IMPACTS OF COASTAL STORMWATER POND NITROGEN CYCLING ON DOWNSTREAM WATER QUALITY

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ABSTRACT

Adam C. Gold: Impacts of coastal stormwater pond nitrogen cycling on downstream water quality
(Under the direction of Michael Piehler)

This thesis investigates the impacts of stormwater pond nitrogen cycling on downstream water quality by using long-term stream water quality monitoring data and nitrogen cycling data from within stormwater ponds collected on Marine Corps Base Camp Lejeune in North Carolina. A comparison of water quality before and after watershed development and stormwater wet pond implementation showed that wet ponds did not mitigate the negative effects of development on water quality. Additionally, wet ponds were shown to be sources of suspended solids and algae and sinks for nitrate. Summer measurements of net N_2 fluxes from the sediment-water interface from a chronosequence of pond sediments showed net nitrogen fixation throughout the summer. Also, the response of net sediment N_2 fluxes to nitrate loading was negatively correlated with pond age.

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PREFACE

This thesis consists of two complete and separate manuscripts written with the intent of publishing both in scientific journals. The overarching research topic of the thesis is the water quality impact of stormwater wet ponds in the coastal plain of the southeastern US. The goal of the thesis is to provide insight for more effective management of wet ponds in the southeastern coastal plain by investigating both the watershed-scale effects and internal nitrogen dynamics of wet ponds. Chapter one focuses on the effectiveness of watershed-scale wet pond implementation to mitigate the water quality effects of stormwater from increased impervious area. Chapter two investigates nitrogen dynamics in wet ponds during summer and fall and the effects of pond age and location in pond on net N_2 fluxes. All research was conducted on Marine Corps Base Camp Lejeune in Jacksonville, NC.

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CHAPTER 1: WATER QUALITY BEFORE AND AFTER WATERSHED-SCALE IMPLEMENTATION OF STORMWATER WET PONDS IN THE COASTAL PLAIN

1. Introduction

Nearly 80% of the US population lives in urban areas, and this percentage is increasing (US Census Bureau, 2010). Concomitantly, the amount of impervious area is increasing due to the expansion of urban and sub-urban areas (Terando et al., 2014). Specifically, the coastal plain of the southeastern US is predicted to experience urban expansion over the next 50 years (Terando et al., 2014). Despite known negative impacts of stormwater runoff from urban areas on coastal stream hydrology and water quality, research on stormwater mitigation techniques in coastal regions is very limited when compared to extensive research in non-coastal regions (Ex. DeLorenzo, 2012; EPA, 2014; Lewitus, 2008; Merriman et al., 2016; Serrano and DeLorenzo, 2008). Coastal stormwater managers apply similar stormwater control measures (SCMs) as managers in non-coastal areas and have the same priorities for water quantity and quality (Collins et al., 2010). To test the assumption that stormwater management in coastal systems and non-coastal systems can be approached the same way, it is necessary to determine the effects of prevalent types of SCMs, particularly wet ponds, on the water quality of coastal watersheds.

The effects of increased watershed impervious area on streams are well-studied and predictable in most geographic regions of the US, including coastal systems (Ex. O'Driscoll et al., 2009; O'Driscoll et al., 2010). As watershed impervious area increases, more runoff is generated from storm events, and evaporation and infiltration within the watershed decreases. Typically, the total volume of water leaving a watershed increases due to an increase in stormflow and a decrease in baseflow (Booth and Jackson, 1997; O'Driscoll et al., 2010; Paul et al., 2001; Walsh et al., 2005), although the effect of increased impervious area on baseflow dynamics can vary (Price, 2011). Changes in catchment hydrology due to development generally leads to lower stream biota diversity, increased loading of nutrients and other pollutants, and channel incision or enlargement (Goetz et al., 2008; Paul et al., 2001; Walsh et al., 2005). Similar effects have been observed in urban areas within the southeastern coastal plain of the US (O'Driscoll et al., 2009; O'Driscoll et al., 2010; Sanger et al., 2013).

Conventional stormwater management has focused on the objectives of flood mitigation and pollutant removal (Burns et al., 2012; Walsh et al., 2016), and most SCMs have focused on detaining stormwater and slowly releasing it to lower peak flows (Collins et al., 2010). The most prevalent kind of SCM is a wet pond, which is designed to hold a large volume of runoff and retain a permanent pool of water (Collins et al., 2010). Wet ponds are primarily intended to mitigate increased surface runoff from impervious surfaces during storms by lowering peak stormflows and extending the hydrograph (Hancock et al., 2010), but the effects of these ponds on downstream water quality are not well constrained. In some cases wet ponds have been shown to offer valuable ecosystem services, such as increased biodiversity (Hassall and Anderson, 2014; Moore and Hunt, 2012), carbon sequestration (Moore and Hunt, 2012), and nutrient and suspended sediment retention (Bettez and Groffman, 2012; McPhillips and Walter, 2015; Rosenzweig et al., 2011). Conversely, some studies have shown that wet ponds failed to meet regulatory goals for stream channel protection (Hancock et al., 2010), increased nutrient loading at times (Duan et al., 2016; Rosenzweig et al., 2011), created longer periods of erosive stormflow (Tillinghast et al., 2011), increased heavy metal concentrations (Stephansen et al., 2014; Wium-Andersen et al., 2013), and grew harmful algae and bacteria (DeLorenzo, 2012; Lewitus et al., 2008).

The implementation of wet ponds may have distinctive effects on water quality in coastal watersheds in the southeastern US due to the landscape's high water table, low relief, soil type, and biogeochemistry. Many coastal watersheds in the southeastern coastal plain have soils and natural hydrologic and biogeochemical processes that produce blackwater streams - streams characterized by large amounts of dissolved organic matter and low concentrations of chlorophyll-*a* and suspended sediments (Meyer, 1990). The optical properties, nutrient concentrations, and suspended sediment concentrations of the blackwater naturally found in coastal streams is significantly different than the water funneled into wet ponds from impervious surfaces (Piehler et al., in prep). Few studies have investigated the effects of watershed-scale implementation of wet ponds on coastal stream water quality, but many of the SCMs in coastal NC counties are wet ponds or dry ponds (NCDEQ, 2017). Previous research on coastal stormwater management has focused on water quality in tidal and brackish water or on single SCMs (Ex. DeLorenzo, 2012; Lewitus, 2008; Merriman et al., 2016; Serrano and DeLorenzo, 2008). Improving and broadening the understanding of watershed-scale stormwater management in coastal areas will have clear implications for coastal water quality, public health, and estuarine ecology.

Another unresolved issue is how pollutant removal functions of coastal wet ponds may change over time. Wet ponds fill in with vegetation and sediment over time, but the establishment of vegetation in deeper parts of the ponds is discouraged (Mitsch and Jørgensen, 2004). The excavation of in-filled areas every few years in wet ponds and wetlands is required to maintain water storage capacity and sediment and phosphorus removal (Hunt and Lord, 2006;

Merriman and Hunt, 2014). This wet pond maintenance, like most SCM maintenance, is often overlooked but recommended (Blecken et al., 2015). Understanding how stream water quality from a coastal watershed outfitted with stormwater wet ponds changes over time will inform plans for excavation to maximize nutrient and suspended sediment removal and demonstrate the need for maintenance in coastal wet ponds. Few studies have investigated how the pollutant removal function of SCMs changes over extended periods of time (ex. Merriman and Hunt, 2014; Merriman and Hunt, 2016), and none have been conducted on wet ponds in a coastal watershed.

Here I examined the effects of watershed-scale wet pond implementation and increased development on coastal stream water quality by analyzing a time series of nutrient, total suspended solids, and chlorophyll-*a* concentration data. Assessing the efficacy of coastal wet ponds through analysis of data before and after wet pond implementation offers a unique opportunity to understand the role these structures play in shaping coastal water quality and mitigating the negative effects of increased development. Our data span seven years, encompassing before, during, and after increased development and concurrent implementation of wet ponds in a developed coastal watershed and parallel sampling in a minimally developed reference coastal watershed aboard US Marine Corps Base Camp Lejeune in coastal North Carolina.

The goals of this study were to:

1) Quantify the changes in stream chemistry that occurred due to increased development and the watershed-scale implementation of wet ponds.
2) Identify trends in stream nutrient and suspended sediment concentrations after development and the implementation of wet ponds.

3) Determine if wet ponds were functioning as sources or sinks for nitrogen and phosphorus, suspended sediments, and chlorophyll-*a*
4) Assess implications for future coastal stormwater management and wet pond management along the US Southeastern coast and other similar systems.

2. Site Description

Study watersheds sampled were located aboard US Marine Corps Base Camp Lejeune in Jacksonville, NC in the coastal plain of North Carolina (Figure 1). Camp Lejeune is the largest US Marine base in the world, employing 170,000 people and covering an area of 640 km^2 (http://www.lejeune.marines.mil/About.aspx). Camp Lejeune surrounds the New River Estuary, and has installed over 200 wet ponds to mitigate negative hydrologic impacts of increased impervious area on coastal streams. The New River Estuary, like many other estuaries in NC, has experienced intense eutrophication in the past due to high levels of nutrient loading (Mallin et al., 2005), so understanding the effects of stormwater management on nutrient dynamics is imperative. The two study streams drain into the New River Estuary but did not experience significant tidal fluctuations or any salinity during the study period.

The developed watershed (70 ha, 28% mean imperviousness (Xian et al., 2011)) for this study is located in a residential neighborhood called Tarawa Terrace on the northern boundary of the estuary (Figure 1). Between January 2009 and March 2011, the existing homes were demolished and completely rebuilt. This development increased the mean imperviousness of the watershed by 5.2% (Figure 2). Seven wet ponds were constructed during this time period, covering 2.4 ha (3.4% of the watershed area) and receiving nearly all surface water drainage from the watershed (97 % of watershed area, 68 ha). By the end of the study, all wet ponds were fringed with marsh vegetation, mainly cattails (*Typha* spp.), and each pond had alligator weed

(*Alternanthera philoxeroides*) established at the permanent pond surface that reached into the open water, covering approximately 30% of the pond surface.

The French Creek watershed (835 ha, 1.2 % mean imperviousness (Xian et al., 2011)) was the reference watershed for this study (Figure 1). The watershed has been partially cleared but contains large areas of woody wetlands and shrubs, has very low levels of imperviousness, and exhibits characteristics of an undeveloped blackwater coastal stream system (Figure 2). This watershed encompasses a bombing range and some gravel roads. The reference watershed is located on the eastern side of the New River Estuary and does not have any SCMs (Figure 1). This watershed maintained its hydrologic patterns and blackwater characteristics during this study.

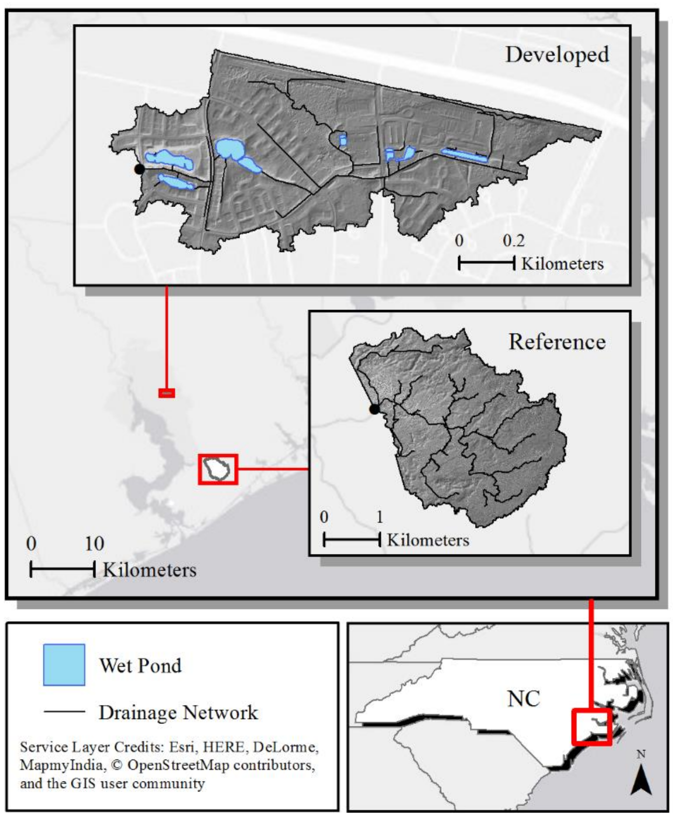

Figure 1. Location of study watersheds within North Carolina and hillshade with drainage network and wet ponds.

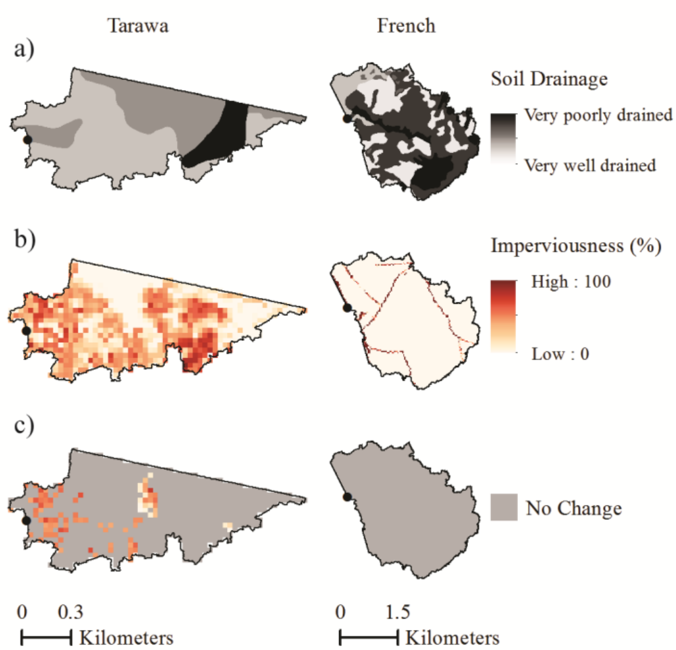

Figure 2. a) Soil drainage of the study watersheds b) 2011 National Land Cover Database percent imperviousness c) 2006 - 2011 change in imperviousness.

The developed watershed's soils are primarily well-drained and moderately well-drained (Soil Survey Staff, 2015), although the soil classification in the developed watershed incorporates the extensive development and storm sewer drainage (Figure 2). There is a patch of very poorly-drained soil near the top of the watershed. The outlet of the watershed is located next to the outlet of two wet ponds, and a third wet pond is located approximately .35 km from the watershed outlet within the stream network (Figure 1). Four more wet ponds are located higher in the watershed. Reference watershed soils are a mix of poorly-drained and well-drained soils (Soil Survey Staff, 2015), and the natural stream drainage network is unaltered (Figure 2).

French watershed was selected as a reference in this study because of its proximity to the developed watershed, its low amount of impervious area, and its lack of disturbance during the time period of construction in the developed watershed, despite distinctions in watershed soil types and watershed area. Although not considered a control, this study uses French as a reference with the aim of comparing temporal trends in nutrient, total suspended solids, and chlorophyll-*a* concentrations in each watershed's stream.

3. Methods

Sampling occurred over a period of seven years, beginning in January 2008 and ending in June 2015 for both the developed and reference watersheds. Water samples from each watershed's stream were collected every two weeks during baseflow and throughout the course of one storm event each month. Samples during storm events were collected using Teledyne Isco automatic water samplers programmed to collect samples after the stream velocity passed a certain threshold that was unique for each stream and paced to provide samples from the rising limb, peak, and falling limb of the storm hydrographs. Storm samples collected by Isco's were transported as quickly as possible (always within 48 hours of the storm event) for sample processing at the University of North Carolina at Chapel Hill's Institute of Marine Sciences (UNC IMS). Water samples were analyzed for concentrations of nitrate-nitrite (NO_x^--N, µM), ammonium (NH_4^+, µM), orthophosphate (PO_4^{3-}, µM), total nitrogen (TN, µM), organic nitrogen (ON, µM), chlorophyll-*a* (chl-*a*, µg/L), and total suspended solids (TSS, mg/L). All data were Log_{10}-transformed before analysis to fit assumptions for parametric statistical testing. A value of 10^{-6} was added to all data before log transformation due to multiple values of zero (below detection limit) in the data set.

Water quality data for each stream were partitioned into three time periods based on the timing of construction in the developed watershed: Pre-Construction (Pre), Construction (Mid), and Post-Construction (Post) (Table 1). This delineation enabled comparison among time periods and between watersheds. This study focused on the differences between the Pre and Post periods. While changes in water quality were evident during the Mid period, this study does not offer conclusions about this period because the effects of disturbance from the construction activities and the effects of wet ponds cannot be differentiated. The Mid period was part of the data record, but was not explicitly analyzed as part of this study.

	Start	**End**	**Tarawa Sample n**	**French Sample n**
Pre	January - 2008	December - 2008	27	94
Mid	December - 2008	March - 2011	-	-
Post	March - 2011	July - 2015	256	234

Table 1. Sampling dates for each period of development. n represents the number of water samples collected for concentration measurements during each period.

A Student's t-test ($\alpha = 0.05$) was performed on the Log_{10}-transformed nutrient, TSS, and chl-*a* data to determine if there were significant differences in any of the water quality variables between Pre and Post development periods for both streams. Nutrient, TSS, and chl-*a* concentration data from the developed stream were parsed into samples collected at baseflow and stormflow, and the same methodology above was used to determine if there were significant differences between time periods for both baseflow and stormflow for each water quality variable.

A linear model was created for each variable measured during the Post period for each stream using the date of sampling as the independent variable and unaltered concentration measurements as the dependent variable. Concentration values were predicted for each variable for each stream using the corresponding linear model for the beginning and end of the Post period. The predicted change in each variable for the reference stream was subtracted from the predicted change in each variable for the developed stream to remove natural trends in concentration data. The reference stream did not experience significant anthropogenic disturbance during this study, so any trends in water quality variables in the reference stream during the Post period were assumed to be trends unrelated to development. These trends in concentration data could hypothetically be driven by changes in precipitation (ex. dilution vs. concentration) or temperature over the course of the Post period. After trends exhibited by the undeveloped watershed were removed, the developed stream's predicted change for each variable was divided by the developed stream's predicted values for the beginning of the Post period and multiplied by 100 to calculate percent relative change.

Finally, a Student's t-test ($\alpha = 0.05$) was performed on Log_{10}-transformed nutrient, TSS, and chl-*a* concentration data to compare baseflow and stormflow concentrations during both the Pre and Post periods. To investigate the role of wet ponds as a source or sink for various water quality variables, a paired Student's t-test ($\alpha = 0.05$) was performed on Log_{10}-transformed nutrient, TSS, and chl-*a* concentration data to compare water quality concentrations at baseflow from the developed stream and a developed watershed wet pond between mid-March 2015 and the end of June 2015.

All statistical analyses were performed in R (version 3.1.2). Maps were created using Environmental Systems Research Institute (ESRI) ArcMap (version 10.2.2). Imagery, elevation data, and SCM data were provided by US Marine Corps Base Camp Lejeune.

4. Results and discussion

4.1 Impacts of wet pond implementation: Comparing Pre and Post periods

Stormwater managers in both coastal and non-coastal areas utilize similar SCMs and overall management goals (Collins et al., 2010), but the coastal plain presents distinct conditions for stormwater management such as flat topography, high water table, proximity to recreational and ecological resources, high cost of land, and complications associated with tidal influences (EPA, 2014). To determine the efficacy of wet ponds in a coastal watershed, I examined changes in stream water quality using long-term data collection in 2 representative coastal plain watersheds, one largely undeveloped and one that was further developed and outfitted with wet ponds during the study.

There were significant changes in the mean concentrations of all variables except NH_4^+ in the developed (Tarawa) watershed's stream between the Pre and Post periods (Figure 3). In the less developed reference (French) watershed's stream, mean chl-*a* and NH_4^+ concentrations both significantly increased between the Pre and Post periods, but the magnitude and percent change in the mean chl-*a* concentration was smaller than in the developed stream and NH_4^+ increased while the developed stream slightly decreased, but not significantly (Figure 3). This multi-year data record indicates that the installation of the SCMs during the construction phase in the developed watershed did not result in water quality on-par with the Pre conditions.

4.1.1. Nitrogen

Human modification of the nitrogen cycle has been extraordinary (Vitousek et al., 1997). In coastal areas, excessive nitrogen loading has led to impairments of many of the world's estuaries (Bricker et al., 2007). In areas where nitrogen loading to estuaries is excessive, any sinks and/or processes that remove nitrate from the system become increasingly important (Brush, 2009). Coastal stream networks have been shown to be significant sinks for nitrogen, reducing the load delivered to estuaries (Thompson et al., 2000). Wet ponds are presumed to be nitrogen sinks and enhance nitrogen removal, but there are few long-term measurements and fewer still in the coastal plain. In nitrogen-sensitive, eutrophic coastal plain ecosystems, sinks for excess nutrients are ecologically and economically valuable (Piehler and Smyth, 2011). In order to determine whether wet ponds are detrimental or beneficial to estuaries in terms of nitrogen processing, I analyzed a record of nitrogen concentrations before and after the installation of stormwater ponds.

In this study, the mean developed stream NH_4^+ concentration did not change significantly between Pre and Post periods (Figure 3). The mean concentration did significantly decrease during baseflow but not during stormflow between Pre and Post periods (Figure 4). The reference stream showed a significant increase in the mean NH_4^+ concentration of .22 $\pm$ 1.05 μM, or 21.84 %, between Pre and Post (Figure 3, Table A.1). The increase of the mean NH_4^+ concentration in the reference stream and decrease in the mean baseflow concentration in the developed stream between Pre and Post indicates that wet ponds or the stream in the developed watershed may have functioned as NH_4^+ sinks (Figure 3, Figure 4). Possible mechanisms for the observed decrease of baseflow NH_4^+ concentration could include the storage of NH_4^+ in pond vegetation (Mallin et al., 2002), uptake by pond phytoplankton (Lewitus et al., 2008), or the transformation of NH_4^+ into NO_x^- via nitrification in the pond or stream (Collins et al., 2010).

The mean NO_x^- concentration increased by 1.97 $\pm$ 4.85 μM in the developed stream, a 51.8 % increase, between Pre and Post periods (Figure 3, Table A.1). The mean concentration of NO_x^- significantly increased during baseflow but not stormflow in the developed stream between Pre and Post (Figure 4). There was no significant increase of the mean NO_x^- concentration in the reference stream (Figure 3). The increased mean baseflow concentration of NO_x^- in the developed stream could be caused by increased impervious and lawn area (Table A.4), which can increase NO_x^- inputs from the atmosphere (Kaushal et al., 2011) and fertilizer (Osmond and Hardy, 2004). The majority of nitrogen export in suburban areas occurs during low flows (Groffman et al., 2004, Shields et al., 2008), indicating that sources of nitrogen within the watershed are exported to the stream by high-frequency, low-intensity storm events that bypass stormwater infrastructure (Groffman et al., 2004). Alternatively, channelization of the stream due to increased runoff or elevated wet pond discharge could disconnect the stream from its floodplain, an important area for NO_x^- removal (Newcomer-Johnson et al., 2014). A third possible mechanism for the increase in the mean baseflow NO_x^- concentration in the developed stream is the conversion of NH_4^+ into NO_x^- via nitrification (Collins et al., 2010) since the mean baseflow NH_4^+ concentration decreased as well. No change in the mean stormflow concentration of NO_x^- in the developed stream indicates that the ponds are not a source of NO_x^- when flushed during storms (Figure 4).

The mean ON concentration in the developed stream increased by 7.15 $\pm$ 10.38 μM, or 57.93 %, between Pre and Post periods (Figure 3, Table A.1). Mean baseflow concentrations and stormflow concentrations significantly increased (Figure 4). No significant change in mean ON concentrations was observed in the reference stream (Figure 3). Wet ponds could be sources of ON during baseflow and when flushed during storm events. Possible mechanisms for this increase could be vegetation and algal biomass supported by ponds.

4.1.2. Phosphorus

Excess concentrations of phosphorus in freshwater, specifically orthophosphate (PO_4^{3-}), can cause eutrophication issues much like those caused by nitrogen in ocean or estuarine waters (Correll, 1998). The New River Estuary has historically experienced eutrophication issues with connections to phosphorus enrichment from sewage treatment plants (Mallin et al., 2005), so keeping phosphorus concentrations low is known to be important for maintaining the health of the estuary. Stormwater ponds are thought to remove phosphorus by enhancing settlement of phosphorus-sorbed suspended sediments (Nairn and Mitsch, 2000) or uptake by vegetation (Kadlec, 2016) and algae (Nairn and Mitsch, 2000). Phosphorus removal is thought to be a major benefit of stormwater ponds, but SCMs have been known to become phosphorus saturated over time (Hunt and Lord, 2006; Merriman and Hunt, 2014) and even become sources of phosphorus during low flows due to anoxic sediments (Duan et al., 2016).

Comparing before and after the implementation of wet ponds, the developed stream mean dissolved PO_4^{3-} concentration decreased by .30 $\pm$ .65 μM, or 32.24% (Figure 3, Table A.1). Mean stormflow and baseflow dissolved PO_4^{3-} concentrations in the developed stream both significantly decreased. There was no significant change of the mean dissolved PO_4^{3-} concentration in the reference stream (Figure 3). The decrease in the developed stream mean dissolved PO_4^{3-} concentration and no change in the reference stream indicates that wet ponds lowered the mean PO_4^{3-} concentration within the stream, especially during storm events (Figure 3, Figure 4). These data show that wet ponds may be effective at reducing mean dissolved PO_4^{3-} concentrations either by sorption to suspended sediments that settle out (Nairn and Mitsch, 2000) or uptake from wetland vegetation (Kadlec, 2016) and algae (Nairn and Mitsch, 2000). However, analysis could have been skewed since sample filtration removed sediment-sorbed phosphorus. This could explain the significantly lower mean concentration of dissolved PO_4^{3-} if more PO_4^{3-} was sorbed to sediments in the Post period than the Pre period. Future research should include measurements of total phosphorus in addition to dissolved phosphorus to determine if wet ponds are actually removing phosphorus or supplying it downstream attached to suspended particles.

4.1.3. Chlorophyll-a

Nutrient management in coastal regions is most often focused on reducing excessive phytoplankton biomass as measured by chlorophyll-*a*. Pristine blackwater coastal streams are generally understood to be sites with low phytoplankton biomass due to naturally low nutrient concentrations and high amounts of dissolved organic material (Meyer, 1992). However, in coastal streams with a developed watershed, increased nutrient loading can create large amounts of algae and negatively impact downstream water quality (Mallin et al., 2004; Wahl et al., 1997).

At our study sites, the mean concentration of chl-*a* increased by 8.23 $\pm$ 14.56 μg/L in the developed stream and .64 $\pm$ 1.93 μg/L in the reference stream, a 349.26 % and 76.35 % increase, respectively (Figure 3, Table A.1). The increase of the mean chl-*a* concentration in the developed stream was approximately thirteen times larger than the increase in the reference stream. The mean chl-*a* concentration in the developed stream significantly increased during both baseflow and stormflow, although the increase in mean concentration was larger during stormflow (Figure 4). The larger increase of the mean chl-*a* concentration in the developed stream relative to the reference stream indicates that the increase in the developed stream was not solely due to environmental conditions. Additionally, the larger increase in mean concentration during stormflow compared to baseflow in the developed stream suggests that there is a flushing of chl-*a* from the watershed during storm events, likely from the wet ponds (Figure 4). Coastal wet ponds have been shown in the past to have high concentrations of algal biomass during certain seasons (DeLorenzo, 2012; Lewitus et al., 2008). As a consequence of design, these ponds appear to provide optimal habitat for algal blooms: sufficient irradiance, low flow velocities, and nutrients that flow into ponds from large areas of the watershed after storm events.

4.1.4. Total suspended solids

Wet ponds are designed to remove suspended solids by slowing down incoming water and allowing suspended solids to settle out of the water column (NCDENR, 2009). Suspended solids, such as sediments and organic matter, reduce water quality by increasing water column light attenuation (Bilotta and Brazier, 2008), changing water temperature (Bilotta and Brazier, 2008), and reducing dissolved oxygen concentrations by adding organic material to the water column and increasing sediment oxygen demand (Waterman et al., 2011).

In the present study, mean TSS concentration increased by 18.99 $\pm$ 41.51 mg/L in the developed stream between Pre and Post periods, which is a 310.65 % increase (Figure 3, Table A.1). After the construction period, the mean TSS concentrations were significantly higher in both baseflow and stormflow in the developed stream (Figure 4). No significant change in the mean TSS concentration was observed in the reference stream between Pre and Post (Figure 3). The main purpose of wet ponds is typically to mitigate altered hydrology from development (Hancock et al. 2010) and capture suspended solids that are eroded from the watershed (NCDENR 2009). It is surprising that the mean TSS concentration in the stream during the Post period was 310% higher than the Pre period (Figure 3, Table A.1). Logically, TSS concentrations will increase while construction is ongoing, but once construction ceased, the wet ponds in this study did not maintain or reduce the mean TSS concentration downstream relative to the Pre period mean concentration. This phenomenon has been documented in the Piedmont of North Carolina by Tillinghast et al. in 2011. They showed that lowering the peak flow from storm events using ponds can increase the amount of time that an SCM's discharge exceeds a level that erodes downstream stream channels. An alternative hypothesis is that the wet ponds were actually sources of TSS due to sediment resuspension within the pond.

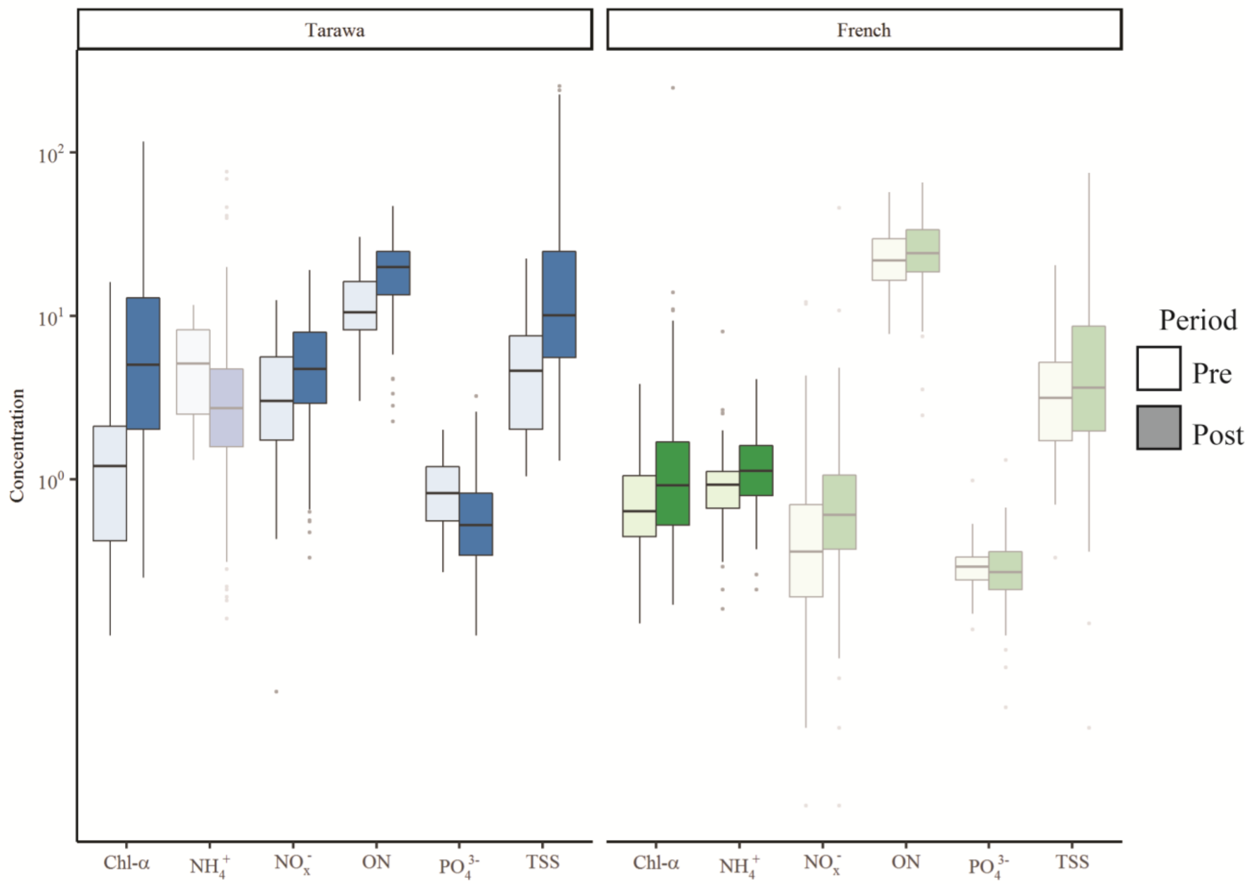

Figure 3. Nutrient, total suspended solid, and chlorophyll-*a* concentrations for the Pre and Post periods of development. Full color boxplots indicate water quality variables that changed significantly between Pre and Post periods based on Student's t-tests ($\alpha = .05$).

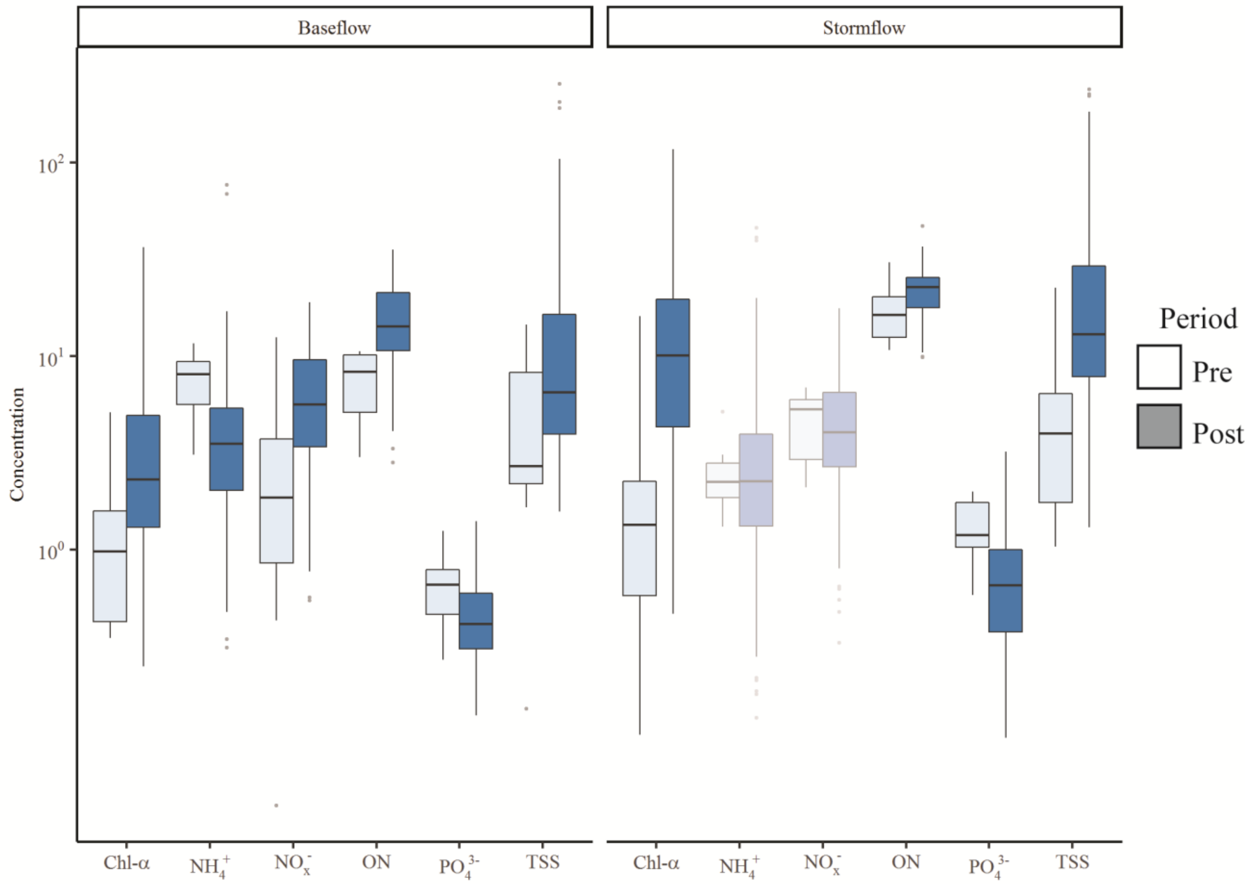

Figure 4. Nutrient, total suspended solid, and chlorophyll-*a* concentrations from the developed stream for the Pre and Post periods of development for baseflow and stormflow water samples. Full color boxplots indicate water quality variables that changed significantly between Pre and Post periods based on Student's t-tests ($\alpha = .05$).

4.2. Trends in stream water quality after wet pond implementation

Comparing the relative change between the beginning and end of the Post period, concentrations in the developed stream decreased relative to the reference stream for chl-*a*, NH_4^+, and ON and increased relative to the reference stream for NO_x^-, PO_4^{3-}, and TSS (Figure 5, Table 2). During this 3 year period, chl-*a* decreased by 14.48 %, NH_4^+ decreased by 48.76 %, and ON decreased by 1.71 % relative to the reference stream (Figure 5, Table 2). Concentrations of NO_x^- increased by 158.23 %, PO_4^{3-} increased by 5.23 %, and TSS increased by 590.08 % (Figure 5, Table 2). Predicted stream water chl-*a* concentrations decreased slightly through the Post period, which may be explained by an increase in pond vegetation cover over time. An increase in vegetation cover within the ponds over time could compete with algae for nutrients and light within the ponds, possibly also explaining the decrease in NH_4^+ concentrations in the stream over time. The increase in NO_x^- concentrations predicted by the linear regression indicates the wet ponds became less effective at removing NO_x^- as time went on, or channel incision and erosion decreased the stream's ability to remove NO_x^- by disconnecting the stream from its floodplain (Newcomer-Johnson et al., 2014). There was no clear trend in ON concentrations between the beginning and end of the Post period. Predicted concentrations of PO_4^{3-} increased slightly through the Post period, which could mean that the sediments in the pond became saturated with PO_4^{3-} within a few years and lowered the pond's ability to remove PO_4^{3-} (Hunt and Lord, 2006; Merriman and Hunt, 2014), or the dissolved oxygen concentrations within the pond decreased over time and allowed particle-bound phosphorus to be released (Duan et al., 2016). Additionally, predicted TSS concentrations increased almost 6-fold during the Post period, indicating that the ponds were removing less TSS over time, having sediments become resuspended within the pond and exported, or scouring material from the streambed.

Considered together, these results indicate that wet ponds in the developed watershed became less effective at removing nutrients and TSS over time or negatively impacted the ability of the stream to remove nutrients and TSS. Alternatively, sources of nutrients and TSS could have increased throughout the Post period. To maximize NO_x^-, PO_4^{3-}, and TSS removal within the wet ponds, this study suggests that wet ponds in coastal areas undergo more frequent excavation. This is in line with the recommendations for wet ponds in the Piedmont of North Carolina and elsewhere that call for sediment excavation every few years to preserve water storage capacity and sediment, nitrogen, and phosphorus removal (Duan et al., 2016; Hunt and Lord, 2006; Sønderup et al., 2016). While stream water concentrations of chl-*a*, NH_4^+, and ON decreased over the Post period, the increases in various water quality concentrations were much larger, percentage-wise, than the reductions (Table 2).

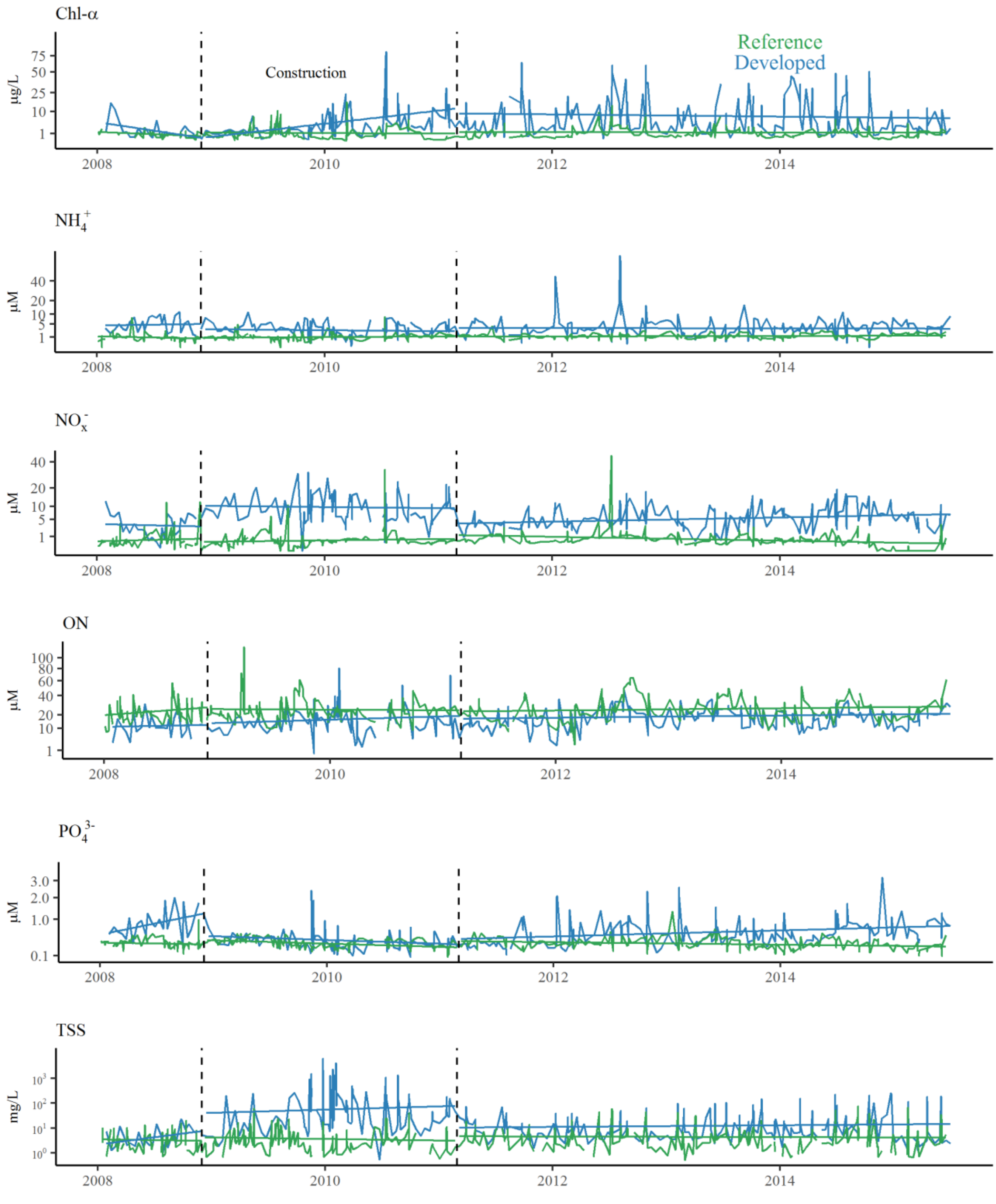

Figure 5. Time series of nutrient, TSS, and chl-*a* concentrations for both reference (green) and developed (blue) streams with linear fits for each period of construction. Area between the dotted lines indicates the construction (Mid) period. All y-axes use a square root scale, except TSS which uses a $\log_{10}$ scale.

Natural trends removed	**TSS** (mg/L)	**NO_x^-** (µM)	**NH_4^+** (µM)	**PO_4^{3-}** (µM)	**ON** (µM)	**Chl-a** (µg/L)
Change	32.35	4.22	-2.66	0.42	-0.31	-1.72
% change	590.08	158.23	-48.76	5.23	-1.71	-14.48
Slope (conc/yr)	7.56	.99	-.62	.10	-.07	-.40
Slope (perc/yr)	137.89	36.98	-11.39	1.22	-.40	-3.38

Table 2. Relative change, percent relative change, and the relative slope of nutrient, TSS, and chl-*a* concentrations at the beginning and end of the Post-Construction period.

4.3. Stormwater wet ponds as a source of algae and sediments and a sink for NO_x^-

Concentrations of water quality variables during baseflow and stormflow were compared for both Pre and Post periods in the developed stream. During the Pre period, concentrations of all water quality variables, except for chl-*a* and TSS, were significantly different during baseflow and stormflow conditions (Figure 6). During the Post period, chl-*a* and TSS concentrations became significantly different during baseflow and stormflow, and in both cases had higher stormflow concentrations than baseflow (Figure 6). NO_x^- concentrations during the Pre period were lower during baseflow than stormflow, but flipped during the Post period to have lower NO_x^- concentrations during stormflow conditions (Figure 6). Additionally, mean concentrations of chl-*a* and TSS were significantly higher and the mean NO_x^- concentration was significantly lower in the wet pond than in the developed stream during baseflow over the sampling period (Figure 7). These data indicate that wet ponds in the watershed were likely sources of both chl-*a* and TSS to the stream and sinks for NO_x^-. The variation in each parameter, except chl-*a*, was higher in the wet pond than in the developed stream (Table A.3). The extremely low concentrations of pond NO_x^- seem to contrast the fact that NO_x^- concentrations significantly increased in the stream after the implementation of wet ponds (Figure 3). Based on this observation, and the fact that almost all of the developed watershed drains to a wet pond, NO_x^- may be effectively removed by the ponds, but NO_x^- within the watershed may be infiltrating to groundwater during small storm events and be released to the stream during baseflow. It is also important to note that this comparison between a wet pond and the stream took place between March and the end of June, so it did not capture variability throughout all seasons. All other nutrients in the pond had mean concentrations higher than the stream, but the differences were not significant due to higher variability in nutrient concentrations within the pond. The negative ecological effects of increased chl-*a* and TSS concentrations within coastal wet ponds should be considered in management decisions.

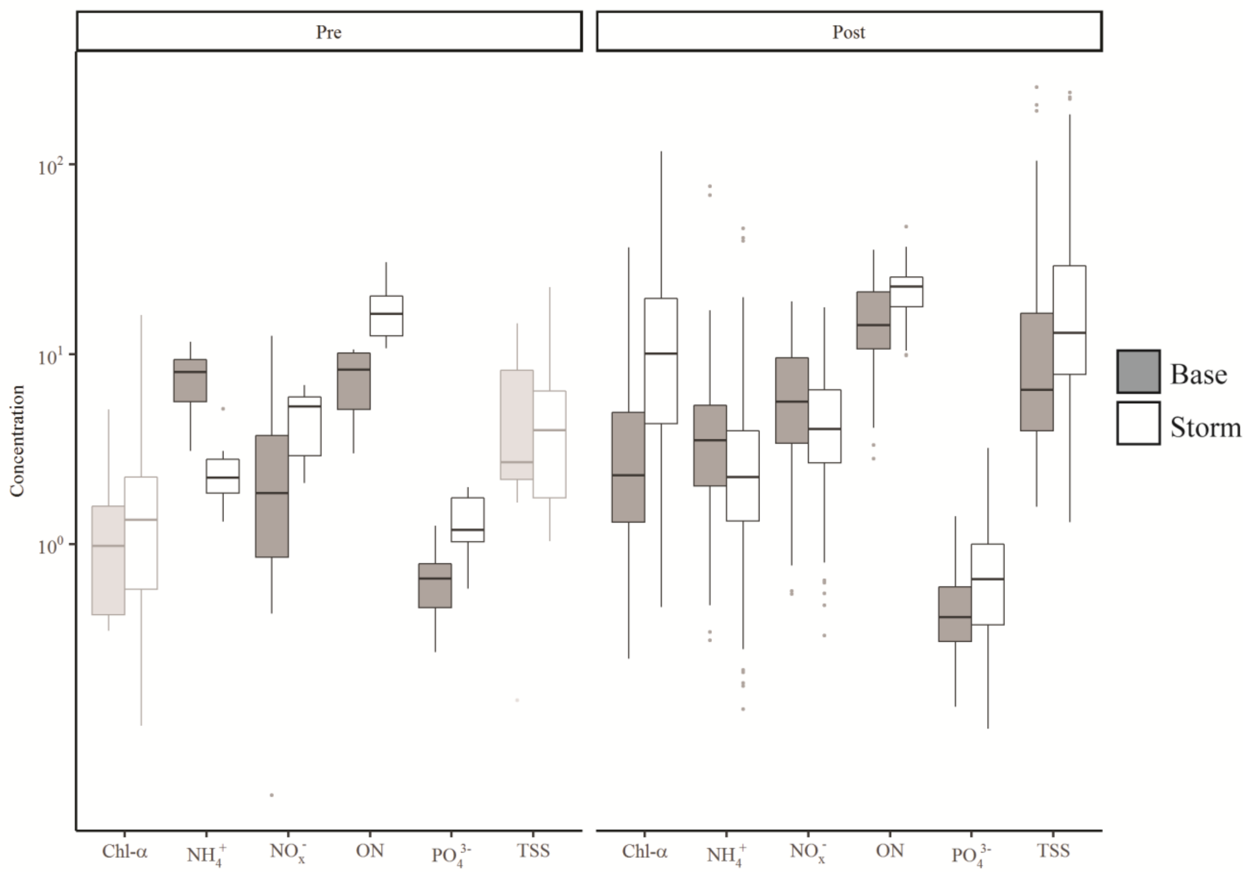

Figure 6. Developed stream concentrations of nutrient, TSS, and chl-*a* during baseflow and stormflow conditions for the Pre and Post Periods. Full color boxplots indicate a significant difference determined by a Student's t-test (α = .05). n = 15 for baseflow and 12 for stormflow during the Pre Period, and n = 114 baseflow and 140 for stormflow during the Post period.

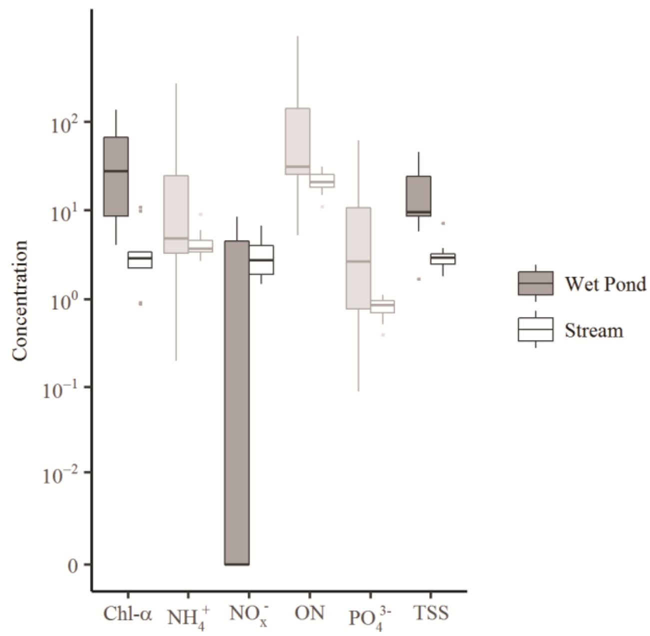

Figure 7. Developed stream and developed wet pond concentrations of nutrient, TSS, and chl-*a*. Samples for both sites were taken within 15 minutes of each other. Full color boxplots indicate a significant difference determined by a paired Student's t-test ($\alpha = .05$). n = 9 for each location.

4.4. Implications for stormwater management in the coastal southeastern US

Conventional stormwater management has focused on narrow management goals (Burns et al., 2012) and relied on large, centralized SCMs, such as wet ponds, that collect water from large areas of the landscape (Collins et al., 2010). While most centralized SCMs are made with the primary goal of mitigating the negative hydrologic effects of development, the results from this study show that this typical method of stormwater management in coastal areas may have some negative effects on downstream water quality.

Wet ponds may not be the best choice for stormwater management in the coastal southeastern US. Overall, the installation of wet ponds that drained 97% of the watershed area was unable to mitigate the negative effects of increased development. This is illustrated by the findings of this study that a wet pond was likely a source for TSS and chl-*a* between spring and summer and that water quality generally decreased further after watershed-scale wet pond implementation with increased development. Undeveloped watersheds on the coast of the southeastern US are drained by blackwater streams (Meyer 1990), but extensive impervious area that accompanies development does not allow precipitation to infiltrate into soils and undergo natural soil biogeochemical processes that supply streams with water rich in dissolved organic matter and low in suspended sediments (Piehler et al., in prep). Rather, the water from developed coastal watersheds have less dissolved organic matter with complex molecular composition (Hosen et al., 2014), more broken-down, bioavailable dissolved organic matter (Hosen et al., 2014), more nutrients (Wahl et al., 1997), and more chl-*a* (Figure 3, Piehler et al., in prep) than natural watersheds.   The installation of wet ponds in the developed watershed did not mitigate many of these negative effects of development, but rather increased them.

Managing stormwater with low-impact development (LID) structures may help restore watershed biogeochemistry and stream water quality by restoring pre-development flow regimes, decreasing surface runoff, and increasing both evapotranspiration and infiltration (Burns et al., 2012; Walsh et al., 2016). Restoring flow paths and biogeochemistry is an optimal approach for improving the water quality of developed coastal watersheds due to importance of dissolved organic matter in streams (Meyer 1990). LID may be more practical than wet ponds in settings represented by the study watershed due to the large amount of open and low-intensity developed area in the watershed (Table A.4) that could support LID infrastructure but not additional wet ponds. Additionally, the higher cost of land and higher water table in coastal areas could make LID more tenable than wet ponds and other large, deep SCMs (EPA, 2014). LID has improved stormwater quality (Dietz, 2007; Dietz and Clausen, 2008; EPA, 2014) and quantity (Jarden et al., 2016) in urban or suburban watersheds and could possibly minimize the negative water quality impacts from wet ponds found in this study by decreasing open water area that can promote algae and sediment resuspension.

Concentrations of NO_x^- , PO_4^{3-}, and TSS increased in the developed stream relative to the reference stream during the period after wet pond implementation, and chl-*a*, NH_4^+, and ON decreased. These changes in stream water quality indicate that the function of wet ponds in the study watershed changed after they were implemented. If other types of SCMs cannot be implemented to replace wet ponds, this study recommends frequent pond excavation to maintain the effective removal of various water quality variables. However, this recommendation may be untenable for some communities due to the high price of maintenance for wet ponds, which are among the most expensive types of SCM to maintain appropriately (Houle et al., 2013). If maintenance cost is not an issue, stormwater wet pond retrofits, such as the implementation of floating wetland vegetation (Tanner and Headley, 2011; Winston et al., 2013), could also be implemented to improve nutrient and suspended sediment removal within a wet pond.

5. Conclusions

After a period of increased development and watershed-scale implementation of stormwater wet ponds in a developed watershed, stream water quality significantly changed and decreased overall. Mean concentrations of chl-*a*, NO_x^-, organic nitrogen, total nitrogen, and TSS in the developed stream significantly increased, while the mean PO_4^{3-} concentration decreased, and the mean concentration of NH_4^+ did not change. Over a three year period after wet pond implementation, the stream water concentrations of NO_x^-, PO_4^{3-} , and TSS increased over time compared to the reference stream, indicating a reduction in pollutant removal efficiency for wet ponds, a negative impact on pollutant removal processes in the stream, or an increase in pollutant sources to the stream throughout the Post period. Concentrations of chl-*a*, NH_4^+, and ON in the developed stream decreased over time after wet pond implementation, but the decreases were much smaller compared to increases of other water quality variables. Comparing baseflow and stormflow water quality concentrations from the developed stream during the Pre and Post period as well as a wet pond within the developed watershed to the developed stream during a single spring and summer showed that the wet ponds were likely functioning as sources of chl-*a* and TSS to the stream and sinks for NO_x^-.

This study demonstrates that the watershed-scale implementation of stormwater wet ponds may not be optimal for nutrient, TSS, and chl-*a* removal in coastal areas within the southeastern US. Distributed stormwater management, such as LID, may be a better method than wet ponds for mitigating the negative effects of development on coastal water quality, but further study of both traditional and LID stormwater structures at the watershed-scale is needed in coastal areas of the southeastern US. In areas where distributed systems cannot be used our finding indicate that both stormwater pond retrofits and frequent pond excavation to maximize removal efficiency may improve nutrient removal performance.

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CHAPTER 2: NITROGEN DYNAMICS IN COASTAL STORMWATER WET PONDS: SEDIMENT NITROGEN FIXATION AND A NITROGEN-CHLOROPHYLL-*A* TRADEOFF DURING THE SUMMER

1. Introduction

The Coastal Plain of the southeastern United States is expected to undergo a steep increase in urbanization over the next 50 years with urban area nearly doubling by 2060 [*Terando et al.*, 2014]. With the expansion of urban area and the resulting increase in impervious surface area comes the need to manage stormwater created by impervious surfaces to protect coastal water quality, human health, and the ecosystem services provided by coastal waters. Specifically, elevated concentrations of nitrogen in stormwater can have negative impacts on coastal waters where nitrogen is typically the limiting nutrient for algal growth [*Howarth and Marino*, 2006]. Runoff containing pollutants has immediate impacts on sensitive and important coastal waters [*Sanger et al.*, 2013], but effective stormwater management should, ideally, reduce these impacts.

Stormwater ponds are the most common type of stormwater control measure (SCM) in the US [*Collins et al.*, 2010], and make up the majority (~ 60%) of SCMs in coastal NC counties [*NCDEQ*, 2017]. Wet ponds, one of the two types of stormwater ponds, collect water from a landscape and retains a permanent pool of water [*Collins et al.*, 2010] with the main purpose of extending the storm hydrograph, reducing peak flows downstream, and reducing TSS loads [*NCDENR*, 2009; *Hancock et al.*, 2010]. Wet ponds are widely acknowledged as a less effective means of reducing nitrogen loads from stormwater than other kinds of SCMs [*Collins et al.*, 2010], and wet ponds can have negative impacts on water quality, including increasing concentrations of algae [*Lewitus et al.*, 2008; *DeLorenzo et al.*, 2012; *Song et al.*, 2015; *Gold et al.*, in prep] and harmful bacteria [*DeLorenzo et al.*, 2012]. Their ability to protect streams from erosion and remove other kinds of nutrients is highly variable and lower than other types of SCMs [*Gold et al.*, in prep; *Collins et al.*, 2010; *Hancock et al.*, 2010; *Tillinghast et al.*, 2011; *Houle et al.*, 2013; *Koch et al.*, 2014].

Denitrification, the microbially-mediated transformation of nitrate into inert N_2 gas and the concurrent oxidation of organic matter [*Seitzinger et al.*, 2006], is thought to be an important nitrogen removal process in SCMs [*Groffman et al.*, 2004, 2009; *Zhu et al.*, 2004; *Collins et al.*, 2010; *Bettez and Groffman*, 2012]. Nitrogen fixation, the transformation of N_2 gas into bioavailable nitrogen [*Howarth et al.*, 1988], can be an important part of the sediment nitrogen cycle in low-nutrient water bodies [*Scott et al.*, 2008; *Newell et al.*, 2016b] or water bodies that are nitrogen limited due to large amounts of phosphorus relative to nitrogen [*Howarth et al.*, 1988]. Previous studies on SCMs in coastal areas have used methods that do not measure these processes or use acetylene reduction assays that significantly alter the microbial community [*Fulweiler et al.*, 2015]. Most studies have calculated concentration changes between the inflow and outflow of SCMs, but very few studies have gone beyond the "black box" approach to directly quantify the processes of nitrogen removal and fixation within SCMs [ex. *Scott et al.*, 2008]. No studies, to our knowledge, have measured net N_2 gas flux in coastal SCMs using intact core incubations, but this methodology is important for gaining a clear understanding of sediment nitrogen cycling. An approach that investigates nitrogen dynamics within SCMs is necessary to better understand their impacts on water quality and inform management decisions [*Collins et al.*, 2010], especially in nitrogen sensitive coastal areas.

Pond and SCM maintenance have been suggested as necessary to maintain pollutant removal over time. As they age, ponds fill in with sediments and vegetation, thus decreasing sediment and phosphorus retention in the pond [*Hunt and Lord*, 2006; *Merriman and Hunt*, 2014; *Sønderup et al.*, 2016]. Because of this in-filling, ponds are routinely excavated to maintain water storage volume and suspended sediment and phosphorus removal [*Merriman and Hunt*, 2014; *Duan et al.*, 2016; *Merriman et al.*, 2016]. While removal of suspended sediments and phosphorus are important for maintaining good water quality and reducing peak flows in some areas, these established practices of wet pond management may not adequately address removal of nitrogen or the effects of unique topography, soil type, and hydrology of coastal areas. Additionally, no studies have investigated intra-pond variation in nitrogen removal processes in coastal SCMs with different vegetation, water level (ex. fringing marsh, shallow water, deep water), and flow path position. Vegetation can increase permanent nitrogen removal by oxygenating soils [*Kreiling et al.*, 2011], thereby fueling coupled nitrification-denitrification in nutrient and organic-rich sediments, but this process remains unstudied in coastal wet ponds. SCM management would be enhanced by better understanding nitrogen processing in wet ponds of different ages and spatial variability within wet ponds. Such knowledge could help maximize nitrogen removal and minimize maintenance costs.

This study directly measured summer net N_2 fluxes from the sediments of five coastal wet ponds of different ages during ambient conditions and during nitrate-enriched conditions similar to those found during stormflow in the pond. This study also measured net N_2 fluxes from sediments in different locations within a single wet pond to determine the effects of depth and vegetation cover on nitrogen cycling. Additionally, a six month time-series of nutrient, suspended sediment, and chlorophyll-*a* concentrations at the inlets and outlet of a single coastal wet pond were monitored to determine removal efficiencies for each of these water quality variables.

The objectives of this study were to:

1. Quantify rates of net N_2 flux from ponds spanning a range of ages.
2. Quantify rates of net N_2 flux in different areas of a wet pond.
3. Determine concentration-based nutrient and suspended sediment removal efficiencies of a coastal stormwater wet pond.
4. Discuss implications for coastal wet pond management and design.

2. Methods

2.1. Site Description

All data were collected on Marine Corps Base Camp Lejeune in Jacksonville, NC between June 2016 and January 2017. ArcGIS was used to select five stormwater wet ponds spanning a range of ages from 3.25 to 10 years with similar land use and soil types (Figure 1). All ponds sampled are located in an on-base residential neighborhood, Tarawa Terrace. A middle-aged pond (6.16 years old, Figure 2) was selected for more intensive study, comparing measurements from the forebay and main pond and different locations within each section (Figure 2).

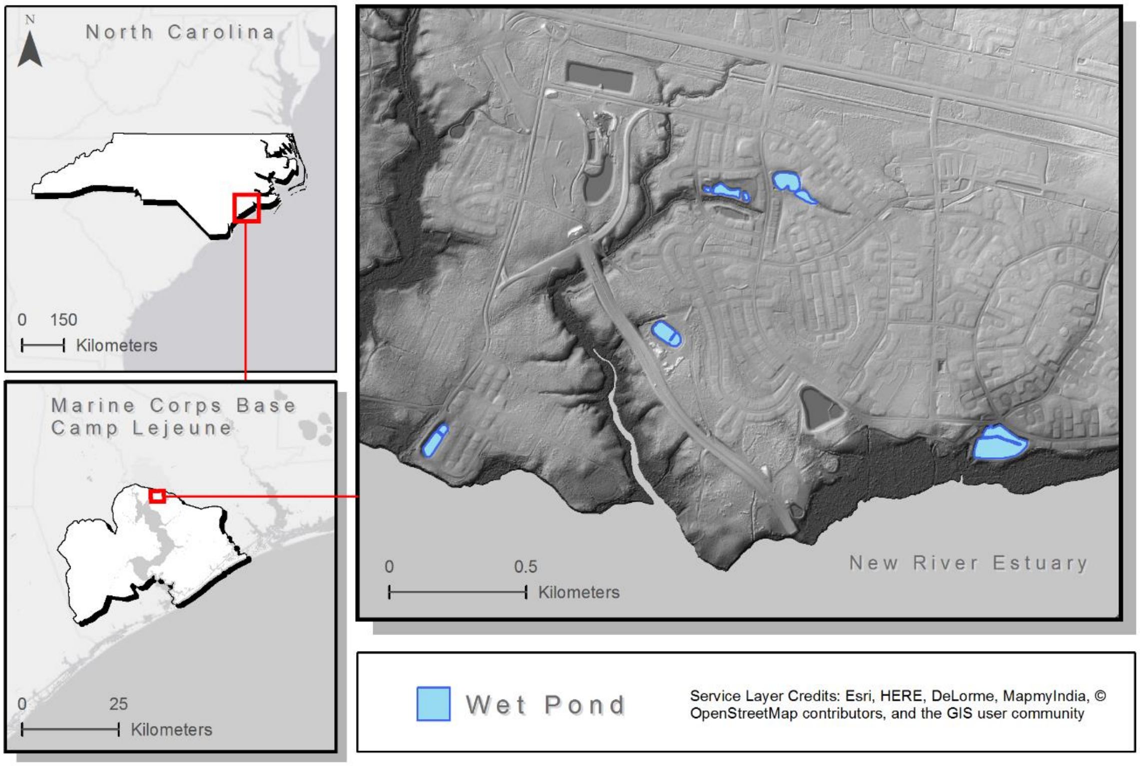

Figure 1. The five stormwater wet ponds sampled, located on Marine Corps Base Camp Lejeune near Jacksonville, NC.

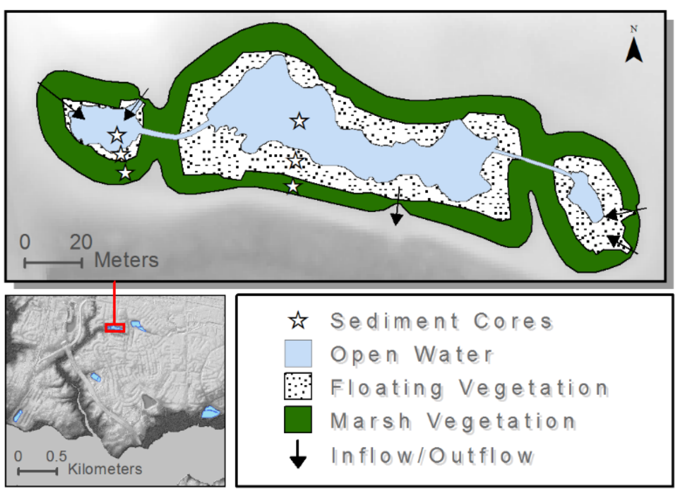

Figure 2. Sample stormwater pond with different vegetation types, location of sediment cores, and location of study site. Sample transects from the forebay (left transect) and main pond (right transect) were compared.

2.2. Five pond flux experiment

2.2.1. Pond conditions

Pond ages were determined from a base-wide SCM shapefile and were rounded to the nearest month. To calculate the percent cover of vegetation for each sample pond, aerial imagery from November 2015 was used to manually delineate marsh vegetation, floating vegetation, and open water area. Sediment cores collected from the deep, main parts of each pond were analyzed for percent organic matter by loss-on-ignition. Surface water samples from the main pools of the ponds were analyzed for nutrient concentrations (NO_3^--N, NH_4^+, PO_4^{3-}, total nitrogen, and organic nitrogen) using a Lachat nutrient auto-analyzer, total suspended solids concentrations by weighing the amount of particulates on a glass fiber filter after filtering a known amount of water, and chlorophyll-*a* concentrations by analyzing sonicated water samples with a fluorometer.

2.2.2. Gas and nutrient fluxes from the sediment-water interface

To measure net sediment N_2 fluxes in ponds of different ages, three replicate sediment cores were collected using an extended sediment corer deployed from a canoe in the deep pools of five different wet ponds in the Tarawa Terrace neighborhood in late June 2016 (Figure 1). Sediment cores with overlying site water were transported immediately to the UNC Institute of Marine Sciences in Morehead City, NC and allowed to equilibrate for 24 hours in site-specific pond water. Sediment cores were capped, excluding air bubbles, and connected to a flow-through system. The flow-through system moved site-specific pond water into the top of the overlying water of sediment cores and out through a tube approximately 2 cm above the sediment surface. Water samples from each core outflow were analyzed for N_2:Ar with a membrane inlet mass spectrometer, and net N_2 fluxes were calculated for each core by difference from dissolved gases in inflow water samples. Following initial incubation, nitrate was added to the feed water, raising it to a concentration of 30 µM NO_x^--N that is similar to NO_x concentrations measured in ponds during storm events (Piehler Lab unpublished data). The incubation equilibrated for 12 hours before they were sampled again for analysis of N_2 fluxes. Water samples from the feed water were collected and analyzed for nutrient concentrations at the same time as water samples were collected and analyzed for nitrate enriched net N_2 fluxes.

The same gas flux methodology was used to measure sediment oxygen demand (SOD) (O_2:Ar), the flux of oxygen into the sediment. SOD was divided by the dissolved oxygen concentration of the water sample from each core to calculate normalized SOD, or the flux of oxygen into the sediment normalized by dissolved oxygen available in the water column. This measurement was used to correct for variation in ambient inflow oxygen concentrations.

Finally, sediment nutrient fluxes were measured using the same flow-through system methodology. Concentrations of NO_3^--N, NH_4^+, PO_4^{3-}, and organic nitrogen were measured using a Lachat Quick-Chem 8000 Nutrient Auto-Analyzer.

2.3. Single pond flux experiment

2.3.1. Pond depth profiles

Measurements of temperature, dissolved oxygen, and turbidity were collected from surface and bottom water from both the forebay and main pond using a YSI 6600EDS-S water quality sonde. Measurements were taken at the same time as pond sediment core extraction, approximately noon in August 2016. Relative thermal resistance to mixing (RTRM) was calculated using temperature measurements from the top and bottom of the water column to determine if the ponds were stratified at the time of sampling (Equation 1, *Wetzel*, 2001).

$$\textbf{(1)}\ \boldsymbol{\varphi} = \frac{(\rho_{z2}-\rho_{z1})}{(\rho_4-\rho_5)}$$

In equation 1, φ = RTRM (dimensionless), ρ indicates water density (kg/m^3) at the bottom of the pond (z2), the top of the pond (z1), at 4 ˚C, and 5 ˚C. Water density was calculated from temperature measurements using an equation derived from reference table values from *Hornberger et al.* [1998] (Equation 2).

$$\textbf{(2)}\ \rho = -.006t^2 + .0383t + 999.92$$

In equation 2, ρ = density (kg/m^3) and t = temperature (˚C). Keeping with previous studies in stormwater ponds and shallow water bodies, a RTRM greater than 50 indicated pond stratification, and a RTRM less than 50 indicated mixed conditions [*Chimney et al.*, 2006; *Song et al.*, 2013].

2.3.2. Gas fluxes from the sediment-water interface

The flow-through core incubation method described above for measuring net sediment N_2 fluxes, excluding the nitrate addition, was repeated later in the summer (early August 2016) on cores from the middle-aged wet pond (6.16 years old, Figure 2). Three replicate cores from six different sites within the pond were extracted and analyzed to determine any relationship between net N_2 fluxes and depth and vegetation type (Marsh, shallow with floating vegetation, and deep) within the pond's forebay and main pond.

Again, the flow-through core incubation method was used to measure SOD of sediment cores from the single pond. SOD was not normalized by dissolved oxygen concentration because all samples were collected from the same pond.

2.4. Single pond monitoring

To monitor concentration-based removal efficiencies for nutrients, TSS, and chlorophyll-*a*, the 6.16 year old pond (Figure 2) was outfitted with Teledyne Isco automatic water samplers in each of the two forebays and the single outlet from the main pond. Each forebay had two inlet pipes, one much larger than the other. One ISCO in each forebay measured discharge from the larger inlet pipes. A ratio between the largest inlet pipes from both forebays was calculated based on the pipe diameter, and this ratio was used to weight and combine the two concentrations into one "inflow" concentration.

Water samples were collected once every two weeks at the inlets and outlet of the pond from July to the end of December 2016. Additional water samples were collected by the Isco auto-samplers to span the rising, peak, and falling limbs of the hydrograph during five storm events. Water samples were analyzed for nutrient (NO_3^--N, NH_4^+, PO_4^{3-}, total nitrogen, and organic nitrogen), total suspended solids, and chlorophyll-*a* concentrations. Concentration data were used to calculate the removal efficiency of each water quality variable for each sampling date under the assumption that the wet pond's stormwater input was approximately equal to the outflow. Wet ponds are not designed to reduce stormwater volume; this assumption is consistent with current NC regulations and wet pond design [*NCDENR*, 2009].

Using the observed change in chlorophyll-*a* (chl-*a*) concentrations through the pond, a chl-*a*:carbon ratio [*Cloern et al.*, 1995], and Redfield's carbon:nitrogen ratio [*Redfield*, 1934, 1958] was used to calculate a theoretical amount of nitrogen assimilated or remineralized by the observed change in chl-*a*. This theoretical amount of nitrogen was deducted from the observed change in dissolved inorganic nitrogen (DIN, NO_x + NH_4) concentrations, and the resulting value was called "change in autochthonous DIN". A positive change in autochthonous DIN signifies an increase in DIN from within the pond that is not related to algal decomposition (ex. remineralization of carbon in sediments, nitrogen fixation), and a negative change in autochthonous DIN signifies a removal of DIN that is not related to algal uptake (ex. denitrification, DNRA, anammox, macrophyte uptake).

3. Results

3.1. Five pond flux experiment

3.1.1. Pond conditions

All ponds were fringed with marsh vegetation, mainly cattails (*Typha* spp.). Floating vegetation in each pond consisted of alligator weed (*Alternanthera philoxeroides*), which was established at the permanent pond surface and reached into the open water. The percentage of the pond surface covered by vegetation exhibited a positive relationship with pond age (Figure 3A, $p < .2$, $r^2 = .5$). Sediment organic matter increased with pond age, but the relationship was not significant (Figure 3B, $p < .22$, $r^2 = .45$). Total nitrogen concentrations exhibited a strong and significant ($\alpha = .05$) negative correlation with pond age (Figure 3C, $p < .02$, $r^2 = .91$), and dissolved inorganic nitrogen:phosphorus (DIN:P) ratios decreased with age (Figure 3D, $p < .12$, $r^2 = .61$). All ponds had a DIN:P ratio of less than 5, indicating that they were nitrogen limited, assuming that nitrogen limitation consists of a DIN:P < 16.

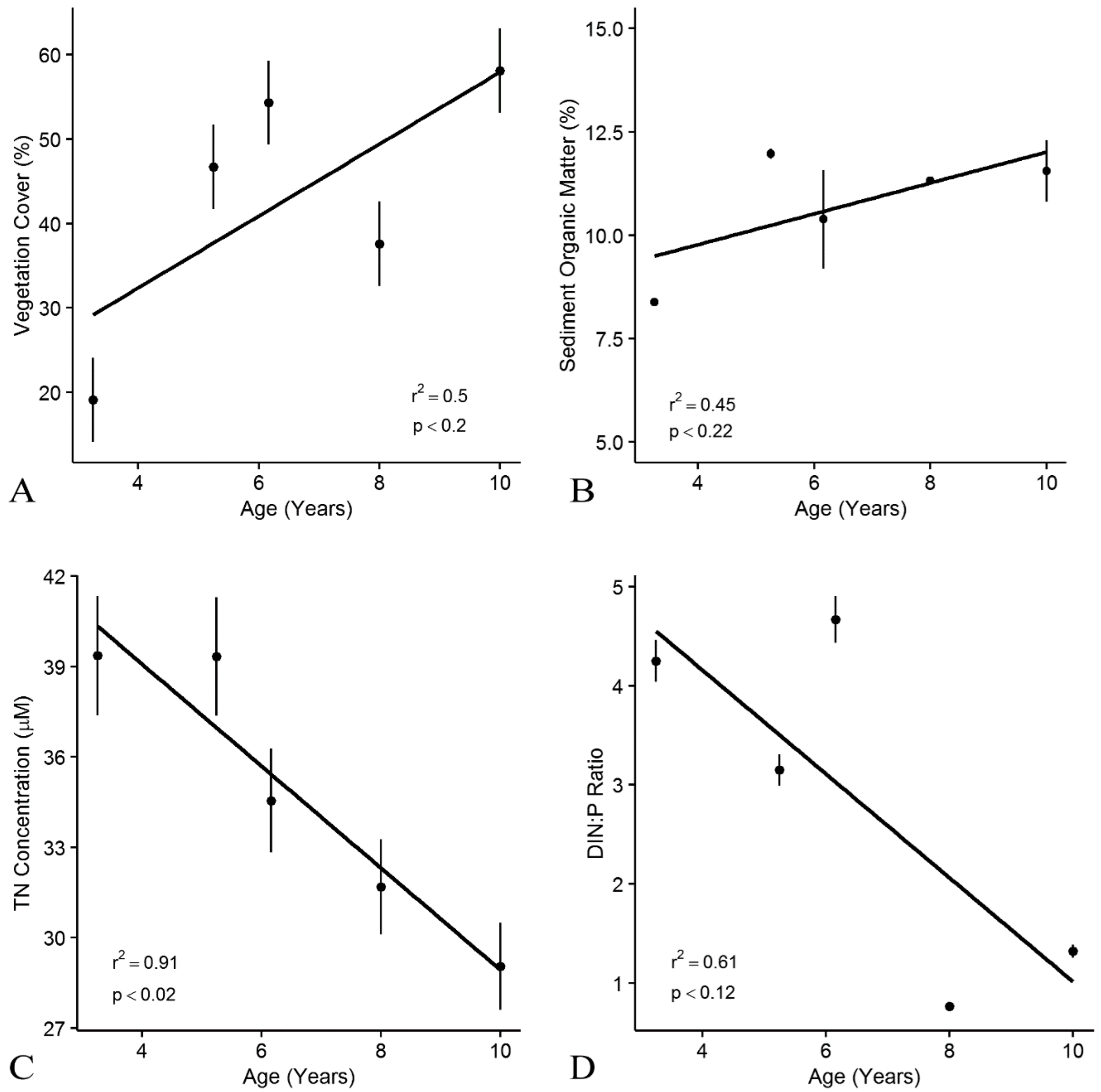

Figure 3. a) Pond vegetation cover as a function of age **b)** Pond sediment organic matter as a function of age **c)** Concentrations of total nitrogen (TN) significantly correlated with pond age **d)** Pond DIN:P ratio as a function of age. Mean ± SE.

3.1.2. Gas and nutrient fluxes from the sediment-water interface

Net N_2 fluxes during ambient conditions exhibited a weak negative relationship with age (Figure 4, $p < .65$, $r^2 = .08$). During nitrate-enriched conditions, net N_2 fluxes exhibited a strong and significant negative relationship with pond age (Figure 4, $p < .05$, $r^2 = .85$). During ambient conditions, nitrate concentrations were below detection limit (.051 µM). In the nitrate-enriched conditions, the fuel for denitrification was introduced, and the net N_2 flux of younger ponds increased while the net N_2 flux of older ponds decreased. There were slight decreases in oxygen concentration within the cores (< 1 mg/L) between ambient and nitrate-enriched net N_2 flux measurements.

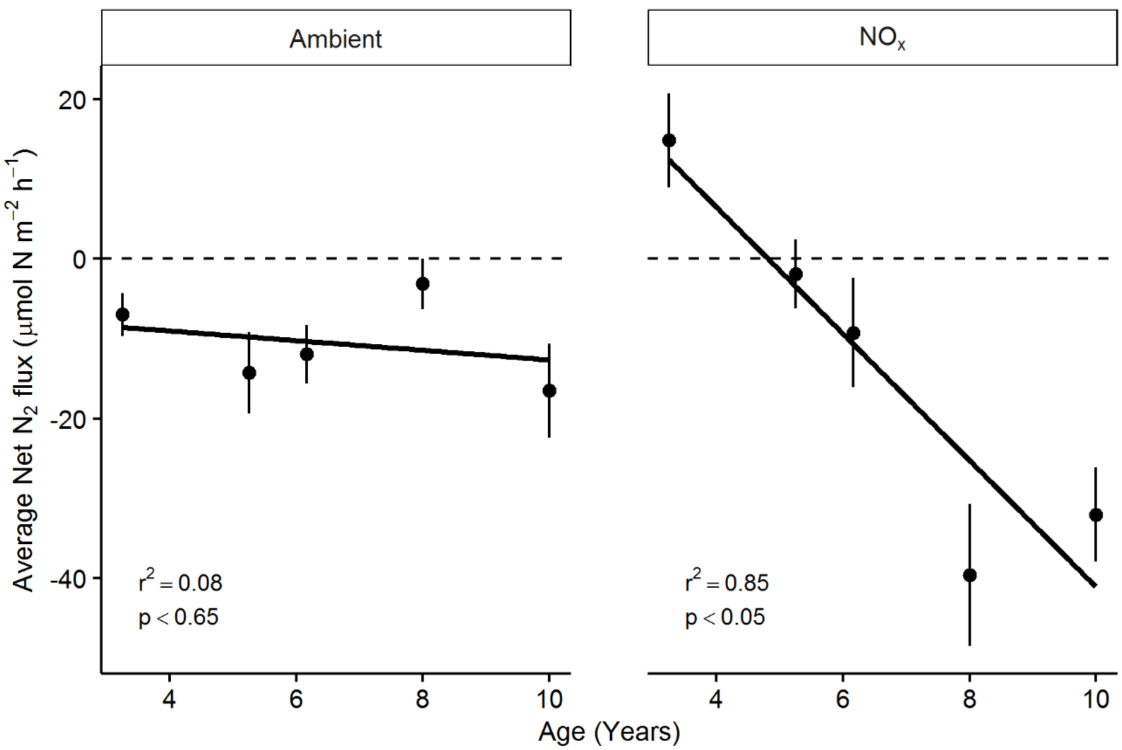

Figure 4. Net sediment N_2 fluxes as a function of pond age for both ambient and nitrate-enriched conditions. Mean $\pm$ SE.

Due to the experimental set up, the amount of nitrate in the inflow of each pond's cores varied based on the amount of nitrate taken up over the 12 hour period between nutrient enrichment and sampling (Figure 5). Ponds that had higher rates of nitrate removal within their inflow water had net N_2 fluxes that became more negative between ambient and nitrate-enriched conditions (Figure 5, $p < .001$, $r^2 = .99$). Older ponds generally had increasingly negative net N_2 fluxes after nutrient enrichment compared to younger ponds (Figure 5). Due to nitrate uptake that occurred in the feed water, cores were subjected to different concentrations of nitrate at the time of sampling. This added a realistic insight into the effect of nitrogen processing in the water column on sediment processing that varied with pond age.

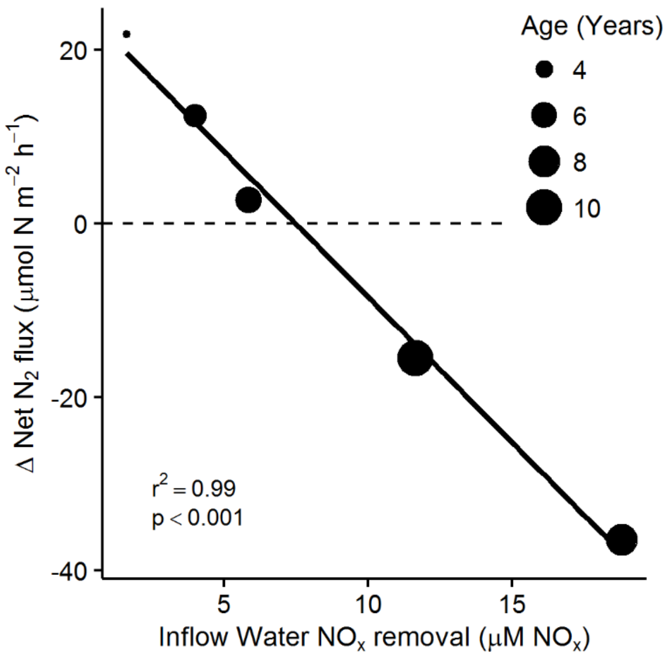

Figure 5. The change in net N_2 flux between ambient and nitrate-enriched conditions as a function of the change in inflow water NO_x concentrations during the 12 hours between enrichment and sampling.

Normalized SOD increased with age (Figure 6, Ambient: $p < .01$, $r^2 = .98$, NO_x: $p < .02$, $r^2 = .92$), indicating an increase in microbial activity as ponds age. Net N_2 flux and normalized SOD had a weak, negative relationship that was not statistically significant (Ambient: $p < .65$, $r^2 = .09$, NO_x: $p < .15$, $r^2 = .6$). SOD is a proxy for organic matter oxidation by the microbial community [*Eyre et al.*, 2013], so the results imply that ponds have more microbial activity as they age.

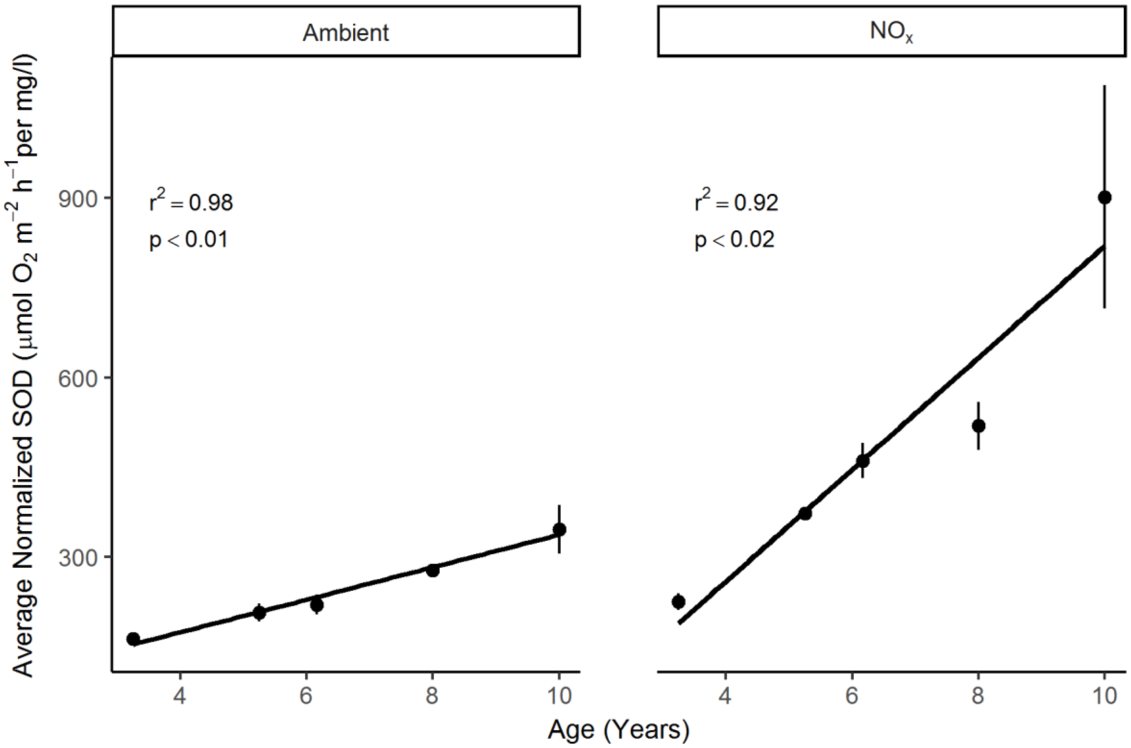

Figure 6. The relationship between normalized SOD and pond age for ambient and nitrate-enriched conditions. SOD is a proxy for microbial activity. Mean ± SE.

During the flow-through core incubation with ambient site water, all but one pond released NH_4 from the sediments, all ponds had extremely small NO_x fluxes, all but one pond released ON from the sediments, and all but one pond released PO_4 (Figure 7).

During the nitrate-enriched phase of the flow-through incubation, the sediments took up large amounts of NO_x and all ponds released PO_4 (Figure 7). The eight year-old pond exhibited aberrant fluxes again, most notably a smaller uptake of NO_x compared to the other ponds (Figure 7).

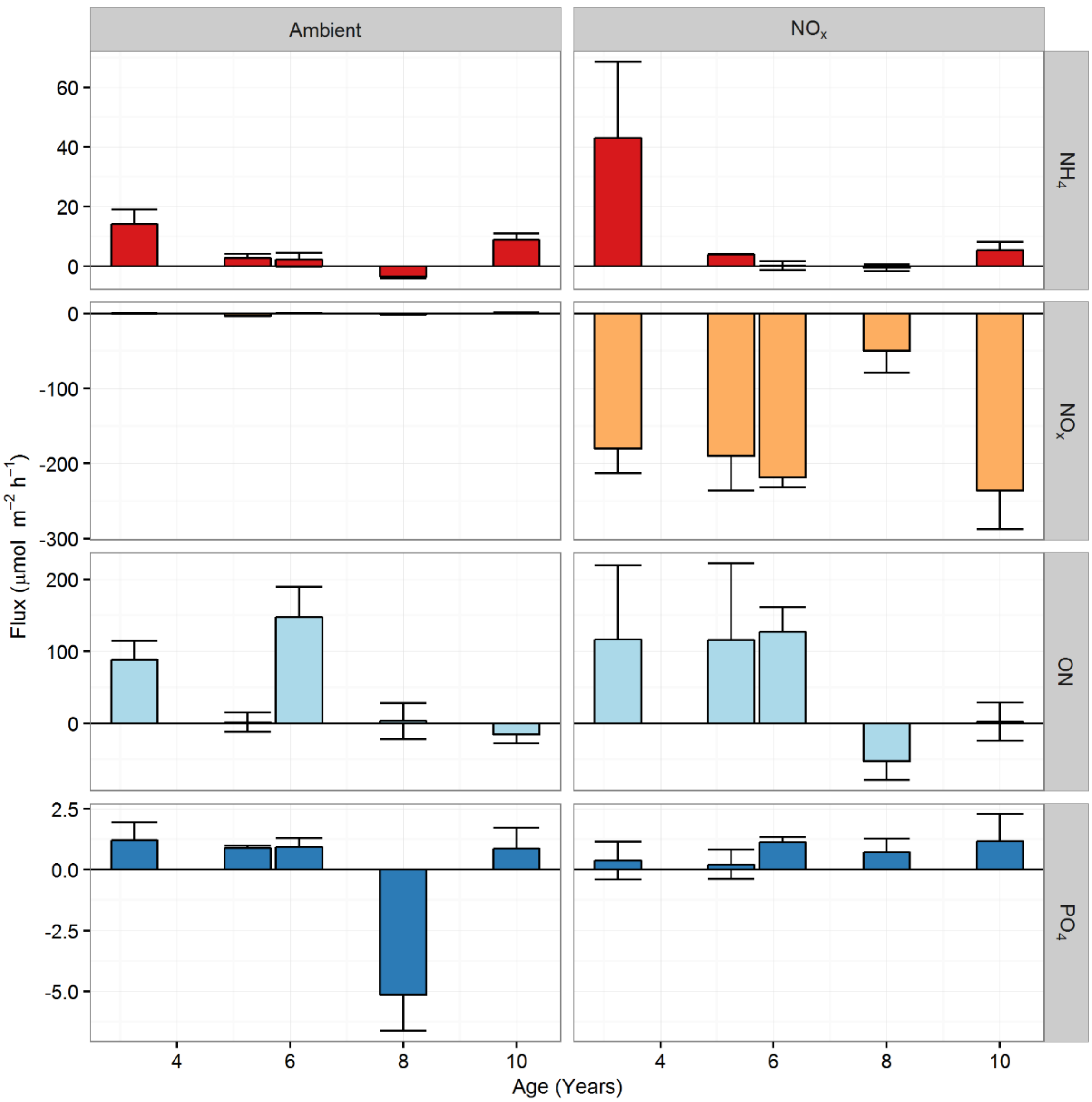

Figure 7. Nutrient fluxes from the sediment surface measured during the flow-through core incubation under both ambient and nitrate-enriched conditions. Mean $\pm$ SE.

3.2. Single pond flux experiment

3.2.1. Pond depth profiles

The main pond during sampling had an RTRM of 60.82, indicating stratified conditions (Table 1). The forebay was mixed with an RTRM of 26.93 (Table 1). Both the main pond and forebay had higher dissolved oxygen concentrations in the top part of the water column and higher turbidity in the bottom of the water column (Table 1).

Table 1. Depth profile measurements from the top and bottom water of the single pond

	RTRM	Condition	Depth (m)	Temperature (˚C)	DO (mg/l)	DO (%)	Turbidity, (NTU)
Main	60.82	Stratified	0.463	28.09	3.61	46.1	3.9
			2.385	25.92	0.71	8.8	22.7
Forebay	26.93	Mixed	0.591	27.26	2.12	26.7	3.3
			1.841	26.29	0.54	6.7	8.2

3.2.2. Gas fluxes from the sediment-water interface

In the six-year-old pond, there were no significant differences in net N_2 fluxes between zones within the forebay, but the shallow and deep sediments from the main part of the pond had significantly lower rates of nitrogen fixation than all other areas sampled (Figure 8, $p < .05$). Comparing the deep-main sediment core net N_2 fluxes to the same site from almost a month earlier during the five pond flux experiment, the magnitude of nitrogen fixation increased five-fold from a net N_2 flux of -11.96 to -60.72 µmol N m^{-2} hr^{-1}.

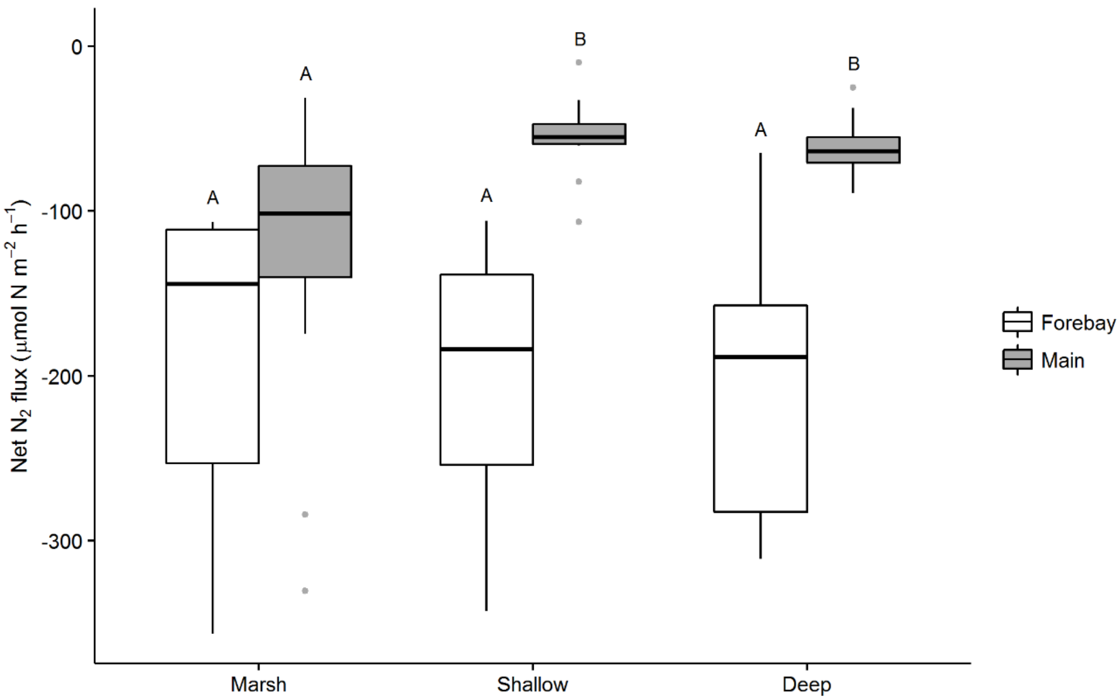

Figure 8. Rates of net N_2 flux in different areas of the pond (forebay, main) and different vegetation covers (marsh, shallow, deep). Letters indicate significant differences ($p < .05$) based on a Tukey HSD post-hoc test.

In the single pond, net N_2 flux and SOD showed a strong and significant negative relationship (Figure 9, $p < .01$, $r^2 = .88$). This relationship is opposite of what has been found in previous studies relating SOD and net N_2 flux in other habitats [*Piehler and Smyth*, 2011; *Eyre et al.*, 2013; *Smyth et al.*, 2013]

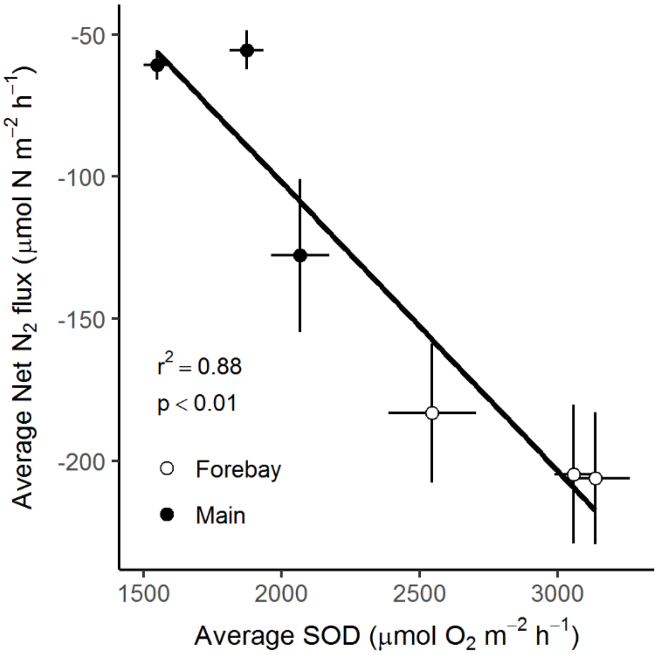

Figure 9. Average net N_2 flux as a function of Average SOD in the single pond flux experiment. Mean $\pm$ SE.

3.3. Single pond monitoring

The time series of nitrogen species and chlorophyll-*a* show effective removal (comparing inflow and outflow) of nitrogen throughout the entire sampling period and a shift in chl-*a* concentrations from net addition during the summer to net removal during the fall (Figure 10). Inflow concentrations of nitrogen species and chl-*a* increased substantially during the fall and then decreased during December.

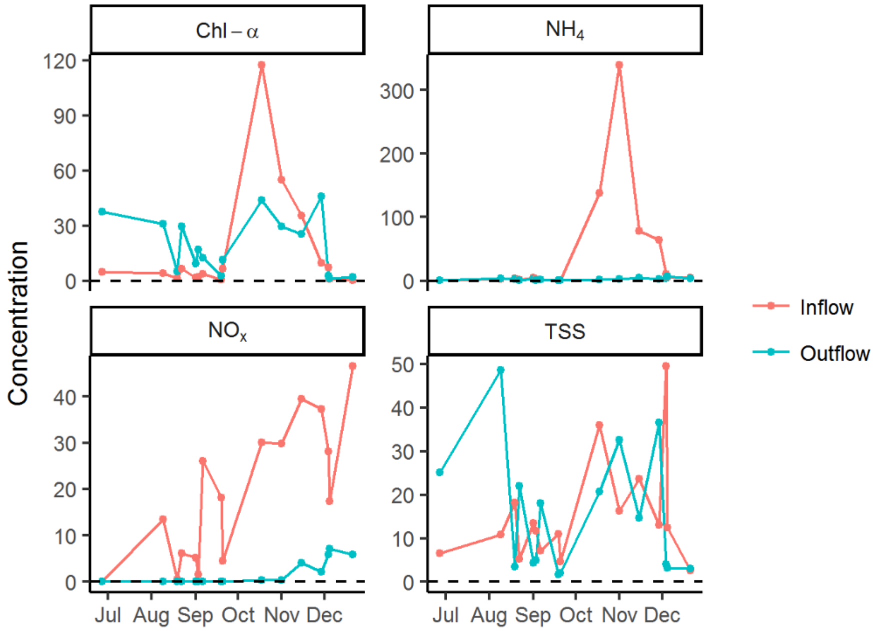

Figure 10. Time series of Chlorophyll-*a* (μg/L), nitrogen species (μM), and TSS (mg/L) from the single pond.

Figure 11 shows a time series of the change in autochthonous DIN concentrations. The change in autochthonous DIN shifted between slightly positive and slightly negative during the summer, indicating small magnitudes of autochthonous DIN production and removal, respectively. Autochthonous removal increased substantially during the fall and decreased during late November and December.

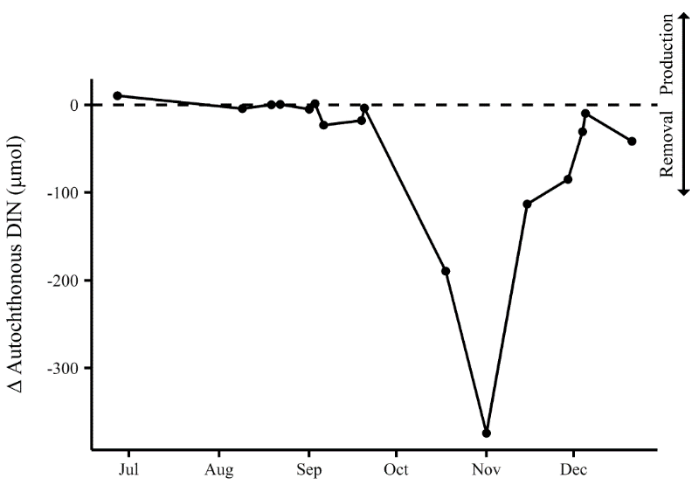

Figure 11. Time series of change in DIN concentration between inflow and outflow that is not explained by theoretical N uptake from observed change in chl-*a* concentration between inflow and outflow.

Combining both baseflow and stormflow measurements of removal efficiency for each water quality variable over the study period showed that the pond was extremely effective at removing NO_x (-93.74%), and effective at removing NH_4 (-49.48%), total nitrogen (TN,-43.77%), organic nitrogen (ON, -7.34%), and orthophosphate (PO_4, -61.34%) (Table 2). The pond increased concentrations of TSS (50.30%) and chl-a (289.56%) (Table 2). There was significant variability between baseflow and stormflow measurements in TSS, with the pond increasing TSS concentrations by 108% during baseflow and decreasing TSS concentrations by 76% during stormflow (Table 2).

Table 2. Mean removal, mean percent removal, and standard deviations for the two means for each water quality variable.

Baseflow	TSS	NO_x	NH_4	PO3	TN	ON	Chl-a
Mean Change	6.96	-20.99	-55.40	-2.52	-81.08	-4.69	2.34
SD	*16.95*	*14.38*	*103.56*	*3.80*	*112.36*	*9.13*	*31.38*
Mean % Change	**108.01**	**-92.75**	**-47.68**	**-58.49**	**-50.97**	**-12.82**	**256.25**
SD	*156.61*	*12.70*	*48.16*	*30.14*	*35.89*	*29.53*	*275.50*

Stormflow	TSS	NO_x	NH_4	PO3	TN	ON	Chl-*a*
Mean Change	-17.07	-9.52	-2.37	-1.47	-11.51	0.38	4.85
SD	*16.15*	*9.98*	*2.60*	*0.70*	*12.87*	*4.22*	*7.03*
Mean % Change	**-76.67**	**-95.74**	**-53.45**	**-67.59**	**-27.93**	**4.72**	**362.86**
SD	*13.67*	*9.19*	*19.65*	*12.54*	*19.24*	*25.59*	*279.57*

Total	TSS	NO_x	NH_4	PO3	TN	ON	Chl-*a*
Mean Change	-1.05	-17.41	-38.83	-2.19	-59.34	-3.11	3.13
SD	*19.92*	*13.95*	*88.30*	*3.16*	*97.82*	*8.14*	*25.90*
Mean % Change	**50.30**	**-93.74**	**-49.48**	**-61.34**	**-43.77**	**-7.34**	**289.56**
SD	*149.97*	*11.40*	*40.70*	*25.82*	*32.85*	*28.75*	*272.12*

4. Discussion

4.1. Wet pond conditions during summer

This study sampled multiple ponds during summer, but the high temperatures and pond stratification created extreme conditions that are likely unique to this season. Stormwater ponds and other shallow water bodies experience long periods of stratification from late spring to early fall [*Wilhelm and Adrian*, 2008; *Song et al.*, 2013] (Table 1). Most of the nitrogen removed during this time is accounted for by an increase in phytoplankton biomass (Figure 11), which suggests that nitrogen from stormwater inflows was not influenced by pond sediments. Assuming some algae is retained within the pond, the production of algae combined with frequent summer stratification leads to a drawdown of dissolved oxygen in the bottom water by microbial activity (Figure 6) and extended hypoxic conditions at the sediment-water interface [*Diaz and Rosenberg*, 2008], promoting the release of sediment-sorbed phosphorus [*Wilhelm and Adrian*, 2008; *Song et al.*, 2013; *Duan et al.*, 2016]. Phosphorus likely builds up in the bottom water of the pond during the summer, further increasing nitrogen limitation, and when the pond mixes in the fall, there is a release of phosphorus [*Song et al.*, 2013] and large increases of nitrogen and chl-*a* concentrations [*Wilhelm and Adrian*, 2008] (Figure 10). Stormwater pond waters have been shown to be dominated by autochthonous processing of carbon rather than by watershed inputs [*Williams et al.*, 2013], so the large increase in nitrogen and chl-*a* concentrations could be from the oxidation of organic matter and re-mineralization of nitrogen that was deposited on the pond bottom during the summer when the pond was stratified. Alternatively, watershed inputs of nitrogen from fertilizers could contribute to increase nitrogen and chl-*a* in the pond during fall.

4.2. Summer nitrogen dynamics

4.2.1. Effects of pond age

Over time, the amount of organic matter in pond sediments increased (Figure 3B, *Moore and Hunt*, 2012). This is likely caused by increased vegetation in the ponds as they age as well as phytoplankton production and subsequent deposition of algal material in the sediments (Figure 3A). As plants increase their areal extent over time, ponds experience a decrease in total nitrogen concentrations and an increase in nitrogen limitation (Figure 3C, Figure 3D). Combined with summer stratification [*Song et al.*, 2013] (Table 1), summer nitrogen limitation in wet ponds becomes more extreme as the ponds age due to increased organic matter accumulation in the sediments, leading to more respiration and anoxic conditions. This idea is supported by an increase in normalized SOD as the ponds age (Figure 6), indicating that older ponds have more microbial activity that would create more anoxic conditions and nitrogen limited conditions.

This extreme nitrogen limitation likely explained the nitrogen fixation observed during ambient conditions in all pond sediments in the five pond flux experiment. N_2 fluxes shifted differentially based on age in response to nitrate enrichment (Figure 4). Net nitrogen fixation in aquatic sediments has been measured in low nitrate or low inorganic nitrogen conditions [*Scott et al.*, 2008; *Fulweiler et al.*, 2013; *Newell et al.*, 2016a, 2016b], but net sediment nitrogen fixation has not been measured before in coastal stormwater wet ponds. Studies have measured denitrification or hypothesized that denitrification is an important removal mechanism for nitrogen in wet ponds [*Groffman et al.*, 2004, 2009; *Zhu et al.*, 2004; *Collins et al.*, 2010; *Bettez and Groffman*, 2012], but results from this study suggest that this may not be true during the summer in coastal wet ponds. While this study did not directly measure net sediment N_2 fluxes during other seasons, monitoring data suggests that denitrification could be important in the fall (Figure 11) and other seasons as well.

It is worth noting that increased nitrogen fixation in older ponds after the nitrate enrichment was in part due to the experimental design and the incorporation of water column processing within the feed water of the flow-through incubation set-up. Much of the nitrate in the feed water was removed over the 12 hours between the nitrate addition and sampling (Figure 5), so the younger pond sediments were exposed to higher concentrations of nitrate because their water column removed less nitrate over that time period. This water column nitrate removal (Figure 5) shows that older ponds have more nitrate removal in the water column than younger ponds, which could be expected due to increased nitrogen limitation with pond age (Figure 3D). This water column processing is likely heterotrophic due to the absence of light throughout the incubation. Given that older pond sediments were subject to lower nitrate concentrations at the time of sampling due to water column removal, the benthic heterotrophic nitrogen-fixers that are likely driving the net sediment nitrogen fixation may not have been inhibited by the nitrate concentrations they experienced in the older ponds.

Benthic heterotrophic nitrogen-fixers are less sensitive to levels of inorganic nitrogen than nitrogen-fixers found in the water column [*Knapp*, 2012], and they can fix substantial amounts of nitrogen at higher levels of inorganic nitrogen in dark, anoxic waters [*Knapp*, 2012; *Farnelid et al.*, 2013; *Foster and Fulweiler*, 2014; *Newell et al.*, 2016a]. One explanation for this phenomenon is that extremely low oxygen concentrations and high carbon availability, the conditions found in this study, may reduce the bacterial community's sensitivity to inorganic nitrogen [*McGlathery et al.*, 1998] and allow a select group of nitrogen-fixing benthic heterotrophs to dominate [*Newell et al.*, 2016a]. This could explain the increase in net nitrogen fixation between ambient conditions (0 µM NO_x) and nitrate-enriched conditions in older ponds (10 - 20 µM NO_x) that would have been expected to shift towards net denitrification due to the availability of nitrate [*Scott et al.*, 2008]. In addition, the decrease in oxygen concentration in the core's overlying water (an increase in SOD) could have caused the increased net nitrogen fixation between ambient and nitrate-enriched conditions (Figure 9 - intra-pond relationship).

In situ, nitrate-rich stormwater flows into the pond and likely takes time to mix with existing water before reaching the sediments. In the summer, pond stratification appears to keep nitrogen from stormwater separated from the sediment-water interface during baseflow conditions (Table 1). If nitrogen from stormflow reaches the sediment-water interface during the summer after a storm event, it is likely that nitrate removal within the water column would result in concentrations too low to inhibit benthic nitrogen fixers. Evidence from this study suggests that wet pond sediments undergo net nitrogen fixation until the stratification ends in the fall.

4.2.2. Effects of location within pond

The net sediment N_2 fluxes from the forebay and main pond sediments of the stormwater pond were significantly different and were driven by SOD (Figure 9). This difference in both SOD and net N_2 fluxes is possibly a result of the forebay's position upstream of the main pond, allowing suspended materials to settle out, increasing sediment organic matter [*Zhu et al.*, 2004], and driving SOD.

The values of net N_2 flux in both the forebay and main pond were an order of magnitude larger than those observed in the five pond flux experiment that took place earlier in the summer (late June compared to early August). This large increase in net nitrogen fixation and the presence of stratification within the main pond later in the summer supports the idea that pond stratification and associated water quality effects, which become more intense as the summer progresses, are the root causes of the net sediment nitrogen fixing seen in this study.

4.3. Pond water quality monitoring

4.3.1. Nitrogen - chl-*a* tradeoff during summer

During summer and early fall, the pond effectively removed all three nitrogen species, but generated large amounts of chl-*a* (Figure 10). It is likely that summer pond stratification that led to sediment nitrogen fixation favored algal uptake of nitrogen in surface waters. The relatively low autochthonous DIN production during the summer was likely from the remineralization of nitrogen from organic matter around the edges of the pond. Autochthonous DIN removal could be the result of denitrification, DNRA, anammox, or plant uptake, which were not assessed.

This nitrogen - chl-*a* tradeoff during the summer presents a problem for water quality managers because labile phytoplankton biomass can be easily washed downstream during storms where it degrades and draws down dissolved oxygen concentrations. Denitrification is the preferred nitrogen removal pathway in the wet pond because bioavailable nitrogen is permanently removed from the system. The wet pond provided ideal habitat for algae to grow during the summer, and this algal growth in the pond essentially pushed the biogeochemical cycles upstream that would normally occur farther downstream in the estuary. By moving algal growth upstream, the nitrate incorporated into algal biomass in the pond bypassed important denitrifying habitats, such as salt marshes and streambeds, that could permanently remove nitrate instead of causing eutrophication. Furthermore, the decomposition of this labile carbon downstream could decrease the health, stability, and denitrification efficiency of these important denitrifying habitats.

4.3.2. Autochthonous DIN removal during fall

The increase in nitrogen concentrations and Chl-*a* concentrations during the fall likely resulted from an increase in loading from the pond's watershed and the remineralization of nutrients resulting from pond mixing. The fall increase in autochthonous DIN removal suggests that microbial processing of nitrogen via sediment denitrification or anammox increased and plant uptake by duckweed (*Lemna* L.) increased. Pond turnover in the fall could increase denitrification or anammox by supplying the sediments with nitrate, and a proliferation of duckweed observed during fall sampling indicated that duckweed was utilizing nutrients in the pond. Duckweed was not measured because water samples were filtered before analysis. Other mechanisms for DIN removal were less likely during this time as macrophytic plant uptake was waning, and DNRA has been shown to be minor relative to other nitrogen removal pathways in freshwater sediments [*Scott et al.*, 2008]. Anammox was also unlikely due to low rates, relative to denitrification, found in estuarine environments [*Koop-Jakobsen and Giblin*, 2009]. This supports the idea that denitrification and uptake by duckweed accounted for the observed autochthonous nitrogen removal.

4.3.3. Nutrient and pollutant removal

Over the 6 month monitoring period, the wet pond achieved its regulatory goal (based on *NCDENR*, 2009) of removing more than 25% of TN (-43.77% change, Table 2) but did not meet its regulatory goal of removing 85% of the TSS that flow into it (50.30% change, Table 2). I did not measure total phosphorus as a part of this study, but the pond removed substantial amounts of orthophosphate (-61.34%, Table 2).

4.4. Implications and suggestions for coastal wet pond management

The net nitrogen fixation measured in stormwater wet pond sediments illustrates that the common design of large, deep stormwater wet ponds may not be the most effective solutions for improving the water quality of urban stormwater in coastal areas of the southeastern US, especially during the summer. While the pond removed many of the water quality pollutants discussed above, it also converted a large amount of nitrogen into algal biomass during the summer that is likely exported during storms, contributes to a buildup of organic matter and phosphorus during the summer, and is possibly remineralized and released during the fall. Large amounts of algal biomass in stormwater wet ponds have been documented on the southeastern US coast [*Lewitus et al.*, 2008; *DeLorenzo et al.*, 2012], suggesting that the nitrogen - chl-*a* tradeoff commonly occurs along the southeastern US coast.

Due to the negative effects on water quality from wet ponds in coastal areas, this study recommends using alternative kinds of SCMs to manage coastal stormwater. Using SCMs with shallower water or no standing water at all could possibly decrease the amount of chl-*a* exported downstream during the summer and decrease the concentrations of nitrogen during the fall because of the lack of stratification. More work is needed on seasonal nitrogen cycling within wet ponds and in alternative SCMs, such as stormwater wetlands and bioretention cells that could be used to replace the water quantity and quality control of wet ponds.

Results from this study and others suggest that the net sediment nitrogen fixation observed during the summer in coastal wet ponds is caused by pond stratification. Also, water quality issues during the summer and fall, such as algal blooms, may be caused by summer stratification and a lack of permanent nitrogen removal by pond sediments. A direct solution to this stratification is to increase pond mixing to break up the thermocline and re-connect the sediment-water interface with oxygenated water and nitrogen from stormwater inflows. However, exposing sediments to too much oxygenated water could reduce the pond sediments' carbon storage capability and reduce denitrification. A possible compromise for carbon storage, nitrogen removal, and phosphorus removal would be to mix the pond periodically to create alternating low and high-oxygen conditions, or design ponds to be shallower to discourage stratification.

Another management action that could improve pond function is more frequent excavation. Frequent excavation is often associated with greater water storage capacity, suspended sediment removal, and phosphorus removal, but this study suggests that frequent excavation could also reduce the amount of nitrogen in ponds during the fall. Excavating organic-rich and phosphorus-rich sediments from ponds could reduce anoxic conditions and phosphorus buildup in bottom water during the summer, which could decrease net nitrogen fixation during summer and reduce the amount of carbon and nitrogen re-mineralization after pond turnover in the fall.

5. Conclusions

Stormwater wet ponds are generally considered important places for denitrification, the permanent removal of nitrogen from the environment, but this study shows that this may not be the case in coastal areas of the southeastern US during the summer months. Based on net sediment N_2 fluxes from five wet ponds in early summer and one pond in late summer, this study found that stormwater wet pond sediments in coastal areas of the southeast US are possible spots of net nitrogen fixation during the summer.

Based on the five pond survey, all ponds had net sediment nitrogen fixation during ambient conditions. After a nitrate addition, rates of net nitrogen fixation were significantly correlated with pond age, with older ponds having higher rates of net nitrogen fixation and younger ponds shifting towards net denitrification. Additionally, nitrate removal in the water column was significantly correlated with pond age, and older ponds had higher removal. In the single pond study, the pond's forebay had significantly larger rates of net sediment nitrogen fixation. Rates of net nitrogen fixation between the five pond study at the beginning of summer and the single pond study towards the end of summer increased by an order of magnitude. In all cases, the cause of this net nitrogen fixation was most likely pond stratification that exacerbated anoxic conditions, phosphorus release, and nitrogen limitation at the sediment-water interface. Monitoring data showed that during the summer, almost all DIN removed from the pond was converted into algal biomass and exported downstream, essentially creating a nitrogen - chl-*a* trade-off. Nitrogen concentrations in the pond substantially increased in the fall, possibly due to the remineralization of organic matter after pond turnover. This increase in nitrogen concentrations spurred large amounts of autochthonous DIN removal, which was most likely denitrification and uptake by duckweed. The removal efficiency of the pond met North Carolina stormwater guidelines for nitrogen but not for total suspended solids.

Stormwater wet ponds may not be the most effective solutions for improving or maintaining water quality in coastal areas because they do not appear to provide any permanent nitrogen removal during the summer. Rather, they undergo net sediment nitrogen fixation, transform nitrogen from stormwater into large increases in algal biomass, and possibly increase nitrogen concentrations in the fall after pond turnover. These pond functions are likely due to persistent thermal stratification during the summer. This study recommends the use of alternative SCMs with shallow water or no standing water instead of wet ponds in coastal areas, increased pond circulation and aeration in existing coastal ponds to decrease stratification and its effects, and frequent pond excavation to reduce anoxic conditions, phosphorus release, and nitrogen limitation in pond sediments.

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APPENDIX

	Developed	% Change	Reference	% Change
Chl-*a*	8.23 ± 14.56 μg/L	349.26	.64 ± 1.93 μg/L	76.35
NH_4^+	-0.71 ± 8.60 μM	-13.61	.22 ± 1.05 μM	21.84
NO_x^-	1.97 ± 4.85 μM	51.80	-	-
ON	7.15 ± 10.38 μM	57.93	-	-
PO_4^{3-}	-.30 ± .65 μM	-32.24	-	-
TSS	18.99 ± 41.51 mg/L	310.65	-	-

Table A.1. Change in mean concentrations of water quality variables between the Pre and Post period and the percent change from the mean concentrations for the Pre period. Only variables that significantly changed are listed.

Tarawa	***TSS***	***NO_x^-***	***NH_4^+***	***PO_4^{3-}***	***ON***	***Chl-a***
3/8/2011	5.48	2.67	5.45	8.07	18.26	11.87
6/18/2015	40.34	5.79	2.95	8.39	21.70	9.84
Change	34.86	3.13	-2.50	0.31	3.44	-2.03
% change	635.69	117.21	-45.90	3.87	18.83	-17.12

French	**TSS**	**NO_x^-**	**NH_4^+**	**PO_4^{3-}**	**ON**	**Chl-a**
3/8/2011	8.55	0.39	-0.87	0.41	26.51	2.23
6/18/2015	11.05	-0.70	-0.72	0.30	30.26	1.92
Change	2.50	-1.09	0.16	-0.11	3.75	-0.31
% change	29.25	-279.82	-17.92	-26.48	14.15	-14.02

Table A.2. Predicted values for the beginning and end of the Post, the change over the Post period, and the percent change over the Post period for each stream.

	F Statistic	P-Value
Chl-*a*	2.0936	0.3163
NH_4^+	38.1077	**8.618e-05**
NO_x^-	124.7187	**1.489e-06**
ON	25.3087	**0.0003381**
PO_4^{3-}	37.7774	**8.875e-05**
TSS	6.704	**0.01433**

Table A.3. Results of F-test between the variance of Developed and a Developed stormwater wet pond for each water quality variable ($\alpha = .05$)

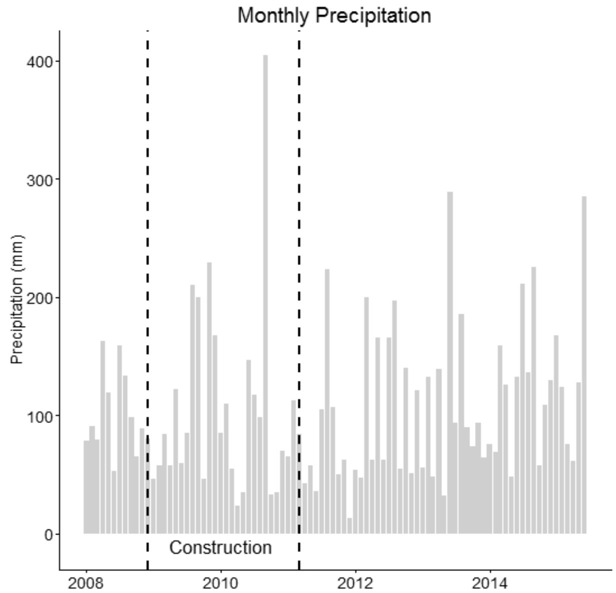

Fig. A.1. Monthly Precipitation for both watersheds for Pre-construction (Pre), Construction (Mid), and Post-Construction (Post).

Developed

	2006	2011	Change
Barren Land	0	0.76	0.76
Cultivated Crops	2.80	2.80	0
Developed, High Intensity	0.64	1.40	0.76
Developed, Low Intensity	37.32	36.43	-0.89
Developed, Medium Intensity	9.94	21.02	11.08
Developed, Open Space	30.32	25.35	-4.97
Evergreen Forest	15.41	10.06	-5.35
Shrub/Scrub	3.44	2.16	-1.27
Woody Wetlands	0.13	0	-0.13

Reference

	2006	2011	Change
Barren Land	4.66	4.90	0.25
Deciduous Forest	0.06	0.06	0
Developed, Low Intensity	2.81	2.63	-0.18
Developed, Medium Intensity	0.01	0.15	0.14
Developed, Open Space	2.09	2.14	0.04
Emergent Herbaceuous Wetlands	17.50	17.77	0.27
Evergreen Forest	6.64	6.59	-0.05
Herbaceuous	12.82	12.71	-0.11
Mixed Forest	0.78	0.78	0
Open Water	0.24	0.33	0.10
Shrub/Scrub	22.83	22.67	-0.16
Woody Wetlands	29.55	29.26	-0.29

Table A.4. Land cover for each study watershed in percent watershed area and the change between 2006 and 2011.