8-15-2023

Drivers of variability in Blue Carbon stocks and burial rates across European estuarine habitats

Inés Mazarrasa

Joao M. Neto

Tjeerd J. Bouma

Tim Grandjean

Jordi Garcia-Orellana

See next page for additional authors

10.1016/j.scitotenv.2023.163957
Mazarrasa, I., Neto, J. M., Bouma, T. J., Grandjean, T., Garcia-Orellana, J., Masqué, P., ... & Juanes, J. A. (2023). Drivers of variability in Blue Carbon stocks and burial rates across European estuarine habitats. Science of The Total Environment, 886, 163957. https://doi.org/10.1016/j.scitotenv.2023.163957

Science of the Total Environment 886 (2023) 163957

Contents lists available at ScienceDirect

Science of the Total Environment

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

Drivers of variability in Blue Carbon stocks and burial rates across European estuarine habitats

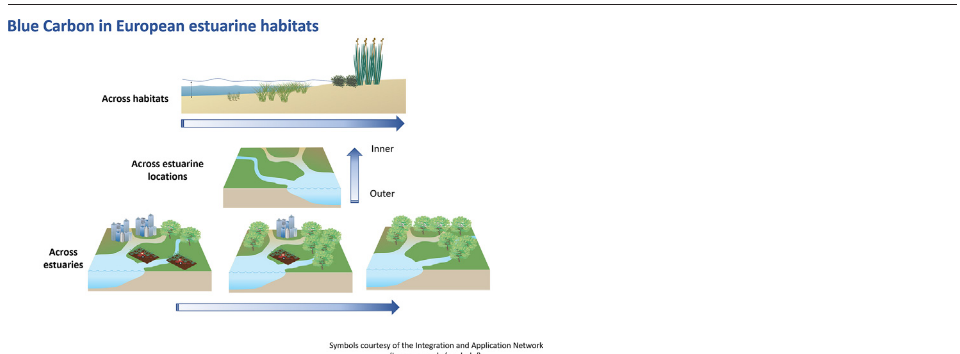
Inés Mazarrasa [a,b,*], Joao M. Neto [c], Tjeerd J. Bouma [d,e], Tim Grandjean [d,e], Jordi Garcia-Orellana [f,g], Pere Masqué [h,i], María Recio [a], Óscar Serrano [b,i], Araceli Puente [a], José A. Juanes [a]

[a] IHCantabria - Instituto de Hidráulica Ambiental de la Universidad de Cantabria, Spain
[b] Centro de Estudios Avanzados de Blanes, Consejo Superior de Investigaciones Cientificas, Blanes, Girona, Spain
[c] MARE- Marine and Environmental Sciences Centre, Department of Life Sciences, University of Coimbra, Portugal
[d] Department of Estuarine and Delta Systems, Royal Netherlands Institute for Sea Research, 4401 NT Yerseke, the Netherlands
[e] Faculty of Geosciences, Department of Physical Geography, Utrecht University, Utrecht, the Netherlands
[f] Institut de Ciència i Tecnologia Ambientals, Universitat Autònoma de Barcelona, 08193 Bellaterra, Spain
[g] Departament de Física, Universitat Autònoma de Barcelona, 08193 Bellaterra, Spain
[h] International Atomic Energy Agency, Monaco
[i] Centre for Marine Ecosystems Research, School of Science, Edith Cowan University, Joondalup, WA, Australia

HIGHLIGHTS

- Soil C_{org} storage across intertidal habitats of 5 European estuaries is examined.
- C_{org} storage was higher in high-marshes and decreased towards the estuaries mouth.
- Unvegetated tidal flats showed comparable C_{org} storage capacity to low marshes and seagrasses.
- C_{org} stocks were positively related to silt & clay content.
- C_{org} stocks increased with higher proportion of natural land in the river basins.

GRAPHICAL ABSTRACT



ARTICLE INFO

Editor: Wei Shi

Keywords:
Saltmarshes
Seagrasses
Tidal flats
Carbon storage
Estuaries
Land use

ABSTRACT

The implementation of climate change mitigation strategies based on the conservation and restoration of Blue Carbon ecosystems requires a deep understanding of the magnitude and variability in organic carbon (C_{org}) storage across and within these ecosystems. This study explored the variability in soil C_{org} stocks and burial rates across and within intertidal estuarine habitats of the Atlantic European coast and its relation to biotic and abiotic drivers. A total of 136 soil cores were collected across saltmarshes located at different tidal zones (high marsh, N = 45; low marsh, N = 30), seagrass meadows (N = 17) and tidal flats (N = 44), and from the inner to the outer sections of five estuaries characterized by different basin land uses. Soil C_{org} stocks were higher in high-marsh communities (65 ± 3 Mg ha^{-1}) than in low-marsh communities (38 ± 3 Mg ha^{-1}), seagrass meadows (40 ± 5 Mg ha^{-1}) and unvegetated tidal flats (46 ± 3 Mg ha^{-1}) whereas C_{org} burial rates also tended to be higher in high marshes (62 ± 13 g m^{-2} y^{-1}) compared to low marshes (43 ± 15 g m^{-2} y^{-1}) and tidal flats (35 ± 9 g m^{-2} y^{-1}). Soil C_{org} stocks and burial rates decreased from inner to outer estuarine sections in most estuaries reflecting the decrease in the river influence towards the estuary mouth. Higher soil C_{org} stocks were related to higher content of silt and clay and higher proportion of forest and natural

* Corresponding author at: IHCantabria - Instituto de Hidráulica Ambiental de la Universidad de Cantabria, Spain.
E-mail address: mazarrasai@unican.es (I. Mazarrasa).

http://dx.doi.org/10.1016/j.scitotenv.2023.163957
Received 2 December 2022; Received in revised form 31 March 2023; Accepted 1 May 2023
Available online 8 May 2023

land within the river basin, pointing at new opportunities for protecting coastal natural carbon sinks based on the conservation and restoration of upland ecosystems. Our study contributes to the global inventory of Blue Carbon by adding data from unexplored regions and habitats in Europe, and by identifying drivers of variability across and within estuaries.

1. Introduction

Ecosystems degradation is the second most important source of anthropogenic CO_2 emissions to the atmosphere (Canadell et al., 2007). Therefore, the conservation and restoration of ecosystems that act as natural carbon sinks have become one of the top priorities to address climate change (Trumper et al., 2009). These strategies have been traditionally focused on terrestrial ecosystems such as forests (e.g., REDD; Soares-Filho et al., 2010). However, the publication of the Blue Carbon report by Nellemann et al. (2009), highlighting the role coastal vegetated ecosystems (e.g., mangroves, seagrass meadows and saltmarshes) play as carbon sinks, triggered increasing interest among countries to include the management of these ecosystems into their climate change mitigation frameworks (e.g., National Determined Contributions) (Herr and Landis, 2016; Herr et al., 2017; Lovelock et al., 2022).

Blue Carbon ecosystems along the European coastline are represented by saltmarshes and seagrass meadows, together with macroalgae that are increasingly being considered as key habitats for climate change mitigation (Krause-Jensen et al., 2018; Macreadie et al., 2019). Within estuarine environments of the Atlantic coast, seagrasses, saltmarshes and unvegetated tidal flats coexist forming a mosaic of habitats distributed along the tidal range. The largest carbon deposits in saltmarshes and seagrass meadows are located in the soil compartment, formed by belowground biomass, plant detritus and/or allochthonous organic carbon (C_{org}) that can accumulate in the anoxic soils over millennia (Mateo et al., 1997; McLeod et al., 2011; Serrano et al., 2019). In addition to C_{org} sequestration, these intertidal ecosystems provide multiple critical ecosystems services to societies, such as the support of biodiversity and fisheries, the protection of coastal areas from erosion and flooding, the regulation of nutrient cycling and water quality, and the provision of social and cultural values (Barbier et al., 2011).

Despite all the benefits these ecosystems provide, vast areas of estuarine ecosystems have been lost in Europe since the beginning of the 20th century due to anthropogenic impacts including land reclamation (Airoldi and Beck, 2007). The loss of saltmarshes and seagrass meadows has led to the loss of significant soil carbon sinks and potentially to CO_2 emissions (Luisetti et al., 2013; Lovelock et al., 2017). The conservation and restoration of estuarine ecosystems in Europe is a timely opportunity to contribute to climate change mitigation (i.e., by enhancing CO_2 sequestration and preventing emissions from disturbed soils) while protecting and restoring all other ecosystem services they provide (Nellemann et al., 2009; EU Biodiversity Strategy for 2030, European Comission COM 2020). However, including the management of estuarine ecosystems into climate change mitigation strategies (e.g., through Nationally Determined Contributions) entails developing and monitoring national carbon inventories and requires a good understanding of the magnitude and variability in C_{org} deposits and burial rates across different spatial scales (Ullman et al., 2013), as well as on the drivers that allow wetlands to continue to develop over long time periods (Temmink et al., 2022).

In Europe, most Blue Carbon assessments focused on seagrass meadows formed by relatively large and mainly subtidal species (e.g. *Posidonia oceanica* and *Zostera marina*; Röhr et al., 2016, 2018; Mazarrasa et al., 2017) whereas estuarine intertidal ecosystems have been overlooked. In particular, the number of studies assessing soil C_{org} deposits and burial rates in European saltmarshes is scarce compared to other regions worldwide (Luisetti et al., 2013; Ouyang and Lee, 2014). In addition, soil C_{org} deposits in the small intertidal seagrass species *Zostera noltei* have been reported only for Ria Formosa (South Portugal) and South of England (Santos et al., 2019; do Amaral Camara Lima et al., 2022), whereas C_{org} burial rate for this species has only been reported in Ria Formosa (Martins et al., 2022) despite being the species that dominates the low intertidal zones in many European estuaries. Finally, although unvegetated tidal flats can significantly contribute to soil C_{org} storage at the estuarine scale, particularly given the large area they usually occupy within estuaries (Gorman et al., 2020; Bulmer et al., 2020), they have received much less attention from the Blue Carbon scientific community (Bulmer et al., 2020), and to the extent of our knowledge, no data is available for Europe.

Soil C_{org} deposits and burial rates can be highly variable across and within intertidal estuarine ecosystems and between estuaries, due to a combination of biotic and abiotic factors that determine the accumulation of belowground biomass, the sedimentation of autochthonous and allochthonous particles, including C_{org}, and the burial efficiency. Some of these factors are the size and life traits of the dominant species, the frequency and duration of the inundation period, the sediment load in the water column, the mud content (% silt and clay) of the sediment which enhances the preservation of soil organic matter (Meyers, 1994), and management practices for resource exploitation within estuaries (e.g., shell fishing, cattle grazing) and beyond (e.g., coastal development and agriculture within the catchment) (Chmura, 2004; Schuerch et al., 2013; Elschot et al., 2015; Dahl et al., 2016; Barañano et al., 2018; Ricart et al., 2020). Within an estuary, factors driving soil C_{org} sequestration vary along two spatial gradients: a vertical gradient determined by the habitats elevation with regard to mean sea level (MSL) and a horizontal gradient determined by the influence of the river and the distance to the estuary mouth (Kelleway et al., 2016; Ricart et al., 2020).

Along a vertical gradient, estuarine intertidal habitats distribute at different elevation with regard to mean sea level (MSL), being subject to marked differences in terms of the frequency and duration of the inundation periods associated to tidal cycles. The upper intertidal zone, only covered by seawater in the highest tide periods, is occupied by saltmarsh communities formed by relatively large and perennial species (e.g., *Halimione* spp. and *Juncus* spp.). The immediately lower level, daily covered by seawater under high tide conditions, is occupied by annual or perennial saltmarsh species, usually of relatively smaller size (e.g., *Spartina* spp., *Sarcocornia* spp.). At the lowest intertidal level that only emerges during the low tide periods, estuarine vegetated communities are represented by intertidal seagrass meadows of small and deciduous species (e.g., *Zostera noltei*) alternated within unvegetated tidal flats (Adam, 2002).

As the frequency and duration of the inundation period increases from upper to lower intertidal, the sedimentation of organic and inorganic particles from the water column tends to increase resulting in differences in sediment accretion and C_{org} burial rates across habitats located at different zones (Stoddart et al., 1989; Cahoon and Reed, 1995; Chmura, 2004). In general, habitats located at a relatively lower intertidal zone show higher sediment accretion rates and C_{org} burial rates than those located at a higher intertidal zone (Temmerman et al., 2003; Ouyang and Lee, 2014). On the contrary, plant above and belowground biomass tend to decrease from habitats located at the upper intertidal zone (i.e. saltmarshes) to those located at the lower intertidal zones (e.g. seagrass meadows and unvegetated tidal flats) (Adam, 2002; Sousa et al., 2017). In addition, the plant chemical composition also differs across the different habitats along the tidal range, leading to different lability of organic matter compounds and thereby, potential for C_{org} storage. For instance, lignin content, which protects organic matter from the action of decomposer organisms, is usually higher in saltmarsh species than in small and deciduous seagrass species (Trevathan-Tackett et al., 2015). As a consequence, larger soil C_{org} deposits (i.e. C_{org} stocks) are usually found in upper intertidal habitats compared to lower intertidal habitats (Kelleway et al., 2016; Macreadie et al., 2017; Santos et al., 2019; Bulmer et al., 2020).

Along horizontal gradients within the estuaries, a decrease in suspended sediments in the water column is usually found from upper estuarine sections towards the estuary mouth, due to a decrease in the influence of the river flow, which is a source of terrestrial organic and mineral particles (Milliman and Syvitski, 1992; Peck et al., 2020). As a consequence, soil C_{org} deposits and burial rates tend to be higher in saltmarshes and seagrass meadows located at the upper estuarine sections compared to those closer to the estuary mouth due to higher allochthonous C_{org} and fine sediment accumulation in the upper sections (Kelleway et al., 2016; Macreadie et al., 2017; Ricart et al., 2020; Gorham et al., 2021).

Soil C_{org} deposits and burial rates in estuarine ecosystems may also vary across estuaries due to differences in basin characteristics including the type of the dominant land use. Land uses at the catchment level are known to influence the pull of suspended sediments transported downstream (Milliman, 2001; Lambert et al., 2017; Sun et al., 2017), which is a key driver in sediment accumulation in estuarine habitats (Kirwan et al., 2011; Peck et al., 2020). For instance, Sun et al. (2017) found that the spread of crops reduced the amount of suspended sediments and particulate C_{org} transported by rivers due to a decrease in runoff associated with water extraction for irrigation, whereas Kirwan et al. (2011) found that deforestation for agriculture expansion led to an increase in sediment supply to estuarine ecosystems due to soil erosion. The presence of urban land can also affect the sediment supply by rivers, by altering fluvial organic matter fluxes towards a more microbial/algal and less plant/soil-derived origin (Lambert et al., 2017), which is more prone to remineralization (Enríquez et al., 1993). Thus, differences in the percentage of agricultural, urban and natural land across estuaries are expected to cause differences in the C_{org} storage of estuarine habitats by affecting the type and amount of suspended particulate C_{org} reaching estuarine environments.

The goal of this study is to identify patterns of variability in the magnitude of soil C_{org} deposits and accumulation rates across and within estuarine intertidal habitats (i.e., saltmarshes, seagrass meadows and tidal flats) in Europe due to differences in species composition, relative elevation with regard to MSL, location across a land-ocean estuarine gradient and drainage basin characteristics (i.e., land uses). We do so through a large-scale sampling effort encompassing saltmarsh communities located at different tidal zones, intertidal seagrass meadows and unvegetated tidal flats, distributed from upper estuarine sections to the estuarine mouths in five estuaries located across a broad latitudinal gradient along the European Atlantic coast. This study significantly increases the data available on Blue Carbon sinks particularly from an underrepresented region, Europe, and underrepresented habitats (e.g. unvegetated tidal flats). In addition, this study contributes to understand the drivers of variability in soil C_{org} deposits and burial rates in estuarine ecosystems, key aspects in order to develop management strategies for climate change mitigation in Europe and elsewhere.

2. Methodology

2.1. Study sites

As part of the European Project LIFE ADAPTABLUES, 5 estuaries distributed from 40.14°N to 51.41°N along the Atlantic coast of Europe were selected for this study: the Mondego estuary (MN) located at the lowest latitude on the west coast of the Iberian Peninsula; the Santoña Marshes (SM), Oyambre (OY) and Santander Bay (SB) estuaries located at an intermediate latitude along the North of the Iberian Peninsula; and the Western Scheldt (WS) located at the highest latitude in the Netherlands (Fig. 1). These estuaries differ in the percentage of land use typologies of their basins and extension of the saltmarsh, seagrass meadows and unvegetated tidal flats (Table 1).

At each estuary, between three to four areas distributed from inner to outer sections were selected (Fig. 1, Table SI 2). In each area, from two to three different habitats located at different elevation with regard to MSL or dominated by different species were sampled along a transect perpendicular to the estuary main channel (Table SI 2). Habitats sampled were classified following Adam (2002) in: "high marshes", formed by perennial saltmarsh species located in the upper intertidal zone (e.g. *Juncus maritimus; Halimione portulacoides)*; "low marshes", formed by annual or perennial saltmarsh species located in the immediately lower intertidal zone (e.g. *Spartina* spp.*; Sarcocornia* spp.); and "seagrass" (*Zostera noltei* species) meadows and "unvegetated tidal flats" (from here on referred as "tidal flats") located at the lowest intertidal zone sampled per area. Thus, in total, 46 sampling stations were selected: 10 in SB, seven in SM, eight in OY, nine in MN and 12 in WS, encompassing different habitats located at different elevation with regard to MSL (i.e. and subject to different frequency and duration of inundation periods) and different estuarine sections (Table SI 1).

2.2. Soil cores sampling and processing

Soil was sampled by extracting three replicate soil cores in each sampling station. All cores were sampled in the summer-autumn of 2019 except the cores in the OY estuary that were sampled in the summer of 2018. After hammering the PVC pipes (7 cm diameter and 60 cm long) into the soil, the length of the PVC tube and the inner and outer distance between the top of the tube and the sediment were measured in order to estimate the soil compression that typically occurs during hammering. The core sediment depths were corrected accordingly (Table SI 1).

Soil cores were preserved frozen until processing in the laboratory. One replicate soil core per sampling station was sliced every 1 cm along the whole soil depth. The other two replicate cores were sliced every 2 cm for the top 20 cm and every 5 cm for deeper layers. Each sediment slice was measured for wet volume (cm^3) and dried at 60 °C until constant weight (dry weight, DW) to estimate sediment dry bulk density (DBD, g DW cm^{-3}) along the soil depth profiles.

2.3. Biogeochemical analysis

Soil organic carbon content (C_{org} %DW) was analyzed in powder subsamples from every other two sediment slices (i.e. one slice analyzed-two slices skipped) from each core in the IHLab Bio laboratory of the Environmental Hydraulics Institute of the Universidad de Cantabria. Organic carbon content was measured using a carbon elemental analyzer (Shimadzu TOC-L + SSM-5000A). From the cores sliced every 1 cm (N = 46), the sediment grain size was measured every 5 cm after digestion with 30 % H_2O_2 at Laboratorio de Sedimentología, Universidad de Barcelona, on a Beckman Coulter LS GB500 and classified according to the Udden-Wentworth grain size scale (Ø: <4 μm, clay; Ø: 4–63 μm, silt; Ø: 63–2000 μm, sand and Ø: 2000–4000 μm, gravel).

Age models of the soil records were obtained based on the excess ^{210}Pb concentration profiles along the accumulated mass at each core. The concentration of ^{210}Pb was measured along the sediment depth profile of 27 cores sliced every 1 cm distributed among the five estuaries of study, encompassing the wide range of habitats, marsh zones and inner and outer sections (Table SI 1). The concentration of ^{210}Pb was determined by alpha spectrometry through the measurement of its granddaughter ^{210}Po, assuming radioactive equilibrium between both radionuclides. About 100–200 mg aliquots of each sample were spiked with ^{209}Po and microwave digested with a mixture of concentrated HNO_3 and HF. Boric acid was then added to complex fluorides. The resulting solutions were evaporated and diluted to 100 mL 1 M HCl and Po isotopes were autoplated onto pure silver disks. Polonium emissions were measured by alpha spectrometry using PIPS detectors (CANBERRA, Mod. PD-450.18 A.M) at Universidad Autónoma de Barcelona and Edith Cowan University. Reagent blanks were comparable to the detector backgrounds. Analyses of replicate samples and reference materials were carried out systematically to ensure the accuracy and the precision of the results. The supported ^{210}Pb was estimated as the average ^{210}Pb

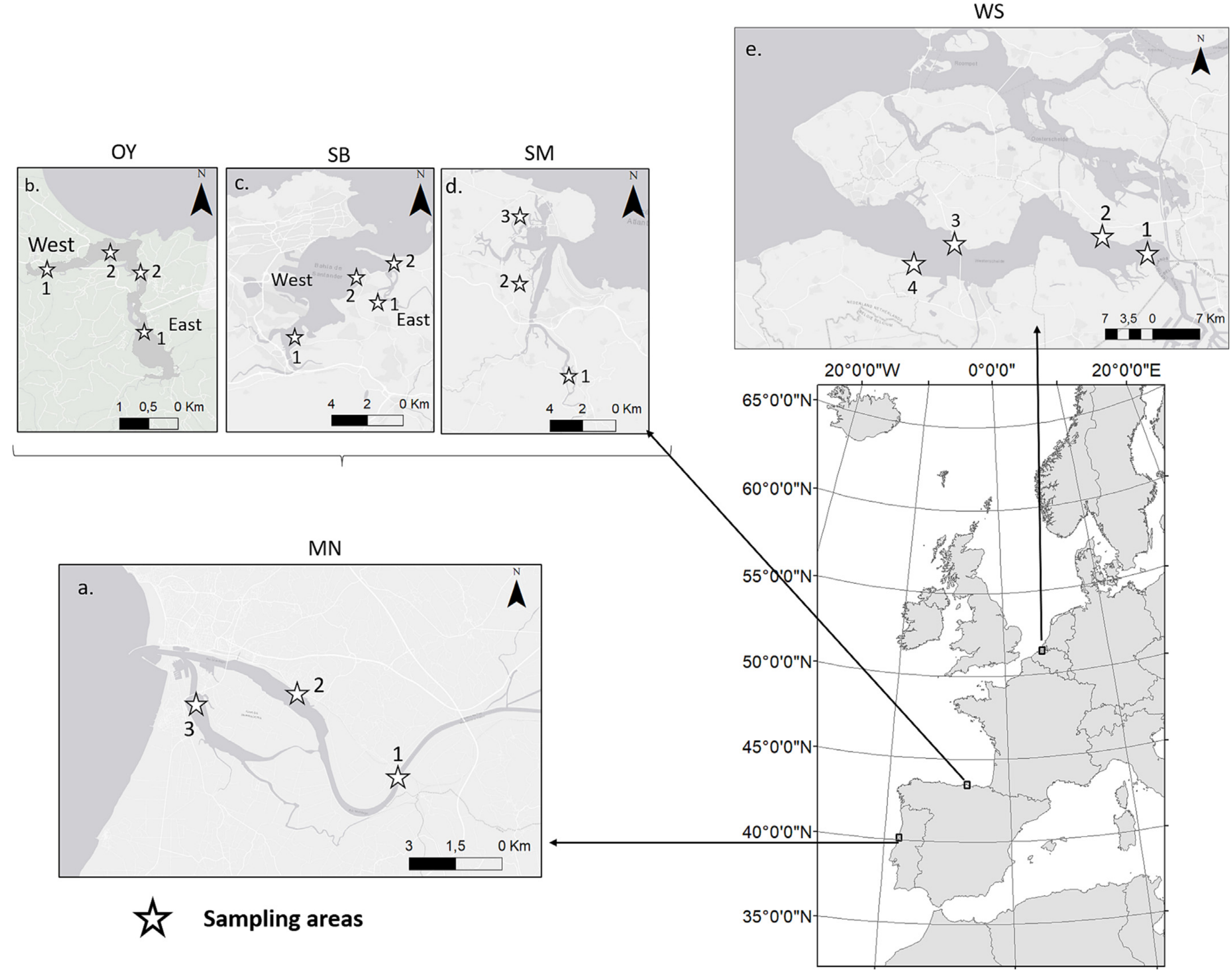


Fig. 1. Location of sampling sites within each of the five estuaries studied: a) Mondego (MN); b) Oyambre (OY) East and West branches; c) Santander Bay (SB) East and West branches; d) Santoña Marshes (SM); and e) Western Scheldt (WS). Increasing numbers (1–4) reflect the relative position of each sampling area within the estuary or estuary branch, from the most inner section to the most outer section. Needs to be updated.

concentration of the deepest layers once ^{210}Pb reached constant values. Then, excess ^{210}Pb concentrations were obtained by subtracting the supported ^{210}Pb from the total ^{210}Pb. The sedimentation rates were estimated using the CRS and CF:CS models (Krishnaswamy et al., 1971; Appleby and Oldfield, 1978) in all sediment profiles except in one from OY estuary, where the age model was obtained from the concentration profile of ^{137}Cs measured along the core by gamma spectrometry, since apparent presence of mixing along the upper 8 cm of the sediment precluded the application of ^{210}Pb aging models.

2.4. Numerical procedures

Soil C_{org} stocks within the top 30 cm of each core were estimated by calculating the cumulative C_{org} stock (g C_{org} cm^{-2}) downcore. C_{org} content (C_{org}, %DW) of not analyzed sections was estimated as the average C_{org} content of the slices above and below. When necessary (i.e., soil cores shorter than 30 cm), we inferred C_{org} stocks below the limits of the reported data to 30 cm by extrapolating linearly integrated values of C_{org} content (cumulative C_{org} stock per unit area), with depth. Linear regression analysis

Table 1
Biogeomorphological characteristics of the estuaries studied, including the extent of Blue Carbon ecosystems.

Estuary	Basin size (ha)	Basin land use (%)					Surface area per studied habitat (Ha)			
		Forests and natural areas (%)	Agricultural land (%)	Artificial surfaces (%)	Water bodies (%)	Coastal wetlands (%)	High marsh[a]	Low marsh[a]	Seagrass	Mud/sand flats[a]
MN	667,000	66.43	26.10	6.66	0.59	0.23	103.80	15.49	27.90[a]	216.48
SB	50,816	37.87	48.69	11.95	1.04	0.49	13.91	6.04	189.24[b]	1120.51
SB	73,981	62.12	35.05	1.94	0.50	0.49	58.84	263.80	278.00[b]	527.57
OY	23,714	48.04	45.81	2.97	2.56	0.62	38.52	2.36	4.00	28.21
WS	2,186,300	26.40	47.13	24.01	1.23	0.11	2110	259	0[a]	100

[a] Data extracted from the project LIFE ADAPTABLUES (https://lifeadaptablues.eu/es/).
[b] Data extracted from NANO Project viewer (https://nano.ihcantabria.com/resultados/visor-nanozoostera/).

Table 2
Mean ± SE SAR and C_{org} burial rates since 1960 estimated in successfully ^{210}Pb-dated cores from saltmarsh, seagrass and tidal flat habitats across the estuaries studied.

Estuary	Location	Habitat	^{210}Pb dating		SAR (cm y^{-1}) since 1960	C_{org} burial (g m^{-2} y^{-1}) since 1960	Core code
			Year closest to 1960	Depth (cm)			
SB	East_1	High marsh	1959 ± 7	11.29	0.19 ± 0.02	51.98 ± 5.67	BS1a1
	East_1	Low marsh	1959 ± 16	11.21	0.18 ± 0.04	41.73 ± 10.36	BS1B1
	East_1	Tidal flat	1968 ± 4	15.5	0.30 ± 0.02	47.50 ± 3.46	BS1C2
	East_2	High marsh	1959 ± 2	5.5	0.09 ± 0.00	20.23 ± 0.73	BS2A1
SM	1	HIgh marsh	1959 ± 4	16.87	0.28 ± 0.02	76.79 ± 5.31	MS1A2
	1	Tidal flat	1963 ± 4	15.5	0.28 ± 0.02	46.44 ± 3.72	MS1C3
	2	High marsh	1959 ± 3	14.5	0.24 ± 0.01	59.77 ± 3.2	MS2A3
	2	Low marsh	1956 ± 3	9.39	0.15 ± 0.01	16.47 ± 0.90	MS2B2
	2	Tidal flat	1958 ± 6	9.5	0.16 ± 0.01	13.23 ± 1.25	MS2C3
MN	1	High marsh	1962 ± 9	7.75	0.14 ± 0.02	42.25 ± 6.81	MM3A1
	2	Tidal flat	1962 ± 9	9.76	0.16 ± 0.03	31.97 ± 6.81	MM2C1
	2	High marsh	1961 ± 12	9.33	0.17 ± 0.03	46.55 ± 7.06	MM2A1
	3	Low marsh	1962 ± 4	16.58	0.29 ± 0.02	70.01 ± 5.25	MM1A1
	3	Seagrass	1961 ± 5	18.86	0.33 ± 0.03	93.94 ± 7.51	MM1B1
OY	East_1	Tidal flat	1966 ± 12	3.91	0.07 ± 0.02	5.94 ± 1.40	RP3
	East_2	High marsh	1963 ± 2	16.22	0.29 ± 0.01	108.89 ± 4.02	NM3
	East_2	Seagrass	1963 ± 5	20.69	0.35 ± 0.03	45.57 ± 3.95	NP2
	West_1	High marsh	1961 ± 9	18.97	0.33 ± 0.05	120.99 ± 18.38	A2M3
	West_2	High marsh	1973 ± 2	2.5	0.06 ± 0.00	32.03 ± 1.33	A1M1
	West_2	Tidal flat	1963	14.41	0.26 ± 0.00	67.29 ± 0.00	A1P1

between extrapolated C_{org} stocks and measured C_{org} stocks in soil cores 30 cm long or longer had an $r^2 = 0.9$ (Fig. SI 1).

To allow comparisons among sites, soil accretion rates (SAR, cm yr^{-1}) and soil C_{org} burial rates were standardized to encompass the period between 1960 (i.e. the oldest year that could be consistently identified across all dated cores) and 2018 or 2019 (i.e., year of sampling). SAR was estimated by dividing the soil depth (decompressed) accumulated since 1960 (or the closest year in the sediment depth profile, Table 2) by the number of years spanned until sampling. Soil C_{org} burial rate was estimated by dividing C_{org} stocks accumulated since 1960 (Table 2) by the number of years passed since sampling year (2018 or 2019).

2.5. Classification of land-use within the catchment

Land uses were classified using greater Corine Land Cover classification level, in: 1) forest and seminatural areas (e.g., natural grasslands

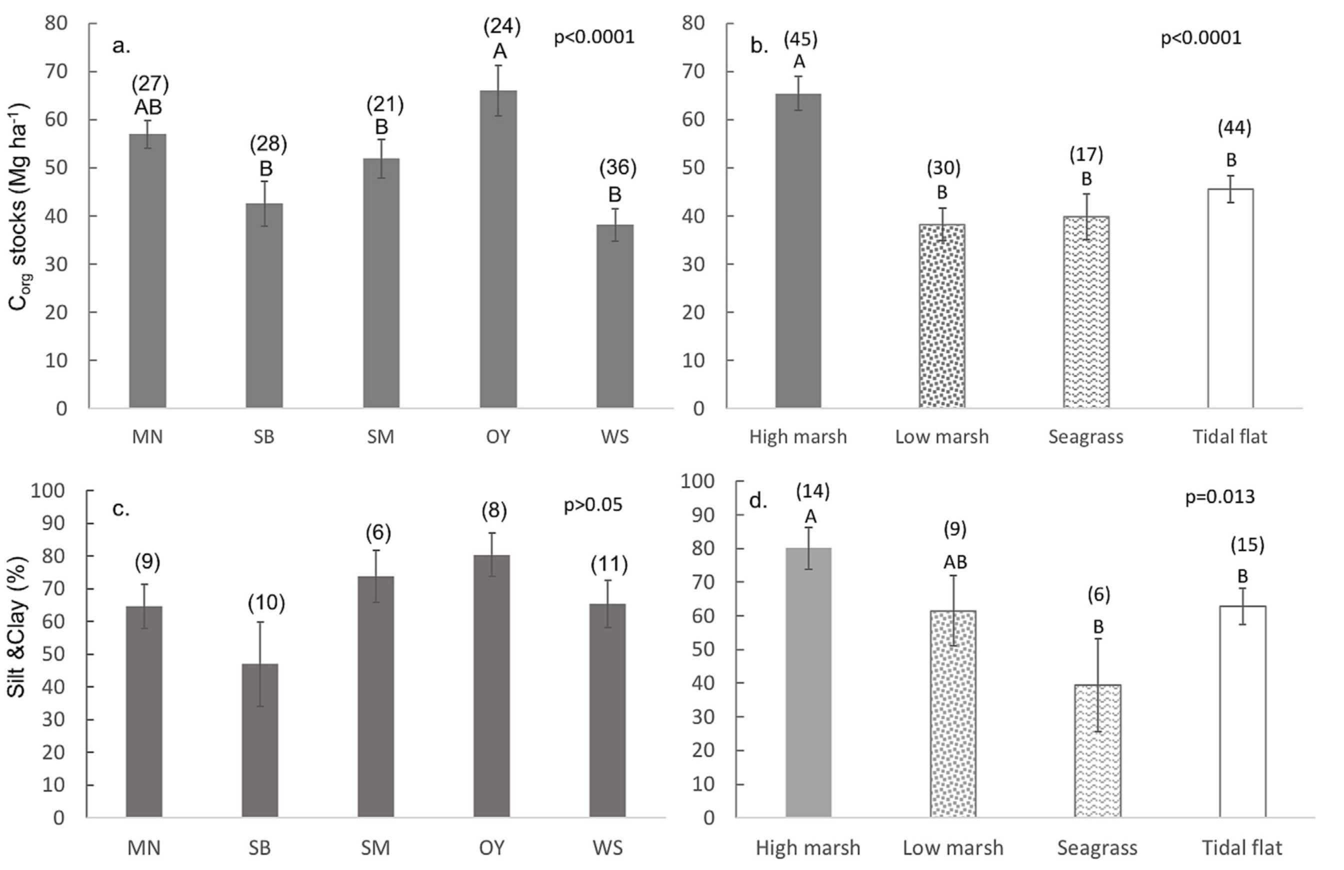


Fig. 2. Mean ± SE top 30 cm soil organic carbon (C_{org}) stocks (Mg C_{org} ha^{-1}) (a, b) and silt and clay content (%) (c, d) across estuaries (MN: Mondego estuary, SB: Santander Bay, SM: Santoña Marshes, OY: Oyambre estuary, WS: Western Scheldt) (a, c); and habitats (b, d). Error bars represent standard error. Different letters indicate significant differences (p < 0.05; Wilcoxon Test) between categories, whereas the numbers in brackets indicate the number of cores studied.

and mixed forest); 2) artificial surfaces (e.g., road and rail networks, urban areas and ports); 3) agricultural areas (e.g., pastures and agriculture); 4) coastal wetlands (e.g., salt ponds and saltmarshes); and 5) water bodies (e.g., sea, ocean and estuaries). Cartographies of estuarine basins were extracted from Feio et al. (2007) for the Mondego estuary, Goemans et al. (2007) for the Western Scheldt estuary, and IHCantabria (2005) for the Oyambre, Santoña Marshes and Santander Bay estuaries.

2.6. Statistical analysis

Differences in soil C_{org} stocks, grain size, SAR, and soil C_{org} burial rates across habitats and estuaries were assessed using Wilcoxon test non-parametric analysis. In the case of soil C_{org} stocks, differences across estuarine sections and habitats within each estuary were also assessed using Wilcoxon tests. It was not possible to assess differences in sediment grain-size properties, soil accretion rates (SAR), and soil C_{org} burial rates across estuarine sections and habitats within each estuary due to limited replicates. Spearman correlation tests were applied to assess correlations between sediment grain size, soil C_{org} stocks, SAR and soil C_{org} burial rates, and between land use patterns and soil C_{org} stocks. All statistical analyses were conducted using JMP statistical software.

3. Results

3.1. Soil C_{org} stocks

Top 30 cm soil C_{org} stocks ranged from 1.9 to 123 Mg C_{org} ha^{-1} (49 ± 2 Mg C_{org} ha^{-1}; mean ± SE) and were significantly different across estuaries, habitats and estuarine locations. The largest stocks were found in OY (66 ± 5 Mg C_{org} ha^{-1}) and MN (57 ± 3 Mg C_{org} ha^{-1}) estuaries, followed by SM (52 ± 4 Mg C_{org} ha^{-1}), SB (43 ± 3 Mg C_{org} ha^{-1}) and the WS (38 ± 3 Mg C_{org} ha^{-1}) (Fig. 2a, Table SI 2).

Across all estuaries studied, high marsh communities showed significantly higher soil C_{org} stocks (64 ± 3 Mg C_{org} ha^{-1}) than low marshes (38 ± 3 Mg C_{org} ha^{-1}), seagrass meadows (40 ± 45 Mg C_{org} ha^{-1}) and tidal flats (46 ± 3 Mg C_{org} ha^{-1}) ($p < 0.001$; Table SI 2; Fig. 2b). Indeed, high marshes also stored significantly higher C_{org} stocks than adjacent habitats (i.e., low marsh, seagrass and/or tidal flats) within each estuary and estuarine section (Fig. 3). Different trends were found across the other three habitats at each estuarine section, with low marshes showing either higher, similar or lower soil C_{org} stocks to those found in adjacent tidal flats or seagrass meadows.

Soil C_{org} stocks were significantly higher at the inner sections compared to sections closer to the estuary mouth in the two estuarine branches of SB, and in the western branch of the OY estuary ($p < 0.05$; Fig. 3, Table SI 2). In

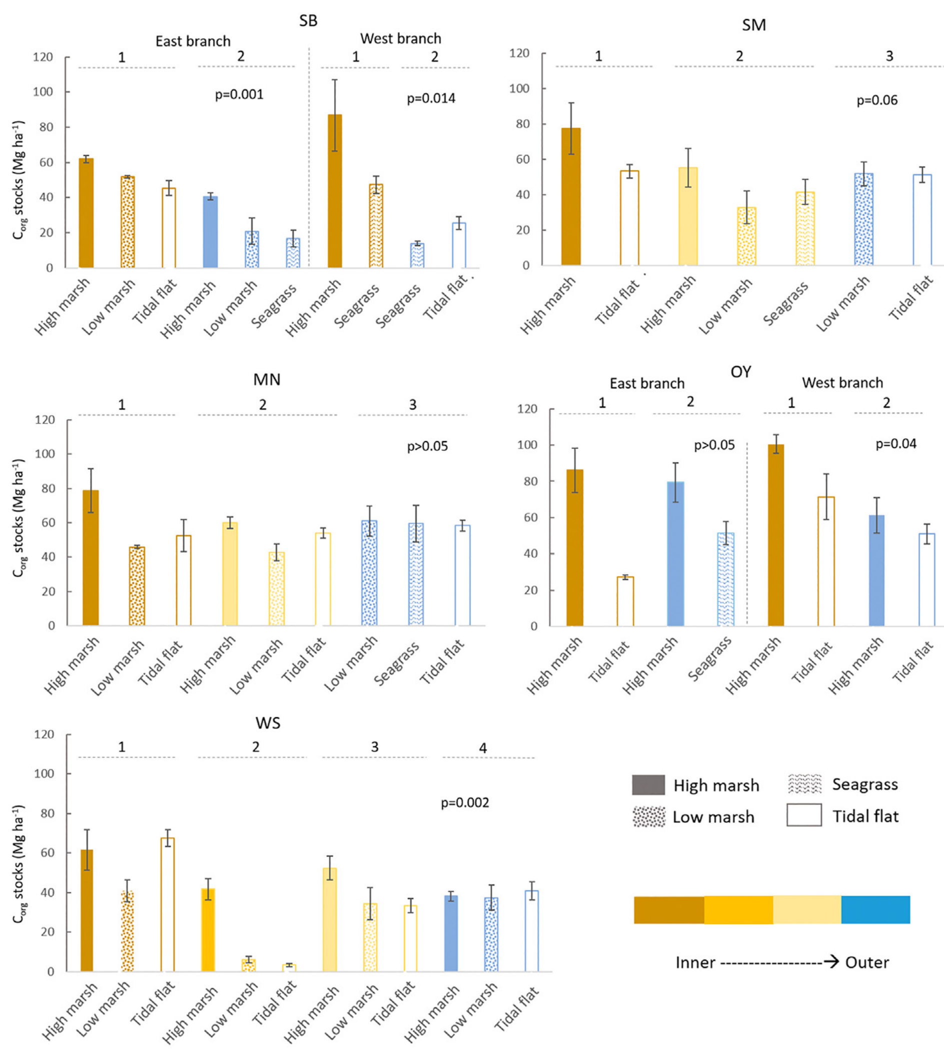


Fig. 3. Mean ± SE top 30 cm soil C_{org} stocks (Mg C_{org} ha^{-1}) per habitat (n = 3 core replicates) within each section per estuary (MN: Mondego estuary, SB: Santander Bay, SM: Santoña Marshes, OY: Oyambre estuary, WS: Western Scheldt). Number above the bars indicates the sampling area within each estuary. Color gradient indicates relative position along the estuary (from inner to outer sections) while bar fill patterns represent different habitats.

the WS, soil C_{org} stocks also tended to decrease from inner sections to sections closer to the estuary mouth although this general trend was disrupted by the second most inner section sampled ($p > 0.05$). No significant differences were found across estuarine sections in the MN estuary and the east branch of OY estuary ($p > 0.05$).

3.2. Silt and clay content

Silt and clay content across all sites ranged from 1.2 to 99 % (26 ± 2 %), and no significant differences were found across estuaries ($p > 0.05$; Fig. 2c, Table SI 2). However, silt and clay content was significantly higher in high marsh communities (80 ± 6 %) compared to the other habitats examined (ranging from 39 ± 14 % in seagrass meadows to 63 ± 5 % in tidal flats; $p < 0.05$; Fig. 2d). Differences in the content of silt and clay across low marshes, seagrass meadows and tidal flats showed different trends across estuarine sections (Fig. 4). The content of silt and clay tended to decrease from inner sections to outer sections of the estuaries in both branches of SB and in the west branch of OY, whereas no clear trends were detected for the other estuaries (Fig. 4).

3.3. Soil accretion rates (SAR) and soil C_{org} burial

The ^{210}Pb concentration profiles along the soil cores allowed obtaining age models in 21 of the 27 cores analyzed, while for the other six cores the data suggested mixing (and an age model could not be obtained) or lack of sedimentation (Table 2). None of the cores analyzed from the WS showed a net accumulation of excess ^{210}Pb and therefore, it was not possible to estimate SAR or soil C_{org} burial. Thus, the following results refer only to the cores sampled in the MN estuary, OY estuary, SB and SM.

SAR and soil C_{org} burial rates since 1960 ranged from 0.06 ± 0.002 to 0.35 ± 0.03 cm y^{-1} (0.22 ± 0.03 cm y^{-1}) and from 5.9 to 121 g C_{org} m^{-2} y^{-1} (55 ± 10 g C_{org} m^{-2} y^{-1}), respectively (Table 2). No significant differences in SAR and soil C_{org} burial rates were found across estuaries ($p > 0.05$; Fig. 5, Table SI 3).

Data from seagrass meadows was excluded from the analysis of differences among habitats because only two seagrass cores could be dated. No significant differences in SAR were found among habitats ($p > 0.05$; Fig. 5, Table SI3) when considering all estuarine sections together neither

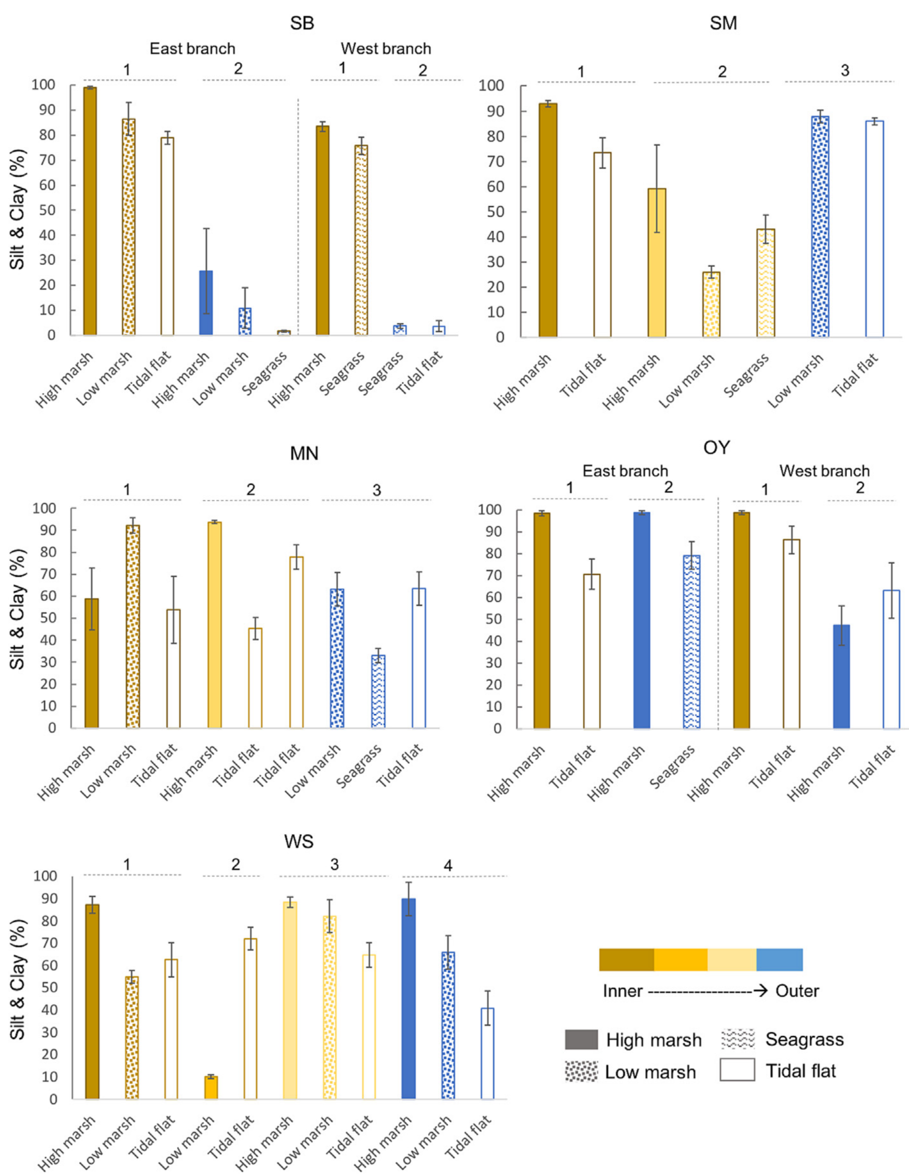


Fig. 4. Mean ± SE content of silt and clay (%) per habitat (n = 1 core) within each section per estuary (MN: Mondego estuary, SB: Santander Bay, SM: Santoña Marshes, OY: Oyambre estuary, WS: Western Scheldt). Number above the bars indicates the sampling area within each estuary. Color gradient indicates relative position along the estuary (from inner to outer sections) while bar fill patterns represent different habitats.

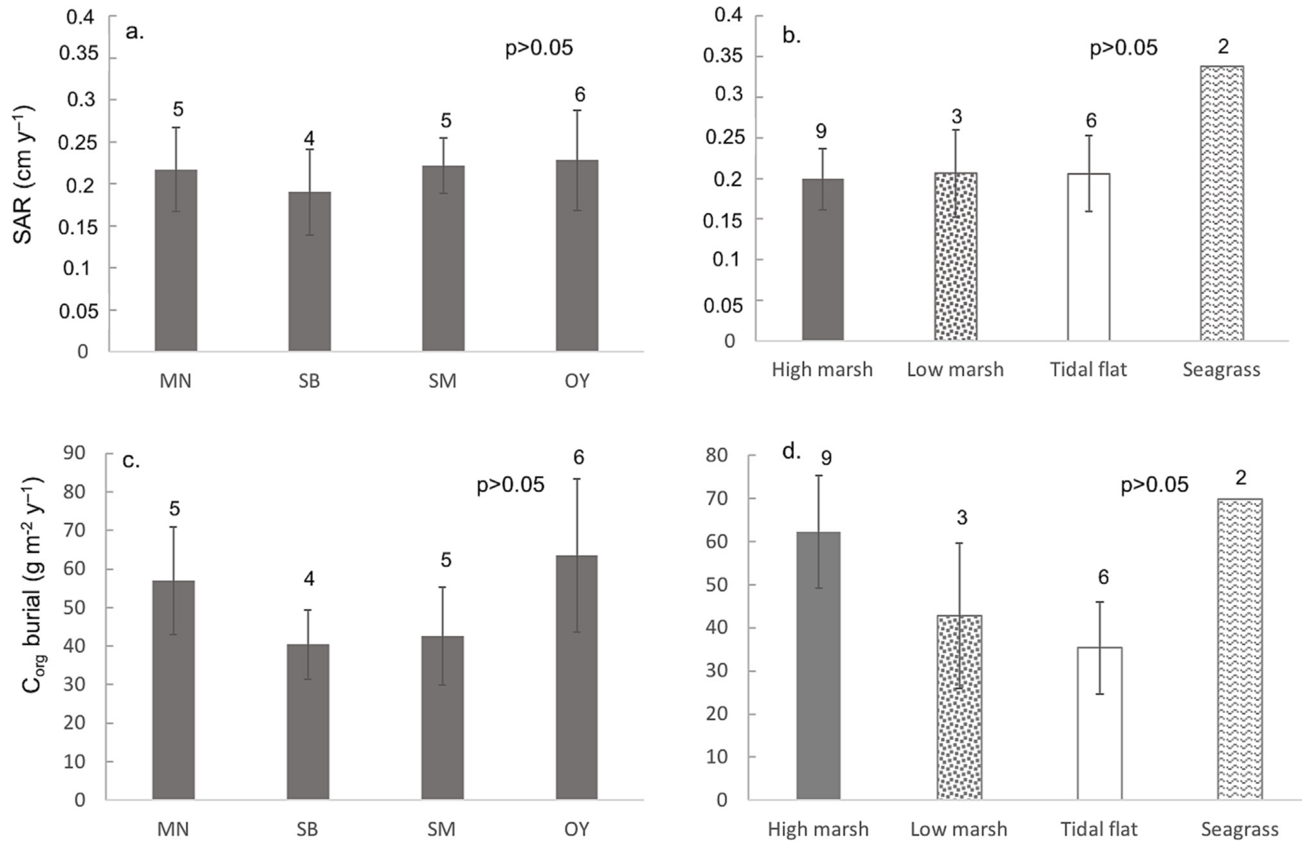


Fig. 5. Mean ± SE sediment accretion rate (SAR, cm y^{-1}) (a, b) and C_{org} burial rate (c, d) across estuaries (MN: Mondego estuary, SB: Santander Bay, SM: Santoña Marshes, OY: Oyambre estuary, WS: Western Scheldt) (a,c) and habitats (b,d). Numbers above bars indicate the number of cores. The *p* value ($p > 0.05$) indicates no significant differences.

within different estuarine sections (Fig. 6a). Soil C_{org} burial rates tended to decrease from high marsh (68 ± 14 g C_{org} m^{-2} y^{-1}) to low marsh (43 ± 17 g C_{org} m^{-2} y^{-1}) and tidal flats (35 ± 11 g C_{org} m^{-2} y^{-1}), although these differences were not statistically different ($p > 0.05$; Fig. 5, Table SI3). When looking at each estuarine section, soil C_{org} burial rates were higher in high marshes compared to adjacent habitats across all estuaries (Fig. 6, Table 2).

SAR and soil C_{org} burial rates tended to decrease from inner estuarine sections to the estuary mouth in SB (from 0.22 to 0.09 cm y^{-1} and from 47 to 20 g C_{org} m^{-2} y^{-1}, respectively), in SM (from 0.28 to 0.18 cm y^{-1} and from 61 to 30 g C_{org} m^{-2} y^{-1}, respectively), and the west branch of OY estuary (from 0.33 to 0.16 cm y^{-1} and from 120 to 50 g C_{org} m^{-2} y^{-1}, respectively). On the contrary, SAR and soil C_{org} burial rates tended to increase from inner to outer estuarine sections in Mondego (from 0.14 to 0.31 cm y^{-1} and from 42 to 82 g C_{org} m^{-2} y^{-1}, respectively) and the east branch of OY (from 0.07 to 0.32 cm y^{-1} and from 5.9 to 77 g C_{org} m^{-2} y^{-1}, respectively) (Fig. 6, Table 2). The number of successfully dated cores was not enough to assess statistical differences in SAR and C_{org} burial across estuarine sections.

3.4. Correlation analysis

Soil C_{org} stocks were positively correlated with the content of silt and clay ($p < 0.05$; Table 3; Fig. SI 2) whereas no significant correlations were found between the content of silt and clay and SAR or C_{org} burial rate ($p > 0.05$; Fig. SI 2). Soil C_{org} stocks across estuaries were positively correlated with the percentage of forest and natural land in the basin ($p < 0.0001$; Fig. 7a; Table 3), and negatively correlated with the percentage of artificial areas ($p < 0.0001$; Fig. 7b) and agricultural land ($p < 0.0001$; Fig. 7c; Table 3).

4. Discussion

The intertidal habitats examined stored an overall average of 50 ± 2 Mg C_{org} ha^{-1} (1.86–123 Mg C_{org} ha^{-1}) in the top 30 cm of soil and vertically accreted and buried C_{org} at overall averages of 0.23 ± 0.02 cm y^{-1} and 55 ± 7 g C_{org} m^{-2} y^{-1} since 1960, respectively. Differences in soil C_{org} storage across habitats, estuarine locations and estuaries were observed as a result of a combination of biotic, hydrodynamic and anthropogenic factors.

4.1. Variability across habitats

The largest soil C_{org} stocks found in high marshes compared to low marshes, intertidal seagrass meadows and tidal flats, are consistent with the relatively larger above and belowground biomass of the saltmarsh species dominating this habitat (i.e. *Halimione* spp., *Juncus* spp.) (Adam, 2002; Sousa et al., 2017; Ewers Lewis et al., 2020; Bulmer et al., 2020). In addition, high marshes have been found to act as detritus sinks, receiving a significant fraction of the organic matter produced by low marsh communities and exported through tides (Bouchard and Lefeuvre, 2000). High marshes soils also showed the highest content of silt and clay across habitats, likely contributing to enhance organic matter preservation and the accumulation of largest C_{org} deposits (Meyers, 1994; Burdige, 2007). The higher content of silt and clay in high marsh communities is indicative of a lower hydrodynamic energy environment compared to the other habitats. This could be attributed to high marsh species having a higher current dampening effect due to structural features such as their higher stem stiffness and biomass (e.g. density) compared to the other habitats examined (Peralta et al., 2008; Sousa et al., 2017). In addition, flow velocity is lower at upper intertidal ranges compared to relatively lower intertidal ranges

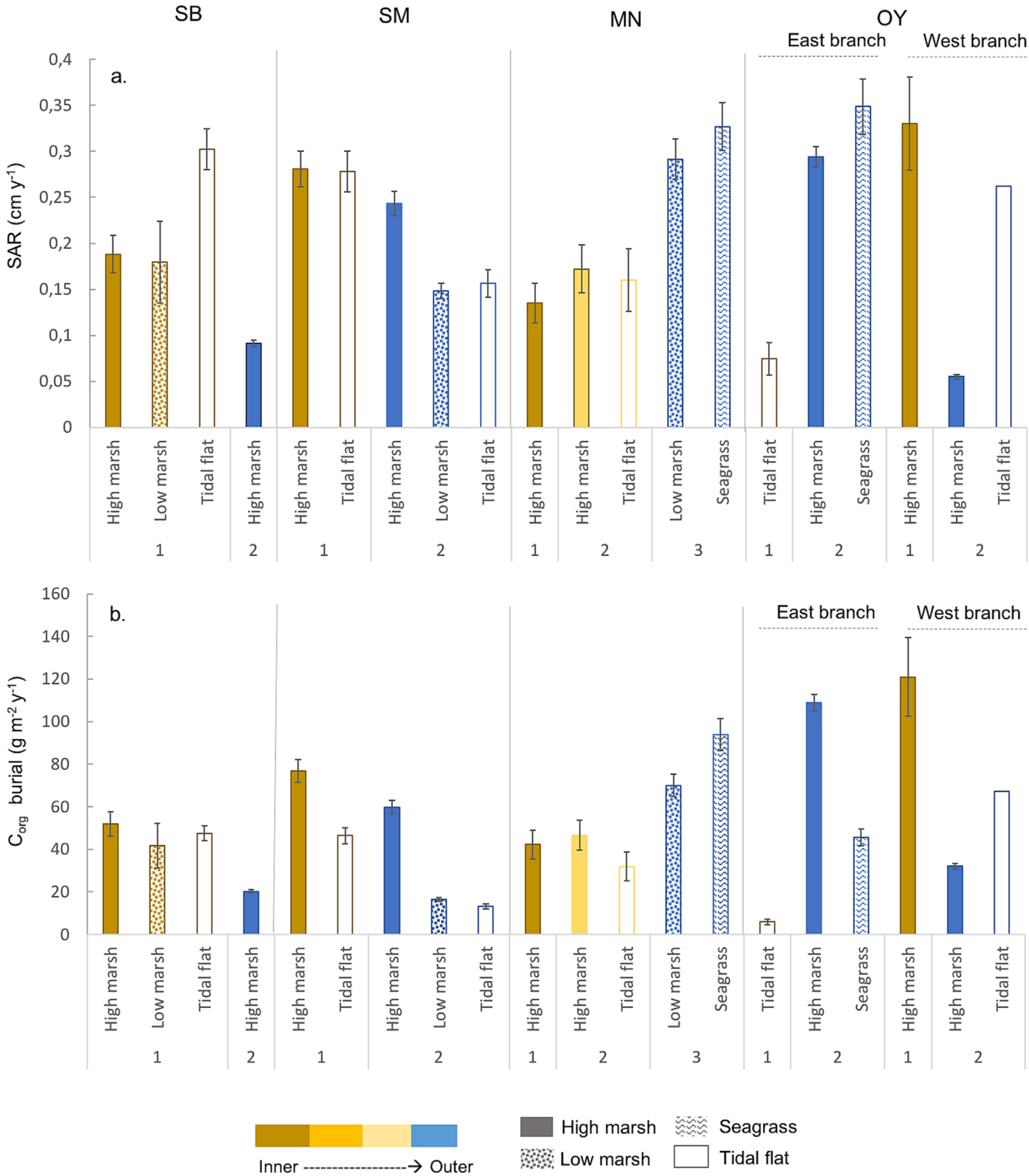


Fig. 6. Mean ± SE a) sediment accretion rate (SAR cm y^{-1}) and b) C_{org} burial rate (g m^{-2} y^{-1}) since 1960 estimated per habitat (n = 1 core) within each section per estuary (SB: Santander Bay; SM: Santoña Marshes; MN: Mondego estuary; OY: Oyambre). Number above the bars indicates the sampling area within each estuary. Color gradient indicates relative position along the estuary (from inner to outer sections) while bar fill patterns represent different habitats.

(Vanderbruwaene et al., 2015), which favors the deposition of more fine organic and inorganic particles at upper intertidal ranges compared to lower intertidal areas (Santos et al., 2019).

Despite remarkable differences in vegetation biomass, low marshes, seagrass meadows and tidal flats had similar soil C_{org} stocks. This result points to other factors in addition to autochthonous productivity driving C_{org} storage in these habitats, such as the efficiency at retaining and burying plant detritus and the accumulation of allochthonous C_{org}. Saltmarshes located at the low marsh zone are low efficient at retaining autochthonous C_{org} since a significant fraction of their biomass is exported by tides to adjacent habitats (Bouchard and Lefeuvre, 2000). On the other hand, seagrass meadows and tidal flats are located at lower elevation zones with regard to MSL compared to low marshes, and thus, are subject to longer inundation periods that favor a higher deposition of organic particles (i.e. allochthonous C_{org}) from the water column compared to low marshes. In addition, the lower intertidal habitats (i.e. seagrass meadows and tidal flats) can accumulate organic matter from adjacent intertidal habitats (e.g. low marshes) as tides favor the redistribution of organic particles

Table 3
Results of Spearman correlation analysis between the biogeochemical variables studied.

Correlation	Spearman ρ	p-value
C_{org} stocks vs. silt & clay %	0.63	<0.0001
% Silt & clay vs. SAR	0.41	0.09
% Silt & clay vs. C_{org} burial	0.41	0.09
C_{org} stocks vs. % forest land and natural areas	0.37	<0.0001
C_{org} stocks vs. % artificial areas	−0.34	<0.0001
C_{org} stocks vs. % agricultural land	−0.36	<0.0001

across adjacent habitats (Bulmer et al., 2020; Asplund et al., 2021). Finally, despite depicted of vegetation, tidal flats can accumulate C_{org} from microphytobenthos which can contribute to the formation of similar soil C_{org} stocks to those found in adjacent vegetated ecosystems, such as intertidal seagrass meadows (Santos et al., 2019; Bulmer et al., 2020).

SAR was expected to increase from habitats located at the lowest intertidal zone (i.e. seagrasses and tidal flats) towards those located at upper intertidal zones (i.e. high marshes) due to the longer and more frequent inundation periods and greater flooding depths in low intertidal habitats, that would allow for a greater deposition of particles from the water column. For instance, SAR has been reported to be higher in low marsh compared to high marsh communities (Temmerman et al., 2003; Chmura, 2004) and in seagrass meadows compared to low marshes (Martins et al., 2022). However the larger vegetation biomass of high marsh communities favor a higher accumulation of plant detritus and below ground biomass that also contribute to sediment vertical accretion (Kelleway et al., 2017). In addition, the denser and stiffer canopies formed by high marshes can create a more depositional environment (Peralta et al., 2008) as suggested by the higher content of silt and clay found in high marshes compared to lower intertidal habitats. In low marshes, subject to longer and more frequent inundation periods than high marshes, canopy attenuate the flow energy but can cause an increase in turbulent kinetic energy and in shear stress, reducing the potential for sediment accumulation and enhancing organic matter export to adjacent habitats (Widdows et al., 2008; Bouchard and Lefeuvre, 2000). As a consequence, SAR can also be higher in high marsh communities compared to lower intertidal habitats (e.g. low marshes), as reported in Saintilan et al. (2013) and Kelleway et al. (2017). In the habitats examined in this study, SAR was comparable across habitats, suggesting that perhaps, the decreasing vegetation biomass and increasing hydroperiods from high marshes to lower intertidal habitats (i.e. seagrasses and tidal flats) have a balancing effect on SAR.

The decreasing trend in soil C_{org} burial rate from relatively higher intertidal habitats (e.g., high marshes) to lower intertidal habitats contradicts previous findings by Ouyang and Lee (2014), that reported higher C_{org} burial rates in low marsh communities than in high marsh communities at a global scale driven by a higher SAR in the former. Our finding also differs to those reported in Martins et al. (2022), which found similar soil C_{org} burial rates between low marsh communities and seagrass meadows, despite higher stocks in saltmarsh communities, due to higher SAR in seagrass meadows. In our study, the differences in soil C_{org} burial rates are directly linked to differences in soil C_{org} stocks as no significant differences in SAR across habitats were found. In fact, cumulative soil C_{org} stocks since 1960 of the cores that could be dated tended to decrease from high marsh to tidal flats, with low marshes showing intermediate stocks (Fig. SI 3).

4.2. Variability within estuaries

We expected to find a decrease in soil C_{org} stocks, the content of silt and clay, SAR and C_{org} burial from upper estuary sections to lower estuary sections due to a higher accumulation of silt and clay contributing to C_{org} preservation (Mayer, 1994; Burdige, 2007) and greater allochthonous C_{org} deposited from rivers (Saintilan et al., 2013; Ricart et al., 2020). Yet, this was only observed in SB and in the west branch of OY estuary, which points at other abiotic factors not considered in this study, acting at different estuarine sections, as key drivers of the carbon sink capacity of intertidal estuarine ecosystems.

In the WS, the inner and outer sampling areas (sampling areas 1 and 4; Fig. 1) are sheltered from the prevailing southwestern winds, whereas the mid sampling areas (sampling areas 2 and 3; Fig. 1) are highly exposed to the southwest wind fetch (Callaghan et al., 2010). These differences in the exposure to wind-driven waves likely interfere with the effect of the river as a source of particles, and results in a lack of a clear decreasing gradient in particles deposition.

In SM, the expected trends were disrupted by the outer sampling area, a sheltered location where tidal regime is partially restricted through a dyke. This area also showed similar C_{org} stocks and silt and clay content to those found in the most inner estuarine area sampled likely due its particular depositional conditions (Figs. 3, 4). Yet, when considering the other two sampling areas in this estuary, the expected decrease in C_{org} stocks, the content of silt and clay and SAR and C_{org} burial rates from inner to outer estuarine sections is evident.

In the MN estuary, the outer section showed similar soil C_{org} stocks but higher SAR and soil C_{org} burial rates compared to the inner sections. The reason might be related to the specific conditions of MN outer section, where flow velocity is significantly lower compared to the other estuarine sections sampled due to human interventions (Ascione Kenov et al., 2012), likely allowing for a relatively higher deposition of particles from the water column and a lower export of plant biomass.

The lack of a decreasing trend in soil C_{org} stocks, content of silt & clay, SAR and C_{org} burial rates from the inner to the outer sections of the east branch of OY estuary could be attributed to the fact that the inner section was a low salinity lagoon for >100 yrs. until 2009, when a tidal barrier was removed. Although the restoration of the natural tidal regime allowed saltmarsh communities to develop (Frau et al., 2014), it is likely that more time is required for these restored saltmarshes to develop comparable soil C_{org} stocks to natural ones (Burden et al., 2019).

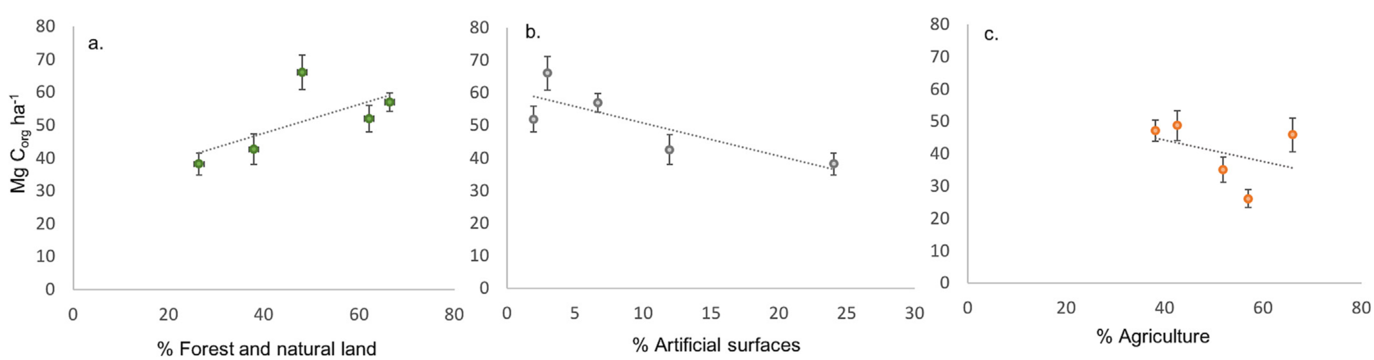


Fig. 7. Relationship between soil organic carbon stocks (Mg C_{org} ha^{-1}) and percentage of a) forest and natural land; b) artificial surfaces; and c) agricultural land within the estuarine basin.

4.3. Variability across estuaries

Differences in soil C_{org} stocks across estuaries seem to respond to differences in land uses within the river basins, with larger stocks being positively correlated with the percentage of forest and natural land and negatively correlated with higher percentage of artificial areas (i.e., higher urbanization) and agricultural land. The influence of land uses at the catchment level in the C_{org} storage of estuarine habitats is likely driven by differences in the amount and typology of suspended sediments transported by rivers (Peck et al., 2020), that are known to be strongly related to land use changes (Lambert et al., 2017; Sun et al., 2017). For instance, a higher presence of agricultural land can reduce the amount of POC transported by rivers compared to forest land due to a decrease in runoff associated to water extraction for irrigation (Sun et al., 2017). On the other hand, higher presence of urban land commonly alters fluvial organic matter fluxes towards a more microbial/algal and less plant/soil-derived character (Lambert et al., 2017), which is more reactive and subject to remineralization and thereby, leading to more labile and poor soil C_{org} deposits (Enríquez et al., 1993).

Previous studies revealed different trends in the variability of soil C_{org} deposits in coastal ecosystems in response to land use patterns of the basins. Young et al. (2021) reported a positive relationship between the degree of human alteration of the catchment and soil C_{org} deposits in Blue Carbon ecosystems across Australia, and attributed it to the increase in sedimentation of organic particles in coastal areas derived from different human activities (e.g., deforestation). On the contrary, Mazarrasa et al. (2021) found that human alteration of the catchment had a negative effect in seagrass soil C_{org} stocks temperate estuarine meadows (the environment most similar to the estuaries examined in this study) but neutral in temperate coastal and tropical estuarine meadows.

The results found in this and previous studies suggest that differences in land uses drive variability in soil C_{org} deposits across estuaries. However, the different results found across studies points to the need to further explore these relationships including a broader spectrum of estuaries and considering other factors that influence the sediment transport from land to coastal ecosystems (e.g., rainfall patterns, river flow) and that cause synergistic and antagonistic interactions with land use. These results also reinforce the concept that processes occurring upland are relevant for the management of Blue Carbon ecosystems (Macreadie et al., 2017). Preserving and restoring natural land cover as opposed to urban sprawling along rivers is one of the key measures needed to improve the ecological status of European rivers (Grizzetti et al., 2017). According to the results found in this study, increasing and conserving natural areas along river course would also have a positive impact in the conservation of the carbon sequestration service provided by coastal and estuarine ecosystems.

4.4. Contribution to the global Blue Carbon inventory

Our study contributes to increase the knowledge about the role estuarine habitats play as C_{org} sinks by providing new data from underrepresented habitats (e.g. tidal flats) and from an underrepresented region in Blue Carbon science (the Atlantic coast of Europe). This study increases the scarce data available on SAR and soil C_{org} burial associated to the small and deciduous seagrass species *Zostera noltei*, that dominates the low intertidal of European estuaries and is presently recovering in many locations. This study also increases the scarce data on soil C_{org} storage available for unvegetated tidal flats. Similar to previous studies (Phang et al., 2015; Bulmer et al., 2020), this study found similar stocks per surface area across tidal flats and intertidal seagrass meadows and reports for the first time, similar stocks per surface area across low marshes and tidal flats. When considering the total surface area occupied by each habitat per estuary examined, unvegetated tidal flats stored significantly higher soil C_{org} stocks across the habitats examined in three of the estuaries of study (SB, SM and MN) and similar to those found in high marshes for the other two (OY, WS) due to the large surface area tidal flats occupied at the estuarine scale (Table 4). Although C_{org} stored in tidal flats have an allochthonous origin, it is sequestered and buried in the soil of this habitat instead of being exported and potentially remineralized somewhere else. Thus, tidal flats act somehow as a carbon sink and reservoir. These results reinforce the idea of tidal flats being significant Blue Carbon deposits despite traditionally neglected in Blue Carbon literature. Saltmarsh communities (high marshes and low marshes) store an average of 55 ± 4 Mg C_{org} ha^{-1} within the top 30 cm of the soil, lower than stocks found in top 30 cm soil of Australian saltmarshes (82 ± 5 Mg C_{org} ha^{-1}; Macreadie et al., 2017). Comparing our results of top 30 cm soil C_{org} stocks with other saltmarshes in Europe and at a global scale is difficult as previous studies are based on a broader range of soil depths, ranging from the top 5 cm (Santos et al., 2019) to >1 m depth (Chmura et al., 2003; Ouyang and Lee, 2014; Santos et al., 2019). SAR and C_{org} burial rates in the saltmarsh communities examined here (0.22 ± 0.03 cm yr^{-1} and 0.61 ± 0.12 Mg C_{org} ha^{-1} yr^{-1}, respectively) are comparable to those estimated for Australian saltmarshes (0.20 cm y^{-1} and 0.55 Mg C_{org} ha^{-1} y^{-1}) (Macreadie et al., 2017). However, the SAR found in the saltmarshes of this study are lower than estimates in other Mediterranean saltmarshes (0.35 ± 0.06 cm y^{-1}; Fennessy et al., 2019) and both SAR and soil C_{org} burial rates are about half of average SAR and C_{org} burial rates in

Table 4
Summary of mean ± SE soil organic carbon (C_{org}) stocks per unit area within the top 30 cm soil in high marsh, low marsh, seagrass and tidal flat habitats, and total soil C_{org} stocks within the top 30 cm soil within each habitat and across the estuaries studied. MN: Mondego estuary, SB: Santander Bay, SM: Santoña Marshes, OY: Oyambre estuary, WS: Western Scheldt.

Estuary	Row labels	Top 30 cm stocks per surface area		Surface area occupied by habitat (Ha)	Total top 30 cm stocks	
		Mg C_{org} ha^{-1}	SE		Mg C_{org}	SE
BS	High marsh	63.09	8.91	13.91	877.58	124.01
	Low marsh	36.31	7.70	6.04	219.16	46.49
	Seagrass	27.46	6.23	189.24	5195.75	1179.9
	Tidal flat	37.31	5.53	1120.51	41,803.74	6191.57
SM	High marsh	66.36	9.50	58.84	3904.21	558.72
	Low marsh	42.33	6.68	263.80	11,167.37	1761.09
	Seagrass	41.59	6.97	278.00	11,561.99	1936.98
	Tidal flat	52.26	2.60	527.57	27,568.96	1373.71
MN	High marsh	69.35	7.21	103.80	7199.03	747.99
	Low marsh	53.43	5.14	15.49	827.68	79.62
	Seagrass	59.47	10.79	27.90	1659.20	301.11
	Tidal flat	51.92	2.95	216.48	11,240.01	639.10
OY	High marsh	81.81	5.97	38.52	3151.10	230.05
	Seagrass	51.37	6.35	4.00	205.48	25.38
	Tidal flat	49.85	7.52	28.21	1406.29	212.17
WS	High marsh	48.47	3.95	2110.00	102,268.41	8333.87
	Low marsh	29.68	4.89	259.00	7687.10	1266.16
	Tidal flat	36.27	7.06	100.00	3627.34	706.02

European saltmarshes (0.75 cm ± 0.1 cm y^{-1} and 3.12 Mg C_{org} ha^{-1} y^{-1}, respectively) and at global scale (0.67 cm ± 0.07 cm y^{-1} and 2.45 Mg C_{org} ha^{-1} y^{-1} respectively; (Duarte et al., 2013; Ouyang and Lee, 2014). The lower average SAR found compared to the recent estimates in Mediterranean saltmarshes could be explained by differences in geomorphological characteristics across sites. The saltmarshes of our study are located in Atlantic macrotidal estuaries whereas saltmarshes studied in Fennessy et al. (2019) are located in a Mediterranean delta system (i.e., Ebro Delta) under a microtidal regime where hydrodynamic processes allow a higher deposition of inorganic and organic particles (Jiménez et al., 1997). In addition, the lower average SAR and C_{org} burial rates found in the saltmarshes of this study compared to previous European and global assessments might be due to an uneven distribution of data across high marshes and low marshes in the latter. In our study, similar to the study by Macreadie et al. (2017) on Australian saltmarshes, most of the saltmarshes SAR and C_{org} burial rates obtained (9 out of 12) correspond to high marsh communities, whereas in previous regional and global assessments most of the data correspond to low marsh communities (Ouyang and Lee, 2014) where the longer and more frequent inundation periods can allow for higher sedimentation from the water column and as a consequence, higher SAR (Chmura, 2004). These results highlight the need to increase the data available on soil C_{org} stocks, SAR and C_{org} burial in different saltmarsh communities from those regions still underrepresented in global assessments, in order to produce more reliable estimates of saltmarsh carbon sinks at regional and global scale.

Estuaries are dynamic systems subject to intense human pressures that can trigger significant habitats loss or degradation. Knowing the C_{org} stocks, SAR and C_{org} burial rates across and within estuarine habitats is critical in order to predict how ecosystem loss or distribution shifts can affect the carbon budget at the estuarine scale. This is particularly relevant in Europe, where approximately 50 % of the original surface of coastal wetlands has been lost due to human impacts (Airoldi and Beck, 2007) and where the restoration of these ecosystems is being encouraged by different climate and environmental policies (EU Biodiversity Strategy; EU Strategy on Adaptation to Climate Change). In addition, identifying management practices that enhance Blue Carbon sequestration by coastal ecosystems is key to develop climate change mitigation strategies that consider these ecosystems. The results of this study significantly contribute to the knowledge on the magnitude and variability of Blue Carbon stocks and burial rates in European estuaries and identify the preservation of natural areas upland as a key strategy to protect and enhance carbon sequestration in estuarine ecosystems.

In memory of Jordi Garcia-Orellana, who left us during the preparation of this manuscript and whose ideas, motivation and help always made this job easy and fun.

CRediT authorship contribution statement

Inés Mazarrasa contributed with study conceptualization; field and laboratory work, formal analysis and writing of the original draft; Joao Neto, Tjeerd Bouma and Tim Grandjean contributed with study conceptualization and field and laboratory work, Jordi Garcia-Orellana and Pere Masqué contributed with biogeochemical analysis (sediment dating); María Recio contributed with funding acquisition, fieldwork and formal analysis, Óscar Serrano contributed with formal analysis and editing and Araceli Puente and José Juanes contributed with funding acquisition and project management. All co-authors contributed with manuscript reviewing and editing.

Data availability

Data will be made available on request.

Declaration of competing interest

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

Acknowledgements

The authors acknowledge the financial support from the Government of Cantabria through the "Fénix Programme". IM was supported by a Juan de la Cierva Incorporación postdoctoral fellowship of the Spanish Ministry of Science, Innovation and Universities (JC2020-045917-I). OS was supported by I+D+i projects RYC2019-027073-I and PIE HOLOCENO 20213AT014 funded by the Spanish Ministry of Science, Innovation and Universities (MCIN/AEI/10.13039/501100011033) and FEDER. The IAEA is grateful for the support provided to its Marine Environment Laboratories by the Government of the Principality of Monaco. PM was supported by the Australian Research Council. J.G.O. acknowledges the financial support of the Spanish Ministry of Science, Innovation and Universities, through the "Maria de Maeztu" program for Units of Excellence (CEX2019-000940-M) and the Generalitat de Catalunya (MERS; 2017 SGR–1588).

This research was carried out with the contribution of the LIFE Programme of the European Union to the Project ADAPTA BLUES (ref. LIFE18 CCA/ES/001160). This document reflects only the authors' view and the Agency/Commission is not responsible for any use that may be made of the information it contains.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.163957.

References

Adam, P., 2002. Saltmarshes in a time of change. Environ. Conserv. 29, 39–61. https://doi.org/10.1017/S0376892902000048.

Airoldi, L., Beck, M.W., 2007. Loss, status and trends for coastal marine habitats of Europe. Oceanogr. Mar. Biol. 45, 345–405. https://doi.org/10.1201/9781420050943.ch7.

do Amaral Camara Lima, M., Ward, R.D., Joyce, C.B., Kauer, K., 2022. Carbon stocks in southern England's intertidal seagrass meadows. Estuar. Coast. Shelf Sci. https://doi.org/10.1016/j.ecss.2022.107947.

Appleby, P.G., Oldfield, F., 1978. The calculation of lead-210 dates assuming a constant rate of supply of unsupported 210Pb to the sediment Es wird eine Methode zur Berechnung des Sedimentalters beschrieben , die auf 210pb. Catena 5, 1–8.

Ascione Kenov, I., Garcia, A.C., Neves, R., 2012. Residence time of water in the Mondego estuary (Portugal). Estuar. Coast. Shelf Sci. 106, 13–22. https://doi.org/10.1016/j.ecss.2012.04.008.

Asplund, M.E., Dahl, M., Ismail, R.O., et al., 2021. Dynamics and fate of blue carbon in a mangrove–seagrass seascape: influence of landscape configuration and land-use change. Landsc. Ecol. 36, 1489–1509. https://doi.org/10.1007/s10980-021-01216-8.

Barañano, C., Fernández, E., Méndez, G., 2018. Clam harvesting decreases the sedimentary carbon stock of a Zostera marina meadow. Aquat. Bot. 146, 48–57. https://doi.org/10.1016/j.aquabot.2017.12.002.

Barbier, E.B., Hacker, S.D., Kennedy, C., et al., 2011. The value of estuarine and coastal ecosystem services. Ecol. Monogr. 81, 169–193.

Bouchard, V., Lefeuvre, J.-C., 2000. Primary Production and Macro-detritus Dynamics in a European Salt Marsh: Carbon and Nitrogen Budgets. Aquat. Bot. 67, 23–42.

Bulmer, R.H., Stephenson, F., Jones, H.F.E., Townsend, M., Hillman, J.R., Schwendenmann, L., Lundquist, C.J., 2020. Blue carbon stocks and cross-habitat subsidies. Front. Mar. Sci. 7. https://doi.org/10.3389/fmars.2020.00380.

Burden, A., Garbutt, A., Evans, C.D., 2019. Effect of restoration on saltmarsh carbon accumulation in Eastern England. Biol. Lett. 15, 0–3. https://doi.org/10.1098/rsbl.2018.0773.

Burdige, D.J., 2007. Preservation of organic matter in marine sediments: controls, mechanisms, and an imbalance in sediment organic carbon budgets? Chem. Rev. 107, 467–485. https://doi.org/10.1021/cr050347q.

Cahoon, D.R., Reed, D.J., 1995. Relationships among marsh surface topography, hydroperiod, and soil accretion in a deteriorating Louisiana salt marsh. J. Coast. Res. 11, 357–369.

Callaghan, D.P., Bouma, T.J., Klaassen, P., van der Wal, D., Stive, M.J.F., Herman, P.M.J., 2010. Hydrodynamic forcing on salt-marsh development: distinguishing the relative importance of waves and tidal flows. Estuar. Coast. Shelf Sci. 89, 73–88. https://doi.org/10.1016/j.ecss.2010.05.013.

Canadell, J.G., Le Quéré, C., Raupach, M.R., et al., 2007. Contributions to accelerating atmospheric CO2 growth from economic activity, carbon intensity, and efficiency of natural sinks. Proc. Natl. Acad. Sci. U. S. A. 104, 18866–18870. https://doi.org/10.1073/pnas.0702737104.

Chmura, G.L., 2004. Controls on Salt Marsh Accretion : A Test in Salt Marshes of Eastern Canada. 27 pp. 70–81.

Chmura, G.L., Anisfeld, S.C., Cahoon, D.R., Lynch, J.C., 2003. Global carbon sequestration in tidal, saline wetland soils. Glob. Biogeochem. Cycles 17. https://doi.org/10.1029/2002gb001917.

Dahl, M., Deyanova, D., Gütschow, S., et al., 2016. Sediment properties as important predictors of carbon storage in Zostera marina meadows: a comparison of four European areas. PLoS One 11, 1–21. https://doi.org/10.1371/journal.pone.0167493.

Duarte, C.M., Losada, I.J.I.J., Hendriks, I.E., Mazarrasa, I., Marbà, N., 2013. The role of coastal plant communities for climate change mitigation and adaptation. Nat. Clim. Chang. 3, 961–968. https://doi.org/10.1038/nclimate1970.

Elschot, K., Bakker, J.P., Temmerman, S., Van De Koppel, J., Bouma, T.J., 2015. Ecosystem engineering by large grazers enhances carbon stocks in a tidal salt marsh. Mar. Ecol. Prog. Ser. 537, 9–21. https://doi.org/10.3354/meps11447.

Enríquez, S., Duarte, C.M., Sand-Jensen, K., 1993. Patterns in decomposition rates among photosynthetic organisms: the importance of detritus C:N:P content. Oecologia 94, 457–471. https://doi.org/10.1007/BF00566960.

Ewers Lewis, C.J., Young, M.A., Ierodiaconou, D., Baldock, J.A., Hawke, B., Sanderman, J., Carnell, P.E., Macreadie, P.I., 2020. Drivers and modelling of blue carbon stock variability in sediments of southeastern Australia. Biogeosciences 17, 2041–2059. https://doi.org/10.5194/bg-17-2041-2020.

Feio, M.J., Reynoldson, T.B., Ferreira, V., Graça, M.A.S., 2007. A predictive model for freshwater bioassessment (Mondego River, Portugal). Hydrobiologia 589, 55–68. https://doi.org/10.1007/s10750-006-0720-0.

Fennessy, M.S., Ibánez, C., Calvo-Cubero, J., Sharpe, P., Rovira, A., Callaway, J., Caiola, N., 2019. Environmental controls on carbon sequestration, sediment accretion, and elevation change in the Ebro River Delta: implications for wetland restoration. Estuar. Coast. Shelf Sci. 222, 32–42. https://doi.org/10.1016/j.ecss.2019.03.023.

Frau, D., Eizaguirre, B.O., Arbeiza, C.G., De La Peña, J.A.J., 2014. The role of the hydrodynamic regime in the distribution of the invasive shrub Baccharis halimifolia (Compositae) in Oyambre Estuary (Cantabria, Spain). Limnetica 33, 1–12.

Goemans, G., Roosens, L., Dirtu, A.C., Belpaire, C., Neels, H., Covaci, A., 2007. Brominated Flame Retardants in Eel From the Scheldt River (Belgium) L7. 69 pp. 445–448.

Gorham, C., Lavery, P., Kelleway, J.J., Salinas, C., Serrano, O., 2021. Soil carbon stocks vary across geomorphic settings in Australian temperate tidal marsh ecosystems. Ecosystems 24, 319–334. https://doi.org/10.1007/s10021-020-00520-9.

Gorman, D., Sumida, P.Y.G., Figueira, R.C.L., Turra, A., 2020. Improving soil carbon estimates of mudflats in Araçá Bay using spatial models that consider riverine input, wave exposure and biogeochemistry. Estuar. Coast. Shelf Sci. 238, 106734. https://doi.org/10.1016/j.ecss.2020.106734.

Grizzetti, B., Pistocchi, A., Liquete, C., Udias, A., Bouraoui, F., Van De Bund, W., 2017. Human pressures and ecological status of European rivers. Sci. Rep. 7, 1–11. https://doi.org/10.1038/s41598-017-00324-3.

Herr, D., Landis, E., 2016. Coastal blue carbon ecosystems. Opportunities for Nationally Determined Contributions. Policy Brief.

Herr, D., von Unger, M., Laffoley, D., McGivern, A., 2017. Pathways for implementation of blue carbon initiatives. Aquat. Conserv. Mar. Freshwat. Ecosyst. 27, 116–129. https://doi.org/10.1002/aqc.2793.

IHCantabria, 2005. Estudio de los recursos hídricos de la vertiente Norte de Cantabria. Gobierno de Cantabria.

Jiménez, J.A., Sánchez-Arcilla, A., Valdemoro, H.I., Gracia, V., Nieto, F., 1997. Processes reshaping the Ebro delta. Mar. Geol. 144, 59–79. https://doi.org/10.1016/S0025-3227(97)00076-5.

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, 865–880. https://doi.org/10.1007/s10021-016-9972-3.

Kelleway, J.J., Saintilan, N., Macreadie, P.I., Baldock, J.A., Ralph, P.J., 2017. Sediment and carbon deposition vary among vegetation assemblages in a coastal salt marsh. Biogeosciences 14, 3763–3779. https://doi.org/10.5194/bg-14-3763-2017.

Kirwan, M.L., Murray, A.B., Donnelly, J.P., Corbett, D.R., 2011. Rapid wetland expansion during European settlement and its implication for marsh survival under modern sediment delivery rates. Geology 39, 507–510. https://doi.org/10.1130/G31789.1.

Krause-Jensen, D., Lavery, P., Serrano, O., Marba, N., Masque, P., Duarte, C.M., 2018. Sequestration of macroalgal carbon: the elephant in the Blue Carbon room. Biol. Lett. 14. https://doi.org/10.1098/rsbl.2018.0236.

Krishnaswamy, S., Lal, D., Martin, J.M., Meybeck, M., 1971. Geochronology of lake sediments. Earth Planet. Sci. Lett. 11, 407–414.

Lambert, T., Bouillon, S., Darchambeau, F., Morana, C., Roland, F.A.E., Descy, J.P., Borges, A.V., 2017. Effects of human land use on the terrestrial and aquatic sources of fluvial organic matter in a temperate river basin (the Meuse River, Belgium). Biogeochemistry 136, 191–211. https://doi.org/10.1007/s10533-017-0387-9.

Lovelock, C.E., Fourqurean, J.W., Morris, J.T., 2017. Modeled CO2 emissions from coastal wetland transitions to other land uses: tidal marshes, mangrove forests, and seagrass beds. Front. Mar. Sci. 4. https://doi.org/10.3389/fmars.2017.00143.

Lovelock, C.E., Adame, M.F., Bradley, J., et al., 2022. An Australian blue carbon method to estimate climate change mitigation benefits of coastal wetland restoration. Restor. Ecol. https://doi.org/10.1111/rec.13739.

Luisetti, T., Jackson, E.L., Turner, R.K., 2013. Valuing the European “coastal blue carbon” storage benefit. Mar. Pollut. Bull. 71, 101–106. https://doi.org/10.1016/j.marpolbul.2013.03.029.

Macreadie, P.I., Ollivier, Q.R., Kelleway, J.J., Serrano, O., Carnell, P.E., 2017. Carbon sequestration by Australian tidal marshes. Nat. Publ. Gr. 1–10. https://doi.org/10.1038/srep44071.

Macreadie, P.I., Anton, A., Raven, J.A., et al., 2019. The future of Blue Carbon science. Nat. Commun. 10, 1–13. https://doi.org/10.1038/s41467-019-11693-w.

Martins, M., de los Santos, C.B., Masqué, P., Carrasco, A.R., Veiga-Pires, C., Santos, R., 2022. Carbon and nitrogen stocks and burial rates in intertidal vegetated habitats of a mesotidal coastal lagoon. Ecosystems 25, 372–386. https://doi.org/10.1007/s10021-021-00660-6.

Mateo, M., Romero, J., Pérez, M., Littler, M., Littler, D.S., 1997. Dynamics of millenary organic deposits resulting from the growth of the Mediterranean seagrass Posidonia oceanica. Estuar. Coast. Shelf Sci. 43, 103–110. https://doi.org/10.1006/ECSS.1996.0116.

Mayer, L.M., 1994. Relationship between mineral surfaces and organic carbon concentrations in soils and sediments. Chem. Geol. 114, 347–363.

Mazarrasa, I., Marbà, N., Garcia-Orellana, J., Masqué, P., Arias-Ortiz, A., Duarte, C.M., 2017. Effect of environmental factors (wave exposure and depth) and anthropogenic pressure in the C sink capacity of Posidonia oceanica meadows. Limnol. Oceanogr. 62, 1436–1450. https://doi.org/10.1002/lno.10510.

Mazarrasa, I., Lavery, P., Duarte, C.M., et al., 2021. Factors determining seagrass blue carbon across bioregions and geomorphologies. Glob. Biogeochem. Cycles 35, 1–17. https://doi.org/10.1029/2021gb006935.

McLeod, E., Chmura, G.L., Bouillon, S., et al., 2011. A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO2. Front. Ecol. Environ. 9, 552–560. https://doi.org/10.1890/110004.

Meyers, P.A., 1994. Preservation of elemental and isotopic source identification of sedimentary organic matter. Chem. Geol. 114, 289–302. https://doi.org/10.1016/0009-2541(94)90059-0.

Milliman, J.D., 2001. Delivery and fate of fluvial water and sediment to the sea : a marine geologist ' s view of European rivers *. Sci. Mar. 65, 121–131.

Milliman, J.D., Syvitski, J.P.M., 1992. Geomorphic/tectonic control of sediment discharge to the ocean: the importance of small mountainous rivers. J. Geol. 100, 525–544. https://doi.org/10.1086/629606.

Nellemann, C., Duarte, C.M., Valdés, L., De Young, C., Fonseca, L., Grimsditch, G., 2009. In: Program, U.N.E. (Ed.), Blue Carbon. The Role of Healthy Oceans in Binding Carbon. Birkeland Trykkeri AS.

Ouyang, X., Lee, S.Y., 2014. Updated estimates of carbon accumulation rates in coastal marsh sediments. Biogeosciences 11, 5057–5071. https://doi.org/10.5194/bg-11-5057-2014.

Peck, E.K., Wheatcroft, R.A., Brophy, L.S., 2020. Controls on sediment accretion and blue carbon burial in tidal saline wetlands: insights from the Oregon Coast, USA. J. Geophys. Res. Biogeosci. 125. https://doi.org/10.1029/2019JG005464.

Peralta, G., Van Duren, L.A., Morris, E.P., Bouma, T.J., 2008. Consequences of shoot density and stiffness for ecosystem engineering by benthic macrophytes in flow dominated areas: a hydrodynamic flume study. Mar. Ecol. Prog. Ser. 368, 103–115. https://doi.org/10.3354/meps07574.

Phang, V.X.H., Chou, L.M., Friess, D.A., 2015. Ecosystem carbon stocks across a tropical intertidal habitat mosaic of mangrove forest, seagrass meadow, mudflat and sandbar. Earth Surf. Process. Landf. 40, 1387–1400. https://doi.org/10.1002/esp.3745.

Ricart, A.M., York, P.H., Bryant, C.V., Rasheed, M.A., Ierodiaconou, D., Macreadie, P.I., 2020. High variability of Blue Carbon storage in seagrass meadows at the estuary scale. Sci. Rep. 10, 1–12. https://doi.org/10.1038/s41598-020-62639-y.

Röhr, M.E., Boström, C., Canal-Vergés, P., Holmer, M., 2016. Blue carbon stocks in Baltic Sea eelgrass (Zostera marina) meadows. Biogeosciences 13, 6139–6153. https://doi.org/10.5194/bg-13-6139-2016.

Röhr, M.E., Holmer, M., Baum, J.K., et al., 2018. Blue carbon storage capacity of temperate eelgrass (Zostera marina) meadows. Glob. Biogeochem. Cycles 32, 1457–1475. https://doi.org/10.1029/2018GB005941.

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.

Santos, R., Duque-nú, N., Santos, C.B.D.L., Martins, M., 2019. Superficial sedimentary stocks and sources of carbon and nitrogen in coastal vegetated assemblages along a flow gradient. Sci. Rep. 1–12. https://doi.org/10.1038/s41598-018-37031-6.

Schuerch, M., Vafeidis, A., Slawig, T., Temmerman, S., 2013. Modeling the influence of changing storm patterns on the ability of a salt marsh to keep pace with sea level rise. J. Geophys. Res. Earth Surf. 118, 84–96. https://doi.org/10.1029/2012JF002471.

Serrano, O., Lovelock, C.E., Atwood, T.B., et al., 2019. Australian vegetated coastal ecosystems as global hotspots for climate change mitigation. Nat. Commun. 10, 1–10. https://doi.org/10.1038/s41467-019-12176-8.

Soares-Filho, B., Moutinho, P., Nepstad, D., et al., 2010. Role of Brazilian Amazon protected areas in climate change mitigation. Proc. Natl. Acad. Sci. U. S. A. 107, 10821–10826. https://doi.org/10.1073/pnas.0913048107.

Sousa, A.I., Santos, D.B., Da Silva, E.F., Sousa, L.P., Cleary, D.F.R., Soares, A.M.V.M., Lillebø, A.I., 2017. “Blue Carbon” and nutrient stocks of salt marshes at a temperate coastal lagoon (Ria de Aveiro, Portugal). Sci. Rep. 7, 1–11. https://doi.org/10.1038/srep41225.

Stoddart, D.R., Reed, D.J., French, J.R., Stoddart, D.R., Reed, D.J., 1989. Understanding saltmarsh accretion , Scolt Head Island , Norfolk , England. Estuaries 12, 228–236.

Sun, H., Han, J., Li, D., Lu, X., Zhang, H., Zhao, W., 2017. Organic carbon transport in the Songhua River, NE China: influence of land use. Hydrol. Process. 31, 2062–2075. https://doi.org/10.1002/hyp.11173.

Temmerman, S., Govers, G., Wartel, S., Meire, P., 2003. Spatial and temporal factors controlling short-term sedimentation in a salt and freshwater tidal marsh, scheldt estuary, Belgium, SW Netherlands. Earth Surf. Process. Landf. 28, 739–755. https://doi.org/10.1002/esp.495.

Temmink, R.J.M., Lamers, L.P.M., Angelini, C., et al., 2022. Recovering wetland biogeomorphic feedbacks to restore the world's biotic carbon hotspots. Science 80, 376. https://doi.org/10.1126/science.abn1479.

Trevathan-Tackett, S.M., Kelleway, J., Macreadie, P.I., Beardall, J., Ralph, P., Bellgrove, A., 2015. Comparison of marine macrophytes for their contributions to blue carbon sequestration. Ecology 96, 3043–3057. https://doi.org/10.1890/15-0149.1.

Trumper, K., Bertzky, M., Dickson, B., van Der Heijden, G., Jenkins, M., Manning, P., 2009. The Naturak Fix?The Role of Ecosystems in Climate Change Mitigation (A UNEP rapid response assessment).

Ullman, R., Bilbao-bastida, V., Grimsditch, G., 2013. Ocean & coastal management including Blue Carbon in climate market mechanisms. Ocean Coast. Manag. 83, 15–18. https://doi.org/10.1016/j.ocecoaman.2012.02.009.

Vanderbruwaene, W., Schwarz, C., Bouma, T.J., Meire, P., Temmerman, S., 2015. Landscape-scale flow patterns over a vegetated tidal marsh and an unvegetated tidal flat: implications for the landform properties of the intertidal floodplain. Geomorphology 231, 40–52.

Widdows, J., Pope, N.D., Brinsley, M.D., 2008. Effect of Spartina anglica stems on near-bed hydrodynamics, sediment erodability and morphological changes on an intertidal mudflat. Mar. Ecol. Prog. Ser. 362, 45–57. https://doi.org/10.3354/meps07448.

Young, M.A., Serrano, O., Macreadie, P.I., Lovelock, C.E., Carnell, P., Ierodiaconou, D., 2021. National scale predictions of contemporary and future blue carbon storage. Sci. Total Environ. 800, 149573. https://doi.org/10.1016/j.scitotenv.2021.149573.