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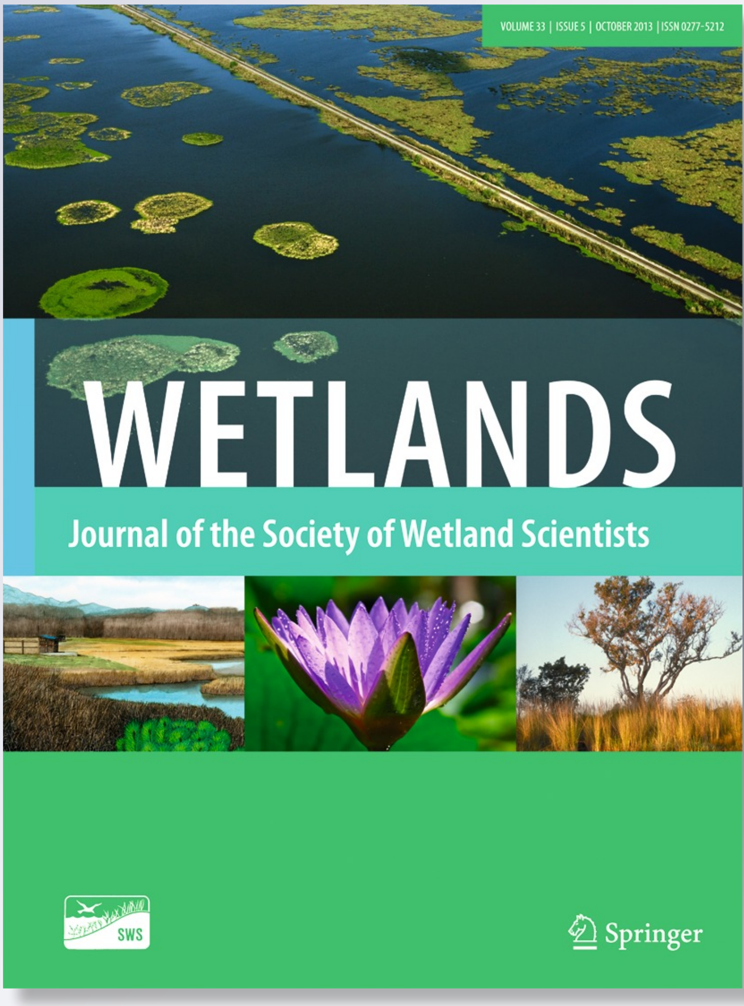

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ARTICLE

A Long-Term Comparison of Carbon Sequestration Rates in Impounded and Naturally Tidal Freshwater Marshes Along the Lower Waccamaw River, South Carolina

Judith Z. Drexler · Ken W. Krauss · M. Craig Sasser · Christopher C. Fuller · Christopher M. Swarzenski · Amber Powell · Kathleen M. Swanson · James Orlando

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Abstract Carbon storage was compared between impounded and naturally tidal freshwater marshes along the Lower Waccamaw River in South Carolina, USA. Soil cores were collected in (1) naturally tidal, (2) moist soil (impounded, seasonally drained since ~1970), and (3) deeply flooded "treatments" (impounded, flooded to ~90 cm since ~2002). Cores were analyzed for % organic carbon, % total carbon, bulk density, and ^{210}Pb and ^{137}Cs for dating purposes. Carbon sequestration rates ranged from 25 to 200 g C m^{-2} yr^{-1} (moist soil), 80–435 g C m^{-2} yr^{-1} (naturally tidal), and 100–250 g C m^{-2} yr^{-1} (deeply flooded). The moist soil and naturally tidal treatments were compared over a period of 40 years. The naturally tidal treatment had significantly higher carbon storage (mean=219 g C m^{-2} yr^{-1} vs. mean=91 g C m^{-2} yr^{-1}) and four times the vertical accretion rate (mean=0.84 cm yr^{-1} vs. mean=0.21 cm yr^{-1}) of the moist soil treatment. The results strongly suggest that the long drainage period in moist soil management limits carbon storage over time. Managers across the National Wildlife Refuge system have an opportunity to increase carbon storage by minimizing drainage in impoundments as much as practicable.

Keywords Carbon sequestration · Impounded · Tidal freshwater marsh · Vertical accretion

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J. Z. Drexler (✉) · A. Powell · K. M. Swanson · J. Orlando
U.S. Geological Survey, California Water Science Center,
6000 J Street, Sacramento, CA 95819, USA
e-mail: jdrexler@usgs.gov

K. W. Krauss
U.S. Geological Survey, National Wetlands Research Center,
700 Cajundome Blvd., Lafayette, LA 70506, USA

M. C. Sasser
Waccamaw National Wildlife Refuge, 21424 North Fraser Street,
P.O. Box 1439, Georgetown, SC 29440, USA

C. C. Fuller
U.S. Geological Survey, National Research Program,
345 Middlefield Rd., MS465, Menlo Park, CA 94025, USA

C. M. Swarzenski
U.S. Geological Survey, Louisiana Water Science Center,
3535 South Sherwood Forest Blvd., Baton Rouge, LA 70816, USA

Introduction

Carbon sequestration is an important ecosystem service provided by tidal marshes, wherein carbon is stored long-term in soils, thus reducing greenhouse gas emissions to the atmosphere. Tidal *freshwater* marshes have been found to have higher rates of carbon storage than brackish or saline tidal marshes (Bridgham et al. 2006; Craft 2007; Trulio et al. 2007). Unlike saline and brackish marshes, however, tidal freshwater marshes are also capable of high methane emissions, which can reduce or possibly even negate carbon sequestration gains due to the much greater global warming potential of methane (25x) vs. carbon dioxide (Kayranli et al. 2010; Poffenbarger et al. 2011). For this reason, it is important to ascertain which types of marshes have the potential to store the most carbon, thereby having the propensity to offset potential gaseous carbon losses. In their comparison of impounded and tidal marshes in Louisiana (salinity regimes not specified), Bryant and Chabreck (1998) showed that tidal marshes had vertical accretion rates and organic accumulation rates that were greater than impounded marshes, whose management included "periodic drying". These results suggest that there may be important differences in carbon storage and other soil formation processes between impounded and naturally tidal marshes. Furthermore, the particular management regime of a marsh may have a strong impact on its rate of carbon storage.

National Wildlife Refuges along the southeastern coast of the United States contain naturally tidal marshes as well as impounded marshes managed for waterfowl and wildlife habitat (Gresham and Hook 1982; Doyle et al. 2007). In some refuges, these different marsh types occur close together and are managed to accomplish a variety of habitat goals. The carbon storage capacities of these marshes *in relation to each other* have yet to be quantified. Such information is needed by refuge managers, who are not only responsible for managing wildlife populations of interest, but are also tasked with strategic habitat management, which includes maintaining and developing the ecosystem services, such as carbon sequestration, provided by refuge habitats. In addition, a greater understanding of carbon sequestration rates would enable managers to reap the potential benefits of a developing carbon reduction market, which could provide much-needed funds for refuges in exchange for enhancing and restoring marsh habitat for the purpose of carbon sequestration.

The goal of this study was to compare soil formation processes in impounded vs. naturally tidal, freshwater marsh habitats in the Waccamaw National Wildlife Refuge and adjacent areas along the Lower Waccamaw River, South Carolina, USA. In particular, we compared rates of carbon sequestration, mass accumulation rate, organic matter accumulation, inorganic sedimentation, and vertical accretion in impounded vs. naturally tidal marshes to better understand how impoundment and associated management activities affect these soil processes.

Study Sites

Land Use History

The study area consists of wetlands and associated river systems in the Winyah Bay drainage basin of South Carolina, USA (Fig. 1). This part of the country has experienced a varied land-use history for the past 300+ years. In the late 1600s, an area of swampland and forest encompassing approximately 40,000 ha was cleared to make way for rice cultivation (Doyle et al. 2007; Edelson 2007). By 1705, the foundation of the Carolina low country economy was rice cultivation (Otto 1987; U.S. Fish and Wildlife Service 2011). Between 1700 and 1720, the first English settlements were established near Georgetown, South Carolina (U.S. Fish and Wildlife 2011). Between 1792 and the 1880s there were 10 rice plantations in operation on Sandy Island, near Georgetown, and, up until the 1940s, Sandy Island rice cultivation continued to be of major economic importance (U.S. Fish and Wildlife Service 2011). In 1840, the district of Georgetown produced nearly half of the total rice crop in the United States (Rogers 1970). After the Civil War, the loss of slave labor caused the South Carolina rice culture to decline (Coclanis 1985; Coclanis and Marlow 1998). A string of hurricanes between 1893 and 1911 damaged much of the infrastructure of the rice fields and devastated the rice economy (U.S. Fish and Wildlife 2011). By 1910 commercial rice production had ended in South Carolina, and by the mid- 20th century, many of the area's plantations were in disrepair (Coclanis 1985; U.S. Fish and Wildlife Service 2011). After being abandoned as rice fields, many plantations slowly returned to tidal marsh. In 1997, the Waccamaw National Wildlife Refuge, which encompasses an area of 21,853 ha, was established (U.S. Fish and Wildlife Service 2011).

Management Regimes

The study sites are freshwater marshes situated along Thoroughfare Creek and the Pee Dee River (Fig. 1). In this part of the Winyah Bay drainage basin, salinities are less than 0.5 ppt (Bennett et al. 1989). The sites included in this study are under three different management regimes or "treatments". Two sites, Sandy Island Tidal (29.3 ha) and Coastal Education Foundation (Coastal EDU) (55.0 ha) are naturally tidal freshwater marshes with a mean tide of approximately 1 m (Conrads and Roehl 2007).

Two sites, Wardlaw I (26.7 ha) and Hasty Point (18.3 ha), are subject to moist soil management, which is designed to drain marshes during the growing season (typically April–October), while maintaining adequate soil moisture to provide an abundance and diversity of seeds, aquatic invertebrates, and other foods for waterfowl and other wildlife (Smith et al. 1995; Strader and Simpson 2005). In this type of management, sites are flooded during the natural flooding cycle of the adjacent river system and, initially, water is often exchanged daily in order to circulate water inside the impoundments. In the Winyah Bay drainage basin of South Carolina, natural flooding usually occurs in late fall. Suspended sediment concentrations typically increase during this time with peak concentrations occurring during the period of highest flooding in late winter and early spring (Conrads and Roehl 2007). The Wardlaw I site has been under moist soil management, largely uninterrupted, since the mid 1960's. The Hasty Point site has been under moist soil management since the early 1970's. These sites have all been routinely disked, burned, and been subject to herbicide application to reduce the density of vegetation and facilitate planting of annual plant species.

The last two sites, Sandy Island Managed (12.5 ha) and Bird Field (6.19 ha), are both under the deeply flooded treatment (~90 cm) in which drainage occurs for a short time each spring, usually to manage vegetation communities.

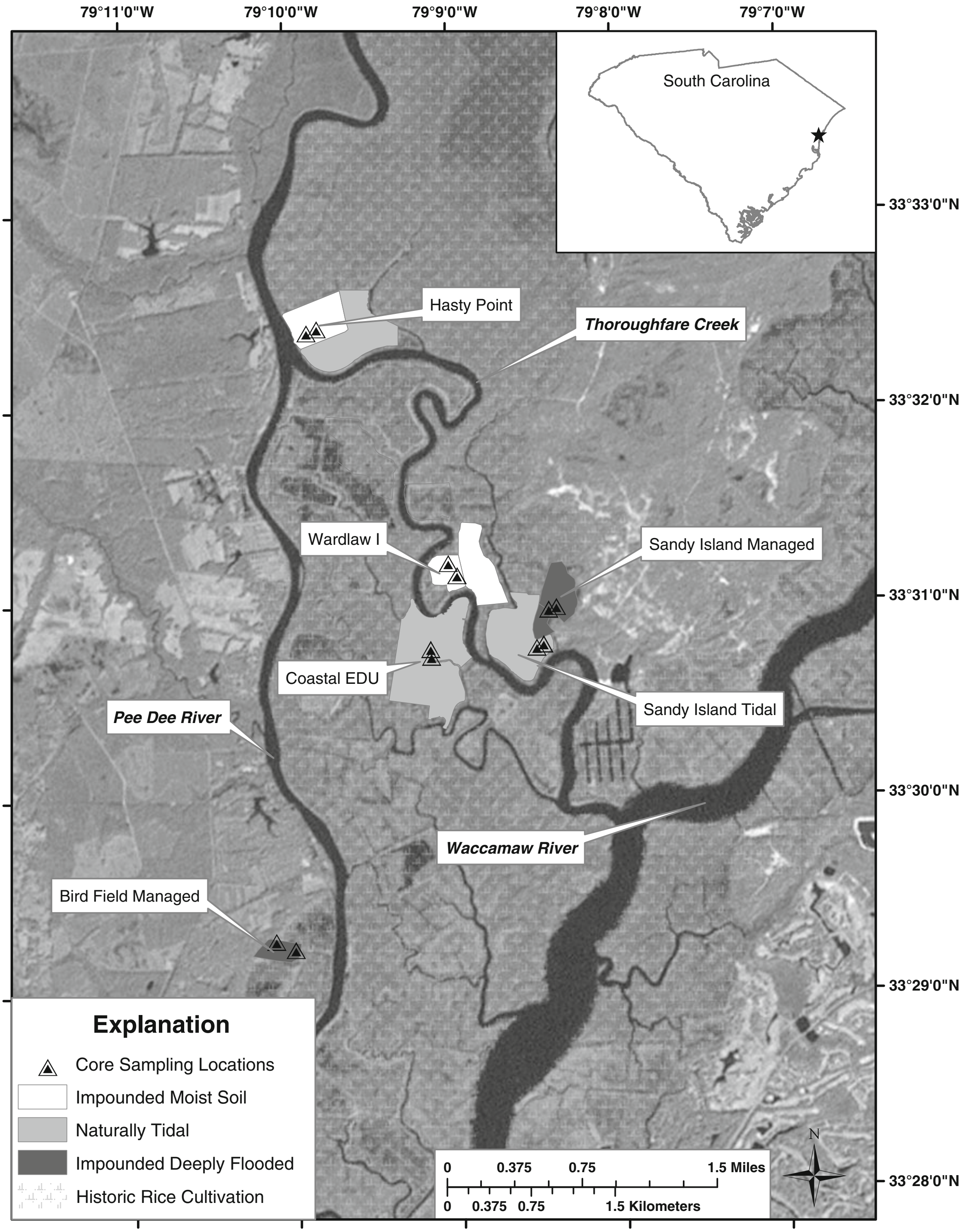

Fig. 1 The location of the study sites along the Lower Waccamaw River near Georgetown, South Carolina, USA. The extent of prior rice cultivation was estimated by identifying historic levees, irrigation and drainage ditches, and differences in vegetation color on aerial images

Similar to moist soil sites, these marshes also provide cover and invertebrate food for breeding waterfowl and typically do not require the on-going management besides drainage that moist soil sites receive. Sandy Island Managed has been under this treatment regime since 2002, while Bird Field has been managed this way since 2000. Bird Field has water control structures designed to receive water from nearby creeks during flooding and high tide. These sites have not received management in the form of disking, burning, or herbicide application since under deeply flooded management, however, they likely received such management before being deeply flooded.

Plant Communities

Freshwater marshes in the WNWR and the Lower Waccamaw River contain a highly diverse assemblage of plant species. Plant species common to all of the marshes in this study include cattail (*Typha latifolia* L.), giant cut grass (*Zizaniopsis miliacea* (Michx.) Döll & Asch.), soft-stemmed bulrush (*Schoenoplectus tabernaemontani* (C.C. Gmel.) Palla), various smartweeds (*Polygonum* spp.) and bulltongue (*Sagittaria lancifolia* L.). Tree species found within these tidal wetlands include baldcypress (*Taxodium distichum* (L.) L.C. Rich.), swamp tupelo (*Nyssa biflora* Walt.), and water tupelo (*Nyssa aquatica* L.). In addition, managed marshes often contain submerged aquatic vegetation (e.g., *Ceratophyllum demersum* L.), which is important for waterfowl forage. The refuge has the northern most nesting area for swallow tailed kites (*Elanoides forficatus*) and provides habitat for large populations of wading birds, wintering waterfowl, and neotropical migratory songbirds.

Methods

Soil Core Collection

Two soil cores were collected in each marsh site during May 3–5, 2010. A strong effort was made to collect cores within similar vegetation communities. In the deeply flooded and moist soil treatments, coring was conducted in areas dominated by giant cutgrass. In the naturally tidal marshes, coring was done in areas dominated by *Schoenoplectus* spp. Cores were collected with a Hargis corer, a razor-edged piston corer (15-cm diameter) that minimizes soil compaction (Hargis and Twilley 1994). After collection, cores were sealed air-tight on both ends within their acrylic collection tubes, laid horizontally on ice for transport, and stored under refrigeration until being shipped overnight to USGS laboratories in Sacramento, California, USA for further processing.

Soil Core Analysis

In the laboratory, soil cores were sectioned into 2 or 3 cm sections, weighed wet, dried at 80 °C, and weighed again. Dry bulk density was determined using the dry weight of each section of soil and the volume of the core section. Samples were then ground to pass through a 2 mm sieve. Total % carbon and % organic carbon were determined using a Perkin Elmer CHNS/O elemental analyzer (Perkin Elmer Corporation, Waltham, Massachusetts, USA) according to a modified version of U.S. Environmental Protection Agency Method 440.0 (Zimmerman et al. 2007). Samples were combusted at 925 °C. For % organic carbon analyses, samples were first exposed to concentrated hydrochloric acid (HCl) fumes in a desiccator for 24 h to remove inorganic carbon. The instrument was calibrated with blanks and acetanilide standards before use. Blanks, replicates, and standards were analyzed every 10 samples to assess instrument stability. Replicate samples were re-analyzed if the relative percentage difference between the two replicates was greater than 20 %. Method detection limit for carbon was 0.01 %.

Subsections of the cores were analyzed at the U.S. Geological Survey in Menlo Park, California, for ^{210}Pb, ^{226}Ra and ^{137}Cs to assign estimated dates to core profiles. Both ^{137}Cs and ^{210}Pb have been used successfully for dating lake sediments and wetland soils (Armentano and Woodwell 1975; Lynch et al. 1989; Appleby et al. 1997). ^{137}Cs, which does not occur naturally, is deposited on the land surface as fallout from nuclear weapons testing and power plant accidents. The first elevated levels of ^{137}Cs appeared in the atmosphere in the early 1950s, with peak concentrations occurring at the height of atmospheric nuclear testing in 1963. Peak activity of ^{137}Cs can usually be clearly identified in a soil profile. After the peak is found, subsequent dates can be determined by assuming a constant rate of soil accretion since that time. ^{210}Pb is a natural isotope of Pb with a half-life of 22.3 years. The total ^{210}Pb pool in soil consists of two parts: (1) a supported ^{210}Pb component produced within the soil via radioactive decay of ^{222}Rn that never diffused to the atmosphere and (2) an unsupported or excess ^{210}Pb component derived from ^{222}Rn that first diffused into the atmosphere and subsequently decayed to ^{210}Pb. The decay of the excess ^{210}Pb with depth is used to date sediments up to approximately 100 years old (Armentano and Woodwell 1975; Bricker-Urso et al. 1989; Thomas and Ridd 2004).

Activities of total ^{210}Pb, ^{226}Ra, and ^{137}Cs were measured simultaneously by gamma spectrometry as described in Baskaran and Naidu (1995), Fuller et al. (1999), and Van Metre et al. (2004). Subsamples of dried sediment samples were counted using a high-resolution intrinsic germanium well detector gamma spectrometer. Samples are placed in the detector bore-hole or well which provides near 4π counting geometry. Sediment samples were sealed in 7-mL polyethylene scintillation vials. The supported ^{210}Pb activity, defined by the ^{226}Ra activity, was determined on each interval from the 352 KeV and 609 KeV gamma emission lines of the short-lived daughters ^{214}Pb and ^{214}Bi daughters of ^{226}Ra, respectively. Self-absorption of the ^{210}Pb 46 KeV gamma emission line was accounted using an attenuation factor calculated from an empirical relationship between self-absorption and bulk density developed for this geometry based on the method of Cutshall et al. (1983). Self-absorption of the ^{214}Pb, ^{214}Bi, and the 661.5 KeV ^{137}Cs gamma emission lines was negligible for the well detector. Detector efficiency was determined from National Institute of Standards and Technology (NIST) traceable standards. NIST and International Atomic Energy Agency reference materials were counted monthly to check detector calibration. The reported uncertainty in the measured activity was calculated from the random counting error of samples and background spectra at the one standard deviation level was typically within ±10 %. The measured activities of replicate analysis of material from the same sample agreed to within ±15 %. Measured activities of ^{137}Cs were decay corrected for the period between sample collection and analyses.

The age-depth relationship of each core was estimated using the constant rate of supply model (CRS) (Appleby and Oldfield 1978; Robbins 1978; Appleby and Oldfield 1983) and uncertainty analysis was conducted following Van Metre and Fuller (2009). The CRS method assumes a constant supply of ^{210}Pb to the accreting sediment or soil surface and allows for accurate dating despite temporal variations in sedimentation rates. The method depends on measurement of the entire core profile of unsupported ^{210}Pb. We did not use the ^{137}Cs peak for dating, *per se*, but only to verify the general position of the 1963 horizon determined with the CRS (^{210}Pb) method. We followed this approach because we were interested in assessing temporal variability in vertical accretion and carbon sequestration rates, which is possible with the CRS method but not the ^{137}Cs method. The dating of a core was deemed successful only if (1) the ^{137}Cs profile had a clearly identifiable peak (assumed to represent the peak of atmospheric bomb testing in 1963) that spread beyond no more than two dated sections, (2) the 1963-dated sections from both the ^{137}Cs and CRS (^{210}Pb) methods were within the same section or adjacent sections and/or the bottom, middle, or top date of the section containing 1963 via the ^{137}Cs method was within 10 years of the bottom, middle or top of the section dated via the CRS (^{210}Pb) method, and (3) the entire unsupported ^{210}Pb profile was available for the core, showing a general decline with depth down to a value of <1 dpm/g. We chose a cutoff date of approximately 1970 for comparing core characteristics, vertical accretion, and carbon sequestration rates for two reasons: (1) the period from 1970 to 2010 was covered by each soil core in the moist soil and naturally tidal treatments, and (2) all the soil intervals dated during this time had acceptable levels of uncertainty (ESM, Table S1). For the sections that contained the 1963 date, the top, middle, or bottom had a mean date of 1962.6 with a standard deviation of 2.1 years.

The age-depth profiles for each core in the moist soil and naturally tidal treatments in conjunction with core characteristics were used to calculate the various measures of soil formation. Mass accumulation rates were determined by dividing the total cumulative mass of core sections within the study period (the closest top, bottom, or mid-section to 1970 up to 2010) by the total number of years in the study period (cumulative $mass_{study\ period}$/study period in years (units of g m^{-2} yr^{-1})). Inorganic sedimentation was estimated similarly except that the top term in the equation, inorganic cumulative mass, was estimated for each section in the study period by subtracting twice the organic carbon content from the total mass (relying on the fact that % C_{org}≈% OM/2; Mitsch and Gosselink (2000), p. 157) and multiplying this term by the total cumulative mass. Organic accumulation was estimated by replacing the top term by organic cumulative mass, or twice the organic carbon content multiplied by the total cumulative mass of each section in the study period. Because organic carbon content and vertical accretion rates were specifically determined for each core section, we calculated these rates by first determining the time-weighted arithmetic mean with the formula, $X_W=\sum wx/\sum w$, where X_w=weighted arithmetic mean, x=% organic carbon content or vertical accretion within a section, and w=time period covered per section for all sections between 1970 and 2010. Carbon sequestration rates were then determined similarly to the above calculations ((cumulative $mass_{1970–2010}$*weighted arithmetic mean of % organic carbon)/study period).

Because the deeply flooded sites were only flooded in 2000–2002, they were excluded from the long-term (40 year) comparisons of soil formation processes between the moist soil and naturally tidal sites. This is because, over a 40 year period, various natural processes such as decomposition and compaction act to alter soil structure and composition, making direct comparison to much younger soils inappropriate. Nevertheless, carbon sequestration rates were calculated for deeply flooded sites in order to provide data on initial rates for this management regime.

Results

Cores ranged from 16 cm to 84 cm in length. Soil density determined the depth to which the Hargis soil corer could be pushed into the substrate to extract a core. Basic soil core characteristics, activity data for the ^{210}Pb and ^{137}Cs analyses, and resulting dates for sections of each core are provided in the ESM, Table S1. The majority of cores collected met the dating criteria described in the methods above, signifying that cores were collected from sites with intact soil stratigraphy. Only two cores, Wardlaw I Core 2 and Bird Field Core 1, did not meet the critiera, and thus, were excluded from the study. Several factors may have affected the soil stratigraphy in these two cores including disturbance from animal burrows and/or impacts from management activities on the sites.

Because the deeply flooded treatment was initiated in 2000–2002, only the top layers of each core encompassed the period from 2000 to 2010 when all three treatments were in effect. These top layers contained higher % organic carbon (mean of top layers) in the deeply flooded (35.3 %, sd=4.60) and naturally tidal (35.5 %, sd=1.25) treatments than the moist soil treatment (27.3 %, sd=9.29). However, these differences were not significant (p>0.05; one-way ANOVA) due to high within treatment variability. Bulk density (mean of top layers) at the sites during this time period was highest in the moist soil (0.13 g cm^{-3}, sd=0.04) followed by the naturally tidal (0.08 g cm^{-3}, sd=0.01) and the deeply flooded treatments (0.05 g cm^{-3}, sd=0.05). A significant difference in bulk density was only found between the moist soil and deeply flooded treatments (Bonferroni pairwise comparisons based on one-way ANOVA, $p<0.05$).

Carbon sequestration rates *over each of the treatment periods* generally ranged from 50 to 200 g C m^{-2} yr^{-1} for the moist soil treatment, 100–435 g C m^{-2} yr^{-1} for the naturally tidal treatment, and 100–250 g C m^{-2} yr^{-1} for the deeply flooded treatment (with one outlier at >400 g C m^{-2} yr^{-1}; Fig. 2). At each of the sites, carbon storage was greatest at the surface and declined with time, likely as a result of decomposition.

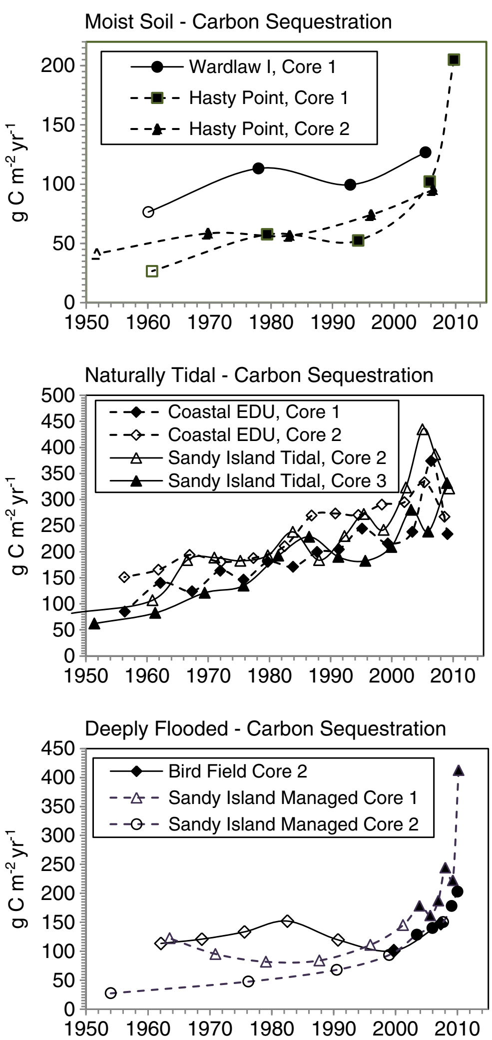

Fig. 2 Carbon sequestration (net carbon storage) in g C m^{-2} yr^{-1} at each site organized by management regime. For moist soil and deeply flooded sites, filled markers represent the time period during which the site was under each management regime. Different y-axis ranges are used to better demonstrate the range of carbon sequestration within each management regime

Because the deeply flooded treatment has only been in effect since 2000–2002, it is premature to discern the overall impact of this treatment on the soil processes of the deeply flooded sites. For this reason, the remainder of the results will be focused on a 40-year comparison between the moist soil and naturally tidal treatments. During the period from 1970 to 2010, when both the moist soil treatments and naturally tidal treatments were in effect, the moist soil sites had greater variability in and slightly less than half the mean organic carbon content of the naturally tidal sites (17 % vs. 32 %; Student's t-test, $p<0.005$; Table 1). Mean bulk density of the moist soil sites was also more variable and more than three times the mean bulk density of the naturally tidal sites (0.33 vs. 0.09 g cm^{-3}; Student's t-test, $p<0.05$; Table 1). The mass accumulation rates in the moist soil and naturally tidal sites were all quite similar (~600–800 g m^{-2} yr^{-1}) during the period from 1970 to 2010. The other measures of soil formation, however, differed significantly between the two treatments (Fig. 3). Inorganic sedimentation rates were significantly greater (Student's t-tests, $p<0.005$) in the moist soil treatment (mean=485 g m^{-2} yr^{-1}) than the naturally tidal treatment (mean=251 g m^{-2} yr^{-1}), and carbon sequestration (and organic accumulation) rates were greater (Student's t-test, $p<0.005$) in the naturally tidal (mean C sequestration=219 g C m^{-2} yr^{-1}) than the moist soil treatment (mean C sequestration=91 g C m^{-2} yr^{-1}; Fig. 3).

The mean vertical accretion rate in the naturally tidal treatment was four times greater than that of the moist soil treatment (0.84 vs. 0.21 cm yr^{-1}, respectively) over the period from 1970 to 2010 (Student's t-test, $p<0.005$; Fig. 4). The variability in vertical accretion rates was much greater in the naturally tidal sites (sd=0.12) than the moist soil sites (sd=0.01).

Discussion

The results of this study show that the naturally tidal, freshwater marshes have higher rates of carbon sequestration (mean=219 g C m^{-2} yr^{-1}) than impounded marshes under a moist soil management regime (mean=91 g C m^{-2} yr^{-1}; Fig. 3). The rates from this study for naturally tidal, freshwater marshes are within the same general range as those measured in the Florida Everglades (86–387 g C m^{-2} yr^{-1}, the higher values reflect an increasing P gradient; Reddy et al. 1993) and greater than the overall mean rate for tidal freshwater marshes in North America (~140 +/− 20 g C m^{-2} yr^{-1} overall for northeast, southeast, and Gulf coast; Craft 2007).

Over a 40 year period, soil formation processes in the moist soil treatment differed from the naturally tidal treatment in several respects. Besides the difference in carbon sequestration rates, vertical accretion in the naturally tidal

Table 1 Mean bulk density and % organic carbon content (sd) of soil sections dated from 1970 to 2010 for the moist soil and naturally tidal treatments and 2000–2010 for the deeply flooded treatment

Management regime	Site	Core	% Organic carbon	Bulk density (g cm^{-3})
Moist soil	Wardlaw Pond I	1	22	0.28
	Hasty point	1	17	0.29
	Hasty point	2	12	0.41
	Arithmetic mean		**17 (5.0)**	**0.33 (0.070)**
Naturally tidal	Sandy Island Tidal	2	32	0.09
	Sandy Island Tidal	3	33	0.09
	Coastal EDU Tidal	1	31	0.09
	Coastal EDU Tidal	2	33	0.08
	Arithmetic mean		**32 (0.9)**	**0.09 (0.005)**
Deeply flooded	Bird field	2	29	0.11
	Sandy Island managed	1	30	0.09
	Sandy Island managed	2	29	0.08
	Arithmetic mean		**29 (0.6)**	**0.09 (0.015)**

The significance of the bold text is to show that these are the means (average values)

sites was four times higher than in the moist soil sites (Fig. 4). Conversely, the inorganic sedimentation rates in the moist soil sites were significantly greater than at the naturally tidal sites (Fig. 3). Overall, however, the total mass accumulation rates in the two treatments were quite similar. This demonstrates that the naturally tidal sites accreted and stored organic matter (which typically occupies a much greater volume than an equal mass of inorganic matter; Drexler et al. 2009) at a much greater rate. In addition, the naturally tidal treatment had lower soil bulk density than the moist soil treatment, suggesting that compaction was greater and/or pore space smaller in the moist soil treatment. The higher inorganic sedimentation rates in the moist soil sites may be due to (1) flooding of the impoundments during late fall when suspended sediment concentrations typically rise

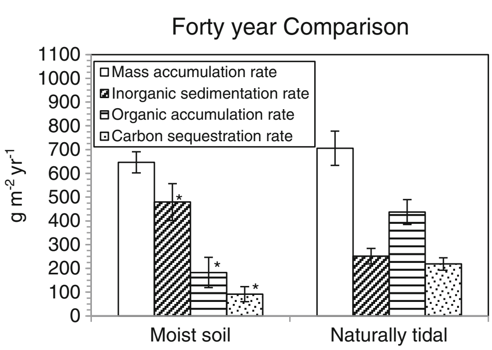

Fig. 3 Comparison of mean mass accumulation, inorganic sedimentation, organic accumulation, and carbon sequestration rates (units in g m^{-2} yr^{-1} for all except carbon sequestration, which is in units of g C m^{-2} yr^{-1}) between moist soil and naturally tidal treatments for the period from 1970 to 2010. No symbol on top of bar for moist soil represents no significant difference between treatments and "*" signifies difference at $p<0.005$ (Student's two-tailed t-tests). Deeply flooded sites were not included because the sites have only been under this treatment regime from 2000 or 2002 to 2010. A longer time period is needed for the ^{210}Pb dating to have adequate resolution to compare the carbon sequestration rates in the deeply flooded sites to the other treatments

and spike due to storm activity (Conrads and Roehl 2007), (2) a period of daily water exchange, which occurred early in the flooding process to circulate water within the impoundments, and/or (3) the long settling period following flooding, which served to retain most of the suspended sediment load.

In their study of impounded and naturally tidal marshes in coastal Louisiana, Bryant and Chabreck (1998) compared numerous soil parameters including vertical accretion rates and organic accumulation. Similar to this study, their work showed that impounded marshes had significantly lower vertical accretion and organic accumulation than naturally tidal marshes. However, in their study, the two marsh types did not differ with respect to bulk density or % organic matter, whereas, in this study, bulk density in the moist soil treatment was over three times that of the naturally tidal treatment and % organic carbon of the moist soil treatment was roughly half of the naturally tidal treatment (Table 1). Bryant and Chabreck (1998) attributed the lower vertical

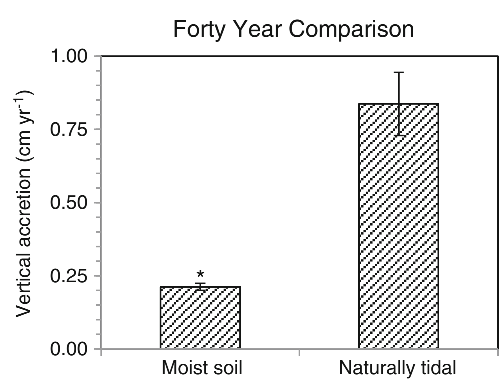

Fig. 4 Vertical accretion rates for the period from 1970 to 2010 for both the moist soil and naturally tidal treatments. The symbol "*" signifies that there is a difference between rates at $p<0.005$ (Student's two-tailed t-test). Deeply flooded sites were not included because the sites have only been under this treatment regime from 2000 or 2002 to 2010

accretion rates in impounded marshes to hydrologic isolation from tidal sediment subsidies and microbial oxidation of soils during hydrologic drawdown periods. In this study, tidal subsidies do not appear to be a causal factor in the difference between treatments, because the naturally tidal treatment had lower inorganic sedimentation rates than the moist soil treatment (Fig. 3). The second causal factor noted by Bryant and Chabreck (1998), hydrologic drawdown, is very likely the chief cause of the difference between carbon sequestration rates in the moist soil vs. the naturally tidal treatments. The simple act of drainage, even for a short period, converts marsh soils from anaerobic to aerobic status, thus initiating microbial oxidation of the organic carbon in the litter and top layers of the soil. The fact that drawdown occurs during warm periods (late spring through summer for moist soil sites) further stimulates the process of microbial decomposition. Any additional management activities such as disking and herbicide application, which are used to reduce macrophyte cover, can further impact organic matter accumulation. By breaking up the soil matrix, disking leads to greater oxidation and loss of organic matter content (Larney et al. 1997; Saggar et al. 2001) and the large machinery required to carry out this work can simultaneously cause soil compaction. Herbicides act to constrain growth to annual plant species, which do not create intricate root mats or lots of surface litter, further limiting organic matter accumulation in the soil. In their study, Bryant and Chabreck (1998) found the only exception to low organic accumulation and vertical accretion in impounded marshes was a permanently flooded freshwater marsh (i.e., no drawdown period). Although the deeply flooded treatment in this study has only been running for 8–10 years, it appears that this treatment may have the potential for greater organic matter accumulation and higher vertical accretion due to the high organic carbon content in the top layers and lower bulk density than the moist soil treatment (see Table 1).

Although hydrologic drawdown (and accompanying microbial oxidation of organic carbon) seems to be likeliest reason for the differences between carbon sequestration rates in the treatments, there are also two other possible causes that should be considered. Differences in local environmental conditions and/or dominant plant species have the potential to influence soil properties. However, with regard to local environmental conditions, we deliberately chose tidal freshwater marshes and impounded freshwater marshes in close proximity to each other to minimize variability in environmental factors such as salinity, climate, and hydroperiod. Therefore, although some local-scale variability is inevitable, it seems unlikely that these factors could explain the *major* differences between treatments. With respect to plant communities, some differences in dominant species between the treatments may possibly be responsible for differences in soil formation processes. The coring sites in the moist soil treatment were dominated by *Zizaniopsis miliacea* and the naturally tidal treatments were dominated by *Schoeonoplectus* spp. Although these two species are both large emergent macrophytes, their above- and/or belowground structures may differ in carbon content and/or biomass. However, the magnitude of difference between the two treatments is so large with respect to carbon sequestration and vertical accretion, that these species differences alone seem unlikely to be able to account for such a discrepancy.

Management Implications

The moist soil treatment used in this study is employed widely in National Wildlife Refuges in the southeast as well as other areas of the United States. There are approximately 65,770 ha of freshwater wetlands under moist soil management in the National Wildlife Refuge system and 14,240 ha in the Southeast Region (U.S. Fish and Wildlife Service 2012). Roughly half of the refuges in the Southeast Region are found along the coast.

The overall result of this study is that carbon sequestration rates are significantly greater in naturally tidal, freshwater marshes than in impounded marshes subject to moist soil management. Managers interested in maximizing carbon sequestration should consider reducing the drawdown time during the growing season to limit the amount of aerobic decomposition of accumulated organic material. Additional research, which quantifies the relationships between drawdown periods and carbon sequestration rates as well as the impact of differing levels of drawdown on habitat quality, is needed to assist managers in effectively tweaking their systems to gain maximum carbon storage while preserving habitat value.

The practice of deeply flooding marshes is not as common in National Wildlife Refuges (such statistics are not maintained by the U.S. Fish and Wildlife Service) as moist soil management. Deeply flooded marshes may possibly be an option for increasing carbon sequestration rates (Bryant and Chabreck 1998; Miller and Fujii 2010, 2011), however, the 8–10 years the treatment was in effect in this study is not long enough to determine the merits of this management regime. A longer time period is needed for the ^{210}Pb dating to have adequate resolution to compare the carbon sequestration rates in the deeply flooded sites to the other treatments.

Although this study demonstrates major differences between naturally tidal and moist soil managed sites with respect to carbon sequestration rates, it does not provide a full carbon balance for the treatments. Because freshwater marshes are capable of high rates of CH_4 emissions, which can reduce or even negate carbon sequestration gains due to methane's much greater global warming potential (25x) than carbon dioxide (Kayranli et al. 2010; Poffenbarger et al. 2011), it is possible that the radiative forcing of these freshwater wetlands may outweigh the amount of carbon that is stored over the same time. Currently, there are few studies quantifying methane fluxes from tidal freshwater wetlands, and in the studies available, methane fluxes have been variable, likely reflecting site-specific conditions (Poffenbarger et al. 2011). Working in a tidal freshwater marsh along a tributary of the Waccamaw River (in the same region as our sites), Neubauer (2013) determined that annual net ecosystem production was positive (295 g C m^2 for the year 2009), indicating that the marshes fixed more carbon through production than was lost by gaseous carbon emissions. Similar results were found in impounded freshwater marshes in the Sacramento-San Joaquin Delta of California, demonstrating that net carbon balance in a marsh can result in more carbon stored than lost if there are high carbon inputs that offset gaseous carbon losses (Miller and Fujii 2010, 2011). Such high carbon inputs can be achieved as a result of high plant productivity, a long growing season, and reduced rates of decomposition due to sustained anaerobic conditions (Miller 2011). Furthermore, new work shows that several types of tropical and temperate freshwater marshes have CO_2 sequestration:CH_4 emission ratios greater than the requisite 25 required to counterbalance the greenhouse gas strength of methane over carbon dioxide, and that these ratios increase substantially over time periods of 100 years or more (Mitsch et al. 2012).

Clearly, more research is needed to better quantify the balance between carbon emissions and carbon storage in tidal as well as impounded marshes at the fresh end of the salinity scale. The only way to definitively determine the difference in carbon storage between the tidal and impounded freshwater marshes in this study is to conduct a full carbon balance, preferably with continuous eddy covariance measurements of carbon fluxes. In the meantime, however, it is worthwhile to consider employing *any* practice that increases carbon storage, particularly in National Wildlife Refuges, which are protected habitats that are managed on a long-term basis for a range of goals including ecosystem services.

This study demonstrates that there is great potential to increase carbon sequestration rates in impounded freshwater marshes within the extensive National Wildlife Refuge system. This may present managers with an entry into the developing carbon reduction market while providing much-needed funds to better manage and conserve their wildlife populations into the future. However, any change in management has the potential to jeopardize other management goals, including adequate maintenance of food sources and cover for a variety of waterfowl and other wildlife. Therefore, further research is needed to better characterize the full carbon balance in a broader range of impounded and natural marshes so that managers have the full suite of information needed to make the best decisions in light of their particular management needs.

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References

Appleby PG, Oldfield FR (1978) The calculation of ^{210}Pb dates assuming a constant rate of supply of unsupported ^{210}Pb to the sediment. Catena (Supplement) 5:1–8

Appleby PG, Oldfield FR (1983) The assessment of ^{210}Pb data from sites with varying sediment accumulation rates. Hydrobiologia 103:29–35

Appleby PG, Shotyk W, Fankhauser A (1997) Lead-210 age dating of three peat cores in the Jura Mountains, Switzerland. Water, Air, and Soil Pollution 100:223–231

Armentano TM, Woodwell GM (1975) Sedimentation rates in a Long Island marsh determined by ^{210}Pb dating. Limnology and Oceanography 20:452–456

Baskaran M, Naidu AS (1995) ^{210}Pb-derived chronology and the fluxes of ^{210}Pb and ^{137}Cs isotopes into continental shelf sediments, East Chukchi Sea, Alaskan Arctic. Geochimica et Cosmochimica Acta 59:4435–4448

Bennett CS, Cooney TW, Jones KH, Church BW, Murray GL (1989) Water resources data South Carolina 1990, vol 1. U.S. Geological Survey Water –Data Report SC-89

Bricker-Urso S, Nixon SW, Cochran JK, Hirschberg DJ, Hunt C (1989) Accretion rates and sediment accumulation in Rhodes Island salt marshes. Estuaries 12:300–317

Bridgham SD, Megonigal JP, Keller JP, Bliss NB, Trettin C (2006) The carbon balance of North American wetlands. Wetlands 26:889–916

Bryant JC, Chabreck RH (1998) Effects of impoundment on vertical accretion of coastal marsh. Estuaries 21:416–422

Coclanis PA (1985) The rise and fall of the South Carolina Low Country: an essay in economic interpretation. Southern Studies 24:143–166

Coclanis PA, Marlow JC (1998) Inland rice production in the South Atlantic States: a picture in black and white. Agricultural History 72:197–212

Conrads PA, Roehl EA, Jr. (2007) Analysis of salinity intrusion in the Waccamaw River and Atlantic Intracoastal Waterway near Myrtle Beach, South Carolina, 1995–2002: U.S. Geological Survey Scientific Investigations Report 2007–5110, 41 p. Available via http://pubs.water.usgs.gov/sir2007-5110

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

Cutshall NH, Larsen IL, Olsen CR (1983) Direct analysis of ^{210}Pb ins sediment samples: self-absorption corrections. Nuclear Instruments and Methods 306:309–312

Doyle TW, O'Neil CP, Melder MPV, From AS, Palta MM (2007) Chapter 1 – Tidal freshwater swamps of the southeastern United States: effects of land use, hurricanes, sea-level rise, and climate change. In: Conner WH, Doyle TW, Krauss KW (eds) Ecology of tidal freshwater forested wetlands of the southeastern United States. Springer, Dordrecht, pp 1–28

Drexler JZ, de Fontaine CS, Brown TA (2009) Peat accretion histories during the past 6,000 years in marshes in the Sacramento–San Joaquin Delta, California, USA. Estuaries and Coasts 32:871–892

Edelson SM (2007) Clearing swamps, harvesting forests: trees and the making of a plantation landscape in the colonial South Carolina Low Country. Agricultural History 81:381–406

Fuller CC, van Geen A, Baskaran M, Anima R (1999) Sediment chronology in San Francisco Bay defined by ^{210}Pb, ^{234}Th, ^{137}Cs, and $^{239,\ 240}$Pu. Marine Chemistry 64:7–27

Gresham CA, Hook DD (1982) Rice fields of South Carolina: a resource inventory and management policy evaluation. Coastal Zone Management Journal 9:183–203

Hargis TG, Twilley RR (1994) Improved coring device for measuring soil bulk density in a Louisiana deltaic marsh. Journal of Sedimentary Research Section A: Sedimentary Petrology and Processes 64:681–683

Kayranli B, Scholz M, Mustafa A, Hedmark Å (2010) Carbon storage and fluxes within freshwater wetlands: a critical review. Wetlands 30:111–124

Larney FJ, Bremmer E, Janzen HH, Johnston A, Linkwall CW (1997) Changes in total, mineralizable and light fraction soil organic soil matter with cropping and tillage intensities in semiarid southern Alberta, Canada. Soil and Tillage Research 42:229–240

Lynch JC, Meriwether JR, McKee BA, Vera-Herrera F, Twilley RR (1989) Recent accretion in mangrove ecosystems of ^{137}Cs and ^{210}Pb. Estuaries 12:284–299

Miller RL (2011) Carbon gas fluxes in re-established wetlands on organic soils differ relative to plant community and hydrology. Wetlands 31:1055–1066

Miller RL, Fujii R (2010) Plant community, primary productivity, and environmental conditions following wetland re-establishment in the Sacramento-San Joaquin Delta, California. Wetlands Ecology and Management 18:1–6

Miller RL, Fujii R (2011) Chapter 1 – Re-establishing marshes can turn a current carbon source into a carbon sink in the Sacramento-San Joaquin Delta of California. In: Contreras DA (ed) River deltas: types, structures, and ecology. Nova Science Publishers, Inc., Hauppauge, pp 1–34

Mitsch WJ, Gosselink JG (2000) Wetlands, 3rd edn. John Wiley and Sons, New York

Mitsch WJ, Bernal B, Nahlik AM, Mander Ü, Zhang L, Anderson CJ, Jørgensen SE, Brix H (2012) Wetlands, carbon, and climate change. Landscape Ecology. doi:10.1007/s10980-9758-8

Neubauer SC (2013) Ecosystem responses of a tidal freshwater marsh experiencing saltwater intrusion and altered hydrology. Estuaries and Coasts 36:491–507

Otto JS (1987) Livestock-raising in early South Carolina, 1670–1700: prelude to the rice plantation economy. Agricultural History 61:13–24

Poffenbarger HB, Needleman B, Megonigal PJ (2011) Salinity influence on methane emissions from tidal marshes. Wetlands 31:831–842

Reddy KR, DeBusk WF, DeLaune RD, Koch MS (1993) Long-term nutrient accumulation rates in the Everglades. Soil Science Society of America Journal 57:1147–1155

Robbins JA (1978) Geochemical and geophysical applications of radioactive lead. In: Nriagu JO (ed) Biogeochemistry of lead in the environment. Elsevier Scientific, Amsterdam, pp 285–393

Rogers GC Jr (1970) The history of Georgetown County, South Carolina. University of South Carolina Press, Columbia

Saggar RN, Yeates GW, Shepherd TG (2001) Cultivation effects on soil biological properties, microfauna, and organic matter dynamics in eutric gleysol and gleyic luvisol soils in New Zealand. Soil and Tillage Research 58:55–68

Smith WD, Rollins GL, Shinn R (1995) A guide to wetland habitat management in the Central Valley. California Department of Fish and Game, Sacramento, CA. Available via http://www.dfg.ca.gov/lands/waterfowl/docs/WetlandinCentralValley.pdf. Accessed 11 Sept 2012

Strader RW, Simpson PH (2005) Moist-soil management guidelines for the U.S. Fish and Wildlife Service Southeast Region, Migratory Bird Field Office Division of Migratory Birds Southeast Region U.S. Fish and Wildlife Service, Jackson, Mississippi. Available via http://www.fws.gov/columbiawildlife/MoistSoilReport.pdf. Accessed 11 Sept 2012

Thomas S, Ridd PV (2004) Review of methods to measure short time scale sediment accumulation. Marine Geology 207:95–114

Trulio L, Callaway J, Crooks S (2007) White paper on carbon sequestration and tidal salt marsh restoration. Prepared for the South Bay Salt Pond Project. Available via http://www.southbayrestoration.org/climate/. Accessed 14 Sept. 2012

U.S. Fish and Wildlife Service (2011) Waccamaw National Wildlife Refuge website. Available via http://www.fws.gov/waccamaw/History1.html. Accessed 26 June 2012

U.S. Fish and Wildlife Service (2012) U.S. Fish and Wildlife Service 2012 refuge annual performance plan data, Available via https://rmis.fws.gov/RAPP. Accessed 22 Sept 2012

Van Metre PC, Fuller CC (2009) Dual-core mass-balance approach for evaluating mercury and ^{210}Pb atmospheric fallout and focusing to lakes. Environmental Science and Technology 43:26–32. doi:10.1021/es801490c

Van Metre PC, Wilson JT, Fuller CC, Callender E, Mahler BJ (2004) Collection, analysis, and age-dating of sediment cores from 56 US lakes and reservoirs sampled by the US Geological Survey, 1992–2001. U.S. Geological Survey Scientific Investigations Report 2004–5184

Zimmerman CF, Keefe CW, Bashe J (2007) Determination of carbon and nitrogen in sediments and particulates of estuarine/coastal waters using elemental analysis. U.S. Environmental Protection Agency Method 440.0, Available via http://www.caslab.com/EPA-Methods/PDF/EPA-Method-440.pdf. Accessed 14 Sept 2012