Limited vegetation development on a created salt marsh associated with over-consolidated sediments and lack of topographic heterogeneity.

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Abstract

Restored salt marshes frequently lack the full range of plant communities present on reference marshes, with upper marsh species under represented. This often results from sites being too low in the tidal frame and/or poorly drained with anoxic sediments. A managed coastal realignment scheme at Abbotts Hall, Essex, UK, has oxic sediments at elevations at which upper marsh communities would be expected. But seven years after flooding, it continued to be dominated by pioneer communities, with substantial proportions of bare ground, so other factors must hinder vegetation development at these elevations. These poorly vegetated areas had high sediment shear strength, low water and organic carbon content and very flat topography, characteristics which occur frequently on upper parts of created marshes. Experimental work is required to assess which, if any, of these is responsible for the ecological differences, although other studies have shown that topographic uniformity is associated with reduced plant β-diversity, lower usage by fish and means that the created marsh has very different visual appearance to a natural marsh. On the upper intertidal sediment deposition rates are low and erosive forces too weak to generate microtopographic variation, even where creeks have been excavated. So without active management intervention, these conditions will persist indefinitely.

Keywords: salt marsh plant species, de-embankment, Blackwater Estuary, abiotic conditions, habitat restoration, managed realignment

Introduction

Salt marshes are being created in increasing quantities worldwide to replace those destroyed by coastal erosion, land reclamation and coastal development (French 2006; Wolters et al. 2005; Zedler 2001). If created marshes are intended to compensate for areas destroyed by coastal development they should, as far as possible, match the ecological characteristics of natural marshes. In Europe such “compensatory measures ... should ... address, in comparable proportions, the habitats and species negatively affected [and] provide functions comparable to those... of the existing site” (European Commission 2008), and in the USA there is a requirement that marsh creation should result in no net loss of wetland area and function (Zedler 2004). In the UK, the majority of replacement salt marshes are created through the process of managed coastal realignment, where sea defences are relocated landward and the old seaward wall is breached to allow tidal inundation (French 2006). The areas to which tidal flooding is restored were often salt marshes before being embanked for use as agricultural land. Salt marshes develop naturally on sheltered sediment at the appropriate tidal elevation. If restored sites are at similar elevations to natural marshes, it might be expected that vegetation would develop on restored sites that is very similar to that on natural marshes (Burd 1995; French 2006; Parker et al. 2006). However, plant community development often fails to replicate the vegetation found on nearby natural reference marshes (Garbutt and Wolters 2008; Mossman et al. 2012b; Thom et al. 2002). Invertebrate communities can also be slow to develop, particularly higher in the tidal frame (Atkinson et al. 2001; Mazik et al. 2010). So most created salt marshes provide an imperfect substitute for natural marshes (Mossman et al. 2012a).

If we are to increase the likelihood of created marshes developing ecological characteristics that are similar to nearby reference marshes, we need to understand the environmental and ecological processes that lie behind this divergence. Some aspects of this problem are already well characterised. Shrinkage and compaction of soils after reclamation has led to areas being low in the tidal frame when reflooded (Crooks et al. 2002; Pethick 2002). At the managed realignment site at Tollesbury in Essex, south-east England, for example, much of the site was too low for marsh development (Garbutt et al. 2006). In addition, at tidal elevations corresponding to low and mid-marsh, the soils on re-flooded areas are often more anoxic than those at the same elevations on reference marshes, and this can still be true on sites flooded accidentally many decades ago (Mossman et al. 2012a). The anoxia results from poor drainage, at least in some cases due to the development of impervious layers or 'aquacludes' (Blackwell et al. 2004; Crooks et al. 1998, 2002). Anoxic or suboxic soils limit plant colonisation, favouring pioneer species that are tolerant of reducing soils and limiting the abundance of others (Davy et al. 2011; Mossman et al. 2012b). However, sections of a number of restored sites at elevations high in the tidal frame remain significantly less vegetated than those at equivalent elevations on natural reference marshes, despite having sediments that are well oxygenated (Mossman et al. 2012a). The reasons for this are unclear. One possible explanation is that hypersaline conditions may develop in summer, as occurs on unvegetated areas of natural marshes (Bertness 1991; Bertness et a. 1992). Most created marshes lack the creeks that occur on natural marshes, and this may also contribute to the ecological differences. On a natural marsh, Morzaria-Luna et al. (2004) found that plant species richness was slightly higher in areas close to creeks, and these areas also had higher topographic heterogeneity. Topographic variation is also associated with increased use of natural marshes by fish (Able et al. 2003). However, an experimental assessment at Friendship Marsh in the Tijuana Estuary showed that while creek excavation did generate topographic heterogeneity by "jump starting" the development of the drainage network (Wallace et al. 2005), the effect of this on plant survival was only modest (O'Brien and Zedler 2006) and although one fish species showed higher abundance where creeks were excavated, a second species occurred at lower abundance (Larkin et al. 2008).

In 2004, two years after restoration of tidal inundation to a managed realignment at Abbotts Hall in Essex, UK, vegetation colonisation was extensive, with only 30% bare ground (Mossman 2007). Much of the re-flooded area was at elevations at which upper marsh communities usually occur, and sediments were oxic throughout much of the site. In addition, creeks had been excavated on part of the site. Seven years after restoration (in 2009), we carried out intensive sampling to assess vegetation development on a site where the most common impediments to vegetation development, such as anoxic sediment and unsuitable elevation, were absent. We expected that the rapid pace of vegetation colonisation would have continued. However, development of typical high marsh communities dominate by perennial species had not taken place. Here we quantify the environmental characteristics at the site to try to understand what may be responsible for this.

Methods

The study area

Abbotts Hall is situated on the north bank of the Blackwater Estuary in Essex, south east England (Fig. 1) (51° 47′ 8″ N, 0° 50′ 43″ E). The was been reclaimed in the 18th Century and used as grazing marsh, then was levelled and converted for arable use between 1943 and 1970 (Essex Wildlife Trust 2003). In 1996, tidal flow was restored to the area shown in Fig 1 via a culvert (Nottage and Robertson 2005). As part of the 1996 scheme, a network of artificial creeks was excavated in an attempt to recreate natural marsh drainage conditions and sediment deposition (Dixon et al. 1998). The amount of seawater reaching the site was lower than expected (ABP 1998) and vegetation development was very poor (Diack 1998), so in 2002, a larger area (49 ha) of intertidal habitat was created by breaching the sea wall in five places, including sections lying outside of the area shown in Fig. 1. The creeks excavated in 1996 were 1-3 m wide and deep, with a total length of 2.2 km, and are clearly visible on Fig. 1. Spoil from these excavations was left in situ in a pile adjacent to the excavated creeks (an example is visible at right hand side of Fig. 2b). The tops of these piles were just above the level of the highest tides. An area of natural reference marsh (approximately 70 ha) occurs adjacent to the managed realignment site. Extensive vegetation colonisation had occurred on the managed realignment two years after the sea wall was breached and sediments then were relatively well oxygenated (Mossman 2007), and five years after breaching the vegetation was described as “similar to the adjacent ancient saltmarsh” based on data from a single transect (Hughes et al. 2009).

We studied three areas of marsh. These were an area where no creeks had been excavated (‘non engineered site’, area A in Figs. 1 and 2), one that included an excavated creek (‘engineered site’, area B in Figs. 1 and 2) and one on an area of natural reference marsh (‘reference site’, area C in Figs. 1 and 2). On both parts of the restored site, the landward part of the marsh consists of a relatively well vegetated and rather flat surface. This is visible as darker areas on Fig. 1 and these can also be seen in Fig. 2a and b. It is located at approximately 2.25 to 2.5m ODN (see also Fig. 6), an elevation at which species-rich marshes dominated by *Puccinellia maritima* and *Atriplex portulacoides* normally occur in this region (Mossman 2007). Seawards of this is an area of mudflat with pioneer vegetation, visible as the lighter coloured parts of the site in Fig. 1. As we wanted to focus on factors influencing the vegetation on the upper parts of restored marshes, the areas sampled were located approximately mid-way between the landward sea wall and the areas of mudflat and pioneer communities. To allow comparison with the restored sites, the reference site was located in an area where the main marsh surface was at an elevation of around 2.25m ODN, but as the reference marsh is quite heavily dissected by creeks, some sampling points are located lower in the tidal frame (c.f. Fig. 6).

Many restored marshes appear to be topographically more uniform than reference marshes and this could be a cause of ecological differences (Grant and Mossman, *pers. obs.,* and compare figure 2c with figures 2a and b). To allow us to quantify spatial heterogeneity of elevation on small and intermediate scales, at each of the sites we used systematic sampling of a grid rather than using random or transect sampling. A square 6 by 6 grid at 10 m spacing was laid out in a 50 x 50m section of each area, with 36 sampling points at the intersections of the grid lines. Eight of these points were randomly selected and additional sampling points were located at 1 m and 2 m N, S, E and W from the focal point; giving an additional 64 sampling points. The same eight positions were used in each site to give consistency in sampling spatial patterns. This sampling design resulted in a total of 100 sampling points at each of the three sites. As noted above, all three areas were at relatively high tidal elevations and the area covered by the sampling grid had vegetation that was visually similar to the whole of that section of the site.

In August 2009, the elevation, vegetation and soil characteristics were assessed at each of the sampling points. The vegetation was assessed by estimating the percentage cover in a 0.25 m^2 quadrat (centralised over each sampling point) of all plant species and bare ground, to the nearest 5%, assigning an abundance of 1% to species present at < 5% area. Plant species observed within each sampling grid but that did not fall into any quadrats were also noted. Plant species nomenclature followed Stace (2001).

At each sampling location, elevation of the marsh surface relative to Ordnance Datum Newlyn (ODN - the UK national topographic datum, based on mean sea level at a site in south west England) was measured at the centre of each sampling point using a differential GPS (Topcon, Newbury, UK), with an accuracy of <2 cm and precision of <1.5 cm. The levels of mean high water neap (MHWN) and mean high water spring (MHWS) tides, measured on-site, were respectively 1.83 and 2.99 m ODN (Mossman et al. 2012c). Substrate redox potential was measured at low tide with a single reading from the centre of the sampling unit, taken after 90 seconds of equilibration at 4 cm below the marsh surface using a BDH Gelplas combination redox electrode with an Ag/AgCl reference. Values are expressed relative to a standard hydrogen electrode (Eh) by adding 204 mV. Redox potential was not taken at two locations on the engineered site because the substrate was too hard for the probe to be inserted. Soil shear strength was measured using a hand-held shear vane with a precision of <1 kPa. Where the readings exceeded the maximum of the scale (>100 kPa), a value of 120 kPa was used in the statistical calculations. Approximately 40 g of soil from the surface 5 cm was taken from the centre of each of the 36 main grid sample points at each of the three sites, placed a sealed bag and stored at 4 °C within 48 h. Approximately 5 g of these un-sieved sediment samples were analyzed in the laboratory for gravimetric soil water content (after drying for 24 hours at 80^{o}C) and organic matter (percentage loss on ignition after 20 h at 390 °C) in accordance with standard methods (Miller 1982). Two replicates of gravimetric water content and loss on ignition were made for each sample and the mean used in statistical analyses. We did not measure sediment nutrient concentrations as these are unlikely to be limiting as the site had previously been used for intensive arable farming and our previous work has found relationships between sediment nutrient concentrations and vegetation on created marshes to be weak or absent (Davy et al. 2011).

Initial data analysis and fitting of LOESS regressions (locally weighted non-linear regressions, using an Epenechnikov kernel) to relationships between elevation and individual plant abundances and environmental variables, were carried out using SPSS version 16 (SPSS Inc, Chicago, Ill.). In order to examine, and test the significance of, differences in vegetation communities between sites, multidimensional scaling (MDS) and one-way analysis of similarity (ANOSIM, which tests whether similarities between groups are smaller than expected by chance using permutation tests) were carried out on a matrix of Bray-Curtis similarities of vegetation data after square root transforming abundances. SIMPER (Similarity Percentage) analysis is a method for assessing which taxa are primarily responsible for an observed difference between groups of samples and was used to identify the species responsible for the differences in vegetation communities between the three sies. MDS, ANOSIM and SIMPER were carried in PRIMER v6 (Plymouth Routines In Multivariate Ecological Research, Primer-E, Plymouth; Clarke and Gorley 2006). Heterogeneity of site elevations was quantified using R (R Development Core Team 2008), writing our own code to calculate horizontal distances and elevation differences between all pairs of sampling points at each site and using the *stats* package to fit LOESS regressions.

Results

Vegetation data

A total of 16 plant species were found, although two of these, *Picris echioides* and an unidentified moss, were essentially non-maritime species that occurred only on the highest part of the engineered site, i.e. on creek berms above the high-tide level. Multidimensional scaling showed that the plant communities on both the engineered and non-engineered parts of the created marsh remain markedly different from those on the reference marsh (Fig. 3). None of the samples from either part of the created marsh plot in the area at the top and centre of Fig. 3 where the bulk of the reference samples are located, although the engineered part of the site showed greater similarity to the reference marsh than the non-engineered part. In addition, a small number of reference samples, mostly from the low marsh, *are* similar to samples from the created marsh. ANOSIM confirms that pairwise differences between the three sites are significant ($P < 0.001$ in all cases). There are a number of species contributing to these differences, and the mean abundances of the commonest species are presented in Fig. 4. The abundance of individual species changed with elevation, as well as showing differences between sites, so we have used LOESS regressions to visualise these patterns in more detail and demonstrate that the differences between the sites are not simply due to differences in elevation (Fig. 5). On the reference marsh there was 100% vegetation cover in most quadrats at elevations greater than 2.1 m ODN, but vegetation cover remained incomplete at elevations above this on both parts of the created marsh, with about 30% bare ground at 2.25 m ODN on the engineered site, and about 60% on the non-engineered part (Fig. 5a). Two species, *Puccinellia maritima* and *Atriplex portulacoides* were common between 2 m and 2.6 m ODN on the reference marsh, with *Puccinellia* dominating between 2.3 and 2.5 m ODN, and abundance of *Atriplex* peaking at around 2.2m ODN. At elevations below 2 m ODN, *Salicornia europaea* was the commonest species, although *Atriplex* and *Puccinellia* were present in a small number of quadrats. *Atriplex* and *Puccinellia* were present at much lower abundance, and occurred at higher elevations, on the engineered site and were rather rare on the non-engineered site (Fig. 5d, e). SIMPER analysis indicates that the abundance of these two species and the occurrence of bare ground made the largest contribution to the differences between sites. The annuals *Salicornia europaea* and *Suaeda maritima* were more common on the realignment site, with the latter particularly abundant on the non-engineered part, peaking at about 80% cover at 2.5 m ODN (Fig. 5b, c).

Other species were present at much lower abundance, so data are not presented graphically. The grass *Elytrigia atherica* occurred at high abundance (> 60% cover) in some quadrats, almost entirely those between 2.5 and 2.75 m ODN, but was absent from the majority of quadrats. It occurred at a cover of > 50% in more quadrats (15) on the non-engineered site than on the engineered or reference sites (5 and 8, respectively), which was significantly different from a uniform distribution (χ^2= 6.2, d.f. = 2, P = 0.04). However, 28% of sampling points on the non-engineered site were between 2.5 and 2.75 m ODN, as compared with only 6 and 7% for the engineered and reference sites, and if we calculate expected values based on these proportions, the difference was not significant (χ^2 = 3.6, d.f. = 2, P = 0.17). Four characteristic mid to upper marsh species (*Sarcocornia perennis*; *Triglochin maritima*; *Plantago maritima* and *Limonium vulgare*) were present on the reference marsh but absent from both parts of the realignment site, and *Aster tripolium* was more than ten times more abundant on the reference site. *Agrostis stolonifera* and *Spartina anglica* occurred only on the engineered site. One-way Anova showed that, apart from *Plantago* ($F_{2,287}$ = 2.8, N.S.) and *E. atherica* ($F_{2,287}$ = 3.5, P = 0.03), all the above differences in mean abundance between sites were significant at P < 0.001.

Environmental characteristics

There were also substantial differences between the sites in their environmental characteristics, displayed visually for elevation in Fig. 6 and using LOESS regressions against elevation for the other parameters in Fig. 7. On the non-engineered area, most of the sampling points fell within a restricted elevation range, with only a few samples close to a former drainage ditch lying below 2m ODN. Sampling points on the engineered area covered a much larger elevation range because creeks were excavated to about 1.25m ODN, and the spoil from this excavation was left as a berm, with a crest that was well above 3 m ODN. On the reference site, the highest sampling points are at similar elevations to the highest parts of the non-engineered site, but the whole marsh is dissected by creeks, so there were many more low lying sampling points.

The sites differed in mean elevation, redox, shear strength, water and organic content (Table 1; $P < 0.001$ in all cases, Anova). The non-engineered parts of the realignment site had the *highest* mean elevation, as the absence of any creeks results in no sampling points on this site being below 1.94 m ODN (Fig. 6). The mean redox potentials at all elevations in all sites were above 0 mV (Fig. 7a). The non-engineered site also had a markedly higher mean redox potential, because this declined steadily with elevation and almost all sampling points below 2 m ODN were on the other two sites. Nevertheless, the redox potential on the non-engineered site was somewhat higher than at similar elevations on the other two sites (Fig. 7a). Sediment shear strength was high throughout the non-engineered site, with most quadrats having shear strengths greater than 80 kPa (Fig. 7b). By contrast, shear strengths on the reference marsh were lower than 40 kPa in sampling points below 2.3 m ODN. They increased on the higher marsh, but very few sampling points exceeded 80 kPa. On the engineered site, shear strengths were higher than those on the non-engineered site above 2.3 m ODN, declining to very low values at the lowest elevations. Sediment water content at 2 m ODN on all sites was in the region of 50%, but declined steadily with increasing elevation on both the engineered and non-engineered realignment sites (Fig. 7d). By contrast, water content increased on the upper parts of the reference marsh, where sediment organic content was also markedly higher than on the realignment sites (Fig. 7c, d).

Environmental heterogeneity

The data on elevation heterogeneity confirmed the visual impression given by the photographs in Fig. 2. On the reference marsh, locations that are 1 m apart horizontally typically differed by 15 cm in elevation, rising to about 30 cm for locations 8 m apart (Fig. 8), reflecting the dissection of the site by an extensive system of creeks (Fig. 2c). By contrast, locations up to 10 m apart on both parts of the realignment site were likely to differ in elevation by only 10 cm, reflecting the flat surface topography visible in Figs. 2a and b. The three sites only showed similar patterns of heterogeneity in elevation at horizontal scales greater than 40 m.

Discussion

Seven years after full restoration of tidal influence, the vegetation at the restored salt marsh at Abbotts Hall continues to be dominated by the two annual species, *Salicornia* and *Suaeda*, with substantial areas of bare ground. There is rather little colonisation by perennial species normally characteristic of mid and upper marshes, despite much of the site being at appropriate elevations. The failure of vegetation communities to develop on restored or created sites within similar timeframes has been noted previously (e.g. Havens et al. 2002; Mossman et al. 2012a; Wolters et al. 2005). At some restored sites, the delay in colonisation by mid and upper marsh species has been attributed to relatively low tidal elevations and poorly oxygenated sediments, which are more suitable for inundation-tolerant, pioneer species, such as *Salicornia* (Crooks et al. 2002; Davy et al. 2011; Mossman et al. 2012b). This is not the case for much of the Abbotts Hall site, which is at elevations suitable for mid and high marsh communities, and has oxic sediments. It is not clear why these communities have not rapidly developed following the restoration of tidal flow.

Initially, Abbotts Hall was rapidly colonised by *Salicornia* and *Suaeda*, and after two years the mean percentage of bare ground was only 35% (Mossman 2007). But in 2009, seven years following tidal restoration, both the engineered and non-engineered parts of the restored site still had very different vegetation from the adjacent reference marsh. A substantial proportion of bare ground remained and the two annual species, *Salicornia* and

Suaeda were also still abundant. Vegetation on the engineered part of the site was more similar to the reference marsh as a result of colonisation of the upper elevations by *Puccinellia* and *Atriplex*. However, the abundance of *Salicornia* and bare ground remained higher than on the reference marsh. This lack of colonisation by higher marsh species is despite the area being slightly *higher* in the tidal frame than the reference site, and having sediments that are slightly better oxygenated at any given elevation than those on the other two areas. So whilst suitable elevations in the tidal frame and moderately oxic sediments are necessary conditions for development of mid-marsh communities, they do not guarantee success. In the UK, most managed realignment sites are relatively low in the tidal frame, but we have observed similar failure of vegetation development on higher sections of some other restored marshes (Mossman et al. 2012a). What might be responsible for this failure?

Perhaps there has simply not been sufficient time for the vegetation to fully develop. It is true that succession can be relatively slow, but on natural marshes the rate of succession from mudflat to pioneer communities to the colonisation by later successional mid and upper marsh species is usually constrained by rates of sediment deposition (Boorman 2003). In locations where natural marsh development has been initiated by a change in the local hydrodynamic or sedimentary environment rather than by gradual sediment deposition, succession takes place rapidly. Species-rich vegetation, including high marsh perennials such as *Armeria maritima; Juncus maritimus* and *Triglochin maritima*, developed in 17 years on an area of previously bare sand in North Wales (Packham and Liddle 1970). In the Netherlands, Olff et al. (1997) found that ten years after initial colonisation, vegetation was dominated by *Spergularia maritima, Suaeda maritima, Limonium vulgare* and *Puccinellia maritima*. Natural recolonisation is also rapid when turf is stripped from existing salt marshes; *Puccinellia maritima* recolonises within a year, even at elevations above its normal range, and re-establishment of full vegetation cover is complete after 3-5 years (Cadwalladr and Morley 1974; Gray and Scott 1977a, b). In contrast to these examples, restored sites flooded accidentally more than a century ago still show marked differences in vegetation communities (Mossman et al. 2012a). Are the “missing” species those that disperse only slowly to created marshes? Some perennial species are rare on created marshes of all ages, even many decades after restoration (Mossman et al. 2012a). Colonisation by some perennials, such as *Limonium vulgare* , *Plantago maritima* and *Triglochin maritima*, may be slowed by low seed viability and long reproductive cycles (Boorman 1967; Hutchings & Russell 1989; Davy & Bishop 1991). But the two species that make the greatest contribution to the difference between the created and reference marshes at Abbotts Hall are *Atriplex portulacoides* and *Puccinellia maritima*. Both of these have rapidly colonised other managed realignment sites and are often dominant on older accidentally restored sites (Mossman 2007).

What other environmental factors might be responsible? Sediments on the created marsh have higher shear strength, and lower water and organic matter contents than the reference marsh. So water availability will be low and it will be difficult for plant roots to penetrate the soil. Establishing which, if any, of these possible factors are responsible will require field experimentation. Addition of organic matter to soils on Friendship Marsh in the Tijuana Estuary increased plant survival rates (O'Brien and Zedler, 2006), although it is not possible to judge the relative importance of the effects of increased organic matter, nutrient content and decreased bulk density. We have not measured soil nutrient concentrations, because these are unlikely to be limiting at a site previously used for intensive arable farming before flooding. At another site, plant species distributions showed weak or non-significant relationships with soil nutrients were measured (Davy et al. 2011).

The reference marsh shows considerable topographic heterogeneity, even on horizontal scales of a few metres. This is typical of natural marshes, and develops, at least in part, during the early stages of plant colonisation of mudflats and results in heterogeneity of sediment redox (Stribling et al. 2006, 2007). This in turn produces heterogeneity of plant communities (Morzaria-Luna et al. 2004; Varty and Zedler 2008). By contrast, the restored marsh surface lacks topographic heterogeneity, as a result of levelling for arable farming. This is true of many managed realignment sites, and means that their visual appearance is rather different to that of natural marshes, in ways that are obvious even to people without detailed knowledge of their ecology (c.f. Fig. 2). As well as being responsible for a lack of ecological heterogeneity, the lack of topographic variation may help to create conditions that are hostile to plant colonisation. When the restored marsh has been flooded by a high spring tide, water will drain away much more slowly than from a marsh with a more heterogeneous surface and natural, or artificial, creeks. Many managed realignment sites that were previously used for arable farming have substantial areas of shallow standing water after high spring tides (pers. obs.). If hot, dry weather follows a period of spring tides, evaporation of this standing water will produce high surface salinities, inhibiting germination and growth, even of halophytes. A similar process was responsible for the slow recolonisation of large open areas created by ice scour observed on New England marshes (Bertness et al. 1993; Shumway and Bertness 1992, 1994), where initial plant colonisation provides shading, reduces evaporation and salt stress and facilitates the colonisation of other plants (Bertness 1991).

If topographic uniformity does contribute to the failure of later successional plants to colonise, then the restored marsh could become more similar to reference sites over time if sediment erosion and/or deposition generate more natural topography. This can occur when the restored sites are at low elevations as patchy vegetation colonisation may also increase topographic heterogeneity (Stribling et al. 2007). But rates of sediment supply are low on the high marsh, so more varied microtopography cannot develop as a result of differential deposition. Erosion could generate heterogeneity on higher sites if sediments are relatively soft. But at Abbotts Hall, high sediment shear strengths will inhibit erosion and high in the tidal frame flooding is less frequent and water velocities, and therefore erosive forces, are lower (Atkinson et al. 2001; Wallace et al. 2005). This may be why excavation of creeks alone has limited impacts on the upper marsh topography (Wallace et al. 2005) and on plant survival there (O'Brien and Zedler, 2006).

It is likely that the high sediment shear strength results from deflocculation of clay minerals in the low salinity conditions during aerial exposure, leading to loss of soil structure. This in turn results in over-compaction when flooded by sea water, a process that has led to the widespread development of over-consolidated sediments in the Holocene estuarine deposits of South-East England (Crooks 1998; Crooks et al. 2002). These consolidated layers can impede drainage in freshly deposited sediment and contribute to the development of anoxic conditions at low lying sites (Crooks et al. 2002), and over-compaction of sediments also seems to have hindered the ecological development of the Seal Sands managed realignment (Evans et al. 1998). Once a marsh has been flooded, large scale sediment treatment to break up over-consolidated sediment is extremely challenging. As noted above, excavation of creeks seems to be of limited value on high marsh sites, and it may be necessary to carry out more aggressive surface treatment to prevent over-consolidated layers forming. This might involve deep ploughing or even the mixing of soil and seawater into a slurry before flooding. Experimental evaluation of the options is required, but if over-consolidated sediments can be avoided, this will result in a created marsh surface that is more likely to undergo small scale erosion and deposition to generate topographic heterogeneity. This will certainly create a marsh with a more "natural" appearance and one that may also be less hostile for plant colonisation and provide microsites suitable for the growth of different species (Varty and Zedler 2008).

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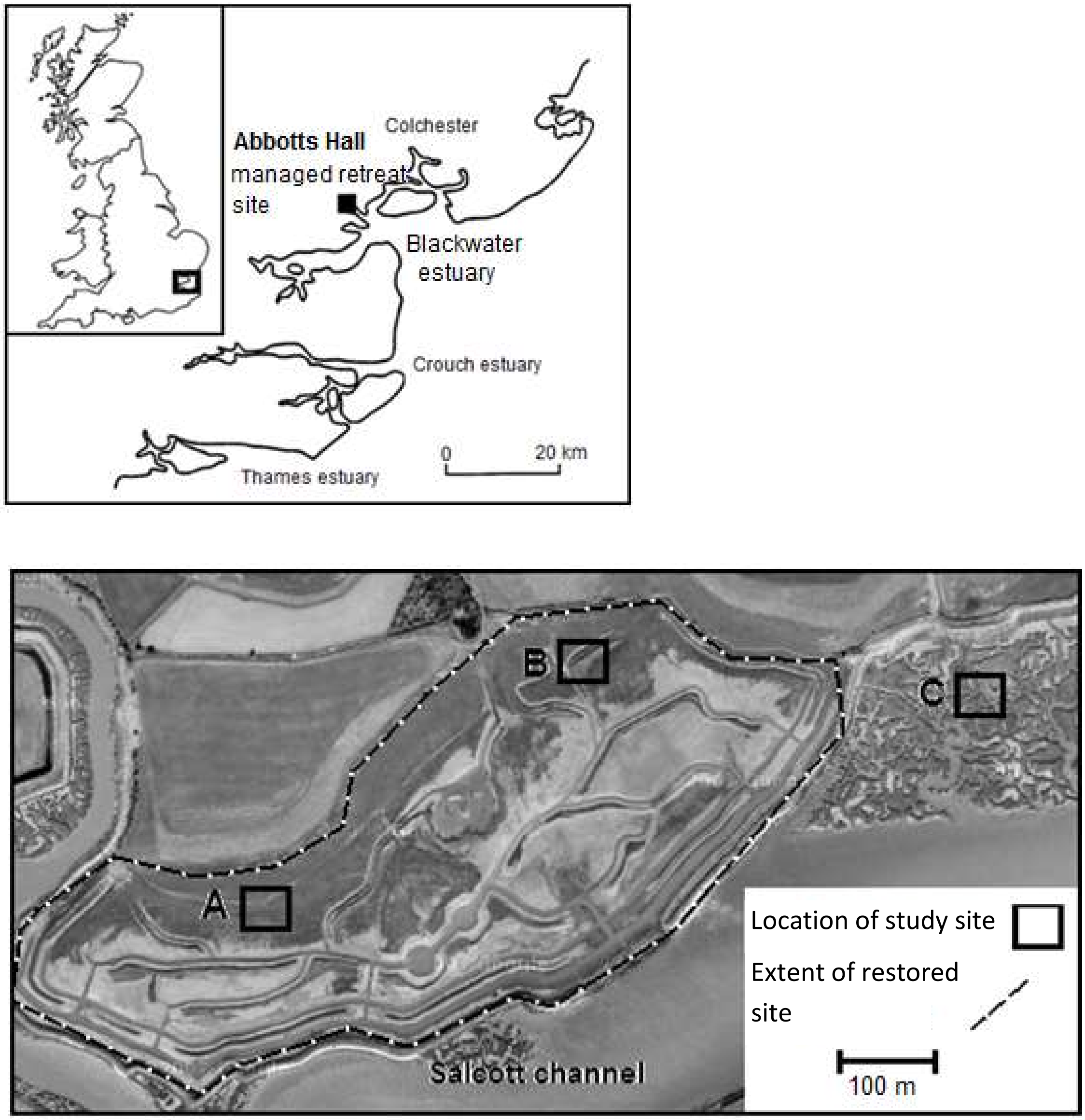


Fig. 1. Location of study site within the UK and location of three sampling areas; A = Non-engineered part of the restored salt marsh; B = Engineered part of the restored salt marsh; C = Reference salt marsh. Aerial photography from Google maps (© Google, 2009)

a

b

c

Fig. 2. Photographs showing characteristics of each site. Taken 22/08/09. A) non-engineered part of the restored salt marsh; b) engineered part of the restored salt marsh, with embankment of spoil from excavating creek to right; c) reference site

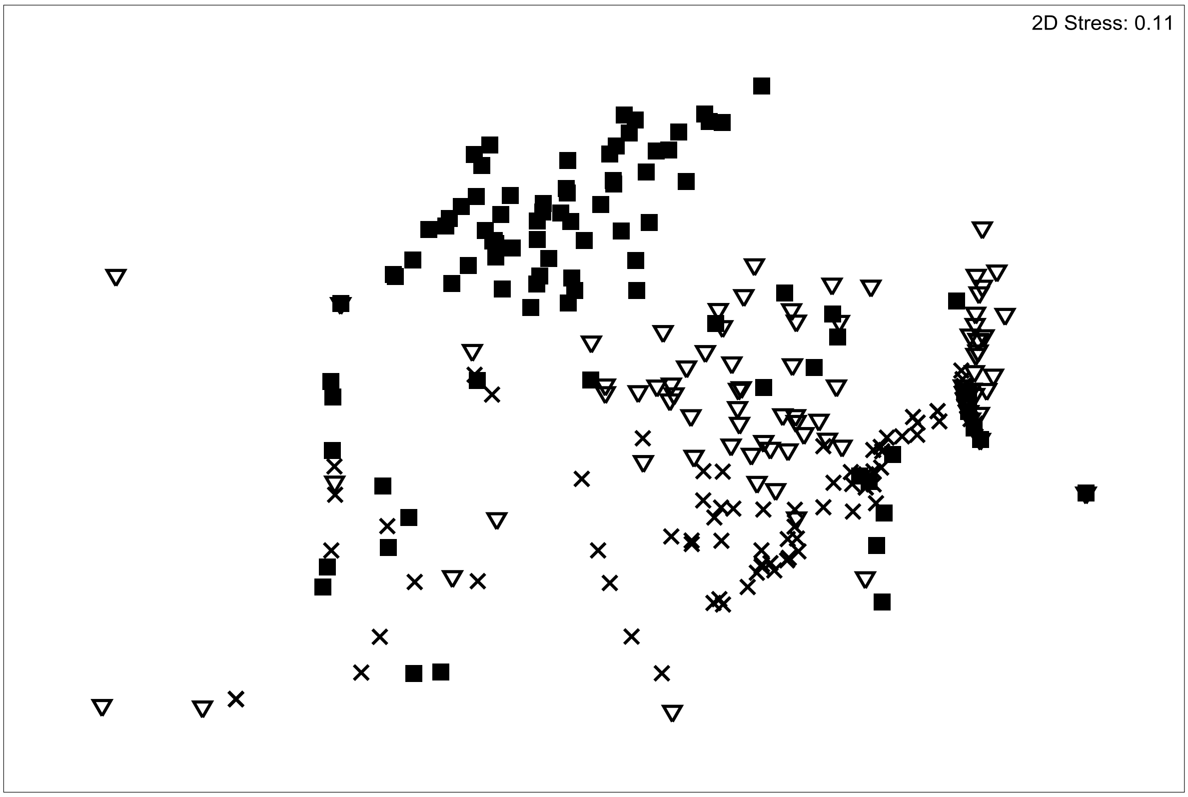


Fig. 3. MDS plot of vegetation data at Abbotts Hall realignment site and adjacent reference marsh. ■ indicates reference site, ∇ engineered part of the restored saltmarsh; × non-engineered part of the restored saltmarsh

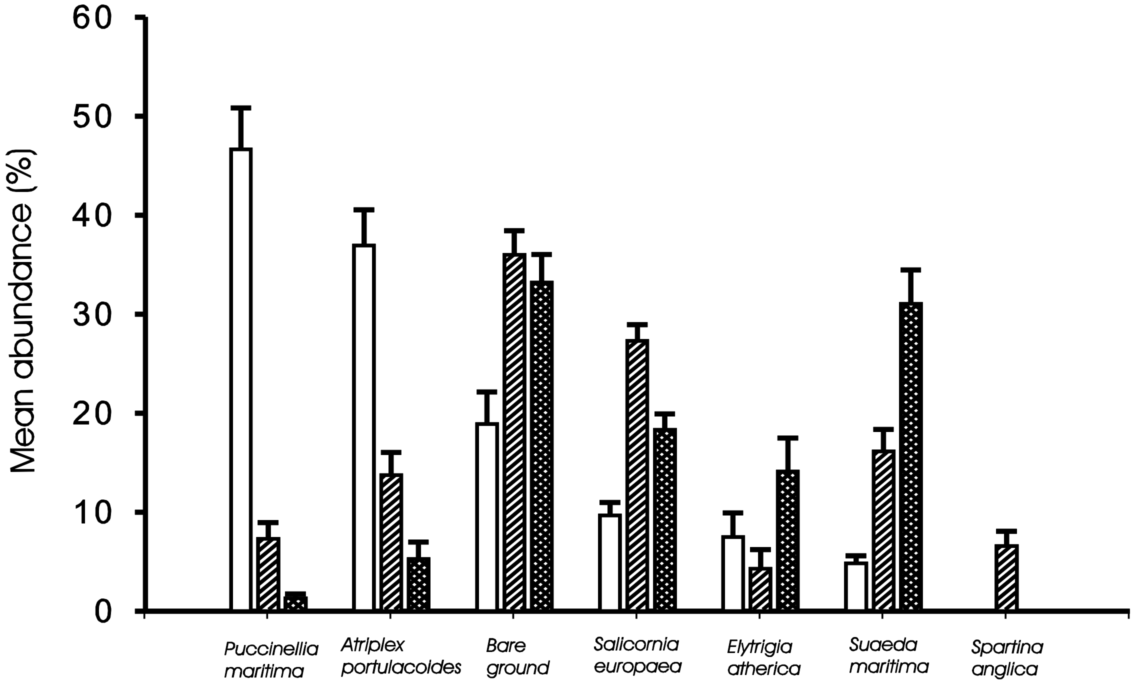
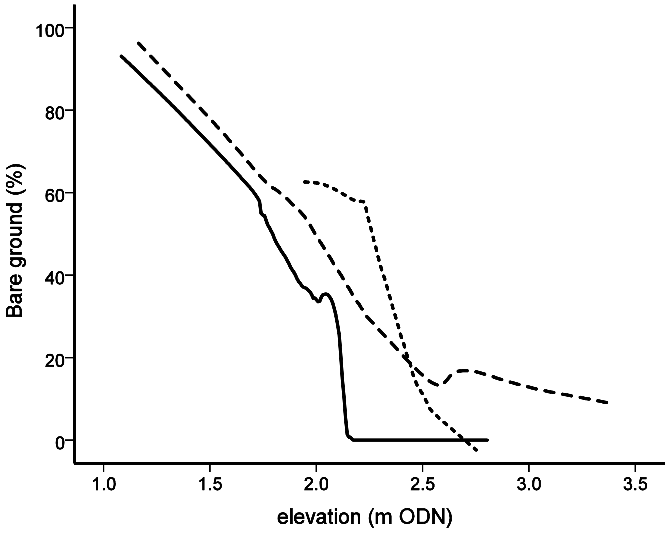
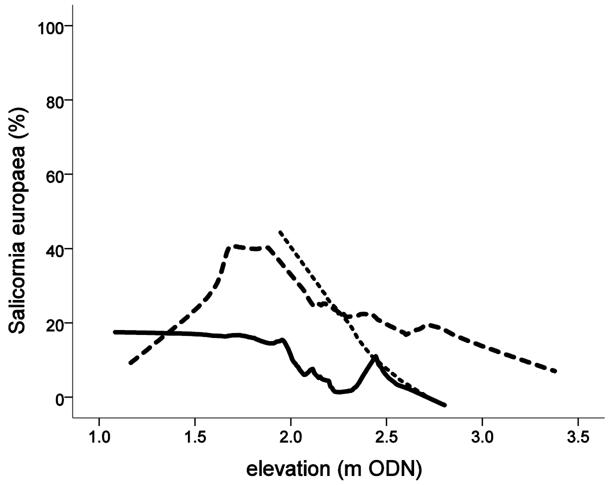


Fig. 4. Mean abundance of six commonest species and bare ground on the reference salt marsh (open bars) and engineered (diagonal) and non-engineered (cross-hatched) parts of the restored salt marsh. All species with mean abundance > 5% in at least one of the three areas are plotted. Error bar is 1 standard error. Differences between sites are significant ($P<0.001$) for all species.

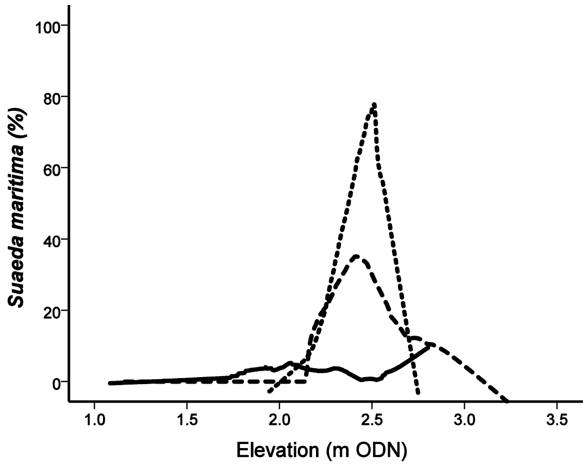
a


Bare ground (%)
100
80
60
40
20
0
1.0
1.5
2.0
2.5
3.0
3.5
elevation (m ODN)

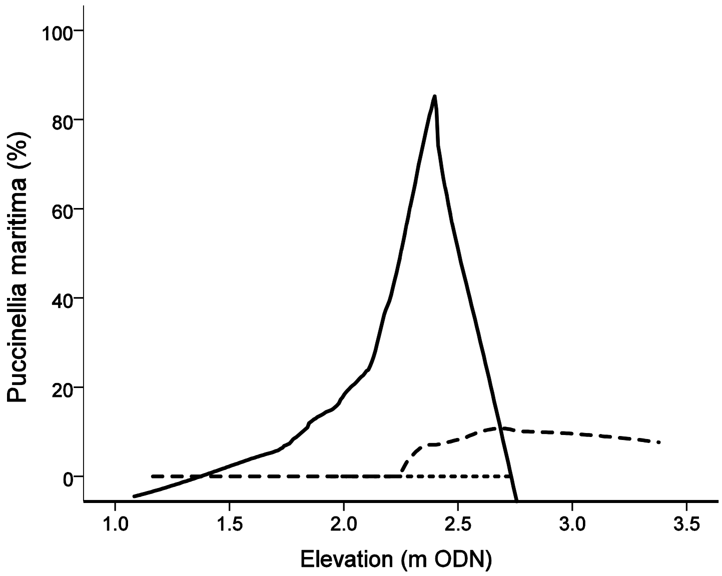
b


Salicornia europaea (%)
100
80
60
40
20
0
1.0
1.5
2.0
2.5
3.0
3.5
elevation (m ODN)

c


Suaeda maritima (%)
100
80
60
40
20
0
1.0
1.5
2.0
2.5
3.0
3.5
Elevation (m ODN)

d



e

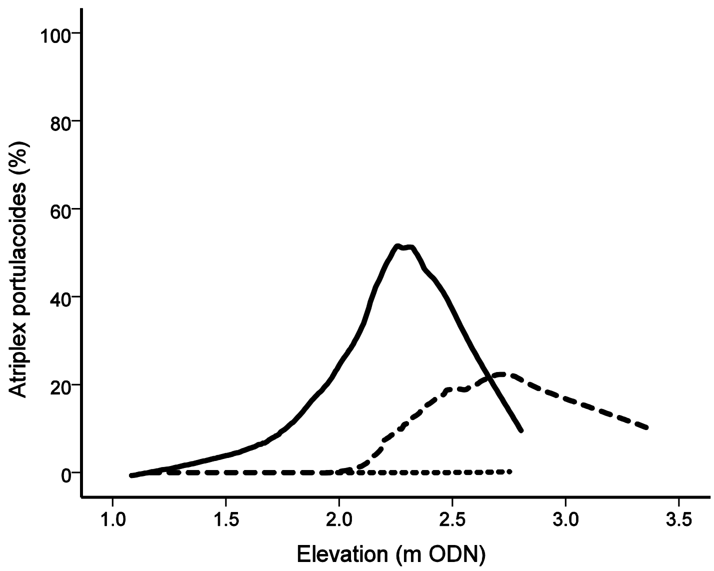


Fig. 5. LOESS regressions showing relationship between percentage cover of individual plant species and elevation at Abbotts Hall restored and reference salt marshes. a) bare ground *,* b) *Salicornia europaea* ; c) *Suaeda maritima* d) *Puccinellia maritima* e) *Atriplex portulacoides.* Solid line – reference site; dashed line engineered part of the restored site; dotted line – non-engineered part of the restored site

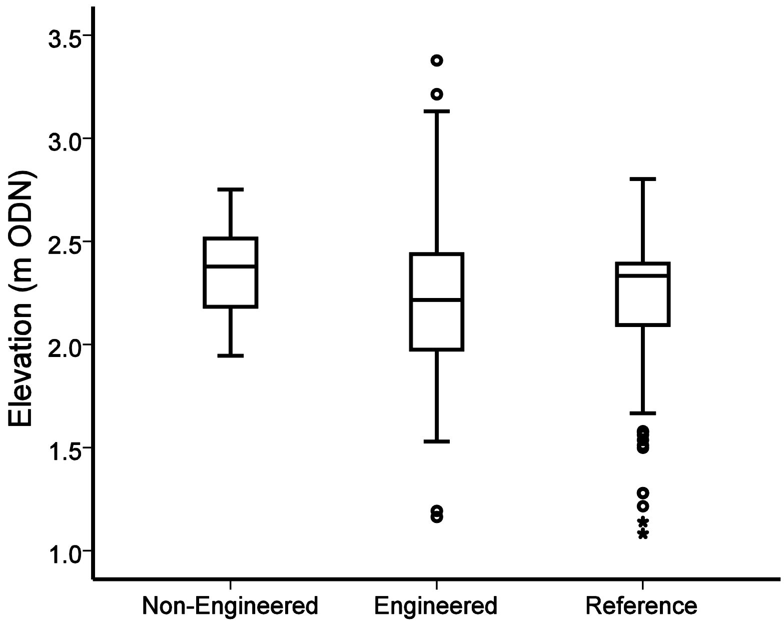
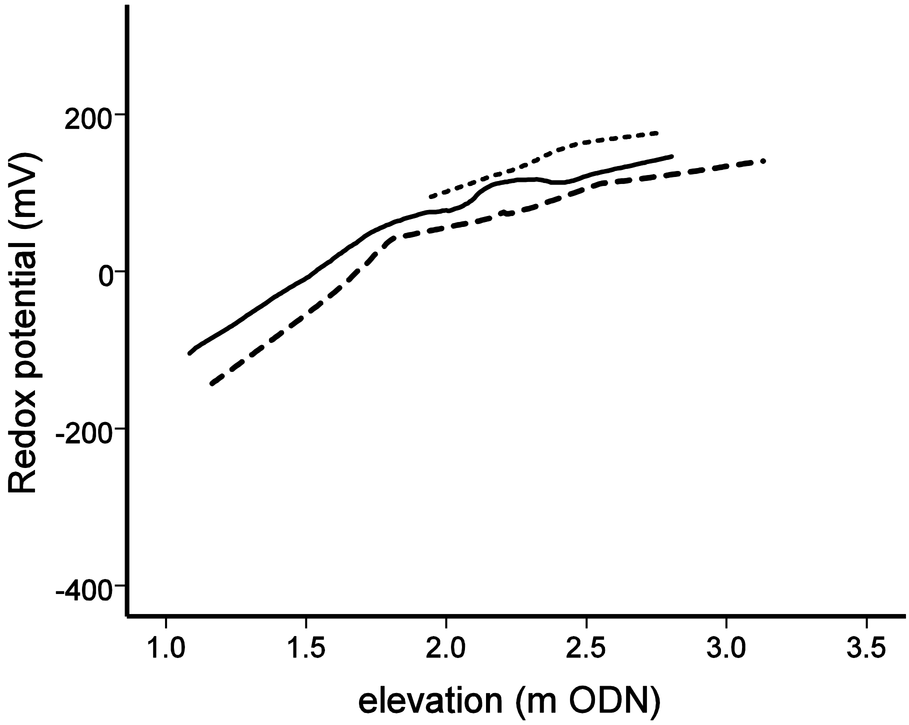
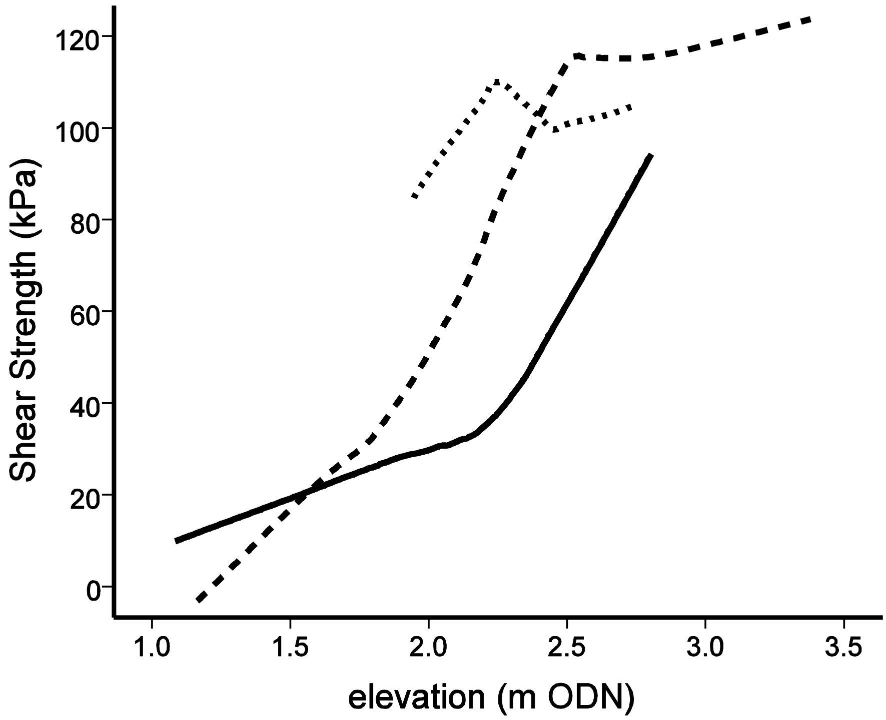


Fig. 6. Range of elevations occurring at the engineered and non-engineered part of the Abbotts Hall restored salt marsh and at the adjacent reference marsh

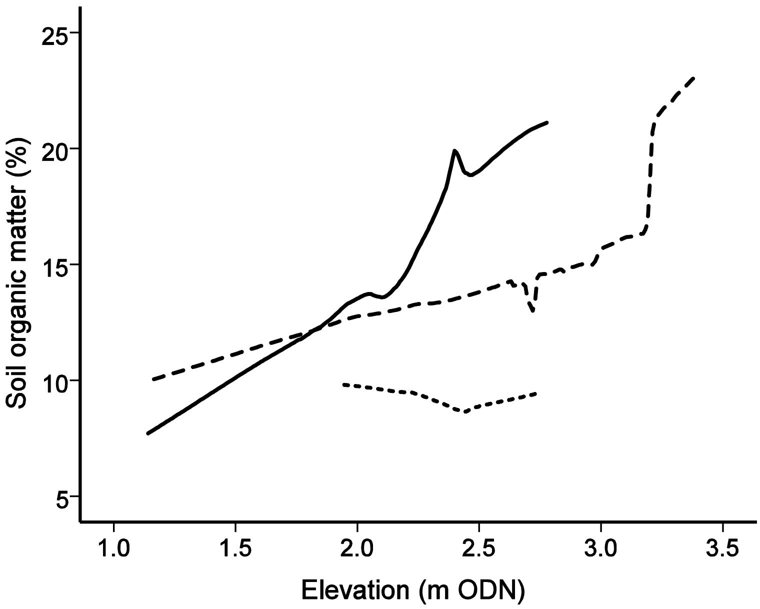
a


Redox potential (mV)
200
0
-200
-400
1.0
1.5
2.0
2.5
3.0
3.5
elevation (m ODN)

b


Shear Strength (kPa)
120
100
80
60
40
20
0
1.0
1.5
2.0
2.5
3.0
3.5
elevation (m ODN)

c



d

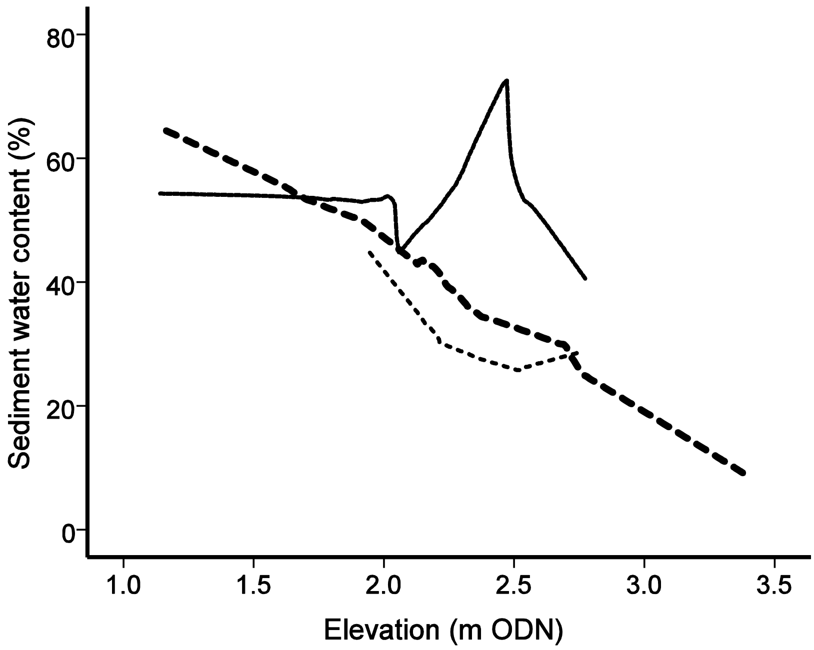


Fig. 7. LOESS regressions showing relationships with elevation of a) redox; b) shear strength; c) soil organic matter and d) water content. Details as Fig. 5.

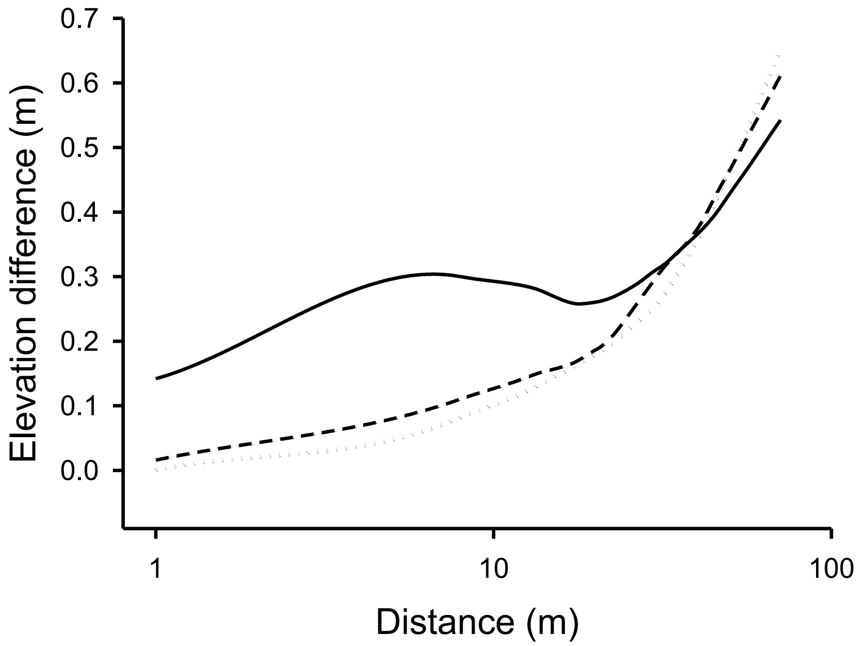


Fig. 8. Spatial scales of topographic heterogeneity on the three sites at Abbotts Hall, quantified using the relationship of elevation difference between pairs of sampling points to the horizontal distance between them. Lines are LOESS regressions fitted to all pairwise differences between sampling points. Dotted line shows relationship for the non-engineered part of the restored salt marsh, dashed line for the engineered part of the realignment site and solid line for the adjacent reference salt marsh.