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Carbon Input and Accumulation in Freshwater to Brackish Marshes on the Barrier Islands of Virginia, USA

Nathan Sedghi[1] · Emily C. Adams[2] · Frank P. Day[1]

Abstract Barrier island marshes are exceptionally dynamic systems. The objective of this study was to quantify above and belowground net primary production, above and belowground decay, sedimentation rate, and soil carbon pools within four interdunal fresh to brackish marshes on Hog Island, Virginia, USA and to compare carbon input and soil accumulation among marshes with varied susceptibility to overwash. Major carbon fluxes were assessed by the following methods: aboveground net primary production by harvesting biomass throughout the growing season, belowground net primary production with in-growth cores, sedimentation by deposition on sediment plates, and belowground and aboveground decay with litterbags. We expected sites more susceptible to overwash events to have higher carbon inputs and more accumulation in the soil. Sites in the direct overwash path had slower belowground decay rates, and one of these sites also had the slowest aboveground decay rate. The sites most susceptible to overwash had higher ANPP, but did not have higher soil carbon pools. Sites most susceptible to overwash had a higher salinity and bulk density, but did not store more carbon than other sites. As these sites age and the island accretes vertically, it is expected that soil carbon stocks will increase as well.

Keywords Carbon · Barrier Islands · Freshwater marshes · Brackish marshes · Primary production · Decomposition

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Introduction

Record breaking global surface temperatures are a direct result of increasing concentrations of atmospheric greenhouse gases, particularly CO_2, from anthropogenic activities such as the burning of fossil fuels and land use changes. Consequences of these increases include extreme weather events, extended periods of drought, shifts in species ranges and migration patterns, increased coastal flooding and higher storm surges, and economic distress associated with individuals having to adapt to climate change effects (Hartmann et al. 2013). Obvious mitigation strategies include reducing the amount of CO_2 released into the atmosphere by developing and using alternative and renewable fuel sources, increasing energy efficiency, and reducing consumption of energy (Lal 2008). A less obvious mitigation strategy includes taking advantage of natural carbon sequestration processes (Hallegatte 2009). While biotic carbon sequestration is a low cost and immediately available alternative for removing carbon from the atmosphere, potentially for millennia, a reduction of fossil fuel emissions would provide a larger global carbon offset (Lal 2008; Mackey et al. 2013). Systems that export or bury a substantial portion of their carbon produced remain important in regulating the global carbon cycle (Duarte and Cebrian 1996).

Prior studies on global carbon budgets have focused primarily on terrestrial ecosystems; however, tidally influenced wetlands have higher sequestration rates and bury carbon more efficiently (Chmura et al. 2003). These high sequestration rates are due to high above and belowground primary productivity, anoxic soil conditions slowing decomposition rates, and dense vegetation trapping suspended matter distributed tidally (Kennedy et al. 2010; Fourqurean et al. 2012; Chmura 2013; Duarte et al. 2013). Thus, these ecosystems

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accrete vertically, allowing carbon that is either produced in situ or imported to persist for centuries to millennia if the system remains unaltered (Mateo et al. 1997; Lo Iacono et al. 2008). Soil microbial content and community diversity both correlate positively with groundwater salinity (Morrissey et al. 2014). Ground water salinity plays a large role in determining soil carbon storage as increasing salinity in tidal wetlands causes increased decay rates, though these systems show high variability (Poffenbarger et al. 2011; Morrissey et al. 2014). Depending upon the salinity tolerance of individual species, elevated salinity can decrease primary production (Howard and Mendelssohn 1999). In marshes in the conterminous United States, salinity is inversely correlated with organic carbon content and positively correlated with soil bulk density (Craft 2007).

Wide barrier islands will support freshwater marshes due to a freshwater lens that forms under the island. However, storm-induced overwashes on coastal barrier islands frequently inundate interior marsh systems with seawater and sediment (Leatherman 1988; Fahrig et al. 1993; Hayden et al. 1995). The salinity of these interior marshes naturally fluctuates and tends to increase over time as salt deposited from overwashes remains when the standing water recedes (Hayden et al. 1995). Some studies have shown that tidal freshwater marshes have higher carbon accumulation rates than more saline tidal marshes due to the growth conditions promoted by oceanic inundation boosting organic accumulation and vertical accretion rates (Craft 2007; Drexler et al. 2013). High vertical accretion allows these marshes to keep pace with moderate rates of sea level rise.

The Barrier Islands of Virginia, USA are some of the most dynamic areas in the world and are regularly affected by storm events, which can convert freshwater to brackish or saltwater marshes for at least short periods, if not longer (Hayden et al. 1991). These marshes are not tidal; however, they receive irregular tidal inundation during storm and overwash events (Hayden et al. 1995). Thus, this study sought to quantify net primary production, decomposition, and the carbon pools in aboveground vegetation, belowground vegetation, and the soil for four marshes on Hog Island. The four marsh sites represented marshes that are protected from overwash events to varying degrees due to geomorphological features, such as sand dunes and an elevated trail berm. The four marsh sites were compared to determine if sites more susceptible to overwash events had higher net primary production and larger carbon pools. Hog Island carbon pools were also compared to literature values of similar systems. We hypothesized that the location of the marsh, its proximity to the ocean, and its elevation would increase tidal frequency, nutrient availability, and salinity, and consequently impact carbon storage. We also hypothesized that these marshes would store less carbon than salt marshes due to a lack of daily tidal action.

Methods

Study Area

The Nature Conservancy's Virginia Coast Reserve (VCR) encompasses the largest chain of undisturbed barrier islands along the East Coast of the United States (Ray and Gregg 1991). The VCR is one of the National Science Foundation's Long Term Ecological Research (LTER) sites and is owned by The Nature Conservancy (TNC) (Hayden et al. 1991). The VCR includes 14 islands of various sizes to the east of the Delmarva Peninsula (Fig. 1) (Shao et al. 1996).

The plant communities on Hog Island are among the more well studied of any of the VCR islands (37°40′N, 75°40′W). Hog Island (11.3 km long × 0.8 km wide and 14 km off the Atlantic coast (Hayden et al. 1991)), is eroding on the southern end and accreting at the northern end. The northern end includes 3 dune ridges that run parallel to the shore, with low-lying swales in between. Communities in the low lying swales include shrub thickets dominated by *Morella cerifera* and freshwater marshes dominated by *Spartina patens* and *Schoenoplectus americanus* (Shao et al. 1996).

During the course of this study, from February to November 2013, four interdunal freshwater to brackish marsh sites were monitored (Fig. 2). A trail berm, approximately 1.44 m above sea level, running from east to west on the island, separated the sites into two northern marshes (2 and 4) and two southern marshes (1 and 3). The western marshes (1 and 2) were located close to an overwash fan, where seawater could flow into the sites. The eastern marshes (3 and 4) did not have an overwash fan and had less exposure to a probable overwash. All the sites had a pond area that persisted even as the water tables dropped in the summer. The marshes were dominated by *Spartina patens* and *Schoenoplectus americanus,* but *Phragmites australis* (Cav.) Trin. had begun to invade the edges of each site. The approximate areas of each marsh were marsh 1 (8.1 ha), marsh 2 (4.2 ha), marsh 3 (1.4 ha), and marsh 4 (2.1 ha). Sample grids were established at a size proportional to the size of the marsh to determine the location of all randomly obtained samples. Previous studies from 1994 and 2012 on the interior marshes on Hog Island established salinities to be between 0 and 2 ppt, indicating that these marshes have previously been predominantly freshwater to slightly brackish (Conn 1994; Blecha 2010).

Groundwater depths for each site were measured in hand dug cores. LiDAR data from a 2010 flight were provided by the Anheuser-Busch Coastal Research Center (ABCRC) and were used to create a bare earth digital elevation model for the island. These LiDAR derived elevations for each marsh indicated that marshes 1 and 2 were lower in elevation than marshes 3 and 4 (Fig. 2). While it is notoriously difficult to correct LiDAR data for salt marshes, the LiDAR data provided by the ABCRC are reported to be accurate to within 0.2 m

of elevation, making these data valuable for relative comparisons. Groundwater salinity measurements were taken at various site locations during each sampling event using an Extech RF20 portable refractometer with ATC.

Aboveground Net Primary Production

Aboveground net primary production (ANPP) was measured via four sampling events that encompassed the growing season of the predominant species, *S. patens* (Windham 2001). The aboveground plant parts were harvested in ten 0.25 m^2 plots per marsh site in February and April (representing pre- to early growing season), in August (representing peak-growing season), and in November (representing post-growing season) (Windham 2001). The aboveground plant parts were sorted into live and standing dead material. The live material was further sorted by species. After sorting, all plant material was oven dried at 70 °C for 48 h and weighed to determine biomass. ANPP was calculated by determining the change in

live biomass from April to August (Fahey and Knapp 2007). This is likely an underestimate, as the shedding of plant parts and herbivory are difficult to determine and were not measured in this study (Ovington 1963).

The carbon concentrations in both aboveground and belowground plant tissues were determined using a Europa 20/20 isotope ratio mass spectrometer with an automated nitrogen and carbon analyzer (ANCA). Reference samples were run every 8 samples and blanks were run at the beginning and end of every batch. The standard was sucrose for carbon. Two tissue samples per species per marsh were run in the mass spectrometer and averaged to determine the percent carbon in each species for each marsh.

Belowground Net Primary Production

BNPP was measured with an in-growth core technique similar to Stevenson and Day (1996). This method has been shown to slightly overestimate production relative to more conservative techniques (Neil 1992; Hendricks et al. 2006; McKee et al. 2007). It is considered useful because it involves minimal vegetation destruction (McKee et al. 2007) and allows identification of time periods of growth (Neil 1992). The technique involved the construction of 140 cm^2 nylon bags using 3 mm mesh. The sampling depth (20 cm) was chosen because
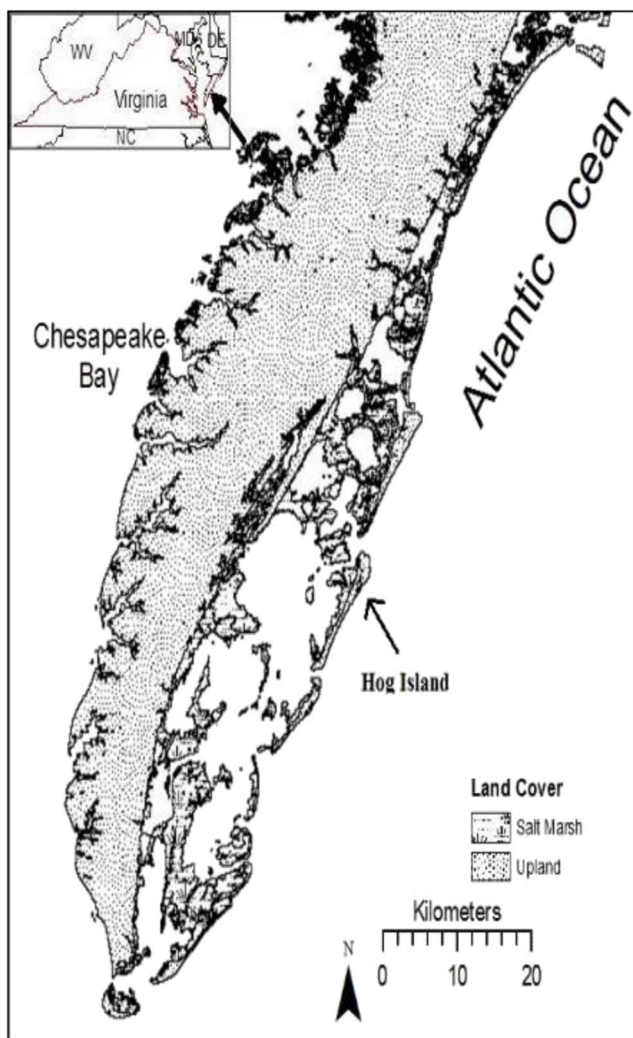
Fig. 1 Map of the Delmarva Peninsula with Hog Island noted
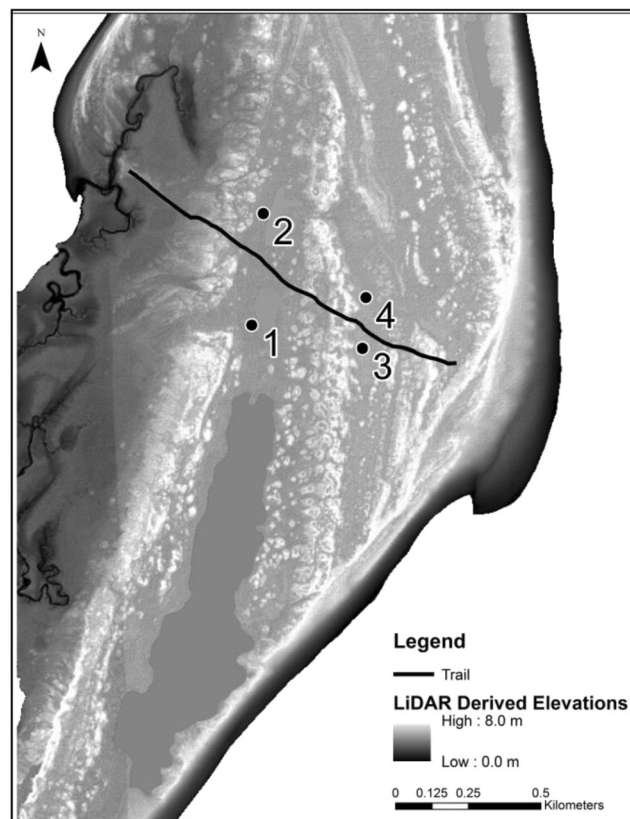
Fig. 2 LiDAR derived elevations (m relative to sea level) for northern Hog Island, Virginia. The trail and locations of the marshes are noted

approximately 85% of the total root biomass is within the top 20 cm of the soil in these marshes (Stevenson and Day 1996). At each marsh site, eight cores were taken using a soil auger, and the bags were inserted into the holes at random locations in each marsh. The bags were then filled with root-free sand from a nearby dune. There was a total of three collection periods: March 15–May 26, May 26–July 31, and July 31–November 17. At the end of each period, the bags were collected, and new bags were installed within 15 cm of the previous core. This sampling method allows determination of growth over growing season intervals to minimize error due to fine root turnover. All growth was clipped, separated in the lab, washed, oven dried until mass was constant, and the weight was recorded. Rhizomatous mass was differentiated from root mass. BNPP was determined by taking the average of all belowground growth for each collection period, and adding means of the 3 collection periods for each site. Results were reported in g dry mass m^{-2} $yr.^{-1}$ to 20 cm depth.

Decomposition of Aboveground Litter

Aboveground decomposition rates were determined by deploying 15 cm by 15 cm litter bags at ground level in February. The litter bags, constructed of nylon netting with a 0.79 mm square mesh, were filled with standing dead stems of *Spartina patens* collected from the marshes in November, 2012. A subsample of the material collected in November was weighed before and after being oven dried in order to determine the air dry/oven dry conversion factor. Each of the four marshes had 5 sites with 6 bags at each site for a total of 120 bags. The locations for the bags were determined using the same grid as the NPP measurements. One bag from each site was collected in April, May, June, August and November and oven dried at 70 °C for 48 h. Decay curves were determined for each marsh site based on percent mass remaining after applying the air dry/ oven dry conversion factor to the stems placed in the litter bags before deployment. Percent carbon lost was determined by multiplying the weight of collected material by the percent carbon determined for *Spartina patens* using the ANCA.

Decomposition of Belowground Litter

Root decomposition was assessed using a vertical litterbag technique, similar to the one described by Tupacz and Day (1989). Vertical insertion of the litterbags represents decay over the soil profile. This technique may underestimate absolute decomposition, but it allows correlation with a variety of measured environmental variables (Wieder and Lang 1982). Litterbags are commonly used to measure decay, allowing easy comparisons with literature values, even if the technique excludes large decomposers and only measures losses of plant materials that are quick to decay.

Five to ten grams of root and rhizome material were placed into each of 100 preweighed, 5 cm by 20 cm litterbags made from 1 mm nylon mesh. A subsample of root material was weighed and dried at 70 °C for 48 h in order to calculate an air dry/oven dry weight ratio. At each of 5 locations per marsh, 5 litterbags were installed. The order in which the litterbags were collected was decided randomly, with 1 litterbag from each location at each marsh site being collected per collection date. All litterbags were installed on March 26, and collections were made on May 29, June 25, July 31, August 21, and November 17. Upon collection, the bags were rinsed, dried at 70 °C for 48 h, and the dry mass remaining was measured. Many fine roots had grown into the 1 mm holes in the mesh, and these were carefully removed with tweezers. Regressions were run on mass remaining versus time; mass loss was found to best fit a negative exponential model.

Sediment Deposition

Sediment deposition was assessed using a simple sediment plate technique. This technique is designed so that as an area floods, the plates remain bolted to the ground. As water recedes, the suspended sediment falls onto the plates and is
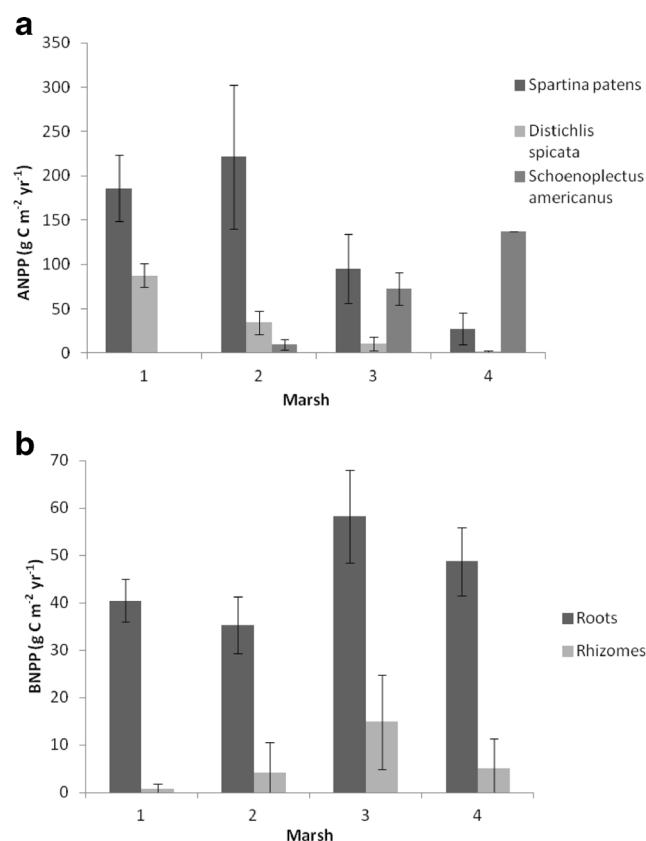
Fig. 3 **a** Aboveground net primary production in g C m^{-2} $yr.^{-1}$ for the four marshes, by species. Error bars indicate one standard error. **b** Belowground net primary production in g C m^{-2} $yr.^{-1}$ for the four marshes, separating rhizomatous growth from root growth. Error bars indicate one standard error
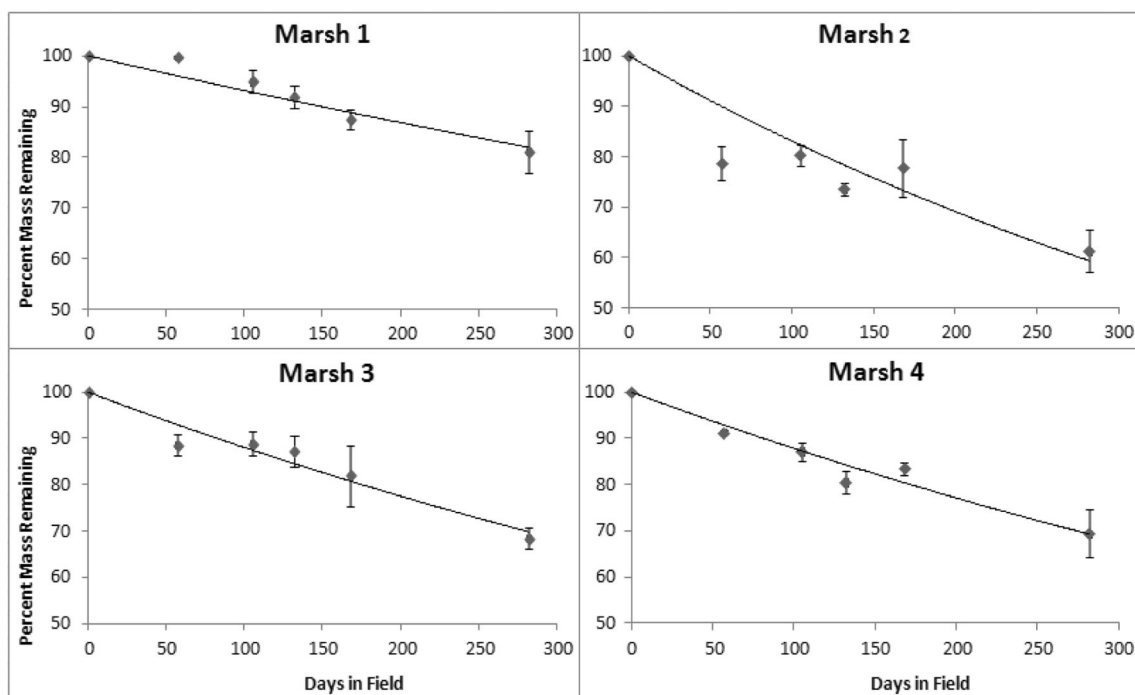
Fig. 4 Aboveground mass loss over the course of the study. Error bars indicate one standard error

collected (Kleiss 1993). The plates (25.5 cm diameter) were made from masonry cutting disks, coated with latex based weather proofing paint, and secured with 5″ hex bolts. At each site, six of the plates were anchored to the ground by hammering hex bolts through a small hole in the middle of them. The sediment plates were placed at the sites on March 3rd and were collected on December 15th. In order to make these values more comparable to literature values, we assumed that what we collected represented an annual value, representing an underestimate of sedimentation as less frequent overwashes occur during the summer months. However, short-term sedimentation measurements may overestimate overall sedimentation rates. As the plates were collected, the material was gently removed using a brush. The material that was collected was oven dried until a constant weight was reached. The mass collected was estimated and a subsample from each plate was run on the ANCA to determine the total carbon deposited on each sediment plate.

Table 1 Aboveground (A) and Belowground (B) decay equations and coefficients of determination (r^2)

| Marsh	Decay Equation	r^2
1 (A)	$y = 100e^{-0.0007x}$	0.927
1 (B)	$y = 100e^{-0.00132x}$	0.551
2 (A)	$y = 100e^{-0.00185x}$	0.781
2 (B)	$y = 100e^{-0.00126x}$	0.775
3 (A)	$y = 100e^{-0.00127x}$	0.951
3 (B)	$y = 100e^{-0.00192x}$	0.951
4 (A)	$y = 100e^{-0.00130x}$	0.947
4 (B)	$y = 100e^{0.00311x}$	0.954

Soil Carbon Pools

During the month of August, three soil cores per marsh site were taken to a depth of 30 cm. The location of each core was determined using the established grid. A 7 cm diameter bucket auger was used, and each core was separated into 10 cm zones (Powell and Day 1991). The cores were refrigerated until processed. Soil pH was not measured but previous studies found a pH of 6.3–6.4 in these swales with no discernible differences by depth or between swales (Conn 1994; Stevenson and Day 1996). Two subsamples of homogenized soil from each depth interval were analyzed using a Europa automated nitrogen and carbon analyzer, which provided percent total carbon. Soil bulk density was determined in each marsh to a depth of 30 cm at each marsh site by extracting three additional 7 cm diameter cores; the cores were dried at 100 degrees C and the mass of each 10 cm increment was divided by its volume. Soil carbon pools were determined by multiplying the bulk density obtained from each marsh by soil carbon concentrations.

Data Analysis

Data were analyzed using an SPSS software package. ANOVAs were run to detect significant differences among marshes, over time, and based on proximity to probable overwash paths. All data were transformed as necessary to meet assumptions of homogeneity of variance, homogeneity of errors, and normality. Post-hoc Bonferroni and Tukey tests were run to determine differences between specific marshes.

Fig. 5 Belowground mass loss over the course of the study. Error bars indicate one standard error
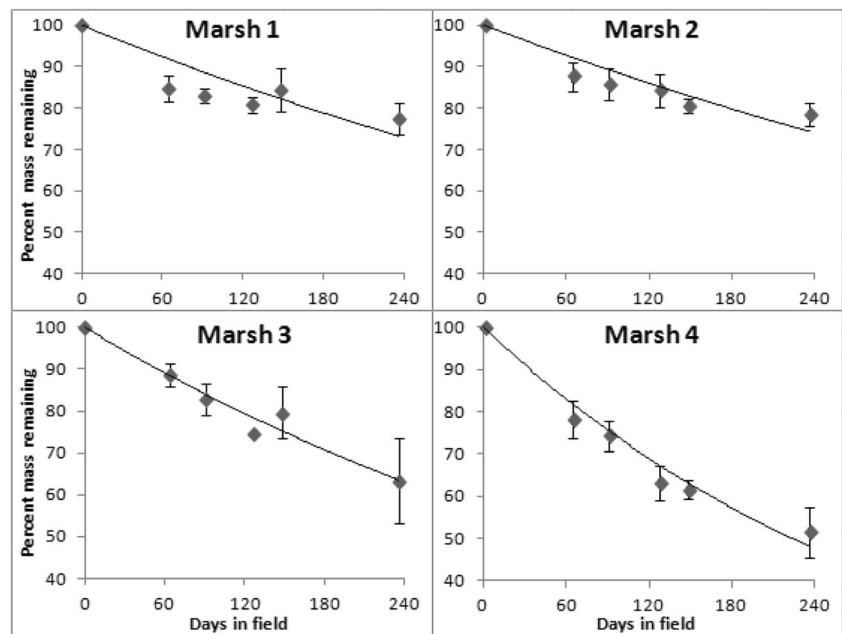
Results

Above and Belowground Net Primary Production

ANPP values were not significantly different among marshes (ANOVA, $p = 0.235$). The trends among the marshes suggest that a site in the direct overwash path but unprotected by the berm (marsh 2) had the highest production while the site least susceptible to overwash (marsh 4) had the lowest (Fig. 3a).

There were significant differences in belowground NPP among marsh sites (ANOVA, $p = 0.002$). Site 3 was the most productive (Tukey, $p_{31} = 0.023$, $p_{32} = 0.003$, $p_{34} = 0.509$), followed by site 4 (Tukey, $p_{41} = 0.326$, $p_{42} = 0.074$) with little separation between sites 1 and 2 (Tukey, $p_{12} = 0.865$). Site 3 was the most variable, with the least variation occurring on site 1. The large variability in samples from site 3 could be a

result of rhizomatous growth, as site 3 had the highest portion of its BNPP from rhizomes at 20.4% (Fig. 3b).

Above and Belowground Decomposition

Aboveground decomposition exhibited a negative exponential decay curve (Fig. 4, Table 1). Significant differences in aboveground decay rate were found among marshes (ANOVA, $p < 0.0005$). Marsh 1 had the slowest decay rate with 81.1% mass remaining after 282 days (Tukey, $p_{12} < 0.005$, $p_{13} = 0.011$, $p_{14} = 0.009$), while marsh 2 had the fastest with 61.3% mass remaining after 282 days (Tukey, $p_{23} = 0.019$, $p_{24} = 0.01$); marshes 3 and 4 had similar rates with 68.5% and 69.5% mass remaining after 282 days (Tukey, $p_{34} = 0.999$).

Differences in belowground decay rates were found among marsh sites (ANOVA, $p = 0.028$), with fastest decay occurring in marsh 4 (Tukey, $p_{41} = 0.042$, $p_{42} = 0.044$,

Fig. 6 Carbon deposited in g C m^{-2} yr.$^{-1}$ during the study period at each marsh site. Error bars indicate one standard error
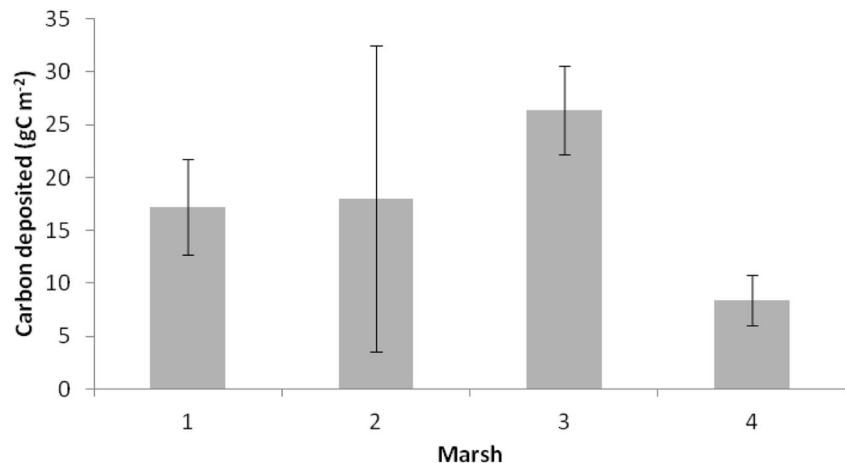
Table 2 Soil bulk density (BD) and carbon density for each marsh for the top 30 cm of soil

Marsh	B.D. (g cm^{-3})	S.E.	Carbon density (g C cm^{-3})	S.E.
1	0.882	0.0635	0.00457	0.00122
2	0.754	0.0405	0.0110	0.00485
3	0.870	0.0546	0.00155	0.000174
4	0.839	0.0853	0.0159	0.00992

$p_{43} = 0.194$). Decay rates were similar between sites 1 and 2 (Tukey, $p_{12} = 1.000$) with litterbags at marsh 1 having 77.5% of initial mass remaining after 236 days and litterbags at marsh 2 having 78.4% of the initial mass remaining after 236 days (Fig. 5, Table 1).
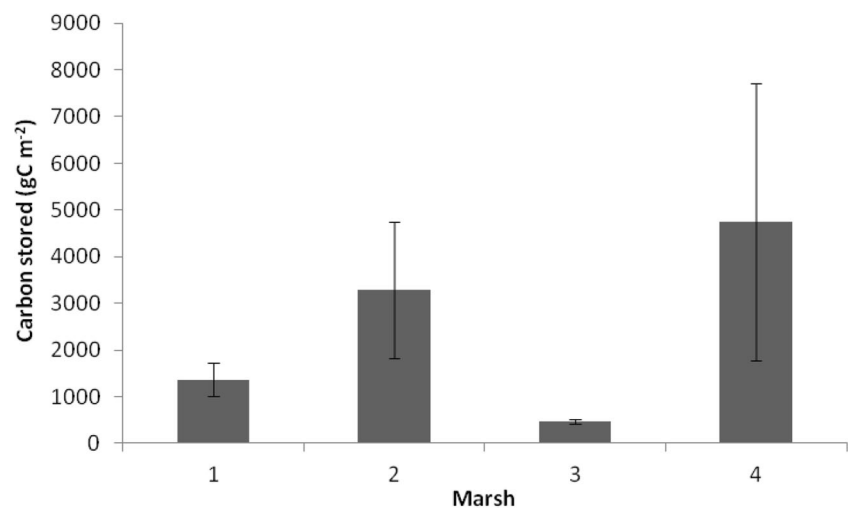
Sediment Deposition

There were significant differences in total carbon deposition among marsh sites (ANOVA, $p = 0.001$). Marsh 3 had a significantly greater deposition than either site 2 or 4 (Bonferroni, $p_{32} = .001$, $p_{34} = 0.005$). While the deposition accumulated in marshes protected by the trail berm (2 and 4) and marshes unprotected by the trail berm (1 and 3) exhibited a statistically significant difference (ANOVA, $p < 0.001$), the sites in the overwash path (1 and 2) had lower deposition rates (ANOVA, $p = 0.053$) than marshes not directly in the overwash path (3 and 4); (Fig. 6).

Soil Carbon Pools

The soil carbon concentrations to a depth of 30 cm were highly variable in each marsh; no significant differences among marshes were found (ANOVA, $p = 0.383$). Soil carbon density by marsh was 0.00457 g C cm^{-3}, 0.0110 g C cm^{-3}, 0.00155 g C cm^{-3}, and 0.0158 g C cm^{-3} for marshes 1–4 respectively

(Table 2). When integrated by depth, there was a trend suggesting that marsh 4 had more carbon stored in the soils, followed by marshes 2, 1, and 3 (Fig. 7).

Site Environmental Differences

There were significant differences in salinity among marshes over time (ANOVA, $p < 0.0005$) (Table 3). Marshes 1 and 2 were significantly more saline than marshes 3 and 4 (Tukey, $p_{12} = 1$, $p_{13} = 0.031$, $p_{14} = 0.028$, $p_{23} = 0.035$, $p_{24} = 0.031$, $p_{34} = 1$). There was a wide range in groundwater salinities over time and generally these marshes were more saline than recorded in earlier studies on the same sites (Conn 1994; Blecha 2010). The values recorded for marshes 1 and 2 in August and November were considerably higher than any other values reported for these marshes (past and present) and could indicate an overwash.

Hand augured wells (Table 4) showed differences among marsh sites (ANOVA, $p = 0.002$) and over time (ANOVA, $p < 0.001$). Marsh site 2 had a higher water table than site 1 (Bonferroni, $p = 0.002$), site 3 (Bonferroni, $p = 0.053$), or site 4 (Bonferroni, $p = 0.009$). The water table was higher in February than it was in May (Bonferroni, $p = 0.003$), August (Bonferroni, $p = 0.005$), or November (Bonferroni, $p < 0.001$). In the summer the water table dropped below the ground surface at all sites, but overall water table height varied by marsh site (ANOVA, $p = 0.003$) with sites 3 and 4 remaining below the ground surface through November.

Discussion

Elevation, water table depth and salinity data support the hypothesis that marshes 1 and 2 are more susceptible to overwash events than marshes 3 and 4. Thus marshes 1 and 2 were hypothesized to have higher net primary production

Fig. 7 Soil carbon pools in g C m^{-2} to a depth of 30 cm in each marsh. Error bars indicate one standard error

Table 3 Mean salinity values (ppt) for each marsh through the course of the study

Marsh	Feb	S.E.	Apr	S.E.	May	S.E.	June	S.E.	Aug	S.E.	Nov	S.E.
1	14	1	20	0	15	0	12	0	25.7	0.67	36.7	0.67
2	13.7	0.67	16.7	0.88	15	0	11.3	0.25	26.3	1.4	30	0.58
3	14	0.58	11.3	0.33	11.7	0.33	10	0.25	11.3	1.4	15	0.58
4	11	0	12	0	10.3	0.33	8	0.58	14.3	0.33	16.3	0.33

rates, lower decomposition rates, higher concentrations of soil carbon, higher sediment deposition and thus higher carbon inputs and soil carbon accumulation. Plant species composition also supports the hypothesis that marshes 1 and 2 are more susceptible to overwash events. Marshes 1 and 2 were dominated by *Spartina patens*, but also had *Distichlis spicata,* a salt tolerant species, indicating that marshes 1 and 2 receive more salt water input from the ocean (Lonard et al. 2013). Marshes 3 and 4 were dominated by *Spartina patens* and *Schoenoplectus americanus,* fresh to brackish water species that would be expected in this environment (Howard and Mendelssohn 1999; Lonard et al. 2010). The sediment plates from marsh 1 and a salinity spike only on marshes 1 and 2 indicate there may have been some incursion of seawater. Marshes 1 and 2 had higher aboveground production and belowground decay rates compared to marshes 3 and 4; however, the trends were not statistically significant in BNPP, aboveground decomposition, sediment deposition, or soil carbon pools.

Soil carbon accumulation and carbon fluxes were compared to literature values reported for salt marshes from similar latitudes. Aboveground net primary production in carbon units ranged from 156 to 284 g C m^{-2}. These values fall within the range of reported aboveground net primary productivity values for salt marshes. Northern Canadian and Alaskan marshes were 60 g C m^{-2} yr.$^{-1}$ while north central Gulf of Mexico marshes, some of the highest salt marsh values for North America, were 812 g C m^{-2} yr.$^{-1}$ (Chmura 2013). According to Drexler et al. (2013), tidal freshwater marshes have been found to store carbon at rates equal to, or in some cases higher than, brackish or saline tidal marshes.

In general, the aboveground decomposition rates for all the marshes were slower than rates reported for salt marsh systems, which indicates that a substantial portion of aboveground biomass may be available for storage at the end of each growing season. Summer months in these marshes are relatively dry, which could boost decomposition rates in the absence of overwash. These marshes are young, approximately 70 years old for marshes 1 and 2 and 30 years old for marshes 3 and 4; as the marshes age, it is likely the soil carbon pool will increase (Harris et al. 2007).

Annual BNPP at each marsh site was less than the average obtained from literature values of salt marshes. The range of the literature BNPP values (676–3500 g dry mass m^{-2} yr.$^{-1}$) for 0–30 cm in Virginia, New Jersey, Massachusetts, and Georgia salt marshes demonstrates both high variability and greater production than marshes on Hog Island (Valiela et al. 1976; Smith et al. 1979; Livingstone and Patriquin 1981; Bellis and Gaither 1985; Blum 1993). There is no uniform approach to measuring BNPP (Schubauer and Hopkinson 1984), as such, the referenced literature values from salt marshes use a variety of techniques.

It was expected that more frequent overwashes will lead to slower decay rates. Belowground decay was lowest in the marshes that are closest to the probable overwash path (marshes 1 and 2), but was still faster than the decay rates found in the literature for Virginia salt marshes for 0–20 cm depth (k = 0.000825 day^{-1}) (Blum and Christian 2004). The decay rates at marsh sites 1 (0.00132 day^{-1}) and 2 (0.00126 day^{-1}) were comparable to but still greater than salt marshes; thus, belowground decay indicates that the sites directly in the overwash path have moderate carbon accumulation capability.

Sediment deposition showed high variance within sites; other studies have shown high variability with sedimentation measurements (Proosdij et al. 2006). Several plates had been completely buried by detritus in the field, thus possibly inflating organic material collected. The dry mass of sediment collected from each marsh site was compared to literature values for nearby Virginia salt marshes. Nearby Phillips Creek and

Table 4 Mean water table heights (cm) in relation to the soil surface for the study period; positive numbers are aboveground, negative numbers indicate the water table is belowground

Marsh	Feb	S.E.	Apr	S.E.	June	S.E.	Aug	S.E.	Nov	S.E.
1	2	1.32	1.3	1.25	No Data	No Data	−21	8.54	0.83	0.44
2	15	2.68	11	2.25	5.4	2.03	−16	2.32	0.5	0.29
3	17	1.04	15	0.58	1.4	0.43	−15	0.88	−14	2.60
4	12	1.53	11	1.45	2.7	1.09	−14	1.20	−28	0.33

Chimney Pole marsh (580 g dry mass m^{-2} yr.$^{-1}$) exhibited much higher sedimentation rates than those observed during the course of this study (Kastler and Wiberg 1996). These nearby Virginia salt marshes receive frequent tidal inundation, naturally leading to high sedimentation rates. During years with large storms and overwash events, sedimentation may be more comparable in the Hog Island marshes. Hog Island marshes did not experience any recorded major storm events during the course of the study, which depressed sediment deposition in these sites relative to what might be expected during a typical year with more frequent overwashes, making comparisons to literature values difficult.

These sites also had a much lower soil carbon density than the average of tidal saline wetlands of 0.039 g C cm^{-3} (Chmura et al. 2003). While the average of tidal saline wetlands is much higher than these sites, marsh sites 2 (0.0109 g C cm^{-3}) and 4 (0.0158 g C cm^{-3}) both fell within the range of recorded soil carbon densities (0.009–0.120 g C cm^{-3}) (Chmura et al. 2003), marsh sites 1 (0.00457 g C cm^{-3}) and 3 (0.00155 g C cm^{-3}) both fell outside this range. The sites with the highest sedimentation rates, sites 3 and 1, also had the highest bulk densities and the lowest soil carbon concentrations, possibly due to high mineral input diluting contributions from plant biomass (Chmura et al. 2003). Increased sea level rise and storm frequency would lead to more frequent overwashes, thus increasing sedimentation and total soil carbon stocks, provided that the island accretes vertically enough to keep pace with sea level rise (Day et al. 2008). If the islands are unable to keep pace with sea level rise, marsh ecosystems will be lost and could potentially be replaced by open water as the island falls below sea level (Fitz Gerald et al. 2007). During this process, the previously inhabited vegetation will die and the stored carbon may be exported from the system through erosion without the protection of the sand dunes.

Previous studies in the Hog Island marshes reported a range of salinities between 0 and 2 ppt (Conn 1994; Blecha 2010); whereas we measured salinities to be highly variable and typically between 8 and 20 ppt, indicating that these marshes may be becoming more saline over time. Measurements from July of 2014 (3–11 ppt) indicate that the 2013 salinity measurements may have been unusually high. A study of the evolution of a similar barrier island marsh in Georgia illustrates a similar pattern (Booth et al. 1999). Barrier island freshwater marshes undergo many substantial changes throughout their history due to storm tides continuously altering hydrology and salinity. The increasing salinities seen on Hog Island are likely due to high spatial and temporal landscape variability; however, it is also likely that the higher values were a result of salt water intrusion from an earlier coastal storm. Total carbon stocks have been shown to be greatest at brackish stages of a marsh compared to saline or freshwater stages, thus making brackish marshes the most efficient at storing carbon (Wieski et al. 2010; Poffenbarger et al. 2011).

It was expected that the sites most likely to experience overwash (sites 1 and 2) would have the largest carbon inputs and soil carbon pools. These marshes had the highest ANPP and the slowest rate of belowground decay, both of which were expected results at the onset of this study. These trends seem to indicate that these sites accumulate more carbon than the sites less susceptible to overwash (3 and 4), but these trends did not show statistical significance. BNPP was highest in marsh 3, possibly due to high rhizomatous growth, and soil carbon accumulation was highest in marsh 4, both of which were unexpected results.

Tidal import and export will vary with the frequency and severity of storm systems, which will change as a result of global climate change (Hartmann et al. 2013). Overwash frequency during this study was not recorded, therefore it is difficult to predict how storms will actually affect the carbon storage capabilities of these marshes. This unpredictability, of both carbon storage due to storms and storm frequency, makes it a high priority to investigate carbon storage capabilities of barrier island marshes over time.

Conclusions

ANPP and aboveground and belowground decomposition rates suggest that the quantity of plant biomass remaining at the end of the growing season is similar between salt marshes and Hog island marshes. However, data from sedimentation, BNPP, and soil carbon pools suggest that the Hog Island marshes do not bury carbon at similar rates to salt marshes. Climate model projections forecast increasing storm frequency and severity, possibly increasing sediment deposition to these marshes, assuming they keep pace with sea level rise (Hartmann et al. 2013). Since these are young marshes that may receive greater future sediment input, the soil carbon stocks will likely increase over time as the island accretes vertically to keep pace with sea level rise. While the potential to accumulate carbon in these systems was determined to be high, the lack of understanding of export limits the ability to make long-term proclamations about the fate of carbon stored. Further work must also consider tidal export, herbivory, and methane emissions from these systems in order to accurately determine a realistic sequestration rate. By understanding the carbon storage dynamics of wetlands, ecosystems that represent a high conservation priority can more readily be identified and protected in an effort to mitigate the accelerated greenhouse effect.

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