Review of the geomorphological, benthic ecological and biogeomorphological effects of nourishments on the shoreface and surf zone of the Dutch coast

M.J. Baptist (Ed.), J.E. Tamis, B.W. Borsje, J.J. van der Werf

Report IMARES C113/08, Deltares Z4582.50

Wageningen IMARES & Deltares

Client:
RWS Waterdienst
Zuiderwagenplein 2
8224 AD Lelystad

Publication Date: 23 January 2009

Wageningen IMARES is a cooperative research organisation formed by Wageningen UR en TNO. We are registered in the Dutch trade record
Amsterdam nr. 34135929,
BTW nr. NL 811383696B04.

A_4_3_2-V5.2

Summary

In The Netherlands, the shore is protected by supplying sand on the beaches and onto the shoreface. To enable predictions of future states of beach ecosystems, and to contribute to the development of effective and sustainable nourishment practices, knowledge on ecological and morphological processes is essential. Furthermore, because these two aspects of the environment are mutually dependent, a coupled understanding of how these influence each other, and how the processes feedback into each other will provide valuable information. A literature review has been conducted, with focus on the macrozoobenthos in and on the sediment of the shoreface and surf zone. Results are described in this report, including an integration of the two disciplines geomorphology and ecology.

The current (2000-2007) policy is to nourish yearly 12 Mm^3 sand in the Dutch coastal system, of which about 60% by means of relatively large (10^6 m^3) shoreface nourishments and 40% by means of relatively small (10^5 m^3) beach nourishments. The largest part (49%) of the 12 Mm^3 of nourished sand is put into the central Dutch coast (between Hoek van Holland and Den Helder); the Wadden and the southwestern coastal system receive 28% and 23%, respectively. The shoreface nourishments typically have a volume of 1-3 Mm^3 (400-600 m^3/m) and are usually placed against the outer breaker bar, at a water depth of 4-8 m.

Geomorphology
Most knowledge on the morphodynamic behaviour of shoreface and beach nourishments originates from data-analysis studies. Numerical modeling tools have been used successfully in hindcasting behaviour of nourishments, but do not yet have the predictive power for reliably forecasting.

Shoreface nourishment affects the autonomous behaviour of the breaker bars. The nourishment, placed against the outer breaker, generally re-shapes itself relatively quickly (within a few months) into a bar with a landward trough. As a result of this, the offshore migration of the original breaker bars is halted. The duration of the impact of a shoreface nourishment is finite and related to its lifetime, which typically lies between the 2 and 8 years. It seems as if the grain size at the nourishment location adjusts itself quickly (within a year). In general, most nourished sand is transported onshore in the end.

Shoreface nourishments display different morphodynamic behaviour at different locations in the Dutch coastal system, i.e.: strong alongshore migration for the Terschelling (1993) shoreface nourishment (probably due to differences in orientation compared to the dominant offshore wave conditions); relatively persistent presence in the southwestern Dutch coastal system (lacking breaker bars); consistent erosion at Egmond in the central Holland coast (possibly due to divergence of alongshore currents at this location).

Ecology
Since the 1980s, several monitoring studies have been conducted on macrobenthos and effects of sand nourishments in the Dutch coastal zone. Some of these involved site-specific projects and some are part of yearly monitoring programs.

The macrozoobenthos is a well investigated group and consitst of molluscs (bivalves and snails); worms; spiny-skinned animals (Echinodermata); and crustaceans (shrimp-like animals). Different ecological zones can be distinguished in the Dutch sandy coast, related to the distance to shore and depth. Also, the number of species in the coastal zone can be related to the grain size and slope.

Ecological effects of nourishment are usually short-term: a reduction of abundance and biomass of species. In general, recovery proceeds rather fast. For most of the species, abundance and biomass will largely recover ca. 1 year after completion of the nourishment. Full recovery of the benthic community and age structure is considered to take 2-5 years.

Biogeomorphology
Some benthic species, such as *Lanice conchilega, Echinocardium cordatum* and *Tellina fabula*, may influence the sediment properties (i.e. they are considered to be bio-engineers). High density aggregations of *L. conchilega*, which is a characteristic species of the ecological zone between the two breaker bars, seem to increase sediment consolidation. *L.conchilega* and *Echinocardium cordatum* are able to stabilize a nourishment, by decelerating the near bed flow and increasing the grain size at the sediment water interface respectively. *Tellina fabula* will destabilize a nourishment by bioturbation activity (mixing the sediment layer).

Future considerations
The influence of grain size on the ecological effects of nourishments is important to consider in future nourishment projects. Another important aspect to consider in predicting the ecological effect of future sand nourishment is the spatial (i.e. ecological zones) and temporal (i.e. reproduction cycle) distribution of nourished sand. Furthermore, bio-engineers could significantly influence the stability of a nourishment. Therefore, to predict the stability of a nourishment, the site-specific species composition should be known, including the change in species composition after a change in physical parameters. Field experiments should be conducted to get real insight in the bio-geomorphological interactions for the stability of nourishments. During these field experiments it is recommended to monitor the recovery of benthos, the change in physical parameters (and corresponding habitat) and the behavior of the nourishment itself. Ecological relevant abiotic parameters within nourishment projects are grain size, layer thickness, oxygen level, turbidity/SPM, depth and distance to shore.

Contents

1 Introduction

Background

In The Netherlands, the shore is protected by supplying sand on the beaches and onto the shoreface. Studies related to the protection of the Dutch coast by sand nourishment, e.g. within the research program 'Kustlijnzorg' (Coastal protection), are mainly focused on sediment transport and geomorphology. Knowledge on the coastal ecosystem and the long-term effects of nourishment is limited (Oranjewoud 2006). However, many geomorphic processes occur in parallel with ecosystem processes at similar spatial and temporal scales, thus making the two mutually dependent (Renschler et al. 2007). Furthermore, the ecological aspects of sand nourishment have drawn attention, e.g. by the implementation of Natura 2000. A coupled understanding of how these two aspects of the environment influence each other, and how the processes feedback into each other will enable predictions of future states of beach ecosystems, and contribute to the development of effective and sustainable nourishment practices. Therefore, an (applied) research study on ecology has been included within the program 'Kustlijnzorg', with this report as result.

This report includes an overview of Dutch monitoring studies and recent (inter)national studies on the effect of sand nourishment on macrobenthos. The review is part of the research program 'Kustlijnzorg', conducted by Deltares and Wageningen IMARES, and integrates two disciplines in sand nourishment, i.e. geomorphology and ecology. Based on that, potential mitigation measures for future sand nourishment will be discussed. Besides providing an overview of sandy beach ecology and the relationship with sand nourishment, results of this study will be used as input for another coastal protection research program: the megasuppletions project carried out in the framework of the 'Stuurboord programme'. Within this program the effects of large (mega) nourishment projects as opposed to smaller and more frequent nourishment projects are investigated, with the aim to develop a long term research program on coastal protection and nature.

Aim

For efficient and sustainable coastal management, understanding of the relationship between sand nourishment and ecology is essential. Therefore, a literature review was conducted with the aim to provide an overview of the relationship between sand nourishment and seabed ecology, with focus on the aspects of sediment mobility and biogeomorphology.

Scope

An exposed sandy shore consists of a coupled shoreface, surf zone, beach and dune system, which together constitute an active zone of sand transport (Brown & McLachlan 2002). The focus of this study is on the shoreface and surf zone with the high tide level as the landward border. The study has a main focus on the macrozoobenthos in and on the sediment. Effects of sand nourishments on other species in the coastal zone, such as birds or sea mammals, fall outside the scope of this study.

2 Nourishment

Author: J.J. van der Werf (Deltares, Marine and Coastal Systems)

2.1 Sand nourished in the Dutch coastal system

Figure 1 shows the volume of sand nourished in the Dutch coastal system from 1990 to 2007. The figure distinguishes different types of nourishments: shoreface nourishments, beach nourishments and other nourishments (e.g. a dune enhancements).

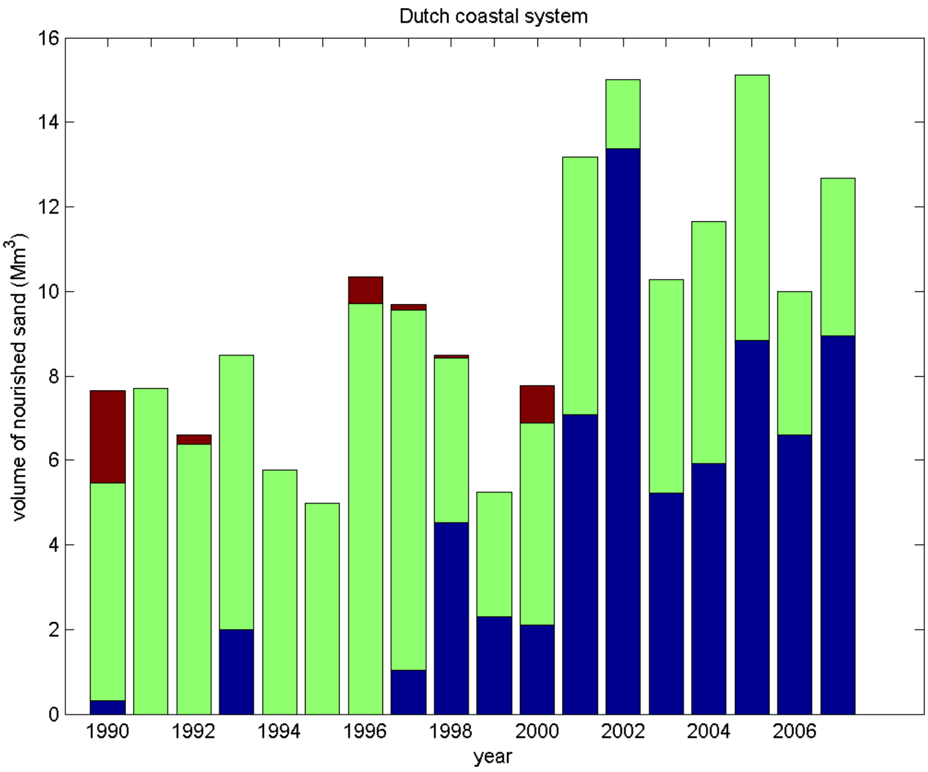

Figure 1 Volume of sand nourished in the Dutch coastal system from 1990 to 2007. Blues bars: shoreface nourishments, green bars: beach nourishments, brown bars: other nourishments. (Data courtesy of Waterdienst.)

On average 9.5 Mm^3 of sand was yearly nourished in the Dutch coastal between 1990 and 2007. The majority, 5.5 Mm^3, was placed on the beach; 3.8 Mm^3 sand was nourished on the shoreface, and 0.2 Mm^3 at another location. The yearly volume of nourished sand increases from the end of the 1990's as a result of an increase in the sand volume associated with shoreface nourishments. The average yearly nourishment volume is 12 Mm^3 for the period 2000-2007, of which 7.3 Mm^3 through shoreface nourishments, 4.6 Mm^3 through beach nourishments, and 0.1 Mm^3 in other ways.

The Dutch coast can be divided into three parts with their own characteristics:

1. The Southwestern Dutch coast from Zeeuws-Vlaanderen to Hoek van Holland. This coastal system is shaped by the estuaries of the rivers Scheldt, Rhine and Meuse and is subject to large anthropogenic influences such as the closure of most of the tidal inlets after the storm surge of 1953 and the construction of the Maasvlakte.

2. The 120 km long closed Holland coast from Hoek van Holland to Den Helder.
3. The open Wadden coast from Den Helder to the Dutch-German border. This coastal system is still affected by the construction of the Afsluitdijk (1932) and the closure of the Lauwerszee (1963).

Figure 2 presents an overview of the volume of nourished sand in each of these coastal systems.

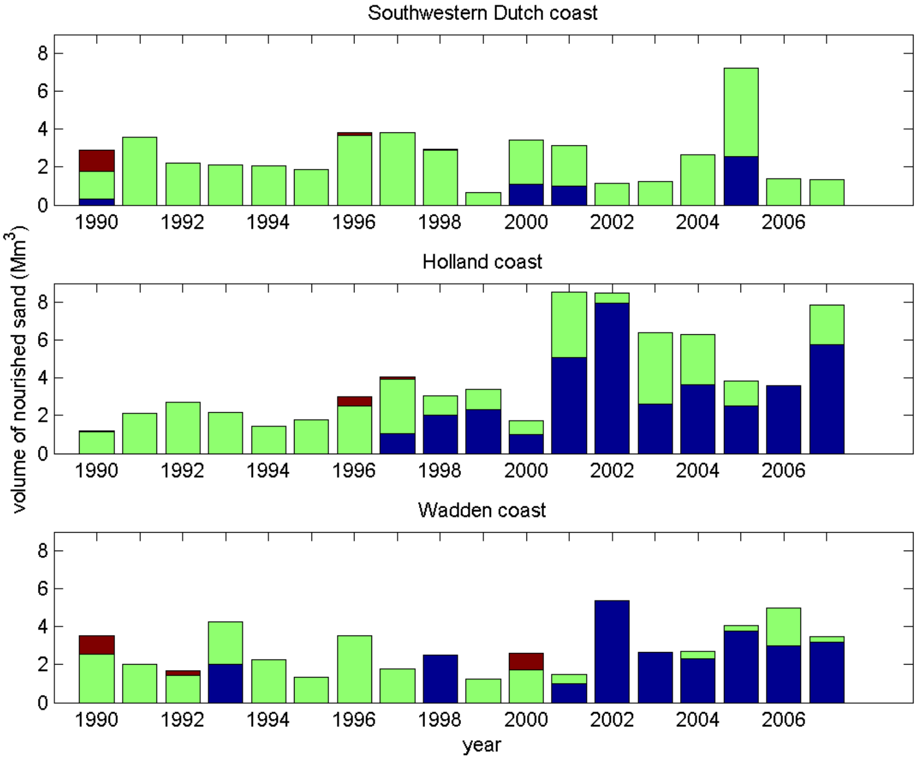

Figure 2 *Volume of sand nourished in the different coastal systems from 1990 to 2007. Blues bars: shoreface nourishments, green bars: beach nourishments, brown bars: other nourishments. (Data courtesy of Waterdienst.)*

This figure shows the following:

1. The yearly volume of nourished sand in the Southwestern Dutch coast is quite constant between 1990 and 2007. In the Holland coast there is a strong increase in the nourished volume starting in 2001, resulting from an increase in shoreface nourishments. The nourished volume increases slightly from 1990 and 2007 in the Wadden coast. After 2000, beach nourishments are almost fully replaced by shoreface nourishments.
2. Between 2000 and 2007 most sand was nourished on the Holland coast (on average 5.8 Mm^3 yearly), followed by the Wadden coast (3.4 Mm^3 yearly) and the Southwestern Dutch coast (2.7 Mm^3 yearly).
3. In the period 2000-2007 the yearly-averaged volumes of sand nourished on the shoreface are largest on the Holland coast (4.0 Mm^3 yearly). In the Wadden coast this is 2.7 Mm^3/year; in the Southwestern Dutch coast only 0.6 Mm^3/year.

Table 1 to Table 3 show the characteristics of all shoreface nourishments carried out between 1990 and 2007.

Table 1 Overview of shoreface nourishments carried out in the Southwestern Dutch coast between 1990-2007. (Data courtesy of Waterdienst.)

Year	Location	Begin (rkm)	End (rkm)	Volume (Mm^3)	Volume per transect (m^3/m)
1990	Cadzand-West (Zeeuws-Vlaanderen)	13.30	14.30	0.3	319
2000	Slufterdam (Maasvlakte)	6.41	10.54	1.1	266
2001	Slufterdam (Maasvlakte)	6.00	10.02	1.0	249
2005[1]	Oostgat (Walcheren)	24.70	26.85	2.6	1218

[1] Studied by Meetadviesdienst Zeeland (2008)

Table 2 Overview of shoreface nourishments carried out in the Holland coast between 1990-2007. (Data courtesy of Waterdienst.)

Year	Location	Begin (rkm)	End (rkm)	Volume (Mm^3)	Volume per transect (m^3/m)
1997	Ter Heijde (Delfland)	113.15	114.85	1.0	605
1998[1]	Noordwijk (Rijnland)	80.50	83.50	1.3	422
1998	Katwijk (Rijnland)	87.50	89.50	0.8	377
1999	Scheveningen (Delfland)	97.73	100.50	1.4	515
1999[2]	Egmond (Noord-Holland)	36.90	39.10	0.9	400
2000	Bergen aan Zee (Noord-Holland)	32.25	34.25	1.0	497
2001	Kijkduin-Ter Heijde (Delfland)	107.40	112.50	3.6	702
2001	Zijpe (Noord-Holland)	11.08	14.01	1.5	515
2002	Wassenaar (Rijnland)	91.00	97.00	3.0	500
2002	Noordwijkerhout (Rijnland)	73.00	80.00	3.0	429
2002	Camperduin (Noord-Holland)	26.50	30.00	2.0	564
2003	Callantsoog-Zwanenwater (Noord-Holland)	10.00	16.00	2.6	429
2003	Groote Keeten (Noord-Holland)	9.13	9.43	<0.1	41

Year	Location	Begin (rkm)	End (rkm)	Volume (Mm^3)	Volume per transect (m^3/m)
2004	Zandvoort-Zuid (Rijnland)	65.75	67.75	0.9	446
2004	Zandvoort-Noord (Rijnland)	62.75	65.75	1.1	371
2004[3]	Egmond aan Zee (Noord-Holland)	36.20	40.20	1.6	402
2005	Monster (Delfland)	108.00	113.00	1.0	200
2005	Bergen (Noord-Holland)	31.50	36.20	1.5	269
2006	Noordwijk-Katwijk (Rijnland)	81.50	89.00	1.1	150
2006	Wassenaar (Rijnland)	89.00	97.00	0.9	109
2006	Callantsoog-Zwanenwater (Noord-Holland)	10.00	15.20	1.6	308
2007	Monster-Hoek van Holland (Delfland)	113.00	118.00	0.8	150
2007	Kop Noord-Holland	0.00	7.10	5.0	704

[1]Studied by Ojeda et al. (2008). [2]Studied by Van Duin et al. (2004) and Cohen & Brière (2007). [3]Studied by Cohen & Brière (2007) and Walstra *et al.* (2008).

Table 3 Overview of shoreface nourishments carried out in the Wadden coast between 1990-2007. (Data courtesy of Waterdienst.)

Year	Location	Begin (rkm)	End (rkm)	Volume (Mm^3)	Volume per transect (m^3/m)
1993[1]	Terschelling-midden	13.7	18.1	2.0	455
1998	Ameland-midden	13.00	21.00	2.5	312
2001	Vlieland-oost	46.20	48.50	1.0	435
2002	De Koog (Texel)	17.00	23.00	5.4	899
2003	Ameland-midden	9.40	13.70	1.4	333
2003	Zuid-west/Den Hoorn (Texel)	9.00	11.48	1.2	489
2004	Eierland (Texel)	25.20	27.80	2.3	892
2005	Vlieland-oost	48.60	50.20	1.2	724
2005	Centrale kust Texel	13.52	16.90	2.6	666
2006	Ameland-midden	12.00	17.00	1.5	300
2006	De Koog (Texel)	17.00	23.00	1.5	250
2007	Ameland-noordwest	1.95	3.02	1.2	1121
2007	Texel-zuidwest	9.00	13.50	2.0	444

[1]Studied by Hoekstra *et al.* (1996) and Grunnet *et al.* (2004).

The following can be observed from these tables.

1. In the Southwestern Dutch coast the shoreface nourishment of 2.6 Mm^3 in 2005 in the tidal channel “Oostgat” is remarkable for its size and the way it has been carried out. The sand was placed against the slope of the tidal channel (“geulwandsuppletie”). Next to the 4 shoreface nourishments, almost 100, mostly small, beach and other types of nourishments have been carried out.
2. In the Holland coast 23 shoreface nourishments have been carried out between 1990 and 2007, as well as more than 80 mostly small beach and other nourishments. Large volumes have been nourished near the coastal towns Egmond, Bergen en Noordwijk, which are so-called erosional hotspots.
3. Ameland and Texel are nourished frequently. In total 13 shoreface nourishments have been carried out in de Wadden coast. Next to these, 37 mostly small beach and other nourishments have been carried out. Compared to the Holland and Southwestern Dutch coast, the volumes of nourished sand per transect are high.

2.2 Morphological effects of nourishment

Section 2.2.1 discusses the general morphodynamic response of the coastal system to a shoreface nourishment. In the sections thereafter specific shoreface and beach nourishments are discussed. This information originates from reports and articles by Kroon et al. (1994), Hoekstra et al. (1996), Van Duin et al. (2004), Grunnet et al. (2004), Cohen & Brière (2007), Meetadviesdienst Zeeland (2008), Ojeda et al. (2008) and Walstra et al. (2008). These are both modelling and data analysis studies.

2.2.1 General morphodynamic response to a shoreface nourishment

The shoreface nourishments carried out in the Dutch coastal system along the Holland and Wadden coast typically have a volume of 1-3 Mm^3 (400-600 m^3/m). They are usually placed against the outer breaker bar at a water depth of 4-8 m.

A shoreface nourishment has the following two effects on the coastal system:

1. *The lee effect*: The artificial sand bar increases wave dissipation, by which the wave height and the alongshore current onshore of the nourishment decrease. As a result, the alongshore sand transport capacity decreases here and therefore sediment accumulates upstream and erodes downstream of the nourishment. When the waves approach the shore perpendicularly, the leeside of the nourishment possibly erodes as a result of divergence of alongshore currents induced by alongshore differences in wave set-up.
2. *The feeder effect*. This refers to the feeding of coastal system onshore of the nourishment with nourished sediment due to cross-shore sand transport processes. The net sand transport in onshore direction is enhanced by the nourishments, because, i) seaward suspended load decreases because the additional wave dissipation by the nourishments reduces the offshore-directed undertow and the wave-induced sediment suspension ii) onshore bed- and suspended load increase at the nourishment due to additional wave skewness related to the lower water depth compared to the no-nourished case.

The shoreface nourishment affects the autonomous behaviour of the breaker bars. The autonomous behaviour of the breaker bars is periodically and consists of the following phases: 1) generation near the beach, 2) net migration in seaward direction through the surf zone, and 3) de-generation at the edge of the surf zone. The latter phase triggers the generation of a new breaker bar (phase 1) and the seaward migration of the now outer breaker bar (phase 2). This cycle is a cross-shore distribution of sand without a significant loss in offshore direction. The number of breaker bars (between 0 and 4) and the duration of this cycle (between 0 and 15 years) vary along the Dutch coast and are, among other things dependent on the steepness of the coastal profile.

The nourishment, placed against the outer breaker, generally re-shapes itself relatively quickly (within a few months) into a bar with a landward trough. As a result of this, the offshore migration of the original breaker bars is halted; sometimes they even temporarily migrate in the onshore direction. During this stop of offshore bar migration, the bars keep their pre-nourished dimensions.

The above-described effects of a shoreface nourishment are temporary, as the volume of a nourishment decreases in time. The duration of the impact of a nourishment is related to its lifetime, which lies between the 2 and 8 years for the shoreface nourishments studied in The Netherlands. The lifetime is, among other things, dependent on the nourishment volume, the grain size, alongshore nourishment length and the location of the nourishment.

2.2.2 Wadden coast: shoreface nourishment Terschelling (1993)

The shoreface nourishment at Terschelling has been carried out between May and November 1993. The surf zone is locally 1400 m wide with a mean slope of 1:180 and contains 2-3 breaker bars. The bar cycle return period is about 11 years (Walstra *et al.,* 2008). The median grain size, D_{50}, is 0.20-0.25 mm in the intertidal area and 0.16-0.18 mm at a water depth of 10 m; i.e. offshore of the nourishment (Kroon et al., 1994). The trough between the middle and the outer breaker bar (water depth between 5 and 7 m) was filled with 2.1 Mm^3 sediment over a length of 4.5 km (450 m^3/m), leading to a sediment layer with a thickness of approximately 2.3 m. The nourished sediment was somewhat coarser (0.18-0.21 mm) and less well-sorted than the original sediment.

Within the NOURTEC project (Hoekstra et al., 1996) this nourishment has been monitored extensively during a couple of years. The most important conclusions are:

- The disturbed bar morphology recovers quickly. After 150 days the original bar-trough system appears again. This rapid adjustment can largely be explained by the heavy storm conditions in the period following the implementation of the nourishment.
- The nourishment itself can hardly be detected after 2.5 years. However, as a result of the morphological adjustment the middle breaker bar grows in height and migrates temporarily in the onshore direction.
- After 2.5 years, 560.000 m^3 sand has disappeared from the nourishment area, of which about 20% in Eastern direction.. At the same time, a landward gain of 1.1 Mm^3 sand occurs, which thus cannot be fully explained by the cross-shore transport of nourished sand. The difference is due to longshore transport gradients, which have a natural cause and/or are related to the placement of the nourishment
- The grain size at the nourishment location adjusts itself quickly in the first half year. Finer sediment is transported in the offshore and Eastern direction; coarser sediment is transported onshore. This results in fining of the sediment in the nourished area and the original grain size seems to re-appear.

Grunnet et al. (2004) have tried to simulate this nourishment with Delft3D, which is a simulation model to study hydrodynamics, sediment transport and morphology e.g. in coastal systems. They conclude that it is potentially possible to simulate the morphodynamic behaviour of a nourishment with a 3D model, but not yet the detailed breaker bar behaviour. The simulations shows flattening of the breaker bars; introduction of a phase difference between sediment transport and bathymetry appear to be essential. Although the hydrodynamics are better simulated by a 3D model, the 3D model does not give a significant better prediction of the morphology. The model is sensitive to free model parameters in the transport modules and should therefore only be applied after a thorough morphodynamic calibration.

2.2.3 Southwestern delta coast: channel slope nourishment Oostgat (2005)

At the southwestern delta coast there is no bar-trough system. Nourishments are placed at the channel slope of tidal channels that tend to migrate to the shore. Just below the coast of southwest Walcheren the deep tidal channel Oostgat is located. Under the influence of the tidal current this channel migrates slowly in the direction of the Walcheren coast, which becomes smaller as a result. The upper part of the coast is kept in place by means of beach nourishments, but under water the upper shoreface erodes due to the onshore migration of the Oostgat. Due to this erosion, the coastal profile steepens. To counteract this steepening, 2.6 Mm^3 sand was nourished on the slope of the channel near Zoutelande between September and November 2005. This “channel slope nourishment” has been placed between rkm 24.7 and 26.85 (1218 m^3/m), from near the line of Low Water to the bottom of the tidal channel (located at a water depth of 22 m) with a slope of about 1:13.

Between March and May 2006 a beach nourishment of 1.4 Mm^3 was carried out between Vlissingen and Westkapelle (rkm 21.8 and 34.7; 108 m^3/m). In November 2006 the channel slope nourishment was extended on the northern side with sand that originated from the natural sills of the Galgeput and the Sardijngeul, which were dredged for navigational purposes. This extension is of the order of 10% of the original nourishment.

The geomorphological behaviour of this channel slope nourishment was extensively monitored, which is written down in the report of Meetadviesdienst Zeeland (2008). Between August 2005 and November 2007 8 bed level measurements were carried out. During the same period ADCP velocity measurements were carried out along three transects from the beach to the bottom of the tidal channel; at the location of the nourishment, north and south of the nourishment.

The median grain size of the sand from the sand mining pit Steenbanken is about 0.4 mm. The grain size of the nourishment has been determined directly after implementation along the above-described transects; the D_{50} varied between 0.32 and 0.47 mm.

In the rapport by Meetadviesdienst Zeeland (2008) the following observations are made.

- The nourishment has not eroded in December 2005 (T1). Close to the beach the nourishment flattened; close to the channel the nourishment steepened. The ebb tidal volumes of the Oostgat channel decreased with 7% to 13%, the flood tidal volumes in the southern and middle transects increased with 1% and 5%, respectively, while in the northern transect it decreased with 11%.
- The T2-situation (March 2006) hardly differed from T1.
- T3 (June 2006) showed a pattern of both erosion and sedimentation compared to T1. Erosion, especially between transects 25.000 and 25.750, is dominant. Furthermore, a deepening of about 3 m (relative to T2) at three locations on the shoreface about 500 m north of the nourishment was observed.
- The January 2007 measurements showed that erosion is dominant on the upper side of the nourishment and sedimentation on the lower side, which could be the result of settling of the nourishment. The loss of sand from the total monitoring area (nourished area + about 1 km in alongshore direction + offshore over the full width of the Oostgat) compared to T1 is estimated to be 5-10%. At a number of locations north of the nourishment, the shoreface steepened. In the central transect (middle of the nourished area) the cross-sectional area decreases with 5% and the tidal volume with 5 to 10%. The cross-sectional area in the northern and southern transect (outside the nourished area) was very similar to T0. The tidal volume decreased with 5% and 13%, respectively. It seems that the small loss of sand from the nourishment is partially due to the fact that the tidal current seems to have moved from the shore.
- The T5 situation (May 2007) hardly differed from the prior measurements. The shoreface north of the nourishment deepens and steepens at a number of locations.
- The last measurement in November 2007 (T6) show that the sediment has settled compared to T1. T6 hardly differs from T4 and T5. However, the nourished area has eroded compared to T5 with 29.000 m^3. In the total monitoring area the erosion is 126.000 m^3. The loss of sand in the original nourished area (T6 compared to T1) is estimated to be 5 to 10%. It should be noticed that this number is affected by the beach nourishment carried out in 2006.

2.2.4 Holland coast: shoreface nourishment Noordwijk (1998)

Between February and March 1998 1.7 Mm^3 of sand was nourished at a depth of 5-8 m over a length of 3 km in front of the coast of Noordwijk. This corresponds to 570 m^3/m. The nourishment was constructed more like a hump of sand instead of well distributed and placed against the outer breaker bar. The nourished sand was coarser than the original sand, 0.40 and 0.25 mm, respectively. In the surf zone, with a slope of 1:150, 2 breaker bars were located; the duration of the bar cycle was about 4 years.

Ojeda *et al.* (2008) have studied the morphological behaviour of this specific nourishment on the basis of analysis of daily Argus video images from mid-September 1998 to mid-July 2004, supplemented with (half)yearly bed level measurements. They have also intercompared the morphodynamics response of the shoreface nourishments of Noordwijk (1998), Terschelling (1993) and Egmond (1999). They concluded the following:

- The Noordwijk nourishment delayed the natural development of the two breaker bars located onshore of the nourishment. First, the migration speed of the outer bar decreased, and thereafter the migration speed of the inner bar, which temporary even migrated in the onshore direction. After 6 years, 1.5 times the duration of the natural bar cycle, the breaker bars had not resumed their autonomous seaward migration. This relatively long period of influence compared to the Egmond and Terschelling nourishments was possibly caused by 1) the coarser grain size, 2) the relative large nourishment volume (m^3/m) compared to the size of the breaker bars, and 3) the location of the nourishments, at a distance of the active coastal zone.

- There was no indication for the growth of 3D patterns; the sinuosity of the breaker bars even decreased in time.
- The nourishment had no effect on the coastline development.
- Shoreface nourishments enhanced the possibility for “bar switching” by the creation of alongshore gradients in the location and depth of the outer breaker bar. Bar switching is a sequence of morphologies during which the bar becomes discontinuous and the flank section decays or becomes attached to an offshore-located bar, while the section of the bar landward of the nourishment becomes attached to a landward-located bar.
- Despite the differences, the nourishments at both Noordwijk, Terschelling and Egmond were absorbed by the natural bar system. Loss of sand offshore and/or alongshore appeared to be of minor importance compared to the onshore processes.

2.2.5 Holland coast: shoreface and beach nourishments at Egmond (1999-2004)

The cross-shore profile at Egmond contains 3 bars: 2 breaker bars in the surf zone and a swash bar. The duration of the bar cycle is about 15 years. The cross-shore slope of the shoreface is about 1:100 and the median grain size is 0.2 mm. Between 1999 and 2005 the following nourishments have been carried out near Egmond:

- between June and September 1999; shoreface nourishment of 0.8 Mm^3 between 36.9 and 39.1 rkm (400 m^3/m);
- between June and July 2000; beach nourishment of 0.2 Mm^3 between 38.0 and 38.8 rkm (259 m^3/m);
- between June and November 2004; shoreface nourishment of 1.6 Mm^3 between 36.1 and 40.2 rkm (402 m^3/m);
- between April and May 2005; beach nourishment of 0.5 Mm^3 between 37.0 and 39.25 rkm (222 m^3/m).

The two shoreface nourishments are both put against the outer seaward side of the breaker bars at a water depth of 6-8 m. According Ojeda *et al.* (2008), the sand of the 1999 shoreface nourishment has a D_{50} of 0.228 mm; the grain size characteristics from the other nourishments are not known.

The first two nourishments were studied by Van Duin *et al* (2004) over the period May 1999 – April 2002 by means of data analysis and simulation with model suites Unibest-TC and Delft3D. They used eight bed level measurements taken in this period. This study confirmed the general response to a shoreface nourishment as described in Section 2.2.1. Furthermore, they concluded the following:

- During the first 2 years the shoreface nourishment hardly changed and did not contributed directly to the beach volume. The nourishment was probably too large and resulted in a slow diffusion. Only after 2 years the nourishment started to diffuse.
- The original inner and outer bar showed a large onshore migration and a trough was created between the outer bar and the nourishment. The nourishment acted as a new outer breaker bar. The final measurements showed that this trend was ending; the system seemed to return to the original 3 bar system.
- The area onshore of the nourishment accreted due to the lee effect. After 3 years the sand volume in the area of interest (0.9x5 km) increased with 0.73 Mm^3, and after three years with 0.475 Mm^3. This corresponds to 45% of the shoreface and beach nourishment.
- The shoreface nourishment has a minimal direct impact on the beach during the first 2-3 years. Apparently, the nourishment needs to be maintained to make sure sand ends upon the beach; accompanying time scales are probably of the order of 5-10 years.

Cohen & Brière (2007) studied the impact of all four nourishments on the intertidal beach (position and volume of intertidal coast line) and the breaker bar behaviour at Egmond by analyzing bed level soundings (JARKUS, (half)yearly information) and Argus video images (monthly information). They distinguish between the period before dynamic preservation policy (1965-1990) and the period directly after the first shoreface nourishment (1999-2006).

They concluded the following:

1. Two years after the 1999 shoreface nourishment (and the 2000 beach nourishment) a positive effect on the intertidal beach volume and beach width could no longer be observed. Two years after the 2004

shoreface nourishment (combined with the 2005 beach nourishments) a clear impact was still observed, probably because in this case wave conditions were milder and the nourished sand volume larger. Sawtooth behaviour was not very obvious due to strong seasonal variations.

2. Shoreface nourishments had the strongest effect on the middle breaker bar; the migration direction changes from off- to onshore. For the first shoreface nourishment this effect could be observed for 1.5-2 years. Two years after the construction of the second nourishment this effect was still ongoing. The outer bar shows a similar behavior, but less pronounced. The inner bar did not seem to be influenced by the shoreface nourishments. Beach nourishments did not seem to have an effect on the bar behavior There was no clear relation between the intertidal beach and the bar behavior.
3. It seemed that the wave damping effect induced by (shoreface) nourishments resulted in steepening instead of flattening of the intertidal beach during storm conditions.
4. The second shoreface/beach nourishment combination (2004-2005) resulted is a more stable trend of the intertidal beach than the first one, which could be due to the stabilizing effect of a shoreface nourishment on a beach nourishment..
5. There is an erosional hotspot, a location with consistent erosion, south of Egmond, which remains present over the whole monitoring period despite the nourishments.

Walstra *et al*. (2008) adopted an integrated approach to study the 2004 Egmond shoreface nourishment to increase the insight into the effect of shoreface nourishments on the coast on a timescale of weeks (storms) to years (breaker bar cycle). The study consisted of 1) literature review, 2) data analysis of the 2004 nourishment, iii) application of Argus/Beach Wizard to investigate the morphodynamic response with a high resolution, iv) calibration and application of Delft3d on the 2004 nourishment and v) laboratory measurements in the Schelde flume

The data analysis used 7 measurements of the bathymetry which were conducted between June 2004 and May 2006. From this analysis, the following was concluded:

- The nourishment was quickly absorbed by the bar system; a trough was formed in the middle of the nourishment in the winter of 2004/2005. The original outer bar grew and a new outer bar was created offshore. During the summer of 2005 nothing much changed due to seasonal effects. During the autumn of 2005 the original bar migrated towards the coast and sand accumulated in the coastal zone. The bars were distorted temporarily. During the winter of 2005/2006 and the spring of 2006 the new outer bar flattened, while the original outer migrates more towards the coast. The nourishment, more flattened and less pronounced, could still be observed.
- Erosion northward of the nourishment (downstream) indicates the blocking of longshore transport by the nourishment (feeder effect). The sand accumulation can especially be observed on the beach during the first year after the nourishment was placed.
- 63% of the nourishment volume was found in a polygon around the nourishment one month after the placement. All of it is found in the larger area (on- en offshore of the nourishment and 1250 m in both alongshore direction). After about 1.5 years all of the sediment (including the beach nourishment volume) still seemed to remain in the area.
- The coastal zone benefited strongly from the shoreface nourishment. After subtraction of the beach nourishment, 0.5 Mm^3 sand (28% of the nourishment volume) accumulated onshore of the nourishment after about one year. As a result, the bed level increased on average with 0.5 m.
- Beach width and dune foot position strongly increased after the nourishment was put in place. It was difficult to assess the contribution of the shoreface nourishment to this, as at the same time a beach nourishment was carried out.

In the Schelde flume, three beach profiles, a reference case and two nourishment designs (a high placed and a low place nourishment corresponding to 400 m^3/m), were subject to average, accretive and severe, erosive wave conditions. The nourishment appeared to be capable of decreasing the erosion in the upper beach profile. The higher placed nourishment had more impact and the nourishment impact was stronger in case of higher waves. Wave measurements showed that the nourishment acted as a wave filter, where only in case of the higher placed nourishment wave damping was observed over the complete coastal profile. Furthermore, the measurements confirmed that the nourishment decreases the offshore, undertow transport and increase the onshore transport due to wave skewness.

The Beach-Wizard data-model assimilation technique aims at producing coastal bathymetries with a high temporal resolution (hours/days) by combining different techniques that operate on different time scale (different type of data and models). This system proved not to be fully capable of producing reliable bathymetries.

It appeared not be possible to simulate the Egmond 2004 shoreface nourishment morphodynamically longer than 3 months with Delft3D. In the simulations, large, physically unrealistic rip channels appeared that after a while affected the breaker bar behavior. It is promising that before the rip channels appeared, the morphodynamic response of the coastal system to a shoreface nourishment was well predicted.

3 Ecology

Authors: J.E Tamis & M.J. Baptist (Wageningen IMARES)

3.1 Monitoring studies

There have been several monitoring studies on macrobenthos and effects of sand nourishments in the Dutch coastal zone (Table 4). Monitoring programs usually determine the species, the biodiversity (the number of different species), the abundance (number of individuals per species per m^2) and the biomass (Ash Free Dry Weight (AFDW) per species per m^2). This data provides insight in, e.g. availability of food for species of a higher trophic level, e.g. birds, fish and biodiversity.

The Dutch National Institute for Coastal and Marine Management/RIKZ (Rijkswaterstaat) and the Institute for Inland Water Management and Waste Water Treatment (RIZA), commissioned by the North Sea Directorate (Rijkswaterstaat), organises a series of national monitoring programmes which together constitute MWTL (Monitoring van de Waterstaatkundige Toestand des Lands). Within this framework, a yearly monitoring programme of macrobenthos in the North Sea, Wadden Sea and the Delta is carried out. The monitoring programme of macrobenthos in the North Sea was generally referred to as BIOMON (from “MON*BIOLOGIE”) (Van Dalfsen et al. 2007).

The effect of sand nourishment on the ecology of the beach has been studied for the first time on Ameland and Texel (Dankers et al. 1983). The ecological effects of a shoreface nourishment were studied for the first time on Terschelling in 1993, within the European RIACON project (Risk Analysis of Coastal Nourishment Techniques). The RIACON project was started in 1994-1996 with the aim to investigate the ecological risk of shoreface nourishment and beach nourishment to the coastal benthic communities and organisms depending on the benthos (Essink 1997).

Under the motto “A Beach Is More than Just a Pile of Sand” a start was made in 2000 with the development of and making the knowledge available about the ecology of the sandy coast. In 2001 and 2002, the Dutch National Institute for Coastal and Marine Management studied the benthos of 9 locations along the Dutch coast (Janssen & Mulder 2004). Because of the physical conditions, the surf zone is relatively difficult to study and thus relatively unknown. Therefore, a special device was developed, which is able to take sediment samples in the surf zone to a depth of ca. 8 meter: the ‘WESP’ (Water En Strand Profiler). With this device, the surf zone was sampled in 2002 at Egmond and Castricum (Janssen & Mulder 2004).

As part of the project ‘Integral Water Management’ of the National Institute for Coastal and Marine Management (RIKZ), a limited field study was conducted in October 2005 in the surf zones of Schiermonnikoog (island coast) and Egmond (mainland coast) (Van Dalfsen 2006). The main objective of the study was to get insight in the ecology of the shallow sandy coast. The macrobenthos was described in relation to morphological differences and differences in the composition of the sediment.

(Van der Wal & Van Dalfsen 2008) studied the benthos in 2007 at four nourishment locations of the Dutch coastal zone (mainland), i.e. Petten, Bloemendaal, Zandvoort and Den Helder/Julianadorp.

As part of the shellfish monitoring program of the Ministry of Agriculture, Nature and Food Quality (LNV), shellfish stocks (e.g. *Ensis directus* (synonym *Ensis americanus*), *Spisula subtruncata* and *Cerastoderma edule*) are estimated yearly in the Dutch coastal waters by Wageningen IMARES (Goudswaard et al. 2008). Spring 2008 was the 14th survey executed in the consecutive survey since 1995.

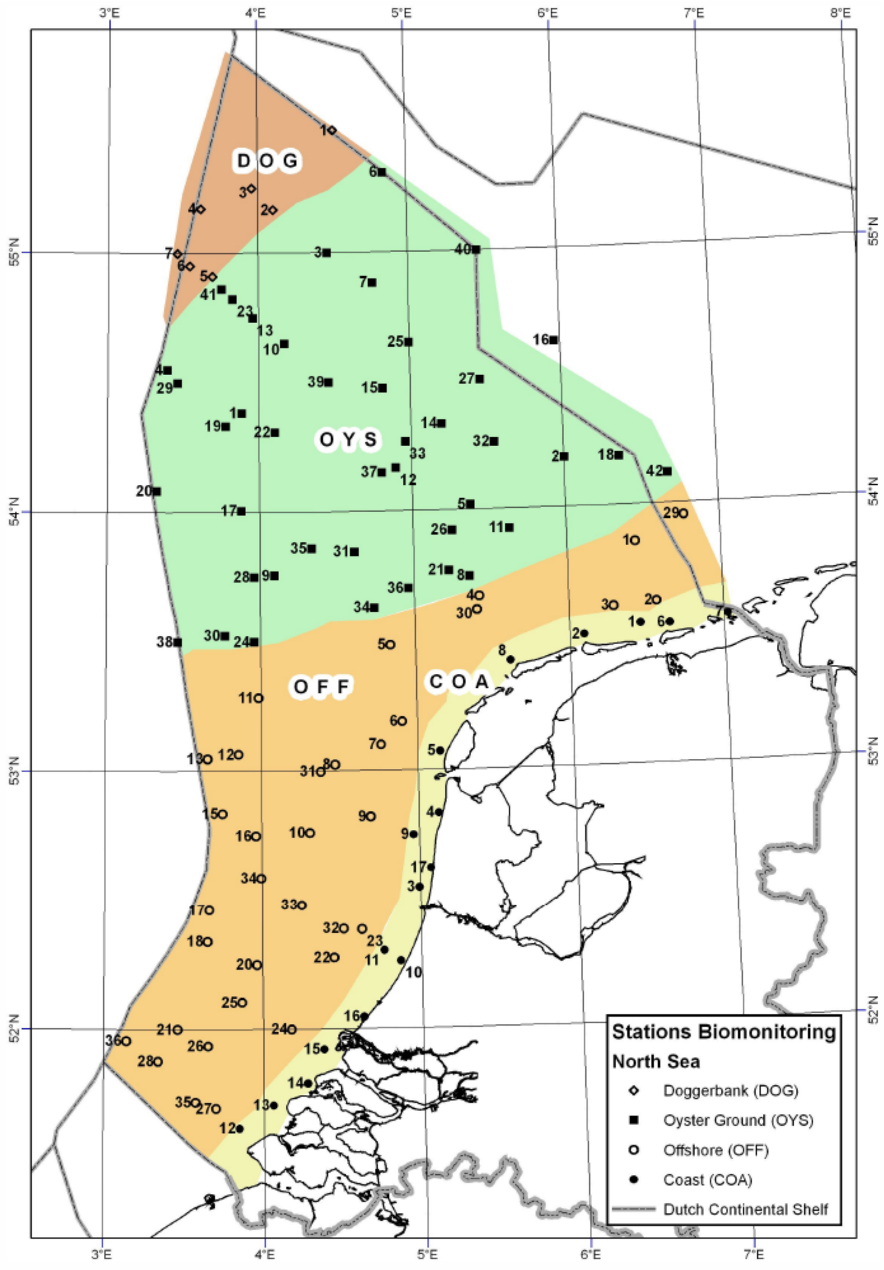

Figure 3 *Locations of the MWTL sampling stations (Van Dalfsen et al. 2007).*

Table 4 Ecological monitoring studies of the Dutch coast

Year of monitoring	**Topic**	**Location**	**Reference**
1980-1981	Ecological effect of beach nourishment	Ameland & Texel (Wadden coast)	(Dankers et al. 1983)
1986 – present	Yearly monitoring program of macrobenthos	North Sea, Wadden Sea and the Delta Estuary	(Van Dalfsen et al. 2007)
1993-1995	Ecological effect of shoreface nourishment	Terschelling (Wadden coast)	(Essink 1997, Van Dalfsen & Essink 1997)
1995 – present	Yearly shellfish monitoring program	Dutch coastal waters (Wadden - and Holland coast), including the Natura 2000 areas "Waddeneilanden/ Noordzeekustzone/Breebaart" and "Voordelta.".	(Goudswaard et al. 2008)
2001	Long-term effects on the benthos of a sand extraction site	Coastal zone near Heemskerk (Holland coast)	(Van Dalfsen & Lewis 2001)
2001-2002	Macrozoobenthos distribution in relation to meso-scale bedforms	Two locations at the Holland coast	(Baptist et al. 2006)
2002	Ecology of the Dutch sandy coast	Nine locations of the Dutch coastal zone (Wadden - and Holland coast): Schiermonnikoog West; and - east; Texel; Egmond; Castricum; IJmuiden North; and – South; Katwijk; and Goeree Overflakkee	(Janssen & Mulder 2004, Janssen & Mulder 2005)
2002-2003	Ecological effect of shoreface nourishment (T0, T1, T2)	North Sea coastal zone at the west coast of Texel (Wadden coast)	(Leopold 2002a) (Leopold 2002b) (Leopold 2003)
2004	Macrobenthos in- and outside harbour	In and near harbour IJmuiden (Holland coast)	(Kaag 2004)
2005	Sediment characteristics and macro fauna of surf zone	Schiermonnikoog (Wadden coast); and Egmond (Holland coast)	(Van Dalfsen 2006)
2007	Benthos at nourishment sites	Four locations of the Dutch coastal zone (Holland coast): Petten; Bloemendaal; Zandvoort; and Den Helder/Julianadorp	(Van der Wal & Van Dalfsen 2008)

To compare the monitoring data of the different surveys, two aspects are important. First, the difference in locations along the coastline where samples were taken is relevant because of local geographical differences between the southwestern Delta coast, the Holland coast and the Wadden coast. As shown in Table 4, most surveys are from the Wadden coast and Holland coast. The Southwest coast is only studied within the shellfish monitoring program (yearly ongoing survey from 1995 (Goudswaard et al. 2008)) and the monitoring survey conducted in 2002 at Goeree Overflakkee (Janssen & Mulder 2004). The shoreface at the Southwest coast has not been monitored yet.

Furthermore, the distance from the shoreline or depth where the samples were taken is a relevant aspect because of the different ecological zones. Table 5 provides an overview of the areas (or zones) of the Dutch coast that have been studied in the past.

The scope of this review is from the low water line to the outside of the last breaker bar, i.e. the shoreface and surf zone. Generally two breaker bars are present, but sometimes a third- or fourth breaker bar is formed. Compared to deeper water and the beach, the ecosystem of the shoreface and surf zone is relatively unknown.

The area further offshore has been studied at two locations, ca. 5 and 10 km from the Holland coast (Baptist et al. 2006). This area does not overlap with (potential) shoreface nourishment sites, as these are conducted much closer to shore (i.e. within one km from the coast). However, the data could be useful to provide an indication of the macrofauna distribution beyond the second breaker bar.

Table 5 Overview of areas (or zones) sampled within ecological monitoring studies of the Dutch coast. The height is relative to NAP (or mean tidal level).

Sample area	**Height (m)**	**Reference**
The beach (from above the high tide mark to slightly below the low tide mark)	+1 to -2	(Janssen & Mulder 2004)
The coastal zone	-2.5 to -19.5	(Van Dalfsen et al. 2007)
The surf zone	-2 to -7	(Janssen & Mulder 2004)
The shoreface	-6 to -9	(Van Dalfsen & Lewis 2001)
From the top of the first breaker bar to deep water	-1 to -13	(Van Dalfsen 2006)
At the seawards side (slope) of the second breaker bar and in deeper water	-5 to -13	(Van der Wal & Van Dalfsen 2008)
Ridge connected to the shoreface and the lower part of the shoreface	-15 to -18	(Baptist et al. 2006)

3.2 Benthic fauna of the Dutch coast

3.2.1 General

The macrozoobenthos is a well investigated group. Benthic organisms play a valuable role in the ecosystem function as they form the link between primary production and fish stocks. Benthos in the Dutch marine waters consist mainly of the following systematic groups (Essink 2005):

- Molluscs (bivalves and snails)
- Worms
- Spiny-skinned animals (Echinodermata)
- Crustacea

The benthos of the Dutch coastal zone is, since many decades, disturbed by fishing activities, e.g. on shrimps, shellfish (*Spisula* and *Ensis*) and bottom dwelling fish (with beam trawlers). Fishing has significant direct and indirect effects on habitat and on the diversity, structure and productivity of benthic communities (Jennings & Kaiser 1998). The benthos as now observed can therefore be considered as the product of several decades continuous disturbance by fishing activities.

In 2002, nine beaches along the Dutch coast were studied and a total of 49 species were found, from 5 to 28 per beach, with abundances ranging from 222 to 4166 individuals per m^2 (Janssen & Mulder 2005). The mean biomass on the Dutch beaches was determined at 2.3 g AFDW m^2.

The most common species on the beaches was found to be the bristle worm *Scolelepis squamata*, an important food source for the Sanderling *Calidris alba*, which is protected under the EU Birds Directive (Janssen & Mulder 2005). Two other characteristic species that were reported are *Bathyporeia pilosa* – the sand digger shrimp, and

Euridice pulchra, the speckled sea louse. Monitoring conducted twenty years before, found the same dominant species (Dankers et al. 1983).

In 2008, shellfish stocks in the Dutch coastal zone were estimated as part of the yearly monitoring program (Goudswaard et al. 2008). Of the *Ensis* spec., the non-native *E. directus* (syn. *E. americanus*) is most abundant on the Dutch coast. A total of more than 65 billion individuals for *Ensis directus* were estimated, with a total fresh weight of 892 million kilogram (based upon shell-size/weight relation). From this total stock, 30 billion individuals were found in the Birds and Habitats Directive Areas (Natura2000); 10 billion in the protection area “Noordzeekustzone” and 20 billion in the “Voordelta”.

A total stock of 5.5 million kilogram fresh weight was estimated for *Spisula subtruncata*. The total stock has slightly increased compared to 2007, but is still low compared to former years (Goudswaard et al. 2008). A stock of 0.7 million kilogram fresh weight was found in the “Bird and Habitat Directive Areas” (Natura2000). The number of juveniles (1-year) was slightly higher compared to 2007, which reflects a year of poor recruitment in 2007. Inside the Natura 2000 areas, a total stock of 2.6 million kilogram fresh weight was calculated for *Cerastoderma edule*. Cockles of 1 and more years were found at the mouth of the Haringvliet. One location had a density above 50 cockles/m^2.

An increasing stock of otter shells (*Lutraria lutraria*) was found along the entire Dutch coast. The number of animals is estimated at 1.7 billion specimen (Goudswaard et al. 2008). Estimating an average weight of 30 grams per shell (low estimate), the total mass is 51 million kilogram fresh weight, almost 10 times more than the total mass for *Spisula*.

3.2.2 Geographical differences

Three regions can be distinguished in the Dutch coastal zone (Figure 4):

- Southwest (from Zeeuws-Vlaanderen up to Hoek van Holland);
- Holland coast (from Hoek van Holland up to Den Helder);
- Wadden coast (from Den Helder up to the German border).

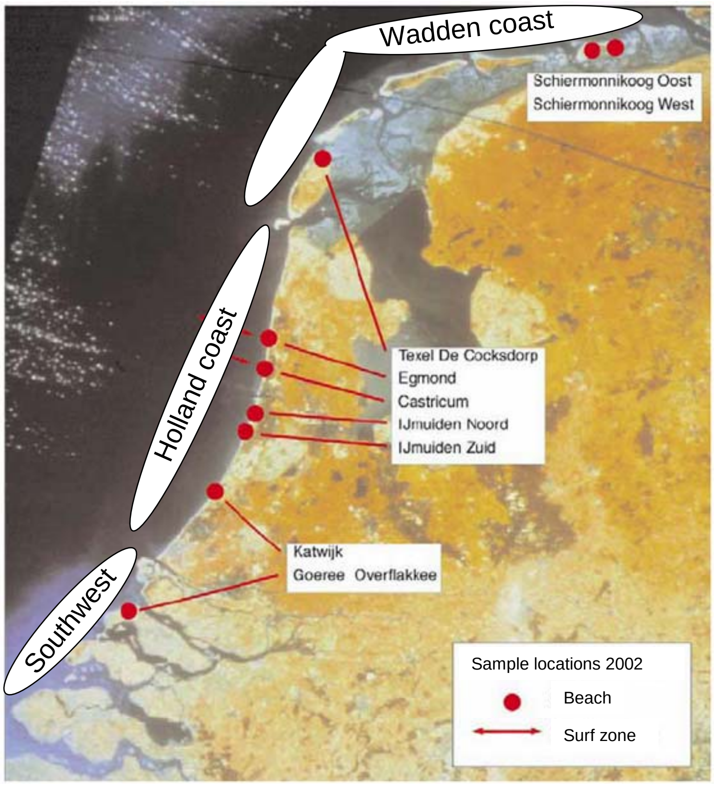

Figure 4 Map of the Dutch coast showing the regions as identified for the scope of this review and the sample locations of the monitoring study conducted in 2002 (based on (Janssen & Mulder 2004)).

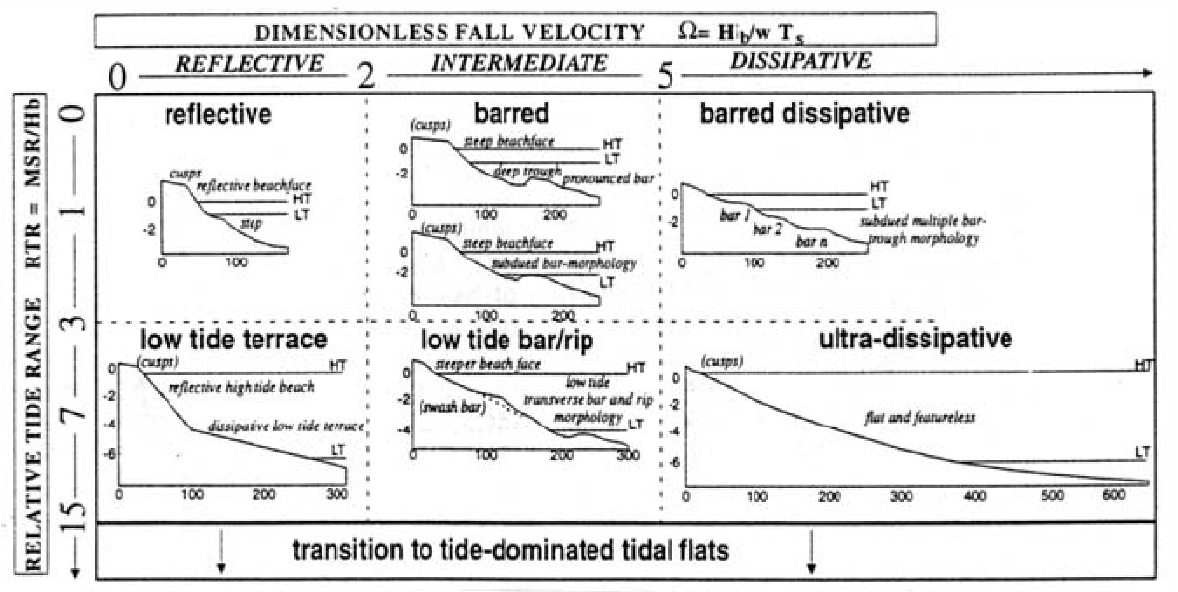

Figure 5 Classification of the morphological condition of beaches with a two-barred dissipative system, used by (Janssen & Mulder 2004) from (Short 1996).

The Dutch beaches are generally described as mesotidal, barred, dissipative, and moderately exposed (Janssen & Mulder 2005). There are clear differences between the beaches of the Dutch mainland coast and those of the Dutch Wadden Sea islands. Based on tidal range and fall velocity of sand, the Holland coast and Southwest coast are considered barred intermediate and the Wadden coast ultra-dissipative (Figure 5). The beach exposure, i.e. dynamics, is of great influence on the occurrence of species (Janssen & Mulder 2004). As tide range or wave energy increases or sand particle size decreases, beaches become wider, flatter and more dissipative. Faunal communities increase linearly in species richness and exponentially in abundance over this range of beach types (McLachlan 1996). The beaches of the Holland coast and Southwest coast are generally more exposed than the beaches of the Wadden coast (Table 6).

The number of species at the Wadden coast (Schiermonnikoog and Texel) is much higher compared to the beaches of the Holland coast and the Southwest coast (Figure 6). As shown in Figure 7, the species abundance (number of individuals per m^2) is highest on the beaches of Schiermonnikoog (Wadden coast) and IJmuiden North (Holland coast). The relatively high abundance at Schiermonnikoog West is mainly determined by the presence of the worm species *Pygospio elegans* (Janssen & Mulder 2004).

Table 6 Parameters determining beach dynamics and the resulting exposure score of several Dutch beaches, based on (Janssen & Mulder 2004).

Location	Wave action	Width surf zone (m)	% very fine sand	Grain size	Slope 1/x 2002	Stable burrow holes	Exposure score*
Southwest							
Goeree	<1	broad	3	218	96	no	11
Holland coast							
Egmond	<1	491	2.4	305	42	no	12
Castricum	<1	452	2.1	314	31	no	12
IJmuiden north	<1	861	3.7	257	77	no	12
IJmuiden south	<1	787	1.9	328	32	no	11
Katwijk	<1	broad		359	36	no	12
Average	*<1*	*648*	*2.5*	*313*	*44*		*12*
Wadden coast							
Schier east	<1	1100	3.6	202	96	no	11
Schier west	nil	1030	3.6	202	89	yes	8
Cocksdorp	<1	22	2.8	261	21	no	11
Average	*<1*	*717*	*3.3*	*222*	*69*		*10*

* A score between 1 and 5 is 'not exposed', between 6 and 10 is 'moderately exposed', between 11 and 15 is 'exposed' and between 16 and 20 is 'highly exposed' (Janssen & Mulder 2004), based on the classification of (Brown & McLachlan 1990).

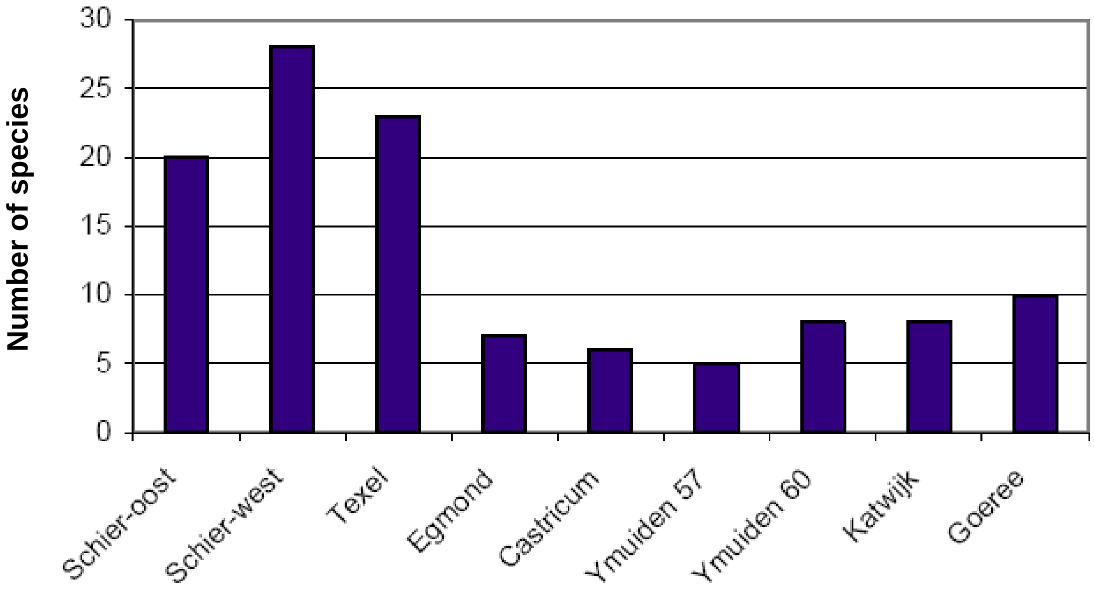

Figure 6 Number of species per beach in 2002 (Janssen & Mulder 2004).

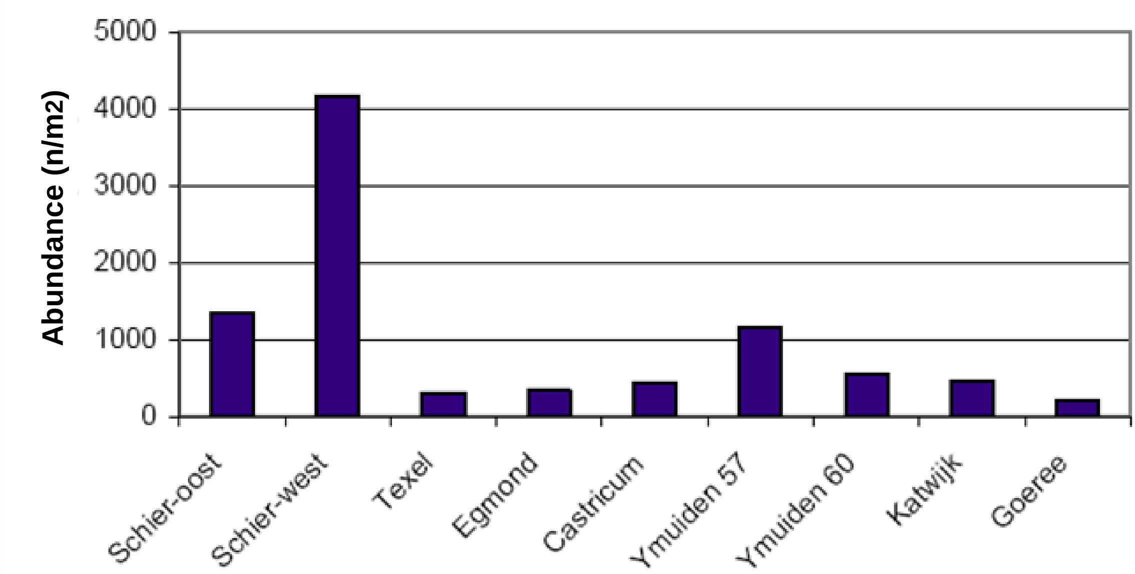

Figure 7 Abundance per beach in 2002 (Janssen & Mulder 2004).

Based on data from 2002, the beaches of the Wadden Sea islands are generally characterized by relatively fine sand with low carbonate content and large variation in grain size and a high number of species and densities, compared to the mainland coast. The sediment can be penetrated more easily (Janssen & Mulder 2004). The degree of penetration is measured by forcing a rode into the sediment, thereby measuring resistance. Again, in 2005, a significant difference was found in the sediment composition between the Holland coast (Egmond) and the Wadden coast (Schiermonnikoog) (Van Dalfsen 2006). The diversity in sediment composition (grain size range and the amount of shells in the through and first slope) was found to be higher at Egmond, resulting in more microhabitats present which enhance the living conditions for more species (Van Dalfsen 2006).

Based on results from MWTL monitoring 2004-2006 eight different macrobenthos communities have been identified in the coastal zone (Table 7), of which some are only found in a certain region (i.e. the Wadden-, Holland- or Southwest Coast). The exotic invader *Ensis americanus* has become a dominant species of the Dutch coast (Van Dalfsen et al. 2007).

Table 7 Characteristics of the communities found in the coastal zone (Van Dalfsen et al. 2007)

Community	Dominant species	Average Density (ind/m2)	Total species	Depth range (m)	Coastal region
C'	*Bathyporeia elegans* and *Magelona spec.*	817	17	8.5	Wadden coast
C1.1	*Ensis americanus, Urothoe poseidonis, Magelona spec.* and *Spiophanes bombyx*.	1330	66	2.5 - 19.5	Wadden-, Holland- and Southwest coast
C1.2	*Ensis americanus, Urothoe poseidonis, Notomastus latericeus, Phyllodoce mucosa, Magelona spec.* and *Lanice conchilega*	1279	55	10.8 - 19.5	Holland- and Southwest coast
C2	*Lanice conchilega, Ensis americanus and Nephtys cirrosa*.	660	17	7.5 - 13.2	Holland- and Southwest coast
C3	*Phoronids, Urothoe poseidonis* and *Ensis americanus*	460	28	8.3 - 19.5	Holland- and Southwest coast
C4	*Ophiura albida, Ensis americanus* and *Owenia fusiformis*	436	11	2.5	Southwest coast
C5	No dominant species	51	3	2.5	Southwest coast
D2	*Bathyporeia elegans* and *Fabulina fibula*	1555	117	16.9 - 21	Wadden coast

3.2.3 Zonation

Besides geographical differences of the Dutch coast (alongside the coastline), there are differences in relation to depth and distance from the low-water line. A relation was found between the macrobenthos distribution (species richness and density) and different zonation, i.e. the breaker bars, the troughs and the slopes. The zonation of the sandy shores in the Netherlands is described as follows (Janssen and Mulder 2005):

- supralittoral zone with insects and airbreathing crustaceans (outside the scope of this study)
- midshore zone with intertidal species,
- lower-shore zone, whose species extend into the shallow surf zone, and
- sublittoral fauna zone, with an abundance peak in the trough between the two breaker bars within the surf zone.

Figure 8 shows the characteristic macrofauna species of these zones. The trough between the two breaker bars consists primarily of large numbers of the sand mason. These worms presumably play an important role in holding on to and stabilising the sediment and thus the coast (Janssen & Mulder 2004, Janssen & Mulder 2005). The surf zone (in between the beach and the outer breaker bank) is generally poor in species but rich in individuals. In this zone the Polychaetes (worms) are dominant. The zone seawards starting from the outer breaker bank hosts a lot more species. They represent primarily the main groups Amphipoda, Bivalvia and Echinoïdea (Janssen & Mulder 2004, Janssen & Mulder 2005). A kite-diagram of all species found in the intertidal zone, during the 2002 survey, is included (Figure 9). Some species of the Dutch beaches are found along the full width of the intertidal zone, such as *Scolelepis squamata, Bathyporeia pilosa, Eurydice pulchra* and *Haustorius arenarius* (Janssen & Mulder 2004). However, most species only inhabit a specific part of the beach. For example, in the higher part of the intertidal zone *Talitrus saltator* and *Bledius sp.* are found and in the lower part *Bathyporeia pelagica* and *Urothoe poseidonis*. Of the 40 species found on the Dutch beaches, seven were classified as common: *Ensis sp*, *Scolelepis squamata, Nephtys cirrosa, Nephtys hombergii, Lanice conchilega, Spiophanus bombyx* and *Spio martinensis*.(Janssen & Mulder 2004).

The relation between the presence of organisms and sediment characteristics as was found for the Dutch beaches, corresponds with what is known from the literature about the ecology of sandy coasts elsewhere in the world (Janssen & Mulder 2004). Figure 10 presents a schematic view of the spatial distribution of taxa found on Belgium beaches and their trophic position. It shows the macrofauna (grazers and detritivores) and the higher trophic level of birds and fish (predators) as present in the supralittoral, littoral and infralittoral zone. The macrofauna species found on the Belgium coast are similar to those found on the Dutch coast. The crustacean *Bathyporeia* spp. are also found throughout the intertidal zone. The bivalve *Donax vittatis* was only found in the surf zone at Castricum.

It is not quite clear whether beyond the surf zone, where species diversity increases, other zones can be distinguished, as is the case for the beaches of south-eastern Africa (Janssen & Mulder 2005). There are, however, indications that different zones do occur beyond the surf zone in the Dutch coast. In the lower shoreface area, differences in assemblage structure were found between the 'steep slope' and the 'low area' (Baptist et al. 2006). The grain size sorting was found to correlate better with the benthic assemblage structure than the depth gradient. The species composition of the lower shoreface was found to be dominated by a mixture of polychaetes and molluscs and differed significantly between seasons. In March 2001 the molluscs *Tellina fabula* and *Spisula subtruncata* characterised the benthic composition, and in September 2001 the mollusc *Ensis americanus* showed a high abundance, together with the polychaete *Lanice conchilega*. Two characteristic polychaete species in March as well as in September are *Nephtys hombergii* and the *Spiophanes bombyx* (Baptist et al. 2006).

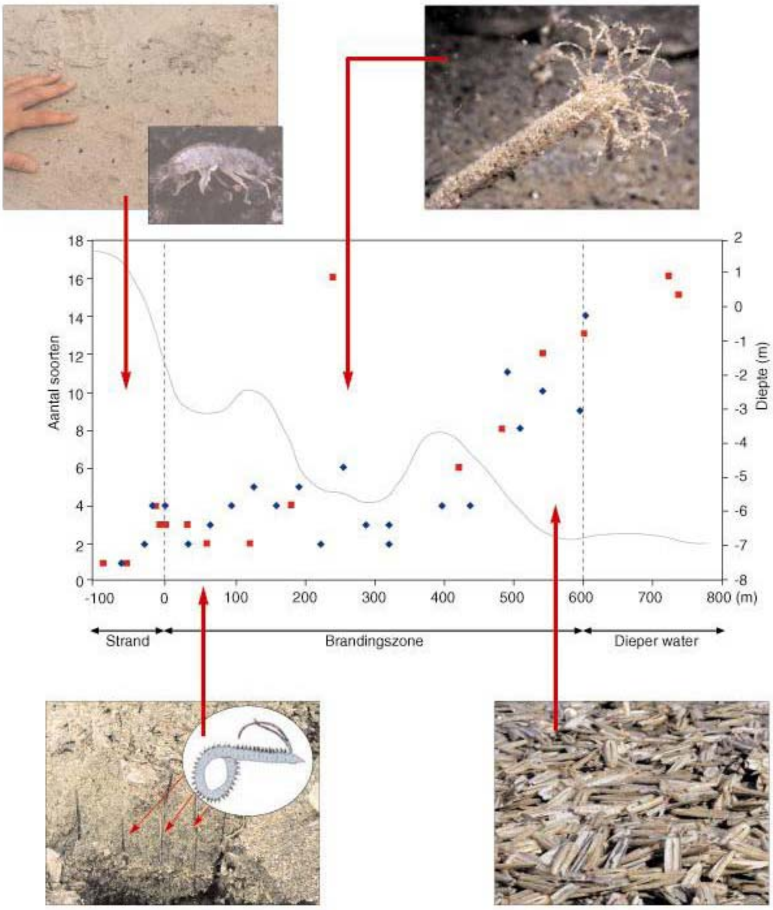

Figure 8 *Number of species in the beach and surf zones in relation to depth and distance from the low-water line. Characteristic species are shown: Talitrus saltator sand hopper (beach); Scolelepis squamata (beach and surf zone); Lanice conchilega sand mason (trough between the two bars); Ensis americanus razor clam (deeper water) (Janssen & Mulder 2005).*

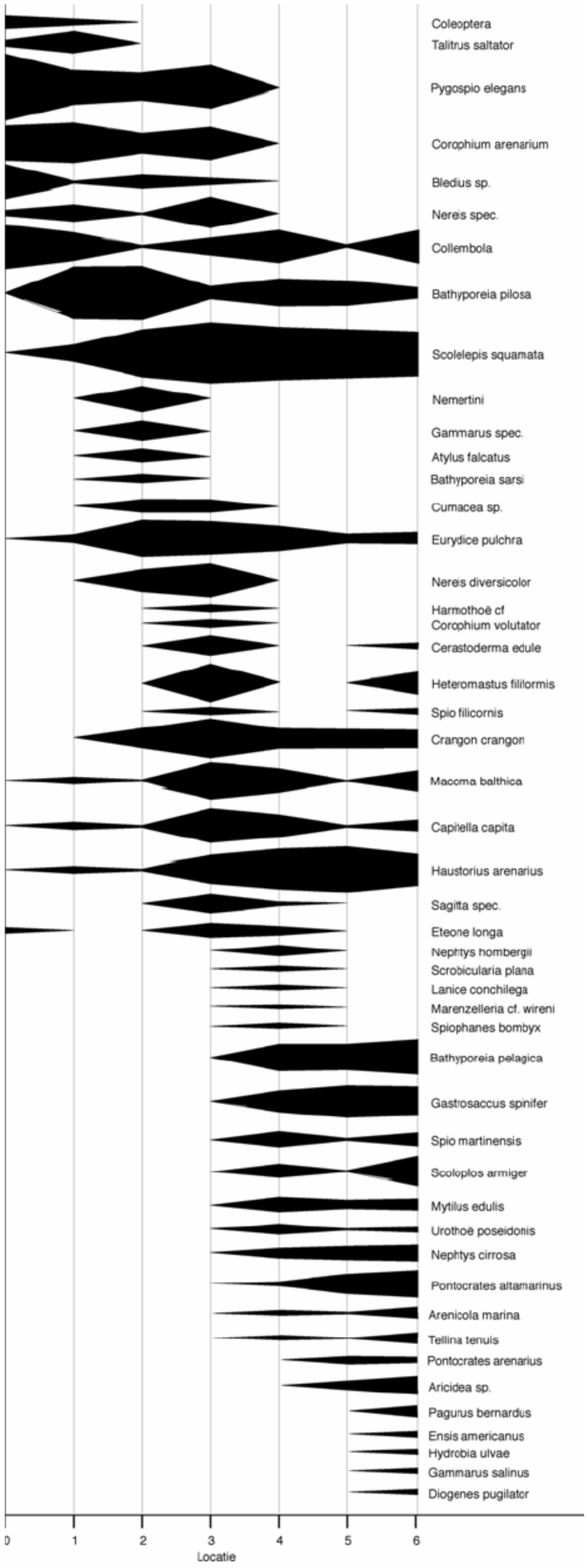

Figure 9 *Kite-diagram of the species found in the intertidal zone. Per species, the log-transformed density per location of all sampled beaches in 2002 was used (1 = high water line; 5 = low water line) (Janssen & Mulder 2004).*

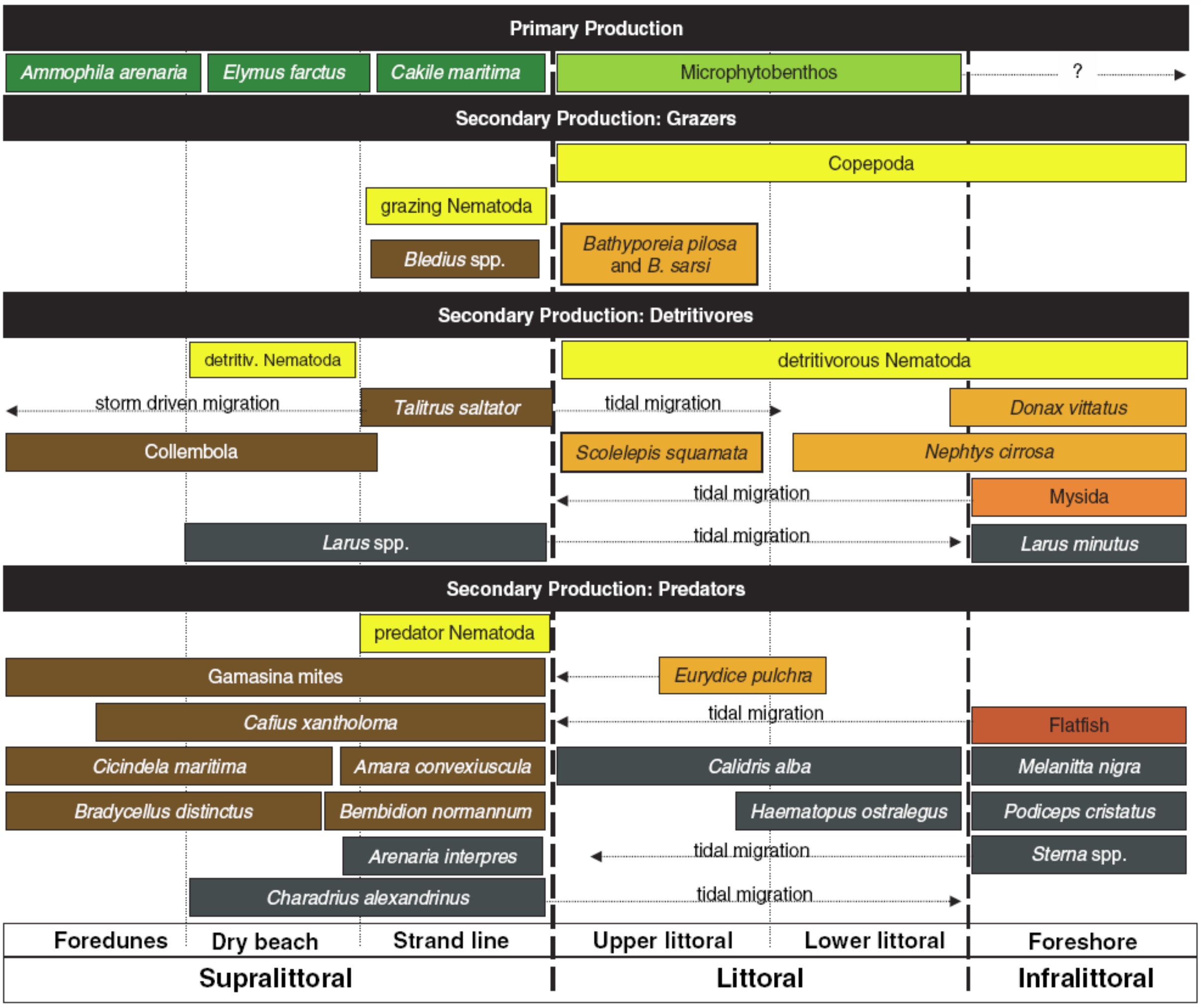

Figure 10 Productivity allocation, represented by some abundant taxa on Belgian beaches (Speybroeck et al. 2008).

A relation was found on the Dutch intertidal beach between diversity and abundance on the one hand, and grain size and slope on the other (Janssen & Mulder 2005), in accordance with the relation given by (Brown & McLachlan 1990): the numbers of species and their abundance both increase as the grain size and slope decrease. The maximum number of species was found between 0 and –1 m relative to the mean tidal level. The peak in abundance was found just above the mean tidal level (Table 8, locality 3), while the biomass reached a maximum at this level. (Brown & McLachlan 1990) have schematically presented the number of species in relation to the seawards distance (Figure 11). Other properties, such as penetration, also differ from the high water line to the low water line. Penetration is an important characteristic of sediment, as it determines the effort for small organisms to burrow themselves in the sediment and at determines the effort for birds to forage on these organisms. Around the low water line the amount of effort to penetrate the sediment is lowest, which enables organisms to easily penetrate the sediment.

Table 8 Grain size, sorting, penetration, carbonate content, slope, diversity, abundance, Shannon-Wiener index and biomass for each of 7 localities on the beach (mean of all nine beaches). HW (high-water line); LW (low-water line) (Janssen & Mulder 2005)

Locality	Position	Grain size [µm]	Sorting [φ]	Penetration [N/m^2]	Carbonate content [%]	Diversity [no. of species]	Abundance [indiv. m^{-2}]	Shannon-Wiener	Biomass [g AFDW m^{-2}]
0	40 cm above HW	265	3.6	516	4.4	7	146	0.5	0.2
1	HW	258	3.7	533	3.9	10	30	1.1	0.2
2	¼ tidal flooding time	253	3.4	564	5.2	14	639	1.4	1.3
3	½ tidal flooding time	270	3.4	486	10.2	19	1581	1.5	5.8
4	¾ tidal flooding time	298	3.2	385	11.6	28	729	1.7	3.7
5	LW	279	3.5	250	6.9	12	624	1.4	2.3
6	40 cm below LW	279	3.4	269	10.9	29	484	2.5	2.5

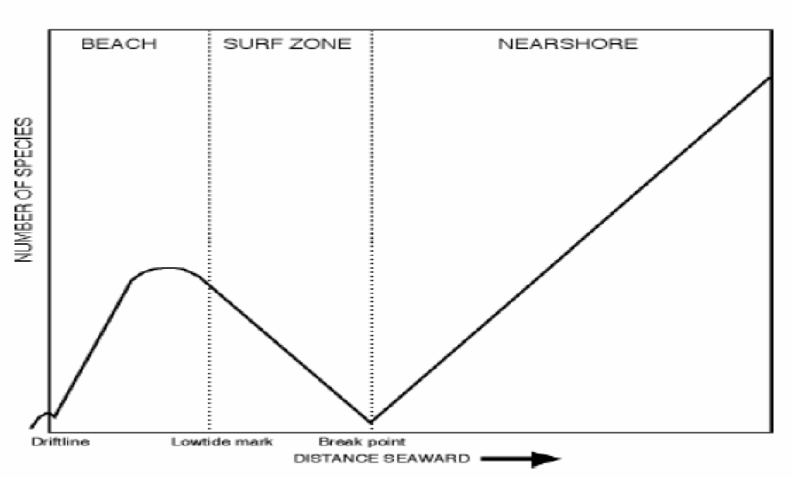

Figure 11 Relationship between the number of species and the distance to shore ((Janssen & Mulder 2004) to (Brown & McLachlan 1990)).

Holland- and Wadden coast

The depth profile of the Holland coast (Figure 12) in 2002 shows two breaker banks, the first one at ca. 100 m from the coast and the second one at ca. 400 m from the coast (Janssen & Mulder 2004). This pattern was also found in 2005, with occasionally the presence of a third breaker bank at Egmond (Van Dalfsen 2006). The Wadden coast (Schiermonnikoog) also shows the pattern of two breaker banks parallel to the shoreline (Van Dalfsen 2006).

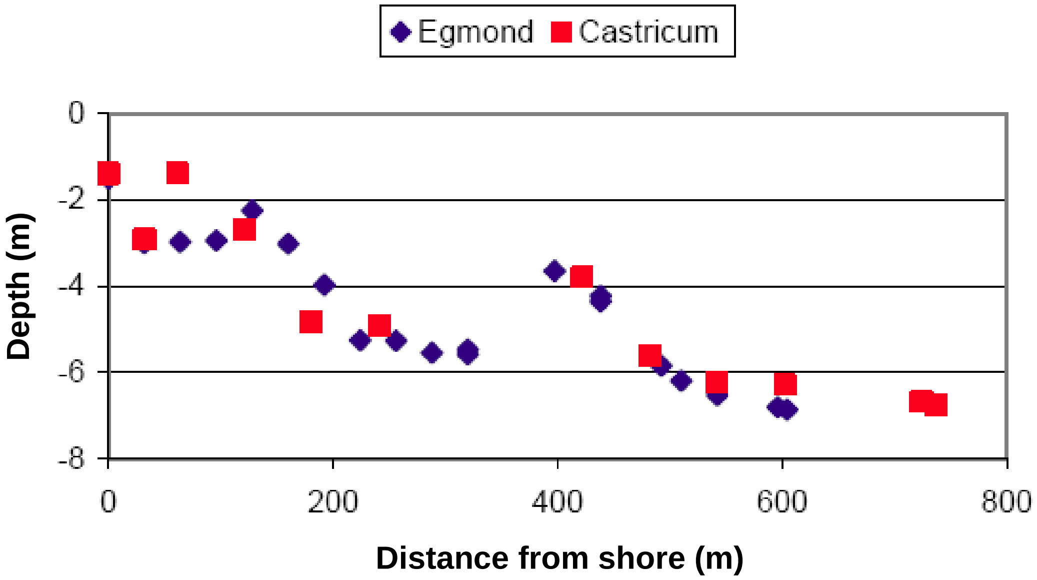

Figure 12 Depth profile of the surf zone at Castricum and Egmond. The distance from shore is presented from the low water line (Janssen & Mulder 2004).

Figure 13 shows the relationship between the number of species and depth, at different locations along the Southwest-, the Holland- and the Wadden Coast. Although there is a difference in the number of species found at the locations, the pattern is similar: the number of species increases from ca. 4 m depth, as shown in the two diagrams at the bottom of Figure 13. Other monitoring studies of the North Sea coastal zone show that the communities in the shallow coastal zones are relatively poor in number of species and that the number of species and the diversity increases with depth from ca. 6 m (Degraer et al. 2003, Van Hoey et al. 2004, Kaag et al. 2005).

The ecological zones (or habitat units) have been geographically mapped for the Holland coast (Figure 14, location Egmond) and for the Wadden coast (Figure 15, location Schiermonnikoog) (Slijkerman et al. 2007). The bathymetrical boundaries for the identified habitats are given in Table 9.

Table 9 Habitat Units and their bathymetrical boundaries (Slijkerman et al. 2007)

Habitat Unit	**Bathymetrical boundaries**
Habitat 1	-3 m. (rel. NAP) and shallower (at or near -0.5 m. or -1.0 m.)
Habitat 2	-3 m. to – 5.5 m.
Habitat 3	-5.5 m. and deeper (maximum depth encountered within the study areas was ca. 15 m)

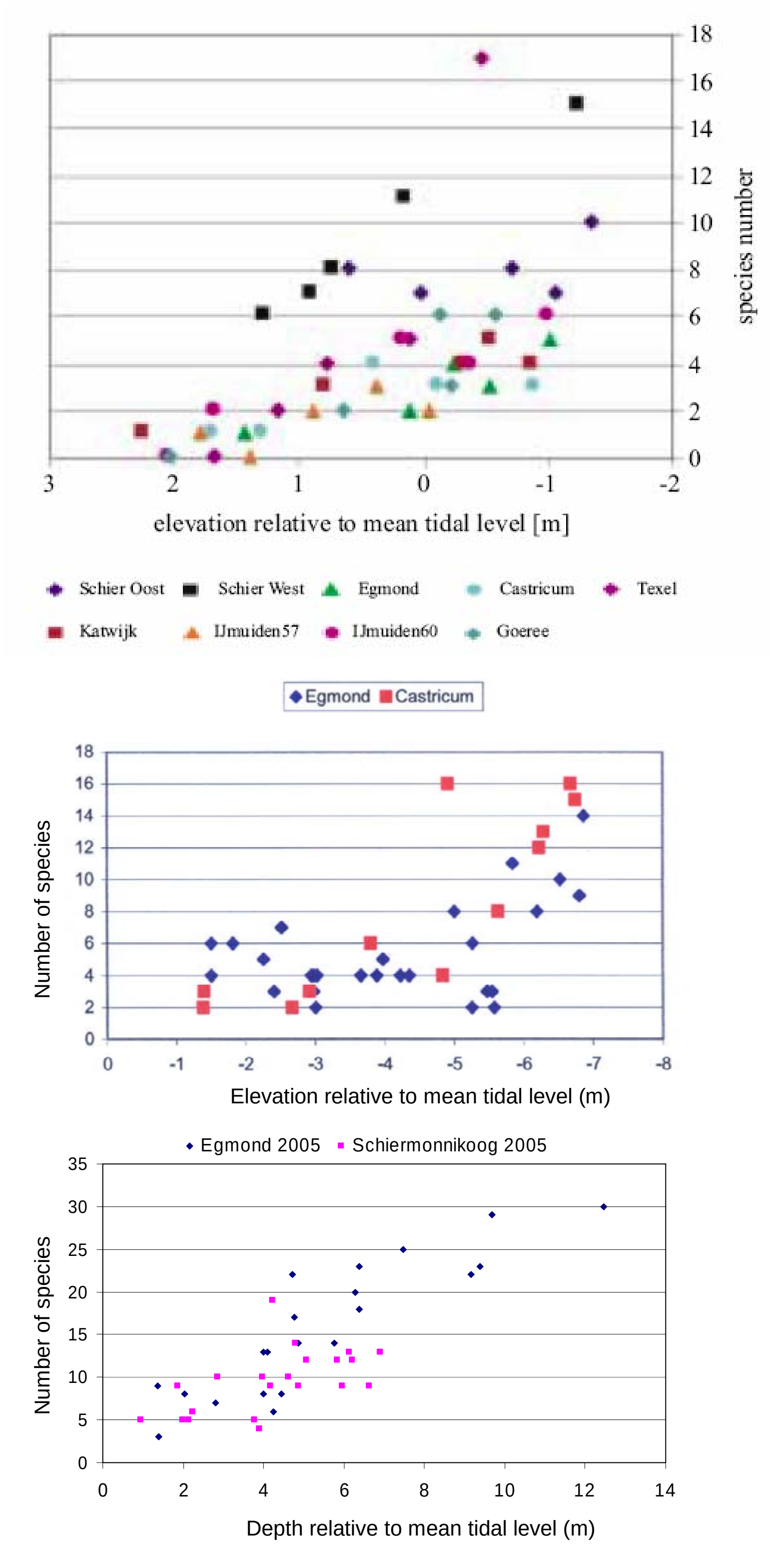

Figure 13 Relation between the number of species and elevations (or depth) relative to mean tidal level, found in 2002 at seven different beaches of the Dutch coast (top) (Janssen & Mulder 2005); found in 2002 at Egmond and Castricum (centre) (Janssen & Mulder 2004) and found in 2005 at Schiermonnikoog and Egmond (bottom) (Van Dalfsen 2006). Note that the mean tidal level (or NAP level) is 0 m.

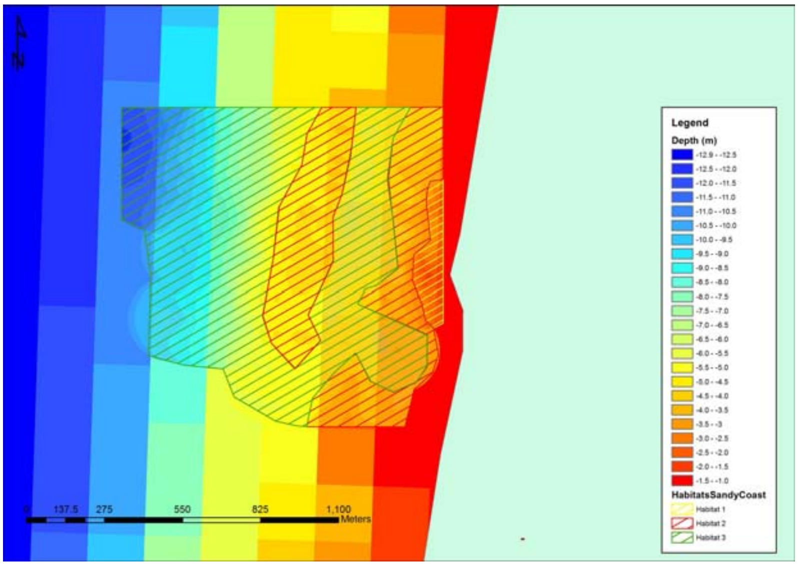

Figure 14. *Habitat map for Egmond in relation to available bathymetry (represented on coarser grid) outside the study area (Slijkerman et al. 2007).*

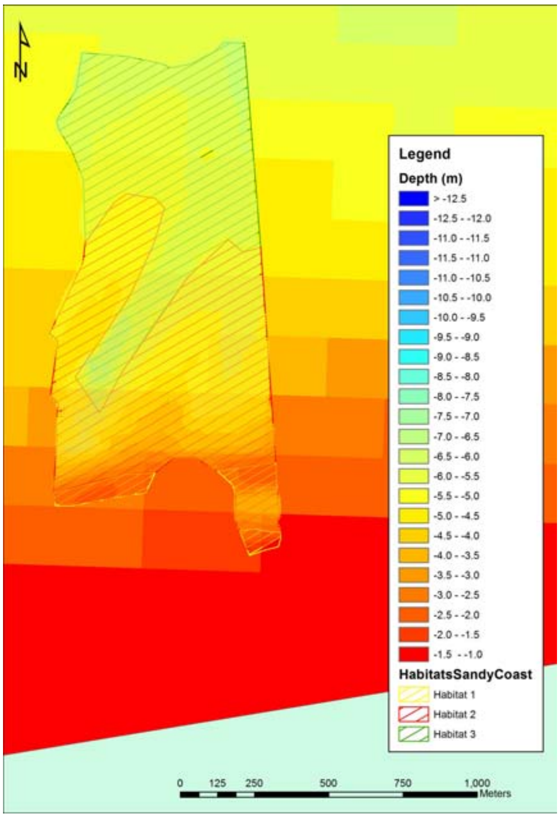

Figure 15. *Habitat map for Schiermonnikoog in relation to available bathymetry (represented on coarser grid) outside the study area (Slijkerman et al. 2007).*

3.2.4 Surf zone

The Dutch surf zone was studied in 2002 at the Holland coast (Egmond and Casticum) (Janssen & Mulder 2004, Janssen et al. 2008). The surf zone of the Holland coast (Egmond) was studied for the second time in 2005 (Van Dalfsen 2006). This study also included a survey at the Wadden coast (Schiermonnikoog). The surf zone of the Southwest coast has not been studied yet.

Polychaetes (bristle worms) and crustaceans dominate the macrobenthic fauna of the Dutch surf zone (Janssen & Mulder 2004). The same was found for the Belgian coast (Speybroeck et al. 2008) and many beaches worldwide. A total of 40 macrobenthic species were found in the Dutch surf zone, of which 7 were classified as common, i.e. they were found in most of the samples: *Ensis sp, Scolelepis squamata, Nephtys cirrosa, Nephtys hombergii, Lanice conchilega, Spiophanus bombyx and Spio martinensis* (Janssen & Mulder 2004). In the trough between the two bars at Castricum, a very high number of species was found. High densities of sand mason were also found at this location (at approximately 5 m depth and 250 m from the coast).

In 2005 the sediment characteristics and macrofauna of the surf zone at two locations of the Dutch coast (Schiermonnikoog and Egmond) were studied (Van Dalfsen 2006). The sediment at Egmond was coarser than the sediment at Schiermonnikoog (median grain size 200-380 µm at Egmond and 165-220 µm at Schiermonnikoog) (Van Dalfsen 2006). This is consistent with the natural difference in grain size between the mainland coast and the islands' coasts, as described above. The values for grain size found at Egmond in 2005 (Van Dalfsen 2006) are similar to those found in 2002 (Janssen & Mulder 2005).
The number of species found in 2005 at Egmond was relatively high compared to Schiermonnikoog (Van Dalfsen 2006). The number of species found in 2002 at Egmond was also much lower than in 2005 (Janssen & Mulder 2005). It was suggested that natural variation and the sampling depth, could explain the difference (Van Dalfsen 2006). In 2005, samples were taken in deeper water compared to monitoring in 2002. The maximum depth at which samples were taken in Egmond was ca. 12 meter in 2005 and ca. 7 meter in 2002. In Schiermonnikoog the maximum sampling depth was both years ca. 7 meter. It has been found that the species diversity increases with depth from ca. 4 meters onwards (Van Dalfsen 2006). This is consistent with other studies (Brown & McLachlan 1990, Janssen & Mulder 2004).

At an area located at the Holland coast near Heemskerk from 6 to 9 m depth, a total of 49 species (or species groups) were found, of which 11 crustaceans, 23 bristle worms and 10 bivalves (Van Dalfsen & Lewis 2001). The number of species is higher compared to the zone up to 7 meters deep (i.e. 40 species, see above), which is consistent with the previous described relationship between depth and species diversity. The common species of the surf zone, as described above, were also found at this location. Other common species were the bivalves *Spisula subtruncata, Tellina fabula, Tellina tenuis* and *Montacuta ferruginosa*, the crustacean *Urothoe poseidonis*, and the polychaetes *Eumida spp.*, *Magelona papillicornis* and *Phyllodoce mucosa*.

Analyses of the macrofauna community at the Wadden coast (Schiermonnikoog, 2005) and the Holland coast (Egmond, 2002 and 2005 and Castricum, 2002), showed four different clusters of macorfauna species (Van Dalfsen 2006). A distinctive habitat was found at the first breaker bar for the locations at Egmond and Schiermonnikoog (first cluster), with the characteristic species *Scolelepis squamata* (polychaeta) and *Haustorius arenaria* (crustacean). Much lower densities of species such as *Ensis americanus, Magelona mirabilis, Spio fillicornis* and *Capitella capitata* were found at the sites of the first cluster. The second group is only found at Schiermonnikoog and the third group only at Egmond. Both groups involve the deeper locations outside from the first and second breaker bar. The last group involves the shallow locations on top of the first and second breaker bar. As significant differences were found between the Wadden coast and the Holland coast, monitoring results from the shore face of the Holland coast do not represent the Wadden coast and vice versa (Van Dalfsen 2006).

A higher species diversity and density in the shore face of the Wadden coast was expected compared to the Holland coast, similar as was found for the beach. However, the species diversity and density was found to be lower at the Wadden coast (Schiermonnikoog) than at the Holland coast (Egmond) (Van Dalfsen 2006).

3.3 Effects of nourishment

3.3.1 Introduction

Possible ecosystem effects of nourishment can be divided into direct and indirect effects. Direct effects are mostly related to burial of benthic species. Indirect effects are caused by change in habitat through the introduction of 'exotic' sediment (i.e. sediment from another location with different properties). Altered sediment properties affect the habitat suitability for benthos, such as: level of the seabed; penetrability; organic matter content; grain size; and silt content. For example, impacts at nourished sites in the southeastern United States were observed during monitoring as a result of nearshore turbidity, direct burial of organisms and extreme habitat alterations (Lankford & Baca 1989),

Most studies assess short term impacts in and near nourishment sites, quantifying the elimination and early recovery of fauna but few studies cover longer periods (Nordstrom 2005). The direct effects of nourishment are therewith most known and can be related to:

- Burial of benthic species and suffocation of nearby benthos;
- Increased turbidity, possibly affecting primary production and/or the foraging success of predating fish and birds;
- Mortality of fish larvae;
- Disturbance by noise and ship maneuvering (outside the scope of this study);

The effects of elevated turbidity and sedimentation on benthic fauna are more significant in environments that have low natural concentrations of fine sediments, particularly in areas dominated by gravelly substratum (ICES 2000). The Dutch sandy coast is thus relatively insensitive to these direct effects of nourishment.

3.3.2 Burial

The following factors are known to determine the effect of burial on species (Kranz 1974, Maurer et al. 1980, Bijkerk 1988, Van Dalfsen 1994, Baan et al. 1998, Essink 1999):

- Depth of burial;
- Tolerance of species (life habitats, escape potential, degree of mantle fusion and siphon formation, low oxygen tolerance);
- Burial time;
- Nature of material (grain size different from native sediment);
- Temperature (mortality rate by burial higher in summer than winter).

Particles that settle onto the seabed are mixed into the top-layer of the sediment as a result of the activity of benthic fauna (bioturbation) and physical processes (stirring by currents and waves). Increased sedimentation can lead to mortality of benthos, which in turn could inhibit this mixing process. Burial can also lead to a chain of other stressors on benthic species communities like oxygen depletion and high sulphide concentrations. In general, the effect of burial mainly depends on the mobility of organisms in the sediment matrix and on the settling rate of particles. Sedentary organisms, which have no or very limited abilities to move, such as attached barnacles or mussels, are very sensitive. Other species with a low capability to move through the sediment, such as certain bivalve species, may eventually suffer from low oxygen concentrations in the sediment (Essink 1999). Most species present in muddy sediments or in high-energy, dynamic sediments are, however, well adapted to changes in their substrate. Especially species with normally active burring behaviour, experience hardly any effect (Bijkerk 1988).

Species which suffer most from burial with a sediment type different from the native one are the infaunal non-siphonate suspension feeders, infaunal mucus tube feeders and labial palp[1] deposit feeders. When buried with native sediment, the mucus tube feeders and labial palp deposit feeders seem to be the least affected groups. The group least affected by burial with exotic sediment are infaunal siphon-feeding bivalves. This could be explained by the fact that the members of this group do not demonstrate any significant burrowing escape (Smit et al. 2006).

---

[1] gathering food particles directly from the sediment with extensions of their labial palps

As listed above, the burial time also determines the effect on benthos. Distinction is made between incidental deposition, where species are buried by deposited material within a short period of time (i.e. minutes to hours) and continuous deposition, where species are exposed to an elevated sedimentation rate during a long period of time.

Incidental deposition (minutes-hours)
For benthic organisms a 'fatal depth' can be defined, which denotes at what depth of incidental burial the organism will not survive. This fatal depth is species dependent, but also differs with the type of sediment. Essink (1993) provides a literature overview of fatal depths for different organisms and two sediment types, silt and fine sand. In general benthic species are more sensitive to burial by silt than by sand. Furthermore, species of a sandy bottom are more sensitive to burial by silt than species of a silty bottom. Larger species are generally more capable of moving upwards than smaller species. However, the adult *Mya arenaria* is exceptionally large and is not able to move at all.

The fatal depth for incidental deposition of silt for a number of benthic species, selected from Essink (1993), is presented in Table 10.

Table 10 Fatal depth (cm) for incidental deposition with silt ((Essink 1993) from: (Bijkerk 1988), and (Smit et al. 2006))

Scientific name	Name	Fatal depth (cm)
Mytilus edulis	Blue Mussel	1
Petricola pholadiformis	American Piddock	3
Mya arenaria	Sand gaper	7
Cerastoderma edulis	Cockle	11
Hydrobia ulvae	Mudsnail	18
Macoma balthica	Baltic Tellin	38
Ensis ensis	Razor shell	43
Ensis Americanus	American Razor Shell	>40
Nephtys hombergii	a Catworm	60
Crangon crangon	Brown shrimp	19

Continuous deposition
A continuous deposition of material onto the bottom can have negative effects when the sedimentation rate is higher than the velocity at which the organisms can move or grow upwards. The sensitivity to long-term continuous deposition again is species dependent and also dependent on the type of sediment. A continuous deposition of silt is in general more lethal than a deposition of sand. Table 11 presents the maximum tolerance for different benthic species for a continuous deposition of silt and fine sand in cm/month.

Table 11 Maximum tolerance for continuous deposition of silt and fine sand in cm/month ((Essink 1993) from: (Bijkerk 1988))

Scientific name	Name	Deposition of silt (cm/month)	Deposition of fine sand (cm/month)
Mya arenaria	Sandgaper	2	5
Cerastoderma edule	Cockle		17
Macoma balthica	Baltic Tellin	15	>17
Arenicola marina	Lugworm	11	>17
Nephtys hombergii	a Catworm	>35	>17
Carcinus maenas	Crab	31	

Species Sensitivity Distributions
The sensitivity of the ecosystem can be represented by a threshold for burial, i.e. the threshold value (depth / layer thickness) for adverse effects caused by burial. Threshold values and risk curves for non-toxic sediment stressors were established to assess the impact of drilling discharges for the offshore oil and gas industry (Smit et al. 2006). A statistical description of the variation in sensitivity (Species Sensitivity Distributions) (SSD) was applied to derive the threshold value (Smit et al. 2006, Smit et al. 2008). Figure 16 shows the cumulative

distribution of the species sensitivity. It is indicated that species are more sensitive to burial by exotic sediment than native sediment (Smit et al. 2006). Therefore, a second SSD for burial by exotic sediment only, was drawn (see Figure 17). From this sensitivity distribution, the probabilistic value, at which 5% of the species are likely to be affected (HC_5), can be calculated. The HC_5 for burial by exotic sediment is determined at a level of 0.65 cm (Smit et al. 2006). This value is expected a better representation of the effects of burial by 'exotic' nourishment sand. However, it should be noted that only 5 of the 32 species on which the SSD is based, are found on the Dutch beach (i.e. *Cerastoderma edule, Crangon crangon, Ensis directus, Mytilus edulis* and *Nereis succinea*).The suggested PNEC (Predicted No Effect Concentration) of 0.65 cm is in the same range as a previous defined threshold level of 1 cm for non-moving sediment species (TNO 1994), and the list of fatal depths in Table 10.

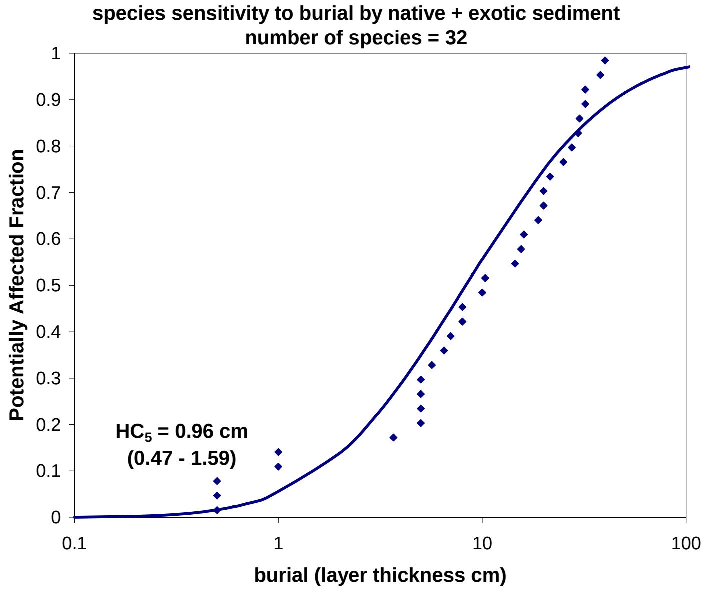

Figure 16 *Species Sensitivity Distribution (SSD) of benthic species based on data on burial by both native and exotic sediments (Smit et al. 2006).*

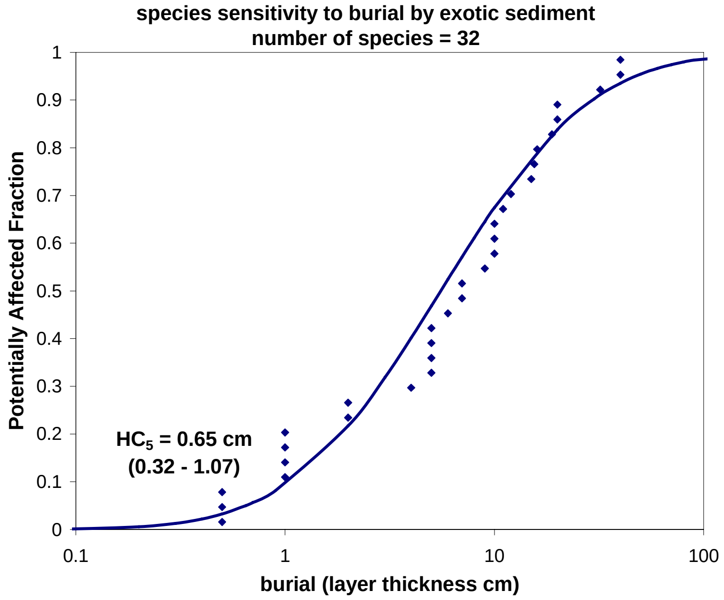

Figure 17 *Species Sensitivity Distribution (SSD) of benthic species for burial by exotic sediment only (Smit et al. 2006).*

Hypoxia

Besides the physical effect of burial, chemical effects of the anaerobic sediment, often together with high sulphide concentrations, play a role. A decreased dissolved oxygen level can amplify the effects of increased sedimentation. Hypoxia (i.e. a lack of oxygen) degrades bottom habitat through a wide suite of mechanisms. Under conditions of limited oxygen at the bottom, rates of nitrogen (nitrate) and phosphate remineralization, and sulfate reduction increase. The resulting production of nitrite, ammonia, and sulfide in combination with low oxygen can be lethal to benthic organisms (Buzzelli et al. 2002). Hypoxia may have several sub-lethal effects on organisms and population by impacting growth, survival, feeding, development, hatching, motion, respiration and settlement of individual benthic organisms. The cleaning of the siphons at increased sedimentation flux will cost more energy, while at the same time the oxygen levels are lower. The tolerance levels for low oxygen levels and high sulphide levels differ between species. A species such as the Brown Shrimp is a lot more sensitive to anaerobic conditions than species that are used to similar situations. In general, the critical dissolved oxygen concentration for survival of most benthic organisms (based on Northern European waters) is around 2.8 mg O_2/l, while certain crustacean and zooplankton species could tolerate 0.5-1 mg O_2/l for several days to weeks (Wu 2002).

The exposure time to anaerobic conditions (< 0.2 mg O_2/l) and for high sulphide concentrations (7 mg/l) at a 50% mortality level is presented in Table 12.

Table 12 Exposure time to anaerobic and sulphide-rich conditions at 50% mortality (Essink 1993)

Scientific name	Name	Exposure time oxygen (hours)	Exposure time sulphide (hours)
Mytilus edulis	Blue Mussel	800	600
Scrobicularia plana	Peppery furrow shell	600	500
Mya arenaria	Sand gaper	500	400
Nereis diversicolor	Ragworm	150	100
Cerastoderma edule	Cockle	100	100
Asterias rubens	Common Starfish	90	70
Carcinus maenas	Beach Crab	40	30
Amphiura filiformis	a Brittle Star	25	30
Crangon crangon	Brown Shrimp	2	2

Many ecosystems have reported some type of decline in dissolved oxygen levels through time with a strong correlation with human activities, as well as inputs of nutrients and organic matter (Smit et al. 2006). The processes that determine the oxygen content in bottom waters are (Smit et al. 2006):

- The consumption of oxygen due to degradation of organic material in the bottom water and sediments. The consumption rate depends on the amount and quality of organic material settling to the bottom and on the temperature;
- Consumption by infaunal organisms;
- The supply of oxygen from vertical mixing and horizontal transport processes. The supply rate depends on the hydrographical processes forced by wind, buoyancy and tides.

Sediments having oxygen-depleted overlying bottom water typically exhibit substantially reduced macrofaunal diversity. Within hypoxic zones the macrofauna exhibit low species richness and very high dominance of a few (tolerant) species. Among the macrofauna, many molluscs, crustaceans, echinoderms (e.g. sea potato), and cnidarians (e.g. sea anemones) appear less tolerant of hypoxia than other taxa, although there are exceptions. No single taxon dominates the macrofauna of low oxygen settings although annelid species are often prevalent (Smit et al. 2006).

A theoretic risk curve was constructed for modelling the risk of oxygen depletion (Smit et al. 2006), see Figure 18. This indicates a threshold level of 20% O_2 reduction (threshold level).

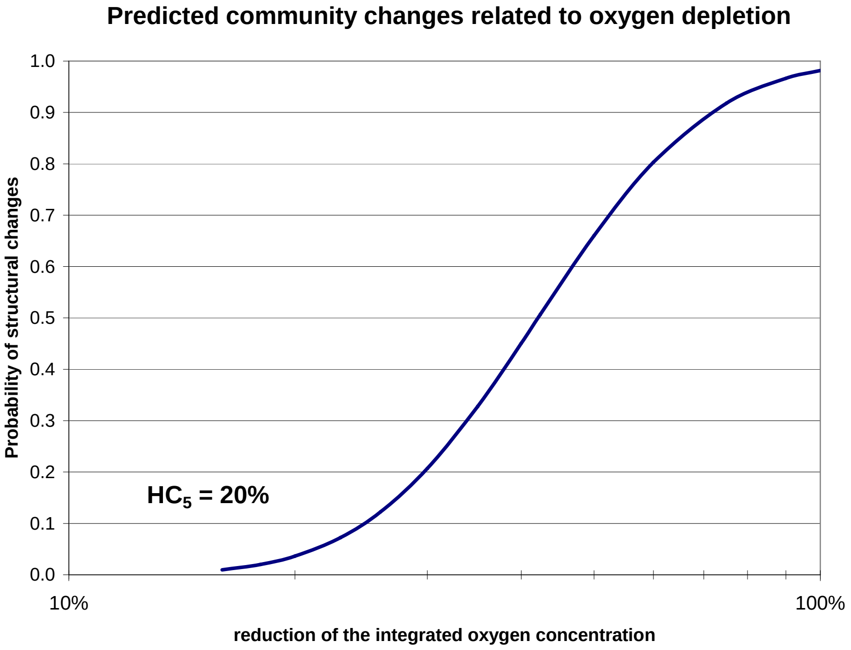

Figure 18 Theoretic risk curve for the reduction of the thickness of the oxygenated layer (Smit et al. 2006).

3.3.3 Turbidity

An increased Suspended Particulate Matter (SPM) concentration is especially harmful to ecological processes in the water column, but it may, directly or indirectly, also affect ecological processes that take place in the coastal zone.

Primary production (PP) in the North Sea is governed by the growth of phytoplankton. Additional turbidity may lead to a decrease in primary production by phytoplankton. When the primary production decreases, less food is available to primary consumers, such as zooplankton and zoobenthos.

As a result of the increased suspended solids concentrations, the food uptake by filter feeders can be negatively affected in two ways. First, the high concentrations of particles can clog the food uptake system and second the food quality (organic to inorganic ratio) may decrease. The extra energy it takes to filter the SPM out of the water can result in a decrease in the growth rate. The increased turbidity may also lead to a decreased concentration of phytoplankton, what in combination with a hindered food uptake can increase the effect on filter feeders.

The decreased food uptake may lead to a reduced growth of filter feeders. The filtering speed of filter feeders shows an optimum curve with SPM concentrations. Research into the filtering capacity of the Blue Mussel (*Mytilus edulis*) has shown that an average Mussel of 3 centimetres of length, reaches the maximum filtering speed at a SPM concentration of 125 mg/l. When the SPM concentration is 225 mg/l, the filtering capacity has decreased to about 30% and at a suspended solids concentration of 250 mg/l filtering will be ceased (Widdows et al., 1979).

Long-term nourishment and/or erosion of nourished sand could affect turbidity sensitive species (Speybroeck et al. 2004) by:

- Damaging gills and eggs due to low oxygen levels;
- Decreasing light penetration and therewith primary production;
- Decreasing sight and therewith inhibition of visual predators (e.g. some fish, crabs and birds);
- Inhibiting filtration (feeding and respiration) of bivalves;
- Slowing down recovery of macrobenthic organisms likely caused by sediment with high mud content (at least 2 years after nourishment (Rakocinski et al. 1996));

- Causing long term release of particulate matter if the sediment contains lumps of mud, which could increase turbidity up to many years after nourishment.

However, some aspects of nourishment of the Dutch coast suggest limited effects: because the sand used for nourishment is usually relatively coarse, the mud content and therewith increased turbidity is expected to be limited. Furthermore, increased suspended particles by nourishment, has in general little effect on the transparency of the water in the Dutch coastal zone, which has already a high turbidity (Lindeboom et al. 2005). This is consistent with other findings: A beach nourishment project in which sediment was pumped up on a US sandy beach (native and pumped sediment characteristics were similar), resulted in elevated total SPM levels in both the surf zone and nearshore up to 400 m from the discharge point. However, these levels were not as high compared to SPM levels after storms (Wilber et al. 2006). In addition, coastal communities are already adapted to a relatively high and variable turbidity of the water due to storms and wave action.

Although the focus of this review is on macrobenthos, the eggs and larvae of fish are recognized as being relatively sensitive to nourishment. Larvae and eggs of fish and shrimp, that are abundant in shallow areas, are sensitive to increased suspended particulate matter concentrations, more sensitive than adults (Sherk et al. 1975, Van Dalfsen 1999). An increased SPM may affect the respiration of larvae and the gas-exchange of eggs. SPM concentrations over 100 mg/l may lead to an increased mortality. An increased SPM concentration may also hinder the functioning of the gills of fish. In general pelagic species are more sensitive than bottom fish.
Besides sensitivity of larvae to the direct effects of increased SPM, they could also be indirectly affected. Fish larvae use light to regulate their vertical migration. Increased turbidity decreases the light penetration in the water which could mislead larvae to shallow and less suitable water (RIKZ 1999).

3.3.4 Grain size

Some studies (e.g. (McLachlan 1996)) have indicated that the grain size is of (great) importance to the ecological effect of the nourishment (Speybroeck et al. 2004). Monitoring results from both nourished and un-nourished sites have shown that more than 50% of the variability in the benthic assemblages was related to variations in the grain size structure of the sediments among shores (Colosio et al. 2007).

When the grain size and composition of the sediment to be applied significantly differs from the original sediment, the vertical migration (locomotion) of the infauna will be inhibited and compaction will be increased. Compacted beaches have a decreased number of burrowing organisms. There are several studies mentioned in the review (Speybroeck et al. 2004) that indicate that when the sediment or sand used for nourishment is of similar properties (grain size and organic matter) as the original sediment, it will result in the least impact on benthic macrofauna and the shortest recovery time. Long-term changes in grain size distribution caused by the ongoing supply of coarser sediment can not be excluded (Mulder et al. 2005).

SSDs for mortality effects of burial (see Figure 16 and Figure 17 of the previous section on burial), based on layer thickness, also reflect the relationship between sensitivity to burial and sediment properties. The potentially affected fraction of species is less for burial by native sediment than by non-native sediment at the same layer thickness (Smit et al. 2006).

Although many studies have revealed a relationship between sediment type and infauna community structure, there is considerable variability in species response to specific sediment characteristics (Smit et al. 2006). The studies suggested that the factors ultimately controlling infauna distributions may not be sediment grain size per se or factors correlated to it (such as organic content), but rather interactions between hydrodynamics, sediments and infauna and how these affect sediment distribution, larval supply, particle flux and pore water chemistry (Snelgrove & Butman 1994). Although acknowledging the complexity of these processes, (Smit et al. 2006) used a simplified approach in which the change in median grain size was taken to represent the overall changes in sediment characteristics. The sensitivity of species to changes in grain size has been derived by the use of ‘the window-of-occurrence’, which is defined as the observed range of median grain sizes per species (Smit et al. 2006). Species with a small window-of-occurrence are more sensitive to changes in grain size than species with a wide window-of-occurrence. The median grain size together with the presence of specific species is frequently measured in field surveys. It was observed that North Sea species are more sensitive to changes in grain size than species from the Norwegian and Barents Sea (Smit et al. 2006). Based on the absolute width of windows-of-occurrence for 300 species a Species Sensitivity Distribution was constructed describing the spread in sensitivity of biota to grain size changes (Figure 19). From the sensitivity distribution presented in Figure 19, the probabilistic value at which 5% of the species are likely to be affected (HC_5) can be derived. This value of 52.7 µm can serve as a threshold value for changes in grain size. According to the overview for some Dutch coastal species (Table 13), the bivalve *Donax vittatis* (which can be found in the surf zone) is found in sediment with a relatively narrow range in grain size. Based on data of the Dutch Continental Shelf, a HC5 of 21% relative to the mean grain size has been determined (Smit 2004). This means that, for the species listed in Table 13, a change in grain size ranging from 53 to 63 µm would be acceptable. The HC5 of 53 µm, as derived from the SSD (Figure 19) seems therefore a suitable protection level.

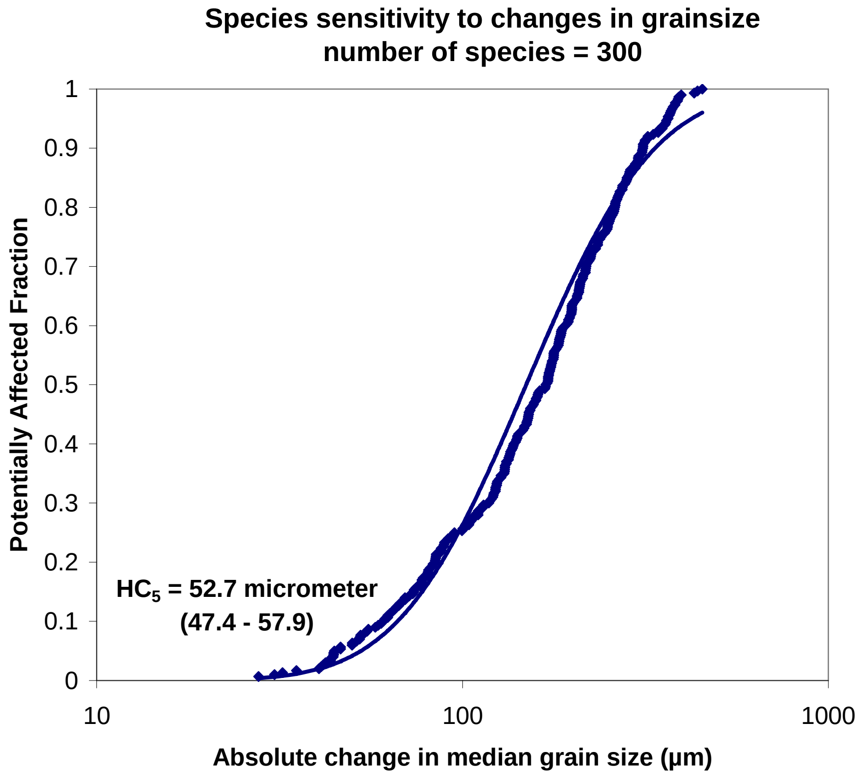

Figure 19 *Species Sensitivity Distribution (SSD) based on the absolute natural grain size window-of-occurrence (95% interval) of 300 North Sea, Norwegian Sea and Barents Sea species (Smit et al. 2006).*

Table 13 *Some Dutch coastal species and the grain size (median and standard deviation) of the sediment*

Species	Class	Median grain size (µm)	St dev
Donax vittatus	Bivalvia	225	44
Macoma balthica	Bivalvia	200	64
Spisula subtruncata	Bivalvia	209	74
Bathyporeia elegans	Crustacea	243	67
Echinocardium cordatum	Echinoidae	227	74
Lanice conchilega	Polychaeta	228	70
Nepthys cirrosa	Polychaeta	262	75
Nepthys hombergii	Polychaeta	189	57
Scolelepis squatama	Polychaeta	265	64
Spiophanes bombyx	Polychaeta	235	75

3.3.5 Field surveys

There are many articles and reports available that describe the impact of beach nourishment, based on either field results ((Dankers et al. 1983, Essink 1993, McLachlan 1996, Rakocinski et al. 1996, Peterson et al. 2000, Van Dalfsen & Essink 2001, Menn et al. 2003, Wilber et al. 2003, Bilodeau & Bourgeois 2004, Janssen & Mulder 2004, Peterson & Bishop 2005, Bishop et al. 2006, Wilber et al. 2006, Colosio et al. 2007, Leopold & Baptist 2007, Jones et al. 2008, Van der Wal & Van Dalfsen 2008, Fanini et al. 2009), or literature review (Greene 2002, Speybroeck et al. 2004, Mulder et al. 2005, Nordstrom 2005, Speybroeck et al. 2006). Some relevant study results are described below.

An extensive study has been conducted on the impact of sand nourishment on the ecosystem of the Belgium coast (Speybroeck et al. 2004). The ecological effects were linked to three different aspects of sand nourishment, i.e. the activities, the quantity of sediment and the quality of sediment. The quality of sediment can be described in terms of: fraction of shells and other coarse material; fraction of mud and organic matter (i.e. particulate matter); grain size distribution; beach morphology and – profile; presence of toxic substances and/or dead organisms. The effects were also found to be influenced by the (a) technique and strategy of nourishment and the (b) location, time and scale of nourishment.

Effects

The effects of sand nourishment were first studied in the Netherlands in 1980-1981, on the islands of Texel and Ameland, see Table 14 for characteristics (Dankers et al. 1983). The main species found at the two sites before nourishment was *Scolelepis squamata* . Four months after nourishment at Texel, there were hardly any individuals left and two years after nourishment the biomass and abundance was recovered above the low water line. Below LWL there were still relatively few adults present, compared to the reference site. At Ameland, there were hardly any effects on benthos. Nourishment at Ameland was conducted high on the beach, close to the dunes. The effects were therefore related to the method of nourishment (Dankers et al. 1983). However, the studies of (McLachlan 1996, Rakocinski et al. 1996, Speybroeck et al. 2004, Colosio et al. 2007) suggest that also the sediment characteristics (i.e. change in median grain size and presence of lumps of peat and clay, see Table 14) could have had a significant influence on the effect. A study cited by (Greene 2002), also found relatively small effects when using sand of similar characteristics compared to the native sediment in combination with application high on the beach, which allows gradual sand redistribution.

Table 14 Overview of nourishment conditions and effects on benthos, based on (Dankers et al. 1983)

Location	Amount	Size	Application	Sediment characteristics	Effects on benthos
Ameland	2.4 Mm^3	2-6 m thick and 40 to 70 m width	High on beach, above NAP	Grain size after nourishment comparable to before nourishment	Negligible
Texel	3 Mm^3	1.5 m thick and 500 to 550 m width	On beach, above and below NAP	Median grain size after nourishment ca. 50 μm smaller than before, average silt content was higher and lime content was lower than before. Lumps of peat and clay were also found after nourishment	Mortality of benthos with recovery above WL after 2 years

In 2007, the benthos at four nourishment sites (Petten, Bloemendaal, Zandvoort and Bollen van Kijkduin) on the Dutch coast was studied (Van der Wal & Van Dalfsen 2008). Samples were taken at depths of 5 to 10 meters at the seaward side of the outer breaker bar. All areas were characterized by fine sand with low mud- and organic matter content. The study focussed on the presence of two bivalve species, *Spisula subtruncata* and *Ensis spec.*, because of their importance as a main food source for some (Natura 2000 protected) bird species and as an important macrofaunal species of Natura 2000 habitat type 1110 (Sandbanks which are (slightly) covered by seawater all the time). Bivalves seem to be particularly sensitive to the effects of nourishment (Colosio et al. 2007). The bivalve species *Spisula substruncata* was not found at any of the locations (Van der Wal & Van Dalfsen 2008). These findings are consistent with other studies, where it was reported that *Spisula subtruncata* was not found at the locations (Perdon & Goudswaard 2007). It was noted that this species is found further from the coast in deeper water. The observed decline in *Spisula* since 2001 could not be related to nourishment activities (Leopold & Baptist 2007). *Ensis spec*. was found in low densities at several locations, but mainly at Bloemendaal (Van der Wal & Van Dalfsen 2008). The yearly shellfish survey also found the bivalve *Ensis spec*. in low densities (Perdon & Goudswaard 2006).

The benthos samples at the nourished sites showed a low species diversity, mainly characterized as a *Nephtys cirrosa* (Polychaeta) community. It is noted that these findings are consistent with the Belgian coast, as described by (Van Hoey et al. 2004), where *Nephtys cirrosa* was also found to be widely spread. Species composition of the four locations was fairly similar. Observed differences were mainly related to abundance and not diversity. The sand mason *Lanice conchilega* was found in low densities at relatively deep sampling sites (ca. 7 m or deeper) (Van der Wal & Van Dalfsen 2008).

Macrobenthic responses to extensive beach restoration at Perdido Key (Florida, USA), was studied during 3 years (Rakocinski et al. 1996). The morphodynamic state of the beach at Perdido Key is comparable to the Dutch beaches, with shallow beach slopes, nearshore bar/trough formation and fine to medium grain sand. The restoration consisted of two phases: (1) beach nourishment, completed over a one-year period between November 1989 and September 1990, where ca. 4 million m3 of dredge material was deposited by a pipeline system along 7 km of shoreline and (2) profile nourishment, completed over a one-year period between November 1990 and September 1991, where ca. 3 million m3 dredge material was deposited subtidally by hopper dredge over a considerable offshore area running parallel with the shoreline for 3.8 km, between 400 and 600 m seawards at ca. 6.7 m depth. The sea floor was raised by ca. 2 m over much of the subtidal disposal area. The beach nourishment extended the beach width, resulting in steep drop-offs with less sandbar development and increased the offshore silt/clay content and grain size distributions. Near shore, typical shallow-water macrobenthic assemblages characteristic of the usual dissipative beach was reduced after beach nourishment from to that of a reflective beach morphometry. Offshore, the macrobenthos was impacted by both beach nourishment and profile nourishment. Macrobenthic impacts (i.e. assemblage structure and species richness) from silt/clay loading still were evident more than two years after beach nourishment. Profile nourishment induced wide fluctuations in offshore (300 – 800 m from the original coastline) macrobenthic populations for more than one year after nourishment activities, presumably from shifting sediments (Rakocinski et al. 1996).

Sediment characteristics

The Dutch coast is exposed to prevailing SW-NW winds and has a fairly great morphodynamics in the shoreface area, caused by the interplay of tidal currents and wave climate. Because of these conditions, the original morphological features of ridges, breaker banks and troughs can be expected to return at a relatively short time scale after sand nourishment, by the exchange of nourished sand with that of the surroundings of the actual nourishment site. This has been observed at Terschelling, where the enhanced median grain size in the nourishment site (180 µm compared to the original of 165 to 178 µm) disappeared within six month after completion of this nourishment operation (Essink 1993). Regarding the relation between the grain size and the morphology of the beach, the individual effects of these two factors on benthos are hard to distinguish in the field (Speybroeck et al. 2004).

It has been observed that an increase in sand particle size (on a beach where tide range and wave energy have remained constant) results in a change in beach state (from dissipative to intermediate) and a decrease in species richness and abundance (McLachlan 1996). Mean sand particle size increased from original values of 110 to 160 µm to values of 500 to 800 µm. The large sand mussel *Donax serra* disappeared in the treated area. A study on the long term (>1 year) effects of onshore beach nourishment of shores along the North Adriatic Sea (Italy) showed a decrease in macrofauna which was related to greater particle size (Colosio et al. 2007). Also nourishment with relatively fine sand and large concentrations of shell hash was found to affect benthos (i.e. *Emerita talpoida* and *Donax spp*.) (Peterson et al. 2000).

Significant coarsening of sediments and associated changes to assemblages of benthic macroinvertebrates was observed 8 months after deposition of dredged material on subtidal shoals at depths of ca. 8 m (Bishop et al. 2006). Impacts to sediments and macroinvertebrates were closely correlated and, although greatest where sediment was directly deposited, extended over a wider (at least 1 km to the east) area than the deposition.

The abundance of the crustacean amphipod *Talitrus saltator* (a proposed indicator of human impact for the Mediterranean) did not seem to be directly affected by beach nourishment on a sandy beach in Italy (Fanini et al. 2009). *Talitrus saltator* is a characteristic species of the Dutch beach and is present in the zone above the low water line. The nourishment consisted of groynes building, creating beach segments, of which a few were filled with marble gravel. Although the abundance of the sand hopper could not be directly related to the presence of nourished material, indirect effects were observed. The beach's physical characteristics were altered by the nourishment and consequently its fauna on different organizational levels (Fanini et al. 2009).

On the long term, a structural change of grain size by repeated nourishments with relatively coarse sand could have adverse effect on the biodiversity of the macrobenthos of the beach (Janssen & Mulder 2004).

A long term study on the impact of beach- and profile nourishment suggests that even well-matched sediments can hamper macrobenthic recovery through sediment shifting (Rakocinski et al. 1996).

The sand used for nourishment in the Netherlands is in general slightly coarser than the original sediment. For 70 nourishment projects that have been conducted in the Netherlands, the grain size before and after nourishment was measured. The average change in grain size was 59 µm. The threshold level of 53 µm was exceeded at 22 different occasions.

Zonation

Because of the different ecological zones (see section 3.2.3), the effect of nourishment on benthos will depend on the location of sand deposition, i.e. beach or shoreface. At nourishment sites the macrofauna community at the shallow zone was also found to differ from that of the deeper parts of the coastal zone (Van Dalfsen 2006). This indicates that the naturally formed ecological zonation, is not significantly affected by nourishment as conducted in the past.

Macrofaunal abundance and species density in the deeper subtidal zone were found to be lower than at the reference site, even nine months after nourishment with 351,000 m3/2 km beach line (Menn et al. 2003). However, the effects were considered not as dramatic compared to natural changes along the shore and between years.

Monitoring results after both beach- and shoreface nourishment indicate that diverse offshore assemblages may be less resilient than contiguous nearshore sandy-beach assemblages (Rakocinski et al. 1996).

Temporal aspects

Because reproduction takes place in spring and fall, the preferable time for nourishment was suggested to be in winter, which would enhance recovery (Janssen & Mulder 2004).

3.3.6 Recovery

Recovery has only been studied on a limited number of (opportunistic) species. No information is available on the degree of recovery on the level of biological communities.

Survival, migration and recruitment may all contribute to the recovery after a disturbance (Van Dalfsen & Essink 2001). The recovery after nourishment therewith depends on many factors, such as the application method/location, the sediment characteristics (influencing both chance of survival and recruitment), the species resistance and resilience and the season of application. Recovery can sometimes be fairly rapid (e.g. some months to <1 year, because of the quick dispersal of sediments and/or the intrinsic tolerance of the assemblages) but can quite often be long-lasting, particularly when the sediments alter the native habitat characteristics, or have high organic loads and/or are highly polluted (Colosio et al. 2007). As the sand used for nourishment of the Dutch coast is not polluted or organically enriched, benthos should be able to recover relatively fast, assuming other sediment characteristics (i.e. grain size and mud content) are fairly similar to the original sediment.

Dutch studies

Some results of studies on the recovery after both sand extraction and sand nourishment projects in the Dutch coastal zone are presented in Table 15 and further discussed below.

Table 15 Recovery after sand extraction and –nourishment projects in the Dutch coastal zone

Type / Location	**Nourished sand**	**Recovery**			**References**
		Sediment composition	**Morphology**	**Biology**	
Sand extraction (Heemskerk)	Not relevant	3 months	< 4 years	< 4 years	(Van Dalfsen & Lewis 2001)
Beach nourishment (Texel)	Finer sand (ca. 50 µm decrease), higher mud content and lower calcium content	Unknown	Unknown	< 2 years	(Dankers et al. 1983)
Shoreface nourishment (Terschelling)	Coarser sand (17-57 µm increase), higher mud- and calcium content	< 6 months	< 2 years	< 4 years	(Van Dalfsen & Essink 1997)

The effects of sand extraction at a site near Heemskerk were studied by (Van Dalfsen & Lewis 2001). The activities conducted in the area had significant effects on the morphology and composition of the sediment and on the benthos. One year after the activities had ended, morphological differences were still observed. The sediment composition however (grain size, organic matter, lime and silt) were already recovered within ca. 3 months. This indicates that human induced changes in this dynamic part of the coast can be undone by nature in a short period of time. After 4 years, morphological differences between the extraction site and reference site could no longer be found. The benthos was also fully recovered within 4 years. Short term recovery of the sediment structure was also observed after shoreface nourishment off the Island of Terschelling (Van Dalfsen & Essink 1997). Six months after the nourishment was completed, the grain size distribution and the median grain size showed no significant changes with pre-nourishment situation.

Recovery of effects of nourishment at Texel in 1979, were the sand was applied on the beach both above the HWL and below the HWL (to ca. 3 meters deep), was after nearly two years (Dankers et al. 1983). When the sand was applied high on the beach (close to the dunes), as was the case at Ameland, hardly any effects on benthos was observed. Besides direct effects from burial, possible (additional) effects may have resulted from habitat alterations: the nourished sand differed in grain size from that of the original sand and contained lumps of peat and clay.

A monitoring study of shoreface nourishment projects at four North Sea sites (including one site in the Netherlands, i.e. Terschelling), found short-term effects: a reduction of abundance and biomass of species (Essink 1997). It was concluded that recovery did proceed rather fast. The import of live benthos with sand from the borrow site was considered not to play an important role. The immigration of specimens from the direct surroundings of the nourished area were considered more important in the hydro/morphodynamic system of ridges and runnels (Belgium) and breaker banks and troughs (Netherlands, Germany and Denmark). It is noted that only in the further process of recovery settlement of recruits becomes important (Essink 1997). For most of the species, abundance and biomass had largely recovered already ca. 1 year after completion of the nourishment. Long living species, such as bivalves (e.g. *Spisula subtruncata*, *Donax vittatus*) and sea urchins (*Echinocardium cordatum*) showed a much slower recovery. For these species, that do not reproduce successfully each year, recovery of total biomass and a normal age structure is considered to take 2-5 years (Essink 1997). Following shoreface nourishment in Dutch coastal waters a short-term opportunistic response of the benthic community was observed followed by an almost complete recovery of community composition and structure after four years (Van Dalfsen & Essink 2001). This is consistent with the observations after sediment disturbance caused by sand extraction (Van Dalfsen & Lewis 2001).

International studies

Long-term (>1 year) offshore macrobenthic impacts have been observed after both beach- and shoreface nourishment (Rakocinski et al. 1996). The sediment used for nourishment contained high amounts of mud which was thought to slow down recovery of macrobenthic organisms. This was also found for intertidal populations of the burrowing ghost shrimp, *Callichirus islagrande*, where high levels of silt/clay loading in the nourished sand appeared to have slowed population recovery (Bilodeau & Bourgeois 2004).

Effects of burial on soft bottom benthic species are temporary. Opportunistic species will quickly recolonise the affected site, but long-living bivalve species or some sea urchins (such as the sea potato, *Echinocardium cordatum*) do not reproduce each year. In general, soft bottom benthic communities show partial recovery in one year and full recovery from 18 to 24 months (Allen & Hardy 1980) up to a maximum of 5 years (Mulder et al. 2005) .

The impact of beach nourishment (by use of pipelines) on abundance of the intertidal amphipod, *Exoediceros fossor,* was very large at both borrow and nourishment locations (Jones et al. 2008). However, recovery started within several weeks and appeared to be complete within a year.

The impact of beach restoration was studied on intertidal populations of the burrowing ghost shrimp, *Callichirus islagrande*. Two years after beach nourishment with sediment different from the native sediment (with respect to silt/clay, sand, and gravel fractions) minimal population recovery (three individuals) was evident (Bilodeau & Bourgeois 2004).

For a few biomorphologic relevant species (see section 5.2), the recoverability is provided (Table 16). Recoverability is defined as the ability of a habitat, community or individual (or individual colony) of species to redress damage sustained as a result of an external factor (Hiscock et al. 1999). Recoverability is only applicable if and when the impacting factor has been removed or has stopped and only refers to the recoverability potential of a species, based on their reproductive biology etc. All species considered are classified with a high recoverability potential. High recoverability indicates that full recovery will occur, but will take many months (or more likely years) but should be complete within about five years (Hiscock et al. 1999).

Table 16 Recoverability of some macrobenthic species

Species	**Common name**	**Life span**	**Reproduction frequency**	**General recoverability**	**References**
Lanice conchilega	Sand mason	Short	Insufficient information	High	(Rabaut et al. 2008) (Ager 2008)
Tellina fabula (synonym *Fabulina fabula*)	Bean-like tellin	2-5 years	Annual	High	(Rayment 2008)
Echinocardium cordatum	Sea potato	10-20 years	Annual episodic	High	(Hill 2008) (Essink 1997)

Based on available information from both national and international studies, it can be concluded that full recovery can be expected within 4 to 5 years.

3.3.7 Mitigation of effects

For most species, the oxygen consumption rate is lower in winter than in summer. This can cause organisms to survive longer in winter after burial. Movement of the organisms, however, is also slower, so it takes longer for the organism to escape from the layer of burial. The influence of the season on the effect of burial is therefore hard to predict. It depends on the species, location and temperature. However, because most species reproduce and establish in spring and summer, nourishment in this period is expected to cause increased effects (Jongbloed et al. 2006). This was found in a study of the bivalve Donax, where it was hypothesised that rate and timing of recovery is affected by the seasonal timing of nourishment (Peterson et al. 2000).

In contrast, effects of increased turbidity on benthic primary production are expected to be higher in winter. This is because the benthic primary production is highest in February (Jongbloed et al. 2006).

Table 17 summarises the effects related to different options in nourishment, i.e. location, type of sediment, season and frequency. Nourishment conducted in winter, and not each year at the same location, is expected to have the least effects on macrobenthos.

Table 17 Management options related to the deposition of sediment with expected effects

Factor / option	**Expected effect or best option**	**Reference**
Location	Mortality of macrofauna depends on the size/amount of the sedimentation (i.e. deposition of sand)	(Jongbloed et al. 2006)
	Many small nourishments (<800 m) are preferred over one large nourishment, as the non-affected area between sites is expected to enhance recovery (depending on species)	(Speybroeck et al. 2004)
Suspended matter	Adverse effect on filtering benthic species expected with more than 20% increase of suspended matter	(Jongbloed et al. 2006)
Sediment characteristics		
Type	Mortality of macrofauna depends on the type of sediment	(Jongbloed et al. 2006)
Grain size	Not smaller than original beach	(Speybroeck et al. 2004)
Amount of shells	Should be low, as they can remain in the area	(Speybroeck et al. 2004)
Amount of mud balls	Should be low	(Speybroeck et al. 2004)
Toxic chemicals	Should not be present	(Speybroeck et al. 2004)
Time		(Jongbloed et al. 2006)
In fall	No adverse effect on the winter migration of juvenile *Macoma balthica* (Baltic tellin).	
	Relatively positive effect on macrofauna	
	Adverse effect on filtering benthic species due to increased suspended matter is not expected	(Jongbloed et al. 2006)
In winter	Adverse effect on the winter migration of juvenile *Macoma balthica* (Baltic tellin).	(Jongbloed et al. 2006)
	Chance of macrofauna to escape after burial is limited	
	Adverse effect on filtering benthic species due to increased suspended matter is not expected	(Jongbloed et al. 2006)
	Best option is nourishment during one winter season, starting after October and ending March, this is also the best season considering tourists	(Speybroeck et al. 2004)
In spring / summer	Macrofauna is more sensitive to burial	(Jongbloed et al. 2006)
Frequence		
Each year	The macrofauna at the site of deposition will not fully recover in this period	(Jongbloed et al. 2006)
Not each year	Better recovery of macrofauna	(Jongbloed et al. 2006)
	Intervals of at least three years are recommended between successive operations at a given site to enable recovery	(Menn et al. 2003)

3.3.8 Environmental regulations for nourishment

The North Sea coast is designated as Natura 2000 (N2000) site and therewith protected under the Habitats and Birds Directive. Besides the morphological component of Habitat type (H) 1110 (sandbanks which are covered by seawater all the time) and H1140B (mud and sand shoals: "wet beach"), there are also biological components, such as the bivalve species *Ensis*, which is an important food source of the N2000 protected bird species Common Scoter, Greater Scaup and Eider. Furthermore, the beach, intertidal zone and shallow coast are protected under the Water Framework Directive (WFD) up to 1 nautical mile from the coastline. Some effects from nourishment in relation to the European policies are described below.

H1140B can be affected by beach nourishments due to burial (and thus mortality) of local benthic organisms. In the N2000 area "North Sea Coast" each 6 year period (1990-2004) a total of >10% has been affected by beach nourishments (pers comm. Gerard Jansen). Upcoming plans for beach nourishments in this area (till 2016) will have the same dimensions. Related to N2000 goals, this habitat type will most probably not be affected more then in previous periods.
Nourishment affects H1110 by coverage of the seabed causing mortality of local benthos. Considering the size and locations of shoreface nourishment in the N2000 area 'North Sea Coastal Zone', nourishment affects only a small part of the total surface of H1110 in the area (Slijkerman et al. 2008). However, shoreface nourishments are expected to increase in future. For the purpose of the N2000 management plan of the North Sea coastal zone, the effects of nourishment on this habitat type will therefore be assessed in detail (Slijkerman et al. 2008). However, it is also noted that nourishment will eventually be beneficial for the conservation of sandbanks, because it keeps the sand balance and morphology in shape and it prevents the necessity of solid constructions.

According to the EU definition for reefs ("*Reefs can be either biogenic concretions or of geogenic origin. They are hard compact substrata on solid and soft bottoms, which arise from the seafloor in the sublittoral and littoral zone. Reefs may support a zonation of benthic communities of algae and animal species as well as concretions and corallogenic concretions*"), the tube dwelling polychaete *L. conchilega* can be classified as a reef builder (Rabaut et al. 2008). Reefs are listed under Annex I of the EC Habitats Directive EEC/92/43 as a marine habitat (Natura 2000 code 1170) to be protected by the designation of Special Areas of Conservation (SACs) (European Commission 2007). As shown in Figure 8, aggregations of *L. conchilega* occur mainly in subtidal areas, but intertidal aggregations close to the MLWS line do exist (Rabaut et al. 2008). Based on remote sensing images, the physical characteristics of these intertidal aggregations are assumed to be comparable with the subtidal ones.

The bivalve species *Spisula subtruncata* is an important food source for many protected bird species foraging in the coastal zone. In the Netherlands, Spisula occurs in highest densities around depths of 5 to 20 meters and hardly occurs in the surf zone. Shoreface nourishment between the two breaker bars will therefore hardly affect this species. The observed decline in Spisula since 2001 could not be related to nourishment activities (Leopold & Baptist 2007). It should be noted however, that recent nourishment practice tends to shift seawards behind the second breaker bar, where *Spisula* could occur in high densities. Potential effects on *Spisula* can therefore not be excluded.

Besides bivalves, fish is also an important food source for protected bird species. Impacts on surf zone fish following beach nourishment operation were observed as 'localized attraction' (northern kingfish) and 'avoidance' (bluefish) (Wilber et al. 2003). Observed effects of nourishment on benthos suggested that nourishment may have caused the loss of a full season of energy transfer to surf fishes at the nourished sites (Peterson et al. 2000). However, considering the quick recovery of benthos and the relatively small size of nourishment areas in comparison to the coastal feeding ground habitat, the risk of shoreface nourishment to demersal coastal zone fish has been considered negligible (Essink 1997). It is however noted that this only implies for the spatial scales that have been studied (nourished areas between 0.24 and 1.7 km^2).

Nourishment projects in the Netherlands, which are conducted to maintain the stability of the Dutch coast and ensure public safety, are not subjected to a permit regime. In contrast, the United States enforces beach nourishment projects to permit requirements. This includes monitoring of biological resources on the beach and at the dredging site. Despite decades of monitoring in the US, much uncertainty persists about the ecological impact of nourishment and how to minimize and mitigate them (Peterson & Bishop 2005). A total of 46 beach monitoring studies conducted in the US, of which most are unpublished, have been subjected to a quality evaluation (Peterson & Bishop 2005). It was concluded that much uncertainty surrounding biological impacts of beach nourishments can be attributed to the poor quality of monitoring studies. The responsible agency was therefore recommended to change their process.

4 Biogeomorphology

Authors: J.E Tamis & M.J. Baptist (Wageningen IMARES)

4.1 Introduction

Wave energy associated with sediment particle size and tidal range has been emphasized as a major structuring force for beach morphology, and for the infaunal communities of these habitats (e.g. (Brown & McLachlan 1990, Menn 2002); Figure 20). Thus, shore morphodynamics may considerably influence the biotic beach system which, in turn, can influence the physical nature of the beach (Menn 2002). Benthic (sea-bed) organisms may be classed as 'ecosystem engineers' or 'bio-engineers' in that their activity has a profound effect on their environment, resulting in a significant alteration of the sediment properties (Mazik et al. 2008).

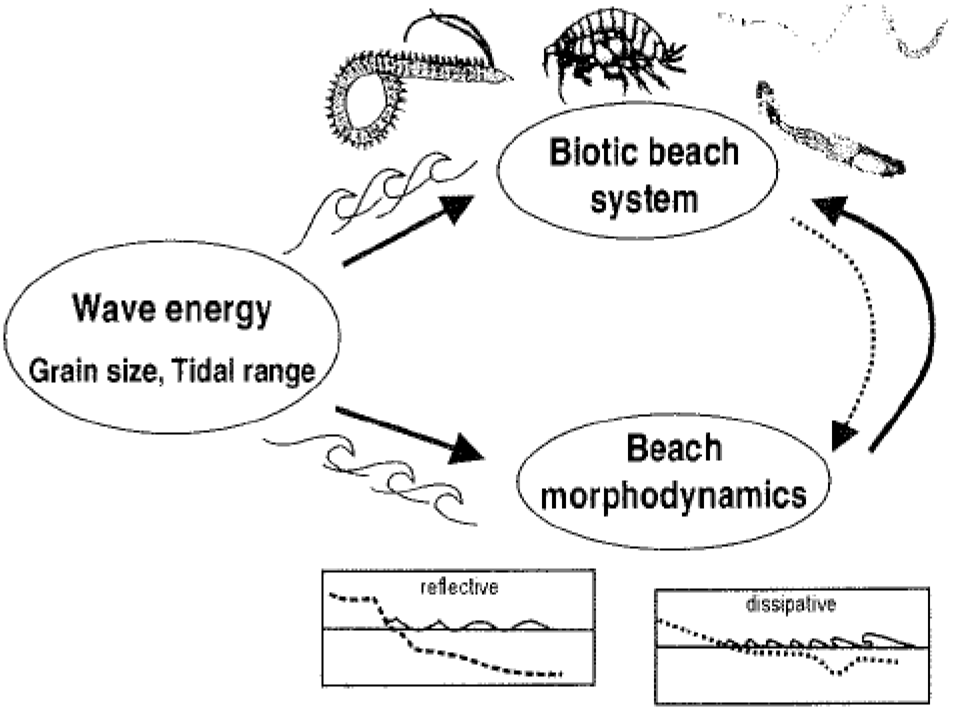

Figure 20 Schematic model of interactions between wave energy, beach morphodynamics and the biotic components on sandy shores (Menn 2002).

Many studies have indicated that biological communities can alter the geotechnical properties of marine sediments, and can therefore impact on the geomorphology of the resulting bedforms (Murray et al. 2002). Several implications following this relationship are described (Murray et al. 2002):

- The importance of community ecology in sedimentological and geomorphological processes. For example, if within a community, one species is replaced by another species and these species differ in their geomorphological impact, the large-scale outcome will be radically different. Some species known to stabilise sediment are the mud-burrowing crustacean *Corophium volutator*, the burrowing polychaete worm *Nereis (Hediste) diversicolor* and the polychaete worm *Lanice conchilega* producing organic cement in tubes;
- The energetics of sedimentary and geomorphological processes. Geomorphological processes are dependent on the balance between driving forces and resisting forces acting on sediment. By incorporating the biomechanical activity of fauna into such processes, it naturally follows that the

chemical energy stored in the tissues of organisms becomes available to do sedimentological or geomorphological work.

Main processes involved with (de)stabilising of sediment by marine organisms are:

- Creation of biogenic sediment (or biological armouring), i.e. skeletons and shells of marine organisms (Murray et al. 2002, Peletier & Janssen 2004), for example mussel- and oyster beds;
- Biological sediment stabilization, i.e. by microbial pads, suspension feeders (beds and reefs), rooting plants (sea grass) (Peletier & Janssen 2004) and animals tubes (Murray et al. 2002), for example mucus secretion of some crustaceans and worms;
- Bioturbation activity by sediment reworkers, i.e. passage - and living tracks, deposit feeders, food relationships and larger predators and grazers (Peletier & Janssen 2004), biogenic bottom features (Murray, Meadows et al. 2002).

4.2 Biogenic structures

Interaction with the hydrodynamics of the sediment–water interface produces altered patterns of sediment erosion, transport and deposition (Murray et al. 2002). Many benthic organisms modify the near-bed flow regime due to the biogenic structures they generate (Friedrichs & Graf 2008). It was shown that these flow interactions follow general trends, but each type of structure also produces individual flow characteristics (Friedrichs & Graf 2008). The resulting flow alterations are expected to enhance the supply of food particles or the exchange of gases, solutes and spawn of benthic organisms with the surrounding water. The authors conclude that this impact on the flow regime also strongly affects substratum stability in soft-bottom environments, i.e. deposition or resuspension of sediment particles. They suggest that passive flow effects generated by benthic biogenic structures need to be taken into account for future studies on the feeding behaviour and bio-engineering capacities of organisms living on the sediment–water interface.

In addition to the wave and current energy, driven by meteorological forces, the roughness of the sea floor determines the shear stress velocity which influences the particle dynamics at the sea bottom (Bobertz et al. 2008). The roughness is not only depending on the grain size of the sediments and sediment bed forms (e.g. ripples) but also by the benthic (macro) flora and fauna (Friedrichs & Graf 2008). Thus, in order to parameterise the sea bed roughness one has to take into account abiotic as well as biotic factors (Bobertz et al. 2008). The biotic effect on sea bed roughness was estimated, based on four key species forming dominant benthic structures in the area of investigation (south-western Baltic Sea), see : *Arenicola marina* (worm, hill like burrows), *Lagis koreni* (worm, protective tube, horizontal), *Mytilus edulis* (mussle, cluster), *Pygospio elegans* (worm, sward like tubes, vertical). Although the area of investigation does not overlap with our study area, the species were also found on the Dutch coast in 2002 (Janssen & Mulder 2004).

4.3 Biological sediment stabilization

Organisms that play a primary role in stabilizing processes are microbial pads, single celled bottom diatoms and fungi (Peletier & Janssen 2004). For example, organisms that have found to stabilize the sediment slope are *Pseudomonas atlantica* (eubacterium) and *Penicillium chrysogenum* (fungus). The two primary mechanisms underlying the role of microorganisms and fungi in sediment erosion processes, are thought to be soft-cementation by extracellular polymeric material (ECPM) and entanglement by filamentous growth (Murray et al. 2002). Although relatively much is known about the presence of these species and their effect on sediment stability in the tidal area, hardly anything is known about the subtidal. Furthermore, because the focus of this study is on the macrobenthos, these processes are not further discussed in this report.

Filter feeders are common in coastal areas. Filter and suspension-feeding benthic animals deposit faeces and pseudofaeces in and on the surrounding sediment; so-called biodeposition (Murray et al. 2002). They are able to modify their environment and create different habitats. With the catchment of sediment and occurrence in high densities these filter feeders enhance sediment stabilisation. Noticeably, polychaetes and bivalves are two major groups of marine fauna that produce sediment biodeposition as a result of their filter-feeding, and are widely distributed in both soft and hard bottom environments. In soft bottom environments, geomorphological effects such as increased sediment accretion and the development of raised areas of the sea floor would be expected (Murray et al. 2002).

Deposit feeders also have a positive influence on sediment stabilization. The excretion products of crustaceans such as *Corophium volutator* and the worm *Nereis diversicolor* stabilize the sediment. The excretion products exist of 1 – 2 µm wide treads which bind the sediment particles together (Peletier & Janssen 2004). The effect of *C. volutator* and *N. diversicolor* on the stability of sediment was studied by (Meadows & Tait 1989). Permeability decreased with increasing density of *C. volutator*, but increased with increasing density of *N. diversicolor*. Water content decreased with increasing density of both species. Shear strength increased with increasing density of both species, but more so with *C. volutator* than with *N. diversicolor*. Mixed-species effects on shear strength were additive, but some of the permeabilities were lower than predicted. Shear strength was negatively correlated with water content. The correlations between shear strength and permeability and between permeability and water content were affected differently by the two species. The mechanism of increased shear strength development with *C. volutator* must therefore be different to that for *N. diversicolor*, since it is not associated with an increase in permeability. It may involve direct "soft-cementation" by *C. volutator* mucus (Murray et al. 2002). Besides affecting sediment by excretion products, there are other processes involved between *C. volutator* and the sediment. Sediment made available by burrowing activities of *C. volutator* could lead to sediment transport. Open burrows can also act as passive sediment traps (Murray et al. 2002).

The relation between the critical bed shear stress, the macrozoobenthos density and the amount of diatoms has been used as a connection between biology and morphology and demonstrated in a case study for the Molenplaat (Western Scheldt estuary) (Holzhauer 2003).

Benthic communities are intimately involved in slope stability processes, both qualitatively and quantitatively. Furthermore, the resultant morphology of sedimentary structures produced by failure is a function of the composition of the faunal, microbial and macroalgal communities within the deposit. Marine sedimentary communities in significant densities are hence potentially very important in controlling the occurrence and geometry of large-scale geomorphological features resulting from slope failure, often on scales of many kilometers (Murray et al. 2002). A UK study has shown that *N. diversicolor* and *C. volutator* are able to increase the initial failure angle of intertidal sediments by around 12° to 16° (Murray et al. 2002). Mucus cementation and pile reinforcement could be the responsible mechanisms.

The constructed tubes of some infaunal species that protrude out of the sediment into the water column can either increase or decrease local erosion. Where tubes occur in low densities, local erosion is likely to occur. In contrast, where tubes are abundant, sediment accretion is likely to occur. Dense stands of animal tubes protruding from the sediment surface can protect the deposit from erosion by so-called skimming flow, in which the water passes over but not through the stand. The significance of the relative abundance of tube-building species on large-scale geomorphological processes is therefore very important. As the numbers of animal tubes increase, the effects of the tubes on large-scale processes change from being erosional to being depositional at some critical tube abundance (Murray et al. 2002).

"Hard" (brittle) organic matter bonding occurs, for example, the biologically produced organic cement in some polychaete worm tubes, such as *Lanice conchilega* (Murray et al. 2002). The proportion of "soft" to "hard" organic matter bonding in sediment could influence the sediments liquid–plastic–brittle transitions, which in turn would have significance for large-scale phenomena, like creep and slope failure. The presence of these structures (hard cemented vertical tubes) may also act as "pile reinforcement". Reinforced sediment by animal tubes would be stabilised (by rigid vertical structures) against mass failure and therefore, the incidence of slope failure would be reduced. A recent study has investigated the physical and temporal characteristics of high density aggregations of *L. conchilega* (Rabaut et al. 2008). The elevation and sediment consolidation of the biogenic mounds of *L. conchilega* was found to be significantly higher compared to the surrounding unstructured sediment. Elevation of the sediment occurs with relatively low *L. conchilega* densities (500 ind m2) and remains the same for increasing densities. Shear strength (indicating sediment stability) inside the aggregations is far higher than immediately outside the aggregations (Figure 21). A clear correlation also appears between the shear strength and the densities of *L. conchilega* tubes. Furthermore, the presence of adult *L. conchilega* changes the hydrodynamic pattern on a very local scale which leads to a settling advantage for juvenile *L. conchilega*. It is therewith suggested that there is a relation between the density of the aggregations and the longevity, as the chance of being renewed with juveniles is higher for high density aggregations.

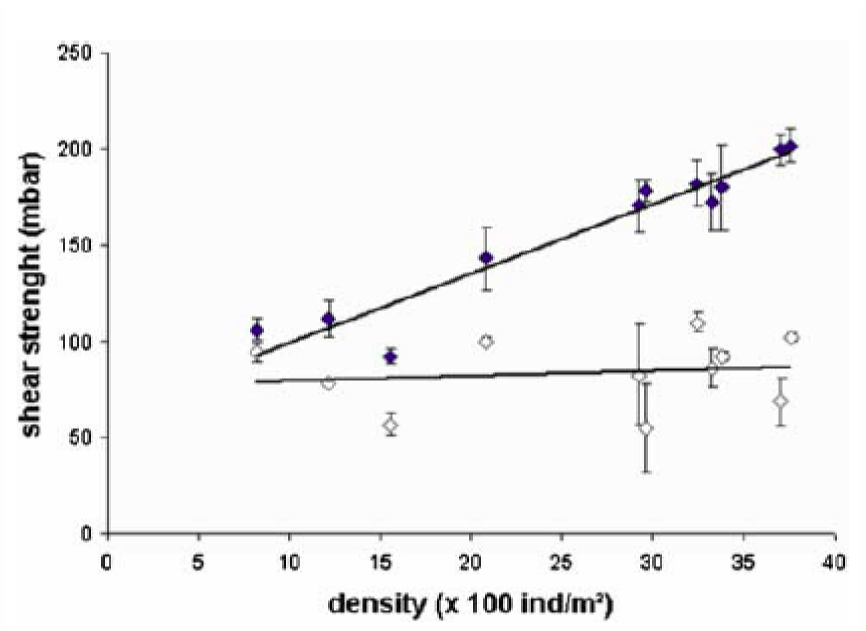

Figure 21 *Shear strength inside the aggregation (filled diamond) and immediately beside each aggregation (open diamond). Visualized error bars are standard errors, calculated on five replicate measurements per aggregation (Rabaut et al. 2008).*

4.4 Bioturbation

The importance of the impact of bioturbation by infaunal organisms, and the biogenic structures they produce, on marine sediment properties and biogeochemical cycling is well known (Mazik et al. 2008). The burrowing activity of benthic invertebrates modifies the physical properties of the sediment in terms of (Mazik et al. 2008):

- particle size distribution,
- porosity and permeability
- compaction,
- surface roughness
- cohesion and adhesion between particles

Biodestabilisation can be regarded as a form of natural disturbance (Van Moorsel 2005). This disturbance can be caused by burrowing organisms (i.e. bioturbation), such as the worm *Nephtys cirrosa*, the shrimp *Crangon crangon*, the crustaceans *Bathyporeia spp*. and *Corophium volutator* and the echinoderm *Echinocardium cordatum* (heart urchin). The lugworm (*Arenicola marina*) disturbs the sediment when feeding. The mud snail (*Hydrobia ulvae*) destabilises the diatom-film and could enhance therewith the erosion of sediment if present in sufficient desnities.

Many burrowers, such as *Nereis* spp., are responsible for sediment mixing, resulting in localised erosion (Murray et al. 2002). Sediment mixing by biological activity can also alter sediment elasticity, as measured by the geoacoustic properties of the sediment (Murray et al. 2002). This has implications for the relationships between bioturbation, sediment geotechnics and geoacoustic properties. In addition, changes in elasticity of the sediment, resulting from biological activity, will also affect wave energy attenuation by the sediment (Murray et al. 2002).
Many soft-bodied benthic invertebrates move vertically and horizontally in sediments, generating considerable pressures within the sediment, which can lead to differential compaction of the sediment fabric (Murray et al. 2002). The magnitude of the compaction effect depends on the activity rates, abundance and biomass of the animals responsible. At low animal densities, local strengthening of sediment due to compaction is expected to result in patchy erosion under appropriate conditions. Sediment mixing by benthic animals can also produce a winnowed deposit. This mixing will continuously bring to the surface fine sediment, which water currents may then erode. The net effect of biogenic mixing by bioturbation would be an increase in sorting and eventual erosion of

the deposit. At high animal densities, local strengthening of sediment due to compaction overlap and result in much less erosion and greater stabilisation.

Biogenic bottom features are determined largely by bioturbation activity, which modifies the topography of the sediment–water interface, particularly when caused by crustaceans, fishes and large polychaetes. The mounds produced by bioturbation are found to be not eroded, possibly by microbial colonization of the mound sediment protecting it from erosion (Murray, Meadows et al. 2002). Some animals secrete biogenic features directly and others use sediment as the building material.

4.5 Relations between ecology and morphology

The tables below present an overview of the relations found between morphological- and ecological aspects of sediment. Table 18 shows some relations between morphology and ecology and Table 19 shows some relations found between nourishment factors and ecological effects. Most of the threshold levels that are provided represent conservative estimations. When these thresholds are exceeded, short-term effects are expected to occur: a reduction of abundance and biomass of species.

Table 18 Relations between morphology and ecology

Morphological aspects	**Ecological aspects**	**References**
Shear strength	Density of *Lanice conchilega*	(Rabaut et al. 2008)
Distance to shore / depth	Ecological zones / habitat units (species composition and abundance)	(Janssen & Mulder 2004) (Janssen et al. 2008)/ (Slijkerman et al. 2007)
Grain size and slope	Number of species	(Brown & McLachlan 1990, Janssen & Mulder 2005)
Roughness of the seafloor	Abundances (ind./m^2) of four key species (*Arenicola marina*, *Lagis koreni, Mytilus edulis, Pygospio elegans*)	(Bobertz et al. 2008)

Table 19 Relations between nourishment factors and ecological effects

Nourishment factors	**Ecological effects**	**Parameter**	**Threshold or guideline**	**Reference**
Thickness of layer, deposition time and oxygen reduction	Burial of benthic species and suffocation of nearby benthos (mortality)	Layer thickness (incidental deposition) Oxygen level	For incidental deposition: 1 cm layer thickness; 20% reduction of integrated oxygen concentration	(Smit et al. 2006)
Quality of sand and application method	Increased turbidity, possibly affecting primary production and/or the foraging success of predating fish and birds	Turbidity/SPM	10 mg/l (behavioural change, i.e. avoidance, by Mackerel)	(Van Dalfsen 1999)
	Mortality of fish larvae	SPM	100 mg/l	(Van Dalfsen 1999)
	Habitat change / impact on benthos	Change in grain size	53 µm change in grain size	(Smit et al. 2006)
Location	Impact on benthos	Depth and distance to shore	None, parameter determines the affected ecological zone	(Janssen & Mulder 2004, Slijkerman et al. 2007, Janssen et al. 2008)
Frequency of nourishment	Recovery	Nourishment/year at the same site	Minimum of 3 years	(Menn et al. 2003)
Time of nourishment	Impact and recovery	Season / month	Best option is during winter season	(Speybroeck et al. 2004, Jongbloed et al. 2006)

5 On the interactions between benthos and sediment dynamics in the foreshore

Author: B.W. Borsje[a,b]
[a] *Water Engineering and Management, University of Twente, P.O. Box 217, 7500 AE Enschede, The Netherlands; b.w.borsje@utwente.nl*
[b] *Deltares, Marine and Coastal Systems, Rotterdamseweg 185, P.O. Box 177, 2600 MH Delft, The Netherlands; bas.borsje@deltares.nl*

5.1 Introduction

The interaction between benthos and sediment dynamics in the intertidal area has been exhaustively documented (e.g. Widdows and Brinsley, 2002, and references therein). They show that benthos are able to influence the strength of the bed by several orders of magnitude, and thereby have a significant influence on the sediment dynamics on a large spatial and temporal scale, either by stabilizing or destabilizing the sediment (Borsje et al., 2008a). Next to this, benthos are also present in the foreshore (Heip et al., 1992; Künitzer et al., 1992; Rabaut et al., 2007). Given the large biological influences on sediment dynamics known in the intertidal areas, researchers search for similar interactions in the North Sea (Borsje et al., 2008b).

In order to examine the hypothesis that biogeomorphological interactions occur in the foreshore, we need a tool to incorporate biological activity into models. The aim of this Chapter is to propose a parameterization in which biological activity is expressed in physical parameters.

5.2 Parameterization of biological activity

Three benthic species are included in the parameterization on the basis of (i) their abundance in the North Sea, (ii) their strong modification of the environment they are living in, and (iii) their contrasting type of feeding and burrowing, and thereby contrasting influence on the sediment and fluid dynamics. The three species selected are *Lanice conchilega*, *Tellina fabula* and *Echinocardium cordatum*.

The interaction between the selected benthos and the environment is schematized in Figure 22. The tube building worm *L. conchilega* protrudes several centimeters from the sediment in the water column, and thereby influences the near-bottom flow. For dense tube assemblages the near-bottom flow reduces, fine sediment will deposit and consequently lower ripples are present (Figure 22B), compared to the default case (Figure 22A). Due to the digging and feeding activities of the bivalve *T. fabula* up to 10 cm deep in the sediment, the properties of the surficial sediment are modified and the sediment is more prone to erosion (Figure 22C). Finally, the sea urchin *E. cordatum* lives in the top 20 centimeters of the bed and mixes sediment in vertical direction (Figure 22D), resulting in relatively coarser sediment in the top layer of the bed.

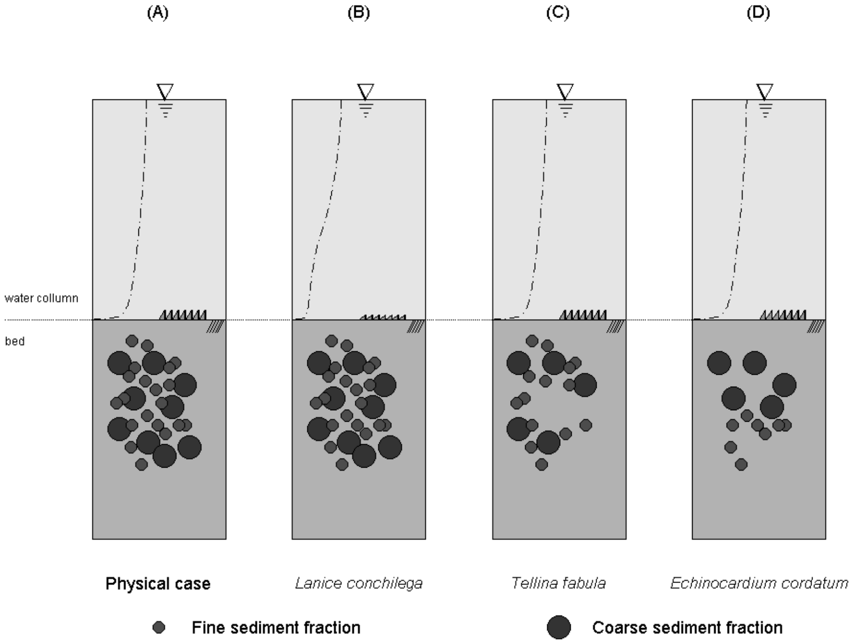

Figure 22. Schematized interaction between the selected benthos and their environment. For the default case (A), the sediment fractions are normally distributed over the bed, ripples are present at the sediment-water interface and the flow velocity profile represents the normal case. Lanice conchilega (B) reduces the near bottom flow, and hence the ripples are lowered. Tellina fabula (C) destabilize the sediment and thereby increases the pore volume. Echinocardium cordatum (D) redistributes the sediment, resulting in a coarser surface layer and a finer sub-surface layer.

5.2.1 Tube building worm – Lanice conchilega

To model the influence of *L. conchilega* on the near bottom flow, we represent the tube building worm by thin piles on the bottom of the seabed. In this way, we are able to include the worms in a vegetation model (Uittenbogaard, 2003). This vegetation model is able to calculate the turbulent flow over and through vegetation (thin piles) in water of limited depth. The vegetation model explicitly accounts for the influence of cylindrical structures on drag and turbulence by an extra source term of friction force in the momentum equation and an extra source term of Total Kinetic Energy (TKE) and turbulent energy dissipation in the k-ε equations respectively. For a detailed mathematical description of the vegetation model see Bouma et al., (2007).

Validation of the model outcome is done for two cases. In both cases *L. conchilega* was mimicked by artificial tubes placed in a regular pattern (Figure 23), and flow velocity profiles in front of the tube field and after the tube field were compared. The first case consists of a flume experiment with an abundance of 350 ind. m^{-2}, a free stream velocity of 0.1 m s^{-1} and a tube height of 10 cm. The modeled flow deceleration of 56% corresponds well with measured reduction of flow velocity of around 60% at a height of 1,5 cm above the sediment surface (Friedrichs, 1997). The second case consists of a more recent flume experiment (Friedrichs et al., 2000), with a free stream velocity of 0.05 m s^{-1}, a variation in the abundance of 490 to 3836 ind. m^{-2} and a tube height of 3,5 cm. Due to the fact that the height of the bottom boundary layer (~ 3 cm) was comparable to the height of the tubes, Friedrichs et al. (2000) discuss that the results may vary with different experimental settings, but that the results are qualitatively correct.

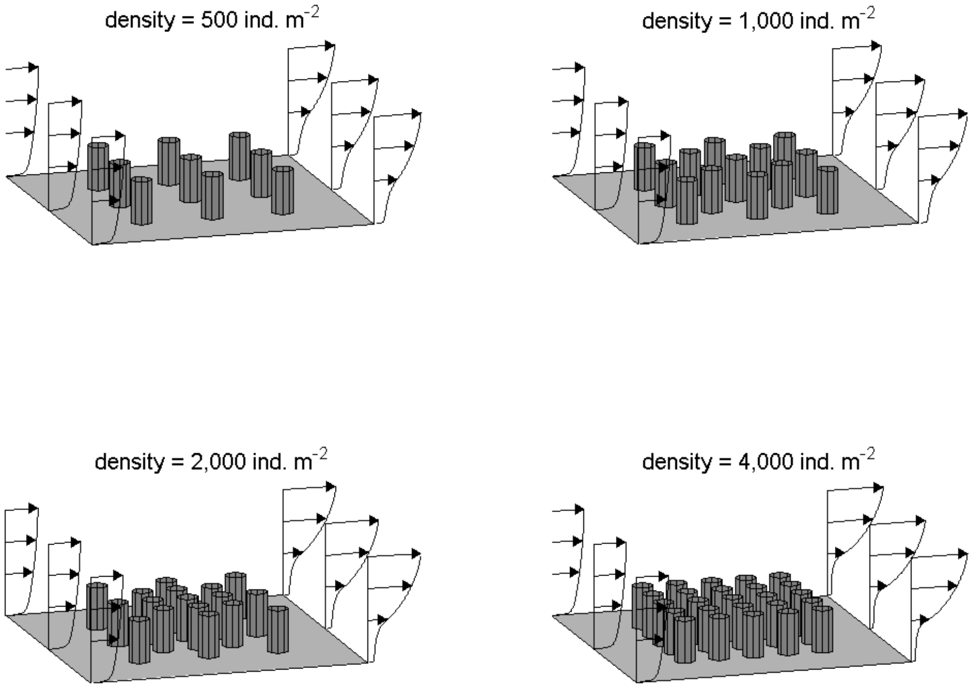

Figure 23 Schematic overview of the model set-up to determine the reduction of the flow velocity by Lanice conchilega. Flow velocity profiles correspondent to the indicated densities.

Nevertheless, the reduction in the near bottom flow velocity modeled with a tube height of 10 cm and a free stream velocity of 0.05 m s^{-1} shows comparable results with the measured flow deceleration in the flume (Figure 24). Comparing these flume and model experiments is acceptable, while the population density is the main determinant in the deceleration of the flow (Nowell and Church, 1979). Population densities used in the flume experiments are comparable to densities found in the field for *L. conchilega*.

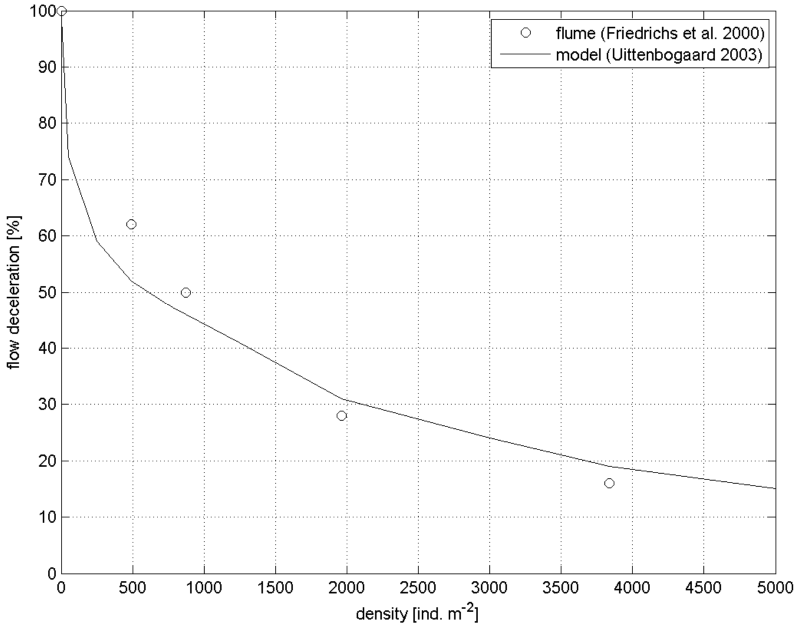

Figure 24 Comparison between the flow deceleration by Lanice conchilega for different densities, mimicked in a flume experiment by thin piles (Friedrichs et al., 2000) and model results, by adopting the vegetation model by Uittenbogaard (2003).

The flow declaration by *L. conchilega* will reduce the ripple height both directly and indirectly. These ripples are the main origin of the sea bed roughness (Soulsby, 1983), and have a height of a couple of centimeters.
Directly, by a decrease in energy and indirectly due to deposition of fine particles in the tube fields and consequently lower ripples. Moreover, another indirect effect is the augmentation of the benthic community with the presence of *L. conchilega* (Rabaut et al., 2007). These bio-engineers burrow and crawl through the top layer of the sediment and in this way break down the ripples. Reduction of the ripple height in the field is site specific (local sediment sorting, amount of suspended sediment and abundance of burrowing and crawling species), and therefore difficult to express in general terms. However, following the empirical relations derived by O'Donoghue et al., (2006), the ripple height will reduce to 60%, given a reduction of the near bottom flow to 30% of the original near bottom flow velocity. A reduction of 70% of the bottom flow is chosen to represent the maximum density of *L. conchilega* found in the North Sea area (Borsje et al., 2008b). Assuming a biological factor of 10%, this will result in a maximum reduction of the ripple height with 70%.

5.2.2 Bivalve – Tellina fabula

Data on the bio-engineer capacity of the bivalve *T. fabula* are scarce, but the sediment modification by the bivalve *Macoma balthica* is much better known. Both bivalves have comparable feeding strategies (selective deposit as well suspension feeding). However, *M. balthica* is mostly found in muddy sediments, while *T. fabula* prefers fine sediments. Therefore, the distribution of *M. balthica* is much more bordered close to the coast, in contrast to *T. fabula* which can be found in all other parts of the North Sea. Nevertheless, the relation used for *M. balthica* to parameterize the sediment destabilization is a good alternative to model the bio-sediment interactions for *T. fabula*. The biomass of the bivalve is related to the critical bed shear stress by a destabilizing factor (T_d), where τ_{cr} is the critical bed shear stress for erosion. The superscript '0' for the critical shear stress represents the values without the influence of biological activity. The destabilizing factor is defined by Borsje et al. (2008a), based on field experiments by Austen et al. (1999), and quantitatively shown in Figure 25.

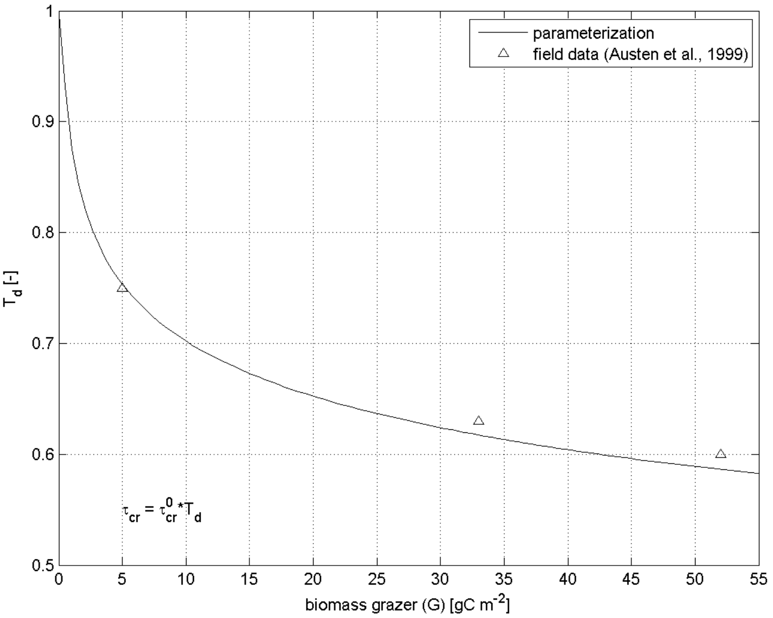

Figure 25 *Destabilization of the bed by Tellina fabula, expressed in a destabilization factor (Td) on the physical critical bed shear stress (τcr0), adapted from Borsje et al. (2008a).*

5.2.3 Sea urchin – *Echinocardium cordatum*

E. cordatum causes a change in sediment distribution in the bed, resulting in a relatively coarser layer at the sediment water interface, and relatively finer layer of sediment underneath this layer.
E. cordatum is regarded as a non-selective deposit feeder (Lohrer et al., 2005). However, due to two reasons, *E. cordatum* will cause a heterogeneous sediment distribution in the top centimeters of the bed. Firstly, because finer particles have a relative larger surface area and have therefore a larger chance to get ingested and brought downward. Secondly, fine sediment is richer in organic matter compared to coarse sediment (Bureno et al., 2003), and therefore *E. cordatum* moves to another spot, while relative larger particles are not ingested and are still at their original position (Cramer et al., 1991). To parameterize the transport of particles from the sediment surface deeper into the sediment by *E. cordatum*, we adopt an active layer concept. In this concept, the probability of entrainment of a particle is defined in a step function, for which the probability of entrainment of a particle has a constant value in an active layer near the bed surface, and vanishes below this layer (Hirano, 1971). The layer underneath the active layer is called the substrate, which is physically covered by the active layer. As a result, the grain size distribution in both layers can be assigned differently. By adopting an active layer thickness which is equal to the area of influence by *E. cordatum* the top layer can be modeled as a bio-turbated layer, while the substrate can be modeled as a non bio-turbated layer. Based on an experimental study for *E. cordatum* in New Zealand, Lohrer et al., (2005) found that *E. cordatum* displaces up to 20,000 cm^3 m^{-2} d^{-1}, suggesting that surface sediment is reworked about every 3 days at sites where *E. cordatum* is abundant.

The heterogeneous vertical sediment distribution is shown in Figure 26, in which the thickness of the bio-turbated layer (l_{bio}) and the multiplication factor for the medium grain size needs to be imposed. Based on measurements of *Arenicola marina* (Baumfalk, 1979), which is a comparable non-selective deposit feeder, the multiplication factor could reach values up to 2, meaning a medium grain size twice as large, compared to the default case. The thickness of the bio-turbated layer could reach values up to 0.2 m (Holtmann et al., 1996).

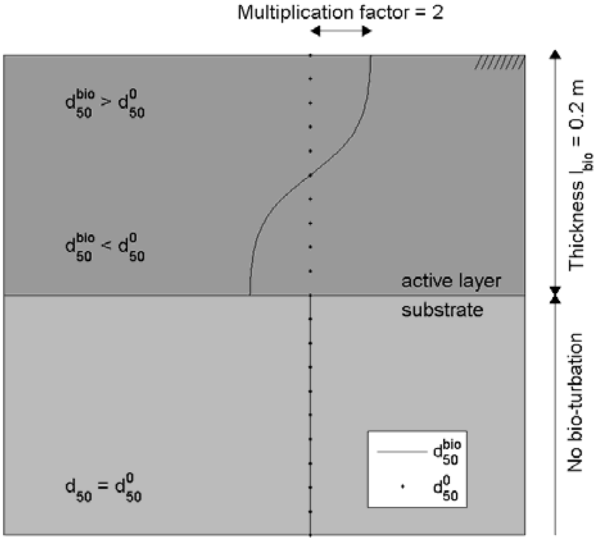

Figure 26 Schematic overview of the influence of Echinocardium cordatum on the grain size distribution in the bed. Resulting in a larger grain size in the surface layer and a smaller grain size in the sub-surface layer (d50bio), compared to the physical case (d500). In the substrate, no bio-turbation occurs.

5.3 Implications for the stability of a nourishment

The parameterization described in the previous sections gives us a first insight in the potential impacts of benthos on the stability of a nourishment. Moreover, given the recoverability potential of *Lanice conchilega*, *Tellina fabula*, and *Echinocardium cordatum* we are also able to discuss the feedback from the nourishment to the recovery of benthos.

5.3.1 Stability of the nourishment

As discussed in Section 3.2.1, some communities can be linked to a certain region along the Dutch coast. For example, *Tellina fabula* is mainly found near the Wadden coast. As a result, a nourishment near the Wadden coast will possibly be destabilized by *Tellina fabula*. However, *Echinocardium cordatum* is able to stabilize a nourishment. The spatial distribution of *Echinocardium cordatum* is very abundant along the Dutch coast. Therefore, it is difficult to say whether a nourishment will be destabilized or stabilized. Nevertheless, by knowing the dominant community before a nourishment, we are able to predict the stability of a nourishment.

Due to a nourishment, also the slope and grain size can be influenced. As discussed in Section 3.2.3, the number of species and their abundance both increase as the grain size and slope decrease. Therefore, it is important to be able to predict the habitat of a certain species and thereby the influence on the stability of a nourishment.

Finally, the roughness of the seafloor can be influenced by a nourishment. The ripples in a *Lanice conchilega* field are significantly lower, compared to the default case. As a result of a nourishment, a *Lanice conchilega* field can be buried, resulting in a rougher seabed, and consequently a less stable seabed.

5.3.2 Recovery of the benthos after a nourishment

As discussed in Section 3.3.6, the recoverability of the parameterized benthos is high, which means that full recovery will occur, and should be complete within about five years. Given the recoverability of *Echinocardium cordatum* (about five years), the stabilization by this species is negligible, while most nourishments are assumed to be effective less than five years (Chapter 2). On the other hand, stabilization of a nourishment by *Lanice conchilega* can be much more effective, given the stability of the tube fields, and the high recoverability of this species.

6 Conclusions and recommendations

6.1 Morphological effects of nourishment in the Dutch coastal zone

The current (2000-2007) policy is to nourish yearly 12 Mm^3 sand in the Dutch coastal system, of which about 60% by means of relatively large (10^6 m^3) shoreface nourishments and 40% by means of relatively small (10^5 m^3) beach nourishments. The largest part (49%) of the 12 Mm^3 of nourished sand is put into the central Dutch coast (between Hoek van Holland and Den Helder); the Wadden and the southwestern coastal system receive 28% and 23%, respectively. The shoreface nourishments typically have a volume of 1-3 Mm^3 (400-600 m^3/m) and are usually placed against the outer breaker bar, at a water depth of 4-8 m.

The impact of the shoreface nourishments on the coastal system is the result of the following effects.

1. The lee effect; wave energy is dissipated on the nourishment as a result of which the alongshore sand transport capacity decreases, which results in sediment accumulation upstream and erosion downstream of the nourishment.
2. The feeder effect; the coastal system onshore of the nourishment is fed with nourished sediment due to cross-shore sand transport processes.

Due to these effects, the shoreface nourishment affects the autonomous behaviour of the breaker bars. The nourishment, placed against the outer breaker, generally re-shapes itself relatively quickly (within a few months) into a bar with a landward trough. As a result of this, the offshore migration of the original breaker bars is halted; sometimes they even start migrating in the onshore direction. During this stop of offshore bar migration, the bars keep their pre-nourished dimensions. The duration of the impact of a shoreface nourishment is finite and related to its lifetime, which typically lies in between 2 and 8 years. The lifetime is, among other things, dependent on the nourishment volume, the grain size, the alongshore nourishment length and the location of the nourishment. It seems as if the grain size at the nourishment location adjusts itself quickly (within a year). The pre-nourished grain size re-appears, since finer sand tends to be transported offshore and alongshore with the current and coarser sand onshore due to wave skewness effects. In general, most nourished sand is transported onshore in the end.

Shoreface nourishments display different morphodynamic behaviour at different locations in the Dutch coastal system.

- Typically for the Terschelling (1993) shoreface nourishment is its strong alongshore migration, which is not found for shoreface nourishments in the other coastal systems. This can be explained by the differences in orientation compared to the dominant offshore wave conditions.
- In the southwestern Dutch coastal system there are no breaker bars. The shoreface nourishments are generally placed on the slope of a tidal channel. After two years, the Oostgat channel slope nourishment does not seem to have changed much, which is in contrast to most other nourishments that are quickly absorbed by the coastal system.
- Typically for the central Holland coast is the consistent erosion south of the town Egmond aan Zee, despite shoreface and beach nourishments. This could be due to divergence of alongshore currents at this location.

Most knowledge on the morphodynamic behaviour of shoreface and beach nourishments originates from data-analysis studies. Numerical modeling tools have been used successfully in hindcasting behaviour of nourishments, but do net yet have the predictive power for reliably forecasting. With the exception of the Terschelling (1993) nourishment, the monitoring and data-analysis focused on morphological parameters relevant to coastal safety, especially the change of sediment volume in the nourished and adjacent areas in time. Investigating the ecological impact of a shoreface nourishment requires both monitoring additional physical processes/parameters (e.g. grain size, flow velocities, suspended sediment concentrations) as well as extracting other parameters from the bathymetric data (e.g. bed slope, thickness of nourished sand). These parameters should be chosen on the basis of their controlling influence on ecology.

6.2 Ecological effects of nourishment in the Dutch coastal zone

A number of aspects on the area affected by sand nourishment are important to consider when evaluating the ecological effects. The Dutch coastal zone is an area of a high fishing intensity, causing significant direct and indirect effects on habitat and on the diversity, structure and productivity of benthic communities. The benthos as now observed can therefore be considered as the product of several decades continuous disturbance by fishing activities. However, due to storms and wave action, coastal communities are well-adapted to a dynamic environment.

Since the 1980s, there have been several monitoring studies on macrobenthos and effects of sand nourishments in the Dutch coastal zone. Some of these involved site-specific projects and some are part of yearly monitoring programs.

The macrozoobenthos is a well-investigated group and consitst of molluscs (bivalves and snails); worms; spiny-skinned animals (Echinodermata); and crustaceans. Different ecological zones can be distinguished in the Dutch sandy coast, related to the distance to shore and depth. Also, the number of species in the coastal zone can be related to the grain size and slope.

Ecological effects of nourishment are usually short-term: a reduction of abundance and biomass of species. In general, recovery proceeds rather fast. For most of the species found under present-day conditions, abundance and biomass will largely recover ca. 1 year after completion of the nourishment. Long living species, such as bivalves (e.g. *Spisula subtruncata*, *Donax vittatus*) and sea urchins (*Echinocardium cordatum*) are expected to recover much slower. For these species, that do not reproduce successfully each year, recovery of total biomass and a normal age structure is considered to take 2-5 years.

Several studies indicate that the nourished sediment characteristics compared to original sediment have great influence on the ecological effects. The influence of grain size on the ecological effect of nourishment is important to consider in future nourishment projects.

High density aggregations of the sand mason *L. conchilega* seem to increase sediment consolidation, as a clear correlation between the shear strength of sediment and the densities of *L. conchilega* tubes has been determined. *L. conchilega* is a characteristic species of the ecological zone in the trough between the two breaker bars.

An important aspect to consider in predicting the ecological effect of future sand nourishments is the spatial and temporal distribution of nourished sand, especially when characteristics (e.g. grain size and mud content) differ from the original sediment. Spatial distribution of nourishment is related to the presence of species (ecological zones). For example, a relation exists between the number of species and the distance to shore. Temporal distribution is related to the reproduction cycle. Affecting species in spring or summer will have more impact because most species reproduce this time of year.

Although grain size has proven to be of great importance, both ecological as morphological, this parameter is not often measured within nourishment projects. It could be recommended to include monitoring of grain size, before and after nourishment as standard practice within nourishment projects. If not feasible, as a minimum data requirement, the grain size of the nourished sediment could be measured and reported, together with the grain size of the original sediment at the nourishment site. Other potential parameters to measure are layer thickness, oxygen level, turbidity/SPM, depth and distance to shore.

6.3 Interactions between benthos and sediment dynamics in the foreshore

We can conclude that bio-engineers may influence the stability of a nourishment significantly. *Lanice conchilega* and *Echinocardium cordatum* are able to stabilize a nourishment, by decelerating the near bed flow and increasing the grain size at the sediment water interface respectively. *Tellina fabula* will destabilize a nourishment, by moving through the sediment and in this way making the sediment more prone the erosion. However, to predict the stability of a nourishment we first need to know the site-specific species composition. More important, to get a general overview we need to know the change in species composition after a change in physical parameters (e.g. after a nourishment the grain size and the slope could be influenced and will this result in a new habitat?). Next to this, the parameterization in this chapter is based on flume experiments. To get real insight in the biogeomorphological interactions for the stability of nourishments we need to execute field experiments. During these field experiments it is recommended to monitor the recovery of benthos, the change in physical parameters (and corresponding habitat) and the behavior of the nourishment itself.

References

Chapter 2 Nourishment

Cohen, A.B., Brière, C., 2007. Evaluatie van de uitgevoerde suppleties bij Egmond op basis van Argus video waarnemingen, WL|Delft Hydraulics report Z4212, the Netherlands. (in Dutch)

Grunnet, N.M., Walstra, D.J.R., Ruessink, B.G., 2004. Process-based modeling of a shoreface nourishment, Coastal Engineering, 51: 581-607.

Hoekstra, P., Houwman, K.T., Kroon, A., Ruessink, B.G., 1996. Morphodynamic behaviour of the Terschelling shoreface nourishment, NOURTEC report, University of Utrecht, The Netherlands.

Kroon, A., Hoekstra, P., Ruessink, G., 1994. Morphological monitoring of a shoreface nourishment NOURTEC experiment at Terschelling, The Netherlands, Proceedings International Conference on Coastal Engineering, Kobe, Japan, pp. 2222-2236.

Meetadviesdienst Zeeland, 2008. Rapportage metingen/signalering vooroeversuppletie Oostgat.

Ojeda, E., Ruessink, B.G., Guillen, J., 2008. Morphodynamic response of a two-barred beach to a shoreface nourishment, Coastal Engineering.

Van Duin, M.J.P., Wiersma, N.R., Walstra, D.J.R, Van Rijn, L.C., Stive, M.J.F., 2004. Nourishing the shoreface: observations and hindcasting of the Egmond case, The Netherlands, Coastal Engineering, 54: 813-837.

Walstra, D.J.R., Van Ormondt, M., Roelvink, D.A., 2004. Shoreface nourishment scenarios. Detailed morphodynamic simulations with Delft3D for various shoreface nourishment designs, WL|Delft Hydraulics report Z3748.21, the Netherlands.

Walstra, D.J.R., C. Brière, A.B. Cohen, A.R. van Dongeren, I.J.P. Elshoff, C. Hoyng, M. van Ormondt, S. Quartel, B. de Sonneville, P.K. Tonnon, L. Uunk, 2008. Monitoring and Modelling of a surface nourishment. VOP Short Term Morphology. Research Report Z4479, WL|Delft Hydraulics, the Netherlands.

Chapter 1 Introduction; 3 Ecology; and 4 Biomorphology

Ager OED (2008) Lanice conchilega. Sand mason. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [on-line]. . Plymouth: Marine Biological Association of the United Kingdom. [cited 26/11/2008].

Allen KD, Hardy JW (1980) Impacts of navigational dredging on fish and wildlife: a literature review.

Baan PJA, Menke MA, Boon JG, Bokhorst M, Schobben JHM, Haenen CPL (1998) Risico Analyse Mariene Systemen (RAM). Verstoring door menselijk gebruik., Waterloopkundig Laboratorium, Delft.

Baptist MJ, van Dalfsen J, Weber A, Passchier S, van Heteren S (2006) The distribution of macrozoobenthos in the southern North Sea in relation to meso-scale bedforms. Estuarine, Coastal and Shelf Science 68:538-546

Bijkerk R (1988) Ontsnappen of begraven blijven. De effecten op bodemdieren van een verhoogde sedimentatie als gevolg van baggerwerkzaamheden., RDD Aquatic Systems.

Bilodeau AL, Bourgeois RP (2004) Impact of beach restoration on the deep-burrowing ghost shrimp, Callichirus islagrande. Journal of Coastal Research 20:931-936

Bishop MJ, Peterson CH, Summerson HC, Lenihan HS, Grabowski JH (2006) Deposition and long-shore transport of dredge spoils to nourish beaches: Impacts on benthic infauna of an ebb-tidal delta. Journal of Coastal Research 22:530-546

Bobertz B, Harff J, Bohling B (2008) Parameterisation of clastic sediments including benthic structures. Journal of Marine Systems doi:10.1016/j.jmarsys.2007.06.010

Brown AC, McLachlan A (1990) Ecology of sandy shores. Ecology of sandy shores

Brown AC, McLachlan A (2002) Sandy shore ecosystems and the threats facing them: Some predictions for the year 2025. Environmental Conservation 29:62-77

Buzzelli CP, Luettich Jr RA, Powers SP, Peterson CH, McNinch JE, Pinckney JL, Paerl HW (2002) Estimating the spatial extent of bottom-water hypoxia and habitat degradation in a shallow estuary. Mar Ecol Prog Ser 230:10

Colosio F, Abbiati M, Airoldi L (2007) Effects of beach nourishment on sediments and benthic assemblages. Marine Pollution Bulletin 54:1197-1206

Dankers N, Binsbergen M, Zegers K (1983) De effecten van zandsuppletie op de fauna van het strand van Texel en Ameland., Rijksinstituut voor Natuurbeheer.

Degraer S, Volckaert A, Vincx M (2003) Macrobenthic zonation patterns along a morphodynamical continuum of macrotidal, low tide bar/rip and ultra-dissipative sandy beaches. Estuarine, Coastal and Shelf Science 56:459-468

Essink K (1993) Ecosystem effects of dredging and dumping in the Ems-Dollard estuary and the Wadden Sea, RWS, RIKZ

Essink K (1997) Risk analysis of coastal nourishment techniques (RIACON). Final evaluation report. , Rijkswaterstaat, National Institute for Coastal and Marine Management/RIKZ.

Essink K (1999) Ecological effects of dumping of dredged sediments; options for management. . J Coastal Conserv 5:12

Essink K (2005) Bodemfauna en beleid. Een overzicht van 35 jaar bodemfauna onderzoek en monitoring in Waddenzee en Noordzee. Report No. RWS/RIKZ-2005.028, RIKZ, Haren

European Commission (2007) Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora (OJ L 206, 22.7.1992, p. 7)

Fanini L, Marchetti GM, Scapini F, Defeo O (2009) Effects of beach nourishment and groynes building on population and community descriptors of mobile arthropodofauna. Ecological Indicators 9:167-178

Friedrichs M, Graf G (2008) Characteristic flow patterns generated by macrozoobenthic structures. Journal of Marine Systems

Goudswaard PC, Kesteloo JJ, Perdon KJ, Jansen JM (2008) Mesheften (Ensis directus), halfgeknotte strandschelpen (Spisula subtruncata), kokkels (Cerastoderma edule) en otterschelpen (Lutraria lutraria) in de Nederlandse kustwateren in 2008, Wageningen IMARES

Greene K (2002) Beach Nourishment:A Review of the Biological and Physical Impacts, ASMFC (Atlantic States Marine Fisheries Commission)

Hill JM, . (2008) Echinocardium cordatum. Sea potato. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [on-line]. . Plymouth: Marine Biological Association of the United Kingdom. [cited 14/07/2008].

Hiscock K, Jackson A, Lear D (1999) Assessing seabed species and ecosystem sensitivities: existing approaches and development, Report to the Department of Environment, Transport and the Regions form the Marine Life Information Network (MarLIN), Marine Biological Association of the United Kingdom, Plymouth.

Holzhauer H (2003) Biogeomorphology. Small activities with large effects?, University of Twente, Department of Civil Engineering, Group of Water Engineering & Management

ICES (2000) Report of the working group on the effects of extraction of marine sediments on the marine ecosystem

Janssen G, Mulder S (2005) Zonation of macrofauna across sandy beaches and surf zones along the Dutch coast. Oceanologia 47:265-282

Janssen GM, Kleef H, Mulder S, Tydeman P (2008) Pilot assessment of depth related distribution of macrofauna in surf zone along Dutch coast and its implications for coastal management. Marine Ecology 29:186-194

Janssen GM, Mulder S (2004) De ecologie van de zandige kust van Nederland. Report No. RIKZ/2004.033, RIKZ, Haren

Jennings S, Kaiser MJ (1998) The effects of fishing on marine ecosystems Advances in Marine Biology:152

Jones AR, Murray A, Lasiak TA, Marsh RE (2008) The effects of beach nourishment on the sandy-beach amphipod Exoediceros fossor: Impact and recovery in Botany Bay, New South Wales, Australia. Marine Ecology 29:28-36

Jongbloed RH, Dankers NMJA, Brinkman AG, Van Dalfsen JA, Smit CJ, Tamis JE (2006) Effecten van storten van baggerspecie in het Marsdiep. Report No. C084/06, IMARES, Den Helder

Kaag NHBM (2004) Beschrijving macrofauna Buitenhaven IJmuiden (nulmeting), TNO

Kaag NHBM, Weber A, Lewis WE, van Dalfsen JA (2005) Ecologische monitoring kustverdedigingsproject Oostende (T0-situatie, fase2): zacht substraat, TNO

Kranz PM (1974) The anastrophic burial of bivalves and its paleoecological significance. . J Geol 82:29

Lankford TE, Baca BJ (1989) Comparative environmental impacts of various forms of beach nourishment Coastal Zone: Proceedings of the Symposium on Coastal and Ocean Management. Publ by ASCE, Charleston, SC, USA, p 2046-2059

Leopold MF (2002a) Nulmeting vogels en benthos in de Texelse vooroever (KUSTADV*NH), Alterra-Texel

Leopold MF (2002b) T-1 meting benthos en vissen in de Texelse vooroever (KUSTADV*NH), september 2002, Alterra-Texel

Leopold MF (2003) T-2 meting: vogels, vissen en benthos in de Texelse vooroever (KUSTADV*NH), februari 2003, Alterra-Texel

Leopold MF, Baptist MJ (2007) De effecten van onderwaterzandsuppleties op het habitat van de Kustzee, Spisula en enkele beschermde soorten zeevogels. Report No. C014/07, Wageningen IMARES, Texel

Lindeboom H, Geurts van Kessel J, Berkenbosch L (2005) Gebieden met bijzondere ecologische waarden op het Nederlands Continentaal Plat. , RWS

Maurer D, Keck RT, Tinsman JC, Leathem WA (1980) Vertical migration and mortality of benthos in dredged material. Part 1: Mollusca. Mar Environ Res 4:21

Mazik K, Curtis N, Fagan MJ, Taft S, Elliott M (2008) Accurate quantification of the influence of benthic macro- and meio-fauna on the geometric properties of estuarine muds by micro computer tomography. Journal of Experimental Marine Biology and Ecology 354:192-201

McLachlan A (1996) Physical factors in benthic ecology: Effects of changing sand particle size on beach fauna. Marine Ecology Progress Series 131:205-217

Meadows PS, Tait J (1989) Modification of sediment permeability and shear strength by two burrowing invertebrates. Marine Biology 101:75-82

Menn I (2002) Ecological comparison of two sandy shores with different wave energy and morphodynamics in the North Sea. Ber Polarforsch Meeresforsch 417

Menn I, Junghans C, Reise K (2003) Buried alive: Effects of beach nourishment on the infauna of an erosive shore in the North Sea. Senckenbergiana Maritima 32:125-145

Mulder S, Raadschelders EW, Cleveringa J (2005) Een verkenning van de natuurbeschermingswet-geving in relatie tot Kustlijnzorg. De effecten van zandsuppleties op de ecologie van strand en onderwateroever, RWS RIKZ

Murray JMH, Meadows A, Meadows PS (2002) Biogeomorphological implications of microscale interactions between sediment geotechnics and marine benthos: a review. Geomorphology 47:15-30

Nordstrom KF (2005) Beach nourishment and coastal habitats: Research needs to improve compatibility. Restoration Ecology 13:215-222

Oranjewoud (2006) Verkennend onderzoek naar kennisleemtes in de ecologie van de kustzone, Oranjewoud

Peletier H, Janssen GM (2004) De levende natuur als ecosysteemvormer in kustgebieden. Report No. RIKZ/2004.005, RIKZ, Haren

Perdon KG, Goudswaard PC (2007) Mesheften (Ensis directus), halfgeknotte strandschelpen (Spisula subtruncata) en kokkels (Cerastoderma edule) in de Nederlandse kustwateren in 2007, IMARES

Perdon KJ, Goudswaard PC (2006) De Amerikaanse zwaardschede, Ensis directus, en de halfgeknotte strandschelp, Spisula subtruncata, in de Nederlandse kustwateren in 2006, Wageningen IMARES

Peterson CH, Bishop MJ (2005) Assessing the environmental impacts of beach nourishment. BioScience 55:887-896

Peterson CH, Hickerson DHM, Johnson GG (2000) Short-Term Consequences of Nourishment and Bulldozing on the Dominant Large Invertebrates of a Sandy Beach. Journal of Coastal Research 16:368-378

Rabaut M, Vincx M, Degraer S (2008) Do Lanice conchilega (sandmason) aggregations classify as reefs? Quantifying habitat modifying effects. Helgoland Marine Research:1-10

Rakocinski CF, Heard RW, LeCroy SE, McLelland JA, Simons T (1996) Responses by macrobenthic assemblages to extensive beach restoration at Perdido Key, Florida, U.S.A. Journal of Coastal Research 12:326-353

Rayment WJ, . (2008) Fabulina fabula. Bean-like tellin. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [on-line]. . Plymouth: Marine Biological Association of the United Kingdom. [cited 14/07/2008].

Renschler CS, Doyle MW, Thoms M (2007) Geomorphology and ecosystems: Challenges and keys for success in bridging disciplines. Geomorphology 89:1-8

RIKZ (1999) Milieu-effectrapport voor de winning van beton- en metselzand op de Noordzee. Een studie naar de effecten in het zeegebied ten westen van Zuid-Holland., RIKZ

Sherk JA, O'Connor JM, Neumann DA (1975) Effects of suspended and deposited sediments on estuarine environments. Est Res 2:18

Short AD (1996) The role of wave height, period, slope, tide range and embaymentisation in beach classifications: a review. Revista Chilena de Historia Natural:16

Slijkerman D, Van der Wal JT, Van Dalfsen J (2007) Mapping the shallow sandy coast of Egmond and Schiermonnikoog, The Netherlands. MESH Action 3.

Slijkerman DME, Tamis JE, Jongbloed RH (2008) Voortoets bestaand gebruik Noordzeekustzone (m.u.v. Visserij en Militaire activiteiten) – Hoofdrapport. Beheerplan Natura 2000 Waddenzee & Noordzeekustzone. Report No. CONCEPT, April 9, 2008, Wageningen IMARES

Smit MGD (2004) Grain size data for DCS species. Unpublished data

Smit MGD, Holthaus KIE, Tamis JE, Jak RG, Karman CC, Kjeilen-Eilertsen G, Trannum H, Neff J (2006) Threshold levels and risk functions for non-toxic sediment stressors: burial, grain size changes, and hypoxia - summary report - TNO

Smit MGD, Holthaus KIE, Trannum HC, Neff JM, Kjeilen-Eilertsen G, Jak RG, Singsaas I, Huijbregts MAJ, Hendriks AJ (2008) Species sensitivity distributions for suspended clays, sediment burial, and grain size change in the marine environment. Environmental Toxicology and Chemistry 27:1006-1012

Snelgrove PVR, Butman CA (1994) Animal-sediment relationships revisited: Cause versus effect. . Oceanogr Mar Biol Ann Rev 32:7

Speybroeck J, Bonte D, Courtens W, Gheskiere T, Grootaert P, Maelfait J-P, Mathys M, Provoost S, Sabbe K, Stienen E, Van Lancker V, Vincx M, Degraer S (2004) Studie over de impact van zandsuppleties op het ecosysteem, Ministerie van de Vlaamse Gemeenschap. Afdeling waterwegen kust.

Speybroeck J, Bonte D, Courtens W, Gheskiere T, Grootaert P, Maelfait JP, Mathys M, Provoost S, Sabbe K, Stienen EWM, Van Lancker V, Vincx M, Degraer S (2006) Beach nourishment: An ecologically sound coastal defence alternative? A review. Aquatic Conservation: Marine and Freshwater Ecosystems 16:419-435

Speybroeck J, Bonte D, Courtens W, Gheskiere T, Grootaert P, Maelfait JP, Provoost S, Sabbe K, Stienen EWM, Van Lancker V, Van Landuyt W, Vincx M, Degraer S (2008) The Belgian sandy beach ecosystem: A review. Marine Ecology 29:171-185

TNO (1994) Ecologische risicoanalyse op basis van overschrijding van kritische verstoringniveaus. Een bijdrage aan het MER Proefboringen naar aardgas in de Noordzeekustzone en op Ameland en het MER Proefboringen naar aardgas in de Waddenzee.

Van Dalfsen J (1994) Effecten van het lozen van baggerretourwater bij Ferwerderadeel (1990-1993).

Van Dalfsen J (1999) Ecologische effecten van grootschalige zandwinning. Werkdocument t.b.v. visieontwikkeling op kustplannen. , RIKZ

Van Dalfsen J, De Kluijver M, Lewis W, van Moorsel G, Tempelman D, van der Wal JT (2007) The Macrobenthic Fauna in the Dutch Sector of the North Sea in 2006 and a comparison with previous data, Grontmij, AquaSense, Ecosub and TNO-IMARES

Van Dalfsen JA (2006) Inventaristatie brandingszone zandige kust, TNO

Van Dalfsen JA, Essink K (1997) Risk analysis of coastal nourishment techniques in The Netherlands. Part A. The ecological effects of shoreface nourishment off the island of Terschelling, Part B. The ecological effects of subaqueous sand extraction North of the island of Terschelling, Part C. Literature references. , National Institute for Coastal and Marine Management RIKZ,

Van Dalfsen JA, Essink K (2001) Benthic community response to sand dredging and shoreface nourishment in Dutch coastal waters. Senckenbergiana Maritima 31:329-332

Van Dalfsen JA, Lewis WE (2001) Lange-termijn effecten op de bodemfauna van een tijdelijke zandwin/overslagput in de kustzone ter hoogte van Heemskerk, TNO

Van der Wal JT, Van Dalfsen JA (2008) Monitoring kustsuppleties. Report No. C014/08, Wageningen IMARES, Den Helder

Van Hoey G, Degraer S, Vincx M (2004) Macrobenthic community structure of soft-bottom sediments at the Belgian Continental Shelf. . Estuarine, Coastal and Shelf Science 59:15

Van Moorsel GWNM (2005) Macrofauna en hydromorfologie van zoute wateren, Ecosub, in opdracht voor het RIKZ

Wilber DH, Clarke DG, Burlas MH (2006) Suspended sediment concentrations associated with a beach nourishment project on the northern coast of New Jersey. Journal of Coastal Research 22:1035-1042

Wilber DH, Clarke DG, Ray GL, Burlas M (2003) Response of surf zone fish to beach nourishment operations on the northern coast of New Jersey, USA. Marine Ecology Progress Series 250:231-246

Wu RSS (2002) Hypoxia: from molecular responses to ecosystem responses. Mar Pollut Bull 45:11

Chapter 5 On the interactions between benthos and sediment dynamics in the foreshore

Austen, I., Andersen, T.J., Edelvang, K., 1999. The influence of benthic diatoms and invertebrates on the erodibility of an intertidal mudflat, the Danish Wadden Sea. Estuarine, Coastal and Shelf Science 49, 99-111.

Baptist, M.J., van Dalfsen, J., Weber, A., Passchier, S., van Heteren, S., 2006. The distribution of macrozoobenthos in the Southern North Sea in relation to meso-scale bedforms. Estuarine, Coastal and Shelf Science 68, 538-546.

Baumfalk, Y.A., 1979. Heterogeneous grain size distribution in tidal flat sediment caused by bioturbation activity of Arenicola marine (polychaeta). Netherlands Journal of Sea Research 13, 428-440.

Borsje, B.W., De Vries, M.B., Hulscher, S.J.M.H., De Boer, G.J., 2008a. Modeling large-scale cohesive sediment transport affected by small-scale biological activity. Estuarine, Coastal and Shelf Science 78, 468-480.

Borsje, B.W., De Vries, M.B., Bouma, T.J., Hulscher, S.J.M.H., Herman, P.M.J., 2008b. Bio-geomorphological interactions in offshore seabed patterns. Proceedings of the 31st International Conference on Coastal Engineering, Hamburg, Germany.

Bouma, T.J., van Duren, L.A., Temmerman, S., Claverie, T., Blanco-Garcia, A., Ysebaert, T., Herman, P.M.J., 2007. Spatial flow and sedimentation patterns within patch of epibenthic structures: combining field, flume and modeling experiments. Continental Shelf Research 27, 1020-1045.

Burone, L., Muniz, P., Pires-Vanin, A.M.S., Rodrigues, M., 2003. Spatial distribution of organic matter in the surface sediments of Ubatuba Bay (Southesatern – Brazil). Annals of the Brazilian Academy of Sciences 75, 77-90.

Cramer, A., Duineveld, G.C.A., Jenness, M.I., 1991. Observations on spatial distribution, metabolism and feeding strategy of Echinocardium cordatum (Pennant) (Echinodermata) and the implications for its energy budget. In: Cramer, A., (Ed.), Benthic Metabolic Acitivy at Frontal Systems in the North Sea. Ph.D. Thesis, University of Amsterdam, Amsterdam, pp. 63-74.

Friedrichs, M., 1997. Report about the characteristics of the NIOO-CEMO flume system-pilot study. NIOO-CEMO, NIOO rapporten 1997-6, Yerseke.

Friedrichs, M., Graf, G., Spring, B., 2000. Skimming flow induced over a simulated polychaete tube lawn at low population densities. Marine Ecology Progress Series 192, 219-228.

Heip, C., Basford, D., Craeymeersch, J.A., Dewarumez, J.-M., Dörjes, J., De Wilde, P., Duineveld, G., Eleftheriou, A., Herman, P.M.J., Niermann, U., Kingston, P., Künitzer, A., Rachor, E., Rumohr, H., Soetaert, K., Soltwedel, T., 1992. Trends in biomass, density and diversity of North Sea macrofauna. Journal of Marine Science 49, 13-22.

Hirano, M., 1971. River bed degradation with armouring. Proceedings, Jap. Soc. of Civil Eng. 195, 55-65.

Holtmann, S.E., Groenewold, A., Schrader, K.H.M., Asjes, J., Craeymeersch, J.A., Duineveld, G.C.A., van Bostelen, A.J., and van der Meer, J., 1996. Atlas of the zoobenthos of the Dutch Continental Shelf. Ministry of Transport, Public Works and Water Management, North Sea Directorate, Rijswijk, 1-244 pp.

Künitzer, A., Duineveld, G.C.A., Basford, D., Dewarumez, J.-M., Dörjes, J., Eleftheriou, A., Heip, C., Herman, P.M.J., Kingston, P., Niermann, U., Rumohr, H., De Wilde, P.A.W.J., 1992. The benthic infauna of the North Sea: species distribution and assemblages. Journal of Marine Science 49, 127-143.

Lohrer, A.M., Thrush, S.F., Hunt L., Hancock, N., Lundquist, C., 2005. Rapid reworking of subtidal sediments by burrowing spatangoid urchins. Journal of Experimental Marine Biology and Ecology 321, 155-169.

Nowell, A.R.M., Church M., 1979. Turbulent flow in a depth-lirnited boundary layer. Journal of Geophysical Research 84, 4816-4824.

O'Donoghue, T., Doucette, J.S., van der Werf, J.J., Ribberink, J.S., 2006. The dimensions of sand ripples in full-scale oscillatory flows. Coastal Engineering 53, 997-1012.

Rabaut, M., Guilini, K., Van Hoey, G., Vincx, M., Degraer, S., 2007. A bio-engineered soft-bottom environment: The impact of Lanice conchilega on the benthic species-specific densities and community structure. Estuarine, Coastal and Shelf Science 75, 525-536.

Soulsby, R. L., 1983. The bottom boundary layer of shelf seas, in Physical Oceanography of Coastal and Shelf Seas, edited by B. Johns, pp. 189–266, Elsevier Sci., New York.

Uittenbogaard, R. 2003. Modelling turbulence in vegetated aquatic flows. In International workshop on RIParian FORest vegetated channels: hydraulic, morphological and ecological aspects, 20-22 February 2003, Trento, Italy, 2003.

Widdows, J., Brinsley, M.D., 2002. Impact of biotic and abiotic processes on sediment dynamics and the consequences to the structure and functioning of the intertidal zone. Journal of Sea Research 48, 143-156.

Justification

Report IMARES C113/08, Deltares Z4582.50
Project Number: IMARES 439.61119.01, Deltares Z4582.50.

To be cited as:
Baptist, M.J., J.E. Tamis, B.W. Borsje, J.J. van der Werf & H. Holzhauer, 2008. Review of the geomorphological, benthic ecological and biogeomorphological effects of nourishments on the shoreface and surf zone of the Dutch coast. Wageningen IMARES report C113/08, Deltares report Z4582.50, 68 p.

The scientific quality of this report has been peer reviewed by a colleague scientist and the head of the department Ecology of Wageningen IMARES.

Approved:	Dr. D.M.E. Slijkerman Scientist
Signature:	
Date:	23 January 2009

Approved:	Drs. F.C. Groenendijk Head Department of Ecology
Signature:	
Date:	23 January 2009

Number of copies:	25
Number of pages	69
Number of tables:	19
Number of graphs:	26
Number of appendix attachments:	0