ABSTRACT

IMPACT OF A FORESTED STATE PARK ON NUTRIENT CONCENTRATIONS IN AN AGRICULTURALLY DOMINATED WATERSHED IN SOUTHWEST OHIO

by Tessa Louise Farthing

Agricultural land cover in the U.S. Midwest is a major source of nutrient pollution that has led to severe degradation of stream water quality. Previous studies have shown that land cover, stream morphology, and hydrology can influence stream nutrient concentrations. This study examines the impact of a forested state park on nutrient concentrations within an agriculturally dominated watershed. Water samples were collected biweekly from eight stream sampling sites along four creeks and processed for total nitrogen (TN), nitrate (NO_3^-), phosphorus (TP), and orthophosphate (PO_4^{3-}). Hydrology, channel morphology, and remotely sensed vegetation data were also collected and analyzed within the study area. An analysis of covariance test (ANCOVA) and a regression coefficient t-test indicated that the state park significantly reduced NO_3^-, PO_4^{3-}, and TP concentrations. The park as a whole did not significantly reduce TN concentrations, however, within one of the four creeks, significant decreases in TN concentrations were detected. Discharge was a significant driving factor for changes in TN, NO_3^-, and TP concentrations within one study creek and change in PO_4^{3-} concentrations within an additional study creek. The normalized difference vegetation index (NDVI) was a significant predictor of reductions in TN concentrations within one of four study creeks, and NDVI was globally correlated with reductions in NO_3^- concentrations. The results of the study suggest that conservation of forested areas within agriculturally dominated watersheds can provide meaningful water quality improvements in the U.S. Midwest.

IMPACT OF A FORESTED STATE PARK ON NUTRIENT CONCENTRATIONS IN AN AGRICULTURALLY DOMINATED WATERSHED IN SOUTHWEST OHIO

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by

Tessa Louise Farthing

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Advisor: Dr. Bartosz Grudzinski

Reader: Dr. Thomas Fisher

Reader: Dr. Jessica McCarty

Reader: Dr. Michael Vanni

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by

Tessa Louise Farthing

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College of Arts and Science

and

Department of Geography

Dr. Bartosz Grudzinski

Dr. Thomas Fisher

Dr. Jessica McCarty

Dr. Michael Vanni

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I. Introduction

Anthropogenic impacts on freshwater environments are a major concern in the Great Lakes region of the U.S. Midwest as agricultural production continues to lead to severe water quality degradation (Tong et al., 2007; Giri & Qiu, 2016; Falcone et al., 2017). Nonpoint source pollutants, including from cropland runoff, are widespread across U.S. Midwestern landscapes, making them difficult to track and regulate (Mello et al., 2018). Excessive nutrient inputs can result in eutrophication, which can lead to increased algal growth, anoxic zones, and the release of toxins (Carpenter et al., 1998; Dodds & Smith, 2016). Prior to widespread anthropogenic land development in the U.S. Midwest and subsequent declines in water quality, these landscapes primarily consisted of forests or grasslands. Remaining undeveloped land areas within agriculturally dominated watersheds may result in nutrient reductions within stream environments.

Many previous studies have shown the important role that land use has on stream nutrient dynamics. Generally, agricultural land cover is associated with higher phosphorus and nitrogen concentrations and loads (Carpenter et al., 1998; Nielsen et al., 2012; Ostrofsky et al., 2017; Mello et al., 2018). Forest cover and riparian zones, in contrast, have been shown to improve water quality through phosphorus and nitrogen removal (Peterjohn et al., 1984; Lowrance et al., 1997; Webber et al., 2002; Lefebvre et al., 2005; Ostrofsky et al., 2017; Mello et al., 2018; Weigelhofer, 2018). Furthermore, native forest cover is one of the most important land cover types to preserve water quality in low order streams (Mello et al., 2018). Larger riparian forested areas typically yield higher nutrient removal rates (Vanni et al., 2001; King et al., 2016). However, due to extensive forest loss throughout the agricultural U.S. Midwest, most previous studies are limited to a focus on the effects of relatively thin riparian buffer strips rather than more substantial forested areas.

In some areas hydrogeological setting within a riparian zone may have a more significant impact on groundwater denitrification rates and hydrological function than riparian buffer width (Vidon & Smith, 2007; Hill, 2017). Seasonal fluctuations in hydrological characteristics can cause changes in stream nutrient levels. For example, increases in precipitation and surface runoff can increase stream discharge and groundwater exchange rates (Vega et al., 1998; Arntzen et al., 2006; Shrestha & Kazama, 2007). Nutrient flux from the watershed can also increase as stream flow increases from higher rates of overland flow (Petry et al., 2002). Sediment mobilization from runoff may release additional sediment-bound nutrients including phosphorus into the water column (Wallbrink et al., 2003). High flows decrease water residence time in fluvial systems due to increased water velocity (Withers & Jarvie, 2008; Rech et al., 2018), while nutrient uptake by vegetation and stream algae can increase during periods of low flow and high water residence time (Jansson et al., 1994; Royer et al., 2004; Bernot et al., 2006; Chen et al., 2010). In the U.S. Midwest, land cover and stream hydrology may have variable impacts on nutrient concentrations. For example, in one study, both agricultural land cover and storm-flow discharge were found to be significant drivers of total nitrogen (TN), orthophosphate (PO_4^{3-}), and total phosphorus (TP) concentrations, but storm-flow discharge was not a significant driver of nitrate (NO_3^-) concentrations (Lazar et al., 2019). Nitrate concentrations may be more heavily influenced by stream flow associated with seasonal changes rather than storm pulses (Kalkhoff et al., 2016). Phosphorus concentrations may also remain more consistent than nitrogen concentrations through seasonal changes in stream flow but have been shown to be more sensitive to larger storm events (Vanni et al., 2001; Kalkhoff et al., 2016). Therefore, peaks in phosphorus input may be higher during months with the greatest precipitation.

Within a forested riparian buffer, vegetation condition or "greenness" is often quantified using the Normalized Difference Vegetation Index (NDVI), derived from satellite imagery as a proxy. Vegetation growth, photosynthetic capacity, and greenness are typically highly correlated with NDVI (Griffith, 2002; Zhu et al. 2015; Robinson et al., 2017). NDVI has previously been shown to be more correlated with vegetation phenology metrics, water quality parameters, and indices of biological integrity than land cover and land use (Griffith et al., 2002). Early growing season NDVI, which is also referred to as the onset of greenness can be a strong predictor of water quality parameters (Griffith et al., 2002). In China, sub-basin NDVI was found to be a significant factor in predicting nonpoint source nitrogen and phosphorus loading (Ouyang et al., 2009). Additionally, the utilization of riparian buffers with high NDVI adjacent to farmlands was found to be an effective management strategy for reducing nonpoint source nutrients (Ouyang et al., 2009). Increased total nitrogen and phosphorus input in the spring and summer can also increase hydrophyte growth and consequently increase vegetation index values as well as nutrient uptake (Chen et al., 2010; Zhu et al., 2015).

Channel morphology and stream size may also impact in-stream nutrient dynamics. Larger streams have a higher water volume to creek bed ratio than smaller streams (Alexander et al., 2000; Royer et al., 2004). This relationship results in less nitrogen and phosphorus exchange with the creek bed and therefore a decreased reduction in nutrients within the water column (Alexander et al., 2000; Bernot & Dodds, 2005; Withers & Jarvie, 2008). Low order streams are also expected to have a shallower depth, which increases light penetration. With increased light penetration primary production rates and nutrient uptake may also increase (Withers & Jarvie, 2008). Increased channel complexity and sinuosity can also increase flow paths, hyporheic exchange, transient storage, and residence time (Roley et al., 2012; Covino, 2017). Streams with a more complex geomorphology may yield the highest rates of denitrification (Alexander et al., 2000; Bernot & Dodds, 2005; Opdyke et al., 2006). Studies have shown that riffles with coarse sediment may have lower denitrification rates than storage zones such as pools with finer sediment, higher organic matter concentrations, and increased residence time (Hill et al., 1998; Opdyke et al., 2006). Pools may also provide short term retention of phosphorus due to sorption by sediment, while long term retention may be a result of biological uptake; however, retention is limited by transport during storm events and high flows (Meals et al., 1999; Withers & Jarvie, 2008). Therefore, channel morphology may have less of an influence on phosphorus retention in comparison to hydrology and biochemical processes associated with phosphorus uptake (Bernot & Dodds, 2005; Withers & Jarvie, 2008).

II. Research Questions and Hypotheses

Research Objective

The primary goal of this study was to determine if a forested landscape, contained within a state park, would provide water quality improvements within an agriculturally dominated watershed in Southwest Ohio. The secondary goal of this study was to identify additional factors that may influence changes in nutrient concentrations within the forested area.

Primary Research Question: Does a large, forested state park impact nitrogen and phosphorus concentrations in an agriculturally dominated watershed in Southwest Ohio?

-Hypothesis I: All nutrient concentrations (TN, NO_3^-, TP, PO_4^{3-}) will decrease within the forested state park.

Secondary Research Question: Do discharge, vegetation condition (NDVI), or stream characteristics impact changes in nutrient concentrations within the forested state park?

-Hypothesis I: An increase in discharge will result in a decrease in nutrient reduction within the state park.

-Hypothesis II: An increase in NDVI will be associated with a greater decrease in all nutrient concentrations due to increased uptake by vegetation.

-Hypothesis III: Creeks with higher residence times will retain more nutrients, and therefore exhibit the greatest decreases in nutrient concentrations.

III. Literature Review

From a biogeochemical standpoint, nutrients can undergo several transformations depending on the surrounding conditions. Understanding these underlying mechanisms is important when trying to determine the different chemical forms of a parameter to measure. The reactive forms of nitrogen commonly associated with aquatic studies are nitrite (NO_2^-), nitrate (NO_3^-), and ammonium (NH_4^+). Nitrite is the intermediate product of nitrification and has a short half-life, which often results in negligible concentrations. Ammonium is rapidly nitrified into NO_3^- via soil microbes. Ammonium also has a positive charge which encourages interaction with the negative charge of soils, making it immobile. Nitrate, on the other hand, is stable and soluble. Nitrate also has a negative charge that does not allow for interaction with soils, which is why it has the greatest potential for loss to waterways through leaching and runoff (Hatch et al., 2002; Figure 1).

Phosphorus can be found in several different forms in water: inorganic, organic, dissolved, colloidal, and particle associated (Leinweber et al., 2002). The phosphorus cycle includes a biological and a geochemical sub-cycle (Mullen, 2005; Figure 2). The geochemical component of the cycle involves the dissolution of phosphorus-containing minerals, reactions to form Fe, Al, Ca, and Mg phosphates, and the sorption of phosphorus clays and oxide soil minerals (Mullen, 2005; Figure 2). The biological component of the phosphorus cycle primarily consists of bacteria and fungi-facilitated mineralization and immobilization and dictates the amount that is readily available to plants (Mullen, 2005; Figure 2). It is important to understand the transformations phosphorus is undergoing to successfully measure it in an aquatic system. In an agricultural setting, colloidal organic phosphorus can enter an aquatic system through overland flow. Organic phosphorus can be converted to inorganic orthophosphate (PO_4^3), the most usable form for plants, because it does not require additional biochemical processing by soil microbes (Mullen, 2005; Weigelhofer et al., 2018; Figure 2). Fine clay sediment with a high sorption capacity will typically be enriched with phosphorus, and this sediment-bound phosphorus must be mobilized through overland flow to enter the water column (Leinweber et al., 2002).

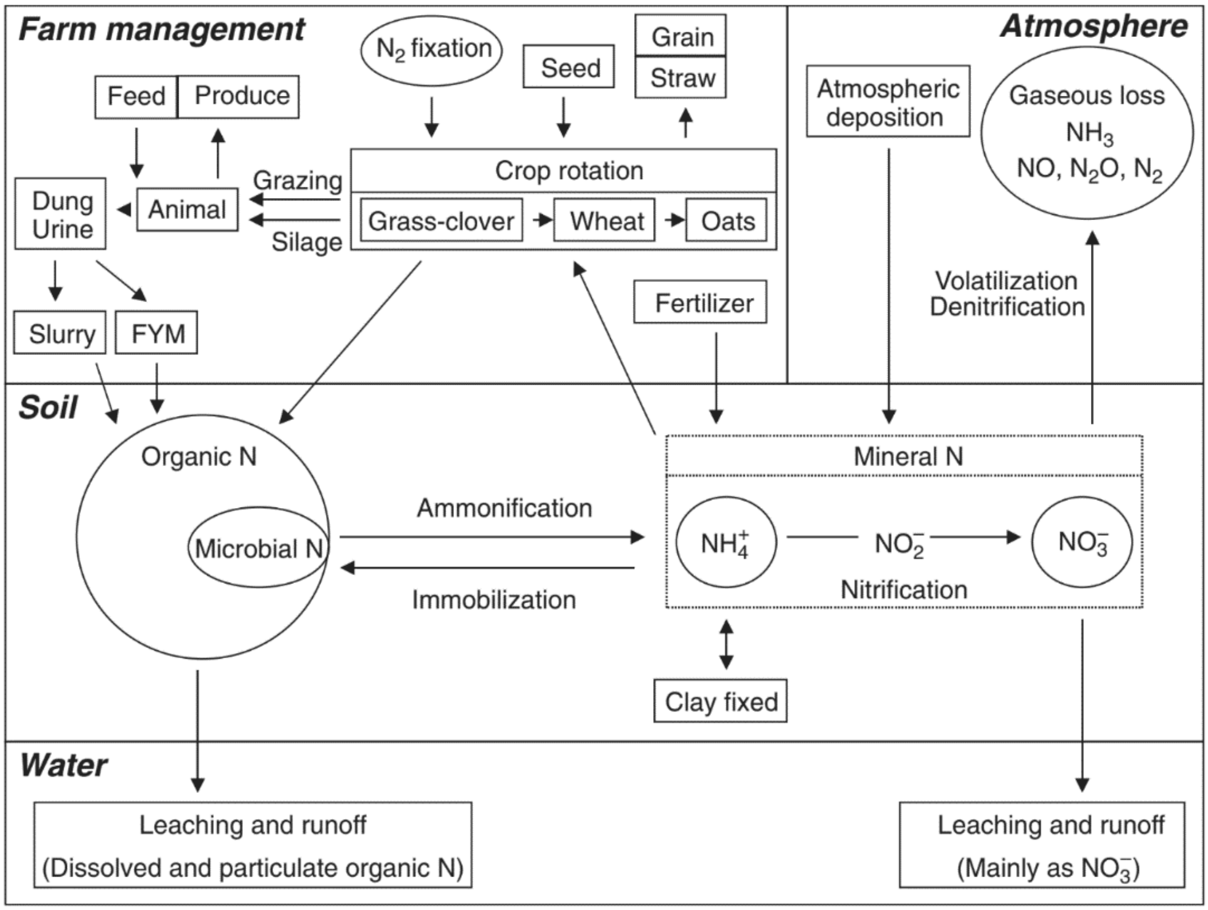


Figure 1: Nitrogen cycling from agricultural and atmospheric environments through soil and aqueous mediums (Hatch et al., 2002).

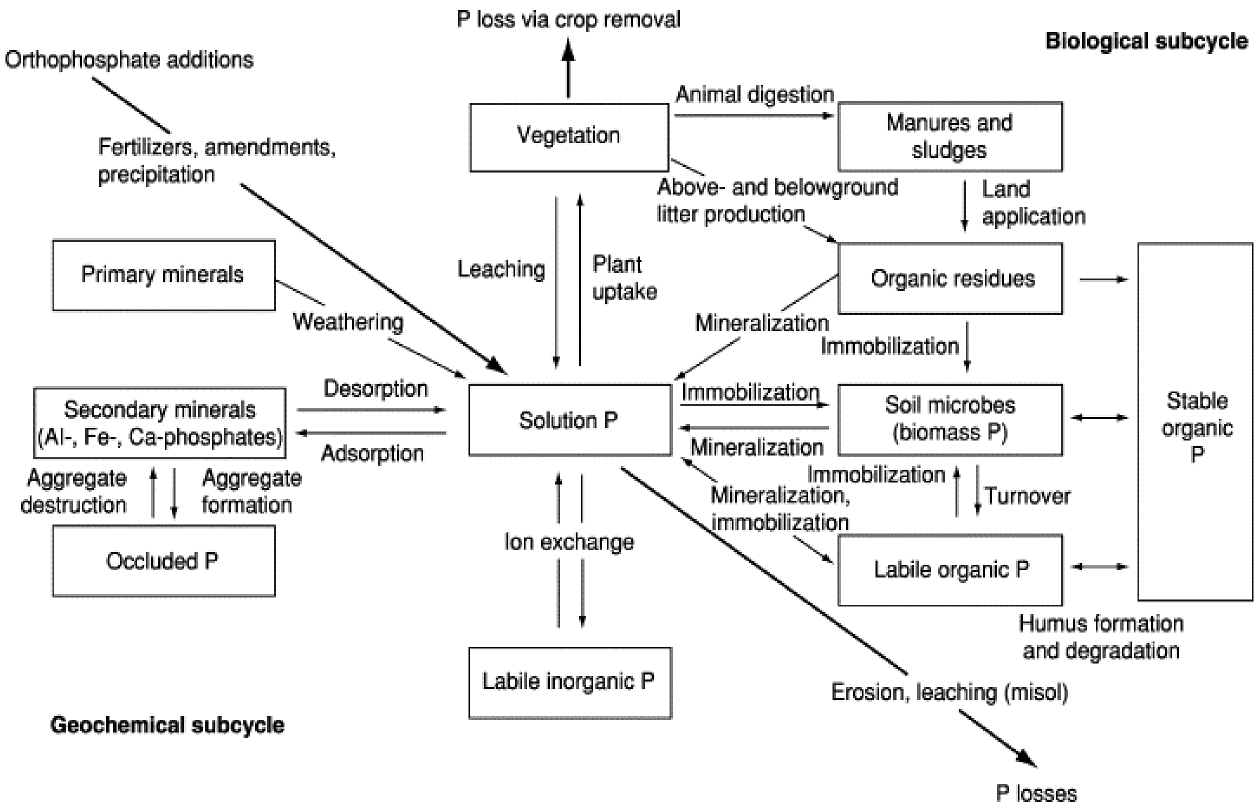


Figure 2: Phosphorus cycle including the biological and geochemical sub-cycles (Mullen, 2005).

To analyze the role of a forested state park on stream water quality, it is also necessary to understand how nutrients will be transported through the system. Most nitrogen input in cropland dominated watersheds comes from management practices including fertilization. Additionally, nitrogen retention rates in cropland are expected to be low, and the vast majority of nitrogen may fall subject to groundwater leaching primarily in the form of NO_3^- (Peterjohn & Correll, 1984; Hatch et al., 2002).

Nitrate is expected to be the most prominent form of total nitrogen (TN) in fluvial settings due to its negative charge, which prevents reactions with clays and organic matter (Hatch et al. 2002). A riparian forest adjacent to cropland is also expected to receive the majority of nitrogen from cropland in the form of NO_3^- via groundwater (King et al., 2016). In a study completed in Canada, NO_3^- removal rates in a riparian buffer were the highest in summer months (Satchithanantham et al., 2019). In contrast, NO_2^- is the reactive, intermediate product of nitrification, and it is not expected to be found in large quantities (Hatch et al., 2002). However, elevated temperatures, poor aeration, and alkaline environments can result in ammonium oxidation, which could then potentially increase NO_2^- accumulation and leaching (Hatch et al., 2002).

Phosphorus is often predominantly exported from cropland through overland flow (Peterjohn & Correll, 1984; Leinweber et al., 2002). Therefore, within the riparian buffer, groundwater flow may not be as influential for phosphorus loss and retention as it may be for nitrogen (Weigelhofer et al., 2018). In a previous study, phosphorus retention rates were greater in a riparian forest than cultivated land but were still generally lower than nitrogen retention rates in riparian zones (Peterjohn & Correll, 1984). A study in Canada also showed NO_3^- retention rates to be higher than dissolved phosphorus retention because PO_4^{3-} retention was dictated more by sorption and desorption in riparian buffers (Satchithanantham et al., 2019). Alternatively, retention rates of sediment-bound phosphorus in riparian buffers have been found to be higher than dissolved phosphorus (Hoffman et al., 2009).

In addition to natural transport paths, artificial connectivity, such as tile drains, are also expected to introduce pollutants into streams (Hoffmann et al., 2009). Tile drains are underground pipe systems that are used to remove excess water from soil to prevent oversaturation. In central Ohio, PO_4^{3-} and NO_3^- were the primary forms of nutrients in tile drain water under corn and soybean production (King et al., 2016). Large storm events can increase TP and PO_4^{3-} loads, but tile drain inputs can also be an important source of PO_4^{3-} (Leinweber et al., 2002; Gentry et al., 2007). Nitrate and NH_4^+ export rates have also been shown to be associated with increases in discharge on tile-drained land (Cuadra & Vidon, 2010). Differences in nitrogen and phosphorus concentration in tile drains can be due to fertilization timing and seasonal changes in discharge (King et al., 2016).

An increase in discharge and decrease in water residence time can reduce the amount of phosphorus uptake, while a lower velocity can increase water residence time and opportunity for interaction between a streambed and biota. This interaction can lead to periphyton proliferation and increased uptake of phosphorus (Withers & Jarvie, 2008). Orthophosphate and TP concentrations are anticipated to be the highest during storm events. Storm events increase overland flow, which may encourage mobilization of sediment-bound phosphorus and downstream transport (Kelly et al., 2019). Overall, phosphorus input may not be as seasonally driven as nitrogen, because nitrogen has been found to follow seasonal fluctuations in discharge and to plateau at larger events (Kalkhoff et al., 2016; Kelly et al., 2019).

Nitrogen flux may be more highly correlated with seasonal flow rates than phosphorus. Previous research in the study watershed showed that nitrogen loads were proportionate to discharge while phosphorus mostly exhibited disproportionate increases in loading during storm events (Kelly et al., 2019). While nitrogen concentrations are not expected to be as highly influenced by large storm pulses as phosphorus, total nitrogen (TN) concentrations may be higher when larger storm events occur than baseflow, and concentrations may be lower in seasons with lower discharge (Covino, 2017; Lazar et al., 2019). Specifically, NO_3^- may exhibit a higher concentration than phosphorus during baseflow relative to storm events, because NO_3^- is primarily mobilized by subsurface flow rather than overland flow (Hatch et al, 2002; Lazar et al., 2019). While NO_3^- could potentially increase with storm events, NO_3^- surface runoff loads can alternatively be diluted by storm events (Kelly et al., 2019). Hysteretic patterns among storm events due to various physical and biological watershed characteristics as well as potential post-drought flushing events are responsible for the diverging possibility of increased nitrogen concentrations versus dilution (Aguilera & Melack, 2018). Ammonium, unlike NO_3^-, reacts with the negative charge of clay and organic soil matter, generally making it immobile (Hatch et al., 2002). Because of this reaction, NH_4^+ can be mobilized by overland flow and large storm events rather than subsurface flow. However, in heavily fertilized watersheds, NH_4^+ is a relatively insignificant contributor to total dissolved nitrogen (Hatch et al., 2002; Kelly et al., 2019).

Modifying tillage practices is one best management strategy for achieving stream nutrient reductions. Conservation tillage, widely used in the Four Mile Creek Watershed, is designed to reduce erosion and nutrient loading (Renwick et al., 2008; Renwick et al., 2018). However, over extended periods of time, conservation tillage can lead to excess phosphorus in topsoil layers, which is highly susceptible to storm mobilization (Kelly et al., 2019). In the Midwest, PO_4^{3-} concentrations can initially decrease in response to conservation tillage to a point, but then eventually increase with long-term conservation tillage (Richards et al., 2009; Jarvie et al., 2017; Renwick et al., 2018). This increase in phosphorus can result in a TN:TP shift towards N-limitation (Kelly et al., 2019). Ammonium and NO_3^- concentrations can also decrease as a result of conservation tillage (Renwick et al., 2018). This decrease in NO_3^- is most likely due to more organic matter in soil and because reduced tillage can reduce oxygen availability and increase denitrification rates in the soil, which reduces NO_3^- leaching (Mkhabela et al., 2008; García et al., 2016).

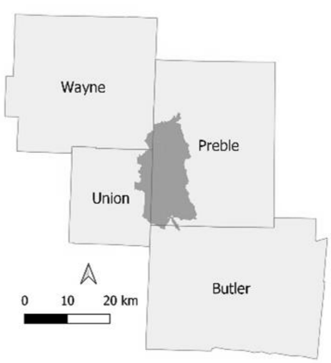
Natural areas such as grasslands, wetlands, and vegetated buffers can remove nutrients from aquatic systems (Baulch et al., 2019). A previous study in the Midwest suggested that watersheds with a higher percent of forested area in the riparian buffer zone had lower PO_4^{3-} stream concentrations, but there was not a clear relationship associated with NO_3^- or NH_4^+ concentrations with the amount of forested area (Webber et al., 2002). Nutrient uptake by plants in these areas is dictated by a number of physiological and biological processes. For example, the membrane potential and electrochemical gradients of plant cells must accommodate the charges associated with inorganic nutrients for successful transport (Reid & Hayes, 2003). Over forty percent of the canopy cover in Hueston Woods State Park is beech trees, which have been shown to exhibit high plasticity based on soil phosphorus availability and will produce as much biomass as possible under increased soil nutrient conditions (Ohio Department of Natural Resources, 2007; Meller et al., 2019). However, the percentage of beech trees within the riparian zone specifically may vary. Nitrogen is taken up by plants in the form of NO_3^- and NH_4^+, and when both forms are available in similar quantities, plants may preferentially uptake NH_4^+, because it requires less energy to use (Reid & Hayes, 2003). While uptake by trees is expected to be dominant in the forest buffer, stream algae, often quantified by chlorophyll-a concentrations, can also contribute to nutrient uptake in the Midwest (Hamilton et al., 2001; Dodds et al., 2002; Bernot et al., 2006). A previous study in the Midwest showed that the uptake of NO_3^- was not higher with higher input concentrations due to saturation, while NH_4^+ and PO_4^{3-} uptake did increase with higher input concentrations (Bernot et al., 2006). These relationships depend on the nutrient budgets in the aquatic system and which nutrients are in the most demand.

Best management strategies will likely face limitations due to the presence of legacy nutrients in soils and groundwater (Weigelhofer, 2018). Land use may not significantly impact phosphorus removal in some areas because of legacy phosphorus deposits buried in sediment. Even in forested areas, phosphorus can be released into the water column with high flow conditions (Kreiling et al., 2019). Long-term accumulation of phosphorus in watersheds is the result of fertilization over extended periods of time (Kusmer et al., 2019). Nitrogen legacy deposits are also a concern. Most of the NO_3^- exports in the Mississippi River are over 30 years old, and these legacy nutrients can lead to a delay in environmental response to management plans (Van Meter et al., 2018; Kusmer et al., 2019). Although, the addition of riparian zones has been shown to alleviate the lag time in nutrient reduction levels to some degree (Van Meter et al., 2018).

IV. Methods

Study Area

This study was completed within the Four Mile Creek Watershed which is located on the border of Southwest Ohio and Southeast Indiana, U.S., and is part of the greater Great Miami River watershed (Figure 3). The four creeks sampled for water quality include Four Mile Creek, Little Four Mile Creek, Marshall's Branch, and Deer's Ear (unofficial name). All four study creeks drain into the northern portion of Acton Lake, within Hueston Woods State Park (Figure 3). Each creek had two sampling locations: one located at the northern park boundary of the state park and farm land and another located downstream within the state park prior to the creeks flowing into Acton Lake (Figure 4). The watershed is agriculturally dominated (Figure 5), but a variety of sustainable practices have been put in place to manage nutrient and sediment runoff (Renwick et al., 2018; Kelly et al., 2019). Despite these efforts, the creeks and lakes within the watershed remain subject to eutrophication.



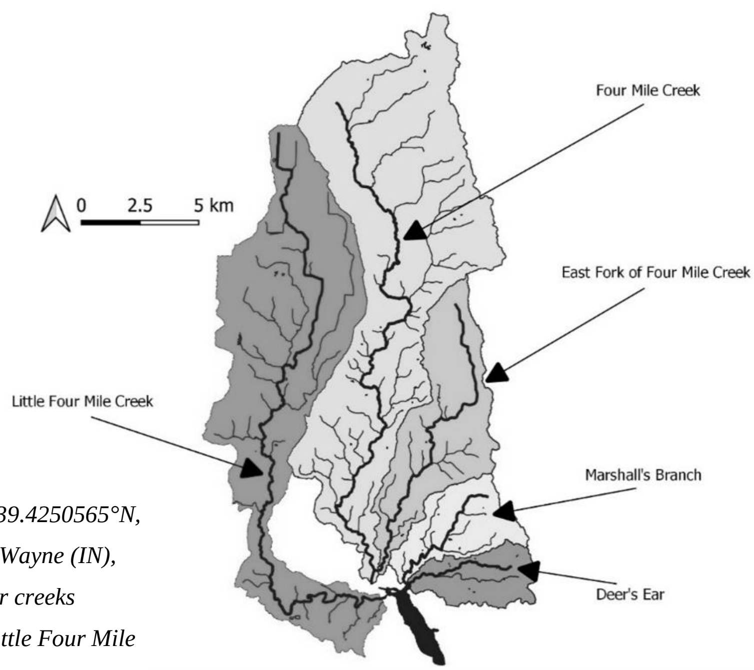


Figure 3: The Upper Four Mile Creek Watershed (39.4250565°N, -84.5435575°W) is contained within four counties: Wayne (IN), Union (IN), Preble (OH), and Butler (OH). The four creeks examined in the study area are Four Mile Creek, Little Four Mile Creek, Marshall's Branch, and Deer's Ear.

The basic geologic units present within the study area are Ordovician bedrock, glacial till, and outwash. The bedrock is mostly lithified limestone or unlithified shale and is mostly overlain by Wisconsin glacial till. Creeks primarily consist of step-pool and pool-riffle geomorphology (Rech et al., 2018), however, the creeks have variable underlying geology, including median creek bed sediment particle size, bed rock exposure, and overall channel morphology (Table 1).

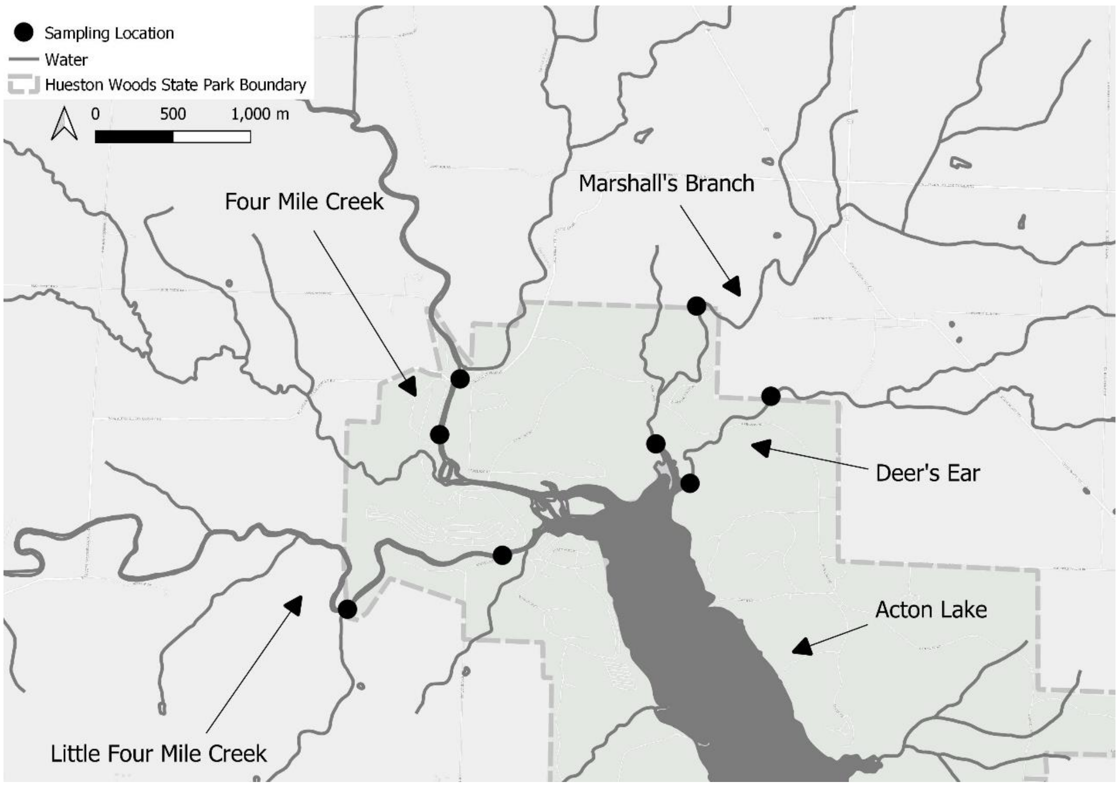


Figure 4: Sampling was completed on the four main creeks that drain into Acton Lake: Four Mile Creek, Little Four Mile Creek, Marshall's Branch, and Deer's Ear. Each creek has a sampling location at the park boundary and a point downstream before water enters the lake.

Four Mile Creek has the largest total watershed drainage area and average channel area, followed by Little Four Mile Creek, Marshall's Branch, and Deer's Ear (Table 1). Four Mile Creek is the widest creek on average while Marshall's Branch is the deepest creek within the state park (Table 1). Four Mile Creek and Little Four Mile Creek have the highest amount of pool volume within the state park and Marshall's Branch has the longest residence time (Table 1). In the study area, the wettest months of the year are generally March through July. As temperature reaches its peak at the end of July and into August, the precipitation begins to decrease. The driest months are August through October and February (Debrewer et al., 2000; Rech et al., 2018; Figure 6). Summer and spring precipitation events tend to be shorter and more intense than those in fall and winter (Debrewer et al., 2000). The highest discharges in the creeks are typically observed in March and April and the lowest discharges are typically observed from August to October.

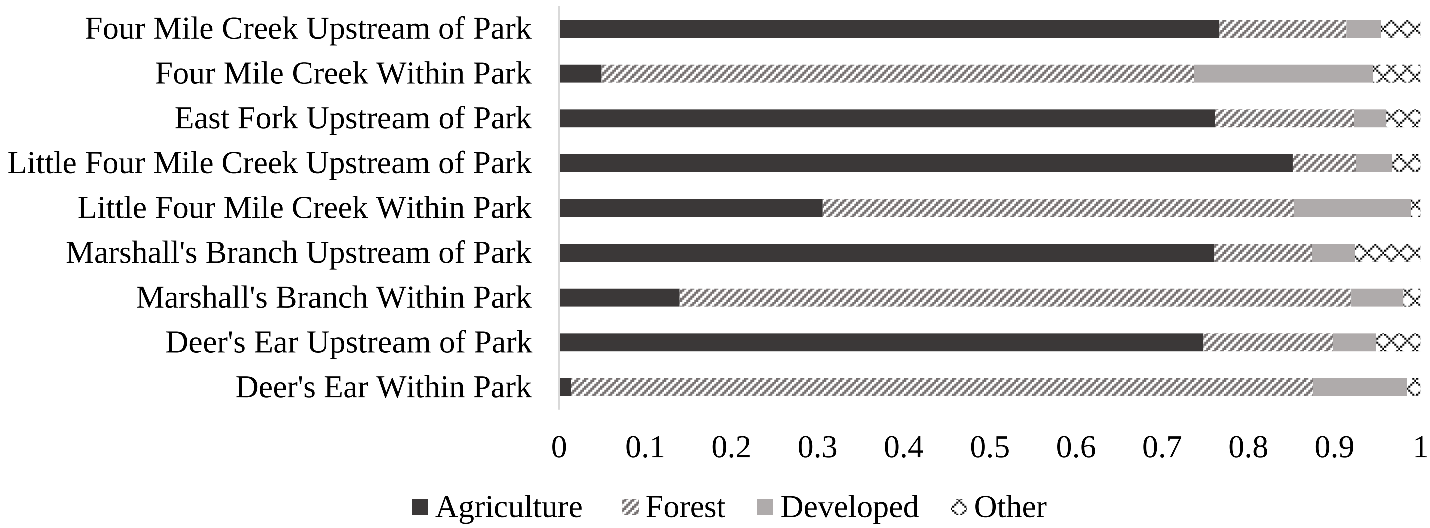


Figure 5: A 500-meter buffer zone was established around each of the creeks to quantify land cover. "Within the Park" refers to the buffer zone between the most downstream sampling point and the park boundary. "Upstream of Park" refers to the buffer area above the park to the top of the watershed. The graph shows the proportion of land use and land cover (LULC) that falls within each buffer zone for each creek in 2020. The LULC data was analyzed based on the United States Department of Agriculture (USDA) National Agricultural Statistics Service (NASS) Cropland Data Layer (CDL).

Table 1

Watershed Characteristics and Stream Morphology

Parameter	Four Mile Creek	Little Four Mile Creek	Marshall’s Branch	Deer’s Ear
Total Drainage Basin Area (km^2)	134.36	83.60	14.33	10.88
Total Pool Volume in Study Area (m^3)	1198.4	1033.5	908.27	347.12
Discharge (m^3/s)	0.51	0.277	0.047	0.051
Baseflow Residence Time (min)	39.54	64.09	323.41	113.19
Average Width (m)	18.36	17.48	12.25	9.09
Average Depth (m)	0.62	0.63	0.48	0.42
Average Channel Area (m^2)	11.43	11.06	6.08	4.20
Average Width: Depth	29.5	27.63	27.03	23.36
Median Creek Bed Particle Size				
D-16 (mm)	4.85	45	22.5	16
D-50 (mm)	22.5	128	60	60
D-84 (mm)	128	Bedrock	128	90

The total drainage basin area (km^2) was measured in ArcGIS Pro 2.7.2 following watershed delineation. The discharge (m^3/s) and pool volume (m^3) were measured in the field during baseflow conditions on 10/04/20. Residence time (RT) is a function of discharge (Q) and pool volume (V) and is calculated as RT=V/Q. The average channel width and depth were also measured in the field during base flow conditions on 12/07/2020. The average channel area (m^2) was calculated by multiplying each cross section width by average depth. The creek bed sediment was also surveyed on 10/04/20.

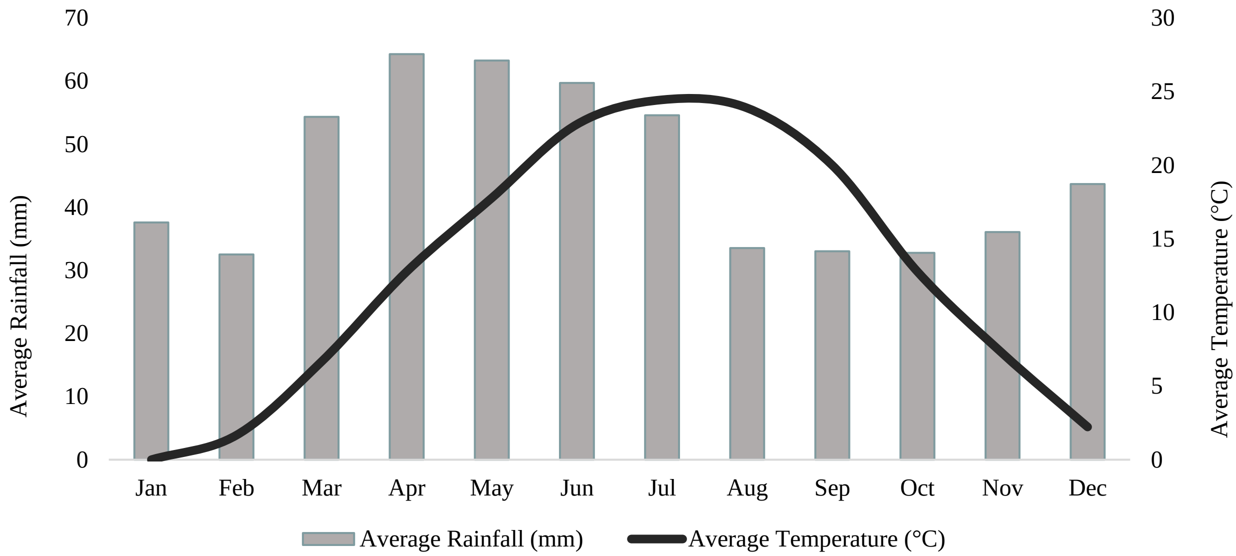


Figure 6: The climate graph for Oxford, Ohio with monthly averages of precipitation and temperature for 2019. Data was provided by the Miami University's Ecological Research Center.

Data Collection and Analysis

Water Sampling

The original study design, which was subsequently modified, included sampling points on farmland north of the park boundary. However, CDL data revealed a meaningful change in land cover from the farm sampling point and the park boundary sampling point (Appendix A). This change in land cover likely contributed to the difference in nutrient concentrations between the upstream farm sampling point and the park boundary sampling point (Appendix A). Therefore, the study was modified to examine changes in nutrient levels within the state park (rather than comparing the changes in nutrient concentrations inside of the state park to the changes outside of the state park).

For this study, water samples were collected every two weeks from December 2019-December 2020 with the exception of a pause from March 8th 2020-May 22nd 2020 due to the COVID-19 pandemic and from August 18th 2020-November 3rd 2020 due to regular seasonal drying. There were a total of eight creek surface sampling points (Figure 4), with two surface sampling points on each creek (Four Mile Creek, Little Four Mile Creek, Marshall's Branch, and Deer's Ear). The upstream sampling points were located at the boundary of Hueston Woods

State Park and the downstream sampling points were located where each creek flows into Acton Lake or where lake water backs up into a creek and thus flow ceases. Samples were collected in 4 L bottles and kept on ice until sampling was completed. A total of 168 samples per nutrient were collected across 21 sampling days throughout the study period.

Nutrient Analysis

Water samples were stored in a temperature-controlled room at 4°C until processing. All samples were processed within 48 hours of collection. The water samples were processed for TN, TP, NO_3^-, and PO_4^{3-}. From each water sample, one 125 mL sub-sample was filtered using a type a/e glass microfiber filter to isolate NO_3^- and PO_4^{3-} present in the sample. All samples were acidified with 188 μL of sulfuric acid for preservation until analysis. Nitrate was measured using the cadmium reduction method (QuikChem® 10-107-04-1-A), and PO_4^{3-} was measured using the molybdate blue method (QuikChem® 10-115-01-1-Q). An unfiltered TN and TP subsample was analyzed following a manual persulfate digestion to convert all the nitrogen and phosphorus into NO_3^- and PO_4^{3-} , respectively. The concentrations of the digested subsamples were then measured using the aforementioned methodology for NO_3^- and PO_4^{3-}. All nutrient analyses were completed using a Lachat QuikChem® 8500 Series II nutrient analyzer.

Hydrological and Morphological Data

Watershed drainage areas were delineated in ESRI ArcGIS Pro 2.7.2 using a 10 m resolution digital elevation model (DEM) and the hydrology toolset. Stream stage (m) was recorded using Onset HOBO® automatic water level loggers, which were installed on Four Mile Creek, Little Four Mile Creek, and Marshall's Branch. Stages were converted to discharges with previously created stage-discharge rating curves (Renwick et al., 2018). For statistical modelling, the stage-discharge value that corresponded with the time of sampling was utilized. The discharge in Deer's Ear which is ungauged was scaled by the relative watershed area of Marshall's Branch. Marshall's Branch and Deer's Ear are adjacent to one another and are similar in size and thus have similar discharge characteristics.

Total pool volume (m^3) was calculated within the park between the upstream and downstream sampling points for the four study creeks during baseflow by summing the volume of individual pools. Pool volume was calculated by multiplying the average length (m), width (m), and depth (m) of each pool within a creek. During the same day, discharge was measured on each creek to calculate residence time. Residence time (min) was calculated as pool volume

(m^3)/stream discharge (m^3/s). It is important to note that residence time will vary within each creek under various flow conditions. However, relative residence time will be consistent between streams. Streams were at a moderate baseflow when residence time was calculated for this study. Discharge was measured in the field using the velocity-area method. For the velocity-area method, velocity and area were measured for at least ten subsections within each creek using a Flow Tracker 2 Acoustic Doppler Velocimeter (ADV®) (Herschy, 1993). Basic morphological measurements were also taken at cross sections on each creek between sampling points. On Little Four Mile Creek, Marshall's Branch, and Deer's Ear, a minimum of ten equidistant cross sections were established, while four cross sections were established on Four Mile Creek due to the shorter distance between upstream and downstream sampling sites and consistent channel morphology. Bank-full width (m) was measured from the top of the lower bank to the equivalent elevation on the opposite bank. Depth (m) measurements were taken along the bank-full width and averaged. The W:D was calculated by dividing the channel width by the average depth. The average channel area (m) was calculated by multiplying the width by the average depth for each cross section. All width, depth, W:D, and channel area measurements were then averaged by each creek (Harrelson et al., 1994; Grudzinski et al., 2018). A Wolman pebble count was completed using a gravelometer to compare creek bed particle size among the different creeks (Wolman, 1954). One hundred substrate samples were measured (mm along the b-axis) at each cross section within the four creeks. The median D-16 (16^{th} percentile), D-50 (median), and D-84 (84^{th} percentile) particle sizes (mm) for each creek were calculated following the Wolman pebble count (Olson et al., 2005; Grudzinski et al., 2018).

Remote Sensing

Vegetation conditions were quantified using the normalized difference vegetation index (NDVI) derived from Sentinel-2 imagery. Image processing was completed within the Google Earth Engine (GEE) cloud computing platform (Gorelick et al., 2017). The NDVI was calculated using red and near-infrared (NIR) wavelengths: (NIR – Red) / (NIR + Red) (Reed et al., 1994). Ten-day composites surrounding water sampling dates (five before and five after) were used to determine vegetation conditions during sampling events and to limit effects of cloud coverage (Appendix A). One-hundred-meter buffer zones were created around the creeks in QGIS v. 3.16.1 (QGIS Development Team, 2009; Figure 7). Mean NDVI was calculated within the buffer zones between the boundary sampling point and the most downstream in-park sampling point using the zonal statistics plugin within QGIS v. 3.16.1.

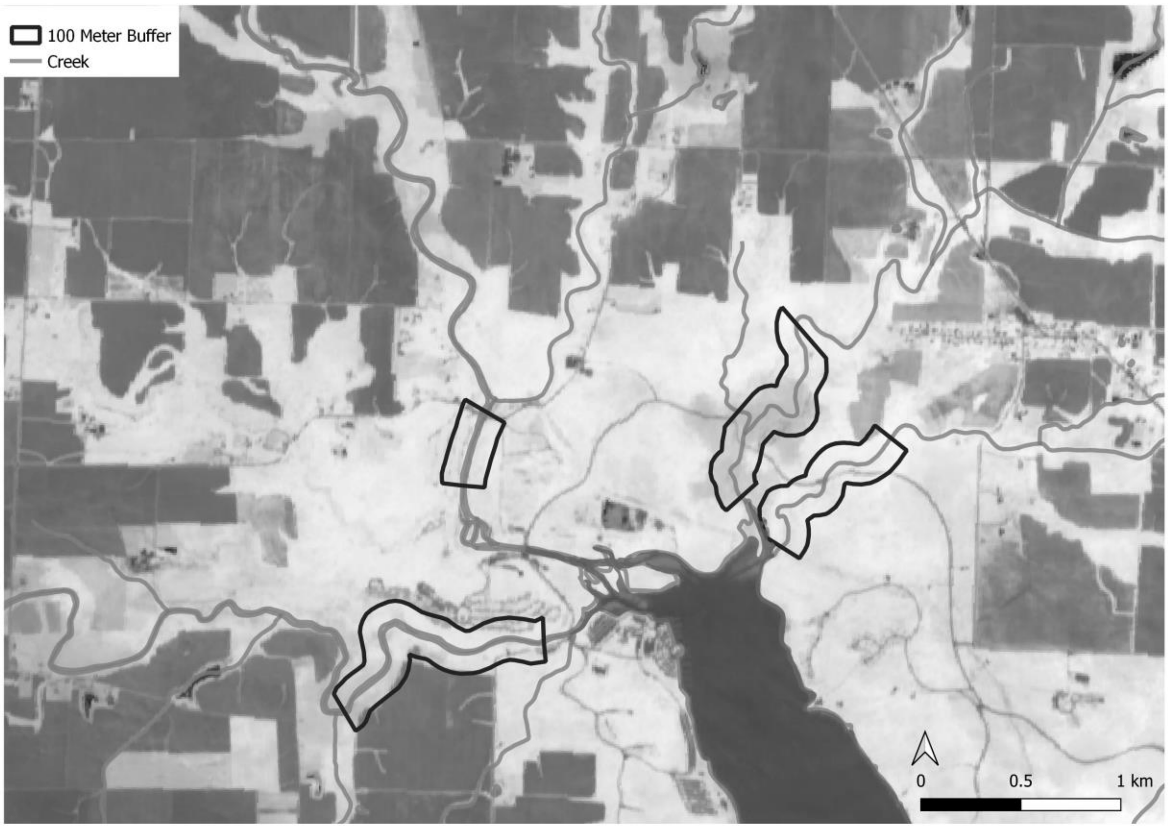


Figure 7: The map shows the 100-meter width buffers established between the downstream park sampling point and the park boundary. The mean NDVI was calculated within each buffer using the zonal statistics plugin in QGIS v. 3.16.1.

Data Analysis

All statistical modeling was completed in R Studio v. 3.6.2. A pre-post analysis of covariance test (ANCOVA) was used to model the nutrient reduction within the park (i.e., park effect). In the model, the "post" nutrient concentration is the downstream sample within the park, which is a function of the "pre" nutrient concentration sampled at the park boundary where each creek enters the park from farmland. The effects of discharge and NDVI are also incorporated into the model. A pre-post ANCOVA is recommended by the United States Environmental Protection Agency (USEPA) to assess changes in water quality as the result of a treatment in a paired-watershed study design (Dressing et al., 2016). Numerous studies have utilized this approach to model upstream versus downstream water quality changes (e.g., Grabow et al., 1999; Bishop et al., 2005; Uwimana et al., 2017).

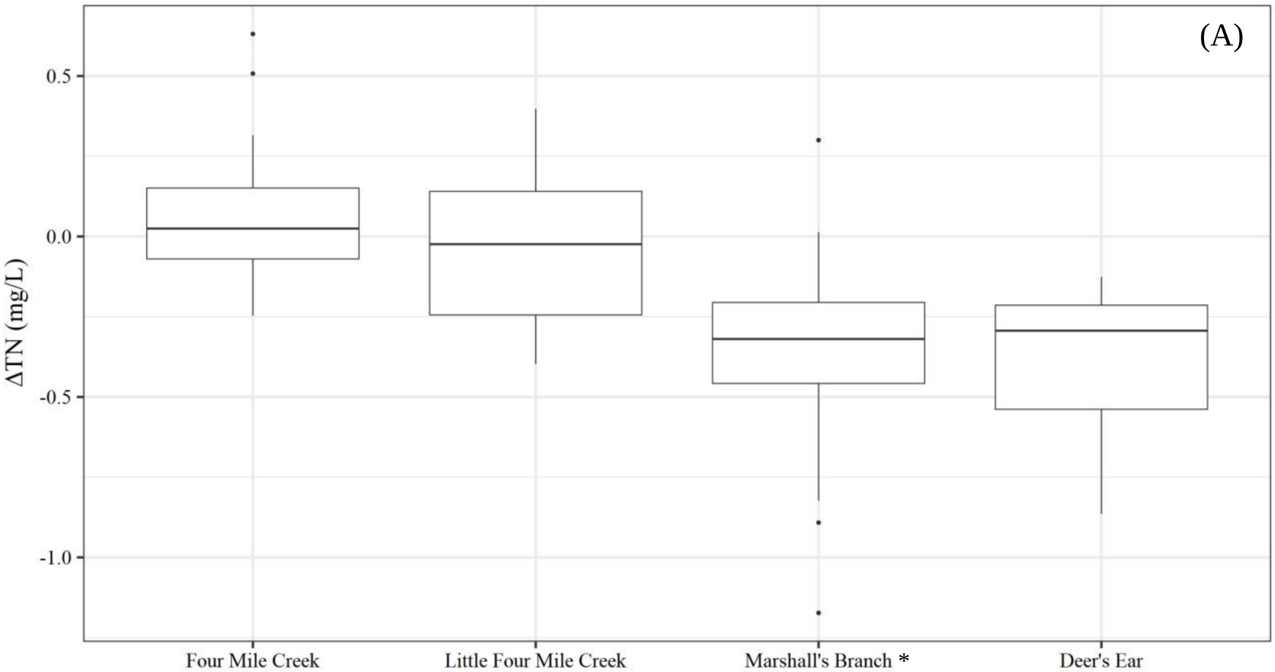
To model the global park effect in this study, the change in concentrations was independent of the different creeks, which were treated as a block term in this model. To assess the impact of the forested state park on nutrient concentrations, a regression coefficient t-test was used to determine if the nutrient concentrations at each downstream sampling point were significantly less than the upstream sample concentrations (Cressie & Whitford, 1986). Akaike Information Criterion (AIC), Bayesian Information Criterion (BIC), and adjusted R^2 values were used to build a more parsimonious model (Burnham & Anderson, 2004; Johnson & Omland, 2004). Confidence intervals were based on the chosen model to examine the influence of the park, discharge, and NDVI within each creek (Bishop et al., 2005; Nielsen et al., 2012; Chaffin et al., 2018). A Bonferroni correction was used to mitigate error from building multiple intervals (Armstrong, 2014). Channel morphology characteristics were not included in the statistical models as the number of creeks would not allow for robust analyses on the impact of these variables.

V. Results

1) Total Nitrogen

The ANCOVA and regression coefficient t-test suggests there was not an overall park effect for TN ($p=0.1177$). However, the tests indicated that the impact of the park, the impact of discharge, and the impact of NDVI on TN concentrations varied by creek ($p<0.001$, $p<0.001$, and $p<0.01$, respectively; Table 2).

The confidence intervals derived from the model showed that the park significantly reduced TN concentrations within Marshall's Branch (CI: 0.779, 0.956, Figure 8). The park did not have a significant impact on TN concentrations within Four Mile Creek, Little Four Mile Creek, or Deer's Ear. Seasonal trends varied across sites, but the greatest reduction in TN was observed in the summer within all the creeks (Figure 8). Within the model, an increase in discharge resulted in a decrease in TN reduction within Marshall's Branch (CI: 1.209, 5.252; Figure 9). There were no statistically significant relationships between discharge and TN reductions within Four Mile Creek, Little Four Mile Creek, or Deer's Ear. An increase in NDVI was associated with increased TN reduction within Deer's Ear (CI: -1.302, -0.051; Figure 10). No significant relationship was detected between NDVI and TN within Four Mile Creek, Little Four Mile Creek, or Marshall's Branch.



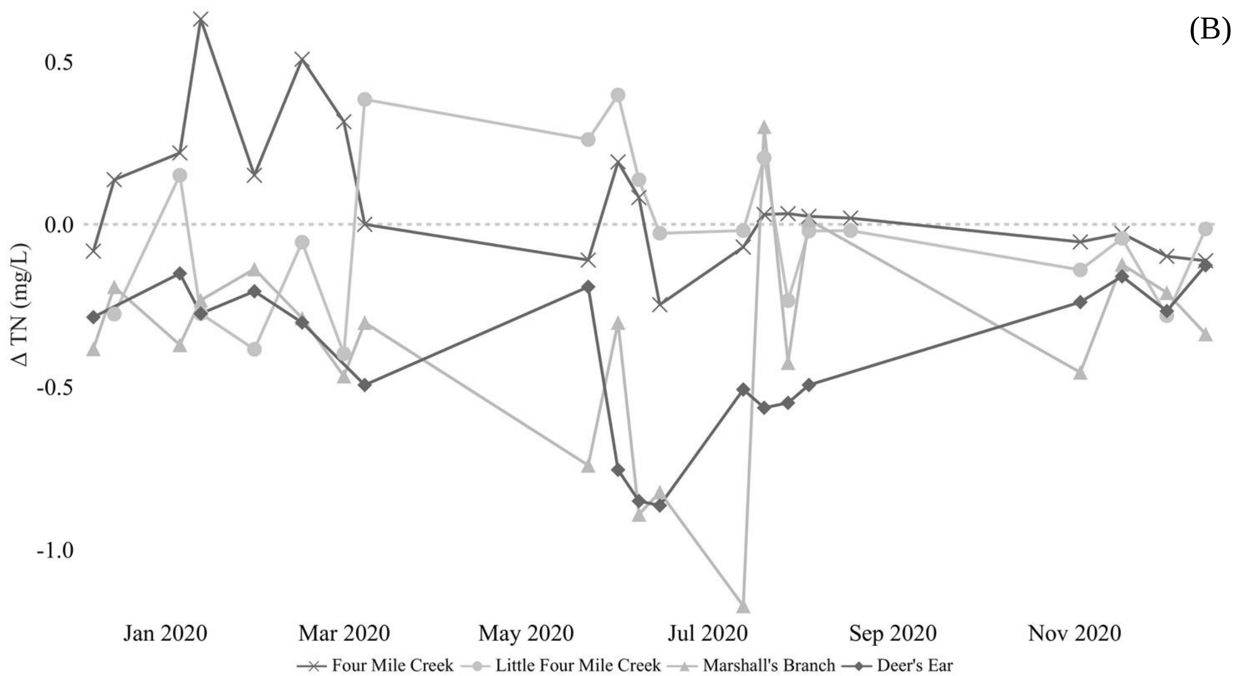


Figure 8: Changes in total nitrogen concentration (mg/L) between study creeks (top, a) and through time (bottom, b).

** Denotes significant reduction in total nitrogen concentrations within creek.*

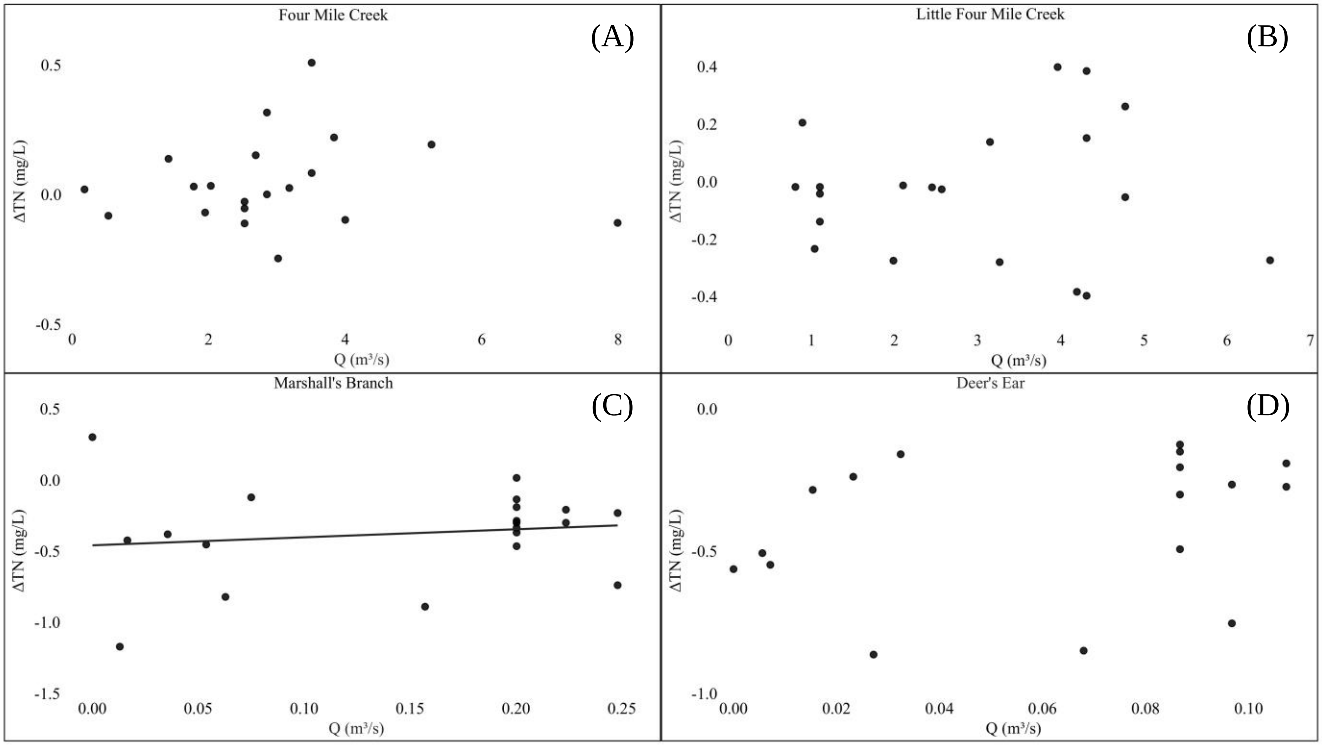


Figure 9: Relationship between the change in total nitrogen concentration (mg/L) and stream flow (m^3/s) within each study creek.

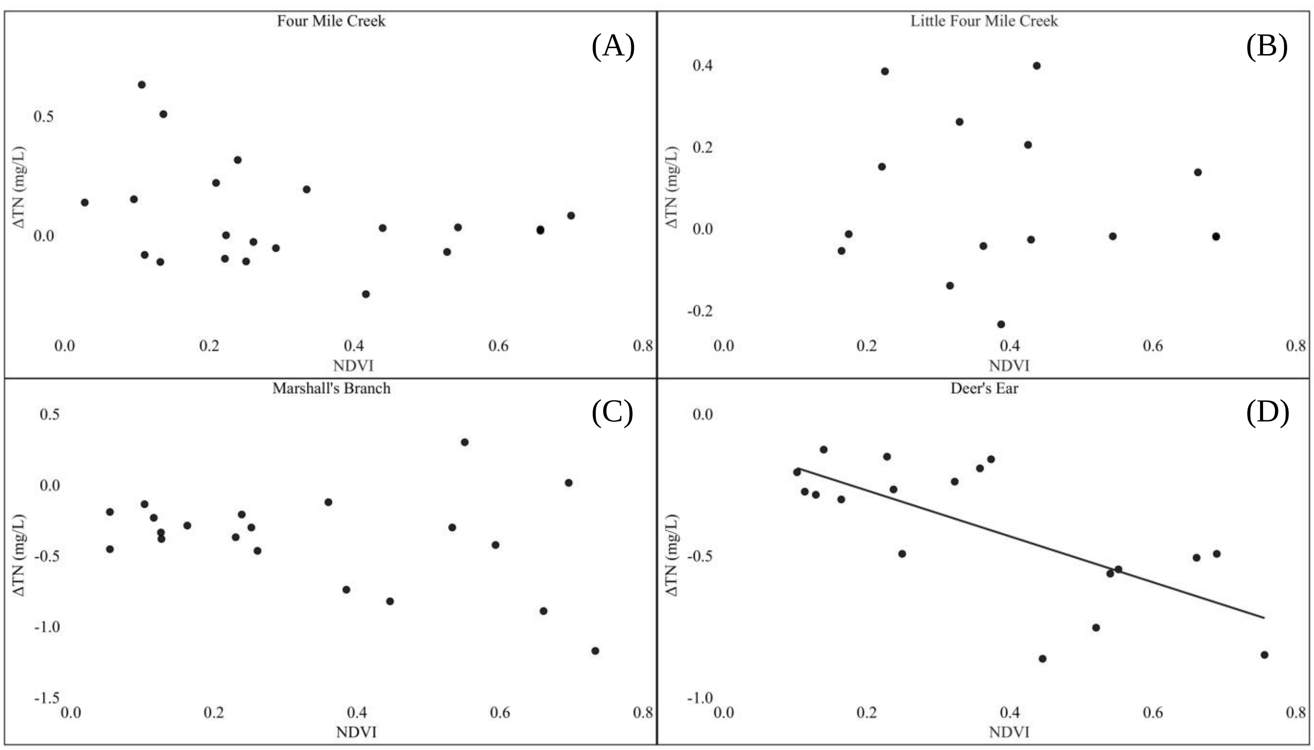


Figure 10: Relationship between the change in total nitrogen concentration (mg/L) and ten-day composite mean NDVI (Appendix A) within each study creek.

Table 2

Type III Sum of Squares

TN		Sum Sq	Df	F	P
	Boundary	**23.0432**	**1**	**635.2527**	**< 2e-16**
	Discharge	**0.6135**	**1**	**16.9132**	**0.0001**
	NDVI	0.1301	1	3.5868	0.0630
	Creek	0.2216	3	2.0360	0.1182
	Boundary: Creek	**0.8496**	**3**	**7.8070**	**0.0002**
	Discharge: Creek	**0.8297**	**3**	**7.6239**	**0.0002**
	NDVI: Creek	**0.4535**	**3**	**4.1669**	**0.0095**
	Residuals	2.2127	61		
NO_3^-					
	Boundary	**24.9682**	**1**	**1248.0967**	**< 2e-16**
	Discharge	**0.3374**	**1**	**16.8671**	**0.0001**
	NDVI	**0.0903**	**1**	**4.5136**	**0.0379**
	Creek	0.0870	3	1.4492	0.2379
	Boundary: Creek	**0.7582**	**3**	**12.6331**	**1.841e-06**
	Discharge: Creek	**0.3720**	**3**	**6.1986**	**0.001**
	NDVI: Creek	0.0686	3	1.1438	0.3391
	Residuals	1.1603	58		
TP					
	Boundary	**0.02686**	**1**	**101.0067**	**1.472e-14**
	Discharge	**0.0055**	**1**	**20.6541**	**2.659e-05**
	NDVI	0.0004	1	1.5939	0.2116
	Creek	0.0010	3	1.2895	0.2861
	Boundary: Creek	**0.0029**	**3**	**3.6656**	**0.0170**
	Discharge: Creek	**0.0070**	**3**	**8.7519**	**6.497e-05**
	NDVI: Creek	0.0005	3	0.6614	0.5790
	Residuals	0.0162	61		
PO_4^{3-}					
	Boundary	**11194.8**	**1**	**133.6301**	**< 2e-16**
	Discharge	**4497.2**	**1**	**53.6815**	**7.562e-10**
	NDVI	162.9	1	1.9448	0.1684
	Creek	650.0	3	2.5864	0.0615
	Boundary: Creek	**2573.5**	**3**	**10.2399**	**1.596e-05**
	Discharge: Creek	**5201.1**	**3**	**20.6947**	**2.776e-09**
	NDVI: Creek	48.5	3	0.1931	0.9007
	Residuals	4942.7	59		

2) Nitrate

The ANCOVA and regression coefficient t-test suggests there was an overall park effect for NO_3^- ($p<0.01$). The ANCOVA model indicated that the influence of the park and the influence of discharge on NO_3^- concentrations varied by creek ($p<0.01$, $p<0.01$, respectively; Table 2), while NDVI was found to be influential across all the creeks within the study area ($p=0.0379$; Table 2).

The park significantly reduced NO_3^- within Marshall’s Branch and Deer’s Ear (CI: 0.786, 0.910; 0.831, 0.991, respectively; Figure 11). There was not a significant reduction from the park shown within Four Mile Creek or Little Four Mile Creek. Seasonal trends varied across sites, but the greatest reduction in NO_3^- was observed in the summer within all creeks (Figure 11). Within the model, an increase in discharge resulted in a decrease in NO_3^- reduction within Marshall’s Branch (CI: 0.905, 3.957; Figure 12). There was no statistically significant relationship detected between discharge and NO_3^- reduction within Four Mile Creek, Little Four Mile Creek, or Deer’s Ear. An increase in NDVI resulted in a reduction in NO_3^- concentrations across all sites (CI: -0.423, -0.066; Figure 13).

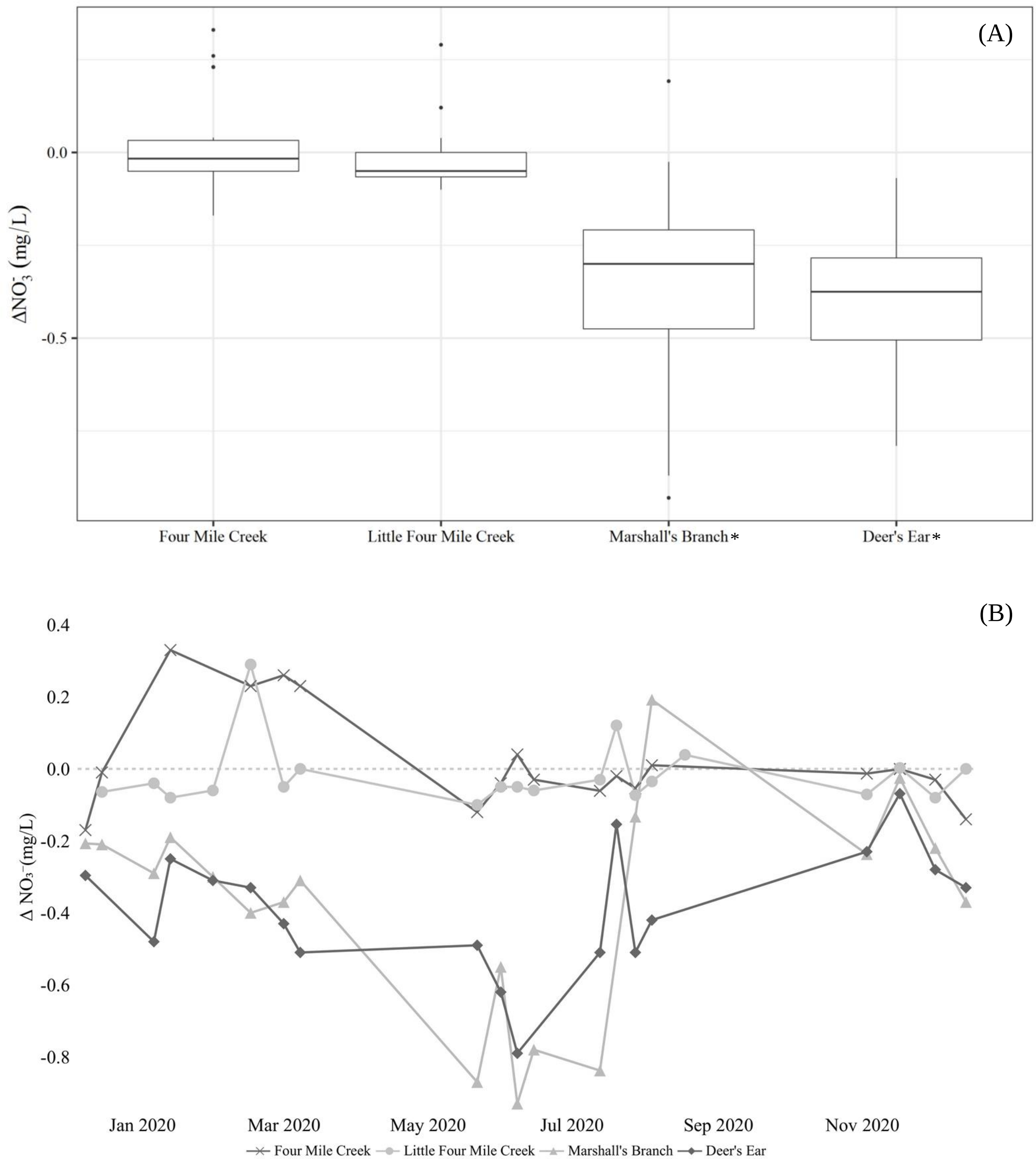


Figure 11: Changes in nitrate concentration (mg/L) between study creeks (top, a) and through time (bottom, b).

** Denotes significant reduction in nitrate concentration within creek.*

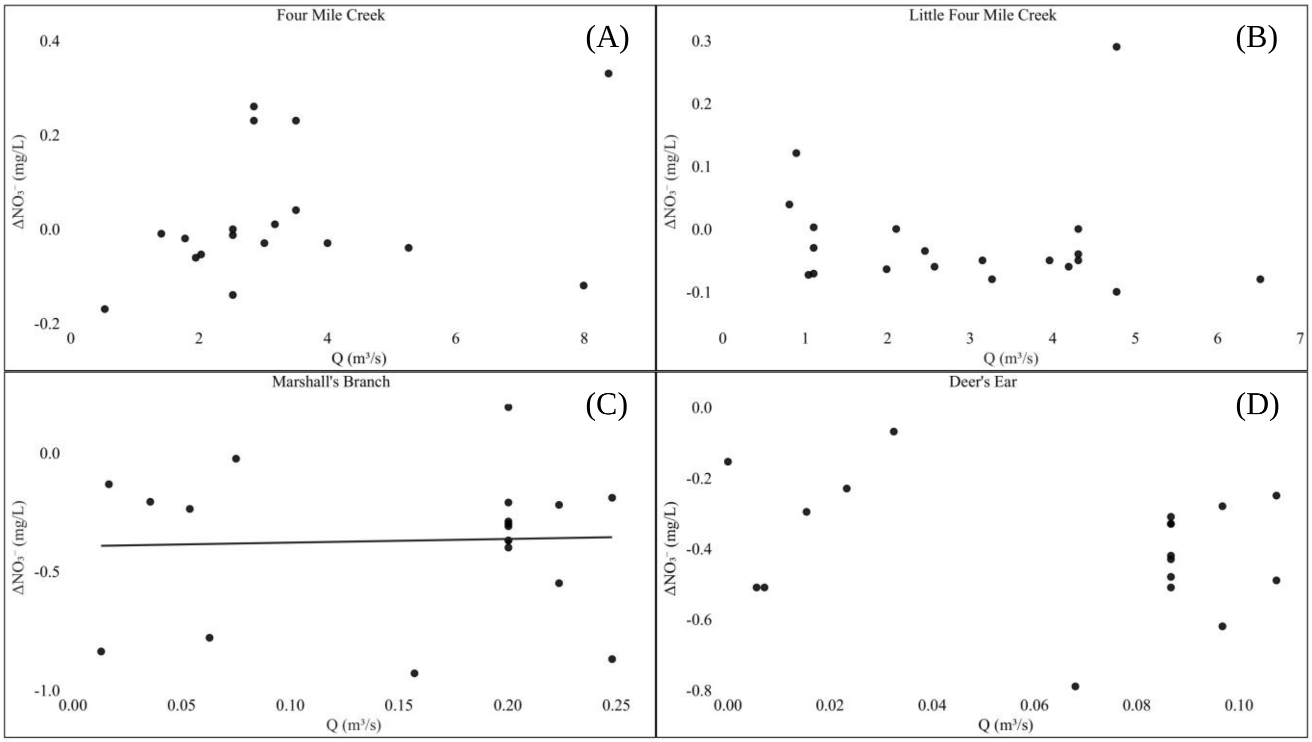


Figure 12: Relationship between the change in nitrate concentration (mg/L) and stream flow (m^3/s) within each study creek (a-d).

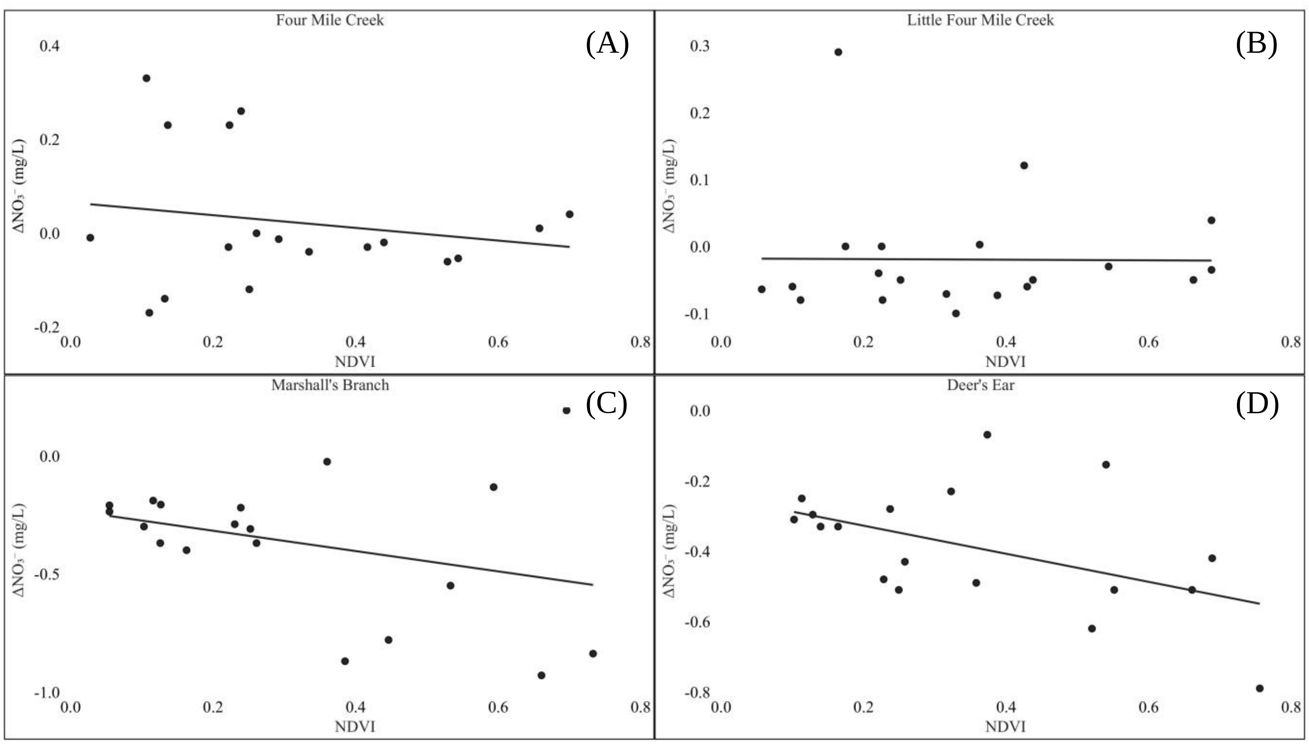


Figure 13: Relationship between the change in nitrate concentration (mg/L) and ten-day composite mean NDVI (Appendix A) within each study creek (a-d).

3) Total Phosphorus

The ANCOVA and regression coefficient t-test suggests there was an overall park effect for TP ($p=0.0146$). The results of the ANCOVA indicated that the impact of the park and the impact of discharge on TP concentrations varied by creek ($p=0.0170$, $p<0.001$, respectively; Table 2).

The confidence intervals derived from the model showed that the park significantly reduced TP concentrations within Marshall's Branch (CI: 0.544, 0.919; Figure 14). Seasonal trends varied across sites, but the greatest reduction in TP within the smallest creeks was observed in late summer. There was not a clear seasonal pattern in TP reduction associated with the larger creeks (Figure 14). Within the model, an increase in discharge resulted in a decrease in TP reduction within Marshall's Branch (CI: 0.096, 0.348; Figure 15). There were no statistically significant relationships detected between TP reduction and discharge within Four Mile Creek, Little Four Mile Creek, or Deer's Ear. There was also no significant relationship detected between NDVI and TP reduction in any of the creeks (Figure 16).

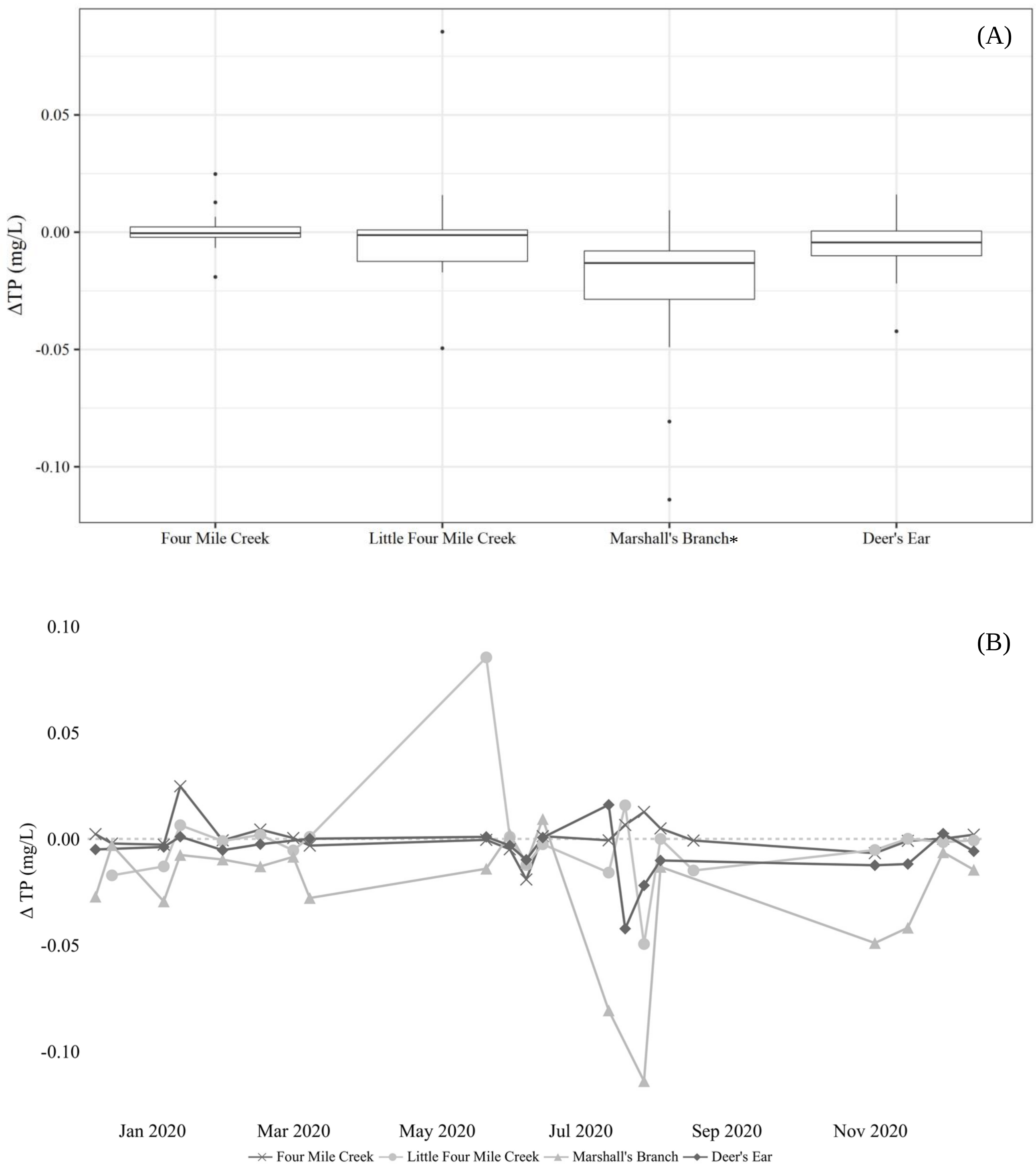


Figure 14: Changes in total phosphorus concentration (mg/L) between study creeks (top, a) and through time (bottom, b).

** Denotes significant reduction in total phosphorus concentrations within creek.*

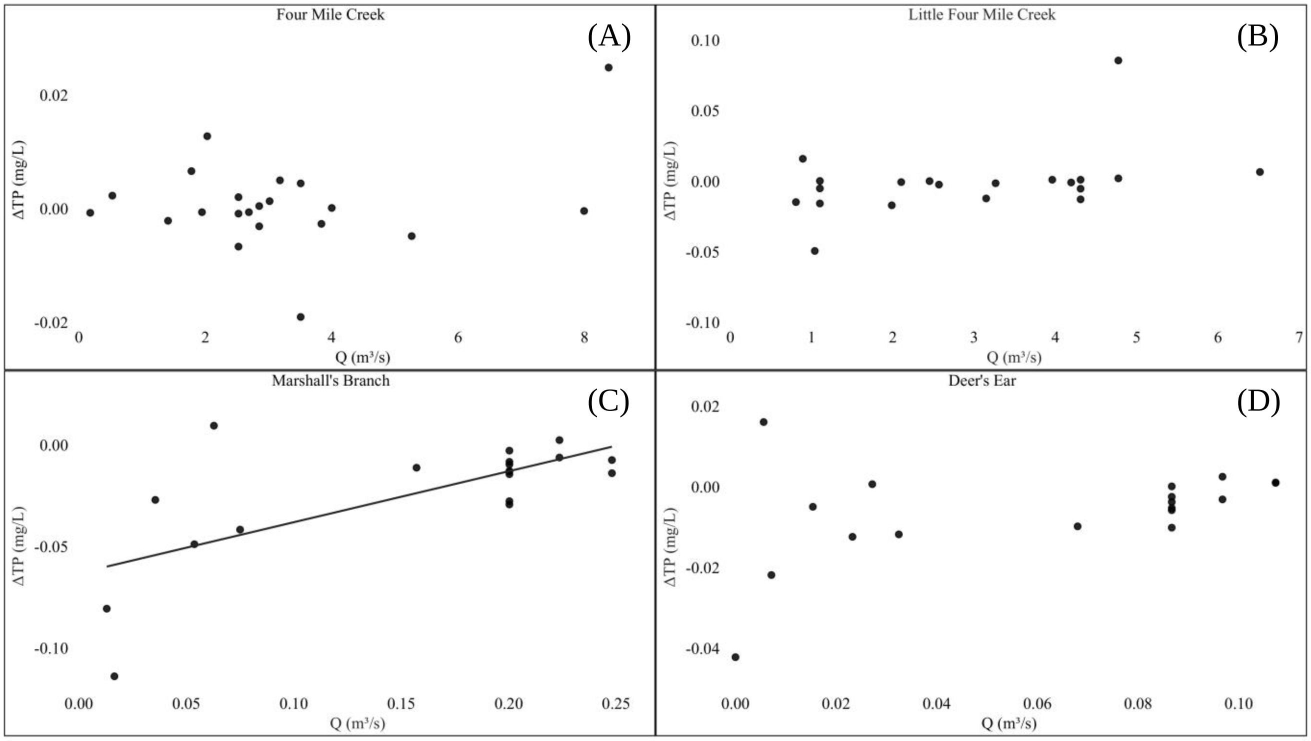


Figure 15: Relationship between the change in total phosphorus concentration (mg/L) and stream flow (m^3/s) within each study creek (a-d).

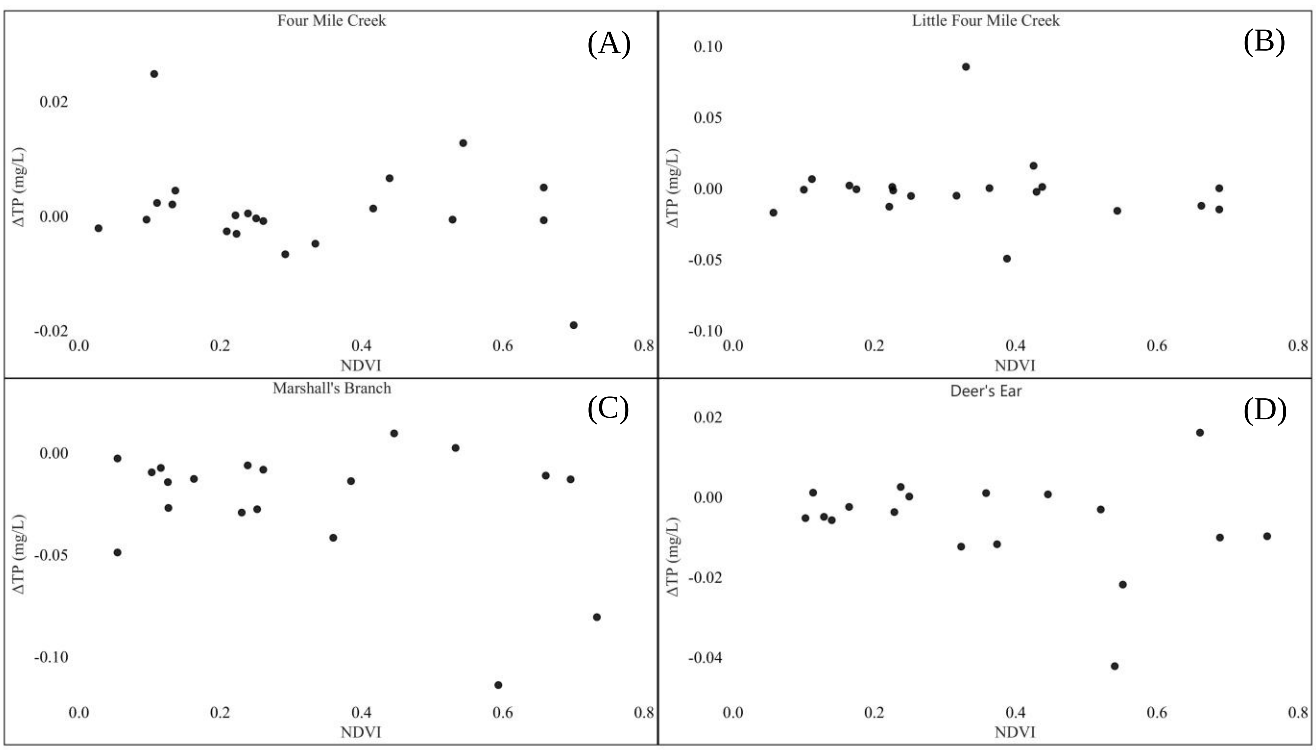
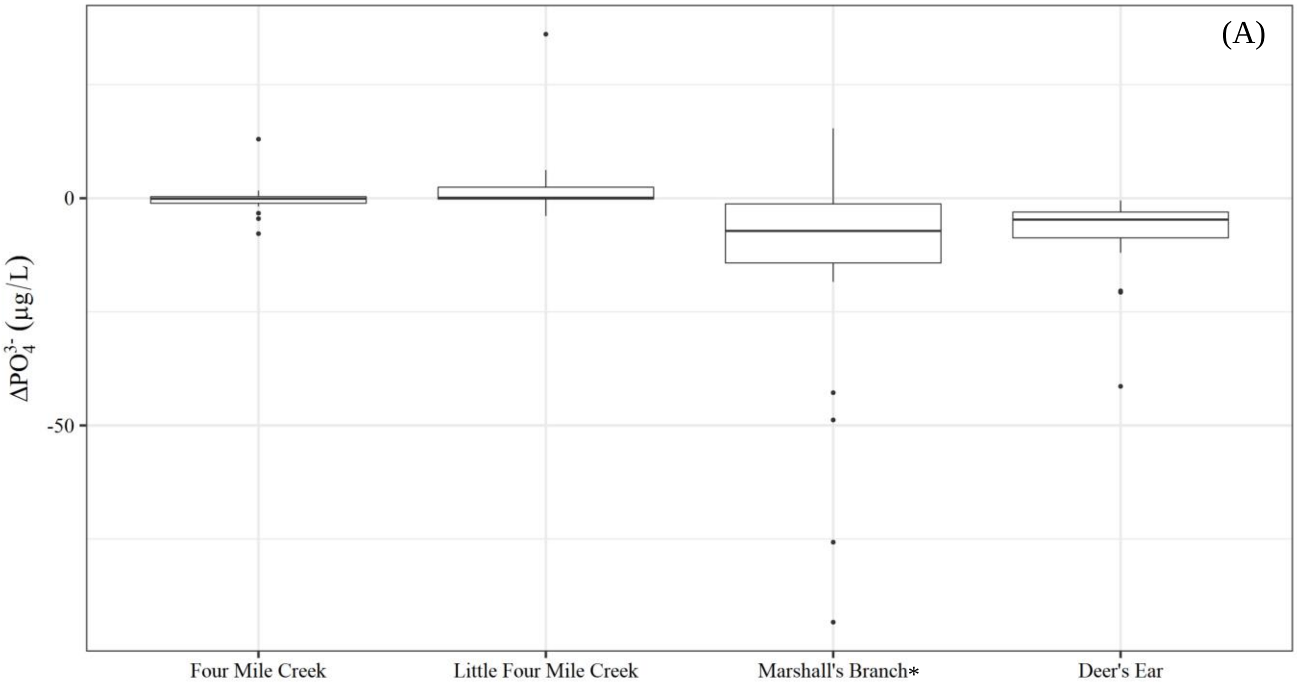


Figure 16: Relationship between the change in total phosphorus concentration (mg/L) and ten-day composite mean NDVI (Appendix A) within each study creek (a-d).

4) Orthophosphate

The ANCOVA and regression coefficient t-test suggests there was an overall park effect for PO_4^{3-} ($p<0.001$). The results of the ANCOVA indicated that the influence of the park and the influence of discharge on PO_4^{3-} concentrations varied by creek ($p<0.001$, $p<0.001$; Table 2).

The confidence intervals derived from the model showed that the park significantly reduced PO_4^{3-} concentrations within Marshall's Branch (CI: 0.479, 0.754; Figure 17). Seasonal trends varied across sites, but the greatest reduction in PO_4^{3-} within the smallest creeks was observed in the late summer. There was not a clear seasonal pattern in PO_4^{3-} reduction associated with the larger creeks (Figure 17). Within the model, an increase in discharge resulted in a decrease in PO_4^{3-} reduction within Marshall's Branch and Deer's Ear (CI: 132.241, 275.701; 2.049, 335.426, respectively; Figure 18). There were no statistically significant relationships detected between discharge and PO_4^{3-} reduction within Four Mile Creek or Little Four Mile Creek. There was also no significant relationship detected between NDVI and PO_4^{3-} reduction in any of the creeks (Figure 19).



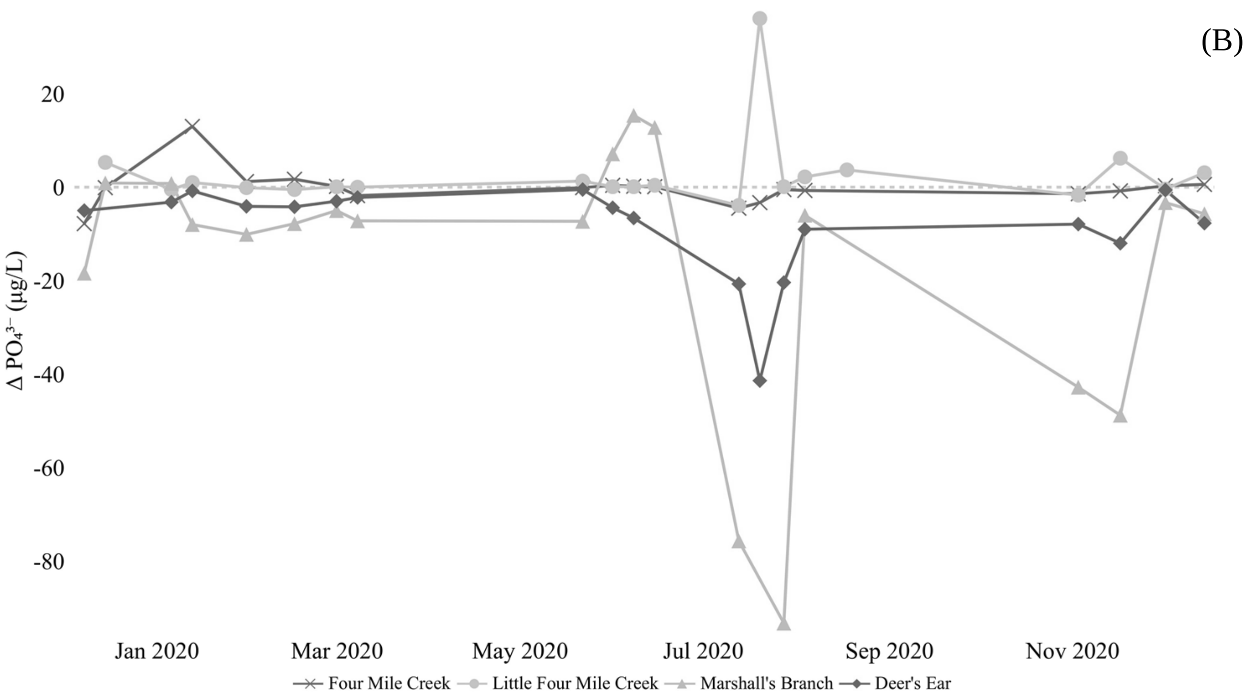


Figure 17: Changes in orthophosphate concentration (µg/L) between study creeks (top, a) and through time (bottom, b).
** Denotes significant reduction in orthophosphate concentrations within creek.*

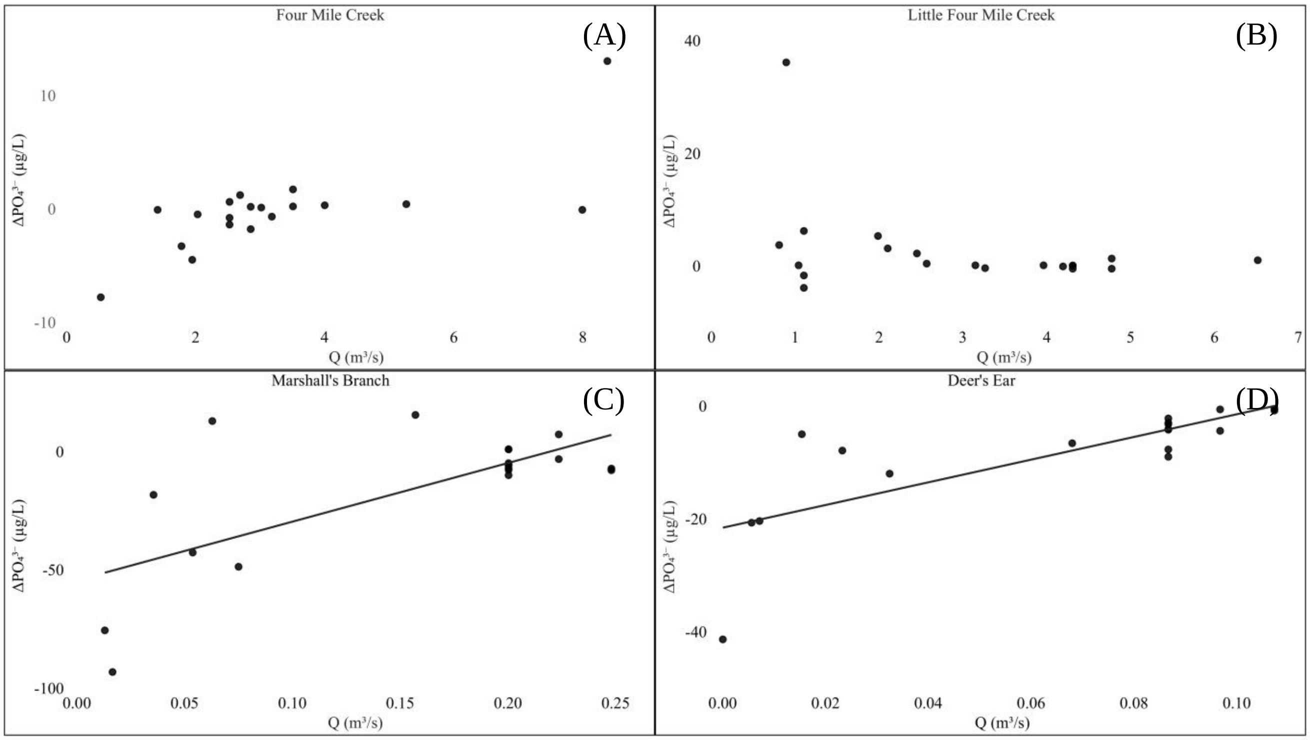


Figure 18: Relationship between the change in orthophosphate concentration (µg/L) and stream flow (m^3/s) within each study creek (a-d).

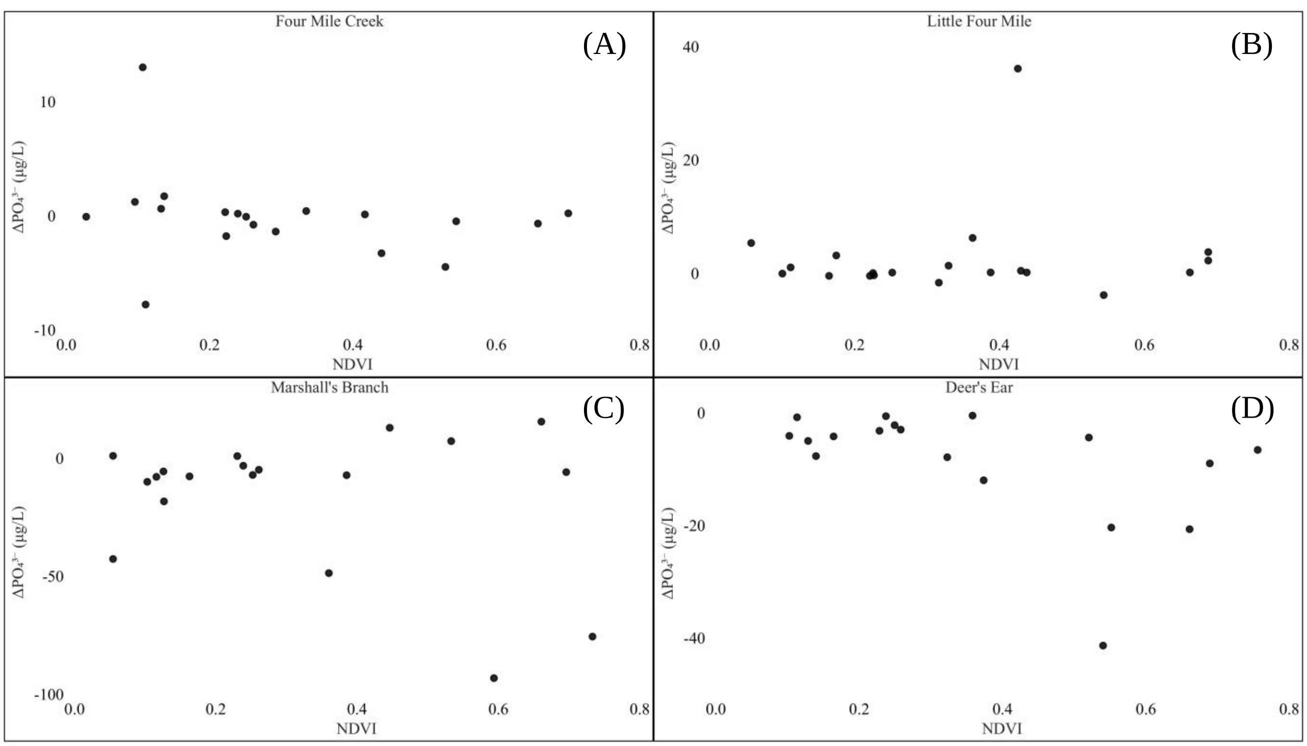


Figure 19: Relationship between the change in orthophosphate concentration (µg/L) and ten-day composite mean NDVI (Appendix A) within each study creek (a-d).

VI. Discussion

The findings of this study demonstrated that a forested state park can improve water quality by reducing nutrient concentrations within agriculturally dominated U.S. Midwestern watersheds. These findings are consistent with other studies that have shown reductions in nutrients from riparian buffers within agricultural watersheds (Peterjohn et al., 1984; Lowrance et al., 1984). The statistical models, which examined the influence of the park, were primarily driven by reductions in TN, NO_3^-, TP, and PO_4^{3-} within the two smallest creeks (Marshall's Branch and Deer's Ear) in the study area. The nutrient reductions resulting from the forested state park within the smaller creeks are likely related to morphological and hydrological features of these creeks. Smaller streams exhibit a smaller water volume to creek bed surface area ratio compared to larger creeks, which increases the potential for benthic exchange and nutrient removal (Alexander et al., 2000; Bernot & Dodds, 2005; Withers & Jarvie, 2008). Additionally, as stream size increases, denitrification has been shown to be a less significant mechanism for reductions in N loads (Bernot & Dodds, 2005). Furthermore, Marshall's Branch and Deer's Ear have the highest residence times which provides even more opportunity for nutrients to settle or be taken up by biota (Hill et al., 1998; Meals et al., 1999; Opdyke et al., 2006; Withers & Jarvie, 2008).

It was expected that TN and NO_3^- would be influenced by seasonal changes in discharge while PO_4^{3-} and TP were expected to be driven by large storm events (Wallbrink et al., 2003; Kalkhoff et al., 2016; Covino, 2017; Lazar et al., 2019; Kelly et al., 2019). Within the model, the interaction between discharge and the individual creeks was significant for all nutrient parameters (TN, NO_3^-, TP, and PO_4^{3-}). Specifically, discharge was predictive of changes in all nutrient concentrations within Marshall's Branch and PO_4^{3-} concentrations within Deer's Ear. While discharge was a significant indicator of the changes in nutrient concentrations within the smaller creeks, discharge was not predictive of the changes in nutrient concentrations within the larger creeks, Four Mile Creek and Little Four Mile Creek. The sampling reach within the larger creeks represented a smaller portion of the overall stream length compared to the smaller creeks, which may limit the influence of discharge at this scale. For example, an increase in discharge and water velocity can reduce residence time and therefore increase the nutrient spiraling length or the average distance traveled before uptake (Doyle, 2005; Ensign & Doyle, 2006; Covino, 2017).

Recent studies have shown that climate change and altered hydrological regimes can also impact nutrient limitation in aquatic ecosystems (Hayes et al., 2015; Williamson et al., 2020). According to the National Oceanic and Atmospheric Administration (NOAA)'s Advanced Hydrological Predictive Service (AHPS), 2019 and 2020 were relatively wet years for the study area, which could also have influenced the impact of discharge on nutrient concentrations. Furthermore, legacy nutrient deposits within the watershed can complicate the relationship between discharge and nutrient concentrations by releasing buried nutrients into the water column during high flow events (Van Meter et al., 2018; Weigelhofer, 2018; Kreiling et al., 2019; Kusmer et al., 2019).

It was expected that NDVI would be associated with the reduction in all nutrient concentrations due to increased nutrient uptake by vegetation, particularly during summer months. NDVI was expected to be predictive of changes in PO_4^{3-} concentrations because the nutrient is readily usable by plants (Reid & Hayes, 2003; Mullen, 2005; Weigelhofer et al., 2018). However, a significant global NDVI effect was only observed for NO_3^-. There was not a global NDVI effect for any of the other nutrient parameters. When considering the influence of the individual creeks, NDVI was found to be predictive of TN concentrations within Deer's Ear given the influence of the park boundary and discharge. The changes in vegetation greenness throughout different seasons in the study area were quantified using NDVI, but the index alone is not fully representative of nutrient uptake by vegetation due to the numerous influences that impact uptake success. There are complex relationships with phosphorus and nitrogen limitation in aquatic ecosystems that examining was outside the scope of this study. For example, algal and fish communities can have a significant influence on nutrient concentrations in freshwater environments (Hamilton et al., 2001; Dodds et al., 2002; Bernot et al., 2006; Williamson et al., 2018; Andersen et al., 2019). Additionally, vegetation can also reach saturation points, which reduces nutrient uptake capacity (Bernot et al., 2006; Schade et al, 2011; Finkler et al., 2018). Remotely sensed data collection was also limited to a one-year time frame. Due to the COVID-19 pandemic, there was a gap in water sample collection and NDVI data from March-May, a period when NDVI experiences significant changes within the study area, which could have masked the impact of NDVI.

This study helps to fill a research gap pertaining to the influence of a forested state park on nutrient concentrations within an agriculturally dominated watershed, while taking anthropogenic and natural drivers of water quality into consideration. The results of the study demonstrate the importance of natural area protection and where preservation efforts could be prioritized. The findings of this study suggest that a state park can be effective at reducing nutrient concentrations in fluvial environments that drain agricultural land. Small order creeks with a high residence time and a large portion of watershed area within the park boundary showed the greatest improvements in water quality. A larger protected area may be needed to allow for reductions in nutrient concentrations to occur within larger creeks as the results of the study did not show the park effectively reducing concentrations in TN, NO_3^-, TP, or PO_4^{3-} within Four Mile Creek or Little Four Mile Creek.

Long-term studies could provide deeper insight into trends associated with nutrient concentrations, discharge, and NDVI that may have been limited by the one-year study period and gap in data. Future studies could also investigate the change in nutrient concentrations once water drains out of the state park and back through farmland to determine how far downstream the observed nutrient reductions are retained. Furthermore, it is not currently understood how Acton Lake, where the creeks drain into, influences changes in nutrient concentrations relative to changes that occur within the creeks. The results of this study could also vary within different regions of the United States that have adopted different agricultural practices and exhibit different climates, vegetation patterns, and land use histories.

VII. Conclusion

A forested state park was effective in reducing nutrient concentrations within the agriculturally dominated study area. The results were primarily driven by nutrient reductions within the smaller creeks in the study area. Discharge was a driving factor for changes in nutrient concentrations in some creeks. The larger residence times in smaller creeks appear to be beneficial at the scale of this study. There was limited evidence in the study to suggest that NDVI was correlated with a reduction in nutrient concentrations, however longer-term data collection may reveal unique trends not detected within this study. Based on the results of this study, conservation of forested areas within agriculturally dominated watersheds can provide improved water quality in degraded landscapes. Stream characteristics related to channel morphology and hydrology should be considered when planning forest conservation if the objective is to improve local water quality.

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Appendix A

The original study design consisted of three surface sampling points on each creek (Four Mile Creek, Little Four Mile Creek, Marshall's Branch, and Deer's Ear). Two of the sampling points were within the Hueston Wood State Park boundary and the third sampling was outside of the boundary on agricultural land. Specifically, one was located at the point where each creek flows into Acton Lake, another was at the park boundary with farmland, and the last point was on farmland upstream of the park (Figure 20).

The LCLU analysis showed that the farm sampling point buffer zone contained more forest cover than further upstream in the watershed (Figure 21), which could have contributed to the observed decreases in nutrient concentrations from the farm to park boundary (Figures 22, Figure 23, Figure 24, Figure -25). For these reasons, the statistical analysis was focused on the changes from the park boundary to the downstream sampling point within the park prior to entering the lake.

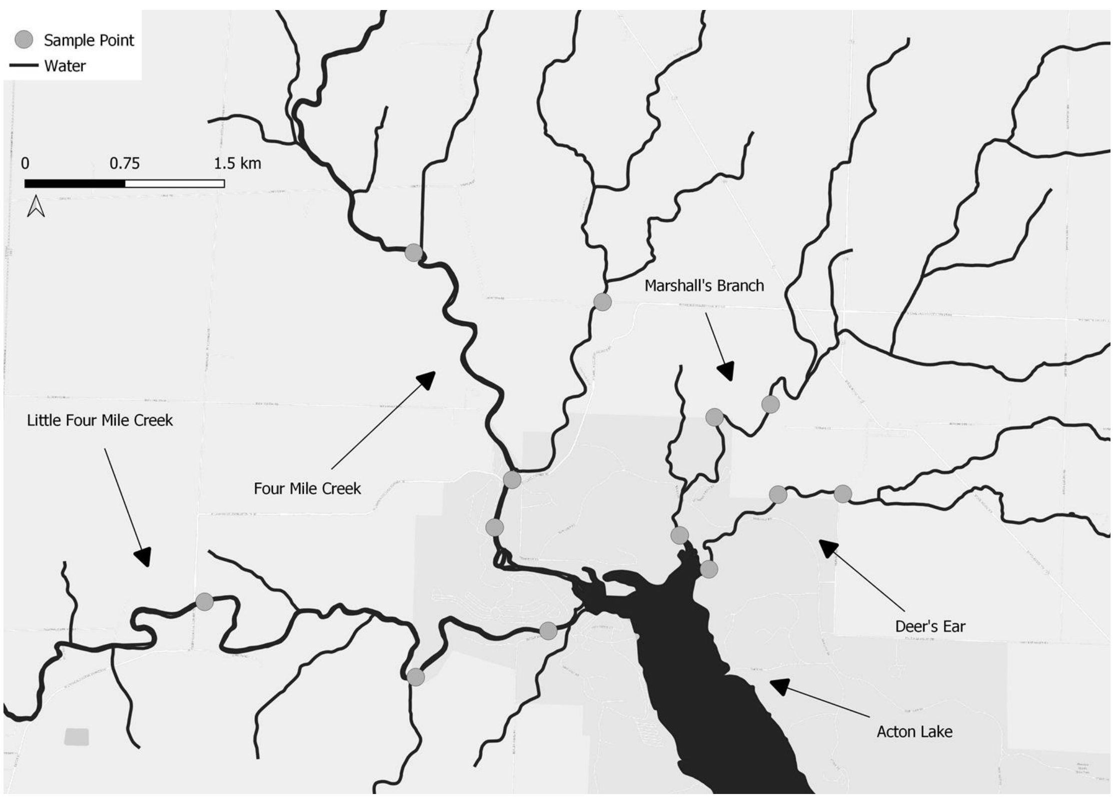

Figure 20: The original study design included three points on each of the creeks within the park boundary as well as sampling points on farmland outside of the park boundary.

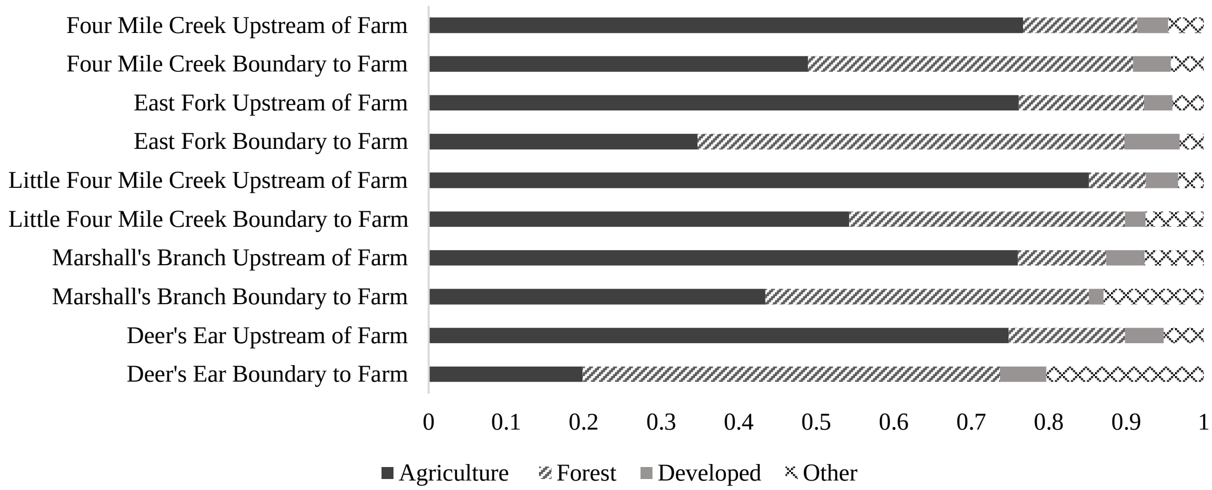

Figure 21: A 500-meter buffer zone was established around each of the creeks. "Upstream of Farm" refers to the buffer zone from the farm to the top of the watershed. "Boundary to Farm" refers to the buffer area between the Hueston Woods State Park boundary and the farm sampling location. The graph shows the proportion of land cover and land use (LCLU) that falls within each buffer zone for each creek in 2020. The LCLU data was analyzed based on the United States Department of Agriculture Cropland Data Layer (CDL).

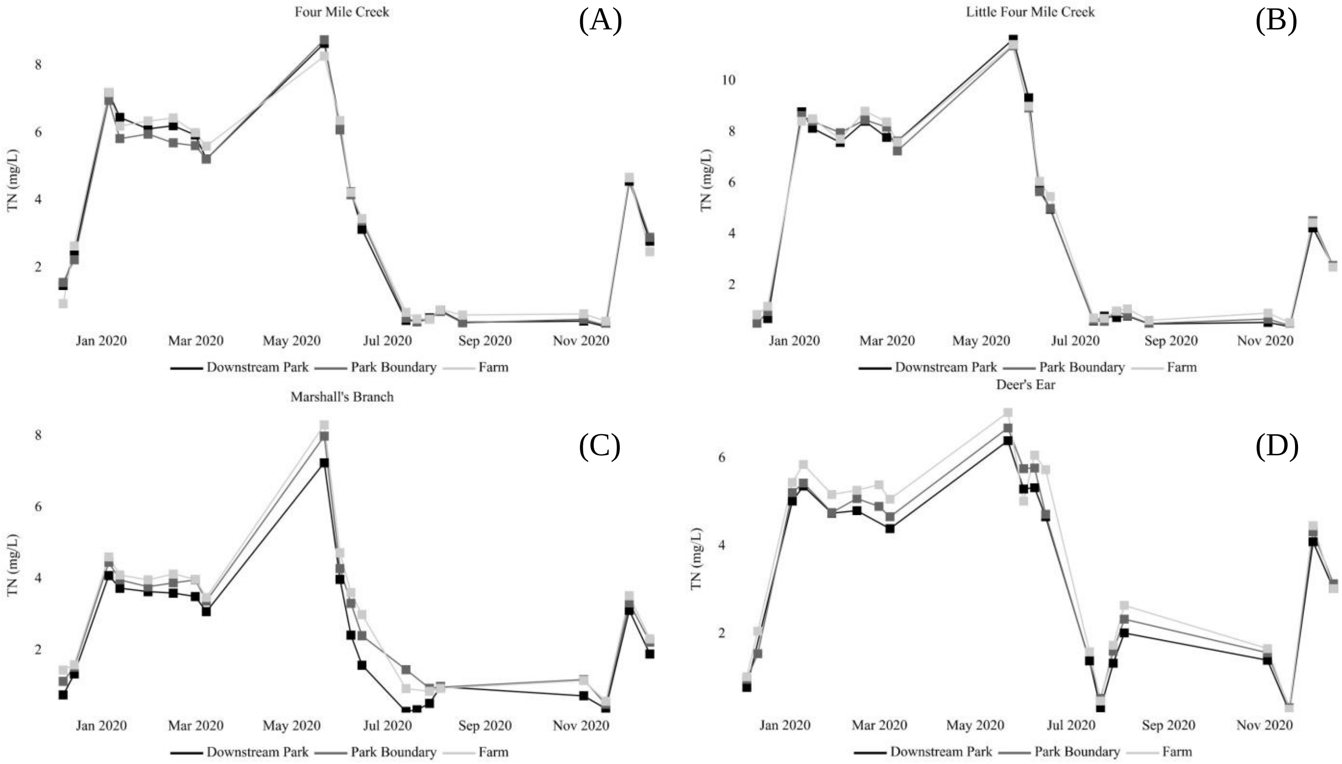

Figure 22: Total nitrogen concentrations (mg/L) are shown over the study period (December 2019-December 20020) at the farm sampling location, the Hueston Woods State Park boundary, and the downstream park location within each sampling creek (a-d).

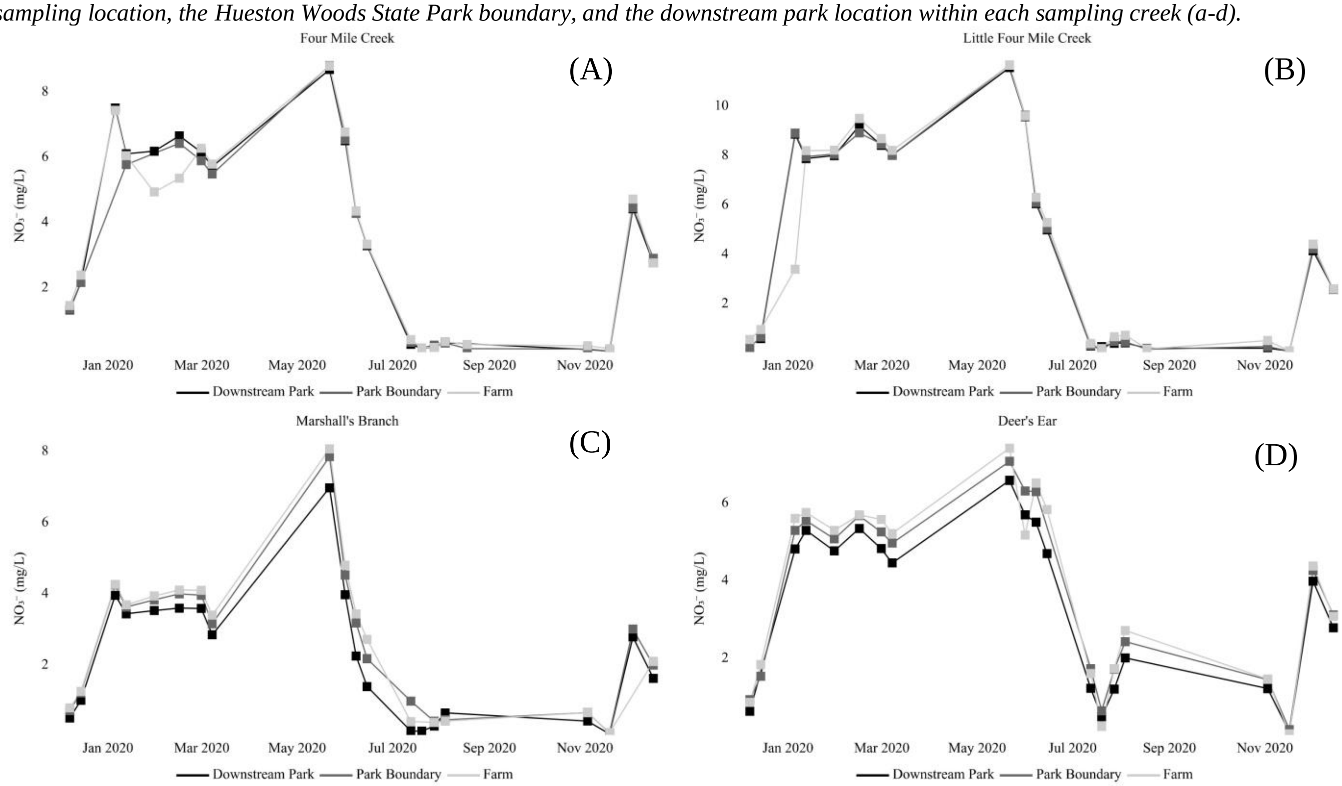

Figure 23: Nitrate concentrations (mg/L) are shown over the study period (December 2019-December 20020) at the farm sampling location, the Hueston Woods State Park boundary, and the downstream park location within each sampling creek (a-d).

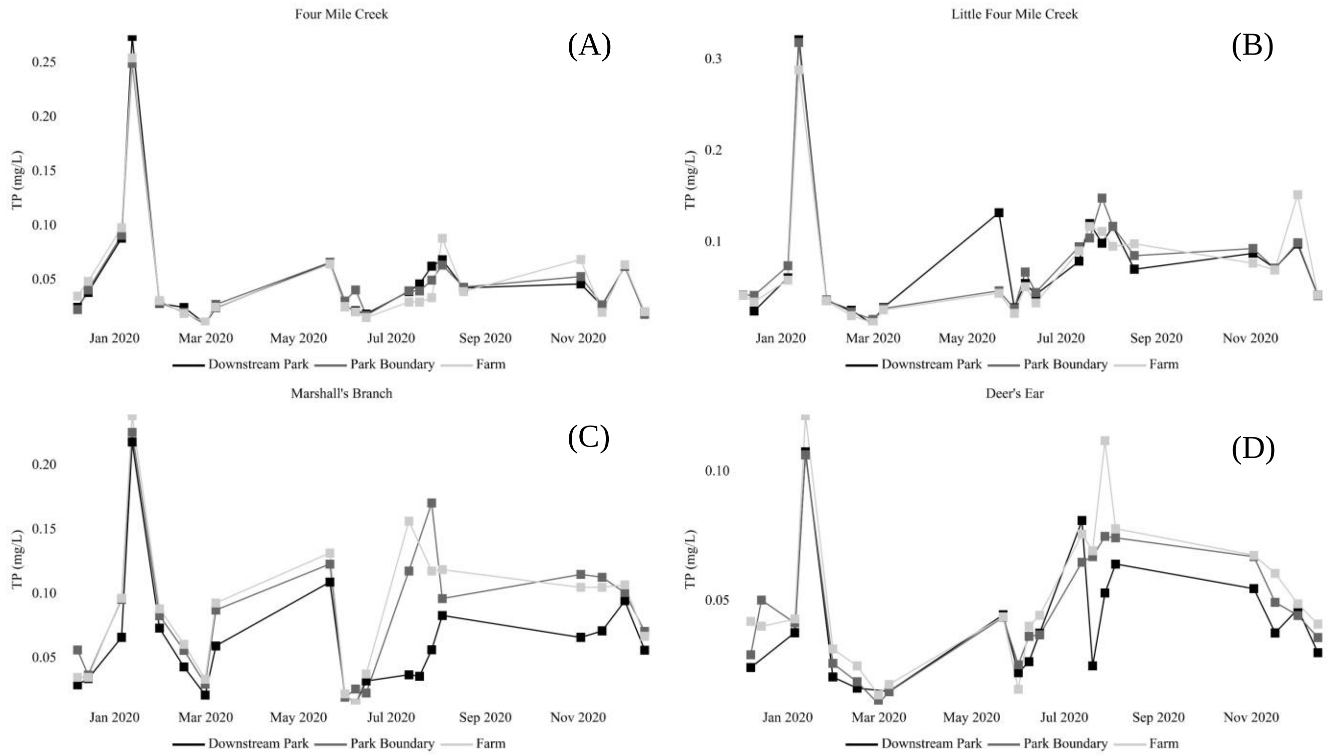

Figure 24: Total phosphorus concentrations (mg/L) are shown over the study period (December 2019-December 20020) at the farm sampling location, the Hueston Woods State Park boundary, and the downstream park location within each sampling creek (a-d).

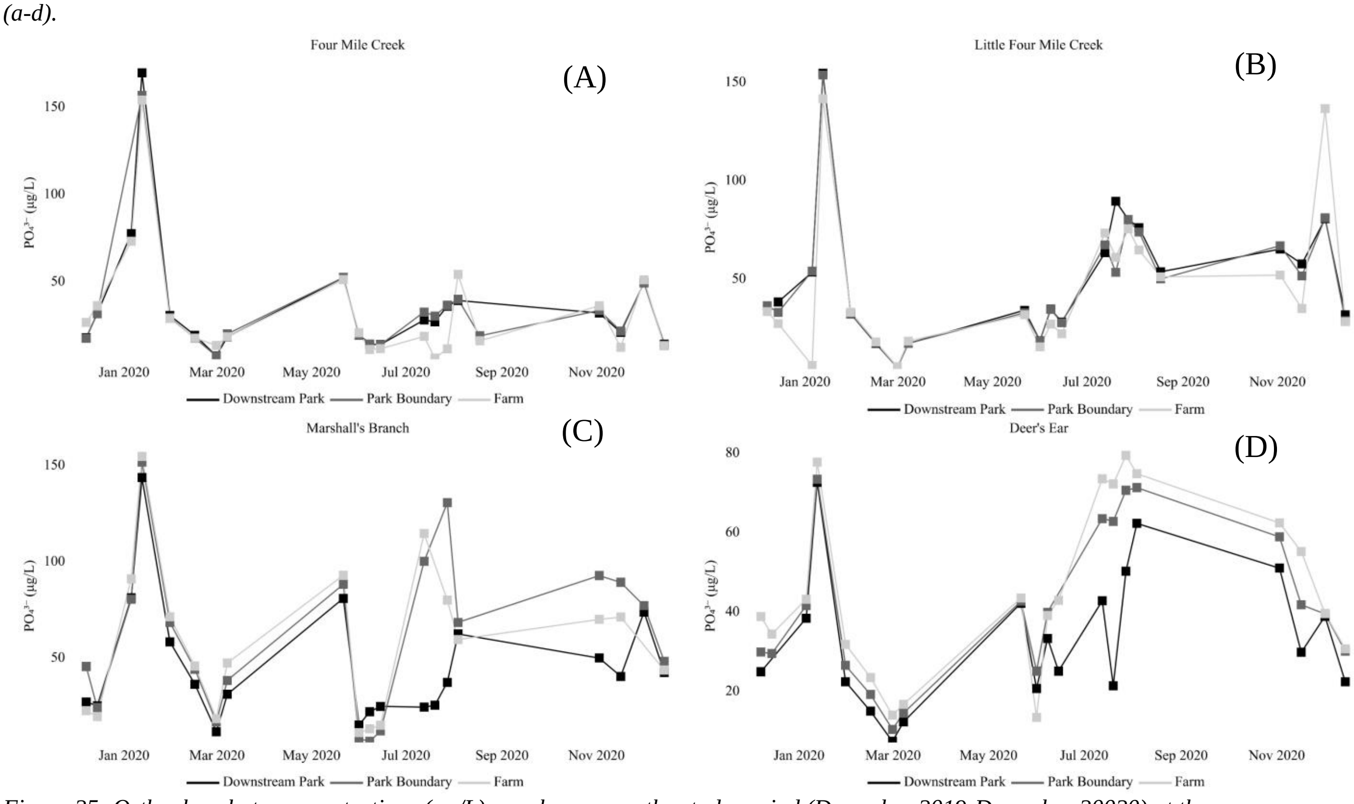

Figure 25: Orthophosphate concentrations (µg/L) are shown over the study period (December 2019-December 20020) at the farm sampling location, the Hueston Woods State Park boundary, and the downstream park location within each sampling creek (a-d).

Table 3

Sentinel-2 Composite Dates for NDVI

Water Sampling Date	Composite Start Date	Composite End Date
12/08/2019	12/03/2019	12/13/2019
12/15/2019	12/10/2019	12/20/2019
01/06/2020	01/01/2020	01/11/2020
01/13/2020	01/08/2020	01/18/2020
01/31/2020	01/26/2020	02/05/2020
02/16/2020	02/11/2020	02/21/2020
03/01/2020	02/25/2020	02/06/2020
03/08/2020	03/03/2020	03/13/2020
5/22/2020	05/17/2020	05/27/2020
06/01/2020	05/27/2020	06/06/2020
06/08/2020	06/03/2020	06/13/2020
06/15/2020	06/10/2020	06/20/2020
07/13/2020	07/08/2020	07/18/2020
07/20/2020	07/15/2020	07/25/2020
07/28/2020	07/23/2020	08/03/2020
08/04/2020	07/30/2020	08/09/2020
08/18/2020	08/13/2020	08/23/2020
11/03/2020	10/29/2020	11/08/2020
11/17/2020	11/12/2020	11/22/2020
12/02/2020	11/27/2020	12/07/2020
12/15/2020	12/10/2020	12/20/2020

Ten-day composites surrounding water sampling dates (five before and five after) were used to determine vegetation conditions during sampling events and to limit effects of cloud coverage.