https://doi.org/10.5194/egusphere-2026-1559
Preprint. Discussion started: 2 April 2026

Organic matter composition and origin differ between restored and natural UK saltmarshes up to 100 years after the breach

Authors

Catrina Gore[1], Peter Mueller[2], Mark Chapman[1], Franck Dal Molin[3], Gerhard Schmiedl[4,5], Yvonne Milker[4,5], Lisa Benson[3], Stefanie Nolte[1,6]

Affiliations

[1]School of Environmental Sciences, University of East Anglia, Norwich, NR4 7TJ, United Kingdom

[2]Institute for Environmental Sciences, RPTU Kaiserslautern-Landau, 76829 Landau, Germany

[3]Centre for Environment, Fisheries and Aquaculture Science, Lowestoft NR33 0HT, United Kingdom

[4]Department of Earth System Sciences, University of Hamburg, 20146, Hamburg, Germany

[5]Earth and Society Research Hub (ESRAH), University of Hamburg, 20146, Hamburg, Germany

[6]Lower Saxon Wadden Sea National Park Authority, Virchowstr. 1, 26382 Wilhelmshaven, Germany

Correspondence: Catrina Gore (c.gore@uea.ac.uk)

Abstract

Atmospheric carbon dioxide uptake by restored saltmarshes can be converted to carbon credits that are sold to offset emissions and provide funding for restoration. While carbon stocks in restored marshes were found to reach equivalence with natural sites, the shares of autochthonous and allochthonous carbon sources entering these marshes are less well known. This question has consequences for carbon crediting because only the former can be attributed to the restoration and legitimately included. To determine if differences do exist, the conventional tools for determining organic matter source – elemental, stable isotope and thermogravimetric analyses – were applied to sediments from four restored saltmarshes of different ages. Comparisons were then made with existing data from nearby natural marshes. Breach depth horizons in cores were successfully identified using foraminifera, unicellular marine protists. The C/N-ratio was persistently lower in restored saltmarshes compared with natural counterparts even one hundred years after the creation of a breach. Additionally, δ^{13}C signals indicated a mixture of organic matter sources to the restored sites and there was often a higher proportion, based upon thermal sensitivity, of recalcitrant material in the organic matter of the restored marshes. Together, these results indicated that a higher proportion of allochthonous organic matter that had undergone the processes of decomposition in an external setting, was entering these restored marshes. Understanding the composition of the organic matter in restored marsh sediment will allow more useful and constructive climate change mitigation methods to be developed.

Keywords: Foraminifera; stable isotopes; radiometric dating; carbon stocks; blue carbon; natural climate solutions, nature-based solutions; de-embankment; managed realignment

1. Introduction

Saltmarshes represent the interface between terrestrial and marine ecosystems, and breaching the seawalls surrounding areas previously claimed for agriculture (i.e. de-embankment) is a method of saltmarsh restoration (French, 2006; Garbutt and Wolters, 2008). Originally, the main aim of de-embankment in the UK was coastal protection, with nature conservation benefits also being emphasised (Pethick, 2001; Wolters *et al*., 2005). Recently, more attention is being placed upon the increased atmospheric carbon dioxide (CO_2) removal resulting from this land use change (Connor *et al*., 2001; Jones *et al*., 2024). Currently, a 'Saltmarsh Code' is in development in the UK that will allow this increase to be quantified and converted to carbon credits that can be sold as a means of funding restoration and of offsetting the emissions of the buyer, further emphasising the importance of this issue (Burden *et al*., 2023). It is the capacity of saltmarshes to continuously accrete sediment and build elevation, in combination with regular tidal inundation producing waterlogged conditions, that creates the sink into which organic matter (OM) is deposited. Part of this OM is preserved, thus ensuring the sequestration of the carbon it contains. Rapid rates of carbon accumulation have been recorded in the years immediately following the creation of a breach (Wollenberg *et al*., 2018; Mossman *et al*., 2022) and organic carbon (OC) stock levels have recovered in as little as 4 to 20 years after restoration in some instances (Santini *et al*., 2019; Poppe and Rybczyk, 2021). High levels of variability within and between restored sites, however, mean there is much uncertainty about when and if restored saltmarshes come to replicate the conditions of natural long-standing habitats of the same type.

https://doi.org/10.5194/egusphere-2026-1559
Preprint. Discussion started: 2 April 2026

The differences shown to exist between restored saltmarshes and natural reference sites are manifold. Agricultural soil layers remain in restored saltmarshes after the creation of the breach, and these relatively compacted layers persist as saltmarsh sediment accumulates on top (Tempest *et al*., 2015). Ultimately, water is unable to penetrate these layers and drainage is impacted (Spencer *et al*., 2017). In turn, the development of topographic variation is prevented and characteristics like elevation and creek density continue to resemble those of agricultural fields and lack the diversity present in natural marshes (Shafer and Streever, 2000; Brooks *et al*., 2015; Lawrence *et al*., 2018). Waterlogging and a lack of elevation gradients mean that plant community diversity is often lower in restored saltmarshes because the small-scale variation required for a mosaic of plant species with different inundation frequency requirements is missing (Mossman *et al*., 2012a; Mossman *et al*., 2012b; Lawrence *et al*., 2022). Given these differences, the potential for differing carbon dynamics in restored saltmarsh is strong.

One key factor in saltmarsh carbon dynamics is the origin of the OM. As well as sediment, OM from outside the marsh, be it from marine, coastal or riverine sources, is delivered by tides, and this is termed allochthonous (Zhou *et al*., 2007). Whether or not this OM should be included in carbon crediting schemes is debated because its storage cannot be directly linked to the restoration action undertaken (Needelman *et al*., 2018; Houston *et al*., 2024; Williamson *et al*., 2025). On the other hand, OM produced *in-situ* (i.e., autochthonous) by the growth of saltmarsh plants provides a carbon input that can only be attributed to the restoration. The contribution of allochthonous OM to saltmarsh stocks is highly variable (Williamson *et al*., 2025). However, in many cases it represents a significant portion, especially in minerogenic marshes (e.g., Gebrehiwet *et al*., 2008; Wang *et al*., 2011; Martins *et al*., 2022; Krause *et al*., 2022; Fu *et al*., 2023; Benjamin *et al*., 2024). This is important because the autochthonous input can be remineralised over a decadal timescale while the allochthonous input is more effectively preserved (Bartholdy *et al*., 2014; Van de Broek *et al*., 2016; Mueller *et al*., 2019a; Komada *et al*., 2022). Therefore, OM provenance is a central question to address when comparing restored marshes with natural counterparts.

Common tools to determine the possible mixture of allochthonous and autochthonous components present in saltmarshes are the ratio of OC to nitrogen (N) and the $\delta^{13}C$ of OC in the sediment (e.g., Jiménez-Arias *et al*., 2020; Xia *et al*., 2021; de los Santos *et al*., 2022). The former is useful due to the high level of protein in marine algae producing a large N content that lowers the ratio compared with terrestrial plants that contain C rich cellulose (Meyers, 1994; Zhou *et al*., 2007; Hulisz *et al*., 2013). Depending on the origin of the OM (i.e., higher plants or algae) and the photosynthetic pathway being used (i.e., C_3 or C_4) different amounts of the heavier carbon isotope are incorporated into tissues during growth, producing distinctive signatures that can be identified with the latter method (Boutton *et al*., 1998; Mueller *et al*., 2019b). Three signals with relevance in saltmarsh ecosystems are marine particulate OM (POM)/algae, terrestrial plants (C_3; be they from a saltmarsh or agricultural field) and Spartina (C_4). The $\delta^{13}C$ values of these sources tend to become increasingly enriched in the order listed. This is rarely a simple task, and consideration must be given to whether the signals at different depth along a sediment core are indicative of a change in OM source or if variation is simply due to OM decomposition (Luk *et al*., 2020). Another important method for characterising the OM present in saltmarshes is via thermogravimetric analysis, whereby the amounts of labile or recalcitrant OM in a sediment sample can be differentiated and quantified (Smeaton and Austin 2022). Labile OM refers to simple molecules that are easily broken down while recalcitrant OM is more complex and stable. These methods are combined to achieve a full and insightful picture of

https://doi.org/10.5194/egusphere-2026-1559
Preprint. Discussion started: 2 April 2026

saltmarsh carbon dynamics and can be used to make comparisons between restored and natural marshes. However, caution is needed when explaining any differences.

A crucial first step when measuring the properties of restored saltmarshes is to differentiate between the pre- and post-breach sediment layers. In addition to making accurate comparisons with natural marshes, this is particularly important when calculating carbon stocks and sequestration rates because of the need to determine the land use of the site at the time of OM deposition (Smeaton *et al.*, 2023). Radiometric dating is often applied to saltmarshes but in the case of restored sites this is often not feasible due to the young age of a site and/or disruption of sediment layers during site construction. Previous efforts have successfully used diatoms and moisture content to determine the breach depth (Spencer *et al*. 2008). Problems with using diatoms, however, include poor preservation making it necessary to identify remains, instead of full valves, and the impossibility of any identification in samples containing coarse silt and sand (Spencer *et al.*, 2008). Foraminifera, unicellular marine protists (Sen Gupta, 2003), have previously been used to find the transition in sediment cores from tidal flat to saltmarsh ecosystems (Mueller *et al.*, 2019a). The potential for using foraminifera to differentiate between the horizons in restored marsh cores is clear from the importance of salinity for most species. However, the usefulness of this method is yet to be investigated.

The aim of this study is to compare restored and natural saltmarshes with a view to identifying and explaining consistent differences in biogeochemical variables and thus determine the potential impact of any differences upon the efficacy of using restoration as a climate change mitigation strategy. We took sediment cores from four restored saltmarshes in the UK and combined the new data we collected with an existing database of natural marsh reference sites. To achieve our overarching goal, the first objective was to determine the depth of the breach in the sediment cores using foraminifera and, in doing so, establishing the reliability of this method. When measuring the biogeochemical variables of interest, it was possible to know which core sections were representative of saltmarsh. We also used the breach depth to calculate carbon stocks for the post restoration sections of the cores to demonstrate the potential usefulness of this technique for carbon crediting schemes. Finally, we make recommendations about how these differences can be understood and utilised.

2. Study Sites

2.1 Restored Saltmarshes

The focus of the study is the east of England (Fig 1a & 1b), where saltmarshes are a common feature along the coastline that provide biodiversity and coastal protection and where several restoration projects have already been implemented (Boorman, 2003).

Often, studies aiming to investigate the development of restored saltmarshes over time turn for examples of older marshes to areas with storm created breaches in their seawalls, given the relatively recent development of deliberate restoration. Differences in the sediment characteristics – such as microporosity and pore connectivity – resulting from the presence of the agricultural soil layer were not identified in sites breached accidentally during storm surges (Spencer *et al.*, 2017); however, plant community differences, such as an absence of perennials and the predominance of the shrub *Atriplex portulacoides* have been identified (Mossman *et al.*, 2012b). Furthermore, recently created unmanaged sites were found to lack topographical diversity (Williams and Dale, 2023). Overall, this suggests that other persistent differences

https://doi.org/10.5194/egusphere-2026-1559
Preprint. Discussion started: 2 April 2026

could exist in unmanaged sites that represent the future development of more recently restored marshes.

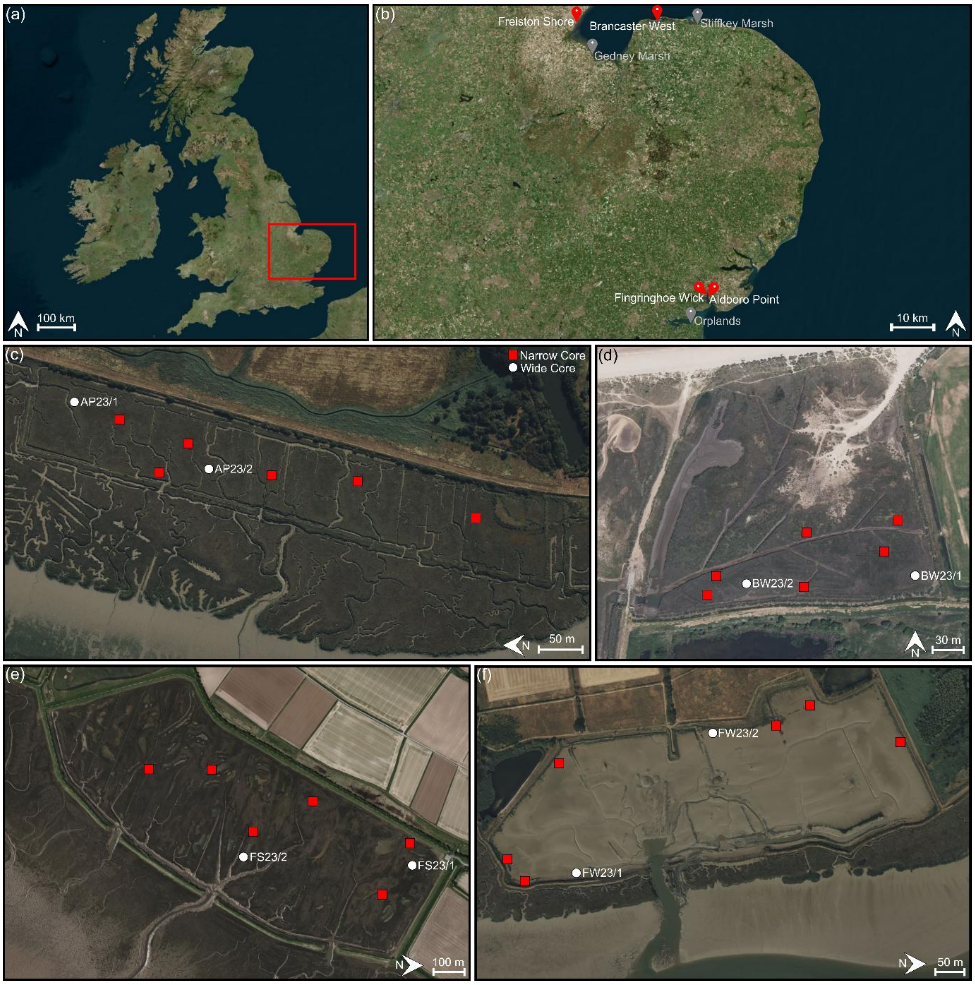

Figure 1. Locations of the sampling sites within (a) the UK and (b) the east of England and of the narrow (red squares) and wide (white circles) cores taken from (c) Aldboro Point, (d) Brancaster West, (e) Freiston Shore and (f) Fingringhoe Wick. The red markers in panel b represent the marshes that were sampled as a part of this study while the grey markers are the natural marsh reference sites sampled by Smeaton and colleagues (2023; 2024). All images are from Bing (https://www.bing.com/maps, image copyright DigitalGlobe, 2020).

2.1.1 Aldboro Point

Aldboro Point lies within the Colne estuary, approximately 12 km southeast of Colchester (51.832375, 0.986063). This marsh represents the oldest “restored” marsh included in the study, with tidal inundation reintroduced in 1921 when a storm created a breach in the seawall

(Burd, 1992; Burd *et al*., 1994; Wolters *et al*., 2005). The reported history of the area prior to the final breach suggested very inconsistent conditions. Enclosure with field boundaries occurred after 1840, by 1874 no seawall was present and by the 1896 Ordinance Survey (OS) mapping indicates the area was again enclosed (Burd *et al*., 1992). It is not definitively known when the final breach occurred, but the 1921 OS map showed a saltmarsh. For simplicity in this study, the breach was assumed to have occurred at the latest possible date, in 1921, as in other studies (e.g., Atkinson *et al*., 2001; van Oevelen *et al*., 2000; Wolters *et al*., 2005). Therefore, at the time of sampling, the marsh was assumed to be 102 years old. The plant species recorded in 2023 at the selected sampling locations (Fig. 1c; Table S1) included *Puccinellia maritima*, *A. portulacoides*, *Limonium vulgare* and small patches of *Spartina anglica* (Table S2). The tidal range is 4.5 m (Wolters *et al*., 2005).

Monitoring of the vegetation at Aldboro Point in 1993 (72 years after the breach; Burd *et al*., 1994) showed that species diversity was low, with only 18 % of the "target" species present (Wolters *et al*., 2005). The most abundant species (41-60 % cover) were *Suaeda maritima* and *P. maritima*. Another study using data collected in 2004 (83 years after the breach) found that species richness in Aldboro Point did not differ significantly from a nearby natural reference marsh but that the plant community in the former was dominated by *A. portulacoides* and the latter by *P. maritima* (Garbutt and Wolters, 2008). Data from the monitoring project of Burd and colleagues (1994) was used to calculate an initial sediment accretion rate of 3.00 mm yr^{-1} and an average of 7.22 mm yr^{-1} by another project (van Oevelen *et al*., 2000).

2.1.2 Freiston Shore

Freiston Shore is located within The Wash, ~8 km east of Boston in Lincolnshire (52.965328, 0.092937). The most recent artificial embankment in Freiston Shore was constructed in 1980 and the three breaches in the seawall were deliberately created in 2002, making the site 21 years old during sampling (Symonds and Collins, 2007). At the selected sampling locations (Fig. 1e; Table S1), the plant species present predominantly included *A. portulacoides*, *P. maritima*, and *Salicornia* spp., with smaller quantities of *S. anglica*, *L. vulgare* and *Aster tripolium* (Table S2). The tidal range experienced by the site is 6.5 m (Wolters *et al*., 2005).

Unlike the other study sites, Freiston Shore has been the subject of many investigations looking at a variety of factors. For example, it was reported that 88 % of the restoration site was vegetated only 4 years after breach creation (Freiss *et al*., 2012), that substantial channel erosion occurred at certain locations (Freiss *et al*., 2014) and that the restored marsh had a wave amplification, as opposed to attenuation, effect (Kiesel *et al*., 2019). Especially relevant to this study was a previous investigation of sediment accretion rates between 2002 and 2006 (Spencer *et al*., 2012). The study by Wolters and colleagues (2005) referred to in sect. 2.1.1 also included Freiston Shore and they reported that this site contained the same proportion (18 %) of target species as Aldboro Point, but only one year after the breach. Furthermore, ten years after the initial breach was created in Freiston, topographic manipulation was carried out to increase habitat diversity and this led to an increase of plant species beta-diversity (Lawrence *et al*., 2022).

2.1.3 Brancaster West

The third selected site, Brancaster West, is situated on the North Norfolk coast, around 15 km West of Wells-next-the-Sea (52.972107, 0.631014). Brancaster West was selected for restoration in 1996, work began in 2001 and was completed in 2002 (Doody, 2002; Myatt-Bell *et* *al*., 2002; Wolters *et al*., 2005). Like Freiston Shore, this site was 21 years old at the time of sampling. Prior to restoration, the area was used as a freshwater grazing meadow and differs from the other sites included in this study as the tide flows in via piped culverts, as opposed to a simple breach dug in a seawall (Doody, 2002). Development of the plant communities in the five years following the breach has been recorded and showed that *Salicornia europaea* and *S. maritima* were generally most dominant (Mossman *et al*., 2012a). In addition, it was shown that the vegetation still differed significantly from the communities in natural reference marshes five years after restoration was implemented, which was put down to strongly reducing conditions in the Brancaster West sediments (Mossman *et al*., 2012a). Most of our sampling locations (Fig. 1d; Table S1) were overwhelmingly dominated by *S. anglica* and *A. portulacoides* but there were more diverse patches in which *L. vulgare, Salicornia* spp., *A. tripolium, P. maritima* and *Triglochin maritima* were present (Table S2). The tidal range in the area is 6.5 m (Wolters *et al*., 2005).

2.1.4 Fingringhoe Wick

The youngest restored marsh selected in this study sits within the Fingringhoe Wick Nature Discovery Park in Essex and, like Aldboro Point, lies within the Colne estuary (51.842106, 0.974856). The site is an area of former agricultural land to which tidal inundation was reintroduced by the creation of two breaches in the seawall in 2015, making it 8 years old during sampling (Vaughan, 2019). Prior to breach the sea wall, the southern section of the area was used as an arable field while the northern part was a pasture (Cai *et al*., 2022). This marsh largely resembles an unvegetated mudflat, but saltmarsh plant communities have begun to colonise the fringes of the site. Only three plant species were found across all the sampling stations (Fig. 1f; Table S1): *S. anglica*, *Salicornia* spp. and *A. portulacoides* (Table S2). The tidal range is 4.5 m (Wolters *et al*., 2005).

2.2 Natural Marsh Reference Sites

Three natural saltmarshes, investigated in the studies conducted by Smeaton and colleagues (2023; 2024), were selected for their proximity to the studied restored saltmarshes and were used as baselines to compare all tested variables of interest. Aldboro Point and Fingringhoe Wick, two sites almost opposite to one another in the Colne estuary, were compared with Orplands, approximately 16 km away. Freiston Shore was paired with Gedney Marsh, also ~16 km away, and Brancaster West with Stiffkey Marsh, ~20 km away. Detailed information about these natural saltmarshes is available elsewhere (Smeaton *et al*., 2023, 2024; https://catalogue.ceh.ac.uk/documents/5cc9ee72-ed88-4e71-bd8d-e3511bb9ed12). To make comparisons possible, results from cores sampled at a 1 cm resolution were binned into the same depth intervals we sampled (see section 3.1).

3. Methodology

3.1 Sample Collection

Fieldwork was conducted during June and August of 2023. From each of the four restored saltmarshes, eight sediment cores were collected – two wide diameter (5 cm) cores extracted with a Russian peat corer and six narrow diameter (2.7 cm) cores extracted with a gouge auger. The former cores were transported whole to the laboratory while the latter were subsampled in the field at the following resolution: every 2 cm in the top 10 cm and every 5 cm over the remainder of the cores. Hundred cm deep cores were collected from Aldboro Point, and 50 cm deep cores were extracted from the other studied sites, as it was anticipated that sediment

https://doi.org/10.5194/egusphere-2026-1559
Preprint. Discussion started: 2 April 2026

accumulation following the breach in the more recently restored sites (those less than 21 years old) would be less than 50 cm.

Usually, it was not possible to reach the maximum corer depth. Three replicate wide-diameter cores were always taken from within the same ~30 cm^2 area to ensure enough material was available for all required analyses. In all cases, the plant species present within the 1 m^2 areas from which cores were taken were recorded. Cores were stored at -20 °C after being returned to the laboratory and until further analyses were conducted. Hence forth, cores from Aldboro Point are referred to as AP23/1-8, Freiston Shore FS23/1-8, Brancaster West BW23/1-8 and Fingringhoe Wick FW23/1-8.

3.2 Foraminiferal Analysis

Foraminiferal analysis was tested to identify the breach horizons. One replicate core from each of the two wide core sampling location were cut into 1 cm slices in the laboratory. Between 3.0 and 3.5 g of sediment was taken from each slice, and the precise wet mass of each recorded. These samples were then sieved and the fraction between 63 and 500 μm retained for analysis. Samples were refrigerated in 5 % isopropyl before analysis to ensure a high level of preservation. Rose Bengal staining was not used to differentiate between living and dead foraminifera, meaning that all counts represent the total foraminiferal population (Scott and Medioli, 1980). This was deemed acceptable because the main purpose of analysing the foraminifera populations was to determine the breach depth and it has been shown that living foraminifera are overwhelming concentrated at the surface (Saffert and Thomas, 1998; Hayward *et al*., 2014). Samples were split into eight equal aliquots and when foraminifera were present, the aim was to count a minimum of 50 individuals for each sample; where this figure was not obtained in the first aliquot, further splits were processed until this benchmark was reached or the entire sample was counted. If foraminifera were absent from the first aliquot, the entire sample was processed to confirm this was the case throughout. Foraminiferal assemblage composition was only assessed in samples containing over fifty tests following Kemp *et al*. 2020. Murray (1979) was followed for taxonomic assignments to species level. In the cases of *Quinqueloculina* spp., *Elphidium* spp, *Ammonia* spp., *Rosalina* spp. and *Haplophragmoides* spp. only genus level identification was performed. Numbers of tests are given relative to the wet mass of the sample prior to sieving.

3.3 Radiometric Dating

Radiometric dating was only conducted with one wide diameter core from Aldboro Point, the oldest site, because of the need for sediment old enough to achieve interpretable ^{210}Pb and ^{137}Cs profiles. When wet subsampling core AP23/1 in the laboratory, material from each centimetre wide slice from the two replicate cores, AP23/1A and AP23/1B, were combined to provide enough materials for radiometric analysis. Total (i.e., unsupported and supported) ^{210}Pb was determined indirectly via alpha spectrometry (ORTEC Alpha Ensemble) by measuring its decay product, ^{210}Po , and assuming secular equilibrium (Ebaid and Khater, 2006). In more detail, approximately 0.3 g of each 1 cm core slice between 0 and 40 cm was spiked with a known amount of ^{209}Po radiotracer (National Physical Laboratory) and digested using 12 mL of a mixture of HNO_3/HF (3/1) in a microwave assisted digestion system (CEM Mars 6). After neutralising any remaining free fluoride ions, each digest solution was transferred to a glass beaker, evaporated to dryness, and taken up to 25 mL using a 0.5 M HCl solution to allow for the spontaneous deposition of ^{209}Po and ^{210}Po on a silver disc for 5 hours at 90 °C. Each silver disc was finally counted via alpha spectrometry for up to 5 days. Selected adjacent slices (0-3, 8-10, 13-15, 16-18, 19-23, 24-26, 27-29, 30-32 and 38-40 cm) were combined to prepare a suitable geometry for gamma spectrometric analysis (ORTEC HPGe detector) and determine levels of ^{226}Ra (i.e., supported ^{210}Pb via ^{214}Pb) and ^{137}Cs at 352 and 662 keV, respectively (Woodward-Rowe *et al.*, 2025).

The R package *rplum*, version 1.0.0 (Blaauw *et al.,* 2023), was used to develop the age-depth model generated from ^{210}Pb and ^{137}Cs data. This negated the need to select an equilibrium depth and allowed for the easy integration of the ^{137}Cs chronohorizon (Aquino-López *et al.*, 2018). Sediment accretion rates (SAR; mm yr^{-1}) were also obtained from the *rplum* model and these were subsequently used to calculate the accumulation rates of recalcitrant OM in Aldboro Point by multiplying SAR by the recalcitrant OM density (g cm^{-3}).

3.4 Bulk Elemental and Isotopic Analyses

OC and N content, as well as dry bulk density, were measured in every sample from all cores, narrow and wide. Similarly to the replicate core used for the foraminiferal analysis, another replicate was sliced into 1 cm slices in the laboratory before being freeze dried along with the narrow core subsamples that were sliced in the field, as detailed in 3.1. After freeze drying, samples were weighed and this dry mass was divided by the volume of the sample prior to drying to calculate dry bulk density (Dadey *et al.*, 1992). Moisture content was calculated for only the wide cores because it was possible to record the wet mass when subsampling in the laboratory; the dry masses were then subtracted from the wet masses and the products divided by the wet masses.

To approximately 1 g of every dried sample, we added 15 mL – split between three 5 mL aliquots – of 6 % sulphurous acid to remove carbonates by producing calcium sulphate dihydrate, water and carbon dioxide (Gibbs, 1977; Caughey *et al.*, 1995). Acidified samples were allowed to stand for 72 hours before being dried again in the freeze dryer. Finally, samples were homogenised for 30 seconds in a ball mill. From these homogenised samples, 5 mg were weighed into tin capsules and their OC and N content determined by elemental analysis with an CE440 elemental analyser (Exeter Analytical). OC/N ratio was calculated by dividing the OC content of a sample by its N content.

Only the wide diameter cores were subject to bulk sediment δ^{13}C analysis and only every other centimetre was processed, except where the OC content showed high variability, in which case every centimetre was analysed in that section. To ensure that an appropriately sized δ^{13}C signal was obtained from the Finnigan Delta plus XP Isotope Ratio Mass Spectrometer (Thermo Fisher Scientific), the amount of sediment weighed into the tin capsule prior to this analysis was based up on the OC content and varied between 0.03 and 1.60 mg. The analytical precision of δ^{13}C analysis was ± 0.10 ‰. Values are given in δ notation versus VPDB.

3.5 Thermogravimetric Analysis

Approximately 20 mg of the same dried and acidified sediment samples discussed in section 3.4 were used to determine total OM, labile OM and recalcitrant OM content with a TRIOS SDT 650 (TA Instruments). The samples were heated from 40 °C up to 1000 °C under a constant stream of N_2. To classify the OM, the methodology presented in Smeaton and Austin (2022) was followed where labile OM is lost in the temperature range between 200 °C and 400 °C, the recalcitrant fraction between 400 °C and 650 °C, and total OM was represented by the entirety of this range.

3.6 Stock and Accumulation Rate Calculations

To calculate OC and OM stocks we used the narrow cores data and followed Howard and colleagues (2014) by multiplying the carbon density of a segment by its length, summing these values for the entire length of a core and, finally, taking an average of the totals of each core within a site. The sections of the cores taken to represent the post-breach marsh was determined upon a case-by-case basis. Accumulation rates for Aldboro Point were calculated by multiplying the radiometric dating derived sediment accretion rates for each cm of sediment with the OC density of each individual cm.

3.7 Statistical Analysis

To determine whether statistical differences existed between the biogeochemical variables (OC, % N, % OM, C/N ratio, dry bulk density, proportion of recalcitrant OM, proportion of OC in OM) measured in our restored sites and those also recorded in the natural marshes, we used linear mixed effects models with sampling site and core identification being random variables. The lmer function from the *lme4* package in R was used (Bates *et al*., 2015). OC and N data were log transformed prior to analyses to obtain normal distributions of residuals, which was determined by the visual inspection of diagnostic plots. Transformation was not necessary for C/N ratios or proportion of recalcitrant OM. All statistical analyses were performed in R version 4.4.1 (R Core Team, 2024). Given the presence of random effects, denominator degrees of freedom were estimated with the Satterthwaite method (Kuznetsova *et al*., 2017).

4. Results

4.1 Age-depth model

The core AP23/1 from Aldboro Point was the only one subject to ^{210}Pb and ^{137}Cs dating. These data were used to create an age-depth model (Fig. 2a) in *rplum* (Blaauw *et al.,* 2023). The peak in ^{137}Cs levels clearly visible between 21 and 22 cm depth was only attributed to legacy discharge from the Bradwell Power Station, approximately 12 km from the site, which peaked in 1975 (European Commission, 1990; modelled as 1975 ± 5 years). Lead-210 activity concentrations decreased from the surface of the core down to 27-28 cm depth at which point concentrations stabilised (Fig. 2c). The level of supported ^{210}Pb activity concentration (i.e.,^{226}Ra determined via its decay product ^{214}Pb at 352 keV by gamma spectrometry) averaged at 24.5 ± 1.9 Bq kg^{-1}, a value derived from the measurement of seven combined slices evenly spaced along the length of the core (Fig. 2c). The 2 σ uncertainties produced by the model were less than 20 years up to a depth of 25-26 cm, below which point uncertainties increased down the core and reached 70 years in the lowermost sample (Fig. 2a).

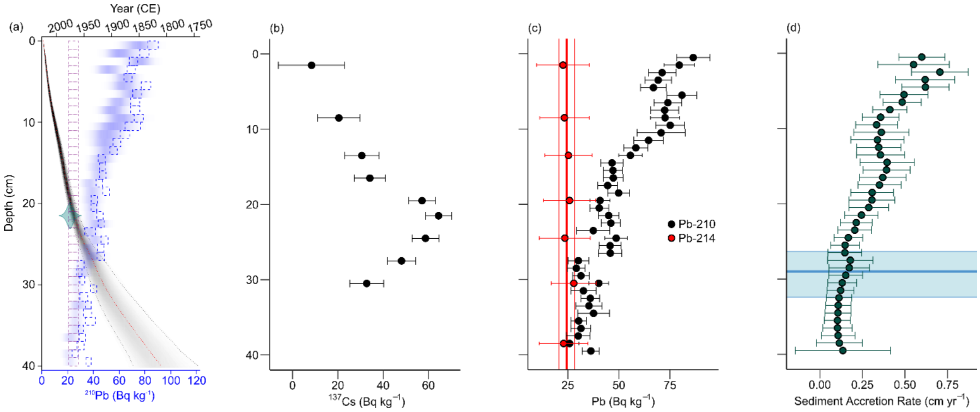

Figure 2. (a) The Bayesian age depth model for AP23/1, with the model represented by the black shaded area and the blue shaded area showing total ^{210}Pb levels. The green shaded area between 21-22 cm depth represents the ^{137}Cs peak at the calendar date 1975 ± 5. The model was developed with the *rplum* package (Blaauw *et al.,* 2023) using (b) ^{137}Cs and (c) ^{210}Pb data. A mean ^{214}Pb value was used for the model, and this is represented by the thick red line with the 2 σ standard deviations represented by the thin lines. (d) Sediment accretion rates for each core slice were obtained from the model. The blue shaded area represents the estimated depth of the breach, based upon the ages assigned by the model to each centimetre of sediment. The thick line shows the depth estimated from the mean ages while the two thin lines show the depths calculated using the minimum and maximum ages (i.e., 2 σ uncertainties). Error bars are ± 2 σ.

Using the estimated ages of deposition assigned to each section of the sediment core, it was possible to determine the depth that represented the year 1921, the year of the breach. From the mean ages, 29.0 cm was assigned as the breach depth, and using the 2 σ uncertainty values, the maximum and minimum breach depths were assigned to 32.4 and 26.4 cm depth, respectively (Fig. 2d). These are the depths represented by the blue lines and shaded areas on all subsequent Aldboro Point figures.

Sediment accretion rate (SAR) values for each centimetre of sediment in the top 40 cm of the core were also derived from the age-depth model developed using *rplum* (Blaauw *et al.,* 2023; Fig. 2d). The average SAR between the bottom slice of the core (39-40 cm) up to 26-27 cm was 0.14 ± 0.28 cm yr^{-1}. Above 26-27 cm, the SAR increased gradually up to the top of the core (Fig. 2d). The beginning of this increase in SAR roughly coincided with the possible breach depth range estimated using the age-depth model. SAR increased gradually from between 27 cm and 14 cm depth before remining relatively constant until 7 cm. Values then continued to increase up to the top of the core, with a maximum SAR of 0.60 ± 0.14 cm yr^{-1} recorded between 2 and 3 cm.

4.2 Foraminifera

4.2.1 Aldboro Point

High numbers (>70, at minimum) of foraminiferal tests were found in almost every centimetre analysed from core AP23/1 (Fig. 3a). Between 31 and 33 cm depth, however, below 40 tests were found, and this immediately proceeded and overlapped the lower limit of the breach depth range. Furthermore, there was a notable decrease in the numbers of tests per gram wet mass between 40-41, 35-36, 33-34 and 30-31 cm. Similar dips in numbers of tests per gram were detected in other parts of the core (i.e., between 90-91, 80-81, 70-71 and 55-56 cm) but these reductions were less sustained (Fig. 3a). The other core taken from Aldboro Point (AP23/2) followed a different pattern: several samples that contained less than sixty tests (Fig. 3b). Additionally, there was generally a lower number of tests per gram in this core, with values no higher than 57 in all samples except for at the surface (398 tests) and between 81 and 75 cm depth (157 - 196). Within the estimated breach depth range for this core, was a sample that contained 42 tests (i.e., 11 tests per gram; from 30-31 cm).

https://doi.org/10.5194/egusphere-2026-1559
Preprint. Discussion started: 2 April 2026

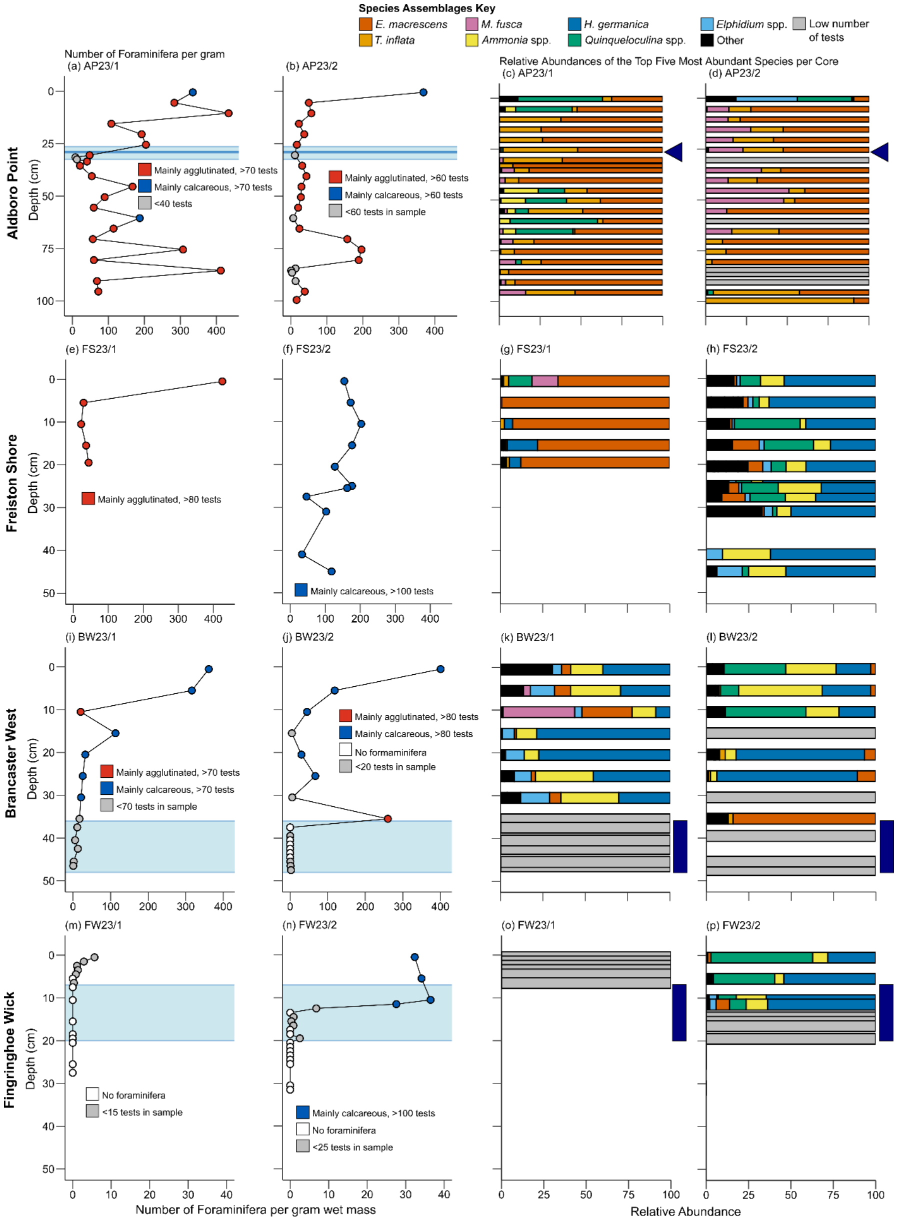

Figure 3. The numbers of foraminifera per gram of sediment and the relative abundances of the seven most abundant species in each centimetre of sediment processed from the two wide-diameter cores taken from (a-d) Aldboro Point, (e-h) Freiston Shore, (i-k) Brancaster West and (m-p) Fingringhoe Wick. All other species were lumped together and classed as "Other". The blue shaded areas and dark blue triangles/boxes represent the depth range in which the breaches could have occurred. This is based upon the age depth model (Fig. 2) for Aldboro Point and upon the foraminifera results for the other sites. Freiston Shore plots do not have these features because established foraminiferal population were found along the entire length of the cores taken.

https://doi.org/10.5194/egusphere-2026-1559
Preprint. Discussion started: 2 April 2026

Almost every sample examined from the Aldboro Point cores had assemblages overwhelmingly dominated by the agglutinated species of foraminifera *Entzia macrescens* (Fig. 3c and 3d). The cores differed, however, in terms of the less abundant species. Between 65 and 66 cm in AP23/1, the calcareous *Quinqueloculina* spp. were one of the most abundant in the sample (0.35), having only been present in another sample prior to this. The *Quinqueloculina* spp. population increased further in the sample from 60-61 cm, meaning that this was one of the few samples in these cores in which calcareous species predominate (Fig. 3a). This species was also present in the next three samples but was slightly less abundant before disappearing entirely until the top two samples (5-6 and 0-1 cm). Between the disappearance and reappearance of *Quinqueloculina* spp. and during the interval in which the breach was thought to have occurred, the agglutinated species *Trochammina inflata* became the second most abundant, having been present in low amounts throughout the lower core depths. The main differences in the assemblages of AP23/2 were the absence of the peak in calcareous species around 60 cm and the relatively high numbers of the agglutinated species *Miliammina fusca* between 66 cm and 5 cm (Fig. 3d). The uppermost sample of this core was predominated by the calcareous *Quinqueloculina* spp. and *Elphidium* spp., which was almost entirely absent from AP23/1. There was no notable change in the assemblages in AP23/2 around the depths at which the breach was thought to occur (Fig. 3d). Based upon the radiometric dating results, the breach depth was identified at 32.4 and 26.4 cm depth, and this was corroborated by the results from the foraminiferal analysis of core AP23/1.

4.2.2 Freiston Shore

The cores from Freiston Shore contained high numbers (>80) of tests in all selected samples, including those from the lowest core depths achieved (19-20 and 45.0 cm in FS23/1 and FS23/2, respectively; Fig. 3e and 3f). The numbers of tests per g was lower in FS23/1, except for in the uppermost sample which contained the highest density value across both cores. The main difference between FS23/1 and FS23/2 was the overwhelming predominance of agglutinated species in the former and calcareous species in the latter. Similarly to Aldboro Point, the agglutinated populations in FS23/1 consisted almost entirely of *E. macrescens* (Fig. 3g). At lower depths (i.e., 19-20, 15-16 and 10-11 cm), the second most abundant species was the calcareous *Haynesina germanica* while at the surface *M. fusca* and *Quinqueloculina* spp. were the second and third most abundant, respectively (Fig. 3g). In the samples from FS23/2, *H. germanica* and *Ammonia* spp. were present in every sample, with the former generally being the most abundant species (Fig. 3h). Between 27-28 cm depth, *Quinqueloculina* spp. reached higher levels of abundance (>10 %) in several samples and between 28and 15 cm higher abundances (6 to 16 %) of *E. macrescens* were seen. It was concluded that the breach depth was not reached and that the cores were representative of saltmarsh sediment only.

4.2.3 Brancaster West

Brancaster West cores contained no or only very low numbers (<60) of foraminiferal tests below 30 cm in BW23/1 and 35 cm in BW23/2 (Figs. 3i and 3j). BW23/1 contained a relatively straightforward assemblage compositions in which the calcareous *H. germanica* and *Ammonia* spp. predominated in all but one of the samples (Fig. 3k). The noteworthy exception was between 10 and 11 cm depth: agglutinated *M. fusca* and *E. macrescens* predominate in this sample and this coincided with a fall in test density to 20 tests per g (Fig. 3k). BW23/2 contained less consistent patterns of test densities and assemblage compositions. The core layer between 37 and 38 cm contained no foraminifera, but the layer above (36-37 cm) had 260 tests per gram (Fig. 3j). Furthermore, this dense population consisted almost entirely of the

https://doi.org/10.5194/egusphere-2026-1559
Preprint. Discussion started: 2 April 2026

agglutinated species *E. macrescens* (relative abundance – 84 %; Fig. 3l). The depth between 30 and 31 cm contained only 19 tests and all remaining samples from higher up the core contained predominantly calcareous species at far lower densities, expect for in the uppermost sample (Fig. 3j). There was, however, clear differences in the species compositions of these calcareous assemblages at higher depths. Populations in samples between 25- 26 cm and 20-21 cm were overwhelmingly predominated by *H. germanica* while further up the core (10-11, 5-6 and 0-1 cm), there were high quantities of *Ammonia* spp., *Quinqueloculina* spp. and *H. germanica* (Fig. 3l).

It was difficult to clearly identify the breach depth range for Brancaster West. The absence of several consecutive samples containing no foraminifera meant that the lowest possible breach depth was assigned to 48 cm, the lowest depth reached by the cores. The highest possible breach depth was at the point directly below the depth at which high numbers (>80) of foraminifera (i.e., an established population) was first detected. Therefore, the breach depth range was between 36 and 48 cm, but this needed to be corroborated with additional information.

4.2.4 Fingringhoe Wick

There was a stark difference between the two cores from the youngest site, Fingringhoe Wick. FW23/1 contained no foraminifera until a depth of 6-7 cm and above this point only very low numbers of tests, 13 at most, were found (0-1 cm; Fig. 3m). On the other hand, FW23/2 contained empty samples until a depth of 19-20 cm, low numbers (1 -21 tests) in most samples between this point and 12-13 cm and then high numbers (>100) in all subsequent samples, bar one (Fig. 3n). Therefore, the possible breach depth for this site was between the first emergences of foraminifera in both cores, meaning a range between 7 and 20 cm.

The assemblages in the samples from FW23/2 that contained a high number of foraminifera are initially predominated by *H. germanica,* but the abundance of this species decreased up the core and was replaced by an increasing quantity of *Quinqueloculina* spp. (Fig. 3p). *Ammonia* spp. remained at a similar level of abundance at all depths.

4.3 Organic Carbon and Nitrogen Content

In the Aldboro Point narrow core segments, OC and N content followed a pattern of gradually increasing values up the cores, similarly to the pattern in the corresponding natural marsh reference site, Orplands (Fig. 4a and 4b). In Aldboro Point values remained relatively stable between the bottom sample to 25-30 cm depth (i.e., near the estimated breach depth of 31 cm). Above this, values increased up the core, reaching an OC peak of 6.55 ± 1.40 % at 4-6 cm and a peak in N of 0.59 ± 0.08 % at 2-4 cm. In the Orplands cores, a similar, steady increase in OC and N content was seen, ending in 5.02 ± 0.71 % OC and 0.48 ± 0.14 % N in the surface sediment.

https://doi.org/10.5194/egusphere-2026-1559
Preprint. Discussion started: 2 April 2026

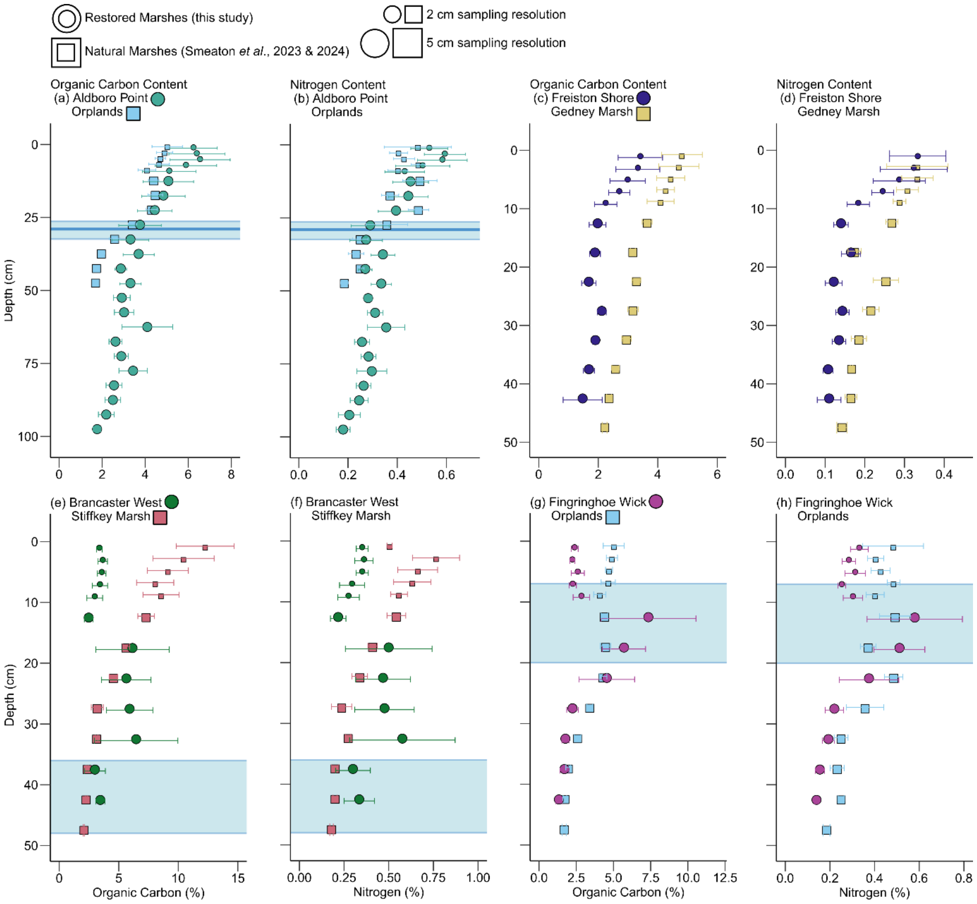

Figure 4. Organic carbon and nitrogen content of the narrow diameter cores and the corresponding natural marsh reference sites: (a-b) Aldboro Point and Orplands, (c-d) Freiston Shore and Gedney Marsh, (e-f) Brancaster West and Stiffkey Marsh and (g-h) Fingringhoe Wick and Orplands. Restored marsh datapoints are always represented by circles while natural marsh data, collected by Smeaton *et al*. (2023; 2024), are represented by squares. Smaller symbols represent 2 cm samples while larger ones represent 5 cm samples. The shaded blue areas denote the depth range at which the breaches may have occurred.

Freiston Shore and Gedney Marsh cores also shared a similar pattern of OC and N content (Fig. 4c and 4d). In the Freiston Shore narrow cores, values remain constant from the bottom to a depth of around 9 cm and from this point an increase was seen, with the uppermost samples containing 3.41 ± 0.75 % OC and 0.33 ± 0.07 % N. The Gedney Marsh values continued to steadily increase up the cores and reached peaks of 4.81 ± 0.69 % OC and 0.33 ± 0.08 % N in the uppermost samples.

In Brancaster West, OC and N content followed a distinctly different pattern (Figs. 4e and 4f). Again, and as was expected, the natural marsh reference site, Stiffkey Marsh, displayed a pattern of steady increase up the core, reaching 8.37 ± 1.3 % OC and 0.51 ± 0.02 % N in the surface samples. The lowermost samples (42-43 cm) from Brancaster contained 3.45 ± 0.38 % OC and 0.34 ± 0.09 % N. Immediately above the breach depth range, however, and in the three successive samples, there was a jump to a maximum of 6.46 ± 3.51 % in OC and 0.58 ± 0.30 in N. From 10-15 cm, values dropped down to 2.48 ± 0.37 % OC and 0.22 ± 0.04 N and small, successive increases were sustained to the surface (Fig. 4e and 4f).

Like Aldboro Point, Fingringhoe Wick was paired with Orplands for a natural reference site and was also compared against the same pattern of steady increase in OC and N up the cores (Fig. 4g and 4h). At the bottom of the Fingringhoe Wick cores (40-45cm), OC content was 1.34 ± 0.21 % and N content 0.14 ± 0.01 %. There followed an increase in both variables, with a clear jump in values from 20-25 cm depth and over the two subsequent depth categories (15-20 and 12-15 cm). Values peak at 7.34 ± 3.20 % for OC and 0.58 ± 0.21 % of N. In this case, the jump in OC and N content began below our estimated breach depth. At 8-10 cm – still within the estimated breach depth – values fell to 2.85 ± 0.56 % OC and 0.30 ± 0.04 % N and similar levels were sustained to the surface (Fig. 4g and 4h).

Overall, there was not a significant difference between the OC and N content of the restored marshes compared with their natural counterparts (Table 1). Marsh type and depth, however, did have a significant interaction, meaning that differences between marsh types were present at certain depth intervals (Table 1). Namely, and as already described, differences in OC and N content between restored and natural marshes were often strongest in the top ~10 cm of the cores.

Table 1. Results of linear mixed effects models that tested the impact of marsh type (i.e., restored or natural), depth and the interaction between the two upon the key variables investigated by this study. All numerator degrees of freedom = 1. Denominator degrees of freedom were estimated using the Satterthwaite method.

Variable	Factor	*p*-value	Denominator *d.f.*	*F*-value
% OC	Marsh Type	0.170	5.900	2.445
	Depth	<0.001	764.170	238.176
	Interaction	<0.001	764.170	30.735
% N	Marsh Type	0.202	6.090	2.041
	Depth	<0.001	587.080	392.538
	Interaction	<0.001	587.080	47.600
Proportion of Recalcitrant OM	Marsh Type	0.047	6.030	6.218
	Depth	<0.001	597.220	168.243
	Interaction	<0.001	597.220	60.463
C/N Ratio	Marsh Type	0.014	6.070	11.733
	Depth	0.409	612.830	0.682
	Interaction	0.003	612.830	8.873

4.4 Recalcitrant Organic Matter

The proportion of OM classed as recalcitrant was high across all sites, restored and natural, with most values being greater than 0.5 (i.e., 50 % recalcitrant; Fig. 5). In Aldboro Point and Freiston Shore, there was a small decrease in recalcitrant OM content up the cores, and, in both cases, this closely matched the patterns of the natural reference sites (Figs. 5a and 5b). Brancaster West samples below 15 cm depth also presented values overlapping with the natural reference site Stiffkey Marsh, but above this point, there was a divergence (Fig. 5c). A share of between 0.64 ± 0.03 and 0.67 ± 0.02 of OM was classed as recalcitrant in Brancaster West from 15 cm to the tops of the cores while in Stiffkey, the share was between 0.45 ± 0.04 and 0.56 ± 0.02. Recalcitrant OM content was higher in the Fingringhoe Wick samples compared with those from Orplands throughout the length of the cores, but this difference became starker from 9 cm and upwards (Fig. 5d). The share of recalcitrant OM in this top section was between 0.67 ± 0.02 and 0.68 ± 0.01 in Fingringhoe Wick and between 0.54 ± 0.02 and 0.55 ± 0.01 in Orplands. In the case of the former, this change roughly coincided with the breach depth, but it happened much later in the Brancaster West cores (Figs. 5c and 5d). Proportion of recalcitrant OM did differ significantly between marsh type but depth and the interaction between depth and marsh type had far stronger effects (Table 1). This lends support to the observation that recalcitrant OM content differed most strongly between samples from the top ~10 cm of the marshes (Fig. 5).

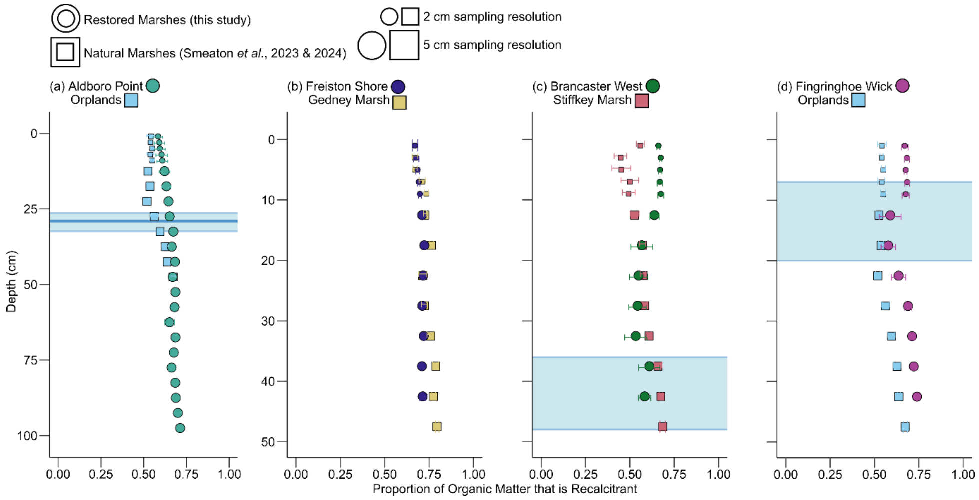

Figure 5. Proportion of total OM classified as recalcitrant by thermogravimetric analysis in the narrow diameter cores and the corresponding natural marsh reference sites: (a) Aldboro Point and Orplands, (b) Freiston Shore and Gedney Marsh, (c) Brancaster West and Stiffkey Marsh and (d) Fingringhoe Wick and Orplands. Restored marsh datapoints are always represented by circles while natural marsh data, collected by Smeaton *et al*. (2023; 2024), are represented by squares. Smaller symbols represent 2 cm samples while larger ones represent 5 cm samples. The shaded blue areas denote the depth range at which the breaches may have occurred.

4.5 Carbon to Nitrogen Ratios and δ^{13}C Content

As shown in Figure 6, the C/N values in restored saltmarshes were consistently lower than those observed in natural saltmarshes. Indeed, across all samples displayed in Figure 6, the mean C/N for the restored samples was 9.95 ± 2.31 (1 σ) and 14.30 ± 1.83 for the natural samples (1 σ).

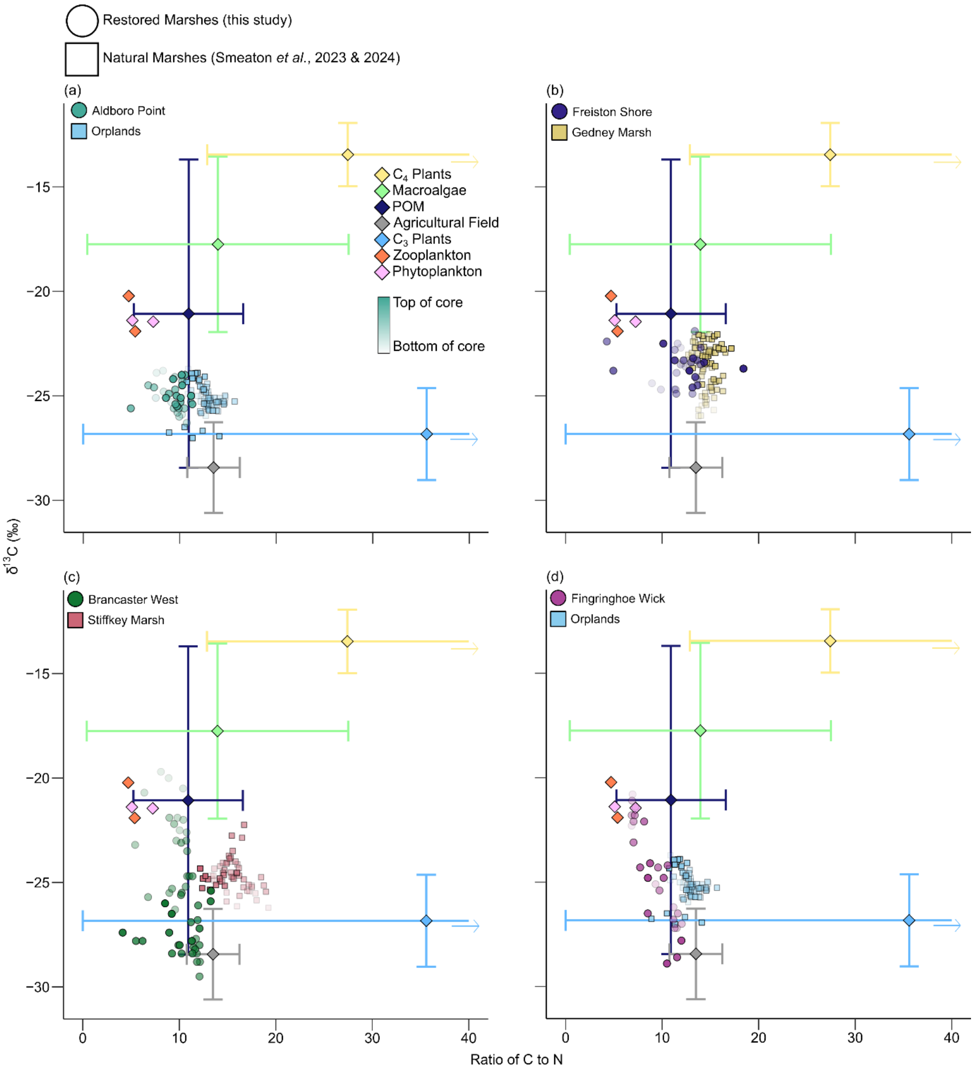

Figure 6. δ^{13}C and C/N values of the post breach sections of the wide diameter cores taken as a part of this study and by Smeaton *et al*. (2024) from (a) Aldboro Point and Orplands, (b) Freiston Shore and Gedney Marsh, (c) Brancaster West and Stiffkey Marsh and (d) Fingringhoe Wick and Orplands. Restored marsh datapoints are always represented by circles while natural marsh data are represented by squares. Values of possible OM sources are adapted from Khan *et al*. (2015) and Smeaton *et al*. (2022).

The C/N ratios in the narrow cores (Fig. 7a-7d) varied with depth. In Aldboro Point, there was a peak of 13.98 ± 1.42 at 25-30 cm that coincided with the breach depth range. Above this, ratios decreased to 11.44 ± 0.87 at the surface (Fig. 7a). Orplands, the corresponding natural marsh reference, also had increasing values up to a depth of 15-20 cm, at which point a peak of 13.60 ± 0.29 was reached before decreasing to 12.95 ± 1.15 at the surface (Fig. 7a). The C/N ratios in Freiston Shore and Gedney Marsh decreased from the lowermost sample to the surface, although variability was greater in the former (Fig. 7b). In Brancaster West, the lowermost sample had a value of 10.70 ± 1.58 and ratios remained around this level, expect for occasional peaks above the breach depth range. The Stiffkey Marsh C/N ratios increased until 25-30 cm depth; above this point, values remained relatively constant to the surface (Fig. 7c). Between 45 and 20 cm in Fingringhoe Wick, values increased from 9.48 ± 0.79 to 11.50 ± 0.45, with the peak occurring below the estimated breach depth range (Fig. 7d). Following this, values fell to 7.44 ± 0.48 at the surface.

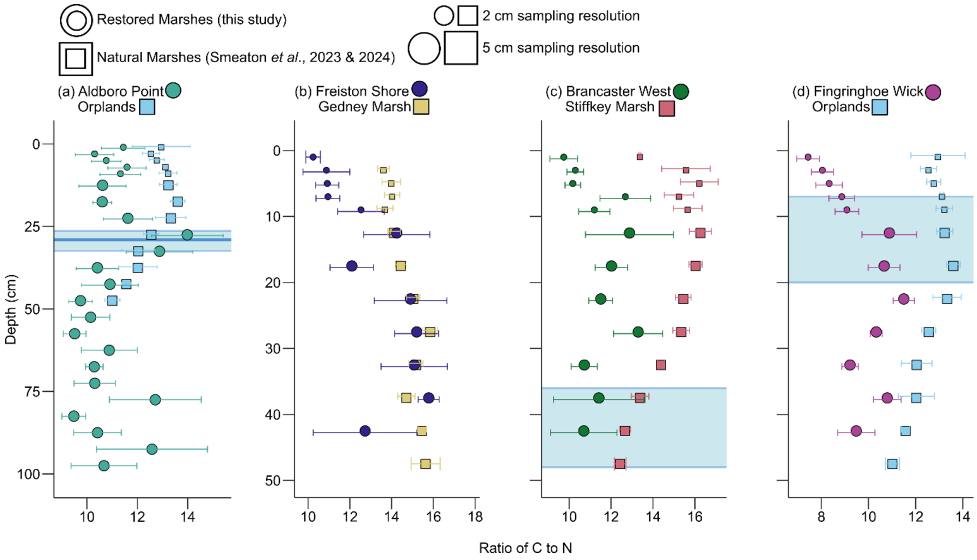

Figure 7. Ratio of C to N in the narrow diameter cores taken as a part of this study and the corresponding natural marsh reference sites: (a) Aldboro Point and Orplands, (b) Freiston Shore and Gedney Marsh, (c) Brancaster West and Stiffkey Marsh and (d) Fingringhoe Wick and Orplands. Restored marsh datapoints are always represented by circles while natural marsh data, collected by Smeaton *et al*. (2023; 2024), are represented by squares. Smaller symbols represent 2 cm samples while larger ones represent 5 cm samples. The shaded blue areas denote the depth range at which the breaches may have occurred.

The $\delta^{13}C$ values in Aldboro Point and Freiston Shore were largely within the same ranges detected in their respective natural marsh reference sites (Fig. 6a & 6b). The former having values within the ranges usually recorded for C_3 plant and/or particulate organic matter (POM) while the latter was more closely clustered within the POM range alone, with some samples crossing over into the macroalgae range. In Brancaster West and Fingringhoe Wick, a wide range of $\delta^{13}C$ values was observed compared with the close clustering of the natural marsh sites (Fig. 6c and 6d). Brancaster West $\delta^{13}C$ values ranged between -29.50 and -19.70 ‰, while at Fingringhoe Wick the range was between -28.90 and -20.80 ‰. To contrast, the range in Stiffkey Marsh was -26.21 to -22.24 ‰ and in Orplands -27.02 to -23.72 ‰. Values in the former two sites, therefore, are closer to the distinctive range of *S. anglica* (-14.97 - -11.94 ‰; Khan *et al*., 2015). Across different depths, Aldboro Point and Freiston Shore $\delta^{13}C$ values were consistent along the entire length of the cores while Brancaster West and Fingringhoe Wick values usually increased from the bottoms of the cores to the tops (Fig. S1).

https://doi.org/10.5194/egusphere-2026-1559
Preprint. Discussion started: 2 April 2026

4.6 Estimate Organic Carbon Stocks and Accumulation Rates

The narrow cores-based OC stock estimates showed an increase from the youngest site, Fingringhoe Wick, through to the two 21-year-old sites, Brancaster West and Freiston Shore (Fig. 8a). In the former, the OC stock estimate was 4.40 ± 1.45 kg m^{-2} while for Brancaster West the estimate was 9.92 ± 2.49 and for Freiston Shore 7.52 ± 1.20 kg m^{-2}. However, it must be emphasised that the foraminifera and OC and N data indicated that the breach depth was not reached by the cores taken from Freiston Shore and therefore, this value was assumed to be an underestimate. In Aldboro Point, the 102-year-old site in which the breach depth was ascertained with ^{210}Pb and^{137}Cs dating, the OC stock was lower than in the 21-year-old sites (5.98 ± 1.28 kg m^{-2}). These restored site stocks were within the range of values reported by Smeaton and colleagues (2023) for Orplands (OR; 8.24 ± 3.75 kg m^{-2}), Stiffkey Marsh (SM; 10.92 ± 3.66 kg m^{-2}) and Gedney Marsh (GM; 5.19 ± 1.32 kg m^{-2}).

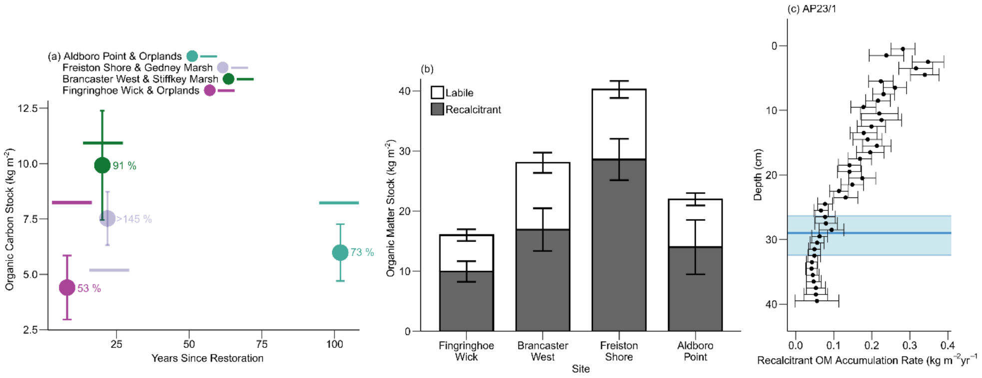

Figure 8. (a) The OC stocks estimated for the four restored marshes included in this study with the horizontal lines representing the stocks calculated by Smeaton *et al*. (2023) for the corresponding natural reference marshes. The Freiston Shore stock estimate is transparent because the breach depth was not reached and so these are necessarily underestimates. With the results of thermogravimetric analysis, it was also possible to calculate (b) organic matter stocks and to differentiate between the proportion of this stock that was recalcitrant versus the labile proportion. Finally, the age-depth model provided SAR rates that were used to calculate (c) recalcitrant OM accumulation rates for core AP23/1. As in previous figures, the blue shaded area represents the estimated breach depth range, also calculated from the age-depth model results.

Across all the restored sites, the recalcitrant OM stock was greater than labile organic matter stock (Fig. 8b). Unlike the OC stock, Brancaster West (16.93 ± 3.55 kg m^{-2}) had a lower recalcitrant OM stock than Freiston Shore (28.58 ± 3.43 kg m^{-2}). Again, the oldest site had a lower stock than both 21-year-old sites (14.02 ± 4.42 kg m^{-2}).

It was possible to calculate recalcitrant OM accumulation rates for the Aldboro Point core (AP23/1; Fig. 8c) using the age-depth model that provided sediment accretion rates for every centimetre of sediment (Fig. 2d). Increased rates of accumulation were apparent above the breach depth range (Fig. 8c). Between 40 and 29 cm depth, recalcitrant OM accumulation rates ranged between 41.82 ± 15.08 and 62.79 ± 20.55 g m^{-2} yr^{-1}. Above 29 cm, the rate increased, and the surface sample presented a value of 281.85 ± 31.85 g m^{-2} yr^{-1}.

https://doi.org/10.5194/egusphere-2026-1559
Preprint. Discussion started: 2 April 2026

5. Discussion

Persistent differences were apparent in the composition of the OM in the sediment of restored marshes compared to natural counterparts, suggesting differences in OM source that continued long after the creation of the breach. Consistency in these differences mean they can be used as indicators in restoration (Table S3). There was often variability between the cores taken from different locations within the same site and this meant that the breach depth ranges for two undated sites, Brancaster West and Fingringhoe Wick, were broad. Furthermore, considering evidence beyond the foraminiferal populations would mean extending the potential breach depth further yet.

5.1 Breach Identification

The expected result from the foraminifera analysis was exemplified in core FW23/2 (Fig. 3n and 3p) in which foraminifera were entirely absent in all samples from the bottom of the core to a depth of 20-21 cm, representing an agricultural field with no tidal inundation. From 20 cm up to 12 cm, samples contained low numbers of tests (<21), and this was taken to represent the period during which the breach was created, and some foraminifera and/or propagules could have washed in by the tides (Weinmann *et al*., 2019). Nevertheless, the conditions were not yet suitable for colonisation to occur, supporting previous evidence of a delay between breach creation and foraminiferal colonisation (Masselink *et al*., 2017; Milker *et al*., 2022). From 12 cm to the surface of the core, more than 100 tests were found in every sample examined, suggesting that foraminifera had successfully colonised this part of the site from this depth upwards. All the other cores analysed were harder to interpret and required greater nuance. Foraminifera were absent from much of FW23/1, the other Fingringhoe Wick core, implying that sediment accretion rates differed substantially between the two locations. In the years immediately following restoration, Pethick (2002) stated that accretion rates could be zero due to a lack of vegetation to hold the sediment in place, especially in the winter months and/or during storms. A possible explanation for the slower rate of accretion in FW23/1, therefore, is the removal of any sediment settling there by the tide due to the closer proximity of this coring location to the breach (Fig. 1d). This is in contradiction to sediment accretion rates recorded in the first 15 months following the breach that were higher in the formerly arable section of the marsh (where FW23/1 was taken) compared with the former pasture (where FW23/2 was taken; Cai *et al*., 2022). Using the bottom of the breach depth range, the sediment accretion rate over the eight years since the breach in FW23/1 was 0.88 cm yr^{-1} while in FW23/2 it was 2.50 cm yr^{-1}. This meant that the breach depth range applied to the Fingringhoe Wick narrow cores was necessarily wide because of a need to reflect the results from both these cores. Variability in moisture content changes in the two cores tends to support this wide range (Fig. S2).

The Brancaster West cores did contain samples entirely absent of foraminifera, but these were interspersed with other samples that contained low numbers of tests, so it was not possible to definitively identify a breach depth, as with core FW23/2. It is possible that the collection of samples from a greater depth would have meant reaching the agricultural layer from which foraminifera are normally absent. However, at the time of the breach, an artificial creek network was dug at this site (Fig. 1c), which could have resulted in a mixing of the agricultural and older saltmarsh layers. The possible breach depth for this site ranged between the lowermost sample analysed and the lowest depth at which foraminifera colonised the marsh (Fig. 3j).

For Freiston Shore, the breach depth was not reached as high numbers of foraminifera were found along the entirety of the cores. Using the mean sediment accretion rate of

1.06 ± 0.20 cm yr^{-1} calculated by Spencer and colleagues (2012) between 2002 and 2006 for the interior of the restoration area suggested that approximately 20 cm of sediment could have accreted in the 21 years since the breach in Freiston Shore. This supported the suggestion that the breach depth in the high marsh core FS23/1 was missed, and the higher rates of accretion recorded closer to the breach (e.g., a rate of 2.54 cm yr^{-1} was recorded near the breach in the four years following restoration) would explain the consistent presence of foraminifera in FS23/2, a 46 cm core. Conversely, the site had not been claimed from the sea for long before the creation of the breach, suggesting the possibility that these assemblages are representative of the pre-1980s saltmarsh. Looking at the foraminiferal species assemblages alone strongly suggests that FS23/1 is representative of a high marsh environment while FS23/2 represents a tidal flat, owing to the dominance of agglutinated tests in the former and calcareous ones in the latter (Fig. 3e & 3f). However, the plant communities in both coring locations were overwhelmingly dominated by *A. portulacoides* (90 % cover; Table S2), meaning that defining the ecosystem type based upon foraminiferal species would have bear no semblance to the reality. It is possible that the lack of topographical variation in restored marshes means the zonation of plants and foraminifera were uncoupled and/or *A. portulacoides*, the plant species often seen to dominate in restored sites, was tolerant of a wider range of inundation frequencies than the foraminiferal community.

At Aldboro Point the many and relatively short-term changes in usage may have meant that there was insufficient time for the area to undergo the processes that lead to agricultural land directly behind seawalls being situated at elevations far below mean high tide (Pethick, 2002). As a result, the site could have quickly recovered after the final breach in 1921, and this explains the lack of empty samples around the time of the breach. In this case, however, it was possible to supplement the foraminifera results with radiometric dating to identify the breach depth. Doing so showed that the fall in test densities in AP23/1 matched the breach depth estimated from the age-depth model, but this was not true of AP23/2. For older sites, this highlights the desirability of employing a myriad of different methods to characterise sediment cores, where possible.

Our results indicated that foraminifera could be a useful tool for finding the breach depth in a restored saltmarsh but the data from Brancaster West and Freiston Shore demonstrated the importance of taking deep cores (>50 cm) - even if a relatively short time (~20 years) has elapsed since the breach - to ensure that the breach depth can be confidently identified. However, the challenge of taking deep cores from some sites, as was the case in the high marsh zone in Freiston Shore, could present a barrier. The results from our youngest site, Fingringhoe Wick highlighted the importance of conducting foraminiferal analysis for multiple cores from different locations within a site due to the possibility of large variations in sedimentation rates.

5.2 Stocks and Accumulation Rates in Restored Saltmarshes

To calculate OC and OM stocks, the lowermost value in the estimated breach depth range was used for Fingringhoe Wick and the uppermost for Brancaster West. This was determined by looking at data from foraminiferal analysis combined with OC, N and C/N data. In the case of Freiston Shore, the entire core lengths were used, with the understanding that this would still lead to an underestimation. The breach depth calculated with the mean ages produced by *rplum* was used for Aldboro Point.

Restored saltmarshes with sufficient elevation relative to the tidal frame achieve OC stocks on a par with natural reference sites around 30 years after breach creation (McMahon *et al.*, 2023) and our younger sites appeared to support this rule (Fig. 8a). After 102 years, however, the OC stock in Aldboro Point was only 73 % that of the natural reference site, Orplands, and so had not reached equivalence as fast as expected (Burden *et al.*, 2019). This suggests either that less OC was entering the sediments in the first place or that the carbon stock is being broken down more quickly in this restored marsh. The high stocks of Brancaster West and Fingringhoe Wick arose from the inclusions of the peaks in OC content – attributed to the pre-restoration agricultural root mat – in both estimates. Another study calculating OC stocks removed the breach depth peaks in OC they identified (Adams *et al.*, 2012). This example was not followed because of the difficulty of identifying the most accurate cut off point, but it is certain that differentiation must be made when quantifying the climate change mitigation benefits of a restoration site as a part of a carbon crediting scheme. Otherwise, the inclusion of these peaks could lead to a significant overestimate of the benefit provided. The lower stock in Aldboro Point was perhaps indicative of the fact that the root mat OM was gradually decomposed and lost in the years after restoration. The higher C/N ratios of restored marshes could support this process by providing more nitrate to drive decomposition (Bulseco *et al.*, 2019).

The OC stock of Freiston Shore was less than that of Brancaster West while the opposite was true of the OM stock (Fig. 8a and 8b). Given that the breach depth was not reached in Freiston Shore, the cores taken from this site represented more recently deposited saltmarsh sediment, despite the site being the same age as Brancaster West. This could indicate that less OC rich OM is accumulated after a certain stage of saltmarsh development. This was supported by the profiles of OC to OM ratios along the cores (Fig. S3), which showed high values around the breach depth ranges of Brancaster West and Fingringhoe Wick but consistently low values, compared with the natural reference site, in Freiston Shore.

Recalcitrant OM accumulation rates often remain largely uniform along the entire length of sampled cores because it represents the OM fraction that remains stable through time (Unger *et al.*, 2016; Gore *et al.*, 2024). However, this was not at all the case in Aldboro Point, in which recalcitrant OM accumulation rates steadily increased up the core (Fig.8c). Higher sedimentation rates have been linked to greater labile OC density (Unger *et al.*, 2016) but in Aldboro Point increasingly recalcitrant OM was delivered to the sediments as the site developed.

5.3 Indications of allochthonous versus autochthonous sources

5.3.1 Organic Carbon and Nitrogen Signals

The clearest and most consistent difference was the lower C/N ratio values in restored sites compared to natural references (Table 1; Fig. 6), suggesting a difference in the sources of OM. Other studies have also found that restored saltmarshes had lower C/N ratios than their natural counterparts and the differences were put down to a predominately allochthonous/marine input to the restored sites (Drexler *et al.*, 2020; Mueller *et al.*, 2020). Values below 10 are indicative of recently deposited marine OM (Hulisz *et al.*, 2013).

OC, N and C/N values were more clearly different in the top ~10 cm of the investigated cores, which was opposite to previously reported results (Havens *et al.*, 1995) and counterintuitive as newly accreting layers of marsh sediment often show signs of becoming increasingly "natural" because of vegetation modifying the surface sediment (Craft *et al.*, 1988; Craft *et al.*, 2002; Hulisz *et al.*, 2013). Lower C/N ratios in saltmarshes can be put down to additional N being delivered by the tide and produced *in-situ* and/or a higher amount of algal biomass being incorporated into the OM (Adams *et al*., 2012). Given that N content did not differ between marsh types, however, the former factor is not likely to be the driver of the lower C/N ratios observed. However, the latter possibility calls for greater consideration. Except for Gedney Marsh, all natural references marshes had C/N ratios that decrease with depth (Fig. 6), indicating OM decomposition (Gebrehiwet *et al*., 2008) and/or the incorporation of N from fungi and bacteria into the OM (Goñi and Thomas, 2000). The lower values of the surface sediments in the restored saltmarshes suggests that part of this signal was produced by OM entering the marshes that had already undergone this process of combination with fungal, bacterial or algal biomass.

There are exceptions to the rule of lower C/N ratios in restored sites. Clear peaks in OC, N and OC/N ratios were seen above and during the breach depth ranges of two younger sites: Brancaster West and Fingringhoe Wick. A peak in OC content at breach depths were also identified in the sites investigated by Adams and colleagues (2012) and was assumed to represent the root mat of the agricultural fields present prior to restoration. The presence of these peaks in the selected sites confirmed the appropriateness of foraminiferal analysis as a forensic tool to help identify breach depth ranges. However, this was weakened by the fact that the peak occurred above the breach depth range in Brancaster West and during the breach depth range in Fingringhoe Wick. If indicative of the pre-restoration field, the OM peaks were expected to appear below the estimated depth of breach. An alternative explanation for the peaks could be the large influx of OC shown to enter a marsh in the first ~five years following the creation of a breach, which necessarily comes from allochthonous sources because of the lack of saltmarsh plants in this early phase (Wollenburg *et al*., 2018; Poppe and Rybczyk, 2021; Mossman *et al*., 2022). However, the higher C/N ratios of the peaks compared with the samples in the top 10 cm of the selected cores suggested this early OM was of a less strongly marine sources than the material entering the marsh later. In Aldboro Point, there were no peaks in OC and N during or near the breach but there was a clear peak in C/N ratio (Fig. 7a), suggesting that the OM had decomposed in the long time since the breach but that its origin was similar.

5.3.2 δ^{13}C values

In Fingringhoe Wick, Brancaster West and, to a lesser degree, Freiston Shore, δ^{13}C values indicated the presence of allochthonous OM, while the signals from Aldboro Point were more difficult to interpret. *Spartina anglica* was the only plant species (70-90 % cover) present in the wide core sampling locations of the former two sites (Table S2) and the δ^{13}C enriched signal of the C_4 plant explains the steadily increasing values towards the surface of the cores. Despite this, the δ^{13}C values were lower than the signal expected from this plant alone, which suggests a contribution from allochthonous material combined with the autochthonous *S. anglica* to produce a mixed signal (Middelburg *et al*., 1997). Terrestrial plants which may have been present before the breach have relatively low δ^{13}C values that are reported to range between -25.0 and -27.0 ‰ (Goni and Hedges, 1995) and sediment from agricultural fields has a mean value of -28.4 ± -2.17 ‰ (Smeaton *et al*., 2022). A similar range was observed within and above the breach depths of Brancaster West (-29.5 – -24.7 ‰ between the bottom and 20 cm depth; Fig. S1) and Fingringhoe Wick (–28.9 to -27.2 ‰ between 33 cm and 18 cm in FW23/2; Fig. S1). This, in combination with the higher C/N values, corroborates that the peaks represented the root mat of the agricultural fields and not OM entering the sites after restoration. The disparity with the locations of the foraminifera identified breach depth ranges points to variability in sediment accretion rates and marsh developments within sites.

https://doi.org/10.5194/egusphere-2026-1559
Preprint. Discussion started: 2 April 2026

Enrichment in δ^{13}C with increasing depth was not seen in Aldboro Point and Freiston Shore (Fig. S1). Values from the bottom of the breach depth ranges to the tops of the cores were relatively stable, ranging between -26.3 and -23.8 ‰ in the former and -24.9 and -21.9 ‰ in the latter (Fig. S1). This implied that the expected changes to the OM components over time due to decomposition (Ehleringer *et al*., 2000) had not occurred. This could mean the OM was relatively stable and could not be broken down. Alternatively, microbial processes could be occurring at certain depths that produce OC with a δ^{13}C signal that allows the maintenance of the overall sediment δ^{13}C despite changes to the OM from the surface (Goñi and Thomas, 2000). Most of the samples from Aldboro Point fall within the overlapping range of values produced by POM and C_3 plants, making it impossible to definitively identify the most likely source. Several Freiston Shore values, on the other hand, are more enriched and thus fall either within the range of POM alone or within the overlap between POM and macroalgae. Therefore, Freiston Shore must also receive an input of allochthonous OM. The coring locations of the natural marshes were dominated by *A. portulacoides* and *P. maritima*, with *S. anglica* representing a minor contribution (<15 % cover) to the populations of one coring location within Gedney Marsh and another within Stiffkey Marsh (Ladd *et al*., 2022). The presence of these C_3 plants explained the close similarity between the δ^{13}C profiles of the reference sites and those of Aldboro Point and Freiston Shore (Fig. 6a and 6b).

5.3.3 Recalcitrant and Labile Organic Matter

The proportion of OM classed as recalcitrant was significantly higher in the restored sites compared with the natural (Table 1; Fig. 5). Differences were starkest in the top 10 cm of Brancaster West and Fingringhoe Wick cores (Fig 5c and 5d). Often, labile OM content is higher in the surface sediments but decreases at lower depths due to the preferential decomposition of this material by microorganisms (Gore *et al*., 2024). Taken together, these factors lend support to the idea that the OM present in these surface sediments had already aged and stabilised. Minerogenic marshes often contain OM of a largely marine allochthonous origin (Mueller *et al*., 2019a; Amann *et al*., 2023). The relative stability of the recalcitrant fraction of OM along the entire length of the cores, restored and natural, supports the hypothesis that OM stabilisation in minerogenic saltmarshes occurs similarly to upland soils whilst organogenic marshes differ strongly from both. The former two are mineral dominated, and the OM present is in the form of microaggregates, meaning it is physically protected from decomposition (Lehmann *et al*., 2007) while OM in organogenic marshes is more vulnerable but protected by the environmental conditions (e.g., Craft *et al*., 2003; Wang *et al*., 2019). When undergoing thermogravimetric analysis, the mineral associated OM is burnt off at higher temperatures and thus classed as recalcitrant while organic macroaggregates are unprotected when removed from their anoxic environment and burn in the labile temperature range (Pallasser *et al*., 2013; Li *et al*., 2023; Ortner *et al*., 2025). The higher recalcitrant OM content of the surface sediments was an additional piece of evidence that suggested a higher allochthonous contribution to the restored marshes, given that the labile component of this material was likely lost during decomposition in an external location before being washed into the restored sites.

Conclusions

A total of thirty-two cores were taken from four restored saltmarshes in the east of England, ranging in age from eight to 102 years. Comparisons were made with the extensive natural marsh dataset developed by Smeaton and colleagues (2023; 2024). The variables that consistently differed across all our sites when compared with their natural counterparts were

OC/N ratios and proportions of recalcitrant OM (Tables 1 & S3). These differences were interpreted as representing a greater input of mineral associated, marine, allochthonous and partially decomposed OM to restored saltmarshes. Up to one hundred years after restoration, C/N values had not recovered, which implied that the origin of OM remained different. Accordingly, determination of whether restored saltmarsh carbon stocks represent CO_2 removal additional to that produced without the restoration should be treated as a crucial step in the carbon crediting process.

The root mat indicative of the agricultural field present prior to restoration was identified in two of our four sites from the C/N ratio but also the $\delta^{13}C$ signal. Based upon the breach depth determined with the foraminifera method, the OC peaks produced by the root mats were included in the stock calculations and thus led to overestimates. Caution should be taken to identify and discount this OC when stocks are being calculated for carbon crediting schemes. Furthermore, the oldest site included in this study had one of the lowest organic carbon stocks – a potential reason being the loss of the root mat layer over time via decomposition.

If the foraminifera method tested in this study is to be applied elsewhere it is vital to consider and account for within site variability by analysing several cores taken from locations widely dispersed across the site. Furthermore, deep cores are often necessary to reach the breach horizon because, despite a time lag between breach and colonisation, small numbers of tests can be washed onto a marsh immediately after tidal reintroduction. Additionally, defining zones, such as high marsh, low marsh and tidal flat, based upon foraminiferal assemblage data from restored marshes is not necessarily an accurate means of tracking the development of a site over time because difference may not extend beyond the highly sensitive foraminifera.

Author Contributions

CG: Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Conceptualization. PM: Writing – review & editing, Supervision, Conceptualization. MC: Writing – review & editing, Methodology, Conceptualization, Supervision. FDM: Writing – review & editing, Supervision, Formal analysis. GS: Writing – review & editing. YM: Writing – review & editing. LB: Writing – review & editing, Supervision. SN: Writing – review & editing, Supervision, Methodology, Conceptualization, Funding acquisition.

Competing Interests

The authors declare that they have no conflict of interest.

Data Availability

The datasets used in this study are publicly available in the online repository Figshare: https://doi.org/10.6084/m9.figshare.31812925

Acknowledgments

We thank Simon Ellis from the University of East Anglia for assistance with fieldwork and Tony Hinchliffe from the University of East Anglia for guidance with the laboratory analyses. Thanks also to the Essex Wildlife Trust, the Environment Agency, the RSPB and Lesley Gunfield for permissions to access the sites. Finally, thank you to the Cefas radioanalytical team for generating the radiometric dating data. This work was supported by the Natural Environment Research Council and the ARIES Doctoral Training Partnership [grant number NE/S007334/1].

References

Adams, C.A., Andrews, J.E. and Jickells, T. (2012) 'Nitrous oxide and methane fluxes vs. carbon, nitrogen and phosphorous burial in new intertidal and saltmarsh sediments', *Science of The Total Environment*, 434, pp. 240–251. Available at: https://doi.org/10.1016/j.scitotenv.2011.11.058.

Amann, B. *et al.* (2023) 'Multi-annual and multi-decadal evolution of sediment accretion in a saltmarsh of the French Atlantic coast: Implications for carbon sequestration', *Estuarine, Coastal and Shelf Science*, 293, p. 108467. Available at: https://doi.org/10.1016/j.ecss.2023.108467.

Andrews, J.E., Samways, G. and Shimmield, G.B. (2008) 'Historical storage budgets of organic carbon, nutrient and contaminant elements in saltmarsh sediments: Biogeochemical context for managed realignment, Humber Estuary, UK', *Science of The Total Environment*, 405(1), pp. 1–13. Available at: https://doi.org/10.1016/j.scitotenv.2008.07.044.

Aquino-López, M.A. *et al.* (2018) 'Bayesian Analysis of 210-Pb Dating', *Journal of Agricultural, Biological and Environmental Statistics*, 23(3), pp. 317–333. Available at: https://doi.org/10.1007/s13253-018-0328-7.

Arias-Ortiz, A. *et al.* (2021) 'Tidal and Nontidal Marsh Restoration: A Trade-Off Between Carbon Sequestration, Methane Emissions, and Soil Accretion', *Journal of Geophysical Research: Biogeosciences*, 126(12), p. e2021JG006573. Available at: https://doi.org/10.1029/2021JG006573.

Atkinson, P. *et al.* 'The success of creation and restoration schemes in producing intertidal habitat suitable for waterbirds', Report to English Nature. ENRR425.

Bartholdy, J. *et al.* (2014) 'On autochthonous organic production and its implication for the consolidation of temperate salt marshes', *Marine Geology*, 351, pp. 53–57. Available at: https://doi.org/10.1016/j.margeo.2014.03.015.

Bates, D., Maechler, M., Bolker, B. and Walker, S. (2015). Fitting Linear Mixed-Effects Models Using lme4. Journal of Statistical Software, 67(1), 1-48. doi:10.18637/jss.v067.i01.

Benjamin, A. *et al.* (2024) 'Understanding sediment and carbon accumulation in macrotidal minerogenic saltmarshes for climate resilience', *Geomorphology*, 467, p. 109465. Available at: https://doi.org/10.1016/j.geomorph.2024.109465.

Blaauw, M.J., Christen, A. and Aquino-Lopez, M.A. (2023). *rplum*: Bayesian Age-Depth Modelling of Cores Dated by Pb-210. R package version 0.3.0. https://CRAN.R-project.org/package=rplum

Boorman, L.A., (2003). Saltmarsh Review. An overview of coastal saltmarshes, their dynamic and sensitivity characteristics for conservation and management. JNCC Report, No. 334.

Brooks, K.L. *et al.* (2015) 'Limited Vegetation Development on a Created Salt Marsh Associated with Over-Consolidated Sediments and Lack of Topographic Heterogeneity', *Estuaries and Coasts*, 38(1), pp. 325–336. Available at: https://doi.org/10.1007/s12237-014-9824-3.

Boutton, T.W. *et al.* (1998) 'δ13C values of soil organic carbon and their use in documenting vegetation change in a subtropical savanna ecosystem', *Geoderma*, 82(1), pp. 5–41. Available at: https://doi.org/10.1016/S0016-7061(97)00095-5.

Bulseco, A.N. *et al.* (2019) 'Nitrate addition stimulates microbial decomposition of organic matter in salt marsh sediments', *Global Change Biology*, 25(10), pp. 3224–3241. Available at: https://doi.org/10.1111/gcb.14726.

Burd, F. (1992) 'Historical study of sites of natural sea wall failures in Essex. Phase I study', Report to English nature. ENRR015. Institute of Estuarine and Coastal Studies, Hull.

Burd, F., Clifton, J., Murphy, B. (1994) 'Sites of historical sea defense failure. Phase II study', Report to English nature. ZO38-94-F. Institute of Estuarine and Coastal Studies, Hull.

Burden, A., *et al*. (2023) 'Recommendations for development of a UK domestic Saltmarsh Code', Report to the Natural Environment Investment Readiness Fund (NEIRF). UK Centre for Ecology & Hydrology, Bangor. 59pp

Burden, A., Garbutt, A. and Evans, C.D. (2019) 'Effect of restoration on saltmarsh carbon accumulation in Eastern England', *Biology Letters*, 15(1), p. 20180773. Available at: https://doi.org/10.1098/rsbl.2018.0773.

Cai, L.L., Helgason, T. and Redeker, K.R. (2022) 'Changes in sediment characteristics in the first year of a realigned saltmarsh', *Estuarine, Coastal and Shelf Science*, 264, p. 107684. Available at: https://doi.org/10.1016/j.ecss.2021.107684.

Calvo-Cubero, J. *et al.* (2014) 'Changes in nutrient concentration and carbon accumulation in a mediterranean restored marsh (Ebro Delta, Spain)', *Ecological Engineering*, 71, pp. 278–289. Available at: https://doi.org/10.1016/j.ecoleng.2014.07.023.

Caughey, M.E. *et al.* (1995) 'Interlaboratory study of a method for determining nonvolatile organic carbon in aquifer materials', *Environmental Geology*, 26(4), pp. 211–219. Available at: https://doi.org/10.1007/BF00770471.

Connor, R.F., Chmura, G.L. and Beecher, C.B. (2001) 'Carbon accumulation in bay of fundy salt marshes: Implications for restoration of reclaimed marshes', *Global Biogeochemical Cycles*, 15(4), pp. 943–954. Available at: https://doi.org/10.1029/2000GB001346.

Craft, C. *et al.* (2003) 'The Pace of Ecosystem Development of Constructed Spartina Alterniflora Marshes', *Ecological Applications*, 13(5), pp. 1417–1432. Available at: https://doi.org/10.1890/02-5086.

Craft, C., Broome, S. and Campbell, C. (2002) 'Fifteen Years of Vegetation and Soil Development after Brackish-Water Marsh Creation', *Restoration Ecology*, 10(2), pp. 248–258. Available at: https://doi.org/10.1046/j.1526-100X.2002.01020.x.

Craft, C.B., Broome, S.W. and Seneca, E.D. (1988) 'Nitrogen, Phosphorus and Organic Carbon Pools in Natural and Transplanted Marsh Soils', *Estuaries*, 11(4), pp. 272–280. Available at: https://doi.org/10.2307/1352014.

Curado, G. *et al.* (2013) 'Native plant restoration combats environmental change: development of carbon and nitrogen sequestration capacity using small cordgrass in European salt marshes', *Environmental Monitoring and Assessment*, 185(10), pp. 8439–8449. Available at: https://doi.org/10.1007/s10661-013-3185-4.

Dadey, K.A., Janecek, T., Klaus, A., 1992. Dry-bulk density: its use and determination. Proceedings of the ocean drilling program. Sci. Res. 126.

de los Santos, C.B. *et al.* (2023) 'Sedimentary Organic Carbon and Nitrogen Sequestration Across a Vertical Gradient on a Temperate Wetland Seascape Including Salt Marshes, Seagrass Meadows and Rhizophytic Macroalgae Beds', *Ecosystems*, 26(4), pp. 826–842. Available at: https://doi.org/10.1007/s10021-022-00801-5.

Doody, P. (2002) 'Coastal Habitat Restoration, towards good practice', Part of the LIFE funded 'Living with the Sea Project'. FST20-48-006 01/0384. Available at: https://webarchive.nationalarchives.gov.uk/ukgwa/20090514092412/http://www.eclife.naturalengland.org.uk/project_details/good_practice_guide/HabitatCRR/ENRestore/home.htm

Drexler, J.Z. *et al.* (2020) 'Carbon Sources in the Sediments of a Restoring vs. Historically Unaltered Salt Marsh', *Estuaries and Coasts*, 43(6), pp. 1345–1360. Available at: https://doi.org/10.1007/s12237-020-00748-7.

Ebaid, Y.Y. and Khater, A.E.M. (2006) 'Determination of 210Pb in environmental samples', *Journal of Radioanalytical and Nuclear Chemistry*, 270(3), pp. 609–619. Available at: https://doi.org/10.1007/s10967-006-0470-5.

Ehleringer, J.R., Buchmann, N. and Flanagan, L.B. (2000) 'Carbon Isotope Ratios in Belowground Carbon Cycle Processes', *Ecological Applications*, 10(2), pp. 412–422. Available at: https://doi.org/10.1890/1051-0761(2000)010%255B0412:CIRIBC%255D2.0.CO;2.

European Commission, Directorate-General for the Information Society and Media, Directorate-General for Environment, (1990) *The radiological exposure of the population of the European Community radioactivity in North European marine waters : Project "Marina"*. Publications Office of the European Union.

French, P.W. (2006) 'Managed realignment – The developing story of a comparatively new approach to soft engineering', *Estuarine, Coastal and Shelf Science*, 67(3), pp. 409–423. Available at: https://doi.org/10.1016/j.ecss.2005.11.035.

Friess, D.A. *et al.* (2012) 'Remote sensing of geomorphological and ecological change in response to saltmarsh managed realignment, The Wash, UK', *International Journal of Applied Earth Observation and Geoinformation*, 18, pp. 57–68. Available at: https://doi.org/10.1016/j.jag.2012.01.016.

Friess, D.A. *et al.* (2014) 'Coastal saltmarsh managed realignment drives rapid breach inlet and external creek evolution, Freiston Shore (UK)', *Geomorphology*, 208, pp. 22–33. Available at: https://doi.org/10.1016/j.geomorph.2013.11.010.

Fu, C. *et al.* (2024) 'Climate and mineral accretion as drivers of mineral-associated and particulate organic matter accumulation in tidal wetland soils', *Global Change Biology*, 30(1), p. e17070. Available at: https://doi.org/10.1111/gcb.17070.

Fu, C. *et al.* (2024) 'Climate and mineral accretion as drivers of mineral-associated and particulate organic matter accumulation in tidal wetland soils', *Global Change Biology*, 30(1), p. e17070. Available at: https://doi.org/10.1111/gcb.17070.

Garbutt, A. and Wolters, M. (2008) 'The natural regeneration of salt marsh on formerly reclaimed land', *Applied Vegetation Science*, 11(3), pp. 335–344. Available at: https://doi.org/10.3170/2008-7-18451.

Gebrehiwet, T., Koretsky, C.M. and Krishnamurthy, R.V. (2008) ‘Influence of *Spartina* and *Juncus* on saltmarsh sediments. III. Organic geochemistry’, *Chemical Geology*, 255(1), pp. 114–119. Available at: https://doi.org/10.1016/j.chemgeo.2008.06.015.

Gibbs, R.J. (1977) ‘Effect of combustion temperature and time, and of the oxidation agent used in organic carbon and nitrogen analyses of sediments and dissolved organic material’, *Journal of Sedimentary Research*, 47(2), pp. 547–550. Available at: https://doi.org/10.1306/212F71D5-2B24-11D7-8648000102C1865D.

Goñi, M.A. and Hedges, J.I. (1995) ‘Sources and reactivities of marine-derived organic matter in coastal sediments as determined by alkaline CuO oxidation’, *Geochimica et Cosmochimica Acta*, 59(14), pp. 2965–2981. Available at: https://doi.org/10.1016/0016-7037(95)00188-3.

Goñi, M.A. and Thomas, K.A. (2000) ‘Sources and transformations of organic matter in surface soils and sediments from a tidal estuary (North Inlet, South Carolina, USA)’, *Estuaries*, 23(4), pp. 548–564. Available at: https://doi.org/10.2307/1353145.

Gore, C. *et al.* (2024) ‘Saltmarsh blue carbon accumulation rates and their relationship with sea-level rise on a multi-decadal timescale in northern England’, *Estuarine, Coastal and Shelf Science*, 299, p. 108665. Available at: https://doi.org/10.1016/j.ecss.2024.108665.

Gulliver, A. *et al.* (2020) ‘Estimating the Potential Blue Carbon Gains From Tidal Marsh Rehabilitation: A Case Study From South Eastern Australia’, *Frontiers in Marine Science*, 7. Available at: https://doi.org/10.3389/fmars.2020.00403.

Havens, K.J., Varnell, L.M. and Bradshaw, J.G. (1995) ‘An assessment of ecological conditions in a constructed tidal marsh and two natural reference tidal marshes in coastal Virginia’, *Ecological Engineering*, 4(2), pp. 117–141. Available at: https://doi.org/10.1016/0925-8574(94)00051-6.

Hayward, B.W. *et al.* (2014) ‘Multi-year life spans of high salt marsh agglutinated foraminifera from New Zealand’, *Marine Micropaleontology*, 109, pp. 54–65. Available at: https://doi.org/10.1016/j.marmicro.2014.03.002.

Houston, A., Kennedy, H. and Austin, W.E.N. (2024) ‘Additionality in Blue Carbon Ecosystems: Recommendations for a Universally Applicable Accounting Methodology’, *Global Change Biology*, 30(11), p. e17559. Available at: https://doi.org/10.1111/gcb.17559.

Howard, J. *et al*. (2014). ‘Coastal Blue Carbon: Methods for assessing carbon stocks and emissions factors in mangroves, tidal salt marshes, and seagrass meadows’, Conservation International, Intergovernmental Oceanographic Commission of UNESCO, International Union for Conservation of Nature. Arlington, Virginia, USA.

Hulisz, P. *et al.* (2013) ‘Chronosequential alterations in soil organic matter during initial development of coastal salt marsh soils at the southern North Sea’, *Zeitschrift für Geomorphologie*, pp. 515–529. Available at: https://doi.org/10.1127/0372-8854/2013/0112.

Jiménez-Arias, J.L. *et al.* (2020) ‘Tidal elevation is the key factor modulating burial rates and composition of organic matter in a coastal wetland with multiple habitats’, *Science of The Total Environment*, 724, p. 138205. Available at: https://doi.org/10.1016/j.scitotenv.2020.138205.

Jones, S.F. *et al.* (2024) 'When and where can coastal wetland restoration increase carbon sequestration as a natural climate solution?', *Cambridge Prisms: Coastal Futures*, 2, p. e13. Available at: https://doi.org/10.1017/cft.2024.14.

Kemp, A.C., Wright, A.J. and Cahill, N. (2020) 'Enough is Enough, or More is More? Testing the Influence of Foraminiferal Count Size on Reconstructions of Paleo-Marsh Elevation', *Journal of Foraminiferal Research*, 50(3), pp. 266–278. Available at: https://doi.org/10.2113/gsjfr.50.3.266.

Khan, N.S. *et al.* (2015) 'The application of δ13C, TOC and C/N geochemistry to reconstruct Holocene relative sea levels and paleoenvironments in the Thames Estuary, UK', *Journal of Quaternary Science*, 30(5), pp. 417–433. Available at: https://doi.org/10.1002/jqs.2784.

Kiesel, J. *et al.* (2019) 'Attenuation of high water levels over restored saltmarshes can be limited. Insights from Freiston Shore, Lincolnshire, UK', *Ecological Engineering*, 136, pp. 89–100. Available at: https://doi.org/10.1016/j.ecoleng.2019.06.009.

Komada, T. *et al.* (2022) '"Slow" and "fast" in blue carbon: Differential turnover of allochthonous and autochthonous organic matter in minerogenic salt marsh sediments', *Limnology and Oceanography*, 67(S2), pp. S133–S147. Available at: https://doi.org/10.1002/lno.12090.

Krause, J.R. *et al.* (2022) 'Beyond habitat boundaries: Organic matter cycling requires a system-wide approach for accurate blue carbon accounting', *Limnology and Oceanography*, 67(S2), pp. S6–S18. Available at: https://doi.org/10.1002/lno.12071.

Kuznetsova, A., Brockhoff, P.B. and Christensen, R.H.B. (2017) 'lmerTest Package: Tests in Linear Mixed Effects Models', *Journal of Statistical Software*, 82, pp. 1–26. Available at: https://doi.org/10.18637/jss.v082.i13.

Ladd, C.J.T.; Miller, L.C.; McMahon, L.; Havelock, G.M.; Smeaton, C.; Garbutt, A.; Skov, M.W.; Austin, W.E.N. (2022). 'Vegetation composition data from UK saltmarshes, 2018 to 2021', NERC EDS Environmental Information Data Centre. https://doi.org/10.5285/7815291a-ee3e-49a7-8ef1-e8baa81f4964

Lawrence, P.J. *et al.* (2018) 'Restored saltmarshes lack the topographic diversity found in natural habitat', *Ecological Engineering*, 115, pp. 58–66. Available at: https://doi.org/10.1016/j.ecoleng.2018.02.007.

Lawrence, P.J., Sullivan, M.J.P. and Mossman, H.L. (2022) 'Restored saltmarshes have low beta diversity due to limited topographic variation, but this can be countered by management', *Journal of Applied Ecology*, 59(7), pp. 1709–1720. Available at: https://doi.org/10.1111/1365-2664.14179.

Li, Y. *et al.* (2023) 'Soil Carbon, Nitrogen, and Phosphorus Stoichiometry and Fractions in Blue Carbon Ecosystems: Implications for Carbon Accumulation in Allochthonous-Dominated Habitats', *Environmental Science & Technology*, 57(14), pp. 5913–5923. Available at: https://doi.org/10.1021/acs.est.3c00012.

Luk, S.Y. *et al.* (2021) 'Soil Organic Carbon Development and Turnover in Natural and Disturbed Salt Marsh Environments', *Geophysical Research Letters*, 48(2), p. e2020GL090287. Available at: https://doi.org/10.1029/2020GL090287.

Martins, M. *et al.* (2022) ‘Carbon and Nitrogen Stocks and Burial Rates in Intertidal Vegetated Habitats of a Mesotidal Coastal Lagoon’, *Ecosystems*, 25(2), pp. 372–386. Available at: https://doi.org/10.1007/s10021-021-00660-6.

Mason, V.G. *et al.* (no date) ‘Blue carbon benefits from global saltmarsh restoration’, *Global Change Biology*, n/a(n/a). Available at: https://doi.org/10.1111/gcb.16943.

Masselink, G. *et al.* (2017) ‘Evaluation of salt marsh restoration by means of self-regulating tidal gate – Avon estuary, South Devon, UK’, *Ecological Engineering*, 106, pp. 174–190. Available at: https://doi.org/10.1016/j.ecoleng.2017.05.038.

McMahon, L. *et al.* (2023) ‘Maximizing blue carbon stocks through saltmarsh restoration’, *Frontiers in Marine Science*, 10. Available at: https://www.frontiersin.org/articles/10.3389/fmars.2023.1106607.

McMahon, L. *et al.* (2023) ‘Maximizing blue carbon stocks through saltmarsh restoration’, *Frontiers in Marine Science*, 10. Available at: https://www.frontiersin.org/articles/10.3389/fmars.2023.1106607 (Accessed: 3 April 2023).

Meyers, P.A. (1994) ‘Preservation of elemental and isotopic source identification of sedimentary organic matter’, *Chemical Geology*, 114(3), pp. 289–302. Available at: https://doi.org/10.1016/0009-2541(94)90059-0.

Middelburg, J.J. *et al.* (1997) ‘Organic Carbon Isotope Systematics of Coastal Marshes’, *Estuarine, Coastal and Shelf Science*, 45(5), pp. 681–687. Available at: https://doi.org/10.1006/ecss.1997.0247.

Milker, Y., Dura, T. and Horton, B.P. (2022) ‘The response of foraminifera to rapid sea-level rise from tidal restoration of Ni-les’tun marsh, Oregon, U.S.A’, *Marine Geology*, 445, p. 106757. Available at: https://doi.org/10.1016/j.margeo.2022.106757.

Mossman, H.L. *et al.* (2012a) ‘Constraints on Salt Marsh Development Following Managed Coastal Realignment: Dispersal Limitation or Environmental Tolerance?’, *Restoration Ecology*, 20(1), pp. 65–75. Available at: https://doi.org/10.1111/j.1526-100X.2010.00745.x.

Mossman, H.L. *et al.* (2022) ‘Rapid carbon accumulation at a saltmarsh restored by managed realignment exceeded carbon emitted in direct site construction’, *PLOS ONE*, 17(11), p. e0259033. Available at: https://doi.org/10.1371/journal.pone.0259033.

Mossman, H.L., Davy, A.J. and Grant, A. (2012b) ‘Does managed coastal realignment create saltmarshes with “equivalent biological characteristics” to natural reference sites?’, *Journal of Applied Ecology*, 49(6), pp. 1446–1456. Available at: https://doi.org/10.1111/j.1365-2664.2012.02198.x.

Mueller, P. *et al.* (2019a) ‘Assessing the long-term carbon-sequestration potential of the semi-natural salt marshes in the European Wadden Sea’, *Ecosphere*, 10(1), p. e02556. Available at: https://doi.org/10.1002/ecs2.2556.

Mueller, P. *et al.* (2019b) ‘Origin of organic carbon in the topsoil of Wadden Sea salt marshes’, *Marine Ecology Progress Series*, 624, pp. 39–50

Mueller, P. *et al.* (2020) ‘Unrecognized controls on microbial functioning in Blue Carbon ecosystems: The role of mineral enzyme stabilization and allochthonous substrate supply’, *Ecology and Evolution*, 10(2), pp. 998–1011. Available at: https://doi.org/10.1002/ece3.5962.

https://doi.org/10.5194/egusphere-2026-1559
Preprint. Discussion started: 2 April 2026

Murray J.W. (1979). British Nearshore Foraminiferids. Academic Press: London; 68 pp.

Myatt-Bell, L.B. *et al.* (2002) 'Public perception of managed realignment: Brancaster West Marsh, North Norfolk, UK', *Marine Policy*, 26(1), pp. 45–57. Available at: https://doi.org/10.1016/S0308-597X(01)00033-1.

Needelman, B.A. *et al.* (2018) 'The Science and Policy of the Verified Carbon Standard Methodology for Tidal Wetland and Seagrass Restoration', *Estuaries and Coasts*, 41(8), pp. 2159–2171. Available at: https://doi.org/10.1007/s12237-018-0429-0.

Negandhi, K. *et al.* (2019) 'Blue carbon potential of coastal wetland restoration varies with inundation and rainfall', *Scientific Reports*, 9(1), p. 4368. Available at: https://doi.org/10.1038/s41598-019-40763-8.

Ortner, M. *et al.* (2025) 'Assignment of Thermogravimetric Mass Losses to Soil Organic Matter, Its Fractions Hot Water–Extractable and Microbial Biomass Carbon, and Organic Matter–Stabilizing Soil Mineral Properties', *Journal of Plant Nutrition and Soil Science*, 188(2), pp. 334–349. Available at: https://doi.org/10.1002/jpln.202400498.

Pallasser, R., Minasny, B. and McBratney, A.B. (2013) 'Soil carbon determination by thermogravimetrics', *PeerJ*, 1, p. e6. Available at: https://doi.org/10.7717/peerj.6.

Pethick, J. (2002) 'Estuarine and Tidal Wetland Restoration in the United Kingdom: Policy Versus Practice', *Restoration Ecology*, 10(3), pp. 431–437. Available at: https://doi.org/10.1046/j.1526-100X.2002.01033.x.

Poppe, K.L. and Rybczyk, J.M. (2021) 'Tidal marsh restoration enhances sediment accretion and carbon accumulation in the Stillaguamish River estuary, Washington', *PLOS ONE*, 16(9), p. e0257244. Available at: https://doi.org/10.1371/journal.pone.0257244.

R Core Team (2024). R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. <https://www.R-project.org/>.

Saffert, H. and Thomas, E. (1998) 'Living foraminifera and total populations in salt marsh peat cores: Kelsey Marsh (Clinton, CT) and the Great Marshes (Barnstable, MA)', *Marine Micropaleontology*, 33(3), pp. 175–202. Available at: https://doi.org/10.1016/S0377-8398(97)00035-2.

Santini, N.S. *et al.* (2019) 'Natural and Regenerated Saltmarshes Exhibit Similar Soil and Belowground Organic Carbon Stocks, Root Production and Soil Respiration', *Ecosystems*, 22(8), pp. 1803–1822. Available at: https://doi.org/10.1007/s10021-019-00373-x.

Saunders, M.I. *et al.* (2020) 'Bright Spots in Coastal Marine Ecosystem Restoration', *Current Biology*, 30(24), pp. R1500–R1510. Available at: https://doi.org/10.1016/j.cub.2020.10.056.

Scott, D.B. and Medioli, F.S. (1980) 'Living vs. total foraminiferal populations: their relative usefulness in paleoecology.', *Journal of Paleontology*, 54(4), pp. 814–831.

Sen Gupta, B.K. (Ed.), 2003. 'Modern Foraminifera', Kluwer Academic Publishers, Dordrecht, 371 pp., Available at: https://doi.org/10.1007/0-306-48104-9.

Shafer, D.J. and Streever, W.J. (2000) 'A comparison of 28 natural and dredged material salt marshes in Texas with an emphasis on geomorphological variables', *Wetlands Ecology and Management*, 8(5), pp. 353–366. Available at: https://doi.org/10.1023/A:1008491421739.

Shiau, Y.-J. *et al.* (2019) 'Carbon storage potential in a recently created brackish marsh in eastern North Carolina, USA', *Ecological Engineering*, 127, pp. 579–588. Available at: https://doi.org/10.1016/j.ecoleng.2018.09.007.

Smeaton, C. and Austin, W.E.N. (2022) 'Quality Not Quantity: Prioritizing the Management of Sedimentary Organic Matter Across Continental Shelf Seas', *Geophysical Research Letters*, 49(5), p. e2021GL097481. Available at: https://doi.org/10.1029/2021GL097481.

Smeaton, C. *et al.* (2023) 'Organic carbon stocks of Great British saltmarshes', *Frontiers in Marine Science*, 10. Available at: https://www.frontiersin.org/articles/10.3389/fmars.2023.1229486

Smeaton, C. *et al.* (2024) 'Organic carbon accumulation in British saltmarshes', *Science of The Total Environment*, p. 172104. Available at: https://doi.org/10.1016/j.scitotenv.2024.172104.

Smeaton, C.; Miller, L.C.; Ladd, C.J.T.; O'Dell, A.; Austin, W.E.N. (2022). 'Bulk elemental and stable isotope composition of organic matter from terrestrial, intertidal, and marine environments, UK, 2016-2021', NERC EDS Environmental Information Data Centre. https://doi.org/10.5285/a445a7a8-528d-4e0b-9094-28cbcd449367

Spencer, K.L. *et al.* (2008) 'Physicochemical changes in sediments at Orplands Farm, Essex, UK following 8 years of managed realignment', *Estuarine, Coastal and Shelf Science*, 76(3), pp. 608–619. Available at: https://doi.org/10.1016/j.ecss.2007.07.029.

Spencer, K.L. *et al.* (2017) 'The impact of pre-restoration land-use and disturbance on sediment structure, hydrology and the sediment geochemical environment in restored saltmarshes', *Science of The Total Environment*, 587–588, pp. 47–58. Available at: https://doi.org/10.1016/j.scitotenv.2016.11.032.

Spencer, T. *et al.* (2012) 'Surface elevation change in natural and re-created intertidal habitats, eastern England, UK, with particular reference to Freiston Shore', *Wetlands Ecology and Management*, 20(1), pp. 9–33. Available at: https://doi.org/10.1007/s11273-011-9238-y.

Symonds, A.M. and Collins, M.B. (2007) 'The establishment and degeneration of a temporary creek system in response to managed coastal realignment: The Wash, UK', *Earth Surface Processes and Landforms*, 32(12), pp. 1783–1796. Available at: https://doi.org/10.1002/esp.1495.

Tempest, J.A., Harvey, G.L. and Spencer, K.L. (2015) 'Modified sediments and subsurface hydrology in natural and recreated salt marshes and implications for delivery of ecosystem services', *Hydrological Processes*, 29(10), pp. 2346–2357. Available at: https://doi.org/10.1002/hyp.10368.

Unger, V. *et al.* (2016) 'Stability of organic carbon accumulating in Spartina alterniflora-dominated salt marshes of the Mid-Atlantic U.S.', *Estuarine, Coastal and Shelf Science*, 182, pp. 179–189. Available at: https://doi.org/10.1016/j.ecss.2016.10.001.

Van de Broek, M. *et al.* (2018) 'Long-term organic carbon sequestration in tidal marsh sediments is dominated by old-aged allochthonous inputs in a macrotidal estuary', *Global Change Biology*, 24(6), pp. 2498–2512. Available at: https://doi.org/10.1111/gcb.14089.

van Oevelen, D. *et al.* (2000) Literatuuronderzoek naar ontpolderingen. IN.R.2000.7. Instituut voor Natuurbehoud, Brussel.

Vaughan, A. (2019) 'Moving away from the coast', *New Scientist*, 243(3245), p. 8. Available at: https://doi.org/10.1016/S0262-4079(19)31601-X.

Wang, F. *et al.* (2019) 'Water salinity and inundation control soil carbon decomposition during salt marsh restoration: An incubation experiment', *Ecology and Evolution*, 9(4), pp. 1911–1921. Available at: https://doi.org/10.1002/ece3.4884.

Wang, J.J. *et al.* (2011) 'Soil Carbon Characteristics in Two Mississippi River Deltaic Marshland Profiles', *Wetlands*, 31(1), pp. 157–166. Available at: https://doi.org/10.1007/s13157-010-0130-y.

Weinmann, A.E. *et al.* (2019) 'Effects of sampling site, season, and substrate on foraminiferal assemblages grown from propagule banks from lagoon sediments of Corfu Island (Greece, Ionian Sea)', *PLOS ONE*, 14(6), p. e0219015. Available at: https://doi.org/10.1371/journal.pone.0219015.

Williams, N. and Dale, J. (2023) 'Unmanaged realignment: Recent examples and the morphological evolution of naturally breached flood defences', *Ocean & Coastal Management*, 242, p. 106715. Available at: https://doi.org/10.1016/j.ocecoaman.2023.106715.

Williamson, P. *et al.* (2025) 'Additionality Revisited for Blue Carbon Ecosystems: Ensuring Real Climate Mitigation', *Global Change Biology*, 31(4), p. e70181.

Wollenberg, J.T., Ollerhead, J. and Chmura, G.L. (2018) 'Rapid carbon accumulation following managed realignment on the Bay of Fundy', *PLOS ONE*, 13(3), p. e0193930. Available at: https://doi.org/10.1371/journal.pone.0193930.

Wolters, M., Garbutt, A. and Bakker, J. (2005) 'Salt-marsh restoration: Evaluating the success of de-embankments in north-west Europe', *Biological Conservation*, 123, pp. 249–268. Available at: https://doi.org/10.1016/j.biocon.2004.11.013.

Woodward-Rowe, H. *et al.* (2025) 'Sedimentary blue carbon around decommissioned oil and gas platforms in the North Sea', *Marine Pollution Bulletin*, 219, p. 118250. Available at: https://doi.org/10.1016/j.marpolbul.2025.118250.

Xia, S. *et al.* (2021) 'Distribution, sources, and decomposition of soil organic matter along a salinity gradient in estuarine wetlands characterized by C:N ratio, δ13C- δ15N, and lignin biomarker', *Global Change Biology*, 27(2), pp. 417–434. Available at: https://doi.org/10.1111/gcb.15403.

Zhou, J. *et al.* (2007) 'Spatial variations of carbon, nitrogen, phosphorous and sulfur in the salt marsh sediments of the Yangtze Estuary in China', *Estuarine, Coastal and Shelf Science*, 71(1), pp. 47–59. Available at: https://doi.org/10.1016/j.ecss.2006.08.012.