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Short & Long-Term Restoration Dynamics of Created Freshwater and Estuarine Wetlands

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SHORT & LONG-TERM RESTORATION DYNAMICS OF CREATED FRESHWATER AND ESTUARINE WETLANDS

by

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B.S., University of Georgia, 2017

THESIS

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On June 29th, 2021

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ABSTRACT

SHORT & LONG-TERM RESTORATION DYNAMICS OF CREATED FRESHWATER AND ESTUARINE WETLANDS

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The restoration of tidal and freshwater wetlands either through compensatory mitigation or voluntary efforts have become a major strategy to conserve remaining wetland resources after historic losses and in the face of current unprecedented threats. Prior efforts of wetland restoration have often fallen short of expectations due to the reliance of a compliance success framework, which requires short monitoring timeframes, misapplies successional theories, and defines success of projects into a yes or no dichotomy. Decades of advancement in restoration ecology theory and its application to wetland ecology and botany have led to the development of a functional success framework to improve outcomes in wetland restoration. Functional success is a framework of thinking in restoration ecology that includes concepts of resiliency and alternative stable-state theory, long-term monitoring requirements, adaptive management, and a view of success as a progression. Elements of functional success were applied to two unique wetland creation studies to improve the understanding of restoration trajectories and baseline expectations for possible outcomes and adaptive management needs.

New Hampshire has adopted a policy for erosion control that establishes living shorelines as the preferred shoreline stabilization method over the past five years. Despite widespread use

on the mid-Atlantic and Southeastern coast in the United States, living shoreline performance has not been well-documented in New England, where shorter growing seasons and ice rafting pose unique challenges that increase with latitude. The vegetation, nekton, and biogeochemical processes were monitored for two years at three living shoreline projects and compared to both references and no-action shorelines to gauge recovery. The recovery of the system was assessed with the Restoration Performance Index and restoration trajectories of the Restoration Performance Index scores and individual metrics were assessed over project age. Living shorelines recovered over 50% of ecosystem structure, function, and services within four years post-constructions. The restoration trajectory followed a logistic regression indicating two phases of recovery: an initial rapid phase driven by nekton and vegetation and a slower, more incremental phase driven by biogeochemical processes. Adaptive management of living shorelines were documented and included wrack removal, measures to prevent herbivory, and replanting of vegetation.

Wetland ecology and restoration historically developed successional models of the vegetation community by studying wetlands of various ages simultaneously. The experimental approach has been cheap, quick, and effective at understanding broad floristic trends, allowing for practitioners to set expectations including the widely held notion that the vegetation community reaches a dynamic equilibrium after 15 years. A created freshwater wetland in Portsmouth, New Hampshire was floristically reevaluated in 2020 to add to a thirty-five-year data set of the species composition and community distribution of the vegetation community. The wetland followed similar documented patterns of being dominated by hydrophytes, perennials, and natives over time. Species turnover, the rate of succession, declined by half between the 7 – 18 and 18 – 35 years post-construction. Conversely, the distribution of wetland

communities experienced drastic change after 18 years post-construction with double the number of wetland communities, major cattail expansion into graminoid meadows and aquatic beds, proliferation of woody vegetation, and development of unique vegetation communities like sedge meadow marshes and red maple swamps.

CHPATER I

WAIST DEEP IN THE MUDFLAT: OBSTACLES AND AVENUES TO IMPROVING OUTCOMES OF WETLAND RESTORATION

Historical Wetland Loss – What Exactly Did We Lose?

Historically wetlands in the United States, whether saline or freshwater, were viewed as either an obstacle to progress or a natural resource to exploit. The conterminous United States has lost roughly 118 million acres of wetlands (~ 53% loss) since colonization by European settlers (Dahl 1990). Forested swamps experienced the highest losses from harvesting of specialized lumber in the 1700 and 1800s and development within the Mississippi River floodplain (OTA 1984, Dahl and Allord 1996). It has been estimated that up to 50% of salt marshes have been lost in the same time period (Kennish 2001). The majority of wetland loss occurred up to 1980; before then federal and state programs encouraged wetland destruction. Once legal protections were enacted and large-scale restorations were funded in the second half of the twentieth century, the rate of wetland drainage decreased drastically, even resulting in a net gain of 192,000 acres during 1998 – 2004 (Dahl 2006).

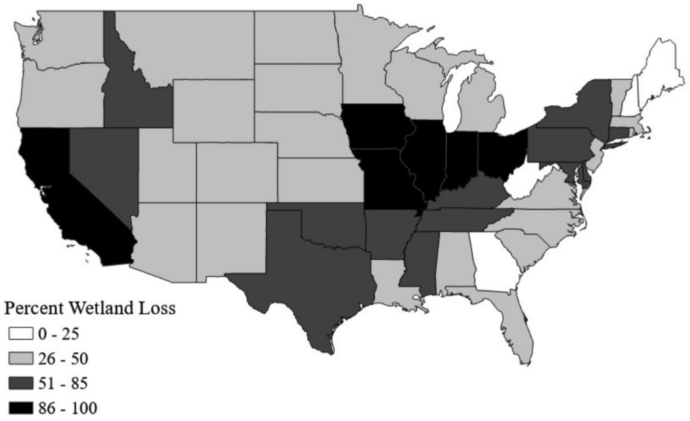

Figure 1: Percent wetland loss per state between 1780 - 1980 of the conterminous United States. Figure was created from data in Dahl 1990.

The federal government directly engaged in or indirectly supported the draining and destruction of wetlands since the mid-nineteenth century to promote westward movement, agriculture, and, later, urbanization and infrastructure. Development and agriculture in wetlands were initially spurred on by the federal government through the transfer of 64.9 million acres of wetlands to states through the Swamp Land Acts of 1849, 1850, and 1860 (Dahl and Allord 1996). The federal government embarked on large-scale wetland drainage projects as watershed improvements and flood control projects through the Reclamation Act (1902), Flood Control Act (1944), and Watershed Protection and Flood Prevention Act (1954). The Department of Agriculture (USDA) provided technical services and financial incentives for installation of drainage tiles and channelization of first order streams in the upper reaches of watersheds (Pavelis 1987, Heimlich et al. 1998). Coordinated efforts across the USDA, Army Corp of Engineers (USACE), and Bureau of Reclamation led to significant losses of wetland acreage in California's Central Valley, Mississippi River floodplain, Dakota prairie pothole region, and Florida Everglades (Pavelis 1987, Dahl and Allord 1996). Freshwater wetlands face a myriad of threats in the twenty first century including conversion to agriculture, modification of hydrology for flood control, urban development, overextraction of aquifers, peat mining, eutrophication, climate change and exotic species invasion (Mitsch and Hernandez 2013, Mitsch and Gosselink 2015).

Salt marshes and other tidal wetlands suffered similar fates in the twentieth century from a combination of local municipality decisions and federal programs. In the 1930's, the Civilian Conservation Corps ditched almost all Atlantic salt marshes for mosquito control (Wolfe 1996), leaving long-term legacy effects on local hydrology (Burdick et al. 2020a). From the mid-1800s to mid-1900s, rapidly growing urban centers found salt marsh systems ideal for expansion of housing, construction of infrastructure (e.g. airports, ports), and conversion to landfills (Kennish

2001, Gedan et al. 2009). For example, Boston may have lost as much as 81% of pre-colonial salt marsh extent from urbanization (Bromberg and Bertness 2005). Widespread salt marsh loss in the Gulf Coast resulted from the dam and lock system of the Mississippi River, soil subsidence from oil and gas extraction, and construction of levees and canals within the wetlands (Turner 1990). The combination of sea level rise, development of shorelines, invasive species, and high nutrient loading continue to contribute to the loss of salt marsh ecosystems (Bertness et al. 2002, Bozek and Burdick 2005, Watson et al. 2016, Kornis et al. 2018a).

The alteration and destruction of wetlands across the United States translates into a loss of valuable ecosystem functions and services. Despite covering a small fraction of the global land surface, coastal and freshwater wetlands supply a disproportionate amount of the world's renewable ecosystem services (Zedler and Kercher 2005, Barbier et al. 2016). In the past 50 years, scientists and ecologists have identified a host of provisioning, regulating, cultural, and supporting ecosystem services directly attributed to wetland systems (OTA 1984, Finlayson et al. 2005). From an economic standpoint, the services provided by freshwater and coastal wetlands were valued at \$25,681 and \$193,843 ha^{-1} yr^{-1} (2007 \$), respectively (Costanza et al. 2014; Table 1). Continuing research have led to a greater understanding of the amount of services provided by wetlands, leading to upward revisions of economic valuations over time, especially tidal wetlands (Costanza et al. 1997, 2014).

Table 1: Global valuation of ecosystem services of selected ecosystems from Costanza et al. 1997. Dollars were adjusted for inflation to 2021 values from 1994 values. The total global value of all ecosystems services was calculated by multiplying the global area of each ecosystem by the value of ecosystem services on a per hectare basis.

Ecosystem	Area (ha x 10^6)	Total Value of Ecosystem Services (2021$ ha^{-1} yr^{-1})	Total Global Value (2021$ yr^{-1} x 10^{-9})
Open Ocean	33200	450	14,979
Forests	4855	1,732	8,411
Grasslands	3898	415	1,619
Mangroves & Salt Marsh	165	17,855	2,945
Freshwater Wetlands	165	34,995	5,775
Lakes & Rivers	200	15,188	3,038
Desert	1925	0	0
Tundra	743	0	0

Freshwater wetlands can play an outsized role on the hydrology of a watershed by reducing peak discharges of storm events, recharging aquifers, and subsidizing groundwater flows during droughts (Mitsch and Gosselink 2015). For example, the USACE purchased and conserved 8,095 acres of riparian wetlands on the Charles River, near Boston, as substitutes for building large flood control structures (USACE 2017). Tidal wetlands have been shown to attenuate wave energy (Morgan et al. 2009, Gedan et al. 2011), abate coastal storm flooding (Danielsen et al. 2005, Arkema et al. 2013), and stabilize shorelines from erosion (Shepard et al. 2011). The restoration of salt marshes and construction of nature-based infrastructure like living shorelines have been advocated to increase coastal resiliency against sea level rise and more intense coastal storms (National Research Council 2007, Bilkovic et al. 2017a).

Through the formation of anaerobic soils and interception of surface and groundwater flows, wetlands can significantly alter biogeochemical cycles and function as natural filters that improve water quality. Wetlands reduce nitrogen in the water column through denitrification in anaerobic soils and the burial of organic matter in peat development (Piehler and Smyth 2011, Mitsch and Gosselink 2015). High productivity and increased sedimentation can remove phosphorus, metals, and organic matter from the water column of both tidal and riverine wetlands (Mitsch and Gosselink 2015). The fact that municipalities utilize engineered wetlands to treat wastewater is a testament of the strength of this ecosystem service (Mitsch and Jorgensen 2003). Additionally, tidal wetlands have been widely recognized for high carbon storage in soils, termed "blue carbon", and regarded as a small yet important tool in combating climate change (Pendleton et al. 2012, Davis et al. 2015).

Wetlands serve as important breeding and nursery grounds for adapted flora and fauna. It has been estimated that over 5,000 species of plants, 190 amphibians, and one-third of all bird species in the United States have adapted to utilize wetland habitats for part of their life cycle (OTA 1984). Salt marshes play an outsized role as nurseries in the early development of commercially important fish and crustaceans (Beck et al. 2001, Johnson and Eggleston 2010). The vegetated salt marsh habitat provides a refuge from predators for juvenile transient nekton and supports infaunal prey for larger commercially important stock (Boesch and Turner 1984, Zimmerman and Minello 1984). The biodiversity of wetland systems can translate into recreational opportunities, such as birdwatching, fishing, and hunting waterfowl, and improved aesthetics for local communities (Barbier et al. 2016, Arkema et al. 2017, Anderson et al. 2018).

Legal Protections & Compensatory Mitigation

Wetland protection is not captured by one explicit federal law but rather a patchwork of various state laws, federal regulations, financial disincentives, and conservation funding programs. The main instruments for wetland protection are the Section 404 program, "no net loss" executive orders, and state-level wetland protection statutes. Congress formally established and endorsed the Section 404 permit program, administered by the USACE and the Environmental Protection

Agency (EPA), through the 1977 amendments to the Federal Water Pollution Control Act, better known as the Clean Water Act (CWA) (Hough and Robertson 2009, Mitsch and Gosselink 2015). The Section 404 permit program regulates permissible damages to wetlands including pollution, dredging, or filling and outlines the requirements for parties to compensate the public for permissible damages. The "no net loss" policy of the H.W. Bush and Clinton administrations reaffirmed the protection and conservation of existing wetland resources in the United States. The Section 404 permit program ensures and enforces the "no net loss" policy and the CWA's mission to "restore and maintain the chemical, physical, and biological integrity of the Nation's waters" (Hough and Robertson 2009).

The Section 404 program lays out a sequence of considerations under wetland mitigation when the USACE is evaluating permits: (1) avoid impacts, (2) minimize damages, and then (3) compensate the public (National Research Council 2001). Despite explicit guidelines for regulators to focus on avoidance and then minimization of damages first, compensatory mitigation has become a widely accepted solution for both regulators and developers (Hough and Robertson 2009). Compensatory mitigation typically takes the form of permittee-responsible mitigation, where an individual party enhances, preserves, restores, or creates wetland resources to compensate for permitted damages (National Research Council 2001). During the permitting process, the USACE or state agencies can require certain mitigation ratios depending on the type of wetland habitat to be impacted. For example, the State of New Hampshire administers its own wetland permitting program under the Fill and Dredge in Wetlands Act (RSA 482-A) and requires a 3:1 creation ratio or 2:1 restoration ratio of tidal wetlands (Pillsbury et al. 2008). Creation is the process of transforming a new wetland in a previously disturbed or upland area by establishing proper hydrology and appropriate vegetation. Restoration is the return of a wetland from a disturbed or degraded condition to a reference or previously documented condition (NRC 1992). For ease of discussion, creation and restoration actions through compensatory mitigation will be henceforth referred as “restoration”.

To ensure legal compliance, the USACE or state agencies assign success criteria for mitigation projects usually in the form of various vegetation community and hydrology metrics, since wetland delineation criteria are based on hydrology, vegetation, and soils (Environmental Laboratory 1987, National Research Council 2001). Common vegetation metrics require a certain threshold of native vegetation cover, species richness, or survival of woody propagules (Bosch and Matthews 2017). Permits typically entail only five years of post-construction monitoring to ensure success. The USACE or state agencies can require monitoring from individual parties or the regulators themselves to ensure the mitigation took place and was successful. If regulatory success criteria are not met during the five years, the USACE or state agency can require further action. After five years, however, the individual party is usually not legally required to enhance restoration efforts (NRC 2001).

In addition to legal protections and mitigation requirements, Congress has enacted financial disincentives for wetland destruction and funding for conservation and monitoring programs. The Swampbuster provision in the 1985 Farm Bill affected agricultural impacts by penalizing farmers who drain wetlands by withholding subsidies and crop insurance payments (Wiebusch and Lant 2017). The federal government has drastically increased conservation and protection of existing wetlands through programs like the Wetland Reserve Program, Coastal Zone Management Program, Land and Water Conservation Fund, National Estuarine Research Reserve System, and National Wildlife Refuge System (Mitsch and Gosselink 2015, Wiebusch and Lant 2017, Anderson et al. 2018). The North American Wetlands Conservation Act alone has awarded over

$1.6 billion to 2,833 projects since its passage in 1989 (Steinwand 2019). The State of New Hampshire actively reinvests mitigation in-lieu fees through the Aquatic Resources Mitigation fund back into restoration projects (Pillsbury et al. 2008). Additionally, the National Wetlands Inventory by the United States Fish and Wildlife Service provides Congress, federal regulators, and state agencies data critical for proactive management and regulatory decisions. Regulatory protection measures, financial disincentives, and conservation actions have led to a significant decrease of the rate of wetland destruction, dropping from 458,000 acres yr $^{-1}$ in 1950 – 70 to 13,800 acres yr^{-1} in 2004 – 09 (Dahl 2013). In a testament to the shift in society's values and government response, the United States achieved a net gain of 32,000 acres $^{yr-1}$ in 1998 – 2004. Reliance on compensatory mitigation in the United States was apparent in the 2004 – 09 NWI as 489,600 acres of upland were converted to wetlands (Dahl 2013).

Documented Regulatory & Ecological Shortfalls of Compensatory Mitigation

Despite successfully reducing the rate of wetland destruction since the early 1970's, the Section 404 program and the concept of compensatory mitigation has been routinely scrutinized and its effectiveness questioned from both regulatory and ecological perspectives. During the full enforcement of wetland protection and mitigation guidelines by the EPA and USACE, the United States still lost 644,000 and 62,000 acres of wetlands during 1986 – 97 and 2004 – 09 periods, respectively. It should be noted that certain losses are out of the purview of the federal government. For example, documented losses of coastal wetlands from sea level rise (Dahl 2013), inability to federally regulate the *draining* of wetlands under the modified Tulloch Rule (73 Fed. Reg. 79, 641 (Dec. 30, 2008)), or the recent narrowing of the definition of Waters of the United States under the Trump Administration (85 Fed. Reg. 22,250 (June 22, 2020)) are outside the purview of Section 404.

Serious concerns have been made public about how federal, state, and local officials run the program and ensure completion and regulatory success of projects. Many of the documented shortcomings stemmed from failure in basic administrative accountability (*i.e.*, paperwork issues), negligibility to ensure projects were constructed properly, and lack of follow-up compliance monitoring (National Research Council 2001). In one high profile case, the USACE was condemned by the Governmental Accountability Office during the second Bush administration for not adequately maintaining basic records and conducting self-imposed compliance visits to mitigation wetlands (GAO 2005). Documented examples of failure include a third party not constructing the mitigation wetland, constructing undersized wetland area, inability to obtain conservation easements, not adhering to agreed-upon site designs, and lack of monitoring after construction. Regulatory shortfalls have been widespread and documented in Florida (Erwin 1991), Illinois (Matthews and Endress 2008), Massachusetts (Brown and Veneman 2001), Michigan (Kozich and Halvorsen 2012), and Tennessee (Morgan and Roberts 2003).

In addition to regulatory shortfalls, researchers have documented declines in ecological performance of mitigation wetlands after the required monitoring period (Matthews and Endress 2008). Dominance of exotics (Spieles 2005, Kozich and Halvorsen 2012), low survivorship of woody and herbaceous propagules (Bosch and Matthews 2017), and insufficient hydrophyte cover (Matthews and Endress 2008, Matthews et al. 2019) are common issues that prevent mitigation wetlands from meeting both legal compliance and ecological parity with references. The deficit in ecosystem services, or "recovery debt", from inadequate restoration and incomplete recovery can be long-term (Gutrich and Hitzhusen 2004, Moreno-Mateos et al. 2017). Although it may be difficult to pinpoint or explicitly demonstrate, communities may be acutely impacted from lost ecosystem services if deficient wetland mitigation is widespread across a watershed or landscape (see Evenson et al. 2018, Barbier et al. 2013).

Compliance Success Framework of Compensatory Mitigation

The ecological shortfalls of compensatory mitigation can be partially attributed to a Compliance Success framework in the success criteria, monitoring framework, and lack of requirements for adaptive management (Quammen 1986, Kentula 2000). Compliance success is an approach to ecological restoration which views the outcome of a project in terms of a yes/no matrix and bases the evaluation of a project on narrow, univariate abiotic and biotic indicators. Within compliance success is a set of ecological assumptions applied to wetland restoration which can lead to less-than-desirable outcomes. The typical five-year monitoring regime assumes that biotic and abiotic metrics can recover to parity with references or achieve success criteria quickly (Mitsch and Wilson 1996, Zedler and Callaway 1999). The recovery of complex ecosystem functions and the overall project can be adequately evaluated with univariate metrics of vegetation and hydrology (Zedler and Callaway 2000). Lastly, if baseline performance criteria are met within five years, created wetlands may continue to recover along predictable successional trajectories and thus maintain (or achieve) restoration goals in the long-term (Zedler and Callaway 1999, Bosch and Matthews 2017).

Research on wetland and restoration ecology over the past few decades have shed light on the insufficiencies of the compliance success framework. The utility of the five year monitoring period has been repeatedly criticized with evidence that abiotic and biotic factors (1) recover at different time scales and (2) typically require loner than five years to recover and stabilize (Craft et al. 1999b, Zedler 2000, Morgan and Short 2002, Moreno-Mateos et al. 2012, Noll et al. 2019).

Extensive research on the vegetation community, one of the core measurements in the monitoring regime (Environmental Laboratory 1987, Staszak and Armitage 2013), has demonstrated the five year monitoring period may fail to fully and accurately capture the recovery of the community. The vegetation community undergoes a phase of rapid change and recovery (0 – 10 years) before it reaches a relative stable-state or a phase of slow, incremental change (Roman et al. 2002, Seabloom and van der Valk 2003, Spieles 2005, Stefanik and Mitsch 2012), which may lead to false conclusions at the end of a five year window (Galatowitsch and Van der Valk 1996, Ahn and Dee 2011, Gittman et al. 2016a). More comprehensive metrics such as species richness, community composition, and floristic quality assessment may require over 15 years to reach reference conditions (Onaindia et al. 2001, Craft et al. 2002, Morgan and Short 2002, Spieles et al. 2006, Gutrich et al. 2009). Certain vegetation communities have been documented to have a significant lag time (> 15 years) after the establishment of proper hydrology including wet prairie meadows (Aronson and Galatowitsch 2008) and forested swamps (Stanturf et al. 2009).

The assumption that the restoration trajectory can be relatively predictable once the hydrology and initial vegetation is established has been misconstrued and needs to be reconsidered or, at least, have additional context included in its application (Zedler and Callaway 1999, Kentula 2000, Matthews and Endress 2008). The assumption is based on the Environmental Sieve Model (ESM), where the present vegetation community is a function of the environmental conditions such as hydrology and soils (Valk 1981), and the Field of Dreams Hypothesis (FDH), which predicts that desired vegetation and fauna communities will naturally colonize a wetland after proper hydrology is established (Palmer et al. 2017). The ESM and FDH are supported from a broad application in wetland ecology that hydrophytic vegetation (*i.e.,* visual cover, prevalence index, species richness) will establish in proper hydrologic conditions. However, specific vegetation communities or species composition are not guaranteed to develop from constraints on connectivity (Suding et al. 2004, Galatowitsch 2006), local and regional species pools (Wolters et al. 2005, Palmer et al. 2017), and seed dispersal and recruitment (Galatowitsch and Van der Valk 1996, Lindig-Cisneros and Zedler 2002). For example, Mulhouse and Galatowitsch 2003 found that restored prairie wetland systems were depauperate of many common wetland sedge species after twelve years post-restoration. A misunderstanding of the application of ESM and FDH may translate into false expectations and inadequate efforts to restore wetlands.

Adoption of Functional Success to Improve Restoration Outcomes

Since compensatory mitigation and voluntary restoration have become increasingly common (Dahl 2013, Gittman et al. 2019), researchers and practitioners have sought to better understand the fundamentals of wetland ecology to improve the outcomes of restoration and creation projects (Mitsch and Wilson 1996, Zedler 1996). One of the biggest shifts in restoration thinking is the shift from Compliance Success to Functional Success in terms of monitoring, evaluation, and adaptive management of restoration projects (Kentula 2000, Neckles et al. 2002, Buchsbaum and Wigand 2012, Chmura et al. 2012). Zedler and Callaway 2000 argued compliance-based success criteria fails to consider the complexity of site history, ecosystem functions, and resiliency and bases monitoring assessments on too few indicators over too short of a timescale, which may not accurately reflect the recovery of the whole system. Functional success incorporates resilience and alternative stable-state thinking into the designs and success criteria of restoration. Functional success differentiates from compliance success by acknowledging (1) nonlinear restoration trajectories and alternative stable-state outcomes (Folke et al. 2004, Bullock et al. 2011), (2) innate site-specific variability (Brudvig et al. 2017), (3) the need for long-term monitoring and adaptive management (Suding 2011, Buchsbaum and Wigand 2012), and (4) success measured as a progression over time (Zedler and Callaway 2000).

Regime Shift and Alternative Stable-state Theory

Unlike compliance success, functional success acknowledges that ecosystems, especially restored systems, are highly dynamic and constantly influenced by multiple environmental and landscape factors. Functional success incorporates the idea that multiple alternative stable-states can exist for a restoration project, ranging across gradients of multiple biotic and abiotic factors (Folke et al. 2004, Walker and Salt 2006). For example, a restored freshwater wetland may have multiple alternative stable-states depending on the duration and frequency of seasonal flooding: tussock sedge meadow, cattail marsh, or open water pond with submerged aquatics. A restoration project, additionally, may take linear or nonlinear pathways to reach any of the alternative stable-states, since biotic and abiotic factors recover at different timescales. Importantly, the pathway of recovery may be different to the one of degradation in terms of both energy inputs and directionality (Suding and Hobbs 2009). Recovery pathways may shift unexpectedly after initial restoration efforts and diverge to undesired stable-states. Vegetation communities, for example, can shift from desired states due to invasion of exotics (Moore et al. 1999, Bosch and Matthews 2017), change in hydrologic regime (Rossell et al. 2009, Kearney et al. 2013), and intense herbivory (Stefanik and Mitsch 2012). Matthews and Spyreas 2010 observed that initially the vegetation community of restored wetlands converged towards natural references as annuals declined (Figure 2a), yet the restored wetlands over time diverged to degraded wetlands as desired and uncommon perennials never recolonized sites (Figure 2b).

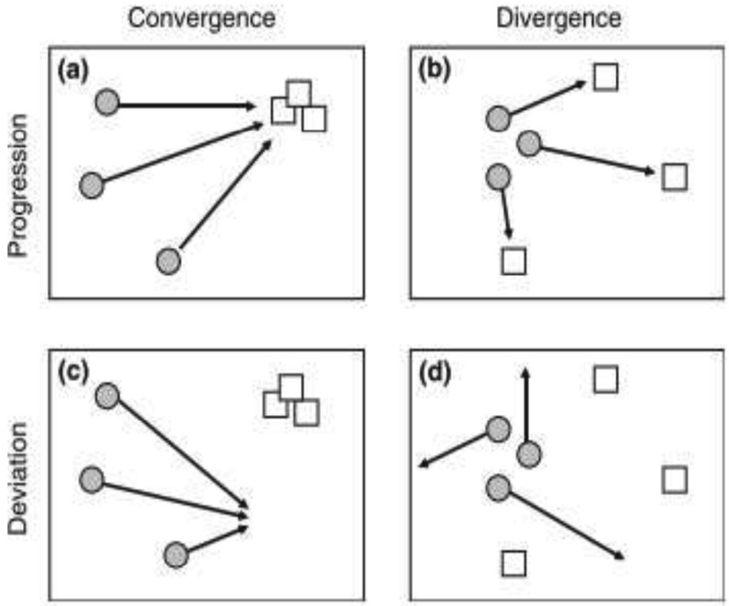

Figure 2: Matthew and Spyreas 2010's hypothetical diagram on the four possible development pathways a mitigation wetland could take post-restoration: **(a)** convergence in species composition among restorations and progression towards the target composition, **(b)** divergence among restorations and progression towards a spectrum of potential targets, **(c)** convergence among restorations but deviation away from intended targets, and **(d)** divergence among restorations and deviation from intended targets. Grey circles are initially restored wetlands, white squares are desired long-term outcomes.

Site-specific Variability

The long-term nature of wetland succession has led researchers to develop prediction models of wetland restorations to accommodate the short-term framework of compensatory mitigation such as trajectory models of individual metrics (Zedler 2000). Restoration, however, will rarely succeed in duplicating the mean response of the reference or original wetland conditions (Palmer et al. 2017), especially community composition of flora and fauna communities (Clewell and Aronson 2013). The functional success framework incorporates natural variability and constraints of predictability into existing alternative stable-state and resilience thinking. Site-specific variability has been observed to be the largest explanatory factor of restoration trajectories in wetland vegetation communities (Matthews and Endress 2010, Stefanik and Mitsch 2012). High site-specific variability can lead to a wide range of outcomes across restoration sites including one where the median response of restoration projects is within the range of success, yet many fall outside the range of success (Brudvig et al. 2017, Figure 3A - Panel B). Site-specific variability which may be difficult to measure or control in restoration projects include, site history, priority effects, inter- and intra-specific interactions (e.g., herbivory, facilitation, allelopathy, etc.), regional species pool, and succession over time (Figure 3B). For example, in the evaluation of invasive species control and native species enhancement efforts of coastal sage communities in southern California, Dickens et al. 2016 documented highly variable responses to similar management efforts and attributed it to the initial vegetation community composition, not environmental factors, as the main explanatory factor.

Practitioners can incorporate this understanding of variability by being deliberate in adopting a hierarchy of predictability for their chosen response variables in monitoring and evaluation assessments. The range of variation of a chosen metric (e.g., vegetation visual cover, avian abundance, soil carbon pool) depends on the number of factors constraining it (Brudvig et al. 2017). The hierarchy of predictability ranks the most commonly measured metrics from most-to-least predictable: vegetation structure > taxonomic diversity > functional diversity > taxonomic composition (Laughlin et al. 2017, Figure 4). In other words, one can easily predict that hydrophytic vegetation will develop with proper hydrology than the exact species composition of the community (see discussion on ESM and FDH). Compensatory mitigation, through the lens of functional success, can set more insightful expectations by (1) increasing the range of acceptable or successful values for metrics based on prior restoration efforts and (2) incorporating a hierarchy of predictability into the chosen success-based metrics.

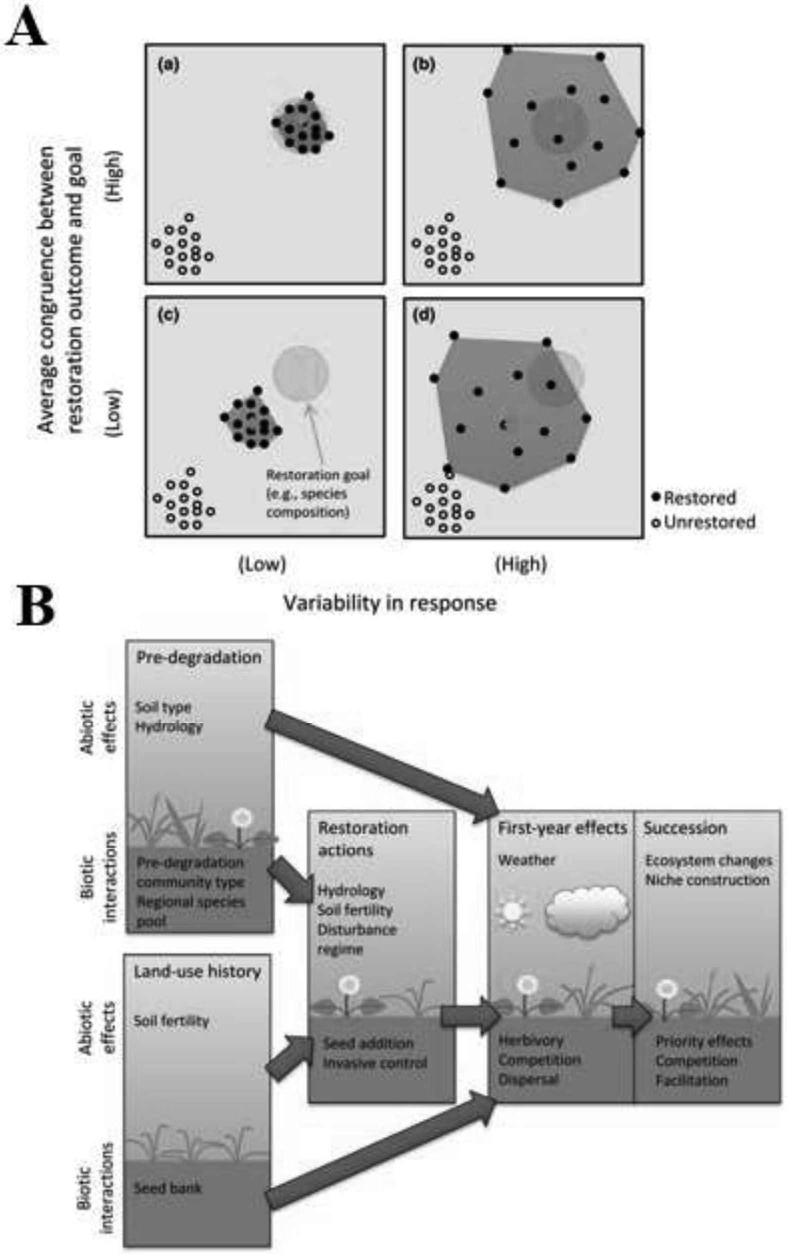

Figure 3: **(A)** Diagram of how the mean response (orange diamond) and variability (blue polygon) of restoration outcomes across multiple different restoration projects may differ from restoration goals (golden circle). Restoration practitioners typically focus on the mean response of restoration outcomes instead of interpreting the variability between restoration outcomes. **(B)** Sources of variability throughout the restoration process from pre-restoration, initial efforts, and succession over time. Both figures from Brudvig et al. 2017.

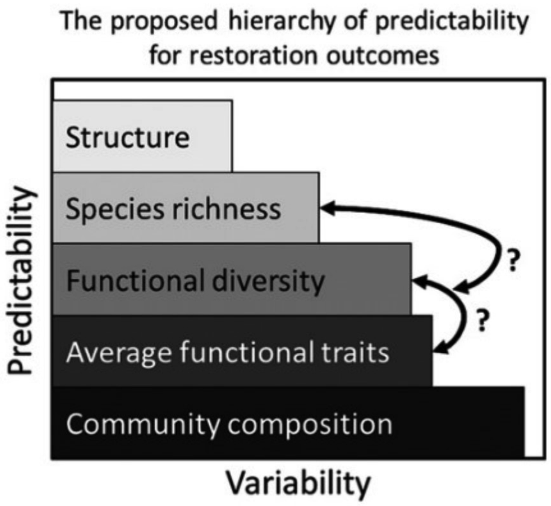

Figure 4: Proposed hierarchy of predictability in terms of both the lability to accurately predict the outcome and the variability of outcome for each metric. Questions remain about the position of functional diversity traits within the hierarchy. Figure from Laughlin et al. 2017.

Long-term Monitoring & Adaptive Management

It has been widely documented that restored systems, including wetlands, do not reach full parity of reference conditions. Restored systems, including wetlands, have been shown to only recover 50 – 75% of biotic community structure and ecosystem functions (Benayas et al. 2009, Borja et al. 2010, Moreno-Mateos et al. 2012, Jones et al. 2018), suggesting fauna and vegetation may require additional efforts to reach restoration goals. Restoration from a functional success perspective understands that outcomes of projects are highly uncertain and the initial restoration actions may not be the last actions to guarantee success (Hackney 2000, Moreno-Mateos et al. 2015).

The long-term success of mitigation wetlands is difficult to quantify under both the five-year regulatory monitoring requirement and the chronosequence experimental design in wetland research. Wetland research has relied on studying chronosequence of wetlands to identify patterns of succession for different communities (see VanRees-Siewert and Dinsmore 1996, Craft et al.

1999, DeBerry and Perry 2012). The experimental design is a quick, cheap, and efficient method of documenting long-term outcomes of wetland restoration. Successional trends of the vegetation community were primarily documented under the chronosequence method, either through large metanalyses or independent studies (Noon 1996, Matthews et al. 2009a, Moreno-Mateos et al. 2015). However, site specific variability may impede the method to strongly detect successional patterns, since multiple studies have documented site-specific variability as the largest explanatory factor to vegetation community development (Matthews and Endress 2010, Stefanik and Mitsch 2012). Additionally, the chronosequence method may not be able to document unique challenges and obstacles to ecological success, which may require intervention and additional restoration efforts.

Long-term monitoring of specific sites could develop unique case study data and identify possible mechanisms for different recovery trajectories, which is difficult under the chronosequence design (Callaway 2005). Monitoring efforts should be consistent and iterative with a focus on metrics related to restoration or research goals (Neckles et al. 2002). Consistent monitoring efforts can provide stakeholders and practitioners with information to inform future restoration efforts to boost recovery rates or remedy unexpected obstacles. Robust data sets of multiple case studies could provide stakeholders with a set of expectations or baseline trajectories for recovery rates and adaptive management needs for future projects.

Long-term adaptive management efforts may be required to ensure propagule success, colonization of desired species, and creation of vegetation communities (Hackney 2000, Aronson and Galatowitsch 2008, Stanturf et al. 2009, Buchsbaum and Wigand 2012). Stakeholders develop adaptive management actions by identifying areas of improvements, stressors (e.g., herbivory, erosion, drought, etc.), or mechanisms of failure from consistent iterative monitoring and assessments (Thom 2000, Buchsbaum and Wigand 2012). Adaptive management efforts can be highly variable and can include repeated seeding and planting of plant propagules, construction of enclosures against herbivores, installation of erosion control structures, and weeding or removal of invasives. Long-term monitoring and adaptive management of projects should be incorporated into the initial designs and budgets of wetland restoration projects in order to ensure goals are reached.

Success Viewed as a Progression

Fundamentally, functional success allows stakeholders to view the success of a project as a progression over compared to a yes/no dichotomy (Hackney 2000, Zedler and Callaway 2000). It should be acknowledged that functional success shifts the understanding of ecological restoration from a zero-sum effort of compliance success to the measurement of “what has been gained” compared to negative controls or degraded areas and “how much recovery remains” compared to reference systems (Moore et al. 2010, Lilian et al. 2021). For example, the Restoration Performance Index groups functional areas or metrics to assess restoration projects on a score of zero to one, with one as the site reaching parity with a local reference (Figure 5). Restoration practitioners should apply progressive evaluations (e.g., RPI; see Raposa et al. 2018) alongside statistical comparisons (e.g., ANOVA, student t test, PERMANOVA,) to have more concrete answers to “what has been gained” and “how much recovery remains” in restoration projects. Additionally, measurements of progression, like the RPI, may be easier to communicate the state of a restoration project to the public and identify adaptive management needs by stakeholders.

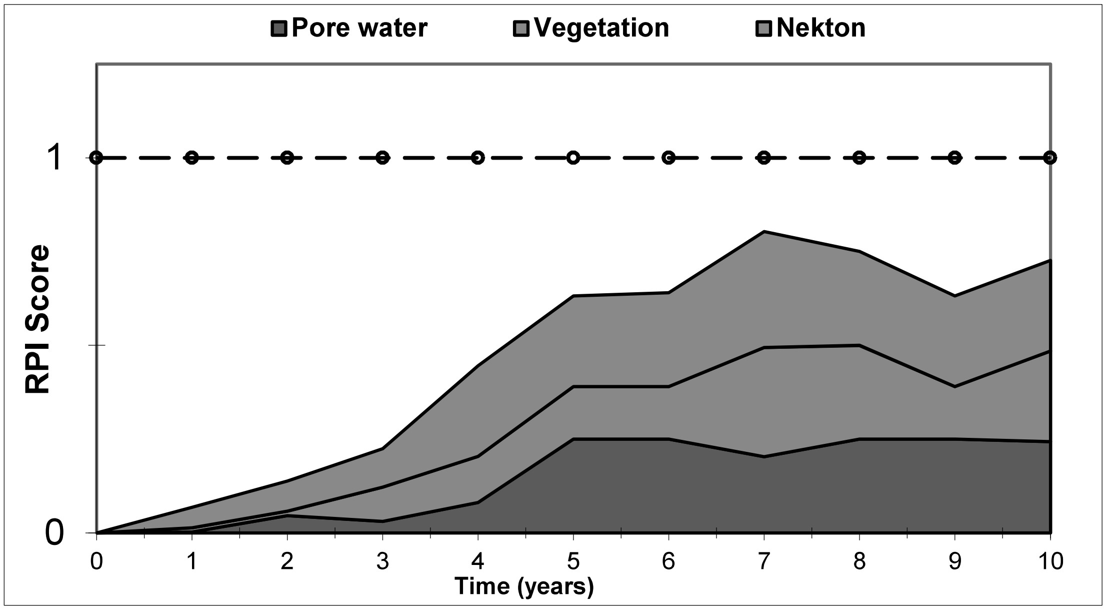

Figure 5: A hypothetical score of the Restoration Performance Index (RPI) of a restored salt marsh over time based on selected metrics of soils, habitat structure, and fauna. The ecosystem structure and function is relativized to a score from 0 – 1, where 1 (dashed line) is complete parity with a selected local reference marsh. Success is based on the stakeholder defined goals of the project. Figure from Moore et al. 2009.

Conclusion

Tidal and freshwater wetland area significantly declined in the United States since colonization of European settlers, with significant losses in the twentieth century. The loss of wetlands, whether tidal or freshwater, on the landscape can translate into large declines in ecosystem services including water filtration, flood control, groundwater recharge, and fisheries support. In the second half of the twentieth century, state and federal governments' roles transitioned from promotion of wetland destruction to conservation, mitigation, and restoration. The CWA, its reauthorizations, and presidential executive orders for "no net loss" transformed the federal government's efforts by enacting compensatory mitigation, financial disincentives, active wetland mapping, and financial support for conservation and restoration projects. As the fields of wetland ecology and restoration ecology have developed over the past four decades since the CWA and Section 404 program, the framework of compliance success and outcomes under compensatory mitigation have come under scrutiny including the zero-sum success evaluations, narrow univariate evaluation metrics, and a five-year monitoring period. To improve the outcomes of mitigation and wetland restoration in general, practitioners should adopt elements of functional success – alternative stable-state theory, site-specific variability, long-term monitoring, adaptive management, and view of success as progression- into the design, monitoring, and evaluation of projects.

This master's thesis applies a functional success perspective in the monitoring, evaluation, and adaptive management of two unique wetland creation case studies, both products of compensatory mitigation. The short-term trajectory of salt marsh living shorelines and the long-term trajectory of a freshwater wetland were characterized to improve the design and expectations of future projects. Although the wetlands in question are quite different (saline vs. freshwater), the fundamentals of wetland ecology, botany, and restoration ecology are applicable to the sampling and analysis of the restoration trajectories and assessment of success (Mitsch and Gosselink 2015). In Chapter 2, I monitored the recovery of vegetation, nekton, and biogeochemical processes at three salt marsh restoration sites to establish baseline trajectories and evaluated the success of the projects using the RPI to better understand the progression of the recovery. Additionally, I compiled possible adaptive management needs for the restoration sites which apply to the geographic context of northern New England. In Chapter 3, I utilized long-term data set spanning 35 years of community composition and wetland community distribution of a created freshwater wetland. I analyzed how successional trajectories of both the species composition and the distribution of wetland communities shifted over time to improve expectations of future mitigation projects. Although "restoration may always be a gamble", as Suding 2011 sagely remarked, incremental advances in knowledge of wetland ecology and adoptions of lessons learned will better our odds of successful wetland mitigation and restoration.

CHAPTER II

LIVING SHORELINES IN NEW HAMPSHIRE: IF YOU BUILD IT, WILL THEY COME?

Introduction

Historically, coastal communities in the United States have possessed few tools except grey infrastructure (e.g., seawalls, bulkheads, and riprap revetments) to combat shoreline erosion and coastal flooding and protect upland infrastructure, property, and livelihoods. Widespread implementation of grey infrastructure has resulted in the armoring of 14% or 22,000 km of the contiguous United States shoreline in the past century (Gittman et al. 2015). The ease of permitting and often approved engineering knowledge allowed coastal communities to consistently turn to traditional grey infrastructure (Sutton-Grier et al. 2015). However, this practice may be inadequate for the consequences of sea level rise: increasing shoreline erosion, coastal flooding, and intense storm events. Hardened shorelines weaken over time, require greater maintenance costs than natural shorelines, cannot adapt to changing environmental conditions, and lack resilience from short and long-term disturbances (Gittman et al. 2014, Scyphers et al. 2015, Sutton-Grier et al. 2015, Smith et al. 2017, 2018).

Additionally, grey infrastructure contributes to declines of faunal and floral communities by disrupting influencing environmental factors. The construction of a concrete or riprap barrier immediately disrupts sediment transport (Bozek and Burdick 2005), geochemical processes (O'Meara et al. 2015), and hydrologic regimes (Plant and Griggs 1992), and increases seaward scour (Kornis et al. 2018b). By redirecting wave energy seaward, they scour both intertidal systems (Defeo et al. 2009, Kornis et al. 2017) and prevent the upland migration of retreating salt marshes and beach dunes, leading to the phenomenon of "costal squeeze" (Doody 2004, Pontee 2013). The resulting ecological consequences are steep declines in nearby flora and fauna diversity and abundance (Bozek and Burdick 2005, Bilkovic and Roggero 2008, Gittman et al. 2016b, Dugan et al. 2018), loss of nekton and bivalve nurseries (Bilkovic and Roggero 2008, Balouskus and Targett 2016, Kornis et al. 2017), prevention of subtidal-upland migrations of crabs (Gehman et al. 2018), and shifts in food web structures (Seitz et al. 2006, Kornis et al. 2018b). The loss of coastal ecosystems can translate into reduced or total loss of valuable ecosystem services including nurseries for commercially important fish and shellfish (Boesch and Turner 1984, Minello and Webb 1997, Minello et al. 2003, Johnson and Eggleston 2010), denitrification (O'Meara et al. 2015), carbon sequestration (Pendleton et al. 2012, Drake et al. 2015), and recreation (Barbier et al. 2016, Arkema et al. 2017). Further, as grey infrastructure disintegrates, large financial investments will be required to maintain natural services such as wave attenuation (Morgan et al. 2009, Gedan et al. 2011) and coastal flood protection (Arkema et al. 2013).

To improve both shoreline resilience and ecosystem services of coastal habitats, coastal communities, regulators, and non-profit organizations have promoted the construction of nature-based solutions for coastal erosion over traditional grey infrastructure since the mid-2000's (National Research Council 2007, Bridges et al. 2014, Restore America's Estuaries 2015). Living shorelines have been promoted as a soft engineering method that utilizes ecological restoration of wetland systems (e.g., salt marshes, tidal wetlands, oyster reefs, and sand dunes) to stabilize shorelines, connect aquatic and terrestrial systems, and improve ecosystem services (Bilkovic et al. 2017b). Living shorelines combine the inherent resiliency of coastal ecosystems with the sheer strength and wave attenuation properties of hardened structures.

One widely implemented living shoreline design is the creation of a fringe salt marsh and the placement of a 3 – 4 ft sill composed of root wads, riprap boulders, or coir fiber logs at the low mean water level (*i.e.,* low marsh edge) (O'Donnell 2017, Woods Hole Group 2017). The sill prevents immediate loss of fill sediment and provides the salt marsh vegetation time to root out, stabilize the sediment, and develop peat. Additionally, the sill attenuates wave energy and may reduce vegetation propagule mortality immediate post-construction.

Living shorelines have been demonstrated to be successful tools for shoreline stabilization. Projects in North Carolina, especially projects older than 5 years, accrete sediment and outpace sea level rise compared to drowning natural marshes (Polk and Eulie 2018). Additionally, living shoreline projects enhanced coastal resiliency by suffering less damage from hurricanes than constructed bulkheads and recovering both vegetation and elevation more quickly than natural marshes (Gittman et al. 2014, Smith et al. 2018).

Research on living shorelines, primarily in the mid-Atlantic and Southeast, has focused on quantifying the recovery rate of individual ecosystem structure, functions, services. Dominant salt marsh grasses, *Spartina alterniflora* and *S. patens*, have statistical equivalency in stem density and canopy height compared with reference fringe marshes within two to three growing seasons (Bilkovic and Mitchell 2013, Gittman et al. 2016a). Nekton and crustacean abundances within the intertidal also reach or exceed densities of reference marshes within 10 years (Currin et al. 2008, Balouskus and Targett 2016). Epibenthic filter feeders, which are important for improving water quality, were found at greater densities on the salt marsh surface in living shorelines with riprap sills than natural marshes (Gittman et al. 2016a). Nekton communities possessed a larger composition of juveniles and transient fish at living shorelines with riprap sills than references

(Currin et al. 2008, Gittman et al. 2016a). The construction of the riprap sill in a soft-sediment benthos shifts the invertebrate community from infaunal deposit feeders to epibenthic filter feeders (Bilkovic and Mitchell 2013, Davenport et al. 2018). Additionally, living shorelines are locations of rapid carbon storage (Davis et al. 2015) and denitrification (Onorevole et al. 2018) immediately after construction, which can aid efforts in combatting climate change and eutrophication.

The goal of ecological restoration is the return of resilient ecosystem structure, function, and services relative to a comparable condition. Individual studies have focused on only a few, specific parameters for living shoreline projects to build case study data for the new coastal resiliency approach. As living shorelines become more widely implemented, coastal communities, regulatory agencies, and researchers will need to monitor for a variety of different parameters to properly evaluate projects and apply possible adaptive management strategies. Monitoring strategies for wetland restoration have shifted from predictive tools like performance curves (Zedler and Callaway 1999, Morgan and Short 2002) to assessments of restoration performance over time based on prior conditions and reference systems (Roman et al. 2002, Chmura et al. 2012, Raposa et al. 2018). The Restoration Performance Index (RPI), first developed by Moore et al. 2010 to compare different salt marsh restoration methods across New England, evaluates restoration projects on the principles of functional success and resiliency. The RPI can incorporate the results of differing monitoring protocols, construction methods, or project locations by relativizing the sampling results of restoration projects against selected local references (target conditions), no action controls, and prior conditions. The tool equally weights each parameter, chosen by stakeholders, to remove subjective bias. The RPI calculates a score from zero to one representing the progress of a restoration project from before/no action control conditions to the desired reference. The tool allows researchers to evaluate the success of a project on a continuum, rather than a “yes/no” dichotomy, and easily communicate the progress of projects to communities and stakeholders (Chmura et al. 2012).

Living shoreline projects have been gaining traction as a tool to improve coastal resiliency and abate shoreline erosion. The tool has been widely constructed in the Gulf of Mexico, Southeast, and Mid-Atlantic (*i.e.,* Chesapeake Bay) (NOAA Restoration Center) and the first projects of New England within the past few years (O’Donnell 2017). Published data on the performance of living shorelines is centered on North Carolina, Delaware, and the Chesapeake Bay region, where the growing season is longer (Kirwan et al. 2009) and ice rafting is not a concern compared to New England (Hardwick-Witman 1985). Since 2016, the New Hampshire Department of Environmental Services has shifted their philosophy of shoreline stabilization from traditional armoring methods to a preference for living shorelines (see NH 604.01(c)). As part of the department’s long-term efforts, three living shoreline projects were constructed in Great Bay Estuary (NH) since 2016 to restore salt marsh habitat at degraded or eroding shorelines. The goal of this research was to evaluate the short-term restoration performance of living shorelines to better understand their potential in northern New England and provide a baseline trajectory for resource managers of future projects. The vegetation, nekton, and biogeochemical processes were monitored over two years and restoration trajectories were calculated across the project age of the living shorelines. Restoration trajectory and adaptive management steps were carefully documented to provide insight for current and future living shoreline projects.

Methods

Site Description

Three living shorelines were constructed in the Great Bay Estuary (NH) within protected coves: North Mill Pond (NMP; 43.0761, -70.7661) in 2016 on the Piscataqua River, Cutts Cove (CC; 43.08442, -70.6568) in 2018 on the Piscataqua River, and Wagon Hill Farm (WHF; 43.1249, -70.8721) in 2019 on the Oyster River (Figure 6 – Left). The living shorelines were constructed similarly including regrading of slopes and introduced sediment within the intertidal zone, plantings of *Spartina alterniflora, S. patens, Distchlis spicata,* and *Juncus gerardii* plugs, and erecting a 3 – 5 ft tall riprap sill at the lower edge of the vegetated marsh. NMP was only built with a coir fiber sill for erosion control, which had completely decomposed by 2019, and only planted with *S. alterniflora* and *S. patens*. A before-after-control-impact (BACI) experimental design was implemented at each site as each possessed a local reference marsh, and a section of a no action control shoreline (Figure 6 - Right). The pre-restoration shorelines at CC and NMP were mudflats in front of historic riprap armoring of steep berms and WHF's pre-restoration shoreline was a rapidly eroding fringe salt marsh adjacent to the living shoreline. Additionally, a salt marsh along the Bellamy River adjacent to Portsmouth Christian Academy (PCA; 43.1585, -70.8546) was included as an additional pristine reference site due to high species richness, presence of a salt panne microhabitat, and well-defined high marsh – upland border.

Ecosystem Structure Sampling

The shorelines were divided between low and high marsh zones, based on vegetation patterns, and ten plots were randomly located within each marsh zone. The pre-restoration shorelines of NMP and CC only represented low marsh due to the elevation of the mudflats compared to surrounding salt marshes. Conversely, the pre-restoration shoreline of WHF was considered only high marsh due to complete erosional loss of the low marsh. Vegetation and pore water chemistry were sampled at plots during low tide in August – September of 2019 and 2020. All vegetation was identified and cover was visually estimated for each species to the nearest 1% within 0.5 m^2 quadrats (Neckles et al. 2002). Epibenthic invertebrates found on vegetation or sediment surface were noted. The average halophyte cover and species richness per plot was calculated for the low and high marsh zones. An additional ten random 0.5 m^2 plots were sampled

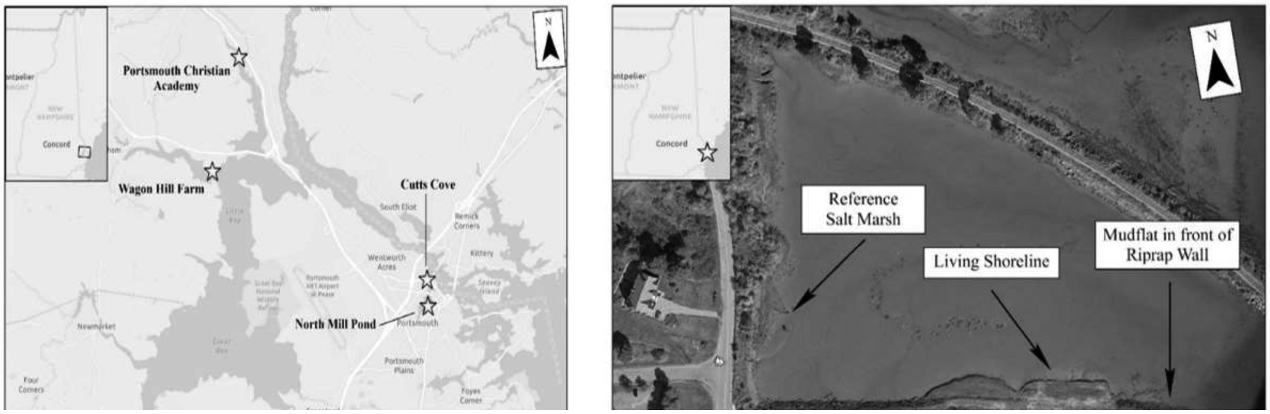

Figure 6: Left - Local site map of living shoreline projects in the Great Bay Estuary of New Hampshire. Study sites included Wagon Hill Farm (Durham) and Cutts Cove and North Mill Pond (Portsmouth). A reference site was located on the Bellamy River adjacent to the Portsmouth Christian Academy (Dover). **Right -** The site map of the Cutts Cove in Portsmouth, New Hampshire as an example of the BACI design at each site. Each site included a reference salt marsh, living shoreline, and pre-restoration comparison as the control. Mud flats in front of riprap armoring functioned as the pre-restoration comparison for Cutts Cove and North Mill Pond. An eroding salt marsh was the pre-restoration comparison for Wagon Hill Farm.

on the riprap and coir fiber sills of the living shorelines to estimate the colonization of algae and macroinvertebrates (O'Connor et al. 2011). All algae species were identified to the species level and cover was visually estimated for each. All macroinvertebrates were identified and counted within all crevices of the sills within the two-dimensional boundaries of the quadrat.

Pore water (*e.g.*, water held within pore spaced in the root zone of the substrate) was collected using a stainless-steel sipper with a plastic syringe inserted between 5 – 20 cm into the root zone at low tide (Portnoy and Giblin 1997a, 1997b). Pore water was immediately analyzed in the field for reduction-oxidation potential (redox potential, mV) with an Orion 290A+ multimeter and redox probe equipped with a platinum electrode (Thermofisher, Waltham, MA). Collected pore water samples of 10 mL were analyzed in the lab for salinity (psu) with a handheld salinity refractometer and pH with an Orion 5000 multimeter and pH probe, respectively (Thermofisher, Waltham, MA). A subsample of 0.5 mL of pore water was fixed in 20% Zinc Acetate solution in the field and then stored at 4°C. Sulfide content (mM) was determined with a colormetric analysis using Cline's reagent (Cline 1969) and a LaMotte SmartSpec Spectrophotometer (LaMotte, Chestertown, MD). The lack of peat development and permeability of sandy soils used to construct the living shorelines prevented pore water collection from most plots ("dry plots"). Soil reduction-oxidation potential was measured for all dry plots with brightened platinum electrodes inserted to a depth of 10 cm and allowed to equilibrate in situ for 45 min before measuring with a double junction reference electrode and millivolt meter (McKee et al. 1988). A correction of +244.0 mV was added to electrode-based redox values to arrive at in-situ redox potential and allow for comparison with pore water redox values (McKee et al. 1988).

To better understand the early development of wetland soil characteristics, soil cores were sampled in the living shorelines to monitor the water content (% water), bulk density (g cm^{-1}), and organic matter content (g). Soil cores (10 cm in length) were just taken outside of each 0.5 m^2 plot with 3 cm diameter corer in late September – early October 2020. The compaction rate of each core was immediately assessed and the actual 10 cm of the soil column kept. Cores were weighed at the lab and stored at 4°C. Cores were dried at 50°C until the weight of the samples remained constant and then reweighed. Loss on ignition was conducted on a subsample of each core at 450°C for four hours to calculate the organic matter (Craft et al. 1991, Morgan et al. 2009).

Use of the salt marsh surface by the nekton community was assessed once over two consecutive spring tide events in mid-September to early October of 2020 (IACUC #190706, Appendix IV). Nekton sampling at NMP and PCA shifted to early October during the second spring tide event, due to Hurricane Teddy (September 23rd, 2020) simulating king tide conditions during the first spring tide event. The early fall timing of sampling was designed to capture a snapshot of the density and size class of resident nekton in the Gulf of Maine such as *Fundulus heteroclitus* (Mummichog) and *Fundulus majalis* (Striped killifish) (Drociak and Bottitta 2005, Raposa and Roman 2006). The nekton community was sampled with unbaited *Gee*'*s* standard wire mesh minnow traps, a common method of sampling salt marsh nekton community (West and Zedler 2000, Carlisle et al. 2002). Minnow traps were constructed out of 6.4 mm square mesh with 20 mm diameter openings. *Gee's* traps are size selective for fish and crustaceans 40 – 110 mm in total length (West and Zedler 2000, Rudershausen et al. 2016). Ten minnow traps were set equally spaced out at each shoreline. Traps were left out for 30 minutes with at least three-quarters of the trap submerged the entire time (Kneib and Craig 2001). All nekton were identified to species and sex and measured for total length (mm). The width of the carapace of crustaceans was measured for total length. For traps with high catch rates, a subset of the first ten males, females, and juveniles of each species were measured for total length. *Fundulus* sp. less than 45 mm were classified as juvenile. The CC and NMP sites were monitored in the low marsh, as there was not high marsh in the no action controls. The PCA and WHF sites were monitored for both the low and high marsh to account for the no action control shoreline at WHF and for comparison across living shorelines.

Ecosystem Function Sampling

Erosion control and decomposition rate were measured as important ecosystem functions for each site and treatment. Erosion control was observed as the loss or gain of sediment (cm) based on permanent stakes in the low marsh zone. For vertical or near vertical shorelines, such as highly eroded fringe marshes or stream banks, three to four rebar rods were hammered flush across the shoreline. For gradually sloped fringe marshes, oak stakes with roofing nails hammered into the top were hammered flush with the soil roughly 1 – 2 m upland from the furthest extent of low marsh vegetation across each shoreline. Stakes and rebar roads were installed in September - October 2019 at every living shoreline, reference, and no action control shorelines. In fall of 2020 the length of the rods or stakes exposed or buried were measured to determine the annual sediment erosion or deposition rates, respectively.

The rate of decomposition was measured over one growing season in 2020. Senesced *Spartina alterniflora* stems and leaves were collected from Adams Point, New Hampshire in October 2019 and dried at 50°C until the weight did not change. Litter bags were constructed with

6 mm mesh fabric and sized at 15 cm X 10 cm (Rubio and Childers 2006). The mesh size was selected to permit macroinvertebrate access like *Melampus bidentatus* (Coffee bean snail) and *Gammarus* sp. (Scud), which play a large role in the brown food web in salt marshes in New England (Zimmer et al. 2014). Litter bags were stuffed with roughly 10 g of dried *S. alterniflora* stems and leaves and weighed for approximate mass in the field when deployed. Litter bags were zip-tied to prevent shredded and smaller pieces of stems from being lost. At each shoreline, eight litter bags were deployed in the low marsh in dense *S. alterniflora* cover in early June 2020. Two litter bags were collected at the following intervals: 1–, 2–, 3–, 5–month intervals from each site. Collected litter bags were gently washed to remove mud, dried at 50°C until no weight change, and then weighed to estimate percent mass loss. The rate of decomposition (k, day^{-1}) was calculated for each site for the slope of percent loss over time (Wilson et al. 1986). An exponential decay regression was fit to decomposition data of sites with at least 3 months of data (> 99 days) to calculate the rate of decomposition (k):

$$Percent\ Mass\ Remaining = a * e^{-k*time}$$

where time is the days passed since deployment and a is a scaling factor.

Data Analysis

The RPI quantified the short-term recovery of the living shorelines based on its progress from the pre-restoration shoreline to the reference. The RPI is an objective and flexible evaluation tool, whose core groups and sub-group metrics can be modified according to a project’s restoration goals (Moore et al. 2010, Raposa et al. 2018). The RPI score for an individual parameters is calculated by relativizing the mean value of the living shoreline by the pre-restoration shoreline and compared to the selected reference (Moore et al. 2010):

$$RPI_{Salinity} = \frac{(T_{Restored} - T_{Pre-Restorati}\ \ )}{(T_{Reference} - T_{Pre-Restoration})}$$

The individual metrics (*i.e.*, salinity and pH) are equally weighted within each core group (*i.e.,* pore water chemistry) and summed to calculate a RPI score for the core group. The core groups are then equally weighted and summed for a total RPI score from zero to one, where zero indicates the restoration projection has not made any progress and one means parity with the reference.

The RPI model was modified from Moore et al. 2009 to incorporate restoration goals based on identified salt marsh values: habitat structure, biogeochemical processes, and fauna use (Figure 7; National Research Council 1992, Neckles et al. 2002). Ecosystem functions of decomposition and erosion rates were not included in the RPI due to incomplete data sets. Individual vegetation parameters included (1) halophyte cover and (2) species richness. Halophyte percent cover was weighted by wetland zone (low and high marsh) and species richness only in the high marsh. Pore water chemistry parameters included (1) salinity, (2) redox potential, (3) pH, and (4) sulfide content. Pore water chemistry parameters were weighted by wetland zone (low and high marsh). The nekton parameters included (1) *Fundulus heteroclitus* trap catch rate and (2) *F. heteroclitus* adult length. The nekton parameters were compared either in the low marsh for CC and NMP or the high marsh for WHF. The RPI scores of the 2019 growing season were calculated with vegetation and pore water chemistry parameters. Nekton was added as a third core group in 2020.

Prior assessments with the RPI on salt marsh systems added hydrology as a core group (e.g., tide depth, marsh surface inundation), since the focus of the restorations were the removal of impediments to proper tidal elevations such as culvert restrictions or berm impoundments (Moore et al. 2010, Raposa et al. 2018). Give that restoration efforts of living shoreline were not directed at correcting the hydrologic regime, this core group was not included into this RPI assessment. Hydrology has been shown to recover incrementally over time without any distinction between restoration activity (Raposa et al. 2018), although incomplete recovery (< 75% RPI score) after 10 years is not uncommon. If monitored, the hydrology would not have boosted the RPI score of the projects (Moore et al. 2010), since the hydrology of the living shoreline, reference, and no action shorelines for each site can be assumed to be the same.

The total RPI and the vegetation metric score were analyzed across a consequence of project ages (WHF: 0 – 1, CC: 1 – 2, NMP: 3 – 4 years) to quantify short-term restoration trajectories. A three parameter logistic regression was applied across the project age to the total RPI score and vegetation metric RPI scores of both reference comparisons (see Morgan and Short 2002, Evans and Short 2005, Matthews et al. 2009):

$$RPI\ Score = \frac{a}{(1 + b * e^{(-c * Project\ Age)})}$$

where a is the slope, b is a scaling factor, and c is the inflection point of the logistic curve. Parameter c represents the maximum threshold for potential RPI scores based on past performance. An additional linear regression was conducted on the total RPI score of the PCA reference comparison based on the shape of the data. Pairwise differences were calculated for each site across both years of all core group scores between the local and PCA reference to estimate the influence of references on the RPI.

Linear regression analyses were performed to determine possible short-term trajectories of individual metrics of (1) halophyte visual cover (blocked by sampling year), (2) halophyte species richness, and (3) pore water redox potential over living shoreline age. Linear regressions were not feasible for evaluation of nekton since sampling only occurred in 2020. Two-tailed t-tests were compared the difference of individual metrics between living shorelines and paired local references: (1) halophyte visual cover (low and high marsh pooled), (2) high marsh species richness, (3) reduction-oxidation potential (low and high marsh separate), (4) nekton catch per trap (low marsh), and (5) nekton adult length (low marsh). One-way ANOVA and post-hoc Tukey's HSD post-hoc test, when necessary, were used to compare *F. heteroclitus* count and adult length between living shorelines projects to investigate differences in sill type and age. A two-tailed paired t-test was used on halophyte cover of local references between 2019 and 2020 to determine if sampling year had an effect on visual cover estimations. The ten vegetation plots of the low marsh of each living shoreline were divided in half based on distance to the seaward edge. Independent two-way t-tests were used to test (1) the difference of halophyte cover between seaward and landward plots across 2019 and 2020 and (2) the difference of algae cover and barnacle density between the CC and WHF riprap sills after one growing post-restoration. Linear regression analyses were performed to determine correlations of aggregated gastropod and crab densities to (1) project age and (2) algae cover. The following transformations were conducted to meet the assumptions of normality: arcsin (sqrt (algae cover)) and $\log_{10}$(*F. heteroclitus* count + 1) (McCune and Grace 2002). Data analyses were conducted in JMP 15 (SAS Institute, Inc., Cary, NC) and figures created using *ggplot2* and *patchwork* packages in R (R Core Team 2013).

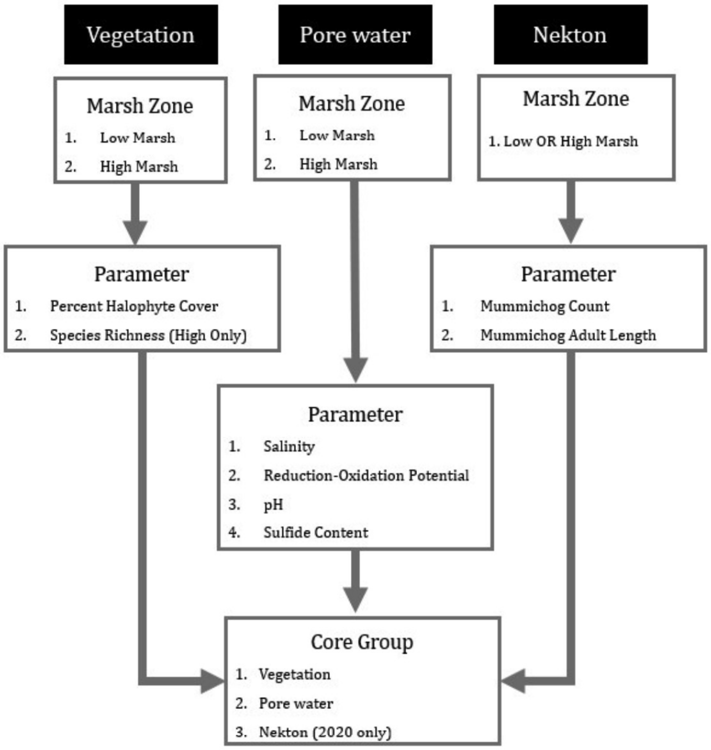

Figure 7: Graphical representation of the RPI calculations. The RPI score for the nekton was evaluated in the low marsh (Cutts Cove and North Mill Pond) or in the high marsh (Wagon Hill Farm) due to constraints by the control shoreline. Figure adapted from Raposa et al. 2018.

Results

The living shorelines recovered roughly 52% of the metrics of ecosystem structure, function, and services within four years after construction when compared to both the local and PCA reference marshes. Site-specific RPI score comparisons between 2019 and 2020 showed conflicting results with strong recovery at WHF and stagnation at NMP. Notably, WHF improved by 0.46 - 0.51 within one year after construction (Tables 2 - 3, Figure 8, Appendix I). The RPI scores of the local reference comparison followed a logistical trajectory with an upper RPI asymptote of 0.52 $\pm$ 0.04 (df = 3, $R^2 = 0.94$, RMSE = 0.06; Figure 8). The RPI scores of the PCA reference followed a moderate, nonsignificant positive linear trend over project age ($F_{1,5} = 3.56$,

$R^2 = 0.47$, $p = 0.13$) with little visual evidence of an upper asymptote (df = 3, $R^2 = 0.71$, RMSE = 0.13, Figure 8). The choice of references had a minor impact on the total RPI scores as the average pairwise comparison differed by 0.06 ± 0.03 (Table 4). The only RPI score to have differed by more than 0.1 was CC in 2020, where the nekton metric score was 62% greater when compared to the local reference than the PCA reference. Additionally, the RPI score of CC remained unchanged in the PCA comparison while improving considerably in the local reference comparison.

Table 2: Calculated scores of the weighted individual and total RPI score for each living shoreline project across both years of sampling. RPI scores are shown for both local and PCA reference comparisons. Individual metrics scores are weighted by number of core groups.

	2019			2020			
Site	Veg	Pore water	Total	Veg	Pore water	Nekton	Total
Local Reference							
Wagon Hill Farm	0.03	0.00	**0.03**	0.15	0.04	0.30	**0.49**
Cutts Cove	0.34	0.00	**0.34**	0.22	0.00	0.29	**0.51**
North Mill Pond	0.31	0.21	**0.52**	0.24	0.08	0.22	**0.54**
PCA Reference							
Wagon Hill Farm	0.04	0.00	**0.04**	0.21	0.00	0.31	**0.55**
Cutts Cove	0.31	0.00	**0.31**	0.22	0.00	0.11	**0.32**
North Mill Pond	0.39	0.07	**0.46**	0.27	0.09	0.18	**0.54**

Table 3: Unweighted individual RPI metric scores for each living shoreline across both years of sampling. Unweighted scores are calculated from 0 - 1. RPI scores are shown for both comparisons of local and PCA reference comparisons to the living shorelines.

	2019		2020		
Site	Veg	Pore water	Veg	Pore water	Nekton
Local Reference					
Wagon Hill Farm	0.06	0.00	0.45	0.12	0.91
Cutts Cove	0.67	0.00	0.67	0.00	0.87
North Mill Pond	0.62	0.42	0.72	0.25	0.65
PCA Reference					
Wagon Hill Farm	0.07	0.00	0.62	0.00	0.92
Cutts Cove	0.63	0.00	0.65	0.00	0.33
North Mill Pond	0.78	0.14	0.80	0.28	0.53

Table 4: The difference in RPI scores between the comparisons of living shoreline to local reference and living shoreline to PCA reference. Differences are reported as absolute values. Individual metric scores are unweighted. Highlighted cells are differences greater than 0.15.

	2019			2020			
Site	Veg	Pore water	Total	Veg	Pore water	Nekton	Total
Wagon Hill Farm	0.01	0.00	0.00	0.17	0.12	0.01	0.06
Cutts Cove	0.04	0.00	0.02	0.02	0.00	0.54	0.19
North Mill Pond	0.16	0.28	0.06	0.08	0.02	0.12	0.01
Average	0.07	0.09	0.03	0.09	0.05	0.23	0.09

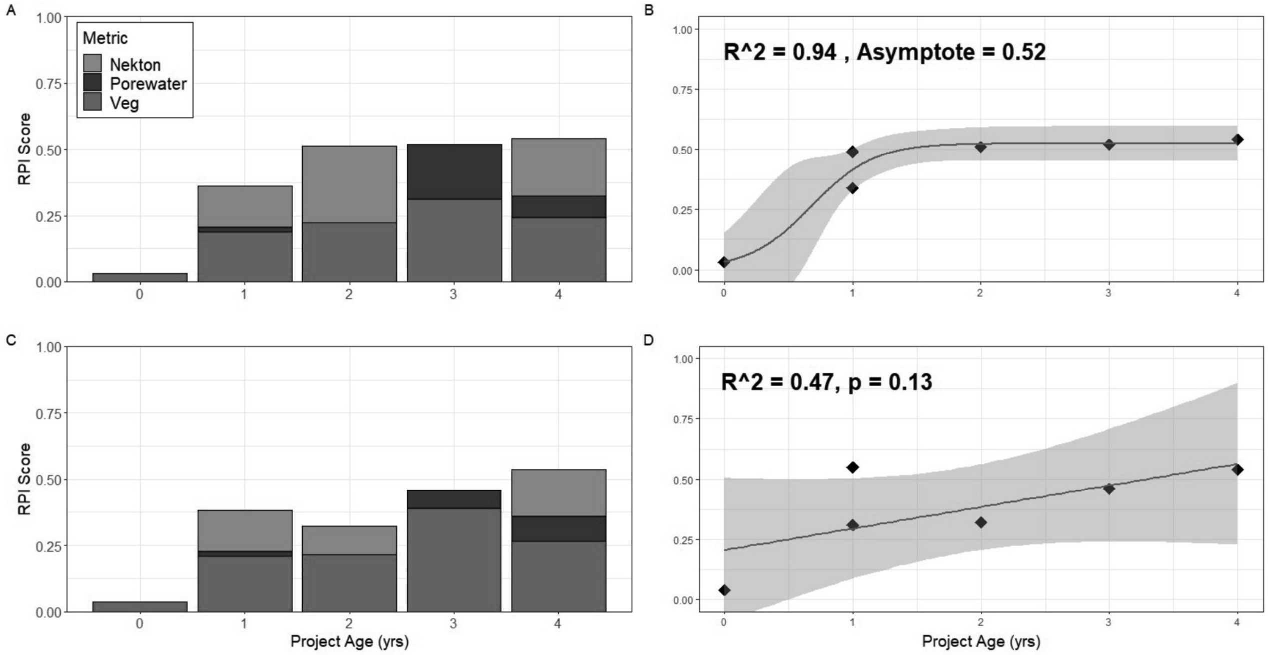

Figure 8: (A) Chronosequence of RPI scores over living shoreline project age of the local reference comparison. RPI score for year 1 was the average of the unweighted metric scores of WHF and CC. **(B)** Three parameter logistic regression of the RPI scores over living project age. Every year for reach site are shown as black diamonds. Grey intervals are ± standard error of regression. **(C)** Chronosequence of RPI scores over living shoreline project age of the PCA reference comparison. RPI score for year 1 was the average of the unweighted metric scores of WHF and CC, and **(D)** Linear regression of RPI score over living shoreline project age of the PCA reference comparison. Every year for reach site are shown as black diamonds. Grey intervals are ± standard error of regression.

The vegetation metric RPI scores in both reference comparisons followed similar logistical trends over project age with upper thresholds near an unweighted RPI score of 0.71 (Local: df = 3, $R^2 = 0.91$, RMSE = 0.099, Upper Asymptote = 0.67 ± 0.06; PCA: df = 3, $R^2 = 0.96$, RMSE = 0.065, Upper Asymptote = 0.75 ± 0.04; Figure 9). There was a moderate, non-significant increase in plot-level species richness in the high marsh of living shorelines over project age ($F_{1,5} = 5.61$, $R^2 = 0.58$, p = 0.077), with a high of 4.3 ± 0.8 species per plot at NMP after three years (Appendix I). All living shoreline sites had statistically similar high marsh species richness to the local references in 2020 (WHF: t $\text{ratio}_{19} = 1.30$, $p = 0.215$; CC: t $\text{ratio}_{19} = 0.474$, $p = 0.642$; NMP: t $\text{ratio}_{19} = 1.34$, $p = 0.197$). The increase in high marsh species richness resulted from natural colonization of halophytes from the local species pool. The NMP living shoreline had the same species composition as the local reference in 2019 including *Plantago maritima*, *Limonium nashii*, and *Puccinellia fasciculata*. Quick dispersal halophytes of *Salicornia* sp. and *Atriplex patula* colonized WHF within one year. NMP living shoreline was the only project to have statistically similar halophyte cover of the local reference (WHF: t $\text{ratio}_{39} = 9.14$, $p < 0.001$; CC: t $\text{ratio}_{39} = 6.11$, $p < 0.001$; NMP: t $\text{ratio}_{39} = 1.66$, $p = 0.11$; Figure 10). There was a non-significant, weak trend for halophyte cover over project age ($F_{1,\,5} = 0.435$, $R^2 = 0.098$, $p = 0.546$), even when accounting for the significant overestimation of halophyte cover 2019 than 2020 (t $\text{ratio}_{198} = -11.71$, $p < 0.001$). Additionally, the five most seaward plots in the low marshes of living shorelines had approximately 25% less halophyte vegetation cover than the landward plots across both years (t $\text{ratio}_{57} = -4.13$, $p < 0.001$).

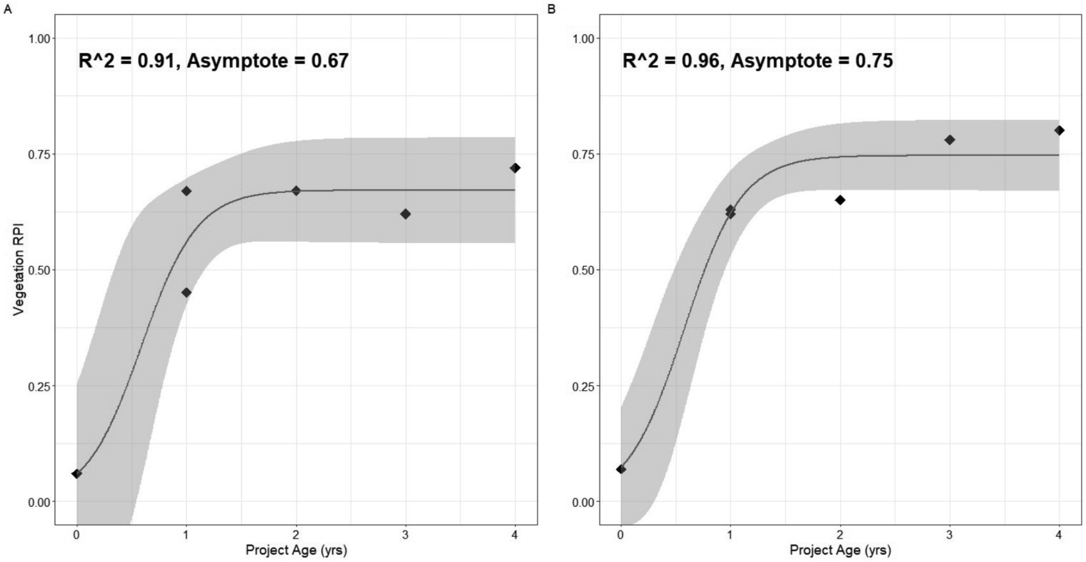

Figure 9: Three parameter logistic regressions of the vegetation RPI metric score for **(A)** local reference comparison and **(B)** PCA reference comparison. Every year for reach site are shown as black diamonds. Grey intervals are ± standard error.

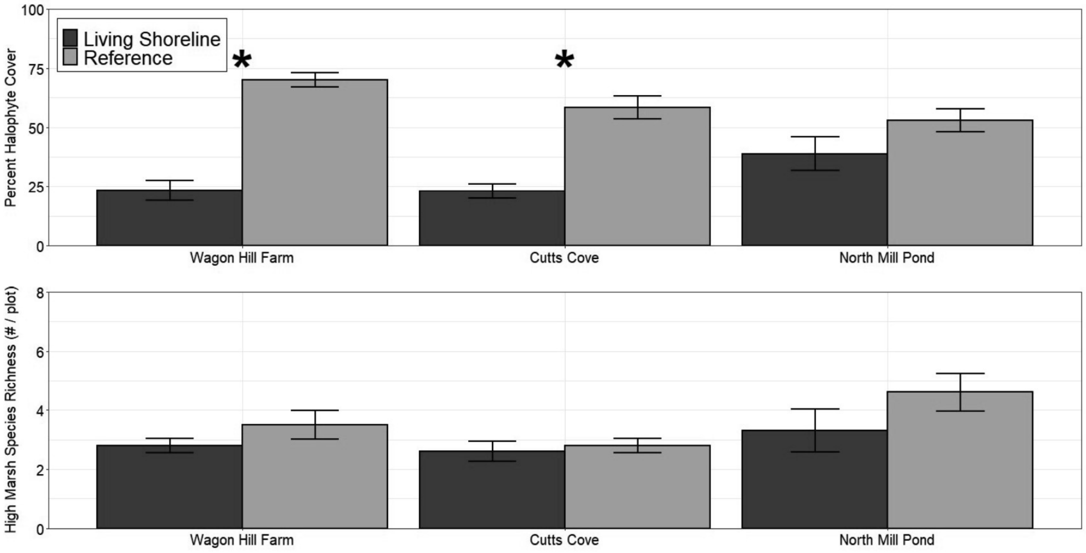

Figure 10: (**Top):** Halophyte cover aggregated across low and high marsh zones in 2020. Significant differences between living shorelines and references are denoted as * and were observed at Wagon Hill Farm ($p < 0.001$) and Cutts Cove ($p < 0.001$). **(Bottom):** High marsh species richness (# per plot) in 2020 of living shorelines and references. No significant difference was observed between all three sites.

Pore water chemistry did not receive a substantial RPI metric score except at NMP for both years three and four when pore water was reliably obtained in the low marsh (Appendix I). The redox potential was significantly higher in the low marsh in all three living shorelines when compared to local references (WHF: t $\text{ratio}_{19} = -5.76$, $p < 0.001$; CC: t $\text{ratio}_{19} = -9.59$, $p < 0.001$; NMP: t $\text{ratio}_{19} = -17.88$, $p < 0.001$). The redox potential of the low marsh zones potential followed non-significant, weak negative trends over project age (Low: $F_{1,5} = 1.15$, $R^2 = 0.22$, $p = 0.344$). Soil redox potential remained highly aerobic in the low and high marshes, with typical values greater than +300 mV and +200 mV, respectively, except in the low marsh of NMP in 2019 (-41.7 ± 10.9 mV). Borderline anaerobic conditions at NMP low marsh translated into low sulfide concentrations both years (2019: 0.4 ± 0.2 mM, 2020: 0.4 ± 0.1 mM). The water content of the soil cores within the top 10 cm in the low and high marsh zones across all living shorelines ranged from 10 – 30% (Table 5). The percent organic matter was well below 0.1 % and bulk density ranged from 1.34 – 1.53 g/cm^3 and 1.15 – 1.62 g/cm^3, respectively.

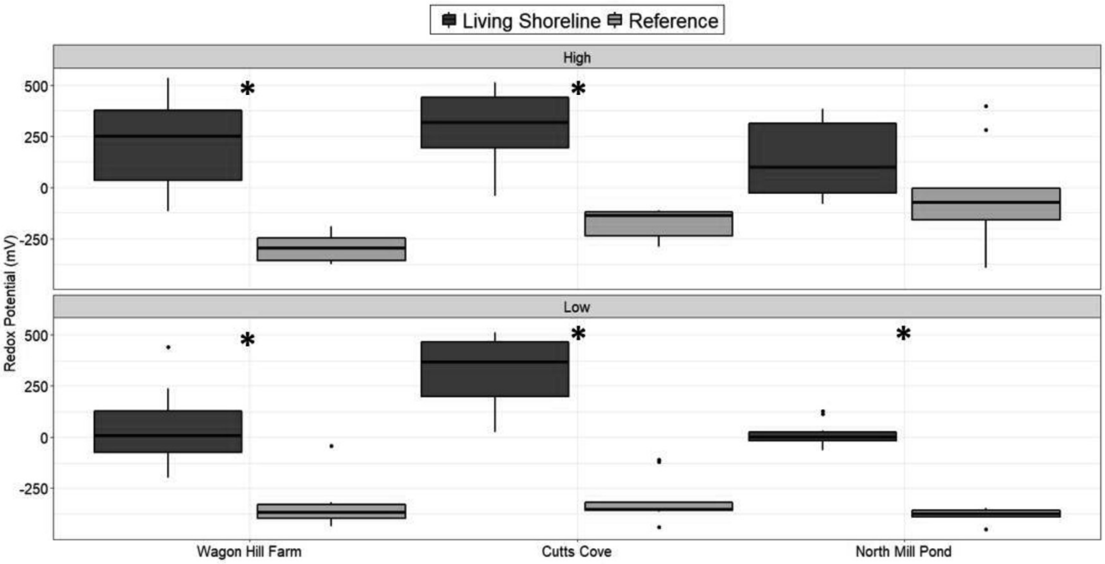

Figure 11: Redox potential (mV) of the living shorelines and reference salt marshes in 2020 divided between low and high marsh zones. The living shorelines had significantly more positive redox potential compared to their respective reference marsh in each zone ($p < 0.05$) except for the high marsh at NMP.

Table 5: Soil metrics of living shoreline projects in low and high marsh in 2020. Soil cores consisted of the top 10 cm of the soil column. Values reported as mean ± standard error.

Site	Marsh Zone	Water Content (%)	Bulk Density (g / cm^3)	Organic Mass (g)	Organic Mass (%)
Wagon Hill Farm	Low	24.0 ± 1.59	1.39 ± 0.07	1.597 ± 0.120	0.062 ± 0.006
	High	30.0 ± 0.75	1.20 ± 0.02	2.417 ± 0.075	0.082 ± 0.002
Cutts Cove	Low	14.1 ± 0.77	1.34 ± 0.04	0.331 ± 0.017	0.011 ± 0.001
	High	10.4 ± 0.37	1.62 ± 0.06	0.376 ± 0.040	0.012 ± 0.001
North Mill Pond	Low	22.2 ± 1.13	1.53 ± 0.02	0.401 ± 0.053	0.013 ± 0.001
	High	28.5 ± 2.38	1.15 ± 0.11	2.501 ± 0.362	0.088 ± 0.013

Fundulus heteroclitus comprised 99.2% of the nekton caught during minnow trapping in 2020. The rest of the nekton were composed of *F. majalis* (0.5%) and *Carcinus maenas* (European green crab, 0.1%). Almost all *F. majalis* were found exclusively at the PCA reference, with one individual captured at WHF living shoreline. The WHF living shoreline had the greatest *F.*

heteroclitus abundance and adult length between all three the living shorelines in the low marsh zone, although there was no significant difference for either metric (Trap Count: $F_{2, 29} = 1.00$, $p = 0.380$; Length: $F_{2, 22} = 3.39$, $p = 0.054$). Trap catch rate of *F. heteroclitus* at each living shoreline was similar or even significantly greater to the respective local reference shoreline (WHF: $\text{ratio}_{19} = -2.31$, $p = 0.036$; CC: t $\text{ratio}_{19} = -0.018$, $p = 0.986$; NMP: t $\text{ratio}_{19} = 0.650$, $p = 0.533$; Figure 11). Additionally, the adult lengths of *F. heteroclitus* were similar between living shorelines and local comparisons (WHF: t $\text{ratio}_{14} = 0.162$, $p = 0.875$; CC: t $\text{ratio}_{11} = 0.650$, $p = 0.533$; NMP: t $\text{ratio}_{16} = 0.665$, $p = 0.517$).

Algae (*Fucus* and *Ascophyllum* sp.) and barnacles (*Semibalanus* sp.) colonized the riprap sills within several growing seasons with noticeable algae cover and barnacle density at CC within two growing seasons (Table 6). Algae and barnacles did not colonize the NMP coir fiber log sills, which had decomposed or been buried by 2019. There was a significant difference of algae cover and barnacle density between CC and WHF after one growing season for each project (Algae: t $\text{ratio}_{19} = -5.73$, $p < 0.001$; Barnacles: t $\text{ratio}_{19} = -3.07$, $p = 0.007$; Figure 12). Non-filter feeder invertebrate density (*i.e.,* gastropod and crab) of riprap sills was significantly correlated with algae cover ($F_{1,39} = 14.71$, $R^2 = 0.28$, $p < 0.001$) but not project age ($F_{1, 3} = 0.21$, $R^2 = 0.10$, $p = 0.689$; Figure 12D).

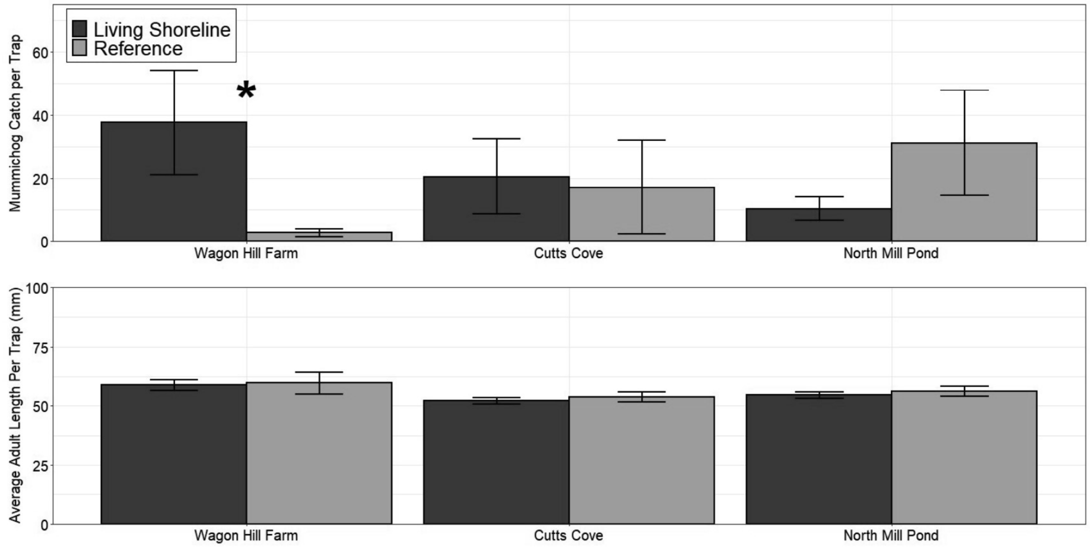

Figure 12: (Top): Mummichog catch per trap of living shorelines and references in the low marsh from minnow trap sampling of 2020. Significant differences of pairwise comparisons are denoted as * and was observed at Wagon Hill Farm ($p < 0.001$). **(Bottom):** Average adult length of the mummichogs from minnow trap sampling of 2020. No significant differences were observed in the pairwise comparisons between living shorelines and references. Error bars are ± standard error.

Table 6: Algae cover and invertebrate density of riprap and coir fiber log sills of living shoreline projects over 2019 and 2020. Values are reported as mean ± standard error.

Site	Sill Type	Year	Algae Cover (%)	Barnacle Density (# / m^2)	Gastropod - Crab Density (# / m^2)
Wagon Hill Farm	Riprap	2019	3.0 ± 1.6	0	0
		2020	8.4 ± 2.5	0.4 ± 0.4	0.2 ± 0.2
Cutts Cove	Riprap	2019	46.5 ± 6.2	71.0 ± 23.0	45.2 ± 8.3
		2020	60.3 ± 5.7	165.0 ± 31.1	16.2 ± 3.7
North Mill Pond	Coir Fiber Log	2019	0	0	0
		2020	0	0	0.4 ± 0.3

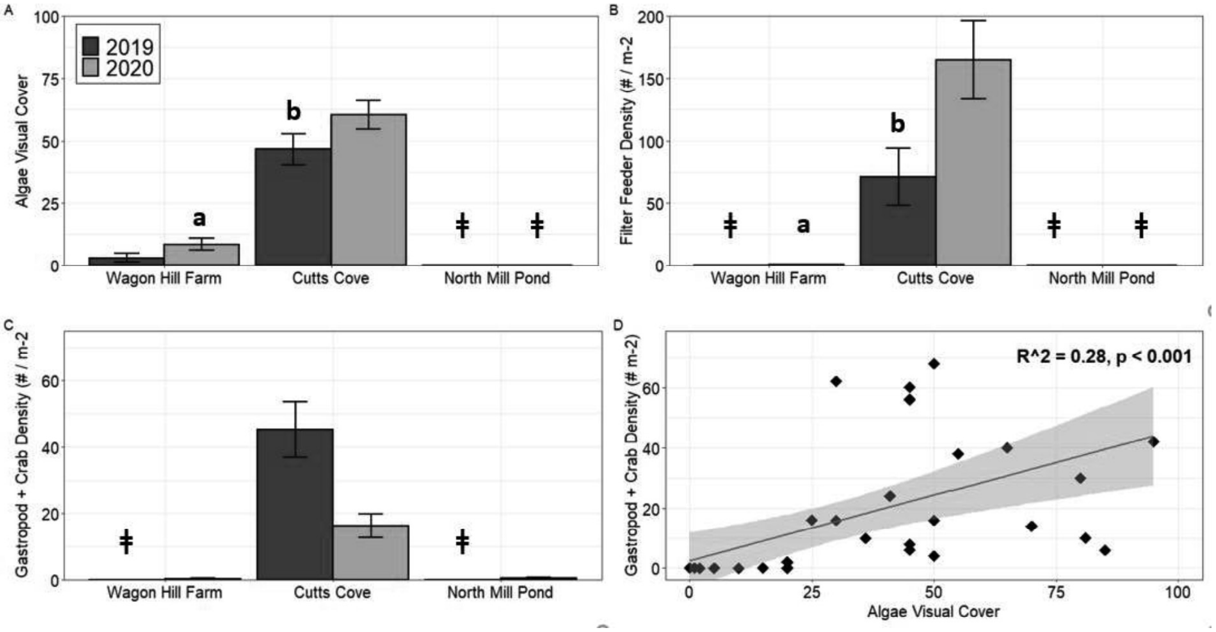

Figure 13: Analysis of algae cover and invertebrate density on living shoreline sills. Wagon Hill Farm and Cutts Cove are riprap sills and North Mill Pond is a decomposed coir fiber sill. **(A)** Algae cover of sills in 2019 and 2020. The algae cover was compared one year after restoration between Wagon Hill Farm (2020) and Cutts Cove (2019). No algae found across all plots is shown as ǂ. **(B)** Filter feeder density (# m^{-2}) of sills in 2019 and 2020. The filter feeder density (*i.e,* barnacles) was compared one year after restoration between Wagon Hill Farm (2020) and Cutts Cove (2019). **(C)** The combined density of gastropods and crabs (# m^{-2}) on the sills in 2019 and 2020. (**D)** Linear regression of combined gastropod and crab density (# m^{-2}) and algae cover. Error bars and grey interval is ± standard error.

The living shorelines had an average decomposition rate (k) of -0.0077 ± 0.0012 day^{-1}, with the CC living shoreline rate almost double the other projects (Table 7). The *Spartina* litter decomposed at a slower rate across the local references (-0.0049 ± 0.0004 day^{-1}) and no action control (-0.0036 day^{-1}) than the living shorelines. The decomposition study was not completed for the entire five months except at CC living shoreline, CC reference, and PCA due to destroyed or missing bags during the study, evidently caused by disturbance from dogs, wildlife, and erosion as the main culprits. An erosion rate was not able to be estimated for the living shorelines (Table 8). Erosion rates estimated for references and no action control shorelines showed high erosion rates except at PCA. The WHF shoreline continues to experience significant erosion rates (>3 cm yr^{-1}) on both the reference and no action control shoreline. A marsh bank erosion (or calving) event was recorded in the NMP reference, where the entire 60 cm (2 ft) erosion pin was found in the bed of a mash creek.

Table 7: Decomposition rate and parameters for the nonlinear fit of percent mass remaining to the exponential decay model. The goodness of fit for each nonlinear regression is given by Root Mean Square of Error (RMSE) and R^2.

Site	Shoreline	Days	K (day^{-1})	p	R^2	RMSE
Wagon Hill Farm	Living Shoreline	100	-0.0064	0.001	0.807	0.063
	Reference	100	-0.0043	0.0106	0.612	0.073
Cutts Cove	Living Shoreline	157	-0.0106	0.0001	0.814	0.129
	Reference	157	-0.0046	0.0001	0.827	0.059
North Mill Pond	Living Shoreline	99	-0.0061	0.0001	0.853	0.093
	Reference	99	-0.0057	0.0001	0.977	0.058
	No action Control	99	-0.0036	0.0305	0.539	0.020
Portsmouth Christian Academy	Reference	157	-0.0052	0.001	0.765	0.071

Table 8: Erosion rates (mm yr^{-1}) for sites with available data. For measurement type, erosion pin measured erosion of vertical shorelines and oak stake measured relatively flat shorelines (*i.e.*, mudflats). Erosion is measured as positive while sediment accretion is measured as negative. Values are reported as mean ± standard error.

Site	Treatment	Measurement Type	Erosion Rate (mm yr^{-1})
Wagon Hill Farm	Reference	Erosion Pin	38.8 ± 20.0
	No action Control	Erosion Pin	75.1 ± 34.2
Cutts Cove	Reference	Erosion Pin	13.5 ± 3.2

North Mill Pond	Reference	Erosion Pin	269.4 ± 240.1
	No action Control	Oak Stake	-5.4 ± 2.9
Portsmouth Christian Academy	Reference	Erosion Pin	5.9 ± 2.4

Discussion

Short-Term Recovery of Ecosystem Structure and Functions

The living shorelines recovered quickly to RPI scores greater than 0.50 within four years. Vegetation and nekton metrics were the primary drivers of the RPI scores across most of the sites as biotic metrics recovered rapidly in the short timeframe. The biogeochemical processes, measured by the pore water chemistry metrics, did not provide a substantial RPI contribution except in NMP in year three.

The WHF living shoreline stood out with a RPI score after one growing season post-construction which was comparable to the two older projects. The relatively quick increase in RPI score was a function of both improving conditions at the living shoreline and deteriorating conditions at the eroding control shoreline. Moore et al. 2010 reported similar immediate increases of RPI (> 0.45 within a year of restoration) including Bridge Creek (Barnstable, MA), Sachuest Point (Middletown, RI), and Meadow Pond (Hampton, NH). The WHF living shoreline over time will represent more substantial marsh habitat and resilient shoreline as the no action alternative salt marsh continues to erode. The rapid recovery of the unweighted vegetation metric score to 0.45 – 0.62 could also be partially attributed to the planting method used at the site. The first planting was conducted with laying a “marsh turf” with pre-installed *Spartina* plugs. The turf method possibly reduced erosion of sediment, increased plug stabilization in the soil, and added nutrients to the coarse sandy fill. The turf of *S. patens* grew well in the first year, resulting in densities over 4 plants ft^{-2}. However many of the *S. alterniflora* did not emerge or died after the first year. Over 10,000 plugs were added to the living shoreline, ultimately resulting in densities over 2 plants ft^{-2} in the low marsh.

The short-term restoration dynamic in this study reflected similar recovery timelines of vegetation, nekton, and soil chemistry of previously documented living shorelines and restored marshes: biotic factors recovered relatively quickly whereas biogeochemical processes will require decades (Craft et al. 2003). Even in northern New England, halophyte vegetation, measured in visual cover and species richness, were nearing comparable levels of local references (> 0.65) within four years. In the mid-Atlantic coast, within a few growing seasons vegetation has been shown to reach equivalency in stem density (Bilkovic and Mitchell 2013, Gittman et al. 2016a) and lag in visual cover by only 11 – 23% (Currin et al. 2008). Vegetation of the living shorelines in this study may require 5 – 10 years to fully reach recover (Craft et al. 1999a, Chambers et al. 2021) as halophyte cover lagged on average 24% and 41% after four years to local references for low and high marsh zones, respectively. Species richness and composition did reach similar levels within three years at the NMP living shoreline, a much quicker recovery than observed previously in constructed salt marshes of Great Bay Estuary (Morgan and Short 2002).

Resident salt marsh nekton recovered very quickly, almost reaching comparable levels to local references based on the RPI score. Additionally, the average trap catch rate was greater at the WHF living shoreline than the local reference. Nekton have been observed to respond to the creation of new salt marsh habitat within several months with increased densities and presence of transient species (Currin et al. 2008). Increased rates of edaphic algae graving, as a result from greater light penetration from less dense *Spartina alterniflora* cover, may explain greater nekton catch rates at WHF than CC and NMP (Seliskar et al. 2002). The nekton community may require at least 3 – 5 years to fully recover as the vegetation or habitat structure continues to improve (Gittman et al. 2016a). Despite lower RPI scores when the PCA reference was compared, the creation of salt marsh at CC and NMP is likely a substantial enhancement of nekton preferred habitat over the pre-existing mudflats and armored shorelines (Davis et al. 2006, Balouskus and Targett 2016). Additionally, the increase in shoreline complexity in the form of a riprap sill, in conjunction with salt marsh vegetation, are preferred by certain species like *Menidia menidia* (Atlantic silverside) and may lead to higher species richness than local references over time (Zeug et al. 2007, Balouskus and Targett 2016).

Biogeochemical processes, measured through pore water chemistry, were a minor component of any site's total RPI score, as unweighted scores never exceeded 0.30. The relatively small contribution of pore water to the RPI score was the result of a lack of pore water in the living shorelines as tidal and groundwater easily drained through the coarse sandy fill material. Peat has not developed in any considerable fashion based on high bulk densities and very low organic matter composition of the soil cores compared to values reported for natural fringe marshes (Craft et al. 1999a, Morgan et al. 2009). Aerobic conditions persisted throughout the low and high marsh zones of all the sites, except NMP low marsh after three years. The persistence of quick draining and aerobic soil conditions partially explains the greater decomposition rates at the living shorelines (Mitsch and Gosselink 2015), even on the surface of the marsh. Long-term monitoring of constructed wetlands have shown that soil development require decadal to century timeframes (Craft et al. 1999a, Brown and Norris 2018, Noll et al. 2019). Chambers et al. 2021 documented the nitrogen and carbon pools in the top 20 cm of the soil column of living shorelines were significantly less than natural comparisons after 16 years. Davis et al. 2015 reported bulk density of living shorelines reached statistical equivalency within 20 years while organic matter composition still lagged by 20 - 45%. The high drainage rates in the coarse sandy soils used in living shoreline construction likely the primary contribution to the long recovery timeframes of certain biogeochemical and soil processes unique to coastal wetlands (Davis et al. 2015, Bilkovic et al. 2021).

Short-Term Restoration Trajectory

Asymptotic trajectories were well-fitted to the total RPI score of the local reference comparison and the unweighted vegetation RPI scores of both comparisons. The asymptotic trajectories of the living shorelines reflect an initial rapid recovery phase followed by a slower, more incremental phase of gains (Jones et al. 2018). For example, the total RPI score asymptote may represent a short-term incremental step within a longer, more gradual recovery as the pore water chemistry and wetland soils slowly recover over decades. Long-term monitoring should be continued during this rapid recovery phase (0 – 10 years) to ensure the living shorelines do not enter an alternative state and provide less ecosystem functions than references (Folke et al. 2004, Borja et al. 2010, Moreno-Mateos et al. 2017). Moreno-Mateos et al. 2012 documented persistent below reference conditions over long timeframes of biotic factors and biogeochemical processes of restored wetlands, resembling asymptotic trajectories in this study. Specific instances of logistic, asymptotic trajectories below reference conditions have been reported in above ground vegetation biomass (Morgan and Short 2002) and soil organic matter (Zedler and Callaway 2000) of constructed salt marshes. Even if the living shorelines never meet reference conditions, the RPI demonstrates that the restoration activities have created substantial value in salt marsh habitat compared to no-action scenarios.

The asymptotic trajectories in this study should be viewed with caution as the timeframe and number of sites are low. No significant or strong linear trajectories were found in the study, except for high marsh species richness ($R^2 > 0.50$, $p > 0.05$). Craft et al. 2003 argued earlier studies of west coast salt marshes (see Simenstad and Thom 1996, Zedler and Callaway 1999) might have been unable to calculate restoration trajectories from monitoring only of projects aged 7 – 12 years. The short timeframe of this study's chronosequence (0 – 4 years) prevented restoration trajectories to be developed for slow recovery metrics of vegetation and biogeochemical processes. The combination of few sites and high natural variation between the sites also contributed to the lack of development of concrete restoration trajectories of living shorelines in the Great Bay Estuary as found in other systems (Simenstad and Thom 1996, Morgan et al. 2009, Brudvig et al. 2017). For example, the WHF living shoreline recovered at a greater rate than the other two sites even when all were compared to PCA. Despite pairwise RPI score comparisons not differing greatly between the two reference comparisons, the path of the restoration trajectories did (linear vs. logistic). The choice of the reference salt marsh for restoration goals can influence both restoration trajectory calculations and expectations of future restoration efforts (White and Walker 1997, Morgan and Short 2002). Additionally, the restoration trajectory of certain factors may not be dependent on time but rather on the development of other biotic and abiotic factors such as the development of the invertebrate community on the riprap sills (Raposa et al. 2018, Lilian et al. 2021).

Effect of Sill Construction on Habitat Heterogeneity and Recovery Rates

The data demonstrates that riprap sills of the living shorelines supported greater abundance of macroalgae and invertebrates compared to coir fiber sills. The algae and invertebrates were able to colonize the three-dimensional hard surface and increase the habitat heterogeneity of the projects. Both riprap sills, despite significant differences, experienced increase in algae cover and invertebrate densities compared to the bare sand of the now decomposed coir fiber logs of NMP. The invertebrate community of the riprap sills was dominated by *Semibalanus* barnacles, measured by abundance of individuals, with a small composition of *Carcinus maenas* and the gastropods *Littorina littorea* and *Littorea saxatilis*. Bilkovic and Mitchell 2013 observed similar trends of the invertebrate community with dominance by epibenthic filter feeders and minor presence of gastropods. It remains to be seen if epibenthic species other than barnacles, like *Mytilus edulis* (Blue mussel), will colonize the sills.

This is the first study to document the colonization of the intertidal macroalgae on the riprap sill. O'Connor et al. 2011 did document slow recovery (> 7 years) of intertidal macroalgae within the lower edge of *Spartina alterniflora* behind the riprap sill. Experimental removal of the intertidal algae from the lower vegetated edge however did not impact the density and canopy height of *S. alterniflora*. The addition of macroalgae on the riprap surface may further enhance the high nekton species richness and species-specific preferences observed in other living shoreline projects (Balouskus and Targett 2016), although the complete recovery of the nekton community may not be realized until 3 – 5 years (Gittman et al. 2016a). Natural analog comparisons to oyster reef restorations have shown the increase in habitat heterogeneity provided refuge for prey from predators (Grabowski 2004), preferential niche space for certain species (Zeug et al. 2007), and more diverse and abundant nekton communities than unpaired natural marsh shorelines (Stunz et al. 2010). Observational data have suggested that riprap sills will provide similar refugia and habitat characteristics of oyster restoration and breakwater projects with complex, three-dimensional surfaces and crevices (Hardaway et al. 2007).

Defining Success with the RPI

The difficulty of determining reliable restoration trajectories highlighted the strength of the RPI as an in-depth evaluation tool of short-term recovery of living shorelines. The RPI incorporated 13 individual metrics into a single, comprehensive score of the progress toward the restoration goals. Although not directly evaluated, the utility of the RPI as a method to quantify restoration trajectories of systems as a whole was apparent. Restoration trajectories were unable to be quantified (linear or logistic) from too few data points (mummichog abundance) or weak and insignificant recoveries (e.g., redox potential of pore water chemistry, halophyte visual cover, and halophyte richness). Although touted theoretically, the efficacy of predictable and reliable restoration trajectories in wetland restoration assessment has been criticized based on empirical evidence (Zedler and Callaway 1999, 2000). However, the criticism of trajectories were based on high variable univariate metrics (e.g., vegetation) or those requiring long timeframes to recover (e.g., wetland soil properties) (Zedler and Callaway 1999). Multivariate or comprehensive metrics such as RPI scores which represent the system as a whole should be more useful for creating reliable restoration trajectories. For example, the floristic quality assessment is a comprehensive metric that measures fidelity of the vegetation to disturbance, which innately accounts for habitat stability, soils, and hydrology (Spyreas 2019). Matthews et al. 2009 quantified strong, significant logistic trajectories of the floristic quality assessment values of vegetation in created freshwater wetlands.

As multiple restoration goals are recommended for projects (Neckles et al. 2002), a tool like the RPI will become essential to communicating progress of a restoration project to stakeholders, researchers, and the general public. The prevalent form of measuring restoration success is paired statistical comparisons where success takes a yes/no dichotomy (Kentula 2000). The RPI's measurement of "what has been gained" compared to "what remains to recover" in conventional thinking shifts the general outlook on restoration to a more positive discussion (Moore et al. 2010, Lilian et al. 2021). Measuring progress against a no action control, though still not widely used in restoration projects (Wortley et al. 2013), can lead to a better understanding of recovery of ecosystem functions and services, especially since abiotic and biotic factors recover at different timeframes. Long-term annual monitoring and evaluations with the RPI will provide more context to understand the speed of which vegetation, nekton, and biogeochemical processes recover in northern New England salt marsh restorations. The flexibility of the tool to include any restoration goals or different types of sampling methods allows it to be used across systems and restoration methods (Raposa et al. 2018). For example, the RPI could be used to evaluate the impact of sill type (riprap vs. coir fiber) on the nekton community recovery with comprehensive sampling efforts over the next few growing season.

Adaptive Management Needs of Living Shorelines in New Hampshire

The living shoreline restoration trajectories experienced multiple stressors that influences RPI scores, yet these stressors were not directly studied and thus quantitatively accounted for, including herbivory, erosion behind the riprap wall, and natural wrack deposition (Figure 13). First, the CC living shoreline experienced heavy herbivory by *Branta canadensis* (Canada goose) in November – December of 2019 and 2020 (Burdick et al. 2021). Geese were able to take advantage of the coarse sandy fill and rip up *Spartina alterniflora* rhizomes for consumption, completely denuding large sections of the low marsh. The herbivory event left large holes scattered throughout marsh and probably erased vegetation gains over the past two years.

Vegetation establishment was difficult at certain locations within salt marsh surface of the living shorelines. *S. alterniflora* plugs experienced higher mortality rates close to the seaward edge. The plugs at the seaward edge experience increased submerged time, tidal wave action, and erosion of fine sediments from the soil surface compared to more landward plants. The loss of *S. alterniflora* was especially noticeable at the NMP site, without a riprap sill, where all planted *S. alterniflora* were dead within 3 – 5 m of the seaward edge. Across all of the living shoreline sites in 2020, halophyte cover was significantly less (~25%) within the first few meters of the low marsh. The lack of dense halophyte cover may lead to increased erosion of sediment at the seaward edge without dense rhizomes and roots for soil stabilization as well as prevent colonization of *Geukensia demissa* (Ribbed mussel) (Bilkovic et al. 2021). In the high marsh, high mortality rate was noticed in WHF and CC living shorelines as well as incomplete recovery at NMP. Drought conditions and plantings at elevations above spring tidal flooding possibly led to moisture stress in newly planted halophytes at WHF and CC especially over the winter months (Burdick et al. 2021). The NMP living shoreline experienced high wrack deposition which smothered and killed high marsh vegetation. The location of wrack deposition was variable each year, preventing the full development of a high marsh vegetation community.

Isolated erosion was observed and addressed at the WHF and CC living shorelines within several growing seasons post-construction. At the WHF site, sandy fill material immediately behind the riprap sill began eroding despite careful anchoring of jute mat, subsequently requiring considerable effort to repair to ensure integrity of the sill edge. At the CC site, runoff from the upland created small gullies through the living shoreline, demonstrating that small even very small catchments can erode and degrade new restoration projects (Burdick et al. 2020b). The observed erosion was not widespread and relatively minor before it was addressed at each site, so it was not believed to have affected the RPI score or the overall recovery of the living shorelines. If unchecked, however, erosion behind the sills or creation of gullies from stormwater runoff may wash out large amounts of sediment and lead to vegetation loss as observed at NMP.

Based on monitoring efforts over the past several growing seasons, the following monitoring and adaptive management efforts should be considered in the designs and budgets of future projects to improve restoration outcomes:

- Monitoring should continue for at least 5- 10 years to encapsulate the full of the vegetation community after the rapid recovery phase of the first 5 years;

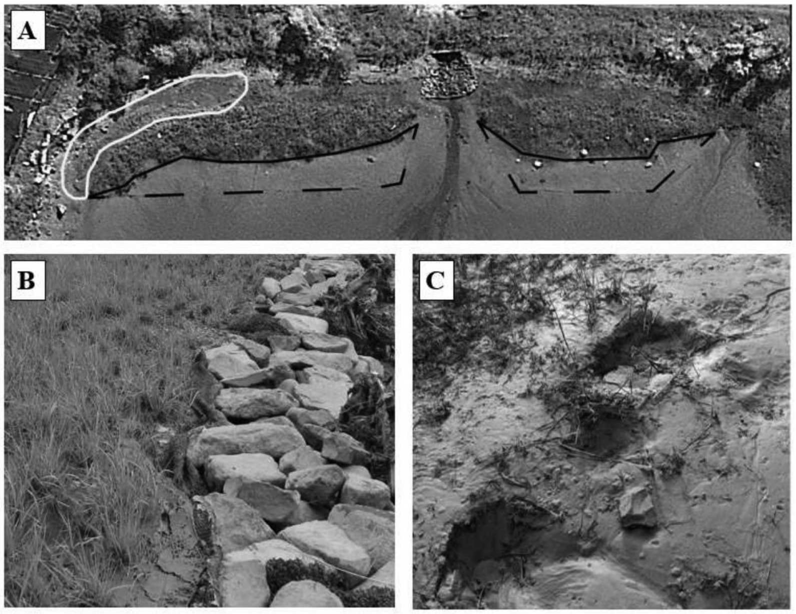

Figure 14: Examples of the three adaptive management considerations documented at each living shoreline project in the study. (A) *Spartina alterniflora* mortality at the seaward edge of North Mill Pond. Dashed lines are the original planted vegetation extent, solid lines are the furthest vegetation in 2020, and yellow lines is extent of wrack deposition of 2020. (B) Erosion of sediment behind the riprap sill of Wagon Hill Farm shown by jute falling into riprap sill. (C) Evidence of geese herbivory of *S. alterniflora* rhizomes at Cutts Cove.

- Monitoring of nekton should include multiple sampling events throughout each growing season and use of different fishing gear (*i.e.,* beach seine) to better understand habitat preferences and salt marsh surface use of transient fish and nekton;
- Monitoring should include inspection for impacts of herbivory from snails (*Littorina littorea*) and Canada geese. Geese prevention measures should be planned every October within the first 5 - 10 years of projects to protect restoration projects from geese herbivory in November – December. If snail impacts are documented, fencing or galvanized mesh may need to be erected around the low marsh edge during the growing season (April – October);

- Multiple plantings should be budgeted and planned within the first five years to fully establish vegetation at the seaward edge and make up losses from other factors negatively affecting planting success such as herbivory, drought, and storm surge;
- Other high marsh and upland border species (*i.e., Solidago sempervirens*, *Plantago maritima*, *Triglochin maritima*, and *Puccinellia* sp.) should not be expected to colonize rapidly as introduced soils lack a native seed bank. Thus, diverse plantings may need to be obtained by vendors or sustainably transplanted from local donor sites to expedite play community development. High marsh species may require irrigation within the first few years post-construction. Sandy soils used as fill material may lack sufficient silt and clay size fractions (*i.e.,* organic matter) to retain tidal water and precipitation. Plantings at elevations above normal tidal flooding may need irrigation since occasional salt water flooding may salinize soils and simulate artificial drought conditions for vegetation as well;.
- Management of excessive wrack deposition should be included in spring maintenance plans. If restoration projects experience high winter wrack deposition, efforts should be made to prevent deposition (*i.e.*, strategically placed fencing) or removal of wrack after major storms and the winter season;
- Erosion control measures should be addressed if issues are noticed during monitoring. Extra or leftover fill material for the living shoreline projects may be stored near the site, if possible, to address erosion problems;
- Collection and placement of rocks or oyster shells with intertidal algae (e.g. *Fucus, Ascophyllum*) from nearby mudflats into the riprap sill may facilitate the colonization by

native and beneficial macroalgae and establishment provision of important habitat for salt marsh invertebrates; and,

- Methods to establish *Modiolus* sp. (Mussels) on the riprap sill should be further investigated to expedite recruitment on the riprap sills and enhance water filtration services of living shoreline projects.

Proposed Monitoring Plan for Living Shorelines

Salt marsh restoration projects typically have similar overarching goals of enhancement of salt marsh habitat, support of local fisheries, development of wetland soils, and carbon storage. The restoration goals of living shoreline projects differentiate from prior salt marsh restoration projects such as *Phragmites australis* removal (invasive species removal), tidal culvert replacement (tidal hydrology connection), and excavation of fill material (tidal elevation correction). In addition to ecological purposes, living shorelines are constructed with the intent purpose of abating shoreline erosion and protecting upland infrastructure and ecosystems. Monitoring plans for living shorelines should then reflect the difference in construction and restoration goals compared to prior salt marsh restoration projects. Ideal five-year monitoring plans of future living shorelines should focus on (1) Shoreline integrity, (2) Habitat structure, (3) Trophic structure, and (4) Wetland soil development. Monitoring plans for living shorelines should include reference salt marshes and, if available, no-action controls to accurately gauge the progress of recovery.

Living shorelines are expected to increase the shoreline integrity by decreasing lateral erosion and protecting upland infrastructure and property. The rate of lateral and surface erosion of the living shoreline can serve as a proper proxy for shoreline integrity. Erosion pins (2 ft rebar rods) should be hammered flush into the vertical face of the sediment fill in gaps of the riprap sill. For surface erosion, erosion pins hammered flush into the salt marsh surface vertically 1 – 2 m behind the riprap sill. Erosion pins should be installed every 10 – 15 m along the shoreline for precise measurements of erosion. To calculate the rate of erosion, the length of the exposed pins should be inspected twice a year in the early spring (after potential ice rafting events in New England) and late fall (after the growing season).

The combination of both ecological restoration and grey infrastructure in living shorelines creates a heterogeneous habitat structure which includes both the halophyte vegetation on the salt marsh surface and the intertidal algae on the riprap sill. Monitoring efforts should attempt to capture the condition of both areas in order to better understand primary production, available niches for fauna, and overall habitat structure of the project. Habitat structure should be monitored annually near the end of the growing season such as August and September in New England. Halophyte vegetation of the salt marsh surface should be measured for abundance (*i.e.*, visual cover, shoot density), species richness, and canopy height. In New England, monitoring of halophyte vegetation should be divided between low marsh, high marsh, and, if possible, marsh-upland transition border due to inherent differences in species identity and canopy height. Intertidal algae colonization of the riprap sills should be monitored for abundance and species richness. The abundance of intertidal algae can be quantified from either a plan or side view based on the slope and height of the riprap sill.

The tropic structure component of the monitoring plan is a multi-tiered focus on how the restored salt marsh habitat supports local fauna and the flow of energy and carbon into the larger, surrounding system. The monitoring plan for trophic structure can be broken down into three components: nekton community, macroinvertebrate community, and decomposition of primary production. The nekton community should be monitored annually for abundance, species richness, and size distribution. Depending on restoration goals, the nekton community can be monitored with minnow traps to sample nekton use on the salt marsh surface or with seines below the riprap sill in the mudflat. The macroinvertebrate community (i.e., snails, crabs, mussels, and barnacles) can be monitored annually on both the salt marsh surface and the riprap sill for abundance and species richness. The invertebrate community can be quantified visually by identifying and counting all individuals found within a given area on the sediment surface and vegetation in the salt marsh surface or within crevices and on intertidal algae in the riprap sill. The decomposition of primary production on the salt marsh surface incorporates the development of wetland soils, invertebrate community, and the nekton community. In the four year of monitoring, senesced *Spartina alterniflora* leaves should be collected, dried, and stored in a cold, dry location. Decomposition bags with senesced *Spartina alterniflora* leaves should be deployed roughly 1 – 2 m behind the riprap sill and collected at regular intervals throughout the growing season. The rate of decomposition can be calculated with an exponential decay and compared to the reference salt marsh.

The development of wetland soil characteristics such as organic matter, bulk density, and availability of porewater is expected to require longer than typical five-year monitoring plans. Living shorelines are constructed with sterile coarse sandy soils, which aerate and drain rapidly. Porewater was not acquired over the two years of sampling except for several plots in the low marsh of NMP four years post-construction. Wetland soil characteristics should be monitored in the fifth and final year of monitoring. Porewater should be collected, if present in the soil column, from the low and high marsh within the top 5 – 15 cm and measured for salinity, redox potential, and pH. Soil cores should be collected of the top 5 cm of the soil column and measured for water content and bulk density. If possible, the build up of organic matter, a proxy for the rate of carbon storage, could be measured by conducting loss on ignition with a subsample of each soil core.

Conclusion

Through the lens of the RPI, restoration activities created ~50% of the ecosystem functions and services of local reference salt marshes within several growing seasons. The substantial enhancement of salt marsh habitat is more encouraging when compared to no-action controls, where salt marshes did not exist or had recently been lost. Vegetation and resident nekton use rapidly recovered within the first several growing seasons. The vegetation community may require 5 – 10 years post-construction to achieve near parity with local references, while the pore water chemistry and soil characteristics may require decades to recover. The recovery of pore water chemistry and wetland soils may depend on other ecosystem components rather than time (*i.e.,* hydrology, invertebrate community, and vegetation development). It is expected the living shorelines will recovery slowly over time after the observed initial rapid recovery phrase. Monitoring efforts should continue after common five-year timeframe to document the full recovery of vegetation and biogeochemical processes. Although the riprap sills were not included in the RPI, they did enhance the habitat heterogeneity of the living shorelines and may enhance nekton and invertebrate communities. Impediments to restoration progress such as geese herbivory, erosion, and wrack deposition, are more easily mitigated if included in the adaptive management budgetary planning at the project scope phase. Living shorelines are a much-needed tool for coastal communities to stabilize shorelines, restore salt marsh habitat, and enhance valuable ecosystem services.

CHAPTER III

SUCCESSIONAL DYNAMICS OF A 35-YEAR OLD FRESHWATER MITIGATION WETLAND IN SOUTHEASTERN NEW HAMPSIRE

Introduction

The goal of wetland mitigation in the United States under the Section 404b program of the Clean Water Act (1977), the federal government's "no net loss" policy, and New Hampshire's Fill and Dredge in Wetlands Act (1969) is the creation of a self-sustaining wetland ecosystem equal in size, structure, and function to the one which was lost (Zedler 1996). Ecologists and regulators have sought to improve the likelihood of ecological success of mitigation by identifying potential causes of failure such as lack of government accountability (Erwin 1991, Brown and Veneman 2001) and improper establishment of hydrology (Matthews and Endress 2008) as well as improvements to restoration strategies such as seeding (Reinartz and Warne 1993, DeBerry and Perry 2012) and incorporation of microtopography (Moser et al. 2007, Rossell et al. 2009). The Section 404b program innately assumes, through monitoring periods of less than five years and lack of universally required management strategies, that mitigation projects will remain ecologically successful long after meeting regulatory success criteria. However, recent studies have revealed declining ecological success of the freshwater vegetation communities 6 – 10 years after wetland construction (Matthews and Endress 2008, Bosch and Matthews 2017), questioning the sustainability of compensatory mitigation as an effective tool compared to avoidance and minimization of wetland impacts.

One of the challenges to restoring ecologically equivalent freshwater mitigation wetlands is the inability to accurately predict the trajectory of the vegetation community. Matthews and

Spyreas 2010 proposed a framework to interpret how a created wetland may converge to or diverge from restoration goals. Convergence results in species composition and relative abundances eventually resembling reference conditions, though the path may be linear or non-linear. Divergence is the process of reaching an alternative stable state, and may occur initially or after a considerable amount of time following restoration efforts. A common documented trajectory is an initial convergence, as early colonizers and annuals are outcompeted, and then a divergence due to the lack of uncommon perennials or formation of alternative wetland communities (Matthews and Spyreas 2010). For example, Aronson and Galatowitsch 2008, through repeated surveys over twenty years, described prairie pothole systems reaching an alternative stable state to references after initially converging with the accumulation of common emergent and floating aquatics. Within 12 years the wetlands had generally stabilized into alternative stable states with lower species richness, lack of representative wet prairie and woody species, and invasion of aggressive exotics.

It has been proposed that the vegetation community of wetlands mature within 15 – 20 years (Mitsch and Wilson 1996) based on stabilization of species composition and richness and declines in the rate of species turnover. Proposed by Noon 1996 first and amended by follow-up studies (Atkinson et al. 2005, Matthews and Endress 2010, Matthews and Spyreas 2010), species composition of the vegetation community is immediately dominated by annuals and shifts over time to perennials and dominant graminoids like *Typha* and *Sparganium*. For example, DeBerry and Perry 2004 observed annuals comprised 60% of the species of 2-year-old created wetlands compared to 4% of adjacent reference wetlands. Second, species richness reaches a maximum between 5 – 15 years and subsequently plateaus or declines (Reinartz and Warne 1993, Stefanik and Mitsch 2012). A decline in species richness has been explained by (1) a lack of equal recruitment of perennial species to annual species loss (Campbell et al. 2002, Matthews and

Endress 2010), (2) hydrophytes outcompeting upland species after proper hydrology establishment (Spieles et al. 2006), and (3) invasive species outcompeting natives and forming monocultures (Aronson and Galatowitsch 2008, Gutrich et al. 2009). Third, the rate of species gain, loss, and turnover, measured as the rate species are lost and replaced over time, is initially high after wetland creation and then declines over time (Anderson 2007, Aronson and Galatowitsch 2008, Matthews and Endress 2010). Although rarely quantified, the rate of species gain, loss, and turnover could describe if a wetland reaches an equilibrium or remains dynamic over time.

Comprehensive, long-term floristic reviews of a site can provide needed context to understand wetland successional dynamics and vegetation restoration trajectories. A common experimental design involves surveying a chronosequence of created wetlands over one or two growing seasons to draw conclusions (see Ballantine and Schneider 2009). The experimental design is cost-effective and has allowed researchers to understand how wetlands recover with a short turnaround. However, large variability between sites has been documented in multiple studies, where site context (inherent factors of landscape management history, landscape location, climate, etc.) has proven to be just as a significant factor in vegetation dynamics compared to landscape, hydrology, or wetland size parameters (Matthews and Endress 2010, Stefanik and Mitsch 2012). Specific site context could provide more details pertinent to understanding the development of unique wetland communities. Consistent, long-term floristic reviews of individual sites could pinpoint mechanisms that explain shifts in restoration trajectories.

In addition to proper habitat structure, a common goal for wetland creation is the support and enhancement of avian communities for conservation or recreation (Anderson et al. 2018). Species composition and distribution of the vegetation community are likely important factors in predicting a wetland's avian community. The diversity and abundance of the avian community can be sensitive to certain vegetation and landscape metrics including wetland area size (Elliott et al. 2019), forested landscape cover (Alsfeld et al. 2010), structural complexity and heterogeneity within the wetland (Muir Hotaling et al. 2002), and open water cover (Hapner et al. 2011). Glisson et al. 2015 found through intensive vegetation sampling efforts that secretive marsh birds were highly sensitive to vegetation metrics. For example, *Porzana carolina* (Sora) preferred greater *Typha* cover while the invasion of *Phalaris arundinacea* deterred use by *Botaurus lentiginosus* (American bittern) and *Rallus limicola* (Virginia rail). Short and long-term shifts in avian use have been attributed to changes in a wetland's vegetation community (VanRees-Siewert and Dinsmore 1996, Brown and Smith 1998). The avian community then is a function of the successional dynamics of the vegetion community and may serve as metrics of habitat quality and mitigation success.

With the federal government's Section 404b program and New Hampshire's Fill and Dredge in Wetlands Act approaching 50 years of age, researchers now have the ability to assess long-term successional dynamics, mechanisms of succession, and the ecological success of created wetlands. Evaluation of long-term restoration trajectories (e.g., $\geq$ 30 years) could determine whether vegetation communities eventually stabilize and converge to desired conditions or diverge to less desired, alternative stable states (Zedler and Callaway 1999, Matthews and Spyreas 2010). Towards this goal, our study analyzed how the vegetation community of a 35-year-old created wetland in southern New Hampshire shifted over time based on three floristic surveys over the last 28 years. The goals of the study were to (1) compare long-term successional trends of the wetland to documented patterns in the literature, (2) determine if the vegetation community converged towards a scrub-shrub swamp or diverged to an alternative state, and (3) evaluate the quality of habitat for wetland birds. We conducted a floristic survey in 2020 of the vegetation community and incorporated species composition and habitat delineation data from 1992 and 2002 to analyze successional patterns. Additionally, we monitored the avian community over one season as an assessment of habitat quality.

Methods

Site description

A 4 ha freshwater wetland was created in an abandoned gravel pit mine, Quarry Pond (43.0234, -70.8004), in the winter of 1985 – 86 in Portsmouth, New Hampshire as off-site compensatory wetland mitigation after the destruction of a similar sized scrub-shrub swamp resulting from site infrastructure expansion (Garlo 1992). Seven pools were excavated to the groundwater table. Peat from the original wetland was excavated, containing an intact seed bank, and deposited at the restoration site. The peat was spread at a thickness of 15 – 30 cm across the restoration area. Surface water runoff, direct precipitation, and groundwater are the primary water inputs into the topographically restricted restored wetland. Several output channels for seasonal flooding connects downgradient to Packer Bog, a 121 ha unfragmented forested swamp northwest of the site (Snyder 2006). The dominant plant community of Packer Bog is a regionally rare Atlantic white cedar – yellow birch – pepperbush swamp. The revegetation plans relied solely on natural colonization and the seed bank from the excavated peat. Descriptions of the vegetation community from the original wetland and first seven years of the Quarry Pond project are unknown (Garlo 1992, Padgett and Crow 1994). The species composition was documented in 1992 and 2002 (Padgett and Crow 1994, Jahr and Crow 2005) and wetland communities delineated in 1992 (Figure 14). Beaver activity was noted for raising the water levels by 1 m after blocking outlet channels (Garlo 1992, Padgett and Crow 1994). Beaver activity was presumed to have been consistent over time and was also observed in 2020, represented by the presence of three beaver lodges, evidence of foraging on *Alnus* shrubs, and creation of new channels.

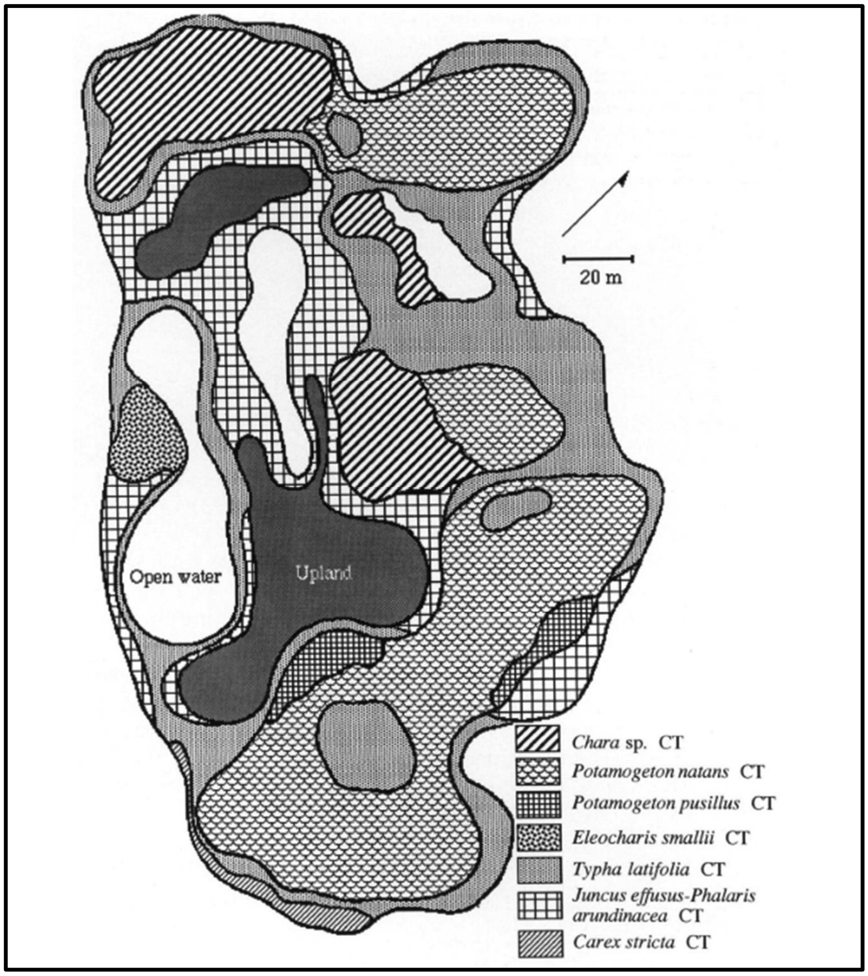

Figure 15: The dominant vegetation map of Quarry Pond in 1992 created by Padgett and Crow 1994.

Vegetation sampling and analysis

Vegetation was surveyed for species composition and distribution of distinct wetland plant communities at the site in two ways. To complete a full inventory of species present, we conducted meander surveys throughout the wetland up to 3 hours biweekly from late May to late October. Meander survey efforts were conducted at roughly the same duration of season and frequency as in 1992 and 2002 surverys. Voucher specimens for each species were collected and accessioned in the New Hampshire Archive at the Albion H. Hodgdon Herbarium at the University of New Hampshire. Vegetation nomenclature is based on Haines et al. 2011. In addition to meander surveys, we completed fixed plot linear transect surveys in June. Vegetation was surveyed every 10 m along four linear transects positioned perpendicular to the southeastern boundary. Each transect was 240 - 290 m in length resulting in a total of 103 plots. Visual cover was estimated to the nearest 1% (maximum of 100%) at three canopy layers: understory (0.5 m^2 square quadrat), shrubs of 2 – 5 m (3 m radius), and trees greater than 5 m tall (5 m radius) (modified from Spencer et al. 2001). The wetland community type was classified for each plot according to Sperduto and Nichols 2011 based on ground, shrub, and tree species composition and cover. Vegetation plot sampling efforts were based on the methods of the initial 1992 survey to accurately compare the distribution of vegetation communities, however the exact locations of the transects and plots from 1992 are unknown. The visual cover and distribution of vegetation communities of 2020 were only compared to 1992, since the 2002 survey methodology did now allow for a direct comparison.

Approximately 120 high resolution images were captured on June 1, 2020 (Figure 15) utilizing a DJI (Los Angeles, CA) Phantom 4 Pro Unmanned Aerial Vehicle (UAV) at an altitude of 200ft and equipped with a DJI 20MP true color (R, G, B) camera. UAV imagery had approximately 80% overlap and at an effective 1.67cm ground resolution. Imagery capture was conducted within two hours of solar noon to maintain consistency of environmental conditions. Resulting images were mosaicked together using Agisoft photogrammetry software and then rectified to ground coordinates. The R, G, B mosaic bands were then stacked and clipped to the bounds of the study area utilizing ArcGIS Pro 2.5 software (ESRI, Redlands, CA). UAV flight planning was completed using DJI Flight Planner software. Flight control was completed with DJI Ultimate Flight v3 and Drone Deploy software (San Francisco, CA).

Using the resulting mosaics and plot sampling data, the areal extents of each wetland community were manually digitized using ArcGIS Pro. The community boundary delineations were groundtruthed by field verification. The dominant vegetation delineations of 1992 were reclassified according to Sperduto and Nichols 2011 georeferenced based on permanent locations of center of pools and wetland perimeter, and manually digitized. The areas of the wetland community types in 1992 and 2020 were calculated based on digital delineations in ArcGIS Pro. The habitat delineation maps of 1992 and 2020 were compared to assess wider community shifts over the past 28 years. Plot-level understory species richness was compared between different wetland communities by aggregating the eight communities based on similar hydrologic conditions and dominant vegetation: aquatic bed, cattail marsh (cattail, cattail – scrub-shrub marsh), graminoid meadow marsh (tall graminoid, tall graminoid – scrub-shrub, and sedge meadow marsh), and woody swamps (scrub-shrub, red maple swamp). Species richness data were power transformed to meet the assumption of normality per the Shapiro-Wilk Test. One-way ANOVA and follow-up Tukey-Kramer tests were used to compare plot-level species richness between the aggregated groups in JMP 15 software (SAS Institute, Inc., Cary, NC).

To determine how plant communities have changed over time, the species composition surveys of 2020 were compared to 1992 and 2002 (Padgett and Crow 1994, Jahr and Crow 2005). Species were assigned wetland indicator designations, native status, growth habits, life history, and coefficients of conservatism. The United States Department of Agriculture (USDA) PLANTS database (National Plant Data Team 2020) was used to classify native status, growth habit (forb/herb, graminoid, and woody), and life history (annual and perennial) for all species. The wetland indicator score (WIS), the probability of the species occurring in a wetland environment, was assigned according to the Northeast Region of the National Wetland Plant List of 2016 (Lichvar et al. 2016). Each WIS is assigned a rank value: obligate (OBL) = 1, facultative wet (FACW) = 2, facultative (FAC) = 3, facultative upland (FACU) = 4, and upland (UPL) = 5. A non-weighted Prevelance Index (PI) was calculated based on WISs to quantitatively assess shifts in hydrophyte and upland species. The lower index values correspond to a presence of more hydrophytic vegetation, with values 1 – 2 representing a dominance of OBL vegetation. The PI was calculated as:

$$PI = \frac{\sum WIS}{S}$$

where S is the number of species in each sampling year.

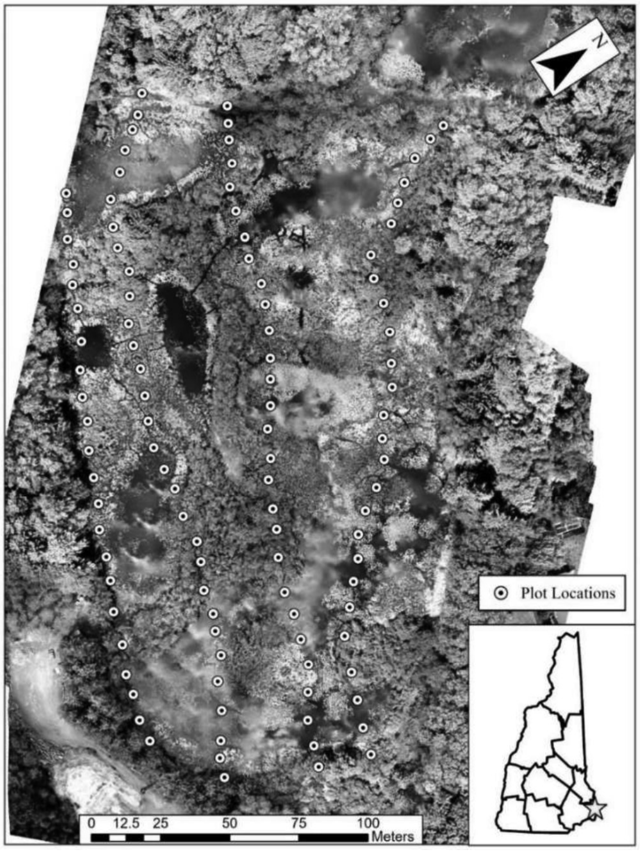

Figure 16: UAV imagery and vegetation sampling plot locations at Quarry Pond in Portsmouth, New Hampshire.

The coefficient of conservatism (CoC), a measure of a species's tolerance to anthropogenic disturbance, was based on the New Hampshire assessment by Bried et al. 2012. CoCs are assigned collaboratively by by regional expert botanists and values range from zero to ten, where zero is highly tolerant to disturbance or an exotic and ten is a species intolerant of distrubance (Spyreas 2019). A floristic quality assessment (FQA) was conducted for each survey to assess the change in conservation value over time. The FQA is a quantitative masurement of a site's relative lack of human disturbance (Bell et al. 2017) and has often been used as a factor in success criteria for mitigation and habitat assessments (Lopez and Fennessy 2002, Deberry and Perry 2015). The FQA is calculated as:

$$FQA = C_{site} * \sqrt{S}$$

where C_{site} is the mean CoC of an entire site and S is the species richness of the site.

Sorensen's Index of Similarity (Sorensen Index) assessed the difference in communitiy composition between the floristic surveys to understand dynamic changes in the vegetation community. The Sorensen Index compares the presence/absence of species between two samples (Sorenson 1948). Sorensen's Index is calculated as:

$$Sorensen's\ Index = \left(\frac{2c}{a+b}\right) * 100$$

where a is the number of species found in the first sample, b is the number of species found in the second sample, and c is the number of species shared between the two samples.

The annual rate of species gain, loss, and turnover from 1992 to 2020 was calculated according to Anderson 2007. Turnover, measured as the rate species are lost and replaced over time, provides insight into whether species composition is stabilizing (Matthews and Endress 2010). The annual proportional species gain (G_p), loss (L_p), and turnover (T_p) was calculated with presence/absence data as:

$$G_p = \frac{G}{(0.5)(S_{t_1} + S_{t_2})(Dt)} \quad (yr^{-1})$$

$$L_p = \frac{L}{(0.5)\left(S_{t_1} + S_{t_2}\right)(Dt)} \quad (yr^{-1})$$

$$T_p = \frac{G + L}{(0.5)(S_{t_1} + S_{t_2})(Dt)} \quad (yr^{-1})$$

where S_t is species richness, G is the number of species observed in t_2 not observed in t_2, and L is the number of species in t_1 not observed in t_2, and Δt is the time interval between surveys. Based on our intensive sampling efforts, it was assumed that a species not found in a survey had disappeared from the wetland and was not an artifact of sampling. Additionally, G_p was calculated for the 1985 – 1992 period based on the assumption that no wetland species were present immediately after restoration due to lack of initial plantings (Garlo 1992, Padgett and Crow 1994).

Avian community sampling and analysis

The avian community was surveyed from early June to late October 2020, encompassing the late breeding, nesting, and fall migration seasons of wetland-associated and wetland-dependent birds (Conway 2011). COVID-19 related restrictions on research activities at the University of New Hampshire prevented surveying during recommended breeding and spring migration seasons of northern New England (Conway 2011). The distribution of *Alnus*, *Frangula*, and *Rhus* shrubs on the constructed islands created natural barriers dividing the wetland into distinct zones. Surveyors created four permanent vantage points along the perimeter and an additional point on an immediately adjacent, hydrologically-connected pond where vegetation was not surveyed (Figure 16). The size of the birding zones ranged from 0.29 – 0.87 ha, and the furthest visual extent within each zone ranged from 75 – 120 m.

We conducted repeated point count surveys to estimate species richness and relative abundance (Nur et al. 1999). Bird surveys were conducted biweekly (at least 10 days apart) and completed between 0700 and 0845 hours. Point count surveys were conducted in the same order with the same personnel every visit. Avian surveys were conducted outside of the wetland, at a distance sufficient to not alter the behavior of the birds. Surveys were not conducted during fog, rain, heavy rain or with loud noise from an adjacent lumber yard. Repeated point count surveys consisted of a 10 minute passive visual and audial surveys (Conway 2011). All individuals seen or heard breeding, nesting, foraging, or resting in the wetland, including the upland islands, or on the immediate edge were recorded.

Each bird species was classified as wetland-dependent, wetland-associated, or upland based on the classification of Brooks and Croonquist 1990. Bird species assigned a score of 5 were classified as wetland-dependent, 3 as wetland associated, and 1 or 0 as upland. The species richness, wetland richness, mean abundance, diversity, and eveness were calculated for the avian community (Nur et al. 1999). Diversity was calculated using a modified Shannon-Weiner Diversitiy Index (*H'*):

$$H' = -\sum p_i \ln{(p_i)}$$

where *H'* is the Shannon-Weiner Diversity Index and H'_{max} is the idealized *H'* of the community where all species are equally abundant (Pielou 1966). where p_i is the relative abundance of each bird species across all of the summer and fall surveys (Shannon and Weaver 1949, Jost 2006). Eveness of the community was calculated using Pielou's J (*J'*):

$$J' = \frac{H'}{H'_{max}}$$

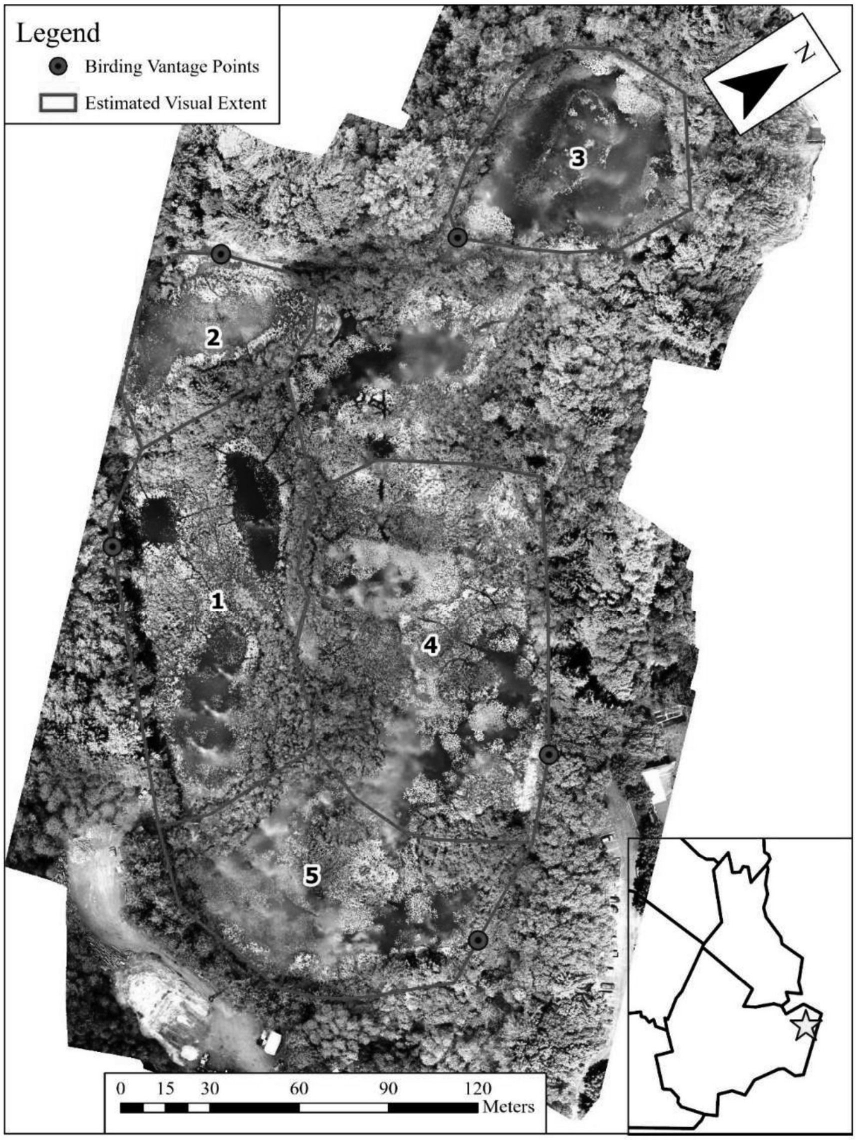

Figure 17: The point count vantage point locations and the estimated visual extent of each vantage point at Quarry Pond in Portsmouth, New Hampshire.

Results

Plant species composition shifts

A total of 129 plant species, comprising 54 families and 85 genera, were identified in the 2020 floral survey (Appendix II). The community experienced an increase of 19 species, 14 genera, and 6 families since the last survey in 2002 (Table 9). Despite a net increase in richness, 21 species observed in 2002 were not observed in the 2020 survey. Additionally, 9 of those species had been present in both 1992 and 2002. The survey identified 12 species that were found in the 1992 yet absent in 2002. Consistent observations have been made across the three survey intervals for two New Hampshire state listed endangered species, *Cardamine bulbosa* and *Potamogeton foliosus*, and one threatened species, *Sparganium eurycarpum* (NHNHB 2020). In our most recent survey, *Typha* x *glauca* was not observed but it's likely that this cryptic hybrid remains due to dense neighboring stands of *T. angustifolia* and *T. latifolia* and its documented presence in 2002.

The species composition of the wetland remained generally similar over the observation period from 1992 to 2020 despite species turnover at each interval. Net changes in the 2002 and 2020 observation intervals were seemingly minor, despite measurable losses and subsequent gains (Tables 10, 11). There was a net gain of 9 new species from 1992-2002; then another net gain of 19 species from 2002-2020. The inclusion of the 2002 survey illustrates steady trends of turnover as the Sorenson Index remained steady between 71 – 75% over the past 28 years (Table 12).

The wetland plant community remained primarily composed of natives, perennials, hydrophytes, and forbs throughout the 28 years of monitoring. Perennials continually dominated after seven years post-construction, fluctuating between 86 – 91% of the community. Natives remained above 90% for all surveys. Forbs interestingly represented exactly 60% of the community for all three surveys. The proportion of graminoids steadily declined by 5% as woody species replaced them. Upland species had a marked increase from 0% to 6% of the community from 2002 to 2020. The eight upland species were represented by a lone individual thus not contributing significantly to the plant community. These species were exclusively found at the upper edges of the wetland on top of *Carex stricta* tussocks. The inclusion of these species in the 2020 survey contributed to the shift in PI from 1.38 to 1.61 between 1992 to 2020.

Table 9: Descriptions of the vegetation community from each floristic survey of Quarry Pond. The percent of the community is reported for each classification of wetland indicator status, native status, life history, and growth habit. Wetland species consist of those classified as OBL, FACW, and FAC. Upland species consist of those classified as FACU and UPL. Coefficient of conservatism reported as mean ± standard error.

Vegetation Metric	1992	2002	2020
Community Description			
Family	46	48	54
Genera	67	71	85
Species	101	110	129
Conservation Value			
Average Coefficient of Conservatism	3.65 ± 0.15	3.95 ± 0.16	3.63 ± 0.16
Floristic Quality Assessment	36.7	41.5	41.2
Wetland Status			
Prevalence Index	1.41	1.38	1.62
Wetland	101	110	121
Upland	0	0	8
Native Status			
Native	96	97	93
Exotic	4	3	7
Life History			
Annual	14	9	14
Perennial	86	91	86
Growth Habit			
Forb/herb	60	60	60
Graminoid	28	25	23
Woody %	12	15	17

Table 10: Descriptions of the species gained between floral surveys of Quarry Pond. The number of species are reported for each classification of wetland indicator status, native status, life history, and growth habit. Wetland species consist of those classified as OBL, FACW, and FAC. Upland species consist of those classified as FACU and UPL. Coefficient of conservatism reported as mean ± standard error.

Metric	1992 - 2002	2002 - 2020	1992 - 2020
New Species	31	40	47
Conservation Value			
Average Coefficient of Conservatism	4.61 ± 0.33	3.38 ± 0.34	4.37 ± 0.27
Wetland Status			
Prevalence Index	1.32	2.15	1.98
Wetland	30	25	39
Upland	1	15	8
Native Status			
Native	30	34	43
Exotic	1	6	4
Life History			
Annual	3	8	8
Perennial	28	32	39
Growth Habit			
Forb/herb	18	21	28
Graminoid	8	10	7
Woody	5	9	12

Table 11: Descriptions of the species lost between floral surveys of Quarry Pond. The number of species are reported for each classification of wetland indicator status, native status, life history, and growth habit. Wetland species consist of those classified as OBL and FACW. Upland species consist of those classified as FAC, FACU, UPL.

Metric	1992 - 2002	2002 - 2020	1992 - 2020
Lost Species	22	21	19
Conservation Value			
Average Coefficient of Conservatism	3.50	4.90	3.87 ± 0.31
Wetland Status			
Prevalence Index	1.41	1.38	1.37
Wetland (OBL + FACW)	20	20	19
Upland (FAC + FACU + UPL)	2	1	0
Native Status			
Native	19	21	19
Exotic	3	0	0
Life History			
Annual	7	0	4
Perennial	15	21	15
Growth Habit			
Forb/herb	13	10	12
Graminoid	9	7	5
Woody	0	4	2

The conservation value of the wetland complex peaked in 2002 and then declined slightly by 2020 (Table 9). The average CoC of the community peaked at 3.95 in 2002 before declining to 3.63, the lowest value of all three surveys. The species gained and lost between 2002 and 2020 had average CoCs of 3.38 and 4.86, respectively. The FQA peaked at 41.5 in 2002 and only declined slightly by 2020 (Table 9). The FQA was supported by an increase in species richness despite a decline in the average CoC. The losses of highly senstivie species and gains of exotic and generalists drove the declines in conservation values.

The rates of change for the wetland community declined over time driven by decreases in species losses and turnover (Table 12). G_p was calculated as 0.286 yr^{-1} for the initial seven years post-construction. G_p (rate of species gain) decreased dramatically after the initial seven years to 0.029 yr^{-1} in 1992 to 2002 and then continued to decline in 2002 to 2020. L_p (rate of species loss) decreased by over half from 0.021 yr^{-1} to 0.010 yr^{-1} in 2002 to 2020. T_p (rate of successional turnover) followed the same pattern decreasing from 0.050 yr^{-1} to 0.028 yr^{-1}, respectively. Overall, the average T_p for the wetland complex was 0.020 yr^{-1} for 1992 to 2020, suggesting low but steady rates of successional turnover after seven years post-construction.

Table 12: Rates of species gains, loss, and turnover of the vegetation community of Quarry Pond. Sorenson Index is a measure of the similarity of the vegetation community between two surveys.

Metric	1986 - 1992	1992 - 2002	2002 - 2020	1992 - 2020
Species Shifts				
Similar Species		79	89	82
New Species	101	31	40	47
Lost Species		22	21	19
Annual Proportional Rate of Change				
G_p - Species Gain (yr^{-1})	0.333	0.029	0.019	0.015
L_p - Species Lost (yr^{-1})		0.021	0.010	0.006
T_p - Species Turnover (yr^{-1})		0.050	0.028	0.020
Community Similarity				
Sorenson Similarity (%)		74.9	74.5	71.3

Wetland Habitat Description of 2020

The floral survey of 2020 detailed a structurally complex and heterogeneous temperate freshwater wetland. We delineated eight wetland communities described by Sperduto and Nichols 2011: cattail marsh, aquatic bed, scrub-shrub swamp, mixed tall graminoid – scrub-shrub marsh, tall graminoid meadow marsh, seasonally flooded red maple swamp, emergent marsh, and sedge meadow marsh. Several areas consisted of a transition stage between cattail marsh and scrub-shrub swamp and were described as mixed cattail – scrub-shrub marsh as a ninth habitat. The upland islands remained a prominent feature and were dominated by *Rhus typhina* shrubs and several mature *Pinus strobus* trees. There was a significant difference in plot-level understory species richness across the four aggregated community groups ($F_{3, 91} = 11.9$, $p < 0.001$). Species richness was divided into two tiers of high richness in sedge meadow marshes (5.7 ± 0.6 SE) and woody swamps (5.1 ± 0.5) and low richness in cattail marshes (3.1 ± 0.3) and aquatic beds (2.7 ± 0.2).

Cattail marsh is the largest community type and is dominant throughout the wetland complex. The habitat was the most diverse community with at least 30 species recorded, though most forbs were found in only one or two plots. The community was dominated by *Typha latifolia* (15.3% mean cover) with notable presence of *Carex* sp. (1.6%), and *Nupha variegata* (1.2%) at pool edges. A small, dense stand of *Typha angustifolia* was found just south of the red maple swamp. The mixed cattail – scrub-shrub marsh habitats were located along channels or at the edges of pools and remained flooded or at least well-saturated during the growing season. The shrub canopy was short in stature (< 9 ft typically), sparse (< 35%), and consisted of *A. incana* (25.8%) and *F. alnus* (6.7%). The patch of mixed cattail – scrub-shrub marsh in the southern end of the wetland consisted mostly of *V. corymbosum*, though plot data captured the few *A. incana* shrubs.

In addition to *T. latifolia* (11.7%), other regularly found species were *Phalaris arundinacea* (2.0%) and *Lythrum salicaria* (1.3%).

The aquatic bed habitat is a series of pools connected by channels maintained by beavers and muskrats. The habitat were dominated by *Nuphar variegata* (16.5%), *Brasenia schreberi* (15.4%), and *Utricularia macrorhiza* (5.5%). Vegetation were primarily located along the edges of pools and deeper channels. Isolated patches of *Potamogeton* sp. were found mixed in with *B. schreberi* and *N. variegata.* Duckweed species of *Lemna minor* and *Wolffia columbiana* were found predominantly at the edge of aquatic bed and cattail marsh. *Vallisneria americana* had colonized deeper portions of the pools, avoiding possible shading by *N. variegata* and *B. schreberi*.

The scrub-shrub swamp community type is found predominantly as a long, narrow band and two additional large patches. The larger thicket in the southwest and a small thicket in the southeast section of the wetland could be described as alder seepage thickets with the presence of *Onoclea sensibilis*, *Impatiens capensis*, *C. stricta*, and other forbs and herbs. The absence of certain characteristic flora like *Caltha palustris* and a well-defined shallow muck soil layer makes classification difficult (Sperduto and Nichols 2011). The channel in the southwest thicket dried out by July and allowed for a diverse herb community to take advantage of the partial open canopy and saturated soil conditions such as *Alisma subcordatum*, *Cicuta maculata*, *Mimulus ringens*, and *Scirpus cyperinus*. Overall, the dominant shrubs in the community were *A. incana* (29.7%), *F. alnus* (18.0%), and *V. corymbosum* (5.3%). The ground cover was primarily *C. stricta* (25.7%) with frequent presence of *O. sensibilis* (3.6%), *F. alnus* seedlings (2.9%), and *Carex* sp. (2.2%).

A small patch of seasonally flooded red maple swamp has developed in a depression in the north. The red maple tree canopy cover comprised enough cover (30.5%) to classify the area as a seasonally red maple swamp. Additionally, several trees are only moderately sized and will only increase the red maple canopy over time. A small berm with an opening allows for seasonal flooding and lasting inundation and soil saturation into the growing season, creating the proper hydrological conditions. The presence of *Symplocarpus foetidus* early in the growing season is an excellent indicator of the hydrology. The red maple swamp possesses a shrub layer mostly dominated by *F. alnus* (21.7%). Few *Acer rubrum* seedlings or saplings were found during meander surveys, bringing the long-term health of the habitat into question.

The meadow marsh communities are scattered isolated patches on raised elevations that experience spring seasonal flooding and slow water drawdown in the rest of the growing season. The true sedge meadow marsh is located on a gas pipline right-of-way at the northest boundary of the wetland complex and was notable for the lack of *Carex strica* compared to the rest of the meadow marsh patches. Seasonal flooding from an outlet channel, constant saturated soil conditions from groundwater fluxes, and annual maintenance mowing creates conditions for characteristic sedge meadow marsh vegetation despite the lack of *Sphagnum* moss present. Dominant vegetation found were *Schoenoplectus pungens* (10.0%), *Thelypteris palustris* (10.0%), *Lysimachia terrestris* (3.0%), and *Carex* sp. (0.5%). Other notable forbs and graminoids included *Juncus tenuis, Juncus canadensis*, *Lycopus americanus*, *Boehmeria cylindrica*, and *Symphiotrichum racemosum*.

The tall graminoid meadow marsh was defined by the dominant presence of *Carex stricta* with little or no presence of *Typha* species. The marshes were dominated by *C. stricta* (47.5%) with the presence of other forbs like *Boehmeria cylindrica* (7.5%) and uncommon graminoids like *Leersia oryzoides* (5.0%). The majority of the mixed tall graminoid – scrub-shrub marsh areas functioned as borders between uplands and cattail marshes or aquatic beds. The saturated shorelines were exclusively dominated by *A. incana* and *F. alnus* with *Carex stricta* as the dominant groundcover. The community were dominated by *C. stricta* (40.0%), *Juncus* species (2.5%), *Phalaris arundinacea* (1.0%) with some shrub canopy cover (< 50%). One patch of this marsh, located near the center of the wetland, functioned as a hostpot for herb diversity such as *Lysimachia terrestris*, *Persicaria amphibium*, *Sium suave*, *Gallium pastre*, *Lycopus uniflorus*, *Scutelleria galericulata*, and two *Triadenum* species. The shrub canopy layer was exclusively *F. alnus* (15.0%) and *A. incana* (8.0%).

Several narrow emergent marshes are present in the southeastern extent of the wetland and one small patch in the west. The emergent marshes act as small borderes between aquatic beds and cattails to uplands or scrub-shrub swamps. The marshes were present as vegetated mudflats exposed by water level drawdown after seasonal spring flooding. The emergent marshes have characteristic vegetation such as *Pontederia cordata* (3.7%), *Proserpinaca palustris* (3.3%), and *Utricularia macrorhiza* (5.3%). During meander surveys later in the growing season and further water level drawdown, *Ludwigia palustris*, *Sparganium americanum,* and *Eleocharis* sp. were additionally found. An emergent marsh community formed after the pool in the west mostly dried from July – October as a result of a severe drought in New Hampshire. This emergent marsh was viewed as an artifact of a drought, yet demonstrated the presence of a robust seed bank for emergent marsh vegetation.

Several common exotic species were present in the wetland during the 2020 survey. *Lythrum salicaria* and *Phalaris arundinacea* were widespread throughout QP with a plot presence frequency of 25 and 27%, respectively. The species do not seem to be outcompeting native species or forming dense monocultures based on low mean covers. In the shrub layer, *Frangula alnus* was present throughout the wetland complex, typically coexisting with *Alnus incana*. *F. alnus* had the highest presence frequency in the shrub canopy of 54% and second highest mean cover of 7.8%.

Additionally, *F. alnus* seedlings and saplings were found extensively, posing a future risk of a monoculture for the shrub canopy. Other exotic shrub species, *Lonicera morrowii* and *Rosa multiflora*, were found in isolated patches of mature scrub-shrub swamps mixed in with *Vaccinium corymbosum* and *A. incana*.

Wetland plant community shifts

The wetland communities were dynamic with major changes in areal extent and distribution in the past 28 years (Figures 17, 18). The wetland complex has become more heterogeneous as the number of communities increased from four (cattail marsh, tall graminoid meadow marsh, marshy moat, and aquatic bed) to nine. The only community to have not persisted was the marshy moat subcommunity in the southwest portion of the site. The cattail marsh expanded its range by replacing 42% and 37% of the tall-graminoid meadow marsh and aquatic bed areas of 1992, respectively. The total area of cattail marsh increased by 80% as the aquatic bed and tall graminoid meadow marsh decreased by 38% and 80%, respectively (Table 13). The areal coverage of the wetland complex expanded by 0.09 ha due to the development of the sedge meadow marsh and red maple swamp outside the original construction boundaries.

Table 13: Change in areas (ha) of wetland communities of Quarry Pond from 1992 to 2020. Interior upland area was excluded from total wetland area calculation.

Community Type	1992 Area (ha)	2020 Area (ha)
Aquatic Bed & Open Water	1.45	0.91
Cattail Marsh	0.64	1.15
Mixed Cattail - Scrub-shrub Marsh	0.00	0.08
Scrub-shrub Swamp	0.00	0.18
Seasonally Flooded Red Maple Swamp	0.00	0.08
Sedge Meadow Marsh	0.00	0.02
Tall Graminoid Meadow Marsh	0.46	0.09
Mixed Tall Graminoid - Scrub-shrub Marsh	0.00	0.13
Emergent Marsh	0.00	0.04
Marshy Moat	0.04	0.00
Interior Upland	0.29	0.29
Total Wetland Area (ha)	**2.59**	**2.67**

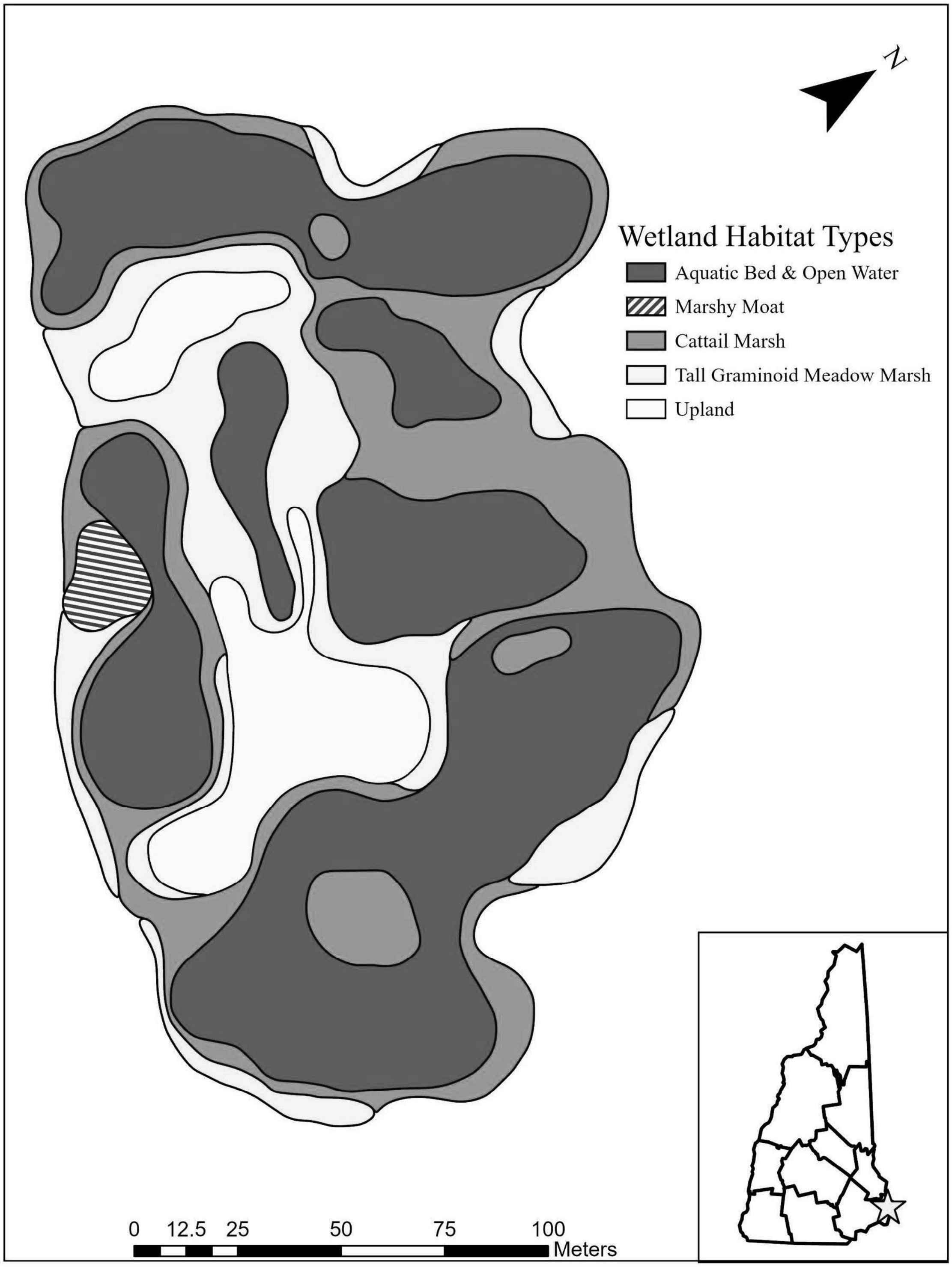

Figure 18: Wetland community distribution map of Quarry Pond in 1992 based on map and descriptions from Padgett and Crow 1994. Wetland community types were reclassified from dominant vegetation classification, georeferenced to drone imagery, and manually digitized in ArcGIS Pro. Wetland channels were not described in detail in 1992.

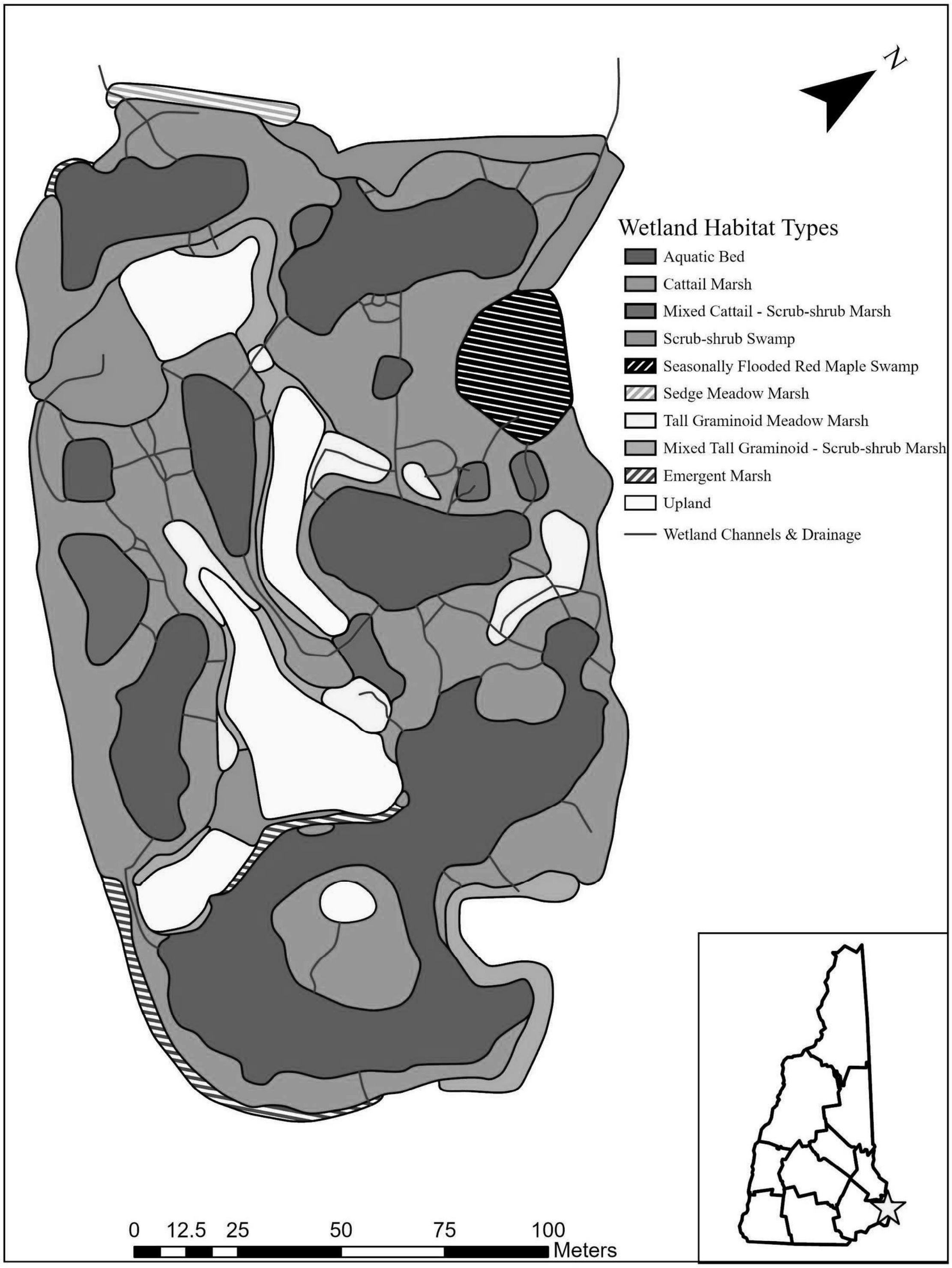

Figure 19: Wetland community distribution map of Quarry Pond in 2020 based on plot sampling and drone imagery. Wetland community types were classified based on plot sampling of ground cover, shrub, and tree canopy. Areas and wetland channels were manually digitized using drone imagery in ArcGIS Pro.

There was a shift of dominant species within the cattail marsh and aquatic bed communities since 1992. The understory herbs and forbs of cattail marshes had shifted from *Juncus effusus, Lemna minor,* and *Phalaris arundinacea* to *Nuphar variegata* and *Carex* sp. (Table 14). The expansion of cattail into permanent shallow waters is demonstrated by the presence of *N. variegata* intermixed within cattail reeds. In the aquatic bed pools, floating leaf aquatics shifted from *Potamogeton natans*, *Potamogeton pusillus*, and *Chara* sp. to a community dominated by *Brasenia schreberi* and *N. variegata*. The 2002 survey noted declines in *Chara* sp. and rise of *B. schreberi*, and the pattern has continued in the last 18 years. The dominant bladderwort species, *Utricularia gibba*, was also replaced with *Utricularia macrorhiza*, which was widespread and the only bladderwort species found in 2020.

Table 14: The average mean cover and plot frequency of understory species in the cattail marsh and aquatic bed communities of the 2020 floristic survey of Quarry Pond. All species with an average mean cover greater than 1.0 are shown. *Carex* sp. includes all *Carex* species except *C. stricta*. *Potamogeton* sp. includes all *Potamogeton* species except *P. natans.* Visual cover is reported as mean ± standard error.

Species	Visual Cover	Plot Frequency (%)
Cattail Marsh		
Typha latifolia	15.3 ± 1.2	97
Carex sp.	1.6 ± 0.8	16
Nuphar variegata	1.2 ± 0.6	13
Aquatic Bed		
Nuphar variegata	16.5 ± 4.2	59
Brasenia schreberi	15.4 ± 4.4	56
Utricularia macrorhiza	5.5 ± 1.7	59
Potamogeton sp.	2.0 ± 0.8	44
Potamogeton natans	1.4 ± 0.5	30

The graminoid meadow marsh communities experienced steep declines in area, shifts in geographical distribution, and turnover of dominant vegetation. In 1992, the meadow marshes would have been classified as tall graminoid meadow marshes due to the dominance of *Carex stricta* or co-dominance of *Phalaris arundinacea* and *Juncus effusus*. Cattail marsh had replaced *Phalaris*-*Juncus* meadow marshes by 2020, continuing a pattern noted in 2002. Additionally, an emergent marsh has replaced the *C. stricta* marsh in the south. The tall graminoid meadow marsh community continued, however, in the form of scattered *C. stricta* patches. The mixed tall graminoid – scrub-shrub form of the community dominate at the upland-wetland ecotone along the outer perimter and shores of interior uplands, where light gaps in the shrub canopy supports a *C. stricta* understory. The mixed tall graminoid – scrub-shrub marshes were not recorded in the first two floral surveys. The combination of both forms of tall gramminoid meadow marshes still represent only 47% of the original area.

The increase in shrub and tree canopy cover led to the development of four new wetland community types and a widspread presence of woody vegetation at the site over time. The combination of all communities with a shrub and tree canopy element would represent the third largest wetland community of 0.47 ha, a major increase since 1992. Previous floral studies did not mention any notable extent of shrub and tree cover. *Alnus incana* and *Frangula alnus* shrubs have expanded into the wetland, especially into cattail marsh demonstrated by the presence of the mixed cattail – scrub-shrub marshes. The narrow mixed tall graminoid - scrub-shrub marshes with *C. stricta* understory has replaced prior interior uplands or *Phalaris-Juncus* meadow marshes. The southestern patch of scrub-shrub swamp has formed on a prior upland island, potentially driven by the creation of a inlet channel by beavers. Lastly, the red maple swamp formed behind a berm (most likely a relict from construction), replacing previous uplands.

The marshy moat community was the only community to have been lost. The marshy moat patch, described in 1992 as peaty soils with *Eleocharis palustris, Vaccinium macrocarpon*, *Drosera intermedia*, and *Triadenum fraseri*, was replaced with a cattail marsh intermixed with *Alnus incana* and *Vaccinium corymbosum*. Characteristic vegetation such as *E. palustris* and *V. macrocarpon* are now missing and were mostly likely shaded out by *Typha latifolia* over time. The presence of *V. corymbosum* with patches of *Sphagnum* moss at shrub bases are most likely remnants of the saturated, peaty soils of the marshy moat community. The area currently is a diversity hotspot with *Spirea* shrubs, *Rumex crispus*, *Persicaria* sp., *Asclepias incarnata*, and *Triadenum* species.

Exotic Species

Two common exotic species, *Phalaris arundinacea* and *Frangula alnus*, followed noteworthy trends over the past 28 years. In 1992, *P. arundinacea* was widespread as *Phalaris-Juncus* meadow marshes comprised 18% of the wetland and the grass occupied 20% of the cover within the community. Although still present in 2020, *P. arundinacea* was sparse being found in only 27% of all non-aquatic plots with a low mean cover of 0.7 ± 0.2 (Table 15). In the shrub layer, *Frangula alnus* was present throughout the wetland, typically coexisting with *Alnus incana*. The exotic shrub exhibited a dramatic expansion since first identified in 2002. *F. alnus* had the highest plot frequency of 54% and second highest mean cover of 7.8 ± 1.3. Additionally, *F. alnus* seedlings and saplings were found extensively in the understory with a plot frequency of 20% and mean cover of 1.0 ± 0.3, posing a future risk of exotic dominance for the shrub canopy. Despite the rise in *F. alnus* abundance, the wetland woody richness only declined by one species from 2002 to 2020.

Table 15: Mean cover and plot frequency of common species of understory and shrub canopy across all non-open water plots of the 2020 floristic survey of Quarry Pond. All species with an average mean cover greater than 0.50 are shown. *Carex* sp. includes all *Carex* species except *C. stricta*. Bolded species are classified as exotic in New Hampshire. Mean cover is reported as mean ± standard error.

Species	Visual Cover	Plot Frequency (%)
Understory		
Typha latifolia	9.7 ± 1.0	72
Carex stricta	8.1 ± 2.2	24
Carex sp.	1.4 ± 0.5	21
Frangula alnus	**1.0 ± 0.3**	**20**
Persicaria amphibia	0.9 ± 0.3	17
Onoclea sensibilis	0.8 ± 0.5	13
Phalaris arundinacea	**0.7 ± 0.2**	**27**
Boehmeria cylindrica	0.6 ± 0.3	6
Nuphar variegata	0.6 ± 0.3	7
Lythrum salicaria	**0.5 ± 0.1**	**25**
Shrub Canopy		
Alnus incana	9.7 ± 1.8	44
Frangula alnus	**7.8 ± 1.3**	**54**
Vaccinium corymbosum	1.1 ± 0.6	6
Rhus typhina	0.9 ± 0.5	7

Avian survey

The 2020 point count surveys detected 430 individuals comprising 11 wetland-dependent, 3 wetland-associated, and 25 upland species (Appendix III). Monitoring efforts identified *Porzana carolina* and *Rallus limicola* as secretive marsh species using the cattail marsh. *P. carolina* is listed as a species of Special Concern for the State of New Hampshire (NHFWGD 2009). Although wetland birdscomprised 45% of total individuals, *Agelaius phoeniceus* (Red-winged blackbird) consisted of 68% of all wetland individuals when flocks of over 30 birds were observed throughout

June and July (Table 16). The wetland was heavily utilized by the avian community with 47.8 ± 9.3 individuals and 12.1 ± 0.9 species observed per survey. The community recorded species diversity of 14.81 H' and eveness of 0.38 J across the entire season of monitoring.

Table 16: The five most common wetland (dependent and associated) and upland species of the avian community monitoring in 2020. Mean abundance is reported as mean ± standard error.

Common Name	Surveys Present	Total Individuals	Proportion of Community (%)	Mean Abundance (indiv. per survey)
Wetland Species				
Red-winged Blackbird	6	132	30.6	14.7 ± 7.3
Mallard	3	19	4.4	2.1 ± 1.3
Wood Duck	3	15	3.5	1.7 ± 1.0
Common Yellowthroat	3	4	0.9	0.4 ± 0.2
Great Blue Heron	4	4	0.9	0.4 ± 0.2
Upland Species				
Common Grackle	5	41	9.5	4.6 ± 2.6
Gray Catbird	9	33	7.7	3.7 ± 0.6
Blue Jay	9	32	7.4	3.6 ± 0.6
Black-capped Chickadee	5	23	5.3	2.6 ± 1.0
Song Sparrow	8	21	4.9	2.3 ± 0.7

Discussion

Drivers of species composition shifts

Plant species successional trends mirrored prior documented patterns in the literature of life history traits, growth habits, and rates of gain, loss, and turnover. Species gain rates during the first seven years of wetland development resembled early successional patterns for restored wetlands based on high rates of turnover and rapid increases in species richness (Reinartz and Warne 1993, Matthews and Endress 2010, Ahn and Dee 2011, Spieles 2014). Annuals initially dominate the species composition of wetlands and decrease over time from competition from perennials and clonal graminoids (DeBerry and Perry 2004, Matthews and Spyreas 2010). In this study, annuals comprised roughly 10 - 15% of the community after seven years. It is unknown if annuals dominated the site during the first six years as we do not have published records of site monitoring that might have occurred.

The permanent shifts from annuals to perennial and woody species are in agreement with other long-term wetland successional studies and models (Noon 1996, Aronson and Galatowitsch 2008). The proportion of different species guilds remained relatively constant as well, except for a small increase in woody species and decrease in forbs. The species composition of the vegetation community is stabilizing and followed documented successional trends based on the species gain, loss, and turnover rates declining by half between 17 and 35 years post-restoration (Anderson 2007, Matthews and Endress 2010). Despite major shifts in the distribution of wetland communities, over 60% of the original species were found 28 years after the initial floristic survey. Atkinson et al. 2005 found that the species composition remained static for 20-year-old restored wetlands, attributing the stability to the resilience of dominant perennials.

The seed bank within the peat from the original wetland most likely allowed for rapid colonization and for seeds to persist until suitable germination conditions developed (Mitsch and Gosselink 2015). Wetland seed banks in general are vertically and laterally heterogeneous and reflect long-term hydrological conditions and successional history (LaDeau and Ellison 1999). Seed banks of prairie pothole systems have been shown to be viable up to 20 years post-draining (Galatowitsch and Van der Valk 1996). Prior restorations utilizing transplanted wetland soils have shown considerable success with high wetland richness and cover in short time frames (Brown and Bedford 1997, Heaven et al. 2003). Although not directly studied, the application of the original seed bank in the restoration process probably led to both rapid colonization in the first seven years and long-term stabilization of the species composition.

Topographical complexity within the wetland might have allowed for persistence of specialized flora. Original construction details were aimed at creating pools connected by channels and gentle sloped mounds throughout the wetland. Efforts to create heterogenous hydrological conditions through manipulation of topography, even on centimeter scales, have been shown to increase species richness in restored wetlands (Vivian-Smith 1997, Moser et al. 2007). *Carex stricta* tussocks of the meadow marshes can function as natural microtopographic hummocks that provide refugia from prolonged flooding, microhabitat variation in moisture and redox conditions, and greater light penetration (Larkin et al. 2006, Peach and Zedler 2006). The *C. stricta* tussock meadow marshes had the greatest species richness and functioned as plant diversity hotspots, where certain species like *Hypericum boreale*, *Campanula aparanoides*, and *Scutellaria* sp. exclusively resided.

Species richness in restored wetlands is predicted to peak and decline, or plateau, within 15 years (Campbell et al. 2002, Mitsch et al. 2012). Species richness in this study continued to increase over time. Upland species partially drove richness gains as their share of the community increased by 6% since 2002. Only one individual was found for all upland species and were all primarily located in the narrow borders of *C. stricta* at the upland-wetland edge. If gains in these upland species are excluded, the species richness in this study seems to be stabilizing as the richness increased roughly the same amount but in double the time (1992 – 2002 vs. 2002 – 2020). Woody species likely took advantage of suitable dry soil conditions in the wetland-upland edge from a prolonged drought period spanning 2016 – 2017 to establish (United States Drought Monitor 2020). Upland annuals found in saturated soils in May and June at the wetland edge are more difficult to explain, but likely will not persist with the continuance of typical hydrologic conditions (Campbell et al. 2002).

Mechanisms of trajectory shifts

From a community perspective, the wetland complex remained dynamic after the proposed maturation period of 15 – 20 years (Mitsch and Wilson 1996). The restoration trajectory of the wetland community structure shifted in the past 18 years based on the floral survey of 2002. The shifts could be seen through (1) cattail marsh replacement of aquatic bed and meadow marsh habitat, (2) successional shifts of dominant vegetation within certain communities, (3) development of woody vegetation, (4) formation of smaller and more specialized niches, and (5) divergent trajectories of the invasives *Phalaris aundinacea* and *Frangula alnus*. Possible mechanisms of long-term shifts in restoration trajectory may be alterations of the hydrological regime and shoreline herbivory from beaver activity and nonlinear development rates of different wetland communities.

Beaver activity can induce major vegetation community shifts within riparian zones and existing wetlands by raising water table elevation, increasing shoreline complexity, removing woody vegetation on shorelines, and altering nutrient cycling dynamics (Naiman et al. 1986, McMaster and McMaster 2001, Cunningham et al. 2006, Hood and Larson 2015). Increased flooding, grazing on macrophytes, and tree felling from beaver activity has been attributed to successful wetland restoration projects (Law et al. 2017). Beaver reintroductions on degraded wetlands can increase plant species richness, within site heterogeneity, and community evenness (Law et al. 2014, 2017). The impacts of beaver activity may not be fully realized until at least 10 years after colonizing a site (Law et al. 2014). It is possible that beaver activity might have shifted the long-term trajectory of the vegetation community at Quarry Pond by altering the hydrological regime and maintaining a sparse shrub canopy. An increase of seasonal flooding depth created proper hydrologic conditions for the red maple swamp and sedge meadow marsh to develop. Additionally, the consumption of *A. incana* maintains scrub-shrub – tall graminoid meadow marsh community on shoreline banks by increasing light availability for *C. stricta* and associated herbaceous understory (Law et al. 2017). Rentch et al. 2015 attributed tree felling in scrub-shrub swamps to the persistence of rare understory flora in open light gaps in West Virginia wetlands.

Exotic species invasion is a common factor in assigning failure for restored wetlands to reach regulatory success and ecological parity with natural references. Common invasives of temperate wetland were present in both the understory and shrub layers (Galatowitsch et al. 1999, Mills et al. 2009). Notably, *Phalaris arundinacea* abundance was reduced consierably since 1992 as tall graminoid meadow marsh habitat declined. Although *P. arundinacea* rapidly colonized the wetland complex within seven years, it has not formed dense monocultures, common in wetland restoration projects (Apfelbaum and Sams 1987). The decline of *P. arundinacea* is encouraging for the future of this restoration project since *P. arundinacea* has been shown to reduce species richness (Perkins and Wilson 2005), prevent the development of graminoid meadow guilds (Brown and Bedford 1997, Price et al. 2018), and increase biotic homogenization across landscapes (Aronson and Galatowitsch 2008, Sueltenfuss and Cooper 2019). The natural *P. arundinacea* decline warrants future research given documented negative impacts on wetland restoration projects and expensive, time-consuming control options (Lavergne and Molofsky 2006).

The shrub layer of the wetland complex experienced a widespread expansion of *Frangula alnus* since its presence was first documented in 2002. Mills et al. 2009 documented a similar expansion within a 20 year timespan in Wisconsin. Possible mechanisms for *F. alnus* invasion are release from disease and herbivory pressure (Zedler and Kercher 2004), avian dispersal of attractive fleshy fruits (Craves 2015), and high tolerance for varying soil and hydrologic conditions (Mills et al. 2012, Berg et al. 2016). Bird species exclusively observed in the exotic shrubs such as *Dumetalla carolinensis* (Gray catbird) and *Cyanocitta cristata* (Blue jay) may be vectors for seed distribution (Craves 2015). Additionally, this study confirms previous findings that *F. alnus* develops a dense understory of seedlings allowing for high recruitment of propagules (Mills et al. 2012). The wetland woody species remained stable as *F. alnus* canopy cover increased in the past 18 years, suggesting the exotic shrub might not be negatively impacting the vegetation community. However, a lag time remains to be seen between the development of *F. alnus* and decline in shrub species richness (Mills et al. 2009).

Avian community as a indicator of habitat quality

The presence of scrub-shrub swamp, cattail marsh, and aquatic bed habitat supported a diverse avian community including 11 wetland-dependent, 3 wetland-associated, and 2 secretive marsh species. Habitat heterogeneity allowed for waterfowl and marsh birds to utilize preferred habitats for foraging, breeding, and roosting. The dominant expanse of cattail marsh supported secretive marsh species *P. carolina* and *R. limicola* and large flocks of *A. phoeniceus* (VanRees-Siewert and Dinsmore 1996, Glisson et al. 2015). Wetland species including *Ardea herodias* (Great blue heron), *Butorides striatus* (Green heron), *Aix sponsa* (Wood duck), and *Anas platyrhynchos* (Mallard) were predominantly found in open water and emergent masrsh habitat. The formation and maintenance of pools by beavers in wetlands has been documented to increase waterfowl abundance and richness (Brown et al. 1996, McKinstry et al. 2001). By raising water levels and grazing, beaver activity might have indirectly benefitted certain waterfowl in this study by preventing further expansion of cattails into pools.

The wetland complex possibly supported the avian community of the immediate upland forests and greater Packer Bog. The most common upland bird species, with the exception of *Quiscalus quiscula* (common grackle), were found primarily on *Alnus* and *Frangula* shrubs on the wetland edge. *F. alnus* fruits are commonly foraged by the common upland species including *Mimus polyglottos* (Northern mockingbird), *Dumetella carolinensis*, and *Zomotrichia albicollis* (white-throated sparrow) (Craves 2015). Additionally, certain upland bird species were predominantly found in wetland habitat such as *Melospiza melodia* (song sparrow) in cattail marsh. Hapner et al. 2011 documented increases in upland species richness and density in created wetlands over time, attributing the increases to expansion of emergent and woody vegetation. As the vegetation structure shifted the avian community likely responded, especially with increases of those reliant on cattail and woody vegetation (VanRees-Siewert and Dinsmore 1996, Muir Hotaling et al. 2002). Long-term monitoring of both habitat structure and avian community response would provide a better framework for approaching wetland restoration projects aimed at boosting waterfowl populations.

Long-term restoration success

Non-linear development of certain wetland communities may initially appear diverging from restoration goals or reference conditions. At Quarry Pond, woody vegetation (scrub-shrub and red maple swamps) and sedge meadow marsh required at least 20 years to develop after increased flooding into the areas from beaver activity. Long-term studies of prairie pothole systems have noted lag times for similar woody vegetation and wet prairie wetland communities (Mulhouse and Galatowitsch 2003, Spieles et al. 2006). Additionally, Sueltenfuss and Cooper 2019 documented that proper hydrology does not lead to similar wetland communities within a watershed even after 15 years. Targeted restoration activities such as planating propagules, seeding, or long-term maintenance may be needed to jump-start, or at least hasten, these late developing communities and prevent permanent divergence from reference conditions (Galatowitsch 2006, Gutrich et al. 2009).

Quarry Pond has developed a diverse vegetation community with prominent open water, herbaceous, shrub, and tree elements within 35 years since creation. The wetland remained primarily composed of hydrophytic natives. The ecological features of the nine wetland subcommunities now support 121 wetland plant species. *Frangula alnus* is the only prominent invasive species in the wetland. The structurally complex wetland supported a diverse avian community including 14 wetland species that actively utilize different herbaceous, shrub, and tree elements. Despite the development of approximately 0.178 ha of scrub-shrub habitat within the restoration site footprint, this mitigation project has not fully replaced the original 4 ha of intact scrub-shrub swamp lost in 1985. However, if the goal was to create a functioning freshwater wetland which supports diverse, native flora and fauna communities, the Quarry Pond compensatory wetland mitigation project would appear to be successful and self-sustaining given present site conditions. Long-term monitoring and flexible restoration goals to account for possible restoration trajectory shifts should be incorporated in future freshwater mitigation projects to avoid anthropogenic bias that limits assigning success.

Conclusion

Freshwater wetland mitigation projects should strive to incorporate long-term monitoring and set flexible restoration goals take into account possible shifts in the vegetation community. Restoration practicioners can utilize a destroyed wetland's seed bank for rapid establishment of hydrophytic vegetation while focusing on the construction of mound and pool topography, inclusion of beavers, and enhancement of *C. stricta* presence to increase niche space and species richness. Goals to restore late-developing communities like scrub-shrub swamps or sedge meadows might require more than 20 years to achieve without targeted initial actions and continued maintenance. Long-term, repeated monitoring of sites can provide guidance for future projects by documenting site-specific successional trends and driving mechanism. It is possible the restoration trajectory of Quarry Pond will converge to the lost scrub-shrub swamp, but nevertheless, the wetland contributes ecological benefits through its functions and values documented herein.

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APPENDIX I: MEANS ± STANDARD ERROR VALUES FOR METRICS USED IN THE RESTORAITON PERFORMANCE INDEX MODEL

Site Characteristics			Vegetation		Pore Water Chemistry				Nekton	
Site	**Year**	**Treatment**	**Halophyte Cover**	**Species Richness (# / plot)**	**Salinity (psu)**	**Redox Potential (mV)**	**pH**	**Sulfide (mM)**	**Average Catch (# per trap)**	**Adult Length (mm)**
Wagon Hill Farm	2019	Living Shoreline								
		Low	25.1 ± 3.7			149.9 ± 32.3				
		High	57.8 ± 5.4	1.7 ± 0.2		281.4 ± 26.4				
		Reference								
		Low	98.5 ± 1.1		28.7 ± 0.5	-286.9 ± 20.8	6.6 ± 0.1	2.4 ± 0.5		
		High	94.4 ± 2.6	3.5 ± 0.4	26.9 ± 1.3	-209.5 ± 42.3	6.5 ± 0.1	1.5 ± 0.5		
		Control								
		High	64.4 ± 13.5	2.3 ± 0.4	21.0 ± 1.6	-136.1 ± 51	6.9 ± 0.1	0.4 ± 0.3		
	2020	Living Shoreline								
		Low	38.2 ± 4.4		35	52.3 ± 58.7			37.6 ± 16.6	58.8 ± 2.3
		High	8.4 ± 1.8	2.8 ± 0.2		235.1 ± 78.9			29.1 ± 19.5	58.0 ± 2.0
		Reference								
		Low	63.0 ± 3.3		35.8 ± 0.9	-343.4 ± 35.6	6.6 ± 0.1	1.5 ± 0.3	2.6 ± 1.3	59 ± 1.6
		High	77.0 ± 4.0	3.5 ± 0.5	31.9 ± 1.1	-295.5 ± 20.9	6.2 ± 0.2	1.6 ± 0.6	2.3 ± 1.1	59.6 ± 4.8
		Control								
		High	22.0 ± 10.7	1.8 ± 0.3	30.0 ± 1.4	-87.4 ± 37.9	6.2 ± 0.0	1.2 ± 0.6	1.6 ± 0.6	54.9 ± 3.4
Cutts Cove	2019	Living Shoreline								
		Low	40.4 ± 8.8			389.3 ± 40.1				
		High	23.7 ± 4.4	2.8 ± 0.3		397.6 ± 34.7				
		Reference								
		Low	96.9 ± 1.4		16.8 ± 1.7	-329.1 ± 6.9	6.8 ± 0.0	3.0 ± 0.3		
		High	85.1 ± 7.9	2.4 ± 0.3	10.9 ± 1.4	-273.7 ± 14.8	6.5 ± 0.1	1.6 ± 0.4		
		Control								
		Low	0		27.6 ± 0.6	6.6 ± 21.6	7.5 ± 0.0	1.2 ± 0.2		
	2020	Living Shoreline								
		Low	13.1 ± 1.7			323.9 ± 56.5			20.5 ± 12.0	52.1 ± 1.4
		High	32.9 ± 3.8	2.6 ± 0.3		277.0 ± 62.6				

Site Characteristics			Vegetation		Pore Water Chemistry				Nekton	
Site	Year	Treatment	Halophyte Cover	Species Richness (# / plot)	Salinity (psu)	Redox Potential (mV)	pH	Sulfide (mM)	Average Catch (# per trap)	Adult Length (mm)
		Reference								
		Low	55.9 ± 4.4		27.2 ± 1.4	-310.4 ± 34.4	6.8 ± 0.1	3.0 ± 0.5	17.1 ± 14.8	53.7 ± 2.1
		High	60.8 ± 9.0	2.8 ± 0.2	24.2 ± 3.4	-179.4 ± 22.6	6.6 ± 0.1	1.7 ± 0.7		
		Control								
		Low	0.0		32.0 ± 0.4	-108.5 ± 86.4	7.4 ± 0.1	2.1 ± 1.0	1.0 ± 1.0	47.3
North Mill Pond	2019	Living Shoreline								
		Low	38.9 ± 14		27.6 ± 0.6	-41.7 ± 10.9	6.7 ± 0.1	0.4 ± 0.2		
		High	63.1 ± 11.5	4.3 ± 0.8		281.1 ± 106.3				
		Reference								
		Low	84.9 ± 4.6		28.6 ± 0.9	-314.9 ± 21.5	7.1 ± 0.1	2.5 ± 0.3		
		High	96.8 ± 1.5	6.2 ± 0.4		179.8 ± 97.5				
		Control								
		Low	0		26.3 ± 0.3	16.6 ± 34	7.4 ± 0.1	0.5 ± 0.1		
	2020	Living Shoreline								
		Low	28.5 ± 8.4		32.9 ± 0.9	14.0 ± 19.9	6.7 ± 0.1	0.4 ± 0.1	10.3 ± 3.8	54.5 ± 1.3
		High	49.1 ± 10.7	3.3 ± 0.7		134.0 ± 84.2				
		Reference								
		Low	51.0 ± 4.3		33.4 ± 0.7	-379 ± 9.4	7.0 ± 0.1	4.6 ± 0.1	31.1 ± 16.7	56.1 ± 2.0
		High	54.9 ± 8.8	4.6 ± 0.6	33.0 ± 1.8	-25.1 ± 79.5	6.8 ± 0.1	0.4 ± 0.1		
		Control								
		Low	0			-57.3 ± 39.3				
Portsmouth Christian Academy	2019	Reference								
		Low	88.8 ± 2.0		20.6 ± 0.6	-273.8	6.6 ± 0.1	3.0 ± 0.2		
		High	92.2 ± 3.5	3.1 ± 0.2	22.2 ± 0.4	-265.8 ± 21.9	6.4 ± 0	2.3 ± 0.3		
	2020	Reference								
		Low	48.1 ± 4.7		29.9 ± 0.8	-302.1 ± 20.6	6.7 ± 0.1	1.9 ± 0.3	76.3 ± 12.4	59.4 ± 1.1
		High	79.1 ± 4.0	3.0 ± 0.2	30.6 ± 1.3	-262.1 ± 27.9	6.2 ± 0.1	2.5 ± 0.5	5.9 ± 1.7	58.6 ± 1.6

The mean ± standard error is reported for each metric in the study. The mean for each metric was the only input into the Restoration Performance Index model.

APPENDIX II: COMPARISON OF SPECIES PRESENCE AND ABSENCE AGAINST THREE FLORISTIC SURVEYS OF QUARRY POND IN 1992, 2002, AND 2020

Taxon	1992	2002	2020
PTERIDOPHYTES			
DRYOPTERIDACEAE			
Onoclea sensibilis L.	X	X	X
EQUISETACEAE			
Equisetum arvense L.	X	X	X
OSMUNDACEAE			
Osmundastrum cinnamomeum (L.) C. Presl		X	X
Osmunda regalis (Willd.) Gray	X	X	X
THELYPTERIDACEAE			
Thelypteris palustris Schott	X	X	X
GYMNOSPERMS			
PINACEAE			
Pinus strobus L.			X
ANGIOSPERMS - DICOTYLEDONS			
ADOXACEAE			
Sambucus nigra L.			X
Viburnum dentatum L.			X
ANACARDIACEAE			
Toxicodendron radicans (L.) Kuntze			X
Toxicodendron vernix (L.) Kuntze		X	X
APIACEAE			
Cicuta bulbifera L.	X	X	X
Cicuta maculata L.			X
Sium suave Walt.	X	X	X
AQUIFOLIACEAE			
Ilex verticillata L.	X	X	X

Taxon	1992	2002	2020
APOCYNACEAE			
Asclepias incarnata L.	X	X	X
ASTERACEAE			
Bidens connata Muhl. ex Willd.	X	X	X
Bidens frondosa L.	X	X	X
Doellingeria umbellata (Mill.) Nees			X
Erechtites hieraciifolius (L.) Raf. ex DC.			X
Eupatorium perfoliatum L.	X	X	X
Euthamia graminifolia (L.) Nutt.	X	X	X
Eutrochium dubium (Willd. ex Poir.) E.E. Lamont		X	
Symphyotrichum lanceolatum (Willd.) Nesom	X		X
Symphyotrichum racemosum (Elliott) Nesom	X	X	X
BALSAMINACEAE			
Impatiens capensis Meerb.	X	X	X
BETULACEAE			
Alnus incana (L.) Moench	X	X	X
BRASSICACEAE			
Cardamine bulbosa (Schreb. ex Muhl.) B.S.P.			X
Cardamine pensylvanica Muhl. ex Willd.			X
Rorippa palustris (L.) Besser			X
CABOMBACEAE			
Brasenia schreberi J.F. Gmel.		X	X
CAMPANULACEAE			
Campanula aparinoides Pursch		X	X
CAPRIFOLIACEAE			
Lonicera morrowii Gray			X
CONVOLVULACEAE			
Calystegia sepium (L.) R. Br.			X
CORNACEAE			
Swida amomum (P. Mill.) Small	X	X	X
Swida sericea (L.) Holub	X	X	

Taxon	1992	2002	2020
DROSERACEAE			
Drosera intermedia Hayne	X	X	
ERICACEAE			
Lyonia ligustrina (L.) DC.	X	X	X
Vaccinium corymbosum L.	X	X	X
Vaccinium macrocarpon Aiton	X	X	
HALORAGACEAE			
Proserpinaca palustris L.	X	X	X
HYPERICACEAE			
Hypericum boreale (Britton) Bicknell	X	X	X
Hypericum canadense L.	X		
Hypericum dissimulatum Bicknell	X		
Hypericum ellipticum Hook.	X	X	
Hypericum mutilum L.	X		
Triadenum fraseri (Spach) Gleason	X	X	X
Triadenum virginicum (L.) Raf.			X
LAMIACEAE			
Lycopus americanus Muhl. ex W. Barton	X		X
Lycopus uniflorus Michx.	X	X	X
Mentha arvensis L.			X
Scutellaria galericulata L.	X	X	X
*Scutellaria lateriflor*a L.			X
LENTIBULARIACEAE			
Utricularia gibba L.	X		
Utricularia vulgaris L.		X	X
Utricularia minor L.	X	X	
LYTHRACEAE			
Decodon verticillatus (L.) Elliott			X
Lythrum salicaria L.	X	X	X
MYRICACEAE			
Morella caroliniensis (P. Mill.) Small	X	X	

Taxon	1992	2002	2020
MYRSINACEAE			
Lysimachia terrestris (L.) B.S.P.	X	X	X
NYMPHAEACEAE			
Nuphar lutea (L.) Sm.	X	X	X
Nymphaea odorata Aiton			X
ONAGRACEAE			
Epilobium ciliatum Raf.		X	X
Epilobium palustre L.	X		
Ludwigia palustris (L.) Elliott	X	X	X
OROBANCHACEAE			
Agalinis purpurea (L.) Pennell	X	X	X
Orobanche uniflora L.			X
OXALIDACEAE			
Oxalis dillenii Jacq.			X
PENTHORACEAE			
Penthorum sedoides L.	X	X	X
PHYRMACEAE			
Mimulus ringens L.	X	X	X
PLANTAGINACEAE			
Callitriche palustris L.	X	X	
Chelone glabra L.		X	
POLYGONACEAE			
Persicaria amphibia (L.) Gray	X	X	X
*Persicaria arifolia (*L.) Haroldson	X		
Persicaria hydropiper (L.) Opiz			X
Persicaria lapathifolia (L.) Gray	X	X	X
Persicaria pensylvanica (L.) Maza		X	X
Persicaria punctata (Elliott) Small	X	X	X
Persicaria sagittata (L.) Gross	X	X	X
Rumex crispus L.	X		X
Rumex pallidus Bigelow		X	

Taxon	1992	2002	2020
Rumex verticillatus L.		X	X
RANUNCULACEAE			
Clematis virginiana L.			X
Ranunculus sceleratus L.		X	X
RHAMNACEAE			
Frangula alnus Mill.		X	X
ROSACEAE			
Potentilla argentea L.			X
Rosa palustris Marshall		X	X
Rosa multiflora Thunb.			X
Spiraea alba Du Roi	X	X	X
Spiraea tomentosa L.	X	X	X
RUBIACEAE			
Galium palustre L.	X	X	X
SALICACEAE			
Salix eriocephala Michx.		X	
Salix lucida Muhl.		X	
Salix nigra Marshall	X	X	X
SAPINDACEAE			
Acer rubrum L.	X	X	X
SOLANACEAE			
Solanum dulcamara L.	X	X	X
URTICACEAE			
Boehmeria cylindrica (L.) Sw.	X	X	X
VERBENACEAE			
Verbena hastata L.	X	X	X
VIOLACEAE			
Viola lanceolata L.	X		
VITACEAE			
Parthenocissus quinquefolia (L.) Planch.			X

Taxon	1992	2002	2020
ANGIOSPERMS - MONOCOTYLEDONS			
ALISMATACEAE			
Alisma subcordatum Raf.	X	X	X
Sagittaria latifolia Willd.	X	X	X
ARACEAE			
Lemna minor L.	X	X	X
Symplocarpus foetidus (L.) Salisb. ex Nutt		X	X
Wolffia columbiana Karst.	X		X
CYPERACEAE			
Carex alopecoidea Tuck.			X
Carex atherodes Spreng.		X	X
Carex canescens L.		X	
Carex comosa Boott	X	X	X
Carex lenticularis Michx.		X	
Carex lupulina Muhl. ex Willd.	X	X	X
Carex lurida Wahlenb.	X	X	X
Carex pseudocyperus L.	X	X	X
Carex scoparia Schkuhr ex Willd.	X	X	X
Carex stipata Muhl. ex Willd.		X	
Carex stricta Lam.	X	X	X
Carex utriculata Boott		X	X
Carex versicaria L.			X
Carex vulpinoidea Michx.	X	X	X
Cyperus strigosus L.	X	X	X
Dulichium arundinaceum (L.) Britton	X		X
Eleocharis acicularis (L.) Roem. & Schult.	X	X	X
Eleocharis elliptica Kunth		X	
Eleocharis obtusa (Willd.) Schult.	X		X
Eleocharis palustris (L.) Roem. & Schult.	X	X	X
Eleocharis tenuis (Willd.) Schult.	X		
Rhynchospora capitellata (Michx.) Vahl	X		

Taxon	1992	2002	2020
Schoenoplectus acutus (Muhl. *ex* Bigelow) A. & D. Love			X
Schoenoplectus pungens (Vahl) Palla	X	X	X
Schoenoplectus tabernaemontani (K.C. Gmel.) Palla	X	X	X
Scirpus atrocinctus Fernald	X	X	
Scirpus cyperinus (L.) Kunth	X	X	X
Scirpus hattorianus Makino	X		X
HYDROCHARITACEAE			
Najas gracillima (A. Braun ex Engelm.) Magnus		X	X
Najas minor All.	X		X
Vallisneria americana Michx.		X	X
IRIDACEAE			
Iris versicolor L.	X	X	X
Sisyrinchium atlanticum Bicknell		X	
JUNCACEAE			
Juncus acuminatus Michx.		X	X
Juncus articulatus L.	X		
Juncus canadensis Gay ex Laharpe	X		X
Juncus effusus L.	X	X	X
Juncus tenuis Willd.			X
POACEAE			
Calamagrostis canadensis (Michx.) Beauv.	X	X	X
Echinochloa crus-galli (L.) Beauv	X		X
Glyceria borealis (Nash) Batchelder		X	
Glyceria canadensis (Michx.) Trin.	X	X	
Leersia oryzoides (L.) Sw.	X		X
Phalaris arundinacea L.	X	X	X
Poa palustris L.	X	X	X
PONTERIDACEAE			
Pontederia cordata L.	X	X	X
POTAMOGETONACEAE			
Potamogeton amplifolius Tuck.	X	X	X

Taxon	1992	2002	2020
Potamogeton bicupulatus Fernald		X	
Potamogeton foliosus Raf.	X		X
Potamogeton natans L.	X	X	X
Potamogeton pusillus L.	X	X	X
TYPHACEAE			
Sparganium americanum Nutt.		X	X
Sparganium eurycarpum Engelm. ex Gray	X	X	X
Typha angustifolia L.	X	X	X
Typha latifolia L.	X	X	X
Typha x *glauca* Godr.		X	X
	101	110	129

APPENDIX III: AVIAN COMMUNITY OF QUARRY POND FROM POINT COUNT SURVEYS OF 2020.

Common Name	Scientific Name	Number of Surveys Present	Total Individuals	Proportion of Community (%)	Mean Abundance (indiv. per survey)
Wetland Dependent & Associated Species					
Belted Kingfisher	*Megaceryle alcyon*	1	2	0.5	0.2 ± 0.2
Blue-winged Teal	*Anas discors*	1	1	0.2	0.1 ± 0.1
Common Yellowthroat	*Geothlypis trichas*	3	4	0.9	0.4 ± 0.2
Double-breasted Cormorant	*Phalacrocorax auritus*	1	3	0.7	0.3 ± 0.3
Great Blue Heron	*Ardea herodias*	4	4	0.9	0.4 ± 0.2
Green Heron	*Butorides virescens*	3	3	0.7	0.3 ± 0.2
Mallard	*Anas platyrhynchos*	3	19	4.4	2.1 ± 1.3
Palm Warbler	*Setophaga palmarum*	1	1	0.2	0.1 ± 0.1
Red-winged Blackbird	*Agelaius phoeniceus*	6	132	30.6	14.7 ± 7.3
Sora	*Porzana carolina*	1	1	0.2	0.1 ± 0.1
Spotted Sandpiper	*Actitis macularius*	3	3	0.7	0.3 ± 0.2
Swamp Sparrow	*Melospiza georgiana*	1	3	0.7	0.3 ± 0.3
Virginia Rail	*Rallus limicola*	1	2	0.5	0.2 ± 0.2
Wood Duck	*Aix sponsa*	3	15	3.5	1.7 ± 1.0
		Individuals	**192**	**44.8**	
		Richness	**14**		

Common Name	Scientific Name	Number of Surveys Present	Total Individuals	Proportion of Community (%)	Mean Abundance (indiv. per survey)
Upland Species					
American Goldfinch	*Spinus tristis*	5	6	1.4	0.7 ± 0.2
American Robin	*Turdus migratorius*	1	2	0.5	0.2 ± 0.2
Barn Swallow	*Hirundo rustica*	1	2	0.5	0.2 ± 0.2
Black-capped Chickadee	*Poecile atricapillus*	5	23	5.3	2.6 ± 1.0
Blue Jay	*Cyanocitta cristata*	9	32	7.4	3.6 ± 0.6
Brown Creeper	*Certhia americana*	1	2	0.5	0.2 ± 0.2
Cedar Waxwing	*Bombycilla cedrorum*	1	1	0.2	0.1 ± 0.1
Chipping Sparrow	*Spizella passerina*	2	10	2.3	1.1 ± 0.9
Common Grackle	*Quiscalus quiscula*	5	41	9.5	4.6 ± 2.6
Cooper's Hawk	*Accipiter cooperii*	1	1	0.2	0.1 ± 0.1
Downy Woodpecker	*Dryobates pubescens*	1	1	0.2	0.1 ± 0.1
Eastern Phoebe	*Sayornis phoebe*	2	2	0.5	0.2 ± 0.2
Gray Catbird	*Dumetella carolinensis*	9	33	7.7	3.7 ± 0.7
Mourning Dove	*Zenaida macroura*	4	5	1.2	0.6 + 0.2
Northern Cardinal	*Cardinalis cardinalis*	6	19	4.4	2.1 ± 0.8
Northern Mockingbird	*Mimus polyglottos*	2	2	0.5	0.2 ± 0.2
Pine Warbler	*Setophaga pinus*	2	2	0.5	0.2 ± 0.2
Red-breasted Nuthatch	*Sitta canadensis*	2	5	1.2	0.6 ± 0.4

Common Name	Scientific Name	Number of Surveys Present	Total Individuals	Proportion of Community (%)	Mean Abundance (indiv. per survey)
Ruby-throated Hummingbird	*Archilochus colubris*	1	1	0.2	0.1 ± 0.1
Song Sparrow	*Melospiza melodia*	8	21	4.9	2.3 ± 0.7
Tree Swallow	*Tachycineta bicolor*	2	4	0.9	0.4 +_0.3
Tufted Titmouse	*Baeolophus bicolor*	3	9	2.1	1.0 ± 0.6
White Throated Sparrow	*Zonotrichia albicollis*	2	11	2.6	1.2 ± 1.1
Yellow Warbler	*Setophaga petechia*	1	1	0.2	0.1 ± 0.1
Yellow Rumped Warbler	*Setophaga coronata*	1	1	0.2	0.1 ± 0.1
		Individuals	**237**	**55.2**	
		Richness	**25**		

APPENDIX IV: IACUC APPROVAL LETTERS

University *of* New Hampshire

Research Integrity Services, Service Building
51 College Road, Durham, NH 03824-3585
Fax: 603-862-3564

22-Aug-2019

Moore, Gregg E
Dept of Biological Sciences
Jackson Lab
Durham, NH 03824-3406

IACUC #: 190706
Project: Living Shorelines in New Hampshire: Restoration Performance of Floral and Faunal Communities

The Institutional Animal Care and Use Committee (IACUC) has reviewed and recommended approval of the protocol submitted for this study contingent upon your response to the following:

1. *The investigator needs to provide full references throughout the application for citations used.*
2. *In Section IV, A of the application, the investigator needs to address the following:*
    a. *Include the mesh size of the lift nets.*
    b. *Add a statement explaining separation of predators and prey.*
    c. *Explain what procedure will be followed in case of animal mortality.*
3. *In Section V of the application, the investigator needs to address the following:*
    a. *Change the USDA pain and stress classification from C to D as the study involves capture/trapping.*
    b. *Change the three-year total number of American Shad and Atlantic Herring to 10.*
    c. *Complete the required literature search in section A.*
4. *In the fourth paragraph of section VI, A, the investigator needs to remove the duplicate sentence "however, we may capture significantly more..."*
5. *In Section VI, D, iv of the application, the investigator needs to change the three-year total number of American Shad and Atlantic Herring to 10.*

As soon as the IACUC receives an appropriate response to its concerns, above, it will issue you an approval letter for this protocol. **You may not commence activities in this protocol involving vertebrate animals until you have received IACUC approval**.

If you have any questions, please contact either Dean Elder at 862-4629 or Susan Jalbert at 862-3536.

For the IACUC,

Julie J Simpson

Julie Simpson, Ph.D.
Director

cc: File
McKown, Grant

University *of* New Hampshire

Research Integrity Services, Service Building
51 College Road, Durham, NH 03824-3585
Fax: 603-862-3564

24-Jul-2020

Moore, Gregg E
Dept of Biological Sciences
Jackson Lab
Durham, NH 03824-3406

IACUC #: 190706
Project: Living Shorelines in New Hampshire: Restoration Performance of Floral and Faunal Communities
Modification Approval Date: 24-Jul-2020
Annual Approval Expiration Date: 20-Aug-2020
Protocol Three-Year Approval Expiration Date: 20-Aug-2022

The Institutional Animal Care and Use Committee (IACUC) has reviewed and approved the requested modification to the protocol for this study:

Change the lift nets to minnow traps

If you have any questions, please contact either Dean Elder at 862-4629 or Susan Jalbert at 862-3536.

For the IACUC,

Julie Simpson

Julie Simpson, Ph.D.
Director

cc: File
McKown, Grant

University *of* New Hampshire

Research Integrity Services, Service Building
51 College Road, Durham, NH 03824-3585
Fax: 603-862-3564

18-Sep-2020

Moore, Gregg E
Dept of Biological Sciences

Jackson Lab

Durham, NH 03824-3406

IACUC #: 190706
Project: Living Shorelines in New Hampshire: Restoration Performance of Floral and Faunal Communities

Next Review Date: 20-Aug-2021

The Institutional Animal Care and Use Committee (IACUC) has reviewed and approved your request for a time extension for this protocol. Approval is granted until the "Next Review Date" indicated above. You will be asked to submit a report with regard to the involvement of animals in this study before that date. If your study is still active, you may apply for extension of IACUC approval through this office.

The appropriate use and care of animals in your study is an ongoing process for which you hold primary responsibility. Changes in your protocol must be submitted to the IACUC for review and approval prior to their implementation.

Please Note:
1. All cage, pen, or other animal identification records must include your IACUC # listed above.
2. Use of animals in research and instruction is approved contingent upon participation in the UNH Occupational Health Program for persons handling animals. Participation is mandatory for all principal investigators and their affiliated personnel, employees of the University and students alike. Information about the program, including forms, is available at http://unh.edu/research/occupational-health-program-animal-handlers.

If you have any questions, please contact either Dean Elder at 862-4629 or Susan Jalbert at 862-3536.

For the IACUC,

Julie J Simpson

Julie Simpson, Ph.D.

Director

cc: File

McKown, Grant