Changes in sediment characteristics in the first year of a Q6 realigned saltmarsh

The corrections made in this section will be reviewed and approved by a journal production editor.

Leda L. Cai* ledalcaiphd@gmail.com, Thorunn Helgason, Kelly R. Redeker

Department of Biology, University of York, YO10 5AT, UK

*Corresponding author.

Abstract

Saltmarsh restoration is described as a cost-effective response to coastal biodiversity loss and flood management, although, to date such conservation practices yield different characteristics than natural marshes, such as in plant communities and sediment properties. The outcome of these differences is thought to be influenced by the initial conditions set within the first year of marsh re-establishment, which are often overlooked or neglected as part of the restoration process. Therefore, our study aimed to address this gap by examining the spatial and temporal variations in sediment characteristics of a newly (<15 months) realigned saltmarsh in Essex, UK and the effects of previous land use (arable and pasture). Sediment properties (bulk density, nutrients and pH) were sampled monthly from the realigned site as well as from an adjacent reference natural marsh. After 14 months from initial inundation the two realigned marsh regions (arable and pasture) behaved similarly despite different starting points in terms of pH, water content, bulk density and nutrient load. Physical aspects of the realigned sediments (bulk density and water content) matched those of the reference natural marsh within 14 months post-breach in the 0–5 cm depth range. By contrast, the lower depth sediments within realigned sites remained more dense and drier compared to natural marsh sediments. Chemical properties of the realigned sediments matched those of the natural marsh throughout the sediment profile 14 months post-breach. Success of restoration is measured in the ability of the site to provide the desired ecosystem services and short-term recoveries and set goals may not imply long term sustainability as these observed differences may have ramifications for future biological community development and the success of a restoration.

Keywords:

Managed realignment, Marine sediment, Saltmarsh restoration, Essex UK

Abbreviations

No keyword abbreviations are available

1 Introduction

Saltmarshes are coastal habitats that provide important ecosystem services, from support for coastal and terrestrial food chains to coastal protection (Beaumont et al., 2008; Hughes and Paramor, 2004; Millenium Ecosystem Assessment, 2005). Tidal marshes are predicted to bury carbon and globally the burial is estimated between 5 Tg (Tg) and 87 Tg C yr^{-1} (McLeod et al., 2011). Saltmarshes provide protection to coastal areas by dissipating wave and tidal energy, reducing the possibility of sea walls being breached, overtopped or undermined (Möller et al., 1999; Möller and Spencer, 2002). Saltmarsh coverage globally (as of 2015) is approximately 5.5 million hectares (Mcowen et al., 2017), across 43 countries in almost all continents.

Climate change threatens coastal habitats through sea-level rise and impacts on sediment availability (Schuerch et al., 2018). Human responses to sea-level rise in order to lower likelihood of flooding of people and property are to either retreat, accommodate or protect (Nicholls, 2011). Installation of sea defences, in service of protecting coastlines, constricts coastal shore-line habitat development and the natural expansion/development of salt marshes, also described as coastal squeeze (Boorman, 2003; Nicholls, 2011). Over the last two decades there has been an accelerated global decline in saltmarsh extent (Millenium Ecosystem Assessment, 2005) and efforts have been initiated to conserve existing marshes and to develop new coastal marsh habitats (realigned or managed saltmarshes). The European Union Habitat Directive (adopted in UK legislation in 1992) maintains a no-net-loss policy that has led to the current strategy of managing coastal marsh areas and the creation of saltmarshes as part of managed realignment schemes. Efforts have been made in England and Wales to identify suitable areas for managed realignment, to estimate the cost of each realignment, and to monitor realignments (DEFRA, 2002; Environment Agency, 2017).

Managed realignment aims to restore saltmarshes by reconstructing new sea walls further inland and deliberately breaching existing fore-shore walls thus allowing tidal inundation of low-lying, often agricultural, land (French, 2006). Such restoration practices have been implemented across the developed Northern hemisphere and, historically, saltmarshes have been reclaimed for either agricultural land or urban development (Adam, 1990). Agricultural practices in reclaimed coastal lands compact and erode soils, and as a result initial elevation of realigned lands are generally below sea level leading to rapid sediment accumulation (Masselink et al., 2017; Oosterlee et al., 2020).

Realigned saltmarshes exhibit different biotic and abiotic characteristics than natural marshes even decades after implementation (Garbutt and Wolters, 2008). Realigned saltmarshes have been shown to experience long term changes to subsurface sediment structure, including collapse of pore space, which leads to poor, or less rapid, marsh drainage (Spencer et al., 2008; Tempest et al., 2015). These changes are not readily reversed once the land is inundated with brackish or seawater, resulting in further downwash of fine particles into the subsurface thus further reducing porosity at depth (Macphail et al., 2010). Restored marshes rapidly become less oxygenated with lowest redox potentials found at lower elevations relative to sea level, sometimes becoming more reduced than sediments in nearby reference natural marshes (Blackwell et al., 2004; Davy et al., 2011; Mossman et al., 2012a). Higher elevations in managed saltmarshes can have drier, more oxygenated sediments relative to natural marshes of the same elevation (Mossman et al., 2012a).

Plant and biodiversity communities in saltmarshes are linked to sediment processes and sub-surface sediment characteristics (Davy et al., 2011; Garbutt and Wolters, 2008; Mossman et al., 2012a, 2012b). Spencer et al. (2017) found that sediment characteristics which influence pioneer species and biodiversity development, including microporosity, pore connectivity and water storage capacity, are often determined by pre-restoration conditions.

Despite the importance of sediments to salt marsh function, a limited number of studies have focused on temporal changes in sediment characteristics within realigned saltmarshes (e.g. Blackwell et al., 2004; Janousek et al., 2021; Masselink et al., 2017; Oosterlee et al., 2020; Spencer et al., 2017, 2008; Tempest et al., 2015; Veenklaas et al., 2015). Many studies have studied salt marshes for limited temporal periods and begin analyses several years after restoration. Changes on short-term time scales, or immediately after breach, are often overlooked even though biotic and abiotic factors that are likely to affect sediment development and therefore vegetation colonization (including sediment pH, salinity, nutrient availability and anoxia (Garbutt et al., 2006, Garbutt and Wolters, 2008, Mossman et al., 2012a,b, Zhou et al., 2016)) are likely to be heavily influenced by initial conditions set within the first year of marsh re-establishment. Of the studies examining the earliest stages of realignment function and form, observed short-term changes (within nine weeks) in pH, conductivity and NH_4^+ concentrations in the topsoil water of a newly realigned marsh while Masselink et al. (2017) primarily focused on water levels over sediment surfaces and the impacts of a self-regulating tidal gate.

This study explores early-stage (<15 months) post-realignment sedimentary processes in two sections (arable versus pasture) of a south eastern UK saltmarsh established on land previously reclaimed for agriculture as compared to a local, natural reference marsh. The objective of the study was to quantify net sediment accumulation immediately following breach and changes in physical and chemical sediment characteristics. We assess the effects of previous land use, in particular ploughing, periodic absence of plant cover and fertilization (arable) versus continuous pasture cover with stock grazing (pasture).

2 Methods

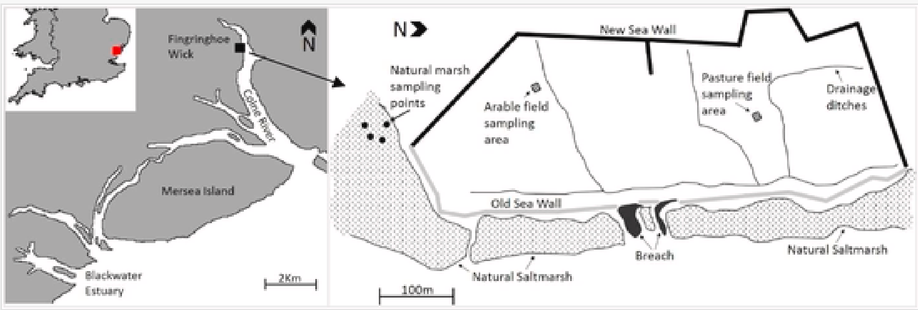
2.1 Study site

The study sites were located within the Fingringhoe Wick Nature Reserve (48.6 ha in total), located in Essex, southeast England. The study site pairs the realigned marsh site (22 ha), which inundation was initiated during this study, with adjacent, natural marsh habitat (8.2 ha). Both natural marsh and realigned marsh are located roughly 8.5 km upstream along the western margin of the Colne river in brackish conditions, and sediment delivery is driven by a combination of river flow and tidal state.

During the course of this study the old sea wall was doubly breached (north and south), to allow tidal inundation of the site from September 2015 (Fig. 1). The double breach intentionally was elevated relative to mean sea level, with the expectation that early inundations would expand and deepen the breach. The implications of this approach meant that the site was fully flooded through high tide inputs for the first several months (September 2015–December 2015) and did not experience incoming and outgoing tidal interactions. The double breach was intended to allow water to flow freely to both northern and southern extents of the realigned fields, however throughout the study period the southern breach was dominant and controlled flow into and out of the site across all tidal ranges. The southern breach elevation reduced by > 0.5m over the course of the study allowing continual tidal flow from January 2016 onwards.

alt-text: Fig. 1

Fig. 1



Location and outline of the new realigned site and natural marsh sampling locations within the Colne Estuary, UK.

The managed realignment site (22 ha) (51°50′25.77″N, 0°58′27.80″E) was previously used for growing wheat and barley in rotation (southern field), and for pasture for grazing by sheep (northern field). The pasture field soils were covered by a thick grass with dense root structure to 8 cm in depth. The arable field was sparsely planted (every 10 cm) with short wheat stubble left after the final harvest before the breach and with relatively limited root density in the top 5 cm. The adjacent natural marsh has creeks and mud pans across its area and a mixed vegetation with the main species being, *Puccinellia maritima, Tripolium pannonicum, Atriplex portulacoides, Limonium vulgare, Suaeda maritima and Spartina anglica.*

Three locations formed the sampling areas for this study, realigned arable field (A), realigned pasture field (B) and adjacent natural mash (C) (mud pans and creeks only). The two locations (arable, A & pasture, B) (each plot was 10 m × 8 m) in the realigned site were measured prior to breaching with a theodolite to be at the same elevation with each other, ± 10 cm. The sampled natural marsh sediments (Fig. 1) are elevated relative to the realigned marshes due to soil erosion and compaction over time in the re-aligned, previously agricultural fields (exact elevation difference was not measured and varied depending on mud pan/creek depth).

2.2 Sampling

A fixed vertical quadrat (60 cm × 100 cm) was placed in each realigned field during November 2015, two months after breach, on the western edge of the sampling areas to measure sediment accretion rate. Rate of accretion was measured monthly thereafter and determined by the difference in the distance from the sediment surface to the top of the vertical quadrat. Ten (10) measurements were taken at 10 cm intervals on the quadrat and their average was used to determine sediment accumulation. The method used is similar to the Surface-elevation change measurement using levelling technique mentioned in Nolte et al. (2013).

Prior-to-breach samples were collected only from the realigned site in March and August 2015. Post-breach sediment samples were collected on a monthly basis, at similar times during the tidal cycle (low tide ±3 h), from October 2015 to November 2016 (excluding December 2015) within the 10 m × 8 m sampling area. No significant differences were found between the two prior-to-breach months (apart from nutrients), thus both samplings were collated and used as a single time point zero reference (except for nutrients where only August was used, although there was a significant variation between the samples for each month). Four sediment cores (4.5 cm inner diameter and ≥20 cm depth) were collected from each location on each sampling date. Core samples were separated into 0–5, 5–10 and 10–15 cm depth sections and used to determine bulk density, sediment water content, sediment nutrient concentration (NH_4^+ and NO_2^-/NO_3^-) and pH. The volume of the core at each depth was adjusted for compression during collection using the difference in depth on the inside and outside of the core before extraction; linear compression was assumed. Cores were stored at 4 °C and processed within 72 h from collection time.

2.3 Core processing

Bulk density and water content samples were taken from the centre (2 cm sub-sections) of each 5 cm (0–5, 5–10, 10–15 cm) section. The top and bottom of each section (e.g. 0–1.5 cm and 3.5–5 cm) were combined and homogenised for nutrient and pH analyses. Any roots present were removed from the sample for the nutrient and pH analyses. Bulk density was determined by drying the central disk at 70 °C for 7 days and calculated by dividing remaining weight by initial (corrected) soil volume. Percent water content was determined from each bulk density sample through measured water loss and is shown as [% water content = (g water/g fresh soil) * 100].

Soil samples for nutrient analysis were prepared according to Houba et al. (1995); 3.0 g of wet soil with 30 ml of 1M KCl, shaken at 200 rpm for 60 min, centrifuged (2000 rpm for 5 min) and the supernatant filtered (Whatman Grade 44 filter, rinsed with DI before sample collection). The filtrate was then stored at −20 °C until analysis. Analyses for NH_4^+ and combined nitrite and nitrate (hereafter NO_2^-/NO_3^-) were performed using a Seal Analytical AutoAnalyzer3 (colorimeter). KCl blanks were run to correct for contamination and/or drifts in extract as well as known concentration standards to ensure the equipment was consistently calibrated. Standards were run every 10 samples. The remaining solid phase from the nutrient samples was used for pH analysis, after in-house analyses demonstrated that pH was consistent between freshly prepared samples and post-extraction samples. In post extraction samples a further 15 ml of 1M KCl solution was added, the samples were placed on a rocker (60 rpm) for 60 min and analysed using a standard pH probe (meter: Jenway 3310, probe: VWR 662–1797). Similar to nutrient analysis standards were used to calibrate the probe before each analysis as well as every 10 samples to ensure no drifts occurred during analysis.

2.4 Statistical analysis

Statistical analyses were conducted using the R statistical language implemented in RStudio (Version 1.1.423). A linear mixed effect model (lmer) (Bates et al., 2015) was used to compare the different sediment characteristics with time (months from breach) between the two realigned sites (arable, pasture) and with the natural marsh. In the mixed effect model sediment accretion was set as a random variable. All variables were tested for normality (Shapiro-Wilk test) and all were normally distributed apart from pH. The pH was transformed out of log scale to meet conditions of normality for statistical analysis. In addition, a time lag analysis was performed to examine whether water content and pH of previous months (up to 8 months prior) affected the concentration of ammonium observed in the sediment. The analysis was performed using a linear mixed effect model (lmer) with accretion set as a random variable for each depth and each field individually; the lag analysis only examines significance up to 8 months prior to allow for sufficient temporal replicates. Significance for both analyses was set at $p < 0.05$.

3 Results

During the sampling period, no saltmarsh vegetation grew within the sampling areas. *Salicornia europaea* and *Suaeda maritima* colonized higher elevation areas within the realigned site but abundance and biomass were not recorded as they were outside (>5 m) our designated sampling areas. The pasture field was still covered with prior-to-breach vegetation (grass) when the breach occurred, however over the first several weeks after the breach it was covered with accreted marine sediment.

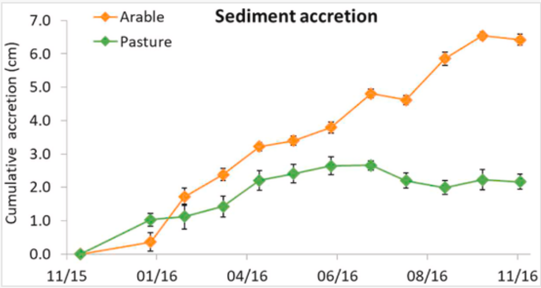
3.1 Sediment accretion

Sediment accretion varied between the two realigned fields with the arable location showing more accretion than the pasture. Total sediment accretion at the last sampling (14 months post-breach, Nov 2016) was 6.4 cm for the arable field and 2.2 cm for the pasture (Fig. 2).

Images are optimised for fast web viewing. Click on the image to view the original version.

alt-text: Fig. 2

Fig. 2



Cumulative sediment accretion in the realigned fields from November 2015 to November 2016. Error bars denote standard error within 10 measurements per time point.

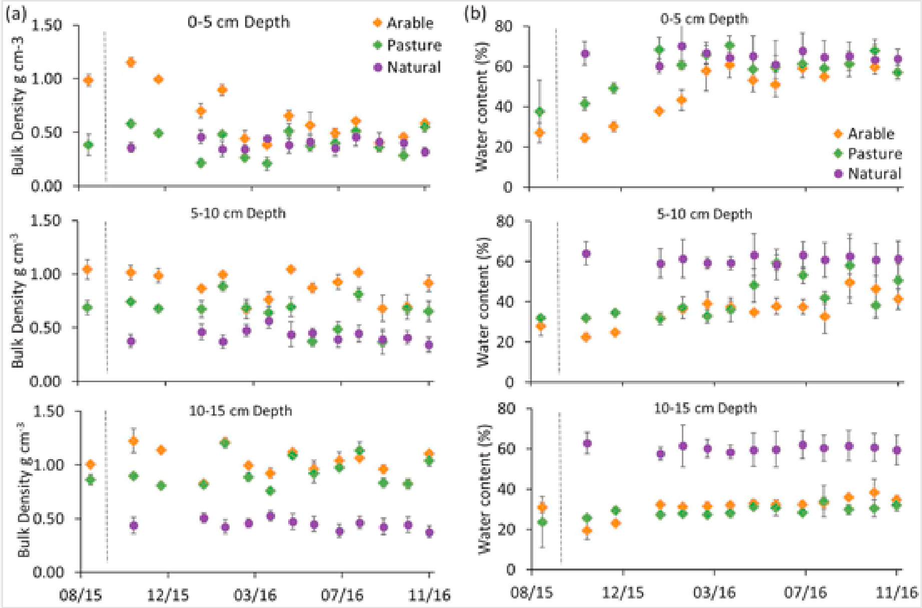
3.2 Bulk density and water content

Bulk density of the top sediments (0–5 cm) within the realigned saltmarsh decreased significantly over 14 months in the arable (from 1.15 to 0.59 g/cm^3, a 51% decrease; $p < 0.001$) and pasture fields (from 0.58 to 0.28 g/cm^3, a 48% decrease; $p < 0.001$) (Fig. 3a). In the 5–10 cm depth range the bulk density between the two fields was significantly different ($p < 0.001$), the density of the arable field decreased over time whereas in the pasture there was little change. There was no significant change in bulk density within the 10–15 cm depth range in either field (Fig. 3a). Bulk density was significantly lower in the natural marsh but over time the top 10 cm within realigned marsh sites eventually approached a similar density. However, the deeper sediments of the realigned sites (>10 cm) remained significantly more dense than those in natural sediments (Fig. 3a).

Images are optimised for fast web viewing. Click on the image to view the original version.

alt-text: Fig. 3

Fig. 3



(a) Bulk density (g/cm3) and (b) Water content (%) over time at each depth in the natural and realigned fields. Breach of sea wall (dotted line) was in Sept'15. Error bars are ±SE, n = 4.

Water content showed similar, but inverse, patterns to those of bulk density patterns in all three fields (Fig. 3b). Water content of the top sediments (0–5 cm) within the realigned saltmarsh increased over time in both fields (arable: from 27 to 60%, pasture: from 38 to 68%, $p < 0.001$) matching natural conditions by the final sampling month. In sediments at 5–10 cm depths, water content was significantly different over time in both fields ($p < 0.001$), increasing towards but not matching natural marsh sediment water content. In sediments at the 10–15 cm depths, no significant change occurred within the realigned fields, which remained at ~30% throughout the sampling period; realigned fields were significantly lower in water content than the natural marsh (~60%) ($p < 0.05$) (Fig. 3b).

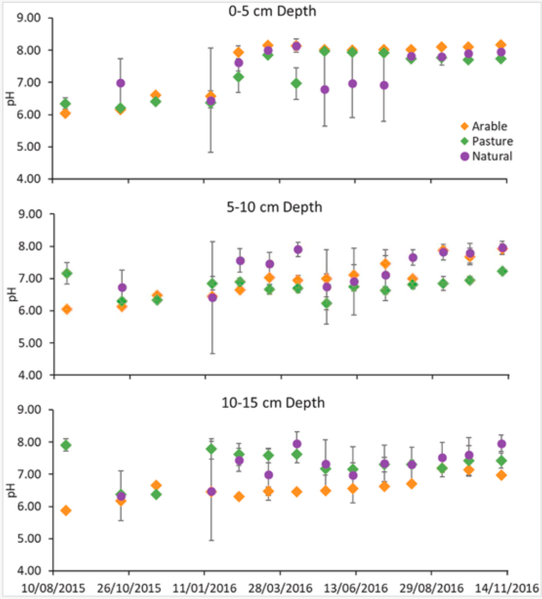
3.3 Sediment pH

The pH of realigned saltmarsh sediments shifted significantly from weakly acidic (between 5.8 and 6) to weakly alkaline (between 7 and 8.2) (Fig. 4), in both fields in the 0–5 cm and 5–10 cm depth ranges ($p < 0.001$). There was no significant difference in those depth ranges between the two fields as they behaved similarly over time. In the 10–15 cm depth range however, there was a significant difference between the two sites over time ($p < 0.001$) with the arable site remaining acidic while the pasture site became more alkaline.

Images are optimised for fast web viewing. Click on the image to view the original version.

alt-text: Fig. 4

Fig. 4



Sediment pH over time at each depth in the natural and realigned fields. Breach of sea wall (dotted line) was in Sept'15. Error bars are ±SE, n = 4.

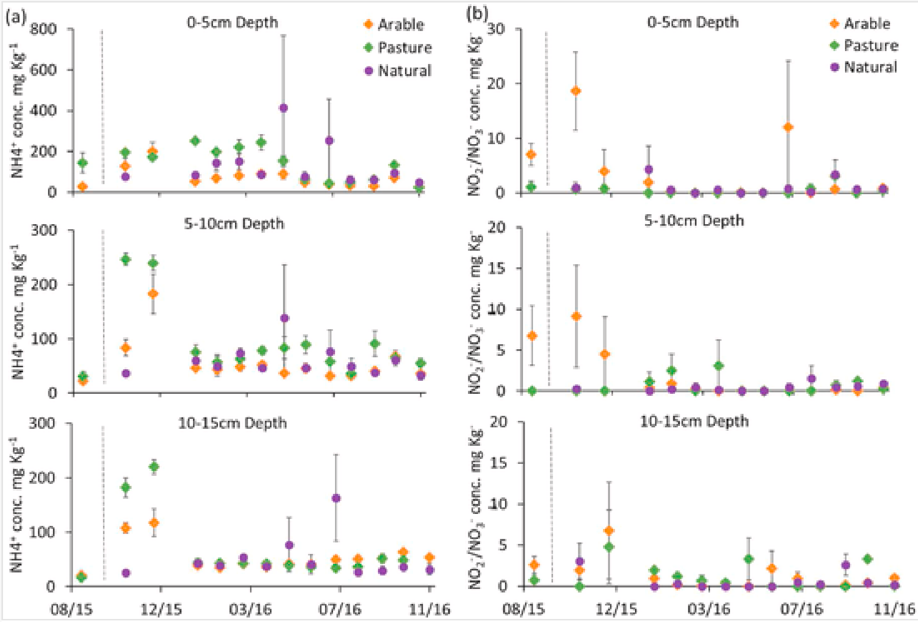
Natural marsh sediments also transitioned from weakly acidic to weakly alkaline over this time frame, although the pH of natural marsh sediments was more variable than realigned sediment during several sampling dates (Jan, May, June, July). The pH readings for natural marsh sediments ranged between 2.95 and 8.30.

3.4 Nutrients

Ammonium (NH_4^+) concentrations in realigned arable sediments increased dramatically after initial inundation (~150 mg/kg in arable and ~200 mg/kg in pasture) and then decreased over time at all depths, with the greatest decrease (~150 mg/kg at both fields) occurring within two months of the post-breach maximum (Nov to Jan) in the 5–10 cm and 10–15 cm depths (Fig. 5a). In the pasture field at the 0–5 cm depth however NH_4^+ concentration continued to increase for 6 months post breach before it began to decrease (Fig. 5a). At the 0–5 cm depth, NH_4^+ was significantly different between each site ($p < 0.001$) and over time ($p < 0.001$). At all depths, the pasture field had greater concentrations of ammonium than the arable field. Natural marsh NH_4^+ concentrations showed similar patterns to realigned sites in fluctuations over time but often with substantially greater variation within site, especially during the May and July sampling periods with readings ranging between 30 and 1470 mg/kg in May and 20 and 850 mg/kg in July.

alt-text: Fig. 5

Fig. 5



(a) Ammonium, NH_4^+ and (b) Nitrate/nitrite, NO_2^-/NO_3^-, concentration in sediment (mg/kg) over time at each depth in the natural and realigned fields. (Note the difference in scales for NH_4^+ and NO_x and the difference in scale in the 0–5 cm depth). Breach of sea wall (dotted line) was in Sept'15. Error bars are ±SE, n = 4.

Nitrate/nitrite (NO_3^-/NO_2^-) concentrations in the sediment were fully depleted by the fourth month after breaching in both fields (Fig. 5b). There was a significant difference only in the 5–10 cm depth between the fields (natural and realigned) over time ($p < 0.001$). Spikes of NO_3^-/NO_2^- concentration (48 mg/kg) were observed in one of our samples of the arable field at the 0–5 cm depth with the other samples being <0.2 mg/kg).

3.5 Lagged environmental drivers for NH_4^+ concentrations

The examination of the potential delayed response of NH_4^+ concentrations to changes in pH, bulk density and water content shows that ammonia concentration was driven by water content and sediment pH (more alkaline led to lower NH_4^+), however the delay period leading to maximum concentrations varied with depth and site.

The ammonia concentrations appear to be driven by pH and water content four months prior to sampling (pH, $p < 0.05$, $r^2 = 0.556$; $\%H_2O$, $p < 0.001$, $r^2 = 0.469$) for the arable field, within the 0–5 cm depth range. While still showing positive correlations with pH and water content, the 5–10 cm depth ammonia content appeared to be most influenced by pH from the same sampling campaign ($p < 0.001$, $r^2 = 0.879$) and water content 8 months prior to sampling ($p < 0.05$, $r^2 = 0.672$). The 10–15 cm depth ammonia concentrations also showed significant correlations with pH and water content, although these were more significant three months (pH $p < 0.01$, $r^2 = 0.583$) and one month prior to sampling ($\%H_2O$ $p < 0.001$, $r^2 = 0.647$), respectively.

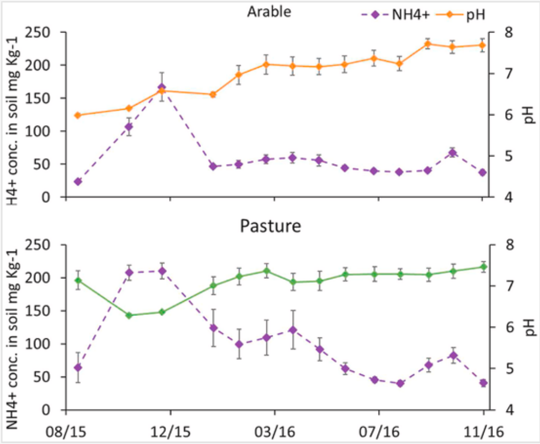
Within the pasture field, the 0–5 cm depth ammonia concentration of sediments was more significantly affected by the sediment pH three months prior to sampling ($p < 0.001$, $r^2 = 0.458$) and water one month prior ($p < 0.001$, $r^2 = 0.701$). In the 5–10 cm depth the ammonia concentration was most strongly correlated with pH of the same sampling campaign and water content from the previous month (pH, $p < 0.001$, $r^2 = 0.864$; H_2O, $p < 0.05$, $r^2 = 0.873$). The 10–15 cm depth ammonia concentrations were best explained by pH and water content of the same sampling campaign (pH, $p < 0.05$, $r^2 = 0.877$; H_2O, $p < 0.01$, $r^2 = 0.919$).

When the lag analysis on NH_4^+ was run without depth separation, pH of the same sampling campaign in both fields showed to affect it most significantly (arable, $p < 0.001$, $r^2 = 0.610$; pasture, $p < 0.001$, $r^2 = 0.350$; Fig. 6). Both arable and pasture fields have shown two distinct time points that were significant for water content's influence on sediment ammonia concentration. For the arable field, water content of the same sampling campaign and 8 months prior (0 months, $p < 0.05$, $r^2 = 0.505$; −8 months, $p < 0.05$), and for the pasture field the same sampling campaign and 1 month prior (0 months, $p < 0.001$; $r^2 = 0.693$; −1 month, $p < 0.001$, $r^2 = 0.740$) (Fig. 7).

Images are optimised for fast web viewing. Click on the image to view the original version.

alt-text: Fig. 6

Fig. 6

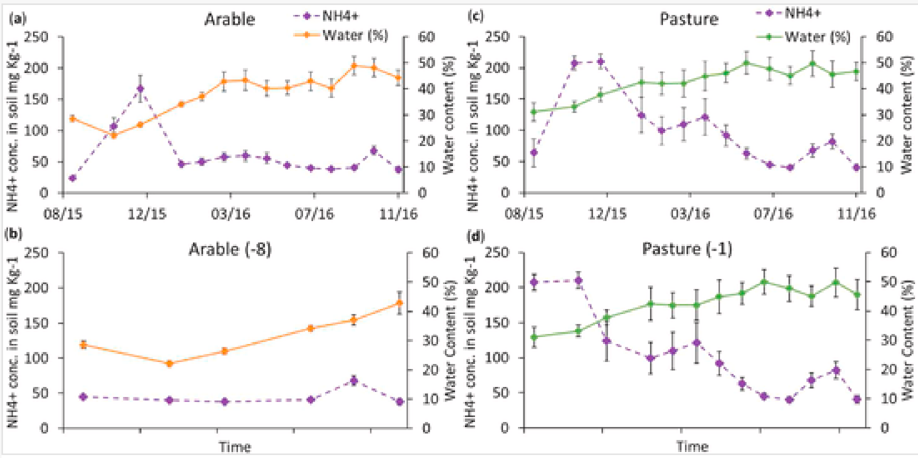


Average ammonium (NH_4^+) concentration in sediment with pH of sediment for the same sampling time point, and all depths collated together. n = 168 for arable and pasture. Error bars ± SE.

Images are optimised for fast web viewing. Click on the image to view the original version.

alt-text: Fig. 7

Fig. 7



Average NH_4^+ (mg/Kg) concentration in sediment with water content (%). (a) arable field for NH_4^+ concentration and water content from same sampling time point (n = 168), (b) arable field with water content of 8 months prior (n = 60), (c) pasture field for NH_4^+ concentration and water content from same sampling time point (n = 168), (d) pasture field with water content of 1 month prior (n = 148). Error bars ± SE.

3.6 Sediment profile

The sediment cores were sampled and analysed to a maximum depth of 15 cm, however due to sediment accretion we did not sample the same location within the original sediment profile over time (Fig. 8). The sediment profile changed as sediment accreted in both fields, resulting in the original sediment surface being shifted lower in depth over time (Fig. 8).

Images are optimised for fast web viewing. Click on the image to view the original version.

alt-text: Fig. 8

Fig. 8

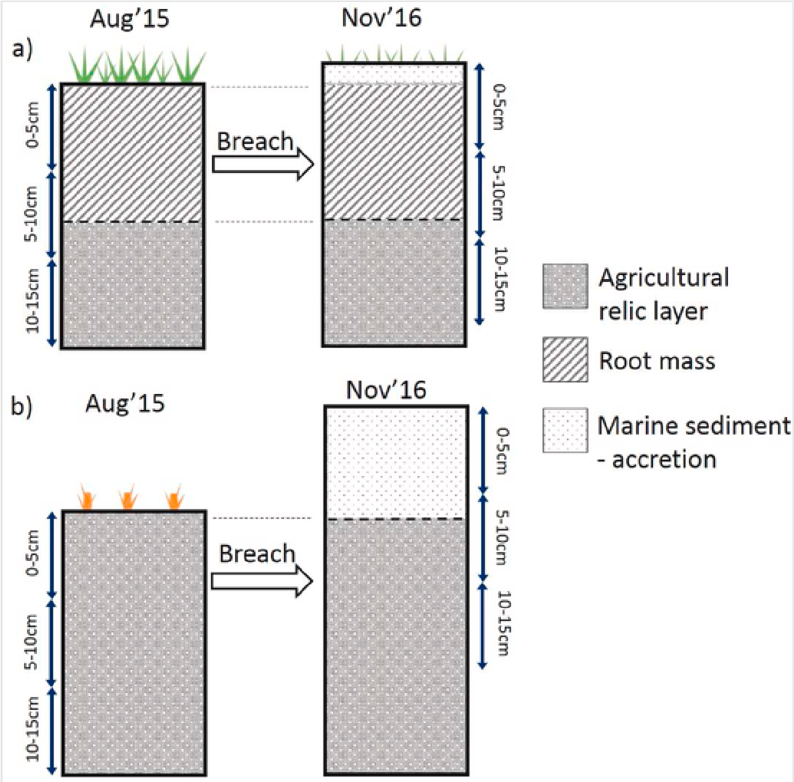


Diagram of the 15 cm sediment profile collected before breach and after 14 months in each site. a) Pasture b) Arable. This shows the difference of the profile that was collected at the two time-points. Horizontal dotted lines indicate where each horizon has remained.

This can more clearly be seen in the arable field. In August 2015 (prior-to-breach) bulk density within the 0–5 cm depth was 0.99g/cm^3, and by November 2016 (final sampling) there was 6.4 cm of sediment accretion and bulk density of the 0–5 cm depth sediments was 0.59 g/cm^3 whereas the 5–10 cm depth sediments retained a bulk density of 0.91 g/cm^3 (Fig. 3).

4 Discussion

In this study we have shown that 14 months post inundation both locations within the realigned saltmarsh sediments matched physiochemical characteristics of the local reference natural marsh but only in the top (newly accreted) sediment. The deeper sediments remained physically unchanged even as nutrient and pH values became similar to those in natural marshes at the same depth.

Realigned salt marshes are being created to restore coastal habitats, occasionally with set functionalities in mind (i.e. coastal protection, biodiversity increase, bird habitat), and success is normally measured against these parameters (Neckles et al., 2002; Strange et al., 2002). For each realignment, recovery and functionality goals are set which determine success of the realignment. However, short-term recoveries and set goals may not imply long term sustainability (Zedler et al., 2001).

4.1 Differences between arable and pasture fields

The differences between the two fields' bulk density at the start of inundation may be primarily attributed to prior use, specifically the pasture field's dense root mass which penetrates down to 8 cm depth. The substantial root presence in the pasture field is likely to be the driving factor for the lower bulk density in the 0–5 cm depth prior to the breach, by favouring fluid transport down the sediment column (Angers and Caron, 1998), in contrast to the dense compact sediment of the arable field. Within the pasture field the extensive root structure may have allowed for more water infiltration to lower sediment depths post breach. Root mediated infiltration may explain the higher water content observed at the 0–5 cm and 5–10 cm depths post breach in pasture sediments.

The cores were sampled and analysed to a maximum depth of 15 cm, however due to sediment accretion we did not sample the same location within the original sediment profile over time (Fig. 8). Thus, when sediments from consistent depths within the prior-use agricultural sediments are compared, we can say that the pre-breach agricultural sediments did not substantially change over time and changes observed are generally from the new sediment accumulating on site. The difference in accretion rates between the two sites may be explained by non-uniform erosion of the double breach (north breach less developed than south) which led to less sediment transported to, and deposited on, the northern field (pasture). The impacts of the breach are also found in the limited accretion at both sites, substantially less than other reported full tidal exchange breached and managed sites (Oosterlee et al., 2020) and closer in sediment accumulation to controlled tidal systems (Masselink et al., 2017; Oosterlee et al., 2020).

The 0–5 cm depth sediments in the November 2016 arable field are representative of the marine accreted sediment and does not represent a change of the agricultural relic layer. This effect is more obvious when we look at the bulk density and water content of sediments found at lower depths (5–10 cm & 10–15 cm) within both fields which did not change significantly over the 14 months of inundation.

The two fields had different starting nutrient concentrations, which could be attributed to their previous land use, where the pasture field was used for grazing by sheep and their excretions increasing the available NH_4^+ within the site (Ma et al., 2007). Post breach, both fields behaved similarly with a rapid decrease of NH_4^+ concentration within the first 2 months, which could be a signal of rapid decomposition of available organic matter (Jordan et al., 1989), and by November 2016 (final sampling) nutrient concentrations were very similar for both field types. The pasture field had greater concentrations of ammonium than the arable field in the top 10 cm, which is likely due to the dense decomposing root mass (Jordan et al., 1989). Spikes of NO_3^-/NO_2^- concentration that were observed which may have been due to bird excrement (Bazely and Jefferies, 1985; Penk et al., 2019), as the newly realigned site was used by bird as feeding grounds.

NH_4^+ and NO_2^-/NO_3^- concentrations in sediment are driven by similar natural factors (i.e. water content and pH) and appear to have similar inundation responses. Higher concentrations of reduction compounds such as ammonia may be indicative of a lower redox potential (Velinsky et al., 2017). We observed that with inundation and higher water content (proxy over time for an anoxic environment) there was an increase in ammonium concentration in the sediment and with the formation of ammonia, pH of the sediment increased to more alkaline.

4.2 Natural vs. realigned marsh

Physical aspects of the realigned sediments, including bulk density and water content matched those of the natural marsh within 14 months post-preach in the 0–5 cm depth range, however the lower depth sediments within realigned sites remained more dense and drier when compared to natural marsh sediments. Similar results were observed by Spencer et al. (2008) and Tempest et al. (2015) at Orplands Farm site (Blackwater Estuary, SE England), where 8 and 17 years after initial inundation the old agricultural relic layer remained unchanged with new marine sediment deposited on top.

Chemical properties of the realigned sediments, including nutrient content and pH matched those of the natural marsh within 14 months post-breach throughout the sediment profile. However, natural marsh sediment pH was more variable between samples (range = 2.95–8.69) when compared against the more homogeneous realigned site sediments pH (range = 5.68–8.49). Nutrients (particularly ammonium) behaved consistently in the realigned site sediments, in that they followed similar patterns across the field over time; but within the natural marsh we observed spikes of very high concentrations of ammonium (1470 mg/kg) and very low (40 mg/kg) during the same sampling time. However, neither field showed the high spike concentrations of ammonium found in previously reported studies (~10 mg/L; Blackwell et al., 2004). Similarly to Burden et al. (2013) it appears that within managed realigned saltmarshes nutrient mineralization rates shift rapidly towards natural marsh rates. Burden et al. (2013) however, concluded that despite reversion of nutrient cycling towards natural rates, C:N ratios remain lower than those in natural sediments decreasing recovery rates of realigned marshes.

Density and moisture of lower sediment depths in the realigned site may play a role in the structural homogeneity of the site since they did not change significantly, likely affecting hydrology withing the managed marshes. Tempest et al. (2015) found that in realigned sites the agricultural relic soils remained the same over time, constricting water movement within the sites. In our realigned sites, the top 5 cm, comprising the newly deposited marine sediment, was where some conditions matched those of the natural marsh, whereas lower depth sediments, especially the deepest depth range of 10–15 cm, showed little change over a 14 month span.

There is a consensus then, that realigned marshes show differences both between each other but also with natural referenced marshes (Garbutt and Wolters, 2008; Garbutt et al., 2006; Lawrence et al., 2018; Mossman et al., 2012a, 2012b; Sullivan et al., 2017; Tempest et al., 2015; Wolters et al., 2005). These differences could be attributed to physicochemical properties of the sediment; such as soil drainage (due to denser lower sediments) (Burden et al., 2013; Spencer et al., 2008; Tempest et al., 2015), nutrient cycling (Burden et al., 2013) and previous land use (Garbutt et al., 2006; Spencer et al., 2017).

5 Summary

This work provides further support to the hypothesis that differences between natural and realigned marshes are attributed to sediment physicochemical properties, and for the first time demonstrates the development of these differences within two realigned sites with prior land use.

Pasture field realignments retain and generate more nutrients (mostly NH_4^+) than arable fields post inundation which could potentially influence development of vegetation. Enhanced plant growth, dependent upon elevated N content within sediments can lead to increased sediment capture, and increase in elevation (Fox et al., 2012). Our realigned sites closely match natural marsh conditions for NO_x and pH at all depths, but bulk density and water content are only similar in the surface 0–5 cm depths, where marine sediments are accreting. Natural marsh sediment however, remained more heterogenous than the realigned sediments cross all depths. The unchanged agricultural layer (in this study >5 cm depth), as also recorded by Tempest et al. (2015), can affect the hydrogeology of realigned marshes, which may provide an explanation for the greater variability of realigned sediments.

Pre-restoration land-use affects the structure of restored salt marshes with implications for functioning and delivering of ecosystem services (Spencer et al., 2017). Success of restoration is measured in the ability of the site to provide the desired ecosystem services, (i.e. biodiversity, coastal protection, habitat creation) (Strange et al., 2002). Short-term recoveries and set goals may not imply long term sustainability (Zedler et al., 2001). Even when species densities within realigned marshes match those of natural marshes (i.e. Garbutt and Wolters, 2008; Mossman et al., 2012b), functional measures often reveal a significant lag of ecological processes recovery, such as nutrient cycling, and microbial communities (Cai, 2018) that are necessary for full functionality of a marsh.

Authors statement

Leda L Cai: Conceptualization, Methodology, Data collection, Data analysis, Writing – original draft, Writing -review and editing.

Thorunn Helgason: Methodology, Writing -review and editing, Supervision.

Kelly R. Redeker: Methodology, Data collection, Writing -review and editing, Supervision.

Q4 ## Uncited references

~~Ford et al., 2016~~; ~~Spencer et al., 2016~~.

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.

Acknowledgements

Q3 The authors thank the staff of the Essex Wildlife Trust, particularly Andrew May and those at Fingringhoe Wick, for access, support, and advice. We would like to thank everyone who helped during field work especially Thomas Rudd. The authors thank Rebecca Sutton and Sylvia Toet for their assistance and use of the facilities at the Environment Department of University of York. Breach of the realigned site was performed by the Environment Agency. The research was supported by Natural Environment Research Council award no. NE/K01546X/1.

References

The corrections made in this section will be reviewed and approved by a journal production editor. The newly added/removed references and its citations will be reordered and rearranged by the production team.

Adam, P., 1990. Saltmarsh Ecology. Cambridge Univeristy Press.

Angers, D.A., Caron, J., 1998. Plant-induces changes in soil structure: processes and feedbacks. Biogeochemistry 42, 55–72. doi:10.1023/A:1005944025343.

Bates, D., Mächler, M., Bolker, B., Walker, S., 2015. Fitting linear mixed-effects models using lme4. J. Stat. Software 67, 1–48. doi:10.18637/jss.v067.i01.

Bazely, D.R., Jefferies, R.L., 1985. Goose faeces : a source of nitrogen for plant growth in a grazed salt marsh. J. Appl. Ecol. 22, 693–703.

Beaumont, N.J., Austen, M.C., Mangi, S.C., Townsend, M., 2008. Economic valuation for the conservation of marine biodiversity. Mar. Pollut. Bull. 56, 386–396. doi:10.1016/j.marpolbul.2007.11.013.

Blackwell, M.S.A., Hogan, D.V., Maltby, E., 2004. The short-term impact of managed realignment on soil environmental variables and hydrology. Estuar. Coast Shelf Sci. 59, 687–701. doi:10.1016/j.ecss.2003.11.012.

Boorman, L.A., 2003. Saltmarsh Review. An Overview of Coastal Salt Marshes, Their Dynamic and Sensitivity Characteristics for Conservation and Management. JNCC Report.

Burden, A., Garbutt, R.A., Evans, C.D., Jones, D.L., Cooper, D.M., 2013. Carbon squestration and biogeochemical cycling in a salt marsh subject to coastal managed realignment. Estuar. Coast Shelf Sci. 120, 12–20.

Cai, L.L., 2018. Saltmarsh Restoration: the Shift from a Terrestrial to a Marine Environment. University of York, UK.

Davy, A.J., Brown, M.J.H., Mossman, H.L., Grant, A., 2011. Colonization of a newly developing salt marsh: disentangling independent effects of elevation and redox potential on halophytes. J. Ecol. 99, 1350–1357. doi:10.1111/j.1365-2745.2011.01870.x.

DEFRA, 2002. Managed Realignment Review Project Report. August 2002.

Environment Agency, 2017. Managing Flood and Coastal Erosion Risks in England: 1 April 2011 to 31 March 2012 1–30.

~~Ford, H., Garbutt, A., Ladd, C., Malarkey, J., Skov, M.W., 2016. Soil stabilization linked to plant diversity and environmental context in coastal wetlands. J. Veg. Sci. 27, 259–268. doi:10.1111/jvs.12367.~~

Fox, L., Valiela, I., Kinney, E.L., 2012. Vegetation cover and elevation in long-term experimental nutrient-enrichment plots in great sippewissett salt marsh, cape cod, Massachusetts: implications for eutrophication and sea level rise. Estuar. Coast 35, 445–458. doi:10.1007/s12237-012-9479-x.

French, P.W., 2006. Managed realignment - the developing story of a comparatively new approach to soft engineering. Estuar. Coast Shelf Sci. 67, 409–423. doi:10.1016/j.ecss.2005.11.035.

Garbutt, A., Wolters, M., 2008. The natural regeneration of salt marsh on formerly reclaimed land. Appl. Veg. Sci. 11, 335–344. doi:10.3170/2008-7-18451.

Garbutt, R.A., Reading, C.J., Wolters, M., Gray, A.J., Rothery, P., 2006. Monitoring the development of intertidal habitats on former agricultural land after the managed realignment of coastal defences at Tollesbury, Essex, UK. Mar. Pollut. Bull. 53, 155–164. doi:10.1016/j.marpolbul.2005.09.015.

Houba, V.J.G., Van der Lee, J.J., Novozinsky, I., 1995. In: Soil Analysis Procedures, Other Procedures, 5B. Wageningen Agricultural University, Wageningen.

Hughes, R.G., Paramor, O.A.L., 2004. On the loss of saltmarshes in south-east Engalnd and methods for their restoration. J. Appl. Ecol. 41, 440–448. doi:10.1111/j.0021-8901.2004.00915.x.

Janousek, C.N., Bailey, S.J., Brophy, L.S., 2021. Early ecosystem development varies with elevation and pre-restoration land use/land cover in a pacific northwest tidal wetland restoration project. Estuar. Coast 44, 13–29. doi:10.1007/s12237-020-00782-5.

Jordan, T.E., Whigham, D.F., Correll, D.L., 1989. The role of litter in nutrient cycling in a brackish tidal marsh. Ecology 70, 1906–1915. doi:10.2307/1938121.

Lawrence, P.J., Smith, G.R., Sullivan, M.J.P., Mossman, H.L., 2018. Restored saltmarshes lack the topographic diversity found in natural habitat. Ecol. Eng. 115, 58–66. doi:10.1016/j.ecoleng.2018.02.007.

Ma, X., Wang, S., Jiang, G., Haneklaus, S., Schnug, E., Nyren, P., 2007. Short-term effect of targeted placements of sheep excrement on grassland in Inner Mongolia on soil and plant parameters. Commun. Soil Sci. Plant Anal. 38, 1589–1604. doi:10.1080/00103620701378516.

Macphail, R.I., Allen, M.J., Crowther, J., Cruise, G.M., Whittaker, J.E., 2010. Marine inundation: effects on archaeological features, materials, sediments and soils. Quat. Int. 214, 44–55. doi:10.1016/j.quaint.2009.10.020.

Masselink, G., Hanley, M.E., Halwyn, A.C., Blake, W., Kingston, K., Newton, T., Williams, M., 2017. Evaluation of salt marsh restoration by means of self-regulating tidal gate – avon estuary, South Devon, UK. Ecol. Eng. 106, 174–190. doi:10.1016/j.ecoleng.2017.05.038.

McLeod, E., Chmura, G.L., Bouillon, S., Salm, R., Björk, M., Duarte, C.M., Lovelock, C.E., Schlesinger, W.H., Silliman, B.R., 2011. A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO2. Front. Ecol. Environ. 9, 552–560. doi:10.1890/110004.

Mcowen, C., Weatherdon, L., Bochove, J.-W., Sullivan, E., Blyth, S., Zockler, C., Stanwell-Smith, D., Kingston, N., Martin, C., Spalding, M., Fletcher, S., 2017. A global map of saltmarshes. Biodivers. Data J. 5, e11764. doi:10.3897/BDJ.5.e11764.

Millenium Ecosystem Assessment, 2005. Ecosystem and Human Well-Being: Wetlands and Water Synthesis. World Resources Institue. doi:10.1017/CBO9781107415324.004.

Möller, I., Spencer, T., 2002. Wave dissipation over macro-tidal saltmarshes: effects of marsh edge typology and vegetation change. J. Coast Res. 36, 506–521 https://doi.org/ISSN:0749-0208.

Möller, I., Spencer, T., French, J.R., Leggett, D.J., Dixon, M., 1999. Wave transformation over saltmarshes: a field and numerical modelling study from North Norfolk, England. Estuar. Coast Shelf Sci. 49, 411–426. doi:10.1006/ecss.1999.0509.

Mossman, H.L., Brown, M.J.H., Davy, A.J., Grant, A., 2012a. Constraints on salt marsh development following managed coastal realignment: dispersal limitation or environmental tolerance? Restor. Ecol. 20, 65–75. doi:10.1111/j.1526-100X.2010.00745.x.

Mossman, H.L., Davy, A.J., Grant, A., 2012b. Does managed coastal realignment create saltmarshes with "equivalent biological characteristics" to natural reference sites? J. Appl. Ecol. 49, 1446–1456. doi:10.1111/j.1365-2664.2012.02198.x.

Neckles, H.A., Dionne, M., Burdick, D.M., Roman, C.T., Buchsbaum, R., Hutchins, E., 2002. A monitoring protocol to assess tidal restoration of salt marshes on local and regional scales. Restor. Ecol. 10, 556–563. doi:10.1046/j.1526-100X.2002.02033.x.

Nicholls, R.J., 2011. Planning for the impacts of sea level rise. Oceanography 24, 144–157. doi:10.5670/oceanog.2011.34.

Nolte, S., Koppenaal, E.C., Esselink, P., Dijkema, K.S., Schuerch, M., De Groot, A.V., Bakker, J.P., Temmerman, S., 2013. Measuring sedimentation in tidal marshes: a review on methods and their applicability in biogeomorphological studies. J. Coast Conserv. 17, 301–325. doi:10.1007/s11852-013-0238-3.

Oosterlee, L., Cox, T.J.S., Temmerman, S., Meire, P., 2020. Effects of tidal re-introduction design on sedimentation rates in previously embanked tidal marshes. Estuar. Coast Shelf Sci. 244, 106428. doi:10.1016/j.ecss.2019.106428.

Penk, M.R., Wilkes, R., Perrin, P.M., Waldren, S., 2019. Nutrients in saltmarsh soils are weakly related to those in adjacent coastal waters. Estuar. Coast 42, 675–687. doi:10.1007/s12237-018-00486-x.

Schuerch, M., Spencer, T., Temmerman, S., Kirwan, M.L., Wolff, C., Lincke, D., McOwen, C.J., Pickering, M.D., Reef, R., Vafeidis, A.T., Hinkel, J., Nicholls, R.J., Brown, S., 2018. Future response of global coastal wetlands to sea-level rise. Nature 561, 231–234. doi:10.1038/s41586-018-0476-5.

Spencer, K.L., Carr, S.J., Diggens, L.M., Tempest, J.A., Morris, M.A., Harvey, G.L., 2017. The impact of pre-restoration land-use and disturbance on sediment structure, hydrology and the sediment geochemical environment in restored saltmarshes. Sci. Total Environ. 587–588, 47–58. doi:10.1016/j.scitotenv.2016.11.032.

Spencer, K.L., Cundy, A.B., Davies-Hearn, S., Hughes, R., Turner, S., MacLeod, C.L., 2008. Physicochemical changes in sediments at Orplands Farm, Essex, UK following 8 years of managed realignment. Estuar. Coast Shelf Sci. 76, 608–619. doi:10.1016/j.ecss.2007.07.029.

~~Spencer, T., Möller, I., Rupprecht, F., Bouma, T.J., van Wesenbeeck, B.K., Kudella, M., Paul, M., Jensen, K., Wolters, G., Miranda-Lange, M., Schimmels, S., 2016. Salt marsh surface survives true-to-scale simulated storm surges. Earth Surf. Process. Landforms 41, 543–552. doi:10.1002/esp.3867.~~

Strange, E., Galbraith, H., Bickel, S., Mills, D., Beltman, D., Lipton, J., 2002. Determining ecological equivalence in service-to-service scaling of salt marsh restoration. Environ. Manage. 29, 290–300. doi:10.1007/s00267-001-0019-X.

Sullivan, M.J.P., Davy, A.J., Grant, A., Mossman, H.L., 2017. Is saltmarsh restoration success constrained by matching natural environments or altered succession? A test using niche models. J. Appl. Ecol. 55, 1207–1217. doi:10.1111/1365-2664.13033.

Tempest, J.A., Harvey, G.L., Spencer, K.L., 2015. Modified sediments and subsurface hydrology in natural and recreated salt marshes and implications for delivery of ecosystem services. Hydrol. Process. 29, 2346–2357. doi:10.1002/hyp.10368.

Veenklaas, R.M., Koppenaal, E.C., Bakker, J.P., Esselink, P., 2015. Salinization during salt-marsh restoration after managed realignment. J. Coast Conserv. 19, 405–415. doi:10.1007/s11852-015-0390-z.

Velinsky, D.J., Paudel, B., Quirk, T., Piehler, M., Smyth, A., 2017. Salt marsh denitrification provides a significant nitrogen sink in barnegat bay, New Jersey. J. Coast Res. 78, 70–78. doi:10.2112/SI78-007.1.

Wolters, M., Garbutt, A., Bakker, J.P., 2005. Salt-marsh restoration: evaluating the success of de-embankments in north-west Europe. Biol. Conserv. 123, 249–268. doi:10.1016/j.biocon.2004.11.013.

Zedler, J.B., Callaway, J.C., Sullivan, G., 2001. Declining biodiversity: why species matter and how their functions might Be restored in californian tidal marshes. Bioscience 51, 1005. doi:10.1641/0006-3568(2001)051[1005:DBWSMA]2.0.CO;2.

Zhou, M., Butterbach-Bahl, K., Vereecken, H., Bruggemann, N., 2016. A meta-analysis of soil salinization effects on nitrogen pools, cycles and fluxes in coastal ecosystems. Global Change Biol. 23, 1338–1352. doi:10.1111/gcb.13430.

Highlights

- Old agricultural relic sediment remains unchanged in realigned saltmarshes.
- Pre-restoration land-use can affect the structure of restored salt marshes.
- Functional measures often reveal a significant lag of ecological processes recovery.

Queries and Answers

Q1

Query: Please confirm that the provided **email** “ledalcaiphd@gmail.com” is the correct address for official communication, else provide an alternate e-mail address to replace the existing one, because private e-mail addresses should not be used in articles as the address for communication.

Answer: Reviewed

Q2

Query: The **citation** ‘Tempest et al. (2013)’ has been changed to match the date in the reference list. Please check here and in subsequent occurrences, and correct if necessary.

Answer: Reviewed

Q3

Query: Have we correctly interpreted the following funding source(s) and country names you cited in your article: Natural Environment Research Council?

Answer: Yes. The research was supported by the grand number NE/K01546X/1. And all acknowlegments are made

Q4

Query: The **Uncited References** section comprises references that occur in the reference list but are not available in the body of the article text. Please cite each reference in the text or, alternatively, delete it. Any reference not dealt with will be retained in this section.

Answer: Done

Q5

Query: Please confirm that **given names and surnames** have been identified correctly and are presented in the desired order and please carefully verify the spelling of all authors' names.

Answer: Reviewed

Q6

Query: Your article is registered as a regular item and is being processed for inclusion in a regular issue of the journal. If this is NOT correct and your article belongs to a Special Issue/Collection please contact y.moreyra@elsevier.com immediately prior to returning your corrections.

Answer: Yes