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The status and future of tidal marshes in New Jersey faced with sea level rise

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Abstract: Salt marshes are key coastal ecosystems that provide habitats for wildlife, including invertebrates, fishes, and birds. They provide ecosystem services such as protection from storm surges and waves, attenuation of flooding, sequestration of pollutants (e.g., blue carbon), and nutrient removal. They are currently under great threat from sea level rise (SLR). We collected information about trends in the horizontal extent (acreage) of New Jersey salt marshes and recent elevation changes compared with the current local rate of SLR in New Jersey, which is between 5 and 6 mm year^{-1}. We found pervasive, although variable, rates of marsh loss that resulted from both anthropogenic disturbance as well as edge erosion and interior ponding expected from SLR. Elevation trends suggest that the current rates of SLR exceed most marsh elevation gains, although some *Phragmites*-dominated marshes keep pace with SLR. Four potential remedies to address current coastal trends of marsh loss were described in the context of New Jersey's regulatory and management environment: protection of marsh inland migration pathways, altered management of *Phragmites*, thin layer sediment placement, and living shoreline installations. Proactive steps are necessary if coastal wetland ecosystems are to be maintained over the next few decades.

Key words: ecosystem services, habitat, surface elevation table, *Phragmites*, thin layer deposition, living shoreline.

Introduction

Salt marshes are key coastal ecosystems that provide habitats for wildlife, including invertebrates, fishes (including important commercial species), and birds. They provide vital ecosystem services to humans, such as protection from storm surges and waves, attenuation of flooding, sequestration of pollutants (including carbon), and nutrient removal. Narayan et al. (2017) projected that the presence of tidal wetlands in the northeastern United States avoided $625 million in flood damage from Hurricane Sandy and estimated that salt marshes reduced annual flood losses by 16%. If marsh acreage is reduced and coastal storms become more intense, coastal communities will lose protection and will be subject to greater storm damage (Sun and Carson 2020). Reduced areas of

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marshes, which are nurseries for some commercial and recreational fish, will likely also lead to reduced fish production.

Key threats to salt marshes include land reclamation, coastal development, dredging, and sea level rise (SLR). While SLR is the major driver of loss, other factors can amplify the vulnerability of tidal wetlands to climate change. Eutrophication can contribute to marsh loss (Deegan et al. 2012) by increasing the above-ground biomass of marsh plants, decreasing root biomass, and increasing decomposition, resulting in plant instability. Top-heavy plants can fall over or be pulled out by waves, causing creek-bank collapse and marsh conversion to unvegetated mud. Overfishing has led to increased populations of herbivorous marsh crabs (*Sesarma reticulatum*), whose excessive consumption of marsh grasses caused marsh die-back in some areas (Bertness et al. 2014). Modification of coastal hydrology through tidal restrictions, construction of dams, drainage ditches ("mosquito ditches"), and ponds for mosquito management can amplify vulnerability to SLR. Such practices may reduce sediment inputs (Weston et al. 2011), contributing to marsh loss through direct habitat conversion (Powell et al. 2020), as well as by enhancing erosion, ponding, or disturbance (Crain et al. 2009). Marshes with a small tidal range and inadequate sediment supply are the most vulnerable to SLR (Fagherazzi et al. 2019).

SLR is by far the largest climate-related threat to salt marshes. The Intergovernmental Panel on Climate Change (IPCC) predicts with medium confidence a global SLR of 0.26–0.98 m over current levels by 2100 (Church et al. 2013). Under the most optimistic IPCC emissions pathway, 60% of the marshes studied will be accreting less than the rate of SLR by 2100. Nicholls et al. (1999) predicted that a 1 m SLR will eliminate 46% of the world's coastal wetlands. For New Jersey, the STAP report (https://climatechange.rutgers.edu/resources/climate-change-and-new-jersey/nj-sea-level-rise-reports) predicted under a high-emissions scenario, coastal areas are likely to see SLR of 1.5–3.5 feet (0.45–1.07 m) between 2000–2070, and 2.3–6.3 feet (0.70–1.92 m) between 2000–2100. Under a moderate-emissions scenario, coastal areas are likely to see SLR of 1.4–3.1 feet (1.43–0.94 m) between 2000 and 2070, and 2.0–5.2 feet (0.61–1.58 m) between 2000 and 2100. The loss of marshes due to rising sea levels is now a critical issue for the present, rather than a future problem.

The rate of SLR is not identical everywhere, and some marshes are better able than others to gain elevation at a rate that matches the SLR. Crosby et al. (2016) synthesized worldwide data and found that many salt marshes did not keep pace with SLR. Marshes in regions experiencing higher local SLR or greater subsidence were less likely to keep pace with the SLR. The issue of SLR in the mid-Atlantic is especially acute, as the local SLR is significantly greater than what is observed globally. This heightened vulnerability is due to apparent subsidence effects caused by sediment compaction, glacio-isostatic adjustment, groundwater overdraft, as well as changes in ocean currents and gravitational effects resulting from shifting masses between land and sea ice and ocean water (Sun et al. 1999; Kopp et al. 2019). Over the last century, sea level in New Jersey has risen 0.45 m versus 0.18 m globally, 250% the global average (Kopp et al. 2019). Over the next century, the rates of SLR in New Jersey are expected to accelerate due to climate change. Median predictions for 2000–2100 range from a SLR of 0.85–1.19 m with the lower prediction resulting from a low emissions scenario, and the higher prediction from a high emissions scenario (Kopp et al. 2019). The rate for 2019–2020 is 5–6 mm year^{-1} (R.E. Kopp, personal communication, 2020).

An assessment of marsh resilience produced by Raposa et al. (2016) found that Pacific marshes were more resilient than Atlantic marshes, with the least resilient marshes in southern New England (none of the marshes studied were in New Jersey). Assessment scores can inform management strategies, with moderate scores calling for enhancing resilience and low scores suggesting seeking opportunities for marsh migration.

Borchert et al. (2018) concluded that migration corridors are particularly important in urbanized estuaries, where low-lying coastal development prevents marshes from moving inland. Migration is also difficult or not feasible in areas with steep slopes, where increased elevation quickly exceeds the range of tidal flooding (Brinson et al. 1995).

The pervasive loss of tidal marshes as an indication of SLR in the New York–New Jersey area was first noted in the early 2000s, with the recognition that Jamaica Bay, NY, had lost a significant portion of its wetlands and marsh islands (Hartig et al. 2002). In Jamaica Bay, 48% of tidal wetlands were found to have disappeared between the 1920s and the 1990s, predominantly due to SLR, which produced edge slumping, tidal channel widening, expansion and coalescence of tidal marsh pools and bare spots, and general fragmentation (Hartig et al. 2002). An analysis of satellite imagery published that same year also showed a high level of deterioration in tidal marshes in Delaware Bay (Kearney et al. 2002). A large percentage of marshlands on the Delaware Bayshore exhibited degraded conditions (62% degraded in New Jersey; 45% in Delaware), and the level of deterioration increased between the 1980s and the 1990s (Kearney et al. 2002). Similar analyses have subsequently reported widespread deterioration of tidal marshes throughout the mid-Atlantic and southern New England (Smith 2009; Cameron Engineering & Associates 2015; Watson et al. 2017; Krause et al. 2020). During a study of Connecticut's and New York's marshes, Hill and Annisfeld (2015) found that declining relative elevation led to increased tidal flooding, particularly in the high marsh. As flooding increased, organic matter accumulation accelerated in all marshes, but mineral deposition was only observed in areas of short-form *Spartina alterniflora*. Pettit et al. (2018) found that urban development in Jamaica Bay greatly reduced inputs of mineral sediment, but an increase in organic matter allowed vertical accumulation to outpace sea level for a time. However, the reduced mineral content caused structural weakness and edge failure. They concluded that marsh survival requires mineral sediment addition to the marsh surface, subsurface channels, and borrow pits.

Climate change is expected to increase the frequency and intensity of extreme rainfall events (Michener et al. 1997), including tropical storms, which can have major effects on salt marshes. Physical damage caused by storms includes breaking above-ground tissue, lateral erosion of marshes, and, in some cases, denudation of vegetation (Hanley et al. 2020). Fragmented or degraded marshes are more vulnerable to disturbances and less resilient to extreme events. However, storms can also deposit new sediments on top of marshes, which may enable them to better keep up with SLR, provided burial does not kill the vegetation (Schuerch et al. 2019). Large storm surges may deposit much of the sediment upland of marshes, and the weight of a storm surge can compress the marsh, negating the benefits of added sediments (Cahoon et al. 2019).

In this overview, we examine the status and trends of marshes in New Jersey and consider four possibilities for mitigating climate change effects to allow for marsh preservation: migration pathways, management of *Phragmites australis*, sediment manipulation, and living shorelines. Migration pathways present a way to open up space behind marshes to allow them to migrate in areas where there is inadequate open space due to development. The reed *P. australis* is an invasive plant that is an "ecosystem engineer", which can have twice as much biomass and carbon storage potential as other marsh plants (Davidson et al. 2018). It also stores more nitrogen (reduces eutrophication) and increases marsh elevation. Rooth and Stevenson (2000) found greater rates of both mineral and organic sediment trapping in *P. australis* due to greater litter production.

Sediment can be added to the marsh surface to increase surface elevation. Thin layer deposition/placement involves spraying sediments onto the marsh surface (Ford et al. 1999), and this approach has been piloted in New Jersey. Plant recovery rates after deposition were variable. Dredged material can be placed in the water at the seaward edges of marshes to provide additional sediment via tides and waves. Another way of altering sediment is to dig small channels (termed runnels) to drain areas of ponding in marshes to promote drainage and encourage revegetation (Wigand et al. 2017). When marshes are eroded at the edge, “living shorelines” (Bilkovic et al. 2017) in the form of oyster reefs, rocks, coir logs, and other materials can be placed at the edge to restore a more gradual vegetated slope and prevent further erosion. Living shorelines can be more resilient to hurricanes than hard edges or natural marshes (Smith et al. 2018). Some living shoreline projects have been installed in New Jersey. Since these are relatively new, their continued effectiveness in the face of SLR remains to be seen.

In this study, we examined the status of New Jersey marshes and analyzed the potential of each of these four remedies to mitigate the detrimental effects of SLR on New Jersey marshes.

Status of New Jersey’s marshes

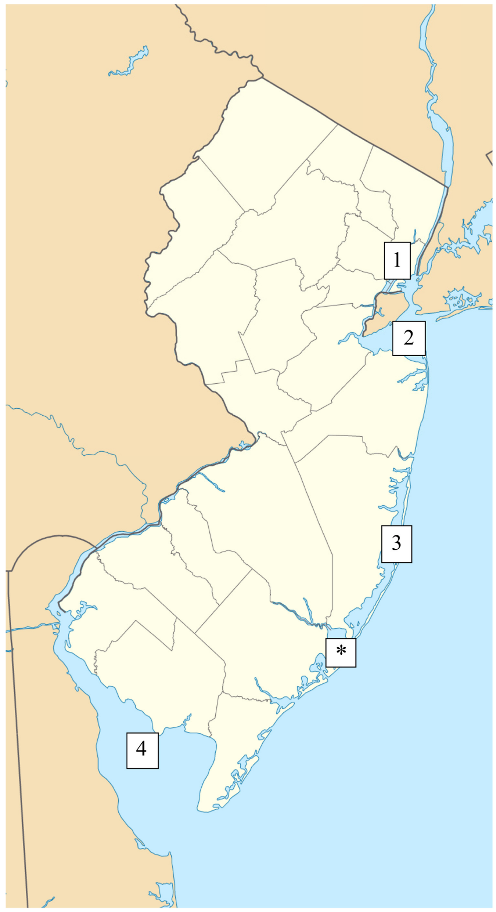
This review considers two indicators of tidal marsh stability and vulnerability to SLR: first, analyses of marsh habitat extent in New Jersey, and second, analyses of marsh accretion and elevation trends. SLR contributes to habitat changes apparent in tidal marshes through several modes: (1) edge erosion, (2) widening and expansion of tidal channels, (3) formation and expansion of interior ponds, and (4) habitat changes along upland transition zones. Increased water levels, in concert with increasing storm frequency and intensity, subject marsh edges to attack by waves, in some cases resulting in extreme erosion rates of 5–20 m year^{-1} (Elsey-Quirk et al. 2019). Because the volume of water moving through tidal channels is in equilibrium with the tidal channel volume (D’Alpaos et al. 2010), when the sea level rises, lateral erosion and headward extension of tidal channel networks occur to accommodate this increased volume (Hughes et al. 2009), which reduces the areal extent of marsh vegetation. Marsh groundwater tables also increase with SLR, and where these water tables are at or above the marsh surface, marsh vegetation is typically not able to survive, and is replaced by bare mud or open water (Andres et al. 2019). Finally, other signs of habitat change related to SLR in marshes are changes at the marsh–upland transition. Over time, maritime forests abutting marshes experience a lack of vegetative regeneration, tree thinning, and the death of mature trees due to intrusion of salt water or a heightened water table (Fagherazzi et al. 2019), permitting marshes to expand into former-upland areas. In developed areas, such as farms and residential developments, marsh vegetation may encroach on fields and lawns, and soil salinization may occur.

Although changes in the areal extent of marsh habitats provide a strong indicator of coastal wetland status relative to SLR, marsh accretion and elevation change are important indicators of marsh status. If the SLR exceeds the capacity of the marsh to accrete vertically, the marsh will drown, and if the SLR is matched by accretion, the marsh will persist (Orson et al. 1985). The installation and monitoring of surface-elevation tables (SETs) provides a non-destructive measure of sediment elevation change in wetlands relative to a fixed sub-surface elevation datum, and typically includes the establishment of feldspar marker beds (MBs), which measure sediment accretion (Lynch et al. 2015). SETs and MBs measure rates of vertical accretion and elevation change and allow partitioning of elevation change into surface sediment deposition versus the elevation produced through belowground biomass production or lost through subsurface consolidation (Lynch et al. 2015).

Marsh habitat changes relative to SLR

Systematic analyses of marsh habitat change relative to SLR in New Jersey (Fig. 1) have focused primarily on marshes in Delaware and Barnegat bays, including analysis of marsh edge retreat and habitat change. Less well-studied are the back-barrier marshes south of

Fig. 1. Map of New Jersey with locations of the Meadowlands (1), Raritan Bay (2), Barnegat Bay (3), and Delaware Bay (4). * indicates Great Bay (see Fig. 2). Map from Wikipedia.



Great Harbor (e.g., Stone Harbor, Ludlam Bay, Great Egg Harbor Bay) on the Atlantic coast of New Jersey and Raritan Bay. Marsh habitat changes have been tracked in the Meadowlands of the New York–New Jersey Harbor Estuary, where long-term losses are related to development, while mitigation and restoration efforts have resulted in some habitat increases (Stinnette et al. 2018). Other parts of the New York–New Jersey Harbor Estuary have not been well studied.

The Meadowlands

In 1889, there were 20 045 acres of tidal wetlands in the Meadowlands, but by the second half of the 20th century, wetland acreage declined due to development pressures (Tiner et al. 2002). By 2019, there were about 8400 wetland acres remaining (supplementary data, Fig S1[1]), of which 3544 acres were conserved (www.njsea.com/master-plan-2020). Although SLR undoubtedly affects the marshes of the Meadowlands, estimating losses related to SLR is difficult because of ongoing development pressures that continue to reduce wetland acreage and ongoing restoration efforts that have increased wetland acreage.

[1]Supplementary data are available with the article at https://doi.org/10.1139/anc-2020-0020.

The New Jersey Sports and Exposition Authority is currently digitizing the wetlands within the Meadowlands District using high-resolution drone imagery to establish a baseline to better track future wetland changes.

Other parts of the New York–New Jersey Harbor Estuary

Four percent of wetlands in the New York–New Jersey Harbor Estuary were lost between 2002 and 2012 (Stinnette et al. 2018). While most losses were forested wetlands, 186 acres of tidal emergent wetlands were lost (Stinnette et al. 2018). About 68% of this loss was due to urban development, while about 15% reflected conversion to open water, which includes salt marsh conversion to unvegetated underwater areas. Between 1986 and 2015, Raritan Bay, including Staten Island and the Kills, showed virtually no change in the acreage of tidal marshes (R.G. Lathrop, personal communication). In the New Jersey portion of the bay, from 1977 to 2010, there was more gain than loss (R.G. Lathrop, personal communication). Published data are lacking and further studies are required.

Barnegat Bay

Analysis of Barnegat Bay wetland change was reported by Lathrop and Bognar (2001) for 1972–1995, using data from the Coastal Change Analysis Program (CCAP) and the Barnegat Bay Partnership for 2007–2012 (Table S1[1]). Wetland change analyses suggest that 11.9% of the tidal wetlands were lost between 1972 and 2012. The highest loss rate was between 1995 and 2007, and the recent loss rate was higher for Barnegat Bay than Delaware Bay, at 2.9% per decade for Barnegat Bay versus 1.1%–1.9% per decade for Delaware Bay (Watson 2019). A higher resolution analysis completed for three marshes in Barnegat Bay found a loss rate of 9.7% between 1975 and 2015. While these values roughly mirror figures from analysis of CCAP cover data, this higher resolution analysis showed relatively large gains (11.3%) and losses (18.4%), with internal marsh fragmentation contributing to losses and upland migration contributing to gains (Watson 2019). Shoreline erosion rates were also determined for Barnegat Bay between 1930 and 2013, through the digitization of more than 100 km of marsh shorelines, using aerial photographs from 1930, 1995, 2002, 2007, 2010, and 2013 (Fig. 2). The median shoreline erosion rates were ~0.5 m year^{-1}, and shoreline erosion has not accelerated over the past decade (Leonardi et al. 2016).

Delaware Bay

By most indicators of SLR vulnerability, Delaware Bay wetlands would appear to be relatively resilient to climate change because of their large tidal and growth range, low slope, adequate sediment supply, and open space to migrate into. However, most analyses suggest that Delaware Bay marshes are eroding and converting to open water (Fig. 3; Table S2[1]). It appears that marsh loss rates in Delaware Bay are 1.1%–1.9% per decade, considerably less than the 4.4% found for coastal New York and Rhode Island (Cameron Engineering & Associates 2015; Watson et al. 2017). The difference between these locations is the significant percentage of marsh area gained through migration into low-lying maritime forests, which was not found in New York or southern New England (Smith 2009; Cameron Engineering and Associates 2015; Watson et al. 2017). The position on the Atlantic coastal plain can confer some adaptive capacity for preserving marshes, depending on the adjacent land use. In addition to marsh loss rates, an analysis of erosion rates was completed for Delaware Bay in 1986, which calculated an average erosion rate of 3.2 m year^{-1} for the entire Delaware Bayshore between 1940 and 1978 (Phillips 1986), with average values of about 2–5 m year^{-1} along different segments.

Fig. 2. Loss of marshes at Great Bay, NJ (* on Fig. 1) from 1995 to 2015 Photo courtesy of Bulletin of the New Jersey Academy of Sciences (Kennish et al. 2016).

Fig. 3. Loss of wetlands at Gandy's Beach, Delaware Bay, NJ (39.2828, –75.2455) from 1930 to 2015. The red dots mark the same location to illustrate the extent of loss. Aerials created by LeeAnn Haaf courtesy of the New Jersey Geographic Information Network (ArcGIS WMS) (Demberger et al. 2017).

Table 1. Accretion rate, subsidence rate, and elevation change of tidal marshes in the Meadowlands over 11 years.

Location	Vegetation	Salinity (PSU)	AR (mm $year^{-1}$)	SR (mm $year^{-1}$)	EC (mm $year^{-1}$)
Lyndhurst	*S. patens*	11.5	3.61	0.58	3.03
Riverbend Patens	*S. patens*	11.5	5.36	0.36	5.00
Riverbend Mixed	*P. australis; S. patens*	15.5	5.21	1.11	4.10
Saw Mill	*S. alterniflora*	13.5	7.80	3.60	4.20
Secaucus	*S. alterniflora*	7.5	5.52	1.97	3.56
Walden Swamp	*P. australis*	4.5	5.45	−6.30	11.75
Eight Day Swamp	*P. australis*	4.0	6.45	−1.72	8.17

Note: AR, accretion rate; SR, subsidence rate; EC, elevation change. Data from the MERI (2019).

Marsh elevation and vertical accretion trends

Over the past decade, SETs have been established across New Jersey to determine whether marsh elevation changes have been keeping pace with SLR. Where rates of SLR (or high-water level rise) exceed rates of marsh elevation change, elevations relative to sea level decline, and eventually marsh drowning is predicted (Orson et al. 1985). Conversely, when the rate of vertical elevation change is greater than that of SLR, marsh persistence is predicted (Orson et al. 1985). In New Jersey, SET establishment has occurred in association with the Mid-Atlantic Coastal Wetland Assessment (MACWA, a partnership of the National Estuary programs in Delaware and Barnegat Bay, with Rutgers University and the Academy of Natural Sciences), the National Wildlife Refuges (Forsythe NWR, Cape May NWR, Sapawna Meadows NWR), the Meadowlands Environmental Research Institute (now New Jersey Sports and Exposition Authority), and National Park Service.

The Meadowlands

SETs and feldspar MBs were set up at seven locations in 2008 across the Meadowlands: Lyndhurst Riverside (LR) high marsh, Riverbend Patens (RBP) high marsh, Riverbend Mixed (RBM) high marsh, Saw Mill (SM) low marsh, Secaucus High School (SHS) low marsh, Walden Swamp (WS) high marsh, and Eight Day Swamp (EDS) high marsh (MERI 2019; Artigas et al. 2021). The first five sites showed accretion rates ranging from 3.61 to 7.8 mm yr^{-1}, but they all subsided at rates ranging from 0.36 to 3.60 mm $year^{-1}$ (Table 1). The net elevation change ranged from +3.03 (LR) to +5.0 (RBP) mm $year^{-1}$. The average subsidence at these sites was 1.5 mm $year^{-1}$, and the average accretion was 5.5 mm $year^{-1}$, so that the average elevation change was 4.0 mm $year^{-1}$. The low marsh showed greater accretion rates (6.6 mm $year^{-1}$) than high marsh (4.7 mm $year^{-1}$), but subsidence rates were also higher in the low marsh because of greater decomposition and compaction. The latter two sites were dominated by *P. australis* (henceforth *Phragmites*) and did not subside. WS had accretion of 5.45 mm $year^{-1}$ and subsidence of −6.3 mm $year^{-1}$, for a net accretion of 11.75 mm $year^{-1}$. EDS had accretion of 6.45 mm $year^{-1}$ and subsidence of −1.72 mm $year^{-1}$ for a net accretion of 8.17 mm yr^{-1}. Based on a current SLR rate of 5–6 mm $year^{-1}$, only the oligohaline *Phragmites* sites keep pace with SLR. It should be noted that of the first five sites, Lyndhurst and Secaucus had previously been *P. australis* before they were "restored" and converted to *S. alterniflora*.

Raritan Bay

Short-term accretion rates in Raritan Bay were measured using sediment plates from May 2018 to November 2019 in four tidal marshes: two *S. alterniflora* marshes and two *P. australis* marshes (J. Quispe, personal communication). Average accretion in the *P. australis* after 18 months was −3.81 mm $year^{-1}$; while average accretion in *S. alterniflora* was

Table 2. Accretion of tidal marshes along the Raritan River over 18 months.

Raritan site	Marsh plot	Vegetation	TA 12 (mm)	TA 18 (mm)
1	1	*P. australis*	−0.36	−1.88
	2	*P. australis*	−3.68	−6.44
	3	*P. australis*	−7.92	−8.84
2	4	*P. australiss*	2.84	—
	5	*P. australis*	0.56	—
	6	*P. australis*	—	—
3	7	*S. alterniflora*	9.40	—
	8	*S. alterniflora*	−5.28	−13.52
	9	*S. alterniflora*	−17.50	−15.65
4	10	*S. alterniflora*	3.76	−6.36
	11	*S. alterniflora*	−24.36	−14.28
	12	*S. alterniflora*	−5.08	−0.80

Note: TA 12, average total accretion at 12 months; TA 18, average total accretion at 18 months. Data from J. Quispe, personal communication, 2019.

−6.53 mm year^{-1}. Although a few plots were accreting over the first 12 months, none of the sites showed overall accretion after 18 months (Table 2). Although data from these sediment plates are not directly comparable to data collected from the other sites, none of the sites appeared to be experiencing sediment accretion and thus could not have elevation gains at rates exceeding SLR. SETs were recently installed in Raritan Bay by Rutgers, and data will become available over the coming years to reveal marsh elevation trends.

Barnegat Bay

Marsh elevation change reported by the U.S. Fish and Wildlife Service (USFWS) ranged from –20 to +154 mm year^{-1} (Table S3[1]), while values reported by the Barnegat Bay Partnership ranged from −2 to +5.8 mm year^{-1}. Several values from the Edwin B. Forsythe National Wildlife Refuge (USFWS) appear to be outliers, and they were identified as such in a USFWS analysis (Ladin and Shriver 2017) and are not credible values relative to those reported in the literature. The anomalous values from Forsythe may reflect anthropogenic disturbances. SETs monitored by the Barnegat Bay Partnership, which were disturbed by pond construction, suggest rapid sediment accretion associated with sediment sidecasts and post-deposition subsidence (Haaf et al. 2021). Of the three sites monitored by the Barnegat Bay Partnership, SETs in Island Beach experienced no net elevation gains, while SETs at other locations increased at rates slightly below SLR rates (Table 3). The Jacques Cousteau National Estuarine Research Reserve has installed SETs where Barnegat Bay borders the Great Bay, but no data are available yet.

Delaware Bay

Marsh elevation change in Delaware Bay ranges from 0.7 mm year^{-1} to 6.9 mm year^{-1} with median values of 4.3 mm year^{-1}, less than SLR. High sediment accretion was found in the tidal freshwater site; other SETs in tidal freshwater stations (in Delaware and Philadelphia) also showed high sediment accumulation rates (Table 4) (Haaf et al. 2021). Two marshes showed high variability in elevation change, while two other marshes showed similar rates of elevation change across sites. There were no strong trends or associations between the vegetation, salinity, or elevation change rates.

Thus, many marshes in New Jersey are losing acreage and are elevating at a rate lower than the SLR. There is a great need for further study in Raritan Bay and other parts of Harbor Estuary.

Table 3. Accretion rate, subsidence rate, and elevation change of marshes in Barnegat Bay. Surface-elevation tables established and monitored for 5–7 years as part of the Mid-Atlantic Coastal Wetland Assessment (MACWA).

Location	Vegetation	Salinity (PSU)	AR (mm year^{-1})	SR (mm year^{-1})	EC (mm year^{-1})
Reedy Creek 1	*S. alterniflora*	20	4.75	0.46	5.21
Reedy Creek 2	*S. alterniflora*	20	6.72	−0.95	5.77
Reedy Creek 3	*S. alterniflora*	20	4.52	−2.28	2.24
Island Beach 1	*S. alterniflora*	27	2.91	−2.28	0.62
Island Beach 2	*S. alterniflora*	27	3.63	−5.60	−1.96
Island Beach 3	*S. alterniflora*	27	2.57	−1.24	1.32
Horse Point 1	*S. alterniflora*	26	5.87	−1.95	3.92
Horse Point 2	*S. alterniflora*	26	5.86	−1.47	4.40
Horse Point 3	*S. alterniflora*	26	5.39	−1.22	4.17

Note: AR, accretion rate; SR, subsidence rate; EC, elevation change. Data from Haaf et al. (2021).

Table 4. Accretion rate, subsidence rate, and elevation change of marshes in Delaware Bay. Surface-elevation tables established and monitored for 5–7 years as part of the Mid-Atlantic Coastal Wetland Assessment (MACWA).

Location	Vegetation	Salinity (PSU)	AR (mm year^{-1})	SR (mm year^{-1})	EC (mm year^{-1})
Crosswicks Creek 1	*Zizania aquatica*	0.10	13.5	−9.41	4.11
Crosswicks Creek 2	*Peltandra virginica*	0.10	8.35	−3.83	4.52
Crosswicks Creek 3	*Nuphar advena*	0.10	9.98	−6.59	3.40
Dividing Creek 1	*S. alterniflora*	17.00	8.16	−3.03	5.13
Dividing Creek 2	*S. patens*; *D. spicata*	17.00	10.10	−3.82	6.28
Dividing Creek 3	—	17.00	6.01	0.89	6.89
Maurice 1	*S. alterniflora*	11.00	7.70	−6.54	1.16
Maurice 2	*S. patens*; *D. spicata*	11.00	3.72	0.05	3.77
Maurice 3	—	11.00	6.81	−1.60	5.21
Dennis Creek 1	*S. alterniflora*	16.00	6.99	0.99	5.85
Dennis Creek 2	*S. patens*; *D. spicata*	16.00	3.78	−4.12	0.74
Dennis Creek 3	—	16.00	5.06	−3.40	1.46

Note: AR, accretion rate; SR, subsidence rate; EC, elevation change. Data from Haaf et al. (2021).

Potential solutions

Wetlands' pathway protection

Salt marshes can migrate inland as sea levels rise if there is an open space landward of the marsh. If there is development, roads, or other barriers, the low marsh can move into the high marsh, but high marsh vegetation has nowhere to go. This problem of "coastal squeeze" may be mitigated by developing pathways and open areas for the marsh to migrate into. Areas that are undeveloped and contiguous with existing wetlands are most likely to become pathways, which can be facilitated by removing berms, resizing bridges and culverts, adjusting the slope of the land, and preventing these areas from developing (Maryland Sea Grant 2019).

Although in many ways supporting coastal retreat may be a simple and cost-effective way to preserve coastal wetlands, there are numerous obstacles in creating and preserving wetland pathways, including unfamiliarity with potential pathways, issues with science/policy translation, competing priorities, insufficient budget, lack of coordination on land acquisition, lack of guidance on land management strategies, and lack of public awareness (Blair Environmental Consulting 2018). Local communities can play a critical role in raising public awareness of these issues. There may be resistance to preserving parcels of land that could be developed. Community engagement is essential for residents to "buy in".

This can be challenging, as there may be inadequate economic or political resources and conflicting cultural norms. A long-term vision of how coastal communities and wetlands can benefit each other is important (Calvin et al. 2018). Education on the benefits of marshes and mechanisms to incorporate marsh migration into long-term plans are critical for local communities and politicians (Maryland Sea Grant 2019).

Prioritization of sites for migration pathways depends on ecological and geophysical, as well as political and social factors. Planners should provide resources to local institutions and community groups to plan and implement projects. Plans for wetland migration should ensure that the benefits and costs are equally distributed (Blair Environmental Consulting 2018). Since it is important to get the support of the community, planners should work with existing organizations, such as community or home-owner meetings, partnering with other community organizations, and social media. Planners, ecologists, and engineers must be open to learning from the community and adapting the project accordingly (Calvin et al. 2018; Maryland Sea Grant 2019).

Policy, regulation, and planning

Planning vehicles might be used to institutionalize pathway protection, such as county master plans, the Coastal Zone Management Program, the New Jersey State Coastal Resilience Plans, Blue Acres, greenhouse gas initiatives, and local waterfront revitalization programs. If restoring spaces for wetland function were included in county hazard mitigation plans, they would be eligible for Federal Emergency Management Agency (FEMA) funding (Calvin et al. 2018). Rolling easements decrease or end continued use of coastal properties as sea level rises. The "Building Ecological Solutions to Coastal Community Hazards" (2017) report for New Jersey advanced the idea of using rolling easements to allow for marsh migration. Examples include: (1) local zoning or state regulations that prohibit building shoreline protection structures; (2) easements that allow continued public access as a beach migrates; (3) future interests that transfer ownership as sea level rises to a predetermined level; (4) migrating property lines, which move inland as the shore erodes; and (5) transferable development rights giving upland acreage to those who yield land to SLR.

Rolling easements should appeal to property owners because they can keep their land as long as it is cost-effective. Economic modeling can be used in combination with SLR modeling to guide the protection of undeveloped lands that maximize the tax base and reduce hazards (Gardner and Johnston 2020).

Status of New Jersey's efforts

New Jersey Adapt (http://www.njadapt.org/) is an online tool that identifies areas where salt marsh migration is impeded or unimpeded based on SLR scenarios (CRSSA 2020). The Nature Conservancy (TNC) tool can also be utilized (https://www.njmap2.com/blueprint/). The New Jersey Blue Acres Program buys out lands and developed properties and manages vegetation; restoration areas are designed to embrace rather than resist encroachment of wetland species as sea level rises. Until now, this program has only bought flood damaged properties, but there is an opportunity for buyouts in migration pathways (Blair Environmental Consulting 2018).

The future of tidal marshes will be influenced by landowners whose decisions could promote or block marsh migration. Conservation easements are unlikely to be sufficient to mitigate losses from SLR, but other strategies, such as restrictive covenants and future interest agreements, may be more likely to be adopted by landowners (Balcom 2019). However, this has not been proven in practice. Failure to factor human behavior into planning can lead to an overly optimistic view of success. Strategies to increase participation in conservation agreements, such as increasing understanding of climate change and ecosystem services provided by marshes, had weak effects on landowner attitudes. In a study of landowners in Connecticut, those with stronger beliefs about increased flooding or marsh migration indicated a greater inclination to build seawalls, potentially leading to greater loss of natural coastline habitats. Overall, 22% of landowners said they were likely to harden their shoreline within 20 years (Balcom 2019).

Phragmites management

The common reed (*Phragmites australis)* is a native wetland plant in North America, although its expansion has been linked to the invasion of a non-native genotype (Saltonstall 2002). It is one of the most aggressively managed plants in the United States (Rogalski and Skelly2012). *Phragmites* is invasive in New Jersey marshes and has been found to reduce plant and bird diversity (Chambers et al. 1999). It is frequently removed with herbicides and replaced with native *S. alterniflora* in restoration projects, which can lower the marsh surface (Rooth and Stevenson 2000). Many restoration practitioners dislike *Phragmites* and want them to be removed. Nevertheless, *Phragmites* does many beneficial things that are generally unappreciated and may provide important benefits in the context of SLR.

Although *Phragmites* has been traditionally viewed to negatively impact wildlife (Weinstein and Balletto 1999), empirical studies predominantly highlight species trade-offs. Many studies have indicated that fish and invertebrates in the tidal creeks of *Phragmites* marshes are approximately as abundant and diverse as those in *Spartina* marshes (Hanson et al. 2002; Posey et al. 2003; Fell et al. 1998; Meyer et al. 2001; Yuhas et al. 2005). Investigators have found that fish assemblages are similar in both types of marshes or less dense in *Phragmites*. However, the killifish, *Fundulus heteroclitus*, which plays an important role in coastal areas, is clearly reduced in *Phragmites* marshes (Able and Hagan 2000). Plant detritus is a food source for many animals, and detritus from *Spartina* and *Phragmites* provide equivalent nutrition for detritivores (Weis et al. 2002). *Phragmites* detritus enters estuarine food webs to support larger fish (Wainright et al. 2000). Bird use of *Phragmites* varies depending on the species, location, stand density and extent, and other plant species. Some birds do not prefer *Phragmites*, but some species prefer it (Chambers et al. 1999; Kiviat 2013). Studies comparing the density or species diversity in *Phragmites* versus other plants have shown variable results (Benoit and Askins 1999). *Phragmites* provides habitat for many other terrestrial organisms, such as mammals and insects (Rogalski and Skelly 2012; Kiviat 2013).

Some of the ecosystem services provided by *Phragmites* have become much more important with climate change and SLR. It is better ~~to sequester~~ pollutants, including nitrogen (Windham and Ehrenfeld 2003, Windham and Meyerson 2003), metals (Windham et al. 2003), and carbon dioxide (Schäfer et al. 2014; Duman and Schäfer 2018; Davidson et al. 2018), ~~better~~ than native plants. By removing nitrogen *Phragmites* decreases algal blooms and eutrophication, and by removing CO_2, it reduces climate change. Although low-salinity *Phragmites* marches also release methane, a potent greenhouse gas (Mozdzer and Megonigal 2013), some studies suggest that it acts as a greenhouse gas sink due to high CO_2 assimilation (Martin and Moseman-Valtierra 2015), while other studies suggest increased emissions (Pendleton et al. 2012).

Phragmites traps more sediments and forms dense rhizomes, which enable a marsh to elevate more rapidly than *Spartina* marshes (Rooth and Stevenson 2000). It builds and stabilizes marsh soils and protects marshes from erosion associated with SLR. At this critical time, a plant that enables a marsh to elevate more rapidly should be valued and not removed. *Phragmites* height and dense growth also provide better protection from storm surges and wind (Sheng et al. 2021).

Management of *Phragmites* in the mid-Atlantic continues to lead to the removal of plants, often with the use of toxic herbicides, but relatively little attention has been paid to the plant communities that follow (Hazelton et al. 2014). This management is very expensive and often does not yield the desired results (Martin and Blossey 2013). The ecosystem services provided by this plant should be weighed against the desired outcomes and costs (financial and environmental) of removal.

Twenty years ago, Rooth and Stevenson (2000) wrote: "Thus *P. australis* may provide resource managers with a strategy of combating sea-level rise, and current control measures fail to take this into consideration." This statement remains true. We should shift from attempted eradication everywhere, to the modification of *Phragmites* stands to create habitat for particular species while retaining its other ecosystem services, especially marsh elevation. This requires an approach tailored to individual areas and site-specific management goals and will require education, a change in the attitudes of major players, and a paradigm shift in how tidal wetlands are managed.

Sediment manipulation

One approach to increase coastal wetland elevation where existing sediment supplies are inadequate is through the repurposing of dredged sediment (Ganju et al. 2017). The addition of sediment can elevate coastal marshes to keep pace with SLR (Morris et al. 2002), reverse losses of salt marsh (Woodhouse et al. 1972), and increase the resiliency of systems that have been degraded by hydrological alterations (Smith et al. 2018). According to Berkowitz et al. (2019), the best way to place dredged sediment on coastal marshes is to add the material in layers up to 30 cm thick to provide "elevation capital" to shallow intertidal areas. This has been called thin-layer deposition (Ford et al. 1999), thin-layer placement (USACE 2019), or sediment enrichment (Slocum et al. 2005). Studies back to the 1980s show a consistent positive marsh response from 2 to 10 years following thin-layer placement (Ray 2007). Another term, marsh nourishment, can refer to either the direct placement of a thin-layer of sediment through spray or hydraulic dredging or from the "spilling" of a thin-layer of sediment over marsh that is adjacent to an uncontained restoration project (LaPeyre et al. 2006).

Engineering and construction approach

Sediment placement in these projects is <30 cm thick, so that plants and invertebrates can readily grow or migrate through the material. Deposits that are too thick could smother and kill existing vegetation, requiring revegetation with new plant material (Ford et al. 1999); those that are too thin are probably not cost-effective because the process will need to be repeated frequently. Cahoon and Cowan (1987, 1988) reported that recolonization by typical salt marsh plant species was underway 14 months after the application of up to 40 cm of dredged sediment. Within 4 years, the sites were very similar to the reference areas (Ray 2007). Mohan et al. (2016) noted that it is ideal to keep the placement between 15 and 30 cm.

Sediment placement typically targets elevations from mean high water to mean higher high water as a biological target elevation. In preparation, the area is usually divided into cells to contain the dredged material, which will be filled to varying thicknesses depending on their baseline elevations. These cells are maintained by structures (e.g., hay bales, coir logs, and geotextiles), which also protect creeks from being filled with sediment. Variability has been observed in the amount of material distributed within each cell due to local topographic features and variability in settling as the material moves away from the discharge pipe. Containment structures are removed after the water has drained and sediments have consolidated, although some have advocated their retention to promote seedling establishment or wave protection (Allen and Shirley 1988). Vegetation re-establishment can occur via planting, mixing seed in with dredge material as it is being sprayed, aerial seeding, or allowing the area to vegetate naturally (Mohan et al. 2016). Natural recruitment is often recommended, although at least 3 years might be required to achieve revegetation (Welp 2015).

Dredged material may be deposited using various methods depending on the distance of the site from the dredging operation, conditions at the placement site (substrate, adjacent depths, wave, wind, and current conditions), and type of material being placed. Approaches to placing sediments include hydraulic placement, spraying, slurry placement, or land-based methods. In traditional hydraulic pipeline placement, dredged material is pumped directly on site to create marsh from open water or to elevate the marsh. Spray-dredge placement, thin-layer placement, and rainbow placement all refer to placing sediments via high-pressure spraying, in which a nozzle is attached to the pipeline and the material is sprayed up to 45–60 m across the marsh surface. Some studies have found that sediment can be sprayed onto living plants without long-term detrimental effects if the sediment thickness is <15 cm (Cahoon and Cowan 1988).

Slurry placement is relatively new. Sediment can flow onto existing marsh with a high fluid to sediment ratio, which facilitates the sediments flowing over long distances (Mendelssohn and Kuhn 2003); this works best with fine-grained sediments. The final category involves placing material from a land-based operation nearshore. Dredged sediments are sometimes moved to create wetlands using a backhoe or "scrape down" methods. The material is then reworked nearshore to obtain the biological target elevation.

Monitoring thin layer projects

Monitoring a thin layer deposition/placement project is critical for success (Mohan et al. 2016). Detailed and regularly scheduled monitoring can identify environmental changes in and around the project area and determine if issues detrimental to success can be corrected. The decision to correct should be addressed through adaptive management, an iterative process of decision-making aimed at reducing uncertainty over time. Depending on project goals, monitoring may include topography, elevation change, water levels, vegetation, epifaunal macroinvertebrates, benthic infauna, nekton, and birds (TNC 2015), and monitoring is often conducted at suitable control or reference sites to better understand the benefits of the project relative to impacted and healthy marshes. Ideally, long-term data should be collected.

New Jersey case studies

Thin layer deposition/placement projects have been successfully implemented across the country in a variety of settings (Ray 2007; Welp 2015; Mohan et al. 2016). In New Jersey, the U.S. Army Corps of Engineers (USACE), New Jersey Department of Transportation (NJ DOT), New Jersey Department of Environmental Protection (NJ DEP), GreenVest, and the National Fish and Wildlife Foundation (NFWF) completed three pilot/demonstration projects in New Jersey at Ring Island, Avalon, and Fortescue from 2014 to 2015. Their objectives were to (1) increase and maintain the optimal tidal elevation for native salt marsh species, (2) increase the cover and health of native salt marsh vegetation, and (3) return all other metrics to baseline conditions unless they are expected to change due to habitat conversion (TNC and NJDEP 2020).

The Ring Island project in Middle Township was completed in 2014. Sand dredged from the New Jersey Intercostal Waterway was placed on a vegetated, short-form *S. alterniflora* salt marsh to create two 1-acre pilot projects. The sand was sprayed from a 14 inch pipe onto the marsh, as it was hydraulically dredged from the channel. Because the sand fell out of the solution quickly, no containment was needed, but the sand piled up unevenly.

In the winters of 2014–2015 and 2015–2016, pilot projects were established at Avalon using fine-grained sediments hydraulically dredged from the New Jersey Intercostal Waterway. The first was a 7-acre test to explore pumping into expanding pools, and the second was a 50-acre project. The sediment was pumped up to a mile from the channel and spread far into the marsh until it hit the containment. Water in the dredged slurry was contained within a ring of coconut coir logs, and pumping had to stop when the site was dewatered. Finer-grained sediment led to an even placement. (Moritzen et al. 2017). In winter 2015, approximately 6 acres of vegetated high and low salt marshes received a mix of sandy and fine-grained sediment from the Fortescue Creek navigation channel. The average depth of placement at all sites was below 30 cm, which was cited earlier as a critical threshold (Moritzen et al. 2017). Placement resulted in widespread loss of plant cover, benthic infauna and epifauna, and changes in sediment chemistry and texture (Moritzen et al. 2017; Taghon 2017). By the second or third year after placement, there was no correlation between the thickness of the added sediment and vegetative or benthic infauna recovery (Moritzen and Zito-Livingston 2018; Taghon 2017). Most of the vegetation recovery was via seed or spreading from the remaining patches of vegetation rather than through the sediment, suggesting that something besides sediment thickness is driving the recovery (M. Yepsen unpublished data, 2020). Although placement was about 5 years ago, not all sites had experienced full recolonization of vegetation, and none had improved over baseline conditions.

Fortescue, where target elevations were not reached and elevation gains were the smallest, had the quickest plant and benthic infauna recovery (Taghon 2017, Moritzen and Zito-Livingston 2018). Conversely, vegetation and benthic infauna recovery was slowest on Ring Island, the site with the thinnest average placement of sediment (15 cm) (Moritzen et al. 2017). This suggests that there may be trade-offs between increasing elevation capital and the temporary loss of ecosystem function that result from restoration activities.

Enlarging marsh islands

In addition to placing sediment on low-elevation marshes, the USACE has initiated marsh island restoration projects as a beneficial reuse for dredge sediments. Two Mid-Atlantic restoration projects (Elders Island, Jamaica Bay, NY, and Poplar Island, Chesapeake Bay, MD) have expanded eroded marsh footprints through the reuse of dredge material (USEPA 2007; USACE 2019). NJDEP is currently planning projects with the USACE in Barnegat Bay. The first sites to receive material are islands that were previously used to manage dredged material from federal navigation channels, which resulted in the creation of a heron rookery and submerged aquatic vegetation habitat.

Other beneficial use techniques

USACE has placed dredged sediment in open water at locations that permit natural forces to transport sediment toward and onto marshes. This practice has been demonstrated to enhance beach volume (Brutsché et al. 2014*a*, 2014*b*). However, most of the sediment dredged by USACE is mixed sand/silt/clay, described as "muddy". The perceived risks associated with muddy sediments and poor understanding of their fate in vegetated environments have not permitted the strategic placement of muddy sediments near marshes in the Unites States.

A federal consistency was issued for marsh/open water placement at Sturgeon Island in Cape May County, NJ, to enhance bird habitats. Construction of the project began in 2020. A small volume of sediment was subject to unconfined placement in open water. Little to no turbidity was observed in the surrounding water during unconfined placement (M. Yepsen, personal communication, January 2021). This project also experimented with the use of runnels to encourage material dispersal across the marsh surface.

Lessons learned

The wide variety of application methods and project objectives complicate the definition of thin layer deposition/placement. In addition, the ability to obtain a specific thickness or target elevation is limited by the placement technique, equipment, project objectives, and site constraints. The thickness of the material is a function of the equipment used, its operation, site conditions, and dredged material physical characteristics (e.g., dispersion or consolidation potential) (Berkowitz et al. 2019).

Mohan et al. (2016) suggest the following actions to ensure a successful thin layer deposition/placement project: conducting baseline surveys, understanding project-specific hydraulics, including a diversity of habitats in the plan, and investing in long-term monitoring. TNC and NJ DEP (2020) discussed lessons learned from the pilot projects that include the items from Mohan et al. (2016) above, plus: consider tradeoffs between higher target elevations and loss of function while the marsh recovers, the grain size of the dredged material is a critical factor, minimizing the use of machinery on the marsh, minimizing the use of containment, and removing containment after sediment solidification.

Runnels

Drainage patterns vary depending on the location within a tidal marsh, with the marsh interior experiencing prolonged soil saturation (Montalto et al. 2006). While marsh plants can tolerate some waterlogging, excessive saturation leads to vegetation die-back and reduction of sediment accretion, which results in conversion of the marsh to pannes or mudflats and ultimately, open water (Watson et al. 2017). Peat collapse associated with plant dieback can lead to further elevation losses (DeLaune et al. 1994; TNC 2018). Runnels (thin channels) can enhance drainage from pools and promote revegetation; however, concerns have been raised about surface subsidence associated with dewatering.

Living shorelines

When the marsh edge is eroding, living shorelines incorporate natural materials such as vegetation, shellfish, and woody debris, as well as in some cases rock, stone, or sand, to provide erosion control, protect, restore, or enhance natural shoreline habitats, and maintain coastal processes.

Challenges implementing living shorelines

There are numerous National Oceanic and Atmospheric Administration (NOAA) and state publications describing installed or proposed living shorelines, but they seldom include long-term monitoring data needed to evaluate various techniques. States with the most mature living shoreline programs include Maryland, Virginia, and North Carolina. O'Donnell's (2016) review of living shorelines, the only review identified in a Web of Science search, describes research findings from the southern United States. Some techniques are similar to traditional engineering approaches, such as planted revetments or breakwaters, while others such as living reefs and reef balls are more novel (NJ DEP 2015*a*, 2015*b*). Researchers note that there are challenges that must be addressed when implementing living shorelines, including site-specific factors and weather (Hardaway et al. 2013; Currin et al. 2017), performance standards (Currin et al. 2010; Sutton-Grier et al. 2015), regional norms (Zhu et al. 2020; Morris et al. 2019; Psuty et al. 2019), and permitting (Restore America's Estuaries 2015).

New Jersey living shoreline projects

Living shoreline installations and programs in the New Jersey–New York region are relatively new and evolving (Rella et al. 2017). The Coastal Zone Management files were revised in 2013 to support the use of living shorelines, and 22 living shoreline projects were permitted. In 2015, the New Jersey Department of Environmental Protection and Stevens Institute of Technology released the Living Shorelines Engineering Guidelines (Miller et al. 2015). Developed for engineers, regulators, and property owners, it describes the design, permitting, and construction of living shorelines. Design considerations include site erosion history, tidal range, SLR, and ecological, hydrodynamic, and terrestrial parameters. Appropriate options are recommended based on site-specific conditions. Assessment tools include a New Jersey interactive website that suggests options for municipalities based on local factors (https://maps.coastalresilience.org/newjersey). The Waterfront Alliance in NYC has launched Waterfront Edge Design Guidelines (WEDG®) modeled on the LEED® green building program and provides a certification system for improving coastal resilience and ecology in urban ecosystems based on earned credits (Waterfront Alliance 2019).

Five permitted living shoreline projects (Atlantic City, Brigantine, Secaucus, Upper Township, Spring Lake) are featured on the New Jersey Department of Environmental Protection website (https://www.state.nj.us/dep/opi/living-shorelines.html). The monitoring time frame for living shoreline projects in New Jersey is only 1–2 years, which is much shorter than the timeframe needed to determine the trajectory of marsh restoration projects. The state is considering rule changes to require longer monitoring. Initially, monitoring was not required because these were new techniques, and it was thought that the cost of monitoring could be excessive and could divert applicants to consider harder shore protection structures, such as bulkheads.

The most extensive living shoreline research and data collection in New Jersey is in Delaware Bay, led by the Partnership for the Delaware Estuary (PDE). The Delaware Estuary Living Shoreline Initiative (DELSI) includes the use of ribbed mussels, coir logs, bagged oysters, and *S. alterniflora* to mimic natural features in low energy areas. Generally, living shorelines have been successful in meeting erosion control goals once they are able to persist beyond the initial settling time. Newly deployed materials are susceptible to wave energy, flow velocity, and storms during the initial stages of deployment. Regarding materials, contained shells (e.g., bagged, caged) showed the greatest stability over time, largely holding its lateral and vertical positions after a brief settlement time. It has been effective in recruiting shellfish, largely because of the refuge provided. Oyster castles have a high degree of initial stability, but if not completely colonized, they may deteriorate, compromising wave attenuation (J. Moody, Partnership for the Delaware Estuary, personal communication). Coirs are generally ineffective owing to their low structural integrity. Coir logs unravel easily with continued movement and become clogged with sediment, but they can be very effective in the area between the waterward edge and pre-existing marsh edge. However, if directly exposed to wave and flow action in all but the lowest energy environments, they tend to deteriorate rapidly (J. Moody, Partnership for the Delaware Estuary, personal communication).

Monitoring results for 2018–2019 suggest that the impact of the Delaware Bay living shoreline is complex. Overall, the oyster reef breakwaters had a localized impact rather than a site-wide benefit. Even though the structures appear to be effective at attenuating incoming waves most of the time, there are times when wave heights inside the breakwaters exceeded 0.5 m (Kerr et al. 2020). The upper intertidal areas have gained sediment, elevation, and vegetation (TNC 2020).

Living breakwaters can be hard, such as concrete reef balls, or softer, such as oyster bags. At the Cape May NWS Supawna Meadows unit, the USFWS developed a living breakwater by manipulating the elevations of a historic breakwater constructed for salt marsh hay farming in the early 1900s. Data on the effects of the project are not yet available.

Subtidal installation of oyster castles or reef balls can reduce the tidal energy that contributes to erosion. NY/NJ Baykeeper installed 600 oyster castles set with oyster spat, where the *Spartina* marsh is eroding in Raritan Bay. Using sediment traps, the rate of deposition was monitored on the shoreward side of the installation. Long-term shoreline monitoring and sediment deposition data collection were used to evaluate the effectiveness of these structures.

A test of a Christmas tree living shoreline is now in progress (2019) in Point Pleasant, NJ. The shoreline of the 13-acre tidal wetland and swamp has eroded 90 m since 1930, and all the low marsh has been lost. The American Littoral Society oversaw the installation of 300 wooden pilings that hold discarded Christmas trees that will form a living breakwater. (https://www.littoralsociety.org/blog/recycled-christmas-trees-will-help-restore-local-shoreline).

Living shoreline recommendations

Toft et al. (2017) synthesized living shoreline lessons learned from long-term perspectives and future needs. It is very important for all project partners (practitioners, regulators, and funders) to agree with the project goals. Critical questions include: What candidate sites and tactics exist to address the project's goals? What data exist to characterize conditions at prospective sites (energy, distance to the opposite bank [fetch], slope, vegetation, soil type, soil firmness, and boating activity)? Which types of living shorelines would be successful at the site (soft, hard, or hybrid)? If the installation is successful, how many acres of tidal wetlands would be protected? Is there a time of year for installation to minimize risks or maximize biological stability? Are there potential drawbacks or habitat tradeoff issues to address, such as the presence of SAV?

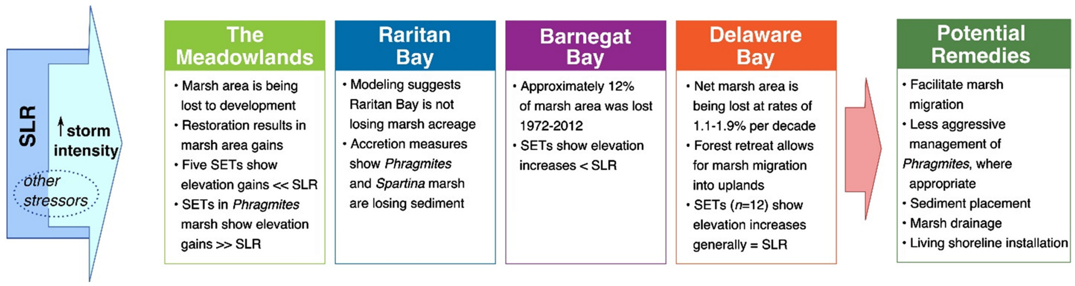
Restore America's Estuaries (2015) recommends regulatory changes that include reduction in traditional "cookie cutter" solutions, and a program that coordinates federal, state, and local regulations to evaluate impacts beyond the project site. The New Jersey Department of Environmental Protection has encouraged a diversity of tidal wetland restoration and enhancement techniques and has approved 66 coastal habitat enhancement projects between 2013 and 2020. Installation must be adaptively managed and frequently augmented. Since Zhu et al. (2020) found that oyster castle installation could increase erosive energies depending on tidal heights and bottom conditions, inadvertently having a negative effect on marsh sustainability, long-term monitoring is recommended for all projects.

Synthesis and summary

This synthesis shows that coastal marshes are losing ground across New Jersey (Fig. 4). Over past decades, coastal wetland loss to development has continued (especially in Raritan Bay, the Meadowlands, and Barnegat Bay), while rates of marsh edge erosion were found to average 0.5 m year^{-1} in Barnegat Bay and 3 m year^{-1} in Delaware Bay. Marsh accumulation rates do not generally exceed the current rates of SLR, predicting that accumulation deficits and marsh drowning will occur (Fig. 2). Rapid acceleration rates of SLR in New Jersey and SET elevation change data suggest that sediment accumulation deficits occur, mirroring deterioration in coastal wetlands in surrounding states (Hartig et al. 2002; Cameron Engineering & Associates 2015; Schepers et al. 2017).

Given these rates of loss, additional data are urgently needed to plan New Jersey's response to this pressing issue. It is important to determine where coastal marsh loss will be most severe, where there is high potential for marsh survival, and where mitigation

Fig. 4. Synthesis of responses and status of the four major wetland systems and potential remedies.



efforts might be most profitably focused (Kearney and Rogers 2010). To advance this approach, the following data sources are valuable.

- Statewide remote sensing analysis of wetland trends to identify areas of deterioration
- Synthesis of consistent and comparable SET and coastal erosion data from across the state to identify emergent patterns and areas of high vulnerability
- Filling gaps in the tide and salinity gauge networks across the state and updated tidal datum calculations from NOAA 2020
- Research on factors that promote marsh resilience vs. vulnerability to SLR

Ideally, these data would guide policy and stewardship responses, in concert with other concerns, such as socio-economic vulnerability, conservation status, and anthropogenic stressors. Basing restoration or intervention actions on strong science will better preserve functioning coastal marsh ecosystems, aid in producing desired results including improved outcomes, and will better utilize scarce restoration funding.

Potential remedies

To preserve coastal wetlands under accelerated SLR rates, a shift in coastal management is needed to extend the lifespan of drowning marshes and facilitate marsh migration into uplands. These approaches may include identifying, preserving, and managing marsh migration pathways, relaxing aggressive *Phragmites* removal policies, interventions such as sediment placement and drainage that build ecosystem resilience, and installation of living shorelines (Fig. 4).

In developed areas, coastal squeeze may be mitigated by developing and maintaining open areas for the marsh to migrate into, creating migration pathways. Creating pathways generally involves local governments working with landowners using tools such as rolling easements. Problems that may arise include lack of knowledge about marshes, coordinating actions among various parties, and providing adequate compensation to landowners, as well as other political, social, and economic issues. The invasive common reed, *Phragmites*, performs important ecosystem services; the most important here is enabling marshes to increase in elevation faster than native plants. Indeed, the marshes that increased their elevation at the greatest rates were the Meadowlands *Phragmites* marshes. Nevertheless, projects continue to remove these plants. Management could be modified to leave some *Phragmites* in place to enable a marsh to keep up with the SLR. Engineering approaches, such as thin layer placement and runnels, can be used to build ecosystem resilience. Important considerations include the timeframe for recovery and the high costs of these approaches. Lastly, living shorelines can reduce erosion at the edge and preserve the extent of the marsh. Options include adding material, such as coir logs, oyster castles, woody debris, or floating wetlands, which dampen wave or storm surge energies. Challenges include site specificity, lack of data on effectiveness, permitting, and substantial costs. Living shorelines need to be adaptively managed and possibly augmented frequently, thus requiring long-term monitoring.

The approaches described here may require changes in attitudes and policies or may be expensive and site-specific. All of these issues require continued attention. While they hold promise for the preservation of coastal marshes under climate change, these techniques require, in some cases, a paradigm shift in coastal management (Wigand et al. 2017). Over the past decades, policies and regulations have prevented the disposal of sediment in marshes and marsh drainage. *Phragmites* removal has been incentivized, and areas upslope of the high tide line have become priority areas for development. Preserving coastal marshes under climate change requires managers to shift their approach to focus on promoting future persistence, rather than simply restoring marshes to a less anthropogenically impacted state.

Competing interests

The authors declare no competing interests.

Authors' contributions

J.S.W. organized the project and was the primary author of the Introduction, Migration Pathways, and *Phragmites* sections. E.B.W. was the primary author of the Status of the Marshes section, C.H. and M.Y. were primary authors of the Sediment section, and B.R. was the primary author of the Living Shorelines section. All authors were involved in editing all the sections.

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