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Research article

# Developing an ecosystem model of a floating wetland for water quality improvement on a stormwater pond

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ABSTRACT

An ecosystem model was developed to assist with designing and implementing a floating wetland (FW) for water quality management of urban stormwater ponds, focusing on nitrogen (N) removal. The model is comprised of three linked submodels: hydrology, plant growth, and nitrogen. The model was calibrated with the data that resulted from a FW constructed and implemented as part of an interdisciplinary pedagogical project on a university campus, titled "*The Rain Project*", which raised awareness of stormwater issues while investigating the potential application of green infrastructure for sustainable stormwater management. The FW had been deployed during the summer of 2015 (i.e., May through mid-September) on a major stormwater pond located at the center of the Fairfax Campus of George Mason University near Washington, D.C. We used the model to explore the impact of three design elements of FW (i.e., hydraulic residence time (HRT), surface area coverage, and primary productivity) on the function of FW. Model simulations showed enhanced N removal performance as HRT and surface area coverage increased. The relatively low macrophyte productivity observed indicates that, in the case of our pond and FW, N removal was very limited. The model results suggest that even full pond surface coverage would result in meager N removal (~6%) at a HRT of one week. A FW with higher plant productivity, more representative of that reported in the literature, would require only 10% coverage to achieve similar N removal efficiency (~7%). Therefore, macrophyte productivity appears to have a greater impact on FW performance on N removal than surface area coverage or pond HRT. The outcome of the study shows that this model, though limited in scope, may be useful in aiding the design of FW to augment the performance of degraded stormwater ponds in an effort to meet local water quality goals.



## 1. Introduction

Runoff from urban areas is increasingly responsible for the transport of nitrogen (N) into natural waterways (Carey et al., 2013; Fletcher, 2004; Fletcher et al., 2013; USGS, 2016; Yang et al., 2011). Over the past two decades, N has become recognized as a pervasive stormwater pollutant and is targeted by the U.S. Environmental Protection Agency's (USEPA) 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 (SCMs), also known as best management practices, 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 our 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 DPWES, 2017). The primary function of 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 (HRT) in the pond (Borne et al., 2013). As sediments build up over time, however, pond function is impaired as retention capacity is reduced and HRT decreases (Verstraeten and Poesen, 2000). Dredging is therefore required every 5–20 years, depending on pond size and

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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, N 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 N 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 artificial islands stocked with wetland plants called a floating wetland (FW) (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 (FW) 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 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 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 of water in the pond (Winston et al., 2013).

The literature suggests there is potential for the use of FW 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).

As part of an interdisciplinary student project called "*The Rain Project*," a small-scale FW ecosystem was designed and deployed on a stormwater pond located on our university campus (Ahn, 2016). In a companion study we investigated the N and sediment capture performance of this FW ecosystem by harvesting and analyzing biomass and accumulated sediments for N content (McAndrew et al., 2016). Although we observed a N uptake rate comparable to the literature, budget constraints precluded the rigorous analysis of water chemistry necessary to quantify the proportion of N removed from the pond by the FW. System modeling, however, presents a means of estimating the N removal performance of our FW.

We present a system model of a FW deployed on an urban stormwater retention pond. The goal of this study was to develop a model to estimate the N removal by plants stocked on our FW and to investigate the impact of FW design and hydrologic regime on N removal performance. This model may provide stakeholders such as private landowners, developers, and municipalities with a tool they may use to guide the design of FW and evaluate potential N removal performance of a given FW-pond ecosystem. The objectives of the model were to (1) estimate the N removal efficiency of the case-study FW and (2) investigate the impact of HRT, surface area coverage, and plant productivity on FW functionality.

## 2. Materials and methods

### 2.1. Mason Pond FW case study

#### 2.1.1. Site description

Mason Pond is a 7100 $m^2$ urban stormwater retention pond located on the Fairfax Campus of George Mason University in the suburbs of Washington D.C. (38°49′44″ N, 77°18′37″ W). The pond is primarily fed by runoff from the heavily urbanized campus surrounding the pond which is dominated by large buildings, expansive parking lots, and heavily manicured landscaping (George Mason University, 2015). Since the pond's construction in 1991, local development has increased the drainage area beyond its design capacity by approximately 10% (George Mason University, 2013). Rigorous analysis of total N (TN) in the pond was beyond the scope of funding allocated for the Mason Pond FW case study, however a campus-wide stormwater assessment during the study period indicated TN concentration ranged from 1.07 to 1.25 mg/L (George Mason University, 2016). A FW experiment conducted in an adjacent watershed reported mean TN concentration of 1.19 ± 0.27 mg/L which suggests that stormwater in this area generally falls below 2 mg/L (Wang et al., 2014). A summary of Mason Pond's physical dimensions and basic water quality parameters during the FW deployment period is presented in Table 1; further details can be found in McAndrew et al. (2016).

#### 2.1.2. Floating wetland construction and deployment

The Mason Pond FW was constructed using a commercial system marketed as Beemat which uses a buoyant foam mat as the foundation of the artificial island (BeeMats LLC, New Smyrna Beach, FL, USA). The foam mat is sold pre-cut with uniformly spaced circular holes that support about 30 plants per square meter. The plants are placed in perforated plastic cups that are inserted into the mat holes such that the roots are submerged below the water while the shoots remain above the water line (see McAndrew et al., 2016).

The FW was designed as two kidney-shaped islands to symbolize the function of the FW as a biological "filter" (Ahn, 2016). At deployment, the combined surface area of both islands was approximately 50 $m^2$ which covered an estimated 0.704% of the pond surface. A mixture of five species of plants were stocked on the FW for a total of 1510 plants. The plant species chosen for the FW included: *Alisma subcordatum* (American water plantain), *Carex stricta* (upright sedge), *Iris versicolor* (blue flag iris), *Juncus effusus* (common rush), and *Pontederia cordata* (pickerelweed). These

**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 |

species were primarily selected because they are native wetland species commonly found in treatment wetlands, however aesthetics were also factored into plant selection (Ahn, 2016; Means et al., 2016; Tanner, 1996).

The FW was deployed on Mason Pond on 5/12/2015. Each kidney was anchored to the bottom of the pond such that the FW was free to move vertically with any changes in pond water level, but would not travel significantly as a result of wind action or water current. Approximately 20 weeks later, the FW was removed on 9/26/2015. Further details regarding the construction and deployment of the FW can be found in McAndrew et al. (2016) and Ahn (2016).

#### 2.1.3. Plant biomass measurement and tissue analysis

Plants from each species were sampled before and after deployment to determine biomass and N content. Roots and shoots were analyzed separately to determine whether differential growth and N assimilation occurred. The results of these analyses are summarized in Table 2 below. These measurements allowed us to estimate the mass of N captured by each species during the deployment period of the FW and thereby identify which species were the most effective. The results suggest that *Carex*, *Iris*, and *Pontederia* exhibited the greatest productivity and, therefore, assimilated the most N. McAndrew et al. (2016), our companion paper, presents a full analysis of the N removal performance of the Mason Pond FW.

### 2.2. Floating wetland ecosystem model

To further investigate the N capture performance of the Mason Pond FW, a system model was developed to estimate the N removal efficiency of the Mason Pond FW and to test performance under a variety of conditions beyond the scope of the Mason Pond case study. The modeling process and underlying rationale is described in the following sections.

#### 2.2.1. 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 developed for this paper 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 N 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 N which is simulated by the N submodel. Lastly, the plant growth submodel simulates the production of biomass on the FW which influences pond N content in the N submodel. Many of the parameters and coefficients contained in the model were calculated from the results of our 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). This study was selected because it reported both biomass production and N removal of a mesocosm-based BeeMat FW which allowed us to verify the structure of each submodel by setting initial conditions to those described in detail in Lynch et al. (2015). A more detailed explanation of the submodels and components is presented in Section 2.2.2.

A review of the limited research available on FW suggest that the key factors affecting N capture are biomass production and HRT (Marimon et al., 2013; McAndrew et al., 2016; Pavlineri et al., 2017). HRT 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 and the proportion of pond surface area covered by the FW. We therefore elected to use the model to test the impact of surface area coverage, HRT, and productivity on N 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. N concentrations in the pond do not limit or drive biomass production;
5. Microbial denitrification by FW biofilms is not included in this model due to a lack of data relating FW plant biomass production and root biofilm development and activity;
6. Knock-on effects of changes in FW surface area coverage (i.e., effect on pond water temperatures or dissolved oxygen levels) were not investigated in this model.

**Table 2**
Summary of shoot and root biomass production and nitrogen content of species planted on the Mason Pond FWs.

| Plant Species | Biomass Produced (g/m$^2$) | Nitrogen Content (%) | |
|---|---|---|---|
| | | Initial | Final |
| *Carex stricta* | | | |
| Shoot | 36.04 | 1.28% | 1.73% |
| Root | 63.34 | 0.53% | 1.86% |
| *Iris versicolor* | | | |
| Shoot | 25.22 | 2.31% | 1.96% |
| Root | 50.45 | 1.34% | 1.44% |
| *Pontederia cordata* | | | |
| Shoot | 43.96 | 2.87% | 2.02% |
| Root | 16.94 | 1.04% | 2.07% |
| *Juncus effusus* | | | |
| Shoot | 26.42 | 0.85% | 1.37% |
| Root[a] | – | – | – |
| *Alisma subcordatum* | | | |
| Shoot | 11.66 | 2.37% | 2.58% |
| Root | 1.05 | 1.77% | 2.32% |

[a] *Juncus* roots showed an apparent loss in biomass during the period of deployment.

#### 2.2.2. Model structure

A conceptual model of the FW-N system is shown in Fig. 1. The differential equations, state variables, forcing functions, and associated variables/parameters that make up the model are summarized in Tables 3 and 4 followed by a detailed description of the model. The model structure and mathematical relationships are

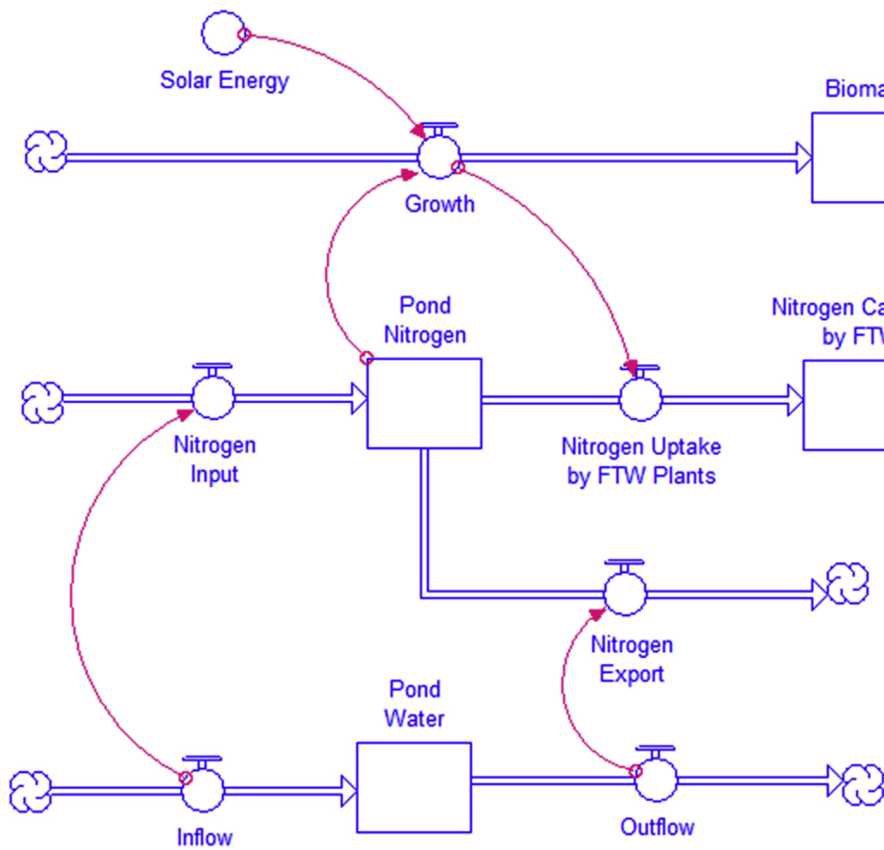


**Fig. 1.** Conceptual STELLA diagram of the FWs-pond ecosystem model.

**Table 3**
Differential equations used in the FWs-pond 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 FWs (g) |
| Root Biomass | Root biomass on the FWs (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) |
| Root Ratio | NORMAL(mean root:shoot ratio, std. dev.) |
| 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) |
| d(Mass N out)/dt = | N out (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 4**
State variables, forcing functions, and variables/parameters used in the FW-pond ecosystem model.

| Name | Description | Value/Units | Source |
|---|---|---|---|
| *State Variables* | | | |
| V | Volume of water in mesocosm or 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 Mason Pond | g | Calculation, Mason Pond |
| Mass N Out | Mass of nitrogen exported from Mason Pond | g | Estimated |
| Plant Captured N | Mass of nitrogen captured by macrophyte biomass production | g | Calculation, Mason Pond |
| *Forcing Functions* | | | |
| Inflow | Water inflow | $m^3$/week | |
| N in | Nitrogen load | g/week | |
| Solar | Average amount of solar energy reaching FW | kcal/$m^2$/week | |
| *Parameters and coefficients* | | | |
| Fill Volume | Volume of water input to pond | 6532 $m^3$ | Calculation, Mason Pond |
| HRT | Hydraulic residence time | Days | 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 (2002) |
| Root Ratio | Ratio of root production to shoot production (R/S) | $1.38 \pm 0.87$ g/g | Calculation, Mason Pond FW |
| N in mean | Mean total nitrogen (TN) concentration of inflow | 1.19 g/$m^3$ | GMU (2016), Wang and Sample (2014) |
| N in SD | Standard deviation of total nitrogen (TN) concentration inflow | 0.27 g/$m^3$ | Wang and Sample (2014) |
| Water N Conc | Concentration of total nitrogen (TN) in Mason Pond | g/$m^3$ | Calculation |
| Shoot N Conc | Nitrogen content of shoot biomass | $1.90 \pm 0.15$% | CHN Analysis of FW Plants |
| Root N Conc | Nitrogen content of root biomass | $1.79 \pm 0.32$% | CHN Analysis of FW Plants |
| Total N Input | Total mass of nitrogen input during FW deployment period | g | Calculation |
| % Plant Capture | Percent of Total N Input captured by plant biomass production | % | Calculation |

adopted and/or modified from the similar models previously developed on STELLA (Spieles and Mitsch, 1999; Wang and Mitsch, 2000; Ahn and Mitsch, 2002). The entire model is built as a simple first order model, a similar approach used as in Wang and Sample (2013) that also assessed nutrient removal performance of floating treatment wetlands through a first order kinetics model.

### 2.2.3. Hydrology submodel

The hydrology submodel (Fig. 2a) 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 we estimated the average HRT of Mason Pond during the study period at approximately 8.5 days, however HRT varies with each storm event. 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). For our model we tested HRTs of 7, 14, and 21 days to test the N 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.

### 2.2.4. Plant growth submodel

The plant growth submodel (Fig. 2b) 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.

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 4; 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, we attempted 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 our 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 3). 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).

### 2.2.5. Nitrogen submodel

The N submodel (Fig. 2c) consists of three state variables: mass N in the pond water (*Water N*), the mass N exported from the pond (*Mass N Out*), and the mass N captured by plant growth (*Plant Captured N*). Water N is calculated as the volume of influent entering the pond (*Inflow)* multiplied by the concentration of N in the influent (*N in Conc*). The influent N concentration is generated with a built-in NORMAL function fed with the mean and standard deviation of the N concentrations reported in the university's latest MS4 (George Mason University, 2016) report and those observed in

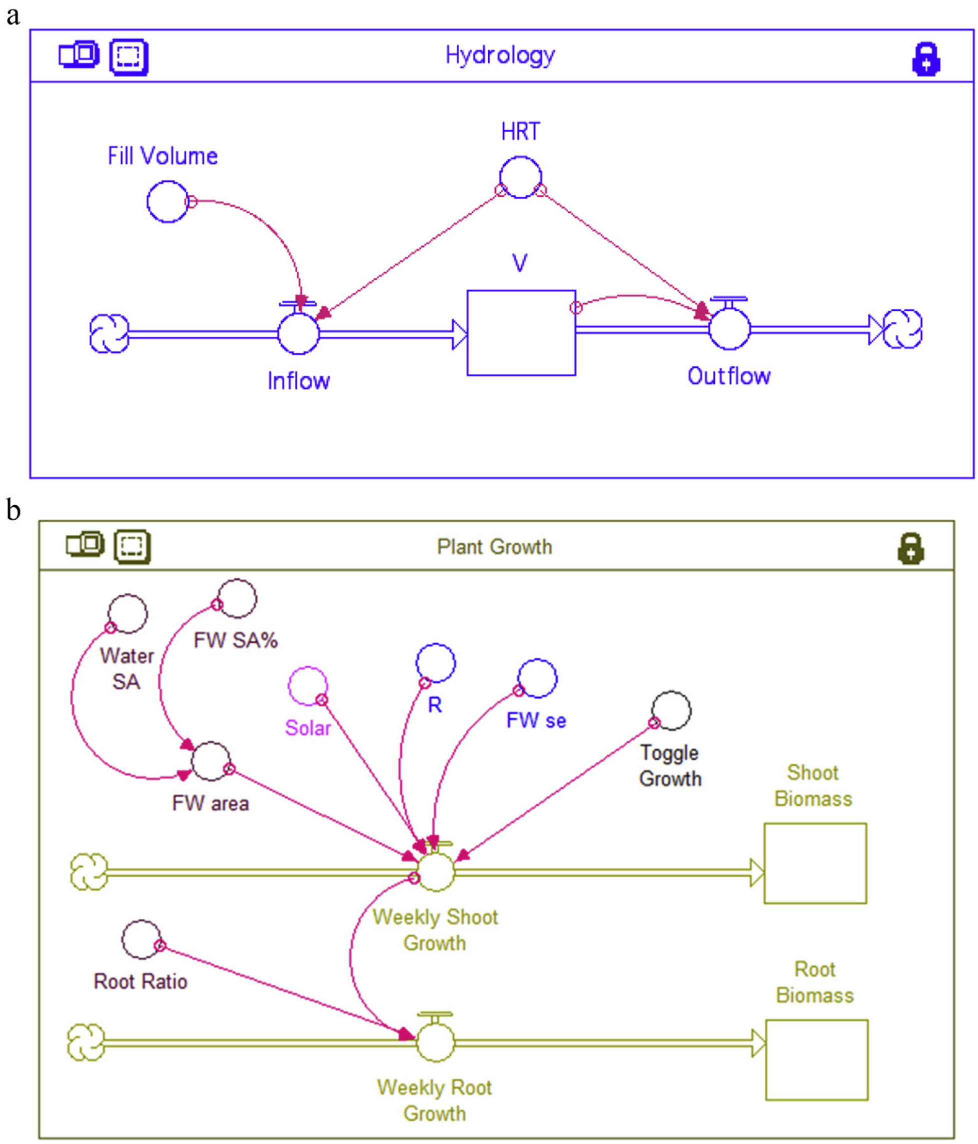


**Fig. 2.** STELLA™ diagrams of a) hydrology submodel, b) plant growth submodel, STELLA and c) nitrogen submodel.

a stormwater pond located approximately 1 mile from Mason Pond (Table 4; Wang and Sample, 2014).

*Plant Captured N* is defined as the mass of N assimilated by biomass in each time-step (*Weekly N Uptake*). N uptake is calculated as the product of biomass production (*Daily Shoot Growth*, *Daily Root Growth*) and the root and shoot N concentration (*Shoot N Conc*, *Root N Conc*). The values for concentration of N in the plant shoots and roots are generated by the built-in NORMAL function fed by the mean and standard deviation for N content we observed in plants harvested from the Mason Pond FW (Tables 3 and 4). The mass of N exported from the pond (*Mass N Out*) is the amount of N remaining at the end of the hydraulic residence interval (*N out*) calculated as a product of effluent volume (*Outflow*) and pond N concentration (*Water N Conc*).

### *2.3. Model calibration and verification*

Model calibration involves altering selected 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 our 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 our FW: *Carex stricta*, *Iris versicolor*, and *Pontederia subcordatum* (Table 2). 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;

c

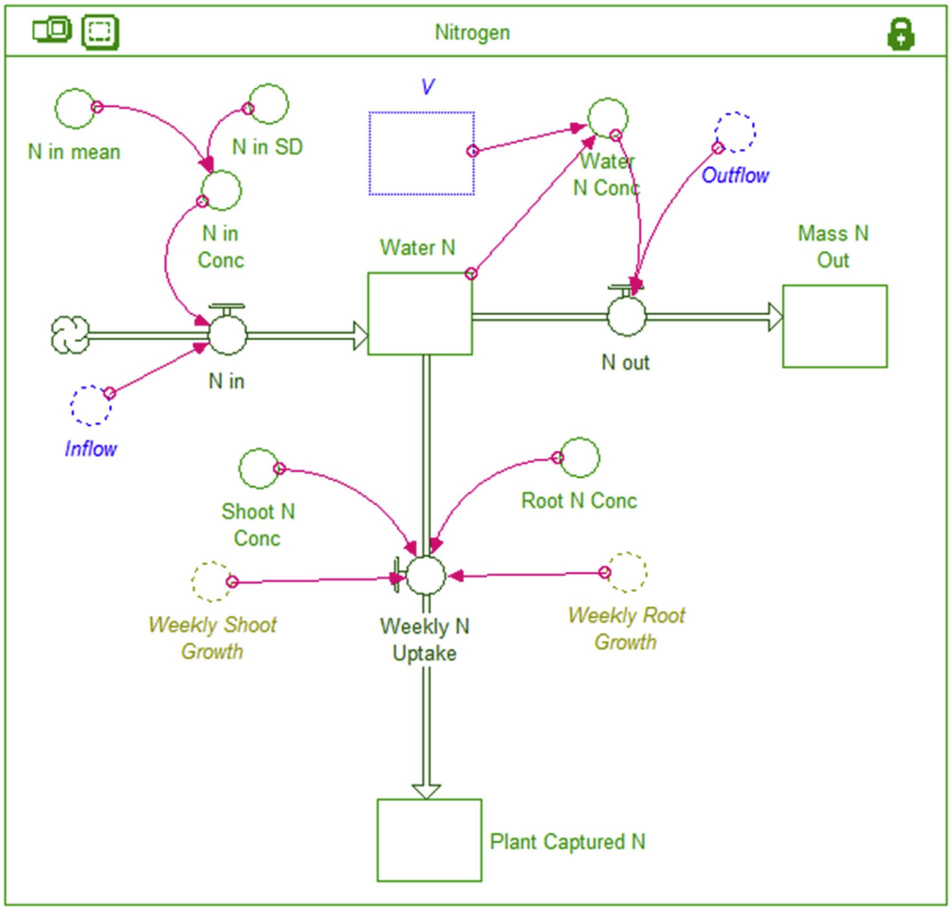


**Fig. 2.** (*continued*).

(2) the constructed wetland models assumed an existing standing stock whereas our FW was stocked with small 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.

Another FW study conducted on a stormwater pond nearby also observed similarly low productivity by softstem bulrush (max. biomass: 41.42 g/m$^2$) which suggests that this estimate of solar efficiency is not unrealistically low (Wang et al., 2015). On the other hand, the same study and several others also report biomass production two to three times higher than we observed on the Mason Pond FW (Wanielista et al., 2012; White and Cousins, 2013). We therefore elected to also simulate a more productive FW, in addition to the simulation based on the Mason Pond FW results, 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 $0.67 \pm 0.06$ g/m$^2$/day which agrees well with the biomass production rate of observed on the FW (0.73 g/m$^2$/day). Similarly, the simulated N uptake rate of 0.010 g/m$^2$/day falls within 10% of the observed N uptake rate of 0.011 g/m$^2$/day.

To test whether our model accurately simulates biomass production on the Mason Pond FW, model verification was performed with data from Lynch et al. (2015) and Wang et al. (2015). The former study was also used to verify the performance of the nitrogen submodel. These studies were selected for model verification as they report more biomass results than other FW studies and include both a mesocosm-based study and a pond-based experiment. To verify the model structure, model variables (e.g., *Fill Volume*, *N in mean*, *Root Ratio*, *Root/Shoot N Conc*, etc.) were set to values reported in Lynch et al. (2015) or Wang et al. (2015) before solar efficiency (*FW se*) was calibrated until biomass production simulated the model reflected the reported results. To simulate the poor growth of softstem bulrush described by Wang et al. (2015), solar efficiency was calibrated to just 0.15% which accurately simulated biomass production to within 2% of reported results. While root:shoot ratio differed between the studies, a solar efficiency of 1.25% was found to accurately simulate biomass production of both *Pontederia cordata* as reported by Wang et al. (2015) and *Juncus effusus* on Beemat as reported by Lynch et al. (2015). The model also accurately simulated N removal to within 10% of the values reported by Lynch et al. (2015).

### 2.4. 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 our model we performed sensitivity analysis on HRT, surface area coverage, and solar efficiency to determine their relationship to FW TN removal efficiency.

Sensitivity analysis was performed on the HRT of the pond to determine whether longer residence times resulted in greater overall N 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 N 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 N 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, we elected to test extreme levels of coverage (>50%) to allow for more direct comparison of our results with the literature. Lastly, we tested N removal efficiency 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.

## 3. Results and discussion

### 3.1. Hydrologic residence time

Model simulations show N removal by the FW increased with HRT (Fig. 3). The low-productivity simulations, calibrated to biomass production reported in McAndrew et al. (2016), estimated mean N 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 N introduced to the system is governed by the HRT while plant uptake of N 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 N removal performance is also apparent in the results from the high-productivity simulations—until FW coverage surpasses 50% at longer HRTs (Fig. 3b). At high surface coverage (≥50%) and HRT greater than one week, N uptake by the plants begins to deplete pond N. An FW-pond ecosystem model developed by Marimon et al. (2013) also showed N 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 FW 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.

### 3.2. Surface area coverage

FW surface area coverage is a key factor influencing N removal as it limits the area available for plant growth. To test the effect of surface area on N removal we performed sensitivity analysis across a range of coverage levels from 0.70% to 100%. As with HRT, N removal efficiency increased with surface area coverage (Fig. 3a). This is because an increase in FW surface area results in a corresponding increase in total plants deployed on the FW and, by extension, N 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. 3b). 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

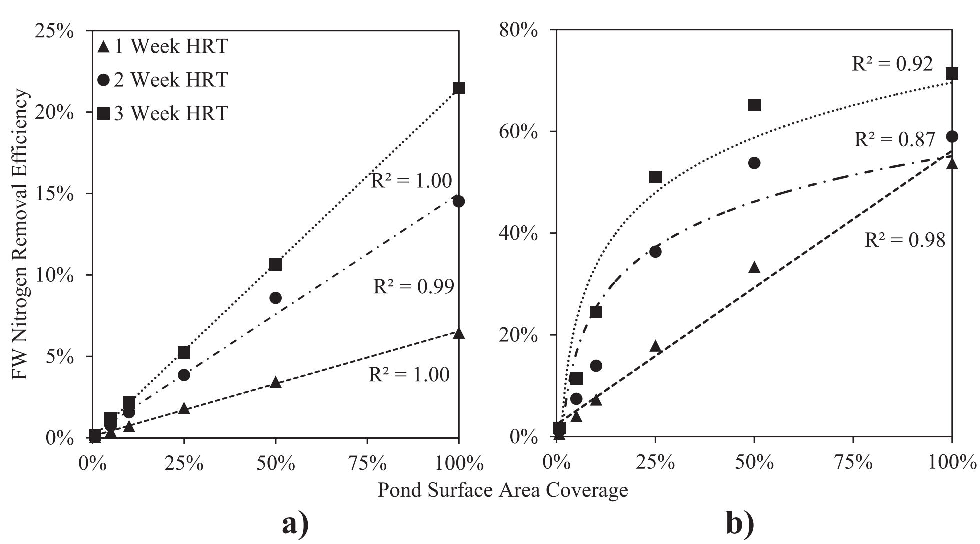


**Fig. 3.** Model results for mean nitrogen removal efficiency ($n = 5$) of Mason Pond FWs at a solar efficiency ratio of **(a)** 0.25% and **(b)** 2.5% across the range of hydraulic residence times (1, 2, and 3 weeks) and FWs surface area coverages (0.70%–100%). The model output shows a positive relationship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

of 45% at full surface coverage (Pavlineri et al., 2017). Our model, however, suggests 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. 3b) 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).

### 3.3. FW solar efficiency

Solar efficiency had the greatest impact on N 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 and 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. 3a). 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. 3b). 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, our 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 a 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 FW.

### 3.4. Application of the model for sustainable stormwater management

The results of our model suggest that while FW can significantly augment the N 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. 3). 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 we performed sensitivity analysis on a wide range of FW surface area coverage up to and including full coverage, it is likely impractical to deploy a 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 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.

### 3.5. Limitations of the model

This model relies heavily on data produced by our own FW case study due to a lack of available research on the performance of FW deployed on real-world stormwater systems as well as a limited amount of data on biomass production by FW based plants. The model is therefore unable to incorporate potential knock-on effects of changes in FW design criteria (i.e., the effect of high FW surface area coverage on dissolved oxygen levels). Furthermore, our FW case study only examined a small FW in a single pond which suggests the model may not accurately simulate FW behavior at higher surface area coverage or at different pond design criteria.

## 4. Conclusions

Our model results suggest that FW 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 N through direct plant uptake alone. Consistent N 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, FW may be able to improve the N 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 ecotechnology in real-world stormwater ponds which limited our ability to incorporate other N removal pathways such as denitrification by root-based biofilms. 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.

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