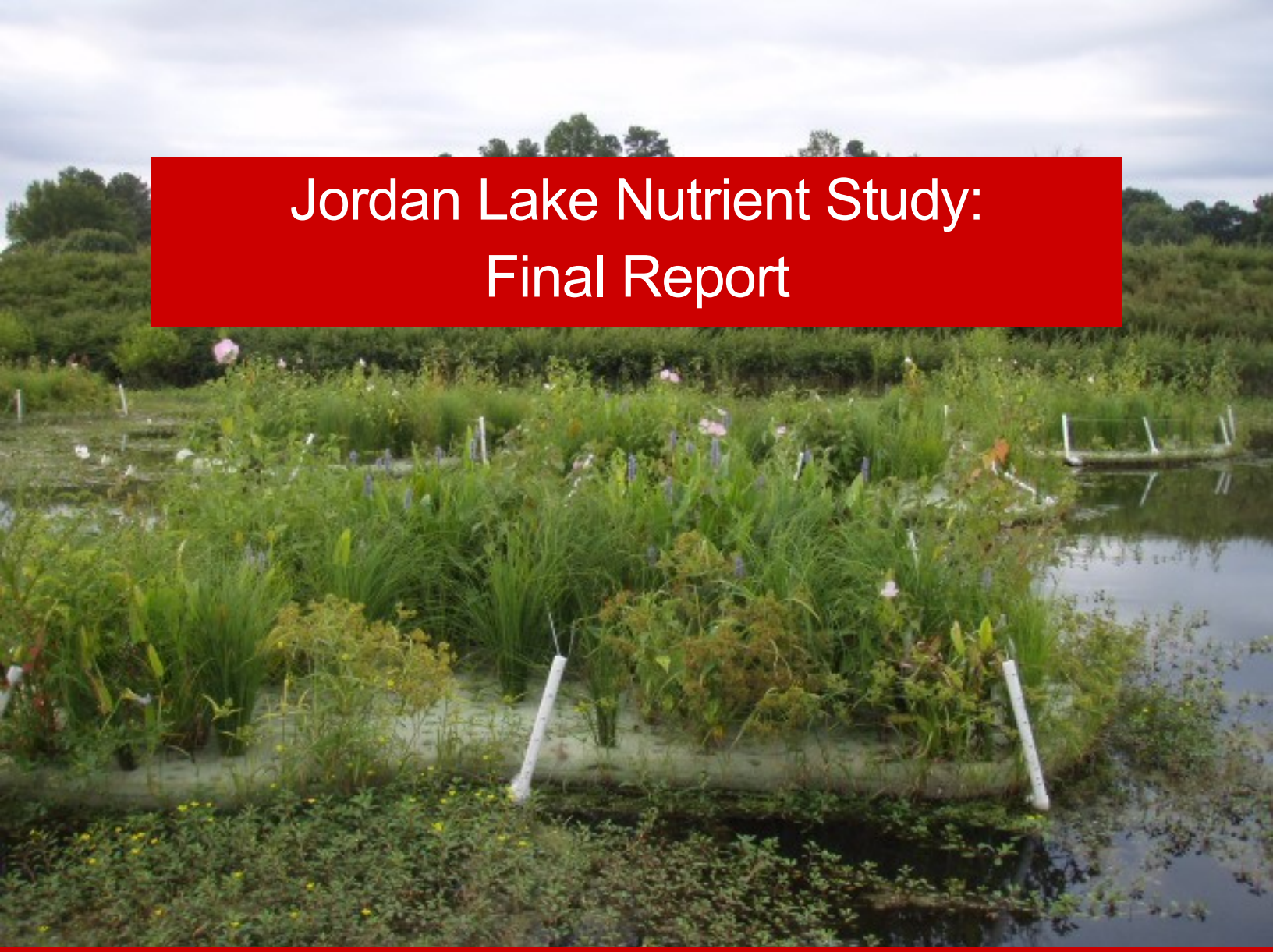

Jordan Lake Nutrient Study: Final Report

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EXECUTIVE SUMMARY

The Stormwater Engineering Group in the Department of Biological and Agricultural Engineering at NC State University conducted a series of research projects and activities as part of the Jordan Lake Nutrient Study and the North Carolina Policy Collaboratory. The following research activities were completed and are detailed below: (1) an assessment of the long-term water quality treatment performance of bioretention, (2) an examination of bioretention media to quantify changes in composition with age and identify maintenance needs, (3) a comprehensive review of literature on floating treatment wetlands, and (4) a column study to assess the potential treatment performance of various blends and depths of sand filters.

The aging bioretention cell sustained, and in some cases improved, nitrogen and phosphorus removal following 17-years of continuous performance. Filter media samples indicated the top 8 inches of filter media were nearing phosphorus saturation, but with 4 feet of filter media, lower depths would most likely continue to provide treatment. The field survey of bioretention media determined that clay and silt portions of bioretention media and phosphorus concentrations significantly increased with age while carbon-to-nitrogen ratio (C:N) significantly decreased. While 32% of survey bioretention cells appeared to be saturated with phosphorus, saturation is likely confined to the upper layers of media allowing deeper depth to continue treatment. Floating treatment wetlands are an increasingly popular retrofit to improve wet pond nutrient removal, particularly because of their ease of installation and relatively low-cost. Nutrient removal appears varied and dependent upon providing enough coverage and proper placement. Lastly, sand filter

columns appeared to outperform the credit assigned to them through NC DEQ, but additional data and field trials are needed to reach conclusions that are more definitive.

LONG-TERM WATER QUALITY PERFORMANCE OF BIORETENTION

The research conducted as part of this study has been as an open access peer reviewed journal article in *Sustainability* (Johnson and Hunt 2019). The findings are summarized below, but the entirety of the research may be found in the article (Appendix A).

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Summary

One of the most popular stormwater practices in (sub-)urban North Carolina is bioretention. While bioretention has been researched intensively to determine the most efficient designs, few long-term studies have attempted to assess the performance of older bioretention. However, previous research and design guidance for bioretention has predicted long-term water quality treatment. This study compared discharged concentrations and loads of nitrogen and phosphorus from a bioretention cell (1) post-construction and (2) following 17 years of treatment. A conventionally-drained bioretention cell with lateral underdrains in Chapel Hill, North Carolina, USA, was first monitored post-construction for 10-months from 2002--2003 and, again following continuous use, for 14 months from 2017-2018. Estimated mass load reductions during the initial monitoring period were 40% for total nitrogen (TN) and 65% for total phosphorus (TP). Mass load reductions were increased 17 years after construction, with reductions of 72% and 79% for TN and TP, respectively. Plant growth, death, and decay over the 17-year life of the bioretention cell are hypothesized to have contributed additional nitrogen assimilation and carbon to the fill media, serving as a catalyst for nitrogen treatment. Phosphorus removal remained relatively unchanged between the two

monitoring periods. Filter media samples indicated the top 8 inches of filter media were nearing phosphorus saturation, but with 4 feet of filter media, lower depths would most likely continue to provide treatment. If designed, built, and maintained correctly, bioretention appears to provide sustained treatment of stormwater runoff for nitrogen and phosphorus for nearly two decades, and likely longer.

BIORETENTION MEDIA STUDY

Introduction

Previous research has typically focused on the hydrologic performance and pollutant removal capabilities of newly installed bioretention cells (BRCs) while basing long-term predictions on post-construction results. As BRCs are a biologically-active filter-based system, changes in performance are expected over the life of an installation. Recent research has begun the process of understanding the impacts of aging on BRCs by comparing the nutrient removal capabilities in aging bioretention cells to post-construction values (Johnson and Hunt 2019; Willard et al. 2017). Outside of water quality treatment in aging BRCs, little research has examined the filter media, and changes in its physical and chemical composition, that provides many of the treatment mechanisms that make bioretention such a popular SCM.

In a comparative study of nutrient removal in a 17-year-old BRC in North Carolina, Johnson and Hunt (2019) attributed improved nitrogen removal to steady cycling of organic matter from the maturation of vegetation planted at the bioretention cells' surface. McPhillips et al. (2018) associated low carbon to nitrogen (C:N) ratios in BRC media with nitrogen loss due to coupled mineralization and nitrification of organic nitrogen and recommended C:N be greater than 20 to promote immobilization. Komlos and Traver

(2012) concluded following media analysis in a nine-year-old rain garden that phosphorus removal would continue for an additional 20 years. Johnson and Hunt (2016) explored the spatial distribution of phosphorus and heavy metals in an 11-year-old BRC in Charlotte, NC, and reported notable accumulation of zinc and phosphorus solely within the pre-treatment area of the BRC. However, BRC media saturated with phosphorus has been shown to be a net source of the nutrient rather than to provide treatment (Hatt et al. 2009b; Hunt et al. 2006).

This study aimed to improve the understanding of dynamic temporal changes in bioretention media composition by exploring (1) the physical characteristics and (2) carbon and phosphorus concentrations in BRCs of various ages through a field survey of 28 BRCs in central North Carolina. Comparing concentrations of nutrients in “new” and “aging” bioretention media will support phosphorus sorption predictions for media life and maintenance recommendations for maximum sustained nutrient removal.

Methods and Site Descriptions

Site Descriptions

Twenty-eight BRCs within the nutrient sensitive Jordan Lake watershed in central North Carolina were selected for sampling (Figure 1; Table 1). Selected BRCs were chosen to create (1) an expansive geographical representation across eight municipalities in the Jordan Lake watershed, (2) a broad swath of bioretention ages, and (3) a relatively equal distribution of design variations for future analysis (e.g., media depth, vegetation type, and presence of forebay; Table 1).

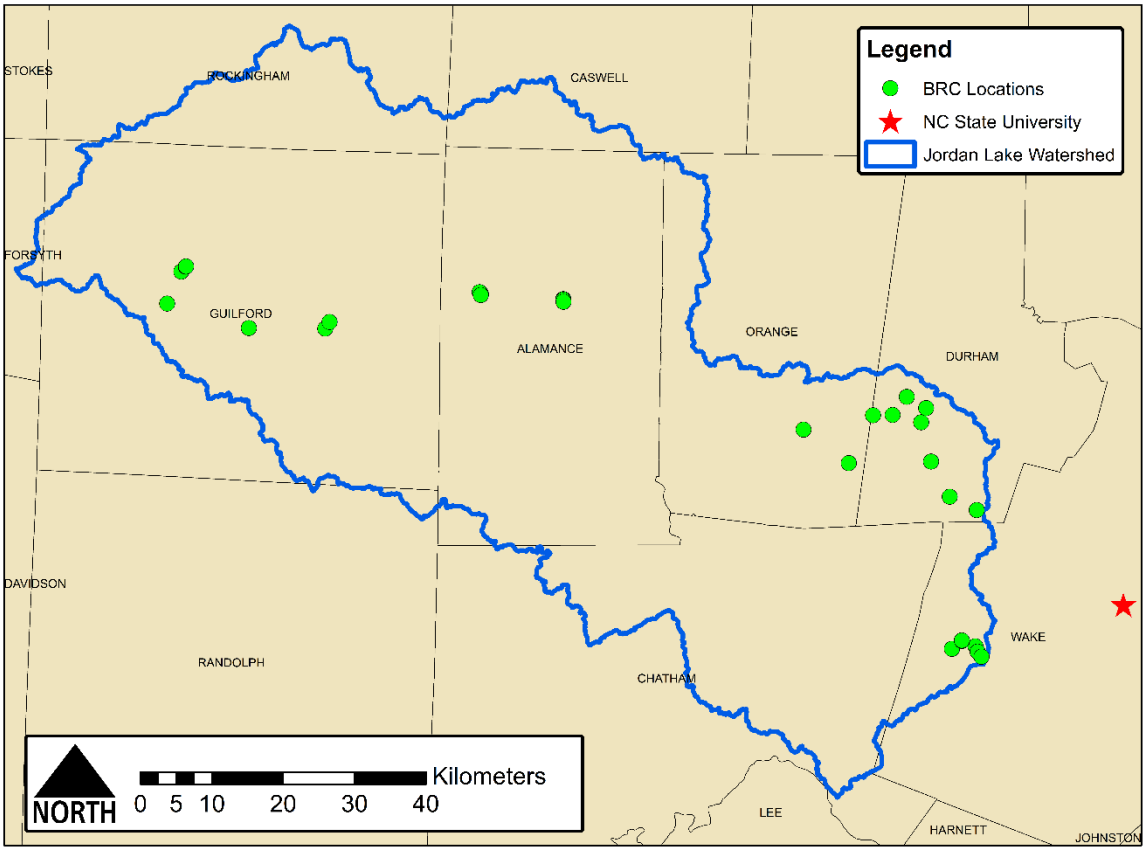

Figure 1. Locations of sampled bioretention cells in central North Carolina.

Table 1. Characteristics for sampled bioretention cells.

Site ID	Year Built	Drainage Area (ac)	% Impervious	BRC Surface Area (ft^2)	Ponding Depth (in)	Media Depth (ft)	Vegetation Type	Forebay
Apex-BCB	2006	1.6	100	720	6	4	Trees	No
Apex-BCE	2005	3.7	100	1,675	15	4	Trees	No
Apex-PUB	2017	2.1	62.3	6,640	9	3	Shrubs	No
Apex-PW	2014	1.2	62.4	4,175	12	3	Shrubs	No
Apex-TO	2002	1.4	50.2	2,640	12	3	Trees	Yes
Apex-WAL	2011	3.5	41.4	7,577	12	2	Sod	Yes
BUR-AEA	2012		75	905	18	2	Sod	No
BUR-LCA	2013	0.8	60	3,819	11	3	Shrubs	No
BUR-LCB	2013	3.3	61	7,607	14	3	Shrubs	No
BUR-LCC	2014	0.8	62.6	2,332	9	2.5	Sod	No
BUR-OE	2015	3.5	56.9	8,591	11	2	Sod	No
CARR-BP	2010	6.8	20	1,526	10	3	Trees	No
CH-PP	2014	24.7	20	27,996	12	3	Shrubs	No
CH-UM	2001	0.3	100	970	9	4	Trees	No
DUR-AB	2008	1.0	63	2,492	12	3	Trees	Yes
DUR-DFA	2015	1.5	66.9	4,790	9	3	Shrubs	Yes
DUR-DFB	2015	2.9	56.5	7,957	9	3	Shrubs	Yes
DUR-JHS	2015	1.4	74.8	4,960	12	3	Trees	Yes
DUR-LW	2013	7.4	36.1	4,380	5	2.5	Trees	Yes
DUR-SHZ	2018	2.5	76	7,090	12	3	Sod	Yes
DUR-TS	2016	0.5	66.7	1,553	9	3	Shrubs	Yes
ELON-MP	2011		55	7,338	18	2	Sod	No
GSO-BC	2009	1.3	90	4,200	12	2	Sod	No
GSO-BFA	2007	1.6	73	5,080	9	4	Shrubs	No
GSO-BFB	2007	1.1	65	3,200	9	4	Shrubs	No
GSO-PH	2007	2.6	65.4	7,301	12	3	Trees	No
MCL-FIRE	2008	1.4	58	4,384	8	2	Sod	No
MCL-WC	2005	2.3	21.5	2,292	6	4	Trees	No

Age, design, and watershed characteristics were collected using several methods. Construction dates, design plans and as-built plans for the sampled BRCs were provided by municipalities. In the absence of design plans, design and watershed characteristics were measured in the field or through GIS. Defining characteristics included age, land use, drainage area, land use, drainage area imperviousness, BRC surface area, drainage area:BRC surface area ratio, surface storage ponding depth, media depth, vegetation type, and whether a pre-treatment device was present. Design characteristics were used for future analysis.

Sample Procedure

Five soil samples were collected at each site using a 12-inch stainless steel soil probe. Bioretention cells were generally trisected length-wise for sampling to avoid skewing of data due to hotspots of accumulation (Johnson and Hunt 2016). In each section, three samples were collected within a 1-foot radius from the top 8 inches of soil media and composited for nutrient analysis as previous research has shown the majority of runoff-borne pollutants to collect in the top 8 inches of soil media (Komlos and Traver 2012; Li and Davis 2008). One additional sample was collected from the top 2-feet at each of the three sample locations. The three samples were then composited into a single sample for nutrient analysis to compare concentrations in the top 2 ft to samples from the top 8 inches. Lastly, an additional sample was collected from the top 2 ft at each of the three sample locations. Those samples were then composited into a single sample for particle size analysis of the soil media.

Sample Analysis

Soil samples were analyzed for physical and chemical composition. Particle size distribution was analyzed following the ASTM hydrometer method at the NC State

University Stormwater Laboratory (ASTM International 2017). Soil sample bulk density was estimated using percent sand, silt, and clay and methods by Saxton and Rawls (2006). Soil samples were analyzed for total organic carbon (TOC), total nitrogen (TN), and total phosphorus (TP) at the Environmental Analysis Laboratory at NC State University using standard methods; Carbon:Nitrogen ratio (C:N), Mehlich-3 phosphorus (M3P), and phosphorus index (P-Index) were calculated. Mehlich-3 phosphorus is a common measure of bioavailable P in North Carolina, and is often reported as an index value (P-Index). Moreover, North Carolina BRC design requirements stipulate that BRC media installed in nutrient sensitive watersheds, such as the Jordan Lake watershed studied herein, be no higher than 30 (N.C. DEQ 2017a). As such, M3P was calculated from TP following guidance by Lammers and Bledsoe (2017). P-Index was then calculated from M3P (Hardy et al. 2014). While calculating M3P from TP and P-Index from M3P introduces uncertainty, both conversions are research based and the resulting P-Index values were within a reasonable range of previously reported P-Index values in North Carolina BRCs (Brown and Hunt 2011; Hunt et al. 2006, 2008; Line and Hunt 2009).

Data and Statistical Analysis

As the analysis herein is examining soil media on a temporal scale rather than a spatial scale, soil samples from the top 8 inches of soil media were averaged to offset potential skewing due to hotspots within individual bioretention cells. Previous research has shown hotspots of nutrient and sediment accumulation to occur near inlets or areas of localized depression and preferential flow paths (Johnson and Hunt 2016; Jones and Davis 2013; Muerdter et al. 2016). For categorical comparisons, BRCs were discretized into three individual groups based on age: (1) 1-5 years (n = 10), (2) 6-10 years (n = 8), and (3) 11+ years (n = 10).

Statistical analysis was performed using R statistical software. Bias corrected and accelerated bootstrapping was performed to determine the 95\% confidence interval about the median. Data were inspected for normality and log-normality and statistical comparisons utilized Wilcoxon signed rank non-parametric statistical analyses to compare differences between accumulation in the top 8 inches and the entire soil media column. Differences between groups were tested using pairwise Wilcoxon rank sum tests. Differences were considered significant at $\alpha = 0.05$.

Results and Discussion

Physical Characteristics

Of the 28 BRCs sampled, the textural classification of media in 22 was sand; the other six were loamy sand. Median percent sand, silt, and clay was 89%, 8%, and 3%, respectively, while mean percent sand, silt, and clay was 88 ± 4%, 8 ± 4%, and 3 ± 1%, respectively. As BRC media is designed to be sand-based, the lack of variation and high sand content in the results were expected.

BRC media specifications in North Carolina prescribe a particle size distribution of 75-85% sand, 8-15% fines, and 5-10% organic matter (N.C. DEQ 2017a). Previous guidance suggested up to 88% sand and 12% fines (N.C. DEQ 2009). The media sampled herein were sandier than NC standards. Twenty-two samples (78%) exceeded the current maximum sand threshold while four samples (14%) exceeded the current percent fines threshold. The greatest percentage of fines was two instances of 22% fines. The physical and chemical composition of bioretention media is a critical component of bioretention performance as it dictates the flow of water through the system and thus, the volume of runoff that can be captured for flood control and water quality treatment. Further, silt and clay content and mineralogy limits maximum permeability to provide

adequate contact time for pollutant removal and chemical sorption sites for additional treatment (Fassman-Beck et al. 2015).

Wardynski and Hunt (2012) examined 41 BRCs in North Carolina and found 71% of sampled medias to be outside the specified ranges for particle size, with 46% of samples being too sandy and 24% too fine. The results herein agree with these authors' assessment, particularly with respect to excess fines. It was anticipated that bioretention cells constructed in more impermeable watersheds would display a higher sand content in filter media as a study in Australia found a strong correlation between watershed imperviousness and sandier sediment in runoff (Perryman et al. 2011); however, neither the correlation nor differences in media physical characteristics and watershed imperviousness were statistically significant.

Previous research has demonstrated the effective capture of sediment and particulate matter at the surface of bioretention, typically within the top 8 inches of media (Asleson et al. 2009; Hatt et al. 2009a; Li and Davis 2008). Wardynski and Hunt (2012) found a significant increasing trend in media fines content with age, which they attributed to the development of media guidance in 2005 defining maximum fines content. In agreement with previous research, fines content in bioretention media significantly increased with age ($p = 0.023$; Figure 2; Table 2). This trend is likely due to a combination of previous hypotheses: (1) sedimentation and trapping of fines in runoff has resulted in the accumulation of additional fines within the media and (2) fines content in newly installed BRCs is lower due to more regimented specifications for media.

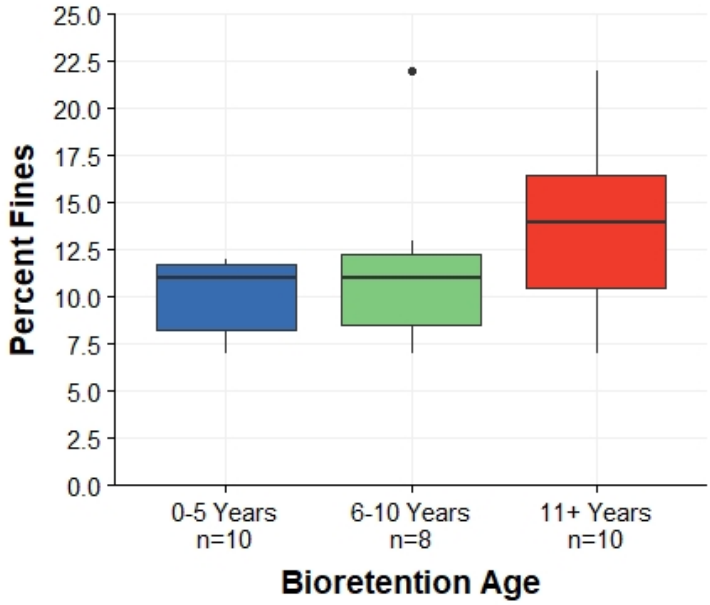

Figure 2. Fines content of bioretention media discretized by age of bioretention.

As the samples collected for particle size analysis in this study were taken from the top 2 ft of media, it is highly likely that the effect of accumulated sediment from the watershed would be diminished. Regardless, a significant trend in fines accumulation with age exists; however, while significant, the accumulation rate of fines is not alarming. The particle size distributions herein would not be deleterious to hydraulic functioning as the four BRCs with less than 85% sand at least 10-years-old and vegetated; mature vegetation has been shown to increase infiltration rates in bioretention, regardless of surface clogging (Hatt et al. 2009b; Virahsawmy et al. 2014). It is possible that accumulated fines could provide additional treatment opportunities for pollutant removal through the addition of sorption sites associated with clay mineralogy (Li and Davis 2016).

Carbon and Organic Matter

TOC content in the soil media ranged from 0.1 to 5.9% by weight. TOC did not vary with depth in the media as average organic carbon content in the top 8 inches of media and the top 2 ft were both 2.0 ± 1.4%. While TOC in bioretention media did not significantly differ with age, median TOC for age categories 1-3 was 8.2%, 15.6%, and 22.7%, respectively (Table 2Table 2. Summary statistics for analyzed characteristics for each age category.).

As organic matter is a prescribed element of bioretention media mix, OM content in sampled BRC media was estimated (Nelson and Sommers 1996) for comparison to NC DEQ (2017a) recommended specifications. Organic matter content ranged from 0.7 to 7.5% by weight with a median of 2.3% and a bootstrapped 95% confidence interval around the median of 1.6-3.6%. As it is estimated from TOC, OM content was higher in older BRCs, albeit not significantly. This increase in TOC and OM over time was expected, and is most likely due to vegetation senescence and the decomposition and routine replenishment of hardwood mulch as BRCs are maintained (Chen et al. 2013)

Organic matter, and thus carbon, is vital to the removal of metals, hydrocarbons, and nutrients (Hunt et al. 2012). In particular, carbon serves as an electron donor in BRC media for denitrification, which is needed to complete the nitrogen cycle and remove nitrogen from BRCs via N_2 gas (Knowles 1982). However, too much organic matter, and OM sourced from compost high in nitrogen and phosphorus, can result in the leaching of nutrients from BRCs (Clark and Pitt 2009; Hunt et al. 2006; Paus et al. 2014). As such, research-based design guidance recommends a BRC media to include 5% organic matter, by weight (Hunt et al. 2012; Peterson et al. 2015). Of the BRCs sampled herein,

only five were within 1% of the target threshold of 5% OM. The nine BRCs exceeding the target threshold contained less than 8% OM by weight.

Table 2. Summary statistics for analyzed characteristics for each age category.

Age Category	Statistic	% Sand	% Clay	% Silt	TOC (% by wt)	OM (% by wt)	C:N	TP (mg/kg)	M3P (mg/kg)	PI
0 - 5 Years	Mean	90	3	7	13.8	2.4	21	138.5	16.2	19
	Median	89	3	8	8.2	1.4	21	65.6	7.7	9
	Std. Dev.	2	1	3	13.4	2.3	6	156.3	18.3	21
	Min	88	2	3	4.3	0.7	12	45.9	5.4	6
	Max	93	4	10	42.9	7.4	32	466.8	54.6	64
6 - 10 Years	Mean	89	3	8	20.3	3.5	17	330.3	38.6	45
	Median	89	3	8	15.6	2.7	15	197.7	23.1	27
	Std. Dev.	5	2	4	11.8	2.0	5	284.6	33.3	39
	Min	78	1	2	6.5	1.1	12	126.2	14.8	17
	Max	93	6	16	40.6	7.0	26	952.5	111.4	130
11+ Years	Mean	86	4	10	23.8	4.1	15	301.4	35.3	42
	Median	86	4	10	22.7	3.9	17	312.9	36.6	43
	Std. Dev.	5	1	4	14.8	2.5	4	130.9	15.3	18
	Min	78	2	3	5.8	1.0	9	140.2	16.4	19
	Max	93	6	18	43.3	7.5	20	467.5	54.7	65

C:N ranged from 8.7 to 32.4 with a median of 17.6 and mean of 18.1 ± 5.7. The highest C:N, 32.4, was observed at a 5-year-old BRC planted with shrubs with 3 ft of media. The lowest C:N was observed at a 14-year-old BRC planted with trees with 4 ft of media. C:N significantly decreased in aging BRCs ($p = 0.007$; Figure 3; Table 2).

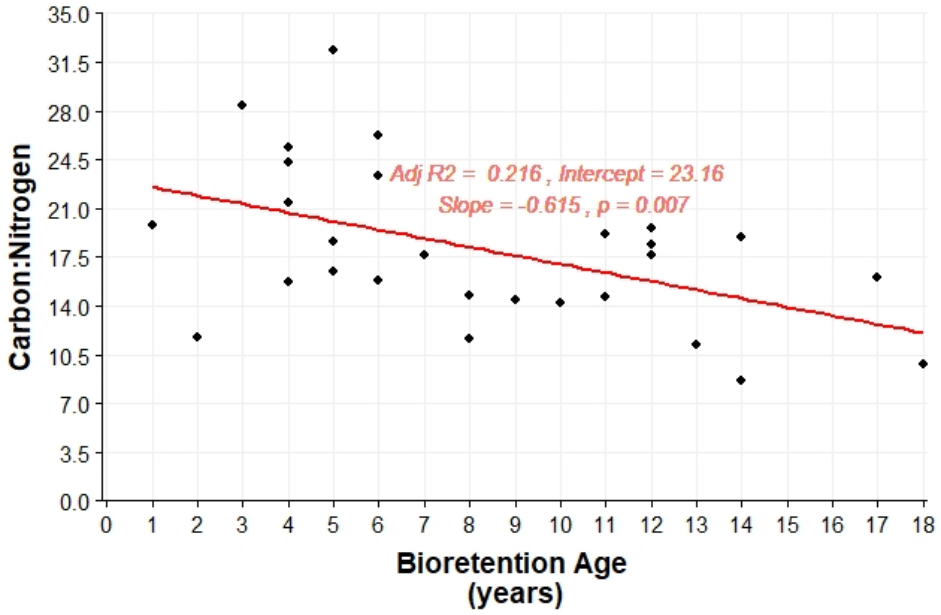

Figure 3. C:N ratio compared to age of sampled bioretention cells.

A decrease in C:N is not surprising, as (1) C:N in surface soils tends to decrease over time until reaching equilibrium with the C:N of microbes in the soil (Janssen 1996; Stevenson 1994) and (2) initial carbon content in the BRC media is quickly utilized in microbial respiration or leached from bioretention media following initial decomposition, particularly in mulched BRCs (Chen et al. 2013). As BRCs are typically vegetated and mulched, watershed inputs and vegetative senescence appears to be replacing carbon and resulting in a somewhat stabilized C:N in older BRCs.

Notably, C:N was anticipated to be greater than what was observed in older BRCs (median of age category 3 C:N = 17). McPhillips et al. (2018) measured C:N in the top 8 inches of a newly-retrofit BRC to be 21.8-25.4, while Willard et al. (2017) reported a maximum C:N at the surface layer of a seven-year-old BRC to be 44 with values in the top 12 inches of media ranging from 24-35; however, leaf compost in the BRC media and a 4-inch layer of mulch at the surface likely contributed to an elevated C:N. Few other

BRC studies have reported C:N values, but several authors do recommend BRC media to have a C:N greater than 20 to promote immobilization of nitrogen rather than mineralization (Chapin et al. 2011; Janssen 1996; McPhillips et al. 2018).

It is possible that routine maintenance of BRC vegetation, such as pruning and/or regularly mowing (with clipping removal), has exacerbated the decline in C:N. Previous research has recommended that vegetation in BRCs be left to grow and naturally decay in-place (Chen et al. 2013; Willard et al. 2017). This would likely replenish carbon in the upper layers of BRC media (Chen et al. 2013; Johnson and Hunt 2019).

That both carbon and C:N are decreasing also suggests that nitrogen is also being removed via nutrient uptake by plants, microbial assimilation, or coupled mineralization/nitrification and mobilization in subsequent storm events. It is likely that the degree of nitrogen loss is less than that of carbon assuming a priori that nitrogen accumulation rates are greater than that of carbon (due to nitrogen loads received in runoff, hence a decrease in C:N rather than an equilibrium). However, the rate at which C:N is decreasing is not substantial. The accumulation of nitrogen is the media is likely insignificant due to (1) mineralization/nitrification and (2) the weakness in sorption between nitrogen and the soil media (Li and Davis 2014).

Phosphorus

Total phosphorus concentrations significantly increased with age in sampled BRCs ($p<0.001$; Figure 4). TP concentrations ranged from 45.9-952.5 mg/kg with a median of 180.8 mg/kg and a mean of 251.5 ± 205.8 mg/kg. The bootstrapped 95% confidence interval around the median was 127.2-377.4 mg/kg. The highest concentration of TP was in an 8-year-old BRC with sod and 2 feet of media. The lowest TP concentration was at a 5-year-old BRC with 3 feet of media.

Median TP concentration in the top 8 inches of media was 193.4 mg/kg and in the top 2 feet was 125.5 mg/kg; however, differences were not significant (Figure 4). While previous research has shown the majority of TP to accumulate within the top 8 inches, the top 2 feet included the top 8 inches, minimizing "dilution."

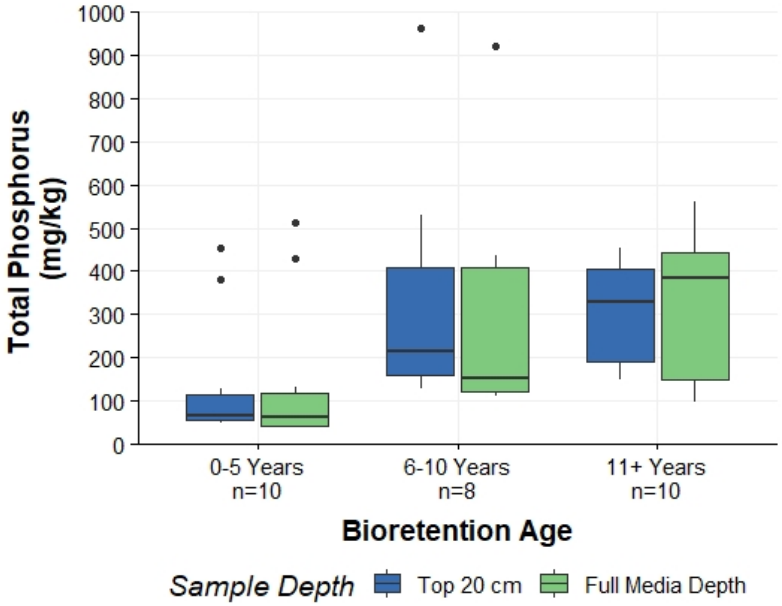

Figure 4. TP concentrations discretized by age group in sampled BRC media.

For comparison to current media specifications, total phosphorus concentrations were converted to a P-Index value following aforementioned methods by Lammers and Bledsoe (2017) and Hardy et al. (2014). P-Index values also significantly increased with age in sampled BRCs ($p<0.001$). P-Index values ranged from 6-130 with a median of 25 and a mean of 35 ± 28. Median P-Index in the top 8 inches of media was 9, 27, and 43 for age categories 1-3, respectively (Table 2). An increase in P-Index with age supports previous research demonstrating this trend in aging BRCs (Johnson and Hunt 2016, 2019). Conversely, Willard et al. (2017) reported reductions in TP in a BRC following 7

years of treatment, which they attributed to a flushing effect due to low cation exchange capacity in the filter media.

As the geochemical removal of dissolved phosphorus is driven by background concentration of P within media and the availability of sites for uptake (Li and Davis 2016; Lucas and Greenway 2008), knowing the current capacity of soil media allows prediction of future performance. As such, P-Index can be interpreted as an indicator of future sorption potential: a P-Index below 50 signifies that BRC media is low in phosphorus and has additional sorption capacity for future P loads, a P-Index between 50 and 100 indicates that the soil media is nearing sorption capacity and may be leaching P, and a P-Index greater than 100 indicates that a media is completely saturated and is leaching P (Hardy et al. 2014; Hunt et al. 2006). In earlier studies, BRCs with a low-medium P-Index have been shown to reduce P loads to receiving waters while high to very high P-Index media have been a net source of P (Hunt et al. 2006, 2008; Johnson and Hunt 2019; Line and Hunt 2009; Passeport et al. 2009).

Of the BRCs sampled herein, nine (32%) had a P-Index value in the high to very high range (Figure 5). It is highly likely that treatment capabilities for these eight BRCs are impaired, and the BRC with a P-Index of 130 is a source of phosphorus to receiving waters. It is promising that the majority of BRCs sampled, including the two eldest, remained in the low-medium P-Index range, indicating sustained P removal of runoff. However, 50% of older BRCs 10-years-old had P-Indexes greater than 50.

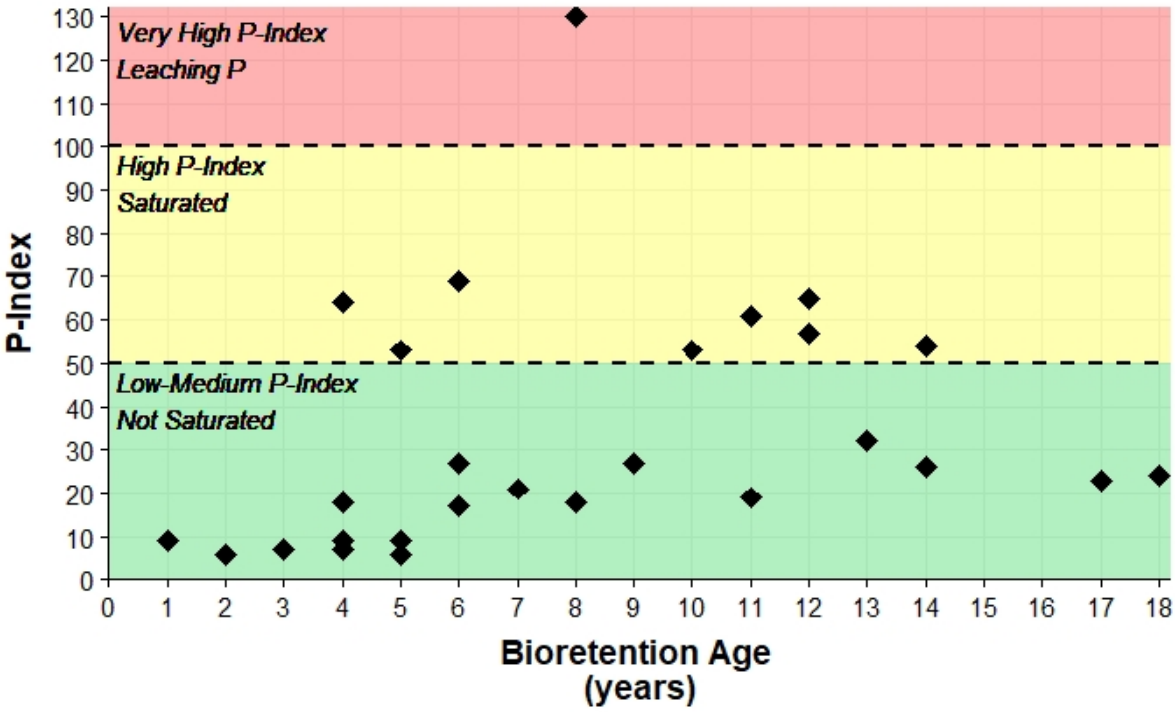

Figure 5. Phosphorus index values for sampled BRCs. Thresholds for phosphorus sorption capacity and potential leaching are indicated.

A key consideration when examining the P-Index values herein is the ability of deeper media to continue P sorption after the upper layers of filter media have reached saturation. Komlos and Traver (2012) examined orthophosphate removal in a rain garden over a nine-year monitoring period and noted at the end of monitoring that sorption capacity had only been reached in the top 4 inches of media while predicting an additional 20+ years of phosphorus treatment in the lower depth of media. A P-Index of 69 is estimated in the top inch of media from Muerdter et al.'s (2016) examination of media in a seven-year-old BRC while samples from 6 to 16 inches were estimated to have a P-Index value of 18, leading those authors to predict continued P sorption capacity in the deeper layers of media for the foreseeable future. As the aforementioned full column P-Index values in this study included the top 8-inches, the top layer likely skewed top 2 feet

P-Index values to be higher, and it is hypothesized that outside of the highest P-Index value reported, most BRCs included in this study should continue to provide long-term phosphorus treatment.

Conclusions and Recommendations

Bioretention media surveyed in this study displayed significant changes in physical and chemical characteristics with age. However, the changes detailed herein show promising results and suggest that with minimal maintenance, bioretention media should continue to provide treatment of stormwater runoff for prolonged periods of time. The following conclusions and recommendations are drawn from these results.

- Silt and clay content in BRC media significantly increased with age, most likely due to recent changes in media specifications and sedimentation of watershed sediment at the surface of BRCs.
    - While the amount of fines increased, 78% of sampled BRCs were sandier than NC DEQ (2017a) media specifications; thus, BRC media likely percolate sufficiently fast, thus providing hydrologic mitigation.
- The median C:N of media samples was 17.6 and significantly decreased with age, below the threshold for nitrogen immobilization (C:N = 20) suggested by McPhillips et al. (2018).
    - It is recommended that a high quality carbon amendment, such as saw dust or biochar, be added to BRC media. The frequency of this addition could be synchronized with the scraping of the surface layer when the latter cakes (Hatt et al. 2008; Johnson and Hunt 2016).

- Phosphorus is accumulating in BRC media; however, generally, P-Index values are less than threshold indicating saturation and possible leaching and phosphorus removal should be sustained as deeper depths of BRC media "pick up the slack" once surface layers saturate.
    - As P removal efficiency and leaching potential can be predicted via P-Indices, it is recommended that testing of media, including depths below 8 inches, should be performed every 5-10 years to ensure media is not reaching sorption capacity.

FLOATING TREATMENT WETLANDS

Introduction

Wet ponds have been utilized throughout North Carolina, humid regions of the United States, and the world. A wet pond includes “a permanent pool of water for removing pollutants and additional capacity above the permanent pool for detaining stormwater runoff.” (N.C. DEQ 2017a). The influent to the pond passes first into a forebay, where initial velocity reduction and particulate settling occurs. Inflow then passes into the main pond, where stormwater mixes with water in the permanent pool, diluting pollutant concentrations and allowing further settling to occur. The basin’s outlet structure controls outflow from the basin such that stormwater is stored and slowly released at pre-construction peak runoff rates. North Carolina design standards require a total storm drawdown in two to five days, such that outflow is slow enough to sufficiently attenuate the runoff flow, yet rapid enough that the basin is emptied before the next storm event (N.C. DEQ 2017a).

Wet ponds are an effective stormwater BMP for attenuating peak flow and thereby reducing downstream flooding; however, their efficacy in improving water quality has been inconsistent in research. For example, Mallin et al. (2002) evaluated the performance of three wet ponds in Wilmington, NC, and found highly variable pollutant removal rates. While one pond showed significant reductions in nearly all pollutants sampled, another pond had no significant removal of nutrients, and yet another actually showed increased nutrient concentrations in the outflow. Similarly, variable results were found in several other NC-based studies, including Borden et al. (1998).

Because ponds treat relatively large watersheds, any means of improving their performance has the potential for widespread effect, particularly in nutrient sensitive watersheds like the Jordan Lake watershed. One relatively proven retrofit of wet ponds is the addition of floating treatment wetlands (FTWs). FTWs are a hydroponic system that employ vegetated floating mats or trays to provide nutrient treatment in surface water settings. FTWs are quickly becoming a popular retrofit to lakes and wet detention ponds as they allow sedimentation, do not compromise existing ability to mitigate peak flows, do not require heavy equipment to install, and provide improved habitat and biodiversity. Moreover, they are a cost-effective and ecologically friendly alternative to a full-scale design/build retrofit to reduce loads from existing wet ponds.

Due to the low-cost/high-reward potential associated with FTWs, extensive research has explored their ecosystem services and treatment performance in recent years. While the preponderance of research features laboratory-based mesocosm studies, multiple field trials have demonstrated the effectiveness of FTWs as a value added retrofit for stormwater runoff (Borne 2014; Borne et al. 2013b; a; McAndrew et al. 2016; Schwammberger et al. 2019; Tharp et al. 2019; Winston et al. 2013).

Design

FTWs are a hydroponic system that employ vegetated floating mats or trays to provide nutrient treatment in surface water settings (Figure 6). At the surface, FTWs are buoyed using a proprietary material or simply from connected sealed barrels or other flotation devices. The body of FTWs is made to support the weight of vegetation and substrate for plants, which are placed within the mats or trays and cover the majority of the surface. Below water level, plant roots are suspended in the water column to

provide treatment. FTWs are held within a general area of a wet pond via tethering to a structure on the shore or by anchoring them to the bottom of the wet pond.

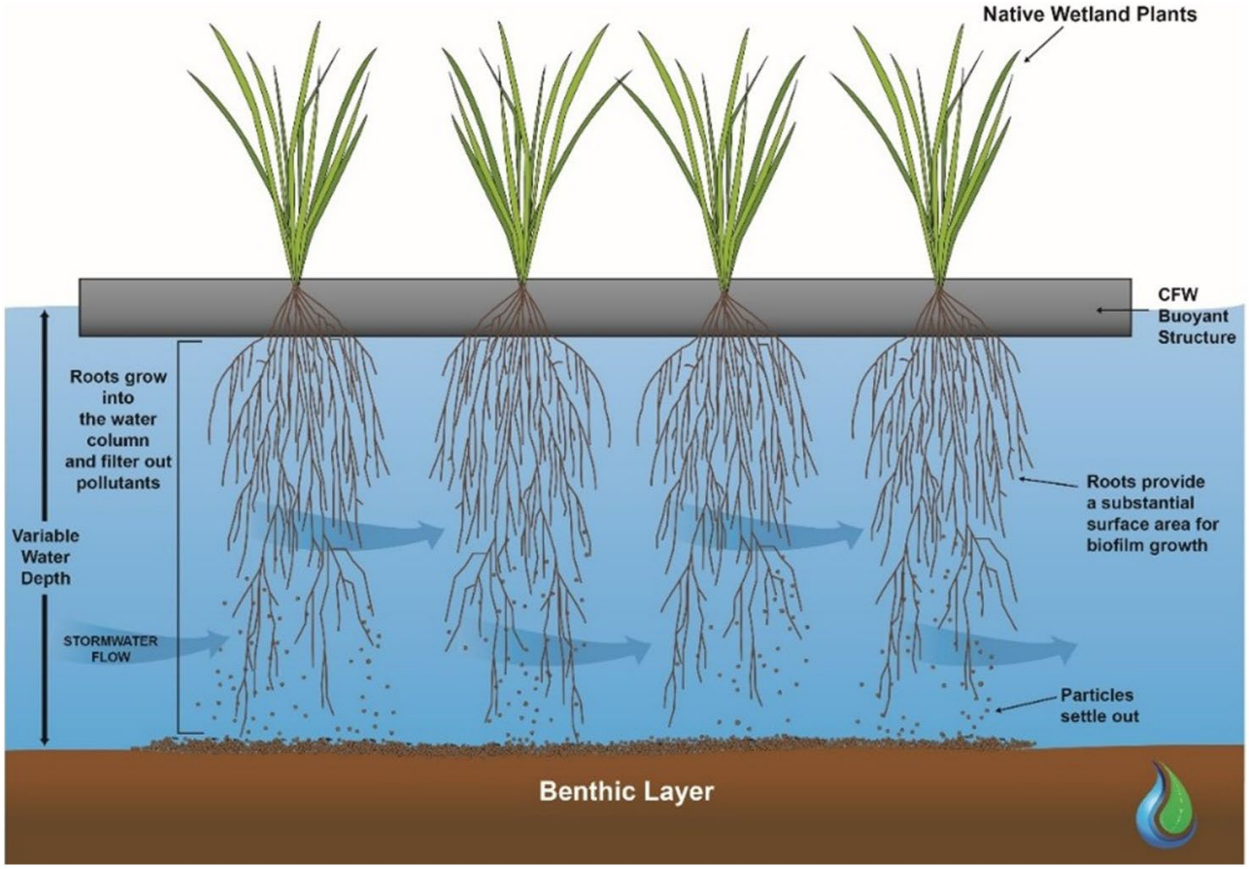

Figure 6. Example cross section of a floating treatment wetland (Lucke et al. 2019).

One of the driving forces behind the growing popularity of FTWs is their ease of construction and customization in design. A simple homemade design consisting of non-toxic plastic containers injected with expanding foam has been shown to provide adequate buoyancy and freeboard for plant roots to grow (Walker et al. 2017). Other successful experimental designs include sealed PVC frames with plastic netting (Figure 7). Typical construction materials include HDPE plastic, marine grade polystyrene foam, and PVC pipe.

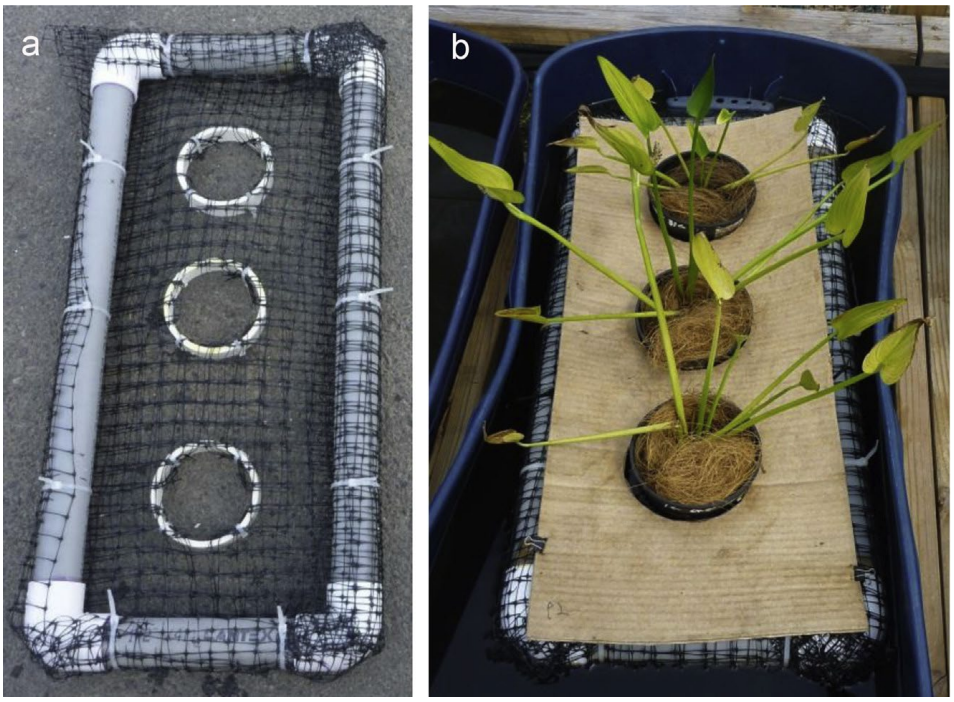

Figure 7. Example of a simple homemade FTW by Wang and Sample (2014).

However, FTWs are also offered as a proprietary device. Two popular versions of proprietary FTWs present in the literature are Beemats (Beemats LLC, New Smyrna Beach, FL, USA) and BioHaven® Floating Islands (Floating Island International, Inc., Shepard, MT, USA) (Lynch et al. 2015). Proprietary FTWs are often a popular choice as they are customizable and are designed to withstand varying conditions seen in field applications.

Pollutant Removal Mechanisms

FTW research has identified four main pathways for pollutant removal: (1) assimilation and nutrient uptake by vegetation planted on FTWs, (2) entrapment of particulate bound pollutants in biomass beneath FTWs, (3) sedimentation beneath FTWs, and (4) biogeochemical transformations occurring within the root zone of FTWs. Each process is depicted in Figure 8 and described in additional detail below.

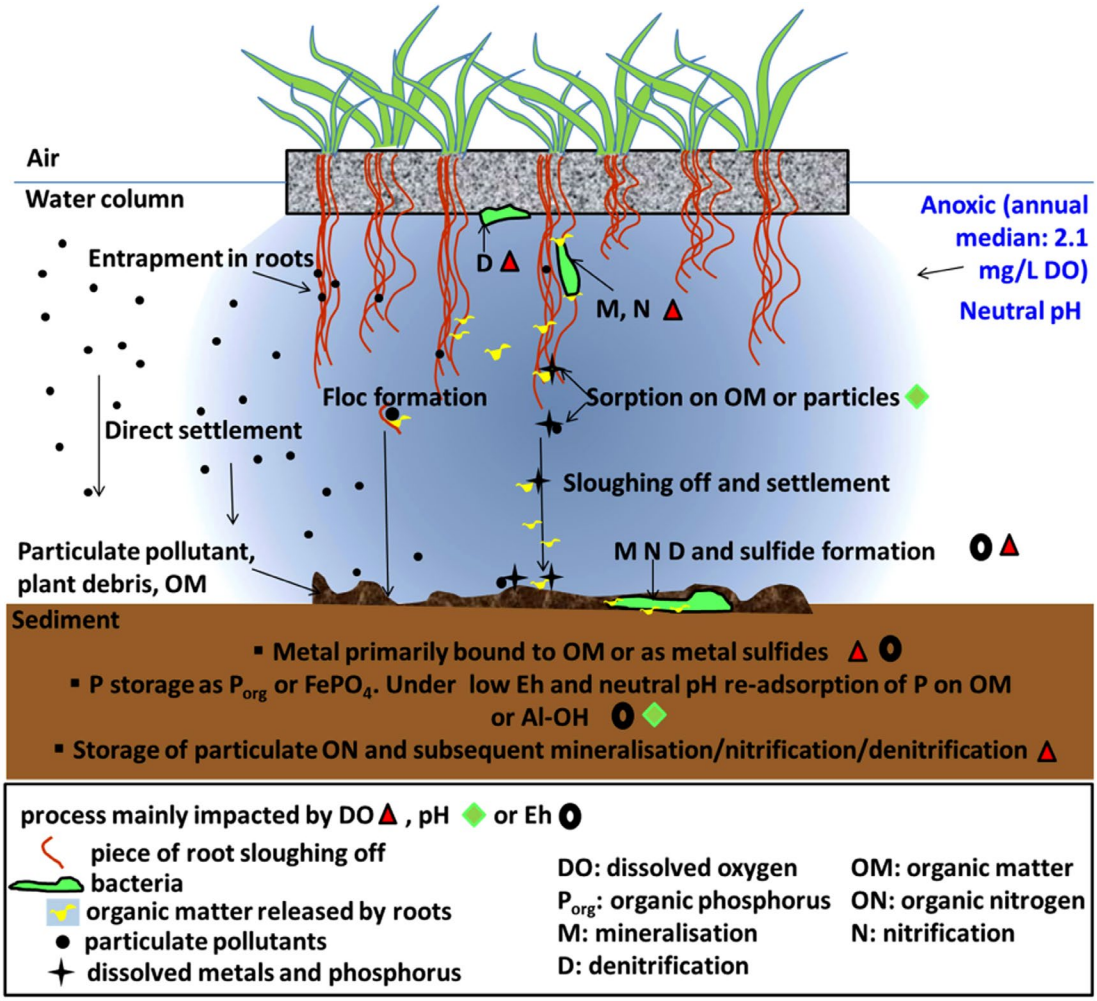

Figure 8. Pollutant removal mechanims of FTWs. Figure from Borne et al. (2015).

Assimilation and Plant Uptake

Assimilation is the biotic uptake of nutrients by plants, bacteria, fungi, and microorganisms and is a major factor in the sequestration of nutrients in FTWs. Plant species such as soft rush, pickerelweed, and yellow flag have been shown to account for upwards of 90% of N removal in FTWs (Keizer-Vlek et al. 2014; Spangler et al. 2019). Some studies have even shown some degree of removal for phosphorus and some metals species (Tanner and Headley 2011).

Biomass is typically examined as a surrogate for nutrient uptake by vegetation on FTWs. As plant roots are suspended in the water column rather than in soil, research has

compared the accumulation of nutrients in biomass “above mat” (e.g., shoots) and “below mat” (e.g., roots). Nutrient accumulation is relatively evenly distributed between above and below mat biomass, but has been shown to vary between sites and plant species (Tanner and Headley 2011; Winston et al. 2013). As the promotion of plant growth is a key design feature in FTWs, opportunities for treatment of N via assimilation are plentiful.

Entrapment

The suspension of plant roots within the water column also provides treatment opportunities through the entrapment of fine particulates to the “sticky” biofilms that grow on the roots. Not only does entrapment remove fine suspended sediment, and what may be colloidal-bound to them, the entrapped fine sediments provide sorption opportunities for phosphorus and metals depending upon their mineralogy (Borne et al. 2013a; Tanner and Headley 2011). A key consideration is the needed residence time for entrapment to occur and depth of roots. Short root depths and high flows will reduce the degree of treatment. Moreover, particularly large flows can wash off sediments that have already been trapped within the root zone (Borne et al. 2013a).

Sedimentation

Sedimentation is also a key pollutant removal mechanism for FTWs. Wet ponds typically include a feature designed to promote sedimentation, forebays, to slow runoff and allow larger sediments to settle before entering the main body of the wet pond. FTWs also promote the stilling of water flow by providing an obstruction (roots) within the water column. Further, the physical barrier of FTWs themselves may also provide some degree of hydraulic resistance. As such, larger particles that may have bypassed treatment within a forebay can be settled near the location of FTWs. This mechanism has been well

documented in prior studies showing reductions in TSS and sediment build up beneath FTWs (Tanner and Headley 2011; Winston et al. 2013).

Biogeochemical Processes

Within the treatment area of a FTW, denitrification can occur within the anaerobic zone created by microbiomes on plant roots or within biofilms. Urakawa et al. (2017) suggested that FTWs provide additional habitat for microbiomes that are linked to the oxic-anoxic gradient that provides the environment for rhizospheric denitrification. In this area, denitrifying bacteria populations can be 10-100 times greater than other areas (Knowles 1982). During respiration, O_2 can be diffused into the anoxic rhizosphere allowing a coupling of nitrification and denitrification.

Research has shown that decomposing periphyton form tight, almost impermeable, mats, called biofilms, on exposed surfaces (e.g., structural components of a FTW) (McAndrew et al. 2016). These biofilms also support communities of denitrifying bacteria due to low O_2 levels resulting from limited permeability of the biofilm.

Performance Review

TSS and Particulate Matter

TSS and particulate matter is primarily removed by FTWs via filtration and trapping in the root zone. FTWs on a partitioned pond in New Zealand significantly reduced TSS concentrations compared to the partition without FTWs (Borne et al. 2013a). Walker et al. (2017) investigated sediment removal from an FTW during a two-year study in Australia and noted a significant 81% reduction in TSS. Two pre- and post-FTW retrofit studies in North Carolina attributed significantly improved sediment removal at one pond to particulate settling due to roots slowing velocities (Winston et al. 2013); the authors

suggested insignificant removal of TSS at the second pond was due to short-circuiting of flow around the FTWs and a low coverage area (9%).

Metals

Previous research has shown the ability of FTW retrofits to effectively remove selective metals species when compared to stand-alone ponds. Borne et al. (2013a) conducted a side-by-side comparison of copper and zinc removal from a wet pond with and without FTWs in New Zealand. The pond with FTWs exhibited a 21% and 16% greater mass removal rate of total copper and total zinc, respectively. Increased removal of metals via FTWs was attributed to anoxic conditions beneath FTWs, flocculation of particulates, and the formation of metal sulfides, particularly during warmer summer months (Borne et al. 2014). Garbett (2005) noted drastic reductions in iron concentrations as a result of the inclusion of the FTW retrofit on a eutrophic reservoir in the United Kingdom.

Mesocosm experiments by Tanner and Headley (2011) describe a 6-fold decrease in copper and additional substantial reductions in zinc through the incorporation of FTWs. The authors concluded that removal was likely enhanced by the bioactive compounds released from FTW plant roots or changing physiochemical conditions in the water column beneath the FTWs.

Nutrients

Nutrient removal via FTWs in research is highly variable, but generally demonstrates the overall ability of FTWs to improve wet pond nutrient removal. Garbett (2005) installed an FTW on a eutrophic reservoir in the United Kingdom and reported substantial reductions in orthophosphate during late spring to mid-autumn, but did not observe a change in nitrate concentrations (Garbett 2005). Walker et al. (2017) also noted

insignificant changes in TN concentrations in a study of FTW performance in Australia, but did find significant 52% reductions in TP concentrations in a wet pond. The authors suggested that re-aeration zones should be included with FTWs to improve nitrogen removal.

A study in Orlando, FL, reported TP and orthophosphate concentration reductions of 47.7% and 79%, respectively, for a wet pond with 9% surface area coverage by FTWs (Chang et al. 2013). Further, nitrogen species were reduced by 15.7%, 20.6%, and 51.1% for TN, NO3-N, and TAN, respectively. The authors attributed phosphorus removal to improved sedimentation via the root zone and nitrogen removal to plant uptake (Chang et al. 2013).

In one of the first studies of its kind, Winston et al. (2013) examined the pollutant removal performance of two wet ponds in North Carolina pre- and post-retrofit with FTWs. FTW surface area coverage for the two ponds was 9% and 18%. The former studied resulted in no significant differences in pollutant removal while the latter significantly reduced TP and orthophosphate.

Deployment

The range of coverage area in FTW research has been a central topic of discussion as field studies have generally not be able to reproduce the removal rates observed in mesocosm studies due to smaller coverage area. Consequently, up to a 50% surface area coverage is recommended in design guidance and crediting documents, but at least 10% surface area coverage is required to receive additional nutrient and sediment removal in the Chesapeake Bay (Lane et al. 2016).

Table 3. Chesapeake Bay crediting for FTWs. Modified from Lane et al. (2016)

Pollutant	FTW Percent Surface Area Coverage in Pond				
	10%	20%	30%	40%	50%
TN	0.8%	1.7%	2.5%	3.3%	4.1%
TP	1.6%	3.3%	4.9%	6.5%	8.0%
TSS	2.3%	4.7%	7.0%	9.2%	11.5%

However, merely prescribing a percentage coverage area may not provide additional treatment. While increasing coverage area has been shown to increase some pollutant removal, a key consideration is forcing water to flow through the root zone. Without strategic placement of FTWs, preferential flow can lead to polluted water bypassing treatment by short-circuiting FTWs. Khan et al. (2013) and Borne et al. (2015) recommend installing FTWs perpendicular to flow and across the entire width of directional flow to prevent short-circuiting. Further, Borne et al. (2015) recommend installing FTWs outside of the forebay of a wet pond to avoid saturating the roots with particulate matter that may be treated by the forebay alone.

An additional key design and deployment consideration is the depth of water available. If ample depth is not given within the water column, FTWs may become rooted to the pond bed and lose their freedom of motion and may become flooded and damaged as water levels rise during large events. FTW root depths up to 4.5 feet have been observed in the literature (De Stefani et al. 2011).

Borne et al. (2015) recommend that FTWs be anchored to desired locations and not allowed to float freely. Anchoring FTWs will ensure they remain perpendicular to flow and prevent FTWs from becoming rooted into the banks of wet ponds. Additionally, tethering FTWs to an outlet structure or posts outside of a wet pond will prevent FTWs from flipping or breaking loose during large storm events.

Maintenance requirements for FTWs are relatively minimal, albeit important. Borne et al. (2015) recommend a minimum of quarterly maintenance to ensure FTWs are operating as intended. More frequent maintenance may be needed during establishment as plants develop. As plants are the governing treatment mechanism, plant survival and protection from predation is important. Unsuccessful vegetation should be removed and replaced at the start of each growing season. Invasive species and small trees should be removed.

Future Work Needed

To date, limited field studies examining pre- and post-FTW retrofit pollutant removal have been conducted. Mesocosm studies have provided an important understanding of the dynamics behind FTW treatment, but the degree of treatment provided in mesocosm studies has yet to translate to full-scale field applications. Additional direct comparisons to baseline treatment performance of wet ponds are needed to accurately credit FTWs.

Other than a prescribed surface area cover, research has suggested strategies on the placement of FTWs within wet ponds. Previous research has, importantly, studied FTW's that were somewhat haphazardly placed (Chang et al. 2013; Winston et al. 2013). Now understanding that that FTW's function best when flow is forced to pass through (or under) their dangling roots, research is needed to test hypotheses on strategic placement. For example, 'protective rings' of FTW's around outlet structures may expose the most treatment to runoff or locating FTW's immediately downstream of the forebay may prove most effective.

Overall, floating treatment wetlands are an increasingly popular retrofit to improve wet pond nutrient removal, particularly because of their ease of installation and relatively low-cost. Nutrient removal appears varied and dependent upon providing enough coverage and proper placement.

SAND FILTER STUDY

Introduction

Sand filters are a commonly used stormwater control measure (SCM) in North Carolina. They are often implemented in highly impervious areas to help municipalities comply with regulations. The North Carolina Department of Environmental Quality (NC DEQ) has identified this technology as a primary SCM or device that can treat runoff from built-upon-areas for water quality without the need of additional SCMs. Due to a lack of North Carolina data, NC DEQ used research from sites located in Florida, Maryland, New Hampshire, and Virginia to establish pollutant removal credits for sand filters (N.C. DEQ 2017b). There is a concern these data are not reflective of North Carolina sand filters and may skew effluent pollutant loads leading to violations of stormwater regulations. Four sand filters in Fayetteville and Greensboro, North Carolina are currently being monitored for water quality and hydrology by NC State University (NCSU). NCSU conducted a column study to identify placeholder effluent credits until monitoring of these four sand filters has been completed.

Methodology

NCSU constructed nine 3 ft tall columns of 4 in SCH 40 PVC pipe. Each column was filled with 18 in of ASTM C33 sand and 3 in of ASTM No. 78 stone above and below the media to prevent preferential flow (Blecken et al. 2010; Lucas and Greenway 2011; N.C. DEQ 2017a). The base of each column was fitted with a cap and sample tube allowing outflow to be collected in a bucket beneath each column (Figure 9). Prior to each trial, these buckets were washed to prevent cross-contamination.

Figure 9. Assembled sand filter columns.

Storm events were simulated using runoff collected from a NCSU parking lot using a contraption developed by NCSU (Figure 10). The runoff was applied to the columns within 48 hours after each storm event. To mimic typical ponding depths, 1 ft of stormwater was applied to three columns and 2 ft of stormwater were applied to another three columns. The remaining three columns received 1 ft of deionized water that served as the study's control.

Figure 10. NCSU runoff collection device.

Prior to application, influent samples were collected from the runoff and deionized water. Composite effluent samples were collected within 24 hours after each trial to mimic typical drawdown periods for sand filters (N.C. DEQ 2017b). Each sample was analyzed at the NCSU Environmental Analysis Laboratory (EAL) for the following parameters: total suspended solids (TSS), total kjeldahl (TKN), nitrate/nitrite nitrogen ($NO_{2,3}$-N), total ammoniacal nitrogen (TAN), and total phosphorus (TP) (Table 4). Total nitrogen (TN) was calculated as the sum of TKN and $NO_{2,3}$-N. For analyses, a value of one-half the reporting limit was used for concentrations reported below the minimum detection limit (Gilbert 1987). SAS version 9.4™ software was used to determine if the data followed a normal distribution, and descriptive statistics were used to characterize each treatment. Meaningful statistical analyses were not performed due to the limited number of trials.

However, plots showing the effluent concentrations with standard errors were used to make inferences about the data.

Table 4. EAL analysis methods and detection limits.

Analyte	Method	Detection Limit (mg/L)
TKN	EPA Method 351.2	0.03
TAN	Standard Method 4500-NH3 G	0.01
$NO_{2,3}$-N	EPA Method 353.2	0.01
TP	Standard Method 4500-P F	0.03
TSS	Standard Method 2540B	0.50

Results

A total of eight trials were conducted; however, two of the trials were omitted from the analyses. Influent data collected from these trials were not reflective of typical North Carolina parking lots (Passeport and Hunt 2009), and the researchers believe including these data would skew the results. Additionally, the researchers discovered the deionized water system used throughout the study was failing, and these data were not used for comparison between the columns receiving stormwater runoff. Refer to Appendix A for the raw data from each trial.

Data were uniformly non-normal. Median effluent concentrations for the stormwater replicates and trials are summarized in Table 5. Differences between the TP concentrations throughout the trials were minimal. TN concentrations ranged from 0.50 to 1.43 mg/L while TSS concentrations were between 0.24 to 7.69 mg/L. The overall median effluent TN, TP, and TSS concentrations were 0.04, 0.73, and 2.17 mg/L respectively. These concentrations are less than the current credits allocated by NC DEQ (2017b); however more data is needed to conducted statistical analyses necessary to identify robust placeholder credits.

Table 5. Median effluent concentrations for stormwater replicates and trials.

Trial	Ponding Depth (ft)	TKN (mg/L)	NH_3-N (mg/L)	NO_3-N (mg/L)	TP (mg/L)	TN (mg/L)	TSS (mg/L)
1	1	0.47	0.12	0.13	0.02	0.58	2.00
	2	0.49	0.14	0.03	0.02	0.51	1.83
2	1	0.78	0.09	0.03	0.04	0.78	1.83
	2	1.10	0.09	0.06	0.04	1.18	2.34
3	1	0.79	0.11	0.03	0.03	0.82	0.25
	2	0.86	0.14	0.01	0.02	0.86	0.25
4	1	0.55	0.05	0.01	0.06	0.57	3.92
	2	0.57	0.07	0.01	0.02	0.60	3.48
5	1	0.49	0.08	0.01	0.04	0.50	5.10
	2	0.66	0.09	0.01	0.06	0.68	7.69
6	1	0.57	0.10	0.71	0.02	1.37	1.77
	2	0.53	0.10	0.84	0.04	1.43	2.75
	Median	0.57	0.10	0.03	0.04	0.73	2.17

Figure 11 through Figure 13 show the median TP, TN, and TSS concentrations with the respective standard error bars and ponding depth for trials one through six. It can be inferred from these plots that if the standard error bars overlap ponding depth and or the trials do not impact effluent concentrations. Figure 11 indicates that after Trial 2, ponding depth appears to impact effluent TP concentrations. Additionally, it can be suggested there are statistical differences between the trials. Figures Figure 12 and Figure 13 indicate ponding depth does not impact effluent TN or TSS concentrations and there are no statistical differences between the trials. However, these inferences should be confirmed through statistical analyses using more data (i.e. 15 trials).

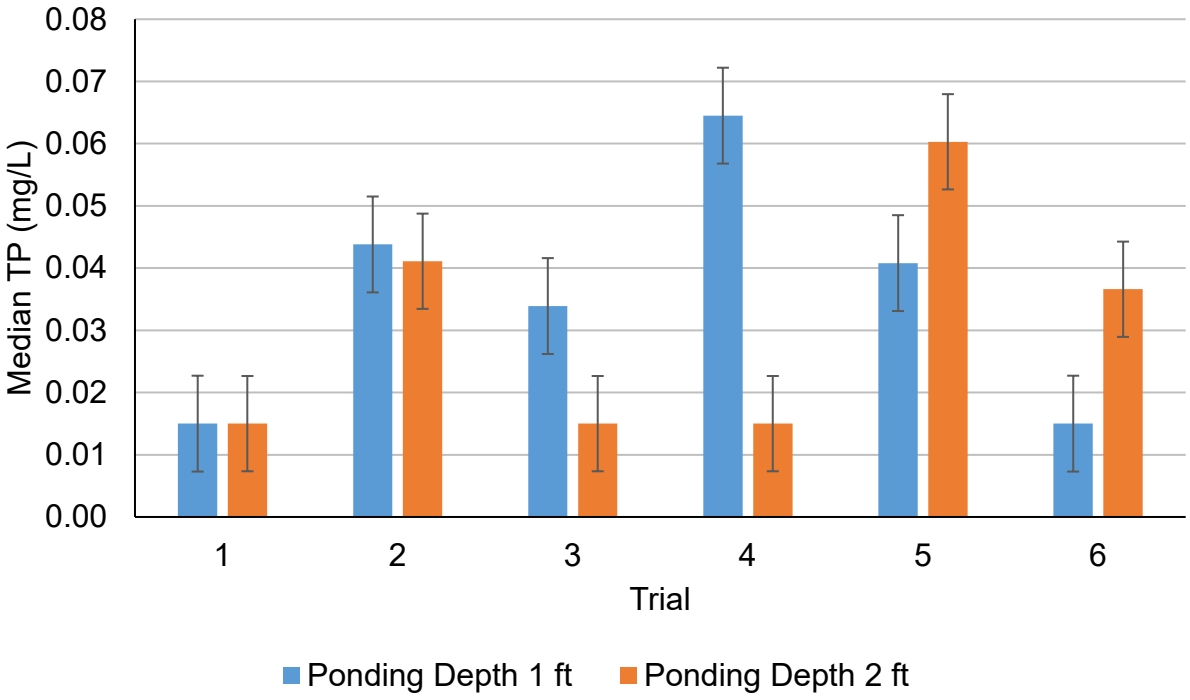

Figure 11. Median effluent TP concentrations with standard error.

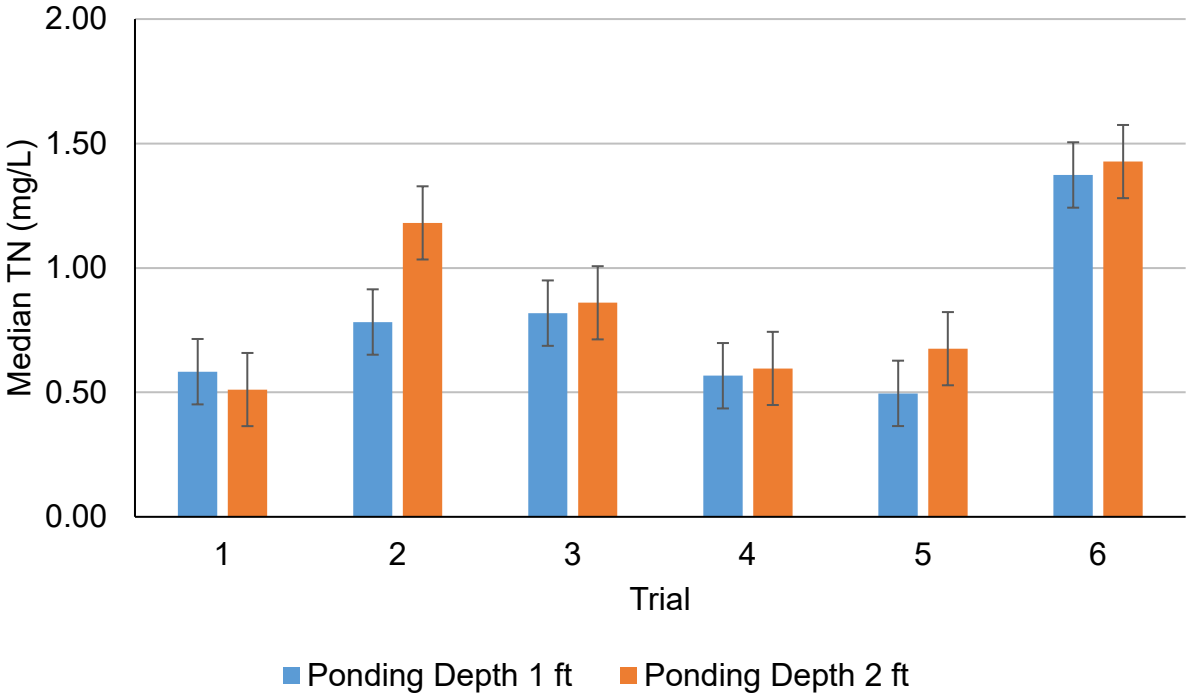

Figure 12. Median effluent TN concentrations with standard error.

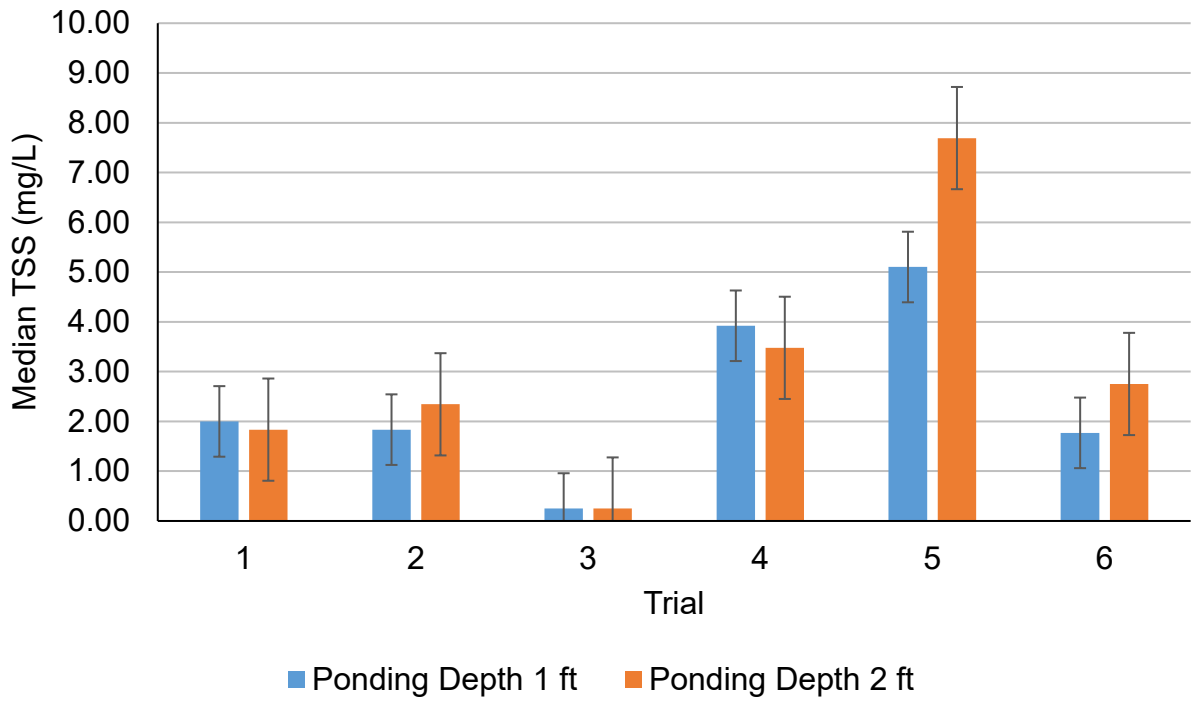

Figure 13. Median effluent TSS concentrations with standard error.

Conclusions

NCSU designed and constructed nine sand filter columns reflective of field scale sand filters. Eight trials were conducted throughout the study; however, trials seven and eight were excluded from the analyses. Influent concentrations from these trials were not reflective of typical parking lots (Passeport and Hunt 2009), and the researchers believe including these data would skew the results. Due to the lack of available data statistical analyses were not performed. Median effluent TP, TN, and TSS concentrations were 0.04, 0.73, and 2.17 mg/L, respectively. These concentrations are lower than the current credits allocated by NC DEQ (2017b). However, the researchers recommend further trials and statistical analyses are conducted to identify robust placeholder credits for sand filters.

REFERENCES

Asleson, B. C., Nestingen, R. S., Gulliver, J. S., Hozalski, R. M., and Nieber, J. L. (2009). "Performance Assessment of Rain Gardens." *JAWRA Journal of the American Water Resources Association*, 45(4), 1019–1031.

ASTM International. (2017). *ASTM D7928-17 Standard Test Method for Particle-Size Distribution (Gradation) of Fine-Grained Soils Using the Sedimentation (Hydrometer) Analysis*. West Conshohocken, PA.

Blecken, G.-T., Zinger, Y., Deletić, A., Fletcher, T. D., Hedström, A., and Viklander, M. (2010). "Laboratory study on stormwater biofiltration: Nutrient and sediment removal in cold temperatures." *Journal of Hydrology*, 394(3–4), 507–514.

Borden, R. C., Dorn, J. L., Stillman, J. B., and Liehr, S. K. (1998). "Effect of In-Lake Water Quality on Pollutant Removal in Two Ponds." *Journal of Environmental Engineering*, 124(8), 737–743.

Borne, K. E. (2014). "Floating treatment wetland influences on the fate and removal performance of phosphorus in stormwater retention ponds." *Ecological Engineering*, Elsevier B.V., 69, 76–82.

Borne, K. E., Fassman-Beck, E. A., and Tanner, C. C. (2014). "Floating Treatment Wetland influences on the fate of metals in road runoff retention ponds." *Water Research*, Elsevier Ltd, 48, 430–442.

Borne, K. E., Fassman-Beck, E. A., Winston, R. J., Hunt, W. F., and Tanner, C. C. (2015). "Implementation and Maintenance of Floating Treatment Wetlands for Urban Stormwater Management." *Journal of Environmental Engineering*, 141(11), 04015030.

Borne, K. E., Fassman, E. A., and Tanner, C. C. (2013a). "Floating treatment wetland retrofit to improve stormwater pond performance for suspended solids, copper and zinc." *Ecological Engineering*, Elsevier B.V., 54, 173–182.

Borne, K. E., Tanner, C. C., and Fassman-Beck, E. A. (2013b). "Stormwater nitrogen removal performance of a floating treatment wetland." *Water Science and Technology*, 68(7), 1657–1664.

Brown, R. A., and Hunt, W. F. (2011). "Underdrain Configuration to Enhance Bioretention Exfiltration to Reduce Pollutant Loads." *Journal of Environmental Engineering*, 137(11), 1082–1091.

Chang, N. Bin, Xuan, Z., Marimon, Z., Islam, K., and Wanielista, M. P. (2013). "Exploring hydrobiogeochemical processes of floating treatment wetlands in a subtropical stormwater wet detention pond." *Ecological Engineering*, Elsevier B.V., 54, 66–76.

Chapin, F. S., Matson, P. A., and Vitousek, P. (2011). *Principles of Terrestrial Ecosystem Ecology*. Springer-Verlag, New York, NY.

Chen, G., Zhu, H., and Zhang, Y. (2003). “Soil microbial activities and carbon and nitrogen fixation.” *Research in Microbiology*, 154(6), 393–398.

Chen, X., Peltier, E., Sturm, B. S. M., and Young, C. B. (2013). “Nitrogen removal and nitrifying and denitrifying bacteria quantification in a stormwater bioretention system.” *Water Research*, Elsevier Ltd, 47(4), 1691–1700.

Clark, S., and Pitt, R. (2009). “Storm-Water Filter Media Pollutant Retention under Aerobic versus Anaerobic Conditions.” *Journal of Environmental Engineering*, 135(5), 367–371.

Dunn, R. M., Mikola, J., Bol, R., and Bardgett, R. D. (2006). “Influence of microbial activity on plant–microbial competition for organic and inorganic nitrogen.” *Plant and Soil*, 289(1–2), 321–334.

Fassman-Beck, E., Wang, S., Simcock, R., and Liu, R. (2015). “Assessing the Effects of Bioretention’s Engineered Media Composition and Compaction on Hydraulic Conductivity and Water Holding Capacity.” *Journal of Sustainable Water in the Built Environment*, 1(4), 04015003.

Garbett, P. (2005). “An Investigation into the Application of Floating Reed Bed and Barley Straw Techniques for the Remediation of Eutrophic Waters.” *Water and Environment Journal*, 19(3), 174–180.

Gilbert, R. O. (1987). “Data Near Detection Limits.” *Statistical Methods for Environmental Pollution Monitoring*, John Wiley and Sons, Inc., New York, NY, 177–185.

Hardy, D. H., Tucker, M. R., and Stokes, C. E. (2014). *Crop Fertilization Based on North Carolina Soil Tests*. *Circular No. 1*, Raleigh, NC.

Hatt, B. E., Fletcher, T. D., and Deletic, A. (2008). “Hydraulic and Pollutant Removal Performance of Fine Media Stormwater Filtration Systems.” *Environmental Science & Technology*, 42(7), 2535–2541.

Hatt, B. E., Fletcher, T. D., and Deletic, A. (2009a). “Hydrologic and pollutant removal performance of stormwater biofiltration systems at the field scale.” *Journal of Hydrology*, Elsevier B.V., 365(3–4), 310–321.

Hatt, B. E., Fletcher, T. D., and Deletic, A. (2009b). “Pollutant removal performance of field-scale stormwater biofiltration systems.” *Water Science & Technology*, 59(8), 1567.

Hunt, W. F., Davis, A. P., and Traver, R. G. (2012). “Meeting Hydrologic and Water Quality Goals through Targeted Bioretention Design.” *Journal of Environmental Engineering*, 138(6), 698–707.

Hunt, W. F., Jarrett, A. R. R., Smith, J. T., and Sharkey, L. J. (2006). “Evaluating Bioretention Hydrology and Nutrient Removal at Three Field Sites in North Carolina.” *Journal of Irrigation and Drainage Engineering*, 132(6), 600–608.

Hunt, W. F., Smith, J. T., Jadlocki, S. J., Hathaway, J. M., and Eubanks, P. R. (2008). “Pollutant Removal and Peak Flow Mitigation by a Bioretention Cell in Urban Charlotte, N.C.” *Journal of Environmental Engineering*, 134(5), 403–408.

Janssen, B. H. (1996). “Nitrogen mineralization in relation to C:N ratio and decomposability of organic materials.” *Plant and Soil*, 181(1), 39–45.

Johnson, J. P., and Hunt, W. F. (2016). “Evaluating the spatial distribution of pollutants and associated maintenance requirements in an 11 year-old bioretention cell in urban Charlotte, NC.” *Journal of Environmental Management*, 184(2), 363–370.

Johnson, J. P., and Hunt, W. F. (2019). “A Retrospective Comparison of Water Quality Treatment in a Bioretention Cell 16 Years Following Initial Analysis.” *Sustainability*, 11(7).

Jones, P. S., and Davis, A. P. (2013). “Spatial Accumulation and Strength of Affiliation of Heavy Metals in Bioretention Media.” *Journal of Environmental Engineering*, 139(4), 479–487.

Keizer-Vlek, H. E., Verdonschot, P. F. M., Verdonschot, R. C. M., and Dekkers, D. (2014). “The contribution of plant uptake to nutrient removal by floating treatment wetlands.” *Ecological Engineering*, Elsevier B.V., 73, 684–690.

Khan, S., Melville, B. W., and Shamseldin, A. (2013). “Design of Storm-Water Retention Ponds with Floating Treatment Wetlands.” *Journal of Environmental Engineering*, 139(11), 1343–1349.

Knowles, R. (1982). “Denitrification.” *Microbiological Reviews*, 46(1), 43–70.

Komlos, J., and Traver, R. G. (2012). “Long-Term Orthophosphate Removal in a Field-Scale Storm-Water Bioinfiltration Rain Garden.” *Journal of Environmental Engineering*, 138(10), 991–998.

Lammers, R. W., and Bledsoe, B. P. (2017). “What role does stream restoration play in nutrient management?” *Critical Reviews in Environmental Science and Technology*, Taylor & Francis, 47(6), 335–371.

Lane, S., Sample, D., Lazur, A., Winston, R., Streb, C., Ferrier, D., Schueler, T., Lane, C., and Wood, D. (2016). *Recommendations of the Expert Panel to Define Removal Rates for Floating Treatment Wetlands in Existing Wet Ponds*.

Li, H., and Davis, A. P. (2008). “Heavy Metal Capture and Accumulation in Bioretention Media.” *Environmental Science & Technology*, 42(14), 5247–5253.

Li, J., and Davis, A. P. (2016). “A unified look at phosphorus treatment using bioretention.” *Water Research*, Elsevier Ltd, 90, 141–155.

Li, L., and Davis, A. P. (2014). “Urban Stormwater Runoff Nitrogen Composition and Fate in Bioretention Systems.” *Environmental Science & Technology*, 48(6), 3403–3410.

Line, D. E., and Hunt, W. F. (2009). “Performance of a Bioretention Area and a Level Spreader-Grass Filter Strip at Two Highway Sites in North Carolina.” *Journal of Irrigation and Drainage Engineering*, 135(2), 217–224.

Lucas, W. C., and Greenway, M. (2008). “Nutrient Retention in Vegetated and Nonvegetated Bioretention Mesocosms.” *Journal of Irrigation and Drainage Engineering*, 134(5), 613–623.

Lucas, W. C., and Greenway, M. (2011). “Hydraulic Response and Nitrogen Retention in Bioretention Mesocosms with Regulated Outlets: Part I-Hydraulic Response.” *Water Environment Research*, 83(8), 692–702.

Lucke, T., Walker, C., and Beecham, S. (2019). “Experimental designs of field-based constructed floating wetland studies: A review.” *Science of the Total Environment*, Elsevier B.V., 660, 199–208.

Lynch, J., Fox, L. J., Owen, J. S., and Sample, D. J. (2015). “Evaluation of commercial floating treatment wetland technologies for nutrient remediation of stormwater.” *Ecological Engineering*, Elsevier B.V., 75, 61–69.

Mallin, M. A., Ensign, S. H., Wheeler, T. L., and Mayes, D. B. (2002). “Pollutant Removal Efficacy of Three Wet Detention Ponds.” *Journal of Environmental Quality*, 31(2), 654.

McAndrew, B., Ahn, C., and Spooner, J. (2016). “Nitrogen and sediment capture of a floating treatment wetland on an urban stormwater retention pond-The case of the rain project.” *Sustainability (Switzerland)*, 8(10).

McPhillips, L., Goodale, C., and Walter, M. T. (2018). “Nutrient Leaching and Greenhouse Gas Emissions in Grassed Detention and Bioretention Stormwater Basins.” *Journal of Sustainable Water in the Built Environment*, 4(1), 04017014.

Muerdter, C., Özkök, E., Li, L., and Davis, A. P. (2016). “Vegetation and Media Characteristics of an Effective Bioretention Cell.” *Journal of Sustainable Water in the Built Environment*, 2(1), 04015008.

N.C. DEQ. (2009). “North Carolina Stormwater BMP Manual.” *North Carolina Department of Environmental Quality*.

N.C. DEQ. (2017a). “NCDEQ Stormwater Design Manual.” *North Carolina Department of Environmental Quality*.

N.C. DEQ. (2017b). “North Carolina Stormwater Control Measure Credit Document.” *North Carolina Department of Environmental Quality*.

Nelson, D. W., and Sommers, L. E. (1996). “Total Carbon, Organic Carbon, and Organic Matter.” *Methods of Soil Analysis: Chemical Methods. Part 3*, D. L. Sparks, A. L. Page, P. A. Helmke, and R. H. Loeppert, eds., Soil Science of America and American Society of Agronomy, Madison, WI, 961–1010.

Passeport, E., and Hunt, W. F. (2009). “Asphalt Parking Lot Runoff Nutrient Characterization for Eight Sites in North Carolina, USA.” *Journal of Hydrologic Engineering*, 14(4), 352–361.

Passeport, E., Hunt, W. F., Line, D. E., Smith, R. A., and Brown, R. A. (2009). “Field Study of the Ability of Two Grassed Bioretention Cells to Reduce Storm-Water Runoff Pollution.” *Journal of Irrigation and Drainage Engineering*, 135(4), 505–510.

Paus, K. H., Morgan, J., Gulliver, J. S., and Hozalski, R. M. (2014). “Effects of Bioretention Media Compost Volume Fraction on Toxic Metals Removal, Hydraulic Conductivity, and Phosphorous Release.” *Journal of Environmental Engineering*, American Society of Civil Engineers, 140(10), 04014033.

Perryman, S. E., Rees, G. N., Walsh, C. J., and Grace, M. R. (2011). “Urban Stormwater Runoff Drives Denitrifying Community Composition Through Changes in Sediment Texture and Carbon Content.” *Microbial Ecology*, 61(4), 932–940.

Peterson, I. J., Igielski, S., and Davis, A. P. (2015). “Enhanced Denitrification in Bioretention Using Woodchips as an Organic Carbon Source.” *Journal of Sustainable Water in the Built Environment*, 1(4), 04015004.

Saxton, K. E., and Rawls, W. J. (2006). “Soil Water Characteristic Estimates by Texture and Organic Matter for Hydrologic Solutions.” *Soil Science Society of America Journal*, 70(5), 1569.

Schwammberger, P. F., Lucke, T., Walker, C., and Trueman, S. J. (2019). “Nutrient uptake by constructed floating wetland plants during the construction phase of an urban residential development.” *Science of the Total Environment*, Elsevier B.V., 677, 390–403.

Spangler, J. T., Sample, D. J., Fox, L. J., Owen, J. S., and White, S. A. (2019). “Data on floating treatment wetland aided nutrient removal from agricultural runoff using two wetland species.” *Data in Brief*, Elsevier, 22(November 2018), 756–761.

De Stefani, G., Tocchetto, D., Salvato, M., and Borin, M. (2011). “Performance of a floating treatment wetland for in-stream water amelioration in NE Italy.” *Hydrobiologia*, 674(1), 157–167.

Stevenson, F. J. (1994). *Humus chemistry: genesis, composition, reactions*. Wiley, New York, NY.

Tanner, C. C., and Headley, T. R. (2011). “Components of floating emergent macrophyte treatment wetlands influencing removal of stormwater pollutants.” *Ecological Engineering*, Elsevier B.V., 37(3), 474–486.

Tharp, R., Westhelle, K., and Hurley, S. (2019). “Macrophyte performance in floating treatment wetlands on a suburban stormwater pond: Implications for cold climate conditions.” *Ecological Engineering*, Elsevier, 136(June), 152–159.

Urakawa, H., Dettmar, D. L., and Thomas, S. (2017). “The uniqueness and biogeochemical cycling of plant root microbial communities in a floating treatment wetland.” *Ecological Engineering*, Elsevier B.V., 108, 573–580.

Virahsawmy, H. K., Stewardson, M. J., Vietz, G., and Fletcher, T. D. (2014). “Factors that affect the hydraulic performance of raingardens: Implications for design and maintenance.” *Water Science and Technology*, 69(5), 982–988.

Walker, C., Tondera, K., and Lucke, T. (2017). “Stormwater treatment evaluation of a Constructed Floating Wetland after two years operation in an urban catchment.” *Sustainability (Switzerland)*, 9(10), 1–10.

Wardynski, B. J., and Hunt, W. F. (2012). “Are Bioretention Cells Being Installed Per Design Standards in North Carolina? A Field Study.” *Journal of Environmental Engineering*, 138(12), 1210–1217.

Willard, L. L., Wynn-Thompson, T., Krometis, L. H., Neher, T. P., and Badgley, B. D. (2017). “Does It Pay to be Mature? Evaluation of Bioretention Cell Performance Seven Years Postconstruction.” *Journal of Environmental Engineering*, 143(9), 04017041.

Winston, R. J., Hunt, W. F., Kennedy, S. G., Merriman, L. S., Chandler, J., and Brown, D. (2013). “Evaluation of floating treatment wetlands as retrofits to existing stormwater retention ponds.” *Ecological Engineering*, Elsevier B.V., 54, 254–265.

APPENDICES

Appendix A

A Retrospective Comparison of Water Quality Treatment in a Bioretention Cell 16 Years Following Initial Analysis

Attached

Appendix B

Supplemental Sand Filter Data

Raw Data (Cells shown in yellow are below the minimum detection limit; columns with bold headings include data adjusted for the minimum detection limit)

Date/Trial	Sample Code	TKN (mg/L)	**TKN (mg/L)**	NH_3N (mg/L)	**NH_3N (mg/L)**	NO_3N (mg/L)	**NO_3N (mg/L)**	TP (mg/L)	**TP (mg/L)**	**TN (mg/L)**	TSS (mg/L)	**TSS (mg/L)**
11/9/2018; 1	R1'1	0.60	0.60	0.13	0.13	0.13	0.13	0.17	0.17	0.74	0.97	0.97
	R1'2	0.33	0.33	0.12	0.12	0.14	0.14	0.00	0.02	0.47	2.00	2.00
	R1'3	0.47	0.47	0.11	0.11	0.12	0.12	0.00	0.02	0.58	4.31	4.31
	R2'1	0.33	0.33	0.14	0.14	0.06	0.06	0.01	0.02	0.39	1.71	1.71
	R2'2	0.49	0.49	0.14	0.14	0.03	0.03	0.00	0.02	0.51	1.83	1.83
	R2'3	1.49	1.49	0.17	0.17	0.02	0.02	0.00	0.02	1.51	2.73	2.73
	DI1'1	0.38	0.38	0.10	0.10	0.03	0.03	0.00	0.02	0.40	2.88	2.88
	DI1'2	1.52	1.52	0.25	0.25	0.33	0.33	0.00	0.02	1.85	0.98	0.98
	DI1'3	0.30	0.30	0.26	0.26	0.32	0.32	0.00	0.02	0.62	0.00	0.25
	IN	1.72	1.72	0.17	0.17	0.00	0.01	0.33	0.33	1.73	6.18	6.18
	DI	0.73	0.73	0.39	0.39	0.16	0.16	0.00	0.02	0.89	0.00	0.25
11/16/2018; 2	R1'1	0.78	0.78	0.10	0.10	0.00	0.01	0.05	0.05	0.78	0.96	0.96
	R1'2	0.75	0.75	0.08	0.08	0.03	0.03	0.04	0.04	0.78	1.83	1.83
	R1'3	0.82	0.82	0.09	0.09	0.04	0.04	0.04	0.04	0.85	3.13	3.13
	R2'1	0.64	0.64	0.09	0.09	0.01	0.01	0.03	0.02	0.64	0.96	0.96
	R2'2	1.14	1.14	0.10	0.10	0.06	0.06	0.05	0.05	1.21	2.34	2.34
	R2'3	1.10	1.10	0.08	0.08	0.08	0.08	0.04	0.04	1.18	3.57	3.57
	DI1'1	0.76	0.76	0.26	0.26	1.00	1.00	0.05	0.05	1.76	1.87	1.87
	DI1'2	0.92	0.92	0.24	0.24	0.56	0.56	0.04	0.04	1.49	1.92	1.92
	DI1'3	0.91	0.91	0.22	0.22	0.61	0.61	0.05	0.05	1.52	0.97	0.97
	IN	1.48	1.48	0.08	0.08	0.00	0.01	0.15	0.15	1.49	40.38	40.38
	DI	0.63	0.63	0.35	0.35	0.15	0.15	0.02	0.02	0.78	0.00	0.25
11/19/2018; 3	R1'1	0.63	0.63	0.11	0.11	0.03	0.03	0.03	0.03	0.66	0.00	0.25
	R1'2	0.79	0.79	0.12	0.12	0.04	0.04	0.06	0.06	0.83	0.00	0.25
	R1'3	0.80	0.80	0.09	0.09	0.02	0.02	0.03	0.03	0.82	0.00	0.25
	R2'1	1.15	1.15	0.11	0.11	0.01	0.01	0.04	0.04	1.16	0.00	0.25
	R2'2	0.44	0.44	0.14	0.14	0.01	0.01	0.03	0.02	0.45	1.25	1.25
	R2'3	0.86	0.86	0.15	0.15	0.01	0.01	0.03	0.02	0.86	0.00	0.25
	DI1'1	0.81	0.81	0.18	0.18	0.51	0.51	0.03	0.02	1.32	0.00	0.25

	DI1'2	0.65	0.65	0.19	0.19	0.53	0.53	0.04	0.04	1.18	0.00	0.25
	DI1'3	0.81	0.81	0.20	0.20	0.56	0.56	0.03	0.02	1.37	0.00	0.25
	IN	2.68	2.68	0.16	0.16	0.03	0.03	0.48	0.48	2.71	55.00	55.00
	DI	0.80	0.80	0.36	0.36	0.17	0.17	0.03	0.03	0.97	0.00	0.25
11/30/2018; 4	R1'1	0.45	0.45	0.04	0.04	0.01	0.01	0.08	0.08	0.47	2.75	2.75
	R1'2	0.84	0.84	0.05	0.05	0.00	0.01	0.06	0.06	0.85	5.77	5.77
	R1'3	0.55	0.55	0.08	0.08	0.01	0.01	0.04	0.04	0.57	3.92	3.92
	R2'1	0.65	0.65	0.08	0.08	0.01	0.01	0.04	0.04	0.67	3.48	3.48
	R2'2	0.57	0.57	0.06	0.06	0.02	0.02	0.02	0.02	0.60	3.77	3.77
	R2'3	0.37	0.37	0.07	0.07	0.01	0.01	0.00	0.02	0.37	1.08	1.08
	DI1'1	0.59	0.59	0.11	0.11	0.89	0.89	0.03	0.03	1.48	2.97	2.97
	DI1'2	0.79	0.79	0.10	0.10	1.28	1.28	0.07	0.07	2.07	1.06	1.06
	DI1'3	0.53	0.53	0.08	0.08	1.31	1.31	0.18	0.18	1.85	0.00	0.25
	IN	5.43	5.43	0.04	0.04	0.03	0.03	1.23	1.23	5.46	64.71	64.71
	DI	0.68	0.68	0.39	0.39	0.18	0.18	0.02	0.02	0.86	0.00	0.25
12/18/2018;5	R1'1	0.48	0.48	0.09	0.09	0.01	0.01	0.04	0.04	0.50	3.74	3.74
	R1'2	0.80	0.80	0.07	0.07	0.01	0.01	0.08	0.08	0.82	5.10	5.10
	R1'3	0.49	0.49	0.08	0.08	0.01	0.01	0.04	0.04	0.49	5.21	5.21
	R2'1	0.64	0.64	0.08	0.08	0.01	0.01	0.07	0.07	0.65	7.69	7.69
	R2'2	1.06	1.06	0.09	0.09	0.00	0.01	0.06	0.06	1.07	34.31	34.31
	R2'3	0.66	0.66	0.10	0.10	0.02	0.02	0.04	0.04	0.68	1.87	1.87
	DI1'1	0.74	0.74	0.10	0.10	0.86	0.86	0.05	0.05	1.60	2.38	2.38
	DI1'2	0.50	0.50	0.13	0.13	0.75	0.75	0.09	0.09	1.26	3.16	3.16
	DI1'3	0.39	0.39	0.10	0.10	0.68	0.68	0.04	0.04	1.07	0.00	0.25
	IN	11.05	11.05	0.17	0.17	0.02	0.02	1.42	1.42	11.07	175.86	175.86
	DI	0.63	0.63	0.46	0.46	0.18	0.18	0.02	0.02	0.80	0.00	0.25
1/25/2019;6	R1'1	0.57	0.57	0.10	0.10	0.80	0.80	0.03	0.02	1.37	2.54	2.54
	R1'2	0.67	0.67	0.09	0.09	0.71	0.71	0.03	0.02	1.38	1.77	1.77
	R1'3	0.47	0.47	0.10	0.10	0.63	0.63	0.02	0.02	1.09	0.00	0.25
	R2'1	0.53	0.53	0.10	0.10	0.59	0.59	0.04	0.04	1.12	3.74	3.74
	R2'2	0.59	0.59	0.10	0.10	0.84	0.84	0.04	0.04	1.43	2.75	2.75

	R2'3	0.48	0.48	0.13	0.13	1.41	1.41	0.02	0.02	1.89	0.00	0.25
	DI1'1	0.56	0.56	0.09	0.09	1.07	1.07	0.06	0.06	1.64	5.88	5.88
	DI1'2	0.44	0.44	0.10	0.10	0.74	0.74	0.16	0.16	1.18	8.24	8.24
	DI1'3	0.27	0.27	0.10	0.10	0.69	0.69	0.05	0.05	0.96	5.71	5.71
	IN	1.96	1.96	0.11	0.11	0.01	0.01	0.25	0.25	1.97	43.06	43.06
	DI	0.52	0.52	0.37	0.37	0.15	0.15	0.03	0.03	0.67	0.00	0.25
4/12/2019;7	R1'1	2.33	2.33	0.06	0.06	0.92	0.92	0.24	0.24	3.25	6.41	6.41
	R1'2	1.46	1.46	0.04	0.04	0.71	0.71	0.21	0.21	2.17	6.25	6.25
	R1'3	2.00	2.00	0.04	0.04	0.83	0.83	0.19	0.19	2.83	7.69	7.69
	R2'1	2.17	2.17	0.08	0.08	1.06	1.06	0.19	0.19	3.23	2.82	2.82
	R2'2	2.54	2.54	0.81	0.81	0.97	0.97	0.20	0.20	3.52	7.25	7.25
	R2'3	2.51	2.51	0.78	0.78	1.35	1.35	0.17	0.17	3.86	8.33	8.33
	DI1'1	1.17	1.17	0.05	0.05	0.34	0.34	0.27	0.27	1.51	9.43	9.43
	DI1'2	1.59	1.59	0.06	0.06	0.25	0.25	0.33	0.33	1.84	25.00	25.00
	DI1'3	2.11	2.11	0.07	0.07	0.26	0.26	0.28	0.28	2.37	15.38	15.38
	IN	9.33	9.33	2.94	2.94	0.03	0.03	0.92	0.92	9.36	23.86	23.86
	DI	0.00	0.02	0.09	0.09	0.02	0.02	0.18	0.18	0.03	0.00	0.25
5/16/2019;8	R1'1	0.91	0.91	0.09	0.09	5.57	5.57	0.05	0.05	6.48	3.23	3.23
	R1'2	0.41	0.41	0.08	0.08	5.99	5.99	0.04	0.04	6.40	3.15	3.15
	R1'3	0.87	0.87	0.08	0.08	5.96	5.96	0.03	0.02	6.82	3.05	3.05
	R2'1	0.61	0.61	0.08	0.08	4.93	4.93	0.00	0.02	5.53	1.56	1.56
	R2'2	1.33	1.33	0.54	0.54	4.80	4.80	0.03	0.03	6.13	3.73	3.73
	R2'3	1.42	1.42	0.36	0.36	4.97	4.97	0.00	0.02	6.39	6.92	6.92
	DI1'1	0.41	0.41	0.07	0.07	0.40	0.40	0.08	0.08	0.81	6.32	6.32
	DI1'2	0.40	0.40	0.07	0.07	0.32	0.32	0.07	0.07	0.72	7.37	7.37
	DI1'3	0.48	0.48	0.08	0.08	0.48	0.48	0.31	0.31	0.96	8.65	8.65
	IN	18.30	18.30	10.40	10.40	0.03	0.03	2.24	2.24	18.33	144.00	144.00
	DI	0.14	0.14	0.07	0.07	0.01	0.01	0.00	0.02	0.15	0.00	0.25