Estuarine, Coastal and Shelf Science 282 (2023) 108223

Contents lists available at ScienceDirect

Estuarine, Coastal and Shelf Science

journal homepage: www.elsevier.com/locate/ecss

Carbon accumulation and storage across contrasting saltmarshes of Scotland

Lucy C. Miller[a], Craig Smeaton[a,*], Handong Yang[b], William E.N. Austin[a,c]

[a] School of Geography & Sustainable Development, Irvine Building, North Street, University of St Andrews, Fife, KY16 9AL, UK
[b] Department of Geography, University College London, North-West Wing, Gower Street, London, WC1E 6BT, UK
[c] Scottish Association of Marine Science, Scottish Marine Institute, Oban, Argyll, PA37 1QA, UK

ARTICLE INFO

Keywords:
Carbon
Wetlands
Coastal
Soil
Belowground
Aboveground

ABSTRACT

Saltmarshes are acknowledged to be "carbon hotspots" due to their capacity to trap and store large quantities of carbon (C) within their soils and potentially have the ability to regulate climate over different timescales. In-turn governments and international organizations are now recognizing the need to include these intertidal ecosystems in national and global C accounting. Yet, in many regions, estimates of organic carbon (OC) storage and the rate at which OC is buried in saltmarsh soils either do not exist or are not at the scale necessary for inclusion in national C budgets. Here we bring together tools from across the geosciences to investigate the quantity of OC held within the soil and above/belowground biomass, alongside estimates of the rate at which OC accumulates and the source of the OC within the soils of four contrasting Scottish saltmarshes. Using radiometric dating techniques it is estimated that OC accumulates at a rate of between 29.1 and 198.1 g C m^{-2} yr^{-1} across the different study sites. In contrast, the source of the OC varies little across the sites with 73%–99% of the OC within the saltmarsh soil originating from terrestrial/in situ sources; marine-derived OC plays a minor role in the development of the saltmarsh OC stocks. Using average values derived from the four sites it is possible to make first-order estimates of saltmarsh OC stocks and accumulation rates for all Scotland's 240 mapped saltmarshes (58.68 km^2). It is estimated that across Scotland saltmarsh habitat stores 1.15 ± 0.21 Mt OC which is supplemented by an additional 4385 ± 481 tonnes of OC each year.

1. Introduction

Blue carbon ecosystems (Nellemann and Corcoran, 2009) such as saltmarshes are valued for their ability to trap and store large amounts of carbon (C) in their soil (McLeod et al., 2011; Duarte et al., 2013). It is estimated that saltmarsh habitats occupy 54,951 km^2 (Mcowen et al., 2017) of global coastlines and store between 0.4 and 6.5 Pg of organic carbon (OC) within their soils (McLeod et al., 2011; Duarte et al., 2013). Additionally, it is estimated that a further 4.8–87.3 Tg C yr^{-1} is trapped and stored in saltmarshes (McLeod et al., 2011). The magnitude of these saltmarsh OC stores and their potential to assist in regulating climate is now widely recognized (Macreadie et al., 2019, 2021). However, saltmarshes are degrading globally at a rate of 1–2% yr^{-1} increasing the likelihood that these globally significant OC resources will be lost by the end of the century unless significant efforts are made to preserve these habitats (Duarte et al., 2008; Crosby et al., 2016; Horton et al., 2018; Schuerch et al., 2018). Quantifying the OC stored and accumulating in saltmarshes is crucial to understanding the coastal C cycle, national carbon accounting, the carbon and climate impact of habitat loss, and supports habitat restoration (Granek et al., 2010; Theuerkauf et al., 2015; Rogers et al., 2019).

Key to these goals is quantifying the amount of OC held with saltmarshes, the rate at which OC accumulates and the source of the OC. Globally, such C assessments have largely focused on tropical/subtropical areas such as Australia (Lovelock et al., 2014; Brown et al., 2016; Kelleway et al., 2016) and the Gulf of Mexico (Thorhaug et al., 2019; Vaughn et al., 2020) with significantly less attention to systems found in temperate and boreal regions, with very limited estimates for NE Atlantic saltmarshes (Mueller et al., 2019a, 2019b). To date, assessments of the total OC stocks of saltmarshes in the United Kingdom have been limited to a single marsh (Porter et al., 2020; Ladd et al., 2022) or focused on understanding the OC stored in the uppermost (normally 10 cm) soil (Ford et al., 2019; Harvey et al., 2019; Austin et al., 2021; Smeaton et al., 2022a). Further, our quantification of the

* Corresponding author.
E-mail address: cs244@st-andrews.ac.uk (C. Smeaton).

https://doi.org/10.1016/j.ecss.2023.108223
Received 7 July 2022; Received in revised form 16 December 2022; Accepted 13 January 2023
Available online 17 January 2023

source of the OC held within these marshes' is poor, with little data available on autochthonous primary production and the import of allochthonous OC from external sources (terrestrial and marine environments) within an UK context. Understanding allochthonous and autochthonous OC contributions to the saltmarsh is crucial for C reporting as only autochthonous OC can be considered sequestered from the atmosphere and to provide a climate regulation service (Mueller et al., 2019b). In contrast the allochthonous OC and its role in climate mitigation is currently unclear. Finally, saltmarshes in the UK lack OC accumulation rate (OCAR) data with most studies (Burrows et al., 2017, 2021; Gregg et al., 2021) applying the global OCAR of 244.7 ± 26 g C m^{-2} yr^{-1} (Chmura et al., 2003; Ouyang and Lee, 2014). Using this global average is likely to result in significant overestimations due to the average OCAR value being largely derived from tropical/sub-tropical saltmarshes. The lack of estimates of the quantity of OC held within saltmarshes and the rate at which OC is accumulating is hindering our ability to quantify Scotland's blue carbon resource and the role these saltmarshes play in global climate regulation.

Saltmarsh habitat is found across the Scottish coastline in highly contrasting environments ranging between fringing systems in rivers to estuarine and embayment marshes (Haynes, 2016). The location of these marshes around the coast results in significant difference in source and quantity of sediment supplied to the marshes. (Ladd et al., 2021). The large geographical spread of the habitat results in marshes being found in settings dominated by both mud and sand which results in high variability in the marshes potential to trap and store OC (Kelleway et al., 2016; Smeaton et al., 2022b). Further, due to changes in Holocene sea level (Shennan et al., 2018) the Scottish saltmarshes are some of the oldest in the UK. The marshes in the North of Scotland formed over the last two millennia compared to marshes in the South of the UK which have only developed in the last 300 years (Barlow et al., 2014; Long et al., 2014). The extended timeframe for marsh development in Scotland potentially allows significant OC stocks to build in comparison to other UK settings. These characteristics make Scottish saltmarshes an ideal location to investigate OC storage and burial dynamics in a North Atlantic setting.

In this study, we (**i**) bring together field and laboratory analyses to quantify the OC stored in the above and belowground biomass alongside the soil of four contrasting Scottish saltmarshes, (**ii**) estimate the rate at which OC is accumulating in the different marsh zones of these saltmarshes using radiometric dating techniques, (**iii**) identify and quantify the source of the OC held within the soil of the marshes by using stable isotopes in conjunction with a Bayesian mixing model, and (**iv**) using geospatial techniques upscale these finding to provide the first national assessment of OC storage and accumulation in Scottish saltmarsh habitats. Through a better understanding of these factors, we aim to provide greater insight into the deposition, accumulation, and storage of OC in Scotland's saltmarsh ecosystems. These insights allow comparisons to be made with other global blue carbon ecosystems and allow the role these saltmarshes play in the C cycle to be better defined, providing a foundation for saltmarshes to be included in national C inventories.

2. Study sites

Saltmarsh occupies 58.68 km^2 of Scotland's coastline (Haynes, 2016, Fig. 1A) which represents ~13% of saltmarsh habitat in the UK (Burden et al., 2020). The geomorphology of the Scottish coastline results in a high number (240 surveyed) of small (mean size: 0.25 km^2) saltmarshes with only nine Scottish saltmarshes extending beyond 1 km^2 in size. The saltmarshes at Dornoch Point (Fig. 1B), Morrich More (Fig. 1C), Skinflats (Fig. 1D) and Wigtown (Fig. 1E) are the focus of this study. These marshes have different physical characteristics (Table 1) and are broadly representative of Scottish saltmarsh habitat in the major estuaries of the east and southwest (Solway Firth) coasts (Adam, 1978; Burd, 1989; Haynes, 2016). Together these four saltmarshes occupy an area of 11.83 km^2, which is equivalent to 20.17% of Scotland's mapped saltmarsh

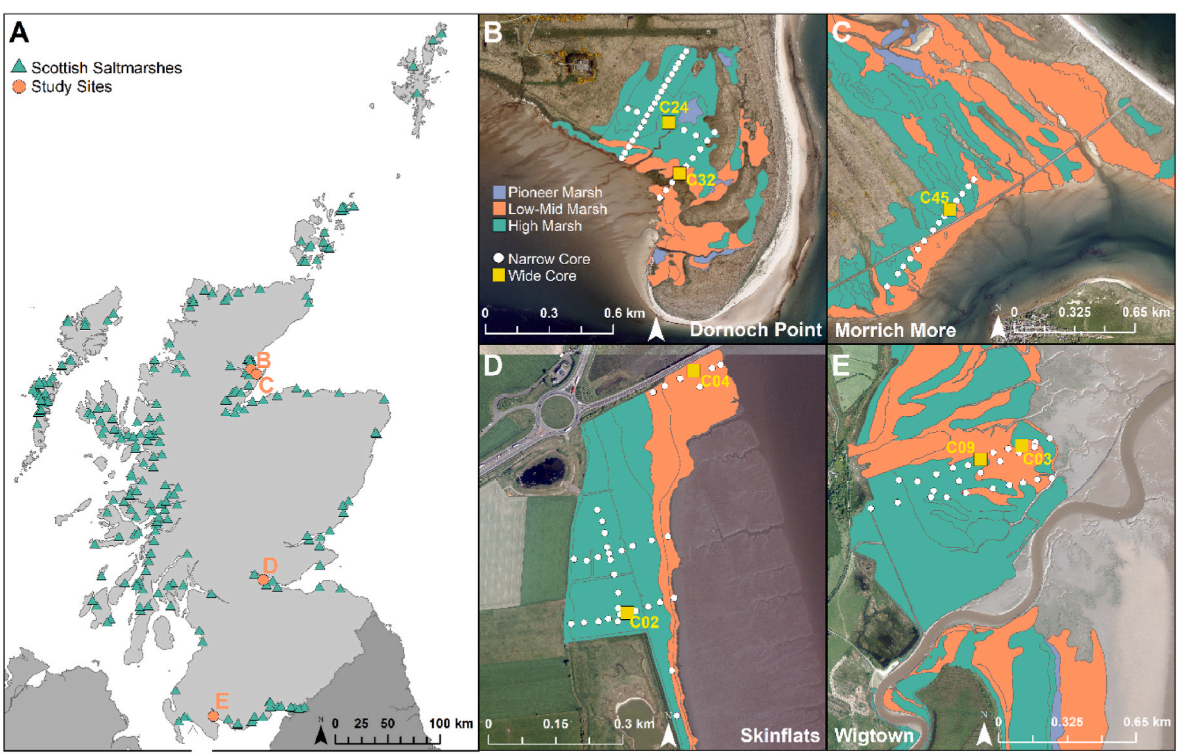

Fig. 1. Location of sampling sites. (**A**) Location of the four study sites in the context of all Scotland's mapped saltmarshes (>0.03 km^2) (Haynes, 2016). Soil core collection sites (*white symbols*) at (**B**) Dornoch Point, (**C**) Morrich More, (**D**) Skinflats, (**E**) Wigtown. Vegetation communities used to classify the marsh zones are detailed in Supplementary Table 1. Locations and the elevations of the wide diameter cores (*yellow labelled squares*) are detailed in Supplementary Table 2. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Table 1
Physical characteristics of the four saltmarshes included in the study.

Saltmarsh	Type	Area of Marsh Zones (km^2)				% of Scotland's Saltmarsh
		Pioneer	Low-Mid	High	Total	
Dornoch Point	Back-barrier	0.02	0.15	0.26	0.42	0.72
Morrich More	Back-barrier	0.06	2.08	2.17	4.30	7.33
Skinflats	Fringing (*Fluvial*)	0.00	0.15	0.22	0.37	0.63
Wigtown	Estuarine	0.57	1.94	4.22	6.74	11.49
Scotland	—	**4.91**	**16.96**	**36.81**	**58.68**	—

habitat.

3. Methods

3.1. Sampling

To estimate saltmarsh soil OC stocks, cores were retrieved in autumn 2019 along three transects on each marsh (Fig. 1B–E). The idealised sampling strategy would have two transects running perpendicular to the shore intersecting the different marsh zones (high, low-mid and where present pioneer zones), and a third transect running diagonally on the shore to intersect the two other transects (Ladd et al., 2022). (Fig. 1B–E; *white points*). At each site, the positioning of the transects was adapted to capture site specific conditions (e.g., geomorphology, hydrology) and avoid restricted areas (i.e., Morrich More), while still providing the samples required to estimate saltmarsh OC stocks and burial rates. Sample locations were spaced evenly and in proportion to marsh width. Coordinates of each coring site were taken using a DGPS to an average accuracy of ~2 cm. Soil cores were extracted using a narrow (3 cm) gouge corer pushed by hand to either a depth of 1 m, or until a resistant basal layer was reached, assuring minimal compaction following the approach described by Smeaton et al. (2020). Soil core profiles were described using the Troels-Smith classification scheme (Troels-Smith, 1955). Cores were sub-sampled in the field at depths of 0–2 cm, 4–6 cm, 10–12 cm, 20–22 cm, 30–32 cm and so on, until 90–92 cm. A total of 671 sub-samples from across the four saltmarshes (97 Dornoch Point; 42 Morrich More; 298 Skinflats and 234 from Wigtown) were collected. Additionally, to calculate OC accumulation at these saltmarshes' wide diameter (60 mm) gouge cores were collected to a depth of 1 m from the low-mid and high marsh zones at each saltmarsh (Fig. 1B–E; *yellow squares*), except for Morrich More where only a high marsh core was collected due to restricted site access (Fig. 1B). These cores were transported back intact to the laboratory.

Samples were collected in the summer months of 2020, to understand the contribution of aboveground (vegetation) and belowground biomass (roots, stolons and rhizomes) to the saltmarsh OC stock. The aboveground vegetation was surveyed within 1 m^2 quadrats at 27 sites across the four saltmarshes. Vegetation composition was described following the British National Vegetation Classification (NVC) scheme (Rodwell, 2000; Sup. Table 1). Within each quadrat living vegetation was cut at soil level from an area of 0.125 m^2 (Harvey et al., 2019). Belowground biomass cores were collected to a depth of 10 cm (Ford et al., 2016) using a wide diameter (6 cm) gouge corer from 11 locations across the low-mid and high marsh zones.

3.2. Soil profiles

On return to the laboratory the wide diameter cores were described according to the Troels-Smith classification scheme (Troels-Smith, 1955). Through the examination of the Troels-Smith (1955) descriptions from both the narrow and wide diameter cores four distinct soil units were identified across the saltmarshes associated with saltmarsh and pre-saltmarsh environments (i.e., intertidal flat):

Basal unit: This unit is formed of either fine silt or sand with low quantities of visible organic matter which is characteristic of the pre-saltmarsh intertidal environment (i.e., mudflat or sandflat).
Transitional unit: The soil in this unit has similarities to the basal unit but with higher quantities of organic matter. This unit is characteristic of pioneer marsh formation and represents the transition between intertidal flat and saltmarsh.
Humified peat unit: This unit is formed of humified organic matter related to the decomposition of vegetation. This unit is found in both the low-mid and high marsh zones but is not found in every core.
Fibrous peat unit: This unit is normally the uppermost in the soil profile and is formed of fibrous peat-like organic matter with large quantities of living roots. This unit is found in both the low-mid and high marsh zones.

The fibrous peat, humified peat and transitional soil units are associated with saltmarsh habitats and together will be referred to as saltmarsh soil in this study. The basal unit represents the pre-saltmarsh environment either in the form of mudflat or sand flat (e.g., Mueller et al., 2019b).

3.3. Soil physical properties

The seven wide diameter cores were sliced at 1 cm intervals down the length of the core resulting in 305 samples in total. The 305 core slices were freeze dried to assure sample integrity for stable isotope analysis, while the 671 narrow core sub-samples were oven dried at 60 °C for three days. Prior to, and post drying the samples were weighed allowing the wet bulk density, dry bulk density and water content to be calculated following standard methodologies (Athy, 1930; Appleby and Oldfield, 1978; Dadey et al., 1992).

3.4. Above and belowground biomass

The harvested aboveground biomass samples were oven dried at 60 °C for three days. After drying, the material was weighed and the aboveground biomass calculated for each harvested quadrat (Harvey et al., 2019; Ford et al., 2019). The belowground biomass cores were gently loosened up by hand, prior to gentle shaking for 3 h in a 5% solution of sodium hexametaphosphate (Penk et al., 2020). Finally, the remaining soil was gently washed from the roots, stolons and rhizomes, through a 500 μm sieve (Harvey et al., 2019). This mesh size permits the removal of fine debris while retaining fine fragments. Fragments on the sieve were combined with the main portion of belowground biomass and dried at 60 °C for three days. Once dry, the material was weighed and the belowground biomass calculated.

3.5. Bulk elemental and stable isotope analysis

The dried soil samples from the narrow diameter gouge cores ($n = 671$), vegetation ($n = 27$) and roots ($n = 33$, *11 samples in triplicate*) were homogenized to a fine powder in preparation for analysis. 30 mg of sample was weighed into a steel crucible and placed into a total organic carbon (TOC) analyser (Elementar "Soli TOC"). The Soli TOC analyser used the temperature gradient elemental analysis method (DIN 19539, 2015; Natali et al., 2020; Smeaton et al., 2021a) to quantify OC and inorganic carbon (IC) from a single untreated sample. This is achieved through ramped heating of the sample at a rate of 70 °C min^{-1} through sequential furnace temperatures of 400 °C, 600 °C and 900 °C. The carbon dioxide (CO_2) evolved at the different temperatures represents the contribution of the various carbon fractions of the sample, i.e.: 0–400 °C TOC; 400–600 °C residual oxidizable carbon (ROC), more commonly referred to as elemental C; and 600–900 °C total inorganic carbon (TIC). The amount of CO_2 produced from each of the temperature windows is measured by infrared spectrometry. For the purposes of this study, OC was calculated as the sum of TOC and ROC, while IC equals the TIC value. The standard deviation of triplicate measurements ($n = 40$) was TOC: 0.06%, ROC: 0.05% and TIC: 0.11%. Further quality control was assured by the repeat analysis of standard reference material B2290 (TOC/ROC/TIC silty soil standard from Elemental Microanalysis, United Kingdom) these analyses of standards deviated from the reference value by: TOC = 0.05%, ROC: 0.07% and TIC = 0.10% ($n = 75$).

The freeze-dried samples from the wide diameter cores were homogenized to a fine powder prior to analysis. To determine the bulk elemental (OC and nitrogen (N)) and stable isotope ($\delta^{13}C_{org}$ and $\delta^{15}N$) composition approximately 12 mg of processed soil was placed into tin capsules and sealed. A further 12 mg was placed into silver capsules. To remove carbonate ($CaCO_3$) the acid fumigation method (Harris et al., 2001) was utilized. The samples encapsulated in silver were placed in a desiccator with a beaker of concentrated (37%) HCl at a temperature of 60 °C for 72 h. To remove excess acid the samples were neutralized by replacing the acid beaker with ~20 g NaOH pellets for 60 °C for 72 h (Bao et al., 2019). The stable isotope analyses were undertaken at the James Hutton Institute using an elemental analyser coupled to an isotope ratio mass spectrometer (EA-IRMS). The acidified samples were analysed for OC and $\delta^{13}C_{org}$, while N and $\delta^{15}N$ values were produced from the tin-encapsulated samples. By analysing the N and $\delta^{15}N$ separately, we reduce the potential risk of altering the isotopic values through the acid fumigation step (Kennedy et al., 2005). Triplicate measurements of samples ($n = 20$) produced standard deviations (1σ) of 0.01% for N and 0.06‰ for $\delta^{15}N$, 0.03% for OC and 0.08‰ for $\delta^{13}C_{org}$. Further quality control was assured by repeat analysis of high OC sediment standard (B2151) with reference values. The reference standards ($n = 28$) deviated from the known values by: OC = 0.07%, $\delta^{13}C$ = 0.11‰, N = 0.02% and $\delta^{15}N$ = 0.13‰. The isotope values are reported in standard delta notation relative to Vienna Peedee belemnite (VPDB) and air. The C/N ratios are reported as molar ratios: C/N = (OC/12)/(N/14).

3.6. Saltmarsh OC stock estimation

3.6.1. Aboveground and belowground OC stock

The above and belowground biomass was combined with the measured OC content of that biomass (Section 3.5) to calculate mean OC storage values (kg C m^{-2}) for each marsh zone across the four saltmarshes. The areal extent of the marsh zones for each saltmarsh was acquired from the best quality existing mapping (Haynes, 2016, Fig. 1). To account for the potential expansion and/or retraction of these saltmarshes (Ladd, 2021; Ladd et al., 2021) since they were surveyed in 2012, an error of ±5% was applied (Austin et al., 2021; Smeaton et al., 2022c). The areal extents of low-mid and high marsh (Table 1) were combined with the OC storage values to calculate the OC stock (eqs. 1-2).

$$\text{Aboveground biomass C stock (kg C)} = \text{area (m}^2\text{) x aboveground biomass C storage (kg C m}^{-2}\text{)} \quad \text{(eq.1)}$$

$$\text{Belowground biomass C stock (kg C)} = \text{area (m}^2\text{) x belowground biomass C storage (kg C m}^{-2}\text{)} \quad \text{(eq.2)}$$

The calculations were carried out in a Markov Chain Monte Carlo (MCMC) framework using the OpenBUGS software package (Lunn et al., 2009). MCMC analysis was applied by taking 1,000,000 out of 100,000, 000 random samples from a normal distribution of each variable (area, aboveground OC storage, belowground OC storage) to populate equations (*eqs. 1-2*). This process generates a significant quantity of solutions which follow a normal distribution. The application of standard descriptive statistical techniques to the pool of generated solutions allows the mean, standard deviation, 5th and 95th percentiles to be calculated.

3.6.2. Soil OC stock

The soil OC stocks were determined for the four saltmarshes following the calculation steps outlined in Smeaton et al. (2020). Briefly, the mean (and standard deviation) depth, dry bulk density and OC content for each soil unit was calculated from the sub-sampled narrow cores. The areal extents of the marsh zones were taken from Haynes (2016), again an error of ±5% was again applied. As the sub-samples from the narrow diameter cores contain belowground biomass the resulting OC stock estimates represent a belowground OC stock (i.e. soil + biomass), to calculate the quantity of OC stored in the soil the calculated belowground biomass OC stock (Section 3.6.1) must be removed.

These calculations (eqs. (3)–(6)) were undertaken to quantify the OC stock for the saltmarsh soil (fibrous peat, humified peat and transitional soil units). Additionally, the soil stock down to a depth of 1 m (saltmarsh soil units and basal unit) was also calculated for each marsh zone and saltmarsh as a whole. Where cores did not extend to the 1 m the basal unit values were extrapolated. These calculations were carried out within the OpenBUGS (Lunn et al., 2009) MCMC framework.

$$\text{Volume (m}^3\text{)} = \text{area (m}^2\text{) x soil unit depth (m)} \quad \text{(eq.3)}$$

$$\text{Mass (kg)} = \text{volume (m}^3\text{) x dry bulk density (kg m}^{-3}\text{)} \quad \text{(eq.4)}$$

$$\text{Belowground OC stock (kg C)} = \text{mass (kg) x OC (\%)} \quad \text{(eq.5)}$$

$$\text{Soil OC stock (kg C)} = \text{belowground OC stock (kg C)} - \text{belowground biomass OC stock (kg C)} \quad \text{(eq.6)}$$

3.7. Understanding the source of the OC

In this study, $\delta^{13}C_{org}$, $\delta^{15}N$ and the C/N values were used in conjunction with the open-source Bayesian isotope mixing model FRUITS to determine the fraction of terrestrial/in situ and marine OC input to the saltmarsh soils (Fernandes et al. (2014); Smeaton and Austin (2017). Unlike traditional binary mixing models (i.e., Thornton and McManus, 1994) FRUITS uses multiple tracers ($\delta^{13}C_{org}$, $\delta^{15}N$, C/N) concurrently within a Bayesian framework, this approach allows the estimation of the proportional contribution of terrestrial/in situ and marine derived OC.

Values representing the OC sources (e.g., in situ production, terrestrial and marine) were determined from samples collected from the catchments of the saltmarshes and coastline of Scotland (Smeaton et al., 2022a; Sup. Table 9). The terrestrial source values were determined from samples of different soils ($n = 37$) and vegetation ($n = 37$) across an array of land uses; the marine source value was calculated from 50 samples of phytoplankton, zooplankton, micro-algae, and macro-algae (Smeaton et al., 2022a). To understand the in situ saltmarsh OC production the samples collected to understand aboveground ($n = 42$) and belowground ($n = 33$) biomass (Section 3.4) underwent bulk elemental and stable isotope analysis to quantify their source values. As observed in other studies (Kennedy et al., 2010; Saintilan et al., 2013; Greiner et al., 2016; Geraldi et al., 2019; Pondell and Canuel, 2022) there is significant overlap in the $\delta^{13}C_{org}$, $\delta^{15}N$ and C/N values of the saltmarsh biomass and soil and vegetation derived from the terrestrial environment (Sup. Table 9) which prevents us from differentiating between in situ OC production and terrestrial OC input to the saltmarshes.

Unlike the west coast of Scotland where recent work has shown minimal (<0.1%) input of fossil/petrogenic C to inshore OC stores (Smeaton et al., 2021b) the catchments where the four saltmarshes are situated are dominated by sedimentary rock. Therefore, there may be potential for fossil/petrogenic C to be incorporated into the saltmarsh soils, potentially altering the $\delta^{13}C_{org}$ values. Currently, it has not been possible to quantify fossil/petrogenic C input or its influence on the $\delta^{13}C_{org}$ values in these systems.

3.8. Sedimentation and OC accumulation

3.8.1. Radiometric dating

Samples from the seven wide diameter cores were analysed for ^{210}Pb, ^{226}Ra, ^{137}Cs by direct gamma assay at the Environmental Radiometric Facility at University College London. The analyses were carried out using ORTEC HPGe GWL series well-type coaxial low background intrinsic germanium detector. The ^{210}Pb was determined via its gamma emissions at 46.5 keV, ^{226}Ra at 295 keV and 352 keV following 3 weeks storage in sealed containers to allow radioactive equilibration. The ^{137}Cs and ^{241}Am were measured by their emissions at 662 keV and 59.5 keV respectively (Appleby et al., 1986). The absolute efficiencies of the detectors were determined using calibrated sources and sediment samples of known activity. Corrections were made for the effect of self-absorption of low energy gamma rays within the sample (Appleby and Oldfield, 1992).

3.8.2. Saltmarsh OC accumulation

The ^{210}Pb, ^{137}Cs and ^{241}Am data were utilized to calculate the linear sedimentation rates (LSR) and mass accumulation rates (MAR) of the saltmarsh soils alongside OCARs. The ^{210}Pb data was used with constant flux–constant sedimentation (CF–CS) model (Krishnaswamy et al., 1971) to calculate LSRs in the seven cores. The peaks in ^{137}Cs and ^{241}Am activity were used to constrain the ^{210}Pb derived LSRs as these peaks can be attributed to fallout derived from nuclear weapons testing (peak - 1964) and the 1986 Chernobyl accident.

Due to the proximity of Wigtown marsh (Fig. 1E) to the Sellafield nuclear fuel reprocessing plant, it is expected that the concentration of radioactive elements derived from the accidental and authorized releases of radioactive waste (Harvey et al., 2007) will overwhelm the natural ^{210}Pb signal as well as the ^{137}Cs and ^{241}Am concentrations derived from weapons testing and Chernobyl. Within the saltmarshes of the Solway Firth peaks in both ^{137}Cs and ^{241}Am corresponding to the maximum radioactive discharge from Sellafield in the mid-1970s (Gray et al., 1995) have been observed in the soils (Mackenzie and Scott, 1993; Pulford et al., 1998; Morris et al., 2000; Harvey et al., 2007), these peaks provide a chronological marker which can be used to calculate LSRs at the Wigtown saltmarsh.

Mass accumulation rates (MARs, g m^{-2} y^{-1}) were calculated by multiplying the LSR by mean soil bulk density values from each core. The OCARs (g C m^{-2} yr^{-1}) were calculated by multiplying the weight fraction of OC, which for each core is represented as the average % OC values of all 1 cm sediment slices (Smith et al., 2015; Ramirez et al., 2016). Similarly, the terrestrial/in situ OC accumulation rates were calculated by multiplying the MAR by the average weight fraction of OC originating from within the saltmarsh and the terrestrial environment determined by the isotope mixing models (Section 3.5). The approach produces globally comparable rates but there are inherent biases that must be recognized. The calculations are based on LSR and average OC values; this approach struggles to take into consideration changes in sedimentation, OC input and preservation conditions potentially leading to over- and under-estimations of the long-term rates.

The second approach utilises the constant rate of supply (CRS) model (Appleby and Oldfield, 1978). The CRS model is based upon the assumption that there is a constant rate of supply of unsupported ^{210}Pb (^{210}Pb$_{supp}$) to the sediment (Appleby and Oldfield, 1978). The model assumes a constant flux of ^{210}Pb to the sediments through time. The initial specific activity is variable and inversely related to MAR with higher MAR leading to lower excess ^{210}Pb (^{210}Pb$_{ex}$) and vice versa. The dating is based upon the comparison of ^{210}Pb$_{ex}$ inventories below a given depth and the integration of ^{210}Pb$_{ex}$ specific activity as a function of the cumulative mass with the overall ^{210}Pb$_{ex}$ inventory in the soil core (Appleby, 2002; Sanchez-Cabeza and Ruiz-Fernández, 2012; Arias-Ortiz et al., 2018). As before, the ^{137}Cs data can be used to check the quality of the CRS model by comparing the known depth and age of ^{137}Cs peaks to the modelled age of that depth horizon derived from ^{210}Pb. The MARs calculated using the CRS model were combined with depth-specific OC values to calculate the OCAR and estimate the quantity of OC originating from the saltmarsh and terrestrial environments that accumulates annually. This approach to calculating OCARs overcome many of the issues of the simpler linear approach, yet care must be taken when interpreting between changes in OC source input versus degradation effects (Arndt et al., 2013; Middelburg, 2018; Larowe et al., 2020).

To quantify the OC that accumulates in the four saltmarshes the OCAR from the cores were combined with the areal extent of the low-mid and high marsh (Table 1) to estimate the total quantity of OC annually accumulating in each zone and marsh as a whole. As a core was not collected for the low-mid marsh at Morrich More a surrogate OCAR value from the nearby Dornoch Point marsh (Fig. 1) was used.

3.9. Upscaling

To estimate the OC stock of all Scotland's 240 mapped saltmarshes (Haynes, 2016), the OC stocks (soil, aboveground biomass, belowground biomass) from the four marshes were combined and mean OC storage values (kg C m^{-2}) were calculated for each soil unit across the marsh zones. The mean OC storage values were combined with the total area of each marsh zone (Table 1) to estimate the OC stocks for the aboveground biomass, belowground biomass, the saltmarsh soil and the soil to a depth of 1 m. To estimate OC accumulation across Scotland's saltmarshes the mean OCAR (g C m^{-2} yr^{-1}) were calculated from the study sites for the different marsh zones. The mean OCAR were combined with the areal extent of the marsh zones across all saltmarshes to estimate the quantity of OC accumulating in Scottish saltmarshes. In both calculations an error of ±5% was applied to the areal extent of the marsh zones to account for changes since the survey occurred. All calculations were carried out within a MCMC framework. However, it should be noted that these OC stocks and accumulation rates are first-order estimates based upon limited data and appropriate caution should be applied in their future use.

4. Results and interpretation

4.1. Saltmarsh OC stocks

4.1.1. Aboveground and belowground biomass OC

The aboveground biomass harvested from the saltmarshes encompassed the main vegetation communities found in the pioneer (*SM8*), low-mid (*SM13*) and high (*SM16, SM28*) marsh zones at the four sites (Supp. Fig. 1) and are characteristic of the vegetation composition of Scotland's saltmarshes (Burd, 1989; Adam, 1978; Haynes, 2016). Across the three marsh zones, the difference in vegetation communities' results in the aboveground biomass increasing from 0.26 ± 0.09 kg m^{-2} in the pioneer marsh to 0.71 ± 0.18 kg m^{-2} in the high marsh (Supp. Table 3). The OC content of the aboveground biomass within the different marsh zones ranges between 35.95 ± 0.07% OC in the pioneer marsh to 39.05 ± 3.67% OC in the high marsh (Supp. Table 3).

Consequently, the C storage data for the aboveground biomass reflects the differences observed in biomass (Fig. 2A), with values of 0.09 ± 0.03 kg C m^{-2}, 0.15 ± 0.09 kg C m^{-2}, 0.25 ± 0.06 kg C m^{-2} being measured from the pioneer, low-mid and high marsh zones respectively (Fig. 2B). The aboveground biomass across the four sites holds a total of 2670 tonnes of OC with the largest quantities held at Wigtown (1542 ± 310 tonnes OC) and Morrich More (944 ± 220 tonnes OC). At Dornoch Point (99 ± 20 tonnes OC) and Skinflats (86 ± 19 tonnes OC) the aboveground OC stocks are significantly less due to the reduced spatial extent of these marshes (Table 1). Across the four saltmarshes, the high marsh consistently holds a greater proportion of the aboveground OC with the high marsh at Dornoch Point, Morrich More, Skinflats and Wigtown holding 74%, 64%, 72% and 77% of the aboveground OC stock, respectively. Full aboveground OC stocks are detailed in Supplementary Table 4.

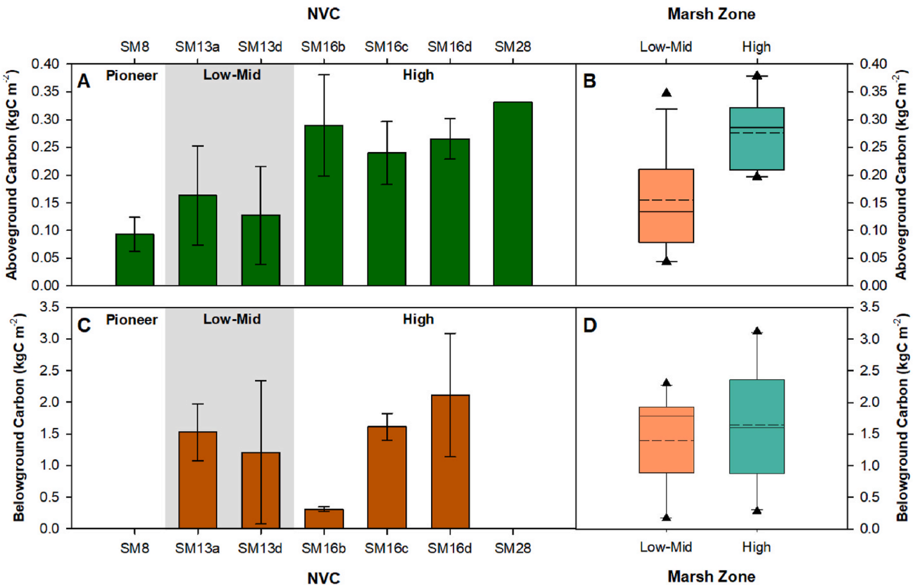

Fig. 2. Aboveground and belowground carbon. (**A**) Aboveground carbon (kg C m^{-2}) for different saltmarsh vegetation communities (NVC; Rodwell, 2000). (**B**) Aboveground carbon values for low-mid and high saltmarsh zones across Scotland. (**C**) Belowground carbon for different vegetation communities, no samples were collected from SM8 or SM28. (**D**) Belowground carbon values for low-mid and high saltmarshes across Scotland. The solid line indicates the median and the dashed line indicates the mean value. The whiskers indicate 1.5 times the range of between the first and third quantile for both extremes; the points represent the 5th and 95th percentile values. Vegetation composition associated with each NVC can be found in Supplementary Table 1. Full breakdown of aboveground and belowground biomass data can be found in Supplementary Table 3.

Belowground biomass samples were collected from the two most common saltmarsh vegetation classes in Scotland (*SM13* and *SM16*) which represent the low-mid and high marsh zones (Fig. 2C). Across the marsh zones the difference in root biomass is negligible with 5.05 ± 2.96 kg m^{-2} measured in the low-mid marsh and 5.58 ± 2.01 kg m^{-2} in the high marsh (Sup. Table 3). Similarly, the belowground OC of the vegetation classes differ little with the exception of SM16b (Fig. 2C). The data for NVC class SM16b was calculated from samples collected from the high marsh at Wigtown (Fig. 1E), this section of marsh is grazed by cattle which potentially explains the lower root OC density values when compared to other high marsh samples (Harvey et al., 2019; Graversen et al., 2022). The high marsh zone is estimated to hold 1.65 ± 0.94 kg C

Table 2
Saltmarsh soil variables used in the calculation of soil OC stocks for each of the four saltmarshes broken down into marsh zone (low-mid and high) and soil unit (fibrous and humified peat, transitional and basal units). Further details on the distribution and relationship between dry bulk density and OC content can be found in Supplementary Figs. 2–5.

Saltmarsh	Marsh Zone	Soil Unit	Area (km^2)	Soil Unit Thickness (cm)	Dry Bulk Density (g cm^{-3})	OC (%)
Dornoch Point	Low-Mid	Fibrous Peat	0.15	14.0 ± 9.6	0.60 ± 0.23	14.75 ± 5.69
		Transitional		11.7 ± 8.6	1.60 ± 0.61	3.49 ± 1.54
		Basal Unit		–	2.09 ± 0.38	0.69 ± 0.12
	High	Fibrous Peat	0.26	9.0 ± 5.1	0.38 ± 0.15	21.38 ± 7.45
		Transitional		13.3 ± 5.6	1.56 ± 0.56	2.69 ± 1.93
		Basal Unit		–	2.08 ± 0.26	0.51 ± 0.14
Morrich More	Low-Mid	Fibrous Peat	2.08	10.7 ± 1.2	0.47 ± 0.15	22.46 ± 9.87
		Transitional		7.5 ± 0.7	1.93 ± 0.38	1.52 ± 0.10
		Basal Unit		–	2.40 ± 0.21	0.45 ± 0.06
	High	Fibrous Peat	2.17	11.5 ± 3.0	0.44 ± 0.07	30.04 ± 4.66
		Humified Peat		6.3 ± 3.2	0.94 ± 0.23	9.11 ± 4.37
		Transitional		20.8 ± 3.6	1.93 ± 0.36	2.00 ± 1.15
		Basal Unit		–	2.26 ± 0.30	0.50 ± 0.18
Skinflats	Low-Mid	Fibrous Peat	0.15	14.2 ± 20.6	0.29 ± 0.13	11.13 ± 3.24
		Humified Peat		10.4 ± 9.8	0.33 ± 0.10	8.99 ± 2.56
		Basal Unit		–	0.44 ± 0.17	6.96 ± 2.01
	High	Fibrous Peat	0.22	10.48 ± 5.4	0.25 ± 0.12	19.20 ± 4.84
		Humified Peat		0.62 ± 2.3	0.30 ± 0.14	16.78 ± 4.46
		Transitional		3.4 ± 10.0	0.62 ± 0.21	11.74 ± 4.76
		Basal Unit		–	0.90 ± 0.24	9.15 ± 3.06
Wigtown	Low-Mid	Fibrous Peat	1.94	10.2 ± 4.1	0.39 ± 0.13	5.19 ± 1.95
		Humified Peat		16.3 ± 6.5	0.62 ± 0.43	3.93 ± 1.23
		Transitional		19.6 ± 7.2	0.69 ± 0.16	2.73 ± 1.00
		Basal Unit		–	0.84 ± 0.15	1.65 ± 0.43
	High	Fibrous Peat	4.22	10.0 ± 5.6	0.41 ± 0.14	6.87 ± 0.98
		Humified Peat		24.5 ± 19.1	0.41 ± 0.16	4.11 ± 2.22
		Transitional		19.5 ± 7.7	0.53 ± 0.17	3.68 ± 2.30
		Basal Unit		–	0.73 ± 0.22	2.25 ± 0.86
Mean	Low-Mid	Fibrous Peat	–	10.7 ± 5.4	0.38 ± 0.15	8.06 ± 5.50
		Humified Peat		16.2 ± 6.3	0.47 ± 0.34	6.53 ± 3.25
		Transitional		13.9 ± 8.5	0.76 ± 0.33	2.72 ± 1.14
		Basal Unit		–	0.86 ± 0.60	3.77 ± 3.10
	High	Fibrous Peat	–	9.9 ± 4.5	0.34 ± 0.15	20.68 ± 7.80
		Humified Peat		10.9 ± 13.0	0.38 ± 0.24	13.49 ± 6.64
		Transitional		16.2 ± 8.4	1.02 ± 0.64	6.00 ± 5.30
		Basal Unit		–	1.24 ± 0.63	5.92 ± 4.70

m^{-2} while the low-mid marsh zone holds 1.33 ± 0.77 kg C m^{-2} within the belowground biomass (Fig. 2D). The belowground biomass at the four sites hold a total of 17,945 tonnes of OC with the largest quantities held at Wigtown (9991 ± 3786 tonnes OC) and Morrich More (6700 ± 2348 tonnes OC). The belowground biomass OC stock at Dornoch Point (661 ± 250 tonnes OC) and Skinflats (593 ± 219 tonnes OC) is dramatically lower due to the difference in areal extent of these marshes (Table 1). The belowground biomass within the high marsh zone consistently holds a greater proportion of the OC, with the high marsh at Dornoch Point, Morrich More, Skinflats and Wigtown holding 68%, 56%, 64% and 73% of the belowground biomass OC stock respectively. As the aboveground and belowground biomass is intrinsically linked, the proportion of OC stored in the marsh zones is driven by the same factors (i.e., vegetation communities and areal extent). Full belowground OC stocks are detailed in Supplementary Table 5.

4.1.2. Soil OC stock

Examination of the soil cores (Section 3.2) highlights that the thickness of the different soil units is highly variable both internally between marsh zones and more widely across the different marshes (Table 2). The dry bulk density values of the basal and transitional soil units are largely dependent on the estuarine setting. The highest dry bulk densities are observed in the basal units of Dornoch Point (2.09 ± 0.38 g cm^{-3}) and Morrich More (2.40 ± 0.21 g cm^{-3}) both these marshes sit within estuaries dominated by sand (Smeaton et al., 2021a, 2022a). In comparison the muddy basal units of Skinflats and Wigtown are characterised by bulk density values < 1 g cm^{-3} (Table 2). As the saltmarsh soil profiles transition towards the organic rich saltmarsh substrates (humified and fibrous peat units), the dry bulk density drops dramatically and becomes far more uniform across all saltmarshes (Table 2).

The lowest OC contents are observed in the sandy systems with 0.51 ± 0.14% OC and 0.50 ± 0.18% OC measured in the basal units of Dornoch Point and Morrich More respectively. In comparison, the finer muddy sediments found at Skinflats (9.15 ± 3.06% OC) and Wigtown (2.25 ± 0.86% OC) exhibit much higher values (Table .2). This is expected as grain size can be strongly linked and used as a predictor for OC content in saltmarsh soils (Mueller et al., 2019a; 2019b; Kelleway et al., 2016) with coarser (e.g. sand) substrates holding less OC than finer material (e.g. muds). The basal soil unit at Skinflats has elevated OC values in both low-mid (6.96 ± 2.01% OC) and high marsh (9.15 ± 3.06% OC) in comparison to other sites (Table 2). Unlike the other marshes Skinflats is situated in the upper reaches of an estuary fed by a major river (River Forth), therefore the saltmarsh and mudflat likely receives large quantities of allochthonous OC from the terrestrial environment (Bianchi, 2011) driving the high OC contents observed.

Unlike the dry bulk density values that are reasonably uniform across the humified and fibrous peat units, the OC content does vary, with fibrous peat values as low as 5.19 ± 1.95% OC being observed in the low-mid marsh at Wigtown and as high as 30.04 ± 4.66% OC in the high marsh at Morrich More. The four marshes have similar vegetation composition (Haynes, 2016; Sup Fig. 1) and Morrich More is the most northerly of all sites (Fig. 1), therefore it is unlikely that primary production within the marsh is the main driver of the differences observed. The position of the marsh within the estuary and sediment supply and source are more likely driving the development of the marshes and the OC content of the soils (Ladd et al., 2019). For example, the marshes in the Solway Firth are known to be expanding both laterally (8800 ± 11, 700 m^2 yr^{-1}) and vertically (15.41 ± 14.53 mm yr^{-1}) (Marshall, 1962; Ladd et al., 2019), yet the OC content of the soil at Wigtown is the lowest of the surveyed sites (Table 2). Wigtown marsh is situated on an open coastline (Fig. 1E) therefore much of the OC rich allochthonous material that potentially could be trapped in the saltmarsh is likely being flushed out to the sediments in the Solway Firth, which potential results in the low OC values.

Both the dry bulk density values and OC content measured across the four saltmarshes are consistent with other studies of saltmarsh soil in Scotland (Marley et al., 2019; Porter et al., 2020; Smeaton et al., 2020, 2022b; Austin et al., 2021, 2022; Ladd et al., 2022).

Morrich More holds the greatest quantity of OC within its soil with 84,991 ± 22,068 tonnes OC held within the saltmarsh soils and an estimated 135,324 ± 22,994 tonnes OC stored within the top 1 m of soil (Fig. 3). In comparison, the saltmarsh soil at Wigtown hold 66,070 ± 27,206 tonnes of OC and a total of 145,401 ± 42,726 tonnes OC within the top 1 m of soil. Wigtown marsh has an aerial extent 36% greater than Morrich More yet stores 22% less OC within the saltmarsh soils. The disparity in OC stocks originates from the OC content of the peat-like soil units, at Wigtown these units have an OC content of 6.87 ± 0.98% OC, in contrast at Morrich More values of 30.04 ± 4.66% OC (Table 2) are observed. In comparison, Dornoch Point and Skinflats are significantly smaller and hold less OC within their soils. At Dornoch Point 7277 ± 3167 tonnes OC is held within the saltmarsh soil units and a total of 13,408 ± 3167 tonnes OC held within the top 1 m of soil (Fig. 3B). Finally, Skinflats the smallest of the four marshes (Table 1) holds 3983 ± 2037 tonnes OC while the 1 m soil OC stock is estimated to be 21,274 ± 7017 tonnes OC. Across the four marshes with the majority of the OC is held with the high marsh zone (Fig. 3C).

Across the four saltmarshes, the quantity of OC held within the top 1 m of soil versus the amount of OC in the saltmarsh soils differ. At Dornoch Point, Morrich More and Wigtown the saltmarsh soils contain 54%, 63% and 45% of the total OC stored to a depth of 1 m respectively (Fig. 3B). Skinflats differs from this pattern with only 18% of the total 1 m OC stock held within the saltmarsh soil (Fig. 3B). The likely driver for this is the elevated OC content of the basal sediments (Table 2) which results in a significant difference in the 1 m soil OC storage value between Skinflats (59.6 ± 19.3 kg C m^{-2}) and the other saltmarshes (Fig. 3D). Howard et al. (2014) and the Intergovernmental Panel on Climate Change (Kennedy et al., 2014) recommend that a standard sampling depth of 1 m should be used when assessing the blue carbon value of coastal ecosystem and to allow for GHG inventories. Whilst a standardised depth allows for easier comparison between ecosystems (Duarte et al., 2013) our results show that marsh deposits do not always reach a depth of 1 m and in Scotland that only half the OC held within the top 1 m is found in the saltmarsh soils with the pre-saltmarsh (mudflat or sandflat) environments holding a significant proportion of the OC (Fig. 3). Including these pre-saltmarsh environments in the saltmarsh soil OC stock assessment risks inflating the saltmarsh soil OC stock (Shi and Lamb, 1991). We therefore recommend that saltmarsh OC stocks should be reported to the true depth of the saltmarsh soil and to the recommended 1 m depth for international reporting standards. Full breakdown of the saltmarsh soil OC stocks can be located in Supplementary Tables 6 and 7

4.2. OC provenance

The OC held within the saltmarsh soils (fibrous peat, humified peat and transitional soil units) has similar isotopic and bulk elemental values to material produced in the terrestrial environment or the saltmarsh itself (Fig. 4; Supp. Fig.9). The isotopic and bulk elemental values observed in the low-mid marsh soil units are more variable indicating a greater mix of OC sources (Fig. 4). Reflecting that this zone has a greater connection to the marine environment due to the zone's position lower in the tidal frame (Balke et al., 2016).

The quantity of OC derived from terrestrial and in situ sources vary between soil units reflecting changes in OC sources and preservation conditions as the saltmarsh develops. As indicated in Fig. 4 the OC in the basal unit has the greatest mix of material from different OC sources with values ranging between 50.0 ± 12.9% (Dornoch Point low-mid marsh) and 89.1 ± 2.1% (Morrich More high marsh) of the OC originating from terrestrial and in situ sources (Supp. Table 10). The transitional soil unit reflects the shift from intertidal flat towards saltmarsh with percentage of the total OC derived from terrestrial/in situ sources

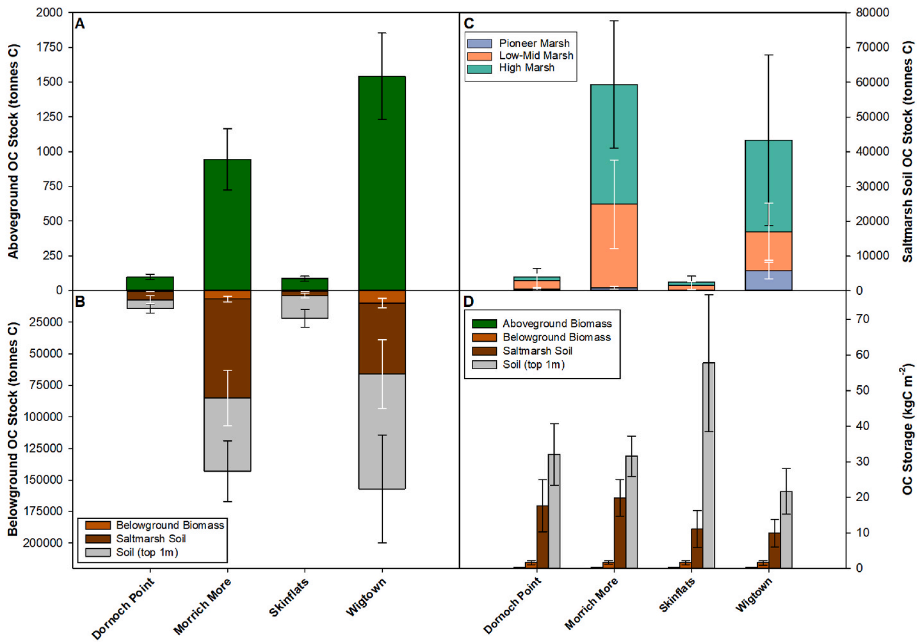

Fig. 3. Organic carbon stocks for the four study sites. (**A**) Aboveground C stock (tonnes C). (**B**) Belowground OC stock (tonnes OC) broken down into belowground biomass, saltmarsh soil (fibrous peat, humified peat and transitional soil units) and the 1 m soil (saltmarsh and basal soil units). (**C**) OC stock (tonnes OC) of the saltmarsh soil split into saltmarsh zone. (**D**) OC storage (kg C m^{-2}) of the OC pools (e.g., aboveground biomass, belowground biomass, saltmarsh soil and the soil down to a depth of 1 m) at each site. Full breakdown of data can be found in Supplementary Table 8.

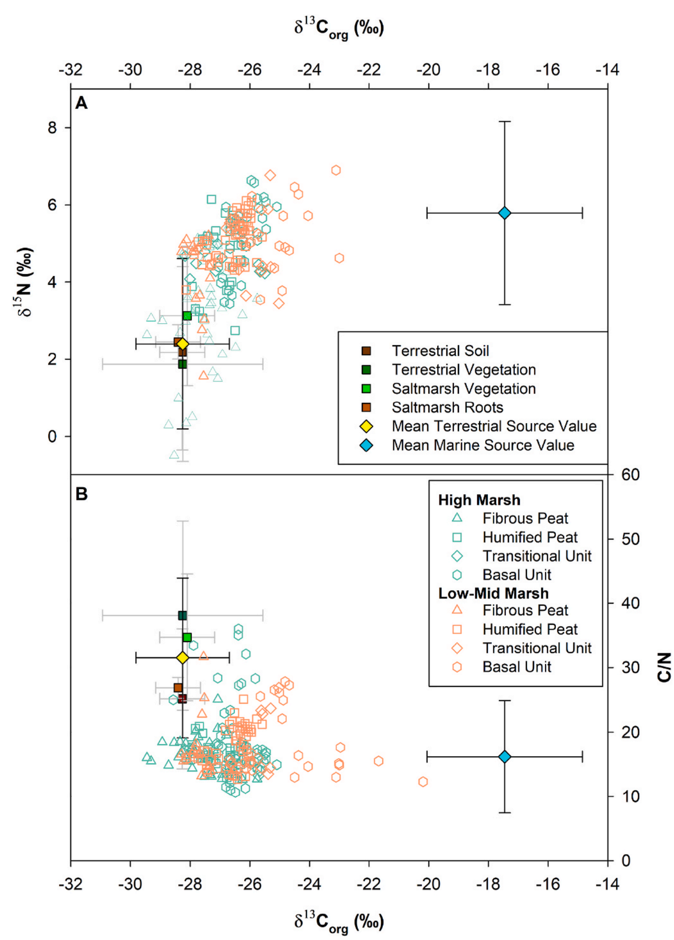

Fig. 4. Cross plots (**A**) $\delta^{13}C_{org}$ versus $\delta^{15}N$ and (**B**) $\delta^{13}C_{org}$ versus C/N for soil samples from across the four saltmarshes. Terrestrial, marine and saltmarsh source values derived from samples collected from across Scotland can be found in Supplementary Table 9.

increasing to between 77.0 ± 4.3% within the low-mid marsh at Dornoch Point and 93.2 ± 4.5% in the high marsh at Skinflats. The fibrous and humified peat units are dominated by terrestrial/in situ OC (Supp. Table 10). Within these soil units, differences between the low-mid and high marshes becomes evident, nowhere more so than at Skinflats where 98.9 ± 5.8% of the OC in the high marsh originates from terrestrial/in situ sources versus 80.6 ± 9.0% in the low-mid marsh.

Though we cannot currently differentiate between terrestrial OC input and in situ OC production it is reasonable to assume that saltmarshes such as Skinflats (Fig. 1D) situated on major rivers (i.e., River Forth) may receive substantial quantities of terrestrial OC. Analyses of the saltmarsh soil of the upper marsh of Skinflats suggests >90% of the OC originates from terrestrial/in situ sources (Supp. Table 10). Further, the basal unit is enriched in OC (Table 2) of which 82.3 ± 5.0% is estimated to be from terrestrial/in-situ sources (Supp. Table 10). While qualitative, the combination of factors suggest that much of this OC originates from the terrestrial environment, as in situ OC production from the saltmarsh vegetation would not directly contribute to the OC rich basal unit at this site. Additionally, it should be noted that marine derived OC is more labile (Smeaton and Austin, 2022) and will be preferentially degraded (Arndt et al., 2013; Larowe et al., 2020; Middelburg, 2018) resulting in more depleted $\delta^{13}C_{org}$ values which potentially leads to an over estimation of terrestrial/in situ OC input to the saltmarshes.

4.3. Saltmarsh OC accumulation

The three cores collected from the saltmarshes within the Dornoch Firth have similar soil profiles with organic rich peat units overlying sandy substrates (Fig. 5). The $^{210}Pb_{unsupp}$ activities indicate increased sedimentation in recent times across all three cores albeit at different rates of accumulation (Fig. 5; Supp. Fig. 5-7). The ^{210}Pb and ^{137}Cs chronologies from core C24 and C45 indicate that the material at 11.5 cm started to form in the year 1872 ± 24 and 1830 ± 22 with the high marsh increasing on average by 0.09 ± 0.02 cm yr^{-1} at Dornoch Point and 0.06 ± 0.03 cm yr^{-1} at Morrich More (Sup. Figs. 11–13; Supp. Table 11). The similarity in LSR results in comparable OCARs across these sites, with average rates of 31.6 ± 14.6 g C m^{-2} yr^{-1} and 29.1 ± 16.1 g C m^{-2} yr^{-1} accumulating in the high marsh of Dornoch Point and Morrich More respectively. The mean LSR in the low marsh at Dornoch Point (C32) is 0.18 ± 0.01 cm yr^{-1}, approximately double that measured

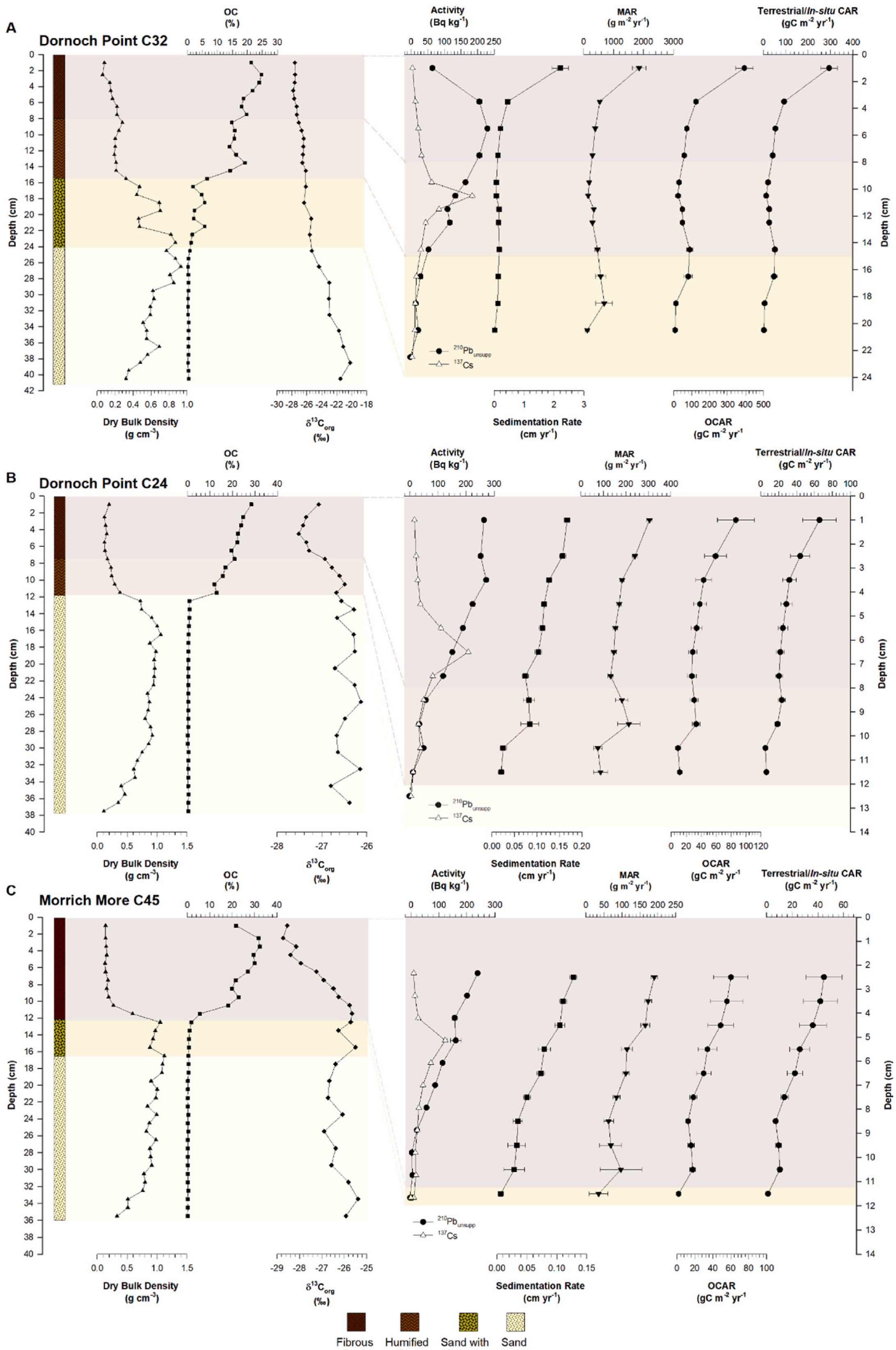

Fig. 5. Downcore plots of data produced from the wide diameter cores from (**A**) Dornoch Point C32; (**B**) Dornoch Point C24, and (**C**) Morrich More C45. The plots include soil profiles, dry bulk density, OC (%), $\delta^{13}C_{org}$ (‰) alongside radiometric dating (^{210}Pb and ^{137}Cs), sedimentation rates (cm yr^{-1}), mass accumulation rates (MAR) (g m^{-2} yr^{-1}), organic carbon accumulation rates (OCAR) (g C m^{-2} yr^{-1}) and terrestrial/in situ carbon accumulation rate (CAR) (g C m^{-2} yr^{-1}). Full evaluation of the radiometric dating can be found in Supplementary Figs. 5–7.

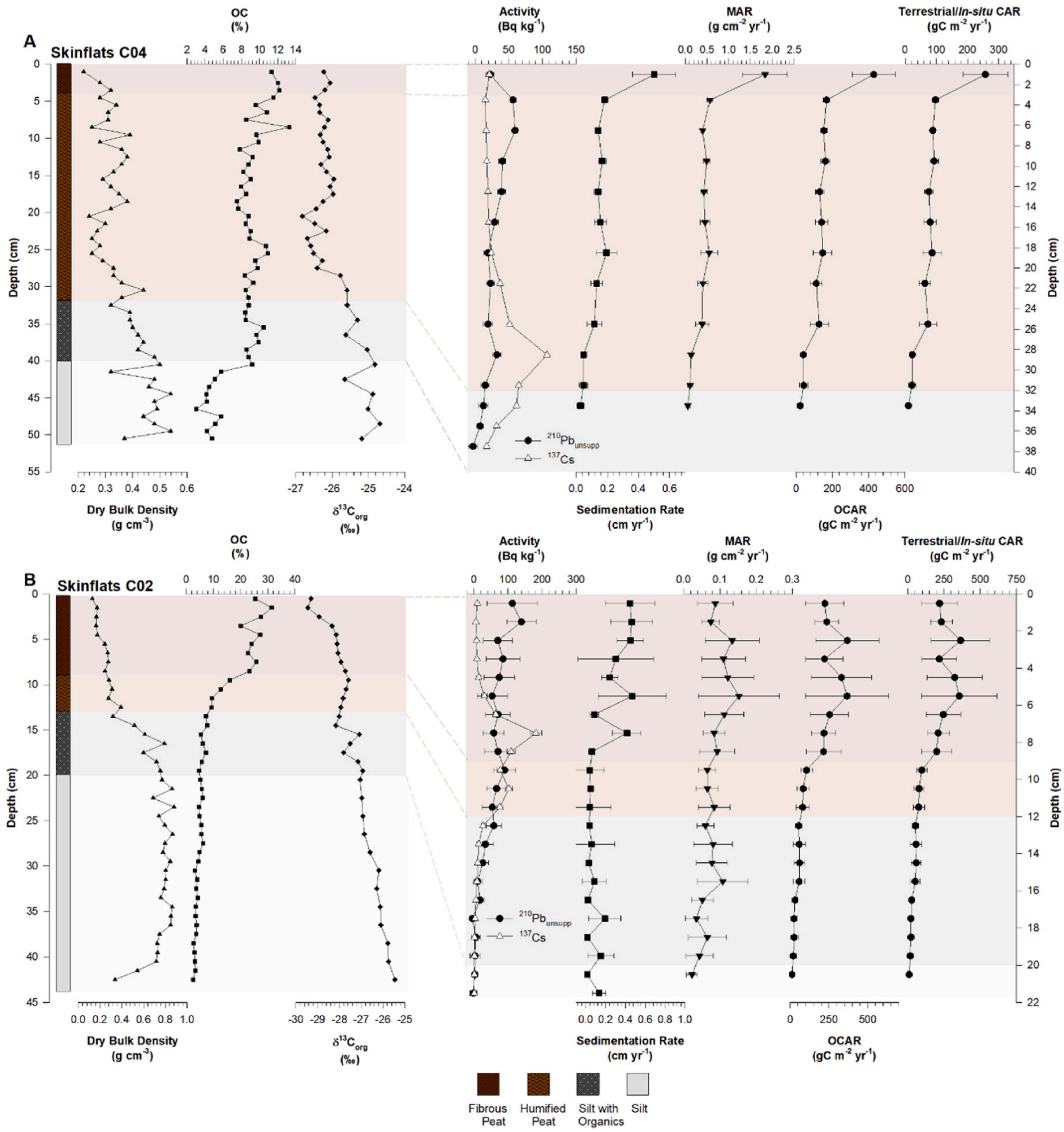

Fig. 6. Downcore plots of data produced from the wide diameter cores from Skinflats (**A**) Core C04 and (**B**) Core C02. The plots include soil profiles, dry bulk density, OC (%), $\delta^{13}C_{org}$ (‰) alongside radiometric dating (^{210}Pb and ^{137}Cs), sedimentation rates (cm yr^{-1}), mass accumulation rates (MAR) (g m^{-2} yr^{-1}), organic carbon accumulation rates (OCAR) (g C m^{-2} yr^{-1}) and terrestrial/in situ carbon accumulation rate (CAR) (g C m^{-2} yr^{-1}). Full evaluation of the radiometric dating can be found in Supplementary Figs. 8–9.

in the adjacent high marsh in-turn resulting in an average OCAR of 61.9 ± 33.0 g C m^{-2} yr^{-1} with 69% of this sourced from terrestrial environment and/or in situ OC production.

The soil profiles of cores C04 (low-mid marsh) and C02 (high marsh) from Skinflats differ from those of the Dornoch Firth with fine silt replacing the sand (Fig. 6). The $^{210}Pb_{unsupp}$ activities in the surface soils are lower than underlying samples, suggesting increased sedimentation in recent times across the cores (Fig. 6; Supp. Figs. 8-9). The ^{210}Pb and ^{137}Cs chronologies from core C04 indicate that the material at 33 cm started to form in the year 1907 ± 20 and 1830 ± 22 (Supp. Fig. 14). The saltmarsh soil units are estimated to begun forming between 1931 and 1956 (28–31 cm) coinciding with the construction of the nearby Kincardine Bridge (Fig. 1D) in 1931, which likely drives the rapid expansion (0.46 cm yr^{-1}) of the low-mid marsh. Within the high marsh, soil began to accumulate in the year 1909 ± 20 (21.5 cm) at a rate of 0.24 ± 0.11 cm yr^{-1} (Supp Fig.15; Supp. Table 11). Calculated OCARs at Skinflats are significantly higher than those within the Dornoch Firth with average rates of 142.5 ± 118.9 g C m^{-2} yr^{-1} and 151.9 ± 19.3 g C m^{-2} yr^{-1} in the low-mid and high marsh, respectively, with 98% of the OC being trapped in the high marsh and originating from terrestrial and in situ sources.

In the low-mid marsh (C03) the peaks in ^{137}Cs and ^{241}Am are found in the basal unit, while in the high marsh they are observed in the humidified peat unit. The location of the peaks indicate that the high marsh formed before the mid-1970s, and low-mid marsh started to form post mid-1970s (Fig. 7). From this single chronological marker, it is calculated that the low-mid and high marsh are vertically accreting at a rate of 0.74 cm yr^{-1} and 0.43 cm yr^{-1} respectively. The low-mid marsh is estimated to accumulate 160.8 ± 61.3 g C m^{-2} yr^{-1}, with the high marsh gaining a further 235.4 ± 62.7 g C m^{-2} yr^{-1} (Fig. 8). Calculating

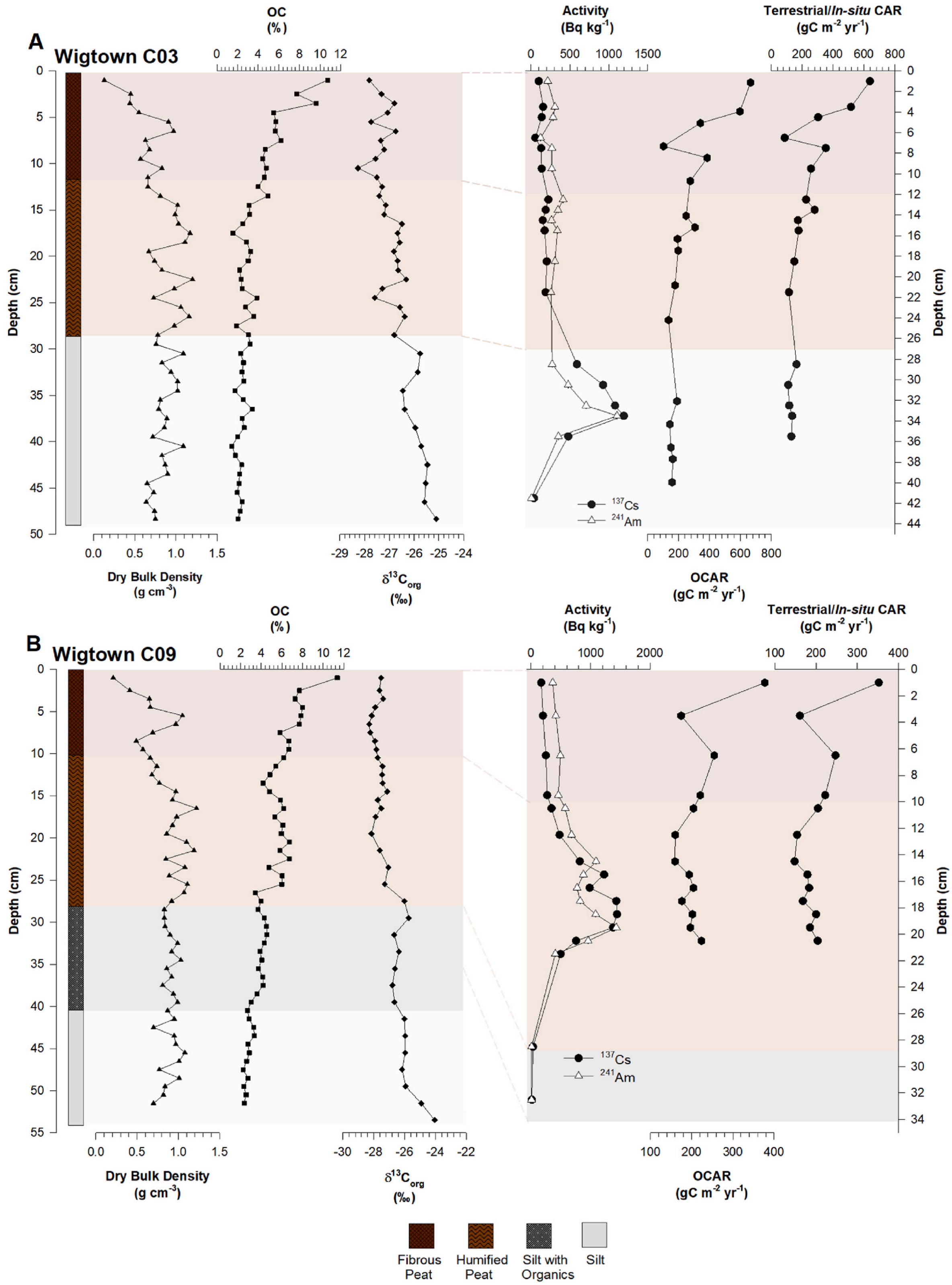

Fig. 7. Downcore plots of data produced from the wide diameter cores from Wigtown (**A**) Core C03 and (**B**) Core C09. The plots include soil profiles, dry bulk density, OC (%), $\delta^{13}C_{org}$ (‰) alongside radiometric dating (^{137}Cs and ^{241}Am), sedimentation rates (cm yr^{-1}), mass accumulation rates (MAR) (g m^{-2} yr^{-1}), organic carbon accumulation rates (OCAR) (g C m^{-2} yr^{-1}) and terrestrial/in situ carbon accumulation rate (CAR) (g C m^{-2} yr^{-1}). Full evaluation of the radiometric dating can be found in Supplementary Fig. 10.

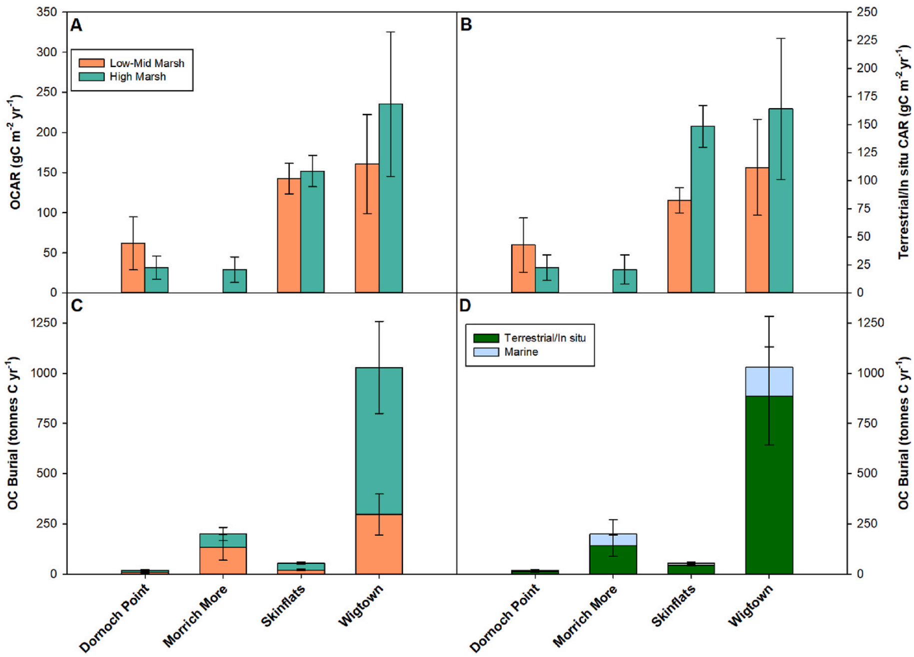

Fig. 8. Organic carbon accumulation rates (OCAR) for the four study sites. (**A**) Organic carbon accumulation rates (g C m^{-2} yr^{-1}) for the low-mid and high marsh of the four sites. (**B**) Terrestrial/in situ organic carbon accumulation rates (g C m^{-2} yr^{-1}) for the low-mid and high marsh of the four sites. (**C**) Annual organic carbon accumulation (tonnes OC yr^{-1}) split into saltmarsh zone. (**D**) Breakdown of the sources (terrestrial/in situ vs marine) of the OC buried annually (tonnes C yr^{-1}).

accumulation rates from a single chronological marker is problematic and likely leads to the higher than average OCARs calculated in these cores. This issue is most acute in C03 where the ^{137}Cs and ^{241}Am spike is found in the basal unit rather than the saltmarsh soils, it is therefore impossible to separate the rate at which the mudflat accreted opposed to the overlying saltmarsh soils (Fig. 7A). For this reason, when upscaling to national estimates of OC accumulation caution should be applied in using the OCARs from this site in any calculation.

Wigtown the largest of the saltmarshes studied, is estimated to accumulate 1029.9 ± 229.3 tonnes OC yr^{-1}, with the high marsh being responsible for 71% of this total (Fig. 8C). In comparison, Skinflats the smallest marsh in the study accumulates 54.8 tonnes OC yr^{-1}, with the high marsh trapping 64% of the OC (Supp. Table 13). The Dornoch Point and Morrich More marshes are estimated to accumulate 18.0 ± 5.8 tonnes OC yr^{-1} and 199.5 ± 72.2 tonnes OC yr^{-1}, respectively. Unlike the muddy systems (i.e., Wigtown and Skinflats) the low-mid marsh plays a larger role, with 54% and 67% of the OC being trapped in this zone at Dornoch Point and Morrich More respectively (Fig. 8D). Across all four saltmarshes, the majority of the OC that is being buried is sourced from terrestrial and in situ sources (Fig. 8E) with marine derived OC playing only a minor role (Supp. Table 13).

4.4. Scotland's saltmarsh carbon resource

It is estimated that 1.15 ± 0.21 Mt of OC is stored within Scottish saltmarshes with 99.9% of this held within the soils, with the above and belowground biomass holding 0.013 ± 0.002 Mt and 0.087 ± 0.033 Mt of OC respectively (Supp. Table 7). Currently, there are no other OC stock assessments that take into consideration the full depth of the saltmarsh soil in the UK. The latest, Scottish saltmarsh OC stock estimates only consider the surficial soils with Smeaton et al. (2022c) estimating a total of 0.37 ± 0.09 Mt of OC in the top 10 cm, while Austin et al., (2021) calculated that 0.52 ± 0.16 Mt of OC is held within the top 15 cm. Beaumont et al. (2014) estimated a total OC stock of 0.57 Mt of OC, with the soil holding 0.45 Mt OC, and the vegetation and roots holding 0.017 Mt and 0.058 Mt of OC respectively. The similarity in aboveground and root OC stocks between the studies provide confidence that these components of the OC stocks are being accurately accounted for.

A recent assessment of Scottish seagrass estimated that between 1.49 and 10.57 kg C m^{-2} is stored in underlying soils, which results in a national OC stock to of 0.088 Mt within the top 50 cm of the soil (Potouroglou et al., 2021). Saltmarsh habitats in Scotland occupy an area 73% larger than that of seagrass and hold 92% more OC owing to them in places storing double the quantity of OC per area unit (Table 3).

Annual OC accumulation was estimated for all Scottish saltmarsh by applying mean OCAR rates calculated for the low-mid (103.4 ± 18.4 g C m^{-2} yr^{-1}) and high (71.5 ± 9.3 g C m^{-2} yr^{-1}) zones from Dornoch Firth, Morrich More and Skinflats, the data from Wigtown was excluded from these calculations due to the issues discussed earlier. Using this approach, it is estimated that annually Scottish saltmarsh habitat accumulate 4385 ± 481 tonnes of OC of which 85% originates from terrestrial or in situ sources (Table 3). Globally, estimates of OC accumulation ranges between 18 and 1713 g C m^{-2} yr^{-1} with the global

Table 3
Summary of OC stocks, density and accumulation rates for the four saltmarshes and estimates for Scotland's 240 mapped saltmarshes.

	Saltmarsh OC Storage	Saltmarsh OC Stock	OCAR	OC Accumulation	Terrestrial/in situ OC Accumulation
Saltmarsh	kg C m^{-2}	Tonnes	(g C m^{-2} yr^{-1})	Tonnes C yr^{-1}	Tonnes C yr^{-1}
Dornoch Point	19.5 ± 7.4	8156 ± 3078	46.8 ± 23.8	18 ± 6	13 ± 4
Morrich More	21.6 ± 5.2	92,659 ± 22,251	29.1 ± 16.1	200 ± 72	141 ± 53
Skinflats	13.0 ± 5.3	4784.9 ± 1933	147.2 ± 19.2	55 ± 6	45 ± 5
Wigtown	11.6 ± 4.0	78,172.8 ± 26,757	198.1 ± 75.9	1030 ± 255	867 ± 244
Scotland Saltmarsh	**18.6 ± 3.9**	**1,048,193 ± 213,530**	-	**4385 ± 481**	**3719 ± 569**

average value of 244.7 ± 26 g C m^{-2} yr^{-1} (Chmura et al., 2003; Duarte et al., 2005, 2013; McLeod et al., 2011; Ouyang and Lee, 2014). In the NW Atlantic region, the mean OC accumulation value is 172.2 ± 18.1 g C m^{-2} yr^{-1} (Ouyang and Lee, 2014) which is higher than that observed in Scottish saltmarshes surveyed (Table 3). The range of OCAR observed in Scottish saltmarshes reflect both mud-rich and sand-rich estuarine systems (Fig. 1). Globally, mud rich marshes have been the focus of much of the drive to understand OC accumulation (Duarte et al., 2013) and when the OCARs from Skinflats (a mud rich system) are compared to these global and regional averages they are equivalent (Table 3). However, it is clear, that sandy saltmarshes, such as Dornoch Point and Morrich More with low OCARs are overlooked in current global data compilations (Chmura et al., 2003; Duarte et al., 2005, 2013; McLeod et al., 2011; Ouyang and Lee, 2014).

The national saltmarsh OC stock and accumulation estimates represent a first-order assessment using the available data. Caution should be applied as the limited number of sites ($n = 4$) investigated in this study makes it difficult to fully constrain the OC stocks and annual OC accumulation across Scotland's 240 saltmarshes.

5. Conclusion

This study provides the first full assessment of OC in Scottish saltmarshes with estimates of both OC stock and accumulation rates. As such, it is the first study of its kind in the UK. Accounting for the variability of soil profiles across Scottish saltmarshes has allowed first-order reporting of both the OC stored in the saltmarsh soils (1.15 ± 0.21 Mt OC) and the full OC stock down to 1 m depth (2.1 ± 0.34 Mt OC) which is a foundational step towards the inclusion of saltmarshes in national and international C reporting (Howard et al., 2014; Kennedy et al., 2014). Annually, these stores are supplemented by a further 4385 ± 481 tonnes of OC with the majority deriving from terrestrial/in situ sources. When compared to seagrass the only other vegetated intertidal system in Scotland with OC stock estimates it is now clear that saltmarsh represents the largest intertidal blue carbon resource of Scotland.

CRediT authorship contribution statement

Lucy C. Miller: Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Data curation. **Craig Smeaton:** Writing – review & editing, Writing – original draft, Visualization, Supervision, Project administration, Methodology, Investigation, Formal analysis, Conceptualization. **Handong Yang:** Writing – review & editing, Methodology, Formal analysis. **William E. N. Austin:** Writing – review & editing, Funding acquisition, Conceptualization.

Declaration of competing interest

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

Data availability

All data produced and used within the manuscript is archived with Marine Scotland Data (Miller et al., 2022a) and the NERC environmental Information Data Centre (Miller et al., 2022b). Links to all data are found in the manuscript.

Acknowledgments

This research was finically supported by the Scottish Blue Carbon Forum and the Natural Environment Research Council funded Carbon Storage in Intertidal Environments (C-SIDE) project (grant NE/R010846/1). We thank Bill Turrell (*Marine Scotland*) for providing comments on an early draft of this manuscript. Additionally, we thank Dr Heather Austin for assistenace with fieldwork and the collection of samples. Finally, we thank the three anonymous reviewers and the associate editor for providing useful comments that improved the manuscript.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecss.2023.108223.

References

Adam, P., 1978. Geographical variation in British saltmarsh vegetation. J. Ecol. 339–366. https://doi.org/10.2307/2259141.

Appleby, P.G., Oldfield, F., 1978. The calculation of lead-210 dates assuming a constant rate of supply of unsupported 210Pb to the sediment. Catena 5 (1), 1–8. https://doi.org/10.1016/S0341-8162(78)80002-2.

Appleby, P.G., Oldfield, F., 1992. Applications of lead-210 to sedimentation studies. In: Uranium-series Disequilibrium: Applications to Earth, Marine, and Environmental Sciences. 2.

Appleby, P.G., 2002. Chronostratigraphic techniques in recent sediments. In: Tracking Environmental Change Using Lake Sediments. Springer, Dordrecht, pp. 171–203.

Appleby, P.G., Nolan, P.J., Gifford, D.W., Godfrey, M.J., Oldfield, F.J.A.N., Anderson, N. J., Battarbee, R.W., 1986. 210Pb dating by low background gamma counting. Hydrobiologia 143 (1), 21–27. https://doi.org/10.1007/BF00026640.

Arias-Ortiz, A., Masqué, P., Garcia-Orellana, J., Serrano, O., Mazarrasa, I., Marbà, N., Lovelock, C.E., Lavery, P.S., Duarte, C.M., 2018. Reviews and syntheses: 210 Pb-derived sediment and carbon accumulation rates in vegetated coastal ecosystems–setting the record straight. Biogeosciences 15 (22), 6791–6818. https://doi.org/10.5194/bg-15-6791-2018.

Arndt, S., Jørgensen, B.B., LaRowe, D.E., Middelburg, J.J., Pancost, R.D., Regnier, P., 2013. Quantifying the degradation of organic matter in marine sediments: a review and synthesis. Earth Sci. Rev. 123, 53–86. https://doi.org/10.1016/j.earscirev.2013.02.008.

Athy, L.F., 1930. Density, porosity, and compaction of sedimentary rocks. AAPG Bull. 14 (1), 1–24. https://doi.org/10.1306/3D93289E-16B1-11D7-8645000102C1865D.

Austin, W., Smeaton, C., Riegel, S., Ruranska, P., Miller, L., 2021. Blue carbon stock in Scottish saltmarsh soils, 12, 13 Scottish Marine Freshwater Sci. 12 (13). https://doi.org/10.7489/12372-1. Marine Scotland.

Austin, W.E.N., Smeaton, C., Ruranska, P., Paterson, D.M., Skov, M.W., Ladd, C.J.T., McMahon, L., Havelock, G.M., Gehrels, R., Mills, R., Barlow, N.L.M., Burden, A., Jones, L., Garbutt, A., 2022. Carbon storage in UK intertidal environments. In: Humphreys, J., Little, S. (Eds.), *Challenges In Estuarine and Coastal Science: Estuarine* and Coastal Sciences Association 50th *Anniversary Volume*. Pelagic Publishing, Exeter. https://doi.org/10.53061/STPP2268.

Balke, T., Stock, M., Jensen, K., Bouma, T.J., Kleyer, M., 2016. A global analysis of the seaward salt marsh extent: the importance of tidal range. Water Resour. Res. 52 (5), 3775–3786. https://doi.org/10.1002/2015WR018318.

Bao, R., McNichol, A.P., Hemingway, J.D., Gaylord, M.C.L., Eglinton, T.I., 2019. Influence of different acid treatments on the radiocarbon content spectrum of sedimentary organic matter determined by RPO/Accelerator Mass Spectrometry. Radiocarbon 61 (2), 395–413. https://doi.org/10.1017/RDC.2018.125.

Barlow, N.L., Long, A.J., Saher, M.H., Gehrels, W.R., Garnett, M.H., Scaife, R.G., 2014. Salt-marsh reconstructions of relative sea-level change in the North Atlantic during the last 2000 years. Quat. Sci. Rev. 99, 1–16. https://doi.org/10.1016/j.quascirev.2014.06.008.

Beaumont, N.J., Jones, L., Garbutt, A., Hansom, J.D., Toberman, M., 2014. The value of carbon sequestration and storage in coastal habitats. Estuar. Coast Shelf Sci. 137, 32–40. https://doi.org/10.1016/j.ecss.2013.11.022.

Bianchi, T.S., 2011. The role of terrestrially derived organic carbon in the coastal ocean: a changing paradigm and the priming effect. Proc. Natl. Acad. Sci. USA 108 (49), 19473–19481. https://doi.org/10.1073/pnas.1017982108. *Oceanographic Research Papers*, 37(1), pp.157–165.

Brown, D.R., Conrad, S., Akkerman, K., Fairfax, S., Fredericks, J., Hanrio, E., Sanders, L. M., Scott, E., Skillington, A., Tucker, J., van Santen, M.L., 2016. Seagrass, mangrove and saltmarsh sedimentary carbon stocks in an urban estuary; Coffs Harbour, Australia. Region. Studies in Marine Sci. 8, 1–6. https://doi.org/10.1016/j.rsma.2016.08.005.

Burd, F., 1989. Saltmarsh Survey of Great Britain: an Inventory of British Saltmarshes.

Burden, A., Smeaton, C., Angus, S., Garbutt, A., Jones, L., Lewis, H., Rees, S., 2020. 'Impacts of climate change on coastal habitats, relevant to the coastal and marine environment around the UK. MCCIP Sci. Rev. 2020, 228–255. https://doi.org/10.14465/2020.arc11.chb.

Burrows, M., Hughes, D., Austin, W.E.N., Smeaton, C., Hicks, N., Howe, J., Allen, C., Taylor, P., Vare, L., 2017. Assessment of Blue Carbon Resources in Scotland's Inshore Marine Protected Area Network: Commissioned Report No. 957. Vol. 957, Scottish Natural Heritage, Inverness. https://www.nature.scot/snh-commissioned-report-957-assessment-blue-carbon-resources-scotlands-inshore-marine-protected-area.

Burrows, M., Moore, P., Sugden, H., Fitzsimmons, C., Smeaton, C., Austin, W., Parker, R., Kröger, S., Powell, C., Gregory, L., Procter, W., Brook, T., 2021. Assessment of Carbon Capture and Storage in Natural Systems within the English North Sea

(Including within Marine Protected Areas), 03 edn. Blue Marine Foundation. https://www.bluemarinefoundation.com/2021/11/08/north-sea-blue-carbon-report/.
Chmura, G.L., Anisfeld, S.C., Cahoon, D.R., Lynch, J.C., 2003. Global carbon sequestration in tidal, saline wetland soils. Global Biogeochem. Cycles 17, 1111. https://doi.org/10.1029/2002GB001917.
Crosby, S.C., Sax, D.F., Palmer, M.E., Booth, H.S., Deegan, L.A., Bertness, M.D., Leslie, H.M., 2016. Salt marsh persistence is threatened by predicted sea-level rise. Estuar. Coast Shelf Sci. 181, 93–99. https://doi.org/10.1016/j.ecss.2016.08.018.
Dadey, K.A., Janecek, T., Klaus, A., 1992. Dry-bulk density: its use and determination. Proc. Ocean Drill. Progr. Sci. Results 126, 37.
DIN 19539, 2015. Investigation of Solids Temperature Dependent Differentiation of Total Carbon (TOC400, ROC, TIC900).
Duarte, C.M., Middelburg, J., Caraco, N., 2005. Major role of marine vegetation on the oceanic carbon cycle. Biogeosciences 2, 1–8. https://doi.org/10.5194/bg-2-1-2005.
Duarte, C.M., Dennison, W.C., Orth, R.J., Carruthers, T.J., 2008. The charisma of coastal ecosystems: addressing the imbalance. Estuar. Coast 31 (2), 233–238. https://doi.org/10.1007/s12237-008-9038-7.
Duarte, C.M., Losada, I.J., Hendriks, I.E., Mazarrasa, I., Marbà, N., 2013. The role of coastal plant communities for climate change mitigation and adaptation. Nat. Clim. Change 3 (11), 961–968. https://doi.org/10.1038/nclimate1970.
Fernandes, R., Millard, A.R., Brabec, M., Nadeau, M.J., Grootes, P., 2014. Food reconstruction using isotopic transferred signals (FRUITS): a Bayesian model for diet reconstruction. PLoS One 9 (2), e87436. https://doi.org/10.1371/journal.pone.0087436.
Ford, H., Garbutt, A., Ladd, C., Malarkey, J., Skov, M.W., 2016. Soil stabilization linked to plant diversity and environmental context in coastal wetlands. J. Veg. Sci. 27 (2), 259–268. https://doi.org/10.1111/jvs.12367.
Ford, H., Garbutt, A., Duggan-Edwards, M., Harvey, R., Ladd, C., Skov, M.W., 2019. Large-scale predictions of salt-marsh carbon stock based on simple observations of plant community and soil type. Biogeosciences 16 (2), 425–436. https://doi.org/10.5194/bg-16-425-2019.
Geraldi, N.R., Ortega, A., Serrano, O., Macreadie, P.I., Lovelock, C.E., Krause-Jensen, D., Kennedy, H., Lavery, P.S., Pace, M.L., Kaal, J., Duarte, C.M., 2019. Fingerprinting blue carbon: rationale and tools to determine the source of organic carbon in marine depositional environments. Front. Mar. Sci. 263. https://doi.org/10.3389/fmars.2019.00263.
Granek, E.F., Polasky, S., Kappel, C.V., Reed, D.J., Stoms, D.M., Koch, E.W., Kennedy, C.J., Cramer, L.A., Hacker, S.D., Barbier, E.B., Aswani, S., Ruckelshaus, M., Perillo, G.M.E., Silliman, B.R., Muthiga, N., Bael, D., Wolanski, E., 2010. Ecosystem services as a common language for coastal ecosystem-based management. Conserv. Biol. 24, 207–216. https://doi.org/10.1111/j.1523-1739.2009.01355.x.
Graversen, A.E.L., Banta, G.T., Masque, P., Krause-Jensen, D., 2022. Carbon sequestration is not inhibited by livestock grazing in Danish salt marshes. Limnol. Oceanogr. https://doi.org/10.1002/lno.12011.
Gray, J., Jones, S.R., Smith, A.D., 1995. Discharges to the environment from the Sellafield site, 1951-1992. J. Radiol. Prot. 15 (2), 99. https://doi.org/10.1088/0952-4746/15/2/001.
Gregg, R., Adams, J., Alonso, I., Crosher, I., Muto, P., Morecroft, M., 2021. Carbon Storage and Sequestration by Habitat: a Review of the Evidence. Natural England Research Report NERR094 (York: Natural England).
Greiner, J.T., Wilkinson, G.M., McGlathery, K.J., Emery, K.A., 2016. Sources of sediment carbon sequestered in restored seagrass meadows. Mar. Ecol. Prog. Ser. 551, 95–105. https://doi.org/10.3354/meps11722.
Harris, D., Horwáth, W.R., Van Kessel, C., 2001. Acid fumigation of soils to remove carbonates prior to total organic carbon or carbon-13 isotopic analysis. Soil Sci. Soc. Am. J. 65 (6), 1853–1856. https://doi.org/10.2136/sssaj2001.1853.
Harvey, M.M., Hansom, J.D., MacKenzie, A.B., 2007. Constraints on the use of anthropogenic radionuclide-derived chronologies for saltmarsh sediments. J. Environ. Radioact. 95 (2–3), 126–148. https://doi.org/10.1016/j.jenvrad.2007.02.005.
Harvey, R.J., Garbutt, A., Hawkins, S.J., Skov, M.W., 2019. No detectable broad-scale effect of livestock grazing on soil blue-carbon stock in salt marshes. Front. Ecol. Evol. 7, 151. https://doi.org/10.3389/fevo.2019.00151.
Haynes, T., 2016. Scottish Saltmarsh Survey National Report. Scottish Natural Heritage.
Horton, B.P., Shennan, I., Bradley, S.L., Cahill, N., Kirwan, M., Kopp, R.E., Shaw, T.A., 2018. Predicting marsh vulnerability to sea-level rise using Holocene relative sea-level data. Nat. Commun. 9, 2687. https://doi.org/10.1038/s41467-018-05080-0.
Howard, J., Hoyt, S., Isensee, K., Telszewski, M., Pidgeon, E., 2014. Coastal Blue Carbon: Methods for Assessing Carbon Stocks and Emissions Factors in Mangroves, Tidal Salt Marshes, and Seagrasses.
Kelleway, J.J., Saintilan, N., Macreadie, P.I., Ralph, P.J., 2016. Sedimentary factors are key predictors of carbon storage in SE Australian saltmarshes. Ecosystems 19 (5), 865–880. https://doi.org/10.1007/s10021-016-9972-3.
Kennedy, H., Alongi, D.M., Karim, A., 2014. Coastal wetlands. In: Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Jamsranjav, B., Kujuda, M., Troxler, T. (Eds.), Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands. IPCC, Geneva, p. 55.
Kennedy, H., Beggins, J., Duarte, C.M., Fourqurean, J.W., Holmer, M., Marbà, N., Middelburg, J.J., 2010. Seagrass sediments as a global carbon sink: isotopic constraints. Global Biogeochem. Cycles 24 (4). https://doi.org/10.1029/2010GB003848.
Kennedy, P., Kennedy, H., Papadimitriou, S., 2005. The effect of acidification on the determination of organic carbon, total nitrogen and their stable isotopic composition in algae and marine sediment. Rapid Communications in Mass Spectrometry. An Int. J. Devote. Rapid Dissemination Up-to-the-Minute Res. Mass Spectrom. 19 (8), 1063–1068. https://doi.org/10.1002/rcm.1889.
Krishnaswamy, S., Lal, D., Martin, J.M., Meybeck, M., 1971. Geochronology of lake sediments. Earth Planet Sci. Lett. 11 (1–5), 407–414. https://doi.org/10.1016/0012-821X(71)90202-0.
Ladd, C.J., 2021. Review on processes and management of saltmarshes across Great Britain. Proc. Geologists' Assoc. 132 (3), 269–283. https://doi.org/10.1016/j.pgeola.2021.02.005.
Ladd, C.J., Duggan-Edwards, M.F., Bouma, T.J., Pages, J.F., Skov, M.W., 2019. Sediment supply explains long–term and large-scale patterns in salt marsh lateral expansion and erosion. Geophys. Res. Lett. 46 (20), 11178–11187. https://doi.org/10.1029/2019GL083315.
Ladd, C.J.T., Duggan-Edwards, M.F., Skov, M.W., 2021. Saltmarsh resilience to periodic shifts in tidal channels. Front. Mar. Sci. 1524. https://doi.org/10.3389/fmars.2021.757715.
Ladd, C.J., Smeaton, C., Skov, M.W., Austin, W.E., 2022. Best practice for upscaling soil organic carbon stocks in salt marshes. Geoderma 428, 116188. https://doi.org/10.1016/j.geoderma.2022.116188.
LaRowe, D.E., Arndt, S., Bradley, J.A., Estes, E.R., Hoarfrost, A., Lang, S.Q., Lloyd, K.G., Mahmoudi, N., Orsi, W.D., Walter, S.S., Steen, A.D., 2020. The fate of organic carbon in marine sediments-New insights from recent data and analysis. Earth Sci. Rev. 204, 103146 https://doi.org/10.1016/j.earscirev.2020.103146.
Long, A.J., Barlow, N.L.M., Gehrels, W.R., Saher, M.H., Woodworth, P.L., Scaife, R.G., Brain, M.J., Cahill, N., 2014. Contrasting records of sea-level change in the eastern and western North Atlantic during the last 300 years. Earth Planet Sci. Lett. 388, 110–122. https://doi.org/10.1016/j.epsl.2013.11.012.
Lovelock, C.E., Adame, M.F., Bennion, V., Hayes, M., O'Mara, J., Reef, R., Santini, N.S., 2014. Contemporary rates of carbon sequestration through vertical accretion of sediments in mangrove forests and saltmarshes of South East Queensland, Australia. Estuar. Coast 37 (3), 763–771. https://doi.org/10.1007/s12237-013-9702-4.
Lunn, D., Spiegelhalter, D., Thomas, A., Best, N., 2009. The BUGS project: evolution, critique and future directions. Stat. Med. 28 (25), 3049–3067. https://doi.org/10.1002/sim.3680.
MacKenzie, A.B., Scott, R.D., 1993. Sellafield waste radionuclides in Irish Sea intertidal and salt marsh sediments. Environ. Geochem. Health 15 (2), 173–184. https://doi.org/10.1007/BF02627835.
Macreadie, P.I., Anton, A., Raven, J.A., Beaumont, N., Connolly, R.M., Friess, D.A., Kelleway, J.J., Kennedy, H., Kuwae, T., Lavery, P.S., Lovelock, C.E., Smale, D.A., Apostolaki, E.T., Atwood, T.B., Baldock, J., Bianchi, T.S., Chmura, G.L., Eyre, B.D., Fourqurean, J.W., Hall-Spencer, J.M., Huxham, M., Hendriks, I.E., Krause-Jensen, D., Laffoley, D., Luisetti, T., Marbà, N., Masque, P., McGlathery, K.J., Megonigal, J.P., Murdiyarso, D., Russell, B.D., Santos, R., Serrano, O., Silliman, B.R., Watanabe, K., Duarte, C.M., 2019. The future of Blue Carbon science. Nat. Commun. 10, 3998. https://doi.org/10.1038/s41467-019-11693-w.
Macreadie, P.I., Costa, M.D., Atwood, T.B., Friess, D.A., Kelleway, J.J., Kennedy, H., Lovelock, C.E., Serrano, O., Duarte, C.M., 2021. Blue carbon as a natural climate solution. Nat. Rev. Earth Environ. 2 (12), 826–839. https://doi.org/10.1038/s43017-021-00224-1.
Marley, A.R., Smeaton, C., Austin, W.E., 2019. An assessment of the tea bag index method as a proxy for organic matter decomposition in intertidal environments. J. Geophys. Res.: Biogeosciences 124 (10), 2991–3004. https://doi.org/10.3897/BDJ.5.e11764.
Marshall, J.R., 1962. The morphology of the upper Solway salt marshes. Scot. Geogr. Mag. 78 (2), 81–99. https://doi.org/10.1080/00369226208735859.
McLeod, E., et al., 2011. A blueprint for blue carbon: towards an improved understanding of the role of vegetated coastal habitats in sequestering CO_2. Front. Ecol. Environ. 9, 552–560. https://doi.org/10.1890/110004.
Mcowen, C.J., Weatherdon, L.V., Van Bochove, J.W., Sullivan, E., Blyth, S., Zockler, C., Stanwell-Smith, D., Kingston, N., Martin, C.S., Spalding, M., Fletcher, S., 2017. A global map of saltmarshes. Biodivers. Data J. (5).
Middelburg, J.J., 2018. Reviews and syntheses: to the bottom of carbon processing at the seafloor. Biogeosciences 15 (2), 413–427. https://doi.org/10.5194/bg-15-413-2018.
Miller, L.C., Smeaton, C., Yang, H., Austin, W.E.N., 2022a. Physical and Geochemical Properties of Scottish Saltmarsh Soils, *Marine Scotland Data*. https://doi.org/10.7489/12422-1.
Miller, L.C., Smeaton, C., Garbutt, A., Austin, W.E.N., 2022b. Physical and Biogeochemical Measurements of Belowground Biomass and Carbon Content from Scottish Salt Marshes, 2021. NERC Environmental Information Data Centre. https://doi.org/10.5285/032627e0-5780-4601-b9b3-e684403cee70.
Morris, K., Butterworth, J.C., Livens, F.R., 2000. Evidence for the remobilization of Sellafield waste radionuclides in an intertidal salt marsh, West Cumbria, UK. Estuar. Coast Shelf Sci. 51 (5), 613–625. https://doi.org/10.1006/ecss.2000.0705.
Mueller, P., Do, H.T., Jensen, K., Nolte, S., 2019a. Origin of organic carbon in the topsoil of Wadden Sea salt marshes. Mar. Ecol. Prog. Ser. 624, 39–50. https://doi.org/10.3354/meps13009. Wadden Sea. Ecosphere, 10(1), p.e02556.
Mueller, P., Ladiges, N., Jack, A., Schmiedl, G., Kutzbach, L., Jensen, K., Nolte, S., 2019b. Assessing the long-term carbon-sequestration potential of the semi-natural salt marshes in the European Wadden Sea. Ecosphere 10 (1), e02556. https://doi.org/10.1002/ecs2.2556.
Natali, C., Bianchini, G., Carlino, P., 2020. Thermal stability of soil carbon pools: inferences on soil nature and evolution. Thermochim. Acta 683, 178478. https://doi.org/10.1016/j.tca.2019.178478.
Nellemann, C., Corcoran, E. (Eds.), 2009. Blue Carbon: the Role of Healthy Oceans in Binding Carbon: a Rapid Response Assessment. UNEP/Earthprint.
Ouyang, X., Lee, S.Y., 2014. Updated estimates of carbon accumulation rates in coastal marsh sediments. Biogeosciences 11 (18), 5057–5071. https://doi.org/10.5194/bg-11-5057-2014.

Penk, M.R., Perrin, P.M., Waldren, S., 2020. Above-to belowground vegetation biomass ratio in temperate north-east atlantic saltmarshes increases strongly with soil nitrogen gradient. Ecosystems 23 (3), 648–661. https://doi.org/10.1007/s10021-019-00428-z.

Pondell, C., Canuel, E., 2022. Composition of organic matter in soils from tidal marshes around the Chesapeake Bay, USA, as revealed by lipid biomarkers and stable carbon and nitrogen isotopes. Estuar. Coast Shelf Sci. https://doi.org/10.1016/j.ecss.2022.108068.

Porter, J., Austin, W., Burrows, M., Clarke, D., Davies, G., Kamenos, N., Riegel, S., Smeaton, C., Page, C., Want, A., 2020. Blue carbon audit of Orkney waters. Scottish Marine Freshwater Sci. 11 (3), 96. https://doi.org/10.7489/12262-1.

Potouroglou, M., Whitlock, D., Milatovic, L., MacKinnon, G., Kennedy, H., Diele, K., Huxham, M., 2021. The sediment carbon stocks of intertidal seagrass meadows in Scotland. Estuar. Coast Shelf Sci. 258, 107442 https://doi.org/10.1016/j.ecss.2021.107442.

Pulford, I.D., Allan, R.L., Cook, G.T., MacKenzie, A.B., 1998. Geochemical associations of Sellafield-derived radionuclides in saltmarsh deposits of the Solway Firth. Environ. Geochem. Health 20 (2), 95–101. https://doi.org/10.1023/A:1006597809373.

Ramirez, M.T., Allison, M.A., Bianchi, T.S., Cui, X., Savage, C., Schüller, S.E., Smith, R. W., Vetter, L., 2016. Modern deposition rates and patterns of organic carbon burial in Fiordland, New Zealand. Geophys. Res. Lett. 43 (22), 11–768. https://doi.org/10.1002/2016GL070021.

Rodwell, J.S., 2000. Maritime Communities and Vegetation of Open Habitats, 5.

Rogers, K., Macreadie, P.I., Kelleway, J.J., Saintilan, N., 2019. Blue carbon in coastal landscapes: a spatial framework for assessment of stocks and additionality. Sustain. Sci. 14, 453–467. https://doi.org/10.1007/s11625-018-0575-0.

Saintilan, N., Rogers, K., Mazumder, D., Woodroffe, C., 2013. Allochthonous and autochthonous contributions to carbon accumulation and carbon store in southeastern Australian coastal wetlands. Estuar. Coast Shelf Sci. 128, 84–92. https://doi.org/10.1016/j.ecss.2013.05.010.

Sanchez-Cabeza, J.A., Ruiz-Fernández, A.C., 2012. 210Pb sediment radiochronology: an integrated formulation and classification of dating models. Geochem. Cosmochim. Acta 82, 183–200. https://doi.org/10.1016/j.gca.2010.12.024.

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

Shennan, I., Bradley, S.L., Edwards, R., 2018. Relative sea-Level changes and crustal movements in britain and Ireland since the last glacial maximum. Quat. Sci. Rev. 188, 143–159. https://doi.org/10.1016/j.quascirev.2014.06.008.

Shi, Z., Lamb, H.F., 1991. Post-glacial sedimentary evolution of a microtidal estuary, Dyfi Estuary, west Wales, UK. Sediment. Geol. 73 (3–4), 227–246. https://doi.org/10.1016/0037-0738(91)90086-S.

Smeaton, C., Austin, W.E., 2017. Sources, sinks, and subsidies: terrestrial carbon storage in mid-latitude fjords. J. Geophys. Res.: Biogeosciences 122 (11), 2754–2768. https://doi.org/10.1002/2017JG003952.

Smeaton, C., Austin, W.E.N., 2022. Quality not quantity: prioritizing the management of sedimentary organic matter across continental shelf seas. Geophys. Res. Lett. 49 (5), e2021GL097481 https://doi.org/10.1029/2021GL097481.

Smeaton, C., Barlow, N.L., Austin, W.E., 2020. Coring and compaction: best practice in blue carbon stock and burial estimations. Geoderma 364, 114180. https://doi.org/10.1016/j.geoderma.2020.114180.

Smeaton, C., Cui, X., Bianchi, T.S., Cage, A.G., Howe, J.A., Austin, W.E., 2021b. The evolution of a coastal carbon store over the last millennium. Quat. Sci. Rev. 266, 107081 https://doi.org/10.1016/j.quascirev.2021.107081.

Smeaton, C., Hunt, C.A., Turrell, W.R., Austin, W.E., 2021a. Marine sedimentary carbon stocks of the United Kingdom's exclusive economic zone. Front. Earth Sci. 9, 50. https://doi.org/10.3389/feart.2021.593324.

Smeaton, C., Miller, L.C., Ladd, C.J.T., O'Dell, A., Austin, W.E.N., 2022a. Bulk Elemental and Stable Isotope Composition of Organic Matter from Terrestrial, Intertidal, and Marine Environments. Environmental Information Data Centre, UK, pp. 2016–2021. https://doi.org/10.5285/a445a7a8-528d-4e0b-9094-28cbcd449367. *NERC EDS*.

Smeaton, C., Ladd, C.T.J., Miller, L.C., Skov, M.W., Austin, W.E.N., 2022b. Physical and Biogeochemical Measurements of Aboveground (Vegetation) Biomass from across Ten Saltmarshes, UK, 2019-2020. NERC EDS Environmental Information Data Centre. https://doi.org/10.5285/f71c9f3e-0ae1-4318-a3ea-1dd30b7af3be.

Smeaton, C., Burden, A., Ruranska, P., Ladd, C.J., Garbutt, A., Jones, L., McMahon, L., Miller, L.C., Skov, M.W., Austin, W.E., 2022c. Using citizen science to estimate surficial soil Blue Carbon stocks in Great British saltmarshes. Front. Mar. Sci. 1461. https://doi.org/10.3389/fmars.2022.959459.

Smith, R.W., Bianchi, T.S., Allison, M., Savage, C., Galy, V., 2015. High rates of organic carbon burial in fjord sediments globally. Nat. Geosci. 8 (6), 450–453. https://doi.org/10.1038/ngeo2421.

Theuerkauf, E.J., Stephens, J.D., Ridge, J.T., Fodrie, F.J., Rodriguez, A.B., 2015. Carbon export from fringing saltmarsh shoreline erosion overwhelms carbon storage across a critical width threshold. Estuar. Coast Shelf Sci. 164, 367–378. https://doi.org/10.1016/j.ecss.2015.08.001.

Thorhaug, A.L., Poulos, H.M., López-Portillo, J., Barr, J., Lara-Domínguez, A.L., Ku, T.C., Berlyn, G.P., 2019. Gulf of Mexico Estuarine Blue Carbon Stock, Extent and Flux: Mangroves, Marshes, and Seagrasses: A North American Hotspot, 653. Science of the total environment, pp. 1253–1261.

Thornton, S.F., McManus, J., 1994. Application of organic carbon and nitrogen stable isotope and C/N ratios as source indicators of organic matter provenance in estuarine systems: evidence from the Tay Estuary, Scotland. Estuar. Coast Shelf Sci. 38 (3), 219–233. https://doi.org/10.1006/ecss.1994.1015.

Troels-Smith, J., 1955. Characterization of Unconsolidated Sediments. Reitzels Forlag.

Vaughn, D.R., Bianchi, T.S., Shields, M.R., Kenney, W.F., Osborne, T.Z., 2020. Increased organic carbon burial in northern Florida mangrove-salt marsh transition zones. Global Biogeochem. Cycles 34 (5), e2019GB006334. https://doi.org/10.1029/2019GB006334.