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Coastal sedge/grass meadow restoration in a peri-urban wetland via alteration of environmental filters: can hydrological constraint be trumped?

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COASTAL SEDGE/GRASS MEADOW RESTORATION IN A PERI-URBAN WETLAND VIA ALTERATION OF ENVIRONMENTAL FILTERS: CAN HYDROLOGICAL CONSTRAINT BE TRUMPED?

by

Eli L. Polzer

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Stabilized lake-level influence on *Typha* x *glauca* has so diminished the extent and richness of Lake Ontario shoreline sedge/grass meadows that they no longer conform to an historic trajectory. These conditions are not likely to change in the foreseeable future, so novel actions may be required to support their preservation. This research investigated the combined effects of a large-scale restoration overlapping multiple revegetation techniques. Excavated spoils from channel and pothole creation in two *Typha*-dominated marshes were reconfigured to create habitat mounds capable of supporting sedge meadow taxa. These mounds supported increased sedge/grass meadow taxa survivorship and richness by altering environmental conditions, such as elevation and soil moisture. However, a higher than expected rate of subsidence and rapidly diminishing elevations point to potentially shifting system dynamics that require further exploration.

Keywords: Lake Ontario coastal wetlands, hydrological regulation, *Typha* invasion, sedge/grass meadow restoration, novel ecosystem, multiple stable states, dredge spoil

PREFACE

Natural lake-level oscillations, integral to the structure and function of Great Lakes coastal wetlands, are no longer an absolute. Effective recovery of shoreline sedge/grass meadow plant communities within these wetlands necessitates an adoption of alternative tactics. This thesis investigates novel restoration methods in coastal wet meadows of Lake Ontario, where the effects of long-term stabilization and other landscape-scale stressors have led to overwhelming dominance by *Typha* x *glauca*.

The following chapters encapsulate a macroscopic overview of project concepts, goals, methodologies, and research, as well as a detailed treatment of executed research and analyses. Specifically, Chapter 1 outlines four coinciding studies conducted across three growing seasons that incorporate a novel restoration approach and active revegetation to support sedge/grass meadow recolonization. Two of these studies—1) examination of sedge/grass meadow plug survivorship atop created spoil mounds and 2) assessment of mound botanical richness as based on biotic and edaphic factors—are described in detail in Chapter 2.

The resulting manuscripts from these chapters will be submitted to the scientific journals *Wetlands* and *Ecological Restoration*.

CHAPTER 1: Lake Ontario coastal sedge/grass meadow restoration under hydrological constraint: a review and description of study methods

ABSTRACT

Across the Lake Ontario Basin, coastal lacustrine sedge/grass meadows remain highly imperiled despite being considered productive wetland ecosystems providing invaluable ecological services. Given the uncertain trajectory of these coastal communities, I summarize the issues related to *Typha* invasion and sedge/grass meadow community (SGM) degradation and review methodologies employed in coastal SGM restoration. I then propose a multifaceted, organism-specific approach based on literature review. This overall approach was implemented through a sequence of restoration actions and multiple ecological studies at a peri-urban coastal wetland in western New York State from 2014 through 2016 to evaluate the outcomes of these efforts.

INTRODUCTION

Coastal wetlands

Estuarine and lacustrine coastal wetlands and shoreline plant communities are intrinsically dynamic systems that function by means of an elaborate interplay between biotic and abiotic constituents and processes (Moffett *et al.* 2015). Spatially complex and temporally variable, they are recognized as playing a critical role in ecosystem function through carbon sequestration, storm-flooding mitigation and minimization of erosion, water quality improvement via waste and pollutant filtration, maintenance of water tables and recharge of aquifers, and providing breeding, nesting, feeding, and nursery grounds for fish and wildlife (Hartig *et al.* 1996; Sierszen *et al.* 2012).

In many coastal wetlands, including those of the Great Lakes, inter-annual water-level fluctuations that permit low water levels support the establishment of less competitive native species while creating a hydrologic barrier to clonal encroachment by invasive exotic taxa (Gathman *et al*. 2005; Wilcox *et al.* 2005, 2007; Hudon *et al.* 2006). However, for over half of a century, Lake Ontario coastal wetland plant communities have been maintained under hydrologic stasis through maintenance of static water levels. Hydroelectric power and waterway navigation demands, as well as pressure by shoreline property owners, have led to a disruption of dynamic coastal wetland processes via inhibition of episodic oscillations in lake hydroperiod. While natural lake-level fluxes suppress succeeding seral stages and initiate cyclical and seasonal shifts in plant community composition and structure, muting natural periodicity in Lake Ontario coastal wetlands facilitates a cascade of deleterious effects (Kelley *et al.* 1985; Wilcox 2004).

Therefore, rather than contributing to a “shifting mosaic,” stabilization yields what can best be described as “coastal squeeze,” a process in which intermediate habitat is effectively wedged between two opposing forces, thereby decreasing spatial extent and habitat quality (richness and diversity) (Doody 2004; Wilcox 2004; Hudon *et al.* 2006; Frieswyk and Zedler 2007; Trebitz and Taylor 2007;Wilcox *et al.* 2008;

Farrer and Goldberg 2009; Vaccaro *et al.* 2009; Cvetkovic and Chow-Fraser 2011; Pontee 2013). In particular, stabilization adversely favors the lakeward expansion of the upland shrub community and landward expansion of the emergent community, particularly the invasive hybrid cattail, *Typha* x *glauca* Godron (Keddy and Reznicek 1986; Wilcox 2004). Moreover, being subject to regional anthropogenic factors, such as urbanization, coastal development, and agriculture, the historical trajectory of Lake Ontario coastal wetland plant communities may now be circumvented or transformed (Cvetkovic and Chow-Fraser 2011; Martina *et al.* 2016).

Organisms of interest

TYPHA

The Laurentian Great Lakes are both a beachhead (*i.e.*, assembly point of species native to coastal North American waters whose first non-marine North American record was in the Great Lakes basin) and a gathering place (first North American records outside of the Great Lakes) for non-indigenous species, some of which become invasive (Rothlisberger and Lodge 2013). Within Great Lakes coastal wetlands, invasive hybrid cattail is the most prevalent, as well as the most dominant member of the *Typha* lineage (Shih and Finkelstein 2008; Elgersma *et al.* 2015).

Invasive hybrid cattail, hereafter *Typha*, is a cosmopolitan wetland invader derived via introgression of parental species, *Typha angustifolia* L. (narrow-leaved cattail) and *Typha latifolia* L. (broad-leaved cattail), capable of promoting monotypic habitat by displacing less aggressive, diminutive floristic taxa and thereby reducing native diversity and overall ecological integrity (Grace and Harrison 1986; Waters and Shay 1990; Frieswyk and Zedler 2007; Shih and Finkelstein 2008; Cvetkovic and Chow-Fraser 2011). *Typha* achieves dominance by heterosis and associated rapid clonal growth (biomass), both increasing invasiveness by means of greater investment in clonal subsidies, a trait observed in many global invasions (Travis *et al.* 2010; Song *et al.* 2013; Bunbury-Blanchette *et al.* 2015; Elgersma *et al.* 2015; Zapfe and Freeland 2015).

Typha dominance is also achieved via feedbacks associated with the production of recalcitrant biomass, which attenuates light (altering edaphic conditions), acts as a mechanical impediment by obstructing plug emergence, modifies biogeochemical pools and cycling, and impacts trophic dynamics (Angeloni *et al.* 2006; Farrer and Goldberg 2009; Tuchman *et al.* 2009; Vaccaro *et al.* 2009; Travis *et al.* 2011; Larkin *et al.* 2012; Martina *et al.* 2016; Lawrence *et al.* 2017). Not only capable of commandeering system resources, *Typha* x *glauca* initiates conditions favorable to its growth, thereby ensuring its sustained expansion (Farrer and Goldberg 2009; Farrer and Goldberg 2014). In this manner, *Typha* is uniquely capable of functioning as both a passenger and a driver of ecosystem change (MacDougall and Turkington 2005).

McNaughton (1966) said of *Typha* communities that “no … community is as ubiquitous and as lacking in floristic complexity.” Some research posits that this is a temporally-mediated outcome, in which certain factors decline (e.g., floristic diversity) and others increase (e.g., microbial denitrifier diversity) according to stand age or time since invasion (Mitchell *et al.* 2011; Geddes *et al.* 2014; Lishawa *et al.* 2014).

Invasive hybrid cattails (as well as invasive taxa overall) are significantly more abundant in Lake Ontario coastal wetlands than in the upper Great Lakes, where regional nearshore development (including shoreline armament) and agricultural inputs reflect greater overall land-use intensity, which inflicts a heavier ecosystem burden (Detenbeck *et al.* 1999; Goforth and Carman 2005; Trebitz and Taylor 2007; Boers and Zedler 2008; Freeland *et al.* 2013; Wensink and Tiegs 2016; Lemein *et al.* 2017). Because wetland response is a unique manifestation of total lake, tributary, and landscape influences, Lake Ontario is classified as more degraded (Trebitz and Taylor 2007; Boers and Zedler 2008; Freeland *et al.* 2013; Lemein *et al.* 2017).

SEDGE/GRASS MEADOW

Lacustrine sedge/grass meadows (sedge meadow, meadow marsh, wet meadow; hereafter, SGM) are dis-climax communities comprised primarily of long-lived perennial (clonal) graminoids and forbs expressly adapted to environmental fluxes, namely intermittent flooding and consequent biogeochemical transformations (Hotchkiss and Stewart 1947; Middleton 2002; Stanley *et al.* 2005). In theory, the morphological and physiological pre-adaptations of particular SGM taxa facilitate a marked resilience and rebound following disturbance removal (Costello 1936; Schutz 2000; Stanley *et al.* 2005).

Coastal sedge/grass meadow plant communities have been disproportionately impacted by the confluence of stabilized hydroperiod, anthropogenically-driven watershed factors (e.g., water quality, sedimentation), and the severe biotic pressure of nuisance species, principally *Typha* spp. (Wilcox *et al.* 1985; Wilcox *et al.* 2008). Collectively, this has resulted in a highly suppressed coastal wetland system that poorly reflects its historical scale and its potential diversity.

Although historically interwoven throughout the entire Great Lakes Basin as a narrow, interrupted coastal fringe, intact native SGMs now only dominate the coastlines of the upper Great Lakes, while coastal wetlands of the lower lakes remain particularly vulnerable, requiring persistent and protracted intervention to sustain a patchwork of valuable habitat for aquatic and wetland fauna (Detenbeck *et al.* 1999; Middleton 2002; DeCatanzaro and Chow-Fraser 2010; Greenhorn *et al.* 2016). Lake Ontario, in particular, has lost an inordinate amount of coastal meadow marsh since the 1960s, principally attributable to lake-level regulation and subsequent halving of its natural annual oscillations (~1.5 m to 0.7 m) (Wilcox *et al.* 2008; Vaccaro *et al.* 2009; Lemein *et al.* 2017).

Occupying the sheltered margins of lacustrine bays and drowned river mouth wetlands beyond the reach of storm surges, these *Carex stricta-Calamagrostis canadensis* wet meadows are now overwhelmingly dominated by ruderal and invasive flora and have a monotypic character (Lemein *et al.* 2017). While these communities appear superficially resilient to enduring pressures, underlying seed banks—more species-rich than extant vegetation—are markedly less resilient under invasive communities, being impacted by stand age and regional context (Frieswyk and Zedler

2006; Hall and Zedler 2010; Lishawa *et al.* 2015). Moreover, complex germination and developmental requirements, in concert with fragmented or isolated habitat, make wet meadows a challenging assemblage to support (Budelsky and Galatowitsch 1999; van der Valk *et al.* 1999; Schutz 2000; Budelsky and Galatowitsch 2004; Leck and Schutz 2005; Kettenring and Galatowitsch 2007a, b; Zukowski *et al.* 2010). These factors collectively reinforce one another, relegating Lake Ontario SGMs to a subordinate status, their absence reflecting diminished ecosystem integrity (Schutz 2000; Wilcox *et al.* 2007).

Spoil amendments in coastal wetlands

To date, many approaches have been used to offset coastal wetland degradation. Marsh sediment amendments are one such system of practices that endeavors to refashion deteriorated or lost wetlands by emulating physical habitat features of natural marshes—notably increased surface elevation and topographical heterogeneity (i.e., elevational pattern over a specific area), both being key factors in supporting ecosystem functioning and the re-creation of conditions adequate to reestablish wetland vegetation (e.g., alleviation of flood stress) and wildlife populations (Comoss *et al.* 2002; Costa-Pierce and Weinstein 2002; Larkin *et al*. 2006; Moser *et al*. 2007; Karstens *et al*. 2016).

The two most common amendment techniques are thin-layer placement and terracing. Thin-layer placement is a contemporary approach often used in tidal and salt marsh restorations, performed via high-pressure application of excavated slurry to a variable depth atop the marsh surface (Ford *et al*. 1999; Mendelssohn and Kuhn 2003; Yozzo *et al*. 2004; Ray 2007; LaPeyre *et al*. 2009; Stagg and Mendelssohn 2011). Terracing is another technique frequently used in estuarine environments, wherein emergent marsh is fashioned by excavating and sculpting sediments into mounded structures (McLellan 1990; Rozas and Minello 2001; Turner and Streever 2002; O'Connell and Nyman 2010; Armitage *et al*. 2014). Both techniques aim to ameliorate the effects of diminished marsh elevations (i.e., excessive flooding and inhibited growth and mortality of marsh vegetation), which thereby improves soil drainage and aeration and consequently vegetation production (Stagg and Mendelssohn 2011). These techniques may contribute to plant establishment and growth in freshwater coastal settings, but the effects of sediment additions on excavated spoil physical properties (e.g., elevation/subsidence, soil moisture, and bulk density), and how these relate to sedge/grass meadow vegetation establishment, have not been studied.

While spoil-supplemented marshes can be functionally equivalent to natural coastal marshes with respect to habitat structure and geomorphology (edaphic conditions and sedimentation rates), they are generally incapable of replacing all lost functions or lost biota (Streever 2000). Amidst rapid increases in aboveground plant biomass, these sediment-nourished marshes frequently sustain reasonable species assemblages (Cui *et al.* 2009; Graham and Mendelssohn 2013); however, not only do the resulting compositional matrices typically differ from those of reference wetlands, but outcomes often demonstrate variable responses in species diversity (Streever

2000; Rozas *et al*. 2005; Feagin and Wu 2006; O'Connell and Nyman 2010; Middleton and Jiang 2013; Bolam 2014).

Presently, there remain sizeable gaps in the scientific literature pertaining to use of spoil in coastal wetland restorations for ecological aims. Well-documented, generally applicable guidance of a non-industrial/non-commercial nature that can be used to target and thereby direct ecological trajectories does not currently exist (Yozzo *et al*. 2004; Berkowitz *et al.* 2017). Furthermore, while use of excavated sediments has long been practiced throughout the Great Lakes Basin, few long-term, quantitative scientific experiments examine their use and subsequent impacts on freshwater Great Lakes coastal wetland landscapes and resident biota.

GOALS

Under the Environmental Protection Agency's Great Lakes Restoration Initiative, The College at Brockport Wetlands Lab created a partnership with Ducks Unlimited, the New York State Department of Environmental Conservation, and the Town of Greece in winter 2014 to advance the Braddock Bay Phase II project, "Invasive Species Control and Wetland Restoration at Braddock Bay Fish and Wildlife Management Area," with the overarching goal to restore habitat for two Lake Ontario species of interest—northern pike (*Esox lucius* L.) and the New York State-endangered black tern (*Chlidonias niger* L.). Black terns are colonial nesting water birds whose populations have diminished significantly across the Great Lakes since the 1960s (Wyman and Cuthbert 2017). Similarly, *Typha* expansion over the last several decades and concomitant reduction in the structural complexity of shoreline vegetation have reduced spawning and nursery grounds for northern pike (Mingelbier *et al.* 2008).

To achieve multiple partner objectives, we collectively decided to perform a multi-measure restoration that would include: 1) strategic excavation of meandering nearshore channels and shallow potholes through the extensive cattail mat at two sites to facilitate shoreline access to sedge/grass meadow by northern pike; 2) subsequent placement of the excavated organic cattail spoil material to create elevated mounds directly adjacent to the landward side of channels where patches of sedge/grass meadow remained intact; and 3) intermittent mechanical and chemical treatment of *Typha*, with the precise timing and methodology following the research of Wilcox *et al.* (2018).

Following site descriptions, specific methodologies of each study are detailed below, described as a synchronized sequence of events from summer 2014 through fall 2016. These studies and associated methods are summarized in Table 1-1.

STUDY SITES

I conducted all studies at Buttonwood Creek and Buck Pond, two coastal wetland units within the Braddock Bay Fish and Wildlife Management Area (FWMA) in Greece, New York, USA (43°18'49.18"N, 77°42'49.55"W). Braddock Bay, an 860-ha coastal wetland embayment on the southern shoreline of Lake Ontario, lies in region V of the six climatic zones recognized by Carter for New York

State (1966). Encompassed by a warm-summer humid continental climate with mean summer temperatures of 13.9 °C and winter lows of 4.1 °C, historic annual mean precipitation exceeds 870 mm (Figure 1-1).

These wetlands consist primarily of emergent invasive cattail coastal marsh, interspersed with occasional floating-leaf macrophytes and other emergent species. Remnant sedge/grass meadow marsh occurs only as depauperate patches along the upland shrub perimeter, where reduced soil moisture inhibits encroachment by *Typha* spp..

Buttonwood Creek (BC) is a shallow, drowned river mouth tributary of Braddock Bay, supporting degraded riparian wetlands (43°17'59.79"N, 77°43'30.13"W) (Makarewicz 1989; New York State Department of Environmental Conservation 2007). Sediment movement within the creek is primarily driven by seiche activity from Lake Ontario and by stream drainage from the catchment. As Lake Ontario water levels dictate creek levels, habitat characteristics generally mimic those at Braddock Bay, with prominent cattail encroachment, *Phragmites* invasion, and greatly diminished sedge/grass meadow community. Underlying substrate is comprised of a mixture of deep, highly organic Edwards muck overlying lacustrine-derived marl and fluvial silts (U.S. Department of Agriculture 1973).

Buck Pond (BP) is a shallow, 75-ha protected embayment of the Braddock Bay FWMA located c. 4750 m downshore to the southeast from Braddock Bay (43°16'55.33"N, 77°39'59.73"W) (NYS DEC 2007) that drains the Larkin Creek Watershed (4366 ha) from the south (Makarewicz and Lampman 1994; Cadmus

Group 2010). It is bounded by Lake Ontario State Parkway to the south, residences along Edgemere Drive to the northeast, and Long Pond Road to the northwest. Although a barrier inhibits direct lake exposure, Buck Pond wetlands remain hydrologically connected to Lake Ontario via a narrow 2.5 to 4 m-wide passage at the southern reach, although this channel closes periodically when sand accumulates at the lakeshore. While habitat characteristics are analogous to those in Braddock Bay and Buttonwood Creek, Buck Pond supports areas of augmented microtopography and sedge/grass meadow marsh. Surficial deposits of lacustrine clays and glacial drift are overlain by buoyant floating mats of *Typha* biomass ≥1 m in thickness.

METHODS

Pre-implementation

SUMMER 2014

Transect vegetation

To document changes resulting from restoration actions, I initiated formal pre-restoration vegetation surveys in mid-July 2014 for a 1-month period during peak flowering and abundance. Surveys included the establishment of ten linear transects at each site oriented perpendicular to the elevation gradient, starting at the upper reach of remnant sedge/grass meadow marsh zones and extending approximately 60 m into either emergent cattail zone or to the open channel (Figure 1-2). I used a stratified random sampling method, with 1 m^2 quadrats at 24 points along each transect, for a sum of 240 quadrats per restoration site and a combined total of 480 quadrats. I identified all vascular species to the lowest taxonomic level possible within each quadrat, with percent cover visually estimated for each taxon. I recorded live ramet counts of *Typha* x *glauca*, as well as water depth, substrate type, organic sediment depth, and global positioning system (GPS) coordinates at each quadrat.

Implementation

WINTER 2015

All activities requiring earth-moving, including spoil excavation and placement, occurred in December 2014 through March 2015 using long-reach excavators operated by private subcontractors. Channels were engineered to mimic naturally occurring Great Lakes shoreline passages that typify access to northern pike habitat (Farrell *et al.* 1996; Mingelbier *et al.* 2008). These channels maintained depths of no greater than 1.25 m with 2:1 side slopes and variable widths of approximately 15 to 20 m. Potholes were created at variable length and width dimensions with depths of upwards of 2 m and 2:1 side slopes. In total, 17 channels comprising 2048 m and 11 potholes encompassing 2.79 ha were created, with 946 m of channels and 1.05 ha of potholes manufactured at Buttonwood Creek and 1102 m and 1.74 ha at Buck Pond (Figure 1-3).

Approximately 6845 m^3 of spoil material excavated for channel creation was placed on the landward side of new channels to manufacture over 4.35 ha of mounds, 1.32 ha at Buttonwood Creek and 3.03 ha at Buck Pond. These mounds were engineered to achieve final elevations of no less than 75.35 m (IGLD 85), within the optimal elevation for sedge/grass meadow marsh vegetation resistance to cattail (75.35 to 75.60 m IGLD 85) (Wilcox and Xie 2007). All mounds were broadcast-seeded immediately following spoil placement with a project-specific, regionally-sourced wet meadow mix prepared by Ernst Conservation Seeds (Meadville, Pennsylvania) and mulched using a standard cereal grain straw. Composition of the final seed mixes and their respective proportions are provided in the Appendix (Tables 1-2 and 1-3).

Cattail was knocked down by long-arm excavators while channeling, potholing, and mounding were being performed.

Post-implementation

SPRING-SUMMER 2015

Plot vegetation

In May 2015, I established 30 random 1 m^2 plots atop the created spoil mounds at each site to evaluate floristic response to novel site conditions. Each mound had at least one and up to a maximum of six plots, each oriented north to south.

I divided each plot into four 0.5 m^2 subplots representing four discrete treatments—i.e., no planting and no *Typha* cutting; planting and *Typha* cutting; planting without *Typha* cutting; and *Typha* cutting without planting. Within each planting treatment subplot, I planted three *Carex lacustris* and three *Carex stricta* plugs. In total, 180 plugs were planted for each species per site for a combined total of 720 plugs across both sites. Cutting treatments for *Typha* consisted of the excision and removal of aboveground biomass at the sediment surface using manual steel blade loppers. Geographic coordinates and elevation of each plot were collected at plot midpoints using a real-time kinetic global navigation satellite system (RTK-GNSS) corrected to the International Great Lakes Datum (IGLD 85). I initiated and maintained a 0.5 m weeded buffer around all plots throughout the growing seasons to inhibit edge effects.

I conducted formal vegetation surveys in mid-July to capture composition, structure, and abundance of all vegetation developing within each of the 240- 0.5 m^2 treatment subplots and summarized data at the plot level (n=60). I classified all extant subplot taxa to the lowest taxonomic level possible and visually estimated percent cover of each species and total recalcitrant biomass at 1% increments from 1 to 20% and at 5% increments thereafter. *Typha* ramet density measurements were captured at each subplot, and leaf heights of the three tallest *Typha* clones were measured using a standard metric measuring tape and averaged per subplot. Species richness was assessed at the plot level and calculated as the total number of unique taxa present.

Transect vegetation

In mid-July 2015, I re-established five of the original linear transects at each site to serve as control transects. These transects measured approximately 60 m in length and persisted outside the boundaries of site restoration actions (spoil excavation and placement, cattail mowing, seeding, and planting). Five linear transects (on average 55 meters in length) were also re-established within the interior of each restoration site where amendments had occurred to serve as treatment transects. Analogous to the controls, each treatment transect was oriented perpendicular to the elevation gradient; however, each extended across the mowed emergent (cattail management) zone, mounds composed of excavated organic spoil, and the newly created channels (Figure 1-4). I sampled vegetation during the period of peak abundance and flowering beginning in early July and continuing until late August following the original survey protocol.

Pothole vegetation

I performed submerged aquatic vegetation surveys in late August 2015 in created potholes at both restoration sites using a random sampling method at 20 points within central and peripheral regions of 11 potholes for a combined total of 220 quadrats. I identified all vascular species within each quadrat to the lowest taxonomic level possible and visually estimated percent cover for each taxon.

Hydrology

I launched a preliminary assessment of *in situ* hydrology on 1 July 2015, which was sustained until November 1 using two methodologies. First, I installed five Decagon EC-5 soil moisture sensors along three of the five treatment transects at both sites on the created spoil mounds. These probes collected hourly soil moisture readings, which were then stored in an on-site Decagon EM-50 data logger (one per treatment mound), allowing me to observe subtle soil moisture fluxes over time (n=15). Secondly, I deployed a total of seven shallow water-table wells outfitted with continuously-reading Solinst level-loggers at each site, with one well in the spoil on first, third, and fifth vegetation transects, one placed in the center of the cattail management zone on the first, third, and fifth transects, and one installed as a control outside of the restoration area. Where necessary, I secured water-table wells to metal conduit driven into solid glacial till to inhibit vertical movement. I collected well elevations using a real-time kinematic (RTK) GPS receiver capable of ± 15 mm vertical accuracy and calibrated them to IGLD 85 at the beginning and end of the season to ensure lack of movement and to evaluate the relationship between Lake Ontario water levels and the water table within the restored sedge/grass meadow marsh zone.

Soil

Finally, to quantify subsidence over time, I documented elevations and fine-scale topography across all mounds at the beginning and end of the growing season using a Trimble R6 RTK-GPS.

Cattail management

In late July 2015, I coordinated with the USFWS to mow the emergent cattail management zone abutting the remnant sedge/grass meadow zone using a hydraulically driven amphibious tracked vehicle fitted with an auxiliary mowing platform with two rotary blades capable of cutting brush and vegetation ≥ 5 cm in diameter (Marsh Master MM-2LX). I intended for the material resulting from this method to be fully fragmented to facilitate sunlight penetration to underlying strata as well as more rapid biomass decomposition. However, as the Marsh Master was functionally inadequate to cut in areas of water exceeding 46 centimeters, I hired a licensed subcontractor to finish cattail cutting using steel blade brush cutters in mid-August 2015. The same subcontractor returned in late September 2015 to apply herbicide to resprouting *Typha* ramets using an aquatic-approved glyphosate formulation. In areas of extensive cattail regrowth, herbicide was administered using pump backpack sprayers; however, hand wicking was conducted in zones having diverse and/or target sedge/grass meadow vegetation. These actions were timed according to documented carbohydrate reduction in *Typha* rhizomes as a result of avid ramet, leaf, and flower production (Linde *et al.* 1976; Sojda and Solberg 1993). Mechanical excision and chemical application during this physiological stage more readily deplete total carbohydrate reserves, which supports their extirpation.

SPRING 2016

Hydrology

I re-installed all hydrologic equipment on-site on 1 April 2016, which remained until 1 November. I supplemented my original design with three additional Decagon EM-50 data loggers with four EC-5 moisture sensors in the remnant sedge/grass meadow zone along vegetation transects at each site to facilitate comparison between the remnant SGM zone and manufactured spoil mounds. This addition increased the number of sensors to 27, with 15 sensors installed atop the spoil mounds and 12 in the remnant sedge/grass meadow. I also re-deployed water-table wells and Solinst level-loggers at eight locations per site. I included the additional well in the transitional zone between the remnant sedge/grass meadow and cattail management zones along the third transect.

SPRING-SUMMER 2016

Plot vegetation

In late May 2016, I evaluated the survivorship of individual plugs at both sites by visually comparing each planted subplot against preliminary planting schematics. A plug was considered extant if at the time of survey it sustained any amount of living aboveground biomass.

I re-sampled planted vegetation plots in mid-August 2016. Species richness was assessed at the plot-level and calculated as the total number of unique taxa present.

Transect vegetation

In early August 2016, I re-sampled vegetation along both control and treatment transects following the original survey protocol.

Pothole vegetation

I performed submerged aquatic vegetation surveys in late August 2016 in created potholes at both restoration sites following the 2015 procedures.

Soil

I continued to document elevations across all mounds at the beginning and end of the growing season in 2016. Additionally, having observed differing patterns of subsidence at each restoration site in 2015, I pursued a small-scale soil bulk density study to assist my understanding of its effect on site hydrology and floristic (re)colonization. To compare the bulk density of the remnant sedge/grass meadow and mound zones, I collected ten soil samples from the remnant sedge/grass meadow at each restoration site and one sample from random locations atop each of the 13 and 7 spoil mounds at Buttonwood Creek and Buck Pond, respectively. Each soil sample was comprised of three individual soil cores of a known volume (each 450 cm^3) collected to a depth of 10 cm. Soil samples were stored out of direct sunlight and on ice until transported to The College at Brockport facilities and refrigerated at 4° C until preparation and analyses were conducted. I performed a rudimentary bulk density analysis following Blake (1965), which required the removal of a 75 cm^3 subsample from each amassed sample, preliminary weighing, drying at 104° C until constant weight was reached, and secondary weighing. I then calculated soil bulk density of each subsample as: soil bulk density (g/cm^3) = dry weight of bulk sample (g) ÷ volume of soil core (cm^3).

Cattail management

In mid-July 2016, I again hired a licensed subcontractor to cut reemerging cattail ramets via steel blade brush cutters. Herbicide was applied in late September 2016 via backpack spraying and wicking.

LIMITATIONS

Although the review portion of this chapter is unique in that it is the first to examine a suite of botanical restoration studies of Lake Ontario coastal wetlands amended with excavated spoil, it may demonstrate one or more limitations. For example, while I attempted to be exhaustive in the literature search, it is possible that some studies may have been missed.

CONCLUSION

This chapter presented the overall project vision and key methods for four floristic restoration studies performed at two coastal wetland units at Braddock Bay WMA along the southern Lake Ontario shoreline. Using a suite of methods to initiate habitat amelioration in shoreline sedge/grass meadow ecosystems has many powerful benefits when faced with hydrological regulation, including increased ecological function. However, these studies are not exhaustive and do not necessarily capture all environmental characteristics essential to a successful outcome. To maximize the impact that the consequent research data have in the applied domain, as well as in the theoretical literature, more research of a similar nature needs to be pursued.

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TABLES

Table 1-1. Summary of habitat restoration methods (actions) and related parameters from 4 floristic studies at Braddock Bay WMA research sites from 2014 to 2016.

Floristic study[a]	**Restoration action**[b]	**Action initiation date**[c]	**Action end date**[d]	**Analytical sampling unit**[e]	**Analytical sample size**[f]
1	Spoil placement	December 2014	March 2015	Mound	20
1	Plot establishment	May 2015	May 2015	Plot	60
1	Subplot establishment	May 2015	May 2015	Subplot	240
1	*Carex* planting	May 2015	May 2015	Plug	720
1	*Carex* survivorship survey	May 2016	May 2016	Plug	720
1, 2	Hydrological survey - water-table well + level loggers	July 2015 April 2016	November 2015 November 2016	Well + Level logger 2015(2016)	14(16)
1, 2	Hydrological survey - soil moisture sensors	July 2015 April 2016	November 2015 November 2016	Sensor 2015(2016)	15(27)
1, 2	Soil bulk density analysis	June 2016	July 2016	Soil sample	30
1, 2	Elevation survey	May 2015 November 2015 May 2016 November 2016	May 2015 November 2015 May 2016 November 2016	Plot	60
2	Spoil placement	December 2014	March 2015	Mound	20
2	Post-implementation SGM vegetation survey	July 2015 August 2016	July 2015 August 2016	Plot(Subplot)	60(240)
3	Pre-implementation	July 2014	August 2014	Transect(Plot)	10(480)

	vegetation survey				
3	Cattail cutting/mowing	July 2015 July 2016	August 2015 July 2016	Hectares	10.0
3	Post-implementation vegetation survey	July 2015 August 2016	July 2015 August 2016	Transect(Plot)	10(480)
3	Elevation survey	July 2015 August 2016	July 2015 August 2016	Plot	240
4	Spoil excavation	December 2014	March 2015	Pothole	11
4	Post-implementation SAV survey	August 2015 August 2016	August 2015 August 2016	Plot	220

[a] Floristic study: 1=survivorship of planted *Carex* species on created spoil mounds; 2=colonization and richness of sedge/grass meadow (SGM) species on created spoil mounds; 3=effectiveness of 3-tiered cattail control sequence; 4=colonization of submerged aquatic vegetation (SAV) in created potholes.

[b] Restoration action(s) associated with respective floristic study.

[c] Action initiation date: date(s) of restoration action(s) in respective study.

[d] Action end date: date(s) of restoration action(s) in respective study.

[e] Analytical sampling scale: sampling unit per restoration action (combination of both sites).

[f] Analytical sample size: number of individuals in the main analysis per floristic study.

Table 1-2. Emergent species seed mix for lower mound elevations at Braddock Bay WMA study sites.

Scientific Name	**Common Name**	**Ecotype**	**Proportion (%)**
Schoenoplectus tabernaemontani	Softstem bulrush	PA	10
Sparganium eurycarpum	Broadfruit burreed	PA	50
Sagittaria latifolia	Broadleaf arrowhead	WI	10
Pontederia cordata	Pickerelweed	PA	8
Alisma subcordatum	American water plantain	PA	12
Bidens cernua	Nodding bur marigold	PA	2
Juncus effusus	Common rush	PA	2
Acorus calamus	Sweetflag	Midwest US	3
Onoclea sensibilis	Sensitive fern	PA	3

Table 1-3. Sedge/grass meadow species seed mix for upper mound elevations at Braddock Bay WMA study sites.

Scientific Name	**Common Name**	**Ecotype**	**Proportion (%)**
Carex comosa	Longhair sedge	PA	15
Carex vulpinoidea	Fox sedge	PA	20
Carex stipata	Awlfruit sedge	PA	6
Carex crinita	Fringed sedge	PA	6
Carex lurida	Shallow sedge	PA	10
Carex lupulina	Hop sedge	PA	6
Carex stricta	Tussock sedge	PA	1
Calamagrostis canadensis	Bluejoint	PA	1
Juncus effusus	Common rush	PA	5
Verbena hastata	Swamp verbena	PA	5
Eupatorium perfoliatum	Common boneset	PA	5
Asclepias incarnata	Swamp milkweed	PA	5
Carex lacustris	Lake sedge	PA	15

FIGURES

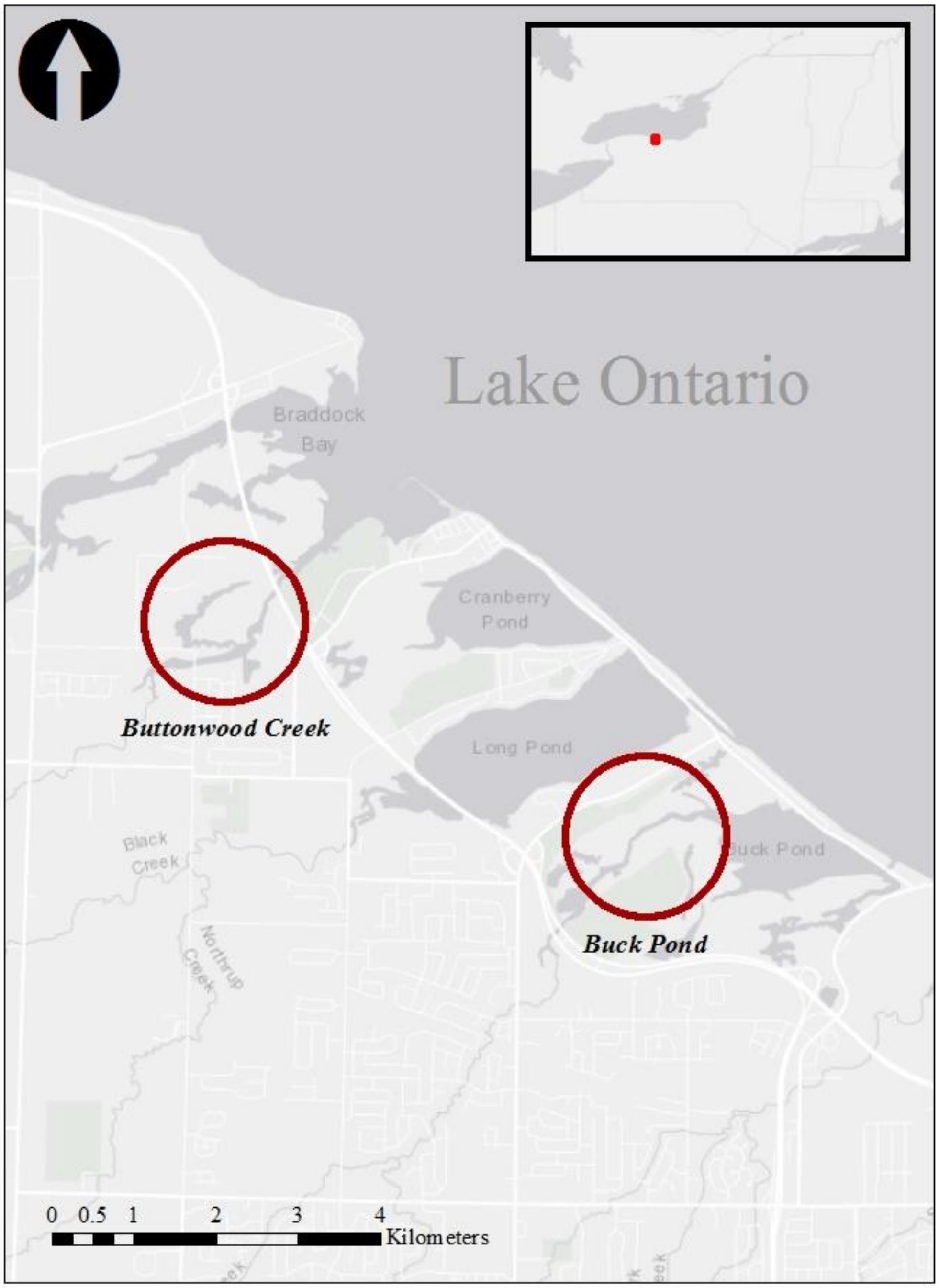

Figure 1-1. Overview map of Braddock Bay WMA study sites, Buck Pond and Buttonwood Creek.

Figure 1-2. Overview of pre-implementation vegetation transect layout at Braddock Bay WMA study sites, Buck Pond and Buttonwood Creek.

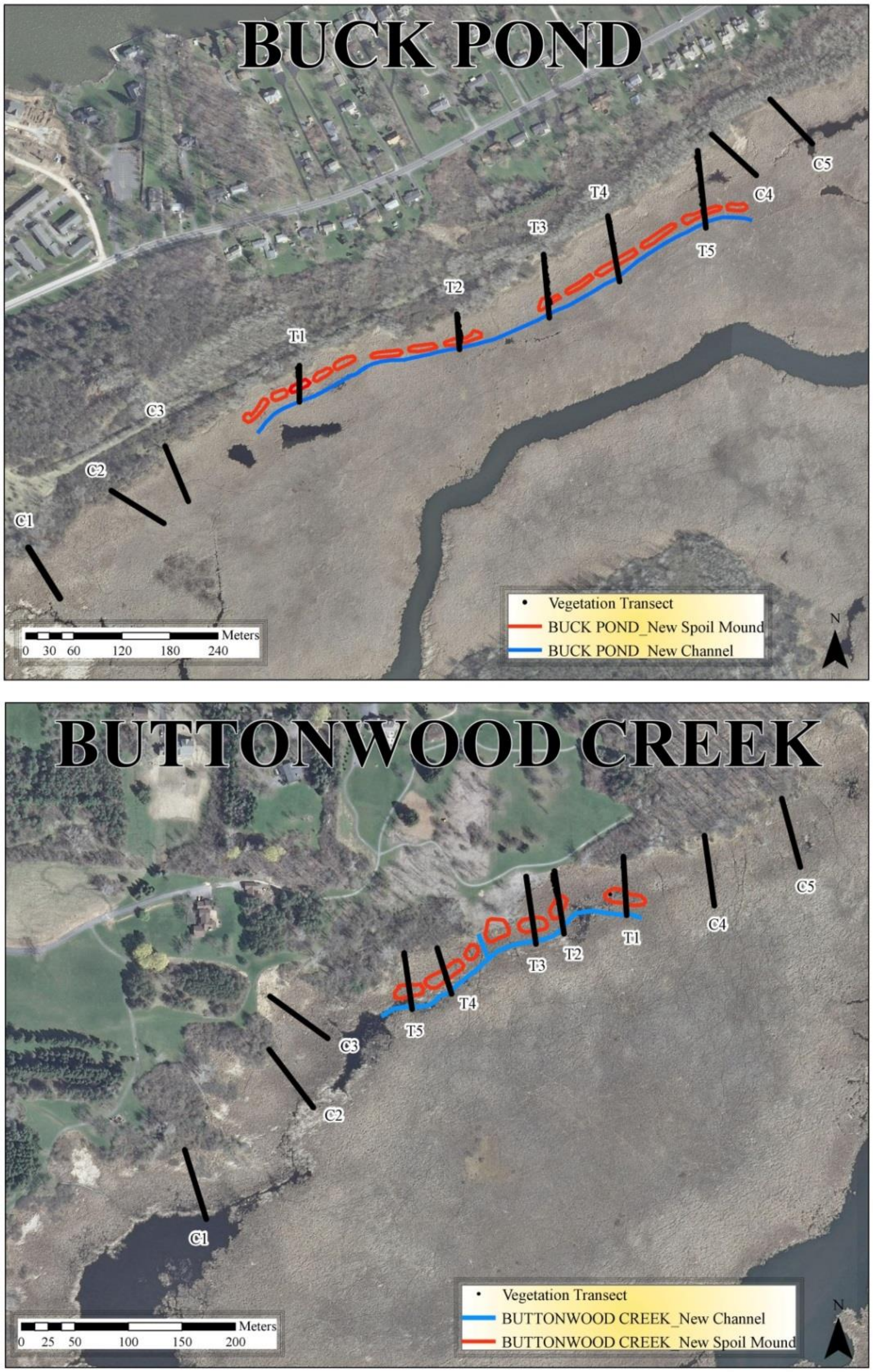

Figure 1-3. Schematic of created channels and spoil mounds at Braddock Bay WMA study sites, Buck Pond and Buttonwood Creek.

Figure 1-4. Overview of post-implementation vegetation transect layout at Braddock Bay WMA study sites, Buck Pond and Buttonwood Creek, respectively (T=treatment; C=control).

CHAPTER 2: Coastal sedge/grass meadow restoration in a peri-urban wetland via alteration of environmental filters: can hydrological constraint be trumped?

ABSTRACT

Degraded, novel conditions often require novel restoration approaches. For this purpose, I implemented a split-plot field study incorporating varying levels of *Carex* plug planting and *Typha* cutting to investigate subsequent patterns in sedge/grass meadow (SGM) marsh species survivorship (across 1 year) and richness (across 2 years) developing on created spoil mounds in two Lake Ontario coastal wetlands. Subplot treatments were as follows: no planting and no *Typha* cutting; planting and *Typha* cutting; planting without *Typha* cutting; and *Typha* cutting without planting. My goal was to evaluate the effectiveness of using excavated organic spoil in concert with active revegetation methods to assist sedge/grass meadow recovery in coastal marshes constrained by a regulated hydrological regime. I hypothesized that abiotic mound conditions would support increased sedge meadow taxa survivorship and richness while reducing *Typha* invasion. Here, I fitted models that included elevation, soil moisture, soil bulk density, and biotic treatments to subplot, plot, and site-level restoration data to test for survivorship and richness patterns. Results indicate that mounds at elevations greater than 75.35 meters are favorable for reducing cattail growth and development but do not consistently produce soil moisture conditions optimal for sedge meadow persistence. As well, plug survivorship and overall richness atop organic-rich excavated spoil mounds can

be readily improved via increased mound elevation in conjunction with decreased soil moisture and soil bulk density, with spoil mound communities floristically richer than those found in many remnant Lake Ontario sedge meadow marshes. Overall, outcomes suggest that excavated spoil-mounding may be a viable restoration method in hydrologically-constrained coastal freshwater wetland systems. This study offers a unique opportunity to examine multi-scalar processes involved in assembly following the use of novel restoration actions and the role of environmental drivers of survivorship and richness in restored coastal sedge/grass meadows.

INTRODUCTION

Peri-urban coastal sedge/grass meadows across Lake Ontario appear to defy a predictable historical trajectory due to a novel hydroperiod created by half of a century of lake-level stabilization. As a consequence of altered hydrology and increased land-use intensity—e.g., urbanization, regional agriculture, and shoreline armament—these communities have been radically diminished in diversity and are now dominated by ruderal and invasive flora—namely invasive hybrid cattail, *Typha* x *glauca* (Boers *et al.* 2008; Wilcox *et al.* 2008; Cvetkovic and Chow-Fraser 2011; Lemein *et al.* 2017). The shoreline habitat has been so transformed as to meet the criteria of a novel ecosystem. Novel systems are those that have progressed beyond some magnitude of similarity with a historical reference point and are thus capable of generating new patterns of assembly, as well as becoming modified in structure (e.g., soil and plants) and function (Hobbs *et al.* 2006; Zedler *et al.* 2012; Radeloff *et al.*

2015; Higgs 2017). Having departed from their historical trajectory, at least in the short term, these systems may defy conventional restoration efforts and thus require unique management considerations (Seastedt *et al.* 2008; Hobbs *et al.* 2014; Wensink and Tiegs 2016).

Many abiotic, biotic, and anthropogenic factors influence plant life-history patterns in Great Lakes coastal wetlands, which then drive plant survivorship and community composition (van der Valk 1981; Wilcox 2004; Farrer and Goldberg 2009; Johnston *et al.* 2010), with survivorship and richness generally decreasing in response to increasing stress (Keddy and Reznicek 1986; Hill *et al.* 1998; Pezeshki 2001; Johnston *et al.* 2008; Kovalenko *et al.* 2014). For example, increased urbanization is typically associated with a decrease in native wetland plant richness, in part by reducing survivorship and habitat area and quality (Findlay and Houlahan 1997; Houlahan *et al.* 2006; McKinney 2008; Larson *et al.* 2016). *Typha* x *glauca* is one of the chief stressors that actively influences sedge/grass meadow dynamics in Lake Ontario coastal systems by competing for preferential habitat, which then shapes dispersal dynamics, survivorship, and community richness (Wilcox *et al.* 2008).

Many restoration practitioners have recognized that *Typha* x *glauca* dominance across Lake Ontario will likely persist until a major overhaul of lake-level management was mandated (Frieswyk and Zedler 2007; Wilcox and Healy 2016; Wilcox *et al.* 2018). However, there are uncertainties as to if and how remnant sedge/grass meadows will endure and whether a shift to a more natural hydroperiod will return the system to its former equilibrium (Wilcox *et al.* 2018). Some researchers are investigating the practice of amending marsh surfaces with spoil material to increase surface elevation and topographical heterogeneity to address similar management issues in other coastal systems (Comoss *et al.* 2002; Costa-Pierce and Weinstein 2002; Larkin *et al*. 2006; Moser *et al*. 2007; Karstens *et al*. 2016; Regan 2016). Mounding is one such method in which marsh is formed or restored by sculpting amassed sediments into topographically variable convex features, typically separated by concave channels or potholes (McLellan 1990; Rozas and Minello 2001; Turner and Streever 2002; Bruland and Richardson 2005; Peach and Zedler 2006; O'Connell and Nyman 2010; Hough-Snee *et al.* 2011; Armitage *et al*. 2014). This technique may contribute to plant establishment and growth in coastal freshwater settings, but the physicochemical behavior of excavated spoil material—e.g., elevation/subsidence, soil moisture, and bulk density—and how these relate to sedge/grass meadow colonization and community development have not been studied.

Despite common use in other sectors and in the restoration of different plant community types, a sizeable gap exists in the scientific literature related to spoil placement as a viable coastal wetland restoration practice for ecological aims (McLellan 1990; Jorgenson *et al.* 1992; Nair *et al.* 2001; Biederman and Whisenant 2011; Gilland and McCarthy 2013; Evans 2017). While use of excavated spoils has long been practiced throughout the Great Lakes Basin, few long-term, quantitative scientific experiments examine their use and subsequent impacts on freshwater Great

Lakes coastal wetland landscapes and resident biota. Moreover, well-documented, generally applicable guidance of a non-industrial/non-commercial nature that can be used to target and direct ecological trajectories does not currently exist (Yozzo *et al*. 2004; Berkowitz *et al.* 2017).

A new Lake Ontario water-level regulation plan (Plan 2014) was initiated in 2017, its aim to permit marginal increases of lake-level variability in a manner that more soundly weighs residential, industrial, and ecological interests (International Joint Commission 2016a, b). While long-term research is required to determine if and how these changes will ultimately impact sedge/grass meadow communities, my expectations are such that arrested shoreline plant assembly in the face of water-level management will be an ongoing dilemma faced across Lake Ontario. Therefore, my study is motivated by an underlying desire to identify alternative means to support sedge/grass meadow communities without necessitating a return to completely natural lake-level fluctuations. My objectives in this study, therefore, were to examine the relative effects of using excavated organic spoil to re-create viable sedge/grass meadow habitat by evaluating biotic and abiotic factors on survivorship and richness of sedge/grass meadow species assemblages. To do so, I used two matrix *Carex* species of natural Great Lakes sedge/grass meadow communities that are adapted to variable conditions and best represent historical community attributes (Costello 1936). I asked the following overarching question: can created mounds produce abiotic conditions that support the restoration of sedge/grass meadow communities by maximizing sedge/grass meadow species survival and richness and minimizing *Typha* growth and development? I hypothesized that 1) spoil mounds at final elevations greater than 75.35 m (IGLD 85) will sustain significantly lower soil moisture and *Typha* x *glauca* growth and development; 2) survivorship of planted *Carex* plugs atop created spoil mounds will be significantly improved in *Typha* removal subplots at elevations greater than 75.35 m; and 3) created spoil mounds will demonstrate native floristic richness equal to or greater than that of remnant sedge/grass meadow communities upon reaching and maintaining optimal environmental thresholds when active revegetation methods are used.

METHODS

Study sites

I conducted the study at Buttonwood Creek and Buck Pond, two coastal wetland units within the Braddock Bay Wildlife Management Area (WMA) in Greece, New York, USA (43°18'49.18"N, 77°42'49.55"W) (Figure 2-1). Braddock Bay is an 860-ha coastal wetland embayment on the southern shoreline of Lake Ontario. It has a warm-summer, humid continental climate with mean summer temperatures of 13.9 °C and winter lows of 4.1 °C, with a historic annual mean precipitation exceeding 870 mm (Carter 1966).

Buttonwood Creek (BC) is a shallow drowned river mouth tributary of Braddock Bay supporting degraded riparian wetlands (43°17'59.79"N, 77°43'30.13"W) (Makarewicz 1989; NYS DEC 2007). Sediment movement within the creek is primarily driven by seiche activity from Lake Ontario and by stream drainage from the catchment. As Lake Ontario water levels dictate creek levels, habitat characteristics generally mimic those at Braddock Bay, with prominent cattail encroachment, *Phragmites* invasion, and greatly diminished sedge/grass meadow community. Underlying substrate is comprised of a mixture of deep, highly organic Edwards muck overlying lacustrine-derived marl and fluvial silts (U.S. Department of Agriculture 1973).

Buck Pond (BP) is a shallow 75-ha protected embayment of the Braddock Bay FWMA located c. 4750 m downshore from Braddock Bay (43°16'55.33"N, 77°39'59.73"W) (NYS DEC 2007) that drains the Larkin Creek Watershed (4366 ha) from the south (Makarewicz and Lampman 1994; Cadmus Group 2010). It is bounded by Lake Ontario State Parkway to the south, residences along Edgemere Drive to the northeast, and Long Pond Road to the northwest. Although a barrier inhibits direct lake exposure, Buck Pond wetlands remain hydraulically connected to Lake Ontario via a narrow 2.5 to 4 m-wide passage at the southern reach, although this channel closes periodically when sand accumulates at the lakeshore. While habitat characteristics are analogous to those in Braddock Bay and Buttonwood Creek, it supports areas of increased microtopography and sedge/grass meadow marsh. Surficial deposits of lacustrine clays and glacial drift are overlain by buoyant floating mats of *Typha* biomass ≥1 m in thickness.

Study implementation

I partnered with Ducks Unlimited, NYS Department of Environmental Conservation, and the Town of Greece to restore habitat for the NYS endangered black tern (*Chlidonias niger* L.) and Lake Ontario species of interest northern pike (*Esox lucius* L.). Preliminary site preparation by subcontractors included channel and pothole creation, placement of excavated organic spoil material to create elevated mounds along the shoreline where patches of sedge/grass meadow remained intact, and cutting followed by chemical treatment of *Typha*, with the precise timing and methodology following the research of Wilcox *et al.* (2018). Mounds were engineered to achieve elevations of no less than 75.35 m (IGLD 85), within the optimal elevation for sedge/grass meadow marsh vegetation resistance to cattail (75.35 to 75.60 m IGLD 85) (Wilcox and Xie 2007). All mounds were broadcast-seeded immediately following spoil placement with a project-specific, regionally-sourced wet meadow mix prepared by Ernst Conservation Seeds (Meadville, Pennsylvania) (Table 2-1).

Biotic manipulation

Carex stricta Lam. and *C. lacustris* Willd. were selected as study organisms due to their significance as sedge/grass meadow matrix dominants of historic and contemporary sedge/grass meadows in Lake Ontario coastal marshes. Plugs were purchased from a local wetland nursery specializing in regional genotypes (Southern Tier Consulting, West Clarksville, New York). *Carex stricta* plugs were ±11.5 cm tall with abundant above-ground shoots when they were transplanted into field plots. *Carex lacustris* plugs were a minimum of 15 cm tall with 1 or 2 vigorous above-ground shoots when they were planted.

Carex plugs were planted to a depth of 7 cm in spring 2015 following a randomized block design. To account for micro-environmental site effects and the influence of plant competition, planting and *Typha* cutting treatments were assigned at random to each of the thirty 1 m^2 plots per site, with three plugs of each species established in each 0.5 m^2 planting treatment subplot, resulting in four discrete treatments—treatment 1: no planting/no *Typha* cutting (control); treatment 2: planting and *Typha* cutting; treatment 3: planting without *Typha* cutting; and treatment 4: *Typha* cutting without planting. In total, 180 plugs were planted for each species per site, for a combined total of 720 plugs across sites. Cutting treatments consisted of the excision and removal of aboveground biomass of *Typha* at the sediment surface using manual steel blade loppers (Fiskars Brands, Inc., Middleton, WI, USA). The geographic coordinates and elevation of each plot were collected at the plot midpoint using a Trimble R6 real-time kinetic global navigation satellite system (RTK-GNSS) (Trimble Inc., Sunnyvale, CA, USA), corrected to the International Great Lakes Datum (IGLD 85).

To maintain aspect uniformity, each 1 m^2 plot was oriented north to south. A 0.5 m weeded buffer was maintained throughout the 2015 and 2016 growing seasons to inhibit edge effects. Plots were watered upon planting and for three weeks thereafter to increase the probability of plug survivorship. Figure 2-2 illustrates the planted plot treatments and orientation atop created mounds. The richness component of the study was maintained for two growing seasons, beginning June 2014 and ending November 2016. The survivorship component of the study was maintained for one growing season, from June 2015 to May 2016.

Field measurements

ABIOTIC FACTORS

ELEVATION

Surface elevation of all planted subplots was captured following plug transplantation in May 2015 in all 60 quadrats, and thereafter at the end of the 2015 and 2016 growing seasons using a real-time kinematic GPS (Trimble R6 RTK-GNSS) (vertical accuracy ±0.02 m), corrected to IGLD85. Analyses were performed using calculated project averages across both seasons. Seasonal and total rates of subsidence of the marsh substrate were calculated for each mound and each site as the change in elevation (m IGLD 85) over time.

SOIL MOISTURE

Hydrological sampling, evaluated as soil moisture content, was implemented to assess the relationship between mound elevation, associated soil moisture, and ensuing effect on *Carex* plug survivorship and plant community richness.

Manual soil moisture measurements (% volumetric water content, vwc) were collected at the subplot-level once per week between June and August 2015 using a portable handheld Delta-T ML3 ThetaProbe soil moisture sensor with HH2 moisture meter (±1% accuracy) (Delta-T Devices Ltd., Cambridge, UK). At each visit, a single measurement was collected in the interior corner of individual subplots following the cardinal directions, and then mean average calculated across each plot. Analyses were performed using seasonal averages calculated at the plot-scale.

SOIL BULK DENSITY

I collected a total of 20 composite soil samples in June 2016 across both study sites (Buck Pond- 13; Buttonwood Creek- 7) for analysis of soil bulk density (g/cm^3) at the mound level. Soil bulk density is defined as the mass of oven-dry sample of undisturbed soil per unit bulk (wet) volume (Ruehlmann and Korschens 2009). Each soil sample was comprised of three individual soil cores collected to a depth of 10 cm from locations directly adjacent to three random planted plots atop each mound, each approximately 450 cm^3 in volume. Soil samples were temporarily stored at 4°C prior to preparation and laboratory analysis. Bulk density analysis followed Blake and Hartge (1986) and consisted of removing a 75 cm^3 subsample from each amassed sample, removal of root and rhizome biomass, preliminary weighing, drying at 105°C until constant weight was reached (minimum of 72 hours), and subsequent weighing. Soil bulk density of each subsample was then expressed in grams of oven-dried soil per cubic centimeter (bulk density = oven-dry soil (g) / soil volume (cm^3)).

BIOTIC FACTORS

PLUG SURVIVORSHIP

I documented survivorship for 360 *Carex stricta* and 360 *C. lacustris* plugs, having planted 180 of each species in both treatments 2 (planting and cutting) and 3 (planting without cutting). In total, 360 plugs were planted within subplots subjected to mechanical removal of *Typha* to eliminate effects of competition (treatment 2) and 360 plugs were planted in subplots receiving no *Typha* amendment to test for competitive effects (treatment 3).

I evaluated the survivorship of individual plugs at both sites from 25 to 31 May 2016, by visually comparing each planted subplot (treatments 2 and 3) against preliminary planting schematics, recording 1 for living and 0 for expired. A plug was considered extant if it sustained any amount of living aboveground biomass at the time of survey.

PLANT SPECIES RICHNESS

Detailed vegetation surveys were performed from 11 to 17 July 2015 and 13 to 16 August 2016 to capture composition, structure, and abundance of all vegetation developing within each of the 240- 0.5 m^2 subplot treatment. I classified all extant subplot taxa to the lowest taxonomic level possible and visually estimated percent cover of each species and total standing dead litter at 1% increments from 1 to 20% and at 5% increments thereafter. *Typha* ramet-density measurements were captured at each subplot, and leaf heights of the three tallest *Typha* clones were measured using a metric measuring tape, and then averaged per subplot. Species richness was assessed at the plot level and calculated as the total number of unique taxa present.

Statistical analyses

ABIOTIC

I generated preliminary summary statistics in R for each abiotic factor, accounting for temporal variations by calculating mean averages, where appropriate (R Development Core Team 2014; RStudio Team 2016). Variables included mean average surface elevation at the plot and mound levels; bulk density at the mound level; and mean average soil moisture at the plot, mound, and site levels. I used nonparametric 2-sample Mann-Whitney tests to draw between-site comparisons of soil moisture and soil bulk density (site-level). Additionally, I used nonparametric Kruskal-Wallis tests to examine differences in surface elevation and soil moisture within mounds (plot-level).

BIOTIC

PLUG SURVIVORSHIP

Generalized linear mixed effect models (GLMMs) provide a viable solution to the constraining issue of normality assumptions for binary survivorship data without the need for transformation. This method allows differences between individuals to be assessed properly using the metric most appropriate to the theoretical context that combines a variety of abiotic and biotic variables occurring at different scales. I performed analyses in the R programming software environment using RStudio version 1.0.153 (version 3.3.2, R Development Core Team 2014; RStudio Team 2016).

In preparation for model fitting, model assumptions were verified by plotting residuals versus fitted values and versus each model covariate. I also tested for multi-collinearity, normality, homogeneity of variance, and outliers among individuals, as well as among groups, assessed appropriateness of parameter estimates and interactions, and confirmed data dependency structure (Graham 2003; Bolker *et al.* 2008; Zuur *et al.* 2010; Zuur and Ieno 2016). Fixed effects were tested for overdispersion.

I fit and selected models using the Laplace approximation in the lme4 package, and used an information criterion approach to compare Akaike Information Criterion corrected for sample size (AIC_c) scores between models (Zuur *et al.* 2009; Bates *et al.* 2015). I used a logistic regression with a binomial (logit) link function to model plug-level survivorship. Fixed effects for the logistic-based models included elevation, soil moisture, soil bulk density, and subplot treatment. Hierarchical random effects included site, mound, and plot (each nested within the next). I included biologically-relevant interactions in preliminary modeling efforts, such as those between subplot treatments, elevation, and *Carex* species, as well as between species and bulk density, although they were not necessarily included in the final models.

I ran all full models and assessed the error structure to remove any random effects to which no variance was attributed. Those near-zero variance random effects (i.e., variance ≤ 0.0001) whose exclusion improved the fit of the model were also removed. Prior to analyses, all fixed effects included in the models were converted to *z* scores, using the scale function in R, to improve the fit of the model and to allow for the direct comparison of the effect size of each predictor variable. After configuring the models, each full model with non-informative random effects was put through an automated model selection process (Bartón 2017). I selected as the top model for each analysis that which had the lowest ΔAIC (0.00).

I calculated conditional averages for effect size to provide the relative importance of each predictor variable of the top models. I then calculated 95% confidence intervals for each predictor included in the set of models <2 ΔAIC_c and identified informative predictors as those not overlapping zero (Arnold 2010; Thiele and Markussen 2012; Kleist *et al.* 2017). To assess goodness-of-fit, I compared all top models to intercept-only null models with random effects and calculated conditional pseudo R^2 (Nakagawa and Schielzeth 2013). I ran models for plug survivorship, which included all explanatory variables and relevant interactions. Finally, I used a nonparametric Kruskal-Wallis test to assess if survivorship differed among mounds.

PLANT SPECIES RICHNESS

Richness analyses closely followed those for survivorship; however, they were adapted for richness count data summed at the subplot level. To evaluate basic spatial and temporal richness trends, I conducted summary analyses by year and at each hierarchical level (site, mound, and subplot). I used a Poisson GLMM to model and analyze changes in total, native, and sedge/grass meadow species richness at three levels of spatial resolution (site, plot, and subplot). Six variables were input as fixed covariates: subplot treatment (categorical with four levels) and several continuous factors—elevation, % litter, % *Typha* cover, *Typha* ramet density, and mean *Typha* leaf height. Plot was included as the sole random effect. Interactions between variables were explored, although were not necessarily included in the final models. Using the Poisson distribution with log link function and the lme4 package in R, I compared AIC_c scores between models, therein generating 95% confidence intervals, fixed covariate effect sizes, and conditional pseudo R^2.

As part of the overall richness analyses, I generated summary statistics in R for three measures of *Typha* x *glauca* abundance within subplots- % *Typha* cover, *Typha* ramet density, and mean height (m) of leaves of the 3 tallest *Typha* ramets. Nonparametric Spearman rank correlations (r_s) were used to evaluate spatial and temporal patterns in *Typha* extent by examining relationships by year, by site, by mound, and by treatment. Additionally, I used a Poisson distribution regression with a log link function to model *Typha* percent cover and ramet density at the plot level. Fixed covariates included subplot treatment (categorical with four levels), site and year (categorical, each with two levels), and elevation (continuous). Plot was incorporated as the only random effect. I explored interactions between variables, but they were not necessarily included in the models. *Typha* model fitting and validation mimicked those of the richness data.

RESULTS

ABIOTIC FACTORS

ELEVATION AND SUBSIDENCE

Plots at Buttonwood Creek had significantly lower surface elevations than those at Buck Pond, both at the beginning (V=7, p<0.0001) and at the close of the first year (V=1, p<0.0001). Following placement of excavated spoil, only 50 percent of plots at Buttonwood Creek were within the optimal elevation range, in contrast to 97% of plots at Buck Pond, demonstrating substantial mound subsidence. This pattern became noticeably more dramatic by the end of year 1, in which only 30 percent of plots (n=9) were within the appropriate range (75.35 to 75.6 m IGLD 85) at Buttonwood Creek, while 90% were in the appropriate range at Buck Pond (n=26) (Figure 2-3). However, neither Buttonwood Creek nor Buck Pond demonstrated any significant differences in within-site plot elevations (H=13.49, df=12, p=0.335; H=15.69, df=12, p=0.206, respectively) (Figures 2-4a and 2-4b).

SOIL MOISTURE

Extreme precipitation patterns were observed throughout the length of the study, with 2015 sustaining exceptionally high water levels relative to the five-year average, while those of 2016 were remarkably reduced (0.1 m lower than normal) (National Centers for Environmental Information 2018). This variability influenced soil moisture patterns, which ranged from 4 to 100 percent saturated (% vwc) across both sites and all mounds during the sampling period. An optimal average soil moisture value of 30.0 (% vwc) was generated by accounting for the greatest proportion of live plugs persisting through a full season within plots that had maintained surface elevations >75.35 m IGLD 85 for the full duration.

Soil moisture decreased across the two-year study period by an average of 24% across plots at Buttonwood Creek, versus an average of 18% at Buck Pond. Seasonal average soil moisture across plots at Buttonwood Creek was significantly greater than that at Buck Pond ($V=433$, $p<0.000$) (Buttonwood Creek mounds—66.53% vwc; Buck Pond mounds—25.11% vwc) (Figure 2-5a). Additionally, seasonal median average soil moisture significantly differed among mounds at Buttonwood Creek ($H=18.54$, $df=6$, $p=0.005$, 17.9 to 88.7%), whereas among-mound soil moisture differences at Buck Pond were not significant ($H=10.28$, $df=12$, $p=0.591$), although they ranged from 12.6 to 56.6 percent volumetric water content (Figure 2-5b).

SOIL BULK DENSITY

Evaluation of edaphic features of mounds at both sites demonstrated that Buttonwood Creek, while underlain by deep organic muck and heavy clays, did not have a median soil bulk density (n=7, 0.098 g/cm^3) significantly greater than that at Buck Pond (W=65.5, p=0.122), which displayed a greater proportion of desiccated, albeit intact, fibric cattail mat “peat” (n=13, 0.076 g/cm^3) (Table 2-2; Figure 2-6).

BIOTIC FACTORS

PLUG SURVIVORSHIP

Overall survivorship across both sites and both species was nearly 75%, accounting for 533 individual plugs of the 720 total planted. Survivorship was similar between species, with 77% of *C. stricta* plugs surviving and 71.4% of *C. lacustris* plugs surviving. Survivorship was significantly greater at Buck Pond than Buttonwood Creek (W=59760, p=0.018, Figure 2-7). Within sites, there were no significant differences in survivorship among mounds (Figures 2-8a and 2-8b). While *in situ* herbivory was observed throughout the 2015-2016 season, it demonstrated no lasting influence on plug survivorship.

Effect of treatments on survivorship

A total of 150 *Carex stricta* plugs survived at Buck Pond following a full year of treatment, with 73 (81.1%) and 77 (85.6%) was no significant difference in survivorship treatments 2 (plug planting with *Typha* cutting) and 3 (planting without cutting), respectively. A total of 130 *C. lacustris* plugs survived at Buck Pond, with 62 (69%) and 68 (76%) in treatments 2 and 3. At Buttonwood Creek, a total of 66 (73.3%) and 59 (65.6%) *C. stricta* plugs and a total of 64 (71.1%) and 63 (70%) *C. lacustris* plugs persisted in treatments 2 and 3, respectively.

No significant difference in survivorship between treatments was detected (W=64440, p=0.866), with 265 (73.6%) and 267 (74.2%) individual plugs remaining at the end of the full year for treatments 2 and 3, respectively. Overall plug survivorship (ignoring species) in treatment 3 (competition with *Typha*) at Buck Pond was significantly greater than at Buttonwood Creek (W=14130, p=0.006). However, there was no significant difference in survivorship between sites across species in treatment 2 (no competition with *Typha*) (W=15750, p=0.551).   Additionally, there were no significant differences in survivorship detected between species across sites in either treatment (treatment 2: W=15030, p=0.121, treatment 3: W=15750, p=0.548) (Table 2-3).

Effect of elevation and bulk density on survivorship

In general, *C. stricta* survivorship was greatest at the uppermost elevations (>75.35 m IGLD 85), particularly at Buck Pond, where 87 percent of plots maintained optimal elevations during the study. Planted plugs at or above the 75.35 m threshold experienced 58% greater survival (n=174) relative to the survival observed at lesser elevations (n=101). In general, *C. stricta* plugs outperformed *C. lacustris*, with the exception of Buttonwood Creek plots located below the 75.35 m threshold (Figure 2-8).

Soil bulk density optima (0.08 to 0.12 g/cm^3) were generated based on the target surface elevation range (75.35 to 75.60 m IGLD 85), and within which the greatest plug survivorship occurred. In general, greater survivorship occurred across plots having a lower soil bulk density. Similarly, plug survivorship of both *C. stricta* and *C. lacustris* was enhanced in substrates with reduced soil bulk densities (Table 2-4).

Effect of soil moisture on survivorship

Buttonwood Creek plug survivorship across plots that maintained the target elevation for the full-year duration and within the established soil moisture range was fairly poor, with extant individuals accounting for only 25% of total survivors (n=64). Conversely, survivorship across plots at Buck Pond capable of maintaining target elevation and moisture optima was substantial, accounting for over 50% of total site survivorship (n=185 individuals). Additionally, both *C. stricta* and *C. lacustris* had a marked preference for reduced soil moisture conditions (Figure 2-11; Table 2-4).

Plug survivorship modeling

Candidate models are presented in Table 2-5. The most parsimonious model derived for survivorship characterized species and increasing elevation as the most significant fixed predictors of plug survivorship (pseudo R^2=0.0784), each increasing survivorship, respectively, by a log-odds average of 0.266 (±0.104 SE, 95% CI: 0.0371 to 0.6554) and 0.345 (±0.158 SE, 95% CI: 0.0615 to 0.4783) (Table 2-5). This model was superior to the intercept-only model that included only random effects (ΔAIC_c=5.52, Table 2-5) (χ^2=11.52, p=0.009). Estimate effect sizes of each parameter, as well as 95% confidence intervals and resulting p-values, are included in Table 2-6.

PLANT SPECIES RICHNESS

Assessment of overall species richness, which included both native and non-native species, yielded 140 taxa across two years of monitoring 240- 0.5 m^2 subplots, with 69 species shared across the two-year span (Table 2-7). Of the 140 species, 35 were exotics, 8 were documented noxious weeds (invasive), 54 were obligate wetland species, and 27 were facultative-wet species (NYS DEC 2014; Lichvar *et al.* 2016; National Plant Board 2018; U.S. Department of Agriculture 2018) (Tables 2-8a and 2-8b). Species richness increased significantly from 2015 to 2016 across all richness measures (total: V=196, p<0.000; native: V=389.5, p<0.000; SGM: V=2514, p<0.000). Native species richness encompassed 64 and 98 taxa in 2015 and 2016, respectively, with herbaceous forbs dominant across both years, followed by graminoids and woody taxa (Table 2-8c). SGM richness was 9 in 2015 and 14 in 2016 and was exclusively comprised of sedge/grass meadow matrix dominants, such as *Carex*, *Calamagrostis*, and *Juncus* species (Table 2-9).

Nine of 13 seeded species were among the 81 taxa found in 2015 sampling, while 12 of 13 appeared among 128 taxa sampled in 2016 (Tables 2-1 and 2-7). These species comprised a decreasing percentage of the total richness across the two years. Certain seeded taxa, such as *Carex lurida*, were not observed in either year. Others, such as *Carex stricta*, were only rarely observed, with the exception of planted plugs. Additionally, other taxa, including *Calamagrostis canadensis* and *Verbena hastata*, were abundant across both years, and differentiating between recruitments as a result of active seeding versus migrant colonization from the adjacent landscape was not possible. Additionally, while site herbivory was observed throughout the study, it demonstrated no lasting influence on species richness.

Median total species richness of the 0.5 m^2 subplots over two years of sampling was positively correlated with average *Typha* cover (S=1.54e+07, r_s=0.167, p=0.0002) and average leaf height (S=1.51e+07, r_s=0.179, p<0.000), but not ramet density (Table 2-10). Conversely, there were no explicit correlations between *Typha* measures and either native or SGM richness.

Effect of treatments on richness

Although species richness increased significantly from 2015 to 2016, values at each resolution were fairly comparable between subplot treatments and between sites (Table 2-11). Across sites, I observed a median total (native plus exotic) and native richness of 6 species, as well as a sedge/grass meadow species richness of 3, in treatment 2 subplots in 2015. These values increased to 12, 9, and 4, respectively, in

2016. Similarly, I observed a median total richness of 8, a native richness of 6, and SGM richness of 3 in 2015, which subsequently increased to 12, 9, and 4, respectively, in the following year. Subplot treatment 3 appeared to consistently generate a slight richness advantage when compared to treatment 2, whether evaluated by year, site, or resolution of richness.

Species richness in subplot treatments 1 and 4 (no planting) maintained somewhat decreased values. Overall richness patterns, from greater to lesser richness, followed the general order—treatment 3_{BC}>treatment 3_{BP}>treatment 2_{BC}>treatment 2_{BP}>treatment 1>treatment 4. Additionally, I identified significant differences in richness as the result of subplot treatments at all resolutions, with treatments 2 and 3 consistently demonstrating increased richness (Figure 2-12; Table 2-12).

Effect of soil moisture and bulk density on richness

Species richness was assessed in tandem with soil moisture at the plot level and with soil bulk density at the mound level, where I detected significant gains in species richness at all levels of resolution as a result of increasing soil moisture. More specifically, Buttonwood Creek demonstrated greater richness at all levels (total- 12.5, native- 9.75, and SGM- 4) throughout 2016 despite displaying greater soil moisture (seasonal median of 65.7% vwc) and augmented bulk density (0.111 g/cm^3). Species richness at Buck Pond was moderated by substantially drier substrate conditions and lower bulk density (0.079 g/cm^3), with a total richness of 10.5, native richness of 8, and SGM richness of 3. Meanwhile, only SGM richness increased significantly as a result of rising bulk density (z=2.195, p=0.028) (Table 2-13).

Species richness modeling

Sedge/grass meadow richness

The optimal model for the increase of sedge/grass meadow richness included treatment, elevation, litter, ramet density, and *Typha* leaf height (pseudo R^2_c=0.2480). Planting treatments 2 and 3 increased richness by a log-odds average of 0.3932 (±0.1107 SE, 95% CI: 0.1783 to 0.9974) and 0.3985 (±0.0772 SE, 95% CI: 0.2478 to 0.5506), respectively; increasing elevation enhanced it by a log-odds of 0.0665 (±0.0314 SE, 95% CI: 0.0043 to 0.1275), leaf height increased it by log-odds of 0.1330 (±0.0551 SE, 95% CI: 0.0258 to 0.2416), and decreasing litter and ramet density increased richness by a log-odds of 0.0728 (±0.0288 SE, 95% CI: -0.1299 to -0.0171) and 0.1283 (±0.0466 SE, 95% CI: -0.2209 to -0.0383), respectively (Tables 2-14 and 2-15).

TYPHA

Typha x *glauca*, growing mostly vegetatively from excavated and mounded spoil material, was more abundant at Buttonwood Creek across all resolutions and both years—percent cover (6.3% compared to 3.1% at Buck Pond) (W=33152, p=0.003), ramet density (mean density per plot was 2.56 versus 1.53 at Buck Pond) (W=31422, p=0.06), and mean leaf height (70.2 cm at Buttonwood Creek, as opposed to 49.5 cm at Buck Pond (W=32078, p=0.02)) (Table 2-16). Additionally, there were no indications that a sizable and/or lasting reduction in *Typha* had occurred between 2015 and 2016 at the site, mound, or plot level.

Effect of treatments on Typha

Subplot *Typha* dynamics followed the general sequence, from greatest to least % cover, ramet density, and average leaf height—treatment 3>treatment 1>treatments 2 and 4 (Table 2-17). While subplot treatments 2 (*Typha* cutting with planting) and 4 (*Typha* cutting without planting) generated marked decreases across all *Typha* parameters, treatments 1 and 3 were accompanied by increased *Typha* cover and ramet density. I observed incremental decreases in *Typha* percent cover as a result of subplot treatments at Buck Pond from 2015 to 2016; however, this reduction was not similarly detected at Buttonwood Creek. Additionally, average *Typha* leaf height increased between years at both sites. Lastly, I identified significant differences in all *Typha* measures as the result of subplot treatments, with no-cutting treatments 1 and 3 consistently demonstrating increased *Typha* cover, ramet density, and average leaf height (Figure 2-13).

Effect of elevation, soil moisture, and bulk density on Typha

Reductions in *Typha* appeared to be scale-dependent, as I was able to detect decreases in *Typha* percent cover, ramet density, and leaf height at the site level, although not at the plot level. Neither elevation nor soil moisture wielded a significant influence on plot-level *Typha* measures across years and across sites (Kruskal-Wallis chi-squared=74.216, df=111, p=0.997; Kruskal-Wallis chi-squared=45.016, df=59, p=0.910). Meanwhile, bulk density had a greater effect (albeit not significant) on *Typha* at Buttonwood Creek in the second year (Kruskal-Wallis chi-squared=24.76, df=18, p=0.132). All *Typha* measures at Buttonwood Creek were elevated above those at Buck Pond, owing to decreased elevation and increased soil moisture and bulk density (Table 2-18). Despite this pattern, however, only *Typha* cover was significantly lessened at Buck Pond as a result of greater elevations and reduced soil moisture and bulk density (H=8.88, df=111, p=0.003).

Typha modeling

The optimal model for the reduction of *Typha* percent cover included elevation, treatment, year, and site (pseudo R^2_c=0.6652) (Table 2-19). Decreasing elevation increased *Typha* cover by a log-odds average of 0.2699 (±0.0868 SE, 95% CI: -0.4456 to -0.1023), treatment 2 diminished it by a log-odds of 2.6751 (±0.1195 SE, 95% CI: -2.9190 to -2.4473), treatment 4 reduced it by a log-odds of 2.9538 (±0.1363 SE, 95% CI: -3.2338 to -2.6955), and the effect of year decreased percent *Typha* cover by a log-odds of 0.1862 (±0.0438 SE, 95% CI: -0.2728 to -0.0999) (Table 2-20).

To determine whether abiotic and applied factors more readily influenced *Typha* ramet density or *Typha* cover over time, I generated secondary Poisson GLMMs for each. Analysis of variance between the cover and ramet density models revealed that *in situ* factors, such as elevation, had a greater effect on *Typha* ramet density than on *Typha* percent cover (χ^2=1952.1, p<0.000) (Table 2-21). The effect of year decreased ramet density by a log-odds average of 0.1580 (±0.064 SE, 95% CI: -0.2865 to -0.0299) while elevation decreased it by a log-odds average of 0.2278 (±0.065 SE, 95% CI: -0.3587 to -0.0992) (Table 2-22).

Spearman's rank correlation coefficients (r_s) among richness indices and *Typha* measures are presented as a matrix in Table 2-12. Significant positive correlations were found among plot-level total species richness and *Typha* across the two-year study period. Both *Typha* cover and average *Typha* leaf height had significantly positive correlations with total community richness, respectively (S=1.536e+07, r_s=0.1667, p=0.0002; S=1.513e+07, r_s=0.1789, p<0.000).

DISCUSSION

Hypothesis 1

While my results provide supporting evidence that greater elevations atop the created mounds, particularly those above the 75.35 m (IGLD 85) threshold, contribute to the reduction of aboveground *Typha* biomass in the short term and that elevation plays a primary role in affecting soil physical properties (i.e., soil moisture) of Lake Ontario coastal wetlands amended with mounded spoil, they do not necessarily substantiate the capability of spoil mounds to sustain adequate elevations indefinitely and thereby maintain soil characteristics most suitable to SGM while restricting *Typha*. Ultimately, I was not able to find a lasting means of eliminating *Typha* x *glauca* atop the spoil mounds. However, this outcome does not preclude the value of using mounding to create or recreate beneficial sedge/grass meadow habitat in Lake Ontario coastal marshes, but it indicates that methodologies require fine-tuning. Above all, I suggest that refinements be made in the evaluation of soil physicochemical properties and their interactions with hydrological variables.

I initially hypothesized that the spoil mounds would maintain soil moisture conditions that diminish *Typha* x *glauca*, thus yielding richness equal to or greater than that of remnant sedge/grass meadow communities. I had hoped to create topographic heterogeneity sufficient to generate a cascade of hydrologic conditions capable of mimicking non-equilibrium processes, such as fluctuating hydroperiods, which would thereafter reduce further need for management practices. These subtle elevational/topographical changes over short distances often yield large environmental variations in soil properties (Kongchum *et al.* 2017).

ELEVATION AND SUBSIDENCE

My data indicate that surface elevation of the created spoil mounds was significantly lower at both sites after one year. Triggered by excavation and subsequent dewatering of spoil material, subsidence is the best explanation for diminishing mound elevations. Dynamic subsidence processes—shrinkage, compaction, and oxidation—occur on both microscopic and macroscopic scales within the sediment profile, the rates predicated on interactions with other physicochemical and hydrological forces, such as depth to the water table, mineral and organic matter content, temperature, and management regime, acting across multiple spatial and temporal scales (Ewing and Vepraskas 2006; Nawaz *et al.* 2013).

The breadth of subsidence rates of organic soils (fibrous peats and/or histosols) can vary from 0.18 cm to 0.47 cm/year in organic Indiana mucklands to between 0.30 cm and 0.74 cm/year in Florida Everglades peatlands and 0.6 to 4.0 cm/year in fluvial-enhanced Sacramento-San Joaquin Delta wetlands (Mirza and Irwin 1964; Miller *et al.* 2008), with the most dramatic rates occurring within the first year following drainage. However, the average observed rate of subsidence across both study sites (7.3 cm/year) extended far beyond these ranges. Buttonwood Creek, on average, demonstrated a much greater rate of soil subsidence (9.0 cm/year) and an associated decrease in surface elevation than did Buck Pond (5.6 cm/year).

SOIL MOISTURE

Measurements across two growing seasons reveal that a vast difference in soil moisture was consistently maintained between the two sites, with lower moisture conditions at Buck Pond supporting over 50% survivorship, but only 25% at Buttonwood Creek. While decreasing soil moisture figured highly in species richness gains across both seasons, it did not demonstrate a significant influence on *Typha* growth and development.

Soil moisture is regulated by several abiotic variables, such as sediment type and particle size, both of which directly pertain to organic matter (OM) concentration, an important factor for water-holding capacity (Garssen *et al.* 2014). Consequently, natural sedge meadows with deep organic-rich soils have higher soil moisture contents—between 55.5 to 86.1% vwc at 0-30 cm within the soil profile— than constructed SGMs on mineral substrates (Bremholm 1993; Lawrence *et al.* 2013).

Furthermore, much of the available experimental SGM research has confirmed that a reduction in soil moisture tends to hamper the ability of *Carex* to establish effectively and compete with emergent invasives, such as *Typha* (Wetzel and van der Valk 1998; Silvertown *et al.* 1999; van der Valk *et al.* 1999; Araya *et al.* 2010; Deng *et al.* 2013). van der Valk *et al.* (1999) also found that the probability of establishing sedge meadow species increased with increasing soil moisture. My research supports this principle, but my data also demonstrate its practical limitations, because soil moisture relationships in Lake Ontario coastal wetlands are affected by the influential driver of flooding. While intact SGMs— chiefly those in the upper Great Lakes and/or in the prairie pothole region— are subject to seasonal fluctuations in which water becomes a limiting factor, degraded wet meadows of Lake Ontario are bound by a static regime in which water is not limiting. Thus, Lake Ontario coastal SGM wetlands, unlike many other coastal marshes, are hampered by excessive soil moisture. Because my principal challenge has been to maintain a reasonably moist environment within a precisely defined threshold, I determined that 30% vwc was the optimal soil moisture for *Carex* and other wet meadow taxa. This threshold is supported by Angeloni *et al.* (2006), who found that intact shoreline sedge meadow communities of Lake Huron demonstrate mean soil water contents of 26.6% ±2.03, unlike *Typha* communities that have 80.2% ±4.51 soil moisture content.

Constricted propagule pressure has often been emphasized as the primary driver of regeneration dynamics in sedge meadow marshes; however, my results indicate that Lake Ontario’s static hydrologic regime often undermines other factors affecting community assembly and structure (Galatowitsch and van der Valk 1996). Additionally, my results are more consistent with studies that recognize that the success or failure of a wetland restoration project depends on the synergy of environmental filters such as water regime and the seed bank, as opposed to dispersal limitation alone (Middleton 1999; Wang *et al.* 2013; Wang *et al.* 2017).

SOIL BULK DENSITY

Within lacustrine-lotic systems, particulate organic matter and suspended inorganic sediments accumulate within depositional zones receiving large volumes of sediment from tributaries or in embayments that amass wind-driven material (Wantzen *et al.* 2008a; Larson *et al.* 2013).

My data reveal that Buttonwood Creek had a bulk density somewhat greater than that at Buck Pond. Buttonwood Creek more rapidly accrues mineral sediments via backflooding and material loads delivered from the upstream catchment (Larson *et al.* 2013). In such hydrogeomorphic settings, mixing of lake-lotic water occurs directly in the receiving basin, which has direct implications for sediment discharge rates into the wetland, as well as subsequent macrophyte productivity and organic matter accumulation and decomposition (Makarewicz 1989; Keough *et al.* 1999; Larson *et al.* 2013). Buck Pond, in contrast, accumulates sediments much more slowly as a consequence of its physiographic setting and poor hydraulic connectivity, which requires a cascade of ideal conditions (strong winds and adequate water movement) to mobilize high-density lake sediments into the marsh (Keough *et al.* 1999). In both cases, land use strongly influences flow regimes, transport sediment dynamics, nutrient loading, and resultant biotic communities, with *Typha* remaining a physical impediment to sediment transport into the SGM zone (Makarewicz 1989; Makarewicz and Lampman 1994; Makarewicz *et al.* 2012).

Following Jing *et al.* (2017), I have now begun to consider how flooding effects from stabilization may operate indirectly on SGM vegetation through its effects on sediment balance. This type of sediment disturbance reduces taxa richness in Great Lakes coastal wetlands (Barry *et al.* 2004). I also recognize the scarcity of information regarding sediment dynamics specific to Lake Ontario coastal wetlands and perceive it as a valuable source of information for future management.

Freshwater marshes are generally observed to have reduced bulk densities relative to other wetlands—like mine, often lower than 0.1 g/cm^3—due to a greater quantity of organic matter and reduced mineral sediment (Wang *et al.* 2017). However, profoundly altered relationships between elevation and soil properties, as reflected in greater bulk densities, are often generated by anthropogenic influence. For example, paleoecological studies of northeastern Lake Ontario coastal marshes have suggested that increases in sediment bulk density are a reflection of land-use alteration initiated in the 19th century, which changed wetland structure and contributed to the conversion of wetlands from sedge meadows to *Typha*-dominated systems (Farrell *et al.* 2010). Rippke *et al.* (2010) further suggest that increased inorganic sedimentation acted in concert with lake-level stabilization to support this shift.

Greater bulk densities have also been observed as an unfortunate consequence of marsh restoration, where values far exceed those of natural wetlands they are intended to replace and/or replicate (Galatowitsch and van der Valk 1996; Craft *et al.* 1999; van der Valk *et al.* 1999; Kettenring and Galatowitsch 2011). While bulk densities of natural SGMs occur between 0.16 to 1.34 g/cm^3 at 2 to 25 cm within the soil profile, those of restored marshes are subject to higher average bulk densities between 0.40 to 2.08 g/cm^3 (Galatowitsch and van der Valk 1996; Hogan *et al.* 2004; Deng *et al.* 2013). Even *C. stricta* tussocks demonstrate a trifold increase in bulk density in restored sedge meadows of the Midwest (Lawrence and Zedler 2013).

TYPHA RESPONSE TO ABIOTIC CONDITIONS

My data reveal that bulk density was a major driver of *Typha* on spoil mounds. Increased elevation and reduced soil moisture were capable of reducing *Typha*, but marked decreases occurred only when in tandem with cutting treatments. While *Typha* was consistently more abundant at Buttonwood Creek, no significant reductions were maintained across sites or years.

Prevailing research supports my observation that elevation influences *Typha* development, specifically as it shapes soil moisture regimes (Frieswyk and Zedler 2007). While saturated wetland soils in a hydrologically regulated system can dramatically augment the density of *Typha* ramets, the induction of periodic and/or sustained drought-like conditions can increase root and shoot mortality and thus be an effective strategy to reduce *Typha* dominance (Li, Pezeshki, and Goodwin 2004; Boers *et al.* 2007; Asamoah and Bork 2010). In stabilized regimes, *Typha* cover can be upwards of 80 percent; however, due to its exceptionally wide moisture tolerance, effective field manipulation of soil moisture levels can be difficult except under rare and prolonged drought conditions (Bedish 1967; Boers *et al.* 2007; Asamoah and Bork 2010). Grace (1985) observed complete elimination of *Typha* in soil moisture conditions below 8%. Similarly, Asamoah and Bork (2010) were able to reduce *Typha* root biomass by 50% when moisture levels were held at ≤5% vwc across a span of 12 weeks in a controlled setting, and this reduction was significantly enhanced when levels were fixed at 1.5% vwc. My models also suggest that not only does increasing elevation effectively reduce *Typha* cover and ramet density, but cutting and removal treatments (2 and 4) serve as a viable means to decrease it as well, particularly when performed as a repeated measure across multiple years.

In a similar study, Regan (2016) observed that spoil mounds in coastal wetlands of the St. Lawrence River maintained surface elevations at a maximum of 75.23 m IGLD85, far below the established threshold (75.35 to 75.6 m IGLD85), which were able to generate a marked reduction in *Typha* ramet density. Therefore, even under the best attempts to control *Typha* via drying, its survival and distribution depend in part on other factors, such as substrate type, nutrient availability, and other site-specific factors (Frieswyk and Zedler 2007; Xu *et al.* 2015). On the whole, distribution patterns are formed by the combined effects of a variety of factors, with hydrological condition playing a decisive role. My ability to manufacture overall drier conditions on the spoil mounds to eradicate *Typha* was only somewhat effective. *Typha* ramets consistently emerged seemingly regardless of soil moisture and elevation. However, of note, these ramets were observed to be more or less fully mature clones, indicating that they were likely derived from the original standing biomass that became integrated into the organic mound material during excavation. While not quantified at the site level, few *Typha* seedlings were observed, which may be perceived as a better measure of *Typha* regeneration. However, my results indicate that upslope invasion of *Typha* does not necessarily require a landscape-scale stabilized hydrologic regime, but can act on the most subtle differences in water regime occurring at exceedingly fine spatial scales (Shay *et al.* 1999; Farrell *et al.* 2010; Raulings *et al.* 2010).

Hypothesis 2

My outcomes fully support the preliminary assertion that applied treatments and site conditions significantly influence survivorship atop organic-rich excavated spoil mounds. Abiotic factors contributing most effectively to *Carex* survivorship can be summarized as follows: increased elevation (75.35 to 75.6 m IGLD 85), decreased soil moisture (<30% volumetric water content), and decreased bulk density. While my study demonstrates a considerable range of elevations and soil moistures capable of supporting substantial *Carex* plug survival in novel wetland conditions, special attention should be paid to soil bulk density, as it may yield greater overall influence, especially in wetter conditions and lower physical settings.

I evaluated the effects of biotic and abiotic variables on the survivorship of *Carex* plugs planted atop mounded spoil. Plant survivorship is dependent on a myriad of environmental factors differing at local and regional scales, which in turn, has cascading effects on community composition and other characteristics (Seabloom and van der Valk 2003a/2003b; Aloisio *et al.* 2017). However, my survivorship results illustrate indirect and unpredictable patterns that necessitate a deeper examination of abiotic variables, as well as synergistic interactions with site and species.

After one year, I observed a substantial effect of elevation and soil moisture on plug survivorship, although the manner by which they influenced survival was not altogether clear. My data indicate that the two factors act in concert to influence local niche characteristics, with increased elevation and lower soil moisture content improving soil aeration and thus plant recruitment and survivorship (Vivian-Smith 1997; Crossle and Brock 2002; Mendelssohn and Kuhn 2003; Xu *et al.* 2015; Campbell *et al.* 2016). Despite reports of extensive reduction in initial survivorship in other sedge meadow restoration studies, I observed substantial plug survivorship after one year (Hellsten *et al.* 1996; Yetka and Galatowitsch 1999; Budelsky and Galatowitsch 2000; Boers *et al.* 2007).

Nonetheless, the modes by which planted plugs interacted with abiotic factors to influence survival at either site remain puzzling. My data reveal that survivorship of *C. stricta* and *C. lacustris* plugs was greater in spoil material of lower bulk density. Moreover, planted plugs fared most poorly at the lowest elevations (and higher soil moistures) when corresponding bulk density was high, indicating some interaction with bulk density.

Seedling survivorship, as a general rule, declines with increased soil bulk density. In constructed and/or restored wetlands with organic-rich soils, bulk density is driven by organic matter, specifically fiber content, which is a proxy measure of the degree of decomposition (Verry *et al.* 2011; Bridgham and Ye 2014). With the breakdown of large plant fibers (>0.01 mm) into amorphous organic material (<0.01 mm), soil bulk density increases, in turn diminishing hydraulic conductivity, and substantially reducing both water storage capacity and drainable porosity while facilitating marked water retention (Verry *et al.* 2011). Therefore, as the bulk density of a soil increases, it imparts greater physiological stress on plant tissues, in turn affecting seedling survival (van der Valk and Pederson 1989). Soil bulk density may have a mediating effect on survivorship when other abiotic factors are held constant. This relationship was observed as a divergence in *C. stricta* survivorship. *Carex stricta* persisted in markedly wetter substrate at lower elevations when bulk density was low; however, when bulk density was high, although at equivalent elevations and soil moistures, plugs perished.

My observations also suggest that nutrient or microbial differences in the substrates at the two sites may have influenced plug growth rates and survival. The general conclusion of many Great Lakes coastal wetland studies is that microbial activity increases with increasing organic matter content of sediments (Hill *et al.* 2006). Therefore, rapid plug establishment and growth as a consequence of greater nutrient availability in microbially-rich organic sediments could explain the differences in enhanced *C. stricta* plug survivorship outside of the optimal elevation and soil moisture ranges.

Although not statistically significant, overall greater survivorship occurred at Buck Pond in both treatments relative to Buttonwood Creek. Contrary to my expectations, plots without *Typha* competition (treatment 2) had lower *Carex* survivorship than plots with competition (treatment 3). More specifically, treatment 2 appeared to influence survival at the two sites in divergent ways, expressed as a subtle increase at Buttonwood Creek and a slight reduction at Buck Pond. Treatment 3 affected survivorship in an equal yet opposite manner, encouraging survivorship at Buck Pond while simultaneously reducing it at Buttonwood Creek. Based on my understanding of site dynamics, greater survivorship at Buck Pond in treatment 3 was likely attributable to the treatment's facilitation of increased shading via *Typha*, which subsequently increased soil moisture to counter water loss via lower bulk density and increased substrate porosity.

During the first growing season, planted *C. stricta* plugs demonstrated a slight survivorship advantage over *C. lacustris* across treatments and across sites, with live *C. stricta* generally favored by treatment 2, and *C. lacustris* survivorship slightly favored by treatment 3. These patterns may illustrate the more pronounced sensitivity of *C. stricta* to competition with *Typha* x *glauca*, and the inverse for *C. lacustris*.

Carex stricta has been shown to be restricted by competition for light at moisture extremes (Wetzel and van der Valk 1998; Budelsky and Galatowitsch 2004; Wilcox *et al.* 2008). Alternatively, this may merely demonstrate the relative importance of elevation and related soil moisture patterns and thus reflect their effect on niche gradients relevant to each of the study taxa (Moeslund *et al.* 2013).

Carex lacustris survivorship within the upper elevations was substantially greater than at the lower elevations, averaged across site and regardless of treatment. At Buck Pond, the greatest mortality occurred in plots at elevations below the 75.35 to 75.60 m IGLD 85 optimum, despite the greater tolerance of this taxon for augmented soil moisture and water depth.

Modeling supported my conclusion that a suite of variables affected plug survivorship on spoil mounds, with elevation being greater and year being slightly lesser in importance. Species as a factor validates the importance of selecting appropriate taxa for *in situ* environmental conditions. *Carex stricta* demonstrated more resilience under the variable conditions at my coastal wetland sites. This pattern supports my conclusion that *C. stricta* plugs are more likely to persist in environments that diminish competitive effects.

Hypothesis 3

My results suggest that active management (revegetation of native taxa via planting and seeding in conjunction with invasive removal, especially over multiple years) and the manufacture of increased elevations increase species richness, and reduce *Typha* aboveground biomass. I also found that the factors contributing most effectively to sedge/grass meadow species richness were nearly identical to those for survivorship, specifically increased elevation, reduced soil moisture, and reduced bulk density, although occasionally the direction of their effect differed or was unclear. In the end, I was able to generate a floristically-rich plant community atop the spoil mounds that surpassed that of degraded, remnant sedge/grass meadows of Lake Ontario. *Typha* cover and leaf height had significantly positive correlations with total community richness.

Similar to Hudon *et al.* (2005), I observed a large increase in native species richness across all years of my study, with most taxa being perennial obligate and facultative wet species. This increase is in direct contrast to some reports of spoil mounds remaining mostly unvegetated after multiple years, even when seeded, with resultant plant communities primarily supporting ruderal (e.g., *Erechtites hieraciifolius*) and/or invasive taxa (e.g., *Typha*, *Lythrum salicaria*, and *Phalaris arundinacea*), and few native species (Hartley 2000; Regan 2016).

While not the primary focus of my study, seeded sedge/grass meadow taxa undoubtedly influenced subsequent richness atop the created mounds two years after implementation. While only 9 of 13 seeded species were readily observed in the first season following broadcasting, a second round of *in situ* cold, moist stratification appeared to facilitate increased germination by stratification-dependent taxa (e.g., *Carex lacustris*), thus contributing to improved richness in 2016. As noted by Leck and Schutz (2005) and Kettenring and Galatowitsch (2007), many temperate carices require a substantial period of cold to break dormancy and successfully germinate. Their research supports my practice of *in situ* seed stratification, suggesting it is a viable means to encourage rapid establishment of *Carex* while concomitantly reducing invasive species (Kettenring and Galatowitsch 2007). Additionally, similar to Wilcox and Healy (2016), I suggest that active transplantation, as well as seeding, can be a fairly accurate predictor of the forthcoming plant community, with clear consideration given to environmental factors, such as soil moisture, that conform to or differ from species requirements, as responses invariably differ among species and with habitat.

Overwhelmingly, regulated lake systems have fewer species than predicted under natural regimes (Hill *et al.* 1998). However, in my study within a regulated system, median native and sedge/grass meadow species richness atop the mounds increased with elevation. As described by Vestergaard (1998), topographic heterogeneity can enhance plant species richness in coastal meadows, which facilitates the release of plants from submergence/moisture stress (Wang *et al*. 2014). This pattern follows Boers *et al*. (2007) and Nishihiro *et al.* (2004), who found that seedling emergence, aboveground biomass, and richness of *C. stricta* and other lakeshore species were positively correlated with elevation. Additionally, Yuan *et al.* (2017) recognized that elevation was also a useful predictor of lakeshore wet meadow *Carex* development and distribution, likely as it mediated soil moisture levels.

The influence of hydrological conditions on floristic richness has been studied extensively, although often using controlled experiments that fail to capture variable conditions fully. At the site scale, there is a pronounced correlation between environmental (and topographic) heterogeneity and species richness as mediated by soil moisture availability and frequency of inundation (Vivian-Smith 1997; Silvertown *et al.* 1999; Moeslund *et al.* 2013). Customarily, and as was observed in my study, while richness indices are positively correlated with soil moisture, they are negatively correlated with extended hydroperiods (Nilsson and Keddy 1988; Nishihiro *et al.* 2004; van Geest *et al.* 2005; Raulings *et al.* 2010; Xu *et al.* 2015).

Research clearly suggests that when exposed to frequent and long-term flooding, local and landscape-scale native richness is often low due to dominance by *Typha* (Boers *et al.* 2007; Boers and Zedler 2008). I observed not only that total species richness, which included ruderals and other exotic and invasive taxa, was positively correlated with *Typha* cover and leaf height, but an increase in SGM richness was accomplished by both decreasing litter and decreasing *Typha* ramet density (in concert with increased elevation).

Modeling indicated variable relationships with soil moisture at different resolutions of richness that changed subtly with soil bulk density. For example, I observed that while total richness is significantly influenced by increased soil moisture, increased moisture together with diminished bulk density enhance native richness. This influence results from the fact that most of these taxa are ruderals and invasives—species well adapted to environmental disturbance. Meanwhile, only SGM richness increased significantly as a result of both elevated moisture and bulk density. These SGM communities in Lake Ontario coastal wetlands now persist under what appear to be highly altered sediment regimes with much greater bulk densities. These taxa (primarily *Carex* and *Juncus*) are more readily adapted to variable hydrology and substrates of more variable composition and structure, as demonstrated by Xiaolong *et al.* (2014), who observed *Carex* persisting in soils of enhanced bulk densities (0.82 to 1.14 g/cm^3) in coastal wetlands of Poyang Lake. For other native species, however, I speculate that changes in bulk density caused by the collapse of soil structure in dewatered spoil material may influence the decrease in species richness by altering the moisture regime.

Likewise, my models suggest that subplot treatments 2 (planting and *Typha* removal) and 3 (planting, but no mechanical removal of *Typha*) increased species richness, indicating that active revegetation successfully supports richness, as well as prevents some amount of competitive dominance by *Typha*. These outcomes mirror those of Carlson *et al.* (2009), who found that invasive cutting and removal treatments had a significant effect on resultant field species richness in a drowned river mouth coastal wetland of Lake Erie, although, too, they determined that saturated soils confounded their results. Furthermore, Mitchell *et al.* (2011) found that species richness significantly decreased with increasing *Typha* stand age as a result of increasing ramet density and organic matter accumulation, indicating that the effect of diminished richness may not become fully apparent until up to 15 years following initial invasion.

Despite these advances, however, I anticipate that, deprived of persistent amendments and with ongoing mound subsidence, this rich plant community will succumb to conditions that motivated my initial actions, likely in a matter of years.

CONCLUSION AND MANAGEMENT RECOMMENDATIONS

Lakes are complex ecosystems composed of distinct habitats coupled by biological, physical, and chemical processes. When lacustrine habitats are severely impacted, effective lakeshore sedge/grass meadow recovery can be highly problematic because establishment and survival of matrix species are strongly yet enigmatically influenced by hydrology and other environmental factors, particularly in novel wetland conditions (Steed and DeWald 2003; Budelsky and Galatowitsch 2004; Leck and Schutz 2005).

In my study, I attempted to manipulate spoil mound elevations in a manner that would allow me to reduce soil moisture predictably to levels capable of supporting SGM community matrix species, as described by previous research (Ewing 2002). My results indicate that sedge/grass meadow seedling survivorship and richness of resident Lake Ontario coastal wetland communities are shaped by a myriad of factors operating at the site scale. In particular, my study experimentally illustrates how species selection combined with experimental manipulation of microsite topographic variables (e.g., elevation of created spoil berms) significantly influence species assemblages.

Similar to Kongchum *et al.* (2017), I observed that the ability of created marshes to meet or exceed their targeted objectives is primarily a function of project design and post-construction surface elevations. For resilience of excavated spoil mounds in Lake Ontario coastal marshes, this means following very conservative engineering specifications (i.e., anticipating an overall average subsidence of approximately ≥7 cm per year, depending on the depth of organic material exposed above the water surface) (Riley 1981). Where no local data on subsidence rates are available, this may mean allowing for initial subsidence estimated at 25 to 35 percent of the designed depth (Riley 1981). However, increasing surface elevations far above the optimum elevation in an attempt to avoid the need for future applications has been cautioned against, as studies indicate that this ultimately delays a vegetation community's ability to achieve stability (Stagg and Mendelssohn 2011). Thereafter, as a rule of thumb, plugs should not be planted at elevations at which subsidence will result in soil moistures of greater than 30% for more than a month during the first growing season. As stated by Budelsky and Galatowitsch (1999), merely replicating environmental conditions found in natural sedge meadows does not appear adequate for the recolonization of sedges, meaning that one should bear in mind that environmental conditions suitable for adult plants are not necessarily optimal for establishment.

Similar to Craft *et al.* (1999), I agree that an important attribute of wetland soil development is the accumulation of organic matter and the concomitant decrease in bulk density resulting from the low particle density of organic matter. I observed first-hand how plugs had the greatest survivorship when planted in substrate with reduced bulk density. The difference in plant growth between low and high bulk density was greatest in the lowest elevations, suggesting that the use of high organic substrate will slightly mediate the other environmental factors, thereby facilitating *C. stricta* and *C. lacustris* plug establishment and survival.

In addition, I recommend, if extensive drawdowns as a result of water-level fluctuations cannot be estimated, or as in my case cannot be achieved, it is better to "err on the dry side" by 1) oversizing original mound/plot elevations and by 2) planting *Carex* plugs at slightly higher elevations. Raulings *et al.* (2010) have suggested that decreasing water depths in wetlands having substantial microtopographical variation may be an effective means to shift variable water regimes to internal areas characterized by shallow water and more frequent exposure of underlying substrate. This arrangement may only be suitable on a case by case basis and is likely not feasible in Lake Ontario coastal marshes.

Research tracking restoration trajectories indicates that any form of ameliorative action is often more effective than a no action option but often does not result in complete recovery (Rey Benayas *et al.* 2009). This trend was unquestionably observed in my study, as well as in a survey of wet meadow restoration projects across Europe, wherein projects increased species richness to only 10-16% of the regional species pool (Klimkowska *et al.* 2007). Also, because time since invasion is an important component of biological invasions, I agree with Mitchell *et al.* (2011), who suggest early intervention with a focus on newer invasions to reduce the likelihood of irreversible impacts to community richness. My sites, unfortunately, are an order of magnitude older, corroborating the intensity and the uncertainty of protracted efforts.

Although many studies have been performed on the effect of elevation, soil moisture, or bulk density on plant survival and growth of sedge/grass meadow species, less attention has been paid to the synergistic effects of the multiple stressors, particularly within the context of novel wetland conditions. With novel conditions come novel questions, and I recognize that commonly understood trends and trajectories may not necessarily apply to this system. For example, a shift in the sediment dynamics of these coastal wetlands may have occurred or be in the process of occurring. Such a shift will likely take place to the ecological detriment of the system, leading to additional modifications of hydrologic and soil properties that have a greater influence on the distribution and fate of materials related to water quality. More research is necessarily required.

Finally, as systems do not always respond in a predictable manner, awareness of the mechanisms of vegetation change maximizes the possibility of more desirable states (Suding *et al.* 2004; Briske *et al.* 2006; Zweig and Kitchens 2009). This awareness provides additional, critical information for restoration management decisions. Identifying the possible states and pathways of change can be used to predict restoration success or the possibility of hysteresis (i.e., upon passing an individually-defined threshold, a system may follow a different path for recovery than the initial trajectory of change, yielding outcomes divergent from historical values)

(Scheffer *et al.* 2001, 2003, 2009; Suding and Hobbs 2009; van de Leemput *et al.* 2017). Therefore, below I close with a consideration of multiple stable states in Lake Ontario coastal wetlands.

FURTHER CONSIDERATIONS

Multiple stable states (MSS) occur in coastal wetlands as a result of autogenic feedbacks, yielding phase shifts (Moffett *et al.* 2015). Such regime-shift phenomena have been explored in many coastal ecosystem types, including tidal coastal wetlands, freshwater lakes, and coral reefs (Moffett *et al.* 2015), but no literature yet exists on MSS in inland freshwater coastal wetlands, such as those in the Great Lakes. Multiple stable state theory is built on a non-equilibrium model that posits that an ecosystem is capable of shifting in vegetative composition and/or function based on a series of factors—e.g., dominant, self-reinforcing feedbacks associated with extreme and/or persistent disturbance, the physicochemical environment, and recruitment patterns (May 1986; Scheffer *et al.* 2001, 2003, 2009; Middleton 2002).

The ultimate implication for a coastal wetland found to conform to MSS theory is that, once degraded, it would be very hard to restore and may require a greater magnitude of ameliorative action for any degree of effective recovery (Scheffer *et al.* 2001; Moffett *et al.* 2015). While not the express focus of this research, the prospect of MSS functioning in regulated coastal *Typha*-sedge/grass meadow systems appears plausible. My speculation follows that of Hill *et al.* (1998), who asked—"Are these differences in vegetation simply the result of differences between hydrological regimes of regulated and unregulated systems or are there irreversible changes associated with the submergence of the plant community by damming? Can hydrological regimes be managed so that diverse shoreline communities establish even along dammed lakes?" While being a global practice, our understanding of the role of water-level regulation in driving regime shifts in lacustrine systems remains limited (Wantzen *et al.* 2008b; Kong *et al.* 2017).

While not explicitly exploring coastal wetland dynamics, Kong *et al.* (2017) did describe the occurrence of regime shifts in Lake Chaohu, China as the result of stabilized water levels and eutrophication using paleolimnological records, environmental data, and system modeling. Similar to Lake Ontario, Lake Chaohu has been regulated since the early 1960s and has experienced enhanced nutrient loading and a severe increase in organic matter for over half of a century (Kong *et al.* 2017). After 1963, the hydrological environment and trophic status of the lake changed sharply, showing an abrupt decrease in vegetation and increase in nutrient levels, which became progressively more severe over time. Within Lake Chaohu coastal wetlands, floristic restoration diminished turbidity and eutrophication, as well as supported enhanced wetland structure; however, these changes remained restricted to individual marshes (Xu *et al.* 1999).

The ecosystem behavior we are currently observing in Lake Ontario coastal wetlands—i.e., chronic internal feedbacks as the result of lake-level regulation, nutrient loading, *Typha* invasion, and shifting sediment dynamics—may very well represent a similar approaching critical threshold, beyond which standard management efforts may prove ineffectual (Suding *et al.* 2004; Kovalenko *et al.* 2014). Thus, it is advisable to consider isolating the associated parameters typically corresponding to transitioning ecosystem dynamics and initiate preliminary investigations.

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TABLES

Table 2-1. Sedge/grass meadow species seed mix for upper mound elevations at Braddock Bay WMA study sites.

Scientific Name	Common Name	Ecotype	Proportion (%)
Carex comosa	Longhair sedge	PA	15
Carex vulpinoidea	Fox sedge	PA	20
Carex stipata	Awlfruit sedge	PA	6
Carex crinita	Fringed sedge	PA	6
Carex lurida	Shallow sedge	PA	10
Carex lupulina	Hop sedge	PA	6
Carex stricta	Tussock sedge	PA	1
Calamagrostis canadensis	Bluejoint	PA	1
Juncus effusus	Common rush	PA	5
Verbena hastata	Swamp verbena	PA	5
Eupatorium perfoliatum	Common boneset	PA	5
Asclepias incarnata	Swamp milkweed	PA	5
Carex lacustris	Lake sedge	PA	15

Table 2-2. Soil bulk density (BD, g/cm^3) and mean elevation (m IGLD 85) of soil samples collected from experimental spoil mounds at Buttonwood Creek (BC) and Buck Pond (BP) in 2016.

Mound	BD	Elevation
BC-M1	0.111	75.145
BC-M2	0.2	75.141
BC-M3	0.242	75.463
BC-M4	0.098	75.434
BC-M5	0.082	75.288
BC-M6	0.071	75.17
BC-M7	0.072	75.186
BP-M1	0.122	75.593
BP-M2	0.133	75.506
BP-M3	0.123	75.505
BP-M4	0.085	75.421
BP-M5	0.081	75.587
BP-M6	0.094	75.663
BP-M7	0.076	75.601
BP-M8	0.071	75.32
BP-M9	0.043	75.524
BP-M10	0.051	75.607
BP-M11	0.05	75.453
BP-M12	0.065	75.364
BP-M13	0.044	75.547

Table 2-3. Summary of plug survivorship by site (BC=Buttonwood Creek, BP=Buck Pond), *Carex* species (CALA=*Carex lacustris*, CAST=*C. stricta*), treatment, and elevation (m IGLD 85) following a full growing season. N/n=quantity of extant individuals; decimal=proportion of extant individuals. For treatment details, see methods. Bold numbers indicate optimal values or ranges for treatments and elevations.

	BC					**BP**				
	Σ (N=360)	**Species** (n=180)		**Treatment** (n=180)		**Σ** (N=360)	**Species** (n=180)		**Treatment** (n=180)	
Elevation		**CALA**	**CAST**	**2**	**3**		**CALA**	**CAST**	**2**	**3**
ALL	252 (0.7)	127 (0.71)	125 (0.7)	130 (0.72)	122 (0.68)	280 (0.78)	130 (0.72)	150 (0.83)	135 (0.75)	145 (0.81)
>75.35 to 75.6	80 (0.22)	37 (0.21)	43 (0.24)	43 (0.24)	37 (0.21)	237 (0.66)	116 (0.64)	131 (0.73)	120 (0.67)	127 (0.71)
≤75.35	172 (0.48)	90 (0.5)	82 (0.46)	87 (0.48)	85 (0.47)	33 (0.09)	14 (0.08)	19 (0.11)	15 (0.08)	18 (0.1)

Table 2-4. Summary of plug survivorship (n) by site (BC=Buttonwood Creek, BP=Buck Pond), *Carex* species (CALA=*Carex lacustris*, CAST=*C. stricta*), elevation (m IGLD 85), soil bulk density (g/cm^3 ±SE), and soil moisture (% volumetric water content ±SE) following a full growing season. Bold numbers indicate optimal values or ranges.

	BC				**BP**			
Elevation	**Species**	**n**	**BD**	**SM**	**Species**	**n**	**BD**	**SM**
>75.35 to 75.6	CAST	43	0.157 ± 0.027	24.593 ± 4.378	CAST	131	**0.082 ± 0.006**	**23.621 ± 2.529**
	CALA	37			CALA	116		
≤75.35	CAST	82	**0.125 ± 0.013**	74.227 ± 3.546	CAST	19	0.089 ± 0.015	45.584 ± 6.390
	CALA	90			CALA	14		

Table 2-5. Candidate model(s) selection table for plug survivorship with estimated parameters and estimated variances; models ranked by AIC$_c$(x); base model: survivorship=species + elevation (E) + bulk density (BD) + (mound/plot); df=degrees of freedom, logLik=logLikelihood, AIC$_c$=corrected Aikake Information Criterion, Δ_i=delta AIC, w=Aikake weight, p-value (Pr (>Chisq), and standard deviation (Std. Dev.). Bold numbers indicate significance and model(s) with the greatest explanatory power.

Fixed Effects

Model	BD	E	Species	df	logLik	AIC$_c$	Δ_i	w	R^2_c
7	**-**	**0.2693**	**+**	**5**	**-472.199**	**954.5**	**0.00**	**0.724**	**0.0784**
8	-0.038	0.2655	+	6	-472.147	956.4	1.93	0.276	
NULL	-	-	-	4	-477.91	961.8	Pr(>Chisq) =**0.009**		

Random Effects

Model	Group Factor	Variance	Std. Dev.	# Observations	# Groups
7	(mound)	0.0745	0.273	720	20
7	(plot)	0.1031	0.321	720	60
8	(mound)	0.0723	0.269	720	20
8	(plot)	0.1039	0.322	720	60
NULL	(mound)	0.0827	0.288	720	20
NULL	(plot)	0.1555	0.394	720	60

Table 2-6. Estimated generalized linear mixed model (GLMM) regression parameters, standard errors (SE), *z*-values, p-values (Pr (>|*z*|)), and 95% confidence intervals (CI) for plug survivorship, a binomial logistic GLMM resulting from modeling; model: survivorship=species$_{\mathrm{CAST}}$ + elevation + (mound/plot). Bold numbers indicate significance.

Model Parameter	**Estimate**	**SE**	***z* value**	**Pr (>|*z*|)**	**95% CI**
Intercept	0.2806	0.1341	2.093	**0.0363**	0.0120 to 0.5522
Species$_{\mathrm{CAST}}$	0.3452	0.1575	2.192	**0.0284**	0.0371 to 0.6554
Elevation	0.2655	0.1039	2.556	**0.0106**	0.0615 to 0.4783

Table 2-7. All taxa growing atop experimental dredge spoil mounds at Buttonwood Creek and Buck Pond in 2015 and 2016; *=exotic taxa; †=planted taxa; ††=seeded taxa.

2015 (N=81; native=64; exotic=17)	**2016 (N=128; native=98; exotic=30)**
Achillea millefolium	*Acer saccharinum*
Acer saccharinum	*Agrostis gigantea**
Apios americana	*Alisma subcordatum*
Apocynum cannabinum	*Alisma triviale*
Asclepias incarnata†	*Apios americana*
Atriplex prostrata	*Apocynum cannabinum*
Bidens cernua	*Artemisia vulgaris**
Bidens frondosa	*Asclepias incarnata*†
Boehmeria cylindrica	*Bidens cernua*
Calamagrostic canadensis†	*Bidens frondosa*
Campanula aparinoides	*Boehmeria cyclindrica*
Carex comosa†	*Bolboschoenus fluviatilis*
Carex lacustris†/††	*Brassica* spp.*
Carex spp.	*Calamagrostis canadensis*†
Carex stipata†	*Calystegia sepia**
Carex stricta†/††	*Campanula aparinoides*
Cephalanthus occidentalis	*Carex comosa*†
*Chamaescye maculata**	*Carex crinita*†
Cicuta bulbifera	*Carex lacustris*†/††
Comarum palustre	*Carex lupulina*†
Cornus amomum	*Carex pseudocyperus*
Cornus sericea	*Carex scoparia*
Cyperus diandrus	*Carex* spp.
Cyperus spp.	*Carex stipata*†
Cyperus strigosus	*Carex stricta*†/††
Decodon verticillatus	*Carex vulpinoidea*†
*Echinochloa crus-galli**	*Cephalanthus occidentalis*
Eleocharis spp.	*Chamerion angustifolium* ssp. *circumvagum*
Elymus glaucus	*Cicuta bulbifera*
Epilobium coloratum	*Cirsium arvense**
*Erechtites hieraciifolius**	*Comarum palustre*
Eupatorium perfoliatum†	*Conyza canadensis*
Fragaria virginiana	*Cornus amomum*
Galium spp.	*Cuscuta gronovii*

Galium trifidum	*Cyperus diandrus*
*Hydrocharis morsus-ranae**	*Cyperus esculentus*
Impatiens capensis	*Cyperus* spp.
Juncus effusus[†]	*Daucus carota**
Juncus spp.	*Decodon verticillatus*
Juncus tenuis	*Eleocharis obtusa*
Lathyrus palustris	*Eleocharis palustris*
Leersia oryzoides	*Eleocharis* spp.
*Lolium perenne**	*Elymus virginicus*
Lycopus americanus	*Epilobium coloratum*
Lycopus uniflorus	*Epilobium leptophyllum*
Lysimachia thyrsiflora	*Epilobium parviflorum**
*Lythrum salicaria**	*Erechtites hieraciifolius**
Mentha arvensis	*Eupatorium perfoliatum*[†]
Oxalis spp.	*Euthamia graminifolia*
Persicaria amphibia	*Festuca rubra**
Persicaria lapathifolia	*Festuca* spp.*
*Persicaria maculosa**	*Fragaria virginiana*
Persicaria punctate	*Fraxinus pennsylvanica*
Persicaria virginiana	*Galium trifidum*
*Phalaris arundinacea**	*Geum canadense*
*Plantago major**	*Hydrocharis morsus-ranae**
Pontederia cordata	*Hypericum punctatum*
Populus deltoides	*Impatiens capensis*
Rhus typhina	*Iris versicolor*
Rosa palustris	*Juncus effusus*[†]
*Rumex crispus**	*Juncus* spp.
Rumex orbiculatus	*Juncus tenuis*
Sagittaria latifolia	*Lathyrus palustris*
Salix eriocephala	*Leersia oryzoides*
Salix spp.	*Lolium perenne**
Schoenoplectus tabernaemontani	*Lonicera* spp.*
Scutellaria galericulata	*Lycopus americanus*
Solidago canadensis	*Lycopus uniflorus*
Sparganium eurycarpum	*Lysimachia ciliata*
*Stellaria media**	*Lysimachia nummularia**
*Taraxacum officinale**	*Lysimachia* spp.
Thelypteris palustris	*Lysimachia terrestris*
Triadenum fraseri	*Lysimachia thyrsiflora*
Typha x *glauca**	*Lythrum salicaria**

Urtica dioica	*Mentha arvensis*
Verbena hastata†	*Myrica* spp.
*Veronica officinalis**	*Onoclea sensibilis*
Vitis riparia	*Oxalis* spp.
	Oxalis stricta
	Persicaria amphibia
	Persicaria hydropiper
	Persicaria lapathifolia
	Persicaria punctata
	Persicaria sagittata
	*Phalaris arundinacea**
	*Phleum pratense**
	*Phragmites australis**
	Pilea pumila
	*Plantago major**
	Pontederia cordata
	Populus deltoides
	Potentilla simplex
	*Ranunculus acris**
	Ranunculus spp.*
	Rhus typhina
	*Rumex crispus**
	Rumex orbiculatus
	Sagittaria latifolia
	*Salix fragilis**
	Salix myricoides
	Salix nigra
	Salix spp.*
	Schoenoplectus tabernaemontani
	Scirpus atrovirens
	Scirpus cyperinus
	Scutellaria galericulata
	Scutellaria lateriflora
	Sisyrinchium spp.
	*Solanum dulcamara**
	Solidago canadensis
	Solidago spp.
	Sparganium eurycarpum
	Stachys palustris
	Symphyotrichum puniceum

	Symphyotrichum spp.
	_Taraxacum officinale_*
	Thelypteris palustris
	Triadenum fraseri
	_Trifolium hybridum_*
	Typha latifolia
	Typha x _glauca_*
	Verbena hastata[†]
	Verbena urticifolia
	Veronica anagallis-aquatica
	Viburnum dentatum
	Viola spp.*
	Vitis riparia
	_Zelkova serrata_*

Table 2-8a. Total, native, and SGM species richness in 2015 and 2016; SGM=sedge/grass meadow.

	Exotic	Invasive	Native	SGM	Total
2015	17	5	64	9	81
2016	30	8	98	14	128
All Years	35	8	108	14	140

Table 2-8b. Species richness by wetland indicator status in 2015 and 2016.

	OBL	FACW	FACU	FAC	UPL	N/A	Total
2015	34	14	11	11	-	11	81
2016	54	26	13	14	2	19	128
All Years	54	27	18	19	2	23	

Table 2-8c. Species richness by growth form and native status in 2015 and 2016.

	Graminoids	Ruderals	Forbs	Woody	Emergent	Ferns
2015 Native	16	2	33	8	4	1
Exotic	4	5	5	1	2	0
2016 Native	26	1	51	11	7	2
Exotic	7	12	5	4	2	0
All Years	39	18	84	18	11	2

Table 2-9. Sedge/grass meadow marsh taxa growing atop experimental dredge spoil mounds at Buttonwood Creek and Buck Pond in 2015 and 2016; †=planted taxa, ††=seeded taxa.

2015 (n=9)	**2016 (n=14)**
Calamagrostic canadensis[††]	*Calamagrostis canadensis*[††]
Carex comosa[††]	*Carex comosa*[††]
Carex lacustris[†/††]	*Carex crinita*[††]
Carex spp.	*Carex lacustris*[†/††]
Carex stricta[†/††]	*Carex lupulina*[††]
Carex stipata[††]	*Carex pseudocyperus*
Juncus effusus[††]	*Carex scoparia*
Juncus spp.	*Carex* spp.
Juncus tenuis	*Carex stipata*[††]
	Carex stricta[†/††]
	Carex vulpinoidea[††]
	Juncus effusus[††]
	Juncus spp.
	Juncus tenuis

Table 2-10. Spearman's rank correlation coefficients (r_s) among richness indices and *Typha* measures for treatment subplots; S=Spearman's rank statistic. Bold numbers indicate significant correlations.

Richness Index	***Typha* % cover**	***Typha* ramet density**	***Typha* leaf height (cm)**	**S**	***p*-value**
Total	0.1667	-	-	15360000	**0.0002**
	-	0.1117	-	16372000	**0.0143**
	-	-	0.1789	15134000	**8.108e-05**
Native	0.0585	-	-	17353000	0.2004
	-	0.0079	-	18286000	0.8629
	-	-	0.0693	17155000	0.1296
SGM	0.0365	-	-	17758000	0.4244
	-	0.0011	-	18412000	0.9815
	-	-	0.0591	17343000	0.1964

Table 2-11. Summary of total, native, and sedge/grass meadow (SGM) richness by subplot treatment atop experimental spoil mounds at Buttonwood Creek (BC) and Buck Pond (BP).

Site Subplot Treatment	Total Richness	Native Richness	SGM Richness
2016			
BC 1	12	10	4
BC 2	12.5	10	4
BC 3	13.5	10	4
BC 4	12	10	4
BP 1	10.5	7.5	2
BP 2	11	8.5	3.5
BP 3	12	8.5	4
BP 4	10	7	2
2015			
BC 1	8	6	2
BC 2	8	7	3
BC 3	8	6.5	3
BC 4	8	7	3
BP 1	6	4	1
BP 2	6	5	2.5
BP 3	7	5.5	3
BP 4	5	4	1

Table 2-12. Effect of subplot treatment on richness (Overall KW=overall Kruskal-Wallis comparison among treatments; MultiC=multi comparison between KW significant treatment pairs); SGM=sedge/grass meadow. Only significant results are displayed.

Richness Index	**Pairs**	**df**	**χ^2**	***p*-value**
Total Richness		**Observed difference**	**Critical difference**	
Overall KW	1-4	3	14.137	<0.003
MultiC KW	3-4	65.996	47.244	0.0015
Native Richness				
Overall KW	1-4	3	15.187	<0.002
MultiC KW	1-2	49.375	47.244	0.0375
	1-3	57.229		0.0073
SGM Richness				
Overall KW	1-4	3	59.452	7.7e-13
MultiC KW	1-2	97.221	47.244	2.9e-07
	1-3	106.958		1.2e-08
	2-4	82.083		1.0e-05
	3-4	91.821		5.3e-07

Table 2-13. Estimated generalized linear mixed model (GLMM) regression parameters, estimates, standard errors (SE), *z*-values, p-values (Pr (>|*z*|)), and 95% confidence intervals (CI) for the effect of soil moisture (SM, % volumetric water content) and bulk density (BD, g/cm^3) on 3 indices of species richness; model: Richness=SM + BD + (plot). SGM = Sedge/grass meadow. Degrees of freedom (df), corrected Aikake Information Criterion (AIC$_c$), logLikelihood (logLik), deviance, chi-squared test statistics (χ^2), and χ^2 p-value (Pr (>χ^2) are provided to display favored models. Bold numbers indicate model(s) with the greatest explanatory power.

Richness Index	**Environmental Parameter**	**Estimate**	**SE**	***z* value**	**Pr(>$\lvert z \rvert$)**	**95% CI**
Model	***df***	***AIC$_c$***	***logLik***	***deviance***	***χ^2***	***Pr(>χ^2)***
Total	Intercept	2.3501	0.0601	39.11	<2e-16	2.229 to 2.468
	SM	0.2427	0.0830	2.92	0.00346	0.077 to 0.408
	BD	0.0904	0.3884	0.23	0.81593	-0.684 to 0.864
Total	*4*	*1196.80*	*-594.4*	*1188.80*	-	-
Native	Intercept	2.0395	0.0693	29.439	<2e-16	1.899 to 2.175
	SM	0.3869	0.0952	4.065	<4.8e-05	0.198 to 0.578
	BD	-0.1763	0.4482	-0.393	0.694	-1.073 to 0.714
Native	***4***	***1117.88***	***-554.9***	***1109.88***	***78.925***	***< 2.2e-16***
SGM	Intercept	0.9812	0.0939	10.454	<2e-16	0.797 to 1.164
	SM	0.2453	0.1286	1.908	0.0564	-0.008 to 0.497
	BD	1.2702	0.5788	2.195	0.0282	0.119 to 2.402
SGM	***4***	***860.89***	***-426.5***	***852.89***	***256.99***	***< 2.2e-16***

Table 2-14. Candidate model(s) selection table for sedge/grass meadow marsh (SGM) richness with estimated parameters and estimated variances; models ranked by AIC$_c$(x); base model: SGM richness=treatment (T) + elevation (E) + litter + *Typha* ramet density (RD) + *Typha* leaf height (LH) + *Typha* cover (C) + (site); df=degrees of freedom, logLik=logLikelihood, AIC$_c$=corrected Aikake Information Criterion, Δ_i=delta AIC, w=Aikake weight, p-value (Pr (>Chisq), and standard deviation (Std. Dev.). Bold numbers indicate significance and model(s) with the greatest explanatory power.

Fixed Effects

Model	T	E	Litter	RD	LH	C	df	logLik	AIC$_c$	Δ_i	w	R^2_c
56	**+**	**0.067**	**-0.073**	**-0.128**	**0.133**		**9**	**-810.3**	**1639.0**	**0**	**0.635**	**0.248**
64	+	0.066	-0.077	-0.169	0.119	0.054	10	-809.8	1640.1	1.1	0.365	
NULL	-	-	-	-	-	-	2	-840.0	1684.0	Pr(>Chisq)		**<1.9e-10**

Random Effects

Model	Group Factor	Variance	Std. Dev.	# Observations	# Groups
56	(site)	0.0548	0.234	480	2
64	(site)	0.0532	0.2306	480	2
NULL	(site)	0.0345	0.1857	480	2

Table 2-15. Estimated generalized linear mixed model (GLMM) regression parameters, standard errors (SE), *z*-values, p-values (Pr (>|*z*|)), and 95% confidence intervals (CI) for sedge/grass meadow matrix taxa richness, a Poisson GLMM described by the equation: SGM richness=treatment$_2$ + treatment$_3$ + elevation - litter - *Typha* ramet density + *Typha* leaf height + (site). Bold numbers indicate significance.

Model Parameter	**Estimate**	**SE**	***z* value**	**Pr (>|*z*|)**	**95% CI**
Intercept	0.8131	0.1805	4.505	**6.6e-06**	0.2312 to 1.3948
Treatment$_2$	0.3932	0.1107	3.553	**0.0004**	0.1783 to 0.9974
Treatment$_3$	0.3985	0.0772	5.162	**2.4e-07**	0.2478 to 0.5506
Elevation	0.0665	0.0314	2.115	**0.03**	0.0043 to 0.1275
Litter	-0.0728	0.0288	-2.531	**0.01**	-0.1299 to -0.0171
***Typha* ramet density**	-0.1283	0.0466	-2.755	**0.006**	-0.2209 to -0.0383
***Typha* leaf height**	0.1330	0.0551	2.417	**0.02**	0.0258 to 0.2416

Table 2-16. Comparison of *Typha* percent cover (% cover), ramet density, and leaf height (cm) atop experimental spoil mounds at Buttonwood Creek (BC) and Buck Pond (BP); W=Wilcoxon rank sum test statistic.

	Site			
***Typha* abundance measure**	**BC**	**BP**	***W***	***p*-value**
% Cover	6.3	3.1	33152	0.003
Ramet Density	2.56	1.53	31422	0.06
Leaf Height	70.2	49.5	32078	0.02

Table 2-17. Summary of *Typha* percent cover (% cover), ramet density, and leaf height (cm) by subplot treatment atop experimental spoil mounds at Buttonwood Creek (BC) and Buck Pond (BP).

Year Site	**Subplot Treatment**	***Typha* % Cover**	***Typha* Ramet Density**	***Typha* Leaf Height**
2015				
BC	**1**	0.095	4	134.6
BC	**2**	0	0	0
BC	**3**	0.1	5.5	122.9
BC	**4**	0	0	0
BP	**1**	0.06	3	100
BP	**2**	0	0	0
BP	**3**	0.05	3	89.2
BP	**4**	0	0	0
2016				
BC	**1**	0.1	4	187.5
BC	**2**	0	0	0
BC	**3**	0.12	5	184.2
BC	**4**	0	0	4
BP	**1**	0.03	2	110.8
BP	**2**	0	0	0
BP	**3**	0.04	3	115.5
BP	**4**	0	0	0

Table 2-18. Summary of *Typha* percent cover (% cover), ramet density, and leaf height (cm) as influenced by elevation (m), soil moisture (% volumetric water content), and bulk density (g/cm^3) atop experimental spoil mounds at Buttonwood Creek (BC) and Buck Pond (BP); H=Kruskal-Wallis between-subject test statistic (degrees of freedom=111).

Site	**Cover**	**Ramet Density**	**Leaf Height**	**Elevation**	**SM**	**BD**
BC	0.035	1.75	77.917	75.23	0.657	0.111
BP	0.02	1	52.083	75.53	0.248	0.079
H	8.8786	2.532	3.177	22.034	73.933	33.833
p-value	0.00289	0.112	0.075	<0.00001	<0.00001	<0.00001

Table 2-19. Primary candidate model(s) selection table for *Typha* cover with estimated parameters and estimated variances; models ranked by AIC_c(x); base model: *Typha* cover=elevation + treatment (T) + site (S) + year + (plot); df=degrees of freedom, logLik=logLikelihood, AIC_c=corrected Aikake Information Criterion, Δ_i=delta AIC, w=Aikake weight, p-value (Pr (>Chisq), and standard deviation (Std. Dev.). Bold numbers indicate significance and model(s) with the greatest explanatory power.

Fixed Effects

Model	Elevation	T	S	Year	df	logLik	AIC_c	Δ_i	w	R^2_c
16	**-0.267**	**+**	**+**	**+**	**8**	**-1188.5**	**2393.3**	**0**	**0.637**	**0.665**
14	-0.352	+	+	+	7	-1190.1	2394.4	1.12	0.363	
NULL	-	-	-	-	2	-2267.8	4539.5		Pr(>Chisq)	**<2.2e-16**

Random Effects

Model	Group Factor	Variance	Std. Dev.	# Observations	# Groups
16	(plot)	0.4316	0.6569	480	60
14	(plot)	0.476	0.6899	480	60
NULL	(plot)	0.5634	0.7506	480	60

Table 2-20. Estimated generalized linear mixed model (GLMM) regression parameters, standard errors (SE), *z*-values, p-values (Pr (>|*z*|)), and 95% confidence intervals (CI) for *Typha* percent cover, described by the model: *Typha* cover=elevation + treatment (T) + site (S) + year + (plot). Bold numbers indicate significance.

Model Parameter	**Estimate**	**SE**	***z* value**	**Pr (>|*z*|)**	**95% CI**
Intercept	2.2174	0.1368	16.214	**<2e-16**	1.9383 to 2.4838
Elevation	-0.2699	0.0868	-3.111	**0.002**	-0.4456 to -0.1023
Treatment$_2$	-2.6751	0.1195	-22.382	**<2e-16**	-2.9190 to -2.4473
Treatment$_4$	-2.9538	0.1363	-21.669	**<2e-16**	-3.2338 to -2.6955
Year$_{2016}$	-0.0186	0.0438	-4.247	**2.2e-05**	-0.2728 to -0.0999

Table 2-21. Candidate model selection table for *Typha* ramet density versus *Typha* percent cover with estimated parameters and estimated variances; models ranked by chi-squared test statistics (Chisq); base models: TC10= *Typha* cover=elevation + year + (plot); TR3=*Typha* ramet density=elevation + year + (plot); df=degrees of freedom, AIC=Aikake Information Criterion, BIC=Bayesian Information Criterion, logLik=logLikelihood, Chi df=chi-squared degrees of freedom, and p-values (Pr (>Chisq). Bold numbers indicate significance and model(s) with the greatest explanatory power.

Model	**df**	**AIC**	**BIC**	**logLik**	**deviance**	**Chisq**	**Chi df**	**Pr(>Chisq)**
TC10	4	4510.4	4527.1	-2251.2	4502.4			
TR3	**4**	**2447.6**	**2464.3**	**-1219.8**	**2439.6**	**1952.1**	**0**	**<2.2e-16**

Table 2-22. Estimated generalized linear mixed model (GLMM) regression parameters, standard errors (SE), *z*-values, p-values (Pr (>|*z*|)), and 95% confidence intervals (CI) for *Typha* ramet density, a Poisson GLMM described by the model: *Typha* ramet density=elevation + year + (plot). Bold numbers indicate significance.

Model Parameter	Estimate	SE	*z* value	Pr (>|*z*|)	95% CI
Intercept	0.6395	0.0749	8.542	**<2e-16**	0.3429 to 0.5846
Elevation	-0.2278	0.0649	-3.509	**0.00045**	-0.3587 to -0.0992
Year$_{2016}$	-0.1580	0.0649	-2.329	**0.015**	-0.2865 to -0.0299

FIGURES

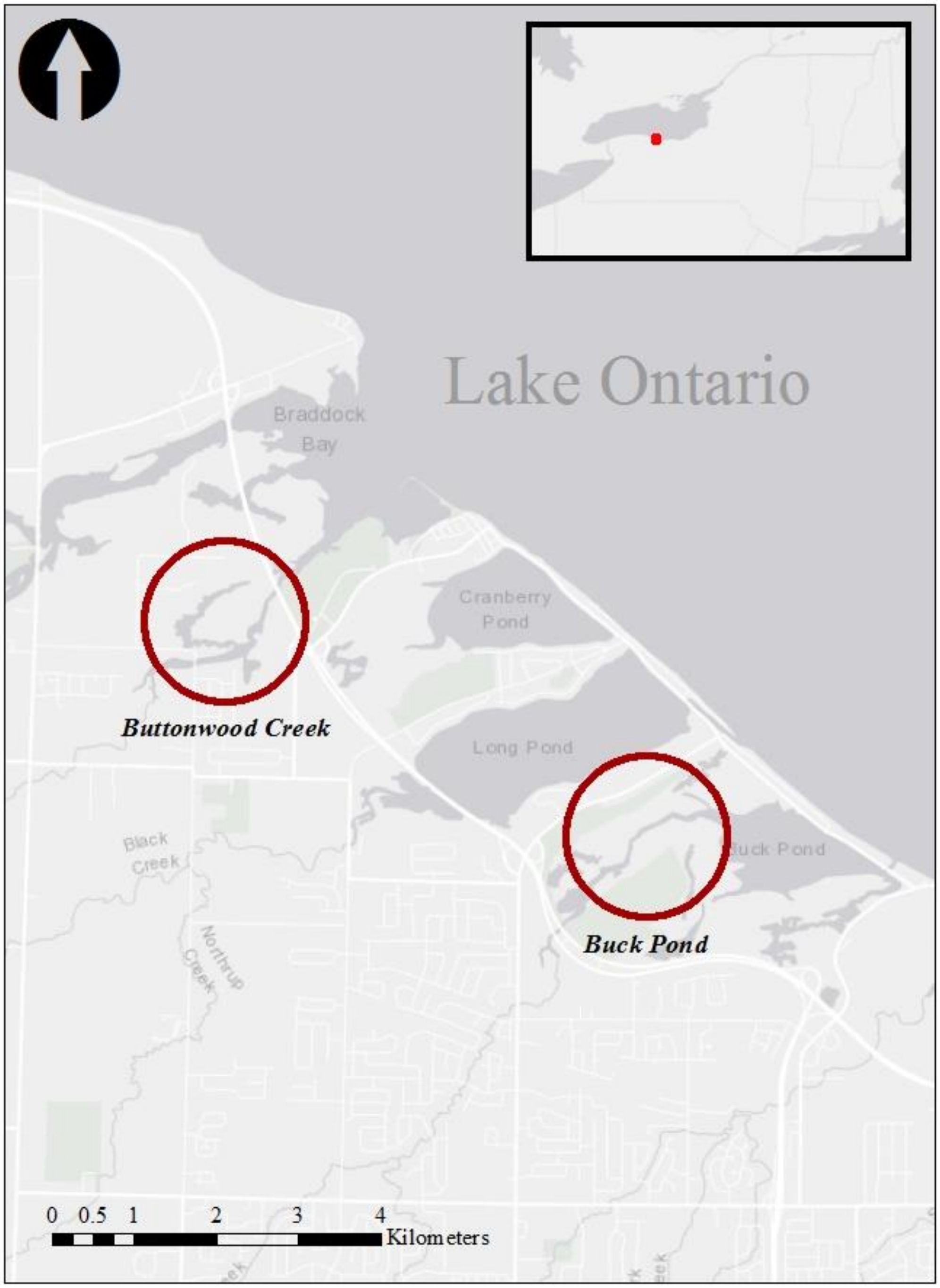

Figure 2-1. Overview map of Braddock Bay WMA and study sites.

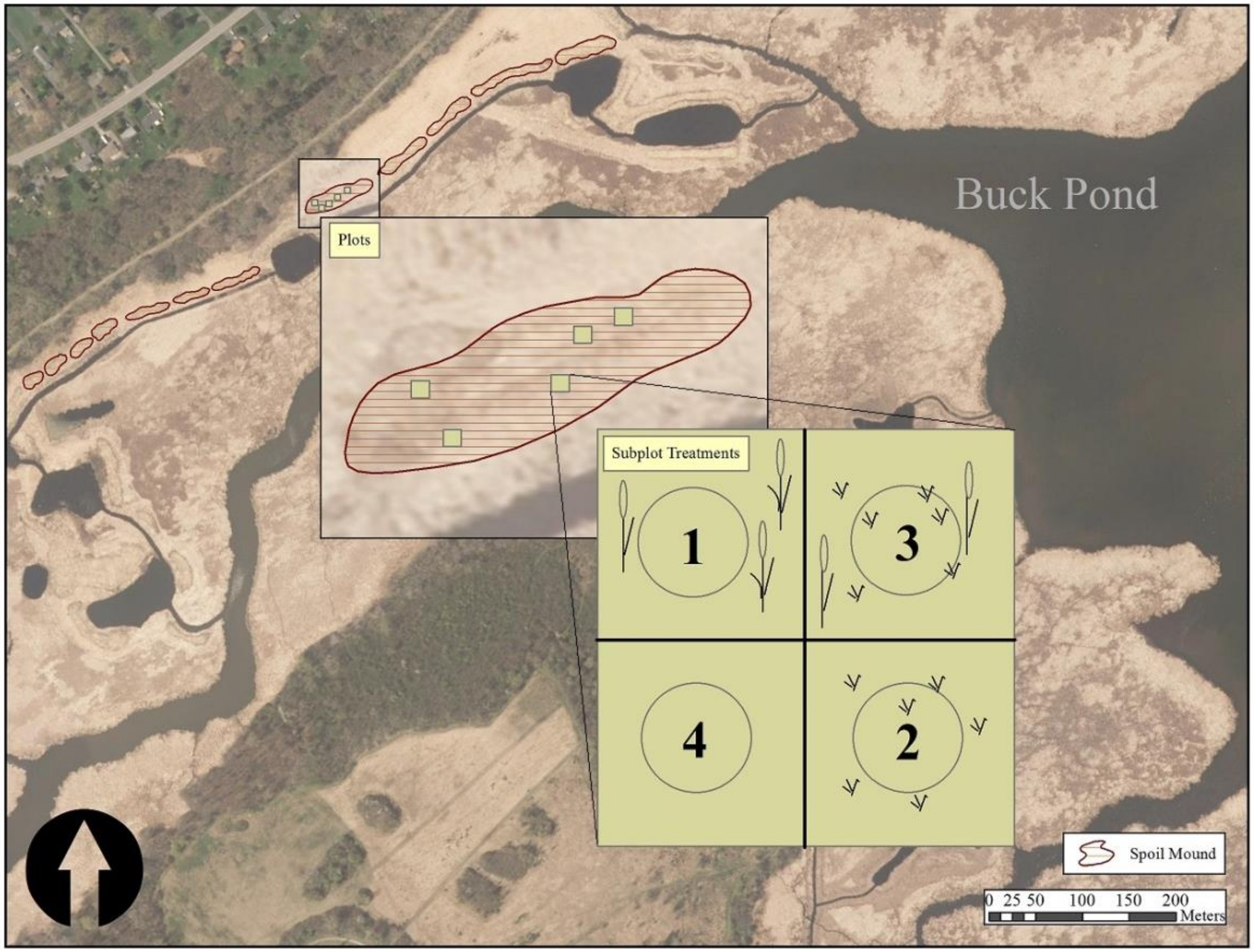

Figure 2-2. Overview of experimental plot layout with planted plot treatments and orientation atop created mounds. For subplot treatment details, see methods.

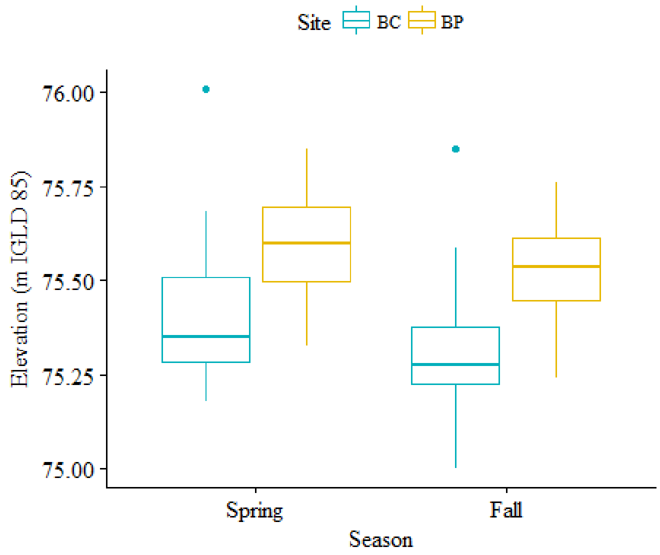

Figure 2-3. Mound elevations (m IGLD 85) at Buttonwood Creek (BC) and Buck Pond (BP) in spring and fall 2016; solid horizontal line=median, box=interquartile range=middle 50% of observations, vertical lines=range encompassing the highest and lowest observed values.

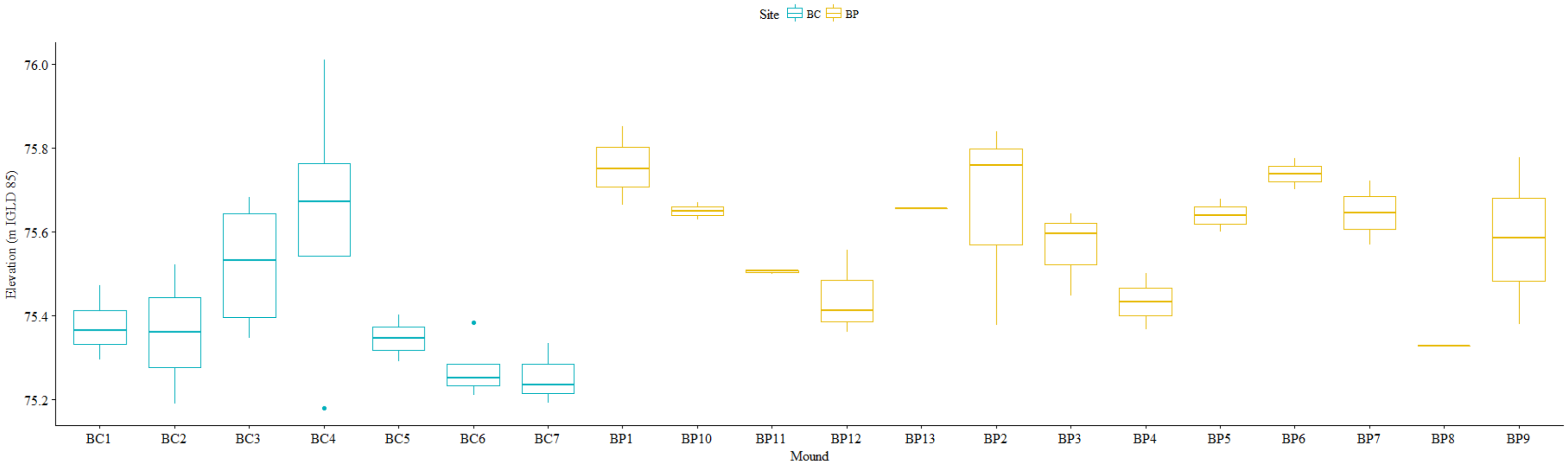

Figure 2-4a. Elevations (m IGLD 85) by mound in spring 2016; solid horizontal line=median, box=interquartile range=middle 50% of observations, vertical lines=range encompassing the highest and lowest observed values.

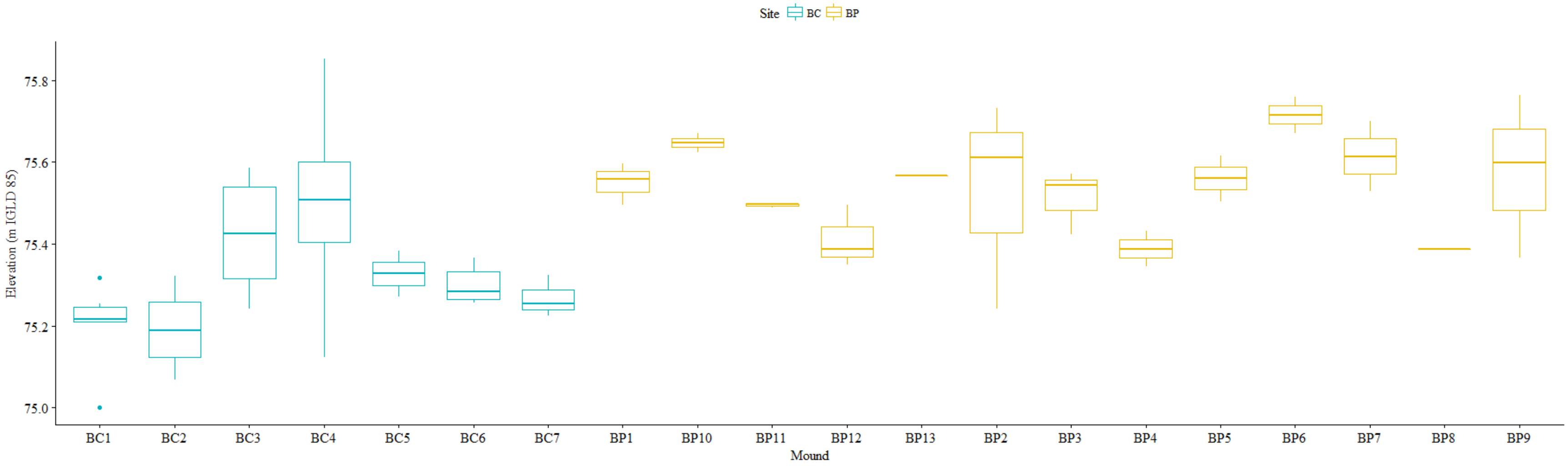

Figure 2-4b. Elevations (m IGLD 85) by mound in fall 2016; solid horizontal line=median, box=interquartile range=middle 50% of observations, vertical lines=range encompassing the highest and lowest observed values.

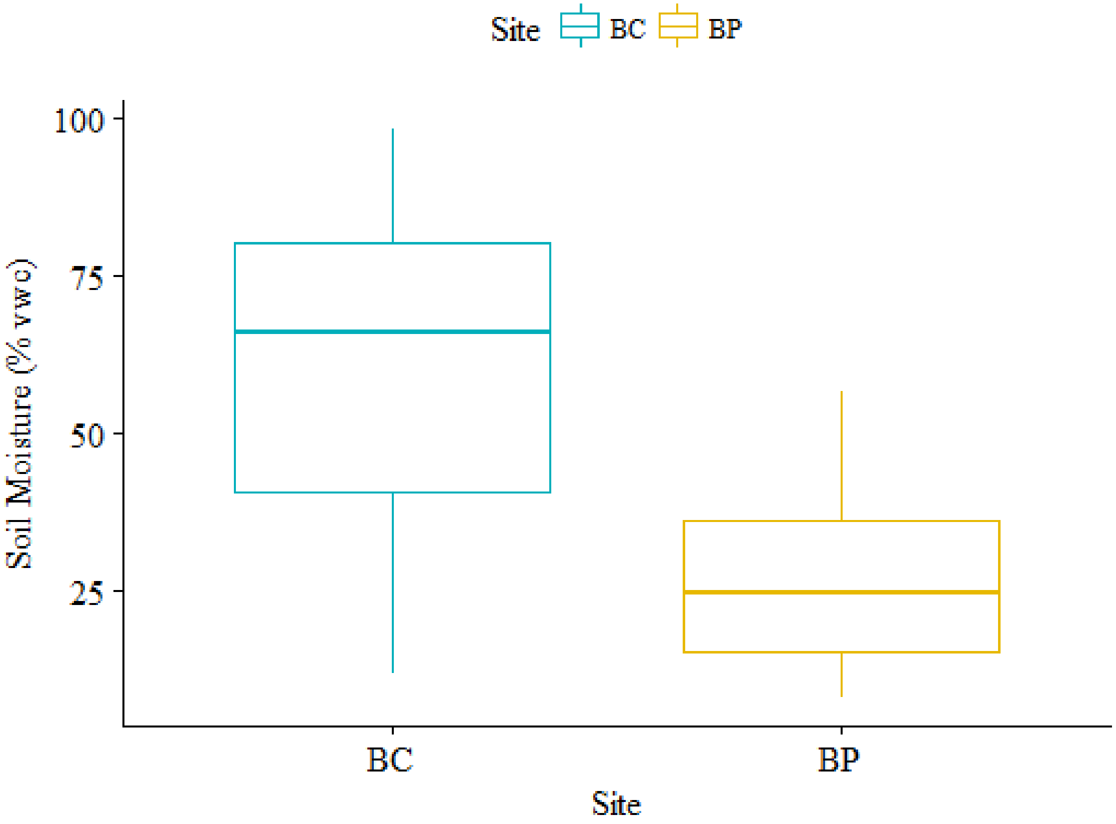

Figure 2-5a. Seasonal mean soil moisture (% volumetric water content) at Buttonwood Creek (BC) and Buck Pond (BP); solid horizontal line=median, box=interquartile range=middle 50% of observations, vertical lines=range encompassing the highest and lowest observed values.

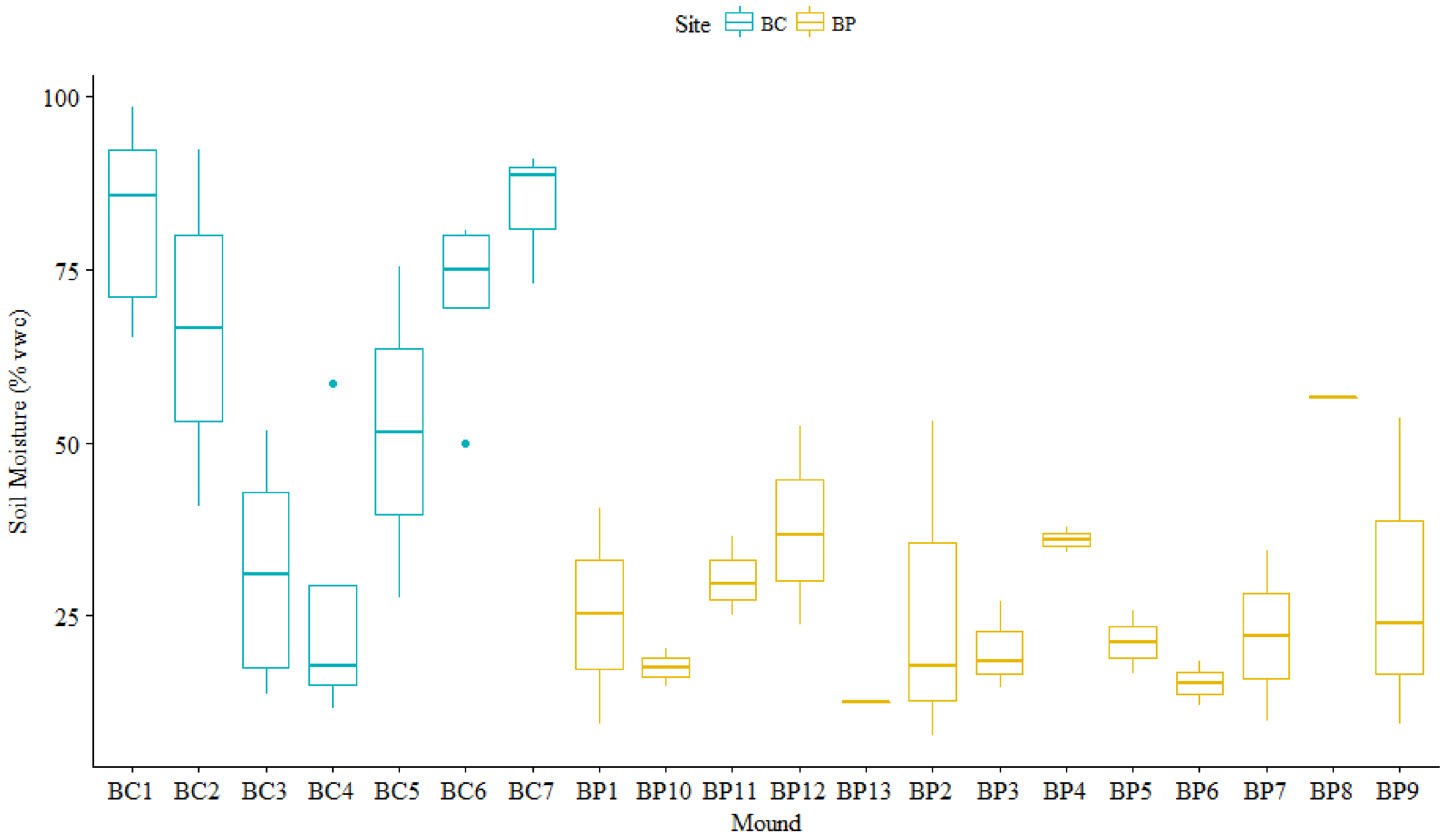

Figure 2-5b. Seasonal average soil moisture (% volumetric water content) by mound; solid horizontal line=median, box=interquartile range=middle 50% of observations, vertical lines=range error bars encompass the highest and lowest observed values.

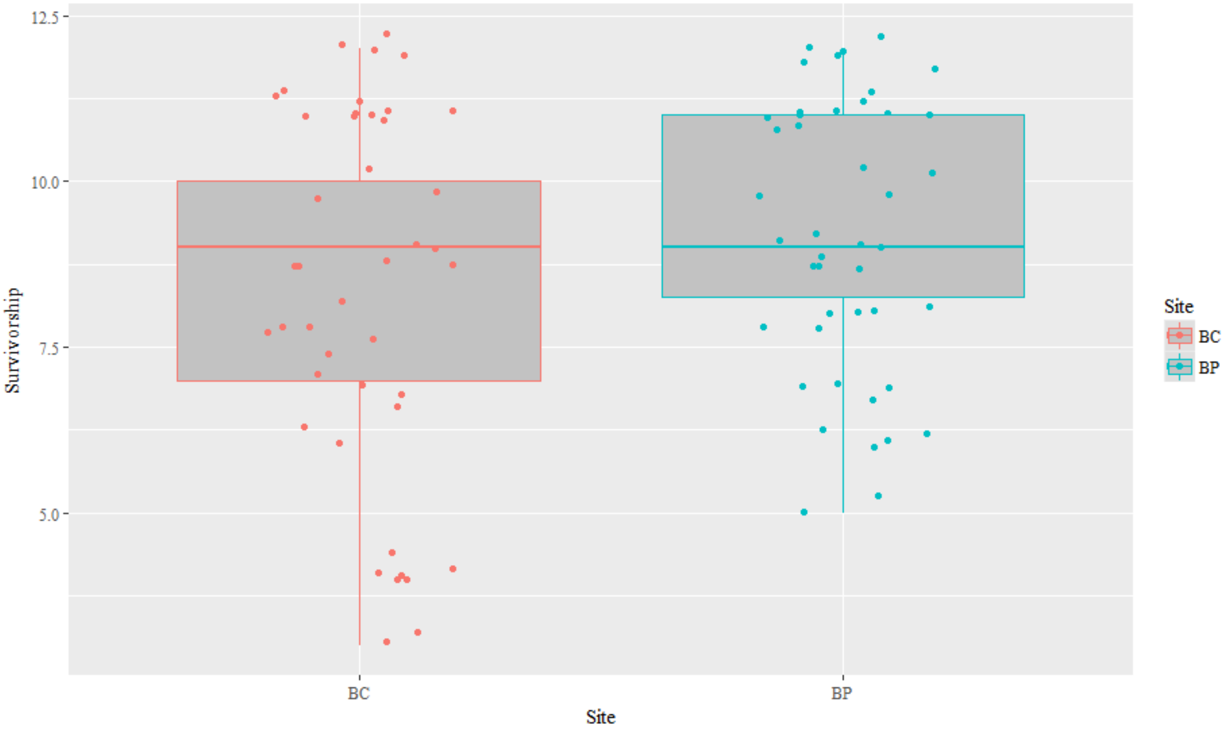

Figure 2-6. Planted plug survivorship at Buttonwood Creek (BC) and Buck Pond (BP); solid horizontal line=median, box=interquartile range=middle 50% of observations, vertical lines=range encompassing the highest and lowest observed values.

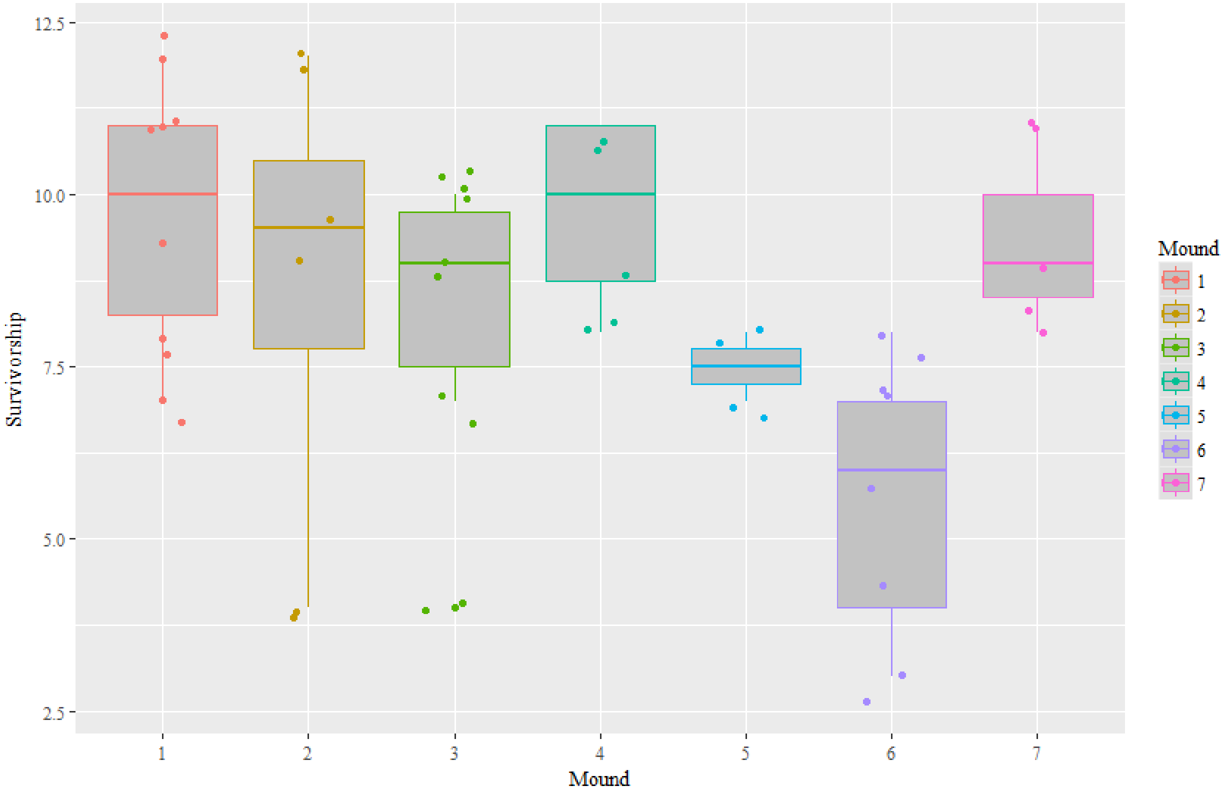

Figure 2-7a. Planted plug survivorship atop experimental spoil mounds at Buttonwood Creek (BC); solid horizontal line=median, box=interquartile range=middle 50% of observations, vertical lines=range encompassing the highest and lowest observed values.

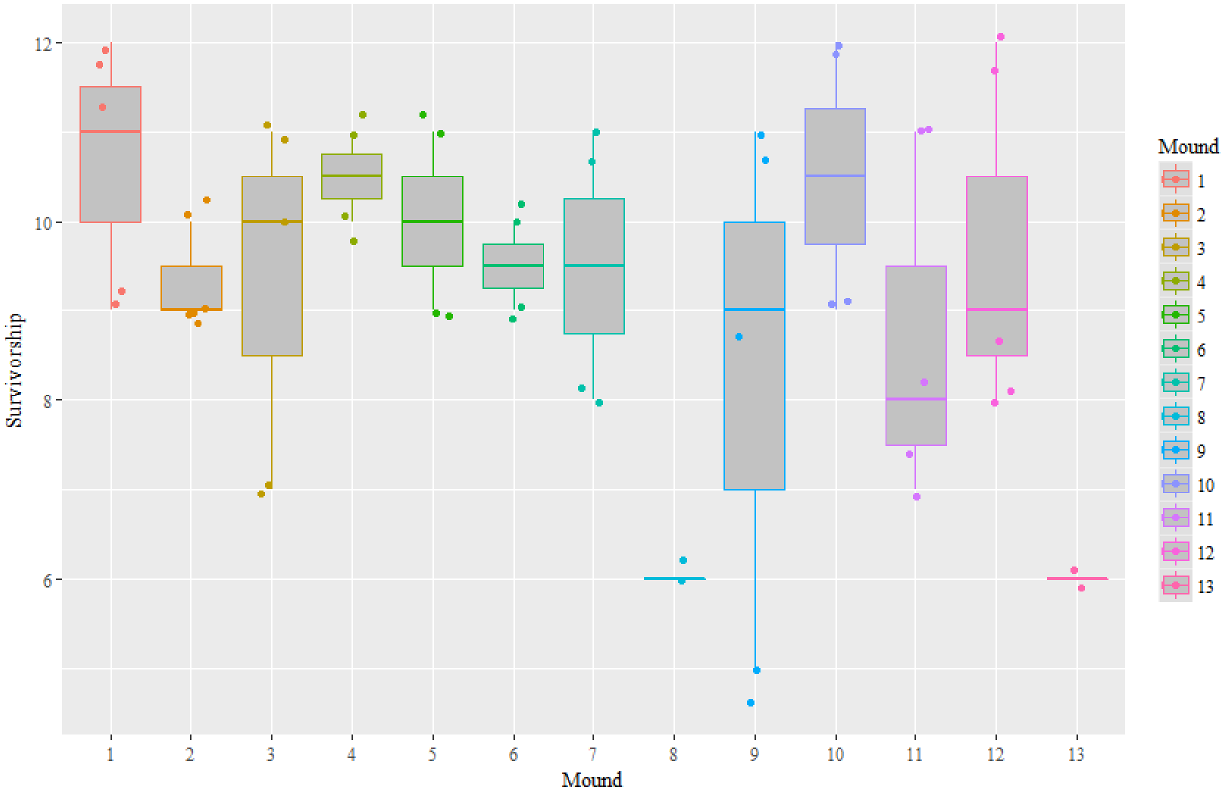

Figure 2-7b. Planted plug survivorship atop experimental spoil mounds at Buck Pond (BP); solid horizontal line=median, box=interquartile range=middle 50% of observations, vertical lines=range encompassing the highest and lowest observed values.

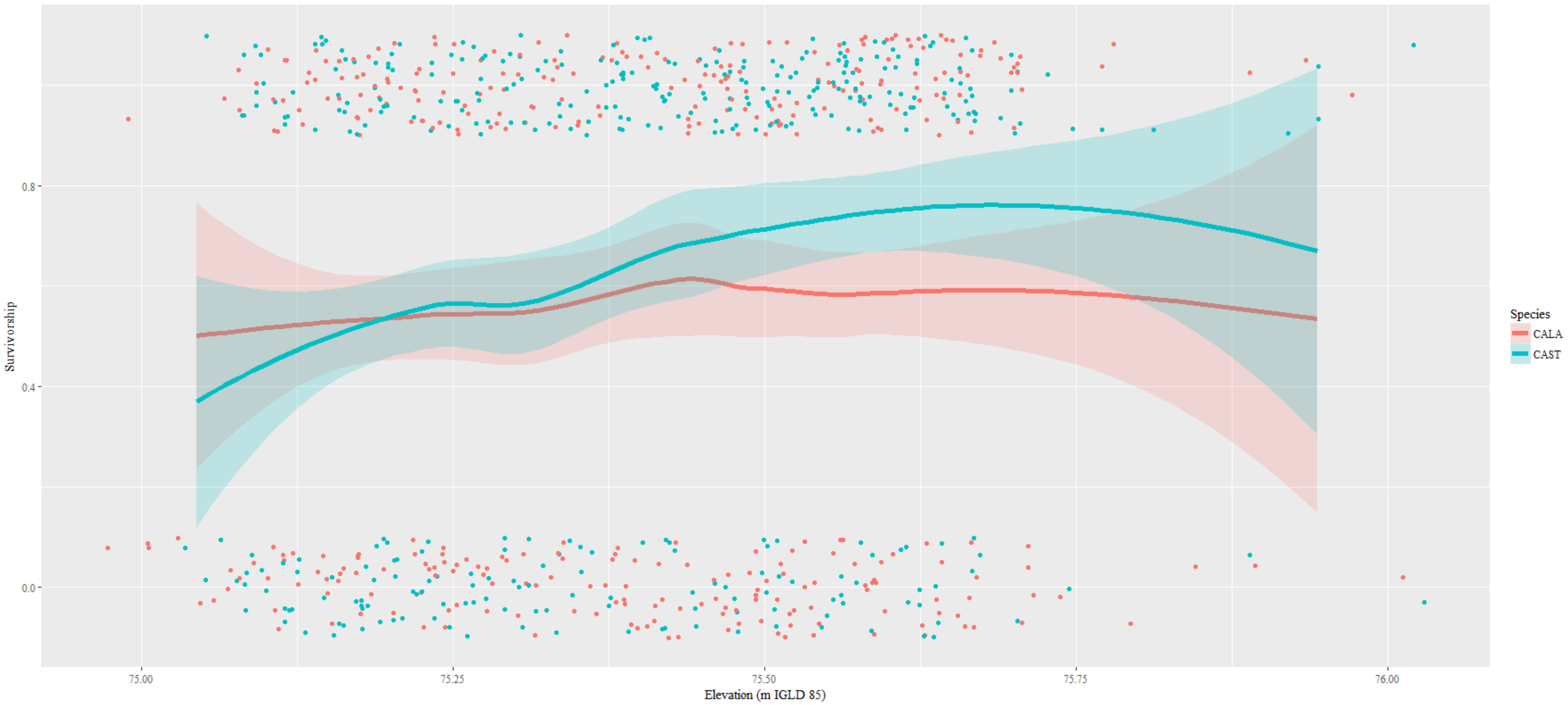

Figure 2-8. Survivorship of planted *Carex lacustris* (CALA) and *C. stricta* (CAST) plugs by elevation (m IGLD 85).

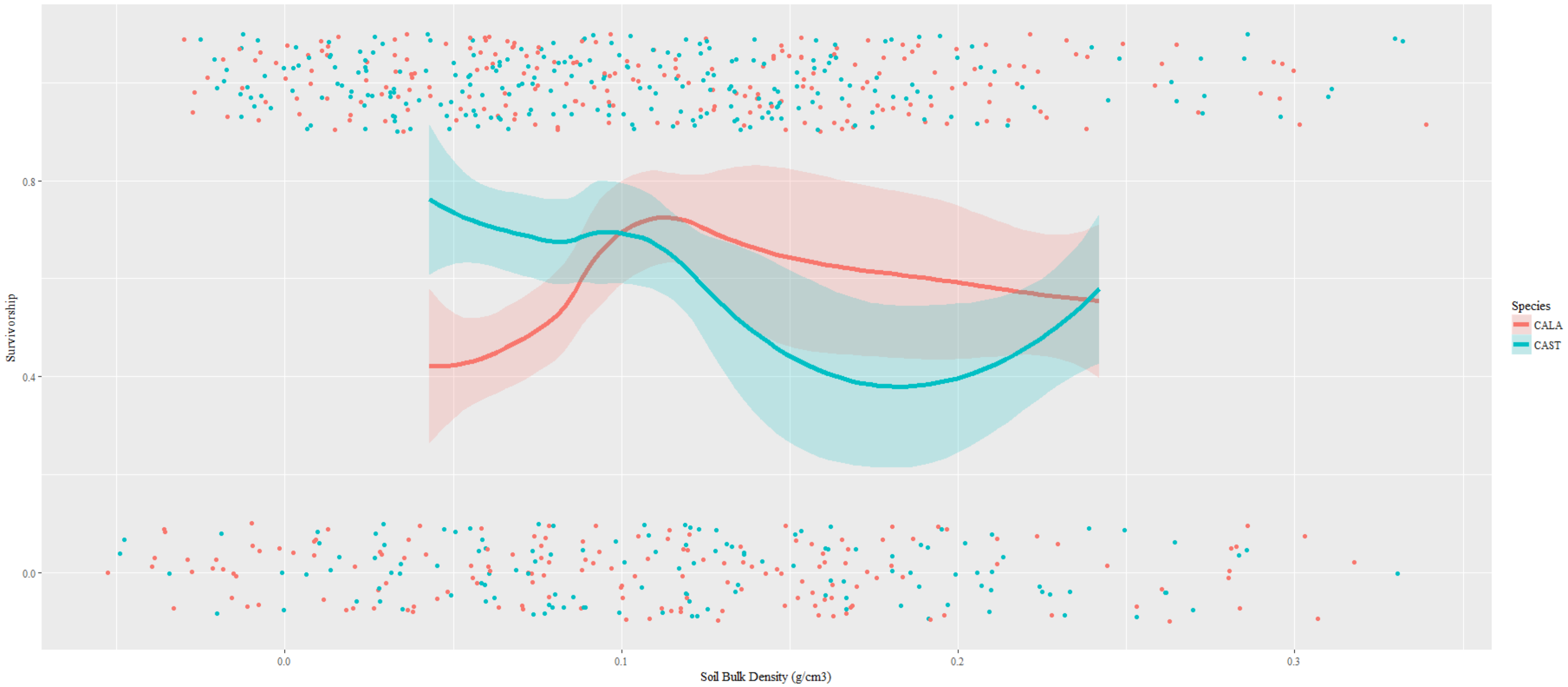

Figure 2-9. Survivorship of planted *Carex lacustris* (CALA) and *C. stricta* (CAST) plugs by soil bulk density (g/cm^3).

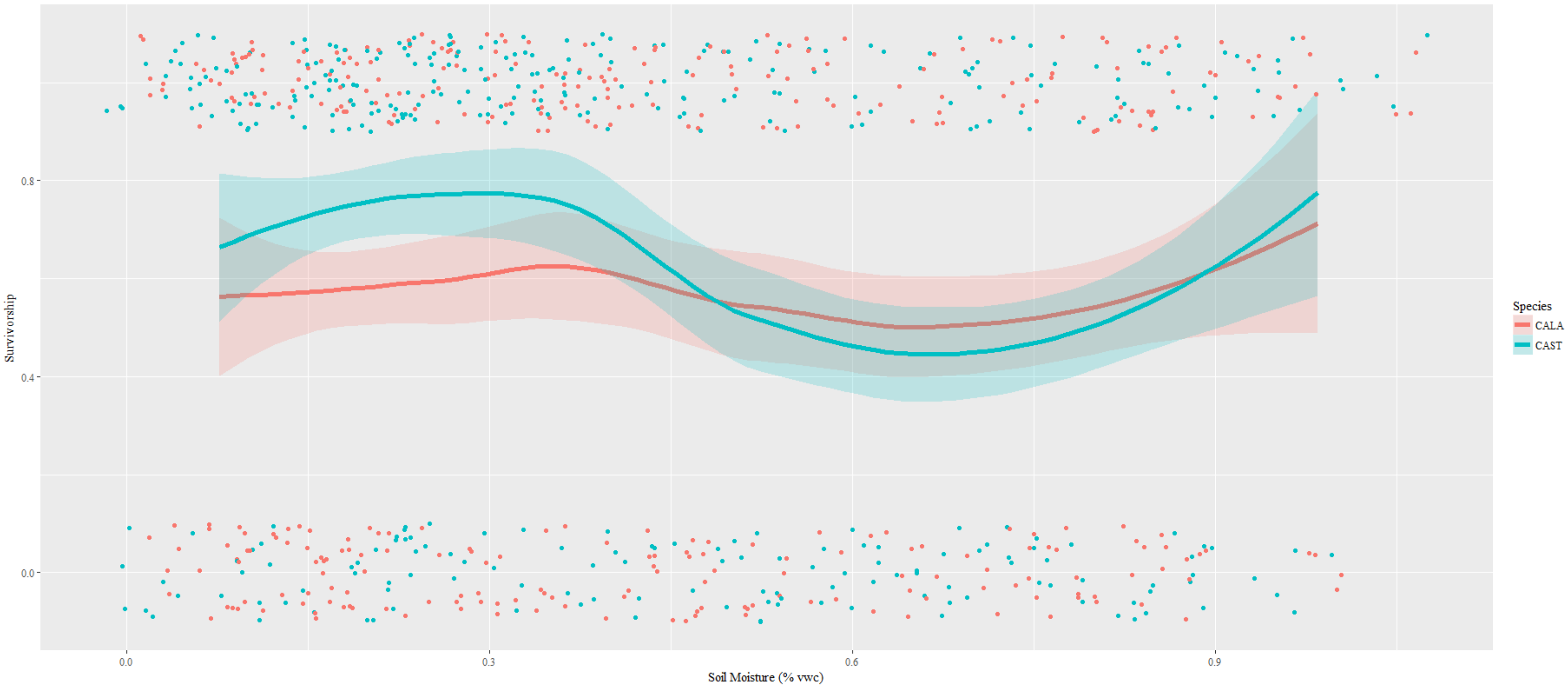

Figure 2-10. Survivorship of planted *Carex lacustris* (CALA) and *C. stricta* (CAST) plugs by soil moisture (% volumetric water content, vwc).

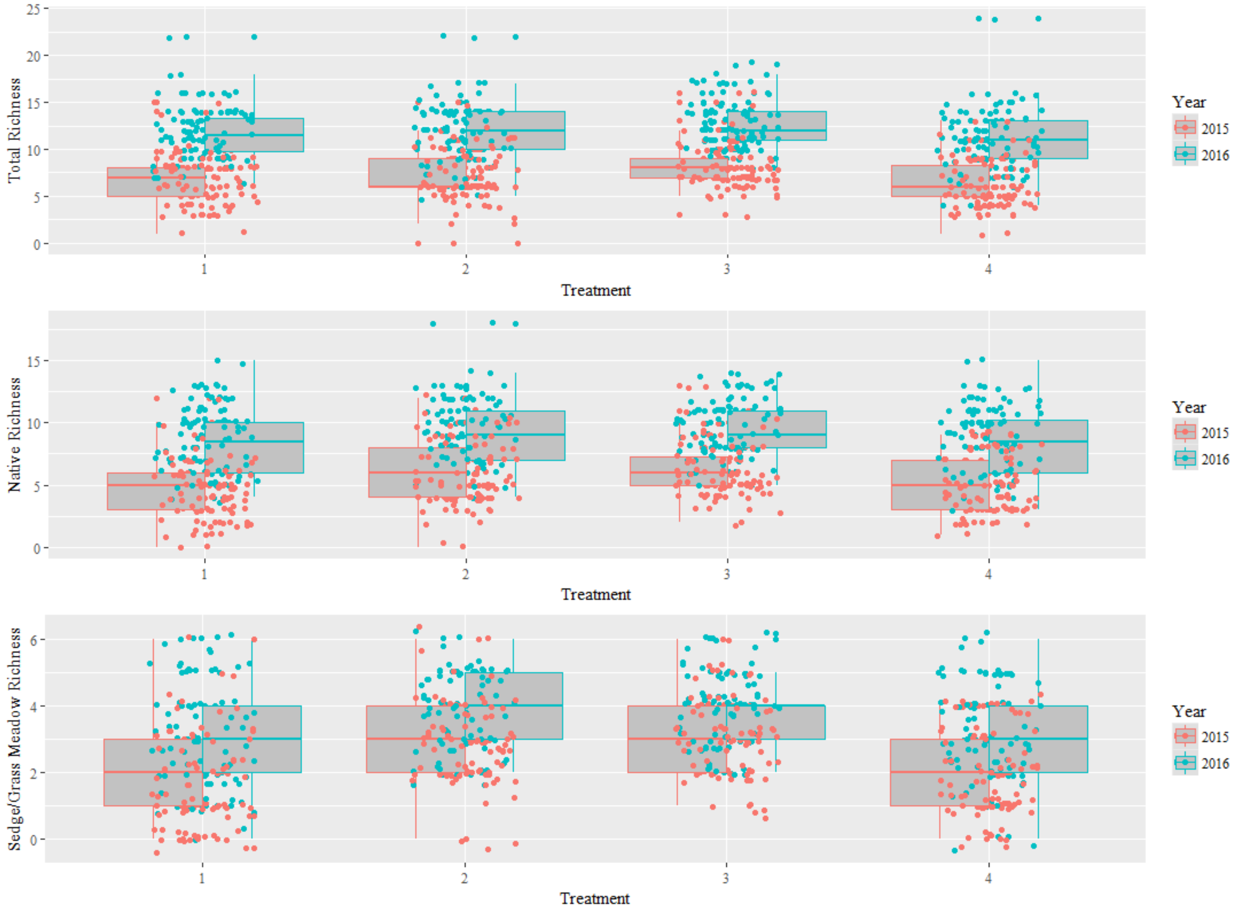

Figure 2-11. Effect of subplot treatment (1-4) on total, native, and SGM richness; solid horizontal line=median, box=interquartile range=middle 50% of observations, vertical lines=range encompassing the highest and lowest observed values. For treatment details, see methods.

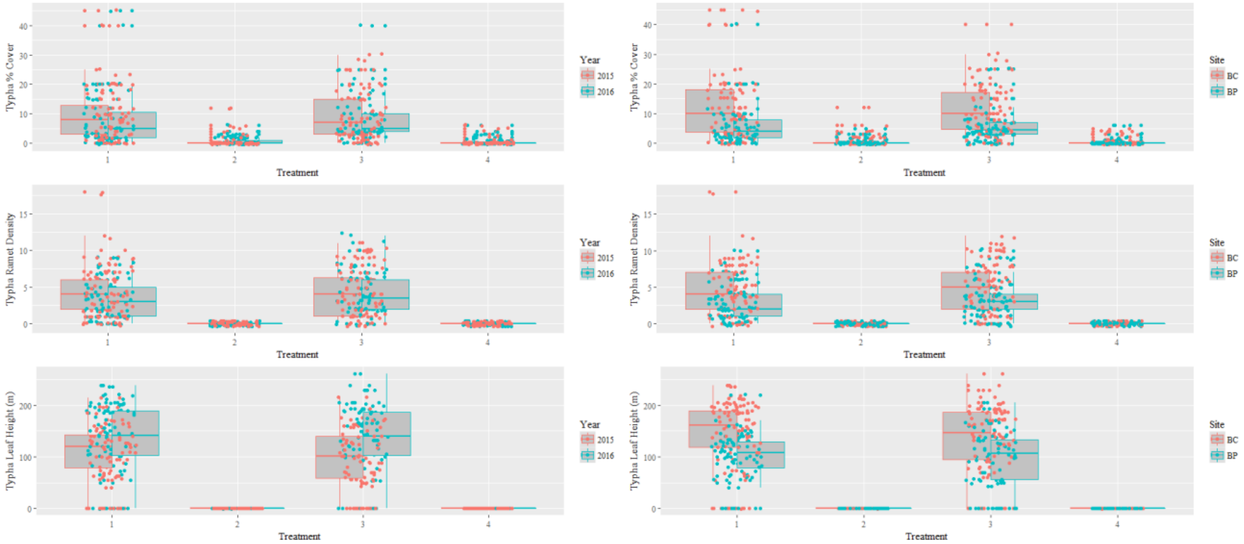

Figure 2-12. Effect of subplot treatment on *Typha* measures; solid horizontal line=median, box=interquartile range=middle 50% of observations, vertical lines=range encompassing the highest and lowest observed values.