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Sediment Accretion in Tidal Freshwater Forests and Oligohaline Marshes of the Waccamaw and Savannah Rivers, USA

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Abstract Sediment accretion was measured at four sites in varying stages of forest-to-marsh succession along a fresh-to-oligohaline gradient on the Waccamaw River and its tributary Turkey Creek (Coastal Plain watersheds, South Carolina) and the Savannah River (Piedmont watershed, South Carolina and Georgia). Sites included tidal freshwater forests, moderately salt-impacted forests at the freshwater–oligohaline transition, highly salt-impacted forests, and oligohaline marshes. Sediment accretion was measured by use of feldspar marker pads for 2.5 year; accessory information on wetland inundation, canopy litterfall, herbaceous production, and soil characteristics were also collected. Sediment accretion ranged from 4.5 mm year^{-1} at moderately salt-impacted forest on the Savannah River to 19.1 mm year^{-1} at its relict, highly salt-impacted forest downstream. Oligohaline marsh sediment accretion was 1.5–2.5 times greater than in tidal freshwater forests. Overall, there was no significant difference in accretion rate between rivers with contrasting sediment loads. Accretion was significantly higher in hollows than on hummocks in tidal freshwater forests. Organic sediment accretion was similar to autochthonous litter production at all sites, but inorganic sediment constituted the majority of accretion at both marshes and the Savannah River highly salt-impacted forest. A strong correlation between inorganic sediment accumulation and autochthonous litter production indicated a positive feedback between herbaceous plant production and allochthonous sediment deposition. The similarity in rates of sediment accretion and sea level rise in tidal freshwater forests indicates that these habitats may become permanently inundated if the rate of sea level rise increases.

Keywords Tidal freshwater wetland · Tidal freshwater forested wetland · Tidal swamp · Tidal freshwater marsh · Sediment accretion · Sea level rise · River geomorphology · Coastal geomorphology

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Introduction

At the interface between rivers and estuaries, riparian tidal wetlands provide critical ecological habitats and affect elemental transport between the world's terrestrial and marine environments. The spatial extent of tidal wetlands and their vegetation are changing as sea level rises, with the most dramatic changes occurring in the freshwater portion of the estuary (Conner et al. 2007 and references therein; Barendregt et al. 2009 and references therein). At the upstream limit of tidal influence, tidal freshwater forested wetlands dominate the riparian zone and are advancing upstream as sea level rises. At the downstream end of this tidal freshwater zone, periodic salinity intrusion and tidal inundation forces succession of riparian forests to oligohaline marshes (Yanosky et al. 1995; Krauss et al. 2009; Cormier et al. 2013). Yet sea level rise, salinity intrusion, and tidal inundation are not the only variables that influence habitat change of these tidal wetlands. Elevation change, due in part to sediment accretion, also affects the relative influence of sea level rise. Sediment accretion in tidal wetlands provides an important mechanism for reducing sensitivity of these habitats and their ecosystem functions (e.g., carbon and nutrient sequestration, sediment trapping) to future sea level rise.

The current study focuses on the accretion deficit in tidal wetlands: the difference between accretion rate and sea level

rise rate (Cahoon et al. 1995). Accretion of sediment in tidal wetlands can be broadly conceptualized in two groups of processes: those occurring within the river channel that dictate the sediment concentration in water entering the wetland, and those processes that occur within the wetland that dictate the accretion of riverine sediment and autochthonous organic matter production and degradation (Fig. 1). The degree to which wetland sediment accretion is coupled with riverine processes is a function of wetland elevation and the water level regime of the channel. Wetland elevation is the primary control on the hydrologic connectivity between rivers and wetlands (Hupp 2000) and results from biological and physical feedbacks occurring within the wetland (Morris et al. 2002; Temmerman et al. 2004; Kirwan and Murray 2007).

Rates of accretion in tidal fresh and oligohaline marshes have been examined in relation to many of the processes outlined in Fig. 1, and are reviewed by various authors in Barendregt et al. (2009; see also Neubauer 2008 and Pasternack 2009). In most of these studies, the rates of marsh accretion are generally greater than the rate of sea level rise throughout the past several decades, indicating the potential for these systems to maintain their elevation relative to rising sea level. These rates of accretion in tidal freshwater marshes were even higher than neighboring brackish and salt marshes in Georgia estuaries (Craft 2007). The existence of individual tidal freshwater marshes over long periods of sea level rise (450 to 2,000 years; Hilgartner and Brush 2006; Pasternack 2009) indicates that accretion rates in these tidal freshwater marshes are sufficient to maintain marsh elevation relative to sea level.

Sediment accretion in tidal freshwater marshes has been studied more thoroughly than in tidal freshwater forests, despite that tidal freshwater forests are more widespread in the USA (110,965 ha) than tidal freshwater marshes (81,908 ha) (Field et al. 1991; Conner et al. 2007). The few accretion rates reported for tidal forests are similar to the range reported for marshes, but long-term rates in tidal forests are lower than the local rate of sea level rise (Table 1). The higher frequency and duration of tidal inundation and lower floodplain elevation in tidal forests than in marshes downstream (Light et al. 2007; Ensign et al. 2012) suggests that accretion may be slower at the upper extent of tidal influence. However, it is difficult to assess the difference between forest and marsh accretion without simultaneous measurements in the same river.

Sediment accretion in tidal wetlands may also differ based on watershed attributes. Data from non-tidal riparian forests suggest that the watershed provenance of a river (e.g., Piedmont versus Coastal Plain physiographic province) may be a major factor influencing the rates of sediment accretion in wetlands (Hupp 2000). For example, Noe and Hupp (2005; 2009) found that tributaries of the Chesapeake Bay originating in the Coastal Plain had a mean floodplain accretion rate of 2.0 mm year^{-1}, whereas rivers originating in the Piedmont had a mean accretion rate of 2.6 mm year^{-1}. In tidal freshwater wetlands subjected to sea level rise, a difference in accretion of 0.6 mm year^{-1} could be enough to differentiate stable wetlands from those that could be permanently inundated as sea level rises. Thus, differences in sediment accretion rates due to watershed geography and geology might influence how sea level rise affects different estuaries.

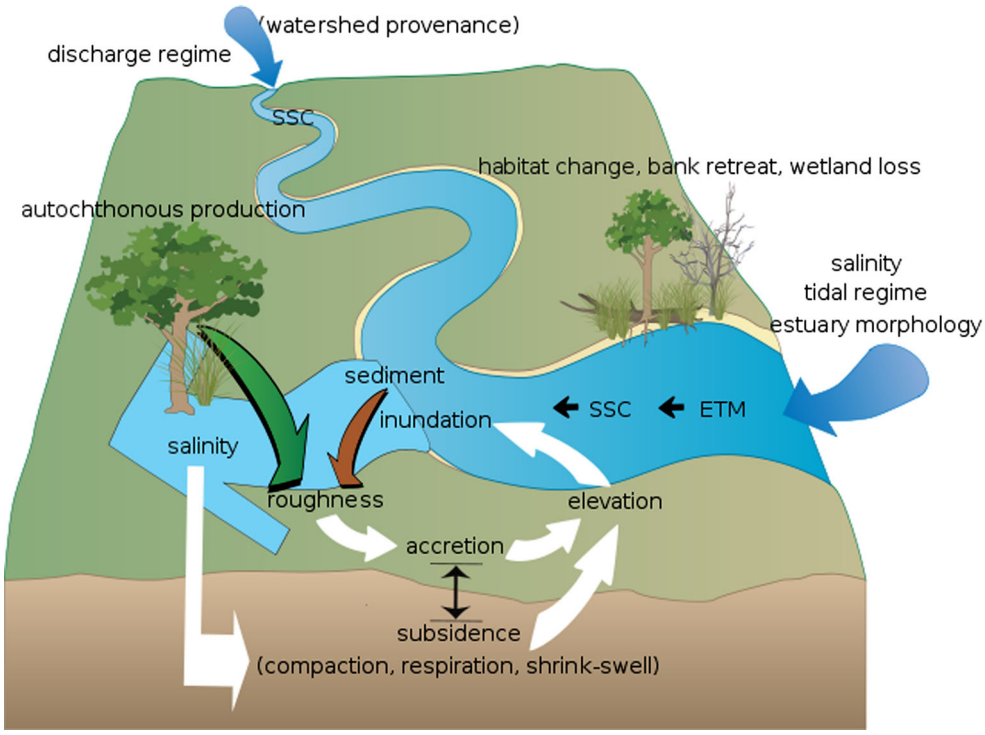

Fig. 1 Conceptual diagram of the main factors governing sediment accretion in tidal riparian wetlands and the feedbacks among these factors. *White arrows* denote feedbacks whereby biologic, biochemical, and geophysical processes influence inundation, sediment deposition, and habitat in tidal forested wetlands. Suspended sediment concentration is abbreviated *SSC*, and estuarine turbidity maximum is abbreviated ETM. Modified from the Integration and Application Network, University of Maryland Center for Environmental Science (www.ian.umces.edu/symbols/)

Table 1 Summary of sediment accretion rate measurements in tidal forests

Study site	Reference	Method	Accretion rate (mm year^{-1})	Period (years)	Local sea level rise (mm year^{-1}; trend ± SE)
Choptank River, MD[f]	Ensign et al. 2013	Feldspar	Mean=16.7, SD=9.9	1	3.52±0.2[b]
Pocomoke River, MD[f]	Ensign et al. 2013	Feldspar	Mean=8.4, SD=6.9	1	3.52±0.2[b]
Altamaha, Ogeechee, Satilla, and South Newport rivers, GA	Craft 2012	^{137}Cs ^{210}Pb	0.7–1.9 1.9–2.5	40 100	3.05±0.2[a]
Pocomoke River, MD	Kroes and Hupp 2010	Feldspar Dendro-geomorphic	3.6 1.3	4 50–100	3.52±0.2[b]
Pocomoke River, MD	Noe and Hupp 2009	Feldspar	4.2	5	3.52±0.2[b]
Nanticoke River, MD	Baldwin 2007	Feldspar	Mean=10.1, SD=3.4	1	3.52±0.2[b]
Pointe au Chene swamp, LA	Rybczyk et al. 2002	Feldspar Feldspar	7.8[c] (SE=1.1) 3.4[e] (SE=0.6)	3 6	12.3±3.4[d]

[a] 1935–1999 at Fort Pulaski, GA (Zervas 2001)

[b] 1943–1999 at Cambridge, MD (Zervas 2001)

[c] Pre-treatment period at treatment site

[d] 1962–1982 (Rybczyk 1997, as cited in Rybczyk et al. 2002)

[e] Control site

[f] Midstream tidal freshwater forested site

A dominant pattern in tidal forests is hummock-hollow topographic patterning, with hummocks existing above the high tide mark and hollows existing below mean low water (Rheinhardt and Hershner 1992; Courtwright and Findlay 2011). Hummocks support oxidized, well-drained sediment with woody and herbaceous growth while hollows consist of reduced, saturated, and more frequently inundated sediments with a higher proportion of herbaceous vegetation in many settings (Courtwright and Findlay 2011; Noe et al. 2013). More tree species associate with hummocks than with hollows (Duberstein and Conner 2009), suggesting benefits to trees positioned a mere 15–20 cm above hollow surfaces. The processes by which hummock-hollow patterning is produced and maintained are unclear, and information on sediment accretion rates in relation to this patterning could help elucidate these eco-hydro-geomorphic feedbacks.

This study investigated the spatial gradients in sediment accretion from freshwater forests to oligohaline marshes in two rivers of the southeastern USA. We tested two hypotheses; first, that sediment accretion is lower in tidal forests than tidal marshes, and second, that sediment accretion is higher in tidal wetlands adjoining Piedmont rivers than Coastal Plain rivers. We also compared sediment accretion between hummocks and hollows and measured environmental characteristics (salinity, inundation dynamics, microtopography, canopy litterfall, and herbaceous production) in four types of vegetative habitats along the tidal gradient to gain insight into the processes influencing sediment accretion.

Methods

Study Sites

Riparian wetlands along a gradient of salinity and tidal influence were studied in the Waccamaw (South Carolina, USA) and Savannah (Georgia, USA) rivers. The Waccamaw River drains a 1,981 km^2 Coastal Plain watershed, and joins the Pee Dee, Black, and Sampit rivers at Winyah Bay estuary near Georgetown, SC. An anabranch channel of the Pee Dee River also enters the Waccamaw River upstream of our study region. The low topographic gradient typical of Coastal Plain rivers such as the Waccamaw generally leads to low suspended sediment transport and extensive bottomland riparian forests (Hupp 2000). Coastal Plain rivers like the Waccamaw are also characterized by a high concentration of dissolved organic material and dark color, and are referred to as "blackwater" rivers. In contrast, the alluvial Savannah River drains a 27,390 km^2 Piedmont watershed and enters the Atlantic Ocean near Savannah, GA. Piedmont sediment sources provide suspended sediment to the Savannah River, although three dams upstream of the tidal zone have altered sediment transport to the estuary.

Four tidal sites were established along a salinity gradient of each river. On the Waccamaw River, the sites included a tidal freshwater forest above the extent of salinity in the river, a moderately salt-impacted forest, a highly salt-impacted forest on Turkey Creek (a second order tributary of the Waccamaw River near Georgetown, SC), and an oligohaline marsh on Turkey Creek (Table 2). The four sites on the Savannah River included a tidal freshwater forest, a moderately salt-impacted

Table 2 Site locations and characteristics of sediment pore water (salinity, mean, and range) and sediment (LOI and bulk density); values are the mean and range

River	Site	Latitude (° N) Longitude (° W)	Salinity	LOI[a] (%)	Bulk density[a] (g cm^{-3})
Waccamaw	Tidal freshwater forest	33.5552	0.09	51.7	0.19
		79.0897	0–0.15	41.6–68.2	0.12–0.31
Waccamaw	Moderately salt-impacted forest	33.4225	1.48	51.1	0.24
		79.2074	0.4–4.85	42.8–75.7	0.16–0.31
Waccamaw	Heavily salt-impacted forest	33.3400	2.27	43.7	0.32
		79.3421	0.93–5.35	12.6–62.4	0.15–0.65
Waccamaw	Oligohaline marsh	33.35003	3.68	34.8	0.28
		79.34491	0.68–9.45	21.9–40.3	0.19–0.38
Savannah	Tidal freshwater forest	32.2387	0.10	34.6	0.27
		81.1563	0.08–0.13	22.0–45.6	0.17–0.35
Savannah	Moderately salt-impacted forest	32.1757	1.43	46.4	0.25
		81.1442	0.35–6.85	33.3–55.9	0.17–0.43
Savannah	Heavily salt-impacted forest	32.1673	3.24	21.2	0.39
		81.1369	1.35–5.33	15.7–30.7	0.35–0.49
Savannah	Oligohaline marsh	32.1672	3.44	31.5	0.30
		81.1364	1.1–7.1	18.8–52.2	0.17–0.37

[a] Data reported in Noe et al. 2013

forest, a highly salt-impacted forest, and an oligohaline marsh (Table 2). Tidal freshwater forests were composed mainly of *Taxodium distichum* L., *Nyssa aquatica* L., *Nyssa biflora* Walt., *Fraxinus* spp, *Acer rubrum* L, and *Quercus* spp. A complete list of herbaceous species at each site is provided in Table 3. Hummock-hollow topography was only present at the tidal freshwater forests. Moderately salt-impacted forests included fewer overstory species, which were restricted to

Table 3 Species composition and net annual primary production (NAPP; ±standard error) of the herbaceous community in each study site

River	Habitat	Species	NAPP (g m^{-2} year^{-1})
Waccamaw	Tidal freshwater forest	*Cardamine hirsuta*, *Carex lupiolina*, *Cicuta maculate*, *Hydrocotyle bonariensis*, *Iris hexagona*, *Justica ovata*, *Leersia oryzoides*, *Murdannia keisak*, *Peltandra virginica*, *Pilea fontana*, *Polygonum arifolium*, *Polygonum persicaria*, *Saururus cernuus*, *Tradenum walterii*	104 (18)
Waccamaw	Moderately salt-impacted forest	*Alternanthera philoxeroides*, *Cardamine hirsuta*, *Cicuta maculata*, *Hydrocotyle bonariensis*, *Hymenocallis floridana*, *Juncus marginatus*, *Muridannia keisak*, *Peltandra virginica*, *Polygonum arifolium*, *Polygonum persicaria*, *Pontaderia cordata*, *Ptilimnium costatum*, *Schoenoplectus tabernaemontani*, *Solidago sempervirens*, *Zizania miliacea*	357 (59)
Waccamaw	Heavily salt-impacted forest	*Lilaeopsis chinensis*, *Saggitaria lancifolia*, *Schoenoplectus montevidensis*, *Scirpus robustus*, *Typha angustifolia*	430 (59)
Waccamaw	Oligohaline marsh	*Typha latifolia*, *Scirpus robustus*, *Schoenoplectus montevidensis*	1,112 (133)
Savannah	Tidal freshwater forest	*Bigonia caprilolata*, *Cardamine hirsuta*, *Clematis crispa*, *Commelina virginica*, *Cyperus* sp., *Decumaria barbara*, *Hydrocotyle bonariensis*, *Hymenocalus floridans*, *Iris haxagona*, *Peltandra virginica*, *Phanopyrum gymnocarpon*, *Polygonum arifolium*, *Ranunculus* sp., *Saururus cernuus*, *Smilax walterii*, *Toxicodendron radicans*, *Triadenum walterii*, *Viola* sp.	187 (35)
Savannah	Moderately salt-impacted forest	*Eleocharis* sp., *Iris hexagona*, *Lilaeopsis chinensis*, *Orontium aquaticum*, *Peltandra virginica*, *Phanopyrum gymnocarpon*, *Polygonum persicaria*, *Pontadera cordata*, *Sarurus cernuus*, *Scirpus robustus*, *Smilax bona-nox*, *Smilax walterii*	194 (38)
Savannah	Heavily salt-impacted forest	*Alternanthera philoxeroides*, *Polygonum glabrum*, *Schoenoplectus tabernaemontani*, *Scirpus robustus*, *Spartina cynosuroides*, *Typha angustifolia*	1,728 (231)
Savannah	Oligohaline marsh	*Carex* sp., *Carex lupoilina*, *Cicuta maculata*, *Murdannia keisak*, *Peltandra virginica*, *Pilea fontana*, *Schoenoplectus tabernaemontani*, *Spartina cynosuroides*, *Tradenum walterii*, *Typha angustifolia*	1,238 (150)

T. distichum and *N. biflora*, and the introduction of oligohaline marsh species. The highly salt-impacted forests were composed of a monoculture of living and dead *T. distichum* and an understory of oligohaline marsh plants. The marshes were composed solely of emergent herbaceous species, and were representative of healthy oligohaline marsh communities in the area. A detailed description of the forest structure at all sites as well as surficial hydrology and biogeochemistry are provided in Cormier et al. (2013), Noe et al. (2013), and Krauss et al. (2009).

Sediment Accretion and Characterization

Accretion of sediment at each site was measured along three transects perpendicular to the river channel and separated by 50–100 m. Transects ranged in length from 50 to 80 m, with five plots located equidistant along each transect, beginning 5 m from the channel. Comparison between accretion in hummocks and hollows was made by use of five paired plots in tidal freshwater forest sites in each river. At each plot, a 2-cm deep, 50-cm wide marker horizon was created by using white feldspar clay powder (Minspar 200, The Quartz Corp, Spruce Pine, NC). The clay becomes a firm, durable marker after it absorbs water, and allows accurate measurement of short-term net vertical accretion rates above the clay surface (Hupp and Bazemore 1993). Feldspar clay also accommodates fine root growth to facilitate sediment particle retention, as would occur naturally. The depth of burial of the marker horizons was determined by slicing vertically through the horizon with a knife and twisting the blade to allow insertion of a ruler to measure the depth of the clay layer. Marker horizons were installed in September 2009 and measured in March 2012. Repeated measurement of vertical accretion by multiple observers was used to develop a 95 % confidence interval of measurement precision. Surficial sediments (0–5 cm) at each site were assayed for percent organic material (loss on ignition, LOI) and bulk density (Noe et al. 2013). Twelve sediment samples from each site were dried to a constant weight at 60 °C and then combusted at 550 °C for 4 h to remove organic matter. Sediment organic fraction and bulk density were used to interpret the contribution of allochthonous mineral matter versus organic matter to sediment accretion.

Topography and Inundation

Vented pressure transducers (Infinities USA, Port Orange, Florida, USA) were installed 1-m deep in a 7.6-cm diameter PVC pipe that was backfilled with gravel around the pipe. One transducer was installed near the middle of each site in a low elevation area. Water depth was logged at 1-h intervals and filtered using a 4-h moving average to remove brief periods of inundation resulting from heavy rainfall that might pond on the wetland. The elevation of each feldspar pad was surveyed relative to the pressure transducer by means of an auto-level and rod. Water level recorded at the well was extrapolated to each pad to allow calculation of the frequency and duration of inundation (submergence) at each pad.

In addition to the water level monitoring wells, four groundwater salinity monitoring wells were installed at each site by use of screened 3.2 cm diameter PVC pipe installed to 0.6 m depth and backfilled with gravel. Two wells were sited 15 m from the channel and two others were sited 35 m from the channel. Salinity was measured monthly by using a YSI Model 30 after the well was first pumped dry and allowed to refill with soil pore water. Further details on pore water sampling are provided in Cormier et al. (2013).

Litterfall, Herbaceous Production, Temperature, and Salinity

Canopy litterfall was measured in ten, 0.25 m^2 traps placed systematically and evenly across each site; leaves, reproductive parts, and branches were collected, dried, and weighted. Traps were emptied monthly from September 2009 through December 2010; sampling was not performed in June 2010 at Waccamaw heavily salt-impacted forest and at Savannah moderately and heavily salt-impacted forests, or in December 2010 at Waccamaw tidal freshwater and moderately salt-impacted forests. Further details on canopy litterfall sampling are provided in Cormier et al. (2013).

Understory production was estimated by harvesting herbaceous plants quarterly over a period of 1 year (April 2011–April 2012) and summing the change in live and dead biomass from all sampling intervals to generate a rate of production (g m^{-2} $year^{-1}$) (Smalley 1959). Herbaceous plants were harvested along two transects that were perpendicular to the river channel, 25 m in length, and separated by 50–100 m. Five 0.25 m^2 plots were randomly located along each transect. Species within the plot were identified before harvest. All plant material was sorted into live and dead biomass, dried at 60 °C for 3–5 days and weighed.

Organic and inorganic sediment accumulation was compared with above-ground autochthonous tree and herbaceous litter production. The annual areal rate of organic sediment accumulation was calculated as the product of LOI/100, bulk density, and accretion rate. Annual areal rate of inorganic sediment accumulation was calculated as the product of 1-LOI/100, bulk density, and accretion rate. Canopy litterfall data from September 2009 through August 2010 were used to calculate an annual average rate, and this was added to the annual herbaceous production rate.

Statistical Analyses

Analysis of variance (ANOVA) was performed to test whether sediment accretion differed between wetland habitats, and

whether sediment accretion differed between rivers draining Piedmont versus Coastal Plain watersheds. An additional variable, distance from the river channel, was added to the analysis to examine its relationship to sediment accretion rate. A paired t test was used to examine the difference between sediment accretion in hummocks versus hollows. Linear regression was used to compare the relationships between sediment accretion rates, accumulation rates, and autochthonous production. Sediment accretion data were normalized by square root transformation and equality of variance was examined by use of Bartlett's test to meet the parametric assumptions. Simplification of the ANOVA models was performed by using sequential F tests ($\alpha = 0.05$) and Akaike Information Criterion (AIC) in order to reduce the number of explanatory variables and thereby increase statistical power of the tests. All statistical analyses were performed by use of R (R Core Development Team 2011).

Results

Sediment Accretion

Mean accretion rates on the Savannah River were 6 mm year^{-1} (tidal freshwater forest), 4.5 mm year^{-1} (moderately salt-impacted forest), 19.1 mm year^{-1} (heavily salt-impacted forest), and 15.2 mm year^{-1} (marsh; Fig. 2). Mean accretion rates on the Waccamaw River were 7.4 mm year^{-1} (tidal freshwater forest), 10.1 mm year^{-1} (moderately salt-impacted forest), 7.5 mm year^{-1} (heavily salt-impacted forest), and 11.4 mm year^{-1} (marsh). On the Savannah River sites, three pads at the tidal freshwater forest and one pad at the heavily salt-impacted forest were not found, and on the Waccamaw River two pads at the moderately salt-impacted forest, three pads at the heavily salt-impacted forest, and three pads at the marsh were not found. Marker pads are sometimes lost to erosion at low elevations, and therefore the mean rates for these sites may be slightly biased (12 out of 125 were not found) toward higher accretion (Cahoon and Lynch 1997).

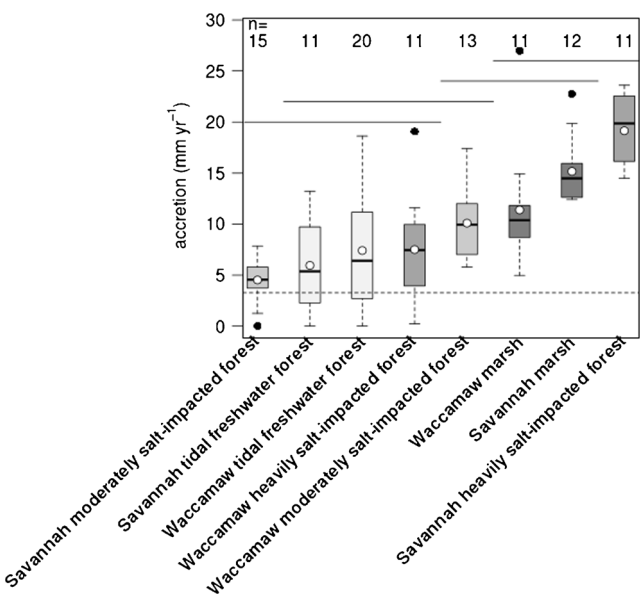

Fig. 2 Distribution of measurements of sediment accretion at each study site; *boxes* represent 25th–75th quartile, *bars* represent the median, *white dots* represent the mean, *dashed lines* represent the minimum and maximum values within 1.5 times the interquartile range, and *black circles* represent values outside 1.5 times the interquartile range. *Horizontal brackets* show groups of sites that did not differ significantly from one another by using a Tukey post hoc test. The *dashed horizontal line* indicates the rate of sea level rise from 1921 to 1999 at Charleston, SC (Zervas 2001)

Distance from the channel was not significant to the model, and exclusion of this factor resulted in a substantial decrease in the AIC of the linear model (Table 4). ANOVA of the simplified model (river × habitat) demonstrated that accretion rates did not significantly differ between the two rivers, but that habitat (the salinity gradient) and the interaction between river and habitat were significant factors. A Tukey post hoc test of accretion differences between each river by habitat combination indicated that accretion in the marshes on both rivers was greater than the tidal freshwater forests on both rivers (Fig. 2). While accretion was greater in the Waccamaw's moderately salt-impacted forest than the Savannah's moderately salt-impacted forest, accretion in the Savannah's heavily salt-impacted forest was higher than the Waccamaw's heavily salt-impacted forest. Accretion in the Savannah's heavily salt-impacted forest was similar to the marshes but greater than all other sites. Within the tidal freshwater forests of both rivers, sediment accretion was significantly higher in hollows (mean = 9 mm year^{-1}) than hummocks (mean = 3 mm year^{-1}; paired t test on square root-transformed accretion rates, $t = -2.6$, df = 9, $p = 0.03$; Fig. 3). The average 95 % confidence interval of measurements made by four individuals measuring 17 pads was 2.6 mm year^{-1}.

Inundation

The tidal freshwater forests had a similar distribution of inundation events and periods of inundation in both rivers, although variability was higher in the Waccamaw (Fig. 4). The moderately salt-impacted forest of the Savannah had a higher number of inundation events than the Waccamaw (although metrics could be calculated for only two sites on the Savannah due to their elevation relative to the water level meters), but the rivers had a similar range in period of inundation. The heavily salt-impacted forest and marsh of the Waccamaw had more inundation events and a longer period of inundation than the same habitats on the Savannah. In the Waccamaw River there was a general increase in variance in accretion rates with increasing

Table 4 Summary of ANOVA examining the influence of river (Waccamaw and Savannah), habitat (tidal freshwater forest, moderately salt-impacted forest, heavily salt-impacted forest, and marsh), and distance from the river channel on the square root of sediment accretion (mm year^{-1})

Model	Term	df	*F*	*P*	AIC
Accretion$^{0.5}$ ~ river × habitat × distance	River	1	0.508	0.478	
	Habitat	3	14.8	<0.001	
	Distance	4	1.59	0.187	320
	River × habitat	3	12.7	<0.001	
	River × distance	4	0.766	0.551	
	Habitat × distance	11	0.773	0.666	
	River × habitat × distance	10	1.09	0.377	
Accretion$^{0.5}$ ~ river × habitat	River	1	0.509	0.477	
	Habitat	3	14.9	<0.001	298
	River × habitat	3	12.8	<0.001	

number of flood events and the percent of time inundated. This inter-site pattern was not apparent at the Savannah River sites. There was not an apparent increase in accretion rate with either metric of inundation on an intra-site basis in either river.

Due to the placement of some pads lower in elevation than the water level pressure sensor, inundation could not be quantified for all plots. Presumably, these lower elevation plots were inundated more frequently and for longer periods. Inundation was calculated for the majority of plots at the Waccamaw moderately salt-impacted forest and marsh, and the Savannah tidal freshwater forest, heavily salt-impacted forest, and marsh. At the Waccamaw tidal freshwater forests and heavily salt-impacted forest, and Savannah moderately salt-impacted forest, inundation dynamics could only be calculated for a minority of plots, and accretion rates at these plots were generally lower than rates at the remaining plots. This indicates that the lower elevation, more frequently inundated plots had higher rates of accretion. In summary, our measurements of pad inundation are biased towards the higher elevation, less frequently inundated plots in each site, and do not represent the full range of inundation dynamics at the sites.

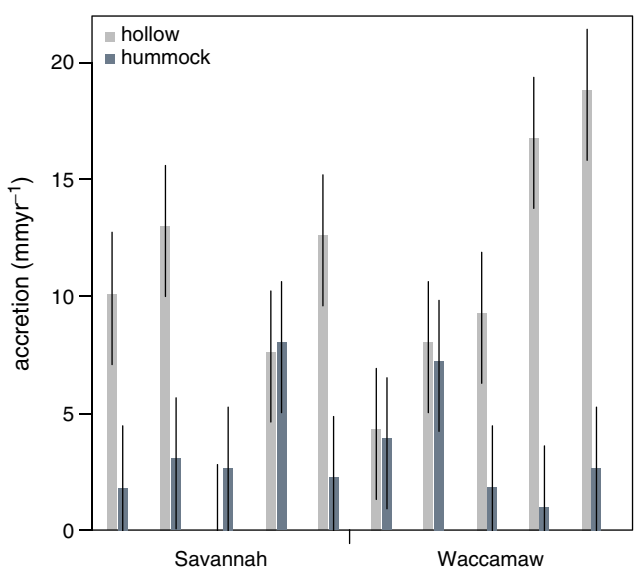

Fig. 3 Comparison of sediment accretion rates at paired hummock-hollows at five plots in the Savannah and Waccamaw rivers. *Lines* indicate the 95 % confidence interval of measurement precision

Litterfall and Herbaceous Production

Mean canopy litterfall during the study period was greatest at the tidal freshwater forests on the Waccamaw and Savannah rivers (324 and 281 g m^{-2}, respectively), followed by the moderately salt-impacted forests (139 and 172 g m^{-2}) and heavily salt-impacted forests (51 and 42 g m^{-2}; Fig. 5a). There was not a significant linear relationship between accretion rate and canopy litterfall (Fig. 5a). Conversely, herbaceous aboveground production was lowest at the tidal freshwater forests on the Waccamaw and Savannah rivers (104 and 187 g m^{-2}, respectively) and generally increased along the salinity gradient and was significantly related to accretion rate (Fig. 5b).

The rates of inorganic sediment accumulation were significantly related to autochthonous production at each site (canopy litterfall + herbaceous production), while organic sediment accumulation was less strongly related to autochthonous production (Fig. 5). The regression between organic sediment accumulation and autochthonous production was close to the 1:1 line, indicating that some allochthonous organic matter input from the river balanced in situ decomposition of autochthonous organic matter and export through erosion. This analysis revealed that the apparent relationship between sediment accumulation and herbaceous production (Fig. 5b) was only partly due to the accumulation of herbaceous plant litter. A much larger proportion of the accumulation in the marshes and lower forest was due to inorganic sediment.

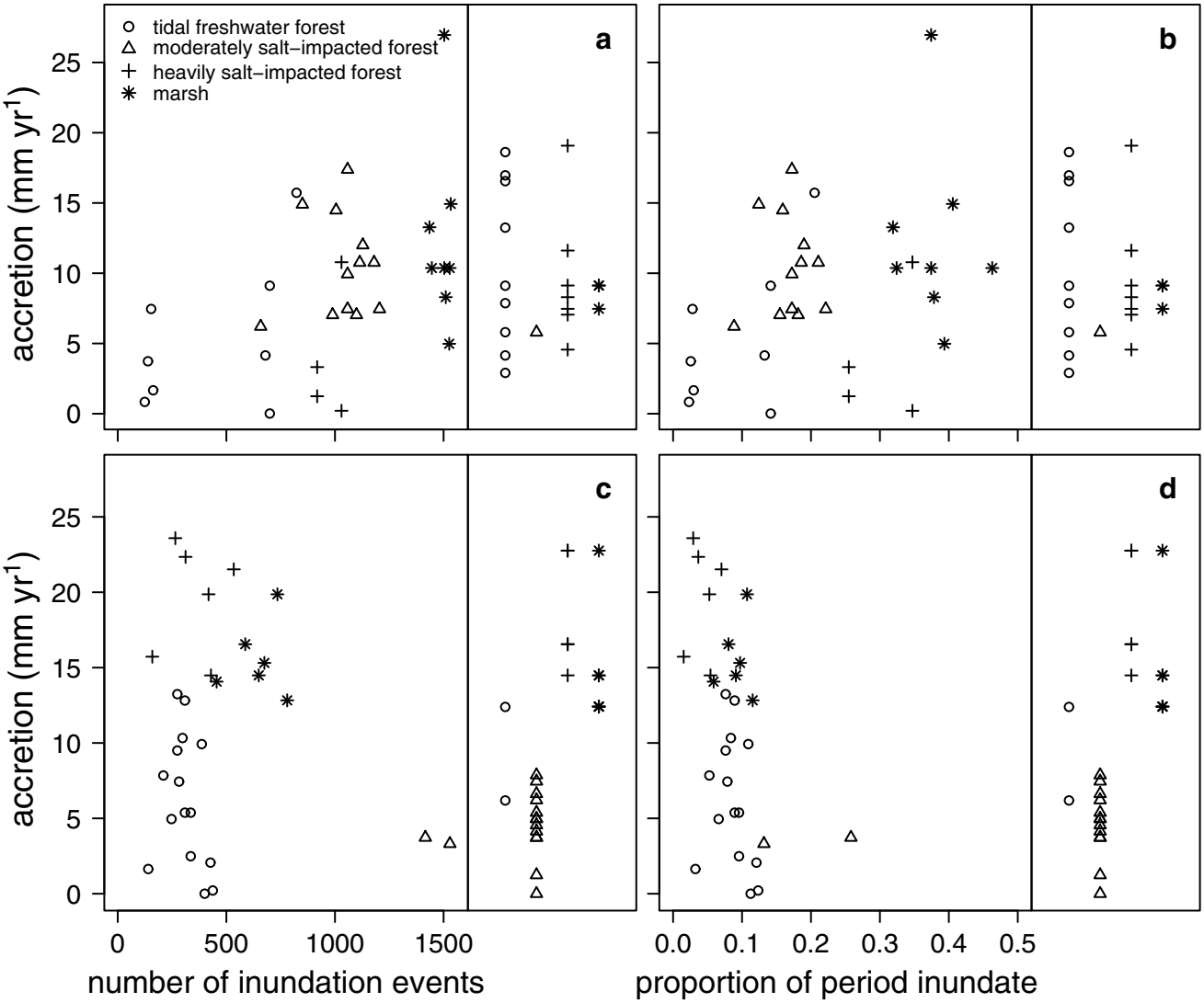

Fig. 4 The number of inundation events that occurred over the study period (**a**, **c**) and the proportion of the time inundated during the study period (**b**, **d**) for plots on the Waccamaw (**a** and **b**) and Savannah rivers (**c** and **d**) in relation to sediment accretion. The adjoining plots in each panel show accretion rates at plots for which inundation dynamics could not be calculated because they were lower in elevation than the monitoring well, but are assumed to be the wettest plots within each site; the *x*-axis in these side panels is categorized by site

Soil and Porewater Characteristics

Organic matter (LOI) was greatest at the Waccamaw tidal freshwater forest and moderately salt-impacted forests and lowest at the Savannah heavily salt-impacted forest (Table 2). The Waccamaw sites had greater mean LOI than the Savannah sites. Mean bulk density was greatest at the Savannah heavily salt-impacted forest and lowest at the Waccamaw tidal freshwater and moderately salt-impacted forests. Mean bulk density was greater at all Savannah sites than their corresponding Waccamaw sites.

Porewater salinity increased from the forests through marshes on both rivers (Table 2). Salinity at the tidal freshwater forests never exceeded 0.5, indicating that no brackish water intruded into those sites during the study period. All other sites exhibited oligohaline conditions (0.5–5), except for pulses of up to seven to nine at the marshes. Salinity at the Savannah moderately salt-impacted forest (mean= 1.4) was lower than at the heavily salt-impacted forest (mean=3.2) despite the proximity of these sites to one another, indicating that the moderately salt-impacted forest has greater subsurface freshwater storage capacity or that freshwater storage in adjoining tributary creeks alters the salinity of water entering this wetland.

Discussion

Sediment Accretion in Tidal Freshwater Versus Oligohaline Wetlands

We found that oligohaline marshes in the Waccamaw and Savannah rivers are accreting sediment at rates 1.5–2.5 times

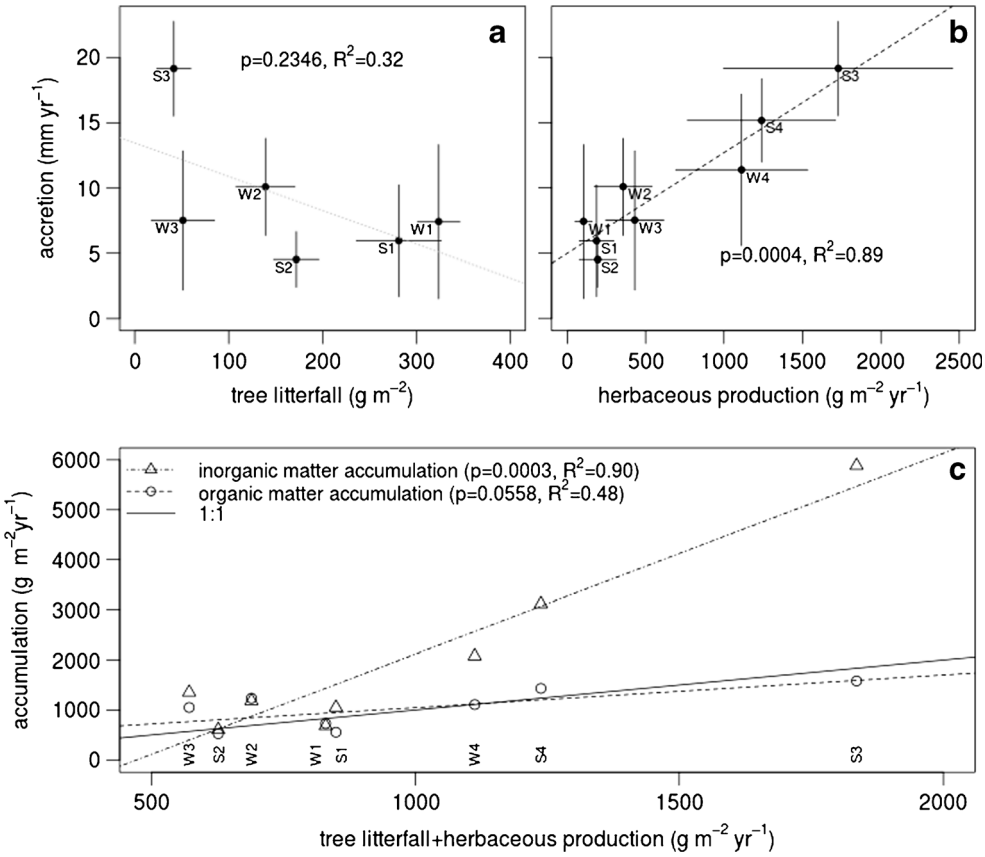

Fig. 5 Mean accretion (September 2009 to March 2012) at a site versus mean canopy litterfall (September 2009 to December 2011) (**a**), mean above ground herbaceous production (**b**), and sediment accretion versus mean annual canopy litterfall + herbaceous production (annual canopy litterfall rate was calculated for the period September 2009 through August 2010 to be concurrent with herbaceous production estimates) (**c**). *Bars* indicate standard deviation. *S* Savannah, *W* Waccamaw, *1* tidal freshwater forest, *2* moderately salt-impacted forest, *3* heavily salt-impacted forest, *4* marsh

higher than the tidal freshwater forests upstream. Sediment accretion at the tidal freshwater and salt-impacted forests were broadly similar to the range of values found in other tidal freshwater forests (Table 1) and non-tidal bottomland forests near the Mississippi River (6–9 mm year^{-1}, Conner and Day 1988). However, rates in this study were generally greater than in non-tidal riparian forests in Chesapeake Bay tributaries (0.2 to 7.2 mm year^{-1}, Ross et al. 2004; 0.4 to 4.8 mm year^{-1}, Noe and Hupp 2009), and Coastal Plain rivers in general (1.5 to 5.4 mm year^{-1}, Hupp 2000).

Our study and most previous studies of accretion in tidal forests (Table 1; except Kroes and Hupp 2010 and Craft 2012) are based on short-term measurements which are generally higher than longer-term measurements (Craft 2007). Short-term measurements may be higher because they do not account for shallow subsidence within the upper several meters of the sediment, and therefore the actual rate of elevation change in these wetlands may be different than the measured accretion rates. Soil compaction, decomposition, root decay, and shrink-swell processes can reduce the elevation of wetlands, and belowground plant production can increase the elevation of wetlands. For example, a mangrove forest in Florida with similar bulk density and soil organic matter as the Waccamaw tidal freshwater forest did not exhibit any subsidence, whereas mangrove sites with bulk density and organic matter similar to the other sites in our study exhibited subsidence rates that were 60–86 % of the accretion rates (Cahoon and Lynch 1997). Even higher rates of subsidence (exceeding 100 % of accretion) were found in salt marshes in Louisiana, Florida, and North Carolina (Cahoon et al. 1995) and riverine mangroves in Micronesia (Krauss et al. 2010).

Accretion rates in some portions of the forested wetlands of both rivers were less than the rate of sea level rise; however, accretion deficit alone is not sufficient for rating sea level rise susceptibility of coastal wetlands (Webb et al. 2013). Sea level at Charleston, SC (roughly equidistant from the Waccamaw and Savannah rivers) has risen at 3.15 mm year^{-1} between 1921 and 2006 (NOAA, Sea Level Trends, http://tidesandcurrents.noaa.gov/sltrends/sltrends_station.shtml?stnid=8665530, accessed 4 November 2013). Given the low bulk density and high organic matter content of the tidal freshwater forests (mean accretion rate of 6.0 to 7.4 mm year^{-1}), the rate of shallow subsidence is likely fairly high (Cahoon and Lynch 1997). Factoring in even a modest subsidence rate of 50 % of accretion (the minimum reported in Cahoon and Lynch 1997), along with measured accretion rates in this study, would reduce the mean rate of accretion to be (at most) 0.56 mm year^{-1} faster than the historic rate of sea level rise at Charleston, SC. We predict that the forested wetlands in the Savannah and Waccamaw rivers are

very susceptible to submergence if the rate of sea level rise increases, a fate also observed for tidal freshwater forests in Georgia by Craft (2012).

Tidal Wetland Sediment Accretion in Piedmont Versus Coastal Plain Rivers

We expected to find differences in accretion rate between a Piedmont (Savannah) and Coastal Plain (Waccamaw) river, but there was no overall difference between these systems. The only significant difference in accretion rate between the rivers involved an interaction between river and habitat type. Our expectations were based on previous comparisons in accretion rate between Coastal Plain and Piedmont rivers (Noe and Hupp 2009), and the corresponding higher sediment load in Piedmont rivers (Hupp 2000). The predicted sediment yield from the Savannah watershed (44 t km^{-2} $year^{-1}$, McCarney-Castle et al. 2010) is far greater than the Waccamaw watershed sediment yield of 14 t km^{-2} $year^{-1}$ (Phillips 1991). These differences occur despite the fact that reservoir trapping following dam construction on the Savannah River was predicted to reduce the sediment load by 69 % (McCarney-Castle et al. 2010). The lower LOI and higher bulk density of floodplain sediments on the Savannah than Waccamaw further highlight the difference in suspended sediment characteristics between the rivers, with the Savannah having a greater mineral versus organic content.

Our inability to detect a difference in accretion rate between the Waccamaw and Savannah rivers may be in part due to three factors which we did not measure. First, sediment load in the portion of the Coastal Plain Waccamaw River we studied is influenced by an anabranch channel of the neighboring Pee Dee River (draining a Piedmont watershed). Thus, our sites on the Coastal Plain, Waccamaw River have a Piedmont sediment source that may have increased the similarity in sediment accretion between the Waccamaw and Savannah. Second, both the Waccamaw (Goñi et al. 2009) and Savannah rivers (Mulholland and Olsen 1992) exhibit tidal flood-dominant sediment transport in their saline reaches, meaning that there is a net upstream movement of sediment from the lower estuary to the upper estuary. Flood-dominant sediment transport is a common condition in many estuaries (Meade 1969), but it is not clear how far upstream this phenomenon extends relative to our study sites and to what extent this process contributes to over-bank deposition of material on the floodplain. Third, the estuarine turbidity maximum (ETM) likely occurs within the oligohaline-to-mesohaline portion of both rivers and may influence in-channel suspended sediment and subsequent overbank deposition. Previous studies of sediment accretion in tidal wetlands have attributed higher floodplain accretion to proximity to the ETM (Darke and Megonigal 2003), and this phenomenon may over-shadow differences in suspended sediment concentration that result from watershed origin.

Relative to non-tidal riparian wetlands, tidal riparian wetlands appear to be buffered from watershed-derived sediment and concomitant riparian sediment deposition. With up to 100 % of the watershed-derived sediment load potentially trapped by non-tidal riparian floodplains (Noe and Hupp 2009), relatively little sediment may be transported through the river network to the tidal zone. In the Pee Dee River, a Piedmont river that joins the Waccamaw near our study sites, only 4 % of the watershed-derived sediment was estimated to reach the estuary (Phillips 1992). The Cape Fear River, a tidal river draining a Piedmont watershed, has little inorganic sediment of Piedmont mineralogy in its floodplain, with the majority of the floodplain sediment originating from the lower Coastal Plain (Benedetti et al. 2006). There are circumstances when significant watershed-derived sediment does reach the tidal zone, such as during intensive agricultural land clearing in the 1800s and more recent silvicultural activity (Hilgartner and Brush 2006), and during extreme river discharge events (Ensign et al. 2013). Future research is needed to identify thresholds in watershed and stream network size, river transport capacity, and floodplain accommodation space that govern the geomorphic connectivity between watersheds and tidal rivers.

Processes Governing Sediment Accretion in Tidal Wetlands

Processes occurring at the habitat scale (the distance separating our study sites, 1–10 km) were the major factors affecting accretion rate. This is best exemplified by the contrast in accretion rates occurring on the Savannah River where the maximum and minimum accretion rates observed in the study occurred between two sites less than 2 km apart. This contrast may be due to the large difference in autochthonous production between these two sites. While autochthonous litter production at the Savannah heavily salt-impacted forest was the highest in the study, organic sediment at that site constituted a small proportion of overall accretion (21 %). In contrast, autochthonous litter production at the Savannah moderately salt-impacted forest was the second lowest in the study but organic sediment at this site constituted 46 % of overall accretion. The higher stem and litter density at the heavily salt-impacted forests may have indirectly contributed to sediment accretion by increasing surface roughness of the floodplain which enhanced deposition of inorganic suspended sediment during tidal inundation (Leonard 1997; Pasternack and Brush 2002).

Inorganic sediment was primarily responsible for the difference in sediment accretion between forest and marsh (Fig. 5c). This indicates greater inorganic suspended sediment concentration (SSC) in water inundating the marshes or more frequent inundation and subsequent deposition. Previous studies have identified sediment concentration of in-flowing water as a determinant of marsh accretion rate (Temmerman et al.

2004), although differences in channel SSC were not measured in this study. We speculate that the oligohaline marshes (and adjoining Savannah heavily salt-impacted forest) were closer to the ETM and this zone of both rivers had flood-dominant sediment transport occurring (Meade 1969) and therefore inorganic SSC was greater at these sites.

We did not find evidence that frequency or duration of inundation were related to accretion of either inorganic or total material. The site with the greatest number of inundation events and longest period of inundation (Savannah moderately salt-impacted forest) was also the site with negligible inorganic sediment deposition. Furthermore, no measure of inundation dynamics exhibited any obvious intra-site relationship with accretion rate. We also investigated the relationship between proximity to the channel and accretion, expecting to find higher accretion at plots closer to the channel and exposed to higher suspended sediment concentration based on previous research in tidal forested wetlands (Ross et al. 2004; Kroes et al. 2007), non-tidal coastal plain riparian wetlands (Hupp 2000), and tidal freshwater marshes (Palinkas et al. 2013). However, proximity of plots to the channel was not a significant factor in the ANOVA model. In reconciling these results with our expectations, it should be emphasized that we measured net accretion. While plots closer to the river may be inundated more frequently and with higher sediment concentration, they are also exposed to potentially higher flow velocity and greater erosion.

There was significantly greater sediment accretion in hollows than hummocks. More anaerobic soils (Noe et al. 2013) and subsequently slower decomposition in hollows than hummocks (Courtwright and Findlay 2011) may explain the differences in sediment accretion we observed. Alternatively, erosion of sediment from higher elevation (hummocks) to lower elevation (hollows) may contribute to higher sediment accretion in hollows, but we expect this redistribution process is counteracted by other mechanisms involving feedbacks between vegetation and biogeochemistry that result in the long-term maintenance of this topographic pattern (Noe 2013). Since greater inputs of new sediment to hollows are opposite to the processes necessary to generate higher hummocks and lower hollows, more information is needed on the factors affecting both shallow subsidence as well as root growth to explain the formation and maintenance of this topographic pattern in tidal freshwater forests.

Feedbacks and Morphodynamics of Tidal Rivers

Feedbacks in ecological and geomorphic systems play a critical role in the evolution of landforms (Murray et al. 2008; Noe 2013). Data from this study provide insight on one potential feedback highlighted in our conceptual model (Fig. 1). Succession of forest to marsh occurs at salinity of ~2, and results in significant declines in canopy litterfall production, lower tree basal area increments, lower tree densities, and a greater number of dead trees (Yanosky et al. 1995; Krauss et al. 2009; Cormier et al. 2013). Simultaneous increases in herbaceous marsh production (Chabreck and Linscombe 1982; McKee and Mendelssohn 1989; this study) and roughness reduce surface water flow velocity and result in higher sediment settling rates (Leonard 1997) compared to the freshwater forests with low ground cover. This influx of inorganic sediment and sorbed nitrogen and phosphorus may stimulate marsh production, organic matter accretion, and subsequent inorganic sediment deposition in a positive feedback on wetland elevation. A similar feedback mechanism has been identified in models of salt marsh elevation change (Morris et al. 2002; Kirwan and Murray 2007).

In contrast, salinity may also contribute to a positive biogeochemical feedback on subsidence in tidal forests. The contribution of sulfate to soil porewater can increase microbial respiration (Neubauer 2013), contributing to shallow subsidence of wetland sediments (Cahoon and Lynch 1997). Tree mortality associated with salinification of tidal forests also stimulates soil nitrogen mineralization and likely enhances shallow subsidence (Noe et al. 2013). As subsidence occurs and inundation frequency increases, increased soil redox cycling (Ensign et al. 2008) drives further mineralization, subsidence, and expansion of saline water into tidal wetlands.

These feedbacks between salinity and wetland elevation occur at the scale of a wetland plant community, marshes in the first example and forests in the second, where saltwater intrusion is an independent factor. At broader spatial and temporal scales salinity intrusion up-river into the tidal freshwater rivers is dependent on river morphology. The volume of the channel and intertidal zone, together with the tidal dynamics and freshwater river discharge, govern the longitudinal salinity profile of an estuary. As tidal wetlands are submerged and shorelines erode as sea level rises, volume of the estuary and salinity intrusion increase. Therefore, feedbacks between salinity and wetland elevation at the habitat level may influence longer-term feedbacks between salinity and estuarine morphology. The low rate of sediment accretion in the tidal freshwater forests relative to sea level rise may result in gradually larger intertidal volume and propagation of salinity upstream. Quantification of these feedbacks through field data, mathematical modeling, and experimentation is needed to improve existing models (e.g., Brinson et al. 1995) of tidal landscape evolution.

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