Control of sediment deposition rates in two mid-Atlantic Coast tidal freshwater wetlands

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Abstract

Eustatic sea level rise and rapidly increasing coastal development threaten tidal freshwater wetlands. Sediment deposition is one process that affects their ability to maintain surface elevations relative to adjacent rivers. Sediment dynamics in salt marshes have been studied extensively, but little is known about the factors that control sediment deposition rates in tidal freshwater wetlands. We examined geomorphic, hydrological, and biotic factors that may influence sedimentation in two tidal freshwater wetlands that fall at opposite ends of the riverine–estuarine continuum. Our data demonstrate that sediment dynamics are highly variable among tidal freshwater wetlands, and are influenced by the location of the wetland on the continuum. Sediment deposition was up to 10 times higher during the growing season at the downstream site than the upstream site. Plant density and height were highly correlated with sediment deposition rates at the downstream site ($r = 0.92$, $p < 0.009$) but not at the upstream site. Elevation, flood depth, and flood duration were correlated with deposition rates only when each site/season combination was considered separately. River suspended sediment and surficial floodwater suspended sediment concentrations were significantly higher at the downstream site ($p = 0.02$ and $p = 0.04$, respectively). These data suggest that vegetation is important in determining sediment deposition rates when river suspended sediment is not limiting, which is not always the case. Longer flood duration increased sediment deposition, but was of secondary importance. Land use and proximity to the turbidity maximum (near the forward extent of the salt water intrusion) appear to be critically important in determining river suspended sediment availability in the tidal freshwater zone of the Mattaponi River, VA.

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1. Introduction

The persistence of tidal freshwater wetlands depends on their ability to maintain surface elevation with respect to adjacent river elevation (Callaway, Nyman, & DeLaune, 1996), which in turn is controlled by the balance of competing processes. Vertical accretion is determined by autogenic primary production and sediment deposition. Conversely, loss of elevation results from erosion and subsidence (Cahoon, Reed, & Day, 1995; Cahoon & Lynch, 1997). Small changes in the elevation of either wetland or river can have far-reaching effects on sediment deposition rates. The loss of wetlands in the Mississippi River delta is a rapid, dramatic example of the negative impacts that may result from perturbations to these processes (Day et al., 2000; Turner, 1997).

Global climate change and local anthropogenic activity threaten the persistence of tidal freshwater wetlands in the mid-Atlantic region (Najjar et al., 2000). Mean global sea level rose 1–2 mm yr$^{-1}$ during the last 100 years and global warming is predicted to increase this rate two to five times by the year 2100 (Cahoon & Lynch, 1997). If sea level rise outpaces vertical sediment accretion, wetland loss will result everywhere that lateral wetland migration is not possible, and salt water intrusion will cause shifts in plant community composition (Burkett & Kusler, 2000; Callaway et al., 1996;
Winter, 2000). Locally, anthropogenic activity such as development of coastal areas, reservoir and water-withdrawal projects, and dam building can affect the complex interaction of sediment deposition, erosion, and subsidence that determine wetland surface elevation (Davis, 1997; Najjar et al., 2000). Sound ecological information is needed for managing sediment accretion in tidal freshwater wetlands so that it keeps pace with rising sea level.

Sediment dynamics in salt marshes have been well studied. Factors that have been implicated in controlling sediment deposition rates include elevation (Christiansen, 1998; Hensel, Pont, and Day, 1999), flood depth and duration (Asselman & Middelkoop, 1998; Cahoon et al., 1995; Christiansen, Wiberg, & Milligan, 2000), surficial floodwater velocity (Christiansen, 1998; Christie, Dyer, & Turner, 1999; Yang, 1998), river suspended sediment load (Asselman & Middelkoop, 1998; Blanton, Clark, Alber, & Kineke, 1999; Callaway et al., 1996; Christiansen, 1998; Christie et al., 1999; French, Spencer, Murray, Arnold, 1993), surficial floodwater suspended sediment load (Christiansen et al., 2000), proximity to sediment source (Brown et al., 1998; Callaway et al., 1996), and the presence of vegetation (Brown et al., 1998; Cahoon & Turner, 1989; Callaway et al., 1996; Christiansen, 1998; Christiansen et al., 2000; Hutchinson, Sklar, & Roberts, 1995; Yang, 1998). These studies suggest that there is a high degree of variability associated with sediment deposition, and that several factors may act synergistically to control rates in any given salt marsh (Leonard, 1997).

In contrast to the wealth of information on sediment dynamics in salt marshes, very little is known about factors that control sediment deposition rates in tidal freshwater wetlands. Several studies suggest that vegetation may play a key role. Sediment deposition rates were highest in the presence of plants in a freshwater tidal flat in the St Lawrence Estuary (Serodes & Troude, 1984). An elevation-based plant association index was highly correlated with sediment deposition in a river-mouth tidal freshwater marsh on the upper Chesapeake Bay (Pasternack, Hilgartner, & Brush, 2000). Sediment deposition was highest on vegetated creek banks during the growing season in a tidal freshwater marsh on the Pamunkey River, VA (Neubauer, Anderson, Constantine, & Kuehl, 2002).

There may be important differences in sediment dynamics in tidal freshwater wetlands and salt marshes due to differences in the relative importance of riverine and marine influences (Odum, 1988). Suspended sediment concentrations, particularly of larger-size particles, may be higher at more fluvially influenced sites due to their greater proximity to upland sediment sources (Bokuniewicz, 1995; Ward, Kearney, & Stevenson, 1998). Differences in cumulative watershed areas and river discharge rates upstream vs. downstream may affect water velocity.

Tidal freshwater wetlands on the Mattaponi River, VA, are representative of relatively pristine mid-Atlantic wetlands, but are threatened by both global and local anthropogenic changes. Because of their location in the Coastal Plain, they are threatened with submergence by eustatic sea level rise. Local groundwater withdrawal contributes to subsidence rates as high as 3.8 mm yr⁻¹ in the area (Davis, 1997), and a proposed water-withdrawal project on the Mattaponi River may alter sediment dynamics in ways that currently are not well understood.

Our objective was to examine factors that may control sediment deposition in tidal freshwater wetlands. We hypothesized that; (1) a combination of geomorphic, hydrological, and biotic factors work synergistically to control sediment deposition rates; (2) that the relative importance of riverine and marine influences will vary depending on the position of an individual wetland along the riverine-estuarine gradient; and (3) summer floods will be the most important to overall sediment deposition due to the combination of high flood-borne sediment loads and abundant sediment-trapping vegetation.

2. Site description

The Mattaponi River originates in the Piedmont physiographic region of north-central Virginia, and flows southeast through the Coastal Plain where it joins the Pamunkey to form the York River, a tributary of the Chesapeake Bay (Fig. 1). The Mattaponi is microtidal

Fig. 1. Location of Gleason Marsh and Walkerton Marsh on the Mattaponi River, VA.
in the study areas, with spring tide ranges occasionally approaching 1.8 m at Walkerton, VA. Astronomical tides are semi-diurnal and are strongly influenced by wind. The Mattaponi watershed is largely forested (63% upland forest and 6% wetland forest), and over 20% of the basin is used for agriculture (14% crops and 8% pasture) (W. McShea, personal communication; Table 1).

We selected two study sites. Walkerton Marsh is located on the southwest bank of the Mattaponi River across from the town of Walkerton, VA, and Upper Gleason (herein Gleason) Marsh is located approximately 19 km downstream, on the south bank adjacent to Wakema, VA. Sandy point bars are prominent on the inner curve of the river meander at Walkerton (Fig. 2). This type of feature occurs as a result of fluvially dominated flow and is built up by bedload movement of coarse sediments (Ritter, Kochel, & Miller, 1995). Both sites are primarily freshwater marsh or shrub–scrub wetland, with small areas of freshwater swamp. At each site, we randomly located five transects roughly perpendicular to the upland margin (Fig. 2). Simple board walks were constructed along each transect to aid access and to minimize disturbance to the wetland surface. The walks were made from 15-cm wide boards supported by 1.9-cm diameter PVC pipes sunk about 3 m into the soil. They extended 80 m from the riverside edge of the vegetated mudflat into the interior. We expected sedimentary processes to be most active in these fringe areas. Twenty sample points were randomly located along each transect to aid access and to minimize disturbance to the wetland surface. The walks were made from 15-cm wide boards supported by 1.9-cm diameter PVC pipes sunk about 3 m into the soil. They extended 80 m from the riverside edge of the vegetated mudflat into the interior. We expected sedimentary processes to be most active in these fringe areas. Twenty sample points were randomly located along each transect. At Gleason transects 1–5 and Walkerton transects 2–5, two additional sample points were randomly located in the vegetated mudflat for a total of 108 sample points at Walkerton Marsh and 110 at Gleason Marsh. No mudflat sample points were located at Walkerton transect 1 because this area was not accessible during low tide.

Vegetation consisted primarily of freshwater species. Mudflat vegetation consisted almost entirely of Nuphar advena at both sites. The dominant low marsh vegetation at the sites included Pontedaria cordata, Leersia oryzoides, and Zizania aquatica var. aquatica, with Spartina cynosuroides occurring only at Gleason Marsh. The higher elevations were dominated by Acorus calamus and Zizaniopsis miliacea at Walkerton, and by Eriochar is quadrangulata and Carex stricta at Gleason. Fraxinus profunda and Nyssa biflora were the dominant vegetation in swamp areas at both sites. Transitional oligohaline-freshwater species were more common at Gleason than at Walkerton; Zizania aquatica was found at both sites, but S. cynosuroides and E. quadrangulata were found only at Gleason. There was no obvious low marsh–high marsh zonation at either site; instead there was an elevation-dependent vegetation mosaic (P.P. Coulling, personal communication).
3. Methods

3.1. Soil characteristics

Soil characteristics were determined for all sample points on transect 3 at each site. In August 1999, three soil cores (0–10 cm) were collected at each sample point and composited in polyethylene bags. Samples were stored on ice during the return trip to the laboratory where they were stored at 4°C. Prior to analysis the samples were mixed thoroughly by hand and coarse root material was removed. Soil pH was determined by pH electrode in a 1:2 slurry of fresh soil : de-ionized water. Soil texture was determined by the Buoyouccos method (Day, 1965) after pre-treatment with H2O2 to remove organic matter (Gee & Bauder, 1986). Percent C and N were determined using a Perkin-Elmer 2400 Series II CHNS/O Analyzer. In April 2000, separate soil cores (approximately 0–28 cm depth) were collected for bulk density determination using a cryogenic coring device (Cahoon, Lynch, and Knaus, 1996). Freezing the soils in place minimized compaction, de-watering, and loss of flocculent material at the water–sediment interface. Frozen samples were stored on ice during the return trip to the laboratory where they were stored frozen until analysis. A band saw was used to slice the frozen cores into 2 cm segments. Because the segments were frequently irregularly shaped, their volumes were determined by water displacement. Displacement was estimated by changes in head pressure using a Microswitch 40 PC pressure transducer connected to the bottom of a graduated cylinder. Segments were dried to a constant weight in a drying oven at 80°C (generally 3 days), cooled and weighed.

3.2. Sediment deposition

We determined sediment deposition rates on several different time scales. Sediment traps were deployed for 24-h collection periods during spring tides when sediment deposition rates are highest (Dyer, 1995; Guzennec, Lafite, DuPont, Meyer, & Boust, 1999; Lindsay, Balls, & West, 1996). These deposition rates were used primarily to identify relationships with hydrological factors that were measured simultaneously. We also deployed sediment traps for 30-day (d) periods to identify seasonal patterns and to estimate annual sediment deposition rates. In order to integrate major episodic deposition and erosion events, we installed feldspar marker horizons to determine vertical accretion rates over a period of approximately 19 months.

Sediment was collected on ceramic tiles (15 cm × 15 cm) placed glazed-side up and flush with the soil surface (Pasternack & Brush, 1998). Preliminary tests showed no statistically significant difference in the amount of sediment deposited on the glazed vs. unglazed-side of the tiles. Where necessary, 20 cm × 20 cm pieces of wire mesh were used to support the tiles. Tiles were deployed for a 24-h period at roughly 2-month intervals beginning in February 1999 and ending in April 2000, for a total of seven sample intervals. There were 20 non-mudflat sample points per transect for a total of 100 samples per site per interval. From March 1999 through April 2000 (except for January 2000) tiles were deployed monthly for approximately 30-d periods at all mudflat and odd numbered non-mudflat sample points on each transect (n = 60 at Gleason; n = 58 at Walkerton). When the tiles were retrieved, fallen vegetation was trimmed at the edges, and the bottoms and edges were brushed free of sediment. The tiles were placed in 41 plastic bags and returned to the laboratory, where they were stored at 4°C until they were processed. Sediment and associated plant material were scraped from each tile into a 2-mm sieve, then washed through the sieve into a pre-weighted aluminum dish (75 ml for 24-h tile deployment and 500 ml for 30-d tile deployment) using a spray bottle filled with de-ionized water. Sediment samples were oven-dried to a constant weight at 80°C, cooled in a dessicator, and weighed.

Vertical accretion was estimated as the accumulation of sediment above feldspar marker horizons (Cahoon et al., 1995) laid on the marsh surface at approximately seven equidistant sample points per transect 1, 3, and 5 at each site between June and August, 1999. This gave a total of 22 sample points at Walkerton and 20 points at Gleason. In January 2001, about 19 months later, we were able to re-locate marker poles of 19 plots at Walkerton and 18 at Gleason. Feldspar horizons were found at all 19 of the remaining plots at Gleason and 14 of the 18 remaining plots at Walkerton. Of the nine missing marker horizons, eight had been installed relatively close to the river, and one was near the upland end of a transect. Although we cannot rule out the possibility that erosion caused the loss of marker horizons (Cahoon & Lynch, 1997), we suspect that some patchy feldspar layers were buried because there were no obvious signs of marsh surface erosion relative to the board walk in those places. Cryogenic cores (Cahoon et al., 1996) were taken through the remaining marker horizons in January 2001, and the depth of the marker below the surface was measured using a digital caliper. Where technical problems prevented cryogenic coring, measurements were taken by vertically slicing the marsh surface with a thin metal ruler to expose the clay layer. A minimum of three measurements were taken at each clay pad.

3.3. Vegetation

The vegetation was quantified approximately bi-monthly on transect 2 at each site from February 1999 through April 2000. The density of stems greater than 5 cm in height was determined in 0.25 m² quadrats.
surrounding each non-mudflat sample point. The median height of stems greater than 5 cm and the percent cover of dead vegetation were visually estimated in each quadrat. Only living vegetation was included in calculations of stem density and median height because senescing vegetation quickly became bent and matted.

### 3.4. Hydrodynamics

Current velocity and flood depth were measured at each sample point on transect 2 of both sites during the time periods when the bimonthly, 24-h sediment-collection tiles were deployed. Measurements were made at 1 h intervals for a total of 6 h, centered on the high tide, with a Marsh-McBirney Flo-mate 2000 portable water flow meter. Velocity was measured at 20 and 60% beneath the surface when the flood depth was greater than 15 cm.

### 3.5. Elevation

The horizontal positions and elevations of the sample points were surveyed with a Trimble 4700 Global Positioning System (vertical precision ±1 cm). Elevation data was not obtained for transect 1 at Walkerton because tree cover interfered with the GPS signal transmission.

### 3.6. Total suspended solid concentration

Surficial floodwater grab samples (250 ml) were taken at each sample point on transect 2 of both sites during the time period that the bimonthly, 24-h sediment-collection tiles were deployed. Samples were taken just beneath the surface when flood depths were greater than 15 cm. Collections were made at 1-h intervals for a total of 6 h centered on the high tide. Water samples were collected approximately 0.3 m below the surface for determining river total suspended sediment concentrations upriver of each site, using an ISCO 3700 C automated water sampler. The ISCO sampler was located on a dock that extended over the mudflat to the river channel at each site. Water samples (250 ml) were collected hourly for 24 h. In the laboratory, the surficial floodwater and river water samples were filtered through 1 μm pre-combusted, pre-weighed Gelman glass fiber filters, rinsed three times with de-ionized water, oven-dried to a constant weight at 60°C, and weighed to 0.1 mg (Franson, 1989; Leonard, 1997).

### 3.7. Statistics

SAS System for Windows, version 8, was used for all statistical analyses (SAS, 1999). Sediment deposition rates, plant density and median height, and surficial floodwater and river suspended sediment concentrations were log10-transformed to normalize the data. To facilitate log-transformation of zero values for plant density and height, 1.0 was added to each value prior to log10-transformation and averaging. Relationships between data sets were analyzed using Pearson’s correlation analyses (proc corr). Site means for general soil characteristics were compared using Student’s t-tests (proc t-test). Overall site means comparisons for seasonal data were made using paired t-tests (proc univariate).

### 4. Results

#### 4.1. Soil characteristics

The study areas fringing the river at Gleason Marsh and Walkerton Marsh differed significantly in basic soil characteristics (Table 2). Walkerton soils were silty and slightly acid while Gleason soils were clayey and circumneutral. The percent C content was higher at Walkerton than at Gleason Marsh. In non-calcareous soils such as those at our sites, the inorganic C content is generally negligible, so we assumed percent C to represent organic C (Nelson & Sommers, 1996). Thus, Walkerton had organic soils with roughly 31% organic matter (estimated by multiplying percent organic C by a factor of 2) and low bulk densities. Gleason soils were mineral (approximately 12% OM) and bulk densities were concomitantly higher. Percent N was higher at Walkerton than at Gleason because most soil N is organically bound (Mitsch & Gosselink, 2000).

#### 4.2. Sediment deposition

Sedimentation rates at Gleason Marsh were 2–10 times higher than at Walkerton Marsh during the growing season, but rates at the sites were similar

<table>
<thead>
<tr>
<th>Site</th>
<th>pH</th>
<th>% Sand</th>
<th>% Silt</th>
<th>% Clay</th>
<th>% C</th>
<th>% N</th>
<th>Bulk density g cm⁻³</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gleason</td>
<td>6.3</td>
<td>3.8 (0.4)</td>
<td>27.3 (0.5)</td>
<td>68.9 (0.5)</td>
<td>5.9 (0.1)</td>
<td>0.3 (0.0)</td>
<td>0.35 (0.03)</td>
</tr>
<tr>
<td>Walkerton</td>
<td>5.8</td>
<td>9.1 (1.2)</td>
<td>77.0 (1.1)</td>
<td>13.6 (0.7)</td>
<td>15.7 (0.8)</td>
<td>1.0 (0.1)</td>
<td>0.19 (0.03)</td>
</tr>
</tbody>
</table>

The two sites were significantly different with respect to all characteristics ($p \leq 0.007$, $n = 11$ for bulk density, $n = 22$ for all other variables). Values are means (SE).

a Statistical analysis applied to [H⁺]; Gleason site mean [H⁺] = $4.98 \times 10^{-7}$ (9.14 × 10⁻⁸) and Walkerton site mean [H⁺] = $1.44 \times 10^{-6}$ (1.96 × 10⁻⁶).
during the non-growing season (Fig. 3). This pattern was evident in both the 30-d and 24-h collection interval data, and it resulted in a significant difference between the sites when the 30-d interval data were averaged across a full year and when the 24-h interval data were averaged across the growing season (Table 3). Short-term deposition rates based on 24-h collection periods were higher and more variable than rates based on 30-d collection periods (Fig. 3). The 24-h collection periods were biased toward higher rates because the sampling occurred only during spring tides, and they did not integrate across as wide a range of conditions as the 30-d collections (Hutchinson et al., 1995). Sediment deposition rates were relatively high in the vegetated mudflat, contributing disproportionately to the overall site means. However, the difference between the two sites in sediment deposition rates did not change when means were calculated using only the non-mudflat sample points. (Table 3).

Vertical accretion, which integrates sediment deposition, autochthonous aboveground organic matter production, and erosion, was significantly higher at Gleason (2.7 ± 0.3 cm yr⁻¹) than Walkerton (1.2 ± 0.2 cm yr⁻¹; Table 3). We were concerned that the measurements we made by vertical slicing with a metal ruler might have had a higher potential for error than those made on cryogenic cores, so we re-calculated vertical accretion using only cryogenic core measurements. Results were similar and again significantly higher at Gleason (3.0 ± 0.4 cm yr⁻¹) than Walkerton (1.0 ± 0.2 cm yr⁻¹; p = 0.0005). Our results may underestimate annual vertical accretion rates because vertical accretion was measured for a 17–19 month period that included two

<table>
<thead>
<tr>
<th>Site</th>
<th>Gleason Marsh</th>
<th>Walkerton Marsh</th>
<th>p-valuea</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vertical accretion (cm yr⁻¹)b</td>
<td>2.7 ± 0.3</td>
<td>1.2 ± 0.2</td>
<td>0.0001</td>
</tr>
<tr>
<td>30-d sediment deposition including vegetated mudflatsc (g m⁻² d⁻¹)</td>
<td>12.4 ± 2.9</td>
<td>3.5 ± 0.4</td>
<td>0.011d</td>
</tr>
<tr>
<td>30-d sediment deposition with out vegetated mudflatsc (g m⁻² d⁻¹)</td>
<td>8.5 ± 1.7</td>
<td>2.9 ± 0.4</td>
<td>0.015d</td>
</tr>
<tr>
<td>24-h sediment depositionc (g m⁻² d⁻¹)</td>
<td>24.7 ± 10.9</td>
<td>6.7 ± 1.8</td>
<td>NS²</td>
</tr>
<tr>
<td>24-h sediment deposition, growing season onlyc (g m⁻² d⁻¹)</td>
<td>43.7 ± 15.0</td>
<td>3.8 ± 1.0</td>
<td>0.049d</td>
</tr>
<tr>
<td>Surficial suspended sedimentc (mg l⁻¹)</td>
<td>21.1 ± 3.2</td>
<td>7.1 ± 0.8</td>
<td>0.040d</td>
</tr>
<tr>
<td>River suspended sedimentf (mg l⁻¹)</td>
<td>18.8 ± 2.4</td>
<td>8.5 ± 3.4</td>
<td>0.021d</td>
</tr>
<tr>
<td>Flood depthc (cm)</td>
<td>13.5 ± 2.9</td>
<td>22.4 ± 5.2</td>
<td>NS</td>
</tr>
<tr>
<td>Flood durationc (h)</td>
<td>3.2 ± 0.6</td>
<td>3.9 ± 0.8</td>
<td>NS</td>
</tr>
<tr>
<td>Flood water velocityc (cm s⁻¹)</td>
<td>2.4 ± 0.3</td>
<td>1.5 ± 0.2</td>
<td>0.007</td>
</tr>
</tbody>
</table>

The sediment deposition rates based on 24 h collection periods were determined at or near spring tides. Surficial suspended sediment, river suspended sediment and flood depth, duration, and velocity were measured during the 24 h sediment-collection periods. Measurements at the two sites were not simultaneous. All values are mean ± SE.

a p-Values for paired t-tests, except for vertical accretion where p-value is for Student’s t-test with pooled variances. NS, no significant difference.
b Site mean ± SE for feldspar markers that could be located at the end of study (n = 14 at Gleason, n = 19 at Walkerton).
c Mean ± SE of monthly site means.
d Data were log-transformed to reduce heterogeneity in error variances.
² Mean ± SE of transect 3 monthly means based on three hourly measurements before and after high tide at each sample point.
³ Mean ± SE of monthly means based on 24 hourly samples taken just upstream of each site.

Fig. 3. Sediment deposition rates at Gleason Marsh and Walkerton Marsh. Values are site means ± SE, calculated from sediment accumulation (a) during approximately 24-h collection periods, at or near spring tides and (b) during approximately 30-d collection periods.
winters, when sediment deposition rates are low compared to the growing season.

4.3. Vegetation

Stem density peaked at both sites in July (Table 4). Values were similar to peak densities reported by Leonard and Luther (1995) for a North Carolina salt marsh (176–370 stems m⁻²). At the end of the growing season, senescing vegetation first became bent and matted, then was largely decomposed or carried away, leaving a patchily distributed, thin wrack layer throughout the year. There was a strong relationship between sediment deposition and plant density or height at Gleason, but not at Walkerton (Fig. 4). When vegetated mud flat values are excluded, the relationships remain the same (r = 0.93, p = 0.0077 for both density and height).

4.4. Hydrodynamics

We used correlation analysis to examine the relationship between sediment deposition and hydrodynamic variables including flood depth, flood duration, and surficial floodwater velocity. These three variables are interrelated because flood depth strongly influences both flood duration and velocity. We found no strong, statistically significant relationships (r ≥ 0.6, p ≤ 0.05) in data pooled across sites and seasons. The relationships varied by site and season. Maximum flood depths measured during the bimonthly 24-h sediment trap deployments were positively correlated with sediment deposition rates at Gleason in the winter, spring, and fall, but not during the summer when vegetation density would have had its strongest influence on deposition rates (Fig. 5). At Walkerton, maximum flood depths were positively correlated with sediment deposition rates in the summer as well as in winter and spring (Fig. 5). There is no obvious reason for the breakdown in the relationship in the fall at Walkerton. Flood duration followed a similar pattern, having a positive correlation with sediment deposition rates in the summer at Walkerton, but the absence of such a relationship at Gleason (Fig. 6). Surficial floodwater velocities were low, generally between 0 and 3 cm s⁻¹ (Table 3), and not significantly associated with sediment deposition rates.

4.5. Elevation

The relationship between mean annual sediment deposition rates (log-transformed) and elevations was stronger within sites (r = −0.64, p < 0.0001 at Gleason; r = −0.70, p < 0.0001 at Walkerton; Fig. 7) than pooled across both sites (r = −0.51, p < 0.0001). Elevation is largely a surrogate for flood depth and duration because higher points on the marsh are generally inundated less frequently and for shorter duration than lower points. The sediment deposition vs. elevation relationship in Mattaponi tidal freshwater wetlands is similar to that found by Pasternack et al. (2000) in a tidal freshwater marsh (r = −0.66, p < 0.0001), and Leonard (1997) in a salt marsh (r = −0.53, p < 0.0001).

4.6. Suspended sediment

Surface water suspended sediment concentrations in the Mattaponi River (Table 3) were similar to those reported for the Hudson River Estuary (4–91 mg l⁻¹; Feng, Cochran, & Hirschberg, 1999). Overall, surficial floodwater suspended sediment concentrations and river suspended sediment concentrations were significantly higher at Gleason than Walkerton (Table 3), suggesting a link between river suspended sediment, surficial floodwater suspended sediment, and sediment deposition rates. However, correlations were generally weak (i.e. r < 0.6) or not statistically significant (i.e. p ≥ 0.05).

5. Discussion

Tidal freshwater wetlands are a distinct class of estuarine ecosystem characterized by biogeochemical cycles and biota typical of freshwater environments, but a flooding regime typical of estuarine environments.
Nevertheless, our results suggest that these wetlands can vary considerably in soil characteristics and sediment dynamics depending on the relative importance of marine vs. fluvial influences. Plant associations identified at Gleason and Walkerton reflect these influences (P. Coulling, personal communication). The downriver plant associations included more freshwater–oligohaline transitional species, reflecting the fact that Gleason Marsh is located closer to the freshwater–oligohaline interface. Upriver at Walkerton, the dominant species were almost entirely freshwater-adapted, indicating a greater fluvial influence.

5.1. Soil characteristics

At Walkerton, marsh fringe soils were more organic and concomitantly had lower bulk densities than at Gleason (Table 2). Lower mineral sediment inputs (Table 3) were likely responsible for the higher organic matter concentrations at Walkerton. The 12% organic matter concentration at Gleason is low and more typical of a salt marsh (10–40%) than a tidal freshwater marsh (20–70%; Odum, 1988). However, it is similar to nearby Sweet Hall Marsh, a tidal freshwater wetland on the Pamunkey River where percent C was approximately 5.8
Soil particle sizes tended to be larger at Walkerton (Table 2) due to the size-sorting that occurs when particles move downstream (Ritter et al., 1995). Larger particles settle out first upriver, in conformity with Stokes’s Law, leaving the smaller particles suspended. Under appropriate conditions of concentration and salinity, floc formation may favor the deposition of smaller clay-sized particles farther downriver that would otherwise remain in suspension (Dyer, 1995; Nichols & Biggs, 1985).

The higher soil pH at Gleason is likely due to larger exchangeable base inputs of marine origin downriver (Table 2). However, organic acid accumulation at Walkerton may contribute to the difference.

5.2. Sediment dynamics

Sediment deposition was spatially and temporally variable in Mattaponi tidal freshwater wetlands. During the year-long study, rates ranged from 0.8 ± 0.2 to 73.2 ± 22.5 g m⁻² d⁻¹ (Fig. 3a), and were comparable to rates in a North Carolina salt marsh (13.8 ± 3.4 to 63.7 ± 10.3 g m⁻² d⁻¹; Leonard, 1997). During the winter and early spring, sediment deposition rates at Gleason Marsh and Walkerton Marsh were low and similar (Fig. 3). During the growing season, sediment deposition increased dramatically at Gleason, but not at Walkerton. A growing season peak in sediment deposition has been observed previously in a tidal freshwater marsh at the mouth of the Bush River, a tributary of the upper Chesapeake Bay (Pasternack et al., 2000; Pasternack & Brush, 2001) and on the Pamunkey River, a tributary of the York River (Fig. 1; Neubauer et al., 2002). Peak sediment deposition at the Pamunkey River site occurred in July 1999, coincident with peak deposition at Gleason Marsh. Similar seasonal trends have been reported in some salt marshes (Leonard, 1997), but not in others (Kastler & Wiberg, 1996).

The seasonality we observed suggests that vegetation plays a role in controlling sediment deposition at Gleason, and indeed there was a strong correlation between sediment deposition rates and plant density or median height at this site (Fig. 4). It has frequently been suggested that vegetation affects sediment deposition
rates in salt marshes (Brown et al., 1998; Callaway et al., 1996; Christiansen et al., 2000; Hutchinson et al., 1995; Leonard, 1997; Yang, 1998) and tidal freshwater marshes (Neubauer et al., 2002; Pasternack et al., 2000) by reducing the surficial floodwater velocity and stabilizing the marsh surface. Direct laboratory manipulation of water flow through real and artificial salt marsh grasses have confirmed that increased plant density reduces water velocity (Burke, 1982; Pethick, Leggett, & Husain, 1990; Shi, Pethick, Burd, & Murphy, 1996). Nonetheless, a direct link between vegetation and sediment deposition rates has not been established in situ experimentally. The apparent site-specific response in Mattaponi tidal freshwater wetlands indicates that other factors must be important. It seems unlikely that small differences in plant community composition and structure would yield such dramatic differences in seasonal patterns of sediment deposition. We cannot rule out the possibility that some plant species may have a somewhat greater baffling effect than others, as found by Hensel et al. (1999) in a salt marsh, or some plants are more likely than others to collect suspended sediment on their stems and leaves (Leonard & Luther, 1995). In any case, sediment deposition rates are often insensitive to the particular plant assemblages present (French, Spencer, Murray, & Arnold, 1995; Leonard, 1997).

We originally expected river suspended sediment concentrations to be higher at the more fluvially influenced site due to its proximity to upland sediment sources (Bokuniewicz, 1995), but found that river total suspended sediment concentrations and surficial floodwater suspended sediment concentrations were higher at Gleason, the more estuarine-influenced site (Table 3). This suggests that the availability of sediment for potential deposition was higher at Gleason. Availability of sediment was an important control on sediment deposition rates in two small UK salt marshes (French et al., 1995) and a freshwater tidal flat in the St Lawrence Estuary (Serodes & Troude, 1984). We propose that dramatically higher sediment deposition rates at Gleason compared to Walkerton were due to an interaction between sediment availability and the seasonal presence of plants. The absence of a growing-season peak at Walkerton was due to an insufficient sediment supply.

![Fig. 6. Seasonal relationship between sediment deposition rates and flood duration at Gleason Marsh and Walkerton Marsh. Details as in Fig. 5.](image-url)
If this is indeed the case, how can we account for the critical difference in sediment availability at these sites? Two possibilities suggest themselves: (1) differences in cumulative sediment sources and land use above Walkerton and Gleason affect the availability of suspended sediment from upriver sources, and (2) the estuarine salinity gradient affects turbidity and flocculation differently at the sites.

5.2.1. Cumulative sediment sources and land cover

The river suspended sediment load at any one point represents the integration of river basin characteristics above that point (Syvitski, Morehead, Bahr, & Mulder, 2000). Sediment dynamics are commonly more active in downstream reaches because cumulative sediment inputs increase in a downstream direction (Ham & Church, 2000). Thus, the difference in sedimentation rates between sites is likely to be related to their location on the upriver–downriver gradient.

Land cover can have a significant influence on sediment dynamics in rivers. The percentage of land cover that is residential or agricultural is about the same above Walkerton as in the reach between Walkerton and Gleason. In contrast, the percentage of marsh and unvegetated mudflat increases nearly three-fold from about 3% above Walkerton to 8% between Walkerton and Gleason (Table 1). Marshes and mudflats are large reservoirs of sediment that can be resuspended, transported, and deposited further downstream (Ham & Church, 2000; Owens, Walling, & Leeks, 1999; Stanley & Hait, 2000). For example, about 40% of the river suspended sediment load of the River Tweed in Scotland is stored in the floodplain, compared to 4% stored in the river channel (Owens et al., 1999). Such sources increase in importance in the reach between Walkerton and Gleason, and may help explain relatively high sedimentation rates at Gleason Marsh. The decreased percent forested wetland downstream of Walkerton is consistent with this view because forested systems are less frequently flooded than marshes and mudflats and are generally more stable. However, because sediment deposition rates downstream were sometimes 10 times higher than at the upstream site, it seems that potential sediment sources cannot explain this difference entirely.

5.2.2. Salt water intrusion

The estuarine salinity gradient is potentially relevant to sediment deposition rates and river suspended sediment concentrations at the Gleason site because it is located near the freshwater–oligohaline transition zone. Although the tidal freshwater environment has, by definition, an average annual salinity ≤0.5, salinity is highly variable across seasons and years (Odum, 1988). For example, the long-term average at nearby Sweet Hall Marsh is 0.5, but the average in 1999 was 2.8 with a range of 0.0 to 15.9 (Neubauer et al., 2002). Estuaries generally have a turbidity maximum where suspended sediment concentrations are higher than they are either landward or seaward, and it is located near the head of the salt intrusion (Nichols & Biggs, 1985; Dyer, 1995). Zhuraleva and Morozova (1999) found that the turbidity maximum usually occurred at a salinity of approximately 2 in the Lower Dnieper River, Russia. The turbidity maximum may shift seasonally as a result of freshwater discharge (Fettweis, Sas, & Monbaliu, 1998), and well-developed turbidity maxima tend to be established during periods of low river discharge (Nichols & Biggs, 1985). If the freshwater/salt water interface was located near Gleason during the study period, the turbidity maximum would have provided large amounts of suspended sediment that was not available at the Walkerton site located 19 km upriver where the fluvial influence was greater. Similarly, Childers, Sklar, Drake, and Jordan (1993) found that sediment deposition in a mid-Atlantic Coast brackish marsh was highest at locations nearest the salt water/freshwater interface.

The turbidity maximum on the York Estuary is normally located near West Point or slightly landward.

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(1999. Between May and July 1999, mean discharges ranged from 0.6 to 1.0 m$^3$ s$^{-1}$ vs. a 57 yr mean of 8.2 to 11.4 m$^3$ s$^{-1}$ for those months (provisional data from R.K. White, USGS). Bottom salinity was 0 at Walkerton, but peaked at 15.1 at a site 13 km below Gleason Marsh on 9/7/99 (the mean during our study period was 6.2, $SE = 1.0$; provisional data from VADEQ). If Gleason was near the freshwater/salt water interface during the study period, the river suspended sediment differences observed at our two sites would have been influenced strongly by the estuarine turbidity maximum.

6. Summary and implications

Although tidal freshwater wetlands are a distinct class of estuarine ecosystem, our data demonstrate that sediment dynamics can vary dramatically and that part of this variability may depend on location along the fluvial–estuarine–marine continuum. Sediment deposition rates were highest at the estuarine end of the gradient because of more sediment sources and proximity to the estuarine turbidity maximum. We concluded that sedimentation rates are influenced by vegetation only when river suspended sediment is abundant. However, conclusive support for the widely held axiom of a vegetation effect on sediment deposition would require direct manipulation of the vegetation. Greater flood depth and duration increase sediment deposition, but they are of secondary importance to site differences such as proximity to the turbidity maximum. Data from additional sites along this riverine–estuarine gradient are needed to confirm these relationships.

It remains to be determined how sensitive tidal freshwater wetlands such as those on the Mattaponi and Pamunkey Rivers will be to sea level rise. Microtidal systems commonly depend on autochthonous organic matter production for vertical accretion because inorganic sediment inputs are low (Stevenson, Kearney, & Pendleton, 1985). Walkerton is a relatively low sedimentation site compared with other tidal freshwater wetlands. The mean sediment deposition rate for Walkerton, 0.13 ± 0.07 g cm$^{-2}$ yr$^{-1}$, is at the low end of the range of <1 to 30 g cm$^{-2}$ yr$^{-1}$ at nearby Sweet Hall Marsh (Neubauer et al., 2002) and an order of magnitude lower than a tidal freshwater wetland on the Bush River (mean sediment deposition rate for HaHa Marsh = 1.2 g cm$^{-2}$ yr$^{-1}$; Pasternack & Brush, 1998).

Walkerton Marsh not only receives a low sediment input compared to other tidal freshwater wetlands, but a large fraction of it is organic (24.0 ± 3.3% at Walkerton vs. 13.5 ± 1.0% at Gleason; J. Morse, personal communication). If the low sediment deposition rates and high soil organic matter content at Walkerton Marsh are typical, most microtidal tidal freshwater wetlands on the Mattaponi River may be highly dependent on autogenic production for keeping pace with sea level rise (Holm, Sasser, Peterson, & Swenson, 2000; Stevenson, Ward, & Kearney, 1986). Brackish marshes in the vicinity of the Blackwater National Wildlife Refuge on the eastern shore region of Maryland are microtidal systems where autogenic production has failed to maintain surface elevation (Stevenson et al., 1985). High rates of marsh loss in these systems are related to the long-term degradation of deeper peat layers in response to subsidence and relative sea level rise. However, the mean accretion rate at Blackwater was substantially lower than at Walkerton, approximately 4 mm yr$^{-1}$ vs. 12 mm yr$^{-1}$, and we do not know how rapid the influx of sediments will have to be in the future to minimize the risk of marsh loss due to sea level rise. It is also uncertain whether primary production will be maintained at current levels during the transition from tidal freshwater wetlands to brackish and monospecific salt marshes. Upriver movement of the turbidity maximum will help maintain marsh elevation by increasing mineral sediment deposition in nearby marshes, but this will not influence the extensive marshes well upstream of the turbidity maximum.

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