ADAPTA BLUES

"Adaptation to climate change through management and restoration of European estuarine ecosystems".

A3.2: Capacity of tidal habitats to path with to sea level rise in European estuaries

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1 INTRODUCTION

Coastal areas are particularly vulnerable to the impacts of climate change, through sea level rise and extreme weather events (e.g. storms, floods) (Wong et al. 2014). Given the high concentration of economic activity and the exposure to hazards, coastal areas are regions under a particularly high disaster risk (Kron 2013; Reguero et al. 2015). The expected scenarios of Climate Change and coastal urbanization increase the need of the world's coastal communities to adapt and manage the risks derived from Climate Change on a sustainable way (Hallegatte et al. 2013; Hinkel et al. 2013; Reguero et al. 2015). In Europe, where approximately one fifth of the EU population lives within 10 km of the coast (EEA 2006), and where many densely populated coastal areas are already below sea level, adaptation strategies are necessary and already in place in many areas.

Thus far, traditional coastal defences have been dominated by built engineering measures, or ´grey´ infrastructure (Mccreless and Beck 2016). However, since Katrina (2005) or Sandy (2012) hurricanes or the Indonesia tsunami (2004) coastal managers worldwide have started to consider alternative solutions based on the natural protection provided by coastal ecosystems, which are known as Nature Based Solutions (NbS, Cohen-Shacham et al. 2016). In particular, estuaries and estuarine ecosystems play a significant role for adaptation to CC in coastal areas. Estuaries serve as buffers of flooding and extreme sea levels and some estuarine ecosystems, such as saltmarshes, seagrass meadows, mangroves and reefs forming organisms (e.g. oysters) protect coastal areas from erosion by dampening wave energy (Temmerman et al. 2013; Ondiviela et al. 2014). For example, saltmarshes prevented over $625 million in flood losses during Hurricane Sandy in the United States (Narayan et al. 2017). In addition, estuarine vegetated ecosystems (i.e. saltmarshes and mangroves) are significant carbon sinks, contributing to CC mitigation through CO_2 sequestration, forming, along with mangroves, the so-called group of Blue Carbon ecosystems (Nellemann et al. 2009). Estuaries also support multiple other ecosystem services relevant for coastal communities, such as fisheries support, biodiversity, water quality improvement and recreational and cultural benefits (Barbier et al. 2011). However, estuaries and coastal ecosystems worldwide, have been threatened by different anthropogenic pressures and climate change related impacts such as sea level rise (Crosby et al. 2016). Estuarine ecosystems are able to self-adapt to sea level rise through two mechanisms: vertical accretion and migration inland (Duarte et al. 2013). The adaptation of these ecosystems to sea level rise assure the maintenance of the ecosystem services provided in future scenarios of climate change.

In Europe, two thirds of European coastal wetlands that existed at the start of the 20th century are lost (Airoldi and Beck 2007). The loss of estuarine ecosystems leads to the loss of all the ecosystem services provided and to an increase in the exposure to climate change hazards. On the contrary, the protection and restoration of estuarine ecosystems provides an opportunity

for coastal climate change adaptation and mitigation while enhancing biodiversity and all other ecosystem services provided (Temmerman et al. 2013; Cheong et al. 2013; Spalding et al. 2014; Sutton-Grier et al. 2015).

The LIFE ADAPTA BLUES project aims to demonstrate that the sustainable management, conservation and restoration of estuarine degraded and reclaimed areas is an efficient strategy to enhance adaptation to climate change in coastal areas of the European Atlantic coast. The ability of intertidal areas to adapt to sea-level rise is one of the most important questions to address for coastal managers to date. The project LIFE ADAPTA BLUES includes a preparatory action (action A3) that specifically aims to provide tools that allow to predict and estimate the resilience of estuarine intertidal habitats in European estuaries to sea level rise through both soil vertical accretion and inland migration.

1.1 Goal

This document presents the results of the work carried out in order to estimate the capacity of intertidal habitats to path with sea level rise through vertical accretion.

2 METHODS

The vertical accretion in coastal ecosystems soils results from a combination of belowground biomass accumulation, in the case of vegetated ecosystems, and the sedimentation of organic and inorganic particles from the water column. Soil vertical accretion may vary across 1) habitats, due to differences in size and biomass of dominant species, and across 2) estuarine locations, due to differences in the influence of the river an important source of suspended particles, the tide (that conditions the duration of the immersion period) (Kelleway et al. 2017; Ricart et al. 2020) or other hydrodynamic components like water currents and waves, that lead to relatively different depositional or erosional conditions. In addition, different human actions at the river basin or estuarine scale such as the building of dams, dykes, intense irrigation practices or deforestation can also lead to differences in sediment vertical accretion rates across estuarine locations and along time, by changing the amount of mineral and organic particles that reach and accumulate in estuarine habitats (Kelleway et al. 2017; Macreadie et al. 2017b; a).

2.1 Study sites

We measured present and historical soil bed level dynamics in different estuarine habitats across five European estuaries from three different regions, distributed along a latitudinal range: the Mondego estuary, in Coimbra (Portugal); the Santoña Marshes estuary, the Miera (Bay of Santander) and Oyambre estuary in Cantabria (Spain); and the Western Scheldt in Zeeland (The Netherlands) (Figure 1).

All estuaries except Oyambre are fed by relatively large rivers. On the contrary, Oyambre is mainly fed by surficial runoff and the input of small creeks, among which Río Turbio that flows into the east branch, is the largest.

In order to reflect differences in sedimentation patterns associated to variability in the river and tidal flow, we considered different estuarine sections, from inner areas to areas closed to the estuary mouth. We considered both vegetated habitats and tidal flats, inhabited by macro benthos, which have received much less attention than vegetated communities (e.g. saltmarshes) despite being critical ecosystems in the estuary (e.g. providing food for migratory species). We analysed the relationship of sedimentary *vs.* erosional trends with natural (e.g. wind) and anthropogenic factors (e.g. dykes) in order to understand the long-term tendency of the estuarine communities and their potential for carbon sequestration and coastal protection in the long-term.

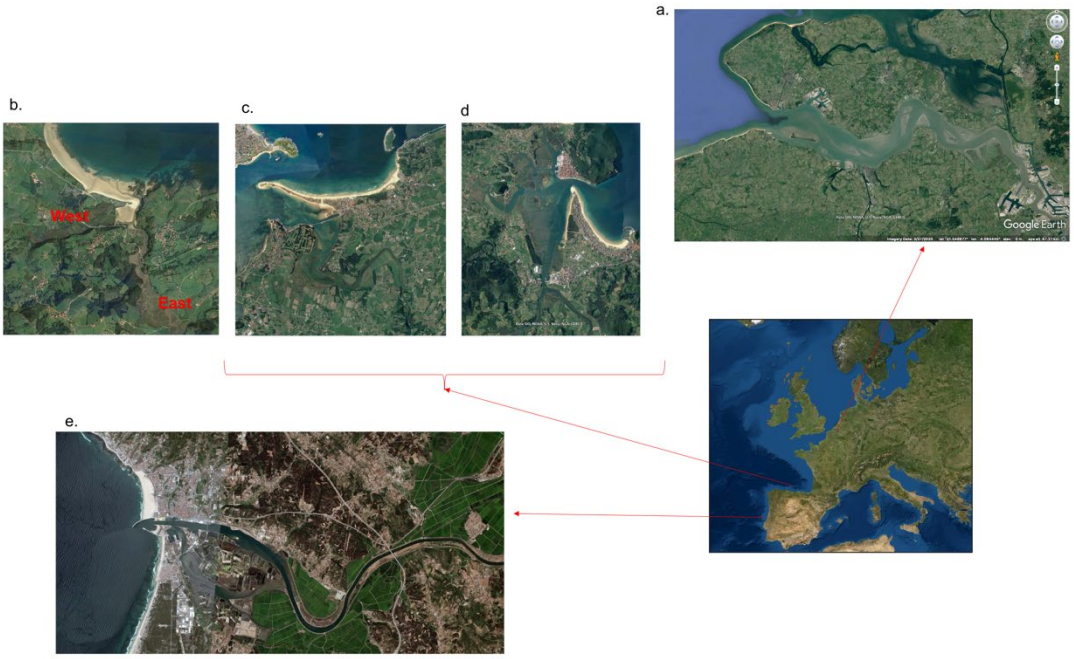

Figure 1 . Location of the estuaries of study in Europe: a) Western Scheldt, in Zeeland (The Netherlands); b)Oyambr east and west branch, c) Santander Bay and d) Santoña Marshes in Cantabria (Spain) and e) Mondego, in Coimbra (Portugal

We combine two techniques that allow estimating present bed level dynamics and historical sediment accretion rates.

2.2 Present bed level dynamics

From previous studies is known that bed level dynamics in estuaries vary spatially across habitats (e.g. saltmarsh communities vs. mudflats) and temporally, across seasons, due to meteorological factors that drive hydrodynamics in estuarine systems (e.g. wind influence) and differences in riverine discharge (e.g. sediment supply). In order to capture present bed level dynamics in estuarine habitats and its spatial and temporal variability, Surface Elevation Dynamic sensors (SED-sensors) were installed during one year at contrasting habitats in the five estuaries of study. Surface Elevation Dynamic (SED) sensors can collect standalone data of sediment dynamics in intertidal habitats for relatively long periods. The sensors can measure bed-level changes continuously with a high vertical and temporal resolution. In addition, spatial differences were analysed by comparing locations under a different exposure to wind energy.

2.2.1 SED-sensors

We used two types of SED-sensors: Optical- (OSED) and Acoustic SED-sensors (ASED).

Optical SED-sensors

The functioning of OSED-sensors is based on the presence of light. Essentially, a pin with 200 light-sensitive cells over a 400 mm measurement section is inserted vertically in the sediment (Figure 2). The transition from aboveground to belowground gives a difference in voltage output, which is used to determine the position of the bed level. The distance between two adjacent cells is 2 mm, so the sensors retrieve 2 to 4 mm accuracy. The sensors collect data when it is not submerged and during the day, as they rely on daylight. The measurement interval in the current study was set to 30 minutes. The difference between measurements tells the user about the bed-level dynamics happened between the measurements. An extensive description and validation of this instrument can be found in Hu et al. (2015).

Figure 2 Exampled of a SED-sensor installed in an Spartina maritima meadow in the Bay of Santander estuary (Spain) (left) and a ASED-sensor installed in the Western Scheldt (right).

Acoustic SED-sensors

In contrast to OSED-sensors, the recently developed ASED-sensors can measure when tidal flats are submerged and thus, most sediment dynamics happen. The ASED-sensor is a device that measures the propagation time of a sound signal reflected by the bed using an acoustic signal with a frequency of 300 kHz. For the sound propagation, water is needed, making the ASED function when most dynamics happen. The accuracy is comparable to the OSED-sensors with 2 to 4 mm. The measurement interval can be set from minutes to hours; we measured every 5 minutes for this campaign.

Given the recent development of the ASED-sensors, no scientific publication on methodology and validation is available. Thus, the sensor was validated within this project. The sensor was validated for 18 contrasting locations in the Eastern Scheldt and Western Scheldt (the Netherlands). For 14 months, manual SEB (Sediment Erosion Bar) measurements were

compared with ASED data for the exact same moment. The majority of the data points fit within a 95% confident range, and the linear regression between both measurements can be explained with 86% variance (R^2 = 0.83) (Figure 3). It is important to highlight that those measurements were done under a range of weather conditions and at locations with contrasting sediment composition (40 µm to 170 µm D50 grain size). The degree of explained variance gives solid confidence in the reliability of the ASED sensors as a suitable method for bed-level monitoring.

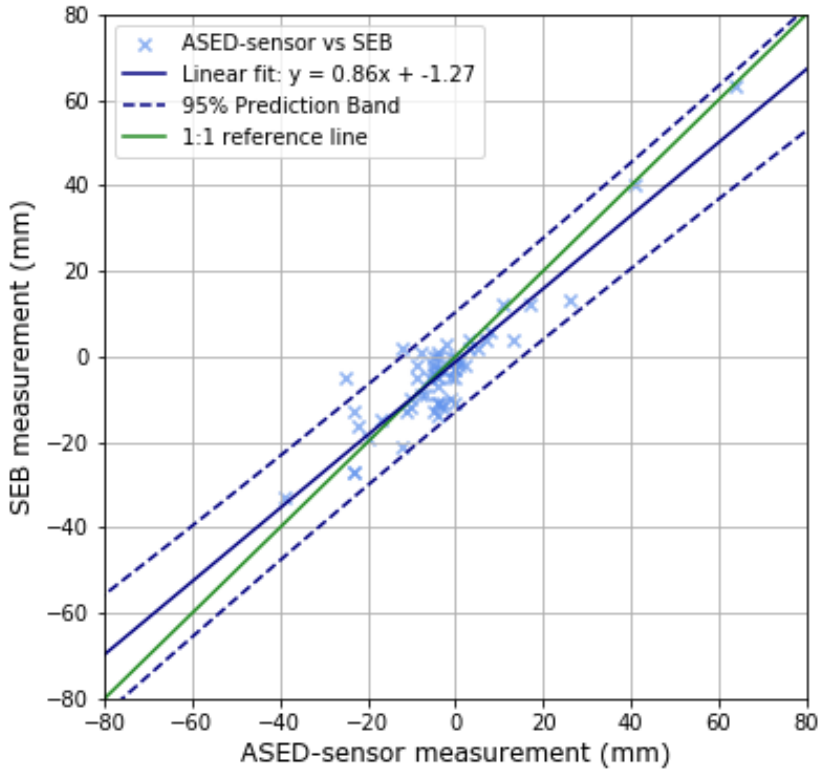

Figure 3. Correlation between SEB- and ASED-measurements for 18 locations in the Eastern Scheldt and Western Scheldt, measured between December 2019 and February 2021.

SED-sensors deployment and data collection

OSED-sensors were installed in Santoña Marshes (n=2), Santander Bay (n=2) and Oyambre (n= 5) estuaries in Spain and in the Mondego estuary in Portugal (n=4). ASED-sensors were deployed in the Western Scheldt (The Netherlands) (n=8) (Table 1-1). The performance of ASED-sensors is significantly better in areas with higher inundation levels compared to the OSED-sensor whereas OSED-sensor perform better in areas with shorter inundation periods and vegetation presence. SED-sensors were installed in both low marsh communities, formed by pioneer species (e.g. *Spartina spp.*, *Sarcocornea spp.*, *Bolboschoenus spp.*) or in unvegetated tidal flats. In the Mondego estuary, one SED-sensor was installed in a *Zostera noltei* meadow and another was installed in a tidal flat where saltmarsh restoration will be carried out within action C2 of this project (Figure 4). This site is referred as pilot restoration site.

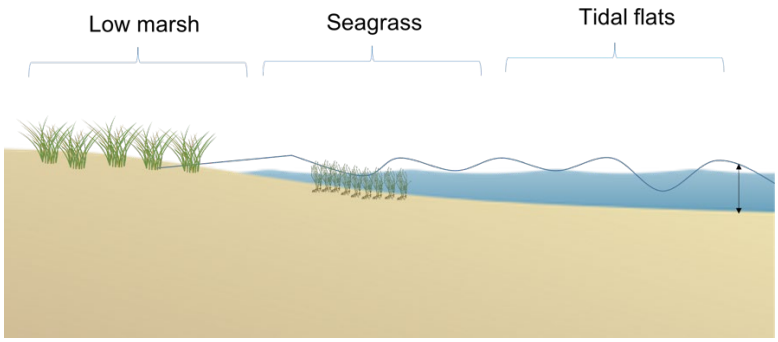

Figure 4 Scheme of different intertidal locations where SED-sensors where deployed in the estuaries of study.

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SENSOR ID / TYPE	*Region*	*Estuary*	*Lat / Long*	*Intertidal zone*	*Dominant spp.*	*Location*	*Other characteristics*	*Deployment date*	*First measurement*	*Last measurement*
NL-1	Zeeland	WS	51,3869205 / 3,8247581	Pioneer zone	*Salicornia spp.*	Outer	Exposed	19/12/2019	20/12/2019	02/02/2021
NL-2	Zeeland	WS	51,3853995 / 3,8237443	Tidal flat	unvegetated	Outer	Exposed	19/12/2019	20/12/2019	02/02/2021
NL-3	Zeeland	WS	51,3510113 / 3,7200605	Low marsh	*Spartina spp.*	Outer	Sheltered	19/12/2019	20/12/2019	02/02/2021
NL-4	Zeeland	WS	51,3522441 / 3,7217453	Tidal flat	unvegetated	Outer	Sheltered	19/12/2019	20/12/2019	02/02/2021
NL-5	Zeeland	WS	51,3970722 / 4,1630143	Low marsh	*Spartina spp.*	Inner	Exposed	20/12/2019	20/12/2019	02/02/2021
NL-6	Zeeland	WS	51,395658 / 4,1636237	Tidal flat	unvegetated	Inner	Exposed	20/12/2019	20/12/2019	02/02/2021
NL-7	Zeeland	WS	51,3642057 / 4,2464466	Low marsh	*Scirpus spp.*	Inner	Sheltered	19/12/2019	20/12/2019	02/02/2021
NL-8	Zeeland	WS	51,3637679 / 4,2440356	Tidal flat	Unvegetated	Inner	Sheltered	19/12/2019	20/12/2019	02/02/2021
SP-1	Cantabria	OY_east	43,383071 / -4,316955	Tidal flat	Unvegetated	Outer		13/01/2020	07/02/2020	22/01/2021

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SENSOR ID / TYPE	*Region*	*Estuary*	*Lat / Long*	*Intertidal zone*	*Dominant spp.*	*Location*	*Other characteristics*	*Deployment date*	*First measurement*	*Last measurement*
SP-2	Cantabria	OY_west	43.383056° / -4.335833°	Tidal flat	Unvegetated	Inner		13/01/2020	13/01/2020	27/02/2020
SP-3	Cantabria	OY_east	43,370445 / -4,310743	Tidal flat	Unvegetated	Inner		13/01/2020	17/01/2020	16/06/2020
SP-4	Cantabria	OY_east	43.383056° / -4.318056°	Low marsh	*Sarcocornea spp.*	Outer		27/02/2020	29/02/2020	16/06/2020
SP-5	Cantabria	SM	43,421134 / -3,4796	Tidal flat	Unvegetated			10/01/2020	10/01/2020	16/06/2020
SP-6	Cantabria	SB	43.433333° / -3.758611°	Tidal flat	Unvegetated			14/01/2020	20/01/2020	17/06/2020
SP-7	Cantabria	SB	43.433333° / -3.758056°	Low marsh	*Spartina spp.*			14/01/2020	Measurements failure	
SP-8	Cantabria	OY_west	43,386047 / -4,32478	Tidal flat	Unvegetated	Outer		13/01/2020	14/01/2020	Lost
SP-9	Cantabria	SM	43,420886 / -3,480808	Low marsh	*Spartina spp.*			10/01/2020	14/01/2020	Lost
P-1	Coimbra	MN	40.118139° / -8.774289°	Tidal flat	Unvegetated	Inner	Sheltered	28/02/2020	29/02/2020	12/11 /2020

SENSOR ID / TYPE	*Region*	*Estuary*	*Lat / Long*	*Intertidal zone*	*Dominant spp.*	*Location*	*Other characteristics*	*Deployment date*	*First measurement*	*Last measurement*
P-2	Coimbra	MN	40.118119° / -8.774519°	Low marsh	*Bolboschoenus spp.*	Inner	Sheltered	28/01/2020	29/01/2020	30/06/2020
P-3	Coimbra	MN	40.140664° / -8.811556°	Pilote site (to restore)	Unvegetated	Mid		28/01/2020	29/01/2020	30/06/2020
P-4	Coimbra	MN	40.140233° / -8.809900°	Low marsh	*Bolboschoenus spp.*	*Mid*		28/01/2020	07/03/2020	03/06/2020
P-5	Coimbra	MN	40.130139° / -8.847281°	Seagrass	*Zostera spp.*	*Outer*		28/01/2020	01/02/2020	30/06/2020
P-6	Coimbra	MN	40.131369° / -8.845461°	Tidal flat	unvegetated	Outer		28/01/2020	29/01/2020	27/06/2020

Table 1 Information about location of SED-sensors deployment in the five estuaries of study.

In the Western Scheldt (Zeeland) SED-sensors were deployed in two inner estuarine locations, one relatively sheltered and one relatively exposed to hydrodynamics, and in two outer estuarine locations, one relatively sheltered and one relatively exposed to hydrodynamic conditions (Table 1; Figure 5). At each site sensors were installed at low marsh and 200 m distance on the tidal flat (Table 1; Figure 5) except in one of the sites (NL-1) where a sed-sensor was installed in a pioneer zone instead of a low marsh due to the absence of the low marsh.

Figure 5 Locations of SED-sensors (labeled in blue) in the Western Scheldt. In green, locations of soil cores sampled for historical accretion analysis.

In the estuaries of the Cantabria region, a total of 9 sensors were deployed. At the Santoña Marshes and Santander Bay estuaries, two OSED-sensors were installed at the same location in two adjacent habitats: a tidal flat and a low marsh dominated by *Spartina spp*. (Table 1, Figure 6, Figure 7).

In Oyambre estuary, four sensors were installed in tidal flats at the inner and outer estuary sections of the east and west branch (4 sensors in total). In addition, one sensor was installed in a low marsh community formed by *Sarcocornea spp.* at the outer section of the east branch (Table 1, Figure 8).

Figure 6 Locations of SED-sensors (labeled in blue) in Santoña Marshes. In green, locations of soil cores sampled for historical accretion analysis.

Figure 7 Locations of SED-sensors (labeled in blue) in Santander Bay. In green, locations of soil cores sampled for historical accretion analysis.

Figure 8 Locations of SED-sensors (labeled in blue) in Oyambre estuary. In green, locations of soil cores sampled for historical accretion analysis.

In the Mondego estuary, a total of six sensors were deployed. Three sites were selected, distributed from the inner estuary to the estuary mouth. In each site two sensors were deployed, in two different habitats (Table 1, Figure 9).

Figure 9 Locations of SED-sensors (labeled in blue) in Mondego estuary. In green, locations of soil cores sampled for historical accretion analysis.

Unfortunately, some of the optical SED-sensors faced technical and logistic failures that limited the data collection (Table 1). One measurement failed, one sensor drowned, two disappeared and several stopped earlier to unclarified problems in a moment when fieldwork was hard to do due to COVID-19 restrictions. Loss and failure of technical instruments in highly dynamic areas are hard to prevent.

2.2.2 Impact of the exposure to wind energy in bed level dynamic rates

Wind data since 1980 was collected to calculate the climatic wind conditions for each site. The European Climate Assessment & Dataset provides daily series of observations at meteorological stations in Spain and the Netherlands. For the Mondego estuary data was retrieved from the website Windguru. However this data was limited to the past 6 years. Wind data was analysed to estimate how often a given wind speed occurred in the long term and during the measurement year of 2020.

As shown in Figure 10, the weather station in the Western Scheldt (in Vlissingen) reported more significant wind speeds than the weather station for the Spanish estuaries (in Santander). Interestingly, during the campaign of 2020, the Western Scheldt was exposed to 20% more strong wind than the climatic average. For the Spanish estuaries, the opposite was happening and seemed to be 8% calmer compared to the long-term. In the end, the data express that the Western Scheldt in the Netherlands was relatively more exposed to wind energy compared to the estuaries examined in Cantabria region, Spain. The Mondego estuary was more or less exposed to the long-term wind conditions during the year of campaign, and in between the Spanish and Dutch sites in terms of exposure.

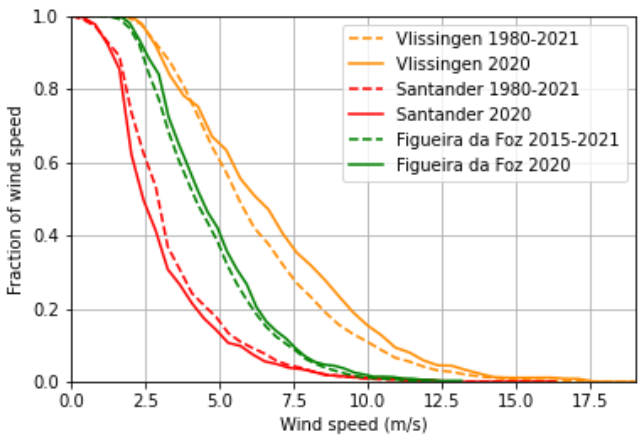

Figure 10 Climatic occurrence of wind speed measured at Vlissingen for the Western Scheldt estuary, Santander for the Spanish estuaries measured from 1980 to 2021, Figueira da Foz for the Mondego estuary measured from 2015 to 2021.

The tidal flats studied are located in areas under different hydrodynamic conditions that condition the long-term sedimentary *vs.* erosional trends. Yet, the level of wind-waves exposure at certain time can impact the short-term behaviour of the bed level dynamics. As we experienced the failure of several SEDs, it is essential to control if the SEDs measured most occurring wind conditions to have a general understanding of the tidal flat response to wind. In our case, for all locations, the SEDs measured at least 99.5% of the climatic wind speed

occurrence. In other words, the bed level trends measured during the deployment period reflect the effect of wind conditions.

The effect of the wind in sedimentary dynamics was assessed through a morphodynamical footprint analysis. The footprint studies the correlation between the measured sediment dynamics and prevailing wind direction and provides insights on the sensitivity of storms for the given location. This analysis allows to understand if bed level trends measured during the deployment period are dominated by certain wind conditions (e.g. calm weather, storm dominated). In this research, the wave exposures are explicitly not included as to do so wave dynamics models or wave measurements are needed, which required the use of expensive wave-measurement devices. A detailed explanation of the methodological procedures of the morphodynamical footprint analysis is provided in Annex I.

2.3 Historical accretion rates

To determine historical sediment vertical accretion rates (SAR) we sampled a total of 21 soil cores across different estuarine locations and habitats, in the five estuaries of study (Table 2, Figure 5, Figure 6, Figure 7, Figure 8 , Figure 9). Some soil cores were sampled nearby SED-sensors were deployed (Table 2).

2.3.1 Soil cores sampling and processing

Soil cores were sampled by manually hammering PVC tubes (60 cm L * 7 cm D). Soil compaction during hammering was measured for each core. Each core was sliced every 1 cm along the whole sediment depth. Each sediment slice was measured for wet volume (cm^3) and wet weight (g) and dried at 60ºC for a minimum of 72 h. Dry weight of each sediment slice was measured and used along with wet volume to estimate sediment dry bulk density (DBD, g DW cm^{-3}) of each slice. Further details on the sampling and processing of the sediment cores can be found in Deliverable A2.

Region	*Estuary*	*Location*	*Lat/Long*	*Habitat*	*Dominant genera*	*Core code*	*SED-sensor nearby*
Zeeland	WS	Inner_exposed	51.397864° / 4.162592°	High marsh	*Puccinellia spp., Halimione spp.*	WS2A1	
		Outer_ exposed	51.387206° / 3.824794°	Pioneer zone	*Salicornia spp.*	WS3B1	NL 1
		Outer_sheltered	51.349617° / 3.719358°	High marsh	*Spartina spp.*	WS4A1	
Cantabria	SB	Inner	43.432096 / -3758772	High marsh	*Juncus spp., Halimione spp.*	BS1A1	
			43.43318 / -3.758215	Low marsh	*Spartina spp.*	BS1B1	SP7

Region	Estuary	Location	Lat/Long	Habitat	Dominant genera	Core code	SED-sensor nearby
		Outer	43.452136° / -3.748134°	High marsh	*Halimione spp.*	BS2A1	
			43.451940° / -3.748752°	Low marsh	*Sarcocornea spp.*	BS2B3	
	SM	Outer	43.419143° / -3.480078°	High marsh	*Halimione spp.*	MS2A3	
			43.419399° / -3.480063°	Low marsh	*Spartina spp.*	MS2B2	
			43.419456° / -3.479795°	Seagrass	*Zostera spp.*	MS2C3	SP5
			43.382817° / -4.318047°	High marsh	*Halimione spp., Juncus spp.*	NM3	
	OY	Outer_east Inner_east	43.383328° / -4.317465°	Seagrass	*Zostera spp.*	NP2	SP1
			43.373796° / -4.315541°	High marsh	*Halimione spp, Juncus spp.*	RM3	
		Inner_east Outer_west	43.374078° / -4.315354°	Tidal flat	Unvegetated	RP3	
			43.385830° / -4.324631°	High marsh	*Halimione spp., Juncus spp.*	A1M1	
		Outer_west Inner_west	43.386026° / -4.324808°	Tidal flat	Unvegetated	A1P1	SP8
			43.382675° / -4.335772°	High marsh	*Halimione spp, Juncus spp.*	A2M3	
		Inner_west Outer	43.383133° / -4.335868°	Tidal flat	Unvegetated	A2P3	SP2
Coimbra	MN						
		Outer	40,13201° / -8,84815°	Low marsh	*Spartina spp.*	MM1A1	
		Outer Mid	40,130783° / -8,846967°	Seagrass	*Zostera spp.*	MM1B1	P5
			40,140467° / -8,8112°	Tidal flat	Unvegetated	MM2C1	P3

Table 2 Location soil cores sampled for historical soil vertical accretion in estuarine communities across the estuaries of study.

2.3.2 Sediment dating

Sediment age along the sediment depth profile was determined applying ^{210}Pb dating techniques, especially suitable for the last 100-150 years (Appleby 2001), the period where the largest impacts in coastal ecosystems have taken place. ^{210}Pb analysis were conducted in the Unit of Physics of Radiations from the Autonomous University of Barcelona and in the Environmental Radioactivity Laboratory of Edith Cowan University (Perth, Australia). The age of the sediment was determined applying CF:CS models (Krishnaswamy et al. 1971). Basically, to estimate the age of the sediment by means of ^{210}Pb dating techniques, decreasing distribution of ^{210}Pb specific activity as a function of depth is required (Arias-Ortiz et al. 2018). This type of ^{210}Pb specific activity profile was found only in 15 cores (Annex I). Thus, the age of the sediment could be estimated in 15 cores whereas in the reminder 6, different sedimentary processes (e.g. mixing, lack of sedimentation) precluded the application of dating models.

Numerical procedures

Sediment accretion rates (cm y^{-1}) were estimated by dividing the decompressed depth of a sediment section by the number of years the section took for accumulating. For each core we estimated net SARs since 1950, the oldest age reached by all the cores that could be dated, in order to analyze differences in long term SAR across habitats and estuarine locations.

3 RESULTS AND DISCUSSION

3.1 Present bed level dynamics

3.1.1 Western Scheldt

The SED-sensors installed in the Western Scheldt worked well and measured during the whole year the deployment was planned.

The results provided by the SED-sensors in the Western Scheldt, revealed several major differences between habitats, between the exposed and sheltered locations and between the outer and inner section of the estuary.

At the outer estuarine area, the tidal flats tend to be more dynamic compared to the higher elevated pioneer zone at the exposed area (Figure 11 A and B) as indicated by the standard deviation of the daily bed-level change, which is almost double in the tidal flat (3.9 mm/day) compared to the pioneer zone (2.1 mm/day). On the contrary, no differences between the tidal flat and the low marsh were found for the sheltered site, that have relatively minor sediment dynamics compared to the exposed site (Figure 11 C and D). The exposed tidal flat at the outer estuary showed an erosional trend towards the year whereas the other locations in the outer estuary seem to fluctuate around a potential equilibrium profile.

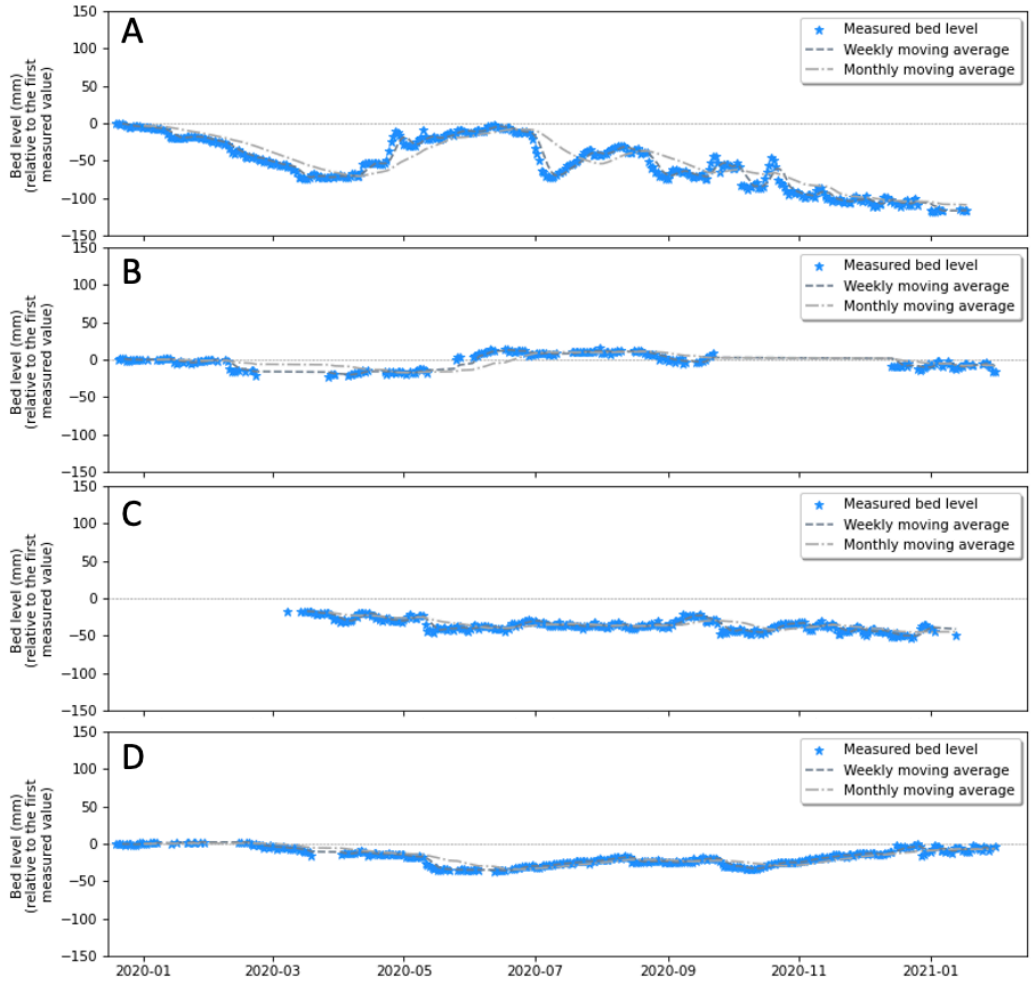

Figure 11 Results of the sediment dynamics at the outer estuarine areas examined for respectively the exposed tidal flat (A), exposed pioneer zone (B), sheltered tidal flat (C) and sheltered pioneer zone (D).

In the inner estuarine section, all locations showed a relatively stable trend along time except the exposed low marsh (Figure 1-12 b) that showed approximately +/- 30 mm accretion in 2020 (Figure 1-12).

In general, exposed and sheltered sites examined at the inner estuarine section (Figure 12), showed higher daily bed-level variability for the two habitats examined (bed level standard deviation of ~3.5 mm/day for low marsh and 2.4 - 6.9 mm/day for tidal flats) that sites examined at the outer estuarine section (maximum bed level standard deviation of 3.9 mm/day and 2.1 mm/day for the tidal flats and pioneer zone) (Figure 11). Whereas differences in grain size are found between the outer exposed site (50 µm) and inner exposed site (170 µm), grain size was similar for the sheltered sites across outer and inner estuarine sections. Thus, differences in sediment accretion rates standard deviation can't be attributed to differences in sediment grain size.

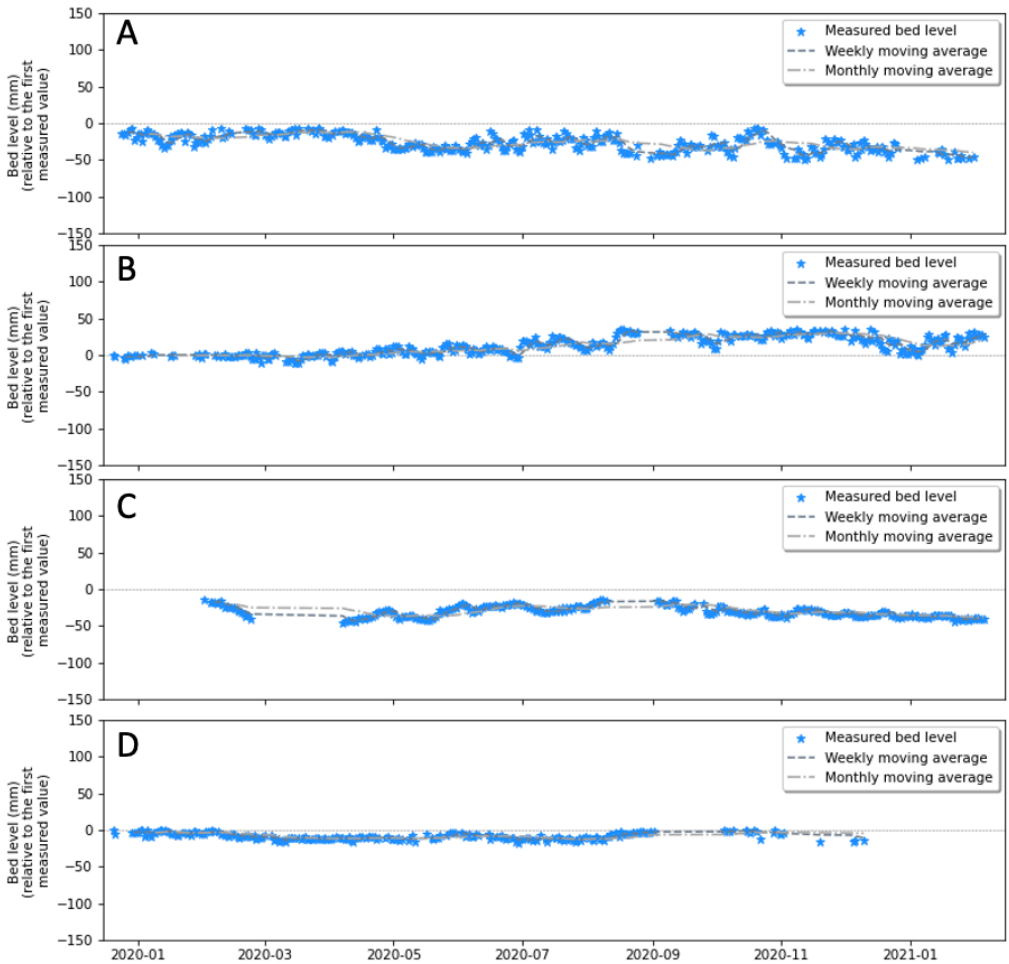

Figure 12 Results of the sediment dynamics at the inner estuarine section for respectively the exposed tidal flat (A), exposed low marsh (B), sheltered tidal flat (C) and sheltered pioneer zone (D).

3.1.2 Santoña Marshes estuary

At the Santoña Marshes, the sensor located at the low marsh (SP-9) got lost during deployment. The sensor deployed in the tidal flat measured for five months (SP-5). The data recorded show a relative stable bed level during the winter and early spring (Figure 13). From May onwards, there was net sedimentation at this location. More bed level dynamic occurred during spring, when bed level showed 5 to 10 mm variability over time (Figure 13).

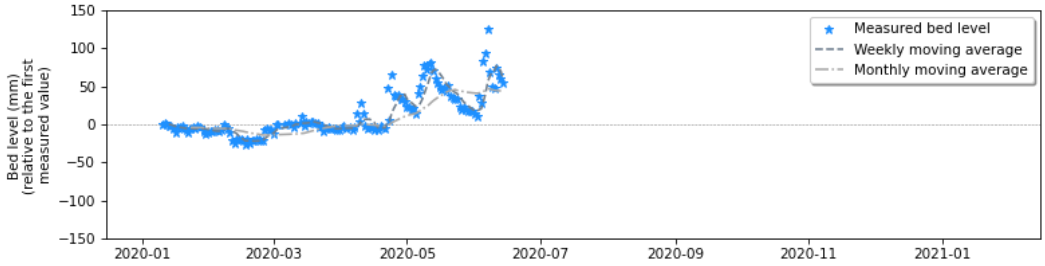

Figure 13 Results of the sediment dynamics at the Santoña marshes for sensor-id SP-5.

3.1.3 Santander Bay

At Santander Bay, two sensors were installed, one at the intertidal mudflat (SP-6) and the other installed at the low marsh (*Spartina spp.*) meadows (SP-7). The one deployed in the low marsh meadow drowned due to unknown reasons. The one deployed at the intertidal mudflat showed a stable bed level with moderate sedimentation during the winter and spring (Figure 14). In May, an extreme sedimentation event occurred, which eroded in the weeks after to the bed level before the event. The general trend tends to be sedimentation.

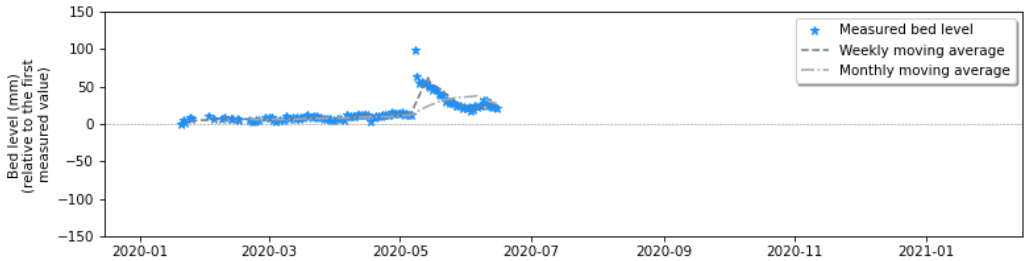

Figure 14 Results of the sediment dynamics at the Santander Bay for sensor-id SP-6.

3.1.4 Oyambre estuary

Five sensors were installed at the Oyambre estuary, of which one got lost. The overall trend is negative for the Oyambre estuary, with an erosion speed of +/- 10 mm/month (Figure 15). More substantial variability in the bed level was measured in the spring months, perhaps caused by the accumulation of green algae.

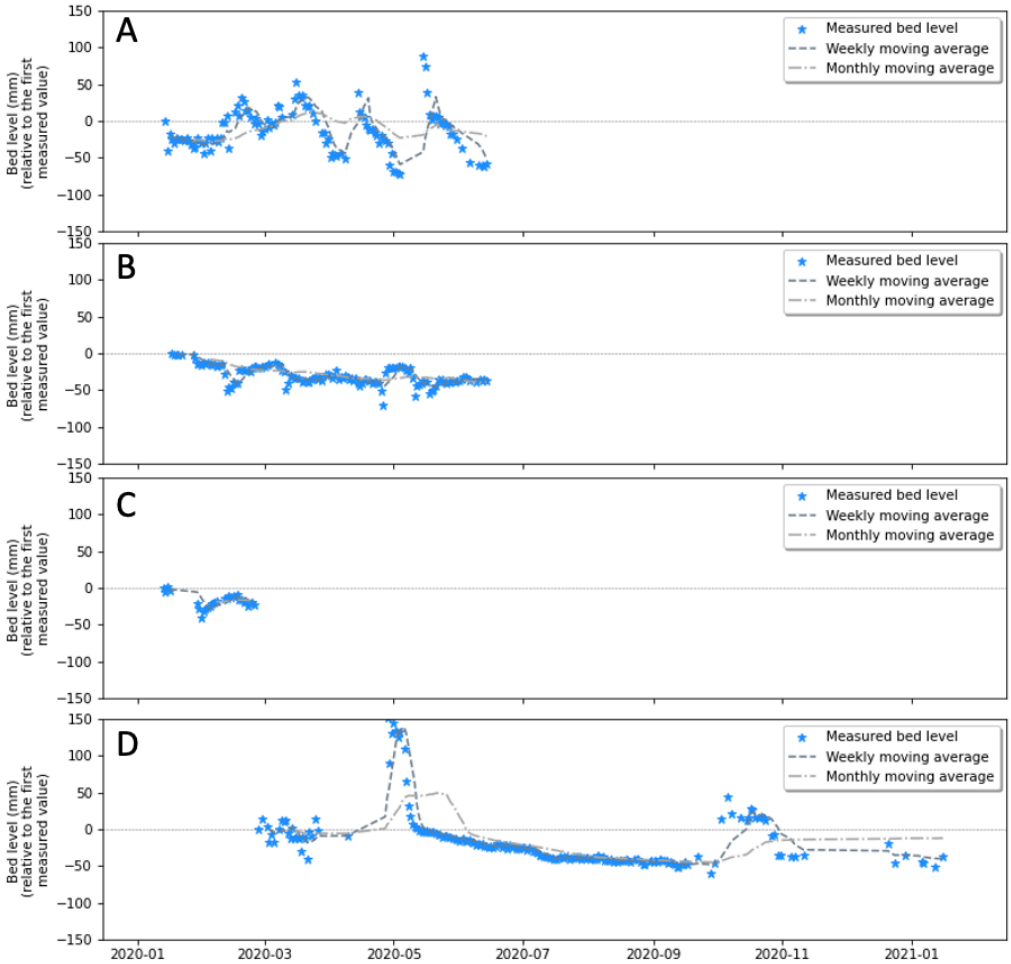

Figure 15 Results of the sediment dynamics at the Oyambre estuary for the tidal flat for the outer section of the east branch (A), tidal flat for the inner section of the west branch (B), tidal flat for inner section of the east branch (C) and the low marsh located at the outer section of the east branch.

The general erosional trend found for the locations in Oyambre estuary can be explained by the fact that just few months before the sed-sensors were installed, a dyke that was restricting the natural tidal flow into the west branch was removed, as part of an implementation action of a previous LIFE project (CONVIVE LIFE). The opening of the dyke has led to a significant change in hydrodynamics in the whole estuary as the tidal prism has increased. It is likely that hydrodynamic energy has increased too, favoring sediment erosion.

3.1.5 Mondego estuary

In the Mondego estuary, different trends were found across estuarine sections and communities within each section. At the inner estuarine section, the tidal flat showed a significant sedimentation event with almost 10 cm increased during April and May and then stabilized to a positive sedimentation trend of +/- 2 cm per month (Figure 16 A). The adjacent low marsh community (*Bolboschoenus maritimus)* showed a more stable bed-level with almost no variability during the measurement interval (Figure 16 B). At the middle estuarine section, no bed level variability was found neither in the low marsh at the control site nor in the bare

tidal flat from the site to restore (Figure 16 C, Figure 16 D). At the outer estuarine section, a sedimentation trend was found in the intertidal seagrass meadows (*Zostera noltei*) during spring and early summer (Figure 16 E) whereas the bare tidal flats showed an erosional trend during winter months and stabilized during spring and early summer (Figure 16 F).

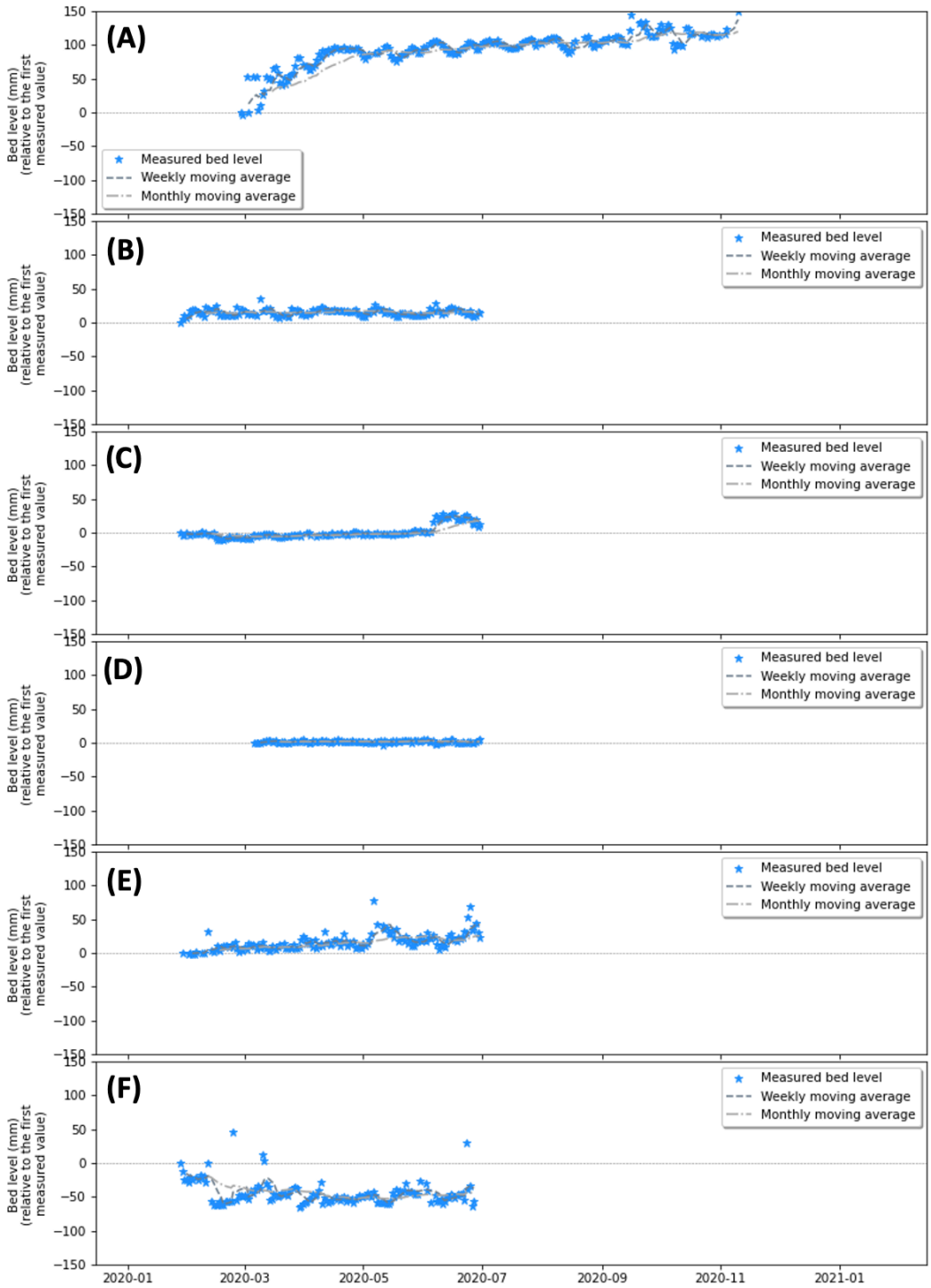

Figure 16 Results of the sediment dynamics at the Mondego estuary for respectively sensor at the bare tidal flat of the inner estuary (A), the Bolboschoenus maritimus marsh of the inner estuary (B), the tidal flat to restore of the mid estuary (C), control Bolboschoenus maritimus marsh of the mid estuary (D), Zostera meadow of the outer estuary (E), bare tidal flat of the outer estuary (F).

3.1.6 Storm sensitivity and general trends.

The morphodynamics footprint analysis conducted for each SED-sensor data (Annex I) revealed the wind conditions under which most bed level dynamics occur at the locations of study (Table 3).

SENSOR ID / TYPE	*Region*	*Estuary*	*Intertidal zone*	*Location*	*Other features*	*Storm sensitivity*	*Sediment dynamics*
NL-1	Zeeland	WS	Pioneer zone	Outer	Exposed	Erosion during storms	Stable
NL-2	Zeeland	WS	Tidal flat	Outer	Exposed	Erosion during storms	Erosion
NL-3	Zeeland	WS	Low Marsh	Outer	Sheltered	No trends	Stable
NL-4	Zeeland	WS	Tidal flat	Outer	Sheltered	No trends	Erosion
NL-5	Zeeland	WS	Low marsh	Inner	Exposed	Erosion during storms	Sedimentation
NL-6	Zeeland	WS	Tidal flat	Inner	Exposed	Erosion during storms	Erosion
NL-7	Zeeland	WS	Low marsh	Inner	Sheltered	No trends	Stable
NL-8	Zeeland	WS	Tidal flat	Inner	Sheltered	No trends	Erosion
SP-1	Cantabria	OY_east	Seagrass meadow	Outer		No trends	Stable
SP-2	Cantabria	OY_west	Tidal flat	Inner		No trends	Erosion
SP-3	Cantabria	OY_east	Tidal flat	Inner		No trends	Erosion
SP-4	Cantabria	OY_east	Low marsh	Outer		No trends	Erosion
SP-5	Cantabria	SM	Tidal flat			Constant sedimentation	Sedimentation
SP-6	Cantabria	SB	Tidal flat			Sedimentation during storms	Sedimentation

SENSOR ID / TYPE	Region	Estuary	Intertidal zone	Location	Other features	Storm sensitivity	Sediment dynamics
SP-7	Cantabria	SB	Low marsh			Measurements failure	
SP-8	Cantabria	OY_west	Tidal flat	Outer		Lost	
SP-9	Cantabria	SM	Low marsh			Lost	
P-1	Coimbra	MN	Tidal flat	Inner		Sedimentation during storms	Sediment ation
P-2	Coimbra	MN	Low marsh	*Inner*		No trends	Sediment ation
P-3	Coimbra	MN	Tidal flat (pilot restorati on site)	Mid		Erosion during storms	Stable
P-4	Coimbra	MN	Low marsh	*Mid*		Erosion during storms	Stable
P-5	Coimbra	MN	Seagrass	*Outer*		No trends	Stable
P-6	Coimbra	MN	Tidal flat	Outer		Sedimentation during storms	Erosion

Table 3 Sediment dynamic trends of each individual site.

3.2 Historical sediment accretion rates

Historical sediment accretion rate could be obtained for 15 out of the 21 cores sampled. No core from the Western Scheldt could be dated as ^{210}Pb profiles suggest a no net accumulation along time (Annex I). Thus, historical sediment accretion rates could not be obtained at the Western Scheldt. Dating was not possible for other two cores analyzed: one sampled in a low marsh community in Santander Bay and the other one sampled in a tidal flat at the west branch of Oyambre estuary (Table 4).

Region	Estuary	Location	Habitat	Dominant genera	Core code	SAR cm y-1 since 1950
Zeeland	WS	Inner_ exposed	High marsh	*Puccinellia spp., Halimione spp.*	WS2A1	no net sedimentation
		Outer_ exposed	Pioneer zone	*Salicornia spp.*	WS3B1	no net sedimentation
		Outer_ sheltered	High marsh	*Spartina spp.*	WS4A1	no net sedimentation
Cantabria	SB	Inner	High marsh	*Juncus spp., Halimione spp.*	BS1A1	0.21 +- 0.01

Region	Estuary	Location	Habitat	Dominant genera	Core code	SAR cm y-1 since 1950
		Outer	Low marsh	*Spartina spp.*	BS1B1	0.20 +-0.02
			High marsh	*Halimione spp.*	BS2A1	0.11 +-0.02
			Low marsh	*Sarcocornea spp.*	BS2B3	mixing
	SM	Outer	High marsh	*Halimione spp.*	MS2A3	0.26 +-0.04
			Low marsh	*Spartina spp.*	MS2B2	0.15 +- 0.02
			Seagrass	*Zostera spp.*	MS2C3	0.26+-0.01
	OY_east branch	Inner	High marsh	*Halimione spp, Juncus spp.*	RM3	0.03 cm y-1*
			Tidal flat	Unvegetated	RP3	0.07 +-0.00
		Outer	High marsh	*Halimione spp., Juncus spp.*	NM3	0.29+-0.01
			Seagrass	*Zostera spp.*	NP2	0.31 +-0.02
	OY_west branch	Inner	High marsh	*Halimione spp, Juncus spp.*	A2M3	0.35 +- 0.02
			Tidal flat	Unvegetated	A2P3	mixing
		Outer	High marsh	*Halimione spp., Juncus spp.*	A1M1	0.06 +- 0.01
			Tidal flat	Unvegetated	A1P1	0.26
Coimbra	MN	Mid	Tidal flat	Unvegetated	MM2C1	0.35 +- 0.02
		Outer	Low marsh	*Spartina spp.*	MM1A1	0.24+-0.02
			Seagrass	*Zostera spp.*	MM1B1	0.47 +- 0.02

**to take with caution due to intense mixing along the sediment depth profile*

Table 4 Historical (since 1950) sediment accretion rates obtained throgh ^{210}Pb analysis of soil core profiles.

The cores that could be dated showed an average sediment accretion rate of the habitats examined of 0.24 ± 0.03 cm y^{-1}.

Sediment accretion rate differed across habitats examined and locations within the estuary (Table 4).

In general, sites located at the inner sections of the estuary showed higher sediment accretion rates than locations at the outer estuarine sections, for all estuaries examined, particularly when comparing across same habitats. In addition, for those locations where more than one habitat has been sampled, the higher sediment accretion rates were found in habitats occupying lower intertidal levels, where the frequency and duration of inundation periods is higher. Such is the case of the outer location of the west branch and both the inner and outer sections of the east branch of the Oyambre estuary and the outer section of the Mondego estuary. In Santoña Marshes, low intertidal seagrass meadows showed equal sediment accretion rate than that found for the high marsh community (0.26 cm y^{-1}) whereas the low marsh, located at intermediate position with regard to the tidal range, showed a lower sediment accretion rate (0.15 cm y^{-1}).

The lowest sediment rates (one order of magnitude lower than other sites) were found in three habitats sampled at the Oyambre estuary. In this estuary the tidal and riverine flow have been intensively modified by human interventions since mid XIX. Significantly lower vertical sediment accretion rates were found in the two habitats, high marsh (core RM3) and unvegetated tidal flat (core RP3), sampled at the inner section of the east branch of the Oyambre estuary. Tidal flow to this section of the estuary was highly restricted since 1850 until 2009 by a tidal mill. In particular, from 1985 to 2009 the tidal flow was completely restricted, and the area turned into a low salinity lake until 2009, when the tidal mill was removed and natural tidal range was recovered. The restoration of the natural tidal regime enhanced the restoration of saltmarsh vegetation communities. The river flowing to this area (Río Turbio) is a relatively small creek compared to rivers flowing to other estuaries of study, and its flow is also partially restricted through an internal bridge. Thus, the input of sediment through the river into the intertidal habitats is low too. Thus, the significantly lower sediment accretion rates found in both the high marsh and the tidal flat of this area is likely due to a combination of low sediment input from the river and the historical restriction of the tide. Although when this area was sampled (in 2019) the tide had flowed naturally for already 10 year since the tidal mill was retrieved, sediment accretion rates are still low.

The other habitat showing a low sediment accretion rate is a high marsh area located at the outer section of the west branch of Oyambre estuary (core A1M1). In 1945 this area was desiccated through the building of a dyke and transformed into a eucalyptus plantation. In 1995 the dyke broke, and the area was partially inundated, leading to the dead of the trees and the formation of saltmarsh communities. By the time this area was sampled, the tidal flow into the area was still highly restricted to low intertidal and subtidal zones. Thus, high marsh communities in this area show significantly lower sediment accretion rates compared to those found in natural sites (where tidal flow is not restricted) whereas the adjacent tidal flat sampled (core A1P1) showed similar sediment accretion rates to those found in other tidal flats.

Surprisingly, the high marsh communities located at the inner section of the west branch (core A2M3) showed a comparable sediment accretion to that found in other high marsh communities, despite also being affected by the same dyke as the outer section, and an additional dyke that existed inner in this branch of the estuary between 1900 and 2009. The most plausible explanation is this area been subjected of an intense surficial runoff as a consequence to geomorphological characteristic of this basin.

4 MAIN CONCLUSSIONS

4.1 Regarding to present sediment bed-levels

The results obtained through the analysis of SED-sensors data indicate a high variability in bed level dynamics and storm sensitivity across habitats and locations, regarding exposure to wind exposure and proximity to estuary mouth.

In general, bed level dynamics tended to be higher in inner estuarine sections compared to outer estuarine sections and in exposed sites compared to sheltered sites. In addition, bed level dynamics were higher in unvegetated tidal flats compared to adjacent low marshes, located at a higher level with regard to the tidal range and covered by vegetation.

Some particularities were found for each of the estuaries of study:

- In the Western Scheldt estuary, the sheltered locations show no sensitivity to storms and low seasonal variation in bed-level dynamics during the spring and summer. In contrast, the exposed locations show eroding trends during storm conditions but locations seem to have a general positive trend in bed level. During the measurement period, the locations were exposed to significant more storms which can interfere with the measured sediment trend by the SED-sensors.
- In Oyambre estuary, most locations within both branches of the estuary (east and west) show erosional trends except the outer location in the east branch occupied by an intertidal seagrass (*Zostera noltei*) meadow, that was stable. The erosional patterns found in this estuary are likely reflecting the recently increase in hydrodynamic conditions favored by the opening of an ancient dyke located at the mouth of the west branch. The locations examined in Oyambre estuary show no sensitivity to storms.
- In the other two estuaries of the Cantabria region, Santander Bay and Santoña Marshes estuaries, only data from one sed-sensor could be obtained. In Santander Bay, an increase in bed level due to sedimentation occurs during storm conditions whereas the site in Santoña shows a continuous sedimentation trend.
- In the Mondego estuary, inner estuarine locations show general sedimentation trend whereas the tidal flat at the outer estuarine section showed a general erosional trend. Habitats at the middle estuary and the seagrass meadow located at the estuary mouth showed no significant trend in the bed level. Habitats respond differently to storms (storm sensitivity). Sedimentation during storms occurs in the tidal flats at the inner and outer estuarine sections whereas erosion during storms occurred at both tidal flat and high marsh community at the middle estuarine section. This section of the estuary is exposed to high hydrodynamic energy but sed-sensors were located within two embankments surrounded by the reminders of stone walls of an ancient salt ponds. The results found for these sed-sensors suggest that, under average conditions, these

locations remind protected from water current and no erosion occurs. On the contrary, under storm conditions, the stone walls are not enough to protect these areas from erosion. The results found for this area within the Mondego estuary are highly relevant for the design of the restoration action it will be implemented within this project under action C2. Bed level in vegetated communities at the inner (low marsh) and outer estuarine sections (seagrass meadow) did not respond to storms.

- Unfortunately, the project collected relative shorter time series of sediment dynamics for some of the locations due to logistic and technical problems (e.g. COVID-restrictions, frost, storms).

4.2 Regarding to historical sediment accretion rates:

The habitats examined showed an average sediment accretion rate of 0.24 ± 0.03 cm y^{1}. Yet, sediment accretion rate differed across habitats and estuarine locations.

- Sites located at the inner sections of the estuary tended to show higher sediment accretion rates than locations at the outer estuarine sections, reflecting the relevant role of rivers as sediment suppliers to estuarine habitats.
- Higher sediment accretion rates were found in habitats occupying lower intertidal levels, where the frequency and duration of inundation periods is higher.
- The restriction of the tide and river flow by historical human interventions, lead to a decrease in sediment vertical accretion, particularly in high marsh communities, limiting their capacity to path with sea level rise.
- The recovery of natural tidal regime enhances the restoration of saltmarsh vegetation in a short time period (5~10 years) but no changes in sediment accretion rates have been detected yet.

4.3 General conclusion:

- Although results provided SED-sensors (i.e. bed level dynamics) and soil cores dating (i.e. long-term vertical accretion rates) are not directly comparable due to differences in time scales encompassed (daily, seasonal *vs.* years, decades). Yet, we found that, in general, lower intertidal areas, where the frequency and duration of inundation periods is higher, experience higher bed level dynamics and higher long-term vertical accretion rates. Similarly, where the natural tidal regime is restricted, lower bed level dynamics and lower long-term sediment accretion rates were found, compared to natural sites. These results can suggest that the amplitude of bed level dynamics are related to the vertical accretion rates on the long-term and thus to the capacity of estuarine habitats to path with seal level rise.

- The results found also highlights that, despite identifying some general trends across estuarine locations and habitats, all estuaries are unique, and characterized by particular hydrodynamic conditions and subject to different human interventions. Thus, is order to understand the resilience of estuarine habitats to sea level rise at a level that allows management, monitoring of bed level dynamics and sediment accretion rates at each estuary is highly recommended.

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6 ANNEX

6.1 Morphodynamical footprint analysis of storm sensitivity.

Storm sensitivity for each SED-sensor location was analyzed through a morphodynamical footprint analysis, based on the time series results and wind data. The data collected by the SED-sensor deployed in the tidal flat at the outer section of the Western Scheldt estuary (NL-2) is used as an example.

The first step was to find the relationship between the daily bed level change and daily wind speeds (Figure 1). For this location, the sediment dynamics are stable and limited to several mm (Figure 1). Dependent on the site there can be a clear difference between lower and higher wind velocities and the related bed level change. However, in most cases it is hard to find a clear relation between bed level dynamics and wind based on these plots.

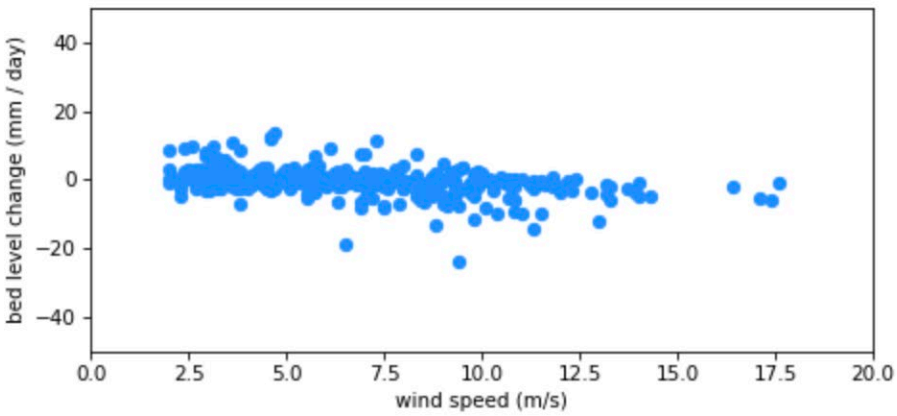


Figure 1. Relation between wind speed and bed level change for the outer estuary exposed tidal flat in the Western Scheldt estuary.

To study in more detail the relation between bed level dynamics and wind, we analyse the relationship between the cumulative bed-level change (blue line) and the trend in bed-level change (orange line) against the wind speed (Figure 2). Figure 2 shows that at lower wind speeds, the tendency is sedimentation; however, from 7.5 m/s, the tendency changes to become more erosive. According to figure 1-10 (main document), during the deployment period, this location has been exposed to higher storm conditions than average conditions. Then it is likely that during the measurement, the bed-level experienced more significant negative bed-level changes than under the average conditions. This result highlights the importance of understanding the storm sensitivity of the site above the available time series of sediment dynamics. In general, it presents a part of the hydrodynamic behavior of the location.

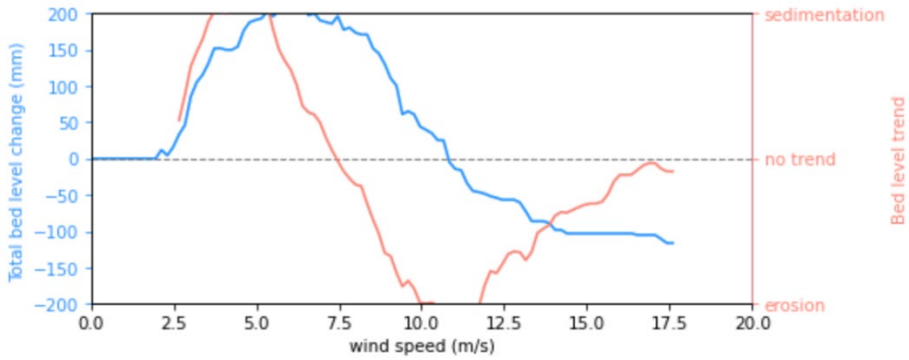


Figure 2. The cumulative bed level change (blue line) and the trend (orange line) against wind speed for the outer estuary exposed tidal flat in the Western Scheldt estuary.

The cumulative bed level change does not take into account when erosion or sedimentation is likely to happen and if wind conditions play a major or minor role. Figure 3, shows the impact of wind on the sedimentation or erosion. The 1:1 line indicate when sediment dynamics are independent of wind speed (no correlation), as 50% of the wind speed can explain 50% of the bed level change. Curves above the 1:1 line are dominated by calm conditions, in this example 50% of the lower wind speeds explains almost 70% of the sedimentation. On the other hand, storm condition dominated curves are under the 1:1 line, like in this example 50% of the lower wind speeds can only explain 15% of the erosion. The example shows that an erosive trend dominates the sediment dynamics during storm conditions. The storm sensitivity can have different scenarios and strengths per location, figure 1-20 explain the possible scenarios.

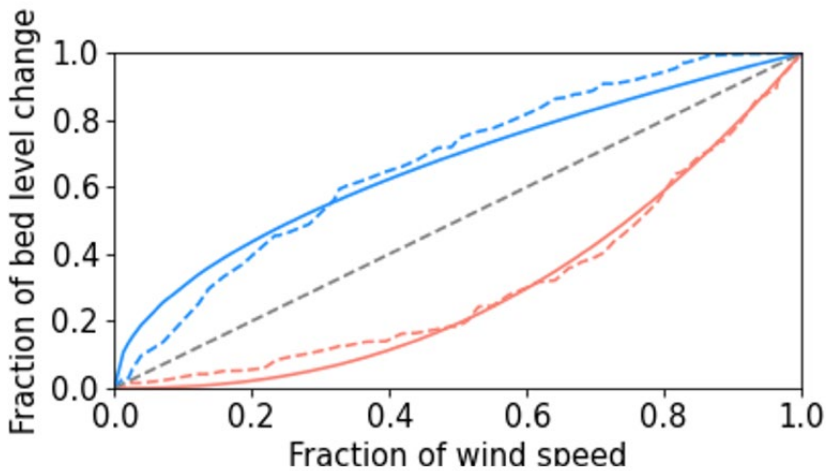


Figure 3. The storm sensitivity for the erosion and sediment dynamics of the tidal flat located at the outer estuarine section of the Western Scheldt estuary. The dashed lines represent the measured data red erosion (in red) and sedimentation (in blue) dynamics, the solid line presents the fitted curve of x^a, a is the variable for the impact of wind.

These analysis was performed for all SED-sensors deployed in the five estuaries of the study (Figure 4)

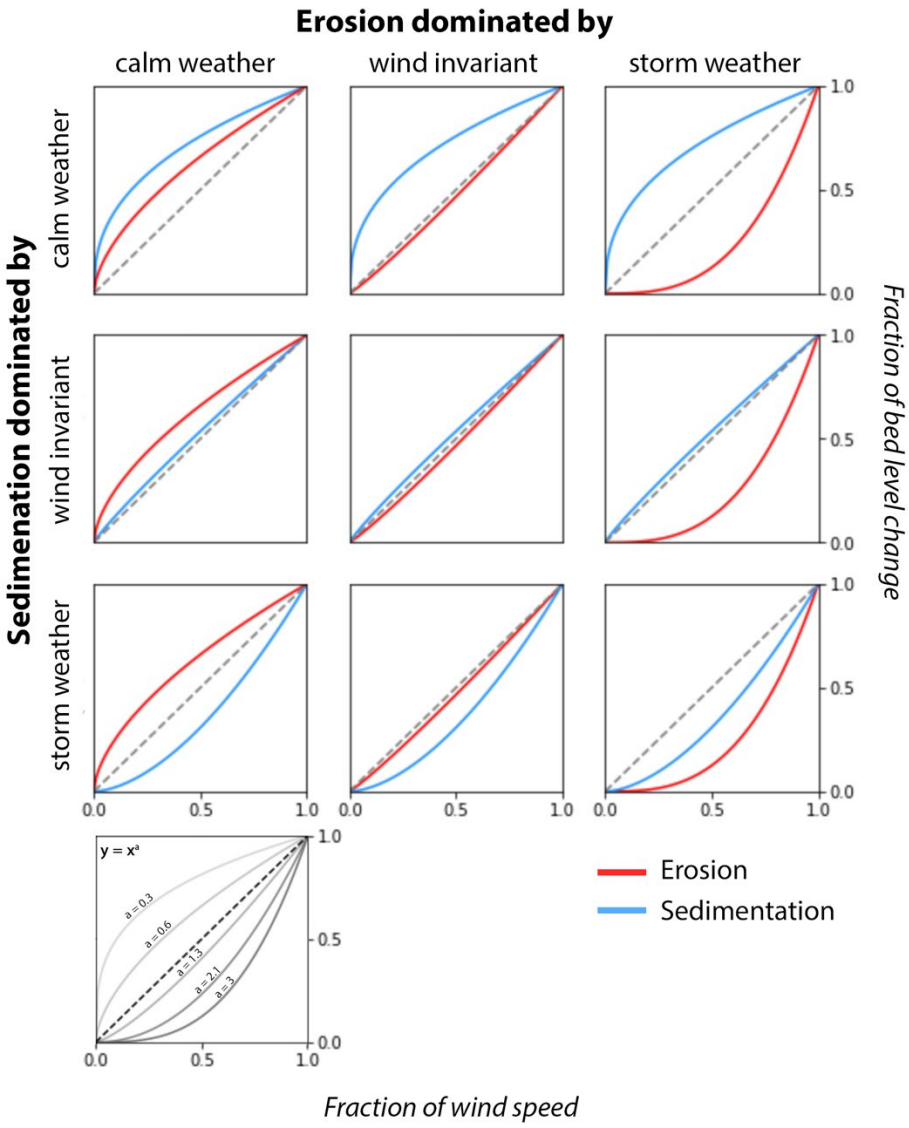
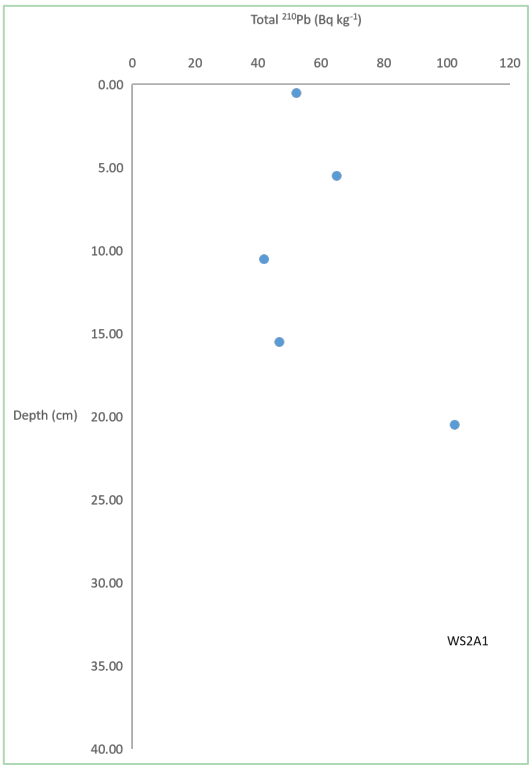
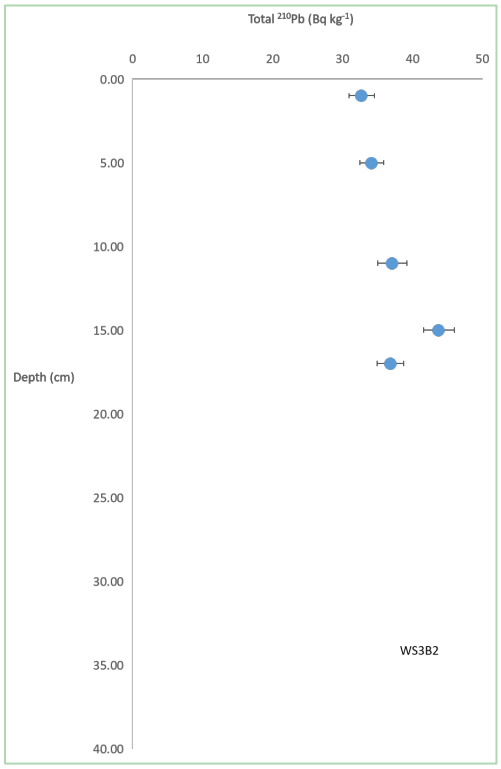


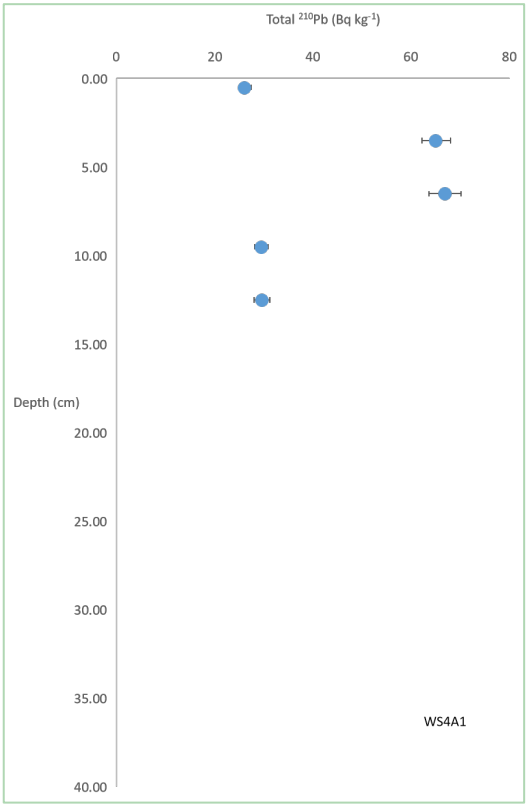
Figure 4. The storm sensitivity scenarios for the bed level dynamics measured with SED-sensors. The left bottom plot, indicates the strength, e.g. a values close to 0 are representative for calm weather dominated dynamics, a values of 1 indicate wind invariant conditions, while a values larger then 1 are an indicator for storm sensitivity.

6.2 ^{210}Pb Activity profiles in sediment cores examined.

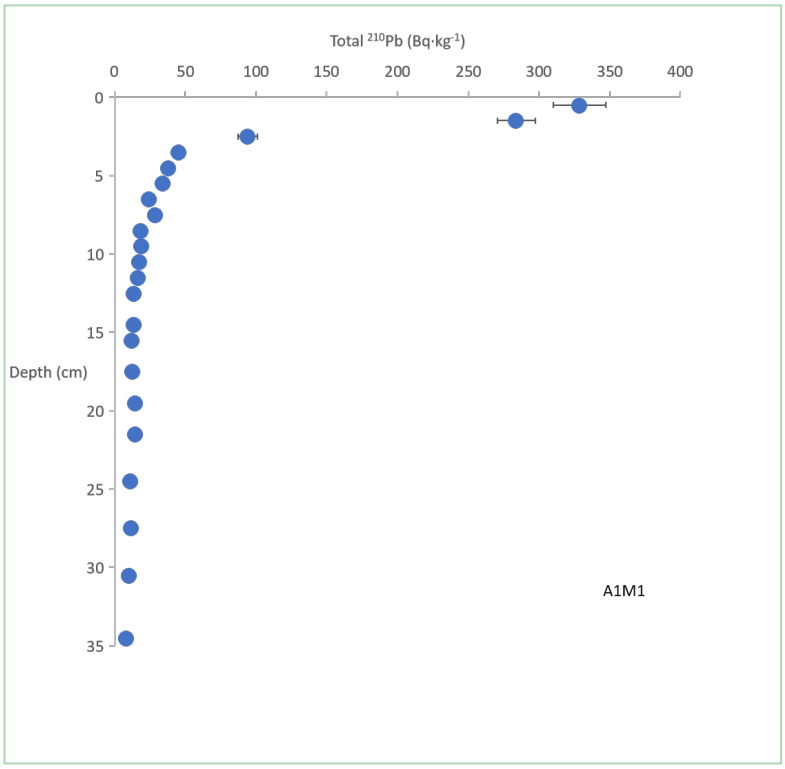
6.2.1 ^{210}Pb Activity profiles of cores sampled in the Western Scheldt (The Netherlands)

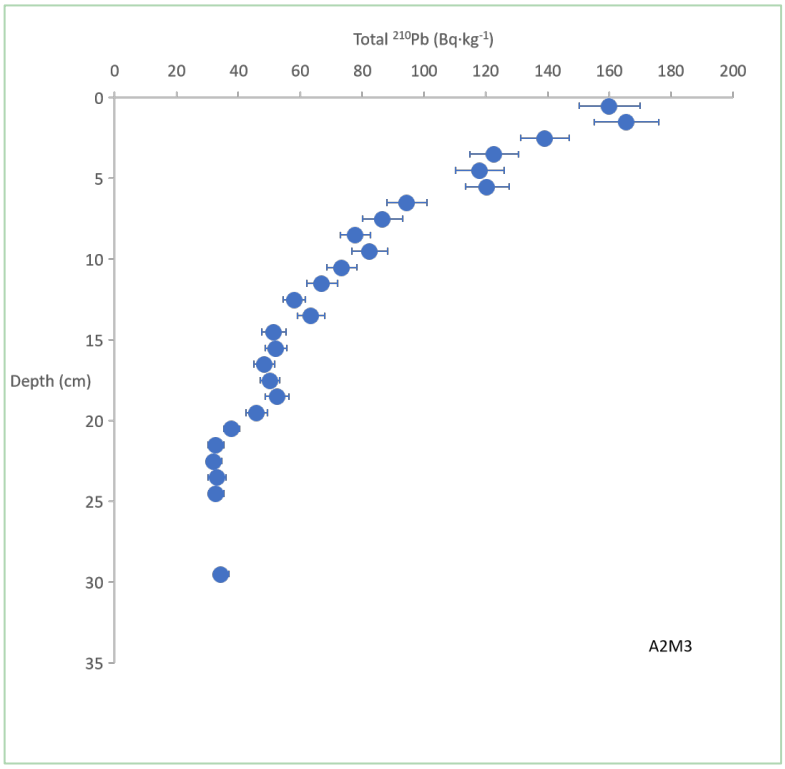


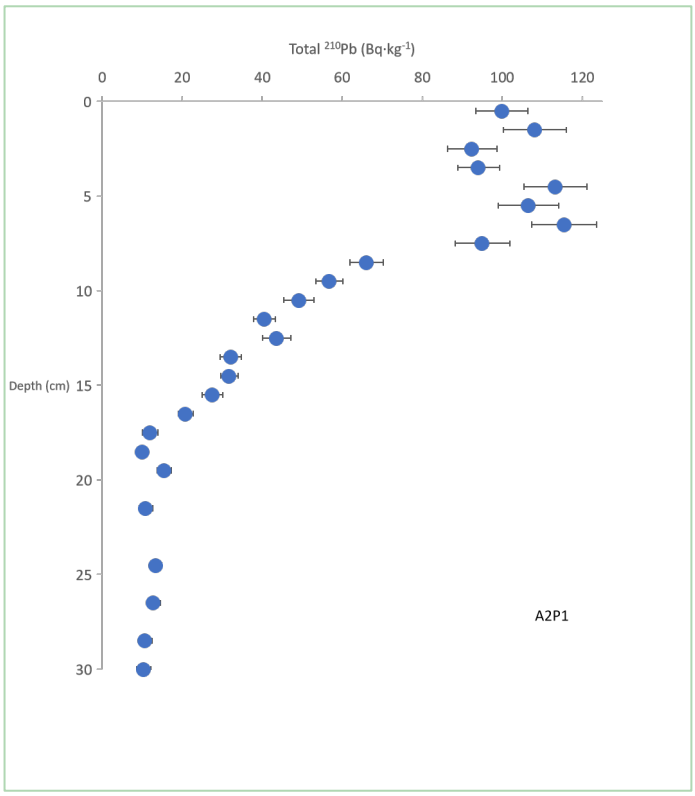


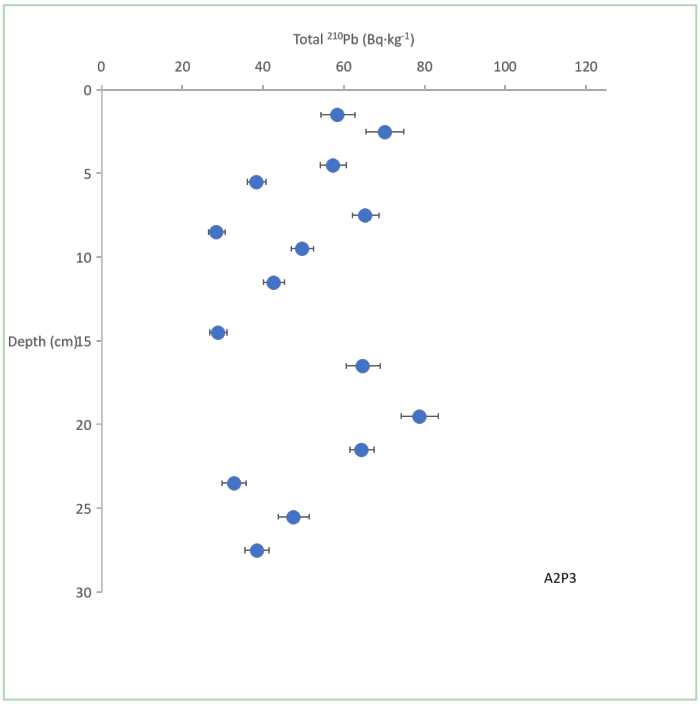


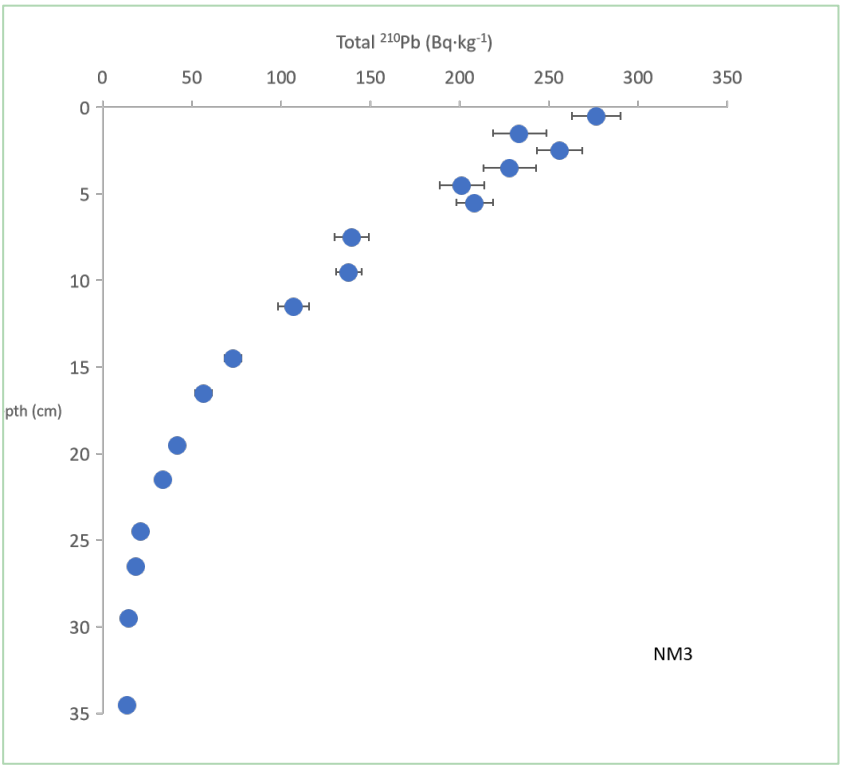
6.2.2 ^{210}Pb Activity profiles of cores sampled in Oyambre estuary (Cantabria)

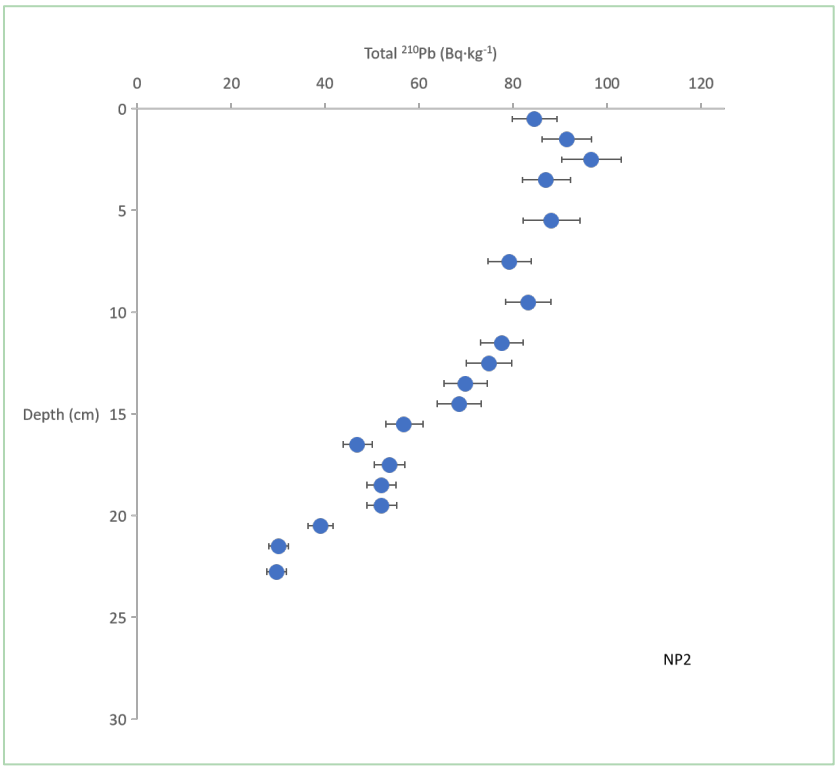


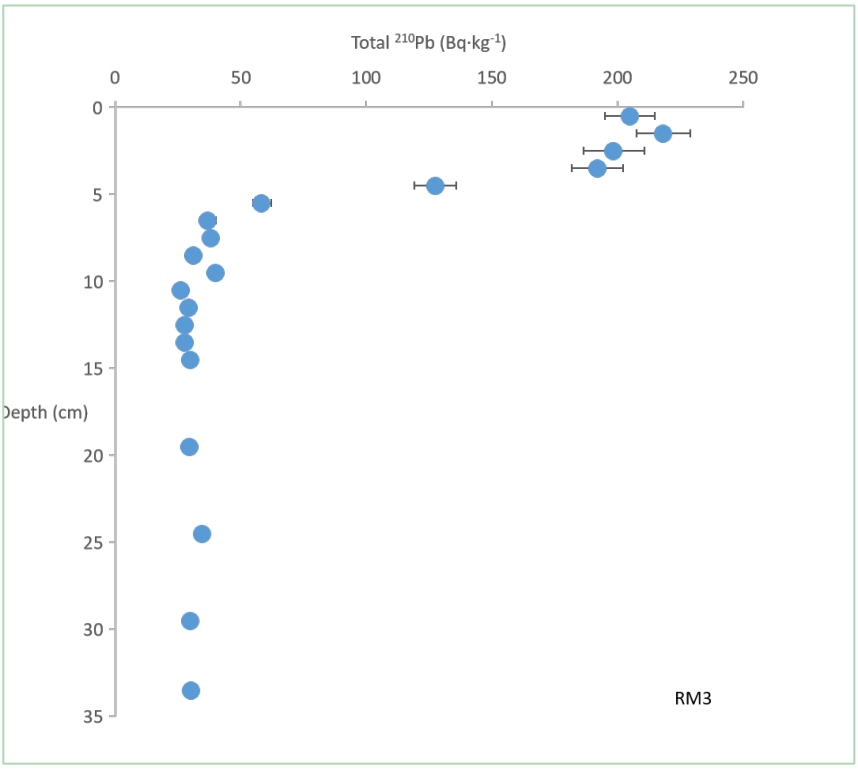


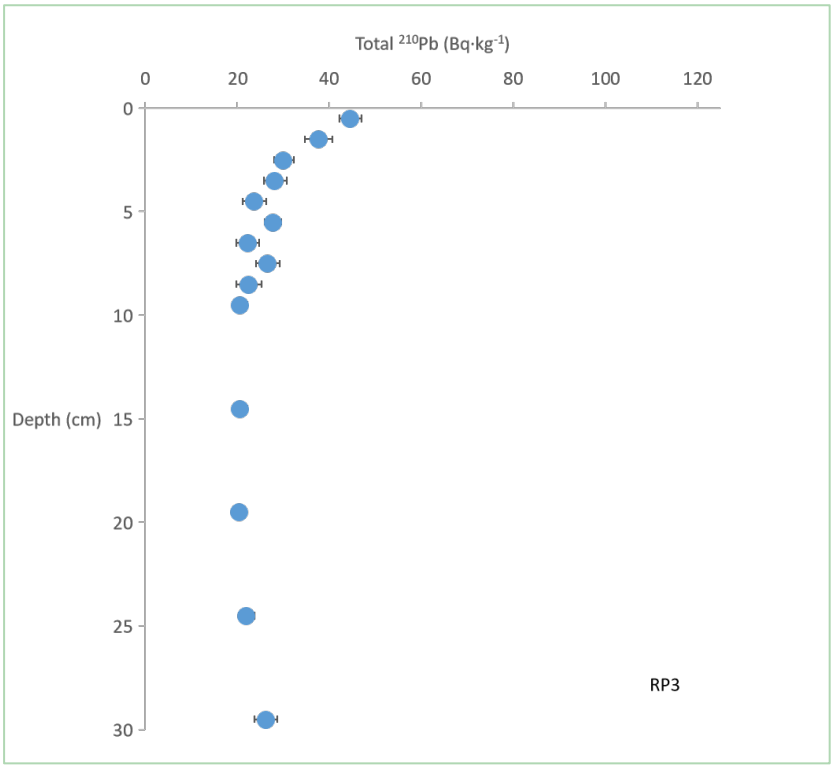

Total ^{210}Pb (Bq·kg^{-1})
0
20
40
60
80
100
120
Depth (cm)
0
5
10
15
20
25
30
A2P1


Total ^{210}Pb (Bq·kg^{-1})
0
20
40
60
80
100
120
Depth (cm)
0
5
10
15
20
25
30
A2P3

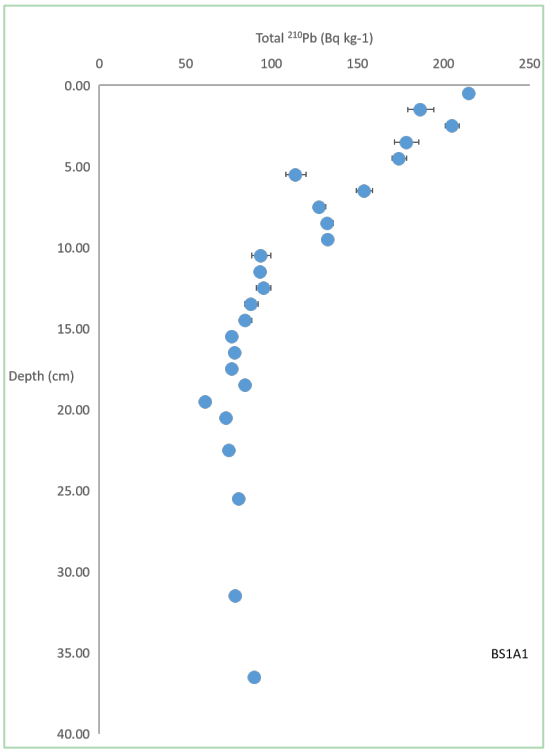

Total ²¹⁰Pb (Bq·kg⁻¹)
0
50
100
150
200
250
300
350
0
5
10
15
pth (cm)
20
25
30
35
NM3

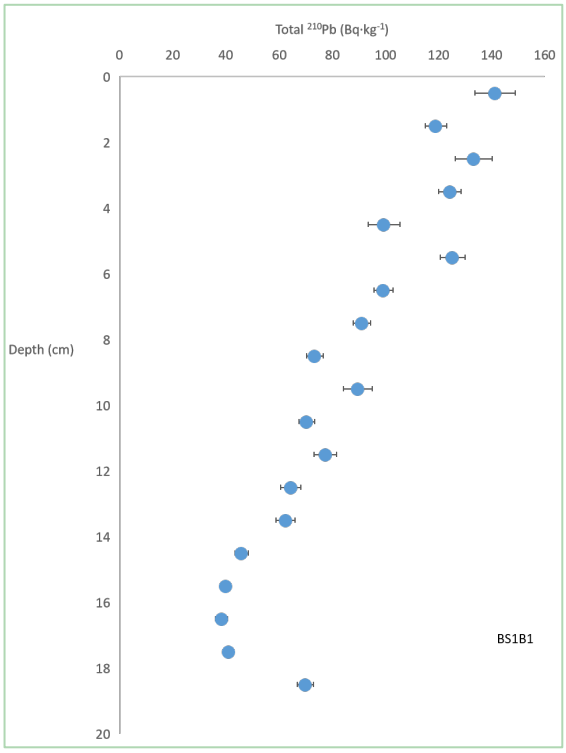

Total ²¹⁰Pb (Bq·kg⁻¹)
0
20
40
60
80
100
120
0
5
10
Depth (cm) 15
20
25
30
NP2

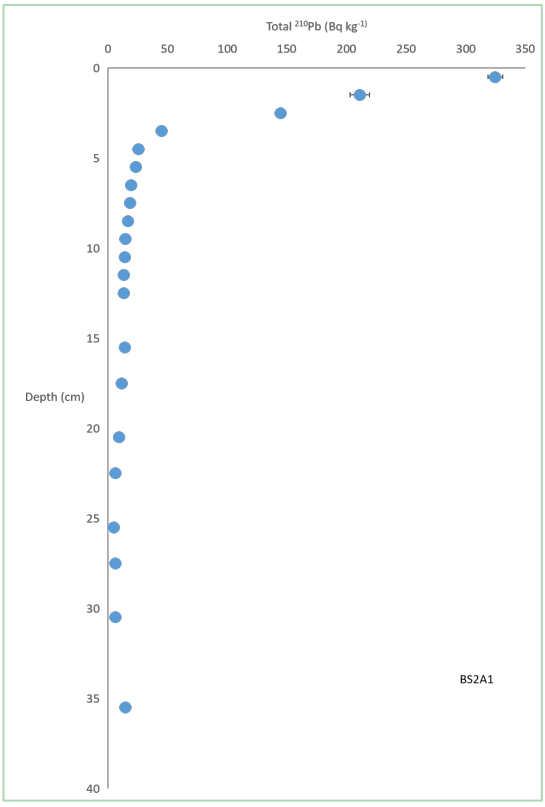

Total ^{210}Pb (Bq·kg^{-1})
0
50
100
150
200
250
0
5
10
15
Depth (cm)
20
25
30
35
RM3

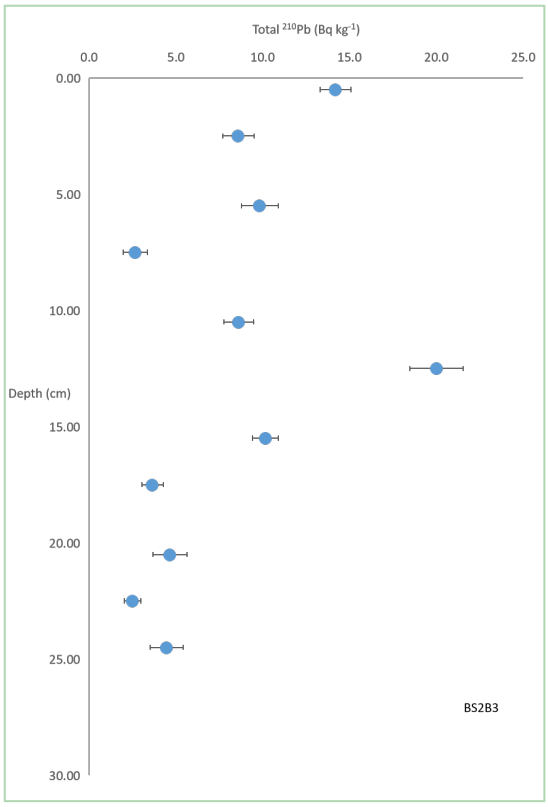

Total ^{210}Pb (Bq·kg^{-1})
0
20
40
60
80
100
120
0
5
10
Depth (cm)
15
20
25
30
RP3

6.2.3 ^{210}Pb Activity profiles of cores sampled in Santander Bay estuary (Cantabria)

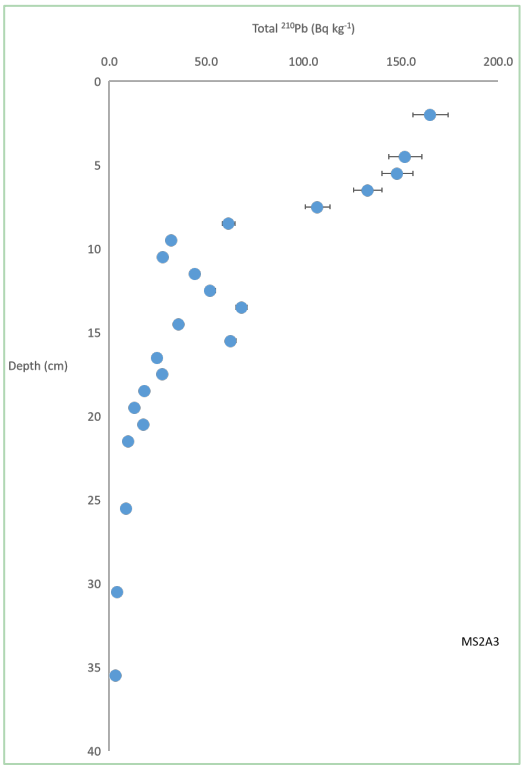


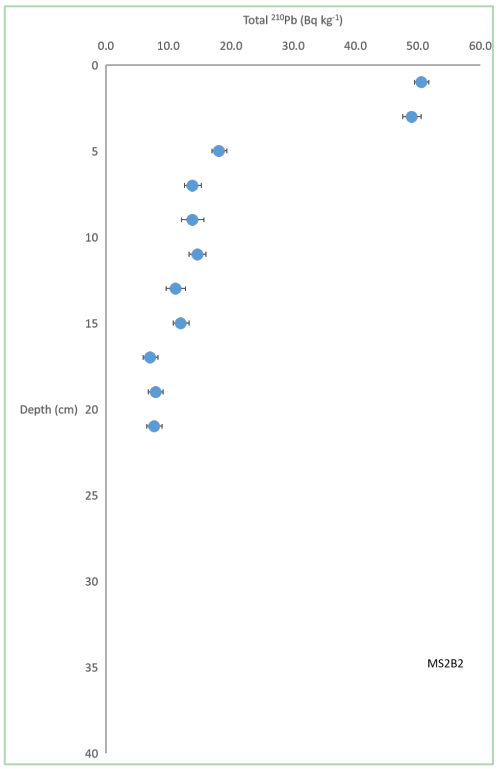


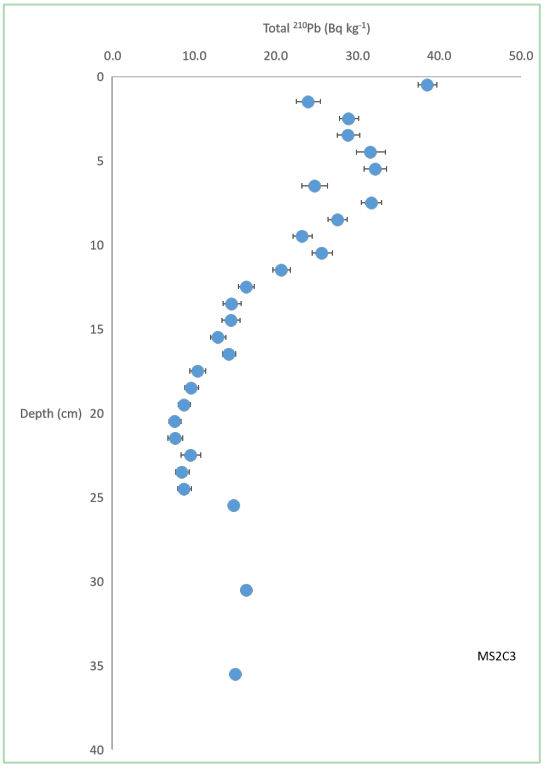




6.2.4 ^{210}Pb Activity profiles of cores sampled in Santoña Marshes estuary (Cantabria)







6.2.5 ^{210}Pb Activity profiles of cores sampled in Mondego estuary (Coimbra)