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Research article

Sedimentary organic carbon and nitrogen stocks of intertidal seagrass meadows in a dynamic and impacted wetland: Effects of coastal infrastructure constructions and meadow establishment time

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ABSTRACT

Seagrass meadows, through their large capacity to sequester and store organic carbon in their sediments, contribute to mitigate climatic change. However, these ecosystems have experienced large losses and degradation worldwide due to anthropogenic and natural impacts and they are among the most threatened ecosystems on Earth. When a meadow is impacted, the vegetation is partial- or completely lost, and the sediment is exposed to the atmosphere or water column, resulting in the erosion and remineralisation of the carbon stored. This paper addresses the effects of the construction of coastal infrastructures on sediment properties, organic carbon, and total nitrogen stocks of intertidal seagrass meadows, as well as the size of such stocks in relation to meadow establishing time (recently and old established meadows). Three intertidal seagrass meadows impacted by coastal constructions (with 0% seagrass cover at present) and three adjacent non-impacted old-established meadows (with 100% seagrass cover at present) were studied along with an area of bare sediment and two recent-established seagrass meadows. We observed that the non-impacted areas presented 3-fold higher percentage of mud and 1.5 times higher sedimentary organic carbon stock than impacted areas. Although the impacted area was relatively small (0.05–0.07 ha), coastal infrastructures caused a significant reduction of the sedimentary carbon stock, between 1.1 and 2.2 Mg OC, and a total loss of the carbon sequestration capacity of the impacted meadow. We also found that the organic carbon stock and total nitrogen stock of the recent-established meadow were 30% lower than those of the old-established ones, indicating that OC and TN accumulation within the meadows is a continuous process, which has important consequences for conservation and restoration actions. These results contribute to understanding the spatial variability of blue carbon and nitrogen stocks in coastal systems highly impacted by urban development.

1. Introduction

Blue carbon, the carbon stored and sequestered in marine ecosystems, especially in vegetated coastal ecosystems such as saltmarshes, mangroves, and seagrass meadows, has received special attention in recent years (Nellemann et al., 2009; Macreadie et al., 2019). These ecosystems can remove large amounts of CO_2 from the atmosphere and store it in sediments for hundreds to thousands of years (Duarte et al., 2005). Currently, the amount of carbon stored by these ecosystems is an active area of research, although with many questions still to be resolved (Macreadie et al., 2019). Seagrass ecosystems are one of the most efficient carbon sinks on Earth, with rates of about 30 times faster than tropical rainforests (Serrano et al., 2011). Therefore, one of the key strategies for climate change mitigation is to conserve and to protect seagrass meadows (Duarte et al., 2013; Lovelock et al., 2017; Macreadie et al., 2021). In addition, seagrass ecosystems are also relevant to the global biogeochemical nitrogen cycle, playing a key role in removing the excess anthropogenic nitrogen in coastal areas (Jordan et al., 2011).

Seagrasses form meadows in sandy and muddy bottoms of coastal areas, both in the intertidal and subtidal zones. The coastal zone encompasses highly dynamic areas that have historically represented priority sites for human settlements and development (Halpern et al.,

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2019). Increasing population and its activity in the coastal zone, together with the effects of climate change (e.g., more frequent and severe storms, and sea level rise), are compromising the health of these ecosystems (Lotze, 2006). Seagrass meadows are currently among the most threatened ecosystems on Earth, with a loss of 19.1% of the surveyed meadow area occurring since 1880 (Dunic et al., 2021), yet there are locations with newly stabilished meadow and/or recovered ones once the disturbance has ceased (de los Santos et al., 2019; Dunic et al., 2021). Poor water quality and coastal development are among the most frequent drivers of seagrass loss (Waycott et al., 2009; de los Santos et al., 2019; Dunic et al., 2021). Dredging, anchoring and boating, shellfish digging and farming, and construction of coastal infrastructures are among those anthropogenic activities (Cabaço et al., 2005; Waycott et al., 2009; Macreadie et al., 2015; Serrano et al., 2020; Román et al., 2022). The result is a partial (or complete) loss of the seagrass standing stock, and the exposure of the disturbed sediment to the water column and air. This disturbance causes the carbon stored over centuries to be released back into the atmosphere (Pendleton et al., 2012; Lovelock et al., 2017).

Few studies have investigated the effect of physical disturbances on both, blue carbon and nitrogen stocks within seagrass meadows. For instance, Macreadie et al. (2014) did not detect changes in sedimentary organic carbon content when *Zostera nigracualis* meadows in Australia were subjected experimentally to small disturbances (representing grazers and anchor damage) over 24 months. However, a significant decrease of 72% in organic carbon stocks was detected in disturbed areas devoid of *Posidonia australis* for 50 years due to physical damage following seismic testing (Macreadie et al., 2015). Also, Thorhaug et al. (2017) estimated an average loss of 21 Mg OC ha^{-1} at 8 sites in the Gulf of Mexico because of dredging and Bourque et al. (2015) estimated that disturbance from vessel groundings resulted in losses of 60 Mg OC ha^{-1} and 4 Mg TN ha^{-1}. Román et al. (2022) also reported a reduction of 53–85% in the sedimentary organic carbon stock of *Z. noltei* meadows due to intensive clam harvesting. In summary, the existing studies point out that seagrass sedimentary organic carbon and nitrogen stocks will be affected to a greater or lesser extent depending on the intensity and type of the disturbance. Although small-scale infrastructures (e.g., pontoons, jetties) are very common in coastal zones, the impacts they may have on carbon and nitrogen stocks in seagrass meadows have not been assessed yet. In addition, coastal zones are also impacted naturally by sedimentary processes leading to dynamic burial and erosion processes affecting seagrass meadows (Peralta et al., 2005; Cunha and Santos, 2009). It is expected that newly established meadows (or patches) in disturbed areas store less organic carbon and nitrogen than long-established ones thriving in unaffected zones. Understanding the effects of both natural and human-driven impacts on seagrass blue carbon and nitrogen stocks is key for a better judgment of their spatial variability and how it translates into the modification of the coastal biochemical cycles.

The aim of this work was to assess the size of carbon and nitrogen stocks of intertidal seagrass meadows occurring in an area subjected to both, human and natural impacts. We selected several sites at the Ria Formosa, a coastal lagoon in South Portugal, which is a highly dynamic system in terms of sedimentary processes and subjected to many physical impacts. In our study, we selected a) meadows impacted by the construction of small-scale coastal infrastructures (e.g. pontoons), b) recently established meadows, c) old established meadows and d) a bare sediment area, to specifically investigate: 1) the effect of the construction of coastal infrastructures on sediment properties and on the organic carbon and total nitrogen stocks of intertidal seagrass meadows; and 2) the size of organic carbon and total nitrogen stocks in relation to the time of meadow establishment (recently and old established meadows). We hypothesized that both the impact of small-scale coastal constructions and the time of establishment of a meadow would influence sedimentary carbon and nitrogen stocks, being lower in impacted areas and in younger meadows than in non-impacted or older meadows, respectively.

2. Materials and methods

2.1. Study area and sampling sites

The Ria Formosa is a coastal mesotidal lagoon located in southern Portugal, extending over 55 km along the coast (Fig. 1A). It consists of seven sand barriers (two peninsulas and five islands) that are connected to the ocean through six inlets. Tides are semi-diurnal with average amplitudes between 1.3 and 2.8 m for neap and spring tides, respectively, and up to 3.5 m in maximum spring tides (Ferreira et al., 2016). Water circulation inside the lagoon is mostly driven by tides (Jacob et al., 2013) and terrestrial sediment supply is limited due to low and episodic riverine inflows into the lagoon (Arnaud-Fassetta et al., 2006). The Ria Formosa is characterized by a network of numerous channels and extensive tidal flats, which are intersected by a high density of shallow meandering tidal creeks. The backbarrier intertidal mudflats are largely colonised by monospecific meadows of the saltmarsh specie *Sporobolus maritimus* (Curtis) P.M. Peterson & Saarela and the seagrass *Zostera noltei* Horneman, the latter covering 45% of the total intertidal area (approximately 1304 ha, Guimarães et al., 2012). The subtidal areas are colonised by seagrass *Cymodocea nodosa* (Ucria) Ascherson and *Z. marina* L. and recently, also by extensive meadows of the seaweed *Caulerpa prolifera* (Forsskål) J.V. Lamouroux (Parreira et al., 2021).

The Ria Formosa lagoon is a highly dynamic system and its coastal vegetated bottoms are physically impacted by natural sedimentary processes such as inlet migration (e.g., Vila-Concejo et al., 2003; Cunha et al., 2005) and by intense anthropogenic activity, namely bivalve farms (Guimarães et al., 2012), clam digging (Cabaço et al., 2005), intense boat traffic, mooring, and anchoring, dredging of the navigation channels (Ferreira et al., 2016), artificial nourishment actions (Dias et al., 2003), opening and stabilisation of inlets (Peralta et al., 2005; Kombiadou et al., 2019), construction of coastal infrastructures (airport, ports, marinas, jetties, pontoons, etc.), and human settlements (Ceia, 2007).

Faro beach (Fig. 1B) is located at the Ancão peninsula (approximately 10 km long), the most western region of the Ria Formosa. It is an urban and touristic area highly developed since the 1960s, thus with a high human pressure, including the above-mentioned physical impacts on the backbarrier shore. The intertidal area facing the lagoon is mostly colonised by *Zostera noltei* in the intertidal fringe, interleaved with low-energy beaches. There are small-scale coastal infrastructures (i.e., in an area of tens of meters) constructed on the intertidal area, impacting directly the *Z. noltei* meadows, namely a bridge from mainland to the peninsula and two pontoons for small boats and ferries. The area is also subjected to natural sedimentary disturbances caused by storm events and inlet migration (Cunha et al., 2005; Cunha and Santos, 2009).

Five intertidal sites were selected along the central part of Faro beach based on aerial images from 1957 to 2019, encompassing: 1) three *Zostera noltei* meadows impacted by coastal constructions (currently having 0% seagrass cover) and three adjacent non-impacted old-established (for at least 14 years) *Z. noltei* meadows (currently having 100% seagrass cover) (ST1, ST2, ST3); 2) a bare sediment area (unvegetated for at least 14 years) in a low-energy beach to be used as control (ST4); and, 3) two recent-established (less than 5 years old) *Z. noltei* meadows (ST5) (Fig. 1C). Sites ST1, ST2 and ST3 were impacted due to the construction of a bridge in 1957, and the construction of two pontoons in 2007 (Fig. 1D), respectively. Based also on the aerial images, we estimated the areas (ha) impacted by these coastal constructions in the intertidal area, by comparing the images before and after the infrastructure construction.

2.2. Sediment sampling and analysis

A total of nine sediment cores were collected at the five sites in October and November 2021, during low tide. Two cores were taken at each ST1, ST2 and ST3 sites, one where the impacted meadow occurred

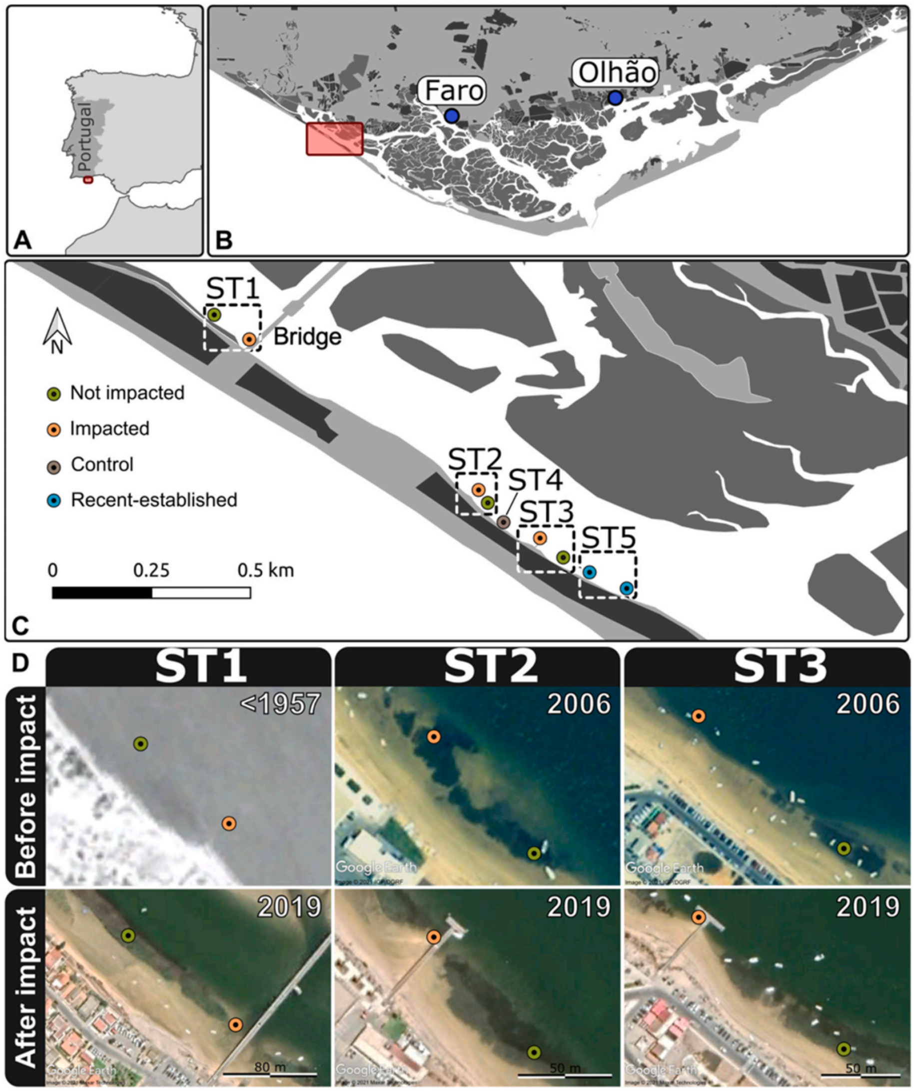

Fig. 1. Location of the Ria Formosa lagoon in South Portugal (A), the study area of Faro beach (B), and the sampling sites, from ST1 to ST5 (C), showing the details of sites before and after the impact of the construction of coastal infrastructures: a bridge in ST1 and pontoons in ST2 and ST3 (D). Source of pictures: Centro de Informação Geoespacial do Exército for ST1 before impact and GoogleEarth for the rest of pictures.

before the coastal construction, and another one in an adjacent non-impacted *Z. noltei* meadow (Fig. 1C–D). Another core was taken in the bare sediment area in ST4 and two cores in the recent-established meadows in ST5, one in each. Sediment cores were sampled by manually hammering PVC pipes (245 cm length, 4.8 cm internal diameter) into the sediment (Table S1). Sediment compaction during coring was measured as the difference in surface sediment elevation inside and outside the core (Glew et al., 2001) and ranged from 7 to 47%. All variables are reported for uncompressed depths. The cores were sealed at both ends, transported to the laboratory in vertical position, immediately halved longitudinally, and sliced every 2 cm (volume of samples after correcting for compaction: 27.4 ± 5.9 cm^{-3}). Samples were frozen at −20 °C until further processing.

Sediment samples were weighted (±1 mg) before (fresh weight, fw) and after (dry weight, dw) oven-drying at 60 °C (48 h) to determine the soil dry bulk density (DBD, g cm^{-3}) and water content (% dw) (Fig. S2). Representative samples at each visual identified sediment layers in each core were selected for biogeochemical analysis, based on the sediment colour and grain size. In dark muddy layers (presumably rich in organic matter), all samples were selected, whereas in light sandy (presumably poor in organic matter), samples were selected at least at the start and end of the layer, and all samples were selected in the transition between layers. A sub-sample (*ca.* 5 g dw) was manually homogenized in a ceramic mortar and pestle, then subjected to loss-on-ignition (LOI) (450 °C, 4 h) to estimate the organic matter content (OM, % dw). A total of 110 paired samples, before and after being subjected to LOI, were selected for CHN analysis using an automated elemental analyser (Elementar, Vario EL III, Central of Technologies and Services, CCMAR), to obtain the organic carbon (OC, % dw) and total nitrogen (TN, % dw) contents in sediment. Then, two linear regressions were built based on our data to estimate TN and OC contents in the remaining samples: OC (% dw) = 0.244 + 0.264 OM (% dw) ($F_{1,108} = 252.3$, $p < 0.001$, $R^2 = 0.698$) and TN (% dw) = 0.072 + 0.02 OM (% dw) ($F_{1,107} = 143.4$, $p < 0.001$, $R^2 = 0.573$) (Fig. S1).

Grain-size analysis was conducted on cores from the bare sediment, impacted and non-impacted meadows. Representative samples from each core (n = 88 in total) were digested with 10% (v/v) H_2O_2 and wet-sieved. The sediment from each sieve was dried in the oven at 60 °C for 24 h and weighed (g dw) to obtain the content of each fraction, according to the Udden-Wentworth scale, in terms of % dw: gravel (>2 mm), coarse sand (<2 mm and >0.25 mm), fine sand (<0.25 mm and >0.063 mm) and mud (<0.063 mm).

2.3. OC and TN stock estimation

Sedimentary stocks of OC and TN (Mg ha^{-1}) were estimated to standardised depths of 1 m and 1.5 m by adding over depth the product of DBD (g cm^{-3}), sample thickness, and OC and TN contents, respectively (Howard et al., 2014). In cores where the depth was less than 1.5 m (bare sediment and old-stablished meadow in ST3), the values of the deepest sample analysed was assumed constant and extrapolated to 1.5 m.

The change in the top 1.5-m OC and TN stocks (stock balance, Mg) at the impacted sites (ST1, ST2 and ST3) was estimated using as reference the stocks (Mg ha^{-1}) of the adjacent non-impacted meadow, as follows: *Stock balance* (Mg) = ($Stock_{impacted}$ - $Stock_{non\text{-}impacted}$ (Mg ha^{-1})) x *Impacted Area* (ha). In addition, the amount of CO_2 equivalents (Mg) potentially emitted by the construction of infrastructures was estimated by multiplying OC stocks by 3.67, the molecular weight ratio of CO_2 to C, considering that the whole carbon stored in the top 1.5-m sediment layer was remineralised.

2.4. Statistical analysis

The relationships between OC and TN contents and mud or OM contents were obtained by linear regression models. The effect of site type (fixed factor with 3 levels: impacted, non-impacted, and bare sediment) and depth of core layer (fixed factor) on grain size fraction (mud, fine sand, coarse sand, gravel) and OC and TN contents was examined using a linear mixed effects regression model, including site as a random factor. Differences found in the models were tested using weighted type III two-away ANOVA. Pairwise comparisons to identify homogenous groups were identified using Tukey's multiple comparison tests. Visual inspection of residual plots did not reveal any obvious deviation from homoscedasticity or normality in the linear models. A critical α level of significance of 0.05 was used. All statistical analysis was conducted in R (R Development Core Team 2021) using the 'lme4' library for the models (Bates et al., 2015).

3. Results

3.1. Sediment profiles

Mud and fine and coarse sand contents along the cores differed among the impacted meadows, non-impacted meadows, and bare sediment site (χ^2 = 49.9, df = 2, p < 0.001, Table 1). The average mud content from the non-impacted meadow was 3-fold higher (31.6 ± 19.6% dw) than that from the impacted areas (9.9 ± 13.8% dw) (post-hoc, z = 6.34, p < 0.001), which did not differ from the bare sediment area (1.7 ± 1.5% dw) (post-hoc, z = 1.16, p = 0.46). Although the mud content was generally low in impacted areas, we observed layers with higher mud contents in ST1 and ST2 (26–64 cm and 60–111 cm, respectively, Fig. 2). The rest of the profile of the impacted sites were dominated by coarse sand (66.8–88.7% dw), similarly to the bare sediment site (84.8% dw). In the sediment profile from the three non-impacted meadows, the highest percentages of mud were found in the shallowest sediment layers, except in ST1 where highest mud contents were detected from 100 to 240 cm depth. Contents of fine sand generally followed those of mud (Fig. 2).

The OC and TN content profiles at the impacted, non-impacted, and bare sediment sites mirrored the observed patterns in the mud content, and TN content was one order of magnitude smaller than the OC content (Fig. 3). Similarly to the mud content, the average OC content was 3 fold -higher at the non-impacted areas (1.09 ± 0.26% dw) than at the impacted areas (0.36 ± 0.07% dw; post-hoc, z = 8.24, p < 0.001), which did not differ from the bare sediment (0.41 ± 0.08% dw; post-hoc, z = 0.54, p = 0.95). The same pattern was observed for the average TN content: higher at the non-impacted areas (0.12 ± 0.02% dw) than at the impacted ones (0.09 ± 0.01% dw; post-hoc, z = 3.50, p < 0.01), not differing the latter from the bare sediment (0.08 ± 0.006%; post-hoc, z = 1.44, p = 0.46). Positive linear relationships were observed between mud content and OC (R^2 = 0.428, p < 0.001, Fig. S3A) and TN contents (R^2 = 0.379, p < 0.001, Fig. S3B), using the whole data set of impacted, non-impacted, and bare-sediment sites.

Table 1
Results of the linear mixed effect models to test the effect of site type (impacted meadow, non-impacted meadow, and bare sediment) and layer depth on the content of grain-size fractions (mud, fin sand, coarse sand, and gravel, % dw), organic carbon (OC, % dw) and total nitrogen (TN, % dw) contents. Significant effects are given in bold.

Variable	Factor	χ^2	df	p-value
Mud	Site type	49.9	2	**<0.001**
	Depth	3.3	1	0.07
Fine sand	Site type	54.7	2	**<0.001**
	Depth	0.6	1	0.43
Coarse sand	Site type	62.8	2	**<0.001**
	Depth	1.1	1	0.30
Gravel	Site type	5.2	2	0.07
	Depth	8.2	1	**<0.01**
OC	Site type	76.8	3	**<0.001**
	Depth	22.5	1	**<0.001**
TN	Site type	20.9	3	**<0.001**
	Depth	26.3	1	**<0.001**

3.2. Organic carbon and total nitrogen stocks

The difference of OC stocks (top 1.5 m of sediment) between impacted and non-impacted areas was clear (Fig. 4 and Table S2). OC stocks from non-impacted areas were 1.5 times higher (86.5 ± 4.8 Mg OC ha^{-1}) than those from the impacted areas (57.6 ± 9.6 Mg OC ha^{-1}), which were similar to the stocks from the bare sediment site (54.9 Mg OC ha^{-1}). The OC stock from the recent-established meadow (61.2 ± 16.4 Mg OC ha^{-1}) were lower than that from the old-established ones (86.5 ± 4.8 Mg OC ha^{-1}), and similar to that from the impacted beds (57.6 ± 9.6 Mg OC ha^{-1}) (Fig. 4). Regarding TN, the effect of the coastal constructions was not so evident since non-impacted and impacted areas presented similar stocks (15.0 ± 1.8 and 14.6 ± 3.4 Mg TN ha^{-1}, respectively), being slightly lower in the recent-established meadow and bare sediment (10.5 ± 0.9 and 12.8 Mg TN ha^{-1}, respectively) (Fig. 4).

The estimated seagrass area lost at sites ST1, ST2 and ST3 after the construction of the bridge and pontoons ranged from 510 to 770 m^2 (Table 2). This represents a loss of 1.1–2.2 Mg of OC due to such coastal constructions, resulting in a potential emission of 3.24–6.67 Mg CO_2 (Table 2). The balance in the TN stock was positive at ST1 (+0.10 Mg TN) and negative at sites ST2 and ST3 (−0.13 Mg TN and −0.02 Mg TN, respectively) (Table 2), that is, impacted area at site ST1 gained TN after the construction of the bridge, while TN stock at sites ST2 and ST3 decreased after the construction of the pontoons.

4. Discussion

This work revealed how small-scale coastal infrastructures (bridge and pontoons) negatively impacted on adjacent *Zostera noltei* meadows by compromising their capacity to sequester OC and TN, as well as reducing the sedimentary carbon and nitrogen stocks. These findings are useful in assessing management strategies, firstly to evidence the impacts on the blue carbon ecosystem service associated to the construction of coastal infrastructures, secondly, to prioritize the protection of old-established seagrass stands to favour the preservation of large carbon stocks, and thirdly, to foster impact mitigation actions through the restoration of seagrass meadows in adjacent areas to favour the continuous sequestration of carbon.

We demonstrated that impacts derived from coastal constructions in Ria Formosa lagoon led to small-scale, yet significant, seagrass total losses and, consequently, to the loss of the services and functions they

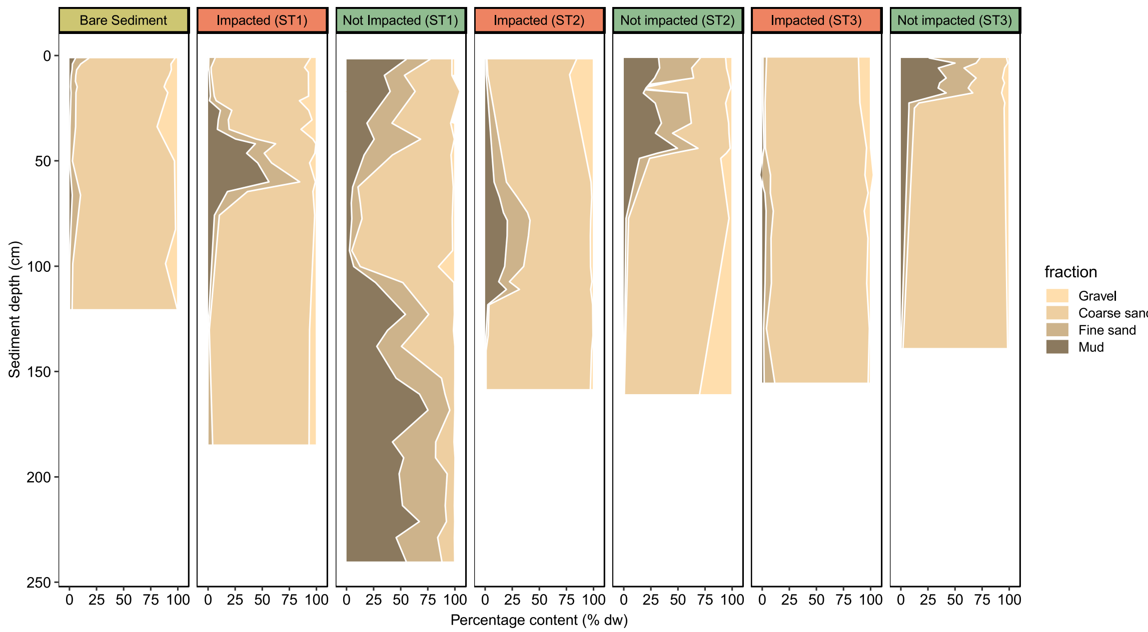

Fig. 2. Grain size depth profiles (gravel, coarse sand, fine sand, and mud contents, in % dw) in the bare sediment site (yellow label) and in the impacted (red labels) and not impacted (green labels) meadows in sites ST1, ST2, and ST3. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

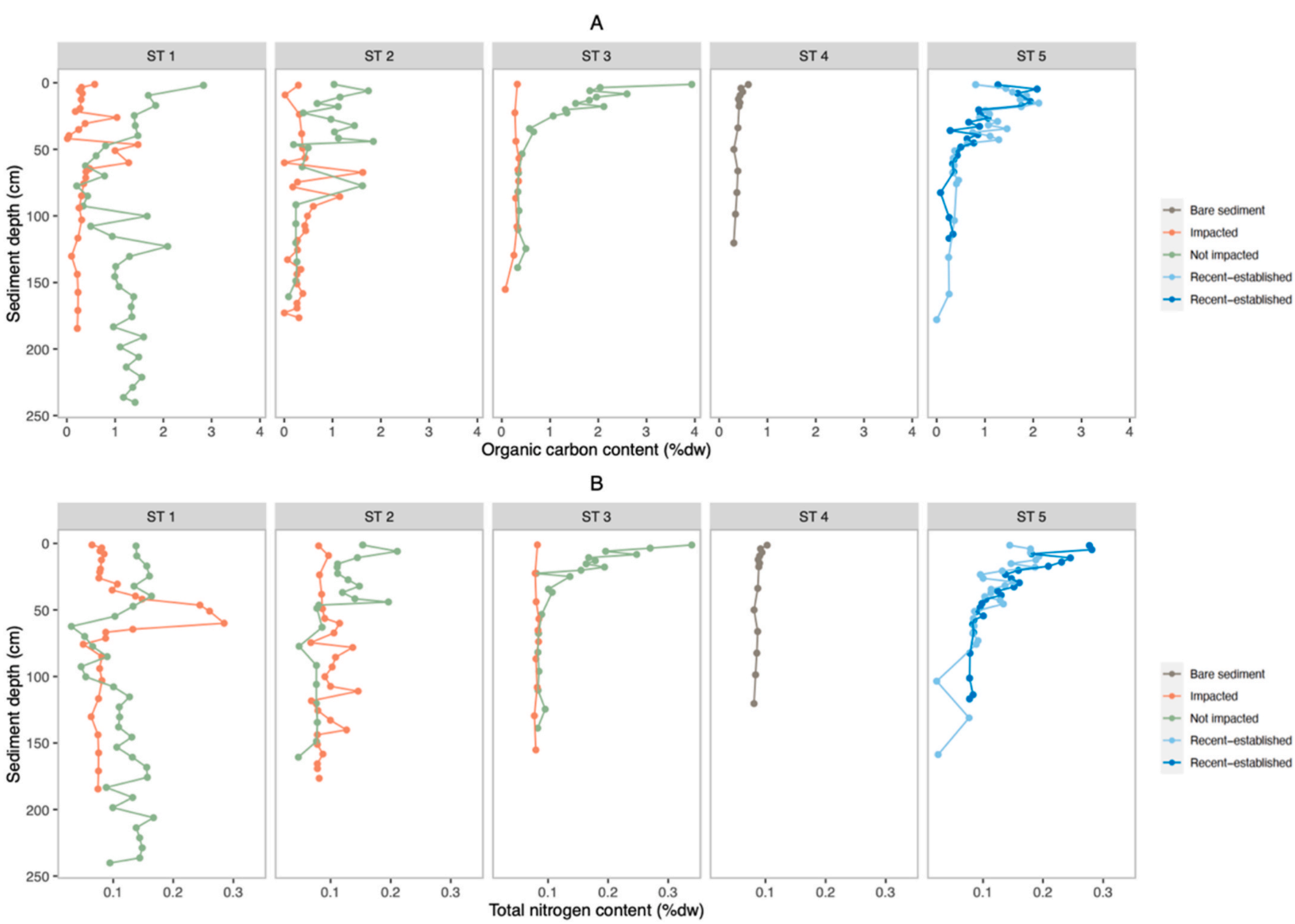

Fig. 3. Sediment depth profiles of organic carbon (OC, % dw) and total nitrogen (TN, % dw) contents in the impacted areas, non-impacted areas (old-established), recent-established meadows, and bare sediment, across sites from ST1 to ST5.

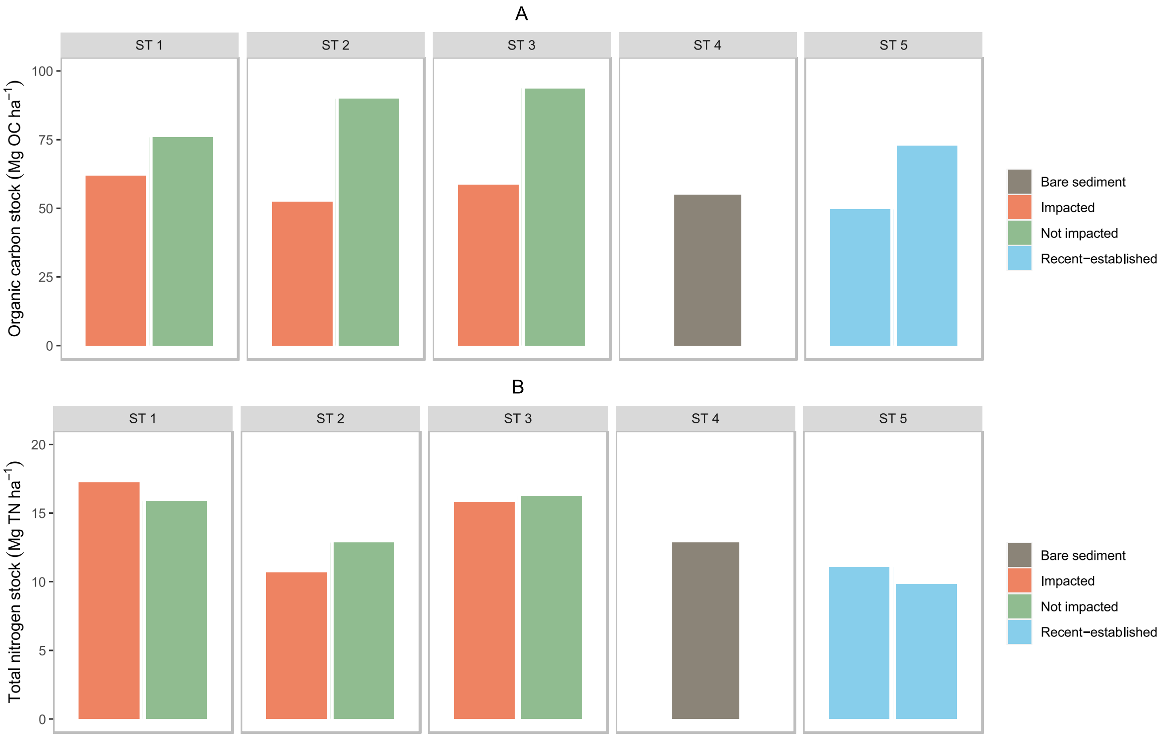

Fig. 4. Stocks of (A) organic carbon (OC) and (B) total nitrogen (TN) (Mg ha^{-1}) in the top 1.5-m sediment layer of impacted meadows, non-impacted old-established meadows, recent-established meadows, and bare sediment.

Table 2
Area impacted (m^2), organic carbon balance (Mg OC), carbon dioxide balance (CO_2) and total nitrogen balance (Mg TN) at the impacted sites ST1, ST2 and ST2.

Impacted site	Impacted area (m^2)	Organic carbon balance (Mg OC)	Carbon balance (Mg CO_2)	Total nitrogen balance (Mg TN)
ST1 (bridge)	770	−1.08	−3.24	+0.10
ST2 (pontoon)	590	−2.22	−6.67	−0.13
ST3 (pontoon)	510	−1.77	−5.30	−0.02

provide. As far as we are aware, this is a pioneering study addressing the effects of coastal infrastructures on blue carbon and nitrogen stocks within seagrass meadows. Our study showed important differences in sediment properties and in OC stocks between the impacted and non-impacted areas, with the mud fraction in sediments from non-impacted areas being 3 times higher than that from impacted ones. The high sand content in the impacted areas may be due to altered hydrodynamics and sediment dynamics by the installed engineered structures (Dugan et al., 2011 and references therein). For example, it is known that coastal infrastructures such as pontoons and jetties can change wave and current regimes, altering the benthic topography and creating, for instance, deep holes and depositional lobes next to the structures (e.g., Sherman et al., 1990; Azarmsa et al., 2009). Furthermore, the linear relationship observed between mud content and either OC or TN content, clearly indicate that the mud-rich areas are also enriched in OC and TN. Such relationships have been already found in several studies predicting that smaller particles are associated with higher OC contents (Bergamaschi et al., 1997; Dahl et al., 2016; Röhr et al., 2016; Mazarrasa et al., 2017) and lower remineralisation rates (Burdige, 2007; Kennedy et al., 2010; Schmidt et al., 2011; Koho et al., 2013). Therefore, as expected, both mean OC content and OC stocks were 1.5 times higher in non-impacted meadows than in the impacted ones. The mean loss of OC stock promoted by coastal infrastructures was 8.90 Mg OC ha^{-1}, almost half the value of the loss estimated in a previous study (20.98 Mg OC ha^{-1}) caused by dredging on seagrass meadows (Thorhaug et al., 2017). Compared to other studies on physical impacts, our estimate is lower than the loss due to vessel groundings (60 Mg OC ha^{-1}, Bourque et al., 2015), which would be consistent as this is a larger physical impact. Other studies have reported lower carbon losses under experimental seagrass removal (2.21 Mg OC ha^{-1}, Githaiga et al., 2019) or even no changes in OC stocks under small-scale disturbances such as grazing and vessel impacts (Macreadie et al., 2014). The effect of coastal constructions on TN stocks was not so evident since non-impacted and impacted areas presented similar stocks. A recent study on *Zostera noltei* and *Spartina maritima* in Ria Formosa estimated TN stocks in the top 1-m sediment layer between 7 and 11 Mg ha^{-1} (Martins et al., 2021), which are similar to those obtained in our study in the top meter (Table S2). Other authors reported a decrease in TN when the *Zostera noltei* meadow was impacted by clam harvesting (50% dw losses) (Barañano et al., 2018), or by wave impacts on *Z. marina* beds with losses of 6.63 Mg TN ha^{-1} (Moksnes et al., 2021).

The estimated loss of sedimentary OC stocks in the impacted areas of the bridge and pontoons, considering the destruction of a seagrass area of 510–770 m^2, rendered a potential loss of 1.1–2.2 Mg OC. Three, non-exclusive, fates are possible for this OC: (1) the organic carbon was buried by the deposition of coarse sediments resulting from hydrodynamic changes caused by the construction, which is supported by the presence of OC-rich mud zones in the impacted areas of ST1 and ST2 at depths of 26–64 cm and 60–111 cm, respectively; (2) fine OC-enriched sediments were eroded, transported by tidal currents and buried elsewhere; and (3) it was remineralised by bacterial activity and returned back to the atmosphere as CO_2. If we consider that all organic carbon was remineralised, we estimate a potential emission of 3.24–6.67 Mg CO_2 in the impacted area. Carnell et al. (2020) estimated an emission of

57.8–104 Mg CO_2 due to sediment erosion induced by sea urchin overgrazing assuming remineralisation of 50–90% of the carbon. Such large values (i.e., 10 times higher than ours), are due to the great area impacted by overgrazing (circa 2.7 ha), while the impacted area we studied was two orders of magnitude lower (0.05–0.07 ha). However, if we extrapolate our results to the area assessed by Carnell et al. (2020), emission values in Ria Formosa are one order of magnitude higher. This indicates that carbon loss resulting from coastal infrastructures is higher than that due to overgrazing.

The impact of constructions on TN varied depending on the studied site, since it was positive for the bridge at ST1 (0.10 Mg TN) whereas it was negative for the pontoons at ST2 and ST3 (−0.13 and −0.02 Mg TN). This result is unexpected, since it may indicate a differential rate of loss or capture for nitrogen in the sediment in ST1 after the bridge construction. In the particular case of ST1, the bridge is transitable with large supporting pillars fully colonized by communities of filter-feeding organisms, such as mussels (personal observation). It is known that biodeposits from bivalves are rich in nitrogen (Kautsky and Evans, 1987) and that sedimentation rates can be also enhanced (Hartstein and Stevens, 2005; Carlsson et al., 2009). Thus, the mussel presence could explain the increase in the TN stock in the bridge area, since large quantities of organic matter rich in nitrogen may be settled down in the area. However, further studies are required to corroborate such hypothesis.

Furthermore, our results showed that the time of the meadow establishment is a significant driver in sedimentary carbon stocks of seagrasses, with a 30% lower OC and TN stocks in the recent-established meadow, 9 years younger than the old-established one. This time effect can be seen in studies on revegetation of seagrass meadow. For instance, Aoki et al. (2020) showed a time-increase in N burial since the onset of a *Zostera marina* meadow restoration reaching a maximum burial rate of 3.52 g N m^{-2} yr^{-1} in mature meadows (>9 years). Marbà et al. (2015) reported that the OC content increased with age at an average rate of 28.2 g OC m^{-2} $year^{-1}$ since the onset of planting *Posidonia australis*, while Greiner et al. (2013) showed that burial rates of OC and TN accelerated 5 years after *Z. marina* planting, reaching rates of 36 g OC m^{-2} $year^{-1}$ after 10 years. Our results are the first quantitative evaluation of OC and TN stocks in natural non-restored seagrass meadows, considering their age or time of establishment. These results emphasise the importance of conservation and maintenance of well-preserved meadows because of their large stocks of OC and TN buried through time. Therefore, restoration of lost meadows as well as conservation of existing meadows becomes necessary to preserve the important function of seagrass beds as carbon and nitrogen sinks (Unsworth et al., 2018).

In summary, the anthropogenic impact of the construction of small-scale coastal infrastructures on intertidal *Z. noltei* meadows resulted in a loss of the seagrass area in the nearby zones to the construction and a loss of OC stocks. This work provides valuable information for future management decisions on coastal ecosystems impacted by anthropogenic activities. The first management option should be the avoidance of the impact by planning infrastructures construction in areas where seagrass meadows do not occur. If that is unavoidable, the impact assessment (IA) should take into account international best practice principles for biodiversity and ecosystem services (Brownlie and Treweek, 2018). For example, IA should aim for a no net loss (NNL) outcome or even aim for net gain (NG), by applying environmental compensatory measures. If the biodiversity and ecosystem service approach is integrated into the IA at the early stages, the direct impacted seagrass area could be used to restore the degraded or lost seagrass area within the same system through transplanting methods.

Credit author statement

Isabel Casal-Porras: Conceptualization, Methodology, Investigation, Formal analysis, Visualisation, Data curation, Writing – original draft, Funding acquisition. **Carmen B. de los Santos:** Conceptualization, Methodology, Investigation, Writing – review and edit, Funding acquisition, Resources, Supervision. **Márcio Martins:** Investigation, Visualisation, Writing – review and edit. **Rui Santos:** Conceptualization, Writing – review and edit, Resources. **J. Lucas Pérez-Lloréns:** Writing – review and edit, Resources, Supervision. **Fernando G. Brun:** Conceptualization, Writing – review and edit, Resources, Supervision.

Declaration of competing interest

Authors declare that there is no conflict of interest.

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Appendix A. Supplementary data

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