Sediment deposition patterns in *Phragmites australis* communities: Implications for coastal areas threatened by rising sea-level

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Abstract

The explosive expansion of the common reed *Phragmites australis* over the last 50 years in the wetlands of the U.S. mid-Atlantic has been of concern to biologists, resource managers and the general public. The replacement of *Spartina* spp. communities by the invasive *P. australis* has been widely reported, but the ecosystem effect of this replacement is poorly understood, especially with regard to sediment accretion processes and elevation change. It is hypothesized that a more detailed understanding of individual plant species and their role in marsh accretion may provide an improved ability to predict the effect of projected sea-level rise in coastal wetlands. Two coastal salt marsh sites on the Eastern Shore of Chesapeake Bay in Maryland (USA) were studied to quantify depositional environments associated with *P. australis*. Short-term sediment deposition (24 hr) and storm deposition (17 d) were measured using filter paper plates, and vertical accretion and elevation change (6 mo.) were measured using a marker horizon coupled with a sedimentation erosion table (SET). Greater rates of mineral and organic sediment trapping were associated with the *P. australis* community in both a subsiding creek bank marsh (34 $g \cdot m^{-2} \cdot day^{-1}$ in *P. australis* vs. 18 $g \cdot m^{-2} \cdot day^{-1}$ in *Spartina* spp.) and a laterally eroding marsh (24 $g \cdot m^{-2} \cdot day^{-1}$ in *P. australis* vs. 15 $g \cdot m^{-2} \cdot day^{-1}$ in *Spartina* spp.). Litter accumulation in *P. australis* stands is responsible for the higher depositional pattern observed. Additionally, below ground accumulation in *P. australis* communities (as much as 3 mm in 6 months) appears to substantially increase substrate elevation over relatively short time periods. Thus *P. australis* may provide resource managers with a strategy of combating sea-level rise and current control measures fail to take this into consideration.

Introduction

Over the past century the earth has warmed about 0.5 °C and surface temperatures are projected to increase by 1.5–4.5 °C when CO_2 concentrations in the atmosphere eventually double (Mahlman, 1997). Consequently, there are expectations that sea-level rise will follow suit and dramatically accelerate over the next century (Titus, 1986; Warrick and Oerlemans, 1990; Gornitz, 1995). While their predictions are undoubtedly high, climate models suggest an acceleration of sea-level rise to 50 ± 25 cm by the year 2100 (Douglas, 1991; Mahlman, 1997). One of the landscape types which will be most negatively affected by this increase in water level will be marshes (Titus

et al., 1984). Considerable marsh loss is expected in sediment starved coastal zones (Stevenson and Kearney, 1996) and especially in deltaic systems where sediment inputs are restricted by dams or levees (Baumann, 1984; Coleman, 1988; Day and Templett, 1989; Boesch et al., 1994). However, the details of which particular types of marsh are most resilient to sea-level change have not been elucidated.

One approach to predicting how sea-level rise will impact marshes is to compare sedimentation and accretion patterns in plant stands of different species. Eleuterius and Eleuterius (1979) postulated from field observations that plant responses to rising sea-level will vary between marsh species, but little quantitative data is available. Evidence does exist that some

marshes confronted with rising water levels decrease in above and below ground organic production, have depressed decomposition rates, and mineral sediment inputs increase as the marsh lowers in the tidal spectrum (Redfield, 1972, Good et al., 1982; Mendelssohn and McKee, 1988; Bricker-Urso et al., 1989; Cahoon and Reed, 1995). As a whole, coastal marsh response to rising sea-level is dependent on the importance of the inorganic and organic components of the sediment, and the effect of increased hydroperiod on net vertical accretion (Reed, 1995). Reed (1995) has hypothesized that by understanding how various plant species contribute to marsh accretionary processes, we can begin to assess the differential effects of sea-level rise on those communities and their sustainability.

The rapid spread of *P. australis* has been documented in Chesapeake Bay over the last 50 years (Rice and Stevenson, 1996; Rice et al., in press), and despite how little is known about its role in the ecosystem, it is being actively controlled on federal and private land throughout marshes of the Bay region (Marks et al., 1994). Field observations suggest the substrate in *P. australis* communities is higher in elevation than in other nearby plant assemblages. Here we report preliminary results (from an ongoing study) of short-term sediment deposition events (24 hour and storm deposition, and six month integrated estimates of vertical accretion and elevation change) in *P. australis* and mixed *Spartina alterniflora*, *Spartina patens* and *Spartina cynosuroides* communities in two coastal marshes of mid-Chesapeake Bay, Maryland. The two study sites were chosen to independently contrast lateral erosion and subsidence (the two primary causes of marsh loss in Chesapeake Bay) (Kearney et al., 1988; Wray et al., 1995; Stevenson et al., 1999) and their effects on sedimentary processes in the two community types. We provide evidence that *P. australis* can have a positive effect, at least in the short term, on stability (or the ability of the marsh environment to exist over time) in the two marshes studied here.

Methods

Study sites

Two marsh sites were selected at the Deal Island Wildlife Management Area (DIWMA) on Maryland's Eastern Shore of Chesapeake Bay (Figure 1A-C). DIWMA is adjacent to Tangier Sound located between Monie Bay and Manokin River. The subsiding creek

bank site is within the Monie Bay National Estuarine Research Reserve of DIWMA (Figure 1A, C) and is composed of three sub-environments that include bank marshes, tidal channel bank marshes, and back marsh areas (Ward et al., 1988). Specifically, the site is in the lower portion of Little Creek, a first order stream, in a tidal channel bank marsh. An increase in number and width of first order streams in the reserve, such as Little Creek, has been documented from aerial photos between 1938 and 1995. Ward et al. (1988) suggest that larger areas of Monie Bay's marshes will be inundated during high tides or storm surges in the future, thereby enlarging existing channels as well as increasing the number of new creeks. A local sea-level rise of 4 mm·yr^{-1} will exacerbate this problem and cause increased submergence at Monie Bay, thereby producing an accretionary deficit that is expected to become more substantial over time (Kearney and Stevenson, 1991). The sedimentary environment at Little Creek is highly organic, and lack of inorganic sediment input is postulated to be the cause of marsh degradation in some areas (Kearney et al., 1994; Ward et al., 1998). In addition, continuing land subsidence from anthropogenic groundwater removal from aquifers is another prevalent cause of the observed increase in sea-level in this region (Kearney and Stevenson, 1994).

The eroding island site, Little Deal Island, is off the tip of Deal Island and is located between Tangier Sound and Manokin River (Figure 1A, B). The site is typical of many of the marsh islands in the Bay region that are degrading from erosive forces and expected to be greatly reduced or totally lost in the coming century (Kearney and Stevenson, 1991; Wray et al., 1995). Unlike Little Creek, which usually has salinity levels below 5, the salinity at Little Deal Island ranges from 7 to 15. Maps of the historical shoreline and erosion rate produced by the Maryland Geological Survey indicate that from 1849 to 1942 the island's coastline retreated in some areas by less than 0.5 m·yr^{-1} and in other areas by more than 2.5 m·yr^{-1} from wave erosion. As wave erosion continues, storm events provide the primary input of inorganic sediment to the marshes at Little Deal Island by depositing sand, silt and clay in storm tides to the marsh interior (as demonstrated by Stumpf, 1983; Reed 1988, 1989).

Both study sites have typical salt marsh vegetative communities which are dominated by *S. alterniflora*, *S. patens, S. cynosuroides*, and *P. australis*. Examination of photos dating back to 1938 suggests that *P. australis* is not particularly aggressive and only constitutes a minor vegetative component of marshes in
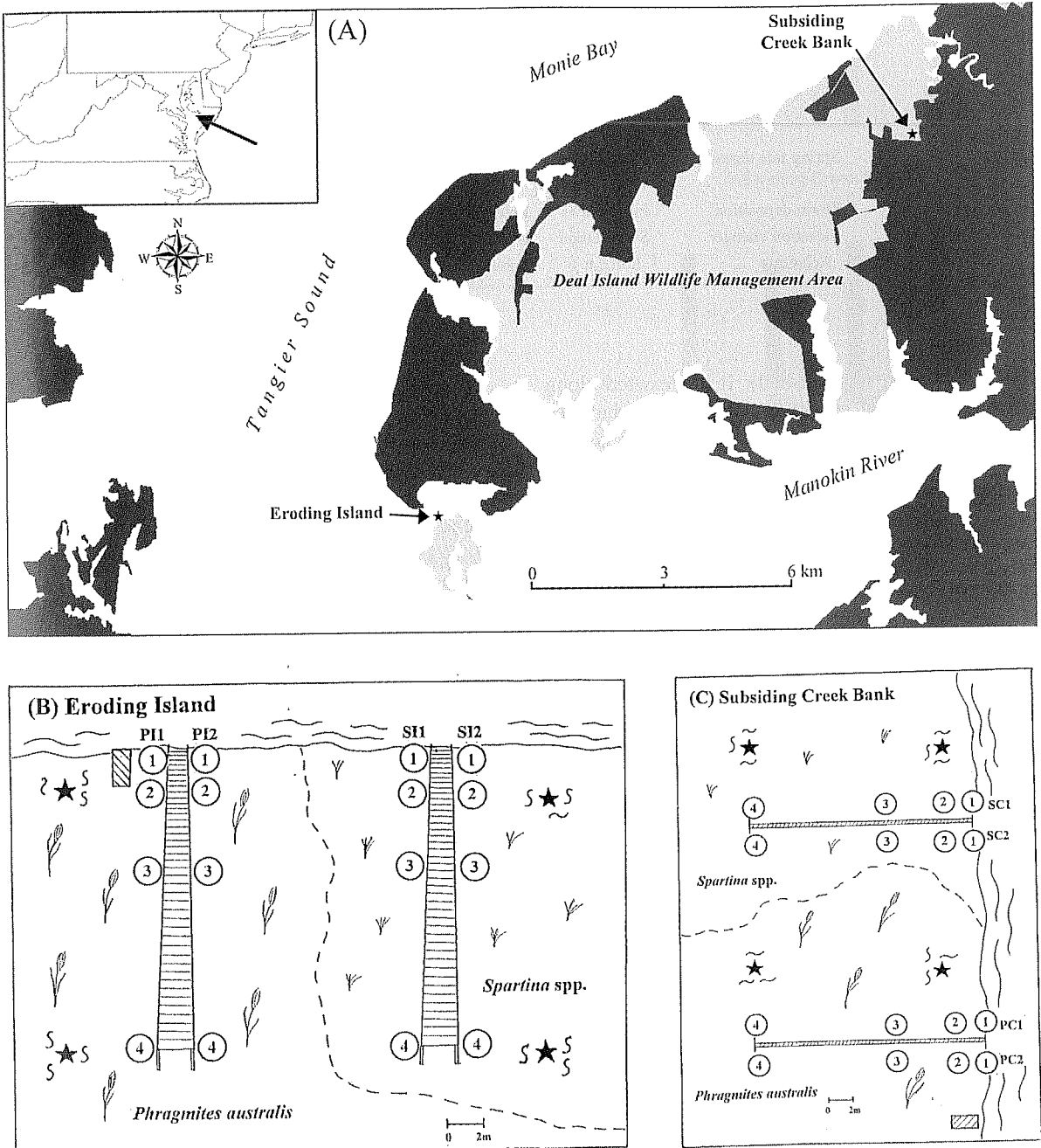
Figure 1. Site map of Deal Island Wildlife Management Area, Chesapeake Bay, Maryland (A), with detailed illustrations of the eroding island (B) and subsiding creek bank (C) sites. Open Circles = sediment trap stations; Filled Stars = SET pipes; ~ = marker horizons; Boxed Diagonal Hatching = tide gauges.

Table 1. Sampling protocol for December 1997 through June 1998.

Parameter sampled	Method	Site	Date
Neap tide deposition	Filter plates	Eroding Island	July 23–24
			Aug 12–13
			Sept 12–13
			Nov 21–22
		Subsiding Creek Bank	Aug 12–13
			Sept 12–13
Spring tide deposition	Filter plates	Eroding Island	Aug 5–6
			Sept 19–20
Storm deposition	Filter plates	Eroding Island	Feb 3–20
Elevation change/ Accretion	SET/Marker horizon	Eroding Island, Subsiding Creek Bank	Dec 1997-Jun 1998

DIWMA (Rooth, unpublished). It has existed along coastlines and tidal creeks in small stands that, for the most part, have not expanded over the last 30 years. Therefore, in this context, *P. australis* is a stable vegetative component of the Monie Bay system.

Sampling schedule and techniques

Short term deposition and elevation change rates were measured over varying time scales from July 1997 to June 1998 at the two sites in each of the vegetative communities of *P. australis* and *Spartina* spp. (Table 1). Two independent methods were used: filter paper plates (Reed, 1989) and a sedimentation erosion table (SET; Boumans and Day, 1993) coupled with a marker horizon (Richards, 1934; Cahoon and Turner, 1989). Plates were left on the marsh for short periods (usually from 24 hours to 2 weeks) in order to determine how seasonal events (spring and neap tides, storms, precipitation, etc.) affected organic and inorganic depositional patterns on the marsh surface (Reed, 1989). To measure deposition over a longer time scale (6 months), we used a SET to make a high precision measurement of elevation change and a marker horizon to measure the vertical accretion (Boumans and Day, 1993; Cahoon et al., 1995). Unlike other studies, our sites were underlain with a firm clay Holocene basement, which was penetrated by the aluminum base pipe, making it a stable datum. At each site, water level was measured at 5 minute intervals throughout the study period with a tide gauge and a data logger (Aquapod Water Level Measurement System, DACOM Technologies, Utah). Coastline loss was monitored at the eroding island via 8 wooden stakes placed in each vegetative community 2 m from

the marsh edge at a distance of approximately 3 m from one another along the coast (16 total stakes). Subsequent loss of shoreline was measured at the conclusion of the study and averaged for all 16 stakes.

The mid-Chesapeake Bay region had several prominent storm events or 'nor-easter's' in the winter of 1997/1998 that were characterized by rain and above-average wind speeds. Of those storms, 3 induced markedly higher water levels at the eroding island and subsiding creek bank site (Rooth, unpublished). The event in November 1997 had maximum sustained winds that exceeded 32 km·hr^{-1} and the two in February 1998 had sustained winds well above 45 km·hr^{-1} (NOAA CLIMVIS Station) that prompted both increased hydroperiod and level of inundation at the sites. This provided an opportunity to briefly examine the significance of storms on deposition patterns in the two different vegetative communities.

Boardwalks were constructed in each vegetative community at both sites to reduce overall marsh disturbance. The communities (each spanning approximately < 40 m along the tidal edge) were adjacent to one another to minimize differences in geomorphology, elevation, hydroperiod and total suspended solids in the water column. We established four permanent stations at 0 m, 2 m, 7 m, and 17 m from MLW along each of two transects running parallel to the boardwalks (labeled as PI1 and PI2 in the *P. australis* and SI1 and SI2 in the *Spartina* spp. at the eroding island; and PC1 and PC2 in the *P. australis* and SC1 and SC2 in the *Spartina* spp. at the subsiding creek bank) (Figure 1B-C). Stations #1 (0 m) and #2 (2 m) are 'coastal stations', and stations #3 (7 m) and station #4 (17 m) are 'marsh interior stations'.

For the eroding island site, sediment deposition was measured on four neap tides between July and November 1997, and two spring tides were sampled between August and September 1997. At the subsiding creek bank site, two neap and two spring tides were sampled between August and September 1997. Additionally, a storm event was sampled in February 1998, with plates deployed for a total of 17 days to encompass the delayed storm surge effects at station #4 at the eroding island site.

The 9-cm glass fiber filter paper plates were rinsed with DI water, dried, ashed and weighed before being fixed to inverted petri dishes with a plastic staple (as described by Reed, 1992). The plates were inserted in the sediment until they were flush with the surrounding surface. Triplicate plates were placed at each station along each transect for a total of 6 plates for each 24 hour measuring period. Some plates were lost due to tidal action or to small animal consumption. After collection, the plates were dried at 60 °C for at least 48 hours and weighed to obtain total deposition. The filters were combusted at 450 °C for 4 hours and re-weighed to determine organic deposition. Plates were averaged over all events for each vegetative type at each station for each site.

To measure the role the aboveground component of each plant species plays in deposition in the two communities, we harvested live and dead (standing + wrack) biomass in August 1997. Three quadrats (0.0625 m^{-2}) of biomass were collected approximately 6 m away from the boardwalk to avoid radically altering the sedimentary dynamics closer to the transects. Biomass was sorted, dried at 60 °C and weighed.

The SET and marker horizons were installed in December 1997 at station #2 and #4 in both the *P. australis*, *Spartina* spp. communities at both sites. Eight permanent base pipes were driven into underlying clay substrate anywhere from 2 m to 6 m below the sediment surface, and the pipes were assumed to be a stable datum for the period of study (Childers et al., 1993; Cahoon et al., 1995). At each pipe, a small sampling platform was erected to reduce disturbance to the marsh surface and measurement protocols followed. A total of 36 measurements per pipe were taken for a baseline elevation in December 1997. They were re-measured in June 1998 (six months after installation). The elevation change data was averaged for the two SET stations in each vegetative community at each site (n = 72).
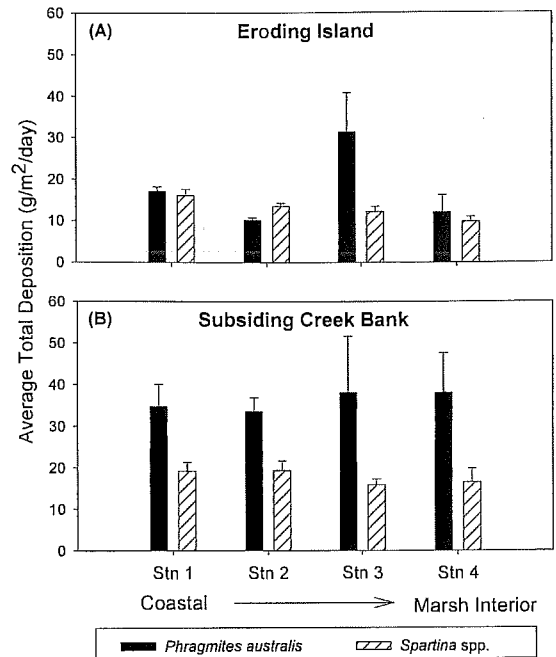

Figure 2. Average total deposition at Little Deal Island, MD, from data averaged from sampling conducted in July through November 1997 (A), and at Little Creek, Monie, MD, from data averaged from sampling conducted August and September 1997 (B), in *Phragmites australis* and *Spartina* spp. communities.

At every SET location, three 0.25 m^{-2} plots of feldspar were established as a marker horizon (Richards, 1934; Cahoon and Turner, 1989). The plots were cryogenically cored coincident with the 6-month SET readings to a depth exceeding the horizon (Cahoon et al., 1996). The depth to the marker horizon was measured to the nearest mm in four locations around the core and averaged to provide a single vertical accretion value for each plot. If no feldspar horizon was recovered, additional cores (up to 3) were collected. The accretion values for the plots were averaged together for both SET stations in each vegetative community at each site (n = 6).

Vertical accretion is measured as the height of accumulated material (mm) over the feldspar marker horizon. The SET measures elevation change over time, and when subtracted from the amount of vertical accretion is an estimate of the amount of marsh subsidence (Cahoon et al., 1995). A one-way ANOVA was performed on the elevation change and vertical accretion data to test for significant differences ($p < 0.5$) between sites and vegetative communities.
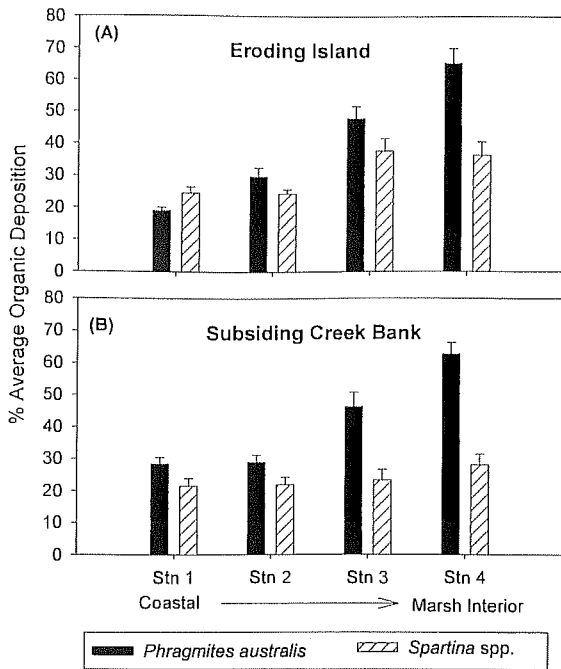
Figure 3. Average % organic deposition at Little Deal Island, MD, from data averaged from sampling conducted in July through November 1997 (A), and at Little Creek, Monie, MD, from data averaged from sampling conducted August and September 1997 (B), in *Phragmites australis* and *Spartina* spp. communities.

Figure 4. Dead biomass averaged from 3 quadrats of 0.0625 m^{-2} collected in August 1997 at Little Deal Island, MD (A), and at Little Creek, Monie, MD (B).

Results and discussion

24 hr and Storm deposition patterns

There was overall greater deposition on the filter paper plates in the *P. australis* stand than in the *Spartina* spp. communities at the eroding island site from July to November 1997 (Figure 2A). This difference, was even more apparent at the marsh interior stations, mostly due to the presence of high amounts of dead biomass on the marsh floor. Dead biomass during August at the marsh interior stations was greater than 2000 g dry weight·m^{-2} in *P. australis* stands compared to less than 500 g dry weight·m^{-2} in the *Spartina* spp. (Figure 4A). More than 50% of the deposition occurring in the *P. australis* marsh interior stations is organic, implying that *P. australis* can effectively entrain its own dead plant material on the surface of the marsh (Figure 3A). It is not uncommon for *P. australis* marshes to contain substantial amounts of decomposed litter that will incorporate itself as a layer on the marsh bottom (Alizai and McManus, 1980). At the coastal stations the dead material was exported by wave action that breaks culms and removes
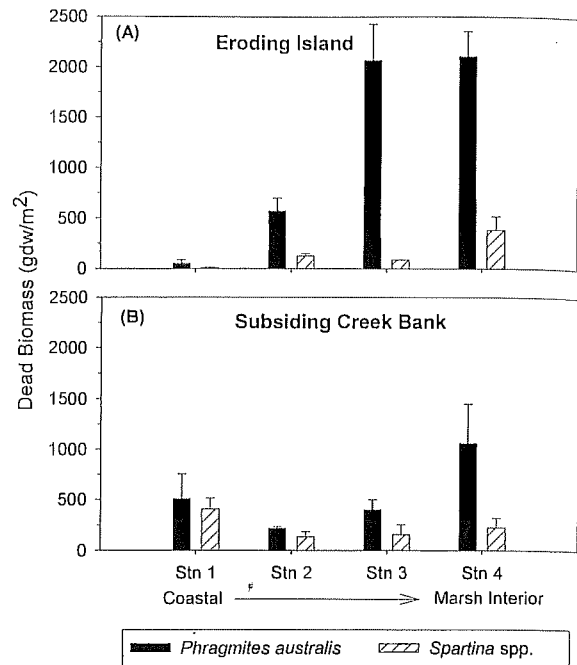
wrack, especially during stormy periods. Therefore, the enhanced deposition in *P. australis* was less evident at the coastal stations, especially at station #2, where wave induced wrack removal reduced total average deposition to just 10 g·m^{-2}·day^{-1}. In contrast, wrack mats from storm events are transported inward from the coast to the marsh interior and subsequently trapped there by the *P. australis* culms (Rooth, pers. obs., Ostendorp, 1989).

Wrack covering the sediment surface can also act as a sediment trap increasing mineral deposition (Alizai and McManus, 1980; Rejmanek et al., 1988). This was observed during a storm event in February 1998 with storm surge effects doubling the water level (from an average of 38 cm to 71 cm) and hydroperiod for nearly a week. Northeast winds exceeded a mean wind speed of 37 km·hr^{-1} over a three day period and caused waves that re-suspended sediment from the bottom of Tangier Sound. The result was very high sediment deposition 17 m into the *P. australis* stand over a 17-day period from February 3–20, 1998. The *P. australis* community collected 307 g·m^{-2}, whereas the *Spartina* spp. community accumulated < 30% of that total amount. The deposition in both communities was primarily mineral,

containing less than 20% organics. We did not convert the storm plates to $g \cdot m^{-2} \cdot day^{-1}$ in order to consider net deposition (deposition-erosion) during the storm period. The pulse of mineral sediment supplied by winter storms has been cited as important in nourishing rapidly deteriorating as well as stable marshes (Reed, 1988; Reed, 1989, Cahoon et al., 1996; Roman et al., 1997). A series of investigators (Baumann, 1984; Rejmanek et al., 1988; Nyman, 1995) have suggested that sedimentation from events like hurricanes can provide partial compensation for subsidence that dominates the marshes of the Mississippi River deltaic plain and elsewhere. Rejmanek et al. (1988) found that a minor hurricane deposited 2.2 cm of sediment in a *P. australis* community and only 0.2 cm in communities dominated by three other vegetative types. They also found a positive correlation between biomass and amount of sediment deposited in each community. Nyman et al. (1995) concluded that greater hurricane sedimentation occurred in *J. roemerianus* than *S. alterniflora* owing this difference to higher stem density in the former. Stem density is lower in *P. australis* than in *Spartina* spp. at the eroding island site, therefore we attribute the higher mineral and organic deposition in *P. australis* to litter accumulation.

A clearer trend of greater accumulation in *P. australis* was observed at the subsiding creek bank in Monie Bay at all stations sampled in August and September 1997 (Figure 2B). At the marsh interior stations the average total deposition was 38 $g \cdot m^{-2} \cdot day^{-1}$ in the *P. australis* whereas in the *Spartina* spp. it was less than half, 16 $g \cdot m^{-2} \cdot day^{-1}$. A large component of what was deposited at those interior marsh sites in the *P. australis* community was organic, and the percentage of organic material deposited at both sites was similar for all stations in *P. australis* (Figure 3A, B). This trend occurred despite the lower quantity of dead biomass harvested at the subsiding creek bank in August, although 1065 g dry weight·m^{-2} litter was present at station #4, the litter totaled only 400 g dry weight·m^{-2} at the other marsh interior station, station #3 (Figure 4B). Decreased biomass in both vegetative communities at the subsiding creek bank is probably due to overall lower production and redox induced by waterlogging (DeLaune et al., 1983; Mendelssohn and McKee, 1988). The presence of a culvert approximately 500 m downstream from the study site restricts water flow on the falling tide and causes this area to be flooded for longer periods. Culverts that are not large enough to allow adequate water exchange (such as maximum flow during the ebb tide) can cause hydro-

logical problems that can translate into deteriorating conditions of the marsh substrate (Bouman, 1998). Intensified flooding at the marsh interior stations made total average deposition similar at all stations within the same vegetative community type (Figure 2B).

The higher rate of sediment deposition at the subsiding creek bank versus the eroding island site is due to a variety of factors. One is the re-suspension of material, primarily very fine organics and particulate sediment, that had previously been deposited inside the marsh. The reduced ebb velocities in conjunction with intensified flooding allow resuspension and easy attenuation of deposited matter back on the sediment surface. Postma (1967) indicates that flood dominated systems, like this subsiding creek bank site, have a slower settling rate of silt material and more is deposited farther in the flood direction than if the sinking took place more rapidly. A longer hydroperiod also contributes to greater daily deposition, and in conjunction with complete inundation past station #4 with every tidal cycle, also explains why deposition exceeded 16 $g \cdot m^{-2} \cdot day^{-1}$ at all stations at the subsiding creek bank site (Figure 2B).

Six-month deposition patterns

Of the 24 marker horizons established at the two sites, 2 were not useable due to erosion removing the feldspar horizon. Both lost plots were located at the eroding island site at the coastal edge of the *P. australis* community. Since erosion cannot be factored in, the marker horizons are biased towards reporting accretion rather than erosional processes (Cahoon et al., 1995). Our vertical accretion values were not significantly different from one another because we lack more than one sampling point in time (Table 2). Sediment accretionary dynamics are highly variable and erratic due to seasonal events and meteorological forcing (Stevenson et al., 1986; Stevenson et al., 1988; Nyman, 1994). Our study period encompasses a particularly stormy period in which both erosional and depositional events were taking place. Because of this, the effects plant communities have on accretion patterns were ambiguous. Slightly greater vertical accretion occurred in *P. australis* (0.64 cm) than in *Spartina* spp. (0.56 cm) at the eroding island. This is the opposite of what we observed at the subsiding creek bank where the six-month accretion rate was 0.56 cm in *Spartina* spp. and 0.40 cm in *P. australis*. Although we demonstrated a large mineral depositional event in the *P. australis* community in February,

Table 2. Change in accretion, elevation, shallow subsidence and below ground accumulation for 2 sites in the Deal Island Wildlife Management Area, MD over a six month period (December 1997–June 1998).

Site	Community type	Vertical accretion (cm)		Change in elevation (cm)		Shallow* subsidence (cm)	Belowground accumulation (cm)
		n	mean ± 1 SE	n	mean ± 1 SE		
Eroding Island	*P. australis*	4	0.64 ± 0.17	62	0.95 ± 0.21	na	0.31
Eroding Island	*Spartina* spp.	6	0.56 ± 0.14	72	0.64 ± 0.15	na	0.08
Subsiding Creek Bank	*P. australis*	6	0.40 ± 0.07	68	0.62 ± 0.13	na	0.22
Subsiding Creek Bank	*Spartina* spp.	6	0.56 ± 0.03	36	0.24 ± 0.24	0.32	na

* Shallow subsidence = (vertical accretion – elevation change).
na indicates this heading is not applicable or this phenomena is not suspected to be occurring.
SE = 1 Standard Error.

it may have been counteracted by strong erosive forces that re-suspended and removed the sediment before we collected cores from the marker horizons in June. The importance of winter storms in promoting vertical accretion over the short term has been demonstrated, but its role in long-term accretion of the marsh is not known (Reed, 1989).

Generally, estimates of vertical accretion are not as accurate as surface elevation due to the complexity of subsurface processes (i.e. subsidence, auto-compaction, shrink/swell from water storage, plant production and decomposition) (Kaye and Barghoorn, 1964). Therefore, it is vital to supplement estimates of vertical accretion with direct measurements of surface elevational change. This allows a clearer understanding of marsh stability, especially in terms of an increasing sea-level rise scenario. Although not statistically significant, the change in elevation is greater for *P. australis* communities at both sites (Table 2). Over six months the elevation of the *P. australis* community at the eroding island increased by 0.95 cm in contrast to 0.64 cm in the *Spartina* spp. The positive elevation change was 2.5 times as great in the *P. australis* as the *Spartina* spp. at the subsiding creek bank. Subsidence was not observed at the eroding island and not in either of the *P. australis* communities at either site. We attribute elevation change that is higher than accretion observed by marker horizons to below ground production. Although, Cahoon and Lynch (1997) attribute greater elevation than vertical accretion to shrink-swell of the highly organic soils during a high flood at their mangrove study locations, this was not the case at either of our sites where measurements were taken during a neap low tide with no water on the marsh surface. The belowground accumulation phenomenon is apparent in both *P. australis* communities and to a small extent in the *Spartina* spp. community at the eroding island. *Phragmites australis* was observed to promote organic soil formation in a Japanese marsh by accumulating dead rhizomes and roots, and by growth of rhizomes (Kudo and Ito, 1988). Haslam (1969) reports that rhizome growth in *P. australis* is highest in the fall. However, studies using short-term in-growth bags indicate that although slowed, production of rhizomes and roots of marsh plants occurs between September and April (Neill, 1992). Therefore, although our study did not capture the effects of maximum below ground growth on elevation change, their still was accumulation evidenced during our measurements.

Implications for coastal areas threatened by rising sea-level

In the past year, approximately 0.5 m of coastline has been lost at the eroding island. This is consistent with Philips' (1986) estimate of a coastline erosion rate of 0.3 m·yr^{-1} for marshes of the Atlantic/Gulf Coastal Plain. The effect of *P. australis* in maintaining or preventing coastline loss in Chesapeake Bay marshes is not known. Storms cause erosion along the coastal edge, and in many cases, provide the interior marsh with higher mineral inputs as a result (Reed, 1988, 1989; Day et al., 1998). It is believed that *P. australis* is a stable component of eroding marsh banks where relatively high tidal energy and coarse sediment tend to prevail (Philips, 1987). In Europe, studies of *P. australis* in an experimental wave tank show a greater reduction of wave loading compared with *Scirpus lacustris* because of increased sediment entrapping (Coops et al., 1996). Other studies show that *P. australis* communities reduce resuspension of fine material (Taked and Kurihara, 1988) and increase retention of sand particles at high stem densities (Knutson, 1988).

Our short-term sedimentation plates suggest that *P. australis* is not increasing rates of deposition more than does *Spartina* spp. where there is active coastal erosion. Instead, it appears that below ground production in the *P. australis* stand may be responsible for the higher elevation change indicated by the SET measurements in the erosive environment. At the subsiding creek bank site, *P. australis'* below ground accumulation appears to be about 30% slower than at the eroding island. It may be the less flushed site has increased sulfide levels that would limit *P. australis* growth (Chambers, 1997). However, *P. australis* unlike the *Spartina* spp. community, is perhaps still able to produce enough peat to keep a positive accretionary balance.

Conclusions

This study demonstrates that at two Chesapeake Bay marshes, where *P. australis* is the primary community type, elevation change can exceed surface deposition rates in the short term. Furthermore, this suggests that below ground accumulation may be the dominant mechanism by which this vegetative type increases substrate level. Direct measurements of deposition over 24 hour periods indicates a greater accumulation of mostly organic material occurs in the interior of *P. australis* stands. In terms of sediment accumulation, both deteriorating marsh types, coastal eroding and subsiding creek bank, appear to benefit from the presence of high densities of litter in *P. australis*. Where there is high wave activity, it is not evident from this study that *P. australis* out performs *Spartina* spp. in terms of trapping material on the marsh surface adjacent to the coast. However, *P. australis* has historically been planted specifically for reduction in wave loading and to protect banks from erosion in Europe (Bonham, 1983). The interior portion of the coastal eroding *P. australis* stand shows a significant ability to accumulate mineral sediment during storm events; perhaps this is important in the long term accretionary budget of this marsh.

Our observations suggest that controlling invasive *P. australis* in Chesapeake Bay marsh areas which are deteriorating (from the effects of relative sea-level rise) may actually enhance erosion. Perhaps removal of *P. australis* should be limited only to areas managed for plant biodiversity or to maintain populations of waterfowl and wildlife. In fact *P. australis* might be valuable in terms of contributing to marsh stability and

could be planted in key areas where erosion is severe. Further studies need to address whether the short term benefits of *P. australis* we observe exist over longer time scales.

Acknowledgements

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