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Sediment Disposals in Estuarine Channels Alter the Eco-Morphology of Intertidal Flats

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Key Points:
- Channel disposals result in a nonlocal sequence of eco-morphological consequences for adjacent intertidal flats
- Sediment disposals in a channel turned the adjacent eroding intertidal flat into an accreting and expanding intertidal flat
- The bed level increase is associated with reduction in grain size, increased quantity of benthic macrofauna, and salt marsh progradation

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Abstract Dredging of navigation channels in estuaries affects estuarine morphology and ecosystems. In the Western Scheldt, a two-channel estuary in the Netherlands, the navigation channel is deepened and the sediment is relocated to other parts of the estuary. We analyzed the response of an intertidal flat to sediment disposals in its adjacent channel. Decades of high-frequency monitoring data from the intertidal flat show a shift from erosion toward accretion and reveal a sequence of cascading eco-morphological consequences. We document significant morphological changes not only at the disposal sites, but also at the nearby intertidal flats. Disposals influence channel bank migration, driving changes in the evolution of the intertidal flat hydrodynamics, morphology, and grain sizes. The analyzed disposals related to an expansion of the channel bank, an increase in bed level of the intertidal flat, a decrease in flow velocities on this higher elevated flat, and locally a decrease in grain sizes. These changes in turn affect intertidal flat benthic communities (increased in quantity in this case) and the evolution of the adjacent salt marsh (retreated less or even expanded in this case). The shifts in evolution may occur years after dredged disposal begins, especially in zones of the flats farther away from the disposal locations. The consequences of sediment disposals that we identify stress the urgency of managing such interventions with integrated strategies on a system scale.

1. Introduction

Sediments are relocated in estuaries for the deepening and maintenance of navigation channels (e.g., De Vriend et al., 2011; Kerner, 2007; Monge-Ganuzas et al., 2013; Zhu et al., 2015). The economic gains come with possible side effects, such as increased suspended sediment concentrations (Dijkstra et al., 2019; Van Maren et al., 2015), increased estuarine circulation (Zhu et al., 2015), water quality deficiencies (Kerner, 2007), destabilization of ebb flood channel systems (Jeuken & Wang, 2010; Monge-Ganuzas et al., 2013; Wang & Winterwerp, 2001), and morphological changes of intertidal flats (e.g., De Vet et al., 2017; Wang et al., 2015). Sediment is preferably retained within these systems, as extraction would weaken their resilience to sea level rise. Together with salt marshes, intertidal flats provide an important buffering function for flood protection (Reed et al., 2018; Temmerman et al., 2013) and are ecologically valuable (Barbier et al., 2011; Smaal & Nienhuis, 1992). Apart from minimizing costs, system managers are increasingly challenged to limit the consequences of sediment disposals such that the values of these areas are preserved for future generations.

To achieve such sustainable sediment management strategies, it is crucial to examine interactions between sediment disposals and the eco-morphological evolution of intertidal flats. However, these relationships have lacked strong field evidence. Identifying these links is complicated in part because morphological impacts may arise long after disposals occur. Furthermore, other anthropogenic and natural pressures affect these estuarine systems simultaneously (Elliott & Whitfield, 2011).

For these reasons, we studied an intertidal flat (in the Western Scheldt estuary, the Netherlands) that experienced significant and abrupt changes in its morphological and ecological evolution after the introduction of dredge disposals in the adjacent channel. These changes were captured by decades of approximately monthly/annual measurements of channel, intertidal flat, and salt marsh morphology, along with

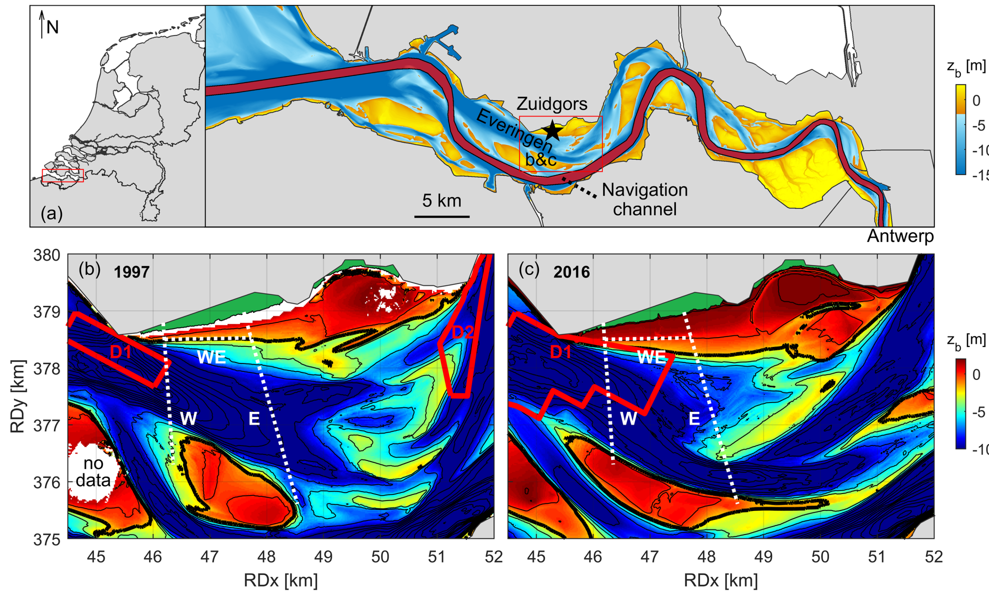

Figure 1. An overview of the Western Scheldt is provided in (a), with its position within the Netherlands. The Everingen channel is indicated, the navigation channel is marked with the red polygon, and the Zuidgors intertidal flat is marked with the star. The area of interest is shown in more detail in (b) with the 1997 bathymetry (single-beam) and in (c) with the 2016 bathymetry (single-beam and LiDAR). In both maps, the 2 m contour lines are indicated in black, with the mean low water contours marked as thick lines and the salt marshes as green polygons. The disposal locations within this area (D1 and D2) are marked with red lines. Transects W, E, and WE (along the 1997 mean low water line) are shown with the dotted white lines. The elevations in the color bars are truncated.

measurements of grain sizes and benthic macrofauna on the intertidal flat. We extended these data with numerical hydrodynamic model simulations to characterize the associated changes in flow velocities. Our analyses revealed a sequence of consequences of channel disposals changing the evolution of the channel and intertidal flat hydrodynamics, morphology, and grain sizes, along with benthic communities and the adjacent salt marsh morphology.

2. Data and Methods

The Western Scheldt (Figure 1a), the Netherlands, is a two-channel estuary. Its navigation channel toward the Port of Antwerp is deepened and maintained (De Vriend et al., 2011; Wang et al., 2015). This results in approximately 10×10^6 m^3 (Mm^3) of sediments deliberately relocated within the system each year. Net extractions from the system are no longer practised. Disposal location D1 in the Everingen channel has been in use since 1997 (Figure 1b). This disposal location was slightly extended in 2000 (Figure 1c). On average, 1.6 Mm^3 has been disposed annually at this location. We consider the effects of these disposals on the nearby intertidal flat Zuidgors (Figure 1a). The mean tidal range in this area equals 4.2 m. The effects of other disposal locations in the Western Scheldt are not part of this study (e.g., as D2 in Figure 1b expired in 2001).

Extensive data on morphology, bed composition, and ecology have been measured in the Zuidgors area by Rijkswaterstaat (Dutch Ministry of Infrastructure and the Environment), and the first measurements predate the start of the disposals. The observations analyzed in this study extend from 1992 to 2017. Bathymetric maps were constructed approximately annually, from single-beam sonar and, in recent years, LiDAR measurements (Marijs & Paree, 2004; Wiegmann et al., 2005). Since 1992, the intertidal sections of transects W and E (Figure 1) have been measured annually with dGPS-RTK equipment and include the edges of the adjacent salt marshes. Additionally, frequent point-elevation measurements have been gathered approximately seven times per year at three locations on transect E, by averaging 15 samples each time (based on sediment erosion bars, and since 2008 RTK-dGPS). Disaggregated grain sizes (D_{50}) have been measured annually at four locations on Zuidgors from 1992 to 2011 (locations are indicated in Figure 3b), as part of the MOVE data set (Rijkswaterstaat, 2006). Benthic macrofauna has been sampled on the Zuidgors intertidal flat at 34 randomly distributed locations (Figure 5a), also as part of the MOVE data set. Every location was sampled once between 1994 and 2008. The bed level corresponding to these samples was derived from the annual

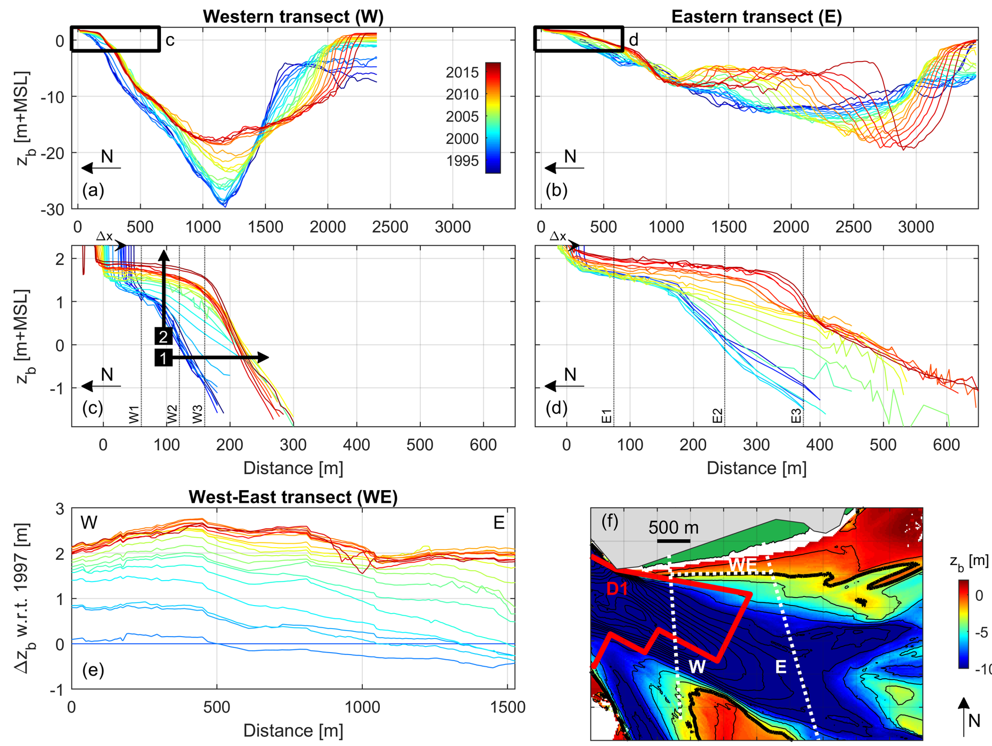

Figure 2. The evolution of transects W, E, and WE (along the 1997 mean low water line) are shown in (a), (b), and (e), respectively, based on single-beam and LiDAR data. The location of these transects are indicated in (f), with the 1997 bathymetry data and the largest extent of the D1 disposal polygon. In (c) and (d) the focus is on the intertidal part of transects W and E (between mean low water and mean high water), only considering available RTK-dGPS measurements. The location of points W1–W3 and E1–E3 are included, for which in Figures 3c and 3d the time series are shown. Also the positive direction of the marsh edge propagation Δx is indicated, see Figure 3e for the time series. The arrows in (c) illustrate the expansion (1) and the successive increase in height (2) of the intertidal flat.

bathymetric maps. Three random cores of 8 cm diameter were taken at the location and then pooled and sieved over a 1 mm mesh in the field. Animals were sorted out in the lab. Undetermined species were removed from the species list (only a few individuals). Finally, annual dredging and disposal quantities have been registered by the Flemish government.

A depth-averaged implementation of the Delft3D hydrodynamic model (Lesser et al., 2004) was used to assess the impact from changes in bathymetry on the flow velocities in the estuary. The model schematization, with a resolution of 40–60 m in the area of interest, covers the full Western Scheldt. The model has been validated to measured hydrodynamics (Van der Werf et al., 2015). For every simulation, all tidal constituents of one representative month (August 2014) were imposed at the boundaries. This approach allows for a focus on differences in flow velocities solely caused by differences in bathymetry.

3. Results

3.1. Observations on Morphology, Grain Sizes, and Ecology

To determine the morphological response of the Zuidgors intertidal flat to the disposal in the Everingen channel, the bed levels at transects W and E (~1.5 km apart) are shown in Figures 2a and 2b for 1992–2017. At both transects W and E, the intertidal area accreted several meters. As shown in more detail in Figures 2c and 2d, the intertidal flat rapidly expanded near the mean low water line with migration of the channel bank within a few years (Figures 2a and 2b); at higher elevations, the response is slower by an order of magnitude (annotations in Figure 2c). The shape of the intertidal flat at transect E changed, such that the profile below MSL is now almost linear, while above MSL a sharp transition to a convex-up profile has formed. The transition between the different shapes coincides with a sharp transition in grain size observed in the field: D_{50} of 150–200 μm for the linear section and approximately 50 μm for the convex-up section.

To analyze the detailed morphological evolution over time and to identify the consequence of the disposals (Figure 3a), time series of the bed elevation of three points on each transect are shown in Figures 3c and

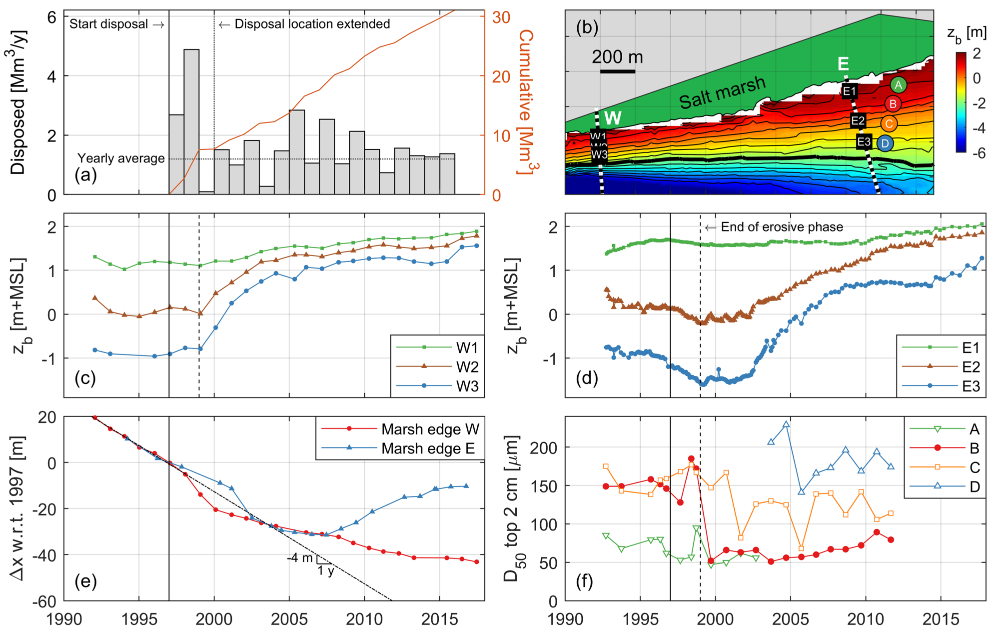

Figure 3. (a) Time series of the annual and cumulative sediment disposal quantities in the D1 disposal location. (b) The locations of transects W and E, elevation measurement points W1–W3 and E1–E3, and grain size sampling points A–D. The 0.5 m contour lines are indicated in black, with the mean low water contour marked as a thick line, based on the 1996/1997 bathymetric data. (c) Elevation time series for points W1–W3, based on a spatial interpolation of the transect data of Figure 2c. (d) Elevation time series for points E1–E3, which are the frequent point-elevation measurements (see section 2). (e) The horizontal propagation of the salt marsh for both transects, based on a spatial interpolation of the transect data of Figures 2c and 2d at mean high water level. (f) The evolution of the median grain size (D_{50}) of the surface sediment (2 cm top layer) of sampling points A–D. The vertical solid and dashed black lines (explained with the annotations) are consistent for the different subfigures.

3d. The elevation of Zuidgors was generally stable or even erosive between 1992 and 1999. After 1999, two years after the start of the disposals but still before the expansion of the disposal location, both transects switched toward accretion. This abrupt trend shift is especially apparent in the frequent point-elevation measurements (Figure 3d). For months the accretion rates exceeded locally 0.05 m/month, and in recent years these still equal to about 0.1 m/year. The peak in accretion rates occurred at transect W more than 3 years earlier than at transect E (~1.5 km eastward of transect W), indicating a significant delay. This is evidence for an eastward propagating accretion front (<0.5 km/year), originating from the disposal location.

This alongshore propagating expansion is captured in more detail by the transect along the original 1997 mean low water line (Figure 2e). The more eastward from the disposal location, the longer the delay until effects arise. Figure 4 shows the boundaries of the area that increased at least 0.5 m in bed level since 1997. Also from this figure, the eastward propagating character of the accretion front is evident, with a smaller celerity for larger distances from the channel (i.e., for higher elevations on the intertidal flat). Bathymetry data were absent for the highest part of the intertidal flat in Figure 4 in 1997, but Figure 3c indicates that W1 (near the salt marsh) reached a 0.5 m bed level increase 13 years after the start of the disposals, while W3 (near the channel) reached this already after 3 years.

The salt marsh edge evolution also changed after the start of the disposals. Van der Wal et al. (2008) traced the position of the salt marsh cliff on transect data (representative for transect W) for the period 1993–2004. In Figure 3e, we extended the analysis with data after 2004 and applied it on both transects. In line with Van der Wal et al. (2008), both salt marsh edges became less erosive around 2000. Salt marsh W retreated with an almost constant rate of 4.0 m/year before 1999, while this rate was only 1.3 m/year between 1999 and 2017. Salt marsh E even expanded after 2008.

Temporal changes of the median grain sizes on the intertidal flat are revealed in Figure 3f, for four locations (~100 m apart) on a cross-shore transect near transect E. Especially at point B, an abrupt but persistent change was observed: a decrease in median grain size from ~150 μm before 1999 (2 years after the start of the disposals), to ~50 μm directly afterward. Closer to the channel, at point C, the grain size reduced less.

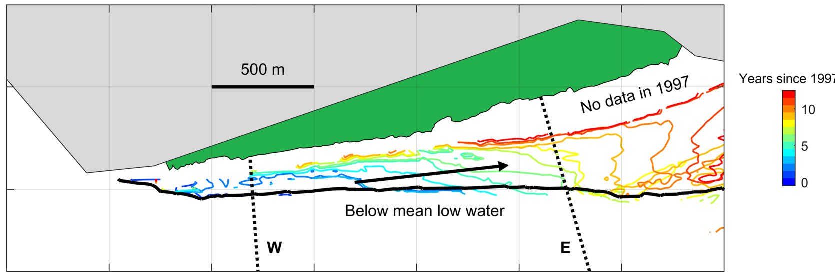

Figure 4. Contour lines of the accretion front, defined as the boundaries of the area where the bed level increased at least 0.5 m compared to 1997, based on the single-beam and LiDAR data. As a reference, transects W and E are shown with the dotted lines, and the 1997 mean low water line is shown with the thick black line. Only the data on the Zuidgors intertidal flat are visualized to improve clarity. The arrow indicates the propagation direction of the front.

Closer to the marsh, at point A, the grain size was already similar to the fine material that settled in the accretion phase at point B.

As benthic macrofauna was not repeatedly sampled at the same locations, we consider the relation between bed elevation and abundance of benthic communities. Changes in elevation of the intertidal flat have important consequences for the benthic communities. Figure 5c indicates, based on randomly distributed samples (Figure 5a), that the abundance correlates with bed elevation (44% of the variance was explained by the bed level) but was not related to the year of sampling ($p > 0.1$). The abundance was on average 7 times larger above MSL + 1 m than below MSL + 1 m. Therefore, for points that substantially heightened (e.g., more than 2.5 m at W3 and E3; Figures 3c and 3d) the abundance of benthic macrofauna increased (approximately by 1 order of magnitude at W3 and E3). Even though the steepness of the intertidal flat initially changed (e.g., Figures 2c and 5b), the area (and hence the abundance of benthic macrofauna) corresponding to bed elevations below MSL + 1 m was in 2016 similar to 1997 (Figure 5b). In contrast, the area of the intertidal flat with bed levels above MSL + 1 m (on average 7 times richer in benthic macrofauna) more than doubled from 1997 to 2016.

3.2. Modeled Hydrodynamics

The accretion on the intertidal flat might be caused by a reduction in tidal energy, resulting from the changed bathymetry (Figures 2 and 6b). To test whether the tidal energy on the intertidal flat reduced, simulated flow velocities using the 1996 and 2014 bathymetry, respectively, are compared. Local peak flow velocities

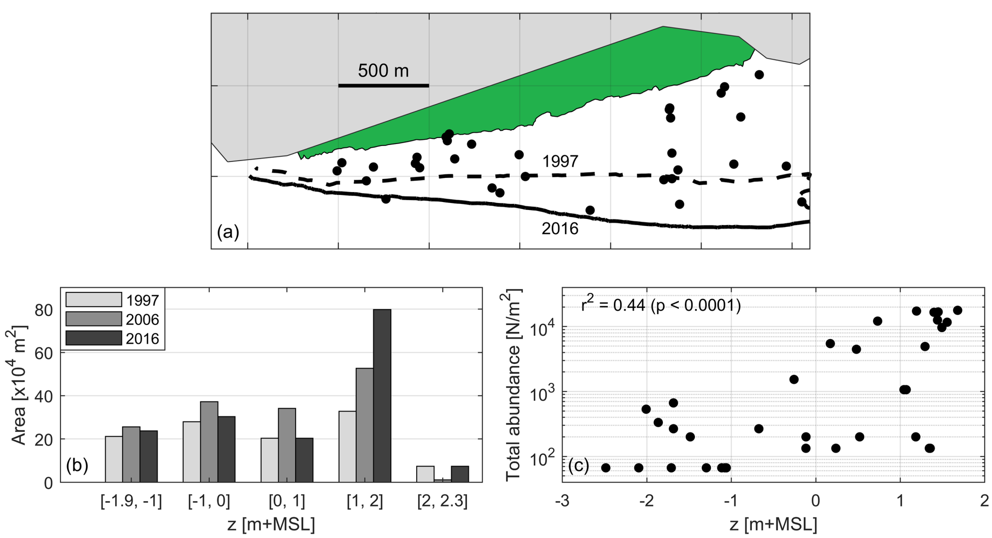

Figure 5. (a) Locations of benthic macrofauna measurements, with the 1997 and 2016 mean low water lines. Every location was sampled once between 1994 and 2008. (b) The area of the intertidal flat in 1997, 2006, and 2016 for 1 m bed level classes (bounded by mean low water and mean high water), based on the single-beam and LiDAR data. Gaps in the 1997 data (Figure 1) were complemented with more recent data. (c) The total abundance of benthic macrofauna versus the bed elevation.

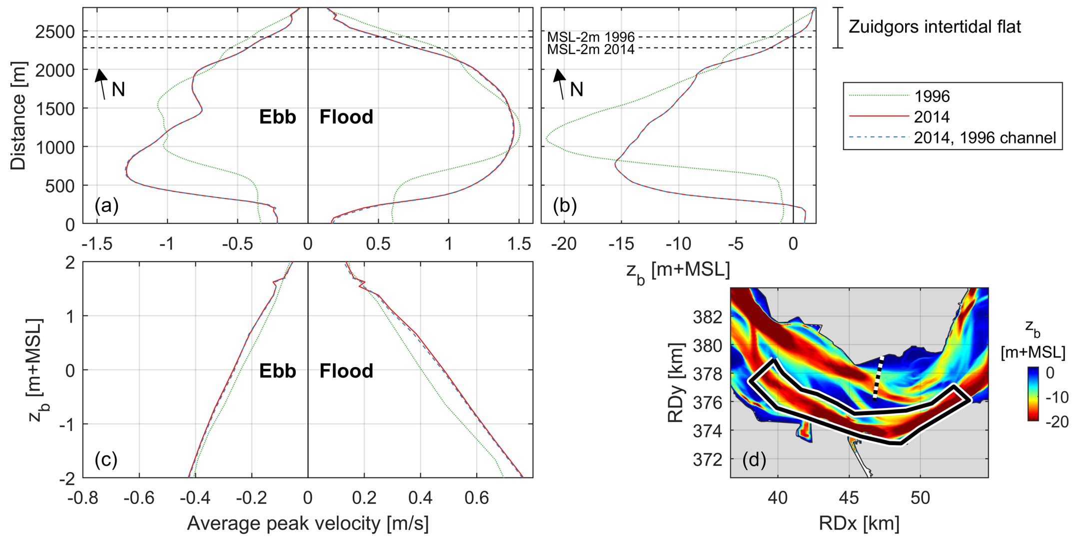

Figure 6. In (a), the average flood and ebb peak velocities are shown along the cross section as indicated in (d). The distance along the cross section increases toward the Zuidgors intertidal flat (which is shown on the top of this graph). These velocities are computed from a 1 month simulation with the 1996 bathymetry, the 2014 bathymetry, and a combination of the 2014 bathymetry with the 1996 navigation channel. The bathymetry data along the cross section are visualized in (b). In (c), the velocities are shown versus the elevation, for the northern part of the cross section (Zuidgors). Finally, in (d) the 2014 bathymetry is visualized with the navigation channel transplanted from 1996 (polygon), with the dotted line indicating the cross section.

on the intertidal flat decreased comparing identical locations (Figure 6a), but increased (especially in flood direction) comparing identical bed levels (Figure 6c). Therefore, the expansion of the intertidal flat coincided actually with an increase in tidal energy on the flat.

To reveal the influence of the deepening of the navigation channel on the intertidal flat hydrodynamics, velocities were simulated also for the 2014 bathymetry with the original 1996 navigation channel (prior to the deepening), see Figure 6d. The implementation of the original navigation channel in the 2014 bathymetry resulted in almost identical flow velocities in the Everingen channel and on the Zuidgors flat as seen with the actual 2014 bathymetry (Figure 6). The navigation channel deepening therefore did not significantly affect the tidal flow velocities on the intertidal flat.

4. Discussion

4.1. Morphodynamic Implications

We showed that sediment disposals in estuarine channels have a direct influence on the channel cross section (shown conceptually in Figure 7; Consequence 1), with consequential changes in evolution of the intertidal areas (channel-flat interaction). The accretion of the intertidal flat was not caused by a reduction in hydrodynamic energy. The velocities on the intertidal flat actually increased considering similar elevations (Figure 6c). The altered velocities result directly from changes in bathymetry caused by the disposals and associated bank and flat expansion, and not from deepening of the navigation channel (Figure 6). Even though dredging activities may eventually induce an estuarine regime shift toward fine sediment import and tidal amplification (Winterwerp & Wang, 2013), the impact on the intertidal flat started within several years and had a more local character. The shift of the channel bank, induced by the adjusted channel geometry, led to intertidal flat expansion (Figure 7; Consequence 2). Our findings suggest that sediment disposals in channel systems can significantly affect proximal intertidal flats if the increased sediment availability impacts channel bank migration.

Intertidal flats do not necessarily respond uniformly in space and with a constant rate to disposals in channels. The adaptations around the mean low water line were an order of magnitude faster (~years) than at higher elevations (~decades). Also, transect E became more erosive for several years before the accretion started (Figure 3d). This lag can be explained by the time required for an accretion front (Figure 4) to propagate along an intertidal flat (Figure 7; Consequence 3)—years, in this case (celerity smaller than

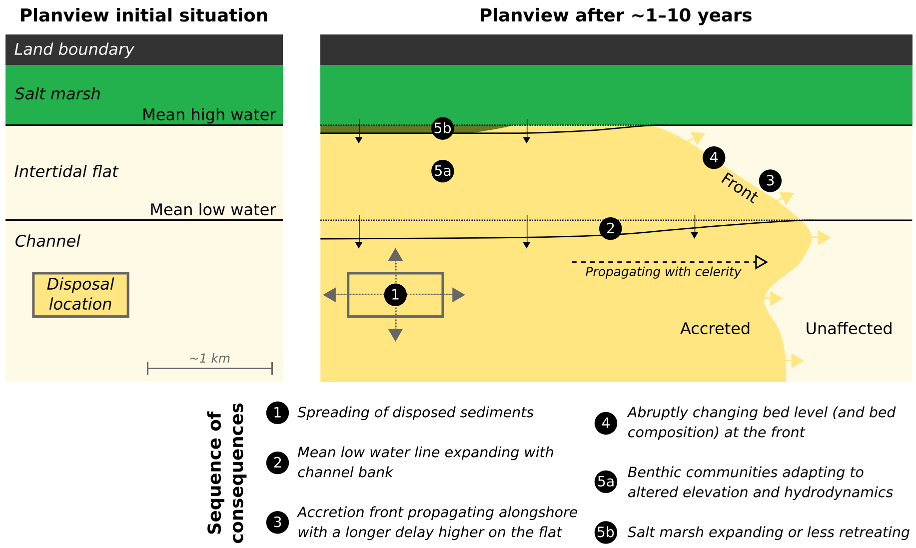

Figure 7. Conceptual diagram on the sequence of consequences of sediment disposals in channels for the eco-morphology of intertidal flats.

0.5 km/year). Such channel-flat systems may thus have an intrinsic adaptation and/or response time of considerable duration. Extreme accretion rates—up to 0.05 m/month in our observations (2 orders of magnitude larger than sea level rise rates)—may then likewise persist for years.

We suggest two mechanisms that might explain changing grain sizes for a changing intertidal flat morphology. On intertidal areas in tide-dominated estuaries, grain sizes typically decrease with reduced current velocities toward higher elevations (Figure 3f; see also Friedrichs, 2011; Yang et al., 2008). Therefore, with reduced tidal velocities on a heightened intertidal flat (Figure 6a), those areas gradually become more amenable to the accumulation of fines. Through this mechanism, accretion of coarser sediments at the lower elevations of a flat could facilitate the accommodation space for fines higher on the flat (see the two stages in Figure 2c). A second mechanism might explain the abrupt changes in grain sizes that we observed. When an erosive intertidal flat becomes accretive, the newly accreted sediments determine the surface bed composition (Figure 7; Consequence 4). In the data, we examined a significant transition in median grain size (from 150 to 50 μm), which occurred abruptly in time (point B in Figure 3f) and coincided precisely with the end of the erosive phase (Figure 3d). The accreted sediment was finer than the sediment originally present in the bed. Abrupt transitions in grain size can occur by changes in morphological evolution during storms (Yang et al., 2008) and seasonal variations (Van der Wal et al., 2010; Yang et al., 2008), but we find the changes induced by disposal persisted. Sediment disposals in channels can be a direct cause of both gradual and abrupt changes in grain size on intertidal areas.

4.2. Ecological Implications

Channel disposals may also affect benthic communities on intertidal flats. Changes in benthic communities have direct ecological consequences, including effects on the foraging habitat for birds, but can also have implications for bed erodibility (Austen et al., 1999; Herman et al., 2001). We showed a significant increase in abundance of benthic macrofauna with bed elevation. Locally, the bed rose vertically as much as 2.5 m, which is associated with increases in quantity of benthic macrofauna (by approximately 1 order of magnitude). Even though the intertidal flat decreased initially in steepness after the start of the disposals, the steepness (i.e., also the area) of the bed level classes below MSL + 1 m was 20 years later again similar as before. Instead, the area of the intertidal flat above MSL + 1 1 m, the richest in benthic macrofauna, more than doubled. There was hence a net increase in total quantity of benthic macrofauna on the intertidal flat. With ongoing heightening, large parts of the intertidal flat may eventually rise above mean high water which implies that intertidal benthic communities may decay again in the future (e.g., by settlement of salt marsh vegetation). Apart from bed level height (i.e., exposure time), Ysebaert and Herman (2002) showed that also grain sizes and current velocity (both relate to bed elevation; Figures 3f and 6c) explain part of the variation in these benthic communities. The sudden decrease in grain sizes from 150 to 50 μm may have induced more sudden changes in benthic communities. Therefore, by altering the morphology, hydrodynamics, and

Figure 8. Photo of the salt marsh edge of the Zuidgors intertidal flat, taken in western direction. The erosive salt marsh cliff is visible with recently settled pioneer vegetation (*Salicornia*) on the intertidal flat in front. Photo taken between both transects, 0.5 km east of transect W and 1.0 km west of transect E, on 26 June 2019.

grain size of an intertidal flat, sediment disposals also can affect the flat's benthic communities (Figure 7; Consequence 5a).

The immediate and diffuse effects of sediment disposals on the evolution of intertidal flats also have direct consequences for the evolution of adjacent salt marshes. In line with Van der Wal et al. (2008) and Mariotti and Fagherazzi (2010), the local geomorphic regime shift of the intertidal foreshore from erosion toward accretion also induced a shift in evolution of the salt marsh edge (Figure 7; Consequence 5b). Cross-shore arrays of wooden poles (~100 m arrays) deployed in 1992 near transect E might have influenced this marsh evolution locally. However, the consistent reduction in salt marsh erosion at both transects, promptly after the start of the disposals, is only explained by these disposals. After the heightening of the foreshore, salt marsh edge erosion was reduced. This can be ascribed to reduced wave forcing (Callaghan et al., 2010; Houser & Hill, 2010; Pethick, 2001), reduced current velocities in front of the marsh (Figure 6a; see also Bouma et al., 2005), reduced inundation times (Balke et al., 2014), related faster recovery from episodic erosion (Van Belzen et al., 2017), and less extreme bed dynamics on higher intertidal forelands (Bouma et al., 2016; Hu et al., 2017). With ongoing accretion of the intertidal flat, it is expected that the full salt marsh will eventually expand. This has been the case for transect E since 2008 (Figure 3e; see also Balke et al., 2014). Even though the salt marsh cliff has still been eroding at transect W, there are signs of a future expansion also there: pioneer vegetation (*Salicornia*) has settled in front of the eroding cliff (observed 26 June 2019; Figure 8).

4.3. Implications for Estuarine Management Strategies

We suggest that strategies for estuarine sediment management need to consider and address, with an integrated approach, the nonlocal sequence of morphodynamic and ecological consequences of channel disposals (Figure 7). This includes disposal strategies that influence the morphology of channel banks or intertidal flats, such as when sediments are disposed directly on the channel edges (e.g., Plancke et al., 2014; Van der Wal et al., 2011). The impact on the full system should be considered, as other locations may experience contrasting consequences: for example, if a reach of channel bank near the disposals retreats (e.g., the channel bank opposing the Zuidgors area in Figures 1b, 1c, and 6b). Our results indicate that changes in morphological evolution, grain size, and ecology can occur simultaneously. Still, these eco-morphological responses can take years to decades to arise, especially farther away from the disposal location. This lag challenges monitoring and management of estuary responses to interventions.

In this section of the Western Scheldt, the state of the estuarine system has changed (e.g., Figures 1b and 1c), largely as a result of the disposals. By extension, stopping the disposals would not guarantee a return to the "natural" historical state. System managers must address the possibility of irreversible consequences of sediment management strategies. There are potentially ways to minimize negative impacts and benefit from effects of dredged disposals (e.g., Baptist et al., 2019; Plancke et al., 2014), but this requires a strategy that can account for eco-morphological consequences and is adaptable under intensive eco-morphological monitoring (Depreiter et al., 2012). In the era in which both human interventions and accelerated sea level rise increasingly pressure the resilience of estuaries and their intertidal habitats (Meire et al., 2005), such a sustainable sediment management strategy is crucial.

5. Conclusions

We identified a sequence of eco-morphological consequences for intertidal flats that results from sediment disposals in an estuarine channel. Following such sediment disposals, increased availability of sediment affects the channel geometry and the channel bank migration. The changes in channel bank migration drive changes in the intertidal flat hydrodynamics, morphology, and grain sizes. The formerly erosive intertidal flat has been increasing in bed elevation, the flow velocities reduced with these higher bed elevations, and the grain sizes decreased locally. Intertidal flat benthic communities and the location of the edge of adjacent salt marshes are affected by these changes. The total quantity of benthic macrofauna increased and the salt marsh edge became less erosive and expanded locally. An accretion front was formed that propagated over the intertidal flat, causing alterations in hydrodynamics, morphology, and grain sizes after years of delay, especially farther away from the disposal location. The identified nonlocal sequence of eco-morphological consequences of sediment disposals calls for integrated sediment management strategies on a system scale, sustained by long-term monitoring.

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