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Salt marsh migration into coastal uplands and application for conservation in New Hampshire

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Cover Photos: Marsh migration into the upland at Shackford Point, Newmarket, New Hampshire in 2023.
Executive Summary

Salt marshes across the East Coast, including New Hampshire, have lost considerable areas of vegetation cover from shoreline erosion and interior pool formation as sea level rise outpaces the marsh’s ability to maintain elevation. Marsh migration, or the inland transgression of halophyte vegetation into upland habitats, is one of the main mechanisms for salt marshes to maintain their areal extent on centennial timescales and recently has been a key strategy discussed by resource managers for shorter term timescales. A literature review was conducted to determine the extent of the research on the topic including biogeochemical drivers, response of the vegetation community, impacts of anthropogenic alterations, and potential indicators of appropriate corridors for marsh migration. Research gaps were identified in the literature and future monitoring protocols and avenues of research were outlined with a focus on New Hampshire and New England.

The conversion of upland habitats to coastal wetlands is driven by increases in flooding, salinity intrusion, sulfide concentrations, and a rise in the groundwater table. Marsh migration is driven by both gradual press stressors of sea level rise and pulse stressors of episodic king tides and coastal storms. Press stressors gradually create sufficient flooding and soil conditions to sustain marsh vegetation while pulse stressors shock intolerant plants and transport marsh propagules (e.g., rhizomes and seeds) into the upland. Coastal forests and *Phragmites australis* function as natural barriers to marsh migration. Forests retreated in a stepwise ratchet model over time, where press stressors would kill off understory upland species and tree saplings and periodic pulse stressors would remove stressed mature trees. *Phragmites australis* have been observed to colonize space under opened-up canopies before other marsh vegetation can become established. On the other hand, marsh halophytes appear to migrate gradually into tidal freshwater wetlands and open fields in response to sea level rise, as increased flooding and salinity allow them to outcompete non-halophytes. Relic agricultural embankments and ditches may delay or restrict marsh migration by preventing flooding into the uplands, detaining freshwater, or rapidly draining surface and groundwater after flooding. A field trip to Shackford Point in Newmarket provided a New England case study to marsh migration and supported numerous lessons from the literature review.

Research gaps in the published literature and interviews with practitioners and ecologists demonstrated a lack of long-term field studies, standardized monitoring protocols, active marsh migration facilitation projects, and specifically, research in the Northeast. Organizations have been proactive in identifying suitable marsh corridors for conservation and active management with empirical models. Several pilot migration facilitation projects are still years away from yielding useful lessons.
Key Lessons Learned

- Primary drivers of marsh migration are increased flooding frequency and duration, salinity, sulfide toxicity, and groundwater table. Gradual, long-term press stressors of sea level rise create biogeochemical conditions which provide marsh vegetation with a competitive advantage over upland and freshwater vegetation. Short-term, pulse stressors of coastal storms and king tides stress intolerant plants while transporting halophyte seeds and rhizomes into the upland.

- Thresholds of annual flooding duration (3 – 5%) and salinity (5 – 10 psu) define the extent of the upland – salt marsh border and should be considered in future migration facilitation projects. These thresholds have been found independently across the nation. The 18.6-year lunar metonic cycle can impact the duration and frequency of upland inundation and should be incorporated into marsh migration conceptual models and facilitation projects where significant.

- The rate and trajectory of marsh migration is strongly affected by the composition of the upland vegetation community – coastal forest, *Phragmites australis*, open meadow and field, and upstream tidal wetlands. Coastal forest and *Phragmites australis* act as barriers to migration by shading out marsh graminoids, and migration can only occur once the vegetation community is removed. Marsh migration through open fields and tidal wetlands is less shade limited but still mediated by competition and long-term shifts in flooding and salinity. A mixed community of halophytes and non-halophytes may form in meadows and wetlands for some time.

- Legacy agriculture infrastructure may delay or outright prevent marsh migration from occurring. Perimeter ditches at the upland – marsh border and into the uplands quickly drains the upland after flooding events and reduces the groundwater table. Embankments reduce the flooding frequency and duration in the uplands, sustaining favorable conditions for upland vegetation.

- Salt tolerant shrubs such as *Clethra alnifolia*, *Morella pensylvanica*, *Iva frutescens*, and *Toxicodendron radicans* have been identified as indicators of marsh migration corridors in upland coastal forests.

- Research gaps in the literature and current projects include (1) lack of long-term observation studies and experimental studies on different aspects of marsh migration, (2) lack of studies in New England, (3) consideration of additional factors in identification of suitable migration corridors in the upland, (4) studies documenting possible lag of marsh migration after upland vegetation retreat, and (5) pilot projects of facilitation projects in New England.
Study Approach and Purpose

Great Bay National Estuarine Research Reserve (GBNERR) alongside local stakeholders of the Piscataqua River Estuary Partnership, Maine Coastal Heritage Garden, The Nature Conservancy, and University of New Hampshire formed a working group to study and develop guidelines for planning, implementing, and monitoring salt marsh migration. GBNERR and fellow land management organizations are seeking practical conservation strategies to sustain and restore coastal wetlands in the Great Bay and Hampton – Seabrook Estuaries of New Hampshire, especially with expected losses of salt marshes in the near future. Marsh migration is a new approach to sustain salt marshes into the future.

Our purpose is to synthesize the collective knowledge of migration mechanics and application of this knowledge to strengthen decision making and conservation actions of GBNERR and local stakeholders. The study used three methods to achieve its goal: (1) Summarize published literature; (2) Investigate the ‘grey literature’ of unpublished projects and anecdotal experiences from researchers, land managers, and regulators; and (3) Observe a site actively experiencing coastal forest retreat and upland marsh migration to provide local context. The report is broken down into four broad sections: (1) Background information on salt marsh ecology, (2) Literature review of the current science of marsh migration with a summary table, (3) Case study at Shackford Point on the Lamprey River of Great Bay, and (4) Knowledge gaps and future research needs for the application of marsh migration.

Functions, Services, and Values of Salt Marshes

Salt marshes are emergent, peat-based wetland systems that develop in temperate latitudes within a narrow elevation band between the subtidal and upland (Mitsch and Gosselink 2014). In New England, salt marshes are classified into different vegetation communities along an elevation gradient dictated by tidal inundation and salinity (Bertness 1991a, b, Sperduto and Nichols 2012). The low marsh is flooded with saltwater twice daily and hydrogen sulfide often accumulates in the peaty soils. Emergent vegetation is primarily comprised of *Spartina alterniflora* (Smooth cordgrass) with some *Salicornia* spp. (Sea glasswort) at higher elevations. The high marsh is found above the mean high-water mark and thus is only inundated irregularly, though completely, during the highest spring tides each month and is vegetated by a mixture of graminoids including *S. patens* (Salt marsh hay), *Distichlis spicata* (Spikegrass), and *Juncus gerardii* (Black grass) as well as short-form *S. alterniflora*. In addition to these perennial graminoid species, the high marsh platform sustains numerous forbs like *Agalinis maritima*
(Salt marsh false foxglove), *Lysimachia maritima* (Sea milkwort), *Triglochin maritima* (Seaside arrowgrass), and *Plantago maritima* (Seaside plantain). The upland edge is characterized by minimal flooding during only the largest high tides and coastal storms and soil salinity is reduced with influx of belowground freshwater from the upland. The vegetation community of the upland border is diverse ranging from shrubs of *Iva frutescens* (Marsh elder), graminoids of *Bolboschoenus* spp. (Bulrush), *Elymus repens* (Quackgrass), *Panicum virgatum* (Switch grass), and *Typha latifolia* (Thin leaf cattail), and forbs of *Solidago sempervirens* (Seaside goldenrod), *Aster novi-belgii* (New York aster), *Teucrium canadense* (American germander), and *Calystegia sepium* (Morning glory).

Salt marshes are increasingly valued, protected, and restored for their host of ecosystem services (Barbier et al. 2011). Coastal wetlands in general have been recognized as some of the most valuable ecosystems with Constanza et al. 2014 estimating $267,023 \text{ ha}^{-1} \text{ yr}^{-1}$ (adjusted to 2023). In New Hampshire, coastal wetlands provided roughly $1.5$ billion worth of services across the $5,711$ acres between the Great Bay Estuary, Hampton-Seabrook Estuary, and the coast (PREP 2023). Physically, salt marshes function as adaptive, resilient features that protect upland property and infrastructure and stabilize shorelines (Gedan et al. 2011, Arkema et al. 2013) by attenuating wave energy (Morgan et al. 2009, Shepard et al. 2011), coastal storm flooding (Leonardi et al. 2018), and erosion of uplands. Salt marshes currently protect 263 miles of tidal shoreline in New Hampshire (Gittman et al. 2015). From an estuarine health-perspective, coastal wetlands act as vital sinks for nitrogen, sediment, and carbon to improve local water quality and regulate gas exchange. Anoxic peat soils paired with oxygenated rhizosphere micro-sites are conducive for denitrification of both groundwater and tidal fluxes (Tobias et al. 2001, Pihler and Smyth 2011). The presence of dense marsh vegetation slows water velocity and allows for particles to settle on the surface ranging from $1 – 100 \text{ g m}^{-2} \text{ day}^{-1}$ (Morgan and Short 2002, Morgan et al. 2009). Due to the unique combination of anoxic and waterlogged conditions with high vegetation productivity, coastal wetlands store carbon (Davis et al. 2015) with estimates of stores ranging $65 – 259 \text{ Mg C ha}^{-1}$ within the top meter (Pendelton et al. 2012, Calarusso et al. 2023). The term “blue carbon” has been coined in the past decade to focus more attention on the ability of coastal ecosystems, including salt marshes, to serve as outsized carbon sinks on the landscape (Pendelton et al. 2012). Lastly, coastal wetlands are juvenile nursery hotspots in estuaries by providing abundant resources and protection from predators (Gittman et al. 2016, Bilkovic et al. 2021).
Impacts of Sea Level Rise on Salt Marsh Resilience

Coastal wetlands maintain their intertidal elevation profile through a negative feedback loop that seeks an equilibrium between elevation loss through decomposition and soil compaction and gains driven by plant productivity and sediment accretion (Figure 1; Redfield 1965, Reed 1990). Greater inundation can spur gains in elevation through increased plant productivity (Nyman et al. 2006), slowing decomposition of peat, swelling of peat soils, and additional sediment accretion (Morris et al. 2002). Sediment capture increases with flooding frequency and inundation as finer particles settle from the water column (Mitsch and Gosselink 2014). Plant productivity has been shown to respond positively to increased flooding by adding more volume of roots and rhizomes into the soil (Nyman et al. 2006), however, this productivity relationship may vary across graminoids (Watson et al. 2017) and peaks after a certain threshold of inundation. Lastly, the biogeochemistry of the peat is altered as the decomposition rate of organic matter slows down with increasing waterlogged conditions, reducing the rate of volume loss. Peat swelling has been documented as a secondary factor of temporary elevation increases as well (Chambers et al. 2019). The negative feedback loop has been able to accommodate historic rates of sea level change (1 – 3 mm yr\(^{-1}\)), however sea level rise (SLR) across the northeastern Atlantic seaboard appears to have outpaced the ability of the salt marsh to maintain elevation (Cronin 2012, Raposa et al. 2016, Maher et al. 2023). In the Great Bay, Payne et al. 2019 estimated salt marshes were building elevation at an average of 2.07 mm/yr but were losing relative elevation at rates of 2.1 mm yr\(^{-1}\) or 5 cm in the prior 25 years (SLR = 4.17 mm yr\(^{-1}\)).

**Figure 1.** Environmental drivers and biogeomorphic processes that influence the negative feedback loop of elevation equilibrium in unaltered coastal wetlands. From Cahoon and Guntenspergen 2010.
In essence, SLR has combined with altered hydrology (due to impairments) to shift dynamic processes into alternative feedback loops propagating mudflat formation at the seaward edge and pool formation on the interior high marsh platform (Figure 2, Cahoon and Guntenspergen 2010, Wang and Temmerman 2013). The loss of interior marshes is widespread in the United States since every marsh from Virginia to Maine was mosquito ditched during the 1930’s and nearly all marshes in New England were altered for colonial agricultural practices (Kennis 2001, Gedan et al. 2009, Adamowicz 2020). In one set of interactions, closely spaced ditches lead to peat oxidation, loss of elevation and conversion to low marsh with tall form S. alterniflora. In the more common situation where extensive areas of marsh platform are waterlogged, inundation frequency and duration increase and short-form S. alterniflora outcompetes and replaces high marsh graminoids (Warren et al. 1993, Raposa et al. 2017). Over time, the high marsh retreats landward and may ultimately be squeezed from natural steep slopes or human-made barriers (e.g., seawalls, roads, and homes). Although broad high marsh platforms are limited in Great Bay, the long-term effects of hydrologic manipulation leading to pool formation are evident at Lubberland Creek Marsh, Adams Point, and Chapman’s Landing (Figure 2). The conversion of high marsh habitat to short-form S. alterniflora pannes and shallow permanent pools is widespread throughout the Hampton-Seabrook Estuary.

Figure 2. (Left) Example of shoreline erosion, calving, and collapse at Moody Point Marsh in Newmarket, NH. Photo by Grant McKown (Right) Example of interior pool formation on the marsh platform as a result of embankments and high density ditching at Lubberland Creek Marsh in Newmarket, NH. Aerial imagery from Google Earth.
Marsh Migration – Passive Conservation of Coastal Wetlands

Coastal wetlands shift landward and replace the upland, non-halophyte community as sea level rises, commonly referred to as marsh migration or transgression. Migration into the upland is the main process that allows the areal extent of coastal wetlands to be maintained on centennial time scales (Scheider et al. 2018, Miller et al. 2021). The availability of improved satellite and aerial imagery has shown that marsh migration can offset large losses of marsh; migration even led to a net gain in the past century during rapid SLR in certain estuaries (Smith et al. 2017 – New Jersey, Burns et al. 2021 - Virginia). Additionally, the rate of coastal forest retreat, a relative proxy for marsh migration, may have increased in the past 30 years compared to historic norms of the 19th and 20th centuries in the mid-Atlantic (Kirwan and Schieder 2019). The ability of coastal wetlands to transgress into the uplands has been historically acknowledged yet rarely studied (see Redfield 1965). Recently it has received heightened attention from researchers and coastal managers as a long-term solution for conserving and sustaining the high marsh platform and its ecosystem functions and services. This literature review focuses on five key areas of research that will help guide future project planning, design, and resource allocation for coastal land managers, governmental agencies, and conservation organizations: (1) Hydrologic and geochemical drivers, (2) Upland slope, (3) Response of the upland vegetation community, (4) Impacts of historic agricultural practices, and (5) Leading indicators of marsh migration. Lastly, we present a conceptual model of marsh migration incorporating important hydrologic, geochemical, and vegetative factors.

Hydrologic and Geochemical Drivers of Marsh Migration

The processes of marsh migration and the retreat of upland vegetation communities are primarily driven by SLR as well as greater frequency and intensity of coastal storms. Coastal storms result in greater inundation, raised groundwater table depth, and soil salinization. Landward expansion of halophytic vegetation is facilitated by the stressors of waterlogging, anoxic conditions, and higher concentrations of salinity and hydrogen sulfides. The interplay of inundation, rising groundwater, and salinity is crucial for creating the environmental conditions for both upland plant species retreat and halophyte establishment (Sacatelli et al. 2022). In the short-term, inundation from coastal storms and extreme tides in the marsh border and immediate uplands temporarily increases salinity and anoxia. In-depth field monitoring has observed that the groundwater table in the upland drains and recovers within 2 – 5 days after episodic coastal flooding events, however soil conductivity remains elevated much
longer: 1 – 3 months (Kearney et al. 2019, Nordio et al. 2023). An inundation threshold of 3 – 5% duration over annual timeframes has been observed to convert the upland border ecotone on both west and east coast marshes (Wasson et al. 2013 – Elkhorn Slough, Anisfeld et al. 2017 – Long Island Sound, Walters et al. 2021 – Chesapeake Bay). However, interannual variation in the flooding frequency of the marsh – upland border ecotone can be high. For example, the marsh – upland ecotone was not flooded once for an entire year three times over the course of a 10-year study at the Elkhorn Slough NERR, California, yet several years were punctuated by multiple days of inundation from coastal storms during spring tide cycles (Wasson et al. 2023).

Over time, increased inundation and SLR can raise the groundwater depth of adjacent uplands, supplying deeper aquifers with saline water (Knott et al. 2019). In a year-long study, Nordio and Fagherazzi (2022) monitored the groundwater table, conductivity, flooding, and temperature in a Chesapeake Bay salt marsh – forest system. They found that soil salinity was better explained by longer-term groundwater table depth and evapotranspiration compared to short-term coastal flooding and storm events. Paradoxically, as the groundwater table increases, the surficial layer of freshwater depletes from a combination of flushing into the adjacent marsh (i.e., greater downhill gradient) and evapotranspiration from trees. The removal of freshwater from the root zone creates an upward hydraulic gradient that draws deeper saline groundwater upward, which is supplied from the adjacent intertidal system (Figure 3). On watershed scales, rising groundwater, driven by SLR, is a major component of coastal flooding and waterlogging in the root zone (Rotzoll and Fletcher 2012). The Sea Level Affecting Marsh Model (SLAMM) has been used widely to alert planners how coastal habitats are likely to transgress under various SLR scenarios, using the term ‘saturation’ to describe groundwater facilitation of marsh migration (Clough et al. 2016). In New Hampshire, Knott et al. 2019 modelled groundwater rising 1.2 m within 1 – 2 km of the coastline under a 2 m SLR scenario (Figure 4). Importantly, the process of soil salinization and waterlogging seems to be a gradual process driven by increasing tidal elevations and frequent minor coastal flooding, not by low frequency, high intensity coastal storms (Wasson et al. 2013, Nordio et al. 2023), however direct observational studies of the interactions of flooding, groundwater, and vegetation in the upland or at the immediate upland – marsh interface remain sparse.
Figure 3. Modelled groundwater table rise at least 1m within 1 – 2 km of the coast of New Hampshire under a 2 m SLR scenario. From Knott et al. 2019.

Figure 4. (Left) Conceptual diagram of hydraulic gradients at the marsh – upland border. (Right) Proposed mechanism for transport of salinity to the root zone of upland forests driven by downward gradients of freshwater coupled with evapotranspiration. From Nordio and Fagherazzi et al. 2022.
One generally overlooked aspect of SLR that will impact future rates of marsh migration is the 18.6-year lunar metonic cycle (Chambers et al. 2003). In the Gulf of Maine, the metonic cycle results in an increase of 20 – 25 cm of the tidal amplitude within a 9.3-year timeframe (Figure 5), further exacerbating the impacts of SLR (Oost et al. 1993, Peng et al. 2019). The lowest tidal ranges will occur in 2024 – 25 and then increase through 2033 – 34. The increase in high tides of 10 - 12 cm elevation will stimulate the main drivers of marsh migration including coastal flooding events, groundwater table depth, and salinity intrusion in the upland. Long-term observations of natural marsh migration or evaluation of facilitation projects need to be viewed within a metonic cycle context.

**Figure 5.** Sea level rise (orange) and change in tide amplitude from 18.6-year metonic cycle (blue) of the Gulf of Maine 1920 – 2022 based on the Portland, Maine tidal gauge (NOAA 8418150). Y axis is either mean sea level (m) for top regression (orange points) or the difference in mean high water and mean low water (m) for the bottom regression (blue points).

**Slope & Artificial Barriers**

Obvious barriers to landward marsh migration are the topographic slope from the marsh platform to the upland (Kirwan and Gedan 2019) and constructed barriers such as seawalls, tidal restrictions, dams, and upland development (Enwright and Osland 2016). Tidal hydrology and slope were originally the main environmental factors in numerical models of marsh migration (Enwright et al. 2016, Kirwan et al. 2016), and are still major factors in more complex modelling or remote sensing studies (Smith et
Slope influences numerous mechanisms of migration including the exposure of upland vegetation to flooding and salinity (Brinson et al. 1995, Kearney et al. 2019), expansion of halophytes from the marsh platform, and groundwater hydrology (Anisfeld et al. 2017). Gentler slopes enhance upland vegetation retreat through greater coastal flooding inundation and salinity intrusion and reduced drainage (i.e., waterlogged soils) while simultaneously providing easier access for vegetative expansion and seed dispersal of marsh halophytes. Flester and Blum 2020 observed that marshes with the lowest slopes in Virginia experienced both the greatest rates of seaward erosion and upland migration. Conversely, steep slopes have also been found to enhance salt marsh migration in limited cases. Smith 2013 observed through historic aerial imagery that Phragmites australis reduced marsh migration on gentler slopes and marsh migration was more successful on steeper slopes. Smith 2013 hypothesized that Phragmites was squeezed between overly dry conditions in the upland vs. high salinity conditions seaward.

Anthropogenic barriers to marsh migration can take numerous shapes and sizes from physical barriers like seawalls and upland development to tidal restrictions such as undersized culverts and dams. Seawalls and upland development create steep, vertical slopes at the upland edge preventing vegetative spread of marsh graminoids and reducing flooding in the upland. Over time, salt marshes may be ‘squeezed’ by sea level rise at the seaward edge and hard barriers at the upland edge (Pontee 2013). Enwright and Osland (2016) estimated that roughly 1500 km$^2$ of potential marsh migration would be prevented by urban development and infrastructure in a 1.2 m SLR scenario throughout the Gulf of Mexico. Inventory of shoreline structures and upland infrastructure will be key for identifying suitable marsh migration corridors (see Torio and Chmura 2013 – Maine, Blondin 2014 – New Hampshire). Second, restrictions to proper tidal flow through undersized culverts and dams reduce or outright reduce flooding, salinity intrusion, and groundwater table rise in the upland. Additionally, tidal restrictions may suitable conditions for invasion of Phragmites australis, adding an additional obstacle to marsh migration (see ‘Response of Upland Vegetation Community’ section). Assessments of tidal crossings should be conducted to better understand their impact on ecosystem functions and marsh migration. For example, the Nature Conservancy and NH Department of Environmental Services completed a comprehensive assessment on the tidal crossings in the state including structural integrity, tidal impact, and inundation risk (Steckler et al. 2017, TNC and NHDES 2019).
Response of Upland Vegetation Community

High marsh graminoids can spread landward through a combination of vegetative rhizome expansion and reproductive seed dispersal in wrack (Michinton 2006), ice rafting (Rabinowitz et al. 2022), and surface water during coastal flooding events (Zhu et al. 2022). For example, Kottler and Gedan (2020) documented *S. patens* and *D. spicata* in a forest seedbank 15 m landward of a Chesapeake Bay marsh. Dowling et al. (2023) identified unique ecotypes of *S. patens* that were more productive in upland soils, compared to peat, highlighting the specie’s ability to drive inland migration. The upland vegetation community has been shown to be a significant factor in the rate and ability of the marsh platform to shift landward, with greater influences on migration rates than slope in at least one study (Schieder et al. 2018). For example, dense overstory canopies of forests and *Phragmites* stands reduce light-availability and subsequent fitness of high marsh graminoids (e.g., *S. patens* and *J. gerardii*) and function essentially as barriers to marsh migration (Brinson et al. 1995, Smith et al. 2013, Field et al. 2016, Kottler and Gedan 2022). The marsh platform has four common types of possible migration corridors: coastal forest, *Phragmites* stands, fields and meadows, and freshwater or brackish tidal wetlands, each with its unique response to SLR and associated pathway for marsh migration.

A large amount of effort has been directed towards understanding the responses of coastal forests at the marsh – upland interface and the underlying processes of ‘ghost forest’ formation (Kirwan and Gedan 2019). Ghost forests are former upland forests which have succumbed to salinity and flooding stress and are distinctly characterized by stands of dead trees and snags. Upland forests may be the most likely option for marsh migration corridors on public and conservation land compared to dense urban shorelines. The bulk of research on forest retreat has been primarily conducted in the mid-Atlantic (North Carolina, Chesapeake Bay, and New Jersey) due to the relative low slope (1 – 2%) across the marsh – upland interface and large ecosystem manipulation experiments (see Walters et al. 2021, Hopple et al. 2023). Forest retreat has been well-documented with remote sensing and use of historic maps and recent aerial imagery across the Mid-Atlantic and Northeast (Smith 2013 & Smith et al. 2017 – New Jersey, Burns et al. 2021 – Virginia, Ury et al. 2021 – North Carolina, Powell et al. 2022 – Delaware). Recent estimates of lateral forest retreat in the Mid-Atlantic have accelerated from historical averages 0.49 m yr⁻¹ to rates ranging from 1.7 to 4.7 m yr⁻¹ (Schieder and Kirwan 2018 – North Carolina to Delaware, Chen and Kirwan 2022 – Virginia).

Originally, many researchers assumed that coastal forest retreat was relatively linear once tidal flooding reached elevation thresholds based on assumptions in numerical (Enwright et al. 2016, Kirwan
et al. 2016), and empirical models (SLAMM). This led to the innate design of monitoring over long-periods in remote sensing studies (see last paragraph). Coastal forest retreat is generally characterized as a gradual process: (1) tree distress and reduced annual growth from flooding and salinity, (2) death of young trees and inability for continued recruitment of saplings, (3) demographics of the tree population skew older over time, (4) ghost forests form as mature trees eventually succumb to stressors of SLR, and (5) canopy opens for the colonization of halophyte species (Brinson et al. 1995, Kirwan and Gedan 2019). Recent research, however, has demonstrated that mature stands of coastal forests are resistant to stressors of inundation and salinity from increased tidal flooding through adaptations (Field et al. 2016). For example, *Juniperus virginiana* (Red cedar), a common coastal forest tree in the Chesapeake region, distributes its roots asymmetrically at higher densities and volumes to avoid salinity toxicity and capture freshwater (Messerschmidt et al. 2021). Marsh migration, therefore, is more likely controlled by a complex set of species-specific interactions and mechanisms rather than a gradual and linear process as originally thought.

The transgression of the forest–marsh ecotone is best described as a non-linear or stepwise ratchet model punctuated by ‘press’ and ‘pulse’ disturbances (Figure 6, Fagherazzi et al. 2019, Sacatelli et al. 2023). Forests can be divided into a zone of regeneration, landward mature forest where seedling and sapling recruitment continues, and a zone of persistence, seaward mature forest demarking the forest–marsh ecotone where seedling recruitment has ceased. The seaward boundary for the regeneration zone is primarily influenced by the pressor disturbance of SLR in the form of increasing salinity and shallowing groundwater table, because germination of seedlings of coastal forest tree species are highly susceptible to flooding, waterlogged soils, and salinity (Kozlowski 1997). Salinity of 5 - 10 ppt (Tolliver et al. 1997, Woods et al. 2020) and flooding duration of 3 – 5% have been identified as potential thresholds for significantly reduced seedling germination in common mid-Atlantic species (Walters et al. 2021). Additionally, recruitment often occurs in short durations of 1 – 2 consecutive years across decadal or centennial timespans during lower astronomical tidal cycles (Williams et al. 1999, Kirwan et al. 2007) or immediately after events that open the canopy like intense coastal storms or logging. In an experimental logging study to accelerate marsh migration, Walters et al. (2021) observed that unless flooding and salinity thresholds are reached, regeneration of the forest will reverse any marsh landward gains. The gradual rise of the groundwater table from SLR and persistent saline soil conditions after coastal flooding events create untenable conditions for young saplings and seedlings and permanently close the already narrow recruitment window (Williams et al. 1999, Kearney et al. 2019).
The persistence zone is a non-resilient width of adjacent forest which maintains its seaward boundary despite numerous abiotic stressors tied to SLR. Mature trees of coastal forests are more capable of managing hydraulic failure and carbon starvation from salinity and anoxic soils than younger trees (McDowell et al. 2022). However, trees become increasingly osmotically stressed from the lack of freshwater or high groundwater table (Desantis et al. 2007, Haaf et al. 2021) and subsequently reduce growth and lose sections of the canopy (Hall et al. 2022). Remote sensing evaluations in the Chesapeake Bay revealed that the persistence zone of coastal forests had significantly lower photosynthetic activity and did not recover from large coastal storms compared to the regeneration zone which were less stressed and highly resilient (Fagherazzi et al. 2019b, Chen and Kirwan 2022). The worsening health of individual trees or entire stands can be compounded by the additional, regional stressors of drought and pest invasions (Desantis et al. 2007, Taillie et al. 2019, Ury et al. 2021). Individual trees become less resilient and are lost to low frequency, high intensity disturbances of hurricanes and wildfires (Rouchibaud and Begin 1997, Williams et al. 1999, Ury et al. 2021). The coastal forest seaward edge may last for several decades longer than predicted based on just SLR (Chen and Kirwan 2023). Over time, the seaward boundary of the forest retreats in a stepwise fashion over decadal timescales punctuated primarily by coastal storms (Figure 6, Fagherazzi et al. 2019a, Miller et al. 2021).
**Phragmites** is a ubiquitous invasive species in coastal wetland settings in the Northeast and the subject of intense management activities and research (Hazleton et al. 2014). From a marsh migration perspective, **Phragmites** can colonize and takeover land after forest retreat but before the marsh can establish itself (Anisfield et al. 2017, Langston et al. 2022). Smith (2013) observed that **Phragmites** displaced over 60% of the available area deforested from retreat in the Delaware Bay, New Jersey, leaving roughly 30% for upland marsh migration. **Phragmites** commonly occupies the upland – marsh ecotone and upper elevations of the high marsh platform due to its flooding and salinity tolerance. **Phragmites** can establish itself under low-light conditions of dense canopies of upland forests receiving tidal and storm flooding, prior to forest retreat (Figure 7, Shaw et al. 2022, Sward et al. 2023). Given that this invader can induce positive-feedback loops to reduce salinity and flooding stress (Burdick and Konisky 2003, Reijers et al. 2019) and build elevation capital relative to sea-level rise (Rooth et al. 2003), marsh migration may proceed non-linearly with its presence. **Phragmites** transgression may only occur until certain flooding, salinity, or sulfide thresholds are reached (Chambers et al. 2002, Buchsbaum et al. 2006, Sun et al. 2007). In addition to embankments, lower tidal amplitudes during the troughs in the 18.6-year metonic cycle and heavy wrack deposition may provide windows of opportunity for **Phragmites** to colonize, establish, and spread in the marsh – upland border (Warren et al. 2001, Minchinton 2002, Chambers et al. 2003).

**Figure 7.** Relationships of *Phragmites australis* density to environmental factors observed along transects in the Chesapeake Bay (Maryland). Phragmites Density Index Value is a binned density measurement ranging 0 – 4 where 0 represents absence and 4 represents a dense, homogenous stand. From Shaw et al. 2011.
Although understudied, lawns, open meadows, and agricultural land adjacent to marshes may serve as vital migration corridors to promote marsh migration more quickly than coastal forests and *Phragmites* stands. Remote sensing studies have shown that marsh losses are more likely to be replaced through migration into agricultural land and open fields rather than coastal forests (Gedan et al. 2020, Osland et al. 2022). In urban regions, parks and residential lawns may be the only option for sustaining marsh platform area (Meixler et al. 2020). The open canopy and gentler slopes of these shorelines better expedite the establishment of high marsh graminoid species. In 2022, Anisfield and colleagues found the leading halophyte vegetation at the upland – marsh ecotone varied significantly with *P. australis* and shrubs of *Iva frutescens* and *Baccharis halimifolia* along coastal forests whereas high marsh graminoids were found invading residential lawns. The adjacent lawns were more conducive for marsh platform development with greater soil organic matter, water content, and porewater salinity than the immediate edge of coastal forests. The lack of a dense, shaded canopy shifts the organizing mechanism of marsh migration to competition between halophyte and non-halophytes. The transition of meadows and agricultural fields to halophyte-dominated systems is more gradual and linear than for coastal forests and *Phragmites* stands. Salinity intrusion and flooding may create temporary novel vegetation communities that contain a mixture of high marsh and salt marsh edge halophytes and salt-tolerant upland graminoids and forbs (Figure 8, Gedan and Fernández-Pascual 2018) as opposed to a relatively uniform landward shift of the marsh – upland ecotone boundary (see Wasson et al. 2013). Over time, as SLR continues, the vegetation community should shift to one closely resembling the high marsh platform as less salt tolerant forbs and graminoids are outcompeted and wetland soils develop (e.g., increased organic matter, anoxia, and sulfide concentrations; Craft et al. 1999, 2002).
Figure 8. Non-metric dimensional analysis of the vegetation communities between the salt marsh ecotone, old agricultural field, and the edge and interior of the salinized, transitional field. Axis values are arbitrary and represent the similarity and dissimilarity between points based on species composition and abundance. From Gedan and Fernández-Pascual, 2018. Plant species are color coded and serve as representative species for each ecosystem. Note the salinized field is in a transitional state between the salt marsh ecotone and the upland agricultural field.

Tidal creeks and upstream tidal freshwater wetlands may serve as longitudinal conduits for marsh migration in addition to lateral shifts upland (Craft 2009), though few studies have been devoted to documenting vegetation shifts from fresh to brackish or saline (Neubauer and Craft 2009). Freshwater wetlands may account for up to two-thirds of salt marsh migration in the conterminous United States, although most such transitions will be concentrated in the Gulf of Mexico and mid-Atlantic (Osland et al. 2023). Long-term increases in tidal inundation, salinization, and root zone waterlogging are the main drivers replacing fresh with salt marsh (Neubauer and Craft 2009). Tidal freshwater and oligohaline riparian wetlands may not build elevation quickly enough in response to SLR (Jarrell et al. 2016), and plant species are not adapted to mitigate long-term increases in anoxia, salinity, and sulfide toxicity (Herbert et al. 2015). Reduced biomass and increased respiration, both signs of stress, begin to materialize at salinities as low as 4 psu for common fresh forbs and graminoids (Sutter et al. 2014). Additionally, S. alterniflora becomes an effective competitor against common freshwater forbs like Peltandra virginica (Green arrow arum) in salinities as low as 1 psu (Sutter et al. 2015). Shifts of vegetation communities upstream in estuaries have been documented on decadal timescales (Schuyler et
al. 1993, Perry 1999, Humphreys et al. 2021). Mesocosm studies have highlighted that species composition (e.g., loss of salt intolerant species) may shift from short pulses of salinity, yet large-scale vegetation community shifts from tidal freshwater to salt marsh are driven by consistently high durations of increased salinity and inundation (Flynn et al. 1995, Li and Pennings 2019, Li et al. 2021).

Interestingly, Mobilian et al. (2023) observed similar results with the soil microbiome as species diversity shifts only took place with continuous inputs of salinity to tidal freshwater soils.

Creeks in the Great Bay and Hampton – Seabrook Estuaries provide numerous opportunities for longitudinal marsh migration into oligohaline and freshwater wetlands. However, fresh and brackish tidal wetlands are some of the rarest habitats in New Hampshire with several rare species like Iris prismatica (Slender blue flag iris), Lilaeopsis chinensis (Eastern grasswort), and Samolus valerandi (Seaside brookweed) which could be threatened with marsh migration (see ‘High brackish riverbank marsh’ and ‘Low brackish riverbank marsh’ communities in Sperduto and Nichols 2012). Steep riparian slopes common to New Hampshire, adjacent upland forests, or invasion of Phragmites may prevent upland migration of brackish species and tidal freshwater wetlands in general.

**Impacts of Past Agricultural Practices on Hydrology and Vegetation**

The widespread agricultural use of salt marshes in the mid-Atlantic and New England and their lasting impacts on tidal hydrology and soil biogeochemistry has not been incorporated into conceptual frameworks or research on marsh migrations specifically. Although the impacts of abandoned agricultural infrastructure at the marsh – upland edge have been rarely investigated, lessons from research on the marsh platform suggest that they may prevent or delay the rate of marsh migration into the upland by ameliorating or serving as refuge from stressful conditions for upland species.

Embankments reduce the frequency and magnitude of tidal flooding landward (Mora and Burdick 2013), preventing fresh water from draining seaward but also delaying development of waterlogged and saline soils in the upland. Embankments may also serve as island refuges from flooding, salinity, and sulfides for upland species and Phragmites australis (Figure 9a-b). One possible explanation for the Phragmites found widely throughout the U.S. at upland edges could be that common reed is taking advantage of less-stressful conditions on perimeter embankments (Bart and Hartman 2000, Philipp and Field 2005). The results of Smith et al. (2013) could be partially explained by the presence of embankments as New Jersey marshes were heavily used for agriculture (Adamowicz et al. 2020, 2021). Ditching in the marsh platform has been shown to increase drainage and oxidation of the surrounding peat soils (Vincent et al.
19, Burdick et al. 2019). Farmers established perimeter ditches at the marsh – upland border to provide drainage after king tides and coastal storms, reducing stressor severity and duration on upland species. Following abandonment in the 19th Century, the highest elevation marsh platforms shifted to red maple or Atlantic white cedar swamps (personal observations; Figure 9c-d). Understanding physical and historical site contexts should inform management of marsh migration at the local level.

![Figure 9a](image1.png) ![Figure 9b](image2.png) ![Figure 9c](image3.png) ![Figure 9d](image4.png)

**Figure 9.** Examples of the impacts of legacy agricultural infrastructure on the marsh platform and upland border. (a) *Phragmites* colonization on an embankment at Shackford Point impeding marsh migration, (b) Embankment restricting drainage of excess tidal waters and creating pools immediately adjacent to the upland at Fairhill Marsh at Odiorne Point, (c) Perimeter ditching surrounding a possible abandoned field which has succeeded into a red maple swamp at Fairhill Marsh, and (d) Example of wide perimeter ditch in the western corner of photograph (c).

**Leading Indicators of Marsh Migration**

Several vegetation and abiotic indicators of marsh migration have been identified through observational field studies including hydrology and soil salinity thresholds, understory species retreat, and facilitation of salt-tolerant shrubs and graminoids. Identification of indicators in the field could predict suitable marsh migration corridors and inform future land management activities. Hydrology and salinity are most likely the most critical indicators of future marsh migration with estimates of 3 – 5%
annual inundation and 5 – 10 psu of salinity driving changes in plant species composition (see ‘Response of Upland Vegetation Community’ section). The retreat of herbaceous species and woody saplings and seedlings in the understory has been identified as one of the first steps to forest retreat in response to flooding and salinity increases (Brinson et al. 1995, Kirwan and Gedan 2019). In the rachet model of coastal forest retreat, the persistence zone bordering the marsh is defined by the lack of tree seedlings and saplings. Working in the mid-Atlantic, Anderson and colleagues (2022) identified key soil salt concentration thresholds for indicator understory forbs and shrubs representative of stable, non-salinized forests including *Mitchella reprens* (Partridgeberry), *Ilex opaca* (American holly), *Symplocos tinctoria* (Sweetleaf), and *Clethra arborea* (Lily-of-the-valley tree, Figure 10).

Increasing dominance of certain salt tolerant shrubs at the edges of coastal forests has also been identified as potential leading indicators of marsh migration. In the coastal plain of the Southeast, remote sensing studies have observed coastal forests routinely convert into freshwater scrub – shrub systems as an intermediate state before shifting to salt marshes or open water (Ury et al. 2021, White et al. 2022). Salt-tolerant shrubs may dominate further into the forest canopy than *Phragmites* after initial introduction of coastal flooding. Sward et al. (2023) observed *Morella cerifera* (Southern wax myrtle), a low salinity tolerant shrub (Woods et al. 2020), colonize and dominate densely shaded forests with porewater salinities of 1 – 10 psu. *Iva frutescens* (Marsh elder), a common shrub in the marsh – upland border, has been observed at moderate flooding and salinity levels in the Southeast and is typically one of the first species representative of the marsh edge to colonize the forest edge and understory (Langston et al. 2017, Anderson et al. 2022). In addition to missing tree seedlings and saplings, presence of salt-tolerant shrubs may be a sign the area of coastal forest transitioned to the state of persistence zone. In New England, shrub and woody indicators of salt intrusion and inundation include *Baccharis halimifolia* (Groundsel tree), *Clethra alnifolia* (Coastal sweet-pepperbush), *I. frutescens*, *M. pensylvanica* (Northern bayberry), *Myrica gale* (Sweet gale), and *Toxicodendron radicans* (Poison ivy).
Figure 10. Indicator species of different states of the transition of coastal forest to salt marsh in Virginia from Anderson et al. 2022. Indicator species were selected based on presence – absence and abundance from salinity concentration thresholds. Species abbreviations are as follows for Forest: MIRE – Mitchella repens, CLAL – Clethra alnifolia, STYI – Symlocos tinctoria, ILOP – Ilex opaca, LITU – Liriodendron tulipifera, and ASTR – Asimina triloba; Transition: LYLU – Lyonia lucida, WOVI – Woodwardia virginica, MOCE – Morella cerifera, PHAU – Phragmites australis, IVFR – Iva frutescens, TORA – Toxicodendron radicans; and, Salt marsh: SCAM – Schoenoplectus americanus, DISP – Distichlis spicata, CLJA – Cladium jamaicense, and SPAL – Spartina alterniflora.

Conceptual Model of Marsh Migration

A conceptual model was created to synthesize current knowledge and contrast marsh migration between the upland vegetation communities as well as differentiate the impacts between pulse and pressor stressors. The organizing mechanism and pathway for marsh migration can be divided between (1) upland forest and Phragmites stands and (2) open meadows and tidal freshwater wetlands. Conceptually, marsh migration into meadows and tidal freshwater wetlands should be viewed as the marsh invading and competing with upland species, whereas migration should be taken from the viewpoint of the upland community retreating from stressful conditions for coastal forests and Phragmites stands. Due to their prevalence on the landscape, agricultural embankments and ditches should be included into conceptual models of marsh migration due to their outsized influence on tidal hydrology and vegetation community composition.

Coastal forest and Phragmites innately function as barriers to marsh migration through their dense overstory canopy, preventing vegetative or reproductive expansion of marsh graminoids. The limiting factor for migration is the availability of adequate sunlight (Kottler and Gedan 2022), despite upland areas having sufficient flooding and soil salinity for marsh vegetation. The marsh – upland border is well-defined and predominantly controlled by episodic or pulse events such as coastal storms or pest infestations. The pathway for migration resembles a stepwise function, instead of a gradual linear progression, where the marsh – upland border remains stagnant for long periods of time and moves inland in short bursts (Figure 11). In the case of Phragmites, as the forest retreats after intense vents, the
giant reed can quickly colonize and occupy a portion of open space before the marsh platform has time to transgress inland. The area for marsh migration is reduced and more time will be required for the original forested area to become a marsh platform. For both habitats, marsh vegetation is able to quickly establish after dense canopy shading is loss, since press stressors of SLR and salinity removed understory upland vegetation. The marsh vegetation faces little competition in newly opened areas.

For freshwater tidal wetlands and open fields, the main organizing mechanism of the vegetation community is competition between upland and halophyte graminoids and forbs. As waterlogging of soils, salinity intrusion, and sulfide toxicity increase from SLR, halophytes gradually outcompete intolerant freshwater species. Although pulse events may not be the stressor that controls the rate of marsh migration, episodic events of coastal storms distribute halophyte seeds and rhizome material for establishment and remove stress-intolerant species. The marsh – upland border remains less defined, compared to coastal forests, as the vegetation community is a heterogenous composition of halophytes and upland species (Figure 11). The marsh – upland border gradually retreats inland in relation to the rate of SLR.

The role of legacy agriculture embankments and ditches, especially at the upland border, should be incorporated into models of marsh migration dynamics. The presence of embankments and ditches at the marsh – upland border may substantially delay the transition of the upland vegetation community (Figure 12). Long-term press stressors of tidal inundation, waterlogging, and salinity are blocked or ameliorated as embankments restrict tidal inundation into the upland and ditches remove excess tidal and groundwater. Large coastal storms, or short-term pulse stressors, and accompanying tidal surges may be mitigated by agricultural infrastructure on the landscape as the initial intensity is reduced by embankments and stressful conditions quickly relieved by perimeter ditching.
Figure 1. Conceptual diagram of the potential pathways for the transition of the upland vegetation community to the salt marsh. (Top Row) Migration into freshwater tidal wetlands and open meadows can be defined as a gradual process of competition by halophyte graminoids driven by long-term, pressor stressors of sea level rise and rising groundwater table. From a landscape-scale, the transition between vegetation communities may be clear cut, however, at site scales, ongoing competition creates heterogeneous vegetation communities and nebulous borders of the marsh and upland. (Bottom Row) The movement of the well-defined marsh – upland border in coastal forests and Phragmites stands is controlled by episodic pulse events of coastal storms that forces the upland vegetation community to retreat after long-term stressful conditions. Marsh vegetation quickly transgresses inland after loss of canopy shading as press stressors of sea level rise removed understory plants.

Figure 12. Conceptual diagram of the potential pathways for the retreat of an upland forest with the presence of agricultural embankments and ditches at the marsh – upland border. Restricted tidal flow to the upland from embankments and drainage of excess tidal flow and groundwater from perimeter ditches reduces salinity and waterlogging stressors on the upland vegetation community and delays migration of the marsh.
Case Study of Marsh Migration – Shackford Point, Newmarket

To observe marsh migration occurring in the Great Bay Estuary, we reviewed aerial photography and selected Shackford Point (Newmarket, New Hampshire) in August 2023 as an example case study. Marsh migration has been documented at other sites in Great Bay, and Shackford Point was selected based on site access and permission and large area of ghost forest currently present. At Shackford Point forested uplands are bordered by a fringe marsh with a well-defined low marsh, high marsh, and upland edge that is adjacent to a gentle sloping ecotone with mixed forest and small patches of Phragmites. Tree mortality was first observed between 2010 – 2012 in aerial imagery along the edges of the salt marsh and has continued inland (Figure 1). The rate of tree mortality has followed the rachet model of episodic declines punctuated by large tree mortality every few years (Figure 13). It is estimated from aerial imagery and delineation of the vegetation community that roughly 1.2 acres of coastal forest has been replaced by salt marsh border, or upland edge, (1.1 acre) and Phragmites (0.1 acre) since 2010 (Figure 14).

Along the perimeter of Shackford Point is a raised earthen berm (~20 – 50 cm height) which once protected the upland from coastal flooding. Tidal flooding has been able to enter the forest through two breaks in the berm, west and south of the point, and at the northern point where the surface expression of the berm is muted. Where the berm is breached, tidal flooding travels through unvegetated swales that terminate in small depressional pools. On the site visit one was bare and dry and the other had standing water. On the southern end, the salt marsh border was primarily composed of Spartina pectinata (Rough cordgrass) with a small mixture of Phalaris arundinacea (Canary reed grass) and Teucrium canadense (American germander). On the northern end, the berm was dominated by Myrica gale shrubs (Sweet gale) and several stressed Juniperus virginiana (Eastern red cedar) and Quercus sp. (Oak) trees. The vegetation of the ghost forest was primarily composed of Spartina pectinata and Typha angustifolia (Thin-leaved cattail) with a minor mixture of Carex scoparia (Pointed broom sedge), Solidago sempervirens (Seaside goldenrod), Toxicodendron radicans (Poison ivy), and Teucrium canadense (Figure 15). The salt-tolerant vegetation community within the ghost forest is reminiscent of seasonally flooded marsh – upland border, suggesting the area is receiving multiple flooding events each year. The Phragmites patch at the northern point was contained on the landward edge and showed no evidence of vegetative expansion based on the presence of only a few short shoots.

Numerous standing dead trees were found throughout the site with several heavily colonized by Smilax sp. (Greenbrier). Few knocked down trees were observed suggesting recent tree mortality, most
likely caused by salinity, sulfide, and inundation stress as opposed to pest infestation or fire. We were only able to find several *Quercus* spp. near the ghost forest – ‘non-impacted’ forest border. A distinct border between the ghost forest and the non-impacted upland forest is highlighted by a dense overstory canopy of trees and understory canopy of *Clethra alnifolia* (Coastal sweet-pepperbush), *Osmunda cinnamomeum* (Cinnamon fern), *Pteridium aquilinum* (Bracken fern), and *Smilax* sp. Several *C. alnifolia* individuals were found in the salt marsh border zone on higher ground. The dense understory at the edge of the ghost forest is a function of the edge effect, where increased sunlight sustains greater biomass and species richness.

The site visit taught several key lessons about salt marsh migration in Great Bay and applicability of the conceptual models from the literature review. First, on gentler slopes or areas without barriers to tidal flow, large swaths of coastal forest can die off rather quickly (< 10 years) and be colonized by salt tolerant graminoids of either the salt marsh border or high marsh. The coastal forest at Shackford Point retreated in steps relatively quickly (< 6 years) based on Google Earth imagery, supporting the rachet model for forest retreat. Second, the presence of historic embankments on the property most likely delayed the transition from coastal forest to marsh border by preventing proper tidal inundation, supporting the conceptual model. The breaking of the embankment could have functioned like an episodic event, similar to large coastal storms in the mid-Atlantic, with large-scale flooding to the forest and subsequent ponding of saline water. The embankment most likely prevents consistent tidal exchange, so only king tides and coastal storm events reached the now ghost forest. Third, *Phragmites* was primarily only found on or immediately adjacent to the embankment, which points to agricultural features being a possible driver of *Phragmites* invasion. It remains to be seen if the *Phragmites* will spread from the embankments into the ghost forest as previous research and the conceptual model suggests. Fourth, there may be a considerable lag between ghost forest formation and establishment of high marsh graminoids. We did not observe any dominant high marsh graminoids (*Spartina patens, Distichlis spicata, and Juncus gerardii*) within the entire salt marsh border zone. Shackford Point most likely requires more regular flooding before conversion of *S. pectinata* and *T. angustifolia* to a well-developed high marsh platform. The conceptual model and research would suggest that those high marsh graminoids would migrate quickly into the open areas. However, regular flooding and insufficient soil conditions, driven by the presence of the embankment, may be lacking. Fifth, *M. gale* and *C. alnifolia* served as leading indicators of marsh migration (see Anderson et al. 2021). *M. gale* was found all along the western berm which experiences salt spray and inundation from king tides. *C. alnifolia* served as an
indicator of canopy opening and soils still supportive of non-salt tolerant plants (reduced salinity, waterlogging, and flooding duration) based on its dense wall at the landward edge of *S. pectinata* and scattered individuals on higher ground within the marsh border area. The lack of salt tolerant shrubs throughout most of the ghost forest highlights the lack of competition marsh vegetation face after the overstory canopy retreats.

**Figure 13.** Formation of a ghost forest at Shaddock Point at the mouth of the Lamprey River, Newmarket. White dashed lines delineate areas of ghost forest in the area. Tree die-off is apparent in 2014, however a clear delineation is not apparent. Timelapse imagery from Google Earth.

**Figure 14.** Map of ghost forest retreat at Shackford Point in Newmarket, New Hampshire with the vegetation community manually delineated from aerial imagery and site survey. The salt marsh border is primarily composed of *Spartina pectinata, Typha angustifolia,* and *Carex scoparia* with minor presence of *Teucrium canadense, Myrica gale,* *Toxicodendron radicans,* and *Smilax* spp.
Figure 15. Vegetation community of Shackford Point (shown in Figure 12) shown from a progression from the salt marsh border – high marsh edge to the marsh border – intact forest border. (a) Marsh border – high marsh edge just south of the northern point with the *Phragmites* patch in the background. Several pools at the ecotone were observed most likely from repeated wrack deposition surrounded by *Schoenoplectus robustus*, (b) 5 – 10 m into the ghost forest facing north (*Phragmites* to the left) highlighting the expanse of *S. pectinata* and containment of *Phragmites*, (c) Center of the ghost forest facing north highlighting several stressed trees and continued expanse of *S. pectinata*, and (d) marsh border – healthy forest border highlighting with a wall of *C. alnifolia* shrubs.
Gaps in Knowledge and Research Needs

Research Gap 1: Pilot Studies to Guide Future Projects

One of the main goals for the literature review was to document successful application of marsh migration projects and highlight potential pitfalls or sources of failure. Documentation of the responses to experimental pilots would also yield insight to useful metrics for monitoring protocols and adaptive management. However, almost no projects have been reported in the published literature (but see Walters et al. 2019) and only a few projects in the grey literature were far enough along to hold any lessons for practitioners (Kenny Raposa and Wenley Ferguson). Important questions remain about the utility and efficacy of different approaches to promote marsh migration such as halophyte plantings, *Phragmites* removal, grading along the upland ecotone, removal of upland vegetation, and change to mowing regimes. Tidal dam removal or culvert replacement projects may provide clues to the response of biogeochemical factors and the vegetation community, however their interpretation for marsh migration is context limited. Removal of barriers to tidal flow may function as a marsh migration project on an expedited timeline (*i.e.*, water elevations will significantly increase in a short period of time). Numerous research questions remain about the implementation, success rate, and influencing factors on active or passive facilitation marsh migration projects.

Research Gap 2: Experimental and field monitoring studies

Marsh migration literature has been dominated by remote sensing analyses, numerical models, and field studies on the upland vegetation community. There are few field studies specifically focused on hydrologic drivers, soil biogeochemistry, and changes at the marsh border. Although several papers have pointed to coastal flooding, salinization, and groundwater rise as mechanisms driving migration, more research is needed to reach useful conclusions across variable geography and topography. Mechanisms, rates, and species-specific considerations of habitat retreat have been largely determined through a series of field surveys and greenhouse experiments. Numerical models and remote sensing studies provide a useful foundation for understanding marsh migration rates, however, field studies documented underlying drivers of marsh migration or the habitat-specific retreat patterns. For example, the non-linear retreat of upland forests over time was largely found through a series of field hydrology and vegetation studies. Development of monitoring protocols and establishment of permanent plots and long-term monitoring of the hydrology, soil biogeochemistry, and vegetation community would benefit the subject greatly.
Observational and experimental trials centered on the marsh platform remain sparse in peer-reviewed literature leaving pilot projects without sufficient knowledge on project design, technique efficacy, and adaptive management needs. There remains a knowledge gap about the differing efficiencies between passive and active facilitation methods. With limited funds and personnel, land management organizations will need to prioritize conservation funds and restoration methods. Additionally, organizations undertaking active marsh facilitation projects will need to understand the extent of adaptive management needs. Potential barriers to marsh migration include invasion of *Phragmites* after upland vegetation removal (personal communication – Kenny Raposa, Narragansett Bay NERR), failure of halophyte establishment, and existing agricultural infrastructure on the landscape. Numerous questions exist about the relationship between *Phragmites* and agricultural infrastructure, which may impede natural or facilitated marsh migration into the upland. Conservation and land management planning will need to understand and incorporate possible adaptive management needs in funding requests and personnel responsibilities.

Field monitoring protocols should focus on the local hydrology and the vegetation community to document changes in the environmental drivers of migration and the actual response of marsh vegetation. Surface and groundwater hydrology should be measured in both the marsh platform and in potential migration corridors throughout the year, if weather conditions permit, to capture coastal storm and king tide flooding events and document where the 3 – 5% flooding thresholds are reached. These data will capture the lasting impacts of flooding events on the groundwater table. Transect-based monitoring would accurately capture the change in the vegetation community from the salt marsh into the upland with at least half a dozen transects distributed across a site. Transect-plot or point-line intercept methodologies are both useful strategies to determine inland extent of salt marsh vegetation (or the retreat of the upland community). When initially setting up a transect-plot design, plots should be located at the furthest landward extent and center of each vegetation community (e.g., at every transition between vegetation communities). Permanent stakes should be installed at vegetation transition zones. Monitoring at appropriate timeframes (i.e., annual, five years, etc.) plots should be located at the new transition zone for each vegetation community. The distance should be measured between the new transition zone location and the original one at the permanent stake. The distribution of the plots captures the distance inland of each vegetation community as well as representative species and abundances for each community. Ideally, monitoring plots will shift inland and be used to calculate the rate of marsh migration and changes in vegetation composition.
Research Gap 3: New England context on coastal forest retreat and marsh migration

The bulk of the published, academic research on marsh migration and coastal forest retreat have been conducted in the Gulf of Mexico, Southeast Coastal Plain, and the Chesapeake Bay. There were very few studies that fully took place or at least possessed one site in New England (see Field et al. 2016, Anisfeld et al. 2017, and Burns et al. 2021). Interviews and discussions with colleagues in the Northeast show that marsh migration projects are being integrated into land management decisions and executed, however, their results are not being published or widely disseminated (Table 1). Examples include experimental tree removal at Narragansett Bay, fill removal in Rhode Island, and elimination of freshwater impoundments in Massachusetts (Table 1). Unique site history, climate, and geography may constrain the type of projects and dictate adaptive management needs for marsh migration. Long-term sentinel site monitoring of salt marshes at National Estuarine Research Reserves (NERRs) may provide some site and regional data on the rate of marsh migration. Current efforts by the National Marsh Synthesis Team (NAMASTE) document a small yet significant annual increase of 0.5 – 1.0% in halophyte cover at the upland edge across New England NERRs. Large meta-analysis projects across the country or regions, though not specifically tailored for marsh migration, may supplement to answer broad questions of species and ecotone shifts.

Different topography and plant species in New England compared to the Chesapeake Bay or Coastal Plain may also impact conclusions on the rate and efficiency of marsh migration. The Chesapeake Bay and Coastal Plain are characterized as generally gentle slopes (1 – 3%) between the marsh and upland compared to more commonly steeper slopes (> 10%) which may constrain marsh migration in New England, especially in the Gulf of Maine. Species may have differing tolerance for flooding and salinity in New England due to the climate and need to be evaluated to better understand the rate and potential indicators of marsh migration. For example, the leading edge of coastal forests in Chesapeake Bay are characterized by *Pinus taeda* (Loblolly pine) whereas the Great Bay Estuary is dominated by *Quercus bicolor* (Swamp white oak), *Q. rubra* (Northern red oak), and *Juniperus virginiana* (Eastern red cedar). Investigations into migration potential on steeper slopes and salinity tolerance of species characteristic of New England forests and tidal wetlands would improve identification of suitable marsh corridors.
Research Gap 4: Marsh Corridor Suitability

Marsh migration corridors within a region are currently identified through a combination of site surveys and geospatial modeling (e.g., SLAMM models, landscape resiliency [Steven et al. 2023]). However, the published literature clearly demonstrates several additional factors like upland vegetation community and local hydrology may have outsized impacts on the rate of marsh migration and the overall success of active migration projects. Only several studies have varied or observed the rate of marsh migration or forest retreat on different vegetation communities (Anisfeld et al. 2017, Walters et al. 2021), slope (Flester and Blum 2020), and hydrology (Kearney et al. 2019). The potential impacts of upland hydrology, vegetation, and topography on the ability of salt marshes to migrate landward have not been fully researched nor have additional factors been explored (i.e., groundwater inputs, soil composition, herbivory, etc.). Documentation of marsh migration or coastal forest retreat were typically observational across similar slopes or vegetation communities. Direct comparisons between varying slopes, upland vegetation, and hydrology may provide valuable information on migration suitability models in conjunction with SLAMM. Other factors which may impact project success have not been explored in the literature, including presence of relic agricultural embankments (Adamowicz et al. 2020), freshwater seepage, intense herbivory, and halophyte plantings in migration corridors. Development of consistent criteria and assessment protocols would aid in the identification of marsh corridors within larger regional contexts such as MarshRAM, which rates migration potential within a 60 m buffer of the upland based on topography, hydrology, vegetation community, and land-use (Kutcher et al. 2022).

The complexity of marsh corridor suitability increases significantly in suburban and urban shorelines and will require in-depth planning, decision-making structure, and public outreach and buy-in. Several municipal and state agencies and land trusts have utilized SLAMM models in identifying future marsh corridors (Table 1). It has recently been recognized that potential pathways for marsh migration contain superfund sites, underground storage tanks, and stormwater culverts (Burman et al. 2023). Martha’s Vineyard Commission is currently mapping out related infrastructure such as roads, pumpstations, and homes in the path of future SLR and marsh migration (Table 1). One major research need across the country will be the development of high-quality marsh migration pathways maps (Schoell 2022). Schoell highlighted needed characteristics in maps including better detailed coastal habitat types, barriers to migration, and conflicting uses of coastal lands. Resource managers could identify future adaptation and restoration projects and locate potential hazards and obstacles to
migration. Stevens et al. (2023b) developed a multi-metric, comprehensive framework to rank salt marsh resilience as a way to evaluate potential restoration projects based on three core groups: current condition, adaptability, and vulnerability to stressors. One of the metrics for their resiliency framework was modelled migration space in a 1.5 m SLR scenario using SLAMM models. The framework was applied to salt marshes of New Hampshire and provided a detailed database and rankings that would guide future restoration and climate adaptation projects for state and local stakeholders.

**Research Gap 5: Rate of upland conversion into high marsh**

One central question to marsh migration as a tool for conservation is the potential lag between marsh loss at the seaward edge from erosion and drowning and marsh gain through migration at the landward edge. Salt marshes adjacent to coastal forest and *Phragmites* may substantially shrink at the seaward edge before expanding into the upland. However, the lack of long-term field monitoring to document the rate of marsh expansion prevents any conclusions (see Wasson et al. 2013). Researchers at USFWS, GBNERR and DU have a manuscript in prep on monitoring ecotone data at four sites New Hampshire and Maine, spanning low marsh into coastal forests, monitored from 2014-2019. Preliminary results show significant changes in the plant communities throughout, including an increase in percent cover of halophytic and brackish species, and a decrease in percent cover of fresh and upland species, as well as a decline in tree canopy condition. This work is part of a larger USFWS effort led by Ralph Tiner, monitoring marsh-upland ecotones throughout New England and the Mid-Atlantic.

Importantly, there is research needed on the development of hydric soil properties (e.g., soil salinity, reduction-oxidation potential, sulfides, nutrient content, and organic matter) and related ecosystem services such as carbon storage (see Smith and Kirwan 2021, Hopple et al. 2022). Prior culvert expansion, freshwater impoundment removal, and living shoreline projects on coastal wetlands may provide clues to help estimate rates of peat development (see Craft et al. 2002) however, questions remain about shifts in soil properties from upland systems. The Coastal Zone Soil Survey Focus Team of the Natural Resource Conservation Service in North Carolina are leading research efforts to conduct soil profiling on both facilitated and natural marsh migration events headed by Dr. Matt Ricker (Table 1). Andrew Payne in Dr. Elizabeth Wasson’s lab at Drexel University is currently conducting research to better understand hydrology and salinity in salt marshes and adjacent upland forests throughout the Northeast (Table 1).
Table 1. Highlighted projects, primarily in the northeast, on the identification, facilitation, and monitoring of salt marsh migration. Projects were facilitated through outreach on listservs (New England Estuarine Research Society, Ecological Society of America) and direct connection.

<table>
<thead>
<tr>
<th>Topic</th>
<th>Contact (Organization)</th>
<th>Location</th>
<th>Project Description</th>
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</thead>
<tbody>
<tr>
<td>Marsh migration facilitation</td>
<td>Wenley Ferguson, (Save the Bay)</td>
<td>Rhode Island</td>
<td>Hydrology restoration, invasive plant management, and native plantings on marsh migration on Palmer River (Warren) with the Warren Land Trust. Active removal of 0.2 ha of upland buffer to facilitate marsh migration. Five years of post-monitoring showed small gains of marsh migration (<em>Spartina pectinata</em>) and mostly colonization by <em>Phragmites australis</em> and nuisance native shrubs. Future facilitation projects included removal of roads and stone walls physical blocking hydrology and an experimental upland removal study. Funding was secured from NOAA for the design and permitting of marsh migration barrier removal.</td>
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<tr>
<td></td>
<td>Kenny Raposa, Mary Schoell (Narragansett Bay NERR)</td>
<td>Rhode Island</td>
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<td></td>
<td>Alex Wolf (Scenic Hudson)</td>
<td>New York</td>
<td>Conservation of 84 acres of farmland along Binnekell River to serve as future marsh migration corridor. No active facilitation projects reported.</td>
</tr>
<tr>
<td>Marsh corridor identification</td>
<td>Liz Durkee (Martha’s Vineyard Commission)</td>
<td>Martha’s Vineyard, Massachusetts</td>
<td>Marsh migration pilot projects on Sengekontacket Pond and Edgartown with a focus on identifying affected human structures, infrastructure, and septic tanks within migration corridor. Seeking future funding for assessments on all salt marshes across the island. Marsh health assessment surveys of fringe marshes in Casco Bay in 2008. Identification of potential areas of marsh migration through simulated sea level rise models. Findings reported in Hayes et al. 2008.</td>
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<td></td>
<td>Mel Cote (Environmental Protection Agency)</td>
<td>Casco Bay, Maine</td>
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<td></td>
<td>Isabelle Stinnette (Hudson River Foundation)</td>
<td>New York – New Jersey</td>
<td>Identification of potential marsh migration corridors through SLAMM models in northern New Jersey.</td>
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<td>Organization</td>
<td>Location</td>
<td>Description</td>
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<tr>
<td>Nava Tabak (Scenic Hudson)</td>
<td>New York</td>
<td>Creation of a comprehensive, multi-faceted resilience model for salt marshes in the state of New Hampshire including marsh migration potential in future sea level rise scenarios. Data can be accessed at Great Bay NERR’s Salt Marsh Plan (see Stevens et al. 2023a).</td>
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<tr>
<td>Rachel Stevens (Great Bay NERR)</td>
<td>New Hampshire</td>
<td>Recently funded ‘Wetland Migration Pathway Planning, Prioritization, and Community Engagement’ project in the NERR system by the NFWF America The Beautiful Challenge in 2022. The purpose of the project is to create a pipeline of tidal wetland migration pathway projects in areas served by NERR in the Northeast (MA, RI), Mid-Atlantic (NJ, NC), and Southeast (SC, FL).</td>
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<tr>
<td>Mary Schoell (Narragansett Bay NERR)</td>
<td>NERR Nationwide</td>
<td>PhD research on movement of salinity and hydrology in salt marsh – upland forest complexes across the Northeast. Continual monitoring of tidal flooding, precipitation, porewater salinity, and hydraulic gradients to better understand interactions of abiotic factors on vegetation. Remote sensing analysis of the forest canopy to detect and understand seasonal fluctuations and impacts of salinity intrusion and coastal pulse events on forest health.</td>
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<tr>
<td>Monitoring</td>
<td>Northeast (NJ, NY, and RI)</td>
<td>Coastal Zone Soil Survey is conducting soil core profiling on abandoned agricultural land experiencing active flooding or salinization. Results available on Web Soil Survey at end of 2023.</td>
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<td></td>
<td>North Carolina &amp; Virginia</td>
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