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Factors influencing the initial establishment of salt marsh vegetation on engineered sea wall terraces in south east England

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ABSTRACT

Sea walls provide vital flood protection for lowland coastal property. We investigated the integrity of a cost-effective method of repairing sea defences, which has potential to create habitat for coastal and salt marsh flora. Experimental stone-gabion and clay-filled terraces were installed as a soft engineered approach to repair damaged sea walls in estuarine embayments in south east England. Changes in the surface heights of sediment and vascular plant colonisation were monitored over a 22 month period. Seven of the 12 terraces were colonised, by 12 species of plant, reaching a maximum of 85% cover. The main drivers of plant colonisation were sediment stability, elevation, exposure and sediment shear strength. Terraces with least change in the surface height of sediments were favourable for plant colonisation. Ordination (Canonical Correspondence Analysis) showed 72% variation in plant distribution explained by elevation (37%), exposure (30%), terrace length and sediment shear strength (5%). Elevation was the most influential variable; recruitment increased as terrace height approached the height of existing marsh ($r^2 = 0.43$). This cost-effective approach has the potential to provide protection to sea walls and create additional habitat for wildlife. Key considerations for the improvement of terrace design and construction are discussed.

1. Introduction

Salt marshes are important coastal habitats (Ford et al., 2013), supporting a range of terrestrial and marine species (acting as nursery grounds for various fish species, Green et al., 2009) and feeding, roosting and nesting sites for various species of shorebirds (Norris, 2000). Salt marshes play a valuable role in attenuating the energy of waves which can otherwise undermine the integrity of sea wall embankments built as flood protection structures (Möller and Spencer, 2002; Möller et al., 2014). Wave attenuation is a complex process affected by the reflection of wave energy on marsh edges (cliff face), wave shoaling and vegetation (Möller and Spencer, 2002; Möller, 2006), and can be as high as 60% even during storm surge conditions (Möller et al., 2014).

In the U.K. there is concern over the loss of salt marsh habitat, with two thirds of salt marsh loss having occurred in the lowland regions of south east England (Dixon et al., 1998; Paramor and Hughes, 2004) at a rate of 40 ha yr^{-1} (Thomson et al., 2011). Coastal squeeze on salt marshes (i.e. the reduction in available habitat space caused by the combined effects of rising sea levels, isostatic land mass adjustments and static sea defences) accelerates the rate of salt marsh loss at the very locations where their presence may be most beneficial (Burd, 1992; Thomson et al., 2011). In eastern England, relative sea level is currently predicted to rise over the next century at an average rate of 5 mm yr^{-1} for a high emissions climate scenario (Thomson et al., 2011). The North Sea coastline of south eastern England is highly modified with flood defence seawalls, many of which were repaired and enhanced after the major storm surge and flooding in 1953. The 1953 flood was the most extreme natural disaster to befall Britain in the 20th century (Baxter, 2005) causing the loss of over 300 lives. In response extensive coastal defences were created, which included the raising of approximately 2,100 km of earthen sea wall embankment around the coasts of England and Wales (Gardiner et al., 2015). These conventional defences will eventually need to be replaced because they are approaching the end of their design life and are challenged by rising maintenance costs (Temmerman et al., 2013).

In addition to the protection of people and property, coastal defence strategies must take into consideration the preservation of

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nature and conform to regulations such as the Conservation (Natural Habitats) Regulations 1994 (Lee, 2001). A desirable solution would be a cost-effective ecosystem-based flood defence strategy that addresses regulatory requirements, while reducing the capital investment required to build and repair conventional hard engineering (Temmerman et al., 2013). The expense of repairing sea wall defences can be significantly reduced if there is salt marsh present, compared to a site with no marsh at all; a 6 m wide fringe of salt marsh can reduce costs by 70% (Adnitt et al., 2007).

Scouring of sediments at the toe of sea defences undermines the structural integrity of sea walls and is a prevalent, serious, and costly problem in the U.K. (Bradbury et al., 2012). One strategy which has been shown to be effective in reducing wave energy involves artificially creating narrow fringes of salt marsh in front of existing sea wall structures (French and Reed, 2001). Salt marsh fringes can be created through the installation of gabions (cages of wire mesh filled with stone) to protect the toe of existing sea walls. Positioned to form a solid margin which is then backfilled with clay or sediment to form a terrace (i.e. a flat strip of raised ground on the seaward face of the sea wall), such terraces have the potential to enhance the local environment by creating new space which can be colonised by salt marsh vegetation. If successfully colonised, vegetated terraces could contribute to the dissipation of wave energy and further protect the sea wall. Gabion terraces have similar initial installation costs; £660 m^{-1} for the present study compared to approximately £635 m^{-1} (D. Gauntlett, pers. comm.) for the concrete blockwork and toe-board protection usually constructed for 'hard' engineered sea wall repairs. Through natural accretion and vegetation growth, the structural integrity of gabion terraces often increases over time, and they can withstand relatively high velocity flows (Miller and Rella, 2009).

By applying knowledge gained from disciplines such as restoration ecology and conservation biology during the design and installation of sediment-filled terraces, it should be possible to increase the potential of these structures to contribute to biodiversity. The zonation and vegetative recruitment of salt marshes is affected by a variety of inter-correlated environmental variables (Davy et al., 2011). Successful recruitment of vegetation is more likely when sites are adjacent to areas of existing and well-developed salt marsh (Wolters et al., 2008; Mossman et al., 2012a), with colonisation by salt marsh plants slow or absent if the availability of plant propagules is limited (Wolters et al., 2005; Dausse et al., 2008). Herbivory, particularly on seeds and seedlings by *Nereis diversicolor*, can also slow the process of colonisation (Paramor and Hughes, 2004), and storm events can result in substantial sediment erosion, which reduces the rate of colonisation by vulnerable seedlings (Boorman, 2003). Height within the tidal frame is a significant variable affecting sedimentation (Marion et al., 2009) and salt marsh vascular plant zonation (Crooks et al., 2002; Davy et al., 2011; Mossman et al., 2012a, 2012b). Elevation affects the duration and frequency of tidal inundation which can alter the community composition of salt marsh plants (Dawe et al., 2000). In addition to water logging of sediments, elevation can influence sediment redox potential, which can in turn determine the composition of vegetation (Crooks et al., 2002; Davy et al., 2011; Mossman et al., 2012a). Sediment water content and un-drained sediment shear strength also affect the capacity for salt marsh to resist erosion (Crooks et al., 2002).

In 2012, the U.K. Environment Agency (EA) (Eastern Region, Essex, Norfolk and Suffolk) piloted a scheme to trial the gabion and clay-infill approach. Soft engineered terraces of this kind have previously only been employed in tidal riverine settings (e.g. Greenwich Penninsula, London EA, 2015); this approach is therefore a novel intervention for sea wall repair on more open coastlines. As part of the ongoing programme of repairing erosion-damaged sea defences, 12 gabion-built terraces were installed at three locations in the Blackwater and Colne estuaries of Essex, S. E. England. The success of the trial would be assessed on the ability of the terraces to protect the "toe" of sea wall blockwork revetment over time, and by the recruitment of salt marsh vegetation onto the new surfaces, in areas where the marsh had been lost to erosion.

In this study, we investigated the rate of vascular plant colonisation and changes in the height and surface topography of clay infill. The aim was to determine the relationships between plant colonisation, terrace durability and controlling factors such as terrace area, height in tidal frame, location and exposure. This study covered a 22 month period which included a storm surge in December 2013, which created the highest water levels since the floods of 1953 (Spencer et al., 2015b). In the longer term, plant colonisation of the terraces could be considered successful if community composition were to converge with those found in existing marshes. However, recognising the considerable timescales (decades) required for stable communities of natural salt marsh vegetation to establish (Mossman et al., 2012b), in this study it was the appearance of pioneer (e.g. *Salicornia* sp.) and early perennial (e.g. *Atriplex portulacoides*) species which were of particular interest. The outcomes of this research could inform further trials of this approach to sea wall repair, and determine whether gabion terraces can provide a cost-effective solution to sea wall maintenance with the additional benefit of providing biodiversity gains for salt marsh communities.

2. Material and materials

2.1. Terrace installation

Twelve soft engineered terraces were installed by the EA at three sites within estuarine embayments along the Essex coastline. Langenhoe and Wellhouse (Mersea Island) both had five terraces, Tollesbury had two (Fig. 1, Table 1). Construction work commenced in January 2012 and the terraces were completed sequentially from June 2012, with the final terraces at the Wellhouse site being finished in August 2012 (Table 1). Individual gabion baskets filled with approximately 2 m^3 of stone were placed in front of the toe-boards at the base of the sea wall. The terraces were created by backfilling clay into the space between the gabion baskets and the sea wall (Fig. 2). The clay for the terraces at Langenhoe and Wellhouse was imported. At Tollesbury the clay was locally sourced by digging linear lagoons, this method involved extracting clay from the grazing marsh and had the benefit of creating additional aquatic habitat. The 12 individual terraces varied in size (from 4 m to 42 m in length and 1.5 m–3.0 m in width) due to variation in the extent to which sections of blockwork and toe needed repair (Table 1). This provided an opportunity for the trial to include size (length) as a factor affecting terrace stability and recruitment.

The Wellhouse site contained three 4 m long terraces (W2, W4 and W5) and two 6 m in length (W1 and W3). The top of the gabions were between 6.24 m (W2) to 6.41 m (W1) above Chart Datum (CD). Five terraces were constructed at Langenhoe, ranging from 5 m (L4) to 42 m (L2) in length. The terraces at Langenhoe were relatively low in the tidal prism, between 5.2 and 5.8 m above CD (Table 1). The Tollesbury site consisted of two terraces; T1 which at 52 m in length and 156.3 m^2 in area was the largest terrace in the study. The Tollesbury terraces also contained the gabion placed at the highest level above CD (T2. 6.85 m) (Table 1). The north-west facing sites at Wellhouse were subject to the lowest wave exposure and wind fetch ($F = 0.05$), Langenhoe was slightly less sheltered ($F = 0.13$), with the most exposed terraces present at Tollesbury (Table 1; $F = 2$). In comparison, the open coast salt marsh at Colne Point which had approximately 270° exposure to the open

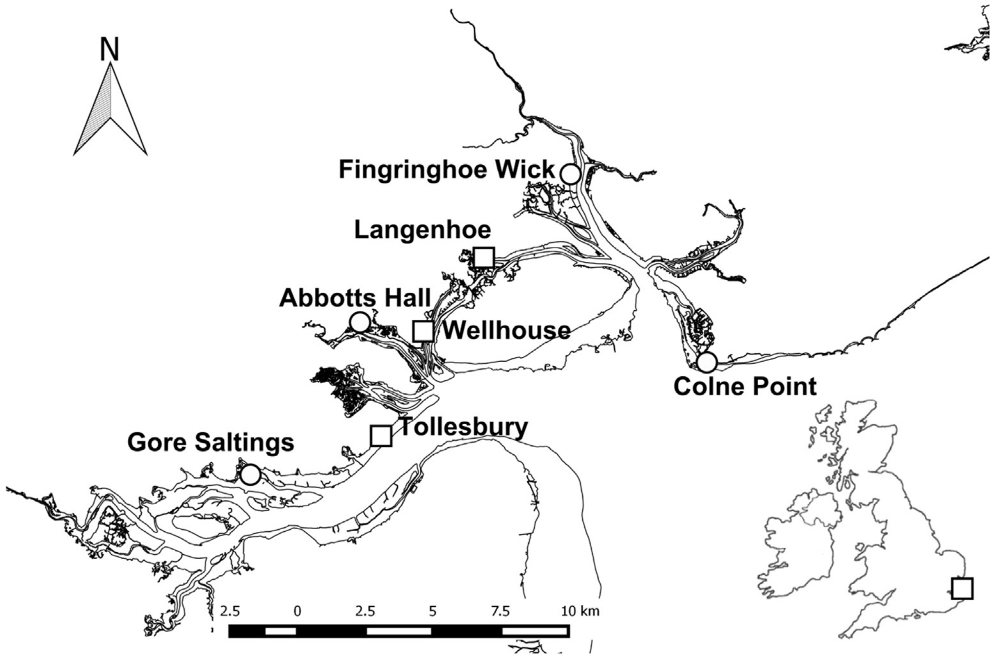

Fig. 1. Three sites in the southeast of England (squares) where experimental gabion terraces were constructed. Reference vegetation data was sourced from surveys conducted at four local salt marshes (circles). The relatively exposed location of Colne Point was used for a comparative reference for an index (F) of average wave fetch exposure (see Table 1 for grid references to individual terraces).

sea (Fig. 1, 51.775431N, 1.0373443E) had an exposure index of $F = 26.06$.

2.2. Surveys

The location of each terrace was recorded using the World Geodetic System 84 (WGS84), coordinates being taken from the seaward left hand corner of each gabion cage. Terraces were surveyed on four occasions (October–November 2012, July 2013, April 2014 and July 2014) using multiple transects. For each terrace, repeated sediment height point measurements were taken using a theodolite and staff at 20 cm intervals along transects extending perpendicular from the seaward edge of the gabion to the point where the infilled sediment met the sea wall. Transects were positioned at either end of each terrace and at intermediate points between. The number of transects (between 4 and 12) depended on the length of the terrace. This survey configuration enabled a grid of terrace surface heights to be derived, with a greater survey point density at the ends of individual terraces, as this was where we expected the greatest potential for erosion. The surface area of individual terraces was derived from transect positions and lengths which allowed polygons to be plotted to represent the perimeter of each terrace. Sediment heights were also taken on the adjacent sea walls and fringing salt marshes, and all heights converted to chart datum (CD) based on actual tide levels and tide table heights on particular days.

The percentage cover of vegetation within quadrats was recorded using the Domin scale (Dahl and Hadac, 1941). Due to the narrow width of the terraces, individual quadrats (size: 1m × 4m) were placed along the length of the terrace. Depending on the terrace length, between one and three quadrats were randomly placed with spacing of no less than 2 m between quadrats. Data for each terrace was pooled to provide a single data set for each site in each sampling period. The un-drained sediment shear strength (N m^{-2}) of terrace sediments was measured using a pocket vane (Model 16-T0174). The vane was inserted into the sediment to a depth of 5–6 cm and rotated until the sediment failed. Three replicate measurements were taken in the centre of each 1 m × 4 m

Table 1
General site description and grid references for 12 experimental soft engineered at three locations on the Essex coastline.

Site	Code	Location (GBNG)		Date of completion	Length of terrace (m)	Height of gabion (m) above chart datum (and height relative to adjacent salt marsh)	Aspect	Exposure (F)
		Latitude	Longitude					
Langenhoe	L1	51.807611	0.939306	06/2012	21.5	5.29 (−0.17)	S	0.13
Langenhoe	L2	51.807639	0.9375	06/2012	42	5.57 (0.06)	S	0.13
Langenhoe	L3	51.807639	0.936667	06/2012	31	5.58 (−0.02)	S	0.13
Langenhoe	L4	51.807556	0.931722	06/2012	5	5.8 (0.01)	S	0.13
Langenhoe	L5	51.809917	0.9235	06/2012	6	5.5 (−0.15)	S	0.13
Tollesbury	T1	51.746528	0.863694	07/2012	52	5.35 (−0.81)	SE	2
Tollesbury	T2	51.748278	0.866147	07/2012	48	6.85 (−0.61)	SE	2
Wellhouse	W1	51.7925	0.914028	08/2012	6	6.41 (−0.58)	NW	0.05
Wellhouse	W2	51.792472	0.913833	08/2012	4	6.24 (−0.27)	NW	0.05
Wellhouse	W3	51.790333	0.91	08/2012	6	6.34 (−0.63)	NW	0.05
Wellhouse	W4	51.789972	0.907694	08/2012	4	6.36 (−0.45)	NW	0.05
Wellhouse	W5	51.789722	0.907333	08/2012	4	6.34 (−0.22)	NW	0.05

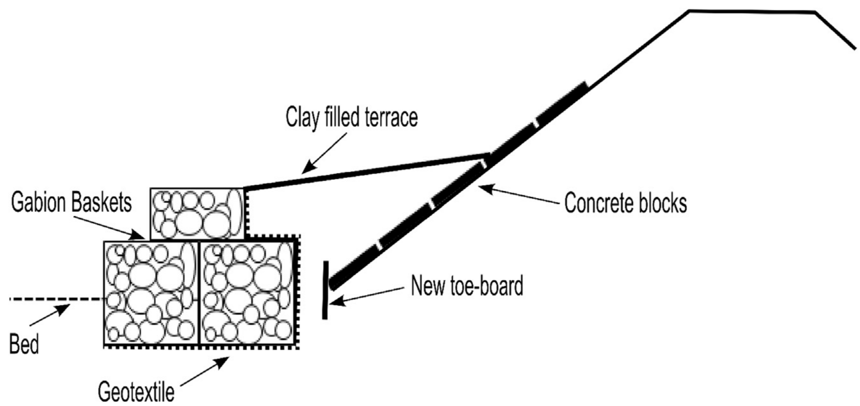

Fig. 2. Schematic cross section of new terraces, backfilled with clay behind stone filled gabions to protect the toe of the sea wall.

quadrat.

At each site, measures of vegetation cover were also taken in September 2015 from existing fringes of saltmarsh. These established marsh "control" comprised four 4 m^2 quadrats placed 10–20 m from terraces and within 4 m of the sea wall toe. An additional set of salt marsh vegetation Domin data comprising 40 National Vegetation Classification (NVC, Rodwell, 2000) quadrat counts from each of four salt marshes (see Fig. 1 for marsh locations) were used as reference data to provide a further comparison between terrace vegetation and that of existing local marshes (Cousins, 2016).

An index for exposure or average wave fetch (*F*) was derived for each of the three sites. Using a Geographic Information System (QGIS 2.4.0) a circle with a 200 km radius was centred over each site then divided into 16 equal segments of 22.5°. The wave fetch, or distance (km) from the site to the nearest land within each segment, was recorded and the index produced by averaging these measurements. Following Burrows et al. (2008) the 200 km radius is a maximum and the comparative value of the index is not affected if landfall is made in less than 200 km in some or all of the 16 segments.

2.3. Data analysis

Surface height measurements were used to create contour (xyz) plots to depict changes in terrace topography between October 2012 and July 2014. Figures were generated in the statistical environment R with package "Lattice" (Sarkar and Deepayan, 2008; R Core Team, 2013). The changes in sediment bed height due to either sediment erosion or deposition resulting from the December 2013 storm surge was measured by comparing replicated point heights recorded on each terrace before (July 2013) and after the storm surge event (April 2014). Normality within the data was not assumed and Mann-Whitney U tests for significant differences were conducted in R utilising the package "stats" (R Core Team, 2013).

Similarity in plant species composition and mean vegetation cover within terraces and controls were compared using Bray-Curtis dissimilarity among the composition of sampled units (Faith et al., 1987). Indices were computed in R using the "vegdist" function within the package Vegan (Oksanen et al., 2013). The relationship between vegetative colonisation and sediment shear strength was investigated with non-linear least squares analysis. Linear regression was applied to determine the effect of elevation within the tidal prism (based on CD) on plant colonisation. Ordination with Canonical Correspondence Analysis (CCA) was used to determine the importance of potential explanatory variables (time since construction, height (CD), sediment shear strength, exposure (*F*) and terrace length) which could influence plant colonisation, following checks for the absence of cross-correlation. All explanatory variables were standardised to zero mean and unit variance to negate the possible effect of different units of measurement. CCA was chosen for its capacity to analyse unimodal responses and to proportion the amount of variation attributable to explanatory variables. Analysis was conducted in R with the package "vegan" (Ter Braak, 1986; Oksanen et al., 2013).

3. Results

3.1. Net sedimentation and erosion over 22 months

Over the three growing seasons, a pattern of sedimentation was seen at seven of the 12 terraces where net sediment deposition occurred towards the sea wall and was eroded towards the edges of the seaward edges of the terraces (Figs. 3 and 4; W2, W3, W4, L1, L3, L4 and L5). Three of the terraces at Wellhouse exhibited erosion just behind the gabion cages (W1, W3 and W4). Overall, at Wellhouse, the changes in sediment surface heights were non-significant, and all five of the terraces maintained constant sediment heights throughout the period studied.

Over the 22 month study period (October 2012–July 2014), Langenhoe terraces L5 and L2 increased in surface height by +8 cm and +12 cm respectively. There was net accretion on Langenhoe terraces L1, L2, L4 and L5 during the period July 2014 to April 2014 (including the tidal surge of December 2013) (Table 2), with only Terrace L3 losing sediment ($U = 2.25$, $p = 0.03$, Table 2). Terrace L2 had a particularly pronounced ($U = -11.02$, $p < 0.01$) +13 cm gain in surface height during this period, which was concentrated towards either end of the terrace (Fig. 4). Terrace L2 had its highest levels of erosion in the centre of the terrace with some deposition on the side furthest from the estuary-mouth and against the sea wall.

From October 2012 to July 2014, both Tollesbury terraces decreased in surface height. Erosion at T1 was lateral, i.e. sediment was scoured from behind the gabions towards the centre of the terrace (Fig. 5). The surface topography of both Tollesbury terraces significantly changed during the period including the 2013 surge event. Terrace T2 gained in height (+6 cm) whilst terrace T1 lost 10 cm of sediment to erosion (Table 2; $U = 5.66$, $p < 0.01$). At T1 most of the deposition occurred in the middle of the terrace, where an L-shaped bend in the sea wall was present, while increased erosion occurred towards the edges of the terrace. T2 showed the highest level of erosion in the centre of the terrace with decreased erosion towards the edges, and high levels of deposition towards

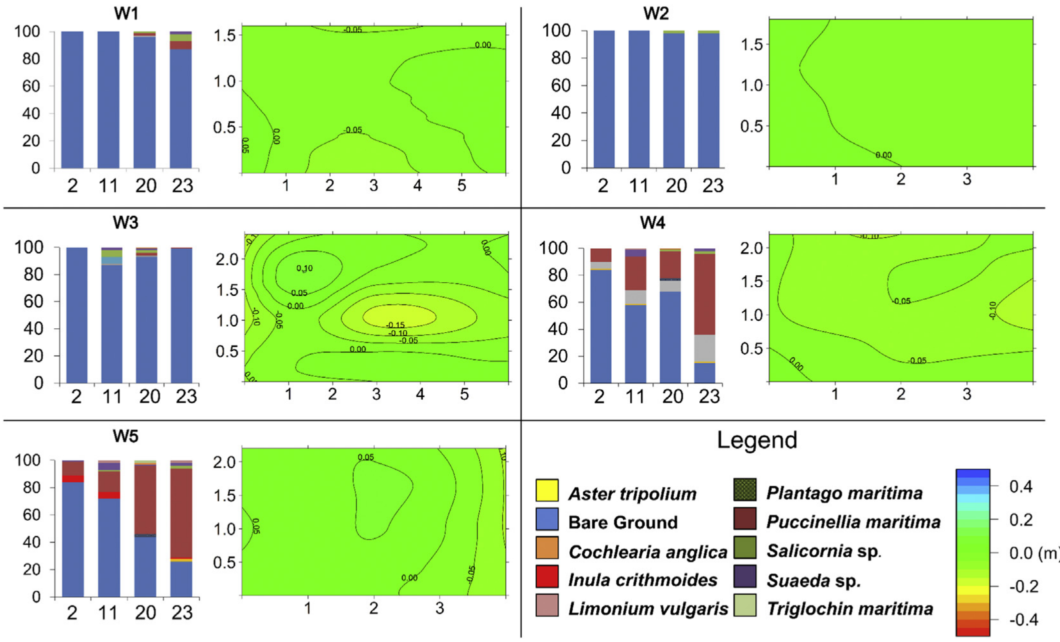

Fig. 3. Vegetation cover and composition and mean changes in surface topography of five terraces at Wellhouse (October 2012 to July 2014). Bars represent percent (%) plant cover at 2, 11, 20, and 23 months after terrace construction was completed. Contour plots illustrate the net effects of deposition and erosion between months 2 and 23, Terrace length (x axes) width (y axes) and height change (z axes) are shown in meters.

the edge least exposed and sea wall side (Fig. 5).

Sediment shear strength at Wellhouse ranged between 2.75 N m^{-2} (standard deviation (sd) = 0.31) on terrace W3, and 4.28 N m^{-2} (sd = 0.51) on terrace W5. Terraces at Langenhoe had both the overall lowest (L1 = 2.23 N m^{-2}, sd = 0.34) and the highest shear strength (L4 = 5.33 N m^{-2}, sd = 0.80). The two terraces at the Tollesbury site had sediment shear strengths of 2.84 N m^{-2} (sd = 1.06) at T2, and 3.67 N m^{-2} (sd = 0.69) at T1. Variation in sediment shear strength between the three locations was insignificant, although sediment shear strength at L4 was significantly higher than at the other sites ($t = -4.16$, $p < 0.05$).

3.2. Colonisation by salt marsh vegetation

Twelve plant species, all of which are common on Essex salt marshes, were recorded colonising the terraces. The largest number of species to occur on any one terrace was six (L1 October 2012 and W5 July 2014). The most frequent species were *Salicornia* sp.,

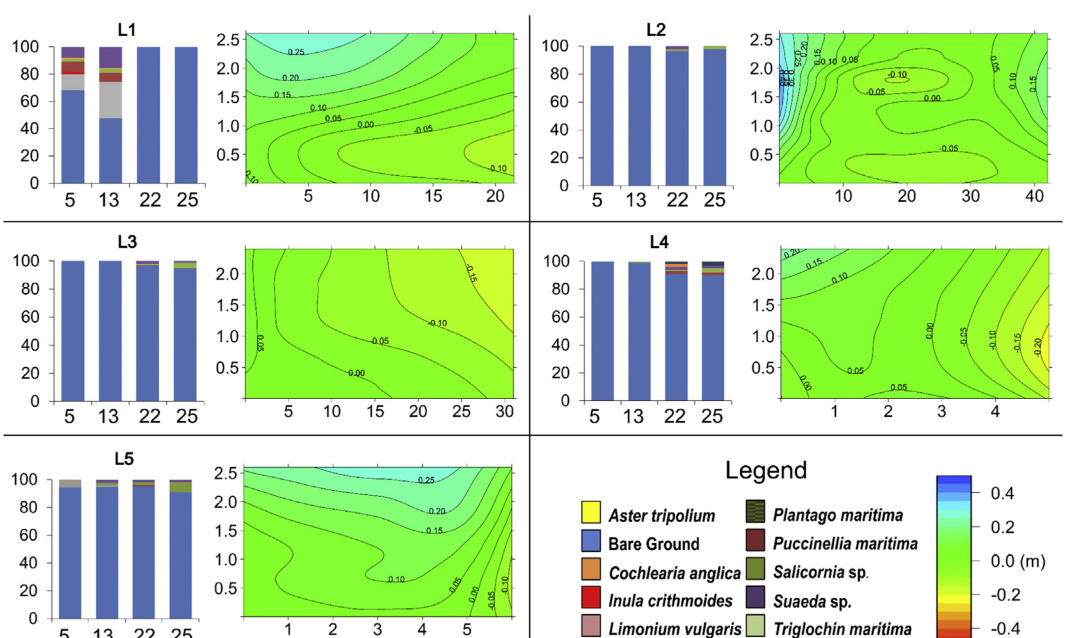

Fig. 4. Vegetation cover and composition and mean changes in surface topography of five terraces at Langenhoe (November 2012 to July 2014). Bars represent percent (%) plant cover at 5, 13, 22 and 25 months after terrace construction was completed. Contour plots illustrate the net effects of deposition and erosion between months 5 and 25, Terrace length (x axes) width (y axes) and height change (z axes) are shown in meters.

Table 2
Comparison of sediment surface height measurements (Mann-Whitney U test) between July 2013 and April 2014, to determine the significance of any erosion and sedimentation during the period including the tidal surge of December 2013.

Berm	Difference (m)	U	df	p value
W1	−0.02	0.63	76	0.53
W2	0.00	−0.12	78	0.90
W3	0.01	−0.75	112	0.46
W4	−0.02	1.47	78	0.15
W5	0.01	−1.40	113	0.16
L1	0.08	−6.48	204	0**
L2	0.13	−11.02	213	0**
L3	−0.03	2.25	247	0.03*
L4	0.08	−4.16	131	0**
L5	0.06	−4.40	189	0**
T1	−0.10	5.66	264	0**
T2	0.06	−2.68	251	0.01*

$p < 0.05$* and 0.01**.

followed by *Puccinellia maritima*, and *Suaeda* sp.

All 12 terraces showed some degree of plant colonisation between at least one pair of observations (Figs. 3–5). The smallest recorded increase was at Tollesbury where terrace T1 remained without vegetation until April 2014 when it had 0.3% cover (Fig. 5). The highest relative vegetation cover was on W4 (85%, July 2014), and W5 with 74% cover in July 2014 and 56% in April 2014 (Fig. 3).

Following the tidal surge in the winter 2013–2014, two of the 12 terraces had decreased plant cover. On Terrace L1, vegetation (cover of 52.3%) recorded in July 2013 was completely removed (reduced to 0%) by the following April (Fig. 4). Vegetation cover on terrace W3 decreased from 13% (July 2013) to 7% (April 2014) and 1% in July 2014.

Seven of the 12 terraces (W1, W4, W5, L3, L4, L5 and T2) were successful in that they experienced plant colonisation and increasing vegetation cover through the period of study (Fig. 6a). Terraces W4, W5, L3 and T2 maintained an equal or increasing number of species for each subsequent observation (Figs. 3–5).

3.3. Comparing vegetation composition with adjacent control samples of existing salt marsh fringe vegetation

Though percent vegetation cover at Wellhouse terraces was lower than within the control sample, terraces were colonised by all species identified within the adjacent marsh. *Puccinellia maritima* was starting to become a dominant species which reflected the dominance of this species within the control (Table 3). Nevertheless, dissimilarity between terrace and control was high (Bray-Curtis = 0.55). The Langenhoe control plots contained seven species, five of which were present on the terraces. *Atriplex portulacoides* was dominant within the control (64.5% cover) and the absence of this perennial combined with the generally sparse cover at the Langenhoe terraces resulted in high dissimilarity between terraces and the control (Bray Curtis = 0.94). Terraces at the Tollesbury site were also highly dissimilar to their controls (Bray-Curtis = 0.91). Within the control at Tollesbury, *A. portulacoides* was the dominant of seven species (58.5% cover). Only two of these species occurred on the terraces, with *A. portulacoides* absent (Table 3).

3.4. Qualitative comparison of terrace vegetation with local salt marshes

Plant communities within the four reference salt marshes (Fig. 1) were sampled with quadrats identical to those employed on the terraces. Assembled from a pool of 17 species, the mean alpha diversity (i.e. species richness within a single sample unit) across the reference sites was six (minimum of three and maximum of 11 species). Percent vegetation cover at the reference sites ranged from 65 to >100% (accounting for vertical structure, the Domin system allows communities to receive scores of greater than 100%). The most frequently occurring and dominant species within the reference sample were *P. maritima* (36% cover), *A. portulacoides* (23% cover), *Salicornia* sp. (11% cover) and *Limonium vulgaris* (10% cover).

By comparison, the cover of vegetation colonising the terraces was sparse, and with the exception of terraces W4 and W5 (Fig. 3), coverage was below the minimum recorded at reference marshes. Nevertheless, pioneer species (e.g. *S. maritima* and *Salicornia* sp.) were beginning to colonise terraces and 12 of the 17 species occurring at reference sites were present after 22 months on the terraces. Two of the terraces had supported a total of six species which matched the average relative richness found on the reference sites.

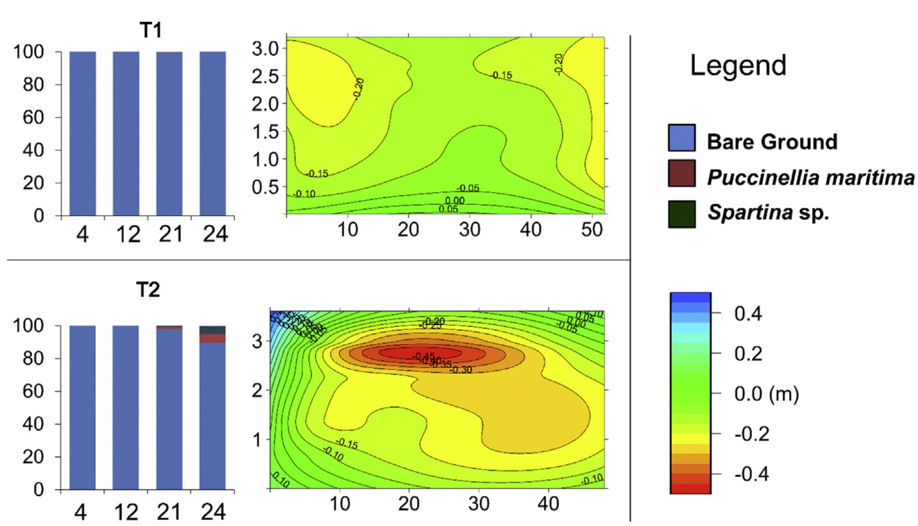

Fig. 5. Vegetation cover and composition and mean changes in surface topography of two terraces at Tollesbury (November 2012 to July 2014). Bars represent percent (%) plant cover at 4, 12, 21 and 24 months after terrace construction was completed. Contour plots illustrate the net effects of deposition and erosion between months 4 and 24, Terrace length (x axes) width (y axes) and height change (z axes) are shown in meters.

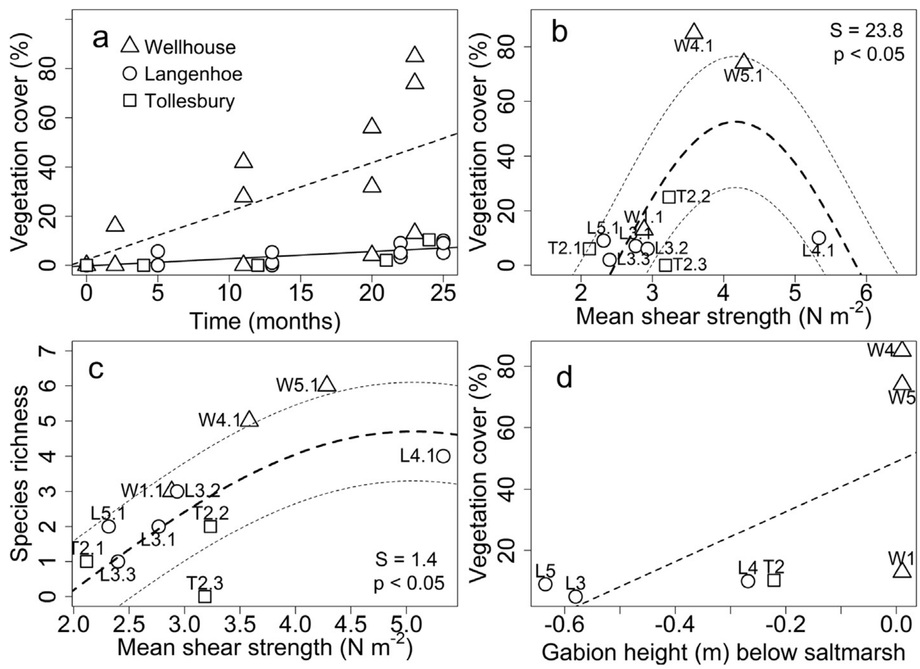

Fig. 6. (a) changes in vegetation cover at seven terraces W1, W4, W5, L3, L4, L5 and T2 over time from construction in 2012. Broken line = line of best fit for Wellhouse, solid line = line of best fit for Langenhoe and Tollesbury. (b) Non-linear relationship between sediment shear strength and vegetation cover, (c) plant species richness of seven successfully colonised terraces in July 2014. Fitted curve (dashed line) and standard deviation (dotted lines). (d) Relationship between vegetation recruitment (% cover) and terrace height in relation to existing adjacent saltmarsh ($r^2 = 0.4$ $p = 0.1$).

3.5. Factors determining plant colonisation

Sediment bed heights at each of the seven terraces showing plant colonisation remained relatively stable over the four observation periods (Figs. 3–5). Species richness and plant cover were greatest at two terraces that had sediments with average undrained shear strengths between 3.5 and 4.5 N m^{-2}. Curve fitting predicted ($p < 0.05$) a shear strength of 4.25 N m^{-2} as optimal for vegetative cover, with plant species richness positively related to sediment shear strength (Fig. 6b and c). There was a positive relationship between the extent of vegetative recruitment and sediment terrace height, with vegetation cover positively related to sediment bed height relative to that of adjacent salt marsh surfaces (Fig. 6d, $r^2 = 0.43$). Five of the seven terraces were fringed with, or adjacent to, existing remnants of salt marsh which offered a potential local source of propagules. Terraces W1 and W5, that has no adjacent marsh, showed low plant colonisation, despite the sediment bed height being within ±6 cm of existing salt marsh within the estuary.

Ordination (CCA) explained 72% of the variation in the distribution of plant species (Fig. 7). Elevation relative to existing salt marsh exerted the greater influence (37%). Weighted averages generated by CCA positioned plant species along a gradient of elevation; with the pioneer species *Salicornia* sp. centred at the lower range of the gradient and *A. portulacoides*at the upper. Exposure (*F*) accounted for 30% of the variation. With the exception of *Spartina* sp., the extent to which plant species were distributed along the exposure axis was less pronounced than for elevation. *Spartina* sp. distribution was centred common to the main centroid with regards to elevation but was strongly influenced by exposure. The remaining five percent of explained variation was due to terrace length and shear strength. Twenty eight percent of the variation was unexplained by the constraining variables.

Analysed independently from elevation, neither terrace length nor surface area had a significant effect on the rate or extent of plant colonisation. Sediment stability and an absence of deposition provided conditions favourable to colonisation. Terraces that experienced the least change in surface height were favourable for plant recruitment. The relationship between the 'terrace averaged' percentage vascular plant cover (July 2014) and the 'terrace averaged' relative change in sediment height over the October 2012–July 2014 period was calculated for all terraces ($r^2 = -0.08$, $p = 0.73$). Though low in statistical power the absence of any significant correlation between changes in sediment surface height and vascular plant colonisation allude to the potential importance

Table 3
Similarity (Bray-Curtis index "**bold**") between the mean (%) cover of vegetation colonising newly constructed terraces and control samples of existing salt marsh fringe at three sites in Essex.

Site		*Aster tripolium*	*Atriplex portulacoides*	*Inula crithmoides*	*Puccinellia maritima*	*Salicornia* sp.	*Suaeda* sp.	*Limonium vulgaris*	*Spartina* sp.	*Elymus* sp.	Index
Wellhouse	Terrace	0.5	3.33	0.17	22.17	2.17	1	0.67	0.33		**0.55**
	Control	11	8.75	0	60.75	4.00	4.50	3.75	9.75		
Langenhoe	Terrace	0.09	0		0.18	2.36	0.73		0.27	0	**0.94**
	Control	4.5	64.5		24.75	1.75	3		1.5	0.5	
Tollesbury	Terrace		0		2.67	0	0	0	2.5		**0.91**
	Control		58.50		10.5	3.25	8.25	8.75	16.25		

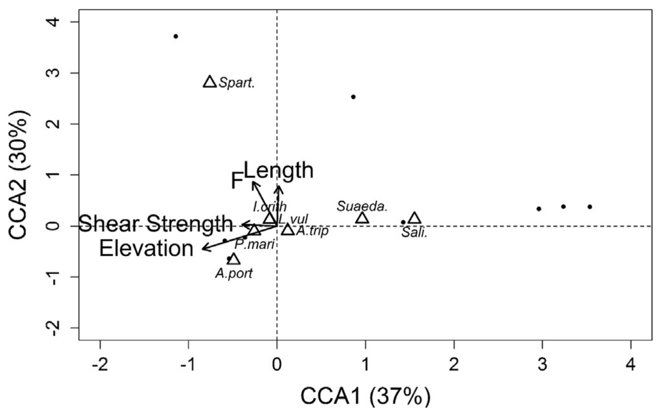

Fig. 7. Distribution of eight species of plant at ten colonised experimental terraces, 23–25 months after construction in relation to environmental factors. Ordination diagram of Canonical Correspondence Analysis (CCA), for axes CCA1 and CCA2 the percentage cumulative proportion of constrained eigenvalues are shown in parentheses. Triangles indicate plant species, dots indicate terrace sites and environmental variables are illustrated by arrows. The plant species are; A. trip = *Aster tripolium*, A. port = *Atriplex portulacoides*, I. crith = *Inula crithmoides*, L. vul = *Limonium vulgaris*, P. mari = *Puccinellia maritima*, Sali = *Salicornia* sp., Suaeda = *Suaeda* sp and Spart. = *Spartina* sp. Environmental variables are; Length = terrace length, F = an index for wind and wave exposure, sediment shear strength (N m^{-2}) and Elevation = the height each terrace was constructed relative to the surface of local remnant salt marsh.

of substratum stability and height in the tidal prism in facilitating plant colonisation.

4. Discussion

During the 22 months since installation, the 12 gabion terraces underwent only minor changes in overall sediment bed height, and varying degrees of colonisation by salt marsh vegetation. The variability in responses revealed some key factors for consideration in order to improve the future success of this approach to sea wall repair.

Deposition of sediment tended to occur towards the sea wall while the erosion was predominately towards the seaward edges of the terraces. Some terraces experienced erosion directly behind the gabion cages, with shallow channels forming along the inward face of the gabion and running to the end of each terrace. This particular form of sediment loss may be preventable if additional gabion cages are used to provide an inward edge to abut the sea wall at the terrace ends. A design adjustment of this kind could be trialled as a potential way to improve terrace integrity. The ability of the trial terraces to maintain sea wall integrity was incidentally and thoroughly tested by a significant storm and tidal surge on the eastern coast of England on 5th December 2013 (Spencer et al., 2015a, 2015b). Each of the 12 terraces remained structurally intact. With the exception of two terraces which had significant sediment erosion, 10 remained unchanged or gained height from sediment deposited by the storm.

Each terrace provided a narrow strip of new sediment substrata that had the potential to support salt marsh vegetation. Increasing colonisation by salt marsh plant species on seven terraces was encouraging and demonstrated how terraces of this type can successfully provide space for wildlife. The extent of sediment compaction was an important factor affecting vegetation recruitment. The present study found that sediment shear strength of approximately 4.25 N m^{-2} was optimal for vegetative recruitment. Shear strength should be considered when deciding on the type of clay to be used and how tightly the fill material is compacted. There are well-recognised interactions between sediment stability, sediment trapping and accumulation, and plant cover (Boorman, 2003; Langlois et al., 2003; Mason et al., 2003). The stability of salt marsh substrata is important for vegetative recruitment and is linked to elevation (Dawe et al., 2000). *Puccinellia maritima* can have a pronounced beneficial effect on sediment stabilisation (Langlois et al., 2003), and is particularly resilient to storm surge stresses (Spencer et al., 2015a). In this study, *P. maritima* colonised terraces at the most exposed site, suggestion that planting of *P. maritima* could offer a biological solution for the stabilisation of terrace clays minimising early erosion of backfill material.

Salt marsh vegetation takes time to develop; a study of the long term recolonisation of previously reclaimed sites estimated time periods in excess of 100 years to reach a fully comparable flora to mature marshes (Mossman et al., 2012b). The successful colonisation across the trial sites by 12 plant species after 22 months was promising. These species were the most common taxa in the mature marshes present in this region. Some terraces had three or more species present which demonstrated potential for further colonisation and suggested these terraces will follow a positive trajectory towards further increases in species richness and cover. It remains to be seen whether terraces can support communities exactly comparable with established salt marshes. Nevertheless, early indications were that certain terraces had the necessary integrity and stability to allow a salt marsh flora to colonise, persist and expand its cover. Our study found that the height within the tidal frame at which terraces were constructed was a significant factor in successful plant colonisation. Elevation within the tidal prism affects the duration and frequency of inundation and is a major determinant of salt marsh plant composition (Davy et al., 2011; Mossman et al., 2012b). Our study has shown that the probability of successful plant colonisation increases if terraces are constructed to a level comparable with adjacent salt marsh. Though the tidal elevations at which middle salt marsh plant communities can exist varies (Rodwell, 2000), pre-construction surveys to establish a location specific height at which nearby plants (e.g. *A. portulacoides*) occur would provide a datum for the terrace surface to which construction engineers could work.

Consistent with Mossman et al. (2012a), proximity to existing salt marsh (i.e. having a local source of propagules) appeared to contribute to the successful recruitment of salt marsh vegetation. Five of the colonised terraces were adjacent to fringes of existing salt marsh. As sea wall repairs may not necessarily be needed in locations with a convenient local source of propagules, planting could be considered as a means of accelerating vegetation cover.

Due to the constraints on the choice of sites used in the trial, this study did not have the statistical power to resolve the importance of aspect-exposure index on plant recruitment and percentage cover. Although all of the terraces were in relatively sheltered estuarine locations, exposure did affect plant composition. Early pioneers at the most exposed site were the grasses *Spartina* sp. and *P. maritima.* The length of terraces was also of interest but without being controlled against other variables, notably elevation, we were unable to satisfactorily test for its effect.

The Water Framework Directive (WFD) requires that repairs to flood defences are completed without deteriorating the ecological potential of tidal waters (Environment Agency, 2015). To meet this requirement, a specific mitigation measure to prevent deterioration within the Blackwater and Colne Estuaries WFD waterbody is to either remove hard bank reinforcement/revetment, or to replace it with soft engineering solutions (Environment Agency, 2015). This pilot study has shown that as a soft engineered approach, these gabion – clay infill terrace installations can contribute towards achieving this mitigation measure and therefore compliance with the European Directive, as well as being a cost-equivalent option for sea wall repair. This method is one of a range of soft engineering solutions available (e.g. faggots or stakes are also used to anchor sediments, Suffolk Coasts and Heaths Org. 2015) that can also enhance biodiversity as well as providing a sea defence function. The theoretical advantages of the gabion-terrace approach are in the double benefits of securing the defence of coastal land while providing additional habitat for wildlife. There is further opportunity for biodiversity gains if consideration is given to the source of construction material. In building the two terraces at Tollesbury, "backfill" materials were locally sourced from inland grazing marsh creating saline lagoons supplemental to the diversity of existing habitat within the estuary. Another possible method is to extract clays from the landward side of the flood defence which, as in the construction of earth sea walls, would create additional pond or borrow dyke habitat.

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