

NUTRIENT, BACTERIA, AND SEDIMENT FLUXES IN A NATURAL WETLAND: AN ECOSYSTEM SERVICE OR DISSERVICE?

by

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Some wetlands have been shown to provide ecosystem services including flood water retention, water quality improvement, and habitat for wildlife and aquatic organisms. Wetlands are commonly created and/or restored to provide these services. Some wetlands though, may be exporters of carbon and other pollutants and thus could provide a disservice to the environment and eventually public health. The goal of this project was to determine if a natural wetland receiving drainage from an urbanizing catchment was a source or sink of nutrients, bacteria, and sediment. Inflow and outflow samples from the wetland were collected monthly for one year and analyzed for total dissolved nitrogen, phosphate, *Escherichia coli* and total suspended solids concentrations. Physiochemical properties of the samples including pH, dissolved oxygen, turbidity, specific conductance, flow, and temperature were also measured in the field. The pollutant treatment efficiency of the wetland was evaluated by comparing differences in inflow and outflow concentrations and loadings. Overall, the wetland reduced masses of total dissolved N (TDN) (51.6% reduction) and phosphate P (PO₄-P) (62.9% reduction). The median outflow concentration of *E. coli* was 50% lower relative to inflow. Flow-weighted turbidity of wetland outflow wetland was 13.2% greater relative to the median inflow. Similarly, median total suspended solids mass increased 14.9% from inflow to outflow. Results from this

research show that the wetland was a sink for N, P and *E. coli* reductions. However, the wetland was a source of sediment (and turbidity), likely because of erosion within the wetland. Efforts to slow urban runoff and stabilize the wetland are suggested to prolong (and perhaps enhance) the ecosystem services it is providing. Such practices may simultaneously inhibit some of the disservices (sources of pollution) caused by the wetland.

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ECOSYSTEM SERVICE OR DISSERVICE?

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LIST OF ABBREVIATIONS

BOD5	Biochemical Oxygen Demand (over 5 days)	3
COD	Chemical Oxygen Demand.....	3
DIN	Dissolved inorganic nitrogen	19
DO	Dissolved oxygen	1
<i>E. coli</i>	<i>Escherichia coli</i>	8
HABs	Harmful algal blooms	9
I-1	Inlet 1	14
I-2	Inlet 2	14
LCW	Lick Creek Watershed	14
LLCW	Little Lick Creek Watershed	14
N	Nitrogen	1
NH ₄ -N	Ammonium-nitrogen	3
NO ₃ -N	Nitrate-nitrogen	2
O-1	Outlet 1	15
O-2	Outlet 2	15
O-3	Outlet 3	15
ORP	Oxidation-reduction potential	12
P	Phosphorus	1
PO ₄ -P	Phosphate-phosphorus	12
SC	Specific conductance	12
TDN	Total dissolved nitrogen	19
Temp.	Temperature	12

TN	Total nitrogen	4
TKN	Total Kjeldahl nitrogen	3
TKN	Total organic carbon	3
TP	Total phosphorus	3
TSS	Total suspended solids	3
Turb.	Turbidity	12

CHAPTER 1: INTRODUCTION

1.1 Wetlands and Ecosystem Services

Wetlands are *ecotones*, or transitional spaces between upland and aquatic environments. There are many types of wetlands, including bogs, swamps, and salt and freshwater marshes (NPS, 2019). All wetlands have a common factor: presence of water at the surface or near the surface, for at least a portion of the year (U.S. EPA, 2015). Wetlands stay saturated long enough for subsoils to become low in dissolved oxygen and lead to a reducing environment (U.S. EPA, 2015). Plants, trees, shrubs, and other vegetation that grow in wetlands have adaptations that allow them to live in soil that is periodically wet based on rainfall patterns and other hydrological controls (Mitsch and Gosselink, 2000). The dynamic saturation of wetlands supports different types of microbes and other organisms, which influence transformations and mobility of nitrogen (N), phosphorus (P), carbon and other elements. Processes resulting from anaerobic decomposition of organic matter often lead to the commonly recognized smells of sulfur in marine environments and methane in freshwater swamps. Previous studies have shown natural wetlands act as sinks for organic pollutants in estuary and marine-fringe locations (Sakari et al., 2014). Wetlands are referred to as “*kidneys of the landscape*” since they filter pollutants from runoff sources including agriculture and urban areas, prior to the runoff entering surface waters (Mitsch et al., 2015). Liu et al. (2004) stated that wetlands are not only rich in biodiversity but provide many positive functions for the environment such as providing habitats for many plants and endangered species; sustainability of wetlands can be achieved through job creation to improve the health of the land and inhabitants. Studies have suggested that larger wetlands in comparison to their drainage sources yield the most effective water quality benefits; restored agricultural wetlands in Illinois, Maryland, and Iowa removed 68% of

NO₃-N and 43% of P in drainage (Woltemade, 2000). Woltemade stated these benefits are highly impacted by flow variability and related to residence time.

Immobilization of nutrients through plant uptake, adsorption of P, sedimentation of soil particles, burial of carbon and P, and N transformations including denitrification may occur in natural wetlands (U.S. EPA 2009; Humphrey et al., 2014; Liu et al., 2019). Engineers have designed and overseen the implementation of constructed wetlands in various settings to mimic the pollutant removal mechanisms observed in their natural counterparts. Wetlands have been designed to reduce nutrient runoff from animal waste operations and agricultural fields, urban stormwater, coal mine drainage, and to treat wastewater (Hammer, 1991; Rousseau et al., 2008; U.S. EPA, 2015). The efficiency of wetlands to treat pollutants are influenced by numerous factors including climate, soil properties (particle size distribution, cation exchange capacity, infiltration rates), design specifications, and usage designation such as natural wetlands, wastewater wetlands, constructed wetlands, and stormwater wetlands (Weiss et al., 2008). These factors will influence adsorption, plant uptake, microbial transformations and/or other treatment mechanisms responsible for pollutant reductions and treatment efficiencies. Therefore, pollutant treatment efficiencies of wetlands may differ greatly if the influencing factors also differ. Designated usage may cause large differences in pollutant removal due to inflow volumes and masses and concentrations of what is entering the wetland, i.e. N and bacteria in designated wastewater wetlands (Table 1). Constructed wetlands are maintained and therefore may have more continued treatment efficiency over time.

Table 1. Literature table of wetland performance from selected sites. BOD5= 5-day biochemical oxygen demand; COD= chemical oxygen demand; TSS= total suspended solids; TKN= total kjeldahl nitrogen; TN= total nitrogen; NH₄-N= ammonium nitrogen; NO₃-N= nitrate nitrogen; TP= total phosphorus.

Author	BMP Type	Findings
Rozema et al. (2016)	21 studies of constructed wetlands in Eastern Canada and Northeastern USA	Average concentration reductions: 81% in BOD5, 83% in TSS, 75% TKN, 76% in NH ₄ -N, 42% in NO ₃ -N, 64% in TP.
Almuktar et al. (2018)	A constructed wastewater wetland in Ingham, Australia	Mass removal: 67% in BOD5, 44% TSS, 74% TN, 65% NH ₄ -N, 91% NO ₃ -N, and 6% TP.
Kao & Wu (2001)	A natural wetland located in McDowell County, NC, USA	More than 80% N removal, 91% of TSS removal, 59% of TP removal, and 66% of COD removal during a storm event.
Jordan et al. (2003)	Agricultural wetland in MD, USA over 2 years	Removal of 59% TP, 38% TN, 41% TOC during 1st year with 3-months of low-flow. Year 2 lacked period of decreased flow, no significant ($p < 0.05$) net removal of TN, P or TSS. Over 2 years: 25% NH ₄ -N, 52% of NO ₃ -N was removed.

Wetlands that have been compromised may contribute to pollution loading to adjacent surface waters. Compromising situations may include high concentrations of toxic substances, metals and nutrient levels in influent waters or extreme inflow volumes that overwhelm wetlands and degrade them to the point of reduced function and storage capacity (Haddaji, et al., 2019; U.S. EPA, 2001). Research in Australia has shown greater “flushing” of nutrients and other pollutants in the wetlands may occur during high-flow periods of the winter (Raisin & Mitchell, 1995). During wet seasons, some wetlands may achieve lower levels of performance, resulting in

increased surface runoff and pollutant transport. This is mainly explained through decreases in hydraulic conductivity in the soil, and reduced residence of water in the wetland (Peterson, 1998). Runoff associated with major storms has been shown to limit pollution treatment efficiency and can cause wetlands to transport pollutants (e.g., NO_3 , bacteria, sediments, P) rather than sequester them. This runoff pollutant transport comes from factors such as saturated soils, and decreased residence time (Mitsch & Gosselink, 2000). Today, the U.S. Government has 5 specific agencies involved with wetland regulations including: U.S. Army Corps of Engineers (to deliver public and military engineering that reduces natural resource issues); U.S. EPA (to protect human health and the environment); The Department of the Interior (to manage America's land, water, and wildlife), U.S. Fish and Wildlife Service (to enforce federal laws to protect wildlife resources); Department of Commerce (to set standards and promote job creation); National Oceanic and Atmospheric Administration, or NOAA (to understand, manage, conserve, and predict climate and marine-related issues); and the Department of Agriculture, Natural Resources Conservation Service, or NRCS (to conserve soil, water and other natural resources by promoting interactive programs).

1.1.1. Common Water Pollutants: Sources and Cycling

N is a dynamic element that may appear in different compounds and oxidative states in the environment. N is the most abundant element in the atmosphere, and N gas (N_2) is the most common form. Most plants can use N in the ammonia (NH_3), $\text{NH}_4\text{-N}$ or $\text{NO}_3\text{-N}$ forms, but only a few are able to use N gas (N_2) (Havlin et al., 1999). Lightning strikes can convert N_2 gas in the atmosphere to N oxides, forms that are useable by most plants (U.S. EPA, 1999). Since most crops typically do not receive enough N naturally from the environment, growth is stimulated by application of fertilizers, animal manure, and biosolids (U.S. EPA, 2019). Organic N in

wastewater can transform into $\text{NH}_4\text{-N}$ through a process known as ammonification, brought on by energy from decaying organic matter in the presence of bacteria. $\text{NH}_4\text{-N}$ is transformed to $\text{NO}_3\text{-N}$ via nitrification, which is brought on by a carbon energy source in the presence of nitrifying bacteria in aerobic environments (Figure 1). N also enters the environment via discharges from municipal sewers and septic systems (U.S. EPA, 2019; Wisconsin Department of Natural Resources, 2018). N in raw wastewater is typically in the form of organic N (ON) and $\text{NH}_4\text{-N}$. Primary and secondary treatment processes at wastewater treatment plants or in the septic tank and drainfield trenches/soil of septic systems may result in transformations of ON to $\text{NH}_4\text{-N}$, and then $\text{NH}_4\text{-N}$ to $\text{NO}_3\text{-N}$. N may be removed through denitrification (Wisconsin Department of Natural Resources, 2018). Denitrification is an anaerobic process that returns $\text{NO}_3\text{-N}$ to atmospheric N, such as N_2 , through reduction of $\text{NO}_3\text{-N}$ via microbial processes to the gaseous N (Martens, 2005). The denitrification process can have negative impacts as well, such as the release of ozone gases N oxide and nitrous oxide (Parton et al., 2001). However, if septic systems or municipal sewer plants are improperly designed, installed and/or maintained, N transformations may not occur and water resources may become enriched with N (U.S. EPA, 2019). Wetlands located between wastewater treatment systems and surface waters may help to reduce N transport via denitrification, if $\text{NO}_3\text{-N}$ is the dominant form of N in groundwater, and the wetland soils are carbon-rich, anaerobic and saturated (Humphrey et al., 2014).

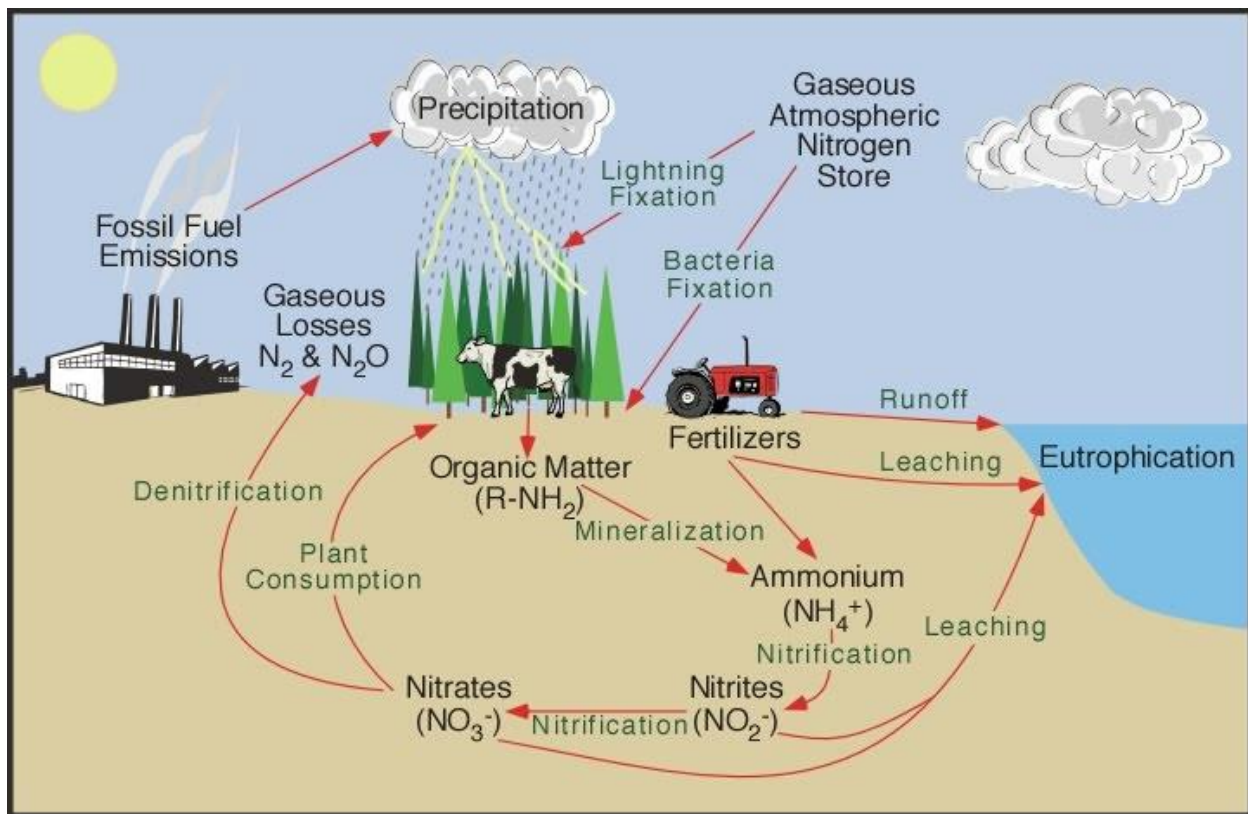


Figure 1. The N Cycle (Pidwirny, 2006).

P is a plant-essential nutrient that often limits productivity on land and in water, and often enters water systems through runoff (U.S. EPA, 2017; USGS, 2019). P is important for energy transfer in plants produced by photosynthesis, promoting root growth and maturity. P enters the environment through many sources including agricultural and lawn fertilizers, wastewater discharges, and animal/wildlife waste (U.S. EPA, 2017). P mobility is limited due to multiple processes including adsorption, mineral precipitation, and immobilization by plants and microbes (Reddy & DeLaune, 2008). These processes are influenced by several factors including soil pH, soil saturation, availability of various elements such as calcium, iron, aluminum, and biological activity (Devau et al., 2009; Reddy & DeLaune, 2008). Humid regions with acidic soils have great capacity to adsorb P via binding with aluminum oxides or precipitation of minerals. However, for this reason, farmers and lawn maintenance specialists may utilize lime on soil to

adjust pH in maintained areas and increase P uptake (Crozier & Hardy, 2018). P may be released and transported to surface waters upon the death of an organism contributing to organic matter or by soil erosion driven by runoff (Parnes, 2019). Excess P that is not used by crops may be transported via runoff to adjacent waterways. Similarly, soil loss through erosion can exacerbate the problem by adding soil-attached P to downstream waters. According to the NRCS (2019), P loss and subsequent pollution into nearby streams typically comes from erosion and runoff, due to P not having an affinity for leaching from root zone areas. This poses a large problem for impaired watersheds that lack stormwater control measures where sediment pollution is occurring (U.S. EPA, 2019). The NRCS further states that P fixation occurs best in the presence of iron at pH levels below 6, then aluminum at pH 4-7, then calcium at pH levels above 6. Depending on pH levels, this may be a contributing factor for P as a pollutant source if levels exceed what is entering the wetland (Figure 2).

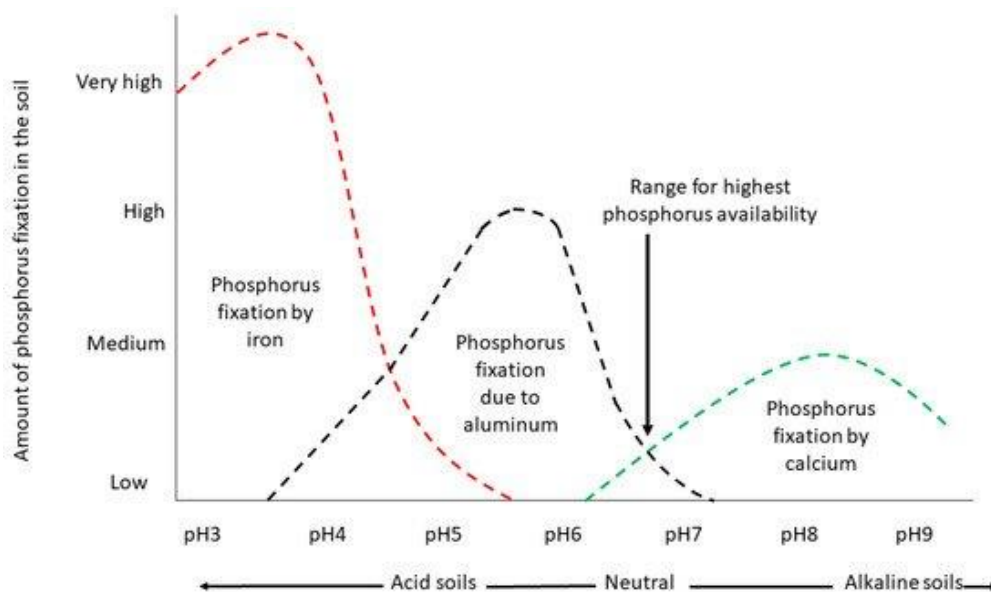


Figure 2. P fixation and pH (Penn & Camberato, 2019; redrawn from Price, 2006).

1.1.2. Fecal Bacteria and Water Quality

Escherichia coli (*E. coli*) is a naturally occurring bacteria in the intestinal track of warm-blooded animals including cows, chickens, hogs, and humans (Ishii & Sadowski, 2008). Some strains are pathogenic, such as the verotoxin-producing *E. coli* O157:H7 (VTEC). This strain is a concern due to its tolerance of high-level pH ranges, making it resilient. VTEC can gain additional virulence when combining with less-toxic strains when deposited or excreted (Elsas et al., 2010). Due to bloody diarrhea potentially experienced by hosts such as humans, there is increased potential for this strain to be introduced to municipal wastewater treatment facilities and septic systems. Malfunctioning septic systems can be a contributor of *E. coli* in groundwater particularly when placed in areas of improper soils (Mallin, 2013). *E. coli* in manure may attach to root systems and leaves in the environment due to splashing from rainfall or irrigation and be transported to waterways during storms (Natvig et al., 2002). The CDC (2015) states that lethal *E. coli* strains such as VTEC can enter drinking water from sewage overflow and storm runoff, particularly from agricultural fields. *E. coli* can come from runoff from livestock feeding areas (Elsas et al., 2010; Verhougstraete et al., 2015). *E. coli* levels can also be present from wild animals in nonhuman managed areas, though levels are lower (Guan et al, 2002). *E. coli* has shown the ability to reach a dormant state of retained viability even in uninhabitable conditions (Elsas et al., 2010). Bacterial growth depends on availability of nutrients (N and P) and an energy source (e.g., organic carbon) (Chekabab et al., 2013). Due to its ability to persist in soil and waters of temperate climates, *E. coli* is used as an indicator of pathogenic contamination in freshwater systems (U.S. EPA, 2013; Ishii & Sadowsky, 2008). Sunlight, UV radiation, and predation are major natural removal methods of *E. coli* in runoff in wetlands (Weber & Legge, 2008). While plants are a major component of wetlands that allow for nutrient and sediment reduction, plants block UV rays, and decaying organic material from the plants may reduce

dissolved oxygen supplies for predator species, such as zooplankton; significant decreases in *E. coli* concentrations were noted in a constructed wetland following removal of shade-providing plants, because loss of shade cover allows increases in temperatures that are more support of *E. coli* (Macintyre et al., 2006). Increased suspended solids in water beyond a certain point leads to decreases in UV disinfection reduction of pathogenic microorganisms, which is totally blocked in areas of darkness (Gonzalez et al., 2020). Other suspected natural methods for removal of *E. coli* in wetlands are adsorption and sedimentation, which may be positively influenced by wetland plants (Boutilier et al., 2009).

1.1.3. Sediment and Water Quality

Sedimentation-related problems first took national notice in the form of erosion in the 1930s during the dustbowl (NRCS, 2019). While programs were enacted more than 70 years ago to help reduce soil loss, sediment is currently the most common pollutant in lakes, streams and rivers and causes an estimated \$16 billion in environmental damages annually (U.S. EPA, 2019). Sediment increases potential for flooding by clogging drainageways. Sediment polluted waters become cloudy and turbid, reducing photosynthesis in submerged aquatic plants (Yang et al., 2008). Harmful algal blooms (HABs) are increasing due to anthropogenic changes to the land, including increased sediment carrying agricultural runoff (Weirich & Miller, 2014). Other facilitators of erosion include deforestation, overgrazing of livestock, and unsustainable agricultural practices (Borrelli, 2017). The U.S. EPA recommends sedimentation prior to urban and agricultural runoff leaving the site, such as: use of straw erosion control blankets on disturbed land, washing items such as vehicles at commercial washes only or in areas where runoff can be adsorbed, use of stormwater control measures, and avoidance of practices such as dumping or mowing in areas near streambeds.

1.2 Impairment of Water Resources

Excess nutrients, fecal bacteria, and sediment in water resources often lead to water use impairment, threaten public and environmental health, and may cause economic hardships. HABs can result from excess nutrients in watersheds and blooms can cause irritation to humans, wildlife and pets that encounter them (Brooks et al., 2016; U.S. EPA, 2019). When the algae die and decompose, there may be a reduction in the concentration of dissolved oxygen in the water, which may result in fish kills. A public health crisis can ensue following HAB presence. Some HABs can create hazards for those consuming shellfish, containing neurotoxins that target the central nervous system, respiratory system and gastrointestinal system, which may be fatal to humans and other animals (NOAA, 2020). Additional negative public health consequences can result from too much N in drinking water. Methemoglobinemia, or “blue baby syndrome”, results from lack of oxygen circulating in the blood due to unavailability of hemoglobin receptors. While this can be caused by certain medications and other factors, overabundance of NO₃-N in drinking water can be a major cause of this as well (CDC, 2002; Knobeloch, et al., 2000). NO₃-N is also a suspected carcinogen; increased risk of colorectal cancer is observed in those consuming increased NO₃-N levels (Ward, et al., 2018). Exposure to pathogens in water supplies and recreational waters normally come from ingestion and can cause gastrointestinal illness, potentially leading to organ failure and death (U.S. EPA, 2013; Pandey et al., 2014; Ramirez-Castillo et al., 2015). Pathogens and nutrients may attach to sediment, thus reducing sediment transport may reduce nutrient and bacteria loading to receiving waters.

Past studies have found that natural wetlands may provide ecosystem services that reduce damages to the public and environment by treating pollution (Davies and Bavor, 2001; UM, 2018). However, other studies have shown that wetlands can act as a source of pollution, such as

the wetlands in the Danube Delta. These wetlands have seen a 67% decline in ecosystem services since 1960, because of developmental pressure and changes in hydrology and pollutant delivery (Gómez-Baggethun, et al., 2019). Gómez-Baggethun and colleagues (2019) suggest that even after decades of restoration efforts, sediment and other pollution generated by anthropogenic means has overwhelmed the wetlands. Additionally, Moss (2008) found that wetlands are effective at removing N, but not P. More work is needed to better understand the factors that influence whether a natural wetland in a nutrient-sensitive watershed is a source or sink for common pollutants such as nutrients, sediment, and bacteria.

CHAPTER 2: STUDY GOALS AND OBJECTIVES

The goal of this project was to gain a better understanding of the pollutant fluxes within a natural wetland that receives drainage from a watershed with a high density of homes and septic systems and to determine if the wetland serves as a source or sink of nutrients, bacteria, and sediment. Humphrey et al. (2016) found that sand filter septic systems in the study area were less efficient at TDN removal in comparison to conventional septic systems. Another study by Humphrey et al. (2016) found that $\text{PO}_4\text{-P}$ was removed by both conventional septic systems and sand filter, though conventional did a much better job at mass removal. Research by Iverson et al (2017) reported elevated concentrations and loadings of *E. coli* in streams draining watersheds with high densities of septic systems in the study area. These studies suggest nutrient and bacteria loadings to the wetland may be elevated due to the number of sand filter-style septic systems and overall septic system density in the watershed upstream from the study site.

Specific objectives were to:

- 1) Determine flow (L s^{-1}) entering and exiting the wetland and characterize the physicochemical properties (oxidation-reduction potential [ORP], pH, temperature, dissolved oxygen [DO], turbidity, specific conductance [SC]) of inflow and outflow using discrete measurements and autonomous loggers.
- 2) Collect water samples from the inlets and outlets of the wetland and analyze the samples for concentrations of total nitrogen (TN), total phosphorus (TP), *E. coli*, and total suspended solids (TSS) monthly for a year.
- 3) Determine if the wetland is a source or sink for TN, TP, TSS, and *E. coli* based on differences in nutrient, sediment and *E. coli* loadings entering and exiting the wetland.

We hypothesized that the natural wetland would be a sink for nutrients, sediment, and bacteria, thus providing ecosystem services by attenuating these pollutants.

CHAPTER 3: METHODS

3.1 Study Site

The study site is a wetland located upslope from a levee on the banks of Lick Creek (Figure 3A). Lick Creek is an impaired stream and a tributary of Falls Lake, which is also an impaired water body (NCDEQ, 2019). Falls Lake is the water supply for the City of Raleigh, the capital of North Carolina. The Lick Creek Watershed (LCW) drains portions of Wake and Durham Counties in the Piedmont of NC and encompasses about 4455 ha (17.2 sq. mi²). The area receives on average about 111.8 cm (44 in) of rainfall annually (Appendix A). The LCW is predominantly composed of the White Store soil series (69%), followed by Creedmoor (9%) and Chewacla Wehadkee (9%) (Durhamnc.gov, 2015). The study area has annual averages of 15°C for temperature, average high and low of 21.2°C and 8.8°C, respectively (U.S. Climate Data, 2019). Relative humidity is typically around 70% (wunderground, 2019). The Inlet 1 (I-1) and Inlet 2 (I-2) watersheds are comprised of a mixture of urban and rural developments. I-2 is adjacent to a new (< 15 years) subdivision and urban development accounts for just over 60% of I-2 land use, resulting in 12% impervious surface. I-1 is semi-rural, with 43% urban development, and 7% impervious surface. Between the I-1 and I-2 watersheds, wetlands account for about 3% of the land use (Appendix B). The U.S. EPA (2015) standard for *E. coli* for Lick Creek, and other fresh waters, is a geometric mean (GM) of less than 126 colony forming units per 100 ml (cfu 100 ml⁻¹). No more than 10% of samples examined should exceed 410 cfu 100 mL⁻¹. In the Little Lick Creek Watershed (LLCW) nearby, poor water quality was exhibited with regards to fecal coliform that exceeded levels for all 10 years, TP for 5 of 10 years, and turbidity levels from 2006 to 2016 (time of the report), deeming the watershed “impaired” by the State of NC, since 2008 (DurhamNC.gov, 2016). The City of Raleigh plans to increase drinking water withdrawals

from Falls Lake, making it even more important to monitor water quality of Falls Lake and its tributaries.

The wetland evaluated in this study receives drainage from a 24-ha catchment. The wetland is roughly 2 ha in size and includes dominant vegetation such as lizard's tail (*Saururus cernuus*), sweet shrub (*Calycanthus floridus*), three species of ground cedar (*Lycopodium* [*flabellum*, *obscurum* and *lucidulum*]) and many trees including sweetgum (*Liquidambar styraciflua*) red maple (*Acer rubra*), and loblolly pine (*Pinus taeda*) (UNRBA, 2020). The LCW is home to two protected species: four-toed salamanders (*Hemidactylium scuatatum*) and Carolina darters (*Etheostoma collis*), adding another element to why maintaining this area is important. There are 44 septic systems in the watershed for the wetland, averaging about 1.9 systems ha⁻¹. The wetland receives drainage from 2 small streams including I-1 and I-2 (Figure 3B, Figure 4A). I-1 drains a 15-ha area with 28 septic systems. I-2 drains a 9-ha watershed with 16 septic systems, 12 of which are sand filter-style septic systems. Streams from I-1 and I-2 dissipate into the wetland, and the morphology of the stream channels change after storms. The outlets are all showing signs of severe erosion (Figure 4B).

3.1.1 Physicochemical characterization of wetland inflow and outflow

Inflow and outflow measurements and samples were collected from the wetland monthly for a year (July 2018 to June 2019). Inlet sampling locations are denoted as Inlet 1 (I-1) and Inlet 2 (I-2); outlet locations are denoted as Outlet 1 (O-1), Outlet 2 (O-2), Outlet 3 (O-3) (Figure 3B). The outlets discharge to Lick Creek.

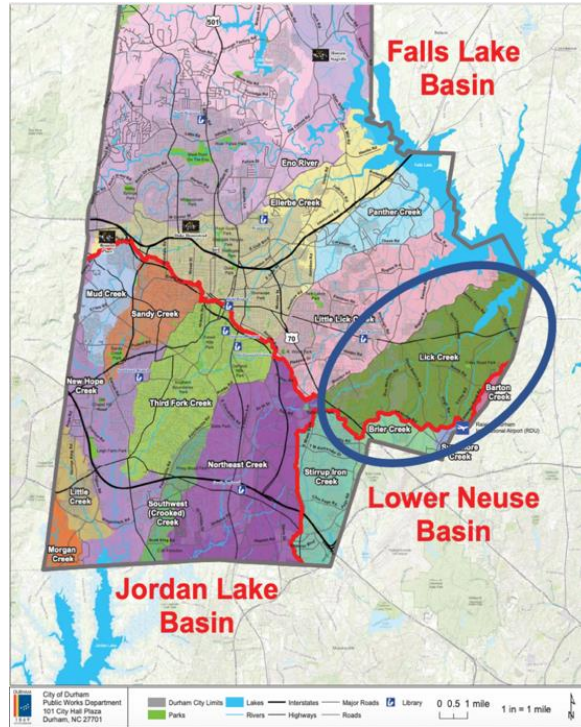


Figure 3A. Lick Creek Watershed (circled). Durham County, NC (durhamnc.gov, 2020)

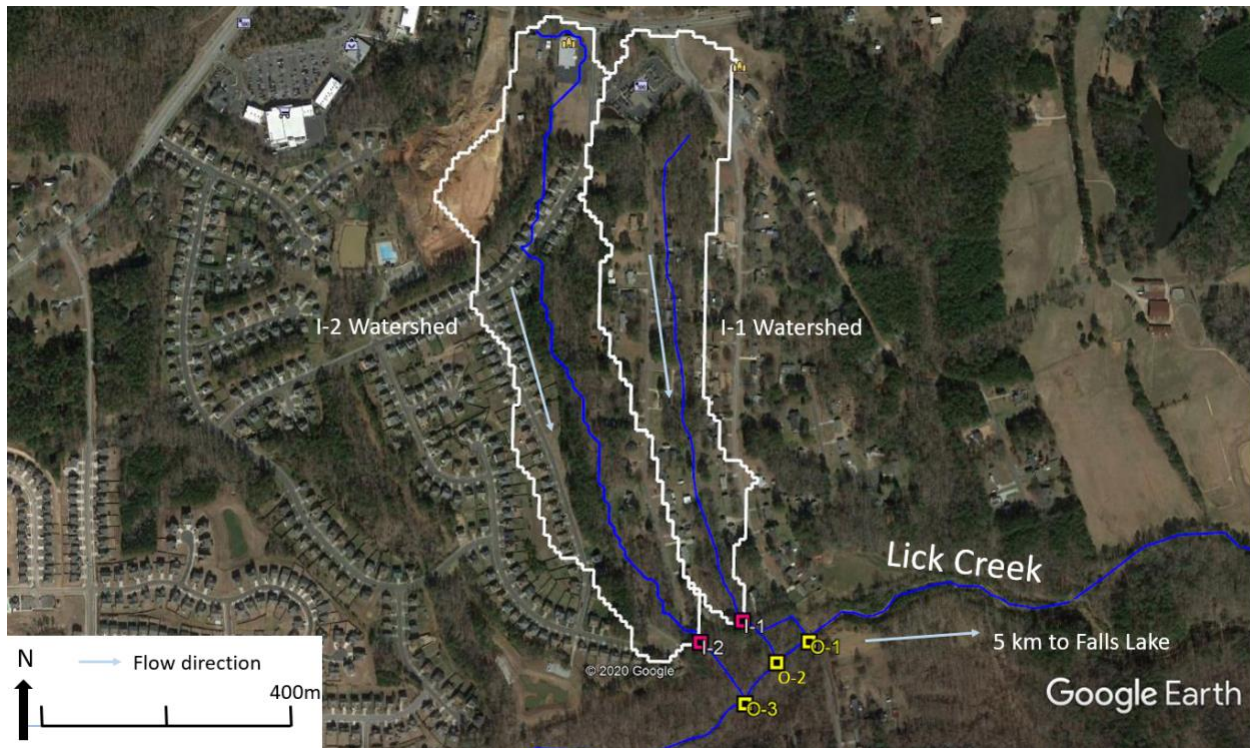


Figure 3B. Lick Creek Wetland and Watershed map. I indicates inlets, O indicates outlets (USGS, 2020).



Figure 4A. Taking measurements at I-2 during baseflow.

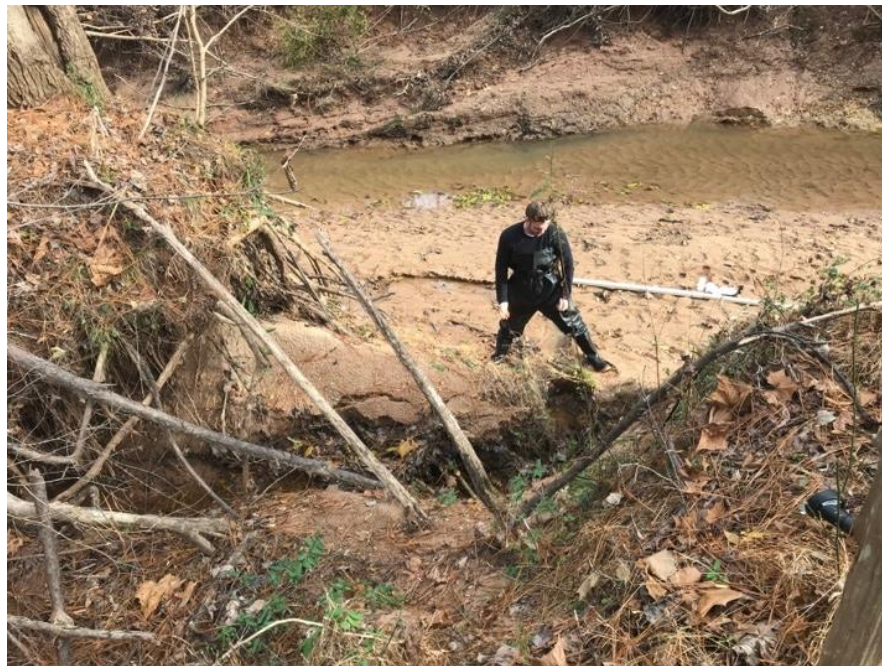


Figure 4B. Erosion at a wetland outlet, such as uprooted trees and sandy slope cut-out. Lick Creek is in the background.

Stream velocity was measured at the inflow and outflow locations using a *Global Water* flow probe. In cases where the flow was too low to be detected with the meter, a xanthene tracer dye (2019) was used to estimate stream velocity by timing how long it took for the dye to travel

a known distance. The process was completed 3 times and the average velocity was reported. Stream width was determined using a tape measure that was pulled from one bank of the stream to the other. Depth of water was measured every 15 cm. along the cross-section. Discharge was calculated by multiplying average velocity (m s^{-1}) by the stream width (m) and average depth (m). Units of discharge were converted to L s^{-1} (Appendix C). A YSI 556 meter was utilized to measure physicochemical parameters of inflow and outflow including ORP, pH, SC, temperature, and DO. A Hach 2100P Turbidimeter was used to measure turbidity. Inflow and outflow water samples were collected for nutrient and TSS analyses by dipping a 500 mL high-density polyethylene (HDPE) bottle under the surface of the water until full at each sampling location. Samples for microbial analyses were collected by dipping sterile 120 mL vessel into the stream and allowing the bottle to fill to at least 100 mL.

3.1.2 Groundwater Characterization

A nest of 2 piezometers was installed in a centralized area of the wetland to assess groundwater levels and groundwater quality. Soil augers were used to create boreholes and the depth to groundwater was noted. Piezometers were constructed by coupling well screen (~ 0.75 m length) to schedule 40 polyvinyl chloride with 5 cm-diameter casing. Deep (~ 3 m) and shallow (1.2 m) depth piezometers were installed. Sand was poured in the annular space surrounding the well screen and the rest of the annular space above the screen and to the soil surface was filled with bentonite and sand mixture for each piezometer. Depth to water in each piezometer was measured using a Solinst TLC meter. Groundwater levels in the 2 piezometers were compared during each field event. The wetland was considered a groundwater recharge zone when water elevations in the shallow piezometer were greater than water levels in the deep piezometer. A discharge zone was indicated when water levels in the deep piezometer were

greater than in the shallow piezometer. Groundwater samples were collected from the piezometers using disposable bailers tied to string. The piezometers were purged several times, allowed to recharge, and then water was transferred from the bailer to sample bottles. The bottles were labeled with the date of collection and sample location. Surface and groundwater sample bottles were placed in a cooler with ice and transported to the labs for analyses.

3.1.3 Nutrient Analyses

Dissolved and particulate forms of N and P were analyzed at the East Carolina University Environmental Research Lab for concentrations (Appendix D). Samples were vacuum-filtered to remove suspended solids. The material on the filter was used to analyze particulate N (PN) and particulate P (PP), and TSS, while the filtrate was analyzed for the dissolved species. A *Unity Smartchem 170* color spectrometer was used to determine the concentrations of $\text{NO}_3\text{-N}$, $\text{NH}_4\text{-N}$, total dissolved P (TDP), and $\text{PO}_4\text{-P}$. A *Shimadzu* TOC/TN analyzer was used to determine concentrations of total dissolved N (TDN) for each sample. TP was calculated by adding TDP to PP. TN was calculated by adding TDN and PN. TDN is comprised of dissolved inorganic N (DIN) and dissolved organic N (DON). DIN was the sum of $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$.

3.1.4 Fecal Indicator Bacteria

Water samples were analyzed for concentrations of *E. coli* at the ECU Environmental Health Water Research Lab within 6 hours of collection. *IDEXX Colilert* media was added to each microbe sample bottle, and thoroughly shaken until the media dissolved. The dissolved media was poured into an *IDEXX Quanti-Tray* that was labeled with the sample identification, the time placed in incubator, and dilution factor when applicable. Sealed trays were placed into a *Fisherbrand* Isotemp and incubated for 24 hours at 35 °C. After incubation, the trays were placed under a blacklight and the number of large and small wells that fluoresced were counted

and compared to the *IDEXX* chart to enumerate *E. coli* concentrations in MPN 100 mL⁻¹ or most probably number per 100 mL- comparable to colony forming units (CFU). If diluted, the *E. coli* concentrations were multiplied by the dilution factor and the concentration was recorded.

Dilutions were performed on samples where results were expected to the maximum detection limit of > 2419.6 MPN 100 mL⁻¹. Scenarios likely to yield max results would include major stormflow events that carried bacteria from up-gradient, where bacteria is trapped even as water dried out, as well as where shaded areas prevent UV light deactivating bacteria.

3.1.5 Total Suspended Solids Analyses

Pre-weighed 1.5µm-pore size 934-AH filters were assigned to each water sample and deionized blank, then vacuum filtered to collect suspended solids from recorded volumes collected in the field. These filters with solids were placed into aluminum boats, with assigned numbers. To remove variability of boat and filter weight, each boat and filter was weighed and later subtracted from the total weight. Samples were then vacuum-filtered and heated at 110°C for 24-hours and weighed once every 24 hours until the dry weight of consecutive weightings was within 5%. TSS concentration was calculated by subtracting the final measured weight by the weight of the filter and dividing by the volume of sample filtered.

3.2. Pollutant Loading

Flow (L s⁻¹) was multiplied by the concentration of nutrients (mg L⁻¹), TSS (mg L⁻¹), and *E. coli* (MPN L⁻¹) to determine pollutant masses or loadings entering and exiting the wetland during each monthly sampling event. Total inflow loading was calculated by adding loads from I-1 and I-2. Total export was calculated by adding exports from O-1, O-2, and O-3. Months when the inlets and outlets were dry, the flow and load/export was reported as “0”. The wetland was

considered a sink when loads entering the wetland exceed exports leaving the wetland. The wetland was considered a source when loads exiting the wetland exceeded inputs.

3.2.1 High-frequency monitoring

HOBO water level loggers (Onset Corporation) were deployed at I-1 and I-2 at the beginning of the study and retrieved and downloaded every 6 months (Figure 5). The loggers were programmed to measure water pressure every 20 minutes. A logger was tied to a tree to measure atmospheric pressure, which was used to correct for water level logger readings related to atmospheric pressure changes. Stilling wells were installed with stage plates attached in which loggers were deployed in them at I-1 and I-2, allowing stream stage to be manually measured during each monthly sampling event. The manual measurements were used to calibrate the water level logger stage readings. The minimum, maximum, and median stream stage based on manual readings and automated logger readings were compared to gain perspective on how representative the monthly manual stage readings were in relation to high-frequency measurements. *Hanna Instruments* 9829 multiprobe sonders were deployed at I-1 and I-2 on 5 different occasions to collect high-frequency (every 10 minutes) physicochemical data including pH, DO, ORP, temperature, SC, and turbidity. The loggers were typically deployed at the start of a field sampling event and removed at the end of the event. On 2 occasions, the loggers were deployed for up to a week. The initial readings at the start of the day were compared to the high-frequency data collected during the deployment to assess variability of the parameters and help determine if the discrete samples were representative of the site for a longer period (6 hours to ≤ 7 days).



Figure 5. Staff gauge with a water-depth logger installed in it.

3.3 Statistical Analyses

The flow-weighted average of physicochemical properties of inflow (I-1 and I-2) and outflow (O-1, O-2, O-3) were calculated for each sampling event and compared to determine if statistically significant differences ($p < 0.05$) were observed in physicochemical properties of water entering and exiting the wetland. Anderson-Darling tests were performed using *Minitab 18* statistical software on water quality data to determine if they demonstrated a normal distribution. If the distribution of the data was not normal, the data were log-transformed, and the tests were repeated. If transformed data exhibited normality, then t-tests were used on the transformed data to determine if differences in pollutant loads entering and exiting the wetland were significantly different. If transformed data did not show a normal distribution, Mann-Whitney tests were used to determine if differences were statistically significant between the inflow and outflows, and

also between the shallow and deep groundwater samples. Concentrations of *E. coli* entering and exiting the wetland were compared to recreational water quality standards and the frequency of exceedance were reported. The 2012 Recreational Water Quality Criteria (RWQC) states that 36 illnesses per 1,000 recreators may occur if the geometric mean (GM) of *E. coli* exceeds 126 cfu/100 mL and if more than 10% of samples exceed the statistical threshold value (STV) of 410 cfu/100 mL (EPA, 2015).

CHAPTER 4: RESULTS

4.1 Wetland Hydrology

4.1.1 Inflow vs. Outflow Hydrology

Stream stage varied during the study (Figure 6) and was likely influenced by factors such as rainfall and season (Figure 7A, 7B and 7C). The median outflow from the wetland (3.30 L s^{-1}) was 33.2% lower relative to the median inflow (4.94 L s^{-1}), but differences in flow were not statistically significant ($p = 0.57$) (Appendix C). The Mann-Whitney test was used in all results due to data being non-normal (Appendix E).

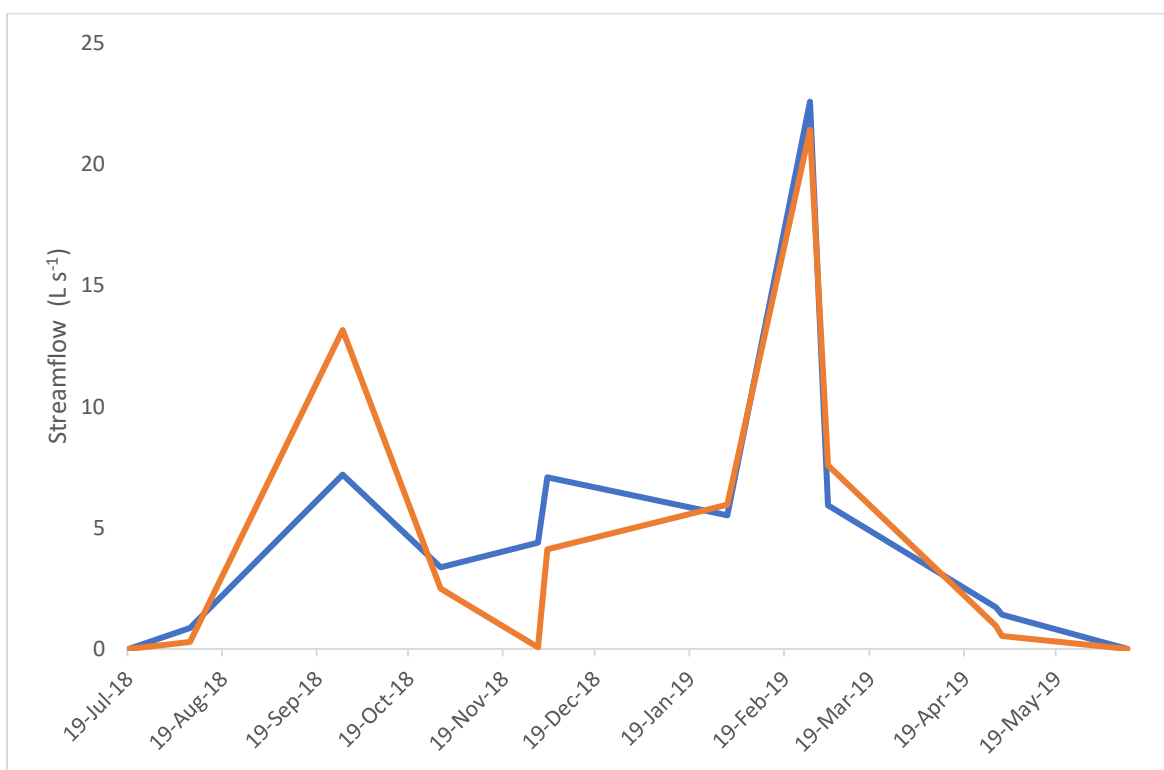


Figure 6. Wetland flow (L s^{-1}). Blue represents inflow and orange represents outflow.

Outflow was lower relative to inflow during 70% of sampling events when flow was measurable. The overall median stage based on water level logger data for I-1 and I-2 was 6.9 cm; the median stage based on manual monthly readings was 11.75 cm for I-1 and 19.5 cm for I-2. I-1 and I-2

medians were 8.7 cm and 24.6 cm respectively, when comparing high-frequency stage readings only during dates when field sampling occurred (Figure 7A).

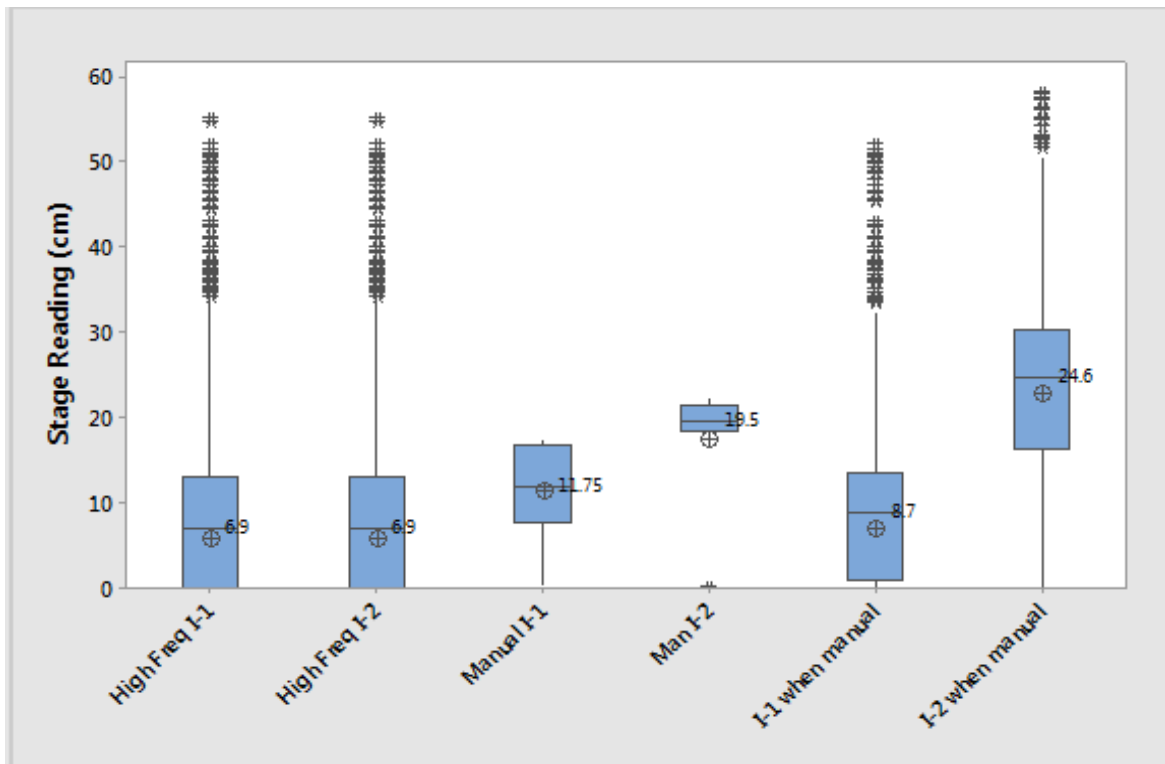


Figure 7A. Stage readings: high-frequency loggers, manual, and logger readings when manual measurements were taken. Crosshairs indicate mean.

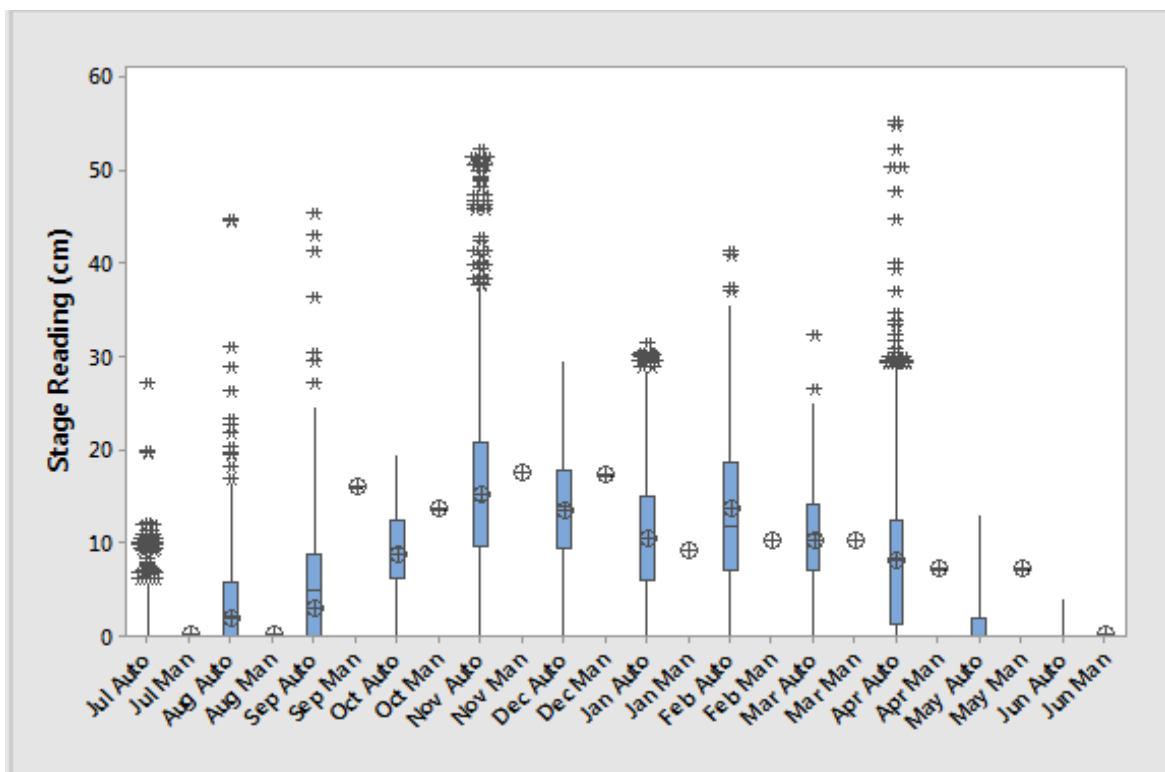


Figure 7B. Inlet-1 high-frequency and manual water level data. Single horizontal dash lines indicate single manual readings. Crosshairs indicate mean.

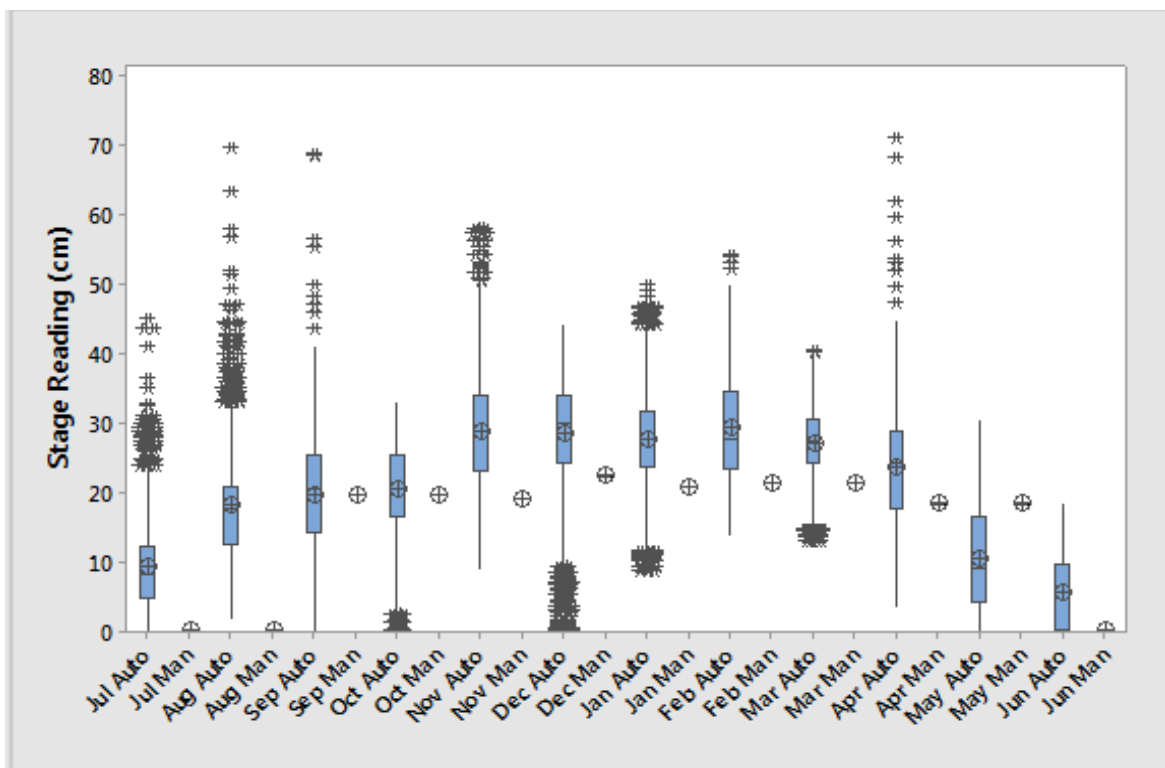


Figure 7C. Inlet-2 high-frequency and manual water level data. Single horizontal dash lines indicate single manual readings. Crosshairs indicate mean.

During the months of July 2018 and June 2019, flow was too low to measure, which may have been due to limited rainfall in the days preceding the sampling events (Appendix A). The sampling events with highest flow were in the fall and winter, when there were cooler temperatures, less day light, and typically lower evapotranspiration rates. More specifically, the highest median stage values based on logger data were seen in November and December 2018 (Appendix A). The single highest flow measured during monthly sampling occurred in winter (February 2019) for both inflow and outflow, at 22.6 L/s and 21.4 L/s respectively (Appendix C). The lowest manually measured flow (inflow and outflow) occurred during a few summer months when the streams were dry and flow was 0 L s⁻¹. There was an almost even proportion of total flow into the wetland from I-1 (48.62%) and I-2 (51.38%). However, the majority of outflow was attributed to O-3 (96.4%), with a much smaller % of flow leaving O-1 (1%) and O-2 (2.6%) (Appendix C). Minimum and maximum manual stage readings for I-1 were 0 cm and 17.4 cm, respectively, and 60.3% of I-1 high-frequency readings (15081 of 25019) were within this range. I-2 minimum and maximum were 0 cm and 21.3 cm, respectively, and 43.6% of I-2 high-frequency readings (10904 of 25019) were within the range. This is likely because of fluctuation in factors such as rainfall and stormflow. I-2 receives runoff from more impervious surface (an urban neighborhood), whereas I-1 primarily receives runoff from an upstream rural catchment. Monthly sampling missed some of the stormflow events.

4.1.2 Groundwater Hydrology

Median groundwater level within the shallow piezometer was significantly higher ($p = 0.01$) in relation to groundwater level in the deep piezometer, suggesting recharge was occurring in the wetland during the study. The median depth to water in the shallow and deep piezometers were 4.88 cm and 15.85 cm below ground surface, respectively (Table 2). Water levels in the

piezometers varied monthly due to precipitation, evapotranspiration, infiltration, and runoff (Appendix I).

Table 2. Median groundwater levels.

Medians	Depth to Water (cm) from Top of Piezometer Casing	Depth to Water (cm) from Ground Surface
Piezometer-S	55.17	4.88
Piezometer-D	110.95	15.85

4.2 Physicochemical Properties

4.2.1 Surface Water Quality

The main differences observed in surface water physicochemical properties were seen in ORP, SC, and turbidity. The median ORP of wetland outflow (8 mV) was 58 mV lower than the median ORP of wetland inflow (66 mV) ($p = 0.02$), and outflow ORP was less than the inflow 90% of the time (Appendix G). The median SC ($296.0 \mu\text{S cm}^{-1}$) of wetland outflow was $51.5 \mu\text{S cm}^{-1}$ lower than the median inflow specific conductance ($347.5 \mu\text{S cm}^{-1}$) ($p = 0.26$), and outflow specific conductance was less than that of inflow 70% of the time (Appendix G). The median turbidity of wetland outflow (33 NTU) was 14 NTU greater than the median wetland inflow turbidity (19 NTU) ($p = 0.16$), and outflow turbidity was greater than inflow 90% of the time (Appendix G).

Overall, DO, pH, and temperature readings for inflow and outflow were similar. Median DO concentrations at the wetland outlet (9.09 mg L^{-1}) were slightly elevated as compared to the inlet DO (8.41 mg L^{-1}), but differences were not statistically significant ($p = 1.0$), and outflow DO was greater than that of inflow 50% of the time (Appendix G). Median pH of wetland

outflow (6.9) based on monthly manual readings was similar the median pH of inflow (6.7) ($p = 0.57$), and outflow pH was greater than that of inflow 60% of the time (Appendix G). Median wetland outflow temperature (14.15 °C) was similar to that of inflow (13.64 °C) ($p = 0.97$) and outflow temperature was greater than inflow 50% of the time (Appendix G). Different brands of field meters were used for manual (YSI and HACH) and automated (Hanna Instruments) physicochemical parameter determination. However, high-frequency logger data showed similar trends to manual readings data (Table 3A, Table 3B). The high frequency data show that inflow into the wetland has a near neutral pH (6.7 to 7.6), and dynamic oxidation reduction potential status (ORP 27 to 498 mV), DO concentrations (2.2 to 21.2 mg L⁻¹), temperature (3.2 to 20.4 C°), and turbidity (4.4 to 924 FNU) (Table 3A and 3B). Physicochemical readings varied during different deployments.

4.2.2 Groundwater Quality

The median pH of groundwater samples collected each month from the deep piezometer (6.6) was similar to the median pH of groundwater collected from the shallow piezometer (6.5) ($p = 0.63$) (Table 1), and shallow piezometer pH readings were greater than those in the deep 63% of the times sampled (Appendix J). The median ORP of groundwater from the deep piezometer (-91 mV) was 148 mV less than groundwater samples collected from the shallow piezometer (57 mV) ($p = 0.13$), and groundwater from the shallow piezometer had ORP values greater than that of the deep 75% of the time (Appendix J). The median temperature of groundwater samples collected monthly from the deep piezometer (15.1°C) differed by only 0.02°C from the groundwater samples collected from the shallow piezometer (14.9°C) ($p = 0.86$), and shallow piezometer groundwater temperatures were less than those of the deep 75% of the time sampled. The median DO of groundwater samples collected from the deep piezometer (3.59 mg L⁻¹) differed by only 0.04 mg L⁻¹ from the groundwater samples collected from the shallow piezometer (3.55 mg L⁻¹) ($p = 0.93$). Groundwater sampled from the shallow piezometer had DO readings greater than those from the deep piezometer 50% of the time (Appendix J). The median specific conductance of groundwater samples collected from the deep piezometer (550 $\mu\text{S cm}^{-1}$) was 317 $\mu\text{S cm}^{-1}$ higher than the groundwater samples collected from the shallow piezometer (233 $\mu\text{S cm}^{-1}$) ($p = 0.001$), and shallow piezometer specific conductance readings were less than those in the deep 100% of the time (Appendix J).

Table 3. Wetland physicochemical medians and ranges (min-max). ORP units are mV, temperature in °C, dissolved oxygen (DO) in mg L⁻¹, specific conductivity (SC) in $\mu\text{S cm}^{-1}$, turbidity in NTU. The piezometer groundwater data shallow (S) and deep (D). Inlet and Outlet refer to the wetland.

Location	SC	DO	ORP (mV)	pH	Temperature	Turbidity
Piezometer (S)	233 (202-350)	3.55 (2.62-5.8)	57 (-95-106)	6.5 (6.2-7.02)	14.9 (5.14-23.14)	N/A
Piezometer (D)	550 (275-612)	3.59 (1.9-5.5)	-91 (-158-134)	6.56 (5.5-7.5)	15.1 (7.18-17.9)	N/A
Inlet	347 (168-525)	8.41 (1.97-13.26)	*66 (-61-120)	6.7 (6.0-8.0)	13.64 (3.37-25.00)	19 (6-132)
Outlet	296 (153-434)	9.08 (2.08-12.34)	*8 (-49-74)	6.9 (6.5-7.3)	14.15 (4.07-24.4)	33 (13-174)

*Note: Asterix denotes statistical significance at $p \leq 0.10$.

Table 3A. Inlet-1 high-frequency physicochemical parameters. “F” indicates first readings and “C” indicates continuous readings along with respective median and range (minimum, maximum). ORP units are mV, temperature in °C, dissolved oxygen (DO) in mg L⁻¹, specific conductivity (SC) in $\mu\text{S cm}^{-1}$, turbidity in NTU .

Deploy #	Duration	Month(s)	F-pH	C-pH	F-ORP	C-ORP	F-Temp	C-Temp	F-DO	C-DO	F-SC	C-SC	F-Turb	C-Turb
1	5 hours	10/29/2018	7	7.09 (6.76-7.14)	160.4	153.45 (149.70-171.60)	12.88	13.55 (12.88-13.78)	5.44	4.12 (3.63-5.81)	568	552.50 (551.00-568.00)	45.6	25.80 (17.20-45.60)
2	5.5 hours	12/03/2018	7.62	6.94 (6.73-7.62)	442.3	449.20 (429.80-487.20)	15.1	16.29 (14.40-20.37)	3.8	3.77 (3.14-4.84)	343	348 (198-354)	7.6	7.55 (5.90-46.00)
3	4 hours	01/31/2019	7.27	7.21 (7.17-7.27)	80.6	77.75 (76.30-80.60)	3.89	4.83 (3.89-5.19)	21.18	14.51 (13.71-21.18)	236	242 (236-244)	7.3	8.30 (7.20-16.90)
4	7 days	02/27/2019-03/05/2019	6.9	6.91 (6.48-7.04)	161.3	463.50 (79.70-498.50)	8.72	9.16 (6.96-12.28)	10.11	7.91 (6.20-12.10)	108	117 (9-219)	76.8	29.80 (10.80-924.00)
5	3 days	04/29/2019-05/01/2019	7	7.1 (7-7.2)	14.7	204.7 (8.9-357)	15.7	17.3 (15.6-20)	2.17	1.5 (1.1-2.2)	246	251 (231-289)	12.8	7.1 (4.4-37.6)

Table 3B. Inlet-2 high-frequency physicochemical parameters. “I” indicates initial readings and “C” indicates continuous readings along with respective median and range (minimum, maximum). DO and pH probes failed on deployment 4, and ORP had a very limited number of readings throughout the deployment period. The automated logging probes for ORP, DO, and pH malfunctioned in April-May of 2019, so some of the high-frequency values were not included during that period. ORP units are mV, temperature in °C, dissolved oxygen (DO) in mg L⁻¹, specific conductivity (SC) in $\mu\text{S cm}^{-1}$, turbidity in NTU

Deploy #	Duration	Month(s)	F-pH	C-pH	F-ORP	C-ORP	F-Temp	C-Temp	F-DO	C-DO	F-SC	C-SC	F-Turb	C-Turb
1	4 days	11/30/2018-12/03/2018	6.73	7.18 (6.73-7.49)	40.3	383.40 (27.40-489.90)	9.17	12.07 (9.08-18.53)	4.68	3.31 (2.62-5.62)	647	461 (22-683)	32.6	19 (6.80-60.10)
2	5.5	01/31/2019	7.17	7.54 (7.12-7.55)	102.8	95.80 (90.40-109.00)	3.2	3.59 (3.20-4.39)	8.1	8.94 (6.75-9.14)	432	463 (432-471)	52.2	14.1 (12.6-195)
3	7 days	2/27/2019-3/5/2019	6.83	7.08 (6.36-7.33)	369.2	441.85 (199.80-531.10)	8.53	8.83 (6.20-12.57)	10.29	9.90 (7.14-13.40)	229	202.50 (12-344)	71.9	46.05 (16.50-743)
4	3 days	4/29/2019-5/1/2019			167.2	232.40 (167.20-312.5-)	15.05	15.97 (15.05-18.24)			545	623 (545-689)	17.3	6.75 (4.50-70.60)

4.3 Nutrient, Sediment, and Bacteria

4.3.1 Surface Water Quality

4.3.1.1 Flow-Weighted Concentrations

The median TN concentration in wetland outflow (0.73 mg L⁻¹) was 0.39 mg L⁻¹ less than the median wetland inflow TN concentration (1.12 mg L⁻¹) ($p = 0.03$). The median PN concentration in wetland outflow (0.11 mg L⁻¹) was 0.01 mg L⁻¹ greater than the median PN concentration entering the wetland (0.10 mg L⁻¹) and differences in concentrations were not statistically significant ($p = 1.00$). The median TDN concentration in wetland outflow (0.59 mg L⁻¹) was 0.45 mg L⁻¹ less than the median TDN concentration in wetland inflow (1.04 mg L⁻¹) ($p = 0.02$) (Figure 8). Flow-weighted inflow TN concentrations were higher than outflow 90% of the time, PN 50% of the time, and TDN 90% of the time (Appendix H).

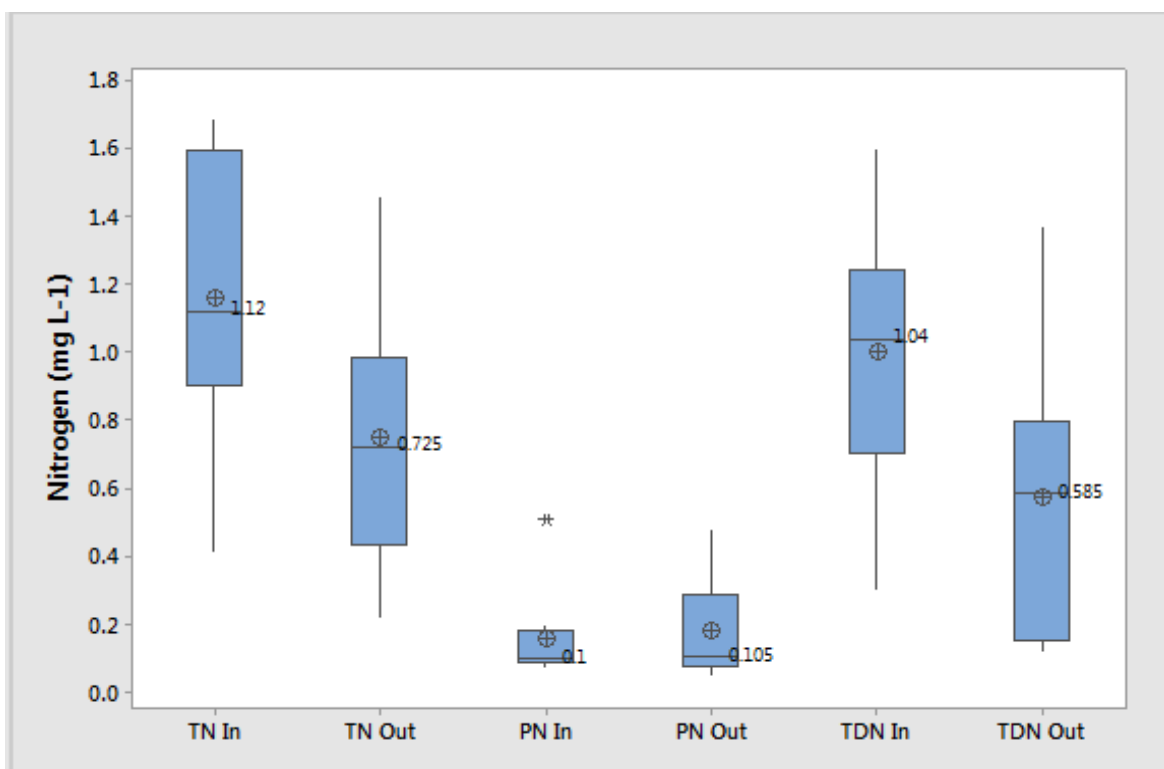


Figure 8. Flow-weighted concentrations of nitrogen by inflow (In) and outflow (Out). Flat lines indicate medians, crosshairs indicate means, and stars represent outlier data. Numerical values are medians. Values are labeled. TN= total nitrogen; PN= particulate nitrogen; TDN= total dissolved nitrogen.

The median $\text{NO}_3\text{-N}$ concentration in wetland outflow (0.44 mg L^{-1}) was 0.38 mg L^{-1} less than the median wetland inflow concentration (0.82 mg L^{-1}) ($p = 0.05$). The median $\text{NH}_4\text{-N}$ concentration in wetland outflow (0.12 mg L^{-1}) was 0.07 mg L^{-1} lower in comparison to inflow (0.19 mg L^{-1}) ($p = 0.21$) (Figure 9). Flow-weighted $\text{NO}_3\text{-N}$ concentrations entering the wetland were higher than outflow 90% of the time. Concentrations of $\text{NH}_4\text{-N}$ in the inflow were higher 60% of the time in comparison to outflow concentrations (Appendix H).

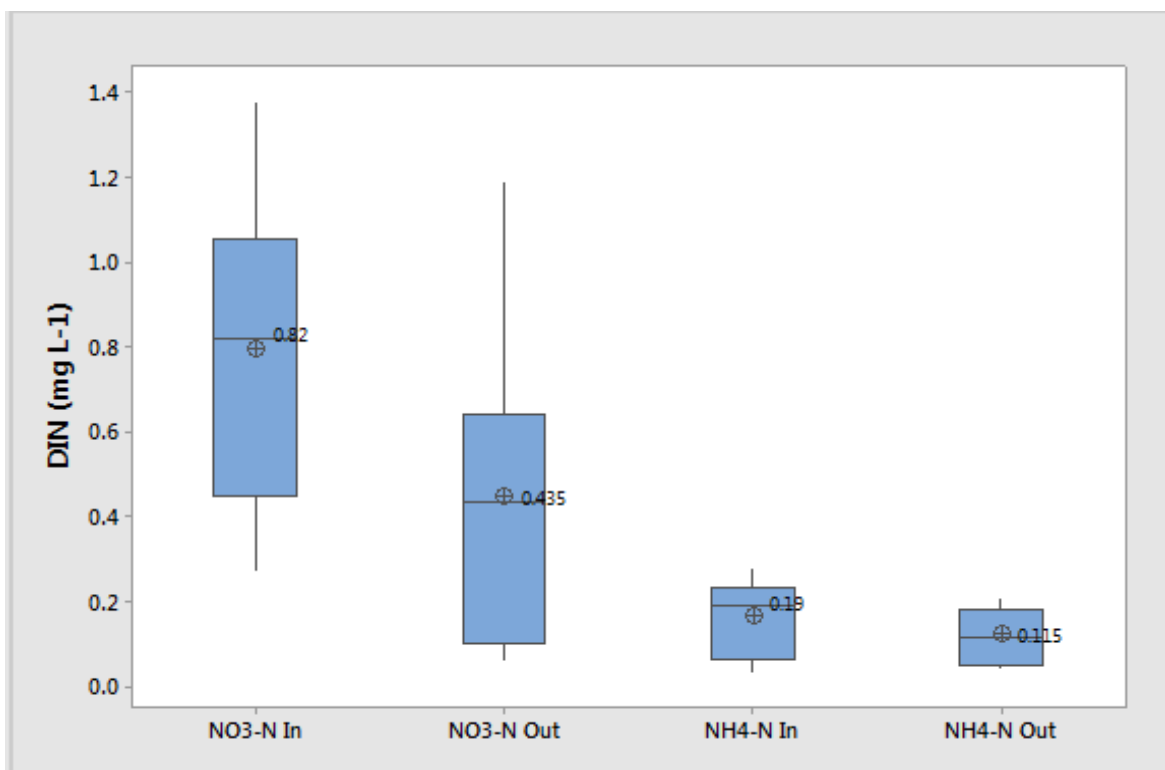


Figure 9. Flow-weighted nitrate-nitrogen (NO₃-N) and ammonium-nitrogen (NH₄-N) concentrations by inflow (In) and outflow (Out). Flat lines indicate medians, crosshairs indicate means, and stars represent outlier data. Median values are labeled. DIN= dissolved inorganic nitrogen.

Median flow-weighted concentration of TDN decreased from inflow to outflow, whereas median PN increased. (Figure 10A). There were similar (within 2%) percentages of DIN as NO₃-N and NH₄-N in wetland inflow and outflow (Figure 10B).

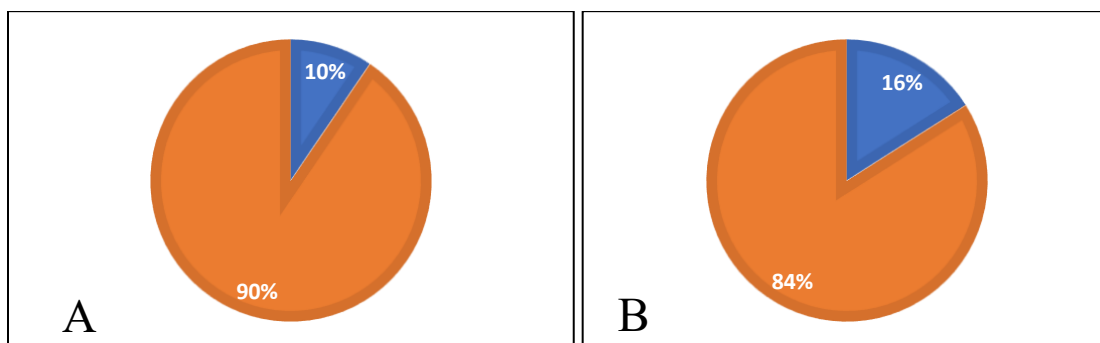


Figure 10A. Inflow (A) vs. outflow (B) TN (total nitrogen) breakdown. Blue represents PN (particulate nitrogen) and orange represents TDN (total dissolved nitrogen).

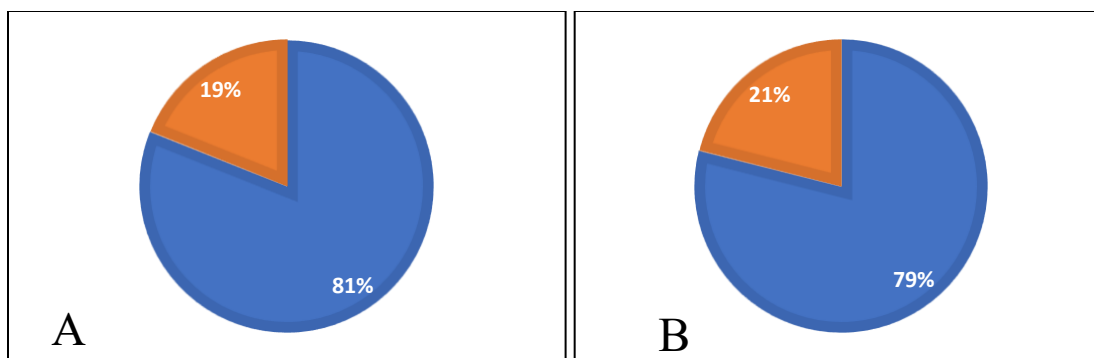


Figure 10B. Inflow (A) vs. outflow (B) DIN (dissolved inorganic nitrogen) breakdown. Blue represents NO₃-N (nitrate-nitrogen) and orange represents NH₄-N (ammonium-nitrogen).

The median PP concentration in wetland outflow (0.058 mg L⁻¹) was 0.005 mg L⁻¹ lower in comparison to inflow (0.063 mg L⁻¹) ($p = 0.50$), and the median PO₄-P concentration in outflow from the wetland (0.022 mg L⁻¹) was 0.007 mg L⁻¹ lower in comparison to inflow (0.029 mg L⁻¹) ($p = 0.24$) (Figure 11). Flow-weighted inflow PP concentrations were higher than outflow 60% of the time, PO₄-P inflow was higher 80% of the time (Appendix H).

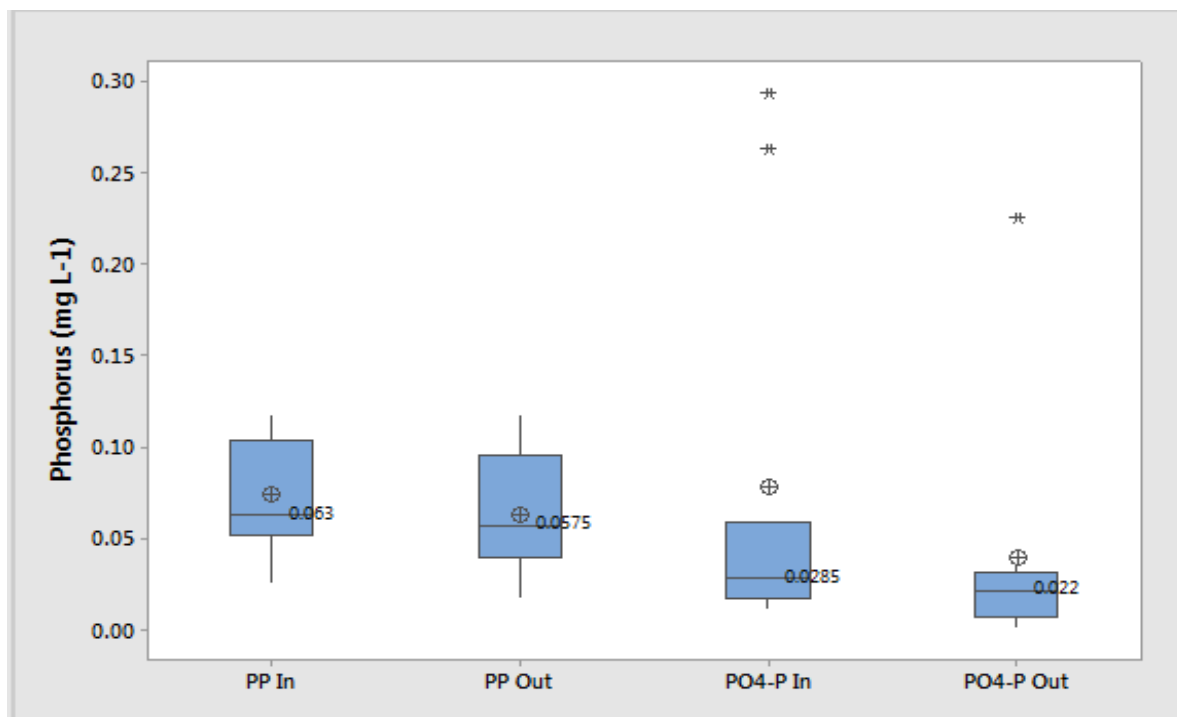


Figure 11. Flow-weighted concentrations of PP (particulate phosphorus) and PO₄-P (phosphate-phosphorus) by inflow (In) and outflow (Out). Flat lines indicate medians, crosshairs indicate means, and stars represent outlier data. Median values are labeled.

Concentrations of TDP were analyzed the final 5 months of the study, but due to zero flow on the June 2019 field day, only February 2019 through May 2019 sampling events were analyzed to measure TP. The median TP concentration in wetland outflow (0.482 mg L^{-1}) was 0.15 mg L^{-1} lower in comparison to inflow (0.497 mg L^{-1}). The median TDP concentration in wetland outflow during those 4 months TDP was analyzed (0.157 mg L^{-1}) was 0.016 mg L^{-1} lower in comparison to inflow (0.173 mg L^{-1}). The median PP concentration in wetland outflow during that time (0.325 mg L^{-1}) was similar to inflow (0.324 mg L^{-1}) (Figure 12A). Concentrations of TDP decreased 9% and PP increased 2% at the outlet relative to the inlet. During those 4 months, the median inflow concentration of $\text{PO}_4\text{-P}$ (0.090 mg L^{-1}) was 40% higher relative to the median concentration in wetland outflow (0.054 mg L^{-1}) (Figure 12B). For months TDP was analyzed, flow-weighted inflow TDP concentrations were higher than outflow 75% of the time, PP inflow was higher 40% of the time, $\text{PO}_4\text{-P}$ inflow was higher 60% of the time (Appendix H).

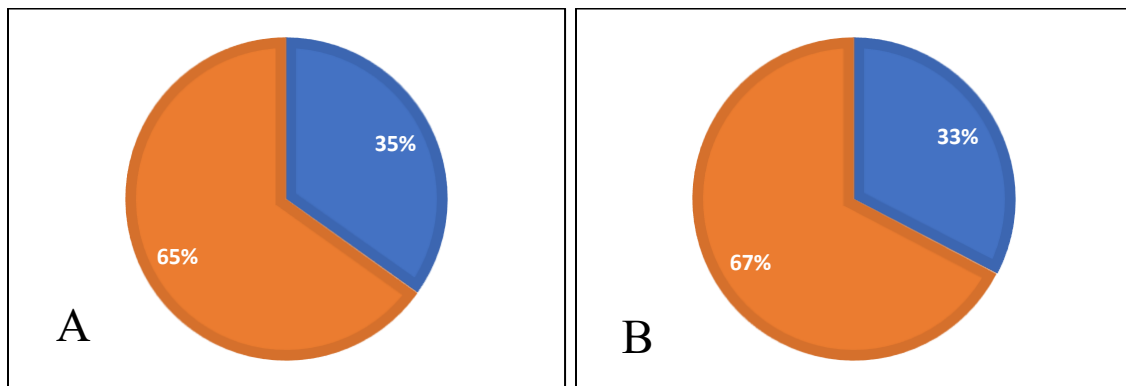


Figure 12A. Inflow (A) and Outflow (B) concentrations of total phosphorus (TP) by dissolved and particulate P species. Blue represents TDP (total dissolved phosphorus) and orange represents PP (particulate phosphorus). Inflow TDP (35%) and outflow TDP (33%), inflow PP (65%) and outflow PP (67%).

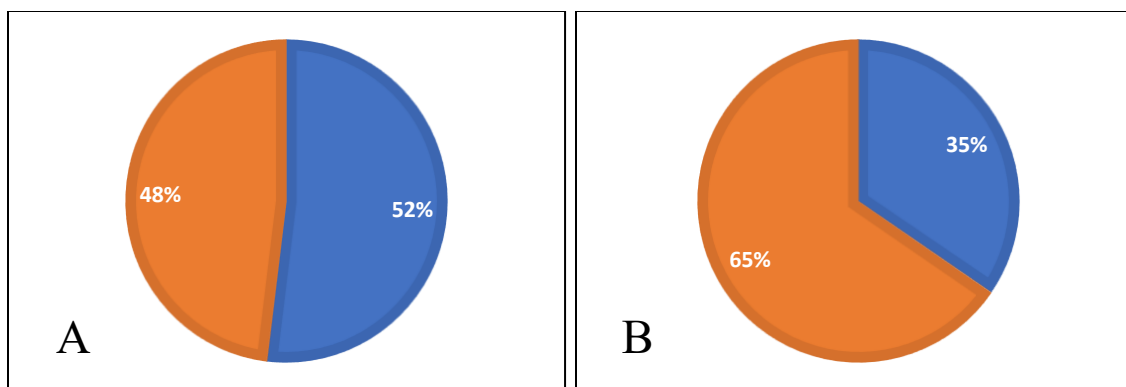


Figure 12B Inflow (A) and Outflow (B) concentrations of total dissolved phosphorus (TDP) species. Blue represents phosphate (PO₄-P) and orange represents other forms of TDP. Inflow PO₄-P (52%) and outflow PO₄-P (35%).

The median TSS concentration in wetland outflow (63.03 mg L⁻¹) was 20.34 mg L⁻¹ greater in comparison to inflow (42.69 mg L⁻¹) ($p = 0.24$); TSS outflow concentrations exceed that of inflow 7 of 10 events (Figure 13).

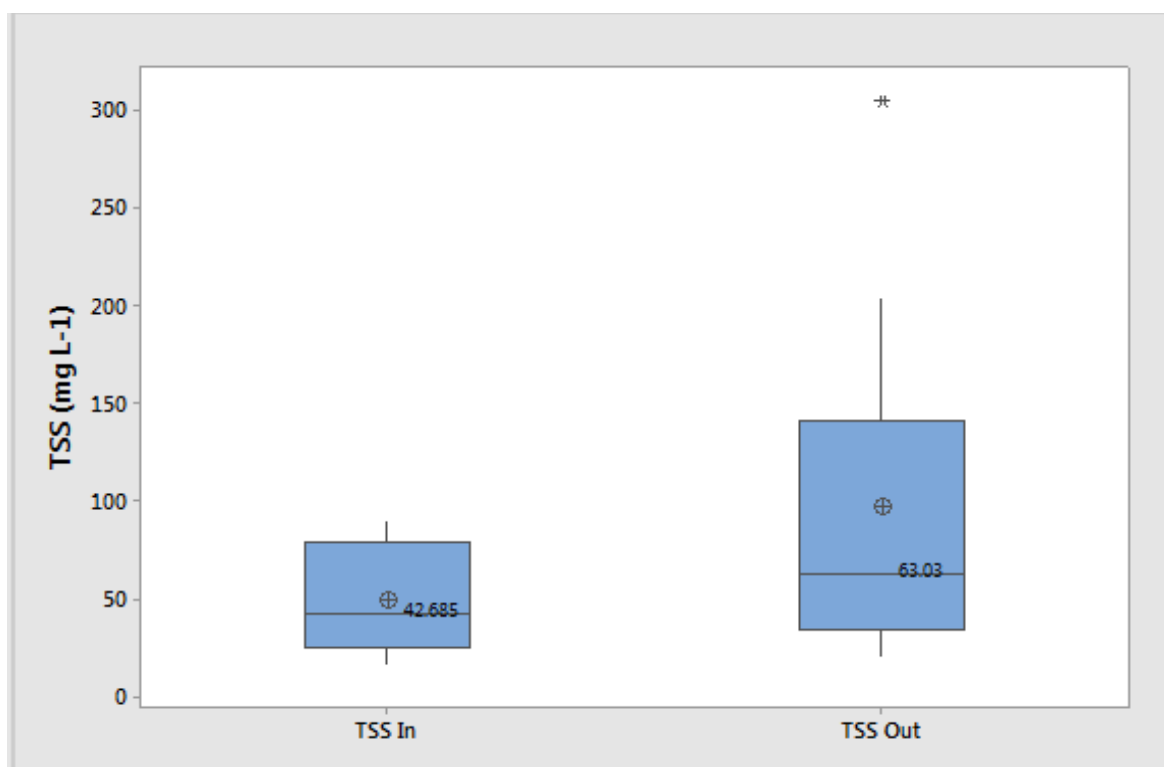


Figure 13. Flow-weighted concentration of TSS (total suspended solids) by inflow (In) and outflow (Out). Flat lines indicate medians, crosshairs indicate means, and stars represent outlier data. Median values are labeled.

The median of monthly manual turbidity readings at the wetland outlet (21.90 NTU) was 2.55 NTU greater in comparison to the median readings at the inlet (19.35 NTU) ($p = 0.39$) (Figure 14).

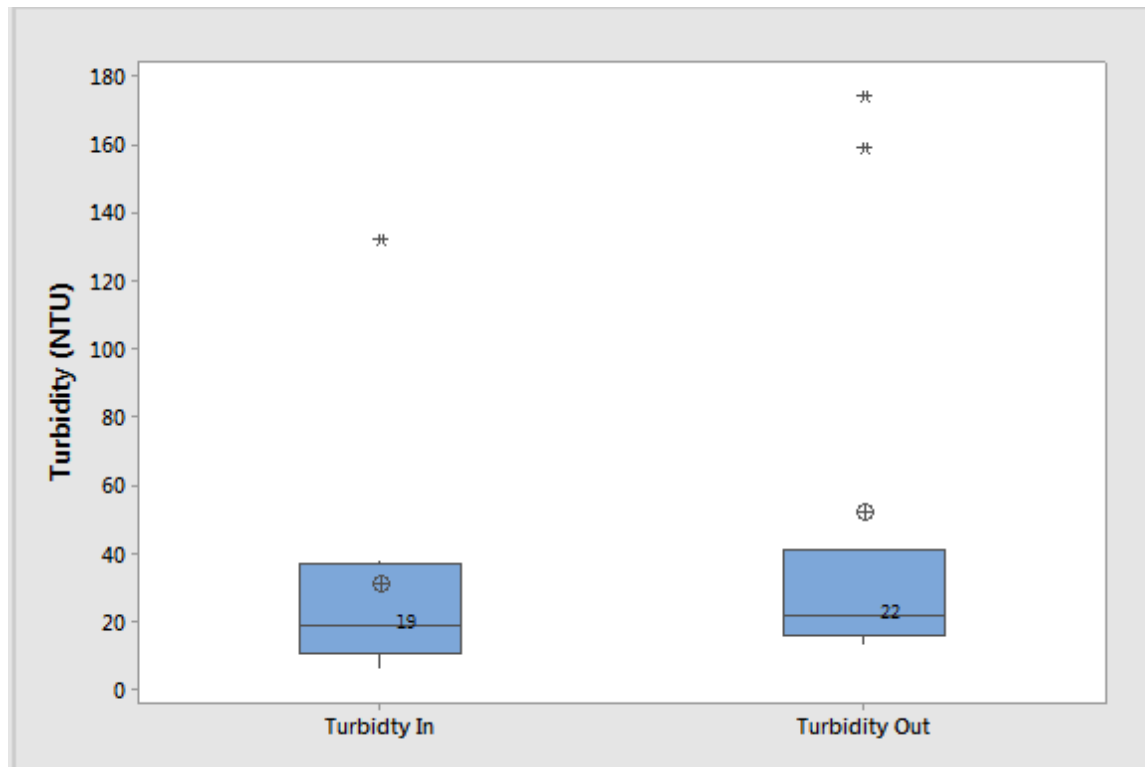


Figure 14. Flow-weighted value of turbidity by inflow (In) and outflow (Out). Flat lines indicate medians, crosshairs indicate means, and stars represent outlier data. Median values are labeled.

The median concentration of *E. coli* at the wetland inlet (1079 MPN 100 mL⁻¹) was approximately double the median concentration at the wetland outlet (545 MPN 100 mL⁻¹) ($p = 0.50$) (Figure 15). Flow-weighted *E. coli* concentrations were higher at the wetland inlet relative to the outlet during 50% of the times sampled (Appendix H).

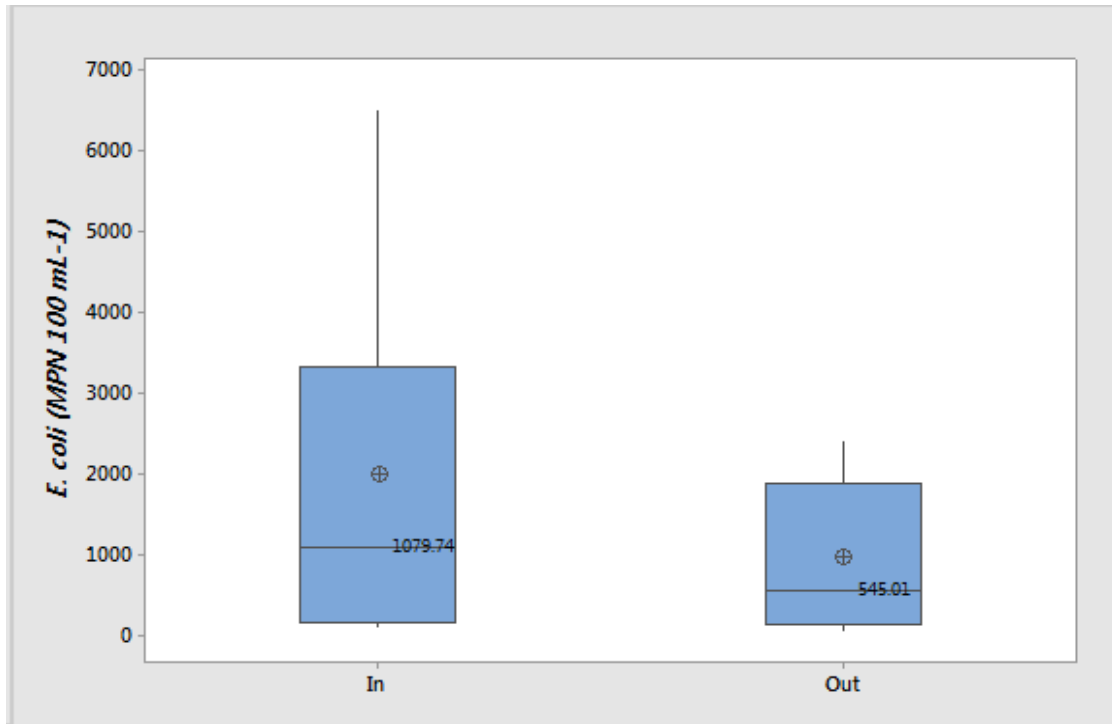


Figure 15. Flow-weighted *E. coli* by inflow and outflow. Flat lines indicate medians, crosshairs indicate means, and stars represent outlier data. Median values are labeled.

4.3.1.2 Mass Loadings

The median TN mass in wetland outflow (244.5 g D^{-1} , or grams per day) was 343.7 g D^{-1} less than wetland inflow (588.2 g D^{-1}) ($p = 0.21$). The TN loading was greater at the wetland inlet relative to the outlet 90% of the time ($n = 9$). The median PN mass exiting the wetland outlet (34.0 g D^{-1}) was similar to the mass entering at the wetland inlet (35.5 g D^{-1}) ($p = 0.79$). The loading of PN into the wetland was greater than the PN loads leaving the wetland 50% of the time ($n = 5$). The median TDN mass discharged from the wetland (220.5 g D^{-1}) was less than half the TDN loads entering the wetland (502.3 g D^{-1}) ($p = 0.19$) (Figure 16). Masses of TDN in wetland inflow were greater in comparison to wetland outflow 90% of the times sampled (Appendix K). Overall, TN and TDN masses were reduced though virtually no reductions in PN occurred.

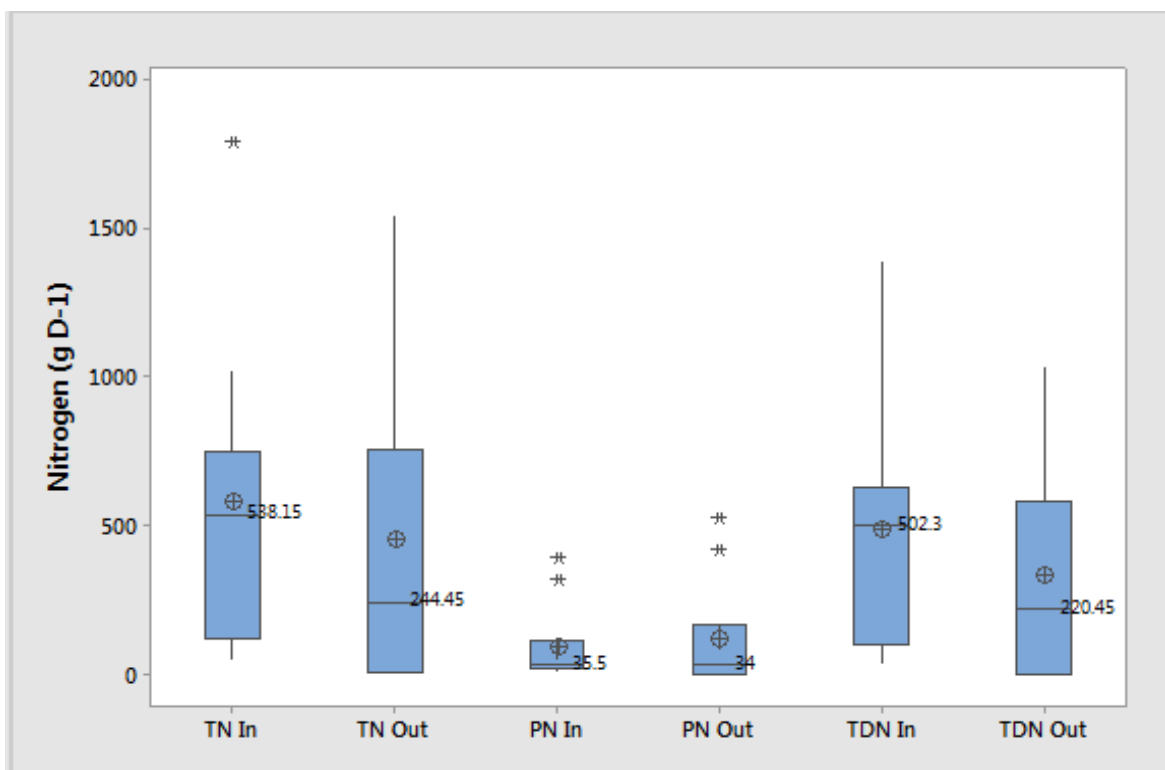


Figure 16. Nitrogen mass exports by inflow (In) and outflow (Out). Flat lines indicate medians, crosshairs indicate means, and stars represent outlier data. Median values are labeled. TN= total nitrogen; PN= particulate nitrogen; TDN= total dissolved nitrogen.

The median $\text{NO}_3\text{-N}$ mass in wetland outflow (179.8 g D^{-1}) was 169.8 g D^{-1} less than the median $\text{NO}_3\text{-N}$ mass entering the wetland (349.6 g D^{-1}) ($p = 0.21$) (Figure 14). The $\text{NO}_3\text{-N}$ mass entering the wetland was greater than the mass leaving the wetland 90% of the time. The median mass of $\text{NH}_4\text{-N}$ exiting the wetland (34.36 g D^{-1}) was 40.55 g D^{-1} less than the median $\text{NH}_4\text{-N}$ mass entering the wetland (74.91 g D^{-1}) ($p = 0.21$) (Figure 17). The $\text{NH}_4\text{-N}$ mass in wetland inflow was higher relative to the outflow 80% of the time (Appendix K).

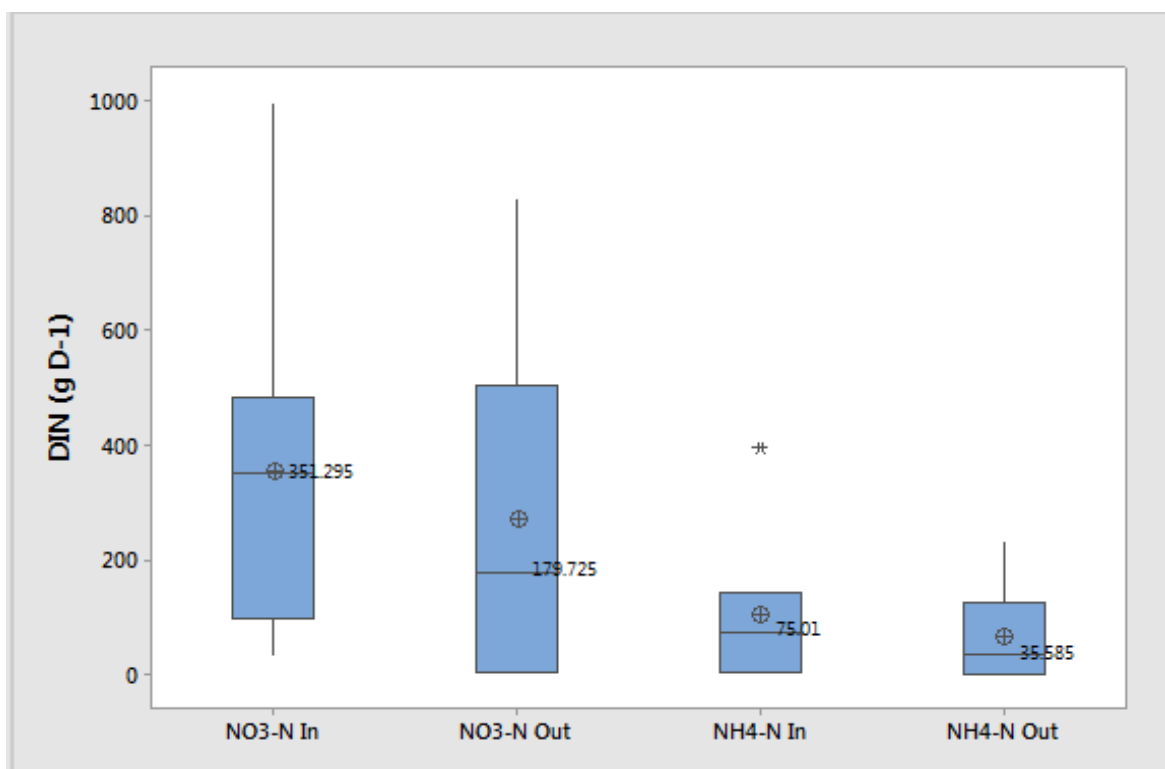


Figure 17. Dissolved inorganic nitrogen (DIN) mass exports by species for inflow (In) and outflow (Out). Flat lines indicate medians, crosshairs indicate means, and stars represent outlier data. Median values are labeled.

The decrease in TN mass was mostly seen in dissolved species, as PN increased as a component of TN in the exports (Figure 16A). The median percentage of DIN that was NO₃-N entering the wetland was lower relative to the percentage of DIN that was NO₃-N exiting the wetland. There was a higher percentage of the DIN that was NH₄-N in wetland inflow relative to outflow (Figure 16B).

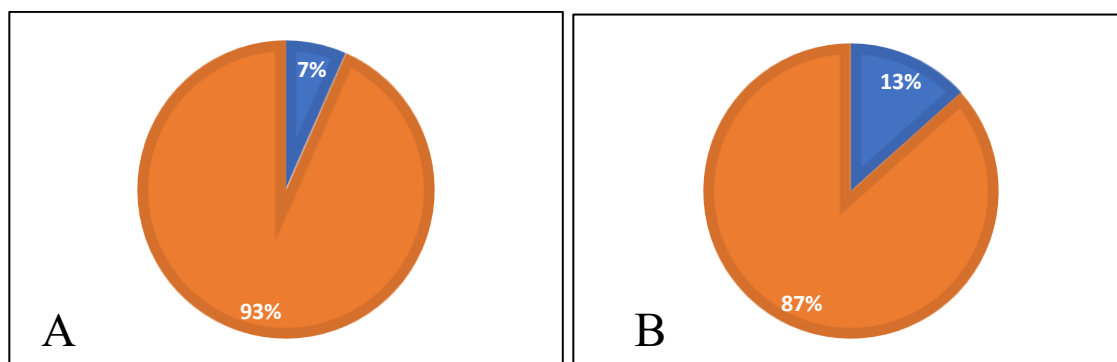


Figure 18A. TN (total nitrogen) mass by species for inflow (A) and outflow (B). Blue indicates PN (particulate nitrogen) and orange represents TDN/DIN (total dissolved nitrogen/dissolved inorganic nitrogen).

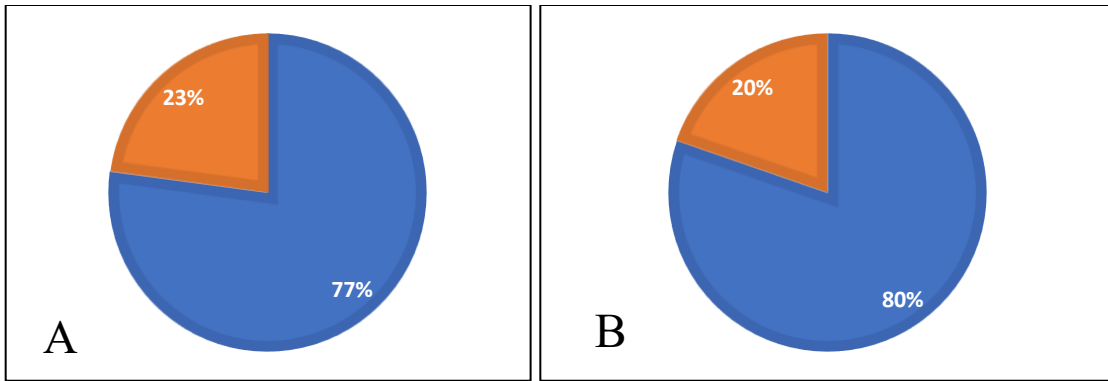


Figure 18B. Dissolved nitrogen mass export by species for inflow (A) and outflow (B). Blue indicates nitrate-nitrogen (NO₃-N) and orange represents ammonium-nitrogen (NH₄-N).

The median PP mass in wetland outflow (13.8 g D⁻¹) was 7.1 g D⁻¹ less than the median PP mass from wetland inflow (20.9 g D⁻¹) ($p = 0.47$) (Figure 17). The PP mass was greater at the wetland inlet relative to the outlet 80% of the time (Appendix L). The median PO₄-P mass leaving the wetland (5.2 g D⁻¹) was 8.8 g D⁻¹ less than the median PO₄-P mass entering the wetland (14.0 g D⁻¹) ($p = 0.19$) (Figure 19). The PP mass was greater at wetland inlet in comparison to the outlet 80% of the times sampled (Appendix L).

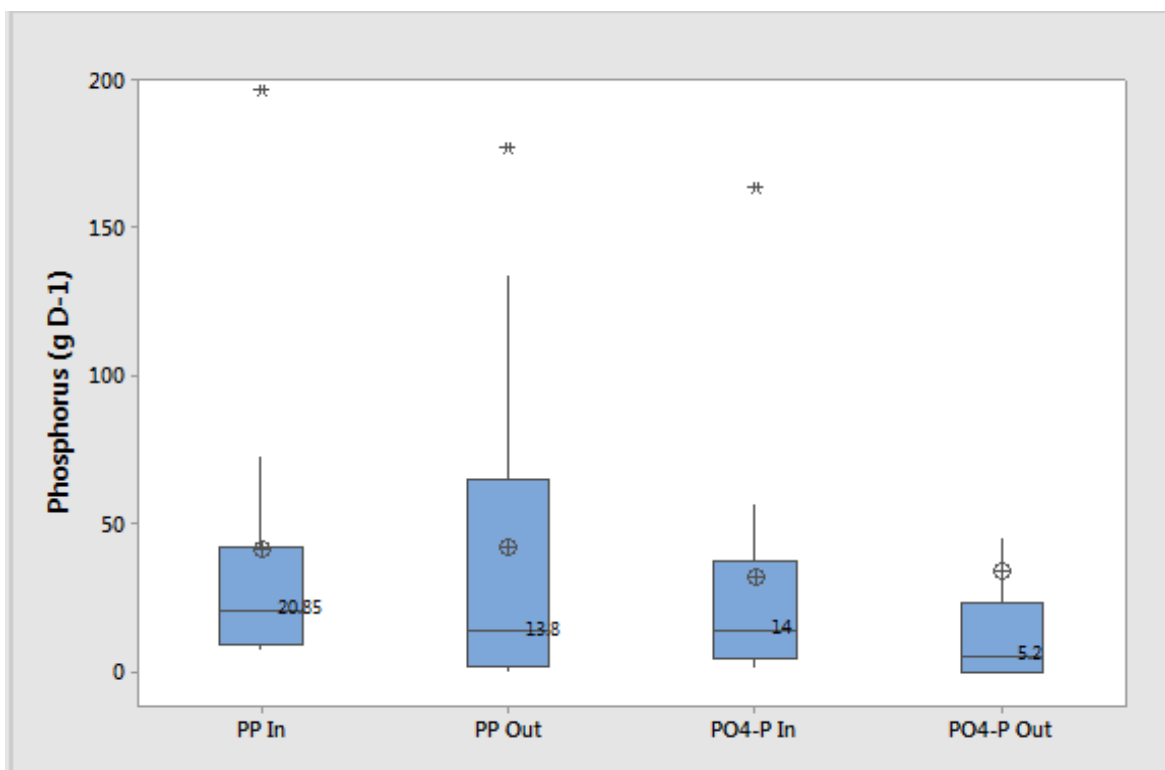


Figure 19. Particulate phosphorus (PP) and phosphate-phosphorus (PO₄-P) mass export by inflow (In) and outflow (Out). Scale was set to remove September 2018 PO₄-P (out) outlier of 255.0 (g D⁻¹) for increased visual acuity. Flat lines indicate medians, crosshairs indicate means, and stars represent outlier data. Median values are labeled.

During the 4 months TDP was analyzed, the median TP mass leaving the wetland (373.4 g D⁻¹) was similar to median TP mass entering the wetland (373.9 g D⁻¹) (Figure 18B). The median TDP mass leaving the wetland (249.3 g D⁻¹) was 124.7 g D⁻¹ greater than the median TDP mass entering the wetland (124.6 g D⁻¹) during the 4-month period. The median PP mass leaving the wetland (230.9 g D⁻¹) was 18.4 g D⁻¹ less than the median PP mass entering the wetland (249.3 g D⁻¹) (Figure 20A). TP mass was greater at the wetland inlet relative to the outlet 50% of the time, PP 75% of the time, and, PO₄-P 75% of the time (Appendix L). A higher percentage of the TP in wetland outflow was TDP (38%) relative to wetland inflow (32%) (Figure 20A). The percentage of TDP that was PO₄-P decreased 15% when comparing wetland inflow (59%) to wetland outflow (44%) (Figure 20B).

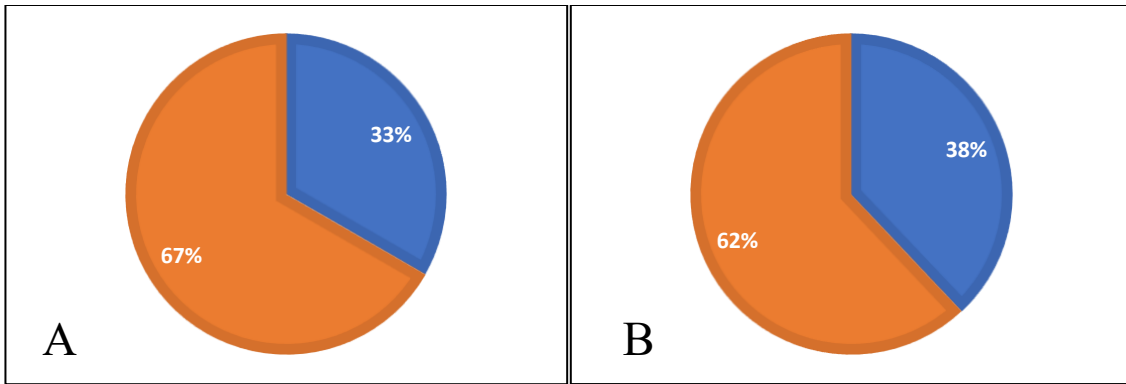


Figure 20A. Inflow (A) and Outflow (B) TP (total phosphorus) mass by particulate and dissolved. Blue represents TDP (total dissolved phosphorus) and orange represents PP (particulate phosphorus). Inflow TDP (33%) and PP (67%), outflow TDP (38%) and PP (62%).

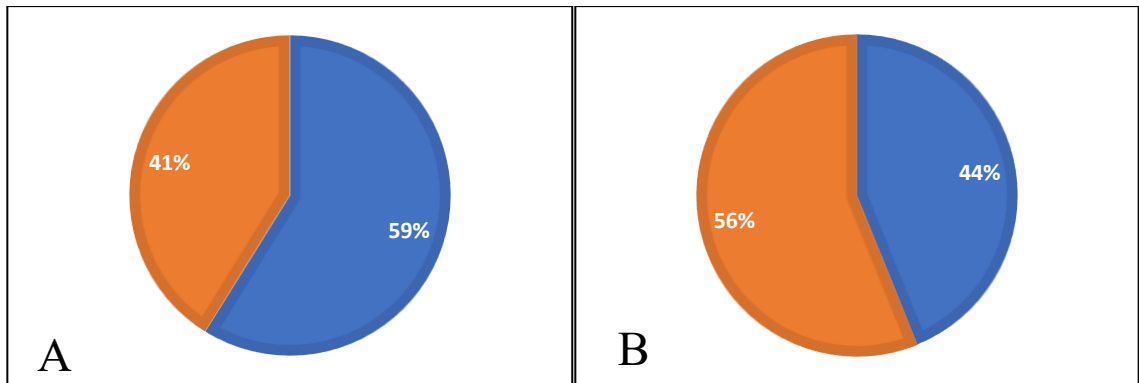


Figure 20B. PO₄-P (phosphate-phosphorus) component of TDP (total dissolved phosphorus) mass by inflow (A) and outflow (B). Blue represents PO₄-P and orange represents other forms of TDP. Inflow PO₄-P (59%) and outflow PO₄-P (44%).

The median TSS mass in outflow from the wetland (12316.3 g D⁻¹) was 1596.8 g D⁻¹ greater than the median TSS mass entering the wetland (10719.5 g D⁻¹) ($p = 0.85$) (Figure 21). During 50% of the sampling events, the TSS mass in wetland outflow was greater relative to the inflow (Appendix M).

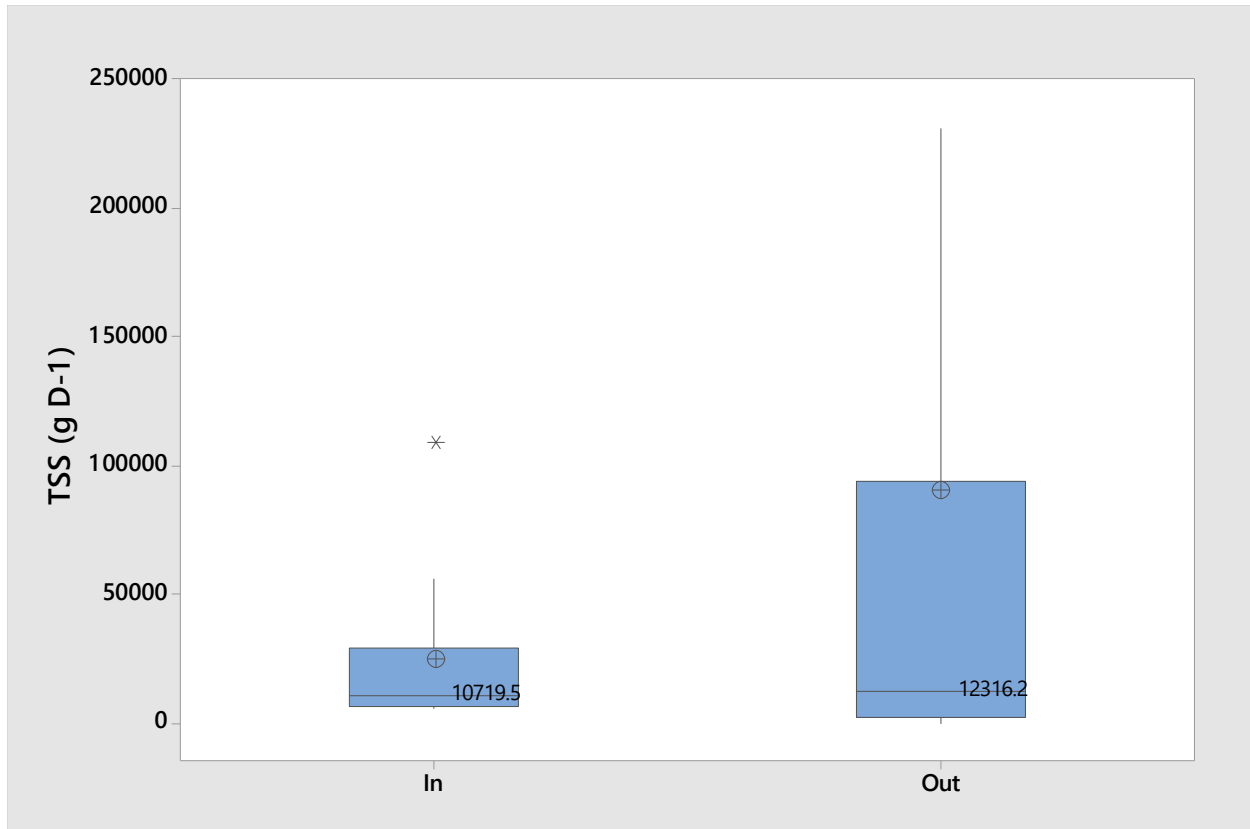


Figure 21. TSS (total suspended solids) mass by inflow (In) and outflow (Out). Scaled to remove outflow outlier of 562747 (g D⁻¹) from February 2019, for visual acuity. Flat lines indicate medians, crosshairs indicate means, and stars represent outlier data. Median values are labeled.

The median *E. coli* loading exiting the wetland (14379 MPN s⁻¹) was 35039 MPN s⁻¹ less than the median *E. coli* loading entering the wetland (49418 MPN s⁻¹) ($p = 0.31$) (Figure 22). *E. coli* loading was greater at the inlets relative to the outlets 50% of the time (Appendix N).

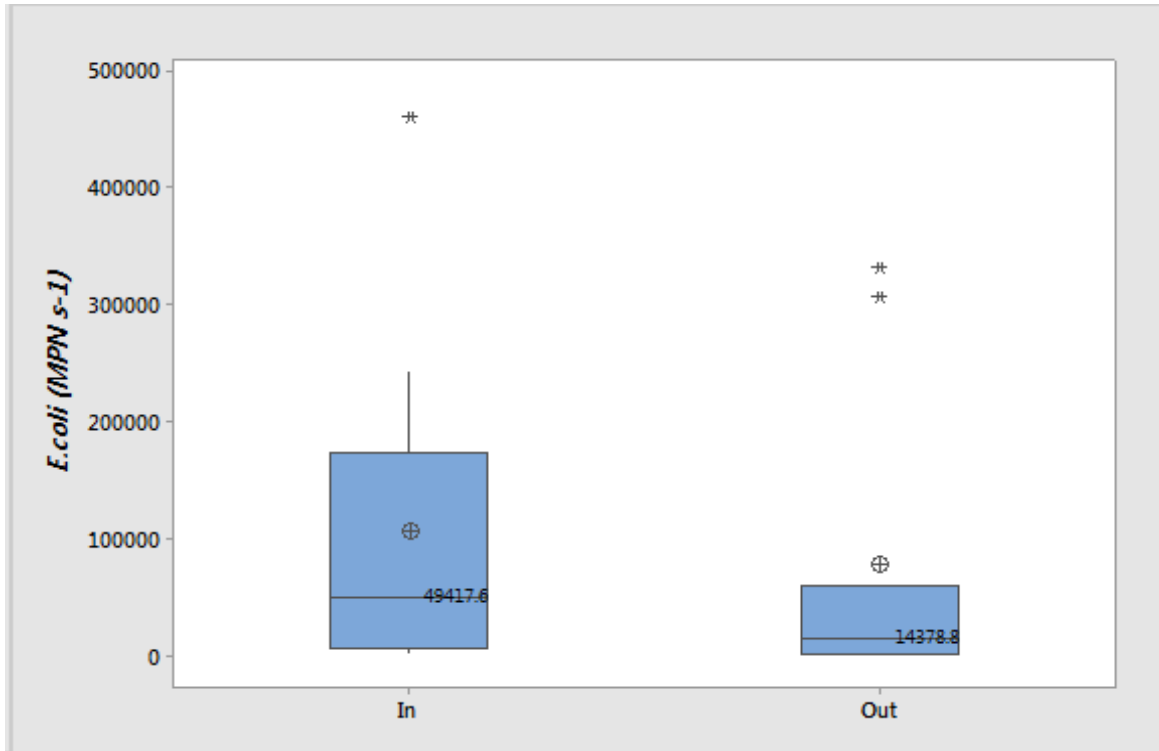


Figure 22. *E. coli* loading by inflow (In) and outflow (Out). Flat lines indicate medians, crosshairs indicate means, and stars represent outlier data. Median values are labeled.

4.3.2 Groundwater Quality

4.3.2.1 Concentration

The median TDN concentration of groundwater sampled from the deep piezometer (0.20 mg L^{-1}) was 0.44 mg L^{-1} less than the median TDN concentration groundwater samples collected from the shallow piezometer (0.64 mg L^{-1}) ($p = 0.03$). Concentrations of TDN in the shallow piezometer ($n = 9$) exceeded the deep ($n = 8$) 88% of times sampled (Figure 23A). The median $\text{NO}_3\text{-N}$ concentration of groundwater sampled from the deep piezometer (0.03 mg L^{-1}) was 0.05 mg L^{-1} less than the median $\text{NO}_3\text{-N}$ concentration in groundwater samples from the shallow piezometer (0.08 mg L^{-1}) ($p = 0.37$). Concentrations of $\text{NO}_3\text{-N}$ in the shallow piezometer exceeded the deep ($n = 4$) 44% of times sampled. The median $\text{NH}_4\text{-N}$ concentration of groundwater sampled from the deep piezometer (0.19 mg L^{-1}) was 0.11 mg L^{-1} less than the

median $\text{NH}_4\text{-N}$ concentration in groundwater from the shallow piezometer (0.30 mg L^{-1}) ($p = 0.17$) (Figure 23A). The shallow piezometer had a higher $\text{NH}_4\text{-N}$ concentration than the deep 67% of the time (Appendix J).

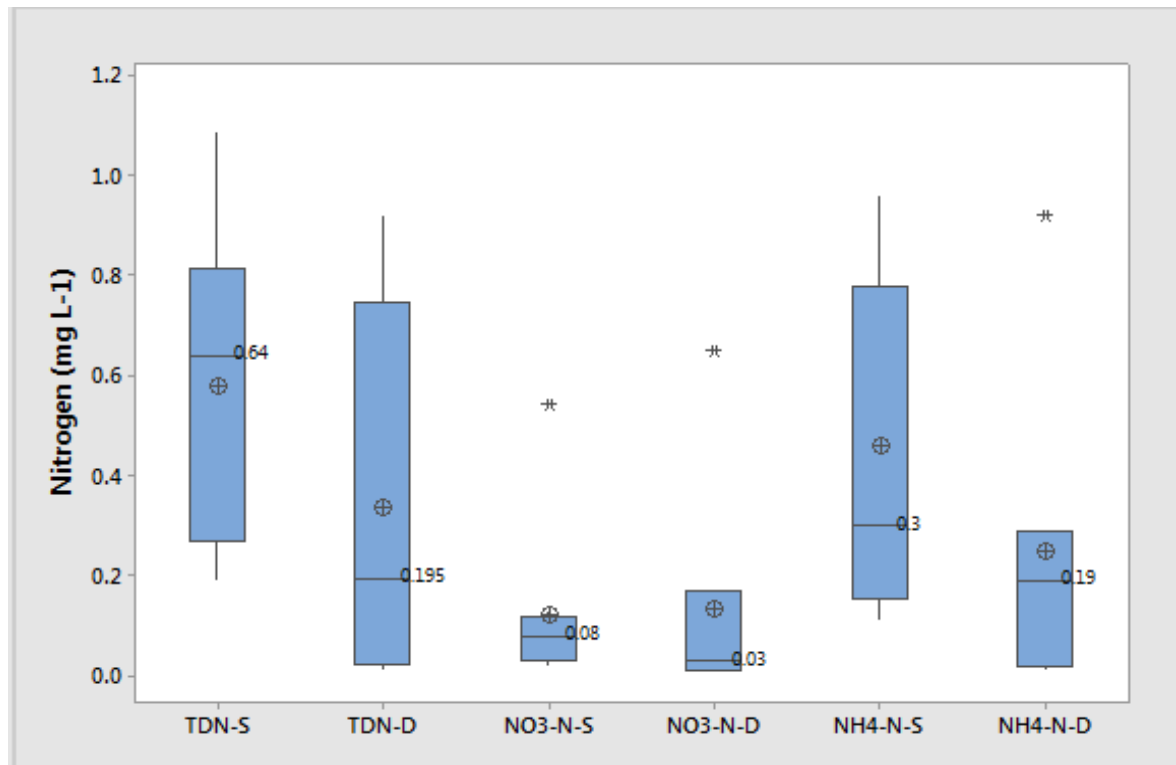


Figure 23A. Concentrations of nitrogen species (mg L^{-1}) by shallow (S) and deep (D) piezometer. Flat lines indicate medians, crosshairs indicate means, and stars represent outlier data. Median values are labeled. TDN= total dissolved nitrogen; $\text{NO}_3\text{-N}$ = nitrate-nitrogen; $\text{NH}_4\text{-N}$ = ammonium-nitrogen.

The median $\text{PO}_4\text{-P}$ concentration of groundwater sampled from the deep piezometer (0.001 mg L^{-1}) was 0.049 mg L^{-1} less than groundwater collected from the shallow piezometer (0.05 mg L^{-1}) ($p = 0.001$) (Figure 23B). The shallow piezometer had a higher $\text{PO}_4\text{-P}$ concentration than the deep 67% of the time (Appendix J).

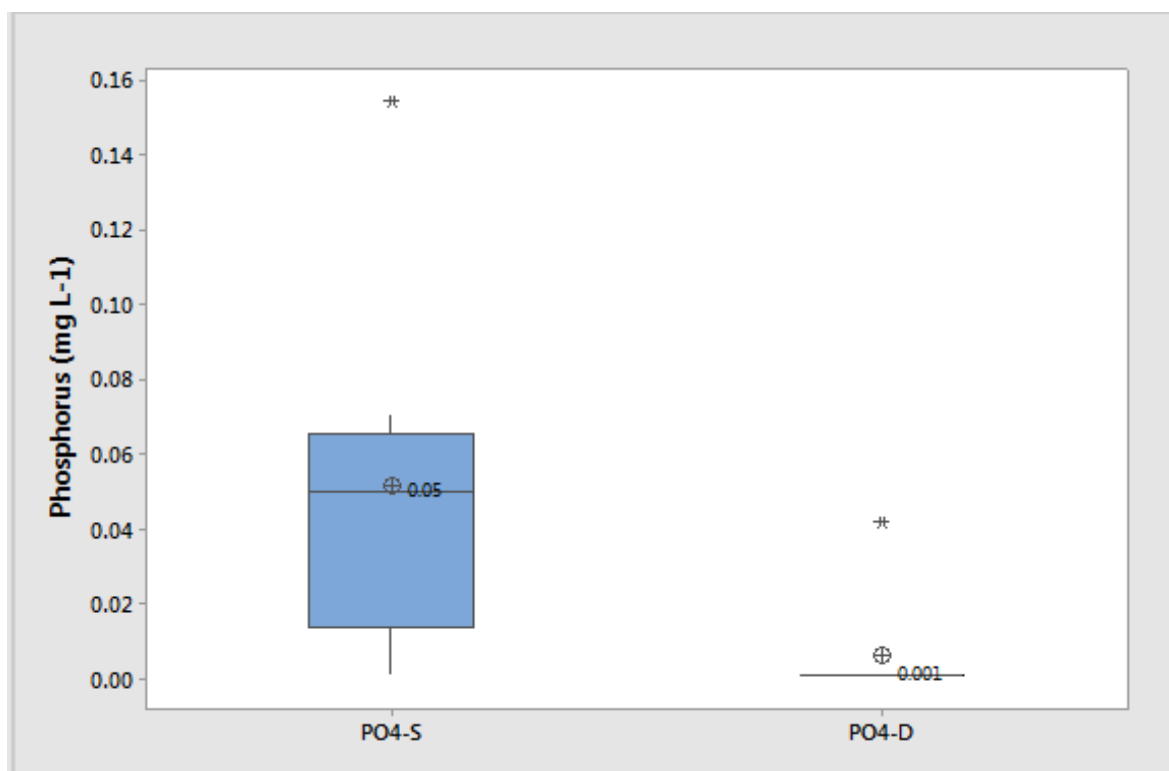


Figure 23B. PO₄-P (phosphate-phosphorus) concentration in the piezometers. PP (particulate phosphate) and TDP (total dissolved phosphate) not detected in piezometers during this study. Flat lines indicate medians, crosshairs indicate means, and stars represent outlier data. Median values are labeled.

The median *E. coli* concentration of groundwater sampled from the deep piezometer (1 MPN 100 mL⁻¹) was 815 MPN 100 mL⁻¹ less than the median *E. coli* concentration in groundwater samples collected from the shallow piezometer (816 MPN 100 mL⁻¹) ($p = 0.01$) (Figure 23C). Groundwater sampled from the shallow piezometer had a higher *E. coli* concentration than the deep 67% of the times sampled (Appendix J).

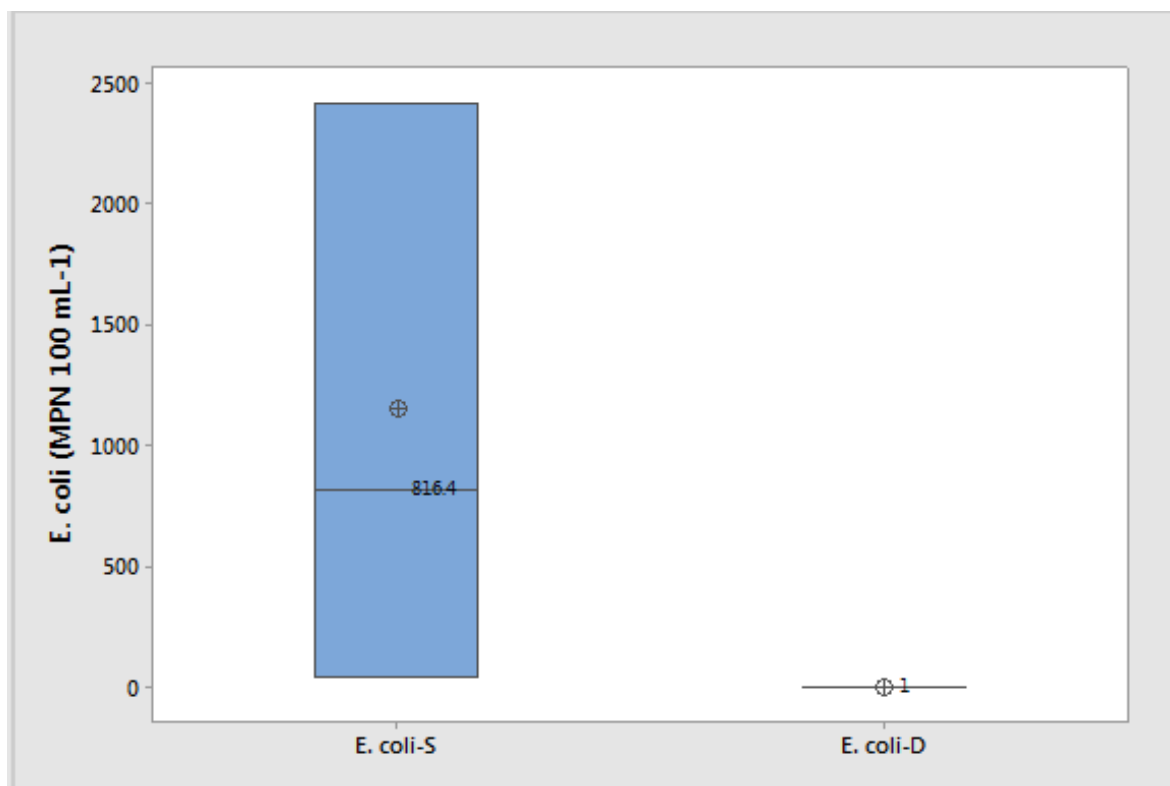


Figure 23C. *E. coli* concentration in the shallow (S) and deep (D) piezometers. The levels of bacteria detected in deep piezometer were almost zero. Flat lines indicate medians and crosshairs indicate means. Median values labeled.

CHAPTER 5: DISCUSSION

The goal of the study was to determine if the natural wetland was providing ecosystem services by reducing the transport of nutrients, bacteria, and sediment.

5.1 Physicochemical Characterization

The range of manual monthly stage readings for I-1 represented about 60.3% of observed logger stage readings and about 43.6% of stage readings for I-2; about 50% of the logger readings exceeded manual state recordings. This shows there was not only variation in flow rates outside of what sampling events captured, but also storm events tend to provide increased flow, such as February 2019 (Appendix A, C). This high flow event resulted in less removal of DIN mass and was one of the events that showed increase in PN and TSS mass in the exports (Appendix M). This may imply treatment efficiencies could be impacted at greater levels than what sampling data suggests due to factors such as decreased residence time during heavier stormflow. Erosion at the outlets seen through loss of sand, deep valleys, and uprooted trees further supports this notion of heavier flow causing decrease treatment efficiency. Heavier storm flow tends to carry increased sediment which may help explain months of higher exports of particulates and TSS than those at inlets.

Groundwater level data from the deep and shallow piezometers suggests the wetland was a recharge zone (Table 3) (Appendix I). Therefore, some streamflow from I-1 and I-2 was infiltrating the soil within the wetland. Groundwater data indicate that the subsurface environment within the wetland was conducive to denitrification, which is a mechanism for removing NO_3 . Denitrification may occur when ORP levels drop below 250 mV (Mitsch and Gosselink, 2000). The median ORP of groundwater collected from the shallow (57 mV) and deep (-81 mV) piezometers also indicated a reducing environment where denitrification may

occur. Other conditions, such as availability of energy sources such as carbon from decaying vegetation matter, and suitable temperature (5-30°C) and pH (7.0-8.5) (Appendix G), are necessary for denitrification to occur (The Water Planet Company, 2020; Trygar, 2009). Median surface water ORP data also suggests this wetland became a more reducing environment toward the outlets, which may influence nutrient transformations and retention (Table 1, 3; Appendix G). Groundwater from the shallow piezometer had a higher median ORP (57 mV) in comparison to the deep (-91mV); the deeper groundwater had a higher SC (550 $\mu\text{S cm}^{-1}$ versus 233 $\mu\text{S cm}^{-1}$), similar DO (3.59 mg L^{-1} versus 3.55 mg L^{-1}), pH (6.56 versus 6.5), and temperature (15.1°C versus 14.9°C) (Appendix J). Physicochemical properties of groundwater were all within the range reported for the high-frequency surface water loggers deployed at the inlets (Table 3).

In surface water, median physicochemical parameter readings at the wetland outlets showed a lower ORP, lower SC, and higher turbidity than wetland inflow, although statistically significant differences were only observed for ORP (Table 1). The median ORP of surface water was 66 mV at the wetland inlet and 8mV at the wetland outlet, both under the 250-mV threshold for denitrification to occur per Mitsch & Gosselink (2000). The wetland was exhibiting obvious signs of erosion, which aligns with the increased turbidity.

Outflow from the wetland was a combination of surface water overflow and groundwater discharge. Median outflow from the wetland was 33% lower relative to the inflow, although there was much variability over the duration of the study. The reduction in outflow could be related to evapotranspiration associated within the wetland (Brooks et al., 2003; Wulschleger et al., 1998). For example, Wulschleger and others (1998) in a review paper, reported that a 5-m tall *Pinus taeda* tree could transpire 40 kg day^{-1} of water. This species of tree was observed in the wetland. The variation in flow observed during the study was likely influenced by rainfall

received a few days prior to a sampling event and evapotranspiration rates (Appendix C). For example, during the June 2018 sampling event, no flow was observed. However, a few months later in September 2018 there was a large increase in outflow relative to inflow, due to sampling being shortly after a major storm event related to the aftermath of a hurricane. The highest inflow and outflow measurements were observed in February of 2019, when temperatures are relatively cold, many plants are dormant and thus evapotranspiration is low.

5.2 Nutrient, Bacteria and Sediment Trends

5.2.1 Nutrients

Median TN concentration in wetland outflow was 35% lower relative to wetland inflow. Most of the reduction occurred via differences in NH_4 (39%) and NO_3 (47%) at the inlet and outlet. In comparison to previous studies, this wetland performed similarly- Rozema et al. (2016) showed reductions in NH_4 (76%), NO_3 (42%); Almukhtar et al. (2018) showed reductions in TN (74%), NH_4 (65%), NO_3 (91%); Jordan et al. (2003) showed a 38% reduction in TN (Table 1). When looking at previous studies, it should be noted that some (Rozema et al, 2016) also had large reductions in TSS (83%) which was not the case in this study and could impact reduction of other pollutants such as particulates. Another potential mechanism for N removal within the wetland was adsorption of NH_4 by the wetland soil, and nitrification. Most of the soil in the study area was Chewacla and Wehadkee and these soils typically have cation-exchange-capacity (CEC) between 4 and 35 meq 100 g⁻¹ (Table 4) (Natural Resources Conservation Service, 2020).

Table 4. Soil types by percent of area of Lick Creek Wetland studied, plus cation exchange capacity (CEC) (USDA, 2020).

Summary by Map Unit - Durham County, North Carolina (NC063)				
Map Unit Symbol	Map Unit Name	CEC (milliequivalents per 100 grams)	Acres in AOI	Percent of AOI
Ch	Chewacla and Wehadkee soils, 0-2% slopes, frequently flooded	9.6	5.9	67.40%
CrC	Creedmoor sandy loam, 6-10% slopes	15.2	0.9	10.60%
MfB	Mayodan sandy loam, 2-6% slopes	13.8	0.7	7.70%
MfC	Mayodan sandy loam, 6-10% slopes	13.8	0.2	2.50%
PfC	Pinkston fine sandy loam, 2-10% slopes		0.6	7.40%
PfE	Pinkston fine sandy loam, 10-25% slopes		0.4	4.40%
Totals for Area of Interest	8.7	100%		

Most of the TDN in this wetland's inflow was comprised of the plant-available species of N including NO_3 and NH_4 . Prior research has shown that riparian plants can immobilize between 15-90 $\text{kg ha}^{-1} \text{ yr}^{-1}$ of N and thus these processes may have also contributed to N removal (Osmond et al., 2002). TDN in this study was 85% of TN inflow and 90% of outflow, whereas DIN was 85% of TDN at inflow and 97% of TDN at outflow. Though there was an overall decrease in median PN mass leaving the wetland in comparison to entering, PN was often exported by the wetland. Visible erosion at the wetland outlets was a likely contributor of PN to Lick Creek. PN could be associated with sediment or plant material. Sediment and plant material may act as a sink for nutrients until transport to surface waters via erosion or decomposition, mineralization overland flow (Grantz et al., 2003). Changes in seasons in the North Carolina region bring changes to temperatures and precipitation level, which impacts the life cycle of plants, in turn impacting plant uptake of pollutants, whereas leaf decomposition and release is also impacted by these same factors (Magee, 1993). Erosion, particularly in heavier stormflow, likely transports sediment and leaf matter from within the wetland to Lick Creek. The N concentration reduction (Table 5) and N mass reduction (Table 5) observed during this study

may change if the study were conducted over a longer period, however the discrepancy in concentration and mass removal can in part be attributed to flow rate. In the case of concentration reductions, treatment efficiency of N by the unmaintained wetland is likely changing over time due to decreased residence time of stormflow in the wetland due to erosion. Perhaps if this natural wetland was stabilized or maintained to control flow and tame erosion, the residence time of stormwater would not decrease as much over time.

Table 5. N concentration (left) and loading (right) treatment efficiencies.

Nutrient	Reduction	Nutrient	Reduction
TN	35%	TN	58%
NO ₃ -N	46%	NO ₃ -N	48%
NH ₄ -N	37%	NH ₄ -N	54%

Natural wetlands may have lower removal efficiencies due to lower concentrations of nutrients and flow entering the wetland. A study of a natural wetland receiving agricultural flow in Ohio showed NO₃-N was reduced 35% the first year and 49% the second year, while TP was reduced 74% the first year and 40% the second (Khanijo, 2020). Average NO₃-N loading did not change much during the study in most storms, though average TP loading did significantly increase. Nichols (1983) stated that increased wastewater nutrient loading over time diminishes wetland treatment efficiency, which is the primary source of nutrient and bacteria pollution in this study- suggesting over time the efficiency of the natural wetland will decrease. A cropland wetland in Beaufort, NC, showed reductions in TN (1.5-4.5 mg/L in the first 6 months, to 1-3 mg/L 24-36 months after) following restoration projects targeting outflow, with methods such as installation of a V-notch weir at outflow locations (Burchell et al. 2011). TP took longer to

decrease in the study, as could be expected because of soil disturbance and erosion during restoration efforts that contribute to soil-bound P loss. The study suggested post-restoration levels of TN would be 1-1.5 mg/L moving forward, compared to 4.5 mg/L in agricultural drainage ditches in eastern North Carolina; median export concentration in this study was 1.87 mg/L for the natural wetland (Appendix B, F). Similar restoration efforts in the natural wetland outlets could decrease flow and thereby mass of exports into Lick Creek.

Decreases in P concentrations were seen in outflow respective to inflow for both PP (60% of the time) and PO₄-P (80% of the time). Concentrations of PP and PO₄-P were reduced by about 8% and 24%, respectively (Table 6). Mass reductions of PP and PO₄-P were nearly 34%, and 63%, respectively (Table 6). Like N, decreases in P are likely related to flow reductions, adsorption of PO₄ onto soil particles, precipitation of P-based minerals, and/or plant uptake. For example, riparian vegetation has been shown to remove between 2-6 kg ha⁻¹ yr⁻¹ for P (Osmond et al., 2002). Similarly, in prior studies P mass removal was between 32% and 71% in a constructed wastewater treatment wetland (Lin et al., 2002); 43% of P mass removal was observed in drainage water in an agricultural wetland (Woltemade, 2000). Clay soils in the wetland, such as the Chewacla and Wehadkee, tend to have aluminum and iron-oxide coatings which provide high surface area, giving great adsorption capacities for various pollutants including heavy metals and nutrients (Tassanapayak et al., 2008; Xie et al., 2018).

Processes in soil and shallow groundwater including adsorption of PO₄ and NH₄, microbial transformations of N species, plant and microbial uptake of N and P, and dilution may have been active during the study and helped to account for nutrient reductions within the wetland. Sampling events of groundwater further supported the treatment occurring as seen by

decreases in TDN (69.5%), NO₃-N (62.5%), NH₄-N (36.7%), PO₄-P (98%), and, *E. coli* (99.9%) from the shallow to the deep piezometer.

Table 6. P concentration (left) and mass (right) treatment efficiencies of the wetland.

Nutrient	Reduction	Nutrient	Reduction
PP	8%	PP	34%
PO ₄ -P	24%	PO ₄ -P	63%

5.2.2 Bacteria

Median *E. coli* concentrations decreased almost 50% in outflow as compared to inflow, but the differences were not statistically significant. *E. coli* in wetland inflow may attach to soils and plants within the wetland during low flow or be transported through the wetland to Lick Creek during higher flows associated with storms and erosion events. Mechanisms for *E. coli* treatment in the wetland are like those for P, such as attaching to soil and roots, but also inactivation by sunlight (González et al., 2019). Therefore, when erosion occurs, transport of *E. coli* may be enhanced. The EPA recommends that the geometric mean of *E. coli* concentrations should not exceed 126 CFU/100 mL and no more than 10% of samples should exceed the statistical threshold value of 410 CFU/100 mL (EPA, 2015). Inflow concentrations of *E. coli* exceeded the statistical threshold value 100% of the time when flow was measurable while outflow concentrations exceeded the threshold 80% of the time (Appendix N). Outflow from the wetland was more turbid than inflow, likely because of erosion of the wetland outlets. The erosion also contributed to higher concentrations of TSS in outflow relative to inflow. The increase in TSS and visible erosion help explain the increases in *E. coli* at months where outflow concentrations

exceeded inflow, likely accelerated by the increased stormflow events those months. The wetland did a good job overall of reducing *E. coli* loading during low flow periods (Table 7).

Table 7. *E. coli* concentration (left) and loading (right) treatment efficiencies.

<i>E. coli</i>	Reduction
Concentration	50%
Loading	71%

5.2.3 Sediment

Though not statistically significant, TSS concentrations and flow-weighted turbidity increased at the outflow respective to inflow in this study. This aligns with the erosion occurring at the outlets into Lick Creek, as they recede further back into the wetland (Figure 4B). Heavy rainfall in unmaintained wetlands can lead to extensive erosion, which results in increased sediment load in receiving waters (Mereta et al., 2020). Managing stormwater flow is important in reducing TSS (Minnesota Stormwater Manual, 2020). Median TSS concentrations and mass loadings both increased throughout the study (Table 8).

Table 8. TSS concentration and loading treatment efficiencies.

TSS	Increase
Concentration	48%
Loading	15%

5.3 The Natural Wetland: Ecosystem Service or Disservice?

The wetland was a recharge zone on most occasions during the study and N species and PO₄-P levels were consistently higher in shallow groundwater relative to deep groundwater. This suggests pollutant removal mechanisms were active in the soil and groundwater system. The deeper well had very little *E. coli* in comparison to the shallow well.

Based on pollutant loading rates at the inlets and outlets, the wetland was a sink for most of the pollutants studied (Table 9), but less effective for PN and TSS due to erosion. While erosion can have some impacts on *E. coli*, we cannot rule out natural sources of bacteria such as wildlife excrement. Microbial source tracking would be necessary to confirm where bacteria originated; one study found that almost 50% of *E. coli* pollution in the Arroyo Colorado watershed was sourced to non-avian wildlife (Jeong et al., 2019). Other studies have shown wildlife to be major contributors to elevated *E. coli* levels, which may contribute to inconsistent treatment results for the bacteria (Meays, 2006; Somarelli et al., 2007).

Table 9. Frequency expressed as a percent (n=10) that wetland sequestered or transported of pollutant. Dry months excluded from data.

Pollutant	TN	TDN	PN	PO ₄ -P	PP	TSS	<i>E. coli</i>
Sink	90%	90%	50%	80%	80%	60%	60%
Source	10%	10%	50%	20%	20%	40%	40%

CHAPTER 6: CONCLUSION

Wetlands can function as areas for recreational purposes such as fishing, boating, and wildlife observation, and they provide habitats for many rare plants and animals. The services provided by some wetlands, such as this natural wetland studied, were not always recognized, or valued. Loss of wetlands translates into loss of ecosystem services, some of which are later regained through costly engineering practices (e.g., flood-control levees, soil renourishment projects, stream restoration and drainage systems, habitat creation, etc.). The Federal government was responsible for such acts as the Watershed Protection and Flood Prevention Act of 1954, which led to intentional draining and destruction of wetlands at an estimated 550,000 acres per year from the 1950s through the 1970s (USGS, 2019). Modern-day loss typically stems from both natural and anthropogenic efforts, such as deforestation, drainage, saltwater intrusion, conversion of wetlands to agricultural uses or residential/commercial development loss, and subsurface changes caused by the oil and gas industries through fracking processes (Morton et al., 2003; Liu et al., 2004; Coleman et al., 2008). Discrepancies over enforcement policies of wetland conservation and misunderstandings of wetland benefits still lead to further destruction, even at the federal level, including conflicting interests and wetland conversion agreements (Votteler & Muir, 2002). Federal wetland protections are provided by section 404 of the Clean Water Act by regulation of dredging and filling functions. State regulatory agencies are critical because federal loopholes allow for exemptions to 404 permit requirements as well as the regulation sometimes failing to protect “navigable waters” (EPA, 2018). The Farm Bill of 1985 directed the USDA to work with conservation districts to decide best practices for wetland conservation for mitigation. The idea behind the Farm Bill was to remove incentives of those converting wetlands, which is also known as “Swampbuster” from the Food Security Act of 1985 (NRCS, 2019). Those

converting to agriculture on previous wetlands between 1985-1990 became ineligible for program benefits. The USDA states the Farm Bill was necessary because 53% of wetlands were lost in the lower 48 states over a 200-year period, 87% directly linked to conversion to agricultural purposes from 1950-1970's (NRCS, 2019). Perhaps seeing the positive impacts wetlands can provide to ecosystems and public health will lead to further conservation of wetlands instead of seeing them as wastelands. The 5 Governmental agencies involved with wetland regulation can each work together to be appeased by the results of well-functioning wetland areas as a result of these efforts for the ecosystem benefits and protection of species inhabiting them, plus protections of public health. As an example of how agencies could work together, the USDA could determine appropriate wastewater runoff- including fertilizer components from farmlands, to working with the Army Corps of Engineers to control storm flow to benefit both humans and wildlife, by conserving natural resources.

The natural wetland studied in this project provides a good example of the ecosystem services provided by wetlands. The natural wetland was acting as a sink for nutrients and bacteria and reduced pollutant transport to Lick Creek. The reduction of pollutant loading can improve water quality and lessen negative impacts from HABs. This is important because Lick Creek is a tributary to Falls Lake, a major water supply and recreation source for the Raleigh, NC area. Higher stormflow reduces the hydraulic residence time within the wetland, bypassing potential immobilization via wetland plants, and reducing opportunities for infiltration of pollutant-rich stormwater. High-frequency loggers showed us that monthly sampling only captures a portion of the picture, implying flow has much more variation than what is seen only on sampling days, as median stage readings varied quite a bit from manual readings on those dates (Appendix O). Reductions in the particulate forms of N and P were less relative to

reductions in dissolved N and P, possible because of erosion and leaf matter transport of particulates during storms. Overall, the wetland was providing ecosystem services for N, P, and *E. coli*, but the services could be lessened or lost due to erosion. Stormwater control measures up-stream from the wetland could perhaps make the biggest differences for reducing nutrient and bacteria inputs to the wetland from upgradient communities. Control measures may also reduce pollutant inputs to Lick Creek, and effectively into Falls Lake, since they are all connected. This information is also important for nutrient management practices in areas downstream of septic systems, point source and non-point source pollution, and for optimizing environmental and human health downstream.

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APPENDIX A: LICK CREEK PRECIPITATION

Table A1. Average monthly precipitation, Lick Creek Watershed 2008-2014 (City of Durham, 2019)

Month	Econet Durham Rain Gauge (Inches)	USGS Rain Gauge Measurement (Inches)
January	2.2	2.1
February	2.6	2.4
March	4.1	4.4
April	2.5	2.7
May	5.5	4.8
June	4.3	4.1
July	3.7	4.3
August	3.7	4.5
September	4.9	5.6
October	2.4	2.2
November	3.4	3.2
December	4.1	3.4
Annual	43.4	43.7

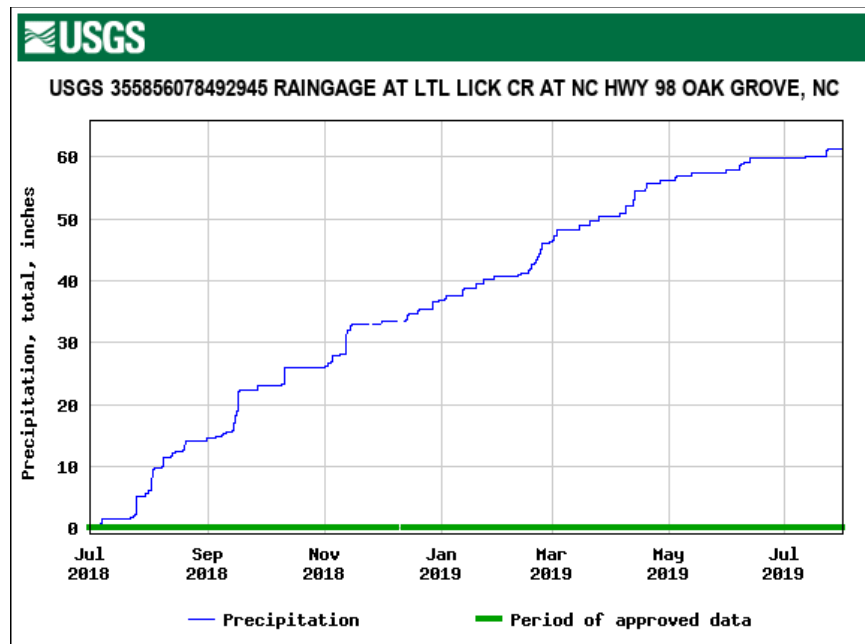
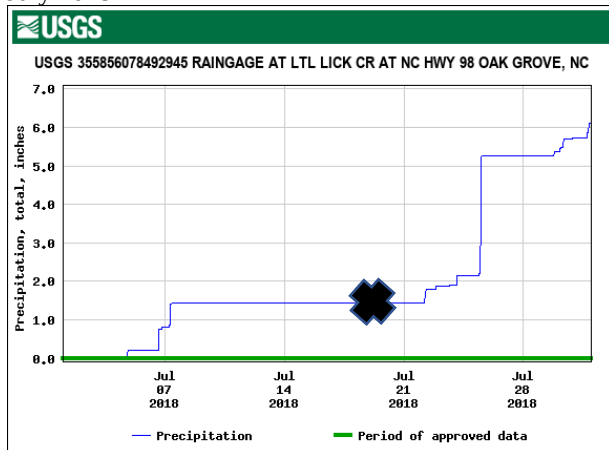
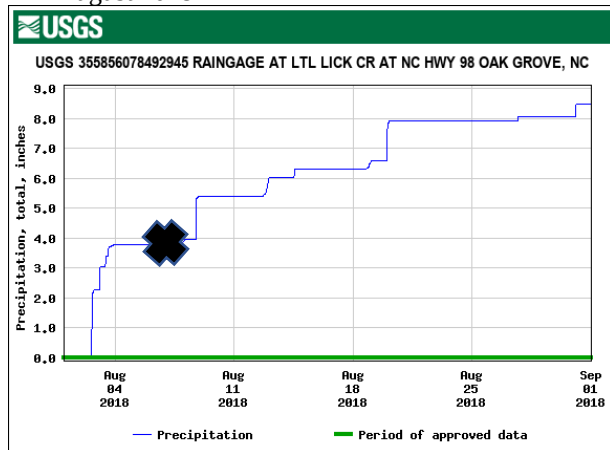


Figure A1. Total precipitation at Lick Creek by month over the sampling period (USGS, 2020)

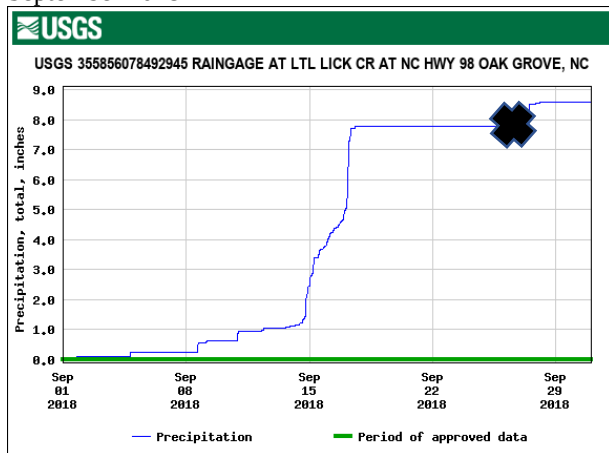
July 2018



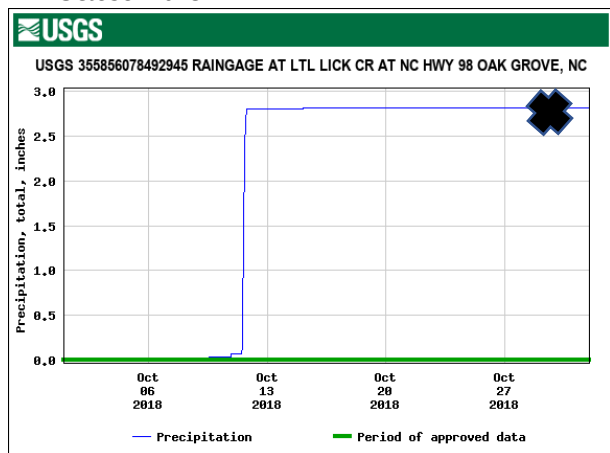
August 2018



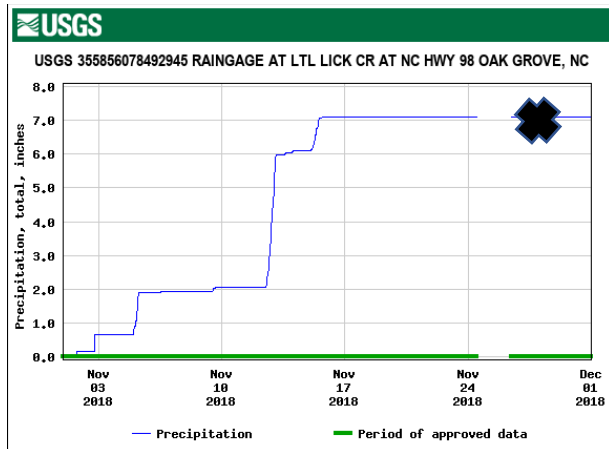
September 2018



October 2018



November 2018



December 2018

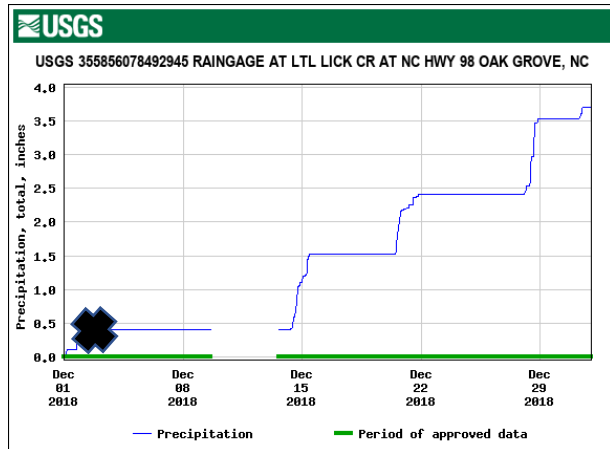
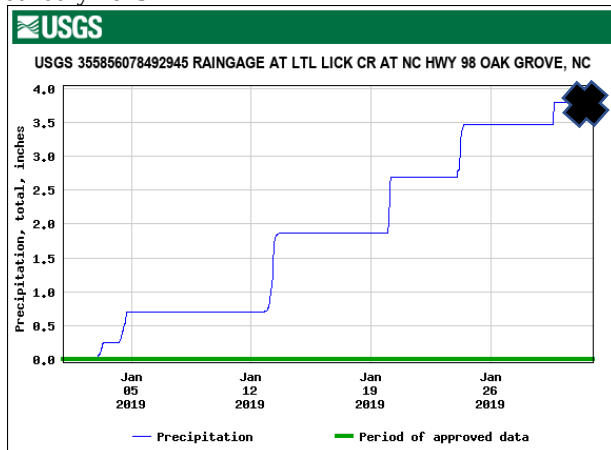
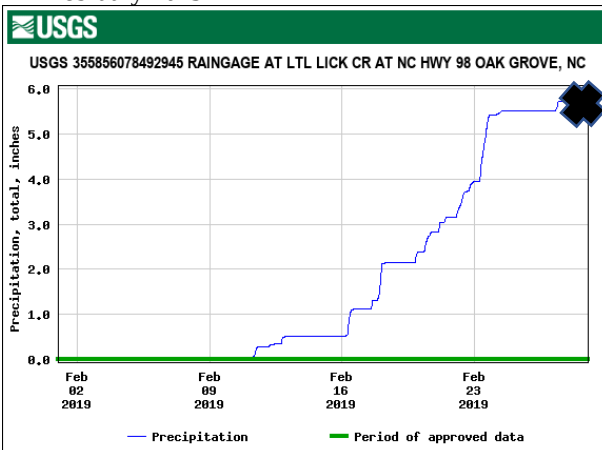


Figure A2. Lick Creek watershed precipitation July 2018 through December 2018.

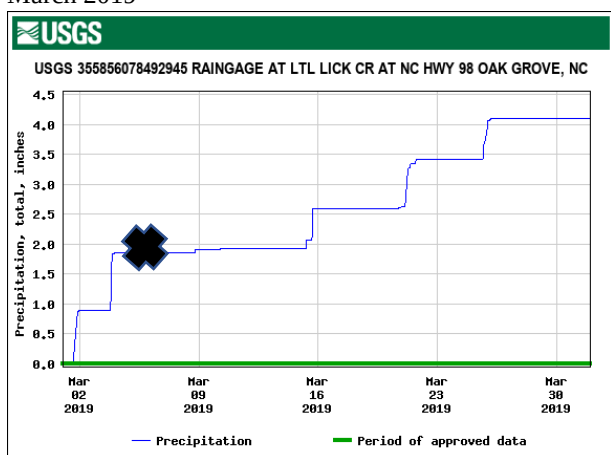
January 2019



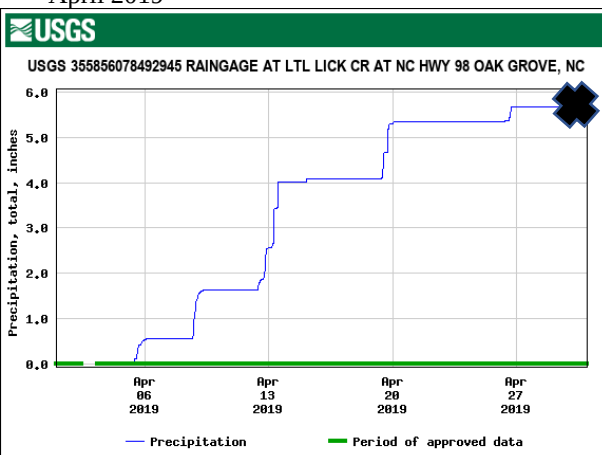
February 2019



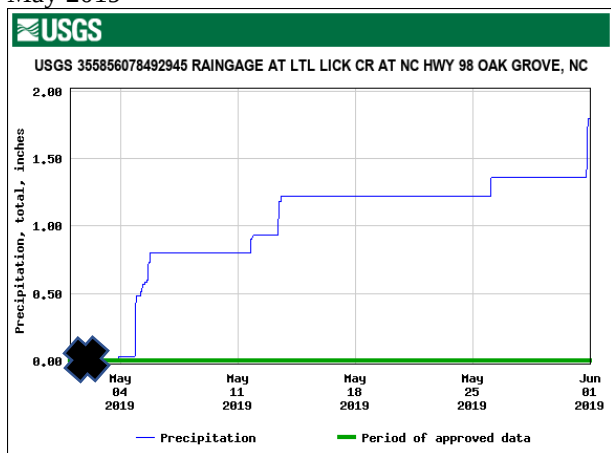
March 2019



April 2019



May 2019



June 2019

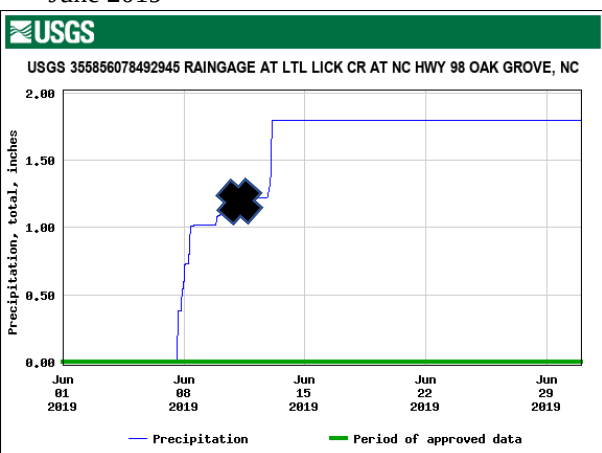


Figure A3. Lick Creek watershed precipitation January 2019 through June 2019.

APPENDIX B: LICK CREEK WATERSHED LAND USE

Table B1. Lick Creek I-1 Watershed Land Use as of 2011 (USGS, 2020).

Land Use, NLCD 2011	Percentage
Forest	55.22%
Developed (urban) land	43.30%
Wetlands	1.46%
Grassland/Herbaceous	0%
Shrubland	0%
Open water	0%
Barren land	0%
Cultivated crops and hay	0%
Average percentage of impervious area	7%

Table B2. Lick Creek I-2 Watershed Land Use as of 2011 (USGS, 2020).

Land Use, NLCD 2011	Percentage
Developed (urban) land	60.60%
Forest	27.30%
Cultivated crops and hay	4.26%
Grassland/Herbaceous	4.06%
Shrubland	2.22%
Wetlands	1.59%
Barren	0%
Open water	0%
Average percent impervious surface	12%

APPENDIX C: WETLAND FLOW

TABLE C1. Wetland flow in liters per second (L/s).

Date	I-1	I-2	O-1	O-2	O-3	Inlet Total	Outlet Total
19-Jul-18							
8-Aug-18	0.85	0.00	0.00	0.00	0.28	0.85	0.28
27-Sep-18	1.70	5.49	0.14	0.34	12.66	7.19	13.14
29-Oct-18	1.70	1.67	0.00	0.00	2.49	3.37	2.49
30-Nov-18	1.53	2.86	0.03	0.03	0.01	4.39	0.07
3-Dec-18	4.30	2.78	0.03	0.20	3.88	7.08	4.11
31-Jan-19	2.58	2.92	1.05	0.00	4.90	5.49	5.95
27-Feb-19	7.22	15.35	1.93	0.14	19.34	22.57	21.41
5-Mar-19	3.06	2.86	0.06	0.01	7.50	5.92	7.57
29-Apr-19	1.05	0.68	0.01	0.00	0.93	1.73	0.94
1-May-19	0.31	1.10	0.00	0.00	0.54	1.42	0.54
11-Jun-19							
Median						4.94	3.30

TABLE C2. Wetland flow fractions by inlets and outlets.

	Weighted Fractions				
	Inlets		Outlets		
Date	I-1	I-2	O-1	O-2	O-3
19-Jul-18					
8-Aug-18	100%	0%	0%	0%	100%
27-Sep-18	24%	76%	1%	3%	96%
29-Oct-18	50%	50%	0%	0%	100%
30-Nov-18	35%	65%	42%	42%	17%
3-Dec-18	61%	39%	1%	5%	94%
31-Jan-19	47%	53%	18%	0%	82%
27-Feb-19	32%	68%	9%	1%	90%
5-Mar-19	52%	48%	1%	0%	99%
29-Apr-19	61%	39%	1%	0%	99%
1-May-19	22%	78%	1%	0%	99%
11-Jun-19					

APPENDIX D: RAW POLLUTANT CONCENTRATIONS

TABLE D1. TN concentrations as mg/L. Inflow is “I”, outflow is “O”.

TN	I-1	I-2	I Total	O-1	O-2	O-3	O Total
19-Jul-18							
8-Aug-18	0.18	0.16	0.34	0.22		0.08	0.30
27-Sep-18	0.53	0.50	1.03	0.28	0.47	0.47	1.23
29-Oct-18	0.09	0.10	0.19			0.09	0.09
30-Nov-18	0.09	0.06	0.15	0.07	0.05	0.04	0.15
3-Dec-18	0.07	0.10	0.17	0.09	0.04	0.08	0.21
31-Jan-19	0.11	0.08	0.19	0.15	0.24	0.09	0.47
27-Feb-19	0.27	0.17	0.44	0.39	0.14	0.21	0.74
5-Mar-19	0.07	0.13	0.20	0.62	0.31	0.12	1.06
29-Apr-19	0.10	0.25	0.34	0.95		0.48	1.42
1-May-19	0.14	0.09	0.23	1.62		0.10	1.72
11-Jun-19							

TABLE D2. PN concentrations as mg/L. Inflow is “I”, outflow is “O”.

PN	I-1	I-2	I Total	O-1	O-2	O-3	O Total
19-Jul-18							
8-Aug-18	0.18	0.16	0.34	0.22		0.08	0.30
27-Sep-18	0.53	0.50	1.03	0.28	0.47	0.47	1.23
29-Oct-18	0.09	0.10	0.19			0.09	0.09
30-Nov-18	0.09	0.06	0.15	0.07	0.05	0.04	0.15
3-Dec-18	0.07	0.10	0.17	0.09	0.04	0.08	0.21
31-Jan-19	0.11	0.08	0.19	0.15	0.24	0.09	0.47
27-Feb-19	0.27	0.17	0.44	0.39	0.14	0.21	0.74
5-Mar-19	0.07	0.13	0.20	0.62	0.31	0.12	1.06
29-Apr-19	0.10	0.25	0.34	0.95		0.48	1.42
1-May-19	0.14	0.09	0.23	1.62		0.10	1.72
11-Jun-19							

TABLE D3. DIN concentrations as mg/L. Inflow is “I”, outflow is “O”.

DIN	I-1	I-2	I Total	O-1	O-2	O-3	O Total
19-Jul-18							
8-Aug-18	1.41	0.12	1.53	0.06	0.00	0.14	0.20
27-Sep-18	1.93	0.90	2.83	0.66	1.44	0.88	2.99
29-Oct-18	2.76	0.42	3.18	0.00	0.00	1.37	1.37
30-Nov-18	1.62	0.57	2.19	0.21	1.01	0.60	1.82
3-Dec-18	1.18	0.42	1.60	0.17	1.08	0.38	1.63
31-Jan-19	1.76	0.59	2.35	0.56	0.25	0.81	1.62
27-Feb-19	1.17	0.50	1.67	0.26	0.58	0.59	1.43
5-Mar-19	1.80	0.54	2.34	0.04	0.00	0.68	0.72
29-Apr-19	0.89	0.39	1.28	0.07	0.00	0.12	0.19
1-May-19	0.87	0.15	1.01	0.02	0.00	0.16	0.17
11-Jun-19							

TABLE D4. NO₃-N concentrations as mg/L. Inflow is “I”, outflow is “O”.

NO ₃ -N	I-1	I-2	I Total	O-1	O-2	O-3	O Total
19-Jul-18							
8-Aug-18	1.34	0.03	1.37			0.06	0.06
27-Sep-18	1.39	0.7	2.09	0.2	1.14	0.68	2.02
29-Oct-18	2.47	0.28	2.75			1.19	1.19
30-Nov-18	1.47	0.42	1.89	0.03	0.83	0.4	1.26
3-Dec-18	0.3	0.21	0.51		0.94	0.26	1.20
31-Jan-19	1.63	0.36	1.99	0.09	0.02	0.67	0.78
27-Feb-19	0.92	0.32	1.24	0.05	0.44	0.49	0.98
5-Mar-19	1.46	0.32	1.78	0.04		0.64	0.68
29-Apr-19	0.84	0.34	1.18			0.08	0.08
1-May-19	0.81	0.12	0.93	0.01		0.11	0.12
11-Jun-19	2.21	0.07				0.02	

TABLE D5. NH₄-N concentrations as mg/L. Inflow is “I”, outflow is “O”.

NH₄-N	I-1	I-2	I Total	O-1	O-2	O-3	O Total
19-Jul-18							
8-Aug-18	0.07	0.09	0.16	0.06		0.08	0.14
27-Sep-18	0.35	0.2	0.55	0.46	0.3	0.2	0.96
29-Oct-18	0.29	0.14	0.43			0.18	0.18
30-Nov-18	0.15	0.15	0.30	0.18	0.18	0.2	0.56
3-Dec-18	0.25	0.21	0.46	0.17	0.14	0.12	0.43
31-Jan-19	0.13	0.23	0.36	0.46	0.23	0.14	0.83
27-Feb-19	0.25	0.18	0.43	0.21	0.14	0.1	0.45
5-Mar-19	0.34	0.22	0.56			0.05	0.05
29-Apr-19	0.04	0.05	0.09	0.07		0.04	0.11
1-May-19	0.06	0.03	0.09	0.01		0.05	0.06
11-Jun-19	0.28	0.26				0.53	

TABLE D6. PO₄-P concentrations as mg/L. Inflow is “I”, outflow is “O”.

PO₄-P	I-1	I-2	I Total	O-1	O-2	O-3	O Total
19-Jul-18							
8-Aug-18	21.5	0.0	21.5	0.0	0.0	0.8	0.8
27-Sep-18	41.8	122.0	163.8	1.5	9.9	243.9	255.2
29-Oct-18	15.5	1.9	17.4	0.0	0.0	8.0	8.0
30-Nov-18	5.8	0.7	6.6	0.0	0.1	0.0	0.1
3-Dec-18	24.5	6.5	31.0	0.0	1.2	5.7	6.9
31-Jan-19	3.8	1.3	5.0	0.2	0.0	3.4	3.6
27-Feb-19	49.9	9.4	59.4	1.7	1.1	43.4	46.2
5-Mar-19	7.9	2.5	10.4	0.0	0.0	16.2	16.3
29-Apr-19	3.6	0.1	3.7	0.0	0.0	0.1	0.1
1-May-19	1.3	0.3	1.6	0.0	0.0	0.2	0.2
11-Jun-19							

TABLE D7. PP concentrations as mg/L. Inflow is “I”, outflow is “O”.

PP	I-1	I-2	I Total	O-1	O-2	O-3	O Total
19-Jul-18							
8-Aug-18	7.3	0.0	7.3	0.0	0.0	0.5	0.5
27-Sep-18	27.9	47.5	75.4	5.4	3.8	120.3	129.5
29-Oct-18	8.8	5.8	14.6	0.0	0.0	12.9	12.9
30-Nov-18	2.6	7.4	10.1	0.1	0.0	0.0	0.1
3-Dec-18	14.9	12.0	26.9	0.1	0.2	13.4	13.7
31-Jan-19	17.7	15.2	32.9	5.4	0.0	21.2	26.6
27-Feb-19	99.8	92.8	192.7	29.9	0.7	150.4	181.1
5-Mar-19	15.9	14.8	30.7	0.4	0.4	38.9	39.7
29-Apr-19	11.8	4.7	16.5	0.3	0.0	8.9	9.2
1-May-19	3.2	3.8	7.0	0.2	0.0	1.9	2.1
11-Jun-19							

TABLE D8. *E. coli* concentrations as MPN/100mL. Inflow is “I”, outflow is “O”.

<i>E. coli</i>	I-1	I-2	I Total	O-1	O-2	O-3	O Total
19-Jul-18							
8-Aug-18	6049	156.75	6205.75	813.75		1716.75	2530.5
27-Sep-18	2419.6	2419.6	4839.2			2419.6	2419.6
29-Oct-18	2419.6	1553.1	3972.7			2419.6	2419.6
30-Nov-18	2419.6	365.4	2785			866.4	866.4
3-Dec-18	9931.5	1203.3	11134.8			579.4	579.4
31-Jan-19	40.2	125.9	166.1			128.1	128.1
27-Feb-19	2615.5	355.25	2970.75			1716.75	1716.75
5-Mar-19		248.9	248.9			547.5	547.5
29-Apr-19	209.8	686.7	896.5		4839.2	172.5	5011.7
1-May-19	156.5	172.5	329			47.1	47.1
11-Jun-19	2419.6	2419.6	4839.2			1046.2	1046.2

TABLE D9. TSS concentrations as mg/L. Inflow is “I”, outflow is “O”.

TSS	I-1	I-2	I Total	O-1	O-2	O-3	O Total
19-Jul-18							
8-Aug-18	78.49	62.86	141.35	170.77	0	51.77	222.54
27-Sep-18	79.02	94	173.02	7300	131.13	126.24	7557.37
29-Oct-18	21.95	40.82	62.77			120	120
30-Nov-18	9.55	24.19	33.74	47.91	25.85	18.33	92.09
3-Dec-18	23.77	33.6	57.37	71.2	16.36	19.62	107.18
31-Jan-19	12.44	19.34	31.78	32.24	1062.73	34.69	1129.66
27-Feb-19	46.4	60.39	106.79	872.24	34.44	249.68	1156.36
5-Mar-19	26.22	55.66	81.88	221.67	1672.68	70	1964.35
29-Apr-19	39.47	149.05	188.52	2398.82		55.56	2454.38
1-May-19	14.55	53.49	68.04	4692		28.11	4720.11
11-Jun-19	365.88	28.8	394.68			668.51	668.51

APPENDIX E: NORMALITY TESTS

Table E1. Wetland flow

Inflow	Outflow
$p < 0.005$	$p = 0.034$

Table E2. Surface water quality

Parameter	Inflow	Outflow
ORP	$p = 0.082$	$p = 0.935$
SC	$p = 0.899$	$p = 0.697$
Turbidity	$p < 0.005$	$p < 0.005$
DO	$p = 0.139$	$p = 0.176$
pH	$p = 0.497$	$p = 0.412$
Temp	$p = 0.826$	$p = 0.861$

Table E3. Flow-weighted concentrations

Pollutant	Inflow	Outflow
TN	$p = 0.571$	$p = 0.416$
PN	$p < 0.005$	$p < 0.005$
DIN	$p = 0.905$	$p = 0.522$
NO ₃ -N	$p = 0.685$	$p = 0.439$
NH ₄ -N	$p = 0.274$	$p = 0.199$
PP	$p = 0.258$	$p = 0.392$
PO ₄ -P	$p < 0.005$	$p < 0.005$
TSS	$p = 0.370$	$p = 0.012$
Turbidity	$p < 0.005$	$p < 0.005$
<i>E. coli</i>	$p = 0.008$	$p = 0.041$

Table E4. Mass/Loadings

Pollutant	Inflow	Outflow
TN	p = 0.109	p = 0.006
PN	p < 0.005	p < 0.005
DIN	p = 0.172	p = 0.023
NO ₃ -N	p = 0.260	p = 0.037
NH ₄ -N	p = 0.026	p < 0.005
PP	p < 0.005	p < 0.005
PO ₄ -P	p < 0.005	p < 0.005
TDP	p = 0.030	p = 0.150
TSS	p < 0.005	p < 0.005
<i>E. coli</i>	p = 0.005	p < 0.005

Table E5. Groundwater levels

Shallow	Deep
p = 0.012	p < 0.005

Table E6. Groundwater quality-physicochemical

Parameter	Shallow	Deep
ORP	p = 0.131	p = 0.075
SC	p = 0.193	p < 0.005
DO	p = 0.572	p = 0.928
pH	p = 0.285	p = 0.663
Temp	p = 0.809	p = 0.185

Table E7. Groundwater quality: pollutants

Pollutant	Shallow	Deep
DIN	p = 0.557	p = 0.065
NO ₃ -N	p < 0.005	p < 0.005
NH ₄ -N	p = 0.105	p = 0.013
PO ₄ -P	p = 0.050	p < 0.005
<i>E. coli</i>	p = 0.117	p = 0.022

APPENDIX F: FLOW-WEIGHTED POLLUTANT CONCENTRATIONS

Table F1. N flow-weighted concentrations in grams per second. Inflow is “I”, outflow is “O”.

	PN		DIN		TN		NH4		NO3	
Date	I	O	I	O	I	O	I	O	I	O
19-Jul-18										
8-Aug-18	0.18	0.08	1.41	0.14	1.58	0.22	0.07	0.08	1.34	0.06
27-Sep-18	0.51	0.47	1.14	0.89	1.65	1.36	0.23	0.21	0.86	0.68
29-Oct-18	0.09	0.09	1.60	1.37	1.69	1.46	0.22	0.18	1.38	1.19
30-Nov-18	0.07	0.05	0.94	0.61	1.01	0.66	0.15	0.18	0.78	0.42
3-Dec-18	0.08	0.08	0.88	0.41	0.96	0.49	0.24	0.12	0.27	0.29
31-Jan-19	0.09	0.10	1.14	0.77	1.23	0.86	0.18	0.19	0.96	0.57
27-Feb-19	0.20	0.23	0.71	0.56	0.92	0.79	0.20	0.11	0.51	0.45
5-Mar-19	0.10	0.13	1.19	0.68	1.29	0.80	0.28	0.05	0.91	0.63
29-Apr-19	0.16	0.48	0.69	0.12	0.85	0.60	0.05	0.04	0.64	0.08
1-May-19	0.10	0.11	0.30	0.16	0.41	0.27	0.03	0.05	0.27	0.11
11-Jun-19										
Median	0.10	0.10	1.04	0.58	1.12	0.73	0.19	0.12	0.82	0.44

Table F2. P flow-weighted concentrations in grams per second. Inflow is “I”, outflow is “O”.

	PP		PO4-P	
Date	I	O	I	O
19-Jul-18				
8-Aug-18	0.104	0.017	0.293	0.032
27-Sep-18	0.118	0.118	0.263	0.225
29-Oct-18	0.052	0.062	0.059	0.037
30-Nov-18	0.026	0.017	0.017	0.018
3-Dec-18	0.042	0.040	0.051	0.019
31-Jan-19	0.068	0.053	0.011	0.007
27-Feb-19	0.101	0.096	0.029	0.025
5-Mar-19	0.058	0.064	0.021	0.025
29-Apr-19	0.108	0.117	0.028	0.001
1-May-19	0.058	0.048	0.012	0.004
11-Jun-19				
Median	0.063	0.057	0.028	0.022

Table F3. TSS and turbidity flow-weighted values. TSS is in grams per second and turbidity in NTU. Inflow is “I”, outflow is “O”.

	TSS		Turbidity	
Date	I	O	I	O
19-Jul-18				
8-Aug-18	78.49	51.77	20.20	25.50
27-Sep-18	90.46	203.67	131.60	158.99
29-Oct-18	31.30	120.00	18.50	16.00
30-Nov-18	19.09	33.79	6.35	12.83
3-Dec-18	27.63	19.81	13.31	18.30
31-Jan-19	16.11	34.26	11.25	41.03
27-Feb-19	55.92	304.25	38.28	174.30
5-Mar-19	40.45	73.53	26.80	39.71
29-Apr-19	82.58	76.67	37.05	17.93
1-May-19	44.92	52.53	8.56	13.19
11-Jun-19				
Median	42.69	63.03	19.35	21.90

Table F4. *E. coli* flow-weighted concentration in MPN 100mL s⁻¹. Inflow is “I”, outflow is “O”.

	<i>E. coli</i>	
Date	I	O
19-Jul-18		
8-Aug-18	6049.00	1716.75
27-Sep-18	2419.60	2330.95
29-Oct-18	1989.99	2419.60
30-Nov-18	1081.06	144.40
3-Dec-18	6510.05	547.43
31-Jan-19	85.70	105.61
27-Feb-19	1078.42	1550.98
5-Mar-19	120.28	542.59
29-Apr-19	397.43	170.95
1-May-19	168.98	46.85
11-Jun-19		
Median	1079.74	545.01

APPENDIX G: PHYSICOCHEMICAL PARAMETER DATA

Table G1. Raw pH raw data by month.

pH	I-1	I-2	O-1	O-2	O-3
19-Jul-18					
8-Aug-18	7.26	6.63	7.35		7.07
27-Sep-18	7.03	7.03	6.97	6.76	7.3
29-Oct-18	6.42	5.67			6.52
30-Nov-18	6.22	5.92	6.75	6.76	6.68
3-Dec-18	6.31	5.58	6.83	6.86	6.57
31-Jan-19	6.89	6.48	6.56	6.83	7.02
27-Feb-19	6.55	6.4	6.9	6.9	6.9
5-Mar-19	6.7	6.6	6.5	6.6	6.6
29-Apr-19	7.76	7.43	7.1		7.1
1-May-19	8.04	7.95	7.7		7.3
11-Jun-19	5.79	5.25			6.32

Table G2. Raw ORP raw data by month.

ORP	I-1	I-2	O-1	O-2	O-3
19-Jul-18					
8-Aug-18	31	31	75		54
27-Sep-18	32.6	32.6	14.3	11.9	19.5
29-Oct-18	113	127			74
30-Nov-18	56	69	34	38	21
3-Dec-18	90	88	43	36	2
31-Jan-19	63.5	59.5	14.9	-80.5	11.5
27-Feb-19	67	84	37	15	-1
5-Mar-19	82	90	5	-44	-21
29-Apr-19	97.6	19.7	39		-40
1-May-19	-66	-60	28		-49.8
11-Jun-19	43.5	-94.2			-101

Table G3. Raw DO raw data by month.

DO	I-1	I-2	O-1	O-2	O-3
19-Jul-18					
8-Aug-18	2.12	1.8	2.7		2.45
27-Sep-18	1.97	1.97	0.83	2.01	2.1
29-Oct-18	7.83	8.55			9.44
30-Nov-18	9.29	9.83	11.3	10.4	9.8
3-Dec-18	7.9	7.6	9.6	9.1	8.7
31-Jan-19	12.7	13.75	8.76	5.34	13.1
27-Feb-19	11.9	12	11.2	11.5	11.5
5-Mar-19	12.2	11.6	11.1	7.7	11.9
29-Apr-19	7.7	9.15	8.1		7.1
1-May-19	7.3	8.9	7.6		7.83
11-Jun-19	4	2.55			1.55

Table G4. Raw temperature raw data by month.

Temp	I-1	I-2	O-1	O-2	O-3
19-Jul-18					
8-Aug-18	25	24.65	26		24.4
27-Sep-18	23.77	23.77	22.89	23.59	23.46
29-Oct-18	13.77	13.16			13.9
30-Nov-18	9.79	9.46	10.91	10.47	10.75
3-Dec-18	13.82	13.78	14.6	15.28	14.36
31-Jan-19	4.1	2.73	4.09	7.11	4.06
27-Feb-19	8.69	8.57	8.55	8.58	9.11
5-Mar-19	7.2	6.5	6.9	7.3	6.4
29-Apr-19	15.56	15.16	15.1		15.19
1-May-19	18.5	19.57	19.5		19.29
11-Jun-19	21.77	20.94			20.69

Table G5. Raw SC raw data by month.

SC	I-1	I-2	O-1	O-2	O-3
19-Jul-18					
8-Aug-18	411	616	240		429
27-Sep-18	168	168	203	188	151
29-Oct-18	289	420			327
30-Nov-18	309	509	240	244	395
3-Dec-18	270	368	199	245	329
31-Jan-19	242	428	292	349	175
27-Feb-19	165	278	153	158	223
5-Mar-19	180	280	214	315	228
29-Apr-19	357	567	336		365
1-May-19	302	588	359		434
11-Jun-19	1418	317			322

Table G6. Raw turbidity data by month.

Turbidity	I-1	I-2	O-1	O-2	O-3
19-Jul-18					
8-Aug-18	20.2	16.2	121		25.5
27-Sep-18	140	129	140	92	161
29-Oct-18	18	19			74
30-Nov-18	7	6	16	12	7
3-Dec-18	9	20	34	61	16
31-Jan-19	7	15	116	60	25
27-Feb-19	41	37	337	52	159
5-Mar-19	21	33	74	1000	38
29-Apr-19	52	14	1000		9
1-May-19	7	9	1000		8
11-Jun-19	16	16			260

Table G7. Flow-weighted physicochemical parameter data by month for inlets and outlets.
Inflow is “I”, outflow is “O”.

	pH		ORP		DO		Temp		SC		Turb.	
Date	I	O	I	O	I	O	I	O	I	O	I	O
19-Jul-18												
8-Aug-18	7.3	7.1	31	54	2.12	2.45	25.0 0	24.4 0	411	429	20	26
27-Sep-18	7.0	7.3	33	19	1.97	2.08	23.7 7	23.4 6	168	153	13 2	159
29-Oct-18	6.0	6.5	120	74	8.19	9.44	13.4 7	13.9 0	354	327	18	74
30-Nov-18	6.0	6.7	64	34	9.64	10.68	9.57	10.7 0	439	268	6	13
3-Dec-18	6.0	6.6	89	4	7.78	8.73	13.8 0	14.4 1	308	324	13	18
31-Jan-19	6.7	6.9	61	12	13.26	12.34	3.37	4.07	341	196	11	41
27-Feb-19	6.4	6.9	79	3	11.97	11.47	8.61	9.06	242	216	38	174
5-Mar-19	6.7	6.6	86	-21	11.91	11.89	6.86	6.41	228	228	27	40
29-Apr-19	7.6	7.1	67	-39	8.27	7.11	15.4 0	15.1 9	440	365	37	18
1-May-19	8.0	7.3	-61	-49	8.55	7.83	19.3 3	19.2 9	525	434	9	13
11-Jun-19												
Median	6.66	6.9 2	66	8	8.41	9.08	13.6 4	14.1 5	347	296	19	33
Min	6.0	6.5	-61	-49	1.97	2.08	3.37	4.07	168	153	6	13
Max	8.0	7.3	120	74	13.26	12.34	25.0 0	24.4 0	525	434	13 2	174

APPENDIX H: HIGH-FREQUENCY PHYSICOCHEMICAL PARAMETERS

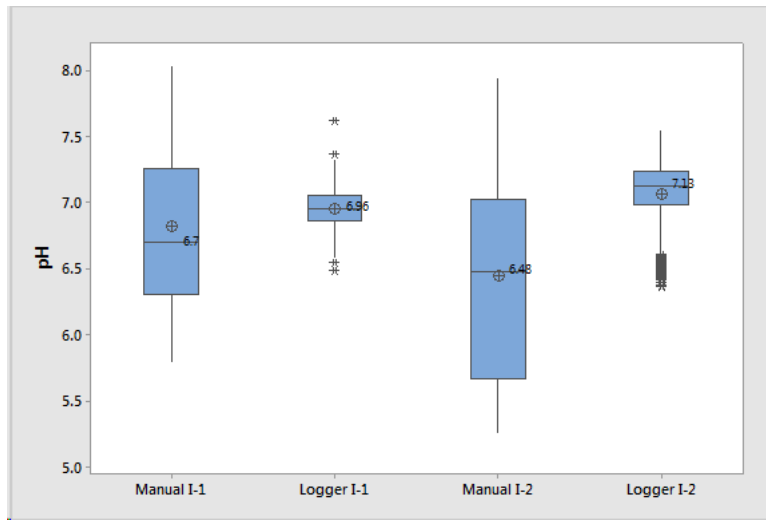


Figure H1. Manual and high-frequency logger comparison of pH.

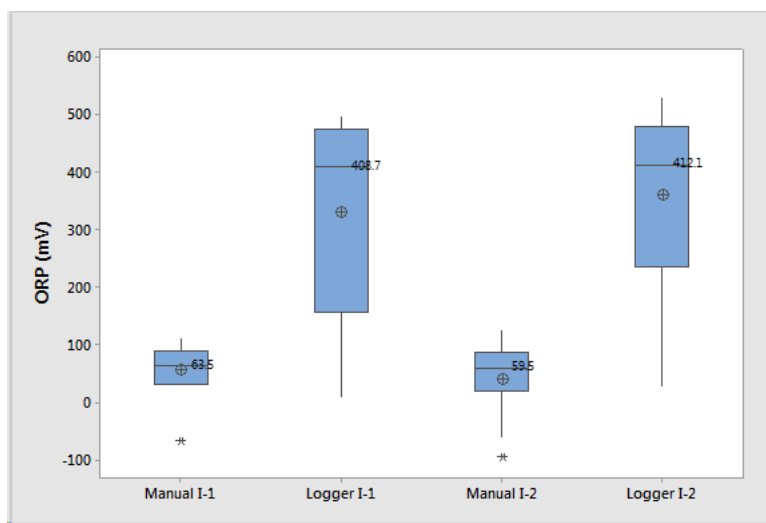


Figure H2. Manual and high-frequency logger comparison of ORP.

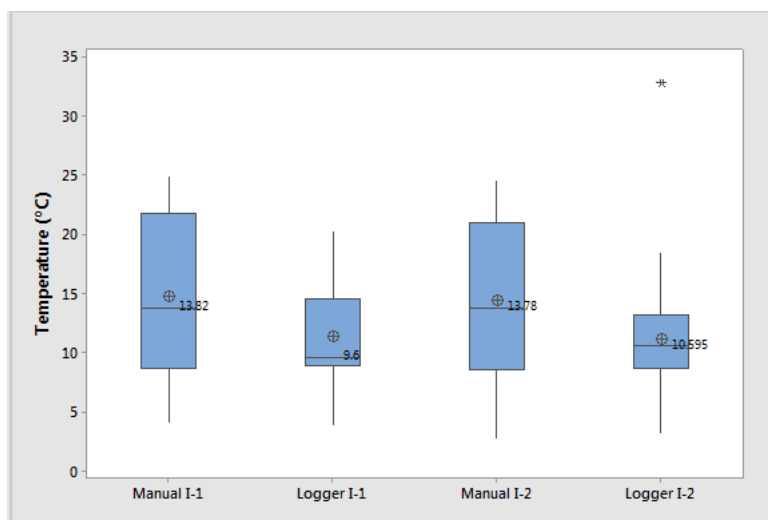


Figure H3. Manual and high-frequency logger comparison of temperature.

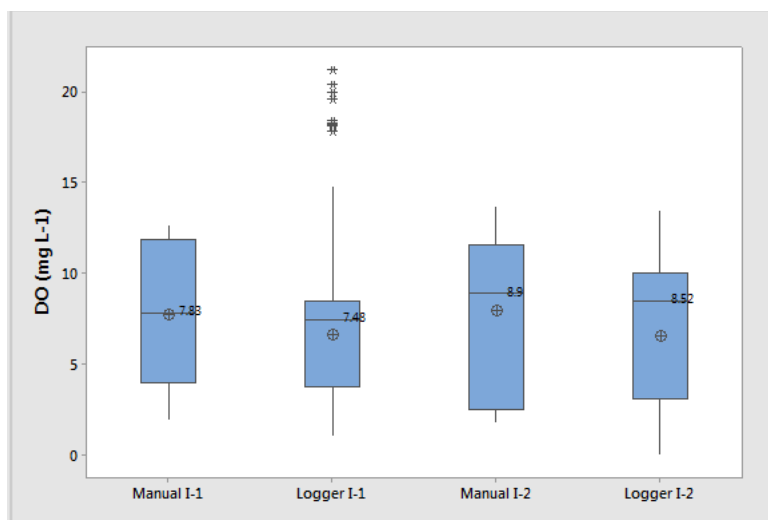


Figure H4. Manual and high-frequency logger comparison of DO.

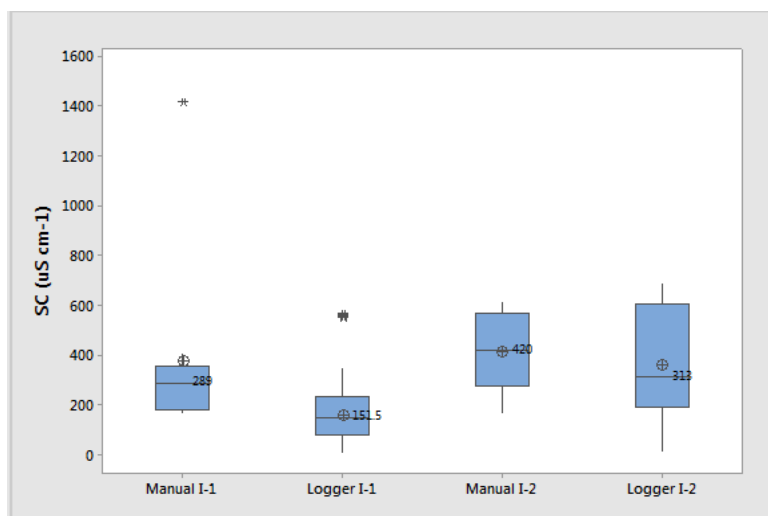


Figure H6. Manual and high-frequency logger comparison of SC.

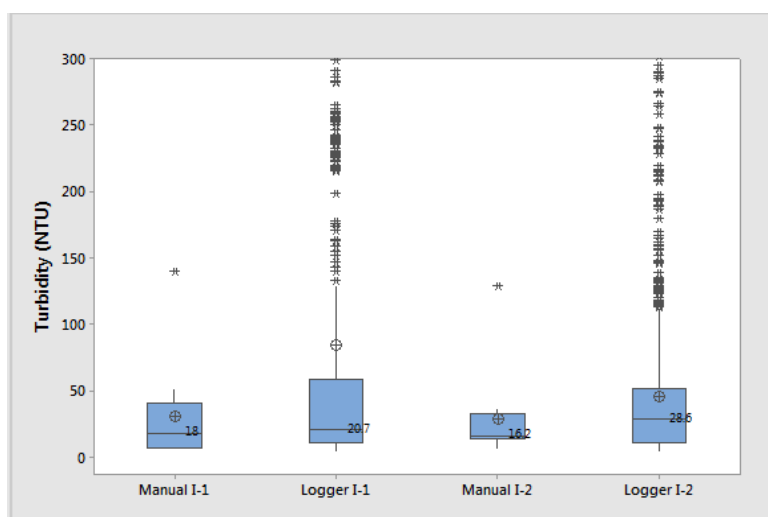


Figure H7. Manual and high-frequency logger comparison of physicochemical parameters. Scale maximum decreased to 300 NTU for visual acuity purposes in turbidity.

APPENDIX I: GROUNDWATER LEVELS

Table I1. Shallow piezometer water level (in feet). Median water level is 0.16 feet below ground level according to these calculations (1.82 feet minus 1.66 feet). Negative values indicate flooding (above ground).

Piezometer-S	Field DTW	Corrected DTW (minus above ground)	Total Well	Above Ground	Well Underground	Water Level
27-Sep-18	1.88	0.23	3.47	1.65	1.82	1.59
29-Oct-18	1.81	0.16				1.66
30-Nov-18	1.62	-0.03				1.85
3-Dec-18	1.6	-0.05				1.87
31-Jan-19	1.71	0.06				1.76
27-Feb-19	1.71	0.06				1.76
5-Mar-19	1.88	0.23				1.59
29-Apr-19	2.46	0.81				1.01
1-May-19	2.75	1.1				0.72
Median	1.81	0.16				1.66

Table I2. Deep piezometer water level (in feet). Median water level is 0.51 feet below ground level according to these calculations (6.88 feet minus 6.37 feet).

Piezometer-D	Field DTW	Corrected DTW (minus above ground)	Total Well	Above Ground	Well Underground	Water Level
29-Oct-18	7.2	4.08	10	3.12	6.88	2.8
30-Nov-18	3.52	0.4				6.48
3-Dec-18	3.69	0.57				6.31
31-Jan-19	3.47	0.35				6.53
27-Feb-19	3.58	0.46				6.42
5-Mar-19	3.56	0.44				6.44
29-Apr-19	4.22	1.1				5.78
1-May-19	4.99	1.87				5.01
11-Jun-19						
Median	3.635	0.515				6.365

* Corrected DTW considers how far down water level is if you remove the above ground portion of the pipe. Water level was calculated by subtracting the corrected DTW from the total well length.

APPENDIX J: GROUNDWATER CONCENTRATIONS

Table J1. Piezometer nutrient (mg/L) and bacteria (MPN/100 mL) raw data. The deep piezometer was created after the shallow piezometer. Empty boxes indicate unattainable data.

Piezometer-S	DIN	NO ₃ -N	NH ₄ -N	PO ₄	<i>E. coli</i>
27-Sep-18	1.09	0.13	0.96	0.027	
29-Oct-18	0.69	0.54	0.15	0.154	2419.6
30-Nov-18	0.41	0.11	0.3	0.048	
3-Dec-18	0.35	0.09	0.26	0.071	816.4
31-Jan-19	0.93	0.05	0.88	0.053	
27-Feb-19	0.19	0.08	0.11	0.05	41.7
5-Mar-19	0.19	0.03	0.16	0.06	39.3
29-Apr-19	0.64	0.03	0.61	0.001	
1-May-19	0.7	0.02	0.68	0.001	2419.6
median	0.64	0.08	0.3	0.05	816.4

Table J2. Deep piezometer nutrient (mg/L) and bacteria (MPN/100 mL) data. Empty boxes indicate unattainable data.

Piezometer-D	DIN	NO ₃ -N	NH ₄ -N	PO ₄	<i>E. coli</i>
29-Oct-18	0.84	0.65	0.19	0.001	
30-Nov-18	0.46	0.17	0.29	0.042	
3-Dec-18	0.28	0.03	0.25	0.001	1
31-Jan-19	0.92		0.92	0.001	
27-Feb-19	0.01	0.01		0.001	0
5-Mar-19	0.03	0.01	0.02	0.001	0
29-Apr-19	0.11	0.05	0.06	0.001	1
1-May-19	0.02	0.01	0.01	0.001	4
median	0.20	0.03	0.19	0.001	1

Table J3. Shallow piezometer physicochemical raw data.

Piezometer-S	SC	DO	ORP	pH	Temperature
27-Sep-18	289		-13.4	6.34	23.14
29-Oct-18	276	5.8	106	6.27	14.9
30-Nov-18	233	4.4	35	6.2	10.32
3-Dec-18	228	4.9	84	6.3	14.9
31-Jan-19	224	2.62	67.5	6.76	5.14
27-Feb-19	225	3.3	57	6.5	8.85
5-Mar-19	202	3.8	77	6.7	7.4
29-Apr-19	350	2.9	-77	7.02	16.71
1-May-19	318	3.3	-95	7	18
Median	233	3.55	57	6.5	14.9

Table J4. Deep piezometer physicochemical raw data.

Piezometer-D	SC	DO	ORP	pH	Temperature
29-Oct-18	275	5.5	134	5.5	17.6
30-Nov-18	527	3.8	-28	6.3	13.7
3-Dec-18	565	3.59	-48.4	6.26	15.1
31-Jan-19	602	2.94	121.1	6.56	7.18
27-Feb-19	612	4.5	-91	6.6	9.95
5-Mar-19	604	2.4	-108	6.6	9
29-Apr-19	540	4.4	-114	6.95	16.3
1-May-19	550	2.8	-146	7.5	16.5
11-Jun-19	539	1.9	-158	6.05	17.9
Median	550	3.59	-91	6.56	15.1

APPENDIX K: N MASS

Table K1. DIN mass movement in grams per day. Empty boxes indicate unattainable data.

Date	I-1	I-2	O-1	O-2	O-3	Inlets	outlets
19-Jul-18							
8-Aug-18	103.3	0.0	0.0	0.0	3.4	103.3	3.4
27-Sep-18	283.3	426.5	8.1	42.4	965.2	709.8	1015.7
29-Oct-18	405.1	60.1	0.0	0.0	295.1	465.2	295.1
30-Nov-18	214.4	140.3	0.5	2.5	0.6	354.7	3.6
3-Dec-18	439.6	99.8	0.4	18.5	126.9	539.4	145.8
31-Jan-19	391.7	148.4	50.5	0.0	342.9	540.1	393.4
27-Feb-19	728.5	663.7	43.4	7.1	988.3	1392.2	1038.7
5-Mar-19	476.1	132.5	0.2	0.0	442.4	608.6	442.6
29-Apr-19	80.3	22.9	0.1	0.0	9.3	103.2	9.4
1-May-19	23.3	14.0	0.0	0.0	7.4	37.3	7.4
11-Jun-19							
Median						502.3	220.4

Table K2. PN mass movement in grams per day. Empty boxes indicate unattainable data.

Date	I-1	I-2	O-1	O-2	O-3	Inlets	outlets
19-Jul-18							
8-Aug-18	13.0	0.0	0.0	0.0	2.1	13.0	2.1
27-Sep-18	77.5	239.5	3.5	13.9	511.6	317.0	529.0
29-Oct-18	13.6	13.8	0.0	0.0	19.1	27.3	19.1
30-Nov-18	11.3	15.3	0.2	0.1	0.0	26.6	0.3
3-Dec-18	26.9	23.6	0.2	0.7	28.0	50.5	28.9
31-Jan-19	24.0	19.7	13.2	0.0	36.4	43.7	49.6
27-Feb-19	166.1	230.4	65.6	1.7	355.0	396.6	422.3
5-Mar-19	17.3	32.4	3.1	0.3	80.2	49.7	83.6
29-Apr-19	8.8	14.5	0.7	0.0	38.4	23.3	39.1
1-May-19	3.9	8.7	0.4	0.0	4.6	12.5	5.0
11-Jun-19							
Median						35.5	34.0

Table K3. TN mass movement in grams per day. Empty boxes indicate unattainable data.

Date	I-1	I-2	O-1	O-2	O-3	Inlets	outlets
19-Jul-18							
8-Aug-18	116.2	0.0	0.0	0.0	5.5	116.2	5.5
27-Sep-18	360.8	666.0	11.6	56.3	1476.8	1026.8	1544.7
29-Oct-18	418.6	73.8	0.0	0.0	314.2	492.5	314.2
30-Nov-18	225.7	155.6	0.7	2.6	0.6	381.3	3.9
3-Dec-18	466.5	123.4	0.6	19.1	154.9	589.9	174.7
31-Jan-19	415.7	168.1	63.7	0.0	379.3	583.8	443.0
27-Feb-19	894.7	894.1	109.0	8.7	1343.2	1788.8	1461.0
5-Mar-19	493.4	164.8	3.3	0.3	522.6	658.3	526.2
29-Apr-19	89.1	37.4	0.7	0.0	47.8	126.5	48.5
1-May-19	27.2	22.6	0.4	0.0	12.0	49.8	12.4
11-Jun-19							
Median						538.1	244.4

Table K4. NO₃-N mass movement in grams per day. Empty boxes indicate unattainable data.

Date	I-1	I-2	O-1	O-2	O-3	Inlets	outlets
19-Jul-18							
8-Aug-18	98.04	0.00	0.00	0.00	1.55	98.04	1.55
27-Sep-18	203.38	333.64	2.45	33.48	740.75	537.02	776.68
29-Oct-18	361.86	39.93	0.00	0.00	256.35	401.79	256.35
30-Nov-18	194.77	102.72	0.08	2.03	0.39	297.49	2.50
3-Dec-18	113.12	49.32	0.01	16.16	87.13	162.45	103.30
31-Jan-19	362.22	91.40	8.56	0.00	285.18	453.61	293.74
27-Feb-19	572.66	424.13	8.72	5.37	813.47	996.79	827.55
5-Mar-19	386.42	78.47	0.19	0.00	412.36	464.89	412.55
29-Apr-19	76.44	19.74	0.00	0.00	6.44	96.19	6.44
1-May-19	21.74	11.37	0.00	0.00	5.03	33.11	5.03
11-Jun-19							
Median						349.64	179.82

Table K5. NH₄-N mass movement in grams per day. Empty boxes indicate unattainable data.

Date	I-1	I-2	O-1	O-2	O-3	Inlets	outlets
19-Jul-18							
8-Aug-18	4.85	0.00	0.00	0.00	1.88	4.85	1.88
27-Sep-18	51.46	92.84	5.64	8.90	221.40	144.30	235.94
29-Oct-18	43.23	20.13	0.00	0.00	38.70	63.36	38.70
30-Nov-18	19.65	37.59	0.44	0.44	0.20	57.23	1.08
3-Dec-18	93.60	50.49	0.41	2.32	39.78	144.09	42.52
31-Jan-19	29.49	56.97	41.97	0.00	57.74	86.46	99.71
27-Feb-19	155.88	239.53	34.67	1.70	174.79	395.41	211.17
5-Mar-19	89.70	54.01	0.01	0.00	30.00	143.71	30.01
29-Apr-19	3.82	3.16	0.05	0.00	2.91	6.99	2.96
1-May-19	1.58	2.59	0.00	0.00	2.36	4.17	2.36
11-Jun-19							
Median						74.91	34.35

APPENDIX L: P MASS

Table L1. PP mass movement in grams per day. Empty boxes indicate unattainable data.

Date	I-1	I-2	O-1	O-2	O-3	Inlets	outlets
19-Jul-18							
8-Aug-18	7.7	0.0	0.0	0.0	0.4	7.7	0.4
27-Sep-18	27.7	45.7	5.4	3.7	125.2	73.4	134.3
29-Oct-18	8.8	6.2	0.0	0.0	13.4	15.0	13.4
30-Nov-18	2.6	7.2	0.1	0.0	0.0	9.8	0.1
3-Dec-18	14.1	11.5	0.1	0.1	14.0	25.6	14.2
31-Jan-19	17.6	14.6	5.8	0.0	21.3	32.3	27.1
27-Feb-19	102.8	93.9	29.2	0.7	147.4	196.7	177.4
5-Mar-19	15.6	13.8	0.5	0.4	41.0	29.4	41.8
29-Apr-19	11.6	4.5	0.3	0.0	9.2	16.1	9.5
1-May-19	3.3	3.7	0.2	0.0	2.0	7.1	2.2
11-Jun-19							
Median						20.8	13.8

Table L2. PO₄-P mass movement in grams per day. Empty boxes indicate unattainable data.

Date	I-1	I-2	O-1	O-2	O-3	Inlets	outlets
19-Jul-18							
8-Aug-18	21.5	0.0	0.0	0.0	0.8	21.5	0.8
27-Sep-18	41.8	121.8	1.5	9.9	243.6	163.6	255.0
29-Oct-18	15.4	1.9	0.0	0.0	8.1	17.2	8.1
30-Nov-18	5.8	0.8	0.0	0.1	0.0	6.5	0.1
3-Dec-18	24.7	6.4	0.0	1.2	5.6	31.1	6.8
31-Jan-19	3.8	1.4	0.2	0.0	3.4	5.1	3.6
27-Feb-19	47.4	9.4	1.1	1.2	43.2	56.8	45.4
5-Mar-19	8.8	2.0	0.0	0.0	16.3	10.8	16.4
29-Apr-19	4.1	0.1	0.0	0.0	0.1	4.1	0.1
1-May-19	1.2	0.3	0.0	0.0	0.2	1.5	0.2
11-Jun-19							
Median						14.0	5.2

Table L3. TDP mass movement in grams per day. Empty boxes indicate unattainable data.

Date	I-1	I-2	O-1	O-2	O-3	Inlets	outlets
31-Jan-19							
27-Feb-19	56.1	39.8	3.3	1.5	100.3	95.9	105.1
5-Mar-19	15.9	0.0	0.2	0.0	32.4	15.9	32.6
29-Apr-19	5.4	2.3	0.0	0.0	3.2	7.8	3.3
1-May-19	2.2	2.9	0.0	0.0	0.5	5.0	0.5
11-Jun-19							
Median						11.8	17.9

APPENDIX M: TSS MASS

Table M1. TSS mass movement in grams per day. Empty boxes indicate unattainable data.

Date	I-1	I-2	O-1	O-2	O-3	Inlets	outlets
19-Jul-18							
8-Aug-18	5761.3	0.0	0.0	0.0	1266.7	5761.3	1266.7
27-Sep-18	11600.3	44615.7	89299.9	3849.8	138063.0	56216.0	231212.7
29-Oct-18	3222.3	5891.7	0.0	0.0	25835.8	9114.1	25835.8
30-Nov-18	1261.1	5976.5	117.2	63.3	17.9	7237.6	198.4
3-Dec-18	8840.9	8056.1	174.2	280.2	6574.7	16897.0	7029.1
31-Jan-19	2770.6	4873.5	2918.9	0.0	14684.4	7644.1	17603.3
27-Feb-19	28947.8	80082.5	145112.6	421.4	417213.1	109030.4	562747.1
5-Mar-19	6929.4	13753.9	1084.6	1636.9	45383.9	20683.3	48105.5
29-Apr-19	3573.3	8751.7	1760.7	0.0	4485.4	12325.0	6246.0
1-May-19	391.5	5103.7	1147.9	0.0	1306.6	5495.1	2454.5
11-Jun-19							
Median						10719.5	12316.2

APPENDIX N: *E. COLI* LOADING

Table N1. *E. coli* loading data in MPN 100 mL-1. Empty boxes indicate unattainable data.

Date	I-2	I-2	O-1	O-2	O-3	Inlets	Outlets
19-Jul-18							
8-Aug-18	51386.5	0.0	0.0	0.0	4861.3	51386.5	4861.3
27-Sep-18	41109.2	132919.7	0.0	0.0	306263.5	174028.9	306263.5
29-Oct-18	41109.2	25947.5	0.0	0.0	60293.5	67056.7	60293.5
30-Nov-18	36998.3	10450.4	0.0	0.0	98.1	47448.7	98.1
3-Dec-18	427467.0	33392.1	0.0	0.0	22477.3	460859.1	22477.3
31-Jan-19	1035.9	3672.0	0.0	0.0	6280.3	4707.9	6280.3
27-Feb-19	188859.6	54522.7	0.0	0.0	332025.9	243382.3	332025.9
5-Mar-19	0.0	7118.5	0.0	0.0	41084.1	7118.5	41084.1
29-Apr-19	2198.1	4666.8	0.0	0.0	1611.9	6865.0	1611.9
1-May-19	487.5	1905.0	0.0	0.0	253.4	2392.5	253.4
11-Jun-19							
Median						49417.6	14378.8

APPENDIX O: LOGGER AND MANUAL STAGE

Table O1. High-frequency logger stage readings at I-1 and I-2. Monthly median logger depth in cm. Logger readings were recorded every 20 minutes. Manual readings missed February 2019, and April 2019.

Date	I-1	I-2
July, 2018	10.25	10.27
August, 2018	10.37	10.38
September, 2018	10.40	10.40
October, 2018	10.44	10.40
November, 2018	10.49	10.49
December, 2018	10.46	10.50
January, 2019	10.43	10.48
February, 2019	10.44	10.48
March, 2019	10.43	10.47
April, 2019	10.40	10.44
May, 2019	10.27	10.29
June, 2019	10.23	10.26

Table O2. Manual monthly readings in cm. Logger readings were recorded every 20 minutes. Manual readings missed February 2019, and April 2019.

Date	Inlet 1	Inlet 2
July, 2018	0	0
August, 2018	0	0
September, 2018	15.8	0
October, 2018	13.4	19.5
November, 2018	17.4	18.9
December, 2018	17.1	18.3
January, 2019	9.1	20.7
March, 2019	10.1	21.3
May, 2019	7.01	18.3
June, 2019	0	0

