

Spatial and temporal patterns of modern (~100 yr) sedimentation in a tidal freshwater marsh: Implications for future sustainability

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Abstract

To maintain their position within the tidal frame, tidal marshes must accrete upward at approximately the same rate as relative sea-level rise (RSLR), typically accomplished through accumulation of mineral (from fluvial, estuarine, and/or coastal waters) and/or organic (plant-derived) sediments. However, accelerating RSLR and declining fluvial suspended-sediment supplies, along with other direct anthropogenic disturbances, may limit the ability of many marshes to keep pace. This study tests hypotheses addressing the relationship of marsh surface elevation and channel influence to marsh accretion and evaluating marsh response to historic mining disturbance and long-term environmental change, using Dyke Marsh Preserve, a tidal freshwater marsh located along the Potomac River estuary (Chesapeake Bay watershed) as a model system. Variability in sediment characteristics (grain size, organic content, bulk density) and decadal-scale accretion rates (using ²¹⁰Pb) across the marsh platform is related to ecogeomorphic setting, anthropogenic activities (mining), and long-term changes in river sediment supply and RSLR. In general, sediment on river/tidal-channel banks accumulates faster and is more influenced by fluvially derived sediment than in the marsh interior. Accumulation rates on the banks can be predicted using proxies for sediment supply and availability most of the time; however, these relationships do not hold for the marsh interior, highlighting the role of sediment redistribution and/or complex sediment-vegetation interactions. Development of temporally variable sediment rating curves reveals that suspended-sediment loads of the Potomac River have decreased since the late 1970s, although current loads appear adequate to support accretion on the marsh platform, if net onshore sediment-transport mechanisms exist.

Tidal freshwater marshes are among the world's most valuable ecosystems (Costanza et al. 1997). They reduce shoreline erosion, improve water quality through filtering land-derived sediments and nutrients bound for the aquatic environment, support commercially important fisheries, and provide critical habitat for migratory birds and waterfowl (Cooper et al. 2008; Craft et al. 2009). Yet, tidal freshwater marshes are also one of the most vulnerable ecosystems, facing the dual challenges of more rapid relative sea-level rise (RSLR) and reduced fluvial sediment supplies (Kirwan and Megonigal 2013; Weston 2014). RSLR and fluvial sediment supply are two fundamental processes driving tidal marsh formation, as they determine accumulation rates of mineral sediments and/or organic material through ecogeomorphic feedbacks (Allen 2000; Friedrichs and Perry 2001; Torres et al. 2006). Tidal marshes accrete upwards at approximately the same rate as RSLR. However, the rate of eustatic sea-level

rise has accelerated over the last two decades (Jevrejeva et al. 2010; Nicholls et al. 2011; Rahmstorf et al. 2012) coincident with decreasing fluvial sediment supplies in urbanized watersheds (Paul and Meyer 2001) and engineered rivers (Syvitski et al. 2005), challenging the ability of marshes to keep pace (Day et al. 2000; Kirwan et al. 2011; Fagherazzi et al. 2012). These challenges are especially prevalent in the Chesapeake Bay, where RSLR may be greater than the global average (Salenger et al. 2012), and where many tributaries have shown declines in suspended-sediment concentrations (SSC) over the years (Weston 2014).

Many studies examining the response of tidal marshes to environmental drivers have focused on salt-marsh environments (Morris et al. 2002; D'alpaos et al. 2007; Kirwan and Murray 2007; Day et al. 2008). As the rate of RSLR increases, the initial decrease in elevation relative to sea level is typically followed by enhanced inorganic deposition. The response of organic matter sequestration is more nuanced—plant biomass tends to respond negatively to a decrease in elevation in *Spartina*-dominated salt marshes (Morris et al.

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2002) but positively in multispecies brackish marshes (D'Alpaos et al. 2007). Freshwater tidal marshes are relatively species-rich and tend to experience lower tidal ranges. Sediment supply to tidal freshwater marshes is more variable because some marshes coincide with the turbidity maxima of estuaries and have ample sediment supply, but others are sediment starved due to watershed processes that decrease sediment inputs or increase sediment export (Coulter et al. 2004; Neubauer 2008; Brakebill et al. 2010). Tidal freshwater marshes are also more likely to be located near urban areas and thus subject to more intense anthropogenic disturbance that could exacerbate environmental pressures (Paul and Meyer 2001).

This study examines decadal-scale sedimentation in one such setting—Dyke Marsh Preserve (DMP), a tidal freshwater marsh in the Potomac River estuary, part of the Chesapeake Bay watershed in the eastern U.S.A. DMP has a complex sedimentary history, with active mining for sand and gravel deposits until the mid-1970s that changed the hydrology of the marsh and also left shorelines exposed to a long southerly fetch (Litwin et al. 2011). As the cessation of mining, the marsh has been protected by the national park service (NPS) from further disturbance. As such, the marsh offers a unique opportunity to study the impacts of human disturbance on sediment accretion and erosion. Sedimentation in DMP is highly variable in space and at bimonthly and seasonal scales, influenced by sediment availability and complex sediment-vegetation interactions (Palinkas et al. 2013). While these short-term observations are relevant to the life-cycle of plants and thus highlight the role of ecogeomorphic feedbacks in marsh accretion, decadal-scale measurements are more relevant to understanding how the marsh responds to longer-term processes such as changes in rates of RSLR and river sediment loads.

Land use in the Potomac, as in many Chesapeake watersheds, has a complex history, from widespread deforestation and agricultural expansion during colonization to industrialization and urbanization in more recent year that has resulted in stabilization of agricultural areas, afforestation, and continued urban sprawl (Brush 2009; Lookingbill et al. 2009). More recently, soil-conservation practices have reduced nutrient loads, and presumably also sediment loads, to the Potomac River (Ruhl and Rybicki 2010; Bricker et al. 2014). While the Potomac is one of the least dam-regulated large river systems in the eastern U.S.A., several small dams exist in upstream reaches that could trap sediment (Shoemaker and Miller 1986).

The goal of this study was to examine the response of a tidal freshwater marsh to historical anthropogenic disturbance (mining) and long-term environmental change (RSLR, sediment supply) within a geomorphic context to gain insights into marsh resilience (“the degree to which an ecosystem can maintain structure and function while withstanding chronic disturbance”; Schile et al., 2014) and

sustainability. This goal is addressed by measuring variability in sediment characteristics and long-term marsh accretion rates (^{210}Pb) across the marsh platform and relating measurements to ecogeomorphic setting, mining activities, and long-term changes in river sediment supply and RSLR. We specifically tested three hypotheses. (H1) Sediment deposition on the marsh platform is subject to complex ecogeomorphic interactions controlling the rate of accretion at individual sites and across the entire marsh platform (e.g., Marani et al. 2006; D'Alpaos et al. 2011; Thorne et al. 2014). (H2) Because mining changed the physical configuration and hydrology of the marsh (Litwin et al. 2013), the sedimentary record of the marsh differs pre- and post-mining, with higher variability in areas close to mining activities and lower variability in areas further removed from mining. This variability decreases toward the present as the marsh reaches a new geomorphic equilibrium. (H3) Decadal-scale changes in fluvial sediment loads directly impact marsh accretion by influencing the amount of sediment available for transport onto the marsh platform. Future marsh resiliency to accelerated RSLR depends on the response of marsh accretion to these changes. Salt-marsh studies have provided valuable insights into some of these hypotheses. However, their applicability to tidal freshwater marshes is uncertain owing to differences in vegetation and geomorphic setting.

Materials

Study site

DMP is a tidal freshwater marsh (salinity < 0.5; [Carter et al. 1994](#)) located along the Potomac River near Alexandria, VA (Fig. 1). DMP is bordered by the Potomac River to the east and the George Washington Parkway to the west. Elevations within DMP range from < 0 m to just over 0.6 m (referenced to the NAVD88 datum). Elevation is highest on a remnant dike at the northeastern border of the central marsh and lowest in the middle of the marsh, with gradually rising elevations toward upland areas to the west. Tidal channels of varying size are prevalent throughout the marsh, the largest of which is Hog Island Gut. Mixed tides at the marsh have a mean tidal range of 0.5–0.9 m ([Carter et al. 1994](#)); ebb discharges in Hog Island Gut average 4.8 m³/s ([Harper and Heliotis 1992](#)). Strong easterly winds coupled with an incoming tide can flood the entire marsh for several hours.

Sediment characteristics vary considerably throughout the marsh, though bimonthly and seasonal-scale sedimentation rates generally increase at lower elevations and with proximity to tidal channels (Palinkas et al. 2013). Sediment availability and spatial differences in the plant community play significant roles in driving these trends (Palinkas et al. 2013). Three major zones/plant communities are present in DMP—tidal freshwater marsh, floodplain forest, and swamp forest. The tidal freshwater marsh comprises ~35% of the area and is partly underwater except at low tide. The emergent marsh

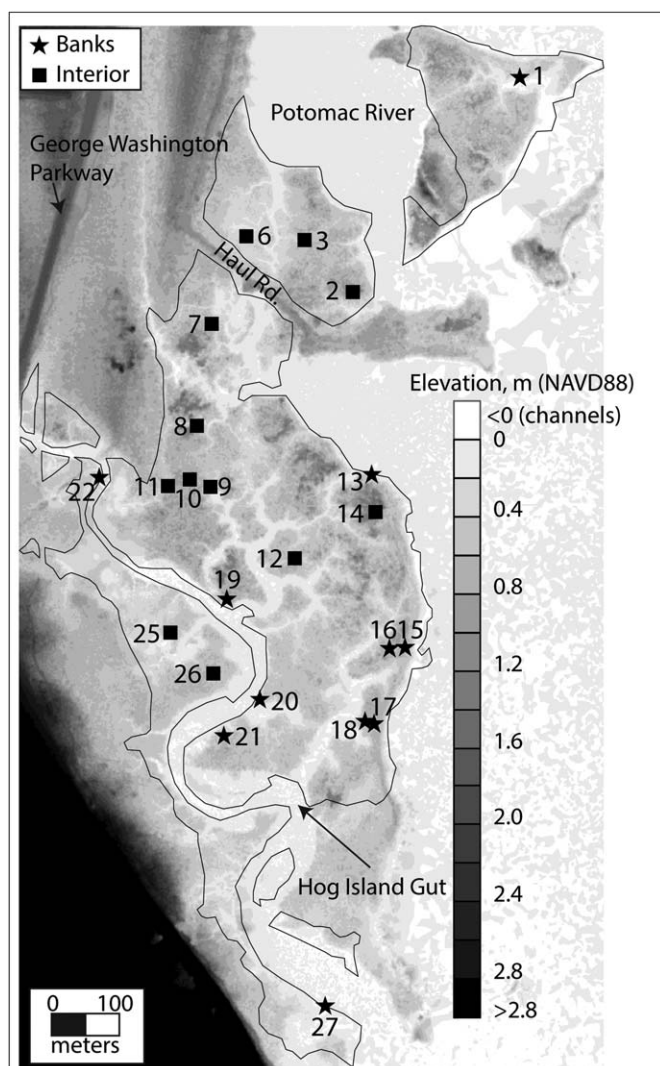


Fig. 1. Map of study sites in DMP overlaid on a 1-m Digital Elevation Model produced from a 2012 LiDAR survey (Elmore et al. in press). Note that the western boundary of the marsh is connected to the George Washington Memorial Parkway (i.e., not open water); similarly, Haul Rd is an elevated trail maintained by the NPS. Sites are colored by elevation (referenced to the NAVD88 datum); symbols indicate whether a site was considered to be on an elevated bank or in the marsh interior.

supports vegetation typical of tidal freshwater marshes along the Atlantic Coast (Simpson et al. 1983; Odum et al. 1984; Leck and Simpson 1995). Dominant species include: *Acorus calamus* (L.), *Amaranthus cannabinus* (L.) Sauer, *Bidens laevis* (L.) B.S.P., *Impatiens capensis* Meerb., *N. lutea* (L.) Sm., *Peltandra virginica* (L.) Schott, *Pontederia cordata* L., *Bolboschoenus fluviatilis* (Torr.) Soják, and *Typha angustifolia* L. (Hopfensperger et al. 2009a,b; Palinkas et al. 2013).

The marsh platform is located on top of alluvium deposits approximately 15 m thick, comprised of sand and gravel with lenticular interlayered units of silt and clay underlain by older crystalline rocks (NPS 1977). The marsh has under-

gone significant anthropogenic alteration, most significantly from mining of the sand and gravel layer. Mining activities occurred from 1938 until 1976 and reduced the area of the marsh from 0.75 km² in 1937 to 0.34 km² by 1976 (Litwin et al. 2011). As cessation of dredging, marsh erosion has continued, driven by wind-driven waves during storm events (Litwin et al. 2013). Previous work found long-term (using ¹⁴C dating) sedimentation rates of 3–5 mm/yr (Litwin et al. 2011), similar to the long-term trend of RSLR but ~1/3 of the rate of RSLR since 1990 (8.7 mm/yr; see Results).

Monitoring data

Several long-term datasets exist for water level, river discharge, and SSC near DMP. Water level has been measured at Washington, DC since 1932 by the National Oceanic and Atmospheric Administration (NOAA; <http://tidesandcurrents.noaa.gov>; station ID 8594900). Potomac River discharge (daily averaged data) has been measured at Little Falls Pump, Washington, DC (station ID 01646500) since 1930 by the United States Geological Survey (USGS; <http://waterdata.usgs.gov/>). SSC data are available from the USGS River Input Monitoring Program (<http://cbrim.er.usgs.gov/>) and are available since 1977 for the Potomac River at Chain Bridge (a bridge crossing the Potomac at Little Falls). Frequency of SSC data collection has varied throughout the years. Generally, data are available for 1 day every other month in the 1980s and 1990s and 1 day every month in the 2000s; however, there are some gaps in the record. Samples are often collected more frequently during storm events.

To assess potential temporal variability of SSC, sediment rating curves were developed for each 5-yr interval of river-discharge data (e.g., 1978–1983, 1984–1989, etc.), following the approach of Warrick (2014). Because few SSC measurements are available for each year, 5-yr intervals were chosen as a compromise between temporal resolution and robustness of the dataset. The main difference between the Warrick (2014) approach and more traditional approaches (e.g., Syvitski et al. 2000) is the use of discharge-normalized data (Q_{GM}) in the regression between log-transformed river discharge (Q) and SSC (C) data:

$$C = \hat{a} \left(\frac{Q}{Q_{GM}} \right)^b \quad (1)$$

where $\hat{a}$ is a vertical offset parameter and has units of kg/m³, equivalent to the SSC of the middle of the sample distribution, b is a unitless rating parameter found from regression (Syvitski et al. 2000) and Q_{GM} is the geometric mean of Q and is uniform for all time intervals.

Sediment field and laboratory data

Cores (~60 cm long) were collected at 23 sites in DMP (Fig. 1) from 2010 to 2011 using a Russian peat corer, which eliminates compaction (Rooth et al. 2003). These sites are identical to those in Palinkas et al. (2013), except that peat

cores were not collected at four locations (sites 4, 5, 23, and 24). Cores were sectioned into 1-cm (top 30 cm) and 2-cm (rest of core) increments in the field, and sediments were placed into sample bags that were sealed and returned to the laboratory for analysis of grain size, organic content, bulk density and ^{210}Pb .

For each 1- or 2-cm increment analyzed, samples were processed for grain size by wet-sieving at $64\ \mu\text{m}$ to separate the fine ($<64\ \mu\text{m}$) and coarse ($>64\ \mu\text{m}$) components. Note that the use of “fine” or “coarse” indicates size class and does not imply the origin (inorganic vs. organic) of material. The fine component was subsequently analyzed with a Sedi-graph 5120 (Coakley and Syvitski 1991) and dried. The coarse component was dry sieved from $64\ \mu\text{m}$ to $500\ \mu\text{m}$, using a standard set of 13 sieves. Fine and coarse data were combined to calculate the median diameter. The dry weights of the fine and coarse components were combined to calculate the bulk density. Note that these data may be underestimated, because some sediment could potentially be lost during this process. Lastly, organic content was determined via combustion at 450°C for 4 h (Erftemeijer and Koch 2001). It should be noted that loss on ignition (LOI) results can vary with differing sample size, exposure time, and ignition temperature (Heiri et al. 2001); the LOI procedures in this study are consistent with previously published organic-content values for DMP (Palinkas et al. 2013).

^{210}Pb (half-life 22.3 yr) is a useful geochronometer of sedimentation processes occurring over the last ~ 100 yr ($\sim 4\text{--}5$ half lives, typically assumed as the detection limit). ^{210}Pb is produced by the decay of ^{238}U and is supplied by precipitation, runoff, and decay of its effective parent ^{226}Ra (Nitttrouer et al. 1979). It has been used extensively in several wetland environments (Cochran et al. 1998; Anisfield et al. 1999; Kolker et al. 2009) to estimate 100-yr accumulation rates, including environments with variable sedimentation rates (Andersen et al. 2011; Ward et al. 2014; Wigand et al. 2014). ^{210}Pb activities were measured via alpha spectroscopy (Canberra Alpha Analyst), using the methods of Palinkas and Nitttrouer (2007). ^{226}Ra -supported activity (“background activity”) was determined via gamma spectroscopy of samples from the base of each core, following the methods of Palinkas et al. (2013). ^{226}Ra activities were determined indirectly by calculating a weighted average of the decays of ^{214}Pb energies (295.2 keV and 351.9 keV) and the ^{214}Bi photopeak (609.4 keV). Samples for all radioisotopic analyses were counted for ~ 24 h and decay-corrected to the time of sample collection. Because ^{210}Pb is preferentially adsorbed to fine particles (Nitttrouer et al. 1979; Goodbred and Kuehl 1998; Andersen et al. 2011), activities were normalized to the fine content (activity divided by percent fines) prior to further analysis to remove the effects of grain-size variations. Depth-integrated ^{210}Pb inventories were calculated for each core using the measured dry bulk density values at each depth horizon analyzed within the core.

Sediment accumulation rates were initially calculated using the constant flux/constant sedimentation model (CFCS; Appleby and Oldfield 1978):

$$P = P_0 e^{-\frac{\lambda}{S}z} \quad (2)$$

where P is the ^{210}Pb activity at depth z , P_0 is the initial ^{210}Pb activity, λ is the ^{210}Pb decay constant (0.031/yr), and S is the sedimentation rate. In the CFCS model,

$$S = -\frac{\lambda}{m} \quad (3)$$

where m is the slope of the line fit to the decrease in the natural log of excess ^{210}Pb activity with depth. However, this approach requires steady-state accumulation, which may not occur in dynamic environments such as wetlands, especially if the sedimentation rate is expected to change due to environmental perturbations. Instead, the constant initial concentration (CIC) model is more desirable (Appleby and Oldfield 1978). This model does not require an assumption of steady sedimentation; rather, the age (t) of sediments at depth z is:

$$t = \frac{1}{\lambda} \ln \left(\frac{A_0}{A(z)} \right) \quad (4)$$

where $A(z)$ is the cumulative inventory of excess ^{210}Pb activity beneath depth z , and A_0 is the cumulative inventory of excess ^{210}Pb activity in the sediment column. The corresponding sedimentation rate can be calculated by dividing the sediment age by its depth; all sedimentation rates thus determined for the core can be averaged to obtain the ~ 100 -yr average sedimentation rate. Mass accumulation rates calculated from the CIC model are emphasized throughout this paper; however, linear rates calculated from the CFCS model are also mentioned at times to facilitate comparison to RSLR.

Ancillary data and data analysis

Topography (elevation) and geometry (proximity to tidal channels) have been shown in field and modeling studies to influence sedimentation patterns on the marsh surface (Pasternack and Brush 2002; Temmerman et al. 2003; Temmerman et al. 2005; D'alpaos et al. 2007). Ancillary data on elevation and distance to channel were published in Palinkas et al. (2013). Briefly, elevation data were obtained via RTK-GPS measurements in 2011 using a Topcon GR-3 Total Station. Previous studies have shown that delivery pathways over the marsh platform are not solely determined by topography and geometry (e.g., distance between a point on the marsh and the nearest channel) but also by hydrodynamic gradients (Marani et al. 2003; Feola et al. 2005). However, the in-depth modeling required to calculate hydraulic distances over the marsh platform (Fagherazzi et al. 2008) is beyond the scope of this article. Instead, channel influence was assessed through the cumulative inverse squared distance

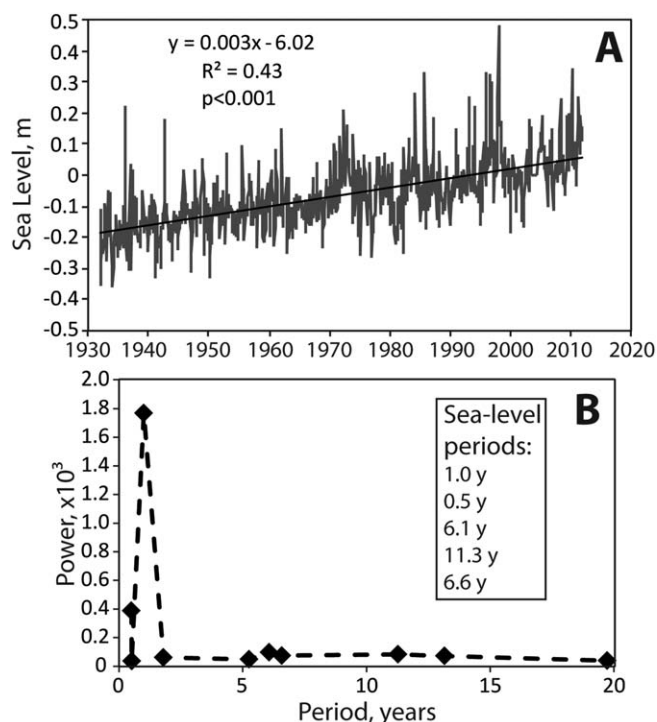


Fig. 2. (A) Monthly average sea level at Washington, DC, measured by the National Oceanic and Atmospheric Administration (NOAA; <http://tidesandcurrents.noaa.gov>; station ID 8594900) from 1932 to 2011. Sea levels are recorded relative to the Mean Sea Level datum established by NOAA's Center for Operational Oceanographic Products and Services (http://tidesandcurrents.noaa.gov/datum_options.html). (B) Results of the Fast-Fourier-Transform analysis on the sea-level data with an inset table listing the dominant periods.

(CISD) function (Sanderson et al. 2001), which is the sum of the inverse distance from each site to each channel multiplied by the order of that channel. A higher CISD value indicates more channel influence. CISD values were determined for DMP by Elmore (2008) using a 2005 QuickBird image and a NPS leaf-off 2004 aerial photo that was coregistered to the repositioned QuickBird image. Digitized channels were categorized according to their apparent size in the images and a ground survey.

Basic statistics were applied to the data where appropriate, using R statistical software. For example, regression lines were fit to ^{210}Pb profiles for the CFCS model (m in Eq. 3). Simple regressions were also used to identify relationships among grain size, organic content, and bulk density. T -tests were used to compare groups of data (e.g., sedimentation rates at sites on the banks vs. those in the marsh interior), and paired t -tests were used to assess change in sedimentation rates over time. The Pearson Product Moment was used to evaluate potential correlations. For all tests, $p < 0.05$ was the preferred level for statistical significance. However, a p value ≤ 0.1 was also accepted to identify trends that may be reflective of physical processes.

Results

Temporal trends in sea-level rise and sediment supply

The long-term trend of RSLR at Washington, DC, shows a relatively consistent rate of increase at 3.6 mm/yr, but with pronounced periodicity (Fig. 2a). A Fast Fourier Transform of the detrended data reveals that the frequencies with the greatest power occur at seasonal (0.5 yr), annual (1 yr), inter-annual (~ 6 yr, likely due to ENSO variability), and decadal (~ 11 yr, likely due to sunspot activity) intervals, with the annual cycle being the most pronounced (Fig. 2b). Mean annual sea level in 2011 (when the last cores were collected) was 0.23 m higher than in 1976 (cessation of dredging), for an average rate of RSLR of 6.5 mm/yr. Mean annual sea level in 2011 was 0.18 m higher than in 1990, approximately when eustatic sea-level rise is thought to have accelerated (e.g., Nakada et al. 2013), for an average rate of RSLR at of 8.7 mm/yr. Mean annual sea level in 2011 (0.18 m, relative to the Mean Sea Level datum established by NOAA's Center for Operational Oceanographic Products and Services; http://tidesandcurrents.noaa.gov/datum_options.html) was the highest since measurements began.

Temporal trends in sediment supply to the marsh are more difficult to discern, since measurements of SSC are available only since 1978, although river discharge has been measured since 1932. Over this time, average annual discharge at Little Pump Falls (1932–2011) is 328 m^3/s , ranging 3.4–12055.8 m^3/s . Potomac River SSC averages (1978–2011) 66.6 mg/L, ranging 0.6–2994 mg/L. The relationship between river discharge and SSC can be evaluated by calculating sediment rating curves for each 5-yr interval since 1978 (Eq. 1) and assessing whether changes in the coefficients ($\hat{a}$ and b) occur over time. Indeed, values of $\hat{a}$ decline (Fig. 3a), indicating that less sediment is currently carried in suspension for an equivalent river discharge than in the past. No significant changes were observed for values of b . The 5-yr rating curves can then be used to calculate daily averaged SSC from daily averaged river discharge measurements made by the USGS (Fig. 3b). Overall, there is a significant decrease in SSC over time (Fig. 3c) that results in a declining sediment load (product of SSC and river discharge; Fig. 3d) over time, although the trend in sediment load is not significant at the annual scale (Fig. 3d). More of the variability in sediment load is explained by time when loads are binned into 5-yr increments ($r^2 = 0.20$), but the trend is not statistically significant ($p > 0.1$). The annual average sediment load from 1978 to 2011 is 8.24×10^5 t/yr. The average annual load from 1978 to 1990 (9.11×10^5 t/yr) is significantly higher (t -test; $p = 0.09$) than the average annual load from 1991 to 2011 (7.71×10^5 t/yr).

Marsh-wide sediment characteristics and accumulation rates

Sediment characteristics (grain size, organic content, and bulk density) vary considerably across the marsh (Fig. 4a,b).

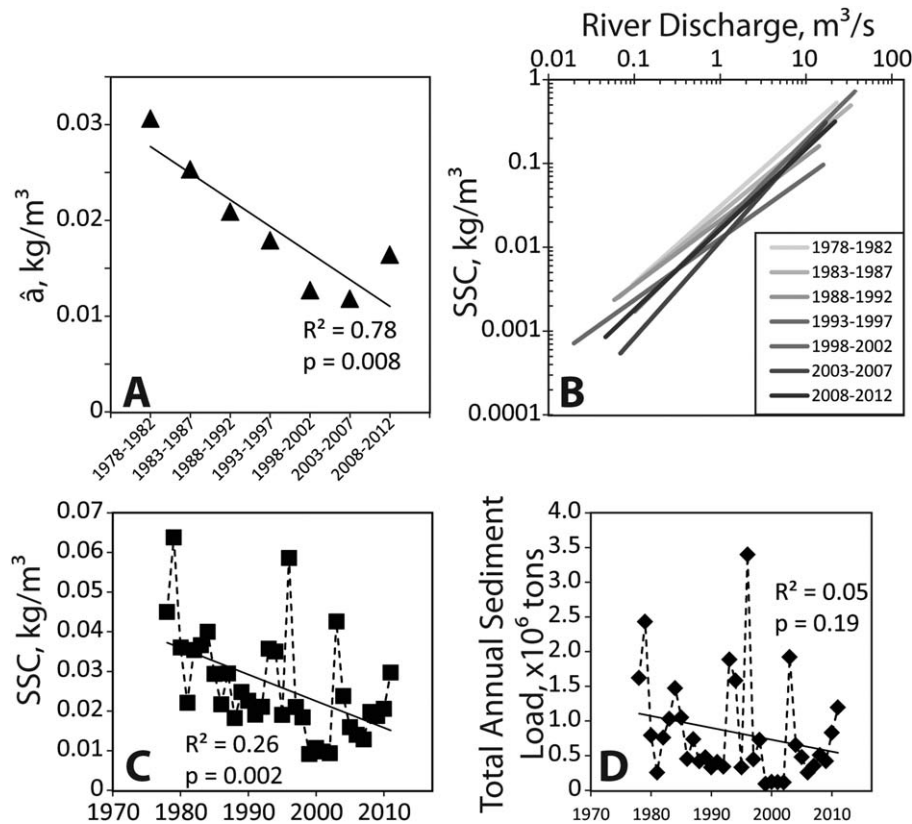


Fig. 3. Results of the sediment-rating-curve analyses. (A) Temporal trend of $\hat{a}$, the vertical offset parameter. (B) Calculated sediment rating curves. Darker colored lines indicate rating-curve relationships for more recent years. Note that data were binned into 5-yr intervals for these analyses. (C) Temporal trend of mean annual SSC from 1978 to 2011 (calculated using the rating curves shown in B). (D) Temporal trend of calculated mean annual sediment load (product of SSC and river discharge).

Surficial (uppermost 1 cm) grain size (expressed as % coarse material) ranges from 4.9% at Site 21 to 93.2% at Site 3, surficial organic content ranges from 19.2% at Site 21 to 61.5% at Site 14, and surficial bulk density ranges from 0.01 g/cm³ at Site 7 to 0.32 g/cm³ at Site 20. Despite their high spatial variability, these parameters are tightly correlated—surficial coarse and organic content are positively correlated ($p = 0.002$, $r^2 = 0.65$), but coarse content and bulk density ($p = 0.04$, $r^2 = 0.51$), as well as bulk density and organic content ($p = 0.07$, $r^2 = 0.47$), are negatively correlated. Core-averaged coarse content ranges from 3.2% at Site 12 to 70.8% at Site 3, organic content ranges 15.2% at Site 1 to 38.1% at Site 3, and bulk density ranges from 0.12 g/cm³ at Site 13 to 0.52 g/cm³ at Site 22. In general, coarse and organic content decrease with depth downward in core profiles, whereas bulk density increases, likely reflecting decreased preservation of organic content over time (Fig. 5). These down-core changes introduce enough variability that correlations among core-averaged measurements are present but not statistically significant, except between bulk density and organic content ($p = 0.001$, $r^2 = 0.73$).

Decadal-scale accumulation rates (Eqs. 2 and 3) average 7.6 ± 1.8 mm/yr, ranging from 2.9 mm/yr at Site 11 to

17.4 mm/yr at Site 25 (Table 1; Fig. 4c). Accumulation rates are generally highest at sites on the banks of the Potomac River and Hog Island Gut and lowest at sites located in more interior regions of the marsh. However, while ²¹⁰Pb profiles typically show a logarithmic decrease in activities with depth, some variability is evident, and the regression fit is not significant ($p < 0.1$) for six of the 23 profiles. At Site 27, excess ²¹⁰Pb activities are only present in the upper 4 cm of the core, limiting the practical utility of the CFCS model. Alternatively, the CIC model calculates sediment ages (and corresponding linear sedimentation rates) for each depth horizon within each core. Averaged over the entire core, these rates are similar to CFCS estimates (paired t -test, $p = 0.78$). CIC-derived linear accumulation rates average 7.7 ± 2.4 mm/yr, ranging from 2.9 mm/yr at Site 18 to 20.3 mm/yr at Site 25. The largest discrepancy is observed at Site 14, where rates are 5.5 mm/yr and 12.2 mm/yr calculated with the CFCS and CIC models, respectively. Because bulk density can change considerably with depth in cores, mass accumulation rates (g/cm²/yr; calculated for each depth horizon with the CIC model; Table 1) are used throughout the remainder of this article. Average mass accumulation rates range from

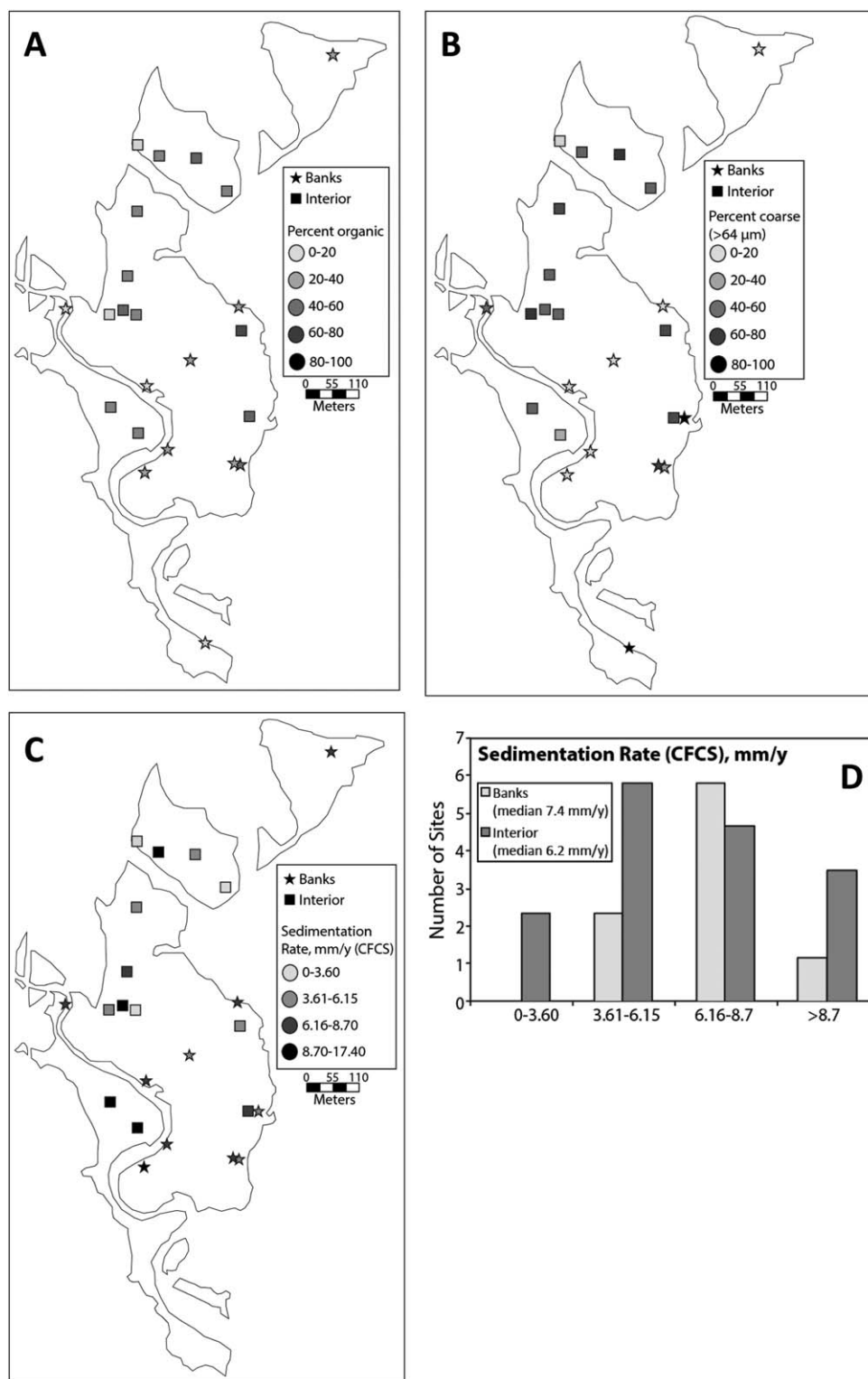


Fig. 4. Maps showing spatial distribution of (A) coarse content (percent > 64 μm); (B) organic content; and (C) linear sedimentation rate from the CFCS model. Note that a sedimentation rate could not be calculated for site 27, and thus it is not mapped. CFCS-derived rates are presented in this figure to highlight the RSLR: lightest color indicates rates less than long-term RSLR (3.6 mm/yr), the intermediary colors indicate rates between long-term RSLR and post-1990 RSLR (3.6–8.7 mm/yr), and the darkest color indicates rates above post-1990 RSLR (>8.7 mm/yr). (D) Histogram of sedimentation rates for sites on the banks (light gray) and in the marsh interior (dark gray). Median values are shown in the figure legend.

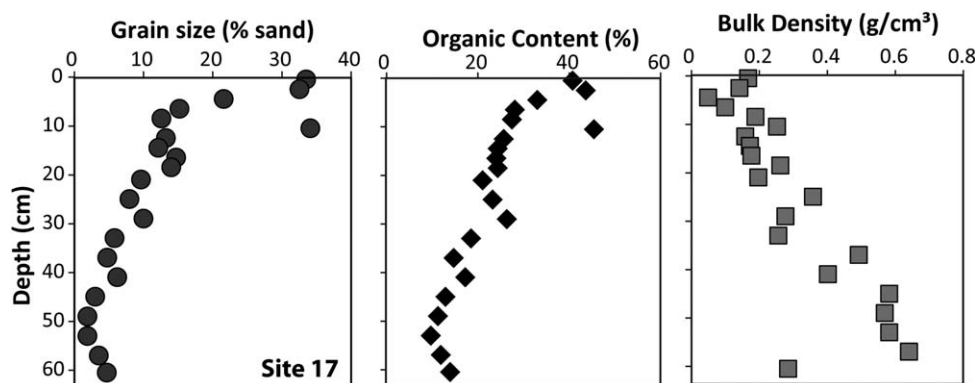


Fig. 5. Typical core profiles of (left to right) grain size (expressed as percent coarse material), organic content, and bulk density.

0.29 g/cm²/yr at Site 11 to 1.74 g/cm²/yr at Site 25 (see Fig. 5 for typical ²¹⁰Pb profiles).

Spatial and temporal variability of sediment characteristics and accumulation rates

The preceding section should make it apparent that sedimentation and corresponding sediment characteristics are variable across the marsh and so understanding the spatial and temporal context of observations is critical for subsequent interpretation. To address spatial variability, sites have been divided into two categories that reflect their general geomorphic setting—those on the banks along the Potomac River or Hog Island Gut and those in the marsh interior. Temporal variability of sediment characteristics and accumulation rates can be investigated with the geochronology established at each site via the CIC model. While there are errors inherent with this approach, potential broad-scale changes between pre- and post-1976 (i.e., pre- and post-sediment mining operations) conditions can be examined. Note that after dividing sites into the geomorphic categories, the sample size for statistical analyses decreases such that the tests lose some robustness. Thus, broad patterns and trends are described instead of relying on statistical tests of significance.

A summary of the coarse- and organic-content data, averaged by category, is given in Table 2. Surficial sediments on the banks generally have less coarse and organic material than those in the marsh interior, resulting in significantly (*t*-test; *p* = 0.06) higher bulk densities. For both categories, sediments emplaced prior to 1976 are less organic and finer than those emplaced after 1976.

Mass accumulation rates are typically highest on the banks and lowest in the marsh interior, evidenced by the median values shown in Table 1 and the histogram shown in Fig. 4d. Regression fits for sites on the banks have lower *r*² values (average 0.38) and higher *p* values (average 0.08) than in the interior (average *r*² = 0.54; average *p* values = 0.003), suggesting that accumulation is steadier over time (less variability in profiles) at sites in the marsh interior.

Indeed, when divided into the pre- and post-1976 eras, average accumulation rates show different trends between the categories—rates decrease at sites on the banks but remain fairly unchanged in the marsh interior. The post-1976 era can be further divided to assess whether there has been a more recent change in sedimentation rate related to the post-1990 increase in the rate of sea-level rise. On the banks, accumulation rates decrease between the pre-1976 and 1976-1990 eras, then remain relatively steady after 1990. In the marsh interior, accumulation rates are similar throughout the record. Note that site 27, located at the mouth of Hog Island Gut, has not been included in these analyses; the average accumulation rate at this site is an order of magnitude lower than the other sites, complicating analysis of its temporal variability within the relatively coarse-scale depth horizons.

The general trends of mass accumulation rates averaged across sites per geomorphic setting are also apparent at individual sites (Fig. 6). Sites on the banks are exemplified by Site 21. Two groups of accumulation rates above and below 25–29 cm emerge in the sediment profile: (1) lower rates in the upper portion of the core (0.21 ± 0.08 g/cm²/yr) and (2) higher rates in the lower portion (0.34 ± 0.14 g/cm²/yr). This horizon occurs between 1975 and 1979, according to the CIC model. Because bulk density changes considerably with depth in cores, mass accumulation rates are cited. However, relative linear accretion rates can be obtained by dividing the mass accumulation rates by the core-averaged bulk density (0.23 g/cm²). This yields average rates of 9.2 mm/yr for the upper portion and 14.6 mm/yr for the lower portion of the core. Although subdivision of the post-1976 era into before and after 1990 is based on few data points, no other obvious changes are present in the ²¹⁰Pb profile. Sites in the marsh interior are exemplified by Site 12. The regression fit has a higher *r*² value and a lower *p* value than Site 21, indicating more steady accumulation. Accumulation rates appear to increase from the base of the core to ~20 cm, then decrease. Although this change

Table 1. Sedimentation rates; for the purposes of calculating the average p -values, $p < 0.001$ was assumed to be $p = 0.001$. Median values ± 1 standard deviation are shown at the bottom.

Site	Elevation, m (NADV88)	CFCS average, mm/yr	R^2	p	CIC average, mm/y (g/cm ² /yr)	Pre-1976, g/cm ² /yr	1976-1990, g/cm ² /yr	Post-1990, g/cm ² /yr	Post-1976, g/cm ² /yr
13	0.57	8.5	0.57	<0.001	12.4 (0.24)	0.40	0.14	0.12	0.24
15	0.67	5.6	0.20	0.31	6.9 (0.11)	0.12	0.09	0.11	0.11
16	0.58	6.9	0.29	0.14	6.8 (0.10)	0.12	0.07	0.06	0.07
17	0.38	4.1	0.71	<0.001	8.0 (0.17)	0.19	0.18	0.17	0.17
18	0.39	6.8	0.56	0.15	2.9 (0.05)	0.06	0.05	0.05	0.05
19	0.37	11.3	0.23	0.08	9.7 (0.23)	0.27	0.20	0.22	0.21
20	0.28	7.8	0.16	0.16	6.0 (0.19)	0.24	0.17	0.08	0.13
21	0.39	8.7	0.17	0.07	9.1 (0.24)	0.30	0.19	0.18	0.19
1	0.15	6.6	0.59	0.005	4.2 (0.09)	0.09	0.07	0.11	0.10
2	0.42	3.6	0.79	<0.001	3.5 (0.08)	0.07	0.08	0.09	0.09
3	0.39	6.1	0.71	<0.001	8.0 (0.10)	0.09	0.12	0.10	0.12
6	0.32	15.5	0.08	0.35	19.7 (0.31)	0.15	0.17	0.42	0.39
7	0.34	6.1	0.54	0.002	5.7 (0.12)	0.08	0.11	0.18	0.16
8	0.47	7.6	0.84	<0.001	5.0 (0.08)	0.09	0.08	0.06	0.07
9	0.41	5.0	0.49	0.004	4.2 (0.04)	0.07	0.03	0.02	0.02
10	0.46	13.0	0.70	<0.001	5.2 (0.11)	0.15	0.13	0.05	0.09
11	0.35	2.9	0.64	0.003	3.8 (0.05)	0.05	0.04	0.05	0.04
12	0.27	5.6	0.47	0.004	4.9 (0.10)	0.09	0.13	0.09	0.21
14	0.40	5.5	0.69	<0.001	12.2 (0.13)	0.11	0.21	0.14	0.14
22	0.40	6.2	0.54	<0.001	5.2 (0.16)	0.12	0.15	0.27	0.23
25	0.35	17.4	0.16	0.23	20.3 (0.33)	0.54	0.40	0.29	0.30
26	0.41	8.5	0.35	0.02	6.7 (0.11)	0.13	0.09	0.06	0.07
27	0.04	NA	NA	NA	NA (0.01)	NA	NA	NA	NA
Banks	0.39 $\pm$ 0.13	7.4 $\pm$ 1.0	0.26 $\pm$ 0.10	0.11 $\pm$ 0.05	7.5 $\pm$ 1.3 (0.18 $\pm$ 0.03)	0.22 $\pm$ 0.05	0.16 $\pm$ 0.03	0.12 $\pm$ 0.03	0.15 $\pm$ 0.03
Interior	0.39 $\pm$ 0.12	6.2 $\pm$ 2.1	0.57 $\pm$ 0.11	0.003 $\pm$ 0.05	5.2 $\pm$ 2.8 (0.11 $\pm$ 0.04)	0.09 $\pm$ 0.06	0.12 $\pm$ 0.05	0.10 $\pm$ 0.06	0.11 $\pm$ 0.05
All Sites	0.39 $\pm$ 0.13	6.7 $\pm$ 1.8	0.54 $\pm$ 0.12	0.004 $\pm$ 0.05	6.4 $\pm$ 2.4 (0.11 $\pm$ 0.04)	0.12 $\pm$ 0.06	0.13 $\pm$ 0.04	0.11 $\pm$ 0.05	0.13 $\pm$ 0.05

occurs near the 1976 horizon (CIC model age), average rates in the upper portion of the core (0.11 ± 0.03 g/cm²/yr) are similar to those in the lower portion of the core (0.09 ± 0.02 g/cm²/yr). The corresponding linear accumulation rates, calculated with the core-averaged bulk density of 0.29 g/cm, are 3.7 mm/yr and 3.3 mm/yr, respectively. Sub-division of the post-1976 era into before and after 1990 reveals an apparent increase in accumulation rates from ~1976 to ~1990, followed by a more recent decrease, though there are few data points in either era.

Relationships among parameters

Sediment characteristics are related to parameters describing the physical setting of sites, i.e., elevation and channel influence. While correlations are not statistically significant, coarse content and bulk density generally decrease with increasing elevation, while organic content tends to increase with increasing elevation. The strongest correlations with channel influence (CISD) occur with surficial and core-averaged bulk density ($p = 0.006$ and $r^2 = 0.43$ for surficial values; $p = 0.005$ and $r^2 = 0.46$ for average values), which both

Table 2. Sediment characteristics averaged by geomorphic unit. Data are presented as the median ± 1 standard deviation.

Category	Surficial coarse %	Surficial organic %	Pre-1976 coarse %	Post-1976 coarse %	Pre-1976 organic %	Post-1976 organic %
Banks	33.5 $\pm$ 14.7	28.9 $\pm$ 5.4	18.0 $\pm$ 13.0	19.4 $\pm$ 15.0	16.1 $\pm$ 9.5	25.4 $\pm$ 7.2
Marsh interior	53.3 $\pm$ 13.1	29.8 $\pm$ 6.0	40.0 $\pm$ 12.6	52.5 $\pm$ 10.8	22.6 $\pm$ 3.4	29.4 $\pm$ 2.9

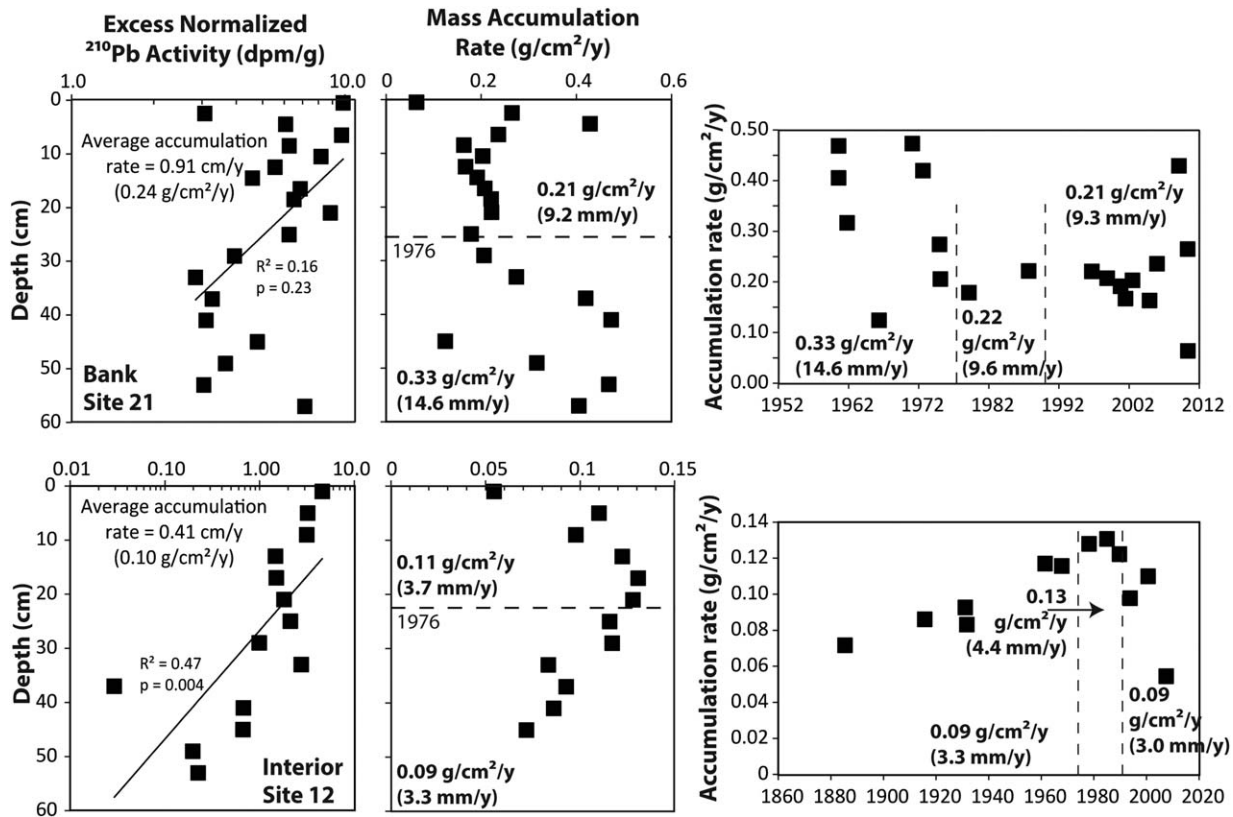


Fig. 6. Example profiles for a site on the banks (top row) and in the marsh interior (bottom row). The left panel shows the excess ^{210}Pb activities measured in the lab, normalized by content of fine sediment. Accumulation rates in this panel were calculated with the CFCS model and translated to mass accumulation rates using the corresponding average measured bulk density. The middle panel shows mass accumulation rates calculated with the CIC model for each depth horizon analyzed in the core. The dotted line corresponds to the calculated 1976 depth horizon; accumulation rates above and below the line are for post-1976 and pre-1976 data, respectively. The right panel shows the temporal trend of CIC accumulation rates. Dotted lines correspond to 1976 and 1990; accumulation rates indicate averages in each time interval. Note that, in all cases, dotted lines are intended to show which data points are included in each time interval and may not match precisely with years shown on the x-axis due to the scale of symbols used.

increase with increasing CISD (higher channel influence). When sites are grouped by geomorphic category, these general relationships are also present but not statistically significant, except for surficial bulk density and channel influence on the banks (positive; $p = 0.04$, $r^2 = 0.81$) and surficial coarse content and elevation on the banks (positive; $p = 0.08$, $r^2 = 0.38$).

No statistically significant relationships exist between sediment accumulation rates and either elevation or channel influence. Some parameters describing sediment character and accumulation rates are related. For example, surficial coarse content is negatively correlated with average mass accumulation rates ($p = 0.07$, $r^2 = 0.16$) and pre-1976 accumulation rates ($p = 0.05$, $r^2 = 0.17$). These relationships also hold true when only sites on the banks are considered ($p = 0.001$, $r^2 = 0.77$; $p = 0.006$, $r^2 = 0.69$, respectively), but no statistically significant correlations were observed with single variables in the marsh interior.

We constructed multiple linear regression models to predict sediment accumulation on banks and the marsh interior using proxies for sediment character (coarse content) and availability (elevation). While alternative choices could be made for suitable proxies (e.g., organic content for sediment character and CISD for sediment availability), interpretations of the resulting models are identical as these variables are interrelated. Thus, we use just two physical variables to eliminate autocorrelations among variables and focus on relevant parameters that are relatively easily measured in the field. Models of the average, pre-1976, 1976–1990, and post-1990 average accumulation rates were built for each geomorphic category. The model explains 89% of the variability in average accumulation rates ($p < 0.001$) on the banks. The bank model also explains 98% of the variability in the pre-1976 accumulation rate ($p < 0.001$) and 63% of the variability in the 1976–1990 accumulation rates ($p = 0.02$). However, only 16% of the variability in post-1990 rates ($p = 0.66$) is

explained. In the marsh interior, however, none of the models are adequate in explaining variability in sediment accumulation across sites. In all models, accumulation rates increase with elevation, but decrease with increasing coarse content.

Statistical models can also lend insight into the influence of sea level and SSC on sediment accumulation rates. The sediment geochronologies developed with the CIC model for each core yield sedimentation rates for discrete years; these were simply averaged together for sites on the banks and for those in the marsh interior. This yields a record of average annual sedimentation rate that can be compared with measurements of average annual sea level and average annual SSC calculated from rating curves. Note that the analysis includes only data from 1978 (start of SSC measurements) to 2011 (core collection). Annual sedimentation rate is negatively correlated with mean annual sea level for sites on the banks ($p = 0.07$, $r^2 = 0.17$) but positively correlated for those in the marsh interior ($p = 0.05$, $r^2 = 0.13$). Annual sedimentation rate is also positively related (though not significantly) to SSC on the banks ($p = 0.11$, $r^2 = 0.09$) and negatively in the marsh interior ($p = 0.25$, $r^2 = 0.05$). Relationships with calculated annual sediment loads were also explored and showed the same trends as SSC.

Annual sedimentation rate can be predicted both on the banks and in the marsh interior with multiple linear regression models that include mean annual sea level and SSC ($r^2 = 0.24$, $p = 0.03$ on the banks; $r^2 = 0.21$, $p = 0.04$ in the marsh interior). However, signs in the equations differ. Intuitively, one would expect accumulation rates to be positively correlated with both sea level (increased opportunity for sedimentation through increased frequency and duration of flooding) and SSC (increased supply of sediments available for deposition). On the banks, sedimentation rate is indeed positively correlated with SSC but negatively correlated with sea level. Conversely, in the marsh interior, accumulation rates are positively correlated with sea level but negatively correlated with SSC.

Discussion

Controls on marsh sedimentary environments: ecogeomorphologic interactions and anthropogenic-disturbance history

The first hypothesis (H1) can be tested by using the observed variability in sediment characteristics and long-term marsh accretion rates across the marsh platform as a natural experiment. We specifically tested whether sediment characteristics and accretion are influenced by elevation and distance to tidal channels and, further, whether the nature of accretion differs between two ecogeomorphic settings—river/tidal-channel banks dominated by species-poor plant assemblages and interior marsh regions populated by diverse plant communities. We found that, in general, sites on

banks have less coarse and organic material, as well as higher average accumulation rates, than those in the marsh interior. The strong correlation of coarse and organic content within marsh sediments indicates that the higher coarse content in the marsh interior is dominated by plant-derived material that generally has lower bulk density than the finer, more fluvially derived sediment comprising the fine fraction. Differences in sediment characteristics across sites can, therefore, be interpreted as reflecting higher sources of fluvial sediment to sites on the banks relative to those in the marsh interior, whose sediment accumulation depends more on plant production. Although the decrease in both coarse and organic content observed in older core sections could reflect changes in plant production and/or sediment supply (e.g., Nyman et al. 2006), it is more likely due to lack of organic-material preservation with depth in sediment cores (e.g., Andersen et al. 2011). Because the potential fluvial sediment supply ($\sim 10^6$ t/yr for the Potomac River) is much higher than possible from plant production (e.g., an average biomass production of 10^3 g/m² (D'alpaos 2011) applied to the area of DMP yields $\sim 10^3$ t/yr), accumulation rates are higher at sites on the banks. Indeed, Mudd et al. (2010) note a maximum organic production rate of ~ 2 kg m²/yr in *Spartina alterniflora* marshes in South Carolina, equating to $\sim 2 \times 10^3$ t/yr when applied to the area of DMP, with any accretion beyond that rate deriving from inorganic sources. This interpretation is also supported by the negative correlation of surficial coarse (plant) content within sediments and accretion rates of the marsh surface.

Site-scale variability of sediment accretion at DMP, as in other systems (e.g., Thorne et al. 2014), can be understood within the context of sediment supply with simple multiple linear regression models. Here, coarse content is a proxy for fluvial- vs. plant-dominated sedimentation, and elevation is a proxy of inundation frequency and length. In these models, accumulation is negatively related to coarse content—less sediment accumulates in plant-dominated environments—and positively related to elevation—more sediment accumulates at sites flooded more often and for longer periods. On the banks, models predict the pre-1976 and 1976–1990 average accumulation rates well, indicating that sediment supply dominates accretion. However, the model loses its predictive ability after 1990, indicating that other processes (linear or nonlinear) affect accretion. These processes could include higher water levels that promote increased deposition and increased exposure to storm-driven waves, enhancing erosion (Litwin et al. 2013), among others. Simple statistical relationships are not adequate when such complex and competing effects on net sediment accretion act on the margins of the marsh. In the marsh interior, the multiple linear regression models fail to predict accumulation rates for any time period, similarly indicating that other nonlinear processes control accretion and/or that elevation and CIRD do not adequately describe the complex

hydrodynamic processes that supply sediment to marsh surfaces. Sedimentation at the interior sites is likely to be dependent on sediment redistribution and/or complex sediment-vegetation interactions. These interactions may include a reduction in turbulence that facilitates particle settling (Cahoon and Reed 1995; Leonard 1997; Morris et al. 2002) or plant community turn-over in space and time (Hopfensperger et al. 2009a), which determines how much recalcitrant plant litter is produced and is incorporated in the soil profile (Windham and Meyerson 2003; Saintilan et al. 2013).

The second hypotheses (H2) can be tested by examining the nature of sediment accumulation before and after mining ceased in 1976 (Table 1). The robustness of CFCS and CIC regression fits to ^{210}Pb profiles shows that sediment accumulation differs between the banks and interior. Higher r^2 values indicate less variability of activities, and hence steadier accumulation over time. Profiles for sites in the marsh interior show this steady sediment accumulation, suggesting that accumulation rates have remained relatively unchanged over time ($\sim 0.14 \text{ g/cm}^2/\text{yr}$). In contrast, profiles for sites on the banks are more variable, showing a general decrease in accumulation rates over time. In particular, accumulation rates decreased significantly after 1976, consistent with other studies that show increased bank erosion (i.e., less accumulation) following the mining of coarse and gravel deposits (Litwin et al. 2013). In the postmining era, average mass accumulation rates are remarkably consistent ($\sim 0.14 \text{ g/cm}^2/\text{yr}$) between the banks and marsh interior, suggesting that the marsh has reached a steady state of sediment accretion (e.g., Friedrichs and Perry 2001), supporting H2. Note that the mass accumulation rates translate to slightly different linear accretion rates, given differences in bulk density between bank and interior sites. Average linear accretion rates are 4.7 mm/yr for the banks (average bulk density 0.30 g/cm^3) and 6.4 mm/yr (average bulk density 0.22 g/cm^3) for the interior. Interestingly, although these rates are lower than the post-1990 acceleration in RSLR, no changes in average sediment accretion have been observed after 1990. The lack of a signal at DMP could indicate that accretion is limited by sediment supply and/or other physical processes (e.g., wave erosion), and/or the signal is obscured with our relatively coarse-scale sampling resolution at the scale of the entire marsh. In any case, marsh accretion is not keeping up with the post-1990 acceleration in RLSR, challenging future marsh sustainability.

Response of marsh accretion to long-term environmental change and implications for future sustainability

The third hypothesis (H3) can be tested by evaluating marsh accretion within the context of environmental change (accelerated sea-level rise, declining fluvial sediment loads). While previous studies have examined the variability of Potomac River nutrient loads (total nitrogen and total phos-

phorus; Boynton et al. 1995; Ruhl and Rybicki 2010), changes in sediment loads and the associated impacts to wetlands have received less attention. Mean sea level at Washington, DC shows a consistent increase (Fig. 2) since measurements began in 1932. Using the same temporal intervals as for sedimentation rates for comparison, average sea level was $-0.06 \pm 0.06 \text{ m}$ before 1976, $-0.006 \pm 0.05 \text{ m}$ from 1976 to 1990, and $0.08 \pm 0.06 \text{ m}$ after 1990. Mean annual sea level is positively related with mean annual river discharge ($r^2 = 0.31$, $p < 0.001$), indicating that both respond to similar storm events (e.g., rainfall and winds). However, they differ in longer-term forcings (e.g., climate and interannual differences in precipitation, respectively), such that there is no consistent longer-term trend in river discharge.

Calculated mean annual SSC have decreased since measurements began in 1978 ($r^2 = 0.26$, $p < 0.001$). Mean annual SSC was $0.03 \pm 0.01 \text{ kg/m}^3$ from 1978 to 1990 and $0.02 \pm 0.01 \text{ kg/m}^3$ after 1990. The relationship of SSC and river discharge has changed over time, evidenced by temporal changes in the sediment rating curves. In particular, there is a change in the vertical offset of these curves (see Fig. 3b), indicating that less sediment is currently carried in suspension than in the past. This decrease in SSC likely reflects decreased sediment production from the watershed, in response to management goals in recent decades. In particular, the Chesapeake Bay Agreement (signed in 1987 by the governors of the states surrounding the Bay, as well as federal partners) targeted a 40% reduction in nitrogen and phosphorous entering the Bay (CBP 1987) and resulted in an emphasis on soil conservation practices. These practices are reflected in reductions in nutrient loading to the Potomac (Bricker et al. 2014; Ruhl and Rybicki 2010) and likely also sediment loading.

Sediment accretion on the banks is positively correlated with SSC but negatively correlated with sea level, as expected. This likely reflects the dominance of erosive processes that remove sediment from the marsh platform, resulting in decreased sedimentation rates at the same time as sea level has been increasing. Conversely, sediment accretion in the marsh interior is positively correlated with sea level but negatively correlated with SSC. Although causation cannot be invoked with these data, we suggest that the relationship reflects coincident changes in other variables, such as the observed shift in the vegetation community in recent years (Palinkas and Engelhardt 2013). While potential errors and unexplored factors may hide drivers of sediment accretion in the marsh interior (e.g., winter ice scouring, animal activity, or lodging of plant litter), it lends insight into potential relationships that should be explored further in future studies. In particular, our analyses link observed changes in sea level and SSC to sediment accumulation on the marsh platform, which is an important step

forward in understanding potential marsh resiliency to environmental change.

The best method of predicting the future sustainability of DMP, however, remains unclear. One approach is to evaluate SSC—computed annual SSC averages are both above and below theoretical thresholds for sustainable marshes (0.02 g/km^3 ; Kirwan et al. 2010), at least since 1978. While river discharge measurements began much earlier, application of any sediment rating curve to those data requires assuming an unknown past relationship of SSC to river discharge. In any case, SSC measurements may be both under- and over-predicted. Data were collected upstream of the marsh, so there may be more sediment in suspension adjacent to the marsh. More importantly, extremely high river discharges that carry the highest SSC are not well represented in the rating curves, resulting in under-predicted SSC.

Another approach is to evaluate whether fluvial sediment loads are capable of supporting marsh accretion in the middle Potomac River (i.e., the portion of river represented by the USGS gauge at Washington, DC and where DMP is located) at approximately the same rate of recent RSLR (8.7 mm/yr). Assuming that an average of the observed values for bank and interior sites is representative for middle Potomac marsh sediments, the “required” mass accumulation rate would be $0.23 \text{ g/cm}^2/\text{yr}$. Applying this value to the area of wetlands in the watershed (69.36 km^2 ; Tiner and Burke 1995) yields an annual “requirement” of 1.57×10^5 tons of sediment, which is $\sim 20\%$ of the computed annual sediment loads that average $0.82 \pm 0.72 \times 10^6 \text{ t/yr}$ (1978–2012). For comparison, sedimentation in marshes accounts for $\sim 20\%$ of the fluvial sediment load in the Corsica River, a tributary of Chesapeake Bay (Palinkas and Cornwell, 2012), and $\sim 45\%$ of the fluvial sediment load in Winyah Bay, South Carolina (Patchineelam et al., 1995). Note that the computed sediment loads for the Potomac River are likely under-predictions due to under-predicted SSC values. Comparison with previous studies (Gellis et al. 2004; Herman and Friedrichs 2010) suggests they are underestimated by $\sim 1/2$ – $2/3$. So, it seems that ample fluvial sediment exists to nourish the marsh platform, especially since the annual “requirement” for DMP includes sediment supplied by plant production. However, current accretion rates on the marsh platform (average of 4.5 mm/yr on the banks and 6.4 mm/yr in the marsh interior) are lower than the rate of RSLR since 1990 (8.7 mm/yr), suggesting that sediment accretion is not keeping up with recent RSLR and may, therefore, be limited by supply and/or physical erosive processes (e.g., wave attack). Thus, the relevant question for future studies needs to go beyond whether there is “enough” sediment carried by the river for future marsh sustainability. Enough sediment may be carried by a river but this sediment may not be enough to ensure marsh sustainability if the sediment is not transported to the marsh and deposited there. Thus, questions that address marsh sustainability also

need to include whether the net flux of sediment onto and away from the marsh platform places the marsh at equilibrium with RSLR, which is expected if the marsh is to remain at the same position within the tidal frame.

Conclusions

Study results can be synthesized into a conceptual model that considers processes within the ecogeomorphic context of tidal freshwater marshes, which are typically located at or near the turbidity maxima of tidal rivers and support high plant diversity and biomass at the mid- to high-elevations of the marsh interior. As such, tidal freshwater marshes differ considerably in structure and function from salt marshes and will, therefore, respond differently to short- and long-term environmental changes. Relative to the marsh interior, sediment on the banks accumulates faster and is more influenced by fluvially derived sediment that is finer (less coarse material) and less organic. Accumulation rates on the banks can be predicted with a multiple linear regression model using proxies for sediment supply (elevation and coarse content) until 1990, when other processes (e.g., sea-level rise and/or wave erosion) likely become more important. Accumulation rates in the marsh interior cannot be predicted with these proxies and are most likely controlled by sediment redistribution and/or complex sediment-vegetation interactions. While accumulation rates on the banks decreased after mining activities, current average accumulation rates are remarkably consistent across the marsh. DMP, like many systems, is threatened by accelerating sea-level rise and declining fluvial SSC over time. However, the suspended load carried by the Potomac River appears capable of supporting accretion on the marsh platform—if processes exist to transport sediment to the marsh surface and erosion is minimized.

This study highlights the need to consider marsh sedimentation ecogeomorphically. It also calls attention to the importance of sediment redistribution and sediment-vegetation interactions in tidal freshwater marshes, especially in the marsh interior, as well as physical processes active at marsh boundaries. Lastly, while SSC may be decreasing in rivers, sediment loads may be adequate to support accretion on the marsh platform. Future studies should focus on determining the sediment flux to and from marshes, rather than absolute values of SSC within a river.

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