Spring 2015

Blue Carbon in Coastal Freshwater/Brackish Marshes on the Barrier Islands of Virginia: Belowground Carbon Dynamics

Nathan M. Sedghi
Old Dominion University

Recommended Citation

Sedghi, Nathan M.. "Blue Carbon in Coastal Freshwater/Brackish Marshes on the Barrier Islands of Virginia: Belowground Carbon Dynamics" (2015). Master of Science (MS), Thesis, Biological Sciences, Old Dominion University, DOI: 10.25777/cqfw-hq52
https://digitalcommons.odu.edu/biology_etds/280

BLUE CARBON IN COASTAL FRESHWATER/BRACKISH MARSHES ON THE BARRIER ISLANDS OF VIRGINIA:

BELOWGROUND CARBON DYNAMICS

By

Nathan M. Sedghi
B.S. December 2011, Old Dominion University

A Thesis Submitted to the Faculty of
Old Dominion University in Partial Fulfillment of the
Requirements for the Degree of

MASTER OF SCIENCE

BIOLOGY

OLD DOMINION UNIVERSITY
May 2015

Approved by:

Frank Day (Director)

Kneeland Nesius (Member)

Rich Whittecar (Member)

ABSTRACT

BLUE CARBON IN COASTAL FRESHWATER/BRACKISH MARSHES ON THE BARRIER ISLANDS OF VIRGINIA: BELOWGROUND CARBON DYNAMICS

Nathan Mohammad Sedghi
Old Dominion University, 2015
Director: Dr. Frank P. Day

Some coastal ecosystems sequester substantially more carbon (blue carbon) than land locked systems due to high net primary production and deposition associated with oceanic influences. Most blue carbon research has focused on mangroves, seagrass beds, and salt marshes. Studies on blue carbon potential of coastal freshwater marshes are less common. Barrier islands frequently flood, with seawater reaching interior ecosystems. I examined brackish/freshwater marshes on Virginia barrier islands for possible roles as blue carbon systems. I studied four interior marshes on Hog Island, which varied in proximity to a direct overwash path and protection by a trail berm that divides northern from southern sites.In order to compare carbon storage rates in these systems to the literature, maximum carbon available for storage at the end of each growing season, or sequestration potential (g C m^{-2} yr^{-1}), was assessed. Sequestration potential includes production, decay, and deposition, but not other processes that affect carbon storage, like nonbiological oxidation and export. Belowground Net Primary Production (BNPP) was lower for sites directly in the overwash path (40.31g C m^{-2} yr^{-1}) than for sites not in the path (63.39g C m^{-2} yr^{-1}). BNPP was greater for sites unprotected by the berm (57.11g C m^{-2} yr^{-1}) than for protected sites (46.59g C m^{-2} yr^{-1}). Decay rates for sites not in the

overwash path ($k = 0.00249$ day^{-1}) were greater than for sites in the path ($k = 0.00129$ day^{-1}). Sediment deposition was greater for sites unprotected by the trail berm than for protected sites. These sites were less productive belowground than salt marshes, and root decay rates were higher than in mid-Atlantic salt marshes, which also had a greater rate of sediment deposition than Hog Island marshes. During this study there were no major overwash events; this lack of overwashes indicates that typical Hog Island sediment deposition rates should be higher than those measured during this study (8.38-26.37 g C m^{-2} yr^{-1}). The calculated rates of carbon sequestration potential (116.6-233.4 g C m^{-2} yr^{-1}) indicated that these sites do not store as much carbon as salt marshes (218 ± 24 g C m^{-2} yr^{-1}), but appear comparable to seagrass beds (138 ± 38 g C m^{-2} yr^{-1}), and therefore have some blue carbon potential.

ACKNOWLEDGMENTS

I would like to thank my committee, Dr. Rich Whittecar and Dr. Kneeland Nesius, for their helpful advice throughout the course of earning my degree. I would also like to thank Dr. Frank Day for tirelessly correcting my mistakes, tolerating my shenanigans, and for advising me on other future plans. My project would not have been possible without help from people such as Emily Adams, Leah Gibala-Smith, Matt Smith, Bryan Gordy, and Jeffrey Rollins for help with sample preparation, field work, and data collection. Dr. Holly Gaff provided valuable help with working through my statistics. The staff at the ABCRC have been very helpful and accommodating while I worked on Hog Island. My parents, Karen Sedghi and Mohammad Sedghi, were the greatest help to me throughout this endeavor; they helped me to prepare materials for field work, supported several decisions to change my field of study, and they kept me going every step of the way. This study was funded by subcontract GA11020-142301 on the University of Virginia's NSF grant DEB-1237733.

TABLE OF CONTENTS

LIST OF TABLES

Table Page

LIST OF FIGURES

INTRODUCTION

Overview

In recent decades, the problems posed by greenhouse gas induced warming have received much attention. Some of these problems, such as increased frequency of storm events, erosion of shorelines, and melting of glaciers, could have destructive consequences for coastal communities (Nellemann et al. 2009). Plants help to combat greenhouse induced warming by capturing CO_2 via photosynthesis. Carbon is stored within living plant tissues as biomass via net primary production (NPP). This carbon may be subsequently stored in soils, decomposed, respired by the plants back to the atmosphere, consumed by herbivores, or exported by flowing water.

Net Ecosystem Production (NEP) refers to the organic carbon produced by an entire ecosystem that is not respired (Lovett et al. 2006). Coastal ecosystems have been shown to be net autotrophic, meaning that these ecosystems store more carbon than is returned to the atmosphere (Gattusso et al. 1998). These systems can also be considered "sinks" in the global carbon cycle. The pattern of biogeochemical cycles can greatly affect whether a system is net autotrophic or net heterotrophic. These cycles occur where organisms exchange nutrients with the environment, catalyze chemical reactions, and the process is continuously reset (Hedges 1992).

A variety of factors control the net biomass produced by an ecosystem; these include water availability, temperature, species composition, nutrient availability, and sunlight availability. A change in any of these factors may lead to higher or lower photosynthetic rates. The combination of these factors culminates in wide production ranges among different ecosystems and even within the same ecosystem (Brinson et al.

1981). Factors within the communities also impact the rate at which carbon can leave a system, shifting the storage balance towards being net heterotrophic. Those ecosystems that export or bury a large portion of their primary production are highly autotrophic, and play an important role in regulating the global carbon cycle (Duarte and Cebrian 1996).

Carbon sequestration refers to the process by which carbon is stored long term. The amount of carbon storeddepends upon how much of the organic material enters the system in the form of refractory (decay resistant) materials (Schlesinger and Lichter 2001). Most of the remains of dead plants are oxidized back to the atmosphere (Hedges 1992) and approximately 50% of the detritus that accumulates on the soil surface is carbon (Houghton 2007), making decay dynamics of a system important in determining sequestration rates. Some ecosystems, particularly wetlands, can be either sources or sinks of carbon, depending upon certain conditions. When the rate of carbon input to the soil is higher than the rate of decay, carbon is buried in the soil. If the rate of decay is higher, then wetlands release greenhouse gases, such as methane and nitrous oxide, to the atmosphere during anaerobic decomposition (Poffenbarge et al. 2011). For this reason, biomass production alone does not adequately suggest potential carbon sequestration rates.

Only a few ecosystems are capable of storing carbon for long timescales, such as millenia (Mateo et al. 1997). Mangroves, salt marshes, and sea grass beds can accumulate carbon faster (Chmura et al. 2003) and store more carbon than terrestrial ecosystems (Mcleod et al. 2011). Due to their oceanic influence, carbon stored in the sediments of these ecosystems is referred to as "blue carbon" (Nellemann et al. 2009). Prior work on blue carbon has not included non-tidal or freshwater marshes (Nellemann et al. 2009; Mcleod et al. 2011; Pendleton et al. 2012). Storm-induced overwashes on the coastal barrier islands frequently inundate interior marsh systems with seawater and sediment (Leatherman 1988; Fahrig et al. 1993; Hayden et al. 1995). These overwashes may supply the nutrients, sediment input, and pathway to organic matter preservation required in order to classify these systems as blue carbon systems. The current study examined the belowground net primary production (BNPP), decomposition, sediment deposition, and carbon storage of freshwater/brackish marshes on a Virginia barrier island, with the goal of determining if these marshes can be considered blue carbon systems.

Blue carbon

Coastal and deep ocean plants have a much smaller cumulative biomass than plants in terrestrial systems, but coastal and oceanic ecosystems cycle roughly the same amount of carbon annually (Houghton 2007; Bouillon et al. 2008; Nellemann et al. 2009). The portion of this carbon which is sequestered, or goes through other pathways, is dependent on both the physical properties of the community and the biological properties of the autotrophs involved (Duarte and Cebrian 1996; Mcleod et al. 2011). Blue carbon sinks are dominated by angiosperms, and thus far, are only identified with salt marshes, seagrasses, and mangroves (Duarte et al. 2005; Nellemann et al. 2009; Mcleod et al. 2011). These systems store more carbon than rainforests, but receive far less attention from the public (Nellemann et al. 2009). The coastal zone covers only about 7% of the ocean's surface area, yet at the interface between land and sea, it is a crucial biogeochemical exchange site. Nutrients from land surface run-off and energy from the open ocean make this area one of the most biologically active ecosystems (Gattusso et al.

1998). These are excellent conditions for plant growth, causing blue carbon systems to be productive and highly autotrophic (Nellemann et al. 2009).

The vegetation of the coastal zone only makes up 4 % of total marine NPP, but contributes ~30 % of the carbon stored in the ocean (Duarte and Cebrian 1996), and is therefore the only part of the oceanic system that represents a long-term blue carbon sink. The carbon stored in blue carbon sediments is roughly 180 times greater than in deep ocean sediment when adjusted for total surface area (Nellemann et al. 2009). The long-term carbon burial rates in blue carbon systems are highly variable (Mcleod et al. 2011), but the maximum burial rates recorded for salt marshes, mangroves, and seagrass beds are 10 times, 6 times, and 2 times the maximum recorded burial rates in Amazon rain forests (Nellemann et al. 2009). The contribution of a system to the carbon cycle is dependent upon what portion of carbon is consumed, as well as how much organic carbon is produced (Gattusso et al. 1998).

Blue carbon mechanisms

The extent to which different variables affect carbon burial rates is not completely understood, but decomposition, growth, and turnover of fine roots all affect soil carbon pools to some degree (Quirk et al. 2011). Blue carbon systems tend to have high productivity, high rates of vertical accretion, and slow decomposition rates (Mcleod et al. 2011). There is a complex interplay between these factors, which contributes to the overall high rates of carbon sequestration in blue carbon systems. The oceans contribute to the carbon sink of blue carbon systems; this occurs directly, by uptaking CO_2 from the atmosphere, and indirectly, by depositing nutrients in blue carbon systems.

Within the oceans, inorganic carbon processes play a much larger role than they do in terrestrial systems (Lovett et al. 2006). The two main processes by which the ocean temporarily stores carbon are called the solubility pump and the biological pump (Mcleod et al. 2011). At the interface between the surface ocean water and the atmosphere, water-soluble CO_2 is regularly dissolved and currents dilute the concentration. CO_2 solubility is higher in colder water, so a greater amount of dissolved inorganic carbon (DIC) exists in the oceans of extreme latitudes (Nellemann et al. 2009) and in deeper ocean waters (Siegenthaler and Sarmiento 1993). As the carbon sinks, Le'Chatelier's principle dictates that the surface water uptake more CO_2 from the atmosphere. The biological pump comes from the fact that phytoplankton based food chains are highly autotrophic, and the uneaten portion of the dead phytoplankton will fall to the bottom of the ocean. Some of this organic matter will be buried in sediment on the ocean bottom (Nellemann et al. 2009). As upwelling occurs, CO_2 is returned to the atmosphere (Gattusso et al. 1998) and only a small fraction of the carbon from the biological pump and solubility pump is preserved long term in ocean sediment (Nellemann et al. 2009). In addition to the effect of upwellings, greenhouse gas induced warming will decrease solubility of CO_2 in ocean water, making the deep ocean sediments short term carbon sinks.

Nutrients typically enter land-based ecosystems through weathering of inorganic stores, nitrogen-fixing bacteria, or import from other sources, such as running water (Killops and Killops 2005). The location of blue carbon systems allows them to intercept nutrients of both terrestrial and oceanic origins (Bouillon et al. 2008). Coastal vegetative systems receive nutrient-rich influxes from the ocean. Any system that buries a substantial amount of organic carbon is likely to be controlled by imported nutrient supply rather than by the rate at which microbes can recycle nutrients (Duarte and

Cebrian 1996). Ammonium adsorbs weakly to the surface of silt and clay particles and is frequently outcompeted by other cations, making it freely available in seawater (Morris et al. 2013). Blue carbon systems receive some type of oceanic flushing, causing ammonium availability to be positively correlated with soil salinity. Seawater contains a variety of other important inorganic ions, including SO_4^{2-}, Mg^{2+}, Ca^{2+}, and K^+ (Killops and Killops 2005). Plants thrive under these high nutrient conditions, leading to increased growth rates (Nellemann et al. 2009), and the subsequent increased input of organic matter can increase the elevation of the ecosystem (McKee et al. 2007).

Ecosystems accrete vertically over time through high organic matter production and storage (McKee et al. 2007), as well as by trapping sediment and preventing resuspension (Kennedy et al. 2010). Blue carbon systems have elevated rates of vertical accretion and sequestration, causing the net amount of carbon stored in their sediments to increase over time (Chmura et al. 2003). This accretion rate can be improved by plants that have vertically growing rhizomes (Romero et al. 1994) and extensive root systems that can store carbon for long periods of time belowground (Romero et al. 1994; Mateo et al. 1997).Salt marshes, mangrove forests, and seagrass beds effectively trap suspended sediment and organic material from external or internal sources (Mcleod et al. 2011). Prior studies have shown that vegetation in coastal communities alters depositional patterns by promoting suspension, as well as by preventing resuspension of sediment (Gacia and Duarte 2001; Kennedy et al. 2010). These blue carbon sinks can provide resistance to water flow, reducing momentum of trapped particles and contributing to vertical accretion. The rate that sedimentation occurs will partly control the rate of decay of organic matter already present in sediment (Killops and Killops 2005).

Many factors affect decay rates, including moisture, temperature, oxygen, pH, availability of electron acceptors, and nutrient content of soil and vegetation (Morris et al. 2013). A change in salinity of a system can lead to increased sulfur availability and, therefore, increased decomposition rates. There is significant temporal and spatial variability for both the sinks and sources of sulfate and methane, thus allowing both processes to occur during anoxic conditions (Poffenbarge et al. 2011). Despite this potential increase in sulfate, overall decay rates should still decrease due to lack of decomposition by obligate anaerobes (Killops and Killops 2005). Some plants, such as mangrove trees, have a low nutrient content, which causes a slow decay rate, even with large nutrient inputs (McKee et al. 2007). It is both the depletion of labile organic material, and the depletion of electron acceptors, that decreases the decay rate during anaerobic conditions (Killops and Killops 2005). Morris et al. (2013) reviewed several studies in which the addition of nutrients reduced belowground organic matter pools by dramatically increasing decay rates. Decreased decay rates cannot be the only mechanism by which blue carbon sinks sequester substantial portions of organic material. High amounts of organic materials must be produced in these systems, and anaerobic conditions must prevail.

The balance between decomposition and production directly affects the quantity of organic matter stored in soil. There are also many feedbacks between different processes in these systems. As aboveground biomass increases, sedimentation will be promoted, and vertical elevation of the system increases (Morris et al. 2013). When this sediment comes with a high concentration of dissolved organic carbon, a priming effect can be triggered, which decreases carbon stored in soils by stimulating microbial decay

(Fontaine et al. 2004). In these cases, the development of anoxic conditions, which can be promoted by high sedimentation rates (Killops and Killops 2005), are essential to preventing the priming effect from decreasing stored carbon.

Organic carbon accumulation is dependent upon both the rate of production and the rate of decay of refractory materials (Morris et al. 2013). Carbon stored can be approximated as net primary production plus the imported carbon, minus the portion of organic material that is decayed, respired, or exported. While there is a variety of different factors affecting carbon storage in blue carbon systems, there is a lack of agreement regarding the relative importance of these factors in the literature (Cebrian and Duarte 1995; Craft 2007).

Carbon fluxes in blue carbon systems

While seagrass meadows, mangrove forests, and tidal salt marshes build up large carbon stores, there is much variation among systems, and even within the same system (Nellemann et al. 2009). Carbon is stored temporarily in living and non-living biomass and may be stored long-term in the underlying sediment (Mcleod et al. 2011).

While these systems produce much of their organic carbon, some have allochthonous origins. Oceanic organic carbon is deposited on these sites, but not all of this carbon is produced in marine systems. Riverine organic matter transported to the ocean is about 0.5 % of all carbon produced by terrestrial systems. Nearly all of the remainder of carbon produced by landlocked systems is remineralized (Hedges 1992; Siegenthaler and Sarmiento 1993) or is stored in the river as sedimentary organic carbon (Aufdenkampe et al. 2011). The oceans also uptake inorganic carbon, forming an enormous dissolved reservoir, which some models have shown accumulate CO_2 at a rate of 2.0 ± 0.6 Gt C yr^{-1} (Siegenthaler and Sarmiento 1993).

The variability in carbon sequestration rates by blue carbon systems is accounted for by differences in hydroperiod, salinity, nutrient availability, and supply of suspended sediments (Mcleod et al. 2011). These ecosystems produce far more organic carbon than is required by the system, with the portion exported and buried representing up to 40 % of total net primary production (Duarte and Cebrian 1996). Romero et al. (1994) speculated that the driving factor in this production excess is allocation of carbon to more refractory structures, such as roots and rhizomes; these structures generally have a high lignin content (Tupacz and Day 1989).

Blue carbon systems are highly productive and store a substantial portion of their carbon in the sediments. Seagrass beds bury 138 ± 38 g C m^{-2} yr^{-1}, mangroves bury 226 ± 39 g C m^{-2} yr^{-1}, and salt marshes bury 218 ± 24g C m^{-2} yr^{-1} (Mcleod et al. 2011). The ultimate fate of these carbon stores depends upon the future conservation or degradation of these systems.

Historical records from ice cores indicate that all previous atmospheric carbon increases of the magnitude of the current increase have led to dramatic climate changes (Siegenthaler and Sarmiento 1993). This makes it important to monitor atmospheric CO_2 concentrations, as well as understand the dynamics of the carbon cycle. Prior global carbon budgets have shown a missing carbon sink, with the carbon sources accounted for (Houghton 2007). It is currently unknown how much of this carbon is stored in blue carbon systems. Recent studies have shown that degraded blue carbon ecosystems can be enormous carbon sources (Pendleton et al. 2012). While there are several proposed methods to preserve blue carbon ecosystems (Nellemann et al. 2009), identification of blue carbon ecosystems must precede conservation, so it can be known which systems are of high ecological importance.

Barrier island marshes

Barrier islands host several ecosystem types, including beaches, dunes, flats, lagoons, and marshes (Leatherman 1988; Hayden et al. 1991). The amount of sand supplied to these islands is critical in determining if barrier islands are able to maintain their position, or if they are unable to keep up with rising sea levels (Leatherman 1988). Barrier islands are ever changing landscapes due partially to frequent disturbances, which control rates of ecological succession (Hayden et al. 1995). Environmental gradients of water availability, salt spray, and water salinity drive patterns of vegetation (Shao et al. 1996). These environmental factors are determined by the elevations of the land, sea, and fresh-water table relative to one another; overwash frequency will affect these variables, as will the biological organisms inhabiting a community (Hayden et al. 1995; Young et al. 2011). The vegetation that grows on barrier islands is adapted to frequent overwashes and will colonize the deposited sediments (Leatherman 1988). These processes are driving factors of elevation changes on barrier islands (Hayden et al. 1995). If overwashes are infrequent, successional changes will occur and the barrier marshes will be replaced by thickets and woodlands (Leatherman 1988).

Geomorphic changes can result in successional alterations and state changes (Hayden et al. 1995). The height of islands depends upon the progress of sand dune formation by wind and overwash processes; new dunes are formed as the shoreline migrates towards the sea (Hayden et al. 1991). These dunes are stabilized by roots of vegetation (Leatherman 1988), which promote vertical accretion of the island and increase habitability (Hayden et al. 1991). Continuous dune ridges, as strong barriers to overwash, are important in promoting successional changes on barrier islands. Storms may breach these dune ridges at points that have been previously eroded (Leatherman 1988).

In addition to land surface elevation and dune morphology, proximity to the ocean, groundwater salinity, and depth to the water table are important ecological factors on barrier islands. The combination of these factors determines frequency of exposure to salt water, as well as available freshwater for plants (Hayden et al. 1995). Plant distribution, and therefore successional development, is highly dependent upon these factors (Hayden et al. 1991). Plant community structure combines with these abiotic factors to control nutrient cycling, productivity, decomposition, and trophic dynamics (Hayden et al. 1995).

A freshwater lens will only develop on barrier islands if the island is of a sufficient width and age to allow freshwater reserves to form. These reserves form underneath sand dunes and increase in height as the dunes grow (Roper et al. 2013). Due to the density difference between salt and fresh water, precipitation will force displacement of salt water, resulting in a freshwater-saltwater interface and a freshwater lens forming on the islands (Collins and Easley 1999). The sand dunes serve as groundwater recharge points as precipitation accumulates beneath them. The extent of freshwater reserves is limited to the area just below the dunes and the surrounding swales. The groundwater tends to increase in salinity with distance from the dunes (Roper et al. 2013). These swale, or slack, systems serve as groundwater discharge points, due to their comparatively low elevation, and many are inundated or have saturated soils (Leatherman 1988).

Freshwater and brackish marshes on barrier islands are controlled by the amount of groundwater discharged from sand dunes; the water flows downwards and laterally before discharging in the wetlands (Leatherman 1988). Freshwater marshes are defined as those that have a salinity of less than 0.5 parts per thousand (Killops and Killops 2005). Brackish salinities tend to occur far from sand dunes on these islands (Roper et al. 2013). Over time, many barrier island wetlands increase in salinity as seawater inundation occurs (Hayden et al. 1995). These marshes are usually not tidally influenced, and as such, receive primary water inputs from precipitation (Leatherman 1988). Due to these processes, it is expected that salinity of barrier island marshes will fluctuate; thus, they are freshwater or brackish at any given time.

Marshes tend to decrease in salinity with increasing distance from the ocean. Marshes of the conterminous United States show a negative correlation between organic carbon and nitrogen content and soil salinity (Craft 2007), though high sediment input will dilute organic matter concentrations in marsh soil (Chmura et al. 2003). This combination causes salt marshes to have much higher bulk densities than freshwater and brackish marshes (Craft 2007). Some studies have shown that tidal freshwater marshes have higher carbon storage rates than more saline tidal marshes (Craft 2007; Drexler et al. 2013), but freshwater marshes emit higher amounts of methane (Drexler et al. 2013). Continental, freshwater marshes rarely receive oceanic inundation (Brinson et al. 1981). Due to the growth conditions promoted by oceanic inundation, organic accumulation and vertical accretion rates are both higher in natural tidal marshes than in impounded marshes (Drexler et al. 2013). Freshwater wetland soils can easily become anoxic during flooding (Brinson et al. 1981), and because of this, increased freshwater input will lead to more organic matter being preserved (Craft 2007).

Barrier islands have frequent "overwash events" that can result in state change and succession on these islands. Overwashes occur when storm surges generate waves of up to 10 m in height (Hayden et al. 1991) that breach sand dunes, depositing sediment, nutrients, and sea water, while burying vegetation (Leatherman 1988; Fahrig et al. 1993; Hayden et al. 1995). The sand deposition creates washover fans that can then be recolonized by vegetation (Leatherman 1988). During these overwashes, the net amount of sediment on an island is conserved, but is moved landward, the extent of which is determined by size of the storm surge (Leatherman1988; Hayden et al. 1991). Overwash frequency for the island's interior depends upon the surface elevation, proximity to the beach, and the proximity of continuous sand dunes (Leatherman 1988; Fahrig et al. 1993). These overwashes often prevent successional stages from advancing to wooded communities (Hayden et al. 1991), and those grasses that dominate barrier flats, such as *Spartina patens*, can completely recover within a few growing seasons of the overwashevent (Leatherman 1988).

Overwashes also dramatically shape the freshwater lens on barrier islands (Roper et al. 2013). Salt water intrusion can diminish the amount of freshwater available in swale environments (Shao et al. 1996). The time for recovery to pre-overwash salinity levels can vary from one location to another, depending upon the size of freshwater reserves and precipitation. Groundwater recharge and precipitation are the only forces that replenish the freshwater lenses on barrier islands (Terry and Falkland 2010). Overwash disturbances are more frequent in the winter than in the summer, so during the summer, groundwater recharge causes an expansion of the freshwater reserves (Roper et al. 2013).

Due to the influx of nutrients from storm surge, overwash, or unusually high tides, it is expected that plants in barrier island swales will have high growth rates. Overwash occurs as a result of storms breaching sand dunes (Leatherman 1988). Storm surge and high lunar tide result in higher tidal range, though storm surge is a result of high winds during storms, and unusually high tides may be driven by the moon's gravity. Barrier island swales are vulnerable to overwash during winter storms, which may result in salt stress (Young et al. 2011). While the salinity intrusion will have a negative effect on plant growth, carbon export from these soils should decrease as well (Krauss et al. 2012). A spike in salinity usually increases concentrations of sulfate, which may increase decay rates if conditions are anoxic (Krauss et al. 2012,) but most of these systems exhibit a reduction in salinity during periods of high groundwater recharge (Terry and Falkland 2010), and this freshwater input should decrease decay rates (Craft 2007). Frequent burial by overwash disturbances, due to inundation and sediment influx, should lead to lower decay rates if anoxic conditions develop. All of these factors should culminate in high rates of carbon storage associated with oceanic input. Therefore, these non-tidal, freshwater/brackish marshes on barrier islands may be blue carbon ecosystems.

Objectives

Do the freshwater/brackish marshes on barrier islands sequester a comparable amount of carbon as known blue carbon ecosystems? Freshwater/brackish marshes on Hog Island, VA (Figure 1) will be compared to salt marshes to evaluate their blue carbon potential. According to a literature review by Craft (2007), tidal freshwater marshes exhibit a lower bulk density and higher organic carbon concentration than salt marshes. This combination should result in a similar amount of stored carbon, but freshwater marshes emit methane at high rates (Poffenbarge et al. 2011). These systems may show similar carbon sequestration rates as salt marshes, and this will be tested by measuring rates of below ground NPP, belowground decomposition, sediment input, and soil carbon concentrations. Utilizing these parameters, carbon sequestration potential will be estimated.

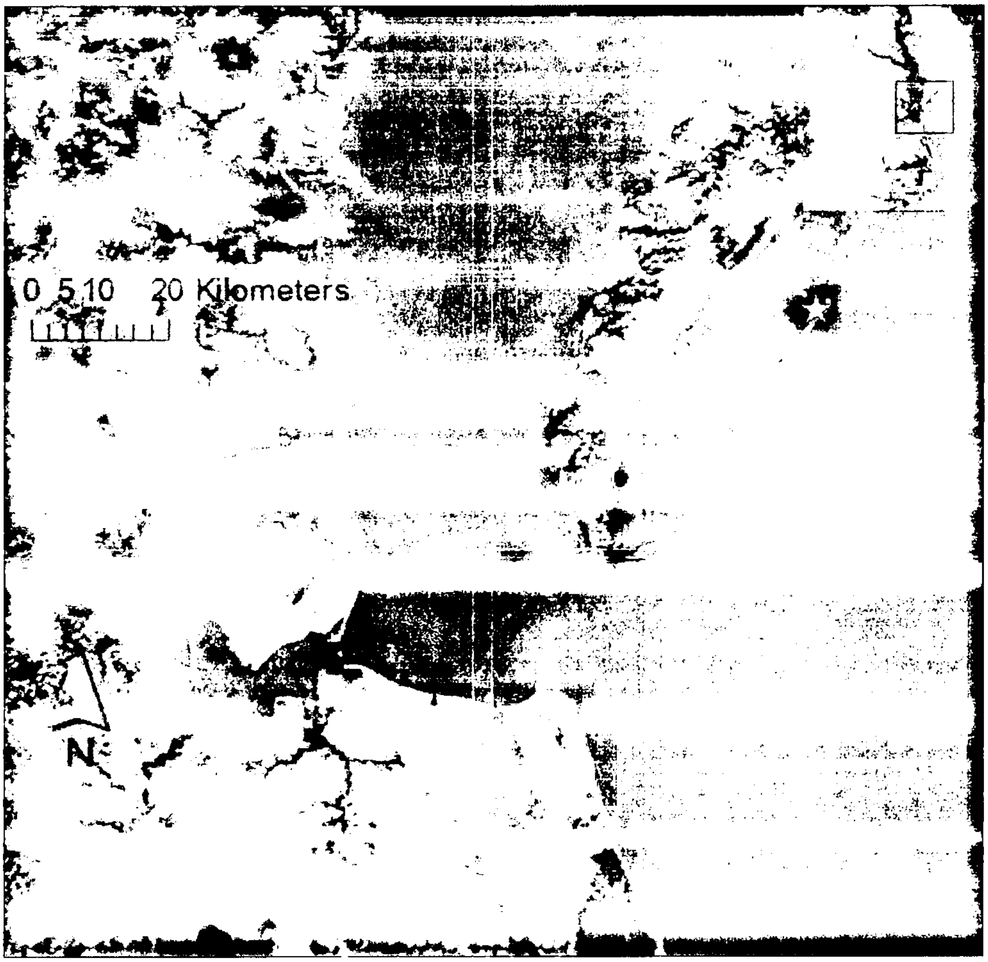

Fig. 1 Hog Island (star) is located approximately 14 km off the Eastern Shore of Virginia.

The study focused on two separate marshes on Hog Island, VA that differ in their vulnerability to storm induced overwashes based upon the elevation of nearby sand dunes and their proximity to the ocean. These marshes are further divided by an elevated trail berm that runs through them (Figure 2). This trail berm should act as a dam and prevent areas beyond itfrom being inundated during minor overwash events. This combination of geographic features causes these sites to vary in their overwash vulnerability. In testing for blue carbon potential, the sites that are most vulnerable to overwash were compared to the sites that were least vulnerable. All of these sites may be potential blue carbon systems, so they were compared to salt marshes as well. The null hypotheses were:

H_{O1}: Sites most vulnerable to overwash have lower BNPP, higher decay rates, and lower sedimentationrates than sites that are least vulnerable to overwash.

H_{O2}: Carbon stocks on study sites are less than carbon stocks in salt marshes.

In testing these hypotheses, it will be determined if these sites are comparable to blue carbon sites, and if the frequency of overwash impacts the belowground carbon dynamics.

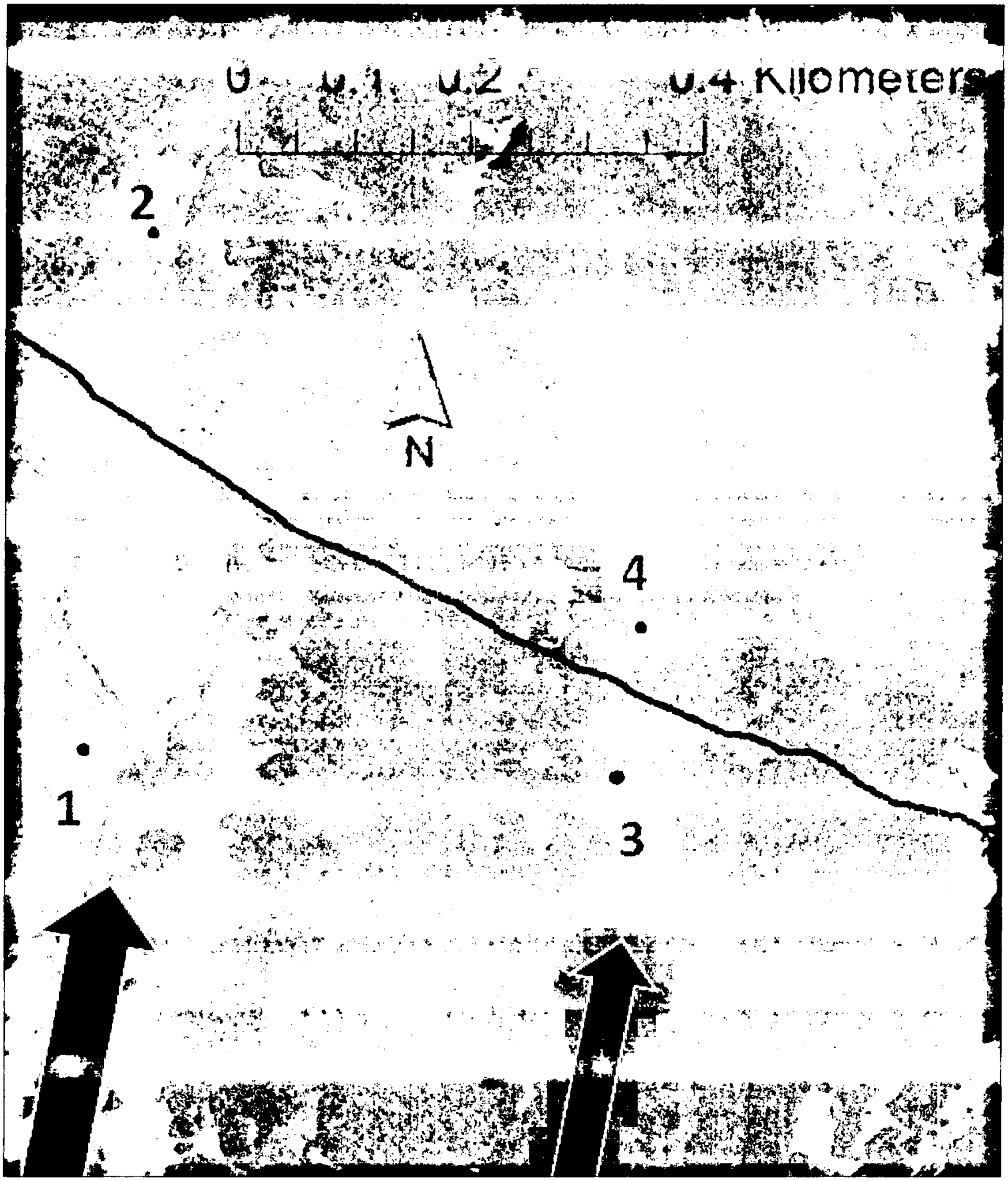

Fig. 2 Northern Hog Island. Probable overwash paths indicated with arrows. The smaller arrow indicates the less likely overwash path. The trail berm separates sites 2 and 4 from sites 1 and 3.

METHODS

Study site

The Virginia Coast Reserve (VCR) site is part of the NSF's Long Term Ecological Research (LTER) program. Fourteen barrier islands are owned and managed by the Nature Conservancy (Hayden et al. 1991) and are inhabited by at least 217 different plant species (Shao et al. 1996). This highly studied system has a high rate of change, mostly due to the frequent disturbance regime (Hayden et al. 1991). During the 2013 growing season, the current study took place on the North end of Hog Island, VA.

The study sites were located at interdune swales that are perched on a freshwater lens. These swales form two interior marshes, which were subdivided by a trail berm, so that there were a total of four study sites. The berm runs east/west and divides the marshes to the north from those in the south and is approximately 5 cm higher than the immediately surrounding marsh area, making it a barrier to only minor overwashes. These marshes also vary in their proximity to a direct overwash path due to differences in the elevation of dunes to the South of these sites. The sites farther to the West (sites 1 and 2) are in a direct path for overwash via a channel, whereas the sites farther to the East (sites 3 and 4) are in an indirect overwash path, though strong overwashes can breach a dune to the East of these sites.Due to the position of these sites relative to the trail berm, and the probable overwash path, it was expected that marsh site 1 would be most vulnerable to overwash, while marsh site 4 would be least vulnerable.

Spartina patens dominates marsh sites 1 and 2, and *Schoenoplectus americanus* dominates sites 3 and 4. *Phragmites australis*is present at the periphery of all of these sites, though it is far more prevalent at marsh 4 than the other sites. At marsh sites 3 and 4, there exists a wide shrub thicket dominated by *Myrica cerifera.* All four of these sites have a freshwater pond at one edge of the site.

The sand dunes on the opposite side of each swale, relative to the current shoreline, have previously been dated by historical photographs and survey charts (Hayden et al. 1991). Prior studies have used the ages of these adjacent sand dunes to estimate the age of the swale sites (Conn and Day 1997; Blecha 2010). Marsh sites 1 and 2 are estimated to have formed in 1871, while marsh sites 3 and 4 are estimated to have formed in 1967. The position of the shoreline has changed several times since the formation of these swales, and the disturbance regime on Hog Island has driven these changes. Some places on the island currently experience frequent overwashes, while others rarely experience overwash (Fahrig et al. 1993), leading to dramatic spatial and temporal variation of soil salinity (Shao et al. 1996). Conn and Day (1997) reported salinity ranges between 0 and 22 ppt at these sites, while Blecha (2010) reported average salinity values of 0 for the 1871 marshes, and 0.73 ppt for the 1967 marshes. Other studies on barrier islands have reported a large change in salinity over time (Van Sickle et al. 1976; Salinas et al. 1986). Due to the constantly changing salinity regime, combined with high freshwater inputs, these marshes are best described as freshwater/brackish.

BNPP

A grid was marked off at each site in an area that appeared to be representative of the vegetation over most of the site. Grid size varied with the area of the marsh, site 1 was 50 m X 50 m, site 2 was 40 m X 40 m, site 3 was 30 m X 50 m, and site 4 was 10 m X 20 m. The grids were treated as a 2-dimensional coordinate system. The location of each sample was determined using a random number generator to determine both X and Y coordinates.

Belowground net primary production (BNPP) was measured with an in-growth core technique, similar to Stevenson and Day (1996). All techniques designed for measuring belowground productivity in wet soils have some drawback, partially due to high labor and time costs (Schubauer and Hopkinson 1984). This method has been shown to slightly overestimate production relative to more conservative techniques (Neil et al. 1992; Hendricks et al. 2006; McKee et al. 2007). It is considered useful because it involves minimal vegetative destruction (McKee et al. 2007) and allows identification of time periods of growth (Neil et al. 1992).

The technique involved the construction of 7 cm X 20 cm nylon bags using 3 mm mesh. The sampling depth (20 cm) was chosen because approximately 85% of the total root biomass is within the top 20 cm of the soil in these marshes (Stephenson and Day 1996). At each marsh site, 8 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. These bags were positioned so that the top of the bags were open and flush with the ground. There were 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.

The roots which were growing into each core represented new growth for the given collection period. All growth was clipped, separated in the lab, washed, oven dried until mass was constant, and the weight was recorded. An attempt was made to separate live from dead roots, but these efforts were abandoned due to a high degree of subjectivity involved in judging live roots from those recently senesced. 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 by dividing the mass obtained by the surface area of the cores (38.5 cm^2). The units are reported on an annual basis, but the data were obtained only during the course of a growing season. It is therefore assumed that the growth occurring between November 18-March 14 was negligible.

Belowground decay

Root decomposition was assessed using a vertical litterbag technique, similar to the one described by Tupacz and Day (1989). The use of litterbags is the most common method of measuring decay dynamics (Wieder and Lang 1982), and the vertical insertion of these litterbags adequately simulates decay over a soil profile (Tupacz and Day 1989). This technique may underestimate absolute decomposition, but it also allows compilation with a variety of independent variables (Wieder and Lang 1982).

The litterbags produced for each site contained a heterogeneous and representative sample of the roots present at each site; a heterogeneous mixture in litterbags is a better approximation of community decomposition than a sample composed only of the dominant vegetation (Day 1982). The roots used were all gently rinsed and air dried in the lab. Five to ten grams of root and rhizome material was placed into each of 100 preweighed, 5 cm X 20 cm litterbags made from 1 mm nylon mesh. A subsample of root material was weighed and dried at 70 °C for 48 hours in order to calculate an air dry/oven dry weight ratio. The bags were marked with the location within the marsh where they were installed, the marsh site number, and the relative position of each bag.

The litterbags were installed vertically into the ground at each randomly determined location. A dibble stick was used to create a slit in the ground for bag installation, and the dibble stick was then placed into the ground 15 cm away and used to push the soil around the bags so each bag was flush with the ground and completely surrounded by the soil. At each of 5 locations per marsh, 5 litterbags were installed. A flag was placed in the ground next to the center litterbag. All other litterbags were placed 15 cm away from the center litterbag. 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 26th, 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 hours, 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.

Upon collecting all samples, regression analyses were done and the line of best fit relating the percentage of mass remaining to timewas determined. A negative exponential decay model was utilized because the trendline fit the data better than a negative linear model, and a single negative exponential function is frequently used for short term decay studies (Wieder and Lang 1982). The rate of decompositionwas calculated using the following equation:

$$\frac{X}{X_i} = e^{-(kt)}$$

Where X indicates the mass remaining at any point in time, X_i indicates the initial mass, t indicates the time elapsed, and k represents the decay rate constant (Olson 1963).

Sediment deposition

Sediment deposition was assessed using a simple sediment plate technique. Sediment plates require little labor and can be used effectively in high depositional environments. 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 collected (Kleiss 1993). The plates are also efficient at capturing wind-blown materials.

The plates were made from masonry cutting disks, coated with latex based weather proofing paint, and secured with 5” hex bolts. While the plates were flat, the faces were rough, which helped prevent sediment from being blown away by the wind (Kleiss 1993). The plates were coated with latex paint in order to prevent them from weathering in the field. 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. As these 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. It is important to note that one site had a much higher portion of fine grained sediment than the other sites; when brushed, this sediment produced dust clouds. In order to collect this sediment without losing a substantial portion, each of these plates had to be washed into a large container, and the sediment was collected from the washings via vacuum filtration. This sediment was dried until a constant weight was obtained. The mass was estimated, and the percent of organic material deposited was measured by using a muffle furnace to measure the loss on ignition (% LOI) at 550° C.

The center hole of the plates measured 3 cm in diameter, while the total diameter was 25.5 cm. The total depositional area, was determined as:

$$A = (\pi r_O^2) - (\pi r_I^2)$$

$$503\ cm^2 = (\pi 12.75^2) - (\pi 1.5^2)$$

Where A represents the depositional area, r_O is the radius of the whole disk, and r_I is the radius of the hole in the middle of the disk. Units were extrapolated to g m^{-2} study $period^{-1}$.

Environmental measurements

On the barrier islands, species composition, community structure, plant biodiversity, exposure to storm surges, and sand burial are controlled by frequency of marine water inundation, depth to the freshwater table, groundwater salinity, and land elevation (Hayden et al. 1995). It is therefore important to measure these variables when performing studies on the barrier islands. Depth to the water table, groundwater salinity, and surface elevation were all measured in order to determine the contribution of these variables to BNPP, root decomposition, and sediment deposition.

During each collection, at each marsh site, a hole was augered and used to measure the distance to the water table. At 3 of the marsh sites, Campbell Scientific pressure transducer wells exist. These wells measure groundwater levels every 15 minutes and produce an hourly measurement. By obtaining these well data from the Anheuser Busch Coastal Research Center (ABCRC), precise water levels were determined and used to validate the measurements from the augered holes. These data were used to produce hydrographs over time for each study site.

Groundwater salinity was measured at several points at each site using a refractometer. Monitoring these data over time was critical because interior marshes of barrier islands can frequently change salinity. In July of 2014, the salinity of the ponds was measured as well. These measurements took place several days after the arrival of tropical storm Arthur (07/04/14). Precipitation data on Hog Island were obtained from the ABCRC in order to determine abnormally dry or wet periods coinciding with field measurements obtained.

Elevation was accounted for by obtaining LIDAR data from the ABCRC. These data were used to measure both the elevation of bare surfaces and of the tops of vegetation. These data were used with GIS to determine the most vulnerable overwash areas.

Carbon sequestration potential

While measuring carbon storage rates (sequestration) would be ideal, several carbon losses were not measured during this study. Variables not measured include methane flux, erosion, non-biological oxidation of organic materials, herbivory, and respiration. A maximum "sequestration potential" was assessed for each site using the measurements available and used as a basis for comparison with literature values. Sequestration potential will be higher than the actual carbon sequestration rates. Root standing stock, aboveground production, and aboveground decay, as well as the carbon and nitrogen content of deposited sediment, vegetation, and soil were obtained from Adams and Day (2015, unpublished).

Soil bulk density was measured at 3 separate locations in each marsh site that varied in their proximity to the freshwater pond. Intact soil cores were extracted with a soil corer and plastic 40 cm length X 5 cm width liners. The corer was driven into the ground with a sledge hammer until the plastic liners were full. Liners were left in a freezer until solid enough to remain intact while being cut. The liners were then cut with a hacksaw at 10 cm intervals. Thecore sections were placed in the oven at 70 °C until a constant weight was obtained. The cores were then weighed, with the mass of the plastic liners subtracted later on. Bulk density was determined by:

$$V = \pi r^2 h$$

$$B.D. = \frac{m}{V}$$

Soil cores were ashed at 550° C and weighed to determine percent organic matter. Soil carbon content was determined with a Europa 20-20 mass spectrometer, featuring a C:N analyzer. A C:N ratio was determined for all samples.

Soil carbon concentrations were determined for depths of 0-30 cm, 31-60 cm, and 61-100 cm. Bulk density was measured to a depth of 40 cm. The bulk densities of the top 30 cm were averaged for the 0-30 cm depth class. Cores from 41-100 cm were examined to determine if the soil bulk density was uniform at these depths. In order to make all measured parameters comparable, soil carbon density was converted from units of g C cm^{-3} to units of g C m^{-2}. Carbon stored in the soil was calculated as bulk density X carbon concentration.

$$Soil\ carbon\ density\left(\frac{gC}{cm^3}\right) = B.D. * C_{conc.}$$

This value was used to estimate carbon stored per unit area, across a vertical profile in the marshes.Since direct measurements of carbon sequestration were not obtained, the relevant variables were combined to estimate the amount of carbon potentially available for storage annually.

The amount of carbon stored can be estimated by an adaptation of an equation from Duarte and Cebrian (1996):

$$S = (ANPP - AD) + (BNPP - BD) + I$$

This equation assumes that import exceeds export, it assumes that herbivory is negligible, and it assumes that vegetative growth of the system is at steady state such that NPP is equal to mortality and loss of plant biomass. The carbon represented by S is not all buried, so the result of this equation is referred to as the carbon sequestration potential, indicating that these values are the maxima possible in these systems. This equation includes aboveground (ANPP) and belowground (BNPP) growth during the study period, plus sediment deposition (I), minus mass loss due to decay over one year (AD and BD). While there will still be decay after this first year ends, an earlier decay study in these marshes measured a root loss of < 5% of the initial mass between the first and second year (Conn and Day 1997), indicating that ignoring additional decay should not have a dramatic effect on estimating sequestration potential.

Statistical analyses

The data were analyzed and compared using an SPSS statistical program. Root decay was compared among the sites with a mixed ANOVA; collection period was used as a within subjects factor and marsh site was a between subjects factor. Bonferroni pairwise comparisons were used, but a Tukey HSD post-hoc test was run for comparisons that were almost statistically significant ($p \geq 0.05$). In order to compare the effect of the trail berm and position relative to the overwash path, as well as differentiate between individual marshes, two separate mixed ANOVAs were run. In both ANOVAs, the percent of initial mass remaining at time of collection was the dependent variable. Some litterbags had greater than 100 % mass remaining at the time of collection; these litterbags were not used in the analyses. The collection performed on July 31st had too many missing samples and was not used in the statistical analyses. With these missing data points, a rank transformation had to be performed to meet the assumptions of normality, homogeneity of variance, and sphericity.

Sediment deposition was compared between the sites with a two-factor ANOVA, comparing protection by the trail berm and overwash path. Another one-factor ANOVA was run in order to compare individual marshes with a Bonferroni pairwise comparison. Before running the analyses, a box plot test was used to detect statistical outliers. In order to meet assumptions of normality and homogeneity of variance, one datum from marsh 2 had to be discarded.

BNPP was analyzed by two separate ANOVAs using the carbon produced by belowground growth as the dependent variable. A two-way ANOVA used collection period and marsh site as independent factors with a Bonferroni pairwise comparison to compare individual marshes. Another two-way ANOVA included positions relative to the trail berm and direct overwash path as independent factors. Before analyses could be run, the dependent variables had to be rank transformed in order to meet assumptions of normality and homogeneity of variance.

Salinity differences were assessed between marsh sites by using a one-way ANOVA with Bonferroni pairwise comparisons. A two-way ANOVA was also run to compare the effect of the trail berm and overwash path. In order to meet assumptions of normality and homogeneity of variance, an inverse transformation was performed on salinity measurements. The difference between salinity in 2013 and 2014 growing seasons was assessed using an independent samples t-test. There were more data points for 2013, so the data had to be rank transformed to meet the assumption of homogeneity of variance.

Elevation was assessed by extracting the bare surface data for 10 points per marsh in ArcGIS. Using a one-way ANOVA with Bonferroni pairwise comparison, elevation was compared between the four marsh sites. Rank transformations were performed in order to meet assumptions of normality and homogeneity of variance.

Groundwater data were analyzed with a mixed ANOVA with marsh site used as a between subjects factor and date as a within subjects factor. Only data from five of the collection dates (02/22, 04/13, 05/29, 08/21, and 11/22) were used in this analysis, as the other collection dates did not have enough hand augered depth to water (DTW) replicates to be used. Rank transformations were performed in order to meet the assumption of homogeneity of variance.

Soil bulk density was compared by using a two-way ANOVA with marsh site and depth as factors. Depth categories were only used from 0-30 cm in 10 cm intervals for this analysis. Bonferroni pairwise comparisons were used to test for individual differences between depths and marsh sites. A three-way ANOVA was used to test for differences based upon position relative to the trail berm and overwash path, depth, and interactions between these factors.

Percentage of carbon in the top 30 cm of soil was analyzed with one-way ANOVA to compare individual marsh sites. Another two-way ANOVA was run to compare differences based upon proximity to the overwash path and position relative to the trail berm. An inverse transformation was performed in order to meet the assumption of normality.

RESULTS

BNPP

There were significant differences in belowground NPP among marsh sites (ANOVA, $p = 0.002$) and collection periods (ANOVA, $p = 0.001$). Interaction of marsh site and collection period was statistically significant (ANOVA, $p = 0.002$); interpretation of simple main effects in SPSS revealed that significant differences among sites varied by collection period.

In general, BNPP increased throughout the growing season; however, the third collection period lasted for 110 days, compared to 68 and 71 days for collection periods 1 and 2, respectively (Figure 3). Marsh sites 3 and 4 represent exceptions to this trend, as marsh site 3 BNPP peaked in the mid-summer collection period while BNPP at site 4 was nearly constant at 17-18 g C m^{-2} yr^{-1} for each collection period. During the March 15-May 26 collection, BNPP was greater at sites 3 and 4 than at sites 1 and 2 (Bonferroni, $p_{13} = .037$, $p_{14} < 0.001$, $p_{23} = 0.132$, $p_{24} < .001$). BNPP was also greater at sites 3 and 4 during the May 26-July 31 period than at sites 1 and 2 (Bonferroni, $p_{13} = 0.005$, $p_{14} = 0.011$, $p_{23} = .003$, $p_{24} = .006$). There were no statistically significant differences among any sites during the July 31-November 17 collection period, and there were no significant differences between sites 1 and 2 or sites 3 and 4 for any collection period (ANOVA, $p \leq .05$).

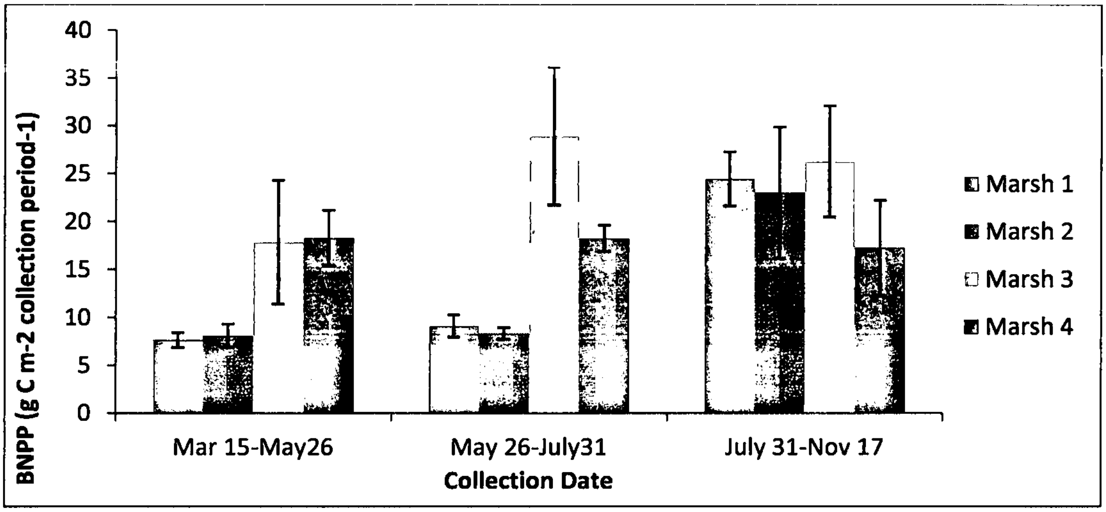

Fig. 3 Belowground NPP (g C m^{-2} collection period^{-1}) from each collection interval. Results are for the top 20 cm of soil. Error bars indicate 1 S.E.

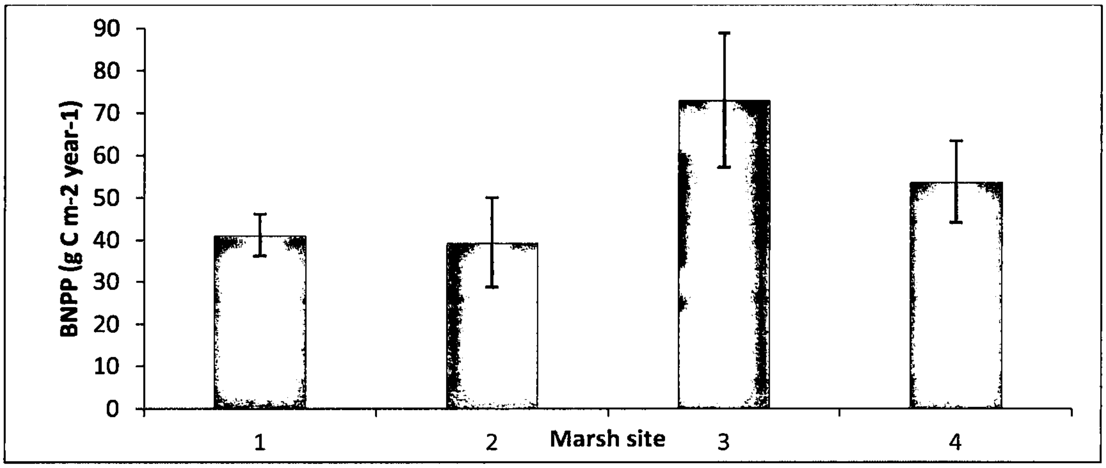

Fig. 4 Annual belowground NPP (g C m^{-2} yr^{-1}) at each site. Results are for the top 20 cm of soil. Error bars indicate 1 S.E.

Annual BNPP (Figure 4) showed that 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 also had the most variability within the data, with the least variation occurring in samples from site 1. The enormous variability in samples from site 3 could be a result of rhizomatous growth, as site 3 had 20.4 % of its total belowground production from rhizomes, compared to just 1.76 % for site 1, 10.5 % for site 2 and 9.38 % for 4. The difference between sites blocked by the trail berm (2 and 4) and unblocked (1 and 4) was not statistically significant (ANOVA, $p \leq 0.05$), sites not directly in the overwash path (3 and 4) were more productive (ANOVA, $p = 0.002$) than sites directly in overwash path (1 and 2), and the interaction of overwash path and trail berm was not significant (ANOVA, $p \leq 0.05$). These results indicate the null hypothesis must be accepted; sites more vulnerable to overwash were not more productive (Table 1).

Table 1 Results of statistical analyses for rank transformed BNPP. *Results of two way ANOVA that compared marsh site to collection period. **Results of two way ANOVA thatcompared proximity to the most probable overwash path and potential protection by a trail berm.

Parameter	Sig.	F	df
Marsh*	.002	5.506	3
Overwash Path**	.002	10.478	1
Trail Berm**	.201	1.662	1
Collection Period*	<.001	7.804	2
Path*Berm**	.733	.117	1
Marsh*Collection*	0.002	3.907	6

Decomposition

Differences in decay rate were found among marsh sites (ANOVA, $p = 0.028$), with fastest decay occurring at marsh site 4 (Tukey, $p_{41} = 0.042$, $p_{42} = 0.044$, $p_{43} = 0.194$). The 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 (Figure 5). Litterbags at marsh 3 had 63.4 % initial mass remaining after 236 days, compared to 51.5 % of the initial mass remaining at marsh 4 after 236 days. While mass remaining over time decreased (ANOVA, $p < 0.001$), significant differences among collection periods were only found between the first and last collection (Bonferroni, $p = 0.007$). Sites directly in the overwash path had lower decay rates than sites not directly in the overwash path (ANOVA, $p = 0.018$). There were no interactions between collection period and marsh site, or between position relative to the overwash path and trail berm (ANOVA, $p \leq 0.05$). Proximity to the direct overwash path resulted in statistically significant differences in decay rates, but the position of marshes relative to the trail berm did not (ANOVA, $p = 0.16$), indicating that the null hypothesis must be accepted; the sites more vulnerable to overwash did not have lower decay rates (Table 2).

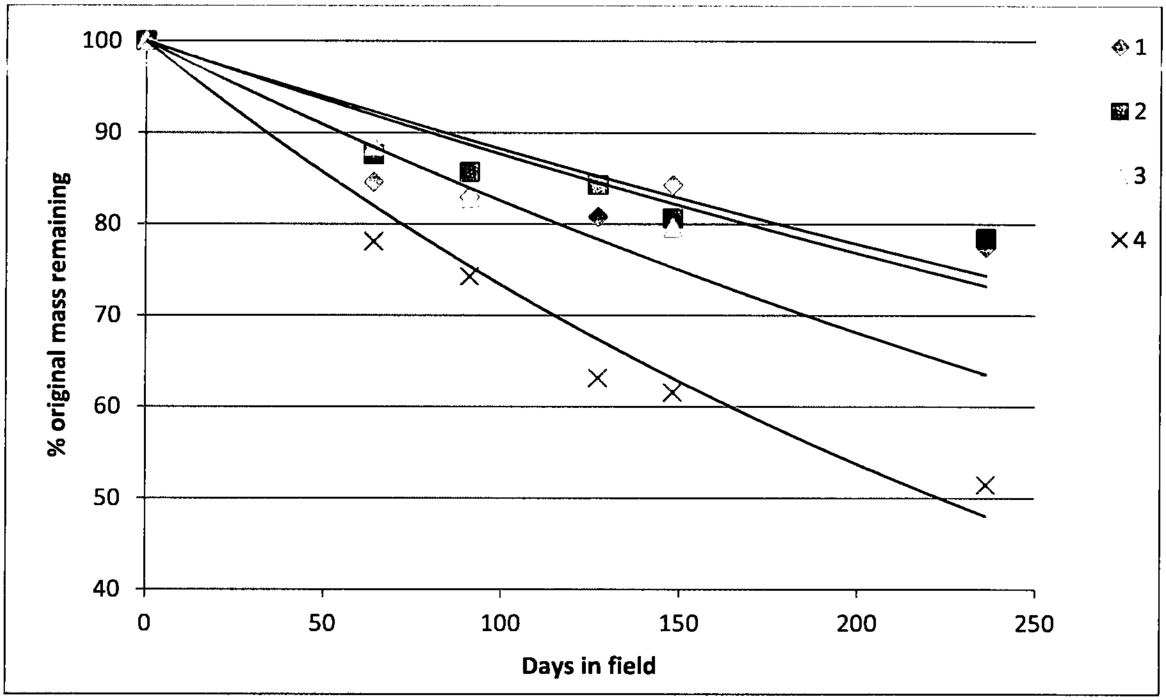

Fig. 5 Negative exponential decomposition of roots in litterbags over time.

Table 2 Statistical analyses of rank transformed decay rates. *Results of mixed ANOVA comparing marsh site and collection period. **Results of mixed ANOVA comparing effect of overwash path and trail berm over time.

Parameter	Sig.	F	df
Marsh site*	.028	4.889	3
Collection period*	<.001	7.536	3
Trail berm**	.16	8.423	1
Overwash path**	.018	2.188	1
Collection*Marsh*	.792	.592	9
Berm*Path**	.173	2.188	1

Deposition

Marsh site 3 had approximately three times as much organic matter (19.79 g m^{-2}) deposited as inorganic matter (6.577 g m^{-2}); all other marsh sites had nearly the same amount of organic and inorganic deposition collected (Figure 6). There were significant differences in total carbon deposition (Figure 7) among marsh sites (ANOVA, $p = 0.001$). Marsh site 3 had a significantly greater deposition than either site 2 or 4 (Bonferroni, $p_{32} = .001$, $p_{34} = 0.005$). While the difference between sites protected (sites 2 and 4) and unprotected (sites 1 and 3) by the trail berm was statistically significant (ANOVA, $p < 0.001$), the sites in direct overwash path (sites 1 and 2) had lower deposition rates (ANOVA, $p = 0.053$) than sites not directly in the overwash path (sites 3 and 4). There was no significant interaction between the overwash path and trail berm (ANOVA, $p \leq 0.05$). These results required accepting the null hypothesis; carbon deposition was not greater in sites more vulnerable to overwash (Table 3).

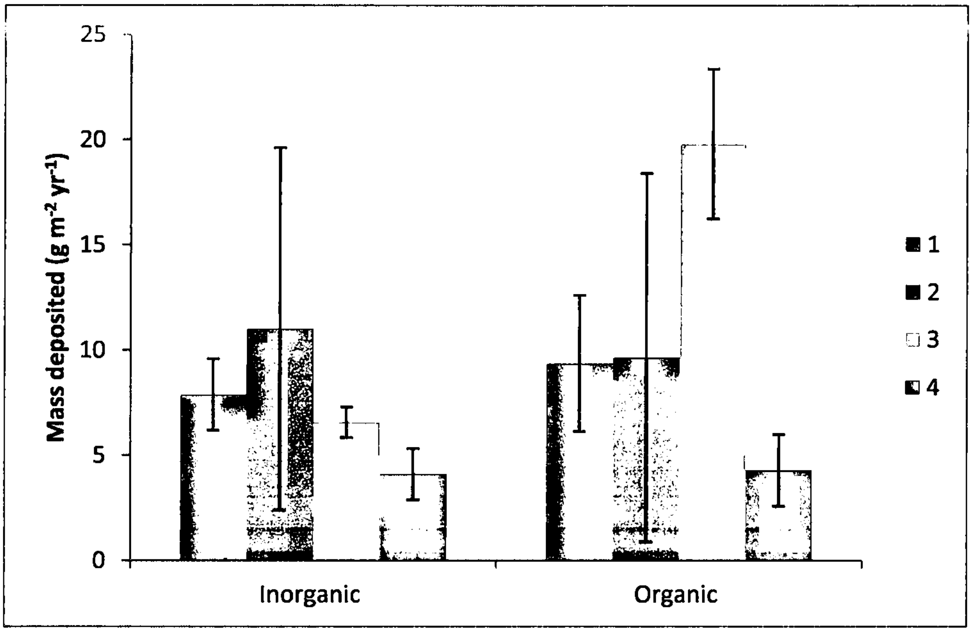

Fig. 6 Organic and inorganic deposition by site. Error bars indicate one standard error.

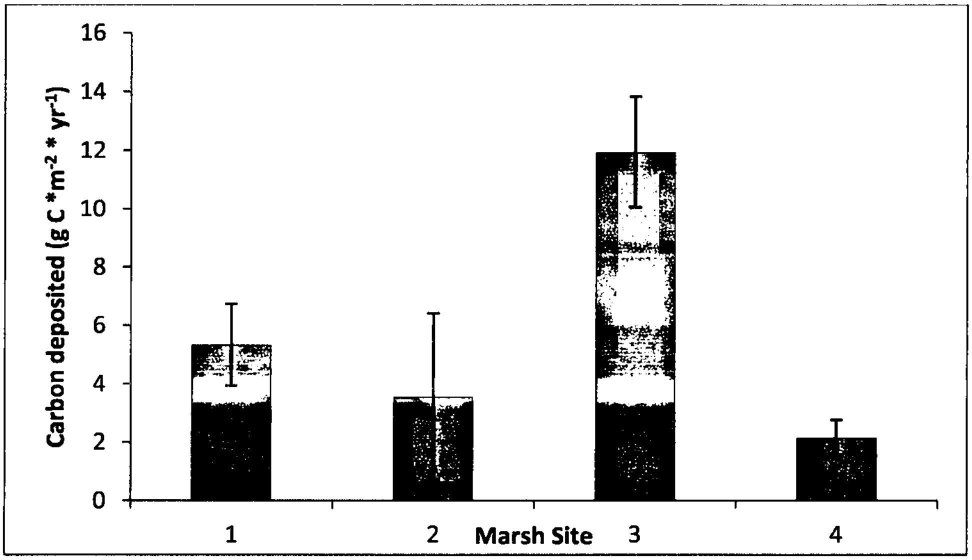

Fig. 7 Carbon deposited at each marsh site. Error bars indicate one S.E.

Table 3 Results of statistical analyses of sediment deposition. *Results of one way ANOVA comparing deposition at each marsh site. **Results of two way ANOVA comparing positions relative to probable overwash path and the trail berm.

Parameter	Sig.	F	df
Marsh*	.001	9.229	3
Trail Berm**	<.001	22.240	1
Overwash Path**	.053	4.293	1
Berm*Path**	.530	.411	1

Environmental variables

The groundwater salinity measurements were taken at various points in time (Table 4). There were statistically significant differences among marshes (ANOVA, $p = 0.006$). Sites directly in the overwash path (sites 1 and 2) had a significantly higher salinity (ANOVA, $p < 0.001$) than sites not in this path (sites 1 and 3), and there was no significant difference between sites blocked (sites 2 and 4) and unblocked (sites 1 and 3) by the trail berm (ANOVA, $p \geq 0.05$). Marsh 1 had a greater salinity than sites 3 or 4 (Bonferroni, $p_{13} = 0.003$, $p_{14} = 0.001$) at 19.1 ppt. Salinity at marsh site 2 was also significantly greater than marsh 3 (Bonferroni, $p_{23} = 0.042$, $p_{24} = 0.017$) and 4 at 15.8 ppt. These sites showed enormous variation throughout the course of the study, with several unusually high spikes in salinity occurring at sites 1 and 2 in late summer and fall. All of these sites had values much closer to freshwater during July of 2014 than in July of 2013; the difference in salinity between these two years was statistically significant (t-test, $p < 0.001$). In July of 2014 the salinity values of the ponds bordering each site were recorded, these ponds followed the same pattern as groundwater salinity, with sites 1 and

2 being the saltiest, followed by 3 and 4 (Table 5).

Precipitation data from the ABCRC were compiled continuously throughout the years of the study (Figure 8 and Figure 9). The rainfall was highest in 2013 on 06/03, 02/08, 10/08, 08/22, and 03/06. The 2014 rainfall was highest on 05/16, 04/25, 07/10, 02/13, and 04/15. Of these dates, only the 07/10 peak was within a week of a date when salinity was sampled.

Table 4 Salinity (parts per thousand) changes over time. *measurement from pond

Date	Marsh 1	Marsh 2	Marsh 3	Marsh 4
02/22	14	13.7	14	11
03/03	16	14	13	14
03/15	15	15	12	12
03/26	15	17	12	11
04/13	20	16.7	11.3	12
05/29	15	15	11.7	10.3
06/25	12	11.3	10	8.3
07/31	25.7	26.3	11.3	14.3
08/21	19	16	13	14
11/22	36.7	30	15	16.3
12/15	29	15	15	14
07/11/14	17	7.3	5.7	3
07/11/14*	9	6	1	3
07/15/14	14.3	7.7	5.3	4

The groundwater data from transducer wells were only available for three study sites (Figure 10). Marsh site 1 only had data from February to July 2013, marsh 2 had data from February to November 2013, and Marsh 3 had data available throughout the duration of the study. All three sites showed a similar pattern of inundation at the start of the year. The water table at site 1 dropped below the ground surface in July, while it fell below the ground surface at sites 2 and 3 in August and July 2013, respectively. Sites 2 and 3 had their lowest groundwater levels in October 2013, which quickly recharged. Hand augered wells (Table 6) 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 on 02/22 than on 05/29 (Bonferroni, $p = 0.003$), on 08/21 (Bonferroni, $p = 0.005$), and on 11/22 (Bonferroni, $p < 0.001$). In the summer the water table dropped below the ground surface at all sites, but recharged at different rates at each marsh site (ANOVA, $p = 0.003$) with sites 3 and 4 remaining below the ground surface on the 11/22 data collection (Figure 11).

Table 5 Statistical analyses for salinity. *Results of one way ANOVA comparing salinity at marsh sites; inverse transformation used. **Results of two way ANOVA comparing positions relative to trail berm and overwash path; inverse transformation used. ***Results of independent samples t-test comparing the 2013 and 2014 measurements; rank transformation used.

Parameter	Sig.	F	df
Marsh*	0.006	8.339	3
Berm**	0.382	0.780	1
Overwash**	<0.001	24.071	1
Berm*Overwash**	0.685	0.167	1
Year***	<0.001	0.580	112

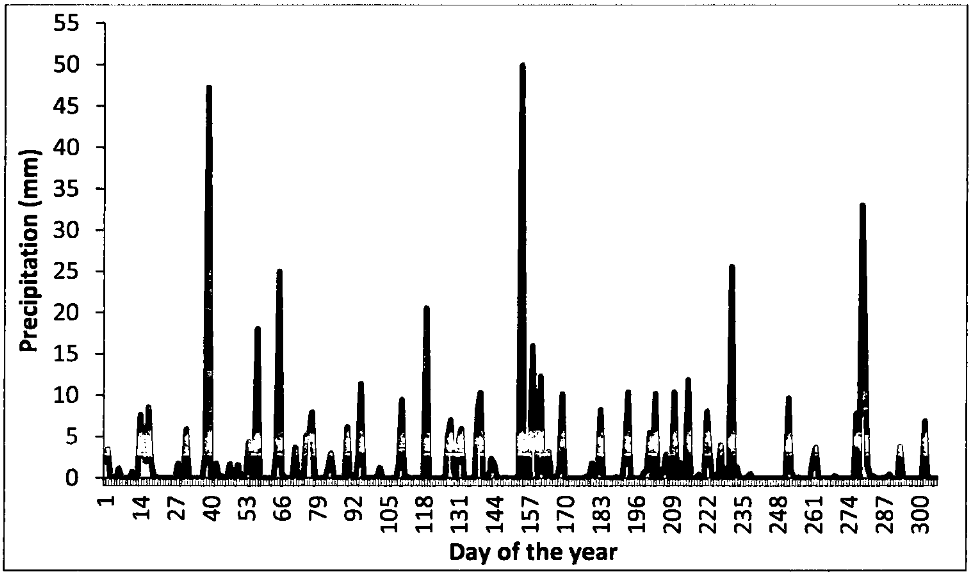

Fig. 8 Precipitation data on Hog island during 2013. Data run from January 1-November 5.

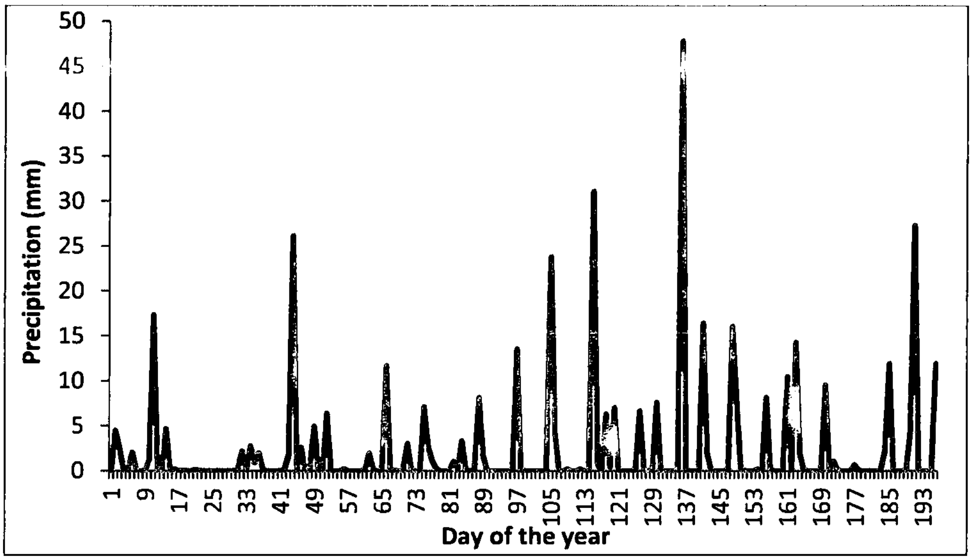

Fig. 9 Precipitation data on Hog island during 2014. Data are between January 1-July 15.

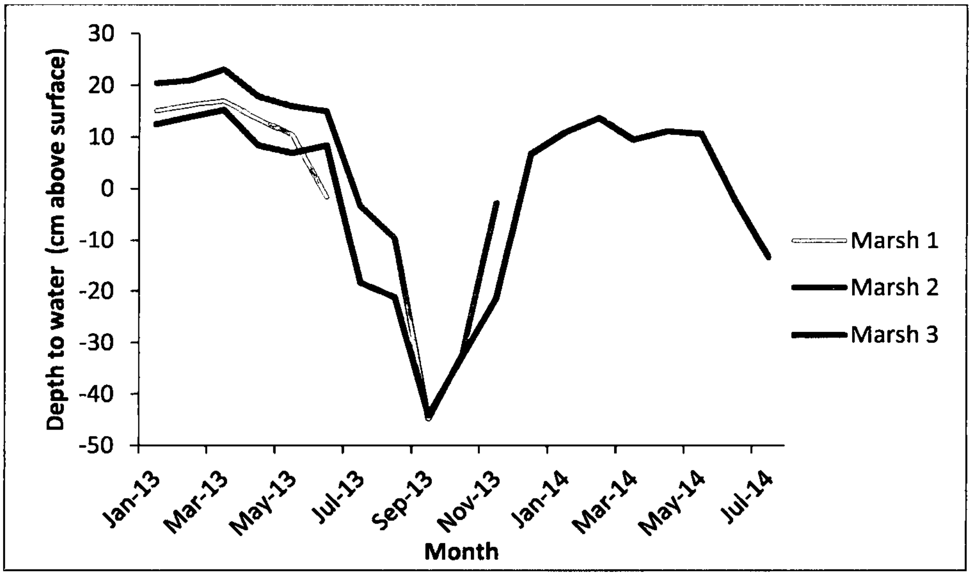

Fig. 10 Depth to ground water from transducer well data for three marsh sites.

Table 6 Statistical analyses for hand augered DTW data

Parameter	Sig.	F	df
Marsh	0.002	19.389	3
Collection date	<0.001	63.882	4
Marsh*Collection date	<0.001	15.294	12

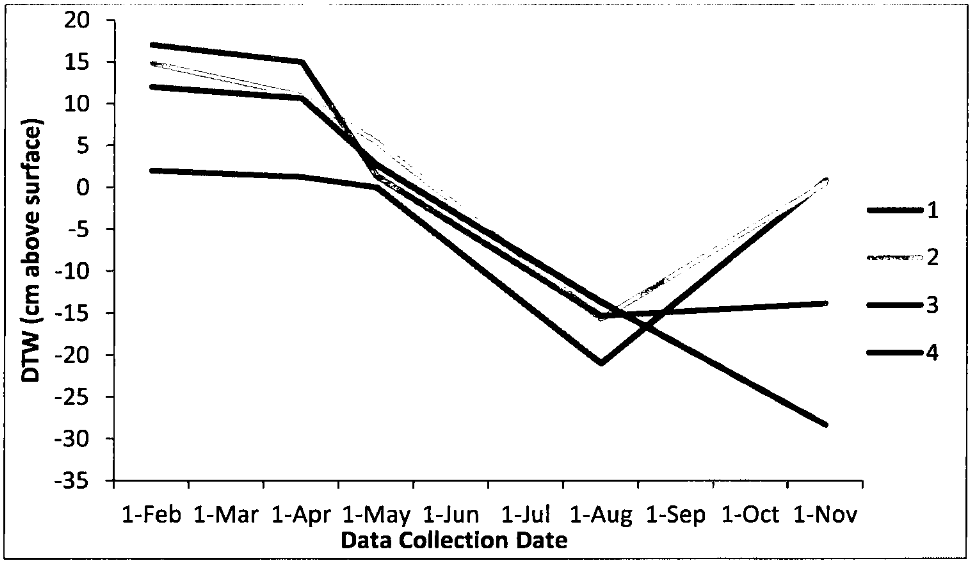

Fig. 11 Depth to ground water from hand dug wells for all marsh sites. Only data from 02/22, 04/13, 05/29, 08/21, and 11/22 are used in constructing this graph.

Differences in ground surface elevation (Table 7) were found among marsh sites (ANOVA, p = 0.003). The average elevation relative to sea level was greatest at site 4 (1.42 m), followed by site 3 (1.40 m), site 2 (1.33 m), and site 1 (1.32 m). Ground surface elevation at marsh site 1 was significantly lower than at marsh sites 3 (Bonferroni, p = 0.039) and 4 (Bonferroni, p = 0.035). Elevation at marsh site 2 was lower than at marsh sites 3 (Bonferroni, p = 0.044) and 4 (Bonferroni, p = 0.039).

Table 7 One way ANOVA results for rank transformed elevation.

Parameter	Sig.	F	df
Marsh	0.003	5.573	3

C Analyses

There were no significant differences in bulk density (Table 8) among marsh sites (ANOVA, $p \leq 0.05$), but there was a significant effect due to depth (ANOVA, $p < 0.001$). Bulk density at marsh site 2 was 0.7545 g cm^{-3}, while it was nearly identical at sites 1 (0.8821 g cm^{-3}) and 3 (0.8704 g cm^{-3}). All depth classes (0-10 cm, 11-20 cm, 21-30 cm) were significantly different from each other (Bonferroni, $p < 0.001$). There was no significant difference based upon position relative to the trail berm (ANOVA, $p \leq 0.05$) or overwash path (ANOVA, $p \leq 0.05$), and there were no interaction effects between any of these factors (ANOVA, $p \leq 0.05$).

Soil carbon content varied among marsh sites (ANOVA, $p < 0.001$; Table 9). Marsh sites 1 (1371.3 g C m^{-2}) and 3 (465.9g C m^{-2}), the sites protected by the trail berm, had lower (ANOVA, $p = 0.001$) carbon densities than sites 2 (3294 g C m^{-2}) and 4 (4749 g C m^{-2}) from 0-30 cm depth. The order of soil carbon density by marsh site is nearly the opposite of the order for soil bulk density. At the sediment surface all four sites had nearly the same soil C:N ratio, but this was not the case from depths of 31-100 cm. Site 4 (11.79) had the lowest C:N ratio, followed by site 3 (12.54), site 2 (23.02), and site 1 (32.50).

The maximum carbon available for storage annually, or sequestration potential, was expected to be different for each site depending upon their position and overwash vulnerability. Marsh site 1 (unblocked by the trail berm and in direct overwash path) had the highest estimated sequestration potential (233.4 g C m^{-2} yr^{-1}), followed by marsh sites 2 (blocked by trail berm and in direct overwash path; 172.8 g C m^{-2} yr^{-1}), 3 (unblocked by trail berm and not directly in overwash path; 153.3 g C m^{-2} yr^{-1}), and 4 (blocked by trail berm and not directly in overwash path; 116.6 g C m^{-2} yr^{-1}). These sequestration potentials almost match the order from highest to lowest carbon density, but marsh site 2 has the second highest soil carbon density.

Table 8 Statistical analyses for rank transformed soil bulk density. *Results of 2 way ANOVA comparing marsh and depth. **Results of 3 way ANOVA comparing depth, and positions relative to berm and overwash path.

Parameter	Sig.	F	df
Marsh*	0.364	1.111	3
Depth*	<0.001	64.707	2
Marsh*Depth*	0.835	0.455	6
Berm**	0.169	2.011	1
Overwash**	0.389	0.711	1
Berm*Overwash*Depth**	0.801	5.444	2

Table 9 Statistical analyses for inverse transformed top 30 cm soil carbon content. *Results of 1-way ANOVA comparing marsh sites. **Results of 2-way ANOVA comparing positions relative to berm and overwash path.

Parameters	Sig.	F	df
Marsh*	< 0.001	10.328	3
Berm**	0.001	15.148	1
Overwash**	0.014	7.285	1

DISCUSSION

BNPP and decomposition

Annual BNPP at each marsh site was less than the average obtained from literature values of salt marshes (Table 10). The range of the literature BNPP values (676-3500 g dry mass m^{-2} yr^{-1}) for 0-30 cm demonstrates both high variability and greater production than marshes sampled on Hog Island. Marsh 3 consistently showed high spatial variability, likely due to the *Schoenoplectus americanus* rhizomes growing in this site. During the first collection, two of these rhizomatous samples collected had many thin hair-like roots growing off of the rhizomes, perhaps as an adaptation to obtain scarce O_2 in the flooded soil (Armstrong et al. 1991). Marsh site 4 showed the most variability during the third collection when several samples had been completely covered by *Phragmites australis* detritus and contained little live growth, indicating that the nearby vegetation was being smothered as the *Phragmites* stand extends further into the marsh site. The final collection included substantial time during the Fall when belowground biomass tends to accumulate before these resources are allocated to aboveground structures in the Spring (Schubauer and Hopkinson 1984).

Production, in both this study and most of the literature values referenced, is based only on one year (Table 10). A study by Blum (1993) measured BNPP of a salt marsh changing from 1,253 to 99 g dry mass m^{-2} yr^{-1} the next growing season, demonstrating that BNPP shows enormous temporal variability depending on growing season conditions; some studies report high variation of BNPP samples collected at the same time from the same location (Blum and Christian 2004). Regardless of the one year sample, salt marshes are more productive than the Hog Island marshes belowground, though they were expected to be for several reasons. Salt marshes typically have a high root:shoot production ratio (Valiela et al. 1976), averaging 4-5.1 in the Northeastern United States (Darby and Turner 2008). There is no uniform approach to measuring BNPP (Schubauer and Hopkinson 1984), as such the referenced literature values from salt marshes use different techniques. The short term in-growth method used in this study is useful for comparisons between sites and over time, but disturbance of nearby vegetation may underestimate absolute root production (Neil 1992; Hendricks et al. 2006). While salt marshes are more productive, these systems may still be considered blue carbon systems if a substantial amount of carbon produced is preserved. The BNPP measured during this study does not indicate blue carbon potential in Hog Island marshes.

Table 10 Dry mass produced at each site and salt marsh BNPP from the literature: (Valiela et al. 1976 (25 cm), Smith et al. 1979 (30 cm), Livingstone and Patriquin 1981 (30 cm), Bellis and Gaither 1985 (30 cm), Blum 1993 (50 cm)). *This value combines different methods in different geographic locations.

Source	BNPP (g dry mass m^{-2} yr^{-1})	S.E.
Marsh 1	125.9	4.934
Marsh 2	112.7	10.68
Marsh 3	178.5	15.80
Marsh 4	125.5	9.645
Salt marshes	676-3500*	-

The decay coefficients were converted to a yearly value in order to be comparable with literature values. Marsh sites 1 (0.4825 yr^{-1}) and 2 (0.4586 yr^{-1}) had the lowest decay coefficients of sampled marsh sites (Table 11). These values are similar, but are not as low as the decay coefficients of salt marshes (0.3013 yr^{-1}) gathered from the literature. These samples showed high spatial variability, like BNPP, samples from site 3 showed the most variation. At one location in site 3, all 5 samples had to be discarded due to gains in biomass, likely indicating that the vegetation at this location had high root growth and the roots could not be removed by tweezers. Several of these samples were discarded as some *Schoenoplectus americanus* rhizomes grew throughout the growing season. There were other litterbags that had gained mass during the study; previous studies have attributed gains in litterbag mass to accumulating microbial biomass or to iron-pyrite mixing with the roots in the marsh soil (Heminga et al. 1988). Two of the litterbags that had gained mass in marsh sites 1 and 2 showed redoxymorphic features, indicating anoxic conditions and potentially accumulation of iron-pyrite. Sediment deposition in the bags, though washed, could have contributed some weight gain.

Environmental parameters on barrier islands can be highly variable over time; the salinity and groundwater measurements depend upon frequency of precipitation and overwash events. Marsh site 3 had the highest water table during the months with available well transducer data. During the Fall collection, the water table remained below the ground surface at sites 3 and 4. Marsh sites 3 and 4 had lower groundwater salinity, but also higher BNPP and a more rapid rate of decay than marsh sites 1 and 2. Since marsh site 3 has the highest water table, one would expect decreased decay rates due to anoxic conditions; this is especially true when combined with decreased salinity, as added SO_4^{2-} from seawater stimulates microbial decay while depressing methane production (Chmura et al. 2003; Poffenbarge et al. 2011). The lower water table could have possibly led to the higher BNPP and decay rates on sites 3 and 4. In addition, it is likely that the priming effect played a role, as organic matter inputs to the soil tend to stimulate microbes, leading to higher decay rates (Fontaine et al. 2004). Some studies have suggested NPP plays little role in long-term carbon storage of wetlands (Cebrian and Duarte 1995; Craft 2007), possibly due to high export from coastal systems.

Recorded salinities during 2013 and 2014 were significantly different, as is to be expected from the dynamic nature of this environment and from the varying salinities previously recorded in these marshes [0-22 ppt. (Conn and Day 1997)]; [0 and 0.73 ppt. (Blecha 2010)]. Salinity data show two spikes for marsh sites 1 and 2, but a steady pattern for sites 3 and 4 with a slight increase at the end of the 2013 growing season. These salinity spikes were detected during the July 31st collection and the November

22nd collection. While the well transducer data were not available for marsh site 1 during either of these collections, well data at marsh sites 2 and 3 showed a marked decrease in groundwater levels during the summer months, with site 2 having a rapid increase during October and November. Typical salinity measurements during this study were 10-19 ppt, making the salinity surges in July (25.7 and 26.3) and November (36.7 and 30) unusual, even for marsh sites 1 and 2. The salinity spike during summer 2013 can perhaps be explained by low precipitation (Figure 8), but the salinity spike during the late fall may be evidence of a minor overwash event. Additional evidence of a minor overwash comes from oxidized iron collected from sediment plates of marsh site 1 only. Any overwash that occurred was insignificant enough to only leave this deposit on 4 of the 5 plates collected from this site. The frequency of overwashes and precipitation can cause freshwater/brackish marshes to suddenly change in salinity. Salt does not leave the soil pores as easily as freshwater and the salt build-up leads to a trend of increasing salinity over long time periods on barrier islands (Hayden et al. 1995; Booth et al. 1999; Mark Hester, personal communication). These salinity fluctuations can alter the plant community, with more salt-tolerant plants invading previously freshwater marshes. Electron acceptors found in seawater increase anaerobic decay rates, meaning increasing salinity will initially decrease carbon sequestration rates. This increase in salinity will also lead to decreased methane emissions, eventually leading to higher sequestration rates.

Decay was lowest at the sites that are closest to the direct overwash path (sites 1 and 2), but was still faster than the decay rates found in the literature for salt marshes from 0-20 cm depth ($k = 0.3013\ yr^{-1}$). A previous study in Hog Island swales found significant differences in belowground decay between marshes along a chronosequence (Conn and Day 1997). This chronosequence reflects differences in age between our sites 3 and 4 versus sites 1 and 2, so the significant difference in this study ($p = 0.018$) has been observed previously. The decay rates measured by Conn and Day (1997) were slower than currently measured rates; however, after approximately 2/3 of a year (the duration of this study), the average percent mass remaining at sites 1 and 2 (77.9 %) is nearly the same as what Conn and Day measured for these marshes (~75%). Average percent of initial mass remaining was much lower at sites 3 and 4 (57.4 %) than what Conn and Day measured (~70 %) for this duration. Since sites 3 and 4 were formed in 1967, it is not surprising to see these decay rates changed more in approximately 20 years than decay rates at the swales formed in 1871 (sites 1 and 2). Given the observed rapid invasion of *Phragmites australis* at marsh site 4, it is possible that the soil chemistry is changing over time due to high annual detrital inputs. The decay rates at marsh sites 1 and 2 were comparable to, but still greater than salt marshes; thus belowground decay indicates that the sites directly in the overwash path have moderate blue carbon potential. The combination of greater BNPP and slower decay rates in salt marshes would result in a larger organic carbon pool leftover at the end of each growing season. While the belowground growth and decay dynamics indicate that these sites do not store as much organic carbon as salt marshes, these marshes still have the potential to store a great deal of carbon.

Table 11 Belowground decay coefficients for sampled marsh sites compared to literature values for Virginia salt marsh (Blum and Christian 2004).

Source	k (yr^{-1})
Marsh 1	0.4825
Marsh 2	0.4586
Marsh 3	0.7020
Marsh 4	1.134
Salt Marshes	0.3013

Sedimentation and soil

Sediment deposition showed high variance within the same sites. Other studies have shown high variability with sedimentation measurements, especially during late fall and winter when storms are more frequent (Proosdij et al. 2006). Several of these plates had been completely buried by detritus in the field. Marsh site 2 had an enormous standard error; only 1 of these plates had been buried in the field, but the other plates collected little material. SPSS boxplots identified this plate as an outlier and it had to be removed in order to meet ANOVA assumptions. The dry mass of sediment collected from each marsh site was compared to the literature values for nearby Virginia salt marshes (Table 12). Sedimentation was only compared to nearby Phillips Creek and Chimney Pole marsh, but these values (580 g dry mass m^{-2} yr^{-1}) were much higher than sedimentation observed during the course of this study (Katler and Wiberg 1996). These salt marshes receive frequent tidal inundation, naturally leading to higher sedimentation rates. During years with large storms and overwash events, sedimentation would be higher in our marshes. The lack of major overwashes during the year of this study depressed sediment deposition in these sites relative to what can normally be expected.

All sampled sites showed a higher surface bulk density (Table 13) than South Atlantic salt marshes (0.52 g cm^{-3}). These sites also had a much lower soil carbon density than the average of tidal saline wetlands (TSWs) of 0.039 g C cm^{-3}; while the average of TSWs is much higher than these sites, marsh sites 2 and 4 both fell within the range of recorded soil carbon densities (0.009-0.120 g C cm^{-3}). Low sedimentation rates will make surface soils have a low bulk density, but high organic carbon concentration (Chmura et al. 2003). This pattern was observed in the study, as the sites with the highest sedimentation rates, sites 3 and 1, also had the highest bulk densities and the lowest soil carbon concentrations.

Soil C:N ratios were nearly identical at the surface sediments, though marsh site 3 was slightly lower than the rest (Table 13). From 31-100 cm depth, however, there were distinct differences between these sites, with the sites listed from highest to lowest C:N ratios as 1, 2, 3, 4. Interestingly, the sites with the highest C:N ratios at the bottom layers of sediment also had the highest BNPP and decay rates. C:N ratios have been previously proposed as driving factors behind decay rates (Hemminga et al. 1988; Blum and Christian 2004). The greater sediment nitrogen concentration at marsh sites 3 and 4 may be due to nitrogen fixation from the surrounding *Morella cerifera*, which cannot grow if the soil is too saturated year-round (Shao et al. 1996). Previous studies have shown a greater concentration of soil nitrates and ammonium in the older marsh sites (1 and 2) than the younger ones (Stevenson and Day 1996). Our results show a higher concentration of nitrogen in the younger sites; combined with the invasion of the nitrophilous *Phragmites australis*, the nitrogen distribution trend appears to now be the opposite of what it once was along the chronosequence. The North end of Hog Island is accreting seaward at about 5 mm every year (Hayden et al. 1991), in the direction of the younger swales. It is possible that marsh sites 3 and 4 are accreting vertically high enough to allow the *Morella cerifera* and *Phragmites australis* to expand into the marsh site, which would explain the difference in soil nitrogen concentration; these sites are converting from low marshes to high marshes, which would also explain the possible lack of an extended period of soil anoxia in marsh site 3 during the study.

Table 12 Sedimentation and standard error from each marsh compared to salt marsh literature values (Katler and Wiberg 1996).

Source	Deposition (g m^{-2} yr^{-1})	S.E.
Marsh 1	17.26	4.510
Marsh 2	18.00	14.42
Marsh 3	26.37	4.171
Marsh 4	8.388	2.388
Salt marshes	580	84.85

Table 13 Soil bulk density, carbon density, and C:N ratios of sampled sites and literature values. Unless otherwise stated, all parameters refer to the top 30 cm of soil. *From Craft 2007 compilation of South Atlantic salt marsh data. **Tidal Saline Wetland values obtained from Chmura et al. 2003.

Source	B.D. (g cm^{-3})	Carbon Density (g C cm^{-3})	C:N (0-30 cm)	C:N (31-100 cm)
Marsh 1	0.8821	0.004571	17.03	32.50
Marsh 2	0.7545	0.01098	17.65	23.02
Marsh 3	0.8704	0.001553	14.85	12.54
Marsh 4	0.8394	0.01583	17.55	11.79
Salt marshes	~0.52 *	0.039**	N/A	N/A

Carbon sequestration potential

Carbon sequestration potential was greatest at marsh site 1 (233.4 g C m^{-2} yr^{-1}), which also had the highest bulk density, but one of the lower soil carbon densities (Table 13). The sequestration potential (Table 14) at marsh site 4 was the lowest of all sites sampled, but the surface soil carbon concentration was the highest. Since sequestration potentials assume that plant biomass is at steady state and do not include processes that occur after organic matter is deposited (such as export and nonbiological oxidation), the sequestration potentials are expected to be higher than the actual annual sequestration values. All of the estimated potentials fit within the global average sequestration range (18-1713 g C m^{-2} yr^{-1}) for salt marshes.

Carbon sequestration potentials followed a pattern that was expected at the start of the study due to known overwash vulnerability differences in these marshes. Marsh site

1 (unblocked by trail berm and in direct overwash path) had a greater sequestration potential than the others; marsh sites 2 (blocked by trail berm and in direct overwash path) and 3 (unblocked by trail berm and not directly in overwash path) were approximately equal; sequestration potential at marsh site 4 (blocked by trail berm and not directly in overwash path) was the lowest (Table 14). The sequestration potential being lowest at sites 3 and 4 appears to reflect higher aboveground and belowground decay rates at these sites. Since the sequestration potentials of these marshes were in the relative order where they were expected to be, but null hypotheses related to overwash vulnerability were accepted, several conclusions can be drawn. Protection by the trail berm does not appear to have a noticeable damming effect when overwashes occur. Of the statistical analyses run, only sediment deposition showed a significant difference between sites based upon their protection by the trail berm. The older sites had a higher sequestration potential than the younger sites, perhaps indicating that the younger sites (3 and 4) will increase in their sequestration potential with age. Being directly in the overwash path appears to have had an influence on sequestration potential, as assessed by both decay and salinity tests. While we may have initially been wrong about the effect of the trail berm, a higher frequency of overwash appears to increase sequestration potential of these marshes and make them more comparable to blue carbon systems. A potential follow up study should assess multiple years of these variables, as well as other processes that affect carbon storage, in order to create a more accurate comparison.

The average carbon sequestration rate for salt marshes (218 ± 24 g C m^{-2} yr^{-1}) is higher than all Hog Island potentials, except for marsh 1, suggesting that the method used for calculating sequestration potential is accurate enough to make realistic comparisons.

By all measures from this study, salt marshes are capable of storing more carbon than the marshes sampled. Comparisons with seagrass beds (138 ± 38 g C m^{-2} yr^{-1}) and mangroves (228 ± 39 g C m^{-2} yr^{-1}) suggest the Hog Island marshes are blue carbon systems (Mcleod et al. 2011). As barrier island wetlands, however, the Hog Island marshes are subject to different successional processes than those that affect typical blue carbon ecosystems. Over time, these marshes may keep pace with rising sea level and build up carbon stores, or they may be lost if sea level rise exceeds the rate of marsh surface accretion. If sea level rises too slowly, the wetlands may disappear as woody vegetation displaces marsh grasses. Given the brief and highly variable nature of barrier island marshes, studying and conserving them should be a high priority.

Table 14 Carbon sequestration potential for all marsh sites sampled compared to compiled literature values of salt marshes (Mcleod et al. 2011).

Site	Sequestration potential(g C m^{-2} yr^{-1})
Marsh 1	233.4
Marsh 2	172.8
Marsh 3	153.3
Marsh 4	116.6
Salt marshes	218

REFERENCES

Armstrong W, Justin SHFW, Beckett PM, Lythe S (1991) Root adaptation to soil waterlogging. Aquatic Botany 39:57-73.

Aufdenkampe AK, Mayorga E, Raymond PA, Melack JM, Doney SC, Alin SR, Aalto RE, Yoo K (2011) Riverine coupling of biogeochemical cycles between land, oceans, and atmosphere. Frontiers in Ecology 9:53-60.

Blecha SB (2010) Inter-island variability in above and belowground plant biomass in interior marshes on the Virginia Barrier Islands. Thesis, Old Dominion University.

Blum LK, Christian RR (2004) Below ground production and decomposition along a tidal gradient in a Virginia salt marsh. In: Fagherazzi S et al. (eds) The Ecogeomorphology of Tidal Marshes. Washington D.C., pp. 47-74.

Booth RK, Rich FJ, Bishop GA, Brannen NA (1999) Evolution of a freshwater barrier-island marsh in coastal Georgia, USA. Wetlands 19(3):570-577.

Bouillon S, Borges AV, Castaneda-Moya E, Diele K, Dittmar T, Duke NC, Kristensen E, Lee SY, Marchand C, Middelburg JJ, Rivera-Monroy VH, Smith III TJ, Twilley RR (2008) Mangrove production and carbon sinks: a revision of the global carbon budget estimates. Global Biogeochemical Cycles 22:1-12.

Brinson MM, Lugo AE, Brown SL (1981) Primary productivity, decomposition, and consumer activity in freshwater wetlands. Annual Review of Ecology Systematics 12:123-161.

Cebrian J, Duarte CM (1995) Plant growth-rate dependence of detrital C-storage in ecosystems. Science 268:1606-1608.

Chmura GL, Anisfeld SC, Cahoon DR, Lynch JC (2003) Global carbon sequestration in tidal, saline wetland soils. Global Biogeochemical Cycles 17(4):1-12.

Collins WH, Easley DH (1999) Fresh-water lens formation in an unconfined barrier island aquifer. Journal of the American Water Resources Association 35(1): 1-21.

Conn CE, Day FP (1997) Root decomposition across a barrier island chronosequence: litter quality and environmental controls. Plant and Soil 195:351-364.

Craft C (2007) Freshwater input structures soil properties, vertical accretion, and nutrient accumulation of Georgia and U.S. tidal marshes. Limnology and Oceanography 52(3):1220-1230.

Darby FA, Turner RE (2008) Below-and aboveground *Spartina alterniflora* production in a Louisiana salt marsh. Estuaries and Coasts 31(1):223-231

Day FP (1982) Litter decomposition rates in the seasonally flooded Great Dismal Swamp. Ecology 63:670-678.

Drexler JZ, Krauss KW, Sasser MC, Fuller CC, Swarzenski CM, Powell A, Swanson KM, Orlando J (2013) A long-term comparison of carbon sequestration rates in impounded and naturally tidal freshwater marshes along the lower Waccamaw River, South Carolina. Wetlands 33:965-974.

Duarte CM, Cebrian J (1996) The fate of marine autotrophic production. Limnology and Oceanography 41:1758-1766.

Duarte CM, Middelburg J, Caraco N (2005) Major role of marine vegetation on the oceanic carbon cycle. Biogeosciences 2:1-8.

Fahrig LB, Hayden BP, Dolan R (1993) Distribution of barrier island plants in relation to overwash disturbance: a test of life history theory. Journal of Coastal Research 9:403-412.

Fontaine S, Bardoux G, Abbadie L, Mariotti A (2004) Carbon input to soil may decrease soil carbon content. Ecology Letters 7:314-320.

Gacia E, Duarte CM (2001) Elucidating sediment retention by seagrasses: sediment deposition and resuspension in a Mediterranean (*Posidonia oceanica*) meadow. Estuarine Coastal and Shelf Science 52:505-514.

Gattuso JP, Frankignoulle M, Wollast R (1998) Carbon and carbonate metabolismin coastal aquatic ecosystems. Annual Review of Ecological Systems 29:405-434.

Hayden BP, Dueser RD, Callahan JT, Shugart HH (1991) Long-term research at the Virginia Coast Reserve: modeling a highly dynamic environment. Bioscience 41(5):310-318.

Hayden BP, Santos MCFV, Shao G, Kochel RC (1995) Geomorphological controls on coastal vegetation at the Virginia Coast Reserve. Geomorphology 13:283-300.

Hedges JI (1992) Global biogeochemical cycles: progress and problems. Chemistry 39:67-93.

Hemminga MA, Kok CJ, de Munck W (1988) Decomposition of *Spartina anglica* roots and rhizomes in a salt marsh of the Westerschelde Estuary. Marine Ecology Progress Series 48:175-184.

Hendricks JJ, Hendrick RL, Wilson CA, Mitchell RJ, Pecot SD, Guo D (2006) Assessing the patterns and controls of fine root dynamics: An empirical test and methodological review. Journal of Ecology 94:40-57.

Houghton RA (2007) Balancing the global carbon budget. Annual Review of Earth and Planetary Sciences 35:313-347.

Kastler JA, Wiberg PL (1996) Sedimentation and boundary changes of Virginia salt marshes. Estuarine, Coastal and Shelf Science 42:683-700.

Kennedy H, Beggins J, Duarte CM, Fourqurean JW, Holmer M, Marba N, Middelburg JJ (2010) Seagrass sediments as a global carbon sink: isotopic constraints. Global Biogeochemical Cycles 24:GB4026.

Killips S,Killops V(2005) Introduction to Organic Geochemistry. Second Edition. Blackwell, United Kingdom.

Kleiss BA (1993) Methods for measuring sedimentation rates in bottomland hardwood wetlands. U.S. Core of Army Engineers.WRP Tech Note SD-CP-4.1.

Krauss KW, Whitbeck JL, Howard RJ (2012) On the relative roles of hydrology, salinity, temperature, and root productivity in controlling soil respiration from coastal swamps (freshwater). Plant Soil 358:265-274.

Leatherman S (1988) Barrier Island Handbook. Coastal Publication Series, College Park, Maryland.

Lovett GM, Cole JJ, Pace ML (2006) Is net ecosystem production equal to carbon storage? Ecosystems 9:152-155.

Mateo MA, Romero J, Perez M, Littler MM, Littler DS (1997) Dynamics of millenary organic deposits resulting from the growth of the Mediterranean seagrass *Posidonia oceanica.* Estuarine Coastal Shelf Science 44:103-110.

McKee KL, Cahoon DR, Feller IC (2007) Caribbean mangroves adjust to rising sea level through biotic controls on change in soil elevation. Global Ecology and Biogeography 16:545-556.

Mcleod E, Chmura GL, Bouillon S, Salm R, Bjork M, Duarte CM, Lovelock CE, Schlesinger WH, Silliman BR (2011) A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO2. Frontiers in Ecology 9(10):552-560.

Morris JT, Shaffer GP, Nyman JA (2013) Brinson Review: Perspective on the influence of nutrients on the sustainability of coastal wetlands. Wetlands 33:975-988.

Neil C (1992) Comparison of soil coring and ingrowth methods for measuring belowground production. Ecology 73(5):1918-1921.

Nellemann C, Corcoran E, Duarte CM, Valdes L, De Young C, Fonseca L, Grimsditch G (2009) Blue Carbon: The Role of Healthy Oceans in Binding Carbon. A Rapid Response Assessment. GRID, Arendal.

Olson, JS (1963) Energy storage and the balance of producers and decomposers in ecological systems. Ecology 44(2):322-331.

Pendleton L, Donato DC, Murray BC, Crooks S, Jenkins WA, Silfleet S, Craft C,Fourqurean JW, Kauffman JB, Marba N, Megonigal P,Pidgeon E, Herr D, Gordon D, Balderal A (2012) Estimating global "blue carbon" emissions from conversion and degradation of vegetated coastal ecosystems. PLoS ONE 7(9):1-7.

Poffenbarge HJ, Needelman BA,Megonigal JP (2011) Salinity influence on methane emissions from tidal marshes. Wetlands 31:831-842.

Proosdij, DV, Ollerhead, J, Davidson-Arnott, RGD (2006) Seasonal and annual variations in the volumetric sediment balance of a macro-tidal salt marsh. Marine Geology 225:103-127.

Quirk TE, Seliskar DM,Sommerfield CK, Gallagher JL (2011) Salt marsh carbon pool distribution in a mid-Atlantic lagoon, USA: sea level rise implications. Wetlands 31:87-99.

Romero J, Perez M, Mateo MA, Sala E (1994) The belowground organs of the Mediterranean seagrass *Posidonia oceanica* as a biogeochemical sink. Aquatic Botany 47:13-19.

Roper T, Greskowiak G, Freund H, Massmann G (2013) Freshwater lens formation below juvenile dunes on a barrier island (Spiekeroog, Northwest Germany). Estuarine, Coastal, and Shelf Science 121:40-50.

Salinas LM, DeLaune RD, Patrick Jr. WH (1986) Changes occurring along a rapidly submerging coastal area: Louisiana, USA. Journal of Coastal Research 2(3):269-284.

Schlesinger WH, Lichter J (2001) Limited carbon storage in soil and litter of experimental forest plots under elevated atmospheric CO_2. Nature 411:466-469.

Schubauer JP, Hopkinson CS (1984) Above and belowground emergent macrophyte production and turnover in a coastal marsh ecosystem, Georgia. Limnology and Oceanography 29(5):1052-1065.

Shao G, Shugart HH, Hayden BP (1996) Functional classifications of coastal barrier island vegetation. Journal of Vegetation Science 7(3):391-396.

Siegenthaler U, Sarmiento JL (1993) Atmospheric carbon dioxide and the ocean. Nature 365:119-125.

Stevenson M, Day FP (1996) Fine-root biomass distribution and production along a barrier island chronosequence. American Midland Naturalist 135:205-217.

Terry JP, Falkland AC (2010) Reponses of atoll freshwater lenses to storm-surge overwash in the Northern Cook Islands. Hydrogeology Journal 18:749-759.

Tupacz EG, Day FP (1989) Decomposition of roots in a seasonally flooded swamp ecosystem. Aquatic Botany 37:199-214.

Valiela, I, Teal, JM, Persson, NY (1976) Production and dynamics of experimentally enriched salt marsh vegetation: Belowground biomass. Limnology and Oceanography 21(2):245-252.

Van Sickle VR, Barrett BB, Ford TB (1976) Baratarian Basin: salinity changes and oyster distribution. Sea Grant Publication No. LSU-76-002.

Wieder RK, Lang GE (1982) A critique of the analytical methods used in examining decomposition data obtained from litter bags. Ecology 63:1636-1642.

Young DR, Brantley ST, Zinnert JC, Vick JK (2011) Landscape position and habitat polygons in a dynamic coastal environment. Ecosphere 2(6):1-15.

VITA

Nathan Sedghi

I am finishing my Masters in Biology at Old Dominion University in May 2015. My Bachelors of Science in Biology, with a Chemistry minor, was also obtained from the Old Dominion University Department of Biology at 5115 Hampton Boulevard Norfolk, VA 23529, in December 2011.

RESEARCH EXPERIENCE

[ODU Department of Biological Sciences]
[Graduate Student] **[Jan 12-Dec 14]**
[ODU Department of Biology]
[Graduate Research Assistant] **[May-Aug 13]**
[ODU Department of Chemistry]
[Volunteer Undergraduate Research Assistant] **[Jan 10 – June 10]**

TEACHING EXPERIENCE

[ODU Department of Biological Sciences]
[Lab Instructor for BIO 105] **[Sept 14-Dec 14]**
[ODU Department of Biological Sciences]
[Lab Instructor for BIO 108, 109, and 111] **[Sept 13-Dec 14]**
[Wyzant.com tutoring]
[Certified tutor in Biology, Chemistry, and Algebra] **[Oct 10-Present]**
[ODU Department of Biological Sciences]
[Undergraduate Teaching Assistant for BIO 116] **[Fall 10]**

PAPERS PRESENTED AT SCIENTIFIC MEETINGS

Sedghi, N. M., and F. P. Day. Blue carbon in coastal freshwater marshes on the barrier islands of VA: belowground carbon pools. Association of Southeastern Biologists Annual Meeting. Spartanburg, SC. April 2014.

Sedghi, N. M., and F. P. Day. Blue carbon in coastal freshwater marshes on the barrier islands of VA: belowground carbon pools. Joint Aquatic Sciences Meeting. Portland, OR. May 2014.

OTHER EXPERIENCE

[Norfolk Naval Shipyard, Portsmouth, VA]
[Physical Science Technician **[May-Sept 12]**
[Jamali, Nichols, and Morales M.D.s]
[Shadowed a Physician **[Dec 08-May 09]**
[Wal-Mart Pharmacy]
[Registered Pharmacy Technician] **[Mar 08-Oct 09]**

The word processor for this Thesis was Microsoft Word.