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Evaluation of Methods and Results in the Braddock Bay Wetland Restoration Project

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Evaluation of Methods and Results in the Braddock Bay Wetland Restoration Project

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Dedication:

I dedicate this paper to my parents, Dieter and Charlene Silva, for all of their guidance and support in life and for always encouraging me to further my education.

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Table of Contents

ABSTRACT

Prior to restoration, Braddock Bay was an open embayment wetland on the southern coast of Lake Ontario, and it is part of the Rochester Embayment Great Lakes Area of Concern (AoC). Braddock Bay was partially protected by two spits that are remnants of the protective barrier beach that has slowly been eroded over time. Without the barrier to protect the shoreline within the bay, the coastal wetland was severely impacted by wave action from Lake Ontario, leading to loss of 43 hectares of wetland. The erosion of the barrier was facilitated by water-level regulations implemented in the late 1950s. A further consequence of water-level regulation was the loss of diversity, as the lack of periodic low water levels resulted in a cattail monoculture and the loss of sedge/grass meadow habitat. Braddock Bay is being restored by the United States Army Corps of Engineers. The plan called for the following: restoration of a portion of existing cattail-dominated wetland by cutting cattails in August (when storage carbohydrates in rhizomes are minimized) and herbicide treatment of new stems; channeling and potholing to improve wildlife access to the wetland; the re-creation of the historical barrier beach using rubble-mound and sand; and the creation of spoil mounds along the channels and potholes to increase the elevation in these areas and discourage the growth of cattail while supporting the growth of sedge/grass meadow species. Two years of data collection were performed following construction activities in 2016. Preliminary surveys showed an increase in an invasive species of concern (purple loosestrife) from year 1 to year 2 across the restoration site. A decrease in cattail across the years was observed in the cattail treatment areas, along

with a slight decrease of *Typha* found in the sedge/grass meadow and spoil mound habitats. Based on this monitoring, construction standards set for the restoration must be met, and adaptive management must occur throughout the project timeline for restorations to succeed. Site-level weighted mean C metrics are recommended for future floristic analyses based on an observed species richness influence on FQAI.

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

1. INTRODUCTION

Wetlands have historically been altered to fit human needs. These alterations have brought about changes in biodiversity and species composition of plants and animals, along with impacts on bird migration and reproduction. More attention is being brought to this issue as the public and government officials become increasingly aware of wetland benefits, like recreational activity, flood mitigation, and water-quality improvement. The alteration or loss of key wetland habitats can be detrimental and can ultimately bring about death and disease to humans and plant and animal communities. Some wetland restoration efforts aim to restore an area to its full historical state, but many aim to transform the impacted area into a sustainable, stable ecosystem that benefits both recreation and wildlife. Techniques to bring about a diverse ecosystem are different for each individual site, making every restoration unique, depending on what needs to be restored. Restoring and aiding in the creation of new wetlands helps maintain benefits, with the intention of sustaining a more diverse ecosystem.

Over 80 species of fish use Laurentian Great Lakes coastal wetlands for a portion of their life cycle, and more than 50 species of fish are dependent on coastal wetlands for the entirety of their lifecycle (Jude and Pappas 1992). In addition to providing important habitat for a large variety of animal and plant species, including some that are rare, threatened, or even endangered, wetlands possess key recreational and economic benefits. Billion-dollar recreational and commercial industries within the Great Lakes region supply jobs, while these industries continue to be supported by the many species that use coastal wetlands (Austin 2007, ASA 2013). Coastal wetlands are also important because of their ability to act as a protective buffer for shorelines throughout the Great Lakes, reducing property and coastal damage from wave action and high water levels. Great Lakes coastal wetlands experience many stressors, including human development, fragmentation, drainage, pollution, and invasive species. Less than 50% of the historical Great Lakes coastal wetlands remain intact today (Krieger 1992). This major loss of wetlands calls for the need to restore degraded areas to their natural functions or to protect existing wetlands from further or possible degradation.

1.1 Water Levels and Regulation

The Laurentian Great Lakes have water-level fluctuations ranging from wind-driven seiches that occur daily to seasonal, annual, and decadal fluctuations that reflect the annual water budget and climatic influences through these cycles (Baedke and Thompson 2000, Johnston et al. 2004, Wilcox et al. 2007). Water levels in Lake Ontario follows a climate-driven pattern similar to that of lakes Michigan and Huron that follow about a quasi-periodic 33-year cycle on top of a larger 160-year cycle (Figure 1a; Thompson and Baedke 1997, Baedke and Thompson 2000). However, the Moses-Saunders hydroelectric dam, which until recently was managed under regulation Plan 1958DD, disrupted these natural patterns in water level fluctuations (Wilcox and Xie 2007). Implemented in about 1960 by the International Joint Commission (IJC), Plan 1958DD reduced lake levels during periods of high water supply and retained higher lake levels during periods of low water supply (Wilcox et al. 2008), with higher summer levels and lower fall, winter, and early spring water levels. Environmental, coastal development, and recreational boating interests, along with traditional navigation, hydropower, and municipal uses were all involved in the adoption of this plan (NRC 2006). The loss of hydrologic variability from Plan 1958DD played a large role in the alteration of wetland plant communities in Lake Ontario. Recently, the IJC approved Plan 2014 for regulating the water levels and flows in Lake Ontario and the St. Lawrence River (IJC 2017A). Plan 2014 was designed to provide more natural variations in water levels to aid in restoring the health of Lake Ontario ecosystems while continuing to mitigate the most extreme highs and lows. Plan 2014 began operation in January 2017, and although it may help restore Lake Ontario wetlands, the previous plan resulted in large areas of cattail-invaded wetlands along the coast that will likely not be reduced by the new regulation plan alone. The previous plan with stable water levels reduced the biodiversity within the bay, especially within the shallow emergent marsh which was invaded by *Typha* and subsequently leading to a decrease in overall area of sedge-grass meadow (Wilcox et al. 2018).

1.2 Vegetation and Invasion Dynamics

Coastal wetland vegetation dynamics have not received as much attention as other, more common, non-tidal marsh systems (Dawe et al. 2000). Great Lakes plant-community dynamics are driven primarily by a periodic lake-level cycle linked to climate cycles (Wilcox 2004). In the Great Lakes, water-level fluctuations function as natural disturbances that promote plant community diversity (Grubb 1977, Huston

1979, Keddy and Reznicek 1986). Without disturbance, monocultures of taxa like *Typha* (cattail), *Phragmites* (common reed), or *Lythrum* (purple loosestrife) can form that degrade the quality of wetlands for fish and other wildlife. For example, wetlands dominated by a cattail monoculture provide few distinct habitats and relatively few available niches, decreasing the amount of potential diversity for the wetland (Frieswyk and Zedler 2007, Wilcox et al. 2008, Bansal et al. 2019). Periodic high water levels in the lake kill dense emergent vegetation and invading shrubs and trees, with lower lake levels allowing for understory species to grow from the seed bank (Keddy and Reznicek 1986, Wilcox et al. 2007). Replenishment of the seed bank relies on individual plant species and their ability to emerge and reproduce based on their environmental resilience to water-level changes. Plant physiology and the unique hydrologic cycle are both driving forces of plant diversity within Great Lakes coastal wetlands (Kozlowski 1984, Wooten 1986). Alongside understory plant regeneration benefits, wetland birds and amphibians prefer particular marsh conditions with varying amounts of interspersion between emergent vegetation and open water (LaPan 2015).

High densities of invasive cattail (mostly the hybrid, *Typha* × *glauca*) have dominated Lake Ontario coastal wetlands following lake-level regulation and continue to spread at the expense of native wetland vegetation. Research has shown that the widespread invasion and dominance of cattail across the Great Lakes is attributable to hydrologic modifications - in this case, management of Lake Ontario's lake levels related to operation of the Moses-Saunders hydroelectric dam (Wilcox et al. 2008). Without cyclical low water periods, increased growth of *Typha* in marsh habitats, such as the sedge/grass meadow marsh, is common and can lead to a decrease in plant diversity (Wilcox et al. 2008, Vaccaro et al. 2009). Dense stands of *Typha* impact many aspects of coastal wetland functioning, including alteration of the rate of nutrient cycling (Farrer and Goldberg 2009), inhibition of germination or growth of native species via increased dead cattail buildup (Vaccaro et al. 2009), and a reduction in habitat quality for various fish and wildlife species (Craigie et al. 2003, Crane et al. 2015). Studies have shown an upslope migration of cattail into meadow marsh in Lake Ontario (Wilcox et al. 2008), suggesting that these changes are due to system-wide, water-level regulation (Wilcox and Xie 2008). Larger *Typha* outcompetes other plant species along the wetland elevation gradient. The upslope migration of invasive cattails has narrowed the area where meadow marsh can occur, drastically reducing meadow marsh habitat throughout Lake Ontario (Wilcox et al. 2008).

Dense *Typha* stands also have the potential to change the response of wetland vegetation to restored water-level fluctuations that historically maintained diverse wetland plant communities (Frieswyk and Zedler 2007). However, *Typha* is not the only invasive species that is transforming wetland composition and function. Other invasive species that have expanded into these coastal wetlands include emergent common reed (*Phragmites australis*) and purple loosestrife (*Lythrum salicaria*), along with aquatic invaders like European frogbit (*Hydrocharis morsus-ranae*) and water chestnut (*Trapa natans*). These various invasive species present in the bay are also impacted from natural water level fluctuations. Common reed forms extensive stands that shade out the species trying to survive below (Hara et al. 1993). Purple loosestrife forms clonal colonies, some of which may result in native food and cover plant species being crowded out (Hovik et al. 2011). Both European frogbit and water chestnut are fast-growing, floating leaved invasives that form a thick mat that prevents sunlight from reaching submersed plants (Methe et al. 1993, Strayer et al. 2003, Zhu et al. 2018).

1.3 Restoration and Monitoring

Ecological restoration is important for returning some function to benefit the future use of an area. The practice of restoring or renewing degraded or damaged habitats through human intervention is key to the future of these ecosystems and every interconnected plant, animal, or microbe (Gann et al. 2006). Restorations can vary widely for purpose and style. In some cases, the aim is to restore a physical barrier to protect the area, where others aim to restore and change the actual wetland to bring back functions or benefits for wildlife. Wetlands provide many beneficial uses for humans, which is why it is imperative to protect the wetlands that are left and restore those that have been damaged (USEPA 2018). These restorations are held together after the work has been done by adaptive management, acknowledging that environmental conditions will always change and that uncertainty is a characteristic of any ecosystem being studied (Stankey et al. 2005). Management of these areas must allow adaptation to the ever-changing environment.

After performing restorations, an essential following step is the continued, long-term monitoring of the sites. These sites will evolve over the years following a restoration; therefore, monitoring should be performed in the 5 to 10 years, or longer, after restoration completion to document the ecosystem response to natural variation adequately (Zedler 1988, Larsen et al. 2003, NOAA 2004, Roegner et al. 2009). Short-term monitoring is beneficial when considering adaptive management, as the initial few years of the restoration can show if any immediate actions need to be taken at the restoration site.

2. STUDY AREA--THE BRADDOCK BAY RESTORATION PROJECT

Braddock Bay is an 860-ha coastal, open embayment wetland (Albert et al. 2005) on the southern shore of Lake Ontario (Figure 2) in the town of Greece, NY and is part of the Rochester Embayment Great Lakes Area of Concern (AoC) (USEPA 2016, NYSDEC 2016A). Braddock Bay has been managed by NYS DEC Region 8 since 1982 and has an ecological and recreational importance to the surrounding Rochester area (NYSDEC 2016B). It has been partially protected from wave attack by two spits that are remnants of the protective barrier beach that was slowly eroded due to wave action from Lake Ontario and loss of sand from littoral drift resulting from shoreline armoring (Figure 3) (USACE 2016A). Without the barrier to protect the shoreline within the bay, there was a loss from erosion of approximately 43 hectares (ha) of emergent wetlands over the past 100 years, decreasing the remaining coastal wetland to about 138 ha (USACE 2016A). Erosion of the barrier was facilitated by Lake Ontario water-level regulation under Plan 1958DD because wave attack on the shoreline occurred within only a narrow elevation range (Wilcox 1993). Two tributaries, Salmon Creek and Buttonwood Creek, connect to Braddock Bay, which in turn, is hydrologically connected to Lake Ontario

The Braddock Bay marsh consists primarily of dense, buoyant *Typha* mats with scattered emergent species that provide limited functional habitat and few benefits for wetland-dependent wildlife and plant life. Remnant sedge/grass meadow marsh only occurs in depauperate patches along the shoreline, where lower soil moisture reduces the rate of *Typha* encroachment (Wilcox et al. 2018). Invasive species of concern in the bay include purple loosestrife, common reed, and water chestnut. Black Tern populations have diminished significantly across the Great Lakes since the 1960s (Wyman and Cuthbert 2017), with historical habitat in Braddock Bay no longer present. *Typha* expansion and the loss of Lake Ontario water-level variability also have reduced the potential spawning and nursery grounds for northern pike (Mingelbier et al. 2008).

Without taking action in Braddock Bay, the diversity and quality of the bay would remain low, and further erosion and loss of wetland area would occur. Restoration activities were conducted at Braddock Bay by the United States Army Corps of Engineers beginning in 2016, with data collection taking place in 2016 and 2017, funded by the Great Lakes Restoration Initiative (USACE 2016A).

2.1 USACE Project Goals

The goal of USACE when conducting the restoration was to improve habitat diversity of the existing emergent marsh currently dominated by cattail and to reduce erosion of the existing emergent marsh, with the overarching objective to delist the Rochester Embayment Area of Concern by addressing the "Loss of Fish and Wildlife Habitat Beneficial Use Impairment (USACE 2016A). As stated by the U.S. Army

Corps of Engineers, the planning objectives were to restore wetland and habitat diversity in Braddock Bay to improve its suitability for fish and wildlife including northern pike, American mink, and the endangered state-listed Black Tern during the planning period of 2015-2065 and protect Braddock Bay wetlands from further erosion during the planning period of 2015 – 2065 (USACE 2017).

Four project constraints to be avoided included negative impact to navigability and operation of marinas within the bay, impacts to nutrient dynamics of Braddock Bay that would worsen eutrophication, negative impacts to the Lake Ontario littoral drift system, and any project activities that would increase the extent of invasive species at the project site (USACE 2017).

To achieve the objectives, the USACE, with the planning assistance of The College at Brockport Wetlands Lab and NYSDEC Region 8, decided to perform a multi-measure restoration that would include: 1) re-creation of the barrier beach to protect the remaining wetland from erosion, 2) excavation of nearshore channels and shallow potholes through the extensive cattail mat on both sides of the bay to facilitate shoreline access to sedge/grass meadow by northern pike, 3) placement of excavated spoil material to create elevated mounds adjacent to the channels and potholes at a certain elevation to hinder cattail growth, and 4) chemical and cutting treatment of *Typha*, with timing and methodology following the research of Wilcox et al. (2017). These were just the primary steps of the restoration, with the key focus of bringing native diversity back to the area with hopes to delist the Rochester Embayment AoC by improving fish and wildlife habitat, which is one of the Beneficial Use Impairments listed for the Rochester AoC (USACE 2016A). Additional steps taken by the Army Corps of Engineers were the creation of the historic barrier beach and a new area of emergent marsh.

2.2 Goal of Thesis Project

My goal was to perform vegetative data collection for use in adaptive management and to identify where success was achieved within the restoration. I evaluated the establishment of different created habitat zones that aim to support different communities across the restoration site, with areas of restoration activities expected to result in an increase in species richness and nativeness. I hypothesize that these different created habitat zones would support separate communities. Restoration activities including mound construction and the treatment areas parallel to the sedge-grass meadow will result in an increase in the amount of native emergent marsh plant species and decrease *Typha* cover. Areas of restoration will have a higher floristic quality, demonstrated by greater assessment scores than untouched cattail-dominated control areas. Lastly, post-restoration, all habitat zones will continue to floristically improve via species richness as the wetland rebounds back to its natural state.

The objective of this study was to evaluate the different aspects of the Braddock Bay restoration project through vegetation surveys and the use of floristic quality assessment index (FQAI) to determine potential success, per contract requirement from the U.S. Army Corps of Engineers. To compare floristic quality between habitats across the years, assessments were based on an FQAI statistic, the standard mean Coefficient of Conservatism (CoC or “C") score, and the cover-weighted mean Coefficient of Conservatism scores. Comparisons of FQAI, mean C, species abundance, and weighted mean C were made across the different grouped zones and from overall year to year.

2.3 Restoration Methods and Standards

I describe this restoration and the specific standards used for the project based on the project goals and reasoning within the U.S. Army Corps of Engineers mission statement and scope of work (USACE 2016A; 2016B). The main concern for Braddock Bay has been erosion of the historical barrier beach, leading to the further erosion of wetland habitat within the bay. As noted above, to combat this, in 2016, the U.S. Army Corps of Engineers re-created the historical barrier beach, which included a 0.52-km-long, continuous rubblemound breakwater spine, a one-hectare headland beach, two 45.7-m-long headland rubblemound breakwaters, and two 24-m-long rubblemound terminal groins (USEPA 2018). New wetland habitat was also created to further improve vegetative diversity and wildlife habitat, but this portion of the project was not created in time to be included in sampling.

Further efforts to restore the bay included digging access channels and habitat potholes, creating habitat mounds, *Typha* control, and sedge/grass meadow restoration (Figure 4). Digging of potholes (Figures 5a and 6) and channels (Figures 5b and 7) aimed to increase the topographic heterogeneity of the wetland, increase the vegetative diversity, and improve habitat suitability for key fish and wildlife species (USACE 2016A). These newly dug channels and potholes, *Typha* treatment areas, and a change in water-level regulation can provide essential habitat and greater access to the wetland for fish and amphibian species in the bay. Cattail-control measures were performed within the sedge/grass meadow treatment areas parallel to the shoreline. This process included cutting cattails when storage carbohydrates in rhizomes were minimized (early August, Wilcox et al. 2017), followed by an herbicide treatment of new stems by hand-wicking with Glyphosate (Rodeo) (Wilcox et al. 2017). Alternating conditions of flooding and dewatering are important for generating diversity in the plant community, along with benefitting certain target species, such as northern pike (*Esox 12ucius* L.), which use the shallowly flooded sedge/grass meadow in early spring to spawn (Crane et al. 2015, Wilcox et al. 2017). After the restoration, the Braddock Bay vegetative community should be more ecologically diverse than the previous *Typha*-dominated emergent wetland and should provide higher quality habitat for many species of fish and wildlife, including American mink (*Neovison vison* Schreb.) and, hopefully, the state-listed endangered Black Tern (*Chlidonias niger* L.), which are target species under the AoC listing (USACE 2016A).

2.3.1 Construction and Excavation Timing

The project began with the channel and pothole excavation and spoil mound placement, concluding in March 2016. In June, the spoil mounds, pothole benches, and channel benches were seeded and planted with vegetative plugs and regionally-sourced seed (Table 1). About 25,000 plugs were planted in the emergent marsh, 38,000 were planted within the sedge-grass meadow, and 8.97kg/hectare of seed were applied within both of these habitats as well. *Typha* was mechanically cut in July, with corresponding chemical treatment with Glyphosate (Rodeo) in September. In August

2016, the barrier beach stone placement was completed, with no further work done in 2016. At this point, the main wetland restoration was basically finished, with the completion of the channel and pothole excavations, the completed seeding and planting, and the base of the barrier beach in place to begin halting wave action. No construction work was completed in 2017 due to high-water conditions. The project resumed in May 2018, with barrier beach sand placement followed by the newly created marsh habitat in June. The second season of *Typha* cutting occurred in August, with the official completion of the Braddock Bay Ecosystem Restoration Project in November 2018. In May through August of 2016 and 2017, I studied the vegetative composition of these channel, pothole, and treatment areas, but the newly created marsh habitat was not yet built at the end of my two-year study.

3. METHODS

3.1 Sampling Design

Fourteen transects were used to sample the overall network of newly created channels (Figures 9 and 10) and habitats, each of which spanned six vegetation zones, with two-to-four quadrats per habitat zone per transect (Figures 5b and 7). The sampled channel zones were separated into sedge/grass meadow (SGM), treated (TR), shallow bench (SB), intermediate bench (IB), mound habitat (M), and channel (C) zones (Figure 7). The 12 newly created potholes and associated spoil piles were sampled with 16 transects (Figures 8 and 9), spanning three identified zones, with one to two quadrats per zone per transect (Figures 5a and 6). Transects ran from the deepest portion of the potholes to the base of the back slope of the spoil pile. The sampled pothole zones were separated into the deep water zone (D), bench (B), and mound habitat (M) (Figure 6). Control quadrats were haphazardly placed within the cattail mat.

3.2 Sampling Methods

The field sampling methods used were developed during the feasibility study and pre-defined in the scope of work for the project in the contract with the U.S. Army Corps of Engineers (USACE 2016B). Multiple steps were taken to ensure that the vegetation surveys covered the entire Bay and each aspect of the restoration. Vegetation surveys in Braddock Bay began in 2016 and were conducted again in 2017 using Chadde's *A Great Lakes Wetland Flora* and Newcomb's *Wildflower Guide* guidebooks as tools to identify individuals to species level. I conducted early season walking surveys in late May and early June 2016 to obtain presence and abundance data for invasive species that are growing in restored areas (Figure 9). While surveying, I specifically looked for three key invasive species that were considered to be the most problematic – *L. salicaria* (purple loosestrife), *T. natans* (water chestnut), and *P. australis* (common reed). All three target species are on the NYSDEC's 2014 New York State Prohibited and Regulated Invasive Plants list (NYSDEC 2014). Locations of all stands encountered were identified with a GPS, and the radii of the invasive stands were estimated (Figure 10). Stem density was estimated within a 3-m control radius extending from the center of the stand. These stands were classified into the following categories: category 1 ranged from 1 to 10 stems, category 2 ranged from 11 to 20 stems, category 3 ranged from 21 to 30 stems, category 4 ranged from

31 to 50 stems, and category 5 included patches with more than 50 stems. Radius was estimated and both sediment, and water depths were taken for each patch of invasive vegetation with a 3-meter depth pole marked at every tenth of the meter.

Sampling of the channels and potholes took place from late June to late August in both years. Each channel transect aimed to have approximately sixteen 1-m^2 quadrats, with two in the channel, two on the intermediate bench, two on the shallow bench, three on the spoil piles, four in the *Typha* treatment areas, and up to three in the remnant sedge/grass meadow. Some areas could only fit fewer quadrats without overlap. Quadrats were placed haphazardly within each of the different habitat zones by blindly throwing the quadrat (over the shoulder) along all 30 transect lines throughout the newly created habitats.

In the potholes, I surveyed vegetation using eight 1-m^2 quadrats per transect, with three in the non-bench deep zone within the center of the pothole, two in the bench, and three on the mound. Quadrats were placed haphazardly within each of the different habitat zones by throwing the quadrat over my shoulder along the transect line. Thirty control quadrats were also sampled throughout the *Typha* mat in the bay (Figures 8 and 9). All quadrats were sampled in the same manner and included identification to species level, percent cover of detritus and total living vegetation, GPS coordinates (Figures 8 and 9), water depth, and depth to hard sediment. Water depth was measured atop the floating *Typha* mat in these locations.

4. STATISTICAL ANALYSES

As directed by USACE, data analyses focused on the Floristic Quality Assessment Index (FQAI) to evaluate the nativeness of the plant community based on plant species present and their Coefficient of Conservatism scores (Reznicek et al. 2014, Faber-Langendoen 2018). This FQAI statistic is currently a favored assessment of wetland plant community health, with many different ways to calculate the statistic (Faber-Langendoen 2018). I made FQAI comparisons across transects and habitat zones both within first year sampling and across the two-year period. I also made similar comparisons between the pre-restoration sampling year 2013 and post-restoration sampling year 2017. This can provide important wetland vegetation dynamics data within single year comparisons, showing plant communities of an eroding coastal wetland that can then be compared across years to determine the relative first-year impact of barrier beach reconstruction along a Lake Ontario coastal wetland.

For plant data obtained from the channel and pothole quadrats, FQAI was calculated by using the New York State preliminary C-score list with reference to the NEWIPCC Northeast Ecoregional C-score list, developed and agreed upon by a number of regional botanists, funded by the U.S. Environmental Protection Agency (EPA) (NEIWPCC 2011, Reznicek et al. 2014, Faber-Langendoen 2018). Each plant species was assigned a corresponding C-score, and all were averaged to determine the mean C-scores for each quadrat, which were then averaged for each zone and were used to calculate the FQAI and weighted mean C. All data representative of each habitat zone were averaged to account for pseudo-replication due to quadrat proximity within each transect. All non-native species received a value of zero, although any quadrat with zero species observed resulted in habitat zones having no C-score, leading me to delete these missing value cases. The FQAI was calculated as followed:

Equation 1: $$\text{FQAI} = \overline{C_t}\sqrt{N_t}$$

Where C_t represents the total mean C and N_t is the total species richness (Freyman et al. 2016, Faber-Langendoen 2018).

This FQAI statistic is used to evaluate the nativeness of an area based on the plant species present. A problem with many diversity measures is the equal weight that each species receives regardless of fidelity to a specific habitat or tolerance to disturbance. The key component of the FQAI statistic is that the quality of a natural community can be evaluated objectively by examining the degree of fidelity or ecological conservatism of each plant within that community (Andreas et al. 2004, Matthews et al. 2015). A C-score of 0 indicates an exotic or non-native species with a widened range of tolerance in terms of environmental limits, with a score of 10 being a very specialized, narrow range of limits that the specific plant species can handle (Swink and Wilhelm 1994, Taft et al. 1997, Rooney and Rogers 2002, Andreas et al. 2004). Even though using non-native species within FQAI calculations has been criticized, it has consistently proven to be reliable (Miller and Wardrop 2006, Kutcher and Forrester 2018). These FQAI metrics may be useful in determining the type and quality of habitat needed for certain species to thrive (Taft et al. 1997). I then averaged these scores to yield a mean FQAI and mean C-score for each transect, which were then grouped into zones. All of these data were averaged to determine mean FQAI and C-scores for all of the channel transects and all of the pothole transects, respectively. Within each transect, the means for total species and other variables observed in each quadrat and zone were calculated similarly. Quadrat-level cover weighted mean C was calculated as followed:

Equation 2: Quadrat-level cover Weighted Mean C $= \sum_{i=0}^{t} C_i \gamma_i \Big/ \sum_{i=0}^{t} \gamma_i$

Where $\sum C_i \gamma_i$ is the sum of C-scores for each species (C_i) multiplied by the percent cover of each corresponding species (γ_i). This is divided by the sum of the percent cover of each species (γ_i) (Freyman et al. 2016, Faber-Langendoen 2018).

By using all three metrics, I examined the differences between the weighted mean C calculation as a comparison to the FQAI and mean C statistics to determine if one calculation is more susceptible to influence by outside factors for use over the other options. Species richness and sample size have been shown to influence on FQAI but do not greatly impact mean C (Matthews 2003, Bourdaghs et al. 2006). In many cases, excluding non-native species has not been shown to influence the FQAI metric (Bourdaghs et al. 2006, Miller and Wardrop 2006). Comparisons of FQAI, mean C, species abundance, and weighted mean C were made across the different grouped zones and from overall year to year. My aim is to use these indices as a representation of the floristic quality within specified habitats and perform comparisons within the first two years of the restoration.

I used the Kolmogorov-Smirnov normality test to determine whether the sample data had been drawn from a normally distributed population, stating whether the data are normal or not, with a significant p-value cut-off $\alpha=0.05$. After running the test, I found that even after transformation, not all of the data met the assumption for a normally distributed population; therefore, not all parametric test assumptions were met and non-parametric statistics were needed for the analyses. The only data shown to follow a normal distribution between both years of sampling were the species count (SPPcount) of 2016 and all of the 2017 variable data other than water-depth (Appendix 2). The rest of 2016 and the 2017 water-depth data did not follow a normal distribution (Appendix 2). Most of the data were shown to be non-normal even after transformation; therefore, I used non-parametric statistics for all analyses for consistency.

Using SPSS statistics software, after averaging together each habitat zone within each transect, I used a multivariate Generalized Linear Model (GLM) comparing the different habitats within each individual sampling year using FQAI, mean C, and weighted mean C as the response variables, with habitat types as factors, and the random variable of transect. All GLMs used normal distribution and the identity link function. Control quadrats were not included in the individual year-by-year analyses. All habitat types, other than the control, were included in the GLM. I used the same GLM model for other analyses including comparing channel overall channel and pothole habitats across the two sampling years. Further statistics included performing non-parametric independent-sample Kruskal-Wallis one-way ANOVAs to compare variables within the different habitats across the two sampling years. With this, I was able to determine the short-term improvements within the vegetative community based on the variables surveyed, with the ability to run a year-to-year comparison for any variables within any habitat type. I was also able to perform multivariate GLMs comparing dependent variables with the different habitat zones as a fixed factor and a transect block. Additionally, using the Primer 7 statistics program, I performed a non-metric multi-dimensional scaling (NMDS) ordination using the factors FQAI, mean C, weighted mean C, species count, and transect number to compare habitat types within each individual year and identify any grouping (1000 runs for each year, 2016 2D stress =0.11, 2017 2D stress=0.1) .

Using pre-restoration data collected by a previous graduate student, Eli Polzer, I extrapolated an FQAI, mean C, weighted mean C, and species richness statistic for the pre-restoration vegetation data collected in 2013. I used the shallow emergent marsh (SEM) zone data, which are from the surveyed portion of pre-restored Braddock Bay where the cattail treatment first occurred in 2016, primarily testing the effect of cattail removal. Half of these data compared between 2013 and 2017 did not follow the normal distribution (Appendix 3); therefore, I used a non-parametric independent-samples Kruskal-Wallis one-way ANOVA for this analysis. This helped to characterize pre-restoration and post-restoration communities to determine if any changes occurred since pre-restoration.

These KW, GLM, and NMDS analyses can provide the answers to the hypotheses being questioned. The analyses included various habitat comparisons across years to determine if the created habitats were floristically distinct, comparing specific restored areas to provide evidence of change in richness and floristic quality, and to determine if these habitat zones are still floristically improving. Percent cover averages and invasive plant abundance were used to determine if the restored areas were still floristically improving and to give further background into what plant species are most abundant and increasing or decreasing in specified areas.

5. RESULTS

5.1 Water Levels

Water depth along the transects ranged in 2016 (a drought year, NOAA 2016) from 0 to 120 cm and ranged in 2017 (a wet year, NOAA 2017) from 0 to 230 cm. Mean water depth in 2017 (60 cm) was significantly greater than in 2016 (23 cm; $p<0.001$,Appendix 4, Figures 12 and 13).

5.2 Walking Surveys for Invasives

Preliminary walking surveys designed to search for three key invasive species (*L. salicaria*, *P. australis*, and *T. natans*) only found *L. salicaria* in the restored areas. All three key invasive species chosen were present within Braddock Bay before the restoration occurred. I performed these surveys In 2016, about 24.7% of the points collected for *L. salicaria* had greater than the maximum cut-off of 50+ stems, which increased to around 52% of the points collected in 2017. All three invasive plant target species were also observed within 2013 sampling data.

5.3 Plant Taxa Observed

A masterlist of all species observed at Braddock Bay during the two-year sampling period is available as Appendix 1. Species described in the text as dominant were designated by a cut-off of 10 percent within each habitat based on the average percent cover that the species had throughout the year. The only species dominant within the 2013 pre-restoration sampling of the shallow emergent marsh were non-native *Typha* × *glauca* (65.9%) and *H. morsus-ranae* (10.6%).

In 2016, 16 species were dominant; five other species were observed with an average above 8%. In the potholes, comprised of the deep zone (D) and the pothole bench (PHB), native *Utricularia vulgaris* (17.7% in D) and non-native *H. morsus-ranae* (25.1% in PHB) were the only dominant species (Table 2). On the pothole mounds (PHM), two non-native species *L. salicaria* (38.8%) and *T.* × *glauca* (16.7%) were dominant, with a lesser abundance of natives *Galium trifidum* (9.6%) and *Impatiens capensis* (9.4%) (Table 2).

In the channel during 2016, comprised of the channel (C), intermediate bench (IB), and shallow bench (SB), three native species *Elodea canadensis* (10.5% in IB), *Lemna minor* (13.8% in IB), and *U. vulgaris* (10.2% in IB and 22.0% in C), with non-native *H. morsus-ranae* (37.1% in IB and 11.0% in C) were dominant (Table 2). On the channel mounds (M), dominant species included non-native species *L. salicaria* (12.5%) and*T.* × *glauca* (11.6%), with similar abundance of two native, ruderal species *Persicaria hydropiper* (19.8%) and*Persicaria lapathifolia* (11.4%,Table 2).

The 2016 treatment area located within the shallow emergent marsh (SEM/TR) was dominated by only non-native *T.* × *glauca* (22.3%). In the sedge/grass meadow habitat (SGM), overhead cover of *Acer saccharinum* (10.0%) and *Salix fragilis* (10.0%), with non-native *T. x glauca* (10.0%) were all dominant, along with native emergent *Carex lacustris* with an average percent cover of 22.1% (Table 2). The only dominant species within the cattail mat control was non-native *T.* × *glauca* at 51.7%, with native *I. capensis* prominent at 9.7% cover.

In 2017, 15 species were dominant; three other species were observed above 8.0 percent cover. In the potholes, comprised of the deep zone (D) and the pothole bench (PHB), native *U. vulgaris* (22.8% in D and 25.5% in PHB) and three non-native species, *H. morsus-ranae* (28.4% in PHB), *L. salicaria* (10.2% in PHB), and *T.* × *glauca* (26.4% in PHB) were the dominant species (Table 3). On the pothole mounds (PHM), three native species *Decodon verticillatus* (12.2%) and *I. capensis* (11.9%), *P. hydropiper* (19.0%), and the two non-native species *L. salicaria* (24.8%), and *T.* × *glauca* (14.2%) were dominant (Table 3).

In the channels during 2017, comprised of the channel (C), intermediate bench (IB), and shallow bench (SB), three native species *Ceratophyllum demersum* (11.7% in C), *U. vulgaris* (25.9% in IB and 19.5% in C), and *Stuckenia pectinata* (11.3% in IB), along with two invasive species *H. morsus-ranae* (20.5% in IB and 23.3% in SB) and *T.* × *glauca* (37.7% in SB) were dominant (Table 3). On the channel mounds (M), dominant species included two non-native species *H. morsus-ranae* (14.1%)and *T.* × *glauca* (13.4%) and the native species *P. hydropiper* (17.9%), (Table 3).

In 2017, the treatment area located within the shallow emergent marsh (SEM/TR) was dominated by non-native *H. morsus-ranae* (18.3%) and native *U.*

vulgaris (12.1%) (Table 3). In the sedge/grass meadow habitat (SGM), the two native species *Calamagrostis canadensis* (10.7%) and *C. lacustris* (13.4%), with the two non-native species of *H. morsus-ranae* (15.8%), and *S. fragilis* (14.2%) were all dominant (Table 3). The only dominant species within the cattail mat control was non-native *T.* × *glauca* at 51.7%, with native *I. capensis* prominent at 9.7% cover.

5.4 Planted and Seeded Species

In total, 30 species were chosen to be seeded and planted using plugs for the restoration (Table 1). Thirteen out of 30 seeded/planted species were observed in pre-construction 2013 surveys. In 2016, a total of 100 different plants were found, 86 of which were identified to species level, with 14 too immature or too herbivorized to identify. Only 13 of the 30 species on the seeded/planted list appeared in the first year of sampling after restoration (Table 1). The following year in 2017, 94 different plants were observed, with 86 identified to species. In this second year of sampling after the restoration took place, 15 seeded/planted species were found (Table 1). Fourteen unique species were found for the two sampling years combined. If all taxa other than known exotics are considered native, then both years had 75 native taxa and 11 non-native taxa observed. The presence of the seeded and plugged plant species in 2016 and 2017 were mostly the same species observed in 2013 (Table 1).

5.5 Non-native Species Observed

Using the NYS C-score list, described earlier, I selected the species listed as “non-native” and analyzed frequency between the two years, where non-native species were given a “0” C-score (Tables 4 and 5). Both years had an equal number of non-native species (n=11), but there was a numerical decrease in cover for some species. For example, *L. salicaria* had an average cover decrease from 38.8% in 2016 to 24.8% in 2017 on the pothole mound habitats (PHM) and from 12.5% to 3.8% on the channel mound habitats (M). Similar patterns occurred with other non-native species, including *T.* × *glauca*, with a mean percent cover decrease from 65.9% in 2013 to 22.3% in 2016 and finally to 3.3% in 2017 within the *Typha* treatment zone (TR) but also observed with an increase from 7.4% to 26.4% on the pothole bench habitats (PHB) (Tables 4, 5, and 6). *H. morsus-ranae* showed a percent cover increase within the SGM (0.0% in 2016, 15.8% in 2017), TR (0.9% in 2016, 18.3% in 2017), M (0.1% in 2016, 14.1% in 2017), and SB (6.6% in 2016, 23.3% in 2017) habitats (Tables 3 and 4). This is primarily due to the extended flooding of these areas caused by the record high rainfall and water-levels that allowed *H. morsus-ranae* to create a mat and block out emergent plant species.

5.6 Transect Sampling

5.6.1 Overall Mean C, Weighted Mean C, and FQAI

The mean C, weighted mean C, and FQAI scores calculated for each quadrat were averaged within each habitat zone and used to compare vegetative differences along the transects and across years. The overall mean C statistics calculated for year 2016 and 2017 were not significantly different (Kruskall-Wallis $p=0.639$, Appendix 4). The overall weighted mean C statistic from year 2016 to 2017 was significantly different across years and showed a slight increase (2016=2.25, 2017=2.68, Kruskall-Wallis $p=0.002$) (Table 7, Appendix 4). The overall FQAI statistic showed an increase in floristic quality from year 2016 to 2017 (2016=6.31, 2017=6.89, Kruskall-Wallis *$p=0.019$*) (Table 7, Appendix 4).

I then looked at the individual years and the potential for the habitat zones to be different within each year. The generalized linear model showed no significant interaction between habitat zones blocked by transects for any metric in both years, separately (Appendix 6 and 7). In both 2016 and 2017, weighted mean C was found to be significantly different across the habitat types ($p<0.000$) (Appendix 6 and 7) but not significant when looking at FQAI across habitat types (2016 $p=0.068$, 2017 $p=0.044$) (Appendix 6 and 7). The ordination analyses show a distinct separation between the habitats included within the pothole areas and all other habitat types (Figure 14). The deep zone (D), pothole bench (PHB), and pothole mound (PHM) habitats appear to separate out from the other habitat zones based on both 2016 and 2017 NMDS ordinations (Figure 14).

Few habitat zones show a statistically significant increase (Appendices 4, 8, 10, and 13). This includes the treatment areas (TR) (Kruskall-Wallis $p<0.001$) (Appendix 8), the 2013 SEM and 2017 treatment (TR) zones (Kruskall-Wallis $p<0.001$) (Appendix 8), overall channel habitat which includes the channel and bench habitats (Kruskall-Wallis $p<0.001$) (Appendix 10), and the mound habitat (Kruskall-Wallis $p=0.003$) (Appendix 13). The overall sampling years also showed a significant increase in species richness (GLM $p=0.050$) (Appendix 4).

5.6.2 Cattail mat – Control vs Treatment

In the first year of sampling, the cattail mat control had a dominance of only *T. × glauca*, with the second year dominated by both *T. × glauca* and *I. capensis.* With the overbearing dominance of *Typha* in the control areas, no change was recorded.

5.6.2.1 Mean C, Weighted Mean C, and FQAI

Mean C was not significantly different within the cattail mat across the two years, which was expected as the control (Kruskall-Wallis $p=0.789$) (Appendix 5). I found that weighted C was not significantly different within the cattail mat across the two years (Kruskall-Wallis $p=0.572$) (Appendix 5). FQAI was not significantly different within the cattail mat across the two years, which was expected for the control (Kruskall-Wallis $p=0.500$) (Appendix 5).

<u>*5.6.3 Cattail Treatment Pre- and Post-Restoration*</u>

To determine the success of the treatment accurately, data from post-restoration 2017, were compared to data collected in 2013 within the pre-restoration shallow emergent marsh. The shallow emergent marsh area in 2013 was dominated heavily by *T. × glauca* and *H. morsus-ranae.* Post-restoration was very similar to the pre-restoration dominance, with dominance of *T. × glauca* in 2016 and *H. morsus-ranae* and *U. vulgaris* in the same areas in 2017. A large numeric decrease in *Typha* occurred within the treatment areas (TR) due to the restoration efforts. However, pre-restoration shallow emergent marsh data for 2013 FQAI, mean C, and weighted mean C were significantly different from the 2017 cattail treatment zone data, found in the same areas (Appendix 8).

5.6.3.1 Mean C, Weighted Mean C, and FQAI

Some of these 2013 and 2017 data do not follow a normal distribution even through transformation (Appendix 3). I found a significant decrease in mean C when comparing 2013 pre- to 2017 post-restoration (2013=3.12, 2017=2.01, Kruskall-Wallis $p<0.001$) (Table 7, Appendix 8). Comparing mean C within the treatment area (TR) from year 1 to year 2 of sampling resulted in no significant difference (Kruskall-Wallis $p=0.262$) (Appendix 9).

Weighted mean C significantly increased from 2013 pre- to 2017 post-restoration (2013=0.42, 2017=2.95, Kruskall-Wallis $p<0.001$) (Table 7, Appendix 8). Weighted mean C within the treatment area (TR) from year 1 to year 2 of sampling resulted in a significant difference (2013=1.61, 2017=2.95, Kruskall-Wallis $p<0.001$) (Table 7, Appendix 9).

FQAI scores significantly increased from 2013 pre- to 2017 post-restoration (2013=4.13, 2017=6.97, Kruskall-Wallis $p<0.001$) (Table 7, Appendix 8). Comparing FQAI within the treatment area (TR) from year 1 to year 2 of sampling showed no significant difference (Kruskall-Wallis $p=0.429$) (Appendix 9).

<u>*5.6.4 Channels*</u>

The three habitat types that make up the overall channel, the channel zone (C), intermediate bench (IB), and shallow bench (SB). In the first year of sampling, the C habitat was dominated by *Elodea canadensis, H. morsus-ranae,* and *U. vulgaris,* the IB habitat was dominated by *E. canadensis, H. morsus-ranae, L. minor,* and *U. vulgaris,* and the SB habitat was dominated by *T. x glauca* (Table 2). The second year C habitat was dominated by *C. demersum, S. pectinata,* and *U. vulgaris,* IB was dominated by *H. morsus-ranae, S. pectinate,* and *U. vulgaris,* and in the SB, H. morsus-ranae and T x glauca were dominant (Table 3). Overall channel habitats had low mean C and weighted mean C scores across the two years (Table 7) but had an FQAI score of 5.7 in 2016 with a significant increase to 7.3 in 2017 (Table 7, Appendix 10). This can be linked to an increase in species count from an average of 4.5 to 5.2, influencing the FQAI equation (Table 8).

5.6.4.1 Mean C, Weighted Mean C, and FQAI

Mean C within the channels was significantly different across the two years of sampling when combining the three habitat zones to create the full channel habitat (2016=2.56, 2017=2.90, Kruskall-Wallis $p=0.040$), (Table 7, Appendix 10). Weighted mean C within the channels also differed substantially and significantly across the two years of sampling (2016=2.15, 2017=2.89, Kruskall-Wallis $p=0.019$) (Table 7, Appendix 10). FQAI within the channels differed significantly across the two years of sampling (2016=5.65, 2017=7.30, Kruskall-Wallis $p<0.001$) (Table 7, Appendix 10).

5.6.5 Potholes

The two habitat types that make up the overall pothole, the deep zone (D) and the pothole bench (PHB). Within the overall pothole habitat, the PHB was dominated by *H. morsus-ranae* and *L. salicaria* and the D habitat was dominated only by U. vulgaris within the first year. In the second year of sampling, *L. salicaria, U. vulgaris, H. morsus-ranae,* and *T. × glauca* were dominant in the PHB habitat, with only *U. vulgaris* dominant in the D zone in the second year. Since dominance remained similar across the sampling years, no changes were seen across the three metrics.

5.6.5.1 Mean C, Weighted Mean C, and FQAI

In the potholes, I found no significant difference for mean C, weighted mean C, or FQAI (Kruskall-Wallis; mean C *p=0.114*, weighted mean C *p=0.578,* FQAI *p=0.962*) (Appendix 11) across the two sampling years.

5.6.6 Mounds

Two mound habitat types were used for these analyses, the pothole mounds (PHM) and the channel mounds (M). The pothole mounds were initially dominated by *L. salicaria* and *T.* × *glauca*, with an abundance of *G. trifidum* (9.6%) and *I. capensis*. In the second sampling year, the pothole mounds were dominated by *D. verticillatus*, *I. capensis*, *L. salicaria*, *P. hydropiper,* and *T.* × *glauca*. The channel mounds had *L. salicaria, P. hydropiper*, *P. lapathifolia*, and *T.* × *glauca* as dominant species in the first year of sampling. The second year showed a dominance of *H. morsus-ranae*, *P. hydropiper,* and *T.* × *glauca.* Composition significantly changed atop the pothole mounds (PHM) but not nativeness or floristic quality, with the opposite occurrence atop the channel mounds (M).

5.6.6.1 Mean C, Weighted Mean C, and FQAI

There were no significant differences across the two years for both mound habitats for mean C (Kruskall-Wallis; PHM p=*0.631*, M *p=0.087*) (Appendix 12, Appendix 13). There was a significant increase across the years for the pothole mounds (PHM) for weighted mean C (2016=2.00, 2017=2.59, Kruskall-Wallis p=*0.008*) (Table 7, Appendix 12) but no difference across years for FQAI (Kruskall-Wallis *p=0.778*) (Appendix 12). The channel mounds (M) had no significant differences across the two sampling years for weighted mean C (Kruskall-Wallis *p=0.560*) (Appendix 13) but were significantly different across the two sampling years for FQAI (2016=6.87, 2017=8.30, Kruskall-Wallis *p=0.002*) (Table 7, Appendix 13).

6. DISCUSSION

I evaluated each different created habitat zone and hypothesize that these created habitat zones will be floristically different. I also evaluated the restored areas individually and expect to see an increase in richness and floristic quality along with an increase in native emergent plant cover and decrease in *Typha* cover. Lastly, post-restoration, all habitat zones will continue to floristically improve via species richness as the wetland rebounds back to its natural state.

6.1 Ability of Restoration Methods to Meet Objectives

Every aspect of the restoration and monitoring efforts must be reviewed in terms of effectiveness for use in future restoration projects. Some issues also occur during the sampling season that call for continued monitoring and adaptive management at any restoration site.

6.1.1 Walking Surveys for Target Invasives

All three target invasive species were found within Braddock Bay prior to the restoration. Purple loosestrife (*L. salicaria*) abundance increased from 2016 to 2017. Several factors might provide insight into why this increase occurred. The initial disturbance of the restoration site allowed for colonization of disturbed areas by different plant species, specifically invasive species that spread by seed (Hovik et al. 2011). The year following the initial disturbance was a drought year, allowing for the potential establishment of invasive species, such as *L. salicaria* and *T. x glauca*, which can tolerate drier conditions that would be unfavorable for native wetland species. The second year of sampling showed establishment success of *L. salicaria* as a 25% increase in data points where the maximum purple loosestrife abundance was observed. Given that these are the first two years of monitoring after the initial disturbance occurred, the area is susceptible to a short-term invasibility of the community (Hobbs and Huenneke 1992).

Within the restoration, *L. salicaria* percent cover decreased from year 1 to year 2 in both pothole mound (PHM) and channel mound (M) habitats. *Galerucella* beetles were released by NYS DEC after the *Lythrum* increase was reported in 2017. I expect the ensuing third year of data under a DEC grant to show a further decrease in abundance of *L. salicaria* due to a combination of the release of *Galerucella* beetles and the dramatic water-level difference between the two years sampled (GLAM 2018).

6.1.2 Floristic Improvement

Based on the analyses of species richness, measured by SPPcount (species count), a limited number of the sampled habitat zones throughout the restoration site are still accumulating more species, adding to the overall biodiversity of the site. This is evidence that the created habitat zones are not floristically diversifying, opposing my hypothesis that the habitats will be floristically distinct. Many habitat zones show a numeric increase in species count (Table 8).Some areas of the restoration seem to be floristically improving through an increase in species richness, but many other habitat zones show no increase in richness, which is a concern when this means these areas are not accumulating more species.

6.1.3 Cattail Treatment Areas

I hypothesized that these restored areas will have an increased richness and floristic quality along with an increase in native emergent plant cover in these areas.Weather also impacted these treatment areas, resulting in an average water depth of 0 cm in 2016 versus about 60 cm in 2017 (Figures 11 and 13). Changes from pre-restoration 2013 and post-restoration 2017 data that occurred within the treatment areas may have been aided by the cattail-cutting treatment performed, which opened up the overhead cover for other plants. This seems to be the case in the invasion of these newly flooded treatment areas by *H. morsus-ranae*, forming their own dense mats, combined with interspersed *U. vulgaris* (Tables 4 and 5). *H. morsus-ranae* is an unwanted invasive species on the New York State DEC's 2014 Prohibited and Regulated Invasive Plants list (NYSDEC 2014) and, at Braddock Bay, has spread rapidly within the disturbed restoration areas (Tables 2, 3, 4, and 5). Best management practices going forward would include continued targeting of high impact invaders (e.g., *P. australis*, *L. salicaria*, *T. natans*) and further additions to the list of species that threaten the bay further (e.g., *H. morsus-ranae*, *Myriophyllum spicatum).*

The increase in the non-native species observed within the SGM and TR areas was accompanied by a decrease of the most abundant native plant species observed in the same areas. This shows that there is no clear increase in native emergent plant cover in these restored areas. Many dominant plant species observed in 2016 decreased within the SGM treatment areas in 2017 while *H. morsus-ranae* abundance increased (Ex. *C. lacustris, U. vulgaris, I. capensis*, Tables 2 and 3). The overall cover of H. morsus-ranae, potentially combined with the accompanying water levels, seems to have also impacted other non-native species, showing decreases in abundance for *L. salicaria* and *T. x glauca* (Tables 2, 3, 4, and 5).

Further monitoring in this shallow emergent marsh treatment area is needed to assess the long-term results of the cattail treatment plan, which at this point show the shift to inundation and potential to block out further cattail growth through observed flooding combined with the cover of the new *H. morsus-ranae* mat present. The resulting invasion, if continued, has the potential to further block out and decrease native emergent plant species in the SGM treatment areas due to the inability to gain sufficient light underneath the *H. morsus-ranae* mat combined with stable, high water levels, not allowing germination of the native seed bank (Keddy and Reznicek, 1986). It can be nearly impossible to predict water-level changes that dramatically change conditions at a restoration site as what is shown here at Braddock Bay. The removal of cattail, combined with treatment and flooding, is a combination that seems to have led to an overall decrease in average percent cover of *Typha* in the treatment areas from 65.9% in 2013 to 22.3% in 2016 to 3.3% in 2017 (Tables 2, 3, and 6) This is a major statistic as it shows that the treatment is working and with an increased affect by the record high water levels. When the cut and treated *Typha* stems are flooded, they no longer are able to transport oxygen to the rhizomes, ultimately killing the plant (Wilcox et al. 2017).

6.1.4 Mounds

No significant change in plant community composition or statistical metrics occurred atop the pothole mound (PHM) habitats from 2016 to 2017, whereas the mound (M) habitat showed a similar trend to the adjacent overall channel habitat. These data from the mound (M) habitat show the FQAI statistic once again being influenced by the species count across the sampling year. In the mound (M) habitat, a low mean C and weighted mean C score across the sampling years was accompanied by a significant increase in FQAI from year 1 to year 2, showing another instance of the FQAI statistic being influenced directly by the species count. This is clearly visible in these data, where a 2.5-point increase to FQAI (Table 7) can be linked to about a 2-point increase in average species count within the overall channel (Table 8). If FQAI continues to show influence by outside factors, specifically species richness, then using one of the other metrics would be more beneficial.

Much variation was shown in spoil pile height of the mounds, which needs to meet a height of 75.35 to 75.60 m (IGLD 85) to discourage *Typha* growth (Wilcox and Xie 2007, Wilcox et al. 2017). During the second year of sampling, with the occurrence of record rainfall and water levels (NOAA 2017, GLAM 2018), both pothole and channel mounds were affected. There were examples of mounds that were built too low that resulted in inundation, many at the prescribed height, or some built above the maximum height with quite a lot of dry spoil above the already high lake-levels. The record high water level reached around 75.80 m (IGLD 85; Figure 1b) during the 2017 sampling season. Although these mound areas were created to discourage the growth of invasive *Typha*, this weather anomaly led to flooding of the mounds and could result in cattail invasion in these areas. This pattern can be seen through the two years of sampling, where, on average, *Typha* was found to have decreased on the PHM habitat by 2.5% average cover but increased on the M habitat by about 2% average cover. These small percent changes did not influence any statistics or calculations. Overall, very little change was observed with *Typha* stands atop of spoil pile mounds.

When building future spoil mounds, another consideration would be to look further into the soils. If the substrate being excavated to create these spoil piles consists of more organic matter, the mounds are expected to settle to a lower elevation, with less settling if the substrate contains a better aggregate, such as clay. In further monitoring and continued cattail treatment work, I expect a decrease in *Typha* cover atop the spoil mounds, especially those built to the prescribed elevation specifications.

6.1.5 Channels

Monitoring showed channels where width was less than needed to prevent filling with sediment or floating pieces of *Typha* mat (Figure 15). In some locations within the restoration site, construction couldn't get every channel, pothole, and mound to the exact planned depth or height, leading to some constructed features not functioning as needed. This is a concern when one of the objectives was creating access to the sedge-grass meadow to allow for fish spawning, specifically the Northern Pike. This does not mean that all channels were blocked or completely impassable by fish, but the overall connectivity of the wetland and the purpose behind the dug channels must be maintained for prolonged success. Channel depths, widths, and mound heights all have specific design standards, and the variability shown at the site can be seen through the variability also found within these data. Some locations within the restoration had channels blocked off by and potholes filled in by *Typha* regrowth.

In the channel habitat, a low mean C and weighted mean C score across the sampling years was accompanied by a significant increase in FQAI, showing another instance of the FQAI statistic being influenced directly by the species count. The FQAI score increased based on the number of species present, despite average mean C and weighted mean C scores in the range of 2.0 to 3.0 (Tables 7 and 8). This is clearly visible in these data, where a two-point increase in FQAI (Table 7) can be linked to a 1.5-point increase in average species count within the overall channel (Table 8). An increase in species count increased the overall FQAI for these channel habitats, but the mean C score was still low, so despite an apparent increase of the floristic quality index, there was no increase in nativeness. Despite the increase in species count, not many of the changes were significant, therefore, the habitats are not still floristically improving by means of species accumulation. There is a numerical increase in *Typha* cover within both the pothole bench (2016=7.4, 2017=26.4) (Tables 2 and 3) and intermediate channel bench (2016=1.2, 2017=4.2) (Tables 2 and 3), but due to the randomness of transect placement, there is no evidence within my data of this occurring within the actual channel or the deep zone of the potholes.

6.1.6 Potholes

Between 2016 and 2017, no significant changes in dominant vegetation, mean C, weighted mean C, or FQAI occurred within the potholes other than access issues caused by the filling of the pothole and the closing of the channels (Figure 15). This is one example of a habitat zone that showed no improvement from 2016 to 2017 through means of species accumulation or floristic quality. Using aerial photography accompanied by ground-truthing at the site from a different entrance point, I was able to determine that the largest excavated pothole has mostly filled in with sediment and regrowth of *Typha* mat toward the center of the pothole (Figure 15). This pothole-filling can be attributed to the timing of the excavation, as this was the last part of the restoration to occur, nearing the end of the winter season and making the sediment softer during excavation. This allowed sediment rebound within the pothole, caused by soil consolidation during construction intensified by construction timing, leaving it shallower than called for in the original plan. This can be of concern with regard to connectivity throughout the wetland, much like in that of the channels, which is important for the different wildlife using these pothole habitats. Post-restoration management should work to clear these blocked channels and filled potholes to restore the intended connectivity.

6.2 Planted and Seeded Species

More information on percentage of planted plugs surviving would be needed to perform meaningful statistics comparing survivorship across the two years of sampling. This would require very intensive surveying and would require a larger crew, as about 65,000 individual plugs were planted. Issues observed with plugs range from wildlife pulling plugs to the planting crew leaving various trays full of plugs unplanted. *In situ* herbivory was observed in both the 2016 and 2017 seasons but is not expected to have lasting effects. Many of the plants observed from the list of seeded and planted species were also present in the preliminary surveying of Braddock Bay in 2013 and may not be a product of restoration efforts. Only half of the planted/seeded species were observed within the second-year post-construction and many of these species were also observed in 2013 before the planting and seeding. There is little evidence to suggest that these species observed within 2016 and 2017 established themselves through the process of planting and seeding, and may point to recruitment from the pre-existing plant community in 2013. There is a visible improvement in floristic quality, but it appears that having a remnant species or seed pool is just as important as planting and seeding efforts.

6.3 Influence of Lake-level on Results

An unpredictable issue while sampling Braddock Bay was the variation of the water levels from year to year. A drought year in 2016 followed by record rainfall and inputs from Lake Erie in 2017 created very different hydrologic conditions (GLAM 2018). The abnormality shows in the data when comparing across years, where the range of water depths measured in the emergent marsh in 2016 was 0 to 120 cm and 0 to 230 cm in 2017 (Figures 11 and 13). This dramatic change in water levels left areas of sedge/grass meadow and treatment areas dry in 2016 versus in greater than 100 cm of standing water in those same areas in 2017, with many spoil piles being submerged, potentially affecting community development. In the second year, with record water levels, a shift to submerged aquatic vegetation showed dominance within treatment areas and even an increase atop the submerged mounds (Table 4 and 5).

Under typical lake-level conditions, the mound habitats should remain dry, with a reduction in *Typha* and establishment of more native wetland vegetation that can withstand dry spells that come with lake-level fluctuation. More native vegetation should establish within the treatment areas due to the reduction in overhead *Typha* cover in these areas (Tables 4 and 5).

The difference of water-levels within the sedge/grass meadow areas between 2013 and 2017 has an impact on the results as well. This comparison takes the shallow emergent marsh area in 2013 and compares it to the high water-levels within the same area in 2017, shifting the community from a *Typha*-dominated monoculture to a submerged aquatic vegetative (SAV) community. The restoration may have attributed to the community change by opening the over story, but there is no indication that the restoration caused the shift to SAVs, which is more likely linked to the dramatic change in water-level, allowing the establishment of SAVs in the area.

6.4 Recommendations

Overall mean C (2016=2.75, 2017=2.77), weighted mean C (2016=2.25±1.35, 2017=2.68), and FQAI (2016=6.31, 2017=6.89) scores for Braddock Bay are still low (Table 7); therefore, more long-term data are needed to determine the long-term success of the restoration. These floristic quality indices can vary widely, with examples along the Northern shore of Lake Ontario ranging from 12.5 in a highly disturbed wetland to 31.8 at a low-disturbance, natural wetland area (n=12, Grabas et al. 2003). In Lake Michigan drowned rivermouth wetlands showed a range of 25.5 to 31.0 (n=6, Wilcox et al. 2002). Lake Superior barrier beach wetlands showed a range of 18.5 to 61.4 (n=6, Wilcox et al. 2002). Higher C-scoring plant species (7 to 10) will generally correspond with higher quality habitat, denoted by high FQAI values (Taft et al. 1997, Andreas et al. 2004). FQAI indicates vegetative quality of the site, with a score from 1-19 considered low quality, 20-35 high quality, and anything above 35 exceptional quality habitat (USFWS 2019). With further monitoring and further cattail treatment, I would expect to see an increase in overall diversity coinciding with the decrease in *Typha* cover atop the spoil piles as well as where the treatment is taking place in the shallow emergent marsh (Wilcox et al. 2017). Adaptive management within the subsequent years would determine if further treatment actions are needed in the shallow emergent marsh. Early research supports management actions involving water-level manipulation and herbicide treatments (Steenis et al. 1959), with further in-depth studies of cattail-control methods that narrowed the potential for successful management (Beule 1979, Wilcox and Ray 1989, Ball 1990, Lawrence et al. 2016).

6.4.1 Mean C, Weighted Mean C, and FQAI

Various combinations of significance in metric results occurred across the three metrics. Some tests showed one of the three metrics proving to be significant, with the other two showing the opposite result, but no discernable pattern is present. Using just the mean C metric is not adequate. The fidelity of the species to the environment in question, that being its role within the environment or necessity for niche habitat, must be considered. Both the weighted mean C and FQAI statistics showed a significant difference between the two sampling years, whereas the mean C statistic used within these calculations showed no significance. The ability of these metrics to use the gradient of nativeness proves beneficial in ranking of each species at each location. The FQAI statistic has proven to be influenced heavily by the species richness of the area, which may be its biggest flaw (Matthews 2003, Bourdaghs et al. 2006). Areas with great diversity may fall victim to a skewed FQAI score if most of the plant species that create that diversity have low C-scores.

My data show that Braddock Bay is low on the overall nativeness C-score scale, but some areas, such as the overall channels and channel mounds, are showing greater floristic quality due to an increase in richness, not an increase in native plants. Weighted mean C may be a better metric to use because it changes a predetermined state-wide or ecoregion-wide conservancy score into a local, site-level C-score. Consideration of the preliminary site vegetative composition may be appropriate when choosing a proper metric to use for future studies. If the location is a monoculture of an invasive species, the site may have a low overall mean C or weighted C score, influenced by the dominant cover of the non-native species (non-native species C-score rating = 0) (Andreas et al. 2004, NEIWPCC 2011, Reznicek et al. 2014, Faber-Langendoen 2018). But, depending on species richness rather than dominance, results using the Floristic Quality Assessment Index may be skewed towards a higher quality rating (Matthews 2003, Bourdaghs et al. 2006). Weighted mean C gave the most reliable result, as it considers the actual abundance of the species at the location surveyed, with hardly any observed influence from species richness like what was seen with the FQAI metric. The weighted mean C metric gives an abundance-based score that is unaffected by species richness, which was observed throughout these results. Short-term studies do not provide as robust comparisons as longer-term, five- or ten-year studies can, since vegetative community composition takes about five to ten years to recover fully after restoration (Haapalehto et al., 2017). Further monitoring is funded and scheduled for the upcoming years.

6.4.2 Project Recommendations

When performing restoration activities, the key factor is following suggested measurements and restoration standards. If weather or other variables impact the ability of the restoration to be performed according to design, then adaptive management to change the plan must occur. Changes to the plan may need to occur throughout the restoration activities being performed, as long as the proper information or guidance to make such changes is available. For restoration work similar to what was done at Braddock Bay, the concern is the ability of the construction crew to maintain an accurate channel and pothole depth/width and spoil mound height, not straying far from the target depth, width, or height. Possibly adding a margin of error or safety to the construction specifications so that the habitats meet a minimum requirement with the uncertainty associated with construction planning and timing. If all of the excavation had been done in a timely manner, some pothole and channel filling may not have occurred. Furthermore, future spoil-mound construction must require a proper soil survey to determine exactly where adjustments need to be made based on substrate composition and settling. A true survivorship study is suggested in future restorations to determine the effect of planted and seeded species in the different created wetland habitats. This study shows that determining the effect can be difficult when the site is so large and not many planted or seeded taxa are observed in the random sampling of the site. Many invasive species can be controlled at restoration sites through mechanical and biological treatment, but the restoration practitioner should be aware of the ability of invasive species to overtake newly disturbed areas. Thankfully, through adaptive management at Braddock Bay, where the influx of invasive *L. salicaria* within the restoration became an immediate concern, *Galerucella* beetles were used as a biological control. It will be necessary to ensure the connectivity of the restoration work to allow the wetland to serve its restored purpose fully, which might include continuously clearing dug channels or continued cattail control. Cattail control efforts reduced the average percent cover of *Typha* observed within the treatment areas, giving reason to continue control efforts on-site and to recommend the methodology used for future restoration activities (Wilcox et al. 2018).

Overall, Braddock Bay currently sits very low on the mean C, weighted mean C, and FQAI scales (Taft et al. 1997, Andreas et al. 2004, USFWS 2019). Further monitoring is needed to give a better representation of how successful the restoration truly will be in the long term. Continued monitoring of Braddock Bay has been funded and scheduled through 2020 thanks to Region 8 of NYS DEC.

Table 1. Listed species seeded and planted at the Braddock Bay Restoration project. If species were observed during the sampling year or during the preliminary 2013 surveys, it is demarcated by a Y for Yes along with blank spaces meaning the species was not observed.

	Observed in 2013	Seed (S) or Plug (P) pre-2016 sampling	Observed in 2016	Observed in 2017
Acorus americana		S		
Alisma subcordatum		S + P		
Asclepias incarnata	Y	S	Y	Y
Bidens cernua	Y	S	Y	Y
Calamagrostis canadensis	Y	S	Y	Y
Carex lacustris	Y	P	Y	Y
Carex lurida		S + P		Y
Carex scoparia		S + P		
Carex stipata		P		
Carex stricta	Y	S + P	Y	
Carex vulpinoidea		S		
Elymus virginicus		S	Y	Y
Eutrochium maculatum		S		
Glyceria canadensis		S		
Iris versicolor	Y	S		
Juncus effusus	Y	S	Y	Y
Leersia oryzoides		S		Y
Mimulus ringens		S		
Persicaria amphibia	Y	P	Y	Y
Poa palustris		S		
Pontederia cordata		P	Y	Y
Sagittaria latifolia	Y	S + P	Y	Y
Schoenoplectus tabernaemontani	Y	S + P	Y	Y
Scirpus atrovirens		S		
Scirpus cyperinus	Y	S		
Scirpus polyphyllus		S		
Sparganium americanum		S		
Sparganium eurycarpum	Y	S	Y	Y

Verbena hastata	Y	S + P	Y	Y
Vernonia noveboracensis		P		Y
Species present =	13	-	13	15

Table 2. Habitat-level dominant vegetation with corresponding mean cover percentages for 2016. Bolded values are over the dominance value of >10 percent cover (non-native species marked with an asterisk*; All values include ± the standard deviation; C=channel, CT=cattail-control, D=deep zone, IB=intermediate bench, M=mound, PHB=pothole bench, PHM=pothole mounds, SB=shallow bench, SGM=sedge/grass meadow, TR =treatment area).

	Pothole Transects			**Channel Transect**						**Control**
	D	**PHB**	**PHM**	**SGM**	**TR**	**M**	**SB**	**IB**	**C**	**CT**
Acer saccharinum	-	-	-	**10.0** ±0.0	1.8 ±0.5	-	-	-	-	-
Calamagrostis canadensis	-	-	-	8.9 ±9.6	0.7 ±8.4	1.5 ±8.6	-	-	-	-
Carex lacustris	-	0.2 ±0.0	0.1 ±0.0	**22.1** ±19.4	1.9 ±1.7	3.3 ±6.1	1.8 ±0.0	0.2 ±2.5	0.3 ±1.3	-
Ceratophyllum demersum	0.3 ±2.0	0.2 ±1.3	-	-	-	-	-	1.4 ±2.6	8.0 ±8.5	-
Elodea canadensis	1.0 ±0.6	-	-	-	-	-	-	**10.5** ±5.5	9.4 ±6.6	-
Gallium trifidum	-	3.4 ±1.4	9.6 ±7.7	1.3 ±9.1	4.2 ±5.1	9.3 ±9.6	8.7 ±9.4	0.3 ±3.8	0.1 ±0.0	1.1 ±1.3
*Hydrocharis morsus-ranae**	1.8 ±1.3	**25.1** ±11.1	0.1 ±0.0	-	0.9 ±9.3	0.1 ±0.00	6.6 ±6.7	**37.1** ±17.8	**11.0** ±1.8	0.2 ±1.3
Impatiens capensis	-	0.2 ±1.3	9.4 ±8.9	1.3 ±4.0	1.3 ±6.2	0.2 ±1.3	1.9 ±9.2	-	-	9.7 ±6.8

Lemna minor	1.2 ±0.7	4.3 ±1.0	0.1 ±0.0	-	0.1 ±0.0	-	0.2 ±0.0	**13.8** ±5.8	5.6 ±7.0	1.3 ±0.8
*Lythrum salicaria**	0.1 ±0.0	9.0 ±5.4	**38.8** ±23.1	1.6 ±1.6	4.1 ±6.1	**12.5** ±8.4	4.1 ±8.1	0.2 ±2.5	-	2.3 ±11.5
Persicaria hydropiper	-	0.8 ±1.9	8.1 ±7.4	1.7 ±0.0	1.7 ±6.2	**19.8** ±15.1	1.6 ±11.5	-	-	0.8 ±1.5
Persicaria lapathifolia	-	-	1.8 ±10.6	0.9 ±1.6	0.5 ±4.8	**11.4** ±9.6	-	-	-	-
*Salix fragilis**	-	-	-	**10.0** ±0.0	-	-	-	-	-	-
Stuckenia pectinata	2.5 ±1.7	-	-	-	-	-	-	8.7 ±8.1	2.8 ±7.3	-
*Typha × glauca**	0.5 ±1.1	7.4 ±7.8	**16.7** ±10.3	**10.0** ±10.5	**22.3** ±7.0	**11.6** ±8.0	**48.9** ±17.3	1.2 ±3.3	0.3 ±1.3	**51.7** ±13.7
Utricularia vulgaris	**17.7** ±9.9	4.2 ±4.9	-	-	-	-	-	**10.2** ±8.8	**22.0** ±16.3	-

Table 3. Habitat-level dominant vegetation with corresponding mean cover percentages for 2017. Bolded values are over the dominance value of >10 percent cover (non-native species marked with an asterisk*; All values include ± the standard deviation; C=channel, CT=cattail-control, D=deep zone, IB=intermediate bench, M=mound, PHB=pothole bench, PHM=pothole mounds, SB=shallow bench, SGM=sedge/grass meadow, TR =treatment area)

	Pothole Transects			**Channel Transect**						**Control**
	D	**PHB**	**PHM**	**SGM**	**TR**	**M**	**SB**	**IB**	**C**	**CT**
Boehmeria cylindrica	-	1.3 ±0.2	8.8 ±10.5	-	-	0.4 ±1.2	-	-	-	2.9 ±0.9
Calamagrostis canadensis	-	-	-	**10.7** ±7.3	0.6 ±5.2	0.2 ±1.3	-	-	-	-
Carex lacustris	-	2.5 ±1.7	2.8 ±8.1	**13**.4 ±10.3	0.2 ±0.0	1.8 ±7.7	1.0 ±6.3	-	-	-
Ceratophyllum demersum	3.0 ±9.1	0.2 ±0.0	-	0.3 ±0.0	0.5 ±1.4	2.1 ±1.5	1.4 ±10.5	7.9 ±4.1	**11.7** ±3.6	-
Decodon verticillatus	-	3.5 ±9.2	**12.2** ±12.9	-	0.2 ±0.0	0.6 ±1.5	0.9 ±0.0	0.2 ±2.5	-	0.5 ±0.0
*Hydrocharis morsus-ranae**	1.3 ±1.3	**28.4** ±19.6	1.3 ±8.7	**15.8** ±19.4	**18.3** ±23.8	**14.1** ±15.4	**23.3** ±8.9	**20.5** ±7.0	2.5 ±3.8	2.8 ±7.4
Impatiens capensis	-	0.5 ±0.7	**11.9** ±25.4	-	-	0.4 ±0.7	0.2 ±0.0	-	-	1.1 ±1.3
*Lythrum salicaria**	4.1 ±1.4	**10.2** ±9.5	**24.8** ±13.2	0.8 ±1.3	2.0 ±1.3	3.8 ±6.4	1.7 ±1.2	0.3 ±3.8	0.4 ±0.0	2.8 ±8.2
Persicaria hydropiper	-	0.2 ±0.0	**19.0** ±10.5	-	-	**17.9** ±23.5	0.2 ±0.0	-	-	-

Stuckenia pectinata	-	-	-	-	0.2 ±0.0	-	1.3 ±11.0	**11.3** ±8.4	8.2 ±9.6	-
*Salix fragilis**	-	-	-	**14.2** ±9.6	-	-	-	-	-	-
Thelyptris palustris	-	4.7 ±8.9	9.4 ±5.9	**-**	-	-	-	-	-	7.1 ±7.5
*Typha × glauca**	1.5 ±1.1	**26.4** ±7.8	**14.2** ±10.3	8.8 ±6.6	3.3 ±3.2	**13.4** ±8.3	**37.7** ±19.6	4.2 ±3.0	-	**48.0** ±27.2
Utricularia vulgaris	**22.8** ±9.9	**25.5** ±4.9	-	5.5 ±6.2	**12.1** ±14.1	0.3 ±0.0	4.6 ±5.7	**25.9** ±9.5	**19.5** ±16.9	-
Verbena hastata	-	0.2 ±0.6	8.6 ±7.1	-	**-**	4.1 ±8.9	-	**-**	**-**	-

Table 4. Mean Cover percentages for each non-native species found in each transect type in 2016, sorted by habitat type along the transect (total non-native species in 2016 = 11; All values include ± the standard deviation; C=channel, CT=cattail-control, D=deep zone, IB=intermediate bench, M=mound, PHB=pothole bench, PHM=pothole mounds, SB=shallow bench, SGM=sedge/grass meadow, TR=treatment area).

	Pothole Transect			**Channel Transect**						**Control**
	D	**PHB**	**PHM**	**SGM**	**TR**	**M**	**SB**	**IB**	**C**	**CT**
Cirsium arvense	-	-	0.8 ±1.7	0.2 ±0.0	0.2 ±1.2	0.9 ±0.8	0.9 ±7.0	-	-	-
Hydrocharis morsus-ranae	1.8 ±1.3	25.1 ±11.1	0.1 ±0.0	-	0.9 ±9.3	0.1 ±0.00	6.6 ±6.7	37.1 ±17.8	11.0 ±1.8	0.2 ±1.3
Lythrum salicaria	0.1 ±0.0	9.0 ±5.4	38.8 ±23.1	1.6 ±1.6	4.1 ±6.1	12.5 ±8.4	4.1 ±8.1	0.2 ±2.5	-	2.3 ±11.5
Myosotis scorpioides	-	-	-	-	0.1 ±0.0	0.5 ±1.2	0.2 ±0.0	-	-	-
Myriophyllum spicatum	1.8 ±5.2	0.5 ±0.0	-	-	-	-	-	5.2 ±4.2	5.6 ±8.7	-
Najas minor	0.2 ±1.25	0.1 ±0.0	-	-	-	-	-	2.1 ±5.1	4.2 ±1.7	-
Persicaria lapathifolia	-	-	1.8 ±10.6	0.9 ±1.6	0.5 ±4.8	11.4 ±9.6	-	-	-	-
Persicaria maculosa	-	-	0.5 ±0.2	0.3 ±0.0	-	0.2 ±0.0	-	-	-	-

Phragmites australis	-	-	-	-	-	-	-	-	-	2.5 ±0.0
Salix fragilis	-	-	-	10.0 ±0.0	-	-	-	-	-	-
Typha x glauca	0.5 ±1.1	7.4 ±7.8	16.7 ±10.3	10.0 ±10.5	22.3 ±7.0	11.6 ±8.0	48.9 ±17.3	1.2 ±3.3	0.3 ±1.3	51.7 ±13.7

Table 5. Mean Cover percentages for each non-native species found in each transect type in 2017, sorted by habitat type along the transect (total non-native species in 2017 = 11; All values include ± the standard deviation; C=channel, CT=cattail-control, D=deep zone, IB=intermediate bench, M=mound, PHB=pothole bench, PHM=pothole mounds, SB=shallow bench, SGM=sedge/grass meadow, TR=treatment area).

	Pothole Transects			**Channel Transect**						**Control**
	D	**PHB**	**PHM**	**SGM**	**TR**	**M**	**SB**	**IB**	**C**	**CT**
Butomus umbellatus	-	-	-	-	-	-	0.9 ±0.0	-	-	0.3
Cirsium arvense	-	0.8 ±0.0	2.5 ±1.7	-	-	3.25 ±0.4	-	-	-	-
Hydrocharis morsus-ranae	1.3 ±1.3	28.4 ±19.6	1.3 ±8.7	15.8 ±19.4	18.3 ±23.8	14.1 ±15.4	23.3 ±8.9	20.5 ±7.0	2.5 ±3.8	2.8 ±7.4
Lythrum salicaria	4.1 ±1.4	10.2 ±9.5	24.8 ±13.2	0.8 ±1.3	2.0 ±1.3	3.8 ±6.4	1.7 ±1.2	0.3 ±3.8	0.4 ±0.0	2.8 ±8.2
Myosotis scorpiodes	-	-	-	-	-	0.4 ±0.7	-	-	-	-
Myriophyllum spicatum	2.4 ±1.3	0.5 ±2.0	-	-	-	-	0.2 ±0.0	1.2 ±2.4	2.4 ±1.5	-
Persicaria lapathifolium	-	-	-	-	-	0.1 ±0.0	-	-	-	-
Persicaria maculosa	-	-	0.9 ±6.4	-	-	0.3 ±1.2	0.2 ±0.0	-	-	-
Salix fragilis	-	-	-	14.2 ±9.6	-	-	-	-	-	-

Taraxacum officinale	-	-	-	-	-	0.1 ±0.0	-	-	-	-
Typha x glauca	1.5 ±1.1	26.4 ±7.8	14.2 ±10.3	8.8 ±6.6	3.3 ±3.2	13.4 ±8.3	37.7 ±19.6	4.2 ±3.0	-	48.0 ±27.2

Table 6. Pre-restoration 2013 Shallow Emergent Marsh (SEM) dominant species.

	SEM
Typha x glauca	65.9
Hydrocharis morsus-ranae	10.6
Impatiens capensis	2.5
Lemna minor	2.2
Salix fragilis	2.0
Typha angustifolia	1.3
Solanum dulcamara	1.2

Table 7. Calculated average FQAI, Mean C, and Weighted Mean C statistics sorted by habitat type and year along with overall year averages (D=deep zone, PHB=pothole bench, PH=OVERALL pothole habitat, PHM=pothole mounds, SGM=sedge/grass meadow, TR/SEM=treatment area in the shallow emergent marsh, M=mound, SB=shallow bench, IB=intermediate bench, C=channel, CH=OVERALL channel habitat, CT=cattail-control; All values include ± the standard deviation).

	2013			**2016**			**2017**		
	FQAI	**Mean C**	**Weighted Mean C**	**FQAI**	**Mean C**	**Weighted Mean C**	**FQAI**	**Mean C**	**Weighted Mean C**
D				4.94 ±1.62	3.40 ±1.27	3.79 ±1.66	4.63 ±1.55	2.51 ±0.88	3.56 ±1.51
PHB				6.97 ±2.32	2.68 ±0.76	1.93 ±0.73	7.26 ±2.21	2.71 ±0.62	2.40 ±0.88
PH (D/PHB)				5.96 ±2.25	3.04 ±1.10	2.86 ±1.58	5.99 ±2.33	2.61 ±0.76	2.96 ±1.36
PHM				7.89 ±2.77	2.80 ±0.76	2.00 ±0.86	7.74 ±2.94	2.84 ±0.87	2.59 ±1.07
SGM				8.33 ±2.04	3.60 ±0.65	3.80 ±0.98	8.09 ±1.87	3.21 ±0.80	3.54 ±1.23
TR/SEM	4.31 ±2.37	2.01 ±1.01	0.42 ±0.55	6.73 ±2.50	2.76 ±0.65	1.61 ±1.09	6.97 ±2.39	2.91 ±0.84	2.95 ±1.56
M				6.87 ±2.04	2.51 ±0.52	2.05 ±0.66	8.30 ±2.41	2.68 ±0.64	2.28 ±1.18
SB				4.99 ±2.16	2.08 ±0.70	1.13 ±0.78	6.51 ±1.73	2.41 ±0.42	1.35 ±0.74
IB				5.99 ±1.29	2.52 ±0.54	1.90 ±1.17	7.72 ±1.49	2.81 ±0.36	3.35 ±0.84

C		5.97 ±1.60	3.08 ±0.83	3.40 ±1.10	7.69 ±1.55	3.54 ±0.59	4.06 ±1.18
CH (SB/IB/C)		5.65 ±1.78	2.56 ±0.81	2.15 ±1.39	7.30 ±1.70	2.90 ±0.63	2.89 ±1.41
CT		5.37 ±3.21	2.40 ±1.30	1.55 ±1.09	4.85 ±2.46	2.34 ±1.05	1.36 ±1.36
OVERALL		6.31 ±2.45	2.75 ±0.96	2.25 ±1.35	6.89 ±2.25	2.77 ±0.75	2.68 ±1.38

Table 8. Calculated average SPPcount (species richness) statistics sorted by habitat type and year along with overall year averages (D=deep zone, PHB=pothole bench, PH=OVERALL pothole habitat, PHM=pothole mounds, SGM=sedge/grass meadow, TR/SEM=treatment area in the shallow emergent marsh, M=mound, SB=shallow bench, IB=intermediate bench, C=channel, CH=OVERALL channel habitat, CT=cattail-control; All values include ± the standard deviation).

	2013	**2016**	**2017**
D		2.73 ±1.65	3.31 ±0.92
PHB		7.16 ±3.02	7.06 ±1.82
PH (D/PHB)		4.94 ±3.29	5.25 ±2.37
PHM		7.81 ±2.21	7.29 ±2.44
SGM		5.76 ±2.47	6.73 ±2.01
TR/SEM	4.27 ±1.87	6.05 ±2.58	6.04 ±2.45
M		7.55 ±2.43	9.38 ±2.55
SB		5.46 ±1.63	7.18 ±1.79
IB		5.75 ±1.05	7.61 ±1.86
C		4.50 ±1.63	5.19 ±2.47
CH (SB/IB/C)		5.24 ±1.55	6.70 ±2.26
CT		4.37 ±2.12	4.25 ±1.51
OVERALL YEAR		5.59 ±2.45	6.33 ±2.42

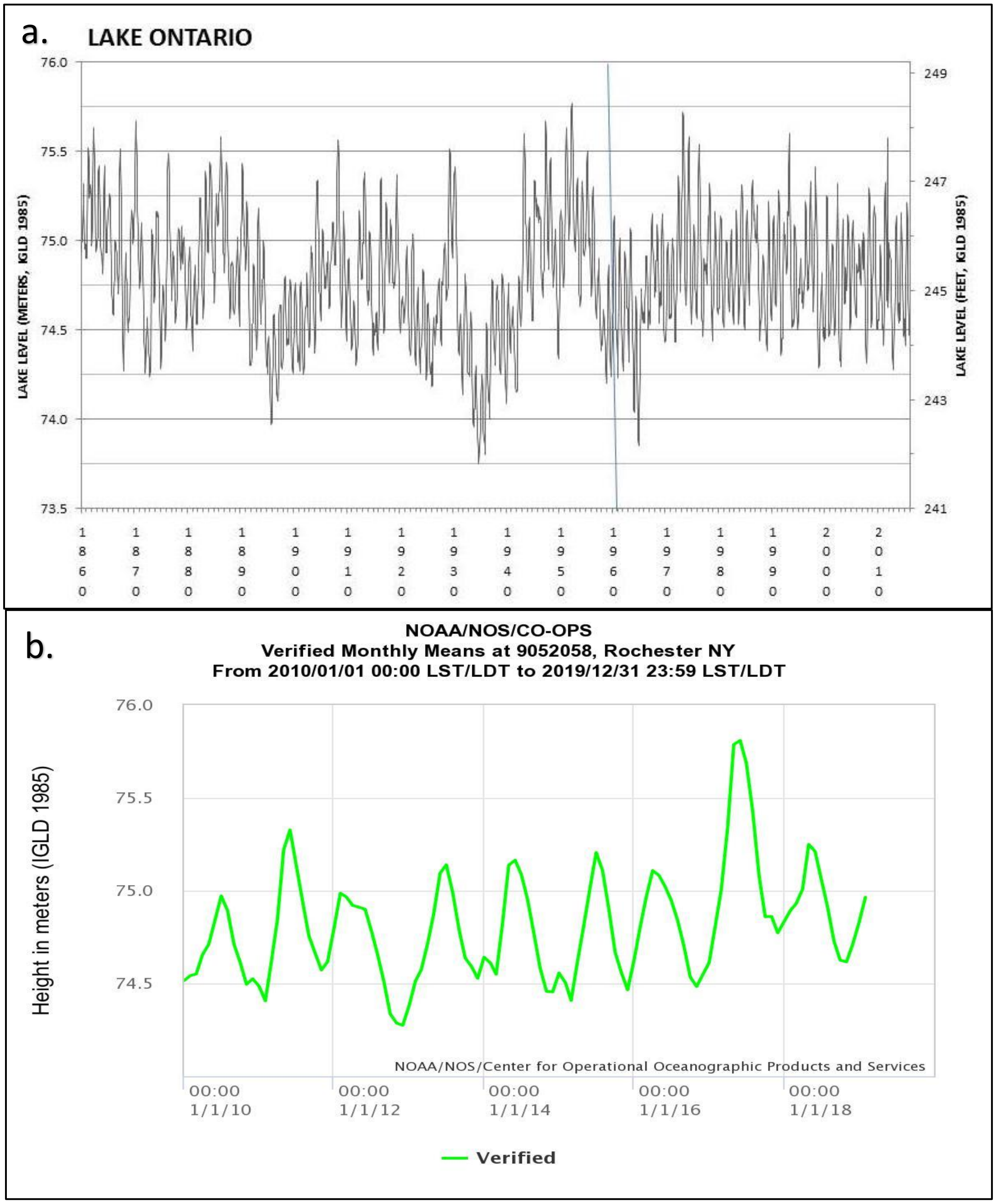


Figure 1. (a) Lake Ontario water level fluctuation from 1860 through 2016 demonstrating the lack of variation after the implementation of lake-level regulation in the 1960s (Wilcox and Bateman 2018). (b) NOAA reported water levels for Rochester, NY from 1 Jan 2010 to 1 Jan 2019 showing the record high water level. Peak in June 2017 at 75.809m (IGLD 1985).

Figure 2. Location in New York State of the Braddock Bay Wildlife Management Area within the Rochester Embayment Area of Concern.

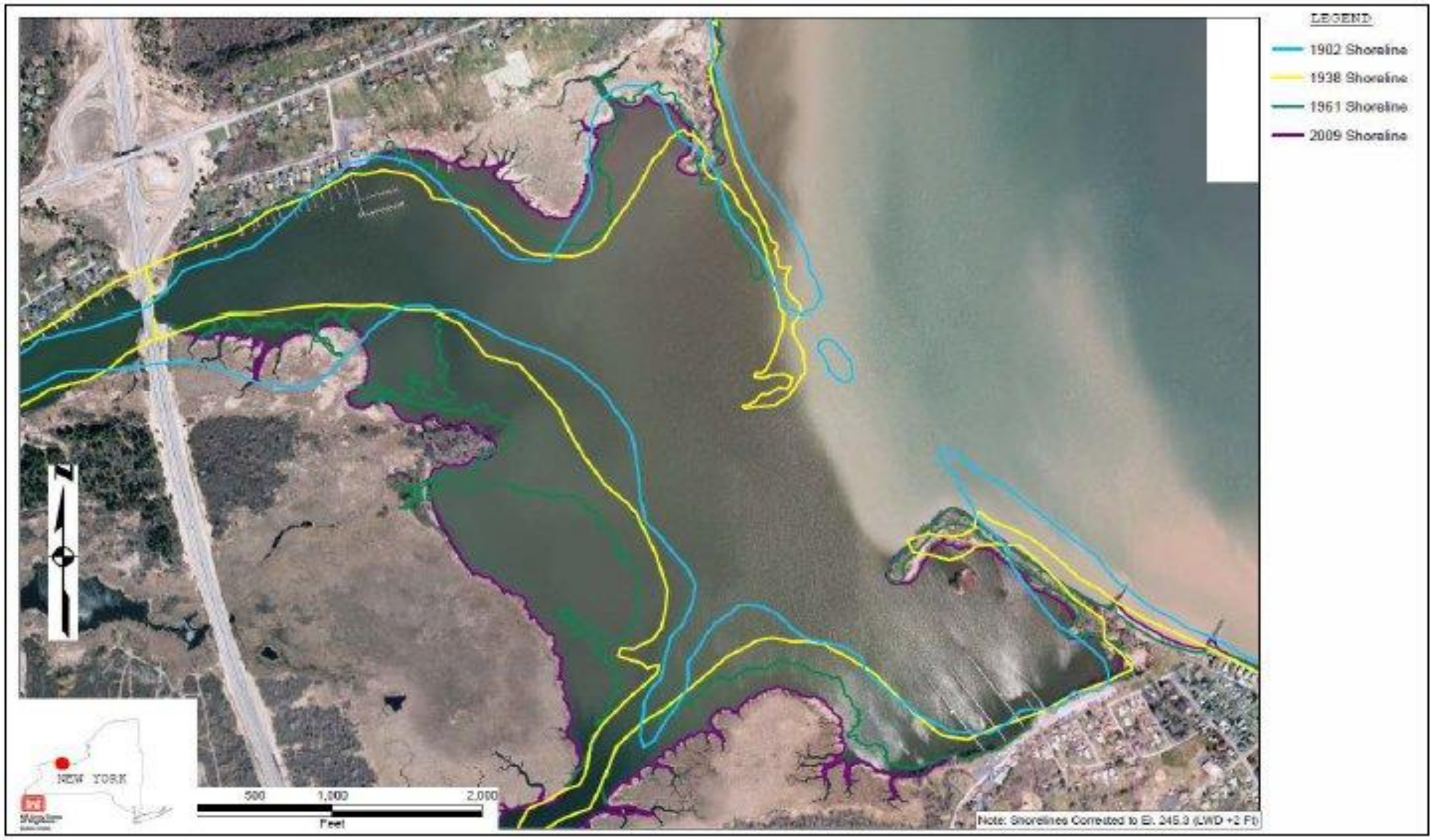


Figure 3. The gradual loss of wetland habitat from wave action-driven erosion within Braddock Bay WMA, illustrating the loss from 1902 to 2009 (USACE 2016A).

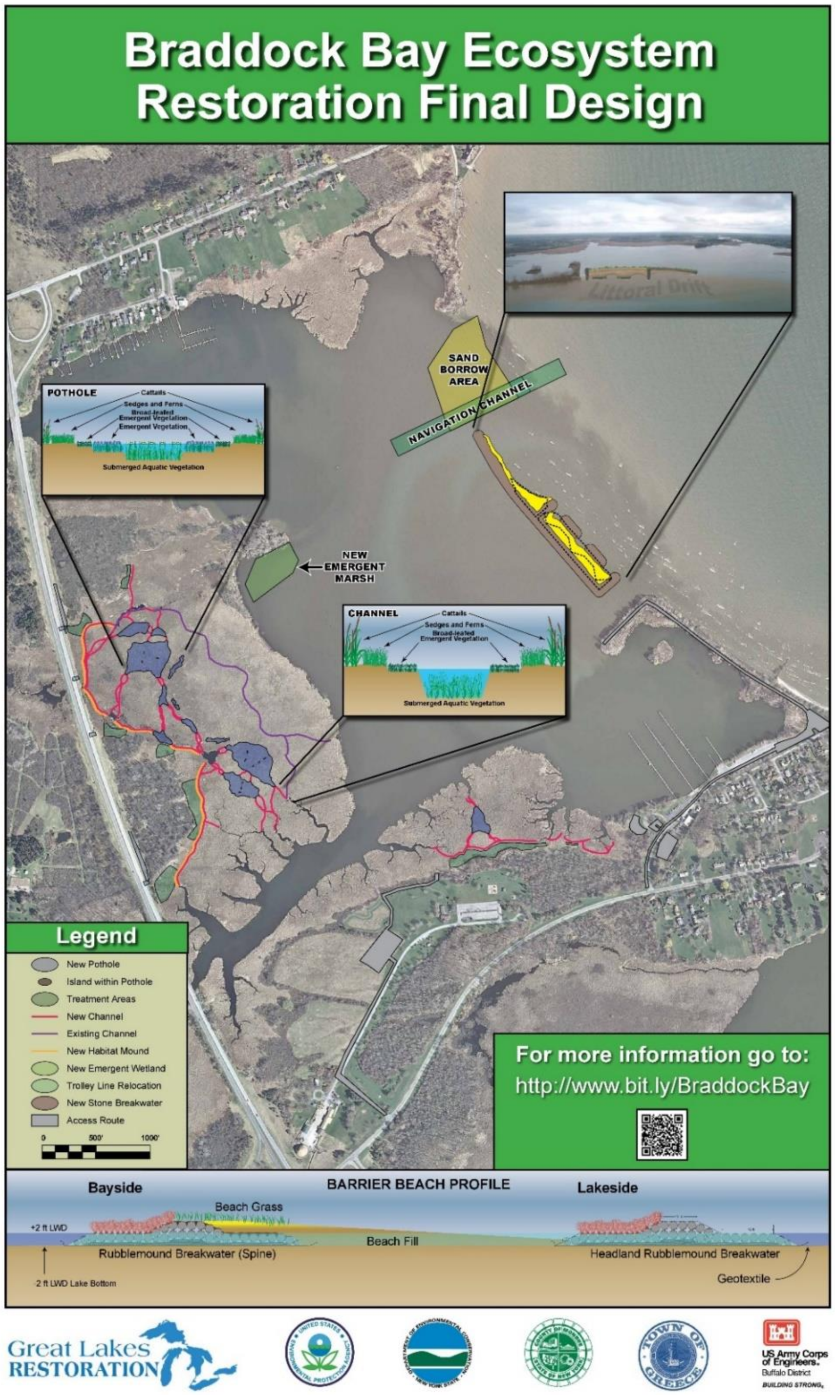


Figure 4. U.S. Army Corps of Engineers plan for the Braddock Bay Restoration Project that shows proposed restoration tactics.

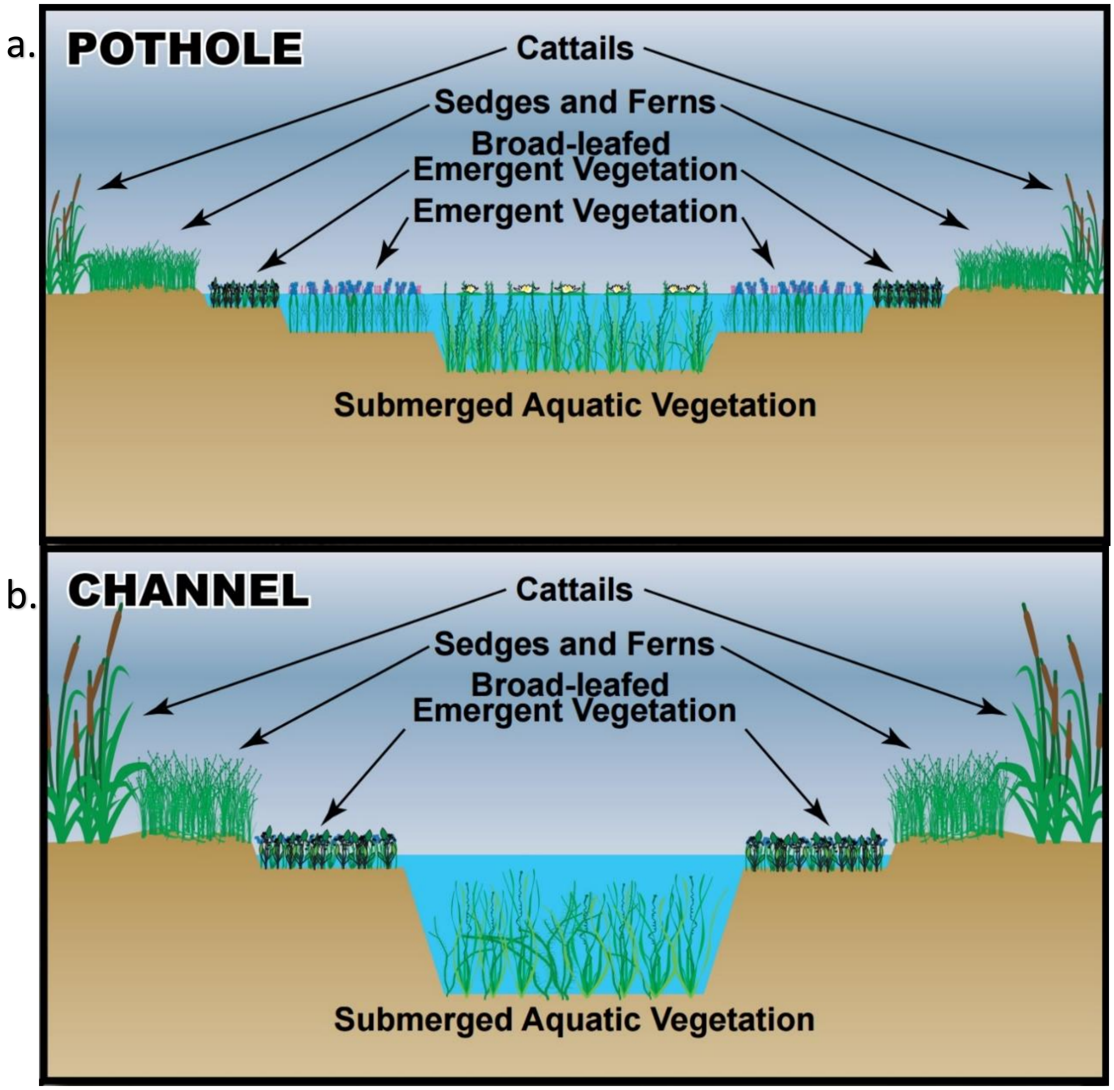


Figure 5(a). A diagram showing the channel formation proposed by the U.S. Army Corps of Engineers (USACE 2016A). (b). A diagram showing the pothole formation proposed by the U.S. Army Corps of Engineers (USACE 2016A).

Figure 6. A photo with the representation of each habitat zone for pothole surveys including the Deep water (D), Bench (B), and the Mound (M) zones.

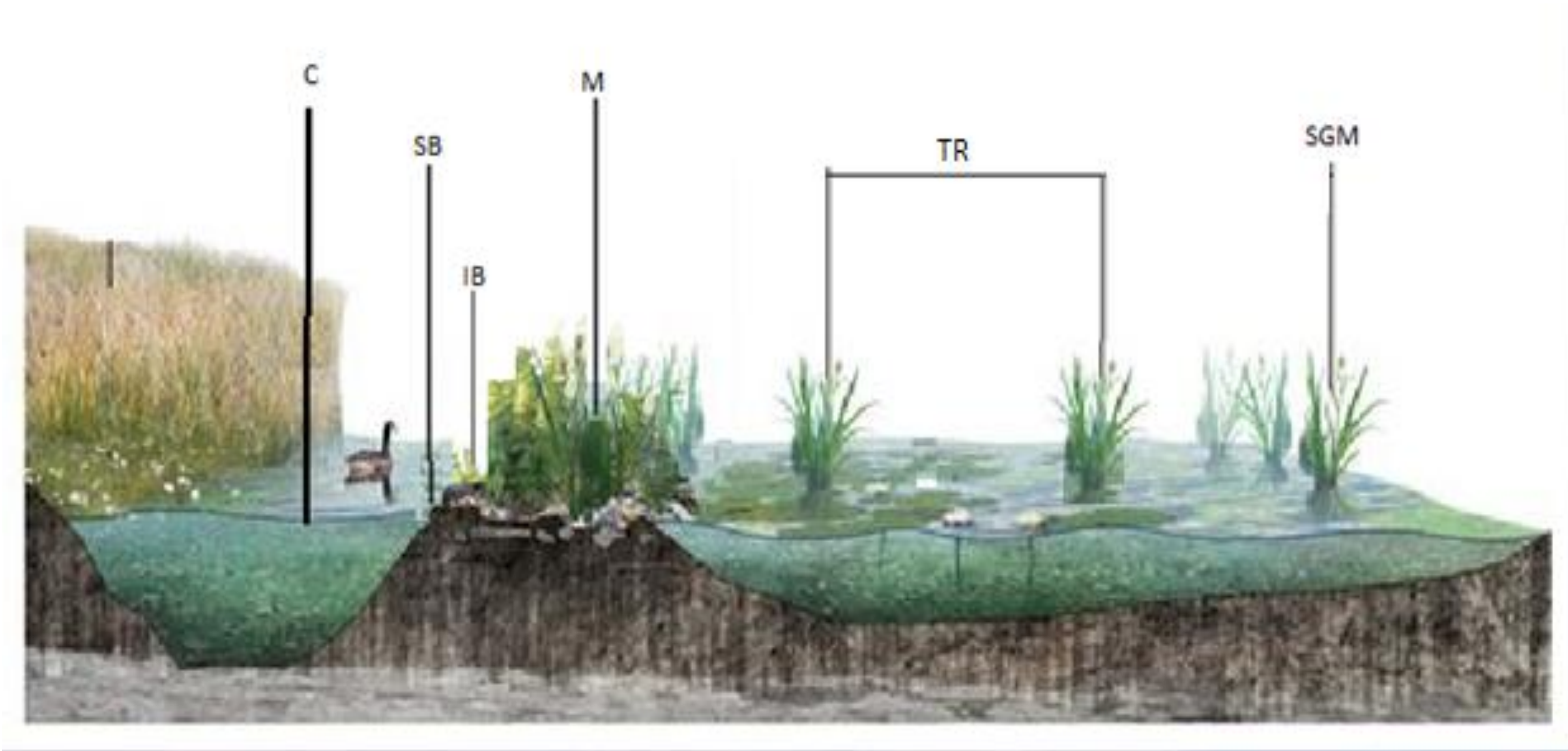


Figure 7. Illustrated example representing each habitat zone sampled for channel surveys including the Sedge/Grass Meadow (SGM), Treated area for cattail (TR), Mound (M), Intermediate Bench (IB), Shallow Bench (SB), and Channel (C) zones. Illustration credit: Kalmaru Arkitektur – wetland section.

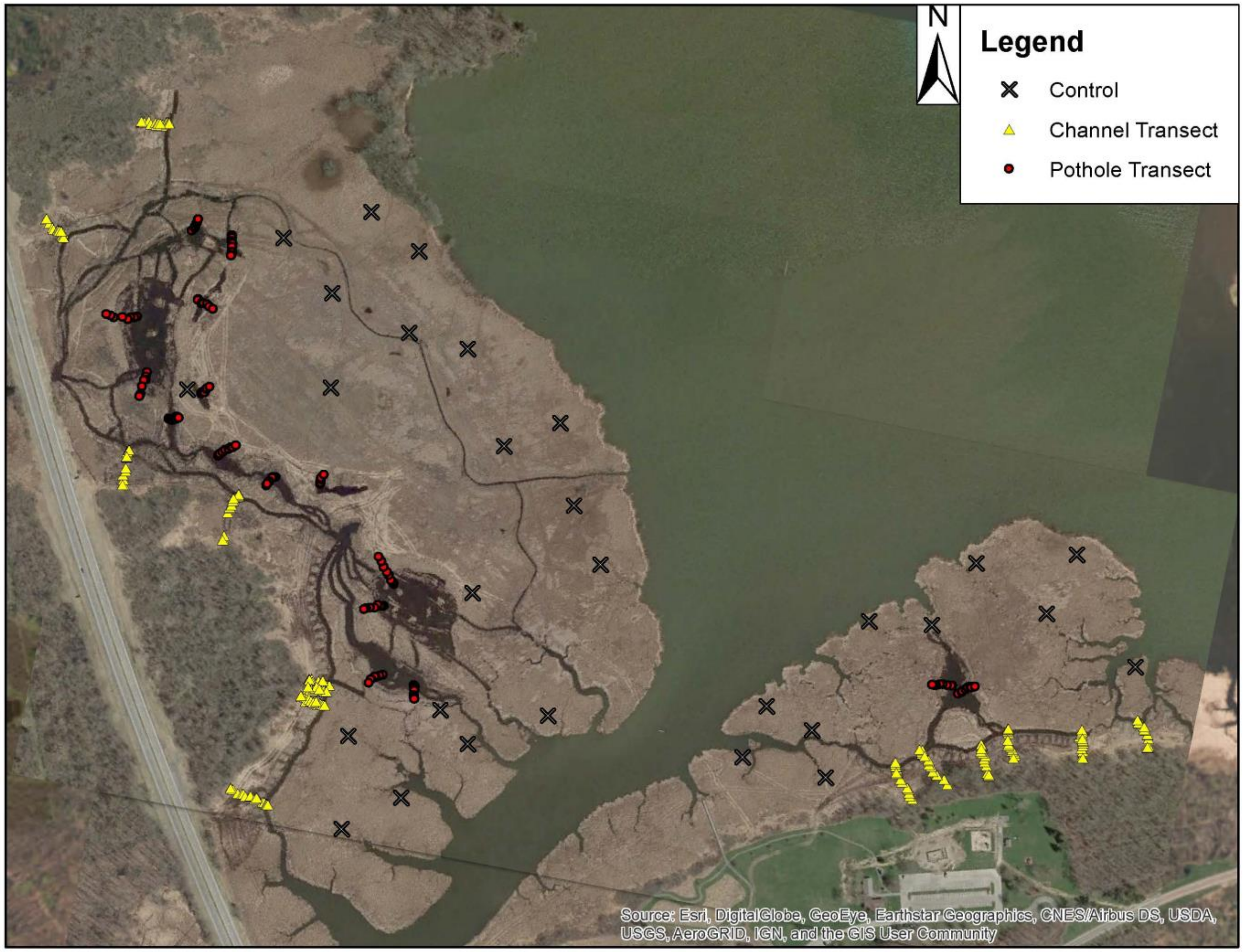


Figure 8. Braddock Bay WMA summer 2016 vegetation sampling pothole transect, channel transect, and control quadrat GPS points.

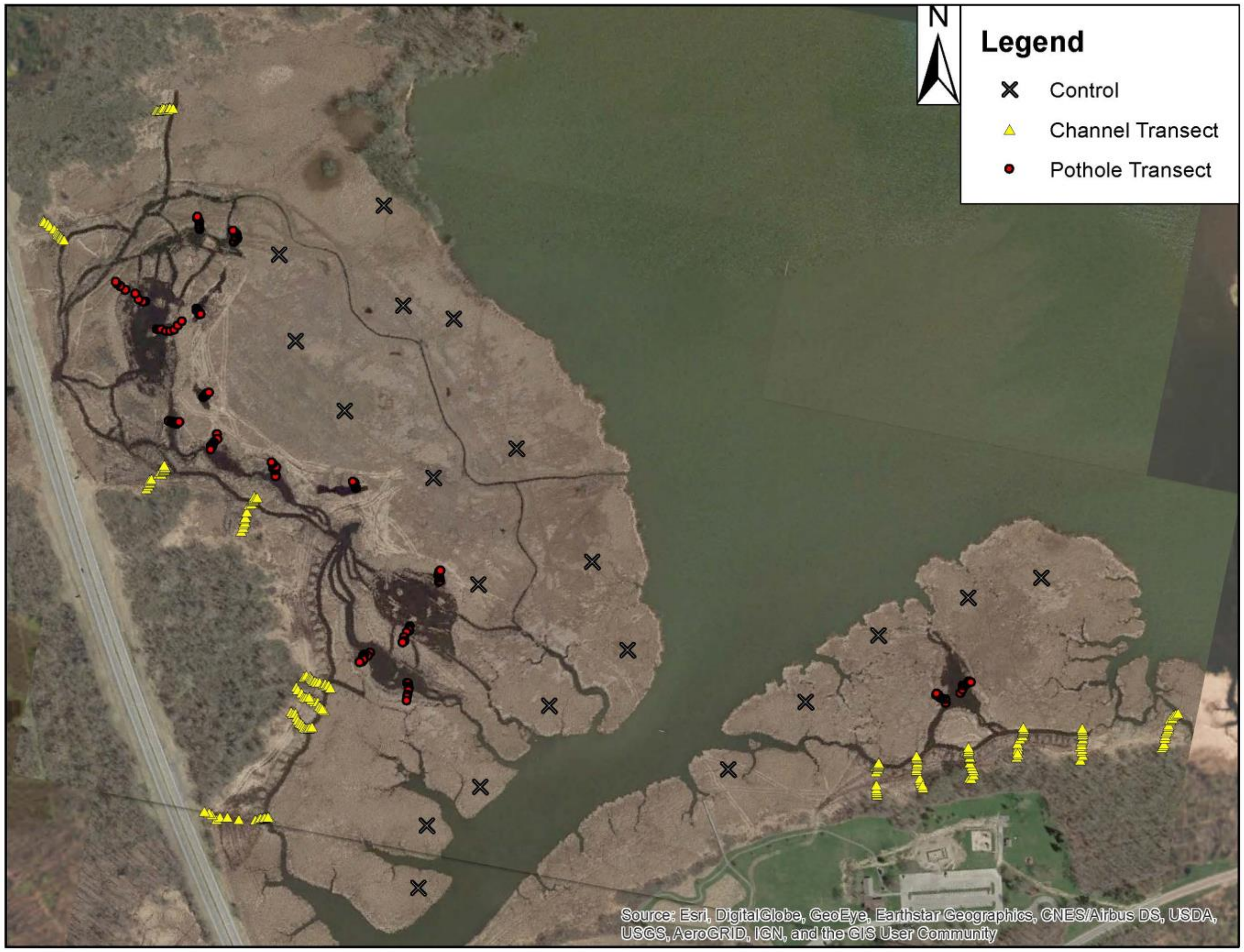


Figure 9. Braddock Bay WMA summer 2017 vegetation sampling pothole transect, channel transect, and control quadrat GPS points.

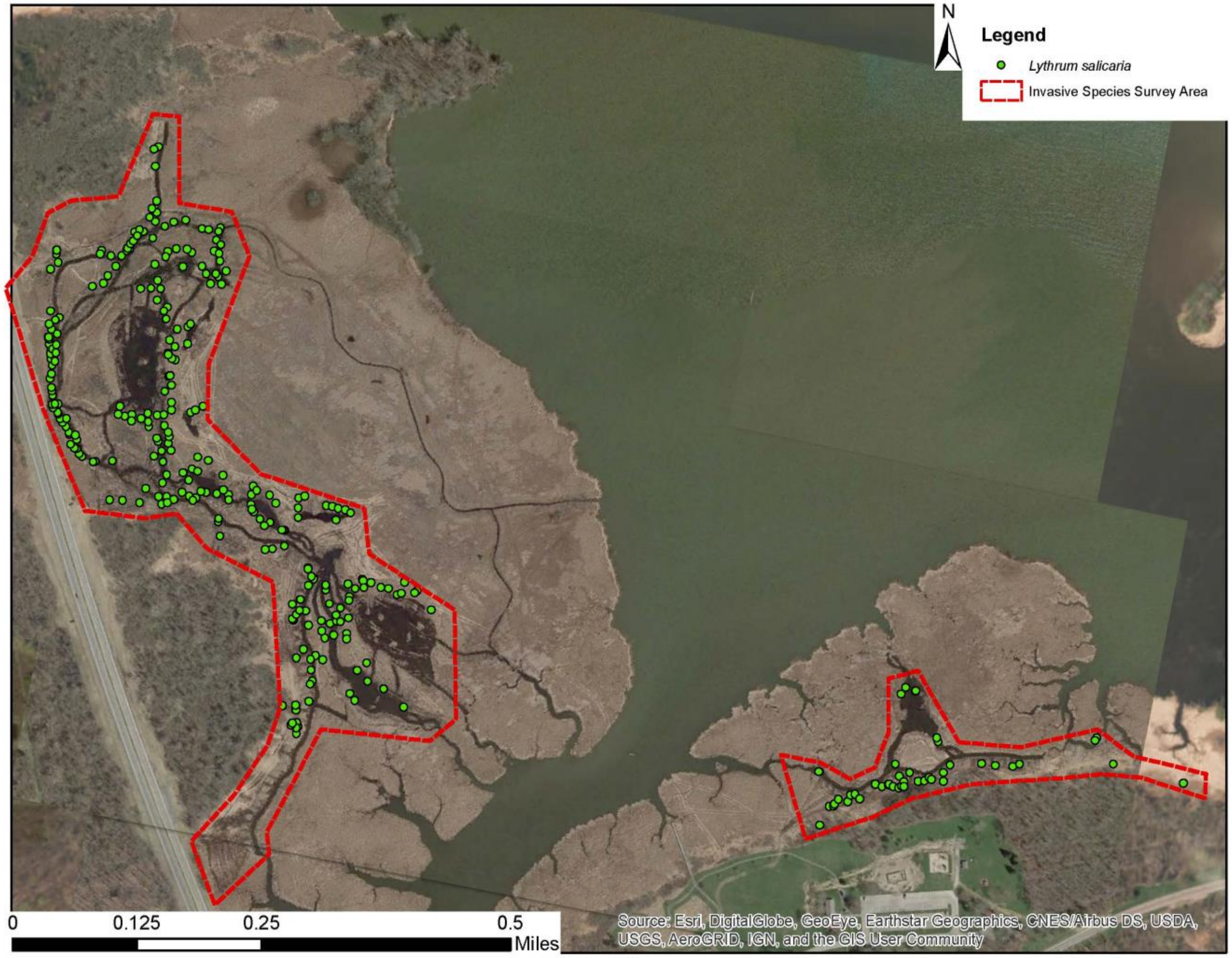


Figure 10. Braddock Bay WMA summer 2016 early season invasive species walking survey GPS points

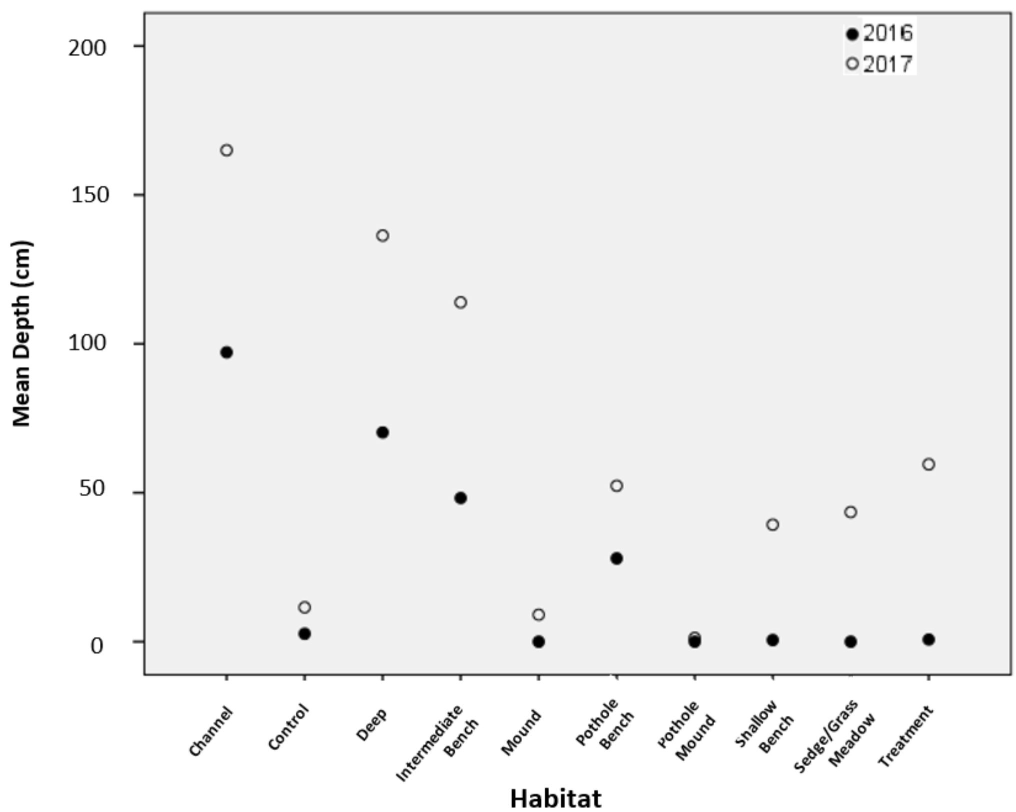


Figure 11. Mean WaterDepth located on the y-axis showing the comparison between the two sampling years (2016 and 2017) within each sampled habitat type represented on the x-axis (C=channel, CT=cattail-control, D=deep zone, IB=intermediate bench, M=mound, PHB=pothole bench, PHM=pothole mounds, SB=shallow bench, SGM=sedge/grass meadow, TR =treatment area).

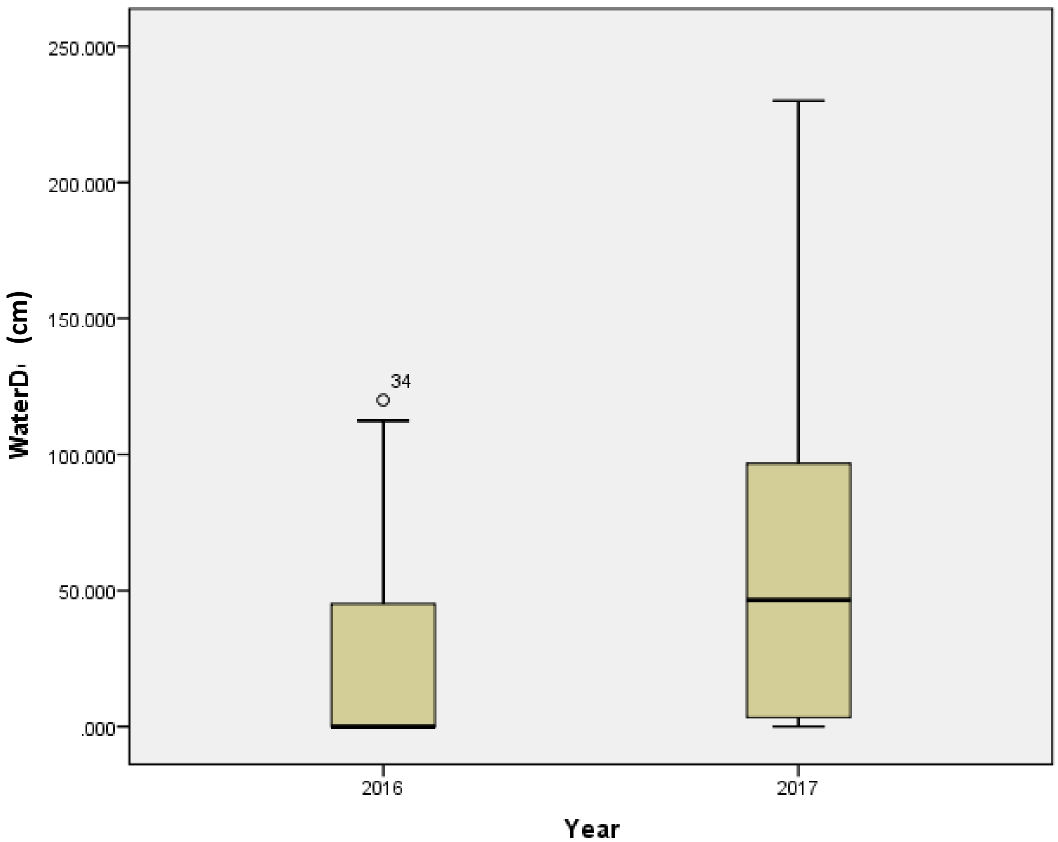


Figure 12. Comparison of Water Depth across the two sampling years of 2016 and 2017.

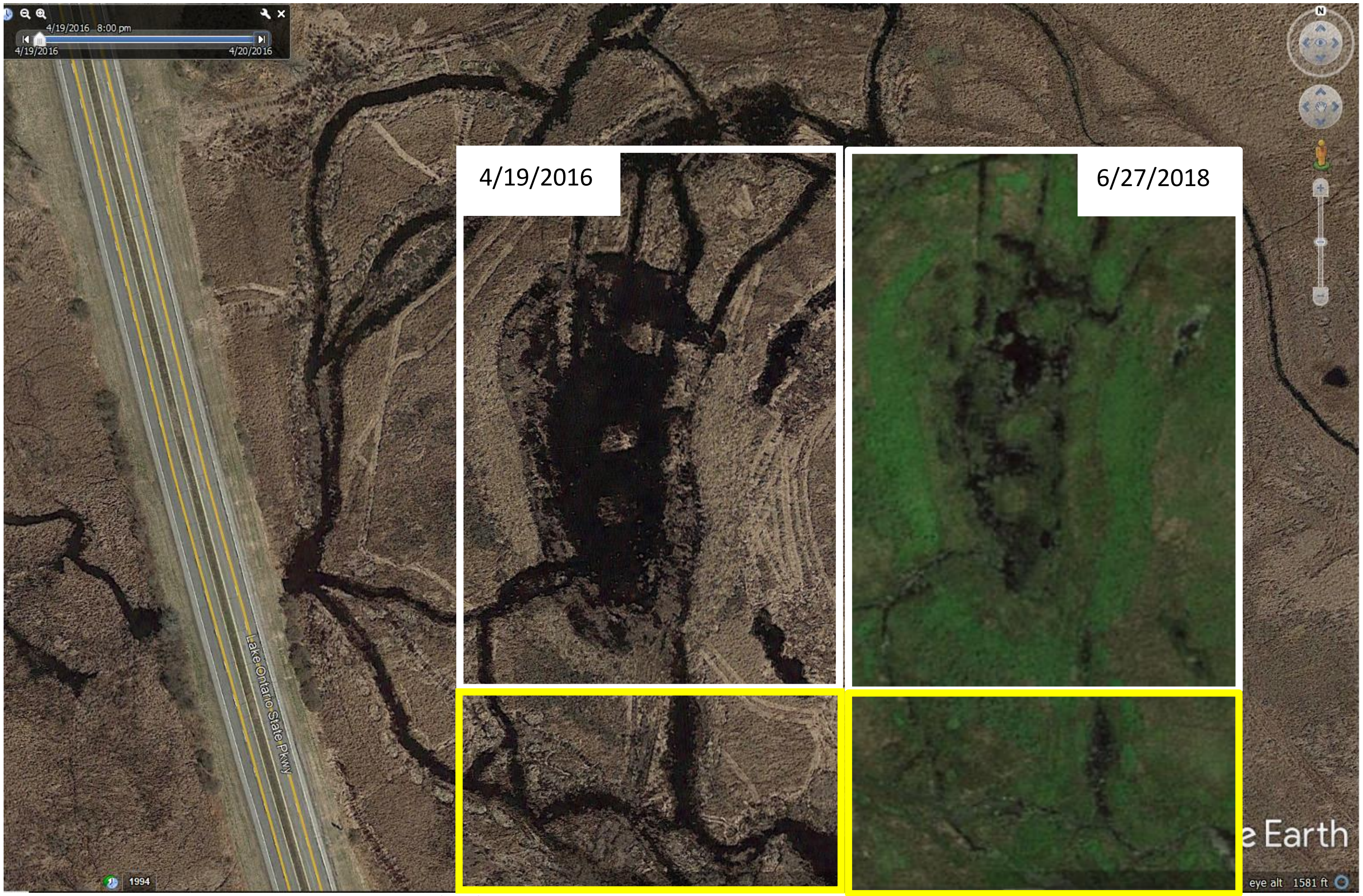
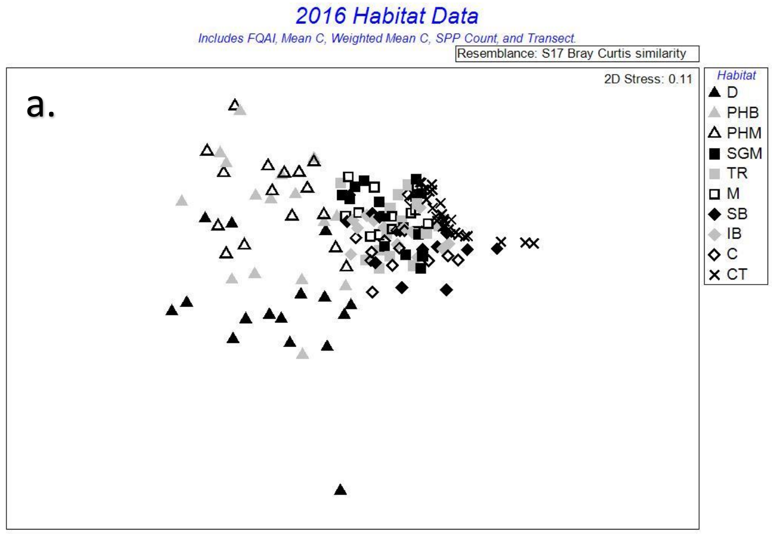


Figure 13. Google Earth aerial photo (4/19/2016) of a portion of the initial excavation of channels and potholes at Braddock Bay. Displayed on the image is a side-by-side comparison of the initial excavation and the most recent Google Earth aerial photo (6/27/2018). Noticeable pothole and channel filling has occurred, creating access issues.



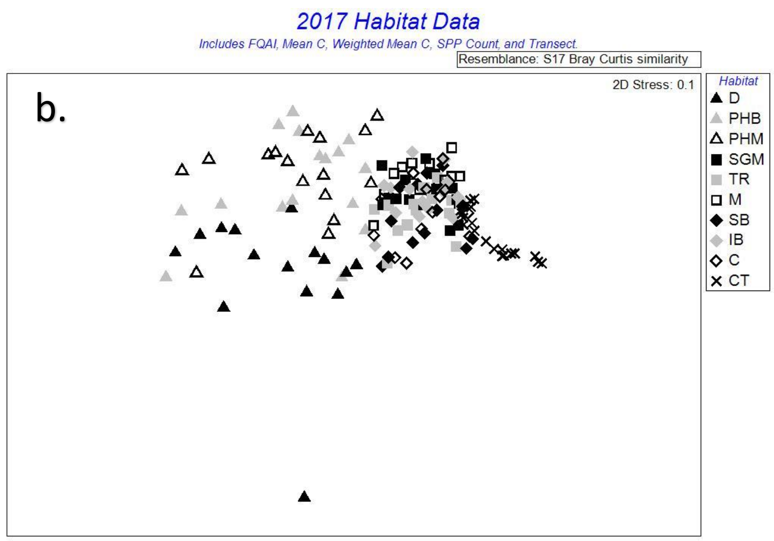


Figure 14. Non-metric multidimensional scaling ordinations for both sampling years, (a) 2016 and (b) 2017, respectively (D=deep zone, PHB=pothole bench, PHM=pothole mounds, SGM=sedge/grass meadow, TR=treatment area in the shallow emergent marsh, M=mound, SB=shallow bench, IB=intermediate bench, C=channel, CT=cattail-control).

Appendix 1. Masterlist of species observed within both sampling years of 2016 and 2017.

Acer saccharinum L.	*Eupatorium perfoliatum* L.	*Rumex orbiculatus* A. Gray
Agrostis stolonifera L.	*Fragaria virginiana* Mill. *ssp. virginiana*	*Sagittaria latifolia* Willd.
Alisma triviale Pursh	*Fraxinus pennsylvanica* Marshall	*Salix fragilis* L.
Apios Americana Medikus	*Galium trifidum* L.	*Schoenoplectus tabernaemontanii* (C.C.Gmel.) Palla
Asclepias incarnata L.	*Hibiscus moscheutos* L.	*Scutellaria galericulata* L.
Azolla caroliniana Kaulf.	*Hydrocharis morsus-ranae* L.	*Sium suave* Walter
Bidens cernua L.	*Impatiens capensis* Meerb.	*Solanum dulcamara* L.
Bidens frondosa L.	*Iris pseudacorus* L.	*Sparganium eurycarpum* Engelm.
*Boehmeria cylindrica (*L.) Swartz	*Juncus canadensis* J.Gay ex Laharpe	*Spiraea alba* Du Roi *var. latifolia* (Aiton) Dippel
Bolboschoenus fluviatilis (Torr.) A. Gray	*Juncus effuses* L.	*Spirodela polyrrhiza* (L.) Schleid
Butomus umbellatus L.	*Lathyrus palustris* L.	*Stachys tenuifollia* Willd.
Calamagrostis Canadensis (Michx.) P.Beauv.	*Leersia oryzoides* (L.) Sw.	*Stuckenia filiformis* (Pers.) Börner
Calystegia sepium (L.) R.Br.	*Lemna minor* L.	*Stuckenia pectinatus* (L.) Börner
Carex comosa Boott	*Lemna trisulca* L.	*Taraxacum officinale* F.H. Wigg.
Carex hystericina Muhl. ex Willd.	*Lycopus americanus* Muhl. ex W.Bartram	*Thelyptris palustris* Schott
Carex lacustris Willd.	*Lycopus virginicus* L.	*Triadenum fraseri* (Spach) Gleason
Carex lurida Wahlenb.	*L. salicaria* L.	*Typha x glauca* Godr. (pro sp.)
Carex stricta Lam.	*Mentha arvensis* L.	*Utricularia vulgaris* L.
Cephalanthus occidentalis L.	*Myosotis scorpioides* L.	*Vallisneria americana* Michx.
Ceratophyllum demersum L.	*Myriophyllum spicatum* L.	*Verbena hastata* L.
Chamerion angustifolium Ség.	*Najas flexilis* (Willd.) Rostk. & Schmidt	*Vernonia noveboracensis* (L.) Michx.
Chara vulgaris L.	*Najas minor* All.	*Vitis riparia* Michx.
Chenopodium glauca L.	*Nuphar lutea* (L.) Sm.	
Cicuta bulbifera (L.) Spreng.	*Nymphaea odorata* Aiton	
Cirsium arvense (L.) Scop.	*Onoclea sensibilis* L.	
Comarum palustre L.	*Persicaria amphibia* (L.) Gray	
Cornus amomum Mill.	*Persicaria hydropiper* (L.) Opiz	
Cornus sericea L.	*Persicaria hydropiperoides* (Michx.) Small	
Cuscuta gronovii Willd. ex Schult. var. *latiflora* Engelm	*Persicaria lapathifolia* (L.) Gray	
Cyperus esculentus L.	*Persicaria maculosa* Gray	
Cyperus fuscus L.	*Persicaria sagittata* (L.) Gross	
Cyperus odoratus L.	*Phragmites australis* Adans.	
Decodon verticillatus (L.) Elliott	*Pontederia cordata* L.	

Appendix 2. Kolmogorov-Smirnov normality tests for sample variables SPPcount, Mean C, Weighted Mean C, FQAI, Total Cover, Detritus Cover, Water Depth, and Sediment Depth in 2016 and 2017.

Tests of Normality

		Kolmogorov-Smirnov[a]			Shapiro-Wilk		
	Year	Statistic	df	Sig.	Statistic	df	Sig.
SPPcount	2016	0.054	162	0.200*	0.985	162	0.088
	2017	0.070	150	0.070	0.985	150	0.113
MeanC	2016	0.079	162	0.016	0.961	162	0.000
	2017	0.066	150	0.200*	0.974	150	0.006
Weighted Mean C	2016	0.075	162	0.027	0.967	162	0.001
	2017	0.053	150	0.200*	0.986	150	0.126
FQAI	2016	0.086	162	0.005	0.973	162	0.003
	2017	0.053	150	0.200*	0.987	150	0.170
Total Cover	2016	0.093	162	0.002	0.968	162	0.001
	2017	0.125	150	0.000	0.952	150	0.000
DetritusCover	2016	0.186	162	0.000	0.860	162	0.000
	2017	0.166	150	0.000	0.863	150	0.000
WaterDepth	2016	0.331	162	0.000	0.694	162	0.000
	2017	0.156	150	0.000	0.879	150	0.000
SedimentDepth	2016	0.286	162	0.000	0.636	162	0.000
	2017	0.085	150	0.010	0.950	150	0.000

*. This is a lower bound of the true significance.

a. Lilliefors Significance Correction

Appendix 3. Kolmogorov-Smirnov normality tests for sample variables SPPcount, Mean C, Weighted Mean C, and FQAI in 2013 and 2017.

Tests of Normality

		Kolmogorov-Smirnov[a]			Shapiro-Wilk		
	Year	Statistic	df	Sig.	Statistic	df	Sig.
SPPcount	2013	0.179	41	0.002	0.920	41	0.007
	2017	0.143	55	0.007	0.951	55	0.025
MeanC	2013	0.089	41	0.200*	0.973	41	0.423
	2017	0.095	55	0.200*	0.961	55	0.073
Weighted Mean C	2013	0.227	41	0.000	0.751	41	0.000
	2017	0.120	55	0.046	0.956	55	0.044
FQAI	2013	0.096	41	0.200*	0.974	41	0.459
	2017	0.079	55	0.200*	0.979	55	0.436

*. This is a lower bound of the true significance.

a. Lilliefors Significance Correction

Appendix 4. A Multivariate Generalized Linear Model (GLM) used to compare the averaged SPPcount, Mean C, Weighted Mean C, FQAI, Total Cover, Detritus Cover, Water Depth, and Sediment Depth across year 2016 to 2017.

Tests of Between-Subjects Effects

Source	Dependent Variable	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	SPPcount	590.855[a]	61	9.686	1.851	0.001
	MeanC	44.594[b]	61	0.731	0.961	0.562
	Weighted Mean C	139.521[c]	61	2.287	1.260	0.113
	FQAI	475.030[d]	61	7.787	1.511	0.015
	Total Cover	42020.895[e]	61	688.867	1.232	0.137
	DetritusCover	149477.665[f]	61	2450.454	3.696	0.000
	WaterDepth	237256.331[g]	61	3889.448	1.643	0.004
	SedimentDepth	1494347.176[h]	61	24497.495	7.385	0.000
Intercept	SPPcount	9197.390	1	9197.390	1757.653	0.000
	MeanC	1952.183	1	1952.183	2565.295	0.000
	Weighted Mean C	1686.463	1	1686.463	929.280	0.000
	FQAI	11177.251	1	11177.251	2169.057	0.000
	Total Cover	892673.079	1	892673.079	1596.488	0.000
	DetritusCover	164381.673	1	164381.673	247.905	0.000
	WaterDepth	519418.058	1	519418.058	219.420	0.000
	SedimentDepth	1193133.731	1	1193133.731	359.684	0.000
Year	SPPcount	20.251	1	20.251	3.870	0.050
	MeanC	.291	1	0.291	0.382	0.537
	Weighted Mean C	9.253	1	9.253	5.099	0.025
	FQAI	13.140	1	13.140	2.550	0.112
	Total Cover	5649.973	1	5649.973	10.105	0.002
	DetritusCover	3420.831	1	3420.831	5.159	0.024
	WaterDepth	79011.034	1	79011.034	33.377	0.000
	SedimentDepth	492742.729	1	492742.729	148.543	0.000
Transect#	SPPcount	298.540	30	9.951	1.902	0.004

	MeanC	25.980	30	0.866	1.138	0.291
	Weighted Mean C	97.313	30	3.244	1.787	0.009
	FQAI	231.071	30	7.702	1.495	0.053
	Total Cover	21376.446	30	712.548	1.274	0.162
	DetritusCover	104293.516	30	3476.451	5.243	0.000
	WaterDepth	97547.198	30	3251.573	1.374	0.100
	SedimentDepth	701109.569	30	23370.319	7.045	0.000
Year * Transect#	SPPcount	259.880	30	8.663	1.655	0.021
	MeanC	18.517	30	0.617	0.811	0.749
	Weighted Mean C	31.459	30	1.049	0.578	0.963
	FQAI	228.740	30	7.625	1.480	0.057
	Total Cover	18321.615	30	610.720	1.092	0.346
	DetritusCover	18458.225	30	615.274	0.928	0.579
	WaterDepth	38835.842	30	1294.528	0.547	0.975
	SedimentDepth	138636.522	30	4621.217	1.393	0.091
Error	SPPcount	1308.192	250	5.233		
	MeanC	190.249	250	0.761		
	Weighted Mean C	453.702	250	1.815		
	FQAI	1288.261	250	5.153		
	Total Cover	139787.017	250	559.148		
	DetritusCover	165770.765	250	663.083		
	WaterDepth	591807.153	250	2367.229		
	SedimentDepth	829293.726	250	3317.175		
Total	SPPcount	12917.076	312			
	MeanC	2611.944	312			
	Weighted Mean C	2475.072	312			
	FQAI	15301.515	312			
	Total Cover	1389282.031	312			
	DetritusCover	674202.648	312			
	WaterDepth	1345713.368	312			
	SedimentDepth	4249233.965	312			
	SPPcount	1899.048	311			

Corrected Total	MeanC	234.843	311			
	Weighted Mean C	593.222	311			
	FQAI	1763.291	311			
	Total Cover	181807.912	311			
	DetritusCover	315248.431	311			
	WaterDepth	829063.483	311			
	SedimentDepth	2323640.902	311			

a. R Squared = .311 (Adjusted R Squared = .143)
b. R Squared = .190 (Adjusted R Squared = -.008)
c. R Squared = .235 (Adjusted R Squared = .049)
d. R Squared = .269 (Adjusted R Squared = .091)
e. R Squared = .231 (Adjusted R Squared = .044)
f. R Squared = .474 (Adjusted R Squared = .346)
g. R Squared = .286 (Adjusted R Squared = .112)
h. R Squared = .643 (Adjusted R Squared = .556)

Appendix 5. Non-parametric Independent-samples Kruskal-Wallis one-way ANOVA comparing the averaged SPPcount, Mean C, Weighted Mean C, FQAI, Total Cover, Detritus Cover, Water Depth, and Sediment Depth for the cattail mat - control habitat (CT) in 2016 vs 2017.

	Test statistic	P-value	df
SPP Count	0.085	0.771	1
Mean C	0.072	0.789	1
Weighted Mean C	0.319	0.572	1
FQAI	0.454	0.500	1
TotalCover	1.541	0.214	1
DetritusCover	12.454	<0.001	1
WaterDepth	6.817	0.009	1
SedimentDepth	24.876	<0.001	1

Appendix 6. A Multivariate Generalized Linear Model (GLM) used to compare individual year 2016 data across habitats using FQAI, mean C, and weighted C as factors blocked by transect. Control quadrats were not included in this analysis.

Tests of Between-Subjects Effects

Source	Dependent Variable	Type III Sum of Squares	df	Mean Square	F	Sig.
Model	MeanC	1326.453[a]	133	9.973	5.678	0.000
	Weighted C	1078.588[b]	133	8.110	6.549	0.000
	FQAI	7116.841[c]	133	53.510	5.006	0.000
Habitat	MeanC	24.107	7	3.444	1.961	0.096
	Weighted C	112.608	7	16.087	12.990	0.000
	FQAI	162.066	7	23.152	2.166	0.068
Transect#	MeanC	18.754	28	0.670	0.381	0.994
	Weighted C	25.561	28	0.913	0.737	0.789
	FQAI	130.828	28	4.672	0.437	0.984
Transect# * Habitat	MeanC	51.854	95	0.546	0.311	1.000
	Weighted C	100.528	95	1.058	0.855	0.720
	FQAI	338.917	95	3.568	0.334	1.000
Error	MeanC	50.934	29	1.756		
	Weighted C	35.912	29	1.238		
	FQAI	309.967	29	10.689		
Total	MeanC	1377.387	162			
	Weighted C	1114.500	162			
	FQAI	7426.808	162			

a. R Squared = 0.963 (Adjusted R Squared = 0.793)
b. R Squared = 0.968 (Adjusted R Squared = 0.820)
c. R Squared = 0.958 (Adjusted R Squared = 0.767)

Appendix 7. A Multivariate Generalized Linear Model (GLM) used to compare individual year 2017 data across habitats using FQAI, mean C, and weighted C as factors blocked by transect. Control quadrats were not included in this analysis.

Tests of Between-Subjects Effects

Source	Dependent Variable	Type III Sum of Squares	df	Mean Square	F	Sig.
Model	MeanC	1212.432[a]	131	9.255	7.948	0.000
	Weighted C	1323.382[b]	131	10.102	5.161	0.000
	FQAI	7753.519[c]	131	59.187	9.279	0.000
Habitat	MeanC	11.197	7	1.600	1.374	0.272
	Weighted C	74.741	7	10.677	5.455	0.001
	FQAI	117.828	7	16.833	2.639	0.044
Transect#	MeanC	13.746	28	0.491	0.422	0.981
	Weighted C	36.903	28	1.318	0.673	0.833
	FQAI	162.101	28	5.789	0.908	0.601
Transect# * Habitat	MeanC	30.034	93	0.323	0.277	1.000
	Weighted C	92.723	93	0.997	0.509	0.982
	FQAI	238.295	93	2.562	0.402	0.998
Error	MeanC	22.124	19	1.164		
	Weighted C	37.190	19	1.957		
	FQAI	121.188	19	6.378		
Total	MeanC	1234.557	150			
	Weighted C	1360.572	150			
	FQAI	7874.707	150			

a. R Squared = 0.982 (Adjusted R Squared = 0.859)
b. R Squared = 0.973 (Adjusted R Squared = 0.784)
c. R Squared = 0.985 (Adjusted R Squared = 0.879)

Appendix 8. Non-parametric Independent-samples Kruskal-Wallis one-way ANOVA for pre-restoration 2013 vegetation data for the Shallow Emergent Marsh zone (SEM) and 2017 treatment area (TR) data from the comparing SPPcount, Mean C, Weighted Mean C, and FQAI. This comparison presents a 2013 pre-restoration shallow emergent marsh compared to post-construction treatment, found within the same locations.

	Test statistic	P-value	df
SPP Count	14.145	< 0.001	1
Mean C	18.377	< 0.001	1
Weighted Mean C	53.178	< 0.001	1
FQAI	23.508	< 0.001	1

Appendix 9. Non-parametric Independent-samples Kruskal-Wallis one-way ANOVA comparing the averaged SPPcount, Mean C, Weighted Mean C, FQAI, Total Cover, Detritus Cover, Water Depth, and Sediment Depth for the treatment habitat (TR) in 2016 vs 2017.

	Test statistic	P-value	df
SPP Count	0.000	0.988	1
Mean C	1.257	0.262	1
Weighted Mean C	19.936	< 0.001	1
FQAI	0.625	0.429	1
TotalCover	0.687	0.407	1
DetritusCover	22.000	< 0.001	1
WaterDepth	89.680	< 0.001	1
SedimentDepth	48.771	< 0.001	1

Appendix 10. A Multivariate Generalized Linear Model (GLM) used to compare the averaged SPPcount, Mean C, Weighted Mean C, FQAI, Total Cover, Detritus Cover, Water Depth, and Sediment Depth across 2016 and 2017 for the overall channel habitat which includes the channel (C), shallow bench (SB), and intermediate bench (IB) habitats.

Tests of Between-Subjects Effects

Source	Dependent Variable	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	MeanC	11.607[a]	27	0.430	0.677	0.865
	Weighted C	40.813[b]	27	1.512	0.616	0.914
	SPPcount	201.602[c]	27	7.467	2.667	0.001
	FQAI	146.896[d]	27	5.441	1.846	0.027
	Total Cover	12778.727[e]	27	473.286	0.898	0.611
	DetritusCover	6550.515[f]	27	242.612	0.292	1.000
	WaterDepth	76412.851[g]	27	2830.106	0.817	0.712
	SedimentDepth	458150.791[h]	27	16968.548	36.920	0.000
Intercept	MeanC	613.492	1	613.492	965.936	0.000
	Weighted C	518.632	1	518.632	211.277	0.000
	SPPcount	2965.056	1	2965.056	1058.948	0.000
	FQAI	3462.099	1	3462.099	1174.684	0.000
	Total Cover	339318.001	1	339318.001	644.020	0.000
	DetritusCover	36280.720	1	36280.720	43.740	0.000
	WaterDepth	480187.792	1	480187.792	138.679	0.000
	SedimentDepth	466608.352	1	466608.352	1015.244	0.000
Year	MeanC	2.246	1	2.246	3.536	0.065
	Weighted C	10.807	1	10.807	4.403	0.040
	SPPcount	47.161	1	47.161	16.843	0.000
	FQAI	56.653	1	56.653	19.222	0.000
	Total Cover	30.720	1	30.720	0.058	0.810
	DetritusCover	1906.597	1	1906.597	2.299	0.135
	WaterDepth	63087.792	1	63087.792	18.220	0.000
	SedimentDepth	421228.089	1	421228.089	916.506	0.000

Transect#	MeanC	5.638	13	0.434	0.683	0.771
	Weighted C	20.379	13	1.568	0.639	0.811
	SPPcount	81.825	13	6.294	2.248	0.019
	FQAI	54.172	13	4.167	1.414	0.183
	Total Cover	9228.434	13	709.880	1.347	0.215
	DetritusCover	2531.446	13	194.727	0.235	0.997
	WaterDepth	6945.169	13	534.244	0.154	1.000
	SedimentDepth	22285.885	13	1714.299	3.730	0.000
Year * Transect#	MeanC	3.571	13	0.275	0.432	0.951
	Weighted C	9.208	13	0.708	0.289	0.991
	SPPcount	77.622	13	5.971	2.132	0.026
	FQAI	35.800	13	2.754	0.934	0.525
	Total Cover	3586.677	13	275.898	0.524	0.900
	DetritusCover	2062.289	13	158.638	0.191	0.999
	WaterDepth	4745.531	13	365.041	0.105	1.000
	SedimentDepth	12579.363	13	967.643	2.105	0.028
Error	MeanC	34.932	55	0.635		
	Weighted C	135.011	55	2.455		
	SPPcount	154.000	55	2.800		
	FQAI	162.099	55	2.947		
	Total Cover	28978.125	55	526.875		
	DetritusCover	45620.833	55	829.470		
	WaterDepth	190441.667	55	3462.576		
	SedimentDepth	25278.125	55	459.602		
Total	MeanC	665.184	83			
	Weighted C	700.557	83			
	SPPcount	3301.750	83			
	FQAI	3776.531	83			
	Total Cover	384835.938	83			
	DetritusCover	87970.312	83			
	WaterDepth	749993.750	83			
	SedimentDepth	941718.750	83			
Corrected Total	MeanC	46.539	82			
	Weighted C	175.825	82			
	SPPcount	355.602	82			
	FQAI	308.996	82			

	Total Cover	41756.852	82			
	DetritusCover	52171.348	82			
	WaterDepth	266854.518	82			
	SedimentDepth	483428.916	82			

a. R Squared = .249 (Adjusted R Squared = -.119)

b. R Squared = .232 (Adjusted R Squared = -.145)

c. R Squared = .567 (Adjusted R Squared = .354)

d. R Squared = .475 (Adjusted R Squared = .218)

e. R Squared = .306 (Adjusted R Squared = -.035)

f. R Squared = .126 (Adjusted R Squared = -.304)

g. R Squared = .286 (Adjusted R Squared = -.064)

h. R Squared = .948 (Adjusted R Squared = .922)

Appendix 11. A Multivariate Generalized Linear Model (GLM) used to compare the averaged SPPcount, Mean C, Weighted Mean C, FQAI, Total Cover, Detritus Cover, Water Depth, and Sediment Depth across 2016 and 2017 for the overall pothole habitat which includes the deep (D) and pothole bench (PHB) habitats.

Tests of Between-Subjects Effects

Source	Dependent Variable	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	MeanC	30.279[a]	31	0.977	1.024	0.474
	Weighted C	49.195[b]	31	1.587	0.557	0.946
	SPPcount	183.029[c]	31	5.904	0.539	0.955
	FQAI	133.683[d]	31	4.312	0.683	0.853
	Total Cover	19624.063[e]	31	633.034	0.723	0.814
	DetritusCover	4729.377[f]	31	152.561	0.447	0.986
	WaterDepth	93172.509[g]	31	3005.565	1.067	0.429
	SedimentDepth	312910.937[h]	31	10093.901	3.473	0.000
Intercept	MeanC	493.631	1	493.631	517.260	0.000
	Weighted C	520.957	1	520.957	182.933	0.000
	SPPcount	1618.485	1	1618.485	147.755	0.000
	FQAI	2207.884	1	2207.884	349.867	0.000
	Total Cover	144356.229	1	144356.229	164.865	0.000
	DetritusCover	14219.342	1	14219.342	41.646	0.000
	WaterDepth	311838.552	1	311838.552	110.705	0.000
	SedimentDepth	580953.455	1	580953.455	199.894	0.000
Year	MeanC	2.959	1	2.959	3.101	0.088
	Weighted C	.098	1	0.098	0.034	0.854
	SPPcount	1.670	1	1.670	0.152	0.699
	FQAI	.003	1	0.003	0.000	0.983
	Total Cover	6238.426	1	6238.426	7.125	0.012
	DetritusCover	719.510	1	719.510	2.107	0.157
	WaterDepth	29414.815	1	29414.815	10.442	0.003
	SedimentDepth	225693.034	1	225693.034	77.656	0.000
Transect#	MeanC	19.253	15	1.284	1.345	0.235
	Weighted C	26.386	15	1.759	0.618	0.838

	SPPcount	73.358	15	4.891	0.446	0.950
	FQAI	77.062	15	5.137	0.814	0.655
	Total Cover	7031.841	15	468.789	0.535	0.900
	DetritusCover	1167.943	15	77.863	0.228	0.998
	WaterDepth	35039.406	15	2335.960	0.829	0.640
	SedimentDepth	22975.840	15	1531.723	0.527	0.905
Year * Transect#	MeanC	8.144	15	0.543	0.569	0.876
	Weighted C	22.512	15	1.501	0.527	0.905
	SPPcount	108.395	15	7.226	0.660	0.802
	FQAI	56.679	15	3.779	0.599	0.853
	Total Cover	6533.850	15	435.590	0.497	0.923
	DetritusCover	2780.509	15	185.367	0.543	0.895
	WaterDepth	27267.923	15	1817.862	0.645	0.814
	SedimentDepth	64550.370	15	4303.358	1.481	0.173
Error	MeanC	29.584	31	0.954		
	Weighted C	88.282	31	2.848		
	SPPcount	339.569	31	10.954		
	FQAI	195.630	31	6.311		
	Total Cover	27143.750	31	875.605		
	DetritusCover	10584.462	31	341.434		
	WaterDepth	87322.222	31	2816.846		
	SedimentDepth	90095.403	31	2906.303		
Total	MeanC	564.216	63			
	Weighted C	670.561	63			
	SPPcount	2156.472	63			
	FQAI	2576.342	63			
	Total Cover	191216.667	63			
	DetritusCover	29904.340	63			
	WaterDepth	495409.028	63			
	SedimentDepth	975800.806	63			
Corrected Total	MeanC	59.863	62			
	Weighted C	137.476	62			
	SPPcount	522.599	62			
	FQAI	329.313	62			
	Total Cover	46767.813	62			
	DetritusCover	15313.839	62			

	WaterDepth	180494.731	62			
	SedimentDepth	403006.340	62			

a. R Squared = .506 (Adjusted R Squared = .012)

b. R Squared = .358 (Adjusted R Squared = -.284)

c. R Squared = .350 (Adjusted R Squared = -.300)

d. R Squared = .406 (Adjusted R Squared = -.188)

e. R Squared = .420 (Adjusted R Squared = -.161)

f. R Squared = .309 (Adjusted R Squared = -.382)

g. R Squared = .516 (Adjusted R Squared = .032)

h. R Squared = .776 (Adjusted R Squared = .553)

Appendix 12. Non-parametric Independent-samples Kruskal-Wallis one-way ANOVA comparing the averaged SPPcount, Mean C, Weighted Mean C, FQAI, Total Cover, Detritus Cover, Water Depth, and Sediment Depth for the pothole mound habitat (PHM) in 2016 vs 2017.

	Test statistic	P-value	df
SPP Count	1.737	0.188	1
Mean C	0.231	0.631	1
Weighted Mean C	7.095	0.008	1
FQAI	0.080	0.778	1
TotalCover	19.904	< 0.001	1
DetritusCover	0.244	0.621	1
WaterDepth	5.215	0.022	1
SedimentDepth	14.908	< 0.001	1

Appendix 13. Non-parametric Independent-samples Kruskal-Wallis one-way ANOVA comparing the averaged SPPcount, Mean C, Weighted Mean C, FQAI, Total Cover, Detritus Cover, Water Depth, and Sediment Depth for the mound habitat (M) in 2016 vs 2017.

	Test statistic	P-value	df
SPP Count	8.626	0.003	1
Mean C	2.937	0.087	1
Weighted Mean C	0.340	0.560	1
FQAI	9.659	0.002	1
TotalCover	5.252	0.022	1
DetritusCover	6.270	0.012	1
WaterDepth	17.691	< 0.001	1
SedimentDepth	43.682	< 0.001	1

Literature Cited

Albert, D. A., Wilcox, D. A., Ingram, J. W., and T. A. Thompson. 2005. Hydrogeomorohic classification for Great Lakes coastal wetlands. Journal of Great Lakes Restoration 31:129–146.

Andreas, B. K., J. J. Mack, and J. S. McCormac. 2004. Floristic Quality Assessment Index (FQAI) for vascular plants and mosses for the State of Ohio. Page 219. Ohio Environmental Protection Agency, Division of Surface Water, Wetland Ecology Group, Columbus, Ohio.

ASA, 2013. Today's angler: A statistical profile of anglers, their targeted species and expenditures. Alexandria, VA, American Sportfishing Association.

Austin, J.C., S. Anderson, P. N. Courant, and R.E. Litan. 2007. America's North Coast: A Benefit–Cost Analysis of a Program to Protect and Restore the Great Lakes. Brookings Institute, Washington, DC.

Baedke, S. J., and T. A. Thompson. 2000. A 4,700-year record of lake level and isostasy for Lake Michigan. Journal of Great Lakes Research 26:416-426.

Ball, J.P. 1990. Influence of substantial flooding depth on cattail control by burning and mowing. Journal of Aquatic Plant Management 28: 32–36.

Bansal, S., S. C. Lishawa, S. Newman, B. A. Tangen, D. Wilcox, D. Albert, M. J. Anteau, M. J. Chimney, R. L. Cressey, E. DeKeyser, K. J. Elgersma, S. A. Finkelstein, J. Freeland, R. Grosshans, P. E. Klug, D. J. Larkin, B. A. Lawrence, G. Linz, J. Marburger, G. Noe, C. Otto, N. Reo, J. Richards, C. Richardson, L. R. Rodgers, A. J. Schrank, D. Svedarsky, S. Travis, N. Tuchman, and L. Windham-Myers. 2019.Typha (cattail) invasion in North American wetlands: biology, regional problems, impacts, ecosystem services, and management. Wetlands 39(4):645-684.

Crane, D. P., L. M. Miller, J. S. Diana, J. M. Casselman, J. M. Farrell, K. L. Kapuscinski, and J. K. Nohner. 2015. Muskellunge and Northern Pike Ecology and Management: Important Issues and Research Needs. Fisheries 40:258-267.

Dawe, N. K., G. E. Bradfield, W. S. Boyd, D. E. C. Trethewey, and A. N. Zolbrod. 2000. Marsh creation in a northern Pacific estuary: Is thirteen years of monitoring vegetation dynamics enough? Conservation Ecology 4:12.

Faber-Langendoen, D. 2018. Northeast Regional Floristic Quality Assessment Tools for Wetland Assessments. NatureServe, Arlington, VA.

Farrer, E., and D. Goldberg. 2009. Litter drives ecosystem and plant community changes in *Typha* invasion. Ecological Applications 19:398–412.

Freyman, W.A., L.A. Masters, and S. Packard. 2016. The Universal Floristic Quality Assessment (FQA) Calculator: an online tool for ecological assessment and monitoring. Methods in Ecology and Evolution 7:380–383.

Frieswyk, C. B., and J. B. Zedler. 2007. Vegetation change in Great Lakes coastal wetlands: Deviation from the historical cycle. Journal of Great Lakes Research 33:366–380.

Gann, G.D., & D. Lamb, eds. 2006. Ecological restoration: A mean of conserving biodiversity and sustaining livelihoods (version 1.1). Society for Ecological Restoration International, Tucson, Arizona, USA and IUCN, Gland, Switzerland.

GLAM (Great Lakes-St. Lawrence River Adaptive Management Committee). 2018. Summary of 2017 great lakes basin conditions and water level impacts to support ongoing regulation plan evaluation. GLAM Report for 2017. Burlington, Ontario, Canada.

Grabas, G., J. Ingram and N. Patterson. 2003. Great Lakes coastal wetland indicators and metrics: a summary of work in Canadian wetlands. Environmental Conservation Branch – Ontario Region, Canadian Wildlife Service.

Grubb, P. J. 1977. The maintenance of species-richness in plant communities: the importance of the regeneration niche. Biological Reviews 52:107-145.

Haapalehto, T., R. Juutinen, S. Kareksela, M. T. Kuitunen, T. Tahvanainen, H. Vuori, and J. S. Kotiaho. 2017. Recovery of plant communities after ecological restoration of forestry-drained peatlands. Ecology and Evolution 7: 7848– 7858.

Hara, T., J. van der Toorn, and J. H. Mook. 1993. Growth dynamics and size structure of shoots of Phragmites australis, a clonal plant. Journal of Ecology 81:47-60.

Hobbs, R. J., and L. F. Huenneke. 1992. Disturbance, Diversity, and Invasion: Implications for Conservation. Conservation Biology 6(3):324–337

Holvick, S. M., D. E. Bunker, C. J Peterson, and W. P. Carson. 2011. Purple loosestrife suppresses plant species colonization far more than broad-leaved cattail: experimental evidence with plant community implications. Journal of Ecology 99:225-234.

Huston, M. 1979. A general hypothesis of species diversity. The American Naturalist 113:81-101.

IJC. 2017. Plan 2014 – Home. [Online] International Joint Commission. Available at: http://www.ijc.org/en_/Plan2014/home

Johnston, J. W., S. J. Baedke, R. K. Booth, T. A. Thompson, and D. A. Wilcox. 2004. Late Holocene lake-level variation in southeastern Lake Superior: Tahquamenon Bay, Michigan. Journal of Great Lakes Research 30:1-19.

Jude, D.J., and J. Pappas 1992. Fish utilization of Great Lakes coastal wetlands. Journal of Great Lakes Research 18:651-672.

Keddy, P. A., and A. A. Reznicek. 1986. Great Lakes vegetation dynamics: the role of fluctuating water levels and buried seeds. Journal of Great Lakes Research 12:25-36.

Kozlowski, T. T. 1984. Responses of woody plants to flooding. *In* T.T. Kozlowski (Editor) Flooding and plant growth. Pages 129-163. Academic Press, New York, New York.

Krieger, K.A. 1992. The ecology of invertebrates in Great Lakes coastal wetlands: current knowledge and research needs. Journal of Great Lakes *Research* 18:634-650.

Kutcher, T. E. and G. E. Forrester. 2018. Evaluating how variants of floristic quality assessment indicate wetland condition. Journal of Environmental Management 217:231–239.

LaPan, S. J. 2015. Bird, reptile, and amphibian response to habitat enhancement of St. Lawrence River coastal wetlands. Ph.D. Dissertation. State University of New York College of Environmental Science & Forestry, Syracuse, New York, USA.

Larsen, D. P., P. R. Kaufmann, T. M. Kincaid, and N. S. Urguhart. 2003. Detecting persistent change in the habitat of salmon-bearing streams in the Pacific Northwest. Canadian Journal of Fisheries and Aquatic Sciences 61:283– 291.

Lawrence, B. A., S. C. Lishawa, Y. Rodriguez, and N. C. Tuchman. 2016. Herbicide management of invasive cattail (*Typha x glauca*) increases porewater nutrient concentrations. Wetlands Ecology and Management 24:457–467

Matthews, J. W. 2003. Assessment of the Floristic Quality Index for use in Illinois, USA, wetlands. Natural Areas Journal 23:53–60.

Matthews, J. W., G. Spyreas, and C. M. Long. 2015. A null model test of Floristic Quality Assessment: Are plant species' Coefficients of Conservatism valid? Ecological Indicators 52:1–7.

Methe, B., R. Soracco, J. Madsen, and C. Boylen. 1993. Seed production and growth of water chestnut as influenced by cutting. Journal of Aquatic Plant Management 31:154-157.

Miller, S.J. and D.H. Wardrop. 2006. Adapting the floristic quality assessment index to indicate anthropogenic disturbance in central Pennsylvania wetlands. Ecological Indicators 6:313-326.

Mingelbier, M., P. Brodeur, and J. Morin. 2008. Spatially explicit model predicting the spawning habitat and early stage mortality of Northern pike (*Esox lucius*) in a large system: the St. Lawrence River between 1960 and 2000. Hydrobiologia 601:55-69.

NEIWPCC. 2011. Northeast regional floristic quality assessment. [Online.] New England Interstate Water Pollution Control Commission. Available at: http://neiwpcc.org/our-programs/wetlands-aquatic-species/nebawwg/nqa/. [Accessed 15 December 2018]

NRC, National Research Council. 2006. *Review of the Lake Ontario-St. Lawrence River Studies*. Washington, DC: The National Academies Press. https://doi.org/10.17226/11481.

NOAA (National Oceanic and Atmospheric Administration). 2004. Monitoring requirements under the Estuary Restoration Act. Available at http://era.noaa.gov/htmls/era/era_monitoring.html

NOAA (National Oceanic and Atmospheric Administration). 2016. State of the Climate: Drought for May 2016. Available at https://www.ncdc.noaa.gov/sotc/drought/201605

NOAA (National Oceanic and Atmospheric Administration). 2017. State of the Climate: Drought for May 2017. Available at https://www.ncdc.noaa.gov/sotc/drought/201705

NYSDEC 2014. NYCRR Part 575 Invasive Species Regulations, New York State Prohibited and Regulated Invasive Plants. [Online.] United State Department of Environmental Conservation, Buffalo, New York. Available at: https://www.dec.ny.gov/docs/lands_forests_pdf/isprohibitedplants2.pdf

NYSDEC 2016A. Rochester Embayment Area of Concern. [Online.] United States Department of Environmental Conservation, Buffalo, New York. Available at: http://www.dec.ny.gov/lands/92771.html

NYSDEC 2016B. Braddock Bay Wildlife Management Area. [Online.] United States Department of Environmental Conservation, Buffalo, New York. Available at: http://www.dec.ny.gov/outdoor/24428.html

Reznicek, A. A., M. R. Penskar, B. S. Walters, and B. S. Slaughter. 2014. Michigan Floristic Quality Assessment Database. Herbarium, University of Michigan, Ann Arbor, Michigan and Michigan Natural Features Inventory, Michigan State University, Lansing, Michigan. Available at: http://michiganflora.net/home.aspx

Roegner, G.C., H.L. Diefenderfer, A.B. Borde, R.M. Thom, E.M. Dawley, A.H. Whiting, S.A. Zimmerman, and G.E. Johnson. 2009. Protocols for monitoring habitat restoration projects in the lower Columbia River and estuary. U.S. Department of Commerce, NOAA Technical Memo. NMFS-NWFSC-97, 63 pages.

Stankey, George H.; Clark, Roger N.; Bormann, Bernard T. 2005. Adaptive management of natural resources: theory, concepts, and management institutions. General Technical Report PNW-GTR-654. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 73 pages.

Strayer, D., C. Lutz, H. Malcom, K. Munger, and W. Shaw. 2003. Invertebrate communities associated with a native (*Vallisneria americana*) and an alien (*Trapa natans*) macrophyte in a large river. Freshwater Biology 48:1938-1949.

Swink, F., and G. Wilhelm. 1994. Plants of the Chicago Region. 4th edition. Indiana Academy of Science, Indianapolis, Indiana.

Taft, J. B., G. S. Wilhelm, D. M. Ladd, and L. A. Masters. 1997. Floristic quality assessment for vegetation in Illinois, a method for assessing vegetation integrity. Erigenia 15:3–95.

Thompson, T. A., and S. J. Beadke. 1997. Strandplain evidence for late Holocene lake-level variation in Lake Michigan. Geological Society of America Bulletin 109:666–682.

USACE. 2016A. Braddock Bay, version 1. [Online.] United States Army Corps of Engineers, Buffalo, New York. Available at: http://www.lrb.usace.army.mil/Missions/Civil-Works/District-Projects/Braddock-Bay/

USACE. 2016B. Braddock Bay Post-Restoration Monitoring. Feasibility study. [Online] United States Army Corps of Engineers, Buffalo, New York, USA.

USACE. 2017. 2016 Braddock Bay Monitoring and Adaptive Management Report. [Online] United States Army Corps of Engineers, Buffalo, New York, USA.

USEPA. 2016. About Rochester Embayment AOC. [Online.] United States Environmental Protection Agency, Chicago, Illinois. Available at: https://www.epa.gov/rochester-embayment aoc/about-rochester-embayment-aoc

USEPA. 2018. Braddock Bay Wetland and Barrier Beach Restoration. [Online.] United States
Environmental Protection Agency, Chicago, Illinois. Available at: https://www.epa.gov/great-lakes-aocs/braddock-bay-wetland-and-barrier-beach-restoration

USEPA. 2018. Why Are Wetlands Important? [Online.] United States Environmental Protection Agency, Chicago, Illinois. Available at: https://www.epa.gov/wetlands/why-are-wetlands-important

USFWS. 2019. Floristic Quality Assessment. [Online.] United States Fish and Wildlife Service, Bloomington Minnesota. Available at: https://www.fws.gov/midwest/endangered/section7/s7process/plants/FQA.html

Vaccaro, L. E., B. L. Bedford, and C. A. Johnston. 2009. Litter accumulation promotes dominance of invasive species of *Typha* (*Typha* spp.) in Lake Ontario wetlands. Wetlands 29:1036–1048.

Wilcox, D. A. 1993. Effects of water-level regulation on wetlands of the Great Lakes. Great Lakes Wetlands 4:1-2.

Wilcox, D. A. 2004. Implications of hydrologic variability on the succession of plants in Great Lakes wetlands. Aquatic Ecosystem Health and Management 7:223-231.

Wilcox, D. A. and J. A. Bateman. 2018. Photointerpretation analysis of plant communities in Lake Ontario wetlands following 65 years of lake-level regulation. Journal of Great Lakes Research 44(6):1306-1313.

Wilcox, D. A., K. Buckler, and A. Czayka. 2017. Controlling cattail invasion in sedge/grass meadows. Wetlands 38:337-347.

Wilcox, D. A., J. W. Ingram, K. P. Kowalski, J. E. Meeker, M. L. Carlson, Y. Xie, G.P. Grabas, K. L. Holmes, and N. J. Patterson. 2005. Evaluation of water-level regulation influences on Lake Ontario and Upper St. Lawrence River coastal wetland plant communities. Report to the International Joint Commission. Ann Arbor, Michigan, USA.

Wilcox, D. A., J. E. Meeker, P. L. Hudson, B. J. Armitage, M. G. Black and D. G. Uzarski. 2002. Hydrologic variability and the application of index of biotic intebrity metrics to wetlands: A Great Lakes evaluation. Wetlands 22(3): 588-615.

Wilcox, D. A., K. P. Kowalski, H. L. Hoare, M. L. Carlson, and H. N. Morgan. 2008. Cattail invasion of sedge/grass meadows in Lake Ontario: photointerpretation analysis of sixteen wetlands over five decades. Journal of Great Lakes Research 34:301-323.

Wilcox, D. A., and G. Ray. 1989. Using living mat transplants to restore a salt impacted bog (Indiana). Restoration and Management Notes 7(1): 39

Wilcox, D. A., T. A. Thompson, R. K. Booth, and J. R. Nicholas. 2007. Lake-level variability and water availability in the Great Lakes: U.S. Geological Survey Circular 1311.

Wilcox, D. A., and Y. Xie. 2007. Predicting wetland plant community responses to proposed water-level regulation plans for Lake Ontario: GIS-based modeling. Journal of Great Lakes Research 33:751-773.

Wilcox, D. A. and Y. Xie. 2008. Predicted effects of proposed new regulation plans on sedge/grass meadows of Lake Ontario. Journal of Great Lakes Restoration 34:745-754.

Wooten, J. W. 1986. Variation in leaf characteristics of six species of Sagittaria (Alismataceae) caused by various water levels. Aquatic Botany 23:321-327.

Wyman, K. E., and F. J. Cuthbert. 2017. Black tern (Chlidonias niger) breeding site abandonment in U.S. Great Lakes coastal wetlands is predicted by historical abundance and patterns of emergent vegetation. Wetlands Ecology and Management 25:583-596.

Zedler, J. B. 1988. Salt marsh restoration: Lessons from California. In J. Caines, (ed.), Rehabilitating damaged ecosystems, Vol. 1, p. 123–138. CRC Press, Boca Raton, Florida, USA.

Zhu, B., C. C. Ottaviani, R. Naddafi, Z. Dai, and D. Du. 2018. Invasive European frogbit (*Hydrocharis morsus-ranae* L.) in North America: an updated review 2003-16. Journal of Plant Ecology 11:17-25.