

Hypsometry of Cape Cod Salt Marshes (Massachusetts, U.S.A.) and Predictions of Marsh Vegetation Responses to Sea-Level Rise

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ABSTRACT

Smith, S.M.; Tyrrell, M.; Medeiros, K.; Bayley, H.; Fox, S.; Adams, M.; Mejia, C.; Dijkstra, A.; Janson, S., and Tanis, M., 2017. Hypsometry of Cape Cod salt marshes (Massachusetts, U.S.A.) and predictions of marsh vegetation responses to sea-level rise. *Journal of Coastal Research*, 33(3), 537–547. Coconut Creek (Florida), ISSN 0749-0208.

The structure and functioning of salt marsh ecosystems are being impacted by sea-level rise, and a major determinant of their vulnerability to this aspect of climate change is their ground surface elevation relative to tide heights (hypsometry). In this study, a comprehensive real-time kinematic (RTK) global positioning system (GPS) survey was conducted within four salt marshes at Cape Cod National Seashore (CCNS) to create digital elevation models, and *in situ* water-level loggers were used to collect tidal data within each system. From these data, marsh surface elevations were calculated relative to mean high tide elevations for 2013 and projected elevation change rates with 50 cm and 100 cm of sea-level rise. Vegetation responses to these scenarios were then modeled based on the relationship of high and low marsh zones to relative elevation. The results suggest that (1) CCNS marshes sit low within their tidal frames, unlike the majority of salt marshes in New England, (2) high marsh areas will be most affected with sea-level rise, with 90–100% losses under both 50 cm and 100 cm sea-level rise scenarios, and (3) total marsh losses of up to 30% could ensue with 100 cm of sea-level rise. Such changes, should they occur, would substantially impact the coastal environment on Cape Cod and profoundly impact the ecosystem services provided by these systems.

ADDITIONAL INDEX WORDS: *High marsh, low marsh, elevation, hypsometry, inundation, NW Atlantic, RTK.*

INTRODUCTION

Salt marshes are some of the most productive ecosystems in the world, and they provide a wide variety of well-documented ecological and socio-economic services (Adam, 1993; Costanza *et al.*, 1998; Nixon, 1982; Nixon and Oviatt, 1976; Shepard, Crain, and Beck, 2011; Teal, 1986; Valiela *et al.*, 2002). Unfortunately, salt marshes have suffered dramatic losses during the last century due to tidal restrictions and various forms of human development (Gedan, Silliman, and Bertness, 2009). These systems are also being impacted by climate change, particularly sea-level rise (SLR; Craft *et al.*, 2009; Reed, 1990).

Hydrology is the primary abiotic factor regulating the structure and function of salt marshes. The extent and duration of tidal flooding control almost every aspect of these systems, including plant species composition, productivity and biomass, carbon sequestration, biogeochemistry, and the movements and distribution of fauna (Bertness, 1991; Kathilankal *et al.*, 2008; Mendelsohn and Morris, 2000; Naidoo, McKee, and Mendelsohn, 1992; Nuttle and Hemand, 1988;

Rozas, 1995; Silvestri, Defina, and Marani, 2005; Watson *et al.*, 2014). Thus, changes in hydrology due to SLR will likely have substantial ecological and physicochemical impacts.

The vegetation of salt marshes generally exhibits distinct zonation that is related to species tolerance to flooding. High marsh plants, which are comparatively flood intolerant, tend to occupy elevations between the mean high and spring high tide lines, while low marsh species dominate between mean low and mean high tides (Bertness, 1991; Bertness and Ellison, 1987). To a certain extent, salt marshes can adjust to SLR by vertical elevation gain (often referred to as accretion) through organic matter (peat) accumulation and sediment deposition (Bastian, Mendelsohn, and Hester, 2012; van Wijnen and Bakker, 2001). However, when accretion rates are at a deficit with respect to increasing water elevations, high marsh will be replaced by low marsh, and the boundary between these zones will shift upslope (Brinson, Christian, and Blum, 1995). With further SLR, plant production will slow (McKee and Patrick, 1988; Mendelsohn and Burdick, 1988; Morris *et al.*, 2002), or marshes may be eradicated quickly due to rapid die-off.

The Intergovernmental Panel on Climate Change (IPCC) predicts a global rise in sea level of 52–98 cm by the year 2100 under a high CO₂ emissions scenario (IPCC, 2013), which is equivalent to a rate of ~6–11 mm/y. Of course, vertical elevation gain during any period of SLR will continuously alter ground elevations relative to tides. While salt marsh

DOI: 10.2112/JCOASTRES1500153.1 received 14 August 2015; accepted in revision 5 April 2016; corrected proofs received 19 May 2016; published pre-print online 14 July 2016.

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accretion rates are highly variable, reported values from the northeast United States range from 1 to 6.4 mm/y, with most values around 2–3 mm/y (Bricker-Urso *et al.*, 1989; Carey *et al.*, 2015; Chmura *et al.*, 2001; Donnelly and Bertness, 2001; Erwin *et al.*, 2006; Orson, Panageotou, and Letherman, 1985; Richard, 1978; Turner, Swenson, and Milan, 2000; Warren and Neiring, 1993).

In marshes of the northeastern United States, the process of vertical elevation gain is mainly attributed to the production of belowground biomass, while inorganic sediment deposition plays a minor role. This is due to the fact that sediment concentrations in tidal waters of this region are typically very low compared to southeastern and Gulf Coast states (Chapman, 1960; Weston, 2014). For example, organic matter accumulation in a Rhode Island marsh accounted for >90% of vertical elevation gain according to Bricker-Urso *et al.* (1989). Turner, Swenson, and Milan (2000, 2002) stated that organic accumulation is five times more important than inorganic accumulation for East Coast salt marshes. Similarly, in the nearby Plum Island Estuary, Massachusetts, Cavatorta *et al.* (2003) suggested that belowground plant production is mainly responsible for marsh accretion. Total suspended solid concentrations on incoming tides have been measured at Cape Cod National Seashore (CCNS) and are in the range of 0.2–18 mg/L, averaging 2.9 mg/L (Stephen Smith, unpublished data), with the higher end of the range generally representing a single marsh that is experiencing crab-related vegetation losses (Smith, 2009) and marsh cannibalism (*cf.* Stevenson, Ward, and Kearney, 1988). The inorganic fractions of these loads are similar to the Plum Island Estuary values of ~20% (Wigand *et al.*, 2015). Furthermore, suspended sediment concentrations are generally decreasing along the Atlantic Coast as a result of best management practices in agriculture, urban development, and vegetation succession on the landscape (Carey *et al.*, 2015; Kennish, 2001; Kirwan and Murray, 2007; Syvitski *et al.*, 2005). As such, they are likely to contribute even less to accretion the future.

Apart from vertical growth, current salt marsh elevations relative to sea level (hypsoetry) will be critical to their responses to future SLR (Couvillion and Beck, 2013; Kirwan *et al.*, 2010). Those at higher elevations relative to tidal range theoretically have a greater potential to survive for a longer period of time—a concept known as “marsh capital” (Cahoon and Guntenspergen, 2010). In this study, the hypsoetry of four salt marshes within CCNS was analyzed using real-time kinematic global positioning system (RTK GPS) elevations and *in situ* water-level data. The current hypsoetry was then used to calculate new hypsoetric profiles under scenarios of +50 cm and +100 cm of SLR (IPCC, 2013) combined with elevation change rates calculated from parabolic curves describing the relationship between plant productivity and elevation relative to mean high tide (hereafter referred to as E_{mht}), and CCNS sediment elevation table (SET) data. These projections were used to predict total losses of marsh as well as changes in the relative extents of high and low marsh.

METHODS

Data on marsh elevation, marsh elevation change rates, and marsh hydrology were collected. Vegetation maps were created

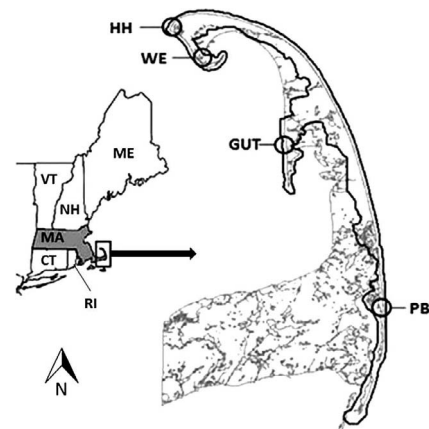


Figure 1. Map of the northeast United States, Massachusetts (shaded), and Cape Cod (within box). A larger view of the enclosed area shows the boundary of Cape Cod National Seashore (dark polygon), within which are the study marshes (HH = Hatches Harbor, WE = West End, GUT = the Gut, PB = Pleasant Bay).

using ARCGIS ver. 10.1 (Smith, 2015a). Additionally, relative elevations (mean high tide elevation – ground elevation = E_{mht}) were calculated, and the seaward extent of the low and high marshes were determined. From these data, marsh elevation maps were created, and subsequent elevation, hydrologic, and vegetation changes were modelled over time under scenarios of 50 cm and 100 cm SLR.

Elevation Surveys

Elevation data (>8000 locations) were collected at four CCNS salt marshes between May and September 2013 using RTK GPS Global Navigation Satellite System (GNSS) receivers (± 1 cm horizontal, ± 2 –4 cm vertical accuracy; Trimble Navigation Limited, Dayton, Ohio). The sites surveyed were Hatches Harbor (Provincetown, 36 hectares), West End (Provincetown, 67 hectares), the Gut (unrestricted Herring River, Wellfleet, 22 hectares), and Pleasant Bay (Orleans, 94 hectares; Figure 1). Surveys were done in base station–rover style fashion, where the base station broadcasts GNSS data out to the rover. Elevations were referenced to nearby benchmarks. The WGS84 ellipsoid model was used to determine vertical and horizontal position. Elevation was surveyed using 20 × 20 m grid spacing across the entire marsh surface to ensure even spacing between points. The survey area within each salt marsh was limited to within the highest and lowest extent of salt marsh vegetation and excluded salt marsh pools or creeks. The National Geodetic Survey (NGS) Geoid 12A (CONUS) model was used to calculate elevations from orthometric heights (North American Vertical Datum of 1988 [NAVD88]), and all points were projected to North American Datum of 1983 (NAD83) Universal Transvers Mercator (UTM) zone 19 using Trimble Business Center software program, version 3.22 (Trimble Navigation Limited, Dayton, Ohio).

Elevation Models

Raster surfaces, or digital elevation models (DEMs), were created from survey points using ArcGIS version 10.1 Spatial

Table 1. Mean high tide elevation (MHT), maximum high tide elevation (M_{xHT}), elevation of the lower limit of *S. alterniflora* (SA LL elev), E_{mht} of the lower limit of *S. alterniflora* (RE SA LL), elevation of the lower limit of *S. patens* (SP LL elev), and E_{mht} of the lower limit of *S. patens* (RE SP LL). All elevations are in meters NAVD88, and values in parentheses are standard deviations of the means (HH = Hatches Harbor, WE = West End, GUT = the Gut, PB = Pleasant Bay; BB = back-barrier marsh, FM = fringing marsh).

Site	Marsh Type	MHT (m)	M_{xHT} (m)	SA LL Elev (m)	RE SA LL (m)	SP LL Elev (m)	RE SP LL (m)
Hatches Harbor	BB	1.23 (0.32)	1.94	-0.07 (0.024)	-1.3	1.12 (0.009)	-0.11
West End	BB	1.52 (0.25)	2.33	-0.21 (0.007)	-1.73	1.09 (0.005)	-0.43
GUT	FM	1.61 (0.26)	2.41	-0.58 (0.009)	-2.19	1.24 (0.006)	-0.37
Pleasant Bay	BB	0.92 (0.18)	1.5	-0.10 (0.009)	-1.02	0.78 (0.008)	-0.14

Analyst (ESRI 2012, Redlands, California). Inverse distance weighting (20×20 m cell size) was used to produce DEMs, using a variable search radius and 12 as the number of nearest points used for each calculation of elevation.

Hydrology

Hydrologic data were collected using four HOBO pressure loggers (Onset Corporation, Onset, Massachusetts) programmed to acquire data every 15 minutes between May and October of 2013. Loggers were placed at the lower (seaward-most) limit of marsh vegetation on the marsh surface to capture every high tide within the study marshes for ~ 90 days. Water depths were calculated through standard conversions of the pressure readings that were compensated for changes in atmospheric pressure. Elevations of the loggers were acquired by RTK GPS so that water surface elevations could be referenced to NAVD88. Mean high tide (MHT) elevations were calculated as the mean values of every high tide (twice per day) during the period of record.

Vegetation Mapping

The extents of high and low marsh vegetation were delineated in ArcGIS (version 10.1) from August 2013 8-band (400–450 nm, 510–580 nm, 585–625 nm, 630–690 nm, 450–510 nm, 705–745 nm, 770–895 nm, 860–1040 nm) georeferenced satellite imagery with 0.5 m spatial resolution (purchased from DigitalGlobe, Longmont, Colorado; Smith, 2015a). In the NW Atlantic, high marsh vegetation is primarily composed of *Spartina patens* (salt marsh hay) and to a lesser extent (estimated at $\sim 5\%$) *Distichlis spicata* (spike grass), whereas the low marsh is dominated by *Spartina alterniflora* (smooth cordgrass). High and low marsh vegetation zones can be readily distinguished as there is typically an abrupt transition between the two, with the former having a very light-colored signature and different appearance due to the shorter stature and growth for *S. patens* as compared to *S. alterniflora*. Interactive supervised classification using training sites ($n = 10$ per type) placed in known areas of high marsh and low marsh was used to delineate the different vegetation zones, which were ground-truthed and exhibited an accuracy of $>88\%$ (Smith, 2015a). The resulting rasters were clipped using masks to constrict the interpolated surface to marsh area only. These rasters were converted to polygon features, and the areal extents of high and low marsh were then computed.

E_{mht} Calculations

Local MHT elevations were calculated for each site from the tidal data. Subsequently, the elevations of all survey points

relative to MHT (E_{mht}) were calculated as the difference between the ground surface and MHT.

Determination of Low and High Marsh Seaward E_{mht} Limits

For delineation of the lower limits of *S. alterniflora* in each marsh, the lowest 20 RTK elevations of the seaward-most vegetated edges were averaged and converted to elevations relative to MHT elevation (Table 1). These points all fell in areas of *S. alterniflora* that were growing below the marsh platform in the muddy or sandy slopes that transitioned to unvegetated mudflat. For delineation of the lower limits of *S. patens*, there were very few survey points that fell on the seaward-most edges of these zones; thus, the elevations of 20 randomly located points (placed using ArcGIS randomization tools) along these borders were extracted, averaged, and converted to E_{mht} (Table 1).

Elevation Change

Elevation change values were derived from a number of different variables. SET data collected within *S. alterniflora* areas for ~ 15 years in three different marshes within CCNS provided the foundation for the calculations of elevation change rates within each system. Average elevation change rates, calculated from three SET instruments in each marsh were: 1.48 mm/y from the Gut (2000–14), 0.95 mm/y for Hatches Harbor (1999–2014), and 4.08 mm/y at Nauset Marsh (1999–2014). Because SET elevation change data were only available for Hatches Harbor, the Gut, and Nauset Marsh (the latter not analyzed in this study), the Gut elevation change rate was used for West End, and the Nauset Marsh rate was used for Pleasant Bay. These pairings were based on the following; the Gut and West End both border Cape Cod Bay, have similar MHT elevations (within 9 cm), and are statistically similar from the standpoint of plant aboveground and belowground biomass and soil organic matter (Smith, 2015b). Nauset Marsh and Pleasant Bay are also comparable in this regard (Smith, 2015b), as both have MHT within 7 cm, and both are older, peaty marshes on the Atlantic Coast only 7.5 km apart.

The projected elevation change rates in this model follow Morris, Sundberg, and Hopkinson (2013), Kirwan and Guntenpergen (2012), Burns (2015), and Janousek *et al.* (2016) who defined parabolic relationships between salt marsh plant productivity and elevation for South Carolina, Maryland, Virginia, and California marshes, respectively. This kind of relationship has also been used by Swanson *et al.* (2014) as part of the WARMER model of marsh change in San Francisco, California. It has been shown that belowground biomass is very closely related to aboveground biomass (Cunha, Asmus, and

Table 2. Parabolic equations used to estimate spatial variations in elevation change rates (y variable in equation is calculated elevation change rate) across each marsh system (SET rate SA = measured elevation change rate within *S. alterniflora*, SET rate SP = derived elevation change rate within *S. patens*, MHT = mean high tide, M_{xHT} = maximum high tide).

Site	SET rate SA (mm/y)	Elevation change rate within SA (mm/y) where $x = MHT$	SET rate SP (mm/y)	Elevation change rate within SP (mm/y) where $x = M_{xHT}$
Hatches				
Harbor	0.95	$y = (-0.0015x^2 - 0.0014x + 0.0006) \times 10^3$	0.48	$y = (-0.0014x^2 - 0.0017x - 4E-05) \times 10^3$
West End	1.48	$y = (-0.0012x^2 - 0.0014x + 0.0011) \times 10^3$	0.74	$y = (-0.002x^2 - 0.0028x - 0.0002) \times 10^3$
Gut	1.48	$y = (-0.0006x^2 - 0.0008x + 0.0012) \times 10^3$	0.74	$y = (-0.0057x^2 - 0.0111x - 0.0047) \times 10^3$
Pleasant Bay	4.08	$y = (-0.0123x^2 - 0.009x + 0.0035) \times 10^3$	2.04	$y = (-0.0089x^2 - 0.0046x + 0.002) \times 10^3$

Costa, 2005; Kang et al., 2013). In fact, Gross et al. (1991) showed that linear regression of above:belowground biomass had correlation coefficients >0.85 at sites ranging from Nova Scotia to Georgia, and belowground productivity itself drives vertical gain in established marshes, particularly in the northeastern United States (Bricker-Urso et al., 1989; Chmura, 2013; Turner, Swenson, and Milan, 2000). Morris, Sundberg, and Hopkinson (2013) found that *S. alterniflora* productivity was maximal at an elevation $\sim 60\%$ of MHT elevation and declined sharply on either side of the maximum. Kirwan and Guntenspergen (2012) reported a similar parabolic relationship for *S. patens* belowground production with elevation, although hydrology data were not collected in their study.

The parabolic elevation change rate curves developed for each marsh in this study were bound on one side by the E_{mht} of the average lower limit of vegetation within each marsh (which was below the marsh platform, thus representing the true physiological limit of *S. alterniflora*), beyond which plant biomass, and therefore vertical elevation change from plant productivity, is zero. The other side of the *S. alterniflora* curve was bound by an upper limit described by Morris, Sundberg, and Hopkinson (2013) of approximately $MHT + 0.76 \times MHT$. In addition, it has been found that this species will not grow at higher elevations with hypersaline soils and infrequent

flooding, regardless of competitive inferiority to *S. patens* (He, Cui, and An, 2012). The other two points on the curve were (1) the average E_{mht} of the SETs (three sites per marsh) (x -axis) with their average elevation change rates (y -axis) and (2) a value on the x -axis at 60% MHT corresponding to a value on the y -axis (elevation change rate) that forced an exact fit of the parabola with an R^2 of 1.0. In other words, the maximum elevation change rate was set so that the parabola peaked at the optimal E_{mht} for plant productivity (as reported by Morris, Sundberg, and Hopkinson, 2013; Table 2; Figure 2).

The elevation change rate in the high marsh (virtually all *S. patens*) was assumed to be half that of *S. alterniflora*, which is supported by the work of Bricker-Urso et al. (1989) (Rhode Island), Morris and Sundberg (2006a,b) (Plum Island Estuary, Massachusetts), and Morris, Sundberg, and Hopkinson (2013) (also in Plum Island Estuary, Massachusetts). Maximum productivity of this species was assumed to occur at the midpoint of its elevation range, with elevation change rates symmetrically falling to zero at the lower elevation limit of *S. patens* (which was below the landward limit of *S. alterniflora*) and at upper elevation limits of *S. patens*. The lower elevation limit for this species was defined as the point at which the high marsh would be flooded by every high tide (i.e. minimum high tide elevation) and therefore could not persist under such

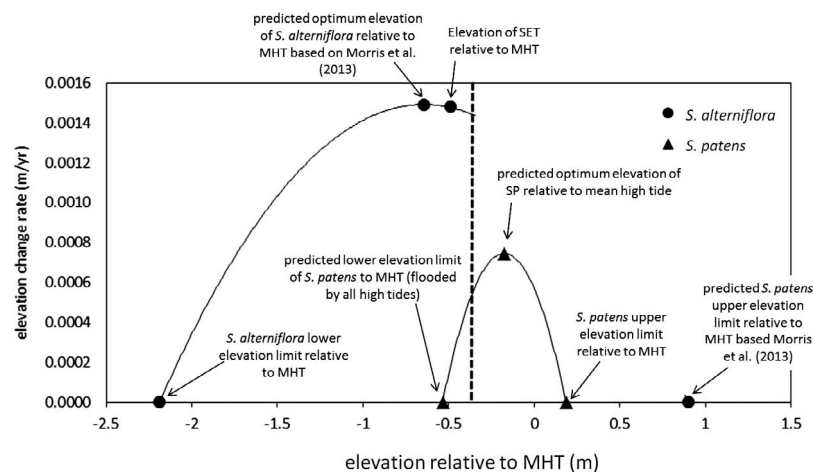


Figure 2. Example of an elevation change rate parabolic curve for the Gut within the *S. alterniflora* and *S. patens* zones. The curve for *S. alterniflora* was set by the field-determined lower limit, the measured accretion rate within *S. alterniflora* at the corresponding SET site (with a known E_{mht}), and a maximum accretion rate that corresponds to 60% of the MHT elevation. The curve for *S. patens* was determined by a theoretical lower limit (elevation that is flooded by every high tide), an accretion rate half that of *S. alterniflora*, and a maximum accretion rate occurring at the midpoint elevation of its range.

conditions (Bertness, 1991). The upper elevation limits for *S. patens* in each marsh were obtained from the field surveys demarking its landward border. Even though there may still be substantial *S. patens* growth here, the likelihood of plant material being permanently added to the soil for elevation gain is extremely low, since it is rapidly decomposed in these very sandy soils that are rarely flooded. In fact, numerous soil cores from these areas have revealed that they contain virtually no organic matter (Smith, unpublished data). This agrees with the finding of Kirwan and Guntenspergen (2012), who documented a decline in *S. patens* belowground productivity at its highest elevations.

A critical point here is that the landward zero-productivity/elevation-gain point for *S. alterniflora* extends upslope well into *S. patens* habitat, noting that from a physiological standpoint, the former can grow further upslope from where it is found in the field. This is simply because *S. alterniflora* is outcompeted by *S. patens* at these elevations (Bertness, 1991) and not because it is incapable of growing there. As such, elevation change rates never reach zero at or near the transition between the two species (although there is a considerable drop in rate when the vegetation changes to *S. patens*) as the model simply switches from *S. alterniflora*- to *S. patens*-based elevation gain when MHT reaches a specific height.

As mentioned previously, there is no separate term for vertical gain through inorganic sediment deposition. However, the elevation change rates from the SET data do include mineral components and are in this way integrated into the model, as these values helped force the parabolas.

E_{mht} and Vegetation Change over Time

The model was run using MS Excel (2010) in 10 year step intervals from 2013 to 2100. The amount of elevation change that occurred over the previous 10 years at each point in the original RTK grid was calculated from the derived elevation change rate from the corresponding parabolic equation for that particular marsh zone (*i.e.* either high or low marsh; Table 1; Figure 2). This amount of elevation change was then added to the previous elevation and then subtracted from the new MHT elevation at that time following a linear rise of either 50 cm or 100 cm of SLR by 2100 to determine the new E_{mht} at each point in time.

The type of marsh vegetation (high or low marsh), which regulates elevation change rate, at each point in time was determined by the E_{mht} of the current lower limits of low and high marsh areas. In other words, where the E_{mht} of a high marsh location fell below the minimum E_{mht} value for high marsh habitat, it then changed to low marsh, and the new elevation change rate for low marsh was then applied for the next 10 years. Over time, locations with E_{mht} values that fell below the current lower limit of *S. alterniflora* were assumed to undergo vegetation loss, with no more capacity for elevation gain.

Losses of high marsh vegetation due to tidal submergence represent net losses of this habitat in this analysis, since no migration into the upland was built into the model. Opportunities for upland migration at CCNS are limited due to the highly compacted soils made unsuitable for plant

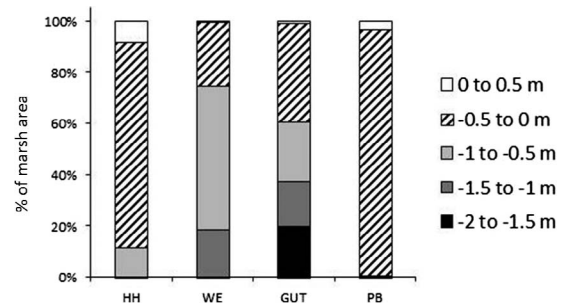


Figure 3. Percentages of marsh area by site in 2013 within 0.5 m E_{mht} interval categories (total range of -1.5 m to 0.5 m relative to MHT).

growth by foot and vehicular traffic at the upper fringes, as well as the steepness of the surrounding dune slopes. Low marsh gains at the landward edge were the inverse of high marsh losses; *i.e.* high marsh was simply replaced by the equivalent amount of low marsh. Net low marsh change was computed as the difference between landward-edge gains and seaward-edge losses. Total marsh loss was calculated from the amount of seaward-edge low marsh loss relative to the total area of marsh.

Calculation of Hydrology

Linear relationships representing 50 cm and 100 cm of SLR were constructed and used at 10 year intervals to spatially define the hydrology of each marsh based on each new ground elevation change.

Calculation of Vegetation Type for the 50 cm and 100 cm SLR Scenarios

New DEMs of each site were created from the final E_{mht} data at the 87 year mark (year 2100), using inverse distance weighting interpolation, and then clipped to the extent of each marsh, minus tidal creeks and pools. The interpolated rasters were reclassified by the lower limits of high and low marsh vegetation and converted to polygon features from which the areal extent of each type was calculated (including conversion of low marsh to mudflat). From these data, percent changes in each marsh type under scenarios of 50 cm and 100 cm of SLR were calculated.

RESULTS

Characterization of baseline hypsometric, hydrologic, and vegetation conditions (*i.e.* 2013) provided a basis for modelling changes in elevation, tide heights, and vegetation change over a period of 87 years (until the year 2100).

Tide Heights and Hypsometry

The pressure logger data showed that MHT elevations varied considerably among sites (Table 1), illustrating the importance of collecting local tide data rather than relying on regional tide gauge data. The NAVD elevation of MHT was highest at the Gut (1.61 m) and West End (1.52 m) and lowest in Pleasant Bay (0.92 m), while Hatches Harbor was intermediate at 1.23 m. The total difference in tidal range among all sites was 0.69 m NAVD. The Gut is a fringing

Table 3. Percent changes in marsh vegetation and total marsh area within the 2013 marsh area footprint with 50 cm and 100 cm of SLR and site (HH = Hatches Harbor, WE = West End, GUT = the Gut, PB = Pleasant Bay; HM = high marsh, LM = low marsh, NM = no marsh or mudflat).

Location	2013		50 cm			100 cm		
	HM	LM	HM	LM	NM	HM	LM	NM
Baseline								
HH	12	88	0.4	100	0.0	0.0	77	23
WE	10	90	0.7	97	2.0	0.0	70	30
GUT	20	80	0.3	92	7.9	0.0	73	27
PB	5	95	3.1	97	0.0	0.0	94	6
Change								
HH			-96	13	0.0	-100	-12	23
WE			-93	8	2	-100	-22	30
GUT			-99	15	8	-100	-8	27
PB			-38	2	0.0	-99	-0.8	6
Average % change			-82	10	2	-100	-11	21
Total change (ha)			-15.2	12.1	2.9	-18.2	-17.5	35.7
Total % change			-86	7	1	-100	-10	18

marsh on the edge of a wide river, whereas Pleasant Bay is a back-barrier marsh lying within an embayment, where hydrography is controlled by a long barrier spit that substantially dampens tidal amplitudes.

In 2013, the largest percentages of E_{mht} values among all sites occurred between 0 and -1 m (Figure 3). Pleasant Bay and Hatches Harbor had the highest percentages of marsh in the -0.5 to 0 m range, while West End and the Gut had substantially more area below -0.5 m. West End and the Gut also had a high percentage of area within the -1 m to -0.5 m range compared to the other sites. The Gut was unique in that it was the only marsh to have any E_{mht} values below -1.5 m. All marshes had less than 8% of their elevations above MHT, and none of the sites had areas above 0.5 m.

Lower Limits of *S. alterniflora* and *S. patens*

The lower limit of *S. alterniflora* ranged between -0.07 m (Hatches Harbor) and -0.58 m NAVD88 (The Gut; Table 1). The lower limit of *S. patens* ranged between 0.78 m (Pleasant Bay) and 1.24 m NAVD88 (The Gut). With respect to MHT, lower E_{mht} limits of *S. alterniflora* ranged between -1.02 m (Pleasant Bay) and -2.18 m (The Gut). The Gut lower marsh limit was similar to the -1.82 m NAVD88 below MHT value found by Redfield (1972) in Barnstable Marsh (also located on Cape Cod Bay; cited in McKee and Patrick, 1988). The lower E_{mht} limits of *S. patens* were less variable, ranging between -0.11 m (Hatches Harbor) and -0.43 m (West End). This indicates that *S. patens* can occur below the MHT elevation, but its elevation distribution varies by site (Table 1), presumably due to variations in edaphic conditions and local competition with *S. alterniflora*.

Vegetation Composition in 2013

Low marsh dominated all four salt marsh sites, accounting for greater than 80% of the area (Table 3). The Gut had the highest percentage of high marsh vegetation, with 20% of its area within this zone (Table 3), while Pleasant Bay had the lowest amount at 5%. High marsh percentages at Hatches Harbor and West End were 12% and 10% of the total area, respectively.

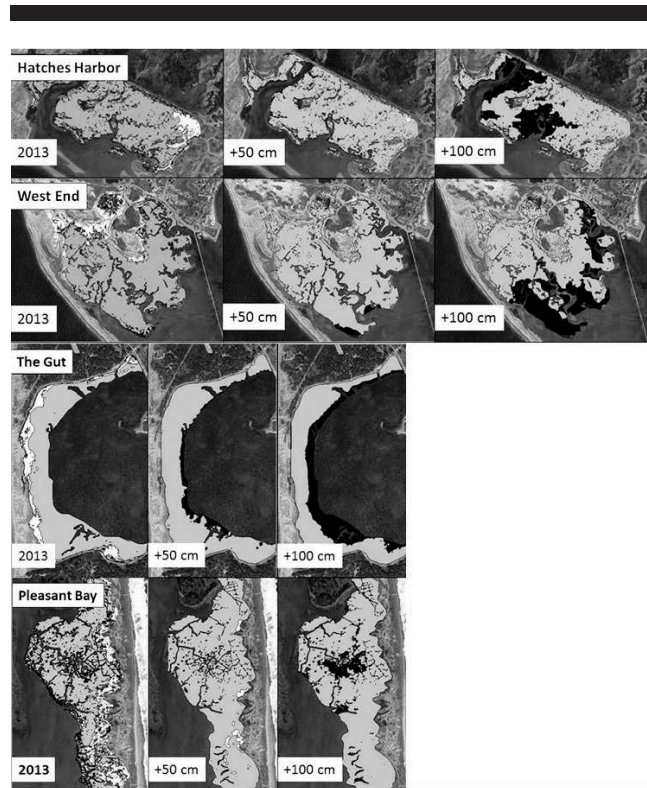


Figure 4. Maps of high marsh (white), low marsh (gray), and no marsh (marsh loss; black) by site in 2013 and after 50 cm and 100 cm of SLR.

Changes in Vegetation in Response to SLR

In the 50 cm SLR scenario, by 2100, Hatches Harbor and West End may lose 96% and 93% of their high marsh area, respectively (Table 3; Figure 4). The Gut may undergo an almost total loss of 99%, while Pleasant Bay may lose 38% of its current high marsh area. In contrast, net low marsh gains of 2% (Pleasant Bay) to 15% (Gut) could occur under this lower SLR scenario. Seaward-edge marsh losses (the conversion of low marsh to mudflat) may occur at the Gut (8% of the total area) and West End (2% of the total area; Table 3; Figure 4).

With 100 cm SLR, all sites are expected to lose ~100% of their high marsh habitat by 2100 (Table 3; Figure 4). Further, under this scenario, all sites may suffer variable net losses in low marsh area as compared to current conditions, ranging between ~1% (Pleasant Bay) and 22% (West End). Seaward-edge low marsh losses are predicted to be similar for Hatches Harbor, West End, and the Gut, ranging between 23% and 30% of the total area. At Pleasant Bay, only 6% (4.9 hectares) of the total marsh area may be converted to mudflat (Table 3; Figure 4).

The average amount of high and low marsh change with 50 cm SLR across all systems is predicted to be -82% and +10%, respectively, while conversion to unvegetated mudflat affected on average over 2% of the total area. With 100 cm SLR, these values are -100% (high marsh), -11% (low marsh), and 21% (low marsh converted to mudflat). In terms of total change (all sites) for the 50 cm SLR scenario, high marsh may be reduced by 86% (15.2 hectares), while low marsh could exhibit a small

Table 4. Marsh vulnerability rankings by SLR scenario (50 cm or 100 cm by 2100) and metric of change (HH = Hatches Harbor, WE = West End, GUT = the Gut, PB = Pleasant Bay; LM = low marsh).

	50 cm	100 cm
Loss of high marsh	GUT > HH > WE > PB	HH = WE = GUT ≥ PB
Loss of low marsh	All sites gain LM (net)	WE > HH > GUT > PB
Conversion to mudflat	GUT > WE > HH = PB	WE > GUT > HH > PB

net gain of 7% (12.1 hectares; but this included a predicted 1% loss from its seaward edge of 2.9 hectares). With 100 cm SLR, 99.8% (18.2 hectares) of the high marsh may be lost, and net losses from the low marsh may cumulatively total 10% (17.5 hectares) of current area. The predicted conversion of low marsh to mudflat constitutes 18% of the original total marsh area (35.7 hectares; Table 2).

Using the model output data, the sites were ranked by their vulnerability to high/low marsh area losses and low marsh area converted to mudflat (which represents overall marsh vegetation loss) under the two different SLR scenarios based on percent change (Table 4). With 50 cm of SLR, the Gut is the most vulnerable to high marsh losses, followed by Hatches Harbor, West End, and Pleasant Bay. None of the sites is vulnerable to net low marsh losses because the amounts of low marsh replacement of high marsh are larger than the losses incurred at the seaward edge. However, in terms of only seaward-edge low marsh loss for the 50 cm SLR scenario, the Gut is most vulnerable, followed by West End and Hatches Harbor, while Pleasant Bay exhibited no loss to total marsh area within the footprint of the existing marsh area.

With 100 cm of SLR, every site undergoes almost a complete loss (99–100 %) of high marsh habitat; thus, all are extremely vulnerable with respect to this vegetation type. West End and Hatches Harbor are most vulnerable to net low marsh losses, followed by the Gut and Pleasant Bay (Table 4). For seaward-edge losses alone, the West End and Gut marshes are most vulnerable, followed closely by Hatches Harbor. Pleasant Bay is the least vulnerable with respect to loss of vegetated marsh habitat at the seaward edge under the higher SLR scenario. The disparities in these ranks reflect the idea that vulnerability is very context-specific.

DISCUSSION

The four CCNS marshes analyzed in this study had variable hypsometric profiles but are generally situated low within their tidal frame and have a very high percentage of low marsh compared to high marsh. This is markedly different from the salt marshes of New Hampshire or Maine, where there is typically a much higher ratio of high to low marsh (Nixon, 1982). The distribution of E_{mht} values was also variable, with the Gut and West End having high percentages of low E_{mht} values. The lower E_{mht} limit of seaward vegetation varied widely among sites, with 1.15 m and 0.32 m total ranges for *S. alterniflora* and *S. patens*, respectively. These limits are likely related to the large differences in tidal amplitudes among systems (McKee and Patrick, 1988), as well as variable soil properties and other geomorphic factors. These factors not-

withstanding, it appears that CCNS salt marshes are exceedingly vulnerable to SLR given the almost total loss of high marsh under the most conservative scenario and up to 30% conversion to mudflat with 100 cm of SLR. Other marsh areas in Rhode Island and New York (Watson *et al.*, 2014) and North Carolina (Voss, Christian, and Morris, 2013) have also been characterized as highly vulnerable.

The lack of an opportunity for salt marsh vegetation to migrate into the upland in these model scenarios affected the results regarding percent and area of high marsh vegetation remaining in 2100. Nevertheless, the projected high marsh losses were extremely large in this study, even with the lowest rate of SLR. High/low marsh zonation shifts in response to SLR have been observed in recent decades in the northeastern United States (Brinson and Christian, 1999; Civco, Kennard, and Lefor, 1986; Donnelly and Bertness, 2001; Orson and Howes, 1992; Smith, 2009; Smith, Medeiros, and Tyrrell, 2012; Warren and Neiring, 1993). Moreover, high marsh vegetation has already declined substantially over the last few decades within CCNS (Smith, 2015a) and nearby in Rhode Island (Raposa *et al.*, 2015). Loss of high marsh vegetation has substantial ecological implications, given that this zone harbors specialized assemblages of plants, insects, macroinvertebrates, reptiles, and is used frequently by numerous mammals and birds (Nixon, 1982). The salt marsh sparrow (*Ammodramus caudacutus*) and four other tidal marsh obligate nesting bird species that require high marsh habitat for reproduction (BirdLife International, 2012) are on the International Union for the Conservation of Nature (IUCN) Red List. In fact, Shriver *et al.* (2015) found that populations of *A. caudacutus* and *Ammodramus nelson* (Nelson's sparrow) even fluctuate with interannual changes in sea level. Many such organisms cannot survive in or properly utilize low marsh habitat due to its flooding frequency. In addition, high marsh species, particularly *Spartina patens* (salt marsh hay), can play an important role in carbon sequestration and marsh accretion (Chmura *et al.*, 2003; DeLaune and Pezeshki, 2003; Linthurst and Reimold, 1978; Morris *et al.*, 2002). Changes in the relative proportions of high vs. low marsh vegetation zonation will therefore result in significant changes in marsh ecosystem services (Craft *et al.*, 2009) and biodiversity.

Aside from species composition shifts and vegetation losses, there are also likely to be changes in the productivity and biomass of salt marsh plants within these systems. The mere persistence of vegetation during accelerated SLR does not necessarily imply healthy physiological conditions, as there are optimal ground-surface elevations relative to tidal elevations for plant productivity (Morris *et al.*, 2002; Morris, 2007; Morris, Sundberg, and Hopkinson, 2013; Scott, 2010). As sea level rises, the productivity of *S. alterniflora* at supra-optimal elevations may initially increase, but then decrease when the flooding frequency becomes physiologically stressful (Morris, Sundberg, and Hopkinson, 2013). Voss, Christian, and Morris (2013) and Watson *et al.* (2014) reported a positive linear relationship between aboveground biomass and elevation for this species in North Carolina and New York–southern New England, respectively. However, because the elevation change rate for *S. alterniflora* switches to that of *S. patens* very close to the peak of the parabolic curves for each marsh, in reality, our

model follows a very similar pattern, *i.e.* declining productivity with decreasing elevation. In other words, there are very few circumstances where there is any significant reduction in productivity of *S. alterniflora* with increasing elevation for the marshes analyzed in this study, simply because it is replaced by *S. patens* before this occurs.

With respect to the latter point, Kirwan and Guntenspergen (2015) reported a linear relationship between aboveground *S. patens* and elevation in mesocosm experiments conducted in a mesohaline setting. However, in an earlier paper (Kirwan and Guntenspergen, 2012), they found a parabolic relationship for *S. patens* belowground productivity and further suggested that “the hump-shaped response of root production to inundation that we measured is likely to be accompanied by a hump-shaped response in the rate of organic accretion and elevation change” (p. 765). Regardless, changes in the areal extent and geographic position of suitable *vs.* unsuitable hydrologic conditions would result in changing patterns of primary production. Moreover, a decline in productivity at suboptimal elevations may result in a positive feedback loop, whereby stressed plants become even less resilient to increasing water levels, and rapid decline or die-off occurs.

Salt marsh vulnerability to SLR also depends upon opportunities to migrate into adjacent uplands. Although not addressed by the study methodology, there is realistically little opportunity for this to occur at the study locations due to recreational use (highly compacted vehicle and foot paths) and topography (steep adjacent dunes and banks). Furthermore, at Hatches Harbor and West End, there are roads and other infrastructure at the landward borders of the marshes that prevent expansion into the uplands. Pleasant Bay is the only marsh within CCNS where there is potential for some transgression in an eastward direction (Smith, 2015a) toward the barrier beach, although the barrier beach itself is continuously eroding in a westward direction due to waves on the Atlantic Ocean side.

To a certain extent, the predictions made here will overestimate marsh losses if primary productivity is stimulated by increases in atmospheric CO₂ concentrations and/or temperature (Kirwan *et al.*, 2010; Langley *et al.*, 2009). In addition, SET rates observed over the last 15 years at CCNS may not portend future rates. For example, changes in precipitation, including summer droughts (White and Alber, 2009), may influence vegetation abundance and therefore organic matter accumulation. Drought was shown to increase the productivity of *S. alterniflora* and *S. patens* at Plum Island, Massachusetts (Charles and Dukes, 2009). It should also be noted that while belowground biomass contributes to soil organic matter accumulation, accretion integrates turnover and decomposition as well, and it is physically influenced by burial and compaction.

One of the limitations of this model is that there is no distinct term for sediment deposition, although the SET-derived change rates (from which all other elevation change values were calculated) do incorporate this parameter. However, as stated previously, inorganic sediment concentrations in northeastern marshes are generally low and will probably continue to decrease with changing land management practices. Even in Louisiana, where sediment inputs are high, Nyman *et al.* (2006) found that marsh accretion varied primarily with

organic accumulation rather than inorganic sediment deposition over a wide range of conditions. Of course, deposition of sand from large overwash events could also increase marsh surface elevations. However, there is no way to predict if, when, and where such depositions may occur in the future.

The way in which marshes respond to SLR may also be influenced by site-specific substrate conditions. Hatches Harbor has relatively low levels of soil organic matter and pore-water hydrogen sulfide concentrations (Smith and Portnoy, 2004). In contrast, these authors reported that Pleasant Bay is highly organic with a thick layer of peat and elevated sulfides. Such variations in edaphic conditions can have a considerable effect on plant growth (Bertness, 1988; DeLaune, Smith, and Patrick, 1983; Deng *et al.*, 2010; King *et al.*, 1982; Padgett and Brown, 1999) and may have variable influences on expected elevation change rates with accelerated SLR. The lowest elevations that were recorded in this study occurred in very sparse *S. alterniflora* growing in sand below the peat platform. However, plants growing on the peat platform may succumb at depths that are shallower than those growing in sand due to the high concentrations of toxic hydrogen sulfide in the former (Howes *et al.*, 1981; Morris, Haley, and Krest, 1996; Smith and Portnoy, 2004). Restricting the lower limits of *S. alterniflora* to the peat platform would affect the model results substantially, resulting in more seaward-edge loss occurring under both scenarios.

While the data reported by Morris, Sundberg, and Hopkinson (2013) for North Inlet (South Carolina) are informative regarding the relationship between *S. alterniflora* production and elevation, more research is needed to determine plant productivity curves and lower limits for other geographic areas, tidal regimes, and edaphic conditions. This study also does not take into account that SLR will likely result in the widening of tidal creeks and the expansion of pools and pannes (Erwin, Sanders, and Prosser, 2004), which would mean additional and permanent losses of vegetation.

Another limitation of this model is that it does not address subsurface hydrology. In this regard, interior marsh areas that may not have surface water but have high water tables within the root zone may experience declines in plant growth as has been documented elsewhere (Gedan, Silliman, and Bertness, 2009; Hartig *et al.*, 2002; Kearney, Grace, and Stevenson, 1988; Reed and Cahoon, 1992). In most Cape Cod salt marshes, which have a sandy substrate and are generally well drained, this probably is not an important factor. However, in peaty marshes like Pleasant Bay, insufficient drainage may be a contributing factor to marsh losses. This study did not include placing piezometers below the marsh surface across the spatial extent of each marsh. However, this would be a worthwhile endeavor to improve the model's prediction capability, especially between peaty and sandy marshes.

Initial elevation (*i.e.* 2013 hypsometric profiles), elevation change, and the composition of high versus low marsh area are all key predictors of how marshes may respond to rising sea level. The core value of this study lies in the acquisition of highly accurate (≤ 4 cm) RTK elevation data collected during a single growing season across a very large area (~ 243 hectares). In contrast, while light detection and ranging (LiDAR) data have been used in other studies (Morris *et al.*, 2005), they can overestimate elevation by as much as 25 cm, mainly due to

interference from marsh vegetation (Chassereau, Bell, and Torres, 2011; Hladik and Alber, 2012; Thorne *et al.*, 2013). Furthermore, water-level data were collected within the marshes themselves rather than using the nearest tide gauges, which may not be representative of the site-specific hydrological conditions.

CONCLUSIONS

Salt marshes on Cape Cod have very low percentages of their surface above MHT, and high marsh habitat appears to be extremely vulnerable to losses under the SLR scenarios modeled here. However, it is clear that site-specific hypsometry may further predispose the marshes to higher or lower vulnerability depending on marsh tide range and opportunities for migration. Thus, the expectation is that some marshes will be more substantially affected than others. The relative vulnerability of individual marshes also depends upon the aspect of change (*e.g.*, high or low marsh loss), metric of change (percentage or hectares), and SLR scenario (50 cm or 100 cm) being considered. Notwithstanding, under both the +50 cm and +100 cm scenarios, major alterations in species composition and total vegetated area are predicted. This may compromise the resilience of these marshes to future storms, accelerated SLR, and other stresses (Gedan, Silliman, and Bertness, 2009), and could ultimately lead to partial or total ecosystem decline, especially if the IPCC projections of SLR used in this study are, as regarded by many, overly conservative (*e.g.*, Rahmstorf, Foster, and Cazenave, 2012). Because these are federally protected marshes, management for migration or thin-layer sediment deposition may be needed to maintain current area and ecosystem services.

ACKNOWLEDGMENTS

This work was supported by the National Park Service, Cape Cod National Seashore, and the Atlantic Research and Learning Center (CCNS). Additional support was provided by the North Atlantic Landscape Conservation Cooperative, and Hurricane Sandy Disaster Mitigation funds.

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