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Restored saltmarshes lack the topographic diversity found in natural habitat

Peter J. Lawrence[a], Graham R. Smith[a], Martin J.P. Sullivan[b], Hannah L. Mossman[a,*]

[a] School of Science & The Environment, Manchester Metropolitan University, Chester Street, Manchester M1 5GD, UK
[b] School of Geography, University of Leeds, Leeds LS2 9JT, UK

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ABSTRACT

Saltmarshes can be created to compensate for lost habitat by a process known as managed realignment (MR), where sea defences are deliberately breached to flood low-lying agricultural land. However, the vegetation that develops on MR sites is not equivalent to natural habitat. In natural sites, surface topography and creek networks are drivers of vegetation diversity, but their development on restored sites has not been well studied. We investigate the topographic characteristics of 19 MR areas, and compare these to nearby natural saltmarshes (representing desired conditions) and to coastal agricultural landscapes (representing conditions prior to MR). From high-resolution LiDAR data, we extracted values of elevation, six measures of surface topography (although two were later excluded due to collinearity), and three measures of creek density. MR and natural marshes differed significantly in all surface topographic indices, with MR sites having lower rugosity and more concave features, with greater potential for water accumulation. MR sites also had significantly lower creek density. MRs and coastal agricultural landscapes were more similar, differing in only one topographic measure. Importantly, there was no relationship between age since restoration and any of the topographic variables, indicating that restored sites are not on a trajectory to become topographically similar to natural marshes. MR schemes need to consider actively constructing topographic heterogeneity; better mirroring natural sites in this way is likely to benefit the development of saltmarsh vegetation, and will also have implications for a range of ecosystem functions.

1. Introduction

Saltmarsh is a valuable intertidal ecosystem that provides habitat for rare species, as well as important ecosystem services such as water regulation, wave attenuation, and recreation (Barbier et al., 2011). Loss of saltmarsh, particularly due to agricultural reclamation, has been substantial, with less than 50% of the extent of historic habitat remaining worldwide (Adam, 2002; Barbier et al., 2011). Although land claim still occurs, one of the major threats currently affecting saltmarsh is sea-level rise (Adam, 2002; Hay et al., 2015; Nicholls et al., 1999), exacerbated by the construction of static, hard sea defences, which prevent the natural landward migration of marshes, so that marshes are trapped between sea defences and rising sea-levels. This coastal squeeze results in loss of saltmarsh (Morris et al., 2004).

New saltmarsh is being created to combat this loss of habitat (Callaway, 2005; Zedler, 2004), partially motivated by legislation requiring its replacement (e.g. European Commission, 2007, USA Clean Water Act). Saltmarsh can be created through the process of managed realignment (MR), where sea defences are deliberately breached following the construction of new defences further inland, to allow tidal waters to flood the land between (French, 2006). Low-lying, coastal agricultural landscapes provide a key location for the restoration of saltmarshes, because much of this was saltmarsh prior to land claim.

Saltmarsh plant and invertebrate species can quickly colonise newly established MR sites (Garbutt et al., 2006; Mazik et al., 2010; Wolters et al., 2005), but community composition and function are often different to that found on natural saltmarshes. For example, plant communities that develop on MR sites are not equivalent to those found on natural saltmarshes (Mossman et al., 2012a). Furthermore, the vegetation on sites established on agricultural land accidentally breached during storm surges remains different to that on natural marshes, even 100 years post flooding (Mossman et al., 2012a). These differences in plant assemblages reduce biogeochemical functions such as carbon storage (Moreno-Mateos et al., 2012) and are likely to have knock-on effects on other plant-influenced ecosystem functions such as wave attenuation (Möller and Spencer, 2002; Rupprecht et al., 2017) and sediment erosion/ deposition dynamics (e.g. Ford et al., 2016), meaning that restored marshes are unlikely to satisfy legal requirements for biological and functional equivalency with natural marshes (Mossman et al., 2012a). Elevation (height above sea-level) is a key determinant of

* Corresponding author.
E-mail addresses: p.lawrence@mmu.ac.uk (P.J. Lawrence), g.r.smith@mmu.ac.uk (G.R. Smith), m.j.sullivan@leeds.ac.uk (M.J.P. Sullivan), h.mossman@mmu.ac.uk (H.L. Mossman).

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the vegetation communities that colonise restored sites because saltmarsh plants have clear elevational niches (Masselink et al., 2017; Sullivan et al., 2017; Zedler et al., 1999). Some restored sites were initially at low elevations because of relative sea-level rise and shrinkage of the land during the years of reclamation, and this may have limited vegetation colonisation (Garbutt et al., 2006).

Plant species also vary in their tolerance of poorly drained, waterlogged sediments (Davy et al., 2011; Huckle et al., 2002), with these conditions more frequent in some MR sites (Sullivan et al., 2017). However, the drivers underlying this increased waterlogging are poorly understood, although in some sites this appears to be due to poor drainage (Masselink et al., 2017). Local variation in surface elevation and shape, i.e. topography, can influence sediment drainage, with flat surfaces draining poorly. Increased topographic variation and complexity could increase the range in potential niches available and thus increase plant diversity (Kim et al., 2013; Moffett and Gorelick, 2016; Morzaria-Luna et al., 2004), which could influence the provision of ecosystem services such as flood defence (Rupprecht et al., 2017). Furthermore, topographic features such as creeks are important to saltmarsh functioning, as they supply sediment and water across the marsh, and provide nursery habitat for juvenile fish (Cavraro et al., 2017; Desmond et al., 2000; Peterson and Turner, 1994). Topography on natural saltmarshes can take many forms, such as hummocks, pans, creeks and levees (Fig. 1; Goudie, 2013). Land management during reclamation, such as ploughing, trampling and channelization of creeks, may reduce surface topography prior to restoration. For example, research at one MR site found reduced heterogeneity in surface elevation compared to natural marshes (Brooks et al., 2015). However, little is known about the topographic diversity of other restored marshes or how this topography develops over time.

We assess the surface elevation, topography, and creek network density and diversity of 19 MR areas, comparing these to natural saltmarsh and local agricultural reference sites. To do this, we use remote sensing (specifically, Light Detection And Ranging [LiDAR]) derived digital elevation models (DEMs), from which we calculate a range of topographic indices and creek network measures that describe the characteristics of the marsh surface. Using this data, we investigate the following questions: (1) Does topography differ between natural saltmarsh, restored saltmarsh (MR), and adjacent agricultural landscapes; (2) Does topography vary with age since restoration and with former land-cover; (3) Are any differences in topography between MR and natural saltmarshes consistent across the intertidal elevational range?

2. Methods

2.1. Study sites

Seventeen MR sites, ranging from 4 to 23 years since the date of breach, were selected along the south and east coasts of the UK (Fig. 2 and Table A1). Two of the MR sites were divided into two hydrologically distinct areas by sea walls or other landscape features, which resulted in a total of nineteen MR areas. MR sites were identified using the ABPmer online database (ABPmer Online Marine Registry, 2014) and aerial photography, and later selected based on the availability of LiDAR data after restoration, as well as to ensure coverage of a range of geographic locations and site ages. Twelve natural saltmarshes and fourteen agricultural plots were sampled as reference sites, representing respectively the desired end-conditions and likely starting conditions of

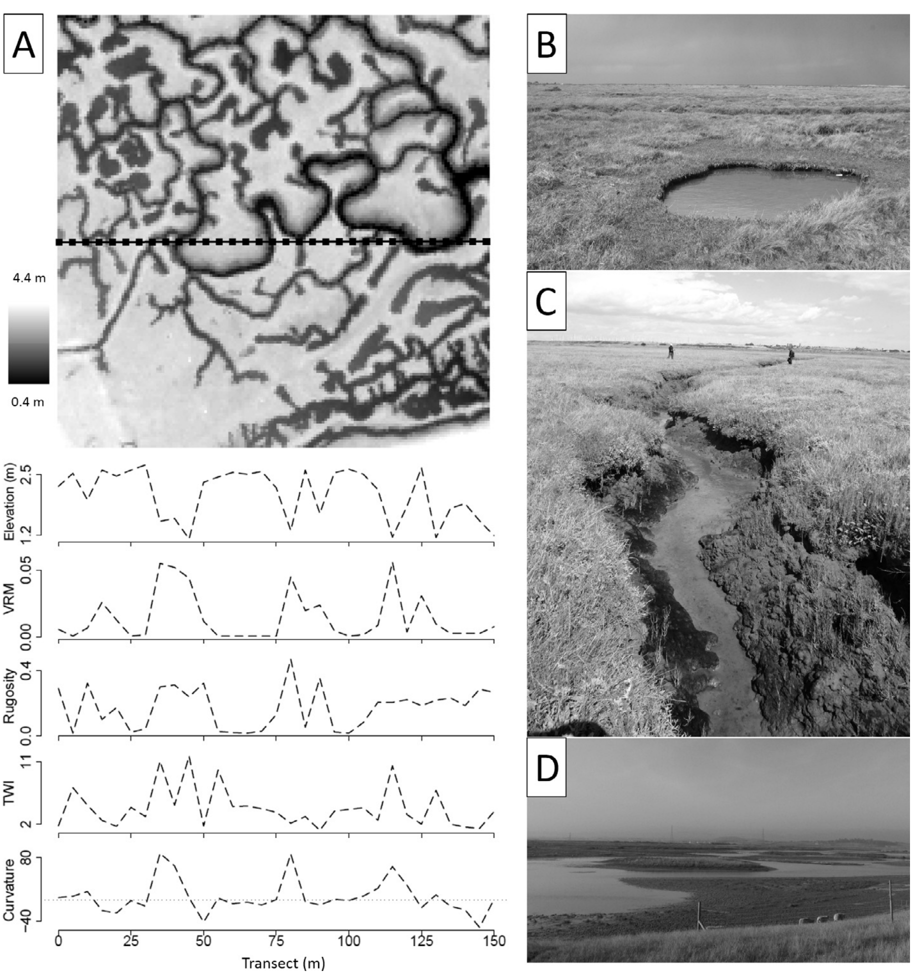

Fig. 1. (A) A sample digital elevation model from Tollesbury (Essex) showing elevation (m ODN). Topographic variables have been illustrated along a seaward transect represented by a dashed line. The five plots below show measurements every 5 m along this transect. From top to bottom these are Elevation, vector rugosity measure (VRM), rugosity (s.d. elevation), topographic wetness index (TWI) and profile curvature. For profile curvature, the dotted line separates convex (−ve) and concave (+ve) scores. Photos illustrate (B) a concave salt pan with high TWI and low rugosity; (C) a creek with variable TWI, concave profile curvature and high rugosity; (D) a constructed hillock at a MR that has low TWI, higher rugosity and convex profile curvature.

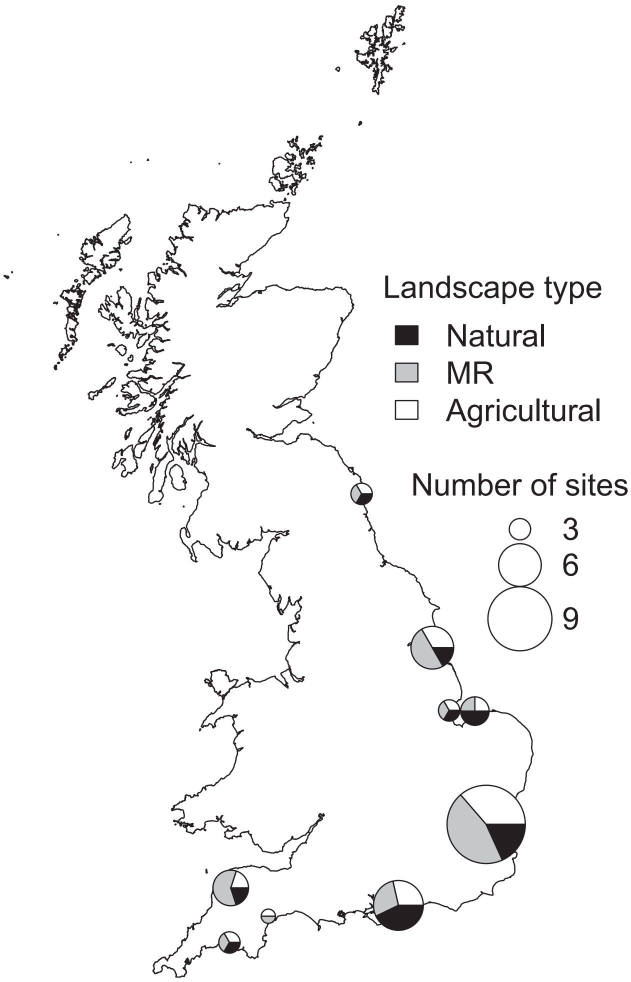

Fig. 2. Location of study sites. Pie charts are positioned at the centroid of clusters of sites within 0.5 degrees of each other, and show the proportion of sites that were natural marshes, managed realignments (MR) and agriculture. The size of each pie chart is proportional to the number of sites sampled. Coordinates of each study site are given in Table A1.

restored sites. Natural saltmarshes were selected to minimise the distance to MR sites (mean distance to MR site = 6.95 km) while ensuring that they were large enough for reference plots of similar size to MR sites to be created. In some areas of the UK, natural saltmarsh is currently undergoing substantial erosion (Cooper et al., 2001). This type of erosion is easily identified by interpretation of aerial photography due to substantial internal dissection and limited vegetation cover; these areas were not sampled. Sites affected by significant anthropogenic structures other than sea walls, such as slipways and groins, were also not selected. Areas of natural saltmarsh were often larger and lacked the clear boundaries of MR sites, which were enclosed by seawalls. In this study, we defined the extent of the sampled natural saltmarshes by using the mean shoreline length of the 19 MR areas. The extent of the marsh perpendicular to the shoreline was defined as the seaward edge of the vegetation, identified from aerial photography. Using these rules, a polygon was digitised within a GIS environment to establish site area of each natural marsh. The mean size of MR areas was 16.5 ha and natural saltmarsh was 18.2 ha. Agricultural reference areas were selected based on the criteria that sites should be as close as possible to MR areas (mean distance = 1.97 km), be adjacent to the coast/estuarine system and be continuous fields (not surrounded by walls or roads as these can be problematic for the flow models used to construct some topographic metrics) that were large enough to create plots of similar size to MR areas (mean size of agricultural areas = 13.8 ha). MR is carried out on both arable and grazed land, so both were included as agricultural reference areas (topography was similar between arable and grazed reference areas, Fig. A1). Each estuarine complex containing a MR area had at least one natural reference (with the exception of the Clyst Estuary where no suitably sized natural reference marsh was available) and one agricultural reference area, ensuring that regional variation in variables such as tidal range and plant community composition were similarly represented in both MR sites and reference sites. Sampling multiple sites this way also enabled us to capture variation in reference conditions (Vélez-Martín et al., 2018).

Previous land cover of MR sites was identified using the land-cover datasets for 1990 and 2007 (Morton et al., 2011), enabling the comparison of topographic variation between different former land cover types. Of the nineteen MR areas, we found eleven were formerly defined as dominated by grazing practises (mown or grazed turf, meadow and semi-natural swards) and eight as 'arable' use prior to breaching (arable and horticulture).

2.2. Quantifying topography

One-metre resolution LiDAR-derived DEM data were downloaded for all sites on 11th February 2016 from the free UK LiDAR resource (UK Government, 2016). These were mosaicked into a continuous gridded raster surface (one for each site rather than a complete coastal DEM for England) in ArcGIS © version 10.2 (ESRI, 2013). The date of the LiDAR survey was noted during download in order to calculate the number of years since restoration that the LiDAR data were collected, i.e. the age of the restored site (Table A1). The stated vertical accuracy (root mean squared error) of the UK LiDAR dataset was between ± 5 cm and ± 15 cm, with values tending to be lower in more recent surveys (Environment Agency, 2016). For each location area, a site boundary polygon was digitised. We then randomly selected 10% of the cells contained within each polygon as our sampling points using a (minimum) separation between points of 1.4 m to ensure no resampling of values. At each sampling point, six measures of topography (including measures of rugosity, curvature, slope and topographic wetness) and three measures of the creek network density and diversity were initially calculated and extracted, with measures selected for their ecological interest whilst also limiting redundancy between measures (Table 1, Fig. 1).

We employed a 3 × 3 cell neighbourhood ($3\,m^2$) with a moving-window to calculate six of the topographic variables. We did not use a larger window as this would artificially smooth the landscapes losing the impact of the smallest features (Liu et al., 2015), such as small creeks often < 1 m in width, thus reducing the biological relevance of values obtained (Grohmann et al., 2011). From this, two indices of local surface heterogeneity, commonly known as rugosity, were extracted. The first measure of rugosity (RUG) was obtained using the standard deviation of elevation in the local 3 × 3 window (Grohmann et al., 2011; Hobson, 1972). The second was the vector rugosity measure (VRM), a 3-dimensional measure of rugosity, calculated as the summed magnitude of variation along x, y and z vectors producing a ruggedness value on a scale of 0–1, with 0 being flat and 1 equating to maximum ruggedness (Hobson, 1972; Sappington et al., 2007). $VRM = \frac{\sqrt{(\sum_{i=1}^{n} X_i)^2 + (\sum_{i=1}^{n} Y_i)^2 + (\sum_{i=1}^{n} Z_i)^2}}{n}$, where X_i = sin(slope) x cos(aspect), Y_i = sin(slope) x sin(aspect), Z_i = cos(slope) and n = cell neighbourhood. VRM has been shown to not be strongly correlated with other topographic variables, thereby helping to avoid collinearity (Sappington et al., 2007). The third index obtained using the 3 × 3 cell neighbourhood was the topographic wetness index (TWI). TWI is defined as the number of cells draining through each point in the context of the local slope, and calculated as TWI = ln (a/tan b) where a = local upslope area and b = local slope in radians. High TWI values indicate drainage depressions and the lowest values centred on the top of ridges (Beven and Kirkby, 1979; Sörensen et al., 2006).

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Table 1

Description of topographic variables selected and their form and functional importance. Note that slope and total curvature were not included in subsequent statistical analyses as they were strongly correlated with other topographic variables.

DEM variable	Topographic relevance	Ecological importance
Elevation[1,2]	Flooding duration	Zonation/sea-level change mitigation
Slope (deg.)[3]	Drainage and niche	Soil hypoxia
Vector rugosity measure (VRM)[4,5]	Micro topography	Metre scale niche detection
Rugosity (RUG)[4,5]	Micro topography	Metre scale niche detection
Total curvature[6]	Creek detection	Creek development, drainage
Profile curvature[6]	Creek detection	Creek development, drainage
Topographic wetness index (TWI)[7]	Local soil moisture	Soil hypoxia independent of slope
Distance to creek[8,9,10]	Drainage	Bio/Chemical sediment transfer
Creek order[8]	Network complexity	Erosion and levee creation (plant niche)
Creek density[11]	Drainage	Vegetation configuration

Reference key: [1] (Bockelmann et al., 2002), [2] (Brooks et al., 2015), [3] (Hladik and Alber, 2014), [4] (Collin et al., 2010), [5] (Sappington et al., 2007), [6] (Moore et al., 1991), [7] (Sörensen et al., 2006), [8] (French and Stoddart, 1992), [9] (Christiansen et al., 2000), [10] (Sanderson et al., 2000), [11] (Moffett and Gorelick, 2016).

Inbuilt functions within ArcGIS were used to calculate surface slope and two measures of surface curvature. Slope is a useful topographic variable measuring in degrees the angle of maximum elevation change within a pre-defined window, in our case 3 × 3 cells. Curvature is also calculated at local-scale and can be derived in several ways. Here, we use curvature following the direction of maximum slope (profile curvature), and an aggregated curvature in all directions (total curvature) (Moore et al., 1991). Negative values of curvature indicate a convex feature, zero a planar surface and positive values a concave feature.

The elevation relative to Ordnance Datum Newlyn (ODN, approximately mean sea-level) was extracted from the DEMs. However, elevation relative to mean sea-level does not account for the variation in tidal amplitude between regions. In order to place the elevation relative to ODN in the context of the local tidal regime, we transformed elevation into relative tidal height (RTH) on a scale of 0–1, where 0 = mean high water neap tide level (MHWN) and 1 = mean high water spring tide level (MHWS). Data for MHWN and MHWS levels were obtained from local port data and those published in Mossman et al. (2012b).

To describe the creek networks, we calculated distance to nearest creek (measured from each sampled point) and two site-scale measures, total creek density and the density of different creek orders. Creek metrics were not calculated for agricultural sites due to the lack of functional comparability with marsh creek networks. To delineate creeks from a DEM, we used flow accumulation threshold set at 1000 cells, as this value resulted in the most reliable delineation of creeks (i.e. without including relic creeks and salt pans). Flow accumulation-based networks can be subject to erroneous creeks in flat areas, so we used a semi-automated methodology to increase accuracy (Lang et al., 2012; Liu et al., 2015). As a result, aerial photography and satellite imagery were used to post-process the flow accumulation model as they have been shown to be effective at identifying creek networks (Goudie, 2013; Moffett and Gorelick, 2016). The creek networks were classified according to Strahler (1957) stream order, with the smallest (source) creeks assigned to first order, and order increments with each downstream intersection. In each site, lengths of all creeks were summed and the total creek density calculated. Creeks were split into the relevant stream order category and the density of each order of creek per site calculated.

Fig. 1 visualises how the surface topographic measures relate to DEM and gives examples of topographic features *in situ*. Fig. 1B shows a salt pan, which would have a positive profile curvature value, indicating it is a concave feature, and a high value for the topographic wetness index. Fig. 1C shows a small creek and Fig. 1D shows a constructed hillock on a MR site, a convex feature with negative profile curvature and low topographic wetness index.

2.3. Statistical analysis

Topographic variables were calculated from the DEMs in the R environment (R Development Core Team, 2012) using the packages ‘raster’ (Hijmans 2015), ‘rgdal’ (Bivand et al., 2016) and ‘rgeos’ (Bivand and Rundel, 2016). Pearson’s product moment correlations were used to identify collinearity between topographic variables; total curvature was omitted due to strong correlation with profile curvature (r = 0.92), and local slope omitted due to correlations with rugosity (RUG, r = 0.97), vector rugosity (VRM) and profile curvature (both $r > 0.5$).

All variables were not normally distributed (Shapiro-Wilks, all $p > 0.05$), so non-parametric analyses were used where possible. Kruskal–Wallis (K-W) tests were used to identify significant differences in the total creek density and densities of each creek order between landscape types. Site averages for each topographic variable were calculated and these were compared between pasture and arable former land cover types with Kruskal–Wallis tests. Spearman’s rank correlations were used to test for correlations between the surface topographic variables and site age, site size, 1st order creek density, total creek density, and distance to nearest creek of MR sites. Linear mixed-effects models (LMMs) were used to test for differences in topographic variables between the three landscape types (natural marsh, MR and agriculture), with site as a random effect, using the R packages ‘nlme’ (Pinheiro et al., 2009) and ‘multcomp’ (Hothorn et al., 2008). Although these assume normality, they are robust to violations of this assumption when sample sizes are large (e.g. Arnau et al., 2013), as is the case with this analysis where tens to hundreds of thousands of data points were used in each analysis. LMMs were used to test whether differences in topography between natural and MR marshes varied across their elevation range, using the R package ‘lme4’ (Bates et al., 2015). To do this, we constructed a LMM with landscape type, relative tidal height and their interaction as fixed effects, and site as a random effect. We assessed the significance of this interaction term by comparing it to a nested model lacking the interaction term using a likelihood-ratio test. Likewise, we tested whether landscape type had a significant additive effect on topography while controlling for the effect of relative tidal height, by comparing a LMM with landscape type and relative tidal height as fixed effects with the nested model only containing relative tidal height as a fixed effect. Finally, we use LOWESS regressions to visualise relationships between topography and elevation in natural and MR marshes. All data were used to calculate LOWESS relationships, but the data visualised are between relative tidal heights of -0.5 and 1.5 (97.8% data) for clarity (total RTH range = -2.54 to 5.23). Confidence intervals around these relationships were calculated by taking 1000 resamples of the data with replacement.

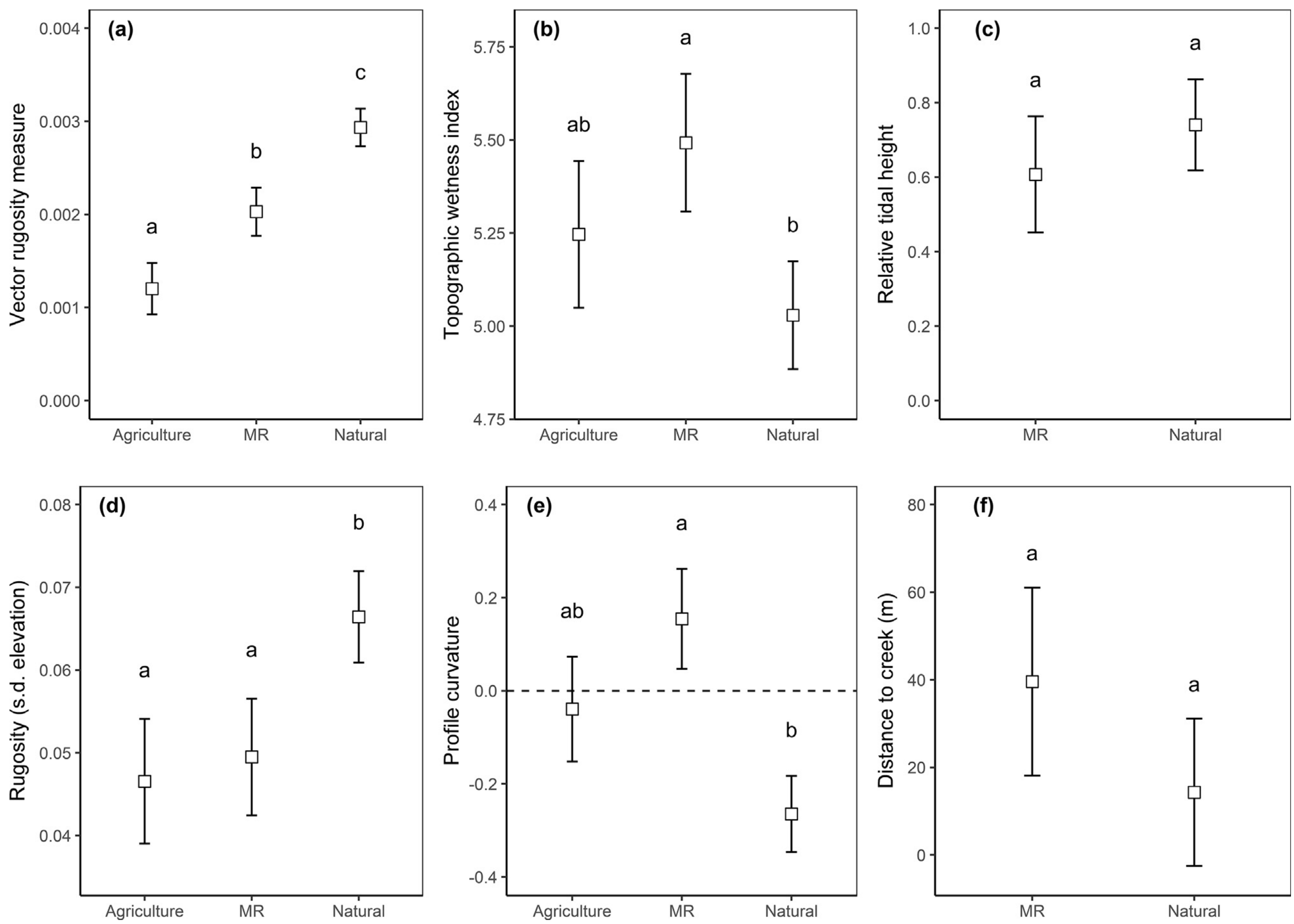

Fig. 3. Mean (± SE) calculated via a GLMM of six topographic indices: (a) Vector rugosity measure, (b) Rugosity (s.d. elevation), (c) Topographic wetness index, (d) Profile curvature the dotted horizontal line in this graph represents a switch from convex (−ve) and concave (+ve) scores, (e) Relative tidal height and (f) Distance to nearest creek. Letters indicate significant differences ($p < 0.05$) between the landscape types.

3. Results

3.1. Comparison of topography between landscape types

All topographic measures, extracted at the randomly located sample points, differed between natural saltmarsh and MR landscape types, except for distance to the nearest creek and relative tidal height (RTH) (Fig. 3). Both measures of rugosity were significantly lower on MR sites (VRM: $z = -3.49$, $p = 0.001$; RUG: $z = -2.40$, $p = 0.043$) and MR sites had significantly higher values of topographic wetness index (TWI: $z = 2.50$, $p = 0.032$), indicating they are flatter and have a greater potential for water accumulation. Profile curvature differed significantly between natural marsh and MR landscape types (Profile curvature: $z = 3.899$, $p < 0.00.1$), with MR being concave on average (mean ± s.e, 0.154 ± 0.107) and natural marshes convex (−0.264 ± 0.081) in the direction of the maximum slope. Total creek density was significantly lower in MR marshes (Table 2, $\chi^2 = 4.62$, $p = 0.03$). This difference was greatest for the smallest creeks (1st order), although differences were not statistically significant for any individual creek order ($p = 0.51$ for 1st order creeks, $p \geq 0.257$ for other creek orders). Topographic wetness index and profile curvature values for the agricultural landscape were between those recorded for MR and natural landscapes respectively (Fig. 3). VRM and RUG were both significantly different between MR and agricultural landscapes, with MR sites having higher rugosity (VRM: $z = -6.23$, $p < 0.001$; RUG $z = -2.64$, $p = 0.022$).

Rugosity was positively correlated with total creek density ($r_s = 0.67$, $p = 0.001$) and density of the 1st order (smallest) creeks ($r_s = 0.74$, $p < 0.001$), but negatively correlated with distance to nearest creek ($r_s = -0.66$, $p = 0.001$). The density of 1st order creeks was negatively correlated with topographic wetness (TWI $r_s = -0.47$, $p = 0.033$), suggesting these smaller creeks must also play a role in reducing up-slope catchments and flat areas.

Table 2
Mean (± standard deviation) density of creek orders (m.ha^{-1}) for the natural marsh and managed realignment.

Density of creeks	Natural marsh (n = 12)	Managed realignment (n = 19)	χ^2	*p*
1st order	127.26 ± 33.33	96.54 ± 42.98	3.78	0.051
2nd order	63.14 ± 21.17	65.43 ± 39.37	0.25	0.611
3rd order	35.07 ± 20.56	27.84 ± 21.17	1.28	0.257
4th order	18.55 ± 19.21	11.45 ± 6.45	0.03	0.855
Total density	233.21 ± 55.81	182.18 ± 71.31	4.62	0.030

One MR site contained a 5th order creek at a density of 1.62 m ha^{-1} omitted from table due to lack of comparison.

3.2. Does topography differ with age since restoration and former land cover?

The age (time since restoration) and area of MR sites were not significantly correlated with any topographic variable (Fig. 4; Table A3). Some individual restored sites overlapped with natural marshes in their characteristics, but there was no trend over time in these characteristics (Fig. 4). There were no significant differences in any topographic variables between pasture and arable land covers prior to

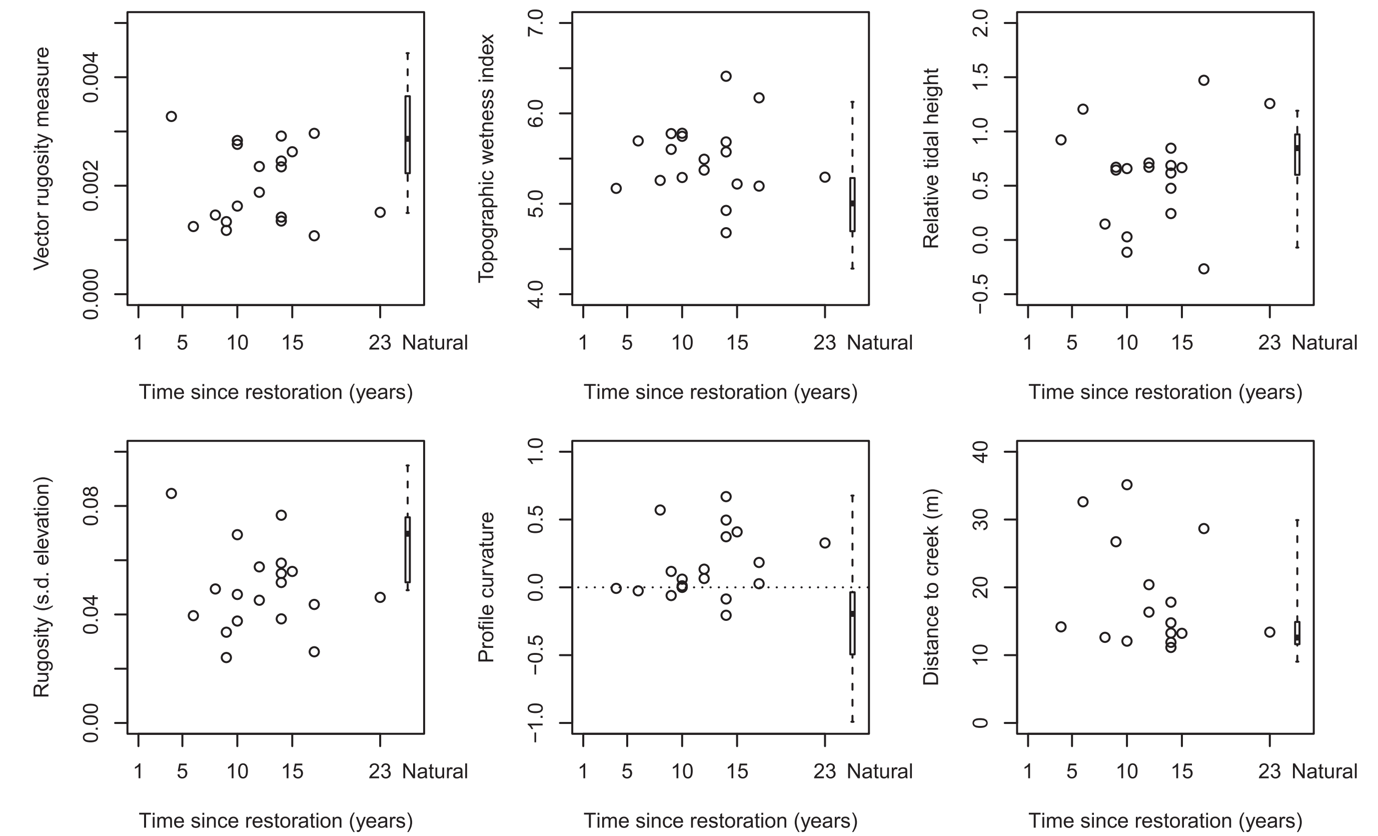

Fig. 4. MR site means plotted against time since restoration in years for each of the six topographic indices: (a) Vector rugosity measure, (b) Rugosity (s.d. elevation), (c) Topographic wetness index, (d) Profile curvature the dotted horizontal line in this graph represents a switch from convex (−ve) and concave (+ve) scores, (e) Relative tidal height and (f) Distance to nearest creek. No relationships were statistically significant.

restoration (Kruskal-Wallis, all $p > 0.05$; Table A4).

3.3. Consistency of topographic differences across elevations

There was a statistically significant interaction between landscape type and elevation for all topographic variables (Table 3). At RTH below zero, MR were flatter (demonstrated by lower VRM and RUG) than natural marshes and with substantially greater potential for water accumulation (higher TWI) (Fig. 5). At these elevations, both natural and MR landscapes were dominated by concave features, with MR being less concave. Furthermore, distance to the nearest creek was the same in both landscapes, but as elevation increased there was divergence between the landscape types, and distance to the nearest creek was substantially greater in MRs than natural marshes above 0.5 RTH. Both rugosity measures were higher in natural than MR marshes between 0 and 1 RTH, but became similar at higher elevations. Between 0 and 0.5 natural marshes were dominated by convex features, whilst MR sites remain dominated by concave features. MR sites briefly become less concave than natural marshes above 0.5 RTH, but above 1.0 RTH MR became strongly concave compared to natural marshes that were moderately concave. MR showed higher potential for water accumulation than natural marshes, except between RTHs of 0.75 and 1.2.

Table 3
Effect of landscape type (restored or natural saltmarsh) and elevation above sea level (relative tidal height RTH) on topographic variables. This is examined as an interaction with relative tidal height, and as an additive term controlling for relative tidal height. The significance of each term was assessed using likelihood ratio tests between a LME model containing the term and a nested model without the term.

DEM variable	Interaction between landscape and RTH		Additive effect of landscape	
	χ^2	p	χ^2	p
Vector rugosity measure	13,364	< 0.001	5.593	0.018
Rugosity (s.d. elevation)	10,795	< 0.001	7.551	0.005
Topographic wetness index	1481	< 0.001	0.812	0.367
Profile curvature	10,564	< 0.001	0.300	0.584
Distance to creek	615.96	< 0.001	1.552	0.212

4. Discussion

4.1. Topography on restored saltmarsh

Saltmarshes restored through managed realignment differ in their topography to natural marshes, and are more similar to the agricultural landscapes they originate from. Compared to natural marshes, they have an enhanced potential for water accumulation (higher topographic wetness index) and lower densities of creeks. Importantly, there was no relationship between age of the restoration and any of the topographic variables. This indicates that, although some individual restored sites overlapped with natural marshes in their characteristics, overall, restored sites are not on a trajectory to become topographically similar to natural marshes over time. We note that, despite the absence of a linear trend, marsh development may exhibit non-linear dynamics (van Belzen et al., 2017), for example, large-disturbance events could alter trajectories of topographic development. The lack of convergence of topography in our dataset is notable as it is mirrored in some other physical, chemical and biological components of restored saltmarshes such as vegetation establishment (Mossman et al., 2012a) and soil edaphic conditions (Hazelden and Boorman, 2001); indeed, topography may act as a driver for these variables (Varty and Zedler, 2008).

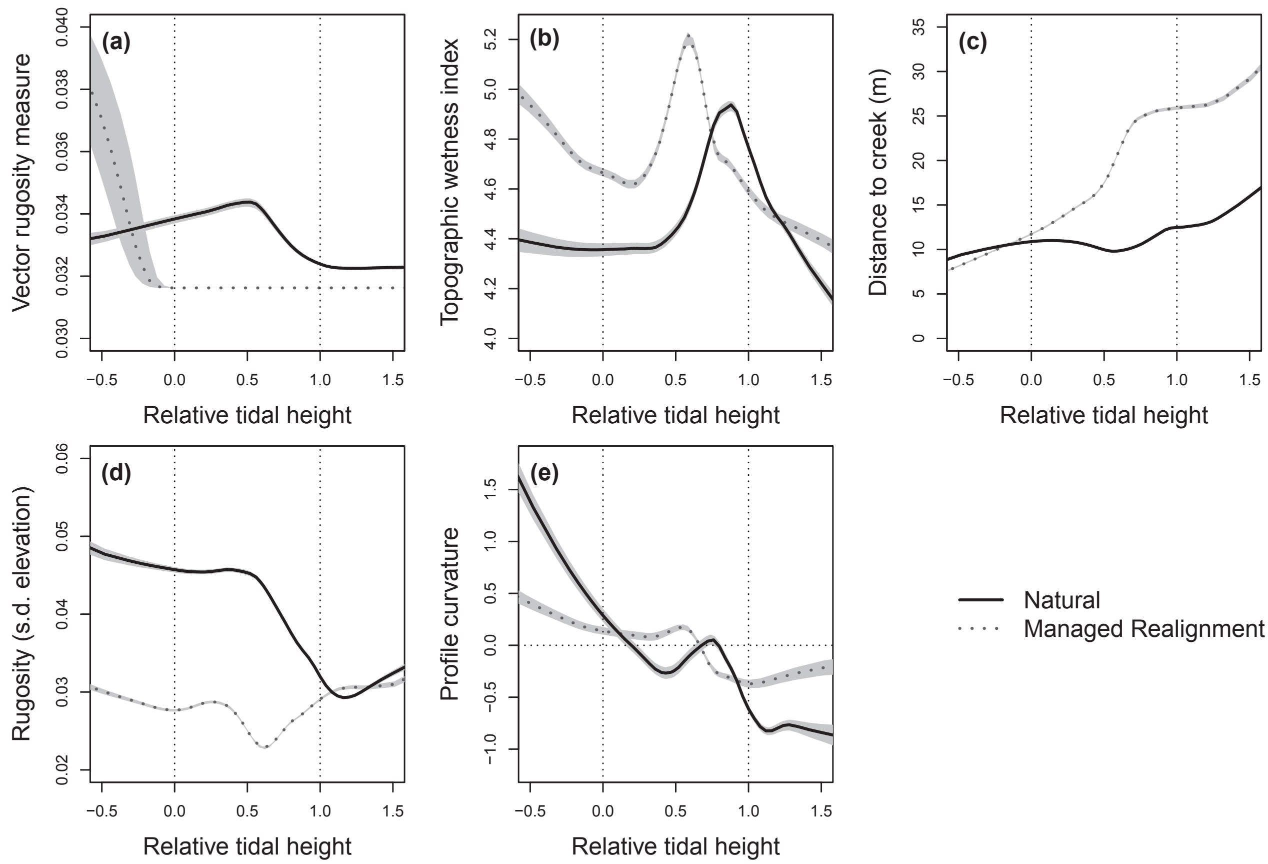

Fig. 5. Relationships (Lowess regressions) between elevation and topographic variables for natural and restored (MR) landscapes. (a) Vector rugosity measure, (b) Rugosity (s.d. elevation), (c) Topographic wetness index, (d) Profile curvature the dotted horizontal line in this graph represents a switch from convex (−ve) to concave (+ve) scores, (e) Relative tidal height and (f) Distance to nearest creek. The elevations at which saltmarsh plants typically occur, 0 and 1 relative tidal height, are marked.

Previous studies have found restored marshes to be lower in the tidal frame, on average, than natural marshes (e.g. Garbutt et al., 2006). In contrast, we found that elevation did not differ between restored and natural marshes. However, all measures of topography varied with elevation and these relationships differed between the landscape types. At low elevations, MR sites were dominated by local depressions (e.g. those surrounding the hillock in Fig. 1D) that often take the form of permanent pools of water or poorly drained areas (indicated by high topographic wetness index), in contrast to natural marshes. This could explain the previous observation that, at low elevations, sediment redox potentials are lower in MR sites than at equivalent elevations on natural marshes (Mossman et al., 2012a). This is because drainage, in addition to tidal inundation, has substantial influence on sediment aeration (and hence redox potential (Armstrong et al., 1985)), and depressions and concave features retain water at low tide, resulting in lower redox potentials at the sediment surface (Varty and Zedler, 2008).

4.2. Implications for vegetation development and ecosystem functioning

Elevation in the tidal frame and redox potential are the major determinants of the niches of saltmarsh plants (Davy et al., 2011). Our finding that restored marshes are flatter will equate to fewer elevational niches being available, and could lead to more homogenous vegetation observed on MR marshes (Collin et al., 2010). Even very small variations in elevation at restored sites resulted in differing vegetation communities (Ivajnšič et al., 2016). This is likely due to changes in immersion time (Masselink et al., 2017), known to impact plant mortality (Hanley et al., 2017). The concave-dominated environments of restored landscapes will generate poorly-drained conditions expected to be suitable for pioneer species, such as *Spartina anglica* and *Salicornia* spp. (Sullivan et al., 2017). Indeed, these species dominate restored marshes (Masselink et al., 2017; Mossman et al., 2012a; Zedler et al., 1999).

In contrast, we find that at elevations typically suitable for mid and upper marsh plants (e.g. RTH 0.75-1.0), natural marshes have a higher potential for water accumulation than restored marshes, with an increase in concave features. These landscape features between RTHs of 0.75 and 1 can increase vegetation diversity by excluding dominant upper-marsh species, allowing plant species more tolerant of harsh conditions to colonise gaps (Sullivan et al., 2017; Varty and Zedler, 2008). The absence of such environmental features at this elevation range on restored marshes may be limiting the establishment and persistence of waterlogging-tolerant mid and upper marsh species, such as *Triglochin maritima* (Fogel et al., 2004), that are rare or absent on restored marshes (Mossman et al., 2012a).

Plant species richness is higher in the areas immediately around creeks (Sanderson et al., 2000), presumably due to the resulting modifications of the abiotic environment, which gives a greater diversity of resulting niches. Our finding that restored landscapes have lower creek densities will therefore have consequences for saltmarsh vegetation. Moreover, creek networks are essential to the use of saltmarshes by fish and crustaceans, including commercially important species (Callaway, 2005; Crinall and Hindell, 2004; Peterson and Turner, 1994). The lower creek density of restored marshes is therefore likely to reduce their ecosystem function as fish nursery grounds (Desmond et al., 2000).

Topographic heterogeneity is likely to influence ecosystem functioning both directly, and indirectly by affecting plant diversity and community composition (Callaway, 2005). Diverse plant communities can enhance sediment stability (Ford et al., 2016) and may increase aboveground biomass production (Doherty et al., 2011), both of which would increase carbon storage. Furthermore, plant species differ in the extent to which they attenuate or withstand wave energy (Rupprecht et al., 2017), so diverse assemblages may enhance flood protection. Topography may also have direct effects on ecosystem functioning. Waterlogging associated with concave topography influences carbon cycling by microbes (Li et al., 2010), while these anoxic sediments are important locations for methane production (Oremland et al., 1982). Finally, wave energy is better dissipated by convex marsh profiles than concave ones (Hu et al., 2015), while the greater rugosity of natural marshes is also likely to increase wave attenuation (Moeller et al., 1996). It is important to note that while these likely differences in functioning would mean that ecosystem service provision by restored marshes is less than by natural marshes, restored marshes will still provide important ecosystem services relative to agricultural land (MacDonald et al., 2017).

4.3. Developing topographic heterogeneity on restored saltmarshes

There are a number of potential explanations for variation in topographic diversity between saltmarsh landscape types. We found no difference in the topography between sites that were arable or pasture prior to restoration. However, other research has found signals from pre-restoration land cover in poor surface drainage and changes to sediment structure, such as the formation of an impermeable layer (aquaclude) (Spencer et al., 2008, 2017). This impermeable layer is an effective barrier to erosion, preventing the scouring required for creek formation (Chen et al., 2012), thereby potentially reducing creek density. This could limit the development of other topographic features in restored landscapes to the depth of newly deposited sediment. Furthermore, high sedimentation rates, as observed in many restored marsh landscapes (Garbutt et al., 2006; Mazik et al., 2010), may fill any existing depressions (Elschot and Bakker, 2016) resulting in a smoothing of the marsh topography. In natural marsh landscapes, the patterns of topography are defined by the accretion of sediment at low elevations very early in marsh development (Elschot and Bakker, 2016). Restored landscapes that are not at suitably low elevations at the time of flooding may miss this window of opportunity for topographic development. Furthermore, limited tidal exchange (e.g. single breaches, regulated tidal exchanges) may impair creek development by reducing scour and sediment accretion (Masselink et al., 2017).

We have shown that topographic differences can be detected from LiDAR-derived digital elevation models across multiple restoration sites, which provides us with the opportunity to use space-for-time substitution to learn lessons from former MR schemes and guide the design of future restored landscapes. Our results suggest that the construction of additional topographic features will be needed to create marshes that are more similar to natural sites, since these features do not develop over time at MR sites. The creation of small creeks and hillocks are likely to be most useful in improving outcomes for vegetation development, as hillocks are likely to be preserved despite high sedimentation and networks of small creeks will increase drainage within sites. Recently constructed managed realignments have included the creation of these topographic features, e.g. hillocks at Steart Marshes, UK (Fig. 1D), and at Hesketh Out Marsh East, UK, small sinuous creeks with bank incisions to promote secondary formation and raised infill areas on the marsh to promote topographic variation (R. Shirres, *pers. comm.*). The functioning and longevity of these features should be monitored.

4.4. Conclusions

We find that within the time scales studied, restored saltmarshes are not on a trajectory to develop topography or creek densities equivalent to those of natural landscapes, and remain similar to the agricultural areas they originate from. These differences have implications for vegetation development and other aspects of restored marsh functioning, such as provision of fisheries habitat. Creation of more topographic features, including hillocks and small creeks, prior to restoration appears to be necessary to ensure restored saltmarshes develop topographic heterogeneity.

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Author contributions

PL, GS and HM conceived the study, PL and GS collected the data, PL, MS and HM analysed the data, PL, GS, MS and HM wrote the paper and approved the final manuscript.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecoleng.2018.02.007.

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