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# A Case for Restoration and Recovery of Zostera marina L. in the Great Bay Estuary

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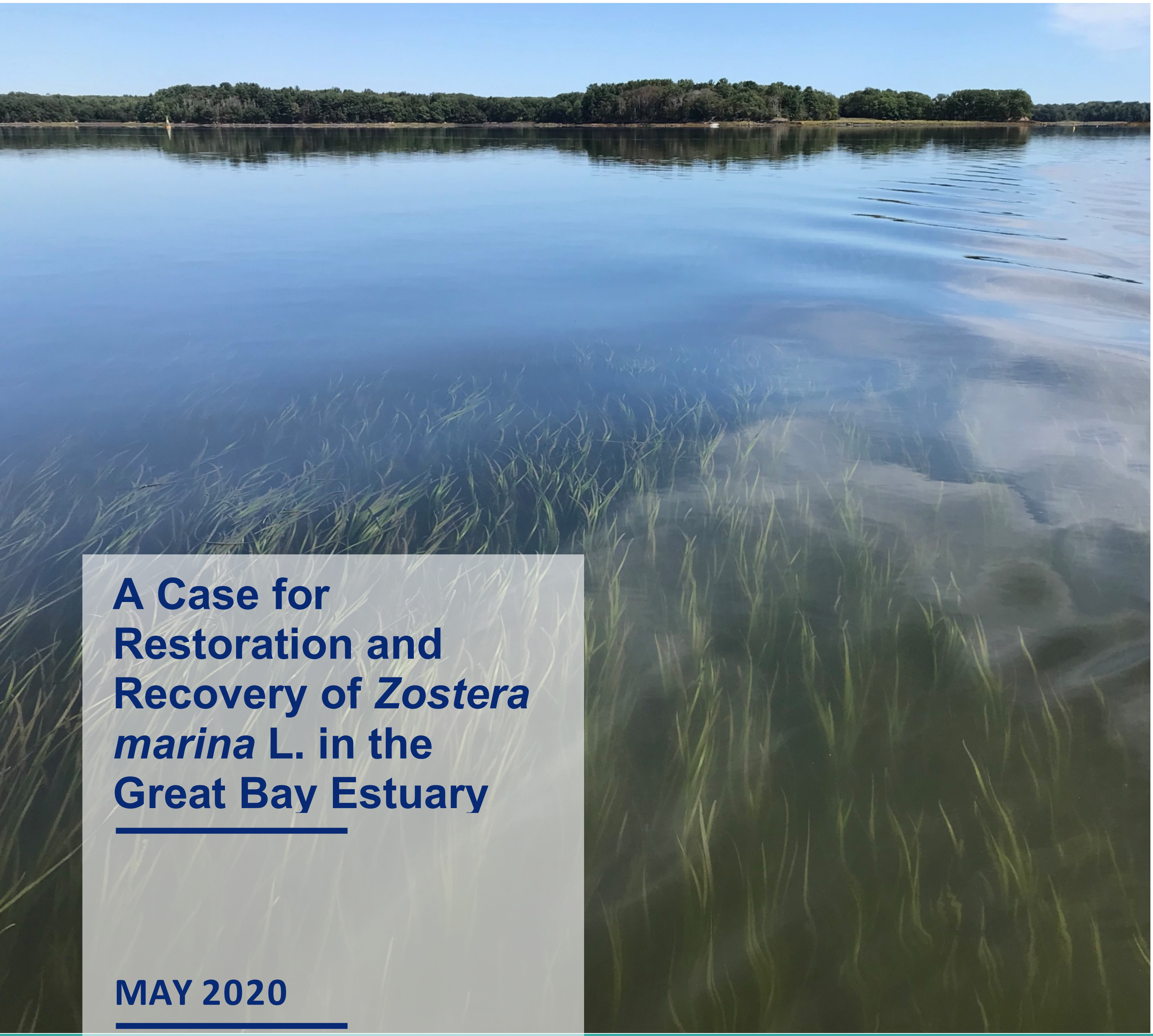

# A Case for Restoration and Recovery of *Zostera marina* L. in the Great Bay Estuary

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## EXECUTIVE SUMMARY

Eelgrass is a cornerstone species of healthy estuaries, providing numerous water quality, habitat, and resilience functions. In recent decades, the Great Bay-Piscataqua Estuary has lost over half of the acreage of eelgrass due to land use and demographic changes, changing precipitation and temperature patterns, and declining water quality. During this same period, eelgrass densities also severely declined at one of Great Bay's largest meadows. With much of the Great Bay Estuary failing to meet state water quality standards, many municipalities in the watershed recently invested in physical upgrades or process improvements in wastewater treatment plants. While there is much more work to be done to improve water quality, there are indications that recent reductions in nitrogen loads are creating conditions more favorable for eelgrass growth in certain areas. This consensus document, created by local scientists and managers, and intended for a general audience, describes the state of eelgrass science in the Great Bay Estuary. By understanding multiple local stressors that impact eelgrass health and utilizing an updated Site Selection Model, we will be able to better identify where trials of eelgrass restoration activity might occur and which transplantation methods may have the greatest likelihood of success. This paper is not a prescription for restoration but sheds light on research questions and data gaps, and establishes a foundation for the development of an eelgrass recovery strategy for the Great Bay Estuary. The authors of this paper believe that restoration efforts have the potential to facilitate the longer-term recovery of eelgrass as well as further the understanding of processes affecting eelgrass survival and growth in Great Bay, ultimately improving the health and resilience of the Great Bay Estuary.

## ACKNOWLEDGEMENTS

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## INTRODUCTION

### Seagrass – a cornerstone of healthy estuaries

Seagrasses are a keystone of healthy estuaries here in New England and throughout the world. They are rooted, flowering perennial plants that grow underwater and are typically submerged, even at low tide. With origins on land, seagrasses evolved to survive in shallow coastal waters about 100 million years ago (Olsen et al. 2016). They are distinct from marsh grasses that grow in the intertidal zone, from dune grasses which generally grow along beach dunes above the high tide line, and from macroalgae, which are not rooted and require more nitrogen in the water column than seagrasses which can access nutrients directly from the sediments through their root system. Also, macroalgae can thrive with much less light than seagrasses due to their thinner cell walls.

Throughout New England, a seagrass commonly called eelgrass, *Zostera marina* L., is the dominant species. When healthy, it forms dense underwater meadows that provide many benefits to a wide variety of organisms and people. Eelgrass meadows provide excellent physical habitat for young fish and shellfish (Thayer et al. 1984, Heck 2019), and because eelgrass is a photosynthesizing green plant, it produces great quantities of oxygen that many marine organisms need to thrive. Eelgrass utilizes nitrogen for growth, removing it from the water column and sediments (Sandoval-Gil et al. 2016). The roots of eelgrass plants hold sediment in place, while the dense canopy of long-bladed leaves slow waves and tidal currents, causing suspended sediments and pollutants to settle out of the water column (Fonseca and Fisher 1986, Short and Short 1984, Ward et al. 1984). This ability to capture sediments helps prevent coastal erosion and improves water clarity. Eelgrass meadows serve as critically important nurseries for finfish and shellfish, as well as feeding areas for fish, invertebrates, and birds (Thayer et al. 1984, Heck 2019). These are but a few of the important functions provided by healthy eelgrass beds that make places like the Great Bay Estuary such unique and valuable coastal ecosystems.

### Eelgrass – a species in decline

Over the past two decades, eelgrass meadows in the Great Bay Estuary (GBE) have severely declined (Short 2016), becoming less dense and shorter, or dying out altogether. Between 1996 and 2014, Great Bay lost 44% of the acreage of eelgrass, and 79% of eelgrass biomass (Short 2016). Eelgrass can die off for many reasons. While eelgrass is known for its ability to tolerate a wide range of environmental conditions related to temperature, depth and salinity, it grows best in high-light areas where sunlight can penetrate through the water (Thayer et al. 1984). Stressors that contribute to a loss of water column light penetration are often important factors in eelgrass loss. Declines in light penetration are associated with increases in nutrients, suspended sediments, Chromophoric Dissolved Organic Matter (CDOM), and organisms like phytoplankton, macroalgae, and epiphytes that also compete with eelgrass for light. Eelgrass is also susceptible to die-off from a pathogenic slime mold (*Labyrinthula zosterae*) which causes wasting disease, among 'Other Stressors' described on page 8. The loss of eelgrass can create a downward spiral in estuaries – with neither roots to hold sediment in place nor plants to attenuate wave action and filter water, a tipping point can be reached where increasingly turbid conditions can further compromise the ability of eelgrass to reproduce and thrive (Moksnes et al. 2018).

**Nitrogen in the Great Bay Estuary**

While loss of light penetration, warming water, disease and competition from other plants all have contributed to eelgrass decline worldwide, there is widespread agreement that elevated levels of nitrogen in many estuaries in the Northeast U.S. have contributed to the plants' downward spiral described above (Valiela et al. 1997, Latimer and Rego 2010, van den Heuvel et al. 2019). All three of these studies show that estuaries with more than 50 kilograms per hectare per year (kg $ha^{-1}$ $year^{-1}$) of nitrogen loading tend to experience significant losses of eelgrass habitat[1]. To put this in perspective, the loading rate for the Great Bay Estuary was 267 kg $ha^{-1}$ $year^{-1}$ in 2006 - before any municipal wastewater treatment facility (WWTF) upgrades and during a very rainy period - and 116 kg $ha^{-1}$ $year^{-1}$ in 2016, after some decreases from WWTFs and in a low-rain year.

A large source of the nitrogen load in the Great Bay Estuary comes from the 17 wastewater treatment facilities in communities that discharge into the system's seven rivers (Trowbridge et al. 2014). In Great Bay, WWTFs account for 50% of dissolved inorganic nitrogen (DIN) loads and 33% of total nitrogen (TN) loads (PREP 2017a). Additionally, nitrogen enrichment is amplified during the drier summer months – the peak growing season for eelgrass, macroalgae, and phytoplankton – when WWTFs represent a dominant source of water inputs to the estuary. During the summer months, over 70% of DIN in the estuary comes from WWTFs (PREP 2017a).

Another major source of nitrogen to estuaries is stormwater, which washes into rivers and the bay from storm events. Nitrogen on the land – from lawn fertilizers, atmospheric deposition, pet waste, and more – can quickly dissolve in rainwater and is carried through streams and rivers into estuaries. With an ever-increasing amount of hardened, paved surfaces in the communities surrounding Great Bay (PREP 2017a), rainwater is less able to percolate into the ground where the nitrogen can be utilized by soil microbes and plant roots, which absorb nutrients before they reach the estuary.

Yet another significant source of nitrogen in the Great Bay Estuary is septic systems, the waste disposal system used by over half of the population in communities around the watershed (Trowbridge et al. 2014). Septic systems are effective at breaking down harmful bacteria in human waste, but are less effective at reducing nitrogen that passes through the leach field and into groundwater, surface streams, and ultimately, the estuary (Barile 2018). Taken together, the nitrogen load from these non-point sources – stormwater and septic systems – accounts for approximately 50% of Dissolved inorganic nitrogen (DIN) loading and 67% of Total nitrogen (TN) loading to the estuary (PREP 2017a).

In other US estuaries including Boston Harbor, Chesapeake Bay, and Tampa Bay, previous management-coupled restoration efforts have shown that controlling nutrient inputs to an estuary through improved wastewater and stormwater management has been an effective strategy to reverse declines in seagrass (Greening et al. 2016, Lefcheck et al. 2018, Taylor et al. 2019), which have contributed to positive feedbacks that further enhance recovery (Lefcheck et al. 2018). Here in Great Bay, municipalities are only beginning to address non-point sources of nitrogen, but are seeing significant declines in the nitrogen load coming from WWTFs. Through both voluntary and court-ordered actions in recent years, municipalities have invested nearly $200 million in upgrades and process improvements at their WWTFs. Piscataqua Region Estuaries Partnership (PREP) estimates

[1] Note that the van den Heuvel et al. 2019 study used nitrate loading rather than total nitrogen

that by 2020, point-source nitrogen loading from WWTFs will be reduced by approximately 70% compared to 2012. (PREP 2017b, pp 36-41). Despite this significant progress, it is important to note that further reductions are attainable: several upgraded facilities are discharging nitrogen effluents (~5mg N/L) higher than limits of technology (3 mg N/L), and the two largest dischargers in the system have the highest rates of 6-9 mg N/L.

**Why Now?**

There is some evidence from 2019 aerial surveys and field monitoring that eelgrass is growing thicker and taller, as well as reappearing in areas where it was previously lost (Barker 2020., Short *in prep*., F. Short per. obs.). With reduced nitrogen loading, improving water quality may be creating conditions more favorable for eelgrass growth in certain parts of the Estuary. It is timely to consider two approaches – whether to wait for eelgrass to recover naturally or to actively undertake restoration.

Though natural recovery may already be happening in some areas of the Estuary as water quality improves, the process can be very slow, especially in locations that are recruitment-limited and cut off from seed supplies due to hydrodynamic factors. Active restoration, on the other hand, is frequently an expensive endeavor that must be approached carefully; reviews of seagrass restoration often point out that failure rates are quite high (e.g., Katwijk et al. 2016.) The authors of this paper believe there are appropriate instances of both approaches within Great Bay.

Pilot restoration projects, along with coordinated water quality monitoring, may add to our understanding of conditions and responses at particular sites (Short et al. 2002a, Moksnes et al. 2018). Determining where eelgrass can thrive, as well as where conditions are not yet suitable, can help inform targeted management decisions related to further pollution reduction and restoration. With parallel tracks of improving water quality and augmenting eelgrass recovery in the Great Bay Estuary with restoration, we may be able to reverse the downward spiral of eelgrass declines observed over recent decades. As eelgrass begins to recover, it can provide better filtration, more nutrient uptake, and reduced turbidity, which may further enhance conditions favorable for eelgrass recovery as well as the many ecosystem services that healthy eelgrass meadows provide. The authors of this paper believe that restoration efforts in the Great Bay Estuary have the potential to contribute in two ways: 1) to further the understanding of processes affecting survival and growth in Great Bay, and 2) facilitate the longer-term recovery of eelgrass.

## CURRENT CONDITIONS AND RECENT TRENDS

**Eelgrass Distribution, Density, and Biomass**

The spatial extent of eelgrass distribution in New Hampshire, including the Great Bay Estuary, has been decreasing since 1996 (Short 2016), when 2900 acres were identified and mapped (Trowbridge 2006). Since 1996, eelgrass acreage in the Great Bay and surrounding coastal areas has decreased by approximately half, with the most recent assessment of eelgrass totaling only 1,678 acres in 2019 (Barker 2020, PREP 2017b). Trends in Great Bay specifically are similar, with measurements in 1996 totaling 2,495 acres (Short and Short 2003, Trowbridge 2006) compared to 2019, which declined to 1,451 acres, a 42% decrease.

When examining trends in eelgrass populations, it is important to consider both eelgrass distribution and other health indicators such as density or biomass. Eelgrass distribution has been measured through aerial surveys in the Great Bay region since the early 1980s. While eelgrass distribution is a commonly used indicator, it has limits in utility because it can only inform us on where eelgrass is present or absent. Eelgrass may be determined to be present but have coverage between 10% and 100%, so the density and biomass of the eelgrass could vary widely but not be reflected in the aerial footprint. Indeed, density at one of the largest beds in the western portion of the Great Bay drastically declined along with distribution. In the late 1990s, this closely monitored seagrass bed averaged over 400 shoots/$m^2$ (Short et al. 1993) whereas in the same area in 2017, density was only 80 shoots/$m^2$ (Short 2017b), a decrease of 80%. Short (2016) reported that the biomass of eelgrass (metric tons) in the Great Bay had also decreased by over 80% since 1996, using an assessment method that converts percent cover categories to biomass categories (PREP 2012).

**Eelgrass Stressors**

In general, worldwide seagrass declines have been caused by multiple, cumulative stressors, with nutrients and sediments most often implicated (Short and Wyllie-Echeverria 1996, Orth et al. 2006, Waycott et al. 2009, Unsworth et al. 2015, Boesch 2019). Other stressors (e.g., wasting disease, sea level rise, temperature, invasive species) are typically more detrimental when coupled with increases in nutrients and sediments (Orth et al. 2006). Below, we describe some of the most critical stressors to eelgrass in the Great Bay Estuary, prioritizing those with chronic, long-term impacts as opposed to those that are more episodic, and focused in time and space (e.g., green crabs, birds, boating).

<u>Eutrophication</u>

As nutrient levels in estuaries increase, a process called eutrophication is set in motion. Algae – including phytoplankton and macroalgae – begin to proliferate and can cause a cascade of problems. First, algae can severely block light (Short et al. 1995, Hauxwell et al. 2001, Collier et al. 2008, Bittick et al. 2018) for eelgrass, which requires greater light than algae (Valiela et al. 1997, Markager and Sand-Jensen 1992). Second, decomposing algae can cause unfavorable biogeochemical conditions (Kennish et al. 2011), such as increased nutrient fluxes from sediment and anoxic conditions that promote sulfide accumulation (Green and Fong 2015) resulting in toxicity to eelgrass roots (Burkholder et al. 2007). Third, excess nutrients promote increased growth of algal epiphytes that adhere to eelgrass leaves, especially in the absence of grazers (Neckles et al. 1993). While some epiphytes are beneficial for food webs (Bittick et al. 2018), too much epiphytic growth on eelgrass can block light from reaching the leaf and diminish growth (Ward et al. 1984). Fourth, excess nutrients have been found to actually weaken the leaf structure of eelgrass, making it more susceptible to physical forces (Soissons et al. 2017). Ample light is needed for eelgrass to survive not only fouling organisms, but grazers and wasting disease.

In the Great Bay Estuary, nitrogen loading peaked between 2005 and 2007, however systematic assessments of nitrogen loading only began in 2003. Previous discrete studies of nitrogen loading estimate 640 tons per year in 1988 (reported in Short 1992) and approximately 715 tons per year in 1996 (Jones 2000). By 2006, loading more than doubled, exceeding 1600 tons per year. Since 2007, nitrogen loading has been steadily decreasing due to lower precipitation as well as substantial improvements at several WWTFs which began in 2014. For the first time since the most recent dataset began in 2003, total nitrogen loading to the estuary was less than 800 tons per year in 2015 and 2016 (PREP 2017a).

Overall in Great Bay there are no significant trends for phytoplankton, but data at the northern end at Adams Point, which date back to the late 1980s, demonstrate higher median values between 2000 and 2010, coinciding with peaks in nitrogen loading, as compared with the 1990s and years after 2010 (PREP 2017a). From 2013 to 2015, the median level for chlorophyll-a was below 5 µg/L, a level considered an indicator of good water quality (US EPA 2012). In the tributaries that feed Great Bay (e.g., Squamscott River and Lamprey River) some median values in recent years have slightly exceeded 5 µg/L. With recent wastewater treatment facility upgrades in Newmarket (2017) and Exeter (2019) not yet reflected in a trend analysis, we anticipate reduced nitrogen load and lower chlorophyll levels in these tributaries.

From 1980 to 2016 in Great Bay, red and green macroalgae increased from a mean percent cover of 8% to 19% (PREP 2017a). A recent report documents that macroalgae in Great Bay may be impacting eelgrass in some areas (Burdick et al. 2019). This report notes that red and green macroalgae have decreased at some intertidal stations in Great Bay over the last several years, but macroalgae remain high, exceeding 20% cover at several other stations, including new subtidal stations. Data on epiphytes, sediment sulfides, and organic content of sediments are either non-existent or have not yet been analyzed.

### Suspended Sediments

Development and the proliferation of hardened, paved surfaces within a watershed can lead to greater contribution of sediments to estuaries. In addition, sea level rise can lead to the erosion of salt marshes and add more sediment to the estuary. When sand, silt, and other particles wash into rivers and the bay, they create turbid conditions that block light and impede oxygen exchange from eelgrass leaves (Brodersen et al. 2017). If unchecked, a positive feedback loop can be set in motion – with less eelgrass to provide a natural filter and stabilizer of sediments, there can be increased resuspension of fine sediments and possible systematic population collapse (van der Heide et al. 2007). Increasingly intense and frequent storms due to climate change can also exacerbate sedimentation, as well as other stressors such as eutrophication and physically uprooting eelgrass (Wang and Linker 2005, PREP 2017a). Total suspended solids (TSS) are monitored at eight locations in the Great Bay Estuary. Between 2012 and 2015, median values at four of these sites were always higher than 15 mg/L, often used as a general threshold for habitat suitability for eelgrass (Kemp et al. 2004). At Adams Point located at the northwestern end of Great Bay, TSS data show a significant increasing trend between 1989 and 2016 (PREP 2017a).

### Chromophoric Dissolved Organic Matter

Chromophoric dissolved organic matter (CDOM) comes from the natural breakdown of plants in both terrestrial and marine environments. In high concentrations, CDOM can block enough light to be an important stressor (Gallegos 2005). Increased macroalgae can lead to increased dissolved organic matter through growth and decomposition (Khailov and Burlakava 1969). Other factors that affect CDOM include water chemistry and precipitation. In the Northeast, climate change dynamics appear to be increasing winter CDOM and dissolved organic carbon (DOC) delivery to the Gulf of Maine

(Huntington et al. 2016). de Wit et al. (2016) predict increased DOC across seasons in a wetter climate.

A complete trend analysis for CDOM has not yet been reviewed for quality assurance and processed for Great Bay. However, in 2007, a study by Morrison et al. (2008) in Great Bay found CDOM partly contributed to light attenuation. They calculated different components of light attenuation through mean daily contributions, finding different sources of attenuation: water 32%, total suspended solids turbidity 29%, CDOM 27%, and Chl-a 12%. These calculations were noted by the authors to vary greatly depending on storms and other conditions. Morrison et al. (2008) highlighted the importance of multiple factors that attenuate light reaching eelgrass (e.g., phytoplankton, TSS, or CDOM) rather than singling out the largest contributor. Furthermore, studies have shown that eelgrass requires more light in degraded conditions, such as in the presence of organically enriched sediments (Kenworthy et al. 2013).

### Wasting Disease

Wasting disease is an infectious disease that weakens and kills eelgrass shoots (Short et al. 1986, Burdick et al. 1993). It is caused by a protozoan and has characteristics of both chronic and episodic stressors (Muehlstein et al. 1988). Outbreaks of the disease in the 1930s caused large scale losses, with eelgrass recovery requiring decades (Milne and Milne 1951) and in some areas, recovery is still incomplete (Muehlstein 1989). Wasting disease is always present in an estuarine ecosystem but can break out and have a large impact when conditions are favorable for the disease (Burdick et al. 1993, Short et al. 1993, Kaldy 2014). While eelgrass needs some saltwater to survive, beds in lower salinity areas decline less and recover faster from this disease (McKone and Tanner 2009). It is important to note that eutrophication, warming waters, decreasing light and herbicides can all aggravate wasting disease (Kaldy 2014, Dawkins et al. 2018, Hughes et al. 2017b).

In the Great Bay Estuary, wasting disease has not been consistently monitored and reported over time, yet was severe enough to be mentioned in annual monitoring reports in 1995, 1999, 2000, 2002, 2003, and 2004. All noted wasting disease outbreaks have been low compared to the 1988 large-scale wasting disease eelgrass die-off (PREP 1997, Burdick et al. 1993, *pers. obs*. F. Short). The most recent reporting of wasting disease was in the 2013 eelgrass distribution report, where it was noted that disease presence was low (Short 2014).

### Other Stressors

Herbivory: Animals such as geese and ducks feed on eelgrass. This has been observed in Great Bay (Rivers and Short 2007). Although the activities tend to be episodic, the impacts can last over many years.

Bioturbation: Benthic organisms such as green crabs, horseshoe crabs, and worms can have a significant negative impact on eelgrass (Davis et al. 1998, Neckles 2015). As with herbivory, these impacts are usually episodic rather than long-term and chronic. Conveniently, green crab foraging appears to be greatest outside of eelgrass's primary growth season (Fulton et al. 2013). Little data has been published on bioturbation specific to the Great Bay Estuary.

Warming Water: Eelgrass experiences stress around 25°C and higher (Short 1980). Studies have documented losses of eelgrass in the Chesapeake Bay (Lefcheck et al. 2017, Shields et al. 2019) as well as in Southern California (Johnson et al. 2003) due to short-term increases in water temperature and interactions with poor water clarity. In the Great Bay Estuary, it is unclear whether warmer waters are stressing eelgrass, and data on temperatures from the shallow areas at the SeagrassNet sites where eelgrass grows have not yet been analyzed.

Extreme Storms: Storms such as the Mother's Day Storm of 2006 in the Great Bay Estuary can have a temporary but significant impact on eelgrass beds. Intense storm events can uproot plants and deposit sediment which can then be easily resuspended, attenuating light and creating a positive feedback cycle detrimental to eelgrass (Kemp et al. 2016). Climate change projections for the Great Bay region forecast more frequent occurrence of extreme storms (Knutson et al. 2013).

Dredging and Filling: Navigational dredging as well as other development-related actions (e.g., installation of submerged cables) can have significant impacts on habitat and water quality. While there have been few dredge and fill projects in the Great Bay Estuary recently, a dredge project at the Turning Basin on the Piscataqua River is currently in the permitting process. If that project proceeds, it will disturb documented eelgrass beds and will be subject to mitigation.

Boating, Docks, and Moorings: Depending on the number and types of such structures, eelgrass can be negatively impacted from light limitation created by docks, and chain scarring from moorings (Burdick and Short 1999). In general, docks and moorings have limited overlap with present and historic eelgrass distribution and have not appreciably changed in the Great Bay Estuary, except in Little Harbor.

Shellfish Restoration and Aquaculture: Many studies have looked at impacts of eelgrass and oyster restoration efforts on each other (see Ferriss et al. 2019 for meta-analysis). While some have shown synergistic effects, there are many examples of shellfish restoration and aquaculture projects negatively impacting eelgrass when not properly managed (e.g., NMFS 2017).

**Data Gaps**

The Piscataqua Region Estuaries Partnership (PREP), a significant convener in the watershed and authors of this paper, is developing a new Integrated Research and Monitoring Plan (due to be completed by early 2021) with a focus on eelgrass as well as other critical biological resources. As of April 2020, PREP had held several meetings involving both local scientists as well as external eelgrass experts, noting critical questions and data gaps. The following list of most critical (not exhaustive) data gaps draws heavily from those conversations.

"Tier 2" Monitoring: as described by Neckles et al. (2012), involves monitoring of intermediate ecosystem variables (e.g., percent cover, canopy height) at a number of sites throughout the estuary. This is in contrast to Tier 1 monitoring, assessing presence/absence of eelgrass throughout the entire estuary, and Tier 3 monitoring, involving detailed assessment on the order of the SeagrassNet protocol (Matso and Short 2019). Although still limited, Tier 2 and 3 monitoring were expanded last year to incorporate four new locations in Great Bay, as part of a new subtidal seaweed monitoring program and a second SeagrassNet site in Portsmouth Harbor, respectively.

Shallow Area Water Temperature: Water temperature is currently obtained from various *sondes* and grab sample stations (Matso and Gregory 2019), but these stations are located in deeper waters where temperatures tend to be cooler. Temperatures are also collected at SeagrassNet sites, but only for a couple locations.

Eelgrass Biomass and Density: These metrics have been monitored in Great Bay Estuary but limited to a couple of sites and not consistently in space and time. For example, biomass data exists for Great Bay (proper) but not for the Piscataqua River or Portsmouth Harbor.

Nitrogen Budget: It is not currently known how much nitrogen stays in our estuarine system or exits to the atmosphere or ocean, versus sinking into the sediments where it can be regenerated later. Understanding the nitrogen cycle in Great Bay Estuary will make it easier to understand the relationship between nitrogen loading dynamics and biological response and time-lags.

Sediment Budget: While TSS levels show increasing trends in some portions of the Estuary (PREP 2017b), a full-scale monitoring plan is needed to understand where the sediment is coming from and where it is ending up. Are sediments coming from riverine inputs and salt marsh erosion, or is it simply sediment that is being resuspended and moved around within the estuary?

Light Penetration: As with water temperature (see above), some light attenuation data exists, but much of it is outside of eelgrass beds. SeagrassNet light attenuation data in the Great Bay (beginning in 2007) exists but has yet to be analyzed.

Phytoplankton: See "Light Penetration" above. In addition, it is acknowledged that phytoplankton levels can change significantly within hours or days; however, accurate phytoplankton sampling only occurs on a monthly basis, potentially missing the dynamic nature of this ecosystem component.

Epiphytes: Epiphytes attenuate light and can indicate nutrient levels or grazing levels, or both. However, data on epiphytes are currently lacking.

Eelgrass sexual reproduction. Eelgrass populations depend to various extents on sexual reproduction to maintain and to recover from environmental stress; some populations behave as perennial plants and rely on vegetative expansion, while other populations behave as annuals and maintain their abundance primarily through sexual reproduction (Jarvis et al. 2014, Johnson et al. 2020). A review of eelgrass distribution and losses in Great Bay since 1996 suggests sexual reproduction is an important component of eelgrass resiliency (Kenworthy et al. 2017). Therefore, greater insight into whether "recruitment failure" is occurring in our estuary is needed - and if so, at what stage of recruitment. In Great Bay, SeagrassNet protocols (e.g., Short 2017b) have documented the number of flowering shoots in Great Bay since 2007. However, we do not have data for the other aspects of sexual recruitment such as seed dispersal, seed viability, burial depth, seed germination, and seedling mortality (Johnson et al. 2020).

Finally, we acknowledge that environmental monitoring directly within seagrass locations, including existing beds and potential future beds through recovery and restoration, is limited. Additional monitoring of water quality and sediment characteristics in these areas would provide further insight on how stressors are interacting with seagrass as well as inform future restoration activities.

Table 1: Summary of Stressors on Eelgrass in Great Bay Estuary, noting data gaps.

| | Amount of Data Available | Influence of Variable as a Stressor | Trends | Current |
|---|---|---|---|---|
| **Nitrogen Loading** | High | High | Peaked in 2007 – 2008; decreasing since then | Still considered high enough to stress eelgrass |
| **Chl-a** | Medium | Unknown | No strong trends; higher between 2000 and 2010 | Mostly low outside of the annual bloom in early spring |
| **Macroalgae** | Medium | High | Increased since 1980 | Recent decrease at some stations; still high at others |
| **Epiphytes** | Low | Unknown | n/a | n/a |
| **Sediment Sulfides** | Low* | Unknown | n/a | n/a |
| **Sediment Organics** | Low-Medium* | Unknown | n/a | n/a |
| **Suspended Sediments** | Medium | High | Increased | High |
| **CDOM** | Low | Unknown | n/a | n/a |
| **Wasting Disease** | Medium | Episodic** | n/a | n/a |
| **Herbivory** | Low | Episodic** | n/a | n/a |
| **Bioturbation** | Low | Unknown | n/a | n/a |
| **Water Temp** | Medium* | Unknown | n/a | n/a |
| **Dredge/Filling** | High | Episodic** | n/a | n/a |
| **Boating Related** | Medium* | Medium | n/a | n/a |
| **Shellfish Restoration & Aquaculture** | Medium | Unknown | Increasing | n/a |

*Data may exist but have not been analyzed or subject to QA/QC.
** Episodic refers to site-specific and time-limited events.

## AN APPROACH TO RESTORATION

As water quality conditions in parts of the estuary begin to improve from upgrades at municipal wastewater treatment facilities and other pollution reduction programs, it is timely to consider the ecological objectives we hope to achieve through eelgrass restoration and recovery activities, where a new wave of such activities should occur, and which methods might provide the greatest likelihood of success. It is also important to fully assess past restoration efforts and clearly understand factors that contributed to their success and failure (van Katwijk et al. 2016).

**Where to restore eelgrass?**

A critical concern in eelgrass restoration is site selection - the determination of where and under what environmental conditions eelgrass planting activities are likely to be successful. Many past eelgrass restoration efforts have been conducted based on incomplete information about the planting site. Given the time, money and effort that is put into transplanting eelgrass and variable success rates, it behooves the restoration team to know as much as possible about the target area so that the site has the highest likelihood of success. Restoration should be based on a review of previous eelgrass restoration, a careful synthesis of all available information to decide what areas are most suitable for transplantation, and address critical data gaps.

Site selection models are utilized to quantify available environmental data and, in some cases, incorporate other critical factors such as historical distribution of seagrass, prior restoration efforts, etc. Models have been developed to process data in different ways, ranging from simple ranking and scoring to robust statistical models (e.g., generalized linear model, classification regression tree; Stankovic et al. 2019). Statistical modeling is a powerful tool that is more objective but requires a large spatial dataset across a range of conditions to be effective.

In Great Bay, a Site Selection Model (SSM) was developed, applied and refined through quantitative and qualitative measures, identifying the most suitable sites for eelgrass restoration (Short et al. 2002a). This SSM uses ranking and a stepwise approach with best available information and specific field measurements to determine suitable locations for long-term transplantation success using a Preliminary Transplant Suitability Index (PTSI) and Transplant Suitability Index (TSI). The PTSI incorporates available parameters such as water depth, current and historic eelgrass distribution, sediment type, wave exposure, and water quality (e.g., salinity, temperature, turbidity, chlorophyll, dissolved inorganic nitrogen, CDOM, etc.). Parameters for each site are qualitatively ranked and totaled for an overall score; higher numbers indicate better potential for eelgrass habitat. This tabulation results in a matrix or map of potential eelgrass habitat (e.g., Figure 1), which is then used as the basis for further location-specific refinement.

The next step in the Short et al. (2002a) model is to collect field measurements and gather input from local experts. Simple field measurements on light and bioturbation from crabs, horseshoe crabs, clamworms, geese, etc. are worth investigation before investing time and resources for a full-scale restoration. From this newly collected information, the TSI is calculated by combining the results of the PTSI with qualitative rankings, as well as light measurements and bioturbation assessments. Further refinement can be made by incorporating information from local researchers, fishermen, and other users of the estuary, which may provide site-specific information relating to other floral and faunal communities (e.g., abundance of resident and drift macroalgae, green crabs, clam worms), as well as bottom type and human activities (oyster restoration and aquaculture, dredging, boating activities, mooring fields). These categories of information are informally considered to screen out sites.

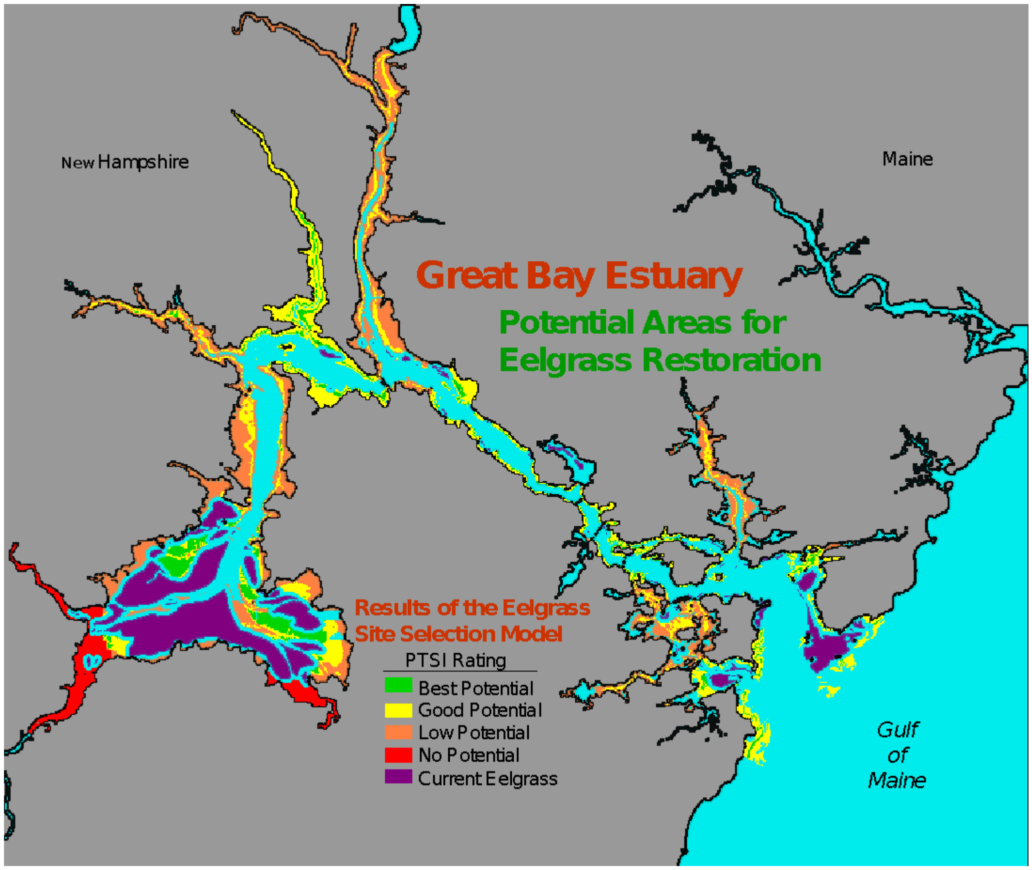


Figure 1 – Example results of the first step (PTSI) in the site selection model for Great Bay, NH based on 2005 data.

The final steps in this site selection model include choosing the most suitable sites based on the results of the PTSI and TSI, and conducting a small-scale test using either transplanted shoots or seeds. Transplants should be conducted using an established site-appropriate method (described in the next section) and routinely monitored. Short et al. (2002a) suggest monitoring shoot survival and eelgrass growth (leaf marking technique) after one month, as well as measuring canopy height, leaf nitrogen content and continuous light data. Routine visits are recommended to observe any sudden eelgrass mortality from bioturbation or other potential causes. The final suitability assessment for eelgrass transplanting at this site incorporates all the steps mentioned along with results from test transplantation and monitoring.

Comprehensive application of the site selection model may not be feasible in some cases due to time and resource constraints. In these instances, it may be beneficial to prioritize information based on ecological importance, data availability, and cost. A good approach to prioritizing information is to begin with parameters in the PTSI (Short et al. 2002a) and work through other steps as resources allow. Arguably, the most important parameters to consider are water clarity, depth, and present-day and historic distribution of eelgrass. Other parameters to consider include current and wave exposure, sediments, and dissolved nutrients, however these are generally less available. The second and third steps of the model become more resource demanding. Gathering quantitative and qualitative information from local experts and users of the Bay is valuable and cost-effective but can be time-consuming. Additionally, specific field measurements of water quality and bioturbation provide valuable information on a smaller scale but can be costly and time-consuming. Finally, pilot transplantations or seeding of eelgrass and monitoring are the most resource intensive part of the process but may also provide the most valuable information. Since transplantation or seed survival and growth are dependent upon all the site conditions evaluated in the PTSI, test plots are the best way to determine if the site is suitable for large-scale restoration.

This site selection model has been used across New England. One example of its use was used in New Bedford, Massachusetts where the highest-recommended site was planted with one acre of eelgrass using TERFS (Transplanting Eelgrass Remotely with Frame Systems; Short et al. 2002b); the site expanded to 14 acres after 10 years. In contrast at Boston Harbor, the site selection model initially showed no sites with high likelihood of transplanting success, but transplanting took place nonetheless at some of the sites rated "fair." Restoration initially failed, but later test plantings and large-scale restoration – coinciding with improved water clarity – were successful (Leschen et al. 2010). Finally, the site selection model was used in Plum Island Sound, Massachusetts where sites recommended by the model failed because high current conditions had not been initially incorporated. Later iterations of the model have incorporated currents.

The SSM was applied in 2005 for the Great Bay Estuary (Figure 1) and identified three sites that ranked high for eelgrass restoration (Short *pers. comm.*):

1. Tricky's Cove in Little Bay, southwest of the Sullivan Bridge, sustained eelgrass prior to peak nitrogen loads and increased storm activity seen in the Great Bay Estuary (2005-2010).
2. "North of Defense Fuels" in the Piscataqua River just north of the Port Authority Pier represents a site that was successfully transplanted during the 1993 NH Port Mitigation Project, which pre-dated the Site Selection Model. Eelgrass survived for many years until 2007, when it completely died off throughout the entire Piscataqua River (Beem and Short 2009). The period between 2005 and 2007 represents the high point for storms and nitrogen loading, so both sediment and nutrient levels were quite high.
3. Fishing Island in Pepperrell Cove on the Kittery side of Portsmouth Harbor was a 15-acre eelgrass flat that was denuded of eelgrass in 2003 by overwintering Canada geese (Rivers and Short 2007). This site is shallow and easily accessible by land, making it suitable for a citizen-based transplanting project.

While these sites ranked highly based on data available in 2005, the authors of this white paper concur that updating the Site Selection Model based on current water quality, sediment and bioturbation data, and conducting pilot transplantations at suitable locations, would be a valuable next step toward the development of an eelgrass restoration and recovery strategy for Great Bay.

**How to restore eelgrass?**

Seagrass reestablishment can occur naturally through vegetative expansion, clonal fragmentation and sexual reproduction, but in some cases requires human intervention to accelerate colonization. There are several methods for active restoration pictured in Figure 2, including anchoring shoots directly to the sediment (Davis and Short 1997; Fonseca et al. 1998; Orth et al. 1999); attaching multiple shoots to planting fabric (Pickerell et al. 2012), shells (Lee and Park 2008) or various types of frames (Short et al. 2002b, Leschen et al. 2010, Kidder et al. 2013) that are weighted or secured to the sediment; or planting plugs of shoots with intact sediments (Fonseca et al. 1998). Bare-root transplantation methods (i.e., methods that involve transplanting eelgrass shoots only, without intact sediments) are preferred for minimizing potential impacts to donor sites (Davis and Short 1997). If utilizing donor plants for restoration, Burdick et al. (2012) found removal of up to 20% of live shoots with rhizomes had no long-term impacts (≥ three months) on shoot density. Rather than using donor beds, detached

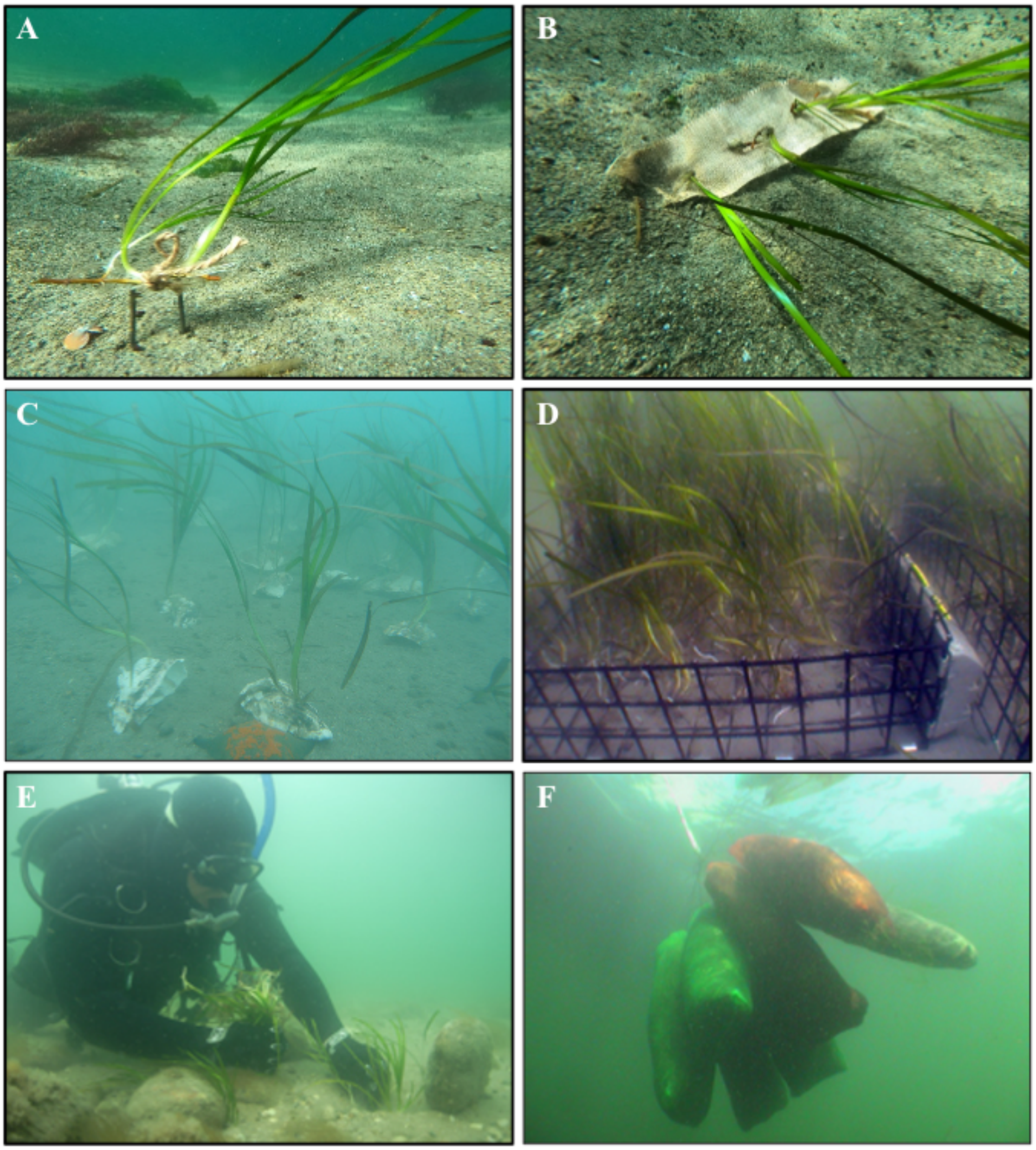


Figure 2 – Examples of common methods of eelgrass restoration. Adult shoots can be anchored to the sediment using materials such as staples (A), burlap fabric (B), shells (C) or TERFS (D); or can be planted directly into the sediment (E). Many other anchoring materials or planting techniques, not shown here, can also be used. Alternatively, eelgrass seeds can be collected and distributed by hand, mechanically, or through methods such as Buoy Deployed Seeding (BuDS; F). Photo credit: Jeff Gaeckle of the Washington State Department of Natural Resources; Long Island's Seagrass Conservation Website (seagrassli.org); and Lee and Park (2008).

shoots drifting in the wrack can be collected, planted and propagated in tanks then harvested as restoration plantings with no impacts. Such plants may have greater genetic diversity than those collected from one donor area. Genetic diversity of plants was positively correlated with resilience to stress associated with eutrophication in a tank experiment (Plaisted et al. 2020), so multiple donor sources may increase survival and performance. Eelgrass may also be propagated from seed or seedlings, which has been demonstrated to be successful in areas of Chesapeake Bay, where natural seed dispersal was limited and environmental conditions were favorable (Orth et al. 2012).

Methods in which eelgrass shoots are attached to a substrate before planting require a considerable investment of time in preparing the planting units, but require less time in the water for planting. TERFS (Transplanting Eelgrass Remotely with Frame Systems) is a method of attaching plants to a frame that can be deployed, retrieved and reused (Short et al. 2002b). TERFS consist of a 0.25 $m^2$ square frame plastic-coated wire grid (lobster pot mesh) in which approximately 50 eelgrass shoots are tied to grid intersections with biodegradable paper, in pairs with the rhizomes pointing in opposite directions (Davis and Short 1997). These transplant frames with attached plants may be carefully dropped into the water off the side of a boat. Another method utilizing remote deployment is attaching eelgrass shoots to shells with holes drilled in them (Lee and Park 2008). Transplanting via shells has the advantage of not requiring SCUBA and utilizing naturally occurring material, but may not be appropriate in high energy systems.

In contrast to remotely deployed transplants, methods in which eelgrass shoots are anchored directly to the sediment require less preparation time but can be more labor-intensive while planting. One such method is the horizontal rhizome method where plants are anchored to the sediment with bamboo shoots (Davis and Short 1997). Rhizomes are aligned parallel to one another and pointing in opposite directions. This method does not require any advance preparation of the planting units but generally requires SCUBA for large-scale planting efforts, especially in deep water. A second method often requiring SCUBA and substantial preparation time involves weaving multiple shoots in a tortilla-sized circular burlap fabric (Pickerell et al. 2012). "Tortillas" hold approximately 10 shoots around the outer edge to allow sediment to be placed in the middle to anchor it to the seafloor and allow for outward expansion of the shoots. Seagrass can also be restored by introducing seeds directly into the restoration target area (Granger et al. 2002; Orth et al. 2012). Researchers have also anchored reproductive eelgrass shoots to a target area as well as encapsulated seeds with a biodegradable coating to facilitate sinking and reduce seed predation (Pickerell et al. 2005).

A variety of transplant methods have been evaluated to determine the most efficient and successful approach for restoring eelgrass and found more or less similar success rates among techniques evaluated in both Casco Bay, ME (Neckles and Short *in prep*) and Chesapeake Bay, VA (Luckenbach et al. 2011). They noted restoration technique, whether bare root, plugs, or seeds, was a less important factor for success than site specific conditions (e.g., current, depth, sediment type, sulfide toxicity, nitrogen loading, biologic activity, disease). Restoration approach is often site dependent, as local factors can highly influence transplanting technique in different ways. For instance, currents can play a major role in inhibiting restoration efforts by exacerbating resuspension of sediments, blocking light penetration, washing away seeds, and uprooting or smothering transplants from increased drift algae (Moksnes et al. 2018). Additionally, benthic infauna has been shown to decrease transplant survival by burying or clipping shoots (Davis et al. 1998, Neckles 2015), but may aid in burying seeds (Blackburn et al. 2014). Evaluation of transplantation techniques should also consider associated costs and environmental impacts.

Although an important factor when considering restoration approaches, reported costs of eelgrass restoration have varied widely, even when the same approaches have been used. Reported costs ranged from a few thousand to over a hundred thousand dollars per hectare depending on multiple factors beyond the approach used, including site selection, time of year, water conditions, planting density, materials, etc. (Busch et al. 2010, Fonseca et al. 1998, Short et al. 2002c; Zhao et al. 2016). Additionally, many estimates likely underestimated the costs of projects, and values in the hundreds of thousands of dollars per hectare over the life of a project may be expected (Fonseca et al. 1998).

Past seagrass restoration efforts in the Great Bay Estuary have primarily been successful using bare root planting methods such as TERFS and the horizontal rhizome method. TERFS have been used at multiple sites around the Estuary including in areas with contaminated sediments (Hoven et al. 1999). Eelgrass trial plantings in Great Bay began in the late 1980s – after severe declines from Wasting Disease – by collecting eelgrass shoots from wrack, planting them in tanks, and then planting into the field. The first major transplant effort in Great Bay was part of the mitigation for the expansion of the NH Port Authority in 1993, where seven acres were planted at multiple sites in the Piscataqua River and Little Bay. Transplanting was done using the Horizontal Rhizome Method (Davis and Short 1997). Two of the locations on the Piscataqua were particularly successful, exceeding 2.5 acres after 5 years. Monitoring of restoration efforts demonstrated general eelgrass expansion (Beem and Short 2009), similar structure (density and biomass) to reference sites, fish habitat creation (Evans and Short 2005), and the development of success criteria for eelgrass restoration (Short et al. 2000), which led to the formulation of the Eelgrass Site Selection model described above (Short et al. 2002a). Overall, the NH Port Authority transplantations were successful. However, eelgrass began to decline in the Piscataqua River and in Little Bay at both transplant and natural reference sites between 2001 and 2005 (Beem and Short 2009). By 2007, all the eelgrass had died, coinciding with peak nitrogen loading and increased storm activity in Great Bay. A second large scale restoration began in 1998 as part of the mitigation for dredging Little Harbor in New Castle, NH. Transplantation of five acres of eelgrass was done using TERFS (Short et al. 2002b, 2006). No long-term monitoring was done for this project, but eelgrass has persisted and thrived in most of this area (F. Short *pers*. *obs*.).

In areas where natural recruitment is limited and traditional restoration techniques have proven unsuccessful, experimental restoration techniques may be warranted. Moksnes and colleagues (2018) recommended using non-traditional restoration approaches in a Swedish Bay, where elevated resuspension of the sediments from a large loss of seagrass as well as increased drift macroalgae made the bay uninhabitable for transplantation. Some of these approaches include dampening wave energy with artificial structures, stabilizing the sediment (e.g., sand capping) to reduce resuspension of newly formed mudflats, and a physical structure for catching drift algae. Other experimental approaches include use of silt fences to prevent resuspension and drift algae, concrete reef structures for wave dampening, restoration of hydrology and sediment topography, or screening and cages to prevent clam worms and green crabs from uprooting transplants, respectively. Oyster reefs or even oyster aquaculture may also provide a reduction in energy and sediment suspension by covering barren sediments, dampening wave energy and filtering water. In Great Bay Estuary, there has been continued oyster restoration and rapid growth of commercial aquaculture in recent years. Many of these experimental approaches require further investigation before applying them to a large-scale restoration.

A monitoring program is recommended in association with all restoration efforts to document: survival and abundance at the restoration site; potential effects to the donor site; and identify similar sites that may be suitable for restoration. In fact, continued monitoring is essential to evaluating the success of restoration and optimizing future efforts (van Katjwik et al. 2017 or Hughes et al. 2017a). Plant survival, density, heights, seed viability, and proportion of reproductive shoots should be tracked over time. Any transplantation effort should be monitored multiple times annually, especially in the first year to help identify potential sources of loss and stress that occur rapidly (e.g., bioturbation). Additionally, monitoring environmental parameters such as underwater light availability, water temperature, salinity, sediment texture and organic content, sediment sulfide concentration, water quality parameters, sediment pH, and green crab abundance would provide important insight on project outcomes that will be helpful to future local and regional restoration efforts.

## CONCLUSION

Eelgrass is a cornerstone species of healthy estuaries, providing habitat, water quality, and resilience functions. In recent decades, the Great Bay-Piscataqua Estuary lost over half of its acreage of eelgrass due to many factors including land use changes and declining water quality. There are indications that water quality – particularly nitrogen loading – may be improving due to significant investment in wastewater treatment by many municipalities around the watershed. By understanding the multiple stressors that impact eelgrass health and incorporating these into a site selection model, we can better identify where trials of eelgrass restoration activity could occur and invest in sites that have the greatest likelihood of success. The coupling of water quality improvements with restoration and recovery of eelgrass could reverse the downward spiral in estuarine health over recent decades and restore the function and resilience of the Great Bay Estuary.

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