The ecological effects of deep sand extraction on the Dutch continental shelf

Implications for future extraction

Maarten Fije de Jong

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Aan mijn ouders, oudere broer Jeroen, mijn nichtjes Marit en Robine en
natuurlijk mijn jongere broertje Rob de Jong

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Chapter 1

General introduction

1 Problem definition

The Dutch coastal zone is characterised by intensive use including activities such as fishing, shipping, wind farming, dredging, disposing of dredged sediment, beach nourishment, and the extraction and transportation of gas, oil and aggregates. These activities have different effects on the marine environment and some are likely to intensify in the future (Jongbloed et al., 2014). Due to this increase, well-considered use of space is necessary to avoid future conflicts.

The main objective of this thesis is to investigate the short-term effects of deep sand extraction and ecological landscaping on macrozoobenthos, demersal fish and habitat characteristics on the Dutch continental shelf (DCS). This project is part of Building with Nature (BwN), a public-private innovation programme which aims at the development of sustainable plans for coastal, delta and river areas.

1.1 Sand extraction in the Netherlands

In the Netherlands, approximately 26 million m^3 of marine sand is used annually for coastal nourishment and construction (Stolk and Dijkshoorn, 2009; ICES, 2014a). Before 1987, less than 5 million m^3 of marine sand was extracted and this increased to 15 million m^3 until 1995. Due to the construction of a seaward harbour extension of the Port of Rotterdam (PoR), Maasvlakte 2 (MV2), the volume of extracted sand increased up to 120 million m^3 between 2009–2012 (Fig. 1.1 in red). An increase in the annually nourished volumes of sand from 12.5 up to 40–85 million m^3 of sand for counteracting effects of future sea level rise is anticipated (Deltacommissie, 2008) (Fig. 1.1 in green).

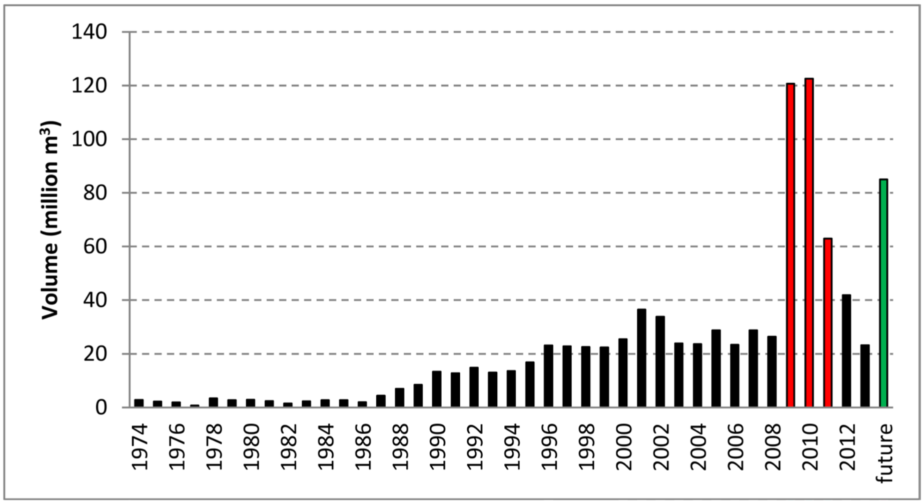


Figure 1.1 Total volume of extracted sand (in million m^3) on the Dutch continental shelf (DCS). Red: peak in volume due to harbour enlargement Maasvlakte 2 (MV2) and in green, anticipated increased annual volume for nourishments. Source Rijkswaterstaat (ICES, 2014a).

A considerable volume of extractable sand is present on the DCS. In the 5000 km^2 area between the 20 m isobath and the 12 nautical mile contour, the layer of sand is several meters thick (Fig. 1.2), and yields almost 2.5 billion m^3 of sand when 5 m of sand is extracted. This means that with the anticipated increase of annual nourishments, the volume of sand may be exhausted within decades. The maximum sand layer thickness is 12 m but the actual thickness may be even larger and extraction depth in the MV2 borrow pit was around 20 m. Generally, only shallow sand extraction down to 2 m below the seabed beyond the 20 m isobath is allowed in the Netherlands (IDON, 2005).

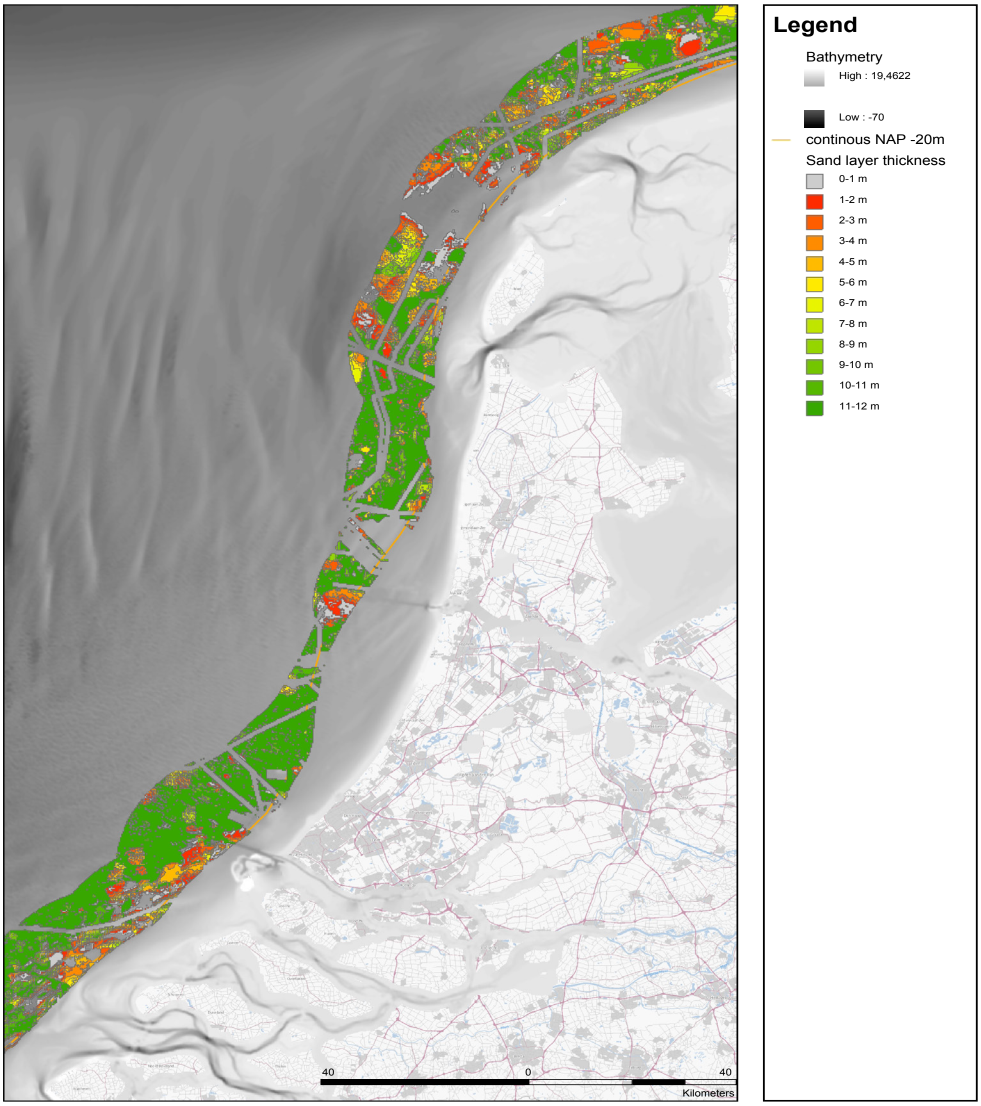


Figure 1.2 Sand layer thickness in the sand extraction area with former extraction areas and pipeline and cable areas. Source: Rijkswaterstaat.

Due to increasing human activities and the sand demand on the Dutch continental shelf, a new sand extraction strategy is formulated in the revised Integrated Management Plan for the North Sea 2015 (IMPNS 2015) to guarantee sufficient supply of marine sand in the intensively used coastal zone at reasonable costs with space for other activities (IDON, 2011). The Dutch government considers sand extraction to be of national importance and gets higher priority than other activities. Starting points of sand extraction are: ecologically responsible, cost-effective, sustainable sand extraction that is smart in relation to supply and properly harmonised in spatial planning terms. The Dutch authorities now allow extraction depths larger than 2 m in areas 2 km seaward from the 20 m depth contour for sand extraction projects larger than 10 million m^3 (IDON, 2011).

The effects of deep sand extraction may be more severe compared to the effects of shallow extraction. Bathymetry, sediment characteristics, and current velocity can be altered which may also have implications for macrozoobenthos and demersal fish. To investigate these possible effects, the Dutch authorities commissioned around 1999 a study in a deep temporary extraction site ‘PUTMOR’, with 6.5 million m^3 of sand extracted, an initial water depth of 23 m and extraction depths between 5–12 m. The study revealed that there were no indications that a borrow pit with a final water depth of less than 40 m would lead to unacceptable effects (Boers, 2005). Due to the short period of investigation, ecological data was not collected during the PUTMOR study and to fill in these knowledge gaps we investigate temporary ecological effects in the deep MV2 borrow pit. We first investigate which environmental variables influence the distribution of macrozoobenthos and determine how many macrozoobenthic assemblages are present in the intensively used area of the Dutch coastal zone prior to the construction of MV2. Short-term effects of deep sand extraction on macrozoobenthos, demersal fish, sediment characteristics and hydrographical variables are investigated. Effects of different sand extraction depths on the DCS are compared and ecosystem-based design rules for future borrow pits are developed which simultaneously maximise sand yields and decrease the surface area of impact.

1.2 Maasvlakte 2 (MV2) borrow pit

For the seaward harbour expansion MV2 of the PoR, approximately 220 million m^3 of sand was extracted from 2009 to 2013, with an average extraction depth of 20 m. The MV2 borrow pit is situated in front of the PoR outside the 20 m isobath (Fig. 1.3, no. 1), and is 2 km long and 6 km wide. For MV2, the Dutch government allowed sand extraction deeper than the common 2 m, primarily to decrease the surface area of direct impact. The surface area of the borrow pit was thereby reduced from 110 km^2 at 2 m extraction depth to only 11 km^2 at 20 m extraction depth. Knowledge on the recolonisation of macrozoobenthos, response of fish and long-term morphodynamic evolution of the seabed is not available.

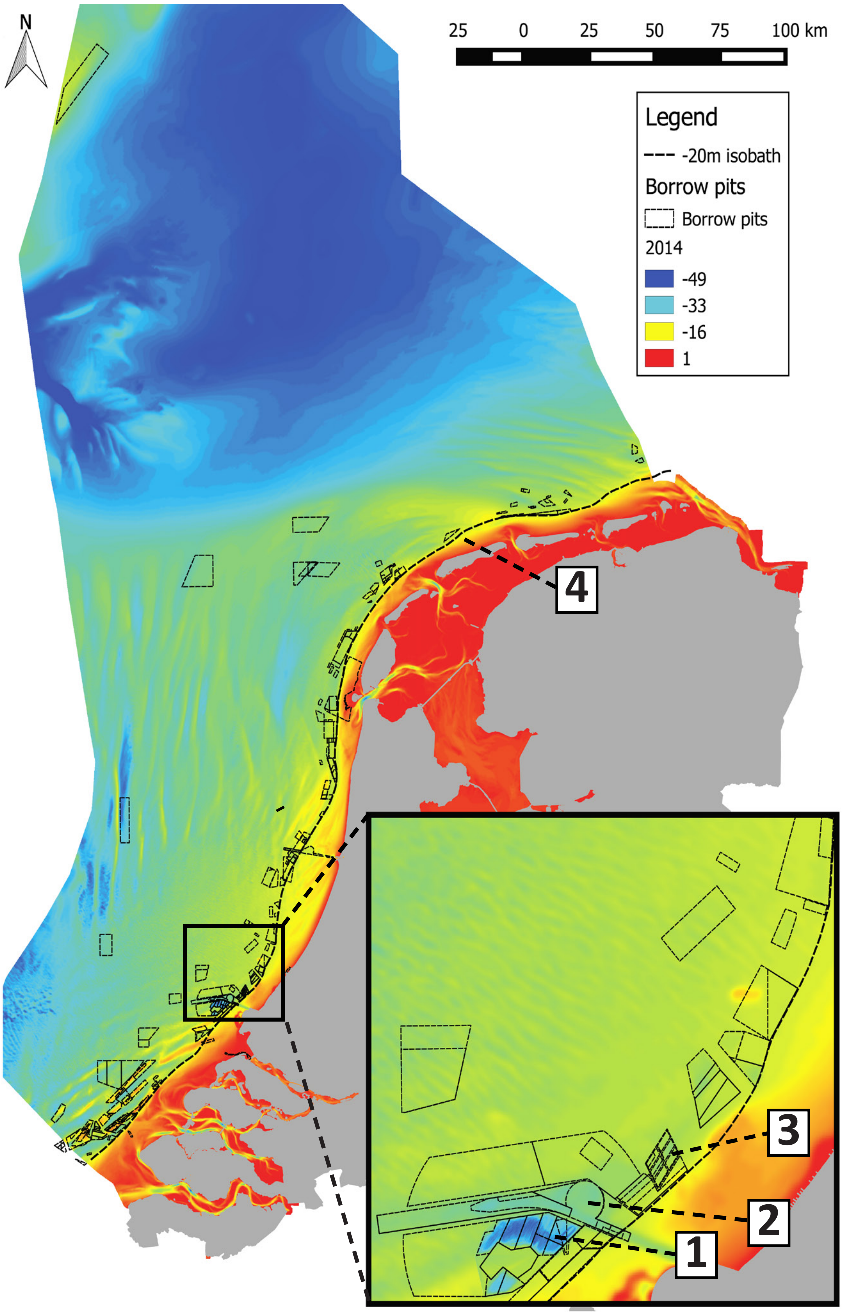


Figure 1.3 Solitary borrow pits (black dashed boxes) with shallow sand extraction depth (2 m) except **1-3** with larger extraction depths. The 20 m deepened Maasvlakte 2 borrow pit is denoted with (**1**), the 8 m deepened shipping lane (**2**), the 5–12 m deep temporary borrow pit (PUTMOR) which is currently used as disposal sites for dredged fine sediment (**3**). The inset shows **1**-**3** in higher detail. The borrow pit which was used to investigate the ecological effects of shallow extraction North of the barrier island Terschelling (Dalfsen et al., 2000; van Dalfsen and Essink, 2001) is denoted with (**4**). Source: Rijkswaterstaat.

1.3 Sand extraction and macrozoobenthos

In general, sand extraction has direct impact since benthic organisms are damaged or removed and bathymetry and sediment composition may change considerably. Indirect effects are increased turbidity, release of nutrients or toxins, and smothering by sedimentation.

Under natural conditions, the distribution of macrozoobenthic biomass and species composition in the North Sea was found to be correlated with variables such as sediment mud content, water depth, water temperature, sediment median grain size, sediment chlorophyll a content, sediment organic carbon content and latitude (Heip et al., 1992; Künitzer et al., 1992; Holtmann et al., 1996; van Hoey et al., 2004; Degraer et al., 2008; Verfaillie et al., 2009). Naturally occurring bed forms influence macrozoobenthos distribution and composition on smaller scale. On sand waves, a zonation exists (Fig. 1.4) where crests accommodate poorer macrozoobenthic communities and slopes and swales richer communities (Baptist et al., 2006; van Dijk et al., 2012). Distinct spatial patterns in biological assemblages on estuarine intertidal flats were related to patterns in bed shear stress (Herman et al., 2001; Bolam et al., 2002). Bed shear stress is the amount of force exerted by flowing water per unit of area of seabed and plays a role in sediment transport processes, the formation of bedforms, and sedimentation or erosion of the seabed.

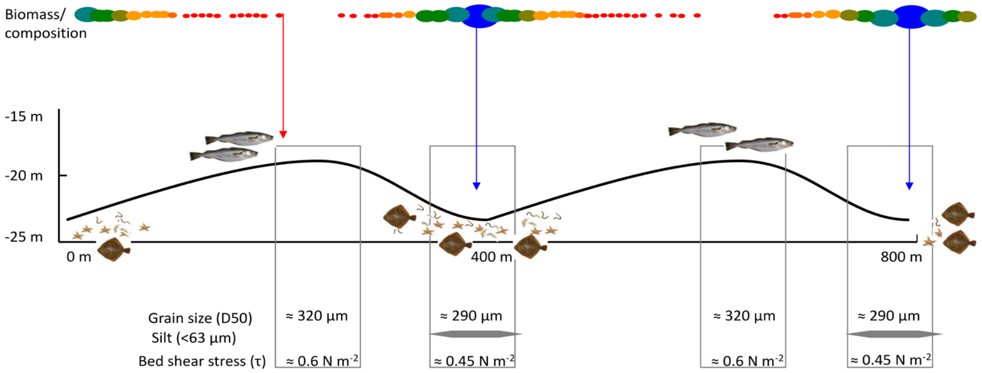


Figure 1.4 Conceptual sand waves with bathymetry, sediment characteristics, macrozoobenthos and demersal fish characteristics.

In 1979, the first studies on the impacts of large-scale sand extraction in the North Sea appeared (de Groot, 1979b; de Groot, 1979a). Since then, many international studies have investigated various ecological aspects of shallow sand extraction (Newell et al., 1998; Seiderer and Newell, 1999; van Dalfsen et al., 2000; Desprez, 2000; Boyd et al., 2003; Boers, 2005; Boyd et al., 2005; Newell et al., 2004; Barrio Froján et al., 2008; Desprez et al., 2009; Le Bot et al., 2010; de Backer et al., 2014). The recovery time of macrozoobenthic assemblages to pre-extraction conditions after shallow sand extraction (2 m) in the North Sea at a site near Terschelling (Fig. 1.3, no. 4) is estimated to be 4–6 years (van Dalfsen et al., 2000; Boyd et al., 2005; van Dalfsen and Essink, 2001). In a region with high extraction intensity off the south-east coast of England, higher variability in the recovery of macrozoobenthos was observed and complete recovery was not reached 11 years after the cessation of extraction (Wan Hussin et al., 2012). For muddy areas, however, Newell et al. (1998) estimated that recovery time is around 6 to 8 months. During the PUTMOR study there were no indications that intermediate sand extraction depths (5–12 m) would lead to unacceptable effects and that recovery of benthic assemblages could be possible.

1.4 Sand extraction and higher trophic levels

Under natural conditions, fish assemblages are linked to biotic and abiotic habitat characteristics, and to meso-scale bedforms (Ellis et al., 2011; Sell and Kröncke, 2013). Ellis et al. (2011) found that species diversity of infauna, epifauna and fish were larger in the silty troughs of sandbanks off the coast of the UK than on the crests. Fish assemblages at North Sea scale were influenced by bottom water temperature, bottom water salinity, tidal stress, water depth, and beam trawl effort (Callaway et al., 2002; Reiss et al., 2010). Large-scale sand extraction was found to have a negative impact on fish in the Yellow sea (Hwang et al., 2013) and a decline of more than 70% of number of fish and total number of species was observed (Son and Han, 2007). Aggregate extraction, however, may also lead to new habitats and may favour macrozoobenthos and fish (Desprez, 2000). Fishing fleets near aggregate extraction sites were not deterred by extraction activities and fishing effort of potters and English dredgers were even positively correlated to aggregate extraction intensity (Marchal et al., 2014). Sand extraction may also positively influence marine mammals (Todd et al., 2014).

1.5 Ecological landscaping

In the UK, gravel-seeding techniques were tested to restore the seabed after gravel extraction (Cooper et al., 2011). In the MV2 borrow pit, the approach of ‘ecological landscaping’ techniques was investigated. Two sandbars were left behind after sand extraction, mimicking natural sand ridges in order to increase habitat heterogeneity and thereby possibly increasing post-extraction macrozoobenthic and demersal fish species richness and biomass.

1.6 Thesis outline

The main objective of this thesis is to investigate the short-term effects of deep sand extraction and ecological landscaping on the Dutch continental shelf (DCS) on macrozoobenthos, demersal fish, and environmental variables. This thesis provides insights of present knowledge gaps and gives recommendations for the MV2 monitoring campaign and potential research areas regarding intermediate sand extraction depths. The effects of deep sand extraction are evaluated in the context of the Marine Strategy Framework Directive (MSFD) and Good Environmental Status (GES). The knowledge can be used by ecologists, coastal morphologists, engineers, coastal zone managers and dredging companies.

Chapter 2 describes which macrozoobenthic assemblages were present and how they were distributed in a 2500 km^2 research area in front of the PoR and in the designated area of the MV2 borrow pit. Relationships between environmental variables and macrozoobenthos are described.

Chapter 3 is the first study which investigates short-term effects of deep sand extraction and ecosystem-based landscaping on macrozoobenthos and sediment characteristics. This chapter describes relationships between macrozoobenthos and environmental variables. Sedimentation rate and sedimentological evolution are monitored and significant differences in ecological effects between shallow and deep sand extraction are presented.

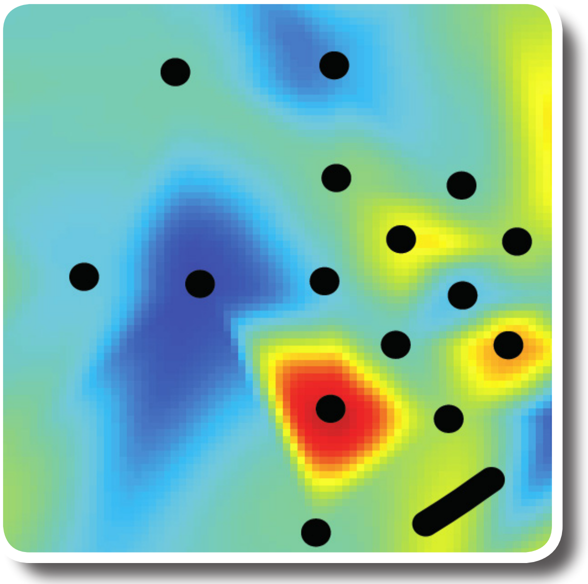
In **Chapter 4,** short-term effects of deep sand extraction and ecosystem-based landscaping on demersal fish and environmental variables are described. Significant differences in demersal fish biomass and species assemblage between the MV2 borrow pit, ecological landscaped sandbars and the reference area are described.

Chapter 5 summarises the ecological effects of different extraction depths on the DCS and combined with estimated bed shear stress values, ecosystem-based design rules are formulated for future borrow pits which simultaneously maximise sand yields and decrease the surface area of direct impact of sand extraction. The potential use of ecosystem-based design rules for the design of borrow pits outside the DCS is discussed.

Chapter 6 summarises the main results of chapters 2–5 and provides insights of present knowledge gaps and recommendations for the MV2 monitoring campaign and potential research areas regarding intermediate and deep sand extraction depths.

Chapter 2

Relationships between macrozoobenthos and habitat characteristics in an intensively used area of the Dutch coastal zone



Maarten F. de Jong[1,2], Martin J. Baptist[1], Han J. Lindeboom[1,2] and Piet Hoekstra[3]

ICES journal of Marine Science, 2015, 72 (8): 2409–2422 with an added paragraph on 'species richness, biomass, and environmental variables'.

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Abstract

Distribution patterns and species composition of macrozoobenthos were investigated in the Dutch coastal zone in a 2500 km^2 research area in front of the Port of Rotterdam (PoR). Relationships between macrozoobenthic assemblages and environmental variables were determined using non-metric dimensional scaling (nMDS) based on 470 boxcore, bottom sledge, and sediment samples collected in spring 2006 and 2008. We investigated two types of benthic assemblages, infaunal assemblages sampled with a boxcorer and epifaunal assemblages sampled with a bottom sledge. Five main in- and epifaunal assemblages were distinguished based on clustering techniques and nMDS ordinations. Macrozoobenthic species composition correlated with combinations of measured sediment variables and modelled hydrographic variables. Macrozoobenthic species richness and biomass were the highest at 20 m deep areas with a grain size of 200 µm, elevated mud and organic matter content, and low mean bed shear stress. Considerable interannual differences in macrozoobenthic assemblage distribution were observed which resulted from an increase of Echinoids, Phoronids, and razor clams. A distinct, highly productive and species-rich macrozoobenthic white furrow shell (*Abra alba)* assemblage coincided in an 8 m deepened shipping lane and near a disposal site for dredged fine sediment. This may be an indication that the benthic system can be changed by these human activities. Modelled bed shear stress is an explanatory variable in addition to sediment variables in explaining distribution patterns in macrozoobenthos.

2.1 Introduction

The Dutch coastal zone in front of the Port of Rotterdam (PoR) is characterised by intensive human use, including activities such as fishing, shipping, wind farming, dredging, disposing of dredged sediment, as well as the extraction of gas, oil and aggregates. These activities have different effects on the marine environment and some are likely to intensify (Jongbloed et al., 2014).

Macrozoobenthos on the North Sea scale is correlated with environmental variables such as sediment mud content, water depth, water temperature, sediment median grain size, sediment chlorophyll a content, sediment organic carbon content, and latitude (Heip et al., 1992; Degraer et al., 2008; van Hoey et al., 2007; Künitzer et al., 1992; Holtmann et al., 1996; Verfaillie et al., 2009; van Hoey et al., 2004; Degraer et al., 1999). Near-bed salinity affected epifauna (EP) in the North Sea (Callaway et al., 2002; Reiss et al., 2011; Reiss et al., 2010), whereas bed shear stress affected macrozoobenthos in intertidal areas (Herman et al., 2001; Ysebaert et al., 2003; Puls et al., 2012). Bed forms also influence small-scale macrozoobenthic distribution and composition (Baptist et al., 2006; van Dijk et al., 2012). Owing to climate and seasonal fluctuations, temporal variations in species composition commonly occur (Straile and Stenseth, 2007; Kröncke et al., 2013).

We investigated distribution patterns, species composition, and temporal differences of in- and epifaunal assemblages in an intensively used Dutch coastal zone in front of the PoR. Relationships between macrozoobenthos, measured environmental variables, and modelled hydrographic variables were determined based on high-resolution data. This study can be used as a baseline for future comparisons and to help in the protection of marine biodiversity. Determined relationships between macrozoobenthos and environmental variables can be used for the prediction of ecological effects due to human activities.

This study aims to answer the following questions:
(i) Which environmental variables influence the distribution of macrozoobenthos?
(ii) Are there significant differences in macrozoobenthic assemblages in the intensively used area of the Dutch coastal zone?
(iii) Are there temporal differences in the distribution of macrozoobenthic assemblages?

2.2 Materials and Methods

Research area

The research area stretches over 2500 km^2 in front of the PoR (Fig. 2.1). The seabed consists of fine (125–250 µm) to medium (250–500 µm) sand deposited during the Holocene. The shallowest part is the shoreface (Fig. 2.1, no. 1), an area with alongshore outer breaker bars and a maximum water depth of 10 m. The northern part of the research area is characterised by the presence of shoreface-connected ridges (Fig. 2.1, no. 2) with crest orientations 20–35° clockwise with respect to the coast-parallel tide (van de Meene and van Rijn, 2000b). These ridges can be up to 30 km long, 2 to 4 km wide, between 2 and 6 m high, and are situated in water depths between 14 and 18 m (van de Meene and van Rijn, 2000a).

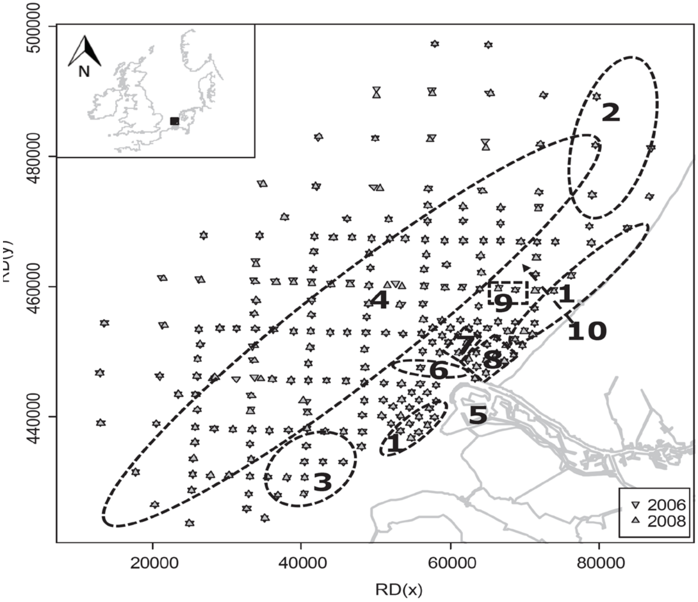


Figure 2.1 Two hundred and thirty-five sampling locations of the PoR baseline study in 2006 and 2008 with four distinct morphological units and three important user functions: (**1**) the shoreface, (**2**) shoreface-connected ridges, (**3**) Zeeland ridges, (**4**) sand waves, (**5**) the PoR with Maasvlakte 2 which was realised after the monitoring, (**6**) 'Euromaasgeul' shipping lane, (**7**) the lowered disposal site (Verdiepte Loswal) for fine sediment, (**8**) disposal site 'North' (Loswal Noord) for coarser sediment, (**9**) disposal site 'Northwest' (Loswal Noord West) and (**10**) sewage treatment effluent discharge (RWZI Houtrust)

The Voordelta region, the southern part of the research area, is characterised by Zeeland ridges (Fig. 2.1, no. 3). Sand waves (Fig 2.1, no. 4) are found in deeper waters, with wavelengths of 100 to 800 m, amplitudes up to 5 m, and crests orientated perpendicular to the tidal current (Hulscher, 1996).

Human activities

A 57 km long and 8 m deepened shipping lane, the 'Euromaasgeul' (de Ronde, 2008), was realised in the 1970s to guarantee access to the PoR (Fig. 2.1, no. 6). Areas near the Euromaasgeul are used for anchoring of vessels. To guarantee accessibility, 2.6 million m^3 of sediment was dredged out of the entrance of the shipping lane and harbour from 2001 to 2005, and relocated to disposal sites 'Northwest' (Fig. 2.1, no. 9) and the lowered disposal site (Fig. 2.1, no. 7). From 2007 to present, dredged coarser sediment (~0.8 million m^3) is relocated in the coastal foundation directly north of the shipping lane at the disposal site 'North' (Fig. 2.1, no. 8) whereas the finer sediment is relocated to the lowered disposal site. The disposal site 'North' was earlier used from 1961 until June 1996 and fine sediment is distributed to the near surrounding of disposal sites (Stronkhorst et al., 2003).

In the Dutch coastal zone, 26 million m^3 of sand is extracted annually from local borrow pits outside the 20 m isobath. A sewage effluent discharge (RWZI 'Houtrust') is located 3.5 km northeast of disposal site 'Northwest' (Fig. 2.1, no. 10). Fishing effort is not uniformly distributed in the study area, higher intensities of beam trawling, seine fishing, and otter trawling are found in near-shore waters. Outside the 12 nautical mile zone, beam trawl fishery is uniformly distributed (WGSFD, 2014).

Macrozoobenthos sampling

We used data from the baseline study of the Environmental Impact Assessment (EIA) for the construction of the seaward harbour enlargement Maasvlakte 2 of the PoR that were collected in spring 2006 and 2008. From this study, 470 boxcore, bottom sledge, and sediment samples collected at 235 sampling locations in a 2500 km^2 research area in front of the PoR were selected (Fig. 2.1). Sample stations were located in a grid. Two grid refinements were implemented in coastward direction, in order to sample more spatial variation in biotic and abiotic variables resulting from the Rhine region of freshwater input (Rhine ROFI) and human activities.

Bottom sledge sampling locations matched boxcore sampling locations. Sampling was executed from 18 April–22 June 2006 and 17 April–17 June 2008. The ships' GPS-system logged position of the sampling locations. We used water depth data from the Netherlands Hydrographic Office (multibeam, 25 x 25 m resolution) with reference level lowest astronomical tide.

The boxcorer was used to sample macrozoobenthic infauna (IN), larger than 1 mm and mostly living in the seabed. The Monitor Taskforce of the Royal Netherlands Institute for Sea Research (NIOZ) executed the boxcore sampling. The boxcorer surface area was 0.0774 m^2, with a maximum penetration depth of 30 cm. Samples were wet-sieved using a 1 mm mesh sieve and the residue was stored in jars with a seawater solution of 40% buffered formaldehyde (Craeymeersch and Escaravage, 2010; Perdon and Kaag, 2006b). Specimens were identified up to species level when possible. Infaunal ash-free dry weight biomass (g AFDW m^{-2}) was analysed by mass loss on ignition (MLOI), 2 days at 80 °C followed by 2 hours at 580 °C. When sea urchins were smaller than 5 mm, identification up to species level was not possible and they were lumped as *Echinoidea* spp.

The bottom sledge was used to sample macrozoobenthic IN and EP with a size range of 0.5–10 cm. Bottom sledge samples are hereafter called epifauna EP, although large IN are collected as well. The Institute for Marine Resources & Ecosystem Studies (IMARES Wageningen UR) executed the bottom sledge sampling. The sledge was equipped with a 5 mm mesh cage. On average, a surface area of 15 m^2 was sampled during each sledge haul of ~150 m length, 10 cm width, and a maximum penetration depth of 10 cm. Wet weight of EP was directly measured (g m^{-2} WW). Razor clam *Ensis* spp. was not captured in whole, so biomass was determined by using regression equations based on previous IMARES Wageningen UR field surveys (Craeymeersch and van der Land, 1998). Sea urchins (*Echinoidea* spp.) are too fragile and were completely damaged after a bottom sledge sampling procedure and therefore not countable.

Sediment sampling

Sediment samples from the upper 5 cm were collected from untreated boxcore samples and kept frozen until analysis. Sediment samples were freeze-dried, homogenised and analysed with a Malvern Mastersizer 2000 particle size analyser. Percentile sediment grain size (D_{10}, D_{50}, and D_{90}) and sediment grain size distribution among the different classes: clay (< 4 µm), silt (4–63 µm), mud (<63 µm), very fine sand (63 µm–125 µm), fine sand (125 µm–250 µm), medium sand (250 µm–500 µm), and coarse sand (500–1600 µm) were measured as percentage of total volume. Sediment grains larger than 1600 µm were not taken into account in this study. Sediment sorting (D90/D10) was calculated and organic matter (OM) was analysed by MLOI and expressed as percentage of sediment mass; sediment samples were freeze-dried and placed for 2 h at 580 °C. The MLOI method used to analyse infaunal ash-free dry weight biomass and OM may be prone to overestimation due to the loss of carbon from the combustion of carbonate above temperatures of 550 °C and from differences in dehydration rates of clays (Santisteban et al., 2004). OM values derived by MLOI were compared with unbiased organic carbon values derived with a CHN analyser (CE Instruments NC2500). This comparison revealed a linear relationship in the range of 0–5% organic matter. Organic carbon values are six times lower than OM values (Dorst, 2012).

Modelling of hydrographic variables

Hydrographical data were modelled for the year 2007 with a model for the southern North Sea (Zuidelijke Noordzee: ZUNO) in the Delft3D FLOW simulation package. The year 2007, is commonly used in earlier reports (de Mesel et al., 2011) and is considered as an average climatological year. The horizontal grid ranged from 6–20 by 5–30 km with finest resolution in the research area (6 x 5 km). Twelve vertical σ-layers were used with finer vertical resolution selected near the seabed and sea surface. From top to bottom, these layers, respectively, represent 4.0, 5.6, 7.8, 10.8, 10.9, 10.9, 10.9, 10.9, 10.8, 7.8, 5.6, and 4.0% of the water depth. Water flow is modelled using time-steps of 5 minutes and forced with open boundary conditions, meteorological, riverine discharge and wave data for the modelled period (de Mesel et al., 2011; Tonnon et al., 2013b). Annual averaged (mean) and maximum values of bed shear stress (N m^{-2}) and near-bed salinity (ppt) were determined from the 2007 model run.

Statistical analysis

Infaunal and epifaunal species composition were analysed using hierarchical agglomerative clustering with Bray–Curtis dissimilarity index based on fourth root-transformed density data and average linkage (Legendre and Legendre, 1998) using the HCLUST function of package ‘vegan’ (Oksanen, 2013). The number of significant macrozoobenthic species assemblages, with the assumption of no a priori groups, was assessed with the similarity profile routine SIMPROF of package ‘clustsig’ (Clarke et al., 2008). Dufrêne–Legendre indicator species analysis was applied using the indval function of package ‘labdsv’ to determine indicator species of species assemblages based on the product of the relative frequency and relative average abundance (Dufrêne and Legendre, 1997). Shapiro–Wilk test, Levene’s test, and diagnostic residuals plot were used to check for normality and homogeneity of the abiotic variables.

The non-parametric Kruskal–Wallis one-way multi-comparison tests (package ‘pgirmess’) was used to determine significant differences in biotic and environmental variables between assemblages. We used non-metric dimensional scaling (nMDS) of package ‘vegan’. Bray–Curtis dissimilarities were used to calculate differences in species composition (Oksanen, 2013; Clarke and Ainsworth, 1993). The lowest stress ordination was selected after repeating the nMDS routine with two-, three- and four-dimensional settings with untransformed, square- and fourth-root transformed data. Stress values below 0.2 are considered as potentially useful, whereas values below 0.1 are regarded as good ordinations (Clarke and Warwick, 2001). Variables were fit to the nMDS ordination using the envfit function in package ‘vegan’ (999 permutations) to find significant correlations. Assemblages in the nMDS ordination are positively correlated when they match with the superimposed correlation arrows of an environmental variable and are negatively correlated when they are in opposite direction. The arrows show the direction of the increasing gradient of the environmental variable, and the length of the arrow is proportional to the correlation coefficient between the variable and the nMDS ordination. The bioenv function in ‘vegan’ was used to determine the best subset of environmental variables, so that the Euclidean distances of scaled environmental variables had the maximum Spearman rank correlation with the macrozoobenthic community dissimilarities. When Spearman rank correlation coefficients between a set of environmental variables exceeded 0.9, one of the variables was dropped (Zuur et al., 2007).

Generalised additive modelling (GAM) was executed with package 'mgcv' using Gaussian distribution and AIC values for selection of variables. Maps of biotic and abiotic variables were made using bicubic interpolation of package 'akima'. For all analyses, we used R: a language and environment for statistical computing, version 3.0.1 (R Core Team, 2013)

2.3 Results

High macrozoobenthic biomass was found just outside the 20 m isobath (Fig. 2.2 a–b and e–f). The highest biomass values were detected in 2006 near the Zeeland ridges in the southern part of the research area (Fig. 2.1, no. 3) and Fig 2.2 c–g).

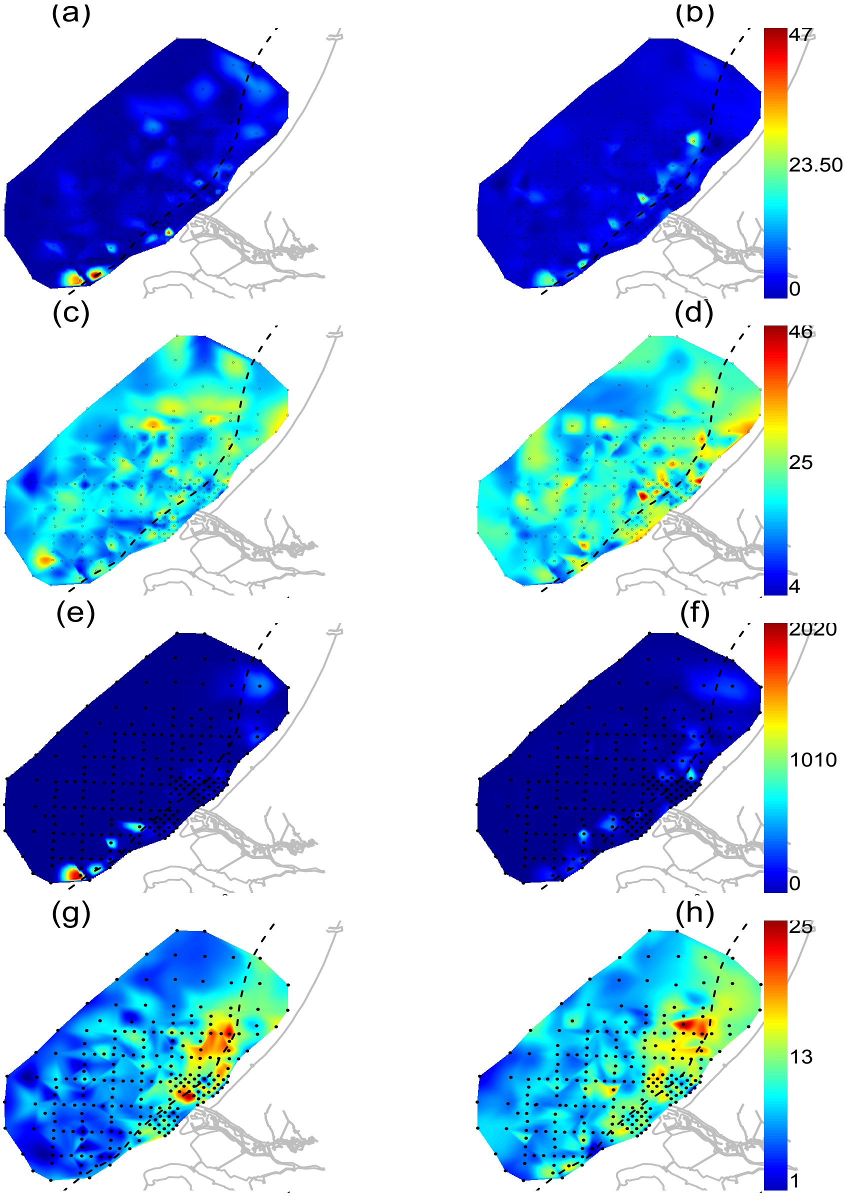


Figure 2.2 Measured biotic variables in 2006 (left panels) and 2008 (right panels), (**a–b**) infaunal ash-free dry weight (g AFDW m^{-2}), (**c–d**) infaunal species richness (number of species per sample), (**e–f**) epifaunal wet weight biomass (WW, g m^{-2}) and (**g–h)** epifaunal species richness. The 20 m isobath is depicted with a dashed line.

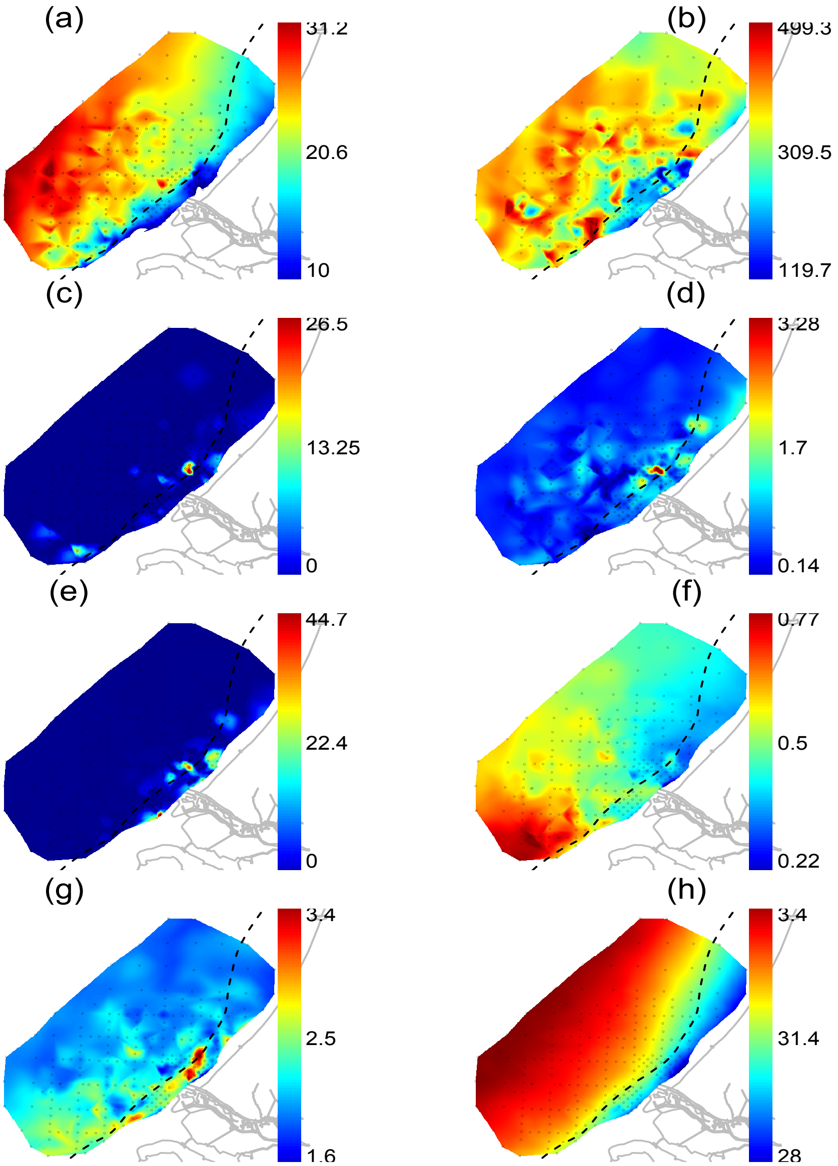


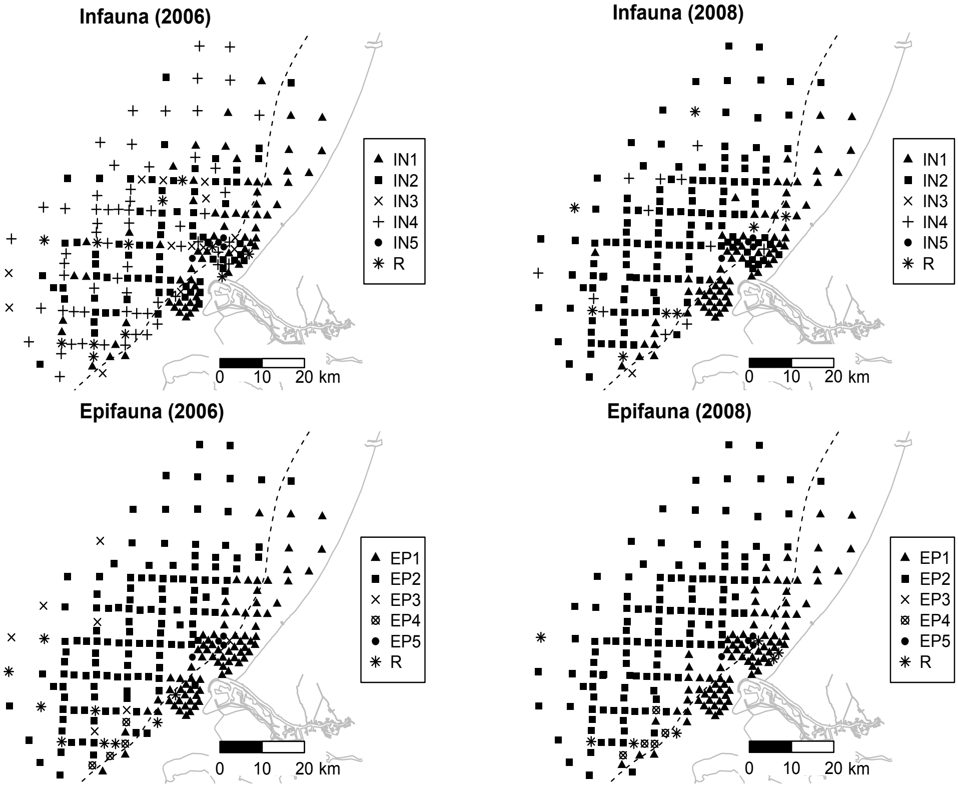
Figure 2.3 Measured abiotic variables in 2008 (**a**) water depth (m), (**b**) median grain size D_{50} (µm), (**c**) mud content(%), (**d**) OM (%), (**e**) very fine sand (%), (**f**) maximum bed shear stress (N m^{-2}), (***g***) mean bed shear stress (N m^{-2}) and (**h**) mean bed salinity (ppt) with 20 m isobath.

The largest D_{50} was found near the Zeeland ridges whereas in front of the PoR the smallest grain size and highest sediment mud content, OM content, and very fine sand fraction were found (Fig. 2.3 b–e). At disposal site 'North' (Fig 2.1 no. 8), the highest bed shear stress values were calculated due to decreased water depths and coarser sediment is found.

Fine sediment rich in mud and OM was found near the lowered disposal site (Fig. 2.1 no. 7) and also 20 km north of the PoR, which may be the result of effluent discharge of a sewage treatment installation (RWZI 'Houtrust', Fig. 2.1 no. 10). Highest mean bed shear stress was found in the southern part of the research area and near-bed salinity decreased in coastward direction. Mean and maximum near-bed salinity and water depth were strongly correlated in 2006 and 2008 with a correlation coefficient of 0.8-0.9 (Appendix 2.I).

Clustering and species composition of IN samples

Based on a 70% Bray-Curtis dissimilarity threshold, 18 infaunal species assemblages were distinguished. In total, 450 sampling locations were grouped into five main assemblages (Appendix 2.II). The remaining smaller assemblages are indicated with an R in Fig. 2.4. According to the SIMPROF analysis, all 18 infaunal assemblages were significantly different.


Infauna (2006)

Infauna (2008)

Epifauna (2006)

Epifauna (2008)

Figure 2.4 Upper panels: distribution of the main infaunal assemblages in 2006 and 2008 (IN1: *Echinoidea spp.–L. conchilega*; IN2: *Echinoidea* spp.–*Phoronida* spp.; IN3: *N. cirrosa*.–*S. bombyx*; IN4: *Spio spp.–N. cirrosa*, assemblage; IN5: ***O. fusiformis*–*A. alba***–*L. conchilega* and R: remaining locations). Lower panels: main epifaunal assemblages (EP1: *Ensis* spp.–*O. albida*–***S. subtruncata***; EP2: *Ensis* spp.–*O. albida*; EP3: *O. albida*; EP4: ***Ensis* spp.**–*O. albida*; EP5: ***A. alba***–*Actiniaria* spp.; and R: remaining locations). Indicator species are indicated in bold.

Infaunal assemblage 1 (IN1) was dominated by subsurface deposit-feeding sea urchins *Echinoidea* spp. and suspension-feeding sand mason worm *Lanice conchilega*. In 2006 and

2008, respectively, 47 and 72 sampling locations belonged to this assemblage. The average biomass of assemblage IN1 was 5.6 g AFDW m^{-2} with 24.5 species per boxcore (Appendix 2.II). Assemblage IN1 occurred also near the sewage effluent discharge location (Fig. 2.1, no.10). Assemblage IN2 was dominated by sea urchins *Echinoidea* spp. and suspension-feeding horseshoe worms *Phoronida* spp., and occurred 94 times in 2006 and 139 times in 2008 (1.6 g AFDW m^{-2}, 19.0 species per boxcore).

The most abundant species of assemblage IN3 were the predatory polychaete *Nephtys cirrosa* and the deposit-feeding polychaete *Spiophanes bombyx* (0.1 g AFDW m^{-2}, 11.7 species per sample). Assemblage IN3 occurred 13 times in 2006 and 1 time in 2008. Assemblage IN4, which showed a high species composition similarity with assemblage IN3, was dominated by the deposit-feeding polychaete *Spio* spp. and *N. cirrosa* (0.4 g AFDW m^{-2}, 14.5 species per boxcore) and occurred 64 times in 2006 and 12 times in 2008. IN2, IN3, and IN4 were found near and in the area of disposal site ‘North’ which received coarser sediment loads. IN4 assemblage was also found near the lowered disposal site which may be induced by recent fine sediment disposal and successive recolonisation. Assemblage IN5 was dominated by the deposit-feeding polychaete *Owenia fusiformis*, the deposit-feeding white furrow shell Abra alba and the sand mason worm. *Owenia fusiformis* and white furrow shell are the only Dufrêne–Legendre indicator species of the infaunal assemblages. Assemblage IN5 is exceptional because of the highest biomass and species richness (8.1 g AFDW m^{-2}, 26.4 species per boxcore). The assemblages were detected four times both in 2006 and in 2008 at identical locations near the lowered disposal site and in the deepened shipping lane ‘Euromaasgeul’ (Appendix 2.III; mud content: 15.5%, OM: 2.1%, and D_{50}: 157.1 µm). The specific part of the shipping lane was not recently dredged and no maintenance dredging was executed before sampling.

For IN in general, the best noticeable difference between the 2006 and 2008 samples was the increase in occurrence of IN2 assemblages (94 times in 2006 and 139 times in 2008) due to higher abundance of *Echinoidea* spp., *Phoronida* spp., and *L. conchilega*. These assemblages replaced the low-biomass and species-poor IN3 and IN4 assemblages inhabiting the deeper part of the study area (Fig. 2.4 and 2.5).
Biomass of assemblages IN1 and IN5 is significantly higher than that of assemblages IN2, IN3, and IN4 for both years (Kruskal–Wallis: $p < 0.05$). Biomass of assemblage IN2 is significantly higher than that of assemblages IN3 and IN4. Assemblage IN5 has the highest species richness followed by assemblage IN1 (24.5 species per boxcore), which is significantly higher than the richness of assemblages IN2, IN3, and IN4. Species composition, biomass and species richness are summarised in Appendix 2.II, accompanying data on sediment variables are summarised in Appendix 2.III.

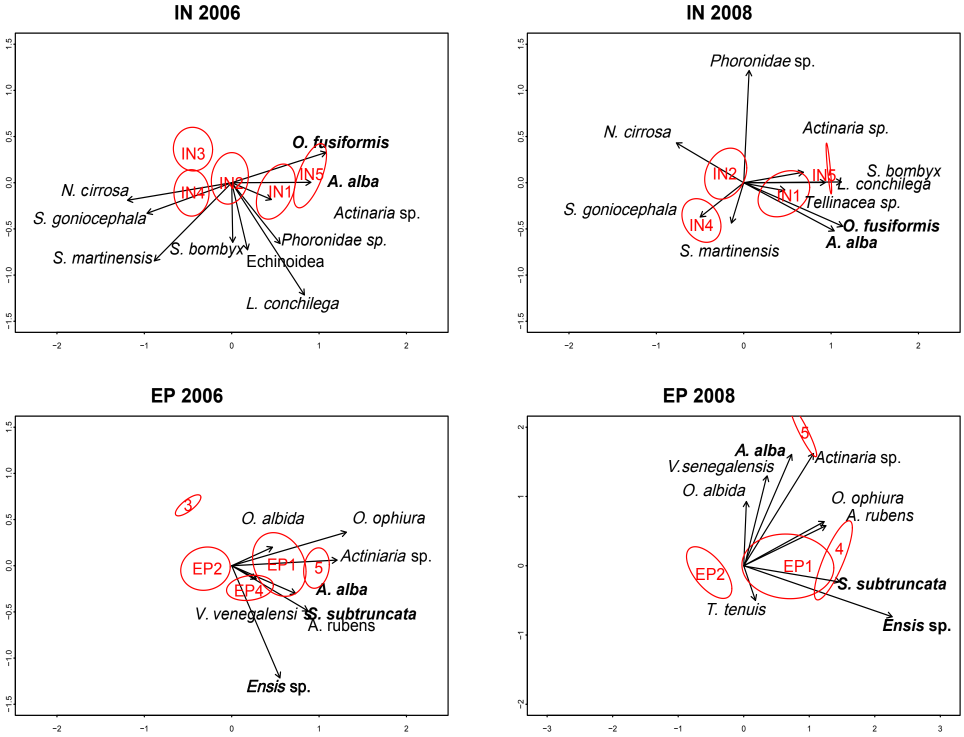


Figure 2.5 nMDS ordination with correlation of most abundant species and Dufrêne–Legendre indicators species (in bold) of determined assemblages depicted with centroids in red. Upper panels: distribution of the main infaunal assemblages (IN1: *Echinoidea spp.–L. conchilega*; IN2: *Echinoidea* spp.–*Phoronida* spp.; IN3: *N. cirrosa.–S. bombyx*; IN4: *Spio spp.–N. cirrosa*, assemblage; IN5: ***O. fusiformis–A. alba****–L. conchilega* and R: remaining locations). Lower panels: distribution of the main epifaunal assemblages (EP1: *Ensis* spp.–*O. albida–**S. subtruncata***; EP2: *Ensis* spp.–*O. albida*; EP3: *O. albida*; EP4: ***Ensis* spp.**–*O. albida*; EP5: ***A. alba***–*Actiniaria* spp.; and R: remaining locations). Only centroids of assemblages are used to maintain readability and in some cases assemblage names are shortened, individual sample points are depicted in Fig. 2.10. Dufrêne–Legendre indicator species are indicated in bold.

Clustering and species composition of EP samples

Ten species assemblages of EP (bottom sledge samples) were distinguished based on a threshold of 60% Bray–Curtis dissimilarity. According to the SIMPROF analysis, all assemblages were significantly different. In total, 455 epifaunal samples were grouped into five main assemblages. EP 1 (EP1) was characterised by the filter-feeding Dufrêne–Legendre indicator species cut trough shell *Spisula subtruncata*, *Ensis* spp. (filter-feeding bivalve), and the scavenging serpent’s table brittlestar *Ophiura albida* (Appendix 2.II). In 2006, 81 and in 2008, 91 samples belonged to assemblage EP1 with an average wet weight biomass of 95.4 g m^{-2} and 13.1 species per sample. Ensis spp. and the brittlestar were also the two most dominant species of assemblages EP2 and EP4. Assemblage EP2 had an average biomass of 15.8 g WW m^{-2} and 8.3 species per haul. Assemblage EP3 was unique by the low biomass and species richness (1.3 g WW m^{-2}, 3 species per haul). *Ophiura albida* was the most abundant species of EP. In 2006 and 2008, respectively, four and five from the troughs of the ridges (Fig. 2.1, no. 3) were grouped in assemblage EP4. This assemblage was unique by the high biomass, low species richness, and high *Ensis* spp. density (670.9 g WW m^{-2}, 6.4 species per haul, 85.8 ind. haul^{-1}). *Ensis* spp. was also indicated as a Dufrêne–Legendre indicator species for this assemblage. Assemblage EP5 contained Dufrêne–Legendre indicator species *A. alba*, the sea anemone (*Actiniaria* spp.), the serpent's table brittlestar (*O. albida*), the pullet carpet shell (*Venerupis senegalensis*), and the suspension-feeding blunt gaper (*Mya truncata*). Assemblage EP5 had the second highest biomass and species richness (139.1 g WW m^{-2}, 11.9 species per haul).

Assemblage EP5 was detected four times both in 2006 and in 2008 near the lowered disposal site for dredged sediment and in the deepened shipping lane 'Euromaasgeul'. For EP, the most noticeable difference between the 2006 and 2008 was the spatial replacement of low-biomass and species-poor assemblage EP3 by assemblage EP2 (Fig. 2.4 & 2.5). Biomass of epifaunal assemblages EP1, EP4, and EP5 is significantly higher than that of assemblages EP2 and EP3 (Kruskal–Wallis: $p < 0.05$). Species richness of assemblages EP1 and EP5 are significantly higher than those of assemblages EP2, EP3, and EP4 (Kruskal–Wallis: $p < 0.05$).

Species richness, biomass, and environmental variables

Infaunal species richness showed a negative correlation with maximum bed shear stress and showed a peak at a water depth of 20 m, a D_{50} of 200 µm, and OM content of 1.75% with 42.7% of the deviance explained (Table 2.1 & Fig. 2.6).

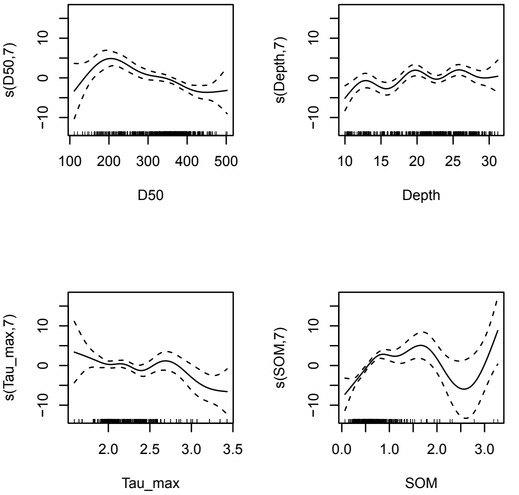


Figure 2.6 Response curves of infaunal species richness and significant continuous variables, dashed lines are 95% confidence intervals.

Infaunal biomass (log transformed) showed a negative correlation with D_{50} (with a peak at 200 µm), a water depth of 20 m, a OM content of 1%, and a mud content of 5%, explaining 34.3% of the deviance. At higher OM and mud contents, the effect on richness and biomass is not clear due to larger variation (Table 2.1 & Fig. 2.7).

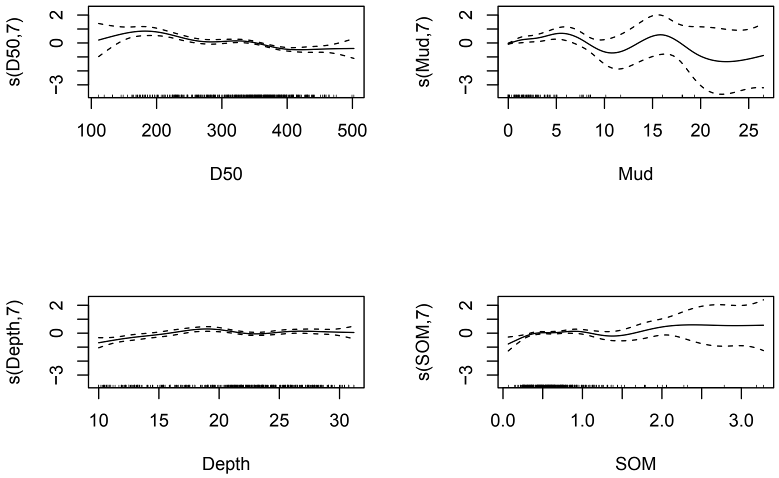


Figure 2.7 Response curves of infaunal biomass (log10 transformed) and significant continuous variables

Table 2.1 GAM output for infaunal species richness and biomass. Set of significant environmental variables, estimated degrees of freedom (edf), Akaike Information Criterion (AIC) and percentage of deviance.

Variable	edf	*p*-value	AIC	Explained deviance
Infaunal species richness				
OM	7	***	2880.4	42.7%
D_{50}	7	***		
Depth	7	***		
Maximum bed shear stress	7	*		
Mean bed shear stress	7	.		
Infaunal biomass (AFDW log10 transformed)				
D_{50}	7	***	909.8	34.3%
Mud	7	**		
Depth	7	***		
OM	7	*		

Species richness of EP showed a negative correlation with mean bed shear stress (peak at a value of 0.4 $\mathrm{N\ m^{-2}}$), a water depth of 20 m, a median grain size of 200 µm, and a maximum shear stress of 2.25 $\mathrm{N\ m^{-2}}$ (Fig. 2.8). A significant difference in species richness between 2006 and 2008 emerged from the GAM analysis (Table 2.2). The average species richness in 2006 was 9.1 and in 2008 10.6 species haul^{-1}. The model explained 56.8% of the deviance.

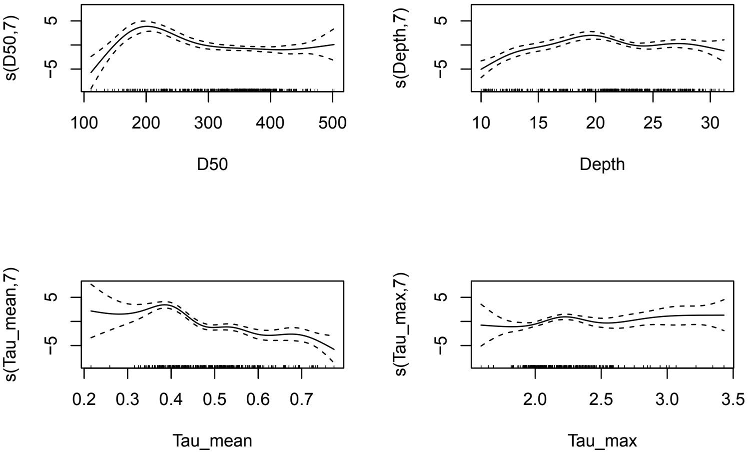


Figure 2.8 Response curves of epifaunal species richness and significant continuous variables

Biomass is lower at a finer grain size with a peak at 200 µm, a water depth of 20 m, 5% mud content, and 0.75% OM. Again, above 5% mud and 0.75% OM, the effect on biomass is not clear (Fig. 2.9). Above a mean shear stress value of 0.7 N m^{-2}, biomass strongly decreases. Also for biomass, a significant difference was present between the two years. In 2006, on average 53.9 and in 2008, 63.6 g WW haul^{-1} was found. The environmental variables in the GAM regression explained 56.8% of the deviance (Table 2.2).

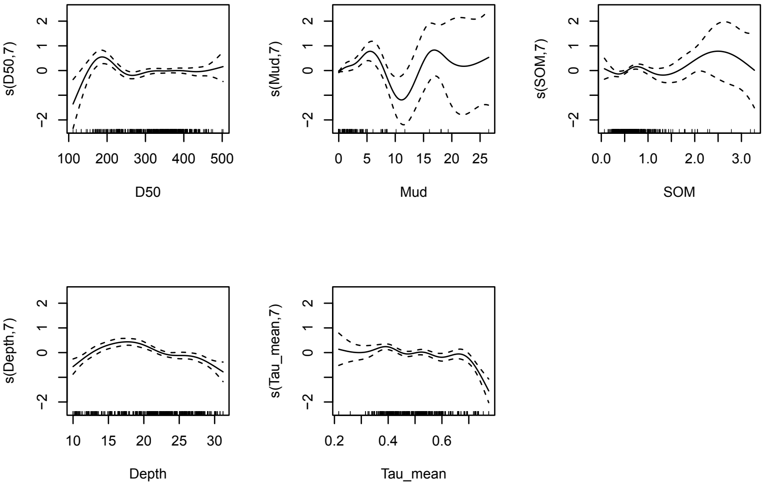


Figure 2.9 Response curves of epifaunal biomass (log10 transformed) and significant continuous variables

Table 2.2 GAM output for epifaunal species richness and biomass. Set of significant environmental variables, estimated degrees of freedom (edf), Akaike Information Criterion (AIC) and percentage of deviance.

Variable	edf	*p*-value	AIC	Explained deviance
Epifaunal species richness				
Mean bed shear stress	7	***	2341.1	56.8%
Depth	7	***		
D_{50}	7	***		
Maximum bed shear stress	7	**		
Year	1	***		
Biomass epifauna (WW log10 transformed)				
Depth	7	***	746.7	46.8
Mean bed shear stress	7	***		
Mud	7	***		
D_{50}	7	***		
OM	7	**		
Year	1	*		

Variables explaining macrozoobenthos distribution

The defined macrozoobenthic assemblages grouped together in the nMDS ordination (Fig. 2.5 & 2.10). For IN, ordination stress with four dimensions was ~0.16 in both years and for EP ~0.12 in 2006 and 0.09 in 2008 (Table 2.3).

Table 2.3 Stress values of two-, three- or four-dimensional nMDS ordinations

Variable	Two-dimensional	Three-dimensional	Four-dimensional
IN 2006	0.25	0.20	0.16
IN 2008	0.25	0.19	0.16
EP 2006	0.21	0.15	0.12
EP 2008	0.20	0.13	0.11

For the distribution of IN in 2006, depth, grain size, OM, mean bed shear stress, and mud were selected with the bioenv function with a combined Spearman's rank correlation of 0.37 (Table 2.4, Fig. 2.10). In 2008, grain size and mean near-bed salinity were selected with a combined Spearman's rank correlation of 0.50. Infaunal assemblage IN1 (*Echinoidea* spp.–*L. conchilega*) was positively correlated with shallow water depth, low near-bed salinity, and higher levels of mud, OM, and very fine sand. Assemblage IN2 (*Echinoidea* spp.–*Phoronida* spp.) showed a correlation with intermediate near-bed salinity, depth, and D_{50} (Table 2.3, Figure 2.8). Assemblage IN3 (*N. cirrosa*–*S. bombyx*) correlated in areas without mud. Assemblage IN4 (*Spio* spp.–*N. cirrosa*) was more abundant in 2006 than in 2008 and generally correlated with highest values of D_{50}, bed shear stress and near-bed salinity.

Assemblage IN5 (*Owenia fusiformis–Abra alba*) was found in the deepened shipping lane 'Euromaasgeul' and disposal site and correlated with the highest percentages of mud, OM, and very fine sand, smallest D_{50}, largest water depth, and smallest bed shear stress.

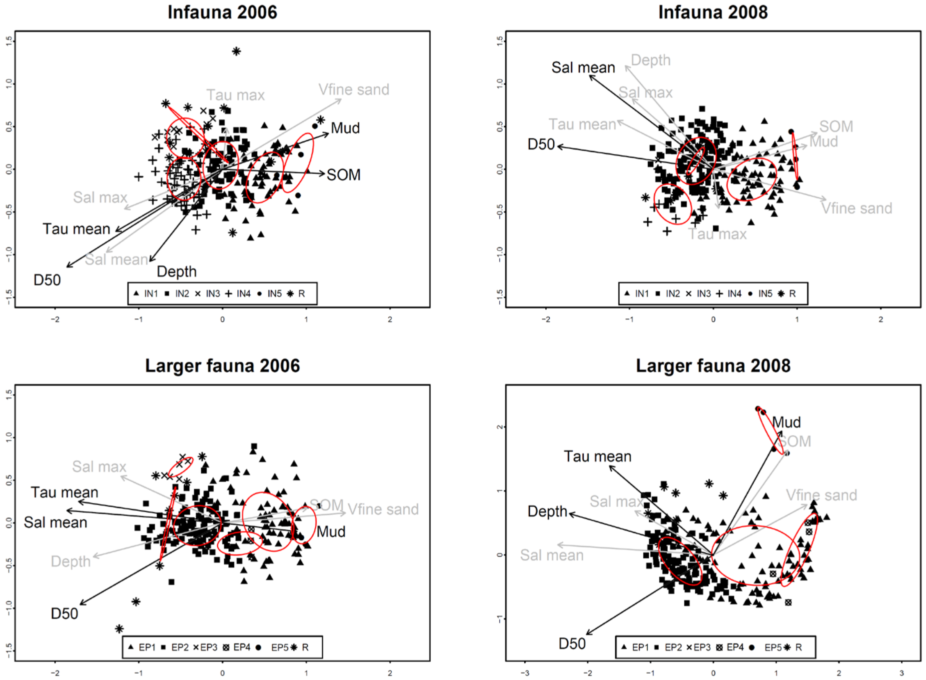


Figure 2.10 nMDS ordination with the first two axis and correlation of all significant environmental variables superimposed in grey (IN1: *Echinoidea spp.–L. conchilega*; IN2: *Echinoidea* spp.–*Phoronida* spp.; IN3: *N. cirrosa.–S. bombyx*; IN4: *Spio spp.–N. cirrosa*, assemblage; IN5: ***O. fusiformis–A. alba***–*L. conchilega* and R: remaining locations). Lower panels: distribution of the main epifaunal assemblages (EP1: *Ensis* spp.–*O. albida*–***S. subtruncata***; EP2: *Ensis* spp.–*O. albida*; EP3: *O. albida*; EP4: ***Ensis* spp.**–*O. albida*; EP5: ***A. alba***–*Actiniaria* spp.; and R: remaining locations). Correlation arrows in bold are the variables selected with the bioenv function

Table 2.4 Multiple regressions of environmental variables and infaunal nMDS scores for four-dimensional ordination , r2 is the squared Spearman's rank correlation coefficient, r2 combined is the squared Spearman's rank correlation coefficient of the best subset, and Pr(>r) is the *p*-value. Significance codes: *$p < 0.05$; ** $p < 0.01$; *** $p < 0.001$. Variables in bold are the subset variables selected with the bioenv function.

Variable	NMDS1	NMDS2	NMDS3	NMDS4	r2	Pr(>r)	r2 combined
2006							
Depth	**-0.44**	**-0.52**	**0.40**	**0.62**	**0.37**	***	**0.37**
D_{50}	**-0.82**	**-0.49**	**-0.01**	**0.31**	**0.59**	***	
OM	**0.99**	**-0.03**	**-0.11**	**0.09**	**0.18**	***	
Mean near-bed salinity	-0.69	-0.47	0.40	0.38	0.41	***	
Maximum near-bed salinity	-0.74	-0.29	0.54	0.27	0.23	***	
Mean bed shear stress	**-0.85**	**-0.49**	**0.05**	**0.20**	**0.26**	***	
Maximum bed shear stress	0.07	0.71	-0.38	-0.59	0.05	*	
Mud	**0.91**	**0.32**	**-0.21**	**0.18**	**0.22**	***	
Very fine sand	0.76	0.43	-0.36	-0.32	0.36	***	
2008							
Depth	-0.56	0.67	-0.18	-0.43	0.54	***	**0.50**
D_{50}	**-0.84**	**0.11**	**-0.21**	**-0.48**	**0.68**	***	
OM	0.9	0.31	0.29	-0.04	0.33	***	
Mean near-bed salinity	-0.75	0.55	-0.26	-0.26	0.63	***	
Maximum near-bed salinity	**-0.70**	**0.60**	**-0.28**	**-0.24**	**0.32**	***	
Mean bed shear stress	-0.84	0.42	-0.19	-0.29	0.32	***	
Maximum bed shear stress	0.07	-0.63	0.52	0.58	0.09	***	
Mud	0.86	0.21	0.41	-0.24	0.27	***	
Very fine sand	0.90	-0.24	0.36	0.06	0.38	***	

Percentages mud, OM, and very fine sand are the highest for assemblages IN1 and IN5 and are significantly higher compared with other assemblages (Kruskal-Wallis: $p < 0.05$). D_{50} of assemblage IN5 is significantly lower than for assemblages IN2, IN3, and IN4 (Kruskal-Wallis: $p < 0.05$). Mean bed shear stress is lower for assemblages IN1 and IN5, respectively, 0.43 and 0.37 $N\ m^{-2}$, than for assemblages IN2 and IN4, both above 0.5 $N\ m^{-2}$ Kruskal-Wallis: $p < 0.05$). Values of environmental variables of infaunal assemblages are given in Appendix 2.IV. In 2006, the distribution of EP correlated with grain size, mean near-bed salinity, mean bed shear stress, and the fraction of mud and very fine sand. In 2008, EP correlated with depth, grain size, mean bed shear stress, and mud. The combined Spearman's rank correlation was, respectively, 0.42 and 0.52 (Table 2.5, Fig. 2.10).

Table 2.5 Multiple regressions of environmental variables and epifaunal nMDS scores for four-dimensional ordination, r^2 is the squared Spearman's rank correlation coefficient, r^2 combined is the squared Spearman's rank correlation coefficient of the best subset, and Pr(>r) is the *p*-value. Significance codes: *p < 0.05; ** p < 0.01; *** p < 0.001. Variables in bold are the subset variables selected with the bioenv function.

Variable	NMDS1	NMDS2	NMDS3	NMDS4	r2	Pr(>r)	r2 combined
			2006				
Depth	-0.61	-0.15	0.76	0.19	0.50	***	**0.42**
D_{50}	**-0.87**	**-0.50**	**-0.01**	**-0.05**	**0.51**	***	
OM	0.63	0.13	0.55	0.53	0.20	***	
Mean near-bed salinity	**-0.77**	**0.06**	**0.63**	**0.04**	**0.56**	***	
Maximum near-bed salinity	-0.70	0.32	0.59	-0.24	0.29	***	
Mean bed shear stress	**-0.87**	**0.15**	**0.45**	**0.14**	**0.44**	***	
Maximum bed shear stress	0.15	0.63	-0.60	-0.46	0.05	*	
Mud	**0.72**	**-0.01**	**0.44**	**0.55**	**0.18**	***	
Very fine sand	**0.84**	**0.07**	**-0.22**	**0.50**	**0.33**	***	
			2008				
Depth	**-0.72**	**0.54**	**-0.02**	**-0.43**	**0.63**	***	**0.52**
D_{50}	**-0.80**	**0.17**	**0.37**	**0.45**	**0.57**	***	
OM	0.49	0.28	-0.07	-0.82	0.37	***	
Mean near-bed salinity	-0.89	0.41	-0.10	-0.17	0.70	***	
Maximum near-bed salinity	-0.84	0.49	-0.21	-0.13	0.33	***	
Mean bed shear stress	**-0.93**	**-0.36**	**0.07**	**0.04**	**0.39**	***	
Maximum bed shear stress	0.25	-0.69	-0.49	0.47	0.18	***	
Mud	**0.40**	**0.30**	**0.09**	**-0.87**	**0.31**	***	
Very fine sand	0.57	-0.17	0.18	-0.78	0.39	***	

For EP, assemblage EP1 (*Ensis* spp.-*Ophiura* spp.-*S. subtruncata*) was positively correlated with higher percentages mud, OM, and very fine sand, and negatively correlated with near-bed salinity, bed shear stress and water depth (Fig. 2.5). Assemblage EP2 (*Ensis* spp.-*O. albida*) showed a negative correlation with mud, OM, and very fine sand, and a positive correlation with near-bed salinity, bed shear stress, and depth. Assemblage EP3 (*O. albida*) occurred in areas characterised by an absence of mud and very fine sand and high bed shear stress. Assemblage EP4 (*Ensis* spp.) was found at the Zeeland ridges in shallow waters with coarse sediments (360.7 µm) and was further characterised by high mean shear stress values. Assemblage EP5 (*A. alba-Actinaria spp.-O. albida-V. senegalensis*) responded in a similar way as infaunal assemblage IN5 and was again found in the deepened shipping lane and disposal site. Percentages mud, OM and very fine sand were highest for assemblage EP5, significantly higher than the values of the other assemblages except for assemblage EP1 (Kruskal-Wallis: p < 0.05).

Mean bed shear stress is significantly lower for assemblage EP1 and EP5 (0.42 and 0.37 N m^{-2}) compared to the other assemblages (>0.50 N m^{-2}). Values of environmental variables of epifaunal assemblages are given in Appendix 2.V, a graphical summary of the characteristics of the assemblages and relationships with abiotic and hydrographic variables is presented in Fig. 2.11.

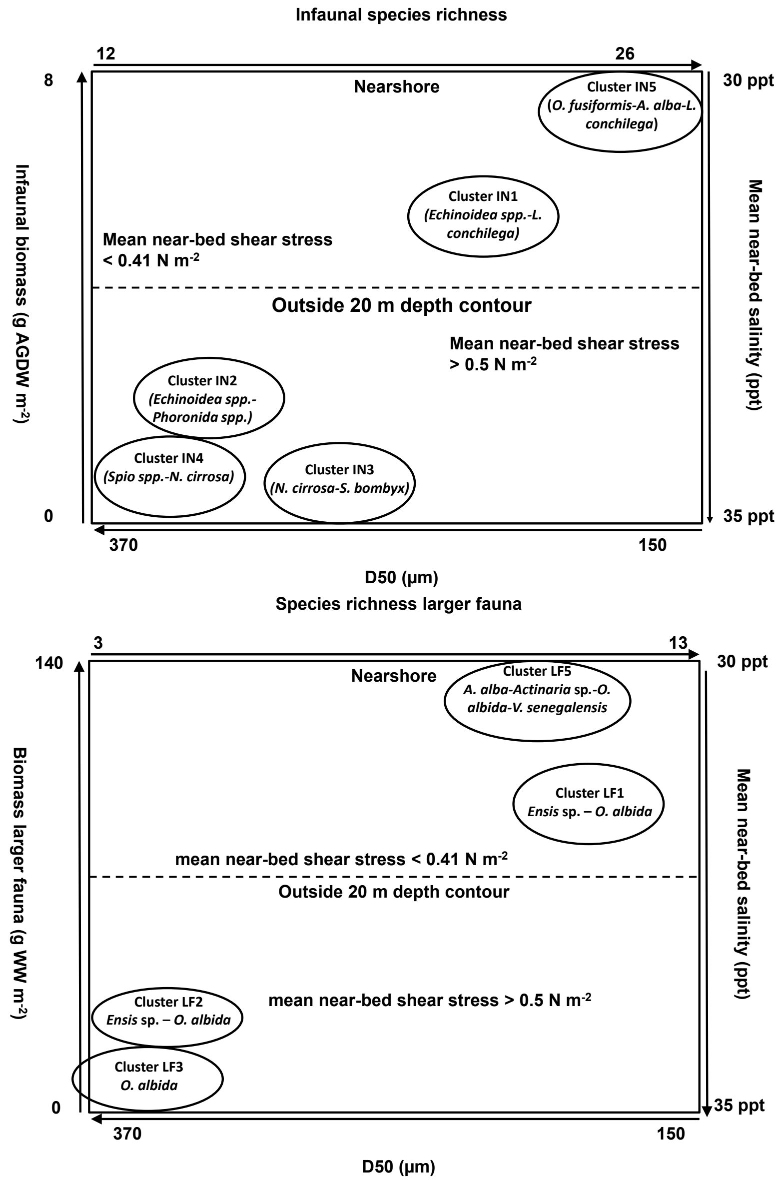


Figure 2.11 Summary of infaunal and epifaunal assemblages with biotic variables in relation to grain size and mean near-bed salinity and bed shear stress variables. Assemblage EP4 is not depicted in the graph due to its extreme combination of abiotic and biotic variables.

2.4 Discussion

The distribution of infauna (IN) in 2006 correlated with water depth, grain size, organic matter (OM), mean bed shear stress, and mud. In 2008, grain size and mean near-bed salinity were selected. The highest species richness was found for assemblage IN5 (*O. fusiformis–A. alba*) in areas with 15.4% mud and 2.1% OM. At these sampling locations, an increase from 21.5 species per sample in 2006 to 31.2 species per sample in 2008 was observed. On the Belgian Continental Shelf (BCS), van Hoey et al. (2004) found that species richness was positively related to sediment mud content and sediment median grain size and negatively related to distance to the coast. Samples with high mud values (> 20%) showed reduced species richness (15–30 instead of 30–50 species). OM was not measured in their study. Species richness in the German bight was found to be negatively correlated with sediment mud content, total organic carbon (TOC), and fishing effort on the short term (Reiss et al., 2009). On the DCS, species richness was found to be negatively correlated with fish trawling intensity, sediment grain size, and primary productivity and positively correlated with biomass (van Denderen et al., 2014).

The risk of reduced species richness from total organic carbon loading is relatively low at TOC values below 10 mg g^{-1}, high at values over 35 mg g^{-1} and intermediate at in between values (Hyland et al., 2005). Our results showed lower species richness at OM values exceeding 3% (30 mg g^{-1}). The highest infaunal biomass was observed in the shipping lane and near the lowered disposal site with sediment with 15.4% mud content. An increase in biomass of infaunal assemblage IN5 from 6.4 g AFDW m^{-2} in 2006 to 9.9 g AFDW m^{-2} in 2008 was observed. Heip et al. (1996) found that infaunal biomass increased consistently in finer sediments and sediments with higher chlorophyll a content. Reiss et al. (2010) found that infaunal biomass in part of the German Bight was positively related to sediment mud content and OM and negatively related to fishing effort. In their study, the highest infaunal biomass (20 g AFDW m^{-2}) was detected at a location with the highest mud content (24%). Total infaunal biomass and *A. alba* biomass were found to be positively correlated with OM (Thiebaut et al., 1997). In the south-western Baltic Sea, large variation in biomass was found for white furrow shell *A. alba*, one of the dominant species of this assemblage (Rainer, 1985). The higher biomass and species richness around the 20 m isobath may also be induced by sand extraction and sediment disposal sites. Although studies on the DCS showed that macrozoobenthos returned within 4–6 years to pre-extraction conditions (van Dalfsen et al., 2000; van Dalfsen and Essink, 2001), in figure 2.2 & 2.3, deviations are visible around the 20 m isobath which overlap with shallow borrow pits, fine sediment disposal site ‘Northwest’, and sediment disposal site ‘North’.

Environmental variables and epifauna (EP)

In 2006, the distribution of epifauna (EP) correlated with grain size, mean near-bed salinity, mean bed shear stress, and the fraction of mud and very fine sand. In 2008, EP correlated with depth, grain size, mean bed shear stress, and mud. Reiss et al. (2010) found relationships with hydrographic variables such as water temperature, near-bed salinity, and wave stress. In our study, epifaunal species richness showed a negative correlation with mean bed shear stress and showed a peak at a value of 0.4 N m^{-2}, a water depth of 20 m, a median grain size of 200 µm and at a maximum shear stress of 2.25 N m^{-2}. Callaway et al. (2002) concluded that epibenthic species richness was negatively correlated with mud content, seabed temperature and beam trawl effort.

Epibenthic biomass was found to be high along the continental coast and a limited number of free-living species was responsible for this pattern (Callaway et al., 2002). In shallow parts of the North Sea along the continental coast, the starfish *Asterias rubens* and the brittlestars *Ophiura albida* and *Ophiura ophiura* were abundant (Callaway et al., 2002). A similar trend was found in our study with high biomass values along the coast with also brittlestars being abundant. In our study, razor clam (*Ensis* spp.) was the most abundant species, which was also found by Tulp et al. (2011).

Number of assemblages

Five infaunal and epifaunal assemblages were distinguished using clustering techniques and confirmed by the nMDS ordinations. A variety of macrozoobenthic assemblages were earlier distinguished within the Southern Bight in the North Sea and the English Channel (Degraer et al., 1999; Künitzer et al., 1992; Duineveld et al., 1991; Holtmann et al., 1996; Desroy et al., 2003; Kröncke et al., 2011; Ghertsos et al., 2000; Govaere et al., 1980). On the BCS, which is most comparable to our research area, four main infaunal assemblages were distinguished in the most recent study: (i) a muddy fine sand *Abra alba–Mysella bidentata* assemblage characterised by high density and species richness; (ii) a *Nephtys cirrosa* assemblage that occurred in well-sorted sandy sediments and is characterised by low densities and species richness; (iii) an assemblage with very low densities and species richness typified by the *Ophelia limacinae–Glycera lapidum* community, which is found in coarse sandy sediments (van Hoey et al., 2004). The fourth macrobenthic assemblage is typical for the upper intertidal zone of sandy beaches and beyond the scope of our study.

We distinguished a comparable 'muddy fine sand' assemblage consisting of *Owenia fusiformis*, *A. alba*, and *Kurtiella bidentata* near the lowered disposal site and in the 'Euromaasgeul' shipping lane with a similar macrobenthic density and species richness (8250 ind. m^{-2}, 31.2 species m^{-2}); the median grain size was smaller, and sediment mud content and OM content were higher. Dumping of dredged fine harbour sediment took place a few hundred metres to the south, maintenance dredging activities were not executed in the last years in the specific sampling area of the shipping lane (Rijkswaterstaat, pers. comm.). A similarity between impacts of sand extraction and sediment disposal was also found on the BCS (de Backer et al., 2014)

Biomass values of comparable *A. alba* assemblages in the Eastern English Channel-southern North Sea in water depths < 15 m were 8.1 g AFDW m^{-2} (Desroy et al., 2003). In the Bay of Seine, biomass was around 25 g AFDW m^{-2} mainly due to polychaetes and echinoderms (Thiebaut et al., 1997) and in Gravelin biomass exceeded values of 100 g AFDW m^{-2} (Ghertsos et al., 2000). Biomass values of comparable *A. alba* assemblages in the Eastern English Channel-southern North Sea in water depths smaller than 15 m were 8.1 g AFDW m^{-2} (Desroy et al., 2003). In the Bay of Seine, France, biomass was around 25 g AFDW m^{-2} mainly due to polychaetes and echinoderms (Thiebaut et al., 1997) and in Gravelin (France) biomass exceeded values of 100 g AFDW m^{-2} (Ghertsos et al., 2000).

We also distinguished a similar *N. cirrosa* assemblage with comparable densities and species richness. This assemblage was also found in earlier research on the DCS (Holtmann et al., 1996; Duineveld et al., 1991; Künitzer et al., 1992). Kröncke et al. (2011) found differences in an *N. cirrosa* assemblage between 1986 and 2000 (assemblages F2 and L2) in the eastern North Sea and in the central Southern Bight to be induced by an increase in the density of Phoronids, the polychaetes *S. bombyx*, *L. conchilega*, *Magelona* spp., and the amphipod *Urothoe poseidonis*, whereas the density of the amphipod *Bathyporeia* spp. and the polychaete *Ophelia borealis* decreased in 2000. Assemblages IN1 and IN2 did not match with an defined assemblage from the literature, but shows some similarity with the *Lanice conchilega* assemblage determined by Degraer et al. (1999).

We found five epifaunal assemblages, whereas Callaway et al. (2002) found three epifaunal assemblages in the southern North Sea using a 2 m beam trawl at water depths < 50 m (Callaway et al., 2002). No similarities in the assemblage are found presumably due to differences in spatial scales and differences in mesh size of the studies. In shallow parts of the North Sea along the continental coast, the starfish *Asterias rubens* and the brittlestars *Ophiura albida* and *Ophiura ophiura* were abundant (Callaway et al., 2002). Epifaunal biomass ranged between 500–1000 near shore and between 1000–8840 g WW m^{-2} at the Zeeland ridges, whereas species richness ranged between 6 and 19. Aggregations of *Ensis* spp. were found in the troughs of the Zeeland ridges.

Interannual differences in assemblage distribution and the NAO winter index

Differences in macrozoobenthic assemblage distributions were found. In 2008, more infaunal samples were grouped as assemblages IN1 and IN2 due to higher density of subsurface deposit-feeding sea urchins *Echinoidea* spp., horseshoe worms *Phoronida* spp., and suspen-

sion-feeding sand mason worms *Lanice conchilega*. These assemblages replaced the low-biomass and species-poor assemblages IN3 and IN4 inhabiting deeper water. The higher densities may be the result of successful recruitment during two successive mild winters after the strong winter of 2006. The Northern Atlantic Oscillation (NAO) winter index was 1.1, 2.9, and 2.1 in 2006, 2007, and 2008, respectively (Hurrell, 2012).
Several species such as *L. conchilega*, *A. alba*, *Bathyporeia* spp., *Urothoe poseidonis*, and *Spiophanes bombyx* increased in density and distribution between 1986 and 2000, which was associated with an increase in sea surface temperature, primary production or food supply, and positive NAO winter index values (Kröncke et al., 2011). Beukema (1985) reported mass mortality of *Echinocardium cordatum* in the Dutch coastal zone during the severe winter of 1979 (NAO winter index: 1.43). Kirby et al. (2007) found evidence that the elevated North Sea temperature after 1987 (NAO winter index: 0.77) favoured the reproduction and survival of *E. cordatum*.

Regarding EP, we found an increase in *Ensis* spp. density in 2008 (e.g. for epifaunal assemblage EP1, 8.4 ind. m^{-2} in 2006 to 22.6 ind. m^{-2} in 2008). This was also observed by Tulp et al. (2010), who noticed an increase in Dutch coastal waters (10 ind. m^{-2} in 2006 to 18 ind. m^{-2} in 2007 and 2008). Low winter temperatures have been shown to negatively affect recruitment success in *Ensis directus* (Dannheim and Rumohr, 2012; Tulp et al., 2010). Epibenthos showed strong seasonal fluctuations due to the severe winter of 1996 (NAO winter index = 1.43), resulting in strong differences in density of the serpent’s table brittlestar *O. albida* and the common starfish *A. rubens* and showing a gradual decrease in density in the following decade (Neumann et al., 2009). We found the highest density of *O. albida* in 2006 (e.g. epifaunal assemblage EP4, 14.15 ind. m^{-2} in 2006 to 1.15 ind. m^{-2} in 2008).

Relevance for management

For human activities, effects on assemblages, biomass and species richness of macrozoobenthos can be predicted with the detected relationships with environmental and hydrographical variables. The information on the distribution patterns and species composition of macrozoobenthic assemblages can function as a baseline for future comparisons and help in the conservation of marine biodiversity. Autonomous temporal variation in species composition has to be taken into account in the assessment of impact of future human activities. This study can be of value for EIAs and in the assessment of Marine Strategy Framework Directive’s (MSFD) Good Environmental Status (GES) descriptors biodiversity, sea-floor integrity, hydrographical conditions, and food webs.

The occurrence of the *A. alba* assemblages near the lowered disposal site for dredged fine sediment and in the deepened shipping lane of the Port of Rotterdam may be an indication that the benthic system can be changed by human activities. Median grain size is considerably smaller; sediment mud and OM content are higher. Macrozoobenthic species composition is significantly different compared with the surrounding assemblages. Smaller shifts in species composition, species richness and biomass may be induced by activities such as shallow (2 m) sand extraction, disposal of coarse dredged sediment and sewage treatment effluent discharges.

Relevance for monitoring activities

Macrozoobenthos was analysed with a boxcorer and bottom sledge. The advantage of a boxcorer is the combination of infauna and sediment sampling. Collecting data about the occurrence of *Ensis* spp. is only possible with the bottom sledge due to their fast escape behaviour. On the other hand, the fragile *Echinocardium cordatum* is severely damaged in the epifaunal sampling procedure with the bottom sledge and gets undistinguishable. Biomass estimates from the bottom sledge can therefore be severely underestimated. To maintain a full coverage of data, a combination of boxcore and bottom sledge data is recommended.

2.5 Conclusion

Highest macrozoobenthic species richness and biomass in the Dutch coastal zone in front of Rotterdam were observed at locations with a water depth of 20 m, a median grain size of 200 µm, elevated mud and OM values, and low mean bed shear stress. Five in- and epifaunal assemblages were distinguished and correlations were found with measured sediment variables and modelled hydrographic variables. Interannual differences in macrozoobenthic assemblage distribution were found resulting from more Echinoids, Phoronids and razor clams in 2008.

A distinct, highly-productive and species-rich macrozoobenthic deposit-feeding white furrow shell *Abra alba* assemblage coincided near a disposal site for dredged fine sediment and in an 8 m deepened shipping lane which may be an indication that the benthic system can be changed by these activities. Smaller changes in macrozoobenthic species composition may be linked to shallow sand extraction, disposal of coarse sediment, and to the discharge of sewage effluent. Modelled bed shear stress is an explanatory variable in addition to sediment variables in explaining distribution patterns in macrozoobenthos.

2.6 Acknowledgements

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Chapter 3

Short-term impact of deep sand extraction and ecosystem-based landscaping on macrozoobenthos and habitat characteristics

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Abstract

We studied short-term changes in macrozoobenthos in a 20 m deep borrow pit with 20 m sand extraction depth. A boxcorer was used to sample macrozoobenthic infauna and a bottom sledge was used to sample macrozoobenthic epifauna. Sediment characteristics were determined from the boxcore samples and bed shear stress and near-bed salinity were estimated with a hydrodynamic model. Two years after the cessation of sand extraction, macrozoobenthic biomass increased 5-fold in the deepest areas. Species composition changed significantly and white furrow shell (*Abra alba*) became abundant. Several sediment characteristics also changed significantly in the deepest parts. Macrozoobenthic species composition and biomass significantly correlated with time after cessation of sand extraction, sediment and hydrographical characteristics. Ecosystem-based landscaped sandbars were found to be effective in influencing sediment characteristics and macrozoobenthic assemblage. Significant changes in epifauna occurred in deepest parts in 2012 which coincided with the highest sedimentation rate. We recommend continuing monitoring to investigate medium and long-term impacts of deep sand extraction.

3.1 Introduction

The demand for marine sand in the Netherlands and internationally is rising (Stolk and Dijkshoorn, 2009; ICES, 2014a). In the Netherlands, 26 million m^3 of marine sand is used annually for coastal nourishments and construction (Stolk and Dijkshoorn, 2009; ICES, 2014a). An increase of annual nourishments from 12.5 up to 40–85 million m^3 for counteracting effects of future sea level rise is anticipated (Deltacommissie, 2008). The potential for deep sand extraction was first investigated in 1999 in a deep temporary extraction site (PUTMOR) in front of the Port of Rotterdam (PoR). 6.5 Million m^3 of sand was extracted in an area with an initial water depth of 23 m and extraction depths between 5–12 m. There were no indications that deep sand extraction would lead to unacceptable effects and recovery of benthic assemblages could be possible (Boers, 2005). For a 20 km^2 seaward harbour expansion Maasvlakte 2 of the PoR the Dutch authorities permitted extraction deeper than the common 2 m extraction depth, primarily to decrease the surface area of direct impact. Approximately 220 million m^3 of sand was extracted between 2009–2013, with an average extraction depth of 20 m. To maintain sufficient supply of marine sand in the intensively used coastal zone, the authorities started promoting deeper sand extraction for future sand extraction projects larger than 10 million m^3 of sand (IDON, 2014).

Ecological impacts of deep sand extraction, however, are largely unknown and still under investigation. In general, sand extraction has direct impacts on the seabed since benthic organisms are damaged or removed and the bathymetry and seabed composition is changed considerably. Indirect effects are increased turbidity, release of nutrients or toxins and smothering by sedimentation. In 1979, the first studies on the impacts of large-scale dredging in the North Sea were published (de Groot, 1979b; de Groot, 1979a). Since then, many international studies have investigated various ecological aspects of shallow sand extraction (Newell et al., 1998; Seiderer and Newell, 1999; van Dalfsen et al., 2000; Desprez, 2000; Boyd et al., 2003; Boers, 2005; Boyd et al., 2005; Newell et al., 2004; Barrio Froján et al., 2008; Desprez et al., 2009; Le Bot et al., 2010; de Backer et al., 2014). The recovery time of benthic assemblages to pre-dredge conditions after shallow sand extraction in the North Sea is estimated to be 4–6 years (van Dalfsen et al., 2000; Boyd et al., 2005; van Dalfsen and Essink, 2001). In a region with high-intensity sand and gravel extraction off the south-east coast of England, higher macrozoobenthic variability was observed and complete recovery was not reached 11 years after the cessation of extraction (Wan Hussin et al., 2012). For muddy areas, however, Newell et al. (1998) estimated that recovery time is around 6–8 months. The recovery time of benthic assemblages after deep and large-scale sand extraction is not yet determined.

In this study, we report on the short-term (0–2.5 y) changes in macrozoobenthos in the deep and large-scale borrow pit for Maasvlakte 2. As an experiment, two sandbars were excavated copying naturally occurring bedforms to increase habitat heterogeneity and benthic species richness (Baptist et al., 2006; van Dijk et al., 2012; de Jong et al., 2014). We tested the hypothesis that deep and large-scale sand extraction and ecosystem-based landscaping approaches lead to significant changes in macrozoobenthic assemblage. Furthermore, we hypothesise that the macrozoobenthic assemblage in the Maasvlakte 2 (MV2) borrow pit will resemble the highly productive and species-rich macrozoobenthic white furrow shell (*Abra alba*) clusters which were found in 2006 and 2008 in the 8 m deepened shipping lane and the lowered disposal sites for dredged fine sediment in front of the PoR (de Jong et al., 2015a).

We aim to answer the following questions:
(i) What are the short-term (0–2.5 y) effects of deep sand extraction on macrozoobenthos?
(ii) What are the changes in environmental variables and bathymetry?
(iii) Which environmental and hydrodynamical variables are influencing macrozoobenthos?

3.2 Materials and methods

Borrow pit and surrounding area

The Maasvlakte 2 (MV2) borrow pit is situated in front of the Rotterdam harbour outside the 20 m isobath and is 6 km long and 2 km wide (Fig. 3.1b). The research area stretches over approximately 100 km^2 in front of the PoR (Fig. 3.1b). The seabed sediment consists of fine to medium sand with small quantities of mud, very fine sand and organic matter (OM).

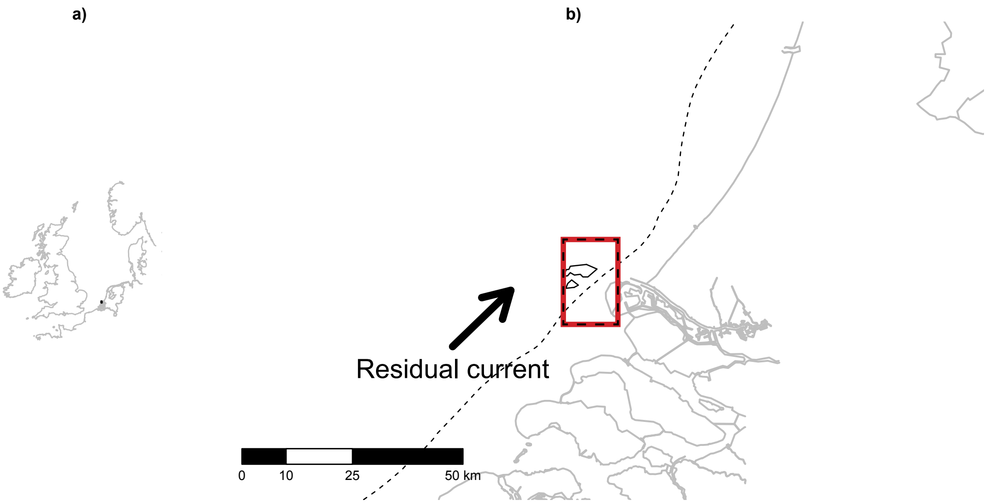

Figure 3.1 (**a**) North Sea, (**b**) Dutch coastal zone with Maasvlakte 2 (MV2) borrow pit and harbour extension with 20 m isobath (black dashed line), residual tidal current and survey area (black and red coloured dashed rectangle).

Approximately 220 million m^3 of sand was extracted between 2009–2013 with an average extraction depth of 20 m. A large northern and smaller southern borrow pit is separated by an exclusion area (Fig. 3.2).

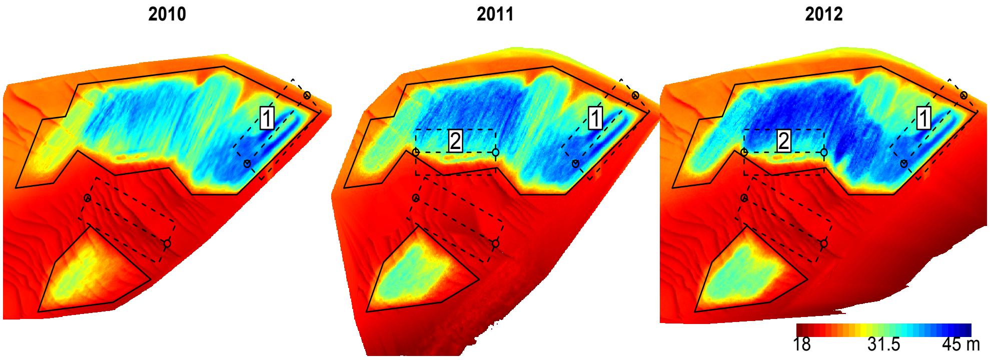

Figure 3.2 Bathymetry of MV2 borrow pit during the sand extraction with the parallel sandbar (1) and the oblique sandbar in 2011 and 2012 (2). The dashed rectangles are the two-dimensional projections of the sand waves (Fig. 3.3), the parallel sandbar denoted with 1 (Fig. 3.4) and the oblique sandbar denoted with a 2 (Fig. 3.5).

Sand waves are present in the surrounding area, with wave lengths of 100–800 m, amplitudes up to 5 m and crests orientated perpendicular to the tidal current (Hulscher, 1996). In the exclusion area, amplitudes are around 4 m (Fig. 3.2 and 3.3).

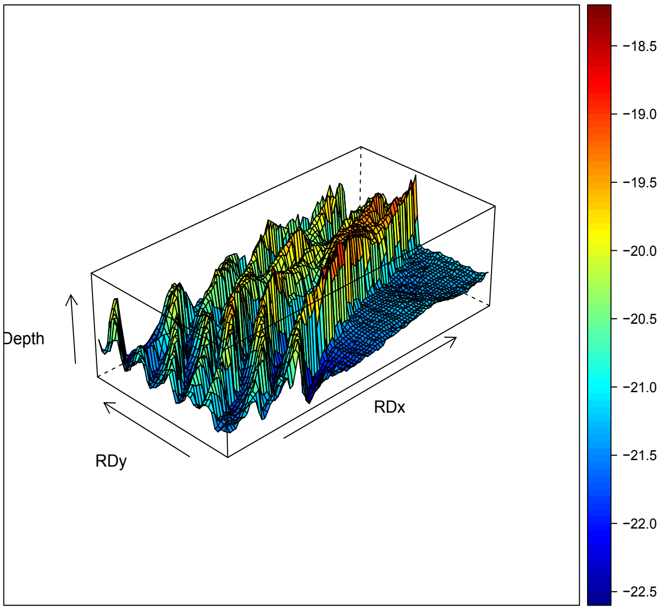

Figure 3.3 Natural sand wave field in the exclusion area of the MV2 borrow pit from NW direction

Macrozoobenthos in the North Sea correlates with a variety of sediment characteristics (Heip et al., 1992; Degraer et al., 2008; van Hoey et al., 2007; Künitzer et al., 1992; Holtmann et al., 1996; Verfaillie et al., 2009; van Hoey et al., 2004). Near-bed salinity affected epifauna (EP) in the North Sea (Callaway et al., 2002; Reiss et al., 2011; Reiss et al., 2010), whereas bed shear stress affected macrozoobenthos in intertidal areas (Herman et al., 2001; Ysebaert et al., 2003; Puls et al., 2012). Naturally occurring bed forms, such as sand waves and shoreface-connected ridges influence macrozoobenthos on smaller spatial scales (Baptist et al., 2006; van Dijk et al., 2012).

To investigate the applicability of ecosystem-based landscaped sandbars in sand extraction projects, two sandbars were excavated on the seabed of the extraction site. One sandbar parallel to the tidal current was completed in spring 2010 (Fig. 3.2: denoted with 1 and Fig. 3.4). The parallel sandbar has a length of 700 m, a width at the crest of 70 m and slopes of 140 m length. The crest of the sandbar is located at a water depth of 30 m and the troughs are more than 40 m deep. In 2011, the second sandbar was completed with an orientation oblique to the tidal current (Fig. 3.2 denoted with 2 and Fig. 3.5). The length and width are similar to the parallel sandbar but, due to time constraints, the difference in depth between crest and trough is less pronounced. The crest is situated at a water depth of 28 m and the northern trough is 36 m deep. A narrow and 32 m deep trench separates the crest from the slope of the borrow pit.

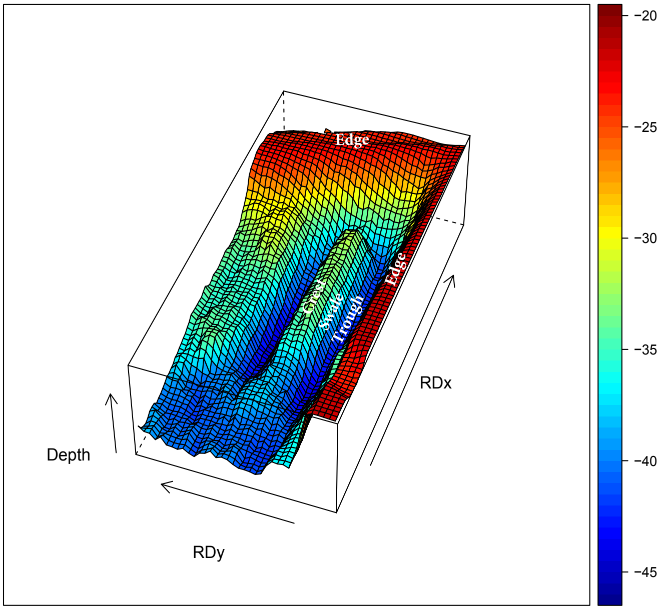

Figure 3.4 Parallel sandbar in the MV2 borrow pit with the edge of the borrow pit and sample locations: crest, slope and trough.

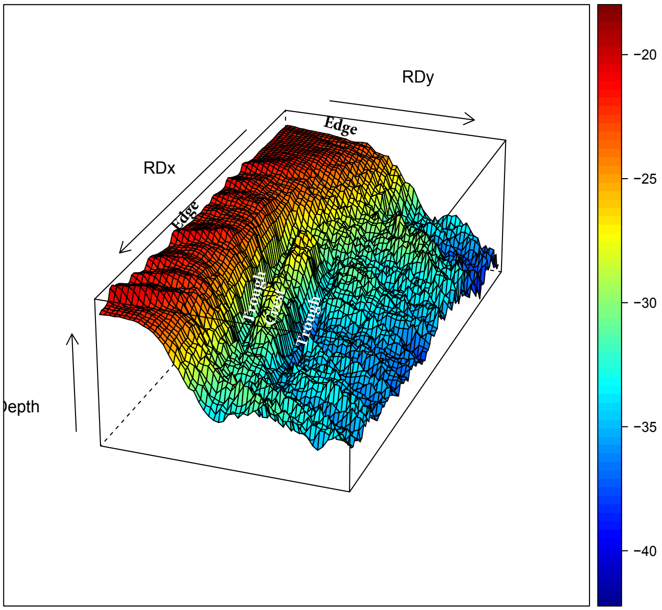

Figure 3.5 Oblique sandbar in the MV2 borrow pit with the southern edge of the borrow pit and sample locations: crest and troughs.

Macrozoobenthos and sediment sampling

To ensure comparable data, sampling was carried out using identical protocols as during the baseline study (chapter 2) and the current recolonisation study of the Environmental Impact Assessment (EIA) for the construction of MV2. A boxcorer with a surface area of 0.077 m^2 was used to sample sediment and macrozoobenthos, larger than 1 mm and mainly living in the seabed. A bottom sledge was used to sample macrozoobenthic in- and epifauna with a size range of 0.5–10 cm. Bottom sledge samples are hereafter called epifauna (EP) although large infauna is collected as well (Perdon and Kaag, 2006a; Craeymeersch and Escaravage, 2010; de Jong et al., 2015a). Sampling with the boxcorer was executed by the Royal Netherlands Institute for Sea Research (NIOZ) on 29–30th June 2010, 2–5th May 2011 and 23–25th April 2012. In 2010 and 2011, 45 and in 2012, 64 boxcore samples were collected. To reach a higher spatial resolution in- and outside the extraction site in 2012, a subsample of the boxcore was analysed which reduced the sampled surface area to 0.015 m^2. Four samples were collected in the deep parts of the extraction site, 14 samples in the reference area (near and far field) and four samples in the shipping lane area (Table 3.1, fig. 3.6). No maintenance dredging was executed in the shipping lane according to Rijkswaterstaat. Specimens were identified up to species level when possible and ash-free dry weight biomass (g AFDW m^{-2}) was analysed by means of loss on ignition, 2 days at 80 °C followed by 2 hours at 580 °C.

Table 3.1 Number and location of boxcore samples. Locations are visualised in Fig. 3.6.

	Reference					Borrow pit							
						Parallel			Oblique		Deep		
Year	-	Cr	Tr	Ed	Sh	Cr	Slo	Tro	Cr	Tr	SE	R	To
2010	-	5	7	13	-	5	6	6	-	-	1	2	45
2011	-	4	4	15	-	4	-	4	4	4	2	4	45
2012	9	4	4	12	4	4	-	3	4	4	7	9	64

-: Undefined, Cr: Crest, Tr: Trough, Ed: Edge, Sh: shipping lane, Slp: Slope, SE: South East, R: Rest and To: Total

Sampling with the bottom sledge for epifauna was conducted by the Institute for Marine Resources & Ecosystem Studies (IMARES Wageningen UR) on 7–8th July 2010, 14–15th June 2011 and 6–7th June 2012. The bottom sledge was equipped with a 5 mm mesh cage. On average, a surface area of 15 m^2 was sampled during each sledge haul of approximately 150 m length, 10 cm width and a maximum penetration depth of 10 cm. In 2010 and 2011, 26 and in 2012, 32 bottom sledge samples were collected. In 2010, 11 bottom sledge samples were collected in the reference area (Table 3.2, fig. 3.6). In 2012, three samples were collected in the shipping lane area of the PoR and one sample in the reference area. Specimens were identified up to species level when possible. Wet weight of epifauna was directly measured after sorting (g m^{-2} WW). Biomass of razor clam *Ensis* sp. was determined by using regression equations based on previous IMARES field surveys (Craeymeersch and van der Land, 1998).

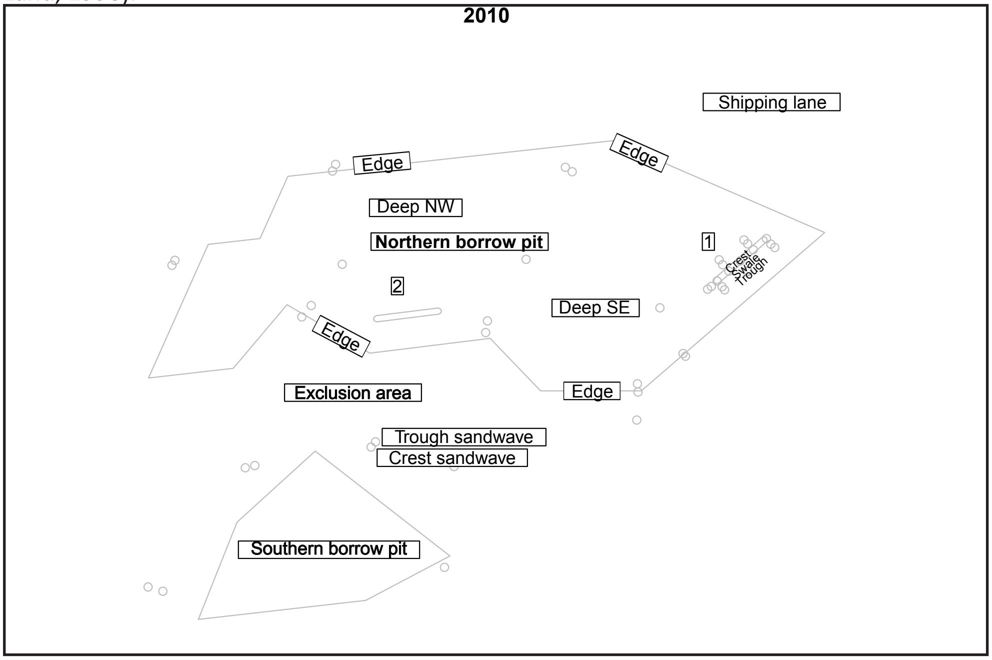

Figure 3.6 Southern and northern borrow pit, with the sampling locations and boxcore sampling positions in 2010. The parallel sandbar (1) with sub locations crest, swale and trough and the oblique sandbar (2) which was not sampled in 2010.

Table 3.2 Number and location of bottom sledge samples. Locations are visualised in Fig. 3.6.

	Reference					Borrow pit						
						Parallel			Oblique		Deep	
Year	Cr	Tr	Sl	Sh	Cr	Sl	Tr	Cr	Tr	SE	R	To
2010	-	1	10	-	2	-	4	2	4	1	2	26
2011	5	1	5	3	2	-	4	2	4	3	3	32
2012	9	4	4	12	4	4	-	3	4	4	7	9

-: Undefined, Cr: Crest, Tr: Trough, Ed: Edge, Sh: shipping lane, Slp: Slope, SE: South East, R: Rest and To: Total

In- and epifaunal biomass values are measured using different protocols (AFDW and wet weight) and are therefore not inter-comparable. Our short-term macrozoobenthic data can be compared with 2006–2008 baseline data and future medium and long-term data of PoR's EIA data. The two sampling campaigns (boxcore and bottom sledge) can be considered as complementary (de Jong et al., 2015a) and the response of infaunal and epifaunal species to disturbances may be different. The ships GPS-system logged the position of the sampling locations. Bathymetric multibeam data were collected by the dredging companies (resolution 20 x 20 m in 2010, 1 x 1 m in 2011 and 2.5 x 2.5 m in 2012) with reference level lowest astronomical tide using ‘Rijksdriehoek’ coordinates (Dutch coordinate system, RD). Closets sediment sample locations were linked to the epifaunal samples. Sediment analysis was identical as in the previous chapter (see also de Jong et al., 2015a).

Modelling of abiotic variables

The MV2 borrow pit and surrounding area has complex hydrographic conditions due to the region of fresh water input (ROFI) from the Rhine, periods of strong haline stratification, up- and downwelling, wind-driven flow, baroclinic cross-shore flows and wind and wave-induced mixing occur frequently (de Boer et al., 2009; van Wiechen, 2011). These complex conditions may lead to considerable fluctuations in salinity and bottom shear stress. We used Delft 3D DD ZUNO, a hydrodynamic model of the southern North Sea consisting of a coarse curvilinear horizontal grid with two grid refinement towards the Dutch coast through domain decomposition (DD). The nested model grid covers an area of about 7.5 km by 7.0 km and the horizontal grid size is about 45 m by 38 m for the research area. To generate appropriate boundary conditions for the nested model (Maasvlakte 2 borrow pit), four open boundaries were used with tangential velocity (Tonnon et al., 2013a). Riemann type boundary conditions were specified within the overall ZUNO DD model (Ye and Morelissen, 2011). Twelve vertical σ-layers were specified, the relative thickness of these has been chosen in such way that near-bed and near-surface vertical gradients were better resolved. From top to bottom, these layers represent respectively 4.0, 5.6, 7.8, 10.8, 10.9, 10.9, 10.9, 10.9, 10.8, 7.8, 5.6, and 4.0% of the water depth. The bathymetry of the nested model (MV2 borrow pit) was interpolated from the multibeam measurements performed by the dredging companies in October 2010. The model was forced with measured meteorological and riverine discharge data for the specific period. In order to keep the calculation time manageable, one single spring-neap cycle with relatively high river discharges was used for validation. The period between 2 March and 17 March 2007 was simulated using a time step of 15 seconds. Mean and maximum values of bed shear stress (N m^{-2}) and near-bed salinity (ppt) were modelled (Appendix 3.VIII).

Time after the cessation of sand extraction

The time after cessation of sand extraction is determined using bathymetric multibeam data. We distinguished areas with 0.5, 1, 1.5, 2, 2.5 and 3 year after the cessation, $t_{0.5}$, t_1 etc. are used. For sampling directly after sand extraction t_{recent} is used. The shipping lane of the PoR is labelled with $t_{not\ recent}$. Surrounding areas of the extraction site without direct influence of sand extraction are labelled as reference or exclusion area. In 2012, four measurement locations in the shipping lane of the Port of Rotterdam were sampled.

Statistical analyses

For all analyses we used R, version 3.0.1 (R Core Team, 2013). Shapiro–Wilk test, Bartlett's test and Levene's test were used to check for normality and homogeneity of univariate variables. When the assumptions for ANOVA were violated, the non-parametric Kruskal–Wallis one-way multi-comparison tests of package 'pgirmess' was used to determine significant differences in biotic and abiotic variables between sub locations. Significance of differences in macrozoobenthic species composition between location and time after cessation of sand extraction was tested with permutational multivariate analysis of variance using distance matrices (ADONIS) of package 'vegan'. Due to the lack of post-hoc multi-comparison tests in the ADONIS function, we manually selected sets of locations and analysed each comparison. We applied Non-Metric Dimensional Scaling (nMDS) using the metaMDS function in package 'vegan', based on Bray–Curtis dissimilarities of macrozoobenthos density data, to visualize differences in benthos assemblages in the extraction site and reference area (Oksanen, 2013). 3-Dimensional (3D) ordinations were used due to high stress values. The bioenv.default function was used to determine the best subset of continuous and categorical environmental variables, so that the Euclidean distances of scaled environmental variables have the maximum Spearman rank correlation with the macrozoobenthic community dissimilarities. The subset of environmental variables were linearly fit onto the 2D ordinations using the ENVFIT function in package 'vegan' (999 permutations). When Spearman rank correlation coefficients between a set of variables exceeded 0.9 one of the variables was dropped (Zuur et al., 2007). Package 'Marmap' version 9.0.2 was used to make the bathymetric 3D plots and setSQL and subsetSQL was used to load the high resolution bathymetry data of 2011 and 2012 (Pante and Simon-Bouhet, 2013). We used package 'mgcv' for the GAM analyses using Gaussian distribution. We selected the best explaining variables with the Akaike information criterion (AIC). When a set of variables was strongly correlated we dropped one of them. We checked the assumptions for GAM with the gam.check function and used log10-transformed biomass values.

3.3 Results

Abiotic data

Decreasing median grain size and increasing mud values were observed in the borrow pit (Fig. 3.7 a–f, Appendix 3.II). Sediment organic matter was only measured in 2012 but shows a similar distribution as for mud (Fig. 3.7 f and g). The conditions in the deep parts of the borrow pit are very similar to those of the 1970's deepened shipping lane area. The circumstances in the shipping lane itself remained very similar during the monitoring campaign. Water depth at the sampling site within the shipping lane in 2006, 2008 and 2012 was, respectively, 29.6, 29.0 and 26.8 m (Appendix 3.II). Median grain size decreased from 228.8 to 193.9 and 135.1 µm and mud content changed from 7.5 to 2.5 and 23.1 vol% and organic matter (OM) from 1 to 1.4 to 4.7 mass%. Due to ongoing sand extraction in the borrow pit, only one t_2 location in the north-western deep part was sampled. The sediment characteristics in this area are similar to the t_2 samples from the south-eastern deep part. The high value of mud and OM on the crest of the oblique sandbar (Fig. 3.2, 2), however, was not related to sedimentation of fines but to remains of old peat or wood fragments. In 2010, significant higher very fine sand values were detected on the parallel sand bar (Fig. 3.2, 1) compared to the reference area (Kruskal–Wallis < 0.05). In 2012, very fine sand and mud values differed significantly between the shipping lane area and the edge and the reference area and between the deep south eastern part, edge and reference area (Kruskal–Wallis < 0.05). The largest water depth was found in the south-eastern trough of the parallel sandbar and was respectively 44.7, 44.4 and 43.2 m in 2010, 2011 and 2012. Between 4 October 2010 and 4 August 2011, the sedimentation rate in the troughs of the parallel sandbar was relatively small (Table 3.3, N: +0.16 and S: +0.05 m). The deep SE area and the crest of the parallel sandbar eroded slightly (-0.02 m and -0.10 m).

The sedimentation rate in the troughs of the parallel sandbar between 4 August 2011 and 20 July 2012, increased considerably (N: +0.43 and S: +0.75 m). In the deep SE area 0.18 m of sediment settled and 0.27 m on the crest of the parallel sandbar.

Table 3.3 Water depth (m) and sedimentation rate at sub locations in the borrow pit (-: erosion, +: sedimentation)

	Location			
Year	Deep SE	Northern trough par. sandbar	Crest par. sandbar	Southern trough par. sandbar
2010	-41.04	-41.89	-35.76	-41.55
2011	-41.06 (-0.02)	-41.73 (+0.16)	-35.87 (-0.10)	-41.50 (+0.05)
2012	-40.88 (+0.18)	-41.30 (+0.43)	-35.60 (+0.27)	-40.75 (+0.75)

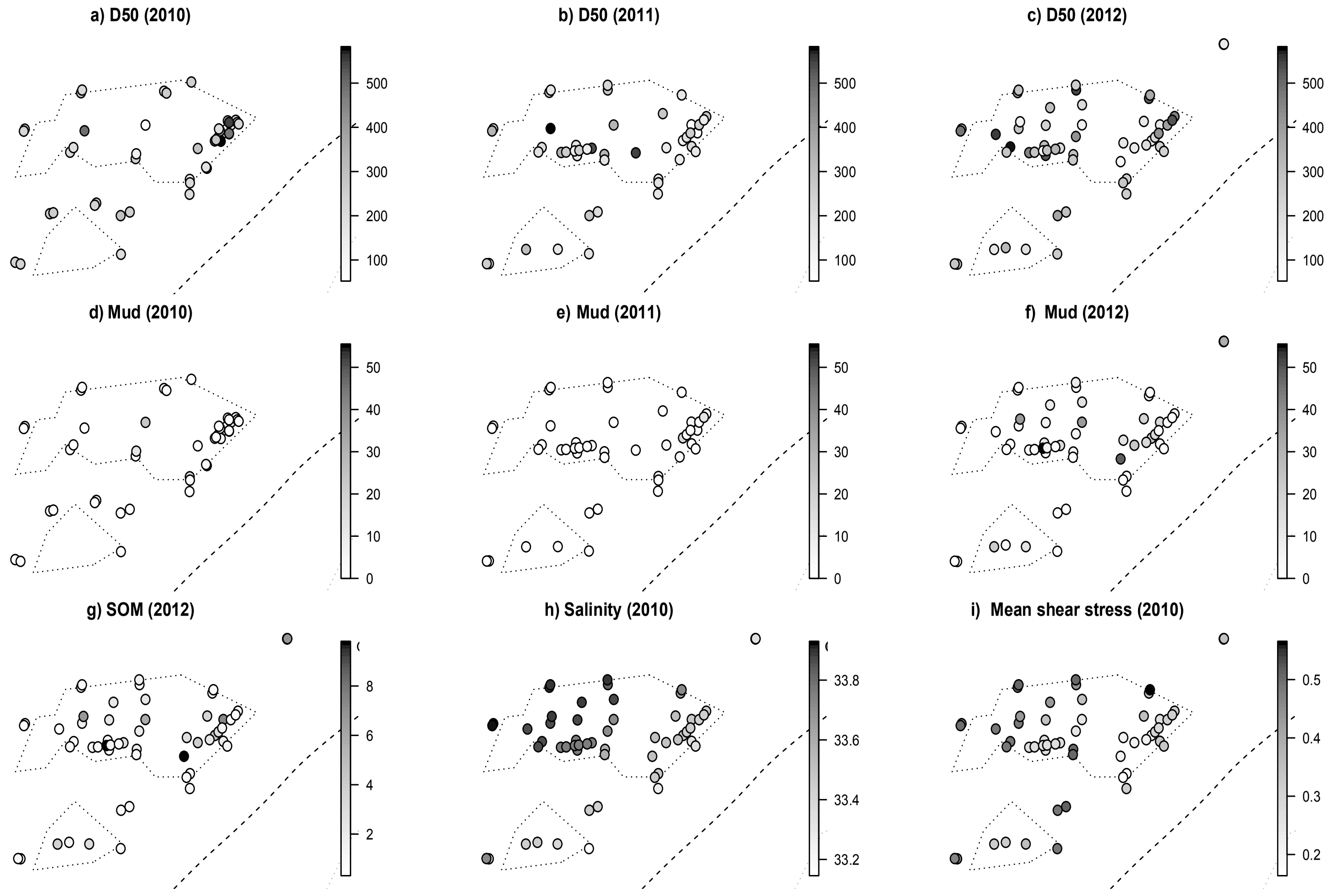

Figure 3.7 Environmental variables. (**a–c**): Median grain size (D50, µm), (**d–f**): Fraction mud (vol%), (**g**): OM for 2012 (mass%), (**h-i**): Mean near–bed salinity (ppt) and mean bed shear stress (N m^{-2}) based on 2007 weather conditions and 2010 bathymetry

Infaunal data

Highest infaunal (IN) species richness was found in the reference area in 2010 (Fig. 3.8a). Species richness in 2010 was significantly lower at the crest and swale of the parallel sandbar when compared to the reference area (Kruskal–Wallis, $p < 0.05$). Species richness in the extraction site in 2012 was similar to the reference area. Directly after sand extraction, species richness in the troughs of the sandbars was significantly lower compared to reference level, respectively, 3.7 instead of 8 species per haul. Comparisons of species richness in the shipping lane in 2006 and 2008 with 2012 is not possible due to the smaller samples sizes obtained in 2012. Species richness peaked at median grain sizes of around 200 µm and 18 m water depth, time after cessation also contributed significantly and the generalised additive model (GAM) explained 40.3% of the total deviance (Fig. 3.9 lower panels, Appendix 3.V, Table 3.2).

Infaunal biomass decreased in the reference area during the monitoring campaign and the 3-year average was 16.1 g AFDW m^{-2} (Appendix 3.IV). Directly after sand extraction, biomass values in the borrow pit are generally significantly lower compared to reference values. In 2012, high infaunal biomass values were detected in the deep parts of the extraction site (Fig. 3.8 panels d–i). Biomass values were significantly higher in the deep SE part of the borrow pit when compared to the edge (Kruskal–Wallis, $p < 0.05$). In 2011, biomass values in the trough of the oblique sand bar were significantly lower than reference values. In the deep SE part in 2012, biomass reached high levels (maximum: 165.7 g m^{-2} AFDW) mainly due to white furrow shell (Fig. 3.8 d and i). Values were significantly higher compared to the reference area and the edge (TukeyHSD, $p < 0.05$). In the middle of the crest of the oblique sandbar, another hotspot in infaunal biomass emerged. The average biomass value was significantly higher than the value at the edge (TukeyHSD, $p < 0.05$). Biomass of the crests of the parallel and oblique sandbars differed, respectively, 14.9 and 51.1 g AFDW m^{-2}. Infaunal biomass showed a significant negative correlation with median grain size and time after sand extraction, which explained 48.5% of the total deviance (Fig. 3.9 upper panel, Appendix 3.V Table 1).

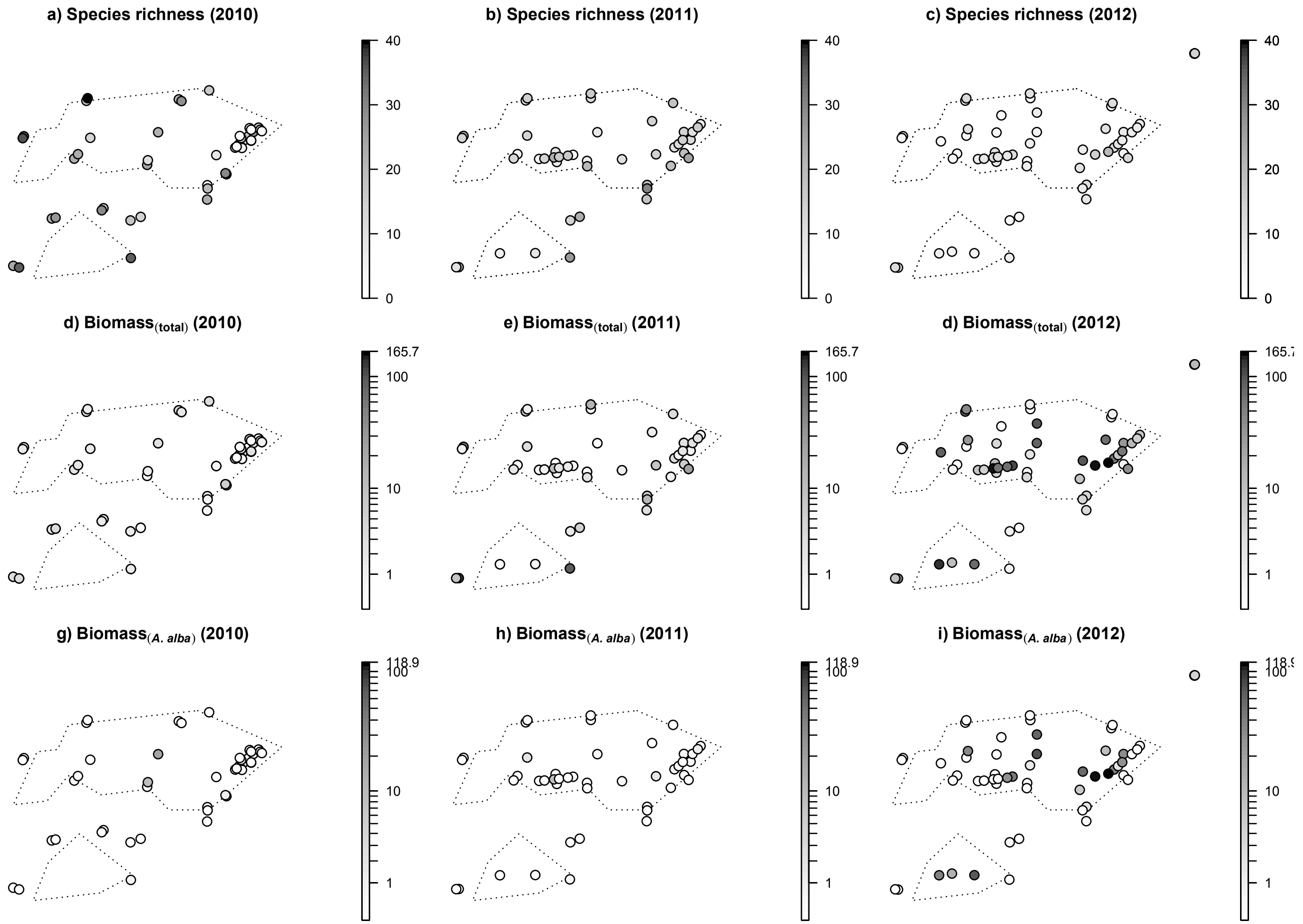

Figure 3.8 Biological parameters of boxcore sampling for 2010, 2011 and 2012. (**a–c**) Species richness (n species boxcore^{-1}), (**d–f**) Total biomass boxcore (g AFDW m^{-2}) and (**g-i**) Biomass *A. alba* boxcore (g AFDW m^{-2}).

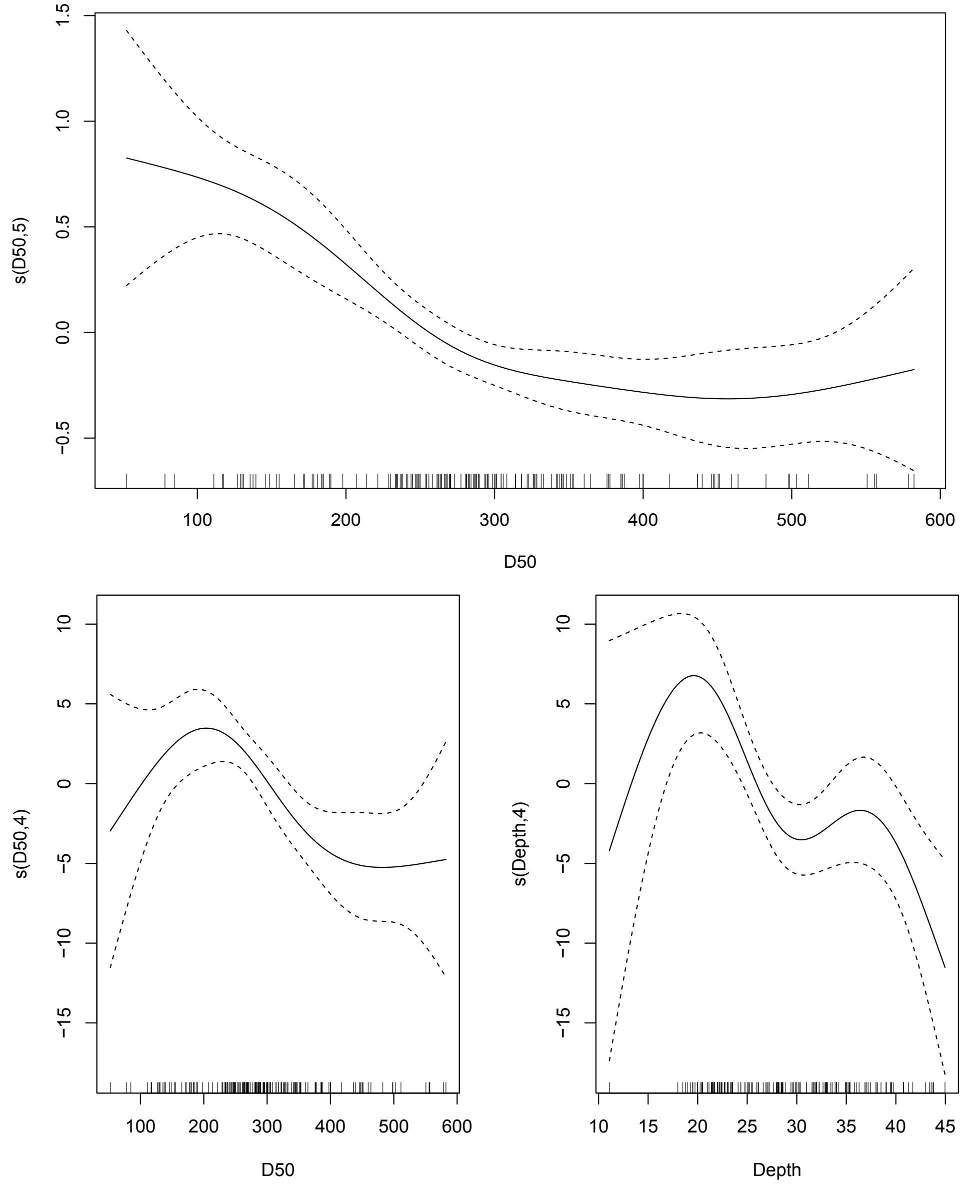

Figure 3.9 Smoothing functions of GAMs with 95% confidence intervals. Upper panel, biomass boxcore (g AFDW m^{-2}) and lower panels: species richness (n species boxcore^{-1})

In total, 109 infaunal species were detected in the reference area and borrow pit. Paddleworms (*Eteone* spp.), bristleworm *Spiophanes bombyx*, spaghetti worms (*Terebellidae* sp.), amphipod *Urothoe poseidonis* and sea urchins (*Echinoidea* spp.) numerically dominated infauna of the reference area. In terms of biomass, infauna was dominated by sea potato (*Echinocardium cordatum*), Atlantic jackknife clam (*Ensis directus*), mud shrimp (*Pestarella tyrrhena*), *Ensis* spp. and swimming crab (*Liocarcinus holsatus*). At the crests of the parallel sandbar, bristleworm *Notomastus latericeus*, white furrow shell (*Abra alba*) and bean-like tellin (*Tellina fabula*) dominated in terms of weight. At the crest of the oblique sandbar, infauna was dominated by white piddock (*Barnea candida*), unidentified bivalves, white furrow shell, the sea potato and *Notomastus latericeus*. Infauna in the troughs of sandbars and the deep areas was dominated by Tellinids and *A. alba*. In the SE trough of the parallel sandbar, boxcore failed due to the observed thick soft muddy bed.

The species composition in the shipping lane area changed considerably between 2006–2008 and 2012. The most abundant species of 2006–2008 were: *Owenia fusiformis* (2010.7 ind. m^{-2}), *A. alba* (862.4 ind. m^{-2}), *Lanice conchilega* (610.5 ind. m^{-2}), Tellinoidea spp. (381.1 ind. m^{-2}), *Heteromastus filiformis* (311.7 ind. m^{-2}), *Caprellidae* spp. (192.2 ind. m^{-2}), *Actiniaria* spp. (271.3 ind. m^{-2}) (de Jong et al., 2015a). In 2012, infauna was dominated by *Heteromastus filiformis* (516.7 ind. m^{-2}), *Tellinoidea* sp. (383.3 ind. m^{-2}) and *A. alba* (200 ind. m^{-2}). Due to differences in sample size in 2012, a statistical analysis was not possible. Species composition differed for all years amongst location and time after cessation of sand extraction (ADONIS, $p < 0.01$). The infaunal composition at the crest of the parallel sandbar differed significantly from the deep SE and NW whereas the trough of the parallel sandbar was not significantly different. Due to the parallel sandbar, significant differences in species composition occurred (Table 3.4). *A. alba* was the most abundant species at the infaunal biomass hotspots reaching values of 118.87 g AFDW m^{-2}. One location on the crest of the oblique sand bar with a biomass value of 129.08 g AFDW m^{-2} was dominated by white piddock (*Barnea candida*) (72.79 g AFDW m^{-2}) and American piddock (*Petricolaria pholadiformis*) (4.35 g AFDW m^{-2}) which both feed on old wood and peat fragments. Infaunal data is summarised in the upper panel of Fig. 3.16.

Table 3.4 Differences in infaunal species composition in 2012 between locations (permutational multivariate analysis of variance using distance matrices (ADONIS). Significance codes: *$p < 0.05$; ** $p < 0.01$; *** $p < 0.001$.

	Ref	Ed	Cr par	Tr par	Cr obl	Tr obl	SE	NW	sh
Ref	-		*	**	*	*	**	**	**
Ed		-					**	**	**
Cr par			-				*	*	
Tr par				-					
Cr obl					-				
Tr obl						-			
SE							-		
NW								-	
Sh									-

Cr: Crest, Tr: Trough, Ed: Edge, Sh: shipping lane, Slp: Slope, SE: South East, par: parallel and obl: oblique

nMDS ordination of IN and correlations with environmental variables

Infaunal data collected in 2012 is analysed with nMDS ordination, the stress value of the 2-D ordination was 0.25. Therefore, 3-D ordination was performed and stress was decreased to 0.17. In 2012, 39.3% of the variance is explained by depth, OM and time after the cessation of sand extraction (Fig. 3.10). The reference locations and the locations from the edge are clustered on the left side of the ordination. The locations from the SE and NW deep area, the troughs and crests group in the right part and are strongly correlated to increased OM levels. The samples from the deep NW part of the extraction site, with recent sand extraction are separated from the SE samples. Three of the four samples from the shipping lane area were grouping in the same area as the SE samples while one sample was more similar to the reference area. Infaunal samples from the troughs of the sandbars are located in the right part of the ordination.

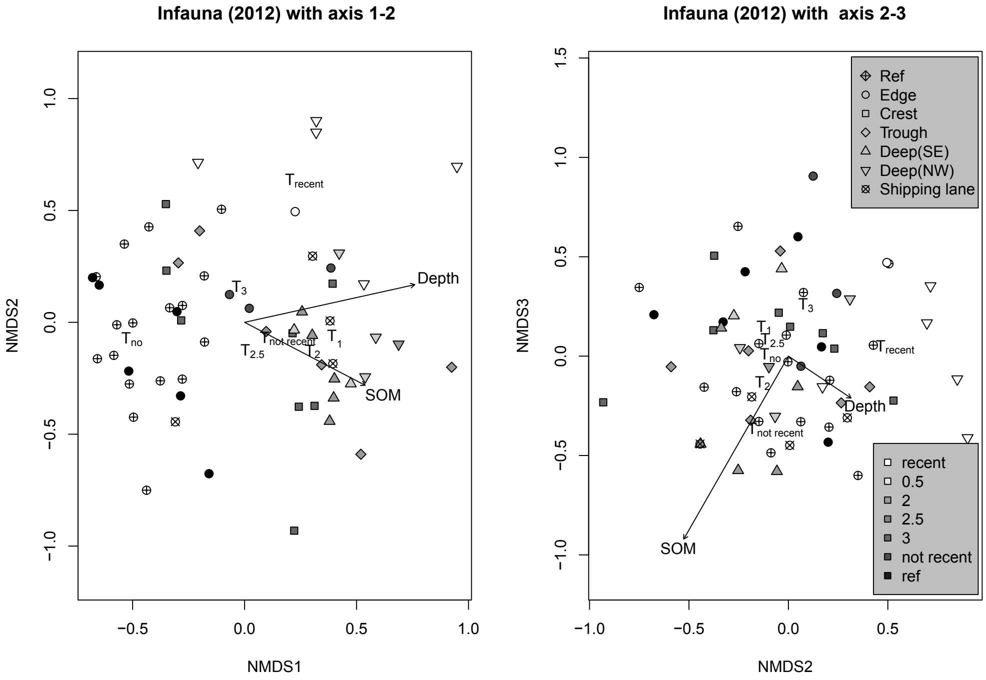

Figure 3.10 2-D ordination plots of 2012 with sites and significant correlations of continuous variables depth and OM and categorical variable time after cessation of sand extraction. The left panel is a ordination with axis 1 and 2, and the left panel axis 2 and 3.

Abra alba, *Actiniaria* spp. and *Owenia fusiformis* grouped together. Piddocks are located in the lowest part of the 2-D ordination, due to the high density on the crest of the oblique sandbar (Fig. 3.11). *Tellina fabula* was more located in the middle due to higher density at the crest of the parallel sand bar. *Asteroidea* spp., *Nepthys cirrosa* and *Phoronida* spp. are found in the reference area and the edge of the extraction site.

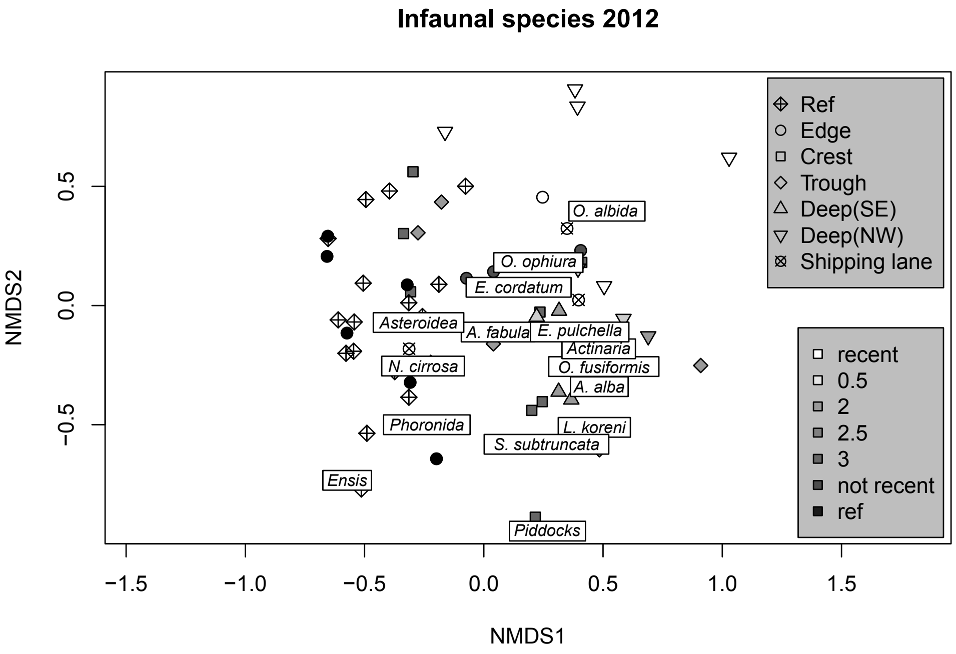

Figure 3.11 2-D ordination plot of 2012 (first 2 axis) with most abundant or characteristic species.

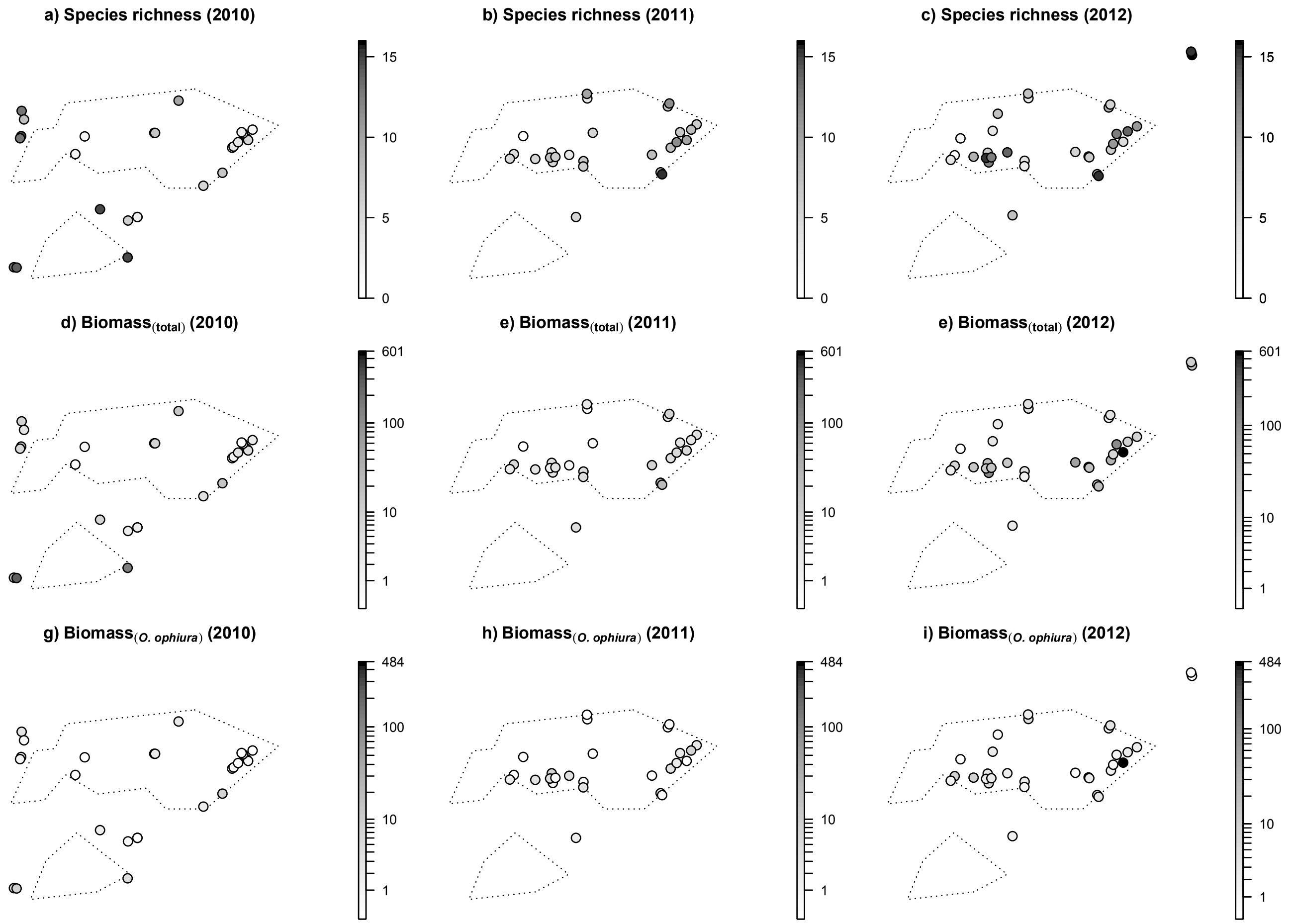

Figure 3.12 Biological parameters of bottom sledge sampling for 2010, 2011 and 2012. (**a–c**) Species richness (n species boxcore^{-1}), (**d–f**) Total biomass boxcore on a logarithmic scale (g WW m^{-2}) and (**g–i**) Biomass *O. ophiura* boxcore on a logarithmic scale (g WW m^{-2}).

EP data

Highest species richness (14.3 species per haul) was observed in the shipping lane whereas for the reference area (exclusion) on average, 7.5 species per haul were observed (Appendix 3.VI). The species richness in the shipping lane was significantly higher compared with the reference area, edge and deep NW part of the borrow pit (Table 3.5).

Table 3.5 Differences in epifaunal species richness in 2012 between locations, based on ANOVA followed by TukeyHSD multi pairwise comparisons. Significance codes: *p < 0.05; ** p < 0.01; *** p < 0.001.

	Ref	Ed	Cr par	Tr par	Cr obl	Tr obl	SE	NW	sh
Ref	-								*
Ed		-							**
Cr par			-						
Tr par				-					
Cr obl					-				
Tr obl						-			
SE							-		
NW								-	**
Sh									-

Cr: Crest, Tr: Trough, Ed: Edge, Sh: shipping lane, Slp: Slope, SE: South East, par: parallel and obl: oblique

The average epifaunal biomass in 2012 in deep SE parts was 32.8, 18.83 at the shipping lane area and 6.6 g WW m^{-2} for the reference area (Appendix 3.VI). Highest epifaunal biomass values (601.4 g WW m^{-2}) was detected in 2012 in the parallel trough mainly due to high wet weight values of *Ophiura ophiura* (483.7 g WW m^{-2}), *O. albida* (46.3 g WW m^{-2}) and *A. alba* (44.9 g WW m^{-2}) (Fig. 3.13 e and i). In 2012, the trough of the parallel sandbar harboured a significantly higher epifaunal biomass compared with the reference area and edge of the borrow pit (table 3.6).

Table 3.6 Differences in (log transformed) epifaunal wet weight in 2012 between locations, based on ANOVA followed by TukeyHSD multi pairwise comparisons of log transformed mean values. Significance codes: *p < 0.05; ** p < 0.01; *** p < 0.001.

	Ref	Ed	Cr par	Tr par	Cr obl	Tr obl	SE	NW	sh
Ref	-			*					
Ed		-		*					
Cr par			-						
Tr par				-				**	
Cr obl					-				
Tr obl						-		*	
SE							-	*	
NW								-	
Sh									-

Epifaunal species richness was significantly correlated with time after extraction, which explained 51.6% of the total deviance (Appendix 3.V & Table 3.3). Epifaunal biomass showed a significant correlation with mud (peaked at 30 vol%), mean salinity and time after sand cessation (Fig. 3.13 & Appendix 3.V & Table 3.4) and explained 53.2% of the total deviance.

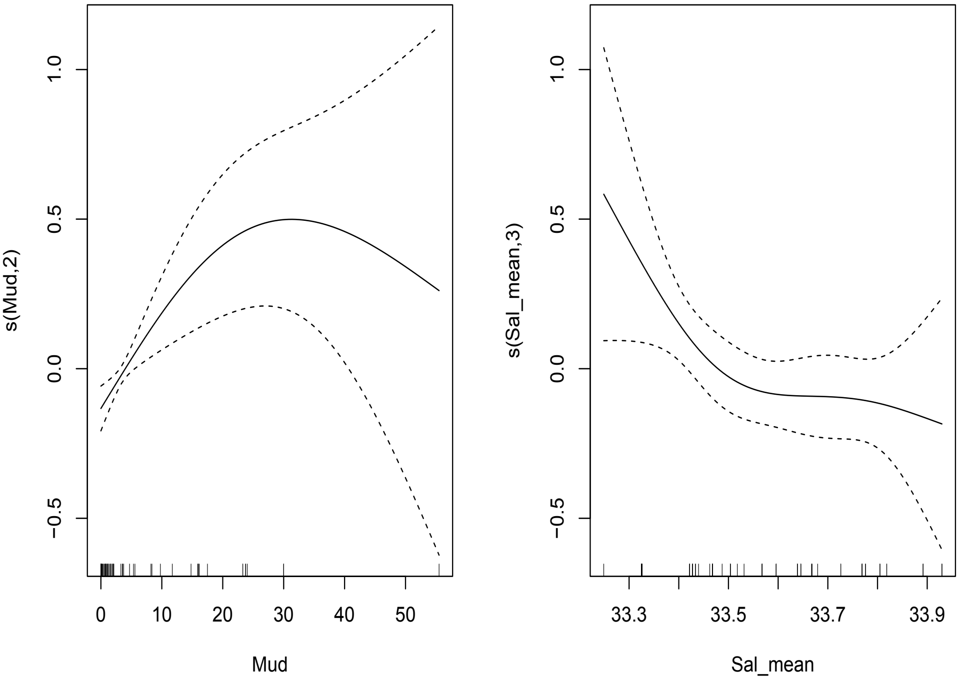

Figure 3.13 Smoothing functions of GAMs. Epifaunal biomass (g WW m^{-2}).

Species composition of Epifauna (EP)

For epifauna, in total, 26 species were detected with the bottom sledge in the reference area and borrow pit. EP in the exclusion area was numerically dominated by brittlestars (*Ophiura ophiura*) and cut trough shell (*Spisula subtruncata*). The edge of the extraction area was numerically dominated by *O. ophiura* and bean-like tellin (*Tellina fabula*) and biomass was 8.0 g WW m^{-2} with nine species. EP at the crest of the parallel sandbar was numerically dominated by white furrow shell (*Abra alba*), *Ophiura albida*, common necklace shell (*Euspira nitida*), *O. ophiura* and netted dog whelk (*Nassarius nitidus*) with a wet weight of 7.6 g WW m^{-2} and 10 species per haul. *Abra alba*, *Tellina fabula*, *O. ophiura*, elliptic cut trough shell (*Spisula elliptica*), *Nassarius nitidus* numerically dominated EP on the crest of the oblique sandbar, wet weight was 17.5 g WW m^{-2} with 12.5 species per haul. Epifauna at the trough of the parallel sandbar was dominated by *O. ophiura*, *Abra alba*, *O. albida* and *Euspira nitida* with a wet weight of 175.4 g WW m^{-2} and 8.3 species per haul. *Abra alba*, *O. ophiura* and *Tellina fabula* dominated the trough of the oblique sandbar with a wet weight of 31.0 g WW m^{-2} and 9.8 species per haul.

White piddock (*Barnea candida*), which was detected with the boxcorer on the crest of the oblique sandbar, was not detected with the bottom sledge but the biomass of EP was also higher. *A. alba*, *Tellina fabula*, hermit crab (*Pagurus bernhardus*), *O. ophiura* and *Nassarius nitidus* dominated the deep south-eastern part and wet weigh was on average 32.8 g WW m^{-2} and 7.7 species per haul. EP in the shipping lane area in 2006–2008 was numerically dominated by *Abra alba* (143.9 ind. m^{-2}), anemones *Actiniaria* sp. (88.8 ind. m^{-2}), *O. albida* (40.7 ind. m^{-2}), pullet carpet shell (*Venerupis senegalensis*) (5.9 ind. m^{-2}) and *Nassarius reticulatus* (3 ind. m^{-2}). In 2012, *Abra alba* dominated EP in the shipping lane area and wet weight decreased from values above 100 g in 2006 and 2008 to 18.8 g WW m^{-2} in 2012. Epifaunal data is summarised in the lower panel of Fig. 3.16.

Table 3.7 Differences in EP species composition in 2012 between locations (permutational multivariate analysis of variance using distance matrices (ADONIS). Significance codes: *p < 0.05; ** p < 0.01; *** p < 0.001.

	Ref	Ed	Cr par	Tr par	Cr obl	Tr obl	SE	NW	sh
Ref	-		*	**	*	*	**	.	**
Ed		-	.	*	.	**	*		*
Cr par			-					.	.
Tr par				-				.	
Cr obl					-				*
Tr obl						-			
SE							-		
NW								-	
Sh									-

The species composition of EP inside the extraction site and shipping lane area differed significantly from the edge of the extraction site and the reference area (Table 3.7). The species composition of the oblique crest differed significantly (ADONIS < 0.05) from the shipping lane area.

nMDS ordination of EP and correlations with environmental variables.

Stress values for the 2-D ordinations of EP in 2012 was 0.08, which is satisfactory. The variables time after the cessation of sand extraction, the fraction very fine sand and mean and maximum bed shear stress explained 62.9% of the variance (Fig. 3.14). Reference samples are located in the right part of the ordination, while the samples from the shipping lane areas ended in the upper left part. Samples from areas with recent extraction are similar to the reference locations. One sample collected in the trough of the parallel sandbar in 2012 ended at the left side of the ordination, and was characterised by the highest epifaunal biomass, 601.36 g WW m^{-2} and *O. ophiura* as most abundant species, 483.67 g WW m^{-2}.

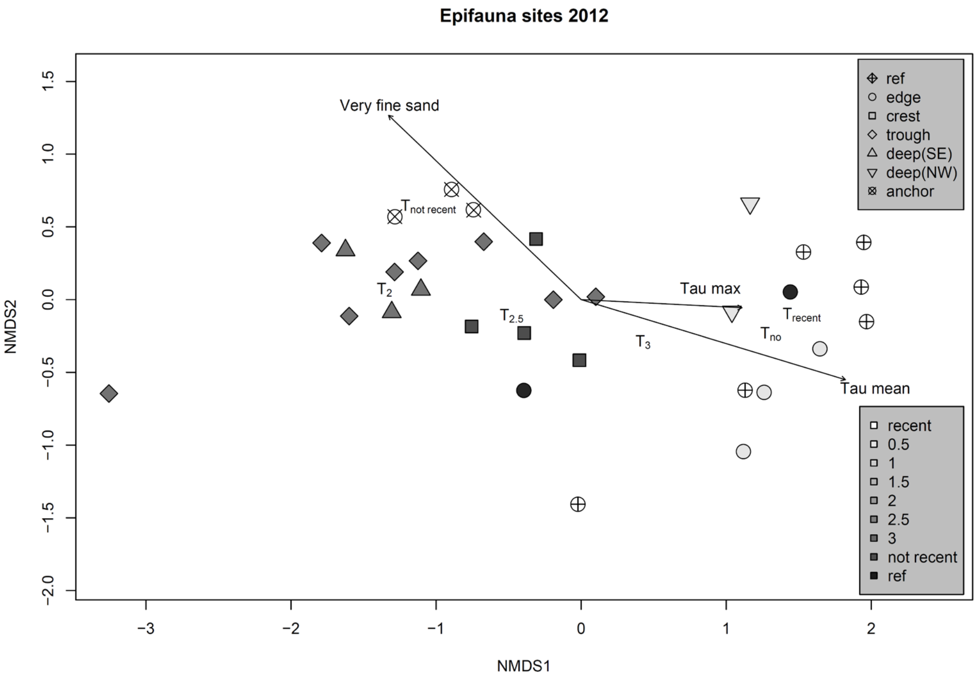

Figure 3.14 2-D ordination plot of EP in 2012 with sites and significant correlations of continuous variables very fine sand, mean and maximum bed shear stress and categorical variable time after cessation of sand extraction.

The samples collected at the shipping lane area were characterised by a higher species richness compared to the samples from the troughs and SE deep parts which are dominated by *Abra alba* (Fig. 3.15). The samples from the crest grouped in the middle of the ordination and harboured Tellinids such as *Tellina fabula* and *Abra prismatica* which are commonly found in coarser sediments (Degraer et al., 2006). Samples from the reference locations grouped in the right part of the ordination and were dominated by *Ophiura ophiura* and *Spisula subtruncata*.

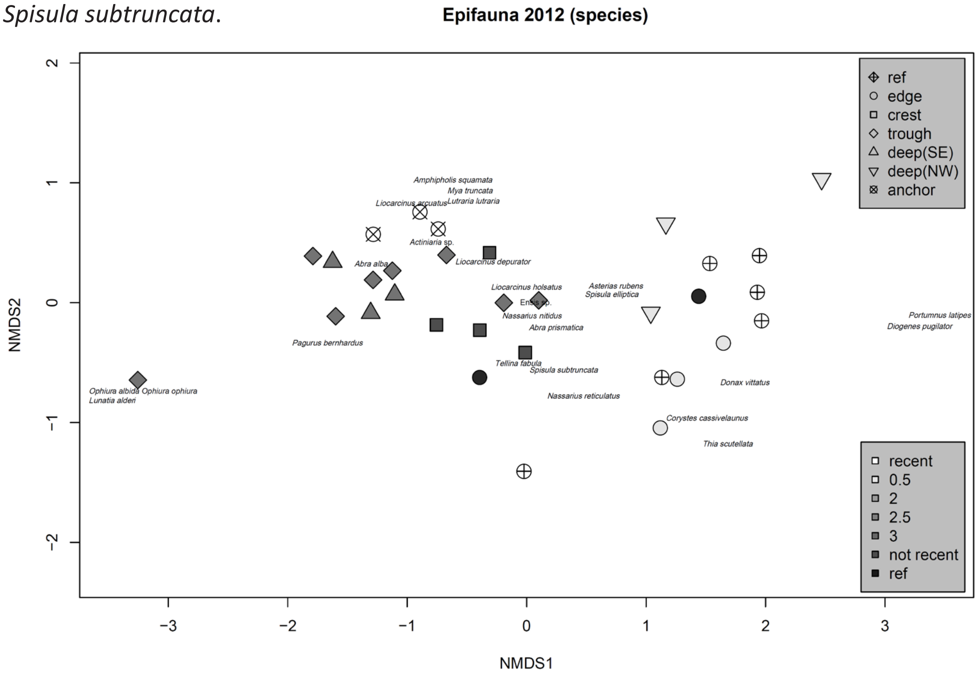

Figure 3.15 2-D ordination plot of 2012 most abundant or characteristic species.

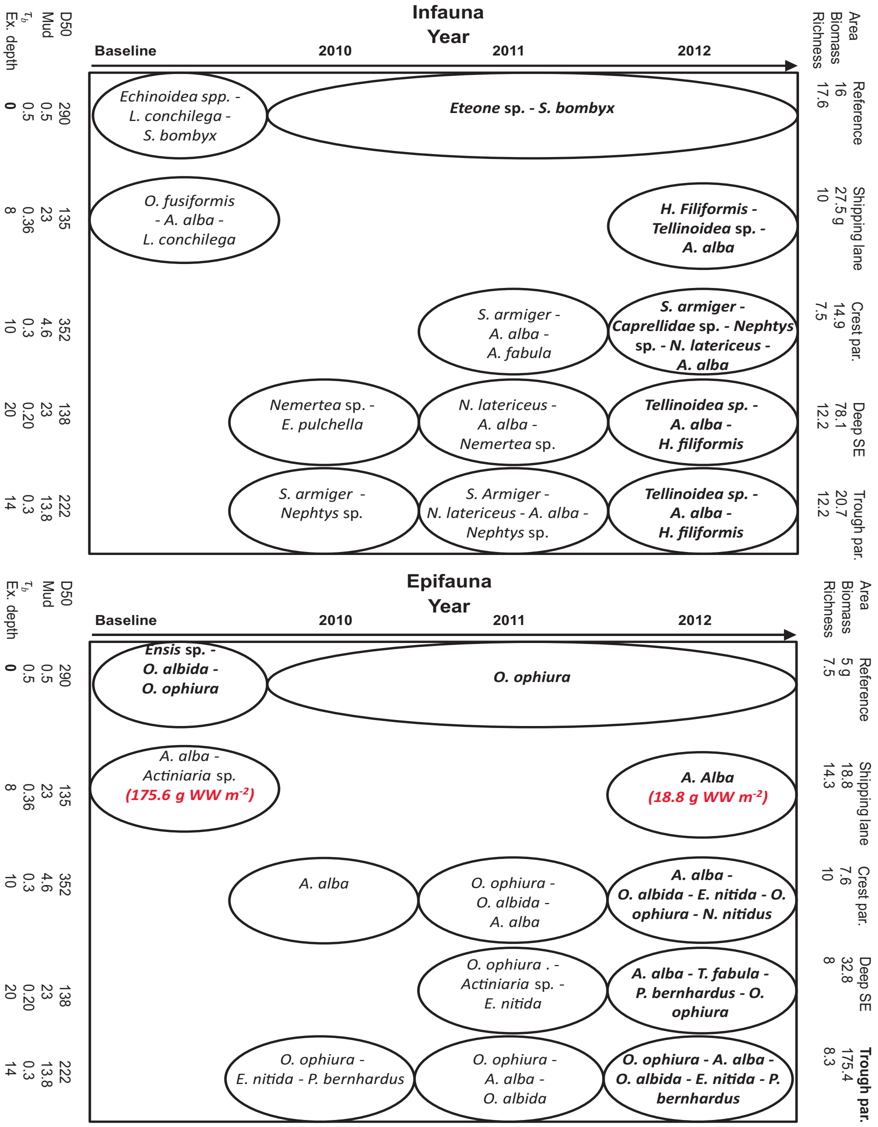

Figure 3.16 Summary of infaunal and epifaunal species composition and biomass in relation to D_{50}, mud content, mean near-bed shear stress (τ_b) and excavation depth.

3.4 Discussion

Two years after sand extraction, macrozoobenthic species composition and biomass differed between the reference area and the deepest parts of the borrow pits and white furrow *shell (Abra alba*) became the most abundant species in the borrow pit. Macrozoobenthic biomass significantly increased on average 5-fold in the deepest areas 2 years after the cessation of sand extraction. Species richness in the deepest areas was slightly lower than in reference area. In the reference area, however, macrozoobenthos consisted of a broad array of organisms such as worms, brittlestars, sea urchins and shellfish.

Macrozoobenthos in the 'Euromaasgeul' shipping lane

The conditions in the Euromaasgeul shipping lane area are similar to the large-scale and deep borrow pit, both in biotic and abiotic characteristics. This may give insight in the long-term development of macrozoobenthos in heavily impacted dredged areas. Species composition in the shipping lane area changed considerably between 2006–2008 and 2012. The most abundant infaunal species of 2006–2008 were: *Owenia fusiformis*, *A. alba* and *Lanice conchilega* (de Jong et al., 2015a). In 2012, IN was dominated by *Heteromastus filiformis, Tellinoidea* and *A. alba*. *L. conchilega* and *O. fusiformis* were absent in the entire research area in 2012. In 2006 and 2008, large differences in the density of *Lanice conchilega* and *Owenia fusiformis* were detected which may be related to variation in winter temperature (de Jong et al., 2015a).

Infaunal species richness in the shipping lane in 2006, 2008 and 2012 was higher than in the reference area. Comparing species richness in 2012 in the shipping lane with baseline data research was not possible due to the decrease in sampled surface. In 2012, species richness in the shipping lane (10 species per boxcore) was higher than in the reference area (7.4 species per boxcore) and IN appeared to be more diverse. Infaunal biomass reached the highest levels in 2012, 27.5 compared with 6.4 and 9.9 g AFDW m^{-2} in 2006 and 2008. EP in the shipping lane in 2012 was dominated by *A. alba* dominated in contrast to the more diverse species composition in 2006 and 2008 (*A. alba*, anemones, *Ophiura albida*, *Venerupis senegalensis* and *Nassarius reticulatus*). There is a possibility that instead of maintenance dredging differences in water depth were levelled by moving crests from ripples and small sand waves in the troughs (pers. comment rijkswaterstaat) which induced the decrease in biomass and species richness in 2012.

Natural deep sea-floor crater

30 years after the formation of a 30 m deep natural sea-floor crater, infaunal species composition and biomass still differed strongly from the surrounding area (Thatje et al., 1999). Maximum infaunal (fresh weight) biomass inside the crater was 487 g WW m^{-2} due to the frequent occurrence of the sea potato (*Echinocardium cordatum*), which accounted for

about 82% of the total biomass. *Amphiura filiformis* and *Ophiura* spp. fluctuated between and within years and *A. alba* was detected but not dominant in terms of density or biomass. In the borrow pit, a maximum epifaunal biomass value of 601.4 g WW m^{-2} was detected in the trough of the parallel sandbar, mainly due to high wet weight values of *Ophiura ophiura* (483.7 g WW m^{-2}), *Ophiura albida* (46.3 g WW m^{-2}) and *A. alba* (44.9 g WW m^{-2}). The very fragile *E. cordatum*, however, is severely damaged in the epifaunal sampling procedure with the bottom sledge and gets uncountable so biomass values may be underestimated. In the deep SE, *E. cordatum* accounted for 11.8% and *A. alba* for 67.2% of the total infaunal AFDW biomass (collected with the boxcore). In the trough of the oblique sandbar, *E. cordatum* accounted for 50.2% and *A. alba* for 28.3%. In the trough of the parallel sand bar, no *E. cordatum* was present and *A. alba* accounted for 70.0%. In the shipping lane, *E. cordatum* accounted for 61.1% and *A. alba* for 19.3% of the total biomass. Biomass of infaunal macrozoobenthos in the deep sea floor crater, shipping lane and troughs of the oblique sandbar was dominated by *E. cordatum*, whereas being absent in the trough of the parallel sandbar.

Ecological landscaping

Infaunal species composition at the crest of the parallel sandbar differed significantly from the deep SE and NW, whereas the trough of the parallel sandbar was not significantly different. The crest of the parallel sand bar was characterised by the highest infaunal species richness (12.5 species per boxcore), high epifaunal species richness (8.25 species per haul), a higher density of bean-like tellin (*Tellina fabula*) and the sediment was becoming coarser in time. The species composition of EP inside the MV2 borrow pit and shipping lane area differed significantly from the edge of the borrow pit and the reference area. Epifaunal composition at the crest of the oblique sandbar differed significantly from the shipping lane area. This may be an indication of the feasibility and effectivity of ecological landscaping.

One sample on the crest of the oblique sand bar, with a biomass value of 129.08 g AFDW m^{-2}, was dominated by white piddock (*Barnea candida*) which both feed on littoral fossilised peat and wood fragments. The presence of this species rich and productive assemblage is, however, coincidental and not the result of ecological landscaping. Little information was found regarding the characteristics of this assemblage, longevity presumably depends on the stock of peat and wood fragments (Budd, 2008).

Sedimentation rate and backfilling in the MV2 borrow pit

Between 2010 and 2011, we only observed small differences in water depth but after 2011 sedimentation rate in the troughs of the parallel sandbar increased considerably, 0.43 m y^{-1} in the northern and 0.75 m y^{-1} in the southern trough. In the deep SE area 0.18 m y^{-1} of sediment settled whereas 0.27 m y^{-1} settled on the crest of the parallel sandbar.

A highly similar sedimentation rate was found inside a natural sea-floor crater, sedimentation rates of 0.50 m yr^{-1} were encountered and mud content increased from 5% to 40%

(Thatje et al., 1999). The morphological model analysis, a sedimentation rate of 5 m non-cohesive sediment in the first 10 year is predicted in the SE area of the borrow pit (Stolk, 2014; Klein and van den Boomgaard, 2013). The PUTMOR study, however, revealed that 20.000 m^3 of fines settled in one year which equals only 0.03 m yr^{-1} (Boers, 2005).

A morphological model study revealed that backfilling of the MV2 borrow pit may take decades or even longer (Klein and van den Boomgaard, 2013). The study showed a higher sedimentation rate in the southern parts of the borrow pit due to the asymmetry of the tide and an average sedimentation rate of 5 m in 30 years. The observation of the decrease in median grain size and increase in mud values in the borrow pit may be originating from suspended load. During storms, settled particles may re-suspend, redistribute and settle again during more calm situations. Density driven transport in coastward direction may lead to higher sedimentation rates in the eastern part of the borrow pit (van der Hout et al., 2015). The discrepancy between the modelled sedimentation rate and observed elevated sedimentation rates in the troughs of the parallel sandbar may be because only non-cohesive sediment was taken into account.

Organic matter enrichment and species richness in the MV2 borrow pit

Hyland et al. (2005) suggested that the risks of reduced infaunal species richness from organic load and other associated stressors in sediments should be relatively low at total organic carbon (TOC) concentrations less than about 10 mg g^{-1}, high at concentrations larger than 35 mg g^{-1}, and intermediate at concentrations in between. Total organic carbon (TOC) was measured with a CHN analyser and some samples as organic matter content by LOI, these values were reduced by a factor of 3 to convert to TOC and correct for the overestimation of organic carbon associated with the LOI methodology (Leong and Tanner, 1999).

In 2012, OM contents were in the range of decreasing species richness. In the deep SE part and at the top of the oblique sandbar values were 9.6% (TOC: 32 mg g^{-1}) and 9.8% (TOC: 33 mg g^{-1}) respectively. The GAM analysis, however, revealed no decrease in infaunal species richness in the region 10–35 mg g^{-1}; species richness is best explained by D_{50} and water depth. A similar response between species richness with D_{50} and water depth was found for the baseline data (de Jong et al., 2015a). Epifaunal species richness in the borrow pit only correlated with time after cessation of sand extraction. In the baseline study, epifaunal species richness showed a negative correlation with mean bed shear stress and peaked at a value of 0.4 N m^{-2}, peaked at a water depth of 20 m, at 200 µm and at a maximum shear stress of 2.25 N m^{-2}. Infaunal biomass peaked at 200 µm.

EP shift from *A. alba* to *Ophiura* spp. in the trough of the parallel sandbar

EP in the SE trough of the parallel sandbar in 2012 strongly differed in species composition and biomass compared to other deep parts with a dominance of *Ophiura ophiura*, *O. albida* and some *A. alba* and a biomass of 601.4 g WW m^{-2} instead of EP being dominated by *A. alba* and an average biomass of 32.8 g WW m^{-2}. Unfortunately, boxcore sampling failed in the SE trough of the parallel sandbar presumably due to the observed thick soft muddy bed. The over-abundance of ophiuroids may be related to factors such as sedimentation rate or increase in sediment organic matter, biological activity, water depth and changes in water circulation. These factors may also lower dissolved oxygen (DO) levels (Eldridge and Roelke, 2011).

Limitations of present study and recommendations

Sedimentation rate and oxygen concentration were not modelled in the hydrodynamic model because model improvements were first needed regarding appropriate nesting. The horizontal grid size for the MV2 borrow pit was decreased from approximately 100 by 100 meters in the first model to 45 m by 38 m in the improved model. The smaller grid enabled determining differences in bed shear stress and near-bed salinity around the landscaped sandbars. For the present MV2 borrow pit we recommend ongoing monitoring including sedimentation rate and oxygen measurements since significant macrozoobenthic changes occurred in the deepest parts of the borrow pit. For future deep and large-scale borrow pits it is recommended to model oxygen concentration and sedimentation rates of cohesive and non-cohesive material in the environmental impact assessments of future borrow pits.

3.5 Conclusion

This is the first study on the short-term impact on macrozoobenthos in a deep and large-scale borrow pit. Contrary to shallow sand extraction, macrozoobenthic biomass significantly increased 5-fold in the deepest areas 2 years after the cessation of sand extraction. Furthermore, species composition changed significantly and white furrow shell (*Abra alba*) became the most abundant species. Macrozoobenthic species composition and biomass correlated with time after cessation of sand extraction, sediment and hydrographic variables. Besides changes in macrozoobenthos, sediment characteristics also significantly changed in the deepest parts. Ecosystem-based landscaping techniques were found to be effective in significantly influencing sediment characteristics and macrozoobenthic assemblage. Significant changes in EP occurred in deepest parts of the MV2 borrow pit in 2012 and coincided with the highest sedimentation rate.

In the case of shallow sand extraction in the Netherlands, macrozoobenthos returns to pre-dredged conditions within 4–6 years. Combining the results of macrozoobenthic species composition in the borrow pit, shipping lane area and natural deep sea-floor crater with sedimentation rate and backfilling time leads to my conclusion that benthos is presumably not returning to pre-dredged conditions within decades. Continuing monitoring including sedimentation rate and oxygen measurements is recommended since significant changes occurred in the deepest parts of the borrow pit.

3.6 Acknowledgements

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Chapter 4

Short-term impact on demersal fish of a large-scale and deep borrow pit with ecosystem-based landscaped sandbars

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Abstract

For the seaward harbour extension Maasvlakte 2 (MV2) of the Port of Rotterdam (PoR) in the Netherlands, approximately 220 million m^3 of sand was extracted between 2009 and 2013. To decrease the surface area of direct impact, the Dutch authorities permitted deep sand extraction, down to 20 m below the seabed. Biological and physical impacts of large-scale and deep sand extraction and ecological landscaping are still being investigated and are largely unknown. For this reason, we investigated the presence of demersal fish in the deep MV2 borrow pit.

Significant differences in demersal fish species assemblages in the MV2 borrow pit were associated with variables such as water depth, median grain size, fraction of very fine sand, biomass of white furrow shell (*Abra alba*), and time after the cessation of sand extraction. Large quantities of undigested crushed white furrow shell fragments were found in all stomachs and intestines of plaice (*Pleuronectes platessa*), indicating that it is an important prey item. One and two years after cessation, a significant 20-fold increase in demersal fish biomass was observed in the deep parts of the borrow pit. However, in the troughs of a landscaped sandbar, a significant drop in biomass down to reference levels was observed two years after cessation. Significant differences in fish assemblage were found between the crests and troughs of the sandbars. Tub gurnard (*Chelidonichthys lucerna*) was indicated as a Dufrêne–Legendre indicator species of the crests. This may be an indication of the applicability of landscaping techniques to induce heterogeneity of the seabed, although it remains difficult to draw a strong conclusion due to lack of replication of the experiment. A new ecological equilibrium is not reached after 2 years since biotic and abiotic variables are still adapting. To understand the final impact of deep and large-scale sand extraction on demersal fish, we recommend monitoring for a longer period, at least for a period of six years or even longer.

4.1 Introduction

The demand for marine sand in the Netherlands and worldwide is strongly increasing. In the Netherlands, approximately 26 million m^3 of sand is used annually for coastal nourishments and construction (Stolk and Dijkshoorn, 2009; ICES, 2014a). An increase of annual nourishments from 12.5 up to 40–85 million m^3 to counteract effects of future sea level rise is expected (Deltacommissie, 2008). For Maasvlakte 2 (MV2), the seaward expansion of the Port of Rotterdam (PoR), approximately 220 million m^3 of sand was extracted between 2009 and 2013, with an average extraction depth of 20 m below the seabed. Generally in the Netherlands, only shallow sand extraction down to 2 m below the seabed was permitted (V&W, 2004; IDON, 2005). For MV2 however, the Dutch authorities permitted sand extraction deeper than 2 m, primarily to reduce the surface area of direct impact.

Fish assemblages on the North Sea scale are mainly influenced by bottom water temperature, bottom water salinity, tidal stress, and water depth (Reiss et al., 2010; Callaway et al., 2002). Furthermore, fish assemblages are linked to biotic and abiotic habitat characteristics, and meso-scale bedforms (Ellis et al., 2011; Sell and Kröncke, 2013). Ellis et al. (2011) found that species richness of infauna, epifauna and fish was larger in the silty troughs of sandbanks off the coast of the UK than on the crests.

Large-scale sand extraction was shown to have a negative impact on fish in the Yellow sea (Hwang et al., 2013), a decline of more than 70% of the total number of fish and the number of species (Son and Han, 2007) and direct and indirect damages to commercial fisheries were observed (Kim and Grigalunas, 2009). On the other hand, aggregate extraction may also lead to new habitats and may favour macrozoobenthos and fish (Desprez, 2000). Fishing fleets in the Eastern English Channel near aggregate extraction sites were not deterred by extraction activities and fishing effort of dredgers and potters was higher than elsewhere, and also positively correlated to extraction intensity with a lag of 0 to 9 months (Marchal et al., 2014). Sand extraction may even positively influence marine mammals (Todd et al., 2014).

Ecosystem-based landscaping techniques are not commonly used to reduce the impact of sand extraction. In the UK, gravel-seeding techniques were tested to restore the seabed after gravel extraction (Cooper et al., 2011). In the MV2 borrow pit, two sandbars were artificially created by selective dredging, copying naturally occurring meso-scale bedforms to increase habitat heterogeneity and thereby influencing demersal fish assemblage (van Dalfsen and Aarninkhof, 2009; van Raalte et al., 2007; Borsje et al., 2011).

In this study, we test the hypothesis that deep and large-scale sand extraction and ecosystem-based landscaping approaches will lead to differences in fish assemblage and we are aiming to answer the following questions:

(i) Are there significant differences in fish species assemblage between reference area and MV2 borrow pit, and within the borrow pit?
(ii) Are there significant temporal differences in fish assemblage, macrozoobenthos and environmental variables during the monitoring campaign?
(iii) Which environmental variables are determining the differences?
(iv) Are ecosystem-based landscaping techniques feasible and effective in influencing fish assemblages?

4.2 Methods

Study area

The Maasvlakte 2 (MV2) borrow pit is situated in front of the Port of Rotterdam (PoR), the Netherlands, outside the 20 m isobath (Fig. 4.1). The borrow pit is 2 km long, 6 km wide with an average extraction depth of 20 m at an initial water depth of approximately 20 m below average sea level. Approximately 220 million m^3 of sand was extracted between 2009 and 2013, of which 170 million m^3 in the first two years (Borst and Vellinga, 2012).

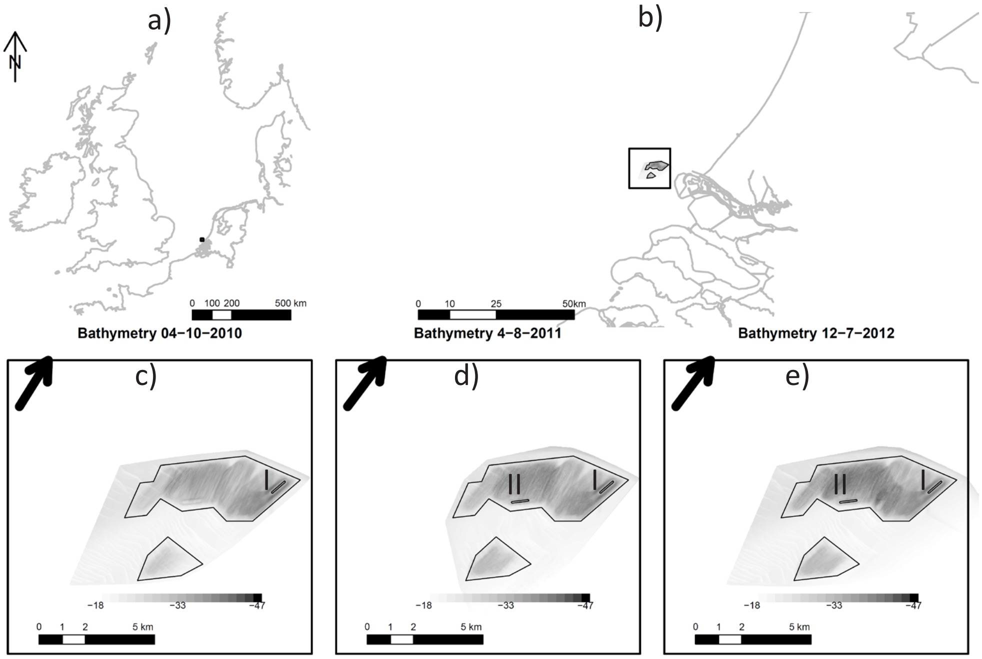


Figure 4.1 (**a**) North Sea, (**b**) Dutch coastal area with the Maasvlakte 2 (MV2) borrow pit and harbour extension Maasvlakte 2 in front of the PoR, (**c**) bathymetry of the borrow pit (date 4 October 2010) with one landscaped sandbar parallel to the tidal current (I). (**d–e**) The borrow pit with two landscaped sandbars, one parallel (I) and one oblique to the tidal current (II), (dates 4 August 2011 and 12 July 2012). The arrows in (c-e) denotes the residual tidal current. Data were derived from bathymetric multibeam surveys of the dredging companies.

Two sandbars were created in the borrow pit to investigate the applicability of ecosystem-based landscaping in sand extraction projects (Fig. 4.1 & 4.2). One sandbar (I), parallel to the tidal current, was left behind in the seabed in spring 2010. This parallel sandbar has a length of 700 m, a width at the crest of 70 m and slopes of 140 m length (Fig. 4.2). The crest of the sandbar is located at a water depth of 30 m and the troughs are more than 40 m deep. In 2011, the second sandbar (II) was completed with an orientation oblique to the tidal current. The length and width are similar to the parallel sandbar but, due to time constraints, the difference in depth between crest and trough is less pronounced. The crest is situated at a water depth of 28 m and the northern trough is 36 m deep. A narrow and 32 m deep trench separates the crest from the slope of the borrow pit. The volume of each sandbar is approximately 1.25 million m^3 of sand with slopes of 1:7–1:10.

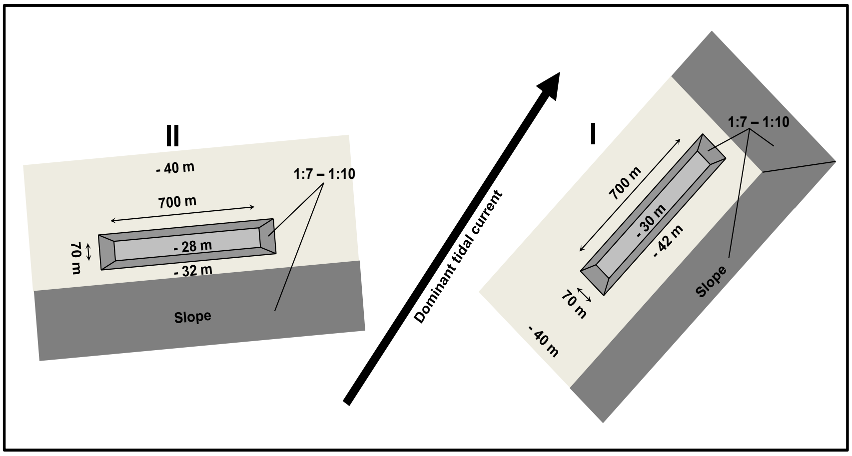


Figure 4.2 Schematic representation of the ecosystem-based landscaped sandbars, left: sandbar II, oblique to the tidal current and right: sandbar I, parallel to the tidal current.

During our surveys in 2011 and 2012, two trailer suction hopper dredgers were active in the centre of the borrow pit, extracting approximately 2 million m^3 of marine sand per week. The water depth increased from 33 m to approximately 40 m (Fig. 4.1) but the areas near the landscaped sandbars remained un-dredged after completion.

Fishing methods

A commercial fishing vessel was used, 'Jan Maria' GO–29, with a length of 23 m, less than 300 horsepower and equipped with a standard commercial 4.5-m beam trawl. The beam trawl, which was towed at 4 knots fishing speed, was equipped with four ticklers, five flip-up ropes and diamond codend mesh size of 80 mm. The ships GPS-system logged the position of the sampling locations, water depth was determined with the ship's depth sounder. The maximum haul distance was one nautical mile in the reference area. Shorter hauls were planned within the borrow pit; at the landscaped sandbars, hauls of approximately 700 m length were applied. Some of the hauls ended before the planned end-coordinates because of difficulties with fishing inside the borrow pit due to large changes in seabed topography and sediment composition. In the surrounding reference area, fishing direction was generally perpendicular to the direction of naturally occurring seabed patterns to ensure heterogeneous sampling of crests and troughs of sand waves. In the borrow pit, fishing direction was generally parallel to the seabed structures to enable comparisons between the different locations. We sampled in the reference area, at the slope of the borrow pit, at two locations in the deep parts of the borrow pit i.e. the south-east and north-west, in the troughs and at the crests of the landscaped sandbars (Fig. 4.3, Table 4.1).

Table 4.1 Sub locations and the number of fish samples

Year	Reference	Slope	Crests	Troughs	Deep (SE)	Deep (NW)	total
2010	4	-	1	2	1	1	9
2011	7	4	3	3	1	2	20
2012	4	2	3	4	2	2	17

The fish surveys were conducted on 14 July 2010, 27 to 29 July 2011 and 13 to 15 June 2012. In 2010, four reference fish samples and five borrow pit samples were collected (Table 4.1 & Fig. 4.2). In 2011, seven samples were collected in the reference area and thirteen samples in the borrow pit. In 2012, four samples were collected in the reference area and thirteen in the borrow pit.

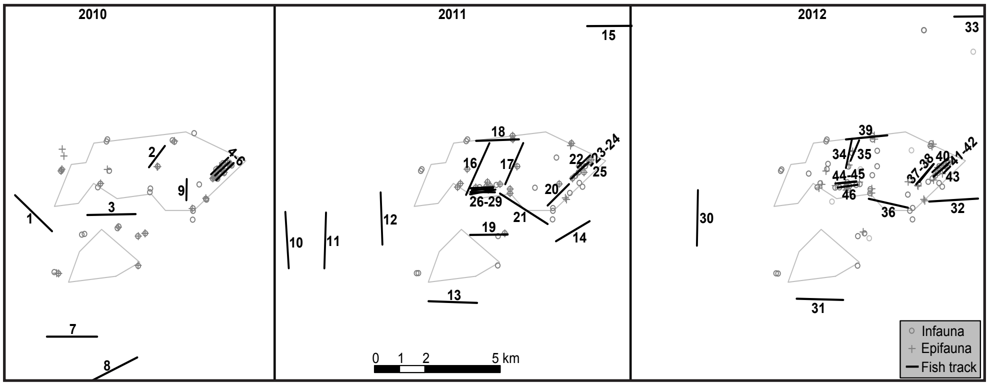


Figure 4.3 The MV2 borrow pit and reference area, fish hauls (in black) and in- and epifaunal samples in grey for 2010, 2011 and 2012. Sandbars were not drawn but are indicated with the fish hauls.

Fish were sorted and length frequency distribution 'the nearest whole cm below the measured length' was determined. Abundance of fish was calculated by dividing the number of fish by the fishing surface area and expressed as number of fish per hectare (length of haul x 4.5 m x number of nets). Species richness of a haul (number of species haul^{-1}) was determined and published length-weight relationships (Robinson et al., 2010; Coull et al., 1988) were used to calculate the weight of fish (WW biomass, kg ha^{-1}). The average length of a fish species was calculated by dividing the sum of all multiplied size classes and their densities with the total density. In 2012, stomach and intestines contents of on average 10 specimens of plaice (*Pleuronectes platessa*), dab (*Limanda limanda*), and shorthorn sculpin (*Myoxocephalus scorpius*) were taken from the south-eastern deep part and troughs to get a quick insight in fish diets.

Macrozoobenthos and sediment

The closest macrozoobenthos and sediment samples locations were selected to link with the fish hauls (Fig. 4.3). A boxcorer was used to sample macrozoobenthic infauna (> 1-mm) and sediment, carried out by the Monitor Taskforce of the Royal Netherlands Institute for Sea Research (NIOZ) on 29–30 June 2010, 2–5 May 2011 and 23–25 April 2012. The boxcorer surface area was 0.0774 m^2 with a maximum penetration depth of 30 cm (Craeymeersch and Escaravage, 2010; Perdon and Kaag, 2006b). Infaunal ash-free dry weight biomass (g AFDW m^{-2}) was analysed by mass loss on ignition (MLOI) (2 days at 80 °C followed by 2 hours at 580 °C). A bottom sledge was used to sample larger macrozoobenthic fauna (0.5–10 cm), executed by the Institute for Marine Resources & Ecosystem Studies (IMARES Wageningen UR) on 7–8 July 2010, 14–15 June 2011 and 6–7 June 2012. The sledge was equipped with a 5 mm mesh cage. On average, 15 m^2 were sampled during each sledge haul (~150 m length, 10 cm width and a maximum penetration depth of ~10 cm). Wet weight of larger macrozoobenthic fauna was directly measured (g WW m^{-2}) and fish were excluded from the samples because they were already sampled with the beam trawl.

The upper 5 cm of sediment from the untreated boxcorer were collected and frozen for later analysis of particle size. Prior to analysis, sediment samples were freeze-dried, homogenised and analysed with a Malvern Mastersizer 2000 particle size analyser. Percentile sediment grain size (D_{10}, D_{50}, D_{90}) and sediment grain size distribution among the different classes: clay (< 4 µm), silt (4–63 µm), mud (<63 µm), very fine sand (63 µm–125 µm), fine sand (125 µm–250 µm), medium sand (250 µm–500 µm) and coarse sand (> 500 µm) were measured as percentage of total volume. Organic matter (OM) was analysed in 2012 by MLOI as percentage of sediment mass.

Time after the cessation of sand extraction and fishing activity

Areas surrounding the borrow pit are labelled as reference area (Ref), t_{recent} denotes sampling directly after sand extraction and for 1 and 2 year after the cessation, t_1 and t_2 are used. Fishing activity in- and outside the borrow pit was derived from Vessel Monitoring through Satellite (VMS) data for the years of the fish survey (2010, 2011 and 2012).

Statistical analysis

Significance of differences in macrozoobenthic species composition between location and time after cessation of sand extraction was tested with permutational multivariate analysis of variance using distance matrices (ADONIS) of package 'vegan'. Due to the lack of post-hoc multi-comparison tests in the ADONIS function, we manually selected sets of locations and analysed each comparison. We applied Dufrêne−Legendre Indicator species analysis using the indval function of package 'labdsv' to determine characteristic species of sub locations. The analysis is based on the product of relative frequency and relative average abundance of a fish species for a certain sub location (Dufrêne and Legendre, 1997). After checking of normality and homogeneity of univariate variables (Shapiro−Wilk, Levene's Test and diagnostic residuals plot) the parametric two-way ANOVA with interaction of variables followed by Tukey's HSD test was used. When the normality and homogeneity assumptions were violated, the non-parametric Kruskal−Wallis one-way multi-comparison tests of package ‘pgirmess’ was used to determine significant differences between locations. We applied Non-Metric Dimensional Scaling (nMDS) using the metaMDS function in package ‘vegan’, based on Bray−Curtis dissimilarities of fish density data, to visualize differences in fish assemblages in the borrow pit and reference area (Oksanen, 2013). Environmental variables were linearly fitted onto the ordination using the envfit function in package ‘vegan’ (999 permutations). We used the ordisurf function to plot a smooth surface onto the ordination in the case of non-linear relationships. When Spearman rank correlation coefficients between a set of variables exceeded 0.9 one of the variables was dropped (Zuur et al., 2007). We used the mantel.correlog function in package ‘vegan’ to check for autocorrelation between the ecological distance matrix and geographic distance matrix (Borcard and Legendre, 2012). For all analyses, we used R version 3.0.1 (R Core Team, 2013).

4.3 Results

Species assemblage, indicator species and biomass

In total, 32 fish species were identified. Fish assemblages in the reference area were dominated by dab (*Limanda limanda*), plaice (*Pleuronectes platessa*), scaldfish (*Arnoglossus laterna*), common sole (*Solea solea*), shorthorn sculpin (*Myoxocephalus scorpius*) and solenette (*Buglossidium luteum*) (Table 4.2). On average, 20.9 ± 12.2 kg WW ha^{-1} of fish and 13.1 ± 1.7 fish species haul^{-1} were caught in the reference area. Species assemblage and biomass at the slope of the borrow pit were not significant different from the reference area (Table 4.4). Plaice, dab, scaldfish, shorthorn sculpin and solenette were most abundant and on average 20.2 ± 6.7 kg WW ha^{-1} and 14.2 ± 1.9 species haul^{-1} were caught. Turbot (*Psetta maxima*) and brill (*Scophthalmus rhombus*) were Dufrêne−Legendre indicator species for the slope of the borrow pit due to a higher relative frequency and average abundance compared to the other sub locations (Table 4.3).

At the crests of the sandbars, plaice, dab, sole, shorthorn sculpin and hooknose (*Agonus cataphractus*) were most abundant and on average, 93.8 ± 47.1 kg WW ha^{-1} and 11.4 ± 2.1 species haul^{-1} were caught. Tub gurnard (*Chelidonichthys lucerna*) is a Dufrêne−Legendre indicator species of the crests of the sandbars (average 4.4 ind. ha^{-1}). Species assemblage at the crest of the oblique sandbar (t_1) significantly differed from the assemblage of the reference area in 2011 (Table 4.4). Biomass at the crests showed a 4-fold increase compared to the reference area (Table 4.2) and was significantly different for the comparisons between crests (t_1 and t_2) and reference area (Table 4.5).

Plaice, dab, european flounder (*Platichthys flesus*), sole and hooknose dominated the troughs of the sandbars. On average, 164.6 ± 205.1 kg WW ha^{-1} and 10.2 ± 2.5 fish species haul^{-1} were caught (Table 4.2). The highest biomass values were found for fish sample 44 and 45 (Fig. 4.3) in the trough of the oblique sandbar (t_1), respectively 522.1 and 484.0 kg WW ha^{-1} (Fig. 4.4). This is a significant 23-fold increase in biomass values compared to the reference area in 2012. Increased biomass of white furrow shell (*Abra alba*) was not detected in the accompanying infaunal sample but in a sample from the trough west of the sandbar, total infaunal biomass reached 61.9 g AFDW m^{-2} with 24.9 g AFDW *A. alba* m^{-2}.

Significant differences in species assemblage between the south-eastern (SE) deep area and the troughs of the parallel sandbar were found (Table 4.4). This difference is also clearly visible in the nMDS ordination (Fig. 4.6) as t_2 samples from the trough ended up in the left region surrounded by reference and deep north-western (NW) samples (t_{recent}). Furthermore, fish biomass values in the troughs of the parallel sandbar (31.4 kg WW ha^{-1}, Table 4.2) differed significantly from the locations sampled one year after cessation, which harboured 175.1 and 413.9 kg WW ha^{-1} (Table 4.5).

The accompanying bottom sledge samples of the fish samples from the troughs of the parallel sandbar was characterised by a high biomass of serpent star *Ophiura ophiura*.

Table 4.2 Total biomass, total species richness, top ten species density (number of fish per hectare) with standard deviation (sd) and environmental and biological variables at reference locations for 2010–2012 and at the sub locations in the sand extraction area for recent sand extraction (t_{recent}) and 1 (t_1) and 2 year after sand extraction (T_2).

	2010		2011		2012		average	
	mean	sd	mean	sd	mean	sd	mean	sd
Reference area								
Water depth (m)	18.8	1.7	21.7	2.8	26.5	5.2	22.2	4.3
D_{50} (µm)	267.6	20.6	290.3	42.0	307.5	55.3	288.8	41.6
Very fine sand (%)	0.6	0.3	0.2	0.2	0.1	0.1	0.3	03
OM (%)	-	-	-	-	0.8	0.1	0.8	0.1
Mud (%)	0.9	0.4	0.1	0.2	0.0	0.0	0.3	0.5
A. alba (g AFDW m^{-2})	0.3	0.3	0.0	0.0	0.0	0.0	0.1	0.0
O. ophiura (g WW m^{-2})	2.3	2.1	3.4	0.9	1.8	0.0	2.7	1.4
Biomass (kg ha^{-1})	17.8	6.2	20.0	13.3	25.7	15.8	20.9	12.2
Species richness ($haul^{-1}$)	12.0	1.8	14.0	1.6	12.8	1.0	13.1	1.7
Limanda limanda	57.4	30.1	100.3	79.8	171.4	123.1	107.8	89.9
Pleuronectes platessa	102.7	59.6	89.0	107.7	54.2	23.3	83.4	78.8
Arnoglossus laterna	12.1	6.59	24.4	13.9	10.0	4.5	17.3	12.0
Solea solea	24.3	26.9	11.3	8.0	5.5	1.9	13.2	15.4
Buglossidium luteum	4.2	3.4	18. 8	21.5	3.4	3.4	10.8	16.3
Agonus cataphractus	7.3	6.7	11.9	21.3	4.9	0.3	8.8	14.6
Myoxocephalus scorpius	2.8	2.0	10.7	4.1	5.9	4.0	7.4	4.9
Platichthys flesus	9.9	6.5	2.9	3.1	4.4	4.1	5.2	5.1
Chelidonichthys lucerna	1.4	1.0	4.4	2.2	3.2	0.8	3.3	2.0
Pomatoschistus sp.	10.8	19.3	0.0	0.0	0.1	0.3	2.9	10.2
Slope of the borrow pit								
Water depth (m)	-	-	23.5	4.4	27.5	0.0	24.8	4.0
D_{50} (µm)	-	-	280.3	47.5	314.2	73.0	291.6	52.2
Very fine sand (%)	-	-	0.7	0.1	0.7	0.6	0.7	0.3
OM (%)	-	-	-	-	0.9	0.1	0.9	0.1
Mud (%)	-	-	0.1	0.1	2.1	3.0	0.7	1.7
A. alba (g m^{-2})	-	-	0.0	0.0	0.0	0.0	0.0	0.0
O. ophiura (g WW m^{-2})	-	-	0.7	0.5	1.2	0.5	0.9	0.5
Biomass (kg ha^{-1})	-	-	17.9	5.6	24.8	8.1	20.2	6.7
Species richness ($haul^{-1}$)	-	-	14.5	2.1	13.5	2.1	14.2	1.9
Pleuronectes platessa	-	-	126	104.5	96.1	64.2	116.0	87.3
Limanda limanda	-	-	69.6	2.8	145.9	53.4	95	46.1
Arnoglossus laterna	-	-	16.6	5.4	12.1	5.7	15.1	5.4
Myoxocephalus scorpius	-	-	12.8	9.8	15.8	0.4	13.8	7.8
Buglossidium luteum	-	-	11.9	7.5	7.3	4.6	10.4	6.6
Solea solea	-	-	8.5	2.9	7.4	1.8	8.1	2.5
Agonus cataphractus	-	-	7.5	6.4	7.0	2.3	7.4	5.1
Scophthalmus rhombus	-	-	7.4	8.2	1.6	0.6	5.5	7.0
Psetta maxima	-	-	7.7	8.2	0.6	0.0	5.4	7.3
Platichthys flesus	-	-	3.1	2.5	6.9	6.0	4.4	3.9
Crests of sandbars								
Water depth (m)	29.0	4.2	29.5	1.8	31.5	1.4	29.9	2.4

D_{50} (µm)	240.8	106.8	270.4	25.7	303.8	16.0	271.5	53.1
Very fine sand (%)	11.9	15.0	5.6	0.2	6.1	1.0	7.5	6.8
OM (%)	-	-	-	-	2.0	0.9	2.0	0.9
Mud (%)	1.7	1.6	7.1	7.5	6.6	0.8	5.4	5.1
A. alba (g AFDW m^{-2})	1.5	2.1	0.2	0.2	0.0	0.0	0.5	1.1
O. ophiura (g WW m^{-2})	0.4	0.2	2.0	0.4	1.2	0.0	1.3	0.7
Biomass (kg ha^{-1})	45.6	21.5	119.3	42.4	103.6	50	93.8	47.1
Species richness ($haul^{-1}$)	11.5	0.7	12	3	10.5	2.1	11.4	2.1
Pleuronectes platessa	272.9	251.4	706.6	348.0	534.1	321.2	533.4	325.1
Limanda limanda	215.4	31.3	477.6	298.1	191.6	49.3	321.0	227.5
Solea solea	48.1	24.5	188.1	202.5	51.1	38.5	109.0	139.7
Myoxocephalus scorpius	58.2	61.6	82.4	58.5	43.4	27.6	64.4	47.2
Agonus cataphractus	19.9	0.8	113.7	153.5	1.4	2.0	54.8	104.6
Platichthys flesus	5.8	2.0	31.6	10.3	37.1	34.6	25.8	20.7
Buglossidium luteum	28.4	31.8	24.8	19.9	0.6	0.9	18.9	21.4
Arnoglossus laterna	23.8	21.2	20.0	12.9	2.0	1.1	15.9	15.0
Merlangius merlangus	5.9	8.3	30.5	31.1	1.3	0.1	15.1	23.3
Trisopterus luscus	33.0	42.5	0.0	0.0	0.0	0.0	9.4	23.7
Troughs of sandbars								
Water depth (m)	35.7	9.3	37.4	3.7	40.0	0.7	37.4	5.4
D_{50} (µm)	296.1	77.2	232.7	62.9	133.9	2.3	231.9	83.2
Very fine sand (%)	2.1	1.6	10.6	11.5	25.9	0.4	11.2	11.7
OM	-	-	-	-	6.0	0.6	3.3	3.0
Mud	0.3	0.3	2.3	1.0	18.0	2.8	5.1	7.5
A. alba (g AFDW m^{-2})	0.0	0.0	0.1	0.1	20.6	1.6	4.6	9.1
O. ophiura (g WW m^{-2})	0.4	0.4	3.5	1.3	242.2	341.5	55.5	160.6
Biomass (kg ha^{-1})	20.8	7.2	339.1	197.1	31.4	26.9	164.6	205.1
Species richness ($haul^{-1}$)	9.7	1.5	11.0	2.4	9.5	4.9	10.2	2.5
Pleuronectes platessa	69.7	103.6	1780.2	504.8	133.6	152.8	844.1	943.6
Limanda limanda	47.7	52.5	567.5	448.4	72.7	62.2	284.3	385.8
Solea solea	23.7	14.6	373.6	510.1	33.3	33.5	181.3	362
Platichthys flesus	18.1	16.4	182.7	161.5	9	0.9	89.2	133.1
Agonus cataphractus	105.3	60.8	114	116.9	4.2	5.9	86.7	90.9
Myoxocephalus scorpius	26.8	24	122.2	84.0	8.3	11.8	65.1	76.1
Merlangius merlangus	8.2	10.2	36	39.5	4.2	5.9	19.7	29.3
Buglossidium luteum	15.4	18.1	21.6	27.1	4.4	1.7	15.7	20.2
Arnoglossus laterna	9	10.7	23.3	28.4	2.1	2.9	13.8	20.5
Callionymus lyra	6.9	12	7.9	9.4	0.0	0.0	5.8	8.9
Deep (SE)								
Water depth (m)	38.0	-	33.5	-	38.5	0.7	37.1	2.5
D_{50} (µm)	338.3	-	233.7	-	126.9	-	206.5	101.3
Very fine sand (%)	0.0	-	14.9	-	26.0	-	16.8	12.3
OM (%)	-	-	-	-	3.9	-	3.9	-
Mud (%)	0.0	0.0	3.6	0.0	23.3	-	12.5	12.5
A. alba (g AFDW m^{-2})	0.0	-	1.4	-	97.4	-	49.1	55.8
O. ophiura (g WW m^{-2})	0.0	-	4.8	-	2.4	0.0	2.4	2.0
Biomass (kg ha^{-1})	5.5	-	413.9	-	409.5	25.7	309.6	203.3
Species richness ($haul^{-1}$)	7.0	-	15.0	-	8.5	0.7	9.8	3.6

Pleuronectes platessa	4.5	-	3520.1	-	1771.4	295	1766.9	1445.3
Solea solea	9.1	-	607.1	-	211.2	78.7	259.7	254.6
Platichthys flesus	2.3	-	330.8	-	299.3	77.6	232.9	160.8
Limanda limanda	18.1	-	653.8	-	48.9	22.0	192.4	308.2
Myoxocephalus scorpius	5.7	-	110.9	-	70.5	5.5	64.4	43.7
Buglossidium luteum	0.0	-	68.1	-	5.9	1.1	20.0	32.2
Agonus cataphractus	6.8	-	38.9	-	0.0	0.0	11.4	18.6
Arnoglossus laterna	0.0	-	7.8	-	3.3	4.7	3.6	4.2
Trisopterus luscus	0.0	-	13.6	-	0.0	0.0	3.4	6.8
Merlangius merlangus	3.4	-	3.9	-	0.6	0.9	2.1	1.8
Deep (NW)								
Water depth (m)	28.0	-	36.5	2.1	35.3	0.4	34.3	3.7
D_{50} (µm)	130.9	-	483.7	139.5	308.3	-	343	163.1
Very fine sand (%)	25.4	-	2.0	1.7	8.3	-	9.2	9.6
OM (%)	-	-	-	-	2.0	-	-	-
Mud (%)	23.8	-	0.4	0.3	8.2	-	8.2	9.5
A. alba (g AFDW m^{-2})	14.0	-	1.5	1.1	0.0	-	3.1	6.1
O. ophiura (g WW m^{-2})	1.6	-	0.1	0.2	1.3	0	0.9	0.7
Biomass (kg ha^{-1})	28.6	-	26.4	30.6	23.6	3.2	25.7	15.5
Species richness ($haul^{-1}$)	8.0	-	11.0	2.8	7.5	2.1	9.0	2.5
Pleuronectes platessa	181.2	-	386.4	508.7	41.4	3.7	207.3	307.7
Limanda limanda	54.4	-	123.6	135.5	128.5	84.1	111.7	86
Myoxocephalus scorpius	9.1	-	7.8	5.0	34.3	10.1	18.7	15.4
Solea solea	9.1	-	10.7	14.3	13.2	10.9	11.4	9.2
Agonus cataphractus	4.5	-	6.4	9.0	9.7	1.9	7.3	5.1
Platichthys flesus	11.3	-	3.4	4.7	4.5	2.5	5.4	4.3
Buglossidium luteum	0.0	-	8.2	9.3	5.2	7.4	5.4	6.8
Arnoglossus laterna	0.0	-	6.6	7.8	4.2	5.9	4.3	5.6
Merlangius merlangus	6.8	-	0.9	0.6	3.1	4.4	3.0	3.3
Pomatoschistus minutus	0.0	-	3.4	4.7	0.0	0.0	1.3	3.0

The most dominant species in the SE deep area of the borrow pit were plaice, sole, european flounder, dab and shorthorn sculpin. European flounder and plaice are Dufrêne−Legendre indicator species (Table 4.3). In 2011 and 2012, fish biomass significantly increased 20-fold and reached 413.9 and 409.5 kg WW ha^{-1} and 15 and 8.5 fish species were caught (Table 4.2 & Fig. 4.4).

Table 4.3 Dufrêne−Legendre indicator species.

Species	Location	Indval	P value
European flounder (*Platichthys flesus*)	Deep (SE)	0.6	**
Plaice (*Pleuronectes platessa*)	Deep (SE)	0.5	*
Turbot (*Psetta maxima*)	Slope	0.6	***
Brill (*Scophthalmus rhombus*)	Slope	0.5	**
Tub gurnard (*Chelidonichthys lucerna*)	Crest	0.3	*

(***) $0.001 < p < 0.01$, (**); $0.01 < p < 0.05$ (*); $0.05 < p < 0.1$ (.)

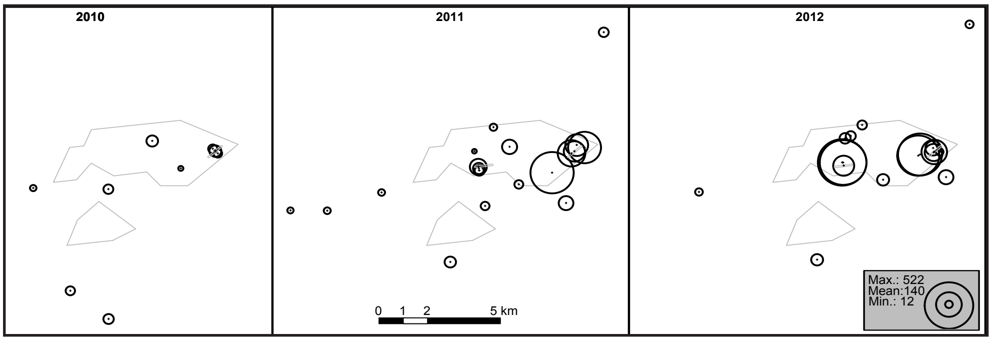


Figure 4.4 Demersal fish biomass in- and outside the borrow pit in 2010, 2011 and 2012. Values are proportional to the radius of the circles in the bubble plot with maximum values converted to bubbles with 1000 m radius. The highest biomass value was found in 2012 at the trough of the oblique sandbar (522 kg WW ha^{-1}).

Demersal fish biomass in the SE deep area remained in 2012 nearly as high as in 2011 but the density of plaice decreased from 3520.1 in 2011 to 1771.4 ind. ha^{-1} in 2012. In 2012, the length of plaice was larger than in 2011, 23.4 cm instead of 15.5 cm, which compensated for the lower observed density in 2011 (Table 4.2 & Fig. 4.5). A similar trend was found in the troughs; in 2012, the average length of plaice was 20.8 cm instead of 15.4 cm in 2011. The average lengths of plaice in the reference area in 2010, 2011 and 2012 was 17.82, 15.29 and 17.11 cm, which means that the deep areas of the borrow pit attracted larger plaice specimen. The length of plaice in the reference area in 2011 was the smallest of the three years, which may explain the smaller length of plaice in the deep areas in 2011. No differences in length of the other dominant fish species were observed.

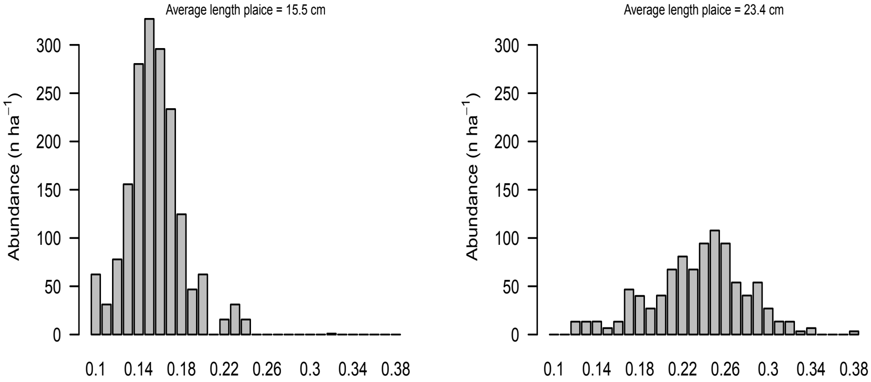


Figure 4.5 Length density distribution of plaice (in m) in the south-eastern deep area of the MV2 borrow pit for 2011 and 2012, respectively 1 and 2 years after cessation ($t_{1 \text{ and } 2}$).

Biomass in the north-western (NW) deep area remained relatively low but just above the reference level, 25.7 ± 15.5 kg WW ha^{-1}. Species richness is significantly lower in the NW deep area, 9 species haul^{-1} compared 13.1 species haul^{-1} in the reference area, (Tukey's HSD, $p < 0.05$). The most dominant species were plaice, dab, shorthorn sculpin, sole and hooknose.

Table 4.4 Differences in demersal fish species composition between locations using permutational multivariate analysis of variance using distance matrices (ADONIS). For the period 2010 - 2012, differences between location, time after sand extraction and combined effects are assessed. For one specific year, only differences between location were tested.

Comparison	*p*-value
Reference : edge 2010 - 2012	
Reference : crest 2010 - 2012	,**,
2010	
2011	*
2012	.
Reference : troughs 2010 - 2012	*,**,
2010	.
2011	*
2012	
Crest : troughs 2010 - 2012	.,*,
2010	
2011	*
2012	
Crest : deep SE 2010 - 2012	*,**,.
2010	-
2011	
2012	
Reference : deep SE 2010 - 2012	**,**,
2010	
2011	
2012	.
Reference : deep NW 2010 - 2012	,*,
2010	
2011	
2012	

***) $0.001 < p < 0.01$, (**); $0.01< p < 0.05$ (*); $0.05 < p <0.1$ (.)

Table 4.5. Tukey's HSD pairwise comparison of log transformed demersal fish biomass among locations with adjusted *p*-values. R: Reference, Sl: Slope, Cr: Crests, Tr: Troughs, SE: South-eastern deep area and NW: North-western deep area. Ref: Reference, 0: Recent sand extraction, 1 and 2: 1 - year after the cessation of sand extraction.

	Ref	SL_0	Cr_0	Tr_0	SE_0	NW_0	Sl_1	Cr_1	Tr_1	SE_1	NW_1	Sl_2	Cr_2	Tr_2	SE_2
Ref	ns	ns	ns	ns	ns	ns	ns	*	***	**	ns	ns	*	ns	***
SL_0		-	ns	ns	ns	ns	ns	ns	**	*	ns	ns	ns	ns	**
Cr_0			-	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	*
Tr_0				-	ns	ns	ns	.	**	**	ns	ns	ns	ns	**
SE_0					-	ns	ns	**	**	*	ns	ns	*	ns	***
NW_0						-	ns	*	***	**	ns	ns	.	ns	***
SL_1							-	ns	***	**	ns	ns	ns	ns	***
Cr_1								-	ns	ns	ns	*	ns	ns	ns
Tr_1									-	ns	*	**	ns	**	ns
SE_1										-	ns	*	ns	*	ns
NW_1											-	ns	ns	ns	*
SL_2												-	ns	ns	**
Cr_2													-	ns	ns
Tr_2														-	**
SE_2															-

(***) $0.001 < p < 0.01$, (**); $0.01< p < 0.05$ (*); $0.05 < p < 0.1$ (.)

nMDS ordinations and associated variables

Fish assemblage and environmental dissimilarities revealed a significant association (Mantel: $p < 0.01$, $r = 0.2181$). Percentage coarse sand was dropped from the analysis because of the collinearity with D_{50}, percentage medium and fine sand and mud content were dropped because of the collinearity with very fine sand. Mantel correlograms analysis showed that autocorrelation was below significance level. All nMDS ordinations had stress values below 0.07, which means an outstanding goodness of fit. For the 2010–2012 survey periods, the ordination showed a significant association with time after the cessation of sand extraction and water depth (Fig. 4.6 & Table 4.6). Median grain size D_{50}, the fraction very fine sand and infaunal white furrow shell biomass were just above the significance level.

In 2010, only water depth showed a significant association with the ordination. In 2011, the very fine sand fraction, time after the cessation of sand extraction, water depth and D_{50} showed significant associations with the ordination. In 2012, infaunal white furrow shell biomass, water depth and the fraction very fine sand showed a strong association with the ordination. The association of the ordination with time after the cessation of sand extraction was just above the significance level. No significant associations were found for total epifaunal biomass and specific species sampled with the bottom sledge, e.g. scavenging brittlestars (*Ophiura sp.*) and predatory flying crab (*Liocarcinus* sp.).

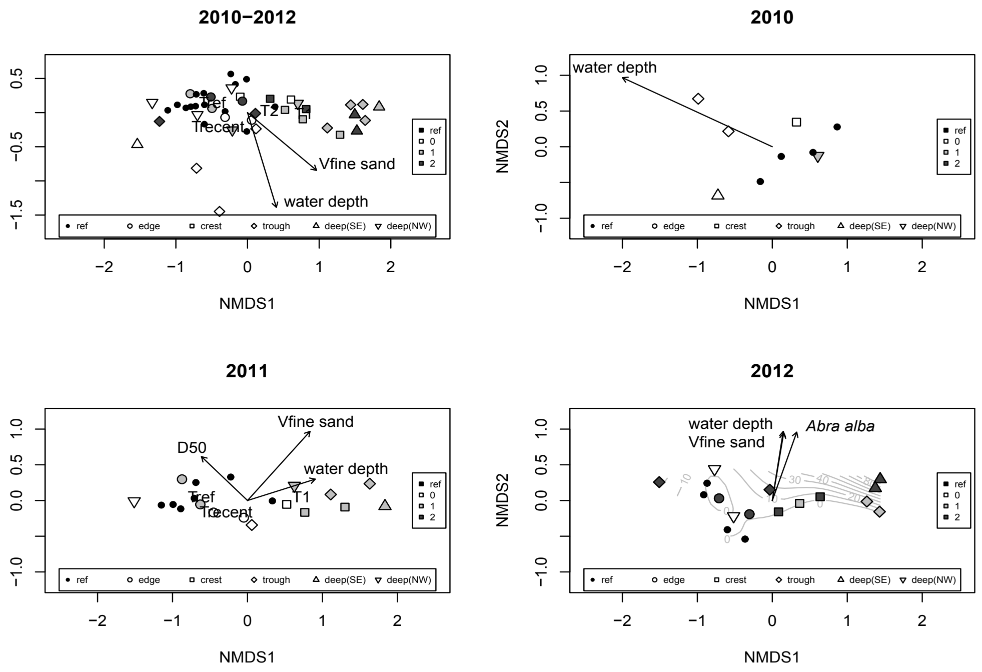


Figure 4.6 nMDS ordination with sites and significant associations with variables. Continuous variables are depicted with arrows and categorical variable time after the cessation, only with text (t_{ref}: reference in black, t_{recent} in white, t_1 in light grey and t_2: dark grey. Sub locations are denoted with symbols, reference as black bullets, slope of the reference area as open large circles, crests of sandbars as squares, trough of sandbars as diamonds, deep (SE) as point-up triangles and deep (NW) as point-down triangles. In 2012, the significant association of the ordination and infaunal white furrow shell biomass is denoted with an arrow and surface plot to show the non-linearity. Stress of all ordinations was below 0.07.

Stomach and intestine contents

Stomach and intestine content of plaice was dominated by undigested crushed white furrow shell remains, dab stomachs end guts were filled with remains of brittlestars (*Ophiura* sp.) and shorthorn sculpin stomachs were filled with whole flying crabs (*Liocarcinus* sp.).

Table 4.6. Multiple regression of environmental variables and nMDS scores for two axis ordination, r^2 is the squared Spearman's rank correlation coefficient and *p*-value.

Variable and period	**r^2 linear**	***p*-value**
2010–2012		
Time after the cessation	0.36	***
Water depth	0.27	***
Very fine sand (vol %)	0.21	*
Abra alba (infaunal)	0.13	·
D_{50}	0.13	·
2010		
Water depth	0.66	*
D_{50}	0.56	·

2011		
Very fine sand	0.58	***
Time after the cessation	0.40	*
Water depth	0.32	*
D_{50}	0.27	*
2012		
Abra alba (infaunal)	0.42	*
Water depth	0.40	*
Very fine sand (vol %)	0.38	*
Time after the cessation	0.41	·

(***) $0.001 < p < 0.01$, (**); $0.01 < p < 0.05$ (*); $0.05 < p < 0.1$ (·)

Colonization of infaunal macrozoobenthos and sedimentary evolution

We observed significant changes in fish assemblage in the borrow pit, which showed a strong association with sediment composition and white furrow shell biomass.

For the fraction of very fine sand, significant differences were found between location and time after sand extraction (Kruskal–Wallis chi-squared = 27.7, df = 5, all $p < 0.001$). In 2011, a significant difference was found between reference and troughs. In 2012, the fraction very fine sand significantly differed between reference area and the SE deep area. In general, the fraction very fine sand decreased at the crests of the sandbars and increased in the troughs and deep areas of the borrow pit. D_{50} at the crests of the parallel sandbar increased from 165 µm in 2010 to 304 µm in 2012 and very fine sand fraction decreased from 22.5% in 2010 to 6.1% in 2012. We observed the opposite for the sediment in the trough of parallel sandbar, D_{50} decreased from 321.8 to 133.9 µm.

Significant differences were found in white furrow shell (*Abra alba*) biomass, between locations and year (2011: Kruskal–Wallis chi-squared = 15.8, df = 5, $p < 0.01$ and 2012: Kruskal–Wallis chi-squared = 12.25, df = 5, $p < 0.05$). In 2011, biomass of white furrow shell was significantly higher at the crests of the sandbars compared to the reference area. White furrow shell is virtually absent in the reference area and the slope of the borrow pit. After 2 years, white furrow shell biomass increased to 20.6 and 97.4 g m^{-2} AFDW for respectively the troughs of sandbars and the south-eastern deep area.

Fishing activity

Based on Vessel Monitoring through Satellite (VMS) data (Hintzen et al., 2013) seabed disturbing fishing activity in the borrow pit was virtually absent in 2010 (Fig 4.7). In 2011, fishing activity mainly occurred in the northern area of the northern borrow pit. In 2012, fishing activity in the borrow pit exceeded the fishing activity level of the reference area. Furthermore, fishing activity in 2012 almost equalled the fishing activity level before sand extraction (Steenbergen and Machiels, 2013).

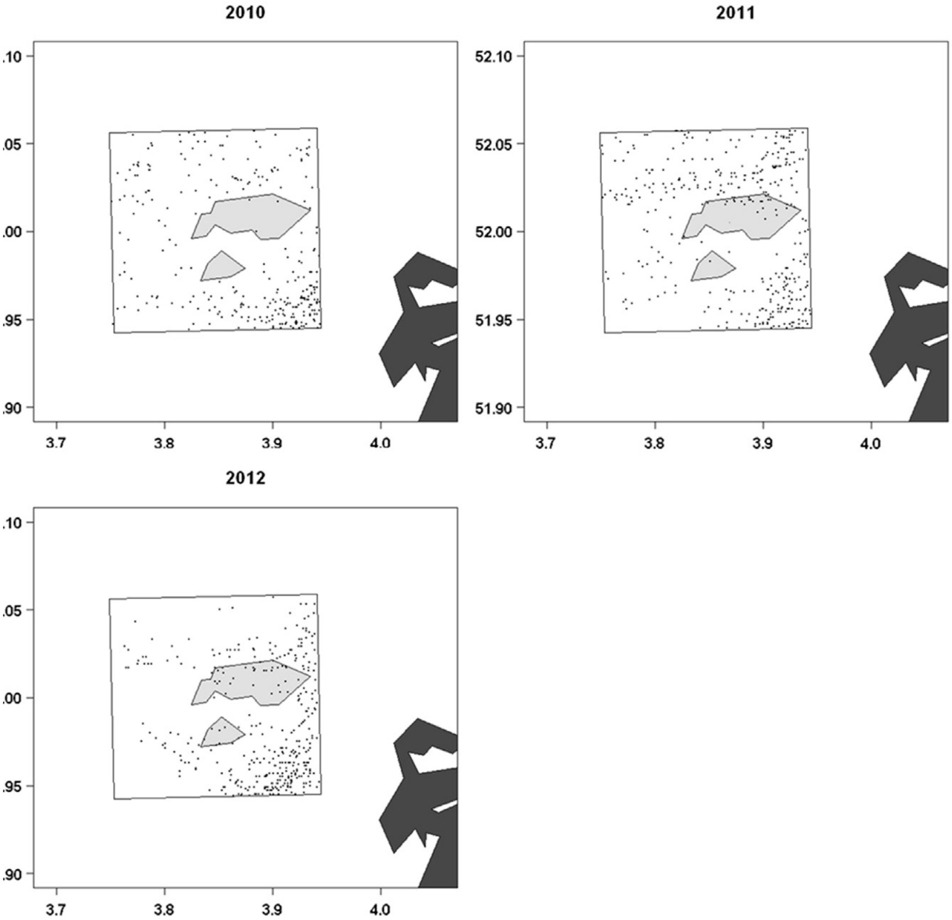


Figure 4.7 Fishing activity, in 2010–2012 derived from Vessel Monitoring through Satellite (VMS) data of all seabed disturbing fish gears Beam Trawl (>300 and <300 horsepower), Bottom Otter Trawl, Bottom Pair Trawl and Fly Shooting Seine. Area in green is the Maasvlakte I area, light grey coloured area are the northern and smaller southern borrow pit.

4.4 Discussion

Fish biomass significantly increased 20-fold in the south-eastern deep area of the MV2 borrow pit and 5-fold on the crests of the landscaped sandbars compared to the reference area and recently extracted areas. This increased biomass is associated with a significant increase in biomass of infaunal white furrow shell (*A. alba*). In the north-western deep area biomass values remained relatively low but just above the reference level, 25.7 ± 15.5 kg WW ha^{-1} probably due to ongoing sand extraction activities and the absence of the increase in white furrow shell biomass.

The highest species richness, 14.2 species haul^{-1}, was found at the slope of the borrow pit with turbot (*Psetta maxima*) and brill (*Scophthalmus rhombus*) as indicator species. These species are at a length of 20 cm known to forage on mobile prey while severely relying on eyesight (Western, 1971; Vinagre et al., 2011; Beyst et al., 1999; de Groot, 1971). Instead of foraging in more turbid conditions, the edge of the borrow pit may be more suitable while still favouring from increased fish biomass in the surroundings. Species richness in the borrow pit is lower compared to the reference area. Inside the borrow pit, on average, 10.5 species per haul were found compared to 13.1 species haul^{-1} in the reference area. The lowest species richness was found in the north-western deep area, 9.0 species haul^{-1} probably again due to continuing sand extraction. Comparisons of species richness between reference area and locations in the borrow pit are potentially biased due to differences in sampled surface area.

Based on a study in an extraction site in France, Desprez (2000) stated that sand extraction on the long term might create new habitats such as the presence of boulders and higher heterogeneity of sediment and favour an increase in the richness of benthic fauna and fish. A qualitative analysis of fish assemblage was however lacking.

Next to differences in biomass and species richness, significant differences in fish species assemblage between reference area and borrow pit were found. Dab (*Limanda limanda*) and plaice (*Pleuronectes platessa*) were the most abundant species in the reference area whereas in the borrow pit plaice was more abundant. This difference is again maybe due to the increase in white furrow shell biomass, which may be a more preferred prey item of plaice. In a smaller borrow pit on the Belgian Continental Shelf with shallow sand extraction dab was also more abundant than plaice although a clear change in fish distribution was not observed maybe due to continuous sand extraction or less pronounced differences in bathymetry and sediment characteristics (Maertens, 1988). Epibenthos in the Belgian borrow pit was dominated by the predatory common starfish *Asterias rubens* and the scavenging serpent star *Ophiura ophiura.*

In 2011, significant differences in fish assemblage between crests and troughs of the sandbars were found and tub gurnard (*Chelidonichthys lucerna*) is an indicator species of the crests which is maybe induced by differences in macrozoobenthic assemblage. The significant difference in fish assemblage may be an indication of the applicability of landscaping techniques to induce heterogeneity of the seabed, although it remains difficult to draw a strong conclusion due the lack of replication of the experiment. Comparable differences in species assemblage also occurred on the crests of natural sandbanks. Early-life history stages of the lesser weever (*Echiichthys vipera*) were found to be more abundant at the crests compared to the troughs (Ellis et al., 2011).

Several environmental variables may be responsible for the differences in fish species assemblage. In general, fishing activity may also play a role but in 2012, activity in the borrow pit was almost equal as in the reference area (Steenbergen and Machiels, 2013). Therefore, differences in fish assemblage and biomass between dredged and non-dredged areas are not likely to be induced by differences in fishing intensity. In 2010, only water depth showed a significant association with the ordination. In 2011, the very fine sand fraction, time after the cessation of sand extraction, water depth and D_{50} showed significant associations with the ordination. In 2012, infaunal white furrow shell biomass, water depth and the fraction very fine sand showed a significant association with the ordination. In 2011, observed differences in the ordination were not yet associated with infaunal white furrow shell biomass, maybe due to the time gap between the two sampling activities. Macrozoobenthos sampling occurred at the end of June 2010, the start of May 2011 and at the end of April 2012 while fishing occurred mid-July 2010, at the end of July 2011 and mid-June 2012.

Sell & Kröncke (2013) concluded that fish assemblages on the Dogger Bank (North Sea) were linked to both biotic and abiotic habitat characteristics, density of specific fish species could be linked with individual in- and epifauna species. In the Western Baltic, white furrow shell comprised 24% of the diet of plaice (Arntz and Finger, 1981; Rainer, 1985) and a comparison of present-day diet and the diet op plaice at the beginning of the 20th century suggested that the preponderance of polychaetes has increased and that of bivalves decreased (Rijnsdorp and Vingerhoed, 2001). White furrow shell is a deposit feeding bivalve and tend to prefer fine-grained sediments with a median grain size between 50 and 250 µm and a mud content of 10–50% (Degraer et al., 2006). An increase of Tellinid shellfish (e.g. white furrow shell), plaice and common sole was found at deposition areas around aggregation sites in France (Desprez and Lafite, 2012). We found that stomach and intestines of plaice were mainly filled with undigested remains of white furrow shell. Stomach content analysis of the other fish species also revealed specific preference of prey items but this was not tested with a statistical analysis due to the lack of enough replicates. Prey items of dab were dominated by brittlestars (*Ophiura spp.*), a similar preference was also found by other researchers (Hinz et al., 2005). All shorthorn sculpin stomachs contained swimming crabs (*Liocarcinus* sp.) and this preference was also found in other studies (Link and Almeida, 2002; Western, 1971).

Time after the cessation of sand extraction is an important variable. Directly after sand extraction, fish biomass values are similar to the reference area. Fish biomass values increased in 2011 and 2012. White furrow shell biomass is showing the same pattern, median grain size in the borrow pit is decreasing and the fraction of very fine sand in the sediment is rising. For shallow sand extraction in the North Sea, recovery time of benthic assemblages is estimated to be 4–6 years (van Dalfsen et al., 2000; Boyd et al., 2005; van Dalfsen and Essink, 2001). In a region with high extraction intensity in Area 222, a large extraction site off the south-east coast of England, greater macrozoobenthic variability was observed and complete recovery was not yet reached after 11 years after the cessation of extraction (Wan Hussin et al., 2012).

A study in a deep temporary borrow pit with 6.5 million m^3 of sand extracted, an initial water depth of 23 m and extraction depth between 5–12 m revealed a sedimentation rate of 3 cm per year (Boers, 2005). Thatje et al. (1999) found a continuous sedimentation rate of around 50 cm per year inside a 700 m wide and 48m deep natural sea-floor crater 20 miles off the coast of Germany with an initial water depth of 34 m. Mud content inside the crater increased from 5% in 1963 up to 40% in 1995 and the benthic community was characterised as a sea urchin–brittlestar association (*Echinocardium cordatum* and *Amphiura Filiformis*). Considering these sedimentation rates backfilling of the MV2 borrow pit may take decades or longer, resulting in a prolonged and more pronounced effect on macrozoobenthos and fish assemblages in the future.

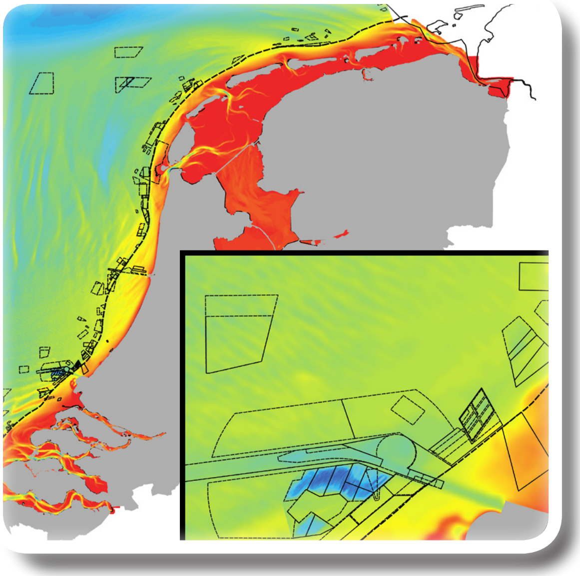
The increase in length of plaice in the troughs and south-eastern deep area (T_2) may be the result of a residing cohort within the borrow pit, plaice with a body size ≈16 cm is 1.5 years old and with body size ≈22 cm is 2.5 years old (Bolle et al., 2004). The increase in length of plaice may also be related to prey-size preferences of juvenile plaice. In one year specimens of *A. alba* reached lengths of 12–14 mm, in two years 13–16 mm with a maximum length of 20–25 mm (Holtmann et al., 1996). All sediment parameters remained in the same range except OM values, which was on average 3.9% in the south-eastern deep area and 6.0% in the troughs of the parallel sandbar. However, the statistical analysis revealed no significant association of the ordination in 2012 with OM. Dissolved oxygen (DO) levels are influenced by factors such as; OM, water depth, temperature and water circulation (Eldridge and Roelke, 2011). The increase in length may also be induced by avoidance of the south-eastern deep area by smaller plaice, which are more sensitive to reduced DO levels (Rabalais et al., 2001; Gray et al., 2002; Miller et al., 1995). DO levels may even be more reduced in the troughs resulting from the specific bathymetry and larger water depth, which may have led to the significant drop in biomass values from the 23-fold increase in 2011 down to reference level and significant change in species assemblage in 2012. Fish biomass was found to be significantly correlated with oxygen concentration and a reduction in fish biomass was observed when oxygen concentration in the bottom water dropped below 3 mg l^{-1} (Petersen and Pihl, 1995). Next to the difference in OM and possible differences in DO levels, differences in macrozoobenthic assemblage were also present. The accompanying bottom sledge samples of the fish samples from the troughs of the parallel sandbar were characterised by high biomass values of serpent star *Ophiura ophiura* (242.2 g WW m^{-2}).

Limitations of present study and recommendations

Studying the distribution of demersal fish for two years is not enough to understand the final impact of deep and large-scale sand extraction on demersal fish. Conclusions on sedimentary evolution are based on a relatively small number of boxcore samples and short monitoring period. More sediment samples from the 2010–2012 surveys will be analysed in future work. On-going sedimentation and a rise of mud content up to 40% can be expected (Thatje et al., 1999). The highest encountered mud content in the second Maasvlakte extraction site was 23.3% in the south-eastern deep part in 2012. Fish surveys were conducted mid-July 2010, at the end of July 2011 and mid-June 2012 while the oxygen concentration reaches a minimum at the end of the summer and a maximum in May (Boers, 2005). The occurrence of temporal hypoxia and possible detrimental effects on fish and macrozoobenthos cannot be excluded with our data. We recommend monitoring of demersal fish and macrozoobenthic assemblage and accompanied sediment variables for a longer period, at least for a period of six years, i.e. the estimated recovery time of shallow sand extraction. Six years of monitoring may be even insufficient because of larger differences resulted from the large-scale and deep sand extraction.

Chapter 5

Ecosystem-based design rules for marine sand extraction sites



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Abstract

The demand for marine sand in the Netherlands as well as globally is increasing. Over the last decades, only shallow sand extraction of 2 m below the seabed was permitted on the Dutch continental shelf (DCS). To guarantee sufficient supply and to decrease the surface area of direct impact, the Dutch authorities started to promote sand extraction depths over 2 m for sand extraction projects over 10 million m^3. The ecological effects of deep sand extraction, however, are still largely unknown. Therefore, we investigated short-term effects (0–2.5 y) of deep sand extraction (20–24 m) and compared these with other Dutch case studies such as, regular shallow sand extraction on the DCS (2 m) and an 8 m deepened shipping lane.

For intercomparison between case studies we used tide-averaged bed shear stress as a generic proxy for environmental and related ecological effects. Bed shear stress can be estimated with a two-dimensional quadratic friction law and showed a decrease from 0.50 to 0.04 N m^{-2} in a borrow pit in 20 m deep water with extraction depths up to 24 m. Borrow pits with a tide-averaged bed shear stress of around 0.17 N m^{-2} may lead to enhanced macrozoobenthic species richness and biomass. Below a tide-averaged bed shear stress of 0.08 N m^{-2}, increasing abundance of brittle stars, and below 0.04 N m^{-2} an over-dominance of epifaunal brittle stars may be expected and detrimental effects such as high sedimentation rates and reduced dissolved oxygen levels may emerge.

Ecological data and bed shear stress values were combined and transformed into ecosystem-based design (EBD) rules. At higher flow velocities and larger water depths, larger extraction depths can be applied to achieve desired tide-averaged bed shear stresses for related ecological effects. The EBD rules can be used in the early-design phases of future borrow pits to simultaneously maximise sand yields and decrease the surface area of direct impact. The EBD rules can also help in implementing the Marine Strategy Framework Directive (MSFD) guidelines and moving towards or preserve Good Environmental Status (GES).

5.1 Introduction

Coastal zones are characterised by many human activities such as fishing, shipping, wind farming, dredging, disposal of dredged sediment, beach nourishments, and the extraction and transport of oil, gas and aggregates. These activities have different impacts on the marine environment and most of them are likely to intensify in the future (Jongbloed et al., 2014). Marine sand extraction in the Netherlands as well as globally is also intensifying (Stolk and Dijkshoorn, 2009; ICES, 2014a; Peduzzi, 2014). In the Netherlands, around 26 million m^3 of marine sand is extracted annually with 12.5 million m^3 used for coastal nourishments (Stolk and Dijkshoorn 2009; ICES, 2014a). An increase up to 40–85 million m^3 of coastal nourishments is anticipated to counteract effects of sea level rise (Deltacommissie, 2008). Considerable volumes are extracted in surrounding countries, in the UK 16.8 million m^3, in France 12 million m^3 and in Denmark 10.5 million m^3 per year. In Belgium, each year almost 4 million m^3 of sand is extracted and 2.5 million m^3 of sand is imported from the Netherlands (ICES, 2014a).

Over the last decades, sand extraction depths were limited to 2 m below the seabed on the Dutch continental shelf (DCS). The potential for sand extraction depths over 2 m was first explored in 1999 during the PUTMOR study, in a deep borrow pit in front of the Port of Rotterdam (PoR) with sand extraction depths between 5–12 m (Boers, 2005). The PUTMOR study concluded that there were no indications that deep sand extraction would lead to unacceptable effects and that recovery of benthic assemblages would be possible (Boers, 2005). Deep sand extraction was therefore considered to be a promising alternative for sand extraction projects over 10 million m^3 of sand. For the construction of Maasvlakte 2 (MV2), a 20 km^2 seaward expansion of the PoR, the Dutch authorities permitted sand extraction deeper than the common 2 m, primarily to decrease the surface area of direct impact. Between 2009 and 2013, approximately 220 million m^3 of sand was extracted from the MV2 borrow pit with an average extraction depth of 20 m under the seabed. To guarantee sufficient supply of marine sand in the intensively used coastal zone, the Dutch authorities now allow extraction depths larger than 2 m when sand extraction projects are larger than 10 million m^3 (IDON, 2014).

Although larger sand extraction depths clearly reduce the surface area of direct impact, the effects on marine life on the DCS is still largely unknown. Our objective is to compare effects of extraction depths on macrozoobenthos and demersal fish, and to recommend on optimised extraction depths for future borrow pits. Ecological effects of three case studies were investigated:
1: regular shallow sand extraction (2 m),
2: a deepened shipping lane (8 m) and
3: deep sand extraction (20–24 m) in a large borrow pit.

Macrozoobenthos in the southern North Sea correlated with sediment parameters (Heip et al., 1992; Künitzer et al., 1992; Holtmann et al., 1996; Degraer et al., 1999; van Hoey et al., 2007; van Hoey et al., 2004; Degraer et al., 2008; Verfaillie et al., 2009).

Next to sediment parameters, salinity (Reiss et al., 2010; Reiss et al., 2011; Callaway et al., 2002), and bed shear stress (Ysebaert et al., 2003; Herman et al., 2001; de Jong et al., 2015a) also influence macrozoobenthos. Bed shear stress is the amount of force exerted by flowing water per unit of area of seabed and plays a role in sediment transport processes, the formation of bedforms, and sedimentation or erosion of the seabed. Bed shear stress also influences grain size, mud and sediment organic matter content of the sediment. Sediment grain size in the Dutch coastal zone is positively correlated with bed shear stress (spearman rank correlation: around +0.4) (de Jong et al., 2015a). On the crests of sand waves, shear stress values are generally higher (~0.6 N m^{-2}) and the sediment is coarser (~300 μm), whereas in troughs, shear stress is lower (0.44 N m^{-2}) and the sediment is finer (~280 μm) (de Jong et al., 2015a). Due to sand extraction, larger differences in bed shear stress can be expected and correlations between sediment parameters and bed shear stress may even become more pronounced. In the UK, suggested limits for acceptable changes in sediment grain size after marine aggregate extraction were based on the natural range found in the wider region of the extraction sites (Cooper, 2012). Sediment characteristics after deep sand extraction continue to change due to sedimentation of fine sediment until the borrow pit is filled (de Jong et al., 2015b; Desprez, 2000; Thatje et al., 1999). Grain size is therefore in our opinion not the best candidate for setting limits for acceptable changes in large-scale and deep borrow pits. For intercomparison between case studies, we therefore used tide-averaged bed shear stress as a generic proxy for environmental and related ecological effects. Ecological data and bed shear stress values were combined and transformed into ecosystem-based design (EBD) rules. These rules can be used in the design of future borrow pits to maximise sand yields and simultaneously decrease the surface area of direct impact for different ecological scenarios.

We aim to answer the following questions:
(i) What are the ecological effects of the different sand extraction depths on the Dutch continental shelf?
(ii) What are the optimized extraction depths to achieve desired bed shear stresses and related ecological effects for different pre-extraction water depths and flow velocities?
(iii) What role can ecosystem-based design rules based on bed shear stress play in the design of future borrow pits outside the DCS?

5.2. Description of different case studies of sand extraction depths on the Dutch continental shelf.

We describe the following case studies with different extraction depths on the Dutch continental shelf (DCS):
(i) regular shallow sand extraction,
(ii) an 8 m deepened 'Euromaasgeul' shipping lane and
(iii) deep and large-scale sand extraction in the Maasvlakte 2 (MV2) borrow pit with ecological landscaped sandbars.

Shallow sand extraction

Before 1987 less than 5 million m^3 of marine sand was extracted annually from the Dutch continental shelf (DCS) which increased to nearly 20 million m^3 in 1995 (Fig. 5.1). From 1996 onwards, around 26 million m^3 of marine sand was extracted annually for coastal nourishments and construction purposes (ICES, 2014a; Stolk and Dijkshoorn, 2009). Generally, only sand extraction to a depth of 2 m below the seabed is allowed and only in the area between the continuous 20 m isobath and the 12 nautical mile boundary (Fig. 5.2) (IDON, 2005; IDON, 2014). Between 2006 and 2014 the surface area impacted by sand extraction increased from 7.5 to 45 km^2 (ICES, 2014a).

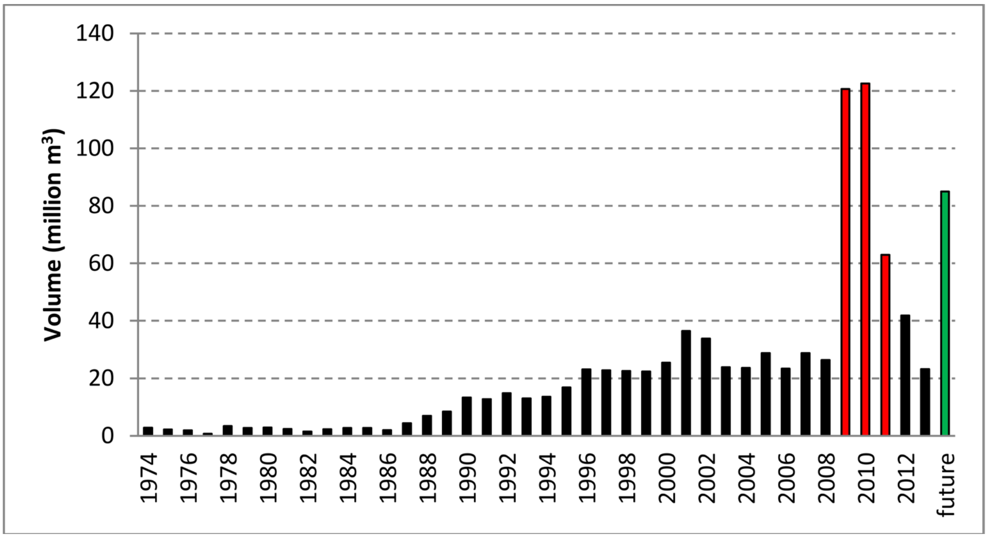


Figure 5.1 Total volume of extracted marine sand in million m^3 per year on the DCS. The red peak in 2009–2012 is due to the large-scale and deep sand extraction for MV2 and the green bar is the anticipated increase for nourishments to counteract sea level rise in the Netherlands. Source Rijkswaterstaat (ICES, 2014a).

Shipping lane

The Euromaasgeul is a 57 km long, 23 m deep shipping lane and located north of the MV2 borrow pit (Fig. 5.2, no. 2). The channel was created in the 1970's to guarantee access to the

PoR. Fine dredged material from the entrance of the shipping lane is dumped at the deepened disposal site ‘Verdiepte Loswal’ (Fig. 5.2, no. 3) and the coarse fraction at disposal site North ‘Loswal Noord’ near the entrance of the PoR. In the specific sampling area within the shipping lane, 8 m sand was extracted.

Maasvlakte 2 (MV2) borrow pit

For the harbour extension Maasvlakte 2 (MV2), approximately 220 million m^3 of sand was extracted between 2009–2012 (Fig. 5.1) with an average extraction depth of 20 m below the seabed (Fig. 5.2, no. 1). This reduced the surface area of the borrow pit from 110 km^2 at 2 m extraction depth to 11 km^2 at 20 m extraction depth (de Jong et al., 2014; de Jong et al., 2015b). The borrow pit is situated in front of the Port of Rotterdam (PoR) outside the continuous 20 m isobath and is 2 km long and 6 km wide (Fig. 5.2, no. 1). An exclusion area, consisting of non-erodible clay covered by a 1 to 4 m thick layer of sand with sand waves, separates a larger northern and smaller southern borrow pit (Stolk, 2014; Klein and van den Boomgaard, 2013). Sediment in the surrounding area consists of fine to medium sand with small quantities of mud, very fine sand, and sediment organic matter content.

Ecological landscaping

In the MV2 borrow pit, ecosystem-based landscaping techniques were used. Two sandbars mimicking natural sand ridges, were left behind after sand extraction, to increase habitat heterogeneity and to influence post-dredging macrozoobenthic and demersal fish assemblages (de Jong et al., 2014, 2015b; van Dalfsen and Aarninkhof 2009; van Raalte et al., 2007).

One sandbar parallel to the tidal current was completed in spring 2010. This parallel sandbar has a length of 700 m, a width at the crest of 70 m, and slopes of 140 m. The crest of the sandbar is located at a water depth of 30 m and the troughs are more than 40 m deep. In 2011, the second sandbar with an orientation oblique to the tidal current was completed. The length and width are similar to the parallel sandbar but, due to time constraints, the difference in depth between crest and trough is less pronounced. The crest is situated at a water depth of 28 m and the northern trough is 36 m deep. A narrow and 32 m deep trench separates the crest from the slope of the borrow pit (de Jong et al., 2014; de Jong et al., 2015b).

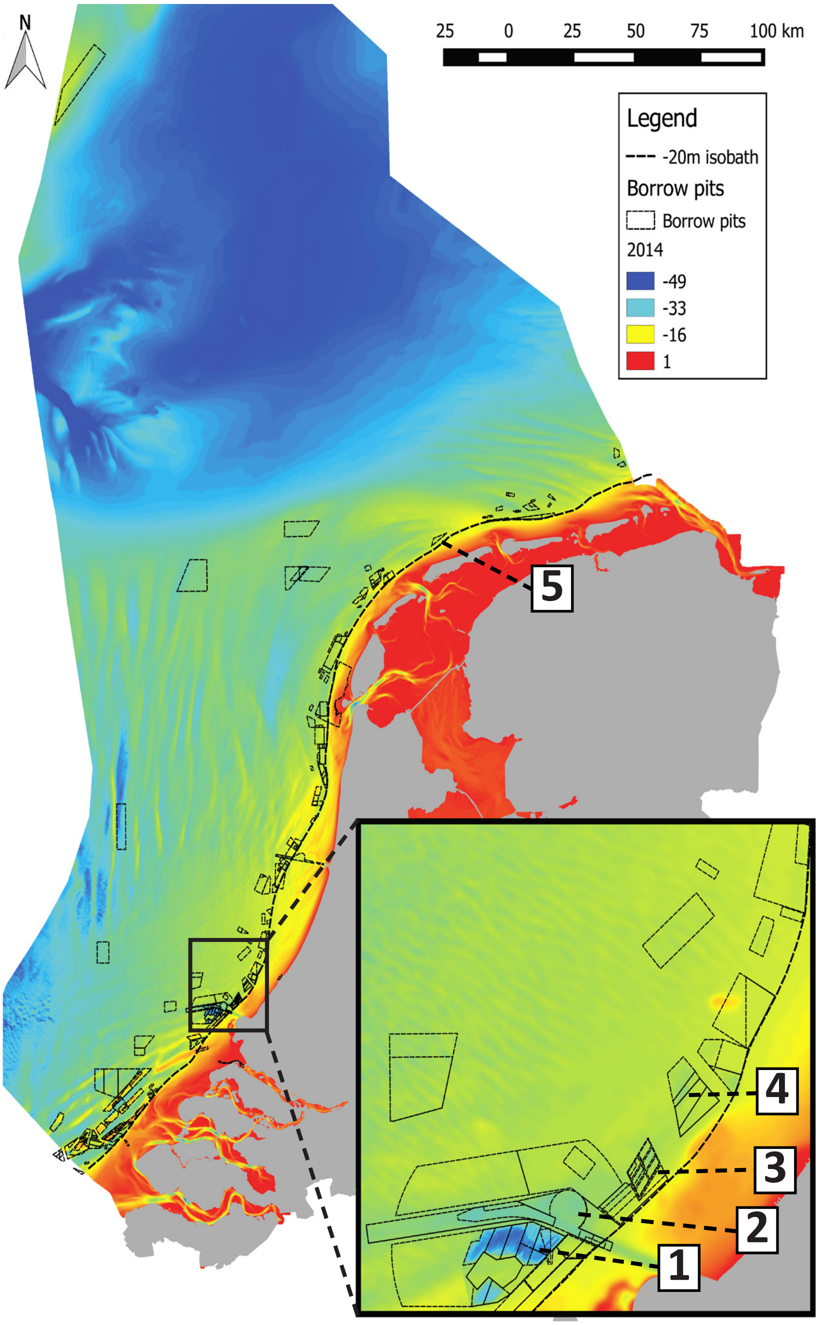


Figure 5.2 Solitary borrow pits (black hatched boxes) with shallow extraction (2 m) except, **1**–**4**. The 20 m deepened Maasvlakte 2 borrow pit is denoted with (**1**), the 8 m deepened shipping lane (**2**), the 5–12 m deep temporary borrow pit (PUTMOR) which is currently used as disposal sites for dredged fine sediment (**3**), a 6 m deep borrow pit for the 'Sand Engine' (**4**) and the shallow sand extraction site North of the barrier island Terschelling (**5**). The inset shows **1**–**4** in higher detail. Source: Rijkswaterstaat.

5.3 Effects of different extraction depths and ecological landscaping on the benthic ecosystem

Here, we summarise the effects of different extraction depths of the case studies and ecological landscaping on the benthic ecosystem and sediment characteristics.

Shallow sand extraction (2 m)

The effects of shallow sand extraction on macrozoobenthos were studied at a site North of the Dutch barrier island Terschelling (Fig. 5.2 no. 5). A volume of 2.1 million m^3 of marine sand was extracted from a 1.4 km^2 large area with pre-extraction water depths of 20–23 m, extraction depths of 1.5 m (van Dalfsen et al., 2000) and depth-averaged flow velocity of 0.5 m s^{-1} (Tonnon et al. 2013). Recovery time of macrozoobenthos to pre-extraction conditions in terms of species assemblage, species richness and biomass at Terschelling is estimated to be 4–6 years (van Dalfsen and Essink, 2001; van Dalfsen et al., 2000). No changes in sediment characteristics after sand extraction were observed, only OM appeared to be lower after dredging (van Dalfsen et al., 2000). Similar recovery times were found for other shallow borrow pits in France near Dieppe (Desprez, 2000; Desprez et al., 2010; Desprez et al., 2014) with water depths of 10–15 m, depth-averaged flow velocities of 1.5 m s^{-1} (Le Bot et al., 2010) and extraction depths up to 2 m (Desprez et al., 2014). In the UK, recovery times at site Area 222, a borrow site 20 miles off Felixstowe at the southeast coast of England were 7 years at low dredging intensity (Boyd et al., 2005) and 15 years at high dredging intensity (Waye-Barker et al., 2015). The water depth in Area 222 varied from 35 m at the high intensity dredging area, 27 m in the low dredging intensity area and 32 m in the reference area and extraction depths were in the same order as for the Terschelling case (Cooper et al., 2013). The depth-averaged flow velocity in Area 222 is around 1.1 m s^{-1} (Boyd et al., 2003).

Extraction in the shipping lane (8 m)

During the baseline study of the PoR in 2006 and 2008, in- and epifaunal samples were collected in the ‘Euromaasgeul’ shipping lane (Fig. 5.2, no. 3). In 2012, additional samples were taken here: 4 in- and 3 epifaunal. Full details on sampling procedures are given in chapter 2–4. Maintenance dredging activities were not carried out in the last years in the specific sampling area (Rijkswaterstaat, pers. comm.). In 2006, 2008 and 2012, a significant distinct and very productive and species-rich assemblage dominated by white furrow shell (*Abra alba*) was observed in the shipping lane. Average biomass of in- and epifauna was respectively 27.5 g AFDW m^{-2} and 18.8 g WW m^{-2} and significantly higher than reference values, respectively 11.7 AFDW m^{-2} and 2.6 g WW m^{-2} (de Jong et al., 2015a,b) (Fig. 5.3). The median grain size of the sediment in the shipping lane in 2006, 2008 and 2012 decreased (respectively 229, 194 and 135.1 µm), and mud (respectively 8, 3, 25%) and sediment organic matter increased (respectively 1, 1, 4%) (de Jong et al., 2015a,b).

Short term effects of deep and large-scale MV2 borrow pit (20–24 m)

In the MV2 borrow pit, macrozoobenthic species composition and biomass was significantly correlated with time after cessation of sand extraction and with sediment and hydrographic characteristics (de Jong et al., 2015b). Two years after cessation of sand extraction, infaunal biomass increased almost 7-fold and epifaunal biomass increased 12-fold in the 40 m deep areas (20 m extraction depth) of the MV2 borrow pit (Table 5.1). Species composition changed significantly and white furrow shell (*Abra alba*) became abundant in terms of density and biomass. Demersal fish biomass increased 20-fold (Table 5.1) and species composition was significantly different in the MV2 borrow pit compared to the reference area (de Jong et al., 2014). Next to ecological differences, sediment characteristics also changed significantly. Sediment grain size decreased and the fraction very fine sand, mud and OM increased. In the deepest parts of the borrow pit (44 m, 24 m extraction depth), infaunal biomass was only 2-fold higher compared to reference values (Table 5.1). Epifaunal biomass, however, increased more than 67-fold and species composition changed from *A. alba* to brittle stars whereas demersal fish biomass dropped almost to reference levels (de Jong et al., 2015b). Next to the large faunal differences, a sedimentation rate of up to 75 cm y^{-1} was observed. The strong changes may also resulted from lowered dissolved oxygen levels (de Jong et al., 2015b). The ecological and environmental effects of the different case studies are summarised in Fig. 5.3 and Table 5.1.

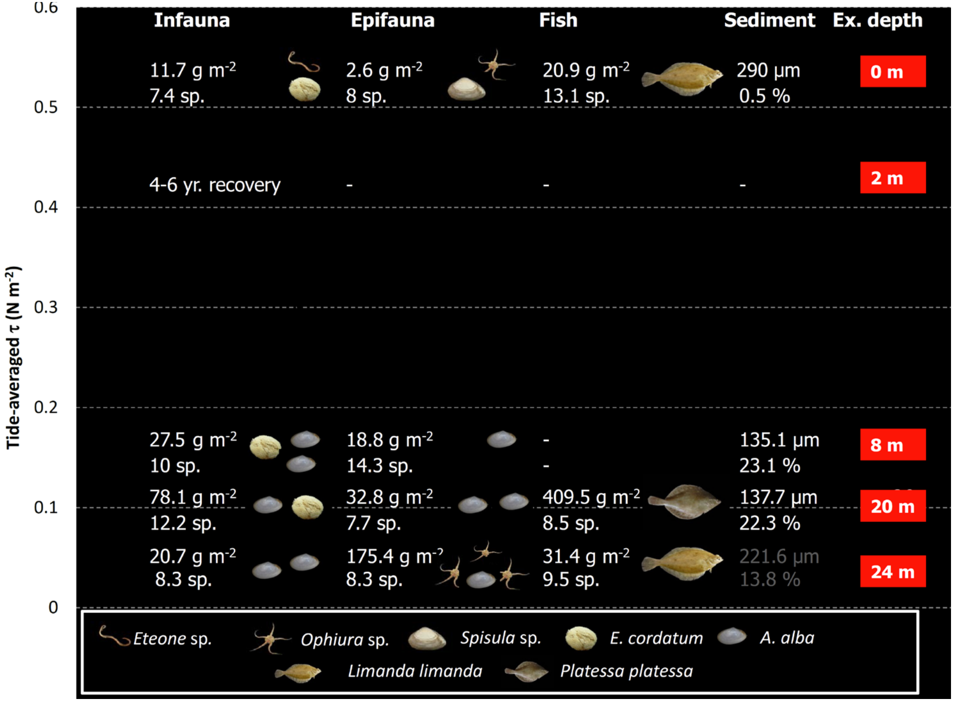


Figure 5.3 Summary of faunal assemblages, median grain size and mud content of the different extraction depths. Infaunal biomass values are measured as ash-free dry weight (AFDW), the others as fresh weight. Grain size and mud content in the deepest area of the MV2 borrow pit may be underestimated due to failure of boxcore sampling.

Short term effects of ecological landscaping in the MV2 borrow pit

The ecological landscaped sandbars significantly influenced species composition of macrozoobenthos and demersal fish and changed sediment characteristics (de Jong et al., 2014 and de Jong et al. 2015b).

Table 5.1 Summary of changes related to different extraction depths.

Extraction depth	Infauna	Epifauna	Demersal fish	Sediment
2	no	nd	nd	Mud: 0.5%< OM
8	*, 2 x	*, 6 x	nd	Grain size: factor 2 decrease and mud 23.1%
20	*, 7 x	*, 12 x	*, 20 x	Grain size: factor 2 decrease and mud 22.3%
24	*, 2 x	*, 67 x	no, 1.5 x	Smaller grain size and mud 13.8%
Landscaping	Significant changes	Significant changes	Significant changes	Significant changes

*: significant changes in species assemblage compared to reference species assemblage (no sand extraction), x means factor of increase in biomass compared to reference values, nd: no data and, no: no change. Grain size and mud content in the deepest area of MV2 borrow pit may be underestimated due to failure of boxcore sampling.

5.4 Ecosystem-based design (EBD) rules for future borrow pits

For intercomparison between the case studies, we used the tide-averaged bed stress as a generic proxy for environmental and related ecological effects. Depth-averaged (2DH) bed shear stress $\tau_{b\ 2DH}$ can be estimated with a quadratic friction law (Eq. 1) (Soulsby, 1997).

$$\tau_{b\ 2DH} = (\rho\ g\ U^2)/C^2 \text{ (Eq. 1)}$$

For the Chézy roughness(C), we used a value of 65 $m^{½}/s$ for the reference area and shallow sand extraction areas, 80 $m^{½}/s$ for the shipping lane of the PoR and MV2 borrow pit due to the high mud content and 110 $m^{½}/s$ (Winterwerp et al., 2004; van Rijn, 1993) for the deepest parts of the MV2 borrow pits due to the soft muddy bed (de Jong et al., 2015b). The density of seawater (ρ) was set to 1023 kg m^{-3} and the gravitational acceleration (g) was set to 9.81 m s^{-2}. The magnitude of the depth-averaged flow velocity (U) in the area of the MV2 borrow pit is 0.65 m s^{-1} (Borsje et al., 2009). From the law of conservation of mass, there is a negative linear relation between the increase in water depth and decrease in depth-averaged velocity. Doubling of the water depth results in a halving of the depth-averaged velocity, in the case of 20 m deep sand extraction in 20 m deep water with a depth-averaged flow velocity of 0.65 m s^{-1}, the depth-averaged velocity is reduced to 0.325 m s^{-1}. When we assume a symmetrical diurnal tide, the tide-averaged bed shear stress equals:

$$\tau_{b\ tide\text{-}averaged} = \tau_{b\ 2DH} / 2 \text{ (Eq. 2)}$$

Applying Eq. 1 and 2 for given parameter values leads to a relationship of tide-averaged bed shear stress ($\tau_{b\ tide\text{-}averaged}$) and sand extraction depth as shown in Fig. 5.4. The tide-averaged bed shear stress shows a strong decrease at small sand extraction depths and smaller decreases at larger extraction depths. The tide-averaged bed shear stress is 0.5 N m^{-2} in the 20 m deep reference area, 0.41 N m^{-2} at 2 m extraction depth, 0.17 N m^{-2} at 8 m extraction depth, 0.08 N m^{-2} at 20 m and 0.04 N m^{-2} at 24 m extraction depth. The stepwise changes in Fig. 5.4 are induced by the changes in Chézy roughness parameter values.

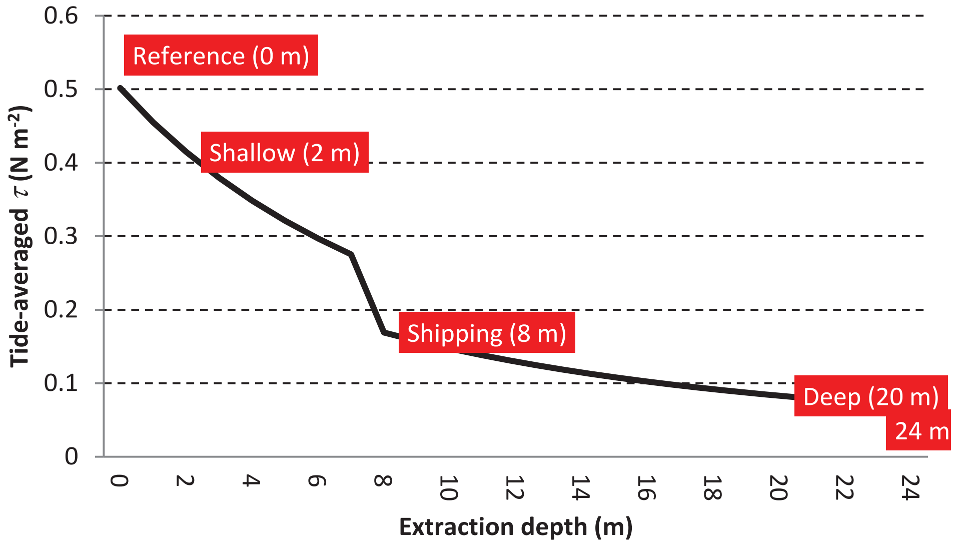


Fig. 5.4 Tide-averaged bed shear stress ($\tau_{b\ tide\text{-}averaged}$) as a function of sand extraction depth at 20 m pre-extraction water depth and using a Chézy roughness value of 65 $m^{1/2}$/s for sand extraction depth interval 0–8 m, 80 $m^{1/2}$/s for 8–20 m and 110 $m^{1/2}$/s for the deepest extraction depth. The reference situation and case studies (shallow sand extraction, shipping lane and deep sand extraction: 20 and 24 m) are indicated with the red text labels.

Extraction depths needed to reach the time-averaged bed shear stress values of the case studies (Fig. 5.4) at different combinations of different pre-extraction water depths and flow velocities can be determined with Eq. 1 and 2 and the resulted EBD graphs (Fig. 5.5). With increasing pre-extraction depth-averaged flow velocities and water depths, larger extraction depths can be applied to reach the tide-averaged bed shear stress values calculated for the DCS case studies (Table 5.2 and 5.3). These EBD rules can be used for the design of future borrow pits to optimise extraction depths in relation to ecological effects on the seabed.

Table 5.2 Influence of higher flow velocities (0.7, 0.75 and 0.8 m s^{-1}) on extraction depths required to reach bed shear stresses of the DCS case studies for 20 m pre-extraction water depths.

DCS case studies		Extraction depth required to reach $\tau_{b\ tide\text{-}averaged}$ of the case studies (m)		
	$\tau_{b\ tide\text{-}averaged}$ (N m^{-2})	0.7 m s^{-1}	0.75 m s^{-1}	0.80 m s^{-1}
Reference	0.50	-	-	-
2	0.41	3.7	5.4	7.1
8	0.17	10.2	12.3	14.5
20	0.08	23.1	26.2	29.2
24	0.04	27.4	30.8	34.2

Table 5.3 Influence of higher pre-extraction water depths (25, 30 and 35 m) on extraction depths required to reach bed shear stresses of the DCS case studies for 20 m pre-extraction water depths.

DCS case studies		Extraction depth required to reach $\tau_{b\ tide\text{-}averaged}$ of the case studies (m)		
	$\tau_{b\ tide\text{-}averaged}$ (N m^{-2})	25 m	30 m	35 m
Reference	0.50	-	-	-
2	0.41	2.5	3	3.5
8	0.17	10	12	14
20	0.08	25	30	35
24	0.04	30	36	42

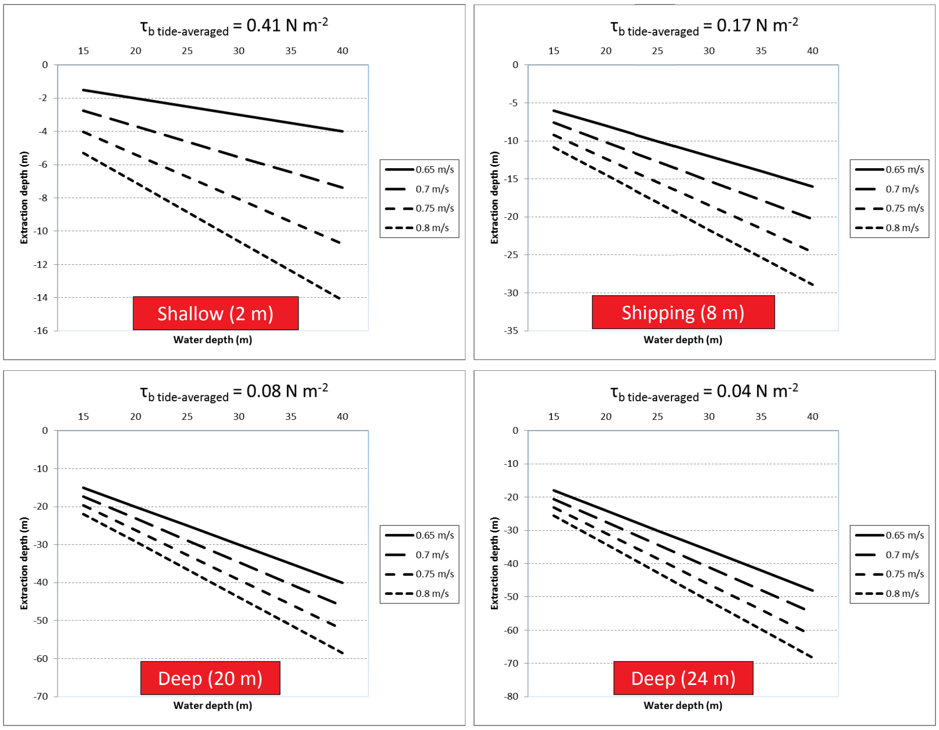


Figure 5.5 EBD graph to reach $\tau_{b\ tide\text{-}averaged}$ (N m^{-2}) values of the DCS case studies (0.41, 0.17, 0.08 and 0.04 N m^{-2}) at the seabed of a borrow pit as a function of extraction depth, at a range of pre-extraction water depths (15–40 m) and varying pre-extraction depth-averaged flow velocity magnitudes (0.65–0.8 m s^{-1}). We used Chézy roughness values 65, 65, 80 and 110 m^{½}/s.

5.5 Discussion

Ecosystem based design rules (EBD rules)

In the UK, limits for acceptable changes in sediment grain size after marine aggregate extraction were proposed based on the natural range, with the aim of ensuring the return of pre-dredge faunal assemblages (Cooper, 2012). Sediment characteristics after deep sand extraction presumably continue to change due to the sedimentation of fine sediment until the borrow pit is filled (Thatje et al., 1999; Desprez, 2000; de Jong et al., 2015b). It is for this reason that grain size is not considered to be the most suitable candidate for setting limits for acceptable change in large-scale and deep borrow pits when a return to pre-dredge conditions is not guaranteed. We therefore developed ecosystem-based design (EBD) rules based on bed shear stress, which can be used to determine extraction depths to reach desirable bed shear stresses and related ecological effects for a range of flow velocities and pre-extraction water depths. In general, with increasing flow velocity and water depth, larger extraction depths can be used.

The EBD rules and ecological landscaping techniques can also help in implementing the European Union's Marine Strategy Framework Directive (MSFD) guidelines and safeguarding or achieving Good Environmental Status (GES) of marine waters. The MSFD consists of 10 GES descriptors: 'Biodiversity', 'Non-indigenous species', 'Commercial fish and shellfish', 'Food webs', 'Eutrophication', 'Sea-floor integrity', 'Hydrographical conditions', 'Contaminants', 'Contaminants in seafood', 'Marine litter' and 'Energy and underwater noise'. When MSFD requires that the seabed has to remain in the original physical condition to enable the return of pre-extraction faunal assemblages, shallow sand extraction with 2 m extraction depth is the best option for 20 m deep pre-extraction areas ($\tau_{b\ tide\text{-}averaged}$: 0.41 N m^{-2}). No significant changes in macrozoobenthos were observed 4−6 year after the cessation of shallow sand extraction (van Dalfsen et al., 2000; van Dalfsen and Essink, 2001). If there are no stringent limitations, 8 m deep sand extraction ($\tau_{b\ tide\text{-}averaged}$: 0.17 N m^{-2}) may be the limit for areas with a pre-extraction water depth of 20 m as only sediment characteristics and macrozoobenthic species composition significantly changed, and biomass increased more than 2-fold. At a tide-averaged bed shear stress of 0.08 N m^{-2}, macrozoobenthic species composition is even more disturbed and biomass increased more than 7-fold. Demersal fish composition changed significantly and biomass increased 20-fold. Below a tide-averaged bed shear stress of 0.04 N m^{-2}, infaunal biomass was only 2-fold higher compared to reference levels. Epifaunal biomass, however, increased more than 67-fold due to a dominance of brittlestar and fish biomass and composition returned to reference conditions. Next to large faunal differences, a high sedimentation rate up to 75 cm y^{-1} was observed.

The most relevant MSFD descriptors for deep sand extraction are: descriptor 1 'Biodiversity' (Patrício J et al., 2014) with the addition of ecological functioning (Bremner, 2008; Törnroos et al., 2015), descriptor 3 'Commercial fish and shellfish', descriptor 6 'Seabed integrity'

(Rice et al., 2012; ICES, 2014b) and descriptor 7 ‘Hydrography’ (OSPAR, 2012). The MSFD can be used to assess the impact of deep sand extraction. The implementation of MSFD into legislation is in progress and therefore maybe not all final criteria of the descriptors are mentioned. The MSFD may have implications for deep sand extraction. The ecological effects of the different extraction depths and ecological landscaping in view of the criteria of the descriptors are summarised in Table 4.

Table 5.4. The ecological effects of the different extraction depths and ecological landscaping in view of the criteria of the MSFD descriptors for the Dutch coastal area with 20 m pre-extraction water depth and a flow velocity of 0.65 m s^{-1}.

	Extraction depth (m)				
	2	8	20	24	Ecological landscaping
Biodiversity	Temporary changes, back to reference conditions in 4–6 y.	Higher diversity, shift to *A. alba*, more deposit feeding	Lower diversity, shift to *A. alba*, more deposit feeding	Low biodiversity, shift to Ophiuroids, only deposit feeding	Increase in heterogeneity, and biodiversity, differences in assemblage
Commercial fish and shellfish	Only temporary changes (?), direct negative for long-living shellfish	Increase in biomass (?), shift to *P. platessa* (?), direct negative impact for long-living shellfish	20-fold increase in biomass, shift to *P. platessa*, negative conditions for long-living shellfish (?)	Biomass back to reference level, *L. limanda*, negative conditions for long-living shellfish (?)	In troughs, increase in biomass, increase in overall biodiversity, negative for long-living shellfish but maybe positive on the long-term (?)
Seabed integrity	Minor changes in bathymetry and sediment characteristics	Smaller grain size, higher mud, very fines and OM	Smaller grain size, higher mud, very fines and OM, high sedimentation rate	Smaller grain, higher mud, very fines and OM, high sedimentation rate	Impact depends on configuration, increase in habitat heterogeneity
Hydrographic circumstances	Decrease in shear stress from 0.50 to 0.41 N m^{-2}	Decrease in shear stress from 0.50 to 0.17 N m^{-2}	Decrease in shear stress from 0.50 to 0.08 N m^{-2}, higher sedimentation	0.04 N m^{-2}, higher chances of hypoxia due to stratification and sedimentation	Impact depends on configuration and circumstances, increase in habitat heterogeneity (stratification, salinity, oxygen, sedimentation rate)

No colour: no changes or only temporary, green: positive impact, yellow: minor negative impact, brown: positive or negative impact, red: negative impact. The question marks indicate possible changes as no data on demersal fish for shallow sand extraction and the shipping lane was collected.

Limitations of ecological data

We compared the ecological effects of different sand extraction depths. The ecological effects due to shallow sand extraction were investigated in the 1990s (van Dalfsen et al., 2000; van Dalfsen and Essink, 2001). We recommend additional research for shallow extraction depths (2 m) on the Dutch continental shelf (DCS) because only one case-study near

Terschelling is described in the literature

The impact of the 8 m deepened shipping lane was investigated in 2006, 2008 and 2012, only a small number of samples were collected and data on maintenance dredging are available but it remains difficult to assess other influences such as anchoring of ships or the influence of movements of ships on sedimentation patterns.
No data were available from case studies with intermediate extraction depths (2–5 m) but the borrow pit that was used for the 'Sand Engine', a 20 million m^3 sand nourishment north of Rotterdam with extraction depths of 6 m (de Vriend et al., 2015), would be very relevant to study. Other interesting cases are the borrow pits used for the beach nourishments 'Zwakke schakels' of the province of Noord-Holland with a total volume of 40 million m^3 and extraction depths between 2 and 8 m.

Macrozoobenthos correlated with time after cessation in the MV2 borrow pit which is an indication that an equilibrium was not yet reached. The full range of ecological effects can only be defined after several years. The most severe effects occurred in the deepest parts of the borrow pit which coincided with significant changes in epifaunal and demersal fish assemblage and it is likely that it will not only be restricted to the deepest areas in the future. Ongoing monitoring is therefore recommended to define medium or long-term effects on macrozoobenthos and demersal fish. Furthermore, the inclusion of sedimentation rate and oxygen content measurements is recommended because two years after cessation of sand extraction, significant differences in epifaunal and demersal fish in the deepest parts of the MV2 borrow pit were encountered (de Jong et al., 2015b).

Limitations of 2D bed shear stress estimates

We applied a 2D approach to estimate the magnitude of bed shear stress. For a more accurate assessment, 3D hydrodynamical modelling approaches are required to cope with the impact of complex bathymetries such as the MV2 borrow pit with polygonal edges (Fig. 5.2, no. 1). Furthermore, complex hydrographic conditions are present with periods of strong density stratification due to the region of fresh water input (ROFI) from the river Rhine (de Boer et al., 2009), density-driven cross-shore flows (van der Hout et al., 2015), up- and downwelling, wind-driven flow and wind and wave-induced mixing. The supply of sediment can also vary between regions and tide-averaged bed shear stresses derived from eq. 1 and 2 assume symmetrical tides but in reality tides are often asymmetrical and create a tide-driven residual current towards the north. Modelling oxygen concentration and sedimentation rates of cohesive and non-cohesive sediment in future borrow pits is also recommended.

In Belgium, effects of sand extraction depths of 5 m were investigated on sandbanks orien-

tated parallel to the tidal current (Bonne, 2010; de Backer et al., 2014). Complex three-dimensional modelling approaches are required to determine bed shear stresses around sandbanks and sandbanks with sand extraction (Briere et al., 2010). Regions with erosion and sedimentation were observed resulting in a variety of ecological responses (Bonne, 2010; de Backer et al., 2014). Due to the orientation of the sandbanks and sand extraction on the crests which are parallel to the tidal current, channelling of the flow can occur resulting in higher flow velocities and bed shear stresses (Roos et al., 2008; Werf van der and Giardino, 2009). The Belgian sand extraction cases are therefore excluded from our present study because flow channelling cannot be predicted with the two-dimensional quadratic friction law.
For the described French case study near Dieppe, the EBD graphs are of limited use due to the complex hydrographic circumstances. Water depths are smaller (15 m) and large tidal amplitudes are present which makes the area prone to the influence of waves resulting in higher bed shear stress values that cannot be predicted with the applied 2D approach. The orientation of the borrow pit and dredge furrows are parallel to the tidal current and instead of decreased current velocities due to the increased water depth, flow contraction may occur (Roos et al. 2008) which may lead to higher current velocities and bed shear stress values. Instead of sediment deposition as in the MV2 borrow pit, erosion may occur. The case study in the UK (Area 222) is in deeper water and therefore EBD rules can be applicable. White furrow shell (*Abra alba)* has a pan-European distribution (Huber and Gofas, 2015) and is therefore also a potential key species for deep borrow pits outside the DCS.

Due to continuing sand extraction on the Dutch Continental Shelf (DCS), a mega-scale trench may emerge. With the orientation of the trench parallel to the tidal current, flow contraction may occur (Roos et al., 2008) resulting in increased flow velocities and bed shear stresses. Instead of sedimentation, erosion of the seabed may even occur. The implications for macrozoobenthos and demersal fish are unknown and deserve attention.

Use of the EBD design rules

This paper is meant to connect ecologists, coastal morphologists and engineers, coastal zone managers and dredging companies in order to maximise sand yields and, simultaneously decrease the surface area of direct impact for different ecological scenarios. Future policy, for example resulting from the EU Marine Strategy Framework Directive (MSFD) presumably requires dredging companies or principals of dredging works to leave the seabed in certain physical conditions to promote or maintain specific ecological conditions. The EBD rules and ecological landscaping techniques can help in implementing MSFD guidelines and safeguarding or achieving Good Environmental Status (GES) of marine waters.

5.6 Conclusions

For intercomparison between sand extraction case studies, we used tide-averaged bed shear stress as a generic proxy for changed environmental conditions and related ecological effects. Bed shear stress can be estimated with a two-dimensional equation using extraction depth and depth-averaged flow velocity magnitude. We developed ecosystem-based design (EBD) rules for future borrow pits based on ecological and bed shear stress data.

I In general, at higher depth-averaged flow velocities and pre-extraction water depths, larger sand extraction depths can be applied to reach certain tide-averaged bed shear stresses.

The EBD rules can be used for early design phases of future borrow pits in the southern North Sea in order to maximise sand yields and simultaneously minimise the surface area of direct impact. The EBD rules can help in implementing MSFD guidelines and safeguarding or achieving Good Environmental Status (GES) of marine waters. For comparable regions in the southern North Sea, EBD rules and ecological landscaping can also be used but ecological data from areas with low shear stress values such as abandoned borrow pits or dredged shipping lanes are a prerequisite. We recommend using 3D modelling approaches for later design phases and regions with complex hydrographical circumstances.

5.7 Acknowledgements

We want to thank Building with Nature (BwN) programme, Pieter-Koen Tonnon and Geert Keetels (BwN HK 2.4), the user group committee and the ICES Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem (WGEXT). We also want to thank Hans Hillewaert for the high quality picture of the North Sea organisms and the anonymous reviewers for their useful comments. This study is part of BwN innovation programme and was funded by several sources, including the 'Subsidieregeling Innovatieketen Water' (SIW, Staatscourant nrs. 953 and 17009), the Dutch Ministry of Transport, Public Works and Water Management and partner contributions of the participants to the Foundation EcoShape. The study received co-funding from the European Fund for Regional Development EFRO and the Municipality of Dordrecht.

Chapter 6

General discussion

The Dutch coastal zone is intensively used for activities such as fishing, shipping, wind farming, dredging, disposal of dredged sediment, beach nourishment, and the extraction and transport of oil, gas and aggregates. These activities have different effects on the marine environment and some are likely to intensify in the future. In the Netherlands, approximately 26 million m^3 of sand is used annually for coastal nourishments and for construction. A possible increase of annual nourishments from 12.5 up to 40–85 million m^3 for counteracting effects of future sea level rise is anticipated. In the last decades, shallow sand extraction up to 2 m below the seabed was employed. Due to the increasing pressure of the activities, well-considered use of space is necessary.

For a 20 km^2 seaward harbour expansion Maasvlakte 2 (MV2) of the Port of Rotterdam (PoR), the Dutch authorities permitted for the first time deep sand extraction to decrease the surface area of direct impact. Approximately 220 million m^3 of sand was extracted between 2009 to 2013, with an average extraction depth of 20 m. The surface area of the borrow pit was reduced from 110 km^2 at 2 m extraction depth to only 11 km^2 at 20 m extraction depth. The ecological effects, however, were largely unknown and therefore investigated in this PhD project.

6.1 Research questions

We first investigated which environmental variables influence macrozoobenthos and we determined how many macrozoobenthic assemblages are present in the intensively used area of the Dutch coastal zone prior to the construction of MV2 **(chapter 2)**. Short-term effects of deep sand extraction and ecological landscaping on macrozoobenthos sediment characteristics and hydrographical variables are investigated (**chapter3**). The short-term effects on demersal fish are described in **chapter 4.** In **chapter 5**, Effects of different sand extraction depths on the Dutch continental shelf are compared and ecosystem-based design rules for future borrow pits are developed which simultaneously maximise sand yields and minimise the surface area of direct impact.

6.2 Main conclusions

Correlations between environmental variables and macrozoobenthos in the Dutch coastal zone in front of Rotterdam were determined using PoR's baseline monitoring data (2006–2008). Next to sediment characteristics, hydrographic variables such as salinity and bed shear stress also partially explained macrozoobenthic distribution patterns. Macrozoobenthic species richness and biomass peaked in 20 m deep areas with fine sediment, elevated mud and sediment organic matter values and low mean bed shear stress. A distinct macrozoobenthic assemblage dominated by white furrow shell (*Abra alba)* coincided in a deepened shipping lane and near a disposal site for dredged fine sediment which may be an indication that intermediate deep sand extraction depths and dredged sediment disposal may have long-term significant impact on macrozoobenthos (**Chapter 2**). Smaller changes in species composition, species richness and biomass were observed around the 20-m isobath which may be induced by activities such as shallow sand extraction and disposal of coarse dredged sediment.

Short-term effects of deep sand extraction and ecological landscaping on macrozoobenthos were investigated from 2009–2012 in the Maasvlakte 2 borrow pit. Macrozoobenthic biomass increased on average 5-fold, species composition changed and white furrow shell *(Abra alba)* became abundant in the deepest parts. Significant changes in sediment characteristics were also observed. Next to sediment and hydrographic variables, macrozoobenthos also correlated with time after cessation which is an indication that an equilibrium was not yet reached. A shift in epifaunal assemblage to brittle stars in 2012 in the troughs of the ecological landscaped sandbars coincided with the highest observed sedimentation rate (**Chapter 3**).

Short-term effects of deep sand extraction and ecological landscaping on demersal fish were investigated with a commercial beam trawl. In the MV2 borrow pit, biomass increased 20-fold and fish species assemblage significantly changed with plaice (*Pleuronectus platessa*) as most abundant species instead of dab (*Limanda limanda*) in the reference area. Increased demersal fish biomass is closely linked to increased white furrow shell **(***Abra alba***)** biomass. Ecological landscaped sandbars significantly influenced macrozoobenthic and demersal fish assemblage and biomass. In the troughs of the landscaped parallel sandbar, however, a significant drop in biomass was observed two years after cessation (**Chapter 4**).

In **Chapter 5**, ecological data of **Chapter 2, 3 and 4** and calculated bed shear stress values were combined and transformed into Ecosystem-based design (EBD) rules. In general, at higher flow velocities and larger water depths, larger extraction depths can be applied to establish certain ecological conditions.

6.3 Limitations of current ecological data

The Dutch government commissioned the PUTMOR study to investigate the potential of sand extraction depths over 2 m in 1999. A deep borrow pit in front of the Port of Rotterdam (PoR) with sand extraction depths between 5–12 m was used (Boers, 2005). The PUTMOR study concluded that there were no indications that deep sand extraction would lead to unacceptable effects such as haline or thermal stratification and oxygen depletion, and that recovery of benthic assemblages would be possible (Boers, 2005). Due to the short period of investigation, ecological data was not collected during the PUTMOR study.

We therefore compared the ecological effects of different sand extraction depths. The ecological effects due to shallow sand extraction (2m) were investigated in the 1990s (van Dalfsen and Essink, 2001; van Dalfsen et al., 2000). The impact of the 8 m deepened shipping lane was first investigated in 2006, later studies were executed in 2008 and 2012 (de Jong et al., 2015a,b). For deep sand extraction in the MV2 borrow pit, only short-term data is currently present (de Jong et al., 2014; de Jong et al., 2015b).

We recommend additional research for shallow extraction depths on the DCS because only one case-study near Terschelling is described in the literature (van Dalfsen and Essink, 2001; van Dalfsen et al., 2000). For the 8 m deepened shipping lane, only a small number of samples were collected and although data on maintenance dredging are available, it remains difficult to assess other influences such as anchoring of ships or the influence of movements of ships on sedimentation patterns.

No data were available from case studies with intermediate extraction depths (2–5 m). The borrow pit which was used for the 'Sand Engine', a 20 million m^3 sand nourishment north of the PoR with extraction depths of 6 m (de Vriend et al., 2015), would be very relevant to study. Other interesting cases are the borrow pits used for the beach nourishments 'Zwakke schakels' of the province of Noord-Holland with a total volume of 35 million m^3 and extraction depths between 2 and 8 m (van Duin et al., 2012).

Another important limitation of the current study is the short period of investigation. Macrozoobenthos correlated with time after cessation which is an indication that an equilibrium was not yet reached. The full range of ecological effects on the seabed of the MV2 borrow pit can only be defined after several years. The most severe effects occurred in the deepest parts of the borrow pit which coincided with significant changes in epifaunal and demersal fish assemblage and it is likely that it will not only be restricted to the deepest areas in the future.

6.4 The use of this study beyond present scope

The demand for marine sand outside the Netherlands is also strongly increasing (ICES, 2013; Stolk and Dijkshoorn, 2009; Peduzzi, 2014). Worldwide, 47 and 59 billion tonnes of aggregate material is extracted yearly (Steinberger et al., 2010). Considerable volumes are extracted in surrounding countries, in the UK 16.8 million m^3, in France 12 million m^3 and in Denmark 10.5 million m^3 per year. In Belgium, each year almost 4 million m^3 of sand is extracted and 2.5 million m^3 of sand is imported from the Netherlands (ICES, 2014a).

In Belgium, effects of sand extraction depths of 5 m were investigated on sandbanks which run parallel to the tidal current (Bonne, 2010; de Backer et al., 2014). Complex three-dimensional modelling approaches were required to determine bed shear stresses around sandbanks and sandbanks with sand extraction (Briere et al., 2010). Regions with erosion and sedimentation were observed after sand extraction resulting in a variety of ecological responses (Bonne, 2010; de Backer et al., 2014). Due to the orientation of the sandbanks and sand extraction on the crests which is parallel to the tidal current, flow contraction can occur (Roos et al. 2008) resulting in higher flow velocities and bed shear stresses instead of decreased current velocities. The Belgian sand extraction cases are therefore excluded from our present study because flow contraction cannot be predicted with our simplified 2D-hydrodynamic approach.

For a described French case study near Dieppe (**Chapter 5**), the EBD rules are also of limited use due to the complex hydrographic circumstances. Water depths are smaller (15 m) and large tidal amplitudes are present which makes the area prone to the influence of waves resulting in higher bed shear stress values that cannot be predicted with the applied current-driven 2D approach which ignores the effect of waves. The orientation of the borrow pit and dredge furrows are parallel to the tidal current and due to the increased water depth, flow contraction may occur (Roos et al. 2008) which may lead to higher current velocities and bed shear stress values. Instead of sediment deposition as in the MV2 borrow pit, erosion may occur. The described case study in the UK (Area 222) (**chapter 5**) is in deeper water and therefore EBD rules may be applicable. Ecological data for the Dutch case studies, however, may not be generally applicable to other regions. Ecological data from regions outside the DCS are needed.

Ongoing sand extraction on the DCS may lead to interconnected individual borrow pits in the area of the 20 m isobath and the 12-nautical mile contour along the entire Dutch coast leading to a mega-scale extraction trench. With the orientation of the trench, parallel to the tidal current, flow contraction may occur resulting in increased flow velocities and bed shear stresses (Roos et al., 2008; van der Werf and Giardino, 2009) and instead of sedimentation, erosion of the seabed may even occur. The potential effects of a mega-scale extraction trench can be very different compared to the findings of this study and therefore deserves thorough investigation and consideration.

6.5 Relevance for coastal zone management

Due to increasing human activities and the sand demand on the Dutch continental shelf, a new sand extraction strategy was formulated to guarantee sufficient supply of marine sand in the intensively used coastal zone at reasonable costs with space for other activities (IDON, 2011). The Dutch government considers sand extraction to be of national importance and the process of sand extraction gets higher priority than other activities. Starting points of sand extraction are: ecologically responsible, cost-effective, sustainable sand extraction that is smart in relation to supply and properly harmonised in spatial planning terms (IDON , 2011).

Keeping in mind the significant effects of deep sand extraction on macrozoobenthos and demersal fish brings me to the conclusion that several uncertainties are still present and some signs of detrimental effects may be already emerging in the deepest parts of the MV2 borrow pit.

In the revised Integrated Management Plan for the North Sea 2015 (IMPNS 2015), Environmental Impact assesments (EIAs) and monitoring and evaluation programmes (MEPs), the focus of deep sand extraction is primarily on cost reduction and decreasing the surface area of direct impact of sand extraction (DHV, 2010; Rozemeijer, 2009; Ellerbroek et al., 2008, IDON, 2011) at the possible expense of ecological values.

The decrease in the grain size and the increase in the mud and organic matter content have led to significantly different macrozoobenthic and fish assemblages. A new ecological equilibrium state is not expected in the short term because of the continuous sedimentation of fine sediments in the MV2 borrow pit that may take decades or even longer. Instead of simple impact calculation rules used in EIAs based on the impacted surface area and 4–6 years of recovery time (van Dalfsen and Essink, 2001; van Dalfsen et al., 2000; Boyd et al., 2005) effects of continuously changing environmental conditions should be considered.

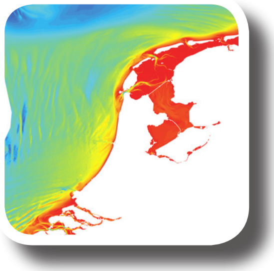
The EBD rules and ecological landscaping techniques can help in balancing the three most important aspects of IMPNS 2015, ecological healthy , safe and profitable North Sea (IDON, 2011). The Marine Strategy Framework Directive (MSFD) demands that each Member State achieves a Good Environmental Setting (GES) of its marine waters by 2020.

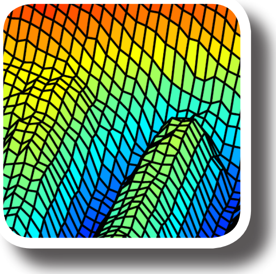
6.6 Recommendations

- Better investigate the medium to long-term ecological effects of varying sand extraction depths by including changing environmental conditions of infilling with fine sediment.
- Reconsider the application of deep sand extraction since significant ecological effects emerged in the MV2 borrow pit, and implement these insights in Environmental Impact Assessments.
- Validate the ecosystem-based design guidelines with medium- or long-term data and new case-studies.

Chapter 7

Summary/Samenvatting





Summary

The Dutch coastal zone is intensively used for activities such as fishing, shipping, wind farming, dredging, disposal of dredged sediment, beach nourishment, and the extraction and transport of gas and aggregates. In the Netherlands, approximately 26 million m^3 of sand is used annually for coastal nourishments and for construction. A possible increase of annual coastline nourishments from 12.5 up to 40–85 million m^3 for counteracting effects of future sea level rise is anticipated (Deltacommissie, 2008). In the last decades, shallow sand extraction up to 2 m below the seabed was employed. Due to the increasing pressure of the activities in the intensively used Dutch coastal zone, well-considered use of space is necessary.
For a 20 km^2 seaward harbour expansion Maasvlakte 2 (MV2) of the Port of Rotterdam (PoR), the Dutch authorities permitted for the first time deep sand extraction to decrease the surface area of direct impact. The ecological effects of deep sand extraction (20 m), however, were largely unknown and therefore investigated in this PhD project.

The main objective of this PhD project was to investigate the short-term effects of deep sand extraction and ecological landscaping on macrozoobenthos, demersal fish and habitat characteristics on the Dutch continental shelf (DCS). Effects of different sand extraction depths on the DCS are compared and ecosystem-based design rules for future borrow pits are developed which simultaneously maximise the sand yield and minimise the surface area of direct impact.

In **Chapter 2** we investigated distribution patterns and species composition of macrozoobenthos in the Dutch coastal zone in front of the Port of Rotterdam (PoR) prior to the realisation of harbour expansion 'Maasvlakte 2'. Relationships between macrozoobenthic assemblages and environmental variables were determined using non-metric dimensional scaling (nMDS) based on 470 box core, bottom sledge, and sediment samples collected in spring 2006 and 2008. Five main in- and epifaunal assemblages were distinguished using clustering techniques and nMDS ordinations. Macrozoobenthic species composition correlated with a set of measured sediment and modelled hydrographic variables. Macrozoobenthic species richness and biomass peaked at 20 m deep areas with a grain size of 200 µm, elevated mud and sediment organic matter, and low mean bed shear stress. Considerable interannual differences in macrozoobenthic assemblage distribution were observed which resulted from more Echinoids, Phoronids, and razor clams. A distinct, highly productive and species-rich macrozoobenthic white furrow shell (*Abra alba*) assemblage coincided in an 8 m deepened shipping lane and near a disposal site for dredged fine sediment. Modelled bed shear stress is an explanatory variable in addition to sediment variables in explaining distribution patterns in macrozoobenthos.

For a 20 km^2 seaward harbour expansion 'Maasvlakte 2' of the PoR, approximately 220 million m^3 of sand was extracted between 2009–2013, with an average extraction depth of 20 m. In **Chapter 3**, we investigated shoret-term effects of deep sand extraction on macrozoobenthos. Sediment characteristics were determined from the boxcore samples, bed shear stress and near-bed salinity were estimated with a hydrodynamic model. Two years after the cessation of sand extraction, macrozoobenthic biomass increased 5-fold in the deepest areas of the MV2 borrow pit. Species composition and sediment characteristics changed significantly and white furrow shell (*A. alba*) became abundant. Macrozoobenthic species composition and biomass significantly correlated with time after cessation of sand extraction, sediment and hydrographical characteristics. Ecosystem-based landscaped sandbars were found to be effective in influencing sediment characteristics and macrozoobenthic assemblage. Significant changes in epifauna occurred in the deepest parts in 2012 which coincided with the highest observed sedimentation rate.

Significant differences in demersal fish species assemblages in the MV2 borrow pit were associated with variables such as water depth, median grain size, fraction of very fine sand, biomass of white furrow shell (*A. alba*), and time after the cessation of sand extraction (**Chapter 4**). Large quantities of undigested crushed white furrow shell fragments were found in all stomachs and intestines of plaice (*Pleuronectes platessa*), indicating the importance as a prey item. A significant 20-fold increase in demersal fish biomass was observed 1–2 year after cessation in the deep parts of the MV2 borrow pit. However, in the troughs of a landscaped sandbar, a drop in biomass down to reference levels was observed two years after cessation.

In **Chapter 5** we combined ecological data with calculated bed shear stress values to determine ecosystem-based design rules for the development of borrow pits. Bed shear stress was estimated with a two-dimensional quadratic friction law and showed a decrease from 0.50 to 0.04 N m^{-2} in a borrow pit in 20 m deep water and extraction depths up to 24 m. Borrow pits with a tide-averaged bed shear stress of around 0.17 N m^{-2} may lead to enhanced macrozoobenthic species richness and biomass. Below a tide-averaged bed shear stress of 0.08 N m^{-2}, increasing abundance of brittle stars, and below 0.04 N m^{-2} further detrimental effects to macrozoobenthos can be expected. At higher flow velocities and larger water depths, larger extraction depths can be applied to achieve desired tide-averaged bed shear stresses for related ecological effects. The EBD rules can be used in the early-design phases of future borrow pits in order to simultaneously maximise sand yields and minimise the surface area of direct impact.

In **Chapter 6**, I discussed the results and limitations of the study and formulated recommendations for further research.

The short period of investigation is an important limitation of the current study and it is recommended to investigate medium or long-term effects of deep sand extraction on macrozoobenthos and demersal fish and to include oxygen content and sedimentation measurements in the MV2 borrow pit. The application of increasing extraction depth need to be reconsidered since it is not merely decreasing the area of direct impact as significant differences in macrozoobenthos and demersal fish assemblage emerged in the MV2 borrow pit. The effects of sand extraction should also be assessed using the Good Environmental Status descriptors of the Marine Strategy Framework Directive (MSFD). I recommend to implement insights of this study in Environmental impact assessments (EIAs) and because long-term effects of deep sand extraction are not yet known, it is advisable to be reluctant when considering larger sand extraction depths. The developed ecosystem-based design rules may be helpful in the design of borrow pits but both ecological and hydrodynamical data need to be validated with medium- or long-term data and new case studies with intermediate extraction depths.

De Nederlandse kustzone wordt intensief gebruikt voor activiteiten als visserij, scheepvaart, windparken, baggerwerkzaamheden, stort van baggerslib, kustsuppleties en het transport en winning van zand, olie en gas. In Nederland wordt ongeveer 26 miljoen m^3 zand gewonnen uit de Noordzee en gebruikt voor kustsuppleties en bouwwerkzaamheden. In de nabije toekomst kan het benodigde volume zand voor kustsuppleties stijgen van 12.5 tot 40–85 miljoen m^3 om de effecten van de zeespiegelstijging tegen te gaan (Stolk and Dijkshoorn, 2009; ICES, 2014a). In de laatste jaren was alleen ondiepe zandwinning toegestaan tot 2 m beneden de zeebodem. Door de toenemende druk van de activiteiten in de intensief gebruikte Nederlandse kustzone, is weloverwogen gebruik van de ruimte nodig.
Voor een 20 km^2 grote zeewaartse havenuitbreiding Maasvlakte 2 (MV2) van het Havenbedrijf Rotterdam (HbR) hebben de Nederlandse autoriteiten voor het eerst diepe zandwinning toegestaan tot 20 m beneden de zeebodem met als doel om het totale ruimtebeslag van de ingreep van de zandwinning te reduceren en daarmee ook de ecologische effecten te beperken. De ecologische effecten van diepe zandwinning zijn echter nog grotendeels onbekend.

De doelstelling van dit PhD onderzoek is om de kortetermijneffecten van diepe zandwinning op macrozoobenthos, bodemvis en habitatfactoren op het Nederlands continentaal plat (NCP) te onderzoeken. We hebben onderzocht of het mogelijk is om deze effecten te mitigeren of te reduceren door middel van ecologische landschapsontwikkeling op de bodem van de put. Effecten van verschillende zandwindieptes op het NCP zijn vergeleken en gecombineerd met berekende bodemschuifspanningen. Dit heeft geleid tot ecosysteem-gebaseerde ontwerpregels voor toekomstige zandwinputten met tegelijkertijd een maximalisatie van de zandopbrengst en een minimalisatie van het ruimtebeslag.

In **Hoofdstuk 2** hebben we de verspreiding en soortensamenstelling van macrozoobenthos onderzocht in de Nederlandse kustzone ter hoogte van Rotterdam. Op basis van 470 box corer, sediment en bodemschaaf monsters die in 2006 en 2008 zijn verzameld door havenbedrijf Rotterdam zijn er met behulp van niet-metrische multi-dimensionale multivariate analyses (nMDS) relaties tussen macrozoobenthos en milieuvariabelen bepaald. Vijf in- en epifauna clusters zijn onderscheiden met clusteringstechnieken. Macrozoobenthische soortenrijkdom en biomassa correleerde met gemeten sediment en gemodelleerde hydrodynamische variabelen. De hoogste waarden aan biomassa en soortenrijkdom zijn waargenomen in 20 m diepe gebieden met een korrelgrootte van 200 µm, verhoogd organisch materiaal en slibgehalten en een lage bodemschuifspanning. Aanzienlijke jaarlijkse veranderingen in soortensamenstelling zijn waargenomen als gevolg van toenamen van zee-egels, hoefijzerwormen en mesheften. Een sterk afwijkend hoog-productief macrozoobenthisch cluster gedomineerd door de witte dunschaal schelp (*Abra alba*) is aangetroffen in de 8 m verdiepte Euromaasgeul en in de nabijheid van bergingslocaties voor baggerspecie. Naast de sediment variabelen is bodemschuifspanning en saliniteit ook een belangrijke habitatfactor.

Ten behoeve van de aanleg van Maasvlakte 2 is ongeveer 220 miljoen m^3 zand gewonnen tussen 2009 en 2013, met een gemiddelde zandwindiepte van 20 meter. In **Hoofdstuk 3** zijn de kortetermijnseffecten van diepe zandwinning op macrozoobenthos en sediment karakteristieken onderzocht. Één tot twee jaar na de zandwinning werd in de diepe delen van de zandwinput een veelvoudige toename in biomassa waargenomen. Verder waren de soortensamenstelling en sediment karakteristieken significant veranderd (kleinere korrelgrootte en meer slib en organisch materiaal) en de witte dunschaal schelp (*A. alba*) was wederom de meest voorkomende soort. Soortensamenstelling en biomassa correleerden met de tijdsduur na zandwinning, en de karakteristieken van sediment en hydrografie. Uitgebaggerde, op het ecosysteem gebaseerde, zandruggen waren effectief in het beïnvloeden van sediment karakteristieken en soortensamenstelling. Significante verschillen in epifauna traden op in de diepste delen van de zandwinput in 2012 wat samenviel met de hoogst waargenomen sedimentatiesnelheid.

Significante verschillen in de bodemvis soortensamenstelling correleerden met variabelen als: water diepte, mediane korrelgrootte, fractie fijn zand, biomassa van de witte dunschaal schelp (*A. alba*) en de tijdsduur na zandwinning (**Hoofdstuk 4**). Grote hoeveelheden onverteerde fijngemalen witte dunschaal schelp (*A. alba*) fragmenten werden aangetroffen in de magen en darmen van schol (*Pleuronectes platessa*) wat een indicatie is dat het een belangrijke prooisoort is. Een tot twee jaar na de zandwinning is er een 20-voudige toename van de biomassa van bodemvis waargenomen. Echter, in de troggen van de uitgebaggerde parallelle zandrug is 2 jaar na de zandwinning de biomassa en soortensamenstelling weer gedaald tot de referentiewaarden.

In **Hoofdstuk 5** hebben we de ecologische kennis aangevuld met berekende bodemschuif spanningen en vertaald naar ecosysteem-gebaseerde ontwerpregels. Bodemschuifspanning is benaderd met een twee-dimensionale kwadratische weerstandsvergelijking. Bodemschuifspanning is de kracht per eenheid van zeebodemoppervlak uitgeoefend door het zeewater en heeft invloed op de korrelgrootte, de bezinking van fijne deeltjes en kan daardoor macrozoobenthos en bodemvis beïnvloeden. Bodemschuifspanning is afhankelijk van de stroomsnelheid, de bodemruwheid, de dichtheid van het zeewater en de gravitatieversnelling. Vanwege de wet van behoud van massa, is er een negatief lineair verband tussen de toename van de waterdiepte, als gevolg van zandwinning, en de afname van de stroomsnelheid. De bodemschuifspanning in een zandwinput in 20 m diep water met zandwindieptes tot 24 m vertoont een afname van 0.50 tot 0.04 N m^{-2}. Zandwinputten met een bodemschuifspanning van rond de 0.17 N m^{-2} leiden lokaal waarschijnlijk tot een verhoogde soortenrijkdom en biomassa van macrozoobenthos en bodemvis. Bij bodemschuifspannin-

gen lager dan 0.08 N m^{-2} is er een toename van slangsterren te verwachten en beneden 0.04 N m^{-2} kunnen er verdere toenamen optreden.
De ontwerpregels kunnen gebruikt worden in de ontwerpfase van toekomstige zandwinputten in de Noordzee om gelijktijdig zandopbrengst en een afname van door zandwinning beïnvloedde oppervlakte te realiseren.

In **Hoofdstuk 6** bespreek ik de resultaten en beperkingen van het onderzoek en geef ik aanbevelingen voor toekomstig onderzoek. De relatief korte onderzoekstijd is de grootste en tevens onvermijdelijke beperking van dit PhD onderzoek. Het verder onderzoeken van midden- tot langetermijneffecten van diepe en grootschalige zandwinning op macrozoobenthos en bodemvis verdient sterk de aanbeveling. Het in kaart brengen van zuurstofconcentratie en sedimentatiesnelheid kan een licht werpen op de ontwikkeling van eventuele negatieve omstandigheden in de put. Een heroverweging van het gebruik van grote zandwindieptes is nodig want het is niet alleen een kwestie van een afname van het ruimtebeslag, zoals nu vaak in milieueffect rapportages vaak wordt aangehaald. Waarschijnlijk zullen er gedurende lange periode significante verschillen in macrozoobenthos, bodemvis en sediment optreden.

De effecten van diepe zandwinning moeten ook aan de hand van de Europese Kaderrichtlijn Mariene Strategie (KMS) en vastgelegde discriptoren van een goede milieutoestand getoetst worden voordat er een eindoordeel geveld kan worden. Omdat de langetermijneffecten van diepe zandwinning nog niet bekend zijn is het aanbevelingswaardig om terughoudend te zijn met grote zandwindiepten. De verzamelde kennis omtrent de effecten van diepe zandwinning op het mariene ecosysteem dienen bij toekomstige projecten te worden geimplementeerd. De opgestelde op ecosysteem gebaseerde ontwerpregels kunnen gebruikt worden bij het ontwerp van toekomstige zandwinputten maar een validatie met ecologische en hydrodynamische langetermijndata is wel aanbevelenswaardig.

Chapter 8

Dankwoord
Curriculum vitae
Publicatielijst

Dankwoord

Na vele omzwervingen en een studie mariene biologie met tropische onderwerpen in voornamelijk Indonesië heb toch gekozen voor een toegepast onderwerp in Nederland. Eind 2009 ben ik bij IMARES Wageningen UR begonnen aan het 'Building with Nature' onderwerp 'Modelling the ecological potential of deep and large-scale sand extraction on the Dutch continental shelf'.

Het voelde in het begin even alsof ik op een rijdende trein sprong maar later werd duidelijk dat de richting niet altijd even duidelijk was en er veel verschillende verwachtingen waren. Co-promotor Martin Baptist, dagelijks begeleider maar ook enorm druk met het opstarten van vele projecten binnen 'Building with Nature' en IMARES bedankt voor alle hulp en je enthousiasme. Promotor Han Lindeboom bedankt voor alle hulp en het was leerzaam om je strijd op het gebied van natuurbescherming op zee te volgen. Een PhD project kan lang duren maar maatregelen om de natuurwaarden van de Noordzee te verhogen vergt tientallen jaren en zelfs dan is de winst soms nog verwaarloosbaar. Promotor Piet Hoekstra dank voor de kritische blik voornamelijk vanuit de wereld van de hydrografie.

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Vanuit Havenbedrijf Rotterdam zijn er vaak veel scherpe vragen gesteld door Wil Borst en Onno van Tongeren waar ik uiteindelijk toch veel aan heb gehad en die ook de kwaliteit van de artikelen heeft verbeterd. Ad Stolk en Sander de Jong, ook jullie bedankt voor alle hulp en informatie, zonder jullie waren de artikelen niet te schrijven. I want to thank all member of ICES WGEXT for the interesting meetings and presentations at very interesting locations, I want to thank in particular Jan van Dalfsen, Michel Desprez (drinking calvados during lunchtime), Jyrki Hämäläinen (thanks for the humour and the Arnold Schwarzenegger imitation 'speak louder' during my presentation ;)), Henry Bokuniewicz, Brigitte Lauwaert, Johan Nyberg, Bryndis G. Robertsdottir, Tammy Stamford, Rebecca Walker and Keith Cooper.
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Ook al heeft dit onderzoek geresulteerd in 4 gepubliceerde artikelen, ik vindt het jammer dat ik alleen de kortetermijnseffecten van diepe zandwinning boven water heb kunnen halen, het voelt alsof ik het eind van het verhaal niet heb kunnen opschrijven. Ik blijf de ontwikkelingen op het gebied van zandwinning en de ecologische effecten in de toekomst volgen. Ik wil me sowieso blijven doorontwikkelen binnen de mariene (tropische) ecologie en start in 2016 bij Arcadis als ecomorfoloog.

Curriculum vitae

When I was a child, I remember that I was immediately hooked by the magic of water while swimming in one of the swimming pools where my father was a manager. During summer holidays we went swimming and snorkelling in the Mediterranean sea where we encountered escaping octopuses in submerged caves releasing enough ink for writing a novel and we hovered in the sea in a floatable truck tire inner tube while people on the shore where trying to alarm us about the risk of large jellyfish surrounding us.

During high school, I developed an interest in Natural sciences but was not stimulated in the right way so I used a long route of studies before finally arriving at Curaçao. During my bachelor environmental engineering, I investigated while snorkelling and diving, the nursery function of shallow bay habitats such as mangroves, seagrass meadows and coral reefs for coral reef fish. This made me decide to study marine biology, which sadly was not possible earlier. My first research was conducted at Wageningen UR where I studied the biological relevance of different branch spacing patterns of Finger coral (*Madracis mirabilis*) using digital particle image velocimetry (DPIV). For my master Marine Biology in Groningen, I investigated advective transport of particulate matter in tropical seagrass meadows in the Spermonde archipelago in front of Makassar Indonesia under supervision of Arie Vonk. With Naturalis we went to Bunaken in front of Manado, Indonesia to collect sponges and search for eusocial shrimps living inside them. All this excitement led to the choice of doing a PhD to further gain knowledge about the marine system. It initially didn't gave me a lot of excitement but finally resulted in 4 scientific peer-reviewed articles and this thesis.

I hope that one day I will return to the tropics to investigate tropical ecosystems again. Maybe in the future, my brother Jeroen and I can work together on topics related to dredging and sand extraction like my little brother Rob was always referring to.

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Chapter 9

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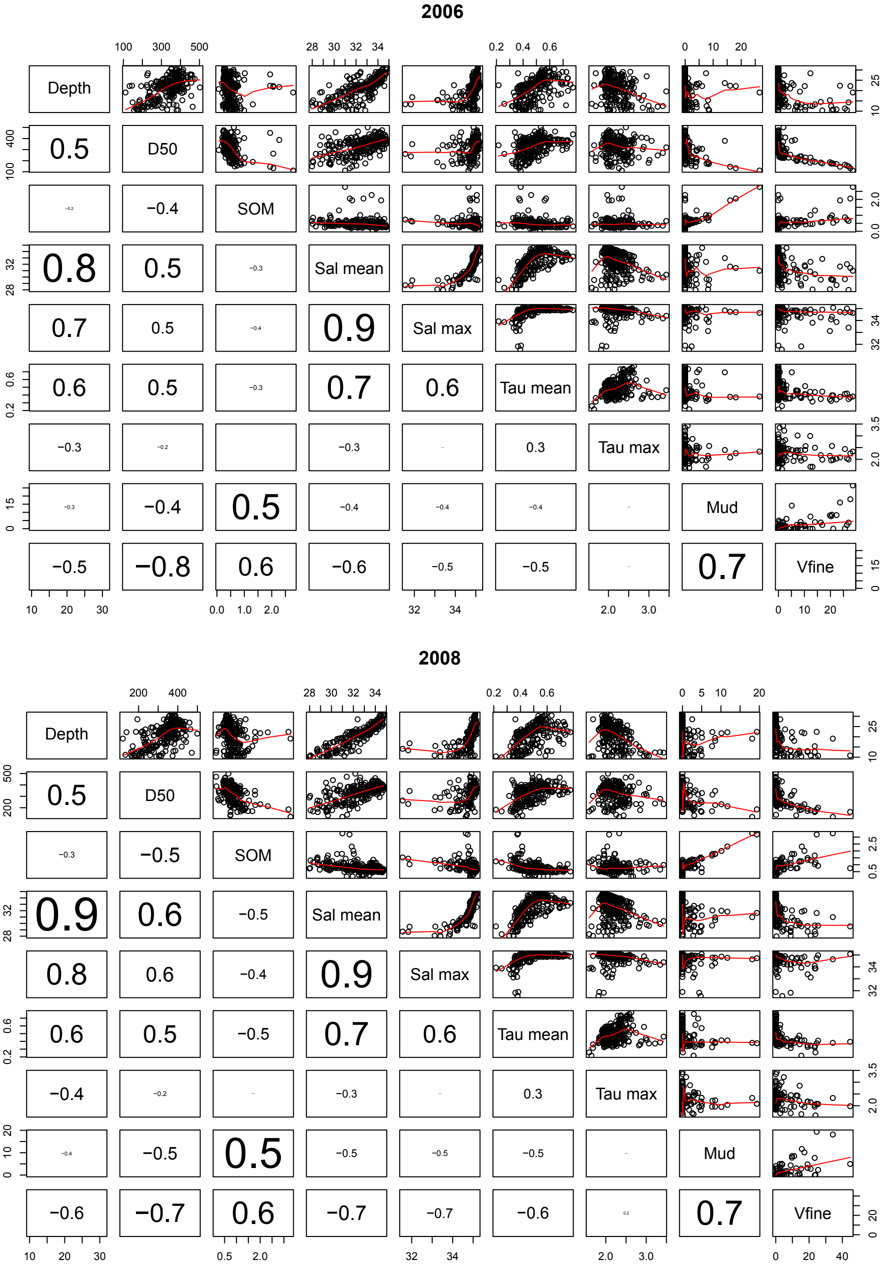
Appendices

Appendix 2.I: Pairplot of variables in 2006 and 2008. The lower diagonal part shows the Spearman's rank correlation coefficient with the size of the font being proportional to the coefficient. D_{50}: median grain size, SOM: fraction sediment organic matter. Sal mean: mean near-bed salinity, Sal max: maximum near-bed salinity, Tau mean: mean bed shear stress, Tau max: maximum bed shear stress, Mud: fraction mud and Vfine: fraction very fine sand.



2006

2008

Appendix 2.II Species composition of infaunal (IN) assemblages (ind. m^{-2}), biomass (g AFDW m^{-2}) and species richness (species per boxcore). Dufrêne–Legendre indicators are indicated in bold.

	2006		**2008**		**Average**	
Variable	**Mean**	**Sd**	**Mean**	**Sd**	**Mean**	**Sd**
Assemblage IN1 (*Echinoidea* spp.–*L. conchilega*)						
Occurrence	47		72		119	
Biomass	6.9	8.1	4.7	5.8	5.6	6.8
Species richness	23.0	4.7	25.5	6.3	24.5	5.8
Echinoidea spp.	1.6	5.8	1069.5	4179.8	647.7	3284.4
Lanice conchilega	232.3	659.1	611.4	1361.9	461.6	1148.9
Spiophanes bombyx	95.9	105.8	641.5	747.9	426	642.4
Owenia fusiformis	85.5	420.3	336.6	693.1	237.4	610.8
Terebellidae spp.	0.8	4.2	279.9	1103.7	169.7	867
Phoronida spp.	75	221.2	176.8	644.4	136.6	521
Assemblage IN2 (*Echinoidea* spp–*Phoronida* spp.)						
Occurrence	94		139		233	
Biomass	2.6	5.3	1.0	1.2	1.6	3.5
Species richness	18.5	5.4	19.4	4.2	19.0	4.7
Echinoidea spp.	6.2	32.3	1283.1	4002.8	767.9	3150.4
Phoronida spp.	160.8	344.8	265.9	594.2	223.5	510.2
Urothoe poseidonis	179.8	252.5	79.4	173.4	119.9	214.2
Spiophanes bombyx	58.8	73.4	131.1	162.2	101.9	138.1
Lanice conchilega	214.7	601.5	24.4	82.6	101.1	397.3
Nephtys cirrosa	65.6	40.6	74.2	44.5	70.7	43.1
Assemblage IN3 (*Nephtys cirrosa*–*Spiophanes bombyx*)						
Occurrence	13		1		14	
Biomass	0.2	0.2	0.0	NA	0.1	0.2
Species richness	12.1	2.4	7	NA	11.7	2.7
Nephtys cirrosa	71.6	29.2	51.7	NA	70.1	28.5
Spiophanes bombyx	26.8	36.7	64.6	NA	29.5	36.7
Urothoe brevicornis	22.9	19.1	38.8	NA	24	18.9
Magelona papillicornis	21.9	27.1	0	NA	20.3	26.7
Pseudocuma spp.	16.9	23.2	0	NA	15.7	22.8
Scolelepis bonnieri	14.9	17.4	0	NA	13.8	17.2
Assemblage IN4 (*Spio* spp.–*N. cirrosa*)						
Occurrence	64		12		76	
Biomass	0.4	0.6	0.4	0.7	0.4	0.6
Species richness	14.7	3.8	13.7	3.3	14.5	3.7
Spio martinensis	153.2	179.1	34.5	60.9	134.5	171.4
Spio goniocephala	96.1	185.1	17.2	21.6	83.6	172.3
Nephtys cirrosa	54.5	28.4	44.1	36.3	52.9	29.8
Phoronida spp.	40	129.6	1.1	3.7	33.8	119.6
Spiophanes bombyx	29.9	40.4	37.7	67.1	31.1	45.2

	2006		**2008**		**Average**	
Variable	**Mean**	**Sd**	**Mean**	**Sd**	**Mean**	**Sd**
Urothoe brevicornis	15.1	27.6	31.2	50.9	17.7	32.5
Assemblage IN5 (***Owenia fusiformis–Abra alba***)						
Occurrence	4		4		8	
Biomass	6.4	5.1	9.9	12.1	8.1	8.8
Species richness	21.5	5.3	31.2	13	26.4	10.6
Owenia fusiformis	**1841.1**	**2511.7**	**2180.2**	**2721.4**	**2010.7**	**2431.2**
Abra alba	**562**	**623.1**	**1162.8**	**1246**	**862.4**	**966.9**
Lanice conchilega	235.8	312.3	985.1	1801	610.5	1261.9
Tellinoidea spp.	0	0	762.3	1315.3	381.1	952.6
Heteromastus filiformis	132.4	124.1	491	793.6	311.7	559.7
Caprellidae spp.	54.9	109.8	487.7	829.7	271.3	594.8
Actiniaria spp.	106.6	196.3	313.3	210.3	209.9	218.4
Kurtiella bidentata	100.1	82	284.2	88.3	192.2	126.1
Notomastus latericeus	142.1	233.5	180.9	144.3	161.5	180.9
Lagis koreni	6.5	7.5	293.9	275.4	150.2	236.9

Appendix 2.III Environmental variables of infaunal (IN) assemblages.

	2006		**2008**		**Average**	
Variable	**Mean**	**Sd**	**Mean**	**Sd**	**Mean**	**Sd**
Assemblage IN1 (*Echinoidea* spp.–*L. conchilega*)						
Occurrence	47		72		119	
Water depth (m)	16.5	4.1	16.2	3.8	16.3	3.9
D_{50} (µm)	257.1	75.8	267.8	66.7	263.6	70.3
Very fine sand (%)	6.9	8.3	5.9	8.6	6.3	8.5
OM (%)	0.7	0.3	1.0	0.4	0.9	0.4
Mud (%)	1.5	2.2	1.7	2.6	1.6	2.5
Mean near-bed salinity	30.6	1.5	30.7	1.3	30.7	1.4
Max. near-bed salinity	34.3	0.9	34.4	0.7	34.4	0.8
Mean bed shear stress (N m^{-2})	0.42	0.08	0.43	0.09	0.43	0.08
Max. bed shear stress (N m^{-2})	2.22	0.32	2.23	0.27	2.22	0.29
Assemblage IN2 (*Echinoidea* spp–*Phoronida* spp.)						
Occurrence	94		139		233	
Water depth (m)	22.1	4.7	23.2	4.3	22.7	4.5
D_{50} (µm)	331.8	52.8	357	44.9	346.8	49.7
Very fine sand (%)	0.7	2.6	0.3	1.2	0.5	1.9
OM (%)	0.4	0.1	0.7	0.2	0.6	0.2
Mud (%)	0.4	1.3	0.1	0.7	0.2	1
Mean near-bed salinity	32.7	1.5	33.2	1.2	33	1.4
Max. near-bed salinity	34.9	0.3	34.9	0.2	34.9	0.2
Mean bed shear stress (N m^{-2})	0.49	0.09	0.52	0.09	0.51	0.09
Max. bed shear stress (N m^{-2})	2.20	0.29	2.18	0.28	2.19	0.29
Assemblage IN3 (*Nephtys cirrosa*–*Spiophanes bombyx*)						
Occurrence	13		1		14	
Water depth (m)	21	6.3	12.9	NA	20.4	6.4
D_{50} (µm)	338.7	58.4	377.4	NA	341.4	57.1
Very fine sand (%)	0.4	1.1	0	NA	0.4	1
OM (%)	0.5	0.5	0.5	NA	0.5	0.5
Mud (%)	0	0	0	NA	0	0
Mean near-bed salinity	32.5	1.7	31.8	NA	32.5	1.7
Max. near-bed salinity	34.9	0.3	34.8	NA	34.9	0.3
Mean bed shear stress (N m^{-2})	0.49	0.11	0.60	NA	0.49	0.11
Max. bed shear stress (N m^{-2})	2.20	0.23	2.33	NA	2.20	0.22
Assemblage IN4 (*Spio* spp.–*N. cirrosa*)						
Occurrence	64		12		76	
Water depth (m)	23	4.8	22.6	7.1	22.9	5.2
D_{50} (µm)	387.3	48.2	378.7	47	385.9	47.8
Very fine sand (%)	0	0.2	0	0	0	0.2
OM (%)	0.4	0.2	0.5	0.2	0.4	0.2
Mud (%)	0	0.3	0	0	0	0.3
Mean near-bed salinity (ppt)	33.2	1.2	33.3	1.5	33.2	1.2

	2006		2008		Average	
Variable	**Mean**	**Sd**	**Mean**	**Sd**	**Mean**	**Sd**
Max. near-bed salinity (ppt)	35	0.2	35	0.2	35	0.2
Mean bed shear stress (N m^{-2})	0.54	0.10	0.55	0.07	0.54	0.09
Max. bed shear stress (N m^{-2})	2.19	0.27	2.30	0.38	2.21	0.29
Assemblage IN5 (***Owenia fusiformis–Abra alba***)						
Occurrence	4		4		8	
Water depth (m)	23	3.9	23.1	4	23.1	3.6
D_{50} (µm)	154.6	51.4	159.6	36.2	157.1	41.3
Very fine sand (%)	**23.1**	**7.5**	**15.4**	**16.7**	**19.3**	**12.6**
OM (%)	**2**	**0.7**	**2.2**	**1.3**	**2.1**	**1.0**
Mud (%)	17	7.8	13.3	9.4	15.4	8.0
Mean near-bed salinity	31.6	0.6	31.6	0.6	31.6	0.5
Max. near-bed salinity	34.7	0.1	34.7	0.1	34.7	0.1
Mean bed shear stress (N m^{-2})	0.37	0.01	0.37	0.01	0.37	0.01
Max. bed shear stress (N m^{-2})	2.09	0.29	2.09	0.29	2.09	0.29

Appendix 2.IV Species composition of epifaunal (EP) assemblages (ind. m^{-2}), biomass (g AFDW m^{-2}) and species richness (species per boxcore). Dufrêne–Legendre indicators are indicated in bold.

	2006		**2008**		**Average**	
Variable	**Mean**	**Sd**	**Mean**	**Sd**	**Mean**	**Sd**
Assemblage EP1 (*Ensis* spp.– *Ophiura* spp.–***S. subtruncata***)						
Occurrence	81		91		172	
Biomass	80.4	148	108.7	144.2	95.4	146.3
Species richness	12.6	4.4	13.6	3.2	13.1	3.8
Ensis spp.	8.4	15	22.6	42.9	15.9	33.5
Ophiura albida	3.8	6.5	2.5	8.4	3.1	7.6
Ophiura ophiura	1.2	1.8	1	1.2	1.1	1.5
Nassarius nitidus	1.2	2.5	1.1	1.9	1.1	2.2
Spisula subtruncata	1.2	1.7	0.9	1.6	1	1.6
Nassarius reticulatus	1	2.4	0.9	1.4	1	1.9
Assemblage EP2 (*Ensis* spp.–*Ophiura albida*)						
Occurrence	130		129		259	
Biomass	10.2	10.6	21.4	32.6	15.8	24.8
Species richness	7.8	3	8.9	2.4	8.3	2.8
Ensis spp.	1.1	1.2	1.5	1.6	1.3	1.4
Ophiura albida	1.5	1.9	1	1.2	1.3	1.6
Thia scutellata	0.2	0.2	0.5	0.3	0.3	0.3
Spisula solida	0	0	0.6	0.5	0.3	0.5
Ophiura ophiura	0.1	0.2	0.2	0.2	0.2	0.2
Spisula elliptica	0.1	0.1	0.2	0.3	0.1	0.2
Assemblage EP3 (*Ophiura albida*)						
Occurrence	6		-		6	
Biomass	1.3	0.7	NA	NA	1.3	0.7
Species richness	3	1	NA	NA	3	1
Ophiura albida	0.7	1	NA	NA	0.7	1
Liocarcinus holsatus	0.1	0.1	NA	NA	0.1	0.1
Ophiura ophiura	0.1	0.1	NA	NA	0.1	0.1
Tellina fabula	0	0.1	NA	NA	0	0.1
Assemblage EP4 (***Ensis* spp.**)						
Occurrence	4		5		9	
Biomass	1099.1	684.9	328.3	312.4	670.9	624.3
Species richness	4.5	1.7	8	3.5	6.4	3.3
***Ensis* spp.**	79.1	62.3	91.1	68.1	85.8	61.8
Ophiura albida	14.1	17.6	1.6	1.6	7.2	12.7
Asterias rubens	1.3	1.9	0.1	0.1	0.6	1.4
Actiniaria spp.	0.2	0.4	0.9	1.9	0.6	1.4
Ophiura ophiura	0.4	0.5	0.2	0.3	0.3	0.4
Lutraria lutraria	0.4	0.5	0	0	0.2	0.4

	2006		2008		Average	
Variable	**Mean**	**Sd**	**Mean**	**Sd**	**Mean**	**Sd**
Assemblage EP5 (***A. alba***–*Actinaria* spp.–*O. albida*–*Venerupis senegalensis*)						
Occurrence	4		4		8	
Biomass	102.7	74.1	175.6	146.4	139.1	114.2
Species richness	13	1.4	10.8	1.5	11.9	1.8
Owenia fusiformis	96.4	114.9	191.5	143.1	143.9	130.5
Abra alba	41.4	36.5	136.1	102.6	88.8	87.4
Lanice conchilega	62.5	42.8	19	18.7	40.7	38.4
Tellinoidea spp.	0.1	0.1	11.8	20	5.9	14.5
Heteromastus filiformis	5.4	9.5	0.7	1.3	3	6.8
Caprellidae spp.	5.9	3.9	0	0	3	4.1
Actiniaria spp.	106.6	196.3	313.3	210.3	209.9	218.4
Kurtiella bidentata	100.1	82	284.2	88.3	192.2	126.1
Notomastus latericeus	142.1	233.5	180.9	144.3	161.5	180.9
Lagis koreni	6.5	7.5	293.9	275.4	150.2	236.9

Appendix 2.V Environmental variables of epifaunal (EP) assemblages.

	2006		**2008**		**Average**	
Variable	**Mean**	**Sd**	**Mean**	**Sd**	**Mean**	**Sd**
Assemblage EP1 (*Ensis* spp.– *Ophiura* spp.–***S. subtruncata***)						
Occurrence	81		91		172	
Water depth (m)	16	3.9	16.5	4.1	16.3	4
D_{50} (µm)	275	76.9	278.1	71.8	276.6	74
Very fine sand (%)	5	7.3	4.9	7.9	5	7.6
OM (%)	0.5	0.2	1	0.4	0.8	0.4
Mud (%)	1.1	2	1.4	2.5	1.3	2.3
Mean near-bed salinity	30.7	1.4	30.8	1.2	30.8	1.3
Max. near-bed salinity	34.4	0.7	34.5	0.7	34.5	0.7
Mean bed shear stress (N m^{-2})	0.42	0.08	0.43	0.09	0.43	0.08
Max. bed shear stress (N m^{-2})	2.22	0.32	2.23	0.27	2.22	0.29
Assemblage EP2 (*Ensis* spp.–*Ophiura albida*)						
Occurrence	130		129		259	
Water depth (m)	24	3.3	24.3	3.1	24.2	3.2
D_{50} (µm)	367.8	46.1	367.7	36.1	367.7	41.3
Very fine sand (%)	0.1	0.5	0	0.1	0.1	0.4
OM (%)	0.4	0.2	0.7	0.2	0.6	0.3
Mud (%)	0.2	1.4	0	0.3	0.1	1
Mean near-bed salinity	33.3	1	33.6	0.7	33.5	0.9
Max. near-bed salinity	35	0.1	35	0.1	35	0.1
Mean bed shear stress (N m^{-2})	0.52	0.09	0.53	0.08	0.53	0.09
Max. bed shear stress (N m^{-2})	2.13	0.21	2.11	0.18	2.12	0.20
Assemblage EP3 (*Ophiura albida*)						
Occurrence	6		-		6	
Water depth (m)	24	6.7	NA	NA	24	6.7
D_{50} (µm)	359.9	35.4	NA	NA	359.9	35.4
Very fine sand (%)	0	0	NA	NA	0	0
OM (%)	0.4	0.3	NA	NA	0.4	0.3
Mud (%)	0	0	NA	NA	0	0
Mean near-bed salinity	33.2	1.7	NA	NA	33.2	1.7
Max. near-bed salinity	34.9	0.3	NA	NA	34.9	0.3
Mean bed shear stress (N m^{-2})	0.56	0.09	NA	NA	0.56	0.09
Max. bed shear stress (N m^{-2})	2.33	0.29	NA	NA	2.33	0.29
Assemblage EP4 (***Ensis* spp.**)						
Occurrence	4		5		9	
Water depth (m)	21.8	1.3	19.1	3.8	20.3	3.1
D_{50} (µm)	312.4	49.8	399.4	41	360.7	62.2
Very fine sand (%)	1.1	1.4	0.6	1.2	0.8	1.2
OM (%)	1	0.7	0.6	0.2	0.8	0.5
Mud (%)	2.6	3.1	0.7	1.5	1.5	2.4
Mean near-bed salinity (ppt)	33.1	1	32.4	0.4	32.7	0.8

	2006		2008		Average	
Variable	**Mean**	**Sd**	**Mean**	**Sd**	**Mean**	**Sd**
Max. near-bed salinity (ppt)	35	0.2	34.9	0.1	34.9	0.1
Mean bed shear stress (N m^{-2})	0.61	0.07	0.64	0.05	0.63	0.06
Max. bed shear stress (N m^{-2})	2.32	0.25	2.53	0.10	2.44	0.20
Assemblage EP5 (***A. alba***–*Actinaria* spp.–*O. albida*–*Venerupis senegalensis*)						
Occurrence	4		4		8	
Water depth (m)	23	3.9	23.1	4	23.1	3.6
D_{50} (µm)	154.6	51.4	159.6	36.2	157.1	41.3
Very fine sand (%)	**23.1**	**7.5**	**15.4**	**16.7**	**19.3**	**12.6**
OM (%)	**2**	**0.7**	**2.2**	**1.3**	**2.1**	**1**
Mud (%)	17	7.8	13.3	9.4	15.4	8
Mean near-bed salinity	31.6	0.6	31.6	0.6	31.6	0.5
Max. near-bed salinity	34.7	0.1	34.7	0.1	34.7	0.1
Mean bed shear stress (N m^{-2})	0.37	001	0.37	0.01	0.37	001
Max. bed shear stress (N m^{-2})	2.09	0.29	2.09	0.29	2.09	0.27

Appendix 3.I Mean values and standard deviation of measured and modelled variables. Average values for location in the MV2 borrow pit are calculated for the time after cessation of sand extraction. Vfine: very fine sand; Sal mean: mean near-bed salinity; Sal max: maximum near-bed salinity; $\tau_{b\,mean}$: mean bed shear stress and $\tau_{b\,max}$: maximum bed shear stress.

Reference								
	2010		2011		2012		mean	
	mean	sd	mean	sd	mean	sd	mean	sd
D_{50}	282.7	26.2	300.4	46.9	300.8	52.6	293.7	42.7
Depth	21.1	2.0	23.0	1.7	22.3	3.6	22	2.7
OM	-	-	-	-	0.9	0.38	0.9	0.4
Mud	0.5	0.8	0.2	0.5	1.1	4.4	0.6	2.7
Vfine sand	0.3	0.3	0.4	0.7	0.9	2.3	0. 6	1.5
Sal mean	33.6	0.2	33.6	0.3	33.0	0. 8	33.4	0.6
Sal max	34.2	0.2	34.2	0.2	34.5	0.4	34.3	0.3
$\tau_{b\,mean}$	0.5	0.1	0.4	0.0	0.4	0.0	0.4	0.1
$\tau_{b\,max}$	1.9	0.3	1.8	0.2	1.9	0.2	1.9	0.2

Edge										
	t_{recent}		t_1		t_2		t_3		t_{no}	
	mean	sd	mean	sd	mean	sd	mean	sd	mean	sd
D_{50}	395	118.2	223.2	54.9	281.8	57.6	310	119.7	263.3	34.3
Depth	32.6	4.7	26.4	4.5	26.4	2.9	28	3.1	22.3	2.1
OM	0.8	0.7	-	-	-	-	1	0.6	0.9	0.3
Mud	0.7	1.6	7.4	8.2	0.5	0.8	1.1	1.9	1.3	3.2
Vfine sand	1.4	2.3	11.1	8.7	1.8	1.5	3	4.8	1.6	4.3
Sal mean	33.7	0.1	33.5	0.1	33.6	0.2	33.5	0.1	33.5	0.2
Sal max	34.3	0	34.2	0.1	34.2	0.1	34.2	0.1	34.2	0.1
$\tau_{b\,mean}$	0.4	0	0.2	0.1	0.4	0.1	0.3	0.1	0.4	0.1
$\tau_{b\,max}$	2.3	0.9	2	0.5	2.2	0.8	1.9	0.5	2.4	0.9

Crest (parallel)						
	$t_{0.5}$		$t_{1.5}$		$t_{2.5}$	
	mean	sd	mean	sd	mean	sd
D_{50}	186.9	45.7	285.2	43.7	352.3	106.8
Depth	30.4	0.5	33.3	0.8	32	1.7
OM	-	-	-	-	1.1	1.1
Mud	3.5	3.3	2.8	5.5	4.6	7.5
Vfine sand	19.2	8.7	5.7	4.9	4.1	5.5
Sal mean	33.4	0	33.4	0	33.4	0
Sal max	34.1	0	34.1	0	34.1	0
$\tau_{b\,mean}$	0.3	0	0.3	0	0.3	0
$\tau_{b\,max}$	1.3	0.1	1.3	0.1	1.3	0.1

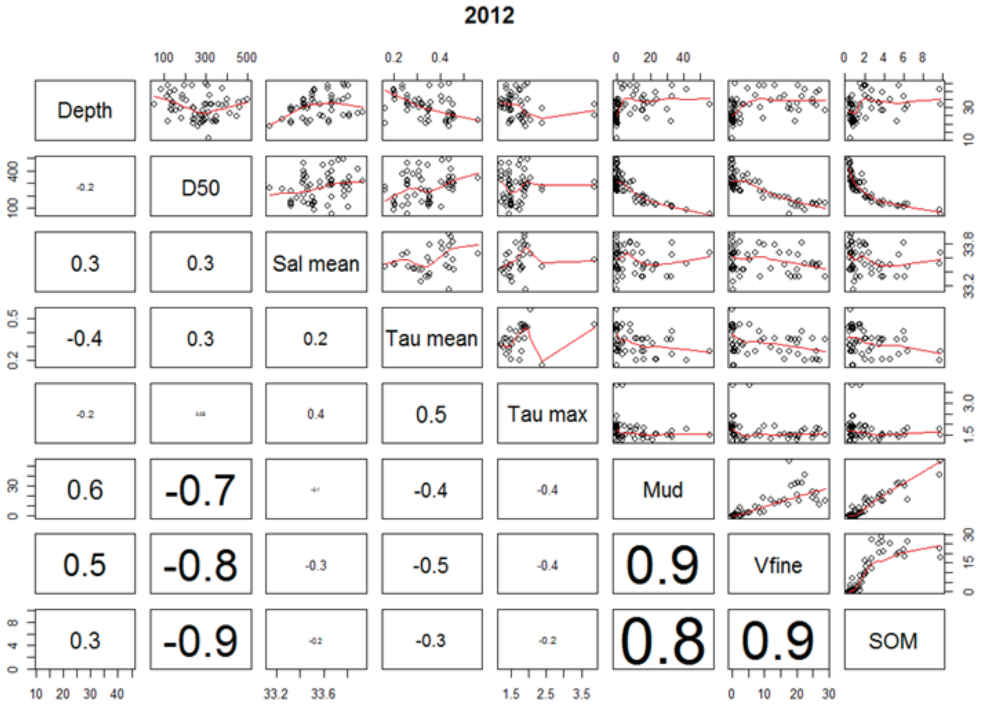
Crest (oblique)						
	$t_{0.5}$		$t_{1.5}$		$t_{2.5}$	
	mean	sd	mean	sd	mean	sd
D_{50}	-	-	279.4	57.8	240.7	126.9
Depth	-	-	27.8	0.4	31.1	2
OM	-	-	-	-	3.1	4.5
Mud	-	-	0.6	0.8	15.8	26.6
Vfine sand	-	-	2.7	2.5	5.3	8.3

Sal mean	-	-	33.7	0	33.7	0
Sal max	-	-	34.2	0	34.2	0
$\tau_{b\ mean}$	-	-	0.3	0	0.3	0
$\tau_{b\ max}$	-	-	1.5	0	1.5	0

Trough (parallel)						
	t_{recent}		t_1		t_2	
	mean	sd	mean	sd	mean	sd
D_{50}	376.9	134.8	231	83.4	221.6	154.7
Depth	37.8	3.3	37.3	3.4	33.9	2
OM	-	-	-	-	3.9	3
Mud	0.4	0.4	5.4	7.3	13.8	11.5
Vfine sand	2.4	2.4	14.1	10	17.6	14
Sal mean	33.5	0	33.5	0	33.5	0
Sal max	34.1	0	34.1	0	34.2	0
$\tau_{b\ mean}$	0.3	0	0.3	0	0.3	0.1
$\tau_{b\ max}$	1.8	0.1	1.6	0.3	1.5	0.3
Trough (oblique)						
	t_{recent}		t_1		t_2	
	mean	sd	mean	sd	mean	sd
D_{50}	-	-	346.1	120.4	341.3	116.3
Depth	-	-	31.3	3.7	34.4	2.4
OM	-	-	-	-	1.1	0.9
Mud	-	-	0.2	0.4	6.3	7.2
Vfine sand	-	-	1.9	3.2	5.6	6.9
Sal mean	-	-	33.7	0	33.7	0
Sal max	-	-	34.2	0	34.2	0
$\tau_{b\ mean}$	-	-	0.3	0	0.3	0
$\tau_{b\ max}$	-	-	1.5	0	1.5	0
Deep SE						
	t_{recent}		t_1		t_2	
	mean	sd	mean	sd	mean	sd
D_{50}	319.5	67.5	311.5	136.6	137.7	37.7
Depth	33.9	5.4	34.6	3.2	38.3	3.3
OM	-	-	2.3	1.7	4.3	3.1
Mud	0	0	5.7	9	22.3	11.6
Vfine sand	2.1	1.8	8.7	8.3	23.3	4.4
Sal mean	33.4	0.1	33.5	0	33.5	0
Sal max	34.1	0.1	34.1	0.1	34.2	0
$\tau_{b\ mean}$	0.3	0.1	0.3	0.1	0.2	0.1
$\tau_{b\ max}$	1.4	0.1	1.4	0.2	1.6	0.2
Deep NW						
	t_{recent}		t_1		t_2	
	mean	sd	mean	sd	mean	sd
D_{50}	358.8	152	221	107.5	116.8	-
Depth	39.9	4.2	38.2	6.2	43.6	-
OM	2.4	2.1	2.2	0.5	5.5	-
Mud	6.1	11.3	12.7	8.3	32.9	-
Vfine sand	5.8	7.4	15.2	8.6	20.5	-

Sal mean	33.8	0.1	33.6	0.2	33.6	-
Sal max	34.3	0	34.2	0.1	34.3	-
$\tau_{b\ mean}$	0.4	0.1	0.3	0.1	0.2	-
$\tau_{b\ max}$	1.8	0.2	1.5	0.3	1.4	-
Shipping lane						
	2006		2008		$t_{not\ recent}$ (2012)	
D_{50}	228.8	-	193.9	-	135.1	15.6
Depth	28.4	-	28.6	-	26.8	3.6
OM	1.0	-	1.4	-	4.7	1.1
Mud	7.5	-	2.5	-	23.1	9.3
Vfine sand	12.3	-	2.7	-	21.9	4.8
Sal mean	32.4	-	32.4	-	33.3	-
Sal max	34.9	-	34.9	-	34.0	-
$\tau_{b\ mean}$	0.36	-	0.36	-	0.35	-
$\tau_{b\ max}$	1.69	-	1.69	-	1.51	-

Appendix 3.II. Pairplot of environmental variables in 2012. The lower diagonal part shows the Spearman's rank correlation coefficient with the size of the font being proportional to the coefficient. D_{50}: median grain size, Sal mean: mean near-bed salinity, Tau mean: mean bed shear stress, Tau max: maximum bed shear stress, Mud: fraction mud, Vfine: fraction very fine sand, SOM: fraction sediment organic matter.



Appendix 3.III: Mean values of infauna in term of biomass and species density. Average values per location and per time after cessation of sand extraction or for the reference area year of sampling.

	Reference							
	2010		2011		2012		Mean	
	Mean	sd	Mean	sd	Mean	sd	Mean	sd
Biomass	19.8	22.6	17.4	24.9	11.68	14.7	16.1	20.3
Species richness	27.5	7.4	18.3	5.5	7.4	3.4	17.6	10.5
	Abundance (n m^{-2})							
Eteone sp.	819.3	1021.6	31	30	0	0	323.6	739.6
Spiophanes bombyx	584.4	792.6	50.4	74	7.8	22.1	240.3	558.3
Terebellidae sp.	444.6	677.7	14.2	28.2	0	0	175	466.8
Urothoe poseidonis	112.5	114.3	173.1	173.7	184.3	295.8	154	211.8
Echinoidea sp.	320	529.6	0	0	0	0	123.6	359.5
Lanice conchilega	247.8	899.8	2.6	5.4	0	0	96.3	562.2
Nemertea sp.	124.6	148.5	71.1	112.3	23.5	32.8	73.4	115.2
Phoronida sp.	31.9	61.8	218.3	365.3	27.5	113.2	72.6	201.3
Heteromastus filiformis	3.8	6.1	253.2	760.6	19.6	45.7	66.6	363.9
Nephtys sp.	37.2	24.1	10.3	20.9	125.5	87.8	65.2	75
	Biomass (g AFDW m^{-2})							
Echinocardium cordatum	5.2	6.5	0.9	2.5	6.2	14	4.6	9.7
Ensis directus	6.4	15.5	7.6	17.3	0	0	4.2	12.8
Pestarella tyrrhena	0.8	1.6	3.1	6	3	8.5	2.2	6
Ensis sp.	2.3	7.9	3.6	11.5	0.5	2.2	1.9	7.4
Liocarcinus holsatus	0.9	3.8	0	0	0	0	0.4	2.4
Lanice conchilega	0.8	1.7	0	0.1	0	0	0.3	1.1
Nephtys cirrosa	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3
Nephtys hombergii	0.2	0.8	0	0.1	0.3	0.7	0.2	0.6
Angulus fabula	0	0	0	0	0.4	1.4	0.2	0.9
Nassarius reticulatus	0.4	1.3	0	0	0	0	0.2	0.8

	Edge									
	t_{recent}		t_1		t_2		t_3		t_{no}	
	Mean	sd	Mean	sd	Mean	sd	Mean	sd	Mean	sd
Biomass	0.1	0.2	4.8	4.6	4.9	5	1.4	2	8	10.4
Species richness	5	5.2	20	12.7	18.2	5.1	5.7	4	17.5	9.9
	Abundance (n m^{-2})									
Spiophanes bombyx	2.6	5.8	1162.8	2074.5	36.6	59.2	0	0	359.6	805.7
Eteone sp.	2.6	5.8	529.7	350.3	15.1	9.7	0	0	207.8	533.6
Urothoe poseidonis	0	0	3.2	6.5	129.2	179.6	0	0	97.3	182.3
Magelona sp.	2.6	5.8	12.9	25.8	12.9	20	0	0	91.7	173.1
Nephtys sp.	0	0	16.1	32.3	4.3	6.7	66.7	115.5	82.5	156.4
Echinoidea sp.	0	0	16.1	24.5	0	0	0	0	80.3	278.9
Bathyporeia guilliamsoniana	2.6	5.8	19.4	38.8	36.6	89.7	0	0	73	167.2
Scoloplos (scoloplos) armiger	29.3	58.5	180.9	278.1	148.6	267.8	177.8	192.4	65	114.9
Nephtys cirrosa	0	0	6.5	12.9	21.5	24.1	0	0	53.5	39.9
Magelona johnstoni	0	0	9.7	19.4	157.2	372.5	22.2	38.5	52	81.4
	Biomass (g AFDW m^{-2})									

Echinocardium cordatum	0	0	0	0	0	0	0	0	3.2	5.6
Corystes cassivelaunus	0	0	0	0	0	0	0	0	1	3.7
Angulus fabula	0	0	0	0	0	0	0	0	1	3.2
Notomastus latericeus	0	0	0	0	0	0.1	0	0	0.4	1.2
Ophiura ophiura	0	0	0	0	0.3	0.7	0	0	0.4	1.1
Nephtys hombergii	0	0	0.1	0.2	0	0.1	0.2	0.3	0.2	0.4
Nephtys cirrosa	0	0	0	0.1	0.2	0.2	0	0	0.2	0.4
Scoloplos (scoloplos) armiger	0	0.1	0.4	0.7	0.6	1.4	0.3	0.5	0.2	0.5
Spisula solida	0	0	0	0	0	0	0	0	0.2	0.7
Spiophanes bombyx	0	0	0.5	1	0	0	0	0	0.2	0.4

	Crests (parallel)					
	$t_{0.5}$		$t_{1.5}$		$t_{2.5}$	
	Mean	sd	Mean	sd	Mean	sd
Biomass	1.5	0.7	1.6	0.7	14.9	11.5
Species richness	21.8	4.4	15	2.6	7.5	1
	Abundance (n m^{-2})					
Scoloplos (scoloplos) armiger	85.3	72.3	122.7	91.1	150	63.8
Caprellidae sp.	31	42.5	0	0	116.7	233.3
Nephtys sp.	12.9	12.9	9.7	12.4	116.7	137.4
Notomastus latericeus	2.6	5.8	6.5	12.9	116.7	113.9
Abra alba	93	78.3	67.8	119.1	116.7	157.5
Angulus fabula	5.2	7.1	19.4	24.7	50	63.8
Tellinoidea sp.	10.3	10.8	0	0	50	100
BIVALVIA	5.2	7.1	42	59.1	33.3	38.5
Nephtys cirrosa	2.6	5.8	6.5	12.9	33.3	38.5
Nephtys hombergii	0	0	9.7	12.4	33.3	38.5
	Biomass (g AFDW m^{-2})					
Notomastus latericeus	0	0	0	0.1	5.6	6
Abra alba	0.8	0.8	0.3	0.5	4.9	8.7
Angulus fabula	0	0	0	0	1.6	2
Ophiura ophiura	0	0	0.1	0.1	0.6	1.2
Nephtys cirrosa	0	0	0	0	0.4	0.6
Ophiura albida	0	0	0	0	0.4	0.8
Nephtys hombergii	0	0	0.3	0.3	0.3	0.5
Euspira pulchella	0	0	0.2	0.3	0.2	0.5
Scoloplos (scoloplos) armiger	0	0	0.3	0.3	0.2	0.1
Bivalvia sp.	0	0	0	0	0.2	0.3

	Crests (oblique)					
	$t_{0.5}$		$t_{1.5}$		$t_{2.5}$	
	Mean	sd	Mean	sd	Mean	sd
Biomass	-	-	3.6	4.7	51.1	53.9
Species richness	-	-	18.5	3.3	9	1.4
	Abundance (n m^{-2})					
Nemertea sp.	-	-	58.1	60.1	300	476.1
Kurtiella bidentata	-	-	25.8	51.7	133.3	180.5
Abra alba	-	-	87.2	165.9	100	158.7
Heteromastus filiformis	-	-	287.5	332.3	66.7	94.3
Barnea candida	-	-	0	0	66.7	133.3

Poecilochaetus serpens	-	-	3.2	6.5	66.7	133.3
Urothoe brevicornis	-	-	9.7	19.4	66.7	133.3
Echinocardium cordatum	-	-	6.5	7.5	50	63.8
Nephtys hombergii	-	-	22.6	30.5	50	100
Pholoe inornata	-	-	0	0	50	63.8
	Biomass (g AFDW m^{-2})					
Barnea candida	0	0	0	0	9.1	25.7
Bivalvia sp.	0	0	0	0	6.3	17.4
Abra alba	0.8	0.8	1.2	2.9	4.8	8.3
Echinocardium cordatum	0	0	0.1	0.2	3.8	8.5
Notomastus latericeus	0	0	0.1	0.2	3	4.8
Angulus fabula	0	0	0	0.1	2	2.2
Nephtys hombergii	0	0	0.2	0.3	0.8	1.7
Petricolaria pholadiformis	0	0	0	0	0.5	1.5
Nassarius sp.	0	0	0	0	0.5	1.5
Spisula subtruncata	0	0	0	0	0.5	1.4
	Trough (parallel)					
	t_{recent}		t_1		t_2	
	Mean	sd	Mean	sd	Mean	sd
Biomass	0.1	0.1	1.4	1	20.7	11.4
Species richness	7.3	2	11.5	3.4	8.3	2.3
	Abundance (n m^{-2})					
Tellinoidea sp.	0	0	0	0	711.1	1231.7
Abra alba	0	0	38.8	44.8	311.1	269.4
Scoloplos (scoloplos) armiger	12.9	11.6	71.1	53.3	222.2	328.9
Nephtys sp.	2.2	5.3	6.5	12.9	177.8	252.4
Abra sp.	0	0	0	0	88.9	101.8
Euspira pulchella	0	0	0	0	44.4	38.5
Nephtys cirrosa	0	0	6.5	12.9	44.4	77
Notomastus latericeus	0	0	67.8	109.8	44.4	77
Pholoe inornata	0	0	0	0	44.4	77
Corophium	0	0	0	0	22.2	38.5
	Biomass (g AFDW m^{-2})					
Abra alba	0	0	0.2	0.2	14.5	12.8
Phyllodoce groenlandica	0	0	0	0	2.6	4.5
Notomastus latericeus	0	0	0.3	0.6	1.4	2.4
Abra sp.	0	0	0	0	0.6	1
Scoloplos (scoloplos) armiger	0	0	0.1	0.1	0.4	0.7
Nephtys hombergii	0	0	0.1	0.1	0.3	0.6
Nephtys cirrosa	0	0	0.1	0.1	0.2	0.4
Euspira pulchella	0	0	0	0	0.2	0.2
Nephtys longosetosa	0	0	0	0	0.2	0.3
Nephtys sp.	0	0	0	0	0.1	0.1
Trough (oblique)						
	t_{recent}		t_1		t_2	
	Mean	sd	Mean	sd	Mean	sd
Biomass	-	-	0.1	0.1	21.9	26.9
Species richness	-	-	6.8	3.9	8.8	2.6

	Abundance (n m^{-2})					
Abra alba	-	-	3.2	6.5	1316.7	2633.3
Tellimya ferruginosa	-	-	0	0	200	356.9
Heteromastus filiformis	-	-	0	0	133.3	224.4
Nephtys sp.	-	-	6.5	7.5	133.3	154
Euspira pulchella	-	-	0	0	116.7	113.9
Urothoe brevicornis	-	-	0	0	116.7	233.3
Echinocardium cordatum	-	-	0	0	100	66.7
Scoloplos (scoloplos) armiger	-	-	3.2	6.5	50	63.8
Angulus fabula	-	-	6.5	12.9	33.3	38.5
Megaluropus agilis	-	-	3.2	6.5	33.3	38.5
	Biomass (g AFDW m^{-2})					
Echinocardium cordatum	-	-	0	0	11.5	15
Abra alba	-	-	0	0	6.2	12.4
Eunereis longissima	-	-	0	0	1.8	3.7
Euspira pulchella	-	-	0	0	1.3	1.1
Angulus fabula	-	-	0	0	0.4	0.7
Nephtys sp.	-	-	0	0	0.2	0.3
Tellimya ferruginosa	-	-	0	0	0.2	0.4
Urothoe brevicornis	-	-	0	0	0.2	0.3
Scoloplos (scoloplos) armiger	-	-	0	0	0	0
Heteromastus filiformis	-	-	0	0	0	0
	Deep SE					
	t_{recent}		t_1		t_2	
	Mean	sd	Mean	sd	Mean	sd
Biomass	0.2	0.3	21.2	35.2	78.1	66.2
Species richness	9.7	2.5	10.8	5.6	12.2	5.9
	Abundance (n m^{-2})					
	Mean	sd	Mean	sd	Mean	sd
Tellinoidea sp.	0	0	0	0	1626.7	1476.2
Abra alba	0	0	159.6	252.3	1453.3	1554.5
Heteromastus filiformis	0	0	2.6	5.8	773.3	1692.2
Abra sp.	0	0	0	0	506.7	674.3
Nemertea sp.	17.2	19.7	47.8	54	200	411
Nephtys hombergii	0	0	7.8	11.6	146.7	109.5
Notomastus latericeus	0	0	191.2	427.6	133.3	170
Euspira pulchella	4.3	7.5	0	0	106.7	101.1
Kurtiella bidentata	0	0	0	0	93.3	208.7
Polynoidae sp.	0	0	0	0	93.3	173.8
	Biomass (g AFDW m^{-2})					
Abra alba	0	0	7.1	9.8	52.5	52.2
Echinocardium cordatum	0	0	10	22.3	9.2	12.8
Notomastus latericeus	0	0	1	2.2	3.8	5.2
Nephtys hombergii	0	0	0	0.1	3.2	2.5
Angulus fabula	0	0	1.2	2.6	1.8	2.6
Eunereis longissima	0	0	0	0	1.8	2.4
Nassarius reticulatus	0	0	0	0	1.6	3.7
Abra sp.	0	0	0	0	1.5	2.9

Ophiura ophiura	0	0	0.4	0.7	0.5	1.2
Lagis koreni	0	0	0	0	0.5	1.1
	Deep NW					
	t_{recent}		t_1		t_2	
	Mean	sd	Mean	sd	Mean	sd
Biomass	8.8	15.7	27.4	21.9	42.3	-
Species richness	7.6	5.7	10	9.4	4	-
	Abundance (n m^{-2})					
Abra alba	64.3	119.4	805	781.5	933.3	-
Euspira pulchella	4.8	13.7	33.3	38.5	66.7	-
Heteromastus filiformis	1.6	4.6	69.9	131.3	66.7	-
Tellinoidea sp.	33.3	50.4	50	63.8	66.7	-
Abra sp.	8.3	23.6	0	0	0	-
Actiniaria sp.	0	0	0	0	0	-
Ampelisca brevicornis	0	0	0	0	0	-
Angulus fabula	0	0	0	0	0	-
Arenicola sp.	0	0	0	0	0	-
Aricidea minuta	0	0	0	0	0	-
	Biomass (g AFDW m^{-2})					
Abra alba	2.7	7.1	21.2	16.7	42.2	-
Euspira pulchella	0.1	0.2	0	0	0.1	-
Tellinoidea sp.	0	0	0	0	0	-
Heteromastus filiformis	0	0	0	0	0	-
Abra sp.	0	0	0	0	0	-
Actiniaria sp.	0	0	0	0	0	-
Ampelisca brevicornis	0	0	0	0	0	-
Angulus fabula	0	0	0	0	0	-
Arenicola sp.	0	0	0	0	0	-
Aricidea minuta	0	0	0	0	0	-
	Shipping lane area					
	2006		2008		2012	
	Mean	sd	Mean	sd	Mean	sd
Biomass	6.4	5.1	9.9	12.1	27.5	40.3
Species richness	21.5	5.3	31.2	13	10.0	5.9
	Abundance (n m^{-2})					
Heteromastus filiformis	132.4	124.1	491	793.6	516.7	989.4
Tellinoidea sp.	0	0	0	0	383.3	459.9
Abra alba	562	144	1162.8	1245.99	200	144
Oligochaeta sp.	3.2	6.5	6.5	-	166.7	290.6
Tellimya ferruginosa	71.1	56.3	67.8	-	116.7	233.3
Euspira nitida	0	0	32.3	-	100	86.1
Kurtiella bidentata	100.1	82	284.2	-	66.7	133.3
Nephtys sp.	48.4	96.9	16.2	-	50	33.3
Abra sp.	0	0	0	-	50	63.8
Echiura sp.	0	0	0	0	33.3	38.5
	Biomass (g AFDW m^{-2})					
Echinocardium cordatum	-	-	-	-	16.8	33.53
Abra alba	-	-	-	-	5.3	4.98

Actiniaria sp.	-	-	-	-	1.7	3.4
Nephtys hombergii	-	-	-	-	1	1.7
Notomastus latericeus	-	-	-	-	1	2.02
Angulus fabula	-	-	-	-	0.8	1.56
Tellimya ferruginosa	-	-	-	-	0.2	0.41
Euspira pulchella	-	-	-	-	0.2	0.14
Nemertea sp.	-	-	-	-	0.1	0.24
Nereis	-	-	-	-	0.1	0.22

Appendix 3.IV

Table 1. GAM output log10 transformed infaunal biomass

Variable	edf	F-value	p-value	Deviance explained
D_{50}	5	10.7	***	48.5
Time	Estimate	Std error	t-value	Pr(>|t|)
Intercept ($t_{0.5}$)	-0.01	0.22	-0.7	
t_{1}	0.55	0.23	2.40	*
$t_{1.5}$	0.53	0.27	1.95	.
t_{2}	1.01	0.23	4.33	***
$t_{2.5}$	1.35	0.28	4.86	***
t_{3}	0.42	0.32	1.32	
t_{no}	0.95	0.23	4.12	***
$t_{\text{not recent}}$	0.49	0.27	1.82	.
t_{recent}	0.37	0.24	1.52	

Significance codes: ‘***’ : 0.001: ‘**’ : 0.01, ‘*’ : 0.05, ‘.‘ : > 0.05, GCV score = 0.22, Scale estimator = 0.21 and n = 165

Table 2. GAM output infaunal species richness

Variable	edf	F-value	*p*-value	Deviance explained
D_{50}	4	4.83	***	40.3
Depth	4	5.48	**	
Time	Estimate	Std error	t-value	Pr(>|t|)
Intercept ($t_{0.5}$)	22.58	3.60	6.28	***
t_{1}	-8.60	3.80	–2.27	*
$t_{1.5}$	-4.20	4.35	-0.97	
t_{2}	-8.77	3.84	–2.29	*
$t_{2.5}$	-9.82	4.39	–2.24	*
t_{3}	–17.14	5.04	-3.40	***
t_{no}	–10.80	4.15	–2.60	*
$t_{\text{not recent}}$	–16.84	4.52	-3.72	***
t_{recent}	-9.87	4.07	–2.42	*

Significance codes: ‘***’ : 0.001: ‘**’ : 0.01, ‘*’ : 0.05, ‘.‘ : > 0.05, GCV score = 58.84, Scale estimator = 52.78 and n = 165

Table 3. GAM output epifaunal species richness

Variable	Estimate	Std error	t-value	Deviance explained
Time				51.6%
Intercept ($t_{0.5}$)	2.00	2.20	0.99	Pr(>|t|)
t_{1}	4.00	2.16	1.86	.
$t_{1.5}$	6.00	2.48	2.42	*
t_{2}	6.50	2.15	3.03	**
$t_{2.5}$	9.75	2.48	3.93	***
t_{3}	7.00	2.86	2.45	*
t_{no}	6.59	2.11	3.12	**
$t_{\text{not recent}}$	12.33	2.61	4.72	***
t_{recent}	1.13	2.15	0.52	

Significance codes: ‘***’ : 0.001: ‘**’ : 0.01, ‘*’ : 0.05, ‘.’ : > 0.05, GCV score = 58.84, Scale estimator = 52.78 and n = 84

Table 4. GAM output epifaunal biomass

Variable	edf	F-value	*p*-value	Deviance explained
Mud	2	5.3	**	53.2%
Sal mean	3	3.2	*	
Time	Estimate	Std error	t-value	Pr(>|t|)
Intercept ($t_{0.5}$)	0.47	0.28	1.69	
t_{1}	0.25	0.30	0.83	
$t_{1.5}$	0.21	0.34	0.63	
t_{2}	0.83	0.30	2.73	***
$t_{2.5}$	0.53	0.36	1.48	
t_{3}	0.29	0.39	0.75	
t_{no}	0.45	0.29	1.52	
$t_{\text{not recent}}$	-0.03	0.42	-0.07	
t_{recent}	0.06	0.30	0.19	

Significance codes: ‘***’ : 0.001: ‘**’ : 0.01, ‘*’ : 0.05, ‘.’ : > 0.05, GCV score = 0.18, Scale estimator = 0.15 and n = 84

Appendix 3.V

Table 1. Multiple regressions of environmental variables and infaunal nMDS scores for 3D ordination for 2012, r2 is the squared Spearman's rank correlation coefficient and *p*-value.

Variable	NMDS1	NMDS2	NMDS3	r^2	Pr(>r)
D_{50}	0.98	0.16	-0.12	0.56	***
Depth	0.68	-0.35	-0.64	0.56	***
Time (centroids)	-	-	-	0.43	***
t_{1}	0.43	-0.11	0.15		
t_{2}	0.30	-0.13	-0.17		
$t_{2.5}$	0.05	-0.08	0.09		
t_{3}	0.05	0.15	0.28		
t_{no}	-0.44	-0.07	0.02		
$t_{not\ recent}$	0.21	-0.00	-0.38		
t_{recent}	0.21	-0.01	-0.38		

Significance codes: '***' : 0.001: '**' : 0.01, '*' : 0.05

Table 2. Multiple regressions of environmental variables and epifaunal nMDS scores for 2D ordination for 2012, r2 is the squared Spearman's rank correlation coefficient and *p*-value.

Variable	NMDS1	NMDS2	r^2	Pr(>r)
Vfine sand	-0.76311	0.64627	0.6056	***
Tau mean	0.94403	-0.32986	0.6622	***
Tau max	0.93892	-0.34413	0.2272	*
Time (centroids)	-	-	0.6941	***
t_{1}	-	-		
t_{2}	–1.2696	0.0285		
$t_{2.5}$	-0.3378	0.1123		
t_{3}	0.5486	-0.2743		
t_{no}	1.4204	-0.2615		
$t_{not\ recent}$	–1.0058	0.6398		
t_{recent}	1.4525	-0.0941		

Significance codes: '***' : 0.001: '**' : 0.01, '*' : 0.05

Appendix 3.VI Mean values of epifauna in term of biomass and species density. Average values per location and per time after cessation of sand extraction or for the reference area year of sampling. Because the number of samples in the reference area decreased after 2010, average values of the exclusion area are given.

	Reference							
	2010		2011		2012		Mean (exclusion)	
	Mean	sd	Mean	sd	Mean	sd	Mean	sd
Biomass	23.6	37.2	3.7	-	2.6	-	5	4.8
Species richness	10.5	3.8	5	-	8	-	7.5	5.6
	Abundance (n m^{-2})							
Ophiura ophiura	0.7	0.6	0.7	-	0.5	-	0.6	0.5
Ensis sp.	1.2	2.1	0	-	0	-	0.0	0.0
Nassarius reticulatus	0.5	0.5	0.1	-	0	-	0.1	0.2
Tellina fabula	0	0	0	-	0	-	0.0	0.0
Spisula subtruncata	0.5	0.6	0	-	0.1	-	0.4	0.7
Ophiura albida	0.3	0.3	0	-	0	-	0.1	0.1
Nassarius nitidus	0.2	0.4	0	-	0	-	0.1	0.3
Thia scutellata	0.2	0.2	0.1	-	0.1	-	0.1	0.1
Pagurus bernhardus	0.1	0.1	0.2	-	0.1	-	0.1	0.1
Corystes cassivelaunus	0.1	0.1	0.1	-	0.1	-	0.1	0.1
	Biomass (g WW m^{-2})							
Ensis sp.	14.12	27.9	0	-	0	-	8.05	20.86
Ophiura ophiura	2.27	2.09	2.45	-	1.42	-	2.11	1.7
Nassarius reticulatus	1.66	1.63	0.13	-	0	-	1.85	2.79
Actiniaria sp.	1.89	6.14	0	-	0	-	0.99	4.45
Spisula subtruncata	1.14	1.3	0	-	0.21	-	0.68	1.07
Corystes cassivelaunus	0.38	0.49	0.99	-	0.34	-	0.58	0.73
Euspira nitida	0.68	0.87	0	-	0	-	0.39	0.71
Pagurus bernhardus	0.36	0.55	0	-	0	-	0.19	0.43
Spisula elliptica	0.3	0.52	0	-	0	-	0.19	0.39
Tellina fabula	0	0	0	-	0	-	0.15	0.69

	Edge									
	t_{recent}		t_1		t_2		t_3		t_{no}	
	Mean	sd	Mean	sd	Mean	sd	Mean	sd	Mean	sd
Biomass	4.5	2.8	7.7	7.0	7.5	7.1	8.0	8.4	8.4	10.4
Species richness	3.2	1.3	5.2	3.0	8.2	1.3	9.0	0.0	7.5	4.3
	Abundance (n m^{-2})									
Ophiura ophiura	1.7	1.6	1.7	1.6	1.6	1.6	1.7	1.6	1.5	1.4
Tellina fabula	0.6	2.4	0.6	2.4	0.6	2.2	0.6	2.4	0.8	2.6
Nassarius reticulatus	0.3	0.4	0.3	0.4	0.3	0.4	0.3	0.4	0.5	0.9
Spisula subtruncata	0.5	1.4	0.5	1.4	0.4	1.3	0.5	1.4	0.4	1.1
Abra alba	0.4	1.3	0.4	1.3	0.4	1.2	0.4	1.3	0.3	1
Nassarius nitidus	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.2
Pagurus bernhardus	0	0	0	0	0.1	0.1	0	0	0.1	0.2
Ophiura albida	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.2
Spisula elliptica	0	0.1	0	0.1	0	0.1	0	0.1	0.1	0.1
Corystes cassivelaunus	0	0	0	0	0	0	0	0	0	0.1

	Biomass (g WW m^{-2})									
Ophiura ophiura	3.3	3.6	3.3	3.6	3.2	3.4	3.3	3.6	2.8	3
Nassarius reticulatus	0.8	1.3	0.8	1.3	0.9	1.2	0.8	1.3	1.4	2.7
Spisula subtruncata	1	2.9	1	2.9	0.9	2.7	1	2.9	0.7	2.3
Ensis sp.	0	0	0	0	0	0	0	0	0.6	1.9
Corystes cassivelaunus	0.1	0.3	0.1	0.3	0.2	0.3	0.1	0.3	0.4	0.7
Actiniaria sp.	0.5	1.5	0.5	1.5	0.5	1.4	0.5	1.5	0.3	1.2
Tellina fabula	0.2	0.9	0.2	0.9	0.2	0.8	0.2	0.9	0.3	0.9
Euspira nitida	0.3	0.6	0.3	0.6	0.3	0.6	0.3	0.6	0.2	0.5
Abra alba	0.3	0.9	0.3	0.9	0.3	0.8	0.3	0.9	0.2	0.7
Asterias rubens	0.2	0.5	0.2	0.5	0.1	0.4	0.2	0.5	0.1	0.4

	Crest parallel sandbar					
	$t_{0.5}$		$t_{1.5}$		$t_{2.5}$	
	Mean	sd	Mean	sd	Mean	sd
Biomass	2.4	0.9	3.7	1.5	7.6	9.7
Species richness	2.0	1.4	9.0	1.4	10.0	12.0
Abra alba	4.5	6.4	0.2	0.3	5.4	3.8
Ophiura albida	0	0	0.8	0.7	1.8	1.9
Euspira nitida	0.1	0	0.1	0	1.2	1.4
Ophiura ophiura	0	0	1.7	1.6	0.9	0.3
Nassarius nitidus	0	0	0	0	0.9	0.9
Tellina fabula	0	0	0	0	0.5	0.4
Spisula elliptica	0	0	0	0	0.4	0.6
Spisula subtruncata	0	0	0.5	0.6	0.4	0.5
Pagurus bernhardus	0	0	0.1	0.1	0.1	0.1
Actiniaria sp.	0	0	0	0	0.1	0
	Biomass (g WW m^{-2})					
Abra alba	0.6	0.9	0.1	0.1	3.9	2.8
Ophiura ophiura	0	0	1.6	1.2	1.2	0.7
Nassarius nitidus	0	0	0	0	1.2	1.4
Euspira nitida	1.7	0.1	0.7	0.2	0.6	0.7
Ophiura albida	0	0	0.2	0.1	0.5	0.6
Spisula elliptica	0	0	0	0	0.4	0.6
Spisula subtruncata	0	0	0.3	0.4	0.4	0.4
Asterias rubens	0	0	0	0	0.1	0.2
Tellina fabula	0	0	0	0	0.1	0.1
Nassarius reticulatus	0	0	0.1	0	0.1	0.2
	Crest (oblique sand bar)					
	$t_{0.5}$		$t_{1.5}$		$t_{2.5}$	
	Mean	sd	Mean	sd	Mean	sd
Biomass			1.2	2.9	17.4	2.2
Species richness			8.0	6.0	12.5	2.1
Abra alba	-	-	0	0	10	4.4
Tellina fabula	-	-	0	0	8.5	3.3
Ophiura ophiura	-	-	0.2	0.1	1.6	0.1
Spisula elliptica	-	-	0	0	0.8	0.9
Nassarius nitidus	-	-	0	0	0.6	0.5
Ensis sp.	-	-	0	0	0.4	0.5

Spisula subtruncata	-	-	0	0	0.2	0.1
Pagurus bernhardus	-	-	0	0	0.2	0
Ophiura albida	-	-	0.2	0.2	0.1	0.1
Euspira nitida	-	-	0	0.1	0.1	0.2
	Biomass (g WW m^{-2})					
Abra alba	-	-	0	0	7.8	1.7
Tellina fabula	-	-	0	0	4.4	1.6
Ophiura ophiura	-	-	0.2	0	2.2	0.3
Nassarius nitidus	-	-	0	0	0.8	0.6
Spisula elliptica	-	-	0	0	0.7	0.8
Liocarcinus holsatus	-	-	0	0	0.4	0.1
Ensis sp.	-	-	0	0	0.3	0.5
Spisula subtruncata	-	-	0	0	0.3	0.2
Actiniaria sp.	-	-	0.6	0.9	0.2	0.1
Donax vittatus	-	-	0	0	0.1	0.1

	Trough (parallel sand bar)					
	t_{recent}		t_1		t_2	
	Mean	sd	Mean	sd	Mean	sd
Biomass	1.6	2.1	3.8	1.4	175.4	284.8
Species richness	3.7	1.5	7.5	1.3	8.3	2.8
	Abundance (n m^{-2})					
Ophiura ophiura	0.2	0.2	1.6	1.1	87.7	172.8
Abra alba	0	0	0.2	0.2	41	30.9
Ophiura albida	0	0	0.5	0.2	37.6	74.7
Euspira nitida	0.1	0.1	0	0	23.9	47.5
Pagurus bernhardus	0.1	0.1	0	0	1.7	3.4
Nassarius nitidus	0	0	0	0	0.4	0.5
Nassarius reticulatus	0	0	0	0	0.3	0.3
Tellina fabula	0	0	0	0	0.2	0.4
Spisula elliptica	0	0	0	0	0.2	0.5
Actiniaria sp.	0	0	0	0	0.1	0.2
Ophiura ophiura	0.2	0.3	2.1	1.4	121.8	241.2
	Biomass (g WW m^{-2})					
Abra alba	0	0	0.1	0.1	33	23.4
Ophiura albida	0	0	0.1	0	11.6	23.1
Euspira nitida	0.5	0.5	0.2	0.4	6.7	13.2
Nassarius reticulatus	0	0.1	0	0.1	0.9	0.8
Nassarius nitidus	0	0	0	0	0.6	0.9
Asterias rubens	0	0	0	0	0.4	0.8
Spisula elliptica	0	0	0	0	0.2	0.3
Tellina fabula	0	0	0	0	0.1	0.2
Actiniaria sp.	0.7	1.2	0.9	1.1	0.1	0.2
	Trough (oblique sand bar)					
	t_{recent}		t_1		t_2	
	Mean	sd	Mean	sd	Mean	sd
Biomass	-	-	2.3	1.2	31.0	23.2
Species richness	-	-	4.0	1.2	9.8	2.2
	Abundance (n m^{-2})					

Abra alba	-	-	0	0	32.8	31.1
Ophiura ophiura	-	-	0.3	0.1	2.2	1.4
Tellina fabula	-	-	0	0	1.3	2.2
Spisula elliptica	-	-	0	0	0.6	0.4
Nassarius nitidus	-	-	0	0	0.5	0.3
Ophiura albida	-	-	0	0	0.2	0.2
Euspira nitida	-	-	0	0	0.2	0.1
Asterias rubens	-	-	0	0	0.1	0.1
Liocarcinus holsatus	-	-	0	0	0.1	0
Actiniaria sp.	-	-	0.1	0.1	0.1	0.1
	Biomass (g WW m^{-2})					
Abra alba	-	-	0	0	23.8	22.6
Ophiura ophiura	-	-	0.5	0.2	4.5	3.1
Tellina fabula	-	-	0	0	0.7	1.3
Nassarius nitidus	-	-	0	0	0.7	0.2
Spisula elliptica	-	-	0	0	0.5	0.3
Liocarcinus holsatus	-	-	0	0	0.4	0.4
Asterias rubens	-	-	0	0	0.2	0.2
Euspira nitida	-	-	0.4	0.5	0.1	0.1
Ophiura albida	-	-	0	0	0.1	0.1
Actiniaria sp.	-	-	0.6	0.7	0	0

Deep SE										
	t_{recent}		t_1		t_2					
	Mean	sd	Mean	sd	Mean	sd				
Biomass	-	-	7.8	-	32.8	10.5				
Species richness	-	-	8.0	-	7.7	2.1				
	Abundance (n m^{-2})									
Abra alba	-	-	0.3	-	41.4	12.8				
Tellina fabula	-	-	0	-	2.2	1.1				
Pagurus bernhardus	-	-	0	-	2.1	3.7				
Ophiura ophiura	-	-	5.2	-	2.1	2.1				
Nassarius nitidus	-	-	0	-	0.9	0.6				
Actiniaria sp.	-	-	0.1	-	0.4	0.4				
Euspira nitida	-	-	0.2	-	0.3	0.3				
Ophiura albida	-	-	0.5	-	0.2	0.3				
Nassarius reticulatus	-	-	0	-	0.1	0.1				
Spisula subtruncata	-	-	0	-	0.1	0.1				
	Biomass (g WW m^{-2})									
Abra alba	-	-	0.1	-	26.6	12.7				
Ophiura ophiura	-	-	4.8	-	3	3.1				
Tellina fabula	-	-	0	-	1.1	0.6				
Nassarius nitidus	-	-	0	-	1	1.1				
Actiniaria sp.	-	-	1.4	-	0.5	0.4				
Asterias rubens	-	-	0	-	0.3	0.5				
Nassarius reticulatus	-	-	0	-	0.1	0.2				
Euspira nitida	-	-	1.3	-	0.1	0.1				
Ophiura albida	-	-	0.1	-	0.1	0.1				
Spisula subtruncata	-	-	0	-	0	0.1				
Deep NW										
	t_{recent}		t_1		t_2					
	Mean	sd	Mean	sd	Mean	sd				
Biomass	1.1	1.3	13.8	9.0	0.4	-				
Species richness	3.8	3.0	7.0	0.0	1.0	-				
	Abundance (n m^{-2})									
Abra alba	0.3	0.4	2.4	1.7	-	-				
Abra prismatica	0	0	0	0	-	-				
Actiniaria sp.	0	0	0	0	-	-				
Amphipholis squamata	0	0	0	0	-	-				
Asterias rubens	0	0.1	0	0	-	-				
Corystes cassivelaunus	0	0	0	0	-	-				
Diogenes pugilator	0	0	0	0	-	-				
Donax vittatus	0	0	0	0	-	-				
Ensis sp.	0	0	0	0	-	-				
Liocarcinus navigator	0.1	0.1	0	0	-	-				
	Biomass (g WW m^{-2})									
Abra alba	0.2	0.3	1.4	1.2	-	-				
Abra prismatica	0	0	0	0	-	-				
Actiniaria sp.	0	0	2	1.9	-	-				
Amphipholis squamata	0	0	0	0	-	-				

Asterias rubens	0	0.1	0	0	-	-				
Corystes cassivelaunus	0.1	0.2	0	0	-	-				
Diogenes pugilator	0	0	0	0	-	-				
Donax vittatus	0	0	0	0	-	-				
Ensis sp.	0	0	0	0	-	-				
Liocarcinus navigator	0	0	0	0	-	-				

Shipping lane area						
	2006		2008		2012	
	Mean	sd	Mean	sd	Mean	sd
Biomass	102.7	74.1	175.6	146.4	18.8	7.0
Species richness	13	1.4	31.3	13.0	14.3	1.5
	Abundance (n m^{-2})					
Abra alba	24.5	18.5	191.5	143.1	16.7	7.8
Lutraria lutraria	0.3	0.4	0.7	0.5	0.8	0.1
Mya truncata	0	0	1.5	2.6	0.4	0.1
Actiniaria sp.	41.4	36.5	136.1	102.6	0.4	0.2
Ensis sp.	2.3	2.5	3.4	3.1	0.3	0.2
Euspira nitida	5.9	3.9	0.0	0.0	0.3	0.2
Spisula subtruncata	0.2	0.2	0.1	0.3	0.3	0.2
Nassarius reticulatus	5.4	9.5	0.7	1.3	0.2	0.1
Ophiura ophiura	2.0	0.9	0.5	0.3	0.2	0
Tellina fabula	0.0	0.0	0.0	0.0	0.1	0.3
	Biomass (g WW m^{-2})					
Abra alba	-	-	-	-	9.5	4.1
Ensis sp.	-	-	-	-	6.2	4.7
Asterias rubens	-	-	-	-	0.9	0.9
Nassarius reticulatus	-	-	-	-	0.6	0.4
Actiniaria sp.	-	-	-	-	0.5	0.3
Spisula subtruncata	-	-	-	-	0.3	0.2
Ophiura ophiura	-	-	-	-	0.3	0
Liocarcinus holsatus	-	-	-	-	0.2	0.2
Euspira nitida	-	-	-	-	0.1	0.1
Liocarcinus navigator	-	-	-	-	0.1	0.1

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- Mixed Linear Models (2010)
- Geostatistics (2010)
- SENSE Writing Week (2011)
- Environmental Research in Context (2011)
- Research in Context Activity: ‘Contributing to the review of IPCC chapter under coordination of the Netherlands Environmental Assessment Agency (PBL) (2013)

Other PhD and Advanced MSc Courses

- Summer School: Estuarine and Coastal Processes in relation to Coastal Zone Management, Netherlands Centre for Coastal Research (NCK) (2011)
- Vloeistofdynamica, Utrecht University (2011)

Selection of Oral Presentations

- *Modelling the ecological potential of sand extraction in the southern North Sea*. ICES Annual Science Conference - Working Group on the effects of extraction of marine sediments on the marine ecosystem (WGEXT), 31 May- 4 June 2010, Djurönäset, Sweden
- *Short-term impact of deep sand extraction and ecosystem-based landscaping on Macrozoobenthos and sediment characteristics*. ICES Annual Science Conference - Working Group on the effects of extraction of marine sediments on the marine ecosystem (WGEXT), 2-5 June 2014, Reykjavik, Iceland
- *Ecosystem based design rules for future sand extraction sites.* 5th EMSAGG conference 'Finding common ground - marine sand and gravel', 4-5 June 2015, Delft, The Netherlands
- *The future of sand extraction in the Dutch part of the North Sea.* VLIZ Young Marine Scientists' Day 2015, 20 February 2015, Brugge, Belgium

SENSE Coordinator PhD Education

Dr. ing. Monique Gulickx

Colophon

Cover design and layout
Dirk-Jan Saaltink and Maarten de Jong

Photography
Maarten de Jong: background front cover and back cover: bathymetry of the Dutch continental shelf, based on surveys done by the Royal Netherlands Navy.

Daan Rijks: small picture front cover, trailer suction hopper dredgers (TSHD's) realising the ecosystem-based parallel sandbar in 2010.

Ulrike Braeckman: small picture front cover, white furrow shell *Abra alba.*

Maarten de Jong: small picture front cover, fishing vessel 'Jan-Maria' GO–29 during a fish survey.

Daan Rijks: cover Chapter 1: TSHDs realising the ecosystem-based parallel sandbar in 2010.

Maarten de Jong: cover Chapter 2, detail of distribution of infaunal species richness.

Ulrike Braeckman: cover Chapter 3, white furrow shell *Abra alba*.

Maarten de Jong: cover Chapter 4, fishing vessel 'Jan-Maria' GO–29 during a fish survey.

Maarten de Jong: cover Chapter 5, bathymetry of the Dutch continental shelf, based on surveys done by the Royal Netherlands Navy and borrow pits (Rijkswaterstaat).

Daan Rijks, Maarten de Jong, Maarten de Jong, Ulrike Braeckman, Maarten de Jong, Hans Hillewart (plaice, *Pleuronectus platessa*) cover Chapter 7.

Maarten de Jong: cover Chapter 8, breathe out during a dry-suit dive in the Mid-Atlantic ridge at Silfra National Park Thingvellir on Iceland.

Maarten de Jong: cover Chapter 9, fishing vessel 'Jan-Maria' GO–29 during a fish survey.

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