Different coastal marsh sites reflect similar topographic conditions under which bare patches and vegetation recovery occurs

Chen Wang[1,2], Lennert Schepers[2], Matthew L. Kirwan[3], Enrica Belluco[4], Andrea D'Alpaos[5], Qiao Wang[6,1], Shoujing Yin[1], and Stijn Temmerman[2]

[1]Satellite Application Center for Ecology and Environment, Ministry of Ecology and Environment / State Environmental Protection Key Laboratory of Satellite Remote Sensing, Fengde East Road 4, Beijing 100094, China
[2]Ecosystem Management Research Group, University of Antwerp, Universiteitsplein 1, 2610 Wilrijk, Belgium
[3]Virginia Institute of Marine Science, PO Box 1346, 1375 Greate Road, Gloucester Point, Virginia 23062, USA
[4]Department of Civil, Environmental, and Architectural Engineering, University of Padova, via Loredan 20, 35131 Padua, Italy
[5]Department of Geosciences, University of Padua, Via Gradenigo 6, 35131 Padua, Italy
[6]Faculty of Geographical Science, Beijing Normal University, Xinjiekouwai Street 19, Beijing 100875, P.R.China

Correspondence to: Qiao Wang (wangqiao@mee.gov.cn) and Shoujing Yin (shoujingy@163.com)

Abstract. The presence of bare patches within otherwise vegetated coastal marshes is sometimes considered to be a symptom of marsh die-back and the subsequent loss of important ecosystem services. Here we studied the topographical conditions determining the presence and revegetation of bare patches in three marsh sites with contrasting tidal range, sediment supply and plant species: the Scheldt Estuary (the Netherlands), Venice Lagoon (Italy), and Blackwater Marshes (Maryland, USA). Based on GIS analyses of aerial photos and LIDAR imagery of high resolution ($\leq$ 2 x 2 m pixels), we analyzed the topographic conditions under which bare patches occur, including their surface elevation, size, distance to and connectivity to channels. Our results demonstrate that, for the different marsh sites, bare patches can be connected or unconnected to the channel network, and that the width of the connecting channels increases with the size of the bare patches, in each of the three marsh sites. Further, pixels located in bare patches connected to channels occur most frequently at the lowest elevations and farthest distance from the channels. Pixels in bare patches disconnected from channels occur most frequently at intermediate elevations and distances from channels, and vegetated marshes dominate at highest elevations and shortest distances from channels. Revegetation in bare patches is observed in only one site with the highest tidal range and highest sediment availability, and preferentially occurs from the edges of small unconnected bare patches at intermediate elevations and intermediate distances from channels. Although our study is only for three different marsh sites with large variations in local conditions, such as tidal range, sediment availability and plant species, it suggests that similar topographic conditions determine the occurrence of bare patches. Such insights may inform decision makers on coastal marsh management on where to focus monitoring of early signatures of marsh degradation.

1 Introduction

Tidal marshes are coastal ecosystems that provide many valuable ecosystem services such as fishery production (Barbier et al., 2011), sequestration of CO_2 (McLeod et al., 2011), protection against shoreline erosion and mitigation of flood risks during storm surges (Barbier et al., 2008; Wamsley et al., 2010; Gedan et al., 2011; Temmerman et al., 2013; Temmerman and Kirwan, 2015). However, tidal marshes and their valuable ecosystem services can be lost when marshes die-off, for instance, as a consequence of sea level rise. Large-scale tidal marsh loss by conversion of marshes into bare tidal flats, open water or bare patches within marshes has been reported from different locations around the world (Baumann et al., 1984; Day et al., 2000; Kearney et al., 2002; Carniello et al., 2009; Kirwan and Megonigal, 2013). Bare patches within marshes, which are often covered by standing water and then referred to as pools, ponds (Stevenson et al., 1985) or marsh basins (Mariotti and Fagherazzi, 2013), are a common feature in salt marshes around the world. In some regions, bare patches are dynamic features that develop but also recover and revegetate (e.g., New England; Wilson et al., 2009, 2010, 2014). In other areas, however, bare patches do not revegetate and are causing permanent marsh loss on a large scale (e.g. Mississippi Delta Penland et al., 2000; Morton et al., 2003).

Marsh loss and recovery is of particular concern because there is growing evidence that vegetated marshes and bare flats behave as alternative stable ecosystem states (Fagherazzi et al., 2006; Kirwan and Murray, 2007; Marani et al., 2007; van Wesenbeeck et al., 2008; Marani et al., 2010; D'Alpaos, 2011; McGlathery et al., 2013; Wang and Temmerman, 2013; Moffett et al., 2015; D'Alpaos and Marani, 2016; van Belzen et al., 2017), which implies that recovery after marsh loss would be very difficult (Hu et al., 2015a; van Belzen et al., 2017). Observations have shown that vegetated marshes and bare flats occupy different elevation ranges (Marani et al., 2007, 2010; Carniello et al., 2009; Wang and Temmerman, 2013) and that shifts from the low-lying bare state to the high-elevated vegetated state occur rapidly once a threshold elevation has been exceeded (Wang and Temmerman, 2013). Moreover, models indicate that the system would shift abruptly between the high elevation vegetated state and low-lying bare state when a threshold value is reached in elevation, sediment input, or rate of sea level rise (Fagherazzi et al., 2007; Kirwan and Guntenspergen, 2010; Marani et al., 2010; D'Alpaos et al., 2011; D'Alpaos and Marani, 2016). Previous studies further suggest that the state shift can be irreversible because of a hysteresis effect (Kirwan and Murray, 2007; Marani et al., 2010; Kirwan et al., 2011), where the threshold conditions to revert the ecosystem back to the original state are far more difficult to reach than the threshold conditions that caused the shift (Scheffer et al., 2001; Scheffer and Carpenter, 2003). Field experiments have also demonstrated that vegetation recovery after disturbance is slower under increased tidal inundation, which further suggests the applicability of alternative stable state theory to vegetated and bare areas in intertidal zones (van Belzen et al., 2017).

The two stable states of marshes and tidal flats can be explained by positive feedback mechanisms which are strongly mediated by the presence or absence of marsh vegetation. As long as vegetation is present on the marsh, waves and tidal currents are effectively attenuated by vegetation-induced friction over several meters of continuously vegetated marsh surfaces (Neumeier and Amos, 2006; Mudd et al., 2010; Vandenbruwaene et al., 2011; Yang et al., 2012; Hu et al., 2014).

As a consequence, suspended sediment is deposited on the marsh surface and marshes can maintain a high position in the
tidal frame, even with sea level rise (Kirwan and Guntenspergen, 2010; D'Alpaos et al., 2011; Fagherazzi et al., 2012).
Above- and belowground plant material further contributes to marsh accretion (Nyman et al., 2006; Kirwan and
Guntenspergen, 2012). When vegetation is absent, however, organic matter accumulation is strongly reduced, and increased
tidal currents and waves may prevent sedimentation or even trigger erosion (Fagherazzi et al., 2006; Kirwan and Murray,
2007; Marani et al., 2007; Mariotti and Fagherazzi, 2010; Temmerman et al., 2012). In large lagoons or extensive tidal
basins, the low elevation of the tidal flats is mainly maintained by wave erosion (Fagherazzi et al., 2006; Mariotti and
Fagherazzi, 2010; Hu et al., 2015b). The existence of these two alternative stable states has been empirically observed on the
large scale of whole tidal basins where large areas (~km ȝ of marshes and tidal flats may coexist next to each other (Marani
et al., 2007; Carniello et al., 2009; Wang and Temmerman, 2013). However, the existence of alternative stable states has not
yet been empirically explored to explain marsh loss and recovery by formation and revegetation of bare patches (~10–100
m ȝ, which is addressed in this paper.

Bare patches are defined here as non-vegetated areas in the interior of otherwise vegetated marshes. Here we consider two
types of bare patches: (i) connected bare patches that have a connection to the tidal channel network and (ii) isolated bare
patches that are separated from the channels by surrounding marsh vegetation. Literature suggests that unconnected bare
patches start as areas with vegetation die-off, by increased flooding stress and inadequate drainage, high salinity stress
(DeLaune et al., 1994; Wilson et al., 2009, 2014), coverage by drifted plant material (Harshberger, 1916; Miller and Egler,
1950; Redfield, 1972), physical disturbance by ice, or herbivory by crabs, nutria, muskrats, geese or snails (Harshberger,
1916; Stevenson et al., 1985; DeLaune et al., 1994; Silliman, 2005; Argow and FitzGerald, 2006). Subsequent elevation loss
due to the collapse of the root structure or decomposition and disintegration of soil organic matter can deepen the bare
patches (DeLaune et al., 1994; Wilson et al., 2014). Connected bare patches form by creek-bank erosion at the creek heads
(Kearney et al., 1988) and subsequent connection of channel heads to bare patches (Redfield, 1972) or by expansion of
unconnected bare patches that ultimately reach a channel and become hydraulically connected to the channel network
(Wilson et al., 2014; Mariotti, 2016).

Nevertheless, it is not fully understood under which topographic conditions connected and unconnected bare patches occur,
and especially under which conditions they recover through re-establishment of vegetation. For example, we may
hypothesize that unconnected bare patches are buffered from tidal currents and waves by the surrounding marsh vegetation,
and therefore are less prone to erosion and more suitable for vegetation recovery. On the other hand, they might also receive
less sediment input since sediment is efficiently trapped by the surrounding vegetation buffer (Mudd et al., 2010; Moskalski
and Sommerfield, 2012). The opposite applies for connected bare patches: they receive direct sediment input through the
channels, but experience higher flow velocities that may cause sediment resuspension and erosion. Some studies show that
marsh plants might recolonize bare patches when they become connected, drain and if vertical accretion elevates the bare
patches sufficiently for plant establishment (Redfield, 1972; Wilson et al., 2009, 2014). However, higher flow velocities and

therefore a decrease in accretion by reduced mineral sediment deposition or erosion may inhibit the recovery of vegetation in connected bare patches (DeLaune et al., 1994; Mariotti, 2016).

Hence, despite the fact that bare patches are often recognized as symptoms of marsh loss (Kearney et al., 1988; DeLaune et al., 1994; Fagherazzi et al., 2013; Mariotti and Fagherazzi, 2013; Wilson et al., 2014; Mariotti, 2016), there are relatively few studies on the dynamics of bare patches. For example, the modelling study by Mariotti (2016) simulates that pond expansion is favored under conditions with low tidal range, low sediment supply and high relative sea level rise. Apart from this study, there is poor empirical evidence on the conditions that determine the presence and/or recovery of bare patches, especially across marsh sites that differ in characteristics such as tidal range, sediment supply and plant species. In this paper, we first study the topographic conditions determining the presence and dynamics of bare patches. Next, we study the topographic conditions determining the marsh vegetation recovery (i.e., the re-establishment of vegetation) in bare patches. To identify the topographic conditions determining the presence of bare patches, we compared the surface elevation, bare patch size and distance to channels for connected and unconnected bare patches in three different sites, located in the Scheldt Estuary (a river-dominated estuary in the Netherlands, 4.8 m tidal range), Venice lagoon (a back-barrier lagoon in Italy, 1.0 m tidal range) and Blackwater Marshes (a submerging tidal marsh, in Maryland, USA, < 0.5 m tidal range). To identify the conditions determining the revegetation of bare patches, we carried out a time series analysis in the Scheldt Estuary, the only site where revegetation was observed and searched for relations between the rate of revegetation of bare patches and topographic conditions including surface elevation, distance to channels and the width of connecting channels. Our hypotheses are that (1) bare patches across all three study sites are found at similar elevation relative to the tidal frame, distance to tidal channels and degree of connectivity to tidal channels; (2) low elevation relative to the tidal frame and wide channel connection lead to larger bare patches that are more difficult to revegetate.

2 Study area

2.1 Saeftinghe marsh, Scheldt estuary, the Netherlands

The Scheldt estuary is a river-dominated estuary located in the southwest of the Netherlands and the northwest of Belgium (Fig. 1). The Saeftinghe marsh (51.33° N, 4.17° E) is a 3000 ha brackish tidal marsh. It is subject to a semi-diurnal tidal regime with a local mean tidal range of 4.88 m, a salinity of 5–18 PSU, and a mean suspended sediment concentration (SSC) of 30–60 mg L^{-1} (Temmerman et al., 2003a; van Damme et al., 2005). In the last 80 years, a long-term rise of mean high water level (MHWL) was observed in the Saeftinghe marsh at a rate of 5.7 mm/yr, while the vegetated marsh regions expanded in area and increased in elevation steadily and continuously (Wang and Temmerman, 2013). Dominant plant species include *Spartina anglica*, *Salicornia europaea*, *Scirpus maritimus, Elymus athericus*, and *Phragmites australis*. Marsh vegetation is observed between -2 m and +1 m relative to MHWL (Wang and Temmerman, 2013). Parts of the Saeftinghe marsh have been converted to bare patches. This is partly attributed to geese grubbing for below-ground tubers

(Elschot et al., 2017). In addition, bare patches are formed at places with poor drainage after high overmarsh tides, especially near the head of the smallest tidal channels. This is the case in the selected study site, covering an area of 35 ha (Fig. 1).

2.2 San Felice marsh, Venice lagoon, Italy

The Venice lagoon is a back-barrier tidal lagoon situated in the northeast of Italy, characterized by a micro-tidal semi-diurnal regime with a mean tidal range of about 1.0 m (Day et al., 1999) and a maximum tidal range of 1.5 m (Rinaldo et al., 1999a, 1999b; Marani et al., 2007). The long-term rate of relative sea level rise varies around 3–4 mm/yr (Carbognin et al., 2004). The marsh area in the Venice lagoon has decreased by about 75 % since 1901, caused by both drowning and lateral erosion of marshes (Tommasini et al., 2019). The San Felice salt marsh (45.48˚N, 12.46˚E) is one of the best preserved marshes in the Venice Lagoon, being capable of keeping pace with current relative sea level rise (e.g., Roner et al., 2016; Marani et al., 2003). The average salinity varies between 24 and 33 PSU (Gieskes et al., 2013; Zirino et al., 2014), and the average SSC is between 10 and 20 mg/l (Zaggia and Ferla, 2005; Defendi et al., 2010; Venier et al., 2014). The salt marsh is occupied by halophytic species, such as *Salicornia veneta*, *Spartina maritime*, *Limonium narbonense*, *Sarcocornia fruticosa*, *Puccinellia palustris*, *Inula crithmoides*, *Juncus maritimus* (Silvestri et al., 2005; Marani et al., 2006). The elevation of the salt marsh ranges from 0 m to 0.7 m relative to mean sea level (MSL). Our study site has an area of 72.3 ha (Fig. 2).

2.3 Blackwater marshes, Chesapeake Bay, USA

The Blackwater marshes (38.40˚ N, 76.08˚ W) are located along the Chesapeake Bay (Maryland, USA). They cover an area of about 6000 ha with an average SSC of about 50 mg/L and an average salinity of 10 PSU (Stevenson et al., 1985; Ganju et al., 2013; Kirwan and Guntenspergen, 2015). Long-term local sea level rise is currently 3.7 mm/yr (NOAA station 8571892, http://tide- sandcurrents.noaa.gov/sltrends, 12/19/2016). About half of the interior marshes have disappeared since 1938, mainly by the development and enlargement of bare patches, which are occurring as interior marsh pools (Stevenson et al., 1985; Kearney et al., 1988; Kirwan and Guntenspergen, 2012; Schepers et al., 2017). The pool expansion has been attributed to submergence by sea level rise, vegetation disturbance by invasive herbivores and subsequent open-water expansion (Stevenson et al., 1985; Kendrot, 2011). Changes in water level are mainly driven by meteorological events (wind and air pressure), while the astronomical tidal range is about 0.25 m at our study site. Brackish vegetation dominates, with species such as *Scirpus americanus*, *Spartina alterniflora, Spartina patens*, *Distichlis spicata, Spartina cynosuroides,* and *Phragmites australis* (Pendleton and Stevenson, 1983; Kirwan and Guntenspergen, 2012). Our study area covers an area of about 699.8 ha (Fig. 3).

3 Materials and data preprocessing

3.1 General procedure

For all three study sites, aerial photographs were digitized, georeferenced and manually classified into vegetated marshes, unconnected bare patches, connected bare patches and tidal channels (Figs. 1-3). Bare patches that were smaller than 1 m^2 were not considered in this study. Given the resolution of the images (see below), bare patches were classified as to be connected to the channel network when the connecting channel was at least 0.5 m wide. Hence our classification of unconnected bare patches may also include patches with a small connecting channel (less than 0.5 m wide). LIDAR data was used to analyze the elevation differences between vegetated marshes, unconnected bare patches, connected bare patches and tidal channels. When bare patches were inundated during the LIDAR survey, the soil surface elevation within the bare patches was measured with field surveys (methods are explained below for the different study sites). Generally, LIDAR data have larger and more homogeneous spatial coverage and higher spatial resolution. Field surveys only include selected locations, but with greater vertical accuracy, especially for vegetated areas where LIDAR partially reflects on the vegetation canopy, and open water where LIDAR reflects on the water surface. All the spatial analyses were done using ArcGIS.

3.2 Saeftinghe

For the Saeftinghe study site, a time series of false-color aerial images was used, from 1990, 1998, 2004 and 2008. The four images were selected considering the data availability and to detect dynamic changes from vegetated marsh portions into bare patches and vice versa. All the photos were processed in a similar way, by scanning, georeferencing and mosaicking them into digital pictures with a minimum resolution of 0.5 m. All the aerial images were provided by Rijkswaterstaat (the Dutch governmental institute for water management) (Huijs, 1995; van der Pluijm and de Jong, 1998; Reitsma, 2006; Bakker and Bijkerk, 2009). From all the available aerial photographs, we extracted two sample areas (Fig. 1) free from drifted plant debris, which were analyzed together. The digitized aerial images in the sample areas were classified into vegetation, water and bare soil based on supervised maximum likelihood classification, and then further classified visually into vegetated marshes, channels, connected bare patches and unconnected bare patches. For elevation data in Saeftinghe, we used a Digital Terrain Model (DTM) with a resolution of 2×2 m, which was obtained from a LIDAR survey performed in 2004 during low tide with a maximum vertical error of 0.2 m (Alkemade, 2004). The measurement point density of the LIDAR survey varied from 1 point per 16 m^2 to several points per m^2. The DTM data were also provided by Rijkswaterstaat. We used only one LIDAR dataset to derive the elevations of bare patches and marshes over the period 1990-2008 because previous research in the area showed that during that period, elevation changes were limited with maximum rates of 1 cm/yr (Wang and Temmerman, 2013). This implies that over the considered time scale (1990-2008), maximum elevation changes (~18 cm in 18 years) are of the same order of magnitude as the vertical error of the LIDAR data (~20 cm). Therefore, we decided to use one LIDAR-based DTM for 2004, which is considered to be representative to characterize the approximate time-averaged

elevation of marshes and bare patches over the period 1990-2008. No field survey data were used for Saeftinghe since all bare patches drain completely during neap tides so that soil surface elevations were recorded by LIDAR.

3.3 San Felice

For the San Felice study site, our analysis was based on a vegetation map classified from a hyperspectral image with a
190 resolution of 1.3 m, which was acquired in 2002 by the airborne CASI sensor (15 bands in the visible and near infrared portion of the spectrum) (Belluco et al., 2006). It was visually reclassified into channels, connected bare patches, unconnected bare patches and vegetated marshes (Fig. 2). For the latter, we consulted a black-white aerial photograph acquired in 2000 with a resolution of 16 cm, and a 1-meter resolution pan-sharpened multispectral IKONOS image acquired in 2006. For elevation data in San Felice, we used both a DTM obtained from a LIDAR survey and field measurements. The
195 LIDAR survey was performed during low tide in 2002 with a mean measurement point density of about 48 points/m^2 and a vertical accuracy better than 0.15 m (Wang et al., 2009). From these data we constructed a gridded DTM with a spatial resolution of 1×1 m. Field elevation measurements from the Venice Water Authority in 2000 were also used, because some bare patches were inundated during the LIDAR survey. Data were collected with stereo aerial photography for marshes, stadia rods with GPS for areas close to marshes and mudflats, and single-beam echo-sounder for shallow waters (Sarretta et
200 al., 2010). In total, 340 elevation measurements were located in vegetated marshes, and 95 measurements in bare patches. The boundary of the study area was delineated by channels and creeks as shown in Fig. 2, considering the availability of data. Since almost no vegetation recovery in bare patches was observable on aerial images from the San Felice marsh, we did not do a time series analysis on vegetation recovery.

3.4 Blackwater

205 In the Blackwater study site, we selected a study area away from the influence of roads and uplands (Fig. 3). The small study area (marked with shading in Fig. 3) was chosen for the field survey. A larger study area (the entire colored region in Fig. 3) was later considered in order to increase the number of bare patches connected to channels wider than 1 m. Bare patches that are connected with narrow channels (< 1 m) and that are located outside of the small study area (blue polygons in Fig. 3) were not considered in the analysis. We used false color aerial photographs with a spatial resolution of 0.3 m obtained in
210 2010 and provided as digitized and georeferenced mosaic by the United States Department of Agriculture (USDA). We classified the photos into vegetated marshes, connected bare patches, unconnected bare patches and channels, using the same method for Saeftinghe. We also used data acquired from a LIDAR survey and a field elevation survey. The LIDAR data were obtained in 2003 with an average area sampling density of about 0.8 points per m^2 and a mean vertical accuracy of 0.14 m. The DTM was provided with a resolution of 2×2 m by the U.S. Geological Survey and Maryland Department of Natural
215 Resources. As most bare patches were covered by water during the LIDAR survey, a field survey was carried out in 2012 in the small study area using RTK-GPS with ±1.5 cm accuracy. In total, 36 elevation measurements were collected in 5 unconnected bare patches, 31 measurements in 5 connected bare patches and 93 measurements in the vegetated marshes. An

overview of the number of data points (LIDAR and GPS measurements) that fall within marshes and bare patches are given for the different study sites in Table 1. We did not do a time series analysis on vegetation recovery because other studies

220 have demonstrated that recovery is absent (Schepers et al., 2017).

4 Data analysis

4.1 Topographic conditions determining the presence of bare patches

In order to identify the topographic conditions determining the presence of bare patches or marsh vegetation, we analyzed the frequency distributions of surface elevation and distance to channels for connected and unconnected bare patches, and

225 compared them with the vegetated marsh portions, for the three study sites. The surface elevation was analyzed using LIDAR data and field data. The distance to channels was calculated as the Euclidean Distance from the edge of channel polygons. Bare patches smaller than 1 m^2 were excluded from the analysis. The nonparametric Mann-Whitney U test was conducted to test whether or not there were significant differences between elevations of vegetated marsh portions, connected and unconnected bare patches.

230 Elevation classes of 10 cm were used, since smaller elevation classes were not deemed to be reasonable considering the vertical accuracy of LIDAR data. Surface elevation relative to the local mean low and high water levels (i.e., the tidal frame), is an important factor for vegetation because it determines the frequency, depth and duration of tidal flooding and is widely considered as a crucial ecological condition for marsh plant growth. Therefore, in order to allow comparisons between the three marsh sites with largely different tidal ranges, we rescaled the surface elevation relative to the tidal frame using the

235 following relationship:

$$RE = \frac{E - MLWL}{MHWL - MLWL} \tag{1}$$

where RE is the relative elevation (a dimensionless proportion of the local tidal frame), E is the actual elevation (in m relative to a fixed datum), $MLWL$ and $MHWL$ are the mean low water level and mean high water level, respectively (in m relative to the same datum). Hence RE is 0 for elevations equal to $MLWL$ and is 1 for elevations equal to $MHWL$.

240 In addition, the frequency distribution of bare patch sizes was calculated and related to the widths of channels that were connected to bare patches. The channel width was measured on the aerial photographs at the connection with the bare patch for each single patch and classified into categories with 5 m spacing. Unconnected bare patches (channel width < 0.5 m) and bare patches connected with small channels (channel width between 0.5 m and 1 m) were classified as two separate categories because of their large number. We combined all bare patches with a connection > 80m in the highest class, since

245 there were only 0, 1 and 2 patches for this category in the Saeftinghe, San Felice, and Blackwater marsh sites, respectively.

4.2 Topographic conditions determining the revegetation of bare patches in Saeftinghe

We studied revegetation of bare patches in the Saeftinghe marsh during the last two decades. We did not include the San Felice and Blackwater marshes in this analysis, because there was almost no revegetation recognizable on the aerial photographs during this period of the last two decades. Between each aerial photograph in 1990, 1998, 2004 and 2008, we identified areas that changed from vegetated to bare surfaces, areas that revegetated from bare to vegetation, and areas that remained bare or vegetated. From this data, we determined the rate of revegetation of bare areas. We made a distinction between the following classes:

(1) permanent bare patches that never revegetated within the considered time period from 1990 to 2008;

(2) rapidly revegetated bare patches, identified as bare in only one image, either 1998 or 2004, and observed as vegetation in the other three images;

(3) permanent marsh areas, classified as vegetation throughout the time series.

In order to identify the topographic conditions for rapid or no revegetation of bare areas, the frequency distribution of elevation was calculated for these three classes (permanent bare patches, rapidly revegetated bare patches, and permanent marsh areas), as well as the frequency distributions of the distance to the closest channel. In addition, we also determined the width of the channels connecting to the bare patches. For permanent bare patches, the channel width is calculated as the mean value for 1990, 1998, 2004 and 2008. For rapidly revegetated bare patches, the channel width is the value when the bare patches occurred, either in 1998 or in 2004. In order to identify the relationship between the rate of revegetation and the width of connecting channels, the frequency distribution of channel widths was compared between permanent bare patches and rapidly revegetated bare patches.

5 Results

5.1 Topographic conditions determining the presence of bare patches

In order to identify the topographic conditions determining the presence of bare patches, we tested relationships between their presence and three topographic variables, which are (1) elevation of the bare soil surface, (2) distance of the bare patches from channels and (3) channel width for bare patches connected to channels (channel width <0.5 m for unconnected bare patches). We first tested whether these three topographic variables are independent from each other. The correlations were low (Pearson's r <0.5) and not significant (p>0.05) between all variables and for all field sites. Only for the Blackwater marsh, the correlation between the elevation and the channel width was high (Pearson's r = -0.9), but this correlation is based on a very low number of connected (n=5) and unconnected bare patches (n=5).

5.1.1 Elevation

275   In Saeftinghe, the connected bare patches, unconnected bare patches and vegetated marshes fall within the elevation ranges of 2.3–3.5 m above MSL, which is close to the local MHWL (relative elevation $RE = 0.91$–1.16) (Fig. 4a). The differences in elevation between the vegetated marshes, connected and unconnected bare patches were statistically significant between each two of the three features ($p < 0.001$ based on the Mann-Whitney test). The peaks of the elevation distribution for the vegetated marshes and unconnected bare patches are 0.1 m higher than for the connected bare patches (or the difference

280   between relative RE, $\Delta RE = 0.02$). The mean elevation of the vegetated marshes is highest (2.97 m above MSL, $RE = 1.05$), whereas this is 0.14 m lower for the unconnected bare patches ($\Delta RE = 0.03$) and 0.23 m lower for the connected bare patches ($\Delta RE = 0.05$).

In San Felice, the connected bare patches, unconnected bare patches and vegetated marshes are situated in different ranges of elevations between -0.5 and +0.7 m relative to MSL ($RE = 0$–1.2, Fig. 4b). The differences in elevation distributions of these

285   three categories are also statistically significant ($p < 0.001$ based on the Mann-Whitney test). The elevation measured in the field is lower than that from the LIDAR survey for both the connected bare patches and vegetated marshes. The peaks of the elevation distribution of the vegetated marshes and unconnected bare patches are about 0.15 m lower than MHWL ($RE = 0.85$) based on LIDAR data, and about 0.3 m or 0.5 m higher than connected bare patches ($\Delta RE = 0.3$ or 0.5) based on LIDAR or field data, respectively. The mean LIDAR elevation of the vegetated marshes is 0.35 m relative to MSL ($RE =$

290   0.85), which is 0.04 m higher than unconnected bare patches ($\Delta RE = 0.04$) and 0.28 m higher than connected bare patches ($\Delta RE = 0.28$).

In the Blackwater Marshes, connected bare patches, unconnected bare patches and vegetated marshes occupy significantly different ranges of elevations ($p < 0.001$ based on Mann-Whitney test) between -0.7 m and +0.5 m relative to MSL ($RE = -0.9$–1.5, Fig. 4c). The peaks of the elevation distribution of the vegetated marshes are 0.1 m lower than MHWL ($RE = 0.8$),

295   0.3 m higher than unconnected bare patches ($\Delta RE = 0.6$) and 0.6 m higher than connected bare patches ($\Delta RE = 1.2$). The mean elevation is the highest for the vegetated marshes (0.13 m relative to MSL, $RE = 0.76$), 0.23 m lower for the unconnected bare patches ($\Delta RE = 0.46$) and 0.6 m lower for connected bare patches ($\Delta RE = 1.2$).

Together these results indicate that connected bare patches, unconnected bare patches, and vegetated marshes tend to occupy different elevation ranges at each site ($p < 0.001$ by Mann-Whitney), with the largest absolute elevation differences in

300   Blackwater, the smallest in Saeftinghe, and intermediate values for San Felice. Connected bare patches always lie within the lowest elevation range, whereas vegetated marshes always dominate the highest elevation range around MHWL. Unconnected bare patches are always found in the intermediate elevation range, which is about 0.1–0.5 m higher than the connected bare patches. The difference in RE (relative to the tidal frame) between the connected and unconnected bare patches is about 0.02 in Saeftinghe, 0.2–0.5 in San Felice, and 0.6–0.8 in Blackwater.

5.1.2 Distance to channels

The frequency distribution of the distance between a bare patch and closest channel shows similar results for the three marsh sites (Fig. 5). Vegetated marshes rather than bare patches occur near channels. With increasing distance from channels, marsh vegetation becomes less frequent and unconnected bare patches become more frequent. Connected bare patches occur most frequently at large distances from the channels. The peak of the distribution is situated at 1.0 m for vegetated marshes in all three sites; at 8 m for unconnected bare patches and over 10 m for connected bare patches in both Saeftinghe and San Felice; and at 82 m and 89 m for unconnected and connected bare patches in Blackwater, respectively.

5.1.3 Bare patch size in relation to connectivity to channels

Bare patch size generally increases with increasing width of connecting channels, whereas the number of bare patches decreases with increasing channel widths (Fig. 6). The unconnected bare patches in Saeftinghe, San Felice and the small study area of Blackwater, occupy 63 %, 36 % and 67 % of the total number of bare patches, respectively, but only 2 %, 1 % and 3 % of the total area of bare patches, respectively. Hence, unconnected bare patches are numerous but small. The number of connected bare patches, in contrast, is in most cases smaller and they become less abundant with increasing width of the connecting channels.

5.2 Topographic conditions determining the revegetation of bare patches in Saeftinghe

The multi-temporal analysis for Saeftinghe shows that bare patches have been dynamically expanding or shrinking between the four images of 1990-1998-2004-2008 (Fig. S4). We focused on bare areas with two extreme rates of revegetation, which are permanent bare areas (which never revegetated throughout the time series) and rapidly revegetated bare areas (only present in 1998 or 2004 and revegetated by the next time step). The spatial distribution of these bare categories (Fig. S4) suggests that the inner portion of big connected bare patches tends to be stable and never revegetated within the studied period, while rapidly recovering bare areas are mainly present at the edge of small bare patches.

5.2.1 Elevation

The elevation distribution showed that permanently bare areas (i.e. remaining bare over the studied 18-year period) occupy the lowest range of elevations, whereas permanent marsh areas have the highest range of elevations (Fig. 7a). At intermediate elevations, bare patches become rapidly revegetated (i.e. within 4 to 6 years after their first appearance (Fig. 7a).

5.2.2 Distance to channels

The frequency distribution of the different bare categories with distance to the channels (Fig. 7b) shows that stable marshes are closest to channels with a peak around 1-2 m from channels. Bare areas that revegetated quickly have an intermediate distance around 8 m from channels, whereas permanent bare areas are located farthest from the channels with a peak at 21 m.

5.2.3 Connectivity to channels

335 Permanent bare areas are always connected to channels, and tend to be associated with wide channels (Fig. 7c). The percentage of bare areas that become revegetated increases with decreasing channel width (Fig. 7c).

6 Discussion

Bare patches within otherwise vegetated coastal marshes are often recognized as symptoms of marsh loss in many places around the world (Kearney et al., 1988; Fagherazzi, 2013; Mariotti and Fagherazzi, 2013; Ortiz et al., 2017; Schepers et al.,
340 2017), but comparative studies among different marsh systems to better understand the conditions that determine their presence and potential vegetation recovery are relatively scarce (e.g., Mariotti, 2016). Fig. 8 provides a schematic summary of our results, and of our interpretations that are discussed here. For three marsh sites with different tidal ranges, sediment input and plant species, we showed that: (1) bare patches connected to channels occur most frequently at the lowest surface elevations and farthest distances from creeks; unconnected bare patches most frequently occupy intermediate elevations and
345 distances from creeks, and are smaller in size and larger in number; and vegetated marshes dominate at the highest surface elevations and closest to creeks. (2) The elevations of connected and unconnected bare patches tend to be lower relative to the tidal frame in sites with a smaller tidal range, although our analysis only included three sites. (3) Recovery of vegetation in bare patches at the time scale of the last two decades was only observed in the site with high tidal range and high sediment input. Here vegetation recovery is hampered by low surface elevations relative to the local tidal frame, by farther distance
350 from channels and by a high connectivity to the channel network. Below we will further substantiate these findings, and discuss interpretations and potential hypotheses that may explain mechanisms of formation and recovery of bare patches.

6.1 Topographic conditions determining the presence of bare patches

Our results suggest that bare patches exist under qualitatively similar topographic conditions across three different marsh systems. We found that marshes have a higher elevation than bare patches, in accordance with previous studies (DeLaune et
355 al., 1994; Erwin et al., 2006; Wilson et al., 2014). In addition to existing insights, we found that unconnected bare patches are most frequently found at higher elevations and shorter distances from channels as compared to connected bare patches (Figs. 4 and 5). Additionally, we found a positive relationship between patch size and the width of the connecting channel (Fig. 6). These different observations may be interpreted as follows. First, the positive relationship between bare patch size and connecting channel width may be due to the difference in tidal prism (i.e. the total water volume that floods into, and
360 drains out of, the bare patches during a tidal cycle). A larger bare patch implies a larger tidal prism, which means that higher volumes of water are transported into and out of the bare patches. Assuming that most of the water is transported through the connecting channel, a larger tidal prism would be associated with larger channel-forming discharges and therefore wider channels (e.g., Rinaldo et al., 1999b; Kirwan et al., 2008; D'Alpaos et al., 2010; Vandenbruwaene et al., 2013).

Secondly, our finding that unconnected bare patches occur most frequently on higher elevations than connected bare patches, may be interpreted as follows. We expect that connected bare patches experience higher incoming and outgoing flood and ebb flow velocities as they are directly connected to the channels. Furthermore, we found that connected bare patches are larger (Fig. 6), and hence we may expect more potential for erosion of surface sediments induced by waves (because of larger wind fetch length). Wave erosion in interior marsh ponds has been found to be related to the size and wind fetch length of marsh ponds (Mariotti and Fagherazzi, 2013; Mariotti, 2016; Ortiz et al., 2017). Hence larger bare patches are likely to experience more wave-induced erosion and are found in this study to be connected through wider connecting channels, which may facilitate the tidal export of the eroded sediments from connected bare patches, and therefore may explain the lower surface elevation of connected bare patches. In contrast, we hypothesize that unconnected bare patches, which are typically smaller (Fig. 6), may be expected to experience less wave erosion (smaller fetch length) and much weaker flow velocities (as flow is obstructed by surrounding vegetation). With respect to the latter effect, we notice that our classification of unconnected patches may also include patches with connecting channels smaller than 0.5 m but impossible to detect on the aerial images. Nevertheless, also in the case of such small connecting channels < 0.5 m wide, one can expect that drainage of the bare patches after overmarsh tides is much slower, with lower ebb flow velocities, as compared to bare patches with wide connecting channels (up to several tens of meters wide, see Fig. 6), facilitating faster drainage, higher ebb flow velocities and potentially leading to larger tidal export of eroded sediments. Unconnected bare patches were also found to occur most frequently at shorter distances from channels as compared to connected bare patches, and this may facilitate higher sediment supply to unconnected bare patches closer to channels, as suspended sediment concentrations typically decrease with increasing distance from channels (Leonard, 1997; Christiansen et al., 2000; Temmerman et al., 2003b). Therefore, higher sediment supply and lower magnitude of waves and tidal currents in smaller, unconnected bare patches at shorter distance from channels, may facilitate the settlement of suspended sediments and reduce erosion, and as such may explain our finding of higher surface elevations of unconnected bare patches as compared to connected bare patches (Fig. 8). This finding is also in accordance with the model of Mariotti (2016) proposing that, in what is called the "pond collapse regime", the depth of connected marsh ponds would be larger than the depth of unconnected ponds.

Thirdly, our results indicate that connected bare patches are predominantly located farther away from channels than unconnected bare patches (Fig. 5). One potential explanation is that connected bare patches are generally larger than unconnected bare patches (Fig. 6), so that a larger fraction of the connected bare patches is located at a farther distance from channels. Presence of bare patches in relation to distance from channels has been previously studied on large regional scales (10^2–10^4 m) considering only large estuarine channels (Turner and Rao, 1990; Kearney and Rogers, 2010). On a smaller scale (10–10^2 m), Redfield (1972) qualitatively reported that big bare patches are located relatively far from channels. Adamowicz and Roman (2005) observed that bare patches were located at around 11 m from the nearest channel in both ditched and unditched marshes in New England. Such a value is similar to that found for Saeftinghe and San Felice, but smaller than the value obtained for Blackwater. The elevation difference between connected and unconnected bare patches probably relates to their difference in distance to channels. Marshes typically have a micro-topography of higher levees

along channels and lower depressions farther away from channels as a consequence of progressive suspended sediment deposition during tidal flooding of marshes from channels (e.g., Reed, 1988; Covi and Kneib, 1995; Leonard, 1997; Esselink et al., 1998; Reed et al., 1999; Allen, 2000; Temmerman et al., 2004; D'Alpaos et al., 2007; Bartholdy, 2012). In accordance with this micro-topography, the lower-elevation connected bare patches are located farther away from channels than the higher-elevation unconnected bare patches (Fig. 8). This micro-topography of levees close to channels and depressions further away from channels, is often associated with an increasing inundation duration after high tides, and decreasing soil drainage/aeration during low tides, with increasing distance from channels (e.g., Ursino et al., 2004). Also, a modelling study suggested that marsh vegetation expansion can lead to increased inundation time, and as such can feedback on increased stress and chance for vegetation die-back (Brückner et al. 2019). This may be all mechanisms that may contribute to increased chance for occurrence of bare patches within marshes at farther distances from channels. In addition, the frequency distribution of distance to the closest channels is observed to be exponential for the vegetated marsh surfaces in all three marsh sites, which is analogous to the results by Marani et al. (2003) and holds only for the vegetated marsh surfaces.

Finally, our results demonstrated that the size of bare patches is negatively related to the number of bare patches (Fig. 6). Such finding has also been observed in other marsh systems (Turner and Rao, 1990; Schepers et al., 2017). This may be indicative for initial formation of many small bare patches that grow and merge together through time, hence leading to a decreasing number of larger patches. This process of merging of initially small bare patches into larger patches has been documented for the Blackwater study site from an analysis of time series of aerial pictures over the period 1938-2010 (Schepers et al., 2017).

In conclusion, we observed qualitatively similar topographic conditions for the presence of bare patches across the three study sites, albeit that the elevations of connected and unconnected bare patches tend to be lower relative to the tidal frame in sites with a smaller tidal range. The latter agrees with earlier findings that micro-tidal marshes have in general a lower surface elevation than macro-tidal marshes (Kirwan et al., 2010; D'Alpaos et al., 2011). Our finding suggests that feedback mechanisms between vegetation and topography are important in regulating the position of the bare patches, and perhaps generalizable across systems. However, we emphasize that our analysis is based on only three study sites, and more research is needed to assess the degree to which this finding is universal.

6.2 Topographic conditions determining the revegetation of bare patches

The comparison between bare patches with two extreme revegetation rates (i.e., permanent bare patches over the studied 18-year period and rapidly revegetated bare patches within 4-6 years) for Saeftinghe suggests that fast revegetation preferentially occurs by expansion of the vegetated edge into small, higher elevation, unconnected bare patches, whereas the central areas of big, lower elevation, connected bare patches tend to remain unvegetated over the considered time period of 18 years. These results are consistent with previous studies. For example, only small bare patches were invaded by vegetation in ditched marshes in Louisiana, although large bare patches were permanent over a study period of 22 years (Turner and Rao, 1990). In several New England marshes, re-establishment of vegetation started within 1-2 years after

unconnected bare patches merged with the channel network and became drained (Wilson et al., 2009, 2014). Additionally, some studies find that unconnected bare patches expand and merge quickly, while connected bare patches are relatively stable (Kearney et al., 1988). These disparate observations in different marsh sites may be due to different environmental conditions, such as differences in relative sea level rise, tidal range, and sediment availability (Mariotti, 2016). In a modelling study, Mariotti (2016) demonstrated that vegetation recovery in marsh ponds is favoured under conditions of slow relative sea level rise, large tidal range, and large inorganic sediment supply.

6.3 Vulnerability for bare patch formation and resilience for bare patch recovery

Previous modelling has suggested that pond formation increases and pond recovery decreases in marsh sites that are subject to a lower suspended sediment availability, smaller tidal range, and lower rate of relative sea level rise (RSLR) (Mariotti, 2016). First of all, we want to emphasize that we only investigated three sites, which it is not enough to fully assess the impact of site-differences, such as in tidal range, sediment supply, and rate of RSLR, on occurrence and revegetation of bare patches. Yet we notice that revegetation only occurred at the site (Saeftinghe) with largest tidal range and sediment supply, while it was not observed at the two other sites (San Felice and Blackwater) with smaller tidal range and sediment supply. As explained in the description of the three study sites (see Sect. 2), the average tidal range and suspended sediment concentrations vary from highest in the Saeftinghe marsh (4.9 m and 30-60 mg/l respectively), intermediate in San Felice marsh (1 m and 10-20 mg/l), to lowest in the Blackwater marshes (0.5 m and 50 mg/l). Long-term RSLR rates in the San Felice and Blackwater marshes are within the same range of 3–4 mm/yr, while mean high water level rise in Saeftinghe is 5.7 mm/yr. In Saeftinghe, marsh elevations are mostly above MHWL, while they are mostly below MHWL in San Felice and Blackwater (Fig. 4). Probably this may explain why the proportion of bare surface area is larger in San Felice and Blackwater (34.33 % and 42.58 %, resp.) than in Saeftinghe (15.72 %).

Marsh resilience inferred by revegetation of bare patches was only observed in Saeftinghe where the mean tidal range is 4.9 m, and bare patches have high elevations relative to the tidal frame (average RE = 1.002 for connected bare patches and average RE = 1.02 for unconnected bare patches; Fig. 4). Revegetation of bare patches has been observed in other systems with high tidal ranges (Millette et al., 2010; Wilson et al., 2014), which facilitates well-drained conditions during low tide and enables vegetation regrowth. In contrast, in Blackwater where the mean tidal range is about 0.5 m, bare patches have a much lower elevation relative to the tidal frame, even below the MLWL (average RE = -0.462 for connected bare patches and average RE = 0.284 for unconnected bare patches; Fig. 4), which means that there is no drainage at low tide so that marsh vegetation cannot recover. Bare patches also tend to be permanent in other systems under low tidal ranges, such as in Louisiana and mid-Atlantic US salt marshes (Wilson et al., 2014; Ortiz et al., 2017). Clearly, the same elevation loss in a marsh with small tidal range will result in a higher increase in tidal inundation frequency and duration, and consequently in more stress on vegetation growth, as compared to a marsh with a large tidal range. Hence, if marsh vegetation and elevation loss occur, it would be easier to recover for marsh vegetation in a higher tidal range environment, such as that of Saeftinghe, as compared to situations with a lower tidal range, such as the Blackwater and San Felice marshes. This interpretation is in

agreement with previous studies. Microtidal marshes were reported to be particularly vulnerable to bare patch formation and
465   expansion (Kearney et al., 1988; Mariotti and Fagherazzi, 2013). Marshes with larger tidal ranges also have bare patches but
they are generally more dynamically forming and recovering, while the whole marsh system is relatively stable (Redfield,
1972; Wilson et al., 2009). The model of Kirwan and Guntenspergen (2010) suggested that extensive bare patches occur,
expand quickly and become permanent under small tidal ranges but not under large tidal ranges, because the elevation range
suitable for vegetation growth is smaller in low tidal range environments. In general, marsh stability is positively related to
470   tidal range (Kirwan et al., 2010; D'Alpaos, 2011), and numerical modelling indicates that high sediment concentrations are
necessary for recovery of bare patches (Mariotti, 2016). However, Mariotti (2016) only considers recovery after connection
to the tidal channel network, not the recovery of isolated bare patches. In our study, we observed that bare patches
unconnected to the tidal channel network all recovered at the Saeftinghe site. Complete drainage of the Saeftinghe bare
patches during ebb tides might explain this apparent discrepancy. We suggest that the close distance to channels (see Figs. 5
475   and 8b) (e.g., Ursino et al., 2004) and coarser sediment associated with channel levees (Allen, 2000) enable the unconnected
bare patches to drain completely in Saeftinghe through subsurface drainage, and allow vegetation recovery.

Finally, our results may be indicative to decision makers on salt marsh management, as the formation of bare patches may be
indicative for marsh degradation towards an unvegetated state that may be difficult to recover. Our study indicates that early
signatures for marsh degradation must be particularly monitored in marsh portions, farthest away from main channels and
480   with lowest surface elevations. Monitoring of early signatures is especially advised in systems with very low tidal range and
suspended sediment availability.

7 Conclusions

In this paper, we studied the topographical conditions for presence and revegetation of bare patches within three coastal
marsh sites that are largely different in tidal range, sediment supply and plant species. The analyses of aerial photographs,
485   LIDAR data and field topographic measurements showed that the topographic conditions (i.e. elevations, distances from
channels and connectivity to channels) for presence of bare patches were qualitatively consistent among the three marsh sites.
We found that bare patches connected to channels occur most frequently at the lowest surface elevations and farthest away
from creeks; unconnected bare patches most frequently occupy intermediate elevations and distances from creeks, and are
smaller in size and larger in number; and vegetated marshes dominate at the highest surface elevations and closest to creeks.
490   Further, we showed that the elevations of connected and unconnected bare patches tend to be lower relative to the tidal frame
with increasing tidal range, although our analysis only included three sites. Revegetation of bare patches was only observed
in one site, which was the site with the highest tidal range and the largest sediment supply. For that site, we found that the
chance of bare patch revegetation decreases with increasing width of channels that connect bare patches to the tidal channel
network. The latter is associated with lower bare patch elevation, farther distance to channels and bigger bare patch size.
495   Finally, in the context of sea level rise, our results suggest that the marsh site with the highest tidal range and highest

sediment input is less vulnerable to bare patch formation and more resilient in terms of revegetation of bare patches than the two other marsh sites with lower tidal range and lower sediment supply. However, we emphasize that our study only included three sites, and that further research comparing much more sites is needed, to further advance our understanding of why certain marsh sites are more vulnerable to others to formation and persistence of bare patches. Such knowledge will be

500 important to inform decision makers on site-specific priorities for marsh conservation.

Data availability

The aerial images and DTM data for Saeftinghe can be downloaded from Rijkswaterstaat (https://geoservices.rijkswaterstaat.nl). The IKONOS data for San Felice can be accessed at Planetek Italia s.r.l. (https://www.planetek.it). The field elevation data for San Felice can be accessed at Venice Water Authority in Italy

505 (http://provveditoratovenezia.mit.gov.it). The aerial images for Blackwater can be downloaded from earthexplorer.usgs.gov. The LIDAR data for Blackwater can be downloaded from https://inport.nmfs.noaa.gov/inport/item/49781.

Supplement link

The supplement related to this article is available online at:

Author contribution

510 ST and CW designed the study. CW prepared the manuscript with contributions from all co-authors.

Competing interests

The authors declare that they have no conflict of interest.

Acknowledgements

We dedicate this paper to the late Jose Busnelli who contributed to parts of the analyses presented in this paper. We thank
515 Rijkswaterstaat (Dick De jong) in the Netherlands, Venice Water Authority in Italy, and United States Department of Agriculture and Maryland Department of Natural Resources in the USA for providing vegetation maps, aerial photographs, as well as elevation data of LIDAR surveys and field surveys. We also would like to thank Glenn Guntenspergen, Melissa Duvall, Patrick Brennand and Kyle Derby for the field measurements in the Blackwater Marshes.

Financial support

520   This work has been supported by the Project 41501116, 51761135022, and 41401413 supported by National Natural Science Foundation of China, the European Union Programme Erasmus Mundus External Cooperation Window (EMECW)–Lot 14–China and by the FWO research community "Functioning of River Ecosystems by Plant-Flow-Soil interactions", Research Foundation Flanders (FWO PhD grant L.S., 11S9614N), the Technology Foundation for Selected Overseas Chinese Scholar from Ministry of Personnel of China (2015), and the U.S. National Science Foundation (1237733, 1426981, 525   1654374, 1832221 and 1529245).

References

Adamowicz, S. C. and Roman, C. T.: New England salt marsh pools: a quantitative analysis of geomorphic and geographic features, Wetlands, 25, 279–288, https://doi.org/10.1672/4, 2005.

Alkemade, I. S. W.: Kwaliteitsdocument Laseraltimetrie, Projectgebied Westerschelde, Ministerie van Verkeer en Waterstaat, 530   Rijkswaterstaat, Delft, NL, 2004.

Allen, J.: Morphodynamics of Holocene salt marshes: a review sketch from the Atlantic and Southern North Sea coasts of Europe, Quat. Sci. Rev., 19, 1155–1231, https://doi.org/10.1016/s0277-3791(99)00034-7, 2000.

Argow, B. A. and FitzGerald, D. M.: Winter processes on northern salt marshes: evaluating the impact of in-situ peat compaction due to ice loading, Wells, ME, Estuar. Coast. Shelf Sci., 69, 360–369, 535   https://doi.org/10.1016/j.ecss.2006.05.006, 2006.

Bakker, R. B. and Bijkerk, W.: Toelichting Bij de Geomorfologische Kartering Westerschelde 2008 op Basis van False Colour-Luchtfoto's 1:10.000. Report, Ministerie van Verkeer en Waterstaat, Rijksinstituut, Adviesdienst Geo-Informatie & ICT, Den Haag, Delft, NL, 2009.

Barbier, E. B., Hacker, S. D., Kennedy, C., Koch, E. W., Stier, A. C., and Silliman, B. R.: The value of estuarine and coastal 540   ecosystem services, Ecol. Monogr., 81, 169–193, https://doi.org/10.1890/10-1510.1, 2011.

Barbier, E. B., Koch, E. W., Silliman, B. R., Hacker, S. D., Wolanski, E., Primavera, J., Granek, E. F., Polasky, S., Aswani, S., Cramer, L. A., Stoms, D. M., Kennedy, C. J., Bael, D., Kappel, C. V., Perillo, G. M. E., and Reed, D. J.: Coastal ecosystem-based management with nonlinear ecological functions and values, Science, 319, 321–323, https://doi.org/10.1126/science.1150349, 2008.

545   Bartholdy, J.: Salt marsh sedimentation, in: Principles of Tidal Sedimentology, edited by: Davis, R. A. and Dalrymple, R. W., Springer, Dordrecht, NL, 151–185, 2012.

Baumann, R. H., Day, J. W., and Miller, C. A.: Mississippi deltaic wetland survival: sedimentation versus coastal submergence, Science, 224, 1093–1095, https://doi.org/10.1126/science.224.4653.1093, 1984.

Belluco, E., Camuffo, M., Ferrari, S., Modenese, L., Silvestri, S., Marani, A., and Marani, M.: Mapping salt-marsh vegetation by multispectral and hyperspectral remote sensing, Remote Sens. Environ., 105, 54–67, https://doi.org/10.1016/j.rse.2006.06.006, 2006.

Brückner, M. Z. M., Schwarz, C., van Dijk, W. M., van Oorschot, M., Douma, H., and Kleinhans, M. G.: Salt marsh establishment and eco-engineering effects in dynamic estuaries determined by species growth and mortality, J. Geophys. Res.: Earth Surf., 124, 2962– 2986, https://doi.org/10.1029/2019JF005092, 2019.

Carbognin, L., Teatini, P., and Tosi, L.: Eustacy and land subsidence in the Venice Lagoon at the beginning of the new millennium, J. Mar. Syst., 51, 345–353, https://doi.org/10.1016/j.jmarsys.2004.05.021, 2004.

Carniello, L., Defina, A., and D'Alpaos, L.: Morphological evolution of the Venice Lagoon: evidence from the past and trend for the future, J. Geophys. Res., 114, F04002, https://doi.org/10.1029/2008jf001157, 2009.

Christiansen, T., Wiberg, P. L., and Milligan, T. G.: Flow and sediment transport on a tidal salt marsh surface, Estuar. Coast. Shelf Sci., 50, 315–331, https://doi.org/10.1006/ecss.2000.0548, 2000.

Covi, M. P. and Kneib, R. T.: Intertidal distribution, population dynamics and production of the amphipod Uhlorchestia spartinophila in a Georgia, USA, salt marsh, Mar. Biol., 121, 447–455, https://doi.org/10.1007/bf00349453, 1995.

D'Alpaos, A.: The mutual influence of biotic and abiotic components on the long-term ecomorphodynamic evolution of salt-marsh ecosystems, Geomorphology, 126, 269–278, https://doi.org/10.1016/j.geomorph.2010.04.027, 2011.

D'Alpaos, A., Lanzoni, S., Marani, M., and Rinaldo, A.: Landscape evolution in tidal embayments: modeling the interplay of erosion, sedimentation, and vegetation dynamics, J. Geophys. Res., 112, F01008, https://doi.org/10.1029/2006jf000537, 2007.

D'Alpaos, A., Lanzoni, S., Marani, M., and Rinaldo, A.: On the tidal prism–channel area relations, J. Geophys. Res., 115, F01003, https://doi.org/10.1029/2008jf001243, 2010.

D'Alpaos, A., Mudd, S. M., and Carniello, L.: Dynamic response of marshes to perturbations in suspended sediment concentrations and rates of relative sea level rise, J. Geophys. Res., 116, F04020, https://doi.org/10.1029/2011jf002093, 2011.

D'Alpaos, A. and Marani, M.: Reading the signatures of biologic–geomorphic feedbacks in salt-marsh landscapes, Adv. Water Resour., 93, 265–275, https://doi.org/10.1016/j.advwatres.2015.09.004, 2016.

Day, J. W., Britsch, L. D., Hawes, S. R., Shaffer, G. P., Reed, D. J., and Cahoon, D.: Pattern and process of land loss in the Mississippi delta: a spatial and temporal analysis of wetland habitat change, Estuaries, 23, 425–438, https://doi.org/10.2307/1353136, 2000.

Day, J. W., Kemp, G. P., Reed, D. J., Cahoon, D. R., Boumans, R. M., Suhayda, J. M., and Gambrell, R.: Vegetation death and rapid loss of surface elevation in two contrasting Mississippi delta salt marshes: the role of sedimentation, autocompaction and sea-level rise, Ecol. Eng., 37, 229–240, https://doi.org/10.1016/j.ecoleng.2010.11.021, 2011.

Day, J. W., Rybczyk, J., Scarton, F., Rismondo, A., Are, D., and Cecconi, G.: Soil accretionary dynamics, sea-level rise and the survival of wetlands in Venice Lagoon: a field and modelling approach, Estuar. Coast. Shelf Sci., 49, 607–628, https://doi.org/10.1006/ecss.1999.0522, 1999.

Defendi, V., Kovačević, V., Arena, F., and Zaggia, L.: Estimating sediment transport from acoustic measurements in the Venice Lagoon inlets, Cont. Shelf Res., 30, 883–893, https://doi.org/10.1016/j.csr.2009.12.004, 2010.

DeLaune, R. D., Nyman, J. A., and Patrick, J. W. H.: Peat collapse, ponding and wetland loss in a rapidly submerging coastal marsh, J. Coast. Res., 10, 1021–1030, 1994.

Elschot, K., Vermeulen, A., Vandenbruwaene, W., Bakker, J. P., Bouma, T. J., Stahl, J., Castelijns, H., and Temmerman, S.: Top-down vs. bottom-up control on vegetation composition in a tidal marsh depends on scale, PLoS One, 12, e0169960, https://doi.org/10.1371/journal.pone.0169960, 2017.

Erwin, R. M., Cahoon, D. R., Prosser, D. J., Sanders, G. M., and Hensel, P.: Surface elevation dynamics in vegetated *Spartina* marshes versus unvegetated tidal ponds along the Mid-Atlantic Coast, USA, with implications to waterbirds, Estuaries Coasts, 29, 96–106, https://doi.org/10.1007/bf02784702, 2006.

Esselink, P., Dijkema, K. S., Sabine, R., and Geert, H.: Vertical accretion and profile changes in abandoned man-made tidal marshes in the Dollard Estuary, the Netherlands, J. Coast. Res., 14, 570–582, 1998.

Fagherazzi, S.: The ephemeral life of a salt marsh, Geology, 41, 943–944, https://doi.org/10.1130/focus082013.1, 2013.

Fagherazzi, S., Carniello, L., D'Alpaos, L., and Defina, A.: Critical bifurcation of shallow microtidal landforms in tidal flats and salt marshes, Proc. Natl. Acad. Sci. U. S. A., 103, 8337–8341, https://doi.org/10.1073/pnas.0508379103, 2006.

Fagherazzi, S., Kirwan, M. L., Mudd, S. M., Guntenspergen, G. R., Temmerman, S., D'Alpaos, A., van de Koppel, J., Rybczyk, J. M., Reyes, E., Craft, C., and Clough, J.: Numerical models of salt marsh evolution: ecological, geomorphic, and climatic factors, Rev. Geophys., 50, 294–295, https://doi.org/10.1029/2011rg000359, 2012.

Fagherazzi, S., Mariotti, G., Wiberg, P., and McGlathery, K.: Marsh collapse does not require sea level rise, Oceanography, 26, 70–77, https://doi.org/10.5670/oceanog.2013.47, 2013.

Fagherazzi, S., Palermo, C., Rulli, M. C., Carniello, L., and Defina, A.: Wind waves in shallow microtidal basins and the dynamic equilibrium of tidal flats, J. Geophys. Res., 112, F02024, https://doi.org/10.1029/2006jf000572, 2007.

Ganju, N. K., Nidzieko, N. J., and Kirwan, M. L.: Inferring tidal wetland stability from channel sediment fluxes: observations and a conceptual model, J. Geophys. Res.: Earth Surf., 118, 2045–2058, https://doi.org/10.1002/jgrf.20143, 2013.

Gedan, K. B., Kirwan, M. L., Wolanski, E., Barbier, E. B., and Silliman, B. R.: The present and future role of coastal wetland vegetation in protecting shorelines: answering recent challenges to the paradigm, Clim. Change, 106, 7–29, https://doi.org/10.1007/s10584-010-0003-7, 2011.

Gieskes, J. M., Elwany, H., Rasmussen, L., Han, S., Rathburn, A., and Deheyn, D. D.: Salinity variations in the Venice Lagoon, Italy: results from the SIOSED project, May 2005–February 2007, Mar. Chem., 154, 77–86, https://doi.org/10.1016/j.marchem.2013.05.011, 2013.

615   Harshberger, J. W.: The origin and vegetation of salt marsh pools, Proc. Am. Philos. Soc., 55, 481–484, 1916.

Hu, Z., Suzuki, T., Zitman, T., Uittewaal, W., and Stive, M.: Laboratory study on wave dissipation by vegetation in combined current–wave flow, Coast. Eng., 88, 131–142, https://doi.org/10.1016/j.coastaleng.2014.02.009, 2014.

Hu, Z., van Belzen, J., van der Wal, D., Balke, T., Wang, Z. B., Stive, M., and Bouma, T. J.: Windows of opportunity for salt marsh vegetation establishment on bare tidal flats: the importance of temporal and spatial variability in
620   hydrodynamic forcing, J. Geophys. Res.: Biogeosciences, 120, 1450–1469, https://doi.org/10.1002/2014jg002870, 2015a.

Hu, Z., Wang, Z. B., Zitman, T. J., Stive, M. J. F., and Bouma, T. J.: Predicting long-term and short-term tidal flat morphodynamics using a dynamic equilibrium theory, J. Geophys. Res.: Earth Surf., 120, 1803–1823, https://doi.org/10.1002/2015jf003486, 2015b.

625   Huijs, S. W. E.: Geomorfologische Ontwikkeling van Het Intergetijdegebied in de Westerschelde, 1935-1989. Rapport R 95-3, Universiteit Utrecht, Utrecht, NL, 1995.

Kearney, M. S., Grace, R. E., and Stevenson, J. C.: Marsh loss in nanticoke estuary, Chesapeake Bay, Geogr. Rev., 78, 205–220, https://doi.org/10.2307/214178, 1988.

Kearney, M. S. and Rogers, A. S.: Forecasting sites of future coastal marsh loss using topographical relationships and
630   logistic regression, Wetl. Ecol. Manag., 18, 449–461, https://doi.org/10.1007/s11273-010-9178-y, 2010.

Kearney, M. S., Rogers, A. S., Townshend, J. R. G., Rizzo, E., Stutzer, D., Stevenson, J. C., and Sundborg, K.: Landsat imagery shows decline of coastal marshes in Chesapeake and Delaware Bays, Eos Trans. Am. Geophys. Union, 83, 173–178, https://doi.org/10.1029/2002eo000112, 2002.

Kendrot, S. R.: Restoration through eradication: protecting Chesapeake Bay marshlands from invasive nutria (*Myocastor*
635   *coypus*), in: Island Invasives: Eradication and Management. Proceedings of the International Conference on Island Invasives (Occasional Papers of the IUCN Species Survival Commission Occasional Papers of the IUCN Species Survival Commission), edited by: Veitch, C. R., Clout, M. N., and Towns, D. R., IUCN, Gland, Switzerland, 313–319, 2011.

Kirwan, M. L. and Guntenspergen, G. R.: Influence of tidal range on the stability of coastal marshland, J. Geophys. Res.:
640   Earth Surf., 115, F02009, https://doi.org/10.1029/2009jf001400, 2010.

Kirwan, M. L. and Guntenspergen, G. R.: Feedbacks between inundation, root production, and shoot growth in a rapidly submerging brackish marsh, J. Ecol., 100, 764–770, https://doi.org/10.1111/j.1365-2745.2012.01957.x, 2012.

Kirwan, M. L. and Guntenspergen, G. R.: Response of plant productivity to experimental flooding in a stable and a submerging marsh, Ecosystems, 18, 903–913, https://doi.org/10.1007/s10021-015-9870-0, 2015.

645   Kirwan, M. L., Guntenspergen, G. R., D'Alpaos, A., Morris, J. T., Mudd, S. M., and Temmerman, S.: Limits on the adaptability of coastal marshes to rising sea level, Geophys. Res. Lett., 37, 58–94, https://doi.org/10.1029/2010gl045489, 2010.

Kirwan, M. L. and Megonigal, J. P.: Tidal wetland stability in the face of human impacts and sea-level rise, Nature, 504, 53–60, https://doi.org/10.1038/nature12856, 2013.

650 Kirwan, M. L. and Murray, A. B.: A coupled geomorphic and ecological model of tidal marsh evolution, Proc. Natl. Acad. Sci. U. S. A., 104, 6118–6122, https://doi.org/10.1073/pnas.0700958104, 2007.

Kirwan, M. L., Murray, A. B., and Boyd, W. S.: Temporary vegetation disturbance as an explanation for permanent loss of tidal wetlands, Geophys. Res. Lett., 35, L05403, https://doi.org/10.1029/2007gl032681, 2008.

Kirwan, M. L., Murray, A. B., Donnelly, J. P., and Corbett, D. R.: Rapid wetland expansion during European settlement and 655 its implication for marsh survival under modern sediment delivery rates, Geology, 39, 507–510, https://doi.org/10.1130/g31789.1, 2011.

Leonard, L. A.: Controls of sediment transport and deposition in an incised mainland marsh basin, Southeastern North Carolina, Wetlands, 17, 263–274, https://doi.org/10.1007/bf03161414, 1997.

Marani, M., Belluco, E., D'Alpaos, A., Defina, A., Lanzoni, S., and Rinaldo, A.: On the drainage density of tidal networks, 660 Water Resour. Res., 39, 5029–5035, https://doi.org/10.1029/2001wr001051, 2003.

Marani, M., D'Alpaos, A., Lanzoni, S., Carniello, L., and Rinaldo, A.: Biologically-controlled multiple equilibria of tidal landforms and the fate of the Venice Lagoon, Geophys. Res. Lett., 34, 224–238, https://doi.org/10.1029/2007gl030178, 2007.

Marani, M., D'Alpaos, A., Lanzoni, S., Carniello, L., and Rinaldo, A.: The importance of being coupled: stable states and 665 catastrophic shifts in tidal biomorphodynamics, J. Geophys. Res., 115, F04004, https://doi.org/10.1029/2009jf001600, 2010.

Marani, M., Silvestri, S., Belluco, E., Ursino, N., Comerlati, A., Tosatto, O., and Putti, M.: Spatial organization and ecohydrological interactions in oxygen-limited vegetation ecosystems, Water Resour. Res., 42, 387–403, https://doi.org/10.1029/2005wr004582, 2006.

670 Mariotti, G.: Revisiting salt marsh resilience to sea level rise: are ponds responsible for permanent land loss? J. Geophys. Res.: Earth Surf., 121, 1391–1407, https://doi.org/10.1002/2016jf003900, 2016.

Mariotti, G. and Fagherazzi, S.: A numerical model for the coupled long-term evolution of salt marshes and tidal flats, J. Geophys. Res., 115, F01004, https://doi.org/10.1029/2009jf001326, 2010.

Mariotti, G. and Fagherazzi, S.: Critical width of tidal flats triggers marsh collapse in the absence of sea-level rise, Proc. Natl. 675 Acad. Sci. U. S. A., 110, 5353–5356, https://doi.org/10.1073/pnas.1219600110, 2013.

McGlathery, K., Reidenbach, M., D'Odorico, P., Fagherazzi, S., Pace, M., and Porter, J.: Nonlinear dynamics and alternative stable states in shallow coastal systems, Oceanography, 26, 220–231, https://doi.org/10.5670/oceanog.2013.66, 2013.

McLeod, E., Chmura, G. L., Bouillon, S., Salm, R., Björk, M., Duarte, C. M., Lovelock, C. E., Schlesinger, W. H., and 680 Silliman, B. R.: A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2, Front. Ecol. Environ., 9, 552–560, https://doi.org/10.1890/110004, 2011.

Miller, W. R. and Egler, F. E.: Vegetation of the wequetequock-pawcatuck tidal-marshes, Connecticut, Ecol. Monogr., 20, 143–172, https://doi.org/10.2307/1943548, 1950.

Millette, T. L., Argow, B. A., Marcano, E., Hayward, C., Hopkinson, C. S., and Valentine, V.: Salt marsh geomorphological analysesviaintegration of multitemporal multispectral remote sensing with LIDAR and GIS, J. Coast. Res., 265, 809–816, https://doi.org/10.2112/jcoastres-d-09-00101.1, 2010.

Moffett, K., Nardin, W., Silvestri, S., Wang, C., and Temmerman, S.: Multiple stable states and catastrophic shifts in coastal wetlands: progress, challenges, and opportunities in validating theory using remote sensing and other methods, Remote Sens., 7, 10184–10226, https://doi.org/10.3390/rs70810184, 2015.

Morris, J. T., Sundareshwar, P. V., Nietch, C. T., Kjerfve, B., and Cahoon, D. R.: 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, 2002.

Morton, R. A., Tiling, G., and Ferina, N. F.: Causes of hot-spot wetland loss in the Mississippi delta plain, Environ. Geosci., 10, 71–80, https://doi.org/10.1306/eg100202007, 2003.

Moskalski, S. M. and Sommerfield, C. K.: Suspended sediment deposition and trapping efficiency in a Delaware salt marsh, Geomorphology, 139-140, 195–204, https://doi.org/10.1016/j.geomorph.2011.10.018, 2012.

Mudd, S. M., D'Alpaos, A., and Morris, J. T.: How does vegetation affect sedimentation on tidal marshes? Investigating particle capture and hydrodynamic controls on biologically mediated sedimentation, J. Geophys. Res., 115, F03029, https://doi.org/10.1029/2009jf001566, 2010.

Neumeier, U. R. S. and Amos, C. L.: The influence of vegetation on turbulence and flow velocities in European salt-marshes, Sedimentology, 53, 259–277, https://doi.org/10.1111/j.1365-3091.2006.00772.x, 2006.

Nyman, J. A., Walters, R. J., Delaune, R. D., and Patrick, W. H.: Marsh vertical accretion via vegetative growth, Estuar. Coast. Shelf Sci., 69, 370–380, https://doi.org/10.1016/j.ecss.2006.05.041, 2006.

Ortiz, A. C., Roy, S., and Edmonds, D. A.: Land loss by pond expansion on the Mississippi river delta plain, Geophys. Res. Lett., 44, 3635–3642, https://doi.org/10.1002/2017gl073079, 2017.

Pendleton, E. and Stevenson, J.: Investigations of Marsh Losses at Blackwater Refuge: Final Report, University of Maryland Center for Environmental and Estuarine Studies, Cambridge, MA, 1983.

Penland, S., Wayne, L., Britsch, L. D., Williams, S. J., Beall, A. D., and Butterworth, V. C.: Geomorphic Classification of Coastal Land Loss between 1932 and 1990 in the Mississippi River Delta Plain, Southeastern Louisiana. USGS Open File Report 00-417, U.S. Geological Survey, Coastal Marine Geology Program, Woods Hole, MA, 2000.

Redfield, A. C.: Development of a New England salt marsh, Ecol. Monogr., 42, 201–237, https://doi.org/10.2307/1942263, 1972.

Reed, D. J.: Sediment dynamics and deposition in a retreating coastal salt marsh, Estuar. Coast. Shelf Sci., 26, 67–79, https://doi.org/10.1016/0272-7714(88)90012-1, 1988.

715   Reed, D. J., Spencer, T., Murray, A. L., French, J. R., and Leonard, L.: Marsh surface sediment deposition and the role of tidal creeks: implications for created and managed coastal marshes, J. Coast. Conserv., 5, 81–90, https://doi.org/10.1007/bf02802742, 1999.

Reitsma, J. M.: Toelichting Bij de Vegetatiekartering Westerschelde 2004 op Basis Vanfalse Colour-Luchtfoto's 1:5000/1:10000, Report, Ministerie van Verkeer en Waterstaat, Rijksinstituut, Adviesdienst Geo-Informatie & ICT, Den Haag, Delft, NL, 2006.

720   Rinaldo, A., Fagherazzi, S., Lanzoni, S., Marani, M., and Dietrich, W. E.: Tidal networks: 2. Watershed delineation and comparative network morphology, Water Resour. Res., 35, 3905–3917, https://doi.org/10.1029/1999wr900237, 1999a.

Rinaldo, A., Fagherazzi, S., Lanzoni, S., Marani, M., and Dietrich, W. E.: Tidal networks: 3. Landscape-forming discharges and studies in empirical geomorphic relationships, Water Resour. Res., 35, 3919–3929,
725   https://doi.org/10.1029/1999wr900238, 1999b.

Roner, M., D'Alpaos, A., Ghinassi, M., Marani, M., Silvestri, S., Franceschinis, E., and Realdon, N.: Spatial variation of salt-marsh organic and inorganic deposition and organic carbon accumulation: inferences from the Venice Lagoon, Italy, Adv. Water Resour., 93, 276–287, https://doi.org/10.1016/j.advwatres.2015.11.011, 2016.

Sarretta, A., Pillon, S., Molinaroli, E., Guerzoni, S., and Fontolan, G.: Sediment budget in the Lagoon of Venice, Italy, Cont.
730   Shelf Res., 30, 934–949, https://doi.org/10.1016/j.csr.2009.07.002, 2010.

Scheffer, M., Carpenter, S., Foley, J. A., Folke, C., and Walker, B.: Catastrophic shifts in ecosystems, Nature, 413, 591–596, https://doi.org/10.1038/35098000, 2001.

Scheffer, M. and Carpenter, S. R.: Catastrophic regime shifts in ecosystems: linking theory to observation, Trends Ecol. Evol., 18, 648–656, https://doi.org/10.1016/j.tree.2003.09.002, 2003.

735   Schepers, L., Kirwan, M., Guntenspergen, G., and Temmerman, S.: Spatio-temporal development of vegetation die-off in a submerging coastal marsh, Limnol. Oceanogr., 62, 137–150, https://doi.org/10.1002/lno.10381, 2017.

Silliman, B. R.: Drought, snails, and large-scale die-off of Southern U.S. salt marshes, Science, 310, 1803–1806, https://doi.org/10.1126/science.1118229, 2005.

Silvestri, S., Defina, A., and Marani, M.: Tidal regime, salinity and salt marsh plant zonation, Estuar. Coast. Shelf Sci., 62,
740   119–130, https://doi.org/10.1016/j.ecss.2004.08.010, 2005.

Stevenson, J. C., Kearney, M. S., and Pendleton, E. C.: Sedimentation and erosion in a Chesapeake Bay brackish marsh system, Mar. Geol., 67, 213–235, https://doi.org/10.1016/0025-3227(85)90093-3, 1985.

Temmerman, S., Govers, G., Meire, P., and Wartel, S.: Modelling long-term tidal marsh growth under changing tidal conditions and suspended sediment concentrations, Scheldt estuary, Belgium, Mar. Geol., 193, 151–169,
745   https://doi.org/10.1016/s0025-3227(02)00642-4, 2003a.

Temmerman, S., Govers, G., Meire, P., and Wartel, S.: Simulating the long-term development of levee–basin topography on tidal marshes, Geomorphology, 63, 39–55, https://doi.org/10.1016/j.geomorph.2004.03.004, 2004.

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

Temmerman, S. and Kirwan, M. L.: Building land with a rising sea, Science, 349, 588–589, https://doi.org/10.1126/science.aac8312, 2015.

Temmerman, S., Meire, P., Bouma, T. J., Herman, P. M. J., Ysebaert, T., and De Vriend, H. J.: Ecosystem-based coastal defence in the face of global change, Nature, 504, 79–83, https://doi.org/10.1038/nature12859, 2013.

Temmerman, S., Moonen, P., Schoelynck, J., Govers, G., and Bouma, T. J.: Impact of vegetation die-off on spatial flow patterns over a tidal marsh, Geophys. Res. Lett., 39, L03406, https://doi.org/10.1029/2011gl050502, 2012.

Tommasini, L., Carniello, L., Ghinassi, M., Roner, M., and D'Alpaos, A.: Changes in the wind-wave field and related salt-marsh lateral erosion: inferences from the evolution of the Venice Lagoon in the last four centuries, Earth Surf. Process. Landf., 44, 1633–1646, https://doi.org/10.1002/esp.4599, 2019.

Turner, R. E. and Rao, Y. S.: Relationships between wetland fragmentation and recent hydrologic changes in a deltaic coast, estuaries, 13, 272–281, https://doi.org/10.2307/1351918, 1990.

Ursino, N., Silvestri, S., and Marani, M.: Subsurface flow and vegetation patterns in tidal environments, Water Resour. Res., 40, 191–201, https://doi.org/10.1029/2003wr002702, 2004.

van Belzen, J., van de Koppel, J., Kirwan, M. L., van der Wal, D., Herman, P. M. J., Dakos, V., Kéfi, S., Scheffer, M., Guntenspergen, G. R., and Bouma, T. J.: Vegetation recovery in tidal marshes reveals critical slowing down under increased inundation, Nat. Commun., 8, 15811, https://doi.org/10.1038/ncomms15811, 2017.

van Damme, S., Struyf, E., Maris, T., Ysebaert, T., Dehairs, F., Tackx, M., Heip, C., and Meire, P.: Spatial and temporal patterns of water quality along the estuarine salinity gradient of the Scheldt estuary (Belgium and The Netherlands): results of an integrated monitoring approach, Hydrobiologia, 540, 29–45, https://doi.org/10.1007/s10750-004-7102-2, 2005.

van der Pluijm, A. M. and de Jong, D. J.: Historisch Overzicht Schorareaal in Zuid-West Nederland, Report Werkdocument RIKZ/OS-98.860 x, Rijkswaterstaat - Rijksinstituut voor Kust en Zee, Utrecht, NL, 1998.

van Wesenbeeck, B. K., van de Koppel, J., Herman, P. M. J., Bertness, M. D., van der Wal, D., Bakker, J. P., and Bouma, T. J.: Potential for sudden shifts in transient systems: distinguishing between local and landscape-scale processes, Ecosystems, 11, 1133–1141, https://doi.org/10.1007/s10021-008-9184-6, 2008.

Vandenbruwaene, W., Bouma, T. J., Meire, P., and Temmerman, S.: Bio-geomorphic effects on tidal channel evolution: impact of vegetation establishment and tidal prism change, Earth Surf. Process. Landf., 38, 122–132, https://doi.org/10.1002/esp.3265, 2013.

Vandenbruwaene, W., Temmerman, S., Bouma, T. J., Klaassen, P. C., de Vries, M. B., Callaghan, D. P., van Steeg, P., Dekker, F., van Duren, L. A., Martini, E., Balke, T., Biermans, G., Schoelynck, J., and Meire, P.: Flow interaction

with dynamic vegetation patches: implications for biogeomorphic evolution of a tidal landscape, J. Geophys. Res.: Earth Surf., 116, 155–170, https://doi.org/10.1029/2010jf001788, 2011.

Venier, C., D'Alpaos, A., and Marani, M.: Evaluation of sediment properties using wind and turbidity observations in the shallow tidal areas of the Venice Lagoon, J. Geophys. Res.: Earth Surf., 119, 1604–1616, https://doi.org/10.1002/2013jf003019, 2014.

Wamsley, T. V., Cialone, M. A., Smith, J. M., Atkinson, J. H., and Rosati, J. D.: The potential of wetlands in reducing storm surge, Ocean Eng., 37, 59–68, https://doi.org/10.1016/j.oceaneng.2009.07.018, 2010.

Wang, C., Menenti, M., Stoll, M. P., Feola, A., Belluco, E., and Marani, M.: Separation of ground and low vegetation signatures in LiDAR measurements of salt-marsh environments, IEEE Trans. Geosci. Remote Sens., 47, 2014–2023, https://doi.org/10.1109/tgrs.2008.2010490, 2009.

Wang, C. and Temmerman, S.: Does biogeomorphic feedback lead to abrupt shifts between alternative landscape states? An empirical study on intertidal flats and marshes, J. Geophys. Res.: Earth Surf., 118, 229–240, https://doi.org/10.1029/2012jf002474, 2013.

Wilson, C. A., Hughes, Z. J., FitzGerald, D. M., Hopkinson, C. S., Valentine, V., and Kolker, A. S.: Saltmarsh pool and tidal creek morphodynamics: dynamic equilibrium of northern latitude saltmarshes?, Geomorphology, 213, 99–115, https://doi.org/10.1016/j.geomorph.2014.01.002, 2014.

Wilson, K. R., Kelley, J. T., Croitoru, A., Dionne, M., Belknap, D. F., and Steneck, R.: Stratigraphic and ecophysical characterizations of salt pools: dynamic landforms of the webhannet salt marsh, Wells, ME, USA, Estuaries Coasts, 32, 855–870, https://doi.org/10.1007/s12237-009-9203-7, 2009.

Wilson, K. R., Kelley, J. T., Tanner, B. R., and Belknap, D. F.: Probing the origins and stratigraphic signature of salt pools from north-temperate marshes in Maine, U.S.A, J. Coast. Res., 26, 1007–1026, https://doi.org/10.2112/jcoastres-d-10-00007.1, 2010.

Yang, S. L., Shi, B. W., Bouma, T. J., Ysebaert, T., and Luo, X. X.: Wave attenuation at a salt marsh margin: a case study of an exposed coast on the yangtze estuary, estuaries coasts, 35, 169–182, https://doi.org/10.1007/s12237-011-9424-4, 2012.

Zaggia, L. and Ferla, M.: Studies on water and suspended sediment transport at the Venice Lagoon inlets, in: Proceedings of the Fifth International Symposium WAVES, Madrid, Spain, 3-7 July 2005, 1–10, 2005.

Zirino, A., Elwany, H., Neira, C., Maicu, F., Mendoza, G., and Levin, L. A.: Salinity and its variability in the Lagoon of Venice, 2000–2009, Adv. Oceanogr. Limnol., 5, 41–59, https://doi.org/10.4081/aiol.2014.5350, 2014.

Table 1. Overview of bare patch number (bare patches smaller than 1 m^2 were excluded from the analysis), Lidar pixels and GPS measurements of the three field sites.

Field site	Type	Number of bare patches	Number of Lidar pixels	Number of GPS measurements
Saeftinghe (NL)	Marsh	-	67729	-
	Unconnected bare patch	97	1722	-
	Connected bare patch	58	12651	-
San Felice (IT)	Marsh	-	361261	340
	Unconnected bare patch	70	2556	-
	Connected bare patch	124	260140	95
Blackwater (USA)	Marsh	-	184871	93
	Unconnected bare patch	255	-	36
	Connected bare patch	227	-	31

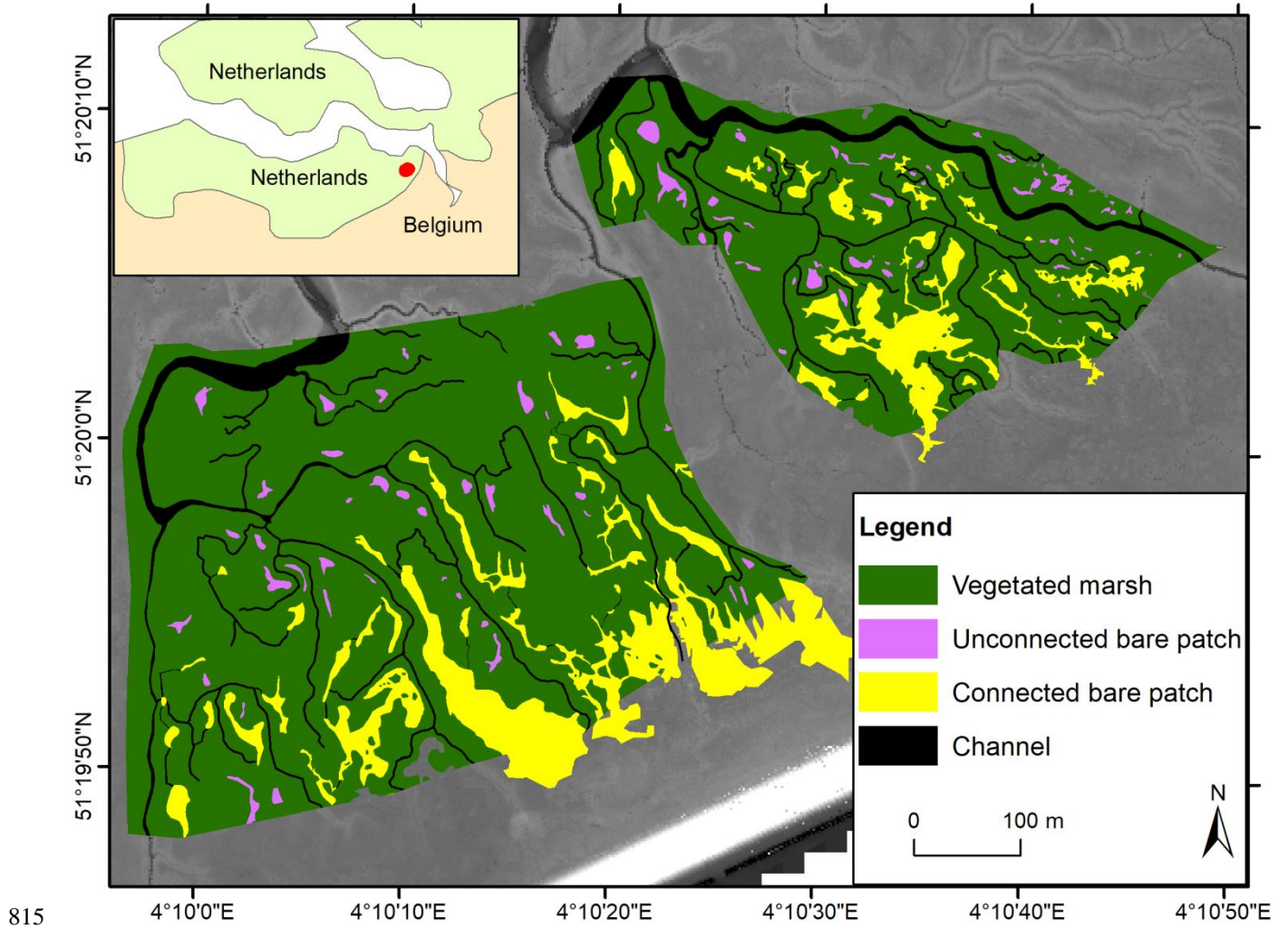



Figure 1: Study area in the Saeftinghe marsh. Spatial distribution of vegetated marshes, unconnected bare patches and connected bare patches in 2004 with LIDAR images as background.

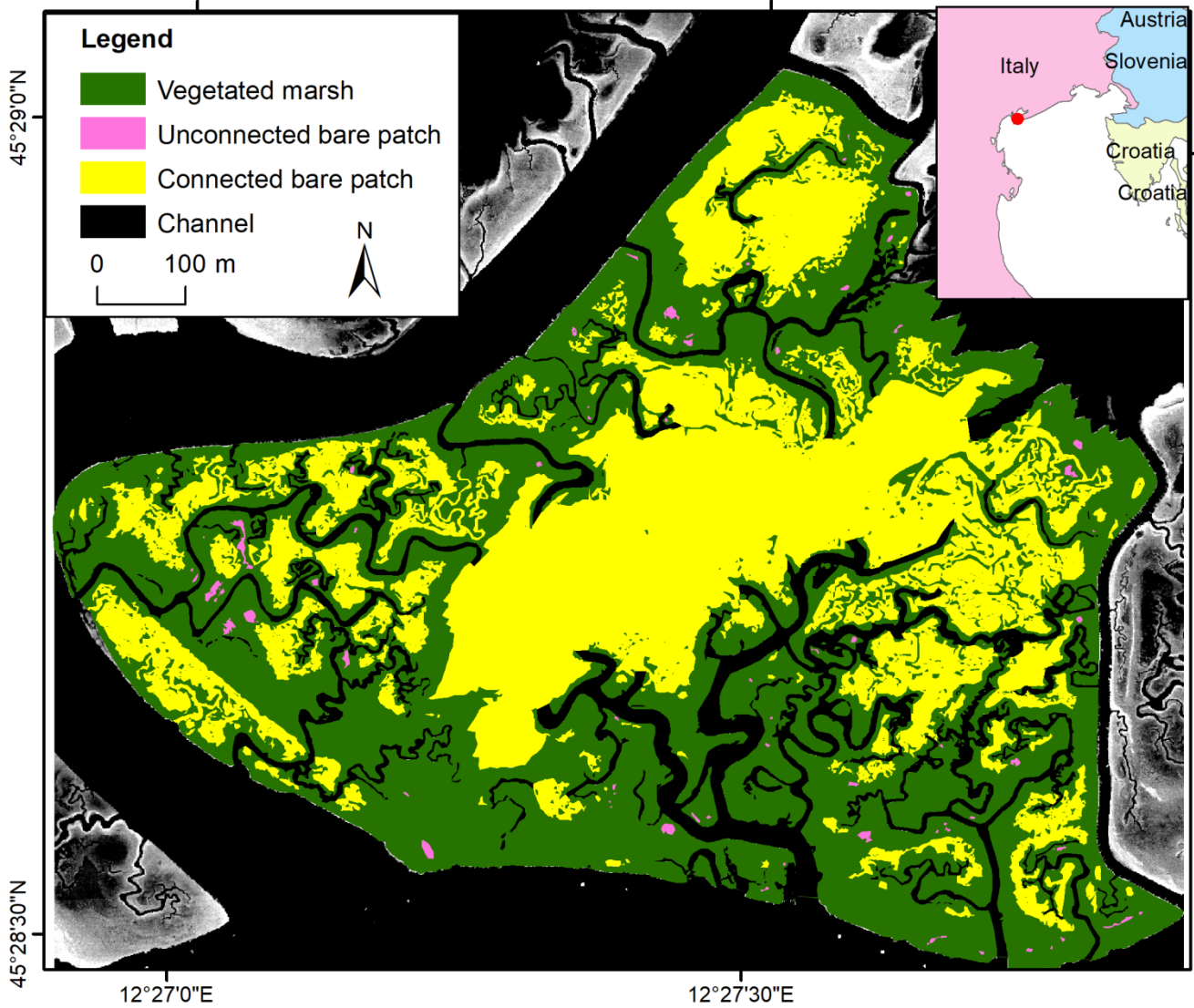



Figure 2: Study area in the San Felice marsh. Spatial distribution of vegetated marshes, unconnected bare patches and connected bare patches in 2002 with LIDAR images as background.

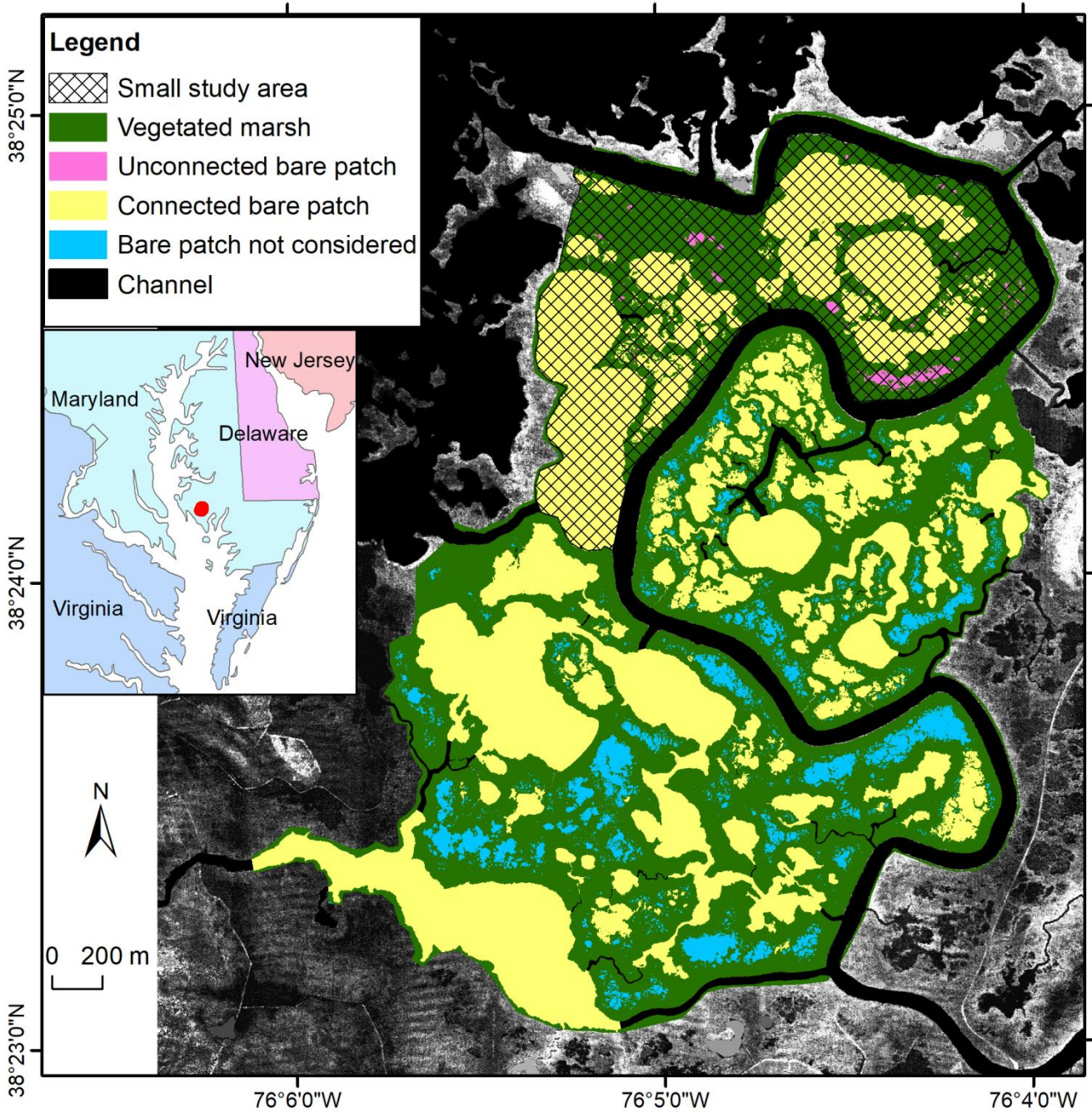



Figure 3: Study area in the Blackwater Marshes. Spatial distribution of vegetated marshes, unconnected bare patches and connected bare patches in 2010 with LIDAR images as background.

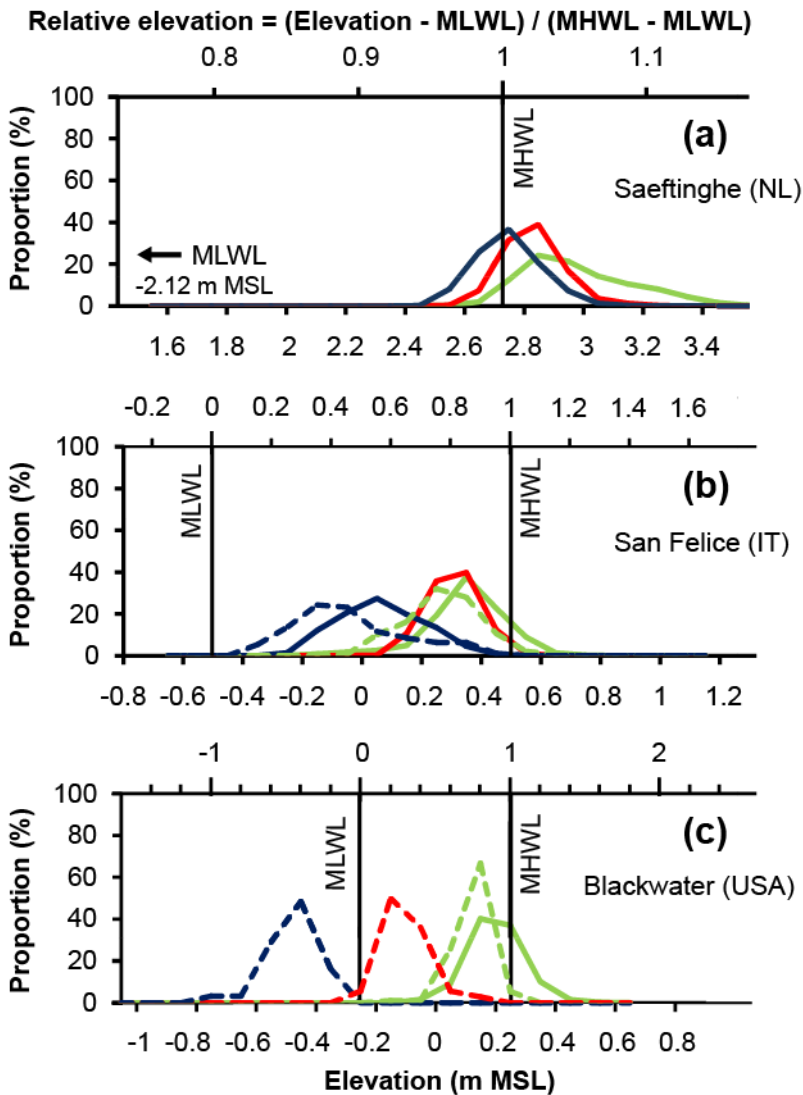



Figure 4: Elevation distribution of vegetated marshes, unconnected bare patches and connected bare patches based on LIDAR surveys and field surveys for (a) Saeftinghe, (b) San Felice and (c) Blackwater. In each figure, the bottom X-axis shows the absolute elevation (in m relative to mean sea level) and the top X-axis shows the relative elevation (dimensionless, as defined in Eq. (1)). The proportion on the Y-axis is calculated based on LIDAR or field measurements as the number of pixels or samples in each elevation class (every 0.1 m) relative to the total number of pixels or samples for each feature. The exact numbers in each category are given in Table 1.

830 The MLWL in Saeftinghe is 2.12 m lower than MSL, which is outside of the range of the main X-axis in Figure 4a. MLWL and MHWL definitions at Blackwater are approximate since water level changes are dominated by meteorological rather than astronomical influences.

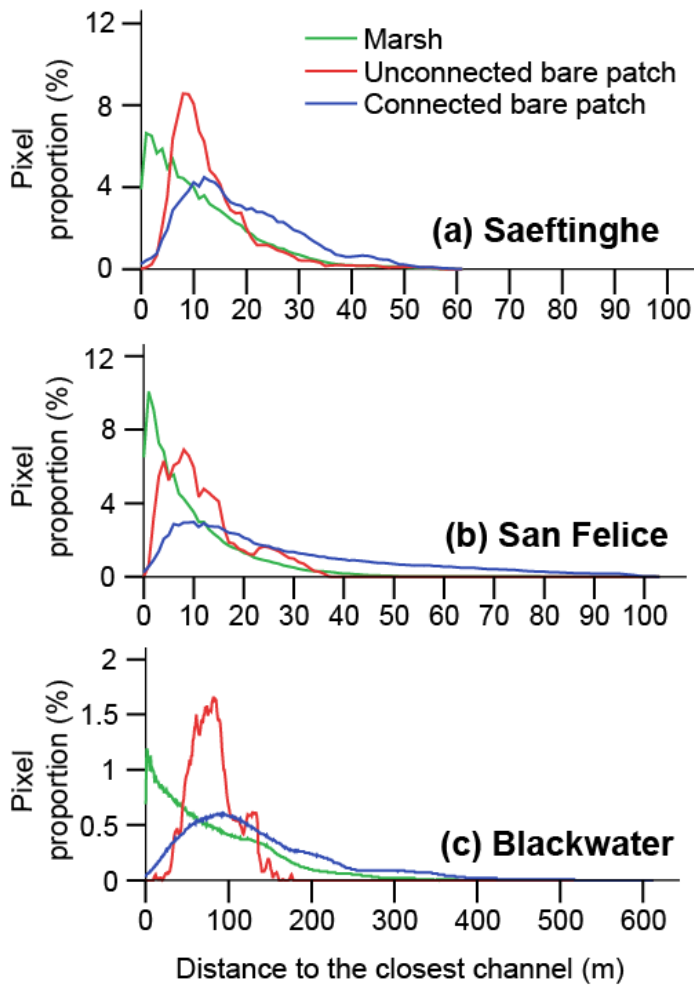



Figure 5: Frequency distribution of distances to the closest channel in (a) Saeftinghe, (b) San Felice, and (c) Blackwater. The proportion is calculated as the number of pixels in each distance class (every 1 m) relative to the total number of pixels for each feature,
835 i.e., vegetated marshes, unconnected bare patches or connected bare patches.

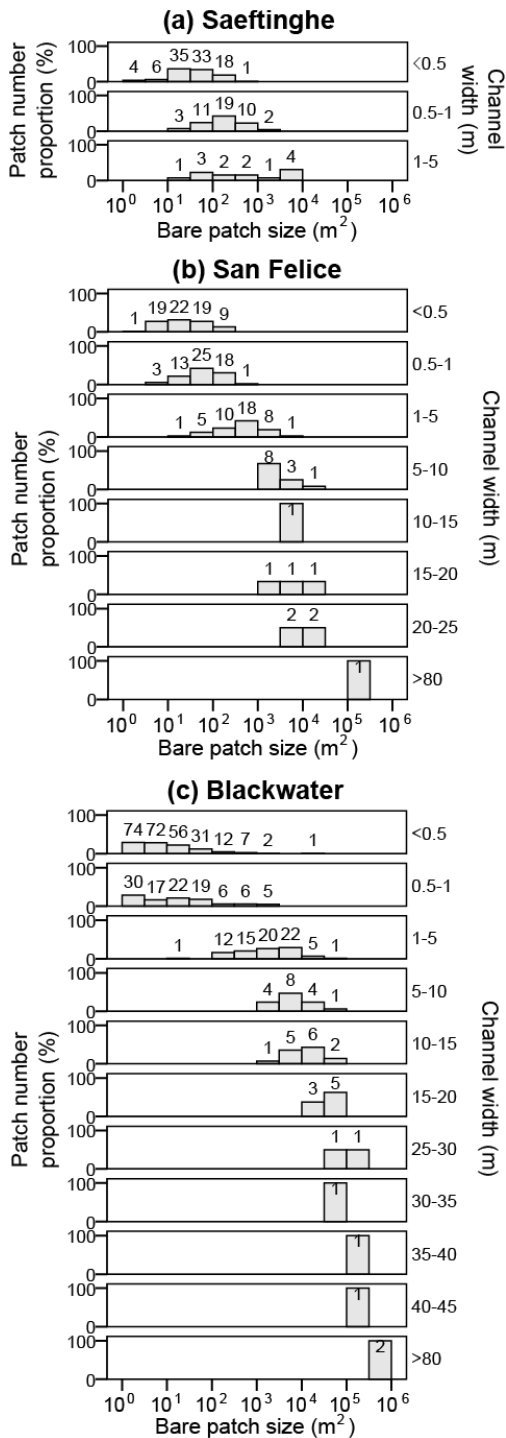



Figure 6: Frequency distribution of bare patch sizes in relation to the connected channel width in (a) Saeftinghe, (b) San Felice, and (c) Blackwater. Note the X-axis is in logarithmic scale. The patch number proportion is calculated as the number of bare patches in each class of bare patch size relative to the total number of bare patches for each category of channel width. The number of bare patches in each size class is labeled at the top of the bars.

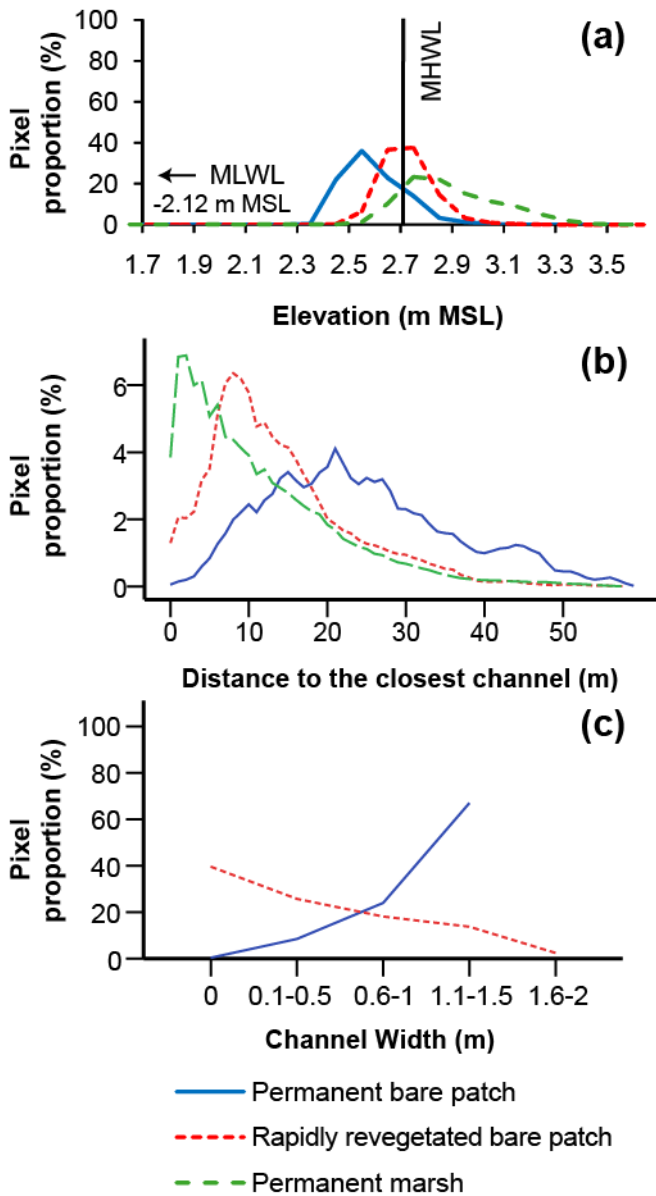
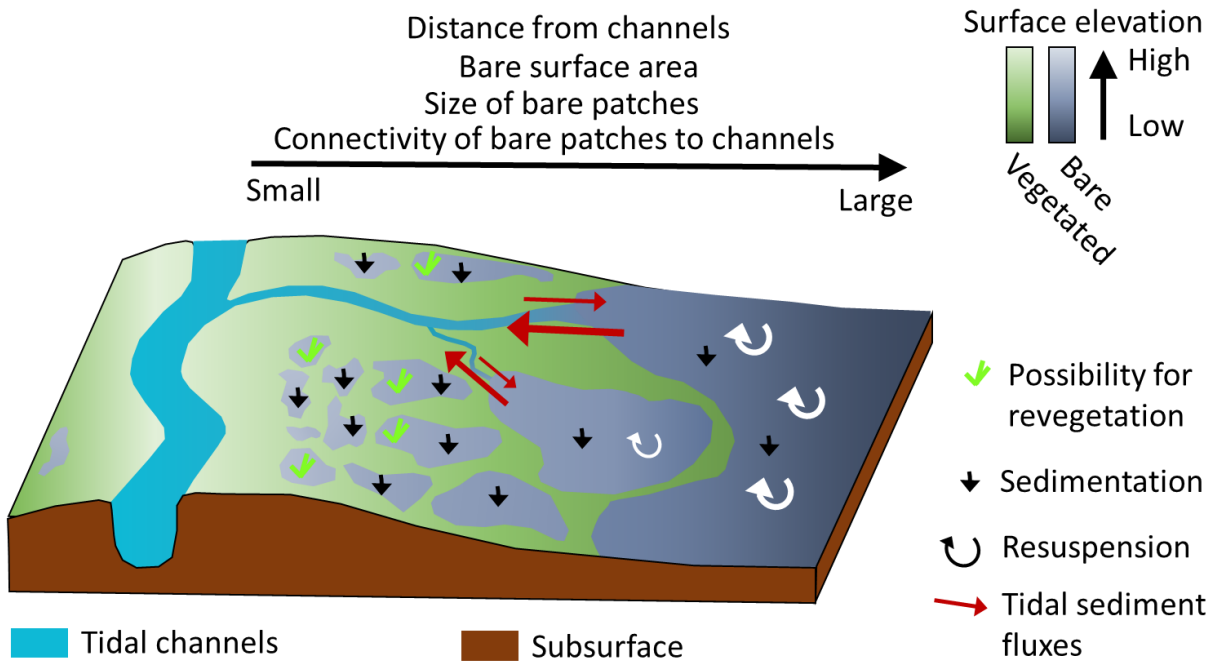



Figure 7: Frequency distribution of (a) elevation, (b) distances to the closest channel, and (c) the connected channel width for permanent bare patches, rapidly revegetated bare patches and permanent marsh areas in Saeftinghe. The elevation is relative to mean sea level and binned into 0.1 m intervals. The proportion in panel (a) is calculated as the number of pixels in each elevation class (every 0.1 m) relative to the total number of pixels for each feature. The proportion in panel (b) is calculated as the number of pixels in each distance class (every 1.0 m) relative to the total number of pixels for each feature. The proportion in panel (c) is calculated as the number of pixels in each class of channel width relative to the total number of pixels for each feature.

850 **Figure 8: Conceptual model summarizing results and interpretations.** The size of the arrows is indicative for hypothetical magnitudes of sediment accretion (black arrows), erosion (white arrows) and tidal sediment fluxes (red arrows). The hypotheses are that the larger bare patches connected to the channels experience stronger incoming and outgoing tidal currents, and more waves during flooding (longer wind fetch length), favoring erosion and tidal export of sediments via the channels; while the smaller unconnected bare patches experience weak tidal currents (because of obstruction by surrounding vegetation), few or no waves (small wind fetch length), limiting erosion and allowing
855 accretion.