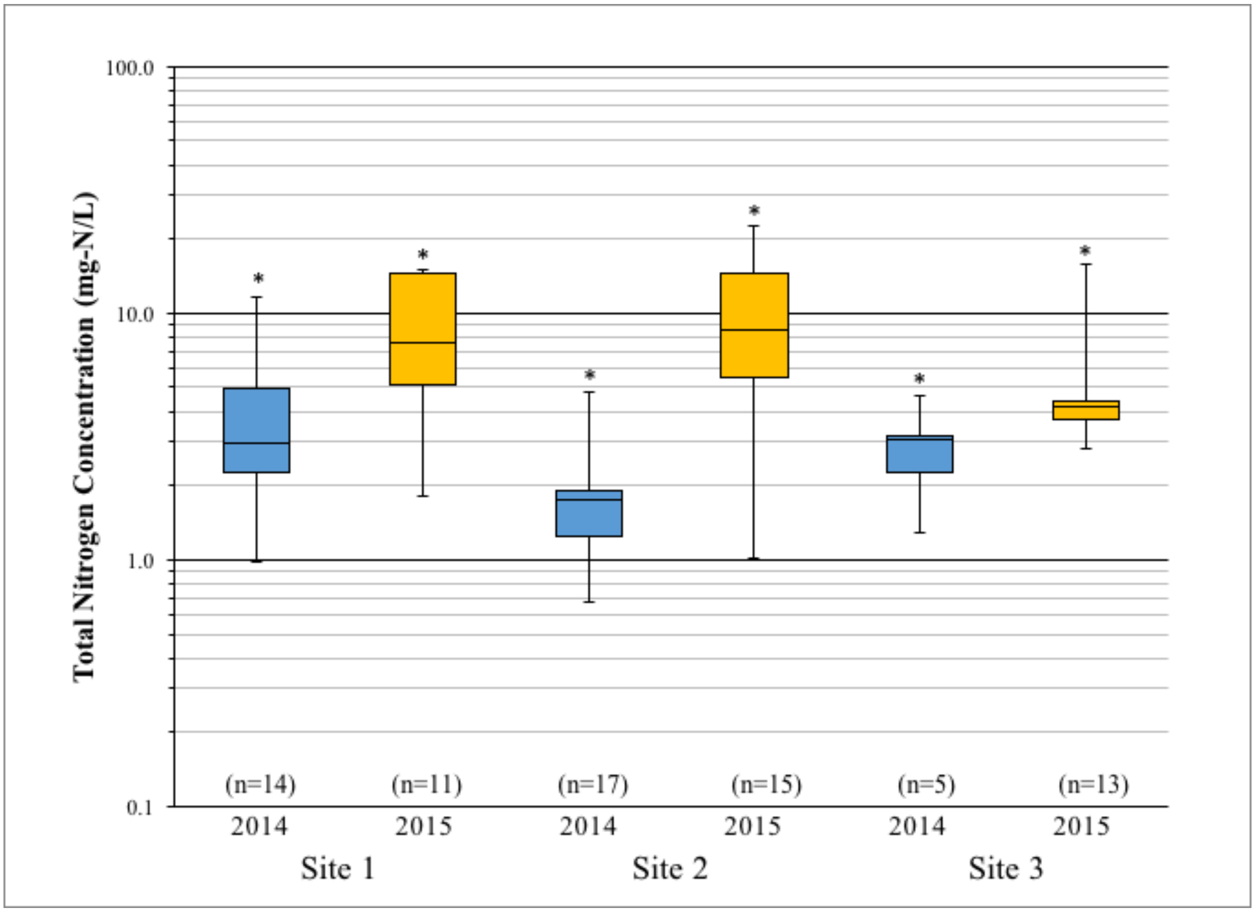


Figure 5. Influent TN concentrations in 2014 and 2015.

Inflow TN pattern. Pollutant loadings from croplands are event driven [28] and depends on the frequency and intensity of intermittent storm events, runoff volume, peak discharge, and pollutant mobilization can be controlled [28,38]. Similarly, our wetlands experienced higher runoff volume and pollutant loading as a response to more frequent, more high-intensity

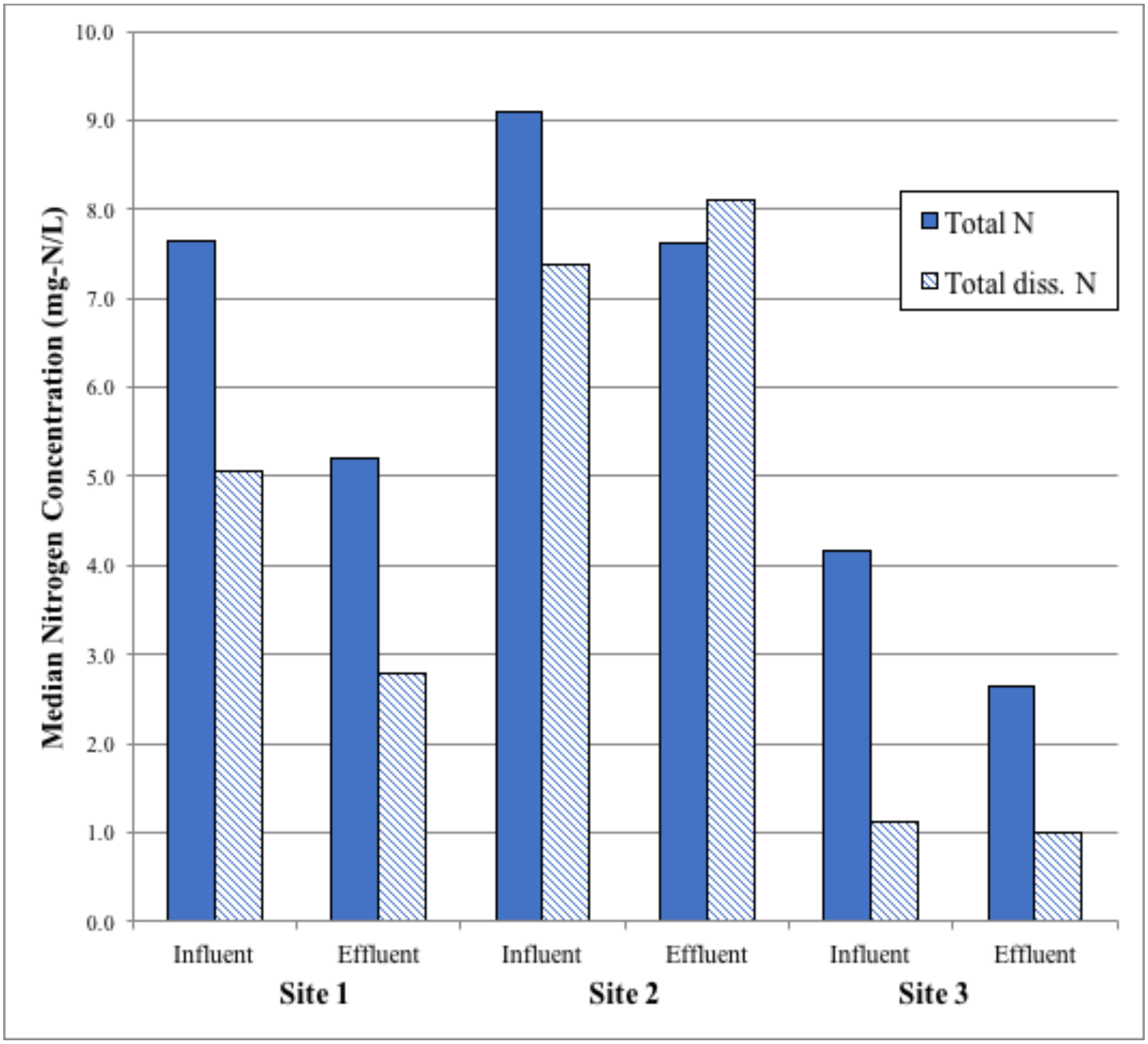


Figure 6. Median total and dissolved nitrogen concentrations at the three study sites in 2015.

storms in the spring season. Figure 7 shows the relationship between rainfall volume and inflow TN concentration at the Harvest Hills sites (sites 1 and 2) from May to September in 2015. As expected, high-intensity storm events were observed more frequently in May, followed by less intense and less frequent storm events. The inflow during the first few moderate-intensity storm events carried the highest TN concentration, but much more diluted concentrations were found during the two heaviest rainfall events in May due to high runoff volume. The overall TN concentration decrease from spring to fall (Figure 7) because of the N-based fertilizer application timing, which was most likely in spring of 2015.

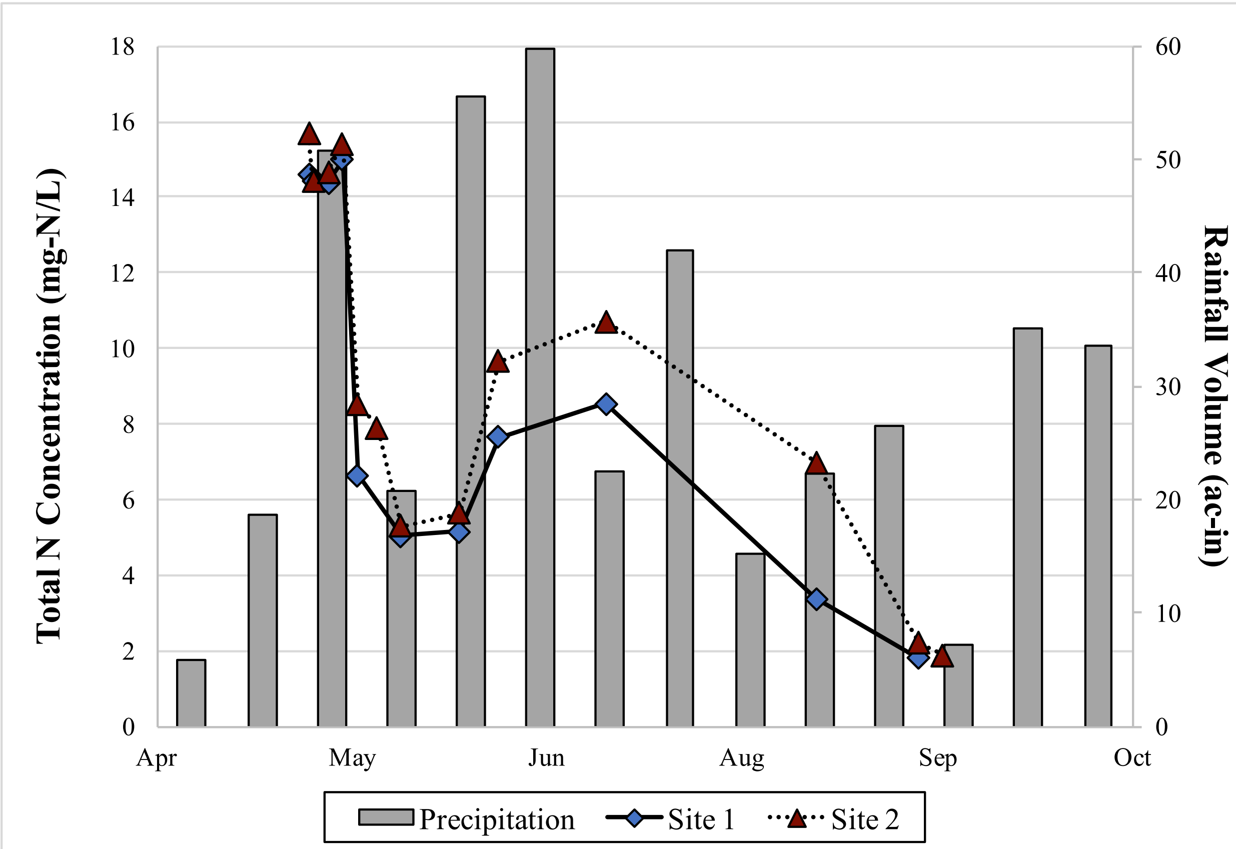


Figure 7. Relationship between rainfall volume and inflow TN concentration at Harvest Hills sites in 2015.

3.2 PHOSPHORUS AND TOTAL SUSPENDED SOLIDS IN TOT RUNOFF

Similar to inflow TN concentrations, inflow concentrations of TP into each wetland showed significant increases from 2014 to 2015 (Figure 8). In 2014, the median inflow TP concentrations were 0.37, 0.20 and 0.34 mg-P/L at site 1, site 2 and site 3, respectively. In 2015, all median inflow TP concentrations were elevated at 0.80, 0.86 and 1.07 mg-P/L. The increase in phosphorus from 2014 to 2015 at sites 1 and 2 was potentially due to higher fertilizer application rates in Fall 2014 (assumed to be approximately 58 lb/ac based on similar application as the Haase site) compared to 2013 (35 lb/ac) at sites 1 and 2, and late crop planting at site 3. In

2015, delayed crop planting, coupled with high-intensity storm events in May, likely resulted in higher sediment erosion. Phosphorus (P) in agricultural runoff is known to correlate to sediment in runoff since phosphorus found in runoff is mostly particulate rather than dissolved P [39-42]. Thus, the elevated TP concentration in 2015 was likely from the increased sediment erosion due to late crop planting, along with higher fertilizer application rates. The patterns of median influent TP and TSS concentrations were similar in 2014 and 2015 (Figure 8 and Figure 9). The only exception was that no significant difference was observed between the median influent TSS concentrations in 2014 and 2015 at site 1 (Figure 9).

Generally, site 2 received runoff with lower TSS concentration than site 1 and site 3 did. This difference contradicts the expectation that sites 1 and 2 will receive similar quality inflow, given that the two sites were almost identical in wetland design, contributing drainage area (CDA) and crop cover. One possible reason for the difference can be traced to an exposed riser on the second terrace at site 1, which caused more sediments, and thus more particulate nitrogen and phosphorus in runoff. For this reason, total soil loss from a plot covered with soybeans both years (site 3) and that from a plot covered with winter wheat then soybeans in 2014 and corn in 2015 (site 1 and 2) should be compared using data from sites 2 and 3 only. In both years, site 3 with soybean coverage and residue yielded higher soil loss, which lead to an observation of higher inflow TSS concentration into the wetland (Figure 9). This occurrence, in which sediment concentration is greater, sometimes even statistically significant, from a system with soybean residue than that with corn residue, was observed in other studies, and was explained by the extent of surface coverage of residue, in conjunction with tillage system and other factors including erodibility factor, and slope [43,44].

As stated above, TP concentration in runoff are known to correlate to sediment

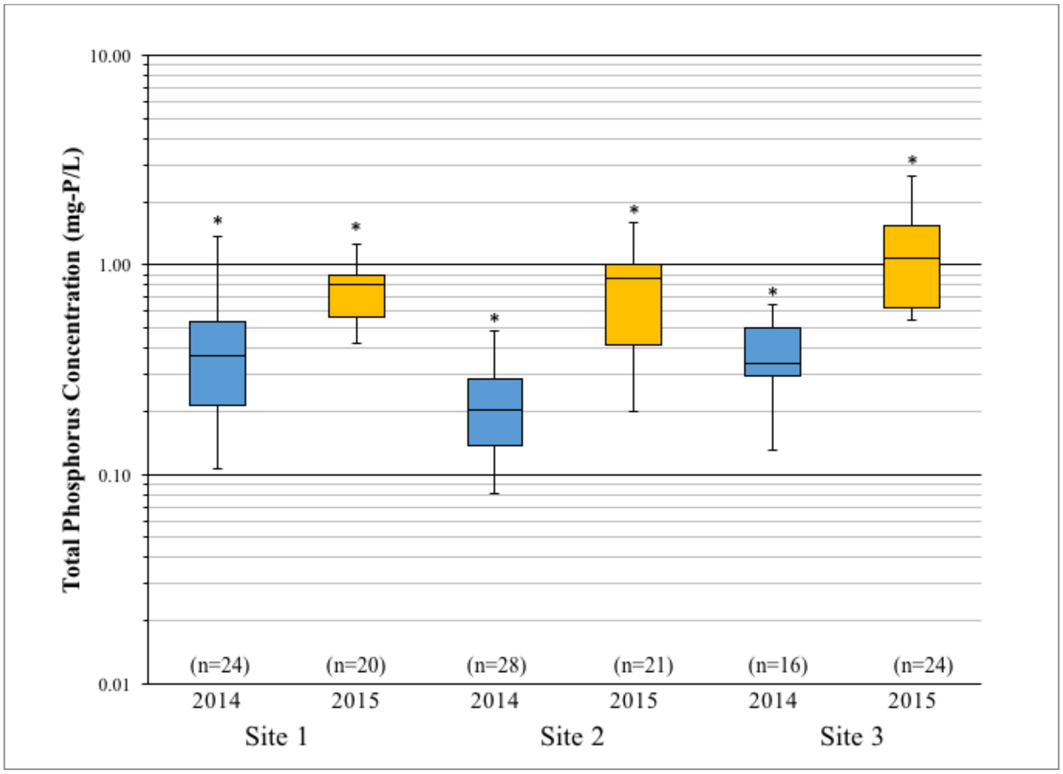


Figure 8. Influent TP concentrations in 2014 and 2015

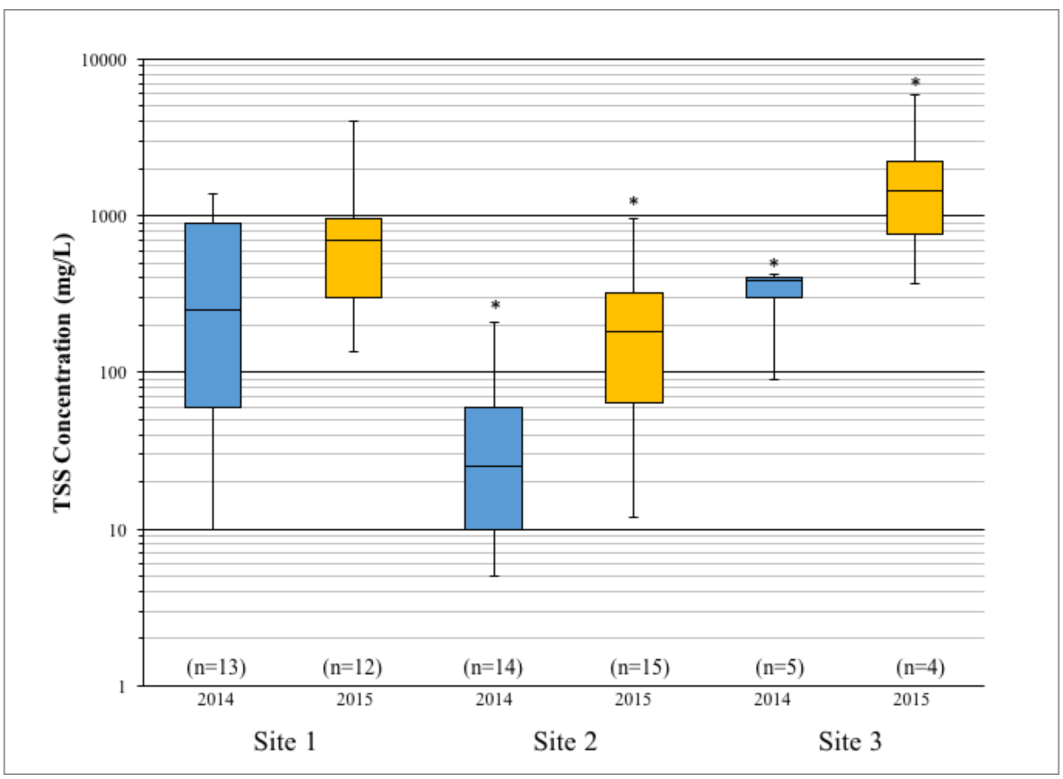


Figure 9. Influent TSS concentrations in 2014 and 2015

concentration in runoff. In this study, inflow TP significantly correlated with TSS ($n = 64$, $R = 0.75$, $p < 0.001$), when any data points with TSS concentrations of 10 mg/L or below were excluded. In 2015, dissolved P concentrations were less than 0.30 mg-P/L at all three study sites' wetland influent whereas total P concentrations were equal to or greater than 0.80 mg-P/L (Figure 10). Similar to median TDN in influent, median influent TDP was the highest at site 2, followed by site 1 and site 3.

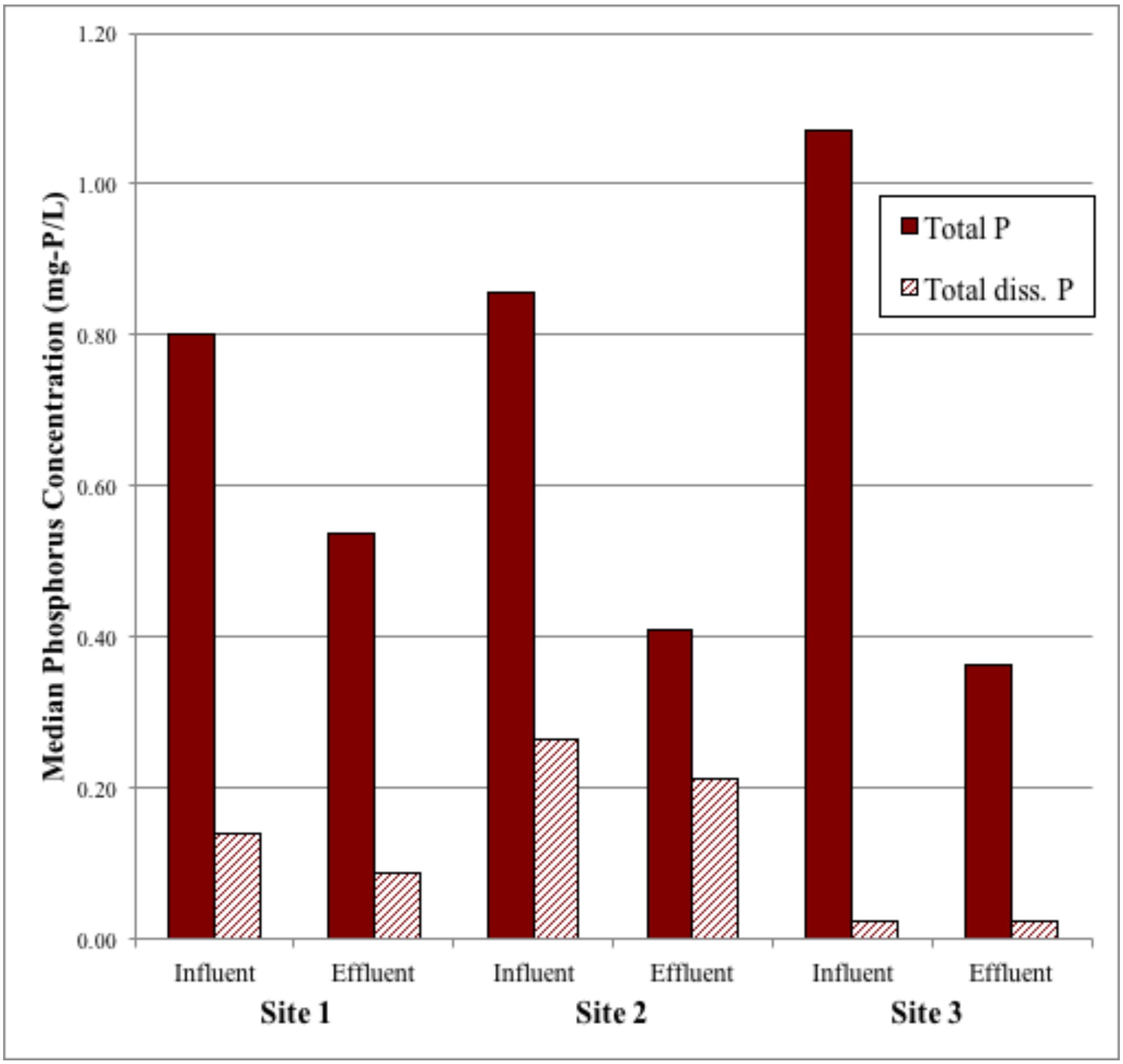


Figure 10. Median total and dissolved phosphorus concentrations at the three study sites in 2015.

3.3 PERFORMANCE OF WETLANDS IN CONSTITUENT REMOVAL

Overall, contaminant removal in the three wetlands varied greatly from 2014 to 2015 (Table 1). Median inflow and outflow values of TN and TP were calculated using all measured values, whereas the concentration removal efficiency (CRE) values for each wetland were calculated using differences between paired inflow and outflow concentrations from individual storm events. The load removal efficiency (LRE) and areal removal rate were calculated differently, using the following equation:

$$LRE = \frac{\sum Load_{IN} - \sum Load_{OUT}}{\sum Load_{IN}} \times 100\%$$

The summation of loads in and out of the wetlands included all values with available flow data (flow volume) and concentration data over the indicated sampling period. Then, the difference between the two summation values was used to calculate the LRE. This approach incorporates storm events for which there was runoff to the wetlands, but no effluent flow from them.

CRE was generally higher in 2015 than that in 2014, with the exception of TN at site 1 and TSS at site 3. At site 1, the CRE value of TN was potentially affected by the data collected from 5/7/15 to 5/14/15, during which no flow data was available for two of the four collections. These missing flow data could imply that the outflow from site 1 at the time period were not resulting from the inflow, but rather from standing water that may be higher in nutrient concentration. At site 3, the number of samples collected during 2014 was very low (n = 3), which inhibits statistical analysis, so comparing the median CRE from those to that from 2015 (n = 10) could have skewed the result. Other than those, the increase in CRE from 2014 to 2015 corresponded to the increase in inflow concentrations of TN, TP and TSS at all three study sites. In addition, the overall increase was not unexpected due to potentially higher plant coverage, which slows the flow and increases the retention time [45,46], and higher organic matter, which increases the denitrification process [3].

At site 1, the median TN CRE values were 8% in 2014, 28% in 2015, and 19% over both years (Table 1). Individual wetlands varied in performance in terms of CRE and site 2, especially, performed poorly compared to the other sites (Table 1). Possible reasons for this poor performance at site 2 are discussed below. TN load inputs ranged from 5.8 to 21.7 kg N in the 2014 growing season (June – October), and 68.4 to 182.0 kg N in the 2015 growing season (May – November) (Table 1). The median TN outputs, which ranged from 1.8 to 8.8 kg N in 2014 and from 57.5 to 85.1 kg N in 2015, were lower than median inputs at all sites and in both years. During the 2-year period, site 3 received the greatest TN load and exhibited the highest load removal efficiency (LRE) of 53% as well as the highest CRE of 38%. In a previous study, three other constructed wetlands demonstrated TN removal of 37% of the overall 4639 kg N during the three-year study period [28], compared to 27% of an overall 369 kg N during the two-year study period here.

In this study, total nitrogen removal, as well as total phosphorus and total suspended solids removal, were calculated considering outlet surface flow as the only outflow from the wetland. It was observed in previous studies that the extent of highly mobile total nitrogen (NO_3-N) removal via seepage flow may account for up to 33%, depending on flow volume, wetland capacity and soil type and condition [28]. A high inflow volume that exceeds the wetland capacity during pulse flow events may result in rapid flow and thus much less seepage [28]. Nevertheless, consideration of combination of seepage and outlet flow in the NO_3-N budget was not evaluated in this study, although it will likely change the overall values of TN removal if seepage is significant as observed elsewhere [47].

Unlike with TN, all three wetlands have been shown to function as sources of TP, usually when the inflow concentration was equal to or below 0.17 mg-P/L. These constructed wetlands performed very much in agreement with an the irreducible TP concentration of 0.2 mg-P/L, as suggested by Schueler [48] . The two largest negative removals occurred at site 1 in 2014 only, of which the -118% removal took place the day after the first heavy rainfall in June of 2014. In 2015, negative removal was seen only once at site 2 in September. Similar to the correlation found between inflow TP and TSS concentrations, the removal of TP and TSS in the wetlands were significantly correlated ($R = 0.95$, $p < 0.0001$).

Table 1. Summary of inflow and outflow concentrations, total loading, concentration removal efficiency (CRE), load removal efficiency and areal removal rate of TN and TP.

Site	Year	Total N		Total N	Concentration	Load removal	Areal N
		Inflow	Outflow	loading	removal efficiency	efficiency	removal rate
		------- mg-N/L -------		kg	%	%	kg ha^{-1} yr^{-1}
1	2014	3.1	1.3	21.7	37%	72%	2.1
	2015	7.6	5.2	83.8	21%	26%	3.0
2	2014	1.8	1.6	5.8	-14%	69%	0.5
	2015	9.1	7.6	68.4	3%	16%	0.4
3	2014	3.1	1.7	15.8	24%	44%	0.8
	2015	4.2	2.6	182.0	46%	53%	13.1
Site	Year	Total P		Total P	Concentration	Load removal	Areal N
		Inflow	Outflow	loading	removal efficiency	efficiency	removal rate
		------- mg-P/L -------		kg	%	%	kg ha^{-1} yr^{-1}
1	2014	0.37	0.24	2.22	27%	62%	0.19
	2015	0.80	0.54	14.3	39%	34%	0.66
2	2014	0.20	0.17	0.42	7%	60%	0.03
	2015	0.86	0.41	8.34	38%	39%	0.44
3	2014	0.34	0.19	1.90	62%	60%	0.10
	2015	1.07	0.36	43.3	72%	56%	3.30

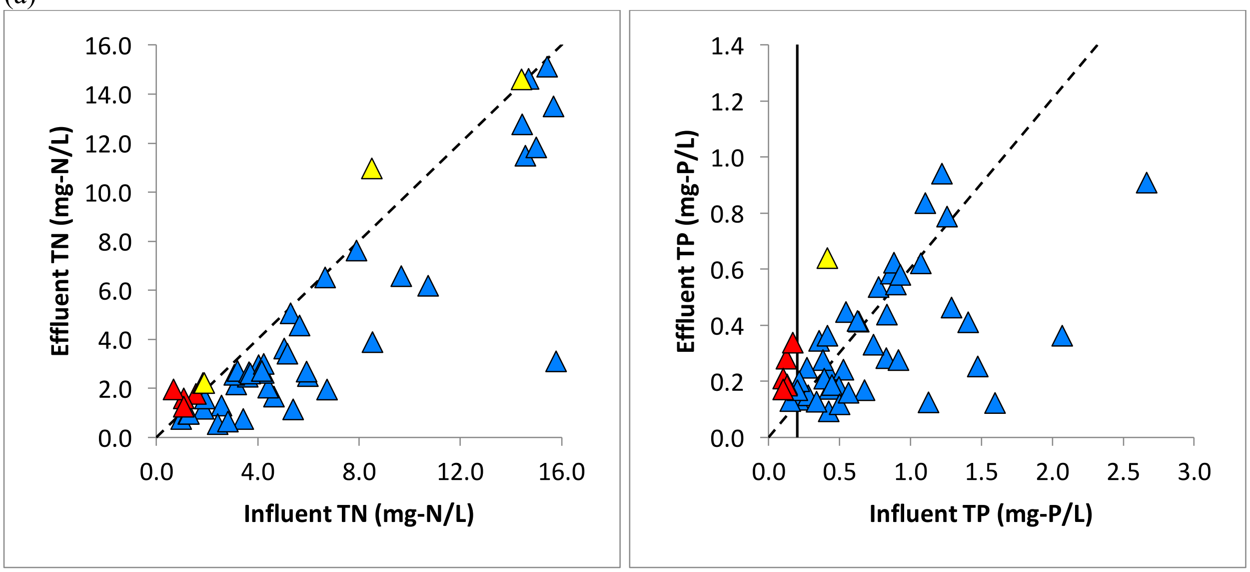
NOTE: Not all parameters were calculated using the same datasets; details on what data were used for each are summarized in the beginning of Section 3.3.

Negative removals (or source). Negative removals of constituents of interest were sometimes observed, which are shown as red and yellow triangles in figures 11 a-d. For TN, negative removals were observed after eight separate rainfall events at site 2 only, of which five occurred in 2014 (red) and the rest in 2015 (yellow) (Figure 11c). The irreducible concentration concept assumes that a given best management practice (BMP) cannot reliably remove pollutants below a given influent concentration [48]. Six out of the eight negative removal events were associated with incoming TN concentrations less than or equal to an irreducible concentration of 1.9 mg-N/L as suggested by Schueler [48], with influent concentrations ranging from 0.7 to 1.9 mg-N/L. In 2014, the largest negative removal of -188% was observed the day after the heaviest storm event (September 1st) of the recorded rainfall events between June and October. The TN concentration of 0.7 mg-N/L in the influent elevated to 1.9 mg-N/L in the effluent. In 2015, the largest negative removal of -28% was observed after the first heavy storm event in May (Figure 7). The inflow TN concentration of 8.5 mg-N/L rose to 11.0 mg-N/L at the effluent. Compared to site 1, which has a similar design, site 2 demonstrated lower contaminant removal, possibly due to poor flow patterns leading to unutilized mixing zones. One possible cause for the better performance of the wetland at site 1 is the buildup of sediments just downstream of the influent pipe. The “island” that built up here due to the sediment accumulation may have allowed better mixing zones and inhibited short circuiting of flow through the wetland.

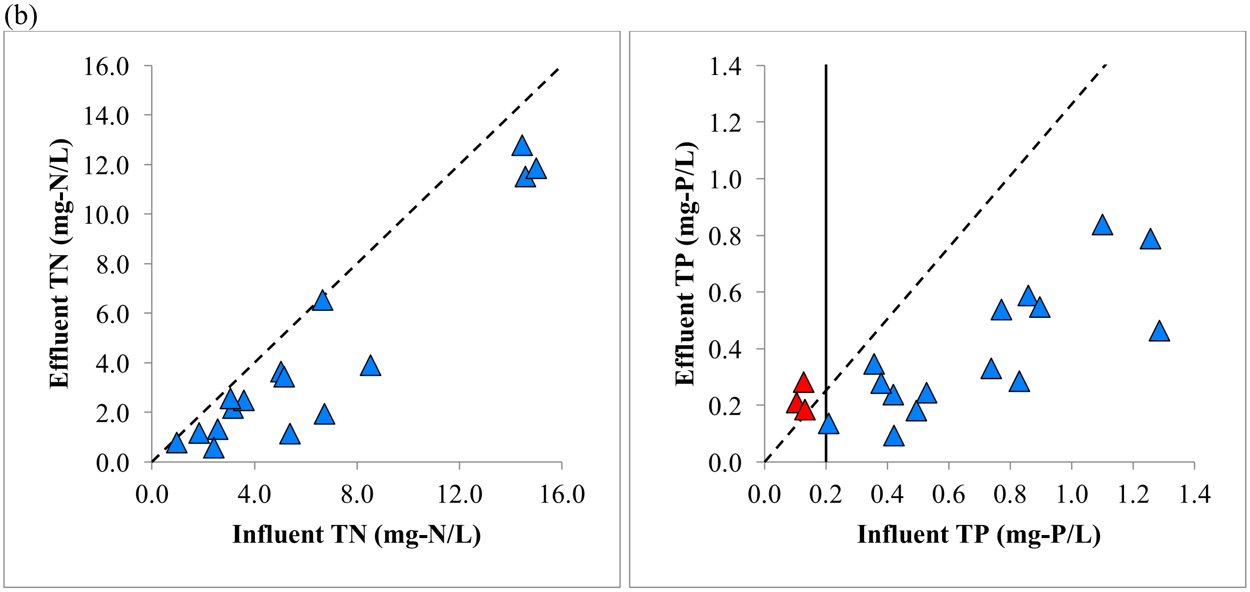
According to Figure 12a, roughly 20% of paired samples indicated negative removal for TN and 40% for TDN. Most of those were associated with inflow TN concentrations of 2.0 mg-N/L or below. As mentioned previously, all eight negative TN removals occurred at site 2, indicating that that specific site was more vulnerable to nitrogen efflux due to poor design. Out of the ten negative TDN removals obtained, 60% occurred at site 3, 30% at site 2 and 10% at site 1. One possible reason as to why 60% of the negative TDN removals were observed at site is that inflow TDN concentrations at this site that ranged from 0.5 to 1.1 mg-N/L, which was significantly lower than those observed at the two other sites that received nitrogen fertilizer application.

Approximately 15% of paired samples demonstrated negative removal for TP and just under 40% for TDP (Figure 12b). Except for the one event with an influent TP concentration of 0.42 mg-P/L (Figure 11c), all storm events with negative removal were associated with influent

(a)



(b)



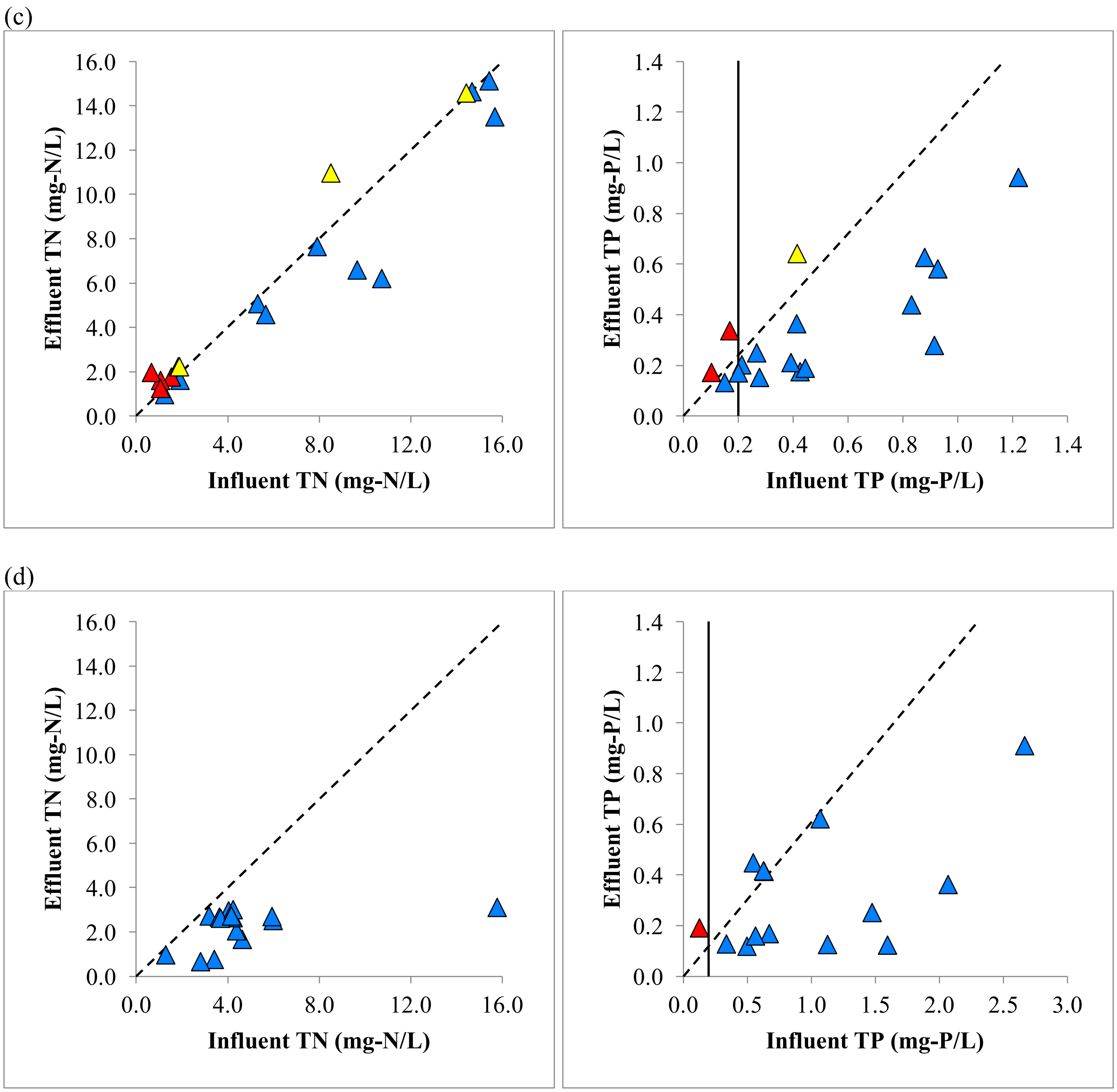
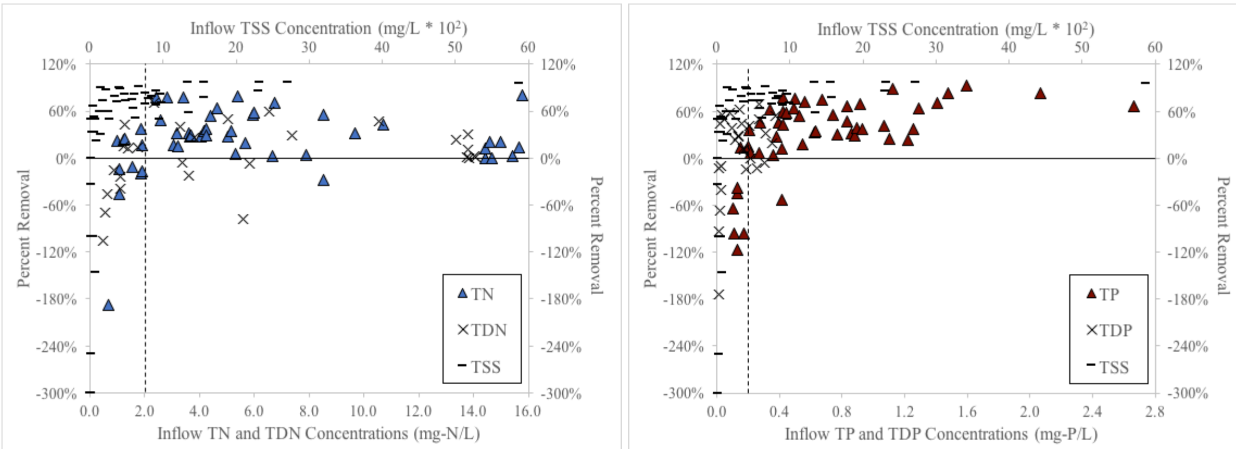


Figure 11. Paired influent and Effluent concentrations of TN (mg-N/L) and TP (mg-P/L) at (a) all sites, combined, (b) site 1, (c) site 2, and (d) site 3.

concentrations ranging from 0.10 to 0.17 mg-P/L, which are slightly less than the suggested irreducible concentration of 0.20 mg-P/L. For TDP, site 3 had six out of nine negative removals, whereas site 2 had three out of ten and site 1 none. All of the incoming TDP concentrations at site 3 were 0.05 mg-P/L or below, which is significantly lower than those at sites 1 and 2, and this could have been the major reason as to why this specific site seemed to have demonstrated poor removal. At site 2, the TDP concentrations in the influent samples were relatively high but all three paired samples that were collected between late May and mid-June demonstrated negative TDP removals.



(a) (b)

Figure 12. Range of percent removal and associated inflow concentrations of (a) TN, TDN and TSS, and (b) TP, TDP and TSS. Note that the unit of inflow TSS concentrations is mg/L * 10^2.

3.4 STREAM MONITORING RESULTS

The range of TN, TP and TSS concentrations in the Haase site and Cain site stream samples are shown in Figure 13. At the Haase outfall discharge point, the median concentrations of TN and TP were 6.5 mg-N/L and 0.93 mg-P/L, respectively (Figure 13 a-b). At the Cain outfall, the concentrations of TN and TP were more than 50% lower, with 3.0 mg-N/L (n = 24) and 0.48 mg-P/L (n = 22), respectively (Figure 13 a-b). Although the nutrient concentrations were lower at the Cain outfall, TSS concentrations were significantly higher at Cain with 315 mg/L compared to Haase with 248 mg/L. This discrepancy may be due to already significantly higher TSS concentrations ($p = 0.02$) at the Cain upstream location compared to the Haase upstream sampling point. While nutrient and TSS concentrations had a wide range at both sites, the median downstream concentrations at the downstream of Cain site were lower than those at the upstream. At Haase, the median concentrations of TP and TSS were equal in the downstream and upstream samples. Median TN concentrations downstream (3.4 mg-N/L) were only slightly higher than that those upstream (3.3 mg-N/L), even though the median outfall TN concentration was 97% higher.

Stream water concentrations of TN, TP and TSS differed substantially between 2014 and 2015 (Figure 14), with upstream concentrations higher by 71, 44 and 131% in the second year. This increase in stream concentration values, especially TSS, can be traced to precipitation pattern, frequency and intensity. During these two years, the heaviest rainfall event occurred in September of 2014 (3.72 inches) followed by June in 2014 (3.36 inches) and June of 2015 (3.16 inches). High intensity storms in the early spring may have disturbed the streambed and acted as a driving force for erosion of sediments. However, it is also possible that the stream samples collected a mixture of both suspended solids and larger sediments that are less likely to be carried long distance (bedload sediments). In an attempt to adjust for this effect, bottles were installed at different heights at each sampling sites, one higher (closer to water surface) and the other lower (closer to streambed). No significant differences were observed between high and low samples, and it is possible that both high and low samples were affected by bedload sediment transport.

Haase Outfall Analysis Results. The Haase outfall samples also showed a difference in water quality from year to year. The TSS concentration values increased, similar to the other stream samples, from 155 to 250 mg/L. However, TN and TP concentrations decreased from 10.1 to 5.6 mg-N/L and from 1.04 to 0.90 mg-P/L, respectively. This is in contrasts to the TOT runoff quality results from the wetland sites, where nutrients increased from 2014 to 2015. However, the crop rotation at the Haase site, with corn in 2014 and soybeans in 2015, was the opposite of the Harvest Hills sites. This further demonstrates that the crop type plays a

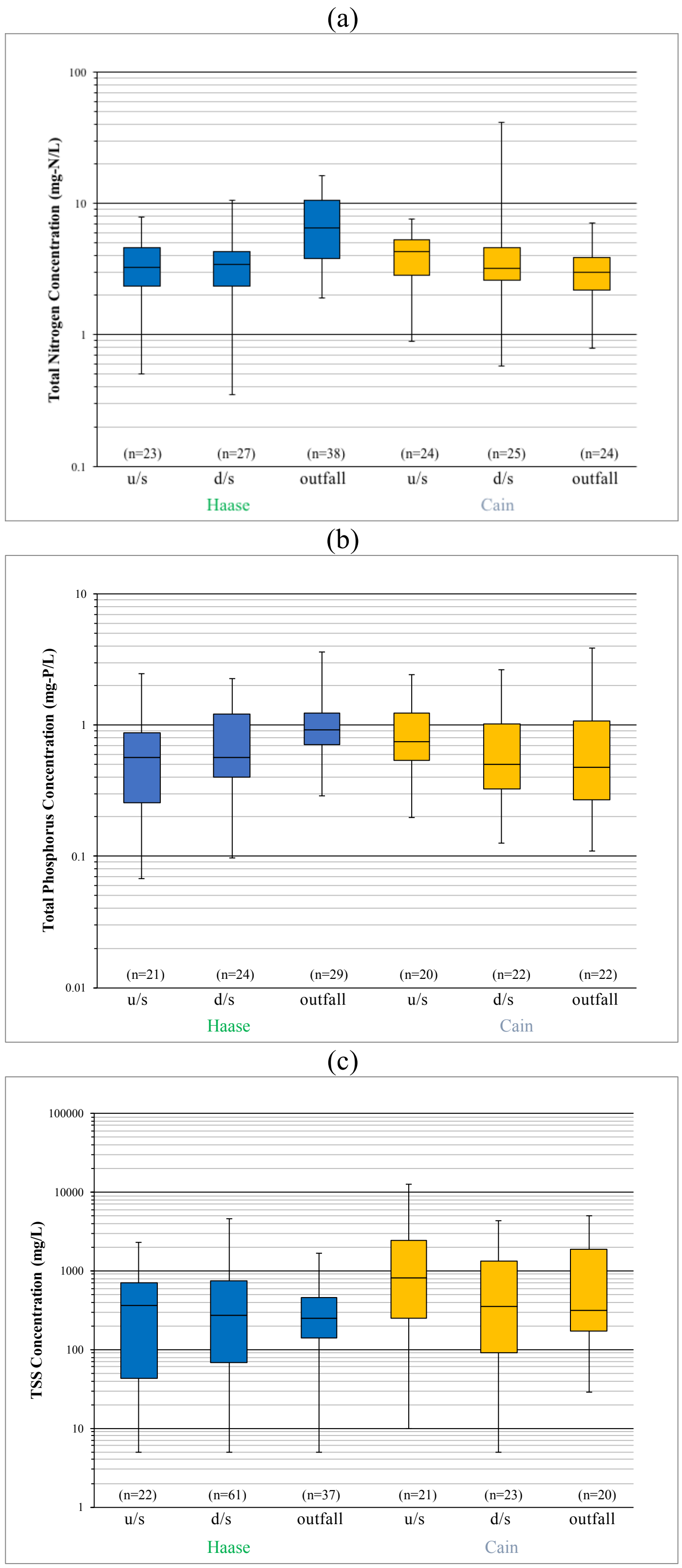


Figure 13. Stream sample concentrations of (a) total nitrogen (TN), (b) total phosphorus (TP) and (c) total suspended solids (TSS) at the Haase and Cain sites.

significant role in determining runoff water quality, especially nutrients and sediments, because of difference in fertilizer application rates and in soil cover levels.

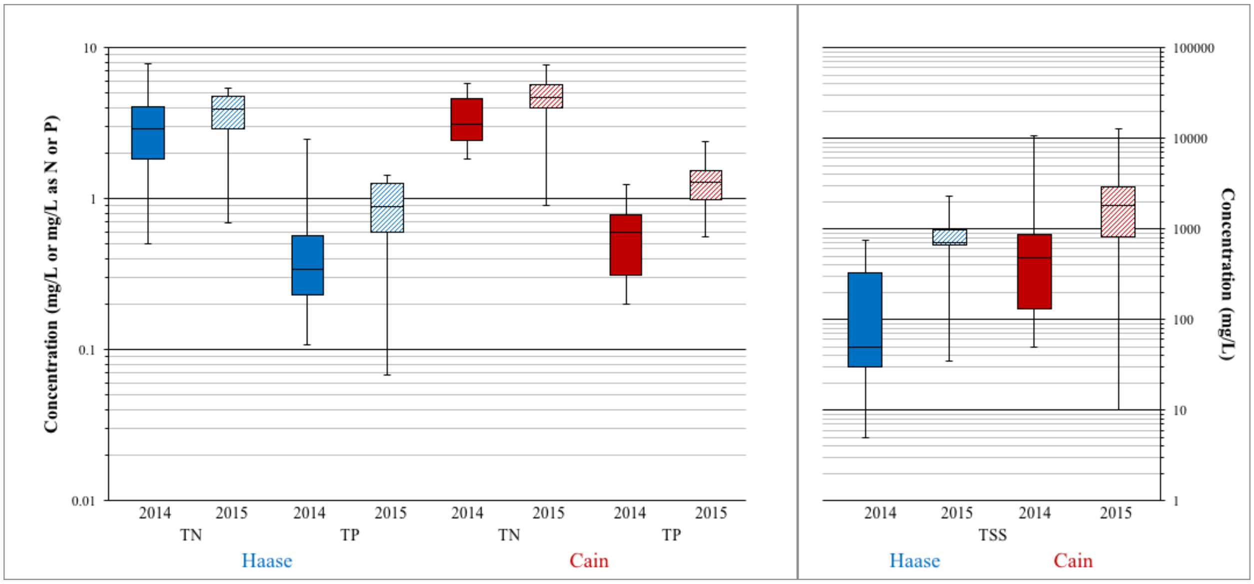


Figure 14. Upstream TN, TP and TSS concentration values at Haase and Cain site streams in 2014 (solid) and 2015 (striped).

Impact of Agricultural Runoff on Stream Quality. The wide range of concentration values observed for nutrients and solids may hinder our assessment of any impact of agricultural runoff on receiving stream quality. To determine whether there was any measurable impact of either untreated or treated runoff discharge on stream quality, median relative difference value was calculated over the whole data set for TN, TP and TSS at each site. Then, the one-sample Wilcoxon Signed-Rank test was used to determine if the median values were significantly different from zero. A positive relative difference would indicate a higher downstream concentration, and a negative value a lower downstream concentration. (Zero indicates no difference.)

The relative differences and confidence intervals are shown in Figure 15. All median values were below zero, which indicates lower concentrations downstream relative to the upstream sample point. However, confidence intervals for all parameters include zero, showing that the difference is not statistically significant. Thus, concentrations of TSS, TN and TP in the two streams showed no consistent change due to either the wetland effluent (Cain site) or the TOT runoff (Haase site). As discussed above, the concentrations entering the streams at the discharge were generally higher, although not significantly, than upstream or downstream values. It may be that the volume of runoff discharge entering the streams is small compared to the stream flow, so any impact would be rapidly diluted out. Another possible reason may be the method of sample collection, which may have collected only the early portion of storm runoff. Collecting samples over a longer period during and after a storm event may show a greater impact of discharge on water quality.

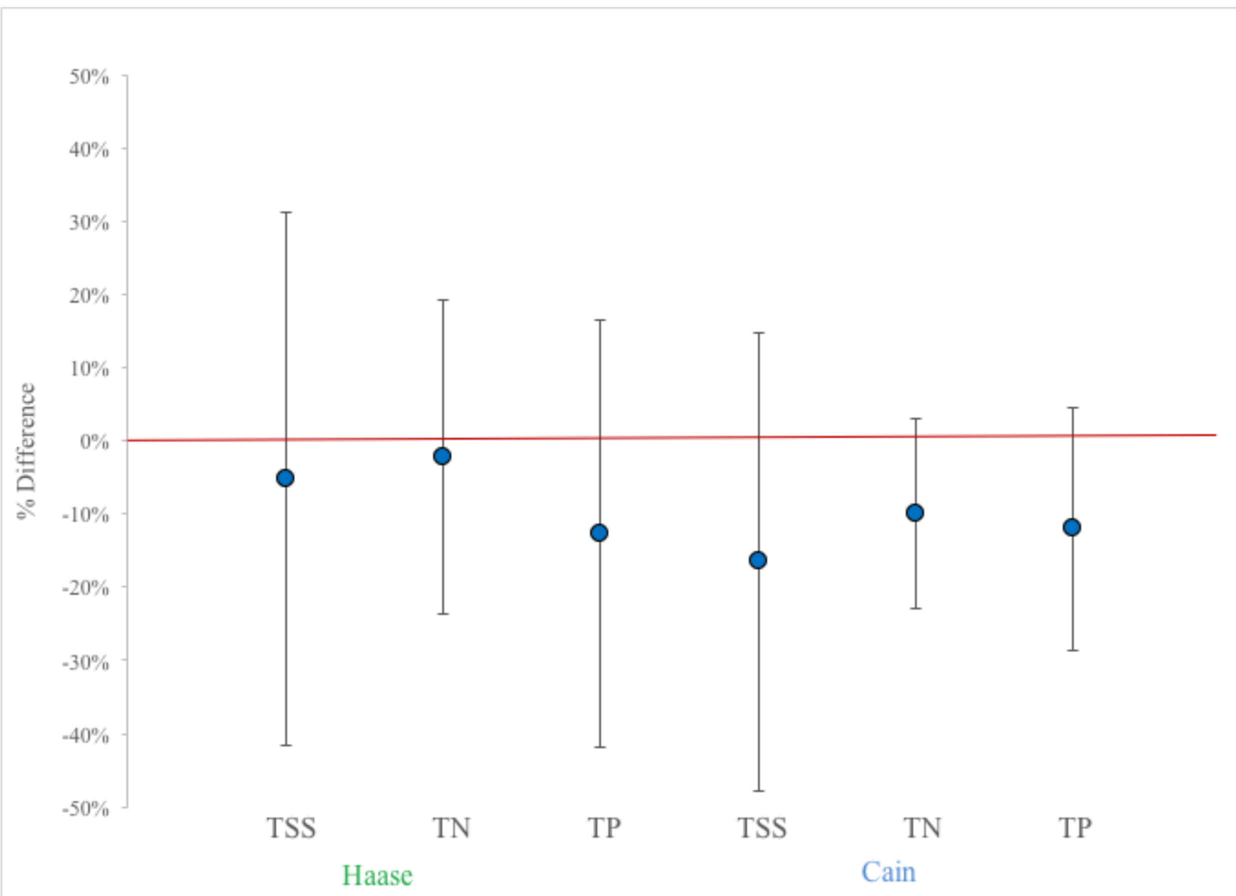


Figure 15. Relative differences between upstream and downstream water quality parameters in the at Haase and Cain streams.

4 CONCLUSIONS

The use of constructed wetlands to treat tile drainage runoff demonstrated a positive removal efficiency in the 2-year study for TSS, TN and TP. The TOT runoff from the three sites (and at the Haase outfall) contained elevated concentrations of nitrogen, phosphorus and suspended solids compared to benchmark or typically observed values for surface waters in the state of Kansas. Benchmark values for streams in EPA Region 7 are 0.9 mg/L for TN and 0.075 mg/L for TP [49]. Late crop planting coupled with earlier, high-intensity storm events were seen to negatively affect agricultural runoff quality, especially for TSS in runoff, and higher fertilizer application rates coupled with moderate-intensity storm events were shown to significantly elevate levels of TN and TP, particularly with dissolved species of N and P.

For wetland performance, higher influent concentrations typically resulted in better removal efficiencies, although the wetland design and inflow volume played an important role, as well. The difference in wetland performance between sites 1 and 2, which had similar design, CDA, and quality and volume of inflow, may be due to the high sediment loading and consequent development of a sediment bed near the TOT influent discharge location in wetland 1, causing better flow distribution. Other potential factors contributing to the performance differences of the three wetlands were wetland seepage rates and extent of vegetation establishment. Analysis of adjacent streams receiving treated (wetland effluent) and untreated (direct) TOT runoff suggest no measurable direct impact of the runoff on stream quality. However, the median wetland influent concentrations were higher than those in the stream samples, and the stream values were higher than reference values for Kansas streams and rivers to begin with. The results demonstrate the reduction of nutrients in agricultural TOT runoff in the wetlands, and hence the use of constructed wetlands is suggested to improve the quality of receiving waterbodies. A further assessment of constructed wetlands is suggested for future studies, through more frequent water collection throughout the year, not just during the growing season, to understand performance under various weather conditions, and through better knowledge of other important factors including hydrology and vegetation. Through more frequent field data collection in combination with modeling studies, determination of important wetland design parameters could be achieved and, consequently, will lead to a better design of future guidelines for treatment wetlands.

5 REFERENCES

1 U.S. EPA. National Water Quality Inventory: Report to Congress. (Tech. rep., Washington, DC: Environmental Protection Agency, Washington, DC, 2009).

2 Baker, L. A. Introduction to nonpoint source pollution in the United States and prospects for wetland use. *Ecological Engineering* **1**, 1-26 (1992).

3 Fink, D. F. & Mitsch, W. J. Seasonal and storm event nutrient removal by a created wetland in an agricultural watershed. *Ecological Engineering* **23**, 313-325 (2004).

4 Kadlec, R. H. & Wallace, S. *Treatment wetlands*. (CRC press, 2008).

5 Ribaudo, M. O., Heimlich, R., Claassen, R. & Peters, M. Least-cost management of nonpoint source pollution: source reduction versus interception strategies for controlling nitrogen loss in the Mississippi Basin. *Ecological Economics* **37**, 183-197 (2001).

6 Smith, D. *et al.* Nutrient losses from row crop agriculture in Indiana. *Journal of Soil and Water Conservation* **63**, 396-409 (2008).

7 Pavelis, G. A. *Farm drainage in the United States: History, status, and prospects*. (US Department of Agriculture, Economic Research Service, 1987).

8 Edward F. Peltier, C. B. Y., LlynnAnn Luellen, Hyunjung Lee. Monitoring effectiveness of tile outlet terrace fields with constructed wetlands for sediment, nutrient, and volume reduction in Northeastern Kansas. Report No. [Draft Version], (Topeka, KS, 2016).

9 Kladivko, E., Grochulska, J., Turco, R., Van Scoyoc, G. & Eigel, J. Pesticide and nitrate transport into subsurface tile drains of different spacings. *Journal of Environmental Quality* **28**, 997-1004 (1999).

10 Royer, T. V., David, M. B. & Gentry, L. E. Timing of riverine export of nitrate and phosphorus from agricultural watersheds in Illinois: Implications for reducing nutrient loading to the Mississippi River. *Environmental Science & Technology* **40**, 4126-4131 (2006).

11 David, M. B., Drinkwater, L. E. & McIsaac, G. F. Sources of nitrate yields in the Mississippi River Basin. *Journal of Environmental Quality* **39**, 1657-1667 (2010).

12 Hatfield, J., McMullen, L. & Jones, C. Nitrate-nitrogen patterns in the Raccoon River Basin related to agricultural practices. *Journal of soil and water conservation* **64**, 190-199 (2009).

13 Algoazany, A., Kalita, P., Czapar, G. & Mitchell, J. Phosphorus transport through subsurface drainage and surface runoff from a flat watershed in east central Illinois, USA. *Journal of environmental quality* **36**, 681-693 (2007).

14 King, K. W. *et al.* Phosphorus transport in agricultural subsurface drainage: A review. *Journal of environmental quality* **44**, 467-485 (2015).

15 Zimmer, D., Kahle, P. & Baum, C. Loss of soil phosphorus by tile drains during storm events. *Agricultural Water Management* **167**, 21-28 (2016).

16 Jaynes, D., Colvin, T., Karlen, D., Cambardella, C. & Meek, D. Nitrate loss in subsurface drainage as affected by nitrogen fertilizer rate. *Journal of Environmental Quality* **30**, 1305-1314 (2001).

17 Bakhsh, A. *et al.* Cropping system effects on NO3–N loss with subsurface drainage water. *Transactions of the ASAE* **45**, 1789 (2002).

18 Bjorneberg, D. L., Karlen, D. L., Kanwar, R. S. & Cambardella, C. A. Alternative N fertilizer management strategies effects on subsurface drain effluent and N uptake. *Applied engineering in agriculture* **14**, 469-473 (1998).

19 Kladivko, E. *et al.* Nitrate leaching to subsurface drains as affected by drain spacing and changes in crop production system. *Journal of Environmental Quality* **33**, 1803-1813 (2004).

20 Kanwar, R. S., Colvin, T. S. & Karlen, D. L. Ridge, moldboard, chisel, and no-till effects on tile water quality beneath two cropping systems. *Journal of Production Agriculture* **10**, 227-234 (1997).

21 Bakhsh, A. *et al.* Tillage and nitrogen management effects on crop yield and residual soil nitrate. *Transactions of the ASAE* **43**, 1589 (2000).

22 Jaynes, D., Hatfield, J. & Meek, D. Water quality in Walnut Creek watershed: Herbicides and nitrate in surface waters. *Journal of environmental quality* **28**, 45-59 (1999).

23 Kanwar, R. S., Baker, J. L. & Baker, D. Tillage and split N-fertilization effects on subsurface drainage water quality and crop yields. *Transactions of the ASAE* **31**, 453-0461 (1988).

24 Mississippi River Gulf of Mexico Watershed Nutrient Task Force. Gulf Hypoxia Action Plan 2008. (2008).

25 Sprague, L. A., Hirsch, R. M. & Aulenbach, B. T. Nitrate in the Mississippi River and its tributaries, 1980 to 2008: Are we making progress? *Environmental Science & Technology* **45**, 7209-7216 (2011).

26 David, M. B. *et al.* Biophysical and social barriers restrict water quality improvements in the Mississippi River basin. *Environmental science & technology* **47**, 11928-11929 (2013).

27 Sprague, L. A., Mueller, D. K., Schwarz, G. E. & Lorenz, D. L. Nutrient Trends in Streams and Rivers of the United States, 1993-2003. (U. S. Geological Survey, 2009).

28 Kovacic, D. A., David, M. B., Gentry, L. E., Starks, K. M. & Cooke, R. A. Effectiveness of constructed wetlands in reducing nitrogen and phosphorus export from agricultural tile drainage. *Journal of environmental quality* **29**, 1262-1274 (2000).

29 Reinelt, L. E. & Horner, R. R. Pollutant removal from stormwater runoff by palustrine wetlands based on comprehensive budgets. *Ecological Engineering* **4**, 77-97 (1995).

30 Rushton, B., Miller, C., Hull, C. & Cunningham, J. in *Proceedings of the Fourth Biennial Stormwater Research Conference, Clearwater Beach, FL.*

31 Groh, T. A., Gentry, L. E. & David, M. B. Nitrogen removal and greenhouse gas emissions from constructed wetlands receiving tile drainage water. *Journal of environmental quality* **44**, 1001-1010 (2015).

32 Hoagland, C. R., Gentry, L. E., David, M. B. & Kovacic, D. A. Plant nutrient uptake and biomass accumulation in a constructed wetland. *Journal of Freshwater Ecology* **16**, 527-540 (2001).

33 KDHE. Upper Wakarusa Watershed Restoration and Protection Strategy: Nine Element Plan. (Kansas Dept. of Health and Environment, Topeka, KS, 2011).

34 KWO. A long-term vision for the future of water supply in Kansas. (Kansas Water Office, Topeka, KS, 2015).

35 NOAA. NOAA Online Weather Data (NOWData). (2014).

36 Federation, W. E. & Association, A. P. H. Standard methods for the examination of water and wastewater. *American Public Health Association (APHA): Washington, DC, USA* (2005).

37 Kumar, K. & Goh, K. Crop residues and management practices: effects on soil quality, soil nitrogen dynamics, crop yield, and nitrogen recovery. *Advances in Agronomy* **68**, 197-319 (1999).

38 Vigon, B. W. The Status of Nonpoint Source Pollution: Its Nature, Exten and Control 1. *JAWRA Journal of the American Water Resources Association* **21**, 179-184, doi:10.1111/j.1752-1688.1985.tb00127.x (1985).

39 Heathwaite, A. & Dils, R. Characterising phosphorus loss in surface and subsurface hydrological pathways. *Science of the Total Environment* **251**, 523-538 (2000).

40 McDowell, R. & Sharpley, A. Approximating phosphorus release from soils to surface runoff and subsurface drainage. *Journal of environmental quality* **30**, 508-520 (2001).

41 Steegen, A. *et al.* Factors controlling sediment and phosphorus export from two Belgian agricultural catchments. *Journal of Environmental Quality* **30**, 1249-1258 (2001).

42 McDowell, R., Sharpley, A., Condron, L., Haygarth, P. & Brookes, P. Processes controlling soil phosphorus release to runoff and implications for agricultural management. *Nutrient Cycling in Agroecosystems* **59**, 269-284 (2001).

43 Dickey, E. C., Shelton, D. P., Jasa, P. J. & Peterson, T. R. Soil erosion from tillage systems used in soybean and corn residues. *Transactions of the ASAE* **28**, 1124-1130 (1985).

44 Ghidey, F. & Alberts, E. Runoff and soil losses as affected by corn and soybean tillage systems. *Journal of Soil and Water Conservation* **53**, 64-70 (1998).

45 Sabokrouhiyeh, N., Bottacin-Busolin, A., Nepf, H. & Marion, A. in *Hydrodynamic and Mass Transport at Freshwater Aquatic Interfaces: 34th International School of Hydraulics* (eds Paweł M. Rowiński & Andrea Marion) 101-113 (Springer International Publishing, 2016).

46 Jadhav, R. S. & Buchberger, S. G. Effects of vegetation on flow through free water surface wetlands. *Ecological Engineering* **5**, 481-496 (1995).

47 Hey, D. L., Kenimer, A. L. & Barrett, K. R. Water quality improvement by four experimental wetlands. *Ecological Engineering* **3**, 381-397 (1994).

48 Schueler, T. Irreducible pollutant concentrations discharged from urban BMPs. *Watershed Protection Techniques* **2**, 369-372 (1996).

49 U.S. EPA RTAG7. Nutrient Reference Condition Identification and Ambient Water Quality Benchmark Development Process: Rivers and Streams within EPA Region 7. (USEPA RTAG7, Kansas City, KS, 2011).