Limnol. Oceanogr., 52(3), 2007, 1220–1230

Freshwater input structures soil properties, vertical accretion, and nutrient accumulation of Georgia and U.S. tidal marshes

Christopher Craft[1]
School of Public and Environmental Affairs, Indiana University, Bloomington, Indiana 47405

Abstract

To identify relationships between freshwater input and marsh soil properties, measurements of bulk density, nutrients (carbon [C], nitrogen [N], phosphorus [P]), accretion, and accumulation were compared in tidal marshes of three estuaries of Georgia that varied in delivery of freshwater. Soil organic C and N (0–30 cm) were two times greater in marshes of the freshwater-dominated Altamaha River than in the salt marshes of Doboy Sound and Sapelo River. ^{137}Cs accretion and accumulation of organic C and N were three to five times greater in freshwater-dominated marshes than in salt marshes. The patterns observed in Georgia marshes were geographically general; data for tidal freshwater and brackish marsh soils compiled from 61 studies in the conterminous United States showed lower bulk density and higher percent organic C and N than salt marshes, regardless of geographic region. Salinity, a proxy for freshwater input, was inversely correlated with percent soil organic C and N and with vertical accretion in Georgia marshes and in marshes elsewhere in the conterminous United States. There was no relationship between above- or belowground emergent plant production and salinity of Georgia marshes but the rate of root decomposition was positively related to salinity, and decomposition rate was negatively related to percent soil organic C and C accumulation. In Georgia tidal marshes and elsewhere, soil organic matter content and accumulation are mediated by freshwater through its effects on decomposition.

The mixing of freshwater and seawater is a fundamental driving force that determines the structure and function of tidal marshes. Freshwater governs the distribution of emergent plant communities and tidal marsh fauna along the estuarine gradient (Simpson et al. 1983; Odum 1988). Spatial patterns in biogeochemical processes such as sediment deposition (Paludan and Morris 1999), P sorption (Sundareshwar and Morris 1999), denitrification (Seitzinger 1988), methanogenesis (Bartlett et al. 1987), and sulfate reduction (Capone and Kiene 1988) also are linked to spatial and temporal variation in freshwater input.

Soils are an important component of tidal marshes. They sequester organic matter, N, and P (Craft et al. 1988), support complex biogeochemical reactions (Capone and Kiene 1988), and contribute to long-term marsh stability through deposition of mineral sediment and accumulation of organic matter (DeLaune et al. 1983; Hatton et al. 1983; Nyman et al. 1990; Craft et al. 1993; Morris et al. 2002). Some work to date has suggested that soil properties vary with freshwater input along the estuarine gradient. Along the Louisiana Gulf coast, percent organic C and N were greatest in tidal freshwater marshes and decreased along the salinity gradient to salt marshes (Hatton et al. 1983). Bulk density was lower in tidal freshwater marshes than in salt marshes (Hatton et al. 1983; Nyman et al. 1990). In the Cooper River estuary (South Carolina), tidal marshes exhibited similar patterns in bulk density and percent C, N, and P along the gradient from tidal freshwater marsh to salt marsh (Paludan and Morris 1999; Sundareshwar and Morris 1999).

Soil accretion, the change in vertical elevation, also is linked to freshwater input (Stevenson et al. 1986; Nyman et al. 1990; DeLaune et al. 2003). Riverine marshes along the Nantikoke River (Maryland) and Savannah River (Georgia) exhibited higher rates of accretion than nonriverine estuarine marshes though much of the difference was attributed to greater sediment input in riverine marshes (Stevenson et al. 1986). Along the Louisiana Gulf coast, vertical accretion was higher in brackish and tidal freshwater marshes than salt marshes (Hatton et al. 1983) or there was no clear trend in accretion among marshes along the salinity gradient (Nyman et al. 1990).

Results from the studies above suggest that tidal marsh soil properties vary along the gradient from tidal freshwater marshes to salt marshes. It is not clear though, whether these patterns are consistent across different geographic regions, nor is there a clear understanding of the processes that drive these patterns. To resolve these issues, I measured soil properties (bulk density; percent organic C, N, P), vertical accretion, and mass accumulation of sediment, organic C, N, and P in nine tidal marshes on the Georgia coast that vary in freshwater input. I

[1] Corresponding author (ccraft@indiana.edu).

Acknowledgments

The idea for this work grew out of two National Science Foundation (NSF)–sponsored workshops, *Regulation of Organic Matter Preservation in Wetland Sediments* held at NSF Long Term Ecological Research (LTER) All Scientists Meeting, Snowbird, Utah (1–5 August 2000) and *Soil Organic Matter in Wetlands* held at Virginia Institute of Marine Sciences, Gloucester Point VA (26–29 July 2001). Thanks to Ken Helm, Sean Graham, and Josh Hall for their assistance in the field; Chrissy Pruett, Jillian Bertram, and Sarah Butler for preparing and analyzing soil samples; and Steve Pennings, Dale Bishop, and Samantha Joye for sharing data. Steve Pennings, Pat Megonigal, Ross Brittain, and two anonymous reviewers provided constructive comments on previous versions of the manuscript.

This work was supported by grant OCE-9982133 from the National Science Foundation and is a contribution of the Georgia Coastal Ecosystems LTER program.

This is Contribution 889 of the University of Georgia Marine Institute.

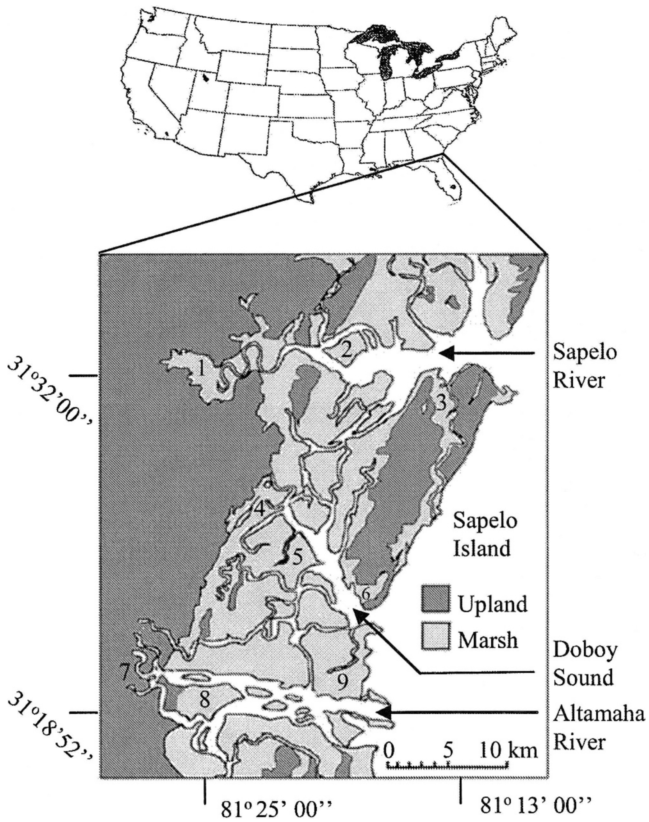

Fig. 1. Location of freshwater- (Altamaha River) and marine- (Doboy Sound, Sapelo River) dominated marshes along the Georgia coast. Numbers on the figure correspond to the stations in Table 1.

hypothesized that enhanced delivery of freshwater would lead to reduced bulk density and greater percent C, percent N, vertical accretion, and accumulation of soil organic matter and nutrients. I also measured root decomposition and productivity in four marshes along the salinity gradient to test the role of these processes in controlling soil organic C stocks and accumulation. In addition, I surveyed soil properties of tidal marshes of the conterminous United States (northeast and southeast Atlantic, Gulf, West coasts) to ascertain whether the patterns observed in Georgia marshes are representative of tidal marsh systems in other geographic regions.

Methods

Site description—Marshes of three estuaries on the Georgia coast, representing low (Sapelo River, Doboy Sound) and high (Altamaha River) inputs of freshwater, were selected for sampling (Fig. 1). Within each estuary, I sampled three marshes arrayed along an onshore–offshore gradient and consisting of upstream, intermediate, and downstream landscape positions (Table 1). I used salinity as a proxy for freshwater input. Along the Altamaha River—the freshwater-dominated estuary—mean surface water salinity (2002 and 2003) in the adjacent river ranged from 0.15 at the upstream location to 16.5 downstream (Table 1). Salinities at comparable locations in Doboy Sound and Sapelo River were substantially greater, ranging from 23 to 30, except for the landward location on Sapelo River (site 1, Eulonia), where salinity was 13.5. Tide range was similar at all sites, including those along the salinity gradient, and was approximately 2.3 m.

Vegetation of the Altamaha River marshes consisted of smooth cordgrass (*Spartina alterniflora* Loisel) at the downstream location, giant cordgrass (*Spartina cynosuroides* (L.); levee), and *Juncus roemerianus* Scheele (marsh plain) at the intermediate brackish marsh location, and giant cutgrass (*Zizaniopsis milaceae*) (Michaux) at the upstream tidal freshwater marsh location (Table 1). In Doboy Sound and Sapelo River, *S. alterniflora* was the dominant species at all landscape positions except for the upstream landscape position of the Sapelo River (site 1), where *J. roemerianus* was the dominant species on the marsh plain.

Soil sampling and lab analyses—In 2001, one soil core, 8.5 cm diameter by 30 cm deep, was collected from the streamside zone and one from the marsh plain at the nine sampling locations. Cores were sectioned into 2-cm depth

Table 1. Site number, name, estuary, dominant water source, surface water salinity, and dominant vegetation of the nine marsh study sites. See Fig. 1 for location of the sites. For each estuary, sites 1, 4, and 7 are farthest inland.

Site No.	Name	Estuary	Water source	Salinity*	Dominant vegetation
1	Eulonia	Sapelo River	Marine influence	12–15	*Spartina alterniflora*, *Juncus roemerianus*
2	Four-mile			25–29	*S. alterniflora*
3	N Sapelo			28–32	*S. alterniflora*
4	Meridian	Doboy Sound	Marine influence	20–25	*S. alterniflora*
5	Folly River			22–25†	*S. alterniflora*
6	Dean Creek			25–29	*S. alterniflora*
7	Carr's Island	Altamaha River	Freshwater influence	0–0.3	*Zizaniopsis milaceae*
8	Alligator Creek			2–6	*Spartina cynosuroides*, *J. roemerianus*
9	Rockdedundy Island			14–19	*S. alterniflora*, *J. roemerianus*

* Annual (2003 and 2002) salinity calculated from daily measurements in the water column of the adjacent estuary.
† Interpolated from measurements at the nearest locations, Meridian and Dean Creek (*see Fig. 1*).

increments in the field and transported to the lab where they were air-dried, weighed for bulk density, then ground and sieved through a 2-mm-diameter mesh screen. Ground soil was packed into 50-mm-diameter by 9-mm-deep petri dishes and analyzed for ^{137}Cs to determine vertical accretion on the basis of gamma analysis of the 661.62 keV photopeak (Craft et al. 2003).

Subsamples of ground soil were analyzed for organic C and total N with a Perkin-Elmer 2400 CHN analyzer. Samples were tested for presence of carbonates by adding a drop of dilute (0.1 mol L^{-1}) HCl then observing whether effervescence occurred. Samples containing carbonates were pretreated with 0.1 mol L^{-1} HCl before CHN analysis. Total P was determined by colorimetric analysis after digestion in nitric-perchloric acid (Sommers and Nelson 1972). Percent sand, silt, and clay (0–10 cm) was determined by the hydrometer method (Gee and Or 2002).

Short-term sediment accretion was measured with feldspar marker plots (Cahoon 1994). Three feldspar plots (0.25 by 0.25 m^2) were established on the marsh plain at each site in June 2003. Small-diameter soil cores (four per plot) were collected in January 2004 after 6.5 months. At some salt marsh sites (2, 3, 6), the feldspar marker layer could not be discerned because of bioturbation by fiddler crabs.

Root decomposition and in-growth—Root decomposition and productivity were measured in a subset of marshes (sites 6, 7, 8 , 9), in which salinity ranged from 0 to 27, following the methods of Blum (1993). Nylon mesh bags (2 by 2 mm), 30 cm long by 7 cm wide, were filled with 10 g of fresh roots collected from the dominant vegetation of the levee and plain of each marsh (Table 1). Sixteen bags containing native roots were deployed on each levee, and 16 were deployed on each marsh plain for a total of 128 bags. Bags were buried to a depth of between 10 and 20 cm in June 2003. Approximately every 3 months for 1 yr (364 d), four bags each were retrieved from the levee and the marsh plain at each site. Bags were dried at 70°C to a constant weight and weighed. Before deployment, a subset of four bags from each marsh location was dried immediately to determine the initial (time 0) dry mass. Soil temperature (n = 5 per marsh location) was measured seasonally at the same time that root bags were retrieved by inserting a temperature probe 10 cm into the soil.

Root production was determined by the in-growth method (Blum 1993). Every 3 months when bags were retrieved, new roots that grew into the bags were carefully removed by hand. Fresh roots were washed with distilled water to remove soil material and dried at 70°C to a constant weight. Root production was calculated as the sum of new roots that grew into each of the four bags collected seasonally during the 1-yr sampling period.

Geographic comparison of soil properties—Soil properties and accumulation of sediment, organic C, N, and P were compared in tidal freshwater, brackish, and salt marshes of the northeast Atlantic (New Jersey to Maine), southeast Atlantic (Florida to Delaware), Gulf coast (Texas to Florida), and West Coast (California to Washington). A total of 61 published studies and two unpublished datasets, including this one, were used in the statistical analysis (*see* Web Appendix 1, Table A1, http//www.aslo.org/lo/toc/vol_52/issue_3/1220a1.pdf). Forty published studies and this study were used in the analysis of vertical accretion and sediment, C, N, and P accumulation (*see* Web Appendix 1, Table A2).

Statistical analysis—Differences in soil properties, accretion, and accumulation among the three estuaries were tested by analysis of variance followed by post hoc comparisons using the Ryan-Einot-Gabriel-Welsch Multiple Range Test (REGWQ. SAS 1996). Differences between the marsh levee and plain were tested by Student's t-test. Statistical analysis of soil properties (n = 18 cores) was performed on mean values integrated over the 30-cm depth (n = 15 depth increments). Because ^{137}Cs profiles from four levee sites were uninterpretable, statistical analysis of accretion; sediment deposition; and C, N, and P accumulation were based on 14 rather than 18 cores.

Analysis of variance and REGWQ also were used to test for differences among tidal freshwater, brackish, and salt marshes of the conterminous United States and for differences among geographic regions for a given marsh type (i.e., salt marshes). I excluded Texas (salt) marshes from the statistical analysis because they differed so much from the Louisiana salt marshes that receive large amounts of freshwater and sediment relative to the Texas marshes. For statistical analyses, I defined tidal freshwater marshes as having salinity < 0.5, brackish marshes with salinity = 0.5–15, and salt marshes with salinity > 15. The salinity ranges also corresponded to the dominant vegetation (e.g., *S. alterniflora* in salt marshes) for each marsh type.

Differences in root decomposition and production were determined with a two-way analysis of variance on the basis of marsh type and location (levee, streamside). Main effects means were separated by REGWQ. Decomposition was modeled by an exponential decay regression of dry mass remaining versus time. Correlation and linear regression were used to explore relationships among shoot and root production, root decomposition, environmental variables (salinity, temperature, root C : N, surface water inorganic N, P, and particulate C, crab burrows), percent organic C and C accumulation. For all statistical analyses, tests of significance were based on $\alpha = 0.05$.

Results and discussion

Vertical accretion—Fourteen of 18 soil cores contained interpretable ^{137}Cs profiles with well-defined peaks (Table 2), and four levee sites (2, 3, 5, 9) lacked distinct ^{137}Cs maxima in the soil profile. Marshes of the freshwater-dominated Altamaha River had higher rates of vertical accretion ($p < 0.05$) compared with marshes of the marine-dominated Doboy Sound and Sapelo River (Fig. 2a). Short-term accretion measured by feldspar marker layers also was significantly greater in the Altamaha River marshes than in the marshes of Doboy Sound (Fig. 2b), and it was positively correlated with ^{137}Cs-based measurements of accretion ($r = 0.85$, $p < 0.05$, $n = 6$ plain sites).

Table 2. Distribution of ^{137}Cs with depth and ^{137}Cs inventories (0–30 cm) in Sapelo River, Doboy Sound, and Altamaha River marsh soils. ^{137}Cs maximum (±1 counting error) of interpretable cores is shown in bold.

	^{137}Cs (Bq kg^{-1})					
	Site 1		Site 2		Site 3	
Depth (cm)	Levee	Plain	Levee	Plain	Levee	Plain
Sapelo River						
0–2	18.5	**8.14 ± 0.7**	0	1.85	4.44	6.66
2–4	**30.34 ± 1.7**	1.11	1.48	2.22	6.29	**7.77 ± 1**
4–6	24.42	0.00	1.85	2.59	5.55	7.77
6–8	22.94	1.11	1.48	2.96	5.92	6.66
8–10	16.28	0.00	2.59	3.70	5.92	5.92
10–12	16.28	0.37	4.81	2.96	6.29	3.70
12–14	11.10	1.48	3.70	2.59	4.44	4.07
14–16	4.81	0	3.33	**6.29 ± 1.0**	5.18	4.44
16–18	3.70	0.74	5.55	5.92	5.18	3.33
18–20	2.96	0	3.70	3.33	5.55	2.96
20–22	1.85	0.74	1.48	4.44	5.92	0
22–24	2.22	0	4.07	2.22	6.29	0
24–26	1.85	1.48	1.48	0.74	5.55	2.96
26–28	0	1.85	0	1.48	6.29	0
28–30	1.85	0.00	0	0	6.29	0
Inventory (Bq cm^{-2}):	80.7	30.7	31.1	37.0	59.2	46.3
	Site 4		Site 5		Site 6	
Doboy Sound						
0–2	18.5	**8.14 ± 1.0**	0.48	4.44	5.92	**14.1 ± 1**
2–4	**30.34 ± 1.0**	1.11	4.81	8.14	5.55	12.95
4–6	24.42	0	7.03	**11.84 ± 1.9**	**31.45 ± 0.9**	10.36
6–8	22.94	1.11	5.55	3.70	4.81	7.77
8–10	16.28	0	4.44	2.59	5.92	4.07
10–12	16.28	0.37	6.29	3.33	3.70	2.96
12–14	11.10	1.48	6.66	2.22	2.22	3.70
14–16	4.81	0	5.92	2.22	2.96	2.96
16–18	3.70	0.74	7.03	2.59	0	2.22
18–20	2.96	0	7.03	0	0	1.85
20–22	1.85	0.74	5.55	0.74	0.74	1.85
22–24	2.22	0	5.18	0	0.37	0
24–26	1.85	1.48	2.59	0	0.37	0
26–28	0	1.85	2.96	0	0	0
28–30	1.85	0	2.96	0	0	0
Inventory (Bq cm^{-2}):	38.5	13.3	43.3	30.0	47.0	51.1
	Site 7		Site 8		Site 9	
Altamaha River						
0–2	23.31	37.00	11.84	14.43	5.92	0
2–4	27.75	41.44	13.32	9.99	5.92	2.22
4–6	32.56	52.54	12.95	21.83	3.33	4.44
6–8	36.26	57.72	13.69	**26.64 ± 2.2**	5.92	**7.92 ± 1**
8–10	40.70	61.42	17.76	23.31	5.18	4.07
10–12	46.62	56.98	20.72	25.53	3.70	4.07
12–14	46.20	**100.27 ± 2.7**	21.09	22.57	4.44	3.33
14–16	50.32	73.63	29.97	22.94	4.00	4.81
16–18	**58.09 ± 2.1**	17.39	37.00	21.09	7.40	4.40
18–20	19.61	3.33	**46.62 ± 2.1**	16.65	5.18	4.29
20–22	2.22	2.22	37.37	7.03	4.81	4.07
22–24	0	0	22.20	5.55	5.55	3.70
24–26	1.11	1.11	17.39	2.96	5.55	3.70
26–28	1.11	0	12.95	0	5.92	0
28–30	0	1.11	9.25	2.59	5.92	0
Inventory (Bq cm^{-2}):	212.7	194.6	202.8	85.1	66.6	43.3

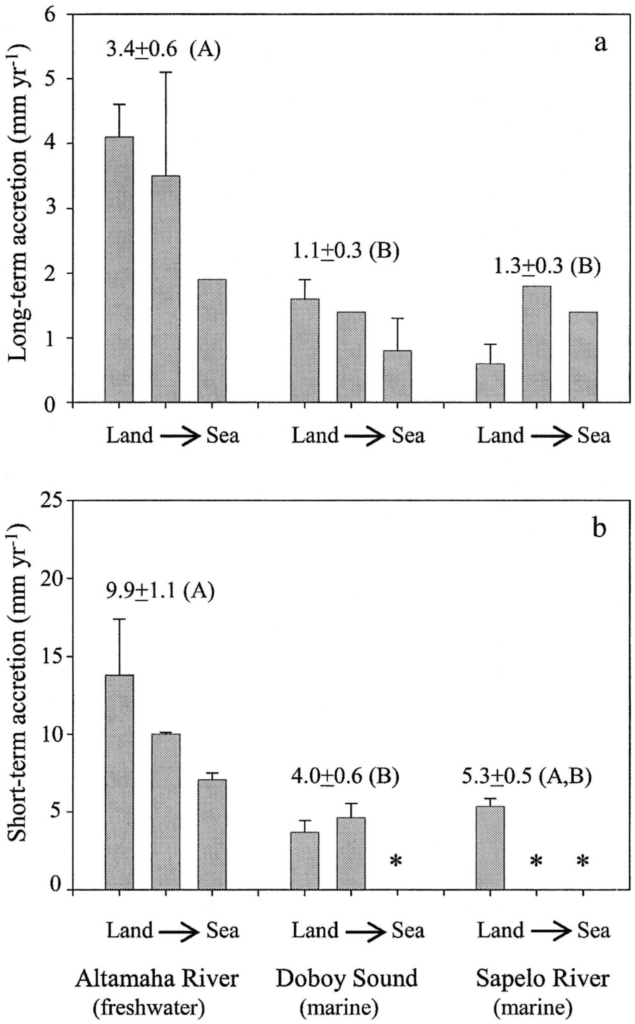

Fig. 2. (a) Long-term (^{137}Cs) and (b) short-term (feldspar marker) vertical accretion of freshwater- (Altamaha River) and marine- (Doboy Sound, Sapelo River) dominated marshes along the Georgia coast. Means ± 1 SE are shown for each estuary. Means separated by the same letter (in parentheses) are not significantly different ($p \leq 0.05$) on the basis of the Ryan-Einot-Gabriel-Welsch multiple range test. Locations marked with an asterisk (*) were omitted: the feldspar marker layer could not be detected because of bioturbation by fiddler crabs.

Sapelo River marshes exhibited statistically intermediate rates of short-term accretion (5.3 ± 0.5 mm yr^{-1}), but mean rates were close to those at Doboy Sound though bioturbation was problematic and reduced the sample size in these estuaries. Among individual marshes, short-term accretion was greatest in the tidal freshwater marsh (14 ± 4 mm yr^{-1}), the upstream location of the Altamaha River (site 7), compared with other marshes (4–10 mm yr^{-1}) (Fig. 2b).

Cs-137 is known to be fixed by micaceous clay minerals in soils and sediments but, in anaerobic lake sediments, can be remobilized by ion exchange with NH_4^+ (Comans et al. 1989). Remobilization of ^{137}Cs is greater initially (2–3% after year 1) but declines with time (0.5% per year after 30 yr) as it is buried by accumulating sediment (Smith and Comans 1996). After 30 yr, up to 20% of the ^{137}Cs inventory may be remobilized. In salt marshes, there is evidence for downward diffusion of ^{137}Cs relative to ^{241}Am, another artificial radionuclide marker that is less mobile than ^{137}Cs, but the ^{137}Cs peak remains in place (Thomson et al. 2002). Also, in salt marshes, it appears that the location of the ^{137}Cs peak is unaffected by differences in redox (oxidized vs. reduced) state (Thomson et al. 2002).

Evidence for ^{137}Cs remobilization in Georgia tidal marsh soils comes from ^{137}Cs inventories (0–30 cm depth) that range from 7% (site 4, plain) to 118% (site 7, levee) of decay-corrected atmospheric deposition (180 Bq cm^{-2}), with an overall mean of 73 Bq cm^{-2} or 41% of atmospheric deposition. ^{137}Cs inventories were greatest in tidal freshwater marshes and brackish marshes of the Altamaha River (Table 2) that had the highest percent soil organic C (Table 3) and that received greater inputs of terrestrially derived riverborne particles (Fig. 2b). The ^{137}Cs inventory was lowest at site 4 (plain), where percent organic matter was low (Table 3). The ^{137}Cs maximum was located at the soil surface (Table 2); hence, accretion was negligible. Similar to this study, ^{137}Cs inventories in marine sediments were greater in areas that receive ^{137}Cs associated with riverborne particles (Su and Huh 2002). And in tidal marsh soils, terrestrial soils, and marine sediments, ^{137}Cs inventories were positively related to organic matter content (McHenry and Ritchie 1977; Park et al. 2004). Regardless of ^{137}Cs remobilization that could result in low ^{137}Cs inventories in Georgia tidal marsh soils, the similarity in trends of ^{137}Cs- and feldspar-based accretion rates provides convincing evidence that freshwater input promotes vertical accretion in these marshes.

Soil organic C and N—Percent organic C and C accumulation were two and three times greater, respectively, in marshes of the freshwater-dominated Altamaha River than in marshes of Doboy Sound and Sapelo River (Table 3; Fig. 3a). Greater percent organic C and C accumulation in low-salinity marshes might be attributed to enhanced plant productivity (NPP) that adds organic C to the soil or suppressed decomposition that preserves C, or both. In situ measurements with buried roots suggest C accumulation is linked to freshwater input through its effect on decomposition. After 1 yr, mass loss of "native" roots, that is roots collected from emergent vegetation growing at each site (*see Table 1*), was significantly greater ($p < 0.001$) in two salt marshes (41–49%) than in the brackish marsh (29–30%) and the tidal freshwater marsh (30–36%). The rate of decomposition (k) was positively related to water column salinity (Fig. 4a) but not to litter quality (e.g., nutrients, lignin), which is known to affect decomposition rates (Melillo et al. 1982; Conn and Day 1997). Although litter quality differed among the source materials, there was no relationship between decomposition expressed as decay rate coefficients (k) and root C : N ($r^2 = 0.01$), one common measure of litter quality (C.B. Craft unpubl. data). Decomposition rate also was unrelated to surface water nutrient concentrations. Saline marshes exhibiting the highest rates of decomposition had lower concentrations of dissolved inorganic N (DIN, 3.8–6 μmol

Table 3. Soil bulk density, organic C, total N, total P, C : N, and N : P (0–30 cm depth) of freshwater-dominated (Altamaha River) versus marine-dominated marshes (Doboy Sound, Sapelo River).*

		Bulk density ($g\ cm^{-3}$)		Organic C (%)		Nitrogen (%)		Phosphorus ($\mu g\ g^{-1}$)		C : N (mol:mol)		N : P (mol:mol)	
Site No.	Estuary	Levee	Plain	Levee	Plain	Levee	Plain	Levee	Plain	Levee	Plain	Levee	Plain
1	Sapelo River	0.23	1.09	11.3	1.4	0.51	0.08	450	260	26	20	25	7
2		0.47	0.43	4.1	4.2	0.23	0.25	580	670	21	20	9	8
3		0.39	0.41	5.1	4.2	0.35	0.30	860	590	17	16	9	11
4	Doboy Sound	1.10	1.03	1.4	2.1	0.09	0.13	100	80	17	19	20	35
5		0.30	0.35	5.5	6.6	0.37	0.36	450	600	18	21	18	13
6		0.41	0.41	5.2	4.6	0.34	0.32	520	520	18	17	15	13
7	Altamaha River	0.32	0.19	8.0	14.7	0.54	0.93	560	520	17	18	21	39
8		0.31	0.22	10.1	16.2	0.65	0.84	640	560	18	23	22	33
9		0.27	0.38	7.9	4.1	0.50	0.40	510	648	19	18	21	9
Mean Sapelo ($n = 6$)		0.50±0.12 a		5.0±1.4 a		0.29±0.06 a		570±80 a		20±1 a		11±3 a	
Mean Doboy ($n = 6$)		0.60±0.15 a		4.2±0.8 a		0.27±0.05 a		380±90 a		18±1 a		19±3 ab	
Mean Altamaha ($n = 6$)		0.28±0.03 a		10.2±1.8 b		0.64±0.08 b		570±20 a		18±1 a		25±4 b	

* Estuary means separated by the same letter are not significantly different ($\alpha<0.05$) according to the Ryan-Einot-Gabriel-Welsch multiple range test.

L^{-1}) and P (0.26–0.37 $\mu mol\ L^{-1}$) than the tidal freshwater and brackish marshes (DIN = 13.5–14 $\mu mol\ L^{-1}$, dissolved inorganic P = 0.38–0.49 $\mu mol\ L^{-1}$; Samantha Joye unpubl. data). The relationship between k and soil temperature (10 cm depth) , another factor that affects decomposition rate, was not significant ($r^2 = 0.35$, $p = 0.12$). Root decomposition has been shown to vary spatially in salt marshes, with greater decomposition at higher elevations on the levee (Hemminga et al. 1988). However, in Georgia marshes, there was no relationship between decomposition and levee versus plain locations (Fig. 4a).

The positive relationship between decomposition and salinity could be linked to the availability of sulfate for anaerobic decomposition or to biotic factors linked to salinity.

Sulfate, a terminal electron acceptor for anaerobic decomposition, is readily supplied by inundation with seawater, and in brackish marshes and salt marshes, sulfate reduction is the dominant pathway of anaerobic organic matter mineralization. In a laboratory study, salinity intrusion of 10 ppt to Georgia tidal freshwater sediments doubled anaerobic organic matter mineralization rates compared with sediments that were exposed to freshwater only. After 4 weeks of exposure, sulfate reduction accounted for 95% of total organic matter mineralization (Weston et al. 2006). These findings contrast with field-based studies that reported no difference in anaerobic organic matter mineralization in brackish versus tidal freshwater marsh soils (Neubauer et al. 2005) and sediments (Kelley et al. 1990).

Of unknown importance is the contribution of aerobic respiration to explaining the patterns of decomposition, percent soil organic C, and C accumulation observed in Georgia marshes. Bioturbation by fiddler crabs could promote aerobic decomposition through burrowing that promotes oxygen diffusion into the soil (Montague 1980; Bertness 1985; Kostka et al. 2002). In this study, the relationship between density of crab burrows and decomposition rate was significant, with greater decomposition in salt marshes where fiddler crab burrows were most abundant (Fig. 4b).

Whereas soil organic C content and accumulation were related to decomposition rate (Fig. 4c,d), there was no relationship between aboveground NPP (Steve Pennings unpubl. data) and percent organic C ($r^2 = 0.01$, $n = 18$) or organic C accumulation ($r^2 = 0.12$, $n = 14$). There also was no relationship between root production and percent C ($r^2 = 0.05$) or C accumulation ($r^2 = 0.09$; $n = 4$ marshes, levee and plain locations), suggesting that marsh soil organic matter dynamics are not determined by inputs of NPP.

Sediment deposition is another factor that could promote accumulation of soil organic matter (DeLaune et al. 2003) by burying it while diluting organic C content of the soil. In Georgia marshes, I observed greater C accumulation in tidal freshwater and brackish marshes, in which sediment deposition on feldspar marker layers was greater, than in salt marshes (Fig. 2b). However, percent organic C content also was greater in the tidal freshwater and brackish marshes, suggesting that burial by itself does not regulate organic matter dynamics of tidal marsh soils.

Finally, allochthonous organic matter deposited on the marsh surface during tidal flooding could conceivably explain the high organic C observed in Altamaha River marshes. However, there was no relationship between particulate carbon measured seasonally in nearby estuarine waters (June 2003–March 2004; Samantha Joye unpubl. data) and percent soil organic C ($r^2 = 0.12$, $p = 0.18$). Also, organic C accumulation in soil was negatively related to water column particulate carbon ($r^2 = 0.50$, $p < 0.01$). Thus, allochthonous C inputs to Georgia tidal marsh soils appear to be low to negligible, which is corroborated by studies from a variety of marsh types and geographic locations, indicating that tidal marsh soil organic matter is derived mostly from accumulation of belowground biomass produced by emergent vegetation (McCaffrey and Thomson 1980; Hatton et al. 1983; Bricker-Urso et al. 1989; Nyman et al. 1990, Blum 1993).

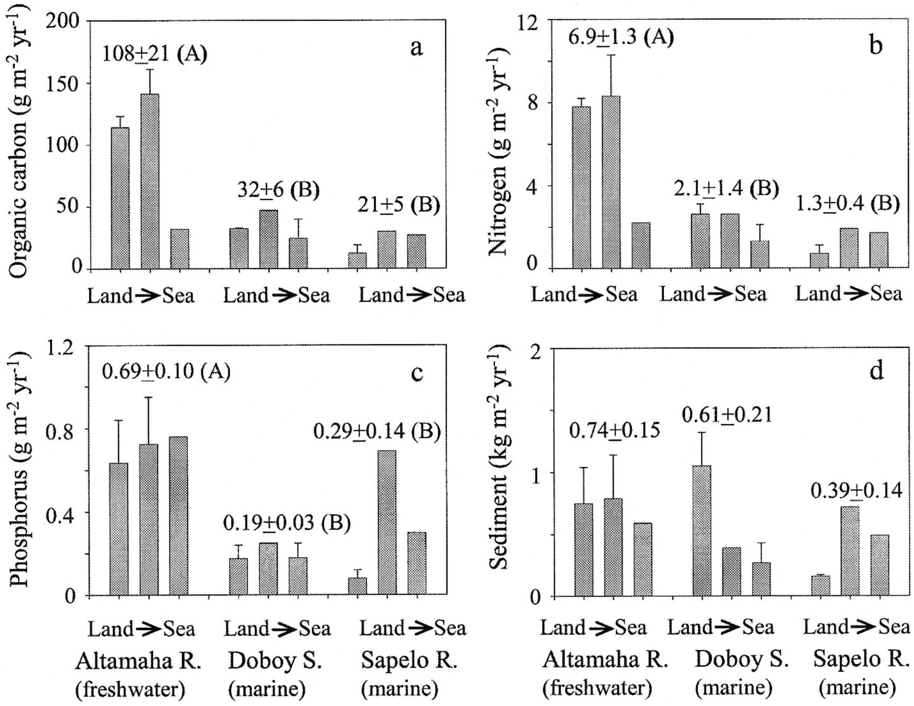

Fig. 3. [137]Cs-based measurements of (a) organic C, (b) N, (c) P, and (d) mineral sediment accumulation in freshwater- (Altamaha River) and marine- (Doboy Sound, Sapelo River) dominated marshes along the Georgia coast. Means ± 1 SE are shown for each estuary. Means separated by the same letter (in parentheses) are not significantly different ($p \leq 0.01$) based on the Ryan-Einot-Gabriel-Welsch Multiple Range Test.

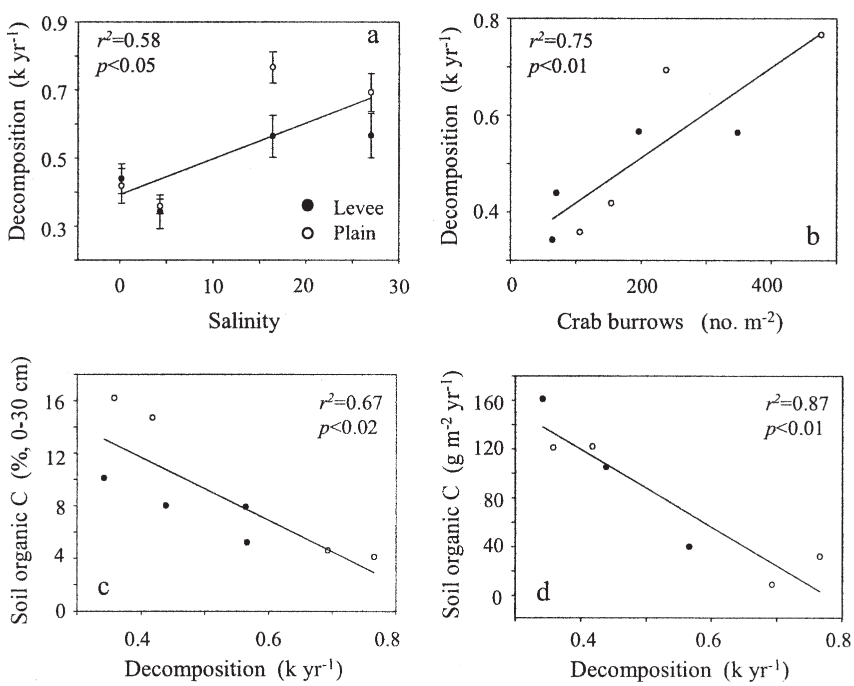

Fig. 4. Relationships between decomposition rate and (a) salinity and (b) crab burrow density, and (c) percent soil organic C and (d) organic C accumulation versus decomposition rate in four tidal marshes exposed to different salinities. Salinity is based on daily (2002 and 2003) measurements in the water column of the adjacent estuary. Burrow density is based on spring and fall 2003 counts of crab burrows (n = 4) per marsh zone per season (Dale Bishop unpubl. data).

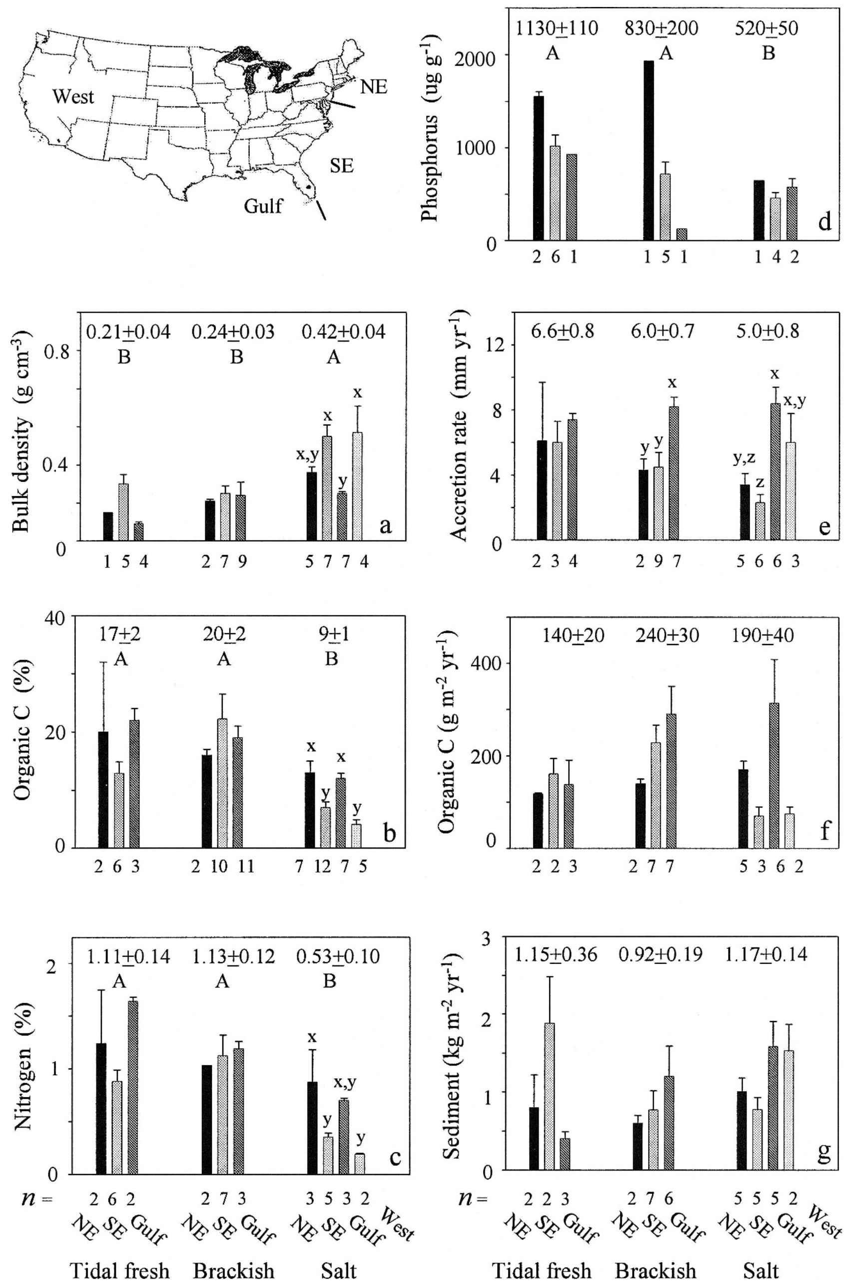

Fig. 5. Comparison of (a) bulk density, (b) percent organic C, (c) percent N, (d) P content, (e) accretion rate, (f) organic C accumulation, and (g) mineral sediment accumulation in northeast (NE) and southeast Atlantic, Gulf coast (excluding Texas salt marshes), and West coast tidal marshes of the continental United States. Bars and error bars represent the mean ± 1 SE. Means separated by the same letter (A, B for marsh type; x, y, z for region within marsh type) are not significantly different ($p < 0.05$) according to the Ryan-Einot-Gabriel-Welsch multiple range test). Data and references used to construct the graphs are presented in Web Appendix 1 (Tables A1, A2).

Soil N, which exists mostly (95%) in organic matter (Craft et al. 1991), and N accumulation also were two and three times greater, respectively, in freshwater-dominated marshes of the Altamaha River than in marshes of Doboy Sound and Sapelo River (Table 3; Fig. 3b). There was no difference in soil C : N among marshes of the three estuaries although N : P was significantly greater in marshes of the Altamaha River compared with marshes of Sapelo Sound (Table 3). Greater N : P in freshwater-dominated marshes was attributed to greater percent N rather than lesser amounts of P (Table 3).

Other soil properties—Bulk density did not differ among marshes of the three estuaries, although bulk density in the

Table 4. Correlations of soil properties with salinity for tidal marshes of Georgia and the conterminous United States.

	Georgia	Conterminous U.S.
Soil properties		
Bulk density (g cm^{-3})	0.25	0.63***
Organic C (%)	−0.70**	−0.53**
Organic C (mg cm^{-3})	−0.70**	−0.11
Nitrogen (%)	−0.73***	−0.49*
Phosphorus (μg g^{-1})	0.03	−0.49*
C : N (mol : mol)	−0.16	0.05
N : P (mol : mol)	−0.63	−0.07
Soil accumulation		
Feldspar accretion (mm yr^{-1})	−0.94**	—
^{137}Cs accretion (mm yr^{-1})	−0.69**	−0.47*
Sediment (g m^{-2} yr^{-1})	ns	−0.06
Organic C (g m^{-2} yr^{-1})	−0.79***	−0.27
Nitrogen (g m^{-2} yr^{-1})	−0.79***	−0.57*
Phosphorus (g m^{-2} yr^{-1})	−0.61**	−0.05

ns, Not significant.
* $p<0.05$.
** $p<0.01$.
*** $p<0.001$.

freshwater-dominated Altamaha River marshes was half that measured in the salt marshes (Table 3). There also was no difference in soil properties related to the mineral (nonorganic) fraction. For example, soil P (Table 3) and percent sand, silt, and clay did not differ between freshwater-dominated marshes and salt marshes (data not shown). In spite of greater vertical accretion in Altamaha River marshes (Fig. 2), there was no difference in long-term (^{137}Cs) sediment accumulation among marshes of the three estuaries (Fig. 3d). Phosphorus accumulation, however, was two to three times greater in freshwater-dominated Altamaha River marshes than marine-dominated marshes (Fig. 3c) and was attributed to greater vertical accretion in these marshes.

Comparison with tidal marsh soils of the conterminous United States—As found in Georgia marshes, tidal fresh and brackish marsh soils across all geographic regions had significantly lower bulk density and greater percent organic C and N than salt marshes (Fig. 5a–c). Tidal fresh and brackish marsh soils also contained more P than salt marshes (Fig. 5d), which contrasts with my findings of comparable P concentrations among Georgia marshes.

In marshes of the conterminous United States, vertical accretion, although not significantly different among marsh types, exhibited the same trend as observed in Georgia marshes: decreasing rate of accretion with increasing salinity (Fig. 5e). Organic C and N accumulation and N : P were greater in tidal fresh- and brackish-water marshes than in salt marshes of Georgia (Fig. 3; Table 3), but I did not see the same trends when comparing tidal marshes across geographic regions (Fig. 5f). There was no difference in sediment deposition among tidal marshes of Georgia or among marshes of different geographic regions (Fig. 5g), nor were differences noted in soil C : N (range 17–20), N : P (23–24), N accumulation (9–15 g m^{-2} yr^{-1}), or P accumulation (0.8–1.3 g m^{-2} yr^{-1}) among tidal freshwater, brackish, and salt marshes of different geographic regions.

Salt marshes were present in the four geographic regions, which allowed for comparison of soil properties among the NE and SE Atlantic, Gulf (Louisiana), and West coasts. Bulk density of Louisiana Gulf Coast salt marshes was half (0.26 g cm^{-3}), and organic C (12%) and N (0.70%) were double, those of salt marshes of the SE Atlantic and West coasts (Fig. 5a–c). Mississippi River flooding also enhanced vertical accretion of Louisiana salt and brackish marshes relative to comparable marshes in other regions (Fig. 5e). Although not significantly different among regions, organic C accumulation (Fig. 5f) also trended higher in Louisiana salt and brackish water marshes. These findings suggest that organic matter is more important to vertical accretion and long-term stability of subsidence-prone Louisiana marshes relative to marshes in other geographic regions.

Differences in soil properties also were evident between Louisiana marshes and other Gulf coast marshes. Brackish and salt marshes of the Louisiana coast contained more organic C than comparable marshes in Texas, Mississippi, and Florida. Percent organic C of Louisiana salt marshes ($12 \pm 1\%$; $n = 7$) was greater than in Texas salt marshes ($4 \pm 2\%$; $n = 2$). Also, brackish marshes of Louisiana contained more organic C ($16 \pm 2\%$; $n = 7$) than brackish marshes of Mississippi and Florida ($9 \pm 2\%$, $n = 4$).

Salt marsh soils of the SE Atlantic coast were more similar to West coast marshes than to NE Atlantic and Gulf coast salt marshes. Bulk density, organic C, and N were comparable in salt marshes of the two regions (Fig. 5–c), both of which have relatively high evapotranspiration and salinity. Salt marshes of the NE Atlantic coast have lower bulk density and high C and N relative to SE Atlantic coast salt marshes (Fig. 5a–c). These marshes are exposed to a cooler climate that probably slows decomposition and preserves soil organic C and N relative to SE Atlantic salt marshes.

In Georgia marshes and in marshes of the conterminous United States, percent soil organic C and N, vertical accretion, and N accumulation were negatively correlated with salinity (Table 4). Soil N : P and C and P accumulation also were negatively correlated with salinity in Georgia tidal marshes, but not in marshes of the conterminous United States. In U.S. tidal marshes, bulk density was positively correlated with salinity, and P concentration was negatively correlated with salinity but I did not see the same trends in Georgia marshes.

In Georgia tidal marshes and elsewhere, freshwater input promotes organic matter preservation and accumulation. In Georgia, short- and long-term accretion, percent soil organic C, N and N : P, and accumulation of organic C and N were greater in tidal marshes of the freshwater-dominated Altamaha River than in salt marshes of Doboy Sound and Sapelo River. In situ decomposition of roots was greater in salt marshes than in the tidal freshwater and brackish marshes and it was positively related to surface water salinity. Percent soil organic C and organic C accumulation were inversely related to decomposition but were unrelated to above- or belowground emergent plant production. Freshwater-driven, landscape-scale patterns of soil properties observed in Georgia tidal marshes also occur in other geographic regions of the conterminous United States. In a survey of 61 published and two unpublished studies, bulk density was lower and percent organic C, N, and P were consistently greater in tidal freshwater marshes and brackish marshes than in salt marshes regardless of geographic region. These findings suggest that freshwater input is important in structuring tidal marsh soils across a wide range of climatic and geomorphic conditions because of its association with lower decomposition rates relative to areas with greater seawater influence.

References

BARTLETT, K. B., D. S. BARTLETT, R. C. HARRISS, AND D. I. SEBACHER. 1987. Methane emissions along a salt marsh salinity gradient. Biogeochemistry **4:** 183–202.

BERTNESS, M. D. 1985. Fiddler crab regulation of *Spartina alterniflora* production on a New England salt marsh. Ecology **66:** 1042–1055.

BLUM, L. K. 1993. *Spartina alterniflora* root dynamics in a Virginia marsh. Mar. Ecol. Prog. Ser. **102:** 169–178.

BRICKER-URSO, S., S. W. NIXON, J. K. COCHRAN, D. J. HIRSCHBERG, AND C. HUNT. 1989. Accretion rates and sediment accumulation in Rhode Island salt marshes. Estuaries **12:** 300–317.

CAHOON, D. R. 1994. Recent accretion in two managed marsh impoundments in coastal Louisiana. Ecol. Appl. **4:** 166–176.

CAPONE, D. G., AND R. P. KIENE. 1988. Comparative microbial dynamics in marine and freshwater sediments: Contrasts in anaerobic carbon catabolism. Limnol. Oceanogr. **33:** 725–749.

COMANS, R. N. J., J. J. MIDDLEBURG, J. ZONDERHUIS, J. R. W. WOITTIEZ, G. J. DE LANGE, H. A. DAS, AND C. H. VAN DER WEIJDEN. 1989. Mobilization of radiocaesium in pore water of lake sediments. Nature **339:** 367–369.

CONN, C. E., AND F. P. DAY, JR. 1997. Root decomposition across a barrier island chronosequence: Litter quality and environmental controls. Plant Soil **195:** 351–365.

CRAFT, C. B., S. W. BROOME, AND E. D. SENECA. 1988. Nitrogen, phosphorus and organic carbon pools in natural and transplanted marsh soils. Estuaries **11:** 272–280.

———, ———, AND ———. 1991. Porewater chemistry of natural and created marsh soils. J. Exp. Mar. Biol. Ecol. **152:** 187–200.

———, E. D. SENECA, AND S. W. BROOME. 1993. Vertical accretion in regularly and irregularly flooded microtidal estuarine marshes. Estuar. Coast. Shelf Sci. **37:** 371–386.

———, AND OTHERS. 2003. The pace of ecosystem development of constructed *Spartina alterniflora* marshes. Ecol. Appl. **13:** 1417–1432.

DELAUNE, R. D., R. H. BAUMANN, AND J. G. GOSSELINK. 1983. Relationships among vertical accretion, coastal submergence and erosion in a Louisiana gulf coast marsh. J. Sediment. Petrol. **53:** 147–157.

———, A. JUGSUJINDI, G. W. PETERSON, AND W. H. PATRICK, JR. 2003. Impact of Mississippi River freshwater reintroduction on enhancing marsh accretionary processes in a Louisiana Estuary. Estuar. Coast. Shelf Sci. **58:** 653–662.

GEE, G. W., AND D. OR. 2002. Particle-size analysis, p. 255–293. *In* J. H. Dane and G. C. Topp [eds.], Methods of soil analysis, part 4: Physical methods. Soil Science Society of America.

HATTON, R. S., R. D. DELAUNE, AND W. H. PATRICK, JR. 1983. Sedimentation, accretion and subsidence in marshes of Barataria Basin, Louisiana. Limnol. Oceanogr. **25:** 494–502.

HEMMINGA, M. A., C. J. KOK, AND W. DE MUNCK. 1988. Decomposition of *Spartina anglica* roots and rhizomes in a salt marsh of the Westerschelde estuary. Mar. Ecol. Prog. Ser. **48:** 175–184.

KELLEY, C. A., C. S. MARTENS, AND J. P. CHANTON. 1990. Variations in sedimentary carbon remineralization rates in the White Oak River estuary, North Carolina. Limnol. Oceanogr. **35:** 373–383.

KOSTKA, J. E., B. GRIBSHOLT, E. PETRIE, D. DALTON, H. SKELTON, AND E. KRISTENSEN. 2002. The rates and pathways of carbon oxidation in bioturbated salt marsh sediments. Limnol. Oceanogr. **47:** 230–240.

MCCAFFREY, R. J., AND J. THOMSON. 1981. A record of the accumulation of sediment and trace metals in a Connecticut salt marsh. Adv. Geophys. **22:** 165–236.

MCHENRY, J. R., AND J. C. RITCHIE. 1977. Physical and chemical parameters affecting transport of ^{137}Cs in arid watersheds. Water Resour. Res. **13:** 923–927.

MELILLO, J. M., J. D. ABER, AND J. F. MURATORE. 1982. Nitrogen and lignin control of hardwood leaf litter decomposition dynamics. Ecology **63:** 621–626.

MONTAGUE, C. L. 1980. The influence of fiddler crab burrows and burrowing on metabolic processes in salt marsh sediments, p. 283–301. *In* V. S. Kennedy [ed.], Estuarine comparisons. Academic Press.

MORRIS, J. T., P. V. SUNDARESHWAR, C. T. NIETCH, B. KJERFVE, AND D. R. CAHOON. 2002. Responses of coastal wetlands to rising sea level. Ecol. **83:** 2869–2877.

NEUBAUER, S. C., K. GIVLER, S. K. VALENTINE, AND J. P. MEGONIGAL. 2005. Seasonal patterns and plant mediated controls of subsurface wetland biogeochemistry. Ecology **86:** 3334–3344.

NYMAN, J. A., R. D. DELAUNE, AND W. H. PATRICK, JR. 1990. Wetland soil formation in the rapidly subsiding Mississippi River deltaic plain: Mineral and organic matter relationships. Estuar. Coast. Shelf Sci. **31:** 57–69.

ODUM, W. E. 1988. Comparative ecology of tidal freshwater and salt marshes. Ann. Rev. Ecol. Syst. **19:** 147–176.

PALUDAN, C., AND J. T. MORRIS. 1999. Distribution and speciation of phosphorus along a salinity gradient in intertidal marsh sediments. Biogeochemistry **45:** 197–221.

PARK, G., AND OTHERS. 2004. Properties of ^{137}Cs in marine sediments off Yangnam, Korea. J. Environ. Radioact. **77:** 285–299.

SAS. 1996. SAS User's Guide. Statistical Analysis Systems Institute.

SEITZINGER, S. P. 1988. Denitrification in freshwater and coastal marine ecosystems: Ecological and geochemical significance. Limnol. Oceanogr. **33:** 702–724.

SIMPSON, R. L., R. E. GOOD, M. A. LECK, AND D. F. WHIGHAM. 1983. The ecology of freshwater tidal wetlands. BioScience **33:** 255–259.

SMITH, J. T., AND R. N. J. COMANS. 1996. Modeling the diffusive transport and remobilisation of ^{137}Cs in sediments: The effects of sorption kinetics and reversibility. Geochim. Cosmochim. Acta **60:** 995–1004.

SOMMERS, L. E., AND D. W. NELSON. 1972. Determination of total phosphorus in soils: A rapid perchloric acid digestion procedure. Soil Sci. Soc. Am. Proc. **36:** 902–904.

STEVENSON, J. C., L. G. WARD, AND M. S. KEARNEY. 1986. Vertical accretion in marshes with varying rates of sea level rise, p. 241–259. *In* D. A. Wolfe [ed.], Estuarine variability. Academic Press.

SU, C., AND C. HUH. 2002. ^{210}Pb, ^{137}Cs and $^{239,240}Pu$ in East China Sea sediments: Sources, pathways and budgets of sediments and radionuclides. Mar. Geol. **183:** 163–178.

Sundareshwar, P. V., and J. T. Morris. 1999. Phosphorus sorption characteristics of intertidal marsh sediments along an estuarine salinity gradient. Limnol. Oceanogr. **44:** 1693–1701.

Thomson, J., F. M. Dyer, and I. W. Croudace. 2002. Records of radionuclide deposition in two salt marshes in the United Kingdom with contrasting redox and accumulation conditions. Geochim. Cosmochim. Acta **66:** 1011–1023.

Weston, N. B., R. E. Dixon, and S. B. Joye. 2006. Ramifications of salinity intrusion in tidal freshwater sediments: Geochemistry and microbial pathways of organic matter mineralization. J. Geophys. Res, Biogeosci. **111,** G01009, doi:10.1029/2005JG000071.

Received: 19 June 2006
Accepted: 4 December 2006
Amended: 28 December 2006