The effect of vegetation height and biomass on the sediment budget of a European saltmarsh

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Abstract

Sediment retention in saltmarshes is often attributed to the presence of vegetation, which enhances accretion by slowing water flow, reduces erosion by attenuating wave energy and increases surface stability through the presence of organic matter. Saltmarsh vegetation morphology varies considerably on a range of spatial and temporal scales, but the effect of different above ground morphologies on sediment retention is not well characterised. Understanding the biophysical interaction between the canopy and sediment trapping *in situ* is important for improving numerical shoreline models. In a novel field flume study, we measured the effect of vegetation height and biomass on sediment trapping using a mass balance approach. Suspended sediment profilers were placed at both openings of a field flume built across-shore on the seaward boundary of an intertidal saltmarsh in the Dengie Peninsula, UK. Sequential removal of plant material from within the flume resulted in incremental loss of vegetation height and biomass. The difference between the concentration of suspended sediment measured at each profiler was used to determine the sediment budget within the flume. Deposition of material on the plant/soil surfaces within the flume occurred during flood tides, while ebb flow resulted in erosion (to a lesser degree) from the flume area, with a positive sediment budget of on average 6.5 g m^{-2} $tide^{-1}$ with no significant relationship between sediment trapping efficiency and canopy morphology. Deposition (and erosion) rates were positively correlated to maximum inundation depth. Our results suggest that during periods of calm conditions, changes to canopy morphology do not result in significant changes in sediment budgets in marshes.

Introduction

The balance between sea level rise and rates of sediment accretion is a key research question in the broader debate as to whether or not marsh surfaces will be able to keep up with near-future accelerated sea level rise (Orson et al., 1985; Kirwan et al., 2010). Sea level rise poses a threat to intertidal saltmarshes due to seawater inundation beyond the physiological tolerance of the vegetation. However, the ability of marshes to accrete vertically through sediment trapping and root growth allows them to maintain their position in the tidal frame as it is translated upwards, promoting their long-term stability and survival (Morris et al., 2002; French, 2006; McIvor et al., 2013). It has been argued that the presence of vegetation enhances sedimentation on saltmarsh platforms both by attenuating wave energy and slowing water flow (Boorman et al., 1998; Temmerman et al., 2005) and by preventing the resuspension of deposited sediments on, and the direct erosion of, saltmarsh surfaces (Fagherazzi et al., 2012).

The presence or absence of vegetation, as well as vegetation parameters such as height and biomass are thought to be key factors in determining rates and patterns of sediment trapping and deposition, although this relationship is non-linear (Nardin and Edmonds, 2014) and may be dependent on wave and flow conditions. The importance of vegetation structure in marsh functioning is well recognised but its incorporation in realistic representations of the interactions between vegetation and sedimentation is complicated by the immense variability in canopy structure on a range of scales. Marsh vegetation shows great inter-specific variability in stem flexibility (Tempest et al., 2015; Rupprecht et al., 2017) affecting plant-flow interactions and sedimentation. Furthermore, marsh vegetation is regularly subjected to both emergent and submerged states and, in the case of the latter, to both 'normal' and extreme 'storm surge' flow regimes. Vegetation height and biomass varies both spatially and temporally on European saltmarshes. Such canopy characteristics vary with intertidal elevation (Silvestri et al., 2005) and with the seasons. Communities are typically composed of a combination of perennial and annual species with little above ground presence during winter (Watkinson and Davy, 1985) when annual species are absent and perennial saltmarsh species biomass is also much reduced (Hussey and Long, 1982; De Leeuw et al., 1990). Biomass reaches a peak at the end of the northern summer growing season (De Leeuw et al., 1990). In the longer term, saltmarsh canopy height and biomass vary as a function of climate change (Arp et al., 1993; Reef et al., 2016) and eutrophication (Deegan et al., 2007).

While slower flow rates in vegetated areas enhance particle settlements and thus deposition (Neumeier and Amos, 2006), the movements of plants when acted on by waves and currents can scour the surface and significantly enhance erosion, particularly in the pioneer zone and along marsh seaward margins (Temmerman et al., 2007; Feagin et al., 2009). Sheehan and Ellison (2015) observed significantly lower accretion and higher erosion rates immediately following the complete removal of a saltmarsh vegetation cover, although the addition of organic matter to the soil substrate over time contributes to erosion-resistant soils (Feagin et al., 2009). Periods of increased erosion in UK saltmarshes coincide with periods of higher winds and wave heights (van der Wal and Pye, 2004; Wolters et al., 2005); this may also cause increased sedimentation on the saltmarsh platform (Schuerch et al., 2012). There are, however, relatively few studies worldwide on the efficiency with which saltmarshes trap tidally advected material in field conditions (French, 2006; Moskalski and Sommerfield, 2012; Spencer et al., 2015b; van der Deijl et al., 2017). In this study, we aim to close this knowledge gap of the role of vegetation structure on deposition *in situ* through the use of a field flume, in combination with a mass balance approach to determine how changes to canopy morphology affect trapping efficiency in a UK saltmarsh.

Methods

Setting and physical environment

The field study was undertaken on the UK east coast at Tillingham, Dengie Peninsula, (51.69425°N 0.94206°E, Fig. 1A) between the estuaries of the Rivers Blackwater and Crouch. The saltmarsh is a near-horizontal platform of clayey silts, ca. 200 m in width, at an elevation of 1.9 - 2.5 m above Ordnance Datum Newlyn (ODN; where 0.0 ODN approximates to mean sea level). The tidal mudflat immediately seaward of the marshes are at elevations of 0.9 - 1.9 m ODN and show a 'mudmound topography' of shore-normal sinuous ridges and runnels in the transition zone between the saltmarsh and the flat tidal mudflat. The runnels narrow shorewards into small creeks which dissect the marsh surface (Möller and Spencer, 2002). The Dengie Peninsula coast is macrotidal, with a mean spring tidal range of 4.8 m (Reed, 1988). The southern North Sea is, however, particularly susceptible to storm surges which raise water levels well above expected tidal levels. Thus the storm surges of 1953 and 2013 reached 4.4 m ODN at West Mersea, Blackwater estuary (Spencer et al., 2015a), resulting in water depths of ca. 2 m over marsh surfaces. The wave climate is moderate with a maximum recorded significant wave height (H_s) of 0.65 m at Sales Point on the northern limit of the Peninsula (Herman, 1999).

The main sediment sources for the marshes in East Anglia are thought to be the erosion of coastal cliffs in Norfolk and Suffolk, and to a much lesser extent, fluvial inputs from East Anglian rivers, and offshore seabed erosion (McCave, 1987). The cliff sediments are unconsolidated Quaternary sediments with a high proportion of inorganic mud. Suspended sediment concentrations in the southern North Sea are highly seasonal, with more than a fourfold increase in sediment concentrations in winter compared to summer (Prandle et al., 1997). Our experiment was undertaken during summer, when sediment supply to the marsh is lowest, but vegetation biomass is highest.

Experimental design

The experiment was carried out during the spring tide period between the 2-9 July 2016. We selected an area towards the seaward fringe of the saltmarsh at Tillingham and enclosed it within a plywood flume channel secured to the marsh surface with wooden stakes. The flume was 1000 mm in width, 1820 mm in length and the vertical walls were 900 mm high, with symmetrical funnel shaped openings at each end (Fig. 1B). It was placed across-shore (E-W), corresponding to the main direction of tidal flow. One profiling turbidity sensor (Argus Surface Meter (ASM) IV) and one pressure sensor (Solnist Levellogger Edge, Model 3001) were placed in the centre of both openings. The ASM IV turbidity profilers are titanium rods consisting of a series of 144 optical backscatter (OBS) sensors arranged as an array, with a 10 mm vertical spacing. The profilers and pressure sensors were programmed to record a depth-averaged turbidity profile (10 measurements over 10 s) and a water depth every 30 seconds

Hydrodynamic measurements

The hydrodynamic conditions during the experiment were primarily deduced from the water levels recorded by the pressure sensors. The continuous (or still) water levels and the maximum tidal inundation depth were derived from a smoothed water level curve, which was calculated using a moving-average filter with a window size of 15 minutes (=30 data records). A proxy for wave heights was calculated for every tidal inundation as the standard deviation of the differences between the recorded and the still water level (smoothed water level curve) and validated with actual wave measurements, using a PTX1830 pressure transmitter (Möller, 2006) during a previous measurement campaign. Validation of the wave proxy resulted in a highly significant correlation between the wave proxy calculated from the smoothed water curve and the measured wave heights measured using the PTX1830 of the form (*wave height (cm)* =6.465 x *wave proxy*, R^2=0.92, $p<0.001$). Wind conditions were obtained from two UK Met Office coastal meteorological stations in Essex, one at Walton-on-Naze and one at Shoeburyness (Southend-on-Sea). Average daily wind speeds (measured at Walton-on-Naze, Essex and at Shoeburryness, Essex) ranged between 10 and 21 km h^{-1} with a predominant W/SW direction (UK Met Office, www.metoffice.gov.uk).

Sediment budget measurements

In order to calibrate the ASM sensor turbidity readings to g m^{-3}, water samples (1 L) were collected at 4 cm above the marsh surface using an automated water sampler (ISCO 6712, Teledyne Isco, Lincoln NE, USA) in the pioneer zone of the saltmarsh during two spring tide periods in April (7-11 April 2016) and July (21-24 July 2016). In each calibration period, three samples were taken, 30 minutes apart, during each inundation over eight consecutive inundations (N=24). Following collection, the samples were filtered through pre-weighed GF/C filters, which were then dried at 105°C for 24 hours and re-weighed. Measured sediment concentrations in the water samples were compared to the simultaneously measured turbidity levels recorded by both ASM IV at 4 cm above the marsh surface. A calibration curve was derived for each of the devices, relating turbidity to suspended sediment concentrations (SSC) (R^2=0.91, Fig. S1).

A sediment budget for each tidal stage (flood and ebb phase) was calculated by numerical integration of the depth-averaged SSC difference between the upstream and downstream ASM sensors (g m^{-3}) over the instantaneous volume within the flume (m^3), using the trapezoidal rule. From the two ASM sensors at each end of the flume, the upstream and downstream sensors were assigned based on the tidal stage (i.e. the seaward sensor was assigned as the upstream sensor during the flood tide and as the downstream sensor during the ebb tide). Due to negligible flow velocities during slack water, we excluded the data from the slack tide period from our analysis. Slack tide was defined as the period where the rate of change of the smoothed water depth curve was $<3.3\times10^{-3}$ cm s^{-1}. Furthermore, data from water depths of <3 cm within the flume were excluded because of increasing relative measurement errors for small water depths.

A positive sediment budget denoted sediment deposition within the flume, whereas a negative value indicated erosion. The sediment budget was defined as the amount of sediment that was actually retained within the flume over the tidal cycle. Assuming an insignificant change in the amount of material held on plant surfaces, this should closely match the deposition data recorded by the GF/C filters deployed on the bed within the flume (see below).

Vegetation canopy height and biomass manipulation

The vegetation within the flume consisted of a typical '*Spartina alterniflora* saltmarsh community' (British NVC, SM5) dominated by *Spartina alterniflora* (50 % cover), *Aster tripolium* (17 % cover) and *Puccinellia maritima* (10 % cover) with some individuals of annual *Salicornia* spp. (5 %), *Suaeda maritima* (3 %) and *Atriplex portulacoides* (1 %). Vegetation ground cover in the flume was 86 %. Initial vegetation height within the flume was on average (± SD) 10.7 cm (± 2.2) with a maximum height of 26.7 cm (± 3.4) (Fig. 2). The 75th percentile was on average 15 cm (± 2.7). Average stem density was 910 stems m^{-2} (± 240) within the flume.

In order to quantify the effect of plant canopy morphology on the saltmarsh sediment budget, we reduced the height of the vegetation in the flume by ca. 5 cm every other inundation (Fig. 2A). The cut material was collected, dried (48 hours at 105°C) and weighed for biomass determination (Fig. 2B). Four cuts were carried out, reducing the mean canopy height from 10.7 cm (range 0-27 cm) to a final mean height of 1.3 cm (range 0-5.4 cm). Canopy height was measured across the entire flume area, using eight, scale calibrated, side-on photographs (Rupprecht et al., 2015) and analysed using line graph analysis (Image J, Schneider et al., 2012). The canopy (green) was easily distinguished digitally from the flume wall background (light tan) using a grey scale threshold. A digitised line plot was then created from the threshold edge so that the y-axis was height above bottom in cm and the x axis was cm from flume opening along the ground. Vegetation stem density was measured by manually counting stems in five 20 cm x 20 cm quadrats within the flume.

Measuring sediment deposition

Five glass petri dishes, inserted level with the soil surface, were evenly distributed within the flume (Fig. 1B). Pre-weighed glass fibre (GF/C) papers (9 cm diameter) were placed on the surface of each petri dish and held in position with small metal pins. The filters were exchanged every other tidal inundation and the dry weight of sediment deposited on the filters was measured after drying at 105°C for 24 hours.

The sediment characteristics (grain size and organic carbon content) were determined using hardened ashless filter papers (Whatman, Grade 540) distributed on the marsh, as above, during the 4-day July calibration period. The filters were then dried at 105°C for 24 hours and weighed, before being combusted at 505°C for 6 hours to burn away the filter paper and to determine sediment loss-on-ignition. The remaining sediment was weighed to determine the loss-on-ignition, representing the organic carbon content and analysed for grain size, using a Malvern Mastersizer 2000.

Statistical analysis

The relationship between vegetation height or biomass and deposition was measured using multiple linear regression with vegetation height (or biomass) and the hydrodynamic variable, maximum water level, as the predictors and deposition as the dependent variable. The two co-variates, maximum water level and vegetation height were not linearly correlated ($p=0.65$), satisfying the assumption of independence. To measure the effect of waves on deposition, we used a multiple regression model with wave proxy and vegetation height as predictors of deposition. Due to the collinearity of wave proxy and maximum water depth, we residualised wave proxy by running a preliminary regression analysis between wave proxy and maximum inundation depth and used the residuals from this analysis (wave proxy_{resid}) in lieu of wave proxy in the multiple regression model. For all linear models, the normality assumption was tested by the visual inspection of the histograms of the model residuals.

Results

Hydrodynamic and sediment characteristics

During the duration of the experiment (2-9 July 2016), we measured 14 tidal inundations over a set of rising and then falling spring tides, peaking on 7 July 2016 with a predicted high water level of 2.01 m (ODN) at Harwich, Essex, 40 km NE of the study site (Table 1). Mean High Water Springs (MHWS) at Harwich is 1.99 m ODN and Highest Astronomical Tide (HAT) is 2.44 m ODN. Due to flume damage on 6 July 2016, we had to remove tides 9 and 10 (6 and 7 July 2016) from the dataset.

Maximum inundation depths in the flume for each tidal inundation during the period of the experiment ranged from 0.14 m to 0.54 m (Table 1). The highest inundation depths were measured during tides 7 and 8 with 0.52 and 0.54 m respectively. With a maximum profiling range of 1.44 m, the observed inundations allowed for the capture of a complete turbidity profile during every tide.

The wave activity during the experiment, as represented by the wave proxy, was closely, positively correlated with the maximum inundation depth ($r = 0.86$), apart for tide 8. The highest wave proxy value occurred during tide 7 with the lowest values being observed during the first two tides (Table 1).

Depth-integrated tidal horizontal velocities within the flume were approximated by the rate of change in water depth between measurements (rising velocity). On average the peak rising velocity during flood tides was 0.64 cm s^{-1}, whereas the average peak rising velocity during ebb tides was -0.57 cm s^{-1} (Table 1). The absolute highest rising velocity recorded was 0.90 cm s^{-1} during the flood of tide 8. During most of the observed tides the maximum rising velocity during flood tides was higher than during ebb tides (flood : ebb ratio >1) indicating a flood dominant inundation regime (Table 1). The most pronounced flood dominance was observed during tide 6, whereas tides 2 and 13 were the only tides showing ebb dominance (flood : ebb ratio < 1).

The sediment that settled on the filters in the flume during the calibration period was primarily composed of very fine sand (D_{50} = 121.9 µm, skewness = -0.19) and contained, on average (±SD), 15.4 % ±4.9 % organic matter.

Table 1: Hydrodynamic characteristics of the different flume inundation periods during the experiment.

Tide	Vegetation Height (m)	Vegetation Cut	Maximum Inundation Depth (m)	Max Flood Velocity* (cm s^{-1})	Max Ebb Velocity* (cm s^{-1})	Flood/Ebb Ratio	Wave proxy
1	0.107	Start	0.143	0.338	0.322	1.049	0.149
2	0.107	Start	0.231	0.470	0.473	0.993	0.232
3	0.107	Start	0.346	0.614	0.587	1.046	0.355
4	0.107	Start	0.366	0.677	0.579	1.169	0.369
5	0.107	Start	0.381	0.684	0.563	1.215	0.386
6	0.073	Cut 1	0.390	0.743	0.608	1.222	0.396
7	0.073	Cut 1	0.524	0.848	0.742	1.143	0.529
8	0.059	Cut 2	0.539	0.916	0.753	1.215	0.342
11	0.039	Cut 3	0.294	0.641	0.542	1.182	0.300
12	0.013	Cut 4	0.402	0.698	0.601	1.161	0.406
13	0.013	Cut 4	0.251	0.461	0.491	0.940	0.257
14	0.013	Cut 4	0.34	0.63	0.58	1.10	0.351

* Tidal horizontal velocities within the flume were approximated by the rate of change in water depth between measurements (rising velocity).

Sediment budget

The sediment budgets (deposition minus erosion of sediment) of the flood and the ebb periods for the flume area were both significantly affected by the maximum water level (multiple linear regression, β = 1.94, $p < 0.001$, R^2 = 0.74 and β = -1.57, $p < 0.001$, R^2 = 0.72 for flood and ebb periods respectively, Fig. 3), but not by the height of the vegetation. Vegetation height was not a significant factor for sediment budgets in either flood or ebb periods (β = 1.2, p = 0.22, R^2 = 0.74 and β = -1.35, p =0.12, R^2 = 0.72 respectively, Fig. 4A). Vegetation biomass, which showed a non-linear reduction in relation to vegetation height (Fig. 2B) was also found to have no significant effect on the sediment budget during flood and ebb flows (multiple linear regression, β = 0.007, p = 0.21, R^2 = 0.74 and β = -0.007, p = 0.14, R^2 = 0.72, respectively, Fig. 4B). The net sediment budget following a full inundation cycle was not affected by maximum inundation depth, vegetation height or vegetation biomass (β = 0.36, p = 0.07; β = -0.09, p = 0.85, R^2 = 0.19; β = -0.00007, p = 0.97, R^2 = 0.19; respectively).

The sediment budget within the flume following each stage of the tide (Fig. 4) indicated sediment import into the flume area during the flood period (42.7 ± 13.53 g m^{-2}), and export from the flume area during the ebb period (-36.6 ± 11.15 g m^{-2}). Import during the flood was consistently greater than export during the ebb periods throughout the measurement period. Hence, the net sediment budget per tide was on average positive (6.1 ± 3.98 g m^{-2}).

A multiple linear regression with the residualised wave proxy and vegetation height as predictors, found that the wave proxy was not a significant forcing on the sediment budgets during flooding (Fig. 5A, multiple linear regression, $\beta = 88.4$, $p = 0.60$, $R^2=0.03$) nor during ebb flows (Fig. 5B, $\beta = -126.3$, $p = 0.35$, $R^2 = 0.11$).

The amount of sediment deposited on the filter traps within the flume was not significantly affected by canopy height ($R^2 = 0.01$, $p = 0.79$). There was no correlation between the mass of the sediment deposited on the filter traps and the mass balance calculation of net sediment budgets within the flume (Pearson's $r = 0.26$), but a high correlation was observed between sediment trapping on the filter and sediment deposition within the flume during flood periods calculated using the mass balance approach (Pearson's $r = 0.72$).

Sediment concentrations were significantly higher closer to the marsh surface than higher in the water column (Fig. 6). The mean SSC in the bottom 5 cm was more than 10 times higher than that measured for the rest of the water column (t-test, $p<0.001$). This pattern was more pronounced for the seaward sensor (Fig. 6A) than for the landward sensor (Fig. 6B). For both sensors, only tide 2 showed the presence of suspended sediment within the middle and upper parts of the water column.

Discussion

Our field study indicates that under calm summer conditions, tidal flooding delivers very fine sand to the lower marsh at Tillingham, Essex, which is deposited at a rate that is not demonstrably affected by canopy height and biomass. The sediment budget of the marsh under the dense (910 stems m^{-2}) *Spartina anglica* dominated plant canopy within the flume was not significantly altered despite a reduction in plant canopy height from an average of 10.7 cm to 1.3 cm. Our findings indicate that morphological changes to the saltmarsh canopy, such as those predicted with nutrient enrichment (Fox et al., 2012), grazing (Elschot et al., 2013; Nolte et al., 2013) or climate change (Reef et al., 2016) might not always have a significant impact on sediment deposition/erosion *in situ*.

Previous studies have compellingly shown that saltmarshes effectively attenuate tidal flows (Christiansen et al., 2000; Neumeier and Ciavola, 2004) and waves (Möller et al., 1999) as a function of vegetation density and height. However, potential linkages between vegetation canopy characteristics and small scale turbulence around plant elements (Widdows et al., 2008), mean that there is not necessarily a direct link between hydrodynamic conditions measured at the larger (metre) scale, sediment trapping, and sedimentation on the marsh surface. Attempts to link either vegetation parameters, or hydrodynamic measures alone, to sedimentation, are unlikely to succeed where the latter is explainable only through the interaction of the former two. Thus, Boorman et al. (1998) found no correlation between vegetation height and sediment accretion at one Essex saltmarsh, but did find a correlation at another marsh, suggesting that the relationship between vegetation structure and sedimentation can be site dependent. Despite flow attenuation, Widdows et al. (2008) even found enhanced erosion and lower sediment accretion rates in the lower sparsely vegetated saltmarsh due to enhancement of near bed turbulence relative to bare mud patches as the flow enters *Spartina anglica* canopies. In a North American saltmarsh, Moskalski and Sommerfield (2012) show that deposition and sediment trapping efficiency are not related to plant stem density but rather to the distance from the creek and suspended sediment properties. Studies on grazing by small and large herbivores on saltmarshes found a significant impact of grazing on vegetation height, but no subsequent effect on sediment deposition (Elschot et al., 2013). Similarly, our study indicates that contrary to theoretical predictions, during calm conditions the role of canopy morphology in areas where vegetation is present, is marginal for sediment accretion *in situ*. Our field based sampling design was such that it might not have been possible to detect a small effect of vegetation structure on deposition due to the dominance of the effects of hydrodynamic forcings and other, unmeasured interacting factors influencing sediment settling and erosion in this environment, such as microtopography (Stribling et al., 2007) and/or bed shear strength characteristics (Howes et al., 2010). Theoretical and lab based flume experiments provide optimal conditions for the detection of vegetation effects, even if they are small, by minimising the naturally occurring variability in other hydrodynamic and geomorphological factors, leading to a possible overestimation of the role of vegetation structure on sedimentation *in situ* under some conditions.

During the duration of our experiment, the missing impact of the vegetation-mediated sedimentation could have been caused by the typically calm summer weather conditions and the wave attenuating impacts of the flume. Significant wave heights in the flume averaged 0.02 m (derived from the wave proxy, Table 1), which is ten times lower than the mean wave heights measured at the saltmarsh pioneer zone at this location over a ten-month period (Möller and Spencer, 2002). The long term climate record for East Anglia wind speeds shows the lowest wind speeds occur in July, and that the wind speeds measured during the experimental period (daily average of 10 to 21 km h^{-1}) are within the range of the long term average for this month. The calm conditions generated low vertical mixing of the water column and led to low vertical mixing of suspended sediments. This resulted in suspended sediment travelling primarily within the lower 5 cm of the water column. The removal of sediment from the flume during ebb flow indicates that there is a highly mobile sediment fraction which is resuspended after initial sedimentation (during flood flow). Our sediment profiler data indicates that this sediment fraction travelled very close to the marsh surface (below 5 cm above the marsh surface, Figure 6). Only the final vegetation cover cutting reduced the mean vegetation height to within this height. Therefore, the imposed changes in vegetation morphology are not likely to have influenced this peak suspended sediment fraction.

Estimated flow speeds in the flume were very low, with maximum rising and falling velocities of less than 1 cm s^{-1}, characteristic of saltmarshes (e.g. Christiansen et al., 2000). Despite removing most of the plant canopy within the flume, surface roughness following the final vegetation cut was still noticeably greater than that of an unvegetated mudflat. Thus it is likely that the turbulent energy and flow structure close to the bed, remained similar following the vegetation cuts. Previous studies have suggested that sediment deposition is more strongly linked with marsh topography rather than with vegetation structure (Coulombier et al., 2012), and it could be that locally more sheltered areas associated with variations in topography caused by belowground vegetation structures trigger sedimentation of the suspended sediment. Our study also supports previous findings (van Eerdt, 1985) that belowground biomass (which remained mostly intact following the aboveground biomass removal) plays an important role in increasing bed-shear strength and preventing erosion (as the cutting of the vegetation canopy in our experiment did not significantly affect sediment loss from the flume during the ebb part of the tidal cycle). In this context, a large scale flume study showed that significant wave dissipation still occurs even when the aboveground biomass of a saltmarsh platform is mowed (Möller et al., 2014).

The net sediment gain per tide was on average only 13 % of the sediment flux entering the flume (Fig. 4). Thus the trapping efficiency of the tidally advected material was significantly lower than the hypothetical maximum and lower than simulations of sediment trapping efficiency in East Anglian marshes, using the numerical mass-balance model MARSH-0D, which predict trapping efficiencies of ca. 50% (French, 2006). Our calculated positive sediment budget of 12 g m^{-2} day^{-1} roughly compares to a vertical accretion rate of 2.7 mm y^{-1} (bulk density of ρ = 1.6 g cm^{-3} was measured for this site, R Reef unpublished data). Long term accretion rates measured using surface elevation tables and marker horizons adjacent to the flume show a net accretion of 7.3 mm y^{-1} (T Spencer, unpublished data). The accretion rates we measured during calm summer days are lower than the long-term average, supporting previous findings that accretion can be seasonal (Spencer et al., 2012), with higher rates during the more energetic winter season, a period when sediment supply is also higher (Prandle et al., 1997). High rates of accretion can also occur during infrequent high energy events (Stumpf, 1983; Schuerch et al., 2013). Our comparison of sediment budgets calculated using filter trap data with those calculated using the sediment mass balance approach suggest that the widely-used filter trap method could overestimate accretion, due to a lower erosion rate from the filter than from the surrounding sediments and/or the sediment removal occurred from surfaces other than those represented by the filter paper.

In conclusion, our findings suggest that during calm summer conditions the rate of sediment trapping by saltmarshes is independent of vegetation height or biomass. This is most likely due to very weak vertical mixing of sediment in the water column during calm sea conditions and the high surface roughness and bed-shear stress due to below ground plant structures. Our conclusion provides support for the simplification of vegetation canopies in numerical models of surface accretion under some hydrodynamic conditions. However, during periods of higher wind/wave energy and stronger vertical mixing of suspended sediments, the role of canopy structure could prove significant.

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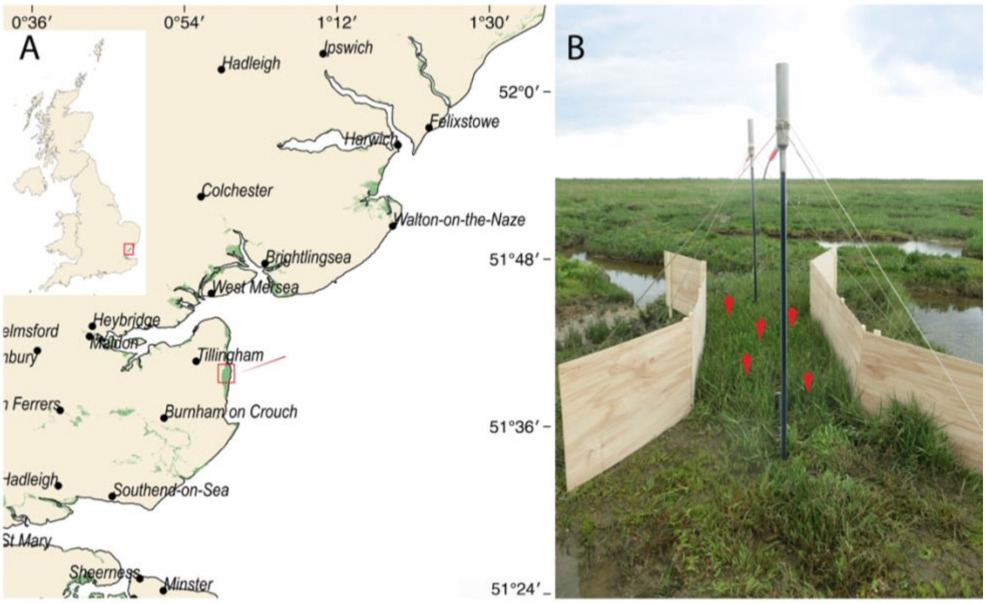

Figure 1: (a) The location of the study site near Tillingham, Essex, UK (51.69425°N 0.94206°E). Green shaded areas are saltmarshes. (b) a west-facing photo of the field flume, with the two ASM turbidity profilers and pressure sensors at the flume openings (the distance between the two ASM turbidity profilers was 1.82 m). Red markers point to the positions of the filters.

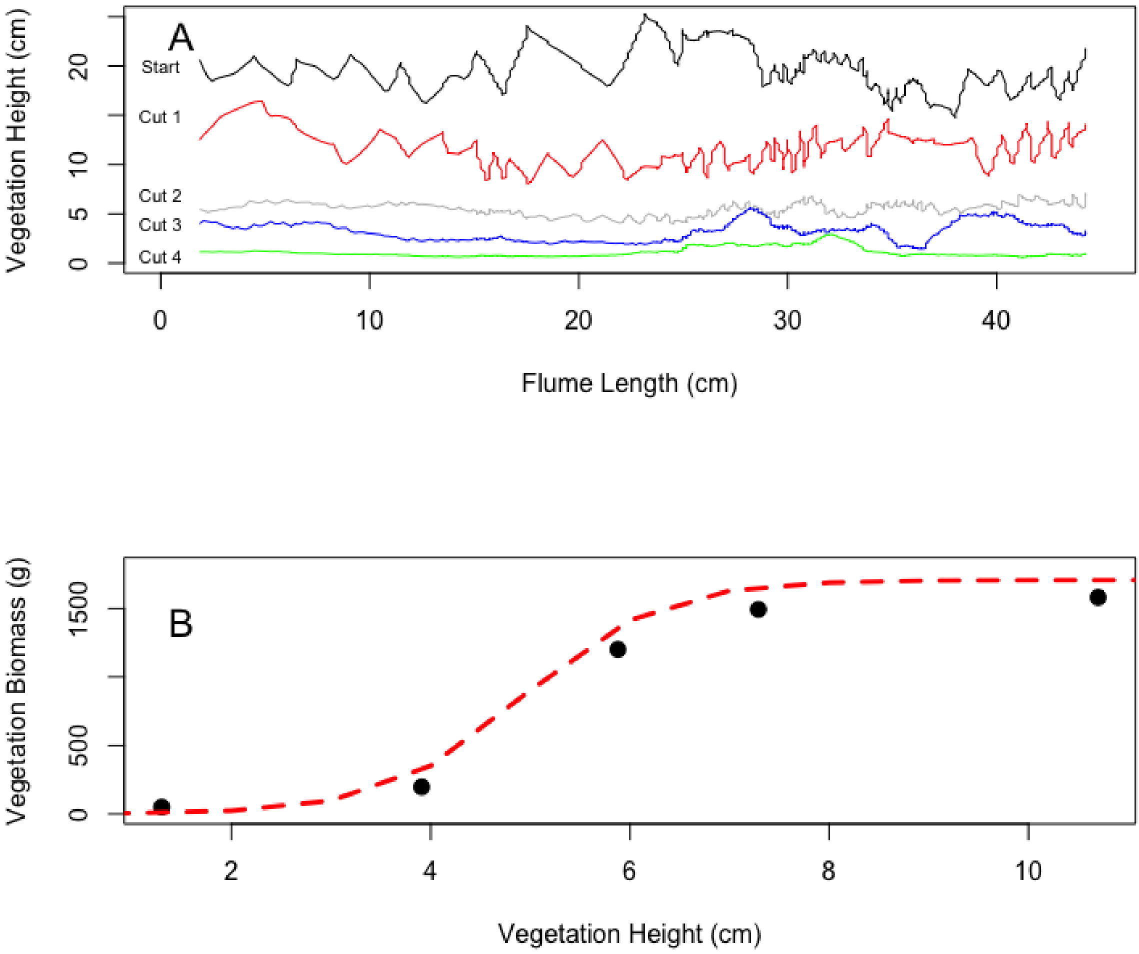

Figure 2: (a) A representative smoothed vegetation height profile along a section of the flume length for the initial conditions (Start) and subsequent vegetation cuts. The saltmarsh community within the flume was dominated by *Spartina alterniflora*, with an average stem density of 910 stems m^{-2} (b) the relationship between vegetation biomass and height is depicted by the logistic population growth function:

$$Biomass = \frac{2150 \times e^{1.46 \times Height}}{1707 + 1.26(e^{1.46 \times Height} - 1)}$$

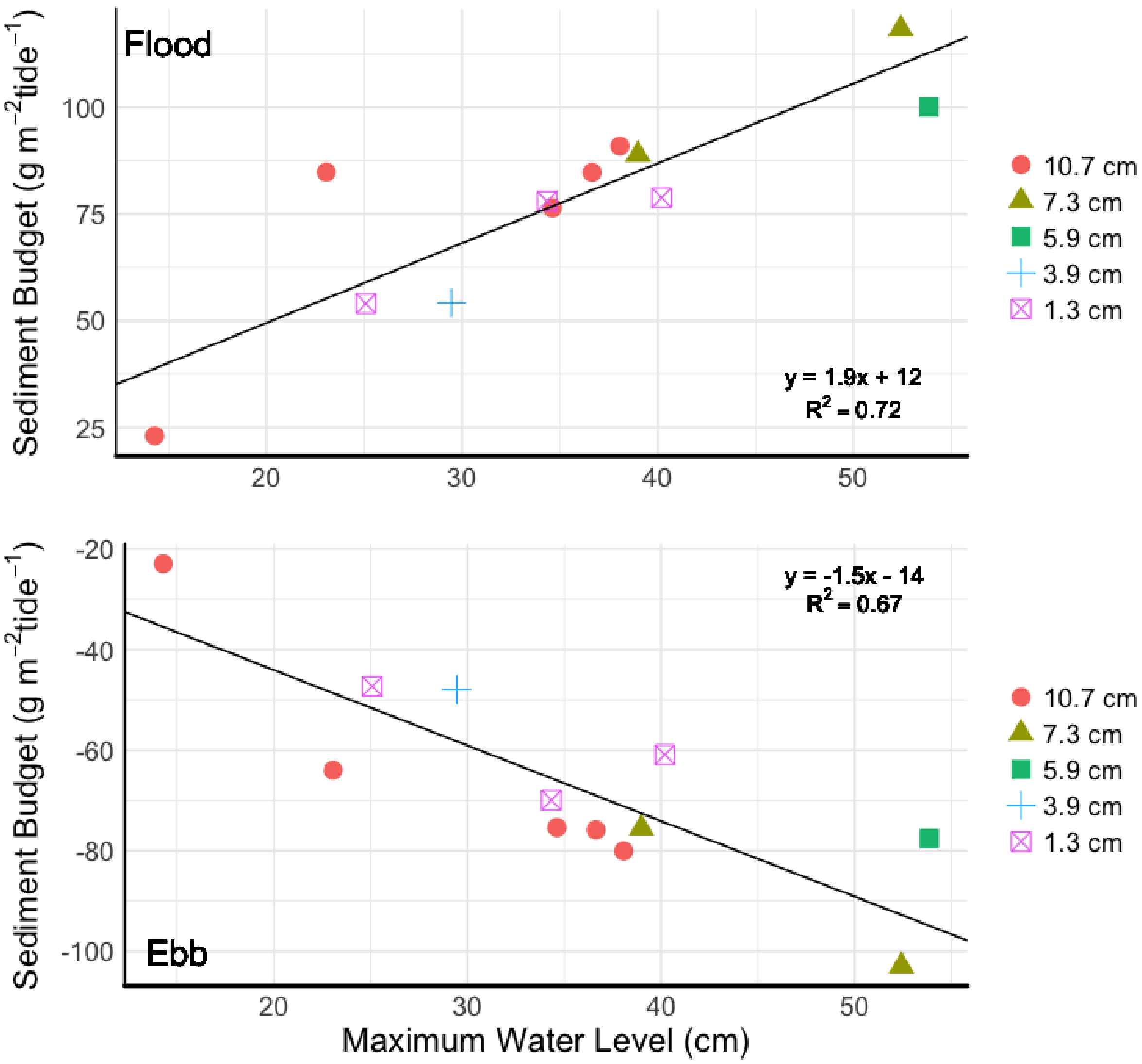

Figure 3: The sediment budget within the flume, calculated from the suspended sediment discharge between the upstream and downstream ASM sensors and the volume of water in the flume during the tidal period, as a function of maximum inundation depth (maximum water level during the tide) during the (a) flood and (b) ebb periods. Symbol shapes/colours correspond to different vegetation heights. A linear regression model was fitted to the data and the equation and fit are presented in the panels.

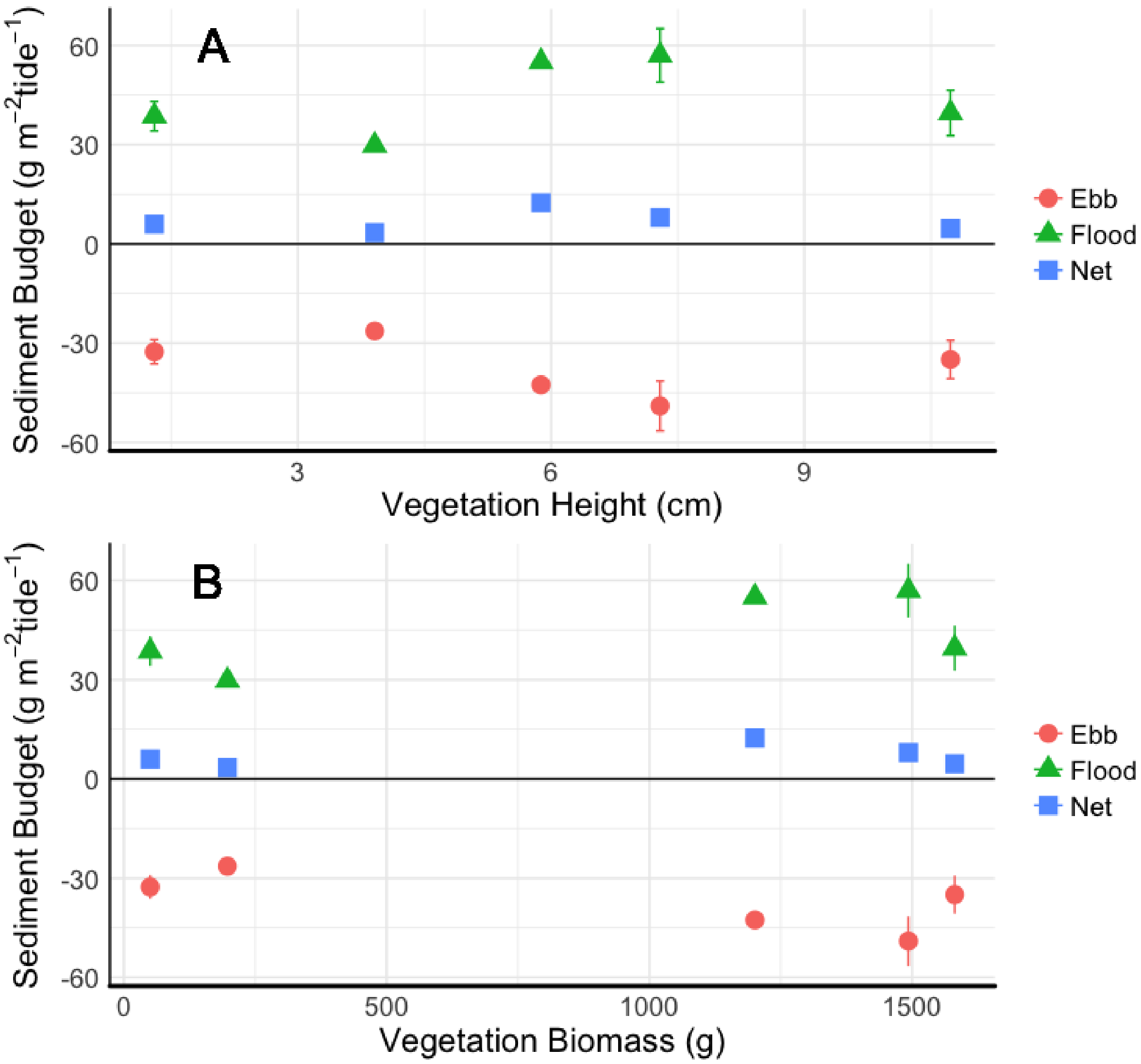

Figure 4: The mean (±SE) sediment budget within the flume, calculated from the difference in the depth-integrated suspended sediment concentration between the upstream and downstream ASM sensors (sediment discharge) and the volume of the water in the flume during the tidal period, as a function of (a) mean vegetation height and (b) vegetation biomass during the flood (green triangles) and ebb (red circles) periods. Blue squares are the mean sediment balance (±SE), with positive values indicating net sediment gain within the flume area and negative values, indicating net sediment loss from the flume area.

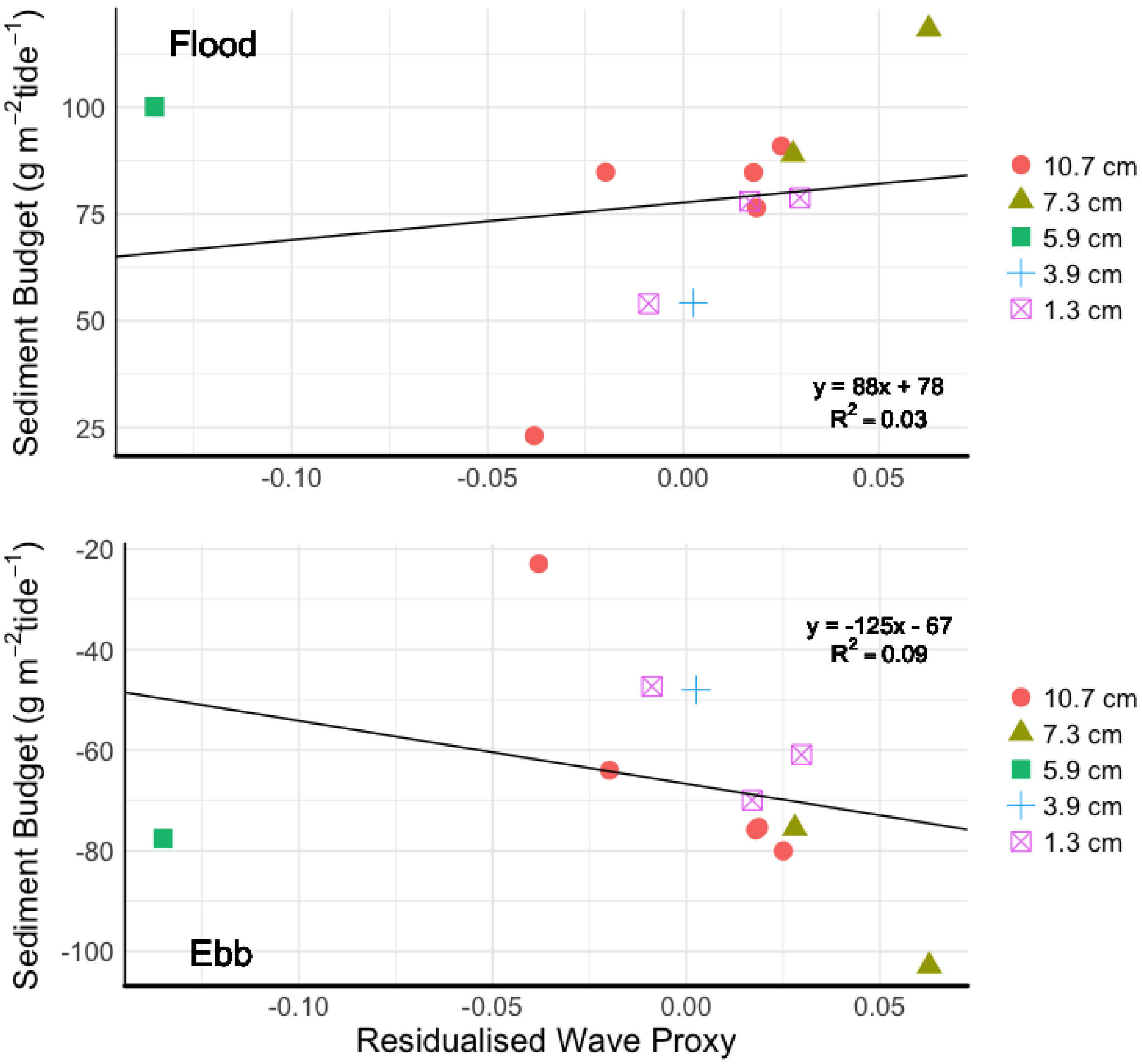

Figure 5: The effect of the residualised predictor wave proxy (corrected for the influence of maximum water level) on the sediment budget within the flume, calculated from the difference in suspended sediment concentration between the upstream and downstream ASM sensors (sediment discharge) and the volume of water in the flume during the tidal period, as a function of the wave proxy during the (a) flood and (b) ebb periods. Symbol shapes/colours correspond to different vegetation heights. A linear regression model was fitted to the data and the equation and fit are presented in the panels.

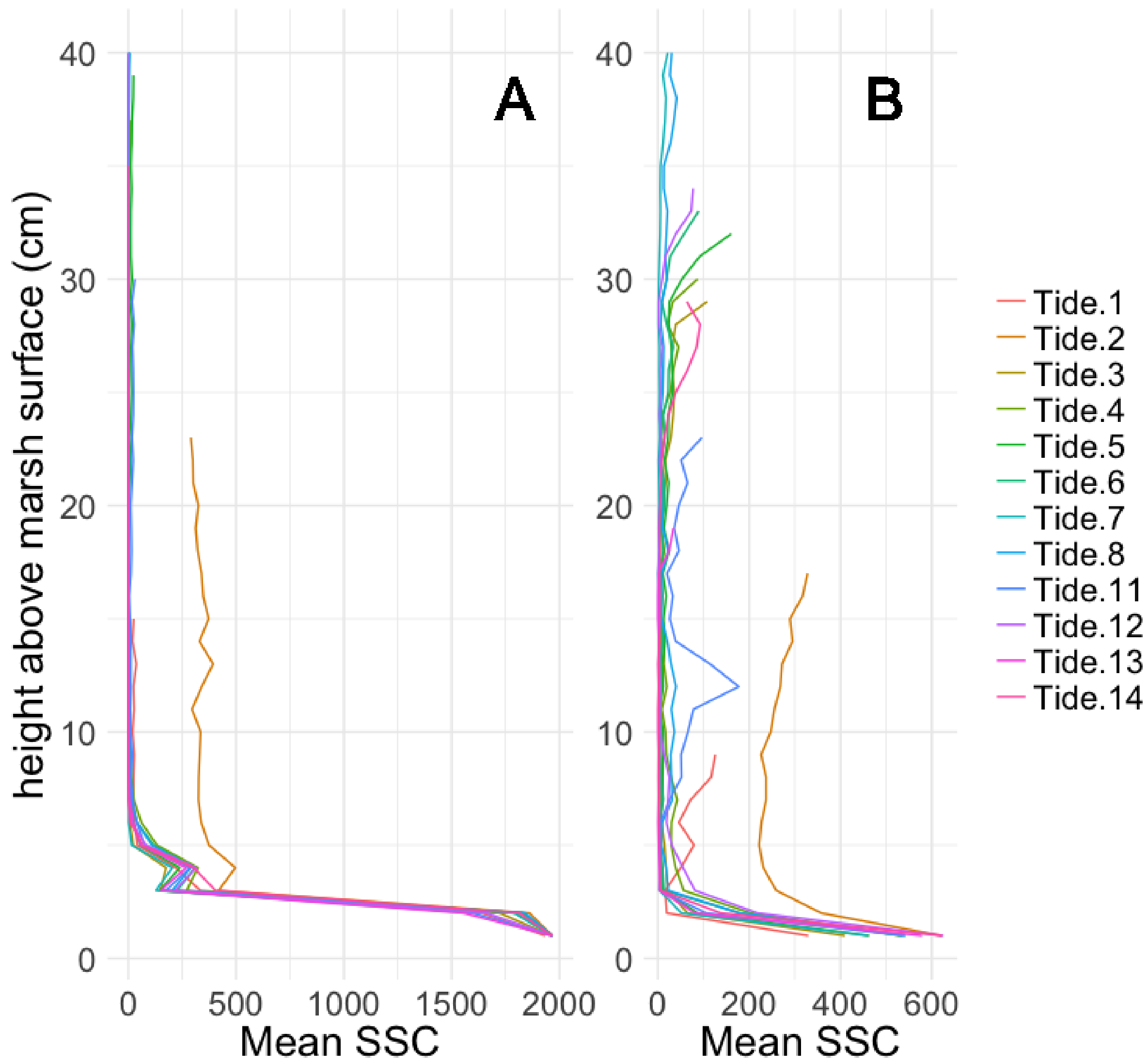

Figure 6: Suspended sediment concentrations (g m^{-3}) measured at different heights above the marsh surface at 1 cm intervals at the (a) seaward and (b) landward openings of the flume. Suspended sediment concentrations were much higher closer to the marsh surface (below 5 cm). Each line represents the mean for an entire inundation period for each sensor. The hydrodynamic conditions of each tide are described in Table 1.

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Figure S1: Calibration curves for the two ASM turbidity profilers used in the flume study. The 24 points of calibration were collected over a four-day period, immediately proceeding the flume experiment. Turbidity, measured by the ASM sensors, 4 cm above the marsh surface, was compared with sediment dry mass in 1 L water samples collected at the same time and from the same depth, using an ISCO automated water sampler and filtered on a GF/F filter. Both ASM sensors and the water sampler intake hose were placed together during the calibration period.