SUSTAINABLE STORMWATER MANAGEMENT USING A FLOATING WETLAND—A SYSTEM APPROACH

by

Brendan McAndrew
A Thesis
Submitted to the
Graduate Faculty
of
George Mason University
in Partial Fulfillment of
The Requirements for the Degree
of
Master of Science
Environmental Science and Policy

Committee:

____________________________________	Dr. Changwoo Ahn, Thesis Director
____________________________________	Dr. Younsung Kim, Committee Member
____________________________________	Dr. Dianna Hogan, Committee Member
____________________________________	Dr. Albert P. Torzilli, Graduate Program Director
____________________________________	Dr. A. Alonso Aguirre, Department Chairperson
____________________________________	Dr. Donna Fox, Associate Dean, Student Affairs & Special Programs, College of Science
____________________________________	Dr. Peggy Agouris, Dean, College of Science
Date: ______________________________	Spring Semester 2017 George Mason University Fairfax, VA

Sustainable Stormwater Management Using a Floating Wetland—a System Approach

A Thesis submitted in partial fulfillment of the requirements for the degree of Master of Science at George Mason University

by

Brendan McAndrew
Bachelor of Science
Christopher Newport University, 2011

Director: Changwoo Ahn, Professor
Department of Environmental Science and Policy

Spring Semester 2017
George Mason University
Fairfax, VA

ACKNOWLEDGEMENTS

I would like to thank everyone that had a role in helping me, directly or indirectly, in completing this degree. A special thank you to my family for supporting me throughout my educational career from kindergarten through graduate school; I would not have been able to get this far without your unending support. Thank you to my friends for helping me stay sane during the ups and downs of graduate school. Enormous thanks to Dr. Changwoo Ahn for taking me under his wing. His patience and kindness as a mentor helped me develop as a scientist while he guided me through the publication two papers. Thank you to the other members of the Ahn Wetland Lab for their help along the way: Mary Means, Alicia Korol, Andy Sachs, Joanna Spooner, and Sarah Macey. Thank you to Dr. Lorelei Crerar and Dr. Larry Rockwood for their support as a teaching assistant and to my fellow TAs for helping each other succeed. Thank you all!

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS AND SYMBOLS

Best Management Practice	BMP
Floating Wetland	FW
Grams	g
Meters	m
Meters, cubic	m^3
Meters, square	m^2
Nitrogen	N
Stormwater Control Measure	SCM
Total Daily Maximum Load	TMDL
Total Nitrogen	TN
Total Suspended Solids	TSS

ABSTRACT

SUSTAINABLE STORMWATER MANAGEMENT USING A FLOATING WETLAND—A SYSTEM APPROACH

Brendan McAndrew, M.S.

George Mason University, 2017

Thesis Director: Dr. Changwoo Ahn

Nitrogen is widely recognized as a chronic urban stormwater pollutant. In the United States, wet retention ponds have become widely used to treat urban runoff for quantity and quality. While wet ponds typically function well for the removal of sediments, nitrogen removal performance can be inconsistent due to poor design and/or lack of maintenance. Renovating ponds to improve their nitrogen capture performance, however, is typically expensive. A relatively untested technology called floating wetlands (FWs) has been proposed as a sustainable means of improving the nitrogen capture performance of stormwater wet ponds. The FWs are comprised of an artificial floating island that supports the hydroponic growth of plants on a pond, lake, or canal. As the plants grow on the floating island, their roots remove nitrogen directly from the water column and may trap waterborne sediments. Few studies have been performed on the effectiveness real-world stormwater systems, however. In this study, the nitrogen and sediment capture

performance of a 50 m^2 floating wetland deployed for 137 days on Mason Pond was investigated. A total of 2684 g of biomass was produced, 3100 g of sediment captured, and 191 g of nitrogen removed from the pond. Although biomass production was relatively low (53 g/m^2), nitrogen uptake rate by the plants (0.009 g/m^2/day) was comparable to contemporary FW studies. A system model was then developed from the collected data to simulate nitrogen removal performance of the FW on Mason Pond. The model was then used to test the nitrogen removal efficiency of the FW over longer deployment periods and with greater surface area coverage. While the literature suggests that FWs must cover at least 10-15% of the pond to significantly aid nitrogen removal, the model suggests only modest nitrogen removal efficiency (~6%) by an FW covering 25% of the surface of Mason Pond. These results may inform municipalities or developers that are considering the use of FWs on stormwater ponds.

CHAPTER ONE

Runoff from urban areas is increasingly responsible for the transport of nitrogen into natural waterways (Carey et al., 2013; Fletcher, 2004; Fletcher et al., 2013; USGS, 2016; Yang et al., 2011). Over the past two decades, nitrogen has become recognized as a pervasive stormwater pollutant and is targeted by the Environmental Protection Agency's (EPA) Total Maximum Daily Load (TMDL) program (USEPA, 1999a). The TMDL program requires that states establish a limit on the amount, or load, of target pollutants that can be discharged into a particular body of water (USEPA, 2015). To meet TMDL goals, urban runoff may be treated by stormwater control measures and "green infrastructure" such as permeable pavement, bioswales, and retention ponds (Collins et al., 2010; Keeley et al., 2013; USEPA, 2008).

After the Clean Water Act's passage in 1972 retention ponds became the most commonly used SCM to manage runoff and they are now a ubiquitous feature of urban development (Winston et al., 2013). The county in which this study took place, for example, is dotted with some 200-300 stormwater retention ponds, the majority of which are located on private property (Fairfax County Department of Public Works and Environmental Services, 2017). The primary function the ponds is to attenuate the flow of stormwater into natural waterways in an effort to reduce stream bank erosion and the risk of flooding downstream (USEPA, 1999b).

Retention ponds can also trap pollutants as the reduced velocity of the water encourages sedimentation of suspended particulates, but this requires adequate hydraulic residence time in the pond (Borne et al., 2013). As sediments build up over time, however, pond function is impaired as retention capacity is reduced and hydraulic residence time decreases (Verstraeten and Poesen, 2000). Dredging is therefore required every 5-20 years, depending on pond size and sediment loading rate, to restore pond functionality (Hunt, 2006; Rollins, 2016; USEPA, 1999b). Nitrogen and phosphorus may also be captured by wet ponds though performance is inconsistent as it is influenced by pond design and influent characteristics (Collins et al., 2010; Marimon et al., 2013; USEPA, 1999b; Winston et al., 2013). While phosphorus is primarily captured by sedimentation, nitrogen is removed by several pathways including microbial denitrification and assimilation into plant biomass. Ponds that feature significant littoral vegetation, for example, are more effective at capturing nitrogen than those lacking a plant buffer (Collins et al., 2010; Mallin et al., 2002). Renovation of retention ponds to improve nutrient removal can be expensive however (est. $20,000–$60,000/acre for dredging alone), prompting research into alternative ways to improve nutrient removal in wet ponds (Northern Virginia Planning District Commission, 2000).

One proposed method of augmenting the nutrient capture function of a wet pond involves the deployment of an artificial island stocked with wetland plants called a floating wetland (Headley and Tanner, 2006; Sample et al., 2013; Wanielista et al., 2012). In wet ponds, vegetation is typically rooted in sediment in the littoral zone which limits exposure of the roots to the water column and slows nutrient assimilation (Tanner and Headley, 2011). Floating wetlands (FWs) employ the hydroponic growth of emergent macrophytes to remove nutrients directly from the water column via root uptake (Headley and Tanner, 2012; Pavlineri et al., 2017). The buoyant nature of the FW ecosystem ensures that the plant roots are fully exposed to the water column at all times regardless of changes in pond depth. In addition to direct plant uptake, recent studies have shown that the FW structure itself and the suspended root matrix may be colonized by microbial biofilms which can contribute to transformation or removal of nutrients (e.g., denitrification) (Kadlec and Wallace, 2009; Osem et al., 2007; Pavlineri et al., 2017). The development of a dense, web-like root matrix beneath the FW can also increase the sediment capture function of a wet pond by physically trapping suspended sediments and encouraging settling of particulates by impeding flow the of water in the pond (Winston et al., 2013).

The literature suggests there is potential for the use of floating wetlands to improve water quality in stormwater systems, however the majority of FW studies have been performed in microcosm/mesocosm environments rather than real-world stormwater systems (Borne et al., 2015; Headley and Tanner, 2012; Pavlineri et al., 2017; Wang and Sample, 2014; Winston et al., 2013). In order to evaluate the extent of this technology's potential it is important to test FW performance in situ where conditions are less predictable (Borne et al., 2015; Headley and Tanner, 2012; Marimon et al., 2013).

One potential drawback of FWs is that the aerial plant tissue must be harvested annually to ensure permanent removal of nutrients from the stormwater system. However, the assumption that aerial tissues accumulate a significant amount of nutrients is based on only a handful of studies investigating the temporal variation in the distribution of nutrients within FW plant tissues (Headley and Tanner, 2012; Marschner, 1995; Wang et al., 2014; Wang and Sample, 2014). Wang et al. (2014), for example, reported that distribution of nutrients within the biomass of FW grown plants varied temporally by both species and target nutrient. In bottom-rooted plants, Marschner (1995) and Williamson (2001) found that nutrient distribution in biomass varied with developmental stage and in response to nutrient availability in the growth medium while Meuleman et al. (2002) found that *Phragmites australis* redistributed nutrients to the roots during the winter, although these results may not be representative of a hydroponic system. Further investigation of nutrient distribution and mobilization is needed in order to optimize harvest strategies for FW systems.

In this chapter, the performance of a small FW on an urban stormwater retention pond near Washington, D.C. was investigated. The FW was designed and deployed as part of "The Rain Project", an interdisciplinary student group research and scholarship project addressing sustainable stormwater management on a university campus (Ahn, 2016). To measure performance, nitrogen uptake via plant growth and sediment accretion on the FW structure itself were measured. The FW was stocked with five species of native wetland plants and deployed for 137 days during the summer of 2015. The objectives were to estimate nitrogen capture performance of the FW through quantification of (1) biomass production; (2) differential nitrogen content of the roots and shoots of each species; (3) physical sediment captured by the roots and FW structure itself; and (4) the nitrogen contained in the sediment accumulated on the FW structure.

This information may inform future management and harvest strategies for FW systems deployed on stormwater wet ponds.

Methods and Materials

Study Site

The FW was deployed on Mason Pond (38°49′44″ N, 77°18′37″ W), a 7100 m^2 urban stormwater retention pond with an average depth of 1.1 m located on George Mason University's (GMU) Fairfax campus. The pond is primarily fed by a heavily incised stream that flows through a small wooded area before entering the pond. During storm events, additional runoff from adjacent parking lots is transported into the pond through two storm sewers. The outflow of the pond is a weir located approximately 100 m from the primary inflow on the opposite end of the pond. The drainage area for Mason Pond covers approximately 0.55 km^2 of land on the urban campus of a university attended by approximately 35,000 students with a large commuter population (George Mason University, 2015). The campus is dominated by large buildings, expansive parking lots, and dwindling pockets of undisturbed land. Since the pond's construction, local development has increased the drainage area beyond its design capacity by approximately 10% (George Mason University, 2013). Total precipitation during the study period (May–September 2015) was 421 mm with a monthly average of 84 mm while air temperature ranged from 5.6 °C in May to 34.4 °C in September (National Oceanic and Atmospheric Administration, 2016).

Table 1. Mason Pond water quality and hydrologic characteristics during the deployment period of the floating wetland. Physicochemical parameters are reported as mean ± standard deviation.

Parameter	Value
Area (m^2)	~7100
Volume (m^3)	~6532
Mean Depth (m)	0.92
Mean Water Temperature (°C)	29.0 ± 2.7
Mean pH	7.51 ± 0.80
Mean Dissolved Oxygen (mg/L)	9.71 ± 2.05
Mean TSS (mg/L)	21.8 ± 10.5
Est. TN (mg/L)	< 2.0

Floating Wetland Construction

FWs can be created using commonly available materials such as polyvinyl chloride (PVC) tubing and plastic mesh or by employing purpose-built FW systems such as Biohaven® or Beemat (BeeMats, LLC, 2017; De Stefani et al., 2011; Floating Islands International, Inc., 2017; Wang and Sample, 2013). The variety of construction materials available allows for flexibility in the size and shape of FWs so that aesthetics may be incorporated. In this study, the FW design incorporated ideas developed by an interdisciplinary group of GMU undergraduate students enrolled in an ecological sustainability course (EVPP/BIOL 378 and 379) in the Spring semester of 2015. The FW was designed to mimic human kidneys for their ability to filter contaminants from stormwater (Ahn, 2016).

The FW was constructed using a commercially available system marketed as BeeMat (BeeMats LLC, New Smyrna Beach, FL, USA). The system is based around a buoyant, 1.3 cm thick ethylene vinyl acetate (EVA) foam mat covered with pre-cut circular holes uniformly spaced 10 cm apart (Fig. 1).

Fig. 1. Top view of the floating wetland (FW) ecosystem deployed on Mason Pond showing cup spacing and waterfowl fence.

The pre-cut holes allow plants, placed in perforated plastic cups supplied with the system (Henry and Henry, 2007), to be suspended from the mat such that the roots were submerged below the water while the shoots remained above the water line as in a hydroponic system (Fig. 2).

Fig. 2. Side view of BeeMat floating wetland showing plant cup system and free-hanging roots. Image courtesy of BeeMats, LLC: (www.beemats.com).

The mats are typically sold in a variety of dimensions with jigsaw-puzzle-style cuts along the edges that allow multiple mats to be joined together like puzzle pieces. Multiple rectangular foam mats were fastened together into two ~30 m^2 rectangles and divided into two identical “kidneys” with a final combined surface area of just over 50 m^2. The FW was then stocked with plants before being placed on Mason Pond in the area with the greatest depth (approximately 1.54 m). Once deployed, the kidneys were tethered together using marine grade rope fed through 1” x 8’ PVC tube to ensure the FWs maintained approximately 2.5 m of separation at all times. Each kidney was then anchored to the bottom of the pond by two cinder blocks such that the FW system was free to move vertically with any changes in pond water level, but did not travel significantly due to wind action or water current. The full-scale floating wetland was deployed on 12 May 2015 (NBC4 Washington, 2015, p. 4).

Canada geese are often found in the pond and have damaged FWs in other studies. To deter waterfowl from landing on the FW and/or grazing on the plants, a simple fence was constructed from fishing line strung between wooden dowels placed around the periphery of each kidney (Fig. 1) (Borne et al., 2015; Wang and Sample, 2014). A piece of reflective plastic flagging was attached to the top of each dowel to create an additional visual/audible deterrent.

Plant Selection

Five species of native wetland emergent macrophytes were chosen for use on the FW: *Alisma subcordatum* (American water plantain), *Carex stricta* (upright sedge), *Iris versicolor* (blue flag iris), *Juncus effusus* (common rush), and *Pontederia cordata* (pickerelweed). Plugs were purchased from Environmental Concern Inc. (St. Michael's, MD, USA); a nursery that specializes in the sale of wetland plant species for research and industrial use. These species were selected because they are common North American wetland native plants that tolerate constant inundation of the root system. While all five species have been previously used in treatment wetlands, *Juncus* and *Carex* have been shown to be highly effective nutrient removers (Means et al., 2016; Tanner, 1996) and, as such, were planted in higher numbers than the other species (Table 2).

Table 2. Planting regime for Mason Pond floating wetland ecosystem showing the number of individuals of each species that were planted on the FW

Plant Species	***Alisma***	***Carex***	***Iris***	***Juncus***	***Pontederia***
# of individuals planted	200	350	170	500	290

Prior to deployment on the FW, the roots of each plug were thoroughly washed in water to remove any potting medium in an effort to limit import of sediment and nutrients into the pond. The washed roots were then gently wrapped with a sheet of coconut coir and inserted into a perforated plastic cup. A plastic fastener was then used to secure the plant in place so that it did not become dislodged or float out of the cup. The plastic cup was then inserted into one of the pre-cut holes on the FW mat immediately prior to deployment.

Plant Biomass Measurement and Tissue Analysis

To determine change in plant biomass (i.e., plant growth) and their nitrogen contents, between 8 and 18 individuals of each species were collected before and after FW deployment. Initial plant samples were randomly selected from the lot delivered by the nursery while post-deployment samples were randomly selected on the day the FW was removed from the pond and all biomass was harvested. Samples were processed within three days of collection. The roots of all samples were gently washed with water to remove any sediment or foreign materials. Sediment washed from roots and cups of plants collected after deployment was saved for sediment analysis (see following section). The roots of post-deployment samples had to be cut away from the coconut coir wrapper because the roots had grown entangled in the coir during the course of the FW deployment. Care was taken to cut the roots off as close to the coir wrapper as possible to minimize the loss of plant material. The coconut coir was then carefully unwrapped and the remaining plant material inside was collected. Scissors were used to separate the roots and shoot of each plant for individual analysis. Individual plant samples were transferred to a pre-weighed brown paper bag, weighed for wet mass, and placed in a Lindberg Blue Mechanical Oven (Model #M01450A) at 50 °C for 36 h or until constant mass was achieved between measurements. Once dry, the bags were removed and weighed to determine dry mass. The individual dried samples were then combined by species and plant structure (root/shoot) and ground into a fine powder using a 60-mesh screen on a Thomas Scientific Mini-Mill Cutting Mill (Model #3383-L10). Three samples of the root and shoot of each species were analyzed for nitrogen content with a Perkin Elmer Series II CHNS/O Analyzer.

Sediment Analysis

After plants were removed from the FW, sediment accumulated on the underside of the Beemat was collected while still wet. Quadrats of approximately 0.05 m^2 were randomly placed on the Beemat and plastic putty knives were used to scrape as much sediment as possible from the surface. Sediment was removed from approximately 3.23 m^2 of the mat surface (excluding pre-cut holes in the mat). The scraped sediment was transferred to a Nalgene bottle and placed in a refrigerator for temporary storage. Within 24 h, the sediment was transferred to a pre-weighed foil packet and dried at 60 °C for 36 h. The mass of the sediment was then calculated by subtracting the mass of the empty foil packet from the mass of the foil packet containing dried sediment. Density of accumulated sediment (g/m^2) was calculated by dividing the mass of dried sediment collected by the area sampled. An estimate for total mass of sediment accumulated by the

FW mat was calculated by multiplying the density by the total area of the FW minus the pre-cut holes.

Sediment and foreign material trapped by plant roots and plastic cups during deployment was collected by species from the roots of the plants randomly sampled from the FW. Tap water was used to wash the trapped material from the roots and cups into a 5-gallon bucket. The mixture of trapped material and water was then passed through a 2 mm sieve to remove large, non-sediment material (i.e., small rocks and twigs) followed by a 0.5 mm sieve to remove finer pieces of non-sediment material (i.e., fine plant material and small aquatic organisms). The material trapped by the sieves was thoroughly washed with tap water to minimize the amount of sediment not accounted for. The remaining mixture was assumed to contain only trapped sediment. To determine the amount of sediment captured by the sampled plants, the total volume of the sediment and water mixture was determined. The mixture was then agitated vigorously to suspend the sediment evenly in the water column. Three random 100 mL samples were then vacuum filtered through Whatman™ 934-AH™ glass microfiber filters (CAT. No. 1827-047). The filters were then dried at 105 °C for 1 h to determine dry mass of sediment filtered. The resultant mass was then divided by the volume filtered to determine the density of suspended sediment. Total amount of sediment trapped by the sampled plants was then estimated by multiplying the density of suspended sediment by the total volume of sediment and water mixture. To estimate the amount of sediment captured per cup, the estimated mass of total trapped sediment was divided by the number of plants sampled for each species respectively. The mass of total sediment trapped by the roots and cups on the FW was then estimated by multiplying the mass of sediment trapped per plant by the total number of plants of each respective species. CHNS/O analysis was also performed for sediment scraped from the FW structure itself as well as the sediment washed from the cup and roots for each species.

Data Analysis

Individual root and shoot dry biomass measurements were combined to estimate whole plant biomass for each species. Mean and standard error of dry biomass was calculated for whole plants, roots, and shoots for each species. Mean change in biomass per plant per species was estimated by subtracting mean initial dry biomass from mean final dry biomass. Total change in biomass per species was estimated by multiplying mean change in biomass per plant by the total number of conspecifics planted on the FW. Similarly, mean and standard error of nitrogen content (as % of dry matter) was calculated per species and plant structure. To estimate nitrogen content of whole plants (roots/shoots combined), results of CHN analysis on roots/shoots were averaged together. IBM SPSS statistical soFWare was used to compare biomass and nitrogen content between pre- and post-deployment with an independent samples T-test ($p = 0.05$). Levene's test for equality of variances was used prior to the T-test to test the equality of variance assumption. Water temperature, dissolved oxygen, and pH were compared between inlet and below-FW using a two-sample T-test ($p = 0.05$).

Results and Discussion

Mason Pond Water Quality

The water quality in Mason Pond (Table 1) was comparable to that of stormwater used in both mesocosm and real-world studies (Lynch et al., 2015; Wang et al., 2014).

There was no significant difference between the water temperature, dissolved oxygen, pH, and TSS when inlet and below-FW measurements were compared ($p > 0.05$). Borne et al. (2015) observed significantly lower DO beneath their FW which, coupled with high water temperature and secretion of organic carbon from plant roots, create ideal conditions for denitrification. The lack of measurable impact on selected water quality parameters is likely due to the small size of the FW (~1% of pond surface area). A model by Marimon et al. (2013) suggests that an FW must cover 10%–25% of pond surface area to significantly improve water quality.

Plant Biomass Production and Nitrogen Uptake

Visual assessment of above-mat growth on the FW suggested that biomass production was poor on the FW. While shoot length was not explicitly measured in this study, no shoots appeared to exceed 25 cm in height; less than half of what was reported in several other FW studies (Lynch et al., 2015; Wang et al., 2014; Winston et al., 2013). Root length, however, appeared to be on par with observations by Wang et al. (2015) ranging from 20 to 50 cm. Over the course of the 137-day deployment, an estimated 53.7 g/m^2 of total biomass was produced by the plants on the FW for a total of 2684 g of biomass. By comparison, Lynch et al. (2015) found that a Beemat based FW (the same system used in this study) stocked with *Juncus effusus* produced 155.3 g/m^2 in total biomass when deployed in mesocosms that contained less than half the TN found in Mason Pond. When biomass production was examined at the species level it was observed that while growth was relatively poor all around, some species were more productive than others. *Carex*, *Iris*, and *Pontederia*, for example, experienced a significant increase ($p < 0.05$) in biomass during deployment, generating an average of 3.29 g, 2.51 g, and 2.02 g in biomass, respectively (Fig. 3a). *Alisma* and *Juncus*, however, showed no change from initial biomass though *Juncus* appeared to have lost mass after deployment. While observed mass per plant values were similar to those reported by Wang et al. (2015), Winston et al. (2013) reported final biomass between one and two magnitudes of order higher than were observed for the same species (e.g., *Carex*, *Juncus*, and *Pontederia*) in this study. In both of these studies, the FWs were deployed between two and four times longer than ours which suggests FWs may require at least one growing season to become acclimated before producing considerable biomass although this did not appear to limit biomass by Lynch et al. (2015).

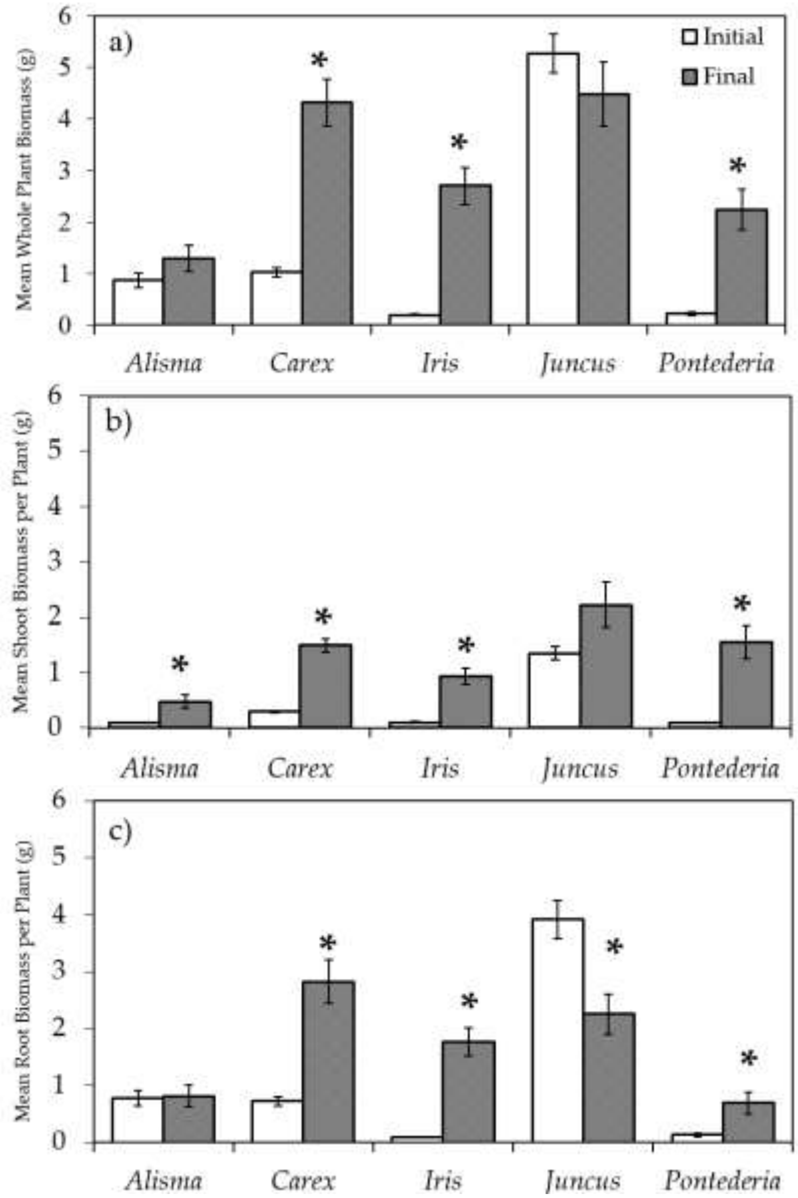


Fig. 3. Initial and final mean biomass per plant for (**a**) whole plant; (**b**) shoots and (**c**) roots of plants deployed on the Mason Pond floating wetland ecosystem. Error bars represent standard error and asterisks represent significant difference between initial and final biomass ($p < 0.05$).

Shoot biomass increased significantly in all species except *Juncus* (Fig. 3b). Similarly, the roots of *Carex*, *Iris*, and Pontederia increased significantly from initial deployment while *Juncus* roots appeared to lose an average of 1.66 g of root biomass which appears to have been caused by inaccurate initial root mass measurements (Fig. 3c). The *Juncus* plugs received from the nursery were severely pot-bound which made it very difficult to wash all soil from the roots without causing damage to the plant. This is supported by the fact that the initial mean root mass of *Juncus* was 3.13 g higher than *Alisma* which had the second heaviest initial root mass (3.91 g vs. 0.78 g respectively). It is likely that any remaining potting material was then washed away during the course of the FW's deployment.

While whole-plant biomass production was mediocre in comparison to other FW studies, oteworthy differences between above- and below-mat biomass production were observed. For example, higher below-mat biomass production was observed on the Mason Pond FW compared to the mesocosm-based FW study mentioned earlier (23.8 g/m^2 vs. 12.4 g/m^2) though above-mat production was nearly five times lower in this study (29.9 g/m^2 vs. 142.9 g/m^2) (Lynch et al., 2015). In both cases, it is likely that below-mat biomass production was higher than observed as it was difficult, if not impossible, to remove all root material for sampling. The ratio of above-mat biomass to below-mat biomass (A:B) on the Mason Pond FW indicated that development of root mass outpaced shoot development in all species except *Juncus* and *Pontederia* (Table 3). Other FW studies, however, have consistently reported more vigorous shoot growth (A:B >1) than was observed in this study (Tanner and Headley, 2011; Winston et al., 2013). While disproportionate root growth has been observed as a physiological response to low nutrient availability, nutrient levels in Mason Pond are not thought to be low enough to trigger such a response (Lorenzen et al., 2001; Marschner, 1995; Williamson, 2001).

Unfortunately, funding prevented periodic measurement of the nutrient levels in Mason Pond to verify this assumption. Given the heavily urbanized location of the pond, however, it is unlikely that the pond waters are oligotrophic. Nutrient removal is a function of the combination of several physicochemical and biological processes, including sedimentation and plant uptake. Further research is needed to investigate more carefully those factors that may interact with or impact growth of FW plants.

Table 3. Post-deployment above- and below-mat biomass (g) and nitrogen content (mg) per plant (mean ± standard error) with ratio between above- and below-mat biomass (A:B). Nitrogen mass per plant estimated by multiplying mean dry biomass with nitrogen content as percent of dry matter. Biomass ratio calculated by dividing above-mat biomass by below-mat biomass. Biomass was compared with a two sample T-test to check for difference in above- and below-mat growth; an asterisk (*) symbolizes significant difference at $p = 0.05$.

	Below-Mat		**Above-Mat**		**A:B**
Plant Species	**Biomass (g)**	**Nitrogen (mg)**	**Biomass (g)**	**Nitrogen (mg)**	**Biomass Ratio**
Alisma	0.82 ± 0.19	18.89 ± 2.15	0.47 ± 0.12	12.18 ± 0.13	0.57
Carex	2.82 ± 0.38*****	52.62 ± 0.25	1.49 ± 0.13*****	25.79 ± 0.47	0.52
Iris	1.77 ± 0.24	25.42 ± 2.62	0.93 ± 0.14	18.23 ± 2.64	0.52
Juncus	2.25 ± 0.35	34.85 ± 1.21	2.22 ± 0.41	30.44 ± 1.89	0.99
Pontederia	0.69 ± 0.19	14.36 ± 2.25	1.55 ± 0.31	31.26 ± 3.57	2.24

The FW plants removed a total of 65.8 g of N at a rate of 0.009 g/m^2/day which is well within the range of nitrogen uptake rates reported by Wang et al. (2014) in a review of 16 studies on FWs and similar technologies (i.e., between 0.0015 and 2.8 g/m^2/day; mean 0.64 g/m^2/day). This result is also very similar to the 0.007 g/m^2/day of nitrogen uptake by a Beemat FW reported by Lynch et al. (2015). Root and shoot nitrogen concentration were quite similar across the board with root concentrations ranging from 1.44% to 2.32% of dry matter and shoot concentrations ranging from 1.37% to 2.58% of dry matter (Fig. 4). *Alisma* exhibited the highest concentration of nitrogen in both the roots and shoots, while root and shoot nitrogen was lowest in *Iris* and *Juncus*, respectively. For comparison, Winston et al. (2013) reported below-mat nitrogen concentrations between 1% and 2% and above-mat concentrations less than or equal to 1% for a Biohaven® based FW. In an investigation of nitrogen accumulation in sediment-rooted wetland macrophytes, Lenhart et al. (2012) reported above- and below-ground nitrogen concentrations of between 0.78%–2.63% and 0.39%–1.94% respectively. These results suggest that nitrogen uptake by the Mason Pond FW was not dissimilar from other FW systems nor bottom-rooted plants. To ensure that nitrogen was not exported from the plants during deployment, initial and final nitrogen concentrations were compared. The analysis indicates that nitrogen content in the shoots of *Alisma*, *Carex*, and *Juncus* increased significantly over the course of the deployment while nitrogen content in the roots increased only in *Carex* and *Juncus* (Fig. 4). This suggests that *Carex* and *Juncus* may have a greater capacity for nitrogen assimilation than the other species sampled.

Several studies have suggested that nitrogen may be redistributed between root and shoot tissue on a seasonal basis or in response to environmental stimuli (Wang et al., 2014; Wang and Sample, 2014). However, it was observed that nitrogen distribution between the roots and shoots of the plants did not differ much after growth on the FW (Table 3). Although *Carex* showed a statistically significant difference in the nitrogen content between roots and shoots, this difference was fairly negligible in terms of overall nitrogen capture (i.e., 1.86% vs. 1.73%). While some studies suggest that temporal variation in the distribution of nutrients within plant tissues should be a major consideration in harvesting strategy, these results suggest that scheduling harvest at peak above-mat biomass may be a simpler solution as it mitigates the risk of release of nitrogen associated with senescence.

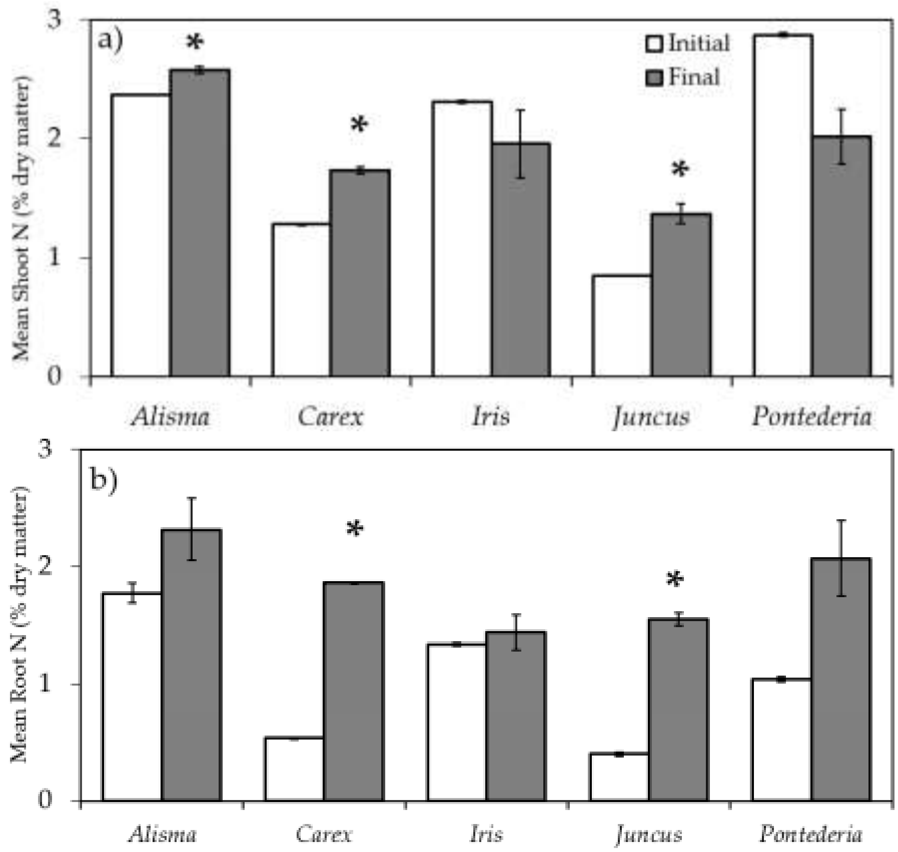


Fig. 4. Initial and final mean nitrogen content as percent of dry matter per plant for (a) shoots and (b) roots for plants deployed on the Mason Pond floating wetland. Error bars represent standard error while asterisks represent significant difference between initial and final nitrogen concentration ($p < 0.05$).

Physical Sediment Capture and Nitrogen Content

During the course of the deployment, the edge of the FW was periodically lifted up to examine root growth. Within several weeks, a brown film had formed on the bottom of the FW mat, the perforated plant cups, and the plant roots themselves indicating accumulation of suspended particulate matter. It was impossible to fully separate sediment captured by plant roots from that captured by the plant cups and they are therefore referred to as a single unit (roots) in this discussion. The FW mat accumulated approximately 30.69 g/m^2 of sediment; a total of 1313 g of sediment captured between the two kidneys (Table 4). This sediment contained 54.4 mg/kg nitrogen indicating that the FW mat removed approximately 71.5 g of nitrogen from the pond (Table 4). Sediment trapped by the roots varied by plant species and ranged from 0.89 g/plant in *Alisma* to 1.44 g/plant in *Carex* with an estimated total sediment capture of 1787 g. Nitrogen content of the root-trapped sediment varied slightly by species and ranged from 2.57% to 3.79%. Extrapolation for all plants on the FW indicated that as much as 1787 g of sediment, containing 54 g of nitrogen, was trapped by the roots. These results suggest that a total of 3100 g of sediment and 125 g of nitrogen was captured by the FW system as a whole at a rate of 66 g/m^2 (Table 4). In a review of floating wetland studies, Headley and Tanner (2012) reported FWs have exhibited sediment capture rates between 20 and 2200 g/m^2 which suggests that the Mason Pond FW performed favorably in this regard. As discussed previously, however, TSS was not significantly reduced near the FW likely due to the small size of the Mason Pond FW. At least one other study has reported significant reductions in TSS in a pond featuring an FW (Borne et al., 2013b). Based on these results, however, it is not unreasonable to assume that FWs that cover a larger proportion of pond surface area may contribute to significant sediment capture through physical trapping and also by encouraging gravity sedimentation.

Table 4. Summary of biomass, sediment, and nitrogen mass captured by the FW. Nitrogen mass is separated by biological (i.e., plant growth) and physical processes (i.e., sediment accumulation on the plant roots and underside of the mat). Sediment was washed from the roots and plant cup simultaneously and is referred to here as "per plant-cup". All calculations were based on surface area of the floating mat (50 m^2) and its deployment period of 137 growing season days. The number of individuals planted per species can be found in Table 2.

	FW Component						
	Plant Species					**Mat**	
	Alisma	***Carex***	***Iris***	***Juncus***	***Pontederia***		**Total**
Biomass (g)							
per plant	0.42	3.29	2.51	−0.79 *	2.02	—	—
FW Total	84.17	1151.79	425.94	−392.80 *	584.78	—	2684.16
Nitrogen—Plant Biomass (g)							
per plant	0.014	0.068	0.042	0.032	0.041	—	—
FW total	3.03	24.77	6.82	19.11	12.06	—	65.80
Sediment (g)							
per plant-cup	0.89	1.44	1.16	1.21	1.04	—	—
FW Total	178.98	503.40	196.75	606.06	301.96	1312.97	3100.12
Nitrogen—Sediment (g)							
per plant-cup	0.03	0.04	0.04	0.03	0.04	—	—
FW Total	6.79	14.00	7.28	15.55	10.48	71.47	125.57

Nitrogen Budget

FW plant uptake was responsible for just 34.4% of nitrogen removal by mass. The remaining nitrogen was contained in sediments captured by the hanging root matrix (28.3%) and accumulated on the FW mat (37.3%). Few other studies have investigated sediment capture through physical processes on FWs and none have analyzed the sediment for nitrogen content so it is not possible to compare these results with others (Kadlec and Wallace, 2009). Regardless, the FW as a whole removed a total of 191 g of nitrogen from the pond at a rate of 0.028 g/m^2/day (Fig. 5).

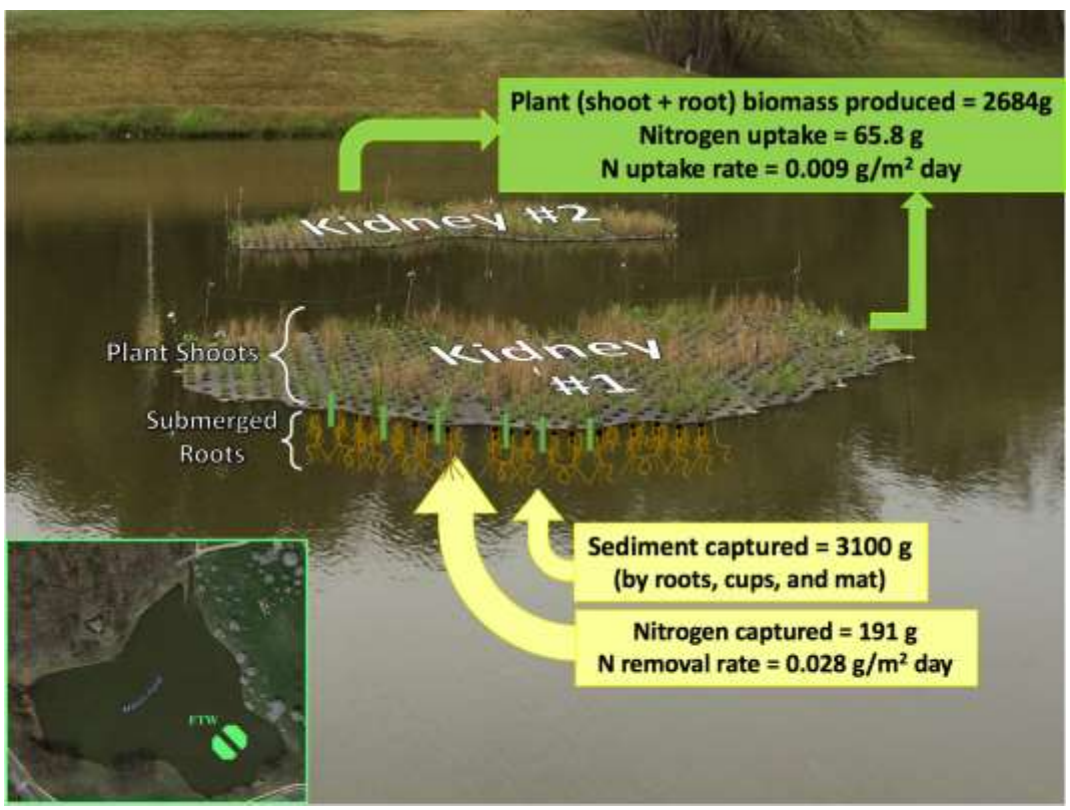


Fig. 5. Summary of FW performance in biomass production, nitrogen uptake, and sediment capture. Nitrogen uptake and removal rates were calculated based on the surface area of the floating mat (50 m^2) and its deployment period of 137 growing season days. Inset: overhead view of Mason Pond showing FW location on the pond.

Implications of the Study and Recommendations

Although the Mason Pond FW was a small case study, it adds further evidence that these systems can augment the performance of stormwater wet ponds as a means of improving water quality of urban runoff. Nitrogen and sediment were captured, albeit in small amounts by a low-cost, low-maintenance system. In addition, the FW provided a new habitat for wildlife in and around the pond. During bi-weekly monitoring, a variety of wildlife was observed utilizing the FW. Two resident green herons, undeterred by the waterfowl fence, quickly learned to use the FW as a platform from which to prey on the schools of small fish that swam around and underneath it. Spiders and ants also appeared to have colonized the artificial island, while dragonflies and damselflies were observed using the waterfowl fence posts as both molting posts and hunting perches. Lastly, frogs and turtles were frequent visitors, sunning themselves on the surface of the mat. It is likely that more wildlife was utilizing the FW than was observed as Wang et al. (2015) reported the presence of at least 22 species of animals and insects on or near their FW (located less than two miles from Mason Pond).

For these reasons, FWs present a sustainable option for improving the function of wet stormwater ponds—especially in cases where funds may be unavailable for expensive retrofits. Despite this, practitioners may face challenges from municipalities or local governing bodies when attempting to deploy FWs as they are a relatively new technology. For example, the project initially faced skepticism and opposition from the campus facilities department when permission was requested to deploy the FW on Mason Pond. Although permission to deploy the FW on Mason Pond was eventually granted, it was only on the condition that the FW was very small and would be removed by the end of the summer. By limiting the size of the FW and length of deployment, the scope of the study was fairly reduced. Once deployed, however, the FW project was well-received by the public and administrators alike, suggesting that the skepticism largely stemmed from unfamiliarity with the technology. In fact, the attention received by the FW provided multiple opportunities for outreach and education regarding stormwater issues several nearby municipalities inquired about the potential for FW use in their watershed. While SCMs and BMPs provide a means of managing and treating stormwater, urban water pollution will not be significantly reduced without an informed public. The Mason Pond FW, located on a pond near the center of campus, provided multiple opportunities to educate the public about urban water pollution and/or sustainable stormwater management, and how they can help reduce it. A local high school class was invited to help deploy and learn about the FW which gave us an opportunity to discuss stormwater related issues in urban watersheds with young students who had little exposure to such issues (see Ahn (2016) for more information). During weekly monitoring after launching the FW, it was not uncommon to receive questions about the FW from curious passersby. While nearly everyone encountered had never heard of FWs before, most people intuitively understood its purpose and were receptive to learning more about stormwater issues.

In this era of ever-increasing urbanization, stormwater issues caused by such development are largely overlooked by the general public. Furthermore, climate change is predicted to increase the frequency of storm events which will further compound issues related to urban stormwater management. It is critical to work together to learn more about and to adopt green infrastructure such as FW to create sustainable ways of managing nutrients and pollutants in both natural and man-made waterways. The Mason Pond FW, developed as the so-called “Rain Project”, presented a successful case of addressing an important contemporary environmental issue (i.e., sustainable stormwater management) through experiential learning, community building, and public outreach (Ahn, 2016; NBC4 Washington, 2015, p. 4).

Conclusions

Although biomass production on the Mason Pond FW was relatively unremarkable, the observed rate of nitrogen uptake by plant biomass was comparable to rates reported in other FW studies. Distribution of nitrogen between the roots and shoots generally did not differ between pre- and post-deployment which suggests that harvest strategies should focus on maximizing biomass removal rather than attempting to capitalize on temporal changes in biomass nutrient distribution. Nitrogen was also found in the sediments trapped by the plant roots and FW structure itself, however it is unlikely that such captured sediments would be regularly harvested as it is a labor intensive process that requires removal of the entire FW system. Furthermore, the plant roots and FW structure may have provided a surface for the development of bacterial biofilms that could contribute to overall nitrogen removal via nitrification and denitrification pathways. Unfortunately, funding limitations prevented us from investigating this aspect of FW performance. FWs represent a promising new technology for the augmentation of stormwater wet ponds, however their long-term effectiveness remains unclear. Further investigation is needed to determine the performance of FW systems over multiple growing seasons, clarify nutrient distributions within plant matter to inform harvest strategies, and examine the contribution of biofilms to overall nutrient removal.

CHAPTER TWO

To further investigate the nitrogen capture performance of the Mason Pond FW system a system model was developed which allowed for the estimation of the nitrogen removal efficiency of the Mason Pond FW and also allowed for the simulation of FW performance under a variety of conditions beyond the scope of the Mason Pond case study described in Chapter One. The modeling process and results are described in the following sections.

Modeling Methodology

Simulation Methods

Using STELLA 10.1, modeling software that allows users to create and test system models with a proprietary "visual programming language," a model was developed for the Mason Pond FW system. Model components used in this study are state variables (rectangles), flows (arrows connecting state variables), and converters (circles) (iSee Systems, 2016). State variables represent mass or volume variables within the system (i.e., pond volume or biomass). Flows contain equations that define the state variables using the output of other flows as well as values contained in converters (i.e., TN concentration). Converters and flows are connected via red arrows.

The FW ecosystem model presented here is comprised of three interdependent and interacting submodels that represent the major components of the FW-pond ecosystem: a hydrological submodel, a plant growth submodel, and a nitrogen submodel. The hydrological submodel simulates the flow of water into and out of the pond. The hydrologic submodel drives the import and export of pond nitrogen which is simulated by the nitrogen submodel. Lastly, the plant growth submodel simulates the production of biomass on the FW which influences pond nitrogen content in the nitrogen submodel. Many of the parameters and coefficients contained in the model were calculated from the results of the Mason Pond FW case study while others were determined by literature review or calculation. For example, the run period of the model was set to mimic the ~20-week deployment period of the Mason Pond FW (week 18-38). The model structure was verified by comparing its output to results reported in another FW study by Lynch et al. (2015). A more detailed explanation of the submodels and components is presented in the following sections.

A review of the limited research available on floating wetlands suggest that the key factors affecting nitrogen capture are biomass production and hydraulic residence time (Marimon et al., 2013; McAndrew et al., 2016; Pavlineri et al., 2017). Hydraulic residence time in stormwater ponds depends on individual pond design and influent/effluent flow rates which are primarily driven by local storm activity. Biomass production is influenced by the productivity of the FW plants as well as the proportion of pond surface area covered by the FW. The model was therefore used to test the impact of surface area coverage, HRT, and productivity on nitrogen removal performance in the FW-pond ecosystem.

Several assumptions were made in the creation of this model to both overcome a lack of available data and as part of the modeling process which requires the simplification a complex system:

1. Mason Pond is assumed to have a batch-fed hydrologic regime similar to the mesocosm set-up used in the majority of FW studies (Pavlineri et al., 2017);
2. FW plants experience an 8-week acclimatization period after deployment before growth begins (Deering, 2016; Lynch et al., 2015);
3. Root biomass production occurs as a ratio of shoot biomass production informed by field observations;
4. Nitrogen concentration in the pond does not limit or drive biomass production
5. Microbial denitrification by FW biofilms is not included in this model.

Model Structure

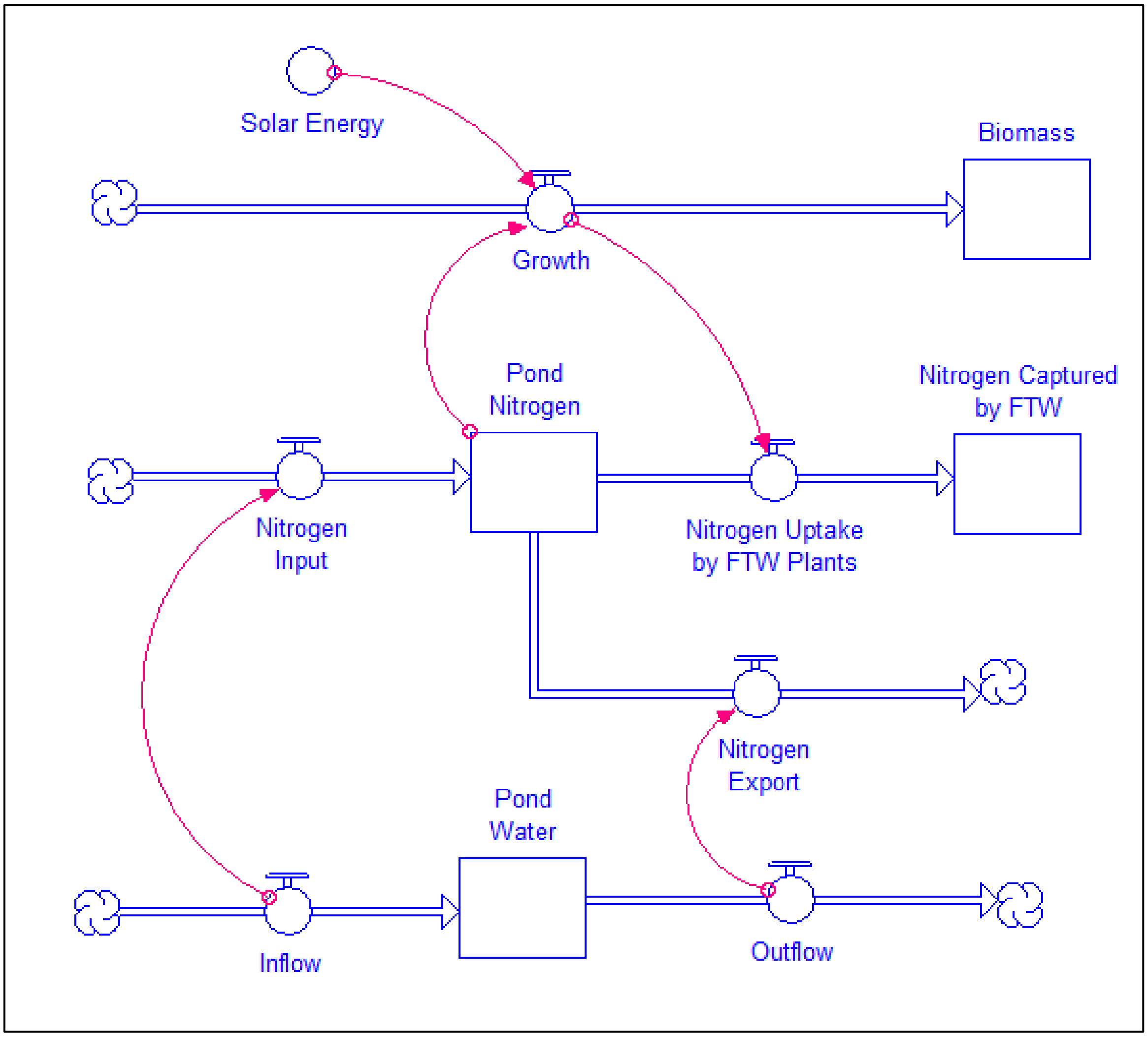


Fig. 6. Conceptual STELLA diagram of the FW-pond ecosystem model.

A conceptual model of the FW-nitrogen system is shown in Fig. 6 above and provides an overview of the interconnectivity of the process within the model. The differential equations, state variables, forcing functions, and associated variables/parameters that make up the model are summarized in Table 5 and Table 6 below followed by a detailed description of the model.

Table 5. Differential equations used in the FW ecosystem model.

Hydrology submodel	
d(Water volume)/dt(t) =	Inflow - Outflow
Where	
V	Volume of water in Mason Pond (m^3)
Inflow	Simulated inflow into Mason Pond (m^3/week) PULSE(Fill Volume, StartTime, HRT)
Outflow	Simulated outflow from Mason Pond (m^3/week) Pond volume fills and empties at the interval set by HRT (hydraulic residence time) PULSE(V, StartTime, HRT)
Plant Growth submodel	
d(Shoot Biomass)/dt =	Weekly Shoot Growth
d(Root Biomass)/dt =	Weekly Root Growth
Where	
Shoot Biomass	Shoot biomass on the FW (g)
Root Biomass	Root biomass on the FW (g)
Weekly Shoot Growth	Solar * FW Area * (1/R) * FW se (g/week)
Weekly Root Growth	Shoot Growth * Root Ratio (g/week)
Solar	(4000-1000*COS(2*PI*TIME/52)) (kcal/m^2/week)
FW se	Solar efficiency of the FW (calibrated value)
R	Energy per biomass ratio (kcal/g)
Water SA	Surface area of Mason Pond (m^2)
FW SA%	FW coverage as a percent of Water SA (%)
FW Area	Water SA * FW SA%
Toggle Growth	Toggles growth after plant establishment period
Nitrogen submodel	
d(Water N)/dt =	N in - Plant N Uptake - N out (g)
d(Plant Captured N)/dt =	Plant N Uptake (g)
Where	
N in conc	NORMAL(N in mean, N in SD) (g/m^3)
N in	Water in * N in conc (g/week)
N out	Water out * Water N conc (g/week)
Plant N Uptake	(Shoot Growth * Shoot N Conc) + (Root Growth * Root N Conc) (g/week)

Table 6. State variables, forcing functions, and variables/parameters used in the FW ecosystem model.

Name	**Description**	**Value/Units**	**Source**
State variables			
V	Volume of water in pond	m^3	Calculation, Mason Pond
Shoot Biomass	Mass of plant shoots on FW	g	Calculation, Mason Pond
Root Biomass	Mass of plant roots on FW	g	Calculation, Mason Pond
Water N	Mass of total nitrogen contained in pond	g	Calculation, Mason Pond
Mass N Out	Mass of total nitrogen exported from pond	g	Estimated
Plant Captured N	Mass of nitrogen captured by macrophyte biomass production	g	Calculation, Mason Pond
Forcing functions			
Inflow	Water inflow	m^3/wk	
N in	Nitrogen load	g/wk	
Solar	Average amount of solar energy reaching earth surface	kcal/m^2/wk	
Parameters and Coefficients			
Fill Volume	Volume of water input to pond	6532 m^3	Calculation, Mason Pond
HRT	Hydraulic residence time	wk	Estimated
Water SA	Surface area of pond	7100 m^2	Field
R	Energy per biomass ratio	4.1 kcal/g	Boyd (1970)
FW se	Solar efficiency of the floating wetland	0.0025, 0.025	Calibrated; Ahn and Mitsch
Root Ratio	Ratio of root biomass produced per shoot biomass produced	1.38 ± 0.87 g/g	Calculation, Mason Pond FW
N in mean	Mean concentration of total nitrogen in inflow	1.19 g/m^3	GMU (2016), Wang and Sample (2014)
N in SD	Standard deviation of the concentration of TN in inflow	0.27 g/m^3	Wang and Sample (2014)
Water N Conc	Concentration of TN in pond	g/m^3	Calculation
Shoot N Conc	N content of shoot biomass	1.90 ± 0.15%	CHN Analysis of Mason Pond FW plants
Root N Conc	N content of root biomass	1.79 ± 0.32%	CHN Analysis of Mason Pond FW

			plants
Total N Input	Total mass load of N input to pond during simulation	g	Calculation
% Plant Capture	Percent of Total N Input captured by plant biomass production	%	Calculation

Hydrology Submodel

The hydrology submodel (Fig. 7) consists of a single state variable, water volume (*V*), defined by the inflow and outflow of a fixed volume of water (*Fill Volume*). The 'Fill Volume' is based on field measurements of the volume of Mason Pond (Table 1). The hydrologic regime of the pond was simplified to resemble a batch-fed bioreactor due to a lack of available data on the inflow and outflow rates of Mason Pond. Using the EPA's National Stormwater Calculator, the HRT of Mason Pond during the study period was estimated at approximately 8.5 days. The EPA's Preliminary Data Summary of Urban Stormwater Management suggests that the HRT of wet ponds should be at least 14 days to achieve moderate sediment capture which suggests the Mason Pond's performance may be somewhat impaired (USEPA, 1999a). HRT values of 1, 2, and 3 weeks were selected to test the nitrogen capture performance of an FW across a range of hydrologic conditions. In the model, a PULSE function drains and then refills the pond volume state variable (*V*) at the interval defined by HRT. This submodel greatly oversimplifies the hydrological dynamics of a real-world retention basin, but it will allow us to gauge the nutrient removal performance of an FW on both degraded and functioning ponds.

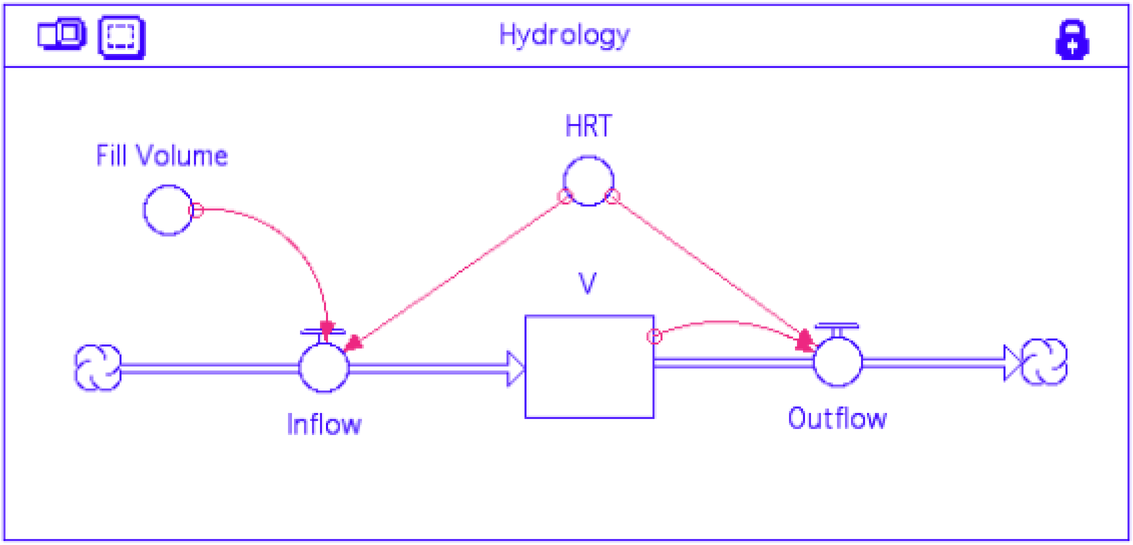


Fig. 7. STELLA diagram of the hydrology submodel

Plant Growth Submodel

The plant growth submodel consists of two state variables that represent the amount of shoot and root biomass produced during the FW's deployment. The structure of this submodel was adapted from a model developed by Ahn and Mitsch (2002) to investigate phosphorus dynamics in constructed wetlands.

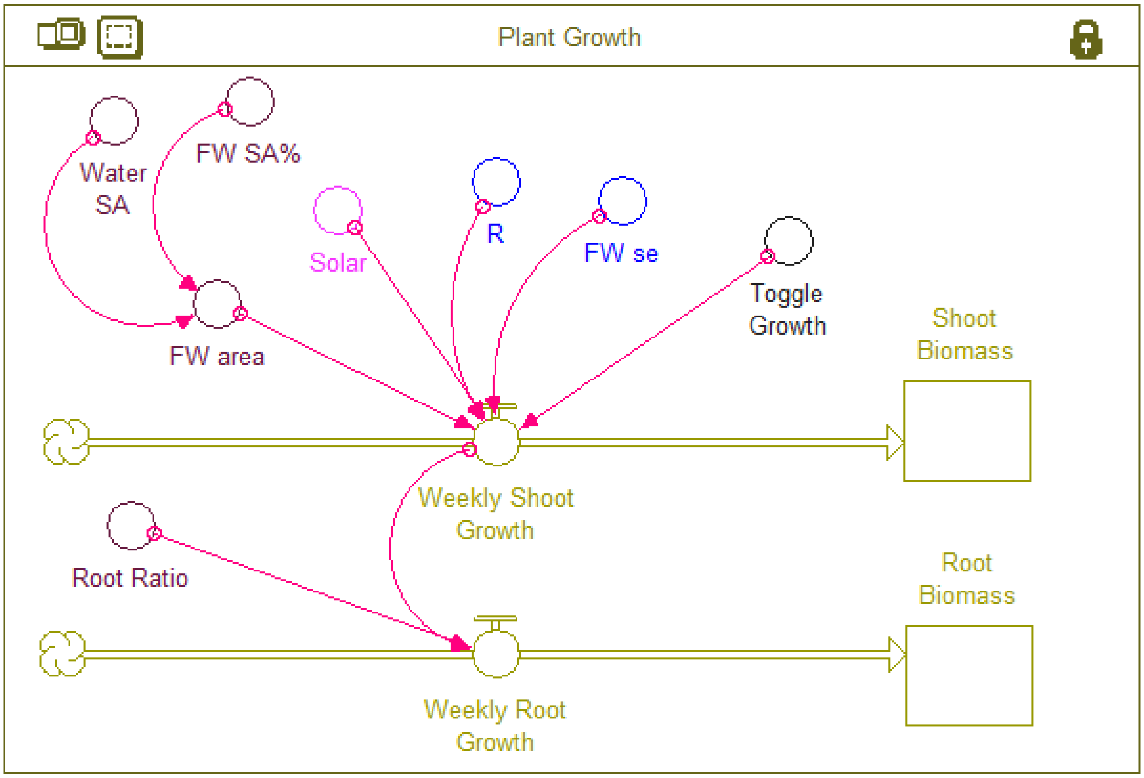


Fig. 8. STELLA diagram of the plant growth submodel

Shoot biomass production in the plant submodel is primarily forced by solar energy (*Solar*) which is approximated by a cosine function to simulate the mean solar energy reaching the earth's surface in Northern Virginia in a week (Table 5; National Renewable Energy Laboratory, 2016). The rate at which this solar energy is converted to biomass is governed by the solar efficiency of the FW (*FW se*).

To convert solar energy (kcal/m^2) into plant biomass (g/m^2) an energy to biomass ratio (*R*) of 4.1 kcal/g was used (Boyd, 1970). This value was determined by calorimetric analysis of wetland macrophytes and was used in two wetland models detailed by Ahn and Mitsch (2002) and Wang and Mitsch (2000). Before using this literature derived ratio, an attempt was made to determine a ratio unique to the biomass harvested from the

FW through analysis at a third party forage testing lab however the reported energy content was not the total calorimetric energy content of the plants. The forage analysis instead reported the energy content that is metabolically accessible to cattle or the "net energy for maintenance." The ratio reported by Boyd (1970) was therefore employed in this model. Areal shoot production was then multiplied by the FW surface area (*FW area*), derived as a proportion of pond surface area (*Water SA)*, to estimate the total biomass produced on the FW. Root production was calculated as a ratio to shoot production (*Root Ratio*) using the built-in NORMAL function fed with the mean and standard deviation in observed root to shoot ratio (Table 6). Lastly, a binary toggle (*Toggle Growth*) is used to activate plant growth after eight weeks of deployment in order to simulate approximately the eight-week long plant acclimatization period observed by Lynch et al. (2015) and Deering (2016).

Nitrogen Submodel

The nitrogen submodel (Fig. 9) consists of three state variables: the mass of nitrogen in the pond water (*Water N*), the mass of nitrogen exported from the pond (*Mass N Out*), and the mass of nitrogen captured by plant growth (*Plant Captured N*). Water nitrogen is calculated as the volume of influent entering the pond (*Inflow)* multiplied by the concentration of nitrogen in the influent (*N in Conc*). The influent nitrogen concentration is generated with a built-in NORMAL function fed with the mean and standard deviation of the nitrogen concentrations reported in the university's latest MS4 (George Mason University, 2016) report and those observed in a stormwater pond located approximately 1 mile from Mason Pond (Table 6; Wang and Sample, 2014).

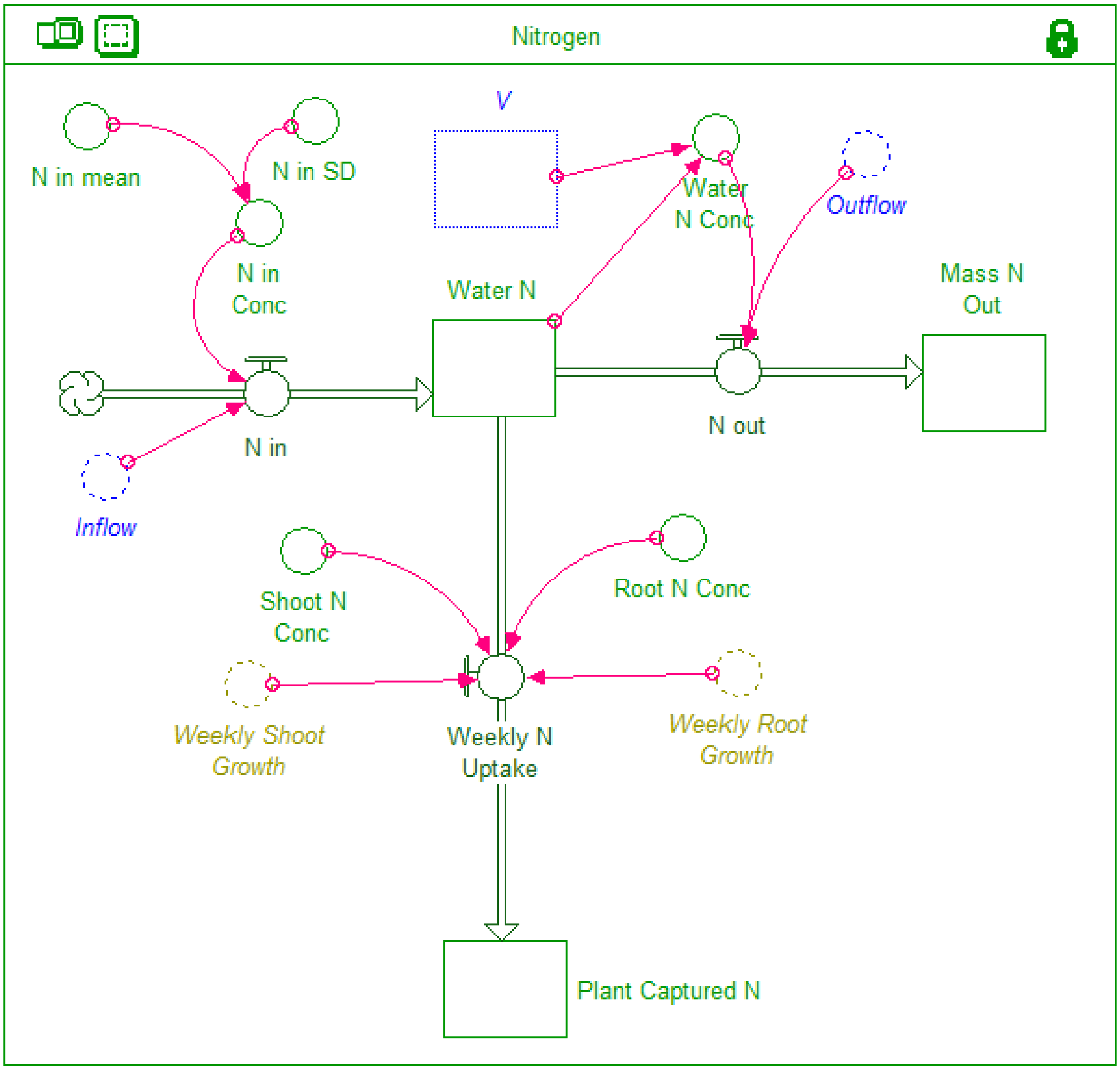


Fig. 9. STELLA diagram of the nitrogen submodel

Plant Captured N is defined as the mass of nitrogen assimilated by biomass in each time-step (*Weekly N Uptake*). Nitrogen uptake is calculated as the product of biomass production (*Daily Shoot Growth*, *Daily Root Growth*) and the root and shoot nitrogen concentration (*Shoot N Conc*, *Root N Conc*). The values for concentration of nitrogen in the plant shoots and roots are generated by the built-in NORMAL function fed by the mean and standard deviation for nitrogen content observed in the most productive species observed the Mason Pond FW (Table 6 and Table 7). The mass of nitrogen exported from the pond (*Mass N Out*) is the amount of nitrogen remaining at the end of the hydraulic residence interval (*N out*) calculated as a product of effluent volume (*Outflow*) and pond nitrogen concentration (*Water N Conc*).

Table 7. Summary of shoot/root biomass production and nitrogen content of most productive species planted on the Mason Pond FW.

Plant Species	**Biomass Produced (g/m^2)**	**Nitrogen Content (%)**	
Carex		**Initial**	**Final**
Shoot	36.04	1.28%	1.73%
Root	63.34	0.53%	1.86%
Iris			
Shoot	25.22	2.31%	1.96%
Root	50.45	1.34%	1.44%
Pontederia			
Shoot	43.96	2.87%	2.02%
Root	16.94	1.04%	2.07%
Mean			
Shoot	35.07	2.15%	1.90%
Root	43.57	0.97%	1.79%

Model Calibration and Verification

Model calibration involves altering select parameters within the model until the simulated output agrees with observed field data or values reported in the literature. For the model described here, the plant growth submodel was calibrated to fit simulated biomass production to production observed on the Mason Pond FW. All of the variables influencing shoot biomass production are in the plant growth submodel are supported by the literature with the exception of solar efficiency. Ahn and Mitsch (2002) utilized a solar efficiency of 2.5% that was originally estimated by Wang and Mitsch (2000) in a model of phosphorus dynamics at the Des Plaines River Wetlands. At this level of solar efficiency, the model simulated biomass production well beyond what was observed on the Mason Pond FW. As a result, solar efficiency was calibrated until modeled biomass production agreed with the mean observed productivity of the most productive species deployed on the Mason Pond FW: *Carex stricta* (Table 7). This calibration process yielded a solar efficiency of just 0.25%. Solar efficiency was likely higher in Ahn and Mitsch's (2002) constructed wetland model for several reasons:

(1) plant density is not fixed in the constructed wetlands whereas the Beemat has a hard limit of ~30 plants per square meter;

(2) the constructed wetland models assumed an existing standing stock whereas the Mason Pond FW was stocked with small, young plugs;

(3) relatively poor growth of the wetland plants on the Mason Pond FW, possibly due to limited nutrients and/or pollutants transported from surrounding parking lots.

While few FW studies have reported biomass production results, at least two studies have reported biomass production two to three times higher than was observed on the Mason Pond FW (Wanielista et al., 2012; White and Cousins, 2013). To reflect this, a more productive FW was also tested using the literature supported solar efficiency ratio of 2.5% which results in greater simulated biomass production on par with values reported by Tanner and Headley (2011) and White and Cousins (2013).

Following calibration, the model estimated FW biomass production at a rate of 0.67 ± 0.06 g/m^2/day which agrees well with the biomass production rate observed for *Carex stricta* (0.73 g/m^2/day), the most productive species planted on the FW (Table 7). Similarly, the simulated nitrogen uptake rate of 0.010 g/m^2/day falls within 10% of the observed nitrogen uptake rate of 0.011 g/m^2/day.

To test whether the model accurately simulated biomass production on the Mason Pond FW, model verification was performed with a Beemat-based FW study that took place in southern Virginia (Lynch et al., 2015). This study was ideal for model verification as it reported results similar to those reported by McAndrew et al. (2016) such as biomass produced and plant nitrogen content. Furthermore, Lynch et al. (2015) used batch-fed mesocosms similar to the hydrologic regime simulated in the model. To verify the model structure, model variables (e.g., *Fill Volume*, *N in mean*, and *Root/Shoot N Conc*) were set to values reported in Lynch et al. (2015) before solar efficiency (*FW se*) was calibrated until biomass production was accurately simulated. As shown in Table 8, the verification simulations produced areal final biomass within 2% of results reported by Lynch et al. (2015) while nitrogen removal efficiency of FW plants fell within 10%.

Table 8. Verification of performance of the FW model when initialized with values reported in Lynch et al. (2015) (i.e., mesocosm volume, N loading, FW surface area). Model output from the verification runs (n = 5) fell within 2% and 10% for mean final biomass and FW N removal reported by Lynch et al. (2015).

	Lynch et al. (2015)	Modeled	% Difference
Mean Final Biomass (g/m^2)			
Shoot	142.9	141.8	0.77%
Root	12.4	12.2	1.61%
FW N Removal (%)	29.0%	20.71%	8.29%

Sensitivity Analysis

Sensitivity analysis is the process by which models are used to test hypotheses. Through alteration of individual variables/parameters in the model and observing the effect on simulation output it is possible to gain a deeper understanding of the system in question and identify key variables that influence output. Furthermore, sensitivity analysis provides a way to predict or estimate system behavior beyond what may be feasible to test experimentally. For this model a sensitivity analysis was performed on hydraulic residence time, surface area coverage, and solar efficiency to determine their relationship to FW TN removal efficiency.

Sensitivity analysis was performed on the hydraulic residence time of the pond to determine whether longer residence times resulted in greater overall nitrogen removal efficiency. FW performance was simulated at pond HRT of 1, 2, and 3 weeks. To determine the effect of FW surface area coverage on nitrogen removal efficiency, surface area coverage was tested at 0.704% (the coverage proportion of the Mason Pond FW), 5%, 10%, 25%, 50%, and 100%. These values were chosen because some studies suggest that FW coverage of 10-20% is enough to achieve significant nitrogen removal performance, however most FW studies have tested performance at 50-100% (Deering, 2016; Marimon et al., 2013; Pavlineri et al., 2017). Such extreme levels of FW coverage are likely unfeasible as stormwater ponds regularly span several acres. Instead, extreme levels of coverage (>50%) were tested to allow for more direct comparison of the model results with the literature. Lastly, nitrogen removal efficiency was tested at two levels of solar efficiency to simulate both the low productivity observed in the Mason Pond FW as well as the high productivity observed in other FW studies.

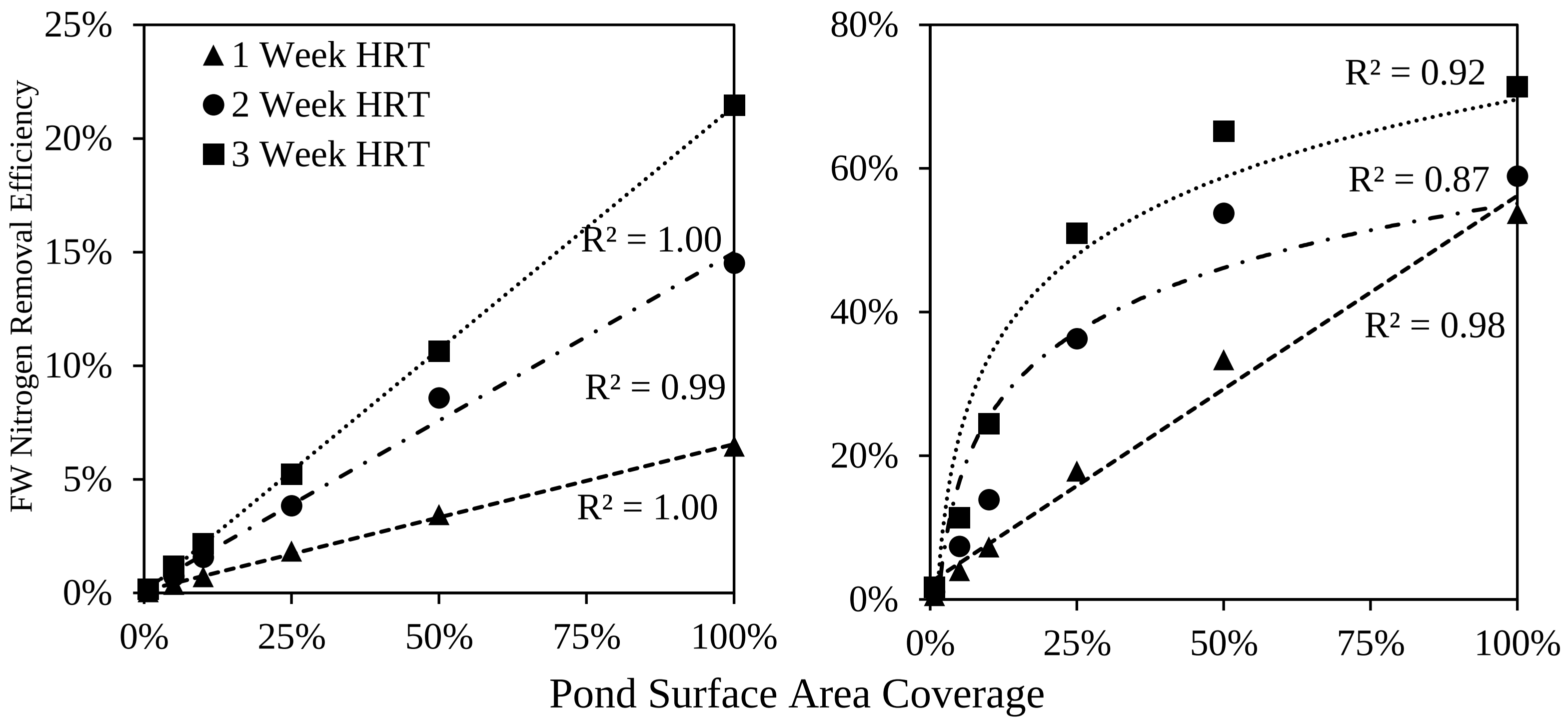
Table 9. Summary of the number of simulations run to develop and test the FW ecosystem model.

Simulation Type	**# of Runs**
Calibrating biomass and nitrogen submodel	
(1) Calibration of the model with Mason Pond FW data	10
Verification	
(2) Verifying model performance with Lynch et al. (2015) data	10
Sensitivity analysis (Mason Pond FW)	
(2) FW surface area coverage	5
(3) Length of deployment period	5
(4) Hydraulic residence time	5

Simulation Results and Discussion

Hydrologic Residence Time

Model simulations show FW nitrogen removal increased with hydraulic residence time (Fig. 10). The low-productivity simulations, calibrated to biomass production reported in McAndrew et al. (2016), estimated mean nitrogen removal efficiency of the Mason Pond FW at approximately 0.052% for an HRT of one week (est. Mason Pond HRT = 8.6 days). When HRT was increased to three weeks, N removal efficiency also increased roughly three-fold to 0.162% suggesting a linear relationship between removal efficiency and HRT. This trend is a product of the batch-reactor-style hydrologic submodel where the total load of TN introduced to the system is governed by the HRT while plant uptake of TN does not vary. For example, at 0.70% surface coverage the model simulations estimate that FW plants captured an average of 75 g N regardless of the length of HRT. On the other hand, the total mass N introduced to the pond over the simulation period decreased from 148 kg N at an HRT of one week to 48.3 kg N at an HRT of three weeks, thus increasing removal efficiency. A positive relationship between HRT and TN removal performance is also apparent in the results from the high-productivity simulations—until FW coverage surpasses 50% at longer HRTs (Fig. 10b). At high surface coverage (≥50%) and HRT greater than one week, TN uptake by the plants begins to deplete almost all nitrogen in Mason Pond. A FW-pond ecosystem model developed by Marimon et al. (2013) also showed nitrogen depletion when FW coverage exceeded 50%, though no HRT was reported for the test ponds. In both cases, N depletion occurs because N removal rate exceeds the N loading rate. Low TN concentration (< 2 mg/L) in the test ponds likely exacerbated this effect (Marimon et al., 2013; McAndrew et al., 2016). It is unlikely that this phenomenon would be encountered in practice as the construction and maintenance of such large FWs is impractical given the large size of most stormwater ponds. Furthermore, ponds that exhibit an HRT of two weeks or more meet EPA guidelines for stormwater pond design for water quality function and would not need such a large FW to meaningfully augment pond performance (USEPA, 1999a). These results suggest that FW design should be pond-specific and take HRT and nutrient loading into account.



a) **b)**

Fig. 10. Model results for mean nitrogen removal efficiency (n = 5) of Mason Pond FW at a solar efficiency ratio of (a) 0.25% and (b) 2.5% across the range of hydraulic residence times (1, 2, and 3 weeks) and FW surface area coverages (0.70%-100%). The model output shows a positive relationship between nitrogen removal efficiency and hydraulic residence time, surface area coverage, and productivity.

Surface Area Coverage

FW surface area coverage is a key factor influencing nitrogen removal as it limits the area available for plant growth. To test the effect of surface area on nitrogen removal sensitivity analysis was performed across a range of coverage levels from 0.70% to 100%. As with HRT, nitrogen removal efficiency increased with surface area coverage (Fig. 10). This is because an increase in FW surface area results in a corresponding increase in total plants deployed on the FW and, by extension, nitrogen removal performance. The low-productivity simulations show a strong linear relationship between FW surface area coverage and TN removal ($R^2 \geq 0.95$, $p < 0.001$). As noted in the previous section, the high-productivity simulations show a linear relationship for an HRT of one week ($R^2 = 0.97$, $p < 0.001$) while a logarithmic relationship emerges when HRT is greater than one week due to N depletion (Fig. 10b). This trend suggests that such high surface coverage may be overkill in some cases. A review of available literature on FW nutrient capture, for example, shows control replicates exhibited an average TN reduction of 20.7% at an HRT of one week or less while FW treatments exhibited a reduction of 45% at full surface coverage (Pavlineri et al., 2017). The model, however, suggests a highly productive FW covering just 25% of a pond with an HRT of one week may be able to augment TN removal by ~18% through plant uptake alone (Fig. 10b) which is in the ballpark of mean N capture at 100% coverage. Furthermore, full surface coverage may impact pond biogeochemical cycles as light penetration and atmospheric oxygen exchange is virtually eliminated (Wetzel, 2001; Zhou and Wang, 2010).

FW Solar Efficiency

Solar efficiency had the greatest impact on nitrogen removal performance of all variables tested with the model as it significantly influenced FW biomass production. The model estimated the biomass production rate at 0.66 $g/m^2/day$ at low productivity and 5.9 $g/m^2/day$ at high productivity. The low-productivity simulations resulted in mean areal N capture of 1.51 g/m^2 at a rate of 0.011 $g/m^2/day$ while the high-productivity simulations estimated capture at 14.14 g/m^2 N at a rate of 0.10 $g/m^2/day$. By comparison, FW studies have observed plant N uptake rates between 0.0015-2.8 $g/m^2/day$ with a median uptake rate of 0.078 $g/m^2/day$ (Wang et al., 2014). While both the low and high-productivity simulations produced N uptake rates within the observed rates, the high-productivity model appears to be more representative of performance reported in the literature. The low-productivity simulations suggest that FW performance is quite poor even at 100% coverage (Fig. 10a). On the other hand, the high-productivity simulations estimate much greater N removal at lower coverage. For a pond with a low HRT ($\leq$ 1 week), FW coverage of just 10-25% may result in N removal on the order of 7-18% by plant uptake alone (Fig. 10b). It is possible that actual removal is higher as biofilm development has been observed on the root zone of FW plants that may also aid in nutrient removal or transformation (Headley and Tanner, 2012).

Plant N uptake rate estimated at low-productivity (0.011 $g/m^2/day$) matched N uptake reported by Lynch et al. (2015) for a mesocosm-based FW with 65% coverage. Despite identical N uptake rates, the model indicates that the Mason Pond FW was less efficient than the FW system described in Lynch et al. (2015) at 4.5% and 11.45% respectively. The difference in efficiency can be attributed to the difference in treatment volume, 0.114 m^3 vs. 6532 m^3, relative to actual surface area covered by each FW. At 65% coverage, the FW utilized by Lynch et al. (2015) sits on a relatively smaller volume of water than an FW covering the same proportion of Mason Pond. This highlights the influence that pond design parameters have on FW performance and why they must be considered to design effective FWs.

Application of the Model for Sustainable Stormwater Management

The results of the model simulations suggest that while FWs can significantly augment the nitrogen capture function of stormwater retention ponds, performance is highly dependent on the productivity of the plants on the FW. For example, an FW with low productivity would have to cover 100% of Mason Pond to achieve about the same removal performance as a highly productive FW covering just 5% of the pond surface (Fig. 10). The literature shows that FW productivity varies greatly because it is influenced or limited by a myriad of factors including climate, plant species, and water quality (Headley and Tanner, 2012; Pavlineri et al., 2017; Wang et al., 2014). Unfortunately, few FW studies report data relating to FW plant productivity which makes it difficult to identify pond conditions and species that typically result in predictable growth [e.g., Lynch et al. (2015) and White and Cousins (2013)]. Instead, FW studies typically gauge FW performance through comparison of nutrient concentration between influent and effluent; often without including a control replicate (Pavlineri et al., 2017). Without plant data, differentiation of nutrient capture by the FW from other processes within the system such as settling or denitrification is difficult or impossible. For this reason, future FW studies should consider including data regarding biomass production and plant nutrient content in addition to water chemistry analyses.

Although sensitivity analysis was performed on a range of FW surface area coverage up to and including full coverage, it is likely impractical to deploy an FW system covering more than 25-30% of the pond surface due to the large size of retention ponds (>1000 m^2). An FW covering 25% of Mason Pond, for example, would span over 1700 m^2. Such a large FW would require significant initial investment as commercial systems cost between \$38 and \$377 per square meter excluding plants and anchoring system (Lynch et al., 2015). While the FW structure itself represents a one-time cost, FW plants may have to be replaced periodically as FW plants must be harvested to permanently remove nutrients from the stormwater system. For this reason, it is important to design an FW of the appropriate size for a given pond to maximize removal and minimize costs. Further research is needed to determine the lifespan of commercial FW systems. This would allow for estimation of the amount of time needed to recoup the costs of an FW system in terms of nutrients removed.

Conclusions

The simulation results suggest that FWs may be a sustainable technology for improving stormwater quality. At high productivity, an FW covering 25-30% of the pond surface can remove up to 20% of pond nitrogen through direct plant uptake alone. Consistent nitrogen removal performance, however, depends on a variety of factors from pond size and hydrology to FW design and primary productivity of the stocked plants. If designed with these factors in mind, FWs may be able to improve the nitrogen capture performance of degraded stormwater ponds and may be employed as part of the patchwork of green infrastructure used to meet TMDL goals.

There is currently a lack of studies that have tested the efficacy and long-term feasibility of this type of technology in real-world stormwater ponds. Further study is needed to determine what factors impact FW productivity and whether this technology is cost-effective over the long-term. Nonetheless, the ecosystem model of a FW described in this paper may provide a useful tool for landowners, developers, or municipalities seeking to estimate the potential performance of an FW to be designed and/or deployed on a given pond as part of sustainable stormwater management.

APPENDIX

STELLA code for the FW-Mason Pond system model:

Mass_N_out(t) = Mass_N_out(t - dt) + (N_out) * dt

INIT Mass_N_out = 0

INFLOWS:

N_out = Outflow*Water_N_Conc

Plant_Captured_N(t) = Plant_Captured_N(t - dt) + (Plant_N_Removal) * dt

INIT Plant_Captured_N = 0

INFLOWS:

Plant_N_Removal =

Shoot_Growth*(Shoot_N_Conc/100)+Root_Growth*(Root_N_Conc/100)

Root_Biomass(t) = Root_Biomass(t - dt) + (Root_Growth) * dt

INIT Root_Biomass = 0

INFLOWS:

Root_Growth = Shoot_Growth*Root_Ratio

Shoot_Biomass(t) = Shoot_Biomass(t - dt) + (Shoot_Growth) * dt

INIT Shoot_Biomass = 0

INFLOWS:

Shoot_Growth = Toggle_Growth*FW_Area*FW_se*Solar*(1/R)

Water_N(t) = Water_N(t - dt) + (N_in - N_out - Plant_N_Removal) * dt

INIT Water_N = 0

INFLOWS:

N_in = Inflow*N_in_Conc

OUTFLOWS:

N_out = Outflow*Water_N_Conc

Plant_N_Removal =

Shoot_Growth*(Shoot_N_Conc/100)+Root_Growth*(Root_N_Conc/100)

Water_Volume(t) = Water_Volume(t - dt) + (Inflow - Outflow) * dt

INIT Water_Volume = 0

INFLOWS:

Inflow = pulse(Fill_Volume, starttime, HRT) {m^3/day}

OUTFLOWS:

Outflow = Pulse(Water_Volume, starttime, HRT)

Combined_Biomass_per_Area =

Shoot_Biomass_per_Area+Root_Biomass_per_Area

Combined_Growth = Shoot_Growth+Root_Growth

Cost_per_gram_N_Removed = if(Plant_Captured_N >

0)then(Total_Cost/Plant_Captured_N)else(Total_Cost)

Cost_per_Kilo_N = Cost_per_gram_N_Removed*1000

Day_of_Week = 1+((time/7)-int(time/7))*7

Depth = Water_Volume/Water_SA

Fill_Volume = 4260

FW_Area = Water_SA*(FW_SA%/100)

FW_SA% = 0.704

FW_se = 0.0004

HRT = 7

Mat_Cost = FW_Area*Mat_Cost_per_Area

Mat_Cost_per_Area = 42.51

N_in_Conc = normal(N_in_mean, N_in_SD)

N_in_mean = 1.19

N_in_SD = 0.27

Percent_Plant_Capture =

if(Total_N_Input>0)then((Plant_Captured_N/(Total_N_Input))*100)else(0)

Plant_Cost = FW_Area*Plant_Density*Plug_Unit_Price

Plant_Density = 30.2

Plant_N_per_Area = Plant_Captured_N/FW_Area

Plug_Unit_Price = 0.8

R = 4.1

Root_Biomass_per_Area = Root_Biomass/FW_Area

Root_N_Conc = 1.86

Root_Ratio = 1.75

Shoot_Biomass_per_Area = Shoot_Biomass/FW_Area

Shoot_N_Conc = 1.73

Solar = (4000-1000*COS(2*PI*(TIME+9)/365)) {kcal/square meter/day}

Toggle_Growth = if(Week_Counter >= 8)then(1)else(0)

Total_Biomass_Produced = Shoot_Biomass+Root_Biomass

Total_Cost = Mat_Cost+Plant_Cost

Total_N_Input = Mass_N_out+Plant_Captured_N

Water_N_Conc = if(Water_Volume>0)then(Water_N/Water_Volume)else(0)

Water_SA = 7100

Week_Counter = int((time-starttime)/7)

REFERENCES

Ahn, C., 2016. A creative collaboration between the science of ecosystem restoration and art for sustainable stormwater management on an urban college campus: Floating wetland ecosystem restoration interdisciplinary learning. Restor. Ecol. 24, 291–297. doi:10.1111/rec.12341

Ahn, C., Mitsch, W.J., 2002. Evaluating the use of recycled coal combustion products in constructed wetlands: An ecologic-economic modeling approach. Ecol. Model. 150, 117–140.

BeeMats, LLC, 2017. BeeMats: Floating Wetlands [WWW Document]. URL http://beemats.com/home.html

Borne, K.E., Fassman, E.A., Tanner, C.C., 2013a. Floating treatment wetland retrofit to improve stormwater pond performance for suspended solids, copper and zinc. Ecol. Eng. 54, 173–182. doi:10.1016/j.ecoleng.2013.01.031

Borne, K.E., Fassman-Beck, E.A., Winston, R.J., Hunt, W.F., Tanner, C.C., 2015. Implementation and Maintenance of Floating Treatment Wetlands for Urban Stormwater Management. J. Environ. Eng. 141. doi:10.1061/(ASCE)EE.1943-7870.0000959

Borne, K.E., Tanner, C.C., Fassman-Beck, E.A., 2013b. Stormwater nitrogen removal performance of a floating treatment wetland. Water Sci. Technol. 68, 1657. doi:10.2166/wst.2013.410

Boyd, C.E., 1970. Amino Acid, Protein, and Caloric Content of Vascular Aquatic Macrophytes. Ecology 51, 902–906. doi:10.2307/1933986

Carey, R.O., Hochmuth, G.J., Martinez, C.J., Boyer, T.H., Dukes, M.D., Toor, G.S., Cisar, J.L., 2013. Evaluating nutrient impacts in urban watersheds: Challenges and research opportunities. Environ. Pollut. 173, 138–149. doi:10.1016/j.envpol.2012.10.004

Collins, K.A., Lawrence, T.J., Stander, E.K., Jontos, R.J., Kaushal, S.S., Newcomer, T.A., Grimm, N.B., Cole Ekberg, M.L., 2010. Opportunities and challenges for managing nitrogen in urban stormwater: A review and synthesis. Ecol. Eng. 36, 1507–1519. doi:10.1016/j.ecoleng.2010.03.015

De Stefani, G., Tocchetto, D., Salvato, M., Borin, M., 2011. Performance of a floating treatment wetland for in-stream water amelioration in NE Italy. Hydrobiologia 674, 157–167. doi:10.1007/s10750-011-0730-4

Deering, E., 2016. Floating Treatment Wetlands in a Northern Climate: Examination of Phosphorus and Nitrogen Removal. UNIVERSITY OF MINNESOTA.

Fairfax County Department of Public Works and Environmental Services, 2017. Stormwater Management Pond Quick Reference Guide [WWW Document]. URL http://www.fairfaxcounty.gov/dpwes/environmental/swm_quickref.htm

Fletcher, T., 2004. Stormwater flow and quality and the effectiveness of non-proprietary stormwater treatment measures: a review and gap analysis. CRC for Catchment Hydrology, Clayton, Vic.

Fletcher, T.D., Andrieu, H., Hamel, P., 2013. Understanding, management and modelling of urban hydrology and its consequences for receiving waters: A state of the art. Adv. Water Resour. 51, 261–279. doi:10.1016/j.advwatres.2012.09.001

Floating Islands International, Inc., 2017. BioHaven® Technology [WWW Document]. BioHaven® Technol. « Float. Isl. Int. URL http://www.floatingislandinternational.com/products/biohaven-technology/ (accessed 9.23.16).

George Mason University, 2016. MS4 Annual Report. George Mason University, Fairfax, VA, USA.

George Mason University, 2015. MS4 Annual Report. George Mason University, Fairfax, VA, USA.

George Mason University, 2013. Stormwater Management Plan - Draft. George Mason University, Fairfax, VA, USA.

Headley, T., Tanner, C.C., 2006. Application of Floating Wetlands for Enhanced for Stormwater Treatment: A Review. National Institute of Water & Atmospheric Research Ltd., Hamilton, New Zealand.

Headley, T.R., Tanner, C.C., 2012. Constructed Wetlands With Floating Emergent Macrophytes: An Innovative Stormwater Treatment Technology. Crit. Rev. Environ. Sci. Technol. 42, 2261–2310. doi:10.1080/10643389.2011.574108

Henry, J.H., Henry, E.T., 2007. Plant root pruning container. US7210266 B2.

Hunt, W.F., 2006. Maintenance of Stormwater Wetlands and Wet Ponds. North Carolina Cooperative Extension Service.

iSee Systems, 2016. STELLA. iSee Systems, Lebanon, NH, USA.

Kadlec, R.H., Wallace, S.D., 2009. Treatment wetlands, 2nd ed. ed. CRC Press, Boca Raton, FL.

Keeley, M., Koburger, A., Dolowitz, D.P., Medearis, D., Nickel, D., Shuster, W., 2013. Perspectives on the Use of Green Infrastructure for Stormwater Management in Cleveland and Milwaukee. Environ. Manage. 51, 1093–1108. doi:10.1007/s00267-013-0032-x

Lenhart, H.A., Hunt, W.F., Burchell, M.R., 2012. Harvestable Nitrogen Accumulation for Five Storm Water Wetland Plant Species: Trigger for Storm Water Control Measure Maintenance? J. Environ. Eng. 138, 972–978. doi:10.1061/(ASCE)EE.1943-7870.0000550

Lorenzen, B., Brix, H., Mendelssohn, I.A., McKee, K.L., Miao, S.L., 2001. Growth, biomass allocation and nutrient use efficiency in Cladium jamaicense and Typha domingensis as affected by phosphorus and oxygen availability. Aquat. Bot. 70, 117–133. doi:10.1016/S0304-3770(01)00155-3

Lynch, J., Fox, L.J., Owen Jr., J.S., Sample, D.J., 2015. Evaluation of commercial floating treatment wetland technologies for nutrient remediation of stormwater. Ecol. Eng. 75, 61–69. doi:10.1016/j.ecoleng.2014.11.001

Mallin, M.A., Ensign, S.H., Wheeler, T.L., Mayes, D.B., 2002. Pollutant removal efficacy of three wet detention ponds. J. Environ. Qual. 31, 654–660.

Marimon, Z.A., Xuan, Z., Chang, N.-B., 2013. System dynamics modeling with sensitivity analysis for floating treatment wetlands in a stormwater wet pond. Ecol. Model. 267, 66–79. doi:10.1016/j.ecolmodel.2013.07.017

Marschner, H., 1995. Mineral Nutrition of Higher Plants. Academic Press.

McAndrew, B., Ahn, C., 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 8, 972. doi:10.3390/su8100972

Means, M.M., Ahn, C., Korol, A.R., Williams, L.D., 2016. Carbon storage potential by four macrophytes as affected by planting diversity in a created wetland. J. Environ. Manage. 165, 133–139. doi:10.1016/j.jenvman.2015.09.016

Meuleman, A.F., Beekman, J. (Hans) P., Verhoeven, J.T., 2002. Nutrient retention and nutrient-use efficiency in Phragmites australis stands after wasterwater application. Wetlands 22, 712–721.

National Oceanic and Atmospheric Administration, 2016. Climate Data Inventories; Vienna Station: Fairfax, VA, USA [WWW Document].

National Renewable Energy Laboratory, 2016. 30-Year Average of Monthly Solar Radiation, 1961-1990 [WWW Document]. URL http://rredc.nrel.gov/solar/old_data/nsrdb/1961-1990/redbook/sum2/state.html

NBC4 Washington, 2015. Students Say Pond Class at GMU Really Floats Their Boats [WWW Document]. URL http://www.nbcwashington.com/news/local/304771321.html

Northern Virginia Planning District Commission, 2000. Maintaining Your BMP: A Guidebook for Private Owners and Operators in Northern Virginia.

Osem, Y., Chen, Y., Levinson, D., Hadar, Y., 2007. The effects of plant roots on microbial community structure in aerated wastewater-treatment reactors. Ecol. Eng. 29, 133–142. doi:10.1016/j.ecoleng.2006.06.003

Pavlineri, N., Skoulikidis, N.T., Tsihrintzis, V.A., 2017. Constructed Floating Wetlands: A review of research, design, operation and management aspects, and data meta-analysis. Chem. Eng. J. 308, 1120–1132. doi:10.1016/j.cej.2016.09.140

Rollins, S., 2016. Neighborhood Stormwater Pond Maintenance Log and Resources. Clemson University.

Sample, D.J., Wang, C.-Y., Fox, L.J., 2013. Innovative best management fact sheet No. 1: Floating treatment wetlands. Virginia Cooperative Extension.

Tanner, C.C., 1996. Plants for constructed wetland treatment systems — A comparison of the growth and nutrient uptake of eight emergent species. Ecol. Eng. 7, 59–83. doi:10.1016/0925-8574(95)00066-6

Tanner, C.C., Headley, T.R., 2011. Components of floating emergent macrophyte treatment wetlands influencing removal of stormwater pollutants. Ecol. Eng. 37, 474–486. doi:10.1016/j.ecoleng.2010.12.012

USEPA, 2008. Incorporating Green Infrastructure Concepts into Total Maximum Daily Loads (TMDLs). Washington, D.C.
USEPA, U.S.E.P.A., 2015. Program Overview: Total Maximum Daily Loads (TMDL) [WWW Document]. URL https://www.epa.gov/tmdl/program-overview-total-maximum-daily-loads-tmdl (accessed 9.16.16).
USEPA, U.S.E.P.A., 1999a. Preliminary Data Summary of Urban Stormwater Best Management Practices (No. EPA-821-R-99-012). Office of Water, Washington, D.C.
USEPA, U.S.E.P.A., 1999b. Storm Water Technology Fact Sheet (No. EPA 832-F-99-048).
USGS, 2016. Runoff (Surface water runoff), USGS Water Science School [WWW Document]. Runoff Surf. Water Runoff. URL https://water.usgs.gov/edu/runoff.html
Verstraeten, G., Poesen, J., 2000. Estimating trap efficiency of small reservoirs and ponds: methods and implications for the assessment of sediment yield. Prog. Phys. Geogr. 24, 219–251. doi:10.1177/030913330002400204
Wang, C.-Y., Sample, D.J., 2014. Assessment of the nutrient removal effectiveness of floating treatment wetlands applied to urban retention ponds. J. Environ. Manage. 137, 23–35. doi:10.1016/j.jenvman.2014.02.008
Wang, C.-Y., Sample, D.J., 2013. Assessing floating treatment wetlands nutrient removal performance through a first order kinetics model and statistical inference. Ecol. Eng. 61, 292–302. doi:10.1016/j.ecoleng.2013.09.019
Wang, C.-Y., Sample, D.J., Bell, C., 2014. Vegetation effects on floating treatment wetland nutrient removal and harvesting strategies in urban stormwater ponds. Sci. Total Environ. 499, 384–393. doi:10.1016/j.scitotenv.2014.08.063
Wang, C.-Y., Sample, D.J., Day, S.D., Grizzard, T.J., 2015. Floating treatment wetland nutrient removal through vegetation harvest and observations from a field study. Ecol. Eng. 78, 15–26. doi:10.1016/j.ecoleng.2014.05.018
Wang, N., Mitsch, W.J., 2000. A detailed ecosystem model of phosphorus dynamics in created riparian wetlands. Ecol. Model. 126, 101–130.
Wanielista, M.P., Chang, N.-B., Chopra, M., Xuan, Z., Islam, K., Marimon, Z., 2012. Floating wetland systems for nutrient removal in stormwater ponds. Florida Department of Environmental Protection, Tallahassee, Florida.
Wetzel, R.G., 2001. Limnology: lake and river ecosystems. Academic Press, San Diego.
White, S.A., Cousins, M.M., 2013. Floating treatment wetland aided remediation of nitrogen and phosphorus from simulated stormwater runoff. Ecol. Eng. 61, 207–215. doi:10.1016/j.ecoleng.2013.09.020
Williamson, L.C., 2001. Phosphate Availability Regulates Root System Architecture in Arabidopsis. PLANT Physiol. 126, 875–882. doi:10.1104/pp.126.2.875
Winston, R.J., Hunt, W.F., Kennedy, S.G., Merriman, L.S., Chandler, J., Brown, D., 2013. Evaluation of floating treatment wetlands as retrofits to existing stormwater retention ponds. Ecol. Eng. 54, 254–265. doi:10.1016/j.ecoleng.2013.01.023

Yang, G., Bowling, L.C., Cherkauer, K.A., Pijanowski, B.C., 2011. The impact of urban development on hydrologic regime from catchment to basin scales. Landsc. Urban Plan. 103, 237–247. doi:10.1016/j.landurbplan.2011.08.003

Zhou, X., Wang, G., 2010. Nutrient concentration variations during Oenanthe javanica growth and decay in the ecological floating bed system. J. Environ. Sci. 22, 1710–1717. doi:10.1016/S1001-0742(09)60310-7

BIOGRAPHY

Brendan McAndrew graduated from Walworth Barbour American International School in Kfar Shmaryahu, Israel, in 2006. He received his Bachelor of Science in Biology from Christopher Newport University in 2011. He was employed as a teaching and research assistant at George Mason University while pursuing a Master of Science in Environmental Science and Policy which he earned in 2017.