

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

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13

14 **Abstract**

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

31

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

34

35 **Introduction**

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

58

59 If we are to increase the likelihood of created marshes developing ecological characteristics
60 that are similar to nearby reference marshes, we need to understand the environmental
61 and ecological processes that lie behind this divergence. Some aspects of this problem are
62 already well characterised. Shrinkage and compaction of soils after reclamation has led to
63 areas being low in the tidal frame when reflooded (Crooks et al. 2002; Pethick 2002). At the
64 managed realignment site at Tollesbury in Essex, south-east England, for example, much of
65 the site was too low for marsh development (Garbutt et al. 2006). In addition, at tidal

66 elevations corresponding to low and mid-marsh, the soils on re-flooded areas are often
67 more anoxic than those at the same elevations on reference marshes, and this can still be
68 true on sites flooded accidentally many decades ago (Mossman et al. 2012a). The anoxia
69 results from poor drainage, at least in some cases due to the development of impervious
70 layers or 'aquaculdes' (Blackwell et al. 2004; Crooks et al. 1998, 2002). Anoxic or suboxic
71 soils limit plant colonisation, favouring pioneer species that are tolerant of reducing soils
72 and limiting the abundance of others (Davy et al. 2011; Mossman et al. 2012b). However,
73 sections of a number of restored sites at elevations high in the tidal frame remain
74 significantly less vegetated than those at equivalent elevations on natural reference
75 marshes, despite having sediments that are well oxygenated (Mossman et al. 2012a). The
76 reasons for this are unclear. One possible explanation is that hypersaline conditions may
77 develop in summer, as occurs on unvegetated areas of natural marshes (Bertness 1991;
78 Bertness et al. 1992). Most created marshes lack the creeks that occur on natural marshes,
79 and this may also contribute to the ecological differences. On a natural marsh, Morzaria-
80 Luna et al. (2004) found that plant species richness was slightly higher in areas close to
81 creeks, and these areas also had higher topographic heterogeneity. Topographic variation is
82 also associated with increased use of natural marshes by fish (Able et al. 2003). However, an
83 experimental assessment at Friendship Marsh in the Tijuana Estuary showed that while
84 creek excavation did generate topographic heterogeneity by "jump starting" the
85 development of the drainage network (Wallace et al. 2005), the effect of this on plant
86 survival was only modest (O'Brien and Zedler 2006) and although one fish species showed
87 higher abundance where creeks were excavated, a second species occurred at lower
88 abundance (Larkin et al. 2008).

89

90 In 2004, two years after restoration of tidal inundation to a managed realignment at
91 Abbots Hall in Essex, UK, vegetation colonisation was extensive, with only 30% bare ground
92 (Mossman 2007). Much of the re-flooded area was at elevations at which upper marsh
93 communities usually occur, and sediments were oxic throughout much of the site. In
94 addition, creeks had been excavated on part of the site. Seven years after restoration (in
95 2009), we carried out intensive sampling to assess vegetation development on a site where
96 the most common impediments to vegetation development, such as anoxic sediment and
97 unsuitable elevation, were absent. We expected that the rapid pace of vegetation

98 colonisation would have continued. However, development of typical high marsh
99 communities dominated by perennial species had not taken place. Here we quantify the
100 environmental characteristics at the site to try to understand what may be responsible for
101 this.

102

103 **Methods**

104 **The study area**

105 Abbots Hall is situated on the north bank of the Blackwater Estuary in Essex, south east
106 England (Fig. 1) ($51^{\circ} 47' 8''$ N, $0^{\circ} 50' 43''$ E). It was reclaimed in the 18th Century and
107 used as grazing marsh, then was levelled and converted for arable use between 1943 and
108 1970 (Essex Wildlife Trust 2003). In 1996, tidal flow was restored to the area shown in Fig 1
109 via a culvert (Nottage and Robertson 2005). As part of the 1996 scheme, a network of
110 artificial creeks was excavated in an attempt to recreate natural marsh drainage conditions
111 and sediment deposition (Dixon et al. 1998). The amount of seawater reaching the site was
112 lower than expected (ABP 1998) and vegetation development was very poor (Diack 1998),
113 so in 2002, a larger area (49 ha) of intertidal habitat was created by breaching the sea wall in
114 five places, including sections lying outside of the area shown in Fig. 1. The creeks excavated
115 in 1996 were 1-3 m wide and deep, with a total length of 2.2 km, and are clearly visible on
116 Fig. 1. Spoil from these excavations was left in situ in a pile adjacent to the excavated creeks
117 (an example is visible at right hand side of Fig. 2b). The tops of these piles were just above
118 the level of the highest tides. An area of natural reference marsh (approximately 70 ha)
119 occurs adjacent to the managed realignment site. Extensive vegetation colonisation had
120 occurred on the managed realignment two years after the sea wall was breached and
121 sediments then were relatively well oxygenated (Mossman 2007), and five years after
122 breaching the vegetation was described as "similar to the adjacent ancient saltmarsh" based
123 on data from a single transect (Hughes et al. 2009).

124

125 We studied three areas of marsh. These were an area where no creeks had been excavated
126 ('non engineered site', area A in Figs. 1 and 2), one that included an excavated creek
127 ('engineered site', area B in Figs. 1 and 2) and one on an area of natural reference marsh
128 ('reference site', area C in Figs. 1 and 2). On both parts of the restored site, the landward

129 part of the marsh consists of a relatively well vegetated and rather flat surface. This is visible
130 as darker areas on Fig. 1 and these can also be seen in Fig. 2a and b. It is located at
131 approximately 2.25 to 2.5m ODN (see also Fig. 6), an elevation at which species-rich
132 marshes dominated by *Puccinellia maritima* and *Atriplex portulacoides* normally occur in
133 this region (Mossman 2007). Seawards of this is an area of mudflat with pioneer vegetation,
134 visible as the lighter coloured parts of the site in Fig. 1. As we wanted to focus on factors
135 influencing the vegetation on the upper parts of restored marshes, the areas sampled were
136 located approximately mid-way between the landward sea wall and the areas of mudflat
137 and pioneer communities. To allow comparison with the restored sites, the reference site
138 was located in an area where the main marsh surface was at an elevation of around 2.25m
139 ODN, but as the reference marsh is quite heavily dissected by creeks, some sampling points
140 are located lower in the tidal frame (c.f. Fig. 6).

141

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

155

156 In August 2009, the elevation, vegetation and soil characteristics were assessed at each of
157 the sampling points. The vegetation was assessed by estimating the percentage cover in a
158 0.25 m² quadrat (centralised over each sampling point) of all plant species and bare ground,
159 to the nearest 5%, assigning an abundance of 1% to species present at < 5% area. Plant

160 species observed within each sampling grid but that did not fall into any quadrats were also
161 noted. Plant species nomenclature followed Stace (2001).

162

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

187

188 Initial data analysis and fitting of LOESS regressions (locally weighted non-linear regressions,
189 using an Epanechnikov kernel) to relationships between elevation and individual plant
190 abundances and environmental variables, were carried out using SPSS version 16 (SPSS Inc,
191 Chicago, Ill.). In order to examine, and test the significance of, differences in vegetation

192 communities between sites, multidimensional scaling (MDS) and one-way analysis of
193 similarity (ANOSIM, which tests whether similarities between groups are smaller than
194 expected by chance using permutation tests) were carried out on a matrix of Bray-Curtis
195 similarities of vegetation data after square root transforming abundances. SIMPER
196 (Similarity Percentage) analysis is a method for assessing which taxa are primarily
197 responsible for an observed difference between groups of samples and was used to identify
198 the species responsible for the differences in vegetation communities between the three
199 sites. MDS, ANOSIM and SIMPER were carried in PRIMER v6 (Plymouth Routines In
200 Multivariate Ecological Research, Primer-E, Plymouth; Clarke and Gorley 2006).
201 Heterogeneity of site elevations was quantified using R (R Development Core Team 2008),
202 writing our own code to calculate horizontal distances and elevation differences between all
203 pairs of sampling points at each site and using the *stats* package to fit LOESS regressions.
204

205 **Results**

206 **Vegetation data**

207 A total of 16 plant species were found, although two of these, *Picris echioides* and an
208 unidentified moss, were essentially non-maritime species that occurred only on the highest
209 part of the engineered site, i.e. on creek berms above the high-tide level. Multidimensional
210 scaling showed that the plant communities on both the engineered and non-engineered
211 parts of the created marsh remain markedly different from those on the reference marsh
212 (Fig. 3). None of the samples from either part of the created marsh plot in the area at the
213 top and centre of Fig. 3 where the bulk of the reference samples are located, although the
214 engineered part of the site showed greater similarity to the reference marsh than the non-
215 engineered part. In addition, a small number of reference samples, mostly from the low
216 marsh, *are* similar to samples from the created marsh. ANOSIM confirms that pairwise
217 differences between the three sites are significant ($P < 0.001$ in all cases). There are a
218 number of species contributing to these differences, and the mean abundances of the
219 commonest species are presented in Fig. 4. The abundance of individual species changed
220 with elevation, as well as showing differences between sites, so we have used LOESS
221 regressions to visualise these patterns in more detail and demonstrate that the differences
222 between the sites are not simply due to differences in elevation (Fig. 5). On the reference

223 marsh there was 100% vegetation cover in most quadrats at elevations greater than 2.1 m
224 ODN, but vegetation cover remained incomplete at elevations above this on both parts of
225 the created marsh, with about 30% bare ground at 2.25 m ODN on the engineered site, and
226 about 60% on the non-engineered part (Fig. 5a). Two species, *Puccinellia maritima* and
227 *Atriplex portulacoides* were common between 2 m and 2.6 m ODN on the reference marsh,
228 with *Puccinellia* dominating between 2.3 and 2.5 m ODN, and abundance of *Atriplex* peaking
229 at around 2.2m ODN. At elevations below 2 m ODN, *Salicornia europaea* was the
230 commonest species, although *Atriplex* and *Puccinellia* were present in a small number of
231 quadrats. *Atriplex* and *Puccinellia* were present at much lower abundance, and occurred at
232 higher elevations, on the engineered site and were rather rare on the non-engineered site
233 (Fig. 5d, e). SIMPER analysis indicates that the abundance of these two species and the
234 occurrence of bare ground made the largest contribution to the differences between sites.
235 The annuals *Salicornia europaea* and *Suaeda maritima* were more common on the
236 realignment site, with the latter particularly abundant on the non-engineered part, peaking
237 at about 80% cover at 2.5 m ODN (Fig. 5b, c).

238

239 Other species were present at much lower abundance, so data are not presented
240 graphically. The grass *Elytrigia atherica* occurred at high abundance (> 60% cover) in some
241 quadrats, almost entirely those between 2.5 and 2.75 m ODN, but was absent from the
242 majority of quadrats. It occurred at a cover of > 50% in more quadrats (15) on the non-
243 engineered site than on the engineered or reference sites (5 and 8, respectively), which was
244 significantly different from a uniform distribution ($\chi^2 = 6.2$, d.f. = 2, $P = 0.04$). However, 28%
245 of sampling points on the non-engineered site were between 2.5 and 2.75 m ODN, as
246 compared with only 6 and 7% for the engineered and reference sites, and if we calculate
247 expected values based on these proportions, the difference was not significant ($\chi^2 = 3.6$, d.f.
248 = 2, $P = 0.17$). Four characteristic mid to upper marsh species (*Sarcocornia perennis*;
249 *Triglochin maritima*; *Plantago maritima* and *Limonium vulgare*) were present on the
250 reference marsh but absent from both parts of the realignment site, and *Aster tripolium* was
251 more than ten times more abundant on the reference site. *Agrostis stolonifera* and *Spartina*
252 *anglica* occurred only on the engineered site. One-way Anova showed that, apart from
253 *Plantago* ($F_{2,287} = 2.8$, N.S.) and *E. atherica* ($F_{2,287} = 3.5$, $P = 0.03$), all the above differences in
254 mean abundance between sites were significant at $P < 0.001$.

255

256 **Environmental characteristics**

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

267

268 The sites differed in mean elevation, redox, shear strength, water and organic content
269 (Table 1; $P < 0.001$ in all cases, Anova). The non-engineered parts of the realignment site
270 had the *highest* mean elevation, as the absence of any creeks results in no sampling points
271 on this site being below 1.94 m ODN (Fig. 6). The mean redox potentials at all elevations in
272 all sites were above 0 mV (Fig. 7a). The non-engineered site also had a markedly higher
273 mean redox potential, because this declined steadily with elevation and almost all sampling
274 points below 2 m ODN were on the other two sites. Nevertheless, the redox potential on the
275 non-engineered site was somewhat higher than at similar elevations on the other two sites
276 (Fig. 7a). Sediment shear strength was high throughout the non-engineered site, with most
277 quadrats having shear strengths greater than 80 kPa (Fig. 7b). By contrast, shear strengths
278 on the reference marsh were lower than 40 kPa in sampling points below 2.3 m ODN. They
279 increased on the higher marsh, but very few sampling points exceeded 80 kPa. On the
280 engineered site, shear strengths were higher than those on the non-engineered site above
281 2.3 m ODN, declining to very low values at the lowest elevations. Sediment water content at
282 2 m ODN on all sites was in the region of 50%, but declined steadily with increasing
283 elevation on both the engineered and non-engineered realignment sites (Fig. 7d). By
284 contrast, water content increased on the upper parts of the reference marsh, where
285 sediment organic content was also markedly higher than on the realignment sites (Fig. 7c,
286 d).

287

288 **Environmental heterogeneity**

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

297

298 **Discussion**

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

312

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

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

330

331 Perhaps there has simply not been sufficient time for the vegetation to fully develop. It is
332 true that succession can be relatively slow, but on natural marshes the rate of succession
333 from mudflat to pioneer communities to the colonisation by later successional mid and
334 upper marsh species is usually constrained by rates of sediment deposition (Boorman 2003).
335 In locations where natural marsh development has been initiated by a change in the local
336 hydrodynamic or sedimentary environment rather than by gradual sediment deposition,
337 succession takes place rapidly. Species-rich vegetation, including high marsh perennials such
338 as *Armeria maritima*; *Juncus maritimus* and *Triglochin maritima*, developed in 17 years on
339 an area of previously bare sand in North Wales (Packham and Liddle 1970). In the
340 Netherlands, Olf et al. (1997) found that ten years after initial colonisation, vegetation was
341 dominated by *Spergularia maritima*, *Suaeda maritima*, *Limonium vulgare* and *Puccinellia*
342 *maritima*. Natural recolonisation is also rapid when turf is stripped from existing salt
343 marshes; *Puccinellia maritima* recolonises within a year, even at elevations above its normal
344 range, and re-establishment of full vegetation cover is complete after 3-5 years (Cadwalladr
345 and Morley 1974; Gray and Scott 1977a, b). In contrast to these examples, restored sites
346 flooded accidentally more than a century ago still show marked differences in vegetation
347 communities (Mossman et al. 2012a). Are the “missing” species those that disperse only
348 slowly to created marshes? Some perennial species are rare on created marshes of all ages,
349 even many decades after restoration (Mossman et al. 2012a). Colonisation by some

350 perennials, such as *Limonium vulgare* , *Plantago maritima* and *Triglochin maritima*, may be
351 slowed by low seed viability and long reproductive cycles (Boorman 1967; Hutchings &
352 Russell 1989; Davy & Bishop 1991). But the two species that make the greatest contribution
353 to the difference between the created and reference marshes at Abbotts Hall are *Atriplex*
354 *portulacoides* and *Puccinellia maritima*. Both of these have rapidly colonised other managed
355 realignment sites and are often dominant on older accidentally restored sites (Mossman
356 2007).

357

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

369

370 The reference marsh shows considerable topographic heterogeneity, even on horizontal
371 scales of a few metres. This is typical of natural marshes, and develops, at least in part,
372 during the early stages of plant colonisation of mudflats and results in heterogeneity of
373 sediment redox (Stribling et al. 2006, 2007). This in turn produces heterogeneity of plant
374 communities (Morzaria-Luna et al. 2004; Varty and Zedler 2008). By contrast, the restored
375 marsh surface lacks topographic heterogeneity, as a result of levelling for arable farming.
376 This is true of many managed realignment sites, and means that their visual appearance is
377 rather different to that of natural marshes, in ways that are obvious even to people without
378 detailed knowledge of their ecology (c.f. Fig. 2). As well as being responsible for a lack of
379 ecological heterogeneity, the lack of topographic variation may help to create conditions
380 that are hostile to plant colonisation. When the restored marsh has been flooded by a high
381 spring tide, water will drain away much more slowly than from a marsh with a more

382 heterogeneous surface and natural, or artificial, creeks. Many managed realignment sites
383 that were previously used for arable farming have substantial areas of shallow standing
384 water after high spring tides (pers. obs.). If hot, dry weather follows a period of spring tides,
385 evaporation of this standing water will produce high surface salinities, inhibiting
386 germination and growth, even of halophytes. A similar process was responsible for the slow
387 recolonisation of large open areas created by ice scour observed on New England marshes
388 (Bertness et al. 1993; Shumway and Bertness 1992, 1994), where initial plant colonisation
389 provides shading, reduces evaporation and salt stress and facilitates the colonisation of
390 other plants (Bertness 1991).

391

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

404

405 It is likely that the high sediment shear strength results from deflocculation of clay minerals
406 in the low salinity conditions during aerial exposure, leading to loss of soil structure. This in
407 turn results in over-compaction when flooded by sea water, a process that has led to the
408 widespread development of over-consolidated sediments in the Holocene estuarine
409 deposits of South-East England (Crooks 1998; Crooks et al. 2002). These consolidated layers
410 can impede drainage in freshly deposited sediment and contribute to the development of
411 anoxic conditions at low lying sites (Crooks et al. 2002), and over-compaction of sediments
412 also seems to have hindered the ecological development of the Seal Sands managed
413 realignment (Evans et al. 1998). Once a marsh has been flooded, large scale sediment

414 treatment to break up over-consolidated sediment is extremely challenging. As noted
415 above, excavation of creeks seems to be of limited value on high marsh sites, and it may be
416 necessary to carry out more aggressive surface treatment to prevent over-consolidated
417 layers forming. This might involve deep ploughing or even the mixing of soil and seawater
418 into a slurry before flooding. Experimental evaluation of the options is required, but if over-
419 consolidated sediments can be avoided, this will result in a created marsh surface that is
420 more likely to undergo small scale erosion and deposition to generate topographic
421 heterogeneity. This will certainly create a marsh with a more “natural” appearance and one
422 that may also be less hostile for plant colonisation and provide microsites suitable for the
423 growth of different species (Varty and Zedler 2008).

424

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428

429 **References**

430 Able, K.W., S.M. Hagan, and S.A. Brown. 2003. Mechanisms of marsh habitat alteration due
431 to *Phragmites*: Response of young-of-the-year Mummichog (*Fundulus heteroclitus*) to
432 treatment for *Phragmites* removal. *Estuaries* 26: 484-494.

433 ABP. 1998. Review of coastal habitat creation, restoration and recharge schemes, Report R
434 909, ABP Research, Southampton, UK.

435 Atkinson, P.W., S. Crooks, A. Grant, and M. Rehfisch. 2001. The Success of Creation and
436 Restoration Schemes in Producing Intertidal Habitat Suitable for Waterbirds. English Nature
437 Research Report No. 425.

438 Bertness, M.D. 1991. Interspecific interactions among high marsh perennials in a New
439 England salt marsh. *Ecology* 72:125-137.

440 Bertness, M.D., L. Gough, and S.W. Shumway. 1992. Salt tolerances and the distribution of
441 fugitive salt-marsh plants. *Ecology* 73:1842-1851.

442 Blackwell, M.S.A., D.V. Hogan, and E. Maltby. 2004. The short-term impact of managed
443 realignment on soil environmental variables and hydrology. *Estuarine, Coastal and Shelf*
444 *Science* 59:687-701.

445 Boorman, L. 2003. Saltmarsh review. An overview of coastal saltmarshes, their dynamic and
446 sensitivity characteristics for conservation and management. JNCC Report, No. 334, JNCC,
447 Peterborough, UK

448 Burd, F. 1995. Managed retreat: a practical guide. English Nature, Peterborough, UK.

449 Cadwalladr, D.A and J.V. Morley . 1974. Further experiments on management of saltings
450 pasture for wigeon (*Anas penelope* L.) conservation at Bridgwater Bay national nature
451 reserve, Somerset. *Journal of Applied Ecology* 11: 461-466.

452 Clarke, K.R., and R.N. Gorley. 2006. PRIMER v6: User Manual/Tutorial, PRIMER-E, Plymouth,
453 UK.

454 Crooks, S. 1998. A mechanism for the formation of overconsolidated horizons within
455 estuarine floodplain alluvium: implications for the interpretation of Holocene sea-level
456 curves. In: *Floodplains: interdisciplinary approaches: 1998*; Norwich: Special Publication-
457 Geological Society of London, Bath; 1998: 197-216.

458 Crooks, S., J. Schutten, G.D. Sheern, K. Pye, and A.J. Davy. 2002. Drainage and elevation as
459 factors in the restoration of salt marsh in Britain. *Restoration Ecology* 10:591–602.

460 Davy, A.J., M.J.H. Brown, H.L. Mossman, and A. Grant. 2011. Colonisation of a newly
461 developing salt marsh: disentangling independent effects of elevation and redox potential
462 on halophytes. *Journal of Ecology* 99:1350-1357.

463 Diack, I. 1998. Botanical monitoring of the saltmarsh option of the habitat scheme 1995-
464 1997. ADAS, Oxford, UK.

465 Dixon, A.M., D.J. Leggett, and R.C. Weight. 1998. Habitat creation opportunities for
466 landward coastal re-alignment: Essex case studies. *Water and Environment Journal* 12:107-
467 112.

468 Essex Wildlife Trust. 2003. Abbots Hall Farm Fact Sheet 1: History, Essex Wildlife Trust,
469 Colchester, UK.

470 <http://www.essexwt.org.uk/uploads/file/AHF%20Fact%20Sheets/FS1%20AHF%20History.pdf>
471 [f](#). Accessed 11/10/09.

472 European Commission. 2007. Guidance document on Article 6(4) of the ‘Habitats Directive’
473 92/43/EEC. Luxembourg: Office for Official Publications of the European Communities. 30pp

474 Evans P.R., R.M. Ward, M. Bone, and M. Leakey 1998. Creation of temperate-climate
475 intertidal mudflats: Factors affecting colonization and use by benthic invertebrates and their
476 bird predators. *Marine Pollution Bulletin* 37:535-545.

477 French, P.W. 2006. Managed realignment – The developing story of a comparatively new
478 approach to soft engineering. *Estuarine, Coastal and Shelf Science* 67:409-423.

479 Garbutt, A., and M. Wolters. 2008. The natural regeneration of salt marsh on formerly
480 reclaimed land. *Applied Vegetation Science* 11:335-344.

481 Garbutt, R.A., C.J. Reading, M. Wolters, A.J. Gray, and P. Rothery. 2006. Monitoring the
482 development of intertidal habitats on former agricultural land after the managed
483 realignment of coastal defences at Tollesbury, Essex, UK. *Marine Pollution Bulletin* 53:155–
484 164.

485 Gray, A.J. and Scott, R. 1977. Biological flora of British Isles - *Puccinellia maritima* (Huds)
486 Parl. *Journal of Ecology* 65: 699-716.

487 Gray, A.J. and Scott, R. 1977. Ecology of Morecambe Bay .7. Distribution of *Puccinellia*
488 *maritima*, *Festuca rubra* and *Agrostis stolonifera* in salt marshes. *Journal of Applied Ecology* 14:
489 229-241.

490 Havens, K.J., Varnell, L.M. & Watts, B.D. (2002) Maturation of a constructed tidal marsh relative to
491 two natural reference tidal marshes over 12 years. *Ecological Engineering*, 18, 305–315. Hughes, R.G.,
492 P.W. Fletcher, and M.J. Hardy. 2009. Successional development of saltmarsh in two managed
493 realignment areas in SE England, and prospects for saltmarsh restoration. *Marine Ecology Progress*
494 *Series* 384:13-22.

495 Larkin, D.J., S.P. Madon, J.M. West, and J.B. Zedler. 2008. Topographic heterogeneity influences fish
496 use of an experimentally restored tidal marsh. *Ecological Applications* 18: 483-496.

497 Mazik, K., W. Musk, O. Dawes, K. Solyanko, S. Brown, L. Mander, and M. Elliott. 2010. Managed
498 realignment as compensation for the loss of intertidal mudflat: A short term solution to a
499 long term problem? *Estuarine Coastal and Shelf Science* 90:11-20.

500 Miller, R.H. 1982. Methods of soil analysis. Part 2: Chemical and microbiological properties
501 (2nd Edition). American Society of Agronomy, Soil Science of America.

502 Morzaria-Luna H., J.C. Callaway, G. Sullivan, and J.B. Zedler. 2004. Relationship between
503 topographic heterogeneity and vegetation patterns in a Californian salt marsh. *Journal of*
504 *Vegetation Science* 15:523-530.

505 Mossman, H.L. 2007. Development of saltmarsh vegetation in response to coastal
506 realignment, PhD thesis, University of East Anglia, Norwich, UK.

507 Mossman, H.L., A.J. Davy, and A. Grant. 2012a. Does managed coastal realignment create
508 salt marshes with 'equivalent biological characteristics' to natural reference sites? *Journal of*
509 *Applied Ecology* 49: 1446-1456.

510 Mossman, H.L., M.J.H. Brown, A.J. Davy, and A. Grant. 2012b. Constraints on salt marsh
511 development following managed coastal realignment: dispersal limitation or environmental
512 tolerance? *Restoration Ecology* 20:65-75.

513 Mossman, H.L., A.J. Davy, and A. Grant. 2012c. Quantifying local variation in tidal regime
514 using depth logging fish tags. *Estuarine Coastal and Shelf Science* 96:122-128.

515 Nottage, A., and P. Robertson. 2005. *The Saltmarsh Creation Handbook: a project manager's*
516 *guide to the creation of saltmarsh and intertidal mudflat*. RSPB, Sandy, UK.

517 O'Brien, E.L., and J.B. Zedler. 2006. Accelerating the restoration of vegetation in a southern
518 California salt marsh. *Wetlands Ecology and Management* 14: 269-286.

519 Olf, H., J. De Leeuw, J.P. Bakker, R.J. Platerink, H.J. Van Wijnen, and W. De Munck. 1997.
520 Vegetation succession and herbivory in a salt marsh: changes induced by sea level rise and
521 silt deposition along an elevational gradient. *Journal of Ecology* 85: 799-814.

522 Packham, J.R., and M.J. Liddle, M.J. 1970. The Cefni Salt Marsh Anglesey and its recent
523 development. *Field Studies* 3: 331-356.

524 Parker, R., S. Bolam, J. Foden, D. Morris, S. Brown, T. Cheshier, C. Fletcher, and I. Möller.
525 2006. Suitability Criteria for Habitat Creation – Report I: Reviews of present practices and
526 scientific literature relevant to site selection criteria. *Defra/Environment Agency R&D*
527 *Technical Report FD1917TR1*.

528 Pethick, J. 2002. Estuarine and tidal wetland restoration in the United Kingdom: policy
529 versus practice. *Restoration Ecology* 10:431-437.

530 R Development Core Team. 2008. R: A language and environment for statistical computing.
531 R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, [http://www.R-](http://www.R-project.org/)
532 [project.org/](http://www.R-project.org/).

533 Shumway, S. W., and M. D. Bertness. 1992. Salt stress limitation of seedling recruitment in a
534 salt-marsh plant community. *Oecologia* 92: 490-497.

535 Shumway, S. W., and M. D. Bertness. 1994. Patch size effects on marsh plant secondary
536 succession mechanisms. *Ecology* 75:564-568.

537 Stace, C.A. 2010. *New Flora of the British Isles*. 3rd Edition. Cambridge University Press,
538 Cambridge, UK.

539 Stribling, J.M., J.C. Cornwell and O.A. Glahn. 2007. Microtopography in tidal marshes:
540 Ecosystem engineering by vegetation? *Estuaries and Coasts* 30: 1007-1015.

541 Stribling, J.M., O.A. Glahn, X.M. Chen, and J.C. Cornwell. 2006. Microtopographic variability
542 in plant distribution and biogeochemistry in a brackish-marsh system. *Marine Ecology*
543 *Progress Series* 320: 121-129.

544 Thom, R.M., R. Zeigler, and A.B. Borde. 2002. Floristic development patterns in a restored
545 Elk River Estuarine Marsh, Grays Harbor, Washington. *Restoration Ecology* 10:487–496.

546 Varty, A.K., and J.B. Zedler. 2008. How waterlogged microsites help an annual plant persist
547 among salt marsh perennials. *Estuaries and Coasts* 31:300–312.

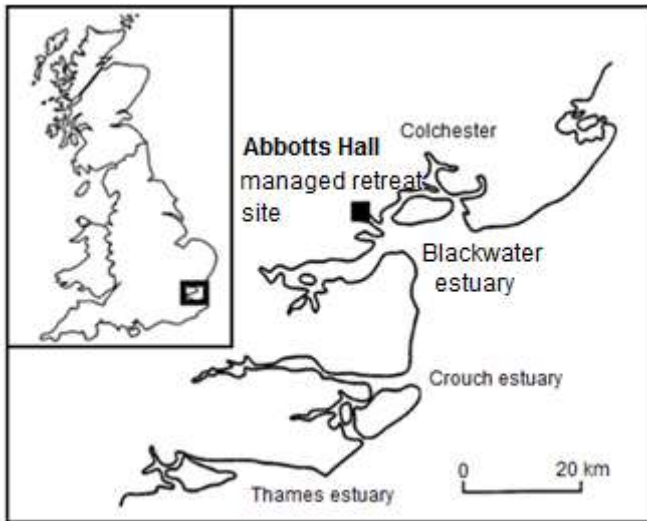
548 Wallace, K.J., J.C. Callaway and J.B. Zedler. 2005. Evolution of tidal creek networks in a high
549 sedimentation environment: A 5-year experiment at Tijuana Estuary, California. *Estuaries*
550 28:795-811.

551 Wolters, M., A. Garbutt, and J.P. Bakker. 2005. Salt-marsh restoration: evaluating the
552 success of de-embankments in north-west Europe. *Biological Conservation* 123:249–268.

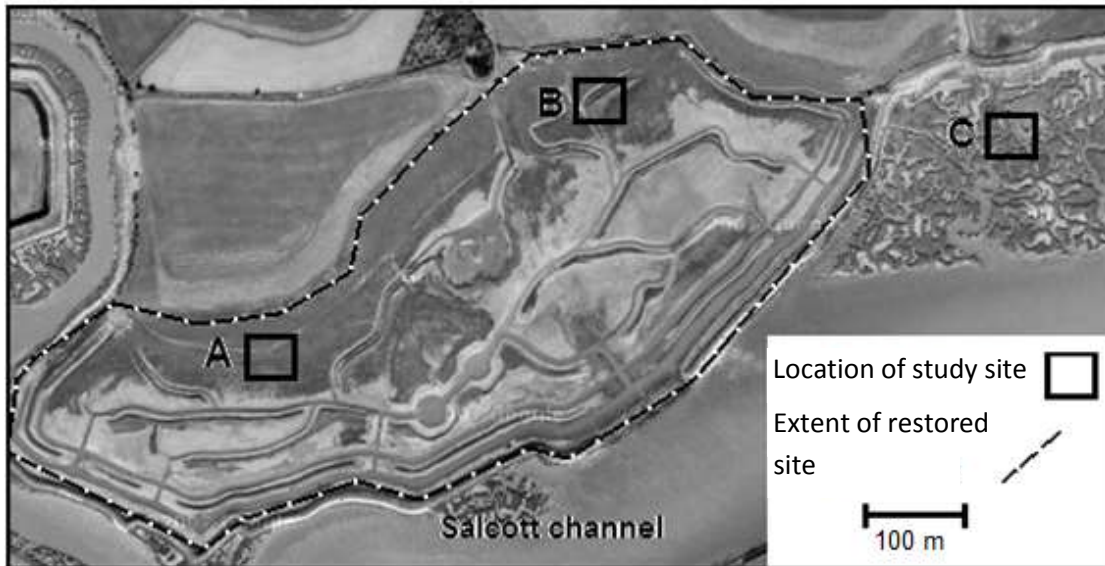
553 Zedler, J.B. (Ed). 2001. *Handbook for Restoring Tidal Wetlands*. CRC Press, Boca Raton,
554 Florida.

555 Zedler, J.B. 2004. Compensating for wetland losses in the United States. *Ibis* 146:92-100.
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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)



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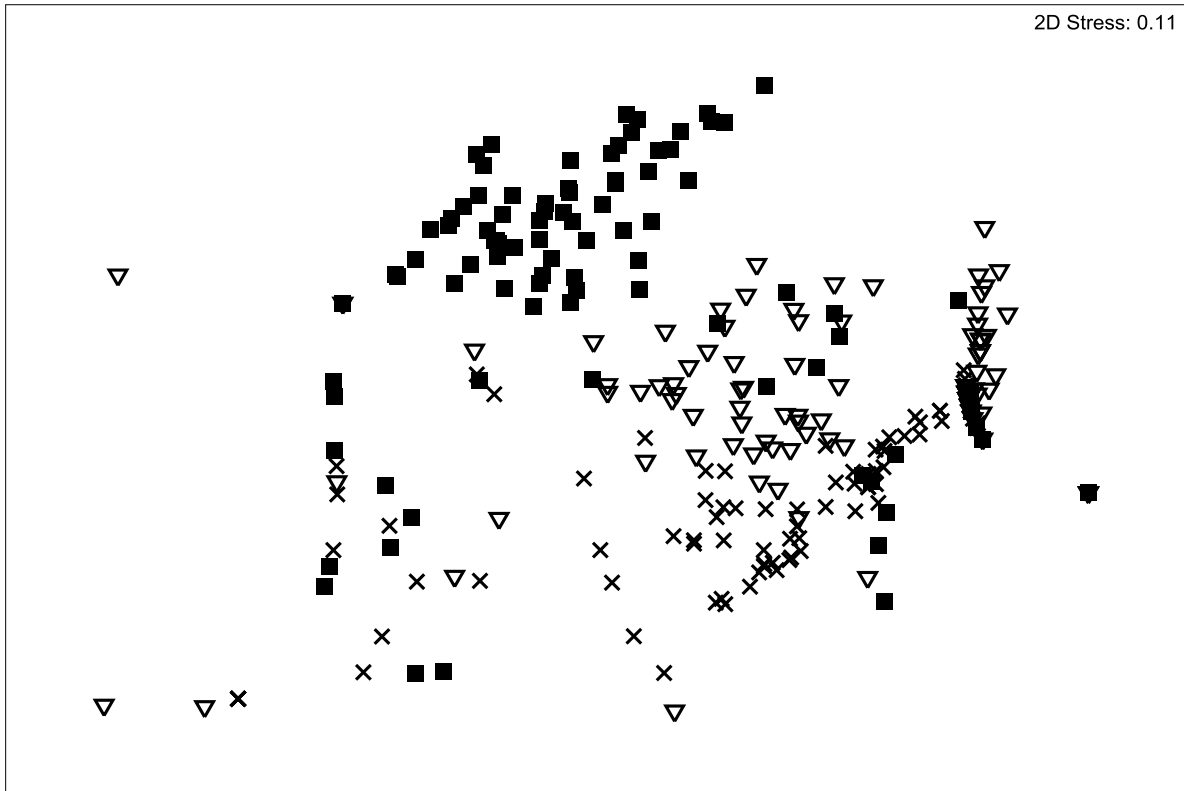


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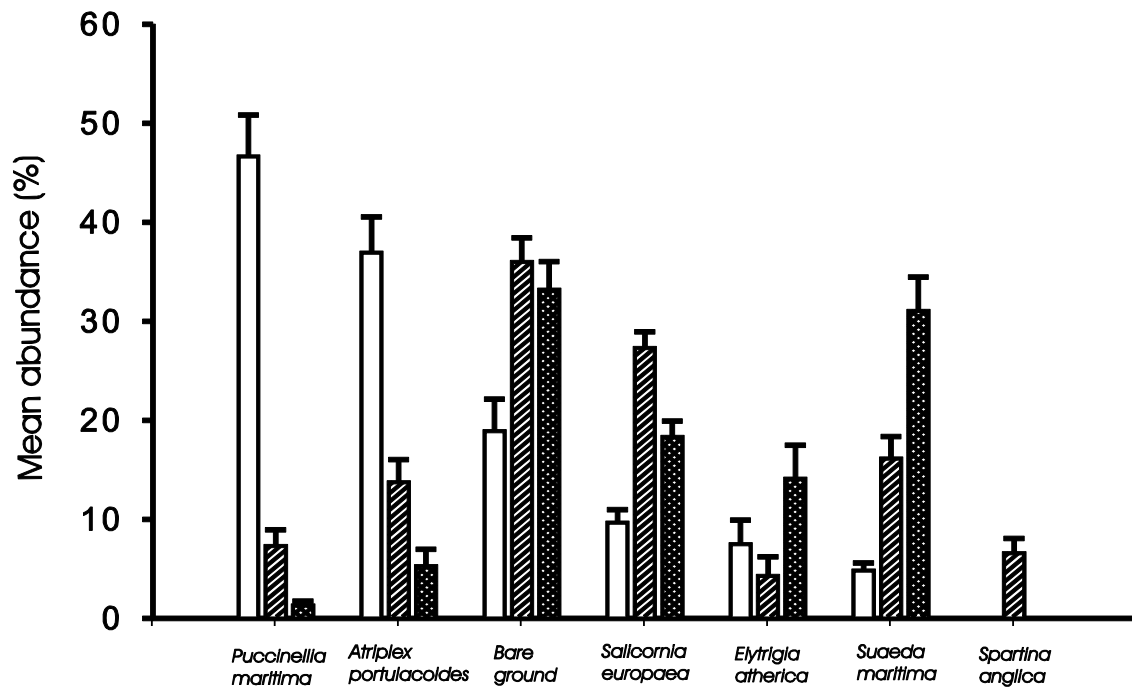
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566 Fig. 2. Photographs showing characteristics of each site. Taken 22/08/09. A) non-engineered
567 part of the restored salt marsh; b) engineered part of the restored salt marsh, with
568 embankment of spoil from excavating creek to right; c) reference site
569



570

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

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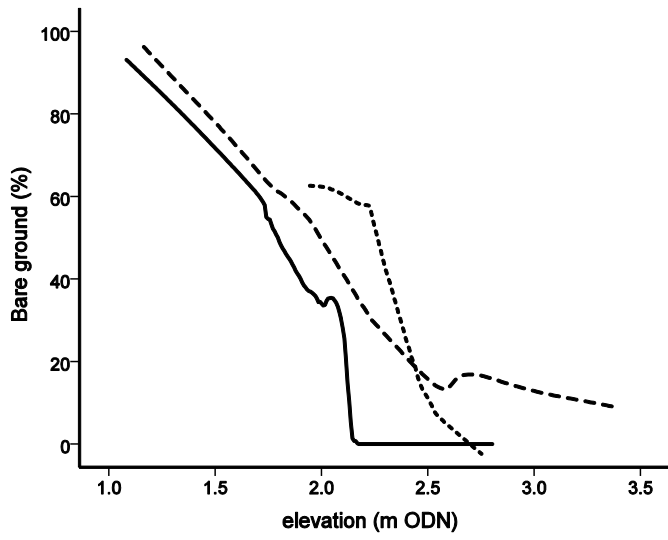
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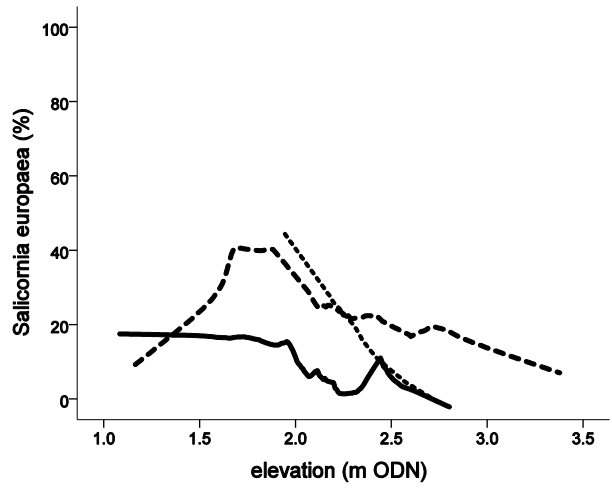
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Fig. 4. Mean abundance of six commonest species and bare ground on the reference salt marsh (open bars) and engineered (diagonal) and non-engineered (cross-hatched) parts of the restored salt marsh. All species with mean abundance > 5% in at least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

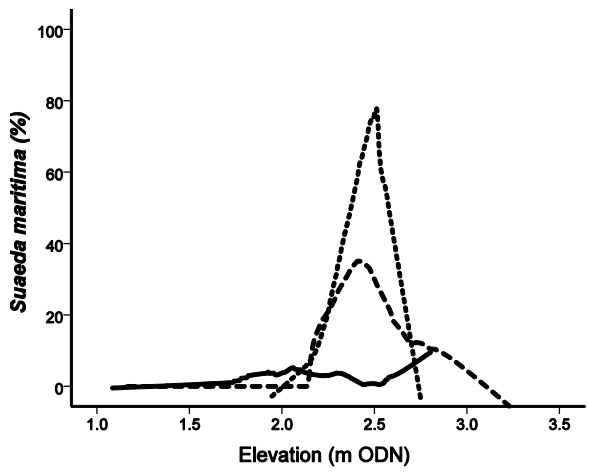
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583 b

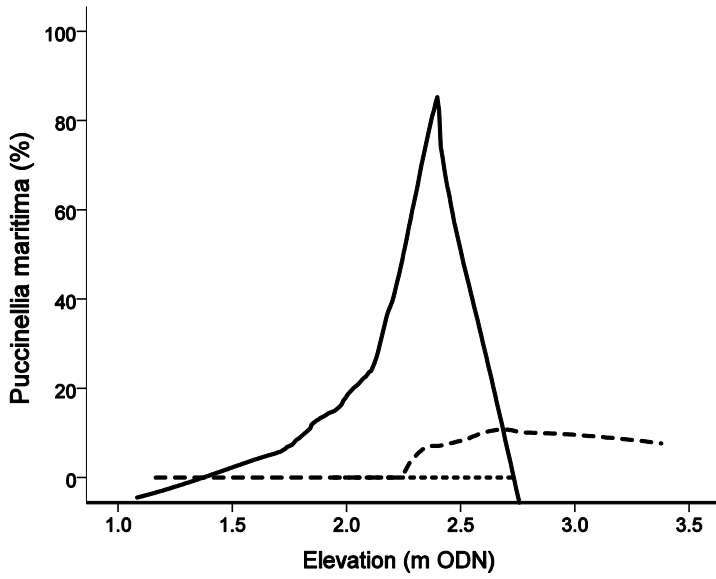


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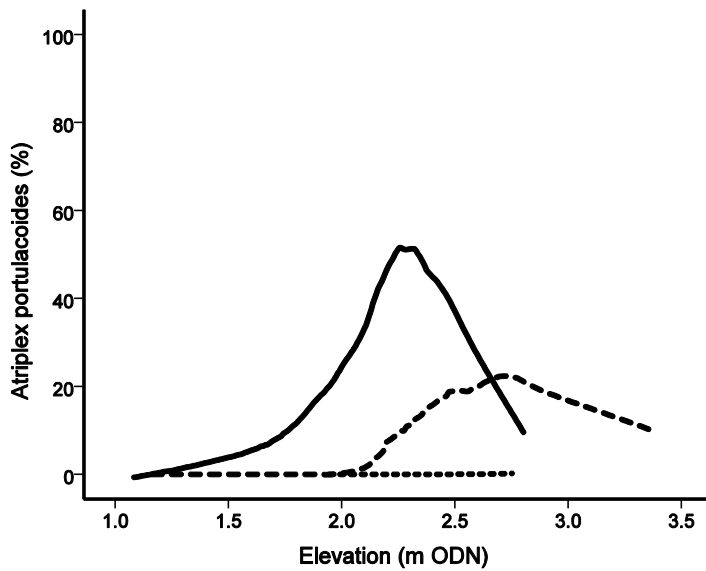


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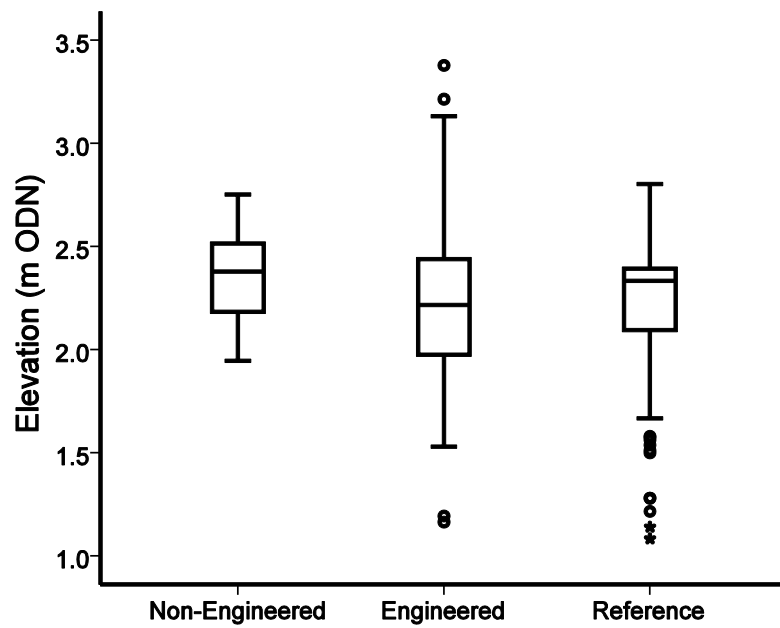


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Fig. 5. LOESS regressions showing relationship between percentage cover of individual plant species and elevation at Abbots Hall restored and reference salt marshes. a) bare ground , b) *Salicornia europaea* ; c) *Suaeda maritima* d) *Puccinellia maritima*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



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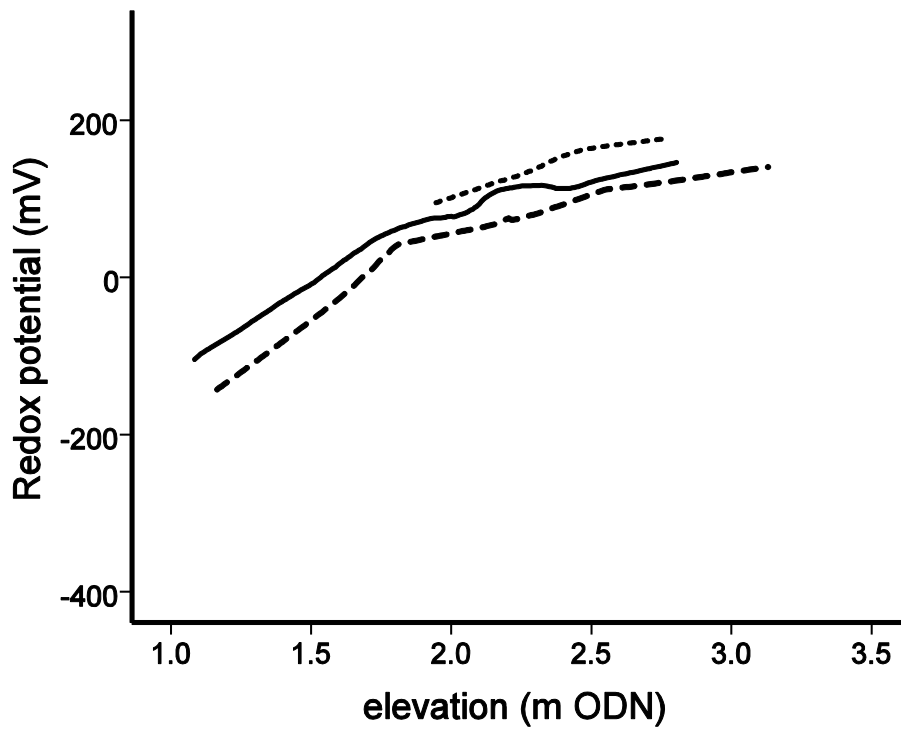
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602 Fig. 6. Range of elevations occurring at the engineered and non-engineered part of the

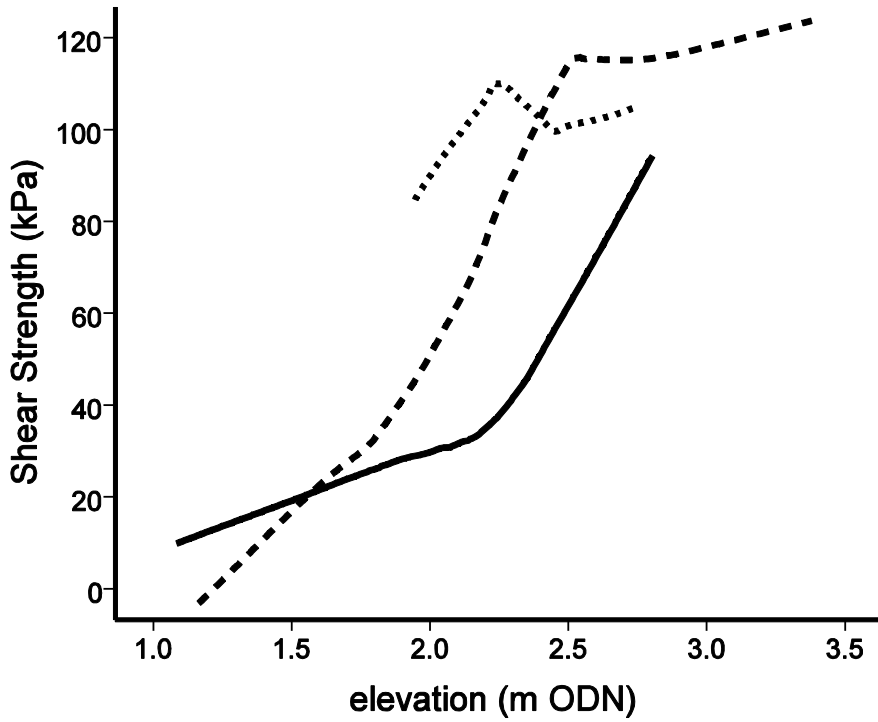
603 Abbotts Hall restored salt marsh and at the adjacent reference marsh

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605 a

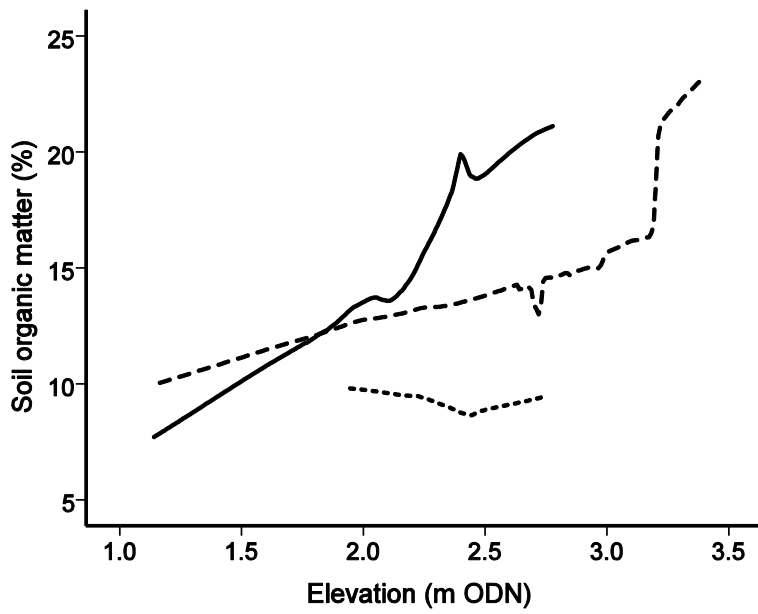


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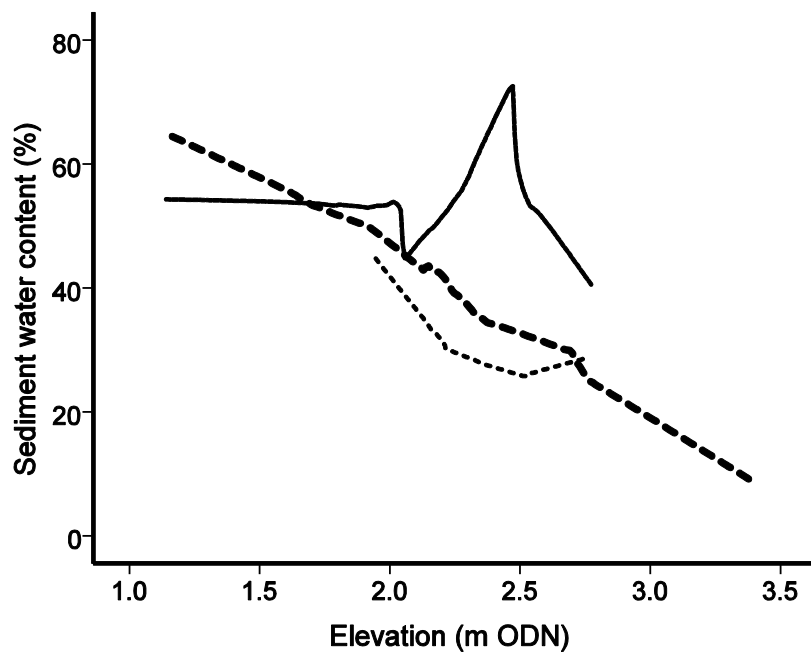


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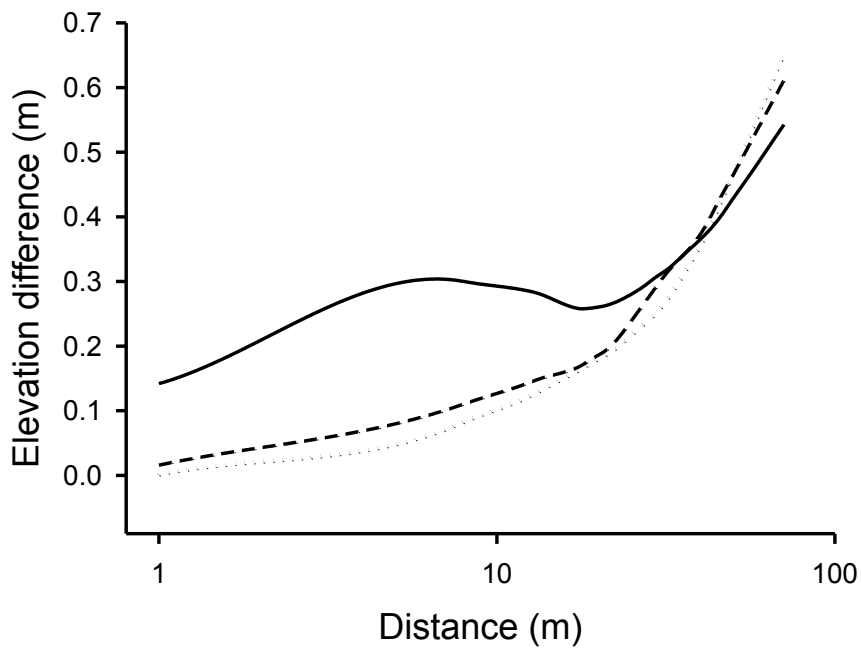


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Fig. 7. LOESS regressions showing relationships with elevation of a) redox; b) shear strength; c) soil organic matter and d) water content. Details as Fig. 5.



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

626