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Application of floating wetlands for the improvement of degraded urban waters: Findings from three multi-year pilot-scale installations

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HIGHLIGHTS

- Evaluated P removal rate for FTW in eutrophic waterbodies.
- Novel evidence of changes in ecological function.
- Zooplankton data show pathway for supporting top-down control of blooms.

GRAPHICAL ABSTRACT

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ABSTRACT

Floating Treatment Wetlands (FTWs) are an emerging ecological engineering technology being applied the restoration of eutrophic urban water bodies. Documented water-quality benefits of FTW include nutrient removal, transformation of pollutants, and reduction in bacterial contamination. However, translating findings from short-duration lab and mesocosm scale experiments, into sizing criteria that might be applied to field installations is not straightforward. This study presents the results of three well established (>3 years) pilot-scale (40–280 m^2) FTW installations in Baltimore, Boston, and Chicago. We quantify annual phosphorus removal through harvesting of above-ground vegetation and find an average removal rate of 2 g-P m^{-2}. In our own study and in a review of literature, we find limited evidence of enhanced sedimentation as a pathway for phosphorus removal. In addition to water-quality benefits, FTW planted with native species, provide valuable wetland habitat; and theoretically improve ecological function. We document efforts to quantify the local effect of FTW installations on benthic and sessile macroinvertebrates, zooplankton, bloom-forming cyanobacteria, and fish. Data from these three projects suggest that, even on a small scale, FTW produce localized changes in biotic structure that reflect improving environmental quality. This study provides a simple and defensible method for sizing FTW for nutrient removal in eutrophic waterbodies. We propose several key research pathways which would advance our understanding of the effects FTW have on the ecosystem they are deployed in.

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1. Introduction

Around the globe, urban waters are both uniquely valuable and uniquely challenged. For cities, urban waters provide flood control, cooling (Wu and Zhang, 2019), and improvements to human health (Tieges et al., 2022) and well-being (Garcia et al., 2016; Jakubiak and Chmielowski, 2020). Remaining riparian habitat, though scarce, offers a critical toehold for native and migratory species (Francis, 2014; Paul and Meyer, 2001) and unique exposure to the natural world for human residents. Simultaneously, urbanization alters the physical, chemical, and hydrologic environment of waterbodies (McGrane, 2016), fueling algal blooms, introducing bacterial contamination (Marsalek and Rochfort, 2004), and degrading biodiversity and ecological function (Le Moal et al., 2019). The development of hardscape and engineered drainage systems create a hydrology in which stormwater transports urban road dust, including heavy metals (Hwang et al., 2016) and organic contaminants (Hwang et al., 2019) directly into receiving waters, bypassing the natural filtration of vegetation and soils (Hobbie et al., 2017; Khan et al., 2021). Under these altered conditions, even natural substances such as leaf-litter contribute substantially to excess nutrient loading and eutrophication (Bratt et al., 2017; Janke et al., 2017). A further challenge to urban waters is the absence of a natural transition from land to water which has often been replaced by a hardened urban edge (Kentula et al., 2004; McCauley et al., 2013) degrading the waterbodies capacity for nutrient cycling and self-purification (D'Arcy et al., 2007).

Many solutions to the challenges of urban wet weather pollution rightly focus on upstream interventions to increase infiltration and reduce nutrient export (e.g. "sponge-city") (Nguyen et al., 2019). Less focus has been placed on within-water modifications to improve water quality and benefit wildlife (e.g., retrofitting seawalls) (Clifton et al., 2022; Sawyer et al., 2020). Floating treatment wetlands (FTW), where macrophytes are grown hydroponically on a buoyant raft, are one such method. FTW provide pathways for nutrient reduction, removal of contaminants, and the provision of wetland habitat (Pavlineri et al., 2017). Because FTW rise and fall with changes in water-level they do not reduce the storage capacity of urban waters, allowing deployment in flood control waters with flashy hydrology.

The ability of FTW to remove nutrients, and specifically phosphorus (P), has been well documented (Colares et al., 2020; Lynch et al., 2015). However, there is disagreement about the relative importance of the sedimentation and plant uptake removal pathways. Recent reviews papers have attributed the majority (50.8 %) of P-removal to sedimentation (Wang et al., 2020) and concluded "settling, caused by the root system is the main route for P-removal" (Pavlineri et al., 2017). In drawing these conclusions, review studies often draw from lab and mesocosm experiments that lacked proper experimental controls and failed to account for un-aided sedimentation. Furthermore, it is not possible to directly apply the results of these studies, where treatment is quantified as a percentage removal (effluent/influent), to well mixed open waterbodies. Consequently, no consensus exists for sizing criteria to achieving specific water quality goals or attain regulatory compliance in eutrophic waterbodies.

While P-reduction is rightly the focus of most efforts to address eutrophication in freshwater (Schindler et al., 2016) it is not the entire story. Specific aspects of differing waterbodies, including biotic structure, will influence how quickly they respond to reduced loading. Some waterbodies may recover suddenly (Ibáñez et al., 2012; Ibáñez and Peñuelas, 2019), while others remain impaired for decades (Smith and Schindler, 2009) trapped in a feedback loop of seasonal algal blooms (Scheffer et al., 1993), pollution-tolerant non-native species, and internal nutrient cycling (Bond et al., 2015; Carey et al., 2012; Ibáñez and Peñuelas, 2019; Weber and Brown, 2009). In such cases, greater emphasis on biotic structure may be needed (Shapiro, 1980). When reducing P loading fails to curb algal growth, there may be a complimentary role to be played by ecological interventions that support top-down control of algal growth through increased herbivory and filter-feeding (Carpenter et al., 2001, 1995).

An important aspect of FTWs in urban waters is the provision of diversified habitat that might support this improved ecological function. For many urban water bodies native vegetation, both riparian and submerged, are scarce to non-existent compared to less altered aquatic ecosystems. In these cases FTW vegetation may mimic natural wetlands and play multiple roles (O'Hare et al., 2018), providing a refuge for small fish and a forage ground for birds (Karstens et al., 2021), a food source for larval insect (e.g. lepidoptera) (Gross et al., 2001), and a complex physical environment on both the micro and macro-scale (Dibble et al., 2006). Regionally, FTW might serve as 'stepping stones' to improve habitat connectivity (Karstens et al., 2022). Beyond these immediate benefits, the provision of vegetated habitat may lead to changes in biotic structure that increase food-web connectivity and might help to accomplish top-down control of excess algal growth.

Conceptually, multiple mechanisms exist through which the presence of FTW might achieve these aims. A direct role is played by vegetation which provides an additional pathway for the cycling of nutrients and direct competition with cyanobacteria and nuisance algae (Wang et al., 2020). The suspended root mass and buoyant matrix of the FTW are thought to provide a surface that is readily colonized by filter feeders including bryozoans, freshwater sponges, and bivalves, as well as shelter for microscopic motile organisms (e.g., bacteria, nematodes, zooplankton). Along with larger grazing zooplankton, these organisms may function as herbivorous control of algae as well as enhancing nutrient cycling. FTW may also be a place of increased predation on zooplankton by harboring planktivorous insects and macroinvertebrates (Sagrario et al., 2009), such as midge larvae which feed on rotifers (Leland et al., 2020) and by acting as "Fish Aggregating Devices," attracting a range of species that may seek refuge or improved food sources (Suresh, 2000). While the provision of habitat has been demonstrated under specific conditions (e.g. nesting terns) (Patterson, 2012; Shealer et al., 2006) the impact of FTW for increasing ecosystem function has been poorly described (Francis, 2014; Hwang and Lepage, 2011).

This study seeks to address these knowledge gaps. We combine results from three field scale installations in Boston, Chicago, and Baltimore to (1) establish a defensible P-removal rate and sizing basis for FTW planted with native species and installed in eutrophic waters; and (2) compare biological monitoring data at experimental and reference sites to provide a basis for future hypothesis driven research into how FTW alter biotic structures and may improve ecological function.

2. Methods

2.1. Project locations

The projects described in this study are in Boston, Massachusetts, Chicago, Illinois, and Baltimore, Maryland (Fig. 1.) FTW were installed between 2016 and 2020 and each have had time to establish mature growth of native wetland species (Table 1.). Information on FTW configurations and detailed project descriptions are provided in the SI.

This Chicago study site is in a side canal on the north branch of the Chicago River that flows along the east bank of Goose Island. The canal was constructed by the first mayor of Chicago in the late 1800s. For over a century it hosted heavy industry along its banks, leaving behind a legacy of contaminated sediments. Today it receives stormwater runoff from the surrounding urban drainage. The canal ranges from 1 m deep at the northern end to 2.5 m deep at the southern end and has a width of 24–37 m. Since the 1980's Chicago has been constructing its Tunnel and Reservoir Plan which seeks to store 67.76B Liters of untreated combined storm- and wastewater during wet weather periods to eliminate combined sewer overflows (CSO) events. Two CSO outfalls near the north (upgradient) end of the canal can discharge into the canal during high rainfall events, sometime occurring with as little as a few centimeters of rain in the area.

The Baltimore study site is a branch of the Patapsco River known as Baltimore's Inner Harbor, a highly urbanized, post-industrial waterfront that now serves as a popular tourist area. Its entire shoreline has been developed and hardened with no opportunity for natural wetland creation. Since 2002, the city has been under a consent decree to update aging sewer infrastructure, limiting the occurrence of Sanitary Sewer Overflows (SSOs) that

Fig. 1. Study map showing location of floating treatment wetlands (FTW) and reference locations.

support the eutrophication cycle. The Baltimore floating wetland is positioned in 6 m of tidal water within a historic shipping channel and secured between two piles allowing it to naturally rise and fall with the tidal changes.

The Boston study site is in the lower Charles River, a heavily used urban water body in recovery from decades of CSO discharge (Determan et al., 2021). The Lower Charles River receives excess nutrients through both culverted base flow and wet weather events (Khan et al., 2021) and experiences frequent cyanobacterial blooms (Rome et al., 2021). The Charles floating wetland is installed in a section of the river that is devoid of emergent vegetation and near the epicenter of these blooms.

2.2. Phosphorus removal

2.2.1. Vegetation

Plant harvesting and dry biomass determinations were performed in September of 2021 (Boston) and repeated in August (Boston) and September (Baltimore and Chicago) of 2022. For each sampling shoots (above ground vegetation including stems, leaves, flowers and fruits) were harvested from 3 to 5 quadrats of 0.1 m^2 each. Quadrats were selected to capture a representative sampling of the dominant plant types on each installation. Samples were transported to the lab and dried at temperature between 40 and 60 °C for a minimum of 72 h or until two subsequent weighing separated by 24 h showed a consistent dry mass. Phosphorus content was determined by the University of Maine's Analytical Lab using the dry-ash method measured with ICP-OES.

2.2.2. Sedimentation

Sedimentation was evaluated at the Boston study site by comparing the Total Suspended Solids (TSS) concentration from samples collected using an 80 μm net at the downstream end of the FTW and at the reference site. TSS were measured following EPA standard method 160.2 by filtering samples 1.5 μ pore size filters and drying at 103–105 °C.

TSS is calculated as the total mass increase on the filter divided by the volume of water filtered. For netted samples, true river TSS was calculated by multiplying the observed TSS by the total volume of the netted sample and dividing by the volume of water the sample came from (133 L per pass of the net).

2.3. Biological monitoring & ecological function

2.3.1. Cyanobacteria

Cyanobacteria were sampled at the Charles FTW and at the reference site during the summers of 2020–2022. Cyanobacteria were quantified using light microscopy following a standard operating procedure from the Handbook of Cyanobacterial Monitoring and Cyanotoxin Analysis (Meriluoto et al., 2017). All counts were performed on live samples using a gridded Sedgewick-Rafter counting chamber. During the bloom period a minimum of 40 filaments were counted and cell density (cells per mL) was calculated based on the total filament count, average number of cells per filament, and the number of counted grid squares. The dominant taxa were determined for each count. Outside of the bloom period, when ambient concentrations were low (<500 cells/mL). BFC were enumerated from a concentrated sample collected with an 80 μm plankton net towed through the upper 2 m of water.

2.3.2. Zooplankton

Zooplankton were sampled at the Charles FTW and at a reference site during the summers of 2020–2022. Sample collection was performed using an 80 μm plankton net towed vertically through the top 2 m of water column. Zooplankton analysis and sample preparation followed a standard operating procedure published by the US EPA (Environmental Protection Agency, 2016). Zooplankton were narcotized using carbonated water and left to settle for 18–24 h. After the settling period, samples were decanted to a volume of 10–50 mL. All counts were performed using a gridded Sedgewick-Rafter counting chamber at magnifications of 40×. Abundance estimation and mean body size measurements were performed for four taxa: rotifers, copepods, cladocerans, and nauplius in 2020. In 2021 and 2022 enumeration was performed for eight taxa: asplanchna, small rotifers, calanoid copepods, cyclopoid copepods, bosmina (including chydoridae family), sididae, daphnia (daphniidae, Ilyocryptidae, Macrothricidae, Moinida families), and nauplius. Throughout this

Table 1
Summary of study sites and water body parameters.

Location	Installation Date	Median [TP] μg/L	Size m^2 (ft^2)	Water Body	Parameters monitored
Chicago	2016	690[a]	140–930 (1500-10,000)	Urban Stream, Freshwater	Vegetation biomass and nutrient content, Fish, Benthic Invertebrates
Baltimore	2017	22[b]	37 (400)	Inner Harbor, Saline	Vegetation biomass and nutrient content, Sessile invertebrates
Boston	2020	55,71[c]	71 (760)	River Basin, Freshwater	Vegetation biomass and nutrient content, Zooplankton, Cyanobacteria, TSS

[a] 2020 Summer median (May-Oct) Source MWRD Station 73.
[b] 2019 summer median (May-Oct.)
[c] 2020, 2021 summer median (May-Oct.) source MWRA station 112.

document common names and taxa groupings are written using lower-case (e.g., daphnia), while binomial names are used to refer to specific species or groupings at the genus level (e.g., *Daphnia* sp.). Whenever possible abundance estimation was based on a minimum of 100 individuals and mean body size was derived from measurements of 20 individuals. For scarce taxa, abundance and mean body size were determined on the number of individuals present in 2–5 mL of concentrated sample (10–30 % of the total sample). Zooplankton biomass was calculated for each taxa based on mean body size and using equations and coefficients specified in the EPA SOP. Abundance for zooplankton for the Charles River is reported as biomass abundance (mg/m^3) to better facilitate compare the abundance of small but populous taxa (e.g. small rotifers) with larger less abundant taxa (e.g. daphnia).

Zooplankton were also collected from the Chicago study site incidentally as part of larval fish surveys conducted in 2019. Light traps (clover leaf style traps with four 5 × 130 mm openings; 250 mm in diameter) set for 1 h, an hour after sunset, were used. Traps were fitted with 500 μm mesh filters and used a green cyalume glowstick as a light source. Traps were set weekly from April 22nd through September 24th. Two traps were set at the FTW, two along nearby overhanging vegetation, and two on a nearby floating dock. All individual zooplankton captured after 1 h were preserved in a >70 % ethanol until analysis. Zooplankton were identified into four taxa: copepod, cladoceran, dipteran pupa, and amphipods. Amphipods were scarce in this data set and excluded from further analysis herein. For the Chicago data, zooplankton body length was not measured, this data is reported as "abundance (n)" which corresponds to number of individuals trapped in 1 h.

2.3.3. Benthic macro-invertebrates

Macroinvertebrates abundance and diversity was assessed at the Chicago study site over two six-week study periods between March–June, and June–July 2021. Sampling at the FTW and a control location was conducted using a 9-plate Hester-Dendy sampler with 3 mm spacing between plates. A total of 16 samplers were deployed (10 at the wetland and 6 at the control site. Samplers were deployed suspended 15–30 cm above the benthic substrate. After a 6-week colonization period, samplers were preserved in 90 % ethanol and refrigerated until analysis. Images of the samples were digitally captured using a dissecting microscope. All organisms encountered on each plate were identified using dichotomous keys (Bojic, 2010) and counted. Organisms from nine taxa were present at the two study sites: amphipoda (Scuds), chironomidae (Midge larvae), isopoda (sowbugs), zygoptera (Mayfly larvae), mollusca, gastropoda (snails), oligochaeta (worms), planaria (flatworms), and hirudinea (leeches).

2.3.4. Sessile macro-invertebrates

Sessile invertebrate communities that recruit seasonally in urban waters were collected in two study locations within the Baltimore Harbor. One location was proximate to the FTW and a second was in a comparable stretch of the inner harbor. In April, an array of 10 cm acrylic disks ("biodisks") was suspended from a buoy at a depth of approximately one meter to recruit sessile life. Biodisk arrays consisted of vertical disks spaced 2 in. apart on a ½ inch PVC pipe. Each month May through December, three disks from each array were sampled by dividing the disk into four arbitrary quadrants. A video clip of each quadrant was recorded for 30–60 s at 7× and 20× magnification under a dissecting microscope (Olympus) and Sony A7W Mirrorless DSLR Full Frame camera using a parfocal adapter. Disks were kept immersed during recording and videos were archived for later viewing to identify and enumerate the invertebrates in each quadrant. During biodisk reviewing, a minimum of three researchers performed counts and identification to ensure quantity and accuracy. Biodisks were returned to the array following each monthly video-recording. Data collection from the biofilm study began in 2016 and continues through 2022.

2.3.5. Fish

Fish assemblages were assessed at the Chicago FTW deployment and four reference locations with unimproved steel bulkhead walls in 2016–2019. Pulsed direct-current electrofishing (generally 120 pulses sec^{-1} targeting 12–14 amps) was carried out by Metropolitan Water Reclamation District personnel. Each sampling event consisted of electrofishing 152 m of shoreline. All fish encountered were netted, identified to species, and returned to the waterway. Sampling occurred three times each year in the months of July–November. A total of 26 species were identified across all sampling events. In addition to these, four groupings were calculated to aid inter-site comparisons and the identification of ecological trends. These additional groupings were "sunfish" (*Lepomis* sp. including bluegill, green sunfish, and pumpkinseed) and "shiners" (spotfin, golden, and emerald shiners). Additionally, analysis was performed on a grouping of small-bodied prey fish species which included shiners as well the less abundant species: bluntnose minnow, blackstripe topminnow, banded killifish, central mudminnow, fathead minnow, and round goby. This grouping represents species which may benefit from enhanced cover associated with the wetland or utilize it for improved spawning grounds.

2.4. Statistical analysis

All statistical analysis was performed using R version 4.2.1. For paired data, with corresponding samples at the FTW and reference site, statistical differences were assessed using the non-parametric paired sample Wilcoxon test. For unpaired data, where sample sizes were unequal between FTW and reference locations the Wilcoxon signed-rank test was used. P values <0.05 are referred to as statistically significant throughout this analysis. To increase the legibility of this analysis across each data set, bar plots are used to show the mean value from a sampling effort. Standard error bars are included to show the variability of the data defined as the standard deviation divided by the number of samples.

3. Results

3.1. Phosphorus removal

3.1.1. Vegetation

Across all study sites plants were well-established by their second full growing season and exhibited robust growth, flowering, and seed production. Biomass of shoots was determined from a total of 17 quadrats. A complete planting and survival list is included in the Supplemental Information (Table SI-1). At the Chicago and Boston study sites, a wide range of plants grew successfully with strong growth from marsh mallow (*Hibiscus moschetus*) and a other flowering perennials (*Verbena hastata*, *Vernonia noveboracensis*) which reached heights of >1.5 m. At these sites, sedges and rushes (e.g. *Juncus effusus, Schoenoplectus* sp.) formed dense clusters but remained short (1–1.2 m). Shoots from Chicago and Boston grew at similar average densities of 1.8 and 2.1 kg m^{-2}. With individual quadrants varied from a minimum of 0.8 kg m^{-2}, for a sparce section planted with a mix of sedges, rushes, and grasses, to a maximum 5.2 kg m^{-2} from an area planted with marsh mallow but including a large volunteer devil's beggartick (*Bidens frondosa*). For comparison, standing biomass was determined for a mature stand of the invasive *Phragmites australis,* collected from Boston's Muddy River. At that location plant stalks stood at a height of >2 m. and grew at a density ranging between 2.6 and 4.0 kg/m^2. At the Baltimore site this reference was exceeded by dense and tall >2 m stands of *Spartina* sp. which grew at an average density of 7.0 kg m^{-2} (Fig. 2, Table SI-2).

P content was determined for a total of 19 samples (Table SI-3). Across all plant tissue samples, P content was relatively uniform (Coefficient of Variation 67 %, Shapiro-wilks W = 0.92) with an inter-quartile range of 0.070–0.164 mg-P/kg-dry weight and a mean concentration of 1.41 mg-P/kg-dry weight.

3.1.2. Sedimentation

Visual inspection of netted samples collected at the Boston FTW revealed a large amount of debris in the size range of 200–1000 μm. During the summer of 2022, TSS >80 μm were slightly higher at the wetland

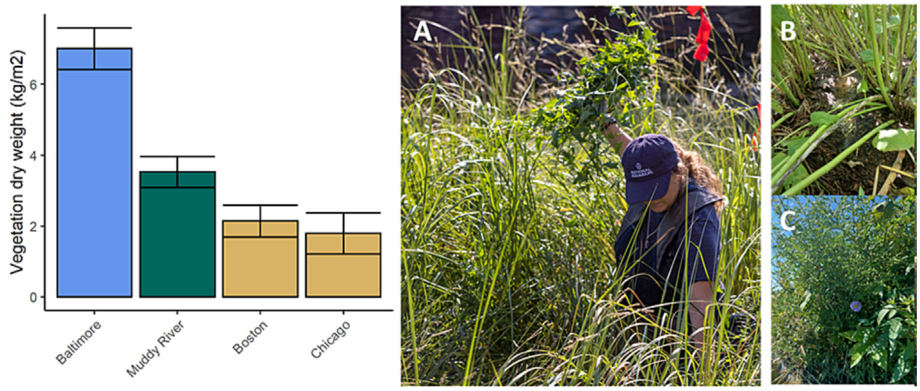

Fig. 2. Comparison of shoots biomass harvested from three FTW locations, and from a mature stand of *Phragmites australis*, harvested from Boston's Muddy River. Images show harvesting of Spartina sp. at the Baltimore study site (A) and illustrate the planting density (B) and height of flowering perennials (C) at the Boston study site.

compared to the reference location (Fig. SI-1). The residual (wetland-control) showed a slightly elevated median and a positive total value (e $=$ 0.006 mg/L, Σe $=$ 0.22 mg/L). This result was statistically significant (Wilcoxon signed-rank test, $p = 0.01$) for the months of July and August, during which the difference was observed in collected samples, but was not significant when considering the entire summer ($p = 0.16$).

3.2. Biological monitoring & ecological function

3.2.1. Zooplankton and cyanobacteria

In the lower Charles River basin (Boston study site), the pelagic zooplankton population changes dynamically over the course of the summer. Over the three-year study period, we observed late-spring and early summer blooms of crustaceans coinciding with an early bloom of diatoms. This was followed by a period of reduced algal and zooplankton abundance. As water temperatures increase bloom forming cyanobacteria (*Aphanizomenon* sp. and *Dolichospermum* sp.) become abundant (Fig. SI-2). During this period zooplankton abundance rebounded with populations dominated by small rotifers and *Asplanchna* sp., Nauplius made an increasing contribution to total abundance toward the end of the summer (Fig. SI-3A, B). When present, daphnia. and bosmina occurred at the highest mean and maximum biomass (Fig. 3).

During the study period (2020 – 2022) at the Boston site, the total abundance of all zooplankton taxa were strongly correlated between the FTW and reference location (Fig. SI-3C) which exhibited similar phenologies (Fig. 4A). In 2020, the first year of the FTW installation in which plants

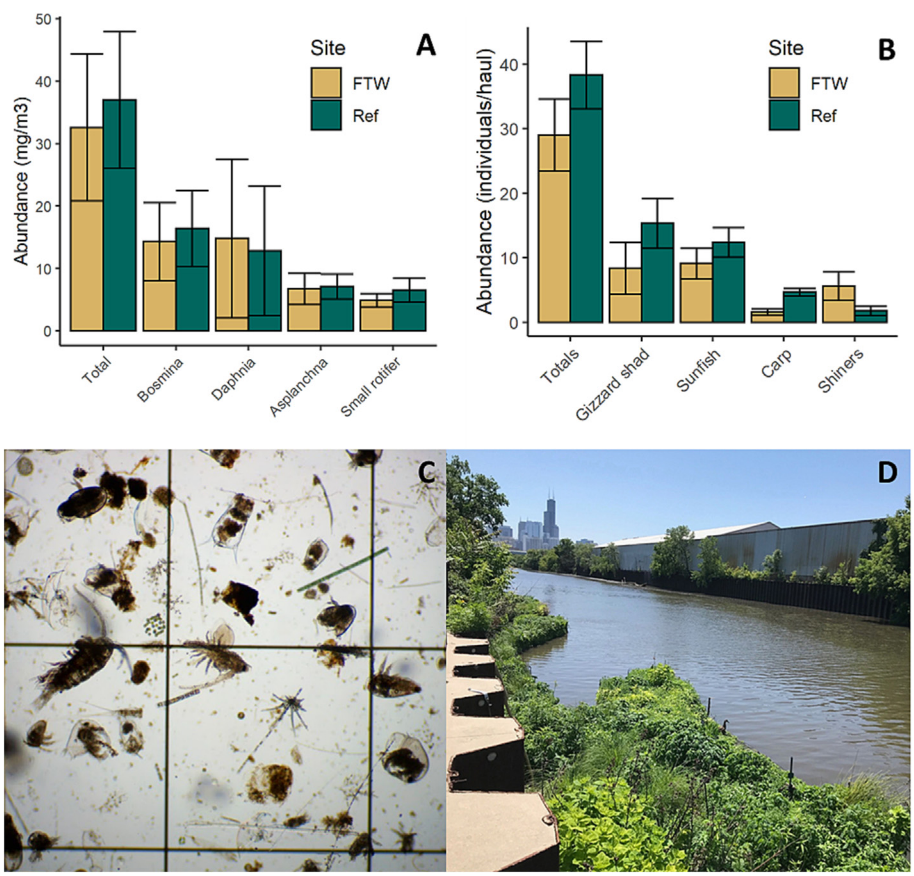

Fig. 3. Mean abundance for most common taxa of (A) zooplankton collected at the Charles River FTW (2021 – 2022) and (B) fish collected at the Chicago FTW. (C) Image of zooplankton from netted sample against 1 mm grid squares. (D) Chicago study site showing FTW and unimproved steel river wall.

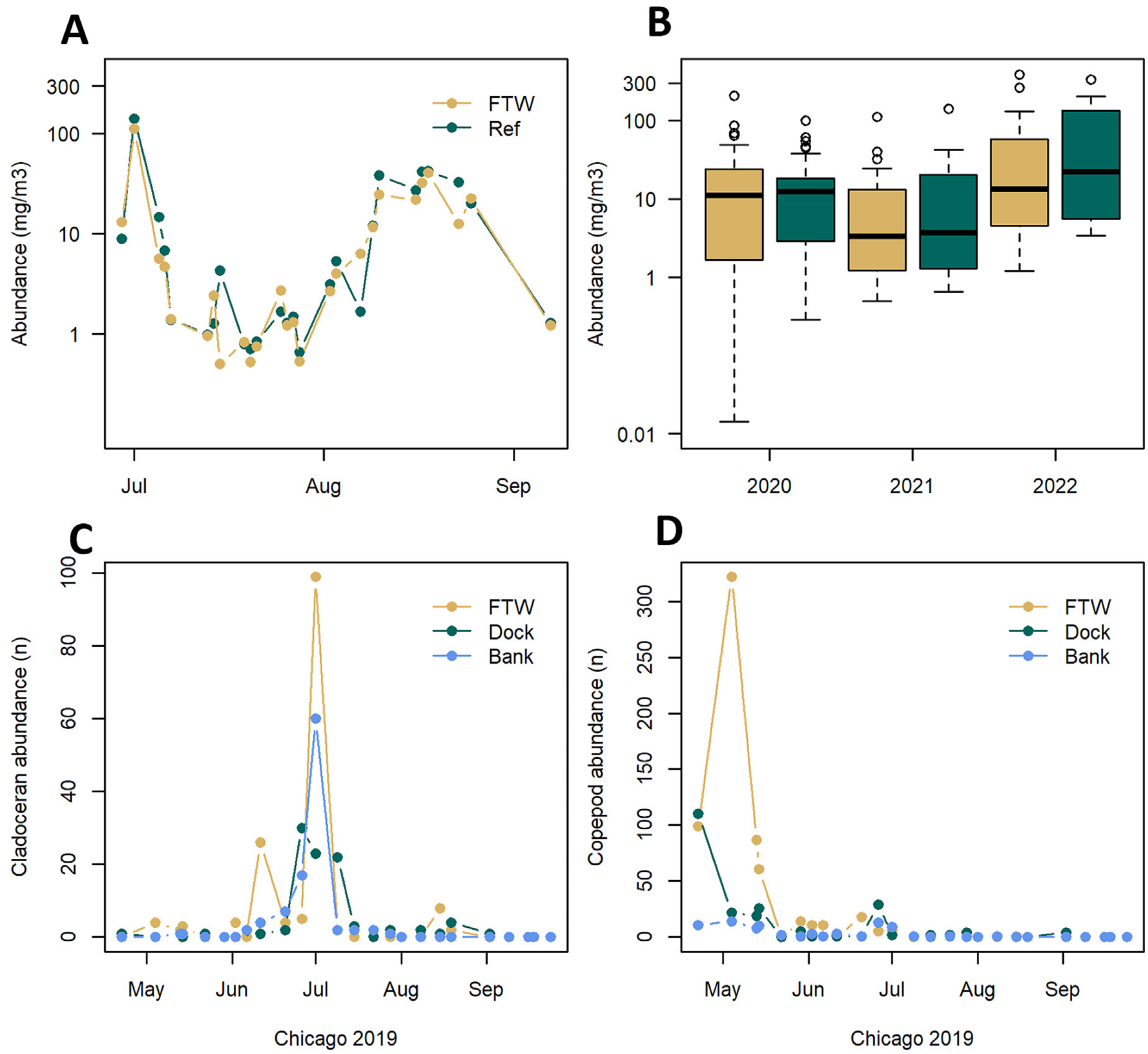

Fig. 4. Fluctuations in total zooplankton abundance at the Boston study sites in 2021 (A) and over the duration of the study (B). Abundance of large bodied (> 500 um) cladocerans (C) and copepods (D) at the FTW and reference locations at the Chicago study sites in 2019.

were not yet established, no statistical difference was found in total zooplankton biomass (mg m^{-3}) between the FTW and reference site ($n = 34$ paired observations, Paired Sample Wilcoxon test: $p = 0.057$). During 2021 and 2022 when plants were well established, on average, wetland samples contained 8.5 % less (relative percent difference) zooplankton biomass compared to the control site (Fig. 4B). This difference was statistically significant for the two-year period ($n = 40$, Paired Sample Wilcoxon test: $p < 0.05$).

A taxa level comparison between the FTW and reference sites showed significant differences between bosmina, asplanchna, and nauplius occurrence. For each of these taxa, abundance was lower at the FTW site (Fig. 3A). Among the largest taxa: daphnia and cyclopods, mean abundance was slightly higher at the FTW compared to the reference site, however this difference was not significant (Table SI-4). Data from Chicago echoes these results, showing that during brief periods of abundance, the FTW was associated with increased peak abundance of large-bodied zooplankton (cladocerans and copepods >500 μm) (Fig. 4C,D). At the FTW the peak and total abundance of copepods was >275 % higher than the highest reference location. For cladocerans, peak and total abundance were > 160 % higher at the FTW. This pattern was not observed for diptera larva which occurred at similar peak abundance across all three reference sites (CV = 13 %). For diptera larva, the greatest total population was observed at the dock, followed by the bank location (Table SI-5).

At the Boston study site, daphnia were observed during the first spring sampling event on 05/16/22, populations grew gradually from moderate abundance (>15,000 m^{-3}) until reaching a peak abundance (>100,000 m^{-3}) four weeks later. The following week, populations collapsed, and no daphnia were observed in netted samples afterwards. Across these 5 sampling events in May–June of 2022, daphnia collected at the FTW had lower mean occurrence compared to the reference site (34,000 vs 42,000) but were slightly larger (631 vs. 516 um) (Fig. SI-5,6). While intriguing this relationship was not statistically significant ($p > 0.5$). However, a direct comparison between the body lengths of 20 daphnia measured at the peak of the bloom showed that daphnia at the FTW were significantly larger at the FTW ($p < 0.05$) while no significant difference was observed for bosmina or small rotifers ($p > 0.5$) which were the other most abundant taxa.

3.2.2. Benthic macroinvertebrates

During the study period, the Hester Dendy samplers for both the FTW and reference site in Chicago were dominated by Amphipoda. In addition to Amphipoda, at the wetland Chironomidae larvae, Isopoda and Planarians were abundant while the samplers from the reference site were primarily colonized by Chironomidae larvae, Gastropoda, and Oligochaeta. A total of 7 taxa were observed at the control site with an additional two taxa (Mollusca and Hirudinea) each appearing in a single sample from the wetland location (Fig. SI-7). Compared to the control location, the wetland had a slightly more organisms, however only the difference in Gastropoda abundance reached a level of significance (Table SI-6).

3.2.3. Sessile macroinvertebrates

A description of the sessile communities that emerged in the vicinity of the Baltimore Harbor FTW and the reference site in 2019 and 2020 are shown in Fig. SI-8. Over the course of each year these populations were dynamic and showed seasonal changes. Early in the summer, smaller organisms, including colonial hydroids and ciliates are abundant. By mid-summer, the growth of barnacles and dark false mussels are evident, with an overlying colonization of colonial bryozoans (Fig. SI-9). Significant differences were observed between the FTW and reference sites among 8 out of 18 taxa (Table SI-7). Most notably the stalked-ciliate vorticella was abundant in both years at the reference site but absent at the FTW. In 2020, anemone were abundant at the reference location but absent from the FTW. In each year stentor, a large single-celled filter feeding ciliate, were among the most abundant taxa collected from the wetland but were scarce at the reference location (Fig. SI-8).

3.2.4. Fish

In Chicago in 2013, a pilot study using subsurface Gee's traps found increased fish abundance across 10 species at a prototype wetland location compared to a nearby dock. While the most abundant species were caught at both locations (e.g. spotfin shiner, bluegill) others including largemouth bass and round goby, were found only at the wetland location. The yellow bullhead and channel catfish were caught only beneath the dock (Table SI-8) (Yellin, 2014).

In 2016–2019 electro-fishing was used to survey the Chicago FTW and four reference sites along the north branch. At each location, gizzard shad were most abundant, followed by sunfish (bluegill, green sunfish, and pumpkinseed), common carp, and shiners (spotfin, golden, and emerald shiners). A comparison between the aggregated reference sites and the FTW site showed similar overall quantities of fish and similarity in the most abundant taxa (e.g. gizzard shad and sunfish) (Fig. 3B). However, the FTW and reference sites differ significantly when it comes to common carp and various shiner species. A statistically significant increased abundance of small-prey fish at the FTW was observed across seven species including three shiner species, bluntnose minnow, blackstripe topminnow, banded killifish, central mudminnow, fathead minnow, and round goby (Table SI-9). For each year (2016–2019) the FTW site had significantly less common carp than the reference site and more of the previously named small prey fish, especially prevalent were emerald and golden shiners, (Fig. SI-10B,C,D)). Throughout the entire dataset there was negative correlation between carp and small prey fish (Spearman $\rho = -0.28$, $p < 0.05$).

With respect to small prey fish, the difference between the FTW and reference sites appears to increase over time with very low abundance in 2016 when <50 m^2 of wetlands are present (only 1 golden shiner collected in 3 hauls), and substantial abundance in 2019 after >280 m^2 of FTW had been installed (average of 17 small prey fish per haul, including >7 golden shiner). 2019 was the first year that the average haul totals were greater at the FTW site compared to the reference sites (Fig. SI-10A).

While this trend is intriguing, the increase in total abundance and specifically small prey fish cannot be definitively attributed to the presence of the FTW habitat. A closer comparison between the FTW and reference sites shows that, while small prey fish are most abundant at the FTW, the difference compared to reference site 1 is not statistically significant (Table SI-10). The same comparison does however show a significant difference between common carp at the FTW and all reference locations (Table SI-11, Fig. SI-11A).

4. Discussion

4.1. Phosphorus removal

In our study, harvesting of native wetland plants represented a viable pathway for P-removal with similar annual removal rates observed across the three study sites. Field data from our Boston study site contained only weak indications of enhanced sedimentation. While our experimental design does not allow us to rule out the importance of this pathway for nutrient removal, our data and review of published literature (Spangler et al., 2019; Wang et al., 2020) does not unambiguously support its inclusion. Instead, area-based removal rates (g-P m^{-2} $year^{-1}$) derived from the growth and harvesting of FTW vegetation should be the main basis of sizing for P removal. From our study the expected removal is ~2 g-P m^{-2} $year^{-1}$ with a possible maximum near 7 g-P m^{-2} $year^{-1}$. This removal rate is comparable to recent studies of full-scale FTW installations on stormwater ponds(> 500 m^2) which have shown plant uptake rates of 0.8–1.63 g-P/m^2 (Schwammberger et al., 2020, 2019) and 3.77 g-P m^{-2} (White, 2021). These rates are also comparable to a results from FTW installations installed for sewage treatment where uptake rates from plant shoots ranged from 2.2 to 12.9 g-P m^{-2} (Huth et al., 2021). Across these studies lower removal rates are observed in water bodies with low P availability (e.g., 60 μg/L for Schwammberger et al., 2019) and higher removal rates are observed in water bodies with higher P availability (e.g., 5000 μg/L in wastewater for Huth et al., 2021).

P-uptake through harvest-able vegetation is the product of plant biomass, and P-concentration. In our study each of these factors varied across study locations and plant species.

Across all sites, mean phosphorus content was 1.4 g-P kg^{-1} dry weight with a variability of 68 % (Coefficient of variation). These values are consistent with a review of non-forest wetland plants which found a similar mean (1.02 g-P kg^{-1} dry weight) and variability (CV = 62 %) across 1253 data sets and126 species (Güsewell and Koerselman, 2002). This value is also consistent with a review of common plant species grown in FTW mesocosm experiments which found a slightly higher mean uptake (1.6 g-P kg^{-1} dry weight) but higher variability (CV = 166 %) among plant tissue harvested from mesocosm experiments with a range of growing conditions and influent characteristics (Table SI-14) (Wang et al., 2020). Average P content was highest in three plant tissue samples from the Chicago site (3.1 g-P kg^{-1} dry weight). Because of the small sample size ($n = 3$) it is unclear whether this difference is caused by increased ambient phosphorus concentrations which are strongly correlated with TP removal rates (Wang et al., 2020), or simply reflects natural variability between samples.

Over the last decade, hundreds of studies have demonstrated FTW capacity to remove nutrients from stormwater ponds and eutrophic waterbodies (Colares et al., 2020). However, no consensus exists on the magnitude of their benefit or on the best method for the sizing and design of FTW installations to achieve specific water-quality goals (Lucke et al., 2019). A recent review of 18 in-situ experiments showed that on average, FTW installations reduce P concentrations by 18 % (Bi et al., 2019). While percentage removal, is useful for comparing the results of multiple studies this number obscures a number of important factors in experimental design including nutrient concentration, duration, growing conditions, and the size of the installation relative to the flow and volume of the waterbody (Wang et al., 2020). An alternate method of comparison is to describe an annual or daily mass removal rate based on area. This method controls for differences in experimental duration and installation size. It can be more easily applied to sizing wetlands to meet regulatory targets that are often expressed in terms of annual mass reduction (e.g. kg-P per year).

A review of multiple studies estimated the P-removal of FTW in eutrophic landscape water as 0.04 (± 0.04) g-P m^{-2} day^{-1} (Wang et al., 2020). Over a 200-day growing season this translates to an annual removal of 8 g-P m^{-2}. Because nutrient removal cannot be measured directly in an open waterbody, this result is extrapolated from multiple studies of variable duration and designs across lab and mesocosm scales. It is critical to understand whether these findings are (1) accurate and (2) applicable to larger open waterbodies where quantifying, and attributing p-removal through sedimentation is difficult.

For example, one important question is whether any P-removal attributed to the presence of a FTW would have occurred in its absence. In mesocosm studies, this question can be answered by using experimental controls, however this is not possible in free-flowing systems. When controls are present in mesocosm or lab set ups, the treatment performance of the FTW is often expressed in terms of gross P-removal and not as the net removal compared to the control treatment (Eq. (1)).

$$\Delta P_{net} = \Delta P_{FTW} - \Delta P_{Control} \tag{1}$$

Of eight studies used by Wang et al., 2020, to estimate wetland nutrient removal rates (g-P m^{-2} d^{-1}), only three included an experimental control. These controlled experiments showed mixed results with respect to P removal. In some cases the FTW treatments contributed substantially increased P-removal compared to unplanted control (e.g. 0.17 vs 0.11 g-P m^{-2}) (Lynch et al., 2015), while in others the difference was quite small or statistically insignificant (Duan et al., 2016; Keizer-Vlek et al., 2014; Wu et al., 2016) (Table SI-12,13). Experiments that include controls have documented substantial un-aided P-removal (24–50 %) in control replicants and have concluded that plant uptake, not sedimentation, accounted for the majority of P-removal (Spangler et al., 2019). While under experimental conditions, FTW root mass can slow down flowing water and decrease turbidity (Borne et al., 2021), from this analysis it appears that FTW are often wrongly credited for P-removal from naturally occurring sedimentation. Consequently, the overall treatment efficacy of FTW is often overstated.

In addition to these attributional errors, quantifying FTW associated sedimentation poses fundamental challenges. First, if enhanced sedimentation is observed in the vicinity of a wetland how can we be confident this material would not have naturally settled elsewhere in the absence of the wetland? Second, what is the long-term fate of settled material; will phosphorus removed from the water-column be stored-in or re-released from the benthos? For these reasons, P-removal through plant uptake and harvesting remains the most straightforward pathway.

4.2. Ecological function

In our study, modest but measurable ecological changes were observed when comparing floating treatment wetlands to reference sites. Decreases in zooplankton abundance (Boston) and increases in native minnows (Chicago) suggest that the wetlands might serve as both a food source and refuge for small fish. Small differences in benthic (Chicago) and sessile macro-invertebrates (Baltimore) between the FTW and reference location may reflect a changed depositional environment.

Zooplankton, for example, are a key link between phytoplankton and higher organisms. In many oligotrophic waterbodies herbivory from large zooplankton grazers achieves top-down control on the growth of algae. In eutrophic waterbodies this control is curtailed resulting in excess algal growth and a shift from larger-bodied cladocerans such as *Daphnia* sp. to smaller taxa such as *Bosmina* sp. and rotifers (Ekvall et al., 2014; Urrutia-Cordero et al., 2016, 2015).

Whole-lake studies have shown that changes in biotic structure (e.g. the removal of zooplankton eating minnows or the introduction of piscivorous fish) can, at least temporarily, reverse these changes (Estes et al., 2011; Søndergaard et al., 2008). In waterbodies where larger zooplankton are shielded from predation, phosphorous loading does not necessarily lead to excess algal growth (Carpenter et al., 1995; Carpenter and Pace, 2018).

The ability of vegetation to act as a refuge for larger zooplankton and contribute to the top-down control of algal growth has been documented (Timms and Moss, 1984). FTW may mimic this dynamic in urban waterways where vegetation is otherwise scarce or absent. The increased abundance of large bodied Cladocerans and Copepods at the Chicago site, and the increased body size of daphnia at the Boston site may reflect such a change. These parameters: abundance of larger zooplankton and daphnia body size, are often cited as important indicators of predation and trophic status (Carpenter and Pace, 2018; Scheffer et al., 1997; Tuvikene et al., 2017). In our study these changes were ephemeral, observed only during the brief period in the summer when large zooplankton became abundant. None-the-less, these changes are notable especially given the small size of the FTW compared to their respective waterbodies, and the presence of noted planktivores (e.g. golden shiner) (Carpenter et al., 1995).

Another consideration is the effect FTW have on the occurrence of native and invasive species. In Chicago, the FTW location was associated with an increase in native minnows and a decrease in non-native common carp. In theory, small prey fish may have been attracted to improved forage or to refuge from predation. The local native predator, largemouth bass, were common across all locations suggesting that predation pressure is at least present. The FTW may also provide spawning habit for species otherwise lacking suitable habitat. For example, spotfin shiner are known to seek out crevices in woody vegetation to hide their eggs, it's possible they are finding suitable habitat in the crevices of the FTW or the floating roots. Notably, the banded killifish, a species which is listed as state threatened in Illinois, was observed most often at the FTW site. Over 4 years of electro-fishing the banded killifish was encountered 5 different times, 4 of which occurred in hauls from the FTW site. While there is no clear rational why FTW might decrease the abundance of common carp the negative correlation between carp and minnows suggests the possibility of interspecies competition. Intriguingly, the pattern of more minnows less carp, echoes broader trends seen among Chicago's fish assemblages over the past three decades as water quality has improved (Happel and Gallagher, 2021). Taken as a whole, these findings in zooplankton, sessile and benthic invertebrates, and fish suggest that while FTW might increase habitat diversity and food-web connectivity. The measurable increases in abundance and mean body size of large zooplankton suggest a viable mechanism by which FTW might contribute to the top-down control of excess algal growth. A key question for future research is whether this effect becomes more durable (allowing daphnia to persist later into the season) as FTW installations are scaled up achieving >10 % for a given river section or basin.

4.3. Limitations of this study

Studying the effect of any ecological intervention in an open waterbody is a significant challenge. Seasonal changes, spatial heterogeneity, and dynamic populations make comparisons between experimental and reference sites quite challenging. In some waterbodies, such as Boston's Charles River, mixing, low flows, and a relatively homogenous physical environment may result in a high level of similarity between two reference locations. In other waterbodies, such as the North Branch of the Chicago River, complex bank conditions and changing water-quality along the length of the river create challenges for direct comparison.

Because rivers and bays are open, flowing, and well mixed, biotic changes that may occur due to experimental changes will also be influenced by surrounding conditions. A local change in zooplankton abundance may be counter-acted by migration of organisms in and out of the experimental site. In our study, biotic communities varied not only throughout the year but substantially between years. A large amount of paired data ($n > 30$) is needed for comparing experimental and reference sites where the effect size may be quite small (<10 %). For this reason, hypothesis driven data collection is needed to supplement periodic monitoring and sampling. For example, the mean body size of daphnia is highly influenced by the presence of planktivorous fish. To evaluate the importance of fish predation additional data should be collected during the late-spring early summer when daphnia are present. Ideally this data would be directly paired with fish collection.

4.4. Recommendations for future studies

Understanding the habitat value and ecological impact of FTW on urban waters is an important goal. Because FTW installations are likely to be small compared to the waterbodies in which they are installed, monitoring these changes requires long-term data collection that would ideally compare an experimental site to a reference site before and after intervention. This study has produced specific findings that can be used to develop hypothesis driven research:

- **Sedimentation**: Our relatively small sample size and methods of data collection were inadequate for fully assessing the importance of FTW enhanced sedimentation. In future studies, sediment traps deployed at FTW and reference sites throughout a season should be used to quantify FTW associated detritus and assess its durability as a store of phosphorus.
- **Habitat:** FTW clearly provide habitat for a range of organisms. Some organisms, like native plants, insects, and sessile invertebrates live directly on the wetland. Other organisms, such as damselflies, moths and butterflies, and birds may visit the wetland using it for forage, shelter, or egg-laying. Specific comparisons should be made between FTW installations, existing urban wetlands, and reference locations to quantify the value of FTW on supporting native wetland species in urban rivers.
- **Scale of impact:** Our study demonstrates that FTW can have a localized effect on biotic assemblages. However, the effect of these installations on the scale of a river segment or throughout an entire waterbody is unknown. Larger installations, paired with new measurement methods like sonar and eDNA, offer an opportunity to assess these effects.
- **Changes in key species:** Our study suggests that changes in fish assemblages and increases in large-bodied zooplankton at the FTW site resemble changes associated with improving water quality. Further studies of key species are needed to understand whether FTW can create pockets of improved aquatic health that anticipate slower processes of ecological restoration.

5. Conclusions

FTW have a clearly demonstrated capacity to absorb nutrients through accumulation in plant tissue. While some additional debris was observed at the FTW site, neither mesocosm studies nor field observations clearly establish FTW enhanced sedimentation as the primary pathway for nutrient removal in eutrophic waterbodies. Instead, for FTWs designed for P reduction, removal rates from harvesting of plant biomass (g-P m^{-2} $year^{-1}$) should be the main basis of sizing. Our study demonstrates that a 500 m2 (5380 ft^2) established FTW can be expected to remove between ~1 kg-P/year. While plant nutrient content may be variable depending on species selection and growing conditions, the most important factor affecting P removal will be the standing density of harvestable vegetation. In our study, a dense planting of saltmarsh (spartina sp.) achieved 2-4× greater vegetation biomass than wetlands planted with a diverse range of herbaceous perennials. While intriguing biotic changes can be seen when comparing FTW to reference locations, these alterations do not support the idea that food-chain changes associated with wetlands should be considered as pathways for nutrient removal.

FTW can be used to locally increase the abundance of wetland plants and to create pockets of native vegetation where none previously existed. FTW habitat is used by a wide range of species including aquatic insects, moths and butterflies, and foraging birds. Beneath the surface suspended plant roots create a unique environment providing cover and forage for small minnows and a substrate for filter feeders.

Results from pilot scale projects suggest that wetlands are capable of locally modulating biotic assemblages compared to reference locations. Locally, wetlands may be associated with a reduction in the biomass of pelagic zooplankton, increases in native minnows, and changes in the abundance of benthic and sessile invertebrates. Observed changes in abundance and size of large-bodied zooplankton suggest a possible contribution to top-down control of harmful algal blooms and raise the possibility that FTW can be used to provide islands of refuge for stressed organisms in recovering waterbodies. Additional research is needed to investigate whether larger scale installations can produce durable reductions in algal growth and contribute to the recovery of native species.

CRediT authorship contribution statement

McNamara Rome: Conceptualization, Formal analysis, Writing – original draft, Visualization, Investigation. **Austin Happel:** Investigation, Writing – review & editing. **Charmaine Dahlenburg:** Conceptualization, Investigation. **Phil Nicodemus:** Conceptualization. **Eric Schott:** Conceptualization, Investigation. **Stephanie Mueller:** Investigation, Formal analysis. **Kathryne Lovell:** Investigation. **R. Edward Beighley:** Supervision, Writing – review & editing, Conceptualization, Methodology.

Data availability

Data will be made available on request.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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