Floating wetlands for urban stormwater treatment

Chih-Yu Wang

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David J. Sample, Chair
Susan D. Day
Daniel L. Gallagher
Thomas J. Grizzard
Theresa M. Thompson

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ABSTRACT

A floating treatment wetland (FTW) is an ecological approach which seeks to reduce point and nonpoint source pollution by installing substrate rooted plants grown on floating mats in open waters. While relatively novel, FTW use is increasing. A review of literature identified several research gaps, including: (1) assessments of the treatment performance of FTWs; (2) evaluations of FTWs in the U.S., particularly within wet ponds that receive urban runoff; and (3) plant temporal nutrient distribution, plant growth rate, and the long-term persistence of the FTWs in temperate regions with periodic ice encasement.

An assessment model, i-FTW model, was developed, and its parameter s fitted based on data from 14 published FTW studies in the first research topic. The estimated median FTW apparent uptake velocity with 95% confidence interval were 0.048 (0.018 – 0.059) and 0.027 (0.016 – 0.040) m/day for total phosphorus (TP) and total nitrogen (TN), respectively. The i-FTW model provided a more accurate prediction in nutrient removal than two common performance metrics: removal rate (mg/m^2/day) and removal efficiency (%). In the second research topic, the results of a mesocosm experiment indicated that FTWs with 61% coverage, planted with pickerelweed (*Pontederia cordata* L.) or softstem bulrush (*Schoenoplectus tabernaemontani*), significantly improved TP and TN removal efficiency of the control treatment by 8.2% and 18.2%, respectively. The pickerelweed exhibited significantly higher phosphorus and nitrogen removal than the softstem bulrush when water temperatures were greater than 25℃. Field observations in the third research topic found that pickerelweed demonstrated higher phosphorus removal performance (7.58 mg/plant) than softstem bulrush (1.62 mg/plant). Based on the observed seasonal changes in phosphorus distribution, harvest of above-ground vegetation is recommended to be conducted twice a year in June and September. Planted perennial macrophytes successfully adapted to stresses of the low dissolved oxygen (DO) concentrations (minimum: 1.2 mg/L), ice encasement, and relatively low nutrient concentrations in the water (median: 0.15 mg/L TP and 1.15 mg/L TN). Systematic observation of wildlife activities indicated eight classes of organisms inhabiting, foraging, breeding, nursing, or resting in the FTWs. Recommendations for FTW design and suggestions for further research are made based upon these findings.

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Chapter 1. Introduction

A floating treatment wetland (FTW) is a developing pollution control practice and have received much attention recently (Borne *et al.*, 2013; Headley and Tanner, 2012). This treatment method provides the potential to reduce point and nonpoint source (NPS) pollution that contribute to eutrophication and hypoxia of water bodies (Chen *et al.*, 2009; Hubbard *et al.*, 2004; Tanner and Headley, 2011; Zak *et al.*, 2011). FTWs integrate biological, ecological, and engineering principles to provide a comprehensive treatment (Todd *et al.*, 2003). Understanding FTW behavior requires a multi-disciplinary approach. FTW techniques are a form of phytoremediation, which is a technology that reduces or immobilizes environmental toxins by using plants (Ladislas *et al.*, 2013; Pilon-Smits, 2005). In a holistic sense, FTWs can be considered as a form of ecological engineering, potentially benefiting nature and human society by reducing negative anthropogenic impacts on the environment, such as stormwater pollution generated from urban areas (Cunningham and Ow, 1996; Hawken *et al.*, 1999). To address water degradation resulting from urban stormwater and the limitation of existing urban stormwater best management practices (BMPs), FTWs are an innovative BMP that exhibits potential for widespread adoption (Chesapeake Stormwater Network, 2012). BMP refers to technologies or combinations of practices, designed to reduce pollutant concentrations and improve urban stormwater quality, such as constructed wetlands and wet ponds (Hsieh and Davis, 2005).

1.1 Urban stormwater pollution

Urban stormwater has been identified as a major non-point source (NPS) pollution contributor (Ladislas *et al.*, 2013; Sun *et al.*, 2010; Winston *et al.*, 2013). Runoff from urban areas and highways is responsible for about 30% of impaired length of streams, and 45% of impaired lake areas in the United States (Novotny, 2003). Urbanization and deforestation have significantly increased impervious areas, which diffuse pollutants including pavement wear, fuel combustion, deicing salts, fertilizer, soil, leaves, and debris. (Burton and Pitt, 2002; Driver and Tasker, 1990; Waschbusch, 1999). Because of large runoff volumes generated from these impervious surfaces, urban stormwater conveys large quantities of nutrients and may facilitate eutrophication or hypoxia of receiving waters (Dougherty *et al.*, 2006). While the mass transported is significant, pollutants concentrations are relatively low due to the large runoff volumes. Table 1-1 shows event mean concentrations (EMC) of total phosphorus (TP) and total nitrogen (TN) in the typical urban stormwater in U.S. nationwide database and three retention ponds close to our study site, Ashby Pond, located at the City of Fairfax, Virginia. The EMC ranges from 0.31 to 0.41 mg/L for TP and 2.39 to 2.61 mg/L for TN. In comparison, TP and TN in domestic wastewater after secondary treatment are typically 2 mg/L and 30 mg/L, respectively (US EPA, 1999).

Table 1-1. Event mean concentrations of typical urban stormwater in the U.S. and wet ponds closed to the study site, Ashby Pond, VA.

Data source / Site	**TP (mg/L)**	**TN (mg/L)**
Nationwide NURP[a]	0.32	2.51[b]
Nationwide NURP, USGS, and NPDES[c]	0.34	2.39[b]
Lakeridge, VA[d]	0.31	2.42
Stedwick, MD[d]	0.37	2.61
Westleigh, MD[d]	0.41	2.55

[a] Smullen *et al.* (1999).
[b] Sum of TKN and oxidized N.
[c] Load (lb/year) / runoff volume (acre-ft/year).
[d] NURP, Washing D.C. report (MWCOG *et al.*, 1983).

1.2 Study area

Ashby Pond is an existing wet pond located in the City of Fairfax, Virginia in a residential area of the Accotink Creek Watershed (Figure 1-1). This watershed has a surface area of 5,689 m^2 and provides non-contact recreational use as a park for property owners in the vicinity. The catchment has an average annual precipitation of 1157 mm and monthly precipitation ranged from 65 to 119 mm between 2003 and 2012 (Figure 1-2). The monthly average temperatures span from 0.7 to 24 ℃ in January and July, respectively (NOAA, 2012).

The watershed draining to Ashby Pond is 0.57 km^2 and is divided into three small basins: west basin, east basin, and the park area, which comprises most of the direct drainage to the water body. The areas of nine land use types and their corresponding coverage of each sub-watershed were measured from aerial photos and Arc GIS 10 software (Figure 1-1 and Table 1-2). The existing land is primarily low density residential for the east basin and high density residential/commercial mixed for the west basin. The park area watershed is mainly covered by forest (36 %) and a small residential area. The impervious area is 38 % for the total watershed, 47% for the west basin, 27% for the east basin, and 12% for the park area. Virginia State Route 236, which passes through the catchment, had an annual average daily traffic of 38,000 in 2010 and is classified as a high traffic street (10,600 vehicles per day) (Steuer *et al.*, 1997; VDOT, 2011). Other roads within the catchment are defined as relatively low traffic streets.

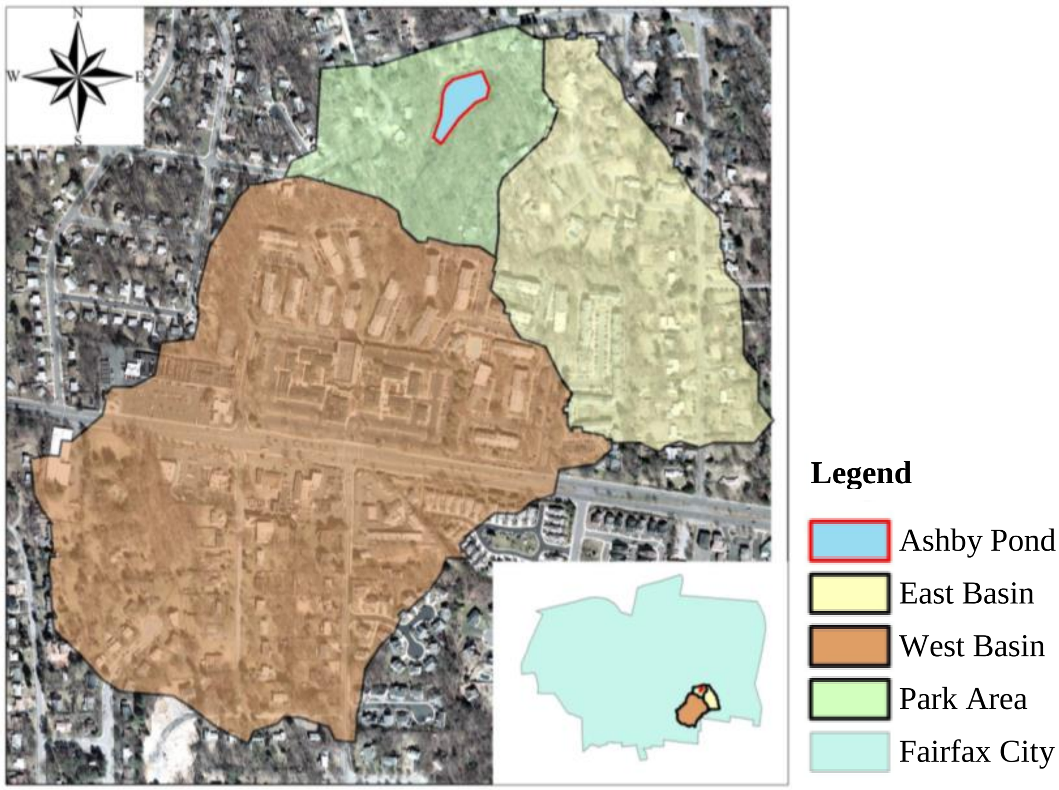

Figure 1-1. Ashby Pond watershed relative location in the City of Fairfax, VA.

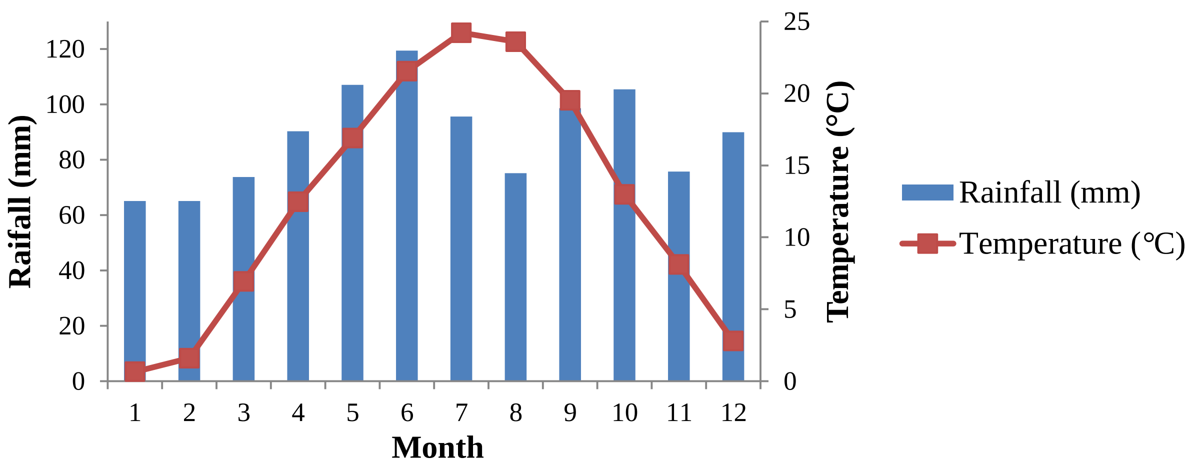

Figure 1-2. Ashby Pond average monthly precipitation depth and temperature from 2003 to 2012.

Table 1-2. Land use characteristics and areas (km^2) of the west basin, east basin, and park area of the Ashby Pond watershed.

Sub-watershed / Land use	West basin	East basin	Park area	Total watershed
Pitched roofs	0.03 (9%)[a]	0.02 (12%)	0.00 (5%)	0.05 (12%)
Flat roofs	0.02 (4%)	–	–	0.02 (4%)
Parking lots	0.07 (20%)	0.01 (4%)	0.00 (0%)	0.08 (19%)
Driveways	0.02 (5%)	0.01 (6%)	0.00 (3%)	0.03 (7%)
Low traffic streets	0.01 (4%)	0.01 (5%)	0.00 (5%)	0.02 (6%)
High traffic streets	0.01 (4%)	–	–	0.01 (3%)
Lawns	0.10 (26%)	0.05 (37%)	0.01 (14%)	0.15 (36%)
Parks/forests	0.10 (27%)	0.05 (36%)	0.05 (66%)	0.19 (11%)
Open water	–	–	0.01 (8%)	0.01 (1%)
Total	0.36 (100%)	0.13 (100%)	0.07 (100%)	0.57 (100%)

[a] Values in parentheses are the percent of each land use type in the corresponding sub-watershed.

1.3 Research problems and goals

Although FTWs may offer great promise to control nonpoint source pollution, several uncertainties still need to be evaluated to insure the success of this technology in urban environments. One example is the stress of low nutrient concentrations on vegetation performance and sustainability of the vegetation and thus the persistence of the FTW itself. Visser and Sasser (2009) found negative effects on natural floating wetlands productivity due to decreased nutrient replenishment, which may have caused insufficient nutrient supply to vegetation. Currently, few studies have investigated the potential and feasibility of FTWs in urban runoff to control anthropogenic nutrient exports. As discussed later in Section 2.4, the initial concentrations of the 15 published FTW studies ranged from 4.55-197 mg/L and 0.1-15 mg/L of TN and TP, respectively. Only three of the experiments had TP concentrations similar to the reported urban stormwater EMC data in Table 1-1 (blue region in Figure 1-3). TN exhibits greater uncertainty because the FTW technology has not previously been evaluated under such nutrient limited conditions.

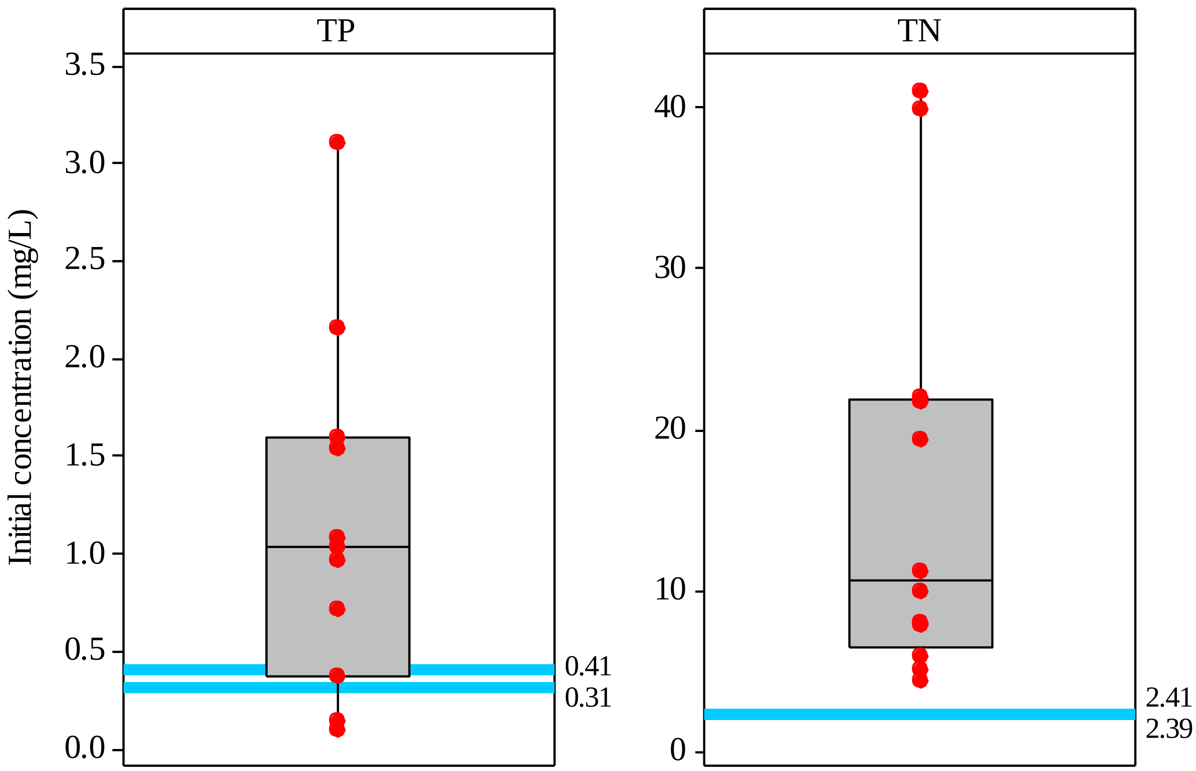

Figure 1-3. The typical urban stormwater TP and TN concentration ranges in Table 1-1 (blue lines) and published FTW studies (red dots) listed in Tables 2-3 and 2-4. Three FTW reports were excluded from the figure due to their high concentrations (TP>4 mg/L, TN>60 mg/L).

Understanding plant performance in the floating raft environment is another research need. Although macrophyte growth and decomposition in wetland environments has been widely investigated, performance of these plants in soilless systems has been studied to a lesser degree. Direct and permanent contact with water is expected to result in different biomass growth and decomposition conditions. It was reported that the plant productivity on floating rafts was 3.8 times higher compared to that of the terrestrial native community, the source of the vegetation on the FTWs (Nakamura *et al.*, 1998). This may be due to more direct and effective nutrient acquisition from water than soil environments, where soil solutions carrying nutrients are not always available. However, not all plants can adapt to a floating environment. Wen and Recknagel (2002) reported that water primrose (*Ludwigia peploides*) showed negligible growth while three other creeping plants grew well on floating rafts. Additionally, the sustainability of FTW vegetation under extended exposure to ice encasement, such as might occur in climates with cold winters, has not been investigated and documented. In contrast with soil-rooted macrophytes, FTW plants have their underground tissues submerged in water, which may become ice in winter in temperate regions, such as Virginia and other mid-Atlantic states. Ice in winter may permanently damage the plants and prevent them from reproducing in the following year (Andrews, 1996).

In terms of management strategies, the nutrients in plant tissues may be recycled and removed from aquatic systems through harvesting practices. Meuleman *et al.* (2002) suggested the nutrient removal efficiency could be increased from 9 to 20% of TN and from 6 to 25% of TP by harvesting aboveground tissues in September instead of winter when most nutrients were translocated to the rhizome/root system. This temporal variation of internal phosphorus distribution was related to their phenology (Ruiz and Velasco, 2010a).

The intent and focus of this research was to investigate research gaps regarding the performance of FTWs in urban stormwater ponds. Most of the published FTW studies have used heavily polluted source water, and were conducted outside the U.S. Evaluations of FTWs in the U.S., and in particular, within wet ponds that receive urban runoff, are rare. Additionally, plant temporal nutrient distribution, growth characteristics, and the long-term sustainability of FTWs remain a research need. Such information is essential for developing management strategies for FTW installations in urban wet pond environments, especially in temperate regions with periodic ice encasement. Last but not least, better assessments of the treatment performance of FTW are critical for design and application. In summary, the purposes of this research are to:

1. Evaluate the capability of FTWs to control phosphorus and nitrogen pollution introduced to a wet pond by urban runoff.
2. Understand terrestrial macrophyte growth in a floating raft environment and the internal translocation of nutrients.
3. Develop an FTW performance assessment model.
4. Provide guidance for design and management of FTWs in urban stormwater treatment facilities.

The hypotheses of this research are:

1. An FTW with 61% coverage can significantly reduce nutrient concentrations in urban stormwater with an average retention time of seven days.
2. Pollutant removals of FTWs could reasonably be described by a compartmental first order kinetic model.

3. Three selected wetland macrophytes can grow and adapt to the environment in the urban retention pond and reproduce in the next year after experiencing ice encasement.

1.4 Study Topics

The hypotheses are examined through a series of research topics organized with major accomplishments.

1.4.1 Topic I

The first topic is the development of an FTW assessment model (Chapter 3), which uses first-order kinetics to provide TP and TN concentration estimates according to given design parameters, such as FTW coverage, water volume, and retention time. Because published FTW studies are primarily mesocosm-scale experiments, it would be unrealistic to interpret full scale FTW performance based on these small physical models. Moreover, full scale performance requires an understanding of the relationship between FTW coverage and pollutant removal efficiency. The assessment model was developed from the FTW literature data and generalized by a bootstrap method. The range of FTW apparent uptake velocity was determined and served as a preliminary FTW model sufficient to guide FTW design.

1.4.2 Topic II

A mesocosm study was conducted at Ashby Pond, Fairfax, Virginia (Chapter 4). Macrophytes were provided to the mesocosm experiment by an in-pond FTW cultural system. There were two native and non-invasive plant species tested in the study. The mesocosm experiment included multiple treatments permitting separation of internal pond processes from those occurring within the FTW, and thus allowing evaluation of the relative water quality improvement due to the presence of the FTWs. Collected water quality and plant growth data were analyzed through statistical methods to understand FTW nutrient removal from an urban wet pond.

1.4.3 Topic III

An in-situ FTW case study was conducted in Ashby Pond (Chapter 5). Experimental data were gathered from the in-pond FTW cultural system in Topic II for a period of 16 months. The vegetation on the FTW and the pond water were sampled periodically in the first six months to understand the phosphorus distribution between above- and below-ground plant tissues and characteristics of the macrophyte growth environment within Ashby Pond. After 16 months of field observations of the in-pond FTW, the ecological benefits associated with the presence of the in-pond FTW and sustainability of the plants on the FTW are evaluated.

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Chapter 2. Literature review

FTWs are also known as artificial floating islands (Nakamura and Mueller, 2008; Shih and Chang, 2006), floating vegetation mats (Smith and Kalin, 2000), floating culture systems (Miyazaki *et al.*, 2000), ecological floating-beds (Li *et al.*, 2010), and ecologically-engineered floating vegetation mats (Strosnider and Nairn, 2010). FTWs function as constructed wetlands in a hydroponic manner (Figure 2-1). Substrate-rooted macrophytes grow on floating rafts with their roots hanging into the water column (Li *et al.*, 2010; Van de Moortel *et al.*, 2010).

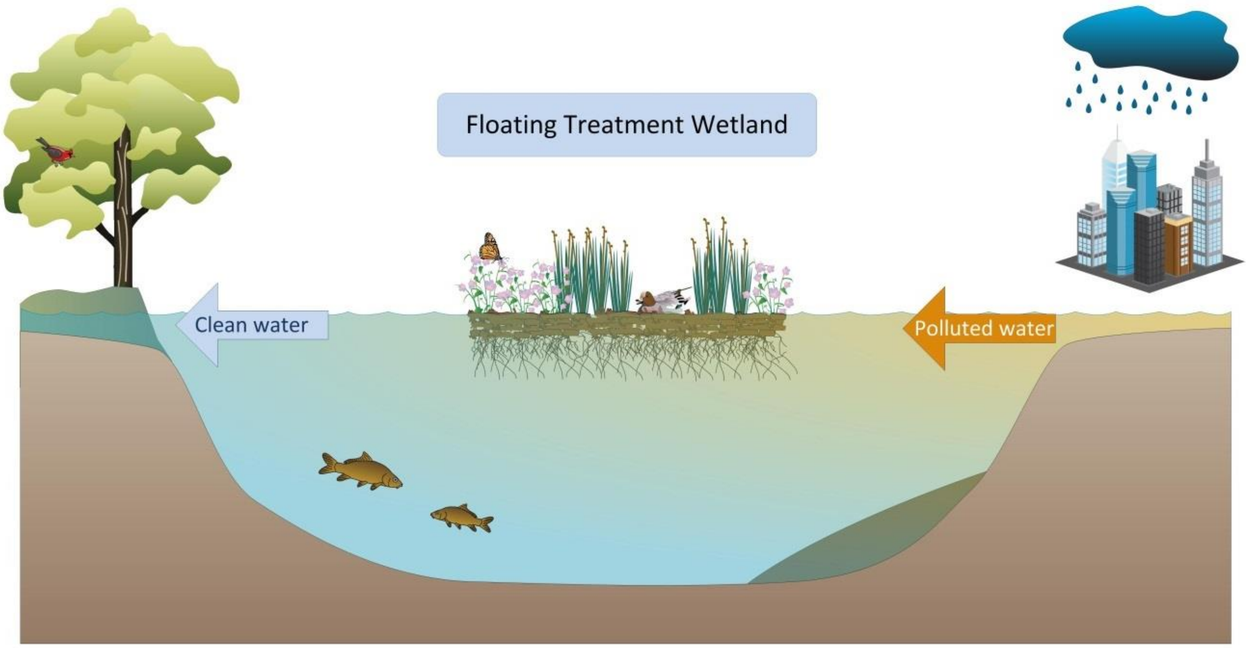

Figure 2-1. Schematic of FTWs in urban wet ponds. Graph is developed by Chih-Yu Wang; symbols are used with the courtesy of the Integration and Application Network, University of Maryland Center for Environmental Science (ian.umces.edu/symbols/)

Appropriately designed FTWs, while artificial, can approach the sustainable function of natural floating wetlands, which are found worldwide in various climates and settings (Guimaraes *et al.*, 2000; Hogg and Wein, 1987; Kadlec, 2009; Mallison *et al.*, 2001; Strosnider and Nairn, 2010; Tsujino *et al.*, 2010). Along the shorelines of lakes, ponds, and estuaries, terrestrial vegetation may proliferate and eventually begin to extend to the water surface by means of flotation. Eventually, the un-rooted vegetation loses the connection with their terrestrial communities and becomes a free floating wetland (Headley and Tanner, 2006; Hubbard *et al.*, 2004; Shih and Chang, 2006). Although growing in a soilless environment, some terrestrial plants in natural floating wetlands could develop self-sustaining systems and respond to various natural and anthropogenic stressors (Krusi and Wein, 1988).

FTWs can also be applied to habitat restoration projects because well-functioning floating wetlands are isolated, secure, and attractive to many species and thus help increase the biodiversity of local ecosystems (American Water Works Association, 2011; Hoeger, 1988; Kato *et al.*, 2009; Kerr-Upal *et al.*, 2000; Nakamura *et al.*, 1995). Billore *et al.* (2007) reported a variety of macroinvertebrates including shredders, grazers, and collectors have been found in artificial floating islands. More fish and shrimp were found under the floating islands than in nearby open water locations (Billore, 2007; Nakamura *et al.*, 1998). In Minnesota, USA, Red-necked Grebes (*Podiceps grisegena*) nested on floating *Typha* spp. island, which was isolated from predators (Nuechterlein *et al.*, 2003). Similarly, artificial floating islands were tested to be a suitable habitat for Dalmatian Pelicans (*Pelecanus crispus*) at Greece (Burgess and Hirons, 1992). Cherry and Gough (2006) stated that floating islands may replenish seed banks and maintain populations of otherwise uncommon plant species. Based on these examples, FTWs can be used for ecosystem restoration in addition to water purification projects depending upon the design.

2.1 Water purification mechanisms of FTWs

FTWs improve water quality by several mechanisms that are based on macrophytes, root systems, microorganisms, and floating rafts (Hubbard, 2010; Li *et al.*, 2010; Shih and Chang, 2006). Similar to a constructed wetland, nutrients and other pollutants are incorporated gradually into biomass, and are thus withdrawn from the aquatic ecosystem. These nutrients may be slowly released back to the environment. However, net storage will continue to increase because plant growth rates are typically higher than decomposition rates for most regions. This is especially true for cooler climates where decomposers are less active (Krusi and Wein, 1988).

2.1.1 Macrophytes

Vegetation takes up pollutants from contaminated water, and some species can enhance aerobic degradation (Billore *et al.*, 2009; Brix, 1997; Van de Moortel *et al.*, 2010). Because they grow hydroponically, macrophytes in FTWs may be more effective in removing nutrients directly from the treated water bodies. In contrast, very little of the phosphorus is assimilated by the macrophytes from the water column in constructed wetlands (Reddy *et al.*, 1999). Additionally, in constructed wetlands, an oxidized rhizosphere is created due to oxygen transportation from the atmosphere into the root zone by certain wetland plants (Brix, 1997). Similarly, this condition also supports bacterial colonies, which may increase aerobic degradation of organics and enhances biological oxygen demand removal in FTWs (Billore *et al.*, 2009).

Another merit of plants on floating rafts is that some species may inhibit pathogens and cyanobacteria (Hoeger, 1988; Nakai *et al.*, 2010; Song *et al.*, 2009). Chen (2011) stated that sites vegetated with great bulrush (*Scirpus zacustris*) were free of pathogens as coliform bacteria and salmonella. It is suggested that the great bulrush exudes microbiocidal substances. Microcystins, which are potent liver toxins, are produced by some species of cyanobacteria, such as *Microcystis* spp., and have been found in drinking water supplies world-wide (Codd, 2000; Westrick *et al.*, 2010). The average *Microcystis* spp. removal efficiency in an aquatic vegetable bed experiment was 78% (Song *et al.*, 2009). The authors asserted that *Ipomoea aquatica* used in the experiment was able to absorb microcystins because microcystins were contained in the roots, stems and leaves. No microcystins were detected in the plant prior to transplanting into the experiment. Song *et al.* (2009) suggested that *Microcystis* sp. is easily trapped or adsorbed by plant roots and degraded by indigenous microorganisms. Net-structure roots are favorable for the acclimation and accumulation of microcystin-degrading bacteria. However, the presence of the microorganisms might not be correlated with the application of aquatic vegetable beds. Nakai *et al.* (2010) also reported that *Typha angustifolia, Scirpus tabernaemontani,* and *Phragmites australis* released anti-cyanobacterial compounds from their roots and enhance water purification.

2.1.2 Root systems

Dense root systems act as filters to trap suspended solids and encourage sediment settling by slowing water flows (Headley and Tanner, 2006; Hoeger, 1988; Nduvamana *et al.*, 2007). The surface area of the root systems may also serve as habitats of microorganisms (Morgan and Martin, 2008; Nduvamana *et al.*, 2007; Osem *et al.*, 2007). Li *et al.* (2010) and Song *et al.* (2011) indicated that abundant nitrifying and denitrifying microorganisms grew on artificial media hanging under a floating raft, like root systems. These microorganisms played an important role in nitrogen reduction. Wang and Sample (2011) reported that artificial roots enhanced TP and TN removal efficiency by 23% and 31%, respectively, when compared to a control.

2.1.3 Microorganisms

Microorganism activities are one of the major pollutant removal processes (Headley and Tanner, 2006; Shih and Chang, 2006; Wen and Recknagel, 2002). The entire underwater surface of the vegetation and floating mats serves as a base for microbial growth (Li *et al.*, 2010). In terms of nutrient removal, the importance of microorganisms may be more significant than plants in FTWs. The first demonstration of this importance is provided in Shih (2006), which reported that 74-95% of TN and 70-92% of TP was removed by microorganisms and other mechanisms in his mesocosm experiment. However, the independent effect of microorganisms was not reported. Protozoa and metazoa were also found to be abundant in the rhizosphere of *Ipomoea aquatica* on an aquatic vegetable bed (Song *et al.*, 2009). These microzooplankton predators were also observed to have a positive effect on removal of cyanobacteria and microcystins. Finally, with the presence of plants, aqueous organic nitrogen and phosphorus are mineralized by the microorganisms into bioavailable forms for plant uptake (Richardson and Vepraskas, 2001). This mechanism could result in more nutrients being removed from impaired water bodies through plant harvest practice.

2.1.4 Raft

Floating rafts on the water surface may block sun light, thus alleviating thermal pollution by decreasing heat gain, and reducing algae populations (Billore *et al.*, 2009; Hubbard, 2010; Li *et al.*, 2007b; Shih and Chang, 2006). Li *et al.* (2007b) reported 89.5% reduction of chlorophyll *a* concentration in an FTW (13.6% coverage) treatment compared with 50% increase in an open water treatment after 25 days. Similarly, the total cell number of phytoplankton in a control plot with no rafts was ten times more than that of other plots with floating rafts (Nakamura and Shimatani, 1997). Although the authors of these two papers ascribed these results solely to nutrient competition between the FTW and algae, they may have ignored the effects of shading from the rafts and overstated FTW nutrient removal. An *in situ* experiment with four shading levels indicated that weak light was unfavorable for phytoplankton growth in the river (Cao *et al.*, 2011). Light availability has been included as a limitation factor of algal growth in several water quality models, such as QUAL2K (Chapra, 2008).

2.2 Advantages of FTWs

FTWs can improve or restore aquatic features in various environmental conditions (Nakamura and Mueller, 2008). The attributes of FTWs effectively control pollutants and address weaknesses of other common BMPs as discussed in the following paragraphs.

2.2.1 Environmental adaptability and variety of applications

FTWs integrate ecological principles and possesses self-organization abilities such as adjusting their community compositions in response to external stresses (Kangas, 2004). For example, a *Typha* spp. floating mat in Canada was reported to be resilient to environmental extremes and recovered quickly from ranges of drying and burning regimes (Krusi and Wein, 1988). The authors concluded that below-ground reserves and nutrients released from damaged plants supported the restoration of the natural floating mat after these significant disturbances.

In addition to their adaptability to a variety of environmental conditions, FTWs have a wide range of possible functions including both nonpoint and point pollution control (Hubbard *et al.*, 2004; Li *et al.*, 2010; Todd *et al.*, 2003). The buoyant rafts provide an ability to float, which is a unique strength of the system. First, FTWs are thus able to be deployed in existing water bodies, such as ponds, lakes, reservoirs, streams, and canals (Cheng, 2006; Headley and Tanner, 2006; Qin, 2009),a significant advantage in areas with high land values and limited space (Li *et al.*, 2009). Second, FTWs are minimally affected by the change of water depth (Van de Moortel *et al.*, 2011), making FTWs an ideal practice in urban areas, where hydrologic conditions vary significantly. In contrast, macrophytes in conventional constructed stormwater wetlands are susceptible to submerged condition during flood events. Such damaged plants may experience chronic dieback and release nutrients into water bodies (Headley and Tanner, 2006). Moreover, FTWs are minimally influenced by low water transparency and high sediment concentrations, which are typical attributes of degraded water bodies (Li *et al.*, 2010). While submerged macrophytes are blocked from sunlight, plants on the rafts continue their photosynthesis processes and absorb nutrients from water for their growth.

2.2.2 Ease of development and management

FTWs could be easily applied not only in developed countries but also undeveloped or developing regions where advanced stormwater and wastewater treatment facilities are lacking, and ecosystems and human are most suffered from degraded water pollution problems (Luo *et al.*, 2011; Pilon-Smits, 2005; Yang *et al.*, 2008). First of all, an FTW is made of floating frames, growth media, and plants. Preparations of the materials and assemblage/operations of the system require no sophisticated technical training according to my experiences in the FTW mesocosm experiment (Chapter 4). Buoyancy calculations of the floating rafts might be the most complicated part of the design processes. Second, minimal or no mechanical/electrical equipment is required, reducing energy consumption and breakdown risk. Third, FTWs may be directly deployed in existing water bodies as described in Section 2.2.1.

Vegetation propagation in FTWs is constrained by the floating raft; in contrast, free-floating plants grow relatively unchecked. While this may be a natural advantage, e.g., water hyacinth and duckweed are naturally buoyant and may also effectively remove nutrients, these free floating plants can proliferate and occupy the entire surface of a water body and may damage aquatic systems (Hubbard, 2010; Wang *et al.*, 2002; Wen and Recknagel, 2002; Wu, 2007). Dark and anoxic conditions under thick floating-plant cover leave little opportunity for the establishment of diverse animal or plant life. Additionally, the floating vegetation can have large negative impacts on fisheries and navigation in tropical lakes (Janse and Van Puijenbroek, 1998; Scheffer *et al.*, 2003). Therefore, the floating raft serves a dual purpose by providing an easy means to manage plant populations and support their growth while extending suitable habitats on the water surface.

The maintenance of FTWs is generally easier to accomplish than constructed wetlands. First, macrophytes may be harvested annually to enhance pollutant removal efficiency and avoid releasing nutrients back into the environment due to plant senescence (Chen *et al.*, 2009). Second, sediments stored in the pond can be easily dredged without excessive damage to the FTW plants (Smith and Kalin, 2000). Moreover, the floating rafts may be assembled or repaired off-site whereas a constructed wetland must be built in place. This property expedites FTW maintenance and enhances its application in water pollution control.

2.2.3 Nutrients recycling and economic returns

The pollutants stored in biomass are removed from aquatic ecosystems by harvesting the vegetation. The harvested biomass could be used as animal or human food if no hazardous substances are present as a result of contamination in the treated water bodies. Vaillant *et al.*

(2003) treated domestic wastewater with *Datura innoxia*, a pharmaceutical raw material. *Chrysopogon zizanioides* tested by De Stefani *et al.* (2011) is grown for animal feed, cosmetics and aromatherapy. Other potential economic returns may include production of biogas, biofertilizer and biomaterial made from the harvested plant tissue (Hu *et al.*, 2010; Li *et al.*, 2007a; Wen and Recknagel, 2002) . *Vallisneria spiralis* and *Eichhornia crassipes* were utilized to remove lignin and metals from paper mill and distillery effluents (Singhal and Rai, 2003). These plants produced significantly more biogas per unit plant dry weight than those grown in unpolluted water.

2.2.4 Economical and natural pollution control technology

FTWs offer great potential to become an economical and sustainable pollution control technology (Todd *et al.*, 2003). First, FTWs could be installed in many existing water bodies (Section 2.2.1); therefore, construction or land acquisition expenses normally associated with stormwater detention and treatment could be significantly reduced (Li *et al.*, 2009; Nduvamana *et al.*, 2007). Second, the major FTW treatment mechanisms are plant growth and microorganism activity. The utilization of these natural processes has been tested and applied in other ecological engineering practices, such as constructed wetlands, for over 15 years because of the low capital cost, very low operating cost, and potential ecological benefits (Billore *et al.*, 2009; Pilon-Smits, 2005; Yang *et al.*, 2008). Although the basic pollutant removal mechanisms are similar, the effectiveness of FTWs is still not well-defined in current studies.

While nutrient removal performance and cost are obviously key bottom line considerations, as we look to the future, it is clear that much greater consideration of our energy resources will be a major concern for scientists and engineers (Shilton, 2005). Energy expenses of FTWs are significantly lower than conventional engineering based practices, because FTWs are driven by solar power (Todd *et al.*, 2003). Todd, et al. (2003) reported that energy requirements of a poultry processing plant dropped 74% after converting a sequencing batch reactor to an FTW system. However, there is insufficient cost data for FTWs in the literature. The cost of a similar ecological engineering practice, phytoremediation, was reportedly 10% of the cost of engineering-based remediation methods, partly due to solar-driven biological processes (Pilon-Smits, 2005).

Depending on the material and personnel costs, the manufacturing and installation costs of FTWs vary (Table 2-1). Some commercial floating rafts used for garden ponds are directly available at local nursery stores or online shops and are 131 to 419 USD/m^2. (MAN, 2011; PGD, 2012). However, it is possible that the costs could be significantly reduced. The floating rafts could be made of lower cost materials (e.g. bamboo stems or recycled plastic bottles), and macrophytes could be transplanted from local habitats (Li *et al.*, 2010). Several community/school FTW projects were built with recycled plastic bottles and free wetland plants by volunteers/students in Taipei, Taiwan (Chen, 2011).

Table 2-1. The manufacturing and installation unit cost of FTW (macrophytes not included)

Data source	**Billore *et al.* (2009)**	**Ashby Pond budget[a]**	**Boutwell (2001)**	**Nursery store (commercial product)[b]**
Country	India	U.S.	U.S.	U.S.
Unit cost (USD/m^2)	60	62	248	131 to 419

[a] A project in this research proposal. This estimation is based on a pilot study (Wang and Sample, 2011).
[b] MAN, 2011; PGD, 2012.

2.3 Disadvantages of FTWs

As with any new treatment technology, there are physical, chemical and biological issues that must be assessed before consideration of FTWs for use in full-scale applications. These limitations should be considered to optimize treatment effectiveness and minimize any negative impacts.

2.3.1 Chemical limitations

Chemical properties of water bodies can be affected by the presence of the floating rafts. Low dissolved oxygen (DO) conditions may occur in the water column under the floating wetland (Headley and Tanner, 2006; Mallison *et al.*, 2001; Tanner and Headley, 2011). This may be caused by reduction in diffusion of oxygen from the atmosphere due to obstruction by the floating mats and insufficient algal photosynthesis activity because of the shading effects of the rafts. However, Nduvamana *et al.* (2007) and Billore *et al.* (2009) reported higher DO values were recorded in an FTW treatment compared with a control in their mesocosm experiments.

Van de Moortel *et al.* (2010) stated that root oxygen release was higher than oxygen diffusion from the air based on redox potential measurement. Therefore, the influence on DO concentration is still uncertain.

2.3.2 Physical limitations

The physical limitations of FTWs include raft structure and buoyancy. Similar to other floating objects, floating rafts are vulnerable to strong waves, which may seriously damage the structure (Strosnider and Nairn, 2010). It was reported that FTWs were severely damaged by a Category 4 hurricane at Liyutan Reservoir, Taiwan (Huang, 2010). Moreover, there is a risk that biomass accumulation may exceed the buoyancy provided by floating rafts. This disadvantage could be overcome by reasonable prediction of the maximum biomass during design stage.

2.3.3 Biological limitations

Water purification efficiency may be limited by the performance of plants and microorganisms. The uptake of pollutants is influenced by the bioavailable fraction of the elements (Pilon-Smits, 2005). Phosphate, nitrate, and ammonium are the major forms of phosphorus and nitrogen taken up by the plants (Marschner, 1995; Reddy *et al.*, 1999), but these directly bioavailable forms are only part of the total nutrients in the water bodies (Richardson and Vepraskas, 2001).

Oil present in the polluted urban stormwater may pose great challenges to biological treatment systems. Cheng (2006) reported *Typha orientalis*, *Ruellia brittoniana*, *Angelonia goyanzensis*, *Celosia argentea* on artificial floating islands were severely damaged by oil from recreational boats and untreated domestic wastewater. The root system was coated with the oil, which resulted in high mortality rate of these species. However, *Hygrophila pogonocalyx* was an exception, as it adapted to the oil stress in the same study.

Macrophyte species selection is critical not only to pollutant removal but also to ecosystem integrity. If invasive species were introduced by FTWs and other practices, these plants could form monotypes and significantly affect biodiversity, ecosystem function, and human uses of the affected environments (Zedler and Kercher, 2004). Some invaders have high nutrient uptake rate and grow rapidly (Sheley *et al.*, 2006). Although invasive species with such attributes can enhance FTW nutrient removal, the negative impacts on the ecosystem or the costs of habitat restoration may be more significant. However, it may be acceptable to use the invaders if they are already presented at FTW application sites. Following are suggested plant selection procedures:

1. <u>Native species</u>: adapt to local habitats.
2. <u>Non-invasive species</u>: prevent local ecosystem degradation.
3. <u>Terrestrial species</u>: control plant propagation (Section 2.2.2).
4. Wetland plants or ones able to thrive in hydroponic environment.
5. <u>Plants with aerenchyma tissue</u>: transport oxygen into water bodies (recommended, Section 2.2.1).

2.4 Applications and Studies of FTW

According to Shih and Chang (2006), the earliest artificial floating islands were designed for a Canada goose habitat restoration project. The practice was also used for biodiversity and habitat restoration (Billore, 2007; Hancock, 2000; Hoeger, 1988; Todd *et al.*, 2003) and shoreline protection (Hoeger, 1988; Nakamura and Shimatani, 1997; Shih and Chang, 2006). The water quality-related FTW literature has been published mainly within the last ten years. Some studies indicate that FTWs can restore or slow the eutrophication process on lakes and reservoirs (Garbett, 2005; Li *et al.*, 2010; Todd *et al.*, 2003; Zhen, 2002), inhibit algae growth (Garbett, 2005; Nakai *et al.*, 2010), reduce microcystin toxins and antibiotics (Li *et al.*, 2010; Song *et al.*, 2009; Xian *et al.*, 2010), treat domestic and agricultural wastewater (Hubbard *et al.*, 2004; Nduvamana *et al.*, 2007; Todd *et al.*, 2003; Wen and Recknagel, 2002; Wu *et al.*, 2006b), and purify urban stormwater (Castro *et al.*, 2005; Chen *et al.*, 2009; Revitt *et al.*, 1997; Tanner and Headley, 2011). These studies were further divided into nine categories based on the source of the polluted water (Table 2-2). Most research focused on the higher nutrient wastewater and/or point source pollution. Only three of 29 published studies addressed FTW performance in urban stormwater (Figure 2-2). These publications focused on the removal efficiency of the following water constituents: nitrogen, phosphorus, biochemical oxygen demand, chemical oxygen demand, total organic carbon, sulfate, chlorophyll *a*, cyanobacteria toxins, suspended solids, and metals.

The TP and TN initial concentrations, removal rates and efficiencies of FTWs from the reviewed literature are summarized in Tables 2-3 and 2-4. The reported removal efficiencies were 18 to 97.7% for TN and -69 to 90% for TP, while the initial concentrations ranged from

4.55-197 mg/L and 0.1-15 mg/L of TN and TP, respectively. In addition to the initial concentrations, the variability in reported removal efficiencies may be caused by other factors. These factors could include chemical forms of pollutants, macrophyte species and ages, raft materials and coverage, experiment locations and seasons, experiment water volumes, detention time, flow condition, and temperatures. It is also possible that the aforementioned FTW performance may have issues in their evaluation and/or have not been accurately reported. Three of the aforementioned publications may overestimate the performance because the solutions were made from chemicals or fertilizers, which provide nutrients mostly in bioavailable forms. For example, Tanner and Headley (2011) used artificial stormwater, which had 85-90% phosphorus in dissolved reactive forms. Moreover, the FTW removal efficiencies may have resulted from very long detention times, such as 60 days in Hu *et al.* (2010). Finally, the extent of coverage by rafts may be a critical factor in evaluating the performance of the system. The coverage of the published FTW experiments ranged from 3.3 to 100% (Miyazaki *et al.*, 2004; van Oostrom, 1995). In this section, a review of published studies demonstrates current progress of FTW research and a high variability of experimental methods/conditions, including water sources, level of initial concentration, reaction time, FTW coverage. This uncertainty requires further investigation to facilitate using FTWs for urban runoff treatment. These gaps are listed in Section 1.3 and studied through modeling, mesocosm, and in-situ experiments in this research.

Table 2-2. Summary of available FTW studies in Figure 2-2.

Solution type	**Number of studies**	**Citations**
Domestic wastewater	7	Billore *et al.*, 2007; Billore *et al.*, 2009; Miyazaki *et al.*, 2000; Vaillant *et al.*, 2003; Van de Moortel *et al.*, 2010; Wu *et al.*, 2006b; Xiong *et al.*, 2011
Reservoir/Lake (eutrophic)	6	Garbett, 2005; Hu *et al.*, 2010; Li *et al.*, 2007b; Li *et al.*, 2010; Shih and Chang, 2006; Wu *et al.*, 2006b
Un-defined	5	Li *et al.*, 2007a; Miyazaki *et al.*, 2000; Miyazaki *et al.*, 2004; Shin *et al.*, 2004; Strosnider and Nairn, 2010
Agricultural runoff	3	Stewart *et al.*, 2008; Wen and Recknagel, 2002; Yang *et al.*, 2008

Table 2-2. (Continued)

Solution type	**Number of studies**	**Citations**
Urban stormwater	3	Chen *et al.*, 2009; Revitt *et al.*, 1997; Tanner and Headley, 2011
Acid mine drainage	2	Fyson *et al.*, 1995; Smith and Kalin, 2000
Aquaculture wastewater	1	Nduvamana *et al.*, 2007
Meat process	1	van Oostrom, 1995
Swine lagoon	1	Hubbard *et al.*, 2004

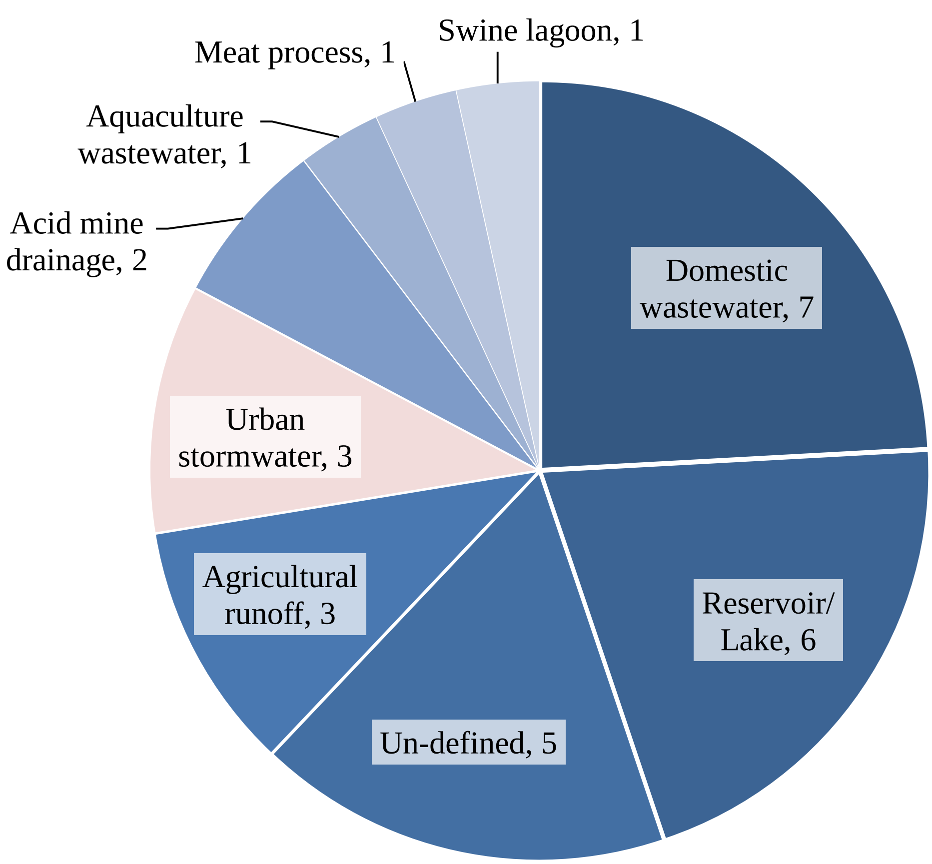

Figure 2-2. Focus areas of published FTW studies (category, paper number). Citations of each study are listed in the Table 2-2.

Table 2-3. Comparison of FTW TP removal performance of 14 studies.

Rank ID[+]	Initial concentration (mg/L)	Removal rate (g/m^2-d)	Removal efficiency (%)[δ]	Retention time (day)	Citation
1[a]	0.10	-	40[*]	7	Tanner and Headley, 2011
2[b]	0.15[*]	0.08	91	20	Li *et al.*, 2007b
3	0.38	0.04[*]	61[*]	60	Hu *et al.*, 2010
4	0.72	-	-69[*]	15	Nduvamana *et al.*, 2007
5[b]	0.97	-	63[*]	7	Li *et al.*, 2010
6	1.04	0.40[*]	-	-	Miyazaki *et al.*, 2000
7[b]	1.09	-	65[*]	0.4	Wu *et al.*, 2006b
8[ab]	1.54	0.11	13	1	Yang *et al.*, 2008
9[a]	1.60	0.62[*]	-	-	Wen and Recknagel, 2002
10	2.16	0.06[*]	22	11	Van de Moortel *et al.*, 2010
11[b]	3.10	0.74[*]	92	50	Chen *et al.*, 2009
12	4.31	-	76	0.8	Wu *et al.*, 2006b
13[b]	9.00	-	38	2	Vaillant *et al.*, 2003
14	15.00	0.35[*]	-	56[*]	Hubbard *et al.*, 2004

[+] Low to high initial concentration
δ Concentration based calculation
a Solutions were made from chemicals. Others used raw solutions directly.
b Integrated systems include aerator, circulator, freshwater clams, and/or biofilm carrier.
[*] interpolated from graph or calculated from available data
- data not available

Table 2-4. Comparison of FTW TN removal performance of 15 studies.

Rank ID[+]	**Initial concentration (mg/L)**	**Removal rate (g/m^2-d)**	**Removal efficiency (%)**[δ]	**Retention time (day)**	**Citation**
1	4.55	0.34[*]	41[*]	60	Hu *et al.*, 2010
2[b]	5.15	-	62[*]	7	Li *et al.*, 2010
3	6.06[*]	1.41[*]	39[*]	2[*]	Xiong *et al.*, 2011
4[ab]	7.94	2.74	64	1	Yang *et al.*, 2008
5[b]	8.06	3.98	84	20	Li *et al.*, 2007b
6	10.00	-	25	-	Billore *et al.*, 2007
7[ab]	11.29	2.86[*]	98	50	Chen *et al.*, 2009
8	19.36	-	18[*]	15	Nduvamana *et al.*, 2007
9	21.80	1.18[*]	42	11	Van de Moortel *et al.*, 2010
10	22.00	-	45	-	Billore *et al.*, 2007
11	39.81[*]	-	34[*]	-	Billore *et al.*, 2009
12[b]	41.00	-	52[*]	2	Vaillant *et al.*, 2003
13	62.93[*]	1.93[*]	47[*]	60	Billore *et al.*, 2009
14	80.00	2.01[*]	-	56[*]	Hubbard *et al.*, 2004
15	197.00	5.30	47	7	van Oostrom, 1995

[+] Low to high initial concentration
[δ] Concentration based calculation
[a] Solutions were made from chemicals. Others used raw solutions directly.
[b] Integrated systems include aerator, circulator, freshwater clams, and/or biofilm carrier.
[*] interpolated from graph or calculated from available data
- data not available

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Chapter 3. Assessing floating treatment wetlands nutrient removal performance through a first order kinetics model and statistical inference

3.1 Abstract

A floating treatment wetland (FTW) is an ecological approach which seeks to reduce point and nonpoint source pollution in receiving waters. This technology has received increasing attention recently. Subsequent studies were conducted at worldwide locations; despite these efforts, reliable estimates of the FTW performance remains a significant research gap. This paper describes the development of a robust and simplified integrated FTW (i-FTW) model that includes uncertainty. The performance of FTWs was separated from other treatment processes ongoing within their respective water bodies. This approach facilitates generalization of the model and allowing parameters to adjust to each applied water body characteristic. A bootstrap method was incorporated to estimate uncertainty and generate more robust predictions of performance.

Water concentration time series data were described by a first order kinetics i-FTW model that generates an FTW performance parameter: FTW apparent uptake velocity (v_f). A comprehensive literature search of i-FTW studies was conducted to collect total phosphorus (TP) and total nitrogen (TN) concentrations data. The v_f values were calculated from these studies and estimated using a bootstrap percentile method. The best estimation of median and expected range with 95% confidence interval of the v_f were 0.048 (0.018 – 0.059) and 0.027 (0.016 – 0.040) m/day for TP and TN, respectively. The goodness of fit (R^2) of the i-FTW model on water concentration time series data of the i-FTW experiments was 0.92 ± 0.30 for TP and 0.86 ± 0.38 for TN data (mean ± SD). This model provides insights into compartmental treatments of i-FTW systems and serves as a preliminary tool to select extent of FTW coverage when designing an i-FTW system. Further research to resolve limitations of the model application is suggested.

Keywords

Floating treatment wetland, Pollution control, First order kinetics, Assessment model, Bootstrap, Uncertainty

3.2 Introduction

Floating Treatment Wetlands (FTW) are an innovative ecological approach to control water quality degradation from point and nonpoint source pollution (Headley and Tanner, 2012; Hubbard, 2010; Rangarajan *et al.*, 2012). An FTW is a system with floating mats and associated ecological communities, such as macrophytes, macro invertebrates, zooplankton, and biofilms (Faulwetter *et al.*, 2011; Hubbard *et al.*, 2004; Kato *et al.*, 2009; Song *et al.*, 2011). FTWs can be applied to most treatment facilities and a range of water bodies (Tanner and Headley, 2011). FTWs and applied water bodies provide a variety of mechanisms that can treat the pollutants within aquatic systems (Figure 3-1). Plant and periphyton uptake and plant root filtration are the contributors in the FTW system (Borne *et al.*, 2013; Headley and Tanner, 2012). Within a water body, pollutants are removed by sedimentation, assimilation by algae/bacteria, and adsorption to bottom sludge (Shilton, 2005; Wetzel, 2001). For the purpose of this study, we are defining an integrated FTW (i-FTW) to be a system that combines separate treatments from the FTW and applied water body, such as retention ponds or experimental water tanks. It is critical to identify differences between FTWs and i-FTWs because most published studies discuss performance of the entire system (i-FTW) instead of the FTW alone.

i-FTW system has recently received increasing attention. The technology was tested in reservoirs (Garbett, 2005), lakes (Hu *et al.*, 2010), urban stormwater ponds (Zhao *et al.*, 2012b), rivers (Zhou *et al.*, 2012), facilities that treat wastewater from mining (Kalin and Chaves, 2003), refinery plants (Li *et al.*, 2012), meat processing plants (van Oostrom and Russell, 1994), swine farms (Hubbard *et al.*, 2011), aquaculture (Nduvamana *et al.*, 2007), agriculture (Wen and Recknagel, 2002), and urban sewer systems (Van de Moortel, 2008). These studies were conducted in 17 countries over five continents. They include Argentina (Torremorell and Gantes, 2010), Australia (Wen and Recknagel, 2002), Belgium (Van de Moortel *et al.*, 2012), Brazil (Guimaraes *et al.*, 2000), Canada (Kalin *et al.*, 2006), China (Xin *et al.*, 2012), France (Ladislas *et al.*, 2013), Germany (Hoeger, 1988), India (Billore *et al.*, 2009), Italy (De Stefani *et al.*, 2011), Japan (Nakai *et al.*, 2010), New Zealand (Tanner and Headley, 2011), Singapore (Chua *et al.*, 2012), Sri Lanka (Weragoda *et al.*, 2012), Taiwan (Shih and Chang, 2006), Uganda (Kansiime *et al.*, 2005), United Kingdom (Garbett, 2005), and the United States (Chang *et al.*, 2013). However, despite the accumulated knowledge of the capabilities of i-FTW in pollution control, reliable estimates of i-FTW performance represent a significant research gap. In particular, a generalized

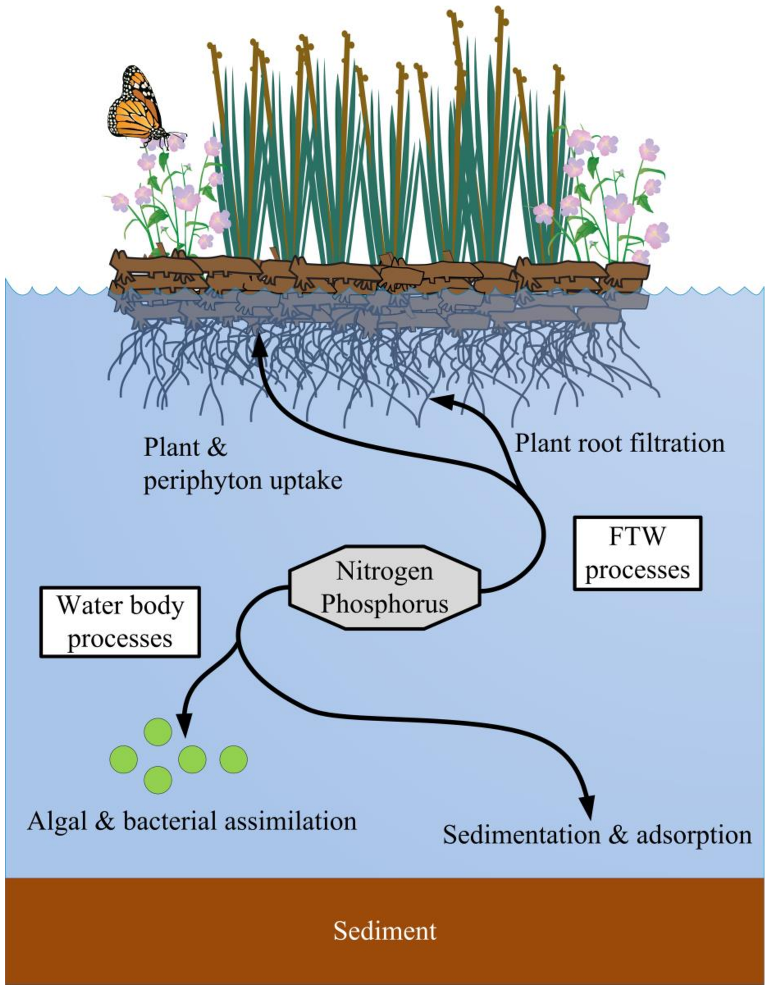

Figure 3-1. Schematic of an integrated FTW system (i-FTW). The phosphorus and nitrogen are removed separately by the FTW and applied water body, such as rivers or retention ponds. Illustrations are used with the courtesy of the Integration and Application Network, University of Maryland Center for Environmental Science (ian.umces.edu/symbols/)

method or model that can predict performance of an i-FTW system is needed. This need becomes more acute because of the growing awareness of FTW technology and its wide applicability.

Published assessment parameters require further analysis to adequately predict performance of i-FTW systems. Removal rate (RR, $mg/m^2/day$) and removal efficiency (RE, %) are the two most commonly reported parameters used to assess the performance of i-FTW systems. The definition of removal rate (RR) equates to a total pollutant mass reduction (mg) of the treated water divided by the FTW surface area (m^2) and reaction time (day) (Tanner and Headley, 2011; van Oostrom, 1995; van Oostrom and Russell, 1994). The removal efficiency (RE) is calculated as a percentage of the concentration reduction of the initial concentration (Borne *et al.*, 2013; De Stefani *et al.*, 2011; Li *et al.*, 2011; Sun *et al.*, 2009; van Oostrom, 1995;

Wu *et al.*, 2006a; Xian *et al.*, 2010; Zhou and Wang, 2010). The RR calculation does not consider water concentration changes and assumes treatment is constant, i.e. zero order kinetics, which may overestimate the performance of i-FTW systems with extended reaction times. Additionally, the RR method attributes all of pollutant reduction to the FTW and ignores contribution of the applied water body. The RE method may not adequately respond to the variation of FTW coverage, water volume, and reaction time. These parameters are absent from the method and thus cannot be used to adjust the RE value. A third assessment approach, plant uptake rate, is similar to the RR method with the total pollutant mass reduction (mg) derived from harvested plant tissues instead of from water concentration reduction (Hubbard *et al.*, 2011; Hubbard *et al.*, 2004; Wen and Recknagel, 2002; Zhu *et al.*, 2011). The third approach may properly represent the plant contributions to treatment, but also exclude any impacts from floating mats and associated periphyton. Because of the aforementioned limitations, there is a need for assessment models that can provide reasonable predictions based on the varied characteristics of the FTW and the water body.

The most widely adapted model for predicting pollutant removal is the 1st order kinetics model (Hendricks, 2006). Developed pollution reduction models use a variety of 1st order kinetics functions to describe behavior of phosphorus (Houng and Gloyna, 1984), nitrogen (Wong *et al.*, 2006), and total suspended solid (Park and Roesner, 2012) in various treatment facilities or natural water bodies (Aumen, 1990; Hvitved-Jacobsen *et al.*, 1994; Torgersen *et al.*, 2004). A compartmental modeling approach has been adapted to simulate multiple processes within aquatic systems such as streams, reservoirs, and wetlands (Aumen, 1990; Wynn and Liehr, 2001). Chapra (1997) described a mass-balance model for a well-mixed lake. The model separates removal processes into several terms, including water reaction and settling processes. The ease of using 1st order kinetics provides great advantages; however, selection of a 1st order reaction rate to generate reliable predictions is difficult because the parameter is sensitive to environmental conditions (Kadlec, 2000). It may be more reasonable to suggest a range of the predictions than a specific number.

We suggest addressing this range, or uncertainties, in model parameters through a statistical approach, known as the bootstrap method. The bootstrap method estimates a confidence interval (CI) of the parameter of interest (Hjorth, 1994). A bootstrap resampling with replacement procedure has several advantages in comparison to parametric statistical methods

(Newman *et al.*, 2000). The computationally intensive bootstrap method can provide estimates without an explicit distribution, and alleviates the difficulty of selecting and validating an appropriate distribution as required by parametric methods (Chernick, 2008). Another advantage of using the bootstrap method is that it offers reasonable evaluations based on limited data, common occurrences in ecological risk assessment (Jagoe and Newman, 1997).

The objective of this research is to develop a robust and simplified i-FTW model that can be widely applied and used for design, further evaluation and assessment of i-FTW systems. The performance of FTWs is separated from the applied water bodies in the i-FTW model. By dividing these two mechanisms, the model can become more general. The water reaction rates are adjusted according to the applied water body characteristics. Ranges of FTW reaction parameters are quantified to provide more robust predictions of the i-FTW performance. For purpose of this paper, we assume that the entire i-FTW treatment processes can simply be described by 1^{st} order kinetics.

3.3 Materials and Methods

3.3.1 Data sources

A comprehensive literature search of i-FTW studies was conducted, and a screening process employed to identify papers that satisfy five criteria to assess the i-FTW model parameters and other analyses (Tables 3-1 and 3-2). The first criterion is a published study of an FTW with experimental data, which data includes TP and TN concentrations, water volumes with dimensions (length × width × depth), FTW size, and reaction time. TP and TN were the most commonly studied constituents. Some data were gathered through direct contact with authors if the information in their papers was insufficient for our purposes (Table 3-2). Second, the experiments included a control group, either with water (C) or with water and floating mats (CM). Third, experimental water was unaltered or sufficiently similar to natural or polluted conditions. Artificially polluted water made from chemicals or fertilizers may not represent real conditions at the field scale because it contains phosphorus and nitrogen predominately in bioavailable forms. A large portion of the readily consumable nutrients may lead to overestimation of the FTW TP and TN removal. For example, particulate and organic phosphorus, two major constituents of TP in natural waters, are non-bioavailable and require more efforts to be removed. Fourth, hydroponic experiments are excluded due to their design are different from those of the i-FTWs. Although plants in both systems remove pollutants directly from water, there are additional floating mat effects which are absent from the hydroponic systems. Pollutants are removed by microorganism growing on the entire underwater surface of the floating mats (Li *et al.*, 2010). The fifth and final criterion was that studies are written in English and published in journals listed in the in Science Citation Index. Exceptions from the five criteria are those papers with concentration time series data, which were used for evaluating goodness of fit of the i-FTW 1st order kinetics model (Section 3.3.3).

Fourteen studies were selected for our analyses. Among the selected experiments, the FTW apparent uptake velocity (v_f) was evaluated from the 12 studies that clearly described their experiment design. TP and TN dynamic changes in the ten publications were used to test the goodness of fit of the i-FTW model. The site locations and annual maximum and minimum air temperatures of the 14 studies were listed in Table 3-1 (Global Support Limited, 2013). The studies were collected from worldwide locations, including Belgium, India, and New Zealand; however, most of the studies were conducted in China, especially in the last three years. The second criterion, requirement of a control group, is the most common reason that other i-FTW experiments were excluded. The experimental water used in the 14 selected studies covered both point and nonpoint source pollution, such as meat processing plant wastewater and river water (Table 3-1). This variety resulted in a wide range of initial concentrations: 0.38 – 2.28 mg/L for TP and 2.99 – 197 mg/L for TN (Table 3-2).

The selected 14 studies include various experiment designs (Table 3-2). While there were 25 plant species tested, the floating mats in the studies were also built from different materials. Plastic foam is the most common substances used in i-FTW studies as growth media. Other materials include coconut coir, natural floating mats, and dredge sediments. The floats were made from PVC (polyvinyl chloride) tubes or plastic bottles. In addition to the diversity of the FTW construction materials, several i-FTW papers evaluated the performance of the modified i-FTW systems, which were combined with other pollution control mechanisms. Li *et al.* (2010) and Song *et al.* (2011) integrated fresh water clams and biofilm carriers into their i-FTW systems. Wang *et al.* (2012) installed aeration pumps to test potential enhancement of the system. The variability of the plant species, floating mat materials, and system design poses a challenge to generalize the i-FTW model parameters. However, the bootstrap method is introduced to address such uncertainty issues (Section 3.3.4).

Table 3-1. Summary of the i-FTW studies and assigned ID codes used in Tables 3-2 to 3-4 and 3-6.

Citation	**Experimental water source**	**Location & Annual max. to min. temperature (℃)***	**Plant species**	**ID[δ]**
van Oostrom, 1995	Meat processing wastewater	Hamilton, New Zealand 24 to 4	*Glyceria maxima*	95-V
Billore *et al*., 2009	River	Ujjain, India 40 to 10	*Phragmites karka*	09-B
Sun *et al*, 2009	River	Guangzhou, China 33 to 10	*Canna* sp.	09-S
Hu *et al*., 2010	Lake	Wuhan, China 33 to -1	*Acorus calamus*	10-H
Li *et al*., 2010	Lake	Wuxi, China 31 to 0	*Ipomea aquatica*	10-L
Van de Moortel *et al*., 2010	Domestic wastewater	Drongen, Belgium 22 to 1	*Carex* spp. (Dominant species)	10-V
Xian *et al*., 2010	Swine farm wastewater	Nanjing, China 32 to -2	*Lolium multiflorum* Lam. 'Dryan'	10-X-d
			Lolium multiflorum Lam. 'Waseyutaka'	10-X-w
			Lolium multiflorum Lam. 'Tachimasari'	10-X-t
Zhou and Wang, 2010	River	Nanjing, China 32 to -2	*Oenanthe javanica* (Blume) DC	10-Z
Li *et al*., 2011	Urban pond	Hangzhou, China 33 to 1	*Lolium perenne* 'Top One'	11-L-t
			Lolium perenne 'respect'	11-L-r

Table 3-1. (Continued)

Citation	**Water source**	**Location & Annual max. to min. temperature (ºC)***	**Plant species**	**IDδ**
Song *et al*., 2011	Urban pond	Nanjing, China 32 to -2	*Ipomea aquatica*	11-S
Li *et al*., 2012	Refinery wastewater	Hangzhou, China 33 to 1	*Geophila herbacea* O Kuntze	12-L-o
			Loliurn perenn ‘Caddieshack’	12-L-c
			Loliurn perenne Topone	12-L-t
			Loliurn perenne L.	12-L-l
Wang *et al*., 2012	River	Tianjin, China 31 to -8	*Phragmites australis* & *Typha latifolia*	12-W
Zhao *et al*.,2012	Urban pond	Hangzhou, China 33 to 1	*Zizania caduciflora*	12-Za-z
			Triarrhena lutarioriparia	12-Za-tl
			Thalia dealbata	12-Za-td
			Vetiveria zizanioides	12-Za-v
			Miscanthus sinensis Anderss sp.	12-Za-m
			Acorus calamus	12-Za-a
Zhou *et al*., 2012	River	Nanjing, China 32 to -2	*Rumex acetosa* Linn.	12-Zo

* Global Support Limited (2013).
δ Year-Author(-Plant species).

Table 3-2. The experimental design details of the i-FTW studies.

ID	Water					Floating mat		
	TP[α] (mg/L)	TN[α] (mg/L)	Volume (m^3)	Depth (m)	Time[β] (d)	Size (m^2)	Cover ratio (%)	Materials
95-V	-	197.00	1.15	0.40	7^r	2.88	100	Natural floating wetland
09-B	-	-	7.80	1.30	60^d	2.00	33	Coconut coir, bamboo, PVC pipes, and galvanized iron wire
09-S[+]	-	8.71	0.48	0.50	6^d	0.96	100	PS foam
10-H	0.38	4.55	7.00	1.12	60^d	0.64	10	Lake sludge, furnace slag, and perlite
10-L[+]	0.97	5.15	12.24	1.70	7^r	1.00	14	PP random copolymer plate and bottles
10-V	2.16	21.80	1.08	0.90	11^d	0.77	64	Coconut coir, plastic pipes, foam, and wire netting.
10-X[+]	1.92	17.10	0.06	0.30	35^d	0.20	100	HDPE foam plate
10-Z[*+]	0.68	12.58	0.04	0.12	35^d	0.20	61	PS foam
11-L	0.56	2.99	-	-	20^d	-	-	PE foam
11-S[+]	-	5.01	1.70	1.50	7^r	0.20	18	PVC plate, nylon net, and bottles
12-L	0.52	15.60	-	-	35^d	-	-	PE foam and peat
12-W	2.20	-	2.00	1.00	7^r	2.00	100	Ceramic pellets, PE foam, and PVC tubes
12-Za	2.28	8.12	0.12	0.60	16^d	0.20	100	PU foams
12-Zo[+]	-	15.07	0.04	0.12	48^d	0.20	61	PS foam

[α] Initial concentration.
[β] r: Retention time (continuous flow), d: Detention time (standing water).
[+] Design details gathered through personal communication with the authors.
[*] Purification phase data.

3.3.2 i-FTW 1st order kinetics model

The 1st order kinetics has been widely adapted to simulate concentration changes in pollution control systems, including ponds/lakes (Huber *et al.*, 2006; Wong *et al.*, 2006) and constructed wetlands (Carleton *et al.*, 2001; Kadlec and Knight, 1996; Wynn and Liehr, 2001). The i-FTW 1st order kinetics model is defined by Equation 3-1 adapted from Chapra (1997). The i-FTW reaction rate ($k_{i\text{-}FTW}$) equates to a general 1st order volumetric rate and includes two terms: water body and FTW reactions. The applied water body processes of sedimentation, algal/bacterial assimilation, and adsorption to bottom sludge are combined into a water body reaction rate (k_w) for simplicity (Shilton, 2005). The FTW nutrient removal mechanisms, including plant uptake and microorganism activities, are lumped into an FTW apparent uptake velocity (v_f) (Headley and Tanner, 2012; Wang and Sample, 2011). Because the presence of the FTW in a water body is similar to a surface area of the sediment-water interface, the FTW pollutant removal is suggested be formulated as a flux across the FTW surface (A_f) similar to a settling process. The complete FTW reaction term is expressed as $v_f \times A_f/V$. The FTW size and system water volume ratio (A_f/V) can be converted to FTW coverage ratio and system average water depth ratio (γ/H) by dividing both the A_f and V by the water body surface area.

$$c_{t,i-FTW} = c_0 e^{-(k_{i-FTW})t} = c_0 e^{-(k_w + v_f \frac{A_f}{V})t} = c_0 e^{-(k_w + v_f \frac{\gamma}{H})t} \qquad \text{Equation 3-1}$$

where

A_f = surface area of the FTW (m^2);

c_t = concentration at time = t (mg/L);

c_0 = concentration at time = 0 (mg/L);

H = average water depth (m);

$k_{i\text{-}FTW}$ = a 1st order reaction rate of the i-FTW system (day^{-1});

k_w = a 1st order reaction rate of the applied water body (day^{-1});

r = FTW coverage ratio = FTW area / water body surface area;

t = reaction time (day);

V = volume of water body (m^3);

v_f = FTW apparent uptake velocity (m/day).

The compartmental i-FTW model in Equation 3-1 describes the FTW and applied water body processes in two separate terms. The water body reaction rate (k_w) is determined from published literature with study site characteristics similar to the applied water bodies. For example, Vollertsen *et al.* (2009) reported the water reaction rate (k_w, day^{-1}) of a high way retention pond at Norway was 0.048 for TP and 0.014 for TN. A second approach is to analyze concentration changes of the applied system (before FTW installation). This method was described in Section 3.3.3 to quantify the water body reaction rates (k_w) of each i-FTW study. Another reaction rate factor in the i-FTW model, FTW apparent uptake velocity (v_f), is evaluated in this study (Section 3.3.3). One limitation of the i-FTW model is that the model ignores effects of the FTW on the applied water bodies and assumes a constant reaction rate (k_w) in the pre- and post-FTW installation condition. Moreover, the i-FTW model was formulated to describe reactions in discrete batches or steady-state plug flow reactors (PFRs) (Chapra, 1997; Wong *et al.*, 2006). The flow conditions of the 14 selected studies (Section 3.3.1) include two types: standing water and continuous flow. The first type is regarded as discrete batches, whereas the later condition is assumed to be PFRs at steady-state because of their long retention time, i.e., seven days in most cases.

3.3.3 FTW apparent uptake velocity (v_f) and i-FTW model evaluation

Performance differences between the i-FTW and control groups indicate treatment contributions of the FTW in each i-FTW experiment. The dynamic TP and TN concentration changes of the i-FTW and control groups were fit to1st order kinetics in order to calculate i-FTW reaction rates ($k_{i\text{-}FTW}$) (Figure 3-2). For the control group without the FTW (A_f = 0), the i-FTW reaction rates ($k_{i\text{-}FTW}$) is equivalent to a water body reaction rates (k_w) of the corresponding study (Equation 3-2). Intercepts of the fitted trend lines were set as initial concentrations (c_0). This step lowered goodness of fit of the trend lines. However, the process standardized the trend lines of the control and i-FTW groups with one initial concentration in the same study and facilitated evaluation of the corresponding FTW apparent uptake velocity (v_f) (Figure 3-2).

To characterize the FTW apparent uptake velocities (v_f) in the i-FTW studies, the water body reaction rate (k_w) in Equation 3-1 is substituted by $k_{w\text{-}C/CM}$ in Equation 3-2 to yield Equation 3-3 and Equation 3-4, where C and CM denote two kinds of the control groups (Section 3.3.1). The C group in six i-FTW studies represents the performance of the applied water body. The $v_{f\text{-}PM}$ in Equation 3-3 is regarded as complete FTW processes, combining treatments from plants, floating mats, and associated ecological communities. In cases of other six studies with the CM control group, the floating mat effect is assigned to the water reaction rate (k_{W-CM}) (Equation 3-4). This is unavoidable due to different control group design (CM instead of C). Such compromises may underestimate the FTW apparent uptake velocity because the v_{f-P} in Equation 3-4 represents the plant effects only. The floating mats remove pollutants through attached microorganisms and should be assigned a separate apparent uptake velocity (v_{f-M}); however, the v_{f-M} could only be calculated through the performance differences between the C and CM groups in the same experiment, which only existed in one of the selected 12 studies (Hu *et al.*, 2010). Equation 3-5 was created for the study conducted by Billore *et al.* (2009). The initial concentrations (c_0) in their experiment were not reported. In this case, the FTW apparent uptake velocity is determined by Equation 3-5, which is formulated by dividing Equation 3-1 with Equation 3-2.

- Evaluation of k_w with control group (C or CM)

$$c_{t,C/CM} = c_0 e^{-(k_w + v_f \frac{A_f}{V})t} = c_0 e^{-(k_{w-C/CM})t} \quad \text{, with } A_f = 0. \qquad \text{Equation 3-2}$$

- Evaluation of v_f with i-FTW group

$$c_{t,i-FTW} = c_0 e^{-k_{i-FTW} t}$$

$$= \begin{cases} c_0 e^{-\left(k_{w-C} + v_{f-PM} \frac{A_f}{V}\right)t} & \text{......studies with C as control.} \qquad \text{Equation 3-3} \\ c_0 e^{-\left(k_{w-CM} + v_{f-P} \frac{A_f}{V}\right)t} & \text{......studies with CM as control.} \qquad \text{Equation 3-4} \end{cases}$$

- Evaluation of v_f with unknown c_0

$$\frac{c_{t,i-FTW}}{c_{t,C}} = e^{-(v_{f-PM} \frac{A_f}{V})t} \qquad \text{Equation 3-5}$$

where

$k_{w-C/CM}$ = a first-order water reaction rate of C or CM group (day^{-1});

$v_{f-PM/P}$ = FTW apparent uptake velocity of plant and floating mat (PM) or plant (P) (m/day);

$c_{t,C/CM/i-FTW}$ = concentration of C, CM, or i-FTW group at time = t (mg/L).

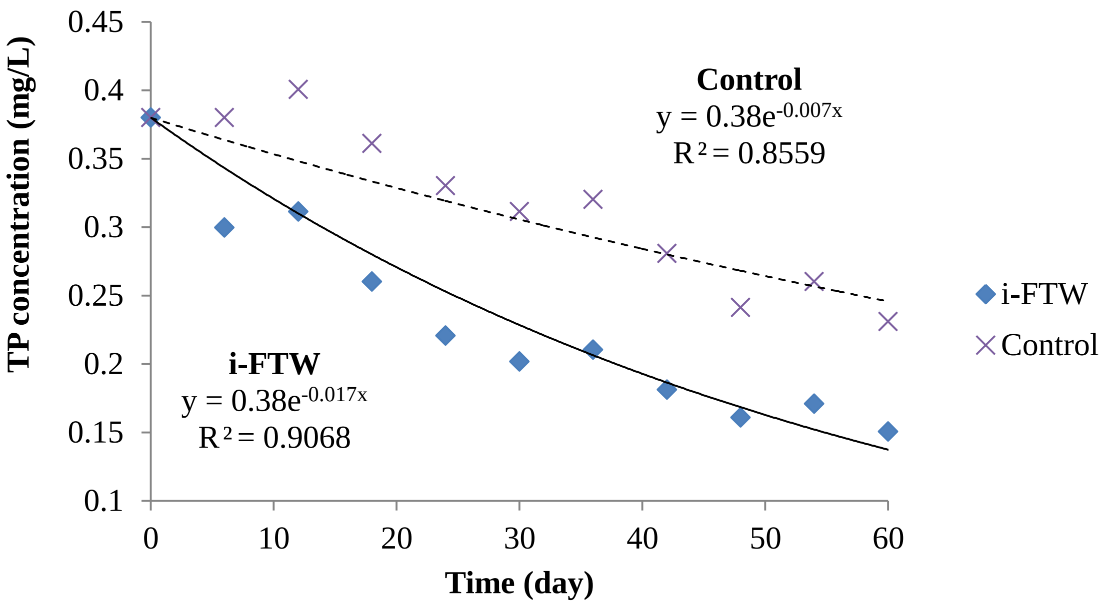

Figure 3-2. The exponential regression lines fit the TP concentration time series data of the i-FTW and control groups (observation data source: Hu et al., 2010).

Changes of TP and TN concentration with time in the i-FTW studies were used to evaluate the adequacy of the i-FTW model. The observation data in C, CM, or i-FTW groups were fit with the 1st order kinetics model to generate trend lines with R^2 values (Figure 3-2). The coefficient of determination (R^2) value indicates the goodness of fit of the trend line. Li *et al.* (2010) and Wang *et al.* (2012) conducted their experiments in continuous flow condition with various hydraulic retention time (HRT). The final concentrations of different HRT treatments were combined and regarded as one time series data. We assumed steady-state plug flow conditions in two continuous flow type experiments. The trend line parameters and coefficient of determination (R^2) was calculated using Microsoft-Excel 2010 software. GetData Graph Digitizer 2.25 software was used to digitize data from figures in the i-FTW studies.

3.3.4 Bootstrap statistics

The bootstrap method estimates the median of the FTW apparent TP/TN uptake velocities calculated from Section 3.3.3. The values were divided into three groups: plant and floating mat ($v_{f\text{-}PM}$), plant only ($v_{f\text{-}P}$), and general FTW ($v_{f\text{-}G}$). The last parameter generalizes the FTW effects by combining both the $v_{f\text{-}PM}$ and $v_{f\text{-}P}$ values as one group. The bootstrap technique estimates the median of the FTW apparent uptake velocity (v_f) with unknown distribution (Chernick, 2008). The 50% and 95% CIs of the median v_f were generated by the percentile interval method (Newman *et al.*, 2000). The available *n* data set were randomly sampled with replacement for *n* times to create a new data set and a corresponding median ($\widehat{M}$). This process was repeated to produce 5000 estimates of median ($\widehat{M}$).

The required number of bootstrap replications varied although most practitioners suggest that the number should be between 1000 and 2000 (Carpenter and Bithell, 2000). We chose 5000 resample times to generate conservative estimates. Chernick (2008) suggested increasing the replication number until the estimate is stabilized. To evaluate the CIs of the median v_f in this study, the 10 median ($\widehat{M}$) calculated from the first 10 bootstrap replications of were ranked, and the values corresponding to 2.5, 25, 50, 75, and 97.5 percentile were regarded as intermediate data of $Q_{2.5}$, Q_{25}, Q_{50}, Q_{75}, and $Q_{97.5}$, respectively. The above process was repeated from the first 10 to 5000 estimates of median ($\widehat{M}$) in increments of 10. The intermediate data ($Q_{2.5}$ – $Q_{97.5}$) were plotted against sample size from 10 to 5000 and became steady as sample size increase. The stabilized intermediate data ($Q_{2.5}$ – $Q_{97.5}$) were used to determine the best estimation (Q_{50}), 50 % CI (Q_{25} and Q_{75}), and 95% CI ($Q_{2.5}$ and $Q_{97.5}$) of the median v_f. The analysis was conducted using Matlab® (Mathworks, 2009) and Microsoft-Excel 2010 software.

3.3.5 Comparing i-FTW pollution control assessment methods

Performance of the i-FTW 1st order kinetics model was compared with three other methods. They are a P-k-C^* model used in an FTW review paper (Headley and Tanner, 2012) and two commonly reported parameters in the i-FTW studies: RE and RR. The P-k-c^* model (Equation 3-6) is adapted from a tank-in series (TIS) 1st order model by relaxing the parameter values in the TIS to become fitting factors (Kadlec, 2003). Model parameters used by Headley and Tanner (2012) were applied in this assessment process, these included: irreducible minimum concentrations (c^*_{TP} = 0.002 and c^*_{TN} = 0.1 mg/L), and the apparent number of tanks in series (P = 3). Although the i-FTW and the P-k-c^* model follows 1st order kinetics, there are several differences between them. The i-FTW model provides temporal solutions of a single tank reactor in batch mode, whereas steady-state solutions of multiple tanks in series with continuous flow conditions for the P-k-c^* model. Therefore, the formulations for these two models are different (Equation 3-1 and Equation 3-6). Additionally, the P-k-c^* model in Equation 3-6 lumps the water body reactions and FTW effects into a single areal reaction rate (k_a), whereas the i-FTW model describes them separately. The water body effects are represented by the reaction rate (k_w) and the FTW effects by the FTW apparent uptake velocity (v_f), FTW size (A_f), and water body volume (V). Therefore, the v_f in the i-FTW model and k_a in the P-k-c^* model are fundamentally different, although their units are the same (m/day). The irreducible concentration (c^*) in Equation 3-6 is a well-adapted parameter in pollution dynamic models (Kadlec and Knight, 1996; Park and Roesner, 2012; Wong *et al.*, 2006). We did not consider this concept in the i-FTW model as limited data could not be used to determine the c^*. The areal reaction rate (k_a) of the P-k-c^* model, the RE, and the RR of each i-FTW experiment in standing water conditions were calculated based on Equation 3-6 to Equation 3-8, respectively, for following comparison processes. The data from the studies with continuous flow group were not used as discussed in Section 3.4.2.

- P-k-c^* model

$$k_a = \left(\sqrt[P]{\frac{c_0 - c^*}{c_{t,i-FTW} - c^*}} - 1 \right) \times Pq \qquad \text{Equation 3-6}$$

- Removal efficiency, RE (%)

$$\text{RE} = \frac{c_0 - c_{t,i-FTW}}{c_0} \times 100 \qquad \text{Equation 3-7}$$

- Removal rate, RR (mg/m^2/day)

$$\text{RR} = \frac{(c_0 - c_{t,i-FTW}) \times V}{A_f \times t} \qquad \text{Equation 3-8}$$

where

c^* = irreducible concentration (mg/L);

k_a = 1st order areal reaction rate of i-FTW system (m/day);

P = apparent number of tanks in series;

q = hydraulic loading rate = average water depth (H) / reaction time (t) (m/day).

A cross validation approach was adapted for the comparison processes. Model parameters (k_w, v_f, k_a, RE, and RR) of one i-FTW experiment (as a “provider”) were selected as input values while the concentration data and experimental design parameters (such as t and A_f) of the other experiments (as “evaluator”) were used to test the performance of the four performance estimation methods. The final concentrations predicted by each method were calculated based on Equation 3-9 to Equation 3-12. For example, the selected input RR value from the i^{th} experiment

(RR_i) was used to predict the final concentration of the j^{th} experiment ($c_t'_{ij}$), while the initial concentration ($c_{0,j}$) and other factors (V_j, $A_{f,j}$, and t_j) was provided from the j^{th} experiment. The *MAX* function in Equation 3-12 is to prevent overestimation of the mass reduction, which results in negative final concentrations. Errors were calculated as the differences between the predicted and observed final concentrations ($c_t'_{ij} - c_{t,j}$). This step was repeated until all the i-FTW groups were selected in turn as the "provider" of the input model parameters (k_w, v_f, k_a, RE, and RR). In one assessment method, the errors ($c_t'_{ij} - c_{t,j}$) were squared and summed to generate an evaluation factor, sum of squared errors (SSE) (Equation 3-13). The assessment model/method with the least SSE value is the most accurate approach to predict concentration changes in the i-FTW experiments.

- Prediction by i-FTW model

$$c'_{t,ij} = c_{0,j} e^{-(k_{w-C/CM,i} + v_{f-PM/P,i}\frac{\gamma_j}{H_j})t_j} \quad \text{Equation 3-9}$$

- Prediction by P-k-c^* model

$$c'_{t,ij} = \frac{c_{0,j} - c^*}{\left(1 + k_{a,i}/Pq_j\right)^P} + c^* \quad \text{Equation 3-10}$$

- Prediction by removal efficiency method

$$c'_{t,ij} = c_{0,j}(1 - \frac{\mathrm{RE}_i}{100}) \quad \text{Equation 3-11}$$

- Prediction by removal rate method

$$c'_{t,ij} = MAX\left[\frac{(c_{0,j} \times V_j) - (\mathrm{RR}_i \times A_{f,j} \times t_j)}{V_j}, 0\right] \quad \text{Equation 3-12}$$

- Discrepancy between observed and predicted data

  SSE (Sum of squared errors of prediction) = $\sum_{i=1}^{n}\sum_{j=1}^{n}(c'_{t,ij} - c_{t,j})^2$ Equation 3-13

where

c_t' = predicted concentration at time = t (mg/L);

i = the provider of reaction factors (k_w, v_f, k_a, RE, and RR), 1 to n;

j = the evaluator (providing concentration data and experimental design parameters), 1 to n;

n = number of the i-FTW experiment with standing water condition. n=14 in TP (Table 3-3) and n=18 in TN (Table 3-4).

3.4 Result and Discussion

3.4.1 FTW apparent uptake velocity (v_f)

Table 3-3 and Table 3-4 summarize the FTW apparent uptake velocities (v_f) of the 12 selected i-FTW studies. The parameters of the i-FTW and other models were calculated based on Equation 3-2 to Equation 3-8.The v_f values are divided into two groups: standing water or continuous flow. The FTW TP apparent uptake velocities in standing water design were 0.04 ± 0.03 m/day for $v_{f\text{-P}}$ and 0.06 ± 0.04 m/day for $v_{f\text{-PM}}$, respectively (mean ± SD). This result is similar to the values of 27 gravity-fed stormwater wetlands (0.03 ± 0.05 m/day) and 20 emergent marshes (0.03 ± 0.02 m/day) with zero irreducible minimum concentration ($c^* = 0$) (Carleton *et al.*, 2001; Kadlec and Knight, 1996). The FTW TN apparent uptake velocities were 0.01± 0.01 m/day for $v_{f\text{-P}}$ and 0.06 ± 0.02 m/day for $v_{f\text{-PM}}$, which are nearly equivalent to those for 30 surface flow marsh wetlands (0.04 ± 0.05) with c^* = 1.5 mg/L (Kadlec and Knight, 1996). Another study evaluated i-FTW performance through another 1st order kinetics approach, P-k-c^* model (Section 3.3.5). Headley and Tanner (2012) evaluated 1st order areal rates (k_a) of "FTWs" (which meet the definition of i-FTW in our study) to be 0.11 ± 0.15 m/day for TP and 0.18 ± 0.21 m/day for TN with c^* = 0.002 and 1 mg/L, respectively. Although the theory (1st order reaction rate) and units (m/day) for the v_f and k_a is the same, fundamental differences exist between the i-FTW and P-k-c^* model as described in Section 3.3.5. Therefore, these two reaction rates (v_f and k_a) are not comparable despite their similarity.

The FTW TP and TN apparent uptake velocities (v_f) were much higher under the continuous flow condition than the standing water one (Tables 3-3 and 3-4). It may in part reflect a fact that the pollutants were actively supplied to the FTWs in the continuous flow experiments. The current effect is recognized as a major mechanism which supplies nutrients to and removes respiration waste from periphyton in another treatment system, algal turf scrubbers (Adey and Loveland, 1998). The plant roots growing under the floating mats and the associated biofilm may receive similar benefits in the continuous flow condition. However, other mechanisms, such as biofilm carriers or aeration pumps, were also used in these studies, except one conducted by van Oostrom (1995) (Tables 3-3 and 3-4). Therefore, it is premature to identify the current effect or the other processes as the primary contributor of the FTW apparent uptake velocity (v_f) improvement in the continuous flow studies.

Table 3-3. The model parameters of the i-FTW model (k_w and v_f), the P-k-c^* model (k_a), removal efficiency (RE), and removal rate (RR) for TP removal from i-FTW studies.

ID	Season$^+$	c_0^* (mg/L)	$c_{t\text{-}i\text{-}FTW}^*$ (mg/L)	t^* (d)	$k_{i\text{-}FTW}$ (1/d)	k_w (1/d)		v_f (m/d)		k_a (m/d)	RE (%)	RR (mg/m^2/d)
						CM$^\delta$	C$^\delta$	P$^\delta$	PM$^\delta$			
Flow type: standing water												
10-H	2, 3	0.38	0.15	60	0.02	-	0.01	-	0.11	0.02	60	42
10-V	2	2.16	1.50	11	0.04	-	0.00	-	0.06	0.03	31	84
10-V	3	2.16	1.53	11	0.04	-	0.01	-	0.04	0.03	29	80
10-V	4	2.16	1.75	11	0.02	-	0.01	-	0.02	0.02	19	52
10-X-d	1	2.29	0.22	35	0.07	0.04	-	0.01	-	0.03	90	18
10-X-w	1	1.49	0.13	35	0.07	0.04	-	0.01	-	0.03	91	12
10-X-t	1	1.75	0.21	35	0.06	0.04	-	0.01	-	0.03	88	13
10-Z	1	0.68	0.16	35	0.04	0.01	-	0.01	-	0.01	76	3
12-Za-z	1	2.28	0.34	16	0.11	0.02	-	0.05	-	0.10	85	73
12-Za-tl	1	2.28	0.37	16	0.10	0.02	-	0.05	-	0.09	84	72
12-Za-td	1	2.28	0.27	16	0.12	0.02	-	0.06	-	0.12	88	75
12-Za-v	1	2.28	0.22	16	0.13	0.02	-	0.07	-	0.13	90	77
12-Za-m	1	2.28	0.21	16	0.14	0.02	-	0.07	-	0.14	91	78
12-Za-a	1	2.28	0.47	16	0.09	0.02	-	0.04	-	0.08	79	68

Table 3-3. (continued)

ID	Season[+]	c_0* (mg/L)	$c_{t\text{-}i\text{-}FTW}$* (mg/L)	t* (d)	$k_{i\text{-}FTW}$ (1/d)	k_w (1/d)		v_f (m/d)		k_a (m/d)	RE (%)	RR (mg/m²/d)
						CM[δ]	C[δ]	P[δ]	PM[δ]			
Flow type: continuous flow												
10-L[α]	2, 3	0.97	0.11	7	0.32	-	0.25	-	0.84	0.80	89	1509
10-L[β]	2, 3	0.97	0.09	7	0.34	-	0.25	-	1.10	0.88	91	1536
10-L[β]	1, 2	0.97	0.36	7	0.16	-	0.05	-	1.34	0.29	63	1067
12-W[γ]	3	2.20	0.36	2.5	0.70	0.54	-	0.16	-	1.00	84	736

[+] 1: spring, 2: summer, 3: autumn, 4: winter.
[*] Concentration at time = 0 (initial) or time = t in the i-FTW group. (t: reaction time).
[δ] C: control with water, CM: control with floating mat, P: plant, PM: plant with floating mat.
[α] FTWs were integrated with biofilm carriers.
[β] FTWs were integrated with biofilm carriers and fresh water clams.
[γ] System were aerated by air pumps.

Table 3-4. The model parameters of the i-FTW model (k_w and v_f), the P-k-c^* model (k_a), removal efficiency (RE), and removal rate (RR) for TN removal from i-FTW studies.

ID	**Season**[+]	**c_0^* (mg/L)**	**$c_{t\text{-}i\text{-}FTW}^*$ (mg/L)**	**t^* (d)**	**$k_{i\text{-}FTW}$ (1/d)**	**k_w (1/d)**		**v_f (m/d)**		**k_a (m/d)**	**RE (%)**	**RR (mg/m²/d)**
						CM$^\delta$	**C$^\delta$**	**P$^\delta$**	**PM$^\delta$**			
Flow type: standing water												
09-B	4	-	33.91	60	-	-	-	-	0.04	-	-	-
09-B	1	-	33.70	60	-	-	-	-	0.04	-	-	-
09-B	1, 2	-	31.63	60	-	-	-	-	0.04	-	-	-
09-S	-	8.71	4.32	6	0.16	-	0.05	-	0.05	0.07	50	398
10-H	2, 3	4.55	2.57	60	0.01	-	0.00	-	0.11	0.01	44	362
10-V	1	21.80	11.30	11	0.07	-	0.02	-	0.07	0.06	48	1339
10-V	2	21.80	11.50	11	0.07	-	0.00	-	0.09	0.06	47	1313
10-V	3	21.80	13.60	11	0.05	-	0.01	-	0.06	0.04	38	1046
10-V	4	21.80	11.30	11	0.07	-	0.03	-	0.05	0.06	48	1339
10-X-d	1	18.00	2.88	35	0.05	0.03	-	0.01	-	0.02	84	130
10-X-w	1	13.50	2.76	35	0.05	0.03	-	0.00	-	0.02	80	92
10-X-t	1	19.50	3.84	35	0.05	0.03	-	0.00	-	0.02	80	134
10-Z	1	12.58	1.16	35	0.08	0.03	-	0.01	-	0.01	91	64
12-Za-z	1	8.12	4.14	16	0.04	0.01	-	0.02	-	0.03	49	149
12-Za-tl	1	8.12	4.52	16	0.04	0.01	-	0.01	-	0.02	44	135
12-Za-td	1	8.12	4.14	16	0.04	0.01	-	0.02	-	0.03	49	149
12-Za-v	1	8.12	3.67	16	0.05	0.01	-	0.02	-	0.03	55	167
12-Za-m	1	8.12	3.26	16	0.06	0.01	-	0.03	-	0.04	60	182
12-Za-a	1	8.12	4.30	16	0.04	0.01	-	0.02	-	0.03	47	143
12-Zo	1, 2	15.37	0.82	48	0.08	0.04	-	0.01	-	0.01	95	60
12-Zo	4	14.84	1.12	48	0.05	0.02	-	0.00	-	0.01	92	56

Table 3-4. (continued)

ID	Season[+]	c_0^* (mg/L)	$c_{t\text{-}i\text{-}FTW}^*$ (mg/L)	t^* (d)	$k_{i\text{-FTW}}$ (1/d)	k_w (1/d)		v_f (m/d)		k_a (m/d)	RE (%)	RR (mg/m²/d)
						CM[δ]	C[δ]	P[δ]	PM[δ]			
Flow type: continuous flow												
95-V	1, 2, 3, 4	197.00	101.00	7	0.10	0.04	-	0.02	-	0.04	49	5486
10-L[α]	2, 3	5.15	1.91	7	0.14	-	0.08	-	0.72	0.30	63	5661
10-L[β]	2, 3	5.15	1.52	7	0.17	-	0.08	-	1.12	0.38	70	6345
10-L[β]	1, 2	5.15	1.96	7	0.14	-	0.03	-	1.28	0.29	62	5578
11-S[γ]	2, 3	5.01	4.08	7	0.03	-	0.01	-	0.18	0.05	19	1113
11-S[α]	2, 3	5.01	3.83	7	0.04	-	0.01	-	0.26	0.06	24	1412
11-S[β]	2, 3	5.01	3.46	7	0.05	-	0.01	-	0.38	0.09	31	1855

[+] 1: spring, 2: summer, 3: autumn, 4: winter.
[*] Concentration at time = 0 or time = t in the i-FTW group. (t: reaction time).
[δ] C: control with water, CM: control with floating mat, P: plant, PM: plant with floating mat.
[α] FTW integrated with biofilm carrier.
[β] FTW integrated with biofilm carrier and fresh water clams.
[γ] FTW integrated with fresh water clams.

The variation of the FTW apparent uptake velocity (v_f) within each flow condition group is expected as the different experimental designs and environmental conditions of each i-FTW study. Although $v_{f\text{-}PM}$ combines plants and floating mats effects and should be higher than $v_{f\text{-}P}$, this relationship only existed in TN but not TP data (Tables 3-3 and 3-4). The phenomenon may be due to variation in the limited i-FTW studies. The plants (P) may remove more pollutants in the experiments with better conditions than the plants with floating mats (PM) in the other studies with less suitable circumstances, such as autumn and winter seasons (Table 3-3, ID: 10-V). Temperature is strongly related to seasonal changes and is a well-recognized driving force of biological reactions (Chapra, 1997). The temperature effects have been incorporated into several wetland and pond models (Kadlec and Knight, 1996; Shilton, 2005). However, this factor could not be incorporated into our i-FTW model because of lack of consistency in reporting temperature data by the authors of the 12 selected i-FTW studies. Mean water or room temperatures are documented in a few studies but not all. Although air temperature statistics could be collected from weather stations (Table 3-1), using air temperature to evaluate the experimental water temperature may involve considerable uncertainty.

The FTW apparent uptake velocities (v_f) were potentially affected by the plant life cycle. As shown in Tables 3-3 and 3-4, only 4 of the 46 i-FTW groups were conducted in winter while the others were mostly in spring. Two i-FTW groups with negative TP performance were not listed in Table 3-3. Their TP apparent uptake velocities were − 0.06 and − 0.03 m/day in winter and spring, respectively (Van de Moortel *et al.*, 2010; Wang *et al.*, 2012). Therefore, the FTW apparent uptake velocities (v_f) listed in Tables 3-3 and 3-4 should be considered representative of FTW performance for the growing seasons.

3.4.2 Bootstrap analysis of apparent uptake velocities (v_f)

The results of the bootstrap resampling analysis of the median FTW TP and TN apparent uptake velocities (v_f) are listed in Table 3-5. The 50 percentile values are the best estimation of the median v_f, whereas 25 and 75 percentile indicate the 50% CI and 2.5 and 97.5 percentile for 95% CI. The v_f values in Tables 3-3 and 3-4 were divided into three groups: plants ($v_{f\text{-}P}$), plants with floating mats ($v_{f\text{-}PM}$), and G (general). The G group combines $v_{f\text{-}P}$ and $v_{f\text{-}PM}$ data and considers that the FTW performance was equally represented by both P and PM groups. Figure 3-3 visualizes the variability of the median FTW TP and TN apparent uptake velocities (v_f) in the

G group. While the 50% CI of the v_f –TP is much smaller than it of the v_f –TN, the opposite condition occurred in the case of 95% CI. The values of the v_f –TP in Table 3-3 are more centralized and resulted in a narrower span of 50% CI comparing with v_f –TN. However, the CI range of the v_f –TP is greatly expanded to cover 95% possibility because of the four extreme values (0.01 m/day, ID: 10-X and 10-Z). There is a limitation that these preliminary results were summarized from the available studies and could not represent a complete population of FTWs.

Table 3-5. Bootstrap analysis of the best estimation of the median FTW TP and TN apparent uptake velocities (v_f, m/day) ranked at 2.5, 25, 50, 75, 97.5 percentile.

Constituent	**Group***	**Percentile**				
		2.5	**25**	**50**	**75**	**97.5**
TP	P (10)	0.008	0.025	0.045	0.052	0.060
	PM (4)	0.024	0.038	0.048	0.065	0.106
	G (14)	0.017	0.040	0.045	0.052	0.058
TN	P (12)	0.006	0.009	0.012	0.015	0.018
	PM (9)	0.043	0.053	0.055	0.062	0.091
	G (21)	0.014	0.017	0.022	0.038	0.043

* PM: plant with floating mat, P: plant, G: general FTW (PM and P data combined). Sample number of each group is in the parentheses.

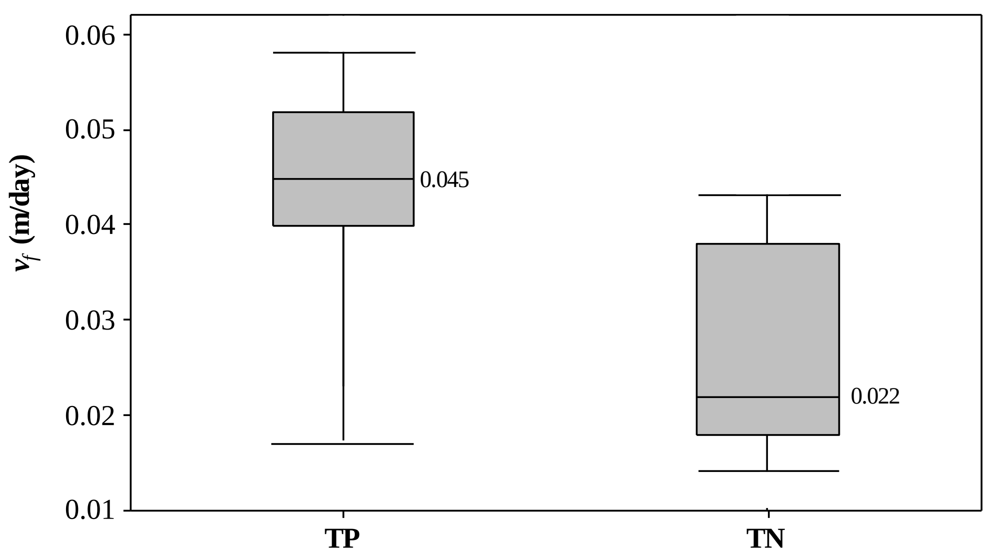

Figure 3-3. The uncertainty analysis of the median FTW TP and TN apparent uptake velocities (v_f) (G group in Table 3-5). The numbers next to the boxes are the best estimations. The 50% and 95% confidence intervals are indicated by the boxes and the whiskers, respectively.

The FTW TP and TN apparent uptake velocities (v_f) in the i-FTW studies with standing water conditions were utilized in the bootstrap analysis. The v_f values in the continuous flow group are very different from those in the standing water design as discussed in Section 3.4.1, and they are fewer than those in the standing water conditions. Therefore, the bootstrap analysis was not conducted for the v_f values in the continuous flow group. An example of determining the best estimators of median $v_{f,PM}$ –TP with 50% and 95% CIs was shown in Figure 3-4. The bootstrap resampling number increased from 10 to 5,000 in increments of 10; however, the curves were already stabilized when the resampling number exceeded 200. Thus, the 2.5, 25, 50, 75, and 97.5 percentile of the median $v_{f,PM}$ –TP were essential determined by 200 resampling times. The same procedure was repeated to evaluate other FTW apparent uptake velocities (Table 3-5). Figure 3-4 only represents a small range between 10 to 1,000 resample times to demonstrate the fluctuations. The required number of resample times varies based on the data set. Thus, visual checks as shown in Figure 3-4 are useful.

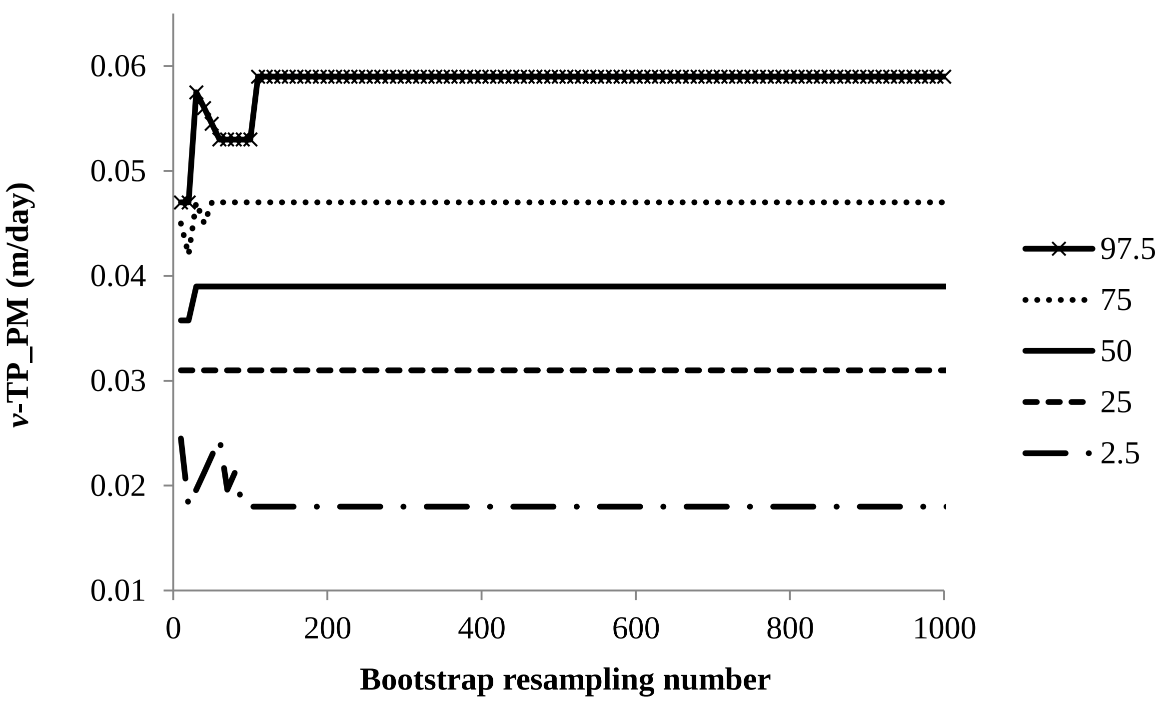

Figure 3-4. The bootstrap resampling number increased from 10 to 5000 in increments of 10. The curves were already stabilized when the resampling number exceeded 200. Thus, the 2.5, 25, 50, 75, and 97.5 percentile of the median $v_{f,PM}$ –TP were essential determined by 200 resampling times.

3.4.3 Adequacy and applicability of the 1st order kinetics model with uncertainty

Table 3-6 lists the results obtained by fitting a 1st order trend line to the concentration time series data of each i-FTW experiment. Ten studies with 71 TP and TN concentration time series data were selected to assess the goodness of fit (R^2) of the i-FTW 1st order kinetics model. The 71 time series data were further divided into 47, 12, and 12 sub-data set of i-FTW, C, and CM groups, respectively. The R^2 values of the total 71 TP and TN concentration data are 0.82 ± 0.38 and 0.78 ± 0.42, respectively (Table 3-6). The goodness of fit of the 47 i-FTW experiments is better with R^2 values 0.92 ± 0.30 for TP and 0.86 ± 0.38 for TN data. These i-FTW experiments were tested with small containers in which the contents could be regarded as well mixed. Such a system with uniformly distributed condition is considered as a single continuously stirred tank reactor (CSTR) in discrete batches (Chapra, 1997), which satisfies the assumptions of the i-FTW model. Additionally, the concentration changes in some studies were the average of replications. It may reduce fluctuations and result in better fitting of the 1st order kinetics curves. Although these results appear to validate the i-FTW 1st order kinetics model, most of the tested data were published from a single region, China. For a more complete evaluation, additional studies from other regions are needed to test the applicability of the i-FTW model.

The data from an i-FTW experiment conducted by Zhao *et al.* (2012b) were utilized to demonstrate the ability of the i-FTW model with uncertainty. The TN concentration time series data of the i-FTW planted with *Miscanthus sinensis* Anderss (sp.) were fit with a 1st order kinetics curve. The trend line and 80% of the observation data fell in a range predicted by the i-FTW model with 95% uncertainty (Figure 3-5). The parameters used in the two predicted boundary lines were the 2.5 and 97.5 percentile of the median v_f –TN in the G group: 0.014 and 0.043 m/day, respectively (Table 3-5). The other i-FTW model parameters: $k_{w\text{-}CM}$ (0.01 day^{-1}) in the control group, initial TN concentration (c_0, 8.12 mg/L), system volume (V, 0.12 m^3), FTW size (A_f, 0. 2 m^2), and reaction time (t, 16 day) were collected from the integrated water body, the experimental tanks in the same study. The predicted TN concentrations on Day 16 ranged from 4.55 to 2.10 mg/L, which included the observations: 3.26 mg/L.

Table 3-6. Goodness of fit of the i-FTW 1st order kinetics model for the TP & TN concentration time series data in the selected studies.

ID	Group*	Season$^{\delta}$	Number of observations	R^2-TP	R^2-TN
09-S	i-FTW	-	19	-	0.77
10-H	i-FTW	2, 3	11	0.91	0.54
10-L	i-FTW	1, 2	4	0.93	0.93
10-V	i-FTW	1	5	0.86	0.61
10-V	i-FTW	2	5	0.84	0.79
10-V	i-FTW	3	5	0.67	0.64
10-V	i-FTW	4	5	0.86	0.79
10-Z	i-FTW	1	6	0.93	0.87
11-L-t	i-FTW	3, 4, 1	5	0.93	0.91
11-L-r	i-FTW	3, 4, 1	5	0.92	0.89
12-L-o	i-FTW	3	6	0.96	0.93
12-L-c	i-FTW	3	6	0.95	0.99
12-L-t	i-FTW	3	6	0.97	0.99
12-L-l	i-FTW	3	6	0.78	0.99
12-W	i-FTW-o$^{\theta}$	3	4	0.99	-
12-W	i-FTW-o$^{\theta}$	3, 4	8	0.88	-
12-W	i-FTW-p$^{\theta}$	3	4	0.97	-
12-W	i-FTW-p$^{\theta}$	3, 4	8	0.96	-
12-W	i-FTW-q$^{\theta}$	3	4	0.98	-
12-W	i-FTW-q$^{\theta}$	3, 4	8	0.96	-
12-Za-z	i-FTW	1	5	0.97	0.99
12-Za-tl	i-FTW	1	5	0.95	0.96
12-Za-td	i-FTW	1	5	0.94	0.96
12-Za-v	i-FTW	1	5	0.93	0.95
12-Za-m	i-FTW	1	5	0.97	0.97
12-Za-a	i-FTW	1	5	0.95	0.97
12-Zo	i-FTW	1, 2	8	-	0.64
12-Zo	i-FTW	4	8	-	0.88

Table 3-6. (continued)

ID	Group*	Season$^{\delta}$	Number of observations	R^2-TP	R^2-TN
10-H	C	2, 3	11	0.86	0.73
10-L	C	1, 2	4	0.40	0.29
10-V	C	1	5	0.69	0.75
10-V	C	2	5	-0.54	-0.33
10-V	C	3	5	0.38	-0.33
10-V	C	4	5	0.48	0.92
10-H	CM	2, 3	11	0.80	0.80
11-L	CM	3, 4, 1	5	0.94	0.96
12-L	CM	3	6	0.95	0.98
12-W	CM	3	4	0.97	-
12-W	CM	3, 4	8	0.86	-
12-Za	CM	1	5	0.81	0.90
12-Zo	CM	1, 2	8	-	0.95
12-Zo	CM	4	8	-	0.97

* i-FTW: floating treatment wetland, C: control with water, CM: control with floating mat.
$^{\delta}$ 1: spring, 2: summer, 3: autumn, 4: winter.
$^{\theta}$ floating mats with different materials. o: ceramic pellets, p: PE foam, q: fibrous fillers.

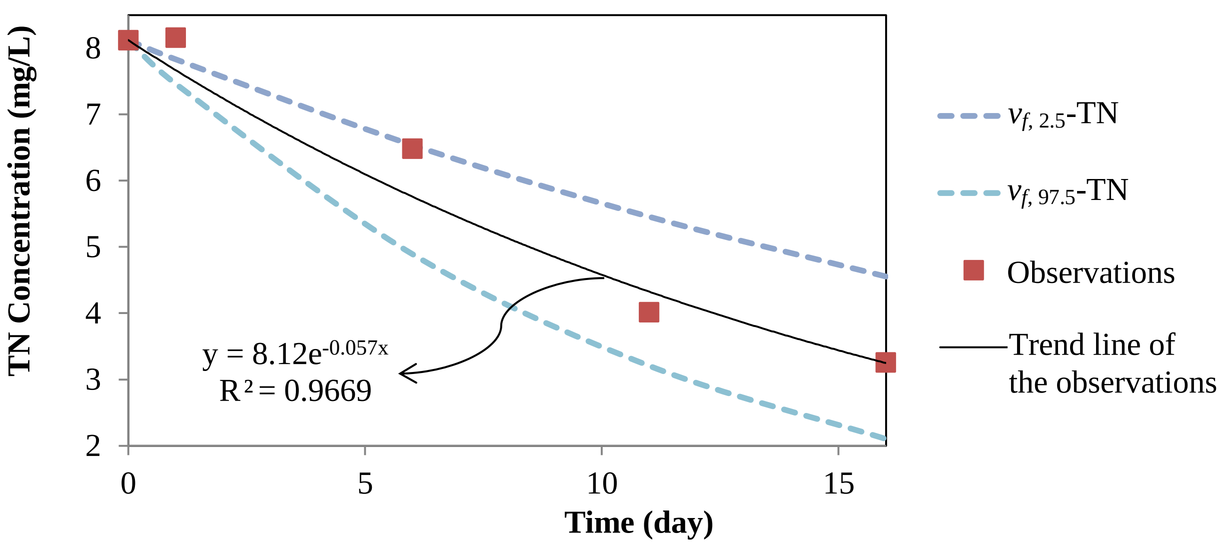

Figure 3-5. Demonstration of the i-FTW model with uncertainty. The observations and the corresponding regression line fell within an area predicted by the i-FTW model with 95% uncertainty. The two boundary lines were calculated with the same water reaction rate (k_w), FTW size (A_f), water volume (V), reaction time (t) of the study (ID: 12-Za-m) but different FTW apparent uptake velocities (2.5 and 97.5 percentile of the v_f–TN, G group in Table 3-5).

3.4.4 Comparison of the four assessment models/methods

Treatment assessment methods are essential tools to design i-FTW systems. Figure 3-6 indicates the result of the validation test of the i-FTW model, the *P-k-c** model, the removal efficiency (RE), and the removal rate (RR) methods. The model parameters used in this process are provided in Table 3-3 and Table 3-4. The result shows that the i-FTW model with the least SSE TP and TN values provided the most accurate concentration predictions among the four assessment methods. The SSE value quantifies the accumulated errors between observed and model predicted concentrations. The model with the best performance (least SSE value) in TP constituent is the i-FTW model and followed by the *P-k-c** model, the RR method, and the RE method. For the TN constituent, the rank is the i-FTW model, the *P-k-c** model, the RE method, and the RR method. Both of the 1st order kinetics models, the i-FTW and the *P-k-c** model, improved the accuracy of concentration predictions considerably compared with the RE and the RR methods. In the validation process of the RR method, 27% of TP and 37% of TN final concentration predictions were overestimated and forced to be zero. The RR method does not automatically incorporate a concept of irreducible concentrations unless specified as we imposed the rule that the minimum concentration is zero.

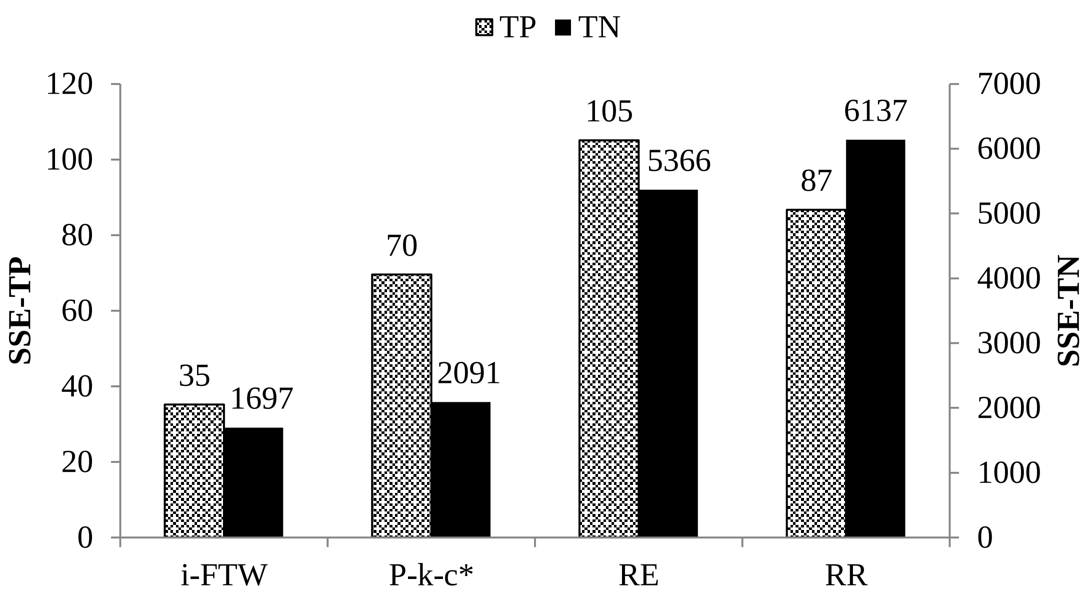

Figure 3-6. Cross validation test of the four assessment methods: the i-FTW model, the *P-k-c** model, the removal efficiency (RE), and the removal rate (RR) methods. SSE: Sum of square error.

The better accuracy of the i-FTW model than the *P-k-c** model may indicate the merit of the proposed compartmental concept which separate the water body and FTW reactions. The factors of FTW size and the system dimension are also considered in the i-FTW model. The nature of the i-FTW experiments used in the validation test may also explain the different SSE values of the two models. Most of the i-FTW experiments were conducted in small volume tanks (< 2 m^3) in batch mode, which better fits the assumptions of the i-FTW model than the *P-k-c** model (Section 3.3.5). Although the four methods were not tested with data from field scale studies, the cross validation results provided in Figure 3-6 compares the four models, and reveals the model with the best performance, the i-FTW 1st order kinetics model.

As a design tool, the i-FTW 1st order kinetics model possesses great advantages compared to other three methods. FTWs apply in systems with various physical attributes, including volumes, depth, and reaction time. These data are basic information when designing a pollution control system. The i-FTW model takes these variables into consideration. The pollutant concentrations can also be estimated under different FTW coverage ratios. However, influences of FTWs on water reaction rate (k_w) of the integrated water bodies were not considered because of limited data. The floating mats create shading and may alter treatment processes of the water body (Cao *et al.*, 2011).

3.5 Conclusion

The i-FTW 1st order kinetics model incorporates compartmental concepts to separate the FTWs' contribution from the effects caused by the integrated water bodies. Therefore, the i-FTW model can be adapted to various treatment systems by changing the model parameters, such as water reaction rates (k_w), system water volumes (V), and reaction time (t). The k_w values are determined from literature reported data or calculated from water concentration changes of the applied systems. The FTW performance is described by another parameter: FTW apparent uptake rates (v_f). The uncertainty of the v_f values was addressed by a bootstrap technique to evaluate the expected range with 95% certainty. Based on the i-FTW model, the ranges of the median v_f values with FTW coverage (A_f) and other factors predict concentrations of the treated water to satisfy the specific TP and TN reduction goals of the designed i-FTW systems. Therefore, the i-FTW model can serve as a preliminary tool to assist i-FTW system designs.

However, it should be understood that there are limitations of the i-FTW model: 1) the FTW apparent uptake velocities (v_f) were evaluated based on the limited data mainly from a small region and during growing seasons; 2) the lack of the water temperature data in the 12 i-FTW studies adds a significant amount of uncertainty; 3) the influences of integrating FTWs into original systems were not considered; 4) the i-FTW model has only been tested by the concentration time series data from the i-FTW studies. Therefore, it is recommended that additional research directions include: 1) conducting i-FTW experiments with control (C group) in various regions to identify the temperature effects on the FTW apparent uptake velocity (v_f); and 2) collecting data from full scale i-FTW treatment facilities to validate the i-FTW model or other developed models incorporated with the v_f values.

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Chapter 4. Assessment of the nutrient removal effectiveness of floating treatment wetlands in urban retention ponds

4.1 Abstract

The application of floating treatment wetlands (FTWs) in point and non-point source pollution control has received much attention recently. Although the potential of this emerging technology is supported by various studies, quantifying FTW performance in urban retention ponds remains elusive due to significant research gaps. Actual urban retention pond water was utilized in this mesocosm study to evaluate phosphorus and nitrogen removal efficiency of FTW. Multiple treatments were used to investigate the contribution of each component in the FTW system with a seven-day retention time. The four treatments included a control, floating mat, pickerelweed (*Pontederia cordata* L.), and softstem bulrush (*Schoenoplectus tabernaemontani*). The water samples collected on Day 0 (initial) and 7 were analyzed for total phosphorus (TP), total particulate phosphorus, orthophosphate phosphorus, total nitrogen (TN), organic nitrogen, ammonia nitrogen, nitrate-nitrite nitrogen, and chlorophyll-a. Statistical tests were used to evaluate the differences between the four treatments. The effects of temperature on TP and TN removal rates of the FTWs were described by the modified Arrhenius equation. Our results indicated that all three FTW designs, planted and unplanted floating mats, could significantly improve phosphorus and nitrogen removal efficiency (%, E-TP and E-TN) compared to the control treatment during the growing season, i.e., May through August. The E-TP and E-TN was enhanced by 8.2% and 18.2% in the FTW treatments planted with the pickerelweed and softstem bulrush, respectively. Organic matter decomposition was likely the primary contributor of nutrient removal by FTWs in urban retention ponds. Such mechanism is fostered by microbes within the attached biofilms on the floating mats and plant root surfaces. Among the results of the four treatments, the FTWs planted with pickerelweed had the highest E-TP, and behaved similarly with the other two FTW treatments for nitrogen removal during the growth period. The temperature effects described by the modified Arrhenius equation revealed that pickerelweed is sensitive to temperature and provides considerable phosphorus removal when water temperature is greater than 25℃. However, the nutrient removal effectiveness of this plant species may be negligible for water temperatures below 15℃. Due to the variation of nutrient distribution between above- and below-ground plant tissues at different stages, aerial part is recommended to

be harvested in June, when most nutrients are located in the shoots and in September before nutrient leached from the senesced tissues. The study also assessed potential effects of shading from the FTW mats on water temperature, DO, pH, and attached-to-substrate periphyton.

Keywords

Non-point source pollution; Nutrient control; Temperature effects; Stormwater, Pickerelweed (*Pontederia cordata* L.), Softstem bulrush (*Schoenoplectus tabernaemontani*).

4.2 Introduction

Sustainable, effective, and economical solutions that address water quality degradation problems are being actively investigated. Urban non-point source pollution has been identified as a major source of water quality impairments, including excess nutrients, organics, sediment, and metals carried by runoff during storm events (Field, 2006). To properly manage the anthropogenic impacts on natural water bodies, a set of technologies known as best management practices (BMPs) have been developed to treat urban runoff. Constructed wetlands are one of the most commonly used BMPs to improve stormwater quality (Carleton *et al.*, 2001; Wynn and Liehr, 2001). However, land acquisition costs limit the broad application of this BMP (Nduvamana *et al.*, 2007). A relatively new and evolving treatment practice may represent a significant opportunity to retrofit existed stormwater facilities by combining the functions of constructed wetlands and conventional retention (or “wet”) ponds. This hybrid system is known as a floating treatment wetland, or FTW (Headley and Tanner, 2012). FTWs use floating mats that sustain and support terrestrial macrophytes and have a wide range of applicability in water bodies, including retention ponds (Headley and Tanner, 2006; Hubbard, 2010; Zhao *et al.*, 2012a).

The combination of FTWs with retention ponds may provide a means to effectively and economically manage urban stormwater quantity and quality. Most urban retention ponds provide flood control, a quantity benefit; whereas, water quality is improved through sedimentation (Shilton, 2005). Conventional methods for improving the pollution control of retention ponds require continuous chemical or energy inputs, such as flocculants and aeration systems. As a potential supplemental treatment practice, FTWs used in urban retention ponds and their associated pollution control mechanisms have been discussed in previous studies

(Headley and Tanner, 2012; Van de Moortel *et al.*, 2011; Wang and Sample, 2011). Nutrients and other constituents are absorbed by macrophytes and microorganisms, which grow on the submerged surface of the floating mats and plant roots (Li *et al.*, 2010; Song *et al.*, 2011). Exportation of pollutants from the ponds could be reduced as the constituents are stored in plant tissues and attached microorganisms on the floating mats rather than algae suspended in the water body. A significant portion of biochemical oxygen demand (BOD) and suspended solids in pond effluent is attributed to algae, which is typically flushed downstream during storm events (Shilton, 2005). The surface area provided by the FTWs may address a key limitation of nitrifiers in retention ponds, i.e., the lack of available surface area in aerobic environments, such as the littoral zone of ponds (Zimmo *et al.*, 2004).

Another potential advantage of FTW application is cost. The cost of installing FTWs on conventional ponds could be relatively lower than the cost of land acquisition and construction of new BMPs. In many cases, modification of the ponds is unnecessary unless sedimentation removal is desired. The total cost of an FTW depends mainly upon floating mats, plants, and labor for harvesting and planting. Billore *et al.* (2009) reported that the manufacturing and installation cost of a floating mat was 60 USD/m^2. Costs can be further reduced if recycled materials, such as plastic bottles are used to construct the floating mats (Chen, 2011; Pelton, 2010). Since retention ponds are one of the most widely used BMPs in the U.S., as reported by the U.S. Environmental Protection Agency, or US EPA (1999), it is possible that a watershed-wide pollutant mass reduction could be economically achieved through FTW application in the existing retention ponds. Potential areas in need of this technology may be the contributing urban watersheds of the Chesapeake Bay, an estuary of national importance. The Chesapeake Bay has experienced significant water quality issues; the severest is a recurring zone of hypoxia which has been attributed to excess sediment and nutrients discharged upstream. Due to the lack of sufficient progress in reducing nutrient and sediment loads to the Chesapeake Bay, the US EPA recently published a Total Maximum Daily Load for the estuary and watershed, requiring significant reductions in nutrient and sediment loads (US EPA, 2010). Treatment technologies such as FTWs, if found to be effective, could assist in meeting these reductions, as they are uniquely suited to removing nutrients from current base loads.

FTWs have been applied to domestic and agricultural wastewater, swine lagoons, and hyper-eutrophic lake waters (Hubbard *et al.*, 2011; Van de Moortel *et al.*, 2010; Wu *et al.*, 2006a;

Yang *et al.*, 2008). Only a few peer-reviewed studies are available that evaluate the performance of an FTW in an urban stormwater application. We contend that FTW applications in urban retention ponds should address the following concerns. First, stormwater is relatively dilute in comparison with most FTW applications in nutrient-rich waters. The total phosphorus (TP) and total nitrogen (TN) concentrations of runoff from mixed urban land uses typically are 0.26 mg/L and 1.8 mg/L, respectively (US EPA, 1999). In comparison, TP and TN in domestic wastewater after secondary treatment are typically 2 mg/L and 30 mg/L, respectively (US EPA, 1999). Second, actual urban stormwater has not been widely tested except for three recently published FTW studies. These include two mesocosm experiments and one in-situ test. Winston *et al.* (2013) monitored inflow and outflow water concentration of two urban retention ponds during storm events. Pre- and post-FTW installation monitoring periods for the same pond were compared and indicated that significant water quality improvement was achieved in the retention pond with 18% coverage. The in-situ experiment reflected the actual behavior of the FTW; however, this approach is necessarily limited by the comparability of the two data sets, which reflect different times. While the inflow concentrations of the pre- and post-FTW installation were found to be statistically similar, the frequency of the loads, the retention time between storm events, and climatic conditions are additional factors that will affect performance, and were not considered in their analysis. In addition, two FTW mesocosm experiments targeted an urban stormwater application (Chang *et al.*, 2012; Tanner and Headley, 2011). In their studies, fertilizers with nutrients were used to create simulated water or provided as supplements in the two urban stormwater FTW studies. A potential disadvantage of this approach is the nutrients are delivered mainly in bioavailable forms such as orthophosphate phosphorus (Tanner and Headley, 2011). orthophosphate phosphorus in fresh water is typically less than ten percent of TP (Wetzel, 2001). These two experiments may reflect the dynamic changes of those readily consumable nutrients; however, removal performance of other forms of nutrients in urban stormwater remains to be addressed. For example, phosphorus in organic forms is processed through biotic decomposition and then absorbed as orthophosphate phosphorus (Kadlec and Wallace, 2009). Third, two kinds of control are suggested to properly evaluate the compartmental effects of the floating mat and macrophyte on water quality according to our pilot experiment in 2010. One type of control is water without a floating mat (C); another is a floating mat without plants (M). Presently, only two peer-reviewed studies are available with the two types of control (Hu *et al.*, 2010; Tanner and Headley, 2011). The differences between the M and C indicate the effects of the floating mats. The microorganisms growing on the submerged surface of the floating mats have been suggested as the main pollutant removal mechanism of FTWs (Headley and Tanner, 2006; Wen and Recknagel, 2002). Shading caused by floating mats may alter other ongoing processes within the water body, such as photosynthesis. These floating mat effects were largely ignored when only one type of the control, C or M, was utilized in the experiments. The differences between planted and unplanted floating mats represent the performance of the macrophytes and associated microorganism. The surface area of the root systems serves as habitats of microorganisms (Nduvamana *et al.*, 2007; Osem *et al.*, 2007). Although nutrient uptake by macrophytes could be estimated through analyses of harvested plant tissue (Hubbard *et al.*, 2011; Wen and Recknagel, 2002; Zhu *et al.*, 2011), it may not include the contribution of the attached biofilms. Additionally, variation of parameters, such as DO and pH, could be caused by shading of the floating mats, plant activities, or both. Without an M control, it is impossible to reliably estimate these effects. Last, effectiveness of FTWs, while reported by many, has not been subjected to a rigorous statistical analysis except a few papers (Van de Moortel *et al.*, 2010; Yang *et al.*, 2008). These results may generate misleading conclusions if the assumptions of the underlying statistical analysis methods are not fully satisfied.

The objective of our study is to evaluate FTW performance in urban retention ponds where water was mostly supplied from runoff during storm events. Our hypotheses are: (1) FTWs can effectively enhance nutrient removal when compared to the control; (2) FTW nutrient removal is temperature dependent; (3) nutrient removal by the macrophytes on the floating mats is associated with the plant life cycle; (4) presence of the floating mats affects physicochemical properties of the associated water bodies; and (5) the shading effect of the floating rafts reduces chlorophyll-a concentrations.

4.3 Materials and Methods

4.3.1 Study site

The experiment was located in the City of Fairfax, Virginia at the Ashby Pond Conservatory Site (38º51” N, 77º17”W), within the Greater Washington, D.C. Metropolitan Area. The pond collects stormwater from its 0.57 km^2 developed drainage area with a mix of residential and commercial land uses. The catchment has an average annual precipitation of 1157 mm and monthly precipitation ranged 65-119 mm from 2003 to 2012. Monthly temperatures span from 0.7 to 24 ℃ in January and July, respectively (NOAA, 2012). The pond water drains to Accotink Creek which empties into the Potomac River, a major tributary of the Chesapeake Bay. As mentioned previously, the Chesapeake Bay is a major water body with eutrophication and ecological degradation problems in North America (Yasuhara *et al.*, 2012).

4.3.2 Plant species selection and preparation

Two plant species, pickerelweed (*Pontederia cordata* L.) and softstem bulrush (*Schoenoplectus tabernaemontani*), were tested in our study. The following criteria were developed for the plant species selection process: (1) native and non-invasive species in Virginia, U.S., (2) perennial plants, (3) terrestrial plant species, (4) wetland plants or the ability to thrive in a hydroponic environment, (5) plants with aerenchyma. Macrophyte species selection is critical not only to the pollutant removal but also to the local ecosystem integrity. Although several invasive species have high nutrient uptake rates, it is likely that their negative impacts on the ecosystem or the costs of habitat restoration may be more significant than their other benefits (Sheley *et al.*, 2006). An interesting exception could be argued if the native invasive plants already exist at the FTW application sites. Propagation of terrestrial plants on the FTWs is controllable compared to free floating plants, such as water hyacinth. The plants with natural buoyancy can proliferate and occupy the entire surface of a water body and may damage aquatic systems (Hubbard, 2010). Aerenchyma is effective in aerating the roots and rhizomes by transporting oxygen from aerial parts of the wetland plants (Cronk and Fennessy, 2001).Softstem bulrush and pickerelweed were selected based on the five criteria and previous research (Chen *et al.*, 2009; Wang and Sample, 2011). Aesthetic considerations were another reason of choosing pickerelweed because our research focuses on FTW application in urban retention ponds and pickerelweed is a flowering plant.

Seedlings of pickerelweed and softstem bulrush were collected from a local nursery. The nutrient-rich media present with the plants from nursery stock was complete removed. The macrophytes were then transplanted into 12.7 cm diameter hydroponic pots and with a bed of coir fibers (bristle coir fiber, RoLanka^{TM} Inc., Stockbridge, GA, U.S.). The coir fiber was elevated from the bottom of the hydroponic cup to prevent direct contact with plant roots (Figure 4-1) and to preserved the integrity of the whole plant when harvesting, especially the roots. In a previous FTW experiment in 2010, it was difficult to completely remove the coir fiber without damaging the plant roots when the macrophytes grew completely within the coir fiber. The bare rooted seedlings held in their individual hydroponic pots were transferred to an in-pond FTW cultural system (Section 4.3.3.3) for acclimation and establishment for 20 days. Before the initiation of the mesocosm experiment on May 17, 2012, stem number and primary stem/root length of 250 plants were measured. Individuals with uniform size and vigor were marked as experimental samples and used in the mesocosm study.

(a)

(b)

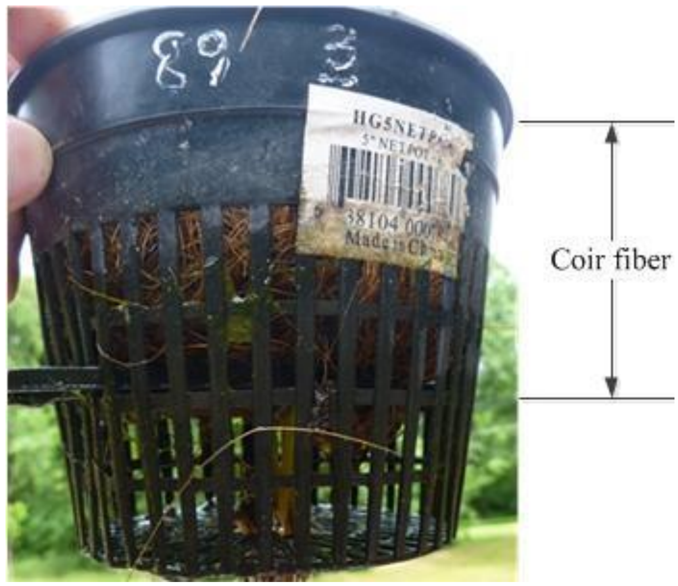

Figure 4-1. Bare root plants in the hydroponic pot with the coir fiber elevated from the bottom of the pot. (a) Top view. (b) Side view.

4.3.3 Experimental design

4.3.3.1 FTW experiment

The FTW mesocosm study was carried out from May 17 to October 31, 2012. Twelve polyethylene tanks with size of 1.11 × 0.55 × 0.46 m (L×W×H) were installed under a clear horticultural plastic shelter to facilitate photosynthesis of macrophytes and to eliminate the effects of rainfall, falling leafs, and bird droppings. The sides of the tanks were shielded by black cloth to block direct sunlight and reduce heat absorption with good ventilation. The water surface was 0.44 m^2 at 0.32 m operational water depth. The floating mat (0.76 × 0.38 m, L×W) was built from 3.81 cm diameter Polyvinyl Chloride (PVC) pipes, plastic mesh, and pot holders. The coverage of the FTW was 61% in the mesocosm experiment. Plastic mesh with three pre-cut holes was tied to the bottom of the floating PVC pipe frame. The pot holders were ring shaped with 10.16 cm diameter and fixed to the periphery of the pre-cut holes (Figure 4-2-a). A sun shade of corrugated cardboard with three pre-cut holes was the same shape as the FTW (0.76 × 0.38 m, L×W) and covered on the PVC frame to prevent sun light penetration within the FTW covered area. Three hydroponic pots with plants were inserted into the pot holders on each floating raft (Figure 4-2-b).

(a) (b)

Figure 4-2. Top view of the mesocosm FTW (0.76 × 0.38 m, L×W). (a) PVC pipe frame with plastic net and pot holders. (b) The FTW with plants in pots and covered by a corrugated cardboard sun shade.

Four treatments with three replicates following a completely randomized block design were used to test the effects of the floating mats and different plant species in our study (Table 4-1 and Figure 4-3). Twelve tanks were lined up in a row along east-west direction and divided into three blocks. The control treatment (C) with pond water was intended to simulate retention pond performance. The floating mat treatment (M) was used to evaluate the influence of the floating mat and associated microorganisms and possible constituents released from the growth media coir fiber. Billore *et al.* (2009) stated the life span of this environmental friendly but degradable material is more than six years in waterlogged conditions. Two plant species: pickerelweed (P) and softstem bulrush (B) were tested in our study as described in Section 4.3.2.

The experiment using the four treatments was conducted in a batch process. Water in each tank was pumped from Ashby Pond directly. After staying in the tanks for seven days (one batch), the experimental water was completely drained to the outlet channel of the pond and replaced by fresh urban stormwater from the pond.

Table 4-1. Summary of experimental treatments.

Treatment	**Description**	**Replicates (tanks)**
C	Control: only pond water	3
M	Floating mat: floating raft with pot and coir fiber	3
B	Floating mat + softstem bulrush (Schoenoplectus tabernaemontani)	3
P	Floating mat + pickerelweed (Pontederia cordata L.)	3
NS	Non-shaded (the same setting as C)	2
S	Shaded	2

(a) (b)

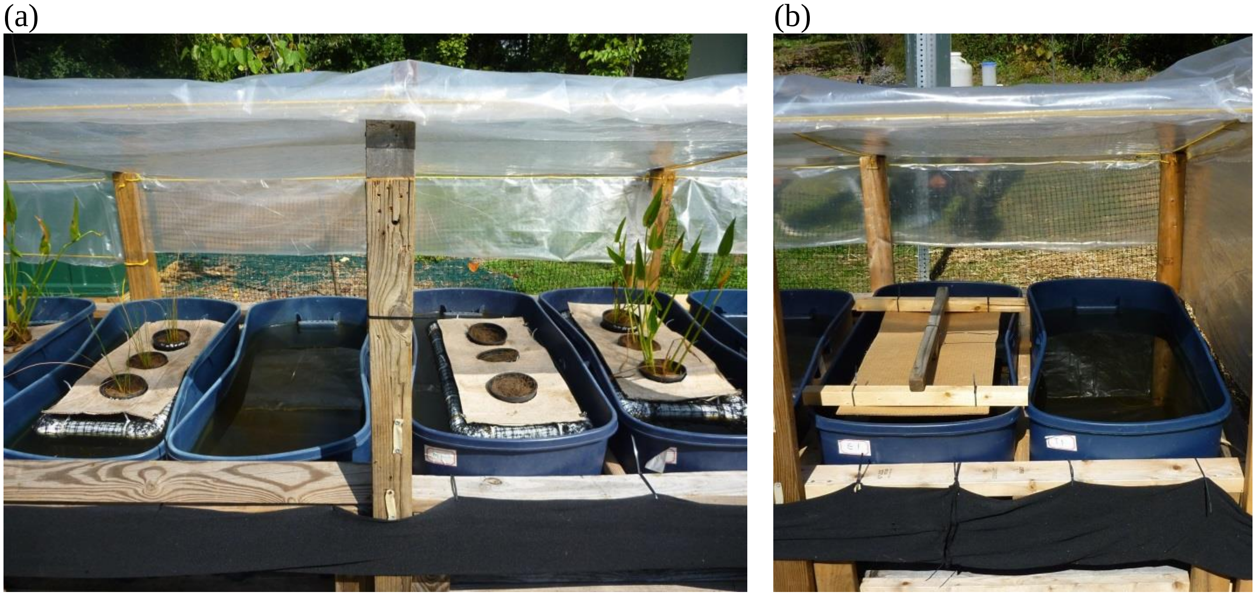

Figure 4-3. Photos of the six treatments in the mesocosm experiment. (a) From left to right: softstem bulrush, control, floating mat, and pickerelweed. (b) Shaded (left) and non-shaded (right). The descriptions of each treatment are in Table 4-1.

4.3.3.2 Shading effect experiment

The shading effect experiment was conducted from September 5 to October 31, 2012 (Stage 5 and 6). Four tanks with two treatments and two replicates for each treatment were carried out to evaluate the floating mat shading effects in the M, B, and P treatments in the FTW mesocosm experiment (Table 4-1 and Figure 4-3). The basic settings of the four tanks were the same as the C treatment in the FTW mesocosm experiment (Section 4.3.3.1). A piece of cardboard with the same size as the floating mat was suspended 14 cm above the water surface in the shaded treatment (S). A non-shaded control treatment (NS) was used to compare with S in the shading experiment.

4.3.3.3 In-pond FTW cultural system

The FTW cultural system was deployed in Ashby Pond to nurture experimental macrophytes and provide plant samples throughout the six months in this study. This system consisted of eight full-scale FTWs (each 1.52 × 0.91 m, L×W) with the same design as those in experiment tanks without pot holders, pre-cut holes on the plastic net, and corrugated cardboard covers (Figure 4-4). The diameter of the PVC pipes for the floating frame was 5.1 cm. The hydroponic pots with plants as described in Section 4.3.2 were placed on the plastic mesh and fixed by a coir mat (BioD-Mat® 40, RoLanka Inc., Stockbridge, GA, U.S.). Each cultural FTW contained 30 plant samples with only one species. The in-pond FTW cultural system was protected by a plastic fence (mesh size: 1.9 × 2.5 cm) from four sides and underneath the floating mats to reduce potential deleterious effects by waterfowl and turtles, which were present during our experiment. The fence stretched from 60 cm above to 30 cm below the water surface. Maintenance was conducted on July 23, 2012 to remove local plant species that colonized on the "new land" floating in the pond. This was another preventive step to reduce potential influences on our experimental samples by competing with these local macrophytes on the cultural FTW. The material cost of the in-pond cultural FTW without growth media was 19 USD/m^2. The cost of the bristle coir fiber (RoLanka™ Inc., Stockbridge, GA, U.S.) as growth media was approximately 52 USD/m^2.

Figure 4-4. The eight full-scale in-pond cultural FTWs protected by plastic fence from four sides and underneath.

4.3.4 Experiment operation and data collection

The FTW mesocosm experiment was conducted from May 17 to October 31, 2012, and was divided into six stages. There were four batches in one stage (28 days) and seven days in one batch. The plants in the mesocosm were replaced in each stage, while the water was exchanged in every batch (i.e., every seven days).

4.3.4.1 Experimental Water

Water levels of each tank were recorded on Day 0, 3, 6, and 7 to evaluate water loss caused by evaporation and evapotranspiration. Reverse osmosis (RO) water was added to each tank to the original water level on Day 3 and 6 to compensate for any evaporative losses. The amount of makeup water used in each tank was documented. As the water loss rate may be different between the four treatments, RO water was added to alleviate potentially inconsistent condensation effects between treatments and facilitate statistical comparison. Between each batch, the tanks were rinsed with water to remove deposited sediments before refilling with new water from Ashby Pond. The practice was modified after the first batch of Stage 3. The tanks were cleaned more thoroughly between batches with a plastic brush to remove excess periphyton grown on tank internal surface area. The rationale for the adjustment is described in Section 4.4.7.

Data of the experimental water were collected on Day 0, 3, 6, and 7 of each batch. The water samples were taken on Day 0 from Ashby Pond (as initial) while refilling the tanks and Day 7 from each tank for laboratory analyses. The analyzed constituents included chlorophyll-a (Chl-*a*), total phosphorus (TP), orthophosphate phosphorus (OP-P), total nitrogen (TN), ammonia nitrogen (NH_4^+-N), and nitrite–nitrate nitrogen (NOx–N) at the Occoquan Watershed Monitoring Laboratory. OP-P, NH_4^+-N, and NOx-N were determined with filtered samples as dissolved nutrients. Chl-*a* and nutrient concentrations were analyzed by a Trilogy® Laboratory Fluorometer (Turner Designs, Sunnyvale, CA, U.S.) and a nutrient auto-analyzer (Astoria Analyzer, Astoria Pacific, Clackamas, OR, U.S.), respectively. All analyses done in the lab followed a quality assurance/quality control (QA/QC) standard (Occoquan Watershed Monitoring Laboratory, 2001). Organic N (Org-N) was estimated as TN – NH_4^+-N – NOx-N. For total particulate phosphorus (TPP), the calculation TP – OP-P was used (Welch and Lindell, 1992). The physicochemical water properties: conductivity (corrected to 25℃), pH, dissolved oxygen (DO), and water temperature were measured by an YSI 556 multi-probe system (Yellow Springs, OH, U.S.) on Day 0 in Ashby Pond and on Day 3 and 6 in the tanks during each batch before added compensation RO water. The field measurement of each tank was conducted between 9:00 am to 12:00 pm Eastern Standard Time. The pH and DO meters were calibrated before each measurement. The parameters of the influent pond water used in the 24 batches (Day 0) are shown in Table 4-2. Two underwater temperature data loggers (EL-USB-1, Lascar Electronics, Erie, PA, U.S.) were used to record water temperatures with five-minute interval from May 22 to October 31, 2012. Data loggers were put into a C treatment tank and an adjacent FTW treatment tank to evaluate the FTW shading effects on the water temperature.

4.3.4.2 Experimental plants

Similar to the experimental water, the plants tested in the mesocosm after 28 days were replaced by fresh ones from the in-pond FTW cultural system in each stage. The purpose of using new plants from the in-pond FTW at every stage was to ensure that plant performance was adequately simulated in the mesocosm experiment. The plants in the greenhouse were expected to acclimate gradually to the indoor environment that is different from Ashby Pond. Following procedure was applied to both the pickerelweed and the softstem bulrush. One day before the beginning of each stage, 15 plants were randomly picked from the marked samples in the in-pond FTW cultural system, transported back to the lab, and thoroughly rinsed with tap water.

The plants were then blotted with absorbent paper and measured for their fresh weights, which are called reference fresh weights (RFW) and used in to estimate plant initial dry weight (Section 4.3.5). Then, these 15 plants per species were randomly separated into two groups. One group with six plants was harvested as initial status representatives (Section 4.3.5). The remaining nine subjects were the test group that replaced the previous samples in the experimental tanks to begin a new stage. The nine fresh plants were arranged in a way to equalize the total plant fresh weight between the three replicates (tanks) of each FTW treatment. When the fresh plants tested in the new stage, destructive analyses were carried out on the harvested samples: six initial status representatives and the replaced samples grown in the mesocosm for 28 days (hereafter referred to as the “INI plant” and “MESO plant”, respectively). The data of the INI plant were used in macrophyte growth rate calculations (Section 4.3.5).

Table 4-2. Physicochemical characteristics of pond water measured at Day 1 (n=24).

Parameter	**Mean**	**±**	**SD**
TP (mg/L)	0.15	±	0.03
OP-P (mg/L)	0.02	±	0.01
TN (mg/L)	1.19	±	0.27
NH_4^+-N (mg/L)	0.05	±	0.09
NOx-N (mg/L)	0.05	±	0.07
Org-N (mg/L)	1.09	±	0.22
Chl-*a* (µg/L)	16.17	±	11.72
pH	6.36	(6.18-6.61)[δ]	
DO (mg/L)	5.09	±	2.00
Temperature (℃)	23.09	±	4.98
Conductivity (us/cm[c])	133.54	±	44.73

[c] Standardized to 25℃
[δ] Converted from the arithmetic mean of hydrogen-ion activity with 1st and 3rd quartile pH values in the parentheses.

The harvested plants were gently washed with tap water and blotted with absorbent paper. Stem, leaf, and primary root length of each plant were also measured. Before measuring fresh and dry weight, the plants were composited to reduce sample number. While the six INI plants randomly separated into two groups (INI-1 and -2), the MESO plants were arranged the same way as they were in the mesocosm experiments, three tanks (MESO-1, -2, and -3). Then, the five composited plant samples were separated into shoot, rhizome (pickerelweed only), and root sections, and weighed for the corresponding fresh weight. All the plant tissues were dried at 70℃ for 48 hours to stabilize the samples and to determine their dry weight (Plank, 1992). The rhizome was sliced into thin pieces before drying to expedite the process. The dried plant tissues were grounded in a Wiley Mill (Thomas Scientific, Model: 3379-K35) and passed through a 35-mesh (0.5 mm) screen. The plant phosphorus content was determined by a wet digestion method conducted at the Occoquan Watershed Monitoring Laboratory (Ebeling *et al.*, 2010; Ruiz and Velasco, 2010b). The five composited samples were further reduced to two groups, INI-t and MESO-t, before the chemical analysis. The same weights of the grounded samples from the two INI samples or from the three MESO samples were composited. The samples from different part of the plant, like shoots and roots, were not mixed. The grounded samples (0.02 g each) were autoclaved with 5.5M sulfuric acid and potassium persulfate at 110℃ for one hour. The digested solution was analyzed by the nutrient auto-analyzer (Astoria Analyzer, Astoria Pacific, Clackamas, OR, U.S.).

4.3.5 Macrophyte growth rate calculation and initial dry weight estimation

Macrophyte growth rates were estimated according to the weight changes of the plants. The dry weights (DW) of the MESO plants at the end of each stage were measured as described in Section 4.3.4.2 (MESO-1, -2, and -3). However, their corresponding initial dry weights were not available. Therefore, the data of the initial status representatives (INI-1 and -2) were used to estimate the MESO plants' initial whole plant DWs. The INI plants were harvested at the time when the corresponding MESO plants were put into tanks for testing (Section 4.3.4.2). The reference fresh weight (RFW) of both the INI and MESO plants are used to calculate the estimated initial whole plant dry weight of the MESO plants (Equation 4-1). The initial shoot dry weight of the pickerelweed could be estimated because every shoot length of the INI and MESO plants were measured (Equation 4-2). Thus, the below-ground DWs of the pickerelweed were calculated as the difference between the estimated whole plant DWs and estimated shoot DWs.

- Whole plant initial dry weight estimation

$$MESO_{i,j}\_W\ initial\ DW = MESO_{i,j}\_RFW \times Ave.\left(\frac{INI_{i,k}\_W\_DW}{INI_{i,k}\_RFW}\right) \quad \text{Equation 4-1}$$

- Shoot initial dry weight estimation

$$MESO_{i,j}\_S\ initial\ DW = MESO_{i,j}\_SL \times Ave.\left(\frac{INI_{i,k}\_S\_DW}{INI_{i,k}\_SL}\right) \quad \text{Equation 4-2}$$

where

Ave.: average of;

DW: dry weight of whole plant (g);

$INI_{i,k}$: initial plant status representatives of Stage *i*, group *k*;

$MESO_{i,j}$: plants tested in the mesocosm for 28 days in Stage *i*, group *j*;

RFW: reference fresh weight (g). INI and MESO plants measured at the same time before the beginning of a new stage (Section 4.3.4.2);

S: shoot (for pickerelweed only);

SL: shoot length (cm) (for pickerelweed only);

W: whole plant (for pickerelweed and softstem bulrush);

i: stage, 1 to 6;

j: group of the MESO plants, 1 to 3;

k: group of the INI plants, 1 to 2.

After the initial DWs of the MESO plants were estimated, the corresponding average relative growth rate (*RG*) is calculated. This parameter represents the new material production efficiency of the plant (Hunt, 1978). The calculation of the *RG* is shown in Equation 4-3.

- average relative growth rate (*RG*)

$$RG_{i,j,\delta} = \frac{ln\left(MESO_{i,j,t_2}\_DW_{\delta}\right) - ln\left(MESO_{i,j,t_1}\_DW_{\delta}\right)}{t_2 - t_1} \quad \text{Equation 4-3}$$

where

$RG_{i,j,\delta}$: average relative growth rate of Stage *i*, group *j*, tissue type δ;

t_1: initial time of Stage *i* (0 days);

t_2: final time of Stage *i* (28 days);

δ: tissue type: (1) whole plant, (2) above-ground tissue, or (3) below-ground tissue.

4.3.6 Performance calculation and statistical analysis

The removal efficiency (RE, %) of each treatment in each batch (seven days) was calculated based on Equation 4-4. The change of the TP and TN concentrations in the FTW treatments was assumed to follow first-order kinetics for simplicity. This method is widely adapted to simulate nutrient removal in constructed wetlands and ponds (Kadlec and Wallace, 2009; Shilton, 2005). The modified Arrhenius equation was used to summarize temperature effects on the reaction rate (Chapra, 1997). The first order reaction and the modified Arrhenius equation are expressed as Equations 4-5 and 4-6, respectively. A regression test was utilized to examine the adequacy of using the modified Arrhenius equation. The tested assumptions included residual normality and homogeneity of variances.

- Removal efficiency (RE, %)

$$\mathrm{RE} = \frac{(c_0 - c_t)}{c_0} \times 100\ \% \qquad \text{Equation 4-4}$$

- First order reaction rate (day^{-1})

$$k_T = \frac{\ln(c_t/c_0)}{t} \qquad \text{Equation 4-5}$$

- Modified Arrhenius equation

$$k_T = k_{20} \times \theta^{(T-20)} \qquad \text{Equation 4-6}$$

where

$c_{0/t}$: initial concentration (time = 0) or final concentration (time = t) in each batch (mg/L);

k_T: first order reaction rate at T ℃ water temperature;

k_{20}: first order reaction rate at 20 ℃ water temperature;

t: reaction time (day);

T: water temperature (℃);

θ: temperature correction factor.

The significance of the removal efficiency differences between treatments was examined through one-side paired sample Wilcoxon signed rank test with a Bonferroni correction (Milton and Arnold, 2008). The one-sided test was chosen to assess whether the performance of one treatment in the four treatments (C, M, B, and P) comparison is greater than the other. For the shading effects experiment, the two-sided test was used to investigate the existence of the differences. The nonparametric method was used because the results from the preferred parametric analysis (Analysis of Variance, or ANOVA) of our data did not meet required assumptions, i.e. residual normality and equal variance. Another reason for choosing nonparametric analysis was the existence of censored data. The values of OP-P, NH_4^+-N and NOx-N concentrations below the detection limit were adjusted to one-half the detection limit. The nonparametric rank-based analysis can treat the censored data as tied with each other. Parametric methods, such as ANOVA, were conducted based on mean, standard deviation and other statistics. These parameters calculated from the censored data could produce less reliable results than ranked-based methods (Helsel, 2012). The Bonferroni correction is an easy and well accepted method addressing error rate related to multiple comparison tests (Milton and Arnold, 2008). One disadvantage of using the Bonferroni correction is it can be overly conservative when tests are correlated (Conneely and Boehnke, 2007). Therefore, the level of significance (α) was less restricted and set to be 0.1 for multiple comparisons between four treatments (C, M, B, and P). The adjustments of the *p*-value were based on Equation 4-7. All statistical analyses were conducted with Minitab® (16.1.0 version) and/or Microsoft Office Excel (2010).

- *p*-value Bonferroni correction (*n* treatments comparison)

$$p = p \times y = p \times \frac{n \times (n-1)}{2} \qquad \text{Equation 4-7}$$

where

n: compared treatments number (four);

p: *p*-value of two treatments comparison;

p: adjusted *p*-value of *n* treatments comparison;

y: the possible pairs of multiple comparison between *n* treatments. For example, *y* is six when *n* is four (Milton and Arnold, 2008).

4.4 Result and Discussion

4.4.1 Physicochemical responses

The pH and DO in the C treatment was significantly higher than the FTW treatments at 99% confidence level (Figure 4-5). Receiving 61% more solar radiation than the FTW treatments, photosynthesis was more active in the C treatment, thus consuming CO_2, raising pH, and releasing O_2 (Shilton, 2005). The range of pH of C was 6.62-8.98, with mean of 7.26 compared

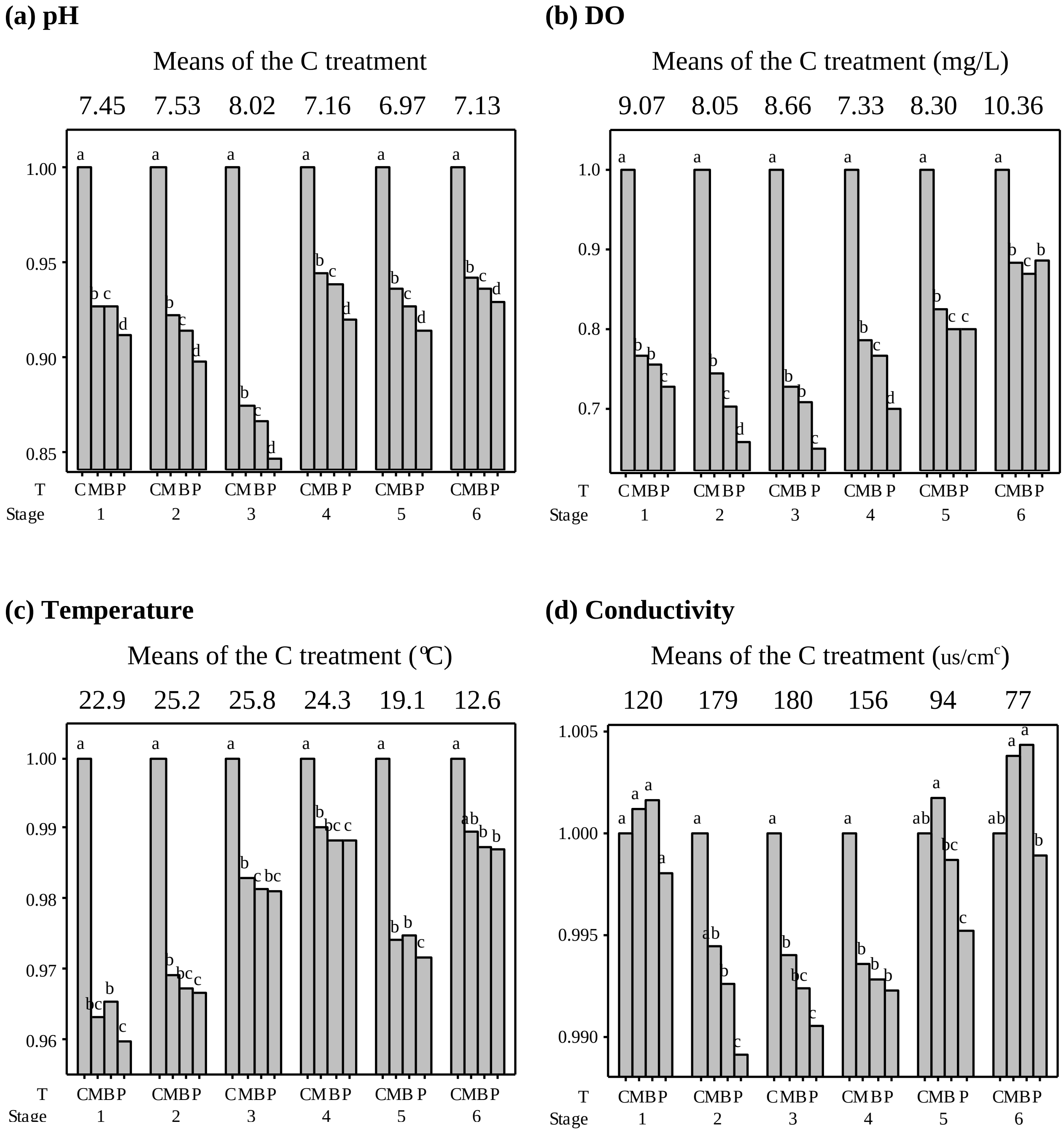

Figure 4-5. Ratio of physicochemical characteristics of the four treatments normalized to control (C). Data were collected on Day 3 and 6 in each batch (seven days). Letters above each bar denote one side paired sample Wilcoxon signed rank test. In each stage, treatments with the same letter are not significantly different from each other (α = 0.1). [c]: Standardized to 25℃.

to 6.62 of the pickerelweed treatment. High pH may increase ammonia volatilization, but may carry a risk of ammonia toxicity, which occurs in basic environments. Azov and Goldman (1982) reported that biological C^{14} uptake ratio reduced from 0.95 to 0.5 as the pH increased from 8 to 8.9 with 0.14 mg/L NH_4^+-N in the solution. Comparison among the three FTW treatments indicates that the pH and DO were significantly lower in the P treatment ($p < 0.05$) except for DO in Stage 5 and 6. Microorganisms, greatly influenced by temperature, are possibly the major cause of the DO variation within the three FTW treatments. Pickerelweed's extensive root structure may provide great surface area for adsorption by biofilms (Tanaka *et al.*, 2012). The respiration of these microbes may cause considerable oxygen consumption, especially at higher temperatures. The significant differences of DO between the M and P treatments were reduced when temperature was dropped to lower than 20 ℃ (Stage 5 and 6). The DO in M and P treatments is considered the same in Stage 6 as the temperatures dropped below 15 ℃. Song *et al.* (2011) reported that total bacteria density reduced from 2.33×10^8 to 0.56×10^8 cells/g dry artificial medium when the water temperature changed from 20 to 5 ℃ in an FTW experiment. The lower oxygen concentrations in the FTW treatments compared to the control treatment was also reported by Tanner and Headley (2011). Although wetland plants could transport oxygen into plant roots, the process is driven by the demand of the external environments adjacent to the roots, which was aerated water in this study (Colmer, 2003). The lower pH in planted FTW compared to the M and C treatments were continuous throughout our experiment. This condition was also observed in another FTW experiment (Van de Moortel *et al.*, 2010). The change of the pH is possibly a result of acidic exudates from plant roots (Coleman *et al.*, 2001). It is likely that higher nitrification effects may also exist in the planted FTW (B and P) thus reducing pH as described in Section 4.4.4. Protons in the ammonia are released through nitrification process (Sawyer *et al.*, 2004).

The variation of conductivity (standardized to 25 ℃) and water temperature was consistent with the seasonal change (Figure 4-5). The water temperature in the four treatments was continuously above 20 ℃ from Stage 1 to 4 (May to August, 2012). The highest average of water temperature in the experiment tanks was 30 ℃ in early July, 2012. The water temperature dropped to below 20 ℃ and 15 ℃ in Stage 5 and 6, respectively. Stage 5 was the beginning of the fall (September, 2012). The conductivity variation between stages was primarily controlled by the experimental water source, Ashby Pond. Conductivity was lower in Stage 1, 5, and 6 than any other period. The pond water was greatly diluted by precipitation in Stage 1, while pond water temperature was reduced considerably in Stage 5 and 6. For the later reason, it is suggested that the microbial activities, correlated with water temperature, may play a dominant role in releasing elements from sediments (Olivie-Lauquet *et al.*, 2001). Therefore, the water conductivity in Stage 5 and 6 was reduced with lower temperature due to the inactivity of the microorganisms in Ashby Pond. Although the differences between the four treatments were minimal at each stage, the nonparametric ranked test indicates that the conductivity in the C treatment was slightly, but significantly higher than B and P in Stage 2, 3, and 4 ($p < 0.05$). For temperature, the variation between the treatments was clear and greatly influenced by the shading effect. The temperature in the C treatment was significantly higher ($p < 0.01$) than those in other three FTW treatments from Stage 1 to 5. With 61% coverage, the greatest temperature difference, 2.51 ℃, occurred between the C and P treatment on June 3, 2012. Van de Moortel *et al.* (2010) reported that water temperature in the FTW treatment with 64.2% coverage was significantly reduced compared to the control. However, this situation may not be reproduced in a small scale FTW field experiment. It is suggested that a high percent coverage is required to reduce water temperatures (Winston *et al.*, 2013). The result in this study indicates the potential of thermal pollution control through the application of the FTW with 61% coverage. Heat accumulation in most retention ponds results in warmer temperature in connected streams and threats local habitats (Ham *et al.*, 2006).

4.4.2 FTW effect on water temperature

The presence of the FTW has the potential to influence the water temperature in the mesocosm experiment according to the records of the underwater temperature data loggers. The variation of the daily maximum water temperature and ranges in the control and FTW tanks were summarized by multiple regression equations. As shown in Table 4-3, the daily maximum water temperature (WM) was reduced by 1.79 °C with the presence of the FTW (PF=1). Depending on the diurnal air temperature variation (AV), the daily temperature fluctuation of the experimental water (WV) in the FTW treatments experienced a 5 to 0 °C reduction compared to the control group. The attenuation of the diurnal fluctuations by the FTW was positively correlated with average water temperatures. The *p*-values for the estimated coefficients of the four predictors were less than 0.01, which indicates they are all significantly related to the dependent variables

(WM and $\sqrt{WV}$) in both equations. The predictors include daily maximum air temperature (AM), daily air temperature fluctuation (AV), present of FTW (PF), and cloud cover (CV). A square root transformation was applied to the values of the water diurnal temperature variation (WV) to linearize parameters and satisfy assumptions of the multiple regression method, including residual normality and equal variances (Krebs, 1999). The variation of the water maximum temperature (WM) can be almost entirely accounted for in the independent variables (R^2-adj=93%). For water diurnal temperature variation (WV), the adjusted coefficient of determination is acceptable (62.6 %). These results indicate that in the case of the mesocosm with a shallow water depth of 0.32 m and 61 % coverage effectively reduces water heat absorption and the potential for thermal pollution. Winston *et al.* (2013) conducted field experiments at two urban wet ponds with 9 and 18 % coverage, respectively. The water temperatures were measured below the center of an FTW and at open water, 2 m away from the edge of the FTW. They concluded little shading benefit from the FTWs because the temperature differences between the open water and under the FTW were insignificant. This may be due to diffusion or convection which equalized or reduced the water temperature differences between the two adjacent sites. The temperature gradient between open water and shaded area (under FTW) would likely cause convection and subsequent mixing. Although the magnitude of the water temperature reduction may be amplified in our FTW mesocosm experiment, it still provided solid evidence of the FTWs' contribution on thermal pollution control.

Table 4-3. Multiple regression equations and goodness of fit (R^2) for daily water temperature data. All estimated slope coefficients are significantly related to WMT and WTF parameters (p <0.01)

Equation*	**R^2 (%)**	**R^2-adj (%)**
WM = - 0.36 + 0.91 AM - 1.79 PF + 0.29 CV	90.4	90.3
$\sqrt{WV}$ = 1.83 + 0.06 AV - 0.54 PF - 0.05 CV	63.1	62.6

* WM: Water, daily maximum temperature (℃); $\sqrt{WV}$: Water, square root of diurnal temperature variation (℃); AM: Air, daily maximum temperature (℃); AV: Air, diurnal temperature variation (℃); PF: present of FTW (1: yes, 0: no); CV: Cloud cover (okta).

4.4.3 Nutrient removal efficiency comparison

The TP, TPP, TN, and Org-N removal efficiency (RE) of the four treatments in the six stages (1 to 6) and growth period (G) with corresponding statistical analysis results are summarized in Figure 4-6. Growth period is defined as Stage 1 to 4 due to a considerable reduction of RE-TP and RE-TN after Stage 4. The RE-TP was lower than 60% of all treatments in Stage 5 and 6. For RE-TN, the value is 40% after Stage 4. This grouping method corresponded with seasonal patterns and temperature changes (Section 4.4.1). The fall season began in September, Stage 5, when temperature dropped below 20 °C (Figure 4-5). The removal efficiency discussion applied to TP, TPP, TN, and Org-N but bioavailable nutrients, OP-P, $NH4^+$-N, and OxN-N. Net reduction of these three constituents was not achieved because (1) they were continuously transformed and replenished from organic phosphorus and nitrogen; (2) their initial concentrations in some batches were lower than the detection limits.

Plant performance is discussed in Section 4.4.3.2 by comparing the three FTW treatments (B, P, and M) in Stage 1 to 6 and G. Planted FTW and floating mat effects compared to the C treatment are evaluated in Stage 1, 3c, 4 to 6, and Gc (Section 4.4.3.1). The time series gap in the later comparison is due to a periphyton outbreak in the C during Stage 2 and the first week of Stage 3 (discussed later in Section 4.4.7). The C treatment could not be used as a control to determine the FTW treatments effectiveness during these five weeks as the periphyton outbreak did not occur in the M, B, and P treatments simultaneously. This five-week data were excluded from the calculations of Stage 3c and Gc.

4.4.3.1 FTW effects compared to the control (Stage 1, 3c, 4 to 6 and Gc)

The results of the FTW treatments compared individually with the C indicate that all three designs (B, P, and M) effectively improved phosphorus and nitrogen removal within urban retention ponds (Figure 4-6, Stage 1, 3c, 4 to 6, and Gc). The one-sided nonparametric paired tests indicates that RE-TP, RE-TPP, RE-TN, and RE-Org-N of the each planted FTWs (P and B) with 61% coverage was significantly higher than the C at the 99% level of confidence in the growth period (Stage Gc). The floating mats alone (M) showed significant enhancement compared to C in RE-TP, RE-TPP, and RE-Org-N ($p < 0.01$) during Stage Gc. For RE-TN, the value of p between the M and C treatments is 0.05. The removal efficiency within the C, M, B, and P in Stage Gc were 63.4, 65.9, 67.5, and 68.6 % for TP, 61.8, 66.8, 67.6, and 70.3 % for TPP,

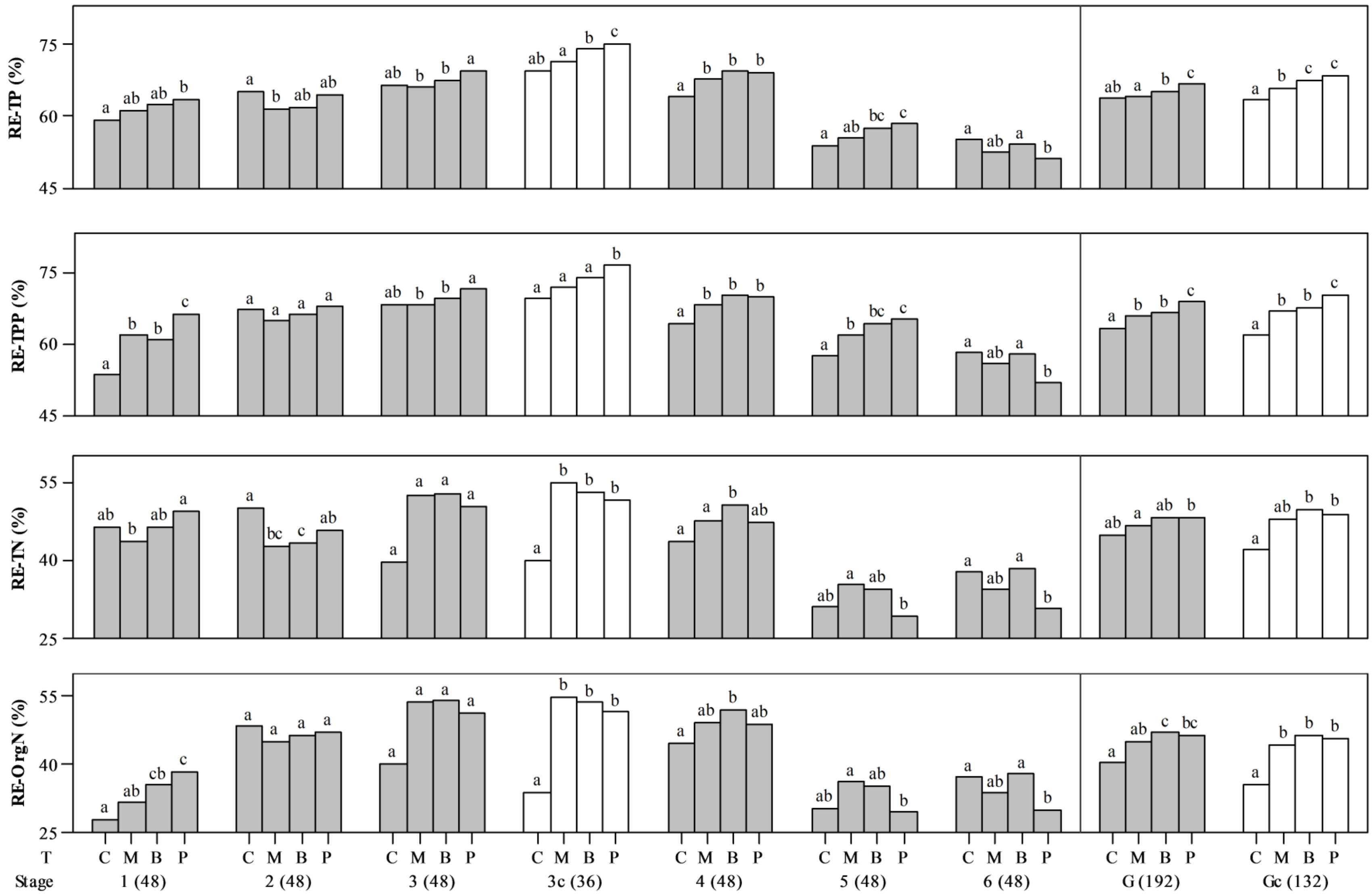

Figure 4-6. Mean removal efficiency (RE) of the four treatments (T) in each stage and growth period (G, stage 1-4). Stage 3c and Gc represent calculations without data from the periphyton influenced period (Section 4.4.3). Description of letters above each bar refers

42.1, 48.1, 49.8, and 49.1% for TN, 35.7, 44.3, 46.4, and 45.8% for Org-N, respectively. With the application of the floating mat with pickerelweed (P) and softstem bulrush (B) in the growing season, the TP and TN removal within C were improved by 8.2 and 18.2 %, respectively. The floating mat alone without plant provided 3.9 and 14.3 % improvement on RE-TP and RE-TN of the control treatment, respectively. The comparison between the four groups identified the treatment with the highest RE-TP and RE-TN in the study. The result suggests that the pickerelweed (P) offered significantly higher phosphorus removal ($p < 0.1$) than the other three treatments (except B in RE-TP). For nitrogen, the removal efficiency within the three FTW treatments (M, B, and P) were similar but significantly higher than the C treatment ($p < 0.1$), except for RE-TN in the M treatment.

For phosphorus, the statistical analysis results in each stage suggested that the floating mat combined with pickerelweed consistently provided the highest TP and TPP removal efficiency among the four treatments and followed by the softstem bulrush from Stage 1 to 5. The paired comparison with the C shows that the P treatment significantly ($p < 0.05$) improved RE-TP and RE-TPP in Stage 1, 3c, 4, and 5, which is equivalent to 79% of the 19-week comparison period. For the B, the differences compared with C are significant in Stage 4 and 5 for RE-TP ($p < 0.01$) and Stage 1, 4, and 5 for RE-TPP ($p < 0.01$). Without plants, the floating mat (M) still provided higher removal than C for Stage 1, 3c, 4 and 5 for TP and TPP. Significant differences between M and C were observed in Stage 4 for RE-TP ($p < 0.01$) and Stage 1, 4, and 5 for RE-TPP ($p < 0.01$).

The improvement of RE-TN by FTW applications was significant but less effective than RE-TP from the perspective of the time span. The FTW planted with the softstem bulrush (B) exhibited significantly higher RE-TN than C in Stage 3C, 4, and 5 ($p < 0.05$), which is equivalent to 58% of the 19-week comparison period and shorter than the time span of RE-TP in the P treatment (79%). For the P treatment, the effective improvement period was Stage 1 and 3c ($p < 0.05$). The floating mat treatment (M) showed higher RE-TN performance than C from Stage 3c to 5, which is one stage less than the period exhibiting similar behavior for RE-TP. The FTW performance for Org-N removal was similar to TN but better in Stage 1. The RE-Org-N of each planted FTW (B and P) was significantly higher than the C treatment ($p < 0.01$) in the first stage. The comparison between the four treatments in Stage 5 and 6 are discussed separately in Section 4.4.3.3.

According to Figure 4-6, the FTW treatments effectively removed more TPP and Org-N than the C treatment, which resulted in significantly higher RE-TP and RE-TN in the M, B, and P treatment. TPP consists of phosphorus in forms of organic matter, minerals, and precipitates from sorption (Robards *et al.*, 1994). Therefore, FTW application may increase the mineralization of the organic compounds into orthophosphate phosphorus (OP-P) and ammonia in urban retention ponds. Organic matter is mineralized through biological transformation into bioavailable forms for uptake by plants and microorganisms (Richardson and Vepraskas, 2001). Both algae and bacteria can utilize OP-P through degradation of organic phosphorus in OP-P deficient water (Robards *et al.*, 1994). Bacteria are suggested to be the major contributor in this study since there were no significant Chl-*a* concentration differences between the four treatments from Stage 2 to 5 (Section 4.4.4). Li *et al.* (2011) reported that alkaline phosphatase increase 67% compared to the control treatment after ten days in the FTW planted with *Lolium perenne* ‘Top One’. Alkaline phosphatase is produced by bacteria and is related to hydrolyzing organic phosphorus compounds (Robards *et al.*, 1994). Additionally, identifying bacteria decomposition as the major FTW improvement mechanism is consistent with the observed temperature variation (Section 4.4.1). Microbial activities are likely the main nutrient removal component in the M treatment. The RE-TPP and RE-Org-N of the M treatment were continuously higher than the C until Stage 6 when the temperature dropped to below 15 ℃ (Figure 4-5). Temperature is regarded as the major factor controlling microbial activities (Kadlec and Wallace, 2009). Li *et al.* (2010) suggested that the submerged underwater surface of the plants may support microbial growth. Biofilms attach on the artificial root or media surface and contribute to nutrient removal (Song *et al.*, 2011; Wang and Sample, 2011). Besides organic matter, other constituents of TPP could also be removed by the FTW treatments. Tanner and Headley (2011) reported that suspended particulate pollutants may adhere to the root–biofilm network of the planted FTW treatments in their standing water experiment, similar to our study design.

4.4.3.2 Macrophytes effects on nutrient removal (Stage 1 to 6 and G)

Statistical analysis results of the comparison between (1) the planted FTWs and the floating mat (M) and (2) between the two plant species are provided in Figure 4-6, Stage 1 to 6 and G. The comparison between the planted and unplanted FTW treatments at the 95% level of confidence shows that both pickerelweed (P) and softstem bulrush (B) provided significantly higher TP, TPP, TN, and Org-N removal than the M treatment in Stage G ($p < 0.05$).

Additionally, the comparison between the B, P, and M treatment identifies the FTW design with highest nutrient removal efficiency in this study. The result suggested that the pickerelweed removed TP and TPP more significantly than the B and M treatment in growth period ($p < 0.05$). In contrast to these results for the phosphorus constituents, there were no significant differences between the B and P treatments regarding TN and Org-N removal efficiency at the 95% level of confidence in growth period (G). However, the RE-TN and RE-Org-N of both planted FTW treatment was significantly higher than those of the M treatment ($p < 0.1$). These differences were not shown in Figure 4-6, which summarized the results with higher confidence level (α_3 = 95%).

The result suggests that both plant species significantly improved the floating mat treatment (M) on phosphorus and nitrogen removal, pickerelweed was consistently the highest performer. Pickerelweed showed better phosphorus removal efficiency than softstem bulrush in our experiment. The growth performance was significantly different between the two species. The pickerelweed produced flourishing aerial tissues, roots, and rhizomes, whereas no significant biomass accumulation was observed in the B treatment according to our observation. The extensive root system of the pickerelweed may better support microorganism activities than the M and B treatments. The nutrient preferences of the plants may result in more significant reduction of phosphorus than nitrogen when compared to the floating mat treatment. Chen *et al.* (2009) reported that the mass based nitrogen: phosphorus (N:P) ratio of the pickerelweed grown hydroponically was 3.7, whereas Hillebrand and Sommer (1999) suggested the value of the periphyton was 7.7. The different N:P mass ratios potentially provide a higher demand for phosphorus uptake and less for the nitrogen in the P treatment. This could cause greater increase of the RE-TP than RE-TN when compared to the M treatment. However, the N:P mass ratio of the wetland plant tissues vary depending on the habitat and environmental factors. Additionally, other plant species with greater nitrogen demands may be utilized in the FTW system when nitrogen reduction is the primary design consideration. Finally, the plant effects may be more significant when applied at suitable environments with higher density. The recording plant density of the pickerelweed was 171/m^2 at the field scale (Sutton, 1991), which is 16.5 times higher than the value in our experiment. The major difference in density is because our plant density was lower as three plants per raft.

4.4.3.3 Plant senescence effects

Compared to the M treatment, plant growth performance considerably influenced the FTW's nutrient removal efficiency in Stage 5 and 6 (Figure 4-6). The information could provide guidance for scheduling maintenance practices, such as plant harvesting. A strong indication of plant senescence of pickerelweed was shown in Stage 5 and 6. The TP and TPP removal efficiency of the P treatment was significantly higher than M from Stage 1 to 5. However, the relationship was reversed in Stage 6 (M>P) but was not statistically significant. The release of nitrogen from the pickerelweed plant tissues may occur one month earlier than the case for phosphorus. The TN and Org-N removal efficiency of the P treatment was significantly lower than M at the 95% level of confidence in Stage 5. The pickerelweed released phosphorus and nitrogen at different times which is consistent with its growth characteristics. The pickerelweed stem growth rate gradually declined and stopped during Stage 5. Therefore, the results may indicate that the nitrogen was released from the pickerelweed when above-ground tissues stopped growing in early September. Observed growing "eyes" on rhizomes were continuously generating during the four weeks of Stage 5. The phosphorus uptake may have been extended for one more stage because of this demand. However, the condition was not observed in the case of the nitrogen. It may be resulted from insufficient supply of phosphorus throughout the experiment. Phosphorus was the limiting nutrient because the average of N:P mass ratio in the P treatment at the end of each batch was 12.2 compared to 3.7 of the pickerelweed tissue reported by Chen *et al.* (2009).

Senescence was not observed in the B treatment during our six months experiment. Furthermore, the softstem bulrush exhibited higher but not significant TP, TPP, TN, and Org-N removal efficiency than M in Stage 6 (October). This indicates that the softstem bulrush may remain active, in contrast with the observed decline of microbial activity in the floating mats (M). The different performance of the pickerelweed and the softstem bulrush in Stage 5 and 6 may reflect their individual nature as herbaceous and evergreen perennials, respectively. Based on the performance of the two plant species, harvesting may be applied to reduce the release of nutrients from senesced plant tissue in September for the pickerelweed; for softstem bulrush, more work across an entire annual period is suggested to evaluate timing effects beyond the range of this study.

The change of the TP and TN removal efficiency with temperature/season has been reported in FTWs, constructed wetlands, and pond treatment systems (Kadlec and Wallace, 2009; Shilton, 2005; van Oostrom, 1995). Zhao *et al.* (2012a) stated that the TP and TN removal efficiency for FTWs from summer-autumn to winter-spring decreased from 43.3 to 17.7 % and from 36.9 to 20.5 %, respectively. Temperature was considered the major factor of the seasonal variation in FTW systems (Van de Moortel *et al.*, 2010; van Oostrom, 1995). As plants experience senescence and decompose in cool temperatures, a portion of the nutrients may return to the aquatic system (Cronk and Fennessy, 2001). Despite this disadvantage, the FTW still provides net nutrient removal on an annual basis if managed appropriately. Additionally, the interception of bioavailable nutrients during the growing season protects downstream water bodies from eutrophication during susceptible periods (Nichols, 1983).

4.4.4 Chlorophyll-a and dissolved nutrients

The Chl-*a* concentration at Day 7 is shown in Figure 4-7. The nonparametric statistical analysis results indicate no significant differences existed between the treatments from Stage 2 to 5. The Chl-*a* concentration in the P treatment was significantly lower than C only in Stage 1 (p = 0.02). However, M and P was significantly higher than C at the 95% level of confidence in Stage 6, which contradicts our hypothesis that the shading of FTWs could reduce Chl-*a* concentration. The results indicate that the shading effects of the experimental tanks with 61% FTW coverage did not inhibit the growth of algae when compared to the C treatment. This statement is supported by the results of our shading experiment (Section 4.4.6). Based on previous studies, nutrient competition between the FTW and algae was considered the major factor of the Chl-*a* variation (Li *et al.*, 2007b; Nakamura and Shimatani, 1997). In Stage 6, the Chl-*a* concentration within both the M and P treatments was significantly higher than that of C ($p < 0.1$). Simultaneously, the TP and TN removal efficiency of the two FTW treatments was lower than C, and different from conditions in the other stages (Figure 4-6). In cases where the two FTW treatments could not effectively remove TP and TN better than the C treatment may have resulted in more untreated nutrients for algae uptake.

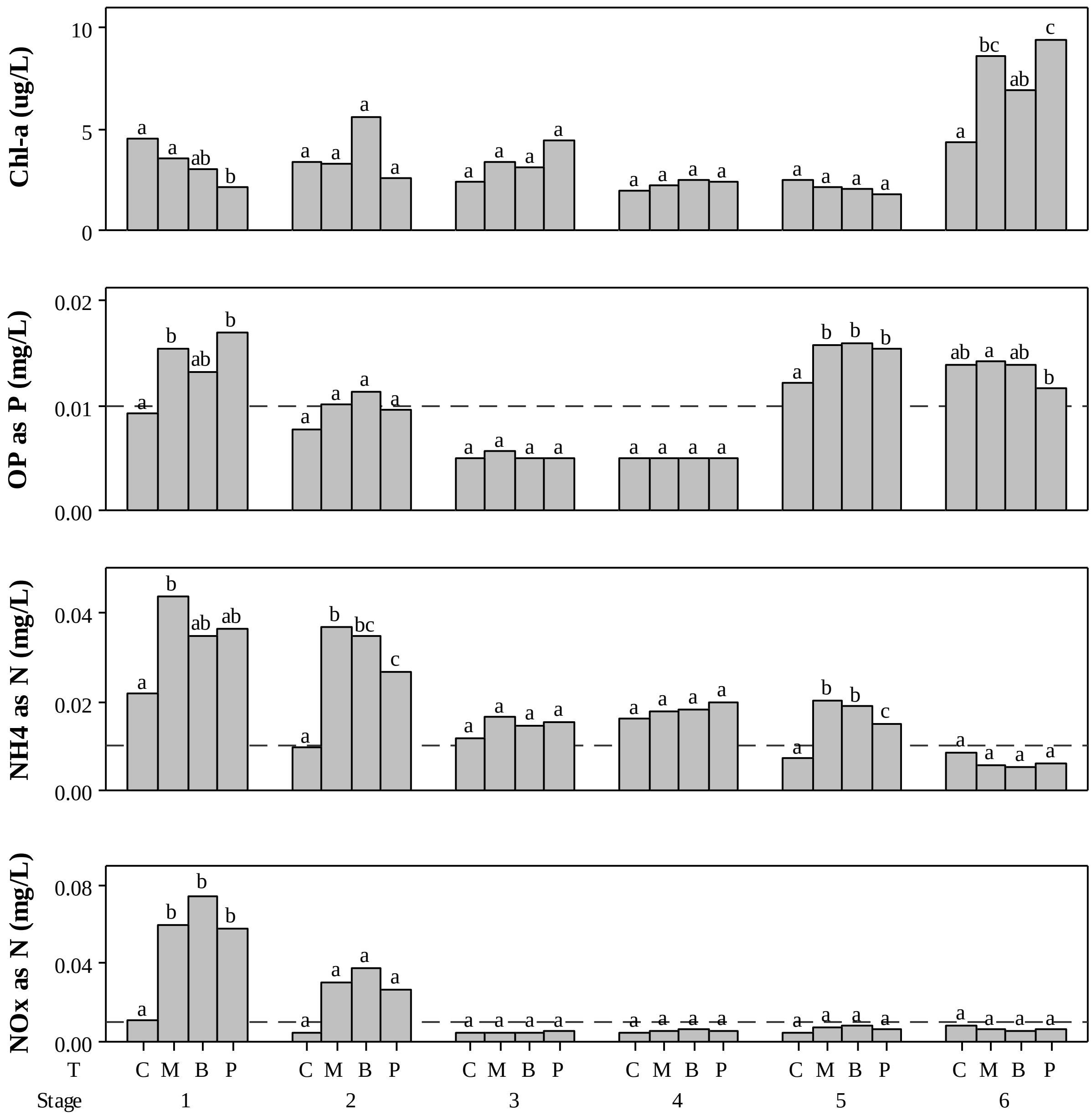

Figure 4-7. Mean final concentration on Day 7 of the four treatments (T) in each stage. Description of letters above each bar refers to Figure 4-5.

The dissolved nutrient concentrations, OP-P, NH_4^+-N, and NOx-N, of the FTW treatments were generally higher than those of the C treatment (Figure 4-7). The relatively abundant periphyton in the C treatment may more effectively utilize bioavailable nutrients than the FTW treatments (Section 4.4.6). Higher OP-P, NH_4^+-N, and NOx-N concentrations in Stage 1 and 2 than other stages were mainly due to the experimental water supplied from Ashby Pond. The initial concentrations on Day 0 were two to five times higher in Stage 1 and 2 than those in the other stages. The OP-P concentration was effectively reduced to below 0.02 mg/L in the four treatments throughout the experiment. As the average water temperature increased to approximate 25 ℃ between Stage 2 and 4 (Figure 4-5), the four treatments could further remove OP-P to lower than 0.01 mg/L.

The higher NH_4^+-N in the FTW treatments was likely produced through ammonification and reduced by volatilization, nitrification, and uptake (Kadlec and Wallace, 2009). The FTW treatments exhibited higher Org-N removal efficiency than the C treatment in Stage 1, 3c, and 4 (Section 4.4.3.1). As a result, the FTW treatments received more NH_4^+-N transformed from Org-N than the C treatment. Additionally, the physicochemical conditions of the C treatment may enhance the NH_4^+-N reduction. Ammonia volatilization is positively correlated with pH and temperature, while DO and temperature contribute to nitrification (Kadlec and Wallace, 2009; Zimmo *et al.*, 2004). The C treatment exhibited significantly higher values of pH, DO, and temperature than the FTW treatments in all stages (Figure 4-5). The differences between the three FTW treatments may be caused by the uptake effects and nitrification. The P treatment significantly reduced NH_4^+-N concentration in Stage 2 and 5 ($p < 0.01$) compared to M, while no differences between the two treatments in other stages. The pickerelweed exhibited robust growth and develop vigorous and flashy roots, which provide a large submerged surface area to support the growth of nitrifiers (Song *et al.*, 2011). The active nitrifier bacteria may result in significantly lower pH of the P treatment than M ($p < 0.1$) as discussed in Section 4.4.1.

The comparison of NOx-N final concentration between the four treatments is unclear as most of the values were less than the detection limit (0.01 mg/L) from Stage 3 to 6. The initial concentrations were the major cause rather than the high removal efficiency. Seventy-five percent of the NOx-N concentration on Day 0 was lower than the detection limit in this four-stage period. The comparisons in Stage 1 and 2 illustrates that NOx-N in the FTW treatments were higher than those in the C treatment. This may be due to nitrification that converts NH_4^+-N to NOx-N or periphyton effects (Section 4.4.6). The assumption of the ammonia transformation is supported by the observation of lower DO and pH in the FTW treatments as the nitrification process consumes oxygen and release of protons. Denitrification is not considered in our experiment as the DO was generally higher than 5 mg/L. The optimum condition for denitrification in ponds is DO < 1 mg/L (Shilton, 2005).

4.4.5 Removal rate constant (*k*) and temperature

The coefficients of the modified Arrhenius equation for FTW TP and TN removal rate constants (*k*) and corresponding statistical results are shown in Table 4-4. To facilitate comparison the reaction rates in our FTW experiment with those reported in constructed wetland studies, the daily volumetric reaction rate (*k*) in this study was converted to the yearly areal reaction rate (k_a) by dividing *k* with the water depth (0.32 m) and timing 365 days of a year. The calculated mean k_{a20} and θ of the three FTW treatments were 15.7 m/yr and 1.033 for TP and 8.55 m/yr and 1.054 for TN, respectively. The results falls within ranges of published constructed wetlands values (Kadlec and Wallace, 2009). The median (range) of the constructed wetlands k_{a20} are 18 (0-96) for TP and 21.5 (4-115) for TN, while θ are 1.006 (0.852-1.086) for TP and 1.056 (0.953-1.130) for TN, respectively. The coefficients of determination (R^2) of the M, B, and P modified Arrhenius equations are 0.35, 0.35, and 0.43 for TP and 0.32, 0.21, and 0.30 for TN. All regression assumptions were satisfied except the residual normality tests of B and P in the TN equations. Within 72 data points in each treatment, only one sample of B and two of P TN removal rates deviated from their population due to their lower values. Therefore, the violation of this assumption is considered acceptable.

Table 4-4. Temperature coefficients of modified Arrhenius equation for TP and TN removal rate.

Parameter	**Treatment**	k_{20} **(1/day)**	k_{a20} **(m/yr)**	**θ**	R^2 **(%)**	**Statistical assumptions test**
TP	M	0.129	15.3	1.031	35.4	Pass
	B	0.135	15.9	1.030	34.8	Pass
	P	0.136	16.0	1.038	42.7	Pass
TN	M	0.073	8.6	1.049	32.3	Pass
	B	0.077	9.1	1.042	20.7	P-Pass*
	P	0.067	7.9	1.070	30.3	P-Pass*

* Residual normality tests failed due to one and two outliers in total 72 samples of the B and P treatment, respectively. Excluding these would result in a pass of the normality test.

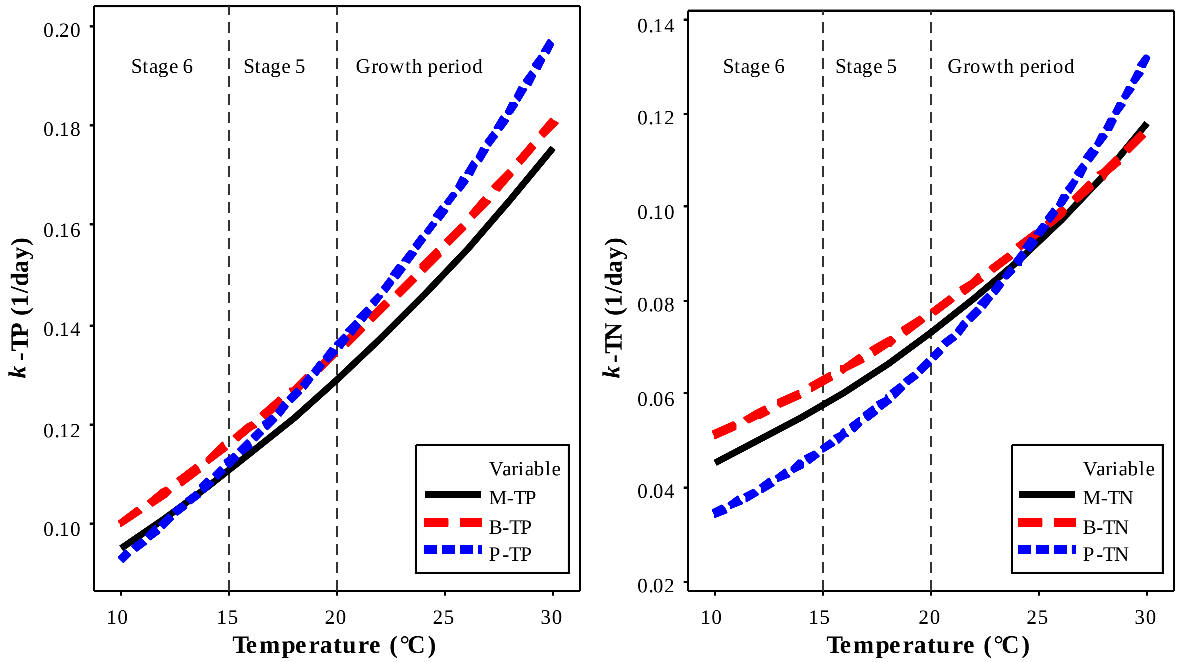

Figure 4-8. The relationship between the temperature and the three FTW treatment first order removal rate (*k*-TP & *k*-TN) described by modified Arrhenius equations.

The relationship between FTW TP and TN reaction rates (*k*) and temperature can be described by the modified Arrhenius equation (Figure 4-8). The θ value in the modified Arrhenius equation indicates the sensitivity of the reaction rate constant to changes in temperature. The pickerelweed treatment has the highest θ for both TP and TN followed by the M treatment. Therefore, the TP and TN reaction rates (*k*) of the P treatment are more sensitive to the temperature variation than those of the M and B treatment (Figure 4-8). The curve of the P treatment in *k*-TP is intercepted by M and B at 15 and 20 ℃, respectively. These interactions may indicate that the pickerelweed is functioning when temperature is above 15 ℃ (compared to the M) and is outperforming the softstem bulrush when temperature is higher than 20 ℃ in the case of phosphorus treatment. The relationship of *k*-TN is B>M>P at lower temperatures (<15 ℃); however, the differences fade and eventually reverse as the temperature increases. The accuracy of the Arrhenius equation prediction is compared with the statistical analysis results (Figure 4-6). The stages are marked on the Figure 4-8 with corresponding temperature regimes. The comparison between the two methods shows that M, B, and P nutrient treatments are more consistent with the Arrhenius equation at higher (>25 ℃, Stage 3) and lower (<15 ℃, Stage 6) temperatures. The modified Arrhenius equation explains 35-43% and 21-32% of the variance of the TP and TN reaction rates, respectively (Table 4-4). Other factors would need to be considered to better predict FTW performance in the moderate temperature regime. However, the modified Arrhenius equation effectively simplified the temperature effects on the FTW TP and TN treatment. These results are consistent with an FTW experiment conducted by Van de Moortel *et al.* (2010), who suggested that temperature was the critical factor controlling the performance in their FTW experiment.

4.4.6 Shading effect and periphyton

The shading factor experiment provided data to clarify the compartmental contribution of the M, B, and P in our experiment (Table 4-5). The pH, DO, and temperature of the non-shaded treatment (NS) was significantly higher than shaded one (S, p-value < 0.01). Because this result is identical to the six stages of the FTW experiment (Section 4.4.1), it suggests that blocking the sunlight was a major factor of lower pH, DO, and temperature in the FTW treatments compared to the C treatment. Additionally, the periphyton on the bottom of NS treatment significantly reduced dissolved nutrients, OP-P and NH_4^+-N, in comparison to those in the S treatment (p-value < 0.05). The NOx-N constituent was not significant because of the low initial concentration factor as discussed in Section 4.4.4. Contrary to the dissolved nutrients, the differences of TP, TPP, TN, and Org-N concentrations between the S and NS treatment were not significant. Based on the results here and in Section 4.4.3, periphyton on the bottom of the NS (and C) treatment considerably improved dissolved nutrient treatment compared to the S treatment.

Chl-*a* concentration was higher in the shaded tanks (p-value = 0.08), which is in opposition to one of our hypotheses. It is suggested that sunlight reduction in S treatment may not effectively limit algal production with 61% coverage. The nutrient competition described in Section 4.4.4 was considered as the major factor of this unexpected result. While shading inhibited the growth of the periphyton on the bottom of the experiment tank (S), the remaining nutrients utilized by the suspended algae, which proliferated by using sunlight from 39% uncover area. Therefore, the shading effect with partial coverage of water bodies may limit the growth of periphyton at the bottom and result in higher Chl-*a* concentration of the applied water bodies. However, it should be pointed out that the reduction of the attached biofilm growth due to the shading effect may not be replicated at the field scale. Periphyton is present in the littoral zone of lakes/ponds but absent from the bottom (Welch and Lindell, 1992). The sunlight in most ponds is prevented from reaching the bottom because of the obstruction by high turbidity and the attenuation of water depth. Therefore, anchoring FTWs at correct locations may not reduce periphyton performance since the establishment of such benthic mechanism is prevented at the dark sediment surface. Secchi disks used to measure water transparency may be a suitable tool to decide the appropriate location. However, the result of the shading effect also suggests that the FTW can limit the productivity of the submerged vegetation and relocate the nutrient removal mechanism from under to above the water surface, perhaps a more manageable space. Based on the periphyton effect found in this study, the accurate definition of the C treatment could be a mesocosm of a shallow urban retention pool with 0.32 m depth. The setting of the S treatment with limited periphyton growth on the tank bottom may create an environment more similar to real ponds than the design of the C treatment.

Table 4-5. Medians of physicochemical characteristics (n=32) and nutrient concentrations (n=16) on Day 7 of the two treatments in the shading effects experiment (Stage 5 and 6). Results of the shaded (S) and non-shaded (NS) treatments comparison by two-sided paired sample Wilcoxon signed rank test results are shown in p-values.

Parameter	**Shaded (S)**	**Non-Shaded (NS)**	**Comparison between the S and NS (*p*-value)**
TP (mg/L)	0.07	0.07	0.41
OP-P (mg/L)	0.02	0.01	0.05^{**}
TPP (mg/L)	0.05	0.05	0.70
TN (mg/L)	0.72	0.76	0.80
NH_4^+-N (mg/L)	0.009	0.006	0.02^{**}
NOx-N (mg/L)	0.01	0.01	0.36
Org-N (mg/L)	0.71	0.75	0.78
Chl-*a* (µg/L)	2.89	2.80	0.08^{*}
pH	6.83	7.12	0.00^{***}
DO (mg/L)	8.80	9.31	0.00^{***}
Temperature (℃)	16.10	16.78	0.00^{***}
Conductivity (us/cm^{c})$^{\delta}$	84.50	85.00	0.85

$^{\delta}$ Standardized to 25℃.
$^{*}\alpha = 0.1$, $^{**}\alpha = 0.05$, $^{***}\alpha = 0.01$.

4.4.7 Periphyton outbreak

Periphyton grew on the experiment tank surface considerably enhanced nutrient removal efficiency of the C treatment, especially in Stage 2 and the first week of Stage 3 (five weeks). This unexpected microbial community in the C treatment accumulated during Stage 1 and developed into significant proliferation of biofilm at the beginning of Stage 2 according to our field observation. The lack of this biofilm in M, B, and P may due to shading effect (Frost *et al.*, 2002; Welch and Lindell, 1992), which was evaluated in our experiment during Stage 5 and 6 (Section 4.4.6). Because the periphyton outbreak only existed in the control tanks, it created an inconsistent environment between the C and three other treatments in the FTW experiment. Therefore, the C treatment is not suggested as a control for the FTW treatments during this five-week period. Regular cleaning of all tanks began at the end of Week 9 to remove the biofilm on the tank surface. A significant change was observed in our data reflecting the functional restoration of the C treatment. The C treatment exhibited significantly higher nutrient removal efficiency in the five-week periphyton affected period; with regular cleaning since Week 9, the performance of the C treatment was restored to its original condition at the beginning stage of the experiment. As shown in Figure 4-6, the continuously higher FTW TP and TN removal efficiency in Stage 3c and 4 than the C treatment was the result the restoration of the homogeneous environment between the four treatments. Similar situation was reported in a study published while our experiment was in process. Chang *et al.* (2012) stated that algae dominated their control treatment, resulting in significant higher nutrient removal than FTW treatments.

4.4.8 Macrophyte growth

The macrophytes were harvested and their morphology characteristics were measured after growing in the greenhouse for 28 days in the six stages. The mean ± SD of the pickerelweed primary root, stem, and leaf length (cm) were 54.3±16.5, 21.1±11.7, and 5.3±3.3, respectively. For the maximum length of the three parts, the values were 85, 52, and 14 cm. The primary root growth was the most significantly in Stage 4 with an average increase of 36.9 cm and followed by Stage 2 (28.2 cm), Stage 3 (27.1 cm), Stage 1 (18.6 cm), Stage 5 (9.8 cm), and Stage 6 (0 cm). Stage 1 to 4 is the growing season (May to August) when the plants are generally more active comparing to fall (Stage 5 and 6). The average primary root length in this study was a multiple of two to four times the values reported in another FTW experiment, which used polluted river water with 0.71 mg/L TP and 23.13 mg/L TN, for the same plant species (Zhao *et al.*, 2011). The concentrations are about five and 20 times higher than the Ashby Pond water, respectively. Because of the limited nutrient supply, pickerelweed in our experiment may allocate more biomass to below-ground tissue to forage needed resources. Brouwer (1961) found shoot growth decreases whereas root growth remains constant when nitrogen supply was interrupted. Additionally, Brix *et al.* (2010) tested *Typha domingensis* and *Cladium mariscus* spp. *jamaicense* with two levels of phosphorus input as a variable. Their results indicated that the fraction of the root biomass of both species was higher in the group with lower phosphorus concentration. Thus, the observed results in our study appear consistent with previously reported FTW observations.

Leaves grown from lateral buds on the pickerelweed rhizomes in Stage 6 are classified as below-ground parts and referred as “submerged leaves” hereafter (Figure 4-9). These end season submerged leaves did not grow into the size of regular shoots and stayed below the water surface and the growth media through the winter of 2012. It is based on the observation of the pickerelweed growing on a demonstration FTW installed at the same time with the in-pond FTW cultural system (Section 4.3.3.3). Because the above-ground part is defined as harvestable tissue, such as stems and leaves, these submerged leaves with the rhizomes and roots were categorized as below-ground parts in the next section.

The growth rate of the pickerelweed in each stage is compared with the accumulated phosphorus and nitrogen removal from water. The result facilitates the evaluation of the FTW performance may also be possible based on field observation as an alternative qualitative method to laboratory analyses. As shown in the Figure 4-10, the relative growth rate (*RG*) of whole, above-ground, and below-ground plant parts exhibited similar variation through six stages. The values were highest in Stage 1 (May), gradually decreased, and became negative in Stage5. The above-ground growth rate may be a better indicator of the pickerelweed performance in term of TP than TN removal from the experimental water. Figure 4-11 describes the weekly accumulated phosphorus and nitrogen mass removal from the experimental water in the P treatment after normalized to the M treatment. It describes the net effect of the existence of pickerelweed, which include plant uptake and associated microorganism activities in the FTW. As the above-ground relative growth rate (*RG*) reduced to lower than zero, the accumulated phosphorus removal by the pickerelweed (Figure 4-11) also reached the maximum value in Stage 5. For accumulated nitrogen removal, the values steadily decreased starting from Stage 4 when the whole plant relative growth rate approaches to zero.

In contrast to the pickerelweed, the observed growth of the softstem bulrush was insignificant in the six stages. Growth rates could not be evaluated appropriately because of the uncertainty inherent in the initial biomass estimate. Thus, further discussion of relative growth performance of the softstem bulrush is omitted. The softstem bulrush may not be suitable according to our data in the first year (six stages, 2012). However, a field observation of a demonstration FTW on September 11, 2013 (the second year) found that the maximum height of the softstem bulrush was about 170 cm, while the value was 70 cm in 2012. The demonstration FTW was installed at the same time with the in-pond FTW cultural system (Section 4.3.3.3). Based on the observation in 2013, multi-year monitoring may be required to reveal the long term performance of the softstem bulrush on the FTW in urban wet ponds.

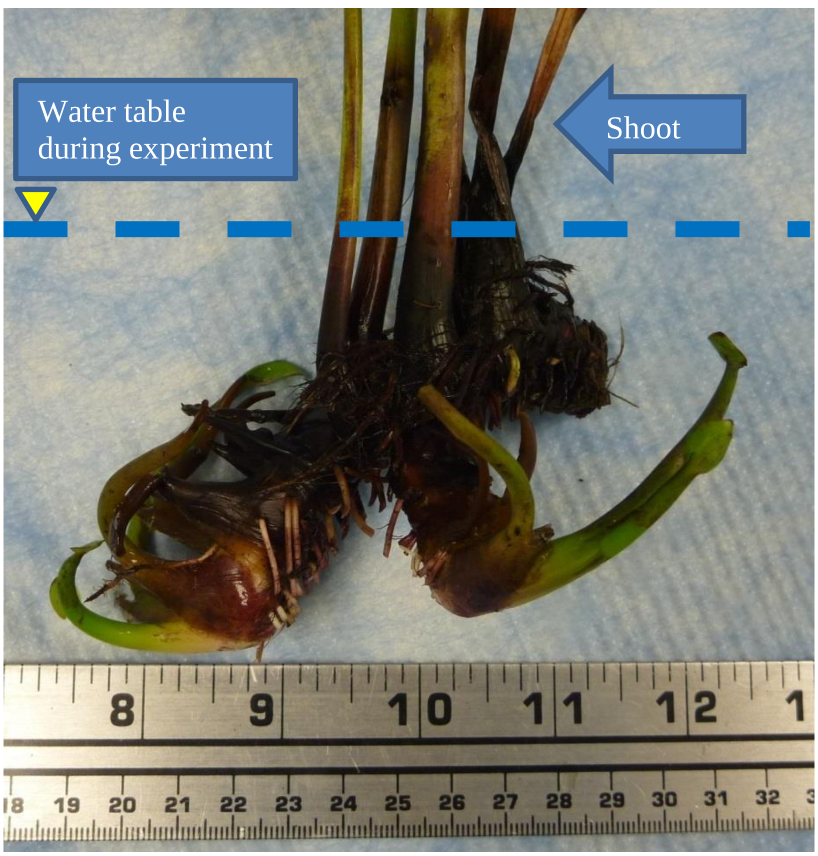

Figure 4-9. Typical pickerelweed submerged leaves emerged from lateral buds on the rhizome in Stage 6 (October). The roots were removed. The leaves stayed below water surface throughout the winter of 2012. The observation was based on the pickerelweed grown on a demonstration FTW installed as the research project begin.

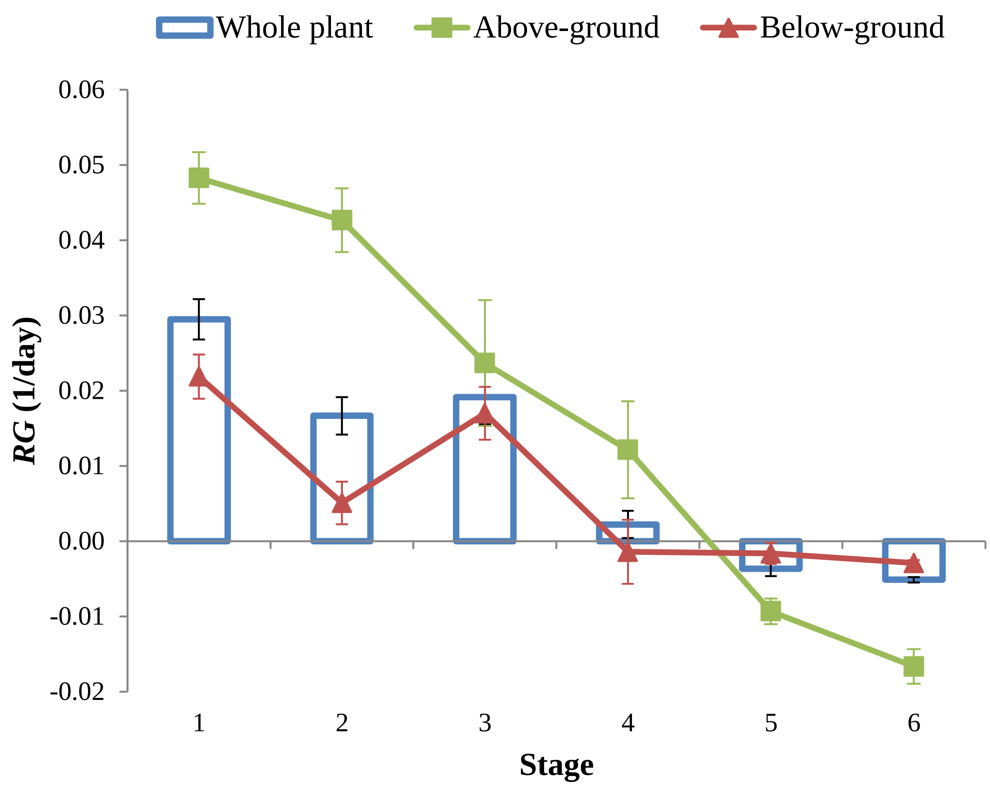

Figure 4-10. Pickerelweed biomass (dry weight) relative growth rate (Mean±SD, *n*=3) of whole plant, above-ground (stem+leaf), and below-ground (rhizome+root+sprout) tissue.

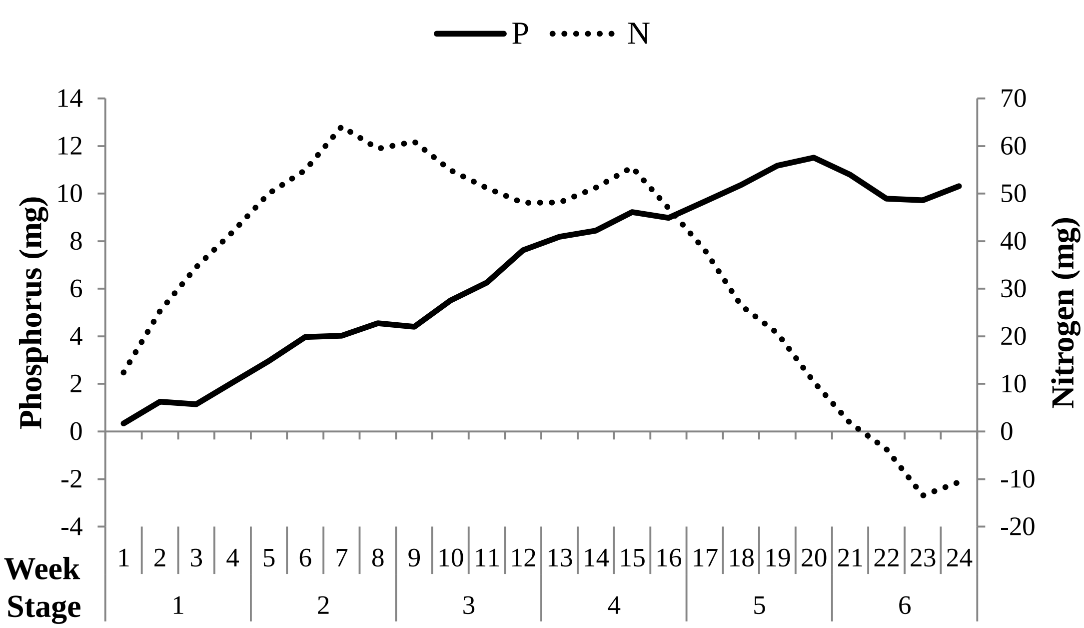

Figure 4-11. Accumulated phosphorus and nitrogen mass removal from water. The values were normalized to the M treatment to reveal net effects of the pickerelweed in the FTW system.

4.4.9 Phosphorus in macrophyte and removal through plant harvest

Phosphorus concentrations (%) in each parts of the whole plant varied during the ontogenesis of the pickerelweed and the softstem bulrush at different life stages and influenced by limited nutrients supply (Figure 4-12). The variation of the phosphorus content in the individual parts, such as the shoots and roots, were similar for both the pickerelweed and the softstem bulrush. The value of the shoot was highest in Stage 1 (May) and steadily decreased to Stage 6 (October). Both below-ground parts, rhizomes and roots, had similar trend in that their phosphorus concentration were higher at initial and final experimental stages and reached the lowest values in Stage 4. For the pickerelweed, the submerged leaves in Stage 6 had a value of 0.12 %, which is similar to that of the shoots at the beginning of the experiment (Stage 1, 0.16%).

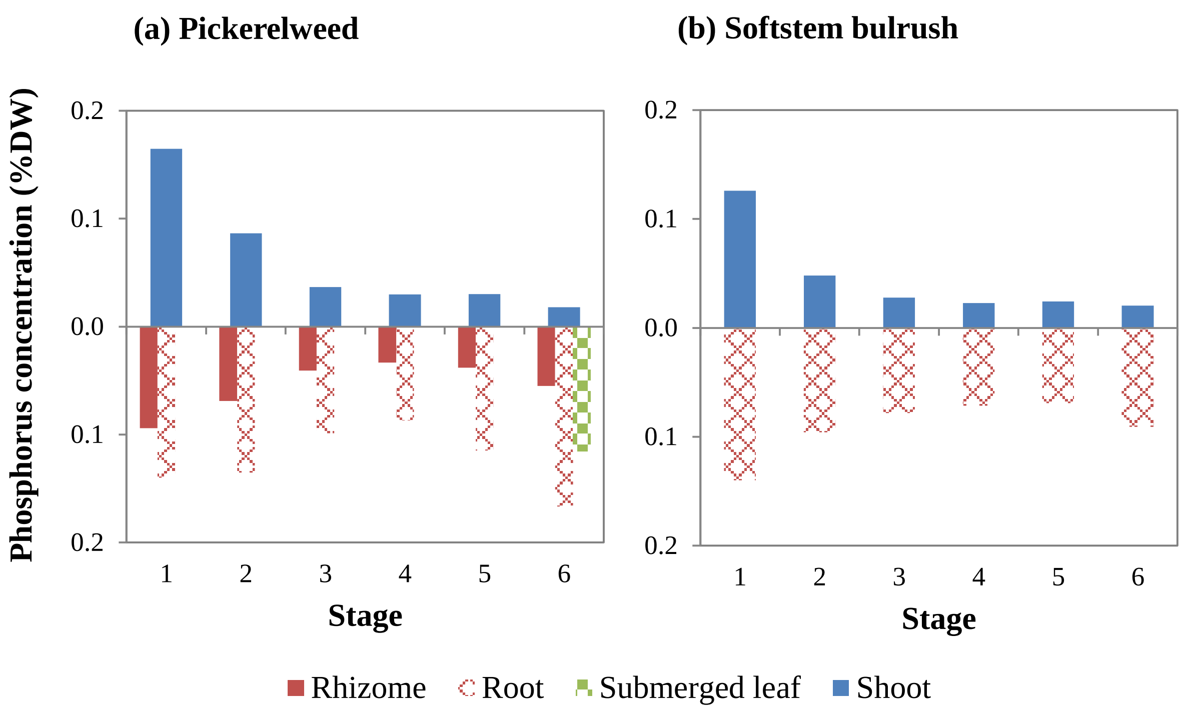

Figure 4-12. Typical dry biomass phosphorus concentrations of (a) Pickerelweed and (b) softstem bulrush in shoot, rhizome, root, and submerged leaf after 28 days mesocosm experiment in each stage (*n*=1). Values were evaluated from composite samples of three replicate tanks (nine plants). The submerged leaves in Stage 6 is classified as below ground part as described in Section 4.4.8. (DW: dry weight)

As shown in Figure 4-12, the phosphorus concentration variation of both species can be separated into two phases: resource use (Stage 1 to 4) and resource remobilization (Stage 5 and 6). During Stage 1 to 4, limited nutrients supply in the experimental water is considered to be the controlling factor in the change of the plant tissue phosphorus concentration. The reduction of the pickerelweed whole plant phosphorus concentration was from 0.13 to 0.04 % during Stage 1 to 4. The measured tissue phosphorus concentrations (%) were lower than those in other FTW studies with higher nutrient sources. Previous studies show that the pickerelweed phosphorus concentrations were 0.4 to 0.5 % (Chen *et al.*, 2009), 0.1 to 0.4 % (Zhao *et al.*, 2011), 0.4 % (Zhao *et al.*, 2012a), and 0.1 to 0.2 % (Winston *et al.*, 2013). Their corresponding experimental water TP concentrations (3.1 to 0.18 mg/L) were 20 to three times higher than our experimental water (0.12 to 0.05 mg/L) from the urban wet pond. Lorenzen *et al.* (2001) reported that the tissue phosphorus contents of *Cladium jamaicense* and *Typha domingensis* is correlated with the phosphorus source concentration. They concluded the stress resulted in higher resource use efficiency in both tested plant species. The pickerelweed and the softstem bulrush in our experiment may experience similar stress as observed in Lorenzen *et al.* (2001), and thus increase resource use efficiency from Stage 1 to 4 continuously as the plants create more dry mass per unit phosphorus absorbed.

The trend of phosphorus content variation was changed when the experimental water temperatures were lower than 20℃ in Stage 5 and 15℃ in Stage 6 (resource remobilization phase). The shoot phosphorus concentrations continuously decreased while the values in the below-ground tissues simultaneously increased. The plants responded to this external environment change by translocated more nutrients to below-ground storage. Nutrients are remobilized from mature shoots to rhizomes and roots, while nutrient supply to the above-ground tissues may be interrupted at this stage (Marschner, 1995). Most nutrients were stored in the under-ground tissue during this period, likely for reproduction in the next year (Ruiz and Velasco, 2010b). This biological reproduction strategy may cause inaccurate plant nutrient removal estimation in vegetated pollution control measures, like constructed wetlands and FTWs, through plant areal part harvest and measurement, if done in different stages of plants' life cycle. Therefore, the whole plant harvest method in our experiment could provide more complete information about the nutrient distribution within the plants, which can be used for developing FTW management strategies, such as aerial partial harvesting.

Nutrient removal through plant aerial parts harvest could be optimized with the knowledge of the nutrient distribution in the above- and below-ground tissues instead of simply evaluating the plant biomass. As previously discussed, plant above- and below-ground tissue growth rates and phosphorus concentrations changed with time and environmental conditions, such as temperature. The maximum amount of the harvestable aerial plant biomass may not be equated with the highest removable phosphorus mass because the phosphorus distribution in plants varies through their life cycle for a variety of reasons, many specific to each species. Figure 4-13 shows biomass (dry weight) distribution in different parts of the pickerelweed and softstem bulrush in the six experimental stages. The pickerelweed had the highest biomass of the shoots in Stage 3 and 4; however, the aerial tissues with the most harvestable phosphorus mass in the experimental period were in Stage 1 and 2 (Figure 4-14). Although the nutrient uptake continued, 97% of the phosphorus mass was stored in the below-ground parts in Stage 6. For the softstem bulrush, the phosphorus mass in the shoot was relatively steady throughout the experiment compared to the pickerelweed. The most absorbed phosphorus was stored in the roots, 90% in Stage 6.

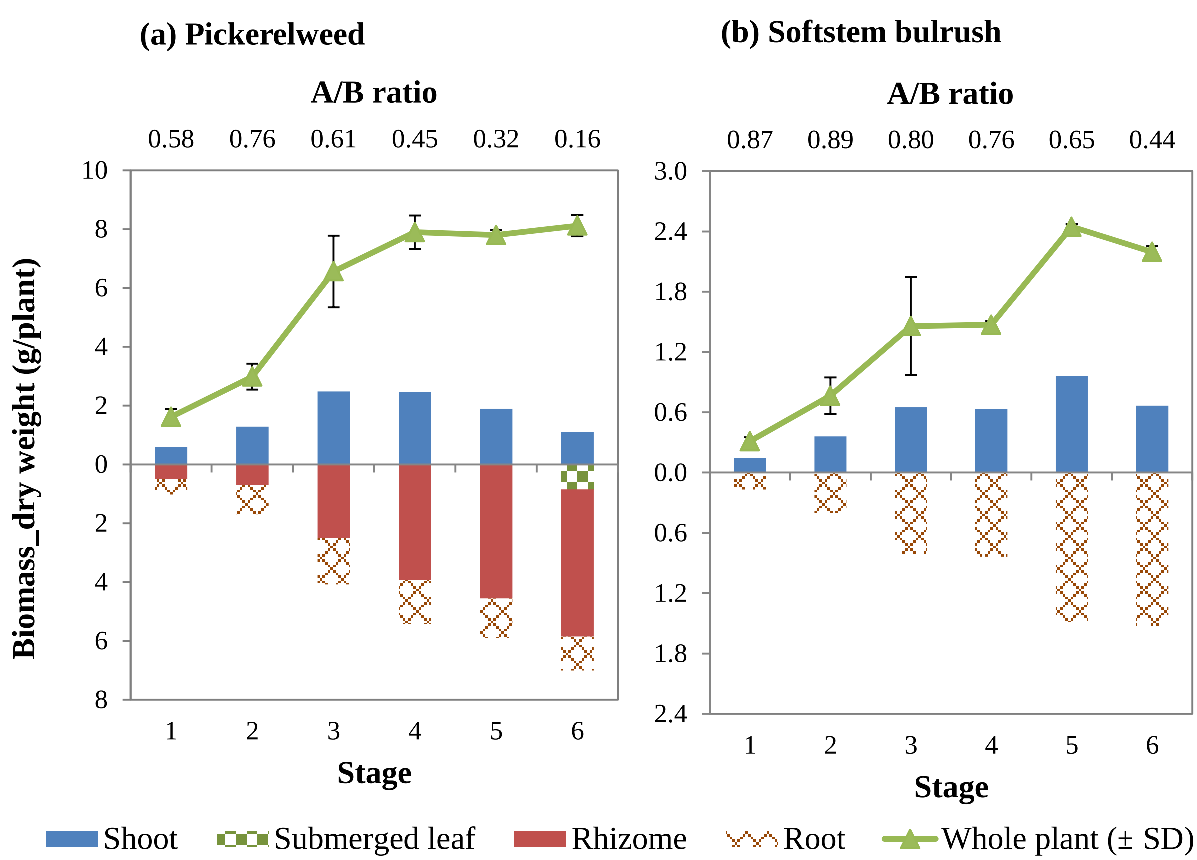

Figure 4-13. Biomass (dry weight) distribution in the (a) pickerelweed and (b) softstem bulrush after 28 days mesocosm experiment in each stage (n=3). A/B ratio: above-ground/below-ground ratio. Values were evaluated from three replicate tanks (nine plants). The submerged leaf in Stage 6 is classified as below ground part as described in Section 4.4.8.

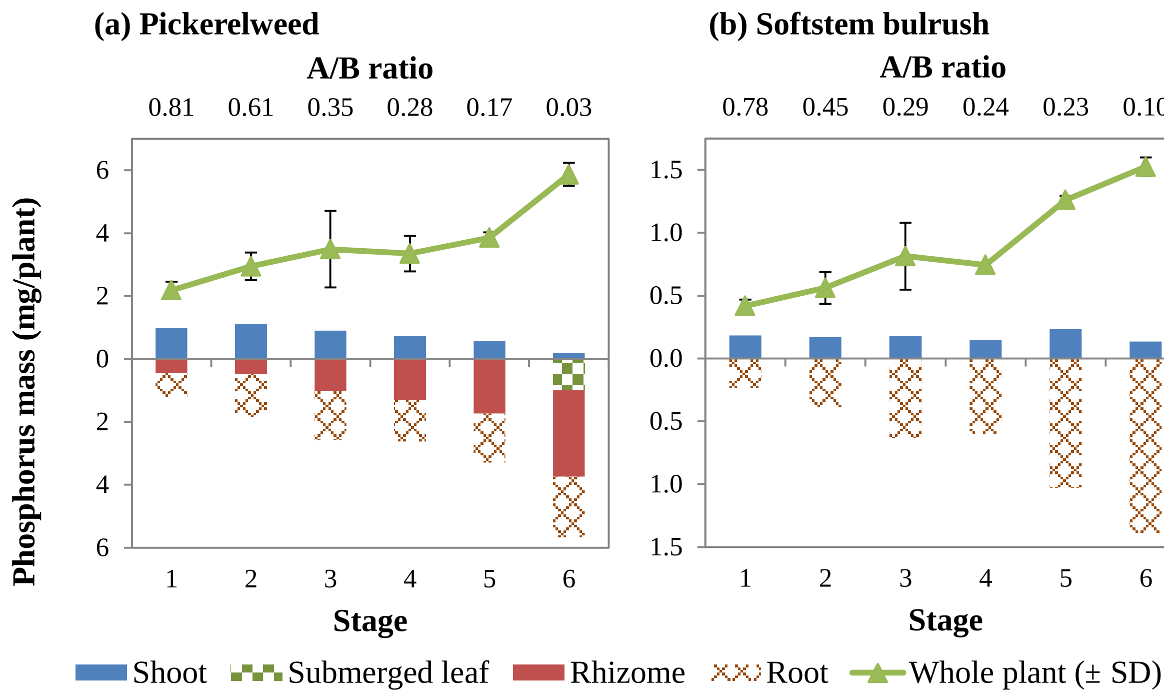

Figure 4-14. Phosphorus mass distribution in the (a) pickerelweed and (b) softstem bulrush after 28 days mesocosm experiment in each stage (*n*=3). A/B ratio: above-ground/below-ground tissue ratio. Values were evaluated from three replicate tanks (nine plants). The submerged leaf in Stage 6 is classified as below ground part as described in Section 4.4.8.

According to the result in Figure 4-14, the aerial parts of the pickerelweed should be harvested in early growing season (Stage 2, June) with lower biomass but high nutrient contents. A second harvest could be applied again at the end of the growing season (Stage 5, September) as discussed in Section 4.4.3.3. Recommendation for the softstem bulrush harvest is unavailable based on the result here. Marschner (1995) described shoots act as nutrient sink and contain most mineral nutrients in vegetative stage. In Singapore, *Chrysopogon zizanioides* and *Typha angustifolia* above-ground tissues were tested with repeated harvests within a year. Chua *et al.* (2012) found the phosphorus and nitrogen content in the shoot tissues increased immediately after each harvest and decreased when the shoots reached maturity. Therefore, the shoot should be removed before the end of the growing season to retrieve higher amount of the nutrients from water bodies. However, this strategy does not consider potential health damage to the macrophytes. In comparison with tropical environments in Singapore, plants in temporal region have a limited window to store nutrients. In Japan, Nakamura and Shimatani (1997) found shoot harvesting too early may cause lower nutrient removal efficiency of their FTW in the following year. This may be due to inability of the plant to remobilize nutrients from the stems to the storage organs, which support new sprouts growth in new growth season. There is an alternative approach that neglects the nutrient distribution in the plants and the reproduction problem mentioned above. The harvest would be conducted at the end of the growing season (Stage 6) if whole plant removal is the case instead of the aerial tissues only. The method is suitable for FTW designs with replaceable growth media and separated floating structures, such the FTW used in our study.

4.5 Conclusion

The application of FTWs in retention ponds to control urban nonpoint source pollution was evaluated in our study. Our five hypotheses were tested and supported by the findings except one: shading effect on the Chl-*a* concentration. The average TP and TN concentration of the studied retention pond water was 0.15 and 1.19 mg/L, which is considered to be at the lower end of expected concentrations and poses a great challenge in evaluating FTW treatment effectiveness. The results indicate all three FTW designs, planted (with two species) and unplanted floating mats, significantly improved phosphorus and nitrogen removal efficiency of a simulated urban retention pond during the growing season, which was May through August. With a retention time of seven days, 66-69 % of TP and 48-50 % of TN reduction were achieved in the three FTW treatments. Compared to the control treatment, the removal efficiency was enhanced by 8.2 % for RE-TP and 18.2 % for RE-TN in the planted FTW treatments. Organic matter decomposition driven by the microbes attached on the floating mats and plant root surface is likely the primary mechanism of nutrient removal by FTWs in urban retention ponds. Comparing the results of the four treatments, the FTWs planted with pickerelweed had the highest RE-TP ($p < 0.1$), and behaved similarly with other two FTW treatments, B and M, in the case of TN during the growth period. Aesthetic merit may also be a reason for using pickerelweed.

Describing FTW TP and TN reaction rates through the modified Arrhenius equation effectively simplified the temperature effects and provided information on possible management strategies. The temperature effect described by the modified Arrhenius equation revealed that pickerelweed is most sensitive to the temperature variation and provides considerable phosphorus removal when water temperature is greater than 25℃. However, the effectiveness of this plant species may be negligible at water temperature below 15℃. The removal rate of the softstem bulrush was more constant throughout this study but less effective than the pickerelweed regarding the phosphorus removal. The information provided by the modified Arrhenius equation demonstrated a great approach on plant species selection based on the most available weather data, temperature, when designing an FTW system.

The phosphorus content and distribution of the pickerelweed and softstem bulrush was influenced by the limited nutrient supply in the mesocosm experiment and water temperatures. The phosphorus use efficiency of the above- and below-ground tissues increased from Stage 1 to 4 that more tissues were produced per unit weight of absorbed phosphorus, i.e. plant phosphorus content decreased. As the experimental water temperatures decreased to below 20℃ after Stage 5, phosphorus was remobilized from the shoots to the storage organs, rhizome and root. Therefore, the aerial tissue harvest should be conducted in June (Stage 2) to remove most plant absorbed phosphorus before they were remobilized to below-ground tissues. Additionally, a second harvest in September could prevent nutrient release from senesced biomass according to data from water analysis. However, the potential damages on plants when harvest in June was not tested in our experiment. Harvesting at different points across a plant life cycle needs further research, to optimize nutrient removal, as well as to protect plant health.

This study evaluated the benefits of FTW utilization but also revealed the potential limitations that require extra consideration when applying this system. TP and TN removal efficiency using water from urban retention ponds was improved. The shading effects of the FTW with 61% coverage could significantly lower water temperatures and reduce thermal pollution. However, the plants and associated microorganisms on the floating mats may reduce pH and DO of the applied water bodies. Additionally, it is suggested that FTW should be anchored in the deep water regime and avoid the littoral zone where benthic periphyton inhabit. This attached-to-substrate microorganism community can provide nutrient treatment benefits and is driven by sunlight, which is blocked when FTWs are presented. This extra consideration may be ignored if the water turbidity is high and benthic periphyton activity is likely insignificant. FTW shading may be utilized to control the productivity of submerged vegetation if such a problem exists.

It should be noted that the finding of this study are restricted to shallow urban retention pond (0.32 m) with significant FTW coverage (61%) and low plant density (10.4 plants/m^2). The experimental tanks used in this study were relatively small and may magnify the FTW treatment effects. The sediments were also absent in experimental tanks to simulate adsorption/release of nutrients in retention ponds. Several uncertainties remain unaddressed in this study. First, FTW performance in the non-growing season was unknown. While the P treatment significantly improved RE-TP of the M from Stage 1 to 5, the performance faded in Stage 6. The exportation of nutrients from the plant tissue in winter is possible but unknown. Second, changes in FTW coverage may alter physicochemical attributes of water bodies with FTWs installed. As FTW coverage increases, more nutrients removal is expected. However, results of the associated impacts, such as reductions in pH and DO, require further investigation. Last, appropriate design of a control treatment to evaluate the FTW contribution in urban retention ponds is critical. An experimental tank with external shading is suggested to serve as a control treatment simulating the nutrient treatments of the retention ponds. The benthic periphyton in our C treatment may provide extra mechanism, which is absent in most urban retention ponds. Therefore, the magnitude of the FTW contribution may be underestimated when compared to the non-shaded control treatment, representing shallow urban retention ponds. As knowledge accumulates on FTW as a treatment BMP, the application of the system in retention ponds and other water bodies may provide an alternative solution to address non-point source pollution at watershed scales, such as within the Chesapeake Bay watershed.

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Chapter 5. Floating Treatment Wetland Nutrient Removal through Vegetation Harvest and Observations from a Field Study

5.1 Abstract

Nonpoint source pollution from urban areas has been identified as a leading contributor to impaired water quality. Floating treatment wetlands (FTW) are cultivated plants growing on floating mats in open water. FTWs can be used to remove pollutants from runoff, but data on their effectiveness is limited. We conducted a field study of FTWs in an enriched urban wet pond to investigate vegetation biomass and phosphorus (P) accumulation/distribution, sustainability under ice encasement stress (which is a concern in temperate regions), and to assess the use of the FTW by species. Planted perennial macrophytes successfully adapted to stresses of the low dissolved oxygen (DO) concentrations (minimum: 1.2 mg/L) in summer, ice encasement in winter, and relatively low nutrient concentrations in the water (median: 0.15 mg/L TP and 1.15 mg/L TN). The P content of the whole plant ranged from 0.06 to 0.24 % dry weight (DW) of the pickerelweed (*Pontederia cordata* L.) and from 0.05 to 0.14 % DW of the softstem bulrush (*Schoenoplectus tabernaemontani*) sampled from May to October, 2012. Pickerelweed demonstrated higher P removal performance (7.58 mg/plant) than softstem bulrush (1.62 mg/plant). Based on the observed seasonal changes in biomass and phosphorus, we recommend harvest of above-ground vegetation is conducted twice a year: in June for maximum P removal and in September to prevent P release due to senescence. Submerged tissues of pickerelweed, softstem bulrush, and yellow iris (*Iris pseudacorus*) survived ice encasement and reproduced in 2013. Additionally, plant diversity increased during the study period through recruitment of both native and exotic wetland plants. Systematic observation of wildlife activities indicated eight classes of organisms inhabiting, foraging, breeding, nursing, or resting in the FTWs. This study suggests above-ground plant harvest can enhance P removal and that softstem bulrush, yellow iris, and pickerel weed can be sustained over winter on the FTW. Future study is recommended to document the possible ecological benefits created by the use of FTWs.

Keywords

Phosphorus removal, Sustainability, Urban wet pond, Pickerelweed (*Pontederia cordata* L.), Softstem bulrush (*Schoenoplectus tabernaemontani*), Yellow iris (*Iris pseudacorus*).

5.2 Introduction

Nutrients, metals, and other pollutants are exported in stormwater from developed urban catchments. As nutrients, nitrogen (N) and phosphorus (P) are required by all living things for survival. However, excessive nutrients can damage biotic integrity and create eutrophication problems in rivers, lakes, estuaries, and coastal waters on a global basis (Anderson *et al.*, 2002; Dodds, 2010). Nonpoint source (NPS) pollution from urban runoff (including excessive nutrients such as N and P), is one of the largest uncontrolled sources of pollution to waterways and has been identified as a leading cause of impaired water quality and eventual decline in aquatic ecological health (Novotny, 2003). In order to mitigate NPS impacts in urban areas, the U.S. Environmental Protection Agency (US EPA) recommends the use of urban best management practices (BMPs), which use a variety of physical, chemical, and biological processes to restore receiving waters. Structural BMPs mitigate stormwater effects by settling and filtering pollutants before they enter receiving water bodies. Retention ponds (also called wet ponds) are one of the most common urban stormwater BMPs (Borne *et al.*, 2013; Winston *et al.*, 2013). Wet ponds are generally effective for coarse and/or heavy particles with attached pollutants, but are much less so for pollutants in dissolved form (Shilton, 2005). Implementation of stormwater BMPs normally requires the use of land, thus incurring an opportunity cost because other land uses are precluded. FTWs are a relatively new stormwater treatment practice that may enhance the effectiveness of retention ponds. FTWs consist of macrophytes growing on floating mats which can be deployed in many existing water bodies (Hubbard *et al.*, 2011). Nutrients and other pollutants are absorbed or filtered by the plants and attached periphyton, which are driven by solar energy through photosynthesis. Although new to management of water pollution, practices similar to FTWs have been used in aquaculture and agriculture for over 50 years in Asia (Li, 1957; Sidle *et al.*, 2007). The first documentation of an FTW-like system, called “floating field” was made in Taiwan, in the Year 1717 (Zhou, 1717). FTWs have been studied across the world with different plant species and environments, from tropical to temperate regions (Chua *et al.*, 2012; Headley and Tanner, 2012). While several studies have evaluated FTW effectiveness using agricultural wastewater or polluted surface water, only a few studies have documented FTW behavior with urban runoff (Headley and Tanner, 2012; Ladislas *et al.*, 2013).

FTWs may be uniquely sustainable and economical as a potential treatment practice with widespread applicability. This attribute of FTWs is due to the scalable nature, relative ease of construction, and potential utilization of locally available (wetland) plants and materials, such as man-made plastic bottles, and natural bamboo. Land acquisition and on site construction expenses associated with other structural BMPs are avoided. FTWs may enhance the performance of existing retention ponds and reduce NPS pollution in the urban areas without land acquisition (Headley and Tanner, 2012; Winston *et al.*, 2013). In addition to lower installation costs, the harvested biomass could provide economic benefits. Vegetation has been grown on FTWs or similar practice for animal or human consumption (De Stefani *et al.*, 2011; Li *et al.*, 2007a). For example, "floating gardens" have been used to cultivate tomatoes and potatoes since early 1960s in Lake Inle, Myanmar (Sidle *et al.*, 2007).

In order to assess the role of aerial tissue harvesting in pollutant removal, knowledge of the temporal variation of nutrient distribution in FTW plant tissues is needed. However, few studies have focused upon this attribute. Macrophytes adjust their biomass growth and nutrient distribution according to external conditions and growth stages. While wetland studies do provide data on vegetative behavior, the information may not adequately evaluate plant performance in soilless and low nutrient environment as is found in FTWs in urban ponds. Typical nutrient concentrations in urban stormwater ponds in the U.S. are low, which may affect plant growth. For example, FTW plants may allocate more resources to below-ground tissues, increase root length, and produce thinner roots to increase their food acquisition efficiency (Marschner, 1995; Williamson *et al.*, 2001). High nutrient use efficiency is another physiological response to low nutrient availability (Lorenzen *et al.*, 2001). As plants experience different stages in their lifespan, absorbed nutrients are remobilized and translocated to different parts of the plant (Marschner, 1995). In a constructed wetland, Meuleman *et al.* (2002) suggested that nutrient removal efficiency could be increased from 9% to 20% of TN and from 6% to 25% of TP by harvesting above-ground tissues in September instead of winter when most nutrients were translocated to the rhizome/root system. However, plant nutrient distribution is another research gap of FTWs, because whole plant harvest is not universally practiced. As the plants grow on the FTW growth media, such as coconut fiber or plastic foam, their roots and rhizomes are embedded in these materials and cannot be easily extracted for analysis (Winston *et al.*, 2013). Therefore, published FTW plant data are typically based on samples from aerial tissues only and sometimes from roots hanging under floating mats (Chua *et al.*, 2012; Tanner and Headley, 2011; Winston *et al.*, 2013).

We investigated FTW nutrient removal performance and the distribution of P within FTW macrophyte tissues over time in an urban wet pond in a temperate region with stormwater as a source water input. Additionally we systematically observed use of the deployed FTWs by wildlife. The objectives of this study were to (1) evaluate P uptake by two native wetland plants and assess the temporal variation of P distribution within the plants, (2) investigate the sustainability of the perennial macrophytes on the FTW, (3) assess the use of the FTW by species, and (4) provide recommendations for management strategies such as harvesting to optimize nutrient removal.

5.3 Materials and Methods

5.3.1 Study site

The FTW experiment was conducted at Ashby Pond (38º51” N, 77º17”W), an existing wet pond located at the City of Fairfax, Virginia in a residential area of the Greater Washington, D.C. Metropolitan Area. The pond has surface area of 5,689 m^2, and is part of a small neighborhood park, which provides non-contact recreational use for property owners in the vicinity. The catchment has an average annual precipitation of 1160 mm and monthly precipitation ranged from 65 to 119 mm between 2003 and 2012. Monthly temperatures ranged from 0.7 to 24 ℃ in January and July, respectively (NOAA, 2012). The watershed draining to Ashby Pond is 0.57 km^2, consisting of approximately 38% impervious surfaces. The land use type in the Ashby Pond watershed is mainly residential and commercial mixed with a high traffic arterial street (average daily traffic of 38,000) (VDOT, 2011).

5.3.2 FTW experiments

There were two kinds of the FTWs used in this study: (1) cultural FTWs, which were designed for plant harvest (first phase experiment); and (2) regular FTWs deployed within the pond to study the sustainability of the vegetation on the FTWs from April 27, 2012 to September 11, 2013 (second phase experiment). In the first phase experiment, plant biomass and P content analyses on plant tissues were conducted on samples collected from May 17 to October 31, 2012. The major difference between the two designs was growth media (coir fiber, RoLanka™ Inc., Stockbridge, GA, U.S.). In the cultural FTWs, the coir fiber was prevented from contacting with the plant below-ground tissues for sampling purposes (Section 5.3.2.1). In contrast, on the regular FTWs, macrophytes grew with their roots and rhizomes completely tangled within the coir fiber.

5.3.2.1 Cultural FTWs (first phase experiment)

The cultural FTWs were used to study plant biomass and phosphorus accumulation/distribution from May 17 to October 31, 2012. Two plant species, pickerelweed (*Pontederia cordata* L.) and softstem bulrush (*Schoenoplectus tabernaemontani*), were grown in the system. They were bare-rooted and transplanted to the cultural FTWs on April 27, 2012 for acclimation (20 days before the beginning of the experiment). There were eight FTWs (each 1.52 × 0.91 m, L×W) built with PVC pipes (square floating frame), plastic mesh (bottom support, mesh size: 1.9 × 2.5 cm), and coir mat (top cover, estimated mesh size: 4 × 4 cm, BioD-Mat® 40, RoLanka™ Inc., Stockbridge, GA, U.S.). Each PVC floating frame has 30 plants with single species grown in their individual hydroponic pots, which sat on the plastic mesh and fixed by the coir mat from the top. The space between hydroponic pots within the FTW was vacant. The plastic hydroponic pot is 13 × 9 × 10 cm (top diameter × bottom dia. × H) with slotted mesh on the side and in the bottom, which allows roots to grow freely and stretch into the water. The upper section of the hydroponic pot was filled with the coir fiber to stabilize the plant, while the lower section was left empty so that the below-ground tissues had direct contact with water. This design prevents the roots and rhizomes from tangling with the growth media and enables the sampling of the complete below-ground tissues. The cultural FTWs was protected by a plastic fence (mesh size: 1.9 × 2.5 cm) from four sides (60 cm above-water to 30 cm below-water) and underneath the FTWs as a deterrent to local wildlife species, such as turtles and waterfowl, that would be likely to disturb the plants.

5.3.2.2 Regular FTWs (second phase experiment)

Vegetation on a regular FTW was used to study the plant sustainability under ice encasement stress as Ashby Pond normally freezes during the winter and did so during 2012-2013. The regular FTW (1.52 × 1.22 m, L×W) was built with recycled lumber (frame), plastic bottles (floats), plastic mesh (bottom support), coir mat (top cover), and coir fiber (growth media). The pickerelweed and softstem bulrush were planted at the same time with the experimental macrophytes on April 27, 2012, while yellow iris (*Iris pseudacorus*) was transplanted from the Ashby Pond littoral zone on September 5, 2012. Additionally, the cultural

FTWs were modified and left in Ashby Pond to study the functionality of the PVC pipe floating structures under ice encasement stress through winter after the previous experiment was completed on October 31, 2012 (Section 5.3.2.1). Four of the cultural FTWs were converted to typical FTWs by filling the gap between the bottom plastic mesh support and top coir mat cover with growth media. Unused macrophytes from the previous experiment were planted on the four modified FTWs on November 22, 2012. These four FTWs were then tied to the regular FTW for the second phase experiment.

5.3.3 Sampling and analysis of pond water and macrophytes

Ashby Pond water was sampled at the outlet of the pond once every seven days from May 17 to October 31, 2012 to understand the growth environment of the harvested vegetation. Analyzed constituents included chlorophyll *a* (Chl-*a*), total phosphorus (TP), orthophosphate phosphorus (OP-P), total nitrogen (TN), ammonia nitrogen (NH_4^+-N), and nitrite–nitrate nitrogen (NOx–N) at the Occoquan Watershed Monitoring Laboratory. Filtered samples were analyzed for dissolved nutrients, including OP-P, $NH^+{}_4$-N, and NOx-N. Chl-*a* and nutrient concentrations were analyzed with a Trilogy® Laboratory Fluorometer (Turner Designs, Sunnyvale, CA, USA) and a continuous flow nutrient analyzer (Astoria Analyzer, Astoria Pacific, Clackamas, OR, USA), respectively. All analyses done in the lab followed a quality assurance/quality control (QA/QC) standard (Occoquan Watershed Monitoring Laboratory, 2001). Organic N (Org-N) was estimated as TN – NH_4^+-N – NOx-N. For total particulate phosphorus (TPP), the calculation TP – OP-P was used (Welch and Lindell, 1992). Physicochemical water properties: conductivity (corrected to 25℃), pH, dissolved oxygen (DO), and water temperature were measured by an YSI 556 multi-probe system on Day 0, 3, and 6 in Ashby Pond during each seven-day period. Field measurement were conducted between 9:00 am to 12:00 pm Eastern Standard Time (EST). The pH and DO meters were calibrated before each measurement.

While the pond water was being sampled regularly over the six month period, harvests of randomly selected plants (six per species) were conducted every 28 days from the cultural FTWs. The six plants were randomly divided into two groups for plant tissue analyses (dry weight and P content). The two composited samples (three plants each) were separated into shoot, rhizome (pickerelweed only), and root sections, and weighed for the corresponding fresh weight. All the plant tissues were dried at 70℃ for 48 hours to stabilize the samples and to determine their dry weight (Plank, 1992), with rhizomes sliced into thin pieces to expedite drying. The dried plant tissues were grounded in a Wiley Mill (Thomas Scientific, Model: 3379-K35) and passed through a 35-mesh (0.5 mm) screen. Plant P content was determined by a wet digestion method conducted at the Occoquan Watershed Monitoring Laboratory (Ebeling *et al.*, 2010; Ruiz and Velasco, 2010b). Equal amounts of the ground tissue from the two groups (three plants each) were further composited into a single sample for each plant part (e.g. shoots and roots). The ground samples (0.02 g each) were autoclaved with 5.5M sulfuric acid and potassium persulfate at 110℃ for one hour. The digested solution was analyzed by the nutrient auto-analyzer (Astoria Analyzer, Astoria Pacific, Clackamas, OR, USA). The data of different tissues were separated into above-ground (shoots) and below-ground (roots and rhizomes) groups for further discussion. Leaves grown from lateral buds on the pickerelweed rhizomes harvested on October 31, 2012 were classified as below-ground parts and referred as "submerged leaves" hereafter. These end season submerged leaves did not grow into the size of regular shoots and stayed below the water surface and the growth media through the winter of 2012. This is based on the observation of the pickerelweed growing on the regular FTW.

Besides the two experimental plant species, weeds that had colonized the cultural FTWs were also harvested for analysis. The weeds were removed on July 23, 2012 to prevent potential nutrient competition with the experimental plants. Beggarticks (*Bidens* spp.) was the dominant weed species at the time of harvesting. After the fresh weight of the weeds was measured, three subsamples (20 to 30 g fresh weight) were randomly selected from the pile of the mixed vegetation and analyzed for fresh/dry weight and P content. The averages of the fresh/dry weight ratio and P content of the subsamples and the fresh weight of the total weeds were used to evaluate the P uptake of these colonized plants grown in the period between April 27 and July 23, 2012.

5.3.4 Wildlife observation

Between May 17 to October 31, 2012, observations were performed twice a week from a nearby pond bank (a distance of approximately 3 m) and once a month when harvesting plants and maintaining the cultural FTWs. The observed wildlife species and associated activities were documented. After the first phase experiment, periodical maintenance was stopped to evaluate the natural evolution of the FTWs vegetation without human intervention through September 11,

2013. The reproduction condition and survivability of the three perennial species in the regular FTWs were evaluated through site visits in 2013.

5.4 Result and Discussion

5.4.1 Ashby Pond water quality

Field measurements and water samples were collected from Ashby Pond from May 17 to October 31, 2012. The in-situ physicochemical data are shown as box plots in Figure 5-1. Water temperatures ranged from 32.3 to 8.48 ℃ during the six months experiment, and varied with seasonal changes. The conductivity values demonstrate the variability of the urban stormwater pond environment (52 to 217 μs/cm) because of alternating between intensive storm events and drought periods. The median dissolved oxygen (DO) concentration was 4.7 mg/L and ranged from hypoxic 1.2 mg/L to supersaturated 12.08 mg/L on July 22, and July 3, 2012, respectively. DO concentrations with a range of 2 to 0.5 mg/L are considered to be moderately hypoxic (Diaz and Rosenberg, 1995). For pH, 90% of the observed values were between 6 and 7. A few elevated values (pH>8) were observed in summer, and may have been caused by algal activity.

The concentrations of phosphorus, nitrogen, and chlorophyll *a* in Ashby Pond water are shown in Figures 2 and 3. The median values of nutrient constituents (mg/L) were 0.15 for TP, 0.13 for TPP, 0.016 for OP-P, 1.15 for TN, 1.06 for Org-N, 0.01 for NH_4^+-N, and 0.005 for OxN, respectively. For chlorophyll *a*, the value was 15 μg/L. The TP and TN concentrations were similar to those of two urban wet ponds with FTW application in North Carolina (NC) (Winston *et al.*, 2013). However, the phosphorus and nitrogen in the current study were mainly in non-bioavailable forms, such as organic/particulate constituents, compared to what was observed by Winston *et al.* (2013). This indicates that the nutrient removal processes were more difficult at Ashby Pond because the fraction of the easily consumable orthophosphate phosphorus, ammonia nitrogen, and nitrate-nitrite nitrogen were significantly lower in our study site. The ranges of the TP and TN concentration at Ashby Pond were from 0.26 to 0.11 and 2.05 to 0.79 mg/L, respectively. These values are considerably lower than those in other published FTW studies (Headley and Tanner, 2012). Additionally, the TP and TN ranges of Ashby Pond overlapped with those of irreducible concentrations in stormwater BMPs, which are estimated to be between 0.15 and 0.2 mg/L for TP and at 1.9 mg/L for TN (Schueler, 2000). The irreducible concentrations are thresholds below which further pollutant reduction is very difficult to achieve, and the values are site and treatment mechanism specific. It is noted, however, that the "irreducible" phenomenon has not been evaluated in the FTW literature, and was not investigated in this study.

The Ashby Pond water quality data demonstrates the variability of the urban stormwater ponds environment because of the variable hydrologic conditions and sources. As shown in Figure 5-4, during the gradual increase in water temperature in May and June, photosynthesis of phytoplankton increased, releasing more oxygen into the water. The response of the algal growth was slower, which resulted in a time lag between the increase of the DO (early June) and the algal population (late June). The chlorophyll *a* concentrations peaked in early July but declined immediately after (Figure 5-4). A symptom of the algal population collapse, pond scum color changes from green to brown was observed on July 24. From August 7 to September 12, the DO concentrations were steadily under 4 mg/L except for two measurements. The pond surface was covered by dead algae and gas bubbles were trapped by the scum at the surface. Carbon dioxide generated from decomposition processes may have been trapped in the bubbles as the DO was 1.31 mg/L on August 22, the day this photo was taken. After the water temperatures decreased to lower than 20 ℃ in mid-September, DO increased to livable conditions above 4 mg/L, an acute mortality limit for aquatic invertebrates (US EPA, 1986). The one month period of low DO concentrations in August could have potentially damaged the FTW ecosystem, such as growth of the macrophytes and associated organisms or relocation of inhabiting aquatic insects. DO concentrations below 4 mg/L creates a stressful environment to aquatic life and may result in fish and invertebrate kills (Cooper, 1993; Diaz, 1995). The presence of the algal bloom and resulting associated problems indicated that Ashby Pond is likely a eutrophic water body. This observation is supported by analysis of the Carlson trophic state index (TSI), which had a value 76 calculated from the median TP concentration (0.15 mg/L) (Cooke *et al.*, 1989). TSI values in this range are classified as a hypertrophic condition. Thus, controlling nutrient inputs is critical to improving the aquatic health of Ashby Pond.

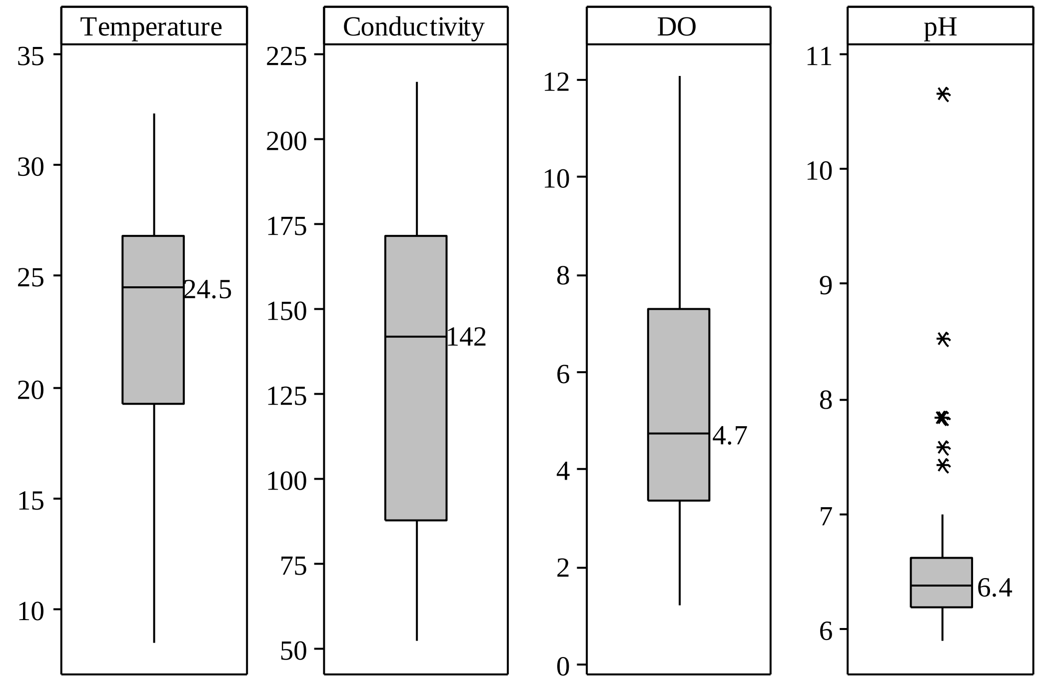

Figure 5-1. Box plots of Ashby Pond temperature (℃), conductivity (µs/cm, standardized to 25 ℃), DO (mg/L), and pH. The numbers in the boxes are the corresponding median values of each constituent (*n*=66).

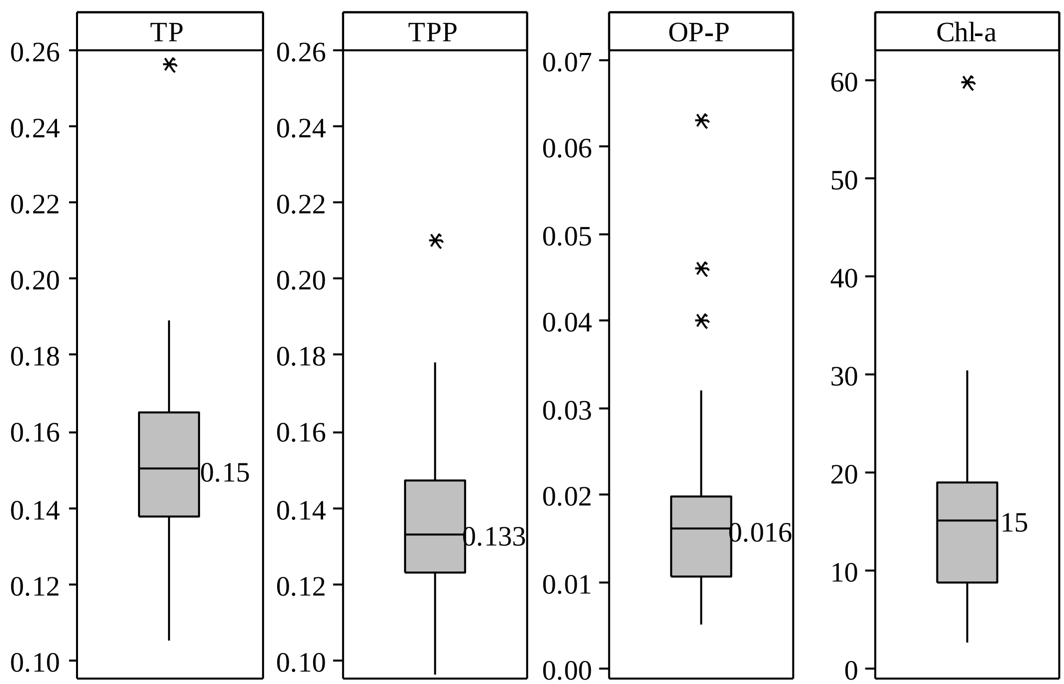

Figure 5-2. Box plots of Ashby Pond total phosphorus (TP, mg/L) total particulate phosphorus (TPP, mg/L), orthophosphate phosphorus (OP-P, mg/L), and chlorophyll *a* (Chl-*a*, µg/L). The numbers in the boxes are the corresponding median values of each constituent (*n*=24).

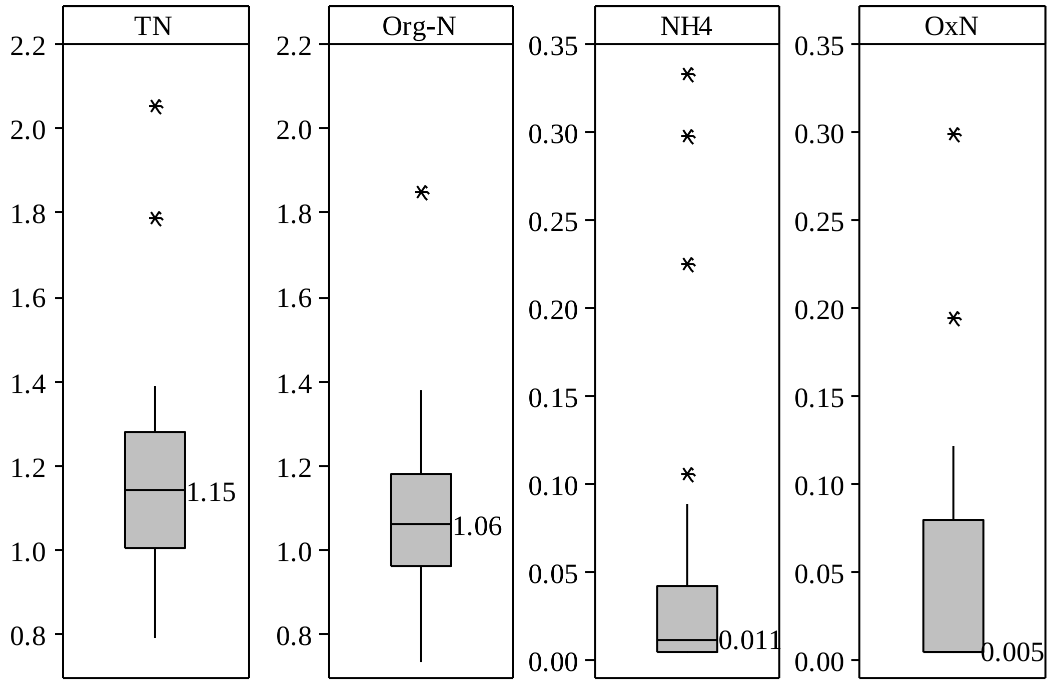

Figure 5-3. Box plots of Ashby Pond total nitrogen (TN, mg/L) organic nitrogen (Org-N, mg/L), ammonia nitrogen (NH_4^+-N, mg/L), and nitrate-nitrite nitrogen (OxN, mg/L). The numbers in the boxes are the corresponding median values of each constituent (*n*=24).

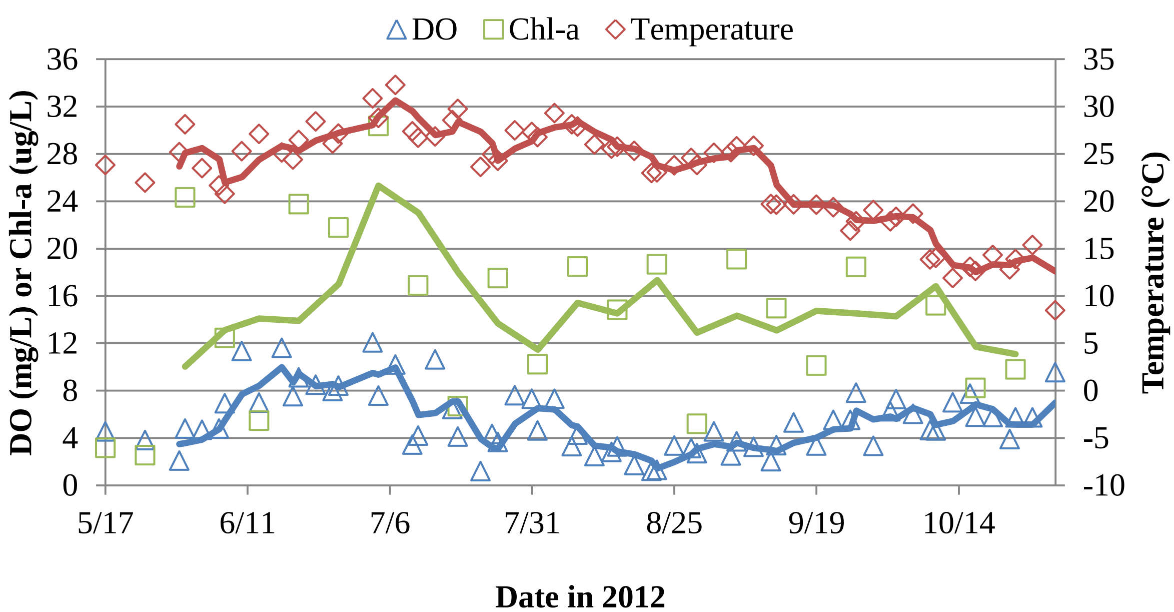

Figure 5-4. Variation of the DO (mg/L), chlorophyll *a* (Chl-*a*, µg/L), and water temperature (°C) of the Ashby Pond. Lines in the figure are three-period moving average trend lines of each constituent. The moving average trend line evens out data fluctuations by connecting the averages of specified number of data points (subset = 3 data points in this figure). The subset starts at the initial number and shifts forward one data point at a time to generate an average as a point in the trend line.

5.4.2 Macrophyte growth study

The growth characteristics of the pickerelweed and softstem bulrush are summarized in Table 5-1. The average shoot length gradually increased from May through September for the softstem bulrush or October for the pickerelweed. For the below-ground tissue, the roots of the pickerelweed grew from 20 to 44 cm and then decreased to 35 cm during the six months experiment. In the case of the softstem bulrush, the trend of the root growth performance was not clear and ranged from 11 to 20 cm. The histograms in Figure 5-5 demonstrate the changes of the two species shoot length distribution during the experiment. Most of the pickerelweed shoot had similar heights, which increased from May 15 to October 3 and declined on October 31. For the softstem bulrush, the shoot heights were more diverse but remained below maximum lengths that also varied with time, similar to the case in the pickerelweed. The different growth strategies between the two species resulted in higher shoot length variation in the pickerelweed across seasons (Table 5-1). Compared to the initial values on May 17, the average shoot length increased 3.6 times in the pickerelweed (from 12 to 43 cm), whereas 2.6 times for the softstem bulrush (from 11 to 29 cm) (Table 5-1). The root growth of the pickerelweed was also more significant than the softstem bulrush that the maximum primary root lengths were 51 and 28.5 cm, respectively. According to the plant morphological data, the pickerelweed grew better than the softstem bulrush after transplanted on the FTW in Ashby Pond.

The temporal whole plant, above-ground tissues, and below-ground tissues biomass DW variation was similar between the two experimental plant species (Figure 5-6). The shoot DWs were highest on September 5, 2.31 and 0.86 g/plant for the pickerelweed and softstem bulrush, respectively. For the below-ground tissues of both species, the highest biomass DW was found 28 days later, or October 3. The biomass DW of whole plant, above-ground, and below-ground tissues steadily accumulated, peaked in fall, and then declined gradually until the last harvest on October 31 (Figure 5-6). The DW above/below ratio (A/B ratio) exhibited different growth patterns that the DW A/B ratio of the pickerelweed increased from 0.35 in May to 0.93 in July and decreased to 0.04 in October, whereas the values of the softstem bulrush were relative stable for the first 112 days (DW A/B ratio: 0.80 to 1.18) but dropped to 0.33 in late fall. The pickerelweed aerial tissues grew rapidly in summer as the plant likely increased resource accumulation through photosynthesis. These shoots then died back in fall faster than those of softstem bulrush. In terms of the softstem bulrush, the growth rates between above- and below-ground parts were similar until October when water temperatures dropped below 20 ℃.

Table 5-1. Growth characteristics of the (a) pickerelweed and (b) softstem bulrush (mean ± standard deviation) from May 17 to October 31, 2012 (*n*=6 for each value). The plants were transplanted to the FTWs on April 27, 2012 for acclimation.

(a) Pickerelweed

Harvest date	Shoot length (cm)	Shoot number / plant	Primary root length (cm)
May 17	12 ± 4	5 ± 1	20 ± 5
Jun. 13	23 ± 7	7 ± 1	27 ± 6
Jul. 11	23 ± 8	5 ± 1	27 ± 8
Aug. 8	36 ± 9	5 ± 1	26 ± 7
Sep. 5	36 ± 15	7 ± 2	44 ± 5
Oct. 3	43 ± 12	4 ± 1	41 ± 6
Oct. 31	24 ± 12	2 ± 1	35 ± 4

(b) Softstem Bulrush

Harvest date	Shoot length (cm)	Shoot number / plant	Primary root length (cm)
May 17	11 ± 6	20 ± 6	15 ± 4
Jun. 13	18 ± 9	11 ± 5	14 ± 2
Jul. 11	21 ± 10	8 ± 2	20 ± 5
Aug. 8	25 ± 12	13 ± 5	11 ± 0
Sep. 5	29 ± 16	13 ± 3	12 ± 5
Oct. 3	22 ± 16	14 ± 3	16 ± 8
Oct. 31	17 ± 13	8 ± 3	11 ± 8

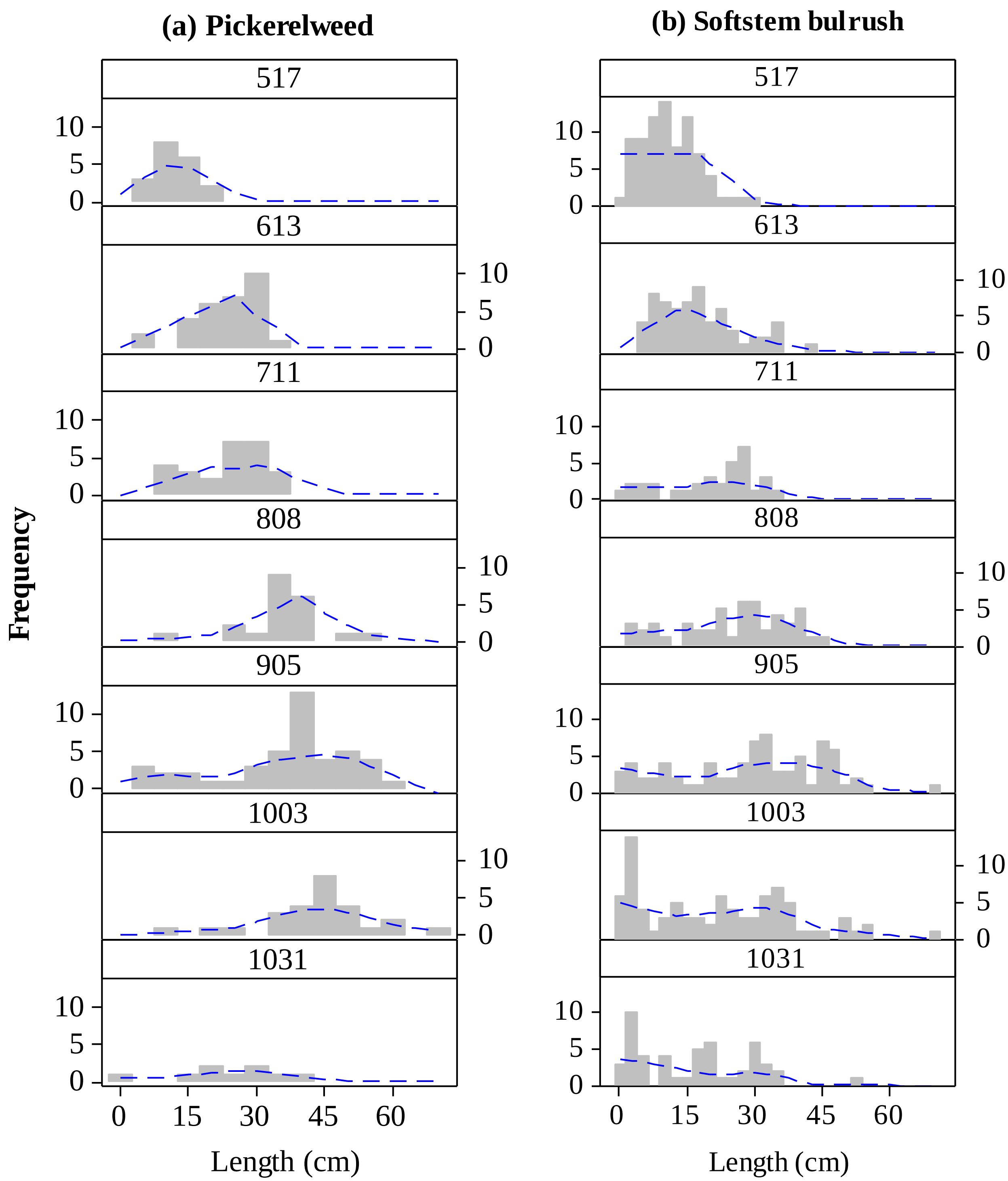

Figure 5-5. Histogram of the harvested (a) pickerelweed and (b) softstem bulrush shoots length distribution (variation) on each sampling date from May 17 to October 31, 2012. The dash lines are two-period moving average trend lines (definition refers to Figure 5-4).

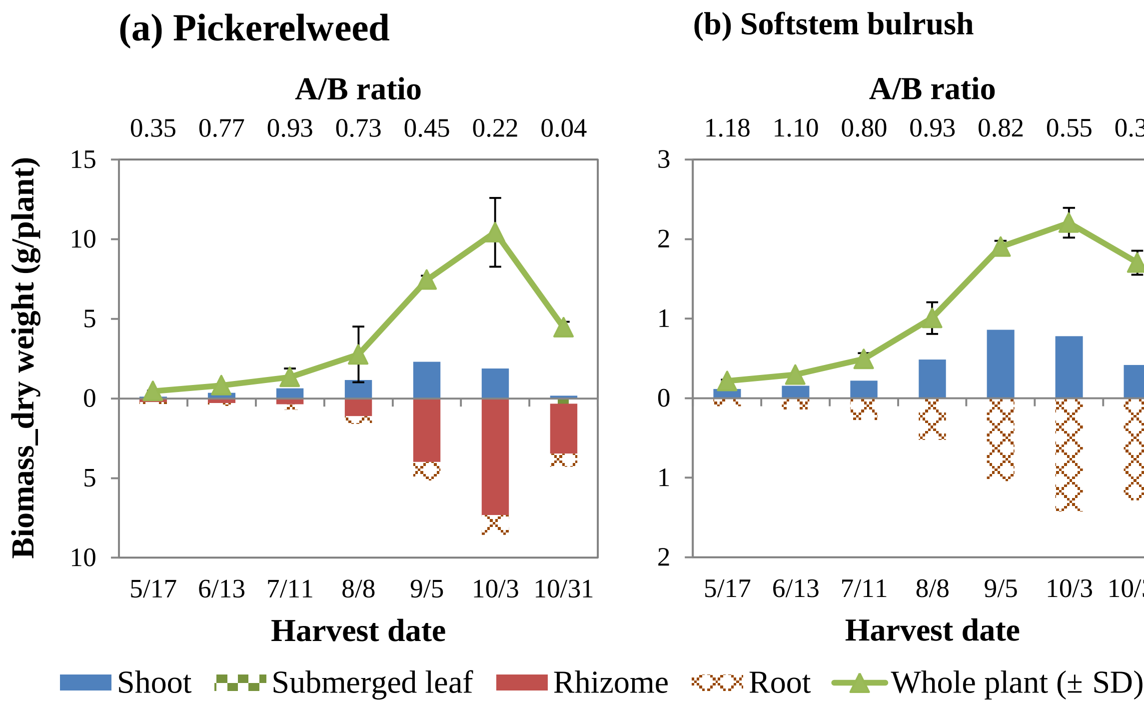

Figure 5-6. Biomass (dry weight) distribution of the (a) pickerelweed and (b) softstem bulrush in Ashby Pond from May 17 to October 31, 2012. A/B ratio: above-ground/below-ground ratio (n=2). Values were evaluated from two groups (three plants each). The submerged leaf on October 31 is classified as below ground part as described in Section 5.3.3.

The low DO condition in Ashby Pond (< 4 mg/L) was most severe between August 8 and September 5. However, significant growth inhibition of both experimental plants was not observed (Table 5-1 and Figure 5-6, Harvest date: September 5). Additionally, the average primary root length of the pickerelweed increased from 26 to 44 cm in the 28 day period. This change of the root morphology is consistent with the results of Zhao *et al.* (2011) who examined aeration effects on the pickerelweed morphology in an FTW experiment. They found that the plant roots in the low DO group (1.07 to 1.13 mg/L) were significantly longer than in the high DO group (2.38 to 2.68 mg/L). The adaptability of the pickerelweed and softstem bulrush to low DO environments is mainly due to the aerenchyma property of these plants, which facilitates oxygen transportation from aerial tissue to under-ground parts and oxidation of the rhizosphere under anaerobic conditions (Colmer, 2003). Moog and Janiesch (1990) investigated responses of three *Carex* species to oxygen deficiency in water culture. The results indicated an additional aerenchyma formation under anaerobic condition (0.3 mg/L DO) for all *Carex* species in the 40 days experiment. Therefore, the aerenchyma may possibly have prevented growth inhibition of the pickerelweed and softstem bulrush during the low DO period in Ashby Pond.

5.4.3 Phosphorus content in the plants

Plant P content and distribution varies temporally, which directly influences the efficiency of P removal through harvest of the aerial fraction of the plant. As shown in Figure 5-7, the P concentration of the whole plant gradually declined from May to late summer, stabilized, and slightly increased in late October for both plant species. While the shoot P concentration steadily declined throughout the experiment, similar trends were shown in the blow-ground tissues during the summer but reversed (increased) starting on September 5. The P concentration of the pickerelweed roots and rhizomes were the lowest in early fall (September 5) compared to late spring (May 17) and late-fall (October 31). For the softstem bulrush, the root P concentration increase in fall was less obvious that in the pickerelweed. This may be caused by life cycle span differences between the two species. According to field observations, the softstem bulrush remained active for one more month into November whereas the aerial part of pickerelweed completely died back in October. Therefore, it is likely that the root P concentration of the softstem bulrush may increase like the pickerelweed in November.

The P mass distribution of the experimental plants in Ashby Pond is shown in Figure 5-8. The whole plant P accumulation was highest in fall for both species. For pickerelweed, the changes of the P mass were similar to those of the biomass DW that more resources and biomass were located at the shoots than the below-ground tissues in summer. But the P mass A/B ratio decreased in fall, which indicates the mass was relocated to the storage organs. In the case of the softstem bulrush, the P mass A/B ratios were gradually reduced from 1.3 to 0.1 in the six months experiment. However, the biomass DW A/B ratios were relative stable for the first 4 months as discussed in Section 5.4.2. The result shows that the P use efficiency of the softstem bulrush was higher in the shoot than the roots during the growing season. After the summer growth season, resource remobilization from aerial to below-ground tissues dominated the mass distribution in the softstem bulrush. The temporal variation P distribution was related to the phenology of the plant (Ruiz and Velasco, 2010b). The water temperature of Ashby Pond continuously decreased from 25 to 15 ℃ in fall (Figure 5-4). The external environment changes signal plant internal mechanisms preparing for the reproduction of the next year (Ruiz and Velasco, 2010b). Nutrients are remobilized from mature shoots to storage organs, rhizomes and roots, while nutrient supply to the above-ground tissues may be interrupted at this stage (Marschner, 1995).

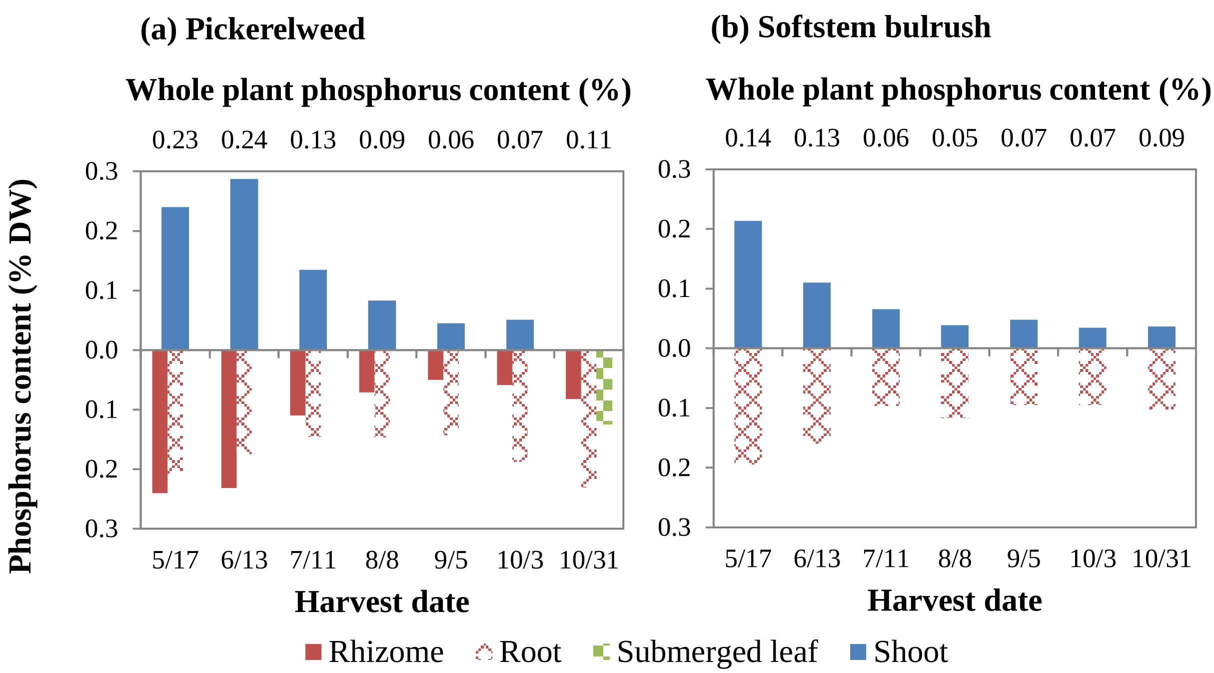

Figure 5-7. Typical dry biomass P concentrations of (a) pickerelweed and (b) softstem bulrush in shoot, rhizome, root, and submerged leaf in Ashby Pond from May 17 to October 31, 2012 (n=1). Values were evaluated from composite samples (six plants). The submerged leaves is classified as below ground part as described in Section 5.3.3. (DW: dry weight)

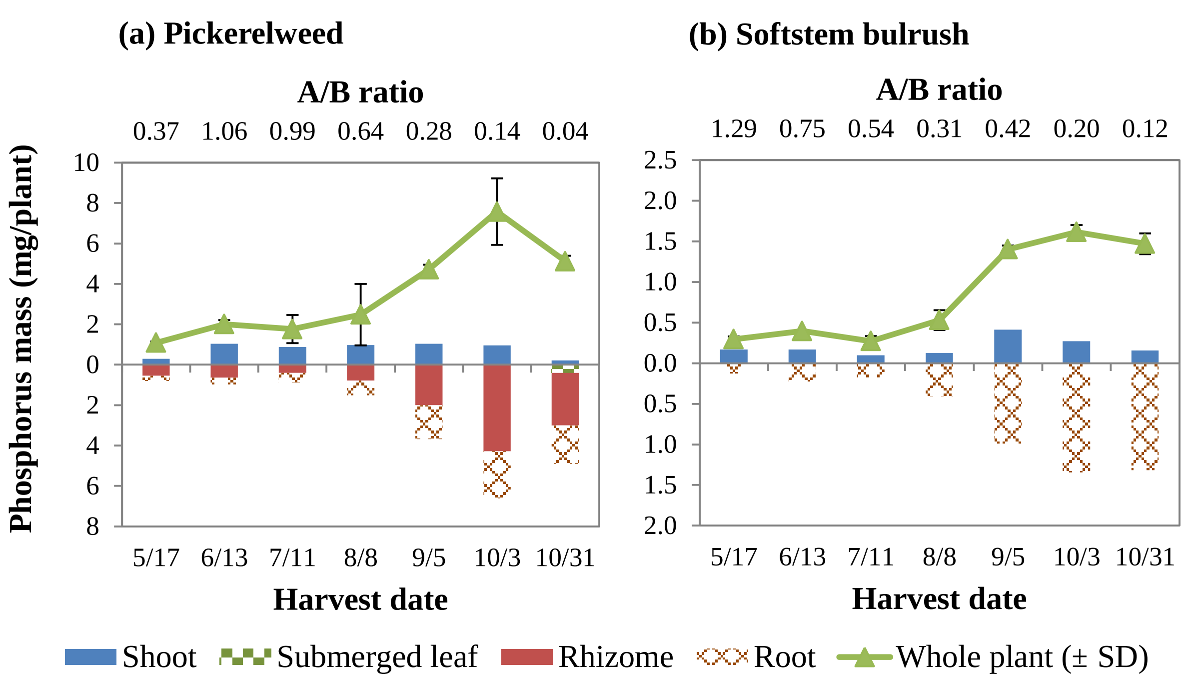

Figure 5-8. Phosphorus mass distribution in the (a) pickerelweed and (b) softstem bulrush in Ashby Pond from May 17 to October 31, 2012 (n=2). A/B ratio: above-ground/below-ground tissue ratio. Values were evaluated from composite samples of two groups (three plants each). The submerged leaf is classified as below ground part as described in Section 5.3.3.

5.4.4 Phosphorus removal through plant harvest

Harvest of the aerial portion of the plant is a common practice in order to remove nutrients from treated water bodies. Because the P content of the shoots varies with time, it introduces uncertainty to the nutrient removal efficiency by harvesting at the time coincident with maximum biomass of the aerial part of plant. As shown in Figure 5-8, the P mass in the shoots reached a plateau from June 13 to October 3 for pickerelweed, whereas this did not occur until the period from September 5 to October 3 for softstem bulrush. The aerial tissue biomass DWs were highest from September 5 to October 3 for both species. Therefore, above-ground tissue harvest in early fall could remove maximum biomass (DW) with highest P mass content according to the results observed in Ashby Pond. On the other hand, pickerelweed harvest in June would result in a similar amount of P mass removal but with less biomass DW. An additional harvest in fall could prevent nutrient release caused by senescence. While this discussion may optimize P removal by harvesting the plant tissues twice, it is important to also prevent negative impacts to vegetation caused by such practices. Without sufficient storage in the below-ground organs, new growth in the next year could be reduced and thus impact nutrient uptake as discussed by Nakamura and Shimatani (1997). Comparisons between the two species show that the pickerelweed accumulated more P in both whole plant and above-ground tissue parameters than the softstem bulrush. The pickerelweed maximum whole plant P mass in (7.58 mg/plant) and above-ground tissue mass (1.04 mg/plant) compared to its initial values on May 17 were 7.0 and 5.4 times higher, respectively. For the softstem bulrush, the values were 3.6 and 2.4 times with the maximum values of 1.62 and 0.41 mg/plant, respectively. Therefore, the pickerelweed demonstrated better P removal performance then the softstem bulrush in the first year.

Numerous non-planted species (weeds) colonized the periphery of the FTWs during the six-month experiment. Their location on the FTWs suggest that propagation of these plant species relied upon hydrochory (dispersal through water) to colonize the FTWs. According to the analysis of the three weed subsamples, the average dry weight/fresh weight ratio was 0.1 with P concentration 0.12% DW. Therefore, it is estimated 0.35g P was contained in the 2.9 kg weed harvested on July 23, 2012. As there was no growth media inside the PVC frames, the weeds could only grow on the outside of the PVC frame covered by the black cloth and coir mats. The weeds were mainly concentrated at the contact area between the FTWs. Therefore, the weed growth area was estimated to be 0.5 m^2. According to these parameters, the P removal rate of the weeds is calculated as 10.5 mg/m^2-day from May 17 to July 23, 2012. Despite the undervaluation, the value is higher than the results of Chua *et al.* (2012) who studied FTWs with three plant species in a reservoir with average 0.22 mg/L TP and 1.6 mg/L TN in Singapore. The average P uptake rates (mg/m^2-day) were 1.57 for *Typha angustifolia*, 0.16 for *Chrysopogon zizanioides*, and 0.4 for *Polygonum barbatum*. However, only mature shoots were sampled in their study. The nutrient uptake by plants is expected to be affected by the nature of the water bodies. For example, Hubbard *et al.* (2004) reported a P uptake rate with 230 mg/m^2-day in the FTW planted with *Typha latifolia* L. Wastewater from a swine lagoon with 30 mg/L TP and 160 mg/L TN was used in their experiment. The weeds on the Ashby Pond in-pond FTWs appeared naturally but also provided nutrient removal benefits, and also suggested that FTW ecosystems evolve through natural succession.

5.4.5 FTW vegetation sustainability in temperate region

The sustainability of the FTW, in terms of its PVC pipe support structure and vegetation due to exposure to ice encasement was examined through field observations from May 2012 to September 2013. The PVC pipe frame remained intact and provided sufficient buoyancy to support the growth of the vegetation in 2013. Two plant species, pickerelweed (*Pontederia cordata* L.) and softstem bulrush (*Schoenoplectus tabernaemontani*) were planted on the demonstration FTW on April 27, whereas yellow iris (*Iris pseudacorus*) was planted on September 5, 2012. Four photos in Figure 5-9 show annual changes of the vegetation on the demonstration FTW. There were 12 pickerelweed plants evenly distributed on the left and right half of the demonstration FTW (six plants each, Figure 5-9-a). However, it was observed that the reproduction rate of the six plants on the left half FTW were lower than those on the right half (Figure 5-9-d). The growth condition of the softstem bulrush was insignificant compared to the pickerelweed in both 2012 and 2013. Despite competition with the other two planted perennial macrophytes and annual weeds, this species was still present on the demonstration FTW in 2013. Moreover, the maximum shoot length of the softstem bulrush in 2013 was more than 160 cm, compared to 70 cm in the first year. The softstem bulrush may require a longer period to adapt to the new habitat and provide nutrient control benefits. The yellow iris was collected from the littoral zone of Ashby Pond and put on the demonstration FTW in September, 2012. Although it only had a short establishment period before entering winter, the iris adapted well to the hydroponic environment. It is suggested the resource accumulated in the rhizome supported the greater performance of this species. The same reason may also explain the difference between the pickerelweed and softstem bulrush, which were planted on the demonstration FTW on the same date. The rhizome formation of the pickerelweed was more obvious compared to the softstem bulrush according to the plant samples harvested on October 3 and October 31, 2012 from the in-pond FTWs. The nutrients in the below-ground storage pool (e.g., rhizomes) are used for the new growth of shoots in early spring (Ruiz and Velasco, 2010b).

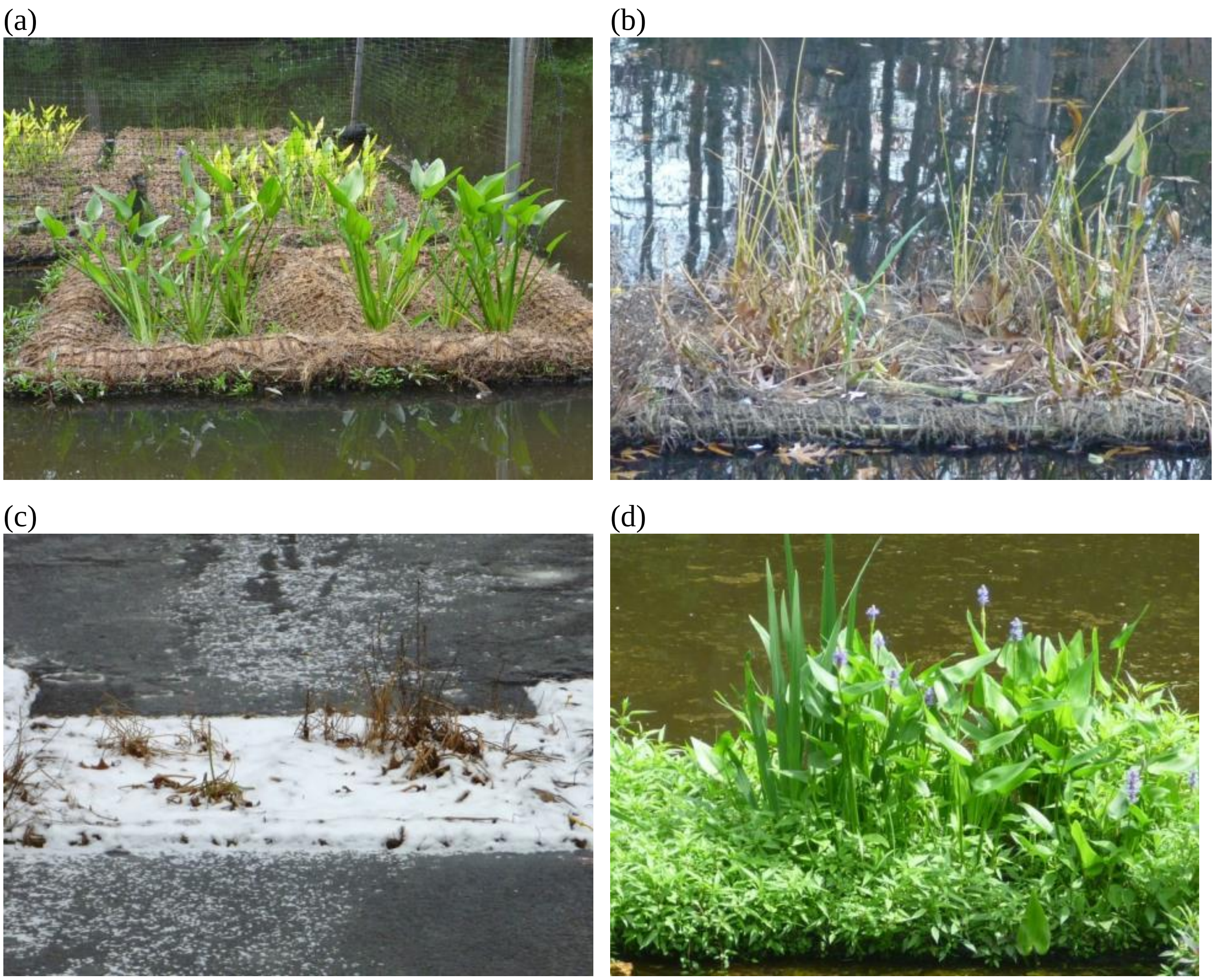

Figure 5-9. Annual vegetation changes of the demonstration FTW: (a) June 19, 2012, (b) November 25, 2012, (c) January 25, 2013, and (d) June 17, 2013. The yellow iris (not shown in (a)) was planted on September 5, 2012.

All three planted perennial species reproduced in 2013 after experiencing ice encasement in January, 2013 (Figure 5-9-c). As perennials, these macrophytes return reliably in soil rooted environments where their reproductive organs persist (Cronk and Fennessy, 2001). However, the reproductive organs of the macrophytes on the in-pond FTW were submerged under the surface and surrounded by water instead of soil. The stress imposed by ice encasement could have caused irreversible damage on cell walls and cell membranes (Guy, 1990). Additionally, ice retards gaseous exchange between water and atmosphere and has the potential to cause severe hypoxia within plants (Andrews, 1996). For winter wheat rooted in soil, plants are often damaged when soil moisture content is higher than 40% due to insufficient pores for aeration. The pickerelweed, softstem bulrush, and yellow iris could adapt to this condition by two strategies. First, the respiration of the ice-encased below-ground tissues could receive oxygen transported by aerenchyma in standing stubble (Colmer, 2003). Second, the roots stretched below the surface ice layer can utilize oxygen dissolved in water (Ehret *et al.*, 2010). The maximum root length was 47 cm from a measurement of the demonstration FTW on October 5, 2012. The results demonstrate the potential of perennial macrophyte reproducibility under ice encasement stress in winter. Because the Ashby Pond is located within a region with temperate winter and infrequent freezing temperatures, FTW studies at areas with more severe winter are needed to evaluate the survivability of perennial macrophytes. The extended ice encasement period may raise the mortality rate of the vegetation on FTWs.

5.4.6 Field observation

Vegetated FTWs in Ashby Pond show great potential for habitat creation or restoration. After the conversion of the in-pond FTW cultural system to regular FTWs on November 22, 2012 (Section 5.3.2), the system was left undisturbed in Ashby Pond. Diverse plant species migrated to the new habitat and thrived. The identified species grown on the FTW included three planted perennial macrophytes, pickerelweed (*Pontederia cordata* L.), softstem bulrush (*Schoenoplectus tabernaemontani*), and yellow iris (*Iris pseudacorus*), and naturally colonized beggarticks (*Bidens* spp.), smartweed (*Polygonum hydropiperoides*), Virginia water horehound (*Lycopus virginicus*), and yellow nutsedge (*Cyperus esculentus*). The beggarticks dominated more than 65% of the FTWs on July 20, 2013. The plant diversity was increased from three (manually planted) to seven species after the installation in May 2012.

The life of diverse organisms was intersected with Ashby Pond FTWs in different times (Table 5-2). Floating rafts and macrophytes appeared to provide needed shelter and possible food sources for these primary and secondary consumers and decomposers. Aquatic macroinvertebrates and tadpoles were found to inhabit in the floating mats, which are similar to littoral zones with ample hidden space and food sources. A female six-spotted fishing spider was spotted nesting on the vegetation and foraging at the adjacent water body while protecting hundreds of newly hatched spiders (Figure 5-10-a). On November 12, 2012, five common carps with maximum total length of 13 to 17 cm were captured in the plastic protection fence with mesh size 1.9 × 2.5 cm (Section 5.3.2). It is assumed that they entered the protection fence at a juvenile stage and stayed at the root zone shelters below the FTWs. A great blue heron and a green heron (juvenile) were found foraging on the FTWs (Figure 5-10-b). The green heron continuously visited the site continuously for 26 days (August 14 to September 9, 2012). Other wild life visited the FTWs for resting or other activities, like sunbathe to regulate reptiles' internal temperature (Figure 5-10-c to f). When there was no or limited vegetation on the FTWs in winter and early spring, Canada geese and mallards occupied the floating surface as a resting area (Figure 5-10-g and h). As discussed previously, live organisms at different trophic levels inhabited or frequently visited the FTWs. It is assume that the interaction between trophic levels may develop a food web with nutrients flowing through the FTW ecosystem.

The objective of this study was to observe species introduced by the FTW ecosystem. A limited number of similar studies of FTW practice have been reported. Seo *et al.* (2013) conducted a field FTW study in an oligotrophic reservoir and found fish eggs attached at plant roots under FTWs. Similarly, fish and shrimp populations were higher under the floating mats than in nearby open water locations (Billore, 2007; Nakamura *et al.*, 1998). A "fish hotel" with similar structures to FTWs was located on the6 Chicago River in downtown Chicago, and attracted fish in the concrete urban channel (Srivastava, 2011). In our study, we document insect, reptile, amphibian, and bird activities within the FTW. Further research could be conducted to evaluate the potential ecological positive and negative impacts of FTWs, and, in particular, the aquatic nutrient subsidy. In this context, subsidy is a term that describes nutrient and energy flow between habitats (Ballinger and Lake, 2006; Marczak and Richardson, 2007). Flux of aqueous mass to terrestrial systems through trophic interactions, or food webs, has been documented in lakes and rivers (Gratton *et al.*, 2008; Marczak and Richardson, 2007). Another potential benefit of FTWs that could be evaluated is fisheries improvement. Fisheries also export aquatic nutrient subsidy and simultaneously provide economic benefits. The Taiwanese aboriginal tribe, Thao, has conducted fishing on its “floating fields” for more than 50 years (Li, 1957).

Table 5-2. Summary of observed wildlife activities at the Ashby Pond FTW.

Class		**Common name**	**Scientific name**	**Activity***
Actinopterygii		Common carp	*Cyprinus carpio*	I[ξ]
Amphibia		American bullfrog-adult	*Rana catesbeiana*	R
		American bullfrog-tadpole	*Rana catesbeiana*	I
		Gray treefrog	*Hyla versicolor*	B+R
		Pickerel frog	*Lithobates palustris*	R
Arachnida		Six-spotted fishing spider	*Dolomedes triton*	F+I+N
Aves		Belted kingfisher	*Ceryle alcyon*	F
		Canada goose	*Branta Canadensis*	R
		Great blue heron	*Ardea Herodias*	F
		Green heron-Juvenile	*Butorides virescens*	F
		Mallard	*Anas platyrhynchos*	R
Clitellata		Freshwater leech	-	I
Gastropoda		Bladder snail	*Physella acuta*	I
Insecta	aquatic	Chironomid-larva	-	I
		Damselfly-larva	-	I
		Dragonfly-larva	-	I
		Water scorpion	-	U
	non-aquatic	Bee	-	F
		Butterfly-adult (skipper)	-	F
		Butterfly-larva	-	F
		Damselfly-adult	-	R
		Dragonfly-adult	-	B+R
Reptilia		Common snapping turtle	*Chelydra serpentina*	U
		Eastern painted turtle	*Chrysemys picta picta*	R
		Northern water snake	*Nerodia sipedon*	R

* B: breeding; F: forage; I: inhabit;; N: nursing; R: rest; U: unknown.
ξ Trapped in the protection fence (Section 5.3.2).
- Unidentified.

(a)

(b)

(c)

(d)

(e)

(f)

Figure 5-10. Wildlife utilization of the FTWs. (a) six-spotted fishing spider; (b) green heron; (c) bullfrog; (d) pickerel frog; (e) eastern painted turtle; (f) northern water snake; (g) Canada goose; (h) mallard.

(g)

(h)

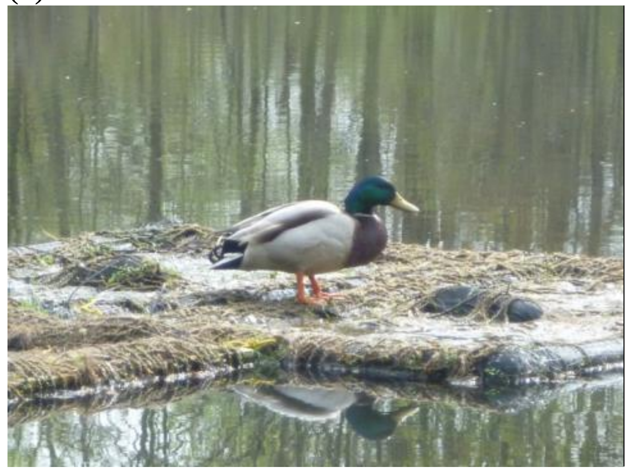

Figure 5-10. (continued)

5.5 Conclusion

This study examined performance of an FTW in an enriched urban stormwater BMP, Ashby Pond, through collection of approximately six months of water and plant samples, subsequent laboratory analyses, and 15 months of field observation data. Planted perennial macrophytes successfully adapted to stresses of the low DO concentrations (Minimum: 1.2 mg/L) in summer and ice encasement in winter of the urban stormwater pond with low concentrations of nutrients (median 0.15 mg/L TP and 1.15 mg/L TN). The comparison between the two experimental species demonstrated that the pickerelweed had a higher P removal performance (7.58 mg/plant, whole plant) than the softstem bulrush did (1.62 mg/plant). For P mass in the harvestable aerial tissue, the values were 1.04 and 0.41 mg/plant, respectively. The biomass and P temporal variability indicated that resources were accumulated at the shoot tissues in summer but were remobilized to below-ground storage organs in the fall. Therefore, the harvest of above-ground vegetation is suggested to be conducted in June for maximum P removal and in September to prevent P release due to senescence. However, the potential damage to the macrophytes caused by harvest was not evaluated in this study and is recommended for future research.

The diverse organisms utilized the Ashby Pond FTW were documented in the field study. The plant diversity was increased from three to seven species one year after the installation. Observed wildlife activities associated with the FTW include inhabitation, forage, breeding, nursing, and rest. We recommend that future research be conducted to evaluate potential benefits, such as aquatic nutrient subsidy and economic returns from fisheries.

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Chapter 6. Conclusions and recommendations

6.1 Conclusion

This research evaluated FTW behavior and its nutrient removal performance in urban ponds through three study topics. First, data were collected from published literature to develop an assessment model, the i-FTW model, using first order kinetics. The i-FTW model simulates the changes of TP and TN concentration in an FTW system with respect to variation in design parameters, such as retention time and FTW coverage size. FTW performance was further examined in the deployed water bodies and evaluated by FTW uptake velocity (v_f) with uncertainty analysis. The i-FTW model not only improves the understanding of FTW nutrient control ability, it may also serve as an important tool when designing FTW at the field scale. Second, the mesocosm experiment utilized a multiple treatments approach to study the effects of macrophytes and floating mats separately and compare them with a control group, which simulated a shallow urban wet pond. Water quality and vegetation data were collected to investigate FTW performance in treating urban runoff. Additionally, while most published studies considered FTW systems as a single unit, the floating mat influence has received little attention and is less understood. Third, the field (in-pond) experiment investigated the plant growth performance and temporal nutrient translocation, vegetation sustainability in temperate region, and ecological/economic benefits associated with the FTW deployed in a eutrophic urban stormwater pond, Ashby Pond. Water and plant samples were collected for approximately six months and analyzed for nutrient content and growth characteristics of FTW vegetation. The span of the field observation was 16 months. The observed wildlife species and associated activities were documented.

The i-FTW 1st order kinetic model incorporates compartmental concepts to separate the FTWs' contribution from the effects caused by the integrated water bodies. Therefore, the i-FTW model can be adapted to various treatment systems by changing the model parameters, such as water reaction rates (k_w), system water volumes (V), and reaction time (t). The k_w values are selected from literature reported data or calculated from water concentration changes of the applied systems. The FTW performance is described by FTW apparent uptake rates (v_f). The uncertainty of the v_f value was analyzed by a bootstrap technique to evaluate its expected range with 95% certainty, 0.017 to 0.058 m/day for TP and 0.014 to 0.043 m/day for TN. The results from the cross validation test show that the i-FTW model better responds to diverse water and system conditions and provides more accurate predictions than the other two common approach: removal rate (mg/m^2/day) and removal efficiency (%).Therefore, the i-FTW model can serve as a preliminary tool to assist i-FTW system designs. However, it should be understood that there are limitations of the i-FTW model: (1) the FTW apparent uptake velocities (v_f) were evaluated based on only 12 i-FTW studies, adding a significant amount of uncertainty; (3) the influences of integrating FTWs into original water bodies were not considered; (4) the i-FTW model has only been tested by the concentration time series data from the i-FTW studies.

The results of the mesocosm experiment indicate that all three FTW designs, planted (with two species) and unplanted floating mats, significantly improved phosphorus and nitrogen removal efficiency of a simulated urban retention pond during the growing season, which was from May through August. The average of initial TP and TN concentration of the experimental water was 0.15 and 1.19 mg/L, which is considered to be at the lower end of the expected concentrations and poses a great challenge in evaluating FTW treatment effectiveness. With a retention time of seven days, 66-69 % of TP and 48-50 % of TN were reduced in the three FTW treatments. Compared to the control treatment, the removal efficiency was enhanced by 8.2 % for TP and 18.2 % for TN in the planted FTW treatments (B and P). Organic matter decomposition driven by the microbes attached on the floating mats and plant root surface is likely the primary mechanism of nutrient removal by FTWs in urban retention ponds. Comparing the results of the four treatments, the FTWs planted with pickerelweed exhibited the best nutrient control performance; this species also provided an additional aesthetic benefit. The modified Arrhenius equation effectively simplified the temperature effects on the plant performance and suggested that pickerelweed was most sensitive to the temperature variation and provides considerable phosphorus removal when water temperature is greater than 25℃. However, when water temperatures were below 15℃, the effectiveness of this plant species was negligible for phosphorus and negative for TN compared to the floating mat group. The temperature effects on the softstem bulrush were similar to those on the floating mat group. For the water physicochemical characteristics, the shading effects of the FTW with 61% coverage could significantly lower water temperatures and reduce thermal pollution. The plants and associated microorganisms on the floating mats may reduce pH and DO of the applied water bodies. It should be noted that the findings of this study are restricted to shallow urban retention ponds

(0.32 m), similar to our mesocosm conditions, with significant FTW coverage (61%) and low plant density (10.4 plants/m^2). Because sediments were absent in our mesocosm tanks, adsorption or release of nutrients from sediments within retention ponds is not part of the simulation.

The field experiment (in-pond) examined the FTW performance in a eutrophic urban stormwater pond, Ashby Pond. The results indicate that the planted perennial macrophytes successfully adapted to low DO concentrations (Minimum: 1.2 mg/L) in summer and ice encasement in winter of the urban stormwater pond with low concentrations of TP and TN (median 0.15 mg/L TP and 1.15 mg/L TN). The comparison between the two experimental species demonstrated that the pickerelweed has higher phosphorus removal efficiency (7.58 mg/plant, whole plant) than the softstem bulrush (1.62 mg/plant). Phosphorus mass in the harvestable aerial tissue were 1.04 and 0.41 mg/plant, respectively. The results of macrophyte phosphorus distribution analysis indicate that nutrient removal through plant aerial tissues harvest should be conducted twice a year. The temporal variation of biomass and phosphorus distribution in both mesocosm and field experiments were similar, which indicate resources were accumulated at the shoot tissues in summer and remobilized to below-ground storage organs in fall. Therefore, the harvest of above-ground vegetation is suggested to be conducted in June for maximum phosphorus removal and in September to prevent phosphorus release due to senescence. The potential damage to the macrophytes caused by harvest was not evaluated in this study and is recommended as a topic for future research.

In terms of plant diversity, the vegetation species on the Ashby Pond FTW increased from three to seven one year after its installation. Systematic observation of wildlife activities indicated eight classes of organisms inhabiting, foraging, breeding, nursing, or resting in the FTWs. As knowledge accumulates on FTW as a treatment BMP, the application of the system in retention ponds and other water bodies may provide an alternative solution to control non-point source pollution at watershed scale, such as within the Chesapeake Bay watershed.

6.2 Recommendations

6.2.1 Future research

6.2.1.1 FTW assessment model (i-FTW)

- A correction factor should be developed to account influences on the water reaction rate (k_w) due to the presence of FTWs. As was observed in this experiment (Chapter 4), the floating mat shading effects altered physicochemical conditions of deployed water bodies.
- FTW experiments with a control group in a variety of regions should be conducted to identify the temperature effects on the FTW apparent uptake velocity (v_f). The reported v_f values in Chapter 3 were evaluated mainly from a single region.
- Data from full scale i-FTW treatment facilities should be collected to further validate the i-FTW model or models incorporating the v_f term.

6.2.1.2 FTW mesocosm experiment

- FTW mesocosm with pickerelweed during the non-growing season should be tested to fully understand the seasonal effects on nutrient removal. The results in this experiment observed high nutrient removal efficiency between May and September and eventual decline and nutrient release in the fall. However, no testing was done during the late fall-winter period. Research efforts should be dedicated to evaluate the duration and magnitude of the nutrient release from the pickerelweed during the non-growing season.
- The effects of FTW coverage changes on physicochemical attributes of water bodies should be further investigated. As FTW coverage increases, more nutrient removal is expected. However, results of the associated impacts, such as reductions in pH and DO, are still uncertain.
- A strategy that harvests above-ground vegetation twice a year (June and September) should optimize nutrient removal from aquatic system in temperate region. However, long term damage to macrophytes should be evaluated to improve the management strategy.

6.2.1.3 FTW field experiment

- The survivability of perennial plants on FTWs in regions with more severe ice encasement stress should be explored. In particular, the duration of ice encasement and ice thickness need to be examined.

- Aquatic nutrient export through aquatic insect emergence should be quantified to evaluate the potential mass flow.
- The long term performance of pickerelweed and softstem bulrush on FTW rafts should be evaluated. The nutrient removal performance of macrophytes on FTWs may gradually improve in following generations that reproduce each year.
- Nutrient removal effectiveness of the colonized indigenous vegetation should be investigated. The dominant species on the Ashby Pond FTWs in 2013 was the beggar's ticks, an annual plant. Despite their fast growing nature, they may significantly release nutrients after senesced in fall and winter. Strategies to ensure the dominance of perennial species on FTWs may be needed.

6.2.2 FTW design, deployment, and operation

- FTW should be anchored in the deep water regime and avoid the littoral zone where benthic periphyton inhabit according to findings documented in Chapter 4. This attached-to-substrate microorganism community can provide nutrient treatment benefits and is driven by sunlight, which is blocked when FTWs are presented. This extra consideration may be ignored if the water turbidity is high and benthic periphyton activity is insignificant. Additionally, FTW shading may be utilized to control the productivity of submerged vegetation if such a problem exists.
- Selection of growth media on FTWs are based on the vegetation harvest strategy. For whole plant removal, floating frames with separate biodegradable growth media, such as coir mats, is suggested as the vegetation and the growth media can be easily replaced as filters. The same floating frame can be reused with new growth media and plants. On the other hand, biodegradable and permanent materials, such as plastic foam, are alternative growth media. However, the growth media replenishment may be necessary when using biodegradable materials. Below-ground plant tissues could develop into mats on FTWs through natural processes similar to those occurring in natural floating islands.
- In addition to vegetated FTWs, it is suggested to build a few floating structures without vegetation and growth media next to them. According to the field observations at the Ashby Pond FTWs, the surface was completely covered by plants in summer 2013. As a result, higher trophic level predators, like birds, could not forage at the more mature FTWs because

of limited space to land. Similarly, reptiles could not visit the more mature FTWs where most sunlight is blocked by vegetation. Therefore, aquatic mass exportation through the trophic interaction and ecological benefits are reduced. The addition of a non-vegetated floating structure may provide a solution for this issue.

- The five criteria for plant species selection were described in Section 4.3.2. Two of them, perennial plants and macrophytes with aerenchyma, are more important than the others according to our finding. First, perennial species remobilize and store nutrients in below-ground tissues for reproduction. The life cycle of annual macrophytes ends in fall, which may cause nutrient release from their senesced tissues. Second, wetland plants with aerenchyma could adapt to hypoxia condition in eutrophic water bodies, like Ashby Pond in summer.
- Perennial vegetation transplantation from local habitats is recommended. As discussed in Chapter 5, indigenous macrophytes ultimately colonized and dominated the FTWs in Ashby Pond. This approach also provides economic benefits because plant purchase costs are reduced or avoided.