Synthesis report on the effects of dredged material disposal on the marine environment (licensing period 2010-2011)

Brigitte Lauwaert[1], Rosalia Delgado[5], Jozefien Derweduwen[2]; Lisa Devriese[2], Michael Fettweis[1], Kris Hostens[2], Job Janssens[5], Chantal Martens[3], Johan Robbens[2], Steve Timmermans[4], Gert VanHoey[2], Toon Verwaest[5]

BL/2011/12

Report by MUMM, ILVO, CD, aMT and WL conform art. 10 of the R.D. of 12 March 2000 defining the procedure for licensing of disposal in the North Sea of certain substances and materials.

Colofon

Synthesis report on the effects of dredged material disposal on the marine environment (licensing period 2010-2011)

Brigitte Lauwaert[1], Rosalia Delgado[5], Jozefien Derweduwen[2]; Lisa Devriese[2], Michael Fettweis[1], Kris Hostens[2], Job Janssens[5], Chantal Martens[3], Johan Robbens[2], Steve Timmermans[4], Gert VanHoey[2], Toon Verwaest[5]

[1] MUMM: Management Unit of the North Sea Mathematical Models, Gulledelle 100, 1200 Brussels

[2] ILVO: Institute for Agricultural and Fisheries Research, Animal Sciences Unit – Fisheries, Section Monitoring, Ankerstraat 1, 8400 Oostende.

[3] aMT: Maritime Access Division, Tavernierkaai 3, 2000 Antwerpen.

[4] AMCS-CD: Agency for Maritime and Coastal Services – Coastal Division, Vrijhavenstraat 3n, 8400 Oostende.

[5] WL: Flanders Hydraulics Research, Berchemlei 115, 2140 Antwerpen.

Also participated in the research:

MUMM: Joan Backers, Matthias Baeye (UGENT) Jean-Pierre De Blauwe, Frederic Francken, Kevin Hyndrickx, Lieven Naudts, André Pollentier, Dries Van den Eynde, Vera Van Lancker

ILVO: Matias Bossaer, Bart Goes, Lode Jacobs, Marc Van Ryckeghem, Stefan Hoffman, Ellen Pecceu, Jan Wittoeck, Annelies De Backer, Hans Hillewaert, Vyshal Delahaut

CODA: Ludwig De Temmerman

aMT: Natasha Blommaert, Vincent Vaninghelandt, Kirsten Beirinckx

WL: Johan Reyns, Katrien Van Der Biest

With acknowledgement to the R/V Belgica crew

Contacts:

b.lauwaert@mumm.ac.be; +32(0)2-7732120

johan.robbens@ilvo.vlaanderen.be; +32(0)59-569850

chantal.martens@mow.vlaanderen.be +32(0)3-2220883

Content

Summary

Dredging and dumping

To conserve the marine access channels to the Belgian coastal harbours and to maintain the depth of the Flemish coast harbours, dredging is needed (Flemish competence) in order to guarantee safe maritime transport.

The competence for dumping at sea of dredged material falls under the federal government. The management of dredged material is Belgium is a mixed competence. A cooperation agreement between the Flemish Region and the federal government has been signed the 12th of June 1990 and was modified the 6th of September 2000.

The legal basis for the permitting procedure is the royal decree of 12th March 2000. According to art. 10 of this royal decree, a synthesis report has to be established per permit period, accompanied by recommendations which support the development of an enforced environmental management.

Dredging and dumping activities

For the previous permit period -1 January 2010 – 31 December 2011, four permits were delivered to the Maritime Access Division (aMT) and three permits were delivered to the Agency for Maritime and Coastal Services (AMCS).

The total average permitted quantity for aMT was 21,045,000 TDM (tonnes dry matter) for 4 dumping sites and 1,170,000 TDM for AMCS for 3 dumping sites. During 2009 and 2010 a total of respectively 11,910,431 TDM and 10,066,736 TDM were dumped by both permit holders. The dredging and dumping intensity maps for these years can be found in annex. For the period 2009-2010 approximately 31,000 m³ was used for beach nourishment (beneficial use).

Physical aspects related to dredging and disposal operations

The morphological and sedimentological effects resulting from dredging and disposal works are numerous and site specific. A good understanding of the site-specific dynamics is needed in order to evaluate environmental impact of dredging and disposal works. The results of different studies are summarized that deal with sediment dynamics and human impact in the Belgian nearshore area and the implementation of monitoring strategies to identify environmental changes induced by these activities.

Natural variability of sediment processes

The Belgian near-shore area is characterized by sediment composition varying from pure sand to pure mud. SPM concentration is high (100 mg/l to a few g/l) in the coastal turbidity maximum between Oostende and the mouth of the Westerschelde. In order to investigate the sediment dynamics in such an area multi-parametric tripod measurements are needed.

Sediment mobility during storms events

Tripod measurements in the high-turbidity area of the Belgian nearshore zone allowed investigating storm effects on near bed SPM concentrations. The data have shown that during or after a storm the SPM concentration increases significantly and that high concentrated mud suspensions (HCMS) are formed. Under these conditions, about 3 times more mass of SPM was observed in the water column, as compared to calm weather conditions.

The following different sources of fine-grained sediments, influencing the SPM concentration signal, have been investigated: wind direction and the advection of water masses; the previous history and occurrence of fluffy layers; freshly deposited mud near the disposal grounds of dredged material, navigation channels and adjacent areas; and the erosion of medium-consolidated mud of Holocene age. Based on erosion behaviour measurements of in-situ samples, the critical erosion shear stresses have been estimated for different cohesive sediment samples outcropping in the study area. The results have shown that most of the mud deposits cannot be eroded by tidal currents alone, but higher shear stresses, as induced by storms with high waves, are needed. Erosion can however occur during storms with high waves. Data suggest that in order to obtain very high SPM concentrations near the bed, significant amounts of fine-grained sediments have to be resuspended and/or eroded. The disposal grounds of dredged material, navigation channels and adjacent areas with freshly deposited mud have been found to be the major source of the fine-grained sediments during storms. This result is important, as it suggests that dredging and the associated disposal of sediments have made available fine-grained matter that contributes significantly to the formation of high SPM concentrations and high concentrated mud suspensions.

Sediment mobility in response to tidal and wind-driven flow

The effect of hydro-meteorological forcing (tidal and wind-induced flows) on the transport of SPM, on the formation of HCMS and on the occurrence of sand–mud suspensions has been studied using long-term multi-parametric observations. Data have been classified according to variations in subtidal alongshore currents, with the direction of subtidal flow depending on wind direction. This influences the position of the turbidity maximum; as such also the origin of SPM. Winds blowing from the NE will increase SPM concentration, whilst SW winds will induce a decrease. The latter is related to advection of less turbid English Channel water, inducing a shift of the turbidity maximum towards the NE and the Westerschelde estuary. Under these conditions, marine mud will be imported and buffered in the estuary. Under persistent NE winds, high-concentrated mud suspensions are formed and remain present during several tidal cycles. Data show that SPM consists of a mixture of flocs and locally eroded sand grains during high currents. This has implications towards used instrumentation: SPM concentration estimates from optical backscatter sensors will only be reliable when SPM consists of cohesive sediments only; with mixtures of cohesive and non-cohesive sediments, a combination of both optical and acoustic sensors are needed to get an accurate estimate of the total SPM concentration.

Cohesive and non-cohesive SPM dynamics

In coastal areas it is not uncommon for benthic sediments to consist of sand and mud mixtures, as is also the case in the coastal area between Oostende and the Westerschelde. The mud-sand ratio influences the transition between cohesive and non-cohesive sediments and has a major influence on the erosion behaviour, on SPM concentration and on the benthic ecological properties. Frequently, mixed sediments occur as an alternation of sand and mud layers. Storms induced waves result in intense sediment mobilization and the deposition of sand dominated layers. In case of high SPM concentration the segregation in sand/mud suspensions occurs when the initial mud concentration is smaller than its gelling point. In such areas, SPM reflects the bed composition and may consist of a mixture of cohesive and non-cohesive mineral particles. Close to a sandy seabed, SPM is likely to also contain resuspended mineral grains, whereas higher in the water column or in muddy environments SPM occurs typically in the form of flocs composed of aggregates of mainly clay minerals, organic matter and water.

The goal of this study was to investigate flocculation and the floc and particle dynamics. SPM concentration and particle size distribution (PSD) were assessed in the Belgian nearshore area during a composite period of 37 days in January–April 2008. PSDs were measured with a LISST 100X and classified using entropy analysis in terms of subtidal alongshore flow. The PSDs during tide-dominated conditions showed distinct multimodal behaviour due to flocculation, revealing that the building blocks of flocs consist of primary particles (<3 μm) and flocculi (15 μm). Flocculi comprise clusters of clay minerals, whereas primary particles have various compositions (calcite, clays). The PSDs during storms with a NE-directed alongshore subtidal current (NE storms) are typically unimodal and characterised by mainly granular material (silt, sand) resuspended from the seabed. During storms with a SW-directed alongshore subtidal current (SW storms), by contrast, mainly flocculated material can be identified in the PSDs. The findings emphasise the importance of wind-induced advection, alongshore subtidal flow and HCMSs as regulating mechanisms of SPM concentration, as well as other SPM characteristics (cohesiveness or composition of mixed sediment particles) and size distribution in a high-turbidity area. The direction of subtidal alongshore flow during SW storm events results in an increase in cohesive SPM concentration, HCMS formation, and the armouring of sand; by contrast, there is a decrease in cohesive SPM concentration, no HCMS formation, and an increase in sand and silt in suspension during NE storms.

SPM concentration as indicator to detect changes in the marine environment

SPM concentration as indicator

Large sets of SPM concentration data from in situ and remote sensing (MODIS) samplings in the Belgian nearshore area are combined in order to evaluate their heterogeneity and the sampling techniques. The heterogeneity has been statistically assessed by comparing the SPM concentration frequency distributions. In situ and MODIS data sets have different distributions and represent a different sub-population of the whole SPM concentrations population. The differences between the data sets are related to meteorological conditions during the measurements; to near bed SPM concentration dynamics, which are partially uncoupled from processes higher up in the water column; to the sampling methods or schemes and to measurement uncertainties. It was found that satellites are biased towards good weather condition or spring-summer seasons (satellite).

Due to the time and spatial variability of SPM concentration in coastal high turbidity areas, greater sampling efforts are necessary, as compared to offshore systems with low SPM concentration. Satellite, tidal cycle and tripod SPM concentrations are very similar at a location situated 20 km offshore at the edge of the coastal high turbidity area. Satellites or low-frequent tidal cycle measurements cannot replace long-term continuous measurements in high turbidity areas, which include all sea state conditions. They consist of a sub-set of the population biased towards good weather condition or spring-summer seasons (satellite). Sediment transport based on these data will thus always underestimate reality. The mean SPM concentrations derived from satellite data set is included within one standard deviation of the mean from the tidal cycle data set at both locations, which indicates a good agreement between the distributions of the two datasets. SPM concentrations from tripod, extrapolated to the water surface and sub-sampled following the satellites sampling scheme, have a mean significantly higher than the mean of satellite surface SPM concentrations.

The reasons are due to uncertainties in the calculation of vertical profiles used to achieve the extrapolation, and to the fact that probably a filtering of high SPM concentrations (>200 mg l^{-1}) occurred, where the visible band saturates.

If the data series collected during different periods have similar log-normal distributions, geometric means and standard deviations, then we could conclude that - within the range of natural variability and measuring uncertainties - these data series represent similar sub-samples of the whole SPM concentration population. Consequently, if disposal of dredged material has a significant impact on SPM concentration then this should be detectable in the differences between the statistical parameters of the sub-sample collected during the dredging experiment and of the whole population. As the heterogeneity and complexity of the SPM concentrations are high in the Belgian nearshore area, due to their natural high variability, statistical methods have been used to characterize temporal SPM concentration variation in a way that it can be used as indicator for changes induced by human activities.

Monitoring the effect of disposal of fine-grained sediments on the SPM concentration

The impact of continuous disposal of fine-grained sediments from maintenance dredging works in the Albert II dock (Zeebrugge) on the SPM concentration in a shallow nearshore turbidity maximum was investigated during dredging experiment. Before, during and after the experiment monitoring of SPM concentration using OBS and ADV altimetry was carried out at a location 5 km west of the disposal site. A statistical analysis, based on the concept of populations and sub-sampling, was applied to evaluate the effect. The data revealed that the SPM concentration near the bed was on average more than 2 times higher during the dredging experiment. The disposed material was mainly transported in the benthic layer and resulted in a long-term increase of SPM concentration and formation of fluid mud layers. The study shows that SPM concentration can be used as an indicator of environmental changes if represenative time series are available.

Conclusions

The effect of winds on SPM concentration is variable and depends also on the wind direction and the availability of muddy sediments. Our data show that high SPM concentrations are often more closely related to advection rather than instantaneous bed shear stress. This confirms the idea that the Belgian coastal area can be seen as a congestion in the residual SPM transport of the southern North Sea, rather than an important source of sediments.

The deposits of fresh mud below the fluffy layer in the navigation channels forms a reservoir of SPM that will only be resuspended during periods with high shear stresses, e.g. caused by storms. The data suggest that a major part of the HCMS could have been resuspended from the very soft mud deposits in the navigation channels and adjacent areas.

Particle size distributions are generally multimodal and consist of primary particles, flocculi, microflocs and macroflocs. The data suggest that two populations of primary particles (<3 µm and flocculi 15 µm) co-exist and are the building blocks of flocs. Flocculi consist of clusters of clay minerals, whereas primary particles are of various composition.

Mixed sediments have been found in suspension at 2 mab during maximum flood currents at spring tide and during storms. Near Zeebrugge this occurs more prominently during winds blowing from SW-W. The size distribution of the local bed sediments is thus only influencing the size distribution of the SPM when HCMS are not present and when turbulence induced by currents or waves is strong enough to bring sand or silt into suspension.

SPM concentration can be used as an indicator of environmental changes, if sufficiently long time series are available that are representative of the natural variability.

Satellites or low-frequent tidal cycle measurements cannot replace long-term continuous measurements in high turbidity areas. The former datasets consist of a sub-set of the population biased towards good weather condition. Sediment transport based on these data will thus always underestimate reality.

An analysis method was used based on the concept of statistical populations in order to evaluate the effects of disposal operations on SPM concentration. The method provides a tool to account for the complexities associated with natural dynamics and the need to evaluate quantitatively human impact. The disposal results in a long-term increase of SPM concentration near the bed at the measuring location. This together with ADV altimetry suggest that fluid mud layers have been formed during whole the disposal experiment rather than being limited to neap tidal or storm conditions as observed during the other periods.

Biological and chemical effects of dredged material disposal

It is important to investigate the effects of dumping of dredged material in the marine environment from ecosystem perspective, because it could lead to different responses of the ecosystem. Therefore, the regular dredging program from ILVO Fishery is evaluating the impact of these dumping activities at different levels by looking at: i) differences in biological characteristics of the ecosystem components macrobenthos, epibenthos and demersal fish ii) The (bio)accumulation of contaminants in the marine ecosystem as investigated by chemical analysis on different biota species and on different matrices iii) biological effects of pollutants on marine organisms as indicated by the prevalence of fish diseases and by the measurement of enzymatic EROD activities in the liver of juvenile dab. Besides this regular monitoring, we also conducted a study on the long term impact of dredged material dumping on the benthic habitats in the Belgian Coastal Zone. Secondly, we explored the benthic life in the dredging area's itself. Thirdly, we invested the effect of some adaptations in the ecological sampling strategy of the routine monitoring program.

Long term impact of dredged material dumping on the benthic habitats

A long term analysis of changes in the benthic habitat characteristics in relation to the dumping intensity is executed at the five dumping sites. At ILVO, a benthic sampling program is running from 1978 onwards and changed strongly over this period, with a clear impact/control sampling strategy at each dumping site in the last years (2004 onwards). Previously, only a monitoring point inside the dumping zone Br&W S2 is followed up. Based on the multivariate analysis, we detected a transition in species assemblage. This changes in species composition can be attributed to the absence of mud loving species in the recent period. The average density and species richness were also significantly lower in the latest period ('01-'08) compared to the previous ones ('85-'90; '93-'00). These observed patterns coincide with a decrease in the dumping intensity in the recent period ('01-'08). We can conclude that dumping at Br&W S2 has a small positive effect, by supply of mud and organic matter to the more sandy environment. This species enrichment pattern at Br&W S2 is observed again in the monitoring of 2010 (see further).

The dumping activities at the five disposal sites have lead to benthic habitat changes at dumping site Br&W S1, while at the other sites; the benthic community seems to cope with the existing dumping regime. Especially in naturally more impoverished areas (dumping site Br&W Zeebrugge-Oost & Br&W Oostende), the impact is less pronounced than in more vulnerable benthic habitats (e.g. *Abra alba* habitat or sandy environments). We suggest that if an impact is detected, this is mainly related to the physical burial of the organisms (smothering, incorporation), or to the properties of the dredged-disposal (mud in more sandy areas), both causing habitat modifications.

Biological and chemical status of the disposal sites: 2009-2010

Biological status

The ecological status of the macrobenthos at the different disposal sites is evaluated with the benthic indicator BEQI. The observed patterns confirm those of the previous years. The medium to high dumping intensity at Br&W Oostende and Br&W Zeebrugge Oost has no effect on the macrobenthic community. At dumping site Nieuwpoort however, the macrobenthic characteristics showed a high variability which could not be attributed to the low dumping activity. At Br&W S1, where the highest dumping activities occur, a steady loss of *Abra alba* habitat could still be observed in the dumping area. The tube building polychaete *Owenia fusiformis* however, has exponentially increased in the surrounding area. Despite the 'good status' of Br&W S2, some remarkable changes in the macrobenthic community appeared. The benthic characteristics of the samples indicate an enrichment of the Northern samples with mud 'loving' species, whereas the samples in the western part are more impoverished (lower diversity). If this could be attributed to the higher dumping intensities over the period 2009 (focus on Northern part) – 2010 (focus on Western part) than in previous years is not unambiguous. Still, this should be confirmed by further detailed analyses.

For the epibenthic and demersal fish fauna, no effects of dumping were clearly visible. This could be explained by the fact that most of those species have a high mobility and are able to flee from the dumping sites. Another possibility is the fact that the statistical power (due to a low number of samples caused by a switch in sampling strategy) was too low to detect any possible effects. Nevertheless, some significant differences in the epibenthic and demersal fish characteristics occurred, probably due to a temporary dominance of certain species (e.g. starfish, brittle star and goby) and/or by the natural variability of the habitat (e.g. dumping site Nieuwpoort). For future research, it is advisable to investigate the effect of dumping on certain functional or sensitive species.

Chemical status

Sediment Assessment

Only limited differences between impact and control assessment are observed for the measured parameters heavy metals, PCBs and pesticides. With exception of dumping site Br&W Zeebrugge Oost, on which the pollution levels of impact sites are elevated compared to control sites. Seasonal variation of these parameters is not significant. Nevertheless, the levels of lead and PCBs must be followed up in future. Other measured heavy metals and persistent organic pollutants do not approach the formulated EAC values (OSPAR, MSFD Task Group 8 Contaminants and pollution effects, Belgisch Staatsblad).

Accumulation of Pollutants in Marine Organisms

Chemical analysis was performed on diverse sentinel species to assess the (bio)accumulation level of persistent organic pollutants and heavy metals. Due to the omnipresence in the marine environment and relevance in the ecosystem, mainly starfish and brown shrimp were used to evaluate the chemical health status of the different dumping sites.

Generally, during the period 2009-2010 no significant trend between control and impact zone assessment was recorded, and the seasonal variation was also rather small. Elevated PAH levels in diverse marine species were observed in the impact areas of dumping sites Br&W Zeebrugge Oost, Br&W Oostende and Br&W S1 compared to the control areas. A higher level of Cu is observed on dumping site Br&W S2 and Br&W Oostende. On dumping site Br&W Oostende a remarkably high CB level was assessed. The measured levels of CBs, PAHs and heavy metals are the lowest on dumping sites Br&W S1 and Nieuwpoort.

Biological Effects of Pollutants

Externally visible fish diseases (e.g. ulcers, skeletal deformations, nodules, lymphocystis) and parasite infection were used as parameter for environmental stress and environmental health status. Most anomalies were due to parasitical infections and did show high variation in spatial and temporal distribution. The observed prevalence of *Glugea stephani* and *Acanthochondria cornuta* in the period 2009-2010 (the dumping site vs. the coastal reference zone) was remarkably higher compared to the mean prevalence over the period 2000-2010. These diseases must be followed up strictly in future.

Secondly, the biomarker EROD (7-ethoxyresorufin O-deethylase) activity is used as an indicator of xenobiotic substance accumulation in the flatfish dab (OSPAR indicator). The EROD induction in the liver of juvenile dab is clearly visible during winter and early spring, while during summer and autumn only a background level is recorded. During the period 2009-2010, no significant higher EROD activity on impact sites versus control sites was observed.

An exploration of the biological life in the dredging areas

One of the aspects that was not studied in the previous decade was the biological life in the dredging areas, especially in the gullies towards the harbour of Zeebrugge and the harbour itself. 48 benthic taxa were recorded in the dredging areas of Zeebrugge, whereof 27 (56%) taxa were only recorded once. Most taxa were found in the gullies towards Zeebrugge and only a few taxa (9; Cirratulidae spp, Oligochaeta spp, Nephtys juvenile, *Mytilus edulis, Streblospio benedicti, Abra alba, Macoma balthica, Anthzoa spp, Crangon crangon*) in the harbour itself. Not one of the observed species was a rare taxon within the benthic fauna on the BPNS. We can conclude that the dredging areas around Zeebrugge were characterised by a very poor benthic community, except for the 'Vaargeul 1' area. Input of benthic animals from the dredging areas towards the dumping areas is possible, but should not lead to species enrichments in the dumping zones, due to the low densities and species richness in the dredging areas.

Optimization of the sampling strategy in the routine monitoring program

In the period 2009-2011, we invested in the optimization of the ecological sampling strategy of the routine monitoring program at the dredge disposal sites. This was carried out to standardize the analysis according to European directives (e.g. MSFD) and to make the monitoring time and cost efficient. We adjusted the sampling protocol of the epibenthos and fish tracks (shortening of the duration) and the benthic sieving procedure (alive instead of fixated). Finally, we introduced quality assurance in the macrobenthos analysis (ISO 16665:2005) by achieving a BELAC accreditation certificate under ISO17025 norm.

Concerning the changes in the duration of the epibenthos and fish tracks, we observed that the rate of overestimation or underestimation varied between tracks and between species groups. This difference plays no part when tracks of similar kind (short) are taken and compared within the same time frame. A clear advantage of using short tracks in the dredge disposal research is the fact that the tracks fit within the borders of the dumping site. Like this, side effects are minimized and the short tracks seem to result in more reliable density and diversity estimates of the area.

Sieving alive has a clear negative influence on the density and species richness of the samples, compared to sieving fixed. Based on analyses, we can trust that data retrieved when sieving alive on a 0.5 + 1mm sieve is comparable with data retrieved for fixed sieving. Therefore, we can consider that this switch in sieving procedure will have a minor influence on the long term trend analysis at the benthic control stations. Since sieving with two sieves is only used at a certain subset of stations, we have to use conversion factors for analyzing a long term trend at the other stations.

Perspectives

In the ecological monitoring program, we will keep the current monitoring strategy, because it seems suitable to evaluate changes within the dumping area and its surroundings. By high dumping intensities, leading to habitat modifications, a clear impact is detected on the overall benthic characteristics (density, diversity). In the future, it is advisable to investigate the possible loss of the ecosystem functioning by this habitat modifications, by using functional traits analysis. Secondly, we have to consider the possibility to determine a critical boundary of dumping intensity leading to a certain impact.

In the chemical monitoring program, it will be a necessity to investigate the general toxicity of the environment. General toxicity tests will give information about the presence and bioavailability of toxic compounds in the environment. Also, special attention must be paid to chemical monitoring of Br&W Zeebrugge Oost, due to fact that elevated contaminant concentrations were detected in the sediments and biota. In addition to the assessment of fish diseases, it would be of main importance to monitor the general health of fish species, e.g. gonadosomatic index (GSI), quality index method (QIM), liver glycogen content (LGC), liver-somatic index (LSI), condition index (k), etc.

Long- to medium-term bathymetry changes

Based on a series of historical navigation channels, the long-term (last 150 years) morphologic evolution of the BPNS has been studied. Additionally, the digitally available topo-bathymetrical data sets of beach, shoreface and near coastal zone have been used to perform a mid-term (from 1997 up till now) morphological trend analysis. Apart from natural trends (banks, though more or less stable in position, seem to have accreted while the troughs between the banks have deepened, especially in the near coastal zone), some anthropogenically induced changes can also be distinguished: the deepening of the Scheur navigation channel and sedimentation of beach and shoreface in the shelter of the Zeebrugge breakwaters being the most clear ones. Dumping activities also seem to have an influence on the local morphology, though this influence cannot be distinguished from natural evolution and does not result always in a clear accretion trend.

Dredging and Dumping: Alternative dumping location

One of the long-term goals of the MOMO project is to develop efficient dredging and dumping strategies to. One way to achieve this is to select dumping zones where the recirculation to the dredging zones is small, thereby decreasing the need for dredging. The recommendations to the Minister in the synthesis report for the period 2008-2009 stated that a field test for an alternative dumping site should be prepared. As a first step, a report was produced, using numerical models to investigate possible alternative locations for a dumping site. Based on the report it was decided to perform a one month field test in autumn of 2012. The field test will consist in replacing the dumping site Br&W Zeebrugge Oost (which is the least efficient) with a new location (Zeebrugge 4) west of Zeebrugge. A comprehensive monitoring program will be set up.

The results of the numerical model predict a 10% reduction in dredging volumes in the outer harbour of Zeebrugge, and 3% in Pas van het Zand. If the results of the analysis of the field test confirm these figures a new landfill site might be established west of Zeebrugge, taking into account both economic and ecological factors.

Research project – Disposal of dredged material from the marina of Nieuwpoort using a pipeline

The coastal harbours and marinas in Belgium are confronted with the never ending problem of shoaling and accretion. Depending on the location and in-situ conditions both sandy and muddy matter can be found.

In the marinas a mean sludge volume of respectively 350.000 m³ is being dredged annually. Currently these quantities have been removed using the classical approach: dredging using a cutter dredger, transfer the spoil in loading containers and dispose this back at sea at the designated dredging spoil disposal sites. The yearly dredging campaign runs from November to April. This pilot project will investigate if the dredging activities in the marinas and depositing the dredge spoil can be done in a more efficient, environmental friendly and economical manner.

Project description

Since the largest quantity of sludge is being removed in Nieuwpoort it is suggested to start a pilot project in this marina. Research will conducted if a pipeline running a short distance can be placed at sea for the execution of the limited pilot project. Using this approach, the dredged material will be retrieved using a cutter dredger and transported to the disposal site a short distance from the shore line. This in contrast with the current method where the dredged material is transported with loading containers and disposed at the disposal site of Nieuwpoort.

In total the dredging will continue for 3 to 4 weeks, with a planned working regime of 24 hours a day. The final disposal site is to be decided in consultation with all partners depending on the feasibility of the pilot project and the possibilities for the final disposal site location.

Advantages

The following advantages of this alternative method can be mentioned.

The efficiency of the dredging activities can be significantly increased. Since time for connecting and disconnecting the loading containers is not needed anymore the cutter can be 1,5 times more productive. Furthermore, bad weather will not delay the work being done.

Since the dredging activities can be conducted quicker, the day and night regime (7 days of 7, 24h a day) is not necessary. The (noise) hindrance during the nights and weekends can be avoided. The period in which the dredging will be conducted will be shortened. The surveillance and control on the works can therefore be intensified.

Navigation between marina and wharf (in this case: Nieuwpoort) is not needed anymore. This means a considerable decline in CO_2 emissions and small dust particles.

Follow up

During the pilot project a monitoring campaign will be done for the sediment transport and ecological impact in consultation with the departments concerned.

MFSD

The MSFD is the first piece of EU legislation aimed at the protection of Europes marine environment as a whole. Its main goal is to achieve Good Environmental Status (GES) in all EU waters by 2020 by applying an ecosystem-based approach to the management of human activities which have an impact on the marine environment.

By the 15th of July 2012, Member States must conduct an initial assessment on the status of their marine waters. By this date they also must define what they understand to be GES in their marine waters and set environmental targets and associated indicators to drive their progress towards achieving GES.

By the 15th of July 2014 Member States will have to have monitoring programmes for all marine waters, adapted to the assessment of progress towards GES.

This new challenge might have implications on the existing monitoring programmes for dumping at sea of dredged material.

1. Dredging and dumping

1.1 Introduction

To conserve the maritime access channels to the Belgian coastal harbours and to maintain the depth of the Flemish coastal harbours, dredging is needed (Flemish competence) in order to guarantee safe maritime transport. This type of dredging is called maintenance dredging and it is the only type of dredging which is covered by this report. Most of the dredged material is being dumped at sea except when the quality is suitable for beach nourishment. The last use is called beneficial use of dredged material.

Dumping at sea of dredged material is carried out in accordance with the federal law of 20th January 1999 and a permit is given in accordance with the procedure defined in the royal decree of 12th March 2000. Corresponding to article 10 of this procedure, a “synthesis report” has to be established for the Minister who has the North Sea under his competences. The synthesis report needs to include recommendations which support the development of an enforced environmental management. Permits were given to the Maritime Access Division who is responsible for maintaining all maritime access channels to the Flemish ports as well as to the Coastal Division of the Agency for Maritime Services and Coasts who is responsible for the maintenance of the Flemish Coastal Marinas.

The international framework for dumping at sea of dredged material is the (regional) OSPAR Convention (1992) and the (worldwide) London Convention (1972) and Protocol (1996). These conventions and their associated guidelines take into account the presence of any contaminants within the sediment and whether some alternative beneficial use is possible. In implementing these guidelines, e.g. action levels (sediment quality criteria) have to be defined, dumping sites have to be chosen and a permanent monitoring and research programme has to be carried out.

1.2 Dredging activities

Since 2008, dredging years are following calendar years and since 2006 a distinction is being made between permits for maintenance dredging (validity 2 years) and permits for capital dredging (these permits are granted for the period of working). The areas to be dredged are divided in accordance with the target depth which is defined in function of the expected vessel types and their maximum draught.

The use of certain dredging technique is dependent upon the site, the hydrodynamic and meteorological circumstances and the nature of the sediment to be dredged. Evaluation is being made on the basis of economical, ecological and technical criteria. In Belgium most commonly trailing suction hopper dredgers are used with a hopper capacity from 5000 to 10000 m³.

In the access channels and Flemish harbours, maintenance dredging is virtually continuous throughout the year. Maintenance dredging in fishing harbours and marinas is taking place before and just after the coastal tourist period. A major port - and its connected access channels - with a diversity of customers may need to carry out a capital project every few years to accommodate changes in the patterns of trade and growth in the size of the vessels to be accommodated.

1.3 Dumping activities

Quantities permitted

In the former licensing period 1January 2010 – 31 December 2011, four permits for maintenance dredging were granted to the Maritime Access Division as well as three permits to the Agency for Maritime and Coastal Services. The maximum and average attributed quantities which may be dumped at sea per year and per dumping area are given in Table 1.1 and 1.2. It should be noted that the permit holder is requested to not exceed the average quantities. The dumping sites are given in the accompanying map in annex.

Table 0.1: Permits for the Maritime Access Division (aMT).

Permit reference	Dredging site	Type dredging	Dumping site	Yearly permitted quantities TDM (tonnes dry matter)	
				average	maximum
M.B. ref. BS/2009/01	* Scheur West * Scheur Oost * Pas van het Zand; CDNB en Voorhaven Zeebrugge	maintenance	Br&W S1	2,300,000 2,300,000 6,400,000	2,800,000 2,800,000 7,150,000
		Total :		**11,000,000**	**12,750,000**
M.B. ref. BS/2009/02	* Scheur West * Scheur Oost * Pas van het Zand; CDNB en Voorhaven Zeebrugge	maintenance	Br&W S2	500,000 375,000 2,000,000	600,000 450,000 2,400,000
		Total :		**2,875,000**	**3,450,000**
M.B. ref. BS/2009/03	* Toegangsgeulen Oostende (Stroombankkil, ingangsgeul Oostende) * Haven Oostende	maintenance	Br&W Oostende	600,000 500,000	900,000 700,000
		Total :		**1,100,000**	**1,600,000**
M.B. ref. BS/2009/04	* CDNB Zeebrugge * Haven en Voorhaven Zeebrugge * Toegangsgeul Blankenberge	maintenance	Br&W Zeebrugge Oost	3,900,000 2,100,000 70,000	5,500,000 3,150,000 100,000
		Total :		**6,070,000**	**8,750,000**
		GRAND TOTAL		**21,045,000**	**26,550,000**

Table 0.2: Permits for the Agency for Maritime and Coastal Services.

Permit reference	Dredging site	Type dredging	Dumping site	Yearly permitted quantities TDM	
				average	maximum
M.B. ref. BS/2009/05	* Jachthaven Oostende – RYCO * Jachthaven Oostende – Montgomery dok	maintenance	Br&W Oostende	50,000 50,000	75,000 75,000
		Total :		**100,000**	**150,000**
M.B. ref. BS/2009/06	* Vaargeul Blankenberge * Vlotdok Blankenberge * Spuikom te Blankenberge	maintenance	Br&W Zeebrugge Oost	100,000 100,000 100,000	200,000 150,000 150,000
		Total :		**300,000**	**500,000**
M.B. ref. BS/2009/ Nieuwpoort	* Toegangsgeul Nieuwpoort * Vaar- en havengeul Nieuwpoort * Oude Vlotkom Nieuwpoort * Nieuwe jachthaven Nieuwpoort * Novus Portus Nieuwpoort	maintenance	Nieuwpoort	70,000 200,000 100,000 200,000 200,000	100,000 300,000 200,000 300,000 300,000
		Total :		**770,000**	**1,200,000**
		GRAND TOTAL		**1,170,000**	**1,850,000**

Quantities dumped.

Despite the fact that dredging years are following calendar years since 2007, gives **Error! Not a valid bookmark self-reference.** an overview of the quantities dumped at sea since 1991 till March 2011 at the different dumping sites, just for continuation of the table in former reports. It should also be noted that the amounts mentioned in table 1.3 are being used for the yearly OSPAR reporting of dumped dredged material, also for continuation in former reporting years. Table 1.4 gives the overview of the quantities of maintenance dredged material dumped yearly since 2007.

Table 0.3: Quantities dumped dredged material since 1991.

Quantities dumped in wet tonnes(*)								
period	Br&W S1	Br&W S2	Br&W Zeebrugge Oost	Br&W Oostende	Nieuwpoort	Br&W R4 (**)	Br&W S3 (**)	Total
April 1991 - March 1992	14,176,222	7,426,064	10,625,173	4,416,386				36,643,845
April 1992 - March 1993	13,590,355	5,681,086	10,901,837	3,346,165				33,519,443
April 1993 - March 1994	12,617,457	5,500,173	10,952,205	3,614,626				32,684,461
April 1994 - March 1995	15,705,346	2,724,157	8,592,891	3,286,965				30,309,359
April 1995 - March 1996	14,308,502	2,626,731	8,432,349	4,165,995				29,533,577
April 1996 - March 1997	14,496,128	1,653,382	7,609,627	2,763,054				26,522,191
Quantities dumped in tonnes dry matter (*)								
maintenance								
capital								
period	Br&W S1	Br&W S2	Br&W Zeebrugge Oost	Br&W Oostende	Nieuwpoort	Br&W R4	Br&W S3	Total
April 1997 - March 1998	6,045,581	1,563,485	6,593,905	745,147				14,948,118
April 1998 - March 1999	7,455,619	482,108	2,976,919	467,107				11,381,753
April 1999 - March 2000	2,885,801	89,556	3,189,077	591,605				6,756,039
	6,187,601	41,583						6,229,184
April 2000 - March 2001	1,684,517	784,343	4,971,782	559,332		310,670	51,150	8,361,794
	3,873,444	614,657						4,488,101
April 2001 - March 2002	2,031,147	329,798	2,623,069	565,938				5,549,952
	2,527,392							2,527,392
April 2002 - March 2003	3,314,115	858,607	2,311,650	491,217	289,949			7,265,538
	2,413,760	208,885	1,369,939					3,992,584
April 2003 – March 2004	5,246306	716,427	3,126,392	646,276	142,420			9,877,821
	829,486	24,896	447,219					1,301,601
April 2004 – March 2005	1,826,561	1,826,033	3,003,397	464,307	71,928			7,192,226
April 2005 – March 2006	3,017,123	1,234,640	2,973,545	599,905				7,890,077
April 2006 – March 2007	3,791,724	505,644	2,394,828	819,665	178,269			7,690,130
	7,930,966	90,673	401,944					8,423,583
April 2007 – March 2008	5,769,680	1,266,266	2,361,012	428,839	201,581			10,027,378
April 2008 – March 2009	4,888,313	59,144	4,603,759	783,545	58,921			10,393,682
	545,907	369,804		335,283				1,250,994
April 2009 – March 2010	5,639,231	2,066,231	4,026,238	182,869	155,716			12,070,285
	1,034,972			476,943				1,511,915
April 2010 – March 2011	3,638,426	2,851,727	2,912,767	629,428	219,399			10,251,747

(*) Before April 1997, the manual "bucket" method was used to evaluate the quantity of dredged material on board a ship. Since April 1997, an automatic measurement device is used which allows directly evaluating the quantity of dry material on board ships. Comparison between both systems is not possible.
(**) Closed for dumping since end 2004

Table 0.4: Quantities of maintenance dredged material dumped at sea per calendar year (tonnes of dry matter).

period	Br&W S1	Br&W S2	Br&W Zeebrugge Oost	Br&W Oostende	Nieuwpoort	Total
2007	5,592,676	127,704	2,219,780	460,167	118,100	8,518,427
2008	4,589,589	80,014	4,667,225	864,863	103,541	10,305,232
2009	6,144,522	1,591,871	3,776,038	241,544	156,456	11,910,431
2010	3,642,577	2,598,212	3,342,526	304,235	179,186	10,066,736

The maps in annex are giving a visual image of the maintenance dredging and dumping intensity during 2009 and 2010. The dredging intensity gives a view on the intensity of dredging at a certain place over a defined period. They show that most places are under sedimentation. The dumping intensity gives a view on where most of the dredged material is being dumped over the surface of the dumping site. Both, dumping and dredging intensity maps are being used for validation of the mathematical models and for defining monitoring stations.

Beneficial use

To keep the access channel to Blankenberge harbour open, maintenance dredging on a regular basis is needed. Wind and current patterns cause a rapid influx of sand from the nearby beaches and a sand plate is being built up. As a consequence of this, the chemical and morphological qualities of this sand are very good. Contamination is virtually non-existent. Within the environmental legislation of the Flemish Region, re-use of dredged material as soil is possible, providing a specific certificate is delivered. Table 1.5 gives an overview of the quantities of dredged material from the access channel to Blankenberge used beneficially to reinforce coastal defence on the nearby beaches.

Table 0.5: Beneficial use of dredged material.

Period	Beneficially used dredged material (m³)
November 2007 – February 2008	69.526
May 2008 – June 2008	18.661
November 2008 – December 2008	30.884
April 2009	9.588
November 2009 – January 2010	21.354
Total	**144.013**

2 Physical aspects related to dredging and disposal operations

The morphological and sedimentological effects resulting from dredging and disposal works are numerous (OSPAR, 2008). The WFD and recently adopted MFSD (see e.g. Borja, 2005; Devlin *et al.*, 2007) identifies human induced changes in the concentration of suspended particulate matter (SPM) as one of the main pollutants. Disposal of fine-grained dredged material at sea has a varying impact on the marine environment (Nichols, 1988; Bray *et al.*, 1996; Hill *et al.*, 1999; O'Connor, 1999; Smith and Rule, 2001; Lohrer and Wetz, 2003; Simonini *et al.*, 2005, Lee *et al.*, 2010; Ware *et al.*, 2010). Deepening of channels and construction of ports increases deposition of fine-grained sediments and has as consequence an increase of maintenance dredging and thus an increase of suspended particulate matter (SPM) concentration (Truitt, 1988; Collins, 1990; Wu *et al.*, 2006; Fettweis *et al.*, 2009; Houziaux *et al.*, 2011). During slack water and after storm periods fluid mud layers may be formed by settling of suspended matter or fluidization of cohesive sediment beds (Maa and Mehta, 1987; van Kessel and Kranenburg, 1998; Li and Mehta, 2000). Massive sedimentation of fine-grained sediments in harbours and navigation channel is often related to the occurrence of fluid mud layers (Fettweis and Sas, 1999; Verlaan and Spanhoff, 2000; Winterwerp, 2005; PIANC, 2008; Van Maren *et al.*, 2009; De Nijs *et al.*, 2009).

Dredging and disposal effects are site specific and require the understanding of the site-specific dynamics in order to evaluate environmental impact of dredging and disposal works. In this chapter results carried out within the MOMO project are summarized that deal with sediment dynamics and human impact in the Belgian nearshore area and the implementation of monitoring strategies to identify environmental changes induced by these activities. In section 2.2 the natural variability of SPM concentration is described focussing on storm and meteorological effects and the cohesive and non-cohesive sediment interactions. The statistical properties of in situ and remote sensing data of SPM concentration and the use of SPM concentration as indicator of changes are presented in section 2.3. These statistical tools are applied to assess the impact of continuous disposal of fine-grained sediments from maintenance dredging works on the SPM concentration during a dredging experiment at Zeebrugge.

2.1 Regional settings and Instrumentation

2.1.1 The Belgian near shore area

The Belgian near-shore area (Figure 2.1) is characterized by sediment composition varying from pure sand to pure mud (Verfaillie *et al.,* 2006) and by high turbidity waters. The SPM concentration in the coastal turbidity maximum ranges between 0.02–0.07 g/l and reaches 0.1 to >3 g/l near the bed; lower values (<0.01 g/l) occur offshore (Fettweis *et al.,* 2010). Tidal regime is semi-diurnal and the mean tidal range at Zeebrugge is 4.3 and 2.8 m at spring and neap tide, respectively. The tidal current ellipses are elongated in the near-shore area and become gradually more semicircular towards the offshore. The current velocities near Zeebrugge vary from 0.2–1.5 m/s during spring tide and 0.2–0.6 m/s during neap tide. South-westerly winds dominate the overall wind climate, followed by winds from the NE sector. Maximum wind speeds coincide with the south-westerly winds; still, highest waves are generated under north-westerly winds. Salinity varies in the coastal zone due to wind induced advection of water masses and river discharges between 28 and 34 (Fettweis *et al.,* 2010).

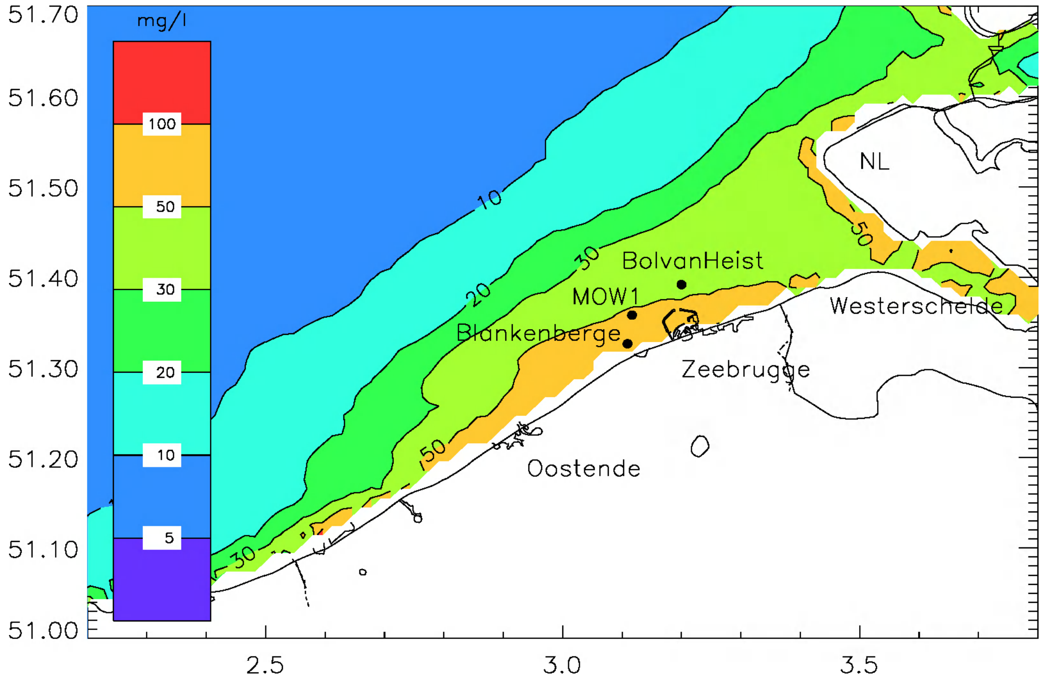

Figure 2.1: Map of the Belgian nearshore area with the in situ SPM concentration measurement station Blankenberge and MOW1, and the meteorological station Bol van Heist. The background consists of the yearly averaged surface SPM concentration (mg/l) from MODIS images (2003–2008).

2.1.2 Instrumentation and measurement data

Long-term multi-parametric measurements with a tripod (Figure 2.2) have been carried out at 2 stations (Blankenberge, MOW1), both situated in the coastal turbidity maximum near Zeebrugge. The measurement station Blankenberge is located on the eastern part of a shoreface-connected sand ridge (Wenduine Bank) at about 1 km from the shore. Sediment samples near this site show variable sediment characteristics with a median grain size of about 150 µm. The MOW1 site is situated about 5 km offshore and is characterised by the occurrence of near bed Holocene medium- consolidated mud, albeit covered with an ephemeral slightly muddy fine sand layer with a median grain size of about 170 µm. The water depth is 9 m MLLWS at MOW1 and about 5m MLLWS at Blankenberge. The tripod was developed for collecting time-series (up to 50 days) of SPM concentration, particle size distribution, salinity, temperature, pressure and current velocity at fixed locations. The instrumentation suite consisted of a 5 MHz SonTek ADV Ocean-Hydra, a 3 MHz SonTek ADP, two D&A OBS, a Sea-Bird SBE37 CT and a Sequoia Scientific LISST-100X. All data (except LISST) were stored in two SonTek Hydra data logging systems. The OBSs were mounted at 0.2 and 2 meters above the bed (hereafter referred to as mab). The ADV velocities were measured at 0.2 mab, while the ADP profiler was attached at 2.3 mab and down-looking, measuring current and acoustic intensity profiles with a bin resolution of 0.25 m. Mean values were obtained once every 10min for the OBS, LISST, and ADV, while the ADP was set to record a profile every 1 min; later on averaging was performed to a 10 min interval to match the sampling interval of the other sensors. The tripod was deployed at three locations from 2004 on. A total of 240 days of data have been collected at Blankenberge, 966 days at MOW1 (until October 2011) and 9 days at the Kwintebank (Figure 2.1). Since November 2009 tripods are continuously measuring at MOW1. The long deployments have ensured accurate sampling of conditions that include complete periods of neap and spring tides, as well as the occurrence of a variety of meteorological events.

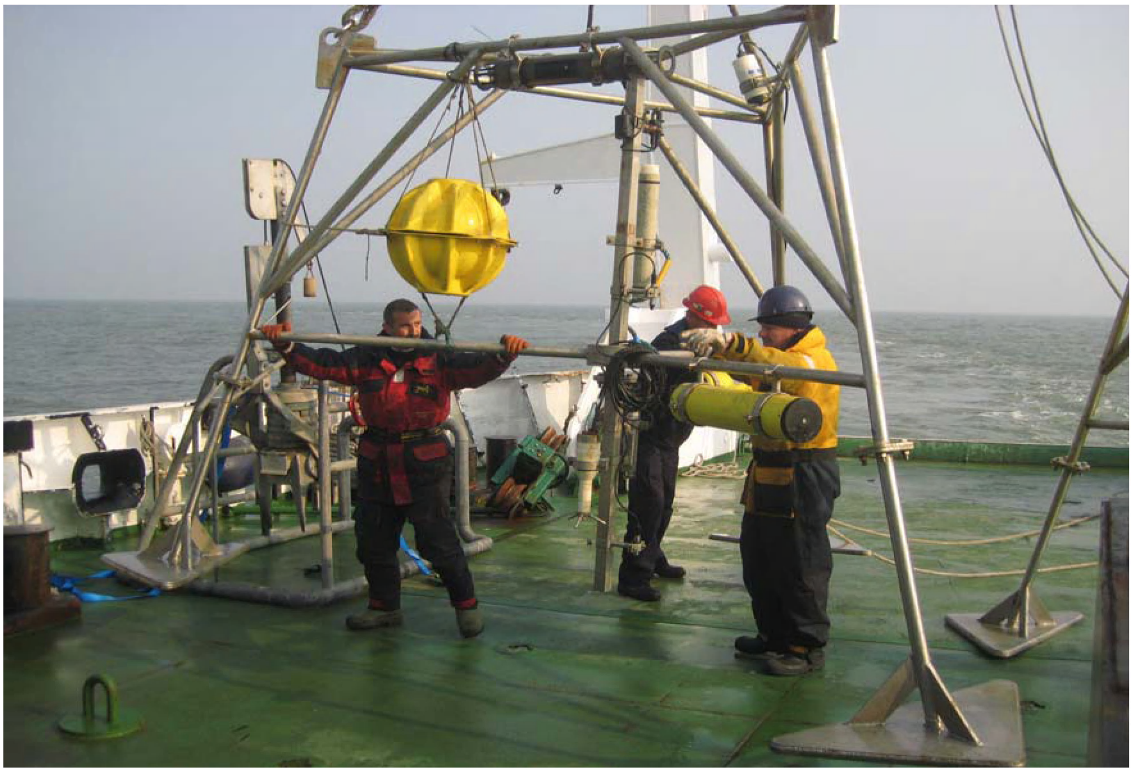

Figure 2.2: Multi-parametric tripod used for data collection at MOW1, Blankenberge and Kwintebank.

During the period 2001-2008, 16 tidal cycle measurements have been carried out at MOW1 and 8 on the Kwintebank (Figure 2.1). During the measurements, the ship remained anchored during one tidal cycle. The Sea-Bird SBE09 SCTD carousel sampling system, containing 12 10 l Niskin bottles and an OBS was kept at about 3 m above bottom (mab). Every 20 minutes a Niskin bottle was closed and every hour the carousel was taken on board of the vessel. Per retrieval of the carousel a vertical profile was measured. About 13 profiles per tidal cycle have thus been collected. In total 198 vertical profiles are available at MOW1 and 103 on the Kwintebank. The measured vertical profiles collected from the vessel during a tidal cycle cover the water column from 3 mab towards the surface. Therefore a linear regression between water depth and the logarithm of the SPM concentration, averaged over depths cells of 0.5 m, was calculated to construct the missing lower part of the profiles. The fitted profiles have been used to calculate ratios between SPM concentration at the surface and at different depths in order to extrapolate surface SPM concentration, measured by the satellite towards deeper water layers and/or near bed SPM concentration, measured by the tripod towards the surface (Fettweis and Nechad, 2011).

MODIS data of level 1A (L1A), covering the period 2003-2008, have been downloaded from the NASA GSFC web site http://oceancolor.gsfc.nasa.gov. The L1A data contain the radiance at the top of the atmosphere, which were geometrically corrected using the SeaDAS software (available from the same NASA web site). The turbid waters atmospheric correction (Ruddick *et al.,* 2000) implemented in SeaDAS was then applied to obtain the marine (water-leaving) reflectance. SPM concentrations were derived from water-leaving reflectance following an algorithm calibrated for turbid waters (Nechad *et al.,* 2010). For Belgian waters about 60 (partially) cloud free images per year are available from each sensor, resulting in total in 460 samples at MOW1 and 502 at the Kwintebank location. 64% of satellite images are during spring and summer and only 36% during autumn and winter. The latter two seasons are characterised by higher SPM concentrations.

2.2 Natural variability of sediment processes

The effect of hydro-meteorological forcing (tides, winds and storms) on the transport of SPM, on the formation of high concentrated mud suspensions (HCMS) and on the occurrence of sand-mud suspensions has been studied using long-term multi-parametric observations. The backscattered signal from OBS and ADP were used to estimate SPM concentration. The voltage of the OBS was converted to SPM concentration by calibration against filtered water samples during several field campaigns (Fettweis *et al.,* 2006). A linear regression between all OBS signals and SPM concentrations from filtration was assumed. The backscattered acoustic signal strength, from ADP, was also used to estimate SPM concentrations. After conversion to decibels, the signal strength was corrected for geometric spreading and water attenuation. Furthermore, an iterative approach (Kim *et al.,* 2004) was used to also correct for sediment attenuation. The upper OBS-derived SPM concentration estimates were used to empirically calibrate the ADP's first bin. In general, the backscattering is affected by sediment type, size and composition. All are difficult to quantify by single frequency backscatter sensors. Limitations associated with optical and acoustic instruments have been addressed in literature (Voulgaris and Meyers 2004; Vincent *et al.,* 2003; Fugate and Friedrichs 2002; Bunt *et al.,* 1999; Hamilton *et al.,* 1998; Thorne *et al.,* 1991). Briefly, the optical sensors tend to underestimate the coarser particles present in the water column. Acoustic devices produce better estimates of mass concentration than optical for the coarser fraction. Besides time series of current velocities and acoustic amplitude, the ADV and ADP was configured to also measure and store the distance between sensor and boundary (i.e., sea bed). The altimetry of the ADV and ADP was used to detect variation in bed level, as also for the identification of deposition and resuspension of fine-grained sediments. For the study site, decreasing distance between probe and bed boundary can correspond to the presence of HCMS acting as an acoustic reflector.

2.2.1 Sediment mobility during storm conditions

Between autumn 2005 and winter 2007, three storm periods have been selected with similar wave conditions from the MOW1 and Blankenberge tripod data, we refer to Fettweis *et al.* (2010) for a more detailed description. The data show that during or after a storm, the SPM concentration increases significantly and that HCMS are formed. SPM concentration is clearly related to high waves and winds (Figure 2.3). The formation of HCMS in wave-dominated areas is well documented in the scientific literature (de Wit and Kranenburg 1997; Winterwerp 1999; Li and Mehta 2000). The occurrence of fluid mud or HCMS on many continental shelves is associated with wave or current-driven sediment gravity flows off high-load rivers (Wright and Friedrichs, 2006). However, the origin of the suspended matter in the southern North Sea and in the Belgian-Dutch nearshore zone has been mainly ascribed to the inflow of fine-grained sediments through the Dover Strait (Gerritsen *et al.,* 2000), as no high-load rivers exist in the area. The fluctuation of SPM concentration with time is complex and it is not always straightforward to identify the origin of some of the variations. Wind direction and advection of water masses, previous history and availability of fine-grained sediments in fluffy layers, the very soft mud deposits around navigation channels, and the erosion of medium-consolidated mud of Holocene age influence the SPM signal. The data suggest that for the generation of very high SPM concentrations near the bed, significant amounts of fine-grained sediments have to be resuspended and/or eroded (Fettweis *et al.,* 2010).

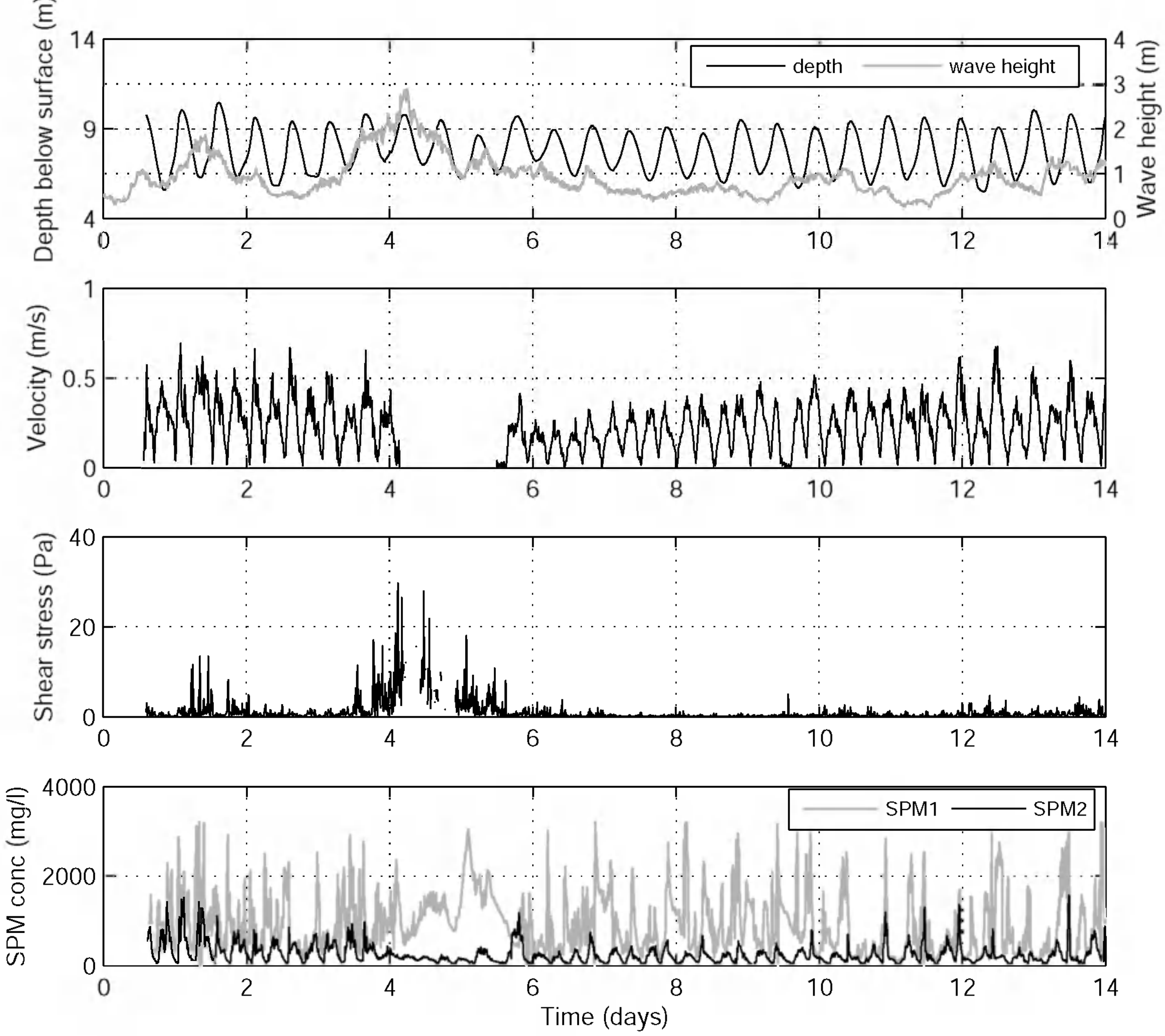

Figure 2.3: Blankenberge site, tripod measurements of 7-20 November 2006. From up to down: depth below water surface (m) and significant wave heights; ADV current velocity (m/s); shear stress (Pa) derived from the ADV; and SPM concentration at 0.2 mab (SPM1) and 2.2 mab (SPM2). Remark the formation of HCMS (SPM1) between day 4-6 corresponding to the occurrence of a storm.

The largest reservoir of fine-grained sediments in the nearshore area consists of the medium-consolidated Holocene mud (Fettweis *et al.,* 2009). Erosion behaviour measurements (Fettweis *et al.*, 2010; Van Lancker *et al.*, 2011) confirm that consolidated cohesive bed layers are difficult to erode by only fluid-transmitted stress as the critical erosion shear stress is about 10 Pa. Near bed shear stresses derived from the ADV data amount up to 40 Pa (Figure 2.3). These values indicate that erosion of Holocene mud by fluid-transmitted shear stress can occur and that SPM could have been released into the water column from the Holocene mud fields under storm conditions. Still, in literature, it is well documented that a cohesive mud bed may be eroded and fluidised by waves (Maa and Mehta 1987; De Wit and Kranenburg 1997; Li and Mehta 2000; Silva-Jacinto and Le Hir 2001). The stresses induced by waves modify the strength of the bed and thus also the erodibility of the sediments. Mud pebbles are an indication of this type of erosion; they have been observed regularly in the area of investigation (Fettweis *et al.,* 2009). Other important erosion mechanisms are due to the mutual interaction of cohesive and non-cohesive sediments (Le Hir *et al.,* 2007): one can distinguish between sand grains moving on a cohesive substrate and erosion of mixed sediments. Thin sand layers on top of Holocene mud layers and mixed sediments have often been observed in the turbidity maximum area and could therefore act as a reservoir for fine-grained material that is only resuspendable under more extreme meteorological conditions.

Numerical model results indicate that, under normal conditions, the bed shear stress in the navigation channels and at location MOW1 and Blankenberge are lower than 4 Pa. The critical erosion shear stresses of in-situ samples in the navigation channels are, below the fluffy surface layer, generally higher than 4 Pa. The deposits of fresh mud below the fluffy layer in these areas forms thus a reservoir of SPM that will only be resuspended during periods with high shear stresses, e.g. caused by storms. The data suggest that a major part of the HCMS, measured at both sites, could have been resuspended from the very soft mud deposits in the navigation channels and adjacent areas. This result is important as it suggest that the deepening of the navigation channels has made available fine-grained matter that contributes significantly to the formation of high concentration mud suspensions.

2.2.2 Sediment mobility in response to tidal and wind-driven flow

The collected current velocity (ADV and ADP) time series from tripod measurements were filtered for the tidal signal using a low-pass filter for periods less than 33 hours and decomposed in an along and cross-shore component (Baeye *et al.,* 2011). The alongshore low-passed flow was used to classify the tidal cycle in terms of wind-driven flow. For each tidal cycle the average value of the alongshore low-passed flow was estimated and subsequently used to characterize the tidal cycle in terms of wind-driven flow. Case I corresponds with purely tidal forcing conditions. The remaining tidal cycles correspond to periods with significant influence of wind-driven flows. Negative values correspond to flows towards the SW, driven by NW-NE winds (Case II), while positive subtidal flows are directed to the NE, corresponding to wind forcing from the SW (Case III). In addition to the above classification, each tidal cycle was classified as neap or spring, in terms of the tidal range of the particular cycle. This classification has resulted in a total of 6 categories of tidal cycles where each category represents both tidal and wind forcing. Tidal cycles, from each category, were ensemble-averaged to create a “typical” representative tidal cycle for each case as described by Murphy and Voulgaris (2006).

Tripod measurements show that near bed hydrodynamics and sediment dynamics, although dominated by tidal forcing, are significantly modified by wind-induced flows with different effects, depending on the wind direction. SPM concentration measurements, under tidal forcing only, show concentration maxima occurring at the end of ebb and at the beginning of flood. Near bed hydrodynamics and SPM dynamics are predominantly dominated by tidal forcing. Generally, SPM concentration is significantly influenced by advection during ebb, whereas during flood local resuspension is more important (Figure 2.4).

SPM measurements, from OBS show that wind-driven alongshore advection has a significant influence on SPM concentration. A significant modification of the tidal forcing results from alongshore advection due to wind-induced flows and influences the position of the turbidity maximum; as such also the origin of SPM. Winds persistently blowing from the NE will increase SPM concentration, due to an increased SPM outflow from the Westerschelde estuary. SW winds will decrease SPM concentrations. The latter is related to the advection of less turbid English Channel water to the measuring location, inducing a shift of the turbidity maximum towards the NE and the Westerschelde estuary. Under these conditions, marine mud will be imported and buffered in the estuary.

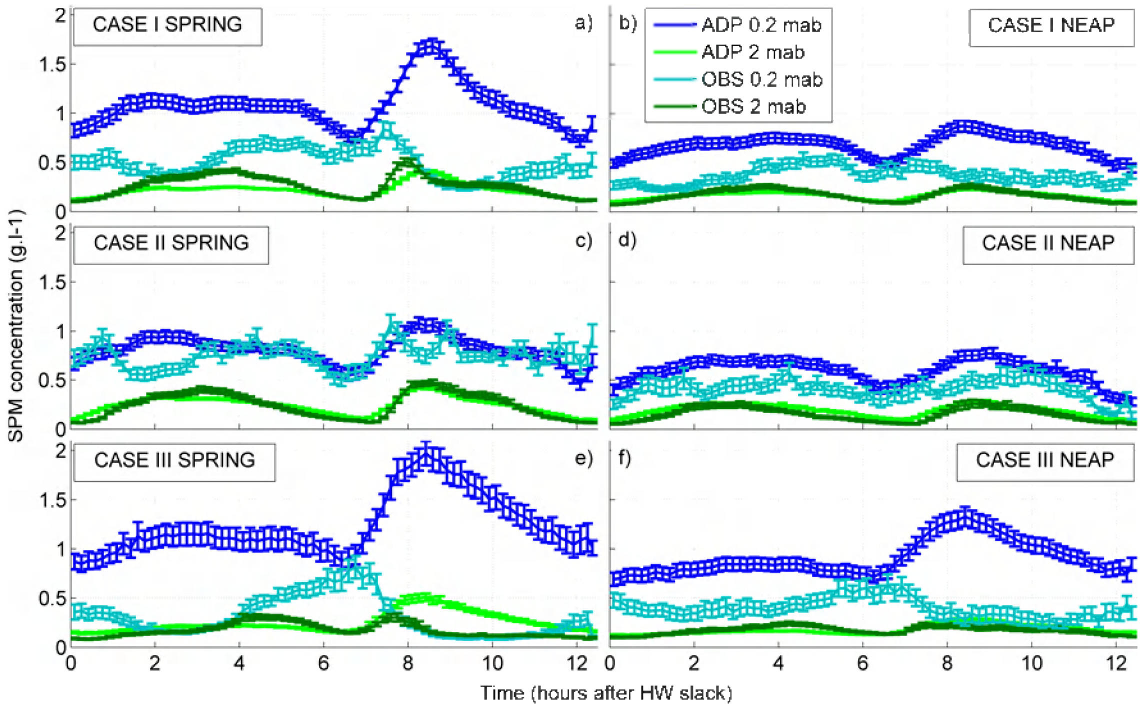

Figure 2.4: Ensemble-averages of tidally varying SPM estimates from OBS and ADP at 0.2 (blue) and 2 mab (green). The error bars indicate standard error. (Case I: no wind, case II: NW-NE wind, case III: SW wind).

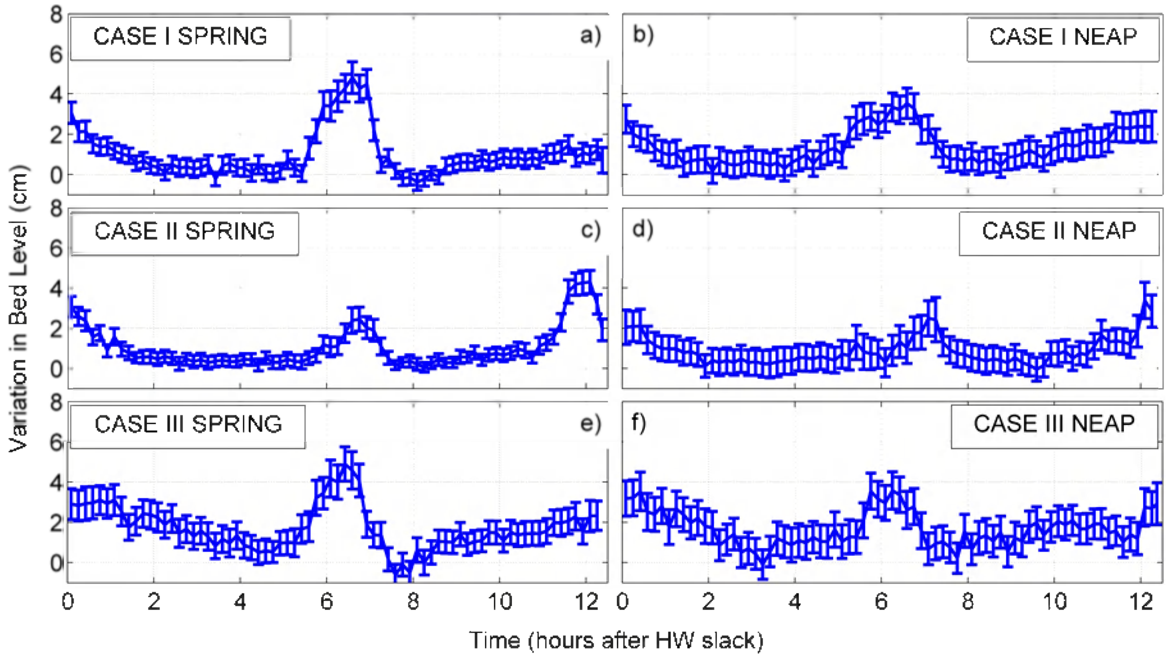

Figure 2.5: Averaged sea bed level change derived from ADV altimetry with standard error bars. (Case I: no wind, Case II: NW-NE wind, Case III: SW wind). An increase of sea bed around slack water (0h, 6h, 12.5h) indicates the formation of HCMS.

Altimetry data, derived from ADV, show bed level variations that can be explained by the formation of HCMS (Figure 2.5). During low wind (Case I) and SW wind (Case III) conditions their occurrence is limited to slack water periods. SPM consists of a mixture of cohesive sediments (flocs) and locally eroded sand grains during high currents (see also section 2.2.3). With prevailing northern winds (Case II), the increase in SPM concentration results in the formation of persistent HCMS. The damping of turbulence by HCMS layers is a major mechanism maintaining these layers during longer time periods (Sheremet *et al.*, 2005; Reed *et al.*, 2009). The results have indicated that these layers mostly remain present throughout the tidal cycle. Inverse armouring occurs, as the sandy bed is sheltered from erosion. SPM consists of cohesive sediments only. These winds are not very frequent and their wind speeds are rather reduced. However, we believe that this type of benthic sediment transport is very important and has implications for object burial and sediment recirculation in and around the port of Zeebrugge.

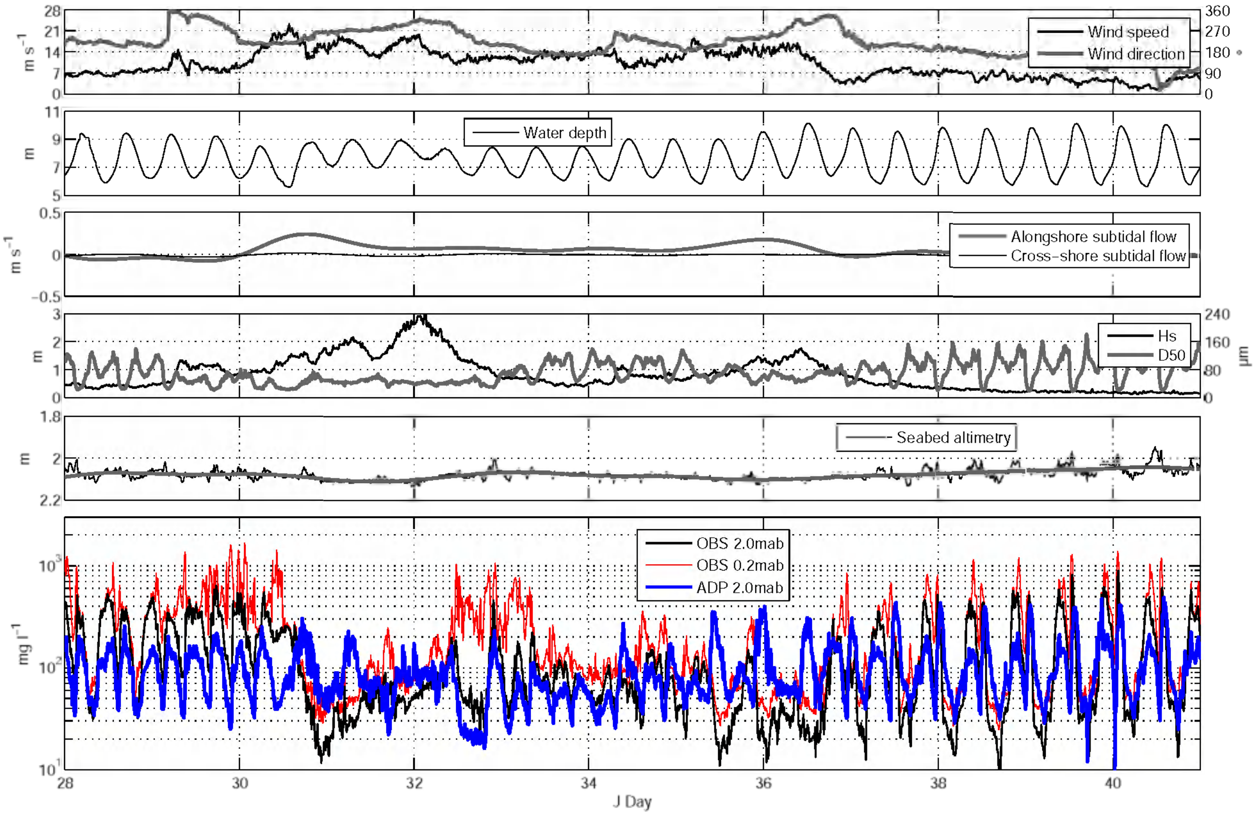

Figure 2.6: January-February 2008 deployment at Blankenberge (28/01 15:38 - 11/02 13:40). Time series of from up to down wind velocity and direction (wind from 0°=N, 90°=E, 180°=S, 270°=W) at Bol van Heist; water depth; subtidal alongshore (positive towards the NE, negative towards the SW) and crosshore flow (positive onshore, negative offshore); significant wave height (H_s); D50 of PSD; seabed altimetry; and SPM concentration from OBS and ADP.

2.2.3 Cohesive and non-cohesive SPM dynamics

Manning *et al.* (2010) have shown under laboratory conditions that flocculation influences the deposition and settling of sand/mud mixtures. Little data are available that deal with resuspension of sand, mud and sand-mud mixtures in natural environments. The aim was therefore to present in situ measurements in an area with mixed bed sediments and to examine the influence of tides, wind and wave effects on the particle size distribution (PSD). PSD provide essential information on floc and particle dynamics as emphasized by Mikkelsen *et al.* (2006). Statistical methods (entropy analysis, fitting of PSD with sum of log-normal functions) and ensemble averaging to classify the PSDs and to establish the link between PSD and the underlying processes have been applied. Similar measuring approaches (e.g. Thorne and Hanes 2002; Fugate and Friedrichs 2002; Voulgaris and Meyers 2004; Hoitink and Hoekstra 2005), statistical methods (Jonasz and Fournier 1996; Mikkelsen *et al.*, 2007) and averaging (Murphy and Voulgaris 2006; Baeye *et al.*, 2011) have been

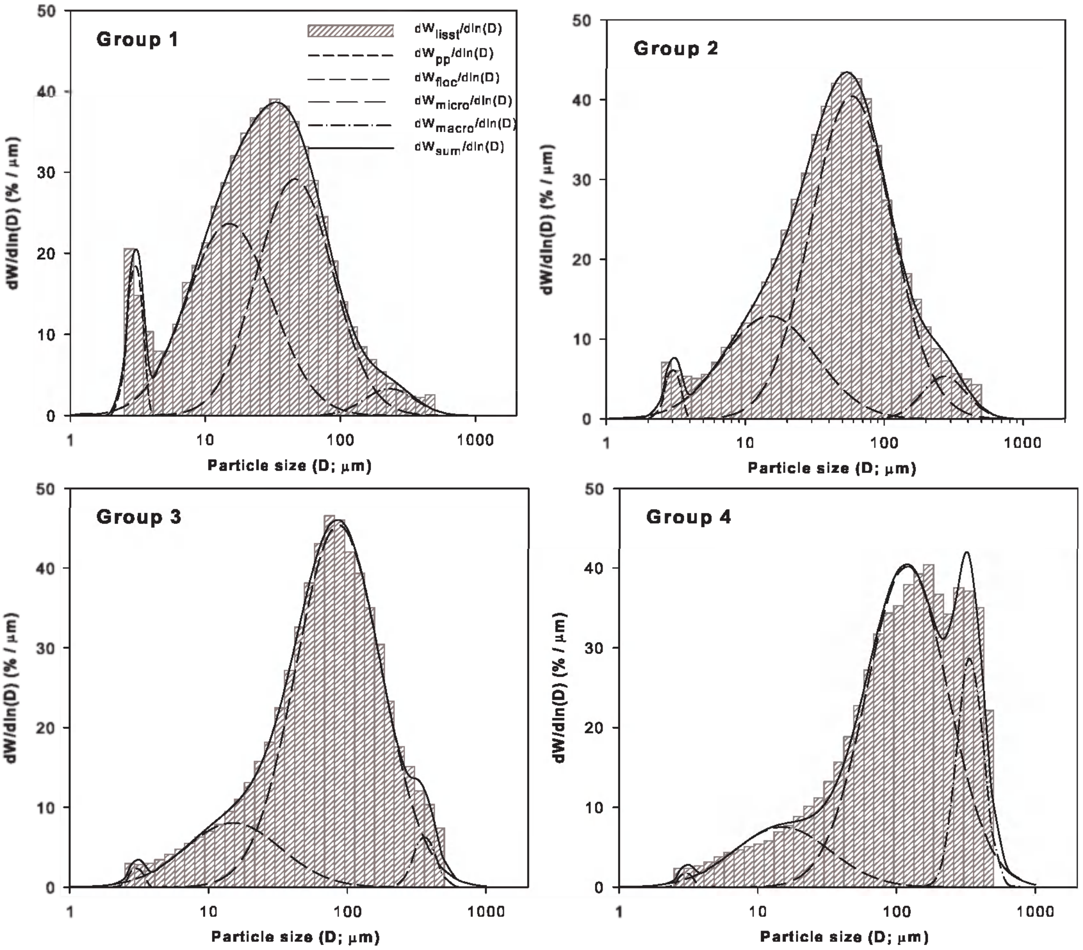

Figure 2.7: January–February and March–April 2008 deployments: averaged PSD for the four entropy groups, $dW_{lisst}/dln(D)$, together with the fitted sum of the four log-normal functions, dW/dln(D).

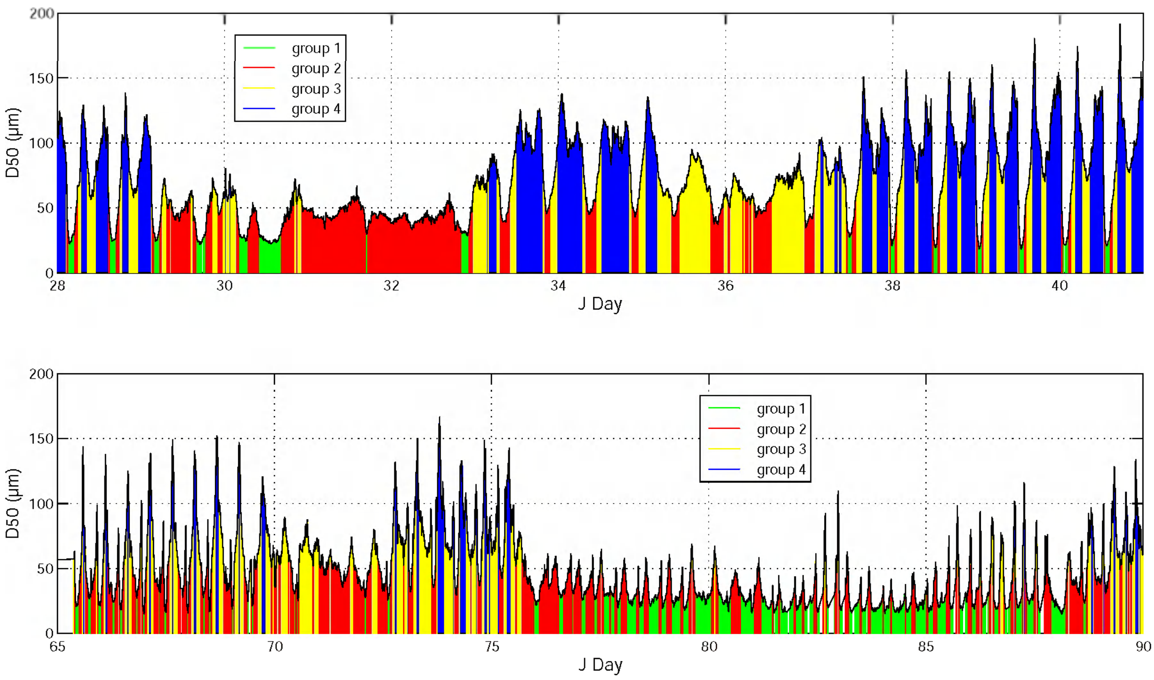

Figure 2.8: January–February and March–April 2008 deployments: temporal distributions of the four entropy groups.

successfully adopted in various marine environments. Entropy analysis evaluates the randomness of an event (such as a particle size distribution) and assigns the event to a group with similar characteristics. Applied to PSD entropy analysis allows grouping the size spectra without assumptions about the shape of the spectra and is therefore suited for analysis of uni-modal, bi-modal as well as multi-modal distributions (Woolfe *et al.,* 1998).

Analysis of the PSDs together with the interpretation of acoustic and optical derived SPM concentration and the altimetry allowed identifying differences in composition of the SPM during storm conditions (Fettweis *et al.*, 2012). The data show the importance of wind induced advection, alongshore subtidal flow and the formation of HCMS as regulating mechanism of SPM concentration, SPM characteristics (cohesiveness or composition of sediment particles) and size distribution in a high turbidity area rather than solely turbulence shear from currents and waves. The former is the clue to explain the different behaviour of SPM concentration from ADP and OBS and the observed differences in PSD during different storms, see Figure 2.6-2.8. Wind direction and strength influences the subtidal alongshore flow and results in an advection of the coastal turbidity maximum. This results in an increase of cohesive SPM concentration, the formation of HCMS and the armouring of sand during northern storms (Case II, see section 2.2.2); or in a decrease of cohesive SPM concentration, no HCMS and increase of sand and silt in suspension during SW storms (Case III, see section 2.2.2). Particle size distributions are generally multimodal and consist of primary particles, flocculi, microflocs and macroflocs. During SW storms (Case III) PSD are, however, uni-modal and consist of mainly granular material (silt, sand) resuspended from the sea bed, whereas during northern storms flocculated material is still being identified from the PSDs. The data suggest that two populations of primary particles (<3 µm and flocculi 15 µm) co-exist and are the building blocks of flocs. Flocculi consist of clusters of clay minerals, whereas primary particles are of various compositions (calcite, clays). Mixed sediments have been found in suspension at 2 mab during maximum flood currents at spring tide and during storms. At the measuring location this occurs more prominently during winds blowing from SW-W (Case III). The size distribution of the local bed sediments is thus only influencing the size distribution of the SPM when HCMS are not present and when turbulence induced by currents (max flood currents) or waves is strong enough to bring sand or silt into suspension.

2.3 SPM concentration as indicator to detect changes in the marine environment

SPM dynamics is complex and is affected by external factors, such as hydrodynamics, waves, availability of SPM sources, biological processes, flocculation, deposition and resuspension. In order to assess its variability, SPM concentration has been defined as a statistical population. By doing so the measured SPM concentration time series can be considered as sub-samples that are characterised by statistical properties, such as median, geometrical mean, standard deviation and probability density distribution. The probability density distributions of the different sub-samples, consisting of the different time series or other sub-samples, were therefore fitted using log-normal distributions, and the χ^2 test probability calculated to assess how well the distribution fits a log-normal one. By doing so statistical properties can be calculated so that inferences or extrapolations from the sub-sample to the population can be made (section 2.3.1). E.g. if the data series collected during different periods have similar log-normal distributions, geometric means and standard deviations, then we could conclude that - within the range of natural variability and measuring uncertainties - these data series represent similar sub-samples of the whole SPM concentration population. Consequently, if a human activity, such as disposal of dredged material, has a significant impact on SPM concentration then this should be detectable in the differences between the statistical parameters of the sub-sample, collected during the dredging experiment and of the whole population (section 2.3.2). The statistical approach provides a tool to account for the complexities associated with natural dynamics and the need to evaluate human impact, quantitatively.

2.3.1 SPM concentration as indicator

In order to evaluate SPM concentration as indicator of detecting changes in the marine environment, a large set of SPM concentration data from MODIS and from in situ measurements (vessel, tripod) was used to evaluate temporal SPM heterogeneity in the Belgian nearshore, see Fettweis and Nechad (2011) for more detailed information. The heterogeneity has been statistically assessed by comparing the SPM concentration frequency distributions. Based on the median and the standard deviation, a representative population of SPM concentration was constructed under different conditions. The probability distributions of the SPM concentration data correspond well with a log-normal distribution (Figure 2.9). The median (x*) and multiplicative standard deviation (s*) of these distributions have been calculated and can be used as a statistically representative background SPM concentration. In general, values of s* vary between 1.5 and 2.9; hence they are included in the most frequently occurring range of approximately 1.4 to 3, observed in various branches of natural sciences (Limpert *et al.*, 2001).

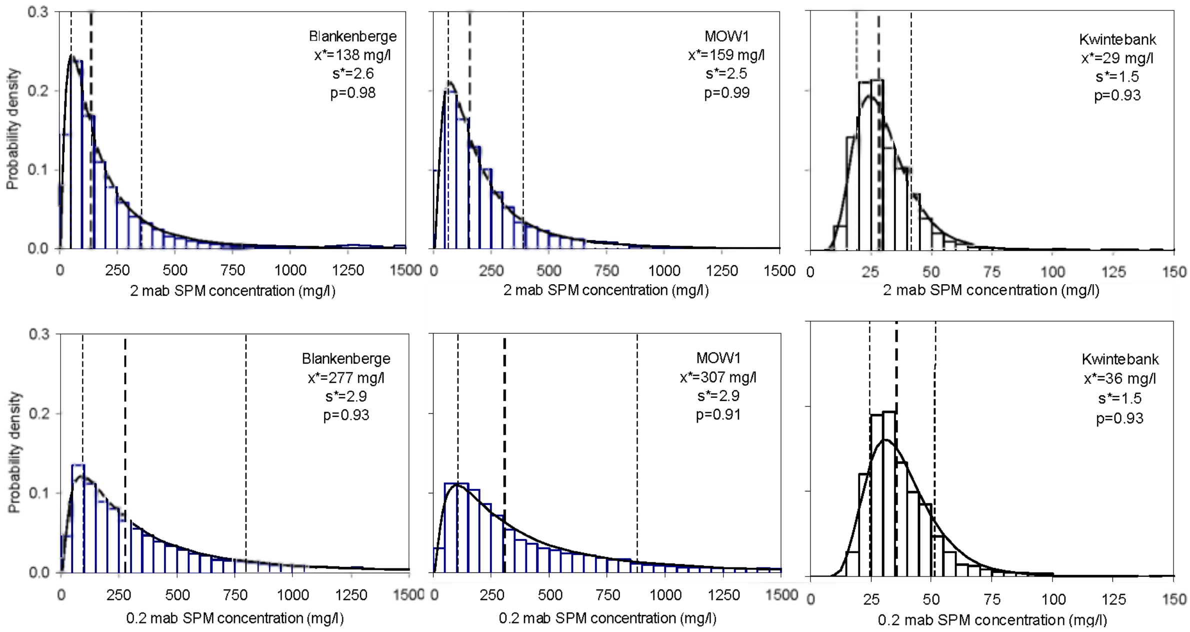

Figure 2.9: Probability density distribution of long-term SPM concentration data at 2 mab and 0.2 mab for the Blankenberge, MOW1 and Kwintebank sites and the corresponding log-normal probability density functions. The data fit the log-normal distribution with a X^2 test probability of $p>0.9$. The dashed lines correspond to the median x* times/over the multiplicative standard deviation s*.

It was found that sampling method (continuous vs. daily) and sampling technique (tripod: in situ near-bed, vessel: in situ water column, MODIS: remote sensing) results in different distributions and that they represent a different sub-population of the whole SPM concentrations population. In order to compare near-bed SPM concentrations from tripod with surface concentration from satellite, correction factors have been constructed based on vertical SPM concentration profiles measured during tidal cycles. The differences between the data sets are related to the different meteorological conditions during measurements; to near bed SPM concentration dynamics, which are partially uncoupled from processes higher up in the water column; to the sampling methods or schemes used to collect the data; to the method of surface correction assuming a logarithmic profiles near the bed and to measuring uncertainties.

2.3.2 Monitoring the effect of disposal of fine-grained sediments on the SPM concentration

The impact of continuous disposal of fine-grained sediments from maintenance dredging works on the suspended particulate matter concentration in the Belgian nearshore area was investigated during a dredging experiment at Zeebrugge (Fettweis *et al.,* 2011a). Before, during and after the experiment, monitoring of SPM concentration using OBS and ADV altimetry was carried out at the Blankenberge location using a multi-parametric tripod, situated about 5 km west of the disposal site. A statistical analysis, based on the concept of populations and sub-sampling, was applied to evaluate the effect. The method provides a tool to account for the complexities associated with natural dynamics and the need to evaluate quantitatively human impact. The measurements indicated that SPM concentration has a very high natural variability (min-max: 10 - >3300 mg/l). SPM concentration near the bed (0.2 mab) were exceptionally high (median was more than 2 times higher) during the dredging experiment. Waves were not identified as being responsible for the high SPM concentrations. During the dredging experiment, a generally higher SPM concentration near the bed during ebb and at 2 mab during flood was observed, suggesting that the disposed material was mainly transported in the benthic layer. The time lag between high wave heights and high SPM concentration suggests further that SPM has been advected towards the measuring location rather than eroded locally. We can conclude that the disposal results in a long-term increase of SPM concentration near the bed at the measuring location (Figure 2.10). This, together with ADV altimetry data, suggest that HCMS have been formed during the whole of the disposal experiment, rather than being limited to neap tidal or storm conditions, as observed during the non-disposal periods.

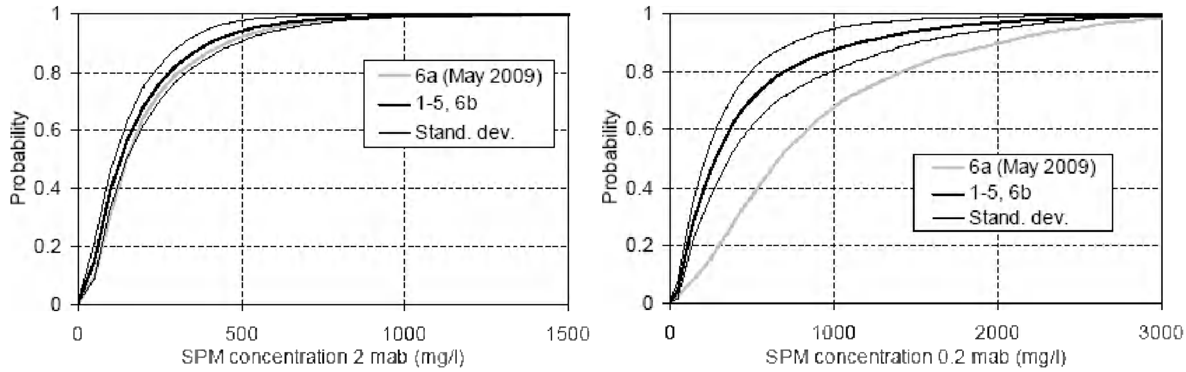

Figure 2.10: Cumulative probability distribution of SPM concentration measured at 2 mab and 0.2 mab. The black line (1 to 5, 6b) shows the data during the non-disposal periods ± one standard deviation (thin black lines); the grey line shows the distributions during the dredging experiment (6a).

2.4 Conclusions

Main conclusions, with relevance to sediment dynamics, influence of dredging and disposal operations and monitoring strategies are:

1. The effect of winds on SPM concentration is variable and depends also on the wind direction and the availability of muddy sediments. Our data show that high SPM concentrations are often more closely related to advection rather than instantaneous bed shear stress. This confirms the idea that the Belgian coastal area can be seen as a congestion in the residual SPM transport of the southern North Sea, rather than an important source of sediments.
2. The deposits of fresh mud below the fluffy layer in the navigation channels forms a reservoir of SPM that will only be resuspended during periods with high shear stresses, e.g. caused by storms. The data suggest that a major part of the HCMS could have been resuspended from the very soft mud deposits in the navigation channels and adjacent areas. This result is important as it suggest that the deepening of the navigation channels has made available fine-grained matter that contributes significantly to the formation of high concentration mud suspensions.
3. Particle size distributions are generally multimodal and consist of primary particles, flocculi, microflocs and macroflocs. The data suggest that two populations of primary particles (<3 µm and flocculi 15 µm) co-exist and are the building blocks of flocs. Flocculi consist of clusters of clay minerals, whereas primary particles are of various composition (calcite, clays).
4. Mixed sediments have been found in suspension at 2 mab during maximum flood currents at spring tide and during storms. Near Zeebrugge this occurs more prominently during winds blowing from SW-W. The size distribution of the local bed sediments is thus only influencing the size distribution of the SPM when HCMS are not present and when turbulence induced by currents (max flood currents) or waves is strong enough to bring sand or silt into suspension.
5. SPM concentration can be used as an indicator of environmental changes, if sufficiently long time series are available that are representative of the natural variability. Due to the time and spatial variability of SPM concentration in high turbidity coastal areas, greater sampling efforts are necessary, as compared to offshore systems with low SPM concentration.
6. Satellites or low-frequent tidal cycle measurements cannot replace long-term continuous measurements in high turbidity areas, which include all sea state conditions. The former datasets consist of a sub-set of the population biased towards good weather condition or spring-summer seasons (satellite). Sediment transport based on these data will thus always underestimate reality.
7. Regarding instrumentation, it is shown that SPM estimates from OBS are only reliable when SPM consists of cohesive sediments only; with mixtures of cohesive and non-cohesive sediments, a combination of optical (OBS) and acoustic sensors (ADP, ADV) are needed to get an accurate estimate of the total SPM concentration.
8. An method was used based on the concept of statistical populations in order to evaluate the effects of disposal operations on SPM concentration. The method provides a tool to account for the complexities associated with natural dynamics and the need to evaluate quantitatively human impact. Disposal results in an increase of SPM concentration near the bed. This together with ADV altimetry suggest that fluid mud layers have been formed during the disposal experiment rather than being limited to neap tidal or storm conditions as observed during the other periods.

3 Biological and chemical effects of dredged material disposal

3.1 Introduction

It is important to investigate the effects of dumping of dredged material in the marine environment from ecosystem perspective, because it could lead to different responses of the ecosystem (Figure 3.1). The dumping itself has its consequences on the state of the water column and the seabed. The main effects within the water column lead to changes in the turbidity, which could be responsible for changes in the primary production and the release of materials (contaminants and organic material). The sediment input towards the seabed leads to habitat creation/modification, smothering of the fauna and changes in bathymetry. A side effect could be the influx of organisms, especially when the fauna in the dredging area is different from the dumping area. The ILVO dredging monitoring program focused on evaluating the following impacts:

- The effect of changes in the contaminants in the sediment and fauna at the disposal sites (bioaccumulation and bio response).
- The effect of the dumping activity on the benthic organisms and their direct predators (epibenthos and demersal fish). Where they negatively affected, due to smothering or niche removal or even favoured by changes in the habitat complexity.
- The effect of the influx of organisms from the dredging areas on the native fauna at the disposal sites.

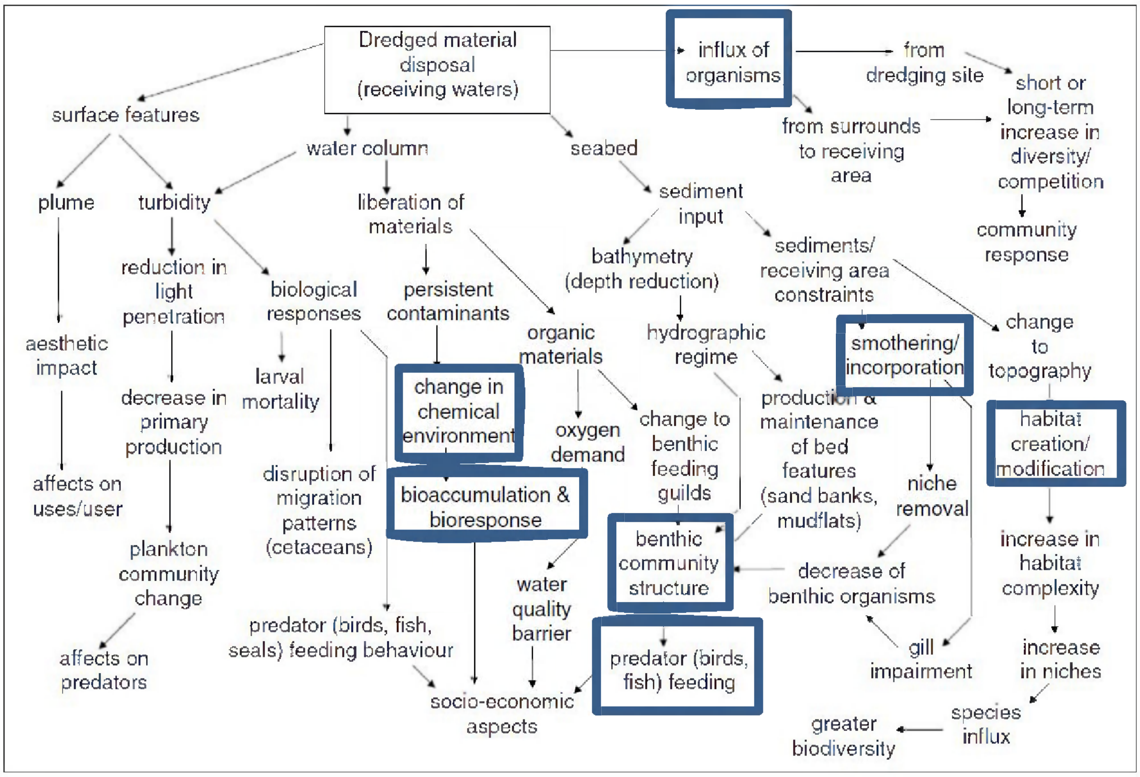

Figure 3.1: The potential environmental impacts of marine dredged material disposal – a conceptual model with indication of the research focus points in the ILVO monitoring program (adapted from Elliott and Hemingway, 2002).

This chapter outlines the results of the research done at the disposal sites by ILVO-fisheries in the context of the study 'biological, chemical and biochemical monitoring of sediment and bottom fauna at the dredged disposal sites of the Belgian Coast' (*cf.* protocol ILVO and MOW-aMT of 5 September 2003) over the period 2009-2010. This study focuses on the evaluation of the above mentioned impacts. In this period, the following tasks were set and the results of it described in this chapter:

- The monitoring program, investigating the effects of dumping of dredged material on the benthic fauna, is running for almost 30 years. The benthic sampling program strongly changed over this period, with a clear impact/control sampling strategy in the last years. A long term analysis of the changes in the benthic macro fauna in relation to the dumping intensity was executed and is descript in section 3.2.
- The results of the regular ILVO monitoring program over the period 2009-2010 are outlined in section 3.3. We describe the differences observed in the biological community parameters of macrobenthos, epibenthos and demersal fish, if possible based on indicator assessments. The level and trends in heavy metals, PCBs, OCPs and PAKs in sediment and biota were measured within the chemical monitoring part. The histo-pathological part focuses on the inventarisation of fish diseases in commercial fish species. Finally, the biochemical part investigates stress-indicators in juvenile dab to see whether the dredged material disposal has repercussion on the safety of the food chain (fish).
- We also investigated the biological life, focusing on the benthos, within the dredging areas of Zeebrugge (section 3.4). The aim was to analyse a possible difference in fauna of the dredging areas and the dumping areas and to examine whether an influx of organisms can occur.
- In section 3.5, we outline the optimisation of the sampling strategy and the sampling handling, with the intention to make the monitoring program more time- and cost efficient and to meet the international sampling and analyse criteria.

3.2 Long term impact of dredged material dumping on the benthic habitats in the BCZ

3.2.1 Introduction

Macrobenthic species are closely associated with the sediment and play an important role in marine ecosystems. They clearly reflect environmental changes in and near the bottom of the sea (Bilyard, 1987). Therefore, they have been shown to serve as valuable bio-indicators to monitor human activities, such as the disposal of dredged material (Wildish and Thomas, 1985; Bilyard, 1987; Soule *et al.*, 1988; Rees *et al.*, 1992; Simonini *et al.*, 2005; Bolam *et al.*, 2006; Rees *et al.*, 2006). A possible impact may be merely of a physical nature, since organisms get buried immediately after a dumping event (cf. changes in foraging capacity, mobility,... Morton, 1977) (Figure 3.1). Additionally, the properties of the dumped sediments (e.g. mud content, median grain size), could contribute to a biological impact (Maurer *et al.*, 1981a; 1981b; 1982). Since benthic species differ in susceptibility towards disturbance, the results of impact evaluation studies will be determined by the benthic community present at a given dumping site. More specifically, it is indicated that species assemblages, adapted to a certain degree of stress, will recover faster after a disturbance event than assemblages occurring in an unstressed environment. The former ones are usually characterised by life-history traits, facilitating the recolonisation process (opportunistic species) (Bolam *et al.*, 2003). The sampling design (number of samples and the degree of coverage of the study area) and the selected parameters contribute to the reliability of the obtained results. All these factors, together with the natural variability of the benthic communities, need to be taken into account, when a conclusion regarding the impact is drawn.

ILVO has a monitoring program to follow up disposal activities at the BPNS. Since 1979, two annual sampling campaigns are carried out, providing a large set of biological and sediment data, which were used for this study. We distinguish three sampling strategies in the history of the monitoring program. From 1979 until 2003, samples were taken at a limited number of stations on the BPNS. Each of these stations functioned as a reference station for nearby disposal sites, although not all disposal sites were operational. One exception is dumping site Br&W S2, since it already had a monitoring point inside the dumping zone since 1979. In 2004-2005, the former strategy was exchanged towards a control-impact strategy. From then onwards, each dumping site had one internal station (=disposal station) and one or two stations outside (=nearby-reference station) the dumping area. At the same time, additional sampling continued at the previously used reference stations. Since 2006, the sampling intensity was markedly increased (7 impact samples; 4-6 nearby reference samples), in function of obtaining a higher confidence of the impact assessment. The aim of this section is to address two main questions and is a summary of Vyshal (2010):

- What is the relationship between the degree of dumping intensity on the one hand and the presence and diversity of the marine benthos on the other hand?
- Do the different sampling designs have a different capacity in reflecting a possible impact of the dumping activity? Where the selected reference stations optimal for impact evaluation?

3.2.2 Material and Methods

The samples used in this study were collected from (1) stations associated with current designated dumping sites (Nieuwpoort, Br&W Oostende, Br&W Zeebrugge Oost, Br&W S1 and Br&W S2) (2) stations associated with two former dumping sites (S3 and R4) and (3) 12 coastal reference stations on the BPNS (Table 3.1, Figure 3.2). The sampling of each station did not start at the same time. In addition, the sampling design did change over the years, as described in detail in Table 3.1.

Macrobenthic samples were collected with a Van Veen grab (0.1 m²) mainly on board of the R/V Belgica. They were immediately fixed with an 8% formaldehyde seawater solution. The samples were sieved on a 1mm sieve. The residue was stained with eosin in order to facilitate further sorting. Species were identified to species level, if possible and counted.

A separate Van Veen sample was taken at each station for granulometric analysis of the sediment, over the period 1979-2005. Since 2006 a single core was taken from each Van Veen sample for further particle size analysis. These samples were dried in an oven at 60°C. From 1979 to 2006, a sieve tower was used for sediment analyses. Since 2007, sediment analyses were conducted using the Malvern Mastersizer 2000 analyzer following a standardized protocol.

The species dataset was standardised by lumping some species (Cirratulidae spp., Spio spp, Anthozoa spp.), and reduced by excluding species that did not belong to the macrobenthos sensu strictu (e.g. Mysida). Nematoda were excluded because of inadequate sampling techniques for quantifying meiofauna.

The multivariate analyses were performed by using version 6 of the PRIMER software package (Plymouth Routines, In: Multivariate Ecological Research, Clarke and Gorley, 2006). Prior to specific analyses, the few outlier samples (10) were identified and removed by conducting a non-metric Multi-Dimensional Scaling (nmMDS). Biological data was fourth-root transformed and the resulting dataset was used to create a Bray-Curtis similarity matrix. In order to create significantly different groups for further analyses, a group average cluster analysis with SIMPROF test (Similarity profile, test for structure in the data) was performed. Characteristic species for the different clusters were identified using the SIMPER function (Similarity/distance percentages, species/variable contributions). Analysis of similarity (ANOSIM) was used to detect possible differences in species composition within and between the sampled locations for four possible structuring factors; cluster name, year, season (winter, autumn) and station type (impact, nearby and far away control stations). In addition, a 2-d ordination plot from the nmMDSs provided a visual representation.

Table 3.1: Information about dumping sites and reference stations. Between brackets, the number of replicates

			Sampling period:		
		Function	1979-2003	2004-2005	2006-2010
Dumping sites	Nieuwpoort	Disposal		2251(3*)	LNP.01-LNP.07
		Nearby Reference		2252(3)&2253(3)	LNP.08-LNP.11
	Br&W Oostende	Disposal		1401(3)	LOO.01-LOO.07
		Nearby Reference		1402(3)	LOO.08-LOO.13
	Br&W Zeebrugge Oost	Disposal		7001(3)	LZO.01-LZO.07
		Nearby Reference		7002(3)	LZO.08-LZO.13
	Br&W S1	Disposal		78001-78011(1)	LS1.01-LS1.11
		Nearby Reference		78012-78016(1)	LS1.17-LS1.22
	Br&W S2	Disposal	7101(3)	7101(3)	LS2.01-LS1.07
		Nearby Reference		7102(3)	LS2.08-LS2.11
	Station Code	Sampling period	Function (historic)		
Far Reference Stations	115	2005-2009	Ref. Nieuwpoort		
	120	1979-2009	Ref. Nieuwpoort		
	140	1979-2008	Ref. Br&W Oostende		
	150	2000-2008	Ref. Br&W Zeebrugge Oost		
	230	2000-2009	Ref. Oostendebank		
	250	1997-2008	Ref. Steendiep		
	315	1997-1998, 2004-2009	Ref. Oost Dyck		
	780	1983-2008	Ref. Br&W S1		
	B031	2000-2008	<2004: Disposal S3, >2004: Ref Br&W S2		
	B032	2004-2008	<2004: NRef S3, >2004: Ref Br&W S2		
	B041	1997-2008	<2004: Disposal R4, >2004: Ref Br&W S2		
	B042	2004-2008	<2004: NRef R4, >2004: Ref Br&W S2		
	B08	1997-2008	Ref. Vlakte van de Raan		
	ZEB	2004-2005	Ref. Zeebrugge Eb		
	ZVL	1979-2008	Ref Br&W Zeebrugge Oost		

Univariate analyses were executed with the software package R, version 2.10.1. The parameters used were; density, species number (S) and Hill's (1973) diversity and evenness indices (N1 and N2), computed from the standardised dataset. ANOVA analyses were carried out in order to detect significant differences in space and time. Post-hoc analysis was performed with the TukeyHSD-test. To meet the assumptions for parametric analysis, data was log-transformed where needed. If assumptions for parametric tests were not met, a

Kruskal-Wallis test was performed, followed by the Wilcoxon-test when the former did identify significant p-values.

Finally, the Benthic Ecosystem Quality Index (BEQI, www.beqi.eu) was used to scale the relationship between dumping intensity and the degree of impact. The BEQI level 3 analyses were based on the parameters, total density (ind/m²), number of species and similarity (Bray-Curtis similarity of fourth-root transformed density data). For this analysis, the disposal stations were compared with the two types of references stations (nearby-reference and far-reference). Together with the BEQI-analysis, a power assessment was performed. The power gives an indication of the chance to detect any impact and is defined as 1-β (β is the probability of a type II-error). The result of this post-hoc power analysis will depend on the variance, the effect size and the choice of the level of significance (here set at 0.05%) (Van Hoey *et al.*, 2007).

Table 3.2: Biological and sedimentological characterisation of the cluster groups, with assignation of the habitat types to each cluster. The number of samples of each station within each cluster is written between brackets.

	group 1	group 2	group 3a	group 3b	group4	group 5a	group 5b	group 6	group 7	group 8a	group 8b
Habitat	outgroup	Macoma balthica	Abra alba	Abra alba	Nephtys cirrosa	Macoma balthica	Abra alba	Nephtys cirrosa-Abra alba	Macoma balthica	Macoma balthica-Abra alba	Macoma balthica
Density (ind/m²)	102 ±8	76±9	242+10	2056±40	328±18	615±21	307±16	148±14	6710±59	164+13	71±7
# species (/sample)	3	3	5	18	8	6	6	6	12	6	4
SIMPER species	Microphthalmus	Spiophanes bombyx	Nephtys hombergii	Nephtys	Nephtys cirrosa	Cirratulidae	Abra alba	Scoloplos armiger	Petricola pholadiformis	Capitella	Oligochaeta
	Pectinaria koreni	Nephtys	Spisula subtruncata	Spiophanes bombyx	Nephtys	Nephtys	Nephtys	Microphthalmus	Polydora	Spio	Macoma balthica
	Gastrosaccus spinifer	Diastylis rathkei	Bathyporeia	Abra alba	Spio	Oligochaeta	Macoma balthica	Oligochaeta	Corophium	Oligochaeta	
	Nephtys	Barnea candida	Nephtys cirrosa	Scoloplos armiger	Magelona johnstoni	Macoma balthica	Spisula subtruncata	Spiophanes bombyx	Alitta succinea	Magelona johnstoni	
	Bathyporeia elegans	Polydora	Magelona johnstoni	Nephtys hombergii	Scoloplos armiger	Nephtys hombergii	Spio	Nephtys cirrosa	Capitella	Scoloplos armiger	
Median grain size (µm)	243±18	86±10	155±18	187±24	223±25	123±12	154±19	269±31	83±11	183±18	241±23
Mud content (<63µm)	20±4	58±10	22±4	11±3	3±1	45±8	23±4	8±2	56±11	19±4	12±3
Samples	120(2)	140(9)	140(1)	**115(20)**	115(2)	115(1)	120(5)	140(3)	140(7)	150(1)	150(11)
	140(2)	LS1(3)	250(1)	**120(145)**	140(1)	140(6)	140(26)	150(3)	B031(3)	LOO(2)	315(1)
	150(4)	ZEB(2)	B041(2)	**140(21)**	150(16)	150(8)	780(1)	B08(12)	B08(3)	LS1(10)	B08(1)
	250(1)	**ZVL(7)**	LS1(2)	**230(5)**	230(2)	702(2)	B041(6)	LNP(2)	**ZVL(51)**	LS2(2)	LOO(1)
	B031(2)		LS2(9)	**250(11)**	250(5)	B041(1)	B08(1)	LS1(17)	LNP(7)	LZO(2)	LS1(5)
	B032(1)		**ZVL(1)**	**780(122)**	**315(39)**	B08(3)	LS1(2)	ZVL(2)		ZVL(1)	LZO(5)
	B08(1)			780C(3)	702(3)	LNP(2)	LS2(2)			LNP(1)	ZVL(2)
	LS1(3)			B041(13)	710(3)	**LOO(87)**	ZVL(1)				LNP(12)
	LS2(7)			**B08(36)**	780(3)	LS1(11)					
	LZO(9)			**LNP(90)**	**B031(44)**	LS2(4)					
	ZEB(2)			LOO(9)	**B032(26)**	**LZO(72)**					
	ZVL(8)			**LS1(30)**	**B041(45)**	ZEB(1)					
				LS2(89)	**B042(27)**	**ZVL(69)**					
				LZO(1)	B08(9)						
				ZVL(2)	LNP(1)						
					LOO(4)						
					LS1(34)						
					LS2(114)						
					LZO(2)						
					ZEB(4)						

3.2.3 Results

Habitat characterisation

The dumping sites and its reference stations could be linked to a certain habitat type based on the number of samples of each disposal site associated with a certain cluster/ habitat type (Table 3.2). Consequently, disposal site Nieuwpoort was situated in an *Abra alba* habitat. 96.84% of the samples from Nieuwpoort-stations were found within clusters with this habitat type (group 3b). The *Macoma balthica*-cluster groups (group 2, 5a, 8b) comprised 87.38% of the Br&W Oostende samples and 86.81% of the Br&W Zeebrugge Oost samples. Therefore the dumping sites Br&W Zeebrugge Oost and Br&W Oostende were situated in the *Macoma balthica* habitat. Disposal site Br&W S2 was situated in a *Nephtys cirrosa* habitat because this cluster existed out of 50.22% of the samples from this site. At first, this assumption may not seem correct, because 44% of the other Br&W S2-samples were found within and *Abra alba* cluster. A closer look revealed that 98% of these samples were taken before 2004. This may indicate a shift in community structure and will be explained lower. Regarding dumping site Br&W S1, the samples were not uniformly associated with one habitat type. Of the 161 samples collected at this site, 25.47% was found within an *Abra alba* habitat, 44% in a *Nephtys cirrosa* habitat, 14.29% in a *Macoma balthica* habitat and 10.56% and 3.10% in respectively the *Abra alba-Nephtys cirrosa* and *Macoma balthica-Abra alba* transition habitats. This habitat heterogeneity can be caused by the dumping activity (see further), but in this area a natural gully-bank system gradient (*Abra alba* habitat in gully, switching towards *Nephtys cirrosa* habitat at the bank) is expected to be found.

Reference stations ascribed to *Abra alba* habitats are the following: 115 (86.96%), 120 (98.68%), 230 (71.43%), 250 (66.67%), 780 (97.62%) and B08 (60.61%). In the *Macoma balthica*-clusters, the reference station ZVL (89.58%) was the only one with a significant amount of samples belonging to this habitat type. Finally, the reference stations found for the *Nephtys cirrosa* habitat are the following; 315(97.50%), B031 (89.80%), B032 (96.30%), B041 (67.16%) and B042 (100%). These analyses were done for all samples collected in the coastal area for dredged material disposal research and correspond with previous findings (Van Hoey *et al.*, 2009).

Impact at the disposal sites

Only, the monitoring data from 2004 onwards were appropriate for an impact evaluation at the five dumping sites, due to the start of sampling with a control/impact design. Impact evaluation is done by using the impact and control samples belonging to the same habitat type for each dumping site. Only for Br&W S1, the control samples of the *Abra alba* and the *Nephtys cirrosa* habitat were used.

Dumping site Nieuwpoort is subjected to a low dumping intensity regime. This is reflected in the benthic characteristics, where no significant differences were seen between the impact and nearby-reference stations over the investigated period with regard to the parameters density, species number, species composition and diversity.

Dumping site Br&W Oostende is characterised by a relative low dumping intensity over the years 2004-2005, with a maximum of 864,863 TDM in 2008. There was no significant difference between the impact and reference samples regarding the benthic characteristics (density, number of species and diversity), except for species composition in the period 2004-2005.

The dumped quantities were relatively high over the years 2004-2008 at the dumping site Br&W Zeebrugge Oost, with maximum of 4,667,225 TDM disposed sediments in 2008. In a few seasons and years (autumn 2006, winter 2007), a significant difference could be found between the impact and reference samples for the parameter density. The number of species and diversity is lower in the disposal site than in the reference locations, but not significantly.

At dumping site Br&W S2, the dumping intensities were high in 2004 (1,826,033 TDM) and in 2005 (1,234,640 TDM), but had a tremendous decrease in the years 2006-2008. No significant differences were seen between the impact and control samples regarding their species composition, density, species number or diversity. Those parameters mostly show a slightly lower value for the impact samples compared to the reference ones.

Dumping intensities were always very high (>1,500,000 TDM) at dumping site Br&W S1 and took place non-stop, throughout the year. Regarding the variables density, species composition and number of species significant lower values were observed for the disposal stations in comparison with the reference stations. A significant difference in diversity between impact and control was not shown. At this site, the bathymetry and morphology strongly changed over the last 10-years, with an extension of the bank towards the gully, leading to a loss of *Abra alba* habitat (characteristic for the gully - flank area).

Indicator assessment

The impact evaluation can be summarized by the use of a benthic indicator (BEQI, level 3). Therefore, we calculate an EQR score, which scales the difference in benthic parameters between the impact and control samples between 0 (bad status) and 1 (high status). The EQR score is calculated separately for each season and year with enough data during the period 2004-2008. The difference between impact and control is calculated within each season and year to avoid temporal trends and to minimize cumulative effects. The indicator shows a clear relation between the yearly dumping intensity (maintenance and capital dredging) and its impact on the health of the benthic habitat (Figure 3.2). We can conclude that yearly dumping intensities, exceeding values of about 2,000,000 TDM, start to affect the benthic characteristics (corresponding with moderate to poor BEQI-values). The effect was smaller for the parameter number of species, than for density.

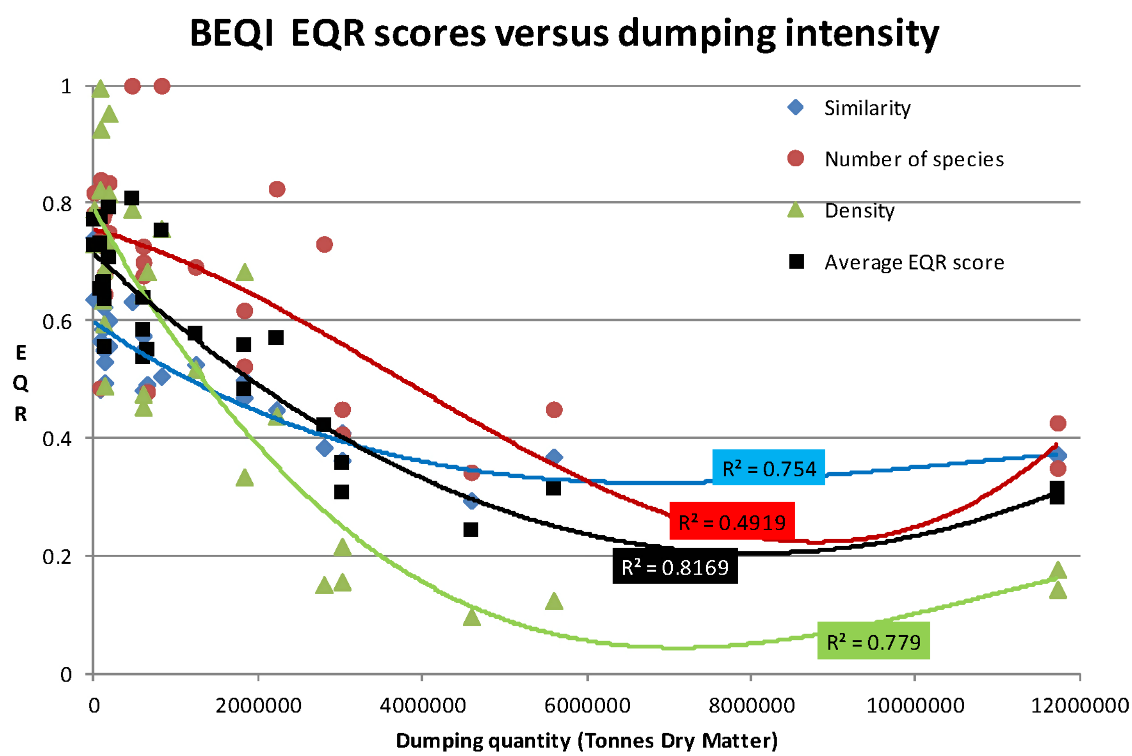

Figure 3.2: Relation between the dumping intensity and the BEQI scores for the parameters similarity (species composition), number of species and density. The BEQI overall score is the average of those three parameters. The trend line is a Polynomial order 3, with indication of the R^2 score.

Long – term LS2-01 (Dumping site Br&W S2)

Station LS2-01 in dumping site Br&W S2 is sampled already over a period of 30 years, which makes a long term analysis suitable. This station is situated at the border of the dumping site and could be a proxy for the changes in the benthic habitat characteristics in this part of the BPNS. The multivariate analysis (Figure 3.3d) indicates a transition in species assemblage over the years. We can roughly distinguish 3 temporal assemblages (period '85-'90; '93-'00; '01-'08). The average density (Figure 3.3a) and species richness (Figure 3.3b) were significantly lower in the latest period compared to the previous ones. The changes in species composition can be attributed to the absence of mud loving species in the recent period, as *Abra alba, Nephtys hombergii, Spisula subtruncata, Capitella* and *Tellina fabula*. The dominant species in the recent period were *Nephtys cirrosa, Ensis directus*, *Scoloplos armiger* and *Bathyporeia guilliamsoniana*, which were species characteristic for more sandy sediments. When comparing the sediment over the different periods, we did not find any significant changes but nevertheless a decreasing trend in mud content is visible from 4 to 2% (Figure 3.3c). This little change in mud content leads to a transition from the more diverse *Abra alba* habitat to the less diverse *Nepthys cirrosa* habitat (Van Hoey *et al.*, 2004). Simultaneously, dumping intensity decreased under the 1 million tons (Figure 3.3e), which is an indication that the seafloor at the dumping site becomes more sandy. This prevents recruitment of 'mud loving' species, causing a decrease in the benthic diversity coinciding with the decrease in dumping.

3.2.4 Discussion

We can conclude that dumping activities have resulted in benthic habitat changes at dumping site Br&W S1. Whereas at the other dumping sites, the benthic community can scope the existing dumping regime. At Br&W S1, mainly the rich *Abra alba* community is affected, normally dominating the gully stations in the disposal area. At Br&W S2, it is possible that dumping has had a small positive effect, by supply of mud and organic matter to the more sandy environment (indicated by the switch between *Abra alba and Nephtys cirrosa* habitat). At dumping site LNP, with lowest dumping intensity, there was no impact. And finally, the dumping activity at site Br&W Zeebrugge Oost and Br&W Oostende had a smaller effect on the benthos, due to the more natural poor benthic environment. Nevertheless, it needs to be kept in mind that this is a control/impact study, without knowledge about the natural situation, before the dumping activities started. Therefore, the characterisation of the dumping sites might not be fully realistic as the dumping events could already have changed the original habitat and nearby environment. The relation between the benthos indicator BEQI and the dumping intensity shows that a benthic community seems to be affected from an average dumping regime of 2,000,000 tons dry matter a year. Based on this analyses, roughly 2,000,000 tons, regardless the surface area where it is dumped on, is proposed as critical boundary. It has to be mentioned that this critical boundary can differ for each habitat type. In other words, it is possible that a sandy or fine muddy sand habitat is more sensitive to it than a muddy habitat (*Macoma balthica* habitat), as some results show, but more detailed investigation is needed. This critical boundary needs to be seen as a boundary where the natural benthic recolonisation and recruitment cannot compensate for the loss of benthos due to smothering by the dumping of dredged material.

Literature provides a lot of explanations for the impact of dredged-disposal on the benthic habitats (Morton, 1977; Elliot *et al.*, 2002; Ware *et al.*, 2010). In the particular case of the five Belgian disposal sites, we exclude large effects of chemicals, since there are specific regulations, which prohibit dumping of heavily polluted sediments. This is confirmed by the Lauwaert *et al.* (2008; 2009.) and Van Hoey *et al.* (2009), which did not mention differences in contamination between disposal- and control stations. Differences in organic matter have come up already as possible explanations (cf. Br&W S2 and Br&W Zeebrugge Oost), but knowledge about the Belgian coastal zone learns us, that the area is characterised by relatively high levels of organic matter, especially in the vicinity of the mouth of the Westerschelde estuary. Therefore a positive contribution of the disposed sediments in increasing the organic matter is less likely, although we could not exclude a short-term effect. We suggest that if an impact is detected, this is mainly related to the physical burial of the organisms (smothering, incorporation), or the properties of the dredged-disposal (mud in more sandy areas). When benthic organisms are buried under a certain amount of sediments, their feeding capacity will get altered or their mobility will decrease (Morton, 1977). Some species will be more resistant to direct burial than others, depending on their capacity to migrate upwards through the sediments. The sediment properties of the dredged material will mainly be responsible for this effect.

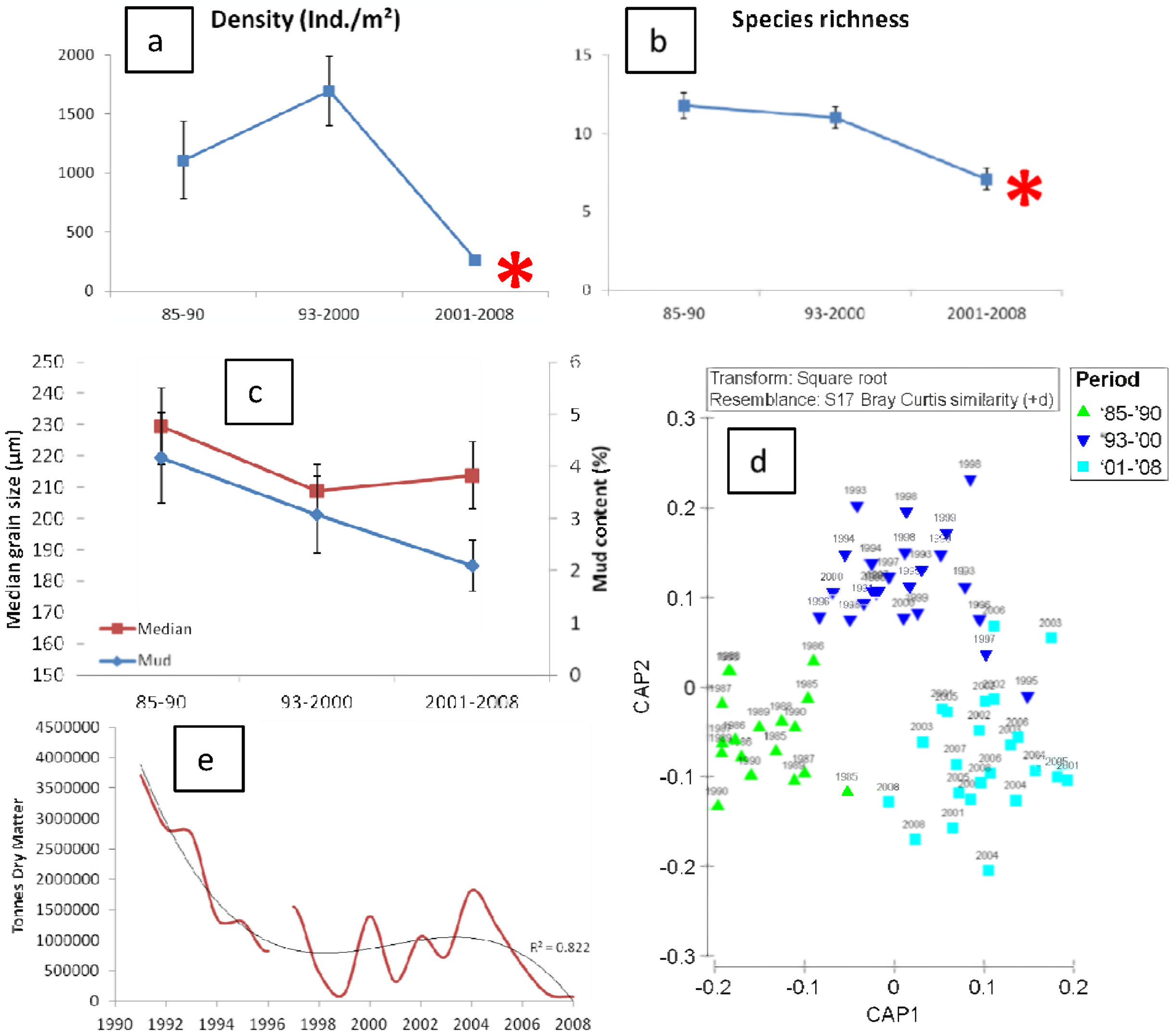

Figure 3.3: (a) the average density with standard error at LS2-01 in the three periods; (b) the average species richness; (c) the average median grain size and mud content; (d) CAP analysis of the species composition with delineation of the three periods; (e) the dumping intensity at dumping site LS2 over the period 1990-2008 in tons dry matter. * significant difference.

3.3 Biological and chemical status analyse of the disposal of dredged material: 2009-2010

3.3.1 Introduction

In this section, the results of the routine biological and chemical monitoring at the five dumping sites on the BPNS were described for the years 2009-2010. Firstly, we analyzed the sedimentology at the five sites, with indication of the sedimentological characteristics (mud content, median grain size) and the pollution (PCBs, PAHs, heavy metals…) in the sediments. Secondly, we described and evaluated the difference or correspondence in biological characteristics of the ecosystem components macrobenthos, epibenthos and demersal fish between the dumping sites and the nearby control area. Thirdly, the (bio)accumulation of contaminants in the marine ecosystem is investigated by chemical analysis on different biota species. Finally, the biological effects of pollutants on marine organisms are investigated by the assessment of the prevalence of fish diseases and by the measurement of enzymatic EROD activities in the liver of juvenile dab. The observed patterns at the disposal sites were evaluated with indicators and threshold values as defined in Belgian or European legislation (e.g. Marine Strategy Framework Directive).

3.3.2 Material and Methods

All sampling locations were distributed in the shallow coastal zone of the BPNS (limited to approximately 20 km offshore), since all dumping sites are situated in this zone (Figure 3.4). The impact samples are situated in the five dumping sites, while control samples are situated just outside the dumping sites and on some overall monitoring locations in the close surroundings (Figure 3.4).

Biological sampling and data analysis

Macrobenthos can be defined as the organisms that live for most part of their life in the sediment, and that are retained on a 1 mm-meshed sieve. The macrobenthos was sampled with a Van Veen grab (0.1 m²), sieved alive and fixed with an 8 % formaldehyde seawater solution. Species were identified to species level when possible and counted. The macrobenthos sampling and analysis protocol is based on the ISO 16665:2005 standard ("Water quality – Guidelines for quantitative sampling and sample processing of marine soft-bottom macrofauna"). From each Van Veen sample, a Perspex core was taken for sediment analyses, which were dried at 60°C and analyzed with a Malvern Mastersizer 2000 following a standardized protocol. Depth and position of each sample were also registered during the campaigns.

Demersal fish fauna and epibenthos can be defined as organisms living on or in close association with the seafloor, and that are caught representatively and efficiently with a beam trawl. Both ecosystem components were sampled with an 8-meter beam trawl, equipped with a fine-meshed shrimp net (stretched mesh width 22mm in the codend) and a bolder-chain but no tickler chains (to minimize the environmental damage) (Figure 3.3). The net was dragged during 30 minutes/3500m (2009) and 15 minutes/1500m (2010) at an average speed of 4 knots over the bottom. Data were recorded on time, start and stop coordinates trajectory and sampling depth in order to enable a correct conversion towards sampled surface units. The fish tracks were positioned following depth contours that run parallel to the coastline, thereby minimizing the depth variation within a single track.

The datasets for all ecosystem components were reduced, and a quality control was performed. The difference in biological characteristics were analysed following the impact/control design. Furthermore, univariate parameters (density, species richness, biomass [wet weight] and diversity) were calculated with the diverse module in PRIMER 6, and statistically significant differences were determined using Permutational MANOVA (an extension of Primer v6), Kruskal Walis and the Mann-Whitney U-test (Statistica 9). Multivariate analyses (cluster, MDS, SIMPER) were used to analyse the community structure for each ecosystem component (PRIMER 6). The benthic indicator BEQI (Benthic Ecosystem Quality Index; www.beqi.eu) was used to assess the macrobenthic status at the five disposal sites.

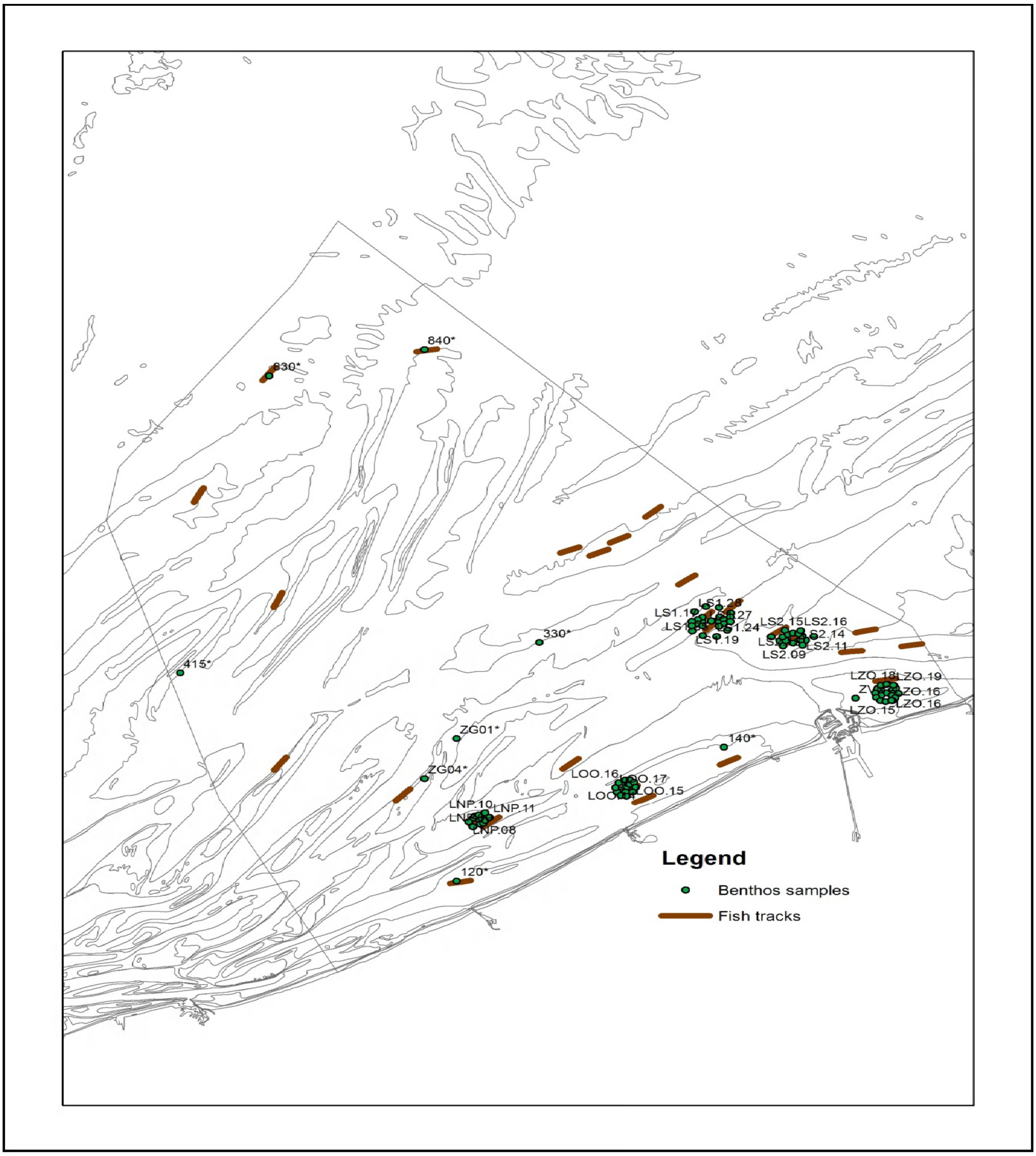

Figure 3.4: Map with sampling done in 2009-2010 for the dredge disposal monitoring program.

Chemical sampling and data analysis

Adsorption of chemical contaminants and nutrients is determined by the presence of small silty particles rather than course sand grains, making silt content an important factor for both abiotic and biotic components. Sediment was sampled using a Van Veen grab (0.1 m²) and sieved to obtain the fine fraction (<63 µm) for chemical extraction and analysis. Analysis of sediment assessed the levels of PCBs, OCPs and heavy metals on the different impact and control sites of the dumping areas. Grain size distribution, organic carbon content and carbonate content were used for normalization of the results. Results for sediment are always expressed on a dry weight (d.w.) basis.

Marine biota species, such as starfish, sea anemone, bivalves, crustaceans and fish species, were caught with a beam trawl as mentioned above for biological sampling. Assessment of marine biota species included the analysis of PCBs, OCPs, PAHs and heavy metals. All results for biota are expressed on a wet weight (w.w.) basis.

The assessment of biological effects of contaminants includes the investigation of the prevalence of fish diseases and the measurement of the enzymatic EROD activity in the liver of dab. For the assessment of fish diseases, a large number of infectious and parasitical anomalies of the epidermis, the gills, the mouth and the intestines were investigated on two sentinel species dab and whiting. ICES guidelines are used as assessment criteria (Bucke *et al.*, 1996). Using long-term prevalence data on fish diseases, changes in prevalence may act as an alert for the environmental health status.

The ethoxy-O-deethylase (EROD) activity is up-regulated by the presence of contaminants and reflects the xenobiotic removal activity of the organism. The EROD activity is measured using a fluorescence spectrophotometer to quantify kinetics of the conversion of ethoxy-resorufin (non fluorescent) to resorufin (fluorescent). The EROD assay is based on the method of Burke and Mayer (1974). The protein level of the sample was determined by the colorimetric detection of bicinchoninic acid and was used as a normalization assay (Smith *et al*. 1985). ICES guidelines were used for the assessment of the EROD activity in juvenile dab liver (Stagg and McIntosh, 1998). The protocol was adapted and participation in an inter-laboratory proficiency test organized by BEQUALM (Biological Effects Quality Assurance in Monitoring Programs) proved good analysis. EROD activities are reported in (pmol)/(min.mg protein).

3.3.3 Biological status

Macrobenthos

The ecological status of the macrobenthos at the different disposal sites is evaluated with the benthic indicator BEQI, which weighed the difference in the parameters density, biomass, species composition (Bray-Curtis similarity) and number of species between the set of impact and control samples (Table 3.3).

Based on the BEQI evaluation, only dumping site Br&W S1, gives a score ‘poor’ for the status of the macrobenthos. This can mainly be attributed to the very low scores for density and biomass, which were much higher in the control samples compared to the impact samples. This difference was caused by the dominance of the tube building polychaete *Owenia fusiformis* in the control samples, with densities up to 46000 ind/m². At this dumping site, the diversity (number of species) was also lower in the impacted area compared to the surroundings.

On average, the status of the macrobenthos was good at the dumping site Nieuwpoort. The lower score for density and similarity can be attributed to the dominance of *Owenia fusiformis* in the impacted area.

At dumping site Br&W Oostende and Br&W Zeebrugge Oost, the benthic parameters deviate not between the impacted area and the surroundings. The evaluation of the parameters density and biomass is less confident because the high variability between the individual samples. Both areas were characterised by a higher heterogeneity in their sedimentological and biological characteristics.

At dumping site Br&W S2, the macrobenthos status was good, which indicates that there is no significant difference observed between the impact samples and the control samples. Nevertheless, some remarkable differences were observed between the samples. The benthic characteristics of the samples indicate an enrichment of the Northern samples with mud 'loving' species, whereas the samples in the western part are more impoverished (lower diversity). At this side, the invasive alien species *Ensis directus* was rather dominant. If this could be attributed to the higher dumping intensities over the period 2009-2010 than in previous years is not unambiguous. From the dumping intensity maps could be derived that the dumping in 2009 was concentrated in the Northern part, whereas in 2010 it was in the western part.

The patterns observed at the five dumping sites in autumn 2010 confirm those of previous years. At the dumping sites Br&W Oostende and Br&W Zeebrugge Oost, naturally characterised by an instable, sandy mud and a poor benthic community, the medium to high dumping intensity does not affect the benthic life. The variability in the benthic characteristics at dumping site Nieuwpoort is more likely natural, and is not related to the low dumping intensity. The changes in the benthic characteristics at site Br&W S1 are still the result of the loss of *Abra alba* habitat within the dumping area. In 2010, we measured the highest densities ever of the tube building polychaete *Owenia fusiformis* in the surrounding area. This side is subjected to the highest dumping intensities over the period 2009-2010, especially in 2009 and is responsible for those ecological changes. Despite, the good status at dumping site Br&W S2, some remarkable differences in benthic characteristics were observed between the western and northern part of the side, which can have a link with the dumping activity over the period 2009-2010.

Table 3.3: Evaluation of the difference between control and impact samples in autumn 2010 at the five dumping sites for the ecosystem component macrobenthos. The values represent the BEQI-index, the colours the status (blue: high; green: good; yellow: moderate; orange: poor; red: bad).

BEQI	Nieuwpoort	Br&W Oostende	Br&W S1	Br&W S2	Br&W Zeebrugge Oost
Density	**0.37**	0.81	**0.06**	**0.75**	0.77
Biomass	**0.97**	0.88	**0.08**	**0.68**	0.76
Similarity	**0.58**	**0.66**	**0.38**	**0.56**	**0.49**
Species	**1**	**0.73**	**0.45**	**1**	**0.74**
Average	**0.73**	0.77	**0.24**	**0.75**	0.68

Epibenthos and demersal fish

The ecological status of the epibenthos and demersal fish was also evaluated for the five dumping sites. To know whether dumping activities had an impact on density, biomass, estimated number of species (ES) and diversity index N1, a Permutational MANOVA-analysis was conducted.

At dumping site Nieuwpoort, several significant differences between control and impact samples could be detected. Concerning the epibenthos, the parameters density and biomass displayed significantly higher values in the control samples than in the impact samples of autumn 2009. This could be attributed to a much higher density (and biomass) of the starfish *Asterias rubens* in the control samples. The estimated number of species (ES) was significantly higher in the impact samples of winter 2009. This pattern was also visible in the other years and seasons but could not be statistically established. For the demersal fish, there was a significant difference between control and impact samples of autumn 2009, both for the density and the diversity index N1. The density of the control samples was higher than the impact samples, which was mainly due to the fact that gobies (*Pomatoschistus sp.*), dragonet (*Callionymus lyra*) and whiting (*Merlangius merlangus*) were more abundant in the control samples. The diversity index N1 showed higher values in the impact samples. In the dumping area of Br&W Oostende, a significant epibenthic density difference between control and impact samples was observed in the winter of 2009. Particularly the brittle star *Ophiura ophiura* was responsible for this pattern, because of its higher abundance in the control samples. The demersal fish community showed no conspicuous changes as a result of the dumping activities.

Table 3.4: Evaluation of the differences between control and impact samples in the period 2009-2010, for the five dumping sites and for the ecosystem components epibenthos and demersal fish. Red = significant difference; green = no significant difference.

	Nieuwpoort				Br&W Oostende				Br&W S1				Br&W S2				Br&W Zeebrugge Oost			
	2009		2010		2009		2010		2009		2010		2009		2010		2009		2010	
	W	A	W	A	W	A	W	A	W	A	W	A	W	A	W	A	W	A	W	A
EPIBENTHOS																				
Density	green	red	green	green	red	green	green		green	green	green	green	green	green	green	green	green	green	green	green
Biomass	green	red	green	green	green	green	green		green	green	green	green	green	green	green	green	red	green	green	green
ES	red	green	green	green	green	green	green		green	green	green	green	green	green	green	green	green	green	green	green
N1	green	green	green	green	green	green	green		green	green	green	green	green	green	green	green	green	green	green	green
DEMERSAL FISH																				
Density	green	red	green	green	green	green	green		green	green	green	green	green	green	green	green	green	green	green	red
ES	green	green	green	green	green	green	green		green	green	green	green	green	green	green	green	green	green	green	green
N1	green	red	green	green	green	green	green		red	green	green	green	green	green	green	green	red	green	green	green

Dumping sites Br&W S1 and Br&W S2 revealed no appreciable effects of dumping on epibenthos and demersal fish. The values of density, biomass and species richness were in most cases lower (but not significant) in the impact area than in the control area. Species like starfish *A. rubens* and brittle star *O. ophiura* are present in lower densities in the impact area. The heart urchin *Echinocardium cordatum* and the blue-leg swimming-crab *Liocarcinus depurator* were even absent in the impact area. For demersal fish however, the diversity index N1 was significantly higher in the Br&W S1 impact samples in winter 2009.

For dumping site Br&W Zeebrugge Oost, the epibenthic biomass data showed a significant difference between the control and impact samples in winter 2009. Again, this difference could be explained by a higher abundance of the brittle star *O. ophiura* in the control samples. The demersal density values differed significantly in autumn 2010, mainly due to the higher occurrence of gobies (*Pomatoschistus sp.)* in the control samples. The diversity index N1 was significantly higher in the impact zone in winter 2009.

There is no clearly visible effect of dumping on the epibenthic and demersal fish fauna. This can mainly be attributed to the fact that most species have a high mobility and are able to avoid the dumping stress. The significant p-values (Table 3.4) can be explained by the temporary dominance of certain species (starfish, brittle star, goby) and/or by the natural variability of the habitat (e.g. dumping site Nieuwpoort). Based on this analysis, a small effect of dumping on the higher trophic level of epibenthos and demersal fish cannot be excluded, because the number of samples per sampling event is too low to have sufficient statistical power. A way to reach a higher power is by combining data of several sampling events, which is not possible over the period 2009-2010 due to a switch in sampling strategy. Since dumping had no or a minor effect on the epibenthos and demersal fish community, it is advisable to investigate the effect of dumping on certain functional or sensitive species.

3.3.4 Chemical status

To assess the chemical status of the different disposal sites, sediment samples of each disposal site and reference site were screened for their chemical content. Environmental assessment criteria (EAC) according to OSPAR, MFSD Task Group 8 Contaminants and pollution effects (Law *et al*., 2010) and Belgisch Staatsblad were used to interpret the observed levels of pollution. Consequently, monitoring data could be scored according to the EAC as presented in Table 3.5. If the assessed level of a specific pollutant reaches the EAC level it is indicated in red, if the level is lower, it is indicated as green. The EAC values for the heavy metals cadmium and lead proposed by the MFSD Task Group 8 are clearly stricter compared to the EAC values of OSPAR and Belgisch Staatsblad. The observed levels of Cd and Pb on each site clearly exceed those EAC values of MFSD Task Group 8, while this is definitely not the case for the other EAC values. A similar observation could be made for the PCBs 118 and 153 by Belgisch Staatsblad and PCB 118 by MFSD Task Group 8. Based on the OSPAR monitoring criteria, only incidentally over range levels of Pb and PCBs are observed (Table 3.5). Nevertheless, the levels of lead and PCBs must be followed up in future. Other measured heavy metals and persistent organic pollutants do not approach the formulated EAC values. It will be necessary to complete this table with monitoring results of the BPNS during the upcoming years. A major remark concerns the availability and relevance of environmental assessment criteria. Extra attention must be paid to the development of those EAC, which will be relevant for monitoring on the BPNS.

Table 3.5: Assessment criteria for sediment analysis according to OSPAR, MSFD and Belgisch Staatsblad (Colour code GREEN = OK according to EAC, Colour code RED = value too high).

		Nieuwpoort								Br&W Oostende								Br&W S1								Br&W S2								Br&W Zeebrugge Oost							
		2009				2010				2009				2010				2009				2010				2009				2010				2009				2010			
		S		A		S		A		S		A		S		A		S		A		S		A		S		A		S		A		S		A		S		A	
		I	C	I	C	I	C	I	C	I	C	I	C	I	C	I	C	I	C	I	C	I	C	I	C	I	C	I	C	I	C	I	C	I	C	I	C	I	C
SEDIMENT																																									
OSPAR	Cd	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G
	Hg	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G
	Pb	G	G	G	G	G	G	G	G	G	R	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	R	G	G	G	G	G	G	G
	Cu	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G
MFSD Task Group 8	Cd	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R
	Hg	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G
	Pb	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R
Belgisch Staatsblad	Cd	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G
	Hg	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G
	Pb	G	R	R	G	G	G	G	G	G	R	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	R	R	G	G	G	G	G	G
	Cu	G	G	R	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G
OSPAR	∑7 CB	G	G	G	G	G	G	R	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	R	G	G	G	G	G	G	G	G	G	G	G	G	G
	pp-DDE	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G
MFSD Task Group 8	CB118	R	R	R	R	R	G	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	G	R	G	R	R	R	R
	CB153	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G
Belgisch Staatsblad	CB118	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R
	CB153	R	R	R	R	R	G	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	G	R	R	R	R
	pp-DDE	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G
	pp-DDT	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G

Based on the results of 2009-2010, only limited differences between impact and control assessment are observed. For dumping sites Br&W Zeebrugge Oost and Br&W Oostende, impact levels seem to be slightly elevated compared to control levels. On Br&W Zeebrugge Oost, levels of Cd, Hg, Cu, Pb, CBs and pesticides were slightly higher on impact sites during 2009-2010 (Figure 3.5). On Br&W Oostende, higher pesticide levels were observed on the impact sites. Dumping site Nieuwpoort wasn't intensively used during the period 2009-2010. Nevertheless, relatively high CB levels and by far the highest DDT levels are observed on this site. Those higher values might be caused by old dredged spoil or by older sediments that were replaced. On sediments of dumping sites Br&W S2 and Br&W Oostende, remarkably higher Cu concentrations were observed compared to the other dumping sites, and this was also reflected by the Cu levels in biota samples.

Generally, no significant trends between impact and control assessment are observed on sediment samples of the different dumping areas. With exception of dumping site Br&W Zeebrugge Oost, on which the pollution levels of impact sites are elevated compared to control sites. Influences by season are small on sediment measurements, still it is important to take into account the sampling period when reporting monitoring data. Given that the seasonal variations are small, chemical monitoring of sediment samples will be executed in future during autumn on a yearly basis.

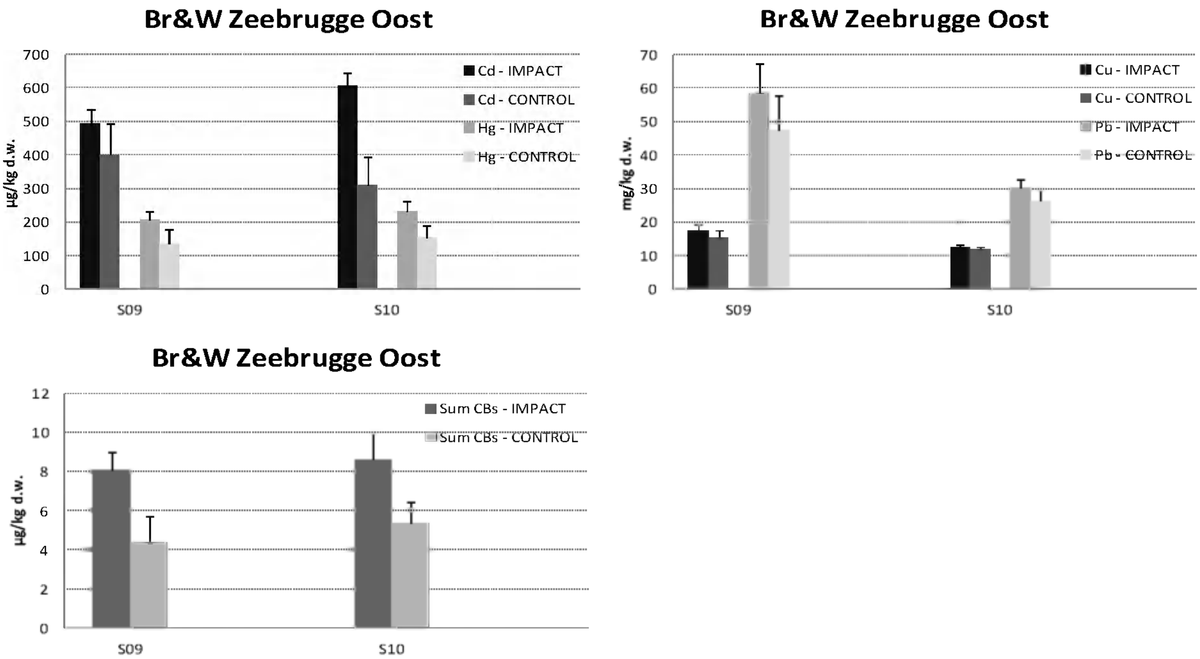

Figure 3.5: Levels of Cd, Hg, Cu, Pb and CBs in sediment samples of Br&W Zeebrugge Oost during spring 2009 and spring 2010 for Impact and Control sites. Concentrations are expressed in µg/kg w.w. for Cd, Hg and CBs, in mg/kg w.w. for Cu and Pb.

In agreement with the assessment strategy on sediments, it would be of main importance to compare the results on biota with environmental assessment criteria from OSPAR, MFSD Task Group 8 or Belgisch Staatsblad. As an illustration, Table 3.6 presents the environmental assessment criteria with colour code for blue mussel on dumping site Nieuwpoort during autumn 2010. Since EAC values are uncommon for most marine biota species, it is not possible to compare the monitoring results with environmental assessment criteria. In the future, more OSPAR EAC values might be available on e.g. brown shrimp, an abundant and important marine organism. EAC values proposed by Belgisch Staatsblad do not discriminate between the different marine organisms and are therefore difficult to interpret and use as assessment guideline.

Generally, during the period 2009-2010 no significant trend between control and impact zone assessment was recorded. Elevated PAH levels in diverse marine species were observed in the impact areas of dumping sites Br&W Zeebrugge Oost, Br&W Oostende and Br&W S1 compared to the control areas. Figure 3.6 reports the levels of different pollutants (CBs, PAHs and Cu) in starfish on the different dumping sites during autumn 2010. In agreement with the results on sediment, a higher level of Cu is observed on dumping site Br&W S2 and Br&W Oostende. On dumping site Br&W Oostende a remarkably high CB level was assessed. The measured levels of CBs, PAHs and heavy metals are the lowest on dumping sites Br&W S1 and Nieuwpoort.

Table 3.6: Assessment criteria for blue mussel analysis according to OSPAR and Belgisch Staatsblad (Colour code GREEN = OK according to EAC, Colour code RED = value too high).

			Nieuwpoort
			2010
			A
			C
Blue mussel			
OSPAR	Pyrene		
	Benzo(a)pyrene		
	Cd		
	Pb		
	Cu		
	Hg		
	pp-DDE		
	∑7 CB		
Belgisch Staatsblad	Hg		
	Hexachlorbenzen		

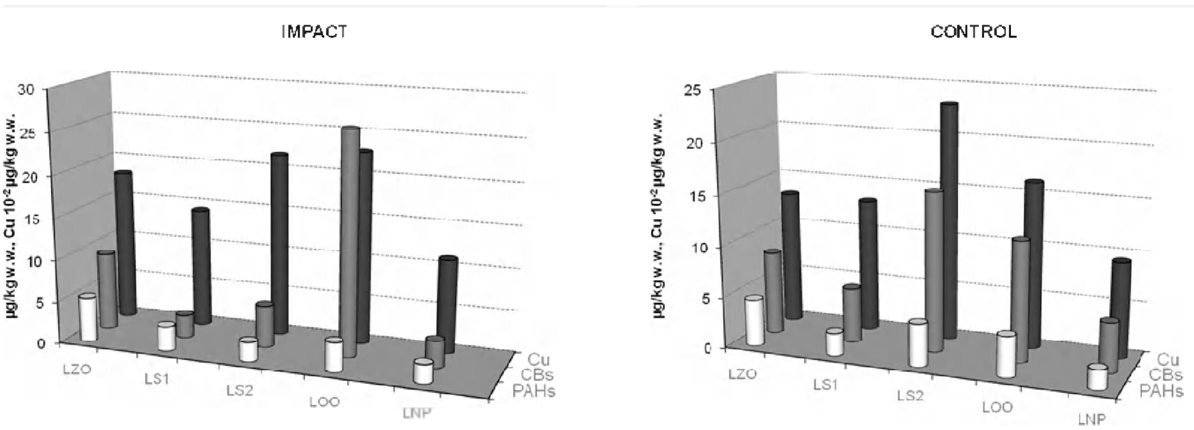

Figure 3.6: Levels of Cu, CBs and PAHs in starfish during autumn 2010 for the Impact and Control sites of the different dumping sites (Br&W Zeebrugge Oost (LZO), Br&W S1 (LS1), Br&W S2 (LS2), Br&W Oostende (LOO) and Nieuwpoort (LNP)). Concentrations are expressed in µg/kg w.w. for the sum of PAHs and the sum of CBs, while concentrations for Cu are expressed in 10^{-2} µg/kg w.w.

In general, seasonal variations are rather small. Nevertheless it is important to take into account the sampling period before evaluating monitoring data. Interestingly, PAH concentrations in epibenthos species e.g. brown shrimp show a seasonal variability with clearly higher values during spring (Figure 3.7).

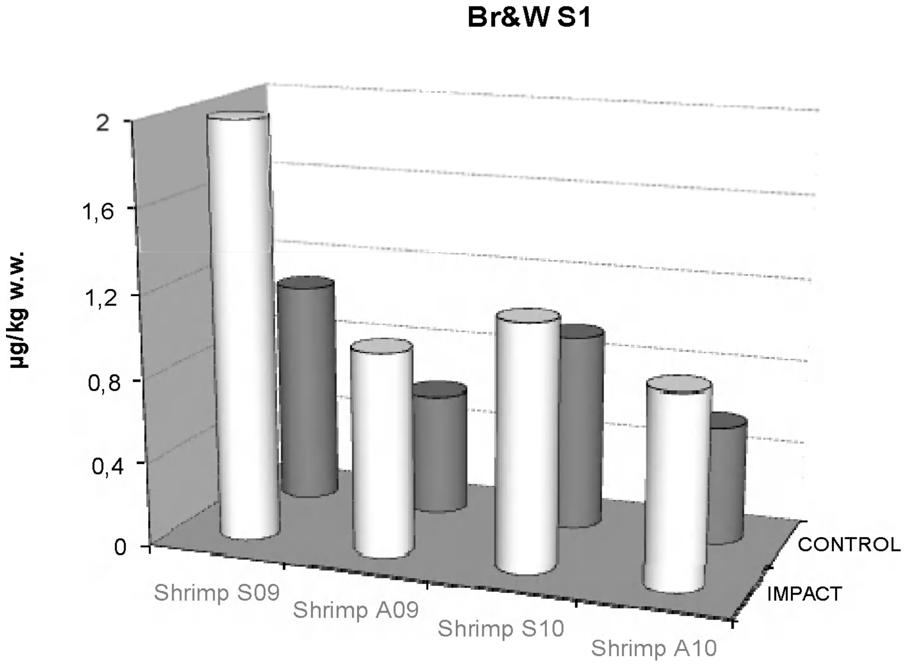

Figure 3.7: Levels of PAHs in brown shrimp during the period 2009-2010. Concentrations are expressed in µg/kg w.w. Impact and Control are presented for each season.

Biological Effects of Pollutants

The assessment of Fish Diseases

Studying the possible harmful effects of pollutants present in the marine environment on biota species is an essential aspect in environmental monitoring. In this part, the relationship between the dumping of dredge spoil with associated pollutants and the biological effect of those pollutants on local fish populations is investigated. It is of major importance to identify if pollutants affect the biological activity of marine organisms or if they are transformed/metabolized and excreted.

Nowadays, it is generally accepted that the investigation of externally visible fish diseases may provide information on the occurrence of environmental stress and consequently the environmental health status. The study of prevalence and distribution of diseases and parasites of wild marine fish became an essential part of integrated monitoring programmes to screen for the effects of environmental changes and marine pollution (Thain *et al.*, 2008; Lang *et al.,* 2009). The aim of the study is to monitor and compare the prevalence of contamination associated diseases and parasites of demersal fish on dredge spoil disposal sites. In this section the observed prevalence of contamination associated diseases and parasites during the period 2009-2010 are discussed.

Especially diseases such as ulcers, skeletal deformations, nodules and lymphocystis can provide valuable information on changes in the environmental health and may act as an early warning system. Those severe diseases are very rare on the BPNS. Moreover, in the period 2004-2010 a clear declining trend in the overall prevalence on dab was observed on the BPNS.

Most of the observed anomalies were due to parasitical infections. The presence of these parasites showed considerable variation in spatial and temporal distribution. The most abundant parasites on whiting comprise the trematode *Cryptocotyle lingua*, the externally attached copepod *Clavella sp.* and the copepod *Lernaeocera branchialis*. The most abundant parasites on dab include the externally attached copepod *Acanthochondria spp.* and the parasite of the intestines *Glugea stephani*. Parasites such as *Lernaeocera branchialis, Cryptocotyle lingua, Acanthochondria cornuta* and *Glugea stephani* tend to show a larger prevalence in coastal areas compared to the more offshore sites. As a consequence of the spatial variation, the dredge disposal sites must be compared to the coastal reference sites. Long-term prevalence data from 1996 until 2010 showed undulating prevalence patterns during the years for all examined parasitical infections. Models can be formulated to predict the prevalence of diseases in future. Those models can be used as assessment guideline or assessment criteria to interpret the obtained prevalence (Devriese *et al.*, 2011). Figure 3.8 illustrates the undulating prevalence pattern of *Clavella sp.* on whiting during the period 1996-2010 with the predicted prevalence using Solver. During the years 2008 and 2009, an exceptionally high *Clavella sp.* prevalence was observed.

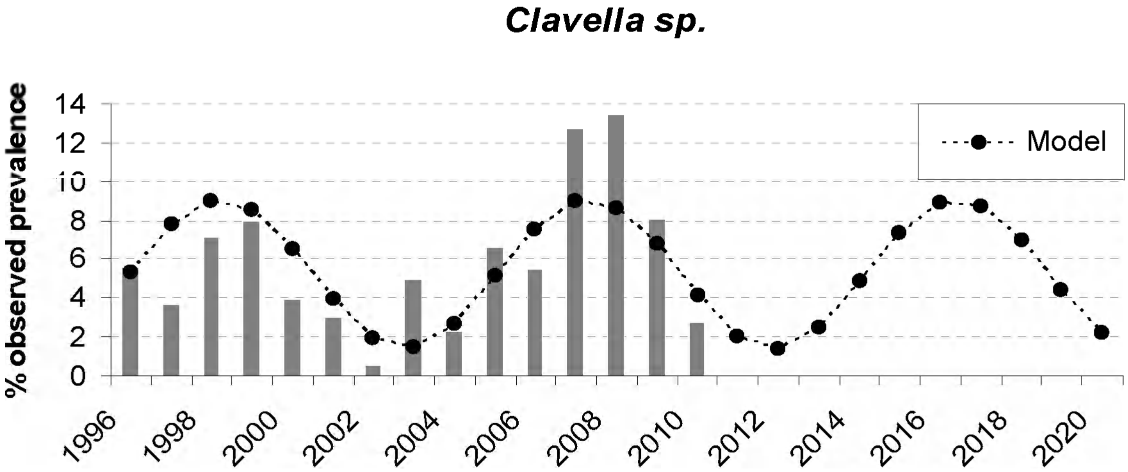

Figure 3.8: Observed prevalence of *Clavella sp.* on whiting during 1996-2010 and the predicted prevalence using Solver until 2020.

In the period 2000-2010, *Glugea stephani* and *Acanthochondria cornuta* on dab (Figure 3.9), and *Lernaeocera branchialis* on whiting showed a slightly higher prevalence on the dumping sites compared to the coastal reference sites. The observed prevalence of *Glugea stephani* and *Acanthochondria cornuta* in the period 2009-2010 was remarkably higher compared to the mean prevalence over the period 2000-2010. Especially these diseases must be followed up strictly in future.

During the following years, the obtained data should be compared with the long-term prevalence data. Significant differences in prevalence between coastal reference zones and dredge deposit areas, and between long-term prevalence data and recent data must be examined thoroughly. According to OSPAR, a fish disease index (FDI) can be formulated to score the different areas to evaluate the impact of anthropogenic induced stresses on fish. Therefore, the assessment should consider the prevalence of externally visible fish diseases and macroscopic liver neoplasm’s, but also liver histopathology in common dab. In addition to the assessment of fish diseases, it would be of main importance to monitor the general health of caught fish species, e.g. gonadosomatic index (GSI), quality index method (QIM), liver glycogen content (LGC), liver-somatic index (LSI), condition index (k), etc.

The EROD assay

The effects of contaminating substances (PCBs, PAHs,…) in living organisms can be studied through the upregulation of enzymatic activities involved in biotransformation of pollutants following their binding on the aryl hydrocarbon receptor (ArH receptor). Upregulated enzyme activities may serve as 'early warning' signals for pollution. Here, the biochemical biomarker EROD (7-ethoxyresorufin O-deethylase) activity is used as an indicator of xenobiotic substance accumulation in the flatfish dab. Assessment of the EROD activity is one of the required indicators of pollution by the Joint Assessment and Monitoring Programme (JAMP) under the coordination of OSPAR.

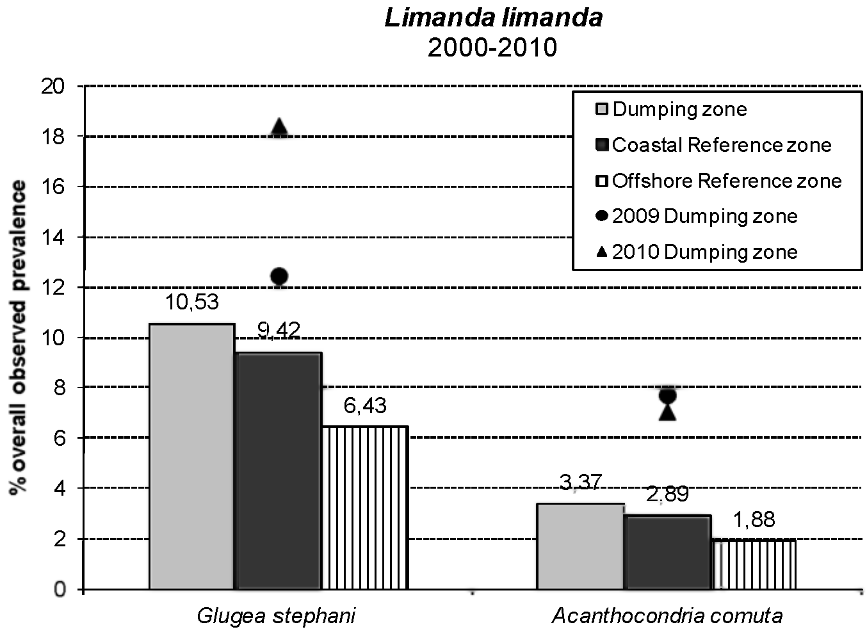

Figure 3.9: The mean observed prevalence of 2 different parasitical infections (*Acanthochondria spp.* and *Glugea spp.)* on dab over the years 2000-2010. The overall observed prevalence for the dredge disposal sites is separately given for 2009 and 2010.

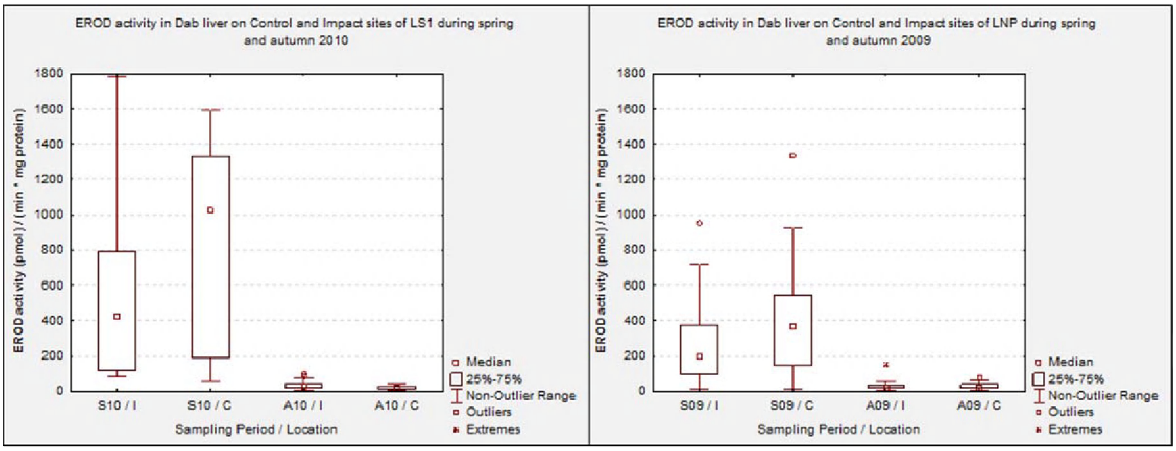

Figure 3.10: The EROD activity, expressed as pmol/(min.mg protein) measured on juvenile dab caught on disposal site Br&W S1 (LS1) and Nieuwpoort (LNP) (Impact versus Control, Spring versus Autumn).

On juvenile dab, no significant difference (α=0.05) was obtained between EROD results on male and female dab and therefore the EROD results for male and female dab were pooled into one group. The EROD results for dumping sites Nieuwpoort and Br&W S1 are presented in Figure 3.10 for the impact and control sites. On Br&W S1, spring and autumn

2010 are presented; while for dumping site Nieuwpoort spring and autumn 2009 are shown. The EROD induction is clearly visible during winter and early spring, while during summer and autumn only a background level is recorded. Only on dredge deposit site Nieuwpoort during spring 2009, a significant difference between impact and control sites was obtained. During the period 2009-2010, no significant higher EROD activity on impact sites versus control sites was observed.

Although the measurement of the enzymatic EROD activity is an important tool to assess the effects of anthropogenic activities, the sample sizes are often too small to allow statistical interpretation, especially during spring. The biochemical EROD assay depends on different parameters such as season, water temperature, size of the fish, maturity of the fish, sex of the fish,... and could be difficult to interpret. It is of main interest to keep searching for other biomarkers (genomic or biochemical) on relevant abundant species. Recently, the first efforts were made to develop a genomic biomarker on starfish.

Conclusion

During the period 2009-2010 no remarkable differences between control and impact zone assessment were recorded. Nevertheless, on sediments of dumping site Br&W Zeebrugge Oost, elevated levels of contaminants are recovered on impact sites compared to control sites. Also elevated PAHs levels were observed in different species on the impact sites of dumping site Br&W Zeebrugge Oost. In future, special attention must be paid to chemical monitoring of Br&W Zeebrugge Oost.

Severe diseases were very rare on the BPNS. Only parasites as *Lernaeocera branchialis* (whiting), *Glugea stephani* and *Acanthochondria cornuta* (dab) showed a slightly higher prevalence on the dumping sites compared to coastal reference sites.

During the period 2009-2010, no significant higher EROD activity on impact sites versus control sites was observed.

To define the chemical health status, it is of main importance to be able to compare obtained monitoring data with available and relevant environmental assessment criteria. The lack of EAC on specific marine biota species and the irrelevance of some EAC values make it so far difficult to apply those EAC values in a monitoring assessment.

Perspectives

In future, it will be a necessity to investigate the general toxicity of the environment. General toxicity test will give information about the presence and bioavailability of toxic compounds in the environment. Once a toxic effect is proven, samples might be screened for pollutants, new and emerging compounds, followed by the assessment of the effect on the different levels of biological organization.

In addition to the assessment of fish diseases, it would be of main importance to monitor the general health of fish species, e.g. gonadosomatic index (GSI), quality index method (QIM), liver glycogen content (LGC), liver-somatic index (LSI), condition index (k), etc.

Although the measurement of the enzymatic EROD activity is an important tool to assess the effects of anthropogenic activities, the small sample sizes and the difficulties for interpretation make it essential to screen for other (genomic or biochemical) biomarkers on relevant species.

3.4 An exploration of the biological life in the dredging areas

One of the aspects, that was not looked after in the previous decade was the biological life in the dredging areas, especially in the navigation channel towards the harbour of Zeebrugge and the harbour itself. This is necessary to make a comparison between the fauna in the dredged material and those occurring on the dumping sites. Species occurring in the dredging areas can be transported to the dumping areas and locally enrich the fauna if they survive (Figure 3.1). This should have a minor effect for the dumping areas on the BPNS, because the dredging areas are expected to be characterised by a poor faunal composition.

3.4.1 Material and Method

Benthic samples were taken in two different years (2007 and 2011). In the first year (2007), mainly samples in the harbour of Zeebrugge were taken, whereas in 2011, most samples were originated from the navigation channels (Pas van het Zand, Scheur, Vaargeul 1).

The samples in December 2007 were taken on board of the Brandaris in coincidence with the sediment monitoring program of the harbours and their entrances (Lauwaert *et al.*, 2008). At 13 of the 42 sampling points an extra benthic sample with a Van Veen grab (12L) was taken (Figure 3.11). In February 2011, a benthic sampling program was executed to monitor the gullies towards the harbour of Zeebrugge. Therefore, a Van Veen grab (22L) was taken at 12 sampling points, which were distributed over this area (Figure 3.4). No biota was found in the samples ZBH6 and BagG11.

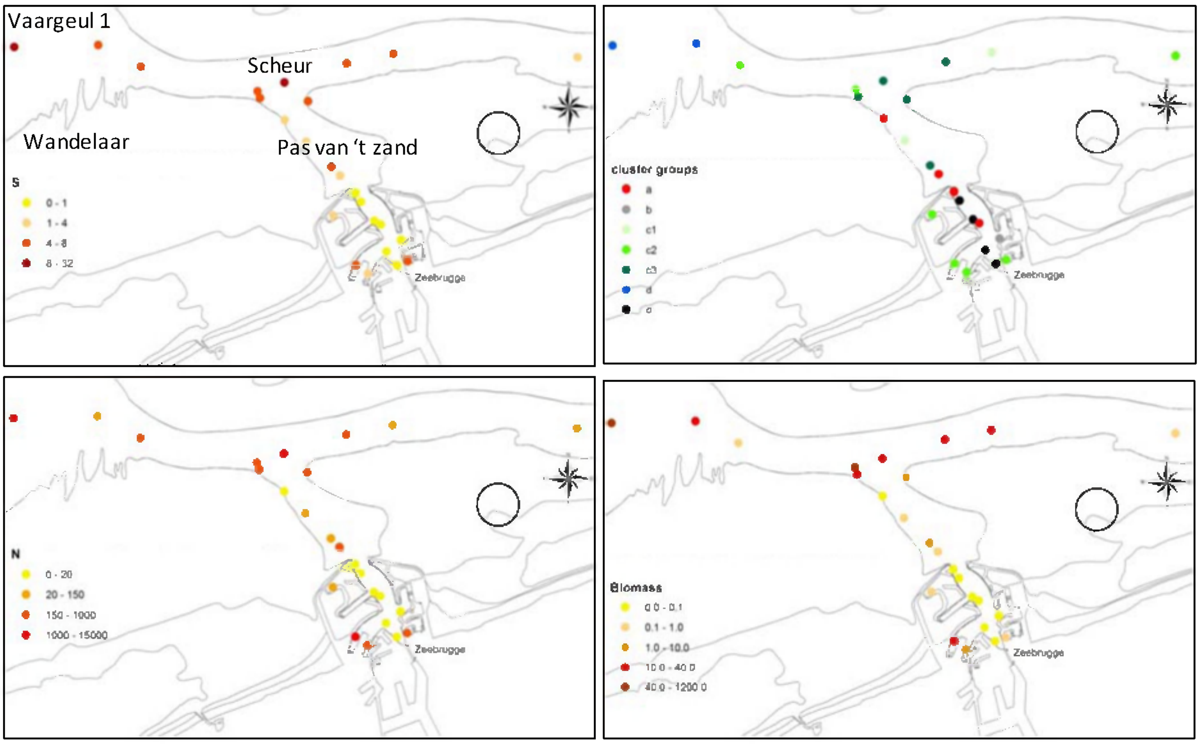

Figure 3.11: Map with indication of the number of species (S, top-left hand corner), density (N, bottom-left hand corner), biomass (bottom-right hand corner) and the cluster groups (top-right hand corner).

All the samples were sieved over a 1 mm sieve to extract the benthic fauna. The samples of 2007 were sieved after fixation, whereas those of 2011 were sieved alive. The species dataset was standardized by lumping some species (Cirratulidae spp, *Spio* spp, Anthozoa spp), and reduced by excluding species that did not belong to the macrobenthos *sensu strictu* (e.g. Mysidacea). Nematoda were excluded because of inadequate sampling techniques for quantifying meiofauna. By this standardization, two more stations were than characterised by the absence of benthic fauna (ZBH8, BagG12).

3.4.2 Results

48 benthic taxa were recorded in the dredging areas of Zeebrugge, whereof 27 (56%) taxa were only recorded once. Most taxa were found in the gullies towards Zeebrugge and only a few taxa (9; Cirratulidae spp, Oligochaeta spp, Nephtys juvenile, *Mytilus edulis, Streblospio benedicti, Abra alba, Macoma balthica, Anthzoa spp, Crangon crangon*) in the harbour itself. Not one of the observed species was a rare taxon within the benthic fauna on the BPNS. Most species were common within the *Abra alba* and/or *Macoma balthica* habitat (Van Hoey *et al.*, 2004). Only the bivalve *Mytilus edulis* is more characteristic for hard-substratum and the specimens found in our samples were mostly recruits. Those were temporally settled or washed-out from the hard substrate around the harbour.

There is a clear pattern in the benthic community characteristics in the dredging areas of Zeebrugge (Figure 3.5). In the central gully within the harbour of Zeebrugge almost no benthic individuals (max 1) were found. Two samples (ZBH16 and ZBH2) within a corner of the harbour were characterised by a higher species richness, respectively 7 and 5 spp./0.054m²), whereas in the other samples the species richness was very low. The highest number of species, densities and biomass were found in 'Vaargeul 1, especially at station ZBS41 (32 spp./0.054m²; 13981 ind./m²; 1124 g/m² Wet weight). The samples in the 'Scheur' area showed low species richness and a moderate density. The high density spot within the harbour of Zeebrugge (ZBH16) is caused by the presence of the spionid *Streblospio benedicti*. The multivariate analyses shows that most samples have a high affinity with each other and three main groups could be characterised. The samples of cluster C, characterised by the presence of Cirratulidae spp., Oligochaeta spp. and *Nephtys* juveniles, was localised in the 'Scheur' area and within the corners of the harbour. The samples of Cluster A, characterised by the presence of *Mytilus edulis* were located in the central part of the harbour and in 'Pas van het Zand. The samples of cluster D consisted of the two samples with the highest diversity and were located in 'Vaargeul 1' area.

We can conclude that the dredging areas around Zeebrugge were characterised by very poor benthic community characteristics, except the 'Vaargeul 1' area. Input of benthic animals from the dredging areas towards the dumping areas is possible, but should not lead to species enrichments in the dumping zones, due to the low densities and species richness in the dredging areas.

3.5 Optimization of the sampling strategy in the routine monitoring program

In the period 2009-2011, we invested in the optimization of the sampling strategy of the routine monitoring program at the dredge disposal sites. This was carried out to standardize the analysis according to European directives and to make the monitoring time and cost efficient. The first is necessary because the ecological and environmental status of the marine waters will be evaluated following the guidelines of the WFD and MSFD. Therefore, the monitoring programs of every Member state have to be adapted in the near future towards these guidelines. ILVO started this process of monitoring standardisation in the previous years. Secondly, monitoring programs are costly and time consuming, especially biological ones. Therefore, some sampling protocols were adapted to collect similar data within less time. In this section, the changes to the sampling protocols of the routine monitoring program of ILVO are outlined and discussed:

- The changes in the duration of the epibenthos and fish tracks from 30 minutes towards 15 minutes.
- Sieving of the macrobenthos samples alive instead of fixated.
- Introduction of quality assurance in the macrobenthos analysis (ISO 16665:2005) to achieve a BELAC accreditation certificate under ISO17025 norm.

3.5.1 Short versus long epibenthos-fish tracks

When evaluating the previous campaigns, it became clear that an adaptation of the sampling method, and more precisely the length and the number of a fish track, was needed. The cause for this specific adaptation was twofold:

1. The length of the fish tracks (3500 m) was too long to fit within the delineation of the dumping sites, whereby fauna within and aside the dumping site was collected.
2. Local effects within the dumping sites are hard to detect using long tracks, since all fauna over a length of 3500 m are pooled in a single catch and information about the small-scale ‘patchiness’ of fauna is largely lost. A shortening of the tracks and an increase in the number of tracks would result in a higher spatial resolution in the analysis, which would decrease the detection level of local changes.

However, when implementing shortened fish tracks in the monitoring program, the confidence of such tracks was first tested experimentally during the autumn campaign of 2009 (Vandendriessche *et al.*, 2010). Secondly, both track types were taken at some monitoring stations on the BPNS during the monitoring activities of 2010. The analysis of both campaigns is shown in this section.

Table 3.7: Average deviation percentages (based on density data) per species groups (epibenthos and demersal fish). Positive values represent overestimation, negative values represent underestimation.

Epibenthos	%	Demersal fish	%
Crangonidae	83	Pleuronectidae	37
Paguridae	66	Soleidae	49
Portunidae	89	Callionymidae	3
Asteriidae	148	Gobiidae	52
Ophiuridae	97		

Effect on density

Comparison of fish density and epifaunal density showed that the standardisation of short tracks to number or weights per 1000m² resulted in an underestimation of fish densities (average 7%) and epifaunal densities (average 36%) compared to the long tracks (Figure 3.12). The rate of overestimation or underestimation varied between tracks and between species groups. Large differences in the estimates of epibenthic densities were found at stations situated in the coastal area (ft120, ft230, ft215). The difference is much lower for tracks on sandy bottoms situated more offshore. The most common epibenthic species groups (Crangonidae, Paguridae, Portunidae, Asteriidae and Ophiuridae) are characterised by an underestimation of the density by taking long tracks (Table 3.7). This underestimation is the highest for sea stars (Asteridae) with a value of 148%. The most common demersal fish species groups (Pleuronectidae, Soleidae, Callionymidae, Gobiidae) are underestimated by long fish tracks. This underestimation is on average very low for Callionymidae.

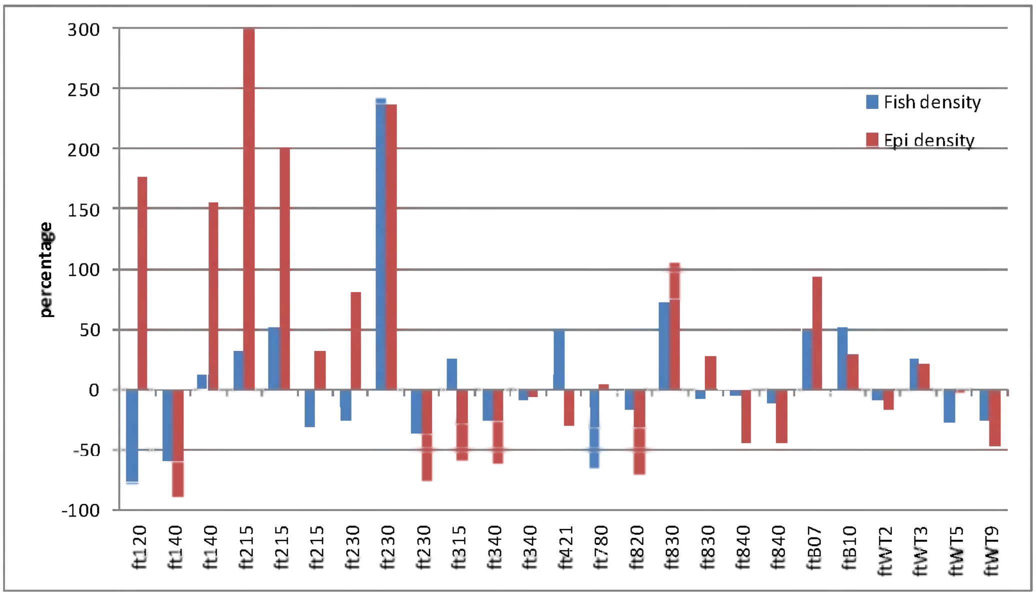

Figure 3.12: Percentage difference between short and long tracks in the density of fish and epibenthic species.

Effect on species richness

Since the number of observed species S strongly depends on sample size (Soetaert and Heip, 1990), this parameter is logically underestimated during short tracks compared to long tracks. The number of species for epibenthos is at average 12% lower, compared to long fish tracks (Figure 3.13). For demersal fish, there is an underestimation of the number of species with 8%. Next to the actual number of species, the expected number of species per 1000 individuals (ES1000) was calculated (Primer 6). This diversity measure is less influenced by sample size (Soetaert and Heip, 1990), and therefore a better proxy for species richness. By the use of ES1000, we found an average underestimation of the number of species for epibenthos and demersal fish of respectively 8 and 4%.

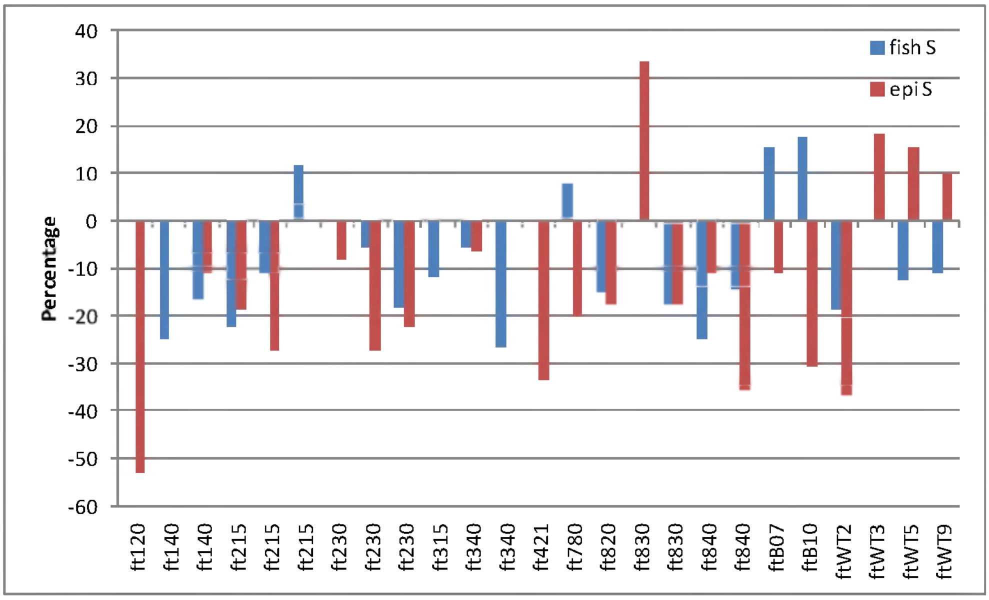

Figure 3.13: Percentage difference between short and long tracks in the number of species for fish and epibenthic species.

Discussion

The rate of overestimation or underestimation varied between tracks and between species groups. For the offshore tracks, a better correspondence between estimates was found compared to the coastal tracks. It seems that in areas with high epibenthic densities (coastal area), the net attained to be filled up, which influenced the net efficiency. Therefore, short tracks gives a better density estimate compared to long tracks. Estimations of diversity of short tracks are reliable, providing that the appropriate measure (expected number of species) is used. Detailed analyses of diversity (e.g. drafting species lists), however, should be based on longer tracks.

For subsequent analyses, the occurrence of overestimations or underestimations compared to standard tracks should be taken into account. Additionally, indices which do not depend on sample size should be adopted. This difference plays no part when tracks of similar kind (short) are taken and compared at the same time. A clear advantage of using short tracks in the dredge disposal research is the fact that the tracks fit within the borders of the dumping site. Like this, side effects are minimised and the short tracks seem to result in reliable density and diversity estimates of the area.

3.5.2 Fixed versus live sieving

Introduction

One of the elements that often vary among macrobenthic studies is the sieving procedure: sieving alive versus sieving after fixation. Before 2010, the ILVO monitoring programs used sieving after fixation as sieving procedure. This is not common in European macrobenthos monitoring and results in the use of high volumes of the carcinogenic formaldehyde. Therefore, from 2010 onwards, we switched to sieving alive. This difference in sieving procedure has its consequences on diversity, species densities and community structure (Degraer *et al.*, 2007). Diversity indices and the density of some species, mainly small, interstitial polychaetes, were negatively impacted. No major impact on the community composition was found.

Due to the effect of the sieving procedure on the benthic characteristics, we tested if sieving alive on a 0.5 mm + 1 mm sieve compensates for the loss of species. It is expected that a smaller mesh size sieve could nullify the loss of specimens due to sieving alive (Degraer *et al.*, 2007). This is important to know whether a change on long-term basis is caused by a change in sampling procedure. In this section, we investigated the difference between sieving alive, sieving on 2 sieves (0.5 mm+1 mm) and sieving fixed, for some long-term monitoring stations.

Material and Method

In winter and autumn 2010 6 replicate samples (whereof 3 were sieved fixed and the other three were sieved alive on a 1 mm and 0.5 mm sieve) were taken at some stations (120, 140, 330, 415, 830, 840, ZG01, ZG04, ZVL), The species were identified to species level when possible and counted. The macrobenthos sampling and analysis protocol is based on the ISO 16665:2005 standard.

The dataset was standardised and the univariate parameters number of species, density and biomass (wet weight) were calculated. The statistical difference in those benthic parameters between the three sieving procedures was tested with a one-way PERMANOVA (PRIMER6).

Results

A difference in the benthic parameters between the three sieving procedures was found. There was a decrease in average number of species, density and biomass from sieving fixed over sieving alive with 0.5+1 mm sieve towards sieving alive on a 1 mm sieve only. An average loss of 29% for diversity and 34% for density was detected when sieving alive compared to sieving fixed. There was also an average loss of 8% for diversity and 16% for density when there is sieving alive on a 0.5+1 mm sieve compared to sieving fixed.

There was no statistical difference between sieving fixed and sieving alive on a 0.5+1mm sieve for the parameters number of species and density (Table 3.8). Sieving alive on 1mm sieve gives significant lower results for the benthic parameters compared to the other methods. For the parameter biomass (wet weight), there was no significant difference between the sieving procedures. For the species composition there was only a significant difference between the fixed sieving procedure and the sieving alive on a 1mm sieve procedure.

Table 3.8: Permanova results (p level) for the parameter density, number of species (N0), biomass and species composition (ssp com) between the three sieving procedures. The non-significant results (p level >0.05).

N0 / Density	Fixed	Alive 0.5+1mm	Alive 1mm		Biomass/ Spp com	Fixed	Alive 0.5+1mm	Alive 1mm
Fixed		**0.4833**	0.0014		Fixed		**0.8373**	0.0199
Alive 0.5+1mm	**0.6551**		0.0039		Alive 0.5+1mm	**0.6975**		**0.2423**
Alive 1mm	0.0012	0.0019			Alive 1mm	**0.7271**	**1**	

Discussion

Sieving alive has a clear negative influence on the density and species richness of the samples, compared to sieving fixed. Therefore, caution is needed, in particular in habitats dominated by small, interstitial and/or larger, slender polychaetes. Larger polychaetes, polychaetes with obvious head capsules and appendages and more rigid species, such as amphipods and bivalves, are less prone to the impact (Degraer *et al.*, 2007). Based on the analysis on the effect of the sieving procedure on the benthic characteristics in the main benthic habitats, we can trust that data retrieved when sieving alive on a 0.5 + 1mm sieve is comparable with data retrieved for fixed sieving. Therefore, we can consider that this switch in sieving procedure will have a minor influence on the long term trend analysis at the benthic control stations. Since sieving with two sieves is only used at a certain subset of stations, we have to use conversion factors at the other stations for analyzing a long term trend.

3.5.3 Quality control macrobenthos analysis

A standardisation and harmonisation of the analysis procedures between the European countries is necessary, due to the requirements of the European environmental legislation (WFD, MSFD) and its monitoring. Due to the fact that macrobenthos is considered as an important ecosystem element for environmental monitoring in different directives, it is necessary to follow the international standards. Therefore, we adopted the ISO 16665:2005 standard for macrobenthos analysis ("Water quality – Guidelines for quantitative sampling and sample processing of marine soft-bottom macrofauna"). This guideline was already adapted in our benthic monitoring, except for the quality control and traceability of the samples procedure. A way to fill this in is to obtain an accreditation for certain analyses and lab working. In 2010-2011, our microscopy lab and macrobenthos analyses have been accreditated under BELAC ISO17025 norm (ILVO – DIER – ANIMALAB; CertificaatN°: BELAC T-315). This label is obtained on 24/05/2011. For the extern control of the counting and determination, we participate in the BEQUALM/NMBAQC scheme (UK).

4. Long- to medium-term bathymetry changes

In the framework of the BELSPO QUEST4D project (Van Lancker *et al.*, 2011) long- to medium-term bathymetry changes in the BPNS and in the coastal zone were investigated.

4.1. Long-term bathymetry changes in the BPNS

Based on historical navigation charts (~150 years ago up till now), the morphological evolution of the BPNS is studied. A selection of charts spanning a large spatial (a large part of the BPNS) and time (last centuries) domain, is being digitized. Table 4.1 presents the selected navigation charts that have been digitized (the most recent chart is a compilation of different datasets, already digitally available).

Table 4.1. Details of the digitized charts (Flemish Authorities, Division Coast, Hydrography).

Date	Title of chart	Surveyed
1866	Carte Générale de Bancs de Flandres compris entre Gravelines et l'embouchure de l'Escaut	?
1908	Mer du Nord Dunquerque – Flessingue	1901 – 1908
1938	Noordzee – Vlaamse Banken	?
1969	Noordzee – Vlaamse Banken	1959 – 1969
2007	(composed from different recent charts)	1997 - 2007

ArcGIS is used to interpolate the digitized charts to grids with a spatial resolution of 20m x 20m, making it possible to study the morphological evolution by visualisation of depth lines chart differencing and trend analysis. For the interpolation, a special methodology was used in order to create grids with smoothly varying depth values. In this methodology, a chart of a specific year is not only based on the interpolation of data points of that year, but also on the chart of 2007, which had a much higher data point density. By doing so, the derived charts not only are the result of an interpolation in space, but also in time, and it is implicitly assumed that the basic morphologic patterns (such as presence of sandbanks and gullies) on the BPNS are approximately stable, which is confirmed by earlier studies (Van Cauwenberghe, 1971, Mathys, 2010) (i.e. it is assumed that the large scale morphological structures present nowadays in the BPNS were also present in the past, on approximately the same location; however, this does not mean that these structures haven't changed in the course of time, e.g. banks or gullies may have accreted or eroded, or shifted over a limited distance). The application of this methodology stems from the observation that straightforward interpolation of the chart data points into a grid using one of the algorithms provided by ArcGIS would result in the artificial creation of pits and peaks on the location of some data points, making it impossible to compare the different grids, since different charts have data points on different locations. The details of the used interpolation methodology are explained in Janssens *et al.* (2011a).

It must still be noted that –given the large uncertainties on the water depths of the data points, especially on the older charts, results of the analysis of the grids must be handled with great care (a quantitative uncertainty analysis was not carried out). Using the ArcGIS bathymetry grids, long-term morphological evolution is studied by visualization of depth contours, chart differencing and trend analysis.

The erosion-sedimentation map of Figure 4.1 is the result of a trend analysis on the historical navigation charts and spans a period from ~150 years ago up till now. Only trends with R^2>0.5 are shown, and the scale is chosen to distinguish only 4 different types of trends (erosion-sedimentation, magnitude lower/higher than 2 cm/year). A finer scale would give a false impression of accuracy, which is not guaranteed by the historical navigation charts.

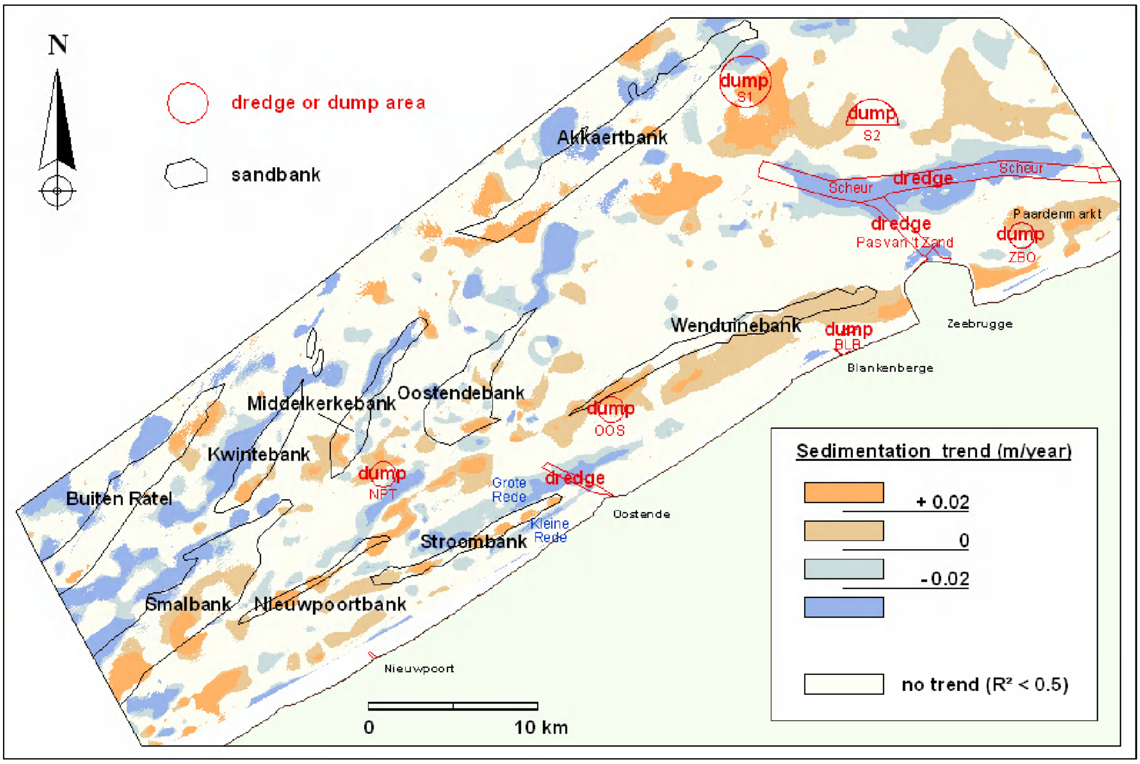

Figure 4.1: Erosion-sedimentation trend along the southwestern part of the BPNS, based on the bathymetric chart differences in the period 1866-2007.

Many significant morphological changes are the result of human activity: deepening due to dredging of the navigation channels towards the harbour of Antwerpen (Scheur) and Zeebrugge (Pas van het Zand) and towards Oostende, where the navigation channel cuts through the Stroombank. A clear sedimentation trend can also be observed in the shelter of the breakwaters of the outer harbour of Zeebrugge. Note that the figure shows the erosion/sedimentation trends for the past 150 years, which explains why the map shows no significant trend for Pas van het Zand, although it is clear that Pas van het Zand has deepened due to dredging works. However, significant dredging started only in the early 70s, which is why the deepening is not revealed as a persisting trend of the last 150 years. The same remarks can be made on the navigation channel Scheur, and on the sedimentation zones east and west of the Zeebrugge breakwaters. But apparently, morphological changes in these areas have been less prone to fluctuations, thus explaining an $R^2 > 0.5$ (the criterion for a "trend").

Results (based on the trend analysis, but also on visualization of depth contours, not shown here) also indicate that there is no significant movement of the sandbanks during the last 150 years, though some banks (especially the Coastal Banks: Smalbank, Nieuwpoortbank, Oostendebank and Wenduinebank) seem to be prone to erosion at the seaward side and to sedimentation at the coastal side; this could indicate minor movement towards the coast. However, taking into account the relatively large uncertainties on both position and depth values of the original data points, this movement cannot be considered significant, though it must be noted that none of the banks showed the opposite effect (seawards movement).

Nevertheless, the Coastal Banks are the most naturally dynamic zones: significant sedimentation trends are observed for the Nieuwpoortbank, the Wenduinebank and the Stroombank, while erosion occurred in the troughs between the banks: between the Stroombank and the coast (Kleine Rede), at the seaward side of the Stroombank (Grote Rede), and north of the Smalbank and of the Kwintebank. It is interesting to note that these long-term trends observed here seem to be confirmed by the medium-term analysis of the coastal zone.

4.2. Medium-term morphological evolution of the coastal zone

Medium-term bathymetry and topography changes along the Belgian coastline are studied, based on the digitally available topography and bathymetry datasets. Various datasets have been gathered at the Coastal Division (Agency for Maritime and Coastal Services) and include the yearly foreshore soundings and dune measurements, the inshore echo soundings and the offshore single beam measurements (from 1.5 to 10 km off the coast). Digital datasets are available roughly from 1995 till now, for beach and shoreface almost on a yearly basis, for the near coastal offshore zone time intervals between two successive datasets are typically 3 years. See Table 4.2 for a list of the available data sets for each zone.

Table 4.2. Overview of data used for the morphological trend analysis of the coastal zone.

Year	Beach	Shoreface	Near coastal offshore
1997-1		✓	Westende – De Haan
1997-2		✓	
1998-1		✓	Wielingen – Scheur
1998-2		✓	
1999-1	✓	✓	
1999-2	✓	✓	
2000-1	✓		Wielingen – Scheur
2000-2	✓		
2001-1	✓		Wielingen – Scheur
2001-2	✓		
2002-1			Zuydcoote – Westende
2002-2	✓		
2003-1		✓	Zuydcoote – Westende
2003-2			
2004-1	✓ **(summer)**	✓	Wielingen – Scheur
2004-2			
2005-1	✓ **(summer)**		Westende – De Haan
2005-2			
2006-1	✓		Zuydcoote – Westende
2006-2			
2007-1	✓	✓	Wielingen – Scheur
2007-2			
2008-1	✓	✓	Westende – De Haan
2008-2			
2009-1		✓	Westende – De Haan
2009-2			
2010-1	✓		

Datasets (mostly plain text files, containing XYZ data) were converted into more manageable bathymetry grids to be analyzed with ArcGIS software. This task was particularly elaborate because of different dataset formats, the use of different coordinate systems and bathymetry reference levels; sometimes clear metadata was lacking. Subsequently, medium-term evolution is analyzed through visualisation of depth contours and cross-sectional profiles, and trend analysis.

Results of the medium-term morphological study of the coastal zone are illustrated by four cases, namely the coastal zone between Oostende and De Haan, the coastal zone between De Panne and Koksijde, the coastal zone west of the port of Zeebrugge, and the area of the maritime access to the port of Zeebrugge and to the Schelde estuary. More details can be found in Janssens *et al.* (2011b).

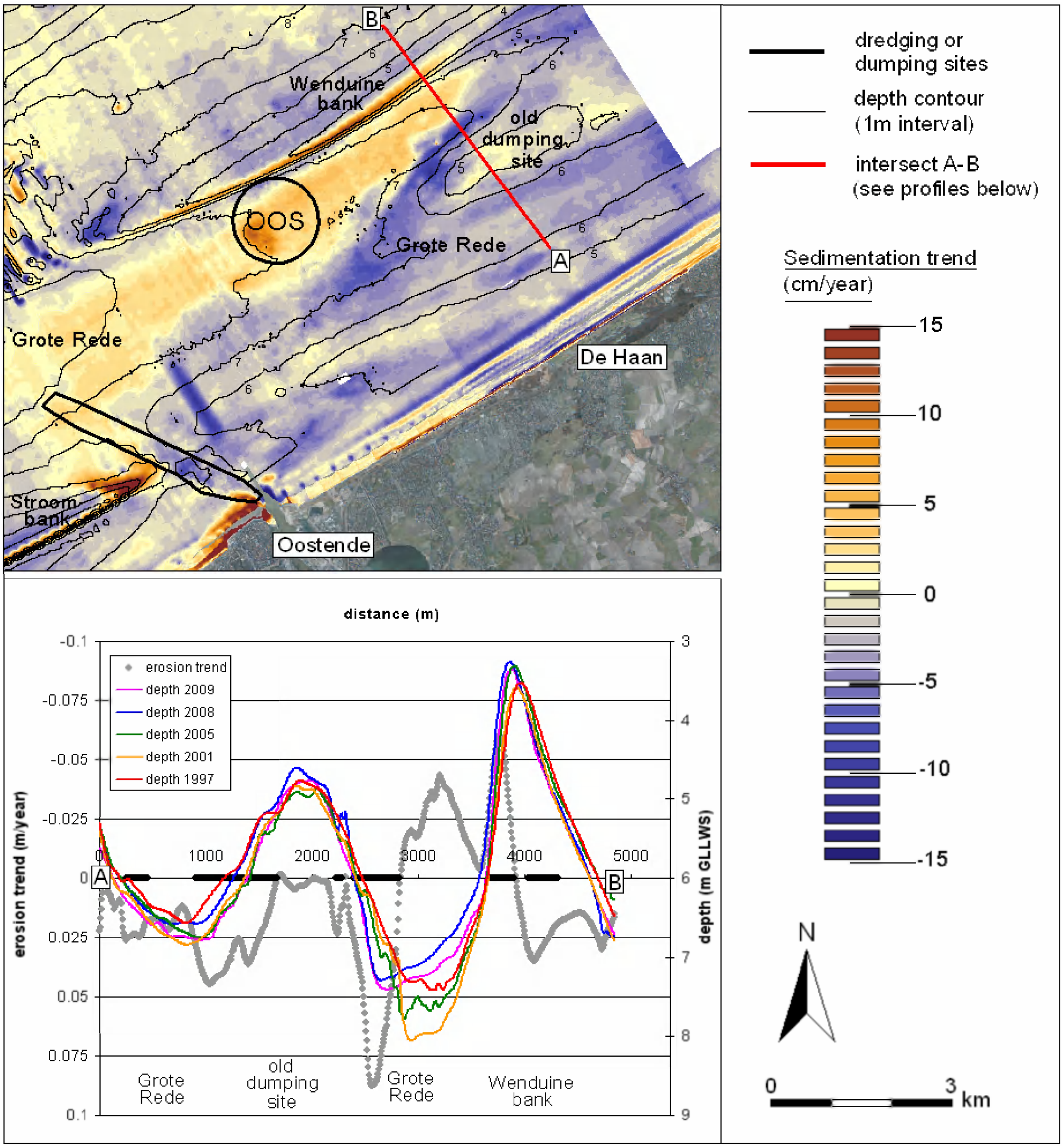

Figure 4.2: Top figure: erosion-sedimentation trend for beach, foreshore and the nearshore between Oostende and De Haan. Bottom figure: depth profiles (solid lines, right vertical axis) and erosion trend profile (dotted line, left vertical axis) along cross section A-B.

4.2.1. The coastal zone between Oostende and De Haan

Figure 4.2 shows the erosion-sedimentation trend of the past 15 years of the seafloor in front of Oostende and De Haan. The map shows a sedimentation trend on the landward side of both the Wenduinebank and the Stroombank, while erosion occurs on the seaward side of both banks. The Grote Rede trough also seems mostly prone to erosion. The figure also shows a time series of profiles along the transect A-B.

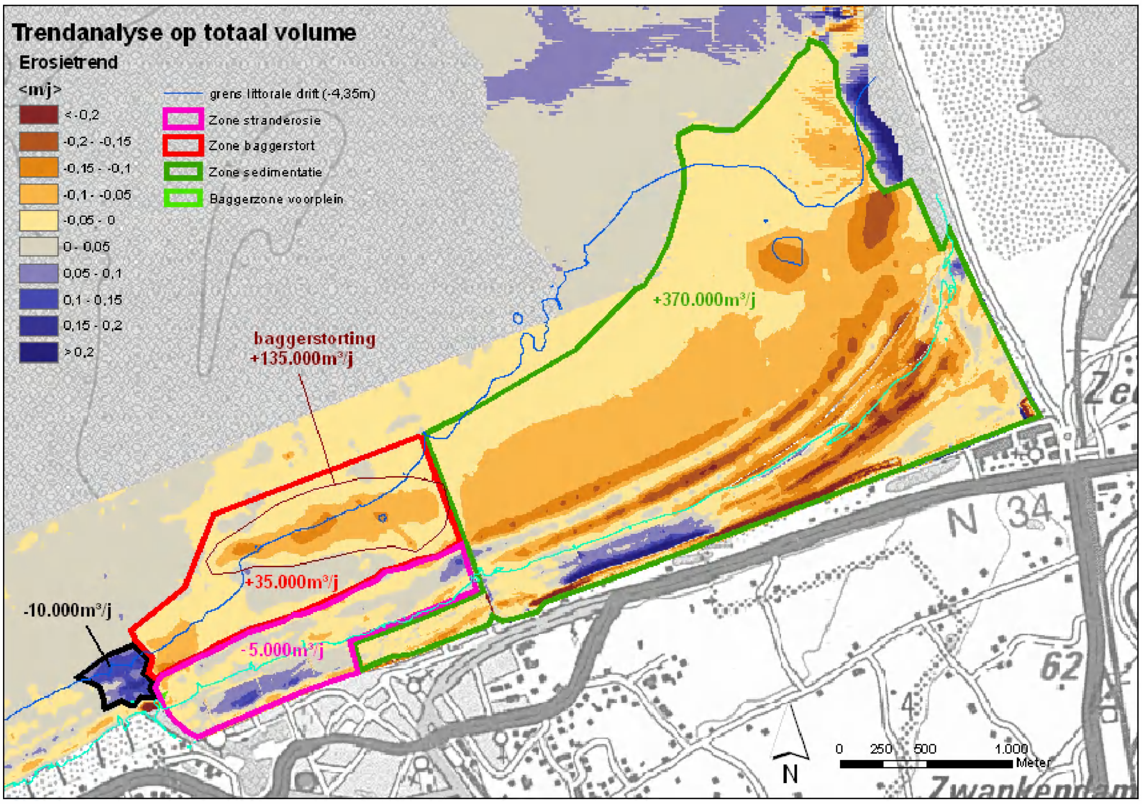

Figure 4.3: Result of the morphological trend analysis to delineate the accumulation area west of the western Zeebrugge breakwater, caused by blockage of the littoral drift. Green polygon: sedimentation due to blocking of the littoral drift. In the bottom left of this zone is a small area with a strong erosive character, which is due to the nourishment of the beach at the “Duinse Polders”. This nourishment was carried out shortly before the period over which the trend analysis was performed. The limited erosive area thus represents the return of the beach system to a more stable morphology, in the period following the nourishment. Pink polygon: downstream erosion due to the response of the system to the presence of Blankenberge harbour. Red polygon: accumulation, mainly due to the presence of the Blankenberge disposal site. The accumulation trend number shown (35×10^3 m^3/year) is the total yearly volumetric trend for this area, note that on average 135×10^3 m^3/year (Teurlincx *et al*., 2009) is dumped at the Blankenberge disposal site, it is therefore hypothesized that most of the dumped material is transported in the eastern direction, where it contributes to the accumulation trend of the green polygon. Black polygon: deepening, due to dredging of the navigation channel towards the Blankenberge marina.

4.2.2. The coastal zone west of the port of Zeebrugge

Based on the medium-term trend analysis of beach and foreshore in the area west of Zeebrugge, an estimation of the littoral drift can be made. The western breakwater of Zeebrugge harbour blocks the longshore transport of sediments along beach and foreshore, which is oriented from southwest to northeast along the Belgian coast. By quantification of the accumulation of sediments in the beach and foreshore area between Blankenberge and Zeebrugge, the magnitude of this littoral drift is estimated. Figure 4.3 shows the result of the morphological trend analysis. Beach and foreshore between Blankenberge and Zeebrugge are divided in different zones, depending on the observed trend and origin of the trend (see caption figure for more details). The total estimate of the net littoral drift of sand from southwest to northeast along the Belgian coast –based on the measured volumetric trends and assuming no sediment is brought into the system from offshore areas– is calculated as 400× 10^3 m^3/year (Verwaest *et al.*, 2011).

4.2.3. The area of the maritime access to the port of Zeebrugge and to the Scheldt estuary

One of the most outstanding features in the Belgian coastal zone is the harbour of Zeebrugge. The area surrounding the harbour shows strong erosion-sedimentation trends; these can be linked to mainly anthropogenic influence (Figure 4.4). Based on this information different zones are delineated, where results of the morphological evolution show very significant trends. In Figure 4.4, areas in green indicate accretion-dominated zones; areas in red, erosion-dominated zones. Areas 1, 2, 3 and 7 show trends that are likely within the influence area of the harbour of Zeebrugge and the dredging and disposal activities in the navigation channels (Pas van het Zand and Scheur) and on the disposal site Br&W S2, respectively. Trends for areas 4 (ebb- dominated trough, in the extension zone of the Westerschelde), 5 (Paardenmarkt shoal) and 6 cannot be directly linked to anthropogenic intervention. These variations in medium-term erosion/deposition rates reveal far field effects of human activities, see Table 4.3. It is quite remarkable that dumping site Br&W Zeebrugge Oost shows no clear accretion trend, notwithstanding the large quantities of dumped material (on average 2.7 million m^3/year (Lauwaert *et al.*, 2009)), on the contrary, a large part of Br&W Zeebrugge Oost seems to be prone to erosion. It appears that dumped material is easily washed away from Br&W Zeebrugge Oost. Possible explanations are the relatively large currents at Br&W Zeebrugge Oost in comparison with other dumping sites (due to Zeebrugge breakwaters), influence of the Westerschelde estuary, and/or the nature of the dumped material (at Br&W Zeebrugge Oost mostly mud).

Table 4.3: Classification of the different zones with significant morphological evolution in the area of the maritime access to the port of Zeebrugge and to the Schelde estuary

Zone	Dominant influence	Hypothesized cause (partly or fully)	Effect
Zone 1	Anthropogenic	Presence of a large breakwater downstream	Blocking of longshore transport
Zone 2	Anthropogenic	Presence of a large breakwater upstream	Accretion
Zone 3	Anthropogenic	Dredging of navigation channels + tidal flow concentration caused by the presence of the large breakwaters	Deepening of the bathymetry
Zone 4	Natural	Influence area of the Westerschelde?	Erosion of an ebb-dominated trough
Zone 5	Natural? Anthropogenic?	Needs further research	Accretion of bank crest that could be interpreted as migration
Zone 6	Natural? Anthropogenic?	Needs further research	Accretion
Zone 7	Anthropogenic	Disposal activities at S2	Accretion

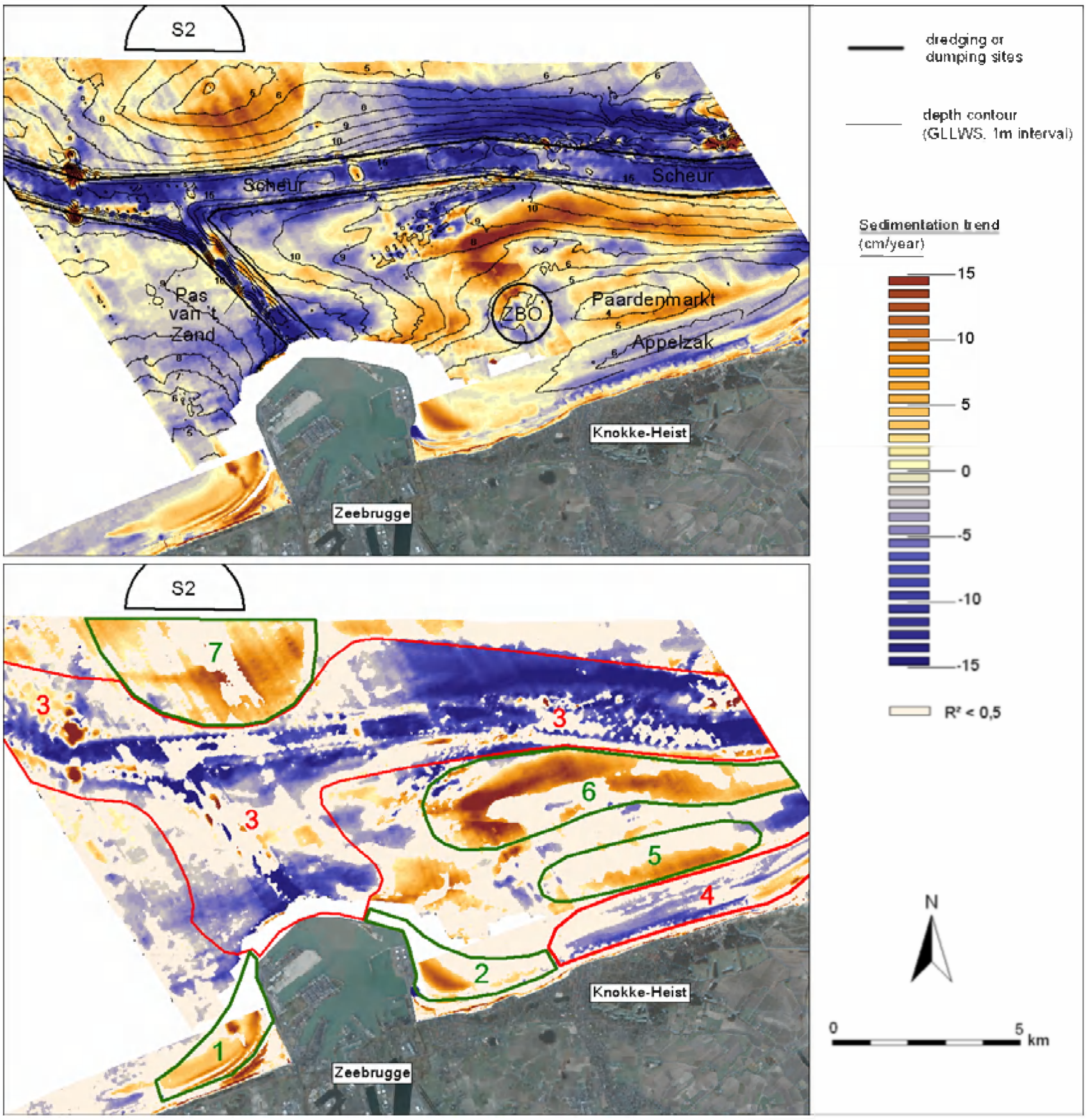

Figure 4.4: Erosion-sedimentation trend in the area of the maritime access to the port of Zeebrugge and to the Schelde estuary. The top figure shows all trends; the lower figure only trends with $R^2>0.5$. Zone 1: This zone extends between Blankenberge and Zeebrugge harbour. The western breakwater of the harbour constitutes a physical obstacle blocking longshore sand transport (which is oriented from southwest to northeast). As a consequence, accumulation of sand at high rates is observed upstream the harbour. Zone 2: This zone concerns the "Baai van Heist" area, east of Zeebrugge harbour. Downstream the harbour, accretion occurs due to (1) blocking of the E-W directed littoral drift during ebb and waves coming from the north; and (2) deceleration of the tidal flow, after being accelerated in the vicinity of the harbour entrance, during flood. Zone 3: This zone includes the main navigation channels towards the Westerschelde (Scheur) and Zeebrugge harbour (Pas van het Zand). It is hypothesised that continuous dredging activities in the navigation channels have an influence on the surrounding seafloor, see e.g. the strong erosion trends extending beyond the dredging boundaries of the Scheur navigation channel (though other mechanisms may be interfering: the large erosion zone north of the eastern part of the Scheur navigation channel will very likely also experience the influence of the Westerschelde). Where this zone nears the harbour entrance, the influence of the breakwater structures on the flow pattern (contraction of stream lines) is perceived; this is associated with erosion in front of the harbour entrance. Zone 7: Located beyond the Pas van het Zand; this zone is likely influenced by disposal of dredged material at disposal site Br&W S2, though it must be stressed that other more important sedimentation mechanisms are certainly at play as well (accretion trend in zone 7: 1.4 million m^3/year, while only about 0.62 million m^3/year is on average disposed at Br&W S2 in the period 1997-2007 (Lauwaert *et al.*, 2008).

5. Dredging and Dumping: Alternative dumping location

One of the long-term goals of the MOMO project (see chapter 2) is to provide ideas and opportunities to develop more efficient dredging and dumping (Fettweis *et al.*, 2011b). One of the ways to achieve this is to select dumping zones where recirculation towards the dredging zones is small, thus reducing the need for dredging. It is clear that the dumping zones also need to be economically (distance to be covered by the dredging vessels) and ecologically (situated in a location where expected impact on the biological level is low) interesting. In the summary report 2008-2009 (Lauwaert *et al.*, 2009), recommendations to the Minister stated that a field test for an alternative dumping location should be prepared. Since then, a report has been prepared by Fettweis *et al.* (2011b), that details the possible locations for an alternative dumping location, and delineates a monitoring programme that will provide the necessary feedback on the results of the test.

5.1 Field experiment – outline

The conclusions from previous tracer experiments (HAECON, 1993a; 1993b; 1995) and numerical simulations (Fettweis *et al.*, 2011b) suggest that Br&W Zeebrugge Oost is inefficient and that a significant amount (approx 35 percent, depending on the meteorological circumstances) of the deposited material recirculates to the dredging sites within a month. Given that sediment transport models are an approximation of reality, a field test will be carried out where dumping will take place in an alternative location to validate the results from the numerical models. The purpose of this experiment specifically is to demonstrate that a reduction in the dredging can be obtained by using alternative dumping site for Br&W Zeebrugge Oost. The choice of the potential alternative dumping locations is based on the results of model calculations, as described in Fettweis *et al.* (2011b). Five possible locations were researched, which are shown in Figure 5.1. ZBR 4 (near Wandelaar) was chosen because the recirculation was smallest, thus providing the best chance to measure the effects from the field experiment.

The field experiment will take place in the autumn of 2012. During one month, all dredged material that normally would be dumped in Br&W Zeebrugge Oost (approx 250000-300000 TDS), will be dumped on alternative dredging site 4. A monitoring programme will be set up before, during and after the field experiment.

5.2 Monitoring programme

The monitoring programme of the field experiment will consist of the following components

- Monitoring of the SPM concentration inside and outside the harbour for a one –year period (including the month-long field experiment) on different locations.
- Monitoring of the effects on the location of the top silt layer in the harbour during the field experiment.
- Monitoring of the effects on the dredging intensity during the field experiment.
- Monitoring of the bathymetrical evolution of the dumping site before and after the field experiment.

A statistical analysis of the SPM concentration will be carried out, following the method as used in the analysis for the dredging experiment in the Albert II dock (Fettweis *et al.*, 2011a). During the field experiment weekly meetings with all involved parties will be held to assure a good follow up of the field experiment.

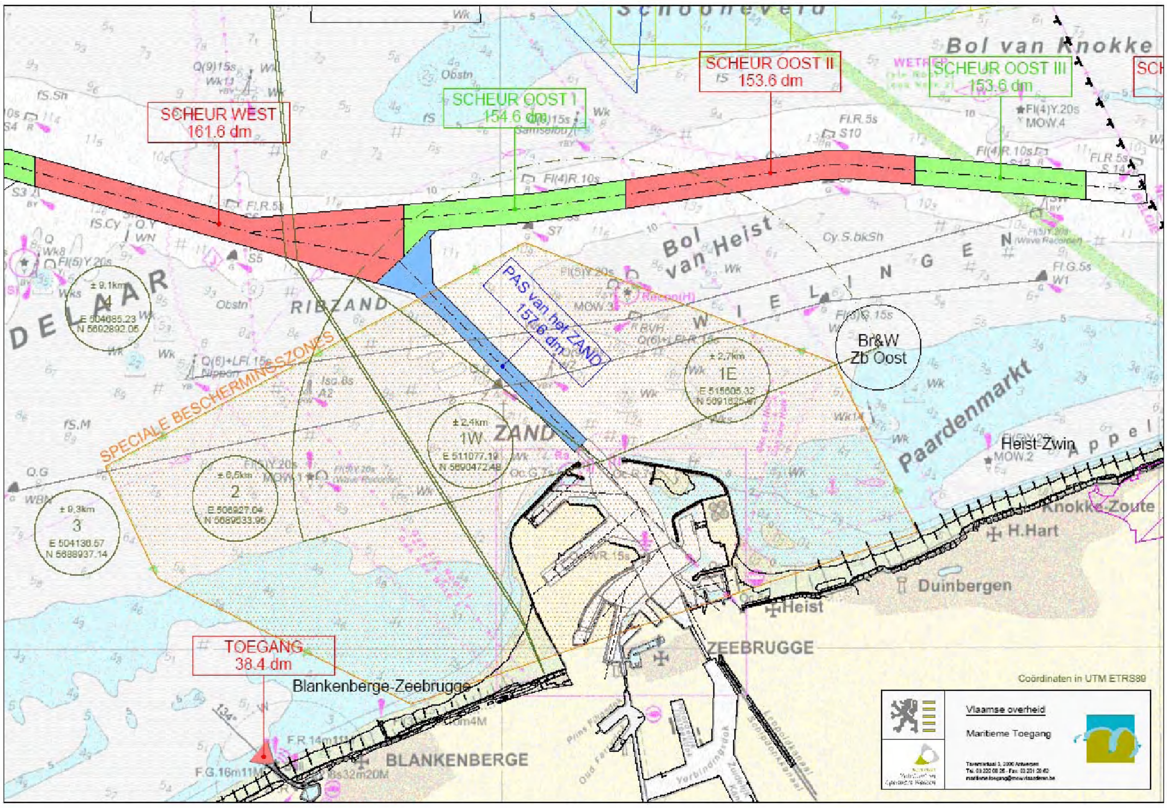

Figure 5.1: location of the potential field test sites. Testing will take place a t site n°4 (near Wandelaar).

5.3 Expected results

The numerical model results predict a decrease in dredging works of 10% for the outer harbour of Zeebrugge, and 3% in Pas van het Zand. If the results obtained by the field experiment confirm the assumptions made based on the results of the numerical modelling, a new and more efficient dumping location could be defined.

This new dumping location does not necessarily need on the exact location as used in the field experiment. For the field experiment, minimisation of the recirculation was the principal parameter to select a location within a reasonable sailing distance from the dredging locations. When a definitive dumping location is designated, a whole set of additional parameters need also to be taken into account, most importantly the ecological and economical factors. In order to prepare for ecological assessment, a quick scan of the biological environment West of Zeebrugge will be carried out by ILVO. This area has a high variation in different habitats, and a good knowledge of the existing habitats and their geographical distribution is essential for a good planning of the possible future dumping site. The selection of the definitive location will consist of an optimisation of the ecological and economical parameters.

6 Research project – Disposal of dredged material from the marina of Nieuwpoort using a pipeline

6.1 Introduction

The coastal harbours and marinas in Belgium are confronted with the never ending problem of shoaling and accretion. This causes the harbour channel and docks to experience difficulties in maintaining their accessibility. The dredged material, being brought in by the sea, has been found to have different levels of thickness and density. Depending on the location and in-situ conditions both sandy and muddy matter can be found.

The Coastal Division of the Agency for Maritime and Coastal Services (Flemish government) is responsible for the accessibility of the marinas of Nieuwpoort and Blankenberge (access channel and docks) and for the harbours of Oostende and Zeebrugge (the docks). In the marinas of Nieuwpoort, Oostende and Blankenberge respectively 250000 m³, 20000 m³ and 80000 m³ are being dredged on average annually. Currently these quantities have been removed using the classical approach: dredging using a cutter dredger, transfer of the dredged material in loading containers and disposal at sea at the designated disposal sites. The yearly dredging campaign runs from November to April. In addition to this a restricted amount of sandy matter, originating from the harbour channels Nieuwpoort and Blankenberge, is used for sea defence (beneficial use). This dredges material is used for foreshore nourishments after receiving a utility certificate.

This pilot project will investigate if the dredging activities in the marinas and depositing the dredge spoil can be done in a more efficient, environmental friendly and economical manner. The pilot will be executed in accordance with art. 25 §3 ii of the Law of the 20th of January 1999 to protect the marine environment in coastal areas in the entitlement of rights of Belgium.

6.2 Project description

Since the largest quantity of sludge is being removed in Nieuwpoort it is suggested to start a pilot project in this marina (Figure 6.1). Research will conducted if a pipeline running a short distance can be placed at sea for the execution of the limited pilot project. Using this approach, the dredged mud will be retrieved using a cutter dredger and transported to the disposal site a short distance from the shore line. This in contrast with the current method, where the dredged matter is transported using loading containers and disposed at the disposal site of Nieuwpoort.

It is proposed that in this pilot project at first only 50,000 to 75,000 m³ is being dredged at that site (compared to the annual dredging mean in Nieuwpoort of 250,000 m³). For organisational purposes this project should be executed either before the classical dredging campaigns in October/November, or in April/May. Using this approach all dredging activities can be performed during one dredging campaign. The way in which the suction hopper brings in the sludge remains the same. The suction hopper moves in the marina to the zones where sludge can be found. The retrieved matter will than be transported to sea by a high pressure pipe that is connected to a solid pipeline. Due to the distance that has to be covered a booster station will be installed ashore.

In total the dredging will continue for 3 to 4 weeks, with a planned working regime of 24 hours a day. The discharge will be around 160 to 200 m³/h, taking in liquid sludge with a density of about 1.07 to 1.09 kg/l. The final disposal site is to be decided in consultation with all partners depending on the feasibility of the pilot project and the possibilities for the final disposal site location.

Figure 6.1: Marina of Nieuwpoort.

6.3 Advantages

The following advantages of this alternative method can be mentioned.

The efficiency of the dredging activities can be significantly increased. Since time for connecting and disconnecting the loading containers is not needed anymore the cutter can be 1.5 times more productive. Furthermore, bad weather will not delay the work being done.

Since the dredging activities can be conducted quicker, the day and night regime (7 days of 7, 24h a day) is not necessary. The (noise) hindrance during the nights and weekends can be avoided. The period in which the dredging will be conducted will be shortened. The surveillance and control on the works can therefore be intensified.

Navigation between marina and wharf (in this case: Nieuwpoort) is not needed anymore. This means a considerable decline in CO_2 emissions and small dust particles.

It needs to be mentioned that the reuse of the sand rich dredge spoil (beneficial use) is equally well possible.

6.4 Follow up

6.4.1 Sediment measurements

During the pilot project a monitoring campaign will be done. At the wharf site the water depth will be measured on a weekly basis by Flemish Hydrography. Parameters in the direct vicinity of the wharf will be measured using a tripod (parameters include salinity, temperature, current velocity and sediment concentrations). These measurements will be carried out by MUMM in cooperation with DAB VLOOT for placing and retrieving the tripod.

This tripod will be placed three times during a period of 1 month, meaning before, during and after the dredging works. Based on the measurements of the tripod a possible increase of sludge concentrations can be determined.

6.4.2 Ecological impact

Due to a point discharge of the dredged material when working with a pipeline, the ecological impact will be local. An adequate ecological sampling design for following up point discharge is to work with a cross gradient design. In this design, you have to take benthic samples at certain distance intervals from the point discharge and this in different directions. A specific monitoring program can be developed within the ecological dredging monitoring program of ILVO. A lot of benthic data of the coastal area nearby Nieuwpoort is available to serve as reference.

7 Marine Strategy Framework Directive

The MSFD is the first piece of EU legislation aimed at the protection of Europe's marine environment as a whole. Its main goal is to achieve Good Environment Status (GES) in all EU waters by 2020 by applying an ecosystem-based approach to the management of human activities which have an impact on the marine environment (Figure 7.1).

The MSFD was published in 2008 and it stipulates that by the 15th July 2010, the Member States of the EU should transpose the Directive's provisions into their national legislative frameworks, and designate the authority (or authorities) competent for the implementation of the Directive.

By the same date, the Commission should have published criteria and methodological standards to be used by the Member States when implementing the Directive, so as to ensure consistency and to allow for comparison between Marine Regions or Sub-Regions. These criteria have been agreed upon by the Commission, but they are still going through scrutiny by the European Parliament, and are therefore only expected to be published in September.

The next challenge relates to a further implementation milestone on the 15th July 2012 whereby Member States must conduct an initial assessment on the status of their marine waters. By this date they must also define what they understand to be GES in their marine waters and set environmental targets and associated indicators to drive their progress towards achieving GES.

The MSFD has great potential to bring about the protection and restoration to a heavily impacted European marine environment. The next stage however is crucial in order to accomplish such objectives: ambitious environmental targets are the key to Member States taking the much needed and necessary actions to restore our ocean and seas to health. Marine biodiversity is under severe pressure:

- Habitat destruction
- Fragmentation and degradation
- Over-exploitation
- Unsustainable practices
- Invasive species
- Ocean acidification
- Pollution
- Climate

Each EU Member State must progressively put in place its own “Marine Strategy” (action plan), consisting of several steps: they must cooperate among themselves and also with neighbouring countries and where possible within Regional Sea Conventions (e.g. OSPAR, Barcelona, Helcom, Black Sea). Member States will have to produce by 15 July 2012:

- Description and assessment of current environmental status, including the environmental impact of human activities & socio-economic analysis;
- Determination of GES to be achieved(precise ecological objectives);
- Establishment of environmental targets and associated indicators.

Member States will have to have monitoring programmes for all marine waters (adapted to the assessment of progress towards GES) by 15 July 2014. By 2015 all Marine Strategies will culminate with a programme of measures towards achieving GES in 2020. Every 6 years there the MSFD will be reviewed and adapted where needed. The descriptors of GES are:

- Biological diversity
- Non-indigenous species
- Population of commercial fish/shellfish
- Elements of marine food web/reproduction
- Eutrophication
- Sea floor integrity
- Alteration of hydrographical conditions
- Contaminants
- Contaminants in seafood
- Marine litter
- Energy incl. underwater noise

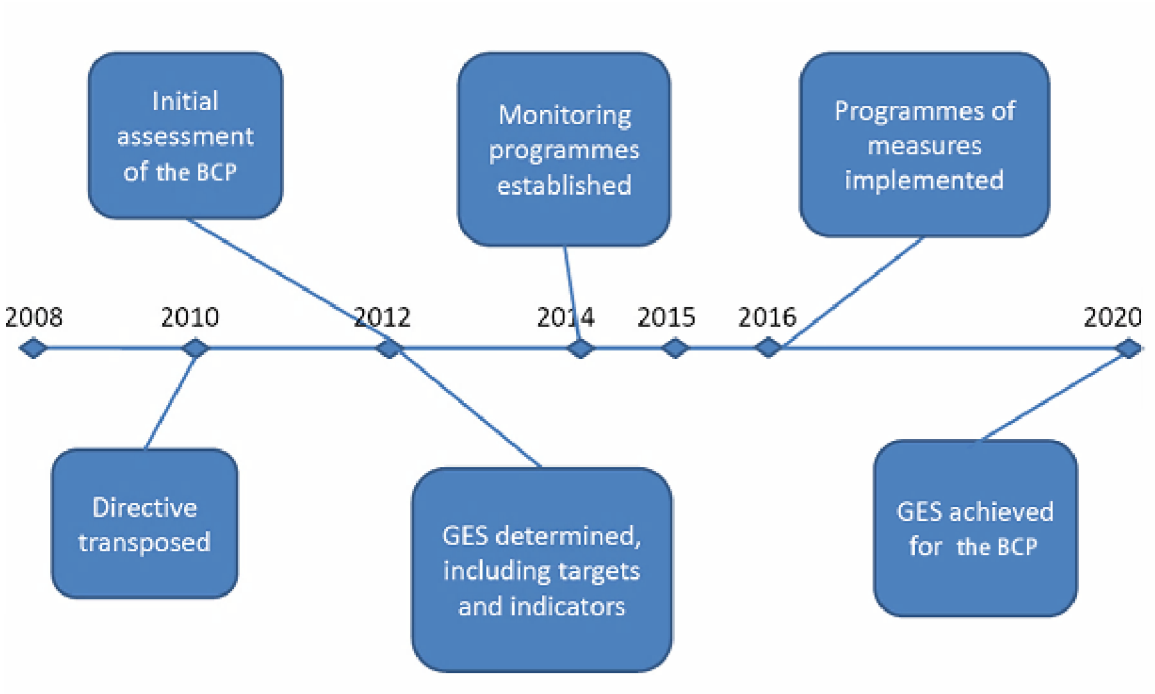

Figure 7.1: Time frame of MFSD implementation.

References

Baeye M, Fettweis M, Voulgaris G, Van Lancker V. 2011. Sediment mobility in response to tidal and wind-driven flows along the Belgian inner shelf, southern North Sea. Ocean Dynamics, 61, 611-622. doi:10.1007/s10236-010-0370-7.

Belgisch Staatsblad N° 209, vrijdag 9 juli 2010.

Bilyard GR. 1987. The value of benthic infauna in marine pollution monitoring. Marine Pollution Bulletin, 18, 581-585..

Bolam SG, Rees LH. 2003. Minimizing impacts of maintenance dredged material disposal in the coastal environment: A habitat approach. Environmental Management, 32(2), 171–188.

Bolam SG, Schratzberger M, Whomersley P. 2006. Macro- and meiofaunal recolonisation of dredged material used for habitat enhancement: Temporal patterns in community development. Marine Pollution Bulletin, 52, 1746-1755.

Borja A. 2005. The new European Marine Strategy Directive: Difficulties, opportunities, and challenges. Marine Pollution Bulletin, 52, 239-242. doi:10.1016/j.marpolbul.2005.12.007.

Bray RN, Bates AD, Land JM. 1996. Dredging and the Environment. In: Dredging: A Handbook for Engineers, Elsevier, 371-387.

Bucke D, Vethaak D, Lang T, Mellergaard S. 1996. Common diseases and parasites of fish in the North Atlantic: Training guide for identification. ICES TIMES N°19.

Bunt JAC, Larcombe P, Jago CF. 1999. Quantifying the response of optical backscatter devices and transmissometers to variations in suspended particulate matter. Continental Shelf Research, 19, 1199-1220.

Burke MD, Mayer RT. 1974. Ethoxyresorufin: direct fluorimetric assay of a microsomal O-dealkylation which is preferentially inducible by 3-methylcholanthrene. Drug Metabolism and Disposition, 2, 583-588.

Collins M. 1990. The behaviour of cohesive and non-cohesive sediments. In: Proc. Int. Seminar Environment. Asp. Dredging (Alzieu C, Galenne B, eds.), Port Autonome de Nantes Saint-Nazaire, Nantes, 15–32.

Degraer S, Moulaert I., Van Hoey G, Vincx M. 2007. Sieving alive or after fixation: effects of sieving procedure on macrobenthic diversity, density and community structure. Helgoland Marine Research, 61, 143-152.

Degraer S, Verfaillie E, Willems W, Adriaens E, Vincx M, Van Lancker V. 2008. Habitat suitability modelling as a mapping tool for macrobenthic communities: An example from the Belgian part of the North Sea. Continental Shelf Research, 28, 369-379.

Degraer S, Braeckman U, Haelters J, Hostens K, Jacques T, Kerckhof F, Merckx B, Rabaut M, Stienen E, Van Hoey G, Van Lancker V, Vincx M. 2009. Studie betreffende het opstellen van een lijst met potentiële Habitatrichtlijn gebieden in het Belgische deel van de Noordzee. Eindrapport in opdracht van de Federale Overheidsdienst Volksgezondheid, Veiligheid van de Voedselketen en Leefmilieu, Directoraat-generaal Leefmilieu. Brussel, België. 93pp.

De Nijs MAJ, Winterwerp J, Pietrzak JD. 2009. On harbour siltation in fresh-salt water mixing region. Continental Shelf Research, 29, 175-193. doi:10.1016/j.csr.2008.01.019

Derweduwen J, Vandendriessche S, Hostens K. 2010. Evaluation of short tracks. In: Degraer S, Brabant R, Rumes B. (Eds.). Offshore wind farms in the Belgian part of the North Sea: Early environmental impact assessment and spatio-temporal variability. Royal Belgian Institute of Natural Science, Management Unit of the North Sea Mathematical Models. 184pp.

Devriese L, Bekaert K, Hoffman S, Cooreman K, Robbens J. 2011. Patterns and Prevalence of Fish Diseases on the Belgian Part of the North Sea, Poster Presentation on PRIMO16, May 2011, CA USA.

Devlin M, Bets M, Haynes D. 2007. Implementation of the Water Framework Directive in European marine waters. Marine Pollution Bulletin, 55, 1-2. doi:10.1016/j.marpolbul.2006.09.020

de Wit PJ, Kranenburg C. 1997. The wave-induced liquefaction of cohesive sediment beds. Estuarine, Coastal and Shelf Science, 45, 261-271.

Elliot M, Hemingway KL, et al. 2002. Links between fish and other trophic levels. In: Elliot M, Hemingway K. Fishes in estuaries, 124-216.

Fettweis M, Sas M. 1999. On the sedimentation of mud in access channels to the harbour of Antwerp. PIANC Bulletin, 101, 53-59.

Fettweis M, Van den Eynde D. 2003. The mud deposits and the high turbidity in the Belgian-Dutch coastal zone, Southern bight of the North Sea. Continental Shelf Research, 23, 669-691. doi: 10.1016/S0278-4343(03)00027-X

Fettweis M, Francken F, Pison V, Van den Eynde D. 2006. Suspended particulate matter dynamics and aggregate sizes in a high turbidity area. Marine Geology, 235, 63-74. doi:10.1016/j.margeo.2006.10.005

Fettweis M, Houziaux J-S, Du Four I, Van Lancker V, Baeteman C, Mathys M, Van den Eynde D, Francken F, Wartel S. 2009. Long-term influence of maritime access works on the distribution of cohesive sediments: analysis of historical and recent data from the Belgian nearshore area (southern North Sea). Geo-Marine Letters, 29, 321-330. doi: 10.1007/s00367-009-0161-7

Fettweis M, Francken F, Van den Eynde D, Verwaest T, Janssens J, Van Lancker V. 2010. Storm influence on SPM concentrations in a coastal turbidity maximum area with high anthropogenic impact (southern North Sea). Continental Shelf Research, 30, 1417-1427. doi:10.1016/j.csr.2010.05.001

Fettweis M, Baeye M, Francken F, Lauwaert B, Van den Eynde D, Van Lancker V, Martens C, Michielsen T. 2011a. Monitoring the effects of disposal of fine sediments from maintenance dredging on suspended particulate matter concentration in the Belgian nearshore area (southern North Sea). Marine Pollution Bulletin, 62, 258-269. doi:10.1016/j.marpolbul.2010.11.002

Fettweis M, Chen P, Francken F, Van den Eynde D, Van Lancker V, Yu JCR. 2011b. MOMO activiteitsrapport 3 (1 januari 2011 – 30 juni 2011). BMM-rapport MOMO/5/MF/201107/NL/AR/3, 29pp.

Fettweis M, Nechad B. 2011. Evaluation of in situ and remote sensing sampling methods for SPM concentrations on the Belgian continental shelf (southern North Sea). Ocean Dynamics, 61, 157-171. doi:10.1007/s10236-010-0310-6

Fettweis M, Baeye M, Lee BJ, Chen P, Yu JCR. 2012. Hydro-meteo influences and multimodality of suspended particle size distributions in a mixed sediment environment. Geo-Marine Letters. doi: 10.1007/s00367-011-0266-7

Fugate DC, Friedrichs CT. 2002. Determining concentration and fall velocity of estuarine particle populations using ADV, OBS and LISST. Continental Shelf Research, 22, 1867-1886.

Gerritsen H, Boon JG, van der Kaaij T, Vos RJ. 2001. Integrated modelling of suspended matter in the North Sea. Estuarine, Coastal and Shelf Science, 53, 581-594. doi: 10.1006/ecss.2000.0633

Hamilton LJ, Z Shi, SY Zhang. 1998. Acoustic backscatter measurements of estuarine suspended cohesive sediment concentration profiles. Journal of Coastal Research, 14, 1213-1224.

Hill AS, Veale LO, Pennington D, Whyte SG, Brand AR, Hartnoll RG. 1999. Changes in Irish Sea benthos: possible effects of 40 years of dredging. Estuarine, Coastal and Shelf Science, 48, 739-750.

Hoitink AJF, Hoekstra P. 2005. Observations of suspended sediment from ADCP and OBS measurements in a mud-dominated environment. Coastal Engineering, 52, 103-118. doi:10.1016/j.coastaleng.2004.09.005.

Houziaux J-S, Fettweis M, Francken F, Van Lancker V. 2011. Historical (1900) seafloor composition in the Belgian-Dutch part of the North Sea: A reconstruction based on calibrated visual sediment descriptions. Continental Shelf Research 31, 1043-1056. doi:10.1016/j.csr.2011.03.010

Janssens J, Verwaest T, Mostaert F. 2011a. Morfologische evolutie van het Belgisch continentaal plat gedurende de laatste 150 jaar: Deelrapport in het kader van het Quest4D-project. Versie WL2011T814_2rev1_0 (deel 1). WL Technische Nota's. Waterbouwkundig Laboratorium: Antwerpen, België.

Janssens J, Verwaest T, Mostaert F. 2011b. Morfologische evolutie van strand en vooroeverin het laatste decennium: Deelrapport in het kader van het Quest4D-project. Versie WL2011T814_2rev1_0 (deel 2). WL Technische Nota's. Waterbouwkundig Laboratorium: Antwerpen, België.

Jonasz M, Fournier G. 1996. Approximation of the size distribution of marine particles by a sum of log-normal functions. Limnology and Oceanography, 41, 744-754.

Kim HY, Gutierrez B, Nelson T, Dumars A, Maza M, Perales H, Voulgaris G. 2004. Using the acoustic Doppler current profiler (ADCP) to estimate suspended sediment concentration. Technical Report CPSD #04-01.

Lauwaert B, Bekaert K, Berteloot M, De Brauwer D, Fettweis M, Hillewaert H, Hoffman S, Hostens K, Mergaert K, Moulaert I, Parmentier K, Vanhoey G, Verstraeten J. 2008. Syntheserapport over de effecten op het mariene milieu van baggerspeciestortingen (vergunningsperiode 2006-2008). BMM, ILVO, aK & aMT rapport, BL/2008/01, 128pp.

Lauwaert B, Bekaert K, Berteloot M, De Backer A, Derweduwen J, Dujardin A, Fettweis M, Hillewaert H, Hoffman S, Hostens K, Ides S, Janssens J, Martens C, Michielsen T, Parmentier K, Van Hoey G, Verwaest T. 2009a. Synthesis report on the effects of dredged material disposal on the marine environment (licensing period 2008-2009). Report by BMM, ILVO, CD, aMT and WL BL/2009/01. 73pp.

Lang T. 2002. Fish disease surveys in environmental monitoring: the role of ICES. ICES Marine Science Symposia, 215, 202-212.

Law R, Hanke G, Angelidis M, Batty J, Bignert A, Dachs J, Davies I, Denga Y, Duffek A, Herut B, Hylland K, Lepom P, Leonards P, Mehtonen J, Piha H, Roose P, Tronczynski J, Velikova V, Vethaak D. 2010. Marine Strategy Framework Directive, Task Group 8 Report: Contaminants and pollution effects. Joint Report prepared under the Administrative Arrangement between JRC and DG ENV (no 31210 – 2009/2010), the Memorandum of Understanding between the European Commission and ICES managed by DG MARE, and JRC's own funding.

Lee D-I, Eom K-H, Kim G-Y, Baeck G-W. 2010. Scoping the effective marine environmental assessment of dredging and ocean disposal of coastal sediments in Korea. Marine Policy, 34, 1082-1092. doi:10.1016/j.marpol.2010.03.008

Le Hir P, Monbet Y, Orvain F. 2007. Sediment erodability in sediment transport modelling: can we account for biota effects? Continental Shelf Research, 27, 1116–1142. doi:10.1016/j.csr.2005.11.016.

Li Y, Mehta AJ. 2000. Fluid mud in the wave-dominated environment revisited. In: Coastal and Estuarine Fine Sediment Dynamics. (McAnally WH, Mehta AJ, eds.), Proceedings in Marine Science, 3, 79-93.

Limpert E, Stahel W, Abbt M. 2001. Log-normal distributions across the sciences: Keys and clues. BioScience, 51, 341-352.

Lohrer AM, Wetz JJ. 2003. Dredging-induced nutrient release from sediments to the water column in a southeastern saltmarsh tidal creek. Marine Pollution Bulletin, 46, 1156-1163. doi:10.1016/S0025-326X(03)00167-X

Maa P-Y, Mehta AJ. 1987. Mud erosion by waves: a laboratory study. Continental Shelf Research, 7, 1269–1284.

Manning AJ, Baugh JV, Spearman J, Whitehouse RJS. 2010. Flocculation settling characteristics of mud: sand mixtures. Ocean Dynamics, 60, 237-253. doi:10.1007/s10236-009-0251-0

Mathys M. 2010. Het onderwaterreliëf van het Belgisch deel van de Noordzee. De Grote Rede, 26, 16-26.

Maurer D, Keck RT, Tinsman JC, Leathem WA. 1981a. Vertical migration and mortality of benthos in dredged material, 1 Mollusca. Marine Environmental Research, 4, 299-319.

Maurer D, Keck RT, Tinsman JC, Leathem WA. 1981b. Vertical migration and mortality of benthos in dredged material, 2 Crustacea. Marine Environmental Research, 5, 301-317.

Maurer D, Keck RT, Tinsman JC, Leathem WA. 1982. Vertical migration and mortality of benthos in dredged material, 3 Polychaeta. Marine Environmental Research, 6, 49-68.

OSPAR. 2000. Quality Status Report, Region II Greater North Sea.

Mikkelsen OA, Hill PS, Milligan TG. 2006. Single-grain, microfloc and macrofloc volume variations observed with a LISST-100 and a digital floc camera. Journal of Sea Research, 55, 87-102. doi:10.1016/j.seares.2005.09.003

Mikkelsen OA, Hill PS, Milligan TG. 2007. Seasonal and spatial variation of floc size, settling velocity, and density on the inner Adriatic Shelf (Italy). Continental Shelf Research, 27, 417-430 doi:10.1016/j.csr.2006.11.004

Morton JW. 1977. Ecological effect of dredging and dredge spoil disposal: a literature review. Technical papers of the US Fish and Wildlife Service 94, 33 S, Washington, DC.

MSFD. Marine Strategy Framework Directive, Task Group 8, Contaminants and pollution effects.

Murphy S, Voulgaris G. 2006. Identifying the role of tides, rainfall and seasonality in marsh sedimentation using longterm, suspended sediment concentration data. Marine Geology, 227, 31-50. doi:10.1016/j.margeo.2005.10.006

Nechad B, Ruddick KG, Park Y. 2010. Calibration and validation of a generic multisensor algorithm for mapping of Total Suspended Matter in turbid waters. Remote Sensing of Environment, 114, 854-866. doi:10.1016/j.rse.2009.11.022

Nichols MM. 1988. Consequences of dredging. In: Hydrodynamics of Estuaries (Kjerfve B, ed.). CRC Press, Florida, 89–99.

O'Connor TP. 1999. A wider look at the risk of ocean disposal of dredged matter. Marine Pollution Bulletin, 38, 760-761.

OSPAR. 2008. Assessment of the environmental impact of dredging for navigational purposes, OSPAR Commission, Publication nr 366/2008, 17pp.

PIANC. 2008. Minimising harbour siltation, Report No 102, 75pp.

Reed AH, Faas RW, Allison MA, Calliari LJ, Holland KT, O'Reilly SE, Vaughan WC, Alves A. 2009. Characterization of a mud deposit offshore of the Patos Lagoon, southern Brazil. Continental Shelf Research, 29, 597-608. doi:10.1016/j.csr.2009.02.001

Rees HL, Rowlatt SM, Limpenny DS, Rees EIS, Rolfe MS. 1992. Benthic studies at dredged material disposal sites in Liverpool Bay. Aquatic Environment Monitoring Report No 28, MAFF Dirct. Fish Res, 21pp.

Rees HL, Boyd SE, Schratzberger M, Murray LA. 2006. Role of benthic indicators in regulating human activities at sea. Environmental Science & Policy, 9, 496-508.

Ruddick KG, Ovidio F, Rijkeboer M. 2000. Atmospheric correction of SeaWiFS imagery for turbid coastal and inland waters. Applied Optics, 39, 897-912.

Sheremet A, Mehta AJ, Liu B, Stone GW. 2005. Wave-sediment interaction on a muddy inner shelf during Hurricane Claudette. Esturaine, Coastal and Shelf Science, 63, 225-233. doi:10.1016/j.ecss.2004.11.017

Silva-Jacinto R, Le Hir P. 2001. Response of stratified muddy beds to water waves. In: Coastal and Estuarine Fine Sediment Processes (McAnally WH, Mehta AJ, eds.). Proceedings in Marine Science, 3, 95-108.

Simonini R, Ansaloni I, Cavallini F, Graziosi F, Iotti M, Massamba N'Siala G, Mauri M, Montanari G, Preti M, Prevedelli D. 2005. Effects of long-term dumping of harbor-dredged material on macrozoobenthos at four disposal sites along the Emilia-Romagna coast (Northern Adriatic Sea, Italy). Marine Pollution Bulletin, 50, 1595-1605. doi:10.1016/j.marpolbul.2005.06.031

Smith PK, Krohn RI, Hermanson GT, Mallia AK, Gartner FH, Provenzano MD, Fujimoto EK, Goeke NM, Olson BJ, Klenk DC. 1985. Measurement of protein using bicinchoninic acid. Analytical Biochemistry, 150, 76-85.

Smith SDA, Rule MD. 2001. The effects of dredge-spoil dumping on a shallow water soft-sediment community in the Solitary Islands Marine Park, NSW, Australia. Marine Pollution Bulletin, 42, 1040-1048.

Soule DF, K GS. 1988. Marine Organisms as Indicators. Springer

Stagg R, McIntosh A. 1998. Biological effects of contaminants: Determination of CYP1A-dependent mono-oxygenase activity in dab by fluorimetric measurement of EROD activity. ICES Times, 23.

Teurlincx R, Van der Biest K, Reyns J, Verwaest T, Mostaert F. 2009. Haven van Blankenberge - verminderen van de aanzanding van de havengeul en het voorplein: Eindrapport. Versie 1_1. WL Rapporten, 643_12. Waterbouwkundig Laboratorium: Antwerpen, België.

Thain JE, Vethaak AD, Hylland K. 2008 Contaminants in marine ecosystems: developing an integrated indicator framework using biological-effect techniques. ICES Journal of Marine Science, 65, 1508-1514.

Thorne PD, Hanes DM. 2002. A review of acoustic measurement of small-scale sediment processes. Continental Shelf Research, 22, 603-632.

Thorne PD, Vincent CE, Hardcastle PJ, Rehman S, Pearson ND. 1991. Measuring suspended sediment concentrations using acoustic backscatter devices. Marine Geology, 98, 7-16.

Truitt CL. 1988. Dredged material behaviour during open water disposal. Journal of Coastal Research, 4, 389–397.

Van Cauwenberghe C. 1971. Hydrografische analyse van de Vlaamse banken langsheen de Belgisch-Franse kust. Ingenieurstijdingen, 20(4), 141-149 .

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, 599-613.

Van Hoey G, Drent J, Ysebaert T, Herman P. 2007. The Benthic Ecosystem Quality Index (BEQI), intercalibration and assessment of Dutch coastal and transitional waters for the Water Framework Directive. NIOO report 2007-02.

Van Hoey G, Hostens K, Parmentier K, Bekaert K, De Backer A, Derweduwen J, Devriese L, Hillewaert H, Hoffman S, Pecceu E, Vandendriessche S, Wittoeck J. 2009. Biological and chemical effects of the disposal of dredged material in the Belgian Part of the North Sea (period 2007-2008). ILVO report.

van Kessel T, Kranenburg C. 1998. Wave-induced liquefaction and flow of subaqueous mud layers. Coastal Engineering, 34, 109–127.

Van Lancker V, Du Four I, Degraer S, Fettweis M, Francken F, Van den Eynde D, Devolder M, Luyten P, Monbaliu J, Toorman E, Portilla J, Ullmann A, Verwaest T, Janssens J, Vanlede J, Vincx M, Rabaut M, Houziaux J-S, Mallaerts T, Vandenberghe N, Zeelmaekers E, Goffin A.

2009. QUantification of Erosion/Sedimentation patterns to Trace the natural versus anthropogenic sediment dynamics (QUEST4D). Final Report Phase 1. Science for Sustainable Development. Belgian Science Policy, 63pp.

Van Lancker V, Baeye M, Du Four I, Degraer S, Fettweis M, Francken F, Houziaux J-S, Luyten P, Van den Eynde D, Devolder M, De Cauwer K, Monbaliu J, Toorman E, Portilla J, Ullman A, Liste Muñoz M, Fernandez L, Komijani H, Verwaest T, Delgado R, De Schutter J, Janssens J, Levy Y, Vanlede J, Vincx M, Rabaut M, Vandenberghe N, Zeelmaekers E, Goffin A. 2011. QUantification of Erosion/Sedimentation patterns to Trace the natural versus anthropogenic sediment dynamics (QUEST4D). Final Report. Science for Sustainable Development. Belgian Science Policy, 93pp.

Van Maren DS, Winterwerp JC, Sas M, Vanlede J. 2009. The effect of dock length on harbour siltation. Continental Shelf Research, 29, 1410-1425. doi:10.1016/j.csr.2009.03.003

Verfaillie E, Van Meirvenne M, Van Lancker V. 2006. Multivariate geostatistics for the predictive modelling of the surficial sand distribution in shelf seas. Continental Shelf Research, 26, 2454-2468. doi:10.1016/j.csr.2006.07.028

Verlaan PAJ, Spanhoff R. 2000. Massive sedimentation events at the mouth of the Rotterdam waterway. Journal of Coastal Research, 16, 458-469.

Verwaest T, Delgado R, Janssens J, Reyns J. 2011. Longshore sediment transport along the Belgian coast. In: Coastal sediments 2011 "Bringing Together Theory and Practice", May 2-6, Hyatt Regency Miami, Miami, Florida (USA), 1528-1538

Vincent CE, Bass SJ, Rees JJ. 2003. Uncertainties in suspended sediment concentration and transport due to variations in sediment size. Proc Coast Sed, Clearwater FL, May 2002, 10pp.

Voulgaris G, Meyers S. 2004. Temporal variability of hydrodynamics, sediment concentration and sediment settling velocity in a tidal creek. Continental Shelf Research, 24, 1659-1683. doi:10.1016/j.csr.2004.05.006

Vyshal D. 2010. Long term impact of dredge spoil dumping in the Belgian Coastal zone. UGent Marelac thesis

Ware S, Bolam SG, Rees HL. 2010. Impact and recovery associated with the deposition of capital dredging at UK disposal sites: Lessons for future licensing and monitoring. Marine Pollution Bulletin, 60, 79-90. doi:10.1016/j.marpolbul.2009.08.031

Wildish DJ, Thomas MLH. 1985. Effects of dredging and dumping on benthos of Saint-John Harbor, Canada. Marine Environmental Research, 15, 45-57.

Winterwerp J. 1999. On the dynamics of high-concentrated mud suspensions. PhD thesis, TU Delft, The Netherlands, 172pp.

Winterwerp JC, 2005. Reducing harbour siltation I: Methodology. Journal of Waterways, Port, Coastal and Ocean Engineering, 131, 258–266. doi:10.1061/(ASCE)0733-950X(2005)131:6(258)

Woolfe KJ, Fielding CR, Howe JA, Lavelle M, Lally JH. 1998. Laser-derived particle size characterisation of CRP- 1, McMurdo Sound, Antarctica. Terra Antarctica, 5, 383-391.

Wright LD, Friedrichs CT. 2006. Gravity-driven sediment transport on continental shelves: A status report. Continental Shelf Research, 26, 2092-2107. doi: 10.1016/j.csr.2006.07.008

Wu J, Liu JT, Shen H, Zhang S. 2006. Dispersion of disposed dredged slurry in the meso-tidal Changjiang (Yangtze River) Estuary. Estuarine, Coastal and Shelf Science, 70, 663-672. doi:10.1016/j.ecss.2006.07.013

Abbreviations and definitions

ADP	Acoustic Doppler Profiler, measured current velocity and turbulence in one point.
ADV	Acoustic Doppler Velocimeter, measured current velocity in a vertical profile.
BEQI	Benthic Ecosystem Quality Index, www.beqi.eu
BPNS	Belgian Part of the North Sea
BCS	Belgian Continental Shelf
d.w.	dry weight
EAC	Environmental Assessment Criteria
Fluid mud	Fluid mud is a suspension of cohesive sediments at a concentration beyond the gelling point (10 to 100 g/l). The suspension behaves non-Newtonian and its dynamics are fairly independent of the flow in the water column (Winterwerp 1999).
HCMS	High Concentrated Mud Suspensions is a suspension of cohesive sediments of a few 100 mg/l up to a few g/l. The suspension behaves Newtonian and is interacting with the turbulent flow field (Winterwerp 1999).
Ind	individuals
LISST	Laser In-Situ Scattering and Transmissometer, measured particle size distribution and volume concentration
mab	meter above bed
MFSD	Marine Strategy Framework Directive
MLLWS	Mean Lowest Low Water at Spring tide
MSL	Mean Sea Level
nmMDS	non-metric Multi-Dimensional Scaling
OBS	Optical Backscatter Sensor, measures turbidity
PSD	Particles Size Distribution
SPM	Suspended Particulate Matter
TDM	Ton Dry Matter
WFD	Water Framework Directive
w.w.	wet weight

Annex: Dredging and dumping intensity maps