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Declining Sediments and Rising Seas: an Unfortunate Convergence for Tidal Wetlands

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Abstract The availability of suspended sediments will be a dominant factor influencing the stability of tidal wetlands as sea levels rise. Watershed-derived sediments are a critical source of material supporting accretion in many tidal wetlands, and recent declines in wetland extent in several large river delta systems have been attributed in part to declines in sediment delivery. Little attention has been given, however, to changes in sediment supply outside of large river deltas. In this study, significant declines in suspended sediment concentrations (SSCs) over time were observed for 25 of 61 rivers examined that drain to the East and Gulf Coasts of the USA. Declines in fluvial SSC were significantly correlated with increasing water retention behind dams, indicating that human activities play a role in declining sediment delivery. There was a regional pattern to changes in fluvial sediment, and declines in SSCs were also significantly related to rates of relative sea level rise (RSLR) along the coast, such that wetlands experiencing greater RSLR also tend to be receiving less fluvial sediment. Tidal wetlands in the Mid-Atlantic, Mississippi River Delta, and Texas Gulf especially may become increasingly vulnerable due to rapid RSLR and reductions in sediment. These results also indicate that past rates of marsh accretion may not be indicative of potential future accretion due to changes in sediment availability. Declining watershed sediment delivery to the coastal zone will limit the ability of tidal marshes to keep pace with rising sea levels in some coastal systems.

Keywords Suspended sediment · Tidal wetlands · Dams · Land use change · Accretion · Sea level rise

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Introduction

Tidal wetlands are highly productive ecosystems that provide important ecosystem services (Barbier et al. 2012), but are under increasing threat from climate change (Reed 1995; Morris et al., 2002) and land use change (Blum and Roberts 2009; Syvitski et al. 2009; Deegan et al. 2012). Tidal wetlands maintain surface elevation at or close to mean sea level through multiple, often nonlinear ecogeomorphic feedbacks between sediment deposition, plant production, organic matter sequestration, and inundation (Morris et al. 2002; Mudd et al. 2009; D'Alpaos et al. 2011; Fagherazzi et al. 2012). The availability of suspended sediment has been identified as one of the major factors influencing wetland development, geomorphology, and response to relative sea level rise (RSLR; Kirwan and Murray 2007; Day et al. 2011; Mudd 2011; Fagherazzi et al. 2012). The water flooding tidal wetlands carries suspended sediment that, when flow slows as the marsh floods, can be deposited on the wetland surface promoting vertical accretion. Increased flooding depth and duration, as would accompany sea level rise, provides greater opportunity for sediment deposition (Pethick 1981; Krone 1985). In addition, stable salt marshes are generally situated supraoptimal to sea level with respect to plant growth, such that as sea level rises, plant biomass increases (Morris et al. 2002). Plant biomass plays an important role in slowing flood waters and promoting sediment deposition (Gleason et al. 1979), as well as supporting organic matter sequestration in wetland soils (Reed 1995; Morris et al. 2002). As sea levels rise, increased trapping of sediments from tidal waters and sequestration of organic material together produce feedbacks that promote vertical accretion and may allow the marsh to keep pace with sea level (Morris et al. 2002; Mudd et al. 2009; D'Alpaos et al. 2011; Fagherazzi et al. 2012).

The maximum rate of RSLR that a wetland can endure is thought to largely be a function of suspended sediment availability in tidal flood waters (Kirwan and Murray 2007;

Day et al. 2011; Mudd 2011; Fagherazzi et al. 2012). Tidal wetlands that exist in coastal systems with low suspended sediment concentrations will be less resilient to increased rates of RSLR (Fig. 1; Kirwan et al. 2010; Fagherazzi et al. 2012). Kirwan and Murry (2007) and others have proposed a relationship between wetland stability, RSLR, and suspended sediment concentration (SSC) where wetland stability is maintained by low rates of RSLR and high SSCs, but wetlands become unstable as rates of RSLR increase and/or SSCs decline (Fig. 1). Recent increases in sea level rise (SLR) (Church and White 2006) and future projections of SLR acceleration (Vermeer and Rahmstorf 2009) have raised concerns about the resiliency of tidal wetlands under future scenarios of climate change (Reed 1995; Morris et al. 2002). As SLR accelerates, the availability of suspended sediment may take on increased importance for sustaining coastal wetlands.

Fluvial delivery of watershed-derived sediments can be an important source of material to coastal wetlands, particularly in tidal freshwater, estuarine, and deltaic wetland systems. While landscape geomorphology largely controls fluvial sediment supply from rivers to the coastal zone (Syvitski and Milliman 2007), human activities can also influence the delivery of sediments from the landscape to rivers that drain to the coast zone (Walling 1999; Syvitski et al. 2005; Syvitski and Milliman 2007; Walling 2006; Saenger et al. 2008). In the Eastern USA, for instance, European colonization resulted in large-scale deforestation and a shift to agricultural land use which is thought to have substantially increased sediment delivery to aquatic systems (Fig. 1; Howarth 1991; Saenger et al. 2008). Indeed, the enhanced delivery of sediment to coastal systems in the East and Gulf Coasts in the eighteenth and nineteenth centuries may have created large expanses of tidal wetlands that did not exist under the pre-settlement sediment delivery regime (Kirwan et al. 2011; Tweel and Turner 2012; Jaffe et al. 2007).

Historical increases in sediment delivery have likely been followed by more recent declines in fluvial sediment transport in many U.S. East and Gulf Coast rivers caused by a shift away from agricultural land use, soil conservation efforts, afforestation, and the creation of artificial reservoirs behind dams (Vörösmarty et al. 2003; Syvitski et al. 2005; Walling 2006; Syvitski and Milliman 2007; Kirwan et al. 2011; Tweel and Turner 2012). A dominant factor in recent reductions in sediment delivery from global rivers to the coastal zone is the damming of rivers (Vörösmarty et al. 2003; Syvitski et al. 2005; Walling 2006). Vörösmarty et al. (2003) estimate that greater than 25 % of the global sediment flux is trapped in reservoirs, with nearly 100 % retention in highly regulated basins such as the Nile and Rio Grande. Climate change may also influence sediment delivery from the landscape to the coastal zone, though these changes will likely be highly regional (Day et al. 2008).

Reductions in sediment supplies have been linked to loss of wetlands in, for instance, the Mississippi River Delta (Blum and Roberts 2009; Tweel and Turner 2012; Day et al. 2011), San Francisco Bay (Jaffe et al. 2007), and the Ebro Delta (Guillén and Palanques 1997). Reduced sediment supplies are a major cause for subsidence and wetland loss in large river deltas worldwide (Syvitski et al. 2009). For instance, substantial wetland area was created in the Mississippi River delta prior to 1930 coinciding with a shift to intensive agricultural land use in the watershed, followed by subsequent loss of wetland area as suspended sediments declined (Tweel and Turner 2012). Blum and Roberts (2009) estimate that 25 % of the Mississippi River delta has been lost due to inadequate sediment supply, delta subsidence, and SLR. Similarly, Jaffe et al. (2007) documented an increase in

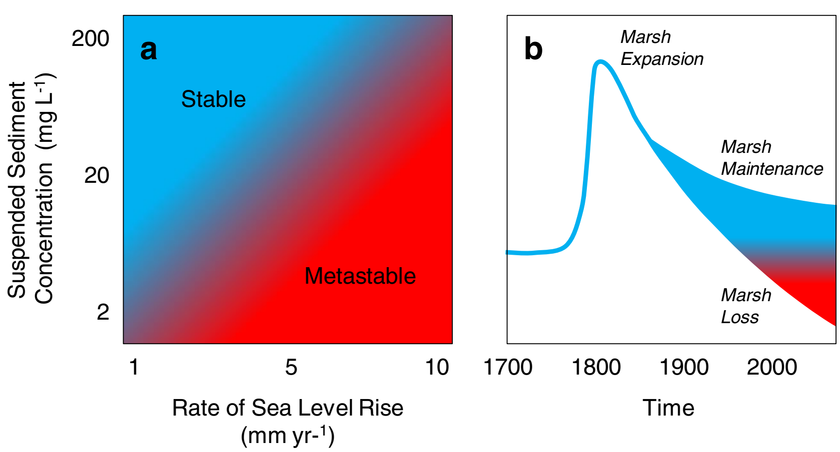

Fig. 1 **a** Conceptual model of tidal marsh stability as a function of suspended sediment concentration and rate of sea level rise (modified from Kirwan and Murray 2007) and **b** conceptual diagram of changes in suspended sediment concentration in a U.S. river over time from pre-development concentrations, increased sediment concentrations following deforestation and agricultural use of the landscape, and subsequent decline in sediment during afforestation, soil conservation, and damming in the watershed

sediment delivery and wetland expansion in San Francisco Bay in the 1800s during the gold mining period, followed by declining riverine sediment delivery, erosion of sediments from the bay, and wetland loss in the 1900s. Following rapid marsh expansion due to human-induced sediment delivery during the eighteenth and nineteenth centuries along the East Coast, Kirwan et al. (2011) suggest that many marshes currently exist in a metastable condition in which they are able to survive but not expand under modern, relatively low sediment concentrations (Fig. 1).

While large delta systems have received attention, less consideration has been given to the role of changing sediment supply on tidal wetlands in smaller coastal estuarine systems. Substantial wetland area is found outside of large delta systems, and fluvial sediment supply may play an important role in structuring future wetland geomorphology in these ecosystems, given the recent (Church and White 2006) and projected acceleration (Vermeer and Rahmstorf 2009) of global SLR. Further, there is no published analysis of recent changes in fluvial sediment delivery along the East and Gulf Coasts of the USA, which is needed for a comprehensive understanding of wetland vulnerability to climate change in these regions. In the current study, water quality monitoring data from the United States Geological Survey (USGS) are used together with information about drainage basin characteristics to evaluate drivers and spatial patterns of changing sediment delivery to coastal systems along the East and Gulf Coasts of the USA.

Methods

Suspended Sediments and Discharge

Concurrent SSC (parameter P80154) and instantaneous discharge (parameter 00061) data were obtained from USGS water quality monitoring stations draining watersheds along the East and Gulf Coasts of the USA of at least 1,000 km^2 and for which at least 50 data points spanning 15 years or more were available. USGS monitoring stations that drained the largest area and were closest to the coastal zone were chosen. No monitoring stations were included that were upstream of other monitoring stations. The drainage basins for each of these monitoring stations were determined using the 1:250,000-scale USGS hydrologic unit data. The lower basin boundaries of USGS water quality monitoring stations that did not fall at the boundary of the predefined hydrologic units were further delineated by geographic information system (GIS) (ArcMap 9.3.1) using 1 arc second elevation data from the National Elevation Dataset (Gesch et al. 2002; Gesch 2007), and the area of the delineated basins was determined.

Average daily discharge (parameter P00060) was also obtained from the same USGS water quality monitoring stations for calculation of the mean river discharge over the period of record corresponding to the SSCs (Table 1) and for determination of the contribution of baseflow to total river flow on a daily basis at each station. Daily discharge data were not available for eight water quality monitoring stations (Table 1), and discharge from alternate USGS, United States Army Corp of Engineers, or International Boundary and Water Commission monitoring stations was utilized. In all cases, the difference in drainage basin size between water quality and river discharge stations was negligible. To evaluate the importance of changes in sediment delivery over time during high discharge events versus normal river flow, and to avoid biasing the analyses with a small number of high discharge events, the amount of baseflow (relatively consistent, background flow attributable to groundwater inputs) in each river was calculated. The amount of baseflow was determined using the Base Flow Index program (v 4.15; Wahl and Wahl 1988) on average daily discharge data for the full period of record. Flow-weighted average SSC (FWA-SSC) was determined for each river at three discharge conditions: all discharges, times when baseflow exceeded 50 % of the total river flow, and again when baseflow exceeded 80 % of the total flow.

To evaluate change in SSC over time, SSC data were linearly regressed against both date (year) and instantaneous river discharge (cubic meters per second) simultaneously to determine significant changes in suspended sediment over time (ΔSSC_{Date}; in milligrams per liter per year) and with river discharge (ΔSSC_{Q}; in milligrams per liter per (cubic meters per second)) at each station:

$$SSC = (\Delta SSC_{Date} \times Date) + \left(\Delta SSC_{Q} \times Discharge\right) + Intercept \quad (1)$$

The relationship between SSC and date and river discharge (Eq. 1) was determined for SSC data for all discharge conditions, >50 % baseflow, and >80 % baseflow conditions. SSCs were found to vary significantly with discharge (i.e., ΔSSC_{Q} was a significant predictor of SSC; $p<0.05$) in 46 of the 61 rivers examined here (for all discharge conditions), and therefore, discharge was included in the regression (Eq. 1) to more accurately predict SSC and to remove any biases in sampling of sediment at different river discharge conditions. If Eq. 1 was found to significantly predict SSC ($\alpha=0.10$) within a given river, the regression coefficient ΔSSC_{Date} (in milligrams per liter per year) was examined to determine if SSC changed significantly with time ($\alpha=0.10$). When the coefficient ΔSSC_{Date} was a significant predictor of SSC in Eq. 1, the change over time was defined by the regression coefficient ΔSSC_{Date} (positive denoting an increase in SSCs over time, negative indicating declining SSCs). Data transformations (Eq. 1 with log transformations of SSC and discharge) were also explored, but untransformed linear regressions are

Table 1 Characteristics of United States Geological Survey (USGS) water quality monitoring stations and their watersheds

River	No.[a]	USGS station	Latitude	Longitude	Average discharge[b] ($m^3\ s^{-1}$)	Drainage area (km^2)	Water yield (mm)	Slope[c] (%)	Temperature[c] (°C)	Precipitation[c] (mm)	Soil erodability[c] k_w
St. Croix	A	01021050	45.17000	−67.29667	75	NA	NA	NA	NA	NA	NA
Penobscot	1	01036390	44.82667	−68.69667	405	20,112	635	7.8	5.4	1,029	0.21
Kennebec	2	01049265	44.47222	−69.68389	261	13,991	589	9.0	5.4	1,039	0.21
Saco	3	01066000	43.80806	−70.78167	76	3,356	714	13.0	5.8	1,036	0.18
Merrimack	4	01096550	42.63889	−71.37139	234	10,720	689	12.1	7.6	1,082	0.20
Connecticut	5	01184000	41.98722	−72.60583	523	24,763	665	15.4	6.2	1,156	0.25
Housatonic	6	01205500	41.38389	−73.16806	80	3,964	638	12.0	8.1	1,099	0.24
Hudson	7	01335770	42.78861	−73.67444	240	11,937	634	10.9	7.5	1,080	0.23
Mohawk	8	01357500	42.78530	−73.70810	173	9,088	600	7.8	7.9	978	0.30
Passaic	9	01389500	40.88472	−74.22611	31	1,986	498	7.8	11.5	1,084	0.24
Raritan	10	01403300	40.55944	−74.52778	36	2,058	550	4.9	11.9	1,091	0.30
Delaware	11	01463500	40.22167	−74.77806	340	17,539	610	10.3	9.0	1,010	0.25
Schuylkill	12	01474500	39.96778	−75.18889	79	4,929	506	11.9	11.4	1,093	0.24
Susquehanna	13	01578310	39.65781	−76.17450	1,135	70,149	510	12.4	10.1	1,034	0.23
Potomac	14	01646580	38.92944	−77.11722	349	29,997	367	15.0	11.5	1,096	0.26
Rappahannock	15	01668000	38.30833	−77.52944	50	4,149	381	6.1	12.5	1,082	0.32
Pamunkey	16	01673000	37.76750	−77.33250	29	2,826	322	7.0	16.5	1,242	0.31
Mattaponi	17	01674500	37.88389	−77.16528	15	1,593	301	7.9	16.1	1,199	0.29
James	18	02035000	37.67083	−78.08611	203	16,190	395	8.1	13.1	1,163	0.31
Appomattox	19	02041650	37.22500	−77.47556	37	3,490	330	6.2	17.0	1,254	0.29
Nottoway	20	02047000	36.77028	−77.16639	40	3,714	342	7.4	16.4	1,212	0.29
Blackwater	21	02049500	36.76250	−76.89861	18	1,595	350	3.6	16.0	1,182	0.29
Roanoke	22	02080500	36.46000	−77.63361	234	21,985	335	7.2	16.6	1,241	0.29
Tar	23	02083500	35.89444	−77.53306	64	5,835	346	4.5	16.7	1,217	0.24
Neuse	24	02089500	35.25778	−77.58556	78	7,039	349	4.7	16.8	1,207	0.25
Contentnea	25	02091500	35.42889	−77.58250	22	1,943	357	2.2	16.5	1,199	0.18
Cape Fear	26	02105769	34.40444	−78.29361	161	13,604	374	6.1	17.1	1,222	0.25
Pee Dee	27	02131000	34.20417	−79.54861	311	22,799	430	10.8	17.2	1,235	0.25
Lynches	28	02132000	34.05139	−79.75417	32	2,685	377	4.3	19.1	1,287	0.17
Black	29	02136000	33.66111	−79.83611	30	3,344	283	2.3	17.6	1,180	0.21
Edisto	30	02175000	33.02778	−80.39167	66	6,948	299	2.7	18.7	1,240	0.14
Savannah	31	02198500	32.52806	−81.26889	334	25,475	413	9.9	17.0	1,261	0.23
Ogeechee	32	02202500	32.19139	−81.41611	71	2,960	759	3.2	18.7	1,209	0.16
Altamaha	33	02226160	31.42694	−81.60556	356	36,604	306	5.2	18.1	1,215	0.18
Satilla	34	02228000	31.22042	−81.86583	74	7,306	319	2.3	19.0	1,272	0.10

Table 1 (continued)

River	No.[a]	USGS station	Latitude	Longitude	Average discharge[b] ($m^3 s^{-1}$)	Drainage area (km^2)	Water yield (mm)	Slope[c] (%)	Temperature[c] (°C)	Precipitation[c] (mm)	Soil erodability[c] k_w
St. Johns	35	02236000	29.00806	−81.38278	80	7,561	333	1.4	22.0	1,397	0.10
Ocklawaha	36	02240000	29.21444	−81.98611	28	6,897	127	2.3	22.0	1,397	0.10
Kissimmee	37	02273000	27.22556	−80.96278	39	7,555	164	1.4	22.0	1,397	0.10
Peace	38	02296750	27.22056	−81.87639	26	3,498	231	7.6	22.0	1,397	0.10
Withlacoochee	39	02313000	28.98861	−82.34972	21	4,486	149	2.2	22.0	1,397	0.10
Suwanee	40	02320500	29.95556	−82.92778	206	20,007	325	3.0	19.4	1,259	0.14
Ochlockonee	41	02329000	30.55389	−84.38417	34	2,813	380	3.3	18.8	1,169	0.16
Apalachicola	42	02359170	29.94917	−85.01556	668	49,802	423	6.0	18.1	1,233	0.18
Choctawhatchee	43	02366500	30.45083	−85.89833	213	11,414	589	4.4	18.8	1,221	0.17
Yellow	44	02368000	30.75278	−86.62917	36	1,587	709	2.4	18.8	1,221	0.19
Escambia	45	02375500	30.96500	−87.23417	190	9,877	606	3.4	19.1	1,411	0.24
Alabama	46	02429500	31.54667	−87.51250	936	56,891	519	10.6	16.2	1,315	0.27
Tombigbee	47	02469762	31.75694	−88.12500	852	47,797	562	5.1	16.9	1,319	0.29
Pascagoula	48	02479020	30.87806	−88.77500	332	17,347	603	4.1	18.6	1,319	0.25
Mississippi	49	07295100	31.00833	−91.62361	15,031	3,203,976	148	10.1	10.7	727	0.31
Atchafalaya	B	07381490	30.98250	−91.79833	6,438	NA	NA	NA	NA	NA	NA
Sabine	50	08030500	30.30361	−93.74361	251	24,003	330	3.6	19.3	1,170	0.35
Neches	51	08041000	30.35556	−94.09306	167	20,494	257	4.6	19.0	1,184	0.32
Trinity	52	08066500	30.42500	−94.85056	232	44,438	165	3.6	19.2	933	0.33
San Jacinto	53	08068000	30.24444	−95.45694	15	2,263	215	4.7	19.8	1,045	0.24
Brazos	54	08114000	29.58222	−95.75750	221	116,768	60	3.5	18.4	726	0.28
Colorado	55	08162000	29.30889	−96.10361	80	108,603	23	3.5	18.6	589	0.24
Guadalupe	56	08176500	28.79278	−97.01278	66	13,531	154	3.1	20.0	819	0.28
San Antonio	57	08188500	28.64929	−97.38486	28	10,127	88	2.7	20.7	761	0.27
Nueces	58	08210000	28.42722	−98.17778	17	40,034	14	3.5	21.7	607	0.25
Rio Grande	C	08475000	25.87639	−97.45417	25	NA	NA	NA	NA	NA	NA

[a] Watershed and river number correspond to the numbers displayed in Fig. 3. Rivers with letters were not included in the watershed analysis due to significant area outside the USA (St. Croix and Rio Grande) or overlap with other watersheds (Atchafalaya)

[b] Average river discharge for several rivers determined from an alternate United States Geologic Survey (UGSG), United States Army Corps of Engineers (USACE), or International Boundary Water Commission (IBWC) river monitoring station: Merrimack (USGS 01100000); Raritan (USGS 01403060); Potomac (USGS 01646502); Altamaha (USGS 02226000); Alabama (USGS 02428400); Pascagoula (USGS 02479000); Mississippi (USACE 01100); and Rio Grande (IBWC 08–4750.00)

[c] Data from Natural Resources Conservation Service STATSGO2 database (http://soildatamart.nrcs.usda.gov)

presented here. To further examine the influence of river flow on the SSC record, ΔSSC_{Date} was examined for the three discharge conditions (all discharges, >50 % baseflow, and >80 % baseflow). An example of SSC and river discharge data including baseflow over time (Fig. 2a) and the relationship between SSC and discharge (Fig. 2b) at three river discharge conditions (all flows, >50 % baseflow, and >80 % baseflow) are shown in Fig. 2. The >50 % baseflow data for both FWA-SSC and ΔSSC_{Date} are considered for the remainder of this analysis, as these data exclude large discharge events but retain the majority (more than half) of the records, whereas nearly 3/4 of the data are lost when considering only the >80 % baseflow data.

Drainage Basin Characteristics

Drainage basin characteristics were determined for the rivers included in this analysis that drained land predominantly in the USA. Data were analyzed using ArcMap 9.3.1. The slope, mean annual precipitation, mean annual temperature, and soil erodibility factor (k_w) were obtained from the Digital General Soil Map of USA (STATSGO2; Natural Resources Conservation Service 2009) and average values for each drainage basin were determined by taking area-weighted averages of the soil units occurring within each basin. The number and normal storage capacity of dams within each drainage basin were determined using data from the major dams of the USA obtained from the National Atlas (2009), which is a subset of the 2005 National Inventory of Dams that includes dams that are 50 ft or more in height and 0.0006 km^3 or more of normal storage capacity. The number of dams erected and the increased storage capacity of reservoirs in each basin during the suspended sediment period of record were also calculated.

Population from the 1960 and 2000 census (U.S. Bureau of the Census 1964; U.S. Census Bureau 2002) by county was used to estimate total population densities in the drainage basins. Similarly, data on the amount of land in farms by county and harvested cropland by state for 1950 and 1997 from the Census of Agriculture (U.S. Bureau of the Census 1952; U.S. Department of Agriculture 1999) were used to estimate changes in agriculture over time in each drainage basin. For counties that were bisected by watershed boundaries, equal population and farm acreage distributions within the counties were assumed when calculating total drainage basin population and agriculture. The purpose of the agriculture data is to define relative changes in agricultural land use by drainage basin and should not taken as a measure of actual agricultural land use in

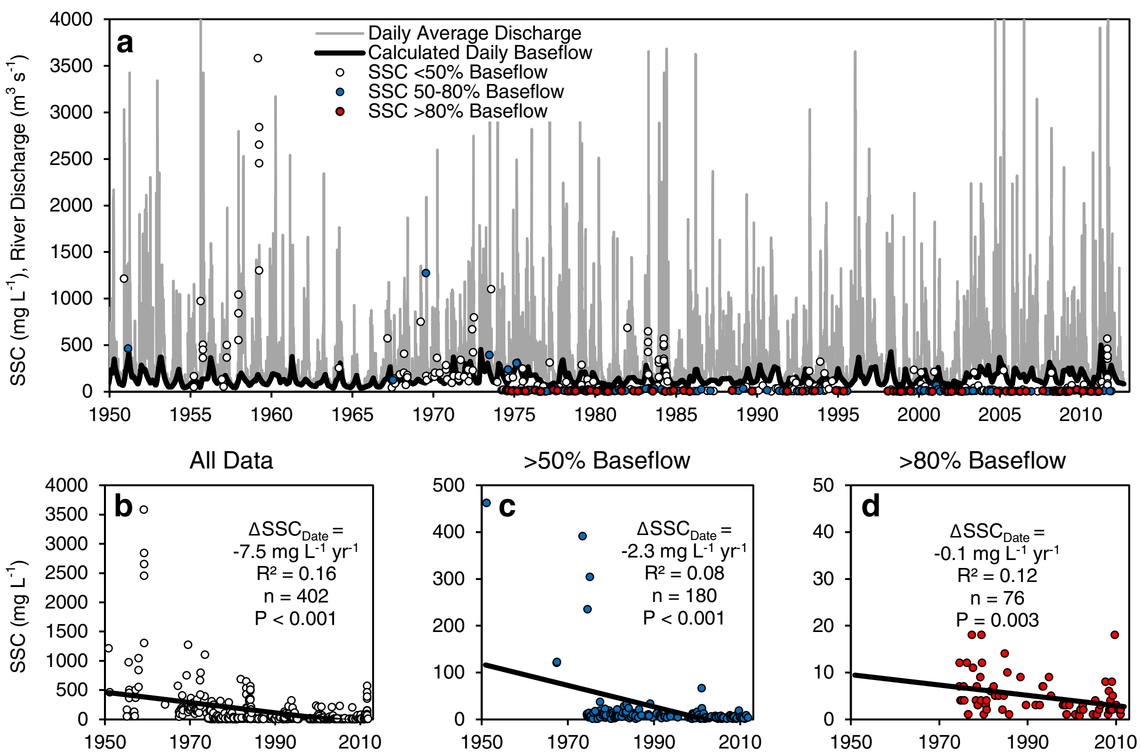

Fig. 2 **a** Example of suspended sediment concentration (*SSC*) and river discharge data from USGS station (01463500) on the Delaware River. The average daily discharge and calculated baseflow are shown, and the SSC data that were measured under discharge conditions with >80 % baseflow, 50–80 % baseflow, and <50 % baseflow are indicated. Best-fit linear regressions of SSC versus date for three discharge conditions: **b** all discharges, **c** >50 % baseflow, and **d** >80 % baseflow. The ΔSSC_{Date} coefficient (Eq. 1), correlation coefficient (R^2), sample number (n), and significance (p) from Eq. 1 are shown

these watersheds. The Multi-Resolution Land Characteristic Consortium's 2001 National Land Cover Dataset (NLCD; Homer et al. 2007) was used to define land use in the drainage basins. The NLCD land classes were compiled into nine land classes to evaluate correlations between watershed land use and suspended sediment concentrations.

Rates of RSLR along the East and Gulf Coasts of the USA were calculated using data from the Permanent Service for Mean Sea Level (2012; Woodworth and Player 2003). The rate of RSLR was calculated from annual mean sea levels for 36 stations for which data were available for at least 20 years and included measurements through 2010 or later. The rate of RSLR closest to the head of tide in each river system was used, though it should be noted that in some cases, sea level gauges were at a significant distance from the estuary into which rivers drained.

Suspended Sediments, Drainage Basin Characteristics, and RSLR

A principal component (PC) analysis was conducted for all watershed parameters, and FWA-SSC (calculated for three discharge conditions: all discharges, >50 % baseflow, and >80 % baseflow) was evaluated against the PCs using Pearson product–moment correlations to evaluate controls on FWA-SSC. To evaluate controls on ΔSSC_{Date}, only the change in dam storage capacity, change in population density, and change in land in farms during the SSC period of record were considered, as these were the parameters evaluated for change over the period of record. ΔSSC_{Date} was linearly regressed against these parameters in a stepwise fashion. In addition, FWA-SSC and ΔSSC_{Date} were related to the spatial pattern of relative sea level rise along the East and Gulf Coasts of the USA using linear regressions. Statistical analyses were conducted with SPSS.

A simple, regional vulnerability index (VI) was calculated:

$$\mathrm{VI} = -\left[(\mathrm{RSLR} \times \Delta\mathrm{SSC}_{\mathrm{Date}})\Big/\mathrm{FWA{-}SSC}\right] \qquad (2)$$

such that decreasing SSCs (negative ΔSSC_{Date}), higher rates of RSLR, and low FWA-SSC contribute to increasing vulnerability (negative VI), while increasing SSCs, low RSLR, and higher FWA-SSC contribute to lower vulnerability (zero or positive VI).

Results

Suspended Sediments and Discharge

Sixty-one USGS water quality monitoring stations were identified that fit the criteria for inclusion in this study (Fig. 3 and Table 1). More than 20 years of suspended sediment water quality monitoring data was available for 46 of the 61 stations included in this analysis and more than 30 years of data for 26 stations (Table 2).

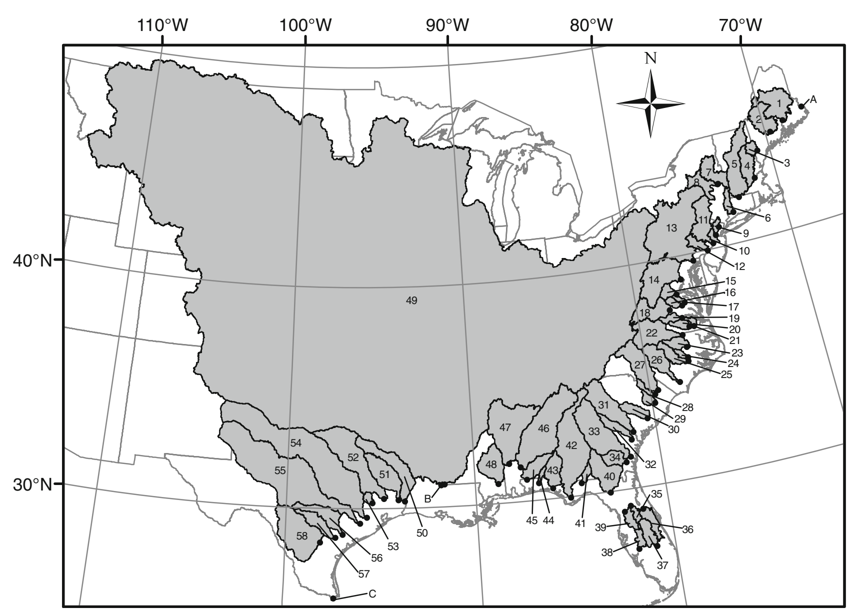

Fig. 3 Map of 58 drainage basins and 61 USGS water quality monitoring stations along the East and Gulf Coasts of the USA. The watershed numbers correspond to the numbered rivers in Table 1. The St. Croix (marked *A* on the map) and Rio Grande (*C*) fit the criteria for inclusion in this study except that the significant portions of the watersheds are outside of the USA and are therefore not included due to data inconsistencies. The Atchafalaya River (*B*) is a distributary of the Mississippi River (*#49*) and is therefore not considered separately. Watersheds were delineated using the United States Geological Survey 1:250,000-scale hydrologic unit maps, with the lower basin boundary further delineated by GIS (ArcMap 9.3.1) using 1 arc second elevation data from the National Elevation Dataset (Gesch et al. 2002; Gesch 2007)

Table 2 Flow-weighted average suspended sediment concentration and relationship between suspended sediment and discharge and time(Equation 1)[a]

River	No.[b]	Period of Record	Years	n	FWA-SSC	R^2	F	$P_{overall}$	ΔSSC_Q	$P_{\Delta SSCQ}$	ΔSSC_{Date}	$P_{\Delta SSCDate}$
St Croix	A	1975–1994	19	88	7.2	0.02	0.8	0.444	−0.3	0.863	−0.14	0.214
Penobscot	1	1979–2011	32	43	10.6	0.00	0.1	0.932	0.9	0.751	−0.12	0.880
Kennebec	2	1978–2000	22	32	3.6	0.01	0.1	0.889	0.2	0.663	0.07	0.709
Saco	3	1974–1995	21	81	6.9	0.05	1.9	0.151	3.4	0.240	0.25	0.099
Merrimack	4	1977–1995	18	39	11.7	0.13	2.7	0.083	0.4	0.831	−1.50	0.027 −
Connecticut	5	1974–2011	37	101	6.4	0.08	4.5	0.014	1.1	0.023	−0.14	0.043 −
Housatonic	6	1974–1994	20	22	9.3	0.11	1.1	0.339	1.5	0.297	0.28	0.248
Hudson	7	1975–2011	36	217	6.7	0.07	7.5	0.001	1.7	<0.001	−0.16	0.020 −
Mohawk	8	1976–2011	35	162	7.3	0.08	6.9	0.001	−1.5	0.332	−0.30	<0.001 −
Passaic	9	1965–2009	44	41	20.9	0.01	0.2	0.852	−15.1	0.624	−0.09	0.812
Raritan	10	1968–2011	43	127	6.2	0.18	13.4	<0.001	2.4	0.609	−0.23	0.000 −
Delaware	11	1950–2011	61	180	24.4	0.09	9.2	<0.001	9.3	0.083	−2.31	<0.001 −
Schuylkill	12	1975–2004	29	45	14.7	0.14	3.4	0.042	14.3	0.147	−0.59	0.047 −
Susquehanna	13	1978–2011	33	146	9.5	0.35	39.3	<0.001	0.5	<0.001	−0.11	0.222
Potomac	14	1973–2011	38	287	12.4	0.17	28.6	<0.001	1.2	0.004	−0.35	0.000 −
Rappahannock	15	1978–2011	33	189	7.9	0.30	39.7	<0.001	16.9	<0.001	−0.10	0.080 −
Pamunkey	16	1974–2011	37	204	13.1	0.09	10.5	<0.001	42.8	<0.001	0.08	0.471
Mattaponi	17	1979–2011	32	201	13.2	0.05	5.6	0.004	11.8	0.001	−0.11	0.273
James	18	1974–2011	37	318	14.2	0.04	6.0	0.003	4.5	0.009	−0.31	0.085 −
Appomattox	19	1978–2011	33	181	8.9	0.34	45.8	<0.001	22.0	<0.001	−0.26	<0.001 −
Nottoway	20	1978–1996	18	58	10.7	0.13	4.2	0.019	0.3	0.942	−0.43	0.006 −
Blackwater	21	1974–1996	22	63	8.3	0.09	3.0	0.056	−1.0	0.803	−0.24	0.017 −
Roanoke	22	1975–2007	32	57	5.7	0.15	4.9	0.011	0.9	0.003	−0.05	0.346
Tar	23	1973–1999	26	113	20.1	0.16	10.6	<0.001	25.1	<0.001	−0.49	0.049 −
Neuse	24	1973–2011	38	246	26.0	0.05	6.6	0.002	9.9	0.001	−0.16	0.345
Contentnea	25	1975–2008	33	170	17.5	0.02	1.4	0.242	4.8	0.689	−0.32	0.102
Cape Fear	26	1973–1999	26	74	17.6	0.06	2.4	0.100	4.3	0.312	−0.48	0.063 −
Pee Dee	27	1977–1995	18	50	27.3	0.20	5.9	0.005	3.8	0.001	−0.18	0.647
Lynches	28	1974–1994	20	123	7.4	0.02	1.2	0.314	3.3	0.133	−0.01	0.915
Black	29	1974–1993	19	97	4.8	0.03	1.6	0.204	−2.6	0.391	0.22	0.126
Edisto	30	1974–2011	37	279	10.1	0.05	6.9	0.001	1.8	0.364	0.41	<0.001 +
Savannah	31	1974–1994	20	124	17.8	0.02	1.4	0.245	0.3	0.563	−0.26	0.146
Ogeechee	32	1958–1995	37	247	12.5	0.04	4.9	0.008	4.9	0.058	−0.36	0.054 −
Altamaha	33	1977–2011	34	132	19.7	0.03	1.8	0.165	−0.3	0.308	0.14	0.084
Satilla	34	1959–1993	34	102	15.7	0.03	1.5	0.229	4.2	0.155	−0.17	0.396
St Johns	35	1979–2002	23	35	13.7	0.07	1.3	0.292	−4.7	0.185	−0.43	0.459

Table 2 (continued)

River	No.[b]	Period of Record	Years	n	FWA-SSC	R^2	F	$P_{overall}$	ΔSSC_Q	$P_{\Delta SSCQ}$	ΔSSC_{Date}	$P_{\Delta SSCDate}$
Ocklawaha	36	1975–1994	19	56	5.4	0.04	1.2	0.305	12.6	0.219	−0.04	0.913
Kissimmee	37	1973–1998	25	11	4.5	0.03	0.1	0.882	−0.6	0.875	−0.06	0.639
Peace	38	1974–2004	30	93	11.4	0.04	1.7	0.181	13.3	0.088	−0.13	0.264
Withlacoochee	39	1974–1995	21	98	2.9	0.10	5.3	0.006	4.1	0.160	0.16	0.002 +
Suwanee	40	1974–1996	22	105	6.8	0.00	0.2	0.823	0.3	0.534	0.02	0.856
Ochlockonee	41	1974–1997	23	62	13.0	0.07	2.4	0.104	9.9	0.118	0.27	0.132
Apalachicola	42	1983–2011	28	1031	31.2	0.18	116.5	<0.001	2.3	0.000	−1.22	<0.001 −
Choctawhatchee	43	1974–1994	20	95	17.1	0.12	6.3	0.003	−1.0	0.676	1.49	0.001 +
Yellow	44	1974–1993	19	84	16.2	0.12	5.3	<0.001	20.1	0.002	0.21	0.272
Escambia	45	1974–1994	20	80	20.4	0.19	8.8	<0.001	7.2	<0.001	0.38	0.092 +
Alabama	46	1973–2004	31	179	20.1	0.40	59.1	<0.001	2.3	<0.001	−0.58	<0.001 −
Tombigbee	47	1974–2008	34	77	23.0	0.10	4.1	0.021	1.5	0.010	0.10	0.492
Pascagoula	48	1973–1995	22	83	66.2	0.02	0.8	0.436	10.8	0.518	3.89	0.225
Mississippi	49	1972–2011	39	639	258.2	0.08	28.4	<0.001	11.75	<0.001	−1.37	0.003 −
Atchafalaya	B	1972–2011	39	323	223.2	0.23	46.7	<0.001	1.7	<0.001	−3.47	<0.001 −
Sabine	50	1974–1995	21	61	27.4	0.03	1.0	0.380	1.5	0.269	0.35	0.427
Neches	51	1960–1994	34	67	50.1	0.01	0.3	0.718	0.0	0.996	−0.51	0.486
Trinity	52	1961–1995	34	89	22.3	0.22	12.3	<0.001	5.5	<0.001	−0.80	0.011 −
San Jacinto	53	1966–2009	43	86	29.4	0.11	5.0	0.009	158.0	0.039	−0.26	0.097 −
Brazos	54	1966–1995	29	59	98.5	0.89	236.7	<0.001	102.3	<0.001	−5.15	0.011 −
Colorado	55	1974–1995	21	78	66.9	0.30	16.1	<0.001	212.7	<0.001	−1.42	0.199
Guadalupe	56	1973–1994	21	119	70.4	0.29	23.4	<0.001	117.5	<0.001	−0.40	0.726
San Antonio	57	1972–2011	39	127	134.5	0.18	13.5	<0.001	592.1	<0.001	−2.24	0.330
Nueces	58	1974–1994	20	71	55.5	0.17	6.8	0.002	204.0	0.008	−1.10	0.076 −
Rio Grande	C	1966–2010	44	60	123.4	0.44	22.0	<0.001	130.9	<0.001	−6.90	0.007 −
[c]Mean					30.3						−0.45	
[c]Median					14.2						0.00	
Min					2.9						−6.90	
Max					258.2						1.49	

[a] Data are shown for >50 % baseflow river discharge conditions only. All discharge conditions and >80 % baseflow discahrge conditions are shown in the online supplementary material (Online Resource 1). The number (n) of suspended sediment concentration data points used to determine the flow weighted average suspended sediment concentration (SSC; in milligrams per liter), the correlation coefficient (R^2), F statistic, and the significance ($P_{overall}$) for equation 1 are given, along with the coefficient ΔSSC_Q[in 10^5 milligrams per liter(cubic meter per second)] and ΔSSC_{Date}(in milligrams per liter per year), and the significance of the coefficients($P_{\Delta SSCQ}$ and $P_{\Delta SSCDate}$, respectively) within the regression. In the final column, the "+" and "-" signs denote a significant positive or negative change in SSC over time, respectively (P<0.10 for both $P_{overall}$ and $P_{\Delta SSCDate}$)

[b] Corresponds to number in Fig. 3

[c] Mean and median values of the change in suspended sediment overtime include zero values for all non-significant changes

There was a wide range in FWA-SSC in the rivers considered here, from less than 10 to over 100 mg L^{-1} (Fig. 4 and Table 2). There was a significant difference in FWA-SSC between regions (Figs. 4 and 5; $F_{(6, 54)}=59.3$; $p<0.001$), with the highest FWA-SSC in the Mississippi and Texas Gulf rivers (Tukey's post hoc, $p<0.05$). Linear regressions of SSC versus river discharge and date (Eq. 1) were significant for 39 (64 %) of the 61 USGS water quality monitoring stations included in this study (at >50 % baseflow; Table 2; all discharge conditions and >80% baseflow conditions are shown in Online Resource 1). Date was a significant predictor of SSC (i.e., ΔSSC_{Date} was significant) at 31 stations, and 25 of these demonstrated significant declines in SSC over time while four indicated significant increases (Fig. 4 and Table 2). Analysis of the data using log-transformed SSC and discharge data yielded 29 rivers with significant declines in sediment (data not shown) indicating the statistical analysis is robust, but the non-transformed relationship (Eq. 1) is presented here for clarity.

The greatest reductions in suspended sediment tended to occur in the Mid-Atlantic, Mississippi, and Texas Gulf regions (Figs. 4 and 5). Smaller declines in sediment were observed in a number of New England and Southeast Coast rivers. There was relatively little change in SSC in Florida and Eastern Gulf rivers (Figs. 4 and 5). Due to differences in the FWA-SSC (Fig. 4 and Table 2), proportional decreases in SSC (in percent per year) were greatest (>1 % year^{-1}) in the Mid-Atlantic, New England, and West Gulf rivers, while East Gulf and Florida rivers experienced small (<1 % year^{-1}) increases in SSC (Fig. 5).

Watershed Characteristics

Fifty-eight watersheds were characterized that drain to the 61 water quality monitoring stations analyzed in this study (Fig. 3 and Table 1). The St. Croix and Rio Grande rivers drain significant land area outside of the USA and were therefore not included in this study. The Atchafalaya River is a distributary of the Mississippi River and was not considered separately. Watersheds ranged in size from 3.2 million km^2 (Mississippi River) to just over 1,000 km^2 (Passaic, Mattaponi, Contentnea, and Yellow), with a median size of 10,002 km^2. River discharge from these watersheds ranged from 15 to 15,030 m^3 s^{-1} (Mataponi and Mississippi, respectively), and the water yield varied considerably across watersheds, ranging from 14 to 760-mm year^{-1} (Table 1), largely as a function of mean annual precipitation (MAP; data not shown) and mean annual temperature (MAT; Table 1) of the drainage basin (Water yield = −30.7(MAT) + 0.48(MAP) + 322.1; $R^2=0.67$; $p<0.001$). Average slopes of the drainage basins ranged from 1.4 to 15.4 %, and soil k_w factors ranged from 0.10 to 0.35 (Table 1).

Six of the watersheds remained undammed, while the estimated residence time of water in reservoirs was over 1 year in another seven drainage basins (Fig. 6 and Table 3). The residence time of water in constructed reservoirs increased by 3 months or more during the suspended sediment period of record in eight drainage basins and did not increase appreciably (<1 day) in 32 watersheds (Table 3). Regionally, there was

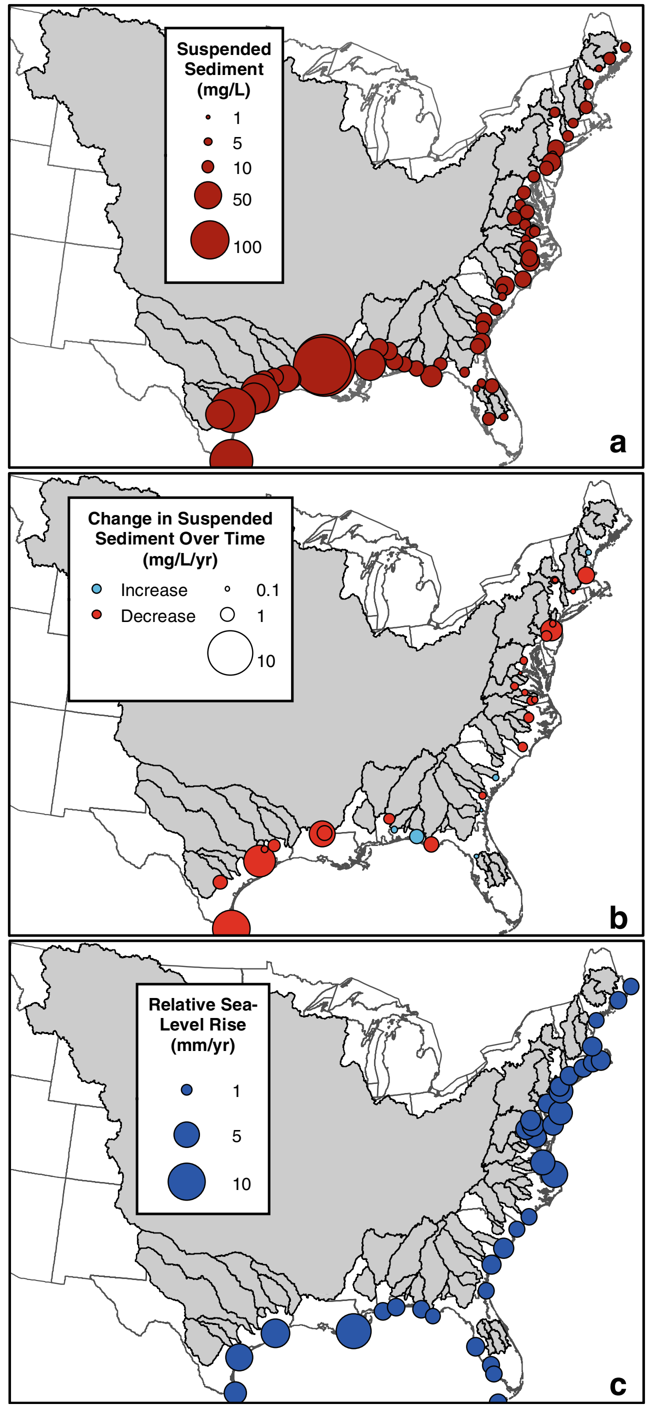

Fig. 4 **a** Average concentrations (FWA-SSC; in milligrams per liter) of suspended sediment in rivers draining to the East and Gulf Coast of the USA. **b** Change in suspended sediment concentration (ΔSSC_{Date}; in milligrams per liter per year; Eq. 1) over time in rivers draining watersheds along the East and the Gulf Coasts of the USA. Map of rates of relative sea level rise along the East and Gulf Coasts as calculated from data obtained from the Permanent Service for Mean Sea Level (PSMSL 2012)

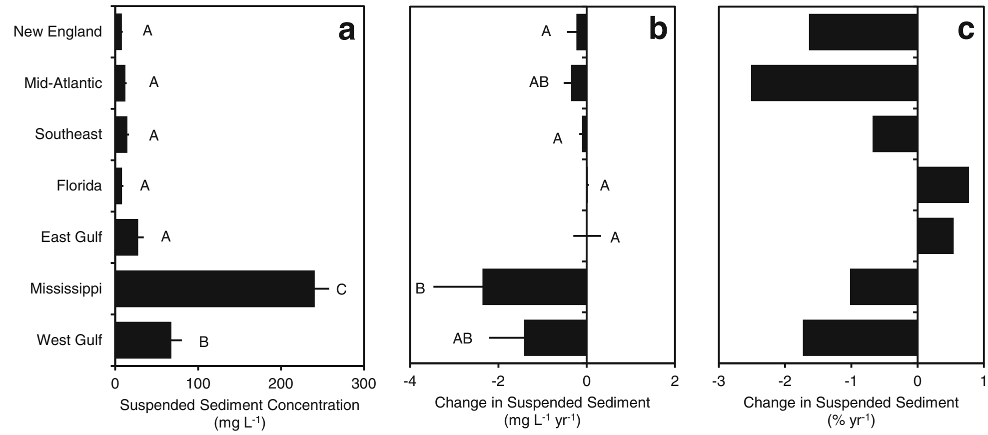

Fig. 5 **a** Regionally averaged suspended sediment concentrations in rivers draining to the East and Gulf Coasts of the USA. Rivers that do not share the same letter have significantly different sediment concentrations (ANOVA; $F_{(6,\ 54)}=59.3$; Tukey's post hoc, $p<0.05$). **b** Regionally averaged changes in suspended sediment concentration over time (in milligrams per liter per year; $F_{(6,\ 54)}=2.852$; $p=0.017$) and **c** proportional change in sediment concentration over time (in percent per year). Note that rivers that did not exhibit significant changes in sediment over time were included in **b** and **c** with zero values

a significant difference in the residence time behind dams ($F_{(6,\ 51)}=4.39$; $p=0.001$), and Texas watersheds tended to have higher residence times (Table 3; Tukey, $p<0.05$).

Population densities in these drainage basins ranged from less than 4 to over 600 persons km^{-2} (Table 4), with increases in population density between 1960 and 2000 ranging from ~0 to 160 persons km^{-2} (Fig. 7 and Table 4). Population density in 2000 was well correlated with developed land use from the NLCD (Table 5; land use=2.52 ln(population density)−6.36; $R^2=0.74$, $p<0.001$), which ranged from less than 2 % to over 25 % of land area in the drainage basins. Change in population density was relatively high in several Mid-Atlantic, Florida, and Texas watersheds (Fig. 7 and Table 4).

Agricultural land use ranged from about 1 % to nearly 40 % of watershed area (Table 4). Summed National Land Cover Dataset (2001) data for cultivated crop, pasture, grassland, and scrub/shrub land classes (Table 5) were well correlated with 1997 Agricultural Census data (U.S. Department of Agriculture 1999) for land in farms (Table 4 and Fig. 8), indicating that while Agricultural Census data included actively cultivated cropland, it also included land that was not actively cultivated. The change in land in farms reported by the Agricultural Census between 1950 (U.S. Bureau of the

Fig. 6 The storage capacity of reservoirs behind dams in the watersheds draining to the rivers evaluated in this study. Dams constructed during the suspended sediment period of record within each watershed are shown, along with dams constructed prior to the period of record (existing). Data from the National Atlas (2009)

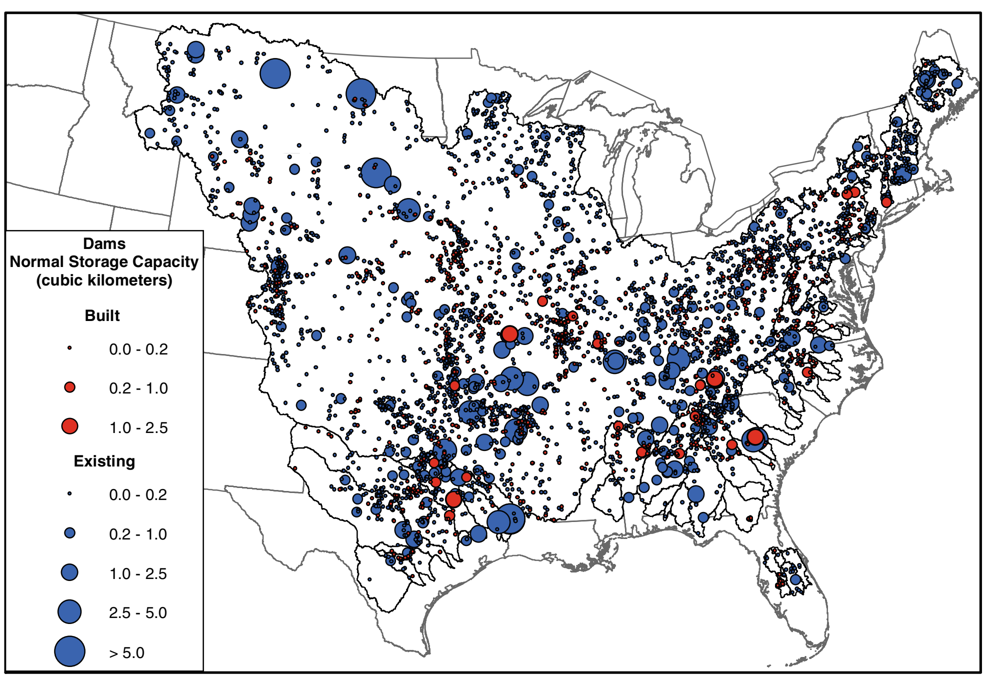

Table 3 The total number, surface area, and residence time of water in reservoirs behind total dams and dams built during the suspended sediment period of record

River	No.[a]	Total dams		Dams built	
		Dams	Residence time (days)	Dams	Change in residence time (days)
Penobscot	1	41	147.4	2	3.6
Kennebec	2	30	201.4	2	6.6
Saco	3	4	10.8	0	0.0
Merrimack	4	35	55.8	0	0.0
Connecticut	5	90	139.9	4	0.1
Housatonic	6	24	289.2	2	30.7
Hudson	7	20	69.0	2	7.1
Mohawk	8	29	40.0	3	2.5
Passaic	9	22	366.8	1	12.5
Raritan	10	6	216.4	0	0.0
Delaware	11	36	64.5	25	43.6
Schuylkill	12	30	24.6	5	16.7
Susquehanna	13	129	25.3	16	0.4
Potomac	14	127	10.7	28	4.9
Rappahannock	15	5	7.5	0	0.0
Pamunkey	16	2	198.1	0	0.0
Mattaponi	17	3	13.7	0	0.0
James	18	37	21.0	16	14.0
Appomattox	19	7	31.6	3	0.4
Nottoway	20	0	0.0	0	0.0
Blackwater	21	0	0.0	0	0.0
Roanoke	22	40	212.5	6	1.0
Tar	23	0	0.0	0	0.0
Neuse	24	7	21.5	6	21.5
Contentnea	25	0	0.0	0	0.0
Cape Fear	26	22	31.1	7	23.0
Pee Dee	27	26	36.5	1	0.3
Lynches	28	2	0.0	2	0.0
Black	29	0	0.0	0	0.0
Edisto	30	1	0.1	0	0.0
Savannah	31	59	511.8	5	45.0
Ogeechee	32	1	0.0	1	0.0
Altamaha	33	43	69.0	14	40.9
Satilla	34	0	0.0	0	0.0
St Johns	35	3	13.1	0	0.0
Ocklawaha	36	2	43.2	1	2.2
Kissimmee	37	12	290.9	0	0.0
Peace	38	66	403.9	12	84.1
Withlacoochee	39	6	59.3	1	6.8
Suwanee	40	27	46.9	12	19.0
Ochlockonee	41	0	0.0	0	0.0
Apalachicola	42	58	96.0	6	0.8
Choctawhatchee	43	1	0.6	0	0.0
Yellow	44	1	0.6	0	0.0
Escambia	45	2	1.3	0	0.0
Alabama	46	144	88.2	35	16.2

Table 3 (continued)

River	No.[a]	Total dams		Dams built	
		Dams	Residence time (days)	Dams	Change in residence time (days)
Tombigbee	47	64	46.0	21	11.3
Pascagoula	48	8	2.3	0	0.0
Mississippi	49	3,383	185.5	555	9.4
Sabine	50	23	1,128.0	2	40.1
Neches	51	13	311.7	2	4.0
Trinity	52	128	466.2	11	131.5
San Jacinto	53	2	400.0	0	0.0
Brazos	54	133	257.7	45	42.0
Colorado	55	78	774.2	10	18.3
Guadalupe	56	14	87.5	7	0.2
San Antonio	57	23	183.3	11	4.6
Nueces	58	1	6.3	0	0.0
Mean		87	132.9	15	11.5
Median		17	44.6	2	0.6
Min		0	0.0	0	0.0
Max		3,383	1,128.0	555	131.5

Data from the National Atlas (2009)

[a] Corresponds to number in Fig. 3

Census 1952) and 1997 (U.S. Department of Agriculture 1999) is assumed to be a good proxy for relative change in agricultural land use in these drainage basins, but should not be taken as a measure of actively worked agricultural land. Land in farms declined in all watersheds over this period of time, ranging from declines of nearly 50 % in several drainage basins in the southeast to negligible losses in agricultural land use in several watersheds in Texas (Fig. 7 and Table 4).

Other land use characteristics of the 58 drainage basins in this analysis ranged considerably (Table 5). 2001 land use from the National Land Cover Dataset (Homer et al. 2007) indicated that wetlands were highest in the Florida watersheds, while forested land cover was highest in northeast and Mid-Atlantic drainage basins. Grassland + scrub/shrub was markedly higher in the Texas drainage basins, and open water was highest in several Florida and Maine watersheds.

Suspended Sediments and Watershed Characteristics

A PC analysis of watershed characteristics yielded five components that explained 80 % of the variance between watershed characteristics (Table 6). FWA-SSC under all three discharge conditions (all discharges, >50 % baseflow discharges only, and >80 % baseflow discharges only) was significantly ($p<0.05$) correlated with the components that were dominated by watershed area and river discharge (PC 3). The amount of watershed area in wetland and open water land classes, soil erodability, precipitation (PC 2), percent scrub/shrub land use, and the retention of water behind dams (PC5) were also significantly correlated with FWA-SSC for all discharges and >50 % discharges (Table 6). In addition, FWA-SSC for all discharges was significantly correlated with PC1 which was loaded by mean annual temperature, watershed slope, and forest, grassland, and agricultural land classes (Table 6). Using the principal components analysis (Table 6) and referring to prior work on controls on fluvial SSC (Walling 1999; Syvitski and Milliman 2007; Schwartz 2008; Pelletier 2012), it was possible to construct a simple linear equation predicting over 80 % of the variation in FWA-SSC (<50 % baseflows) between drainage basins using three variables: watershed area (10^6 km^2), agricultural land use (LU_{Ag}; percent of land in farms from 1997; Table 4), and soil erodability (k_w; Table 1) [FWA-SSC = 64.8(area) + 0.71(LU_{Ag}) + 87.2(k_w) − 23.5; $F_{(3, 54)}=70.9$; $R^2=0.80$; $p<0.001$].

A linear regression of ΔSSC_{Date} in all rivers versus changes in dammed reservoir retention time (Table 3), population density, agricultural land use (Table 4), and mean suspended sediment concentration indicated that only increased retention behind dams was associated with declining suspended sediment [$\Delta SSC_{Date} = -0.009(Dam_{RT}) - 0.007(\text{FWA-SSC}) + 0.009$; $F_{(2, 55)}=6.16$; $R^2=0.18$; $p=0.004$], but that changes in agricultural land use and population density were not ($p>0.05$).

Table 4 Human population densities and the land area in farms

River	No.[a]	Population density (km^{-2})			Land in farms (% of watershed area)		
		1960[b]	2000[c]	Change	1950[d]	1997[e]	Change
Penobscot	1	6.7	7.3	1	13.7	3.7	−9.9
Kennebec	2	6.8	9.0	2	18.3	4.8	−13.5
Saco	3	9.9	19.0	9	17.7	4.5	−13.2
Merrimack	4	71.1	112.3	41	34.4	7.8	−26.6
Connecticut	5	40.4	51.1	11	41.1	11.9	−29.2
Housatonic	6	104.2	137.7	33	39.6	12.7	−26.8
Hudson	7	22.9	34.2	11	25.7	9.4	−16.4
Mohawk	8	51.5	52.2	1	48.1	21.9	−26.2
Passaic	9	552.8	679.0	126	23.7	7.4	−16.3
Raritan	10	211.4	374.0	163	51.1	24.2	−26.9
Delaware	11	60.2	95.1	35	46.6	17.7	−28.9
Schuylkill	12	255.8	315.6	60	56.0	28.9	−27.1
Susquehanna	13	47.1	56.3	9	53.2	27.8	−25.4
Potomac	14	36.4	76.3	40	65.0	38.4	−26.6
Rappahannock	15	12.6	29.3	17	69.1	44.1	−24.9
Pamunkey	16	14.0	40.3	26	60.0	26.2	−33.8
Mattaponi	17	10.4	39.2	29	48.8	17.6	−31.2
James	18	18.1	27.0	9	53.6	28.1	−25.4
Appomattox	19	15.8	32.9	17	64.0	32.5	−31.5
Nottoway	20	15.2	15.7	1	62.3	30.5	−31.8
Blackwater	21	23.0	27.3	4	62.7	33.1	−29.6
Roanoke	22	32.4	43.8	11	76.7	37.1	−39.6
Tar	23	31.3	39.9	9	84.5	43.5	−41.0
Neuse	24	56.8	139.8	83	77.8	37.5	−40.3
Contentnea	25	46.9	67.1	20	85.2	54.8	−30.4
Cape Fear	26	52.9	100.1	47	65.8	27.0	−38.7
Pee Dee	27	40.7	69.7	29	70.1	30.5	−39.6
Lynches	28	25.8	36.6	11	69.4	29.1	−40.3
Black	29	37.6	46.0	8	63.7	33.9	−29.8
Edisto	30	22.5	45.2	23	66.1	28.2	−37.9
Savannah	31	23.4	39.4	16	68.0	24.2	−43.7
Ogeechee	32	23.8	30.5	7	80.7	33.7	−47.0
Altamaha	33	26.4	65.1	39	73.9	30.1	−43.8
Satilla	34	11.0	15.8	5	63.7	33.3	−30.4
St Johns	35	42.8	183.3	141	53.1	36.2	−16.9
Ocklawaha	36	11.9	42.5	31	48.0	27.6	−20.4
Kissimmee	37	24.4	85.1	61	71.8	56.9	−14.9
Peace	38	22.6	55.9	33	91.9	65.1	−26.8
Withlacoochee	39	14.5	75.3	61	64.1	34.9	−29.3
Suwanee	40	12.1	18.8	7	60.2	31.5	−28.6
Ochlockonee	41	21.5	34.2	13	83.8	48.7	−35.1
Apalachicola	42	32.7	64.7	32	71.6	30.7	−40.9
Choctawhatchee	43	13.6	19.4	6	69.7	35.2	−34.5
Yellow	44	16.0	27.6	12	50.4	23.0	−27.4
Escambia	45	13.2	14.1	1	57.6	23.7	−33.9
Alabama	46	25.3	43.2	18	66.2	23.8	−42.4
Tombigbee	47	25.2	30.5	5	67.4	25.8	−41.6

Table 4 (continued)

River	No.[a]	Population density (km^{-2})			Land in farms (% of watershed area)		
		1960[b]	2000[c]	Change	1950[d]	1997[e]	Change
Pascagoula	48	14.5	17.5	3	53.7	18.2	−35.5
Mississippi	49	18.3	23.8	6	79.0	68.0	−11.1
Sabine	50	13.0	23.1	10	50.8	32.8	−18.0
Neches	51	11.6	21.7	10	53.2	36.9	−16.3
Trinity	52	42.8	119.1	76	77.3	61.0	−16.3
San Jacinto	53	9.2	65.5	56	51.7	38.6	−13.1
Brazos	54	9.9	17.8	8	86.5	84.5	−2.0
Colorado	55	7.5	15.1	8	96.1	88.8	−7.3
Guadalupe	56	9.0	23.6	15	89.0	81.3	−7.7
San Antonio	57	70.4	144.1	74	85.9	72.8	−13.1
Nueces	58	3.9	7.6	4	92.0	79.3	−12.7
Mean		43	71	28	62	35	−27
Median		23	40	14	64	31	−28
Min		4	7	1	14	4	−47
Max		553	679	163	96	89	−2

[a] Corresponds to number in Fig. 3

[b] U.S. Census of the Bureau (1964)

[c] U.S. Census Bureau (2002)

[d] U.S. Bureau of the Census (1952)

[e] U.S. Department of Agriculture (1999)

Suspended Sediments and Sea Level Rise

Rates of RSLR were determined for 36 stations along the East and Gulf Coasts of the USA (Fig. 4). RSLR was highest along the Mid-Atlantic, Mississippi, and Western Gulf coasts (Fig. 4). Both FWA-SSC and ΔSSC_{Date} were significantly correlated with the rates of RSLR (Fig. 9). FWA-SSC was positively correlated with RSLR (SSC (mg L^{-1})=17.16·RSLR−31.8; R^2=0.44; p<0.001; Fig. 9), while ΔSSC_{Date} was negatively related to the rates of RSLR (ΔSSC_{Date} (mg L^{-1})$year^{-1}$=−0.22·RSLR+0.36; R^2=0.11; p=0.011; Fig. 9).

Discussion

Data Considerations

Concentrations of suspended sediment (SSC), rather than loads, were used in this analysis of changing sediment delivery to the East and Gulf Coasts of the USA. For most of these rivers, there was not enough suspended sediment data collected to reliably calculate sediment loads. Typically, sediment loads are determined from rating curves using river discharge and sediment data. When attempting to evaluate changes in sediment over time, however, the use of rating curves will obscure alterations in sediment loads due to changes in SSCs and are therefore not appropriate in this analysis (Walling 2006). Simple regressions of instantaneous sediment load values over time will also likely obscure changes in sediment or lead to erroneous results, as instantaneous sediment loads are very discharge dependent. Other methods of evaluating sediment loads, such as double-mass plots (Walling 2006), require a greater sampling frequency than available for most rivers in this analysis. Therefore, the concentration of suspended sediment was chosen as the variable of interest in this analysis.

Numerical models that consider the complex ecogeomorphic feedbacks of wetland response to sea level rise and altered sediment supply typically utilize concentrations of sediment rather than sediment load (Fagherazzi et al. 2012; Mudd et al. 2009; D'Alpaos et al. 2011; Kirwan et al. 2010; D'Alpaos 2011). While the horizontal extent of a tidal wetland system may be determined, in part, by the sediment load, the vertical accretion potential of any given patch of wetland will be a function of the SSC in the flood water. Further, mean annual SSC is a good predictor of annual sediment load (Day et al. 2011). Fluvial SSC may not be the sole source of suspended sediments available for deposition on tidal marsh surfaces. Substantial amounts of erodible sediment may reside within estuaries that, upon sufficient energy within the estuary (from tidal currents or wind), may be resuspended and available for deposition in the marsh (Schoellhamer 2011). However, the

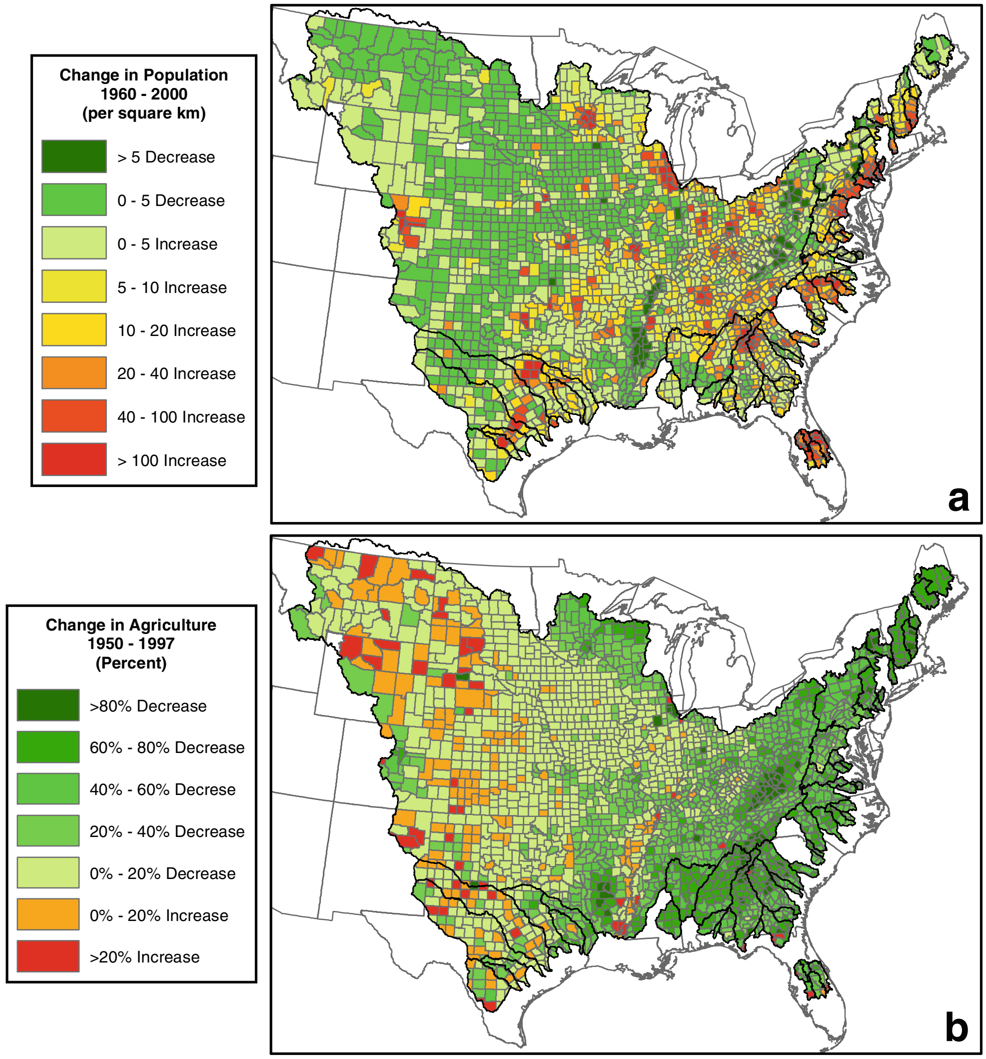

Fig. 7 **a** Change in population density by county from 1960 to 2000 as calculated from the U.S. Census for the 58 watershed included in this study (U.S. Bureau of the Census 1964; U.S. Census Bureau 2002). **b** Change in agricultural land use by county from 1950 to 1997 as calculated from the change in land in farms from the U.S. Agricultural Census (U.S. Bureau of the Census 1952; U.S. Department of Agriculture 1999)

long-term supply of erodible sediment will be depleted without resupply from fluvial or oceanic sources (Schoellhamer 2011). SSC is therefore assumed to be a valid measure of long-term sediment availability to many, though certainly not all, coastal wetlands in this analysis of changing sediment in U.S. East and Gulf Coast rivers.

To avoid biasing the evaluation of change in SSC over time (ΔSSC_{Date} in Eq. 1), linear regressions of SSC versus time and discharge (Eq. 1) were conducted for each river at three discharge conditions (>50 % baseflow; Table 2; all discharges and >80 % baseflow; Online Resource 1). An example of data for these three discharge conditions is shown in Fig. 2. The rationale is that a small number of high discharge events may unduly bias the analysis of ΔSSC_{Date}. The number of data points necessarily declined under increasingly stringent selection of data (Online Resource 1). Nevertheless, there were 23 rivers with significant declines in SSC over time (negative ΔSSC_{Date}) using all discharge conditions, 25 rivers using only >50 % baseflow data, and 15 rivers with >80 % baseflow conditions (12, 4, and 5 with positive ΔSSC_{Date}, respectively).

In general, the ability of river discharge to predict SSC (ΔSSC_Q in Eq. 1) declined as data were selected with increasing contribution of baseflow (Online Resource 1), as would be expected as the range of discharge values declined. Regressions using transformed data (log-transformed SSC and discharge) were also explored, which yielded a higher number of rivers with statistically significant declines in sediment (29 rivers at all discharge conditions; data not shown). The decision was made to use untransformed data (Eq. 1) for clarity and to derive a meaningful, parametric estimate of the change in SSC over time (ΔSSC_{Date}). The consistent patterns observed in this analysis of the data under various discharge conditions (Fig. 2, Table 2 and Online Resource 1) indicate that the findings presented here are robust.

Suspended Sediments and Watershed Characteristics

There was a clear regional pattern of average FWA-SSCs across rivers included in this study, with higher sediment in Mississippi and Gulf Coast rivers (Figs. 4 and 5). Past work

Table 5 Land use for 58 drainage basins derived from the National Land Class Dataset (Homer et al. 2007)

River	No.[a]	Open Water (Percent of watershed area)	Developed[b]	Forest[c]	Shrub/scrub	Grassland	Agriculture[d]	Wetlands[e]
Penobscot	1	5.1	1.5	72.1	7.5	0.7	1.2	11.4
Kennebec	2	6.0	3.0	69.3	9.0	1.0	3.6	7.6
Saco	3	3.0	4.1	82.4	2.2	0.3	2.4	5.2
Merrimack	4	4.5	9.9	73.7	1.4	0.3	5.2	4.6
Connecticut	5	1.9	6.9	79.0	1.4	0.2	6.9	3.5
Housatonic	6	2.6	10.4	66.4	1.6	0.2	13.6	5.0
Hudson	7	3.4	4.7	70.5	1.3	0.2	9.3	10.5
Mohawk	8	1.7	7.5	50.5	3.6	1.6	24.7	10.3
Passaic	9	3.5	30.1	49.8	0.5	0.3	2.7	12.7
Raritan	10	1.0	20.3	38.7	0.2	0.1	31.7	5.2
Delaware	11	2.0	9.3	67.3	0.5	0.3	16.5	3.6
Schuylkill	12	1.0	19.4	39.5	0.0	0.0	38.0	1.2
Susquehanna	13	1.1	7.4	61.7	1.4	0.5	26.4	1.2
Potomac	14	0.7	8.7	58.1	0.0	0.0	31.7	0.4
Rappahannock	15	0.4	4.5	55.7	0.0	0.0	38.1	0.6
Pamunkey	16	1.9	2.3	63.0	0.3	0.4	28.1	1.9
Mattaponi	17	0.4	1.5	68.8	0.0	0.0	22.2	3.6
James	18	0.7	6.4	76.5	0.5	0.6	14.8	0.4
Appomattox	19	1.0	3.7	65.9	2.1	3.1	19.6	4.3
Nottoway	20	0.4	3.5	62.3	1.1	2.8	21.4	7.3
Blackwater	21	1.1	1.5	51.3	0.0	0.0	31.6	12.2
Roanoke	22	2.2	7.3	61.9	2.1	3.7	21.4	1.2
Tar	23	0.6	7.5	47.2	1.8	7.0	27.6	8.1
Neuse	24	1.6	16.2	36.7	1.6	7.7	28.1	7.9
Contentnea	25	1.1	9.0	25.0	1.0	7.3	43.6	13.0
Cape Fear	26	1.5	13.6	46.9	2.7	8.3	20.5	5.9
Pee Dee	27	1.0	11.0	49.4	2.2	6.2	25.1	4.9
Lynches	28	0.3	5.4	37.8	1.5	12.3	26.3	16.3
Black	29	0.2	7.2	21.5	3.0	8.3	33.7	26.0
Edisto	30	0.5	6.0	35.8	2.0	13.1	24.6	17.9
Savannah	31	3.2	8.1	55.3	1.1	10.1	15.6	5.7
Ogeechee	32	0.3	4.9	43.7	1.7	11.4	22.9	14.8
Altamaha	33	1.2	9.5	50.2	0.9	9.7	17.2	10.7
Satilla	34	0.3	6.7	35.3	1.6	11.6	23.2	21.2
St Johns	35	3.9	15.8	12.5	3.4	2.6	19.9	41.8
Ocklawaha	36	14.6	14.3	14.9	1.6	4.2	23.0	27.0
Kissimmee	37	9.4	14.7	4.1	11.1	1.9	27.5	30.9
Peace	38	4.7	13.8	2.0	1.2	7.0	39.8	29.6
Withlacoochee	39	1.1	12.3	19.5	2.9	2.4	24.3	37.1
Suwanee	40	0.5	6.0	35.6	0.7	9.8	21.4	25.8
Ochlockonee	41	0.6	6.5	39.8	0.2	7.5	31.0	14.4
Apalachicola	42	1.8	9.3	48.1	4.1	5.6	21.7	9.1
Choctawhatchee	43	0.8	5.9	44.2	14.2	0.7	22.8	11.1
Yellow	44	1.0	5.8	54.3	10.4	1.2	20.6	6.6
Escambia	45	0.7	4.2	60.3	11.5	0.5	14.6	8.0
Alabama	46	1.9	7.9	59.4	5.9	3.3	16.2	4.8
Tombigbee	47	1.8	6.2	52.7	8.6	1.3	19.1	10.3
Pascagoula	48	1.0	5.7	56.0	12.8	0.3	12.9	11.3

Table 5 (continued)

River	No.[a]	Open Water (Percent of watershed area)	Developed[b]	Forest[c]	Shrub/scrub	Grassland	Agriculture[d]	Wetlands[e]
Mississippi	49	3.1	9.8	40.1	12.5	4.6	22.5	5.3
Sabine	50	4.5	7.0	36.6	9.4	6.7	18.6	17.1
Neches	51	3.0	6.9	39.8	8.6	4.4	17.5	19.6
Trinity	52	3.7	13.9	16.9	3.8	27.9	27.6	6.0
San Jacinto	53	4.1	10.3	32.0	7.8	6.9	24.8	13.9
Brazos	54	0.8	4.5	11.1	20.0	30.7	30.9	1.8
Colorado	55	0.5	2.9	8.4	58.3	15.4	13.8	0.6
Guadalupe	56	0.6	5.8	26.7	35.1	7.8	20.9	3.0
San Antonio	57	0.7	13.8	22.6	29.4	6.6	24.6	2.1
Nueces	58	0.2	3.4	8.1	62.8	10.8	12.7	1.9
Mean		2.1	8.4	45.1	6.8	5.2	21.5	10.4
Median		1.1	7.1	47.6	2.0	3.2	21.9	7.8
Min		0.2	1.5	2.0	0.0	0.0	1.2	0.4
Max		14.6	30.1	82.4	62.8	30.7	43.6	41.8

[a] Corresponds to number in Fig. 3

[b] Developed open space and low, medium, and high intensity developed land classes

[c] Deciduous, evergreen, and mixed forest land classes

[d] Cultivated crop and pasture land classes

[e] Emergent and herbaceous wetland land classes

has more explicitly examined the factors influencing time-averaged sediment delivery at the reach- (Schwartz 2008) and watershed-scale (e.g., Walling 1999; Syvitski and Milliman 2007; Pelletier 2012) than is considered here. The Spatially Referenced Regression of Watershed Attributes model for suspended sediment developed by Schwartz (2008) examined sediment dynamics along more than 60,000 river reach segments and indicated that land use type, slope, soil permeability, erodibility, and rainfall in stream catchments are important in determining sediment delivery from the catchments to the stream reaches. In-stream processes such as settling of sediments in reservoirs and mobilization of sediments in streams with greater velocities were also important in predicting sediment loads (Schwartz 2008). The watershed-scale sediment delivery model developed by Syvitski and Milliman (2007) predicts sediment delivery from watersheds based on basin area, slope, temperature, runoff, lithology, ice cover, and human activities. A more recent, process-based model included slope and soil texture to predict long-term sediment yield from rivers (Pelletier 2012). Models of sediment delivery in rivers tend to share some attributes (such as slope and soil characteristics), though the specifics differ from model to model.

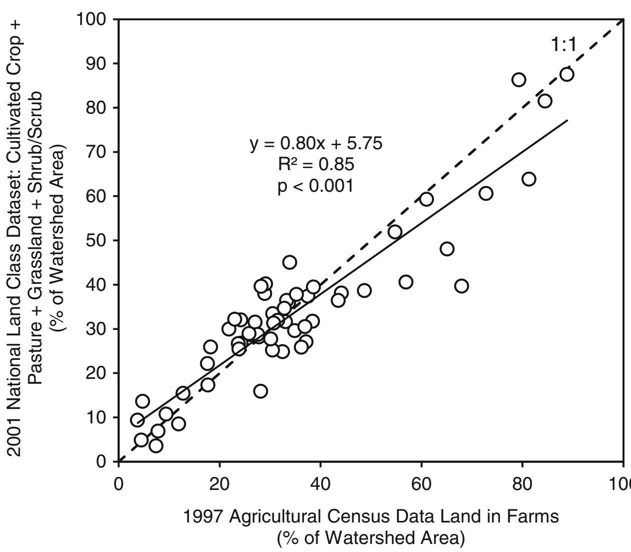

Fig. 8 The relationship between the 1997 Agricultural Census Land in Farms (U.S. Department of Agriculture 1999; Fig. 7) and the 2001 National Land Class Dataset (Homer et al. 2007; Table 5) composite agricultural, grassland, and scrub/shrub land classes for the 58 watersheds included in this analysis

Results from the PCA analysis indicate that many of these same parameters determine FWA-SSCs in rivers analyzed in this study (Table 6). Rivers with high average discharge draining large watersheds clearly tended to have higher concentrations of suspended sediment at all discharge conditions (Table 6). Precipitation, soil erodability, retention of water behind dams, and several land use classes (wetland, open water, and shrub/scrub) also contributed to differences between FWA-SSC (Table 6). Additional watershed parameters (temperature, slope, and additional land class types) were related to FWA-SSCs only when all discharge conditions were

Table 6 Principal component (PC) analysis for watershed characteristics (Table 1), dam residence time (Table 3), population density (Table 4), and land use (LU; Table 5; note that land uses have been combined into seven types for this analysis) using varimax rotation (n=58)

	PC 1	PC 2	PC 3	PC 4	PC 5
LU—forest	**−0.89**	−0.22	−0.09	−0.21	−0.24
Mean annual temperature	**0.85**	0.25	−0.13	−0.08	−0.03
Watershed slope	**−0.81**	−0.25	0.15	0.11	−0.02
LU—grassland	**0.68**	−0.19	0.01	−0.08	0.28
LU—agriculture	**0.62**	−0.14	0.09	0.19	−0.53
LU—wetland	0.37	**0.83**	−0.02	0.08	−0.11
k_w	−0.22	**−0.74**	0.11	0.03	0.15
LU—open water	−0.17	**0.70**	0.11	0.18	0.40
Mean annual precipitation	−0.07	**0.64**	−0.28	0.03	**−0.57**
Watershed area	−0.02	−0.06	**0.99**	−0.02	0.05
Average discharge	−0.08	−0.06	**0.99**	−0.02	0.02
LU—developed	0.09	0.19	0.05	**0.95**	0.02
Population density	−0.10	−0.06	−0.09	**0.94**	0.04
Dam retention time	0.12	0.01	0.02	0.21	**0.72**
LU—shrub/scrub	0.39	−0.30	0.04	−0.27	**0.66**
Eigenvalues	3.82	2.84	2.15	1.75	1.44
Percent of variation	25.45	18.95	14.31	11.66	9.59
All discharges					
r	**0.33**	**−0.37**	**0.33**	−0.06	**0.30**
p	**0.010**	**0.004**	**0.010**	0.663	**0.022**
>50 % baseflow discharges					
r	0.22	**−0.29**	**0.77**	−0.03	**0.29**
p	0.094	**0.029**	**<0.001**	0.804	**0.030**
>80 % baseflow discharges					
r	0.20	−0.24	**0.79**	−0.04	0.25
p	0.134	0.076	**<0.001**	0.756	0.062

The dominant partial correlations for each PC with eigenvalues >1.0 are noted in bold, and the percent of variance explained by each PC is noted (80 % cumulative). The eigenvalues and percent of variation for each PC are shown. The Pearson product–moment correlation coefficients (r) and significance (p) for flow-weighted average suspended sediment concentrations (FWA-SSC) determined for three different discharge conditions (all discharges, >50 % baseflow discharges only, and >80 % baseflow discharges only) for each watershed PC are shown. Significant (p<0.05) correlations are noted in bold

evaluated, suggesting that SSC during higher discharge events may be associated with watershed variables different from those that control SSC during baseflow (Table 6).

Suspended sediment concentrations declined in 25 of the 61 rivers examined and increased in only four (based on >50 % baseflow data; Fig. 4; Table 2). Of the variables examined as potential drivers of change in sediment over time (dams, population density, and agricultural land use), only the increase in water retention time in reservoirs behind dams (Dam_{RT}, days; Table 3) was significantly correlated with ΔSSC_{Date} [ΔSSC_{Date} = −0.009(Dam_{RT}) − 0.007(FWA-SSC) + 0.009; $F_{(2, 55)}$=6.16; R^2=0.18; p=0.004]. The construction of artificial reservoirs behind dams has been recognized as a major alteration to the global sediment cycle (Vörösmarty et al. 2003; Syvitski et al. 2005; Walling 2006). More than 25 % of the global sediment flux is trapped in reservoirs behind dams (Vörösmarty et al. 2003). Sediment reduction due, in part, to dam construction is a major cause for wetland loss in the world's large river deltas (Syvitski et al. 2009). Tweel and Turner (2012) estimate that sediment supply from the Mississippi River has declined by more than 75 % from peak values in the late 1800s, and this has corresponded to increased reservoir capacity in the watershed and declines in delta wetland area. Of the 58 watersheds examined here, seven remain undammed. During the period that SSCs were monitored by the USGS in these rivers (Table 2), there were 882 dams built in the watersheds that drain to these rivers (more than 60 % in the Mississippi River watershed; Fig. 6 and Table 3). These dams increased the residence time of water in 34 of the 58 watersheds, with the largest increase (132 days) in the Trinity River watershed (Table 3). The damming of rivers in watersheds is contributing to the reduction in fluvial sediment supply to wetlands along much of the East and Gulf Coasts. The rather weak predictive power of the change in reservoir retention time (R^2=0.18), however, indicates that watershed-specific factors outside the scope of this study influence the change in fluvial SSCs.

Surprisingly, declines in agricultural land use do not correlate with declines in suspended sediment concentration in rivers. Agricultural land use especially mobilizes surface soils and can be responsible for large increases in sediment delivery (Howarth 1991; Walling 1999; Dearing and Jones 2003; Saenger et al. 2008). The lack of correlation between changes in sediment and agriculture may be due, in part, to

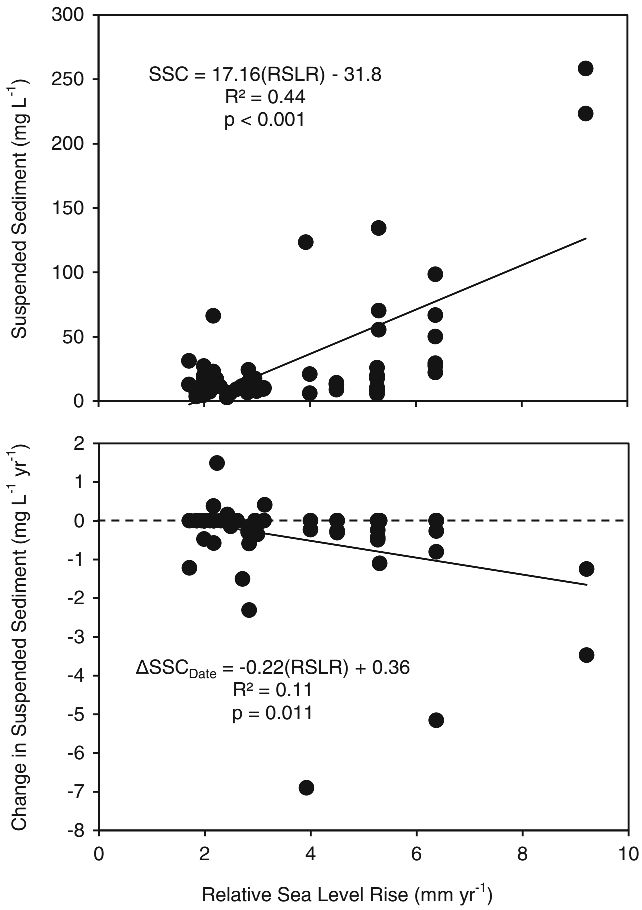

Fig. 9 Relationship between rates of relative sea level rise (RSLR) and (**a**) flow-weighted average suspended sediment concentrations (FWA-SSC) and (**b**) change in suspended sediment concentrations over time (ΔSSC_{Date}) in rivers draining to the East and Gulf Coasts of the USA

the difficulty in determining changes in agricultural intensity at the watershed-scale over time. The "land in farms" metric used here to estimate changes in agricultural land use correlates well with the NLCD cultivated crop, pasture, grassland, and shrub/scrub land classes (Fig. 8), indicating that much of the land in farms in these watersheds is not actively cultivated or grazed. The decline in land in farms (Fig. 7) may therefore not accurately reflect declines in agricultural land use at the watershed scale. The "harvested cropland" entry (U.S. Bureau of the Census 1952; U.S. Department of Agriculture 1999) was also examined as potentially a better metric for active agricultural land, but this variable was not a better predictor of ΔSSC_{Date}.

SSCs in rivers along the East and Gulf Coasts are likely lower in the period of record available here (Table 2) than they were during the large-scale deforestation and shift to agricultural land use in the eighteenth and nineteenth centuries (Fig. 1; Tweel and Turner 2012). Clearing of forested land and initiation of agricultural activities would have resulted in a peak of sediment delivery to river systems, followed by a slow recovery towards a new baseline concentration of suspended sediment as agricultural land use gave way to reforestation, increasing retention behind dams, and soil conservation practices (Walling 1999; Saenger et al. 2008; Tweel and Turner 2012; Fig. 1). The changes evaluated here, spanning the past 20 to 60 years, likely represent relatively smaller ΔSSC_{Date} than in the previous two centuries.

Tidal Wetlands, Sea Level Rise, and Sediment Delivery

Accretion in tidal wetlands is driven by both sediment deposition and organic matter accumulation, and the relative importance of these processes varies within and between wetlands (Neubauer 2008). Organic matter preservation and peat formation may drive vertical accretion in some tidal wetlands (Turner et al. 2000; Nyman et al. 2006; Neubauer 2008). However, the maximum rate of organic matter accretion is thought to be limited, and as SLR accelerates, organic matter preservation alone will likely not keep pace with sea level in many tidal wetland systems (Morris et al. 2002; Kirwan et al. 2010; D'Alpaos 2011). The rate of wetland accretion, and the maximum rate of sea level rise that a wetland can endure, therefore becomes strongly influenced by the availability of suspended sediments as SLR accelerates (Kirwan et al. 2010; Day et al. 2011; D'Alpaos et al. 2011).

Currently, rates of SLR have increased to about 3 mm-year^{-1} (Church and White 2006) and rates are projected to accelerate in the coming decades (Vermeer and Rahmstorf 2009). Tidal wetlands will need to accrete material at a faster rate than they have over the past several thousand years to avoid permanent inundation. In sediment-rich coastal regions, ecogeomorphic feedbacks will likely allow tidal wetlands to keep pace with current rates of SLR (Kirwan et al. 2010; D'Alpaos et al. 2011). However, concerns have been raised about the maximum rate of SLR with which tidal wetlands can keep pace, and the maximum rate has been shown to be largely a function of SSC and tidal range, such that decreasing SSC will result in lower resiliency of tidal wetlands to accelerated SLR (Kirwan et al. 2010; D'Alpaos 2011). Historical increases and more recent declines in sediment delivery from watersheds have been linked to coastal wetland expansion and subsequent loss in the Mississippi River Delta (Blum and Roberts 2009; Tweel and Turner 2012; Day et al. 2011), San Francisco Bay (Jaffe et al. 2007), and the Ebro Delta (Guillén and Palanques 1997). Similar reductions in sediment have been observed in many other large rivers worldwide, along with evidence of increased coastal flooding and delta submergence (Syvitski et al. 2009). The current analysis indicates that sediment reductions have occurred in many rivers draining watersheds of various sizes along the East and Gulf Coasts (Fig. 5), and that sediment delivery to many coastal wetlands is therefore likely declining. Reductions in sediment supply together with accelerating RSLR may exert increasing pressure on many tidal wetlands.

Interestingly, the rates of RSLR over the past decades are higher along the Mid-Atlantic and Western Gulf coasts (Fig. 4), resulting in a significant relationship between RSLR and FWA-SSC (Fig. 9). Land subsidence along the Atlantic Coast is linked to collapse of the proglacial forebulge following retreat of the Laurentide Ice Sheet with maximum subsidence rates centered in Maryland (Englehart et al. 2009), resulting in the pattern of relative SLR observed along the East Coast (Fig. 4). Sediment compaction in the Mississippi River Delta (Törnqvist et al. 2008) and groundwater and hydrocarbon withdrawal along the Louisiana and Texas coast (Morton et al. 2006; Kolker et al. 2011) result in high rates of coastal subsidence and account for high rates of relative SLR in the Western Gulf region (Fig. 4). Sediment supply from the Mississippi River, land subsidence (due to sediment compaction in the delta and lack of new sediment now that sediment supplies have dwindled and been largely diverted offshore), and high RSLR are closely linked in coastal Louisiana. In contrast, the link between suspended sediment concentrations and RSLR (Fig. 4) is incidental and not causal in the other coastal systems examined here.

Implications

Given the complex ecogeomorphic feedbacks operating within any tidal wetland ecosystem (Morris et al. 2002; Mudd et al. 2009; D'Alpaos et al. 2011; Fagherazzi et al. 2012), the importance of plant production in determining accretion, and other factors such as the geomorphic setting of any particular wetland, the patterns of FWA-SSC and ΔSSC_{Date} described here should not be used solely to evaluate wetland vulnerability to climate change. For instance, fluvial sediment supply can play an important role in tidal freshwater, estuarine, and deltaic wetlands, but relatively little accretion is attributable to fluvial sediment in some back-barrier and fringing wetlands. Nevertheless, these results may assist in identifying regional wetland vulnerability to coupled RSLR and sediment reductions. A simple VI was calculated to identify regions in which rapid RSLR, low FWA-SSC, and declining SSCs (negative ΔSSC_{Date}) might together increase the vulnerability of coastal wetlands (Eq. 2; Fig. 10). The index indicated that the Mississippi and West Gulf regions are increasingly vulnerable due to decreases in sediment (Fig. 5) and high rates of RSLR (Fig. 4), but because of large proportional declines in sediment in Mid-Atlantic rivers (Fig. 5) and elevated rates of RSLR along much of the Mid-Atlantic coast (Fig. 4), Mid-Atlantic wetlands are also increasingly vulnerable (Fig. 10). In contrast, the Florida and East Gulf regions are less vulnerable because of slight increases in sediment (Fig. 5) and lower rates of RSLR (Fig. 4).

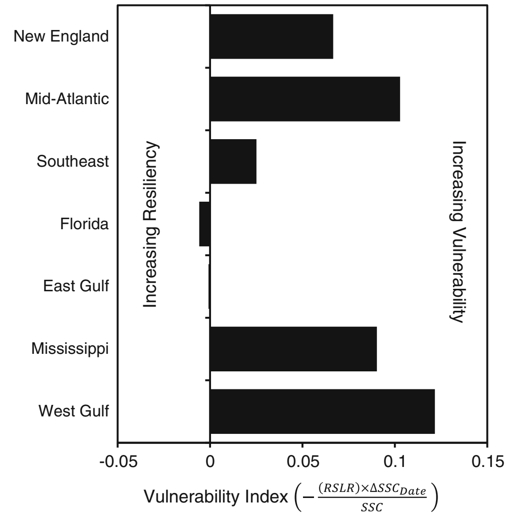

Fig. 10 Vulnerability index (Eq. 2) calculated from rates of relative sea level rise (RSLR), flow-weighted average suspended sediment concentrations (SSC), and change in suspended sediment over time (ΔSSC_{Date}) for regions along the East and Gulf Coasts of the USA

It is perhaps fortuitous that tidal wetlands in the Mississippi and West Gulf regions, which are experiencing the highest rates of RSLR (Fig. 4), also have rivers with generally the highest concentrations of suspended sediment (Figs. 4 and 5). High sediment supplies may provide the material to support vertical accretion allowing these wetlands to keep pace with rising sea levels. In an analysis of soil accretion and elevation trends, Cahoon et al. (2006) found the highest regional accretion rates along the Gulf Coast. High rates of RSLR together with high SSCs contribute to high rates of sediment accretion, provided a threshold is not reached beyond which the wetland is converted to open water (Kirwan and Murray 2007; D'Alpaos et al. 2011). However, wetland loss has been attributed to declines in sediments in some coastal systems that are relatively sediment rich, such as the Mississippi River delta (Blum and Roberts 2009) and in Galveston Bay (Trinity River; Ravens et al. 2009). The regions that are receiving the greatest concentrations of fluvial sediment are also experiencing the most rapid decline in SSC (Figs. 4 and 5), and there is a significant negative correlation between RSLR and ΔSSC_{Date} (Fig. 9). Declining SSC together with high RSLR along the Mid-Atlantic, Mississippi, and West Gulf coasts will increase the vulnerability of tidal wetlands in these regions to future climate change (Fig. 10).

Predictions about the future viability of tidal wetland ecosystems under various scenarios of climate change and SLR are often based on past rates of accretion. ^{210}Pb and ^{137}Cs radionuclide dating is the primary tool for determining past wetland accretion rates, and they give average values over the past approximately 100 and 50 years, respectively (DeLaune et al. 1989). In tidal wetland systems that are experiencing declines in SSC and concomitant reductions in sedimentation, measurements that reflect accretion over the past century may not be indicative of future accretion potential. Accretion rates obtained using these methodologies should therefore be evaluated in light of altered fluvial sediment supplies within any tidal wetland system.

Human activities enhanced the delivery of sediments to coastal waters along the U.S. East and Gulf Coasts in the eighteenth and nineteenth centuries (Fig. 1), resulting in substantial expansion of tidal wetland extent (Jaffe et al. 2007; Kirwan et al. 2011; Tweel and Turner 2012). Increased sediment supply altered the geomorphology of coastal wetlands during a period of relatively stable sea levels, and the expanses of tidal marsh that we observe in many coastal systems today may be in part a product of past human alteration to the fluvial sediment supply. Rates of SLR have increased over the past century and may accelerate in the future which, when coupled with declining sediment supply as observed in a number of rivers in this study (Figs. 4 and 5), may place many tidal wetlands at increased risk of submergence (Fig. 10). The preservation of the current extent of tidal wetlands, which reflects accelerated accretion due to past increases to the fluvial sediment supply, may not be a viable management goal in many coastal systems due to more recent declines in sediment supply and accelerating SLR (Fig. 4). The loss of some portion of wetlands in many coastal ecosystems may be largely unavoidable.

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