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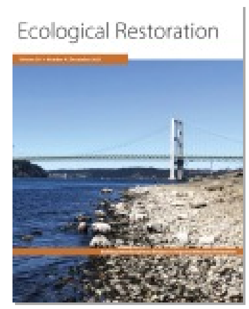
Wetland Restoration in *Typha* -Dominated Braddock Bay of Lake Ontario



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RESEARCH ARTICLE

Wetland Restoration in *Typha*-Dominated Braddock Bay of Lake Ontario

Alexander O. Silva, Douglas A. Wilcox and Eli L. Polzer

ABSTRACT

The barrier beach at the Braddock Bay wetland on Lake Ontario was lost to erosion. Without the protective barrier, the area of vegetated wetland was reduced by wave attack. Lake-level regulation implemented in 1960 resulted in cattail (primarily hybrid *Typha* × *glauca*), invasion and loss of sedge-grass meadow habitat. As part of the Rochester Embayment Great Lakes Area of Concern, Braddock Bay was targeted for restoration by the U.S. Army Corps of Engineers. The plan included reducing coverage by cattails, construction of channels and potholes to improve wildlife access to the wetland, creation of spoil mounds along the channels to discourage growth of cattail while supporting the growth of sedge-grass meadow species, re-creation of the barrier beach, and construction of new emergent marsh. We collected vegetation data for three years following the 2016 construction activities. Post-restoration results showed that cattail cover decreased greatly across years in the cattail treatment areas, decreased in lower elevation constructed habitats, and gradually increased in higher elevation habitats. Opening of the canopy resulted in increased floating and submersed species, and constructed mounds hosted wet meadow species. Site-level weighted mean C is recommended for future data analyses, rather than FQAI or mean C, because it has no observed influence from species richness. Restoration results were affected by high lake levels in 2017, identified problems in seeding and planting, and meeting construction plans for some channel and pothole depths and mound elevations. Pre-restoration soil surveys are recommended to reduce construction problems, and adaptive management should include invasive species control.

Keywords: cattails, channel excavation, pothole excavation, spoil mounds, *Typha* control

Restoration Recap

- Braddock Bay on the south shore of Lake Ontario was impacted by lake-level regulation and erosion, resulting in loss of wetland and invasion by cattails.
- Channels and potholes were excavated, habitat mounds created, and dense cattails treated by cutting when storage carbohydrates in rhizomes were reduced, with herbicide treatment of regrowth.
- Three growing seasons after implementation, treated cattails were greatly reduced, opened channels and potholes supported floating and submersed vegetation, and mounds contained wet meadow species. Weighted mean C was better than FQAI or mean C in tracking native wetland character.
- Problems identified included failure to meet planned depths/widths in some channels and potholes and planned elevations of some mounds, as well as failure to plant many trays of wetland plants.
- Suggestions for future efforts include soil surveys to map organic versus mineral soils to alleviate elevation problems due to settling after excavation and adaptive management to control invasive plants.

Climate-driven, quasi-periodic fluctuations in lake levels (Baedke and Thompson 2000, Wilcox et al. 2007) maintain the diversity of wetland plant communities in the Laurentian Great Lakes (Keddy and Reznicek 1986, Wilcox 2004). High lake levels eliminate upland invaders and canopy-dominating emergent plants, and less competitive understory plants grow from seeds or propagules during low lake levels (Keddy and Reznicek 1986, Wilcox 2004). Sedges and grasses hold a competitive advantage over more robust plants such as cattails at higher elevations in the wetlands because they can tolerate low lake-level periods when soil moisture is low (Wilcox et al. 2008).

Supplementary materials are freely available online at: http://uwpress.wisc.edu/journals/journals/er-supplementary.html

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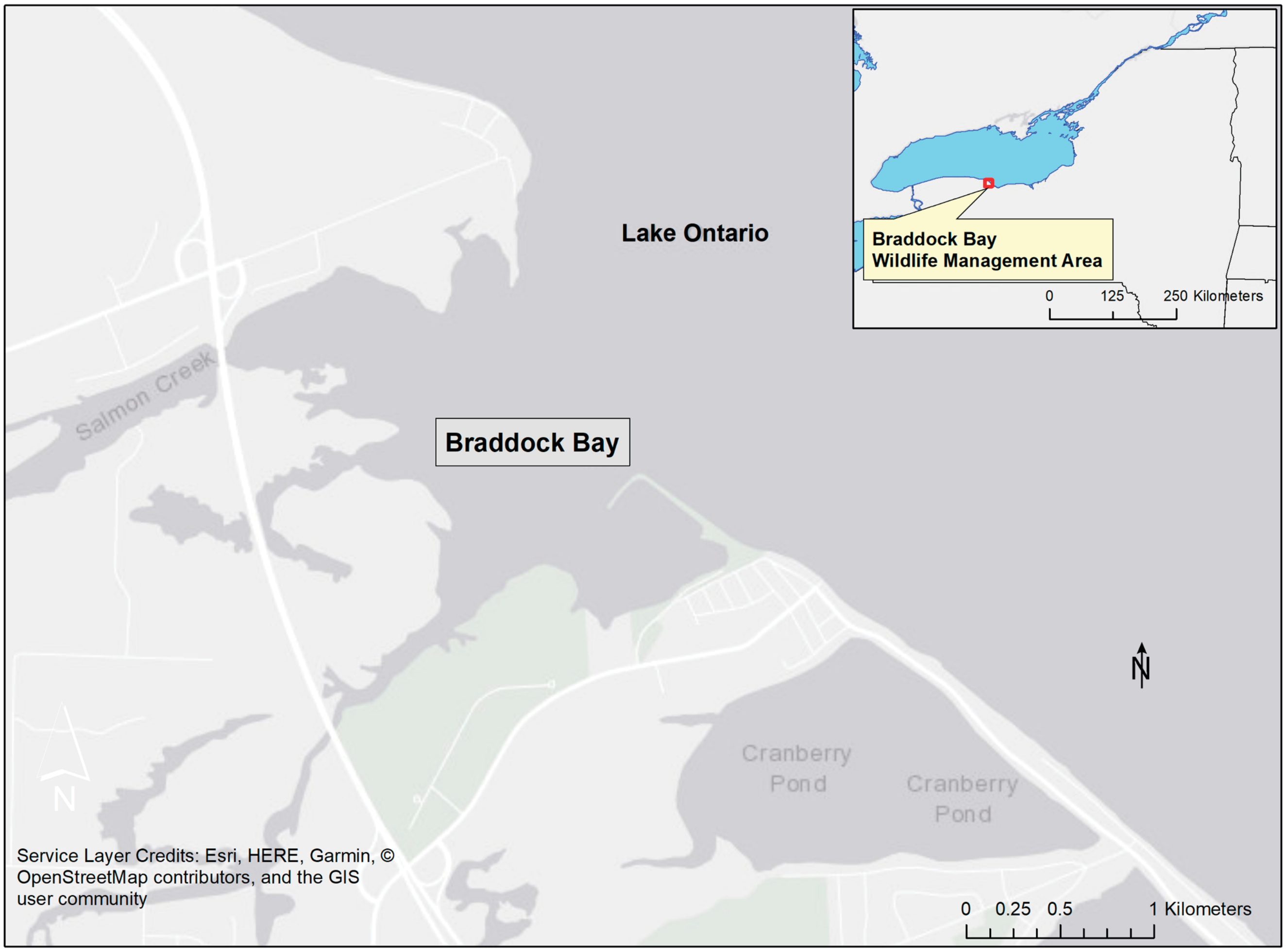

Figure 1. Map of pre-restoration Braddock Bay along the south shore of Lake Ontario.

Regulation of Lake Ontario water levels under Plan 1958DD that began with operation of the St. Lawrence Seaway in 1960 made large reductions in the range of fluctuations from approximately 1.5 m to 0.7 m and eliminated years with low lake levels (Wilcox and Xie 2007). Sedge-grass meadow habitat along the shores then decreased substantially, as sedges and grasses at higher elevations lost their competitive advantage and were replaced by cattails that were never subjected to drier soil conditions. State and federal agencies thus placed an emphasis on restoring this habitat type, and funding through the federal Great Lakes Restoration Initiative brought about projects such as at Braddock Bay.

Study Area—The Braddock Bay Restoration Project

Braddock Bay is an 860-ha coastal embayment (Albert et al. 2005) on the southern shore of Lake Ontario (Figure 1) in the town of Greece, NY and is part of the Rochester Embayment Great Lakes Area of Concern (AoC) (NYSDEC 2016a, USEPA 2016). Braddock Bay has been managed by New York State Department of Environmental Conservation—Region 8 (NYSDEC) since 1982 and has an ecological and recreational importance to the surrounding Rochester area (NYSDEC 2016b). Historically, the embayment was protected from wave attack by a barrier beach, which was slowly eroded due to wave action from Lake Ontario and loss of sand from littoral drift resulting from shoreline armoring (USACE 2016). Without the protective barrier, approximately 43 ha of emergent wetlands was eroded, decreasing the remaining wetland in the embayment to about 138 ha (USACE 2016).

The Braddock Bay marsh consists primarily of dense, buoyant cattail mats with scattered emergent species that provide limited functional habitat and reduced benefits for wetland-dependent fish and wildlife. Reduced structural complexity alters fish spawning and nursery habitat and affects macroinvertebrate communities important as food for many fish and wildlife species. Loss of forage-producing plants also affects many wildlife species (Bansal et al. 2019). 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 and leading to AoC-listing as a species of concern. Cattail expansion into sedge-grass meadow also reduced the potential spawning and nursery grounds for northern pike (Mingelbier et al. 2008), which was also listed as an AoC species of concern. Remnant sedge-grass meadow marsh occurs only in depauperate patches along the shoreline, where lower soil moisture reduces the rate of cattail encroachment (Wilcox et al. 2008). In addition to cattails (primarily hybrid *Typha* × *glauca*), invasive species of concern in the bay include *Lythrum salicaria* (purple loosestrife), *Phragmites australis* (common reed), and *Trapa natans* (water chestnut).

Without taking action in Braddock Bay, the diversity and quality of plant habitat in the bay would remain low, and further erosion and loss of wetland area would occur. Restoration activities were thus conducted by the United States Army Corps of Engineers (Corps) beginning in 2016, with data collection taking place in 2016–2018 funded by the Great Lakes Restoration Initiative (USACE 2016) and NYSDEC.

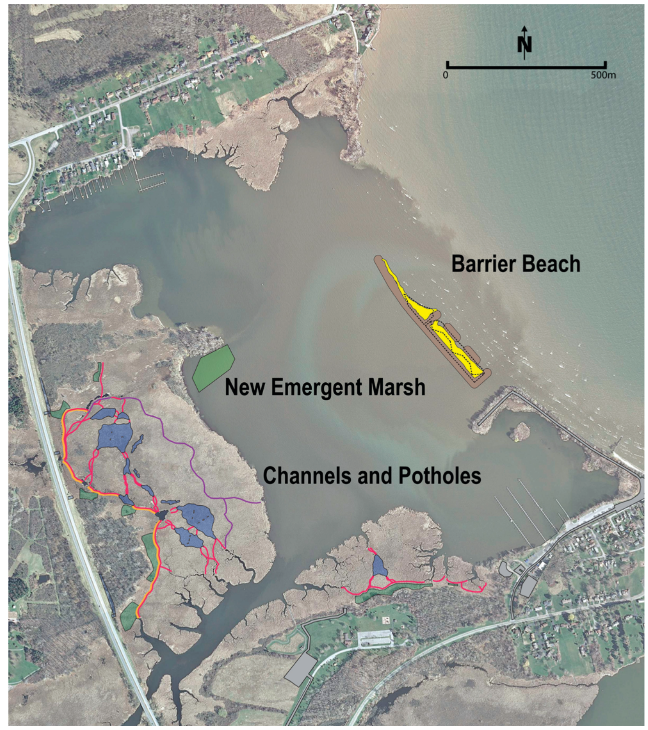

Figure 2. U.S. Army Corps of Engineers plan for the Braddock Bay Restoration Project showing proposed restoration implementations, with created Barrier Beach in yellow near the lake and New Emergent Marsh in green near the wetland margin, as well as locations of Channels (orange and red) and Potholes (blue) where the study was conducted (USACE 2016).

Corps Restoration Project Goals and Implementation

The overarching goal of the Corps was to improve habitat diversity of the existing emergent marsh and reduce erosion of the existing emergent marsh, with ultimate hopes to delist the Rochester Embayment AoC by addressing the "Loss of Fish and Wildlife Habitat Beneficial Use Impairment" (USACE 2016). The overall objectives of the restoration included protecting wetlands from further erosion and improving habitat suitability for fish and wildlife species of concern (northern pike–spawning and nursery; Black Tern–nesting) (USACE 2017). Contaminant load in American mink was also listed as an impairment in the AoC but was not part of this project, as AoC-related work was completed in 2005 (J. Haynes, pers. comm). To achieve these objectives, the Corps, with planning assistance from SUNY Brockport and NYSDEC8, scoped a multi-measure restoration. The plan included 1) re-creation of the barrier beach to protect the remaining wetland from erosion, 2) excavation of nearshore channels and shallow potholes within the extensive cattail mat to increase quantity and quality of fish and wildlife habitat by replacing the function of natural features that had been lost to cattail invasion, 3) placement of excavated spoil material to create elevated mounds adjacent to the channels at an elevation that would support sedge-grass meadow and hinder cattail growth, 4) cutting and herbicide treatment of cattail, and 5) creation of a new area of emergent marsh.

The new emergent marsh was created on dredge spoil placed behind a rock barrier (Figure 2). To re-create the historical barrier beach, a 0.51-km-long, continuous rubblemound breakwater spine was constructed and overlain with sand, with two 42-m-long headland rubblemound breakwaters and two 55-m-long rubblemound terminal groins attached (Figure 2) (USACE 2016, USEPA 2018). Approximately 3.5 km of access channels and 2.7 ha of potholes (Figure 2) were created with long-arm excavators operating from platforms placed on frozen cattail mat. The channels were approximately 1-m deep and 4.2-m wide in a trapezoidal shape with a shallow bench along the edges to encourage emergent vegetation. Twelve potholes of varying size up to 0.4 ha were excavated to approximately 1-m depth along the channels, also with a shallow bench. Spoil from channel excavation was placed in mounds on the landward side of the channel adjacent to existing sedge-grass meadow with a target elevation of 75.6 m International Great Lakes Datum 1985 (IGLD1985) to encourage sedge-grass colonization without cattail invasion (Wilcox et al. 2008). Spoil from the potholes was side-cast as mounds on adjacent areas to create micro-topography for emergent plants. Cattail-treatment measures were performed on approximately 3.6 ha parallel to the shoreline and adjacent to the sedge-grass meadow. Treatment included removal of past-years' growth by mowing in winter, cutting cattails with steel-bladed trimmers when storage carbohydrates in rhizomes were minimized (Sojda and Solberg 1993), and subsequent herbicide treatment of new stems by hand-wicking with Glyphosate (Rodeo, Dow Agrosciences, Indianapolis, IN) (Wilcox et al. 2018).

The project began with channel and pothole excavation and spoil mound placement in January 2016, concluding in March, when the spoil mounds, pothole benches, and channel benches were seeded at 9.1 kg/ha with regionally-sourced seed (later shown in Table 4). In June and July, about 25,000 vegetative plugs were planted in the treated emergent marsh, and 38,000 were planted within the sedge-grass meadow. Cattail was cut in July, with corresponding herbicide treatment of regrowth in September. In August, the barrier beach stone placement was completed, with no further work done until 2018 due to high lake levels (completed in August 2018). A second season of cattail cutting occurred in August 2018. Creation of new emergent marsh was completed in September 2018 and was not part of this study. Fish and wildlife assessment is ongoing by others (USACE 2017).

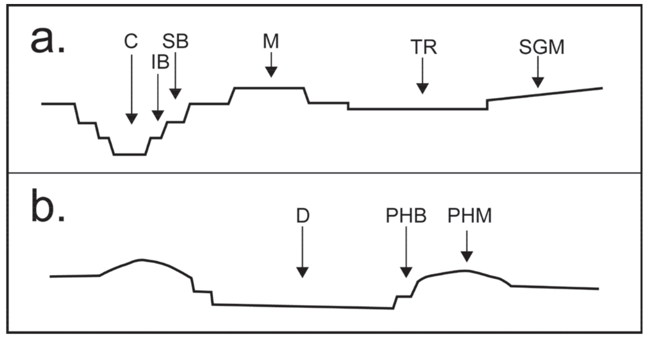

Figure 3. Illustrations (not to scale) showing habitat zones sampled for a) channels, including Sedge/Grass Meadow (SGM), Treated area for cattail (TR), Mound (M), Intermediate Bench (IB), Shallow Bench (SB), and Channel (C); b) potholes, including Deep water (D), Bench (PHB), and Mound (PHM).

Goals of This Study

Our goals were to perform vegetative data collection and evaluate assessment tools to identify where intended plant-community restoration goals were achieved and to provide information for use in adaptive management. We evaluated the establishment of different created habitat zones that aim to support different communities across the restoration site, with expectations of an increase in species richness and nativeness. We hypothesized that the different zones would accordingly support different communities. We used the Floristic Quality Assessment Index (FQAI) (Faber-Langendoen 2018) to determine potential success by comparing floristic quality between habitats across years. We also tested mean coefficient of conservatism (C) and cover-weighted mean C scores to determine if they provided better assessments. A further objective was to use assessment results to evaluate restoration implementation successes and failures and to suggest adaptive management needs.

Methods

Pre-Restoration Plant Sampling

As part of the feasibility study conducted prior to restoration, we sampled plant communities from June to July 2013. With concurrent ground-truthing, we used aerial photographs from 2012 to delineate wetland plant community boundaries by photointerpretation. Stratified sampling by plant community type was conducted in 1 m × 1 m quadrats placed haphazardly by blind toss over the shoulder. We estimated percent cover for each taxon, generally to species level, by single digits to ten and then by increments of five. Although four community types (successional shrubland, sedge-grass meadow, shallow emergent marsh, deep emergent marsh) were sampled, we report on only the 40 quadrats sampled in approximately 7.5 ha of the shallow emergent marsh (SEM), which represents the area in which cattail-control measures were implemented.

Invasive Plant Surveys

In late May and early June 2016 following restoration, we conducted walking surveys that traversed the breadth of all cattail-treated, channel, and pothole restored areas to obtain presence and abundance data for invasive purple loosestrife, common reed, and water chestnut that were growing in those areas. We identified locations of all stands encountered with a GPS, and the radii of the invasive stands were estimated. We estimated stem density within a 3-m control radius extending from the center of the stand so that stands could be classified into five categories with stem counts ranging up to 10, 20, 30, 40, and 50. Sampling was repeated in 2017 and 2018.

Post-Restoration Plant Sampling

We conducted our primary post-restoration vegetation surveys in 2016, 2017, and 2018. Fourteen transects were used to sample the network of newly created channels and associated mounds and cattail-treatment areas. Transects spanned six vegetation zones: sedge-grass meadow (SGM), treated (TR), mound (M), shallow bench between the mound and channel (SB), intermediate bench within the channel (IB), and channel (C) (Figure 3a). The 12 potholes and associated mounds were sampled with 16 transects, spanning three identified zones: deep water (D), bench (PHB), and mound (PHM) (Figure 3b). Transects ran from the deepest portion of the potholes to the base of the back slope of the spoil mounds. Sampling of untreated cattail mat as a control was conducted in quadrats placed haphazardly.

We sampled from late June to late August in all years. Each C-IB-SB-M-TR-SGM transect aimed to have sixteen

1-m² quadrats (C = 2, IB = 2, SB = 2, M = 4, TR = 4, SGM = 2); however, some areas could only fit fewer quadrats without overlap, so fewer quadrats were sampled. Along all transects, quadrats were again placed haphazardly within each of the different zones, and percent cover for each taxon, generally to species level, was again estimated. We also recorded water depths for each quadrat. In the potholes, we surveyed vegetation using eight 1-m² quadrats per transect (D = 2, PHB = 2, PHM = 4). Quadrats were again placed haphazardly along the transects within each zone and sampled similarly. Thirty control quadrats were sampled throughout the cattail mat area in the bay.

Data Analyses

Initial data analyses included calculation of mean water depths from quadrat sampling by habitat type, as well as calculation and comparison of percent cover for species encountered. We analyzed Floristic Quality Assessment Index (FQAI) to evaluate the nativeness of the plant community based on plant species present and their Coefficient of Conservatism scores (C) (Reznicek et al. 2014, Faber-Langendoen 2018). We made FQAI comparisons across transects and habitat zones both within first year sampling and across the three-year period. Similar comparisons were made between the pre-restoration sampling year 2013 and post-restoration sampling year 2018.

FQAI was calculated by using the New York State preliminary C-score list with reference to the NEIWPCC Northeast Ecoregional C-score list (NEIWPCC 2011, Reznicek et al. 2014, Faber-Langendoen 2018), calculated as follows:

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

where C_t represents the total mean C and N_t is the total species richness (Freyman et al. 2016, Faber-Langendoen 2018). The total mean C value represents the mean C metric for each quadrat within a habitat zone.

We then averaged these scores to yield a mean FQAI and mean C-score for each transect, which were grouped into zones. All of these data were averaged to determine mean FQAI and mean C-scores for all of the channel transects and all of the pothole transects, respectively. Quadrat-level, cover-weighted mean C was calculated as follows:

Equation 2: Quadrat level cover weighted mean $C = \sum_{i=0}^{t} C_i \gamma_i / \sum_{i=0}^{t} \gamma_i$

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). We made comparisons of species abundance, FQAI, mean C, and weighted mean C across the different grouped zones and overall year to year.

We used the Kolmogorov-Smirnov normality test to determine whether the sample data had been drawn from a normally distributed population, with significant p-value = 0.05. After transformation, many of the data did not meet the assumption for a normally distributed population; therefore, non-parametric statistics were used for the analyses. With SPSS statistics software, after averaging together each habitat zone within each transect, we used a multivariate General Linear Model (GLM) to compare the different habitats within each individual sampling year using FQAI, mean C, and weighted mean C as factors blocked by transect. Further analyses included non-parametric independent-sample Kruskal-Wallis one-way ANOVAs to compare variables within the different habitats across the three sampling years. We then ran year-to-year comparisons for the FQAI, mean C, and weighted mean C variables within each individual habitat type. We created a multivariate GLM comparing dependent variables with the different habitat zones as a fixed factor and a transect block. Specifically, this compared 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.

Using pre-restoration data, we calculated FQAI, mean C, weighted mean C, and species richness statistics for the 2013 shallow emergent marsh (SEM) pre-restoration data (the portion of Braddock Bay where the cattail treatment later occurred) to test the effect of cattail treatment. Half of these data compared between 2013 and 2018 did not follow normal distribution; therefore, we used a non-parametric independent-samples Kruskal-Wallis one-way ANOVA for this analysis.

Results

Invasive Plant Surveys

On walking surveys designed to search for invasive purple loosestrife, common reed, and water chestnut, we found only purple loosestrife in the restored areas, although all three species were present in Braddock Bay before the restoration. In 2016, we sampled a total of 336 points of purple loosestrife, and 24.7% of them had greater than the maximum cut-off of 50+ stems (category 5). In 2017, 52.0% of the total 546 points were category 5; in 2018, 8.1% of 422 points were category 5.

Water Depths from Post-Restoration Plant Sampling

Maximum water depth along transects in 2016 (a year with normal precipitation patterns and average Lake Ontario water levels) was 120 cm, 230 cm in 2017 (a wet year with record lake levels), and down to about 145 cm in drier 2018 with lower lake levels. Mean water depth in 2017 (60 cm) was significantly greater than in 2016 (23 cm; p < 0.001) but then decreased significantly in 2018 (30 cm;

$p < 0.001$). Maximum depth to the hard sediment within the deep zone of the potholes was about 130 cm in 2016, 235 cm in 2017, and 265 cm in 2018.

Floristics from Pre- and Post-Restoration Plant Sampling

We observed 111 taxa across all years sampled (Appendix Table 1). Dominant species are designated by a cut-off of 10% cover within each habitat. The only species dominant within the 2013 pre-restoration sampling of the shallow emergent marsh were *T.* × *glauca* (65.9%) and *Hydrocharis morsus-ranae* (common frogbit) (10.6%). The only dominant species within the cattail mat control (CT) was *T.* × *glauca* at 51.7% cover (2016), 48.0% (2017), and 53.7% (2018) (Tables 1, 2, 3).

In post-restoration, the C habitat in the channels was dominated by *Utricularia vulgaris* (common bladderwort) and *H. morsus-ranae* in 2016; *U. vulgaris* and *Ceratophyllum demersum* (coontail) in 2017; and *Spirodela polyrhiza* (common duckmeat), *C. demersum, Elodea canadensis* (Canadian waterweed), and *H. morsus-ranae* in 2018 (Tables 1, 2, 3). The IB habitat was dominated by *H. morsus-ranae*, *Lemna minor* (common duckweed), *E. canadensis*, and *U. vulgaris* in 2016; *U. vulgaris, H. morsus-ranae*, and *Stuckenia pectinata* (sago pondweed) in 2017; and *T.* × *glauca* (53.9% cover) in 2018—an increase from 1.2% and 4.2% in 2016 and 2017. The SB habitat was dominated by *T.* × *glauca* (48.9 % cover) in 2016 and 2017 (37.7%). *Hydrocharis morsus-ranae* was also dominant in 2017 and 2018 with additional 2018 dominants *S. polyrhiza, C. demersum*, and *E. canadensis*, while *T.* × *glauca* was reduced to 1.1% cover.

On the channel mounds (M), dominant species in 2016 were *Persicaria hydropiper* (marshpepper knotweed), *L. salicaria*, *Persicaria lapathifolia* (curlytop knotweed), and *T.* × *glauca* (11.6% cover), which increased to 13.4% in 2017 (Tables 1, 2). Also dominant in 2017 were *P. hydropiper* and *H. morsus-ranae*. In 2018, *T.* × *glauca* increased to 21.3% cover; other dominants were *Rorippa palustris* (bog yellowcress), *Scutellaria galericulata* (marsh skullcap), and *Verbena hastata* (swamp verbena) (Table 3). In 2016, the only dominant in the cattail-treatment area (TR) was *T.* × *glauca*, with 22.3% cover, which was reduced to 3.3% in 2017 and increased to 9.5% in 2018. Other dominants in 2017 were *H. morsus-ranae* and *U. vulgaris*; there were no other dominants in 2018.

In the sedge-grass meadow habitat (SGM), *Carex lacustris* (hairy sedge) was dominant in 2016, along with *Acer saccharinum* (silver maple), *Salix fragilis* (crack willow), and *T.* × *glauca* (10.0% cover) (Table 1). In 2017, *T.* × *glauca* was reduced to 8.8% but increased to 15.8% in 2018 (Tables 2, 3). Other 2017 dominants were *C. lacustris*, *Calamagrostis canadensis* (bluejoint), *H. morsus-ranae*, and *S. fragilis*. Also dominant in 2018 was *C. canadensis*.

In the potholes, *U. vulgaris* was the only dominant species in the D habitat in 2016 and 2017 but was joined by *C. demersum*, *Potamogeton foliosus* (leafy pondweed), and *S. polyrhiza* in 2018 (Tables 1, 2, 3). In the PHB habitat, *H. morsus-ranae* was the only dominant in 2016, with *T.* × *glauca* at 7.4% cover. *Hydrocharis morsus-ranae* was joined by *U. vulgaris*, *L. salicaria,* and *T.* × *glauca* (26.4% cover) as dominants in 2017. In 2018, *T.* × *glauca* was reduced to 11.4% cover, and other dominants were *S. polyrhiza, H. morsus-ranae, U. vulgaris,* and *C. demersum.*

On the pothole mounds (PHM), *L. salicaria* and *T.* × *glauca* (16.7% cover) were dominant in 2016 (Table 1). *Typha* × *glauca* cover was14.2% in 2017 and 15.0% in 2018. Other dominants in 2017 were *L. salicaria*, *P. hydropiper, Decodon verticillatus* (swamp loosestrife), and *I. capensis* (Table 2). Dominants in 2018 also included *Boehmeria cylindri*ca (smallspike false nettle), *D. verticillatus*, and *I. capensis* (Table 3).

Planted and Seeded Species

In 2016, we found 100 different plant taxa, 86 of which were identified to species level. Only 13 of the 30 seeded/planted species appeared in 2016 sampling (Table 4). In 2017, 86 of 94 different plants were identified to species, 15 of which were seeded/planted. In 2018, 85 of 91 different plants were identified to species with 17 observed seeded/planted species. The majority of species seeded and planted were also present within the wetland before treatment, and we cannot confirm if seeding or planting had an effect on establishment in the restored areas.

Floristic Quality of Plant Communities

The overall FQAI statistic across all transects sampled showed a significant increase in floristic quality from year 2016 to 2017 and a slight decrease in 2018 (2016 = 6.31, 2017 = 6.89, 2018 = 6.70 GLM $p = 0.007$). The overall mean C statistics calculated for 2016, 2017, and 2018 were not significantly different. The overall weighted mean C statistic was not significantly different across years.

We also looked at individual years and the potential for the habitat zones to be different within each year. The GLM showed no significant interaction between habitat zones blocked by transects for any metric in all years separately. FQAI was not significant across habitat types in 2016 but was significant in 2017 ($p = 0.044$) and 2018 $p < 0.001$). Mean C was not significant across habitat types in 2016 or 2017 but was significant in 2018 ($p < 0.001$). In 2016, 2017, and 2018, weighted mean C was significantly different across the habitat types (2016 $p < 0.001$, 2017 $p = 0.001$, 2018 $p < 0.001$).

FQAI differed significantly across the three years of sampling of the channel transects (2016 = 5.65, 2017 = 7.30, 2018 = 6.09, GLM $p = 0.001$). This can be linked to a change in species count influencing the FQAI equation. Mean C

Table 1. Habitat-level dominant vegetation with corresponding mean cover percentages for 2016 (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 Transects						Control
Plant Taxa	**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	—
Galium 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 ± 0.0	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 ±11.1	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 ± 17.1	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 2. Habitat-level dominant vegetation with corresponding mean cover percentages for 2017 (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 Transects						Control
Plant Taxa	**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	—	—	—	—	—	—
Thelypteris 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 3. Habitat-level dominant vegetation with corresponding mean cover percentages for 2018 (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 Transects						Control
Plant Taxa	**D**	**PHB**	**PHM**	**SGM**	**TR**	**M**	**SB**	**IB**	**C**	**CT**
Boehmeria cylindrica	—	—	16.8 ± 24.2	0.3 ± 1.25	—	9.2 ± 25.4	—	—	—	—
Calamagrostis canadensis	—	—	—	21.6 ± 22.4	3.1 ± 11.9	0.5 ± 2.5	—	—	—	—
Ceratophyllum demersum	19.4 ± 24.1	10.8 ± 10.5	—	—	0.3 ± 0.0	—	19.4 ± 25.1	—	25.9 ± 32.0	—
Decodon verticillatus	—	2.8 ± 5.6	17.7 ± 19.2	—	—	4.7 ± 11.7	0.5 ± 0.0	2.0 ± 8.2	—	—
Elodea canadensis	5.0 ± 3.9	3.7 ± 5.8	—	—	—	—	15.5 ± 13.7	—	11.8 ± 17,7	—
*Hydrocharis morsus-ranae**	3.6 ± 5.3	14.5 ± 8.2	0.4 ± 0.0	—	8.5 ± 33.7	—	37.7 ± 25.4	8.0 ± 7.5	11.5 ± 14.4	1.3 ± 2.6
Impatiens capensis	—	—	32.1 ± 30.2	0.1 ± 0.0	0.1 ± 0.0	6.4 ± 13.7	—	—	—	1.3 ± 2.5
Potamogeton foliosus	26.9 ± 29.6	1.7 ± 3.2	—	—	—	—	0.6 ± 1.2	—	1.0 ± 2.7	—
Rorippa palustris	—	—	0.6 ± 2.5	0.7 ± 0.0	0.1 ± 0.0	14.7 ± 20.8	—	—	—	—
Scutellaria galericulata	—	—	5.0 ± 18.3	—	0.8 ± 9.4	17.4 ± 23.5	—	0.2 ± 0.0	—	0.1 ± 0.0
Spirodela polyrhiza	13.9 ± 14.9	24.5 ± 23.4	0.2 ± 0.0	—	1.0 ± 10.6	0.1 ± 0.0	34.5 ± 18.6	5.2 ± 9.2	29.5 ± 25.3	—
*Typha x glauca**	—	11.4 ± 13.6	15.0 ± 11.5	15.8 ± 16.2	9.5 ± 8.0	21.3 ± 13.8	1.1 ± 1.1	53.9 ± 15.1	—	53.7 ± 17.2
Utricularia vulgaris	22.0 ± 29.3	13.4 ± 23.0	—	—	0.2 ± 0.0	—	7.5 ± 6.4	—	5.8 ± 6.3	—
Verbena hastata	—	—	3.1 ± 18.1	—	—	23.8 ± 22.5	—	—	—	—

Table 4. 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, they are demarcated by a Y for Yes along with blank spaces meaning the species was not observed.

Plant Species	Observed in 2013	Seed (S) or Plug (P) pre-2016 Sampling	Observed in 2016	Observed in 2017	Observed in 2018
Acorus americana		S			
Alisma subcordatum		S + P			
Asclepias incarnata	Y	S	Y	Y	Y
Bidens cernua	Y	S	Y	Y	Y
Calamagrostis canadensis	Y	S	Y	Y	Y
Carex lacustris	Y	P	Y	Y	Y
Carex lurida		S + P		Y	
Carex scoparia		S + P			
Carex stipata		P			
Carex stricta	Y	S + P	Y		Y
Carex vulpinoidea		S			
Elymus virginicus		S	Y	Y	Y
Eutrochium maculatum		S			
Glyceria canadensis		S			
Iris versicolor	Y	S			
Juncus effusus	Y	S	Y	Y	Y
Leersia oryzoides		S		Y	Y
Mimulus ringens		S			Y
Persicaria amphibia	Y	P	Y	Y	Y
Poa palustris		S			Y
Pontederia cordata		P	Y	Y	Y
Sagittaria latifolia	Y	S + P	Y	Y	Y
Schoenoplectus tabernaemontani	Y	S + P	Y	Y	Y
Scirpus atrovirens		S			
Scirpus cyperinus	Y	S			
Scirpus polyphyllus		S			
Sparganium americanum		S			
Sparganium eurycarpum	Y	S	Y	Y	Y
Verbena hastata	Y	S + P	Y	Y	Y
Vernonia noveboracensis		P		Y	Y
Species present =	13	–	13	15	17

and weighted mean C for the channel transects were not significantly different across the three years of sampling.

We found no changes in dominant species across years on the pothole transects. FQAI and mean C differed significantly across years (GLM; FQAI 5.96, 5.99, 8.48 respectively, $p = 0.002$; mean C 3.04, 2.61, 7.34 respectively, $p = 0.018$), but weighted mean did not differ significantly.

We used two mound habitat types for these analyses—pothole mounds (PHM) and channel mounds (M). The channel mounds (M) were significantly different across the three sampling years for FQAI (2016 = 6.87, 2017 = 8.30, 2018 = 9.98, Kruskall-Wallis $p < 0.001$), mean C (2.51, 2.68, 3.16 respectively, $p < 0.001$), and weighted mean C (2.05, 2.28, 3.11 respectively, $p < 0.001$). There were no significant differences across years for FQAI or mean C for pothole mounds (PHM). However, there was a significant increase for weighted mean C (2016 = 2.00, 2017 = 2.59, 2018 = 3.35, Kruskall-Wallis $p < 0.001$).

Cattail Treatment Pre- and Post-Restoration

Mean cover of *T.* × *glauca* in the treatment areas decreased from 65.9% (SEM) in 2013 to 22.3% in 2016 and 3.3% in 2017, with an increase to 9.5% in 2018. In comparing pre-restoration (SEM) data for 2013 FQAI, mean C, and weighted mean C with 2018 data from two years post-restoration, significant differences were found in the cattail-treatment area data from the same areas. FQAI scores increased significantly from 2013 pre- to 2018 post-restoration (2013 = 4.13, 2018 = 7.01, Kruskall-Wallis $p < 0.001$) along with a significant increase in mean C (2013 = 2.01, 2018 = 3.12, $p < 0.001$) and weighted mean C (2013 = 0.42, 2018 = 3.17, $p < 0.001$).

Cattail Mat Control

With the overbearing dominance of *Typha* in the control areas, we found no significant change in mean C, but significant changes in assessment were recorded for FQAI and weighted mean C (Kruskall-Wallis: FQAI $p = 0.009$; weighted mean C, $p = 0.016$).

Discussion

Changes in Cattail

Along the channel transects, percent cover of *Typha* decreased from pre-restoration levels but increased from 2016 to 2018 at higher elevation SGM, IB, and M, likely influenced by disturbance from excavation and high lake levels in 2017. Cattail cover decreased from 2016 to 2018 in lower elevation SB and was negligible in deeper C. In the potholes, *Typha* increased in PHB from 2016 to 2018, also a probable result of excavation disturbance and 2017 high lake levels. Although created mounds (M and PHM) and higher elevation SGM hosted wet meadow species, they also had *Typha* seedlings.

Mean cover of cattail in treatment areas (TR) decreased greatly from pre-restoration 2013 to 2016 to 2017 and remained low in 2018. Cutting and herbicide treatments thus seem to parallel the results of Wilcox et al. (2018) in experimental studies upon which these treatments were based. However, additional data collection is needed to assess the longer-term results of the cattail treatments because lake levels likely influenced results also.

Other Vegetation Changes

Changes in non-cattail vegetation within TR areas from pre-restoration 2013 to post-restoration 2018 were likely related to the cattail treatment. Opening of the cattail canopy by excavation, creation of standing water, and higher lake levels resulted in increases in floating and submersed species. Such seems to be the case for invasion of dense mats of *H. morsus-ranae* and interspersed *U. vulgaris*. *Hydrocharis morsus-ranae* is an undesirable invasive species on the New York State DEC 2014 Prohibited and Regulated Invasive Plants list (NYSDEC 2014), and it has spread rapidly within the disturbed restoration areas at Braddock Bay. Robichaud and Rooney (2020) found *H. morsus-ranae* as a secondary invader when stands of *P. australis* that had been suppressed by herbicide treatment.

Evaluation of survival of planted plugs and seeded species was not conducted because it would require very intensive surveying by a large crew, as about 63,000 individual plugs were planted and seeding was extensive. Issues observed with plugs included herbivory and the contracted planting crew leaving various trays of plugs unplanted. Many of the seeded and planted species observed were also present in the pre-restoration sampling in 2013 and may not be a product of restoration efforts. Only half of the planted/seeded species were observed within the third year post-construction.

Influence of Lake Level on Results

An unpredictable issue while sampling Braddock Bay was the variation of lake levels from year to year. Average water depth in the treatment areas increased from 0 cm in 2016 to about 60 cm in 2017 and receded to about 5 cm in 2018. Cattails are sensitive to flooding (e.g., Grace and Wetzel 1981, Ball 1990, Bansal et al. 2019), and record high lake levels in 2017 likely restricted post-treatment cattail growth. Future water-level changes cannot be predicted, but additional years with high waters, such as 2019, would likely accentuate cattail reduction in the treatment areas.

Sedge-grass meadow (SGM) that was dry in 2016 had greater than 100 cm of standing water in high-water 2017, and many of the mounds (M and PHM) were then submerged. There was a shift to submerged aquatic vegetation, including atop the submerged mounds. In lower water 2018, *C. canadensis* increased in SGM and *C. lacustris* abundance increased in the PHB habitat but also had a slight decrease in abundance within the SGM and PHM habitats. Under less extreme lake-level conditions, the sedge-grass meadow and mound habitats should remain drier, with a reduction in cattail and establishment of more native wetland vegetation that can withstand dry spells (Wilcox et al. 2008).

Use of FQAI, Mean C, and Weighted Mean C

The FQAI statistic is currently a favored assessment of wetland plant community health used to evaluate the nativeness of an area based on the plant species present. There are different ways to calculate the statistic (Faber-Langendoen 2018), with mean C and weighted mean C as other options. 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 (Taft et al. 1997, 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). FQAI, mean C, and weighted mean C metrics may be useful in determining the type and quality of habitat needed for certain species to thrive (Taft et al. 1997).

Various combinations of significance in results occurred across the three metrics. Some tests showed only one metric to be significant, but no discernable pattern was present. The channels had low mean C and weighted mean C scores across the sampling years but a significant increase in FQAI that was influenced directly by an increase in species count. Mean C score remained low, so there was no increase in nativeness. No overall significant changes in FQAI, mean C, or weighted mean C occurred within the potholes. However, the mounded PHM habitat showed the FQAI statistic once again influenced by species count, with a significant increase across years.

Using just the mean C metric (the fidelity of a species to the environment) is not adequate because its role within the environment or niche habitat must be considered. The FQAI statistic showed a significant difference among the three sampling years, whereas the mean C and weighted mean C statistic used within these calculations showed no significant difference. The FQAI statistic is 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 have a lower FQAI score if most of the plant species that create that diversity have low C-scores. The ability of these metrics to use the gradient of nativeness proves beneficial in ranking of each species at each location.

Our data show that Braddock Bay is low overall for C-score, but some areas, such as the overall channels and channel mounds, show greater floristic quality due to an increase in richness, not an increase in native plants. We suggest that weighted mean C is a better metric to use because it changes a pre-determined state-wide or ecoregion-wide conservancy score into a local, site-level C-score, with little observed influence from species richness. 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 (Andreas et al. 2004, NEIWPCC 2011, Reznicek et al. 2014, Faber-Langendoen 2018). However, depending on species richness rather than dominance, results using the FQAI may be skewed toward 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 no observed influence from species richness like that 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 and provided the best comparisons.

Construction Methods—Evaluation and Recommendations

In some locations within the restoration site, construction could not get every channel, pothole, and mound to the exact planned depth or height. We found channels with reduced width that filled with sediment or floating pieces of cattail mat, thus affecting the restoration objective to create access to the sedge-grass meadow preferred by northern pike for spawning and nursery (Casselman and Lewis 1996, Cooper et al. 2008). However, larval pike were netted in spring sampling, so some access was gained (USACE 2017). Using aerial photography accompanied by ground-truthing at the site from a different entrance point, we determined that the largest excavated pothole had mostly filled in with sediment and regrowth of cattail mat toward the middle of the pothole. 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. As only 2.7 ha of potholes were created, they were not sufficient to provide suitable nesting habitat for Black Terns (>18.9 ha preference; Naugle et al. 2000). However, potholes were used by spring and autumn migrating waterfowl (C. Mitchell, unpublished data).

Mound heights also varied. They were intended to range from 75.35 m to 75.60 m (IGLD1985) to discourage cattail growth (Wilcox and Xie 2007, Wilcox et al. 2008, 2018). During 2017, with extreme high water levels (75.80 m), we found mounds built too low and also at the designed height that were flooded, while some built above the maximum height had dry spoil above the high lake levels.

If projects of this type are planned in the future, these observations suggest that excavated channels should be wider than the 4.2 m and perhaps deeper than the 1 m used at Braddock Bay. Even if pothole size is varied, depth should be greater than 1 m. Operators of excavation equipment should also take extra care to avoid creating fragments of cattail mat that might later obstruct channels. Wider and deeper channels or potholes would create more spoil material for constructing mounds. When building future spoil mounds, soils should be assessed prior to excavation. If the substrate to be excavated is highly organic, the mounds may settle and should be built higher. Less settling should occur if the substrate contains a mineral aggregate, such as clay.

Adaptive Management Needs

When performing restoration activities, adaptive management plans should be made. For restoration work at Braddock Bay, if all of the excavation had been done in a timely manner, some pothole and channel filling may not have occurred. Accordingly, future spoil-mound construction should be preceded by a soil survey to determine where adjustments should be made based on substrate composition and settling. Post-restoration management should work to clear blocked channels and deepen filled potholes to restore the intended connectivity.

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. *Lythrum salicaria* was a dominant in higher elevation M and PHM in lower water 2016, as the initial disturbance of the restoration site allowed for colonization by seed (Holvik et al. 2011). It decreased in high water 2017 but decreased more dramatically in 2018 despite lower lake levels. The likely cause was release of *Galerucella* beetles by NYSDEC after increases in the walking surveys were reported in 2017; *Galerucella* presence and heavily herbivorized *L. salicaria* were observed. Best management practices going forward would include continued targeting of cattails and other high impact invaders (e.g., *P. australis*, *L. salicaria*, *T. natans, H. morsus-ranae*) and making further additions to the list of species that threaten the bay (e.g., *Myriophyllum spicatum).*

Overall, Braddock Bay currently sits very low on the FQAI scale (Andreas et al. 2004, USFWS 2019). As recommended for most restoration projects (e.g., Galatowitsch 2012, Clewell and Aronson 2013), further data collection is needed to provide a better representation of how successful the restoration truly will be in the long term. With this in mind, continued monitoring of Braddock Bay by other researchers has been funded and scheduled through 2022 thanks to Region 8 of NYSDEC.

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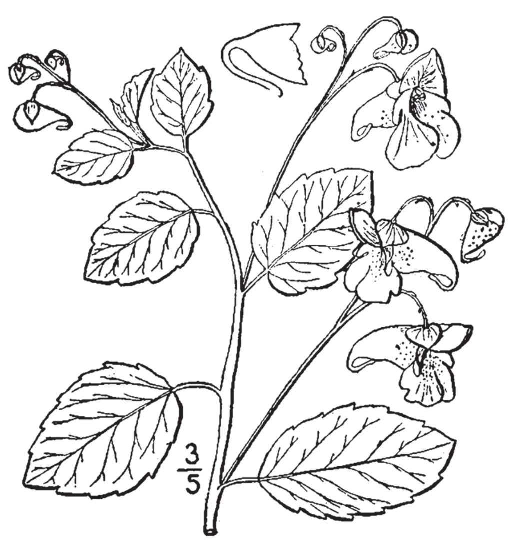

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