

Chapter 3: The monitoring of ‘ecological’ beach nourishment impacts on macrobenthos, within a Special Area of Conservation (SAC) along the Belgian coast

Sarah Vanden Eede, Dries Bonte, Magda Vincx (in prep.) The monitoring of ‘ecological’ beach nourishment impacts on macrobenthos, within a Special Area of Conservation (SAC) along the Belgian coast.

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Abstract

The expected increase in intensity and frequency of storm events combined with the undeniable rise in sea water level, both connected to climate change, poses a threat to coastal low lands. As it safeguards the natural dynamics of the coast, beach nourishment has rapidly become the preferred protective and restoring measure in coastal zones worldwide. We tested whether optimizing the main technical aspects of a nourishment according to ecological recommendations, e.g. a gentle sloped beach with nourished sand resembling the original sediment very closely (average median grain size of 224 μm), leads to an ‘ecological’ nourishment with minor to no impact effects on the macrobenthos.

Monitoring of nourishment impact effects on the macrobenthic community structure can be regarded as a ‘natural experiment’ and is based on a BACI (Before/After, Control/Impact) design. A putatively impacted area and a matching control site are monitored in a time series (2000 – 2012) straddling the impact event (2009). Impact effects should then show up as interactions between the temporal and spatial factors. A wider, higher and flatter intertidal beach with coarser sediment (from $215.89 \pm 3.63 \mu\text{m}$ in 2008 to $280.23 \pm 8.94 \mu\text{m}$ in spring 2010) was created and no return to the pre-nourishment conditions in sedimentology was visible three years after nourishment. The sediment grain size distribution had changed as well, showing slow recovery in the three post-nourishment years. The analysis of the macrobenthos community structure showed that the nourishment under ecological optimal conditions on the beach of Lombardsijde yielded no significant effects on both the intertidal and shallow subtidal beach ecosystem 6 months after the nourishment. Within this time frame, the macrobenthos community had seemingly recovered from the impact of the ecological nourishment. Ecological nourishment thus proves to be the least ecologically damaging way of combating erosion, compared to all other coastal engineering activities.

Keywords: beach nourishment, macrobenthos, monitoring

1. Introduction

During the last decade, climate change has become a much debated topic. The expected increase in intensity and frequency of storm events combined with the undeniable rise in sea water level, both connected to climate change, poses a threat to coastal low lands. Beach erosion is likely to accelerate, driven by predicted consequences of climate change and coastal development (Brown & McLachlan 2002; Feagin et al. 2005; Slott et al. 2006; Schlacher et al. 2012). This diminishes the natural buffer function of beaches, leaving the hinterland less protected from scour, inundation and wave erosion (Young & Bryant 1992; Defeo et al. 2009).

Traditionally, coastal defence focused on beach stabilization by building hard structures of stone, concrete, wood, steel or geotextiles such as groynes, breakwaters, seawalls and dykes (Charlier et al. 2005). However, these structures alter the natural hydrodynamic system of waves and currents, thereby affecting sand transport rates, which in turn control the erosion dynamics of beaches (Defeo et al. 2009). There has been a gradual change from hard to soft coastal defence techniques, like beach nourishment. Beach nourishment is in essence the process of mechanically or hydraulically placing an amount of sand directly on an eroding shore to restore or form, and subsequently maintain, an adequate protective or desired recreational beach. As it safeguards the natural dynamics of the coast, beach nourishment has rapidly become a widely applied protective and restoring measure in coastal zones worldwide (Finkl & Walker 2002; Greene 2002; Hamm et al. 2002; Hanson et al. 2002; Speybroeck et al. 2006a; Cooke et al. 2012) for short-term urgencies (i.e. storm-induced erosion) as well as long-term issues (i.e. structural erosion and relative sea-level rise).

Even though beach nourishment is considered as the more ecologically sound option, an impact of such magnitude can be expected to affect the entire beach ecosystem (Speybroeck et al. 2008a). Coastal engineers prefer coarser grain sizes, obtained from marine extraction zones or nearby channel dredging, to produce a steeper, more stable and longer lived nourishment (Finkl & Walker 2002). Due to the forces of winds and waves, nourished sand will move between the dunes, the intertidal beach, the shallow subtidal and offshore zones until a stable equilibrium profile is achieved (Zeidler 1987), thus affecting the entire beach ecosystem and not only the nourished site. Unfortunately, there are still many uncertainties concerning the effects of beach nourishment. Natural perturbances, like storms, natural variability and spatial patchiness of organisms on sandy beaches may even obscure any but the largest effects. Long term data and large-scale datasets, field experiments and monitoring programmes can reveal insights into natural dynamics, or at least dynamics when unimpacted (Stauble & Nelson 1985; Defeo et al. 2009; Gray & Elliott 2009). Most studies documenting qualitative changes in the beach community have attributed these differences to natural variation (Culter & Mahadevan 1982; Saloman et al. 1982; Turbeville & Marsh 1982; Grober 1992), urging to interpret impact study results with caution.

The biotic composition of macrobenthos is an important indicator of the environmental quality and health of the benthic ecosystem (Goldberg 1988), playing a key role in the wider beach ecosystem and food web. Macrobenthos is generally defined as the organisms measuring over 1 mm long and living buried in the sediment. This group of bivalves, polychaetes, crustaceans and echinoderms feed primarily on faunal detritus and to a lesser extent on algal benthos and detritus (Sundbäck & Persson 1981; Josefson et al. 2002) and they are a major food source for birds and epibenthos. Owing to the highly dynamic nature of their environment, mainly determined by waves, tides and winds, macrobenthos species have a high, but not limitless tolerance towards several forms of environmental stress (Jaramillo & McLachlan 1993; Moffett et al. 1998). Therefore, according to many authors, nourishment should cause only minor and/or temporary damage to the ecosystem (Löffler & Coosen 1995; Miller et al. 2002). However, questions of recovery are less studied than those of short-term impacts (Schlacher et al. 2012). The currently accepted time frame for recovery of a single year may thus not reflect the return to a climax stage or the return to pre-nourishment conditions (Jones et al. 2008; Leewis et al. 2012; Schlacher et al. 2012).

Optimizing the technical aspects of future nourishment projects is indispensable to maintain an ecologically healthy beach ecosystem. For ecologically good practice of beach nourishment, it is advised (1) to choose nourishment sands with a sediment composition comparable to that of the natural sediment, (2) to avoid short-term compaction by ploughing immediately after construction, (3) to execute the nourishment in a period of low beach use by birds and other mobile organisms, (4) to choose a number of smaller projects rather than a single large nourishment project and (5) to select the nourishment technique with respect to the local natural values. The preferred time of nourishment entirely depends on the nature and location of the beach and the species inhabiting or exploiting it, but in temperate regions, the ideal period is the winter season (Speybroeck et al. 2006a).

This study aims at unravelling the *in situ* ecological effects on the soft-sediment macrobenthos of the beach nourishment on the Belgian beach of Lombardsijde. This nourishment was performed from March until September 2009, under optimal ecological conditions, e.g. phased nourishment project with nourished sand closely matching the original sediment and only moderate beach profile changes. The soft substrates of this beach have been extensively monitored prior to (Welvaert 2005; Van Ginderdeuren et al. 2007; Vanden Eede et al. 2008), during (Vanden Eede & Vincx 2010) and after the nourishment (Vanden Eede et al. 2013). To account for natural variation, the parallel monitoring of a control site, e.g. Nieuwpoort, was included in the research and several seasonal studies were performed between 2004 and 2012. This paper has three aims: (1) to quantify *in situ* ecological nourishment effects on macrobenthos, (2) to determine whether or not the macrobenthos has recovered within a period of 3 years and (3) to ascertain and confirm that ecological nourishment is the way forward within the Belgian coastal defence policy.

2. Material and methods

2.1 Study Area

According to the subdivision of the shallow Belgian coastal zone, Belgian beaches have a clearly defined intertidal and shallow subtidal zone (figure 1). The landward boundary follows the high water mark obtained by LIDAR observations of the Belgian coast in 2011 (data provided by the Agency for Maritime and Coastal Services: Coastal division – MDK) and the seaward boundary for the shallow subtidal foreshore is the 1 nautical mile from the zero depth (0 m) bathymetric line (figure 1).

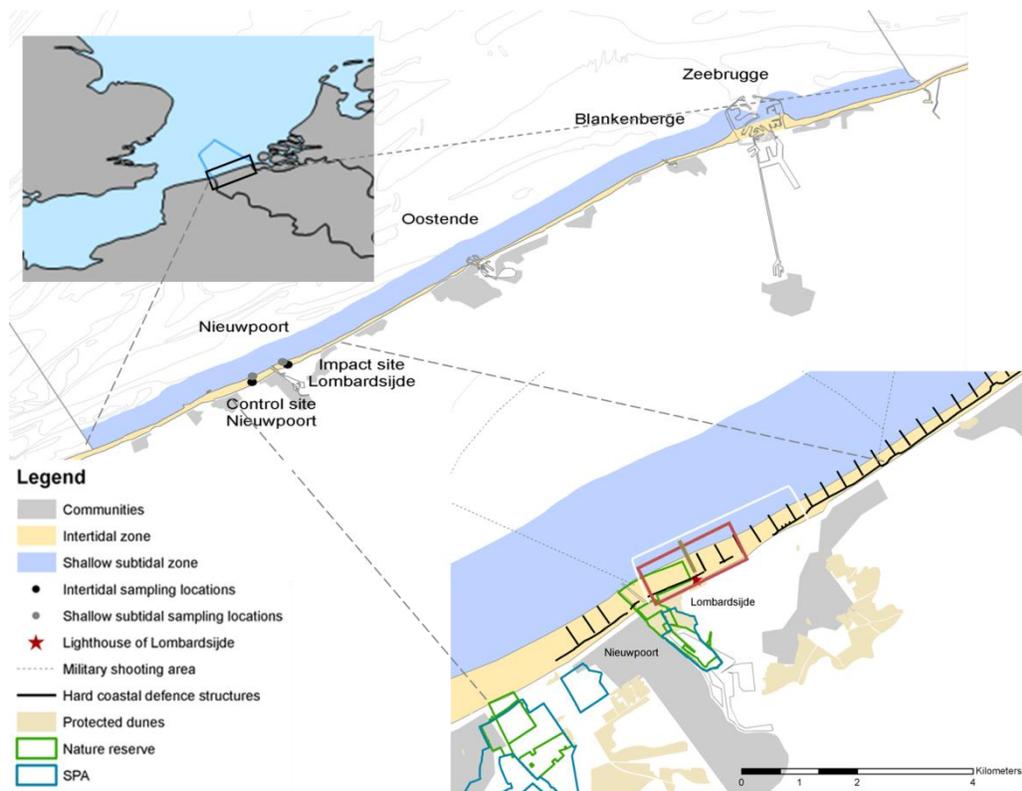


Figure 1: The Belgian coastal zone, with a distinction between the intertidal (light brown) and shallow subtidal zone (blue) at impact site Lombardsijde and control site Nieuwpoort, showing the location of the nourishment site (red box) within the nature reserve ‘Ijzermunding’ (green shape) and the Special Area of Conservation (SAC, white borders)

The beach of Lombardsijde is situated at the eastern side of the fairway to Nieuwpoort (figure 1). The whole beach is part of a Special Area of Conservation (SAC). The western part falls within the nature reserve ‘Ijzermunding’ and is located in front of a Special Protection Area (SPA) and protected dunes while the eastern part falls within a military domain. The intertidal zone of Lombardsijde beach was nourished from March until September 2009, under optimal conditions. Approximately 650 000 m³ of

sand, dredged from the fairway to Oostende, was deposited on top of the beach over a distance of around 1 200 m (15 m beach width at + 5.5 m followed by a slope of 1/70). The accumulated sand was moved with a bulldozer to profile the beach after deposition works finished. The sand resembled the original sediment, with a grain size between 200 and 250 µm. Hence, 250 m wide wet beach and 50 m wide dry beach were created. The aim of the Flemish government is to enclose part of Lombardsijde beach as a nature reserve for both birds and seals. The control site of any impact site should be located out of reach, preferably updrift, of the actual impact, e.g. the beach nourishment. The climate of flood currents, waves and winds at the Belgian coast is dominated by a southwest to northeast direction, narrowing the selection to all beaches located southwest from Lombardsijde. The beach of Nieuwpoort was chosen as a control site for Lombardsijde. It is situated in front of the municipality of Nieuwpoort at the western side of the fairway to Nieuwpoort (figure 1).

2.2 Data

Since 1997, the Marine Biology Research Group of Ghent University coordinated numerous macrobenthic studies along the Belgian coastline. The integrated database for the impact site Lombardsijde and control site Nieuwpoort, comprises all available relevant data gathered during the period 2008 – 2012 in the intertidal and shallow subtidal zones (table 1). Sampling was conducted once before, two times during and six times after nourishment (3 years, spring and autumn season).

Table 1: Sampling locations used for the integrated macrobenthic database. Only data collected in intertidal and shallow subtidal zones of the Belgian coast (key: S=spring (grey zones); A=autumn; a (Vanden Eede et al. 2008); b (Vanden Eede & Vincx 2010); c (Vanden Eede & Vincx 2011b); d (Vanden Eede et al. 2013); red = nourishment phase)

INTERTIDAL ZONE	2008	2009	2009	2010	2010	2011	2011	2012	2012
	S	S	A	S	A	S	A	S	A
Nourishment time	t0	t	t	t1	t2	t3	t4	t5	t6
Study	a	b	b	c	c	d	d	d	d
Nieuwpoort (control)	26	26	26	15	15	15	15	15	15
Lombardsijde (impact)	25	39	26	15	15	15	15	15	15
Total amount of samples	51	65	52	30	30	30	30	30	30

SHALLOW SUBTIDAL ZONE	2008	2009	2009	2010	2010	2011	2011	2012	2012
	S	S	A	S	A	S	A	S	A
Nourishment time	t0	t	t	t1	t2	t3	t4	t5	t6
Study	a	b	b	c	c	d	d	d	d
Nieuwpoort (control)	25	25	25	15	15	15	15	15	15
Lombardsijde (impact)	25	25	24	15	15	15	15	15	15
Total amount of samples	50	50	49	30	30	30	30	30	30

2.3 Sampling method

Intertidal sampling always started at high tide and followed the receding water down the beach, as the organisms tend to cluster here very close to the water line, ending at low tide. Samples were taken by excavating a quadrat frame (surface area, 0.1026 m²) to a depth of 0.15 m. The samples were immediately sieved (\emptyset 1 mm) and preserved in an 8 % formaldehyde-seawater solution. Next to each biotic sample, one core sample (\emptyset 3.6 cm) for sediment analysis was collected. At every sampling location, a beach profile (Vanden Eede & Vincx 2011b) and the geographic position (geographic wgs84) were noted. As we always sample at the water line, we can deduce the real elevation of the sample locations to the water surface (MLW, mean low water level) using the M2 reduction model (Van Cauwenberghe et al. 1993).

The shallow subtidal zone was sampled with a small vessel at high tide. At every sample location, a Van Veen grab was lowered to take a sediment sample (0.1026 m²). Simultaneously, the coordinates (UTMwgs84), time and depth of each sample were determined. The depth was afterwards corrected (compared to MLW) using the M2 reduction model. Before opening the Van Veen grab, a core sample (\emptyset 3.6 cm) for sediment analysis was taken. The samples were immediately sieved (\emptyset 1 mm) and preserved in an 8 % formaldehyde-seawater solution.

2.4 Lab procedures

The sieved samples for faunal analysis were stained with Rose Bengal and elutriated ten times to collect the macrobenthos. The remaining material was examined to collect the larger organisms that were too heavy to be floated off by elutriation. All macrobenthic organisms, except for Oligochaeta, Actiniaria and Nematoda, were identified to species level, where possible, and counted. Faunal abundance was extrapolated to the number of individuals per m². Biomass (gram Ash Free Dry Weight or g AFDW) estimates were obtained by loss of mass on ignition (480 °C for 2 h) of oven-dried samples (110 °C for 24 h) (Van Ginderdeuren et al. 2007; Vanden Eede et al. 2008; Vanden Eede & Vincx 2010, 2011b; Vanden Eede et al. 2013). The biomass was calculated on higher taxon level except for the dominant species, being *Bathyporeia pilosa* and *Bathyporeia sarsi*, *Ensis* juveniles, *Eurydice* species, *Lanice conchilega*, *Macoma balthica* juveniles, *Nephtys cirrosa*, *Owenia fusiformis*, *Scolelepis squamata* and *Spisula subtruncata*.

After drying the sediment samples, the grain size distribution was determined with a Malvern Mastersizer 2000G laser with diffraction module (measuring range: 0.02 – 2000 μ m). In this paper, sediment was characterized by median grain size (μ m) and silt fraction (< 63 μ m). The percentage of shell fragments was determined by means of the percent volume of sediment remaining on the 1 mm sieve

(carbonate content). The values for Total Organic Carbon (TOC) were calculated with an automatic element analyzer 1500 Carlo Elba. The percentage of Total Organic Matter (TOM) was obtained by loss of mass on ignition, using the biomass analysis strategy.

2.5 Data analysis

In total, 348 intertidal and 329 shallow subtidal samples were gathered between 2008 and 2012. 171 species were identified from which 81 species were taken into account for further macrobenthic analysis (50 species in the intertidal zone and 80 species in the shallow subtidal zone). The species excluded from the analyses clearly attributed to other benthic groups, e.g. meiobenthos, hyperbenthos or epibenthos, and/or were gathered in a non-representative way, e.g. a frequency of occurrence of less than 0.5 % and a maximum of 5 individuals per sample (Appendices – Chapter 3 – table A).

Monitoring of the human-induced impact effects on the macrobenthic community structure can be regarded as a “natural experiment”. It is however still an observational study and not an experiment, in the strict statistical sense of randomly allocating treatments to experimental units. This study is based on a BACI (Before/After, Control/Impact) design (Underwood 1991, 1992; Green 1993; Underwood 1994; Smith 2002). A putatively impacted area and a matching control site are monitored in a time series straddling the impact event. Impact effects should then show up as interactions between the temporal and spatial factors.

All tables and basic graphs were made in Microsoft Excel 2010. The sampling location map was created with ESRI ArcMap Version 9.3. Wilcoxon signed-rank tests were performed in R 2.14.2 to observe statistical differences between the mean values of the abiotic and biotic variables of impact and control site.

Multivariate analyses in Primer v6 were used to detect patterns in the distribution of macrobenthic assemblages, without transforming data. Multidimensional scaling was based on Bray-Curtis similarity, a statistic used to quantify the compositional dissimilarity (abundance data per species) between different sampling locations. Samples within a cluster are more similar than samples of different clusters (Clarke et al. 2008).

Regression analysis was used to estimate the relationship between median grain size (μm ; dependent variable) and three independent variables, e.g. Elevation (m versus MLW), Time ($t_0, t_1, t_2, t_3, t_4, t_5, t_6$) and Treatment (impact or control site). The term ‘Treatment’ was used in the analyses rather than ‘Location’ since the key comparison is between the experimental treatment (i.e. nourishment) and the control. This is the full model: Grain size = Elevation + Treatment + Time + Elevation*Treatment +

Elevation*Time + Treatment*Time + Elevation* Treatment*Time. For each pairwise comparison (t0-t; t0-t1;t0-t2 etc.), a significant interaction term Elevation*Treatment shows an average impact of the nourishment on the sediment grain size distribution. The relation between grain size and elevation has changed between the two treatments over time when the p-value of the interaction term Elevation*Treatment*Time is significant ($p < 0.05$). This means the sediment grain size distribution changed between the two beaches over time, and this could be due to changes at different times after the nourishment. If these changes become less clear over time, the sediment grain size distribution might have evolved back to original levels, e.g. recovery of the sediment grain size distribution has taken place.

The variance in the macrobenthic community structure was analysed with the adonis function of the Vegan package in R 2.14.2. This technique fits permutative linear models (e.g., factors, polynomial regression) to Bray-Curtis distance matrices and allows for nonparametric analysis of variance using these distance matrices. We follow the same approach as described by Schlacher et al. (2012). Under this approach, an impact is indicated by statistically significant Treatment*Time interactions ($p < 0.05$). All factors (i.e. Treatment and Time) were fixed. This is the full model: Community Structure = Elevation + Treatment + Time + Elevation*Treatment + Elevation*Time + Treatment*Time + Elevation*Treatment*Time. The final model excluded the interaction terms Treatment*Time and Elevation*Treatment*Time.

3. Results

Multidimensional scaling (MDS) of the macrobenthic community composition of all sampling data did not indicate a clear separation of the macrobenthic data based on spatial (different treatments) or temporal (different years, seasons, nourishment phases) variables, except for samples from the intertidal or shallow subtidal zone (Figure 2). Since these two habitats are substantially different to harbor other macrobenthic communities (Van Hoey et al. 2004) and only the intertidal zone was nourished, we treated the intertidal and shallow subtidal data separately. Appendices – Chapter 3 – table B1 and B2 give an overview of the abiotic and biotic mean values of the impact site per year and season for the intertidal and shallow subtidal zones separately while Appendices – Chapter 3 – table C1 and C2 provide the same overview for the control site. The Wilcoxon-values between impact and control site for all these abiotic and biotic factors per season in the intertidal and shallow subtidal zone can be found in Appendices – Chapter 3 – table D.

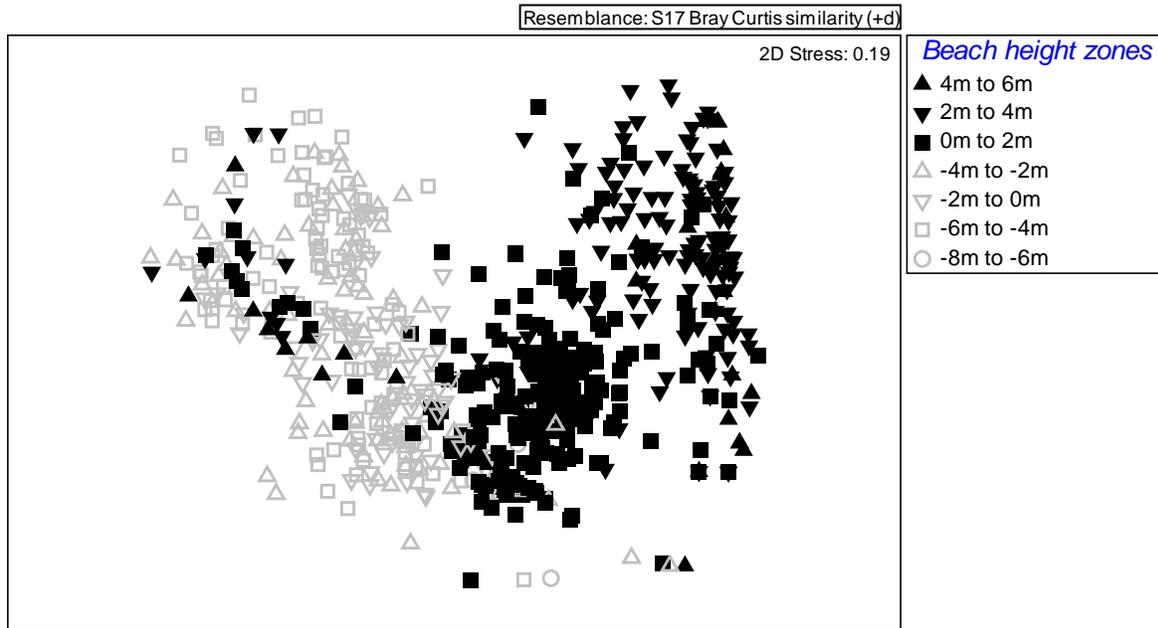


Figure 2: MDS plot of all data, showing a partitioning into two groups, being intertidal (black) and shallow subtidal (grey) samples with overlap around mean low water level, and to a lesser extent into beach elevation zones

3.1 Habitat changes

The nourishment activities created a wider and flatter intertidal beach (red line in figure 3 left), heightened over its entire width by the added sand (red dotted line in figure 3 left). Natural variation in accretion and sedimentation along the Belgian coastline can be detected by the similar evolution in the beach profiles of the control and impact site over the studied years and seasons. The final nourished beach slope (red dotted line in figure 3 left) resembled the original slope, although the entire beach was heightened. Moreover, it resembled the beach profile of the control site (red dotted line in figure 3 right) almost perfectly. Three years later, the impacted beach has still not returned to its original lower pre-nourishment profile.

Prior to the nourishment, the median grain size of the impact and control site did not differ significantly (table 2 and figure 6). Even though the nourished sand resembled the original sediment very closely with an average median grain size of 224 μm , the intertidal mean median grain size became coarser (from 215.89 \pm 3.63 μm in 2008 to 280.23 \pm 8.94 μm in spring 2010) than the value of the control site (stable mean around 200 μm) during and following the nourishment (table 2; figure 4 and 6), except in autumn 2010 when the control site had the coarsest sediment (impact site: 168.12 \pm 4.13 μm and control site: 259.69 \pm 4.35 μm). In the shallow subtidal zone, finer sediments were found at the impact site prior to nourishment. During the nourishment, the impact sediments became slightly coarser than at the control

site and this was also found in spring 2011. In autumn 2010, a coarse peak value could be detected in the control site ($244.79 \pm 4.31 \mu\text{m}$), followed by the lowest registered value (spring 2011: $137.51 \pm 14.47 \mu\text{m}$).

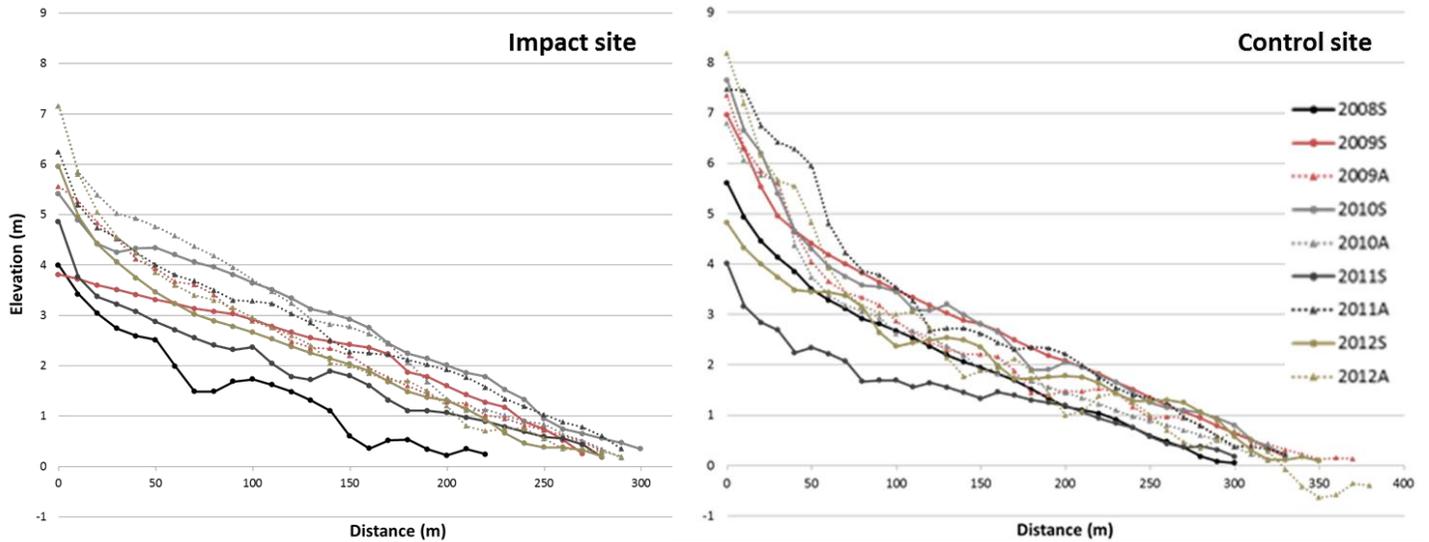


Figure 3: Intertidal beach profiles of the impact site (left) and control site (right) from spring 2008 (2008S) onwards

Table 2: Comparison of mean median grain size \pm SE (μm) between treatments (control and impact), time (years) and levels of the beach (intertidal and shallow subtidal)

Year	INTERTIDAL			SHALLOW SUBTIDAL		
	Impact Site	Control Site	Impact-Control	Impact Site	Control Site	Impact-Control
2008	215.89 \pm 3.63	206.35 \pm 3.55	9.53	182.30 \pm 2.35	186.48 \pm 2.01	-4.18
2009	212.98 \pm 3.94	197.53 \pm 3.02	10.38	188.42 \pm 2.69	182.33 \pm 2.38	6.09
	236.84 \pm 8.53	196.24 \pm 2.83	40.60	195.89 \pm 6.62	181.50 \pm 2.03	14.39
2010	280.23 \pm 8.94	201.75 \pm 5.32	78.48	158.35 \pm 12.17	182.32 \pm 2.57	-23.97
	168.12 \pm 4.13	259.69 \pm 4.35	-91.57	190.12 \pm 2.20	244.79 \pm 4.31	-54.68
2011	262.80 \pm 6.84	202.29 \pm 5.63	60.51	177.04 \pm 2.36	137.51 \pm 14.47	39.53
	252.39 \pm 9.54	206.81 \pm 8.25	45.58	168.94 \pm 5.46	178.84 \pm 4.30	-9.89
2012	267.89 \pm 5.69	208.63 \pm 6.97	59.26	172.98 \pm 1.39	184.06 \pm 2.81	-11.08
	247.86 \pm 2.87	201.97 \pm 1.76	45.89	174.73 \pm 0.35	178.92 \pm 0.78	-4.19

The coarsest median grain size can thus be detected high on the beach (figure 4). Since the same trend can be observed in both the impact and control site, this decrease can't be caused by the beach nourishment. The dispersion of the data during and following the nourishment is however higher in the impact than in the control site, especially in the intertidal zone, with more coarse samples having a median grain size higher than $300 \mu\text{m}$. The regression analysis between median grain size (μm ; dependent variable) and Elevation (m versus MLW), Time ($t_0, t, t_1, t_2, t_3, t_4, t_5, t_6$) and Treatment

(impact or control site) was done with all possible interaction terms (table 4; Appendices – Chapter 3 – table E). The highly significant interaction term Elevation*Treatment showed an average impact of the nourishment on the sediment grain size distribution. The interaction term Elevation*Treatment*Time was highly significant in all post-nourishment times, except t3 (spring 2011), indicating that the sediment grain size distribution has changed between the two treatments over time. Moreover, these changes become less clear over time (e.g. p-value increase over time), hinting that the sediment grain size distribution might have evolved back to original levels.

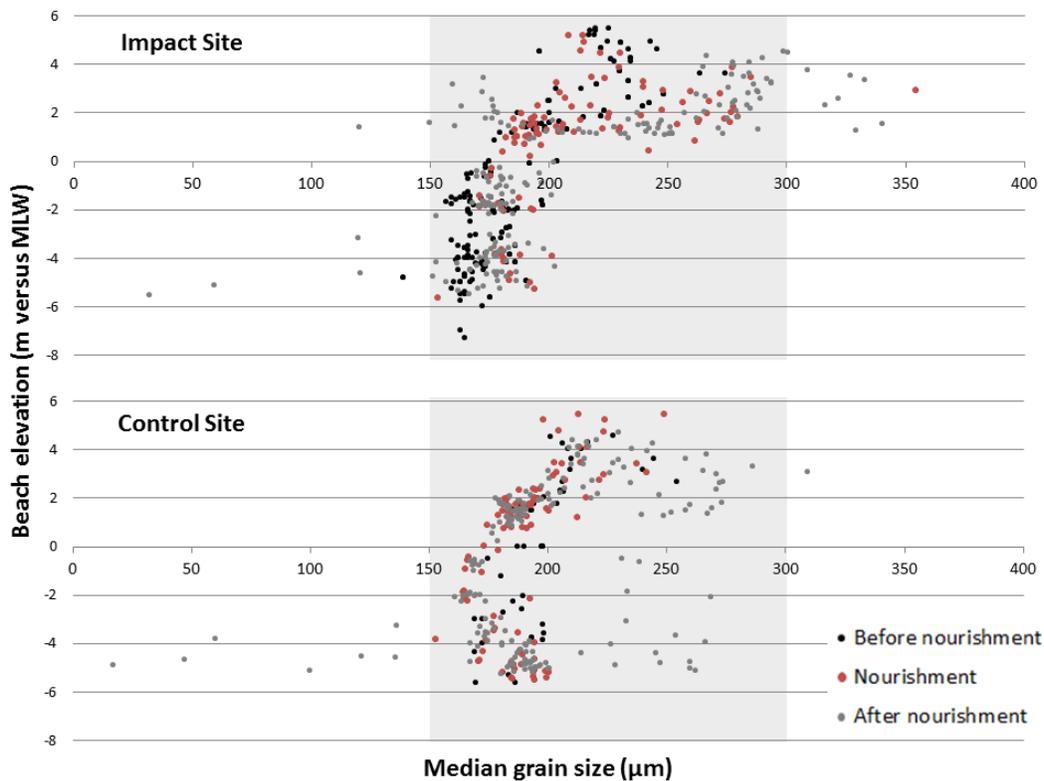


Figure 4: Beach elevation (m versus MLW) versus median grain size (μm) of the impact site (top) and the control site (bottom) before (2008; black), during (2009; red) and after the nourishment (2010, 2011 and 2012: grey); the grey area indicates the normal range of median grain size found on Belgian beaches

Table 3: The interaction terms ‘Elevation*Treatment impact’ and ‘Elevation*Treatment impact*Time’ of the regression analysis between median grain size (μm ; dependent variable), Elevation (m versus MLW), Time (t0, t, t1, t2, t3, t4, t5, t6) and Treatment (impact or control site) with p-values (significance: $p < 0.05$) (Appendices – Chapter 3 – table E)

Interaction terms	Estimate	SE	t value	Pr(> t)
Elevation*Treatment impact	6.0904	1.2027	5.064	5.41E-07
Elevation*Treatment impact*Time t0	-3.7051	1.9643	-1.886	0.05972
Elevation*Treatment impact*Time t1	11.8099	2.2811	5.177	3.04E-07
Elevation*Treatment impact*Time t2	-11.5074	2.3882	-4.818	1.82E-06
Elevation*Treatment impact*Time t3	-3.4478	2.2877	-1.507	0.13229
Elevation*Treatment impact*Time t4	7.4492	2.4568	3.032	0.00253
Elevation*Treatment impact*Time t5	4.8311	2.2836	2.116	0.03478
Elevation*Treatment impact*Time t6	4.9633	2.3879	2.079	0.03807

3.2 Impacts on macrobenthos and recovery

When looking at the intertidal and shallow subtidal abundance data separately, the data did not separate into groups according to treatment (impact and control clusters) or nourishment time (t0 to tx) or a combination of both (figure 5). The interaction terms Treatment*Time and Elevation*Treatment*Time were not included in the final model (table 4). There is however a difference in the zonation patterns of the community structure between the impact and control site (Elevation x Treatment), as well as a difference in time (Elevation* Time).

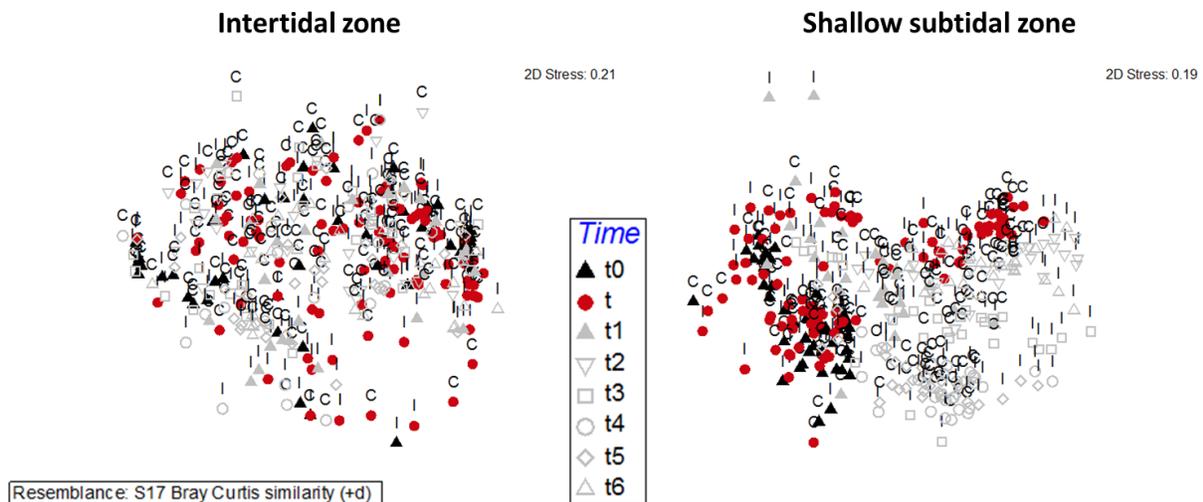


Figure 5: MDS plot of the data per treatment (impact=I and control=C) and per nourishment time (t0=black, t=red and t1=grey; t2 to t6=open grey symbols) in the intertidal and shallow subtidal zone

Table 4: Summary of a permutative linear model based on a Bray-Curtis distance matrix, partitioning multivariate variation in macrobenthic community structure in the intertidal and shallow subtidal zone zone

Main Model	Intertidal zone					Shallow subtidal zone				
	df	SS	Pseudo-F	R ²	P(perm)	df	SS	Pseudo-F	R ²	P(perm)
Elevation	204	85.601	2.442	0.709	0.001	253	99.427	2.324	0.808	0.001
Treatment	1	0.937	5.455	0.008	0.001	1	0.632	3.735	0.005	0.001
Time	7	4.846	4.028	0.040	0.001	7	9.095	7.682	0.074	0.001
Elevation x Treatment	45	12.744	1.648	0.106	0.001	28	6.018	1.271	0.049	0.019
Elevation x Time	46	12.156	1.538	0.101	0.001	27	6.070	1.329	0.049	0.007
Residuals	26	4.468		0.037		11	1.861		0.015	
Total	329	120.753		1.000		327	123.103		1.000	

The macrobenthos community structure will be further characterized by its species richness, abundance (number of individuals.m⁻²) and biomass (g AFDW.m⁻²) (figure 6). The evolution of the intertidal and shallow subtidal species richness and the shallow subtidal biomass in the impact site is almost identical to the evolution in the control site.

Figure 6 shows lower intertidal mean abundance, species richness and biomass values in the impact site compared to the control site. In autumn 2010 a peak value for abundance was observed in the impact site (121.92 ± 33.68 individuals.m⁻²) while in both sites, a peak value for species richness was observed (impact site: 6 ± 0.50 species and control site: 7 ± 0.36 species). The intertidal mean biomass on the other hand only exceeded 1 g AFDW.m⁻² in autumn 2012 (impact site: 1.10 ± 0.10 g AFDW.m⁻²). The impact site showed another, though smaller, peak value in autumn 2010 (impact: 0.67 ± 0.51 g AFDW.m⁻²). In autumn 2010, the high mean intertidal abundance, species richness and biomass in the impact zone coincide with the sudden drop in mean median grain size (168.12 ± 4.13 μm).

In the shallow subtidal zone, the mean abundance values before the nourishment (2008) were almost identical in both sites. Since then, there appears to be a difference between the impact and control values. When comparing the two seasons, peak abundance values could generally be observed in autumn, leading to a larger contrast between impact and control values in this season in the years 2009 (control site: 560.78 ± 90.76 individuals.m⁻²), 2010 (impact site: 839.54 ± 201.25 individuals.m⁻²) and 2012 (impact site: 844.70 ± 37.65 individuals.m⁻²), with 2011 as the only exception (218.60 ± 42.55 individuals.m⁻²). Consistently higher numbers of species were found in the control site, compared to the impact site. Moreover, the species richness increased over time in both sites and this increasing trend did not seem to level off by autumn 2012 (impact site: 16.20 ± 0.39 species and control site: 19.13 ± 0.43 species). An even higher increase in species richness is to be expected. Two mean biomass peak values could be detected in autumn 2010 (impact site: 85.23 ± 16.95 g AFDW.m⁻² and control site: 59.56 ± 15.24 g AFDW.m⁻²) and autumn 2012 (impact site: 74.23 ± 2.57 g AFDW.m⁻² and control site: 54.18 ± 2.33 g AFDW.m⁻²).

The dominant species in the intertidal and shallow subtidal zone of the impact and control site were determined by taking the five species that contributed the most to the total abundance before, during and after the nourishment (Table 5). The juvenile status of the bivalves *Ensis* species and *Macoma balthica* is clearly stated as they were significantly smaller than the adult individuals. In general, the intertidal zone of the impact and control site was dominated by two amphipod species (*Bathyporeia pilosa* and *Bathyporeia sarsi*), one isopod (*Eurydice pulchra*) and two polychaetes (*Scolelepis squamata* and *Nephtys cirrosa*). The dominance of either one of them shifted over the nourishment periods with the two amphipods taking turns in being the most dominant species. Before and after the nourishment *Bathyporeia pilosa* is clearly the most dominant species while during the nourishment *Bathyporeia sarsi* takes its place, in both the impact and control site. In the shallow subtidal zone of the impact site, *Donax vittatus* and *Nephtys cirrosa* dominate the pre-nourishment situation but their dominance decreased during the nourishment and they were eventually replaced by other dominant species, being *Ensis* juveniles and *Macoma balthica* juveniles. The *Ensis* juveniles also dominate the shallow subtidal zone of the control site during the nourishment so this dominance might be due to natural variation while *Macoma balthica* juveniles only appear dominant in the control site after the nourishment. *Magelona* species was the only species found in the impact site before and after the nourishment and in the control site in all three nourishment periods. *Cirratulidae* species became dominant after the nourishment in both the impact and control site.

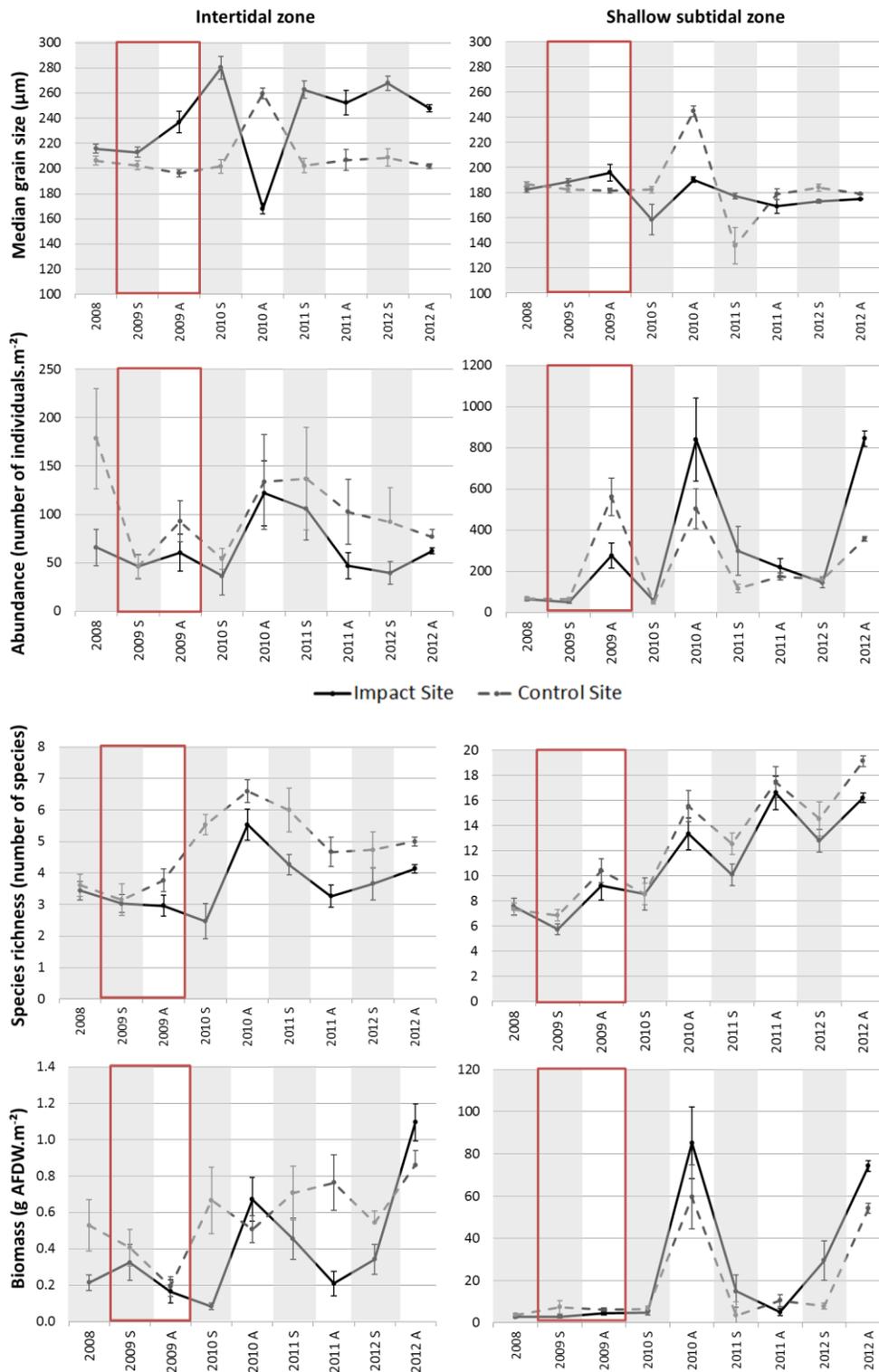


Figure 6: Mean median grain size (μm), species richness (number of species), abundance (number of individuals $\cdot\text{m}^{-2}$) and biomass (g AFDW $\cdot\text{m}^{-2}$) of the impact (black line) and control site (grey dotted line) per year with standard error, for the intertidal (left) and shallow subtidal zone (right) separately

Table 5: The dominant species in the intertidal and shallow subtidal zone of the control and impact site before, during and after the nourishment; the two most dominant species have been highlighted

Impact site	Before	%	During	%	After	%
Intertidal	<i>Bathyporeia pilosa</i>	38.69	<i>Bathyporeia sarsi</i>	33.33	<i>Bathyporeia pilosa</i>	34.06
	<i>Bathyporeia sarsi</i>	18.35	<i>Bathyporeia pilosa</i>	28.28	<i>Scolelepis squamata</i>	23.15
	<i>Scolelepis squamata</i>	17.64	<i>Scolelepis squamata</i>	14.48	<i>Bathyporeia sarsi</i>	15.36
	<i>Nephtys cirrosa</i>	7.40	<i>Nephtys cirrosa</i>	12.02	<i>Eurydice pulchra</i>	9.11
	<i>Eurydice pulchra</i>	7.25	<i>Eurydice pulchra</i>	4.51	<i>Nephtys cirrosa</i>	5.81
Shallow subtidal	<i>Donax vittatus</i>	56.38	<i>Ensis juveniles</i>	60.71	<i>Macoma balthica juveniles</i>	37.48
	<i>Nephtys cirrosa</i>	11.60	<i>Macoma balthica juveniles</i>	20.17	<i>Ensis juveniles</i>	19.51
	<i>Magelona species</i>	8.97	<i>Donax vittatus</i>	3.51	<i>Cirratulidae species</i>	13.81
	<i>Polydora species</i>	7.00	<i>Magelona species</i>	2.79	<i>Owenia fusiformis</i>	9.67
	<i>Nephtys hombergii</i>	4.67	<i>Nephtys cirrosa</i>	2.20	<i>Macoma balthica</i>	3.41
Control site	Before	%	During	%	After	%
Intertidal	<i>Bathyporeia pilosa</i>	46.22	<i>Bathyporeia sarsi</i>	31.67	<i>Bathyporeia pilosa</i>	31.87
	<i>Scolelepis squamata</i>	31.45	<i>Bathyporeia pilosa</i>	24.78	<i>Bathyporeia sarsi</i>	26.49
	<i>Bathyporeia sarsi</i>	10.95	<i>Scolelepis squamata</i>	19.34	<i>Scolelepis squamata</i>	12.37
	<i>Nephtys cirrosa</i>	3.51	<i>Ensis juveniles</i>	5.79	<i>Eurydice pulchra</i>	6.62
	<i>Eurydice pulchra</i>	3.45	<i>Nephtys cirrosa</i>	4.57	<i>Nephtys cirrosa</i>	4.79
Shallow subtidal	<i>Donax vittatus</i>	45.17	<i>Ensis juveniles</i>	85.89	<i>Ensis juveniles</i>	41.46
	<i>Magelona species</i>	22.44	<i>Magelona species</i>	2.88	<i>Magelona species</i>	13.03
	<i>Polydora species</i>	8.93	<i>Donax vittatus</i>	2.46	<i>Macoma balthica juveniles</i>	10.17
	<i>Nephtys cirrosa</i>	7.53	<i>Lanice conchilega</i>	2.15	<i>Spio species</i>	8.80
	<i>Nephtys hombergii</i>	5.02	<i>Polydora species</i>	1.27	<i>Cirratulidae species</i>	5.33

4. Discussion

4.1 Impacts on habitat characteristics

The nourishment resulted in measurable abiotic changes at the nourishment site. A wider and flatter intertidal beach was created, heightened over its entire width by the added sand (figure 3). No return to the originally lower elevated pre-nourishment profile was visible three years after nourishment. Even though the nourished sand resembled the original sediment very closely with an average median grain size of 224 μm , the intertidal sediment became significantly coarser during and after the nourishment. More intertidal samples had a median grain size higher than 300 μm , during and directly following the nourishment (table 2 and 3; figure 4 and 5).

The intertidal beach is not only closely linked to adjacent beaches that are down-drift but also to the shallow subtidal and offshore zone at the one side and to the dry beach, coastal dunes and hinterland on the other side of the beach ecosystem continuum, through the storage, transport and exchange of sand.

Therefore impacts on beaches have consequences for these adjacent habitats (Speybroeck et al. 2006a; Defeo et al. 2009; Komar 2009). Hence, the shallow subtidal sediments at the impact site did become slightly coarser during the nourishment and t3 (spring 2011). The coarsest median grain sizes were always detected high on the beach (table 3 and figure 4), regardless of the nourishment activities. However, the sediment grain size distribution had changed between the two treatments over time, starting directly after the nourishment (t1) and showing slow recovery in the three following years (tx). The highly significant interaction term Elevation*Treatment in the regression analysis assigns the nourishment as a cause for this impact effect.

Ignoring the effects of latitude and exposure, habitat characteristics define the immediate environment experiences by macrobenthos. An increase in sand particle size on a beach where tide range and wave energy have remained constant results in an irrevocable change in beach state and a decrease in species richness and abundance, at least in the medium term (McLachlan et al. 1996a). Even ecological nourishment introduces slightly coarser sediments (Veloso & Cardoso 2001; Defeo & McLachlan 2005) which might thus lead to a decrease in macrobenthos abundance and species composition. However, the nourished beach had a flatter beach slope over a wider beach and the median grain size stayed within the normal limits found on Belgian beaches (150 – 300 μm) so the effects of the ‘ecological’ nourishment on the macrobenthos should be small to negligible compared to usual nourishment practice.

4.2 *In situ* impacts on macrobenthos and recovery

The MDS analysis (figure 5) did not show groups according to treatment (impact and control clusters) or time (t0 to t6) nor to a combination of both. Conclusively, no significant effect of nourishment on the macrobenthos community structure could be detected in both the intertidal and shallow subtidal zone (table 4). There was however a difference in the zonation patterns of the community structure between the impact and control site (Elevation*Treatment), as well as a difference in time (Elevation*Time).

Figure 6 gives yet another affirmation of the lack of nourishment effect as it does not reveal any patterns that are clearly caused by the nourishment. The evolution of intertidal mean abundance, species richness and biomass values was almost identical in the impact and control site before, during and after the nourishment, with the values of the nourished site being lower (figure 6). In autumn 2010 peak values were recorded in mean abundance and mean species richness and to a lesser extent in mean biomass, coinciding with a sudden drop and rise in mean median grain size in the impact and control site. High numbers of both *Bathyporeia* amphipods (*B. pilosa*: < 250 μm and *B. sarsi*: 250 – 350 μm) and the polychaete *Scolelepis squamata* (125 – 500 μm) were detected, explaining the peak values in this period (Speybroeck et al. 2007; Van Tomme et al. 2012b).

In the shallow subtidal zone, the impact and control site were almost identical prior to nourishment (figure 5). The mean abundance values always peak in autumn since 2009, leading to a larger contrast between impact and control values in this season, except in 2011. In autumn 2010, peak values at both sites could be detected in mean abundance and mean biomass, coinciding with a sudden rise in mean median grain size in the control site and to a lesser extent in the impact site. The mean species richness was always higher in the control site and increased over time in both sites with no signs of leveling off by autumn 2012 (impact site: 16.20 ± 0.39 species and control site: 19.13 ± 0.43 species). *Donax vittatus* and *Nephtys cirrosa*, both preferring medium coarse sediment, dominated the pre-nourishment shallow subtidal zone but their dominance decreased during the nourishment and they were eventually replaced by other dominant species, being *Ensis* juveniles (natural variation since same pattern can be found in control site) and *Macoma balthica* juveniles (appeared later in the control site) (table 5). These two juvenile bivalves showed peak abundance values in autumn 2010, explaining the overall abundance and biomass peaks in that period. *Cirratulidae* species became dominant after the nourishment in both the impact and control site. Both bivalves and *Cirratulidae* species prefer finer, almost muddy sediments and they appeared when the silt fraction increased in both the impact and control site.

We observed short-term declines in species richness and biomass in the intertidal zone between the start of the nourishment (2009S) and 6 months after its completion (2010S). Literature reviews of usual beach nourishment practice (Nelson 1993; Hackney 1996; Peterson et al. 2000; Speybroeck et al. 2006a; Defeo et al. 2009; Schlacher et al. 2012) report such short-term declines in macrobenthic abundance, biomass and species richness following beach nourishment. Local benthos may be affected by burial and by changing sediment properties (McLachlan 1996; Essink 1999; Van Tomme et al. 2012b) as their limited mobility makes an active escape unlikely, often leading to large mortality (Schlacher et al. 2012). The recovery of the macrobenthic community then relies on the dispersal of benthic macroinvertebrate species of nearby beaches entering the water column and their planktonic larvae (Günther 1992; Cummings et al. 1995), both conditioned by the rate and direction of currents along the coastline (Hill 1991; Defeo 1996). Settlement will depend on the amount of suitable habitat and available space on adequate substratum within the nourished site (Defeo 1996). Connectivity patterns between sandy beach populations linked by larval dispersal are an unexplored field of research, and the mechanisms influencing larval distribution are poorly understood (McLachlan et al. 1996b; Defeo & McLachlan 2005; Schlacher et al. 2008). Long living organisms that don't reproduce successfully each year, e.g. Bivalvia and Echinodermata, need sufficient time to recover from the impact of usual beach nourishment practice. If no successive nourishment projects are planned, complete recovery can be attained within maximal 4 to 5 years. If the nourishment altered the habitat characteristics too drastically, especially median grain size and beach profile, complete recovery might never take place and a shift in the macrobenthos beach community can be visible.

These results show that at least in some cases nourishment under ecological optimal conditions can show no significant effects in the macrobenthos community structure 6 months after the nourishment (2010S). Within this short-term time frame, the macrobenthos community recovers from the impact of the ecological nourishment, showing no dispersal or recruitment limitations.

Designation of recovery time should as such reflect the long-term biological, ecological and physical processes controlling recolonization and succession (Oliver & Slattery 1976). Complete recovery will only be achieved when the recovered communities resemble those found prior to nourishment. One might consider a benthic community to be recovered when at least 80 % of the species diversity or biomass has been restored (Newell et al. 1998; Essink 1999), bearing in mind that biomass elicits recovery at a more rapid rate than the species diversity (Adriaanse & Coosen 1991). Caution is warranted as focus on a single parameter to measure recovery can lead to a biased view of the actual recovery. Polychaetes, abundant and cosmopolitan benthic invertebrates, recolonize quickly because of pelagic larval availability, variable food gathering behaviour and the ability to confine activities to the surface layer of the sediment (Hayward 1994; Hartmann-Schröder 1996). Pioneer populations of polychaetes may actually exceed the original populations in numbers of individuals and diversity (Grober 1992). High abundance, species richness and biomass values may as such be attributed to one or a few opportunistic species, misleadingly suggesting recovery.

4.3 Guidelines for beach management and coastal defence policy

A growing awareness of environmentally-sensitive approaches to beach management (Peterson et al. 2000; Speybroeck et al. 2006a; Schlacher et al. 2008; Defeo et al. 2009; Dugan et al. 2010) requires a good knowledge of the ecological effects of beach nourishment, especially of the factors influencing the size of the impact effects (figure 7, based on (Speybroeck et al. 2006a)). In general, the following ecological recommendations for beach nourishment have been formulated: (1) sediment sand should be non-contaminated (Essink 1999) and possess comparable hydraulic properties and characteristics as the original sediment, including clay/silt portion, shell content (Peterson & Manning 2001) and sediment grain size (McLachlan 1996; Hamm et al. 2002), (2) increase in beach slope should be kept as small as possible and the nourished beach profile should resemble the original profile (Short & Wright 1983; Defeo & McLachlan 2005; McLachlan & Dorvlo 2005), (3) nourishment should be done slowly in a sheeting spray of sand and water (rainbow spraying) to allow organisms to keep up with the sediment overburdens as they are applied (Grober 1992), (4) the total amount of nourished sediment should be kept as small as possible (Speybroeck et al. 2006a), limiting the thickness of deposits to 10 cm or less per single application (Schlacher et al. 2012) (5) nourishment should be done during winter season to disturb the reproduction and recruiting cycle as little as possible (Speybroeck et al. 2006a) and (6) several short nourishment projects in time (minimum one week in between) and space (leaving beach strips

unnourished) are preferred over broad-scale, long lasting ones, especially in areas where short term morphological changes are unpredictable (Hillen & Roelse 1995).

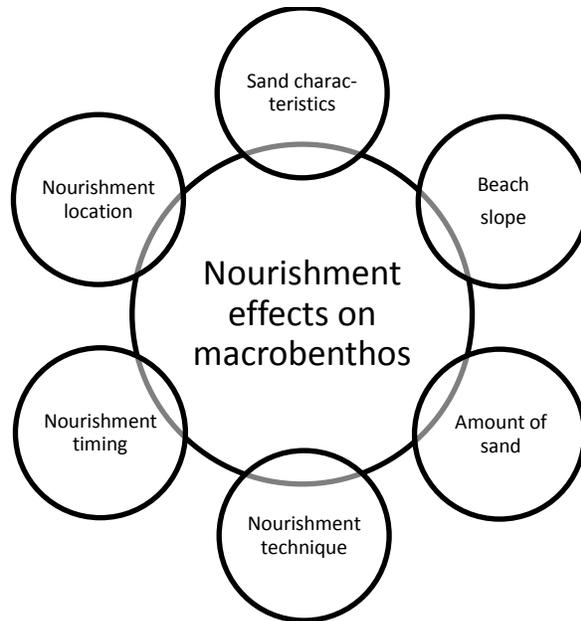


Figure 7: Factors causing ecological impact effects of beach nourishment (Speybroeck et al. 2007)

The nourishment on the intertidal part of the Belgian beach of Lombardsijde was an attempt to perform best-practice ecological beach nourishment. The optimal ecological conditions consisted of (1) creating a gentle sloped beach with at least a 250 m wide wet beach and a dry beach of 50 m wide, (2) using nourished sand with sediment characteristics resembling the original sand as close as possible, (3) establishing a natural transition from dunes to beach, (4) sand strengthening measures in the form of a row of wooden poles, some dune vegetation and fencing, and (5) heightening the harbour dam at the east side of the eastern palisade of Nieuwpoort to prevent the nourished sediment from relocating into the alongside harbour inlet. Unfortunately, the nourishment was postponed to spring and summer 2009 since winter 2008 – 2009 was a heavy storm season (October 15 – April 15) inducing large sand losses.

Nourishment under these conditions has no detectable impact effects on both the intertidal and shallow subtidal beach ecosystem, in particular the macrobenthos. Ecological nourishment thus proves to be an effective way of combating erosion while being less ecologically damaging than any other coastal engineering activity (Peterson et al. 2000; Speybroeck et al. 2006a; Schlacher et al. 2008; Defeo et al. 2009). Alternative locations for nourishment have been suggested though. For instance, nourishing in the shallow subtidal zone would result in a slow distribution of the nourished sand across the intertidal beach by hydrodynamic transport. It is likely that subtidal macrobenthos have a greater and more mobile pool of animals to supply recruits from which recovery can occur. However, the environmental

effects of this strategy are unknown (Schlacher et al. 2012) and caution should be taken as the shallow subtidal is a known nursery area for juvenile fish and epibenthos (Beyst et al. 1999a; Beyst et al. 2001b).

Other approaches to battle inundation, scour and erosion include restoring and supporting the natural development of sand dunes or creating natural buffer zones between the sea and the area at risk, e.g. coastal communities. A combination of soft and hard coastal defence structures, where appropriate, has been proposed as well. In some coastal areas, this seems to be the most sensible approach, especially when vast efficient hard structures are already in place. Looking at the Belgian coastal zone, it would be feasible to keep nourishing the touristic, (semi-)urbanized, top priority coastal defence beaches up to a high protective level as long as enough beach zones are protected in an adequate manner. These ‘pristine’ beaches could then evolve into healthy beach ecosystems and provide a stock of animals for recolonisation when those sacrificed beaches do become viable again. Retreat and relocation are still the most ecologically favorable option. The shoreline is then left to erode, while buildings and infrastructure are relocated further inland. Resistance to regulation and the economic value of coastal communities currently prohibits the establishment of sound long-range retreat policies (Schlacher et al. 2008; McLachlan et al. 2013).

5. Conclusion

The nourishment on the intertidal part of the Belgian beach of Lombardsijde was an attempt to perform best-practice ecological beach nourishment. A wider, higher and flatter intertidal beach with coarser sediment was created and no return to the pre-nourishment conditions was visible three years after nourishment. The sediment grain size distribution had changed as well, showing slow recovery in the three post-nourishment years. The analysis of the macrobenthos community structure shows that at least in some cases nourishment under ecological optimal conditions can show no significant effects on both the intertidal and shallow subtidal beach ecosystem 6 months after the nourishment. Within this time frame, the macrobenthos community recovered from the impact of the ecological nourishment. Ecological nourishment thus proves to be the least ecologically damaging way of combating erosion, compared to all other coastal engineering activities.