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Carbon Sequestration in Tidal Salt Marshes of the Northeast United States

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Abstract Tidal salt marshes provide important ecological services, habitat, disturbance regulation, water quality improvement, and biodiversity, as well as accumulation and sequestration of carbon dioxide (CO_2) in vegetation and soil organic matter. Different management practices may alter their capacity to provide these ecosystem services. We examined soil properties (bulk density, percent organic C, percent N), C and N pools, C sequestration and N accumulation at four marshes managed with open marsh water management (OMWM) and four marshes that were not at U.S. Fish and Wildlife National Wildlife Refuges (NWRs) on the East Coast of the United States. Soil properties (bulk density, percent organic C, percent N) exhibited no consistent differences among managed and non-OMWM marshes. Soil organic carbon pools (0–60-cm depth) also did not differ. Managed marshes contained 15.9 kg C/m^2 compared to 16.2 kg C/m^2 in non-OMWM marshes. Proportionately, more C (per unit volume) was stored in surface than in subsurface soils. The rate of C sequestration, based on ^{137}Cs and ^{210}Pb dating of soil cores, ranged from 41 to 152 g/m^2/year. Because of the low emissions of CH_4 from salt marshes relative to freshwater wetlands and the ability to sequester C in soil, protection and restoration of salt marshes can be a vital tool for delivering key ecosystem services, while at the same time, reducing the C footprint associated with managing these wetlands.

Keywords Salt marsh · Radiometric dating · Carbon sequestration · Management · US National Wildlife Refuge · Carbon trading

Introduction

Tidal salt marshes provide critical ecological services such as habitat for estuarine organisms, shoreline protection, organic carbon and nitrogen sequestration, and ecosystem stability through organic matter accumulation (Barbier et al. 2011). Tidal salt marshes also play an important role in the global carbon cycle by storing an estimated 4.8–87.2 Tg C per year (Mcleod et al. 2011). This ability to act as a carbon sink is due to high sedimentation rates, high soil carbon content, and burial of organic matter (Bridgham et al. 2006). Furthermore, tidal salt marshes are unique in that methane emissions are low relative to freshwater wetlands (Poffenbarger et al. 2011), which create an opportunity to reduce carbon footprints through habitat creation and preservation.

Tidal marsh formation and the associated ecological benefits, including carbon sequestration, are based on soil accretion and tidal patterns. These marshes perform self-maintenance by accreting soil at rates comparable to rates of sea level rise (SLR). This process involves sediment being brought in by tides and is aided by vegetation, which acts as a baffle to increase sediment deposition (Morris et al. 2002). Vegetation also contributes litter and roots that decompose and add organic matter to the soil.

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In spite of their benefits, many coastal wetlands have been modified and destroyed by human activities. Anthropogenic alterations to New England coastal marshes have occurred since the 1600s with the arrival of Europeans. Modifications resulted from cattle grazing, salt hay farming, diking for impoundments, and ditching for mosquito control (Sebold 1992). The height of parallel grid ditching occurred during the 1930s, with 95 % of salt marshes from Maine to Virginia being ditched (Bourn and Cottam 1950) and maintained for the next three decades (James-Pirri et al. 2008). Negative impacts, such as lowered water tables, resulted in transition from typical salt marsh grasses (*Spartina alterniflora*, *S. patens*, and *Distichlis spicata*) to vegetation better adapted to drier conditions (*Iva frutescens* and *Phragmites australis*), significantly altering habitat and support functions for wildlife (Diaber 1986; Wolfe 1996, 2005).

During the 1960s, a new mosquito control technique was developed in New Jersey called open marsh water management (OMWM; Ferrigno and Jobbins 1968). Through the creation and excavation of ponds and plugging of grid ditches, fish would have access to low marsh mosquito breeding areas to feed on the larvae. The restoration of more natural hydrology was also intended to provide additional habitat for fish and waterfowl (James-Pirri et al. 2008). OMWM can be implemented using open systems with connection to tidal channels, closed systems with no direct connection to tidal influences, or sill systems, where higher tides create a partial connection to ponds and ditches (James-Pirri et al. 2008). Methods of altering hydrology for OMWM vary by region but include ponds, radial ditches, sills to partially retain tidal waters, and ditch plugging to retain water in existing grid ditches. Currently, open and closed OMWM systems are both predominantly used in the Mid-Atlantic States, whereas ditch plugging is more common in the New England region (James-Pirri et al. 2008). These alterations return water to previously drained areas of the marsh, which have the potential to alter vegetation and decomposition patterns and thus carbon sequestration and storage.

We measured concentrations and accumulation of organic carbon and nitrogen in eight saline tidal marshes, four where OMWM was implemented and four where OMWM was not implemented, of the New England and Mid-Atlantic regions. Our goals were to determine if OMWM techniques altered the marshes' ability to store carbon and to characterize rates of C sequestration as a potential offset for carbon emissions from management of the wetlands. We also explore how this information may be relevant to natural resource management policy in the United States. Incorporation of ecosystem services into management of public lands has been a particular matter of interest in recent years (Ingraham and Foster 2008; Olander et al. 2012; Sutton-Grier et al. 2014).

Materials and Methods

Site Description

One tidal salt marsh managed with OMWM and one that was not were each selected from four U.S. National Wildlife Refuges (NWRs) along the NE coast (Fig. 1). Managed salt marshes incorporated various OMWM techniques, including radial ditches, ditch plugging, and pond creation. Non-OMWM salt marshes were chosen to closely match managed salt marshes in soil, vegetation, and marsh type but with minimal disturbance from human activities such as dredging, filling, ditching, or dominance by invasive species.

The four refuges were Rachel Carson (Wells, Maine), Parker River (Rowley, Massachusetts), Wertheim (Shirley, New York), and Forsythe (Oceanville, New Jersey) (Table 1). OMWM techniques varied among refuges. Rachel Carson primarily used ditch plugging; Parker River used some pond and radial ditch creation, as well as ditch plugging in a closed system; Wertheim used grid ditch plugging; and Forsythe used a closed OMWM system that incorporated ditch plugs, pond creation, and radial ditches. Non-OMWM sites had various levels of management, but did not incorporate OMWM. At Rachel Carson, there was no management at the non-OMWM site; at Parker River, ditches were present; at Wertheim, there was ditching dating back to the 1950s; and Forsythe was in a wilderness area that had historical grid ditching (1930s) that had been removed.

Vegetation consisted of a mixture of low marsh and high marsh species, such as smooth cordgrass (*Spartina alterniflora*) and salt meadowgrass (*Spartina patens*). Silverling (*Baccharis halimifolia*) and common reed (*Phragmites australis*) were found at Parker River and scattered throughout Wertheim. Soils were primarily histosols (Table 1) (NRCS 2012). Tide range varies between 0.25 and 2.8 m. Open marsh water management techniques at the sites were implemented between 1990 and 2004.

Soil Sampling and Analysis

In June 2012, ten soil cores were collected from an area of about 3 ha in each marsh, for a total of 80 cores. Sampling sites were randomly selected from the marsh plain, which represented about 70–80 % of the area of each system. Cores were 8.5 cm by 60-cm deep and sectioned in the field into 0–10, 10–30, and 30–60 cm increments for physical and chemical analysis. Soil increments were air dried, weighed for bulk density, then ground and sieved through a 2-mm mesh screen. Samples were analyzed for organic carbon (C) and Nitrogen (N). Bulk density was calculated from the dry weight per unit volume for each depth increment using a sub-sample that was dried at 105 °C to correct for moisture content (Blake and Hartage 1986). Organic C and N were measured using a

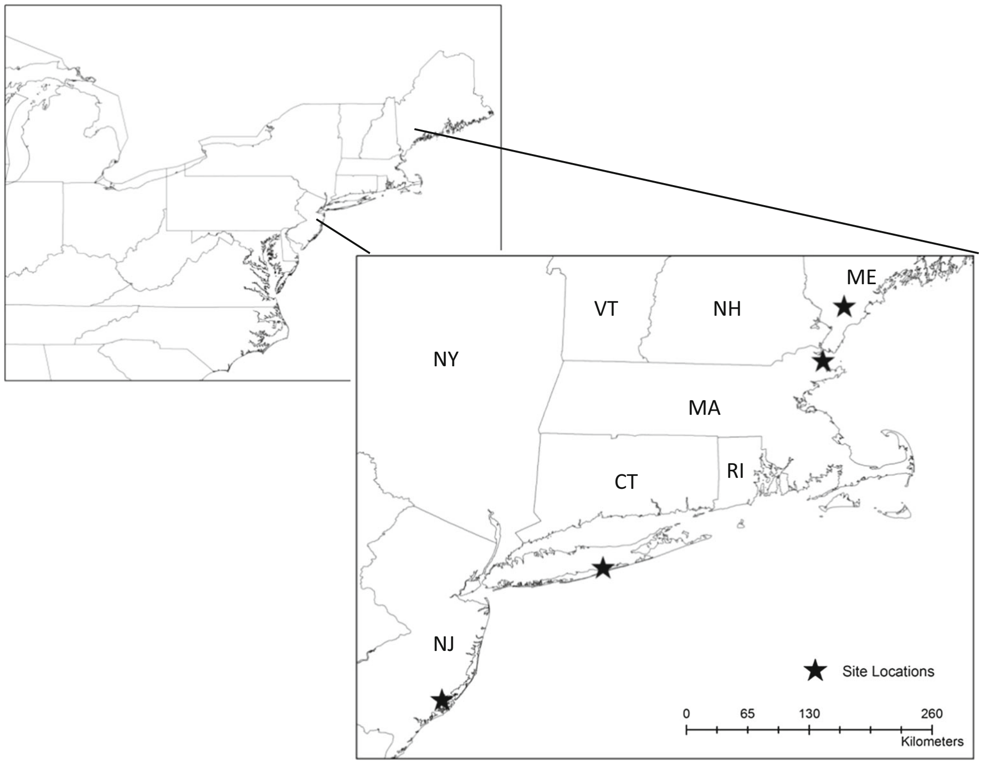

Fig. 1 Study area identifying the locations of each the NWRs within Maine, Massachusetts, New York, and New Jersey

Table 1 Plant community structure, soil taxonomy, tidal range, and initial year of open marsh water management of the four study marshes along the Northeast Coast

Site	Plant species	Soils[a]	Tide range	Mean salinity[b]	OMWM	Location	
						OMWM	Non-OMWM
RC	*Spartina alterniflora* *Spartina patens*	SU—Sulfihemists	2.8 m	28	2000	43°15.942N, 70°35.607W	43°19.022N, 70°34.306W
PR	*Baccharis halimifolia* *Distichlis spicata* *Phragmites australis* *Spartina alterniflora* *Spartina patens*	IW—Ipswich and Westbrook mucks (euic, mesic typic Sulfi-hemists)	2.6 m	20	1994	42°45.770N, 70°48.400W	42°46.941N, 70°48.628W
W	*Baccharis halimifolia* *Phragmites australis* *Spartina alterniflora* *Spartina patens*	Tm—Tidal Marsh (histosols)	0.25 m	12	2004	40°46.198N, 72°53.300W	40°46.193N, 72°53.211W
F	*Iva frutescens* *Spartina alterniflora* *Spartina patens*	TrkAv—Transquaking mucky peat (histosols)	1.2 m	18	1990	39°30.432N, 74°25.415W	39°32.129N, 74°25.609W

[a] Soil Conservation Service 1975, 1978, 1982, 1984

[b] Based on data collected by US FWS between 2001 and 2014

Perkin Elmer 2400 CHN analyzer (PerkinElmer Corp., Waltham, MA). Analysis of an internal lab salt marsh soil standard (mean ± SE 6.106 ± 0.08 % C, 0.365 ± 0.003 % N) yielded recovery rates of 107 % for C and 99–104 % for N.

An additional soil core was collected from each marsh and sectioned in the field into 2 cm increments for the top 30 and 5 cm increments for the lower 30 cm to measure rates of vertical accretion. Ground and sieved soils were packed into 50-mm diameter by 9-mm deep petri dishes, and analyzed for vertical accretion using ^{137}Cs and ^{210}Pb gamma analysis (Craft et al. 2003). ^{137}Cs was measured at the 661.62 keV photopeak, while ^{210}Pb was measured at the 46.5 keV photopeak. The rate of carbon sequestration for each core was determined using vertical accretion rates, bulk density, and organic C concentrations from the 0–10-cm depth of the bulk soils.

Statistical Analysis

Bulk soil properties (bulk density, percent organic C, and percent N) and pools of carbon and nitrogen (g/m^2) in managed and non-OMWM marshes were evaluated using a three-way analysis of variance (ANOVA) based on treatment (OMWM and non-OMWM), site, and sample depth. Normality was tested for using a Kolmogorov–Smirnov test and, when necessary, transformations were made to improve normality of distributions. Organic carbon was transformed using ln(organic carbon), and nitrogen was transformed using sqrt(nitrogen). Transformations did not improve normality for bulk density. Transformations were applied to data from all sites. Post-hoc comparison using the Ryan–Einot–Gabriel–Welsh multiple range test was also used to determine if means were significantly different from each other. All statistics were performed using SAS version 9.3 (SAS Institute Inc. 2011). All tests were made at $\alpha = 0.05$. Because of the limited sample size ($n = 8$ cores, one per marsh), we did not statistically compare the effects of marsh management on rates of C sequestration.

Results

Bulk Soil Properties

Bulk density ranged from 0.13 to 0.40 g/cm^3 among sites with no significant difference between managed (0.22 ± 0.01 g/cm^3) and non-OMWM (0.20 ± 0.01 g/cm^3) marshes (Table 2). Among sites, Wertheim contained significantly lower bulk density than the other sites (0.17 vs. 0.22–0.23 g/cm^3). Among depths, the 30–60 cm increment had generally higher bulk densities than the 0–10 and 10–30-cm depths (Table 2). Soil organic carbon ranged from 8.2 to 23.9 % among managed/non-OMWM marshes, sites, and depths (Table 2). As with bulk density, there was no significant difference among managed (15.0 ± 0.6 % OC) and non-OMWM marshes (16.3 ± 0.5 % OC). Percent organic carbon generally decreased with depth. Soil N ranged from 0.49 to 1.46 % among managed/non-OMWM marshes, sites, and sampling depths. Like bulk density and organic carbon, there was no difference among managed (0.98 % N) and non-OMWM (0.99 % N) marshes (Table 2). Similar to percent organic C, N decreased with depth at all sites. Carbon:nitrogen (C:N) ratios ranged from 14.5 to 24.4 and were comparable among managed (18.1 ± 0.56) and non-OMWM marshes (19.1 ± 0.81; Table 2). There was no consistent change in C:N with depth.

Carbon and Nitrogen Pools and Accumulation

Soil organic carbon pools (0–60-cm depth) ranged from 14.4 to 17.7 kg/m^2 across sites (Table 3). There was no significant difference among cores from managed (15.9 kg/m^2) and non-OMWM marshes (16.2 kg/m^2) overall or at individual sites. On a depth weighted basis, more carbon was sequestered in the 0–10-cm depth than in the 10–30 and 30–60-cm depths. Nitrogen pools (0–60-cm depth) ranged from 753 to 1340 g/m^2 across sites with no difference among cores from managed (1032 g/m^2) and non-OMWM marshes (1035 g/m^2) (Table 3). The Rachel Carson site contained more nitrogen (1202–1340 g/m^2) than the other marshes, while the Forsythe marshes had the smallest nitrogen pools (753–920 g/m^2).

Soil accretion based on ^{137}Cs and ^{210}Pb ranged from 1.3 to 5.3 mm/year (Table 4). All eight cores contained a well-defined ^{137}Cs maximum, whereas six of the eight cores contained interpretable ^{210}Pb profiles (Fig. 2). In uninterpretable profiles, even though activity of ^{210}Pb showed an exponential decrease, the goodness-of-fit regressions of

Table 2 Means for bulk soil characteristics (bulk density, organic carbon concentrations, and nitrogen concentrations) for 60 cm soil cores

Refuge	Depth (cm)	N	Bulk density (g/cm^3)	Organic C % (w/w)	Nitrogen % (w/w)	C:N ratio
Non-OMWM	0–10	40	0.19 ± 0.01	20.1 ± 0.7	1.25 ± 0.04	18.7 ± 0.8
	10–30		0.17 ± 0.01	16.0 ± 0.5	0.95 ± 0.03	19.7 ± 1.1
	30–60		0.24 ± 0.02	12.8 ± 0.6	0.78 ± 0.03	18.9 ± 2.3
Managed	0–10	40	0.21 ± 0.02	17.8 ± 1.1	1.19 ± 0.08	17.5 ± 0.3
	10–30		0.20 ± 0.01	14.9 ± 0.8	1.00 ± 0.06	17.8 ± 1.3
	30–60		0.24 ± 0.01	12.1 ± 0.6	0.78 ± 0.05	18.9 ± 1.2
Mean non-OMWM	0–60	120	0.20 ± 0.01	16.3 ± 0.5	0.99 ± 0.03	19.1 ± 0.8
Mean managed	0–60	120	0.22 ± 0.01	15.0 ± 0.6	0.98 ± 0.04	18.1 ± 0.6

Values represent mean ± standard error

Table 3 Total carbon and nitrogen pools at each site in the top 60 cm

Refuge	Depth (cm)	C pool (g/m^2)	N pool (g/m^2)
Rachel Carson non-OMWM	0–60	17719 ± 923	1340 ± 93
Rachel Carson managed	0–60	16414 ± 1287	1202 ± 89
Parker River non-OMWM	0–60	16270 ± 1924	1112 ± 125
Parker River managed	0–60	14404 ± 1795	982 ± 124
Wertheim non-OMWM	0–60	16264 ± 355	936 ± 26
Wertheim managed	0–60	17106 ± 1257	1024 ± 75
Forsythe non-OMWM	0–60	14560 ± 885	753 ± 46
Forsythe managed	0–60	15797 ± 634	920 ± 45
Mean non-OMWM	0–60	16203 ± 589	1035 ± 53
Mean managed	0–60	15930 ± 649	1032 ± 46

Values represent sums of the 0–10, 10–30, and 30–60-cm depths

Table 4 Rates of carbon and nitrogen accumulation at each site

Refuge	Cs-137			Pb-210		
	Accretion rate (mm/year)	C (g/m^2/year)	N (g/m^2/year)	Accretion rate (mm/year)	C (g/m^2/year)	N (g/m^2/year)
Rachel Carson non-OMWM	1.3	51	3.5	1.0	41	2.8
Rachel Carson managed	2.3	75	5.2	2.9	95	6.7
Parker River non-OMWM	2.7	87	5.7	3.9	116	7.7
Parker River managed	3.1	100	6.6	5.3	170	11.3
Wertheim non-OMWM[a]	2.7	93	5.6	–	–	–
Wertheim managed	2.7	80	5.3	3.8	114	7.6
Forsythe non-OMWM	1.9	67	3.8	4.3	152	8.8
Forsythe managed[a]	3.5	118	7.6	–	–	–
Mean non-OMWM	2.2	75	4.7	3.1	103	5.4
Mean managed	2.9	93	6.2	4.0	126	8.5

Accretion rates are calculated using bulk density, percent carbon, and percent nitrogen based on the ^{137}Cs and ^{210}Pb accretion rates in the 0–10-cm depth as presented in Fig. 2

[a] These ^{210}Pb profiles were not interpretable and therefore were not used in accumulation calculations

excess ^{210}Pb were <0.8 so we did not use them. Accretion rates were slightly higher at managed (2.9–4.0 mm/year) than non-OMWM marshes (2.1–3.1 mm/year). Overall, ^{137}Cs accretion rates were comparable (1.3–3.5 mm/year) to those determined using ^{210}Pb (1.0–5.3 mm/year).

The mean rate of ^{137}Cs C accumulation at managed marshes was 93 g/m^2/year and 74 g/m^2/year at non-OMWM marshes (Table 4). For ^{210}Pb, managed marshes sequestered 126 g C/m^2/year, whereas non-OMWM marshes accumulated 103 g C/m^2/year. Nitrogen accumulation in soil exhibited trends similar to C with somewhat greater accumulation in managed than non-OMWM marshes (Table 4). Accumulation of N based on ^{210}Pb was greater than those based on ^{137}Cs (Table 4) and was driven by higher ^{210}Pb accretion. Overall, N accumulation was an order of magnitude less than C accumulation.

We used the acreage of salt marsh in each refuge along with the C sequestration rates measured in this study to estimate refuge-wide for the four NWRs (Table 5). We assumed C sequestration to be equal across the whole marsh. As we focused on the marsh plain (70–80 % of the system), we feel that our scaled-up estimates of C sequestration are a reasonable approximation of refuge-wide C sequestration. Wertheim, with the smallest acreage of salt marsh (186 ha), had the lowest area-wide C sequestration (178 tonnes/year). Forsythe, with the greatest acreage of salt marsh (12,469 ha), had the highest C sequestration (14,007 tonnes/year). Collectively, the four refuges sequestered 15,680 tonnes C annually (Table 5).

The total amount of carbon sequestered by the four sites is equivalent to over 1.7 million gallons of gasoline C emissions per year (EPA 2014). This is enough gasoline to power an average car around the equator more than 1600 times. We used Rachel Carson as an example site to determine whether the annual carbon sequestration was able to offset the emissions on site. We found that their annual gas usage (about 5200 gallons) is equivalent to

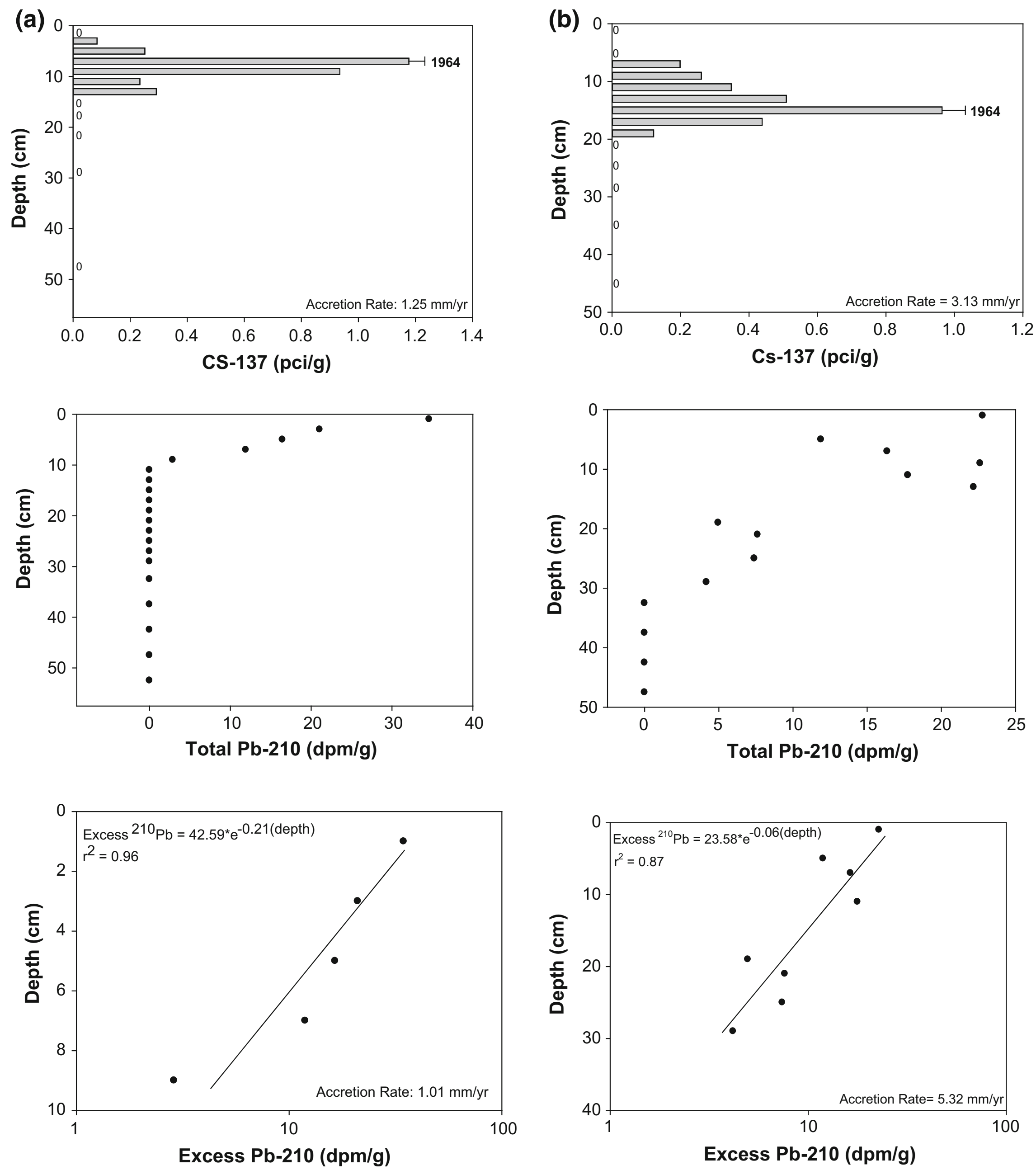

Fig. 2 Depth distribution of ^{137}C, total ^{210}Pb, and excess ^{210}Pb in for **a** Rachel Carson Non-OMWM **b** Parker River Managed **c** Wertheim Managed, and **d** Forsythe Non-OMWM. The r^2 of excess ^{210}Pb of the cores not shown ranged from 0.72 (Parker River Non-OMWM) to 0.97 (Rachel Carson Managed)

about 46.8 tonnes of CO_2/year in emissions, or 12.8 tonnes C/year, far less than the 515 tonnes C/year being sequestered annually in soil. This site is of an intermediate size relative to the other three refuges. For smaller refuges, such as Wertheim NWR, offsetting carbon usage with salt marshes may be more difficult, while for larger sites, such as Forsythe NWR, storage is likely to be much greater than greenhouse gas emissions from management activities (Table 5).

Discussion

Bulk Soil Properties

We observed no consistent differences in soil properties, bulk density, or carbon and nitrogen content between marshes that had been managed for OMWM and those that had not. By contrast, Vincent et al. (2013) examined the influence of two specific types of OMWM features, ditches

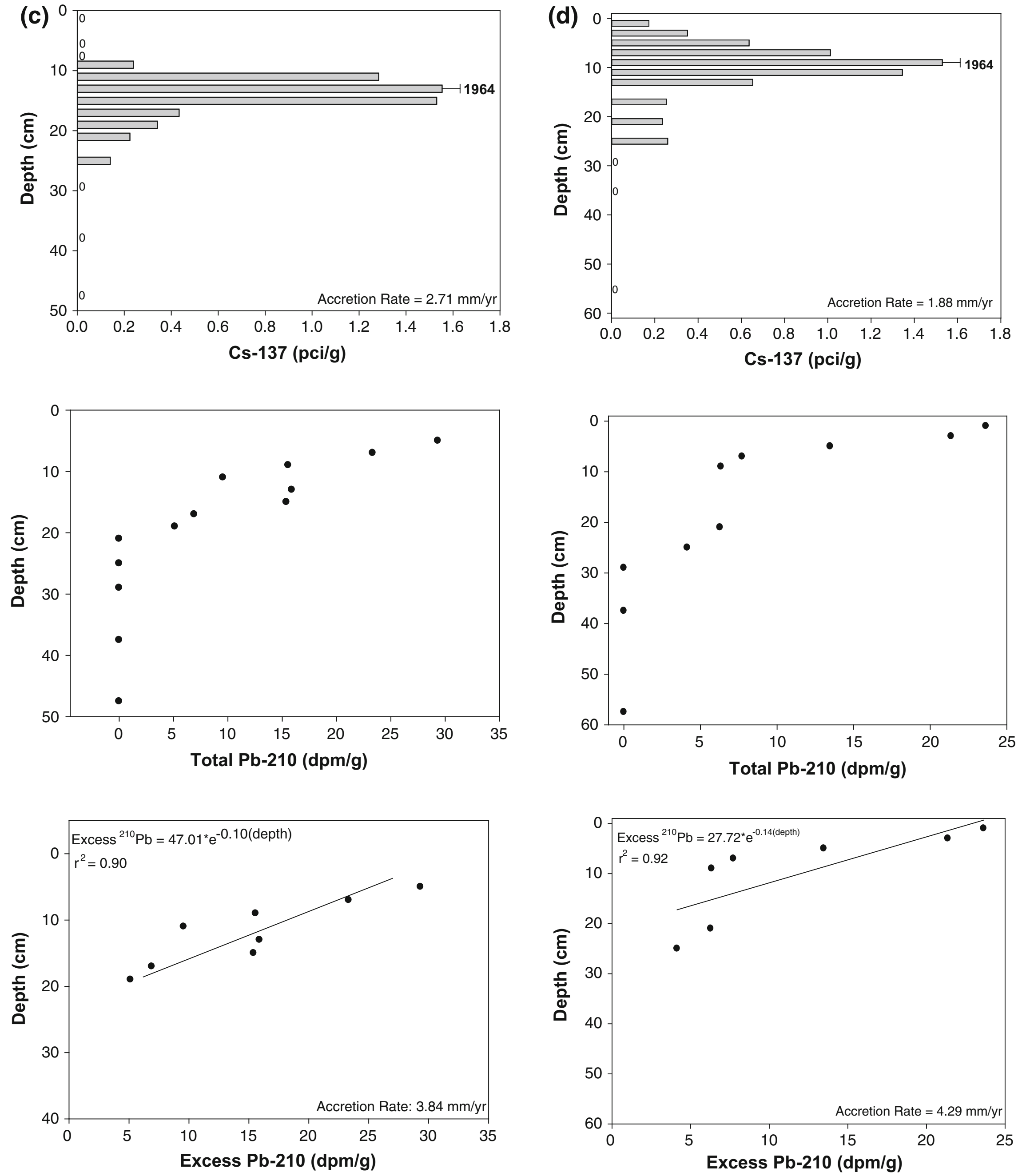

Fig. 2 continued

and ditch plugs, in comparison to creeks and natural pools, and found significantly higher bulk density in natural areas than in OMWM-managed areas. Significantly lower carbon storage was also found in ditch plug areas than in areas with ditches, creeks, and natural pools. This was attributed to changes in hydrology causing increased organic decomposition and plant stress that reduced biomass, leading to increased loss of sequestered carbon (Vincent et al. 2013). The absence of differences in our study may be attributed to the broad range of management techniques, including radial ditches, ditch plugging, and pond creation, and the variable timing of initiation of the projects (Table 1). Site differences, such as tidal range or historical treatment, and geographic range also may be contributing factors.

A comparison of our results with published studies of tidal marsh soils in the Northeast U.S. revealed that bulk density was on the low end of the range, 0.20–0.22 versus

0.1–0.84 g/cm^3 for published studies (Table 6). Percent organic carbon, however, was well within the range of published studies (15.0–16.3 % OC vs. 4.6–40 %) (Table 5) as was percent N (0.99–0.98 % N vs. 0.36–1.5 % for other studies) (Craft 2007). C:N ratios also were comparable (19.1–18.1 vs. 15–22; Craft 2007).

One source of variation may be varying soil depths sampled among studies (Table 6). Our results show a trend of increasing bulk density and decreasing organic carbon content with increasing depth (Table 2), and to some extent similar findings can be seen in previous results (Table 6). Other sources of variation may be geographic location, inundation time, and suspended sediment supply (Chmura et al. 2003). Additionally, tidal wetlands have in the past been noted to have large variability even in nearby marshes (Chmura et al. 2003). Historical efforts to drain marshes may also be a factor. The extent to which salt marshes in the Northeast have been altered over the past 300 years means that there are very few entirely natural sites in the region (Crain et al. 2009).

Table 5 Refuge-wide carbon sequestration rates for the four NWRs sampled in this study

Refuge	Salt marsh area (ha)	C sequestration (tonnes/year)	
		Range[a]	Mean
Rachel Carson	786	322–747	515
Parker River	915	1066–1556	1,080
Wertheim	186	149–213	178
Forsythe	12,469	8354–18,953	14,007
Total	14,358	9892–21,469	15,781

[a] Range of C sequestration based on ^{137}Cs and ^{210}Pb-based measurements

Table 6 Summary of results of previous studies conducted in Northeast salt marshes

Source	Location	Depth (cm)	Bulk density (g/cm^3)	Method	Organic C %	% OM	Accretion rate (mm/year)	C sequestration (g/m^2/year)
Vincent et al. 2013	ME–MA	20	0.551–0.762		7.3–10.8[a]			
Portnoy and Giblin 1997	MA	45	0.1		32.5–37.5[a]	65–75		
Roman et al. 1997	MA	10	0.23–0.38	Pb-210	15[a]	30	2.6–4.2	89.7–239.4[d]
				Cs-137			3.8–4.5	159.6–256.5[d]
Anisfeld et al. 1999	CT	30–40	0.84 ± 0.03	Cs-137			3.7 ± 0.3	170 ± 15[a]
			0.84 ± 0.03	Pb-210			3.6 ± 0.7	
Orson et al. 1998	CT	20	0.362–0.398	Cs-137			1.7–4.0	
				Pb-210			0.91–3.56	
Bricker-Urso et al. 1989	RI	100		Pb-210			0.15 ± 0.05–0.58 ± 0.02	
Clark and Patterson 1984	NY	8–26		Pb-210			4.3	
Armentano and Woodwell 1975	NY	30–85	0.2-0.35	Pb-210	9–13.5[b]	20-30	4.7–6.3	146–196
Church et al. 1981	DE	56.8		Pb-210	4.6–22.7[a]	9.2–45.4	4.7	
Kim et al. 2004	DE	18		Pb-210	10–40[a]	20–80[c]	2.6 ± 0.2–3.9 ± 0.6	
				Cs-137			2.7 ± 0.6	
Artigas et al. 2015	NY	30		Cs-137			1.4	192.2
This paper	ME–MA	60	0.20 ± 0.01	Cs-137	16.3 ± 0.45		2.14	74
				Pb-210			3.1	103
			0.22 ± 0.01	Cs-137	14.97 ± 0.55		2.92	93
				Pb-210			3.58	126

[a] Estimated using % C (or % LOI) = (0.5) % OM

[b] Estimated using % C = (0.45) % OM

[c] % LOI

[d] Calculated from BD, % OM, and sediment accretion rate

Carbon and Nitrogen Accumulation

Carbon sequestration ranged from 74 to 127 g C/m^2/year and N accumulation varied from 4.7 to 8.5 g N/m^2 year (Table 4). This range is on the low end of rates previously reported for the region (Craft 2007; Table 6), but still demonstrates that carbon is being sequestered annually by the marshes. This emphasizes the value of these environments for sequestering C for climate change mitigation, discussed further below. The long-term value is further shown by the carbon pool data, which ranged from 14.4 to 17.7 kg/m^2 and show the extent to which carbon has already been sequestered in the top 60 cm of soil (Table 3). Our results show a general trend of declining carbon and nitrogen pools and increasing C:N moving from north to south occurring at all depths. We speculate that this trend may be related to cooler temperatures in the north reducing rates of decomposition in salt marshes (Chmura et al. 2003; Craft 2007; Kirwan and Blum 2011). However, other site variables, such as adjacent land use (Silliman and Bertness 2004) and varying tidal range, which impacts the length and depth of inundation, may also influence C sequestration (Table 7).

Soil Accretion Rates and Sea Level Rise

Soil accretion ranged from 1.3 to 3.5 mm/year when measured with ^{137}Cs and 1.0–5.3 mm/year based on ^{210}Pb (Table 4). This is consistent with results from previous studies of salt marshes on the northeast coast, which found a range of 1.7–4.5 mm/year with ^{137}Cs methods and 0.15–6.3 mm/year with ^{210}Pb methods (Table 6).

As discussed above, climate change is expected to increase the rate of SLR, potentially outpacing the ability of marshes to maintain their elevation within the tidal frame. To investigate this at our sites, we compared our findings on soil accretion to rates of SLR near each marsh (Table 7). At all sites except Forsythe, marsh accretion is keeping pace with SLR, which indicates that marsh self-maintenance under current rates of SLR is occurring. Accretion measurements based on ^{137}Cs appear to more closely approximate rates of SLR, while ^{210}Pb measurements are generally higher. The relatively low accretion rates at the Forsythe site suggest that the marshes there are at risk of loss due to SLR (Table 7). These marshes are located in southern New Jersey, nearby the Delaware Bay, where atypical accelerated marsh loss due to SLR has been noted previously and partially attributed to geologic subsidence in the region (Phillips 1986). It should also be noted that Vincent et al. (2013) found significant levels of subsidence in ditch plugged habitats (8.6 cm) that were associated with greater decomposition. This may be a sign that OMWM-managed marshes are less able to perform self-maintenance than non-OMWM marshes, despite adequate rates of soil accretion (Vincent et al. 2013).

Management Implications

Our study sites were shown to contain carbon pools ranging from 14.4 to 17.7 kg/m^2 in their top 60 cm and to be sequestering carbon annually at a rate of 74–126 g/m^2 year. Vegetated coastal ecosystems (marshes, mangroves, and sea grass) sequester carbon for very long periods of time relative to many terrestrial ecosystems (Mcleod et al. 2011), which emphasizes the importance of these ecosystems to global greenhouse gas mitigation. Global estimates of carbon sequestration in salt marshes occur on the order of 100–200 g/m^2 year (Chmura et al. 2003; Duarte et al. 2005). Additionally, compared to freshwater wetlands, temperate salt marshes produce relatively small amounts of CH_4 (Bartlett and Harriss 1993), and marshes with salinity greater than 18 have been found to have significantly lower methane emissions than other marshes (Poffenbarger et al. 2011). Methane is approximately 21 times as potent a greenhouse gas as CO_2, making coastal systems particularly valuable to C sequestration on the global scale. Mean

Table 7 Comparison of sea level rise rates with soil accretion rates

Relevant site	Gage location	Mean range (m)	SLR rate[a] (mm/year)	Mean accretion rate (mm/year)	
				Cs-137	Pb-210
Rachel Carson	Portland ME	2.78	1.82 ± 0.17	1.8	1.95
	Portsmouth NH	2.63	1.76 ± 0.30		
Parker River	Portsmouth NH	2.63	1.76 ± 0.30	2.9	4.6
	Boston MA	2.89	2.63 ± 0.18		
Wertheim	Port Jefferson NY	1.96	2.44 ± 0.76	2.7	3.75
	Montauk NY	0.63	2.78 ± 0.32		
Forsythe	Atlantic City NJ	1.23	3.99 ± 0.18	2.7	4.3
	Cape May NJ	1.48	4.06 ± 0.74		

[a] (NOAA 2013)

salinities at our sites ranged from 12 to 28, with only one site having a mean salinity below 18 (Wertheim, 12).

Carbon sequestration and storage are placed at risk from rising sea levels in areas where there is no room for marshes to migrate due to limitations such as developed shorelines or abrupt elevation changes. Instead, conversion to open water can cause sequestered carbon to be released to an adjacent estuary and eventually oxidized and converted to CO_2 (DeLaune and White 2012). Globally, the value of lost tidal marshes due to climate change may be billions of U.S. dollars per year (Pendleton et al. 2012). This emphasizes the economic importance of protecting the salt marshes in this study, which collectively sequester 9892–21,469 tonnes C/year (mean 15,781 tonnes/year) (Table 5). Loss of these NWR salt marshes would mean loss of these valuable carbon sequestration services. Protection of coastal lands makes it possible for land managers to increase biological carbon sequestration while simultaneously preserving and creating habitat. Public land management agencies such as the U.S. Fish and Wildlife service, which oversees extensive estuarine and freshwater wetlands, have the opportunity to apply this on a very large scale.

Although current U.S. policy does not directly address the value of carbon sequestration in wetlands (Pendleton et al. 2013) or greenhouse gas mitigation on public lands, there is future opportunity if these areas are protected, such as from sale of carbon offsets or participation in carbon trading programs (Olander et al. 2012). Such a program is already underway in the U.S. in the Chicago Climate Exchange (DeLaune and White 2012). A similar program, the Regional Greenhouse Gas Initiative (RGGI), exists in the northeast, through which it is currently possible for companies to earn offsets through reforestation, improved forest management, or avoided forest conversion (http://www.rggi.org/market/offsets/categories/forestry-afforestation) and which could potentially be expanded in the future to include carbon sequestration by wetlands.

In addition, wetland management protocols for C sequestration have recently been developed. For example, the American Carbon Registry has a protocol for wetland restoration in the Mississippi Delta and is developing a similar one for California (http://americancarbonregistry.org/carbon-accounting/standards-methodologies). Similarly, there is a movement toward increased recognition of the carbon sequestration value of these ecosystems internationally (UNEP and CIFOR 2014). However, challenges remain, including quantifying C balance (e.g., C sequestration vs. methane emissions), determining whether C is sequestered permanently, and how different management practices affect C sequestration (Emmett-Mattox et al. 2010). Additional research that provides insight into how C sequestration is affected by different land management techniques and permanence in the face of accelerated SLR is needed.

In conclusion, our data showed that open marsh water management had no consistent effects on soil properties (bulk density, percent organic C, and percent N), carbon stocks, or the rate of carbon sequestration and N accumulation in soil. Soil organic C pools (0–60-cm depth) among sites were sizable, 14.4–17.7 kg/m^2, and decreased from north (ME) to south (NJ). Overall, the presence of large carbon pools and ongoing annual carbon accretion of 74–126 g/m^2/year shows the value of maintaining these ecosystems for ongoing carbon sequestration and storage to offset local (and regional) C footprints associated with management of refuge lands.

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