



Hybrid engineering incorporating salt marsh terraces into sea wall repair maintains their defence function and creates new habitats

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ABSTRACT

Sea walls are widely-used engineered structures designed to protect low lying land from flooding. Rising sea levels and coastal erosion threatens sea wall integrity and requires maintenance and repair. As an alternative to conventional repairs, a hybrid engineering design incorporating sediment-filled terraces to allow the development of salt marsh has been trialled to protect sea walls in the Colne-Blackwater Estuary complex, Essex, U.K. Over a 10 year period, three different trajectories of development were measured. Salt marsh halophytes colonised 7 out of 15 terraces, with % plant cover ranging between 5% and 170%. Halophyte species richness on terraces ranged from 1 to 8 species (average = 1.7 per 4 m²), compared to existing salt marsh adjacent to the repairs (max. species richness 9, average 6.85), with positive significant relationships between % halophyte cover and halophyte richness and between % halophyte cover and sediment water content (19–51% on terraces). Organic carbon content was significantly lower on the terraces compared to existing salt marsh (7–19 % AFDW on terraces, 17–24% in marsh sediments). A sediment shear strength of 30 kPa was optimal for % plant cover on terraces. The height of the terraces relative to tidal inundation was a key determinant of successful halophyte colonisation. A second trajectory (3 of 15 terraces) resulted in development of macro- and microalgal mats, up to 100 % cover. Both these trajectories resulted in up to 25 cm of vertical sediment accretion. Shell banks formed on 2 terraces. Seven terraces were subject to erosion, associated with desiccation and absence, or loss, of halophyte cover. Eroding terraces showed both surface lowering (20–30 cm) and massive sediment failure (blocks >50 cm deep falling away). After a decade, protection of the sea wall was still provided by 14 of the 15 terraces, with only one having to be rebuilt. Utilising a nature-based approach incorporating sea wall terraces resulted in the formation of three types of linear habitat (salt marsh, microbial mats, shell banks) providing some ecosystem services that would not have existed if a conventional hard-engineered repair had been used.

1. Introduction

Nature-based solutions (NbS) as an approach to addressing the joint challenges of delivering sustainable environment management and countering losses of biodiversity have gained prominence in the last decade (Narayan et al., 2016; Seddon et al., 2020a, 2020b, 2021, 2020b; Stafford et al., 2021). Nature-based solutions (approaches that work with and enhance nature to help address societal challenges, Seddon et al., 2020a) can provide multiple ecosystem services by harnessing natural processes, increasing biodiversity and ecosystem functioning, and increasing natural resilience to external pressures. Within coastal environments these ecosystem services include provision of food and resources, nutrient cycling and clean water, blue carbon capture, and

flood and storm surge coastal defence (Narayan et al., 2016; Waltham et al., 2020; Guerry et al., 2022; Environment Agency, Chief Scientist's Group, 2023). Protection or restoration strategies for coastal or marine habitats to enhance NbS are included within the Nationally Determined Contributions (NDCs) for the Paris Agreement of the United Nations Framework Convention on Climate Change (UNFCCC) of 37% of coastal states within the UN (Seddon et al., 2020b). Within the U.K., such approaches are now adopted by agencies and organisations, for example, the statutory body for nature conservation, Natural England, has launched a Green Infrastructure Framework which aims to increase the area of green cover in towns and cities to 40% (Natural England, 2023) and the UK Environment Agency propounds increased use of NbS to provide future flood and coastal erosion resilience (Environment

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Agency, 2020; Environment Agency, Chief Scientist's Group, 2023).

The North Sea coastline of south eastern England is highly modified with flood defence sea walls. A major storm surge and flooding event in 1953 which caused over 300 fatalities resulted in repair and enhancement of these, including building approximately 2,100 km of earthen sea wall embankment around the coasts of England and Wales (Gardiner et al., 2015). These high levels of coastal modification are common on many coastlines across the world (Sutton-Grier et al., 2015; Evans et al., 2019). Many of these defences are now approaching the end of their design life and are challenged by rising maintenance costs (Temmerman et al., 2013) and pressure from accelerating rates of relative sea level rise (between 14 and 25 cm rise by 2050 for the Essex and South Suffolk Shoreline Management Plan region, Sayers et al., 2022). In addition, the 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). Having a well-developed coastal fringe of mudflat, salt marsh and seagrasses (*Zostera* spp.) can provide significant wave attenuation benefits for sea defence (Möller et al., 2014; McKinley et al., 2020; Gardiner, 2022). 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). If successfully colonised, vegetated terraces could contribute to the dissipation of wave energy and further protect the sea wall. 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). The Institute of Civil Engineering (ICE) manual for Blue-Green Infrastructure (BGI) contains a chapter on the implementation of BGI for coastal and estuarine ecosystems that includes the establishment of salt marsh terraces. This type of BGI is specifically referenced in several Welsh and Scottish statutory policies. The benefits of such hybrid-engineering approaches are often lower cost compared to engineered solutions, and they increase the provision of habitat and ecosystem services (Sutton-Grier et al., 2015; Naylor et al., 2018). Sea wall terraces have been installed within the UK and also worldwide: along the Thames Estuary ("The Millennium Terraces", Naylor et al., 2023) and in Tokyo Bay, the latter installed to promote the colonisation of *Aster tripolium* (Kuramoto et al., 2017; Gardiner et al., 2019; Mishima et al., 2019).

In Essex, as part of a sea wall reconditioning programme being undertaken in spring 2012 by the EA (with a contractor doing the works), new sea wall blockwork revetment was protected by installing salt marsh terraces in front of the defences to fill in the gaps where salt marsh had been eroded and the new blockwork would be exposed. Potential salt marsh fringes were created through the positioning of gabions (cages of wire mesh filled with granite blocks) forming a solid margin which was then backfilled with clay to form a terrace (i.e. a flat strip of raised ground on the seaward face of the sea wall) (See Fig. 2 in Cousins et al., 2017). The aim was to protect the toe of existing sea walls and to enhance the local environment by creating new space which could be colonised by salt marsh vegetation. Gabion terraces had similar initial installation costs; £660 m⁻¹ for the present study compared to approximately £635 m⁻¹ for 'hard' engineered sea wall repairs to concrete blockwork and toe-board protection (2012 prices, D. Gauntlett, Env. Agency engineer, pers. comm.).

The original engineering design life was estimated to be for 10 years (D. Gauntlett, pers. comm.); Cousins et al. (2017) reported on the first two years of this experimental approach. During the first 25 months, elevation, exposure and sediment shear strength were found to be key factors in the primary succession of halophyte colonisation and success of the engineered terraces. Here we report on the establishment of halophyte plant communities, sediment shear strength, algal mat cover, and success of the terraces in protecting the existing sea defence structures over a 10 year period, from summer 2012 to summer 2022. Sites were compared with the natural salt marsh communities growing adjacent to the engineered terraces, to determine the trajectory towards

achieving a salt marsh typical of the local conditions. We identify features of this BGI design that were associated with both success and failure and make recommendations about the continued application of this salt marsh terrace approach.

2. Methods

Fifteen soft engineered terraces were installed by the U.K. Environment Agency (statutory body for flood defence) at three locations where sea wall repairs were needed within embayments of the Blackwater and Colne estuaries in Essex, England. The Blackwater and Colne estuaries are typical ebb-dominated macrotidal coastal plain estuaries (tidal range 5.2–5.8 m). The estuaries are hypernutrified and well-mixed, with strong dilution gradients and freshwater inputs predominantly at the head of each estuary (Chesman et al., 2006; McMellor and Underwood, 2014; Nedwell et al., 2016). The terrace locations were all in the euhaline region of the estuaries (Fig. 1, Table 1). Construction work commenced in January 2012 and the terraces were completed sequentially from June 2012. The terraces were constructed from sets of 1 m³ wire gabion baskets filled with granite stones, placed 2–4 m from the sea wall toe. The terraces were created by backfilling clay into the space between the gabion baskets and the sea wall, with the clay for the terraces at Langenhoe and Wellhouse imported, while at Tollesbury the clay was locally sourced by digging linear lagoons to provide additional habitat on the grazing marsh. The Wellhouse site was on an eroding frontage on the south side of the Strood channel (Fig. 1). This site has been subject to other attempts to prevent salt marsh erosion, with a set of abandoned polders present on the intertidal mudflats adjacent to the sea wall (Fig. 1B). The five terraces at Wellhouse were between 4 and 6 m in length (Table 1). The Langenhoe sites were located on the north side of Langenhoe Creek which separates Mersea Island from the mainland (Fig. 1C). This was a sheltered site (exposure index 0.13, see Cousins et al., 2017, Table 1), with the longest fetch (6 km) being eastwards (opposite to the main direction of prevailing winds) along the Pyefleet channel to the Colne estuary. There had been significant erosion by 2012 of the fringing salt marsh at this location, with the toe of the sea wall exposed in a number of locations and salt marsh lost or substantially eroded. Seven terraces were constructed at the Langenhoe site, with lengths from 5 to 42 m long. Initially five terraces were surveyed (Cousins et al., 2017). Two further terraces built in 2012 (L-i = 14 m long and L-ii = 11 m long), to the east of the original set were included in surveys after 2014 (Table 1). Three terraces were constructed at the Tollesbury site in 2012 (Table 1). This site was located on the wave-exposed north side of the Blackwater Estuary (Dearing et al., 2006), where considerable salt marsh loss has occurred since the 1970s (196.6 ha loss 1973–1997, –22.3%, –8.19 ha yr⁻¹; Thomson et al., 2011). Tollesbury had the highest exposure index of the three sites (Table 1), and the sea walls at this site subject to repair in 2012 were not fronted by salt marsh, but mixed shorelines of cohesive silt and clay, sand, mixed shingle and shell banks (Fig. 1D).

A set of terraces were monitored at each site from 2012 to 2014 (Cousins et al., 2017). From 2016 (month 49) the survey was extended to include an additional three terraces (two at Langenhoe and one at Tollesbury) that had not been surveyed in Cousins et al. (2017). From 2016, floristic surveys in the *Atriplex* and *Puccinellia* zones (the dominant National Vegetation Classification (NVC) salt marsh communities in Essex, Rodwell, 2006) in the established salt marsh at each site adjacent to the terrace structures were carried out. Additional funding from the Environment Agency allowed for a more comprehensive survey in 2017. Further unfunded surveys in 2020 and 2022 were limited to floristic and sediment shear strength measurements.

Sediment bed height of individual terraces at each location were surveyed from 2012 to 2017, using multiple "mini-transects" across the terraces, from seaward extent (gabion edge) to the base of the sea wall, recording the elevation (to a 0.5 cm resolution) at regular 20 cm horizontal intervals using a theodolite located on the top of the sea wall. A

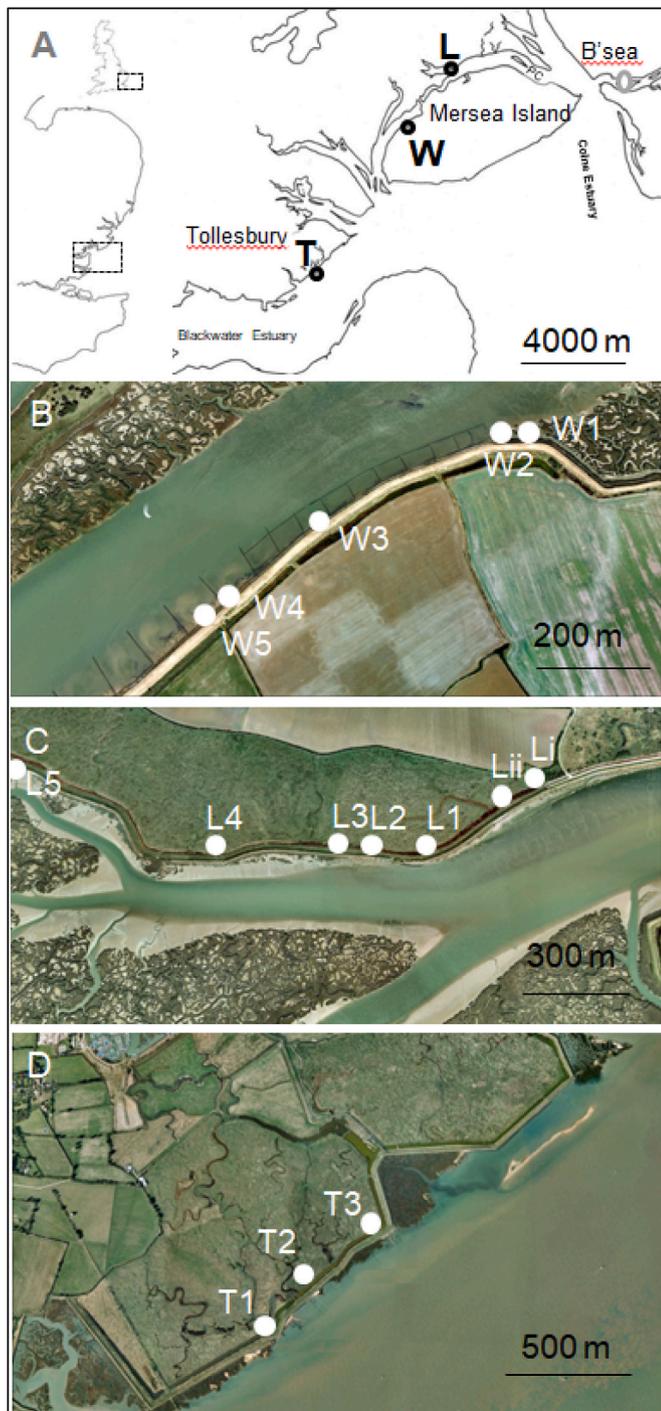


Fig. 1. (A) Location of the three sites in Essex, S. E. England, U.K. where salt marsh terraces were constructed in 2012. Sampled terrace sites and associated salt marshes at W, Wellhouse; L, Langenhoe, T, Tollesbury and salt marsh at Cinderly Island Brightlingsea (B'sea) (B–D): Images from Google Earth (2016) with location of individual terraces shown; (B) Wellhouse; (C) Langenhoe, (D) Tollesbury. See Table 1 for terrace description and GPS locations. Information taken from Google Earth (2023) Data IO, NOAA, US. Navy, NGA, GERCO).

variable number of mini-transects were conducted depending on the length of the terrace, with a minimum of 4 (Wellhouse) and maximum of 9 (Langenhoe, Tollesbury). Elevation data was also collected for the *Atriplex* zone of adjacent areas of natural marsh to allow comparison of the height of the terraces in relation to established salt marsh. Heights were converted to chart datum (CD) on the basis of surveyed height, actual tide height levels, live tidal gauge data from Harwich, U.K. and

published tide tables. The conversion from Chart Datum to Ordnance Datum based on Felixstowe, Suffolk, is -1.95 m.

The areas of clay-infilled terraces between the gabions and the bottom (toe) of the sea wall seaward slope were surveyed for vegetation cover using a standard NVC style $2\text{ m} \times 2\text{ m}$ quadrat. The number of replicates per terrace depended on terrace size, from 1 to 3 per terrace, with quadrats being a minimum of 2 m away from each other. The quadrats were assessed for percentage cover of individual plant species (except for the *Salicornia europaea* species complex, which is recorded as *Salicornia* spp.), cover of macroalgal mats and algal biofilms and bare ground. Following taxonomic revision, Common Cordgrass is now classified as *Sporolobus anglicus* (Peterson et al., 2014), but in this paper we use the widely accepted synonym *Spartina anglica* to facilitate reference to the wider literature. The Domin scale of cover/abundance was used for cover less than 4% (Rodwell, 2006). Due to the narrow width of some of the terraces, individual quadrats (size: $1\text{ m} \times 4\text{ m}$) were placed along the length of the terrace. Data for each terrace was paired with other site data, and then average plant community data created for each site in each sampling period. The undrained sediment shear strength (kPa) of terrace sediments (including root effects, Brooks et al., 2023) was measured using a pocket shear vane (Model 16-T0174, Zhuozhou Tianpeng Ltd. China). The vane was inserted into the sediment to a depth of 3–5 cm and rotated until the sediment failed. Three replicate measurements were taken in the centre of each $1\text{ m} \times 4\text{ m}$ quadrat. The flora and coverage in adjacent salt marsh sites was determined using NVC quadrats placed randomly within the *Atriplex portulacoides* zone of the marsh. This species is a characteristic feature of East Anglian salt marshes and forms where drainage is good but also helps accrete sediment (Steers, 1954). Sediment shear strength ($n = 3$) was determined in the centre of these quadrats. An index for exposure or average wave fetch (F) was derived for each of the three sites, as described in Cousins et al. (2017).

During 2017, additional measurements of sediment chlorophyll *a* (Chl *a*), water content and organic content (loss on ignition, or ash-free dry weight, AFDW) were taken from all the terraces and natural marsh sites to determine potential changes in substratum characteristics. In each quadrat paired minicores (diameter 2.84 cm^2 , top 5 mm of sediment surface, n given in Table 1) were taken from randomly selected areas. One sample from each pair was used for measurement of water content ($105\text{ }^\circ\text{C}$ for 24 h) and AFDW ($550\text{ }^\circ\text{C}$ for 1 h) in the top 5 mm of sediment. The second sample in each pair was frozen at $-20\text{ }^\circ\text{C}$, subsequently freeze-dried and analysed for Chl *a* concentration (conc.) according to Hanlon et al. (2006).

Figures and statistical analyses were created and conducted in the statistical environment R with packages dplyr, tidyverse and ggplot (R Core Team, 2022). Similarity in plant species composition and mean vegetation cover within terraces and controls were compared using Bray-Curtis dissimilarity nonmetric multidimensional scaling (NMDS) of the percentage cover data of samples using the package Vegan (Oksanen et al., 2022). Wilcoxon signed rank test was used to determine differences in species richness between marsh and terrace samples (R Core Team, 2022). Sediment shear strength data were compared using a 3-way ANOVA. Contour maps of terrace sediment height were derived from transect data, with contours of equal height interpolated using the package metR (Campitelli, 2021). The relationship between vegetative colonisation and sediment shear strength, and other relationships were investigated with non-linear least squares analysis. Stepwise multiple linear regression was applied to determine the effect of sediment bed height, time and plant and algal cover on plant colonisation. ANOVA was used to test differences between sites, time and treatments. Where data did not meet the requirements for normality, it was transformed, e.g. values of shear strength data were log-10 transformed.

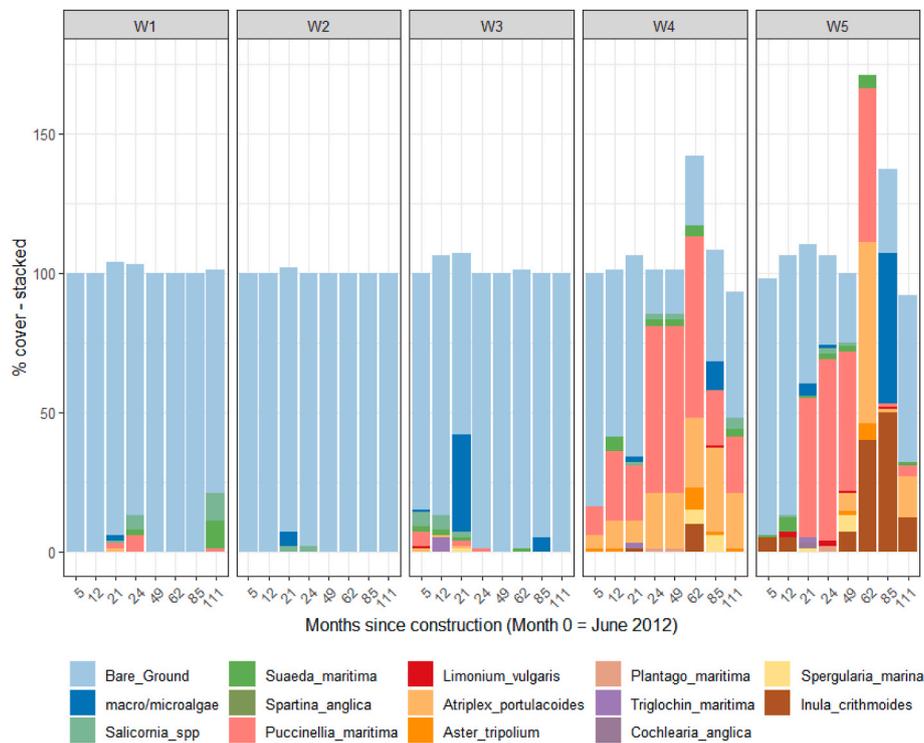


Fig. 2. Vegetation % cover and species composition (including bare ground, cover by macro- and microalgae mats, and halophyte plant taxa) of the sediment surface of five engineered sea wall terraces (W1–W5) constructed at Wellhouse, Essex, sampled from October 2012 (month 5 since construction) to August 2022 (month 111). Samples consisted of standard NVC 4 m² quadrats (n = 1) on each terrace at each sampling point.

3. Results

3.1. Changes in salt marsh flora and sediment properties on the terraces over 10 years

The establishment of salt marsh vegetation varied significantly between the 5 terraces at Wellhouse (Fig. 2). Terraces W1, W2 and W3 all showed initial colonisation and increases in total plant cover (up to month 24), but by 2022 (111 months), two terraces (W2 and W3) were non-vegetated and had suffered from significant sediment loss (Table 1, Fig. S1A). Plants colonised and became established within five months on Terraces W4 (Fig. 3A) and W5, with plant cover exceeding 100% cover by month 62 (2017) with up to 8 different taxa, including perennial species *Atriplex portulacoides*, *Aster tripolium* and *Inula crithmoides* (Fig. 2). The natural fringing salt marsh at Wellhouse was narrow but well established, with *Puccinellia maritima*, *Limonium vulgare*, *A. portulacoides* and *A. tripolium* the dominant species (Fig. 4). The plant cover on both W4 and W5 terraces had declined by 2020 and further declined by 2022 (months 85–111), when there was erosion of blocks of sediment occurring behind the stone gabions (Fig. 3B, Fig. S1B, Fig. S2A). Wellhouse had a low exposure index (0.05, Table 1, Fig. 1), but collapse of entire blocks of sediment also occurred in the natural fringing salt marsh fronting the sea wall during the period of this study (Table 1 Fig. S1C).

Terrace L-i at Langenhoe showed significant plant colonisation and establishment, with the flora dominated by *P. maritima*, *A. portulacoides* and the annuals *Suaeda* sp. and *Salicornia* sp. (Fig. 5). There was a high percentage cover of macro- and microalgal mats on L-i, and also on L-ii (which otherwise only supported annual species in month 49, Fig. 3C). L-ii had zero halophyte cover by 2022. Langenhoe terrace L1 was colonised by vascular plants in the first year but then lost almost all of its plant cover, and became dominated by macro- and microalgal mats (Fig. 3D). These mats were predominantly *Ulva*, and a mix of benthic diatom biofilms and mixed diatom-cyanobacterial mats (Figs. 5 and

3D). A similar failure to develop salt marsh also occurred on Terrace L2 and to a lesser extent on L3 (annual plant species establishing less than 10% overall cover, and some *Spartina anglica* colonisation by month 85), and extensive algal mats developed (Fig. 5). In contrast, Terraces L4 and L5, which were smaller, and surrounded by more established salt marsh, showed successful development of salt marsh, dominated by annual species, but also including perennials such as *P. maritima*, and *A. tripolium* (Figs. 3E and 5). The Langenhoe sites (particularly terraces L-i, L-ii, L1, L2, and L3) experienced a significant increase in grazing pressure after 2016 (month 49). A change in land ownership resulted in many more cattle and sheep being stocked in the adjacent grazing marshes, with the animals having access to the sea walls and fringing marsh. The natural salt marsh flora (initially dominated by *A. portulacoides*, *P. maritima* and *Limonium vulgare*, Fig. 4) was heavily grazed, with a decrease in perennial forbs and an increase in the percentage of bare ground present and abundant hoof damage (Fig. 3F, Fig. S1D). The remaining sward (mainly *P. maritima*) consisted of short, stunted plants (Fig. 3F). The impact of increased grazing was also reflected with the decline of vascular plants from month 49 onwards on terraces L-ii, L1, L2 and L3 (Fig. 5).

At Tollesbury, Terrace T1 was constructed around a convex bend in the sea wall, and showed no salt marsh development over the 10 year period (Fig. 6). The terrace supported macro- and microalgal mats, and became quite waterlogged, with standing water by 2020 (month 85) (Fig. 3G). Terrace T1 showed a consistent pattern of sediment erosion and collapse of the Essex-block bank during the period 2012–2017 (Figs. S1E and F). The toe of the sea wall at this site was not protected from 2017 to 2020. During 2021 the exposed surfaces of T1 and the face of the sea wall were reconstructed with a cover of polygon concrete blocks (Fig. S1G). Only a short (5 m) section of the original terrace remained in 2022, which consisted of bare clay and algal mats (Fig. 6). Terrace T2 supported a number of small patches of plant colonisation (*S. anglica* and *P. maritima*) and algal mats in the first two years, but coverage did not increase in the following years. From 2020 onwards,

Table 1
 Details of individual terraces sampled at three sites in the Blackwater-Colne estuary, Essex, U.K. Sediment water content and AFDW, s.e = standard error (n = sample size). Samples with the same superscript (A, B, C, etc) are not significantly different from each other (ANOVA).

Site	Code ^a	DMS Lat	DMS Long	Terrace length (m)	Exposure index (<i>F</i>) ^b	Period sampled	Sediment water content % (2017) mean ± s.e. (n)	Sediment AFDW content % (2017) mean ± s.e. (n =)	Notes from 2022 (month 111) survey
Wellhouse	W1	51°47'2.97"N	0°54' 50.4"E	6	0.05	2012–2022	26.58 ± 1.78 (5) ^C	8.78 ± 0.76 (5) ^C	Sediment block failure over 30% of berm, –40 cm
	W2	51°47'3.09"N	0°54' 49.77"E	4	0.05	2012–2022	29.32 ± 2.36 (5) ^C	9.43 ± 0.55 (5) ^C	Sed. block failure, clay pieces, –30 cm lower than gabion
	W3	51°47'25.29"N	0°54' 33.49"E	6	0.05	2012–2022	29.32 ± 3.40 (5) ^C	10.22 ± 1.63 (5) ^C	Only base clay, –50 cm below gabion baskets
	W4	51°47'25.28"N	0°54' 33.48"E	4	0.05	2012–2022	30.52 ± 3.70 (5) ^C	14.87 ± 1.83 (5) ^A	Sed. losses – 50 cm at each end, 40% terrace eroded
	W5	51°47'23.94"N	0°54' 27.57"E	4	0.05	2012–2022	41.53 ± 3.60 (5) ^C	17.13 ± 1.91 (5) ^A	Sed. losses behind gabion, - 40 cm, 50% terrace eroded
Wellhouse <i>Atriplex</i> zone salt marsh (sites adj. to above)						2016–2022	58.11 ± 1.29 (15) ^A	22.18 ± 1.05 (5) ^B	Sed. block failure of marsh along sea walls.
Langenhoe	L-i	51°48' 34.54"N	0°56' 30.48"E	14	0.13	2016–2022	51.09 ± 1.47 (10) ^A	19.18 ± 1.97 (10) ^A	gabion baskets corroded. Holes behind gabions
	L-ii	51°48' 33.32"N	0° 56' 26.67"E	11	0.13	2016–2022	42.93 ± 1.86 (10) ^B	13.68 ± 0.79 (10) ^A	continuous linear erosion behind gabion baskets
	L1	51°48' 28.89"N	0°56' 15.22"E	21	0.13	2012–2022	42.40 ± 2.86 (10) ^B	10.83 ± 1.15 (10) ^A	+40 cm accretion in areas of microbial mats
	L2	51°48' 29.28"N	0°56' 08.89"E	42	0.13	2012–2022	41.45 ± 2.41 (10) ^B	13.1 ± 1.0 (10) ^A	gabion baskets corroded, berm level with gabions
	L3	51°48' 29.30"N	0°56' 05.84"E	31	0.13	2012–2022	34.09 ± 2.47 (10) ^B	13.63 ± 0.70 (10) ^A	sediment level > gabion height, extensive mats
	L4	51°48' 28.77"N	0°55' 48.15"E	5	0.13	2012–2022	29.07 ± 2.90 (5) ^B	9.83 ± 0.64 (5) ^A	mainly bare clay and hoof marks, +10 cm sed.
	L5	51°48' 37.55"N	0°55' 18.54"E	6	0.13	2012–2022	32.31 ± 3.02 (5) ^B	12.74 ± 0.517 (5) ^A	Soft sediment accumulation up to seawall.
Langenhoe <i>Atriplex</i> zone salt marsh (sites adj. to above)				n/a	0.13	2016–2022	60.21 ± 3.37 (10) ^A	24.4 ± 2.95 (10) ^B	Increased <i>Spartina</i> in creek channel, esp. L3-L5.
Tollesbury	T1	51°45' 01.37"N	0°52' 03.65"E	52	2.0	2012–2022	26.32 ± 1.23 (10) ^C	7.17 ± 0.39 (10) ^C	rebuilt with blocks and concrete infill
	T2	51°44' 55.69"N	0°51' 51.91"E	46.5	2.0	2013–2022	22.57 ± 1.13 (10) ^C	7.53 ± 0.23 (10) ^C	50% eroded clay –20 cm level, exten. shell bank
	T3	51°45' 01.37"N	0°52' 03.65"E	30	2.0	2016–2022	19.30 ± 1.16 (10) ^C	8.55 ± 0.20 (10) ^C	Sediment 20–30 cm below gabion level
Tollesbury <i>Atriplex</i> zone salt marsh (sites adj. to above)						2016–2022	54.63 ± 3.93 ^A	17.5 ± 1.58 (15) ^B	Wet patches and a lot of shell and other debris

^a Terrace codes as given in Cousins et al. (2017) with new terraces added.

^b *F* = average wave fetch of location, see Cousins et al. (2017).

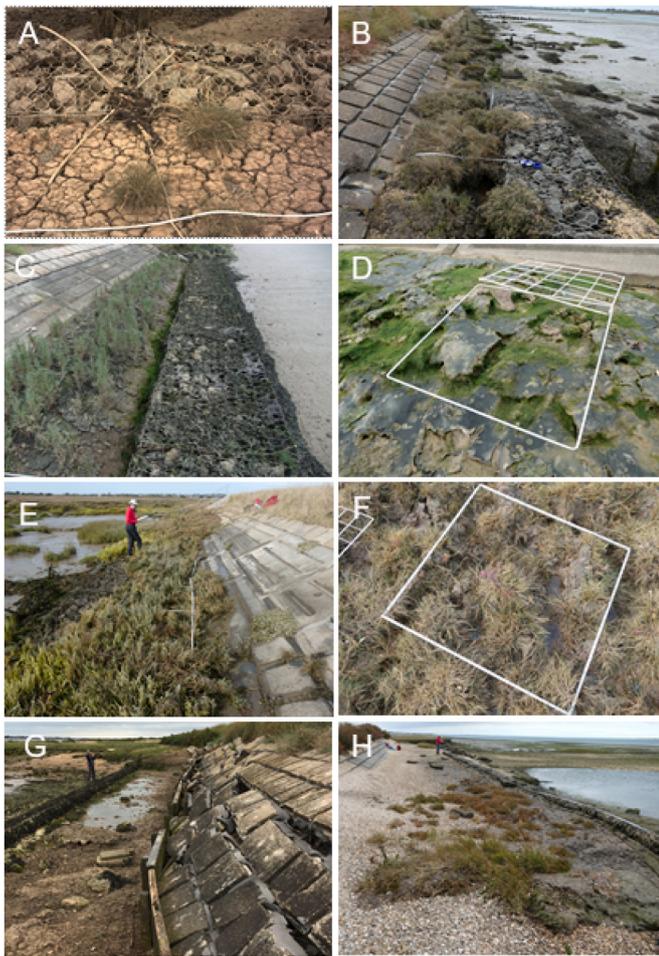


Fig. 3. (A) Wellhouse Terrace W4 five months after installation with colonisation by *Puccinella maritima*; (B) W4 Terrace surface at 111 months, halophytes established, evidence of sediment block failure on W4 and adjacent salt marsh; (C) Langenhoe gabion and terrace, Lii, 49 months after installation, with pioneer halophytes *Salicornia* spp and *Suaeda maritima* and microalgae, with *Fucus* sp. on gabion; (D) Langenhoe Terrace L1 in 2022, accretion of terrace with microalgal mats and *Ulva*, halophytes absent; (E) Langenhoe L5 terrace integrating into adjacent salt marsh, L5 colonised with a mix of vascular plants 111 months after creation; (F) salt marsh at Langenhoe in 2022 affected by grazing, hoof marks evidence of compaction, increased bare ground and changed species composition; (G) Tollesbury Terrace T1 85 months after construction, gabion intact but evidence of damage to terrace and seawall; (H) Tollesbury Terrace T3 in 2022 with shell bank covering toe of sea wall and terrace and *Spartina anglica* colonisation.

most of this terrace (>80%) was covered in a dense bank of shells (4–5 cm deep towards the back and ends of the terrace) consisting mainly of *Cerastoderma edule*. Terrace T3 was only surveyed from 2016. A number of patches of mainly *Salicornia* sp. became established at the south-west end of T3, along with mixed algal mats, but the majority of the surface area of this terrace remained bare clay and shell banks in the period 2016 to 2022 (Fig. 3H). The established salt marshes at the Tollesbury site were diverse, with a significant proportion of *S. anglica*, *P. maritima*, *A. portulacoides* and *A. tripolium* (Fig. 4, Fig. S1H). These marshes had areas of bare sediment, shell banks and small pools of water.

When constructed in 2012, the surface of the terraces at Wellhouse were between 6.0 and 6.5 m above chart datum, an equivalent height to that of the *Atriplex* zone in the adjacent natural salt marsh (Fig. 7A and B). Terraces W1–W4 showed a decrease in average sediment bed height between 2012 and 2017 (Fig. 7A). These changes in average height mask the existence of some areas of substantial sediment loss: during the first 2 years, three of the terraces at Wellhouse (W1, W3 and W4) exhibited

deep erosion features just behind the gabion cages and at the unprotected lateral edges (see Fig. S2A for W4). Terrace W5 showed only minor changes in surface elevation up to 2017 (Fig. S2B). Erosion continued at Wellhouse, affecting all of the terraces and the adjacent salt marsh fringe by 2022 (Figs. S1A, S1B, S1C, Table 1).

The Langenhoe and Tollesbury terraces were constructed lower in the tidal frame, between 5.4 and 6.0 m C.D. (Fig. 7A). This was 0.5 m lower than the *Atriplex* zone of the established salt marsh at the Langenhoe site, but equivalent to the height of the existing salt marshes at Tollesbury (Fig. 7B). Terraces L1, L4 and L5 at Langenhoe accumulated between 10 and 25 cm sediment (Fig. 7A) and increased in bed height over the period 2012–2017 (gaining sediment along the length of the terrace, with greatest accumulation adjacent to the sea wall, see Figs. S2C and S2D). Some of the longer terraces at Langenhoe were susceptible to lateral erosion occurring at the ends of the structures (e.g. L1, Fig. S2C). In contrast, at Tollesbury, an exposed site, the average sediment bed height of T1 and T2 decreased by almost 50 cm during this time (Fig. 7A, Fig. S1E), with erosion across the whole terrace, starting along the line between the clay infill and the gabions (Figs. S2E and F) with T1 showing the greatest losses of sediment by 2017 (Fig. 7A, Fig. S1F). This erosion continued through to 2020, and the terrace was rebuilt in 2021 (Fig. S1G).

Sediment shear strength was significantly different between sites for both salt marsh and terrace locations (ANOVA, type*site, $F_{2,496} = 62.5$, $p < 0.001$), with shear strength decreasing between 2016 and 2017, and then increasing in years 2020 and 2022 (type*year $F_{3, 496} = 3.28$ $p < 0.05$) (Fig. 7C). The sediment shear strength of the natural salt marsh was significantly higher than on the corresponding engineered terraces at both Wellhouse (W) and Langenhoe (L). The salt marsh sediments at Tollesbury (T) had significantly lower shear strength than sediments at the other three natural marsh locations (Fig. 7C, $F_{5,97} = 11.24$, $p < 0.001$). The sediment shear strength of terraces T1, T2 and T3 were significantly higher than that of the natural marsh at Tollesbury, and also significantly higher than the sediment shear strength of terraces W1 to W5 and L-i to L5 from 2016 to the end of the study in 2022 ($F_{2,399} = 36.9$, $p < 0.001$).

3.2. Factors influencing the establishment of salt marsh flora on the terraces

Ordination (NMDS) revealed major differences in the plant and algal community composition between terrace and established salt marsh samples in the early months of the project (Bray–Curtis NMDS stress value 0.131, Fig. 8). Terrace and reference marsh sites were significantly separated along MDS axis 1 (Kruskal–Wallis of sample type axis scores, $p < 0.001$) for all three sites (Fig. 8). Positive MDS axis 1 scores indicated high percent cover of macro- and microalgae and bare ground (applies to many of the terrace sites, particularly early in the study) and negative MDS1 scores indicated samples with higher cover by vascular halophyte species (natural salt marsh samples and terrace samples with increased halophyte colonisation in later months). There was no significant separation between natural marsh and terrace samples on MDS axis 2. Positive scores on MDS2 represent increased macro- and microalgae cover, *Salicornia* sp. and *Spartina anglica* cover, indicative of a lower marsh community, with negative MSD2 values indicating increased abundance of *Puccinellia maritima*, *A. portulacoides*, *I. crithmoides*, *Triglochin maritima*, and *Cochlearia anglica* (a set of middle marsh species whose % cover was correlated with each other, Fig. S3). There was a trajectory of increasing similarity with time between the halophyte communities on terraces W4, W5 and the Wellhouse salt marsh samples (Fig. 8). A similar trajectory on increasing compositional similarity between the developing halophyte vegetation on terraces L3, L4, L5 and L-i, and the adjacent salt marsh was present, with overlap of these sample groups in months 62, 85 and 111 due to the changed grazing regime. There was no trajectory of salt marsh development on the terraces at Tollesbury, shown by the substantial separation in the MDS between the

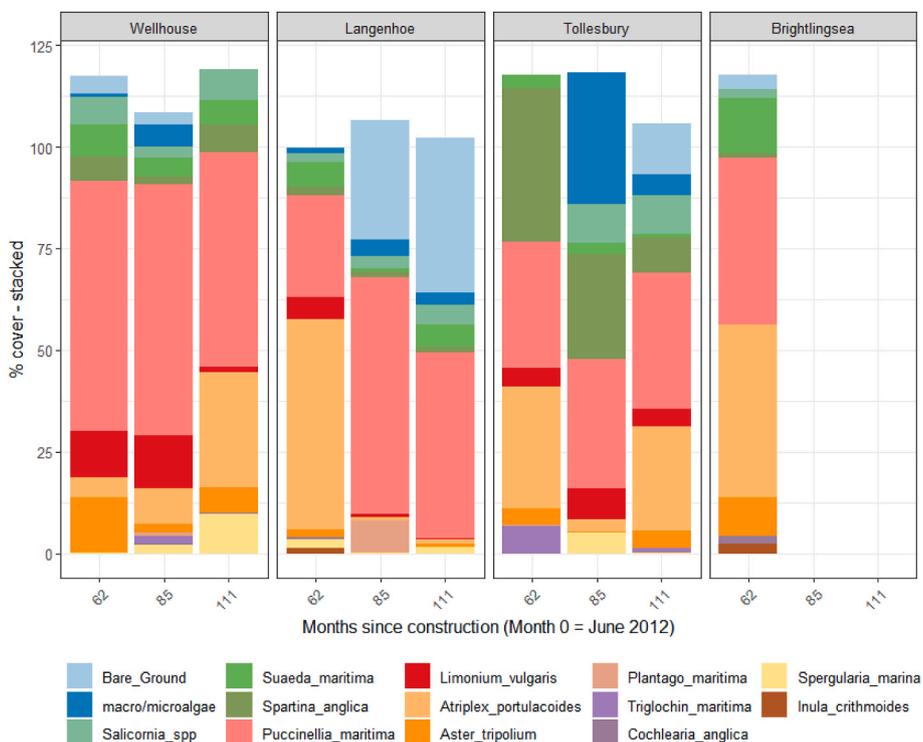


Fig. 4. Vegetation % cover and species composition (including bare ground, cover by macro- and microalgae mats, and halophyte plant taxa) determined using multiple 4 m² quadrats (averages, n between 2 and 7 at each sampling event) in the *Atriplex* zone of natural salt marsh at four estuarine coastal sites in Essex, U.K., sampled in 2017, 2020 and 2022 (months 62, 85 and 111 respectively).

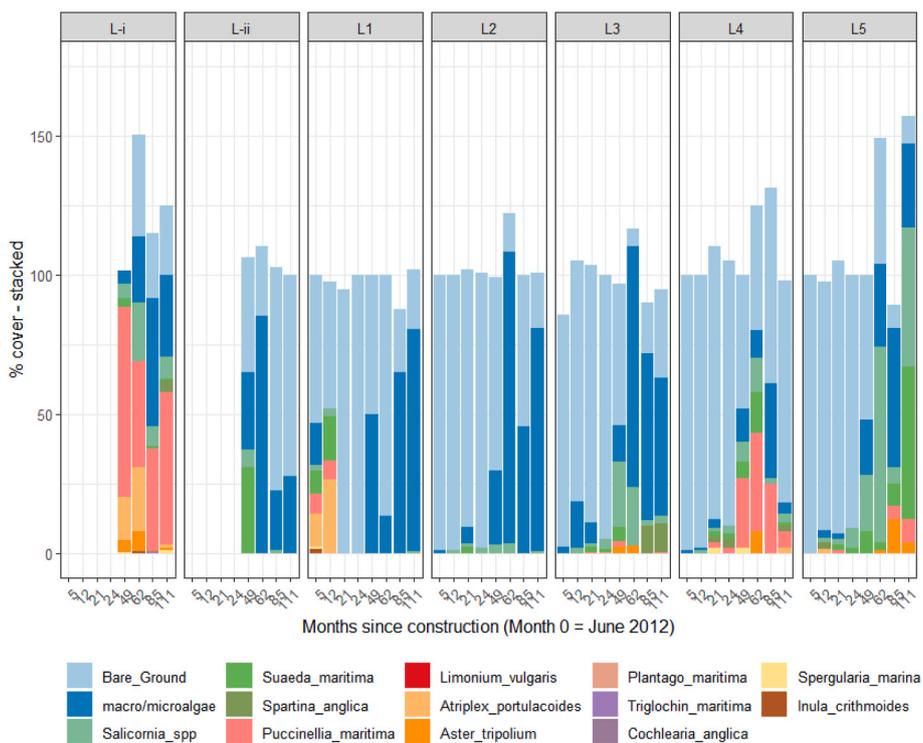


Fig. 5. Vegetation % cover and species composition (including bare ground, cover by macro- and microalgae mats, and halophyte plant taxa) of the sediment surface of seven engineered sea wall terraces (L-i, L-ii and L1 to L5) constructed at Langenhoe, Essex, sampled from Oct. 2012 (month 5 since construction) to August 2022 (month 111). Samples consisted of standard NVC 4 m² quadrats, (averages, n = 3 on L-i, L1, L2, L3, = 2 on L-ii, and = 1 on L4, L5) on each terrace at each sampling point.

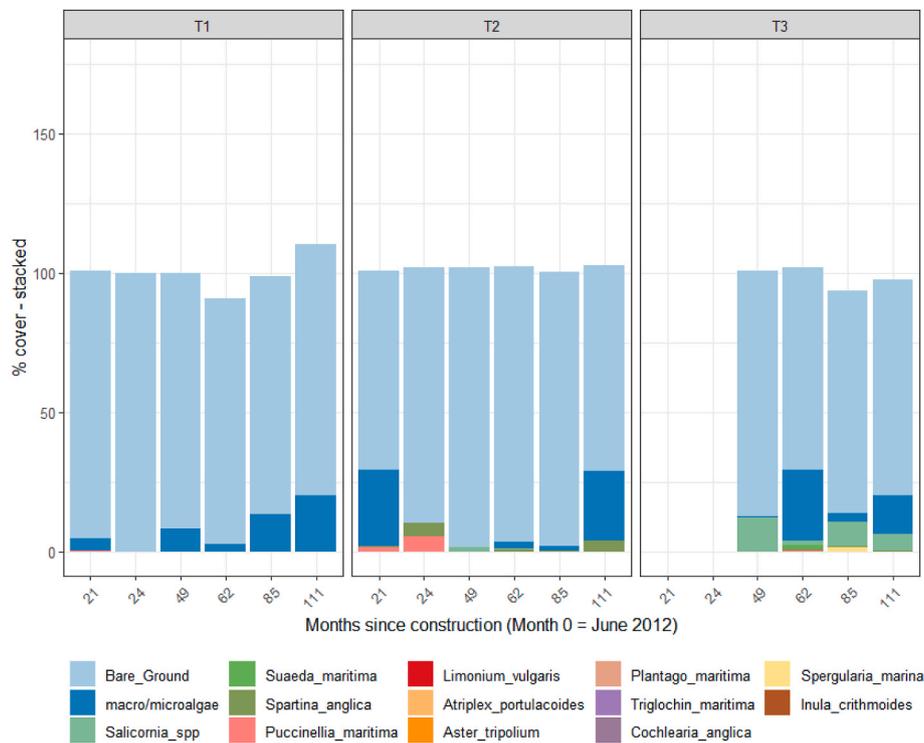


Fig. 6. Vegetation % cover and species composition (including bare ground, cover by macro- and microalgae mats, and vascular saltmarsh plant taxa) of the sediment surface of three engineered sea wall terraces (T1, T2, T3) constructed at Tollesbury, Essex, sampled from April, 2014 (month 21 since construction) to August 2022 (month 111) (T3 only from 2016). Samples consisted of standard NVC 4 m² quadrats, (averages, n = 3 on T1, T2, T3) on each terrace at each sampling point.

terraces and the adjacent salt marsh samples. The natural salt marsh at Tollesbury had higher amounts of *Spartina* and *Salicornia* (positive NMDS2 scores) than the salt marshes at Langenhoe or Wellhouse (Fig. 8).

The number of species of vascular plants on the terraces was positively correlated with the total plant cover ($r = 0.83$, $p < 0.001$), and with the amount of cover of *P. maritima* and *Atriplex portulacoides* ($r = 0.85$, 0.65 , $p < 0.001$ respectively, Fig. S3). There was a significant relationship ($F_{1,254} = 1261$, $p < 0.001$), applicable to both the natural salt marsh sites and the terraces, between the species richness of salt marsh plants and total plant cover (Fig. 9A). The natural salt marshes surveyed contained between 4 and 9 species of vascular plant per 4 m² quadrat (average all sites and samples = 6.85) compared to an average of 1.7 (min = 0, max = 8) for all Terraces and times. Some terraces at Wellhouse and Langenhoe attained plant species richness values equivalent to their adjacent salt marsh (Fig. 9A), but generally plant richness was significantly lower on the terraces compared to adjacent marshes ($W = 281.5$, $p < 0.001$). Considering only the Terraces, sediment bed height was significantly correlated with total plant cover ($r = 0.36$, $p < 0.001$, Fig. S3), and with the percentage cover of *P. maritima* and *Limonium vulgare* ($r = 0.42$, 0.33 , $p < 0.001$ respectively) and *A. portulacoides* ($p < 0.05$, Fig. S3). Total plant cover was positively related ($F_{1,37} = 24.7$, $p < 0.001$) with sediment water content (Fig. 9B) across all of the terraces and natural salt marsh samples (2017 data only). The natural salt marshes had significantly higher ($F_{2,30} = 61.2$, $p < 0.001$) sediment water content (55–60% water content) than the engineered terraces (except L1, L2 and L3), with the greatest differences for those terraces consisting of basal clay sediment (e.g. T1, T2, T3, 20–25% water content; W1, W2, W3, 26–30% water content, Table 1). Sediment water content was positively correlated ($r = 0.39$, $p < 0.05$) with sediment Chl *a* conc. on the terraces (Fig. 9C), and in the natural salt marsh. Terrace sediment Chl *a* concentrations were also positively correlated ($r = 0.44$, $p < 0.05$) with the % cover of macro- and microalgal mats. Fringing salt marsh had significantly higher sediment

organic content (%AFDW) than the terraces at all sites (Table 1, $F_{2,30} = 25.9$, $p < 0.001$), with Langenhoe salt marsh having higher %AFDW than both the Wellhouse and Tollesbury salt marshes (Table 1).

There was a significant non-linear relationship (LSD, $p < 0.001$) between % plant cover and sediment shear strength for terraces with $\geq 1\%$ plant cover (Fig. 9D), with an optimal shear strength of ≈ 30 kPa for maximal halophyte cover. There was a similar significant relationship (LSD, $p < 0.01$) between plant richness and sediment shear with an optimal shear strength of ≈ 35 kPa for maximal plant species richness, but there was significant variability around this relationship (data not shown). There was no significant correlation between terrace sediment bed height and sediment shear strength, but sediment height change was negatively correlated with sediment shear strength ($r = -0.32$, $p < 0.01$) on the terraces.

Stepwise regression (using both forward and backward selection) found that the most significant model to predict total plant cover on the engineered terraces required data on sediment bed height and time (both positive factors) and cover of micro- and macroalgae, and sediment bed height changes (all negative).

Total plant cover = $39.1(\text{height in m above CD})^{***} + 0.44(\text{time, months since construction})^{**} - 0.06(\% \text{ cover algal mats}) - 1.3(\text{change in sediment bed height m}) - 223.9^{***}$.

($F_{4,130} = 7.9$, $p < 0.001$, r^2 (adjusted) = 20.0%; significance of t-value of individual coefficients $p < 0.01^{**}$, $p < 0.001^{***}$).

The best model to predict plant species richness on the terraces utilised data on sediment bed height and time (both positive factors) and cover of micro- and macroalgae, sediment shear strength, and sediment bed height changes (all negative).

Plant species richness = $5.3(\text{height in m above CD})^{***} + 0.02(\text{time, months since construction}) - 0.005(\text{sediment shear strength, KPa}) - 0.007(\% \text{ cover algal mats}) - 0.18(\text{change in sediment bed height m}) - 29.6^{***}$.

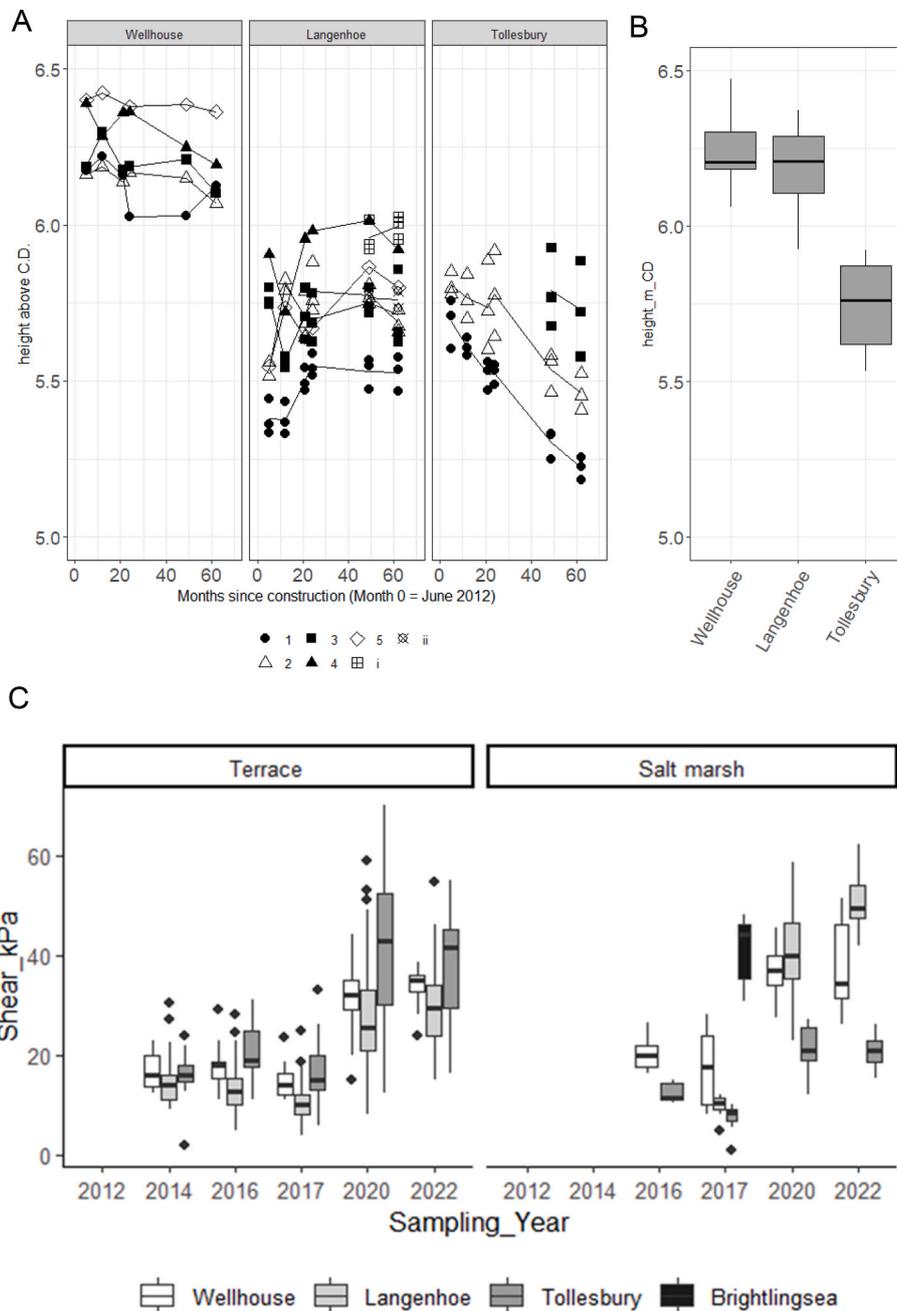


Fig. 7. (A) Changes in sediment bed height (m relative to chart datum C.D.) on 15 engineered terraces and (B) the height of the *Atriplex* zone in adjacent salt marsh, at Wellhouse, Langenhoe, and Tollesbury in the Colne-Blackwater estuary complex, Essex, between 2012 and 2017. (C) Sediment shear strength (kPa) of the terraces and salt marsh at the same sites measured between 2014 and 2022.

($F_{5,72} = 15.5, p < 0.001, r^2$ (adjusted) = 48.4%; significance of t-value of individual coefficients $p < 0.001^{***}$).

4. Discussion

The expectation during the design phase was that the terraces would be colonised and develop a relatively extensive salt marsh flora. This would provide ecosystem services in the form of habitat provision and wave attenuation, protecting the sea wall from erosion. From an engineering perspective, the installed gabion terraces had a 10-year design life and, with the exception of T1, they all successfully protected the sea wall defence from further deterioration, indicating that the gabion terraces were a cost-effective repair (Table 2). In terms of establishing salt marsh, halophyte colonisation was influenced by tidal height and

sediment characteristics. Early settlement of pioneer species did not always lead to increasing plant cover, but could also lead to algal mat dominance or losses of sediment and erosion. Using Water Framework Directive criteria for salt marsh percent cover of suitable intertidal (SMAi, Water Framework Directive, 2014), 7 of the 15 terraces generated salt marsh ranging from high to poor quality status in 2022 (Table 2). Including terraces with microbial mats and shell banks, then 12 of the 15 terraces provided biological habitats after 10 years, where a default engineered solution would have been an abiotic concrete block wall.

Initial colonisation of the terraces was relatively fast (Cousins et al., 2017), mainly with pioneer successional species such as *Salicornia* and *Suaeda maritima*. However, after 24 months since construction, none of the sites had accumulated sufficient cover of plants to provide any

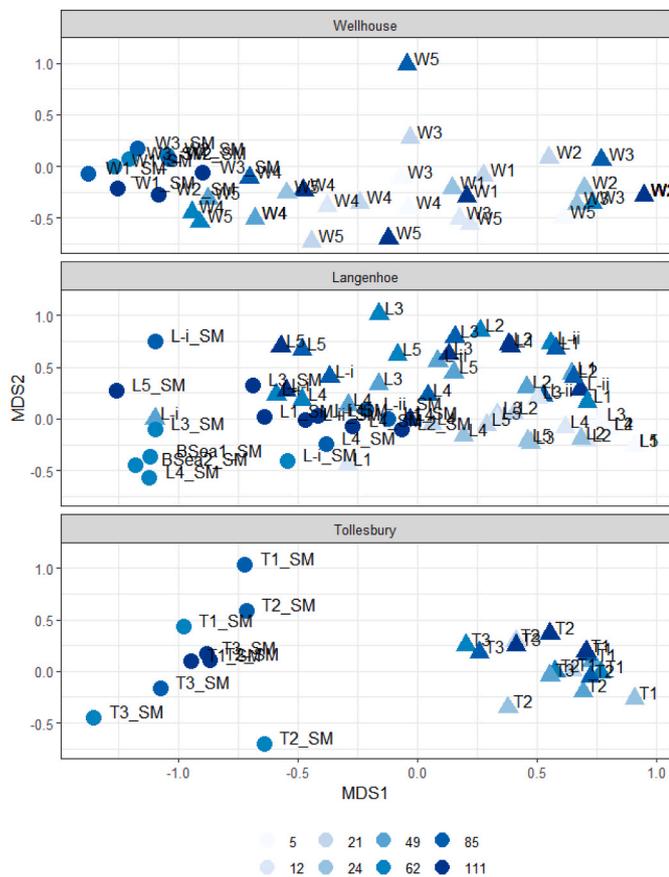


Fig. 8. NMDS plot of sample scores (MDS1 and MDS2) of individual terrace (triangles) and salt marsh (circles) plant community composition (% cover, including bare ground and algal mats) at three locations in the Colne and Blackwater estuaries, Essex, U.K., sampled at eight times (colours) over a 111 month period (2012–2022). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

potential wave attenuation or habitat provision services. The only terraces to exceed 50% plant cover by 24 months after construction were W4 and W5, both of which had significant colonisation by *P. maritima*. Germination and growth of *Salicornia* has been recorded in the early months of a sediment-filled managed realignment scheme at Allfleet's Marsh, Wallasea Island (Kadiri et al., 2011) and at other managed realignment sites in Essex (Underwood, 1997; Wolters et al., 2005a; Garbutt et al., 2006). However, initial *Salicornia* and *Suaeda* colonisation of the terraces did not necessarily lead to a greater subsequent plant cover. Terraces followed one of three different trajectories over the 10 year period; (i) the development of a salt marsh flora, with 4 of the 15 terraces having >25% cover of desired halophyte species by the end of the study (ii) the development of extensive algal mats which increased in extent over the 10-year duration of this study (both trajectories i and ii included sediment accretion), and (iii) exposure and erosion of the infilled clay, leading to degradation of the terraces.

4.1. Development of salt marsh flora (trajectory i)

The vegetative recruitment and resulting zonation of salt marshes is affected by a variety of inter-correlated environmental variables (Davy et al., 2011; Spencer et al., 2017; Mossman et al., 2012a, 2020). Height within the tidal frame is a significant variable affecting salt marsh vascular plant zonation (Crooks et al., 2002; Davy et al., 2011; Mossman et al., 2012a; McMahon et al., 2023). Elevation affects the duration and frequency of tidal inundation which can alter the community composition of salt marsh halophyte plants (Davy et al., 2011). On this basis, the

Wellhouse terraces should have been most successful in developing a halophyte flora, though this was only partly the case. On terraces undergoing halophyte colonisation, plant cover and species richness matched that of the adjacent salt marsh after between 6 and 10 years, timescales similar to other studies (Wolters et al., 2005a, 2005b; Garbutt et al., 2006; Mossman et al., 2012b). As halophyte plant cover increased, so did species richness, therefore the succession of salt marsh plant assemblages was clearly evident (see Fig. 9A). 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., 2005a; Dausse et al., 2008). There was no evidence for shortage of propagules due to the presence of adjacent salt marsh at all sites (Wolters et al., 2005a; Mossman et al., 2012a), such that colonisation was fast. Herbivory, particularly on seeds and seedlings by *Hediste (Nereis) diversicolor*, may slow the process of colonisation (Paramor and Hughes, 2004). However, in the current study there was no evidence for extensive invertebrate colonisation on any terraces so it is unlikely that invertebrate herbivory was a significant issue holding back salt marsh development.

The existing adjacent marshes in this region of Essex, dominated by *Atriplex portulacoides*, contained an average of 6.85 halophyte species per 4 m². This level of species richness is similar with that for natural marsh in south Essex (8 species, Kadiri et al., 2011; Ford et al., 2016; McMahon et al., 2023). Higher species richness is found when upper salt marsh zones are included (Wolters et al., 2005b; Garbutt et al., 2006) but at our sites, reclamation and coastal squeeze meant that these upper shore halophyte communities were not present (Gilbertson et al., 2020). The NMDS revealed classical halophyte species zonation, with *Aster*, *Limonium* and *Inula* present on the highest sites (e.g. Wellhouse) and lower sites (e.g. Tollesbury) supporting a higher abundance of *S. anglica* on both the terraces and marshes, with *Spartina anglica* growing on the Langenhoe terraces. *Spartina anglica* is a lower marsh species often growing below MHWN (Garbutt et al., 2006; McMahon et al., 2023), and its pattern of occurrence fits with the tidal elevation of the Langenhoe and Tollesbury sites. Both *Puccinella* and *Atriplex* are dominant low to mid salt marsh species, (NVC codes SM13 and SM14 respectively, Rodwell, 2006) in the Essex estuaries (Fig. 4) and *Atriplex* is an important plant species promoting marsh productivity and sediment carbon stocks (Ford et al., 2019; McMahon et al., 2023). Achieving a mix of these NVC communities could be considered a successful end point for terrace salt marsh development, with Terraces W4, W5 and L4 approaching this point after 10 years. The other vegetated terraces L3, L5, had halophyte communities more typical of lower tidal elevations (*Spartina*, *Salicornia*).

Key factors affecting the colonisation of the terraces by salt marsh plants were height in the tidal frame, time, sediment strength, the presence of algal mats and loss of sediment. Height within the tidal frame is also a significant variable affecting sedimentation (Marion et al., 2009), with greater potential sedimentation lower in the tidal frame (due to more frequent tidal cover). The Langenhoe terraces were lower than the adjacent salt marsh and significant sediment accretion associated with increasing plant cover was measured on L4 and L5 in particular. Plant cover also can promote marsh accretion through a reduction of water flow and trapping of sediment on leaves enhancing sedimentation (Mason et al., 2003; Reef et al., 2018), though this is not necessarily a linear relationship, with differing species composition and seasonality important controls of sedimentation (Reef et al., 2018).

Terrace height alone was not sufficient to explain the differences in salt marsh development. Many studies have shown that elevation covaries strongly with sediment redox potential, and redox potential is a strong selective factor for halophyte colonisation and growth (Davy et al., 2011; Mossman et al., 2012a, 2020; Spencer et al., 2017). More open, structured sediments, containing roots, pore spaces and organic carbon promote sediment drainage, oxygen penetration and biogeochemical processing (Spencer and Harvey, 2012; Spencer et al., 2017). Species like *Atriplex* and *Suaeda* need oxygenated sediment with positive

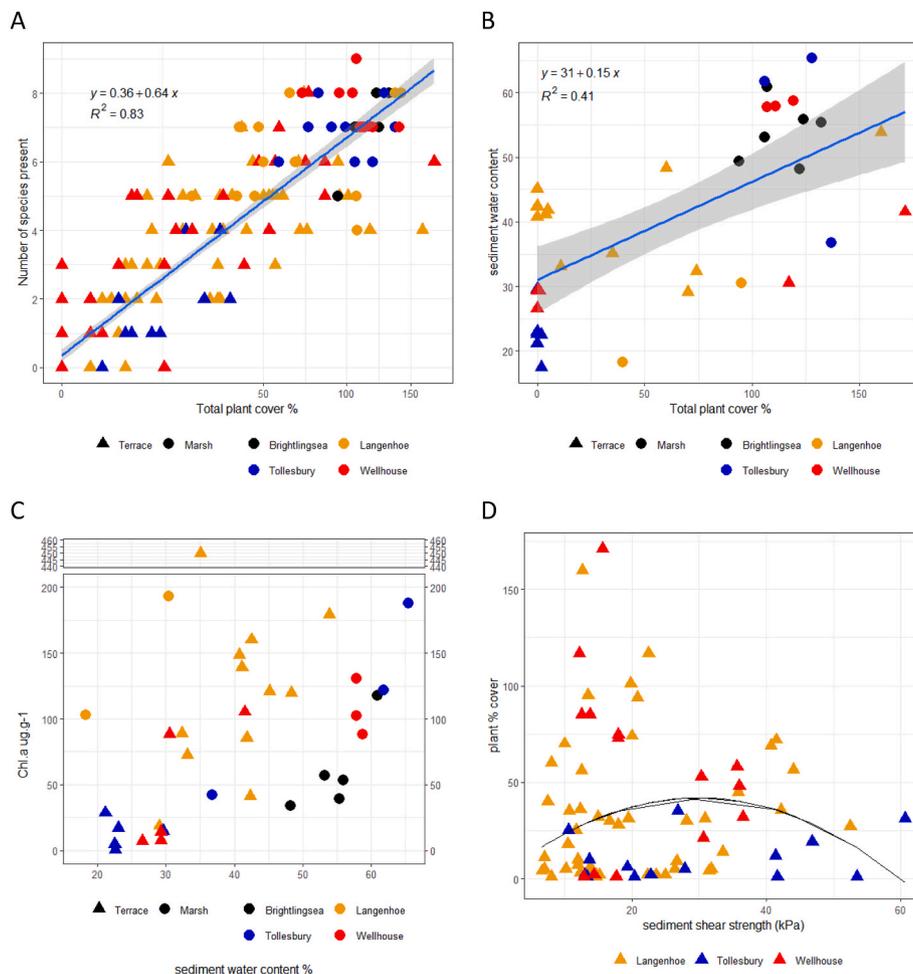


Fig. 9. Relationship between (A) total plant cover and species richness on terraces and natural salt marsh (note X axis square-root transformation), (B) total plant cover and sediment water content, (C) sediment water content and sediment Chl a content (both 2017 data, note split Y axis), and (D) sediment shear strength (kPa) and plant species richness, on terraces and natural salt marsh at the four locations studied between 2012 and 2022.

Table 2

Summary of outcomes for 15 engineered salt marsh sea wall terraces constructed at three sites in the Blackwater – Colne estuary complex, Essex, U.K., 10 years since installation (2012–2022): criteria; continued provision of protection of existing sea wall, generation of salt marsh habitat (scored using WFD TraC criterion, SMAi *Saltmarsh extent as proportion of the intertidal*), generation of alternative habitat.

Location ^a	Sea wall protection	Salt marsh quality ^b category	Other habitat provision number
Wellhouse (n = 5)	5/5	good	2
		bad	3
Langenhoe (n = 7)	7/7	high	2
		moderate	2
		bad	3
Tollesbury (n = 3)	2/3	poor	1
		bad	2

^a See Fig. 1.

^b SMAi categories (% cover): high = ≥ 50 –100, good = ≥ 25 –< 50, moderate = ≥ 10 - <25, poor = ≥ 5 - <10, bad = 0–5, (Water Framework Directive – United Kingdom Advisory Group (WFD-UKTAG) 2014).

redox values, while *Salicornia* and *Puccinellia* can tolerate low redox (anaerobic) conditions (Steers, 1954; Davy et al., 2011). Compacted clay sediments have reduced pore spaces, resulting in poor drainage and anaerobic conditions (Spencer et al., 2017). Though we did not measure

sediment redox, sediment bulk shear strength is partly a measure of compaction, and we observed an optimum shear strength for plant richness of approximately 35 kPa. Shear strength on the Terraces and in the adjacent marshes was higher than those measured in north Norfolk marshes (mean 10 kPa, Davy et al., 2011), but similar to values obtained from salt marsh at Tillingham on the Essex coast south of the Blackwater estuary (Evans et al., 2022; Brooks et al., 2023). Salt marsh sediment shear strength measured by shear vanes can be highly variable, and is affected by multiple variables, including sediment grain size, water content and the presence of plant roots (Evans et al., 2022; Brooks et al., 2023). The compacted clays on the Tollesbury terraces had much higher shear strength than the adjacent vegetated salt marsh, and were not colonised by halophytes. Clay is preferred by engineers for sea wall repairs, due to its high strength, but this can prevent successful colonisation of halophytes. Enhancement of sediment shear strength is also provided by halophyte vegetation, due to an extensive network of plant roots (Evans et al., 2022). Though similar shear strength values were measured between terraces and marsh locations in 2017, sediment water content was significantly higher in the salt marshes. This reflects biostabilisation by plant roots, resulting in increased sediment shear in sediments with higher water content (Watts et al., 2003; Gillen et al., 2021; Evans et al., 2022). It was noticeable that terraces that accreted sediment and were colonised by plants often had higher water content and lower sediment shear strength. It would be expected that shear strength would increase over time on those terraces as the halophyte root matrix becomes more established.

4.2. Development of algal mats (trajectory ii)

An alternative route for terrace development was the colonisation by extensive *Ulva* (macroalgae) and micro-algal biofilms. These were especially prevalent on some of the Langenhoe and Tollesbury terraces which had lower profiles in the tidal frame (Fig. 7). The development of macroalgal biofilms is an increasing problem in the Essex estuaries in the last decade, with macroalgae smothering seagrass meadows on mudflats (Gardiner, 2022). The proliferation of green algal mats is closely linked to eutrophication from agricultural fertiliser run-off, input of sewage plant effluent and likely combined with an increasingly hot spring and summer climate which may be responsible for the increase in macroalgal cover (Wasson et al., 2017; Gardiner, 2022). Similar to the possible fragmentation effect on seagrass (Gardiner, 2022), it is probable that macroalgal biofilms smother (cutting off light penetration and creating anoxic conditions) and disintegrate salt marsh plant species on the terraces and prevent establishment of halophytes (Wasson et al., 2017; Hosier, 2018).

Microalgal biofilms and cyanobacteria mats grew extensively on a number of terraces and were also present on the natural marsh sediments. Microalgae are recognised as early stabilisers of sediments that can facilitate halophyte settlement (Coles, 1979), and a set of characteristic epipellic and epiphytic species are found on salt marsh sediments where the plant cover allows for light penetration (Zong and Horton, 1998; Underwood, 1997; Redzuan and Underwood, 2020). Microalgal biofilms and mats promote sediment accretion by trapping sediment particles between their filaments (cyanobacteria) or through the secretion of mucilaginous extracellular polymeric substances (EPS), mainly by diatoms (Stal, 1995, 2010; Underwood and Paterson, 2003). The combination of sediment trapping and the presence of biofilm matrices (Chl a conc.) were correlated to increased sediment water content (see Fig. 9C, Fig. S3). Microalgal biofilms were abundant in the natural marshes of Tollesbury, which were wetter and had a more open plant sward including *Spartina* (Fig. S1H); similar to the salt marshes of the U. S. eastern seaboard (Hosier, 2018). Microbial mats are also resistant to desiccation (McKew et al., 2011; Stal et al., 2019), and cyanobacterial mats grew extensively on the infilled borrow-dykes containing compacted sediments at the Northey Island managed realignment site between 1992 and 1995 (Underwood, 1997). The Northey Island study found a significant negative relationship between microalgal mat and halophyte cover (Underwood, 1997), similar to the pattern observed on the terraces. High rates of primary production in microalgal biofilms stimulates bacterial secondary production, resulting in anaerobic and highly-reduced conditions just below the mat surface (Underwood et al., 2022). Sulphide rich sediments were often observed on the terraces with extensive and thick microalgal mats. Such conditions would lead to the exclusion of redox sensitive halophytes, such as *Suaeda* (Davy et al., 2011; Mossman et al., 2020). Progressive loss of halophyte cover and increasing algal mat cover was seen during the 10 year study particularly on the Langenhoe terraces. Despite the absence of halophyte vegetation, the presence of micro- and macroalgal mats did result in sediment accretion, often with a lower sediment shear strength, such that the terraces still maintained their sea defence function. While it was not possible to predict the alternative state of the terraces at the onset of the terrace trial it appears that eutrophication and macroalgal biofilm development is a serious problem for the establishment of salt marsh flora on blue-green infrastructure installations. Future research to determine the conditions which drive a change from algal mats into salt marsh cover is needed.

4.3. Exposure and erosion of the infilled clay, leading to degradation of the terraces (trajectory iii)

The terraces were initially constructed with a clay backfill, placed flush with the surface of the stone filled gabion basket. These surfaces did not erode initially and began to recruit vegetation. However, there

was evidence of desiccation breccia at Wellhouse and Tollesbury (particularly T1) from early in the study, and where there was little vegetation; the infilled clays became hard and polished, and exposed to weather. Sparse halophyte cover can actually increase erosion by disturbing water flows across the surface, and the sweeping action by vegetation can physically prevent seedling establishment (Widdows et al., 2008; Reef et al., 2018). Though the Wellhouse terraces W1, W2 and W3 were placed high in the tidal frame such that elevation should not have been an issue for plant colonisation, they were only very sparsely colonised by salt marsh species, and despite high shear strength of the clay, lost physical integrity by 2017 (62 months). There was limited salt marsh adjacent to the terraces at Wellhouse which led to sediment block failure and erosion at the terrace ends. Given the small size of the Wellhouse terraces, this significantly reduced the sediment volume behind the gabion baskets that could be colonised, particularly as they were on an eroding site. The positive outcome is the seawalls were protected for 10 years at Wellhouse and at the end of the study the gabions were intact and most of the terraces were still in place, despite having lost a large proportion of the clay (W1, Fig. S1B).

Sediment shear strength increased on both terraces and marshes from circa 20 kPa between 2014 and 2017, to nearer 40 kPa in 2020–2022; values in the same range as for other vegetated salt marsh sediments (Watts et al., 2003; Gillen et al., 2021; Evans et al., 2022). Sediment water content and un-drained sediment shear strength affect the capacity for salt marsh to resist erosion (Crooks et al., 2002; Gillen et al., 2021). Davy et al. (2011) recorded a median sediment shear strength of 10 kPa (range 1–60 kPa) four years after a marsh realignment (MR) in Brancaster, Norfolk U.K., similar to the early periods of our study and to values from Tollesbury MR site 6 years after realignment (Watts et al., 2003). As halophytes become established and their roots create a network within the sediment, the relationships between sediment shear strength, sediment consolidation, water content and erodability become complex (Evans et al., 2022; Brooks et al., 2023). Lower values would indicate accumulation of deposited sediment that have not yet consolidated or been strengthened by root masses. The terraces in this study contain both the original clay infill, plus any new accreted sediment on top of that. The increase in sediment shear strength over the period of study was also mirrored on the existing salt marshes. Hot dry summers such as experienced in 2018 and 2022, coupled with summer periods of reduced tidal ranges limiting periods of tidal immersion (Smith et al., 1998; McKew et al., 2011) and a lack of plant cover to shade the surface, may have contributed to desiccation breccia on the terraces and increases in sediment shear strength. Increased temperatures (Christidis et al., 2020), droughts and marine heatwaves are having serious negative impacts on coastal ecosystems (Shields et al., 2019; Adams, 2020), and present a risk to establishing salt marsh on sea wall terraces as blue-green infrastructure.

At Tollesbury, shell banks had become established on the terraces and may have protected the clay substratum from desiccation. Halophyte seeds/propagules have to be able to successfully germinate and penetrate the sediment which is not possible when a shell bank develops. It's possible that the shell banks may prevent the colonisation of the desired salt marsh vegetation but may themselves protect the flood defence from wave action if they are maintained *in situ*. Terrace T1 supported no shell cover and the sea wall behind was subject to erosion and had to be repaired, whereas terraces T2 and T3, with shell cover, maintained their structural integrity for the full 10 year period.

An additional factor contributing to terrace failure was water ponding on Tollesbury T1. Erosion of the sediments across the surface of T1 resulted in the gabion edge acting as a barrier and water ponding on the site. Waterlogging impairs halophyte plant growth, particularly through its impact of oxygen penetration and inducing low redox potential (Wolters et al., 2008; Garbutt et al., 2006; Kadiri et al., 2011). The lack of halophyte colonisation reduces the development of an open sediment structure containing both organic matter and pore spaces that can promote drainage (Spencer and Harvey, 2012; Spencer et al., 2017).

Developing a natural profile of sediment properties (texture, biogeochemical conditions, and drainage) is clearly necessary for successful salt marsh colonisation (Steers, 1954; Wolters et al., 2005b; Mossman et al., 2012a; Spencer and Harvey, 2012; Spencer et al., 2017).

4.4. Other considerations

The detailed survey in 2017 found that the organic content (measured as AFDW) of the terraces was lower than that of adjacent marshes. The natural marshes had AFDW values between 17 and 25% loss on ignition, similar to other marshes in the Essex estuaries area (Kadiri et al., 2011; Spencer et al., 2017). The terraces had lower sediment carbon content, lowest on the eroded clay sediments and highest where vegetation was establishing. Organic content of cyanobacterial mats at Northey was between 13 and 20% AFDW (Underwood, 1997). Other salt marsh recreation schemes find similar, lower organic carbon values, (Kadiri et al., 2011; Davy et al., 2011; Mossman et al., 2012a), except where existing terrestrial vegetation remained on the site (Underwood, 1997). The trajectory for new locations to reach a comparable salt marsh organic carbon content appears to be greater than 20–40 years (Burden et al., 2013, 2019), and could rely on the establishment of particular salt marsh species, e.g. *Atriplex portulacoides* (McMahon et al., 2023).

A more significant and unexpected factor impeding salt marsh development on the terraces was grazing and trampling of newly established plants. We observed livestock grazing to lead to the deterioration of terrace plant abundance at Langenhoe, with hoof damage to the clay substratum. The adjacent marsh halophyte vegetation was also heavily grazed by cattle, to a sward height of c. 10 cm height, with a loss of *Atriplex*, *Limonium* and *Aster* in particular. Cattle and sheep hooves also compacted the surface sediment, creating fragmented vegetation cover. In some regions of the U.K. grazing of salt marshes is a traditional practice resulting in grass dominated communities and varying negative impacts on soil compaction and soil carbon (Nolte et al., 2013; Gilbertson et al., 2020; Ford et al., 2016; Davidson et al., 2017). However, the shrubby halophyte communities present on marshes in the south and east of England is significantly impacted by livestock grazing. It may be important to restrict livestock grazing near terraces or to establish them away from grazed sea walls.

Storm events can result in substantial sediment erosion, which reduces the rate of colonisation by vulnerable seedlings (Boorman, 2003). In December 2013 the U.K. was hit by the largest storm surge in 60 years. The storm (Xaver) which hit the U.K. on the 5th and December 6, 2013 resulted in the most serious tidal surge since the disastrous floods of 1953 (Spencer et al., 2015a,b). Record sea levels were observed in several locations due to the combination of low pressure interacting with the high tide (Spencer et al., 2015a,b). Fortunately, in eastern England, a largely offshore, westerly wind meant that aggressive wave action on flood defences was avoided. After the 2013 storm surge, 10 of 12 terraces being studied remained unchanged or gained height (between 6 and 13 cm) from sediment deposited, with two terraces (L3, T1) suffering erosion (Cousins et al., 2017). Plant cover is important in protecting marsh surfaces from storm surge events. *Puccinellia* plants are highly flexible, and coped better with storm surges, with *Atriplex* and other species breaking in flume experiments (Rupprecht et al., 2015; Spencer et al., 2015a,b). The early colonisation and increased cover of *Puccinellia* on the terraces will have provided a surface protection to storm surge erosion. Further severe storms in 2014, 2020, 2021 and 2022 may have caused wave damage to terrace vegetation and sediment loss. The major erosive impacts of storm surges and associated waves are on exposed marsh edges (Möller and Spencer, 2002; Spencer et al., 2015a,b), indicating that storms could potentially be a cause for the block failures of the narrow fringing marshes and terraces at Wellhouse and at Tollesbury T1.

4.5. Conclusions

Like many coastlines around the world (Evans et al., 2019), the majority of the Essex coastline is highly modified with sea defence structures and other fixed installations. This hard engineering disrupts the functioning of natural landforms and contributes to losses of biodiversity, habitats and ecosystem services (Naylor et al., 2023). Rising sea levels continue to threaten the integrity of these engineered structures, requiring interventions to maintain their function. The Environment Agency incorporated salt marsh terraces in a repair programme designed to include green elements into this infrastructure, termed “greening-the-grey” (Naylor et al., 2018, 2023).

It is clearly important that halophyte vegetation becomes established quickly in order to set a terrace on the desired trajectory towards salt marsh. Successful planting of halophytes has been trialled in some schemes, particularly in the U.S. (Silliman et al., 2015). Stepped salt marsh terraces were built on the Thames estuary in 1997, including a terrace at MHWN tide level planted with *Phragmites*, *Bolboschoenus maritimus* and *Aster tripolium* to encourage salt marsh development (Naylor et al., 2023; Estuary Edges, 2023). These designs had mixed success, with salt marsh establishing on 50% of the site, but erosion occurring on 50% of the site, and gravel and shingle smothering plant growth on parts of the terraces. Planting and use of matting was susceptible to being lifted up during periods of high flow and wave activity, and it was concluded that this was not helpful. Plant diversity was lower, becoming dominated by the (planted) common reed *Phragmites*. *Aster tripolium* and *Cochlearia anglica* occurred, but not necessarily where originally planted. Planting has also been trialled at other MR sites, but with limited success (Garbutt et al., 2006; Kadiri et al., 2011; Mossman et al., 2020). Failure of salt marsh planting schemes appears to be closely linked to the difficulty in recreating correct sediment conditions for successful halophyte growth (Spencer et al., 2017; Mossman et al., 2020).

Exposed clay sediments appeared particularly vulnerable to summer drying, even in the first year (2012–13). Once clays have hardened, it appears difficult for halophytes to colonise. Building terrace structures that incorporate a clay substructure but providing softer surface sediments (perhaps using dredging material, as was done at (Kadiri et al., 2011) may provide a better substratum for colonisation. Constructing an ideal proto-marsh sediment structure could be challenging, given changes caused by subsequent settlement, waterlogging and potential anoxia preventing a good soil structure developing (Spencer et al., 2017), but unless the sediment is suitable, halophytes will not establish.

The ecosystem functions and services provided by habitats do not necessarily scale linearly with size (Koch et al., 2009). While wave attenuation is greatest at the edge of a feature, the width of the habitat is an important scaling feature (Koch et al., 2009; Narayan et al., 2016; Möller et al., 2014). The terraces in this study were between 2 and 3 m wide, and in the absence of extensive halophyte cover, probably not wide enough to significantly attenuate wave energy. Small terraces were also vulnerable to lateral erosion; the most successful sites were embraced by existing marsh. Building terraces where the ends can be embedded into existing stable salt marsh is important. Alternative models could be to build lower in the tidal frame and encourage *S. anglica* growth, or to use a stepped set of terraces, so that the overall hybrid-engineered system is wider and offers mutual protection (Sutton-Grier et al., 2015). *Spartina anglica* salt marsh is wetter and stores less organic carbon than *Atriplex* or *Puccinellia* dominated marsh (McMahon et al., 2023), and is less biodiverse, and may not be as effective at attenuating wave energy (Spencer et al., 2015a,b; Widdows et al., 2008) so there could be trade-offs between sea-defence effectiveness and the generation of desired ecosystem services (Narayan et al., 2016). Nevertheless, the creation of linear vegetated features on a hard engineered structure will also provide aesthetic services and soften the impact of the necessary hard engineered defence (Sutton-Grier et al., 2015; McKinley et al., 2020; Naylor et al., 2023). The Essex sea wall

terrace project has shown that hybrid-engineered structures can provide sea defence capability and provide additional intertidal habitats that would not have been present with a conventional sea wall repair. With highly engineered coastlines, and an ongoing biodiversity and climate crisis, incorporating salt marsh terraces into sea defence structures can make a positive contribution as a nature-based solution to sea wall repair.

CRedit authorship contribution statement

Nicola J.D. Slee: Writing – review & editing, Writing – original draft, Investigation, Formal analysis, Data curation. **Tim Gardiner:** Writing – review & editing, Funding acquisition, Conceptualization. **Graham J.C. Underwood:** Writing – review & editing, Writing – original draft, Resources, Project administration, Investigation, Funding acquisition, Formal analysis, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

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