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Ecological evaluation of an experimental beneficial use scheme for dredged sediment disposal in shallow tidal waters

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ABSTRACT

An experiment was performed to test an alternative dredging strategy for the Westerschelde estuary. Clean sand dredged from the navigation channel was disposed seawards of an eroding intertidal flat in order to modify morphology and hydrodynamics, improving the multi-channel system with ecologically productive shallow water habitat. Five years of intensive monitoring revealed that part of the disposed sediment moved slowly towards the flat, increasing the very shallow subtidal and intertidal area, as planned. The sand in the impact zone became gradually finer after disposal, possibly due to reduced current velocities. Nevertheless, no changes in macrobenthic biomass, density, species richness and composition were detected in the subtidal zone, also demonstrating rapid macrobenthic recovery. In the intertidal zone, no ecological effects could be revealed superimposed on trends associated with long-term sediment fining. Thus, despite morphological success and absence of detected negative ecological impacts of the experiment, new beneficial habitat was not created.

1. Introduction

In many shallow coastal waters, estuaries and harbours, dredging is carried out to maintain or increase the depth of navigation channels. Both dredging and disposal of dredged material is an environmental concern throughout the world (Van Dolah et al., 1984; Hall, 1994; Wilber et al., 2007). Benthic macrofauna is often used as an indicator for the ecological impact of such disturbances (Roberts et al., 1998; Borja et al., 2000), partly because the macrobenthos integrate the changes in environmental conditions (e.g., Gray, 1974) and partly because of their essential role in the food chain, causing changes in the macrobenthic community to translate into functional changes in the ecosystem (Pearson and Rosenberg, 1978; Warwick, 1986). In many cases, effects of disposal on the benthic community are near-field and short term (Smith and Rule, 2001; Cruz-Motta and Collins, 2004; Fredette and French, 2004; Powilleit et al., 2006; Wilber et al., 2007), although prolonged effects on macrofaunal biomass and composition have been reported (Wildish and Thomas, 1985; Jones, 1986; Harvey et al., 1998; Blanchard and Feder, 2003; Fraser et al., 2006; Skilleter et al., 2006). Magnitude of the impact and recovery depends on the thickness, area and configuration of the disposed layer that buries the benthos, frequency and timing of the dredging operation, the material characteristics of the discharged material (such as organic enrichment, pollutants and sediment grain-size), but also on the characteristics of the receiving habitat (such as sediment characteristics, water depth and hydrodynamic regime) and the community composition and life history and mobility of the species at the disposal site (see reviews by Newell et al. (1998) and Bolam and Rees (2003)).

In recent years, dredged material is increasingly regarded as a potential resource useful for shoreline protection or for creation or restoration of habitats (in particular mudflats and saltmarsh areas) in so-called beneficial use schemes (Ray, 2000; Yozzo et al., 2004; Bolam and Whomersley, 2005). To date, most of the implemented beneficial use schemes are small-scale trials with uncontaminated sediment carried out in the intertidal zone due to concerns over the subsequent movement of material by tidal current and wave action (Widdows et al., 2006). Few attempts have been made of beneficial use schemes in the shallow subtidal zone (Bolam et al., 2006a). However, when carefully planned, beneficial use schemes can allow for transport of disposed material. Thus, material disposed in the shallow subtidal zone may be transported to nearby intertidal zones, enhancing both shallow subtidal and intertidal habitats.

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In line with such developments, an alternative disposal strategy for the Westerschelde (southwest Netherlands) was proposed (Peters et al., 2001) to make beneficial use of dredged material. The Westerschelde is a site of ecological importance, characterized by a multi-channel system with productive intertidal flats (Fig. 1) that accommodate high biomass and diversity of benthic macrofauna, supporting shorebirds, demersal fish and humans. The estuary is of economic importance, providing access to, among others, the port of Antwerp. Continuous maintenance dredging is required to guarantee accessibility and capital dredging is carried out occasionally to deepen and widen the navigation channel to accommodate increasingly larger ships. Until recently, dredged material was disposed in the secondary channels, thereby sustaining the need for dredging, as the material returned to the navigation channel. Model calculations showed that this practice may destabilize the multi-channel estuarine system, if the disposed volumes exceed approximately 10% of the total transport capacity of a macroscale cell composed of a flood and ebb channel surrounding an intertidal flat (Wang and Winterwerp, 2001). Collapse of a multi-channel system into a single channel system would imply loss of ecologically valuable intertidal flats, besides important hydrographic changes. In contrast, the alternative dredging strategy developed for forthcoming dredging operations involves the disposal of material near (eroding) tidal flats, allowing the material to move slowly towards the flats. By reshaping these areas, a more effective ebb–flood current distribution would be created so that the multiple channel system is sustained and dredging efforts could be reduced in the long-term. In addition, current velocities would be reduced on the shoal, allowing finer sediments to settle, further improving the very shallow subtidal and intertidal habitat for macrofauna. After an extensive feasibility study (Flanders Hydraulics Research, 2003), a small-scale in situ disposal test was executed near the Plaat van Walsoorden, an intertidal flat at the polyhaline/mesohaline transition (mean salinity ca 20) (Fig. 1). For the experiment, clean sand was used from maintenance dredging of the navigation channel, both northwest and southeast of the Plaat van Walsoorden. Samples taken at these source sites prior to dredging (February 2004) show that in situ clay (fraction <2 μm) content of the sediment was 1.3 ± 0.1%, and organic matter content was 0.14 ± 0.04%, and that trace metals and organic micro-pollutants were all below the maximum tolerable levels for distribution of this material, as defined for the Netherlands. Overflow during filling of the hopper dredgers would further reduce the amount of fine sediment. In November–December 2004, 500,000 m^3 of this sand was dredged using a hopper dredger and transported through a floating pipeline to a pontoon, from which it was accurately deposited in the shallow waters near the intertidal flat with a diffuser. Due to the morphological success of the test (Plancke et al., 2006), a second small-scale experiment was executed in January–February 2006, northwest of the initial disposal site, disposing another 500,000 m^3 of sand from maintenance dredging, followed by another 900,000 m^3 in the period September 2006 to March 2007 (Vos et al., 2009) using hopper dredgers (Fig. 2). A comprehensive programme was implemented to monitor the morphological and ecological impact of the experiments.

This paper focuses on the ecological effects of the disposal trials. We hypothesized that near-field effects in the subtidal zone would entail a rapid recovery (months) followed by an enhancement of the macrofaunal community on the long-term (years). In the intertidal zone, a slow net siltation was desirable, but excessive sedimentation of either mud or sand would not be acceptable. Negative ecological effects on the intertidal zone were defined as deviations from natural trends of elevation, mud content of the sediment and macrobenthic biomass and species richness (Vos et al., 2009). We used a BACI (Before and After, Control and Impact) design (Underwood, 1991, 1992), trend analysis and multivariate analysis to evaluate the impact of the disposed sediment on ecologically relevant abiotic variables and on the macrobenthic community.

2. Material and methods

2.1. Field sampling and laboratory analysis

Samples were collected both in spring (either April or May) and autumn (September or late August) from 2004 to 2009. In the subtidal zone, sampling started before disposal at site I1, at an undisturbed control site C1, and at a site that experienced long-term disposal of dredged sediment C2 (Fig. 1). Sampling at the 2006 disposal site (I2) started in spring 2006. At each site, 20 sample stations were selected at random for each campaign. In addition, 5

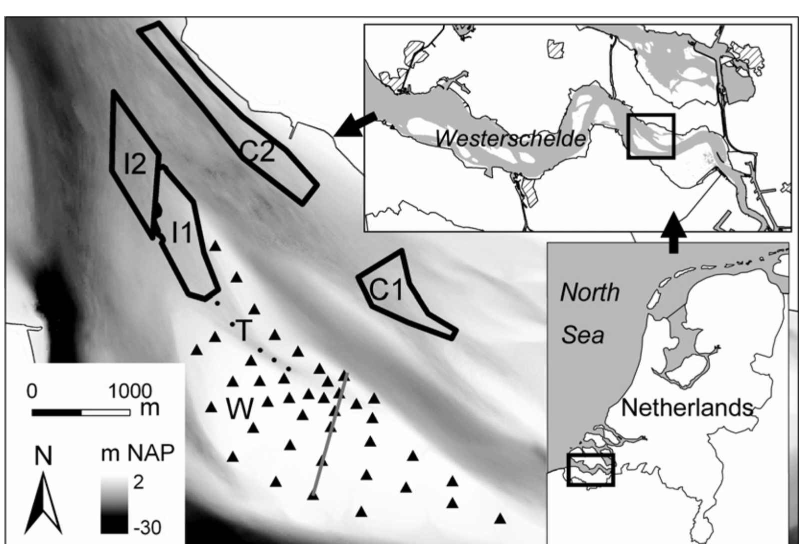

Fig. 1. Study site in the Westerschelde estuary, southwest Netherlands. Subtidal impact sites (I1 and I2) and control sites (C1 and C2) for random sampling and fixed stations (black dots) in subtidal transport zone T. Black triangles show fixed stations in the intertidal zone Plaat van Walsoorden (site W). Grey line in site W is the transect for long-term measurements of bed characteristics. Backdrop shows bathymetry in m relative to NAP (Dutch Ordnance Datum).

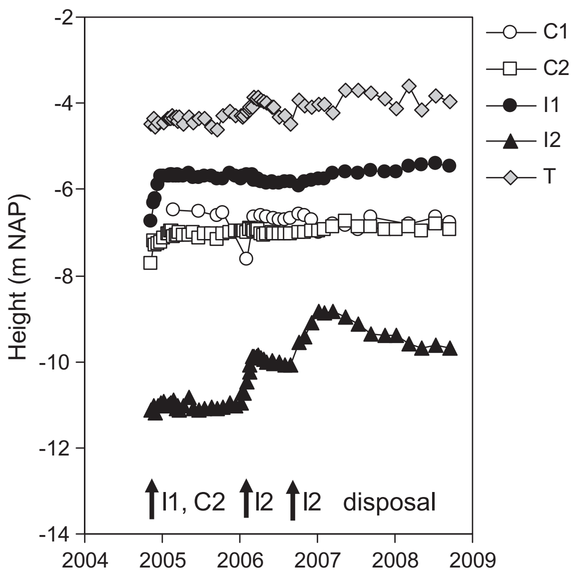

Fig. 2. Changes in elevation in the subtidal zones, based on a large number of points extracted from sequential multibeam surveys carried out by Eurosense. Arrows indicate the start of the disposal events.

fixed stations (T) in the minor flood channel, i.e., the subtidal transport zone between site I1 and the intertidal flat, were repeatedly sampled. In the intertidal zone, 40 fixed stations were sampled (Fig. 1). At each intertidal station, material from three cores (30 cm depth, 8 cm diameter each) was pooled for macrobenthic analysis; at the subtidal stations, three such cores were taken from a Reineck box-corer sample and pooled. Sediment was collected from the upper 3 cm of the surface for granulometric analysis.

The macrofaunal material >1 mm was fixed in formaldehyde. Animals were identified and counted at species level in the laboratory and density was expressed in individuals/m^2. The animals were dried at 80 °C for 2 days, then at 100 °C for 1 day and then ashed for 2 h at 580 °C to determine biomass of each species (ash-free dry weight, in mg/m^2). Species richness was defined as the total number of species in each sample.

The sediment samples were freeze-dried, and material <1 mm was analysed using a Malvern Mastersizer 2000, capable of detecting 0.02 μm to 1 mm grains, to derive values for median grain-size d50 (μm), mud (percentage particles <63 μm) and sand size fractions very fine sand (63–125 μm), fine sand (125–250 μm), medium sand (250–500 μm) and coarse sand (500–1000 μm).

Height (relative to m NAP, which is Dutch Ordnance Datum, approximately mean sea level) was extracted at the sample stations in the subtidal zone from shipborne multibeam surveys (Fig. 2) with an accuracy of the order of centimetres (Leys et al., 2006). In the intertidal zone, height was derived from annual airborne LIDAR surveys with a vertical accuracy of ca 0.05 m (van der Wal et al., 2008).

Apart from the intensive field campaigns, an extensive, but long-term time series of the bed characteristics of the intertidal zone was available (MOVE data-base, Rijkswaterstaat, 2006). Height (using sedimentation–erosion frames with mm accuracy) and estimated clay content of the surface were surveyed at ca 6 fixed intertidal stations along a transect (Fig. 1). Surveys in March, May, September and December in the period 1998–2007 were selected to guarantee a balanced dataset for the two variables of interest. As the Westerschelde sediments have a constant silt: clay ratio (Winterwerp and Van Kesteren, 2004), the estimates of clay content are linearly related to mud content (van der Wal et al., 2010).

2.2. BACI variance and trend analysis

The response to the first dredging disposal event (autumn 2004) in the subtidal zone was identified using a BACI (Before-After Control Impact) design, with I1 as the impact site and C1 as the control site. A two-way factorial ANOVA was carried out on the main effects Site and Time and their interaction term. A priori contrasts were applied to verify comparability between I1 and C1 before the impact (autumn 2004). Short-term (one year) effects of the first disposal were evaluated on the interaction term Site × Time by contrasting data before (autumn 2004) and after (spring and autumn 2005) the impact for I1 and C1, respectively. Long-term (four years) effects were evaluated by contrasting data before (autumn 2004) and after (spring 2005 until autumn 2008) the disturbance for I1 and C1.

To evaluate the effects of all perturbations and identify trends, a one-way ANOVA and subsequent posthoc HSD Tukey test was carried out for each subtidal site separately, with abiotic (sediment or height) or biotic (biomass, density or species richness) variables as dependent and Time as the categorical predictor. In the intertidal zone, changes in macrobenthos and environment were analysed with ANOVA on Year, Season, Year × Season and Station (with Station as random factor), and a posthoc HSD Tukey test.

To warrant homogeneity of variance, values for macrobenthic biomass and density were transformed following $\ln(x+1)$, and grain-size percentages following arcsin $\sqrt{(x/100)}$. For all ANOVA analyses, the level of significance α was taken at 0.05.

2.3. Multivariate analysis of the macrobenthos community

Patterns and trends in macrobenthic community were identified using the software package PRIMER (Clarke and Warwick, 1994). Multidimensional scaling (nMDS) plots were constructed based on Bray-Curtis similarity matrices (Clarke, 1993) from log-transformed biomass of the macrobenthic species. Samples that contained only one or no species were omitted from the analysis. Significance of differences both between sites and between periods was formally tested using an analysis of similarities (ANOSIM); this yielded an R-value (ranging from −1 to 1), denoting dissimilarity (with $R > 0.5$ indicating clear differences between groups), and a probability P (α again taken at 0.05). SIMPER analysis identified which species contributed most to the distinction of groups.

3. Results

3.1. Impact on the sediment characteristics and height in the subtidal zone

In autumn 2004, before the disposal of the sediment, the sediment characteristics in site I1 did not significantly differ from those in control site C1, except for a higher percentage of very fine sands in I1 (Table 1). The year after the impact, only the percentage of fines and medium sand diverged significantly from autumn 2004 between impact site I1 and control site C1. In the long-term, all sediment parameters (except mud percentage) diverged significantly (Table 1, Fig. 3).

Both before and directly after the disposal, the sediment at site I1 was mainly composed of fine and medium sized sand. Before disposal (autumn 2004), it contained 1.32 ± 0.54% mud, 1.69 ± 0.26% very fine sand, 49.98 ± 1.39% fine sand, 45.74 ± 1.50% medium sized sand and 1.28 ± 0.29% coarse sand, on average. After disposal (spring 2005), it contained 0.22 ± 0.15% mud, 1.90 ± 0.43% very fine sand, 52.03 ± 2.23% fine sand, 44.46 ± 2.16% medium sized and 1.40 ± 0.38% coarse sand. Median grain-size ($F_{8,171} = 2.89$, $P = 0.01$) and the amount of mud ($F_{8,171} = 2.07$,

Table 1
Comparison of impact site I1 and control site C1 before the impact (autumn 2004), and short-term (autumn 2004 versus spring/autumn 2005) and long-term (autumn 2004 versus spring/autumn 2005–2008) BACI effects based on ANOVA with a priori contrasts. Sediment fractions and macrobenthic biomass and density values were transformed prior to analysis (see text).

	A priori contrasts		
	Before: I1 versus C1	BACI: short-term effects	BACI: long-term effects
Sediment			
Mud	0.73	2.48	0.48
Very fines	4.68*	1.81	3.99*
Fines	0.40	3.94*	5.71*
Medium sand	2.32	4.12*	6.78**
Coarse sand	0.53	1.96	6.13*
d50	3.43	2.97	7.79***
Macrobenthic biomass			
Total	0.23	2.93	0.42
H. filiformis	0.03	1.08	5.47*
M. balthica	0.92	1.41	0.00
N. cirrosa	5.94*	0.52	0.00
B. pilosa	0.22	0.02	0.11
Macrobenthos density			
Total	0.21	3.20	0.42
Species richness	0.28	2.16	0.10

Numbers are $F_{1,342}$ values.
* Significance $P < 0.05$.
** Significance $P < 0.01$.
*** Significance $P < 0.001$.

$P = 0.04$), fine sand ($F_{8,171} = 3.68$, $P = 0.01$), medium sand ($F_{8,171} = 3.25$, $P = 0.00$) and coarse sand ($F_{8,171} = 3.00$, $P = 0.00$) varied significantly between campaigns in impact site I1, particularly due to fining of the sand fraction and a reduction in mud content in spring and autumn 2006, autumn 2007 and spring 2008 compared to autumn 2004. Height at the sample stations changed significantly in site I1 ($F_{8,171} = 2.39$, $P = 0.02$); the stations randomly selected in spring 2008 were shallower than those selected in autumn 2004, spring 2005 and spring 2007.

Sampling in I2 started in spring 2006, after the first phase of disposal operations at this site (cf. Fig. 2). At this time, sediment in I2 consisted of very fine sand (1.54 ± 0.40%), fine sand (46.50 ± 2.88%), medium sized sand (49.46 ± 2.75%) and coarse sand (2.50 ± 0.67%), but no mud. Median grain-size ($F_{5,114} = 4.07$, $P = 0.00$) and the amount of very fine ($F_{5,114} = 4.79$, $P = 0.00$), fine ($F_{5,114} = 4.53$, $P = 0.00$) and medium sand ($F_{5,114} = 5.30$, $P = 0.00$) varied between campaigns in site I2, generally with the coarsest sand in autumn 2008. Changes in elevation (cf Fig. 2) were not significant between the successive sets of random sample stations ($F_{5,114} = 1.81$, $P = 0.12$). In the transport zone T, neither granulometric parameters nor elevation changed significantly at the sample points (all $P > 0.05$). In control site C1, mud content ($F_{8,171} = 2.48$, $P = 0.02$) and the amount of very fine sand ($F_{8,171} = 2.50$, $P = 0.01$) varied significantly between campaigns.

3.2. Impact on macrofauna in the subtidal zone

Before the impact, subtidal zones C1 and I1 had similar macrofaunal biomass, density and species richness (Table 1). No significant effects of the impact were detected, neither on the short-term nor on the long-term (Table 1). An exception was the biomass of the polychaete *Heteromastus filiformis*, which showed lower values in I1 than in C1 on the long-term ($F_{1,342} = 5.47$, $P = 0.02$; Fig. 4).

One-way ANOVA for site I1 revealed variations in total macrobenthic biomass ($F_{8,171} = 2.07$, $P = 0.04$) and density ($F_{8,171} = 3.42$, $P = 0.00$), but a posthoc HSD Tukey test could not identify any trends. Species richness did not vary with time ($F_{8,171} = 1.42$, $P = 0.19$). Of the four most abundant species, the polychaetes *H. filiformis* ($F_{8,171} = 9.14$, $P = 0.00$) and *Nephtys cirrosa* ($F_{8,171} = 3.56$, $P = 0.00$) varied significantly between campaigns; biomass of *H. filiformis* was particularly high in 2004 and 2005, whereas biomass of *N. cirrosa* was low in spring 2006 and high in spring and autumn 2008. Total biomass, biomass of key species and species richness showed a synchronous development in site I1 and I2 (Fig. 4). However, in site I2 and T, the biotic parameters did not vary significantly between campaigns (all $P > 0.05$), with the exception of the biomass of *H. filiformis* in site I2 ($F_{5,114} = 15.08$, $P = 0.00$), which was high in spring 2008 in particular. Macrobenthic biomass ($F_{8,171} = 11.31$, $P = 0.00$), density ($F_{8,171} = 12.14$, $P = 0.00$) and species richness ($F_{8,171} = 4.32$, $P = 0.00$) varied significantly between campaigns in site C1, especially due to low values in autumn 2005 (Fig. 4). The biomass of *H. filiformis* ($F_{8,171} = 7.11$, $P = 0.00$) and the bivalve *Macoma balthica* also differed significantly between campaigns ($F_{8,171} = 4.340$, $P = 0.000$) in site C1.

Multivariate analysis revealed that site I1 and C1 had a different macrobenthic community before the impact ($R = 0.05$, $P = 0.03$); the biomass of *M. balthica*, *N. cirrosa* and *H. filiformis* explained ca 60% of the dissimilarity between the two sites. After the impact, the two sites still differed, both on the short-term ($R = 0.07$, $P = 0.04$) and long-term ($R = 0.13$, $P = 0.00$). However, no changes in macrobenthic community before and after the impact were detected in impact site I1, neither on the short-term ($R = -0.10$, $P = 1.00$) nor long-term ($R = -0.19$, $P = 1.00$). Likewise, no changes were detected in control site C1, neither on the short term ($R = -0.04$, $P = 0.84$) nor long-term ($R = -0.19$, $P = 1.00$).

Fig. 5 summarizes temporal trends in communities by expressing the dissimilarity between sites C1 and all other sites, and between I1 and all other sites, respectively. The dissimilarity between the macrobenthic community in site I1 and C1 increased

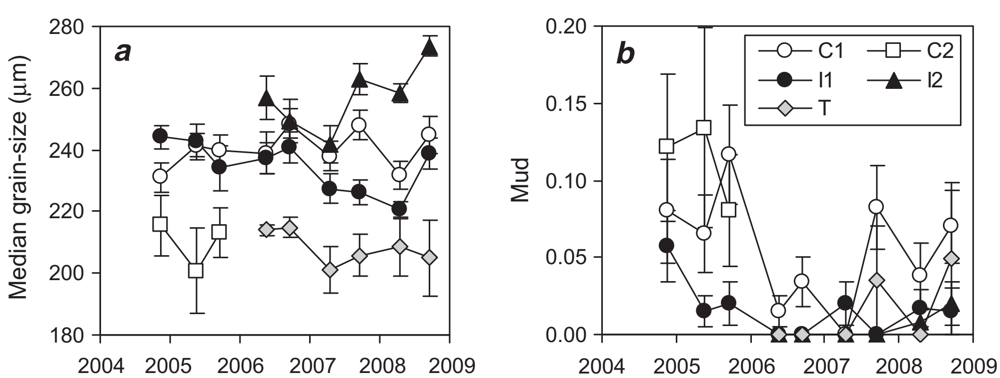

Fig. 3. Time-series of average (a) median grain-size and (b) mud content of the sediment in impact sites (I1, I2), transport site (T) and control sites (C1, C2) in the subtidal zone. Mud is expressed as $\arcsin\sqrt{(x/100)}$, where x is mud percentage. Error bars indicate SE.

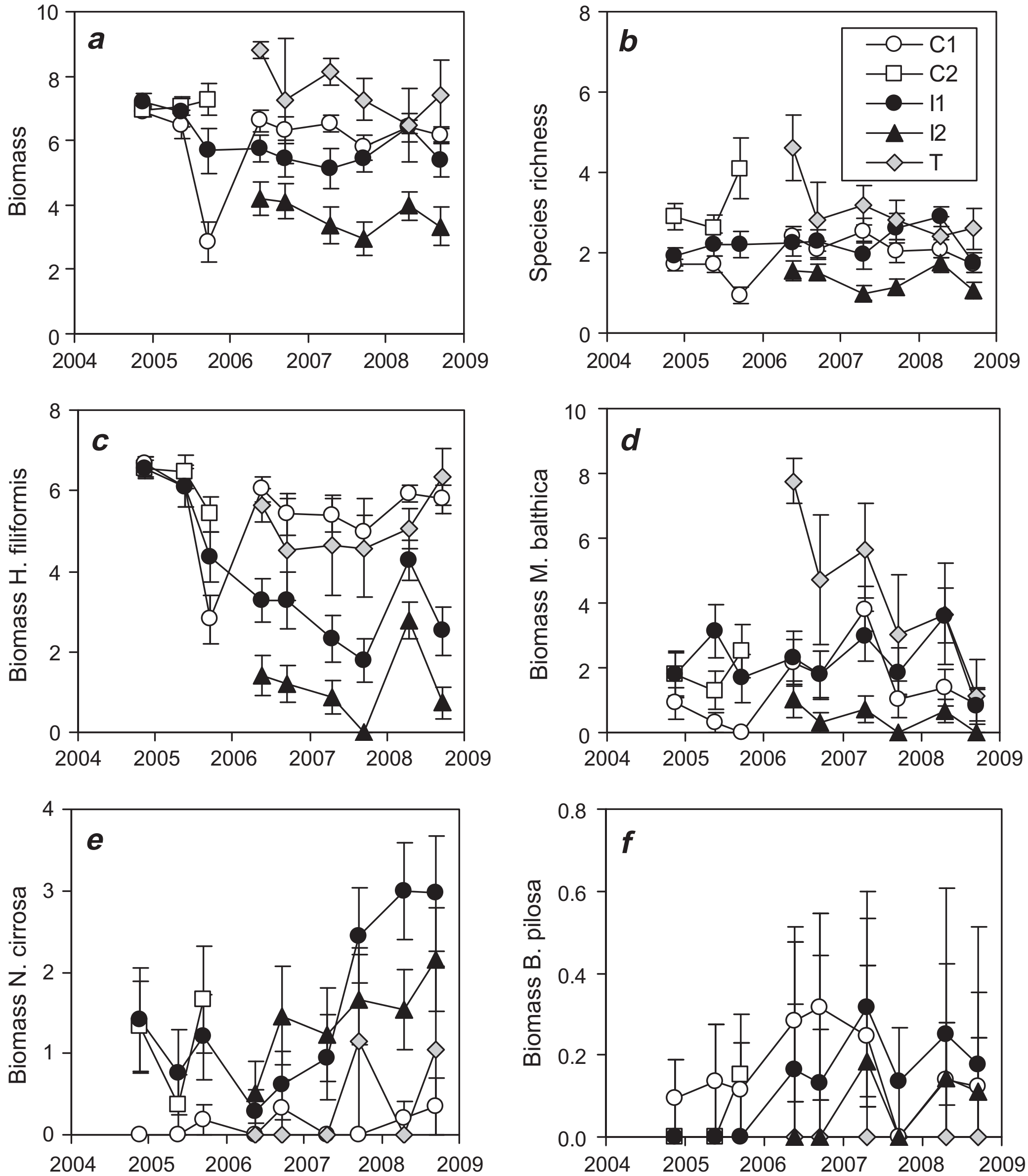

Fig. 4. Time-series of average (a) total macrobenthic biomass, (b) species richness and (c–f) biomass of abundant species in impact sites (I1, I2), transport site (T) and control sites (C1, C2) in the subtidal zone. Biomass is expressed as ln (x + 1), where x is biomass in mg/m^2. Error bars indicate SE.

after the impact, but a similar increase in dissimilarity between C1 and C2 was observed (Fig. 5). Moreover, site I1, I2 and T all showed a similar, synchronized development relative to site C1, suggesting that the changes in macrobenthic community in the subtidal zone near the Plaat van Walsoorden were not driven by site-specific impacts. The dissimilarity between the macrobenthic community in zone I1 and T was significant throughout the sampling period. Dissimilarity in macrobenthic community between I1 and I2 was significant only in autumn 2006 and spring 2008 (Fig. 5), and could be attributed mainly to fluctuations in the biomass of *H. filiformis*, *N. cirrosa* and *M. balthica.*

3.3. Impact on sediment characteristics and height in the intertidal zone

Intensive monitoring of the granulometry of the intertidal sediment in the period 2004–2009 (Fig 6) revealed a decrease in median grain-size ($F_{4,348}$ = 5.13, P = 0.00), an increase in mud content ($F_{4,349}$ = 4.78, P = 0.00) and a decrease in the fraction medium sand ($F_{4,349}$ = 3.28, P = 0.01) with year. Height from LIDAR surveys did not change significantly ($F_{4,156}$ = 0.68, P = 0.61). In contrast, height from long-term (1998–2007) in situ measurements varied significantly with year ($F_{8,196}$ = 11.49, P = 0.00), with alternating periods of vertical accretion and erosion (Fig. 7). The increase in clay content, starting well before the disturbance (Fig. 7), was also significant ($F_{8,190}$ = 12.63, P = 0.00).

3.4. Impact on macrofauna in the intertidal zone

In the intertidal zone, total biomass and density of macrobenthos did not change with year; only species richness increased significantly over the years ($F_{4,350}$ = 7.10, P = 0.00). However, a number of abundant species showed significant changes over the years, including fluctuations in *H. filiformis* ($F_{4,350}$ = 3.22, P = 0.01), an increase in the polychaete *Nereis diversicolor* ($F_{4,350}$ = 6.23, P = 0.00), and a decrease in the amphipod *Bathyporeia pilosa* ($F_{4,350}$ = 3.50, P = 0.01), but not *M. balthica* ($F_{4,350}$ = 0.30, P = 0.88) (Fig. 8).

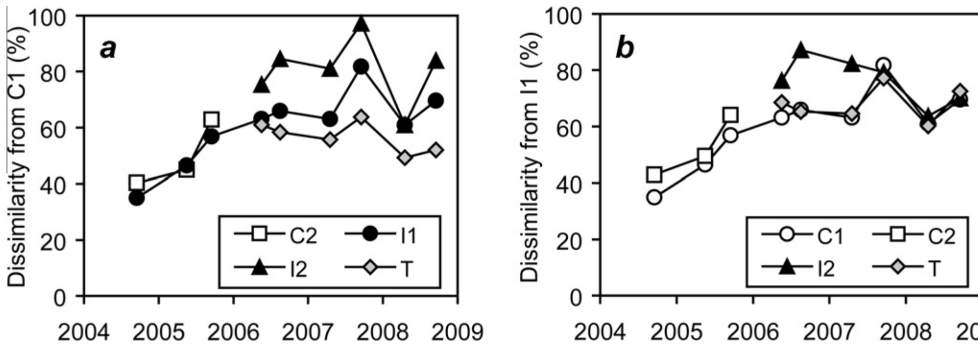

Fig. 5. Dissimilarity in macrobenthic communities in subtidal zones (a) relative to control site C1 and (b) relative to impact site I1.

ANOSIM revealed significant differences in macrobenthic communities between years, albeit with a large overlap in species ($R = 0.03$, $P = 0.00$). The change was gradual, as differences between subsequent years were not always significant ($P > 0.05$ for 2004–2005, 2005–2006 and 2007–2008). *N. diversicolor, M. balthica, H. filiformis, B. pilosa* and *Pygospio elegans* contributed most (ca 40%) to the changes in macrobenthic community.

3.5. Response of macrobenthos to environmental conditions

A multiple regression analysis was carried out to evaluate the response of macrobenthos to abiotic parameters (median grain-size, mud content and height) for all campaigns and all sites. Macrobenthic biomass depended significantly on median grain-size and height ($R^2 = 0.28$, $n = 930$, $F_{2,927} = 176.89$, $P = 0.00$), but median grain-size had most effect (partial regression coefficient $\beta = -0.29$ for median grain-size and $\beta = 0.27$ for height). Species richness depended significantly on all three parameters ($R^2 = 0.54$, $n = 930$, $F_{3,926} = 360.76$, $P = 0.00$), revealing most effect of height ($\beta = -0.26$ for median grain-size, $\beta = 0.10$ for mud content and $\beta = 0.46$ for height).

As a result, a clear gradient from ecologically poor (site I2, deepest and sandiest) to rich (W, shallowest and finest sediment) can be observed (Fig. 9). The graph also shows the potential sensitivity of macrobenthic biomass and species richness to changes in sediment characteristics and height due to disposal of dredged sediment. For example, the intertidal zone (W) is much richer in species than the very shallow subtidal zone (T), but they have comparable biomass. Thus, when enhancing habitats near Walsoorden, most gain in macrobenthic *diversity* is to be expected from an increase in height and reduction of sediment grain-size (cq. reduction in hydrodynamic energy) in the very shallow subtidal zone. In contrast, an increase in total macrofaunal *biomass* is not to be expected from such changes in this zone. The largest increase in total macrobenthic biomass is anticipated when the conditions in the deepest, most dynamic, zones, such as I2, can be ameliorated.

4. Discussion and conclusions

When the thickness of the disposal is of the order of metres rather than several decimetres, the macrobenthos community cannot recover by vertical migration of buried individuals (Maurer et al., 1986; Essink, 1999; Wilber et al., 2007). Instead, colonization mainly occurs via immigration of adults and juveniles from nearby undisturbed or less disturbed areas and via larval recruitment (Günther, 1992). Recolonization following defaunation by disturbance is typified by a rapid increase in abundance of opportunistic species, including small polychaetes (such as *H. filiformis*) and motile crustaceans (such as *B. pilosa*), slowly shifting to a richer macrobenthic community with a greater proportion of longer-lived, slower-growing ‘equilibrium’ species (such as the mobile polychaete *N. cirrosa*) (McCall, 1977; Pearson and Rosenberg, 1978; Van Dolah et al., 1984; Rhoads and Germano, 1986; Harvey et al., 1998; Newell et al., 1998). In general, little impact has been reported when native and disposed sediment are similar, especially in uncontaminated sand with negligible organic enrichment (Smith and Rule, 2001; Bolam et al., 2004; Simonini et al., 2005; Wilber et al., 2007). Macrobenthic communities may be most resilient (recovering within months rather than years) in ecosystems where the magnitude and frequency of natural perturbations are high (Van der Veer et al., 1985; Flemer et al., 1997; Bolam and Rees, 2003; Fredette and French, 2004; Simonini et al., 2005).

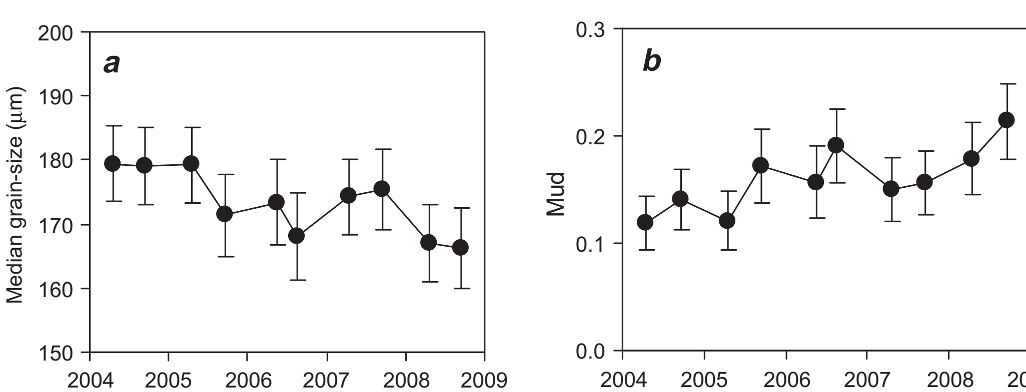

Fig. 6. Time-series of average of (a) median grain-size and (b) mud content of the sediment of the intertidal zone (site W). Mud is expressed as arcsin$\sqrt{(x/100)}$, where x is mud percentage. Error bars indicate SE.

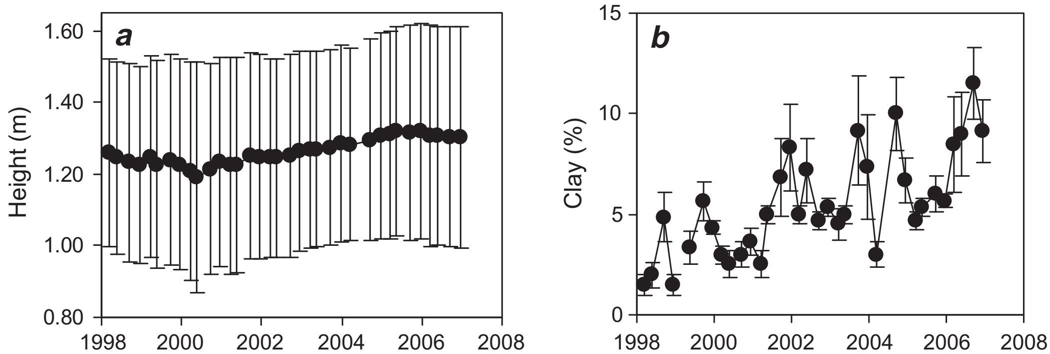

Fig. 7. Time-series of average (a) height and (b) clay percentage of the sediment of the intertidal zone (site W), based on campaigns in March, September, May and December of each year. Error bars indicate SE.

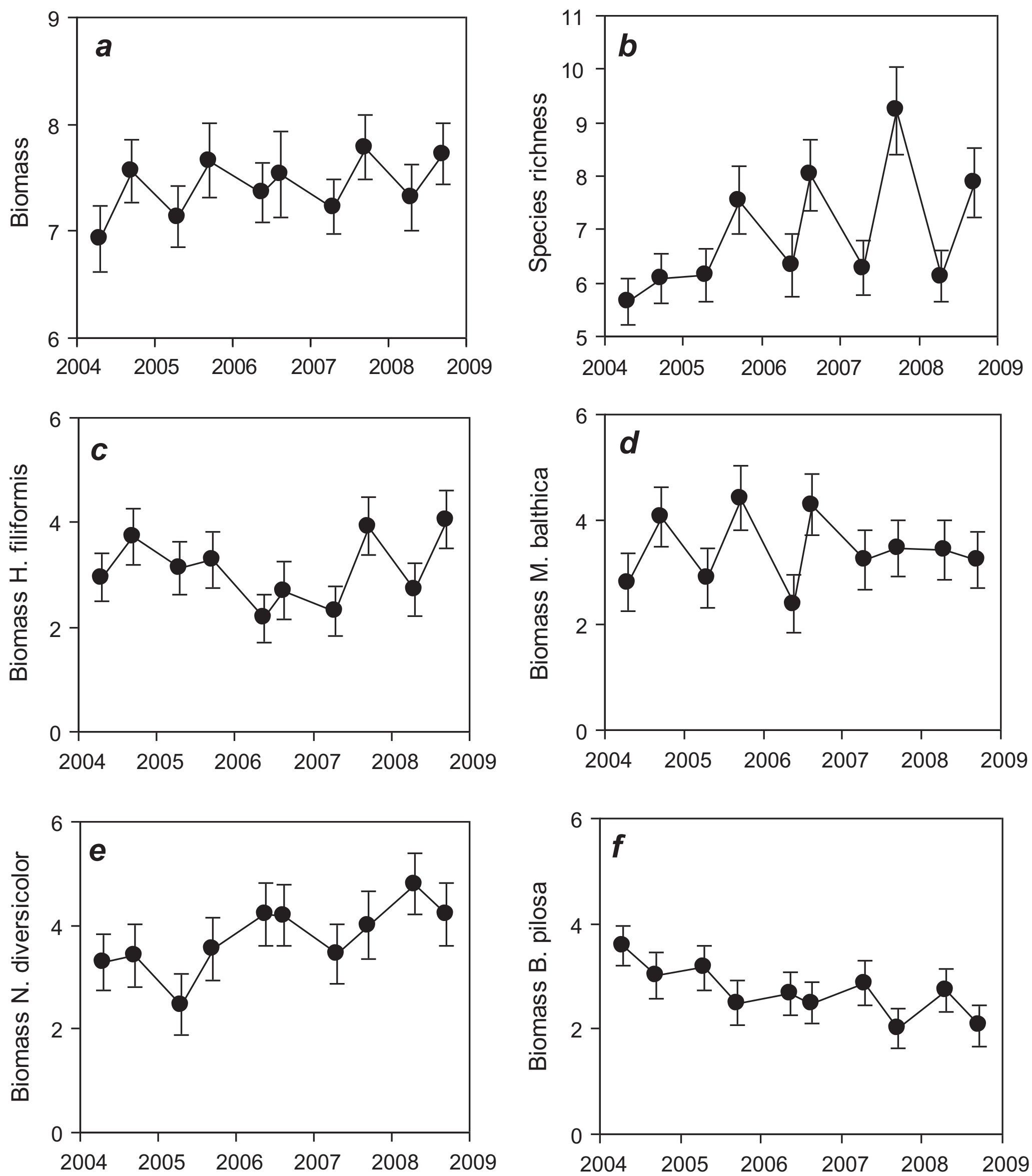

Fig. 8. Time-series of average (a) total macrobenthic biomass, (b) species richness and (c–f) biomass of abundant species in the intertidal zone (site W). Biomass is expressed as $\ln(x+1)$, with x is biomass in mg/m^2. Error bars indicate SE.

Nevertheless, previous experiences with beneficial use schemes for habitat enhancement have shown that macrofaunal communities do not always fully recover or compare to those in nearby reference situations (Ray, 2000; Bolam et al., 2006b). In addition, there

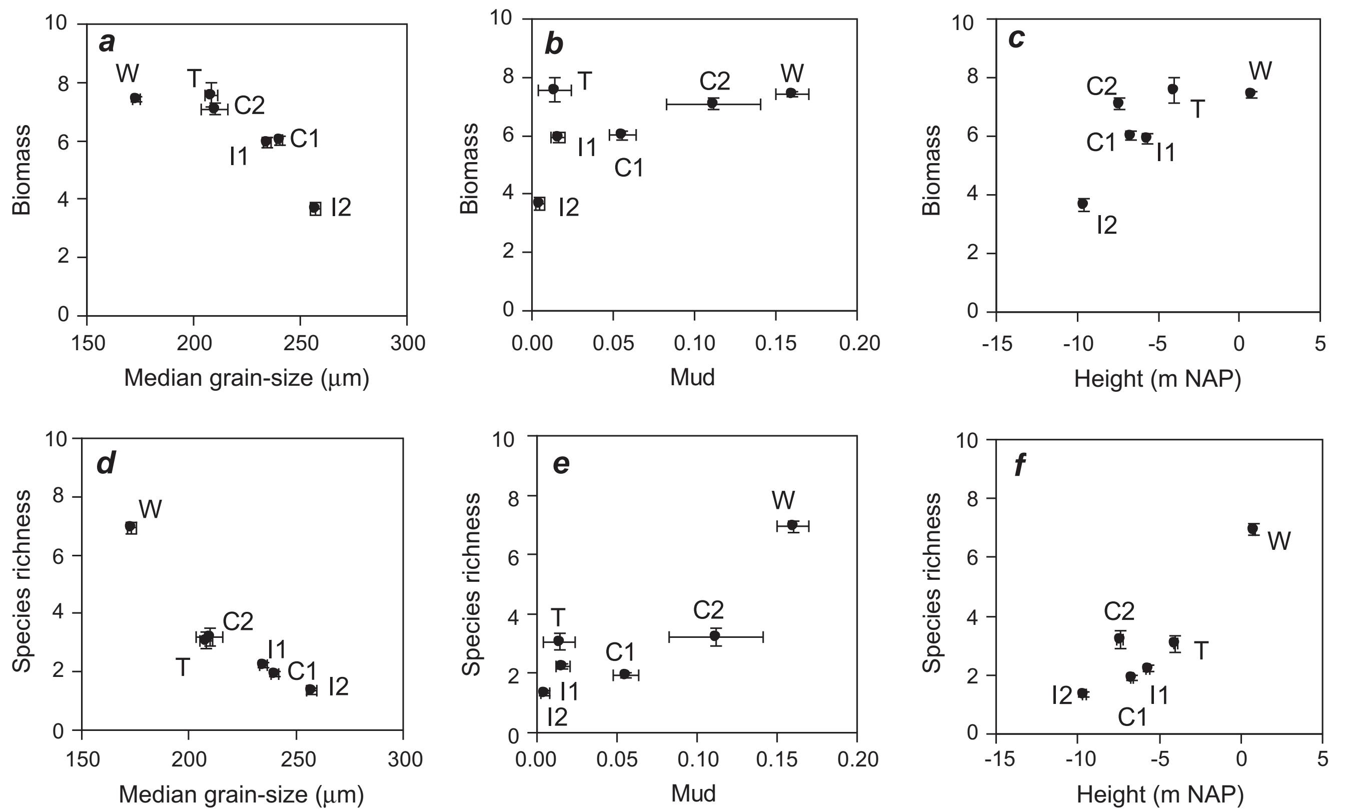

Fig. 9. Macrobenthic biomass ln (x + 1), with x is total biomass in mg/m^2, and species richness as a function of median grain-size of the sediment, mud content arcsin($\sqrt{x}$/100), where x is mud percentage, and height. Labels refer to sites (Fig. 1). Error bars indicate SE.

is still little experience in changing environmental conditions by disposing dredged material in such a way that shallow subtidal and intertidal habitats are *enhanced* for benthic macrofauna.

Our BACI and multivariate analyses did not reveal significant differences in macrobenthic biomass, density, species richness and composition before and within a year after the 2004 disposal experiment near the Plaat van Walsoorden, suggesting that the macrobenthic community had recovered within a year. The subtidal macrobenthic community in our study site was dominated by the opportunistic species *H. filiformis*, with a very limited abundance (i.e., a few individuals in part of the samples) of especially *M. balthica, N. cirrosa* and *B. pilosa.* We did not observe significant changes in biomass/density or changes in the length distribution of *M. balthica* (on average ca 1 cm), suggesting that active or passive adult migration rather than larval recruitment was the dominant mode for recovery. Individuals may have immigrated by crawling and swimming (notably *B. pilosa* and *N. cirrosa*) or individuals (and possibly *H. filiformis* egg capsules) may have been washed in from elsewhere. At some sample points the thickness of the disposal was less than a few decimetres, so that adults could have survived by vertical migration through burrowing (cf. Essink, 1999). Our findings concur with studies that show a fast recovery of most or all ambient species (e.g., Zajac and Whitlatch, 1982; Dauer, 1984; Bolam et al., 2004; Cruz-Motta and Collins, 2004; Simonini et al., 2005; Powilleit et al., 2006), rather than a succession (McCall, 1977; Pearson and Rosenberg, 1978; Harvey et al., 1998). The synchrony in changes in benthic community at the 2004 and 2006 impact sites also indicate that long-term developments were not a result of succession following sediment disposal.

Indirect effects of the disposals on the macrobenthic community in the subtidal zone were expected as a result of changes in elevation, sediment characteristics, current velocities and sediment dynamics (e.g., Hall, 1994; Flemer et al., 1997; Miller et al., 2002; Van Colen et al., 2010). In the Westerschelde, highest macrobenthic biomass and species richness occur in low dynamic, very shallow subtidal and intertidal waters (Ysebaert et al., 2002). Indeed, macrobenthic biomass and species richness were positively correlated with height, and negatively correlated with sediment grain-size in our study. The multibeam surveys were accurate enough to detect changes in height not only due to sediment disposal, but also due to subsequent sediment transport: zone T started infilling (at a rate of ca 20 cm/year) from Sep 2005 onwards (due to transport from I1, and later I2), zone I1 changed from a site of net erosion to a site of net accretion from Sep 2006 onwards (due to transport from I2 after the 2006 disposal) and zone I2 continued to erode after sediment disposal (Fig. 2), although changes in height were not always significant between the successive sets of (random) sample points. Both ambient and dredged sediment were mainly composed of fine and medium sized sand. However, a gradual decrease in median grain-size of the sediment (attributed to a fining of the sand fraction) was found in impact site I1. This decrease may be related to a modified current regime in the impact site as a result of the disposal (in accordance with the accretional trend in I1 since Sep 2006, see Fig. 2). Yet, no unidirectional changes in the macrobenthic community (biomass, species richness and composition) were detected in the subtidal zone on a time-scale of years following the different disposal events, except for the decrease in biomass of *H. filiformis* in the first impact zone.

Far-field changes in biomass and community of macrobenthos in the intertidal zone were anticipated via potential modification of the sedimentation/erosion rate of the intertidal area or a change in sediment grain-size of material that was being deposited on the intertidal flat. Some ecological changes were detected on the intertidal flat, but they can not be attributed unequivocally to the disposal experiment. The intertidal flat had been subject to a gradual fining of the sediment. This complies with the increase in species typical for fine sediment (e.g., *N. diversicolor*) and decrease in species typical for coarser sediment (e.g., *B. pilosa*), resulting in an overall increase in species richness, as detected in the period 2004–2009. The increase in mud content of the sediment, occurring especially on the higher parts of the intertidal flat (cf. van der Wal et al., 2008), started well before the disposal experiments. Thus, it seems unlikely to have been largely impacted by transport of mud from the spoil during and after disposal, or from any reduction in hydrodynamics caused by the disposals.

The amount and type of disposed material, as detected shortly after disposal, would have been sufficient to cover the entire intertidal flat (i.e., an area of ca 4 km^2 above -1 m NAP) with a layer of ca 0.45 m of sand, and only a negligible amount of mud (<1 mm). However, large-scale deviations from long-term sedimentation/erosion rates of the intertidal zone were not observed, despite sufficient accuracy of the methods to detect vertical changes in elevation of the order of millimetres (sedimentation–erosion plots) to centimetres (LIDAR surveys). Rather than being spread onto the intertidal flat, bathymetric and LIDAR surveys demonstrate that part of the disposed sediment is still present at the disposal sites after five years (cf. Fig. 2), while part of the disposed sediment has been transported in the direction of the intertidal zone and has amalgamated with the intertidal zone, as planned (Roose et al., 2008; Vos et al., 2009). Thus, the very shallow subtidal zone and intertidal zone have increased in area (counterbalancing the trend of erosion) at the expense of the deeper subtidal zone. In this way, the multiple channel system of the estuary, with ecologically productive intertidal areas, is sustained. Thus, by at least maintaining these productive areas, the strategy is favourable compared to the traditional dredging practice from an ecological perspective.

The study did not reveal significant changes in biomass, density, species richness and composition of the macrobenthic community that could directly be attributed to the disposal experiment. On the positive side, this implies that we did not detect an adverse impact of the disposal on the macrobenthos. On the negative side, the macrobenthic community did not improve either as a result of the dredging experiment, neither in the subtidal zone nor in the intertidal zone. Thus, no new beneficial habitat was created for benthic macrofauna. This may particularly be due to the fact that the impact sites were in a dynamic environment (megaripple areas with mobile clean sand and limited food availability), with an associated poor macrofaunal community with low biomass and only a few common species adapted to such conditions (cf. Ysebaert et al., 2003). Both the very shallow subtidal zone and the created intertidal zone were still highly dynamic a few years after the disposals. The disposal may not have altered abiotic conditions sufficiently for the habitat to change due to the limited scale of the experiments. Alternatively, it can not be excluded that the slight amelioration expected based on general trends was counteracted by disturbance, either natural or as a result of the morphological unbalance by the disposal.

Environmental conditions and macrobenthic communities are inherently variable in estuaries, both in space and time (Underwood, 1992; Hewitt et al., 2001), and the impact and re-adjustment time of disturbances may vary accordingly, stressing the need for case by case monitoring of beneficial use schemes. Potential for enhancing shallow subtidal and intertidal areas for benthic macrofauna may be greatest in areas that can be transformed from high dynamic to low dynamic, even though such an enhanced community may take longer to establish. In addition, while recovery of the macrobenthic community after disposal could be fast, we have shown that indirect effects as a result of sediment transport may take years to manifest, stressing the need for prolonged monitoring.

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