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Natural Resource Stewardship and Science

Salt Marsh Vegetation Change at Cape Cod National Seashore between 2003 and 2018

Natural Resource Report NPS/CACO/NRR—2020/2157

ON THE COVER
Photograph of Nauset marsh (photo taken by Stephen Smith)

Salt Marsh Vegetation Change at Cape Cod National Seashore between 2003 and 2018

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Executive Summary

Seven salt marshes within Cape Cod National Seashore (CACO) were surveyed for their plant communities in 2018, constituting the most recent survey within the framework of CACO's long-term ecological monitoring program. The main objective of this report is to describe structural and taxonomic change in salt marsh plant communities that have occurred between the first (2003) and most recent (2018) surveys. The analyses include key marsh attributes of vegetation composition, structure, and soils.

Distinct changes in CACO salt marshes have occurred over the last ~15 years. For instance, the dominant low and high marsh species (*Spartina alterniflora* and *S. patens*, respectively) have both decreased, with losses exhibiting large spatial variation among and within marsh sites. These were mostly due to sea level rise (SLR), crab herbivory, and/or lateral wave erosion. The ratio of *S. patens*: *S. alterniflora* and the abundance of transitional species (marsh border and terrestrial species) also declined, leading to reduction in species richness. Soils analyses showed that the two Atlantic-side marshes (Nauset and Pleasant Bay) both have the highest densities of organic carbon in their substrates. They also make up the largest portion of the 18,000 metric tons (MT) of carbon estimated to be stored in CACO marshes.

Although the results of regression analyses were not statistically significant, the declining patterns in these species were reasonably well-defined, considering they consist of only four data points (representing the four survey years). Presently, the statistical power to detect trends remains relatively low but this will quickly improve with subsequent surveys. Notwithstanding, the data suggest that ground-level monitoring of CACO salt marsh vegetation is tracking the kinds of plant community shifts that are expected to occur in response to SLR. Moreover, this report provides further evidence that long periods of monitoring are necessary to elucidate trends in this resource—particularly for species in transitional and upland areas that cannot be identified by remote sensing. Finally, this report suggests that salt marsh migration is currently not keeping up with losses given that reductions in the cover of *S. alterniflora* and *S. patens* were generally observed across the plot network.

Introduction

Salt marsh ecosystems are an important natural resource at Cape Cod National Seashore (CACO). In addition to their aesthetic value, their role in supporting a wide variety of flora and fauna has been well documented and coastal communities receive considerable benefits from marshes, including attenuation of nutrient inputs to the marine environment, protection of shorelines from erosion, dampening of storm surges, and abundant recreational opportunities, (Nixon and Oviatt 1973, Bertness 1999, Roman et al. 2001).

Salt marshes comprise roughly 7% of the total land cover at CACO. Three marshes have been directly impacted by hydrologic restrictions to tidal flow (e.g., Hatches Harbor, Provincetown; East Harbor, Truro; Herring River, Wellfleet), which have cause severe physical, chemical and biological degradation (Roman et al. 1984, Portnoy and Giblin 1997, Roman and Burdick 2012). Restoration of tidal flow was undertaken at Hatches Harbor in 1998 and East Harbor in 2002, while plans are currently in development for Herring River restoration. Most of CACOs marshes, however, are hydrologically unimpaired (unrestricted) and these systems are the subject of this report. Threats to their integrity include accelerating sea-level rise, nutrient inputs, herbivory, erosion, and various aspects of climate change. The vegetation of six such marshes has been monitored at CACO since 2003. Two other marsh areas were added to the sampling network in 2004 and again in 2008 to expand the monitoring network to 8 sites. This report synthesizes these datasets to provide an understanding of how these ecosystems have changed over the last ~15 years.

Methods

Study areas

In 2003, a network of plots along randomly located transects oriented perpendicularly along the main axes of five unrestricted salt marshes was established, following the guidelines of Roman et al (2001). The vegetation was surveyed, and data was also collected for several physico-chemical variables such as soil organic matter content (Smith and Portnoy 2004). The plots span the entirety of each marsh from the seaward to upland edges. Plots were placed at uniform distance from each other along transects but this distance varied between 20 and 100 m depending on the size of the marsh (see Appendix I). These marshes were Hatches Harbor (HH; Provincetown), West End marsh (WE; Provincetown), Middle Meadow (MM; Wellfleet), Nauset marsh (island portion = NI; mainland portion = NM; Eastham), and Pleasant Bay marsh (PB; Orleans and Chatham) (Figure 1). In 2004, the Gut (GU; Wellfleet) was added to the monitoring network and surveyed for plant cover by species, although no plant heights, peat depths, or sediment cores were collected during this survey. Accordingly, all analyses involving GU data in this report were limited to species composition shifts between 2004 and 2018 for that site. Similarly, the monitoring network was further expanded in 2008 to include Jeremy marsh (JM; Wellfleet). More information about monitoring procedures can be found in the previous reports of Smith and Portnoy (2004), Smith et al. (2009), and Smith (2015).

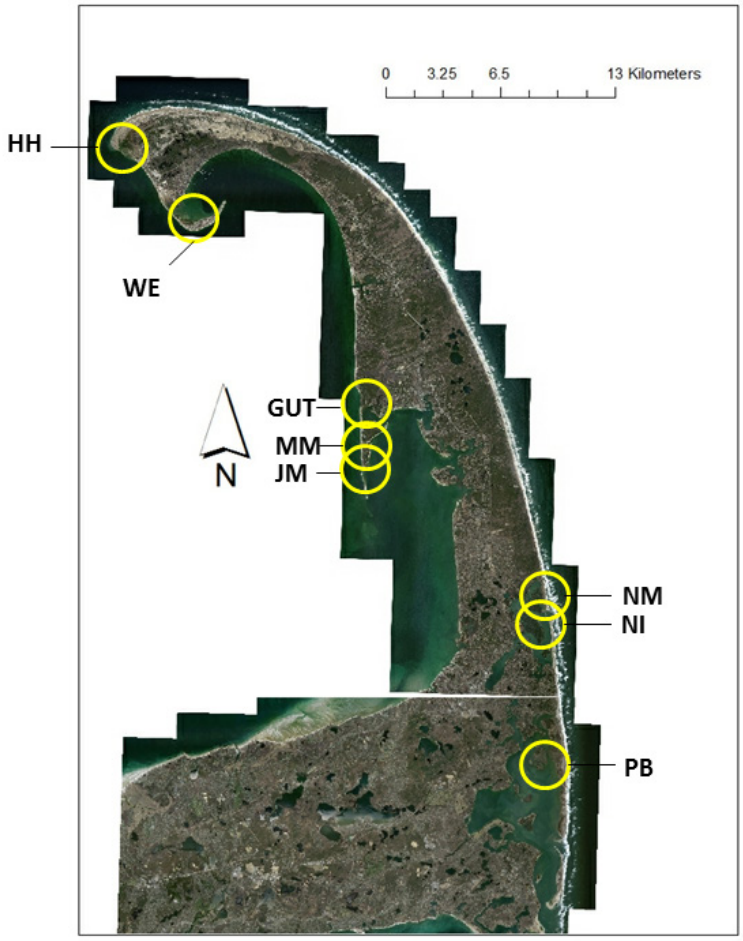

Figure 1. Map of all unrestricted marsh monitoring sites (HH-UR=Hatches harbor (unrestricted portion), WE=West End, GUT=The Gut, MM=Middle meadow, JM=Jeremy marsh, NI = Nauset marsh (island), NM = Nauset marsh (mainland), PB=Pleasant Bay marsh).

Vegetation

Species composition

In 2008, 2013, and 2018, vegetation cover by species was visually assessed in 1-m^2 plots based on a modified Braun-Blanquet scale (0=0%, 1=<1%, 2=1–5%, 3=6–10%, 4=11–25%, 5=26–50%, 6=51–75%, 7=76–100%). This differs from 2003, when cover was assessed by point-intercept counts and converted to cover classes based a slightly coarser scale (0=0%, 1=<1%, 2=1–25%, 3=26–50%, 4=51–75%, 5=76–100%) (Smith and Portnoy 2004). In order to obtain the longest time series for analysis, the 2008–2018 data were converted to the 2003 categories. Mid-point percentage values of those cover scores representing the corresponding ranges (1=0.5%, 2=12.5% 3=37.5%, 4=63.5%, 5=87.5%) were also determined analyzed. In plots where *Spartina alterniflora* (smooth cordgrass) was present, the heights of the five tallest plants were measured to the nearest cm and the average value calculated. Analyses of the data from both surveys (2003 and 2018) were conducted on different levels, ranging from detection of change across the entire network of plots (i.e., CACO-wide) to the responses of individual species in subgroupings of plots in individual marshes.

Environmental variables

Soil organic matter

Samples of sediment from an area adjacent to each plot were obtained by coring to resistance using a 5-cm diameter butyrate tube, which was inserted into the soil until resistance. This was done to capture the entire organic layer for analysis and determinations of carbon storage capacities, instead of just the top 20 cm as has been done previously. Before coring, a serrated knife was used to cut around the outside of the tube to prevent compaction. The cores were placed in zippered bags, transported back to the laboratory, and dried in a convection oven at 105°C for 48 hours. For organic matter determination, a ~10 ml subsample of the dried sediment was placed in a pre-weighed ceramic dish and weighed. The dish was then placed in the muffle furnace set to 500°C for 5 hrs. After letting the dish and sample cool down for at least 1 hr., the dish was weighed again to determine the amount of weight loss after burning (which equals the amount of organic matter). Total organic matter in the core samples was then calculated. Subsamples were sent to Brookside Laboratories, Inc. (Bremen, OH) for total carbon and nitrogen analysis using auto-sampler (Elementar™, Ronkonkoma, NY) based on the methods of Nelson and Sommers (1996). From these values, rough estimates of total carbon storage per marsh based on the formula: mean carbon concentration (g/cm^3) from cores x mean core volume x area of marsh.

Data analysis

All non-parametric analyses of species cover data were done using Primer ver. 7 software (Plymouth Marine Laboratory, UK). Analysis of Similarities (ANOSIM), using Bray-Curtis similarity indices generated from cover class scores, provided a non-parametric statistical test of floristic changes between survey years, which was illustrated using non-metric multidimensional scaling (nMDS). Similarities Percentages (SIMPER) was used to assess which taxa contributed most to dissimilarities among survey years. Temporal trends in specific variables were assessed by regression analysis. Vegetation cover data was handled by converting cover class ranks to their corresponding midpoint percentage values, log-transforming the data to improve normality, and regression analyses (Excel ver. 10). Analysis of Variance (ANOVA) followed by Tukey tests were used to distinguish differences between specific comparisons of mean values (JMP ver. 10).

Results

Environmental variables

Peat depths

Peat depths across the marshes exhibited large differences among sites (all were statistically different from each other) but remained similar to those reported in Smith and Portnoy (2004), with NS and PB having the thickest peat layers and HH the thinnest (Figure 2). This makes sense since NS and PB are the oldest marshes of the seven sites (Uchupi et al. 1996). No comparisons were made with 2004 peat depths given that the margin of error in measuring peat depths is too high to detect a few cm of change over the course of 15 years.

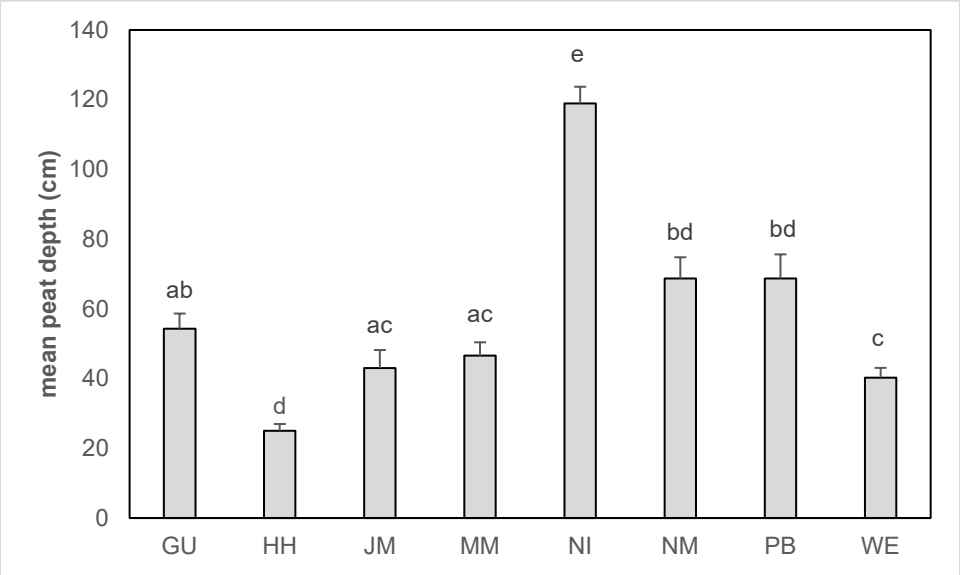

Figure 2. Mean peat depths by site in 2018 (error bars are standard errors of the means; histograms sharing a letter are statistically similar at α=0.05; no peat depths were measured in GU in 2004).

Soil organic matter and nutrient content

Organic matter content was much higher in NS and PB marshes (>10%) than all other sites, which had values below ≤5% (Figure 3). This was originally reported in Smith and Portnoy (2004) and likely reflects the age of these systems and the rate of peat accumulation, with the latter having formed more recently (Uchupi et al. 1996).

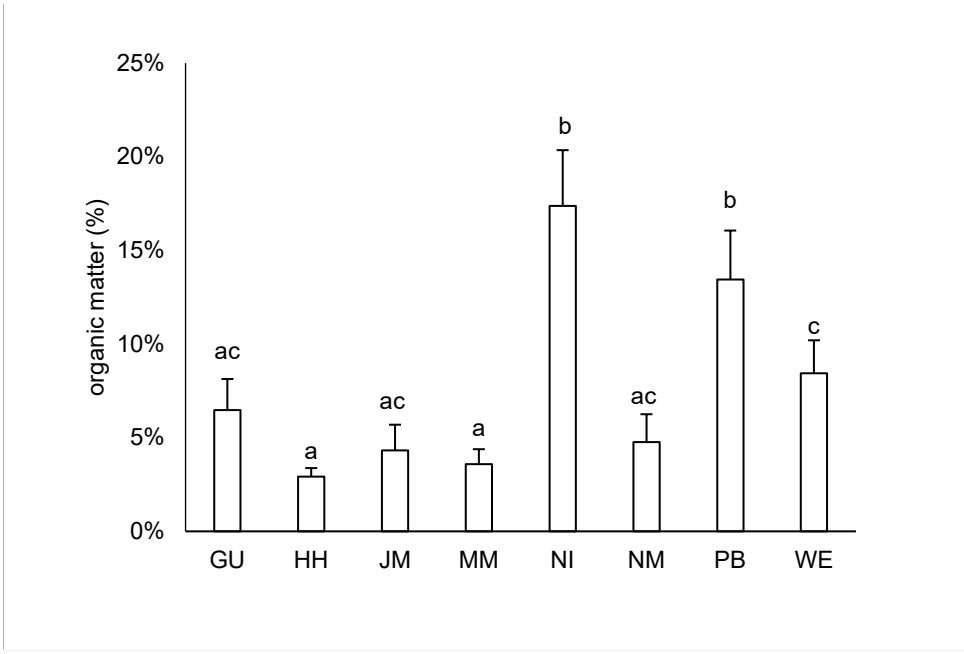

Figure 3. Organic carbon content of 2018 salt marsh soils from samples cored to resistance.

Carbon (C) and nitrogen (N) concentration of marsh soils generally followed the same patterns as OM content. In fact, both carbon and nitrogen concentrations were closely, and positively, related to organic matter content (Figure 4). Mean % C and N among marsh sites are listed in Table 1. NS, PB, and WE had high concentrations of both compared to the rest.

Table 1. Carbon, nitrogen, and C/N ratio of salt marsh soils cored to resistance by marsh in 2018.

Site	Carbon (%)	Nitrogen (%)
GU	1.74	0.18
HH	1.83	0.18
JM	1.04	0.12
MM	0.91	0.09
NI	4.00	0.39
NM	1.01	0.11
PB	3.86	0.33
WE	3.09	0.29

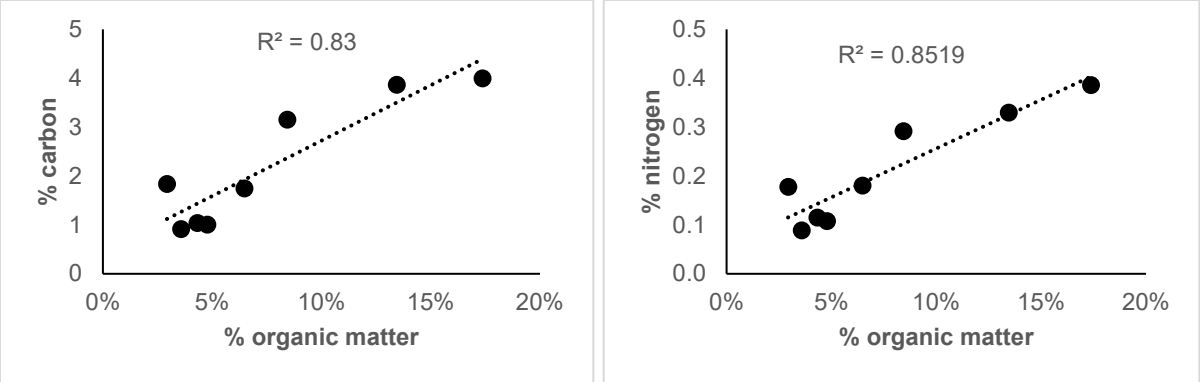

Figure 4. Relationships between organic matter content and carbon (left) and nitrogen (right) concentrations.

Carbon content of CACO marshes

NI and PB had the highest estimated carbon content of CACO marshes, JM had the least. The total amount carbon stored in CACO marshes based on these calculations is 17,686 MT. This is equivalent to the amount of carbon stored in 23,097 acres of "US forest" based on the Environmental Protection Agency's (EPA) greenhouse gas equivalencies calculator (https://www.epa.gov/energy/greenhouse-gas-equivalencies-calculator).

Table 2. Estimates of total carbon in metric tons (MT) by marsh in 2018.

Site	Total C (MT)
GU	285
HH	665
JM	72
MM	339
NI	5749
PB	9226
WE	1351
Total	17686

Vegetation

S. alterniflora heights

Mean plant heights of *S. alterniflora* exhibited significant differences among survey years, but there were no temporal trends—either collectively or at individual sites (Figure 5). More informative, perhaps, are the spatial differences in plant heights that have persisted through all surveys. NS, NM, and PB, which are the oldest marshes and have the highest content of organic matter and porewater H_2S in sediments, have much shorter vegetation than the rest. In contrast, plants at GU, HH, and WE are more robust, since they grow in sandier, well-oxygenated sediments v(Smith and Portnoy 2003).

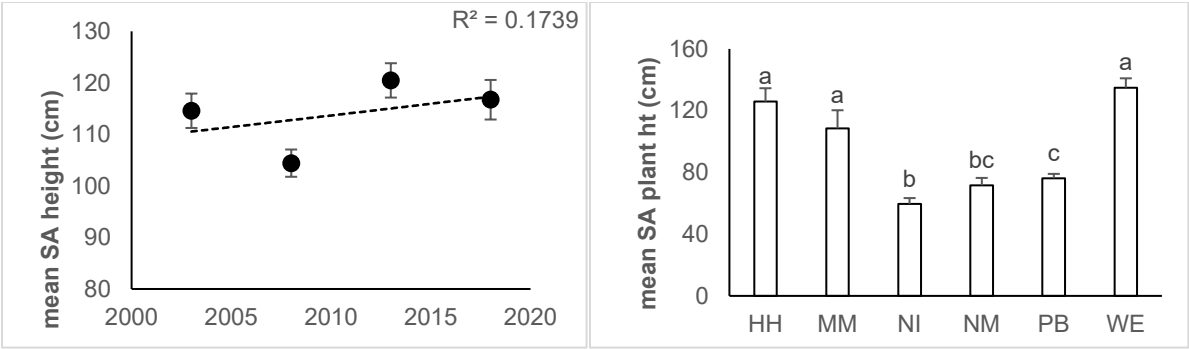

Figure 5. Mean plant height (all sites pooled) by survey year (left) and at individual sites in 2019 (error bars are standard errors of the means; histograms sharing a letter are statistically similar at α=0.05; plant heights were not collected at GU in 2004).

Species richness

Species richness at the individual plot level remained statistically similar between 2003 and 2013 but then decreased in 2018 (Figure 6). Total species richness (all sites) exhibited an obvious declining trend, but it was not statistically significant (Figure 6). These losses were largely a consequence of transitional and upland taxa disappearing from the higher elevation plots along transects. When individual sites are analyzed, species richness appears to have been declining since 2008 at HH and NM, but variability is high and there were no obvious temporal trends at other sites (data not shown). Regardless, species richness in 2018 was significantly lower than in most other years at many sites.

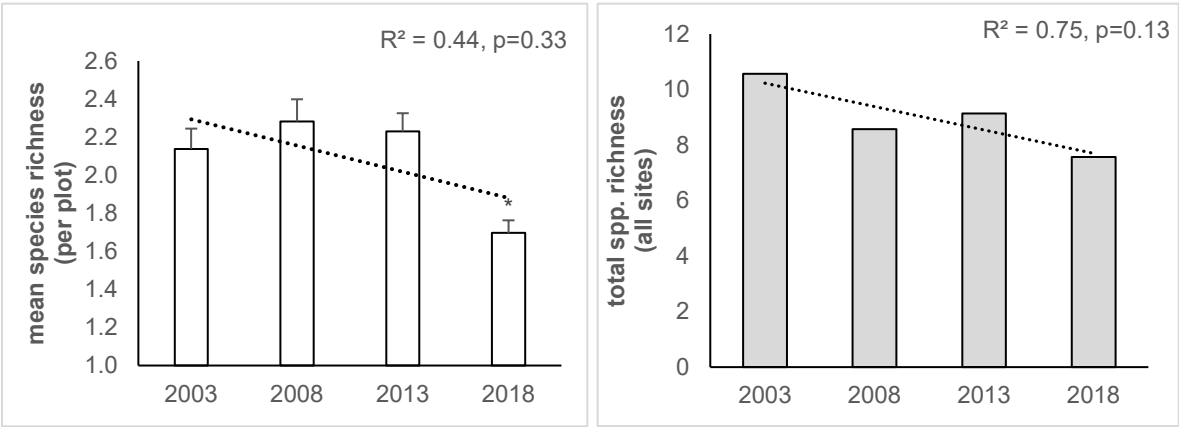

Figure 6. Mean species richness per plot (all sites pooled; upper left) and total species richness across all sites by year (upper right) (error bars are standard errors of the means; asterisks indicate a significant difference between 2003 and 2018 values at α=0.05).

Overall plant communities

There was a significant shift in vegetation overall (ANOSIM R = 0.03, p=0.1%) due to altered spatial patterns and declining abundances of the two foundation species *S. alterniflora* and *S. patens* (Figure 7; Table 3). The outlier NM plots are due to vegetation recovery occurring across the overwash area. The PB outlier was due to the disappearance of four upland species from that location.

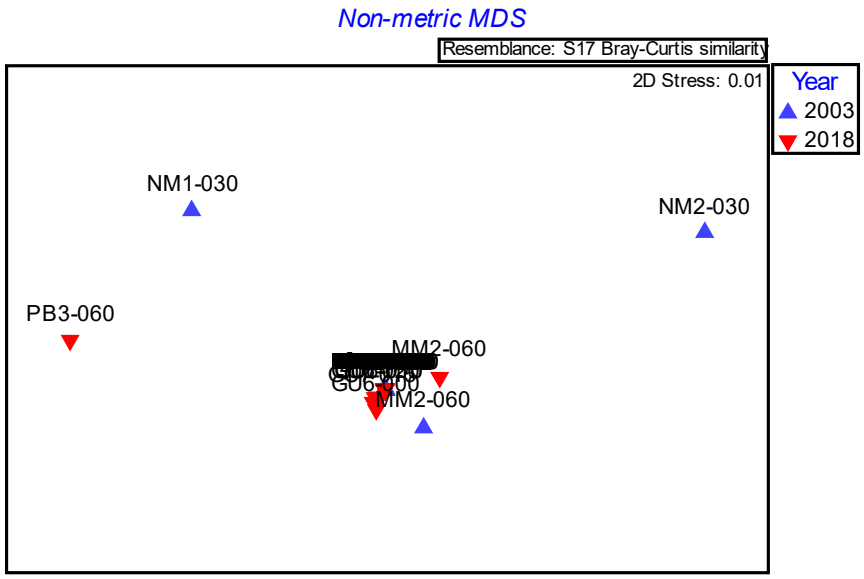

Figure 7. nMDS of species composition of all sites in 2003 and 2018.

Table 3. SIMPER results indicating the species most responsible for taxonomic dissimilarities between 2003 and 2018 plant communities (Cum.% = cumulative percent variance explained by taxon shifts in abundance and distribution).

Species	2003	2018	Cum.%
Spartina alterniflora	3.7	3.0	37.5
Spartina patens	1.0	0.6	57.2
Salicornia virginica	0.7	0.7	73.4

Plant communities of individual marshes

The Gut

At GU, there was no statistically significant change in the vegetation community between 2003 and 2018 (ANOSIM R = −0.007; p=49.5%) (Figure 8).

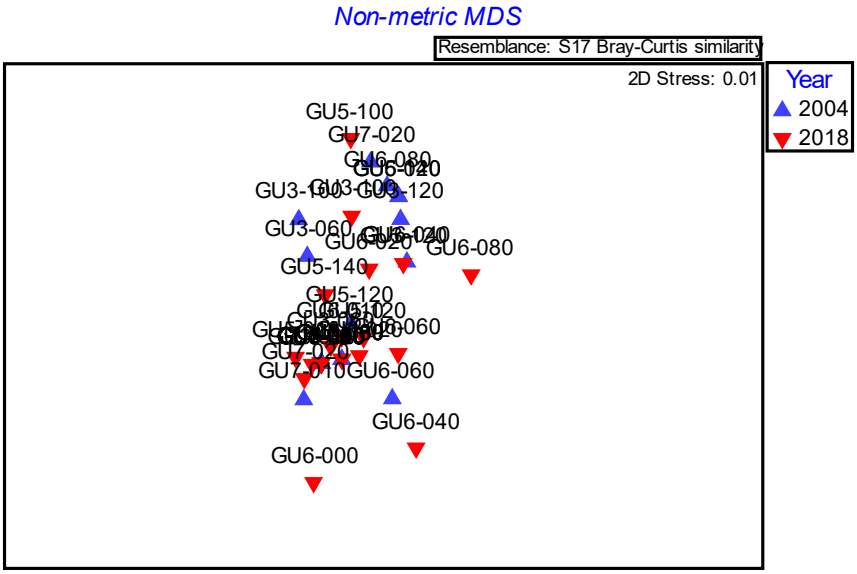

Figure 8. nMDS of species composition at GU in 2004 and 2018.

Hatches Harbor

The plant community at Hatches Harbor shifted significantly between 2003 and 2018 (ANOSIM R=0.13, p=0.006), due mainly to large decrease in abundance of *S. alterniflora*, and to a lesser extent, *Salicornia virginica* (Figure 9, Table 4).

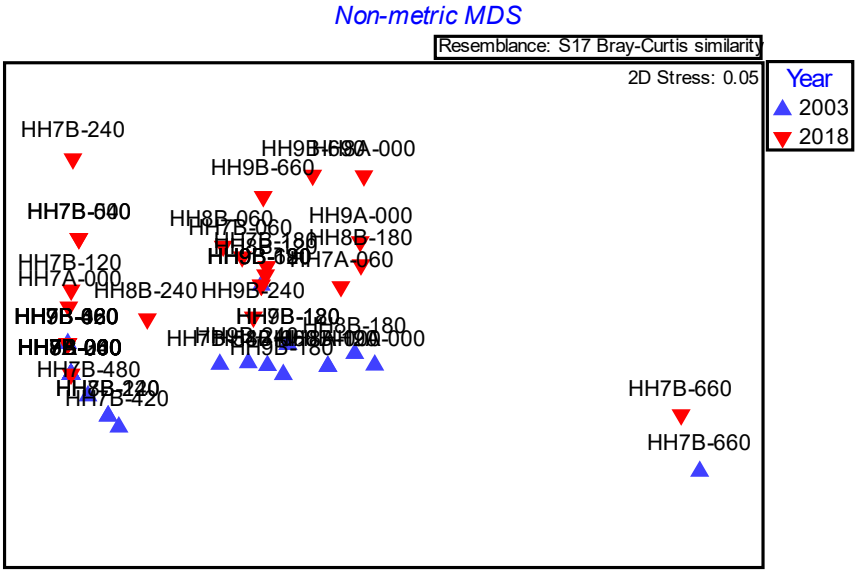

Figure 9. nMDS of species composition at HH in 2003 and 2018.

Table 4. SIMPER results indicating the species most responsible for taxonomic dissimilarities between 2003 and 2018 plant communities at HH (Cum.% = cumulative percent variance explained by taxon shifts in abundance and distribution).

Species	2003	2018	Cum.%
Spartina alterniflora	3.4	2.1	33.4
Salicornia virginica	2.1	2.0	63.7
Salicornia maritima	1.4	0.0	79.2

Middle Meadow

Vegetation change was also observed in MM between 2003 and 2018 (ANOSIM R=0.072, p=0.02). This change was mainly a consequence of reductions in, and spatial shifts in, *S. alterniflora* and *S. patens* (Figure 10, Table 5).

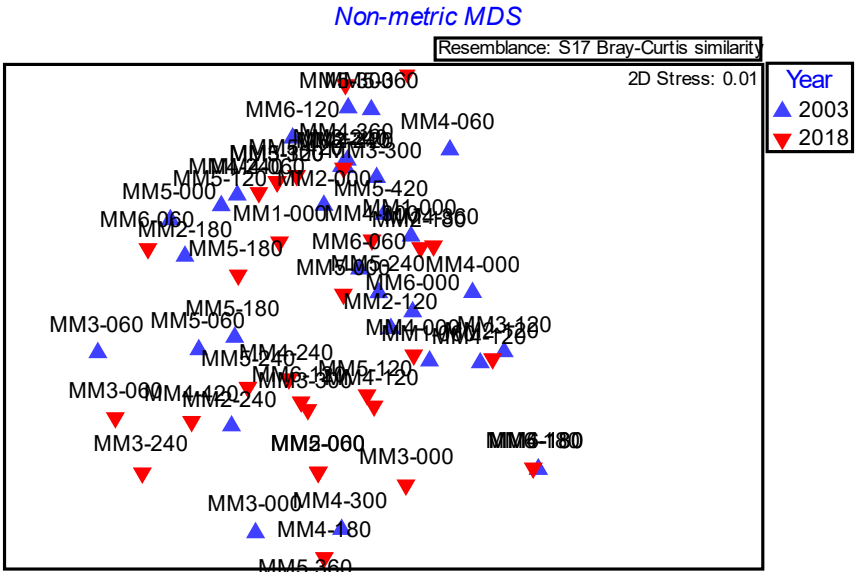

Figure 10. nMDS of species composition at MM in 2003 and 2018.

Table 5. SIMPER results indicating the species most responsible for taxonomic dissimilarities between 2003 and 2018 plant communities at MM (Cum.% = cumulative percent variance explained by taxon shifts in abundance and distribution).

Species	2004	2018	Cum.%
Spartina alterniflora	3.4	3.2	44.9
Spartina patens	1.4	0.7	72.2

Nauset Island

No major taxonomic shifts occurred at NI over the 15-year time period of analysis (ANOSIM R=0.01; p=0.24), primarily because this system is almost entirely dominated by *S. alterniflora*. The minor amount of *S. patens* in the plot network remained similar between years although its distribution shifted significantly as this taxon disappeared from some plots but appeared in others where it had not been recorded previously (NI3-200 and NI3-400) (Figure 11, Table 6).

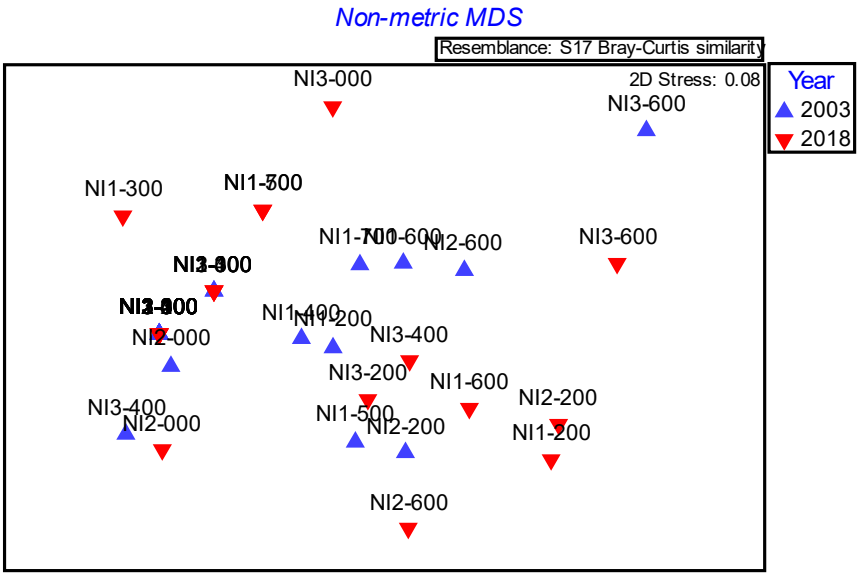

Figure 11. nMDS of species composition at NI in 2003 and 2018.

Nauset Mainland

At NM, much of the plot network was located across am extensive overwash fan that was present in 2003. Since then, vegetation has been recolonizing this sand plume. Species composition remained statistically similar between 2003 and 2018 (ANOSIM R=0.01, p=0.30) (Figure 12).

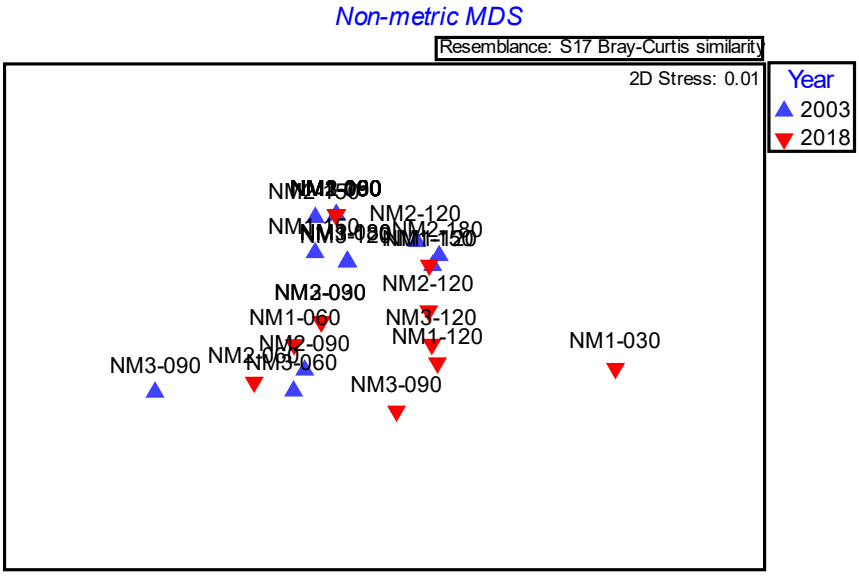

Figure 12. nMDS of species composition at NM in 2003 and 2018.

West End

Reductions in *S. alterniflora*, *S. virginica*, and *S. patens* contributed most to the significant change in vegetation observed in WE between survey years (ANOSIM R = 0.12; p=001) (Figure 13, Table 6).

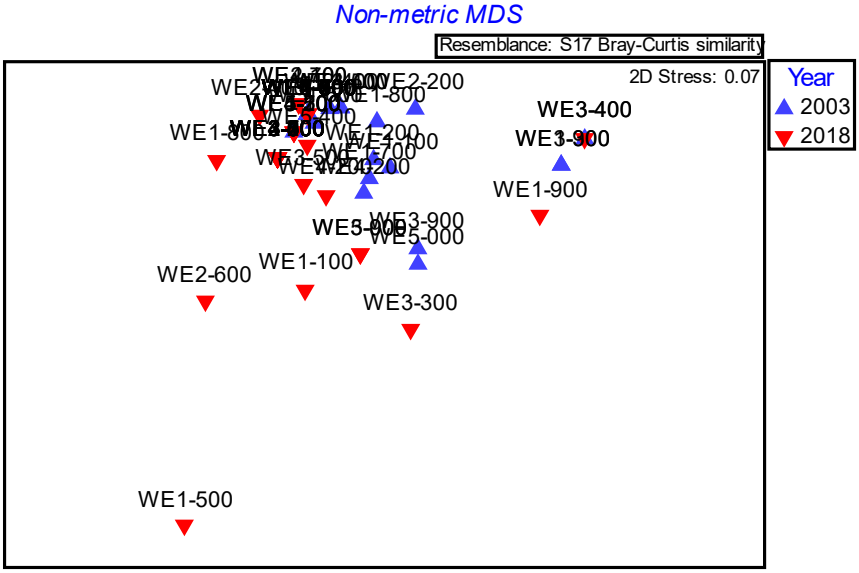

Figure 13. nMDS of species composition at WE in 2003 and 2018.

Table 6. SIMPER results indicating the species most responsible for taxonomic dissimilarities between 2003 and 2018 plant communities at WE (Cum.% = cumulative percent variance explained by taxon shifts in abundance and distribution).

Species	2003	2018	Cum.%
Spartina alterniflora	4.4	3.1	44.4
Salicornia virginica	0.8	0.5	62.3
Spartina patens	0.5	0.2	77.4

Pleasant Bay

In PB, species composition was statistically similar between 2003 and 2018, as indicated by ANOSIM (R = −0.005; p=0.61) (Figure 14).

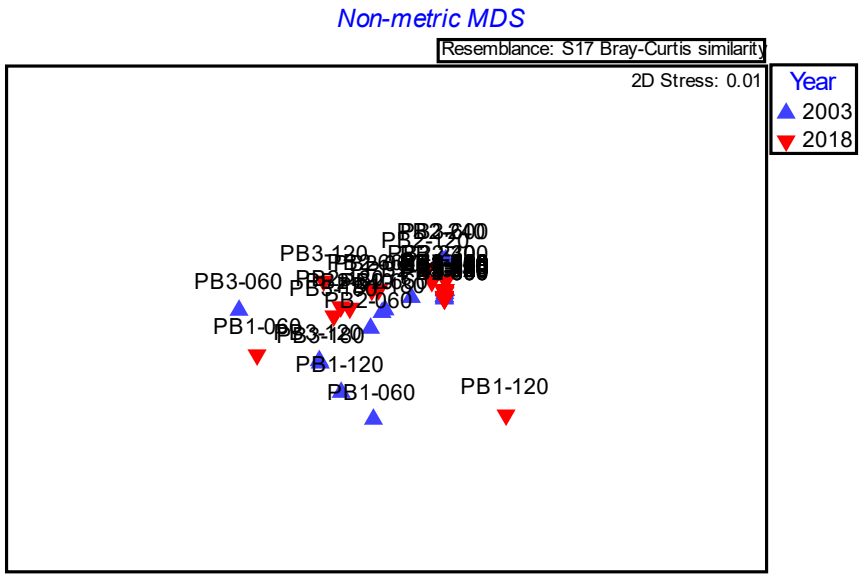

Figure 14. nMDS of species composition at PB in 2003 and 2018.

Changes in S. alterniflora and S. patens

None of the temporal changes analyzed below met the criteria for statistical significance. However, the power to detect change is still greatly limited by the fact that there are only 4 years of survey data (i.e., n-4). Hence, it is unsurprising that what may be real and long-term trajectories are not statistically significant.

Highest elevation plots

When the ten highest elevation plots in each marsh were analyzed across all sites (GU data from 2004, no JM data; n=70), there was a reduction in the cover of *S. alterniflora*, and the difference between 2003 and 2018 was significant. In contrast, there was high variability, very low cover, and no temporal change in *S. patens* (Figure 15).

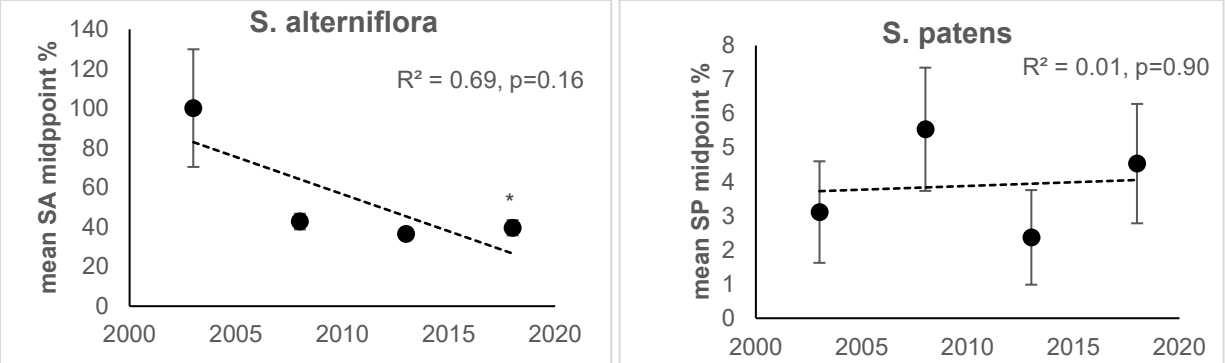

Figure 15. Average midpoint percentage values of *S. alterniflora* (left) and *S. patens* (right) in the highest elevation plots of all transects and sites by year (error bars are standard errors of the means; asterisks indicate a significant difference between 2003 and 2018 values at α=0.05).

<u>Lowest elevation plots</u>

In the lowest elevation plots (10 lowest from each site; GU data is from 2004; n=70), *S. alterniflora* cover declined, although not significantly (Figure 16). Regardless, cover was significantly lower in 2018 than in 2003. *S. patens* exhibited he same trend, although it was statistically much stronger.

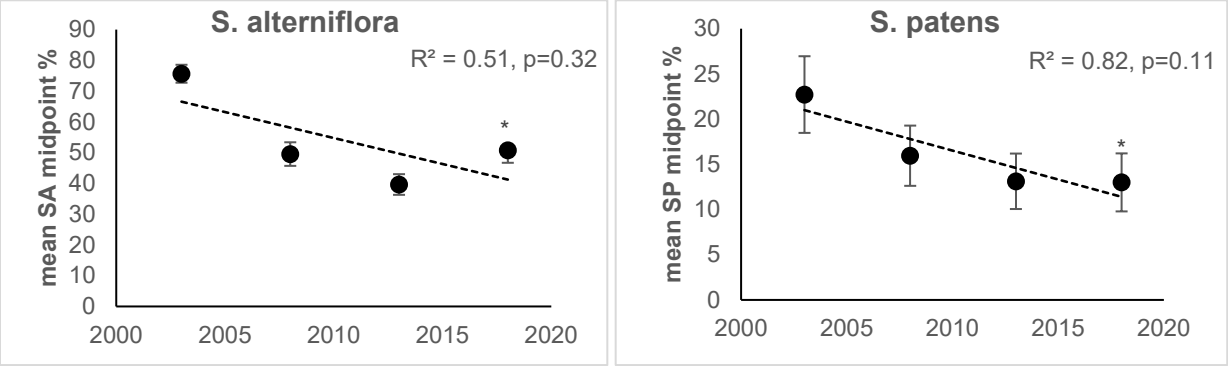

Figure 16. Average midpoint percentage values of *S. alterniflora* (left) and *S. patens* (right) in the lowest elevation plots of all transects and sites by year (error bars are standard errors of the means; asterisks indicate a significant difference between 2003 and 2018 values at α=0.05).

In platform-type marshes, the seaward-most edges are well above the adjacent mudflat and substrantially above mean low tide (MLT) (Smith et al., 2016). However, other marshes are gently-sloped and eventually reach an elevation that experiences too mucvh flooding to survive. In these marshes, the seaward edge of S. alterniflora is typically close to or even below mean low tide—so much lower than in platform marshes. Thus, plants at these locations should be the most sensitive to changes in sea leve. When only these data are analyzed (n=17; plots from GU, MM, and WE), *S. alterniflora* declines precipitously and significantly between 2003 and 2018 (Figure 17).

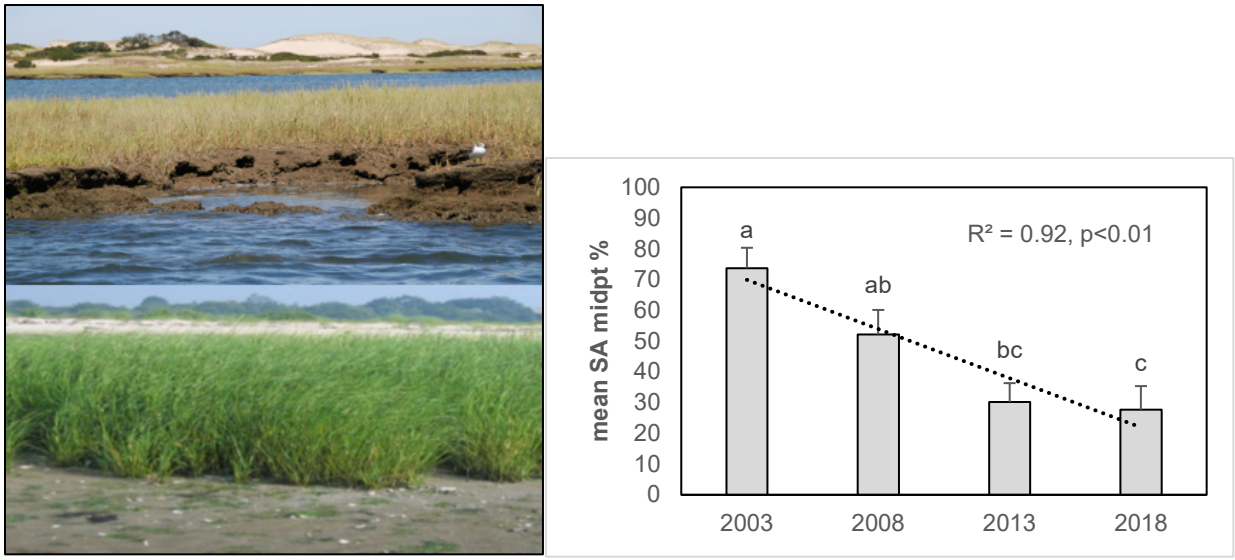

Figure 17. Examples of a platform (top left) vs. sloping (bottom left) and average midpoint percentage values of *S. alterniflora* in the lowest elevation plots of the latter (right) (plots are from GU, MM, WE) (error bars are standard errors of the means; asterisks indicate a significant difference between 2003 and 2018 values at α=0.05).

<u>All plots</u>

When *S. alterniflora* and *S. patens* were analyzed across all plots (all sites), declines in both taxa were observed, resulting in significant differences between 2003 and 2018 values (Figure 18).

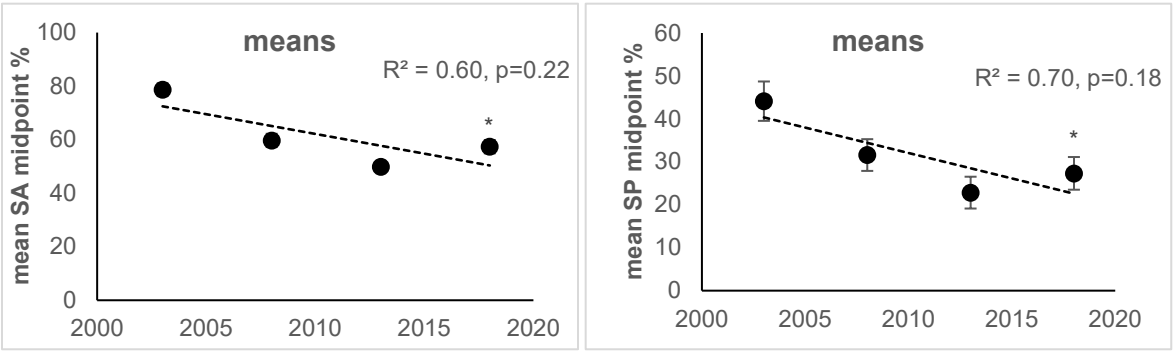

Figure 18. Average midpoint percentage values of *S. alterniflora* (left) and *S. patens* (right) in all plots of all transects and sites by year (error bars are standard errors of the means; asterisks indicate a significant difference between 2003 and 2018 values at α=0.05).

Individual sites

<u>S. alterniflora</u>

Although 2018 cover values for *S. alterniflora* were significantly lower than 2003 and HH, NI, and WE, there was considerable variability about the trend lines (none of which were statistically significant) (Figure 19).

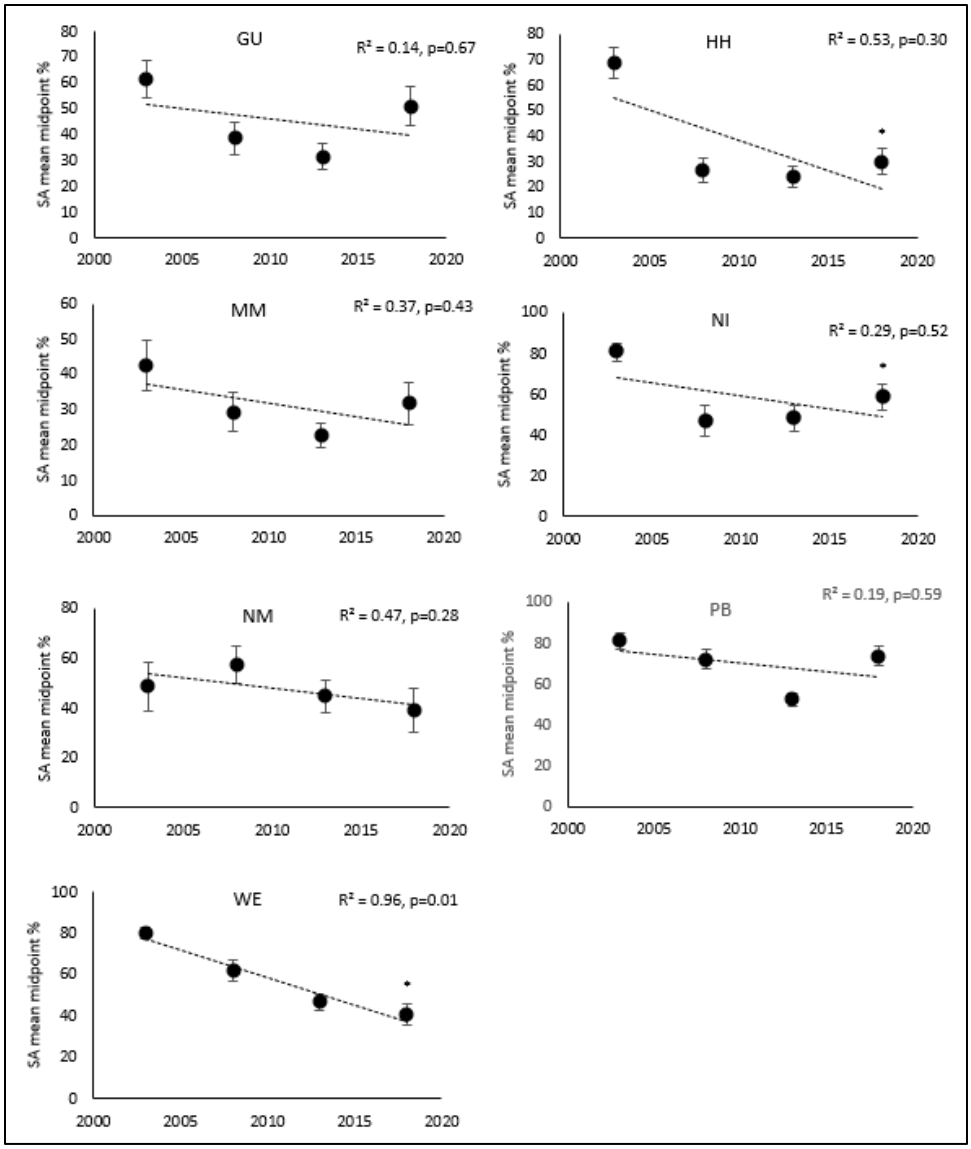

Figure 19. Mean midpoint percentages of *S. alterniflora* (SA) between 2003 and 2018 by site (error bars are standard errors of the means; asterisks indicate a significant difference between 2003 and 2018 values at α=0.05).

S. patens

S. patens declined substantially at MM and WE but demonstrated high temporal variability at the other sites (Figure 20).

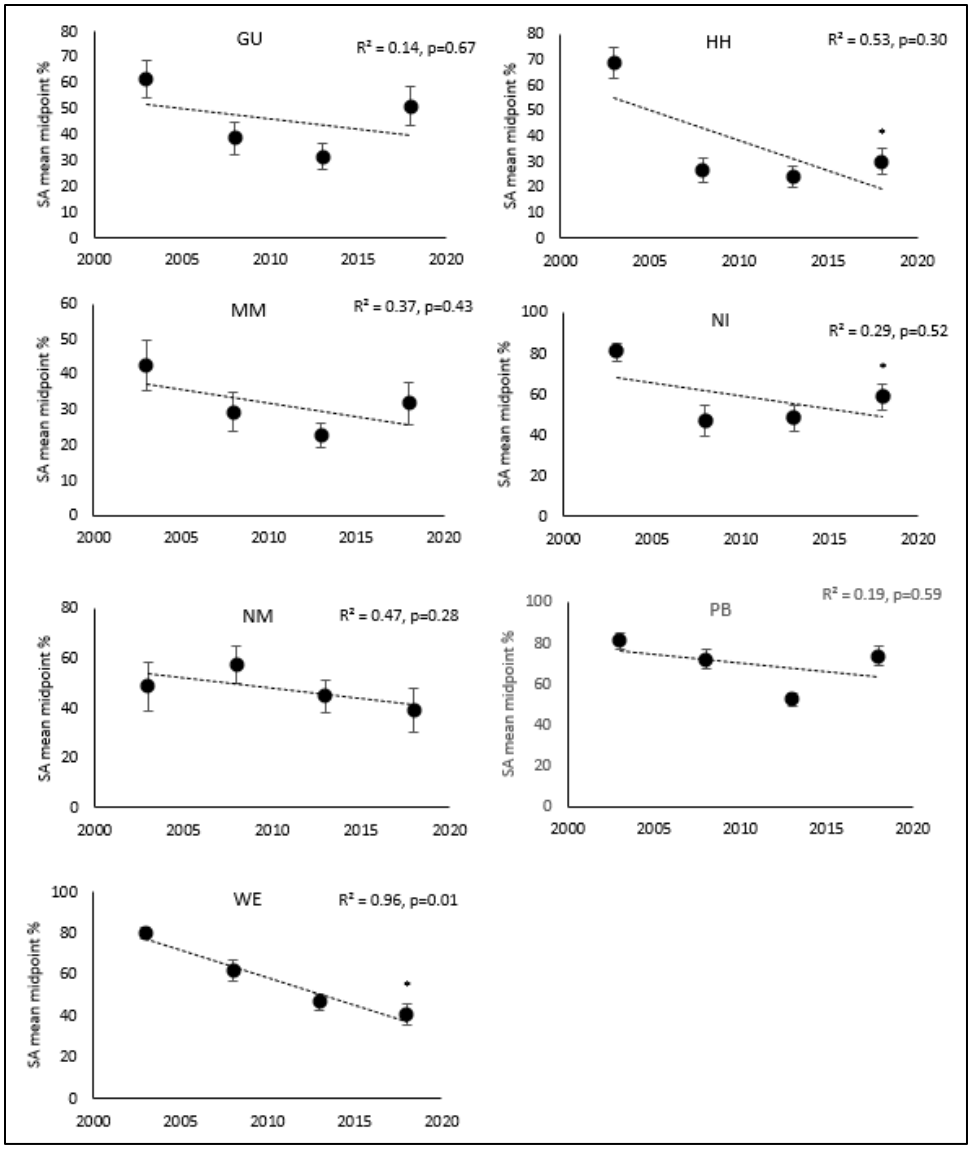

Figure 20. Mean midpoint percentages of *S. patens* (SP) between 2003 and 2018 by site (error bars are standard errors of the means; asterisks indicate a significant difference between 2003 and 2018 values at α=0.05).

Across all sites, the ratio of *S. patens* to *S. alterniflora* in plots where there was a transition from one to the other or a mixture of both, fell quite considerably. This reflects the encroachment of *S. alterniflora* into the high marsh zone, sometimes accelerated by *Sesarma* herbivory, as sea level continues to rise (Figure 21).

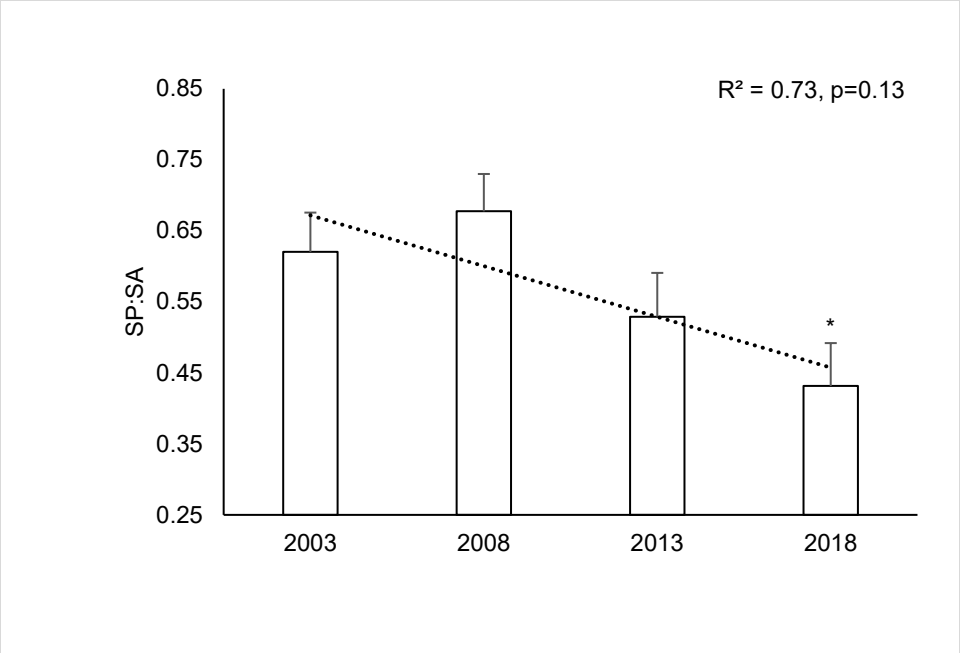

Figure 21. Ratios of *S. patens* (SP):*S. alterniflora* (SA) based on midpoint percentage values between 2003 and 2018 (error bars are standard errors of the means; asterisks indicate a significant difference between 2003 and 2018 values at α=0.05).

Transitional and upland species trends

The survey data make it clear that transitional species inhabiting upper elevations between marsh and terrestrial habitats, as well as species considered solely terrestrial, have been declining since 2003 (Figure 22). Table 7 below lists the frequency of occurrences of such taxa and the change in these values over the last 15 years.

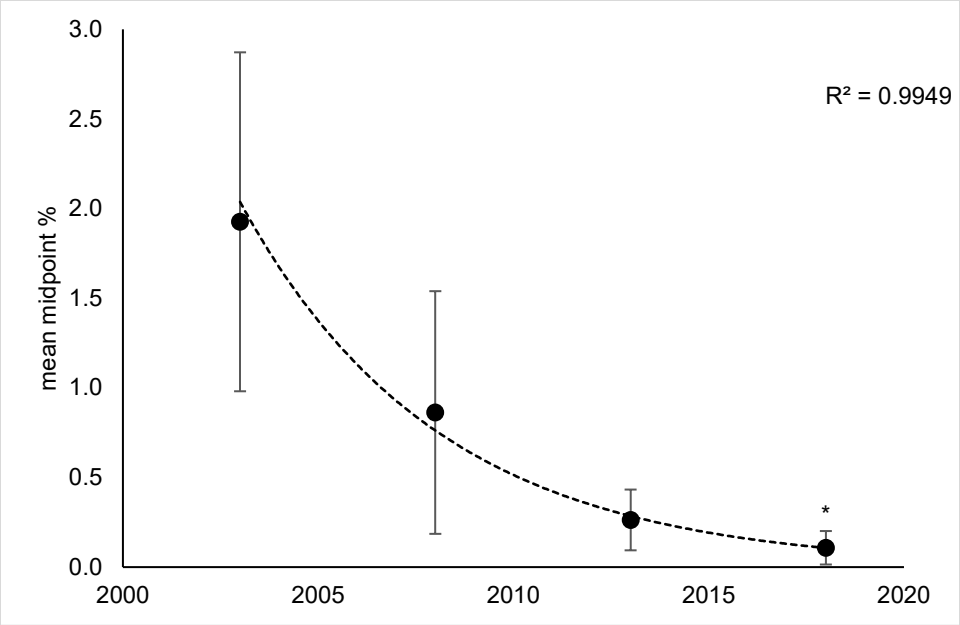

Figure 22. Mean midpoint percentage values for transitional/upland taxa (summed percentages of all taxa) by year (error bars are standard errors of the means; asterisks indicate a significant difference between 2003 and 2018 values at α=0.05).

Table 7. Mean midpoint percentage values of transitional/upland taxa (all species pooled) mean midpoint % cover by year.

Taxon	2003	2018	Difference
Ammophila breviligulata	1.2%	0.6%	−0.6%
Atriplex prostrata	1.2%	0.0%	−1.2%
Baccharis halimifolia	0.6%	0.0%	−0.6%
Cakile edentula	0.6%	0.0%	−0.6%
Elymus repens	1.8%	0.0%	−1.8%
Festuca rubrum	0.6%	0.0%	−0.6%
Juncus gerardii	3.6%	0.6%	−3.0%
Limonium carolinianum	16.4%	11.7%	−4.7%
Morella pennsylvania	0.6%	0.6%	0.0%
Plantago sp.	1.2%	0.0%	−1.2%
Puccinellia maritima	1.2%	0.0%	−1.2%
Rosa rugosa	0.0%	0.6%	0.6%
Salicornia biglovii	5.5%	0.0%	−5.5%
Solidago sempervirens	1.2%	0.0%	−1.2%
Spergularia spp.	1.8%	0.6%	−1.2%
Toxicodendron radicans	0.6%	0.0%	−0.6%

Discussion

Vegetation monitoring since 2003 indicates that CACO's salt marshes have exhibited shifts in taxonomic composition and changes in the abundance of their foundation species, *S. alterniflora* and *S. patens*. Cover values for both taxa are a manifestation of opposing processes that cancel each other out to a certain extent. While these species are being lost from low elevations or creekbank edges, they are also expanding upslope in response to SLR. The net result may be little or no net change. Relative to the interior platform, creekbank edges tend to accumulate sediments faster and are much better oxygenated for plant (particularly root) growth. This tends to make them slightly higher in elevation. However, marsh edges experience significant lateral erosion which explains why *S. alterniflora* appears to be declining at elevations otherwise suitable for growth with respect to flooding stress. Lateral erosion has been most obvious at HH and WE. In contrast, it is disappearing rapidly from the lowest elevations of gently sloped marshes, where the lower limits of vegetation are at the physiological threshold for flooding tolerance. Still more changes across CACO salt marshes can be attributed to *Sesarma* crab grazing, which is a confounding factor in interpreting vegetation changes where they are abundant.

S. patens is mainly disappearing from its seaward edge (which roughly corresponds to mean high tide levels) while being replaced with *S. alterniflora*, or in some cases bare ground. These changes are almost certainly due to SLR, although *Sesarma* grazing is an additional factor at GU and MM. Since 2003, mean high water (MHW) and mean sea-level (MSL) have increased by ~15 cm and 12 cm, respectively (NOAA, Nantucket Harbor) (Figure 23). With limited opportunity to migrate upslope (Smith, in press), high marsh habitat will continue to decline as it has over the past several decades (Smith et al. 2016). Within the long-term monitoring plot network, reductions in *S. patens* were most extensive at MM and WE.

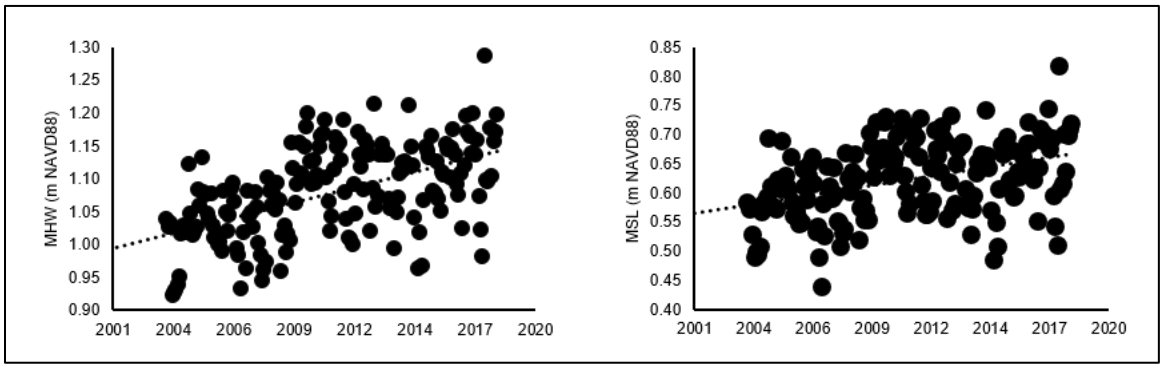

Figure 23. Trends in mean high water (MHW) and mean sea-level (MSL) (NAVD88) between 2003 and 2018.

While there were no statistically significant 15-year trends in *S. alterniflora* plant heights, the spatial variability in this parameter among marshes has been persistent and is informative. Differences in plant heights are likely a manifestation of physiological stress related to site hydrology and sediment properties (Mendelssohn and Seneca 1980). For example, *S. alterniflora* is very short in NI and PB.

These are the oldest marshes within CACO, having formed long before the others (Uchupi et al. 1996). They have the thickest peat accumulations and highest concentrations of hydrogen sulfide (H_2S) in porewaters (Smith 2004). Thus, the vegetation in these areas is probably under considerable stress, and this may render them even more susceptible to SLR (Morris et al. 2013). In contrast, plants in well-drained, highly oxygenated sandy sediments are large and tall, and may be more physiologically robust to SLR. There was a significant increase in plant heights at NM, but this is due to vegetation recolonization and regrowth across the barrier beach overwash that occurred prior to 2003). This analysis also revealed that species richness is declining. This is mainly due to reductions or disappearances of transitional and upland species—ostensibly due to SLR. In most cases, they are being replaced by monospecific *S. alterniflora* as it migrates landward. Some of these species have known wildlife value, such as the *Limonium carolinianum* (sea lavender) (Sei and Porter 2003).

In a broader sense, the power to detect temporal trends in all marsh variables is still limited by sample size given that there are only four data points for regression analyses (representing the 2003, 2008, 2013, and 2018 survey years). Thus, variability about the regression line is still high. One consistent trend, however, was that cover values for almost all floristic variables were very low in 2013. While reasons for this are unknown, clues may lie in climate and sea level conditions during the proceeding years. Based on NOAA tide level data for Nantucket Harbor (tidesandcurrents.noaa.gov), a 5-year rolling means in sea level reveals an interesting temporal trend (Figure 24). Between the first (2003) and third (2013) surveys, MSL increased rapidly, then largely flattened out (including a very low level in 2015) between 2013 and 2018. Thus, it is possible that the steep decline in *S. alterniflora* and *S. patens* between 2003 and 2013 was a consequence of this rapid rise, while a slight "recovery" has occurred during the much gentler rise between 2013 and 2018. Of course, all this is speculation, but it is known that marsh vegetation responds to interannnual fluctuations in sea level (Gross et al. 1990, Teal and Howes 1996). Thus, flooding stress during successive high-water years may be manifested as reduced cover (Mendelssohn and Seneca 1980).

It is unclear whether trends in mean daily temperature and growing-season precipitation have impacted salt marsh vegetation in any way. Data from NOAA (Station KCQX, https://www.ncdc.noaa.gov) shows that between 2008 and 2013 there were no years where precipitation was abnormally high or low. Growing season (April–August) rainfall was relatively high in the 2013 survey (Figure 24), but it is unclear how this would negatively influence growth since rainfall can reduce salinity stress (Gross et al. 1990). Mean daily temperature during April–August was high between 2008 and 2013, but *S. alterniflora*'s geographic range extends south to Florida, so these temperatures are well within the species tolerance range (although the local genotypes are presumably adapted to a colder climate). Notwithstanding, the power to make stronger conclusions about correlations between variables, or lack thereof, will increase substantially as more surveys are conducted every 5 years.

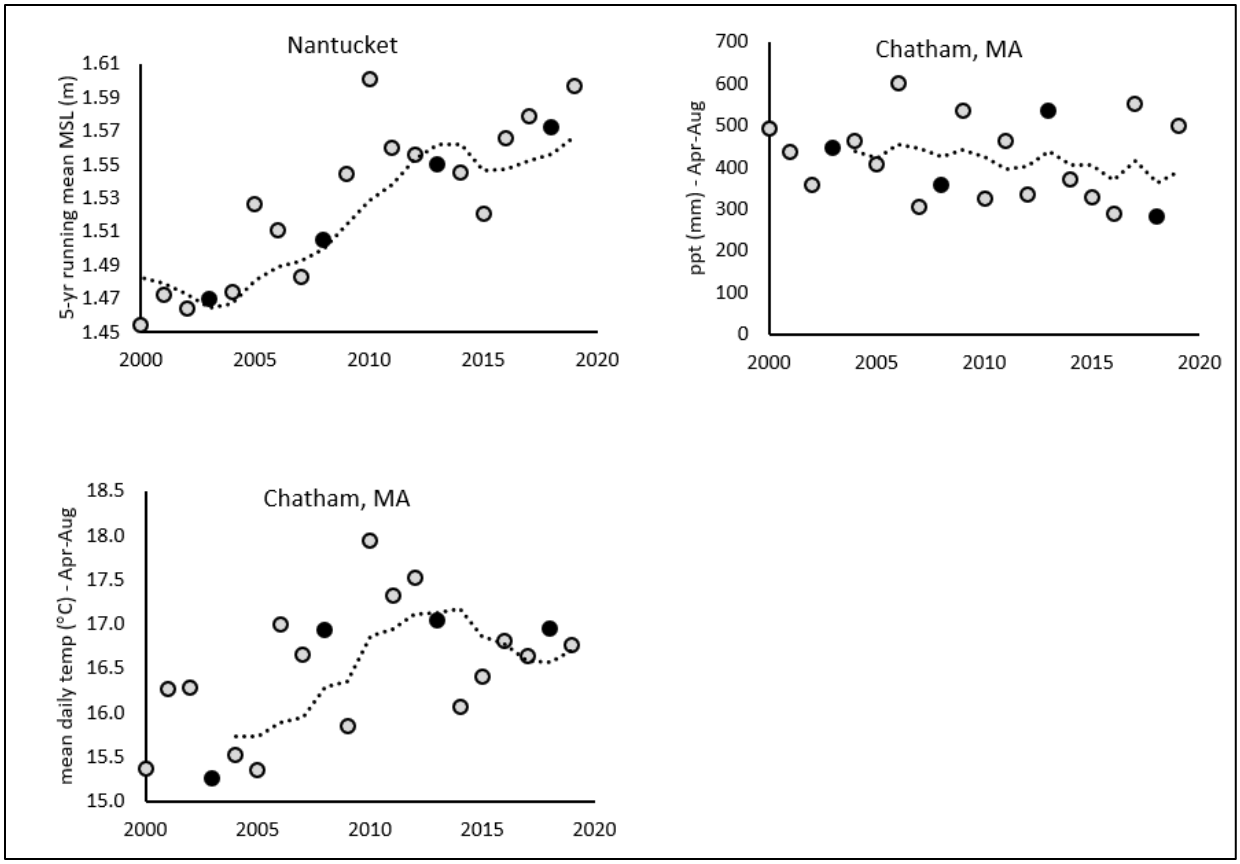

Figure 24. Five-year rolling averages of mean sea level (MSL; top left), growing season precipitation (top right), and mean daily temperature between 2003 and 2018 (bottom left). Black data points correspond with vegetation survey years (2003, 2008, 2013, 2018).

The floristic changes described in this report mirror landscape-scale changes captured in aerial imagery (Smith and Green 2015). For example, Figure 25 below (Middle Meadow, Wellfleet) illustrates the shift rom predominatly high marsh vegetation (light-colored areas; *S. patens*) to low marsh (dark colored areas; almost exclusively *S. alterniflora*). It should be noted that some of this change is the result of *Sesarma reticulatum* herbivory, which also exacerbates the effects of SLR (Smith et al. 2012).

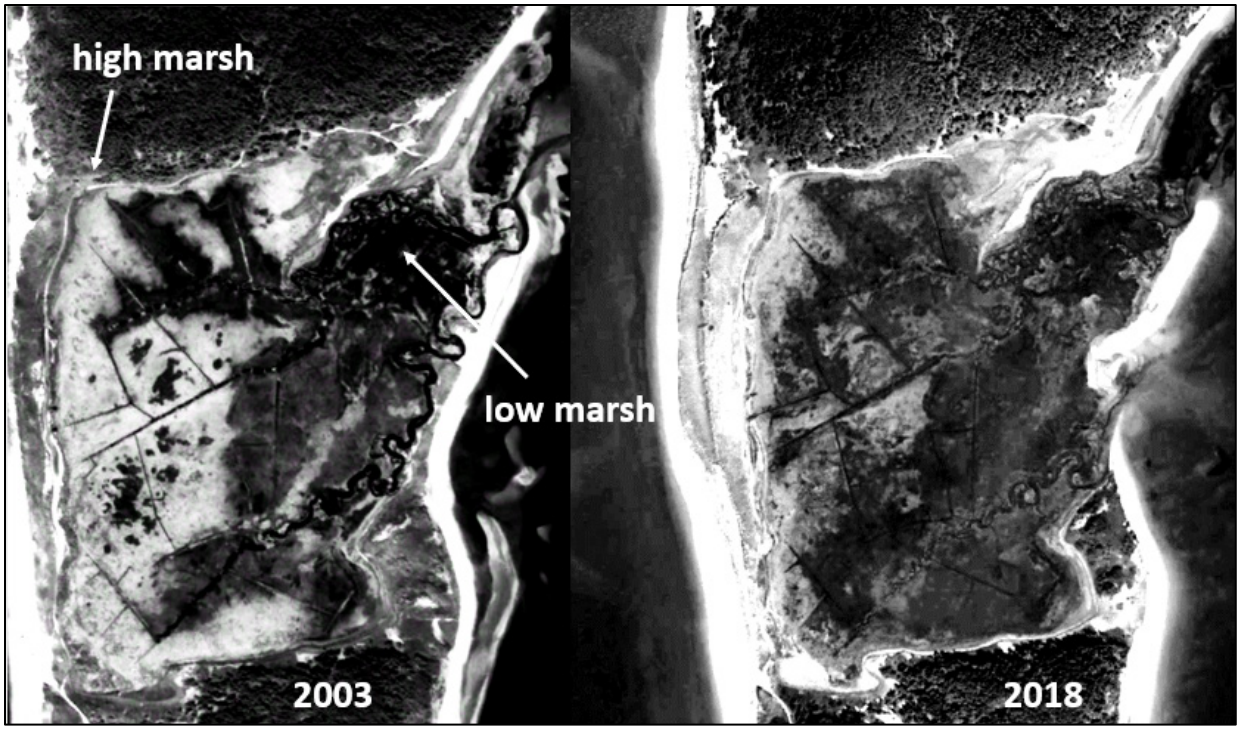

Figure 25. Aerial photography of Middle Meadow in 2003 (left) vs. 2018 (right) showing the replacement of high marsh with low marsh during 15 years of rise in mean sea-level and mean high water of ~12 cm and 15 cm, respectively.

Conclusions

The ground-level field monitoring conducted every five years between 2003 and 2018 has yielded valuable information on salt marsh vegetation dynamics in CACO salt marshes. The power of this protocol to detect trends is still limited by the fact that only four surveys have been completely thus far and that cover values originate from visual approximations within relatively coarse cover class categories, which are then converted to midpoint percentages. With the accumulation of more data points through continued monitoring, however, these trends should strengthen if they are indicative of real long-term, directional change.

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Appendix I: Marsh sites with transects and permanent plot locations

West End marsh

West End marsh is a ~172-acre marsh located at the tip of the Cape Cod peninsula in Provincetown within Cape Cod Bay (Figure 26). A permeable stone dike runs NW-SE along its eastern margin. While the dike has virtually no impact on high and low tide heights it produces a lag in their timing (based on tidal data collected in 2005 by CACO). Currently, there are 7 transects and 58 plots in this marsh.

Figure 26. West End marsh depicting current extent of monitoring plot locations.

Hatches Harbor

Like West End marsh, Hatches Harbor lies at the tip of Cape Cod in Provincetown where the Atlantic Ocean meets Cape Cod Bay (Figure 27). It is approximately 3.3 km northwest of West End. Hatches Harbor is bisected by an earthen dike built in the 1930s, but the 90-acre part of the marsh discussed here is seaward of the dike and hydrologically unrestricted. There are 3 transects and 41 plots within this area.

Figure 27. Hatches Harbor marsh depicting current extent of monitoring plot locations.

Nauset Island

Nauset marsh is a ~700-acre marsh located on the Atlantic side of CACO in the town of Eastham and Orleans (Figure 28). There are different parts of the marsh that are themselves defined by broad tidal channels. The largest section is ~270 acres and is termed Nauset Island (NI) in this report. There are 3 transects and 22 plots within this area.

Figure 28. Nauset Island marsh depicting current extent of monitoring plot locations.

Nauset mainland

A much smaller section (~48 acres) of Nauset marsh extends out westward from the barrier spit connected to the mainland (Figure 29). This section has been termed Nauset mainland (NM) in this report. It has 3 transects and 18 plots.

Figure 29. Nauset mainland marsh depicting current extent of monitoring plot locations.

Middle Meadow vegetation

Middle Meadow is a small (~54 acre) marsh that lies midway between the Gut and Jeremy marshes on the Great Island peninsula along the western edge of Wellfleet in Cape Cod Bay (Figure 30). It is well protected from wave action behind a relatively narrow tidal inlet. There are 34 plots along 6 transects within MM.

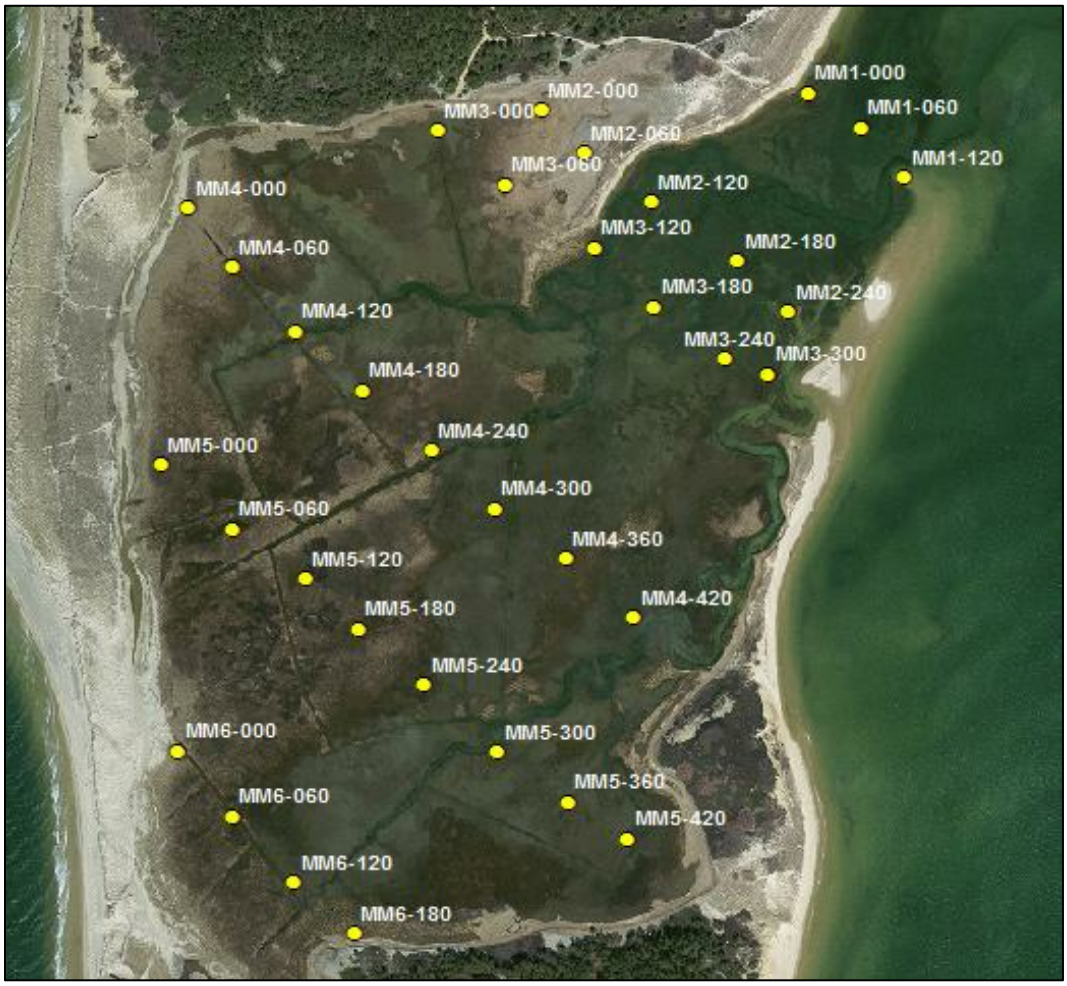

Figure 30. Middle Meadow marsh depicting current extent of monitoring plot locations.

The Gut

The Gut (GU) is a 55-acre marsh also located on the Great Island peninsula but is more of a fringing marsh that is fully exposed to the open waters of Wellfleet Bay (i.e., it does not lay behind a tidal inlet or barrier beach) (Figure 31). There are 7 transects and 47 plots in this marsh.

Figure 31. Gut marsh depicting current extent of monitoring plot locations.

Jeremy marsh

Jeremy marsh is the southernmost marsh on Great Island (Figure 32). At ~11 acres, it is the smallest of all areas monitored. There are 24 plots along 3 transects here.

Figure 32. Jeremy marsh depicting current extent of monitoring plot locations.

Pleasant Bay

Pleasant Bay is the largest marsh within CACO's monitoring network. It is located on the Atlantic side, south of Nauset marsh in the towns of Orleans and Chatham, extending westward from the Nauset spit which encloses Pleasant Bay proper (Figure 33). The marsh is roughly 425 acres and there are 92 plots along 14 transects there.

Figure 33. Pleasant Bay marsh depicting current extent of monitoring plot locations.

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