Science of the Total Environment 846 (2022) 157410

Contents lists available at ScienceDirect

Science of the Total Environment

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

Salt marsh fragmentation in a mesotidal estuary: Implications for medium to long-term management

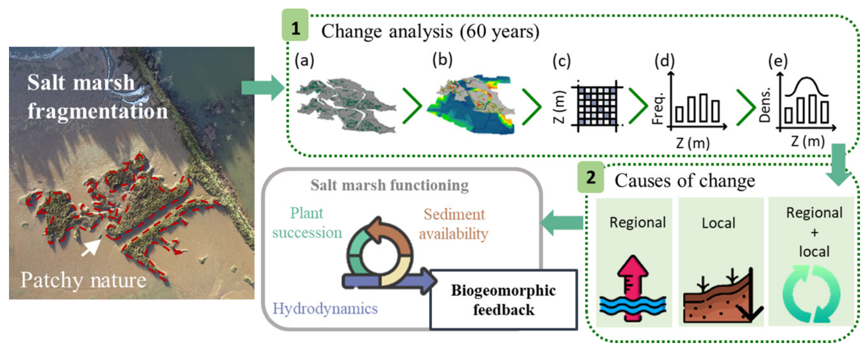
M. Aranda [a,*], G. Peralta [b], J. Montes [a], F.J. Gracia [a], G.S. Fivash [c,d], T.J. Bouma [c,d,e], D. van der Wal [c,f]

[a] Department of Earth Sciences, Faculty of Marine and Environmental Sciences, University of Cádiz, Avenida República Árabe Saharawi, s/n, 11510 Puerto Real, Cádiz, Spain
[b] Department of Biology, Faculty of Marine and Environmental Sciences, University of Cádiz, Avenida República Árabe Saharawi, s/n, 11510 Puerto Real, Cádiz, Spain
[c] Department of Estuarine and Delta systems, NIOZ Royal Netherlands Institute for Sea Research, 140, 4400 AC Yerseke, the Netherlands
[d] Groningen Institute for Evolutionary Life Sciences, Community and Conservation Ecology Group, University of Groningen, Nijenborgh 7, Groningen 9747 AG, the Netherlands
[e] Department of Physical Geography, Faculty of Geosciences, Utrecht University, Princetonlaan 8a, Utrecht 3584 CB, the Netherlands
[f] Faculty of Geo-Information Science and Earth Observation (ITC), University of Twente, 217, 7500 AE Enschede, the Netherlands

HIGHLIGHTS

- Salt marsh fragmentation evolution in a mesotidal estuary with anthropogenic pressures
- Long-term patterns of patchiness combined with topography is a tool to evaluate stress.
- The biogeomorphic feedbacks are responsible of the salt marsh functioning.
- Local factors, rather than regional ones, are the main responsible of the vegetation loss.

GRAPHICAL ABSTRACT



ARTICLE INFO

Guest Editor: Helena Calado

Keywords:
Salt marshes
Estuaries
Eco-geomorphology
Resilience
Nature-based solutions

ABSTRACT

During the last decades many salt marshes worldwide have suffered important losses in their extent and associated ecosystem services. The salt marshes of San Vicente de la Barquera estuary (N Spain) are a clear example of this, with a drastic reduction in vegetation surface over the last 60 years. This paper provides insights into the main factors controlling salt marsh functioning in sheltered estuarine areas. Regional and local factors have been disaggregated to identify the main drivers controlling the functioning of the salt marsh to develop appropriate management measures according to the evolution of the system. These factors have been studied in their spatial context through detailed maps of change in vegetation cover combined with topographic data obtained from UAV and RTK-DGPS surveys. The results demonstrate that in this estuary the salt marsh area is declining following a fragmentation process. No clear pattern of vegetation loss/gain with elevation has been identified. However, the results point to increased hydrodynamic stress in the area, with stronger currents inside the estuary. This is probably the major factor responsible for the decline of the salt marshes in the San Vicente de la Barquera estuary. Furthermore, several human interventions during the 20th century (local drivers) have also probably contributed to a lower resilience against SLR (regional driver). This work demonstrates that both natural and human drivers of change need to be considered when characterizing the evolution of salt marshes, wherever efficient management strategies need to be designed.

1. Introduction

Salt marshes are highly complex systems of great importance for their ecological value and the abundant ecosystem services they provide

* Corresponding author.
E-mail address: maria.aranda@uca.es (M. Aranda).

http://dx.doi.org/10.1016/j.scitotenv.2022.157410
Received 3 March 2022; Received in revised form 7 June 2022; Accepted 12 July 2022
Available online 16 July 2022

(e.g. buffer against energetic events, nurseries for consumers, or carbon and sediment storage on a geological time scale; Barbier et al., 2011; Bouma et al., 2014; Donatelli et al., 2018). These systems are strongly relevant for their role as a natural coastal defence against coastal hazards (Arkema et al., 2017; Ganju, 2019; Kirwan and Megonigal, 2013). The services provided by salt marshes are considered part of the so-called nature-based solutions (NbS). Despite their great importance, in the last decades numerous salt marshes worldwide have suffered major losses in extent and, therefore, in their associated ecosystem services. It is therefore important to quantify long-term changes to understand the processes and timescales driving the evolution of the salt marshes (Sturdivant et al., 2017), in order to recover or conserve the ecosystem services they provide. The conservation and restoration of salt marshes should be a priority for the management of coastal adaptation to climate change (Ganju, 2019; Temmerman et al., 2013).

The threat to salt marshes is widely outlined. Recent studies have focused on the quantification of changes over time (Laengner et al., 2019; and references therein), and others also on the associated drivers of change. Some authors' conclusions are that salt marshes drown with sea-level rise (SLR) effects (Bartholdy et al., 2004; Bouma et al., 2016; Kirwan and Temmerman, 2009; Kirwan et al., 2010) if the sediment accretion rates are lower than the rates of SLR (Kirwan and Megonigal, 2013). This process could lead to a progressive transformation of salt marshes into bare mudflats or partially vegetated by seagrasses and macroalgae. Kirwan et al. (2016) concluded that salt marshes will be able to adapt to future SLR rates only if there is a balanced sediment input. However, other studies indicate that the main SLR threat is not the risk of drowning, but the lateral erosion of the salt marsh edges (Marani et al., 2011; Mariotti et al., 2020; Mariotti and Fagherazzi, 2010). Besides natural processes, changes derived from anthropogenic actions play a crucial role in their evolution and require monitoring, since the combination of both natural and anthropogenic pressures may be responsible for salt marsh deterioration and disappearance (Mcowen et al., 2017). Human pressure and impact on estuaries are very high, including the increasing urbanization concentrated in the coastal zone (Barragán and de Andrés, 2015). Three of the most described anthropogenic actions in estuarine areas are dredging, damming and land claim. Channel deepening for navigation is particularly relevant in large estuaries near important human settlements and ports, but it is also common in smaller estuaries for local activities. All these interventions can strongly modify the estuarine morphodynamics by altering the tidal regime (Donázar-Aramendía et al., 2020), but also the biological and biochemical properties of the intertidal zone. Nevertheless, one of the most described impacts is the effect of dumping the dredged material, which increases turbidity and changes the structure of the sediment, directly affecting the benthic environment. Dredging decreases shallow areas with low hydrodynamics (e.g. mudflats and shallow water, with low physical stress) and increases areas with strong hydrodynamics (e.g. deep water and sand flats, with high physical stress; Gallego Fernández and García Novo, 2007). Therefore, it is an important activity to monitor when facing coastal management plans (e.g., van der Wal et al., 2011). Some studies have focused on the extent and time of the impact these actions have on the system, and most concluded that the adverse impact is noted nearby and in the short term (a few hundreds of meters and <5 years in most cases; Fredette and French, 2004).

Intertidal habitats are also often affected by dike construction. They protect the land from coastal impacts or prevent local flooding (Coops and Van Geest, 2007), but often result in a significant reduction of intertidal areas. Besides, the dike construction at the river mouths hinders the free lateral migration of the beach-dune systems in estuaries (barrier systems), channelizing the fluxes and often accelerating them. The effects of dikes are also widely described and can be summarized in: (1) reduction of flood area and reduction of the tidal prism, (2) slower tidal current velocities, (3) fewer possibilities of channels to migrate, and (4) higher water levels inside the estuary due to an increase in tidal amplification (Pye and Blott, 2014).

The study of the resilience of salt marshes requires an understanding of the dynamic processes associated with them, as well as the pressures they suffer, at a landscape level (Mariotti and Fagherazzi, 2010; van de Koppel et al., 2005). Keeping this in mind, this paper presents a detailed study of 60-year changes in the salt marshes of the San Vicente de la Barquera estuary and evaluates their potential causes. San Vicente de la Barquera (SVB hereinafter) is an estuary in the North of Spain (Gulf of Biscay) with a small catchment area that has suffered a strong decline in the salt marshes during the last decades (Aranda et al., 2020). This study aims to quantify (1) the changes and (2) the fragmentation level in the tidal flat area, and (3) to attribute potential causes to these changes. Furthermore, spatial patterns related to feedback interactions between salt marshes and their abiotic environment are also evaluated. Discussion on the potential causes of salt marsh decline will be evaluated at different spatial scales, considering three hypotheses: (H1) effects of regional factors, i.e. SLR; (H2) effects of local factors, such as dredging and construction works; and (H3) a combination of both regional and local factors.

2. Study area

The study site is located on the Cantabrian coast, the North-Atlantic region of the Iberian Peninsula, at the confluence of two minor rivers. In the Eastern zone, the *Escudo* River contributes to the development of extensive tidal flats (Fig. 1; approx. 200 ha, 4.38°W-43.37°N), with salt marshes and mudflats (bare and colonized by seagrasses); the Western zone is influenced by the smaller *Gandarilla* River – which is not included in the present work.

The pioneer vegetation (i.e. first horizon of vegetation in the salt marsh colonizing the lowest areas) is mainly dominated by the perennial small cordgrass (*Spartina maritima*) and the annual glasswort *Sarcocornia* spp. The upper salt marsh is colonized by perennial vegetation composed mainly of scrub (*Aster tripolium*, Arthrocnemum macrostachyum, *Halimione portulacoides*, *Limonium humile* and *Suaeda* spp. – if present). In addition, some points of the upper salt marsh also exhibited small patches of the exotic invasive species *Spartina densiflora*.

2.1. Hydrodynamic forcing

The SVB is a tide-dominated estuary (Flor-Blanco et al., 2015), whose hydrodynamic regime belongs to the totally-mixed estuary type, i.e. the system has enough energy to mix the entire water column, breaking down the vertical salinity stratification (Cavalcante, 2016; Dalrymple et al., 1992). It is located on a mesotidal-semidiurnal coast with a Mean Spring Tidal Range (MSTR) of 3.94 m and a Highest Astronomical Tide (HAT) that reaches 4.85 m (Santander tide gauge, Fig. 1; National Port Authority, 2019). Regarding tidal wave propagation along the estuary, both the salt wedge and tidal wave reach similar distances from the mouth, 5.8 km and 6.4 km respectively (Flor-Blanco, 2007). The mouth of the estuary was channelled by two jetties, which affected its morphodynamics and sedimentary characteristics (Flor-Blanco et al., 2015). The sediment input in this estuary is low, with a limited river contribution and an external contribution also limited due to the configuration of the coast.

2.2. San Vicente de la Barquera estuarine management

The SVB estuary underwent land reclamation from the mid-20th century with consequences for the hydrodynamic pattern within the estuary, which was partially interrupted by >2500 m of retaining walls (Hoyos Cordero, 2018). The SVB estuary is part of the Oyambre Natural park (1988), and declared Natura 2000 site in 1997. As a consequence, since 2016 retaining walls are being gradually removed to renaturalize the area (Fig. S1). The past drying process, almost a century ago, and the actual recovery of the tidal regime (renaturalization), were (and are) expected to generate strong structural changes in the hydrodynamic functioning of the system. In addition, repeated dredging of the main channel for navigation is also expected to have affected tidal wave propagation throughout the system, suffering almost 30 dredging works since 1933 (Fig. S2; Gobierno de Cantabria, 2018).

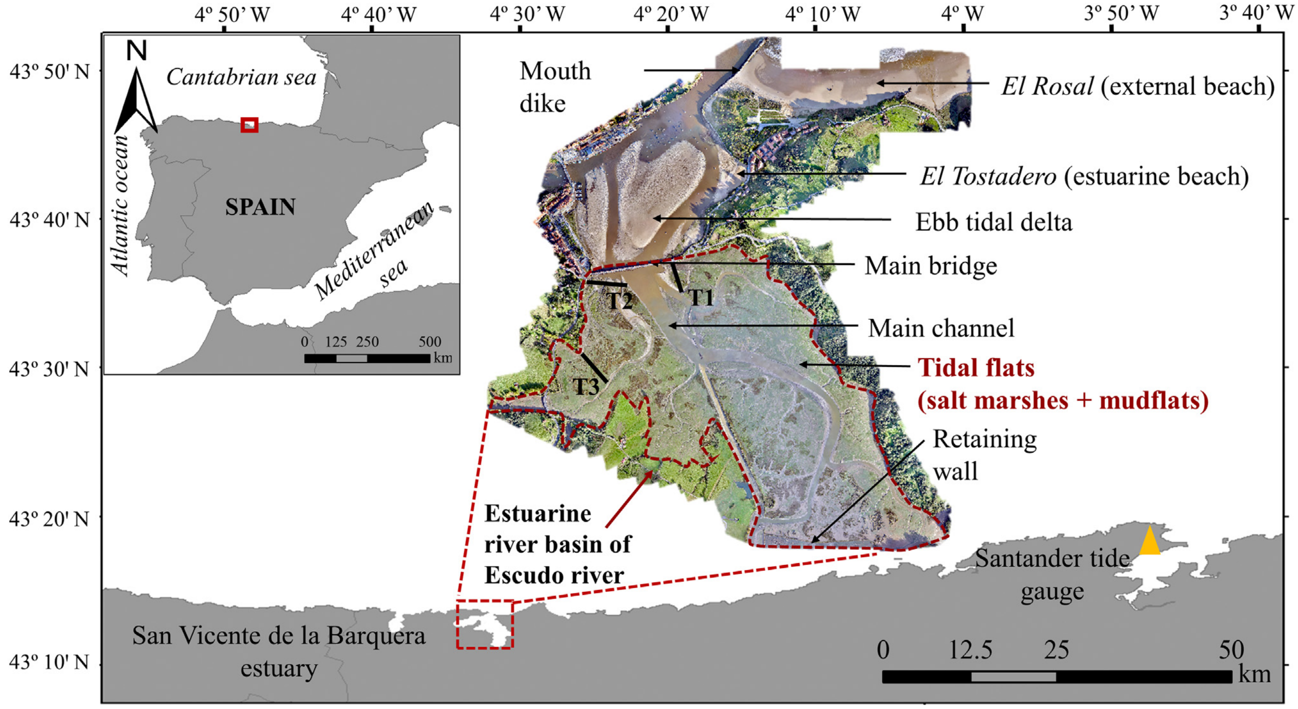


Fig. 1. Location of San Vicente de la Barquera estuary (N Spain) and location of the Santander tide gauge (yellow triangle). An orthomosaic of the estuary with the location of the main features named in this study is shown. Studied features are in red. Defined transects are also indicated (T1,T2, T3).

According to Flor-Blanco (2007), the main effect of these changes is the increase in tidal wave height. This effect has been attributed to the combination of a resonance effect after the reduction in depth, and the hydrodynamic changes caused by the construction of the main bridge that crosses the estuary (Fig. 1).

The river *Escudo* has small anthropogenic pressures, both in number and magnitude. The main pressure is the channelization in urban areas upstream, which retains the supply of fluvial sediments, altering the local hydrodynamics and the sedimentary dynamics of the system (Hoyos Cordero, 2018) and, therefore, the dynamics of the marshes.

3. Methods

The cause and extent of salt marsh decline in the SVB estuary have been evaluated over the last 60 years with the development of 7 geomorphological diachronic maps. These maps were produced based on orthophotographs taken in 1956, 1988, 1997, 2003, 2010, 2014 and 2017; specifications on the image sources and classification methods can be found in Aranda et al. (2020). The analyses carried out for the processing of the different data sources can be divided into two blocks: (1) the analysis of changes in tidal flats over the last 60 years and (2) the identification of the drivers of the changes.

3.1. Changes in tidal flats over the last 60 years

Changes in SVB salt marshes were quantified based on the geomorphological maps described in Aranda et al. (2020). In the present study, the geomorphological maps were converted into binary rasters using the Map Algebra tool in ArcGIS (Fig. 2. a). Subsequently, change maps were generated by overlapping these binary rasters. The changes were categorized into 4 defined classes: (1) stable vegetation areas (*No change in vegetation*; green), (2) *Vegetation gain* areas (yellow), (3) *Vegetation loss* areas (red), and (4) *Mudflats* (grey). Following the methodology from van der Wal et al. (2008), the analysis highlights the qualitative changes, showing them directly on the maps, and provides the corresponding quantitative values, by calculating the total extent of changes that occur in different periods in each defined class.

Moreover, the change maps were combined with a digital terrain model (DTM) to identify changes in the elevation of vegetation classes over the years. The DTM was obtained from a UAV flight conducted in this area in 2018, with a maximum vertical error of 0.18 m. As there was no previous elevation data available for this area (with an acceptable error), the elevation of the DTM was considered a fixed elevation. The DTM values were filtered between −2 and 3 m (reference MSL in Alicante, Spanish reference datum), as the lowest limit of the mudflats is 2 m below MSL and the highest limit of the salt marsh is always under 3 m above MSL. The elevation of each pixel was extracted by merging every layer of the corresponding change map with the DTM (Fig. 2. b). The final outputs were databases (one for each time interval between two consecutive orthophotos), where the rows are pixels and the columns correspond to class and elevation.

To characterize the elevation of each class and the evolution throughout the study period, the probability density function of the elevations was calculated for each class and every defined time interval. This process requires the calculation of the histograms of each class in every time interval (the bin-width was fixed at 0.1 m), in order to obtain the frequency counts of observations. From this frequency, the probability density was estimated according to the Kernel Density Estimation (KDE, Eq. 1). The KDE is a non-parametric way to estimate the probability density function (pdf). It estimates the probability density based on the Kernel normal function and evaluates equally spaced points, creating a smooth curve given a set of data (Baxter et al., 1997).

$$\hat{f}(x) = \frac{1}{nh}\Sigma_{i=1}^{n} K\left(\frac{x - X_i}{h}\right) \tag{1}$$

where $x_1, x_2, \ldots, x_n$ are the samples in the distribution, n is the sample size, K is the smoothed Kernel function and h is the bin-width, which is analogous to the bin-width of the histogram, and affects the smoothness of the resulting curve. This process provides the density curves of frequencies for each class concerning the elevation (Fig. 2. e). The median of these curves

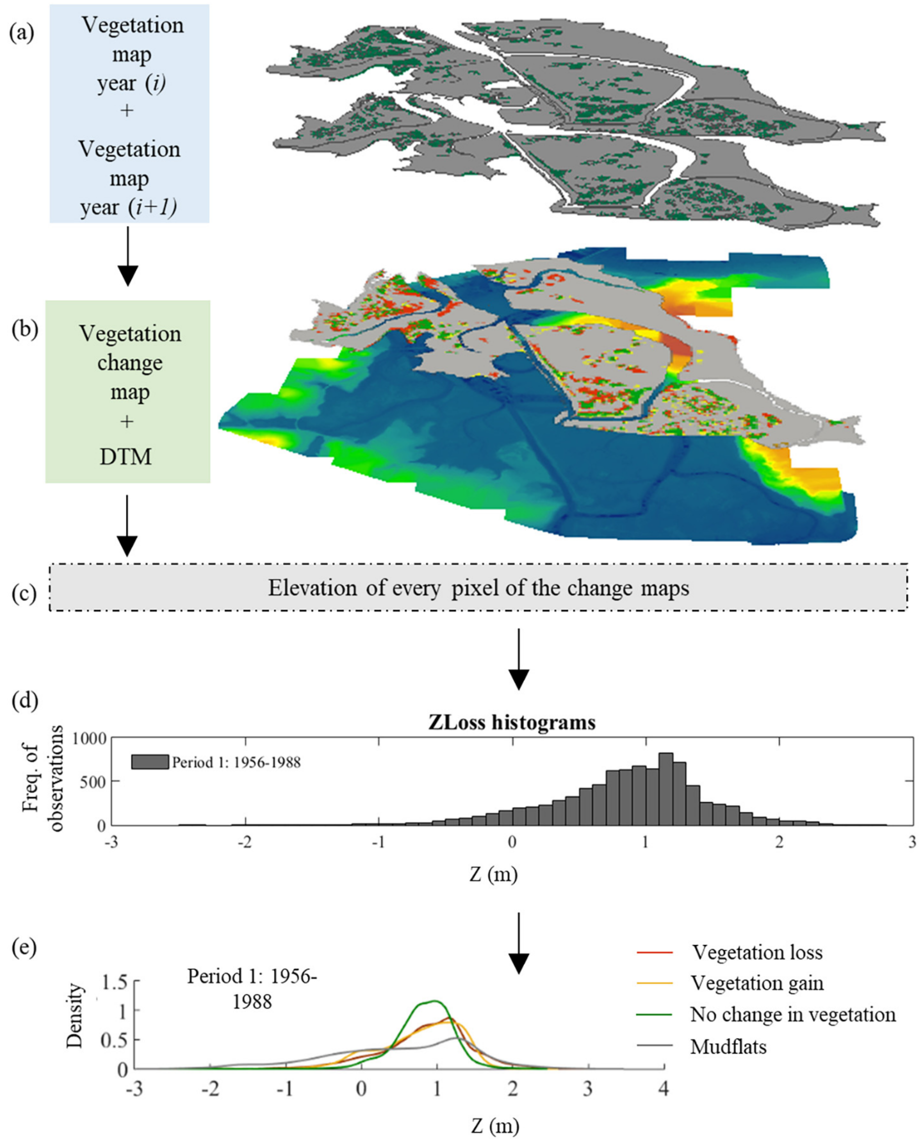


Fig. 2. Implemented workflow to obtain the change maps with elevation per pixel. (a) Combination of geomorphological maps of the tidal flat in two consecutive time intervals to obtain the change maps (salt marshes: green; mudflats: grey); (b) Combination of the change map, from the previous step, with the DTM; (c) Results: data base with the class and elevation of every pixel; (d) Example of the histogram for period 1 for the Vegetation loss class using data from step (c); (e) Example of the density curves for period 1 using data from step (d) for each defined class.

was used as a proxy to quantify the change in elevation for each class during the study period.

3.1.1. Landscape metrics analysis

To better understand the causes of salt marsh degradation, the evolution of the four defined classes was studied using landscape metrics. This procedure is frequently used in landscape analyses (Modica et al., 2012), using the *patch* as the minimum spatial unit, which is defined as a homogeneous area (polygon in GIS) that differs from its surroundings (Forman, 1995). To this end, the dynamics of the landscape and class levels were analysed in each change map using FRAGSTATS v.4.0 (McGarigal, 2015). To enable the synoptic analyses over the past 60 years (1956–2017) with a FRAGSTATS environment, the change maps (raster format) were converted to grids with the same pixel size (7.5 × 7.5 m). The changes in the landscape structure were evaluated by using independent metrics (Cushman et al., 2008), which included the *Number of patches* (NP), *Patch Density* (PD), *Shannon's Diversity Index* (SHDI) and *Mean patch size* (Area_MN). The SHDI describes the landscape diversity, i.e., the heterogeneity in the distribution of patches of the different classes through the years (McGarigal, 2015).

Once the landscape distribution was analysed, the patch size distribution was used to get information on the spatial structure of the vegetation patches and detect causes for the degradation of the salt marshes. According to Bouma et al. (2016), salt marshes are controlled by two-way processes, namely biological (vegetation growth) and physical processes (sediment transport), that when combined form the biogeomorphic feedbacks. Biogeomorphic feedback leads to complex spatial self-organized systems with different responses to environmental forcing (Balke et al., 2014; Marani et al., 2010; van de Koppel et al., 2005). Based on this concept, numerous ecological studies describe how spatial self-organized patterns may be an indicator of stress (van der Heide et al., 2010), as an increase in patchiness (i.e. smaller patches) could be a proxy of ecological transition (Kéfi et al., 2007), to a new state of the system. To identify possible self-organized processes in the estuary, an evaluation was made as to whether the patch distribution could be explained by a power law distribution (Taramelli et al., 2018; Zhao et al., 2019), since this pattern can be interpreted as a symptom of the consistent trend of large patch fragmentation (Kéfi et al., 2014). The fit to the power law function was performed through the 'poweRlaw' package (R v. 3.6).

3.2. Causes of change

Studying variations in plant communities is a useful approach to link habitat degradation to possible regional causes (i.e. SLR), or local causes in the area (i.e. anthropogenic effects). Therefore, both factors are evaluated here.

3.2.1. Regional factors

As a consequence of the low sediment input in the estuary, the SVB salt marshes have limited vertical accretion (Flor-Blanco et al., 2015), pointing to the changes in tidal dynamics inside the estuary and the SLR as the possible main factors involved in the fragmentation of the salt marsh. A way to assess this factor is by evaluating the temporal displacement of the pioneer saltmarsh areas. According to Balke et al. (2016), pioneer plants only colonize areas that emerge during the tidal cycle. The longer the emersion time, the more likely these areas are occupied by salt marsh plants, as long as the equilibrium requirement between sediment supply and water level is met. Based on this, the spatio-temporal trend of the availability of the pioneer zone can be evaluated by determining maps of inundation duration (%) and exposure frequency (%). These maps were based on time series of sea level according to the nearest tide gauge (Santander's gauge; Fig. 1), and the DTM of the area. For each elevation and time interval, the number of flood events was estimated and divided by the number of tidal cycles that occurred (Balke et al., 2016).

Inundation duration and exposure frequency maps were complemented with in situ topographic surveys (11/2018, 04/2019 and 09/2019). The in situ evaluation was based on three transects perpendicular to the main channel (Fig. 1), measured with a DGPS-RTK (vertical errors <0.04 m), where species composition was visually evaluated. The presence and species identification of plants were recorded every 1 m, approx., to characterize marsh zonation and the corresponding elevation ranges (Table S1).

3.2.2. Local factors

Quantification of local factors requires the availability of databases on dredging, drying or renaturalization works for the area. Unfortunately, this information is not available for SVB estuary and required some deep searching on grey literature, most of the time from sources lacking scientific rigour. Therefore, the effects of the potential local drivers were only evaluated qualitatively. The review of local factors included searches in Google Scholar and ResearchGate databases, and the compilation of both regional and national reports from authorities possessing competences in the study area. The search of documents was based on keywords, including location and work type. Location keywords included (1) *San Vicente de la Barquera,* (2) *N Spain,* (3) *Cantabrian coast*, and (4) *north Spain estuaries*, whereas work type keywords included (5) *dredging,* (6) *drying*, (7) *retaining walls*, and (8) *renaturalization works*. Any document including the combination of

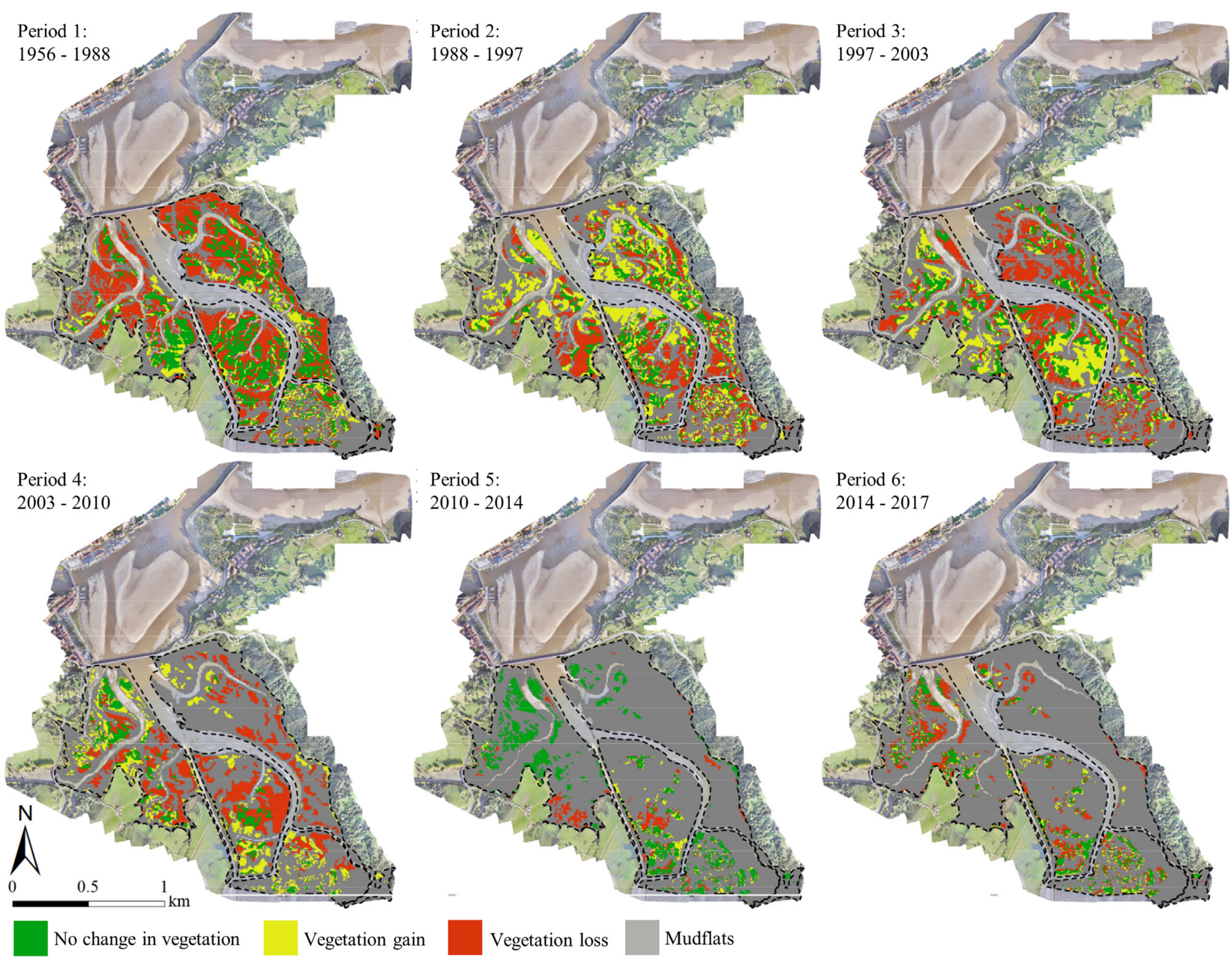


Fig. 3. Change maps for the San Vicente de la Barquera tidal flats in each time interval studied. Dotted lines delimit the boundaries of the study area.

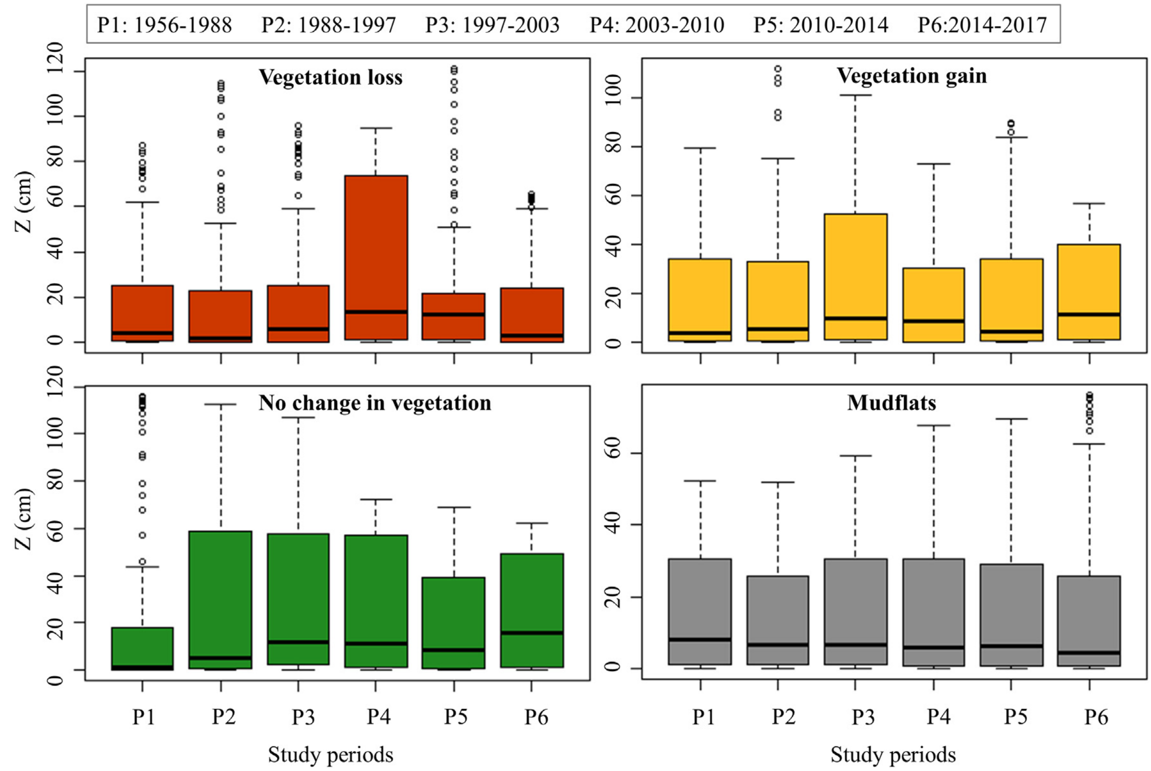


Fig. 4. Changes in elevation of each defined class through the study periods obtained from the combination of the change maps with the DTM of the zone. Slight variations on median values (Q2 on the box) can be inferred for vegetation classes. Boxes represent interquartile range (25th and 75th percentiles), and outliers the maximum values. Minimum values cannot be noted on this scale. Z data are referred to MSL in Alicante, the Spanish Reference datum, and this is plotted in cm in order to better visualize the smaller values.

any location keyword with any work type keyword was reviewed, as well as any competent authority report into anthropogenic activities on the area.

4. Results

4.1. Changes in the tidal flats over the last 60 years

Over the last 60 years, the salt marshes in the SVB estuary have suffered a surface reduction of almost 20 % from the surface in 1956 (corresponding to approximately 85 ha; Fig. S3). The lost area has gradually turned into mudflats (Fig. 3).

The greatest change in habitat distribution occurred at the beginning of the study period (1956–1988), with the loss of large areas of salt marsh vegetation (Fig. 3, red patches). Nevertheless, in these early periods, large areas of *No change* (green patches) and *Vegetation gain* (yellow patches) classes were also observed. Another major change occurred in periods 4 and 5, when most patches of the salt marsh turned into *Mudflat* (grey patches), being extremely low in the abundance of *Vegetation gain* patches during period 5.

The analysis of the median values of density curves (Fig. 4) reveals the topographic sequence for the pattern observed in Fig. 3. Firstly, most vegetation losses were observed at the lowest elevations (red line in Fig. 4), and this loss continued to move upwards through the study periods, as far as period 4 (2003 – 2010) with a sharp increase in the elevation of patches with vegetation loss, after which a decrease in the affected elevation was observed for the last 2 periods (2010–2014 and 2014–2017). The distribution of *Vegetation gain* and *No change in vegetation* patches (yellow and green lines in Fig. 4, respectively) showed a similar temporal pattern, the elevations increasing where they occur with time. On the contrary, the elevation of the mudflats showed a quite constant median over time, taking place at lower elevations through the study period (according to mean values in Fig. 4). Although the loss of vegetation appears to be the dominant process over time (Fig. 3), it is not until period 4 (2003–2010) that the elevation of *Vegetation loss* patches exceeded those of *Vegetation gain* and *No Change in vegetation* patches. Therefore, for period 4, vegetation was lost at higher elevations than gained or maintained. Contrary to expectations, in periods 4 and 5 (2003–2010 and 2010–2014, respectively), the elevation patterns in *Vegetation loss*, *Vegetation gain* and *No change in vegetation* patches did not tally with the other periods, when the losses concentrated at the lowest elevations. *Mudflats'* patches increased in density and occupied lower elevations throughout the entire study period (Fig. 3 and Fig. 4). In the last period (2014–2017), the loss of vegetation occurred at its lowest elevation (Fig. 4), even lower than the areas occupied by mudflats, again denoting that vegetation disappears at low elevations or that zones previously recolonized are eroding.

Table 1
Values of landscape metrics for all time intervals under investigation. NP: Number of Patches; PD: Patch Density; SHDI: Shannon's Diversity Index; Area_MN: Mean Patch Size (ha).

Metric	Period 1: 1956–1988	Period 2: 1988–1997	Period 3: 1997–2003	Period 4: 2003–2010	Period 5: 2010–2014	Period 6: 2014–2017
NP	1291	1236	814	602	551	1031
PD	854.9	828.8	524	375.8	318.4	647.1
SHDI	1.28	1.24	1.16	0.97	0.57	0.63
Area_MN	0.12	0.12	0.19	0.27	0.31	0.15

4.1.1. Metrics analysis at landscape and class levels

The landscape metrics captured the ongoing fragmentation process of the SVB tidal flats (Table 1). At the landscape level and throughout the studied periods, the decrease in the number of patches (NP) was constant, with the exception of an upturn at the final interval. The reduction in NP seems to correlate well with the increase in mudflat surfaces over the years, with patches of mudflat with time. This trend is also supported by patch density (PD), which decreased from 855 to 647 patches/ha. However, the best metric capturing the trend of patches in the SVB estuary was the SHDI. This index decreased sequentially from 1.28 to 0.57 over defined time intervals, as a consequence of a decrease in patch heterogeneity over the years, i.e. low density of very large patches of mudflats and very abundant, but much smaller patches from other classes. Finally, the Area_MN was the most stable metric, with a quite constant value for mean patch size (0.12 at the initial period and 0.15 at the end), as a result of the decrease in patch size for vegetation classes and the increase in patch size of mudflats.

When focusing the analysis on the evolution and fragmentation of the surface at the class level (Fig. 5. a), it can be observed that the salt marsh loss rate was highest in the first interval (1956–1988), decreasing with time and total available area. The number of patches (NP) decreased along the studied period independently of the considered class, with exception of the final period, when a remarkable increase in NP was noted. The final increase in NP was especially large for *Vegetation loss* and *Gain* classes, reaching values similar to the initial ones, but with a much smaller patch size (Fig. 5. a). Finally, the pattern of the *Mudflats* class, with an increase in total surface and a decrease in the number of patches, is consistent with a reduction in fragmentation with time.

4.1.2. Patch size distribution

The distribution of patch sizes does not fit a power law in most cases (see *p*-values in Fig. 6; main stats are specified in Table S2). In general, when comparing the patch size distribution for the initial and final periods, the results show that the frequency of occurrence decreases with size for *Vegetation loss*, *Vegetation gain* and *No change in vegetation* classes. However, for classes of vegetation in period 6, the fit is generally steeper than in previous periods, increasing the quality of the fit. A steeper fit implies a rapid decrease in large patches (a large number of smaller patches), which is contrary to the pattern observed for *Mudflats* patches, showing an increase over time in the frequency of large patches.

4.2. Causes of changes

The existence of changes in the duration of inundation and the exposure frequency affecting the SVB salt marsh was evaluated over the last 25 years, using the time series of the Santander tidal gauge (1993–2018). Both variables showed small changes over time (Fig. S4), slightly larger for the exposure frequency. The complete time series reveals that there is an oscillation (i.e. the curve movement is oscillatory, as expected for these types of sea level data series). However, there is a net displacement to the right, which means that for the same elevation, the inundation duration is higher, i.e. this specific elevation is less time-exposed than before. The direct consequence of this change is that the elevation necessary for vegetation to colonize mudflats increases, approx. 10 cm in the last 25 years (initial and final years of the time series, Fig. S4, zoomed area).

The combination of inundation duration and exposure frequency data with the DTM (2018) of the estuary allows to transform the curves (Fig. S4) into a map of exposure frequency (Fig. 7. a; based on MDT of 2018). The map shows the zones where saltmarsh vegetation can establish itself (areas with a minimum of 35 % exposure frequency in Fig. 7. a), matching with observations made in the field. The exposure frequency map can be used as a proxy for saltmarsh expansion probability, where warmer areas are prone to be colonized by salt marsh species. The lowest limit of vegetation presence observed in the field, i.e. the pioneer zone, was around 0.46 m above MSL. At this elevation, the exposure frequency is around 7 %. For the upper marsh, however, the limit was not that

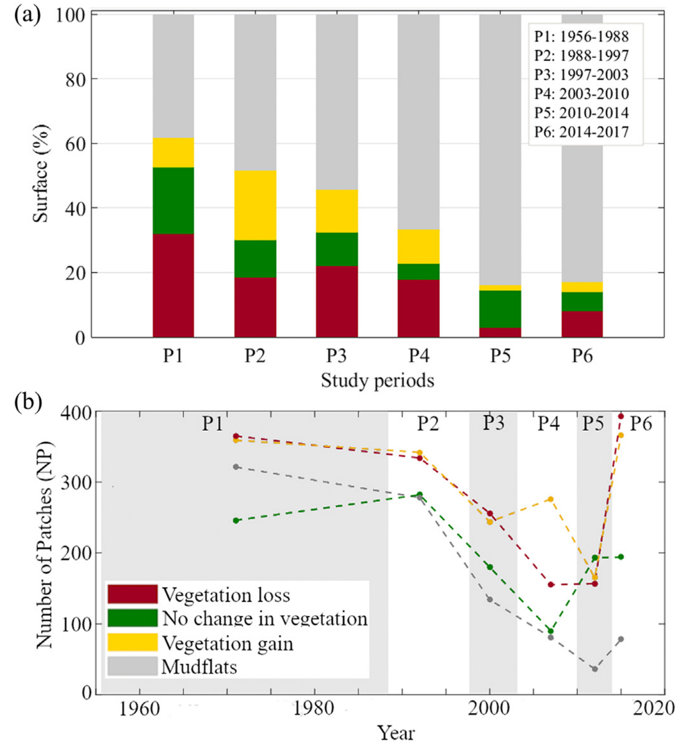


Fig. 5. Landscape composition of the study area by class throughout the study periods. (a) evolution of the occupied surface of each class [in %] with respect to the total surface of the tidal flat; (b) evolution of number of patches (NP) by class during the study periods. Study periods are indicated by alternating shaded areas.

clear, with remarkable differences across the in situ transects (Fig. 7. b, c and d; transect location is indicated in Fig. 1).

The combination of field distribution with the corresponding exposure frequency reveals that there is no clear zonation pattern in the SVB salt marsh, since the pioneer zone and the upper marsh can occur in almost the same elevation range (Fig. S5). The only difference is observed in the lowest limit of the distribution, where the pioneer zone (0.46–1.70 m above MSL; Table S1) extends circa 10 cm lower than the upper salt marsh (0.53–1.77 m above MSL; Table S1). Moreover, the presence of micro-cliffs on the edges of the vegetation patches (Fig. 7. b, c and d) suggests an erosive environment that makes the establishment of plants difficult for both pioneer and upper-zone horizons.

5. Discussion

According to the results presented, the estuary of San Vicente de la Barquera is suffering a decline in the area covered by salt marsh vegetation. Change maps and landscape metric analysis (Fig. 3 and Fig. 4) reveal vegetation loss at low elevations, as expected, but also at high elevations. The elevation of the areas affected by vegetation loss has increased over time, except in the most recent periods in which a reverse trend is observed (Periods 5 and 6 in Fig. 4). The process of plant colonization (Fig. 4, *Gain class*) increased in elevation over time. This pattern could be related to the increase in flood elevation (Fig. S4), as a 3 % exposure frequency seems to be the lowest limit of plant colonization in this estuary. In SVB, the limit between salt marsh and mudflats seems to be around 0.5 m above MSL, and coincides with Balke et al. (2016) in that exposure frequency, rather than inundation duration, is the best variable to define the lowest limit for the establishment of pioneer vegetation. With these boundary conditions, only a small area of the entire intertidal surface seems currently suitable for plant colonization and, therefore, for salt marsh vegetation recovering (West yellow tones in Fig. 7). Around the main bridge (Fig. 1), the ground elevation has increased in recent years

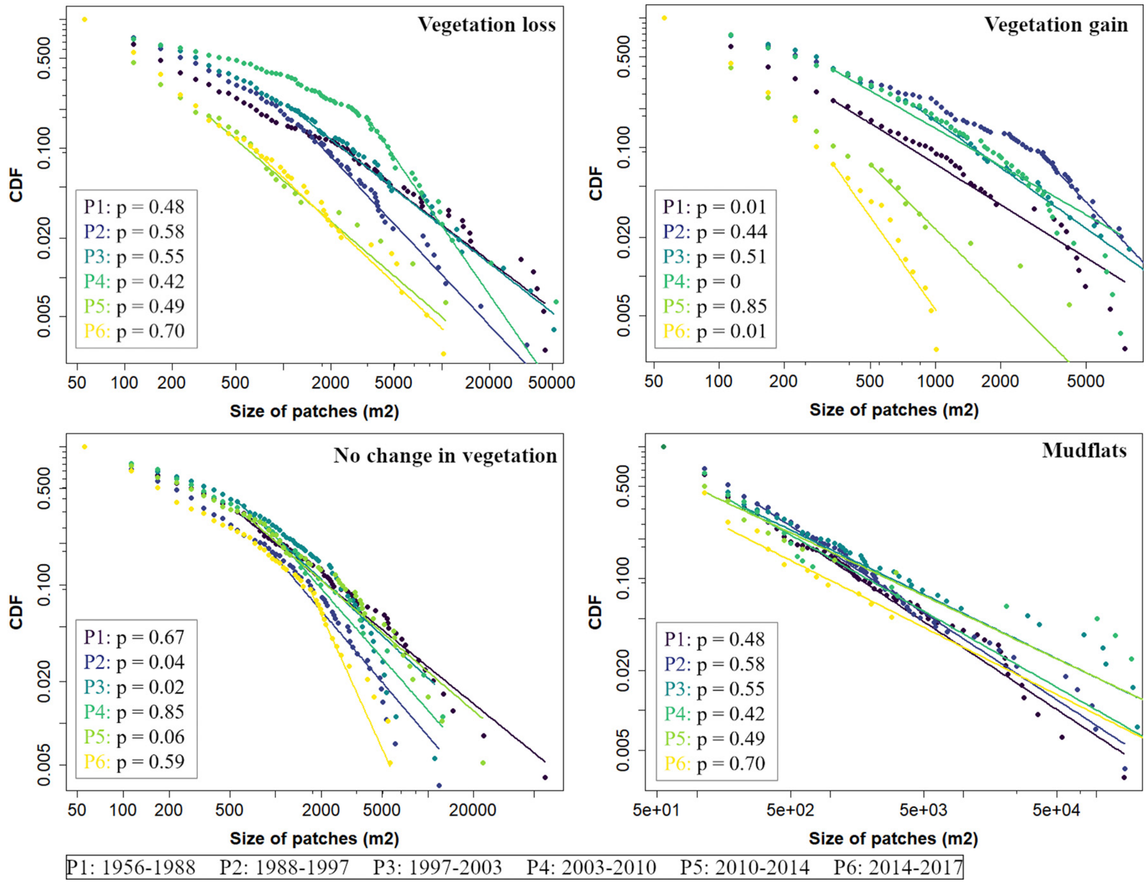


Fig. 6. Patch size distribution by class and time in each study period. Points are observed values and lines the fitted power law model. Axes are represented in a log-log scale. The x-axis represents the patch size (m^2) and the y-axis represents the cumulative distribution function (CDF). The *P*-values obtained are given.

due to sand deposition as a consequence of increased energy in coastal storms, but also due to the remobilization of dredging works, which redistribute the sand to this part of the estuary (Flor-Blanco et al., 2015). Strong energy events can remobilize sediment from the ebb-tidal delta (Fig. 1) and transport it into the estuary via tidal currents. This is also noticeable in the topographic transects that show a slight increase in the elevation of the highest areas of the transects throughout the three field campaigns (Fig. S6). These data illustrate the relevance of sediment mass budget on SVB salt marsh (Ganju et al., 2020), and suggest effects of small temporal-scale processes on SVB salt marsh behaviour. As a consequence, seasonality is expected to have significant effects and consideration of the temporal sequence of the data (orthophotographs and in situ data) will be important in interpreting the results. Without a doubt, the interpretation of salt marsh processes depends heavily on the quality of the elevation data. In the case of SVB, no previous DTM was available, therefore in this study the estimated DTM was used as a constant condition over the time interval. The assumption of constant elevation introduces uncertainties in the analysis of changes in coastal wetlands. Salt marshes are highly dynamic systems (Fitzgerald and Hughes, 2019) fed by external factors (physical, geomorphological and biological), such as tides, sedimentation, erosion and progradation (Adam, 1990). Therefore, considering a constant DTM is similar to assuming drivers of change, like sedimentation, to be static. This assumption has been used in previous studies, arguing the need for computational and interpretative simplifications (Murray et al., 2014; Rahbani, 2011). In intertidal areas where access and measurement difficulties translate into data paucity, as in the study case presented here, simplification is the only way (Aranda, 2021). To compensate for simplification, some authors combine several models to provide a more general perspective of marsh functioning (Mariotti, 2020). Nevertheless, understanding salt marsh evolution requires quantifying vertical ground changes in future works (briefly shown in Fig. S6 only on three transects in the estuary), which are likely to be partly responsible for changes in vegetation.

The dynamic of estuarine pioneer areas can be evaluated by the behaviour of the patch-size distribution (van Wesenbeeck et al., 2008). It has been previously demonstrated that a power law fit distribution of vegetation patches suggests the existence of self-organizing processes controlled by biogeomorphic feedback mechanisms (Rietkerk and van de Koppel, 2008; Taramelli et al., 2018). According to van de Koppel et al. (2005), self-organizing processes improve the functioning of salt marshes over short time scales. However, it also increases physical stress on vegetation edges over longer time scales, favouring habitat change and fragmentation (Fig. 3). In SVB, the spatial pattern of vegetation patches is unclear, as their distribution fits a power law only in some periods (Fig. 6).

Ecosystem functioning also depends on landscape connectivity (Taylor et al., 1993). Characterizing changes in landscape connectivity requires two perspectives: structural and functional connectedness. Structural connectedness refers to the physical continuity of patches within the landscape (i.e. large patches instead of small ones), whereas functional connectedness comprises ecological processes within habitats. In this sense, the SVB salt marshes show an evident decline in structural connectedness (Table 1 and Fig. 5), while functional connectedness could be stressed by self-organizing processes in the long term (Fig. 6).

The decline of SVB salt marshes cannot be attributed to a single cause, but to multiple factors including regional and local ones, often interacting simultaneously (Kirwan and Megonigal, 2013; Thorslund et al., 2017). Nevertheless, in this particular system, local factors seem to be more relevant than regional ones, since there have been anthropogenic pressures acting for a long time (mainly dredging works) that have probably strongly deteriorated the resilience of the salt marsh. The consequences of deepening

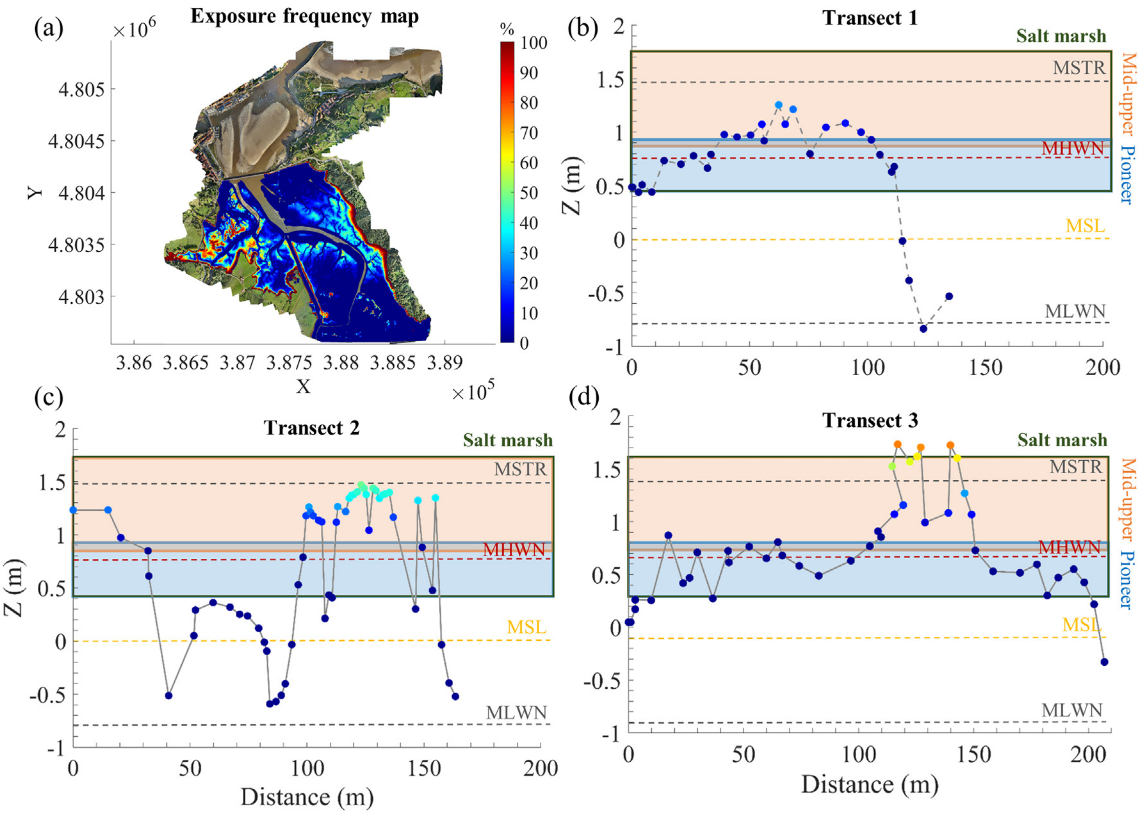


Fig. 7. (a) Exposure frequency map (2018) of the San Vicente de la Barquera estuary; (b, c, d) Exposure frequency (E F, %) of transect 1, 2 and 3, respectively, measured in the field. Dotted lines indicate the MSTR, MHWN, MSL and MLWN (1.475, 0.747 m, 0 m and −0.792 above and below MSL in Alicante, Spanish Reference Datum, respectively). Salt marsh area, i.e. vegetation presence, is indicated by the green box. All dots outside the green box correspond to bare mudflats. The beginning of the transect (0 m in the x-axis) corresponds to the landward zone. Locations of the transects are indicated in Fig. 1.

channels to facilitate maritime transport have been described in other estuaries worldwide, e.g. Westerschelde and Scheldt estuaries in the Netherlands and the Elbe estuary in Germany, among others (Meire et al., 2005; Kerner, 2007). Overcoming differences in spatial scale, these previous works support the conclusion that local pressures have major effects on estuarine functioning, affecting the inner dynamics and processes of estuaries. In SVB, dredging works have decreased since the 1990s and, despite the general negative trend, this may have contributed to a partial recovery of the sheltered and marginal parts of the salt marsh. Topographic aggradation was only detected near the bridge. However, the topographic profiles also exhibited clear symptoms of erosion with the presence of micro-cliffs (Fig. S6), which could be explained by the stronger hydrodynamic conditions described near the bridge due mainly to resonance effects, forming a standing wave, with the node at the bridge and the antinode in the inner part (Flor-Blanco et al., 2015). The particular hydrodynamic conditions near the bridge support the conditions for simultaneous erosional (micro-cliffs) and sedimentary (aggradation) processes in a small space. This pattern is also supported by an irregular plant zonation pattern only explainable by adjacent areas of erosion and recolonization (Singh Chauhan, 2009; van der Wal and Pye, 2004).

Regarding the land reclamation areas, the renaturalization of river flood-plain systems is being widely implemented in Europe at both large and small scales (for instance, river Cole and Skerne in the UK, Brede in Denmark, Danube delta, river Elbe, river Severn or Zeeschelde; Stelk et al., 2017). The removal around 2016 of the retaining walls in the SVB estuary (Fig. S1) has returned the tidal regime to land reclaimed from the sea. Currently, the recovered area is partially renatured, but was previously occupied by a *Eucalyptus* forest. *Eucalyptus* are considered invasive alien species (IAS) and were intentionally introduced into Spain in the XIX century (Silva-Pando and Pino-Pérez, 2016). Therefore, the recovery of the tidal regime has double benefits: the recovery of the previous functioning of the estuary, but also the regulation of an IAS. The recovery of tidal regimes has been practised in other estuaries of the Cantabrian coast since 1959. These experiences have demonstrated that the tidal regime facilitates the reestablishment of ecological functions, e.g. Santoña, Plentzia and Urdaibai (García-Artola et al., 2017). However, in previous cases, the recovery of the functioning was also supported by high sedimentation rates (average 15 mm yr^{-1}), enough to allow the recovery of the vegetation of the salt marshes in periods of <10 years, currently found almost completely in a natural state (Cearreta et al., 2013).

Accelerating salt marsh decline has been widely described, for both North American and European estuaries (Kirwan et al., 2008). Based on previous studies along the US coast, Ganju et al. (2015) described how healthy functioning marshes tend to retain sediment, while degraded ones export it. Thus, the loss of plant biomass leads to a loss of soil elevation, since the system capacity to trap sediment is reduced, but compaction processes are maintained (Belliard et al., 2017; Donatelli et al., 2020; van der Wal et al., 2008). It seems evident that salt marshes will persist only if they can grow vertically or migrate horizontally upland to adjust to the local SLR rates (Day et al., 2008; Kirwan and Megonigal, 2013; Morris et al., 2002). The SLR rates described for adjacent areas to SVB (Muskiz, in the Basque Country) are 0.26 m and 0.59 m for the years 2050 and 2100, respectively, under the RCP 4.5 scenario of SLR; and 0.29 m (2050) and 0.75 m (2100), under RCP 8.5 (Sainz de Murieta et al., 2018). Merging these trends on SLR with the current functioning of the SVB estuary and the anthropogenic pressures above-described, the loss of the SVB salt marsh is expected to accelerate, with the total disappearance of vegetation in 15–20 years (Fig. S3). In this particular case, the presence of rigid infrastructures and geological constraints make landward migration of the salt marsh impossible, creating conditions for a coastal squeeze process (Bouma et al., 2016). The loss of the vegetation would generate a loss of functionality and therefore the loss of ecosystem services.

Understanding the vulnerability of the salt marsh in response to the SLR also requires understanding the dynamics of marsh edges (Marani et al., 2011), essential for the lateral expansion/contraction of tidal marshes. Despite the extensive literature describing the loss of wetlands around the world (Boorman, 2003; Donatelli et al., 2020; Farris et al., 2019; Laengner et al., 2019; Wasson et al., 2019; among others), the uncertainty on expected SLR effects is still very high, mainly due to the lack of knowledge on the interactions between hydrodynamic conditions and eco-geomorphological processes over time (Rodríguez et al., 2017). In short, a deep study is still required on processes favouring/inhibiting the lateral expansion of the tidal salt marshes (Bouma et al., 2016).

Management plans for the adaptation of wetlands to accelerated SLR must take into account eco-geomorphological and hydrodynamic knowledge. The goals of this type of project must be clear, as well as the appropriate methods for measuring the temporal progress. Thus, restoration plans should involve the recovery of prevailing ecological interactions, by allowing the self-organizing systems to rebuild (Gallego Fernández and García Novo, 2007).

6. Conclusions

The causes of the SVB fragmentation process have been assessed at regional and local scales. Local pressures seem to be the most direct causes of change in the SVB salt marshes, modifying the coastal and riverine dynamics and, therefore, diminishing the resilience of the system to future threats at the regional level, such as the SLR. Main local pressures in the SVB estuary include the scarcity of sediment input and the repeated dredging of the navigation channel. Thus, although data are not fully conclusive as to the ultimate causes of degradation and loss, the present work suggests that the most relevant factors in SVB salt marsh decline are local, helping to focus future research efforts. Assessment of SVB estuary responses to tidal recovery or future climate change scenarios still requires additional efforts to be mad towards understanding long-term erosion/accretion patterns.

CRediT authorship contribution statement

M. Aranda: Conceptualization, Data curation, Formal analysis, Funding acquisition, Investigation, Methodology, Project administration, Software, Validation, Visualization, Writing – original draft, Writing – review & editing. **G. Peralta:** Conceptualization, Formal analysis, Funding acquisition, Project administration, Resources, Software, Supervision, Writing – review & editing. **J. Montes:** Formal analysis, Software, Visualization, Writing – review & editing. **F.J. Gracia:** Funding acquisition, Project administration, Resources, Supervision, Writing – review & editing. **G.S. Fivash:** Formal analysis, Methodology, Software, Writing – review & editing. **T.J. Bouma:** Conceptualization, Supervision, Writing – review & editing. **D. van der Wal:** Conceptualization, Formal analysis, Methodology, Software, Supervision, Writing – review & editing.

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

M. Aranda has a postdoctoral contract, funded by the Programme for the Promotion and Encouragement of Research Activity at the University of Cádiz. This work is a contribution to the Andalusia Research Groups (P.A.I.) RNM328 and RNM214, and is supported by an EMAS grant (Earth and Marine Sciences joint PhD programme between the University of Cádiz, Spain, and the University of Ferrara, Italy). This work was partially developed in the Royal Netherlands Institute for Sea Research (NIOZ, Yerseke, The Netherlands). J. Montes has a postdoctoral contract Margarita Salas at the University of Cadiz from the Ministry of Universities of Spain, funded by the European Union-NextGenerationEU.

Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2022.157410.

References

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

Aranda, M., 2021. Geomorphological and Environmental Characterization of Three Estuaries on the Spanish Coast. University of Cádiz.

Aranda, M., Gracia, F.J., Peralta, G., 2020. Estuarine mapping and eco-geomorphological characterization for potential application in conservation and management: three study cases along the Iberian coast. Appl. Sci. 10, 25. https://doi.org/10.3390/app10134429.

Arkema, K.K., Griffin, R., Maldonado, S., Silver, J., Suckale, J., Guerry, A.D., 2017. Linking social, ecological, and physical science to advance natural and nature-based protection for coastal communities. Ann. N. Y. Acad. Sci. 1399, 5–26. https://doi.org/10.1111/nyas.13322.

Balke, T., Herman, P.M.J., Bouma, T.J., 2014. Critical transitions in disturbance-driven ecosystems: identifying windows of opportunity for recovery. J. Ecol. 102, 700–708. https://doi.org/10.1111/1365-2745.12241.

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

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. Ecol. Monogr. 81, 169–193. https://doi.org/10.1890/10-1510.1.

Barragán, J.M., de Andrés, M., 2015. Analysis and trends of the world's coastal cities and agglomerations. Ocean Coast. Manag. 114, 11–20. https://doi.org/10.1016/j.ocecoaman.2015.06.004.

Bartholdy, J., Christiansen, C., Kunzendorf, H., 2004. Long term variations in backbarrier salt marsh deposition on the Skallingen Peninsula - the Danish Wadden Sea. Mar. Geol. 203, 1–21. https://doi.org/10.1016/S0025-3227(03)00337-2.

Baxter, M.J., Beardah, C.C., Wright, R.V.S., 1997. Some archaeological applications of kernel density estimates. J. Archaeol. Sci. 24, 347–354. https://doi.org/10.1006/jasc.1996.0119.

Belliard, J.P., Temmerman, S., Toffolon, M., 2017. Ecogeomorphic relations between marsh surface elevation and vegetation properties in a temperate multi-species salt marsh. Earth Surf. Process. Landforms 42. https://doi.org/10.1002/esp.4041.

Boorman, L., 2003. Salt marsh review. An overview of coastal salt marshes, their dynamic and sensitivity characteristics for conservation and management. JNCC Rep. 1–99. https://doi.org/10.2108/zsj.19.1329.

Bouma, T.J., van Belzen, J., Balke, T., Zhu, Z., Airoldi, L., Blight, A.J., Davies, A.J., Galvan, C., Hawkins, S.J., Hoggart, S.P.G., Lara, J.L., Losada, I.J., Maza, M., Ondiviela, B., Skov, M.W., Strain, E.M., Thompson, R.C., Yang, S., Zanuttigh, B., Zhang, L., Herman, P.M.J., 2014. Identifying knowledge gaps hampering application of intertidal habitats in coastal protection: opportunities & steps to take. Coast. Eng. 87, 147–157. https://doi.org/10.1016/j.coastaleng.2013.11.014.

Bouma, T.J., van Belzen, J., Balke, T., van Dalen, J., Klaassen, P., Hartog, A.M., Callaghan, D.P., Hu, Z., Stive, M.J.F., Temmerman, S., Herman, P.M.J., 2016. Short-term mudflat dynamics drive long-term cyclic salt marsh dynamics. Limnol. Oceanogr. 61, 2261–2275. https://doi.org/10.1002/lno.10374.

Cavalcante, G.H., 2016. Well-mixed estuary. In: K, M.J. (Ed.), Encyclopedia of Estuaries. Encyclopedia of Earth Sciences Series. Dordrecht, pp. 734–737. https://doi.org/10.1007/978-94-017-8801-4_60.

Cearreta, A., García-Artola, A., Leorri, E., Irabien, M.J., Masque, P., 2013. Recent environmental evolution of regenerated salt marshes in the southern Bay of Biscay: anthropogenic evidences in their sedimentary record. J. Mar. Syst. 109–110, S203–S212. https://doi.org/10.1016/j.jmarsys.2011.07.013.

Coops, H., Van Geest, G., 2007. Ecological Restoration of Wetlands in Europe - Significance for Implementing the Water Framework Directive in the Netherlands. 59.

Cushman, S.A., McGarigal, K., Neel, M.C., 2008. Parsimony in landscape metrics: strength, universality, and consistency. Ecol. Indic. 8, 691–703. https://doi.org/10.1016/j.ecolind.2007.12.002.

Dalrymple, R.W., Zaitlin, B.A., Boyd, R., 1992. Estuarine facies models; conceptual basis and stratigraphic implications. J. Sediment. Res. 62, 1130–1146. https://doi.org/10.1306/D4267A69-2B26-11D7-8648000102C1865D.

Day, J.W., Christian, R.R., Boesch, D.M., Yáñez-Arancibia, A., Morris, J., Twilley, R.R., Naylor, L., Schaffner, L., Stevenson, C., 2008. Consequences of climate change on the ecogeomorphology of coastal wetlands. Estuar. Coasts 31, 477–491. https://doi.org/10.1007/s12237-008-9047-6.

Donatelli, C., Ganju, N.K., Zhang, X., Fagherazzi, S., Leonardi, N., 2018. Salt marsh loss affects tides and the sediment budget in shallow bays. J. Geophys. Res. Earth Surf. 123, 2647–2662. https://doi.org/10.1029/2018JF004617.

Donatelli, C., Zhang, X., Ganju, N.K., Aretxabaleta, A.L., Fagherazzi, S., Leonardi, N., 2020. A nonlinear relationship between marsh size and sediment trapping capacity compromises salt marshes' stability. Geology XX 1–5. https://doi.org/10.1130/g47131.1.

Donázar-Aramendía, I., Sánchez-Moyano, J.E., García-Asencio, I., Miró, J.M., Megina, C., García-Gómez, J.C., 2020. Environmental consequences of dredged-material disposal in a recurrent marine dumping area near to Guadalquivir estuary, Spain. Mar. Pollut. Bull. 161, 111736. https://doi.org/10.1016/j.marpolbul.2020.111736.

Farris, A.S., Defne, Z., Ganju, N.K., 2019. Identifying salt marsh shorelines from remotely sensed elevation data and imagery. Remote Sens. 11. https://doi.org/10.3390/rs11151795.
Fitzgerald, D.M., Hughes, Z., 2019. Marsh processes and their response to climate change and sea-level rise. Annu. Rev. Earth Planet. Sci. 47, 481–517. https://doi.org/10.1146/annurev-earth-082517-010255.
Flor-Blanco, G., 2007. Características morfosedimentarias y dinámicas y evolución ambiental de los estuarios de Tina Mayor, Tina Menor y San Vicente de la Barquera (Costa Occidental de Cantabria). University of Oviedo.
Flor-Blanco, G., Flor, G., Pando, L., Abanades, J., 2015. Morphodynamics, sedimentary and anthropogenic influences in the San Vicente de la Barquera estuary (North coast of Spain). Geol. Acta 13, 279–295. https://doi.org/10.1344/GeologicaActa2015.13.4.2.
Forman, R.T., 1995. Land Mosaics: The Ecology of Landscapes and Regions. Cambridge University.
Fredette, T.J., French, G.T., 2004. Understanding the physical and environmental consequences of dredged material disposal: history in New England and current perspectives. Mar. Pollut. Bull. 49, 93–102. https://doi.org/10.1016/j.marpolbul.2004.01.014.
Gallego Fernández, J.B., García Novo, F., 2007. High-intensity versus low-intensity restoration alternatives of a tidal marsh in Guadalquivir estuary, SW Spain. Ecol. Eng. 30, 112–121. https://doi.org/10.1016/j.ecoleng.2006.11.005.
Ganju, N.K., 2019. Marshes are the new beaches: integrating sediment transport into restoration planning. Estuar. Coasts 42, 917–926. https://doi.org/10.1007/s12237-019-00531-3.
Ganju, N.K., Kirwan, M.L., Dickhudt, P.J., Guntenspergen, G.R., Cahoon, D.R., Kroeger, K.D., 2015. Sediment transport-based metrics of wetland stability. Geophys. Res. Lett. 42, 7992–8000. https://doi.org/10.1002/2015GL065980.
Ganju, N.K., Defne, Z., Fagherazzi, S., 2020. Are elevation and open-water conversion of salt marshes connected? Geophys. Res. Lett. 47. https://doi.org/10.1029/2019GL086703.
García-Artola, A., Cearreta, A., Irabien, M.J., 2017. In: Finkl, C., Makowski, C. (Eds.), Recent Agricultural Occupation and Environmental Regeneration of Salt Marshes in Northern Spain. Springer, p. 33. https://doi.org/10.1007/978-3-319-56179-0_2.
Gobierno de Cantabria, 2018. Dragados de mantenimiento de puertos de Cantabria. Puertos de Cantabria.
Hoyos Cordero, J., 2018. Proyecto de las obras de abrigo del nuevo puerto deportivo de San Vicente de la Barquera. Escuela Técnica Superior de Ingenieros de Caminos, Canales y Puertos. Universidad de Cantabria.
Kéfi, S., Rietkerk, M., Alados, C.L., Pueyo, Y., Papanastasis, V.P., ElAich, A., De Ruiter, P.C., 2007. Spatial vegetation patterns and imminent desertification in Mediterranean arid ecosystems. Nature 449, 213–217. https://doi.org/10.1038/nature06111.
Kéfi, S., Guttal, V., Brock, W.A., Carpenter, S.R., Ellison, A.M., Livina, V.N., Seekell, D.A., Scheffer, M., Van Nes, E.H., Dakos, V., 2014. Early warning signals of ecological transitions: methods for spatial patterns. PLoS One 9, 10–13. https://doi.org/10.1371/journal.pone.0092097.
Kerner, M., 2007. Effects of deepening the Elbe Estuary on sediment regime and water quality. Estuar. Coast. Shelf Sci. 75, 492–500. https://doi.org/10.1016/j.ecss.2007.05.033.
Kirwan, M.L., Megonigal, J.P., 2013. Tidal wetland stability in the face of human impacts and sea-level rise. Nature 504, 53–60. https://doi.org/10.1038/nature12856.
Kirwan, M.L., Temmerman, S., 2009. Coastal marsh response to historical and future sea-level acceleration. Quat. Sci. Rev. 28, 1801–1808. https://doi.org/10.1016/j.quascirev.2009.02.022.
Kirwan, M.L., Murray, A.B., Boyd, W.S., 2008. Temporary vegetation disturbance as an explanation for permanent loss of tidal wetlands. Geophys. Res. Lett. 35. https://doi.org/10.1029/2007GL032681.
Kirwan, M.L., Guntenspergen, G.R., D'Alpaos, A., Morris, J.T., Mudd, S.M., Temmerman, S., 2010. Limits on the adaptability of coastal marshes to rising sea level. Geophys. Res. Lett. 37, 1–5. https://doi.org/10.1029/2010GL045489.
Kirwan, M.L., Temmerman, S., Skeehan, E.E., Guntenspergen, G.R., Fagherazzi, S., 2016. Overestimation of marsh vulnerability to sea level rise. Nat. Clim. Chang. 6, 253–260. https://doi.org/10.1038/nclimate2909.
Laengner, M.L., Siteur, K., van der Wal, D., 2019. Trends in the seaward extent of saltmarshes across Europe from long-term satellite data. Remote Sens. 11, 1–25. https://doi.org/10.3390/rs11141653.
Marani, M., D'Alpaos, A., Lanzoni, S., Carniello, L., Rinaldo, A., 2010. The importance of being coupled: stable states and catastrophic shifts in tidal biomorphodynamics. J. Geophys. Res. Earth Surf. 115, 1–15. https://doi.org/10.1029/2009JF001600.
Marani, M., D'Alpaos, A., Lanzoni, S., Santalucia, M., 2011. Understanding and predicting wave erosion of marsh edges. Geophys. Res. Lett. 38, 1–5. https://doi.org/10.1029/2011GL048995.
Mariotti, G., 2020. Beyond marsh drowning: the many faces of marsh loss (and gain). Adv. Water Resour. 144, 103710. https://doi.org/10.1016/j.advwatres.2020.103710.
Mariotti, G., Fagherazzi, S., 2010. A numerical model for the coupled long-term evolution of salt marshes and tidal flats. J. Geophys. Res. Earth Surf. 115, 1–15. https://doi.org/10.1029/2009JF001326.
Mariotti, G., Spivak, A.C., Luk, S.Y., Ceccherini, G., Tyrrell, M., Gonneea, M.E., 2020. Modeling the spatial dynamics of marsh ponds in New England salt marshes. Geomorphology 365, 107262. https://doi.org/10.1016/j.geomorph.2020.107262.
McGarigal, K., 2015. Fragstats. Fragstats 1–182. https://doi.org/10.1016/S0022-3913(12)00047-9.
Mcowen, C.J., Weatherdon, L.V., Van Bochove, J.W., Sullivan, E., Blyth, S., Zockler, C., Stanwell-Smith, D., Kingston, N., Martin, C.S., Spalding, M., Fletcher, S., 2017. A global map of saltmarshes. Biodivers. Data J. 5. https://doi.org/10.3897/BDJ.5.e11764.
Meire, P., Ysebaert, T., Van Damme, S., Van Den Bergh, E., Maris, T., Struyf, E., 2005. The Scheldt estuary: a description of a changing ecosystem. Hydrobiologia 540, 1–11. https://doi.org/10.1007/s10750-005-0896-8.
Modica, G., Vizzari, M., Pollino, M., Fichera, C.R., Zoccali, P., Di Fazio, S., 2012. Spatio-temporal analysis of the urban-rural gradient structure: an application in a Mediterranean mountainous landscape (Serra San Bruno, Italy). Earth Syst. Dyn. 3, 263–279. https://doi.org/10.5194/esd-3-263-2012.
Morris, J.T., Sundareshwar, P.V., Nietch, C.T., Kjerfve, B., Cahoon, D.R., 2002. Responses of coastal wetlands to rising sea level. Ecology 83, 2869–2877. https://doi.org/10.1890/0012-9658(2002)083[2869:ROCWTR]2.0.CO;2.
Murray, A.B., Coco, G., Goldstein, E.B., 2014. Cause and effect in geomorphic systems: complex systems perspectives. Geomorphology 214, 1–9. https://doi.org/10.1016/j.geomorph.2014.03.001.
National Port Authority, 2019. Public Agency of National Ports, Ministry of Transport, Mobility and Urban Agenda. http://www.puertos.es. (Accessed November 2019).
Pye, K., Blott, S.J., 2014. The geomorphology of UK estuaries: the role of geological controls, antecedent conditions and human activities. Estuar. Coast. Shelf Sci. 150, 196–214. https://doi.org/10.1016/j.ecss.2014.05.014.
Rahbani, M., 2011. Numerical Modelling of the Coastal Processes in Dithmarschen Bight Incorporating Field Data. Christian-Albrechts-Universität zu Kiel.
Rietkerk, M., van de Koppel, J., 2008. Regular pattern formation in real ecosystems. Trends Ecol. Evol. 23, 169–175. https://doi.org/10.1016/j.tree.2007.10.013.
Rodríguez, J.F., Saco, P.M., Sandi, S., Saintilan, N., Riccardi, G., 2017. Potential increase in coastal wetland vulnerability to sea-level rise suggested by considering hydrodynamic attenuation effects. Nat. Commun. 8. https://doi.org/10.1038/ncomms16094.
Sainz de Murieta, E., Abadie, L.M., Galarraga, I., 2018. Aumento del nivel del mar en la costa vasca : una aproximación probabilística.
Silva-Pando, F., Pino-Pérez, R., 2016. Introduction of Eucalyptus into Europe. Aust. For. 79, 283–291. https://doi.org/10.1080/00049158.2016.1242369.
Singh Chauhan, P.P., 2009. Autocyclic erosion in tidal marshes. Geomorphology 110, 45–57. https://doi.org/10.1016/j.geomorph.2009.03.016.
Stelk, M.J., Christie, J., Weber, R., Lewis, R.R.I., Zedler, J., Micacchion, M., Harcarik, T., Cowan, L., Famous, N., Teal, J., Bostwick, P., Skidmore, P., Pruitt, B., Urban, L., Merritt, J., 2017. Wetland Restoration. Contemporary Issues & Lessons Learned.
Sturdivant, E.J., Lentz, E.E., Thieler, E.R., Farris, A.S., Weber, K.M., Remsen, D.P., Miner, S., Henderson, R.E., 2017. UAS-SfM for coastal research: geomorphic feature extraction and land cover classification from high-resolution elevation and optical imagery. Remote Sens. 9. https://doi.org/10.3390/rs9101020.
Taramelli, A., Valentini, E., Cornacchia, L., Monbaliu, J., Sabbe, K., 2018. Indications of dynamic effects on scaling relationships between channel sinuosity and vegetation patch size across a salt marsh platform. J. Geophys. Res. Earth Surf. 123, 2714–2731. https://doi.org/10.1029/2017JF004540.
Taylor, P.D., Fahrig, L., Henein, K., Merriam, G., 1993. Connectivity is a vital element of landscape structure. Oikos 68 (3), 571–573.
Temmerman, S., Meire, P., Bouma, T.J., Herman, P.M.J., Ysebaert, T., De Vriend, H.J., 2013. Ecosystem-based coastal defence in the face of global change. Nature 504, 79–83. https://doi.org/10.1038/nature12859.
Thorslund, J., Jarsjo, J., Jaramillo, F., Jawitz, J.W., Manzoni, S., Basu, N.B., Chalov, S.R., Cohen, M.J., Creed, I.F., Goldenberg, R., Hylin, A., Kalantari, Z., Koussis, A.D., Lyon, S.W., Mazi, K., Mard, J., Persson, K., Pietro, J., Prieto, C., Quin, A., Van Meter, K., Destouni, G., 2017. Wetlands as large-scale nature-based solutions: status and challenges for research, engineering and management. Ecol. Eng. 108, 489–497. https://doi.org/10.1016/j.ecoleng.2017.07.012.
van de Koppel, J., van der Wal, D., Bakker, J.P., Herman, P.M.J., 2005. Self-organization and vegetation collapse in salt marsh ecosystems. Am. Nat. 165. https://doi.org/10.1086/426602.
van Der Heide, T., Bouma, T.J., van Nes, E.H., van De Koppel, J., Scheffer, M., Roelofs, J.G.M., van Katwijk, M.M., Smolders, A.J.P., 2010. Spatial self-organized patterning in seagrasses along a depth gradient of an intertidal ecosystem. Ecology 91, 362–369. https://doi.org/10.1890/08-1567.1.
van der Wal, D., Pye, K., 2004. Patterns, rates and possible causes of saltmarsh erosion in the Greater Thames area (UK). Geomorphology 61, 373–391. https://doi.org/10.1016/j.geomorph.2004.02.005.
van der Wal, D., Wielemaker-Van den Dool, A., Herman, P.M.J., 2008. Spatial patterns, rates and mechanisms of saltmarsh cycles (Westerschelde, The Netherlands). Estuar. Coast. Shelf Sci. 76, 357–368. https://doi.org/10.1016/j.ecss.2007.07.017.
van der Wal, D., Forster, R.M., Rossi, F., Hummel, H., Ysebaert, T., Roose, F., Herman, P.M.J., 2011. Ecological evaluation of an experimental beneficial use scheme for dredged sediment disposal in shallow tidal waters. Mar. Pollut. Bull. 62, 99–108. https://doi.org/10.1016/j.marpolbul.2010.09.005.
van Wesenbeeck, B., van De Koppel, J., Herman, P.M.J., Bouma, T.J., 2008. Does scale-dependent feedback explain spatial complexity in salt-marsh ecosystems? Oikos 117, 152–159. https://doi.org/10.1111/j.2007.0030-1299.16245.x.
Wasson, K., Ganju, N.K., Defne, Z., Endris, C., Elsey-Quirk, T., Thorne, K.M., Freeman, C.M., Guntenspergen, G.R., Nowacki, D.J., Raposa, K.B., 2019. Understanding tidal marsh trajectories: evaluation of multiple indicators of marsh persistence. Environ. Res. Lett. 14. https://doi.org/10.1088/1748-9326/ab5a94.
Zhao, L.X., Xu, C., Ge, Z.M., Van De Koppel, J., Liu, Q.X., 2019. The shaping role of self-organization: Linking vegetation patterning, plant traits and ecosystem functioning. Proc. R. Soc. B Biol. Sci. 286. https://doi.org/10.1098/rspb.2018.2859.