The Benthic Ecosystem Quality Index (BEQI), intercalibration and assessment of Dutch coastal and transitional waters for the Water Framework Directive

- FINAL REPORT -

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LEGEND

1. Summary

Introduction

The Water Framework Directive (WFD) of the European Union (December 2000) aims at the protection of all water types (rivers, lakes, coastal, and transitional waters) and defines a ‘good ecological status’ as the objective to be reached for all European waters by 2015. This ‘good ecological status’ corresponds with a more or less undisturbed state. The ecological quality status is assessed with ecological evaluation tools that are based on the integration of well-defined biological quality criteria. Each of these quality criteria has to support a classification (bad to high) aiming at measuring the 'health' of the system against that described for reference (high level) conditions. In the case of heavily modified water bodies, the WFD uses a Maximal Ecological Potential (MEP) as reference, and a Good Ecological Potential (GEP) as objective (similar to a good ecological status).

WFD compliant bio-indicators and classification tools detecting impacts on structure (biodiversity) and functioning of aquatic ecosystems (i.e. fish, aquatic plants and algae, and benthic invertebrates) are needed for the ecological status assessment of surface waters. This report deals with the biological quality element ‘benthic invertebrates’ in coastal, transitional waters and saline water lakes.

The Benthic Ecosystem Quality Index (BEQI), has been developed by the Netherlands Institute of Ecology on behalf of the Rijkswaterstaat/Institute for Coastal and Marine Management. It is based on an ecosystem functioning approach (Ysebaert & Herman, 2004), which aims to give an indication about ecosystem structure and functioning, and about biological relationships. BEQI evaluates at the scale of a whole water body, contrary to methods applied by other member states that evaluate the ecological status per sampling station.

The Dutch monitoring system of water bodies (MWTL) and ecological monitoring (since the nineties) also aims to assess the status of a water body as a whole. The initial developed BEQI method (2004) was compared with other Member States WFD Benthos methods under development and valuated as the most appropriate to develop further and to apply to all Dutch coastal and transitional water bodies the BEQI method for WFD benthos classification.

The BEQI is a multi metric method distinguishing three scale levels to asses overall ecosystem functioning.

The first level is the ecosystem level (the whole water body). Central at this level is the role of the macrobenthos in ecosystem functioning. The macrobenthic community is an important link in the food web between primary production and predatory invertebrate, fish and bird populations. As such it exerts top down control on the primary producers and bottom up control on the higher trophic levels while the reverse, control by primary producers and predatory consumers on the macrobenthic community act just as well. The dynamics of this interplay is fundamental to ecosystem functioning.

The second level is the habitat level. The macrobenthic communities in the ecosystem depend on the physical and chemical characteristics of the habitat. In transitional waters, strong gradients of physical energy due to waves and tides, as well as strong chemical gradients in salt and other constituents shape the habitat structure of the system. Thus, a diversity of habitats is typically found, and this diversity, characterized by the occurrence and the relative surface area of different habitat types, is a major factor determining occurrence, density and biomass of macrobenthos. Additional biogenic structures like the eco-elements mussel beds, oyster reefs and seagrass beds modify conditions at a local and possibly larger scale, and are also included in the second assessment level.

The third level concerns the within habitat macrobenthic community composition. Total biomass, total density, number of species and species composition are community characteristics that respond to different types of stress. The type of response may serve as a first indicator of the stressor type. Detection of changes in within-habitat communities may also help understanding responses at the ecosystem level.

The BEQI multimetric integrates the information of the three levels and primarily aims at providing a signal that is capable of showing significant **deviations** from a defined reference state. The aim of this report is to further develop the method proposed by Ysebaert and Herman (2004), apply it to the Dutch coastal and transitional waters and saline lakes, to compare this method with the assessment methods proposed by other member states (intercalibration), and to formulate advices to improve (adaptation of the evaluation method, new insights, new technologies) the WFD monitoring for the Dutch waters.

The BEQI method

BEQI references definition

According to the WFD guidelines, a water body should be evaluated relative to a reference condition. Ideally, such a reference state should be either representing the actual state in historic times where influence was minimal, or a site being located outside the influence of human activities. In Dutch coastal and transitional waters, there are no true un-impacted areas and historical data from ‘pristine’ conditions do not exist. Strong human interference, dating back several millennia, has reshaped the morphology and hydrography of these transitional waters. Thus, ‘pristine’ conditions cannot even be defined theoretically. Therefore, the definition of reference conditions for the Dutch water bodies is based on data from areas and/or time periods where human pressures are at a minimum and which reflect the spatial and temporal variability of the benthos. However, in practice human pressures are not well quantified, observed effects are not unambiguously attributable to a certain pressure and human pressures may remain unnoticed while having an effect. This greatly complicates the definition of an unaffected reference condition. Moreover, data shortage severely limits the choice of reference conditions.

Because of this complexity, a pragmatic choice in the reference setting was made. BEQI reference settings are based on the oldest third of the available data. This means that BEQI reference settings should **not** be interpreted as absolute ecological targets. A BEQI quality status is a measure of divergence from the defined reference conditions. A good status means that there is a good agreement between the reference and the assessment conditions, a poor status means that the assessment and reference conditions are differing to such a degree that it is very unlikely that this has arisen by chance alone. Such deviation should lead to further investigations of the cause of the deviation. If it can be shown that the deviation is not amenable to management, or that the present status represents an improvement compared to the reference (because, e.g. it responds to effective abatement of pollution or eutrophication), BEQI reference settings can and should be adapted to incorporate these new insights

Also further transnational research to improve the identification of the reference could deliver new insights. This need to reconsider references is also acknowledged in the draft intercalibration decision document of the Commission.

Applicability of BEQI metrics, class boundaries and definition of risk of miss classification

The normative definitions of the WFD (Annex V. (Table 1.2)) provide the basis for classifying the ecological status or potential for surface water bodies. The BEQI classification system conforms to these definitions while taking into account the constraints in the reference data sets.

The BEQI level 1 ecosystem functioning evaluation is based on the relation between macrobenthic biomass and system primary production (the sum of pelagic production by phytoplankton and benthic production by microphytobenthos). An earlier comparison between estuaries revealed a linear relationship where macrobenthic biomass is about 10% of the system yearly primary production (Herman et al. 1999). BEQI level 1 was applied to all Dutch water bodies. For most water bodies there is no good or recent estimate of the system primary production (including both pelagic and benthic production) and benthic biomass (e.g. the lack of estimates of the total biomass of mussel or oyster beds). Therefore, the evaluation at this level is for some water bodies based on expert judgement.

At BEQI level 2, habitat and eco-element (e.g. mussel- and oyster beds) evaluation, surface areas are in principle compared with a reference situation. For the Wadden Sea and for the heavily modified transitional waters (e.g. Westerschelde and Eems-Dollard), the identification of reference levels of habitats requires an interactive science policy process. This study only takes into account available historic information and for some water bodies classification at this level is based on expert judgement. Changes in habitats are caused by land reclamation, dredging, hydrodynamic problems and the occurrence of invasive species *Crassostrea gigas*.

More historical information on the surface area of habitats and on the spatial coverage of the eco-elements is needed to improve the metric at level 2.

At level 3, the changes in the benthos compared to the reference situation are evaluated based on four parameters: density, biomass, species richness and species composition. This set not only covers the WFD requirements (composition and abundance of benthic fauna) but biomass is an additional parameter, which is important in the implementation of the Birds and Habitats Directives reflecting also possible pressures related to fisheries. The assessment was done per habitat within a water body. The determination of the reference situation (with a good spatial and temporal coverage) is mostly based on the selection of the first 1/3 of the available data period for a water body. The reference conditions for each parameter are described by a probability distribution, which is obtained by permutation over 2000 samples. These samples with a specific sample size (in BEQI presented as sample surface) are drawn at random and with replacement from the reference data set. The distribution of the 2000 sample values is described by the median and percentiles. The percentiles define the reference value boundaries, which are linked to fixed ecological status class boundaries (see Table). The probability distribution is described for a range of sampling efforts to match with different assessment efforts (sampling surfaces). The probability that the combined index based on the four parameters leads to misclassification is, similarly, well below 5 % but the exact probability level is unknown.

Ecological status class boundary	**Reference value boundaries**	
	Number of species Species composition changes	Density Biomass
High/good: 0.8	median	25^{th} and 75^{th} percentile
Good/moderate: 0.6	5^{th} percentile	2.5^{th} and 97.5^{th} percentile
Moderate/poor: 0.4	2/3 of Good/mod value	2/3 and 4/3 of Good/mod value
Poor/Bad: 0.2	1/3 of Good/mod value	1/3 and 5/3 of Good/mod value

This procedure allows estimating, for any given sampling surface, the reference value that can be expected to reach a certain ecological status class. Because the variability in the parameter values will decrease with increasing sampling surface (finally it will reach an asymptotic value), it is required to determine a degree of assessment precision. Therefore, for each habitat and water body, a minimal, optimal and maximal required sampling surface is determined to define the assessment precision. These sampling surfaces can be used to determine the required number of samples per habitat and water body for the monitoring.

Application of the BEQI to the Dutch coastal and transitional waters and saline lakes

The assessment results and conclusions are summarized separately for the different Dutch water bodies. The BEQI method detects and evaluates changes compared to the selected reference period and is expressed as Ecological Quality Ratio or EQR.

The different parameters per level are summarized in one score by averaging and, additionally, the three levels are -after weighing- combined in one score. The scores and ecological status classifications for each parameter and level are visualized separately, which make it possible to immediately detect the parameter or level where the changes occurred. This makes the BEQI method transparent. In this summary, only the overall EQR scores are mentioned and more details can be found in the text.

Coastal waters: open polyhaline and euhaline

All North Sea coastal water bodies (Zeeuwse kust, Noordelijke Delta kust, Hollandse kust, Waddenkust en Eems-Dollard kust are characterized as natural water.. The main pressure for the coast is the fisheries activity. The selected reference period for the coast is 1978-1990 (Delta area) and 1988-1990 (Hollandse kust, Waddenkust and Eems-Dollard kust). The assessment is only done for the benthic Q1 community (fine muddy sand habitat) and the Hollandse kust and Waddenkust. This Q1 is the dominant community, other communities and habitats had insufficient data sets. Consequently, it is advisable to increase the sampling effort for the coastal area in the future. The Hollandse kust and Waddenkust are respectively evaluated as moderate and poor on level 3, and for level 1, both are evaluated as good. The observed changes are mainly caused by the dominance of the invasive species *Ensis directus*.

The assessment data set from the Dutch Monitoring network of the coastal water bodies Zeeuwse kust, Noordelijke Delta kust en Eems-Dollard kust was insufficient for for WFD classification.

Coastal waters: sheltered polyhaline

The Oosterschelde, a heavily modified water body, is evaluated as GEP (Good Ecological Potential) at level 1. Level 2 is evaluated as moderate due to the disappearance of the intertidal and natural mussel banks, and due to the increase in the occurrence of the invasive species, the Japanese oyster *Crassostrea gigas*. At level 3, the Oosterschelde is not drastically changed (GEP status) compared to the reference period (1990-1994).

The Wadden Sea is characterized as natural water body. Level 1 of the Wadden Sea is evaluated as good, whereas level 2 is evaluated as moderate (decline in the surface area of mussel beds). At level 3, mainly biomass changes and also shifts in species composition within the habitats are observed due to the occurrence of invasive species. However, the overall ecological status is still good. The reference period contains the years preceding 1983.

Transitional waters

The main pressures for the benthos in the Westerschelde, a heavily modified water body, are habitat loss and dredging, which are clearly detected on level 2 (Ecological status moderate). Level 1 is evaluated as GEP. At level 3, no drastic changes are observed between the reference period (1979-1996) and the assessment in the benthic parameters (ecological status GEP).

For the Eems-Dollard, also a heavily modified water body, the evaluation is based on data from only one habitat (current Eems-Dollard monitoring concentrated on the Heringsplaat), due to the absence of monitoring data with a wide spatial coverage. This habitat is evaluated as GEP at level 3. At level 2, real changes in the surface areas of the habitats are observed and therefore the ecological status is evaluated as moderate. No evaluation is made for level 1 due to the absence of good estimates for system primary production and benthic biomass.

Saline lakes

The main benthos pressures for the saline lakes, Lake Veere and Grevelingen, both heavily modified water bodies, are stratification and oxygen depletion. Due to this oxygen depletion, the overall status of Lake Veere is moderate (level 1 is GEP, but level 3 is moderate). This problem is already recognized by the management authorities and actions to improve the water exchanges have been started (Katse Heule). In Grevelingen, no drastic changes are reported, and consequently the evaluation at level 3 shows that the present situation is more or less similar to the reference period. However, some parameters show a moderate status, which indicates changes (mainly in the species composition).

Monitoring advices

The BEQI method requires a spatially stratified approach, with habitats within a water body as strata. A representative sample of the habitat has to be collected at randomly assigned locations. Because of the inherent temporal variability of macrobenthic communities caused by year-to-year recruitment variation, a yearly surveillance monitoring during the entire assessment period is required. This reduces the risk of misclassification due to natural temporal variation and increases the power to detect temporal trends. Sampling effort should be adjusted according to the required detectable effect size and equal to the minimal required surface after one year and equal to the maximal required surface after 3 or 6 years, to get an acceptable assessment. Small detectable effect sizes and heterogeneous habitats will require a larger effort than large effects sizes and homogeneous habitats. At the biological sampling stations important environmental variables like sediment composition and depth need to be measured as well. For WFD classification not all habitats under the same pressures need to be monitored. It is cost effective to restrict monitoring to those habitat/community types being the most sensitive for pressures.

Intercalibration

The intercalibration process under the WFD Common Implementation Strategy program is aimed at consistency and comparability of the classification results of the WFD assessment methods operated by each Member State for the biological quality elements. The intercalibration exercise must establish values for the boundary between high and good status and for the boundary between good and moderate status which are consistent with the normative definitions of those class boundaries according Annex V of the WFD.

The intercalibration (comparison) of the BEQI method results with four other international methods available (the Danish DKI method, the UK IQI method, the SP/PT m-AMBI multi-metric and the Norwegian NQI index) is done at the community (within–habitat) level for selected common Intercalibration types of coastal water NEA 1/26 and NEA 3/4,.

Level three of BEQI results are compared with the averages of station assessments per habitat from the international methods.

The Intercalibration results in this report are described following the format of the Intercalibration Technical Document as supplied to the Commission through the ECOSTAT Working group. A short description of the method, the reference conditions, the class boundaries and the comparison results are given.

First conclusions are that within water bodies BEQI and the other methods show little agreement (low correlation). The sub-metrics correlate best with the m-AMBI. In a multiple water body comparison between m-AMBI and level 3 of BEQI there is agreement in the general trend but BEQI is more sensitive (evaluates over a larger ranges of states) than m-AMBI.

Overall conclusions and recommendations

1. The BEQI method is useful for making a further integration with other biological or even physical quality elements (on level 1 and 2). Level 3 of the BEQI method is capable to detect and evaluate changes in the benthic community in the Dutch coastal and transitional waters compared to a certain period, and can consequently fulfill the requirements of the WFD. Level 2 has been updated and developed to make it useful for different water bodies; it is equally capable to detect changes. For level 1 a good or recent estimate of the system primary production (including both pelagic and benthic production) and benthic biomass (e.g. the lack of estimates of the total biomass of mussel or oyster beds) is needed to evaluated the ecosystem functioning.
There is, however, a need for further investigations to define the `reference state` of the water bodies, as well as to finalize the assessment for some habitats or water bodies (due to the lack of adequate recent data).

2. The intercalibration exercise has clearly identified the need for further work to compare assessment methods at the level of the water body. Then the BEQI method can becompared comprehensively with other Member States methods, that means including a spatial aggregation at water body level.
At present no common international expert view exists on the valuation of invasive species.

3. The BEQI method requires for some coastal and transitional water bodies innovations in monitoring frequency and sites (habitats representation) relating sampling surface to frequency and spatial variability. In order to make these affordable investigations in new monitoring techniques are suggested. The monitoring recommendations will be taken into consideration for the 2008 Benthos monitoring planning in order to get a sound status assessment for the WFD Management plans of 2009.

2. Introduction

The Water Framework Directive (WFD) from the European Union (December 2000) aims at the protection of all water types (rivers, lakes, coastal, and transitional waters) and defines a ‘good ecological status’ as the objective to be reached for all European waters by 2015. This ‘good ecological status’ corresponds with a more or less undisturbed status defined as the reference for the water system to be evaluated. It is the responsibility of the member states to define the ‘good ecological status’ for each of their water types/systems. This is achieved with the development of systems for ecological evaluation that are based on the integration of well-defined biological quality criteria. Each of these quality criteria has to support a classification (bad to high) aiming at measuring the 'health' of the system against that described for reference (high level) conditions.
WFD compliant bio-indicators and classification tools detecting impacts on structure (biodiversity) and functioning of aquatic ecosystems (i.e. fish, aquatic plants and algae, and benthic invertebrates), are needed for the ecological status assessment of surface waters. This report deals with the biological quality element ‘benthic invertebrates’ in coastal and transitional waters.

With the introduction of the legislative initiatives of the Water Framework Directive, the role played by ecological indicators has fundamentally changed. Whereas these indicators are previously used as descriptors of the system state for scientific and public communication purposes, they have now obtained legal value. When the ecological status of certain systems is judged insufficient (based on the use of the accepted indicator), action must be taken and large costs and efforts may be involved. It is thus of utmost importance that the indicators used in the European context are thoroughly tested, well justified from a scientific point of view, and generally accepted throughout the EU.

The implementation of the WFD has provoked a large debate on the use of benthic bio-indicators and indices to determine the quality of the estuarine (transitional) and coastal waters in Europe (see e.g. Borja et al., 2007; Dauvin, 2007b). Although a wide variety of benthic bio-indicators already existed (review in Diaz et al., 2004), several member states have developed their own nationally assessment methods. Member states such as United Kingdom, Spain, Portugal, and Denmark have developed methodologies that are very similar to each other, using a multimetric that basically combines the relative abundance of sensitive species in a sample (e.g. AMBI, ISI) with a diversity component (Borja et al., 2007). An intercalibration of these Member States' classification systems was performed on subtidal samples from coastal water type NEA1 (Borja et al., 2007). This intercalibration, as required by the WFD, aims at a harmonization of the ecological quality targets (i.e. a common definition of the 'good' ecological quality status for protection and restoration of all European surface waters). Not surprisingly, the intercalibration exercise of Borja et al. (2007) showed good agreement between the different classification systems, due to the similar indices applied. At the time of this intercalibration exercise, the assessment method of the Netherlands was still under development. This method differs from the other methods as it does not evaluate the ecological status sampling station by sampling station, but rather uses a set of parameters that take into account the different scales of variability in coastal and transitional waters and evaluates the water body (ecosystem) as a whole.

The method (BEQI: Benthic Ecosystem Quality Index) which is further elaborated in this report aims at a next step towards a sound ecological assessment of coastal and transitional waters for the WFD (for a first outline on the method see Ysebaert & Herman, 2004). The method adopts the concept of ecosystem approach (see Box 1, also www.biodiv.org for more details) and aims at putting these conceptual ideas to the test in developing a new generation of indicators that summarize the ecosystem dynamics in a more synoptic view. The method aims at an indication about ecosystem structure and functioning, and biological relationships. Although this report primarily deals with the assessment of benthic macro-invertebrates, the method aims at an integration of the different biological quality elements.

This method uses a set of parameters that take into account the different scales of variability in coastal and transitional waters and aims at evaluating the water body (ecosystem) as a whole. Briefly, on the level of the whole ecosystem (e.g. a water body) (ecosystem level) one can evaluate if the benthic macrofauna fulfils the functional role one might expect given the current ecological circumstances. At this level also integration with the evaluation methodologies of other quality elements is most appropriate, and information on the water body can be summarised. On the subsequent level (habitat level) the distribution of habitats (habitat completeness and complexity) can be evaluated. Finally the biological quality of each distinguished habitat based on benthic macrofauna can be evaluated (community [within-habitat] level), with parameters that are sensitive to different types of stress and that can explain possible deviations. The overall metric primarily aims at providing a signal that is capable of showing significant **changes/deviations** from a certain reference state. The whole approach is elaborated in more detail further in this report.

> Box 1 – Ecosystem approach
>
> *(from the Convention on Biological Diversity – www.biodiv.org)*
>
> The Ecosystem Approach is a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way. The Ecosystem Approach is the comprehensive integrated management of human activities, based on best available scientific knowledge about the ecosystem and its dynamics, in order to identify and take action on influences which are critical to the health of the marine ecosystems, thereby achieving sustainable use of ecosystem goods and services and maintenance of ecosystem integrity.
> An ecosystem approach is based on the application of appropriate scientific methodologies focused on levels of biological organization, which encompass the essential structure, processes, functions and interactions among organisms and their environment. The Ecosystem Approach should be undertaken at the appropriate spatial and temporal scales. Recognizing the varying temporal scales and lag-effects that characterize ecosystem processes, objectives for ecosystem management should be set for the long-term. Management must recognize that change is inevitable.

As a consequence, the intercalibration of the Dutch assessment method with the assessment methods of the other member states (e.g. Borja et al., 2007) can only be done at the lowest level and per habitat within a water body, i.e. the evaluation of the biological quality of a habitat based on macrobenthic indicators. This further requires that sampling stations are from a single, well-defined and ecologically relevant habitat.

The aim of this report is to further develop the method proposed by Ysebaert and Herman (2004) in order to make it applicable to the Dutch coastal and transitional waters and saline lakes and to compare this method with the assessment methods proposed by other member states (intercalibration) and formulate advices to improve (adaptation to the evaluation method, new insights, new technologies) the monitoring for the Dutch waters for the WFD.

The outline of the report is as follows:

Chapter 2: Description of the Benthic Ecosystem Quality Index:

In this chapter the scientific background of the method and its relation to human pressures is described, the further details and computation of the different sub-indicators and the overall ecological quality assessment for the biological quality element benthic macrofauna.
Some remarks and thoughts about the precision, power and risk of misclassification using the BEQI and problems encountered setting reference conditions are discussed.

Chapter 3: Application of the BEQI to the Dutch coastal and transitional waters and saline lakes

In this chapter the BEQI method is applied and tested on the different Dutch coastal and transitional waters and saline lakes. A short description is given for each water body including main human pressures present in the water body and, if available, a habitat typology. Reference conditions are set for the different water bodies and the ecological quality of the water body is evaluated using a recent assessment data set. To highlight specific problems associated with a certain water body extra information is presented (e.g. time series of EQR values).
A full application of the BEQI method is often not feasible because of lack of data and general recommendations are being made to improve this.

Chapter 4: Consequences for monitoring

Based on the BEQI approach a general framework for monitoring benthic macrofauna for the WFD is proposed and practical recommendations are made for the different water bodies, based on the current monitoring efforts (e.g. MWTL monitoring) for each of the water bodies.

Chapter 5: Intercalibration

Prior to the implementation of WFD assessment, any proposed methodology must be intercalibrated between the member states within an eco-region. Each MS divides the EQR scale for their monitoring system into five ecological status classes (high, good, moderate, poor, bad) by assigning a numerical value to each of the class boundaries. The value for the 'high/good' and the 'good/moderate' class boundaries should be established through an intercalibration exercise. This is to ensure that the established class boundaries are consistent with the normative definitions of the WFD and are comparable between the different member states.

For coastal water – type NEA 1/26 – the quality element benthic invertebrates are intercalibrated by five Member States (Denmark, Spain, Norway, Portugal and United Kingdom) during a first phase of the intercalibration (Borja et al., 2007). To fulfill the WFD requirements, the lowest level of the BEQI method is compared/intercalibrated with the assessment methods developed by the other Member States. Due to the fact that the BEQI-method directly evaluated the entire water body, whereas the others on sample level, the comparison is done with the average of the EQR scores of the samples within a habitat of a water body. For each Dutch coastal water body (type NEA1), data of the period 2002-2004 are used to assess its ecological quality using the different assessment methods. The derived ecological quality ratios for each assessment method are presented and discussed. The results will be incorporated into the Intercalibration Technical Report and presented to ECOSTAT. Belgium also adopted the BEQI method at the third level for the Belgian coastal waters and will report this to ECOSTAT. At this moment in Germany it is decided to use the m-AMBI for coastal NEA1 waters and for the Wadden sea (NEA3/4). The BEQI method is applied (third level only) to German data. For transitional waters it was decided by the NEA-GIG Benthic Expert Group to postpone the intercalibration to the next phase.

Chapter 6: Plan for further research

Although that clear improvement has been made with respect to the ecological assessment of coastal and transitional waters using a classification system that is based on an ecosystem approach, there is still a big need in further innovating the use of indicators in coastal and transitional waters. Challenges are:

- Develop indicators that cover the full range of known reactions of coastal ecosystems to the different relevant anthropogenic pressures
- embedding the indicators in an ecosystem view, and dynamically link a number of ecosystem variables into a meaningful indicator system
- deploy new monitoring techniques combined with operational modeling as the basis of the indicator system

This is only feasible in a European context and a first outline for a research proposal is presented in this report.

3. Method description

3.1 BEQI (Benthic Ecosystem Quality Index)

Taking into consideration the large intrinsic variability of estuarine and coastal systems and the importance of ecosystem functioning within a water body, Ysebaert & Herman (2004) advocate a multilevel scale-dependent approach for the classification of the quality elements in coastal and transitional waters (Table 1). The method adopts the ecosystem approach which encompasses the essential structure, processes, functions and interactions among organisms and their environment. Basically the method aims at integrating two aspects that are neglected by many other approaches: the functional role of benthos (e.g. highly productive benthic populations may be very important for the conservation of bird populations) and, secondly, the vulnerability to physical changes in the environment (e.g. dredging, land reclamation, harbor construction etc.).

A habitat specific approach is furthermore strongly recommended by Prior et al. (2004) in their guidance document for the application of the WFD to marine benthic communities, because coastal and especially transitional waters are characterized by highly variable physicochemical and hydro-morphologic conditions, resulting typically in a mosaic of different habitats. These habitats differ in (community) structure and function, and as such will show wide variations in statistics or measures between habitats. As such, the water body can not be evaluated as one identity, but the evaluation has to focus on the different habitats for the ecological assessment of the benthic communities.

Table 1. Multilevel scheme illustrating the different levels of the classification system and the associated indicators and links at each level as proposed by Ysebaert & Herman (2004)

Level	Evaluation for macrobenthos	Used to assess	Links to
Whole water body	Functional: biomass, feeding types, …	System integrity, functions performed in land-ocean interaction, functions for carbon and nutrient dynamics, production for higher trophic levels	Other quality elements (chemical, phytoplankton, …): aims at integrating view. Provide constraints for functions related to nature conservation, relevant to Bird and Habitat Directives
Habitat	Spatial organization: surface area, connectivity	System completeness in terms of habitats and community development. Possible developments under appropriate management. Morphodynamic equilibrium and impact of physical stressors	Morphodynamic information Evaluation of habitats and their persistence /conservation (Habitat directive)
Community (within- habitat)	Community structure, based on density, biomass, species number and species composition changes	Completeness and full development of the biological communities within habitats. Occurrence of stress symptoms, comparing species indicator value to expectations valid for the specific habitat	Local stressors Biogeochemical stressors Effects of invasive species

Therefore the multilevel approach proposed by Ysebaert & Herman (2004) consists of three levels (Figure 1):

(1) On the level of the whole ecosystem (level 1; e.g. a water body) one can evaluate if the benthic macrofauna fulfils the functional role one might expect given the current ecological circumstances. At this level also integration with other quality measures is most appropriate, and information on the water body can be summarised.

(2) On the subsequent level (level 2) the distribution of habitats (habitat completeness and complexity) can be evaluated. The size, shape, and spatial relationships of these habitats influence the dynamics of populations, communities, and ecosystems.

(3) Finally (level 3) the biological quality of each distinguished habitat based on benthic macrofauna can be evaluated (community (within-habitat) level), with indicators that are sensitive to different types of stress and that can explain possible deviations. The overall metric primarily aims at providing a signal that is capable of showing significant changes/deviations from a certain reference state. The different levels will be explained further on in detail.

This approach is abbreviated as BEQI (Benthic Ecosystem Quality Index). The BEQI-method uses easily interpretable parameters that are evaluated separately. This allows a transparent assessment method and for any deviation of the overall ecological quality ratio from the reference condition the underlying responsible parameters can be easily traced back and evaluated individually. Therefore, the BEQI-method primarily aims at detecting and evaluating possible ***changes*** in the assessment data compared to the determined reference condition for each parameter, within a habitat of a water body. Based on this the managing authorities can decide to install an operational or investigative monitoring program or to take measures.

The method has already been elaborated in a case study for the Westerschelde (Escaravage et al., 2004), but has been subject to further developments within this project. Details about the method are outlined below.

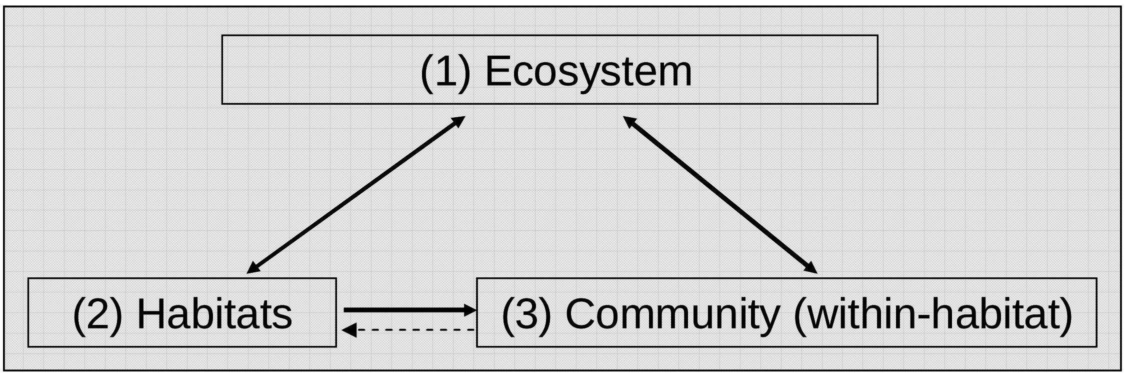


Figure 1. The multilevel approach of Ysebaert & Herman (2004)

3.1.1 Metric at ecosystem level

Principle:

For benthic macro-invertebrates, simple parameters can be used at this level, such as the mean total biomass (adjusted for primary production, which in turn is a function of the amount of light and nutrients). Parsons et al. (1977) showed a dependence of system-averaged benthic biomass on the magnitude of the spring phytoplankton bloom. This relation strongly suggests dependence between benthic biomass and system productivity. Herman et al. (1999) compiled data on benthic biomass and system productivity from the own databases on the Dutch delta area and from published estimates (Figure 2). The relationship between the system-averaged macrofauna biomass (B, g AFDW/m^2) and the system primary production (g C/m^2 year) becomes [$B = 2.85 + 0.08\ P$ ($R^2 = 0.68$)], what remains close to the [B:P = 1/10] line (Figure 2) that could be reasonably kept as the standard ratio between the system primary production and the macrobenthic biomass.

<u>Box 2 – Level 1: Scale of the water body – functional</u>

Aim: evaluate functional role of the macrobenthos in the water system

At the large scale the overall functioning of the water body (i.e. its functions in carbon and nutrient pathways) is evaluated. For estuaries in particular, emphasis is placed on whether the systems perform their filter and buffer function between land and ocean.

At this level, evaluation should preferably not be based on each biological quality element separately, but should be based on an integrative evaluation of the different quality elements. At the scale of the ecosystem, ecological relations between primary producers, macrobenthos, birds and fish enforce correlations between the quality measures based on these groups (either within WFD, or other legislation). Consistency between these measures is not guaranteed. It can be anticipated that, at least in some systems, eutrophication abatement may lead to a decrease of carrying capacity for shellfish and birds. Therefore, there is an urgent need for consistency check at the level of the system.

For benthic macroinvertebrates, this can be summarised by relatively simple indicators, such as the mean total biomass (adjusted for primary production, which in turn is a function of the amount of light and nutrients), the ratio benthic : pelagic grazing on the phytoplankton. These indicators are sensitive for factors that are limiting the growth of the macrobenthos, like eutrophication and dystrophy, shortage of ecotopes/habitats or poisoning.

This ratio may represent a state of equilibrium where the sum of pelagic and benthic production is adequately matched by the biomass of grazers that are present in the system (i.e. macrobenthos and zooplankton). Deviations from this relation could point at unbalanced ecosystem functioning. Such unbalance is illustrated with two examples in the literature (Escaravage et al., 2004). The first is from the upper estuary of the San Francisco Bay, where the invasion by the Asiatic clam (*Potamocorbula amurensis*) after 1987 resulted in a clear top-down effect through the grazing of the phytoplankton by this suspension feeder (Nichols et al., 1990, Alpine & Cloern, 1992). Secondly, the Seine estuary illustrates the alterations to estuaries due to human activities: heavy releases of pollutants of various origins and significant morphological changes (dredging) beginning in the middle of the 19^{the} century (Rybarczyk & Elkaim, 2003). These data are plotted on graph in Figure 2 but left out of the regression estimate.

The escape of primary producers from grazing control was also studied in mesocosm experiments (one month summer incubation) where mussel was experimentally manipulated (Prins et al., 1995). Therein could be concluded that the pattern in nutrient concentration points out that phytoplankton was either heavily controlled by the highest mussel biomass or escaped grazing by the lower biomass. In the intermediate treatment, phytoplankton growth was in equilibrium with the grazing pressure. The B_{graz}:P_{prim} ratio corresponding to this last situation was equal to 1/9 and thus rather close to the relation shown by the systems in Figure 2. Conversely, the situations with excess and lack of grazers gave B_{graz}:P_{prim} ratios of 1/2 and 1/83 respectively. The lines supporting these ratios that are plotted on the graph in Figure 2 are close to the data from San Francisco Bay and from the Seine. The correspondence between the field and the mesocosm data points at the fundaments ruling the interaction between primary producers and their grazers.

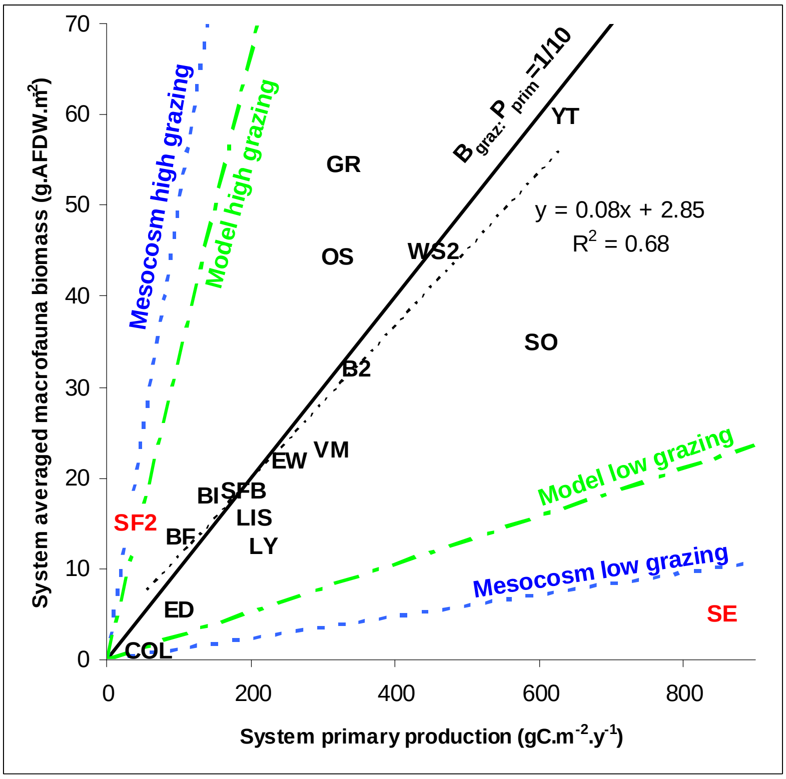


Figure 2. Relation between system-averaged macrobenthic biomass and primary production of shallow well-mixed estuarine systems adapted from Herman et al. (1999). The regression line is a predictive linear least-squares line. Data are indicated by the abbreviation of the name of the system: YT-.Ythan estuary, GR-,Grevelingen, OS-.Oosterschelde, BI-. Balgzand (70s), B2-. Balgzand (80`s), BF-.Bay of Fundy, EW-. Eems outer side, ED-. Eems inner side, VM-. Lake Veere, SFB-. San Fransisco Bay, SF2-.San Francisco Bay after invasion by *Potomocorbula*, LY-. Lynher estuary, WS2-. Wadden Sea, COL-. Columbia river, LIS-. Long Island Sound, CB-. Chesapeake Bay, SO-. Somme estuary, SE-. Seine estuary, the labeled lines materialize the B_{graz}:P_{prim} ratio value of 1/10 and the outputs of mesocosm and model experiments (see text for details)

A condition under which phytoplankton production was either controlled by or escaped from benthic grazers was simulated in numerical models by Herman & Scholten (1990). The ratios found with the model are close to those found in the previous mesocosm experiments in similar situations with respect to the producer/grazer relationships, as illustrate with the proximity of the corresponding lines (Figure 2).

The linearity of the relationships between the primary production and the macrofauna biomass points at the high capacity by these systems to absorb the effects of nutrient enrichments without noticeable alteration of the ecosystem functioning. As a results, shifts of the B_{graz}:P_{prim} equilibrium along the 1/10 line should not be considered as indications of unbalance but rather as a translation towards a new equilibrium. Conversely, systems that are shifting away from the line could become unbalanced. Therefore, this relation can be used to give a robust estimate of possible shifts in the ecosystem functioning of the water body. The B_{graz}:P_{prim} relation will be used as parameter on the ecosystem level and boundaries can be set in accordance to the WFD requirements.

Boundary settings:

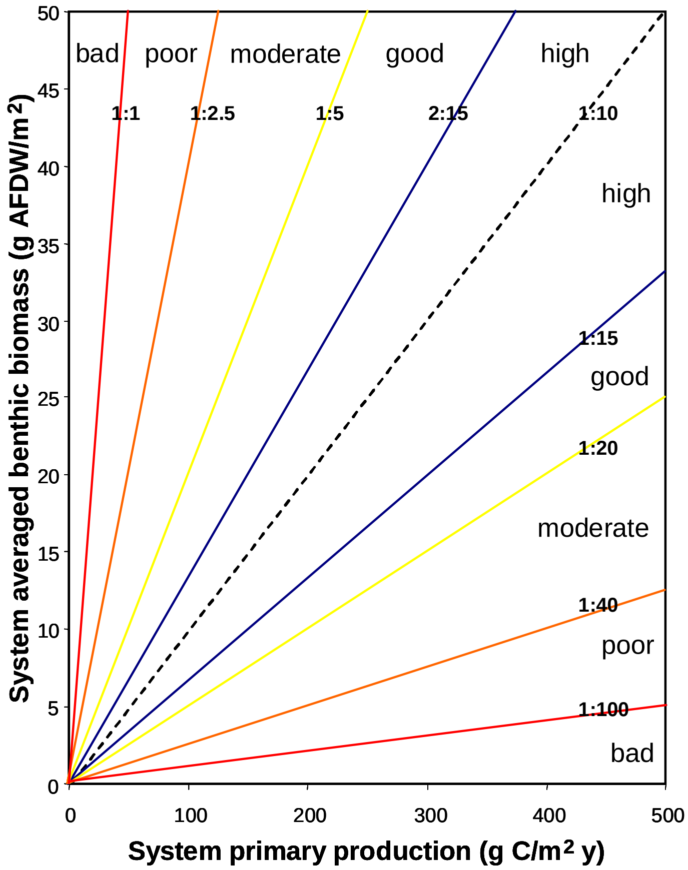


Figure 3. Scaling of the B_{graz}:P_{prim} relation to be used as classification for the ecosystem level

From the field observations, model and experimental results, extreme B_{graz}:P_{prim} ratios could be considered as 1/1 and 1/100 and should be the bad/poor boundary (Figure 3). The model outputs by Herman & Scholten (1990) supply two limits (about 1/2.5 and 1/40) for the areas of poor functioning where overgrazing and phytoplankton escape leads to states of critical unbalance and these limits are selected as the poor/moderate boundaries. These are also the areas where the San Francisco Bay after the invasion by *Potamocorbula* and the Seine are found. Around the 1/10 line standing for the optimal ratio between the macrofauna biomass and the primary production, the interval delimited by the 1/5 and 1/20 ratios corresponds with the areas where systems are properly functioning (good). The Grevelingen and the Somme are two systems that are close from the outer limits of this interval with ratios of 1/6 and 1/18 respectively. The high status, situated around the line of optimal functioning (1/10), was arbitrarily extended on both sites of this line at half distance (2/15 and 1/15) from the outer limits of the good.

3.1.2 Metric at habitat level

At this level the size, shape and spatial relationships of habitats or ecotopes can be evaluated, because they influence the dynamics of populations and ecosystems.

Firstly, the terminology used with respect to define habitats and ecotopes within a water body will be clarified. A substantial literature exists on the definition of \`habitat\` and related concepts, such as ecotope or biotope (Klijn, 1994). Habitats are defined by EUNIS (European Nature Information System) as \`plant and animal communities as the characterizing elements of the biotic environment, together with abiotic factors (soil, climate, water availability and quality, and others), operating together at a particular scale\`. In other words, habitats are defined from a biological point of view and in their relation to the physical environment. In contrast, ecotopes are

> <u>Box 3 – Level 2: Habitats (ecotopes) in the water body – spatial organization</u>
>
> *Aim: Are all representative habitats present in the water body that might be expected based upon hydro- and morphodynamic characteristics of the system.*
>
> At the second level the spatial distribution of habitats or ecotopes within the water body is considered. At this level one addresses the diversity of habitat types, and compares the availability and spatial organisation of these types to the expected possibilities, based on the physical boundary conditions in the system.
>
> Important criteria at this level are the presence/absence of expected habitats or ecotopes, but also their spatial organisation, measured in terms of surface area and connectivity. Within a habitat or ecotope a relation between the surface of the habitat and species richness exists, but these species-area relationships are still poorly known in estuarine systems. Surface area and connectivity are therefore expected to be indicative for the possibility of developing species-rich communities.
>
> Based on geomorphological theory, system characteristics (e.g. bed dimensions, elevation) and driving forces (e.g. tidal conditions, river flow, sediment input), it must be possible to construct expectations about the distribution of the different ecotopes and characterise the system in terms of surface area of ecotopes (e.g. surface of gullies, shallow areas, sandy or muddy intertidal areas). Examples of such simple predictive measures are the equilibrium relations between tidal volume and morphometrics of gullies in tidal systems. This type of measures can be used to test if significant deviations from the morphodynamic equilibrium are present in a water system, e.g. due to hydraulic management (e.g. construction of a storm surge barrier) or infrastructural works (deepening of a gully).

identified as `the environment of a community that is defined similarly as habitats by a combination of several abiotic parameters` (Ruiter & de Jong, 1997). In other words, ecotopes are defined as ensembles of physical (and chemical) conditions that constitute the biotope, with the biological communities that may simply inhabit the physical biotope, but in other cases may also shape it. Thus, muddy, intertidal sands with their typical community would constitute an ecotope, but so would an intertidal natural mussel bank, or a seagrass meadow, the latter two being shaped by the 'ecological engineers' (Jones et al., 1994) that modify the habitat (biogenic structure building species). Ecotopes are more or less homogeneous units on the scale of the landscape, identifiable by their similarities and differences in geomorphologic and hydrologic characteristics, and characterised by a benthic community linked to the above-mentioned abiotic conditions.
Both approaches are used in this report because of different availability of information, but the term habitat – a more widely used term – is used throughout the whole document. For water bodies where habitat information is available, the term habitat is used as a proxy for the ecotope. In the case, where the habitat typology is determined based on species assemblages, the term community is used to characterize the habitat or ecotope.

This level delivers also the opportunity to include the occurrence and distribution of important ecological engineers. Ecological engineers modify the habitat (biogenic structure building species) and are mostly characterized by a specific benthic community. For the Dutch waters, the most important ecological engineers are mussels and oysters, besides some tube-building species (e.g. *Lanice conchilega*). Mussel and oysters banks provide substrate for epiflora and epifauna, while the matrix provides interstices and refuges for a diverse community of organisms. These structures stabilize the sediment, profoundly modify the substratum and increase the turnover of nutrients and organic carbon in water systems. Due to their importance, it is advisable to map their distribution and monitor their specific benthic community.

Due to the fact that there is no uniform habitat classification for all different water body types, the different geomorphological drivers in the different water bodies and (most importantly) lack of information in many water bodies, it was not opportune to make a standardized evaluation (uniform parameters) tool at this level. Possible parameters that can be used at this level are:

- surface area coverage of habitats
- completeness and connectivity
- eco – elements (e.g. mussel beds, oyster beds, sea grasses, …)

These parameters can be evaluated against a certain `historical` reference period, expert judgement or against the management objectives for a certain water body.

3.1.3 Metric at community (within-habitat) level

The BEQI-method on the third level evaluates the state of the benthos within a habitat based on four parameters: number of species, density, biomass and species composition changes. These parameters reflect the normative definitions as defined by the WFD. Any proposed WFD classification scheme must include methodologies that address these parameters defined for assessing the benthic quality status: `the level of diversity` (BEQI parameter: number of species) and `density of invertebrate taxa` (BEQI parameter: density). The WFD parameter `proportion of disturbance-sensitive taxa` is not entirely the same in the BEQI methodology (BEQI parameter: species composition changes), because it does not classify species in disturbance sensitive taxa classes, but reflects wider species composition changes (see further). Our WFD classification scheme also includes biomass as a parameter, which is not directly required by the WFD, but which can be considered as additional estimate of the abundance. Biomass data reflect the partitioning of resources within the community better than density data and for shallow coastal and estuarine systems the biomass is a relevant parameter to link with the higher trophic levels such as birds and fish. The parameter ‘species composition changes’ evaluates changes in the species composition of the macrobenthic community belonging to a certain habitat. This approach allows detecting changes in the dominance of species, the disappearance of species from the community and the appearance of new (e.g. invasive) species in the community. In this case, the

<u>Box 4 – Level 3: Habitats and benthic communities – Molenplaat example</u>

It is well known that the composition of benthic communities is highly dependent on factors depending directly or indirectly on hydrology and morphology in estuarine and coastal systems (e.g. Ysebaert et al., 2002). One cannot expect the same benthic diversity, biomass, density or species composition in a highly dynamic part of a tidal flat characterized by sand waves and coarse sediment as in a sheltered, muddy part (Herman et al., 1999) (see figure 1).

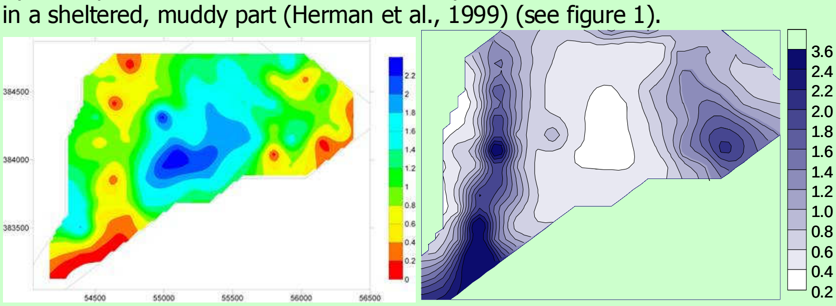


Figure 1. Interpolation of benthic biomass (left) and shear stress (right) on the Molenplaat (Westerschelde), which made it clear that the biomass is related to the shear stress (Herman et al., 1999). High benthic biomass is observed in the central part of the tidal flat where shear stress is low.

A single indicator system could never cope with these two different systems, both of which are a natural part of a tidal estuary, without at least a correction for the physical circumstances. However, merely correcting for the circumstances neglects the problem that, due to anthropogenic stress, the spatial distribution of these habitat types may change fundamentally. An indicator should therefore reflect the distribution of habitat types (level 2), as well as the distribution of species within these habitats (level 3). Otherwise, it will be insensitive (or worse: evaluate positively) for the degradation of estuarine habitats due to physical interferences.

species are not classified in disturbance-sensitive taxa, but all species are treated similarly. These four parameters are calculated and evaluated per habitat within a water body instead of per sample within a water body.

The parameter results strongly depend on the sampling effort (sediment surface) that is deployed. Therefore, the expected reference values for the parameters are calculated per habitat from permutations (KRW program, version 1.0 developed by Peter Herman in FORTRAN) executed over increased sampling surfaces. This allows estimating, for any given sampling surface, the reference value that can be expected. This program is based on bootstrapping with replacement. In statistics, bootstrapping is a modern, computer-intensive, general purpose approach. It is used for estimating the sampling distribution of an estimator (parameter) by re-sampling with replacement from the original samples, most often with the purpose of deriving estimates of standard errors and confidence intervals of a population parameter like the median (Efron, & Tibshirani, 1993).

The basic algorithm of this permutation test works as follows (also illustrated in box 5):

(1) A reference dataset is chosen, that is large enough to represent the main factors of variability in the habitat (temporal and spatial variability). In the reference dataset, all samples may have the same sample surface or not, but the surface sampled is taken into account in all subsequent calculations

(2) For a range of sample surfaces (the calculation restarts for every level of surface sampled), approximately 2000 random samples are drawn with replacement from the reference database. This is done by choosing a sample (at random) from the reference database, adding its density, biomass etc. and its surface to intermediate storage variables, and continuing this process until the desired surface is reached. For example, when doing the permutation test for an 'assessment' sample of 1 m^2, sufficient samples are draw from the reference dataset so that their surface amounts to the desired surface of 1 m^2, and sum all species abundance and biomass data into one species list with density and biomass attributes for this one random sample. Subsequently, the procedure is repeated 2000 times for this 1 m^2 surface. The end result is then a set of 2000 artificial random assessment samples with a surface of 1 m^2.

(3) For each of the 2000 random samples with a particular surface, their total density and total biomass is determined, as well as their species richness and similarity to the species composition of the complete reference data set. This yields 2000 values for each of the parameters density, biomass, species richness and similarity. A number of relevant percentiles of this distribution is determined that will serve a purpose in determining critical limits (see below for the procedure for the different variables).

(4) This whole permutation calculation is repeated for the next level of surface of the assessment sample, until an assessment surface equal to the cumulative reference surface has been calculated.

Reference values for each parameter, based on those permutations, are determined for each ecological status class boundary (Figure 4).

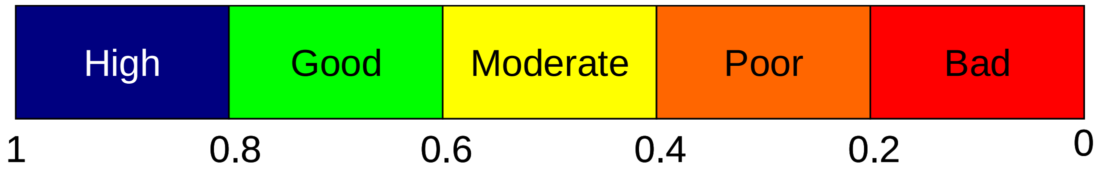
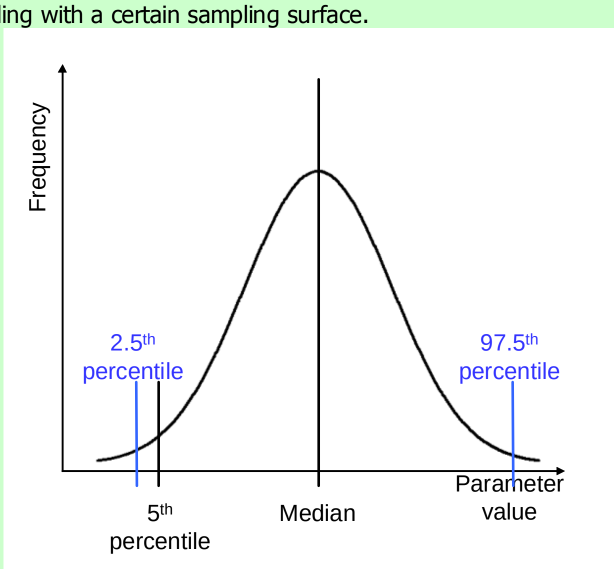


Figure 4. Ecological status class boundaries for the Netherlands

The good/moderate boundary is important within the WFD because it marks the boundary between action and no action to improve the status of the evaluated water body. When the status of the water body is good, no action has to be taken, while some action has to be taken when the water body is evaluated as moderate, poor or bad. For the different parameters, the reference value to be expected in the case of a good status corresponds with the 5^{th} percentile value out of the permutation distribution of each parameter (details in box 5). The 5^{th} percentile is a statistically accepted level, which is not too restrictive and which accounts for the variability within the reference data. Therefore, this level is considered to be suitable as criterion for the important good/moderate

<u>Box 5 – Randomisation and defining of reference boundaries.</u>

As described higher, for each sampling surface, at random 2000 samples are drawn with replacement from the reference database. Such result is illustrated in the figure below. This distribution is used to define the reference boundary values corresponding with a certain sampling surface.



For each level the 5^{th} percentile is selected as the good/moderate reference boundary, which means that for a one side approach (as for number of species and similarity), the 5% lowest values are deviating from the reference situation. The median value of this distribution is selected as the good/high reference boundary. For a two sided evaluation (as for density and biomass), the 2.5% lowest and highest values are selected as the boundaries at which there is a deviation from the reference situation.
These values of the randomization are calculated for each sampling surface and are visualized in Figure 5 and Figure 6.

boundary, following the WFD. Concerning the boundary setting, it has to be mentioned that in artificial and heavily modified water bodies, the high and good boundary are respectively defined as maximum ecological potential (MEP) and good ecological potential (GEP). In the further analysis, those boundaries are all named as high and good, independent of the water body type.

The different parameters and the calculation of the reference values and boundaries are described per parameter in the following section and summarized in Table 2 .

Number of species (species richness)

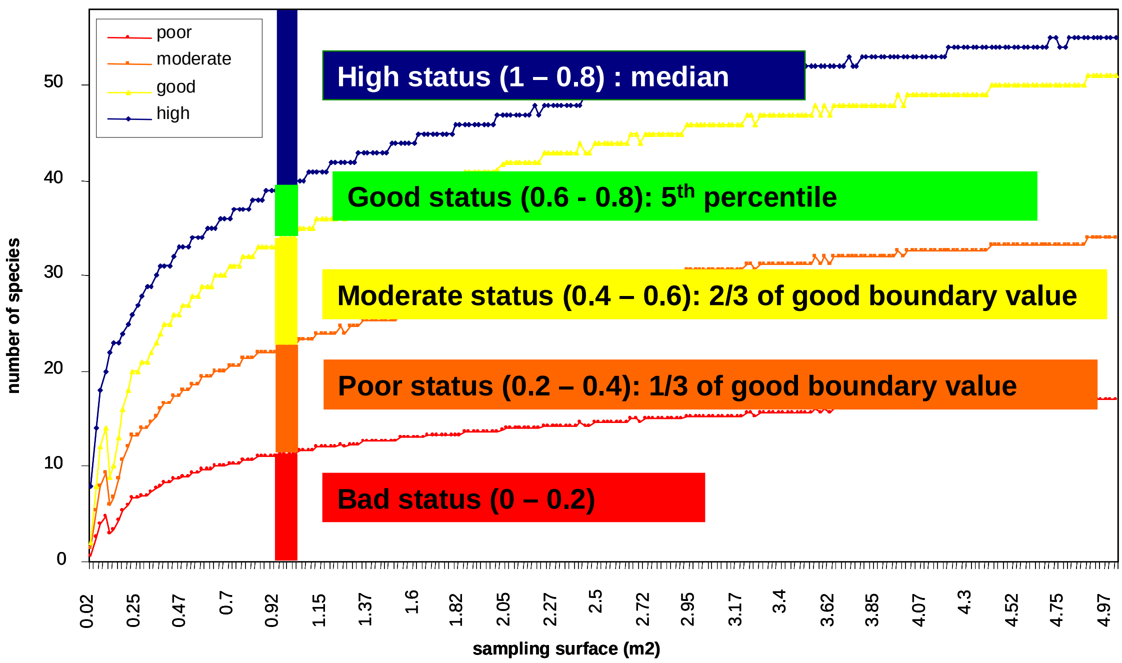


Figure 5. The reference boundary settings for number of species (used as example in the figure) and similarity in relation to sampling surface

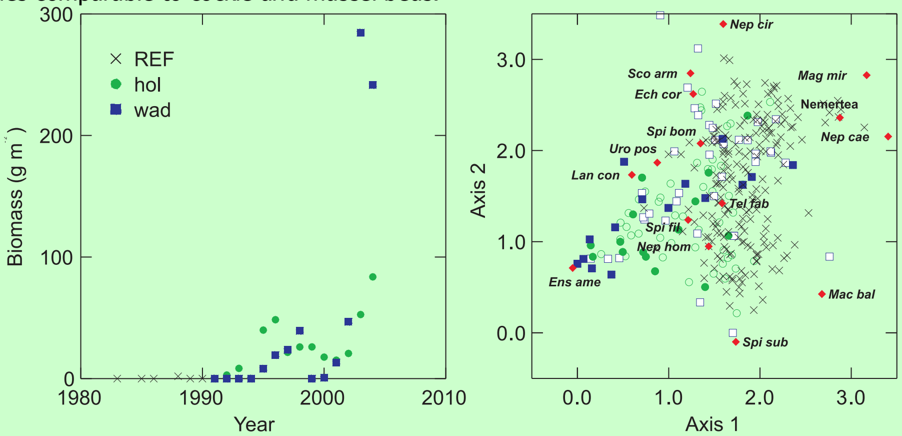
The number of species is an important diversity parameter in evaluating the status of a habitat and is widely used as an indicator for environmental changes or disturbance (Pearson & Rosenberg, 1978; Borja et al., 2007). Therefore, this parameter is selected to evaluate the diversity changes for the different habitats within a water body.

The permutation of the number of species allows estimating the range in the number of species that can be expected to be collected for any given sampling surface (Figure 5). Within this range, the median value is used as the lowest number of species to be expected in the case of a high status. The number of species corresponding with the 5th percentile is used as the lowest number of species to be expected in the case of a good status. The moderate/poor and poor/bad boundaries are scaled in equally intervals relative to the number of species measured for the good/moderate boundary and are respectively 2/3 and 1/3 of the number of species of the good/moderate boundary. The interval of 0.2 between the ecological status boundaries is equally scaled in correspondence to the interval between the number of species of the corresponding reference value boundaries.

Species composition change

<u>Box 6 – Species composition change – an example of an invading species *Ensis directus*</u>

The American Jacknife *Enis directus* (synonim *Ensis americanus*) was introduced in the North Sea presumably through ballast water. The first reported case of *E. directus* is from 1979 in the German Bight at the mouth of the river Elbe (Von Cosel et al. 1982). The invasive history was reconstructed and estimated that *E. directus* spread rapidly through the southern North Sea by approximation 125 km per year (Armonies 2001). In the past twenty years it has become a dominant species of the macrobenthic community of the coastal zone and the sub-tidal sediments of the Wadden Sea. Biomass and density can reach values comparable to cockle and mussel beds.



The left panel of the figure above is the average biomass of *E. directus* at sampled stations plotted against time first for the reference period and then separately for Waddenkust and Hollandse Kust. The right panel is a figure of the macrobenthic community composition of the Waddenkust and Hollandse Kust six mile coastal zone. It is a plot of the first two axis (in standard deviation units) of a detrended correspondence analysis (DCA) of species biomass. Only top ten species in biomass or abundance are included in the analysis, totaling 16 species accounting for 94% of the total biomass. Species are plotted as red diamonds with abbreviated names. The reference stations sampled from 1983 to 1990 are shown as black crosses. Open blue squares are Waddenkust assessment station samples from 1991 until 2001, filled squares from 2002 until 2004. Green circles are for Hollandse Kust assessment stations, open between 1991 and 2001 and closed from 2002 until 2004.
In the reference period from 1983 to 1990 for the North Sea coastal waters *E. directus* was mostly absent (no crosses near *E. directus* in the graph) and the community was dominated by other species. Changes in the species composition have taken place and in the period 2002-2004 *E. directus* is the most important species at several assessment stations.

This parameter aims at evaluating changes in species assemblage structure (species dominance, occurrence of new species and disappearance of species) between the assessment and reference situation. The evaluation of those changes is based on the Bray-Curtis similarity between the two datasets, after a fourth root transformation. In this case, the species are not classified in disturbance-sensitive taxa, but all species are treated similarly.

A similarity distribution of a certain sampling surface is calculated based on randomization of the reference samples to a selected sampling surface and on the calculation of the similarity of those randomly selected samples to the total reference sample (all samples), which results in a similarity range for a certain sampling surface (Figure 5). The 5th percentile of this similarity range is used as boundary for moderate/good. The median value is used as the lowest similarity to be expected in the case of a high status. The moderate/poor and poor/bad boundaries are scaled in equally intervals relative to the similarity measured for the good/moderate boundary and are respectively 2/3 and 1/3 of the number of species of the good/moderate boundary. The interval of 0.2 between the ecological status boundaries is equally scaled in correspondence to the interval between the similarities of the corresponding reference boundaries.

Density and Biomass

Both the macrofauna density (ind.m^{-2}) and biomass (g AFDW.m^{-2}) are treated in a similar way. The description below refers to density but also fully applies to biomass.

The permutation of density allows estimating the range in density that can be expected to be collected for any given sampling surface (Figure 6). A lower as well as a higher deviation (two sided evaluation) of this range is evaluated as negative, because an increase in density does not always indicate an improvement of the ecosystem. In fact, density increase is classically considered an indicator for organic enrichment problems (e.g. Rosenberg, 2005). Therefore, for each boundary a minimal and maximal density is defined for any given sampling surface. When the average density in the assessment falls between the 25th and 75th percentile of this range, a high status is reached. The range in density that can be expected in the case of a good status corresponds with the 2.5th and 97.5th percentiles. Due to this two-sided evaluation, the 2.5th and 97.5th percentiles are selected to border the good/moderate boundary, instead of the 5th and 95th percentiles to get a same total 5% selection of deviation of the reference state for the four parameters. The moderate/poor and poor/bad boundaries are scaled in equal intervals relative to the minimal or maximal density measured for the good/moderate boundary and are respectively 2/3 and 1/3 of the minimal density value and 4/3 and 5/3 of the maximal of the density value of the good/moderate boundary. The interval of 0.2 between the ecological status boundaries is equally scaled in correspondence to the interval between the minimum or maximum densities calculated for the corresponding reference boundaries.

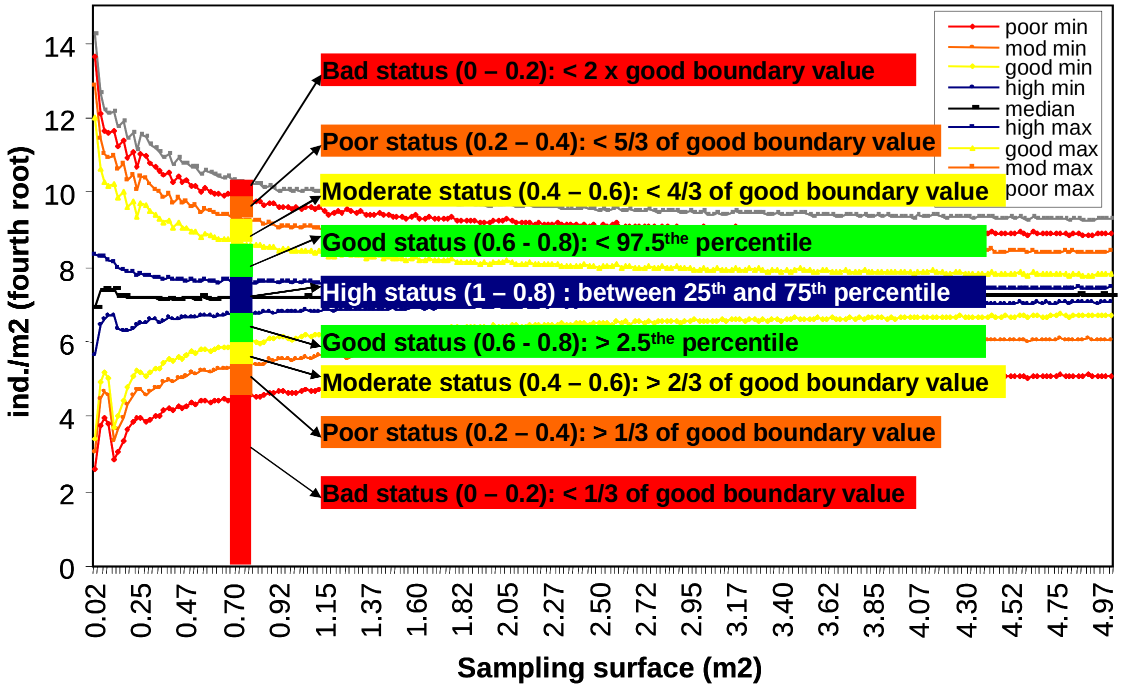


Figure 6. The reference boundary settings for density in relation to sampling surface. A similar protocol is applied for biomass

Table 2. Overview of the reference value boundary settings coupled to the fixed ecological status class boundaries

Ecological status class boundary	Reference value boundaries	
	Number of species	Density
	Species composition changes	Biomass
High/good: 0.8	median	25th and 75th percentile
Good/moderate: 0.6	5th percentile	2.5th and 97.5th percentile
Moderate/poor: 0.4	2/3 of Good/mod value	2/3 and 4/3 of Good/mod value
Poor/Bad: 0.2	1/3 of Good/mod value	1/3 and 5/3 of Good/mod value

Risk of misclassification

Precision and confidence of the classification method are crucial elements for decisions to invest large sums to improve the ecological quality (Carstensen, 2007). Within the WFD, the good/moderate boundary is important, because it marks the boundary between action and no action to improve the status of the evaluated water body. When the status of the water body is good, no action has to be taken, while some action has to be taken when the water body is evaluated as moderate, poor or bad. Due to the selection of the 5th percentile, there is a chance of 5% that a parameter is evaluated as bad/poor or moderate, while in fact it is good. Note, however, that this percentage applies to each of the parameters separately, and that it is much more difficult to quantify the risk of misclassification for the combined index. When the four parameters would be independent, the risk of misclassifying all of them is negligible (order 10^{-6}), but independence is unlikely and the real risk will be between virtually zero and 5 %. The probability that the combined index based on the four parameters leads to misclassification is, similarly, well below 5 % but the exact probability level is unknown.

Overall EQR of the community level

The overall Ecological Quality ratio (EQR) of a habitat within a water body is calculated by averaging the Ecological quality ratios of the four parameters (density, biomass, number of species, species composition changes) of that habitat. When the average is situated between 1-0.8, 0.8-0.6, 0.6-0.4, 0.4-0.2 or <0.2, than the Ecological status of the water body at the community (within-habitat) level is respectively high, good, moderate, poor or bad.

Interpreting the EQR values at the community level

The aim of the community level evaluation of the BEQI method is to detect and evaluate changes in the assessment data compared to the determined reference condition for each parameter, within a habitat of a water body. These results are scaled in accordance with the requirements of the WFD.

The outcome of these calculations can be as follows:

→ The overall Ecological Quality ratio (EQR) and the EQR of the parameters of an habitat within a water body are evaluated as good or high.

This should be interpreted as a status quo in the Ecological Quality status in comparison with the reference situation.

→ The overall EQR was evaluated as good or high, but one or more EQR values of the parameters are evaluated as moderate, poor or bad.

This should be interpreted as a first warning of possible changes in comparison with the reference situation. A more detailed investigation (e.g. through operational or investigative monitoring) in the changed parameter is advisable.

→ The overall EQR was evaluated as moderate, poor or bad.

This should be interpreted as a strong change in comparison with the reference situation. This requires action of the government in co-operation with the scientific community to investigate and tackle those changes. Based on a detailed investigation of the results the managing authorities – in close cooperation with the scientific community – can decide to take measures or to install an operational or investigative monitoring program.

3.1.4 Overall classification of the water body

To give a reflection of the ecological status of the water body, the ecological status classification at the three levels of the BEQI method has to be summarized in one overall ecological status class. In the BEQI method priority is given to both **transparency** and **simplicity** and means that each step of integration will remain visible and editable for the purpose of management priorities. At the level of the ecosystem, one parameter value is obtained, but at the other two levels more parameters are calculated and the overall EQR value of that level is obtained by averaging.

At the highest level of integration between our three main metrics, the relative sensitivity of the metrics to stressors is considered as a weighing factor by the averaging into the overall metric for the whole system. As said earlier, community shifts occurring within a habitat are a response to a change in a parameter that is not used for the definition of the habitat; that could be for example sediment and or water chemistry, temperature. On the other hand, changes in the habitat distribution (size and proportion) will also induce changes in community at the scale of the water body but should then be rather interpreted as response to changes in the morpho-/hydrodynamic conditions than to an effect of deteriorated water quality.

The parameters representative for the habitat and the community (within-habitat) level could then be interpreted as parameters for disturbances either related with morpho-/dynamic or water or sediment quality conditions. As such, both metrics should have an equal contribution to the overall metric.

The parameter at the ecosystem scale represents the interaction term between the processes acting at and within the habitat level but also in relation with the rest of the ecosystem (mostly the pelagic system). As such it integrates the whole spectrum of interactive processes that tend to buffer the various disturbances affecting the system. As a consequence, the B_{macrof} : P_{prim} is rather robust as disturbances will have to exceed the buffer capacity of the system (its resilience) before a significant shift in this metric will be induced. With respect to its rather low sensitivity the parameter at the ecosystem scale should have a lower weight to the overall metric than the parameters at the habitat and community level. This lower sensitivity should not be interpreted as a lower informative value for this level that has clear signal function for the whole system through its integration of the main functional processes at play in the ecosystem.

Finally, there is proposed to attribute a weighing factor equal to 2 for the metrics at and within the habitat scale and equal to 1 for the metric at the ecosystem scale.

3.2 *Assessment precision and power*

The precision of and confidence in the classification method are crucial elements for making decisions to invest large sums to improve the ecological quality (Carstensen, 2007). At level 3 of our assessment method (the community evaluation), a confidence of 5% is reached for the different parameters (see higher).

The precision of defining the ecological quality status for a water body depends on the available reference and assessment data, mainly the following factors:

- The size of the reference dataset
- The total sampling surface and number of samples in the reference and assessment dataset
- The homogeneity or heterogeneity of the data set and the habitat

Therefore, it is opportune to give a certain status to the assessment results, which reflects the degree of precision. This degree of precision is developed for level 3 of the BEQI approach.

First of all, the size of the reference dataset of a habitat has to be large enough to cover the temporal and spatial variability within that habitat. Based on an *a posteriori* evaluation of the different species-area curves, a minimal total sampling surface of 1 m^2 was chosen as a limit for the reference dataset. In the case this requirement was not fulfilled for a certain habitat, no reference settings could be determined for that habitat.

Secondly, for all habitats for which the above requirement is fulfilled and for which the reference analysis was done, a certain degree of assessment precision is determined. The basic approach for this analysis is as follows. For every parameter, the range of values that will give rise to the classification ‘good’ or better, will decrease with increasing sample surface. Figures 5 and 6 illustrate this effect. The reason for the narrowing down of the range is that with increasing assessment sample surface, it is easier to detect a significant deviation from the reference. It can also be observed in these figures that the range decreases rapidly with sample surface for small surfaces (left side of graphic), but little or not for large surfaces. The increase of precision with increasing sampling surface is not linear, but levels off as sampling surface increases. This is equivalent to the law of decreasing added value: as sample surfaces become larger, it is less and less ‘profitable’ in terms of precision to increase the sampling surface even more. Based on this principle, one can make a distinction between sampling efforts where the addition of an extra sample yields much improvement in precision and sampling efforts that are already so big that adding yet an additional sample adds little or nothing to the precision.

Assessment precision

The following assessment precision classes are used to qualify the precision of the assessment results presented in this report.

- **Unacceptable**: a good assessment analysis cannot be done, because the variability of the parameters in the reference datasets is too high. In this case, the score and status obtained should not be included in the overall EQR score and status classification.
- **Minimal**: the assessment score is accepted, but the result has to be interpreted with caution, due to a strong increase of the precision with increasing sampling effort; the precision is low.
- **OK**: no direct assessment problem, but it is advisable to increase the sampling surface in the future to increase the precision of the assessment, because the precision still slowly increases with increasing sampling effort.
- **Optimal**: no assessment problem; a higher sampling surface will not increase the precision drastically, but the asymptotic value is still not reached.
- **Maximal**: no higher sampling surface needed, because the precision will not increase with sample surface any more (close to the asymptotic value).

The procedure to determine the assessment precision classes per habitat is as follows:

- For each sampling surface of the reference value distributions, the changes in the \`good/moderate\` boundary value (5th percentile) with sampling surface was evaluated with a local regression encompassing 6 neighboring sample surfaces. The slope of these regressions was plotted against sample surface. The higher the slope value is the stronger the change in precision with increasing sample surface, and the lower the precision of the actual determination of the parameter at this surface (Figure 8).
- For each parameter, boundaries for the assessment precision classes are determined (based on the slope values) (Figure 7). These boundaries determine the sampling surface needed to have a certain assessment precision. They are chosen in such a way as to make a distinction between visually different classes of precision.
- For each assessment class and parameter the sampling surface corresponding with the boundary values are listed.
- The sampling surfaces obtained for each parameter are compared and the highest is selected to determine the minimal sampling surface needed for a certain precision class.

Table 3. The boundary determination for the different assessment precision classes based on the slope values for the different parameters. Values given are for the slopes of the local regressions of parameter values with sample surface

Assessment power class	slope value Biomass	slope value Density	slope value Species richness	slope value Similarity
unacceptable	high variability	high variability	> + 6	
not advisable	> + 0.06	> + 0.6	> + 2	> + 0.06
OK	+0.06 <> + 0.02	+0.6 <> + 0.2	+1 <> + 2	+0.06 <> + 0.02
optimal	+0.02 <> + 0.01	0.2 <> 0.1	< + 1	+ 0.02 <> 0.01
maximal	< + 0.01	< + 0.1		< + 0.01

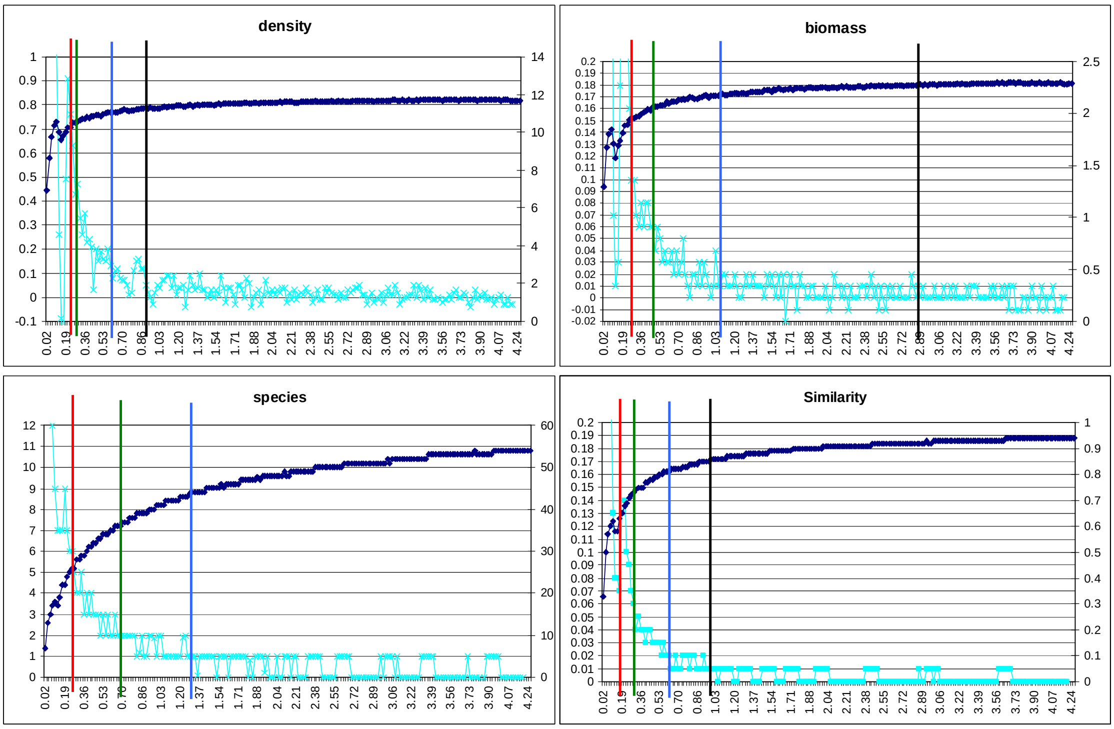


Figure 7. Example of the determination of the sampling surfaces corresponding with the different assessment precision classes for the different parameters for a habitat of the Westerschelde. In dark blue the `good` boundary line is plotted (values on right axis), whereas the light blue line corresponds with the changes in slope of the `good/moderate` boundary line (slope value) (values on left axis; biomass (fourth root gAFDW/m^2), density (fourth root in/m^2), species (number of species) and similarity (Bray-curtis)). The straight lines correspond with the assessment precision class boundaries: Red line: between unacceptable and minimal; green line: between minimal and OK; blue line: between OK and optimal; black line: between optimal and maximal

The dependence of assessment precision on sample surface varies with the type of parameter, habitats (heterogeneous or homogeneous) and the reference data set. A larger total sampling surface is needed in habitats with a low number of individuals and a high number of species, than in habitats with a high number of individuals and a low number of species. Figure 7 illustrates the principle for habitats of the Westerschelde. It plots the required sampling surface to obtain the "OK" status with the ratio of abundance over species diversity.

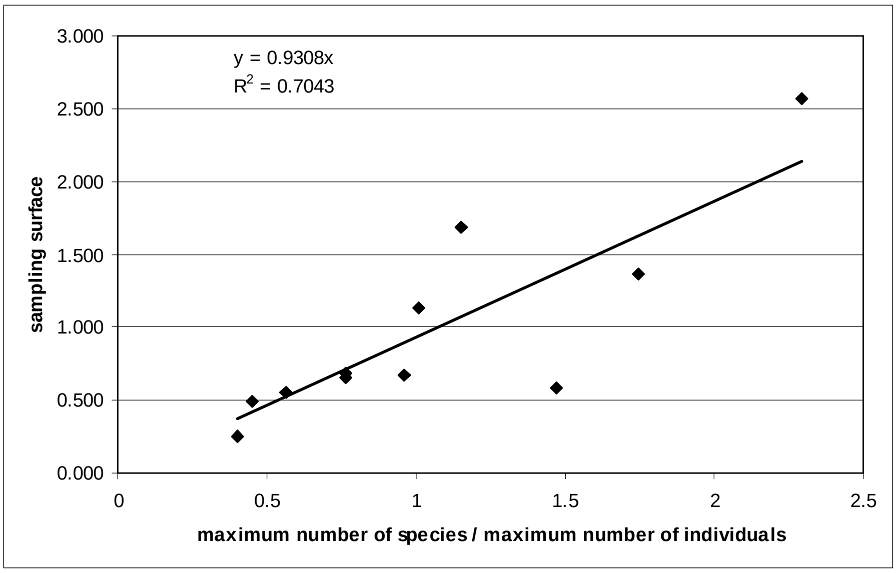


Figure 8. Relation between the ratio of the maximum number species and the maximum number of individuals and the sampling surface for the habitats (black points) of the Westerschelde

Monitoring effort estimations

These calculations can also be used to make estimations of the minimal, optimal and maximal monitoring effort that is needed for a certain habitat within a water body to get a reliable assessment. The assessment precision class can be translated into these monitoring efforts:

- Minimal monitoring effort: sampling surface corresponding with the boundary between \`minimal\` and \`OK\` assessment precision class.
- Optimal monitoring effort: sampling surface corresponding with the boundary between the \`OK\` and \`optimal\` assessment precision class.
- Maximal monitoring effort: sampling surface corresponding with the boundary between the \`optimal\` and \`maximal\` assessment precision class

These sampling advices for the monitoring can be further modified by expert judgement to get a more uniform monitoring effort between certain habitats within the water body. Considering monitoring not only the total obtained sampling surface is of importance, but also the number of samples needed to obtain that total sampling surface. More details will be given in the chapter on the monitoring.

Power of assessment

It is advisable to determine the effective power of the assessments, which means to detect which is the chance to find a certain deviation from the reference value. This will be outlined in this section, with a brief example on the present assessment results. This is an issue that needs further attention and investigations in the development of monitoring programs and strategies.

In a statistical test aiming at determining the significance of a difference between two populations, two different types of error can be committed. Type I error is concluding that there is a difference, while in fact there is none. As stated above, the probability of such errors depends on the significance level used. In this study, this probability level is 5 %. Type II error is the reverse: concluding that there is no difference, while in fact there is one. The probability of a Type II error depends on the variance, the effect size and the choice of the level of significance, the α which is set to 0.05. The *power* of a test is defined as 1-β, where β is the probability of a Type II error.

The BEQI method at level 3 describes the variance of the average reference conditions of the four parameters as a range of probability distributions along an axis of sampling effort. This distribution at every sampling surface is described by a median, quartiles and percentiles. This distribution is estimated with a permutation technique, by randomly drawing samples from the reference data set. Any average assessment value that falls outside the 95% of the random distribution around the median (which is outside the 2.5% and the 97.5% percentile borders) is defined to be significantly different from the reference conditions (at α=0.05). With a sample drawn from the same population as the reference population the chance is 5 % that it is rated significantly different and 95 % that is rated the same.

The power of the assessment is just the chance that the average of an assessment value falls outside the 2.5 and 97.5 borders of the distribution of the variance. When the assumption is made that the variance structure remains unaltered when the median of a population changes (in other words the average of the assessment may be different from the reference but the distribution of the quartiles around the median of the variance remains the same) it is easy to estimate the amount of overlap between the distribution of the reference and the distribution of the assessment.

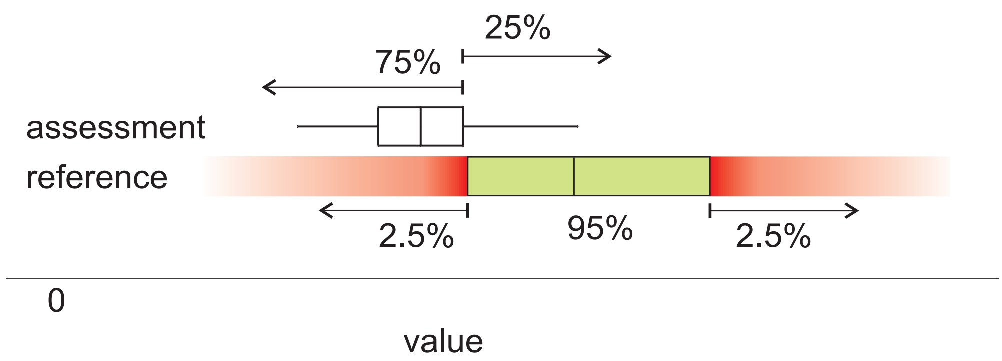


Figure 9. Example of the estimation of the statistical power to detect a significant difference between a reference sample and an assessment sample. The assessment box is like in a regular box plot, showing the median and the second and third quartile. The reference box shows the median with the 2.5 percentile and 97.5 percentile as the box edges. Any value falling in the red area is significantly different at the 5% level (α=0.05)

In Figure 9 an example is given. The reference condition for a certain sample surface is described along an axis by the median and the green box around the median. The range of the reference box is 95 % of the distribution running from the 2.5 percentile to the 97.5 percentile. Any average value from a sample drawn from a population that falls outside the range of the green box is from an (assessment) population with a median significantly different from the median of the reference. The power is the chance to find a mean value outside the range of the green box. This is the same as the fraction of the distribution of the variance of the assessment sample (the white box) that falls outside the range of the green box. In the figure the white box shows the second and third quartiles of variance distribution around the median. There is still about 25% overlap with the reference box. The chance to find a significant difference in this particular case between the reference and the assessment is 75 %. It is assumed that the variance structure does not change with a change in the average of the population. This means that the quartile width of the assessment sample is the same as that of the reference.

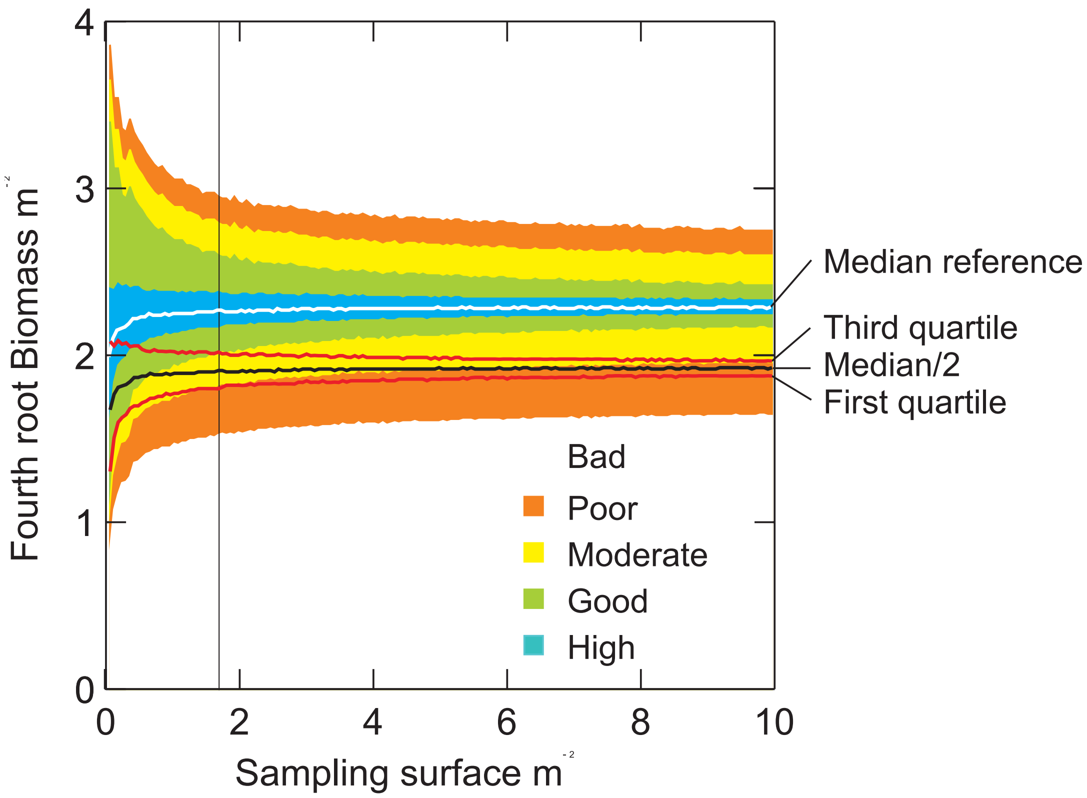


Figure 10. Example of the estimation of the sample size required to find an effect size of 2 (in this case a two times reduction) with a power of 75%

The effect size in this case is the difference between the median of the assessment and the median of the reference. This is the sum of the difference between the median of the reference and the 2.5 percentile and the difference between the median and the 75 percentile of the assessment.

In Figure 10, it is explained what a sampling effort (in surface units) is required to detect a significant difference with a certain effect size with a chance of 75%. In this figure, the variance of the reference biomass (around the median, white line) is given of the Q1 habitat of the coastal zone of the Hollandse Kust, Waddenkust and Eems-Dollard kust. The significance level runs along the border between good (green) and moderate (yellow). The black line is a median which is twice as small as the median of the reference (in other words an effect size 2). The red lines are the first and third quartile borders derived from the reference distribution. At the sampling surface where the third quartile border crosses the border between good and moderate (at the black vertical line) 75% of the distribution of the "assessment" sample falls below the 2,5 percentile. This means that the chance is 75% to find a significant difference with an effect size of 2, is at a sampling surface of about 1.75 m^{-2}.

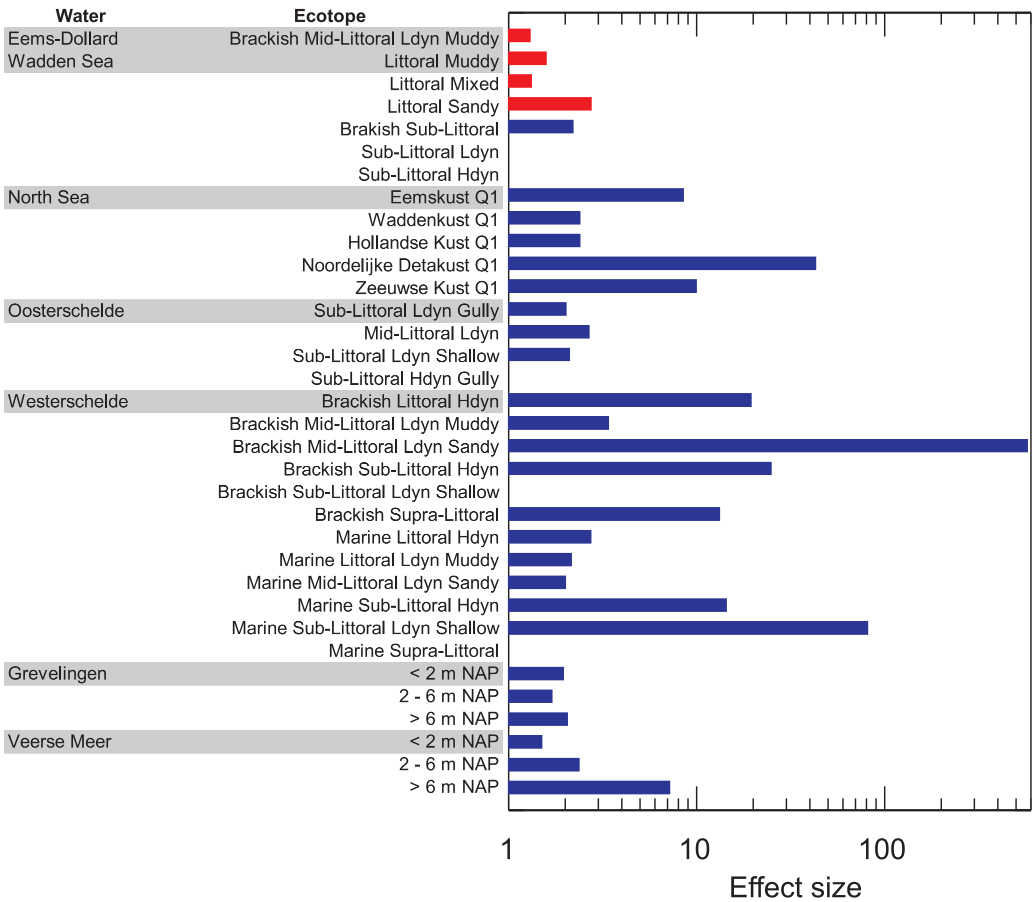


Figure 11. Effect size of a reduction in biomass detectable with power 0.75 at the realized sampling size of assessment samples used for the present qualification of the habitats within every water body

The other way around it is also possible to calculate the detectable effect with a power of 0.75 at a given sampling surface. This is done by summing the difference between the median and the

2.5 percentile or 97.5 percentile and the difference between the median and the 25 or 75 percentile.
There is a large difference between habitats in the effect size that can be detected with a power of 75% using the available assessment samples (Figure 11). For example in Lake Veere in the habitat deeper than 6 m a seven times reduction of the biomass reference value can be detected with a power of 75%. In the same lake in the shallow stratum <2 m there is already a fair chance to find a significant difference between the reference and the assessment when there is a drop of a little more than 30% of the biomass, to 67% of the reference value.
This means that changes in biomass are judged differently between habitats in the same water body and also between water bodies. This is an effect of heterogeneity of the habitats but also of the sampling design and the number of samples used. It is clear from Figure 11 that in some water systems (e.g. Ems-Dollard, Waddensea) a considerable sampling effort is spent on one or a few habitat types only. In addition, this effort is also spatially concentrated and thus only reflects one particular realization of the habitat type. The advantage of high power within this well-studied (sub) system comes with the disadvantage that no real insight is given into trends on a system-wide basis. No information is available on many other habitat types, nor even on the same type at other locations. At the same time, the very high power in the selected habitat may result in a different appreciation of any possible change, compared with other water systems in the country.

In other water systems, Figure 11 illustrates that some habitat types are undersampled, with a very poor power as a direct consequence. This undersampling may be caused by the relatively high heterogeneity of the habitat types. Usually it concerns habitat types with low densities. A solution may be to increase individual sample size in these habitats, thereby decreasing the inter-sample variability. However, there may also be a need to reconsider the number of habitat types to be distinguished, by lumping some of the more detailed types into a single type.
A better streamlining of the sampling effort over water systems and habitat types within water systems must be the aim of improving the monitoring strategy. Results as shown in Figure 11 can be a guidance for this optimization of the monitoring.

3.3 Reference conditions

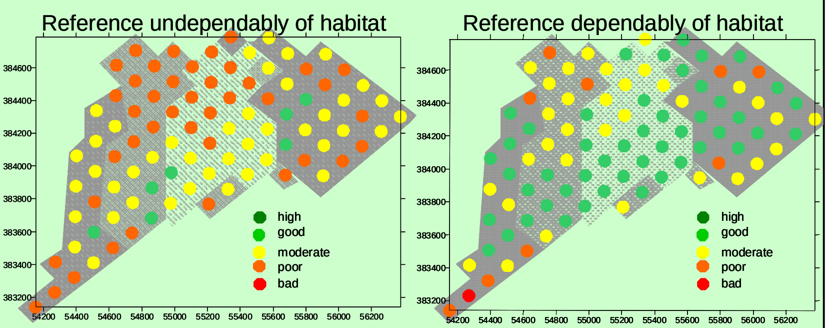
According to the WFD guidelines, quality class boundaries should be set in relation to a reference condition. Ideally, such a state should be either before, or spatially outside the influence of human activities. In Dutch (and most European) waters, no measurements for the period before the 1950s exist that allow assessment of species diversity or sensitivity with the methods, which are used today. Furthermore, it is difficult to imagine that there exist un-impacted areas. The remaining option to establish something similar to a reference condition or at least a good condition is to select data that represent from areas/ time periods where/when human impact was at a minimum. Also it is important to incorporate into the reference condition the natural variability (both spatial and temporal) that characterizes benthic communities. The following procedure is applied to obtain a reference data set for the Dutch water bodies:

1. For each reference area the best 'historical' dataset which was available for a certain habitat (see separated water body sections for more details) was selected.
2. Definition of the reference data set: Of this dataset, approximately the first one third of the years was used to extract the reference conditions. This period is eventually adapted after discussion. If during the selected years, a known (described) anthropogenic disturbance has occurred during a certain year, this year was eliminated in the reference dataset.

At this moment this is a very pragmatic approach, because often there is no evidence that these data are free from anthropogenic disturbance. In some cases it is even clear that some kind of disturbance was already present in the chosen reference period. However, in the absence of data on a period without this pressure, the selection criteria for the BEQI reference period are reduced to finding a period at the start of the observations, with a fair representation of the temporal

Box 7 – Reference conditions per habitat

To avoid the insensitivity of the index and reference settings to natural physical conditions, it is important to distinguish the benthic communities (habitats) within the water body and to define the reference conditions for each habitat within a water body. This problem is illustrated in the figure below, based on m-AMBI calculations of samples on the Molenplaat (Westerscheldt). These are done once with the reference values determined independently (left figure) of the habitat characteristics (1 reference value for the entire water body) and once dependent on habitat (right figure) (reference value determined per habitat within the water body).



The EQR value (poor – moderate) of the habitats are lower with an overall reference value compared to a reference value determined per habitat. This is due to the fact that the benthic ecological potential (species richness, density, biomass) depends on the physical characteristics of a habitat and is not equal between different habitats.

and spatial variability of the habitat. Selection of the oldest part of the data as reference has at least the advantage that deviations from previous conditions in recent years can be detected clearly. There is no point in evaluating present-day data against a present-day reference, since the outcome would always be ‘good’ ecological status. Nevertheless, in cases where the present situation deviates strongly from the chosen reference, additional research may be needed to determine whether the observed deviations are caused by present-day human pressures, rather than by historical pressures that have influence the reference more than the assessment conditions. Another option is to improve the reference data set in the future with `historical` data of similar habitats in other neighboring countries.

In the case of heavily modified water bodies, the WFD uses a Maximal Ecological Potential (MEP) as reference (the highest ecological potential of the water body), and a Good Ecological Potential (GEP) as objective (similar to a good ecological status). The distraction of the MEP and GEP for heavily modified water bodies is based on the Praagse method, which used the present situation as starting point. Based on the present situation, there is described which measures are realistic to increase the ecological quality of the water body. Measures which cause significant damage to the present functions or the environment and measures with a low cost-effectivity are excluded. The results obtained with these measures are defined as the Good Ecological Potential. Details of the WFD – measures and aims for defining the MEP and GEP for the saline waters of the Netherlands are described in Sierdsma et al. (2007). The MEP and GEP objectives for the benthos are defined in this report and based on the same principle as defining the reference conditions (good and high status) for natural waters. But for the heavily modified water bodies high and good status are respectively named as the maximal (MEP) and the good ecological potential (GEP).

The time period and settings of the reference condition differs between all water bodies, due to the difference in data availability and knowledge and will be described in detail in the separate sections.

4. Application of the BEQI to the Dutch coastal and transitional waters and saline lakes

4.1 Introduction

4.1.1 Dutch coastal and transitional waters and saline lakes

The Netherlands distinguish several water types and water bodies within the coastal and transitional waters. Saline lakes are a separate category. Both Dutch typology acronyms and comparable intercalibration acronyms are indicated (Figure 12, Table 4).

Three types of coastal waters are discerned: open polyhaline (Hollandse kust, Noordelijke Delta kust and Eems-Dollard kust, open euhaline (Zeeuwse kust and Waddenkust), sheltered polyhaline (Oosterschelde and Wadden Sea)

Transitional waters include two water bodies, the Eems-Dollard estuary in the north and the Westerschelde (Dutch part of the Schelde estuary) in the south of the Netherlands.

Two saline lakes) are considered, lake Veere and lake Grevelingen, both situated in the Delta region in the southwestern part of the Netherlands (Figure 12).

The water bodies Haringvliet, Nieuwe Waterweg and Nieuwe Maas are not investigated in this report.

The description of the results on the application of the BEQI in the paragraphs below does not follow the (final) Dutch typology nor the intercalibration types but simply follows the coastline starting with the waterbodies of the Delta region in the south-west and ending with the saline lakes. Not the types but the water bodies are where possible starting point for the assessment.

Table 4. List of water types and water bodies for which an ecological quality assessment for benthic macroinvertebrates is presented

	Name Water body		River basin	Dutch type	Intercalibration type
	Coastal waters: open polyhaline			**K1**	**CW-NEA 3**
1	Eems-Dollard kust	(Ems-Dollard Coast)	Eems		
3	Hollandse kust	(Holland Coast)	Rijn-West		
4	Noordelijke Delta kust	(Northern Delta Coast)	Maas		
	Coastal Waters: open euhaline			**K3**	**CW-NEA1**
2	Waddenkust	(Wadden Coast)	Rijn-Noord		
5	Zeeuwse kust	(Zeeland Coast)	Schelde		
	Coastal waters: sheltered polyhaline			**K2**	**CW-NEA4**
6	Waddenzee	(Wadden Sea)	Rijn-Noord		
7	Oosterschelde	(Eastern Scheldt)	Schelde		
	Transitional waters			**O2**	**TW-NEA 11**
8	Eems-Dollard	(Ems-Dollard)	Eems		
9	Westerschelde	(Western Scheldt)	Schelde		
	Saline lakes			**M32**	No NEA type
10	Grevelingen	(Lake Grevelingen)	Schelde		
11	Veerse Meer	(Lake Veere)	Schelde		

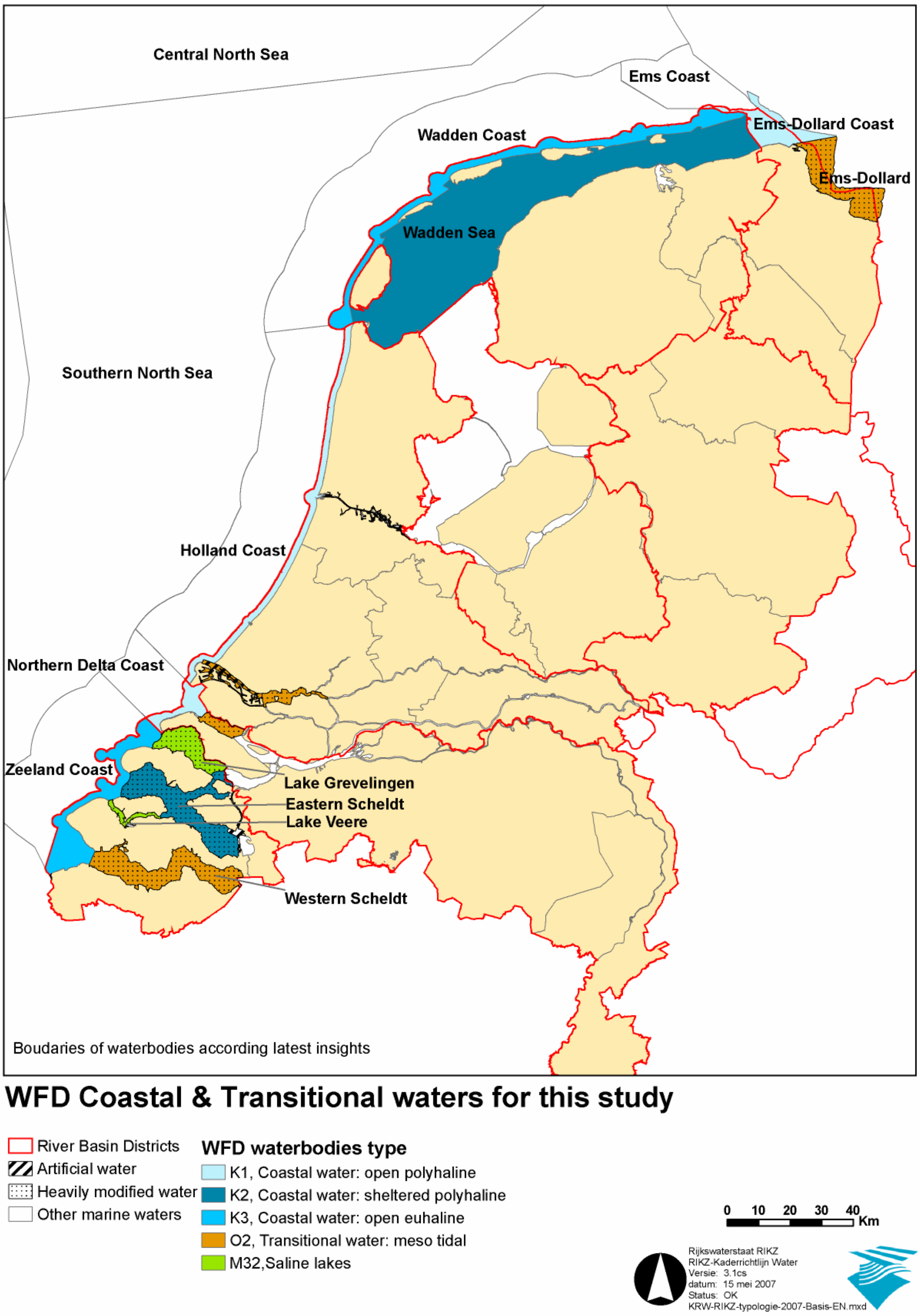


Figure 12. Dutch WFD water bodies (coastal & transitional waters and saline lakes) reported in this study. Dutch types with Intercalibration types (in brackets) : K1 (CW-NEA3), K2 (CW-NEA4), K3 (CW-NEA1), O2 (TW-NEA11)

4.1.2 Benthos data

Data origin

The available data to set the reference conditions and to do the assessment for the different water bodies is summarized in Table 5. All the available data was compiled in a database, named `Database WFD - biotic and abiotic information per waterbody.mdb` which is maintained by NIOO in the purpose of the KRW benthos project. The data is originating from different projects and time periods and collected with different sampling devices and strategies (see monitoring, section 5.3 for more information). The benthic data treatment is uniform for all datasets (using of 1 mm sieve and determination to species level when possible). Due to the variability in data availability between the water bodies, the exact selection of the required data for the reference and assessment analysis is described in the separated chapters per water body.

The problem of difference in sampling devices and strategies is solved by analyzing the changes in the parameters in function of the sampling effort. Also important is the selection of samples belonging to one single season to exclude seasonal effects from the datasets. The autumn period is the period when the highest numbers of species and biomass are found compared to the spring period. Therefore, it is advisable to do the reference and assessment analysis on data from the same season. The autumn period was chosen in most areas, due to the fact that this was the most intensively sampled period and the highest species richness and biomass are found. Only for the coastal area, where the assessment is based on spring samples and for the Wadden Sea, where the reference and assessment is based on spring samples, another season was used (more details in the separated sections).

Databases

All the date used to define the reference conditions and to do the assessment is stored in one database (`benthos database WFD report`, included in the appendix). For each water body, 4 tables are included in the database, which store the species info and metadata for the assessment and reference data. This data belongs partly to the NIOO or NIOZ and the use of it is limited to the control of the calculations within this report. In this database 11718 sample records and 111421 species records are included.

Besides, a database which store the basic data, also a database to calculate the level 3 of the BEQI method and other international WFD metrics (m-AMBI, IQI, NQI, DKI) is developed (`Database for WFD calculations`, included in the appendix) as also a program to calculate the permutations (`KRW.exe`, included in the appendix).

Table 5. Overview per water body of the data origin and sampling strategy

Water body	Projects	Time period	Sampling devices
Coastal waters: polyhaline			
Zeeuwse kust	BIOMON Wijnhoven et al. (2006)	1990 – 2005 1983 - 2004	Reineck boxcorer Van Veen, Reineck boxcorer, Cores of different diameter
Noordelijke Deltakust	BIOMON Wijnhoven et al. (2006)	1990 – 2005 1983 - 2004	Reineck boxcorer Van Veen, Reineck boxcorer, Cores of different diameter
Hollandse kust	MILZONE BIOMON	1987-1990 1990 - 2004	Reineck boxcorer
Waddenkust	MILZONE BIOMON	1987-1990 1990 - 2004	Reineck boxcorer
Eems-Dollardkust	MILZONE BIOMON	1987-1990 1990 - 2004	Reineck boxcorer
Coastal waters: sheltered and polyhaline			
Oosterschelde	BIOMON	1990 - 2005	Cores of different diameter
Wadden Sea	Balgzand Piet Scheveplaat Groningerwad Sub-littoraal	1969 - 2005 1988 - 2005 1988 - 2005 1990 - 2005	Cores Cores cores cores
Transitional waters			
Westerschelde	BIOMON MOVE ZEEKENNIS (Ysebaert et al., 2000)	1990 – 2005 1994 – 2005 1978 - 1999	Reineck, Cores of different diameter Reineck, Cores of different diameter Cores of different diameter, Van Veen, Reineck
Eems – Dollard	Dollard (Heringsplaat)	1988-2006	core
Saline lakes			
Lake Veere	BIOMON	1990 - 2005	Cores of different diameter
Grevelingen	BIOMON	1990 - 2005	Cores of different diameter

Data truncation rules

It is very important to standardize the taxonomy within the datasets, especially when data is derived from different projects or laboratories, before you calculated the results. Therefore, the various reasons for truncation of the data and the applied rules are summarized in this section.

- Removal of non benthic invertebrate taxa (e.g. fish and algae)
- Inconsistencies in identification level between laboratories and projects, or the impossibility to identify to a lower taxonomical level (broken animals, no adequate identification keys); this in particular can affect classification tools as it can give a false impression of species richness in a sample. Therefore, following rules are applied:

Grouping to the highest taxonomical level present in the dataset. If some individuals within a certain taxonomical group are not identified to species level (e.g. if *Eteone sp., Eteone longa* and *Eteone flava* are present in a dataset, they have to be grouped to *Eteone*).

  - o Some exceptions to this rule:
    - ▪ When the number of samples, in which the higher taxonomical level is found is lower than 2, that taxonomical level is excluded.
    - ▪ The taxonomical inconsistence is conserved when the loss of detailed information would be too high (e.g. *Nepthys*, *Nereis*).

- Removal of non-soft sediment taxa (hyper- and epibenthic taxa). This again has an impact on the richness of a sample.
- New taxonomical developments can lead to a more accurate identification compared to \`historical\` datasets. Therefore, some species are grouped to a higher taxonomical level (e.g. the distinction between *Magelona papilicornis, M. mirabilis, and M. johnstoni* was not consistent throughout the years).
- Some taxonomic groups have to be combined to the level specified to negate the problem of inconsistent levels of identification: Oligochaeta, Nemertea, Echiura, Sipuncula, Phoronida, Priapulida, Holothuroidea, Archiannelida.

The application of these rules made the datasets for the reference and assessment analysis more consistent. The used taxa (species) per water body are summarized in the appendix under the section species lists, because differences in species truncation between the water bodies exist.

4.1.3 Primary production data

At the level of the water body a functional evaluation is used based on the relation between system-averaged macrobenthic biomass and system-averaged primary production (pelagic and benthic). This relation is used to get a robust estimate of possible shifts in the ecosystem functioning of the water body. The B_{graz}:P_{prim} relation is used as parameter on the ecosystem level and boundaries are set in accordance to the WFD requirements.

Primary production is an important ecosystem attribute, providing the energy for ecosystem processes. Primary production fuels the food web. The BEQI method uses system primary production for the assessment at the first, ecosystem, level. Although primary production is of overriding importance in ecosystems, measurements are at present not included in the MWTL (BIOMON) program. Available data are therefore scarce and Table 6 gives an overview of the available primary production data used in this study.

Table 6. Overview per water body of the available primary production data used in this study.

waterbody	primary production gC/m2 year	period	origin
Dutch coast			
ZK + NDK	520-680	2000	modeled (REFCOAST)
HK, WK	233	1988-1989	measured
WK	345	1998	modeled (Blauw & Los, 2004)
HK	227	1988	modeled (Blauw & Los, 2004)
			no recent estimates
Oosterschelde	241	2003	modeled value (pers com Jacco kromkamp)
	see ref for values	1990-2000	Geurt van Kessel et al., 2003
			no recent estimates
Waddensea	98,05	1974	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
	92,96	1975	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
	275,5	1986	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
	212,39	1990	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
	238,86	1991	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
	207,64	1992	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
	166,25	1993	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
	304,34	1994	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
	159,46	1995	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
	149,29	1996	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
	204,25	1997	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
	214,43	1998	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
	128,59	1999	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
	134,7	2000	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
	120,45	2001	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
	165,91	2002	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
	177,79	2003	measured at the NIOZ jetty (Cadée and Hegeman, 2002)
Westerschelde	100		Herman et al., 1999
	150		estimated used in BEQI report
	298 (bruto prim prod)	2001	Kromkamp & Peene, 2005
			no recent estimates
Eems-Dollard	200	years '70	Colijn, 1983
			no recent estimates
Lake Veere	229-377	1980-1983	Wattel, 1984
	300		Nienhuis, 1992; Herman et al., 1999
			no recent estimates
Grevelingen	320		pelagic prod + benthic prod + sea grasses and macro algae
			Nienhuis, 1992
	300		Herman et al., 1999
			no recent estimates

4.2 Coastal waters: open polyhaline and euhaline

The entire coastal area is classified as a natural water body. Therefore, the WFD requires determining the high and good ecological status for these water bodies.

4.2.1 Short description

All coastal open polyhaline and euhaline water bodies receive fresh water river outflow (e.g. Schelde, Rhine). The WFD defines the coastal zone as the zone between the beach and 1 nautical mile off the coast (Van Splunder et al., 2006). In this zone hardly any information is available (AquaSense, 2003). This means that at this moment, the beach and adjacent surf zone are not included in the evaluation of the coast, despite the fact that this zone is characterized by its specific benthic community (Van Hoey et al., 2004) and also has specific human pressures (e.g. beach nourishments). Therefore, the coastal zone, for this study, is extended to 6 nautical miles off the coastline, wherein the BIOMON coastal monitoring points are located.

4.2.2 Human pressures and environmental problems

The main anthropogenic pressures on the benthos in the coastal zone are chemical and mechanical disturbances.

The occurrence of heavy metals is strongly reduced since the 80`s and a few micropollutants cause problems. Eutrophication plays a role as well, phosphor reached the highest levels in the 80`s and is reduced later on, whereas the nitrogen concentrations are less reduced (Figure 13). Therefore the N:P ratio is shifted towards nitrogen. Philippart et al. (2007) showed for the Wadden Sea that long-term variations in limiting nutrients (phosphate and silicon) are weakly correlated with biomass and more strongly with community structures of phytoplankton, macrobenthos and estuarine birds.

Mechanical disturbance is caused by fishery and sand suppletions for the coastal defenses. The coastal fishery mainly exists of shrimp and beam trawl fishery. The shrimp fishery appears to have only minimal effect on the macrobenthos (Rijnsdorp et al., 2006). The beam trawl fishery with Eurokotters is mainly situated in the Hollandse kust (Figure 14) and has a strong effect on the macrobenthos (Rijnsdorp et al., 2006). The intensity of the fishery activities remains more or less stable.

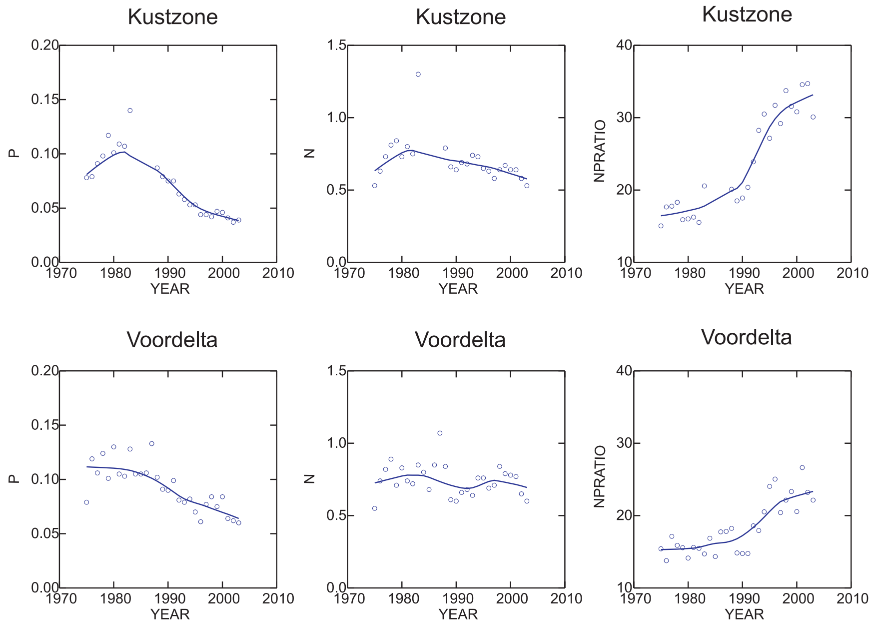
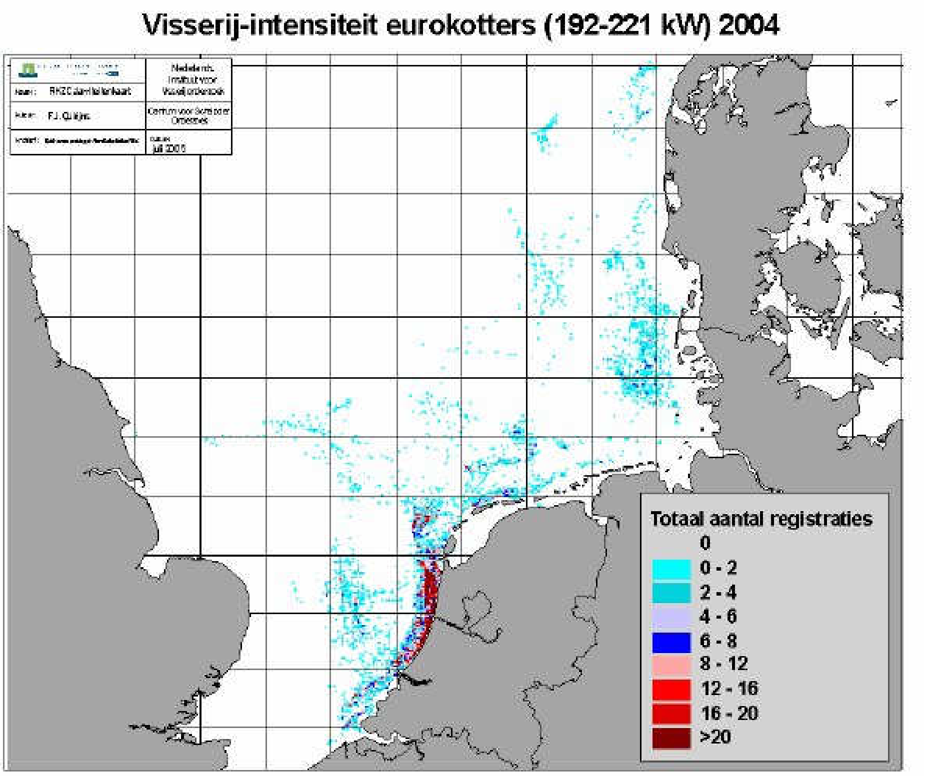


Figure 13. Time series of concentrations (in mg l^{-1}) of Phosphorous (left) and Nitrogen (central) and the N-to-P ratio (right) in the northern part of the coastal zone (called Kustzone) and the southern part (Voordelta). Data from www.waterstat.nl



Figuur 1. Visserij-intensiteit van eurokotters (192-221 kW) in 2004 op basis van VMS-gegevens. Totaal aantal registraties per vakje van 1 bij 2 zeemijl.

Figure 14. Fishery intensity in the North Sea (Steenbergen et al., 2005b)

Trawling reduces biomass production and species richness of the benthic fauna. Estimates for the North Sea by model calculations are a 56% reduction in biomass and a 21% reduction in production due to trawling. Effects are largest in already disturbed habitats in comparison with less disturbed habitats (Hiddink et al., 2006). Also the functional structure of the macrobenthic community is affected by beam trawl disturbance, areas with intensive fishing are characterized by a higher relative biomass of mobile animals and infaunal and scavenging invertebrates, compared to lightly disturbed areas which are richer in filter feeders and larger animals (Tillin et al., 2006). In the Dutch coastal zone a reduction of the biomass of the benthos is estimated on 10-20% by beam-trawl fishery, whereas the shrimp fishery leads to a reduction of less than 3% (Rijnsdorp et al., 2006). However, more research is needed to fully understand the effect of beam trawl fishery on the macrobenthos.

Coastal defenses with sand suppletions occur mainly on the beaches until the end of the 90`s (Figure 15). After 1997, this way of working has partly changed in suppletion of the sub-tidal zone in front of the beaches. The benthic community in the area are sand is deposited is strongly affected, most species do not survive. After suppletion high densities of opportunistic species like *Scolelepis squamata* may develop. Juveniles settling in the disturbed area reestablish the community. Recovery takes about five years (Essink, 2005).

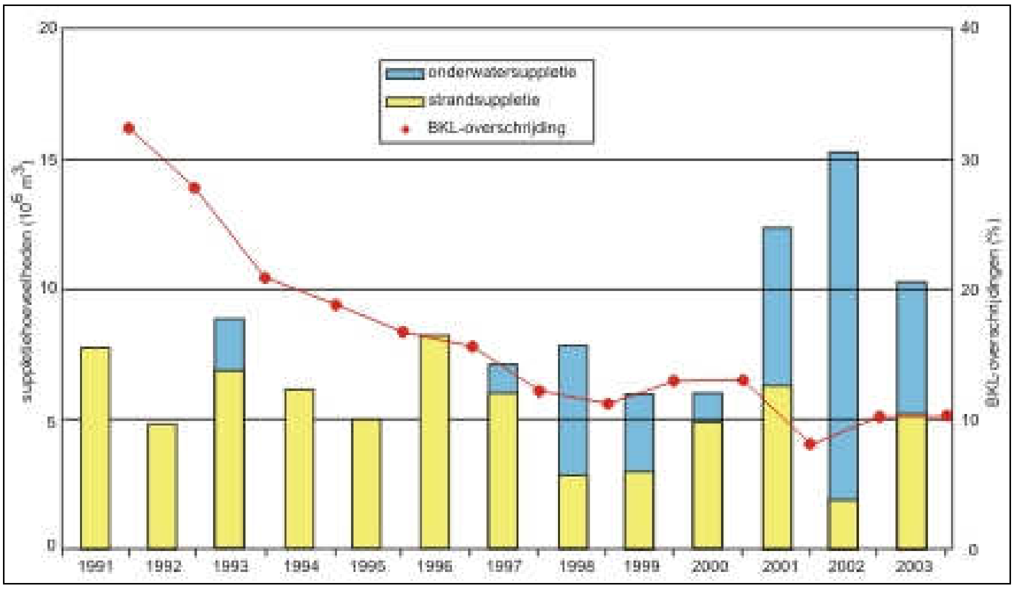


Figure 15. http://www.rikz.nl/thema/kust_en_veiligheid/Beheer/zandsuppleties.html

Main pressure Dutch open coast: fishery

4.2.3 Habitat typology

4.2.3.1 Habitat classification

The composition of the macrobenthic community depends on a multitude of factors. In the first place the physical environment determines the likelihood of occurrence of species. Differences in the physical environment like grain size of the sediment, current velocity, temperature or water depth will have an effect on the species composition of the macrobenthic communities. Knowledge of the relationship between the macrobenthic communities and the physical environment gives us some predictive power for what type of community to expect under certain conditions (Degraer et al., 2002, Van Hoey et al., 2004). It is this relationship that is fully utilized to determine the benthic habitats in the Dutch coastal zone.
To come to a good description of the relationship between the physical environment and the macrobenthic community two approaches can be used. In one approach, the spatial variation in species composition is statistically organized in clusters of similar species composition. It is then assumed that at least a part of the similarity in species composition between different sites is caused by a similarity in the physical environment. In the other approach, physical environmental variables important for species composition are mapped. By combining several physical variables, habitats are described and mapped (Bouma et al. 2005). Due to the lack of a fully developed habitat map (physical and biological information combined) of the coastal zone (within 6 nautical mile), the first approach is used to determine the benthic habitats in the Dutch coastal zone. This benthic community characterization is compared with the habitats of the habitat map of the North Sea (Figure 20).

To describe the variation in species composition of the macrobenthos of the Dutch coastal zone, an extensive dataset combined from several coastal macrobenthos monitoring programs was used. From the Dutch sector of the North Sea, 5259 samples with information on density and biomass of macrobenthic species are available. These are collected in 24 years spread over the period from 1962 until 2004. The sampling effort was not evenly distributed over space and time. Most samples are from the Delta coast in the southwest of the Netherlands, with large spatial and temporal sampling variability. The other part of the Dutch coast is less intensively sampled, with only one spatial covering campaign in 1988 - 1989 (MILZON-project). To get a data set, which covers the temporal and spatial variability, for determining the biological communities and reference conditions, the data of the period 1983 - 1990 was used.
To include at least all the 15 Coastal assessment stations of the BIOMON macrobenthic monitoring program, a coastal zone with a width of six nautical miles was defined (WFD requires only one mile). During the period from 1983 until 1990, 1805 samples (of which 1479 are situated

in the Delta coast) are collected along the Dutch coast, analyzed, and the results stored in the database.

4.2.3.1.1 Cluster analysis

The analyses of similarity between stations are performed on a subset of the total species pool. Species are only included if they are found at least in 5% of the samples or if they represented at least 20% of the numerical density or biomass in one of the samples. From 271 species found in the samples over the years, 125 are included according to the criteria. Only data on the numerical density are used for the analyses. Per species the density per square meter was ln(x+1) transformed. In the multivariate analyses, every station was treated as a sample and every species as a variable. For the similarity analyses, the computer program PRIMER for windows version 5.2.9 was used.

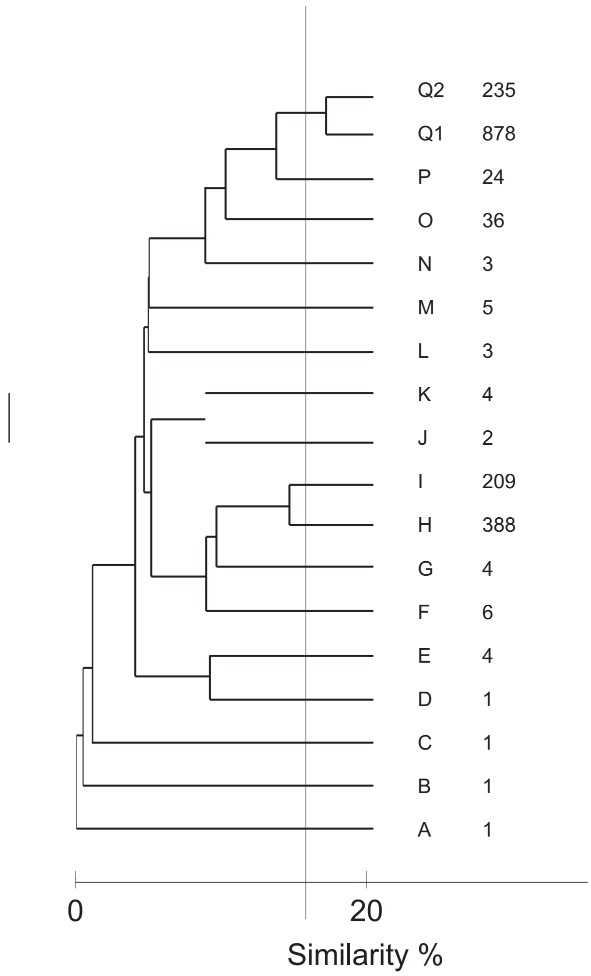


Figure 16. Cluster diagram/dendrogram of a similarity analyses of the macrobenthos communities in the North Sea Dutch coastal zone. Number of stations in each cluster is mentioned behind the cluster names

Similarity of the stations was analyzed in a Bray-Curtis similarity matrix. With a Hierarchical Cluster analysis on the similarity matrix of the samples, a cluster diagram was made (Figure 16). Figure 16 only shows the diagram up to a similarity of 16 %. An additional node is shown for Q dividing this very large cluster in Q1 and Q2. With this method six clusters (see Table 7) are defined representing more than 10 sampling stations. The other clusters only contain one or a few stations and can be considered as outliers.

Table 7. The species composition of the 6 largest clusters. The contribution of a species is the average percentage per species of the total numerical density of all species in the group. Only the cumulative contribution up to 70% is shown

Group	**Q1**		**Group**	**Q2**	
Average	similarity:	31.45	Average	similarity:	26.78
Species	Contrib%	Cum.%	Species	Contrib%	Cum.%
Nephtys cirrosa	10.83	10.83	*Nephtys cirrosa*	40.83	40.83
Spiophanes bombyx	10.74	21.57	*Pontocrates altamarinus*	11.52	52.35
Scoloplos armiger	9.83	31.4	*Bathyporeia elegans*	10.65	63
Nephtys hombergii	8.55	39.95	*Bathyporeia*	5.89	68.89
Spio filicornis	8.05	48	*Spio filicornis*	5.25	74.14
Urothoe poseidonis	6.35	54.35			
Magelona mirabilis	4.41	58.76			
Tellina fabula	3.86	62.62			
NEMERTEA	3.69	66.31			
Spisula subtruncata	3.57	69.88			
Capitella	3.12	73			
Group	**H**		**Group**	**I**	
Average	similarity:	51.4	Average	similarity:	28.23
Species	Contrib%	Cum.%	Species	Contrib%	Cum.%
Nereis diversicolor	26.76	26.76	*Heteromastus filiformis*	37.55	37.55
Corophium volutator	18.63	45.39	*Aphelochaeta marioni*	11.94	49.49
Pygospio elegans	18.21	63.59	*Pygospio elegans*	10.77	60.26
Hydrobia ulvae	12.67	76.27	*Macoma balthica*	7.17	67.43
			Cerastoderma	6.93	74.36
Group	**O**		**Group**	**P**	
Average	similarity:	32.1	Average	similarity:	30.57
Species	Contrib%	Cum.%	Species	Contrib%	Cum.%
Nephtys hombergii	67.91	67.91	*Scoloplos armiger*	72.77	72.77
Macoma balthica	21.24	89.15			

The species composition of the main clusters is presented in Table 7. In both Q clusters, the polychaete *Nephtys cirrosa* is the numerically most dominant species, but the species has a larger contribution in Q2. Besides *Spio filicornis,* all the other species are different between the groups. In group I and O, bivalves have a relatively strong contribution. In group P, *Scoloplos armiger* by itself makes up more than 70% of the density.

Table 8 shows the results of a pair wise comparison between the six most important clusters. Every comparison is significantly different. It is concluded that the clusters defined are different from each other (high R values).

Table 8. Results of pairwise tests of similarity between the main clusters

Groups		R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
Q1	H	0.958	0.1	Too Many	999	0
Q1	I	0.796	0.1	Too Many	999	0
Q1	Q2	0.561	0.1	Too Many	999	0
Q1	O	0.779	0.1	Too Many	999	0
Q1	P	0.655	0.1	Too Many	999	0
H	I	0.764	0.1	Too Many	999	0
H	Q2	0.947	0.1	Too Many	999	0
H	O	0.945	0.1	Too Many	999	0
H	P	0.955	0.1	Too Many	999	0
I	Q2	0.796	0.1	Too Many	999	0
I	O	0.652	0.1	Too Many	999	0
I	P	0.69	0.1	Too Many	999	0
N	Q2	0.799	0.1	2218636	999	0
N	O	0.811	0.1	9139	999	0
N	P	0.686	0.2	2925	999	1
Q2	O	0.791	0.1	Too Many	999	0
Q2	P	0.716	0.1	Too Many	999	0
O	P	0.595	0.1	Too Many	999	0

The spatial arrangement of the clusters is given in Figure 17 and Figure 18. The largest part of the area falls within the cluster Q1. Most variation is found in the Delta area. At the mouth of the Haringvliet, a separate cluster I and H is found. At the Westerschelde entrance, the benthic community is very dissimilar. The stations along the rest of the Dutch coast mainly belong to the Q1 cluster. Along the coast of Texel and in the north of the Hollandse kust, several Q2 cells are found. This pattern is similar but not the same as results from a twinspan analysis in Holtmann et al. (1996).

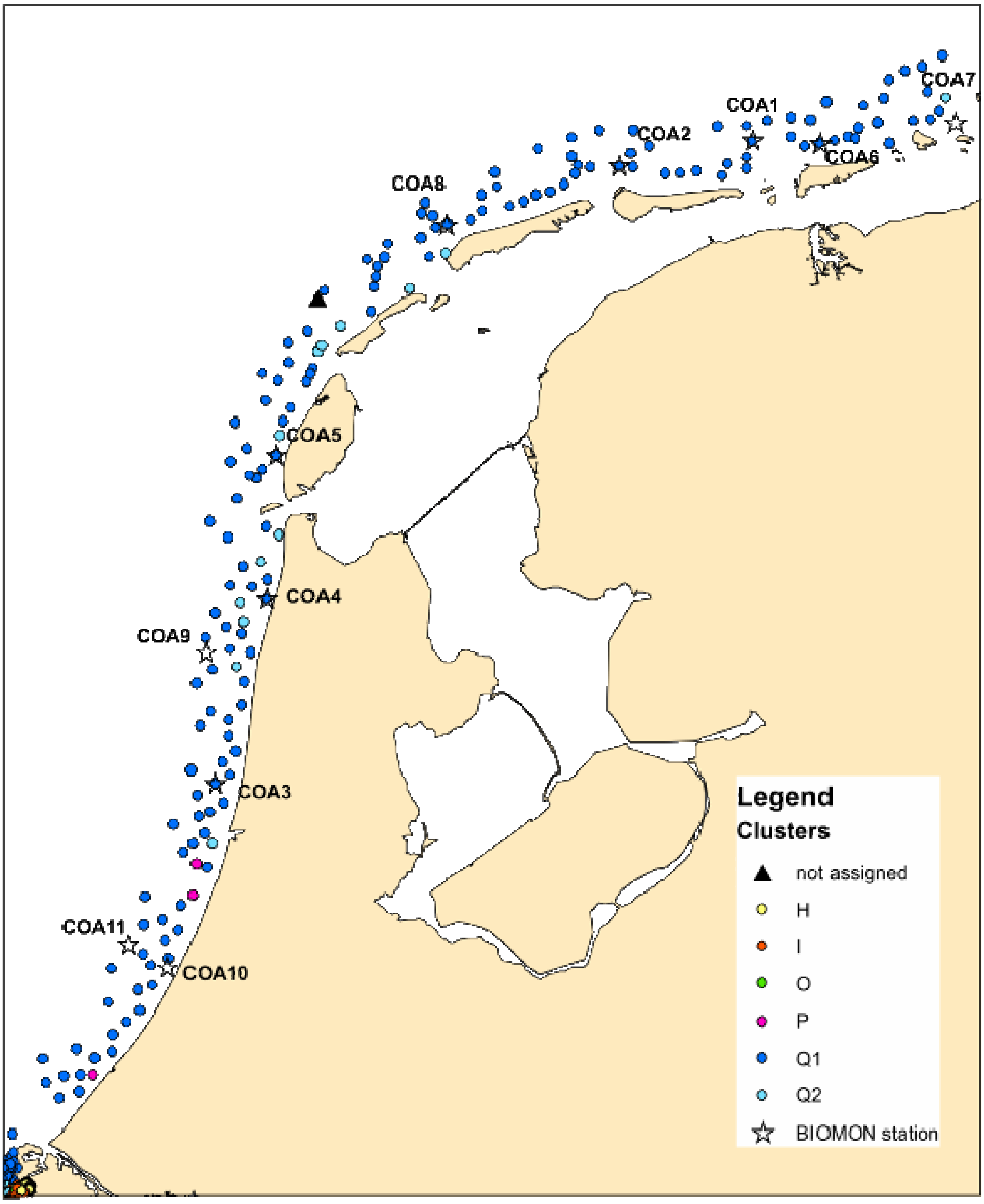


Figure 17. Assignment of stations to clusters for the northern part of the Dutch coast, Hollandse Kust, Waddenkust and Eems-Dollard kust

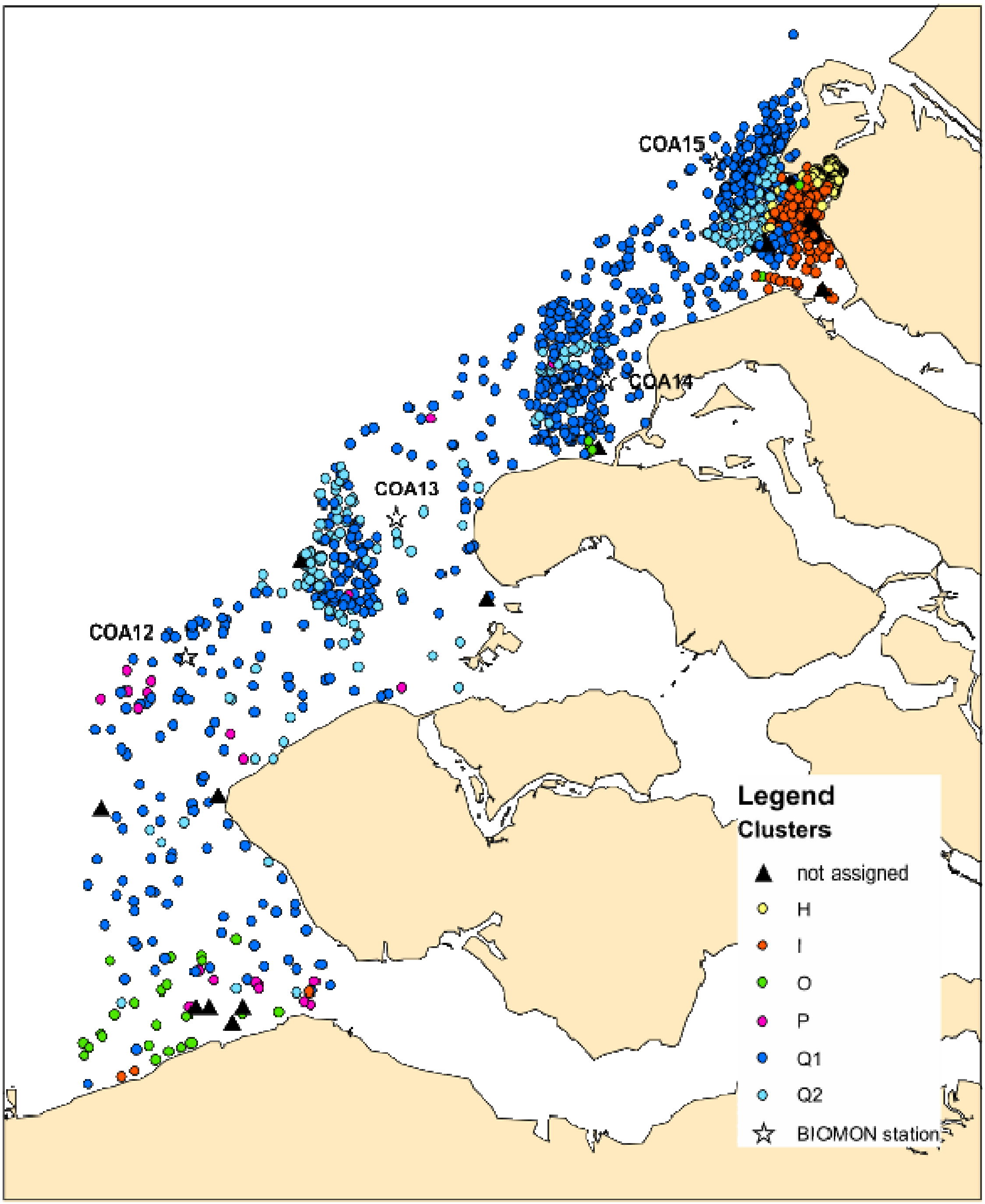


Figure 18. Assignment of stations to clusters for the southern part of the Dutch coast, Noordelijke Deltakust and Zeeuwse kust

4.2.3.1.2 Comparison with TWINSPAN

For comparison the same numerical density data ln(x+1) transformed are analyzed with a Two-Way Indicator Species Analysis (TWINSPAN) (Hill, 1979). The analyses was done with the WinTWINS program version 2.3 available at http://www.ceh.ac.uk/products/software/wintwins.html. Four default cut levels are used, 0, 2, 5 and 10 ln (n+1) m^{-2}. Two division levels are used with a minimum group size for division of 10 samples. The result of the analysis in this configuration is four groups. Every division yields a variable number (max 7) of indicator species which are important contributors to the contrasts between the divided groups (Figure 19).

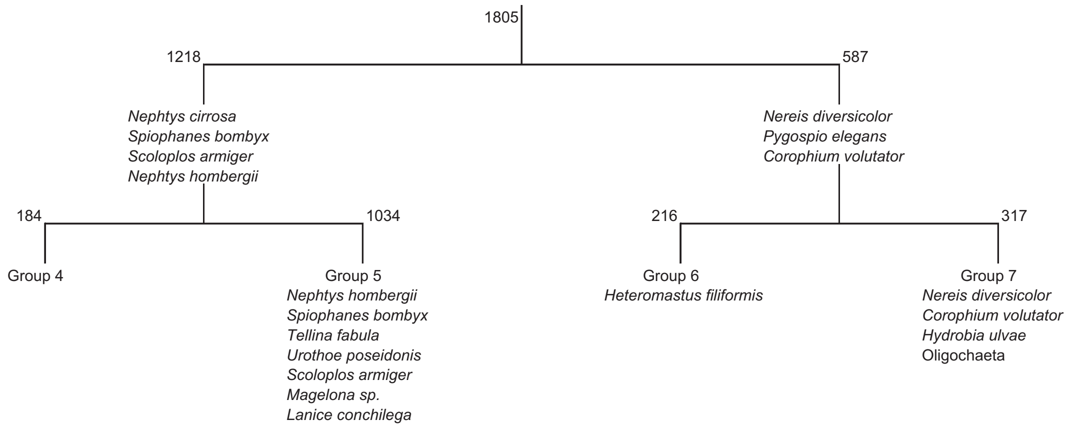


Figure 19. Dendrogram of a TWINSPAN analysis on macrobenthos data of 1805 stations in Dutch coastal waters up to six nautical miles from the coast. Group sizes are indicated as well as indicator species

The results of the two multivariate analyses are compared in Table 9. There is a reasonable correspondence between the two results. Group 4 overlaps with cluster Q2, group 5 with clusters Q2, O and P, group 6 with cluster I and group 7 with cluster H.

Based on the most abundant species in the clusters (Table 7) and the resulting indicator species from the TWINSPAN analysis (Figure 19) some likely habitat differences can be predicted for the assemblages. The first division in the TWINSPAN is between assemblages with marine and brackish indicator species. *Nereis diversicolor* and *Corophium volutator* are species that reach their highest chance of occurrence at salinities below 10 psu. *Pygospio elegans* occurrence peaks at salinities around 20 psu (Ysebaert et al. 2002). Stations belonging to the clusters H and I, (groups 7 and 6) are located in the area with fresh water outflow from the Haringvliet (Figure 18). The further division in the brackish group may be partly due to differences in sediment composition. Especially *Nereis diversicolor* and *Corophium volutator* are found in fine sediments (Ysebaert et al. 2002). The area of cluster H (group 7) is expected to have finer sediments than the area of cluster I (group 6). The division of the marine group in groups 4 and 5 has indicators for group 5 which are found in relatively fine sediments compared to the main species in group 4, which are given in Table 7 (cluster Q2). Based on the species composition of the defined groups or clusters, the factors salinity and sediment composition are expected to be useful for the physical habitat characterization.

Table 9. Comparison of the Hierarchical Cluster analysis and the TWINSPAN analysis. The number of stations for every combination of cluster and TWINSPAN group are given

TWIN group / Cluster group	4	5	6	7	total
A			1		1
B			1		1
C				1	1
D	1				1
E		3	1		4
F		1	5		6
G		1		3	4
H			**22**	**366**	388
I	**3**	**29**	**176**	**1**	209
J			2		2
K		2	2		4
L		3			3
M	3	1	1		5
N		3			3
O	**1**	**32**	**3**		36
P	**1**	**23**			24
Q1	**26**	**851**	**2**		879
Q2	**149**	**86**			235
Total	184	1035	216	371	

4.2.3.1.3 Comparison with the habitat map of the North Sea

The habitat map of the North Sea was already produced and made available on the web at www.noordzeeatlas.nl (Figure 20). In the six mile coastal zone, four habitat types are described. These are based on sediment composition and depth. Two sediment types are distinguished: coarse sand (median grain size between 250 and 2000 µm) and fine sand (<250 µm). Depth categories are shallow (<20m) and relatively deep (between 20 and 30 m). Most sampling locations are in the shallow area, which forms the largest part of the six mile zone. The variation in species composition and abundance does not fit the environmental categorization in the coastal zone. The results of an analysis of similarity are given in Table 10. Only the difference between the coarse and fine habitats in the relatively deep part of the coastal zone turns out significant. From the data set, only 24 samples fall in these two areas.

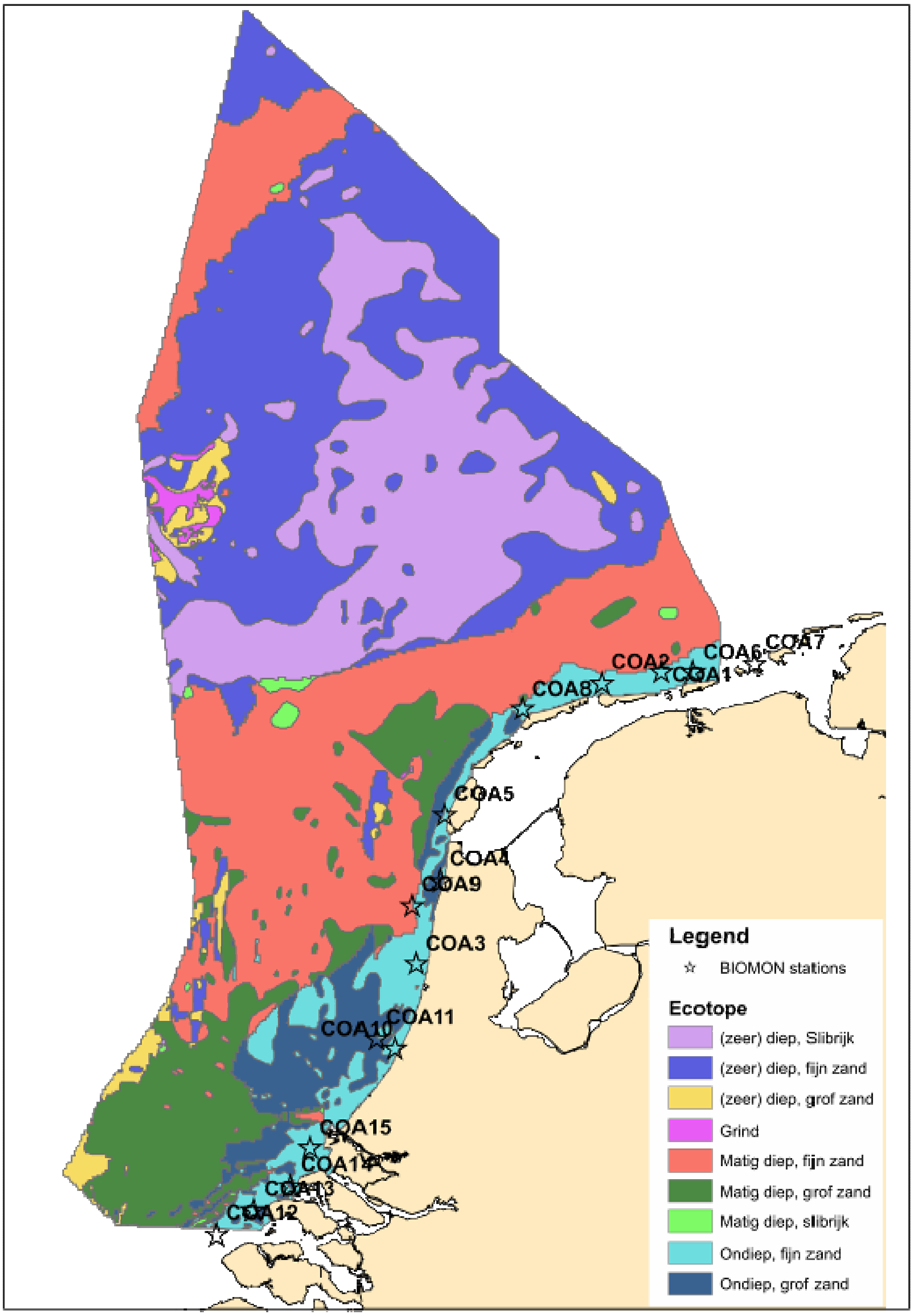


Figure 20. The habitat map for the Dutch continental shelf. Most of the coastal zone is either shallow with fine sand or shallow with coarse sand

Table 10. Results of an analysis of similarity between the habitats described for the Dutch coastal zone of six nautical miles. Densities of 125 species at 1559 stations are used for this analysis, 246 samples are excluded because no habitat was described at the sampling site

Groups	Compared	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
C	D	-0.062	100	Too Many	999	999
C	B	0.02	38	Too Many	999	379
C	A	-0.072	93.1	Too Many	999	930
D	B	-0.006	51	Too Many	999	509
D	A	-0.129	97.3	Too Many	999	972
B	A	0.45	0.1	1307504	999	0

A depth relatively deep (20-30 m) sediment fine (<250 µm) n = 15
B depth relatively deep (20-30 m) sediment coarse (250-2000 µm) n = 9
C depth shallow (< 20 m) sediment fine (<250 µm) n = 1379
D depth shallow (< 20 m) sediment coarse (250-2000 µm) n = 156

The habitats defined for the six nautical mile coastal zone of the North Sea do not match with the variation in species composition of the macrobenthos in our selected data. It is likely that the habitat map designed for the entire Dutch sector of the North Sea does not include enough detail to distinguish biologically relevant sub areas in shallow fine sediments. Another reason may be that the habitat in this area may be rather uniform, and that the variation in species composition is the result of other factors than the ones used to construct the habitat map.
With the environmental information available per sampling station, the biota are matched. Only depth and distance to the shore are available, which correlate with each other. Only applied to depth, it was calculated that the similarity between station matches with the depth variation with a correlation coefficient of 0.34. It is likely that sediment composition (median grain size, mud content) will also correlate with the biotic variation, possibly better than with depth. Choosing different boundaries for sediment categories may eventually lead to a better match between the defined habitats and the composition of the biological community.

4.2.3.1.4 Discerned habitats

Based on the cluster- and twinspan analysis the Dutch coastal area was characterized by five different benthic communities (habitats).

- Q1: muddy fine sand community
- Q2: sand community
- H: brackish muddy sand community (Haringvliet)
- I: brackish sand community (Haringvliet)
- O + P: impoverished Q1 community (mouth Westerschelde)

Of these communities only Q1 and Q2 are found along the entire Dutch coast, whereas the others are situated in specific areas in the Delta (Figure 17 and Figure 18).

Table 11. The number of samples and total sampling surface (m^2) of the soft-bottom habitats of the Zeeuwse kust and Noordelijke Deltakust. Ass ZK: Assessment for Zeeuwse kust and Ass. NDK: assessment for Noordelijke Deltakust. Nr, is the nummer of the community in the database

		Number of samples			Total sampling surface		
Community	Nr	Reference	Ass. ZK	Ass. NDK	Reference	Ass. ZK	Ass. NDK
Q1	**16**	**612**	**6**	**3**	**41.616**	**0.408**	**0.204**
Q2	17	200	0	0	13.6	0	0
H	7	387	0	0	26.316	0	0
I	8	202	0	0	13.736	0	0
O and P	14-15	46	0	0	3.128	0	0

Table 12. The number of samples and total sampling surface (m^2) of the soft-bottom habitats of the Hollandse kust(HK) Waddenkust (WK) and Eems-Dollard kust (EK). For these water bodies the same reference is used. Ass.: Assessment. Nr, is the nummer of the community in the database

		Number of samples				Total sampling surface			
Community	Nr	Reference	Ass. HK	Ass. WK	Ass. EK	Reference	Ass. HK	Ass. WK	Ass. EK
Q1	**16**	**200**	**15**	**15**	**3**	**13.6**	**1.02**	**1.02**	**0.18**
Q2	17	16				1.09			
O and P	14-15	3				0.204			

Table 11 and Table 12 gives an overview of the number of samples available for setting the reference conditions and doing the assessment. Only for habitat Q1 an evaluation can be made. All BIOMON assessment stations fall within cluster Q1, only for monitoring station COA13 is not defined to which cluster it belongs (Figure 17 and Figure 18). Assessment samples for the other habitats are not yet available and they have to be included in the monitoring in the future (extra effort needed).

4.2.4 Reference data/settings

The pressures discussed above make it difficult to define a 'natural, not impacted' reference period for the Dutch coast. For setting the reference conditions the Dutch coast is divided into two sub-areas: (1) the Zeeuwsekust and Noordelijke Deltakust (Delta area) and (2) the Hollandse kust, Waddenkust and Eems-Dollard kust. This is because the Delta area is expected to be slightly different compared to the rest of the Dutch coast. The benthic data from before 1990 are selected as reference data, because:

- This corresponds with the overall strategy of selecting a reference dataset (approximately the first one third of the years of the available data period).
- It is the only period for which a spatial dataset for the Hollandse kust and Waddenkust is available. The disadvantage is that for the Hollandse kust and Waddenkust almost no temporal variability is included in the reference settings, because these data is gathered between 1988 and 1989.

- For the Delta area, the reference setting is well characterized by a temporal and spatial dataset.
- A disadvantage is that the reference period is characterized by eutrophication, but the BEQI method will evaluate if the present situation is changed compared to the more eutrophic period.

For each reference area (Zeeuwse kust and Noordelijke Deltakust; Hollandse kust, Waddenkust- and Eems-Dollard kust) the reference values are calculated for each of the four parameters (number of species, density, biomass and species composition change) in relation to the surface area sampled. The reference distributions presented are for community Q1 (appendix).

4.2.5 Assessment

Assessment data (except for one station in the Zeeuwse Kust) all belonged to the community Q1 and therefore evaluation of other communities is for the moment not possible. The assessment is presented separately for the five different water bodies within the coastal area. An assessment is done for level 1 and level 3 of the BEQI method and level 2 is excluded at this moment. This is because the coastal area is a very open and dynamic system, where the natural sediment transport has a great influence on the habitat types. Where the natural hydrodynamics in the system are not impacted by human influences, the physical characteristics, as depth and sedimentology will not be influenced. Also no historical information on surface area of habitats is available and it is difficult to assign habitat changes to anthropogenic impacts (more investigations needed). Harbor constructions (e.g. Rotterdam) along the coast can change locally the habitat morphology, as also the exclusion of estuarine influences (e.g. Haringvliet) and the effect of fisheries on the abiotic environment, but this all need more investigations and historical information. Due to the lack of both, the habitat level is not included in the assessment classification of the coast for the moment.

The years that are selected for the assessment for the coastal area are the last three years available from the monitoring, i.e. 2002, 2003 and 2004.

4.2.5.1 Zeeuwse kust and Noordelijke Deltakust

4.2.5.1.1 Level 1: ecosystem

At this level, assessment values for system primary production and average macrofauna biomass is needed to construct the ratio $B_{benthos}/P_{prim}$ and estimate the score and status, based on the proposed boundary settings of Figure 3.

System primary production estimates for the Zeeuwse kust can be derived from the REFCOAST project (Van Damme et al., 2006), which modeled the primary production in the coastal zone nearby the mouth of the Westerschelde. The model estimates that the primary production for the year 2000 lays between 520 – 680 gC/m^2 year. Primary production estimates, based on values transformed from chlorophyll measurements shows a significant year-to-year variability in primary production (Van Damme et al., 2006). This means that more investigations are needed to define good standard primary production estimates.

The average macrofauna biomass of the assessment period (2002-2004) was estimated as 79.79 g AFDW/m^2 for habitat Q1. This value is based on a few samples, which are not representative for the entire Zeeuwse kust and also no current estimate for the other habitats was available. Due to the uncertainty in the primary production and biomass estimates, no ecological status score will be calculated, but based on the precaution principle set on good (EQR score: 0.7).

4.2.5.1.2 Level 3: community (within-habitat)

At this level the changes in species richness, species composition, density and biomass are evaluated for habitat Q1 for the water bodies Zeeuwse kust and Noordelijke Deltakust.

Due to the low assessment surface for both water bodies, the assessment analysis is not acceptable. The assessment of the Noordelijke Deltakust is only based on 3 samples from one location, whereas the assessment of the Zeeuwse kust is based on only 6 samples from two locations. The results of the assessment are showed in Table 14, but no conclusion on level 3 of the BEQI method could be made for those two water bodies, because the precision of the assessment is too low (a high risk of misclassification) (Table 13). An option is to take the score of a similar neighboring water body, which is in this case the Hollandse kust (overall EcoQ of 0.462) until a better assessment can be done (based on new sampling).

Table 13. The minimal and optimal sampling surfaces needed to get an acceptable assessment analysis for the Zeeuwse kust and Noordelijke Deltakust for habitat Q1

Habitat	Assessment surface	minimal surface	OK surface	optimal surface	Assessment precision class
Q1, Zeeuwse kust	0.41	0.95	1.50	4.56	unacceptable
Q1, Noordelijke Deltakust	0.204	0.952	1.496	4.56	unacceptable

Table 14. The assessment of level 3 for habitat Q1 for the different Dutch coastal water bodies, with indication of the assessment sampling surface, parameter values, the reference boundary values and finally the EQR score and status. The assessments with an acceptable sampling surface are set in grey

Marine habitats	parameter	Assessment surface	Assessment value	Reference boundary values: Poor min	Mod min	good min	high min	Median	high max	good max	Mod max	Poor max	Max spp.	EQR score	EQR status
habitat Q1	biomass	0.408	58	1	3	4	11	18	29	77	102	128		0.663	
Zeeuwse kust	density	0.408	1609	303	606	908	2513	4588	9261	28914	38551	48188		0.706	
	similarity	0.408	0.46	0.16	0.31	0.47	0.61							0.584	
	species	0.408	38	12	23	35	46						138	0.655	
average of parameters for Zeeuwse kust														0.652	
habitat Q1	biomass	0.204	102	1	1	2	7	15	30	114	152	191		0.620	
Noordelijke Deltakust	density	0.204	2753	135	271	406	1657	3702	8077	33215	44286	55356		0.922	
	similarity	0.204	0.42	0.11	0.22	0.33	0.50							0.708	
	species	0.204	40	7	14	21	33						138	0.813	
average of parameters for Noordelijke Deltakust														0.766	
habitat Q1	biomass	1.02	86	5	9	14	21	26	32	52	70	87		0.217	poor
Hollandse kust	density	1.02	1482	861	1723	2584	3797	4633	5650	7975	10633	13291		0.353	poor
	similarity	1.02	0.52	0.25	0.49	0.74	0.78							0.423	moderate
	species	1.02	72	20	40	60	68						124	0.813	high
average of parameters for Hollandse kust														0.452	moderate
habitat Q1	biomass	1.02	212	5	9	14	21	26	32	52	70	87		0.000	bad
Waddenkust	density	1.02	2667	861	1723	2584	3797	4633	5650	7975	10633	13291		0.615	good
	similarity	1.02	0.53	0.25	0.49	0.74	0.78							0.432	moderate
	species	1.02	53	20	40	60	68						124	0.530	moderate
average of parameters for Waddenkust														0.394	poor
habitat Q1	biomass	0.18	9	2	4	6	14	21	35	95	126	158		0.695	
Eemskust	density	0.18	620	347	694	1041	2456	3935	6331	13206	17608	22010		0.365	
	similarity	0.18	0.17	0.15	0.30	0.45	0.56							0.300	
	species	0.18	13	9	17	26	35						124	0.232	
average of parameters for Eems kust														0.398	

4.2.5.1.3 Integration of the different levels

For the overall assessment of the water bodies (Zeeuwse kust and Noordelijke Deltakust) the ecological score and status obtained for each of the three levels (level 2 not used for the evaluation of the coastal waters) are averaged into a metric representative for the whole water body. The averaging is done with a weighing factor for each level, 1 for the ecosystem level and 2 for the community level. The average of the two levels is 0.541, which corresponds with a moderate status for the Zeeuwse kust and Noordelijke Deltakust (Table 15). It is advisable to update these values by extra sampling in the following years to have a specific assessment for the habitats of the Zeeuwse kust and Noordelijke Deltakust.

Table 15. Ecological quality ratio score and *preliminary* status for the Zeeuwse kust and Noordelijke Deltakust, by using at level 3 the score and status of the Hollandse kust, due to the absence of an acceptable assessment

	EQR score	EQR status	Remark
Level 1: ecosystem	0.7	Good	Expert judgment
Level 3: community	0.462	Moderate	value of Hollandse kust
Overal EcoQ	0.541	Moderate	

4.2.5.2 Hollandse kust, Waddenkust and Eems-Dollard kust

4.2.5.2.1 Level 1: Ecosystem

At this level, assessment values for system primary production and average macrofauna biomass is needed to construct the ratio $B_{benthos}/P_{prim}$ and estimate the score and status, based on the proposed boundary settings of Figure 3.

Primary production estimates are available for the Hollandse Kust and the Waddenkust during the reference period in 1988 and 1989 (average of 233 g C m^{-2}) and there is a model estimate for 1998 (Blauw and Los, 2004)). The primary production model values for the Waddenkust and the Hollandse kust in 1998 are 345 g C m^{-2} and 227 g C m^{-2} respectively. Based on these estimates for the Waddenkust the biomass-to-primary production ratio was 1:5.75 (status good); for the Hollandse Kust the ratio was 1:4.25 (status moderate). But it must be noticed that these calculations are based on different sampling years for the primary production and the macrobenthos biomass. At this moment there are no clear indications that the coastal zone would be out of balance, so therefore it was decided to evaluate it for the moment as GEP (Good Ecological Status, average score 0.7). However, more research is needed to evaluate this relation more precisely, and if e.g. the strong increase of *Ensis directus* might have an effect on the ratio between primary production and macrobenthic biomass.

4.2.5.2.2 Level 3: Community (within-habitat)

Precision of the assessment of the Waddenkust and the Hollandse Kust are minimal, of the Eems-Dollard kust unacceptable (Table 16).

Waddenkust and Hollandse Kust are respectively qualified as poor and moderate (Table 14). There are large differences between the scores of the sub-metrics of the two water bodies. At both sites there is a trend for high biomass scores in combination with relative low density scores. The parameter number of species is declined in the Waddenkust, but not really changed at the Hollandse kust. In both areas the parameter similarity (reflection of species composition changes) indicates a change. The observed changes in the parameters at both sites have a common cause, the occurrence of the invasive species *Ensis directus*.

Eems-Dollard kust is only represented by one station that is visited three times during the assessment period of 2002 - 2004 and the results are not reliable because of a very limited precision with only three samples. An option is to take the score of a similar neighboring water body, which is in this case the Waddenkust (overall EQR of 0.394) until a better assessment can be done (based on new sampling). An extra effort is needed to make an assessment for the Eems-Dollard kust possible.

Table 16. The minimal and optimal sampling surfaces needed to get an acceptable assessment analysis for the Hollandse kust, Waddenkust and Eems-Dollard kust for habitat Q1.

Habitat	Assessment surface	minimal surface	OK surface	optimal surface	Assessment precision class
Q1, Hollandse kust	1.02	0.6	1.08	3.36	minimal
Q1, Waddenkust	1.02	0.6	1.08	3.36	minimal
Q1, Eemskust	0.18	0.6	1.08	3.36	unacceptable

4.2.5.2.3 Integration of the different levels

Table 17. Ecological quality score and status obtained by average the parameters at each level. The Eems-Dollard kust has get the same score as the Waddenkust, due to the absence of an acceptable assessment.

	Hollandse kust		Wadden- and Eemskust		
	EQR score	EQR status	EQR score	EQR status	Remark
Level 1: ecosystem	0.7	good	0.7	good	Expert judgment
Level 3: community	0.462	Moderate	0.394	Poor	
Overal EQR	0.541	Moderate	0.496	Moderate	

For the overall assessment of the water bodies (Hollandse kust , Waddenkust and Eems-Dollard kust) the ecological score and status obtained for each of the three levels (level 2 not used for the evaluation of the coastal waters) are averaged into a metric representative for the whole water body. The averaging is done with a weighing factor for each level, 1 for the ecosystem level and 2 for community level. The average of the two levels is 0.541 and 0.496 for respectively the

Hollandse kust and Waddenkust (Eems-Dollard kust, similar as Waddenkust), which corresponds with a moderate status for both (Table 17).

4.2.6 Discussion

Reference settings

The BEQI method, like all other assessment methods within the WFD, relies on a well-defined reference condition. Ideally, such a condition should be either before, or outside the influence of human activities. For the Dutch coast it was not possible to find such an unaffected area or data from before the influence of human activities. For the Voordelta, only quantitative and qualitative data are available from 1983 onwards, whereas for the rest of the Dutch coast this was available from 1988 onwards. Obviously, there was already human influence on the communities during those years. The Delta works, fisheries, euthrophication and pollution are some of these possible problems. The choice of reference conditions has therefore been very pragmatic. For the BEQI method is aimed at defining a reference period of a `historical` dataset that takes into account the natural spatial and temporal variability that characterizes shallow soft-sediment benthic communities of coastal and estuarine ecosystems. Within a certain habitat, a sufficient spatial coverage should account for the often patchy distribution of many benthic species and a multi-annual coverage should account for temporal effects such as severe winter effects.

For the Dutch coast the reference data set does not always fulfill even these minimal requirements. Spatial coverage is not too bad, but for the Hollandse Kust, Waddenkust en Eems-Dollard kust a good temporal coverage is lacking, as most data are from a single year. Some caution should be exerted therefore in interpreting the assessment results. However, as will be shown further, the major effect picked up by the index is the invasion by *Ensis directus*, and the existence or importance of this effect does not depend on the time coverage of the reference period. Adding 10 more years of data before the invasion would not have changed the essence of the assessment.

Assessment analysis

Besides a representative reference data set, every evaluation requires a representative assessment data set. The same requirements are needed, i.e. the assessment data set should contain a number of samples that sufficiently characterize (both spatially and temporally) the benthic community of a certain habitat within a certain water body. It is clear that the actual surveillance monitoring of the Dutch coast – the BIOMON sampling stations – fail to fulfill the above mentioned requirements. First of all, the BIOMON stations do not cover all habitats. Secondly, the spatial coverage for most water bodies is very poor, with in some cases only one fixed sampling station being surveyed every year. Therefore, the evaluation should be handled with care as explained below and it is advisable to improve the coastal sampling effort in the future. Compared to the relatively weak temporal coverage by the reference, we think that the limited spatial coverage by the assessment data base is potentially a bigger problem.

The BEQI method has evaluated the state in 2002 - 2004 of the different areas of the Dutch coast compared to the determined reference situation (state in 1983 - 1990). Only for the Hollandse kust and Waddenkust an assessment could be done, because the other areas have no sufficient assessment data to result into a powerful assessment. An explanation for these results will be given in this section.

For the Zeeuwse kust, Noordelijke Deltakust and Eems-Dollard kust an extra monitoring effort is needed to draw up a BEQI based ecological status classification..

The poor and moderate classifications of respectively the Waddenkust and Hollandse kust reflect that there are large and significant changes in the coastal ecosystem. This assessment result needs further analysis in order to understand the causes for these large changes. From a multivariate analysis, it appears that the dominance of the invasive species *Ensis directus* is very likely responsible for the changes in the species composition (Figure 21). The species stands widely apart from the rest of the community, and seems to be the major cause for the observation that the samples in the assessment period fall well outside the range occupied during the reference period. It also appears from this graph that the presence of *Ensis directus* is correlated with that of some accessory species, and thus that the invasion has also influenced the rest of the species composition.

The assessment analysis has furthermore shown that the change in species composition is accompanied by a significant change in biomass. It seems likely, then, that there may also be changes in the ecological functioning of the benthic community.

Because the invasion by one dominant species seems to be the essence of the community changes recorded, it is unlikely that the assessment results are determined by the choice of the reference. If a more pristine reference had been used, the results would have been the same because it is impossible that the invasive species *Ensis directus* would have had an equal dominance in the reference samples compared to the assessment samples.

Based on the assessment results by the BEQI method, a classification has been obtained that calls for closer investigation of the changes in the coastal zone. This investigation will have to answer a number of questions. First, since the number of assessment samples is small, a better spatial image of the distribution of *Ensis* will be needed. This will also allow to assess whether the increase in biomass in the available samples is representative for the whole coastal zone. Second, consequences for the rest of the ecosystem will have to be investigated. The former *Spisula* banks, which seem to have been replaced largely by *Ensis*, are an important food item for birds (ducks). An appraisal of the food value of the new community for higher trophic levels will be needed. Third, a central question is whether this invasion and the ensuing changes in community can be influenced by management. In particular, it should be investigated whether certain human pressures (fisheries, changed nutrient ratios, dredging,..) have influenced the success of the invader or not. Finally, this investigation can be the basis on which to decide whether a new reference should be made. If (a) management has (had) no influence on the invasion or (b) there is no marked negative influence on the rest of the ecosystem due to the invasion, a logical decision would be to change the reference to a situation including the invader. If however none of these conditions is fulfilled, appropriate management measures should be found to bring the system back to previous (better) conditions. None of these questions can be fully answered at the present day. However, the important point to note is that the BEQI assessment has successfully picked up a major change in the benthic community, and provides a good basis for closer study that could lead to changes in management.

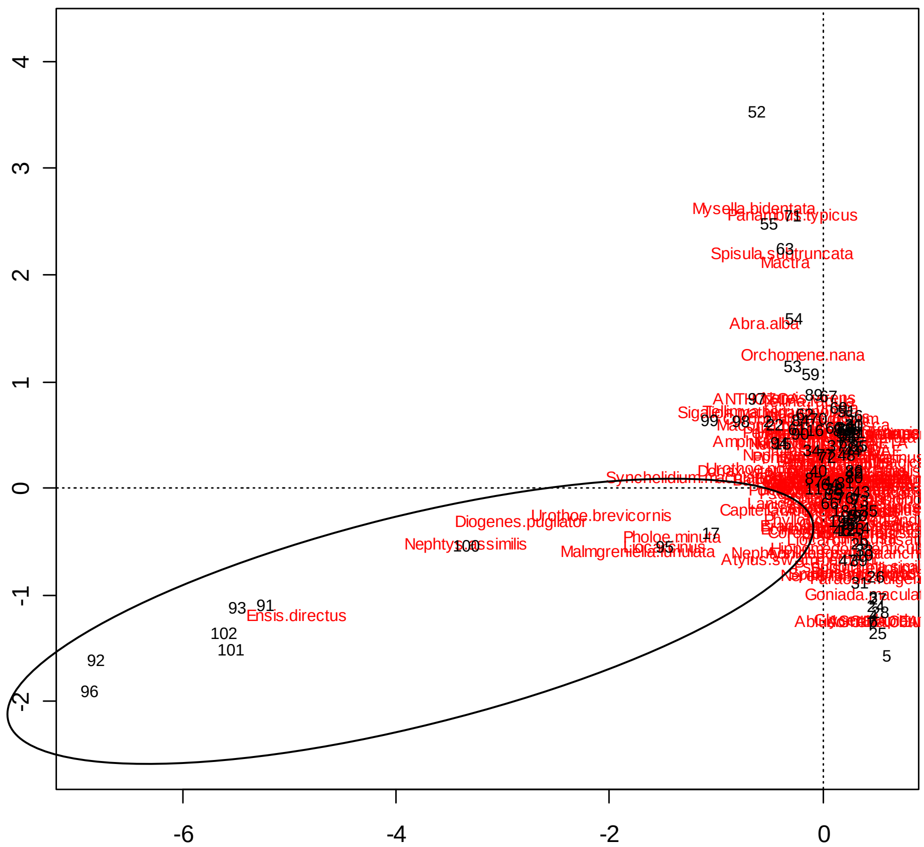


Figure 21. A CA (correspondence analysis) plot of species and samples of the reference (black) and assessment (encircled) dataset of the Waddenkust. X-as: CA1 and y-as: CA2

4.2.6.1 Conclusion

Dutch coast

- Pressure: fishery
- Evaluation:
  - Level 1: Good (for all water bodies) (based on expert judgement)
  - Level 2: not yet developed and included in the assessment for coastal waters
  - Level 3: Moderate for Hollandse kust and poor for Waddenkust and could not be done for the other coastal water bodies due to a low amount of assessment data. In the mean time, the ecological status of neighboring water bodies is copied.
- A better surveillance monitoring is needed to get an acceptable assessment for the different coastal habitats and water bodies.
- The obtained results are mainly caused by the dominance of the invasive species *Ensis directus*. Further investigations on the effect of *Ensis directus* on the ecosystem functioning of the coast are advisable.

4.3 Coastal waters: sheltered polyhaline

4.3.1 Oosterschelde

The Oosterschelde is characterized as a heavily modified water body. Therefore the WFD requires naming the high and good status as respectively the maximal (MEP) and the good ecological potential (GEP).

After a description of the area, the human pressures will be summarized, followed by the habitat typology. In a following section, the reference settings for the different levels of the BEQI method will be explained, followed by an assessment for the period 2003 - 2005 for the Oosterschelde.

4.3.1.1 Short description

Figure 22. Oosterschelde, with indication of the different sub-areas and the in the text mentioned places (Google Earth)

The Oosterschelde is characterized as a sheltered polyhaline water body (for some characteristics see Table 18).

After the flood disaster of 1953, which triggered the start of the Delta plan, the Oosterschelde has been seriously modified. Firstly, Lake Veere was disconnected by building the Zandkreekdam (1960), followed by the Grevelingen (1960-1964) and Volkerak (1969). Later on (1976), it was decided to build a storm surge barrier in the mouth of the Oosterschelde (instead of closing off completely the Oosterschelde), which was finished in 1986. The barrier is closed if the water level exceeded a critical value and would otherwise be open, thus allowing the tide to go in and out most of the time. To reduce the volume of the Oosterschelde and to hold a maximal tidal difference, the Oesterdam (1986) and Philipsdam (1987) are built, which made the Delta plan for the Oosterschelde complete. The characteristics of the Oosterschelde are changed as demonstrated in Table 18 and therefore the Oosterschelde is characterized as a heavily modified water body.

Table 18. Characteristics of the Oosterschelde (Geurts van Kessel, 2004)

	Before the storm surge barrier	After the storm surge barrier
Total surface	452 km^2	351 km^2
Water surface	362 km^2	304 km^2
Intertidal area	183 km^2	118 km^2
Average tide	3.70 m	3.25 m
Maximum current speed	1.5 m/s	1.0 m/s
Residence time	5-25days (west)	10-50 days (west)
	75->100 days (east)	150–>200 days (east)
Tide volume	1230 million m^3	880 million m^3
Fresh water input by rivers	50-100 million m^3/s	10 million m^3/s
Salinity	16.9 (west)	17.1 (west)
	15.4 (east)	16.7 (east)

The Oosterschelde has a large intertidal area, a nature area of international importance and therefore assigned as a national park. The Oosterschelde is protected under the EU Bird and Habitat Directive and the Ramsar convention.

4.3.1.2 Human pressures and environmental problems

The hydro- and geomorphological structure of the Oosterschelde changed drastically by the construction of the storm surge barrier and the compartmentalization dams. Due to the small chance that those constructions will be removed in the future, it is advisable to optimize the construction so that the previous situation is approached. Presently, there are some problems caused by this construction.

“Zandhonger”

After the construction of the storm surge barrier the hydro-morphological equilibrium was drastically disturbed, because tidal amplitude and tidal prism strongly decreased. The ebb and flood gullies are too broad for the volume of water that was entering through the storm surge barrier. The reduction of the tidal amplitude in the Oosterschelde has led to a significant reduction of the flow velocities, especially in the deeper channels (Geurts van Kessel et al., 2003). In a morphologically active tidal basin, this must lead to a morphological response. The gullies are slowly filling up until a new equilibrium in the morphological system will be reached. This “Zandhonger” (sand hunger) of the Oosterschelde is estimated at about 400-600 million m³ sand. The import of sand from the sea is limited because of the storm surge barrier. Because of this there is a redistribution of the sand present in the system: the sand stored on the sand flats, mudflats and salt marshes is transported slowly to the gullies. As a consequence the intertidal area is declining and disappearing under water (Geurts van Kessel, 2004). When the habitat map of 2001 is compared with earlier maps, the following changes are visible: the surface area of low intertidal area is increasing, the high-intertidal is disappearing, and the mid-littoral is decreasing (Figure 27).

Crassostrea gigas

Another major environmental problem is caused by the spreading of the Japanese oyster *Crassostrea gigas*, which is an introduced ‘invasive’ species and which nowadays occupies large parts of the intertidal and subtidal areas. This has its consequence for other naturally occurring bivalves (*Ostrea*, mussel, cockle), which are declining (Geurts van Kessel et al., 2003) and large suitable sites for these species are now occupied by these Japanese oysters. The Japanese Oysters are considered as a feeding competitor of the other bivalve species as they filter a large part of the available phytoplankton; they possibly also filter out the larvae of other bivalve species. There are indications that the Japanese Oyster is also responsible for a change in the phytoplankton composition of the Oosterschelde (Geurts van Kessel et al., 2003). Whereas cockles and mussels are important food sources for birds, the link to the higher trophic levels is rather limited for oysters. More research is needed to quantify and predict the effect of the Japanese Oyster on the ecosystem of the Oosterschelde.

Euthrophication

The problem of euthrophication before the construction of the storm surge barrier has now disappeared, due to the reduction of nutrient-rich fresh water input (Wetsteyn et al., 2003). The concentration of nitrogen and silicate is slightly increasing again in the period 1990 – 2000. At this moment the decrease of nutrients has no or very limited effect on the phytoplankton growth (Wetsteyn et al., 2003, Geurts van Kessel et al., 2003). However, the primary production in the Oosterschelde decreased in the period 1990-2000 and is significantly related to an increase in turbidity in the same period (Geurts van Kessel et al., 2003). The reason of this increase in turbidity is not yet understood and requires further research. The reduced nutrient load to the

Oosterschelde did not lead to a decrease in the blooms of *Phaeocystis*. Number and duration of the blooms of *Phaeocystis* are in the nineties in the same order as in the eighties.

Cockle fishery

Another pressure on the benthos is the cockle fishery, which depends entirely on wild stocks living mainly on the tidal flats. The cockle fishery varies in space and time due to management decisions (closing of some areas in some years) (Geurts van Kessel et al., 2003). In the period 1997 – 2000 the entire Oosterschelde was closed for the cockle fishery, due to the low amount of cockles that are reserved as food for birds, and re-opened in some parts from 2001 on. A big debate on the effect of the mechanical cockle fishery on the macrobenthos is still ongoing; a recent study of NIOO showed no effect on the occurrence of macrobenthos when comparing sites in the Dortsman before and after a cockle fishery event (besides of course the large cockles that are caught by the cockle fishery). On the other hand, in the Wadden Sea clear negative effects are described. More research is needed to quantify the effect of the cockle fishery on the macrobenthos.

Main pressures Oosterschelde: decline of intertidal areas (the `zandhonger`) and the spread of the invasive Japanese oyster

4.3.1.3 Habitat typology

The habitat typology of the Oosterschelde is based on physical characteristics (ecotopes). This ecotope system has been derived from the ZES-ecotope system (Zoute wateren Ecotopenstelsel) developed by RIKZ for the Dutch coastal and transitional waters (Bouma et al., 2003). This system has a hierarchical structure that includes the five following levels: salinity < substrate < elevation < hydrodynamics < elevation/depth < sediment characteristics. Threshold values are defined for each parameter delimit condition classes, wherein rather homogeneous benthic communities are expected to occur (4.3.1.3.1). The validation of these habitats, in other words to investigate if the physical boundaries match with the biological ones, has still to be done.

4.3.1.3.1 Habitat classification parameters

In this section, the parameters and variables relevant for the habitat classification of the Oosterschelde are presented; for a full description of the ZES – ecotope classification see Bouma et al. (2003) and Twisk (2003) for more detailed information on the Oosterschelde ecotope classification.

Salinity

The parameter salinity plays no role in the habitat classification, because the entire Oosterschelde in this period is characterized by a relatively constant salinity (Twisk, 2003).

Substrate

The second level in the ZES system deals with the nature of the substrate, hard substrate or soft-sediment. In the present case only the second category has to be considered.

Elevation

At the third level of the ZES classification distinction is made between areas as function of their vertical position relative to the tidal range. For the Oosterschelde, a different boundary, than which is advised by Bouma et al. (2003), between sub-littoral and littoral is used, namely the GLW-line instead of the GLWS-line (Twisk, 2003). The sub-littoral is situated below the intertidal zone and remains permanently submerged. The littoral domain corresponds with the area between GLW and GHWD (each tide submerged) and the supra-littoral is situated above the GHWD (not each tide submerged). Within each of these littoral areas sub-divisions are made according to the vertical position.

Hydrodynamics

The maximum current speed by an average spring tide is used as an indicator for the intensity of hydrodynamics (independent from ebb or flood). The boundary between low- and high dynamic situations is set at 0.8 m/s for the sub-littoral. The discrimination between low and high dynamic situations in the intertidal is base on geo-morphological maps (based on aerial photographs).

Elevation/depth

The elevation/depth information is used to discriminate between shallow water and gullies (5m below GLWS). Within the littoral three sub-areas are distinguished according to the frequency of exposure with the low-littoral being exposed from 0 up to 25% of the time, the mid-littoral with emergence frequencies between 25 and 75% and the high-littoral that is exposed to air more than 75% of the time.

Sediment characteristics

The sediment characteristics are only determined for the map of 1983 and 2001, by discriminating mud (> 25%) or not, based on expert judgement on aerial photographs (Twisk, 2003). These mud rich areas are visualized in the eco-element maps (not shown in this report) and are scarce. The boundary of 25% mud in the sandy Oosterschelde is not good to explain the variation in the benthos and therefore further investigations are needed to update this level.

Oosterschelde soft-bottom habitats

By combination of these parameters and excluding the habitats, which does not exist in the Oosterschelde, the following habitats are discerned (Twisk, 2003):

- Sub-littoral, high dynamic, gully (sub-littoral[hdyn]_gully)
- Sub-littoral, low dynamic, gully (sub-littoral[ldyn]_gully)
- Sub-littoral, low dynamic, shallow (sub-littoral[ldyn]_shallow)
- Littoral, high dynamic (littoral[hdyn])
- Low-littoral, low dynamic (low-littoral[ldyn])
- Mid-littoral, low dynamic (mid-littoral[ldyn])
- High-littoral, low dynamic (high-littoral[ldyn])
- Salt marsh
- Littoral water (littoral areas where no distinction in hydrodynamics can be made)

For the Oosterschelde three habitat maps are constructed, namely for the year 1983 (before the storm surge barrier construction), 1990 (shortly after the storm surge barrier construction) (Figure 23) and 2001 (recent situation) (Twisk, 2003). The biological validation of the different habitats is not yet done and is advisable. Also the lack of a sedimentological deviation in the present habitat classification is a negative point, because it is one of the most important physical parameters in determining the macrobenthic community structure in marine systems.

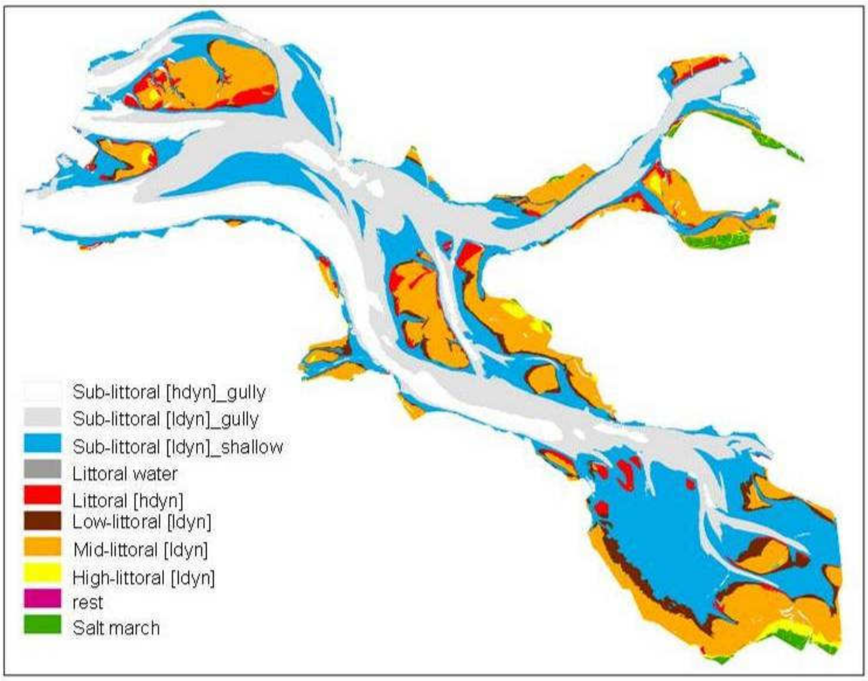


Figure 23. Example of the habitat distribution in the Oosterschelde (habitat map of 1990)

4.3.1.3.2 Eco - elements

The Oosterschelde contains two bivalve species, mussels and oysters, which can dominate in a habitat and because of their ecosystem engineering function, it form its own characteristic, associated benthic community. Therefore, they are indicated as eco-elements. Large mussel and oyster banks occur in the Oosterschelde and shall be taken into account in the evaluation at level 2 of the BEQI method. An example of the eco-element map is given in Figure 24.

Mussel banks

Almost the entire surface of mussel banks in the Oosterschelde is part of the mussel aquaculture. In the postbarrier period, cultivation methods have been slightly modified. Owing to the coastal engineering project, reduced current velocities allowed shellfish culture to be extended to previously unstable sites. Nowadays 22.5 km² is in use as subtidal bottom sites for mussel culture, 3.4 km² for re-watering, and 11 km² for bottom culture of oysters. All the mussel aquaculture areas are mapped in Figure 24, but only on half of the registered mussel culture plots mussels are grown (Kater & Kesterloo, 2003). Nowadays, there are almost no 'natural' mussel banks occurring in the Oosterschelde. Intertidal mussel banks have disappeared and are an important loss for higher trophic levels, e.g. as foraging area for birds.

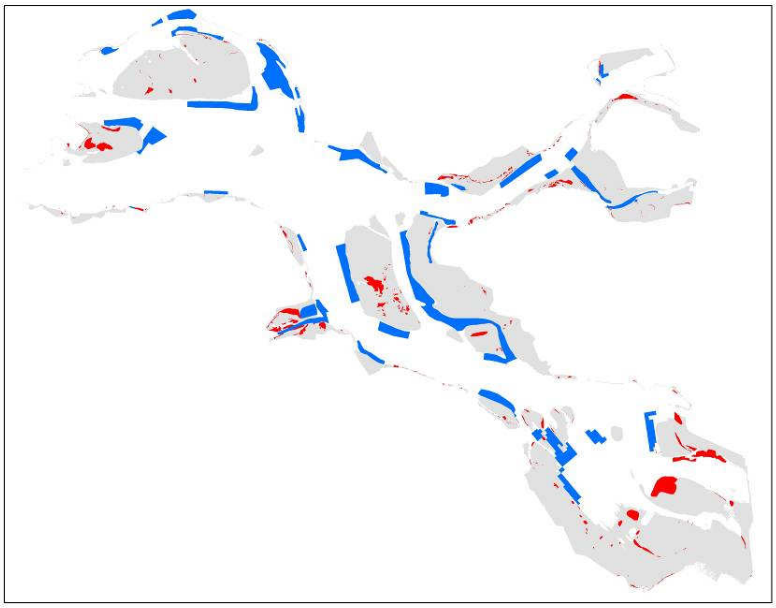


Figure 24. Example of an eco-element map of the Oosterschelde (eco-element map of 2001), with in blue the mussel aquaculture areas and in red the *Crassostrea gigas* beds

Oyster banks

Originally, the `platte oester` *Ostrea edulis* occurred in the Oosterschelde, which was also cultivated. After a mass mortality, due to a severe winter (1962-1963), man started to cultivate the Japanese oyster *Crassostrea gigas* in 1964 (Geurts van Kessel et al., 2003). With the introduction of this oyster, also a disease (Bonamia) was introduced which further reduced the occurrence of *Ostrea edulis*. Although it was anticipated that Crassostrea would not reproduce in the Oosterschelde, this invasive species reproduced and spread over the entire Oosterschelde and further along the Dutch coast and estuaries during the last decades (Nehring, 2006). Nowadays, *Crassostrea gigas* became a dominant species and they form large banks, both intertidally and subtidally. Due to the problems (see further), the *Crassostrea gigas* banks will be evaluated as eco-element at level 2 of the BEQI method.

These two bivalve species occur in banks in high densities and biomass and changes the values of the community parameters of the habitat wherein they occur. They form a distinct benthic community compared to the surrounding bottom. Therefore, it is advisable to take these eco-elements as a distinct habitat at level 3 of the BEQI method.

Sea grass

In previous periods, large areas of sea grass occurred in the Oosterschelde and are nowadays almost disappeared. Sea grass can also be included as eco-element on level 2, but is incorporated as another quality element in the WFD evaluation of a water body and is excluded in this analysis.

4.3.1.3.3 Discerned habitats

The monitoring strategy followed in the Oosterschelde is based on a sampling in four a priori defined strata. These strata are based on the depth distribution, rather than on a habitat classification. Therefore, it is reasonable to expect that many of the habitats of the Oosterschelde will not be quantitatively sampled (i.e. less than 20 records) on a regular basis with the current monitoring program. A selection of habitats that are quantitatively represented in the available data is necessary. The samples used for the analysis are taken in autumn to exclude seasonal effects. The habitat maps of 1990 and 2001 are used to realize the coupling between respectively the reference data (< 1995) and assessment data (2003-2005) of the nine available habitats. Due to the unequal distribution of data between the habitats, only 3 habitats can be considered for setting the reference conditions and the assessment analysis (Table 19). Also the two eco-elements, mussel and oyster bank are included as separated habitat, but cannot be evaluated due to few data availability.

Table 19. The soft-bottom habitats of the Oosterschelde, with in bold the habitats that are evaluated. The total number of samples and total sampling surface for the reference and assessment analysis are indicated

		Number of samples		Total sampling surface	
Habitat	Nr	Reference	Assessment	Reference	Assessment
sub-littoral[ldyn]_gully	**1**	**191**	**112**	**2.865**	**1.68**
high-littoral[dyn]	2	3		0.045	
low-littoral[ldyn]	3	6	6	0.591	0.09
littoral[hdyn]	4	14	9	0.21	0.135
mid-littoral[ldyn]	**5**	**71**	**62**	**3.57**	**0.93**
sub-littoral[ldyn]_shallow	**6**	**217**	**121**	**3.255**	**1.815**
sub-littoral[hdyn]_gully	7	40	33	0.6	0.495
salt marsh	8	0	0	0	0
littoral water	9	0	0	7	0.105
Mussel bank	10	4	3	0.06	0.045
Oyster bank	11	2	2	0.03	0.03

It is clear from Table 19, that especially the sub-littoral habitats are sampled, compared to the intertidal habitats. In Figure 25 and Figure 26, the distribution of respectively the reference and assessment samples is plotted. The reference samples are taken with a randomized sampling strategy every year, whereas the assessment samples are taken on fixed points every year.

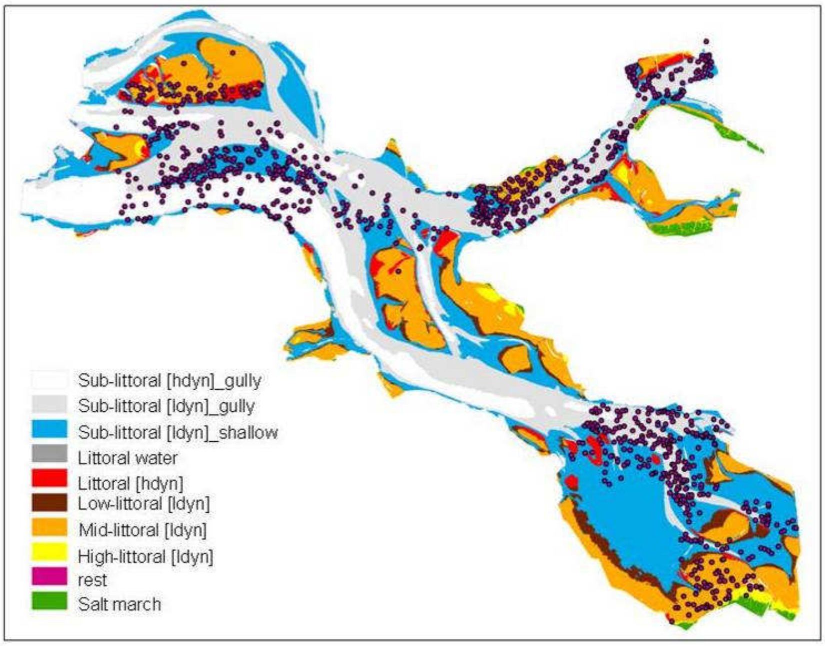


Figure 25. Position of the reference samples in the Oosterschelde on the habitat-map of 1990

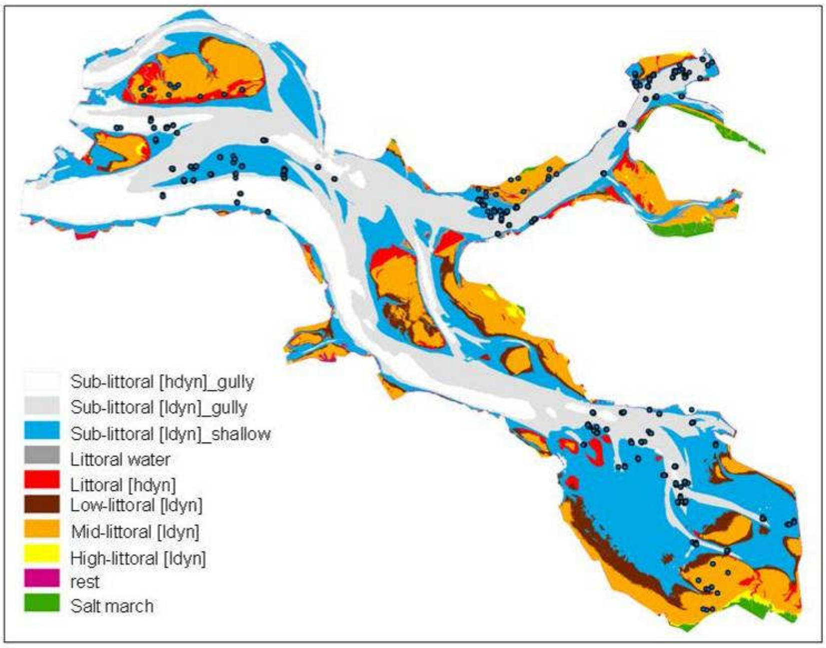


Figure 26. Position of the assessment samples in the Oosterschelde on the habitat map of 2001

4.3.1.4 Reference data/settings

The above discussed aspects made it difficult to define a 'natural' not impacted' reference period, which does not exist for the Oosterschelde, especially due to the man-made constructions, which changed the Oosterschelde from an estuary to a sheltered, polyhaline sea-arm. Therefore the reference settings for the different levels of the BEQI method are separately outlined and are mainly based on a selection of the best available data.

4.3.1.4.1 Reference settings at level 1 (ecosystem)

The boundary settings (high = MEP; good = GEP; moderate; poor; bad) and the optimal reference state (B/P ratio = 1/10) for the evaluation at level 1 of the Oosterschelde are the same as defined in the overall explanation of level 1 of the BEQI method (see Material and Method) (Figure 3).

4.3.1.4.2 Reference settings at level 2 (habitat)

At level 2 (evaluation of the habitats and eco-elements) of the BEQI method, there is no general set-up for the different water bodies and therefore the approach at level 2 for the Oosterschelde will be in detail described in this section.

For the Oosterschelde, three habitat maps are made for the years 1983, 1990 and 2001. From these maps the changes in the habitats after the construction of the storm surge barrier are obvious. The main change after the construction of the storm surge barrier is the increase of the low-dynamic subtidal with more than 10000ha at the expense of the high-dynamic subtidal, due to the decrease in the flow velocities (Figure 27). Also, the intertidal habitat is decreasing (Geurts van Kessel, 2004). Estimations of the decrease in intertidal habitat is presented in the report of Geurts van Kessel (2004) and they estimated a decrease of 11ha/year in the mouth area, 12ha/year in the central part, 8ha/year in the northern part and 21ha/year in the 'kom' area for the period 1983 to 2001 (a total of 1000 ha intertidal disappeared in a period of 18 years). These drastic changes are obvious, but there are no predictions yet available that predicts the surface of the intertidal habitat when the Oosterschelde reaches its new equilibrium. And will this changing pattern disappear when the sand balance problem for the Oosterschelde is solved, e.g. by restoring the sediment input from the sea? Therefore, it is at this moment not possible to set reference surface area values for the different habitats.

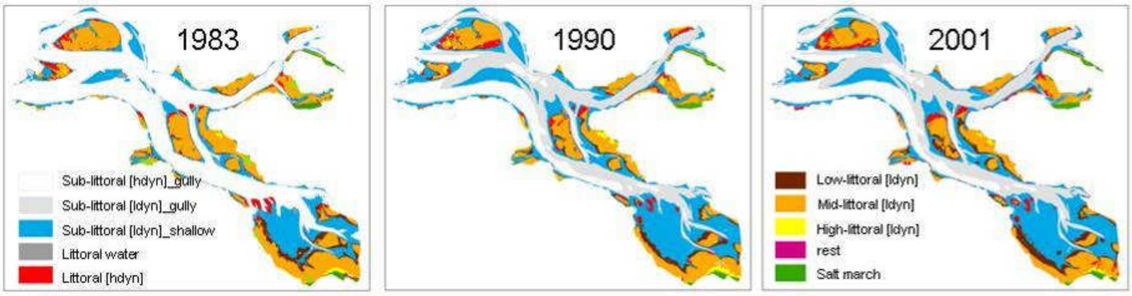


Figure 27. Habitat maps of 1983, 1990 and 2001 to show changes in habitat surface areas

Another important issue at level 2 for the Oosterschelde is the occurrence and changes in the eco-elements (Mussel, Oyster, sea grass). Sea grasses are disappearing in the Oosterschelde, but are not taken into account in this study, due to the fact that it is another quality element for the WFD. Mussel banks nowadays are almost entirely cultured in the Oosterschelde. These culture plots are nowadays situated in the sub-tidal part of the Oosterschelde (Figure 7). Natural mussel banks disappeared almost entirely from the Oosterschelde. The reason for this is still not clear and should be further studied. In the future natural mussel banks have to be separately evaluated from culture plots.

Another eco-element in the Oosterschelde is the occurrence of the Japanese Oyster, which is now widespread (Figure 28) and becomes dominant above the other naturally occurring bivalves. In the study of de Kluijver and Dubbeldam (2003) it was concluded that the diversity (Simpson index) decreased in the sub-littoral communities with increasing surface area of the Japanese Oyster. Therefore the Japanese Oyster beds need to be included as eco-element for the evaluation at level 2.

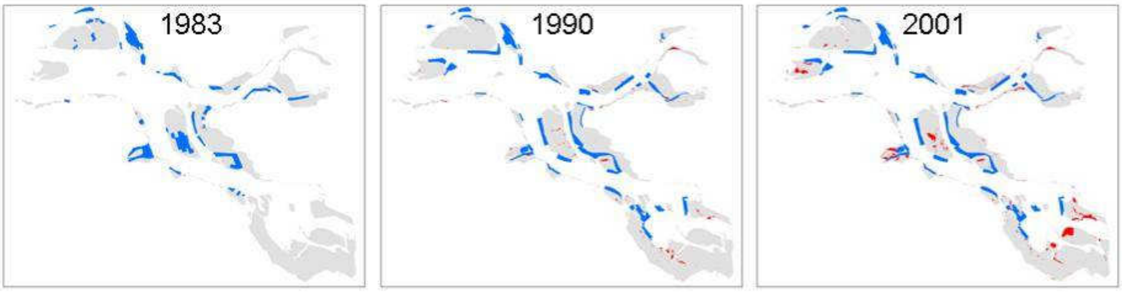


Figure 28. The eco-element maps of the Oosterschelde, with in blue the mussel aquaculture areas and in red the *Crassostrea gigas* banks

Because no reference surface areas for the habitats and eco-elements are available, the evaluation at level 2 for the Oosterschelde will be done by expert judgement.

4.3.1.4.3 Reference settings at level 3 (community; within-habitat)

At the community level, only benthic data, for this project, from 1990 untill 2005 was available and it is from after the construction of the storm surge barrier. This period is not characterized as free from anthropogenic pressure, nevertheless the years before 1995 will be selected as the reference period, because:

- This corresponds with the overall strategy of selecting a reference dataset (approximately the first one third of the years of the available data period).
- In the reference period, the Oosterschelde is sampled with a randomized sampling strategy, which improves the spatial knowledge of the system. A good spatial coverage is required for the reference data set.
- The hydro- and geomorphology of the Oosterschelde is still changing, as well as the turbidity. Also an increase of the invasive species *Crassostrea gigas* is observed. Therefore the beginning of the 1990's was selected to observe changes in the benthos related to these pressures and changes.

Practically, at this level for each selected habitat (Table 19) the reference values and related boundaries are calculated for each of the four parameters (number of species, density, biomass and species composition change) in relation to the surface area sampled out of the reference data set (period 1990-1994). The samples selected for the reference setting, are taken in autumn and are coupled to the habitat map of 1990 to link each sample to a certain habitat.
The plots of the reference values in relation to the sampling surface are visible in annex. The reference boundary values used for the assessment are given in Table 24.

4.3.1.5 Assessment

4.3.1.5.1 Level 1: ecosystem

At this level, assessment values for system primary production and average macrofauna biomass is needed to construct the ratio $B_{benthos}/P_{prim}$ and estimate the score and status, based on the proposed boundary settings of Figure 3.

System primary production

The primary production in the Oosterschelde was both measured and predicted by modeling (the latter showed more or less equal values with the observed values) at 5 locations in the 4 sub-areas for the period 1990-2000 (Geurts van Kessel et al., 2003). These calculations are also done for 2002-2003, but these are not yet published (pers. comm. Jacco Kromkamp). The averaged modeled value of the different locations for the Oosterschelde is 241 g C.m^{-2} yr^{-1} for 2003. This is the most recent estimate.

Average macrofauna biomass

The average macrofauna biomass of the assessment period (2003-2005) was estimated as 21.51 g AFDW/m^2. This value corresponds to a plain average of all sampling points considered in the present study without consideration of the surface area of the different habitats. The biomass of the most important bivalves is estimated at 6 million kg AFDW (mussels, oysters and cockles) for the entire Oosterschelde (estimate based on Geurts van Kessel et al., 2003). This means an average of 19.4 g AFDW.m^{-2} that has to be added to the average macrofauna biomass, which makes a total macrofauna biomass estimate of 40.91 g AFDW.m^{-2} for the Oosterschelde.

Ecological status at level 1

The ecological status at level 1 (ecosystem level) is evaluated as GEP. The value of primary production is obtained by modeling and has to be checked with the measured values. This means that the ecological score and status can change in the future.

Table 20. Evaluation of the ecological score and status at level 1 (ecosystem level). B: average macrofauna biomass (g AFDW/m^2); P: system primary production (gC/m^2 year)

B	P	B/P	Boundary class	EQR score	EQR status
40.91	241.00	0.170	1:5 < B:P < 2:15	0.691	GEP

4.3.1.5.2 Level 2: habitat

The disappearance of the intertidal habitat (with almost 1000 ha in 18 years) is a negative effect for the ecological functioning of the Oosterschelde, because the intertidal is characterized by a unique and valuable fauna and flora that support important bird numbers (link to Habitat and Bird Directive). Therefore, based on expert judgement, it was decided to evaluate the parameter intertidal area as moderate (EQR value: 0.5).

Natural mussel banks have almost disappeared and all culture plots are nowadays situated in the subtidal part of the Oosterschelde. As a consequence, no intertidal mussel beds occur in the Oosterschelde, but due to the presence of aquaculture mussel beds the characteristic fauna is still present in the ecosystem and therefore the parameter mussel bank area is evaluated as moderate by expert judgement (EQR: 0.5).

The oyster bank areas, created by *Crassostrea gigas*, are estimated as a negative influence for the ecological functioning of the Oosterschelde due to the fact that they cause a lot of problems (see e.g. Geurts van Kessel et al., 2003). Therefore the parameter oyster bank is evaluated as moderate by expert judgement (EQR: 0.5).

Table 21. The Ecological Quality score and status for the parameters at level 2, based on expert judgement.

Level 2: habitat	EQR score	EQR status	Remark
intertidal area	0.5	moderate	expert judgement
mussel bank	0.5	moderate	expert judgement
Japanese oyster bank	0.5	moderate	expert judgement

4.3.1.5.3 Level 3: community (within-habitat)

At this level, it was tried to evaluate the changes in species richness, species composition, density and biomass for some habitats of the Oosterschelde. The results of the assessment of the four parameters are summarized in Table 24. For the three habitats, which could be evaluated, of the Oosterschelde, the assessment surfaces are OK to get an acceptable assessment analysis.

Table 22. The minimal and optimal sampling surfaces needed to get an acceptable assessment analysis for the different habitats of the Oosterschelde.

Habitat	Assessment surface	minimal surface	OK surface	optimal surface	Assessment power class
sub-littoral[ldyn]_gully	1.680	0.36	1.13	1.86	OK
mid-littoral[ldyn]	0.905	0.453	0.905	2.665	OK
sub-littoral[ldyn]_shallow	1.815	0.405	1.155	2.52	OK

In the Oosterschelde, all discerned habitats reached a GEP status, which means that they are not seriously changed compared to the reference situation. Only the parameter similarity shows that there are changes in the species composition, because it reaches a moderate status for all habitats.

When the average is taken of the ecological status scores of the discerned habitats of the Oosterschelde, an overall EQR for level 3 of 0.729 (GEP status) is obtained. It has to be mentioned that not all habitats are evaluated due to an insufficient amount of data. Some of these habitats where important in surface area (sub-littoral [hdyn]_gully and littoral[hdyn]). An extra sampling effort is needed to fill this gap.

4.3.1.5.4 Integration of the three levels

For the overall assessment of the water body the ecological score and status obtained for each of the three levels are averaged into a metric representative for the whole water body. The averaging is done with a weighing factor for each level, 1 for the ecosystem level and 2 for the habitat and community level. The average of the three levels is 0.630, which correspond with a GEP status for the Oosterschelde (Table 23)

Table 23. Ecological quality score and status obtained by average the parameters at each level.

	EQR score	EQR status	Remark
Level 1: ecosystem	0.691	GEP	
Level 2: habitat	0.5	moderate	expert judgement
Level 3: community	0.729	GEP	
Overall EcoQ	0.630	GEP	

Table 24. The assessment of level 3 for the habitats of the Oosterschelde, with indication of the assessment sampling surface, assessment parameter value, the reference boundary values and finally the EQR score and status. The habitats with an acceptable sampling surface for assessment are set in grey

	parameter	Assessment		Reference boundary values										EQR	
Habitats		surface	value	Poor min	Mod min	GEP min	MEP min	Median	MEP max	GEP max	Mod max	Poor max	Max spp.	score	status
sub-littoral[ldyn]_gully	biomass	1.680	31	5	10	15	21	25	30	39	52	65		0.774	GEP
	density	1.680	3601	833	1665	2498	2982	3267	3589	4326	5768	7210		0.796	GEP
	similarity	1.680	0.74	0.29	0.59	0.88	0.91							0.506	Moderate
	species	1.680	103	32	63	95	102						120	0.811	MEP
average of parameters for sub-littoral[ldyn]_gully														0.722	GEP
mid-littoral[ldyn]	biomass	0.905	23	4	8	12	21	26	32	42	56	70		0.870	MEP
	density	0.905	16874	1365	2730	4096	6190	7675	9375	13256	17674	22092		0.433	Moderate
	similarity	0.905	0.73	0.25	0.50	0.75	0.83							0.582	Moderate
	species	0.905	53	12	25	37	47						67	0.860	MEP
average of parameters for mid-littoral[ldyn]														0.686	GEP
sub-littoral[ldyn]_shallow	biomass	1.815	39	7	13	20	28	34	40	54	72	90		0.854	MEP
	density	1.815	2812	769	1537	2306	2687	2950	3232	3895	5193	6491		0.896	MEP
	similarity	1.815	0.77	0.29	0.59	0.88	0.90							0.526	Moderate
	species	1.815	114	34	68	102	110						131	0.838	MEP
average of parameters for sub-littoral[ldyn]_shallow														0.779	GEP

4.3.1.6 Discussion

Reference settings

The reference period is not free from pressure influences, but the period after the construction of the storm surge barrier is a good starting point to evaluate the changes within the Oosterschelde ecosystem due to the main pressures (need for sand, influence of the Japanese oyster). It was not opportune to use the benthic data from before the construction of the storm surge barrier, because this construction will not be removed. Detailed observations and investigations will be needed to determine the new equilibrium state of the Oosterschelde and the related ecological potential. This can help to improve the reference settings for the Oosterschelde in the future.

Assessment analysis

An assessment analysis of the Oosterschelde is done for the three levels. The score obtained for the first level is subjected to changes (updates). It is advisable to improve not only the primary production estimate, but also the total benthic biomass estimate.

At level 2 it is concluded that the Oosterschelde is changing (especially the decrease of the intertidal area) due to the 'zandhonger' phenomena. Also the invasive species *Crassostrea gigas* is dominating the system more and more and causes certain problems in ecosystem functioning. Therefore the expert judgement at level 2 is moderate. This gives a signal that further investigations are needed.

The Oosterschelde is characterized by a high number of macrobenthos taxa (compared to the Westerschelde, Lake Veere and Grevelingen) of which are a high amount unique for the Delta (Schaub et al., 2003). In total biomass no significant changes are observed, whereas for the density a slight increase was observed. The species composition of the benthos seems not really stable, but further trend monitoring is needed to confirm this. The occurrence and density of the cockle is declining (Geurts van Kessel et al., 2003), whereas some invasive species are more frequently found (*Crassostrea, Corbula gibba, Ensis directus*) (Schaub et al., 2003).

These trends in the benthos are reflected in the BEQI evaluation method. In general the overall parameter at level 3 for the different habitats shows a GEP status (Figure 29), but some habitats could not be evaluated. The density and biomass did not drastically change. The number of species is still high, thus the diversity (species richness) is not really changed, but changes in the species composition are detected (similarity indicator). The problem of invasive species is most expressed in the occurrence of the Japanese oyster and this is therefore evaluated as moderate at level 2.

The main pressure at this moment is the need for sand and thus the decline in intertidal area and increase of the sub-tidal habitats. Because in these habitats different benthic communities occur, it is expected that a decline in the diversity of habitats will lead to a decline in the diversity of the benthic communities and benthic species (also consequence on higher trophic levels are expected). Changes in species composition are observed, but can nowadays not directly be linked to a certain pressure.

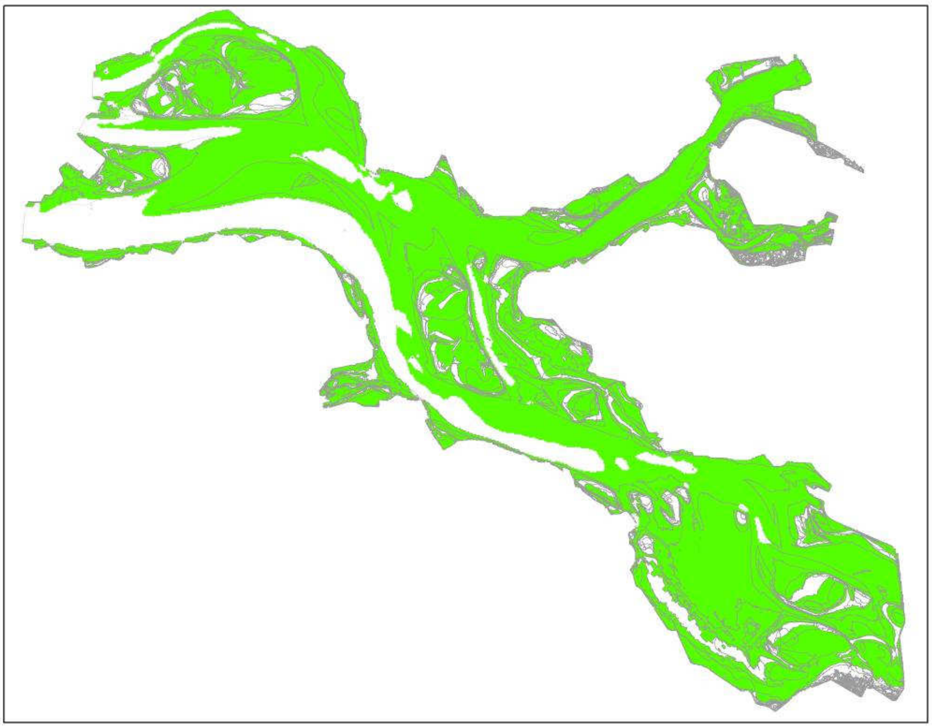


Figure 29. Chart of the Oosterschelde, with indication of the Ecological quality status at level 3 of the BEQI method for all habitats: light green: GEP; white: no assessment possible. In the figure the presence of oyster and mussel banks are not taken into account

Advices

The following advices for the Oosterschelde can be formulated:

- To make a biological validation of the habitat maps and improve the habitat maps (e.g. by taken into account sediment characteristics)
- To incorporate at level 3 and thus also in the monitoring, the eco-elements (mussel and oyster beds) as a different habitat.
- To improve the total assessment surface area for some habitats (see chapter on monitoring).
- To investigate how much the decrease in intertidal area will be, before the need for sand problem is solved (new equilibrium reached). Also the change in the subtidal habitats (e.g. the silting up) requires further research. Tests with sand suppletions are planed and needs good monitoring.
- The effect of a further spreading of the Japanese Oyster on the functioning of the Oosterschelde ecosystem requires further research.

4.3.1.7 Conclusion

Oosterschelde:

- pressure: decrease in intertidal area ('zandhonger') and spread of Japanese oyster
- Evaluation:
    - Level 1: GEP; level 2; Moderate; level 3: GEP
    - The pressures in the Oosterschelde are mostly observed and evaluated at level 2, whereas the benthic community characteristics are presently not drastically changed compared to the reference period.
- Advice:
    - A monitoring program should be installed to evaluate further the impact of the different pressures on the ecosystem of the Oosterschelde. This monitoring program should integrate measurements on primary production (level 1), habitats (level 2) and macrobenthos (level 3). For some specific questions (e.g. the effect of the Japanese Oyster) an investigative monitoring might be required.

4.3.2 Wadden Sea

4.3.2.1 Short description

The Wadden Sea is a large shallow sheltered sea behind a row of barrier islands (Figure 30). It consists of several more or less separated tidal basins. At low tide extensive intertidal flats emerge. It extends from Den Helder in the Netherlands to Esbjerg in Denmark. It is an important stop over and staging site for large flocks of shore birds, which mainly feed on benthic invertebrates. It also serves as a nursery area for flat fish from the North Sea. The Dutch Wadden Sea is designated as protection area under the Ramsar Convention**,** and by the Dutch government it is included as part of the EU Habitat Directive as well as of the EU Bird Directive (and included in NATURA 2000).

Figure 30. Dutch part of the Wadden Sea

4.3.2.2 Human pressures and environmental problems

Main pressures are or are fisheries, euthrophication and the introduction of invasive species. Also land reclamation, closure of sea arms, and construction of dikes have impacted the system and confine the Wadden Sea of to human defined borders.

Land reclamation

The Wadden Sea has developed during the last 6000 years because a rising sea level. Land behind the beach barrier became flooded. Over the ages humans became more effective in stopping the sea progressing landward. In the last century serious plans are made and studied to completely close of the Wadden Sea from the North Sea and reclaim the land. It has not come that far but large dams have been built creating the IJsselmeer and the Lauwersmeer. This has influenced the hydrodynamics, for instance the tidal amplitude in the western Dutch Wadden Sea increased, at some places almost 100 cm with the closure of the Afsluitdijk (Figure 31). This was a severe impact and will have had its effect on the macrobenthic community and still has e.g. through salinity fluctuations caused by sluicing.

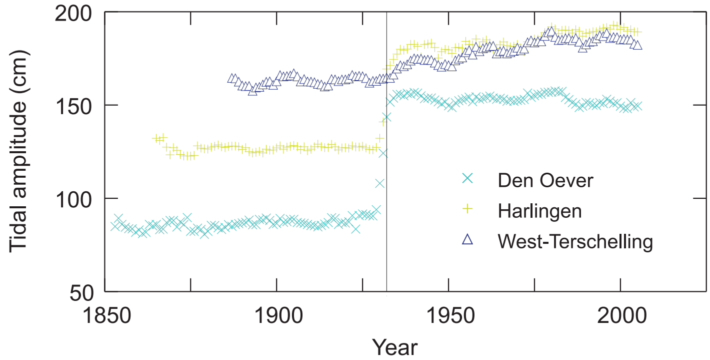


Figure 31. Tidal amplitude in the Western Wadden Sea at three locations over more than hundred years. The effect of the Afsluitdijk is clearly visible, the line shows the moment of closure

Fishery

Until 2004 a large part of the Dutch Wadden Sea area was open to mechanical cockle (*Cerastoderma edule*) dredging. This has negatively impacted the benthic fauna. Recruitment success of bivalves was lower in dredged areas (Piersma et al. 2001). Also the body condition of new recruited cockles in dredged areas was lower than in unaffected areas (Figure 32, Van Gils et al. 2006). It is thought that the bottom disturbance by dredging reduces silt content and disturbs the benthic microphytobenthos, which is an important food source for recruiting bivalves, also for suspension feeders like cockles (Kang et al. 1999). The Wadden Sea is now closed for mechanical cockle dredging but other sources of bottom disturbance like possibly caused by shrimp fisheries are still operating.

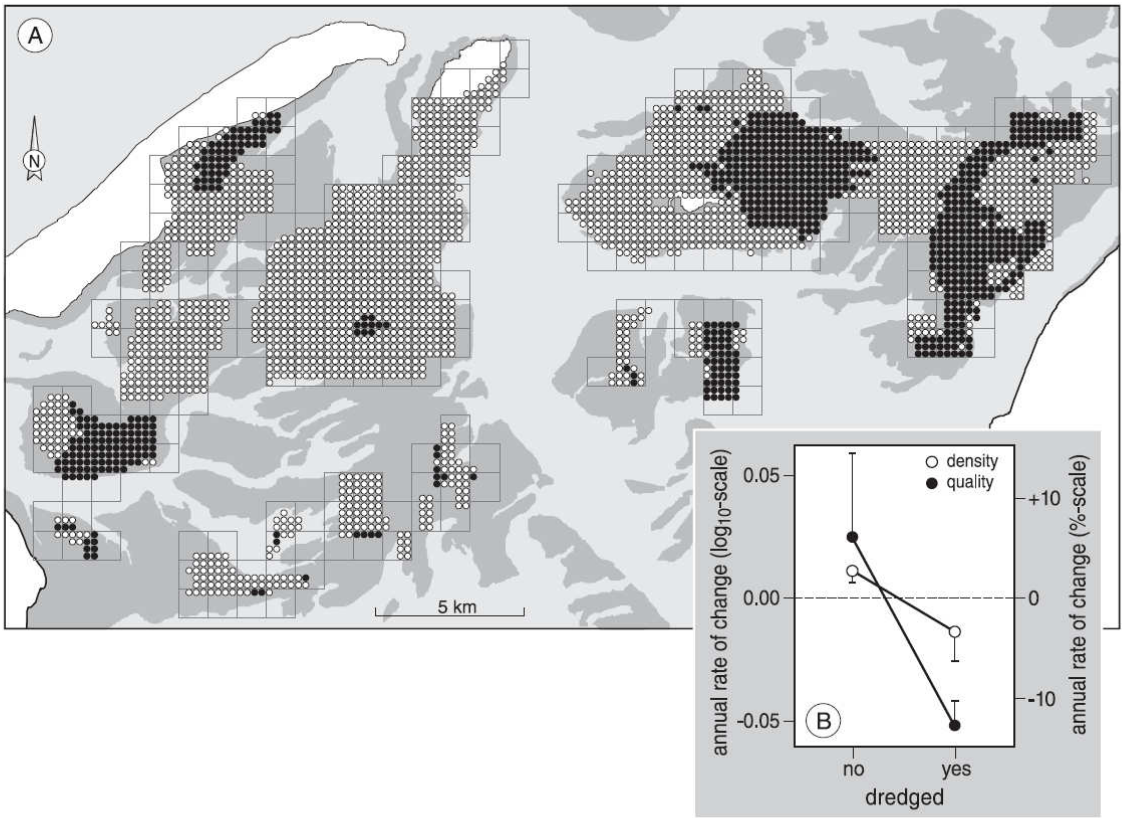


Figure 32. Large scale systematical sampling covering a large area in the western Wadden Sea. Areas at least visited once in the period 1998-2002 are marked with black dots (A). In these areas quality (flesh mass/shell mass) of newly settled cockles declined with 11% per year (B), (from Van Gils et al. 2006)

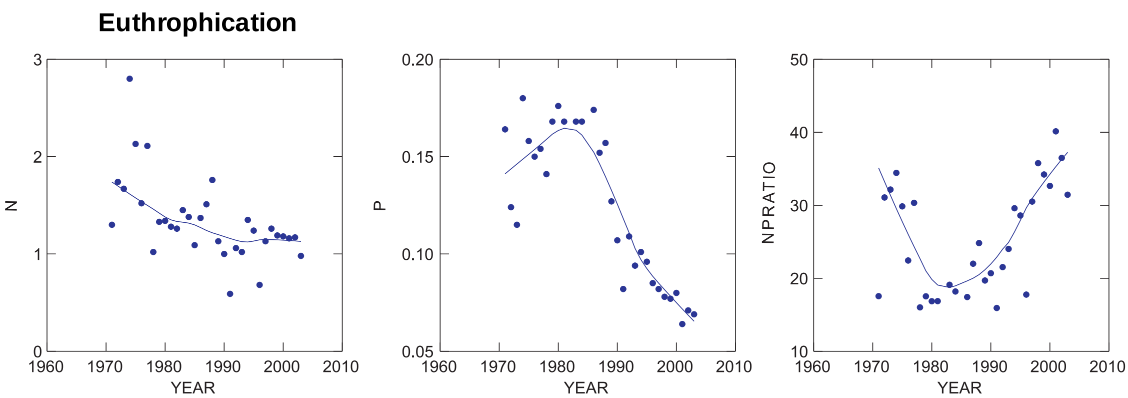


Figure 33. Concentrations of nitrogen (N) and phosphorous (P) in mg l^{-1} and the NP ratio (molar) in the western Dutch Wadden Sea. Data from the DONAR database at www.waterstat.nl

Due to increased use of fertilizers and phosphate in detergents nutrient loads entering the Wadden Sea started to increase, halfway the twentieth century (Van Raaphorst and De Jonge 2004). Phosphate levels are highest in the eighties and decreased in a short period to levels about a third of peak levels. Nitrogen was more constant during the last approximate forty years.

This means that the fluctuations in phosphate had a strong effect on the N to P ratio (Figure 33). The N to P ratio has a strong effect on the species composition of the phytoplankton (Philippart et al. 2000), which could influence its macro benthic consumers. The main effect ascribed to eutrophication in the Wadden Sea is the increase in biomass of the macro benthos (Figure 34; Beukema & Cadée 1997; Beukema et al. 2002). The increase in biomass fueled by eutrophication is persistent even with clear reductions in nutrients in the last 15 years. In a recent study, covering a 30-year observation period, Phillipart et al. (2007) showed that long-term variations in limiting nutrients (phosphate and silicon) are weakly correlated with biomass and more strongly with community structures of phytoplankton, macrozoobenthos and estuarine birds. Relationships between nutrient loading and the primary and secondary producers are very dynamic and still not completely understood. A better insight into these basic ecosystem processes primary- and secondary-production is needed for further improvement of the BEQI method at the first level.

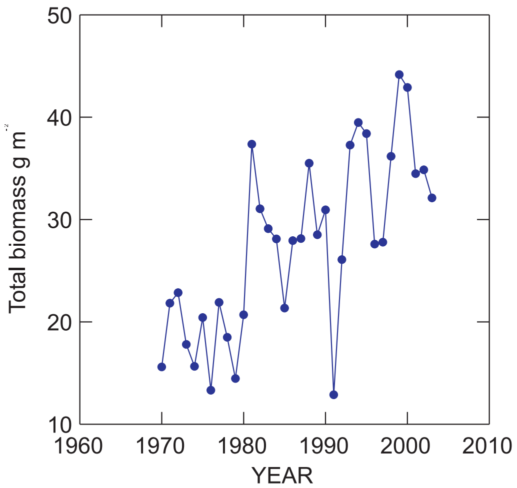


Figure 34. Total biomass of the macrobenthos on the Balgzand from 1970 to 2006. The increase in biomass spans the entire study period. Average of 12 transects and 3 quadrants sampled in late winter/early spring

Invasive species

In the last few decades several new species arrived in the Wadden Sea. In a short period the razor clam *Ensis directus* (syn. *Ensis americanus*) and the polychaete *Marenzelleria viridis* became an important part of the macro benthos. In 1983 the first occurence of the pacific oyster *Crassostrea gigas* was reported in the Dutch Wadden Sea. Almost 25 years later it proves a successful new inhabitant. Oyster reefs are formed at different sites among them mussel beds or old mussel beds (Fey et al. 2007). The mussel *Mytilus edulis* now has a severe competitor for hard substrate for settlement. The oyster forms massive reefs. With its large filtering capacity this so-called ecosystem engineer can have a strong top down control on the primary production of the phytoplankton.

Main pressures Wadden Sea:
Eutrophication, fisheries, invasive species and land reclamation

4.3.2.3 Habitat typology

4.3.2.3.1 Habitat classification parameters

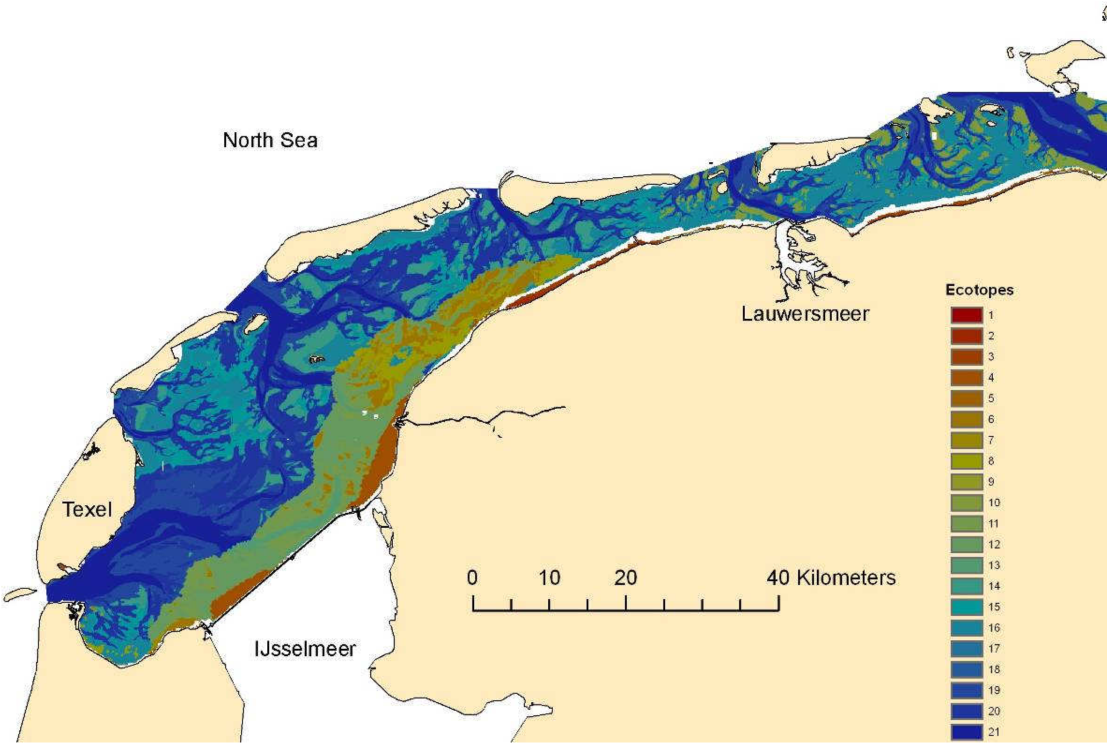


Figure 35. Ecotope map of the Wadden Sea. The legend to the ecotope numbers is presented in Table 25. In total 21 ecotypes are characterized including salt marshes and seagrass beds

For the Wadden Sea a ZES ecotope description is available (Wijsman and Verhage 2004), like for the other habitats discussed in this report. For the inter-tidal and sub-tidal areas The Wadden Sea ecotope classification is based on the parameters depth, salinity and hydrodynamics. Sediment characteristics are not taken into account. There are 21 ecotopes defined, including salt marshes and sea grass beds (Figure 35, Table 25). For the BEQI method a reduction of the number of ecotopes would be practical. Dividing the limited number of assessment station over many ecotopes will leave very little assessment power per ecotope. Furthermore the ecotope classification of the Wadden Sea has not been tested for biological relevance. and it is very well possible that the structuring of the biological community does not show so many levels as suggested by the ecotope map.

Table 25. ZES ecotopes in the Wadden Sea and their surface area

Habitat nr	description	area (ha)
1	Sea grass	144
2	Salt marsh	6609
3	Pioneer vegetation	2143
4	Variable salt	6465
5	Fresh	578
6	Brackish high dynamic littoral	8017
7	Brackish low dynamic low littoral	3241
8	Brackish low dynamic middle littoral	11590
9	Brackish low dynamic high littoral	27
10	Brackish low dynamic supra littoral	8
11	Brackish high dynamic shallow water	9661
12	Brackish low dynamic shallow water	17163
13	Brackish gully	3803
14	Marine high dynamic littoral	28664
15	Marine low dynamic low littoral	21106
16	Marine low dynamic middle littoral	37597
17	Marine low dynamic high littoral	267
18	Marine low dynamic supra littoral	368
19	Marine high dynamic shallow water	37601
20	Marine low dynamic shallow water	48876
21	Marine gully	35636
	not assigned	12956
	total	292520

4.3.2.3.2 Biological validation

For an analysis of the biological community structure the same data as the data for the reference description are used. These are “historical” data from before 1983. They are collected in a few large surveys. The first was a Dutch Wadden Sea wide survey where 99 inter-tidal transects are visited from 1971 to 1974, reported in Beukema (1976). In the western part of the Wadden Sea 42 of these transects are visited for a second time in 1977 (Beukema et al. 1978). During 1981 and 1982 a survey of the benthic fauna in the sub-littoral of the western Dutch Wadden Sea was made. In this project total 461 stations are visited (Dekker 1989). Finally the data of the late winter - early spring of the Balgzand monitoring program up to 1982 are used. Locations of stations and transects are given in Figure 39A.

A priori the data are divided in sub-tidal and inter-tidal based on the sampling method, walking transects at low tide is inter-tidal, using a ship with box-corer is sub-tidal. A TWINSPAN analysis using biomass data was done for each tidal level (resulting dendrogram inter-tidal, Figure 36; sub-tidal, Figure 37). For the inter-tidal 19 species are selected based on dominance in biomass, together they accounted for more than 99% of the total biomass. In the subtidal 16 species are selected using the same criteria as for the inter-tidal. The analysis was run with the software WinTWINS 2.3 freely available at http://www.ceh.ac.uk/products/software/wintwins.html, using the default cut values and only two division levels. Each analysis resulted in 4 groups. The mussel *Mytilus edulis* dominated one group in the inter-tidal (group 7,Figure 36) and one group in the subtidal (group 14, Figure 37). It turned out that the three remaining intertidal groups occur under distinct tidal levels and sediment composition (Figure 38). Group 4 is found highest in the tidal zone with highest mud contents above 5%. Group 5 is at intermediate tidal levels and silt contents roughly below 5%. Group 6 is low in the inter-tidal with hardly any silt. The result of this division of inter-tidal habitat is similar to the habitat typology applied in the German Wadden Sea. Germany distinguishes three types of inter-tidal flats; based on sediment, muddy, muddy sand and sand (Meyer and Ragutzki 1999). For the subtidal no sediment information was available to relate to the resulting clusters. Depth did not explain the biological clustering. A plot of the stations assigned to groups on a map (Figure 39B) suggests that group 15 lies in the vicinity of the sluices in the Afsluitdijk, where salinities are lower and more variable than in the rest of the area. Group 16 is relatively to group 17 found in the lower dynamic sub-tidal and group 17 is nearer the tidal inlets and in the larger gullies, which are clearly in more dynamic areas.

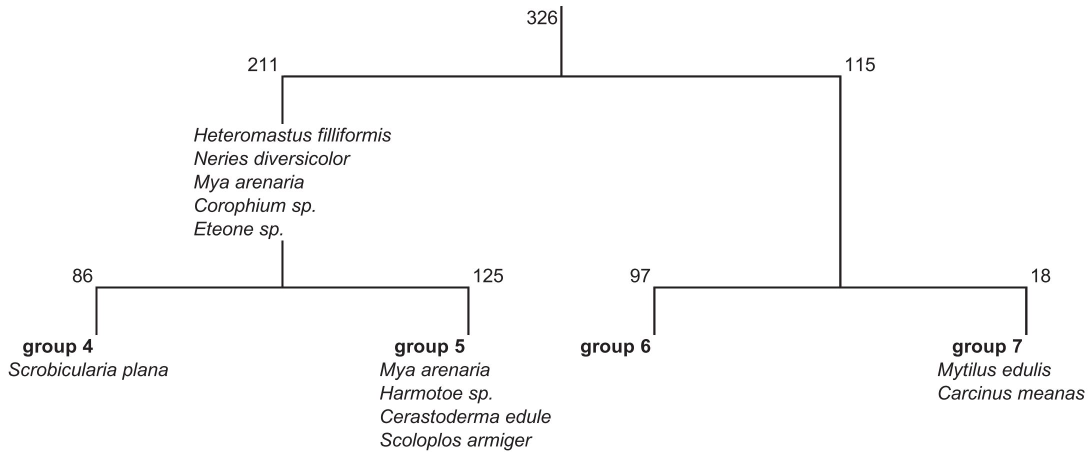


Figure 36. Dendrogram of a TWINSPAN analysis of intertidal stations of the reference dataset for the Wadden Sea. Group sizes and indicator species are shown

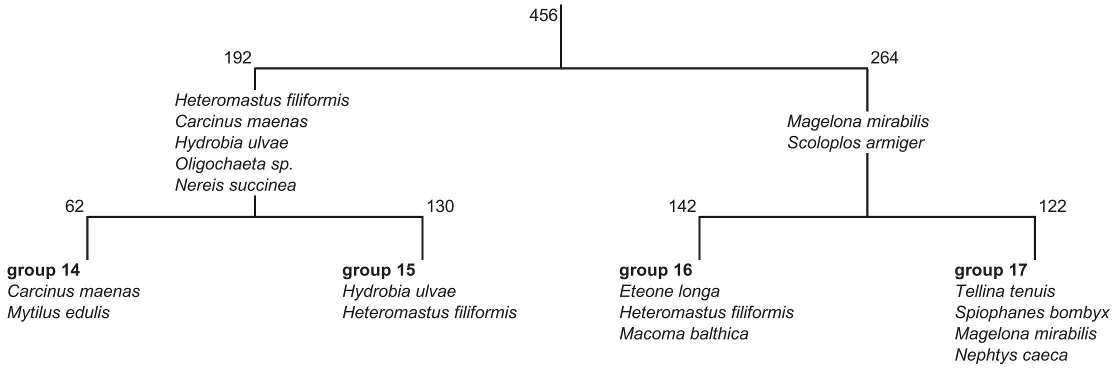


Figure 37. Dendrogram of a TWINSPAN analysis on data from 465 subtidal stations in western Wadden Sea visitid in 1981 and 1982. Indicator species and group sizes are given

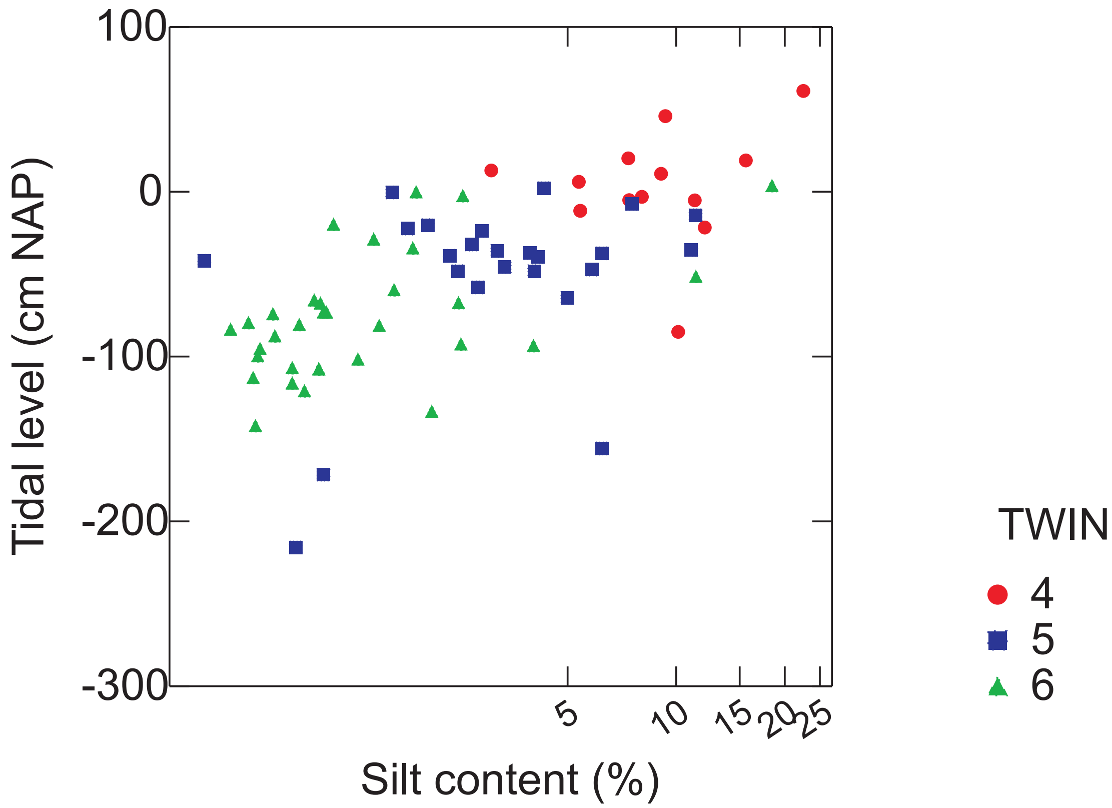


Figure 38. Plot of tidal level against silt content from sampling stations analyzed with TWINSPAN analysis on biomass of macrobenthic species.. The grouping nicely divides the gradient in three parts. Sediment and tidal level data information are extracted from Zwarts 2004

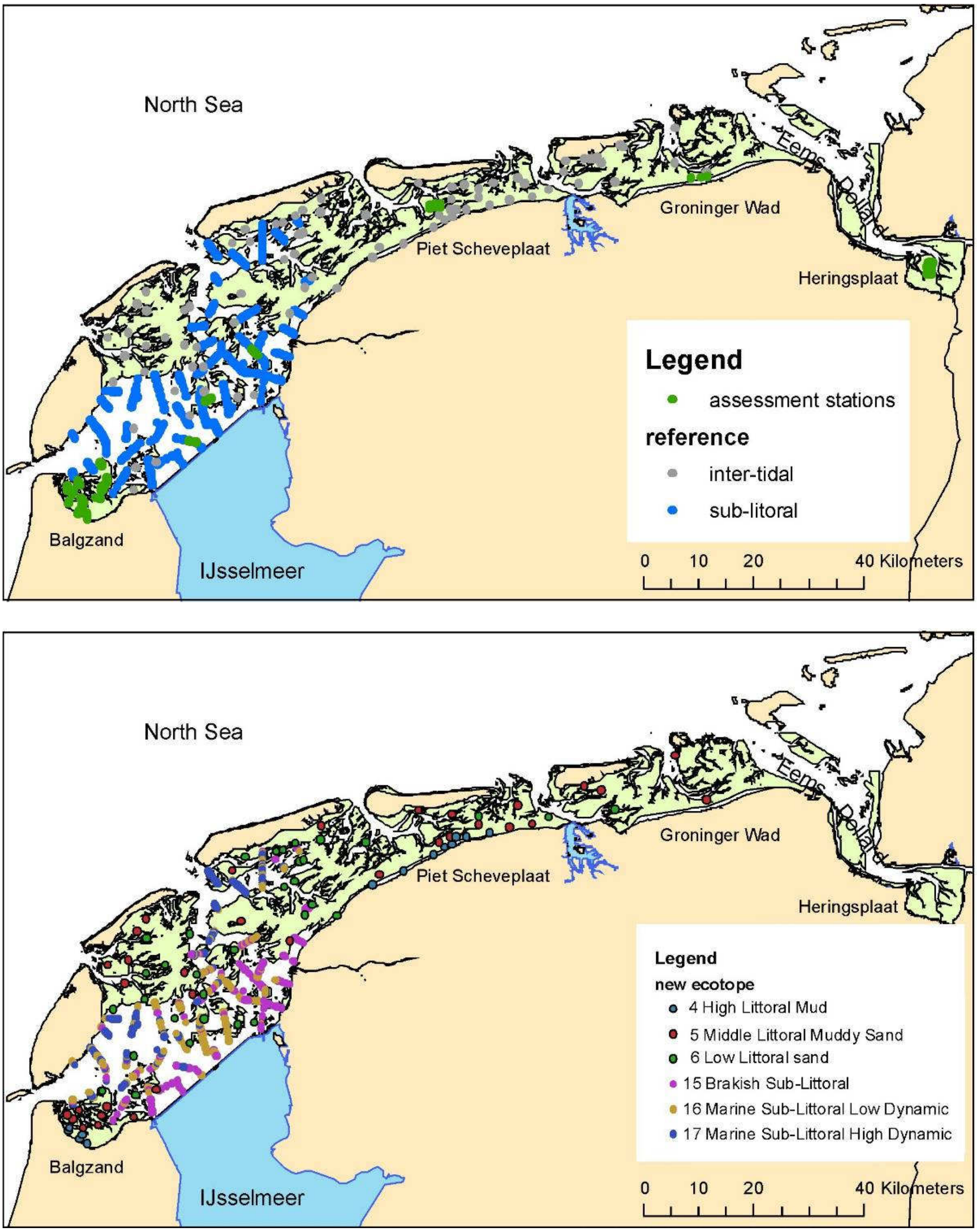


Figure 39. (A) Map of the Wadden Sea with intertidal and subtidal sampling stations used for habitat typology and reference description. Intertidal stations are transects, sub-tidal reference stations are individual sample points. The assessment stations are shown in green, these are all transects or permanent quadrants. (B) The habitat ascribed to each reference station based on a TWINSPAN analysis of the inter-tidal and sub-tidal data separately

Table 26 Comparison of the new defined habitats based on biological clustering and the ZES ecotopes. In total 771 stations are assigned to a ZES ecotope and a new habitat. The counts of the ecotope combinations are shown

new- habitats Zes ecotope	14	15	16	17	4	5	6	7	Sum rows
3					3				3
4	3	19							22
6	1	9	3			3	6		22
7		1				2			3
8		1					1		2
11	10	18	21	1		1	5		56
12	8	**41**	20	1					70
13	3	2	4						9
14		3	5	3		30	16	1	58
15			1		10	**55**	11	8	85
16					**66**	15	9	3	93
17					4	2			6
19	11	10	**40**	**45**	1		**34**	2	143
20	22	25	32	38	2	15	15	4	153
21		1	14	30					45
sum columns	58	130	140	118	86	123	97	18	

The ZES ecotopes and the clustering based on a large inter-tidal and a large sub-tidal sample agree reasonably well with each other (Table 26) Group 15 has the largest number of samples assigned to the Brackish low dynamic shallow water habitat (12). The major part of group 16 and also group 17 falls in ZES ecotope marine high dynamic shallow water (19). But group 16 has more stations in brackish high and low dynamic waters (11 and 12) while group 17 has more Gully stations (21). The intertidal groups 4 and 5 match good with the ZES ecotopes, 4 links up with middle littoral low dynamic (16) and group 5 corresponds with marine low dynamic low littoral. Group 6 rated as intertidal based on the sampling method mainly matches with habitat 19, which is marine high dynamic shallow water. The ZES ecotopes capture part of the variation in the biological community, perhaps less levels are already sufficient.

4.3.2.3.3 Discerned habitats

It was decided to use the grouping of stations based on the biological data rather than the ZES ecotopes for the further assessment. This has several advantages:

- One that the sparse assessment stations are divided over less levels (Compare Figure 35 and Figure 39B).
- Second that it corresponds better to the less complex German typology of the Wadden Sea which makes comparisons between the countries easier.
- Third that it is sure that the grouping is biologically relevant (see higher).

Six habitats are characterized as described above. For only four habitats data are available for an assessment. These are all the three defined littoral habitats and only the brackish sub-littoral habitat. In Table 27 details are given on the reference and assessment samples per habitat. The positions of the assessment stations are shown as green dots in Figure 39A. These are partly overlapping reference stations at the same location. Overall the spatial coverage of the assessment stations is much less than that of the reference stations. The assessment period covers three years from 2003 to 2005. Samples used for the assessment are all collected in spring.

Table 27. Habitats distinguished in the Wadden Sea, with number of reference stations and total sample surface per habitat. Tran. , pulled samples from transects; point, individual samples

		Number of samples		Sample surface m-2	
Habitats Wadden Sea	nr	reference	assessment	reference	Assessment
High Littoral Mud	4	86(tran.)	24(tran.)	85.70	15.94
Middle Littoral Muddy Sand	5	125(tran.)	29(tran.)	105.53	28.69
Low Littoral Sand	6	97(tran.)	6(tran.)	70.09	5.78
Brackish Sub-Littoral	15	130(point)	9(tran)	19.79	8.19
Marine Sub-Littoral Low Dynamic	16	142(point)	0	18.95	0.00
Marine Sub-Littoral High Dynamic	17	122(point)	0	19.32	0.00

4.3.2.3.4 Eco-elements

Mussel beds are important eco-elements in the Wadden Sea. In the beginning of the nineties of the last century the largest part of the inter-tidal mussel beds have been removed by fisheries. In the eastern Wadden Sea the stocks have returned, in the western Wadden Sea inter-tidal mussel beds have not reached historical levels any more. Natural sub-tidal mussel beds are not mapped in the same detail as littoral beds and less is known about sub-tidal mussel beds. Fisheries on sub-tidal mussels has not been restricted to the same extend as the fisheries on inter-tidal mussels.

The population of Pacific Oyster *Crassostrea gigas* is gradually increasing and is expected to become more dominant in the Wadden Sea ecosystem. Therefore, mapping of this species becomes important in future monitoring of the Wadden Sea.

4.3.2.4 Reference data/settings

The reference description is based on four data sets from before 1983. The datasets from the Groninger Wad and the Piet Scheveplaat are not used because part of the species are only semi-quantitatively estimated, in number e.g. *Heteromastus filiformis* and in biomass like *Arenicola marina* and *Mya arenaria*. After 1988 the same measuring procedures are followed in all available monitoring series. Therefore the Groninger Wad and Piet Scheveplaat data series will only be used for the assessment.

The datasets used for the reference description are from:

- A Wadden Sea wide littoral survey by J.J. Beukema (NIOZ) between 1971 and 1974 (Beukema 1976), 99 transects in total.
- A second littoral survey by J.J. Beukema (NIOZ) in 1977 covering the western part of the Dutch Wadden Sea with 42 transects (Beukema 1978).
- A sub-littoral survey of 461 stations in the western Dutch Wadden Sea during 1981 and 1982 by R. Dekker (NIOZ) (Dekker 1989).
- The Balgzand monitoring series. Data of the late winter/early spring campaigns from 1969 to 1982 from 15 stations (12 transects and 3 quadrants).

The chosen period is the first third of period with data available. This period is also still relatively unaffected by eutrophication. Furthermore this is the only period that detailed information on species composition and biomass is available at a reasonable spatial coverage. After 1982 no data with the required detail are available with a good spatial coverage.
A disadvantage is that the sub-tidal reference is based on individual sampling points while the assessment is based on points combined along a transect. This difference in procedure can influence the variance structure making the reference less appropriate for comparison with the assessment. Similarly differences in spatial scale between reference and assessment may also bias the reference assessment comparison.

4.3.2.5 Assessment

4.3.2.5.1 Level 1: ecosystem

At the ecosystem level the assessment is only done for a subsection of the Wadden Sea. Macrobenthic biomass from the Balgzand is compared to the primary production measurements of the phytoplankton at the NIOZ jetty (Cadée and Hegeman 2002). A plot is given in Figure 40.
Over the years there is a considerable amount of variation in primary production and biomass. A large number of estimates fall outside the moderate section. Applying time lags does not improve the relationship. It should be noticed that this comparison is based on primary production estimates of phytoplankton only, whereas in the relation described by Herman et al. (1999) the system primary production is taken, which is the sum of the phytoplankton and microphytobenthos primary production. When taking benthic primary production of the microphytobenthos into account – based on only three years (red dots in the figure) (Philippart and Cadée 2000)– these data points shift more towards the regression line of Herman et al. (1999). It is clear that an assessment on this level cannot be done without an estimate of the benthic primary production. When this is included points in the graph will shift to the right out of

the moderate zone into the good and high zone. A further complication is the effect of transport organic material into the system, produced elsewhere. Changes in the relative contributions of North Sea production and Wadden Sea likely cause variability in the biomass primary production relationship.

Disregarding the benthic primary production the most recent years are characterized by relatively high biomass and low primary production. This might suggest an imbalance of the system, possibly caused by changes in the macro benthic species composition. Future research is needed to be able to further clarify the observed patterns. As for the other water bodies, based on the available estimates and expert judgement, the Wadden Sea is evaluated as good (score 0.7).

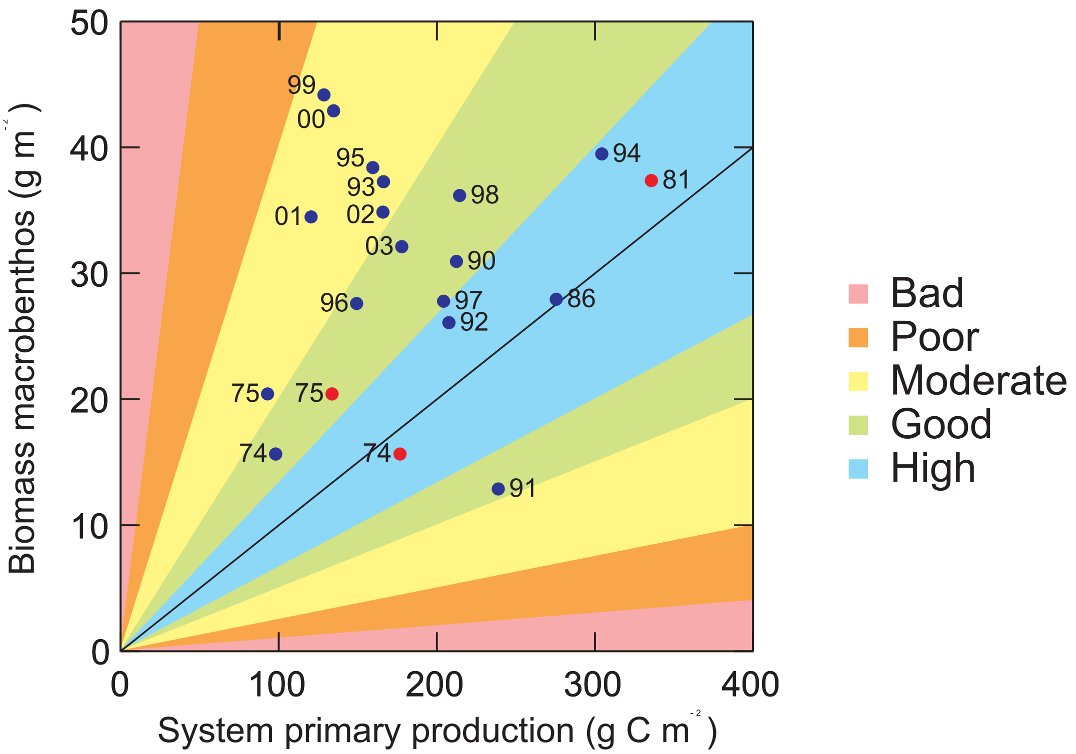


Figure 40. Biomass of the macrobenthos of the Balgzand plotted against the system primary production; blue dots estimated from the production of phytoplankton in the Marsdiep measured at the NIOZ pier (Philippart et al. 2007) and red dots are estimates of primary production based on primary production measurements of phytoplankton and microphytobenthos (data from Philippart & Cadée 2000)

4.3.2.5.2 Level 2: habitat

There are no clear targets set for the proportions of different habitats required in the Wadden Sea. It is recognized that the system is limited in its hydromorphological potential and over time the system changes. The distribution of sediment types has moved over the past decades (Zwarts 2004). In a comparison of silt content between the period 1950-1980 and the period

1980-2000 it was noted that the silt content in the central western Wadden Sea had declined while the silt content nearer to the mainland coast had increased. For a thorough habitat analysis at the second level more information on habitat distribution is needed, also reference values have to be defined.

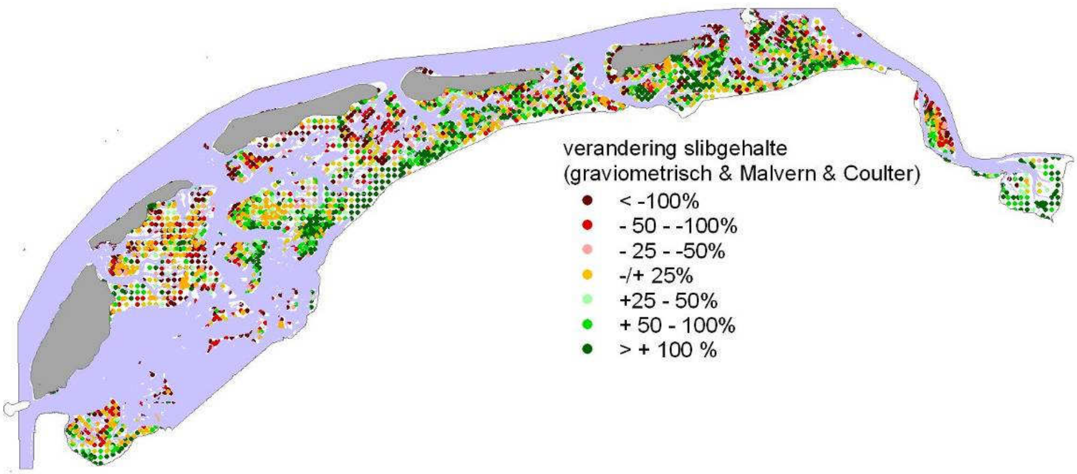


Figure 41. Changes in sediment composition between two periods, 50-80 and 80-2000. the central flats of the Wadden Sea have become coarser and the flats along the shore increased in silt (from Zwarts 2004)

The eco-element intertidal mussel beds are mapped yearly. A Wadden Sea wide mapping of intertidal mussel beds was done for 1975, based on areal photographs. This mapping led the estimate of mussel beds covering 4200 hectare in the Dutch Wadden Sea (Dijkema et al.1989). It is reasonable to use this estimate as reference condition as it is the best available estimate within the reference period 1969-1982 also applied to the third BEQI level. The boundary settings are equally scaled from this value (Table 28)

The estimates over the years 2004 – 2006 (assessment period) do not show large variations, and the mussel bed surface area is estimated on 2327 ha (average of these three years) (Steenbergen et al., 2005 and 2006).

Table 28. Assessment surface area inter-tidal mussel beds in the Wadden Sea

Level 2: habitat	Mussel banks
Areal coverage (assessment)	2327
EQR score	0.443
EQR status	moderate
Min coverage for high status	4200 ha
Min coverage for good status	3150 ha
Min coverage for moderate status	2100 ha
Min coverage for poor status	1050 ha

In the future, also the surface area of *Crassostrea edulis* has to be included as an eco-element.

4.3.2.5.3 Level 3: community (within-habitat)

Assessment was done with spring data only. Sample precision was appropriate for all habitats (Table 29). In the Table 30, the scores of the four assessed habitats are given. The high littoral mud and the brackish sub-littoral are both classified moderate. The middle littoral muddy sand habitat and the low littoral sand are both rated good. The sub-scores for biomass are bad for high littoral mud and moderate for middle littoral muddy sand and the brackish sub-littoral. In all three cases this is because the biomass is significantly higher than in the reference conditions. The brackish sub-littoral habitat has low species diversity and similarity scores as well. Similarity has a moderate score in all habitats assessed. This means that the community composition has changed significantly since the reference period.

Table 29. of assessment surface and precision class

Habitat	Assessment surface	minimal surface	OK surface	optimal surface	Assessment precision class
High Littoral Mud	15.94	1	3.98	6.97	optimal
Middle Littoral Muddy Sand	28.69	0.84	3.98	6.75	optimal
Low Littoral Sand	5.78	2.17	3.61	7.23	OK
Brackish Sub-Littoral	8.19	2.19	3.51	10.67	OK

Table 30. The assessment of level 3 for the habitats of the Wadden Sea, with indication of the assessment sampling surface, assessment parameter value, the reference boundary values and finally the EQR score and status

	parameter	Assessment		Reference boundary values										EQR	
		surface	value	Poor min	Mod min	Good min	High min	Median	High max	Good max	Mod max	Poor max	Max spp.	score	status
Brackish Sub-littoral	biomass	8.19	86	8	15	23	29	34	46	69	92	115		0.44	moderate
	density	8.19	40479	9008	18006	27012	40773	49288	58921	81573	108761	135950		0.80	high
	similarity	8.19	0.73	0.30	0.60	0.60	0.92							0.48	moderate
	species	8.19	31	11	22	33	37						43	0.56	moderate
average of parameters for Brackish sub-littoral														0.57	moderate
High Littoral mud	biomass	15.94	42	3	5	8	10	12	13	15	20	26		0.170	bad
	density	15.94	4546	505	1011	1516	2320	2855	3479	4588	6117	7646		0.610	good
	similarity	15.94	0.67	0.29	0.58	0.87	0.90							0.460	moderate
	species	15.94	35	6	13	19	22						27	1.000	high
average of parameters for high littoral mud														0.560	moderate
Middle littoral muddy sand	biomass	28.69	49	9	18	27	30	33	35	40	54	67		0.470	moderate
	density	28.69	2776	312	623	935	2003	2793	3778	5755	7674	9592		0.996	high
	similarity	28.69	0.68	0.30	0.60	0.90	0.93							0.450	moderate
	species	28.69	44	9	17	26	26						37	1.000	high
average of parameters for Middle littoral muddy sand														0.729	good
Low littoral sand	biomass	5.78	14	2	4	6	9	11	14	21	27	34		0.850	high
	density	5.78	368	42	84	126	198	291	765	3878	5167	6459		0.960	high
	similarity	5.78	0.58	0.23	0.47	0.70	0.79							0.490	moderate
	species	5.78	26	6	11	17	21						33	0.880	moderate
average of parameters for Low littoral sand														0.795	good

4.3.2.5.4 Integration of the three levels

For the overall assessment of the Wadden Sea, the ecological score and status obtained for each of the three levels are averaged into a metric representative for the whole water body. The averaging is done with a weighing factor for each level, 1 for the ecosystem level and 2 for the habitat and community level. The average of the three levels is 0.58, which correspond with a moderate status (Table 31).

Table 31. Ecological quality score and status obtained by average the parameters at each level

	EQR score	EQR status	Remark
Level 1: ecosystem	0.7	good	Expert judgement
Level 2: habitat	0.443	moderate	
Level 3: community	0.653	good	
Overal EQR	0.578	Moderate	

4.3.2.6 Discussion

Reference settings

Reference settings are based on a reasonable dataset. Spatial and temporal coverage is good for the littoral habitats. The sub-littoral reference condition is based on only one a survey in a period of two years. Spatial coverage is good, temporal variation is not well represented in this reference.

Assessment analysis

Species composition has changed in the past decades in the Wadden Sea. Invasive species have established themselves, *Mya arenaria* has increased in importance, and *Macoma balthica* has declined (Philippart et al. 2007). Besides species composition changes also the total biomass has increased. This is probably an effect of eutrophication, however no response of a decline in nutrients is seen yet.

After the reference period two dominant species have established themselves in the Wadden Sea. These are *Ensis directus* and *Marenzelleria viridis*. In a Detrended Correspondence Analysis (DCA) plot the community variation in the brackish sub-littoral habitat is represented in a two-dimensional plane (Figure 42). The reference stations are shown as crosses. They more or less fill a triangular area between *Heteromastus filliformis*, *Hydrobia ulvae* and *Scoloplos armiger* centered on *Macoma balthica* and *Arenicola marina*. These species are dominant in the reference period. The development of the three sub-littoral assessment stations is shown from 1990 onwards as colored lines. In 1990 the assessment stations are already at the outer edge of the variation of the reference conditions. In the years thereafter they moved more in the direction of a

community dominated by *Mytilus edulis*, *Nephtys hombergii*, *Neries virens*, and *Mya arenaria* and not in the last place the invasive species *Ensis directus* and *Marenzelleria viridis* are.
In all habitats changes in species composition have taken place. This is at least partly due to new species that established themselves in the system during the last few decades. Also shifts in the distribution of species has occurred changing the species composition per habitat (Beukema & Dekker 2005)

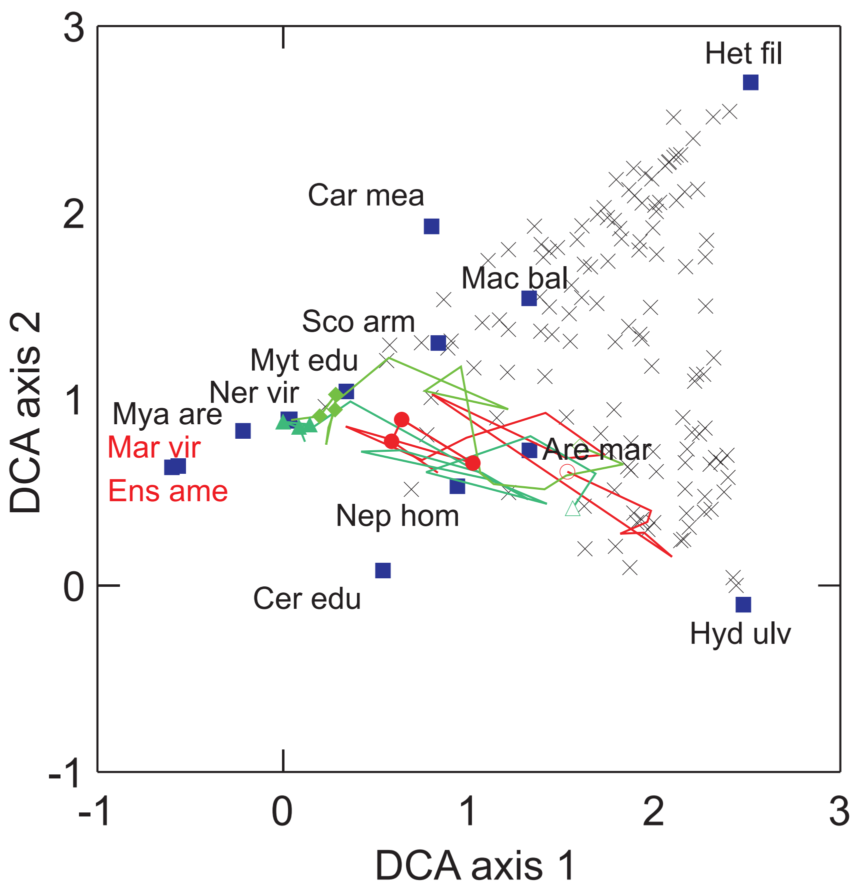


Figure 42. Detrended correspondence analysis of the macrobenthic community in the brackish sub-littoral based on biomass. *Marenzelleria viridis* and *Ensis directus* are two invasive species that cause large dissimilarities with the past. Black crosses show the brakish sub-littoral reference samples. The colored lines connect the consecutive samples at the three sampling transects in the Brackish sub-littoral from 1990 till 2006. The values for the assessment years are indicated with colored dots

The low scores are partly due to the long period between the reference description and assessment period. It is however difficult to give straight answers to what the reasons are. Climate variability has an influence on the recruitment of bivalves and will certainly have played a role, besides the earlier eutrophication. Shifts in habitat as mentioned at level two will also had an effect, especially because there was no correction applied for the possibility that stations changed from habitat.

Spatial coverage of the assessment stations is not very good, especially the sub-tidal is poorly covered. A better spatial coverage will average out possible, local effects that now could be partly causing the moderate score.
Things did not stay the same in the Wadden Sea during the past 30 years.

Advices

The following advices for the Wadden Sea can be formulated:

- The habitats, without an evaluation has to be assessed, by starting a more stratified, random sampling strategy in the future monitoring.
- The ecotope map of the Wadden Sea needs to be updated with sediment information. Based on present biological information less ecotopes may be sufficient to cover variation community structure.
- Investigations on the relation between primary production and the benthic biomass have to include benthic primary production as well as pelagic primary production. In both an assessment of the spatial variation has to be made.

4.3.2.7 Conclusion

Wadden Sea:

- pressures: eutrophication, fishery, invasive species
- Evaluation
    - Level 1: 0.7; level 2: 0.44; level 3: 0.65, overall 0.59 :Moderate
    - Mainly biomass changes and also shifts in species composition within the habitat is observed, due to the occurrence of invasive species.
    - The surface area of mussel banks has also strongly declined since the reference period.
- Advice:
    - A surveillance monitoring program is needed that better covers the spatial variability of the Wadden Sea. This monitoring program should integrate measurements on primary production (level 1), habitats (level 2) and macrobenthos (level 3). Effects of invasive species should be studied (investigative monitoring program).

4.4 Transitional waters

4.4.1 Westerschelde

The Westerschelde is a transitional water, which is characterized as a heavily modified water body. Therefore the WFD requires naming the high and good status as respectively the maximal (MEP) and the good ecological potential (GEP).

After a description of the area, the human pressures will be summarized, followed by the habitat typology. In the following section, the reference settings for the three levels of the BEQI approach will be explained, followed by an assessment for the period 2003-2005 for the Westerschelde estuary.

4.4.1.1 Short description

Figure 43. The Westerschelde (Google Earth)

The Westerschelde is the Dutch, downstream part of the Schelde estuary, with the meso-/oligohaline zone and the freshwater tidal zone of this estuary being situated in Belgium. This report only deals with the Westerschelde.

The Westerschelde (55 km long) is a well-mixed macrotidal coastal plain estuary in the southwest of the Netherlands. The estuary is tide-dominated, and experiences a semi-diurnal tide; the mean tidal range increases from 3.8 m near the mouth of the estuary to 5.0 m near the Dutch–Belgian border. Salinity ranges from a marine zone between the mouth of the estuary and Hansweert, to a brackish zone reaching beyond the Belgian border. The estuary is characterized by a complex network of flood and ebb channels surrounding intertidal flats. These flats are composed of muddy to sandy sediment. The surface area of the Westerschelde is 310 km^2, with the intertidal area accounting for 35% of the area. The average channel depth is approximately 15–20 m.

The Schelde estuary is a typical estuary, situated in a very densely populated area with a very intens economic activity, which is conflicting strongly with the high biological value (important migration route for water birds, importance of tidal flats and salt marshes) (Meire et al., 2005). Estuaries have a unique functional and structural biodiversity, which is certainly true for the Westerschelde, because it is one of the last large estuaries in NW-Europe with a multi-gully structure, large intertidal flats and salt marshes. Therefore it is important to protect this natural value. The Westerschelde is still a nature area of international importance and it is protected under the EU Bird and Habitat Directive and the Ramsar convention.

4.4.1.2 Human pressures and environmental problems

History

Since the beginning of the middle-ages, land was systematically claimed on the Schelde estuary through the building of dykes and creation of polders. The present overall form and circulation pattern of the Westerschelde as the only outflow for the river Schelde was reached after the embankment of the Kreekrak (1867) and the raising of the Sloedam (1871). As a consequence it can be stated that the geographical extent and functionality of the Westerschelde estuary as we know it today date from the beginning of the twentieth century. The year 1900 could then reasonably be used as a 'natural' reference for the geomorphologic characteristics for the estuary.

The embankment of the Braakman, a large sea-arm west of Terneuzen, in 1952 was the last major land reclame executed in the Westerschelde whereas thresholds in the navigation channels to the Antwerp harbour are regularly deepened since the beginning of the 20^{th} century.

Today's pressures

Dredging

Especially during the last two decades the dredging of the main shipping channel and subsequent dumping of the dredged material, mainly within the estuary itself, has increased substantially. The most recent large enlargement (deepening [from -14.5 to -16 NAP] and broadening) of the navigation channels was undertaken between July 1997 and July 1998 to increase the attainability of the harbor of Antwerp for larger sea-ships. In quantitative terms the yearly maintenance dredging works involve nowadays about 7 to 9 million cubic metres in the Netherlands and 3 to 4 million cubic metres in Belgium. The next enlargement of the navigation channels is planned for 2008-2009. This extension operation means clearing about 7 million m^3 from the Westerschelde (estuary as far as the Belgian-Dutch border) and roughly 7 million m^3 from the Beneden-Zeeschelde (between Belgian-Dutch border and Antwerp). This probably also means a further increase of maintenance dredging activities in the future and further changes in the geomorphologic structure of the Westerschelde estuary. A considerable amount of sand is also being extracted in the Westerschelde (about 2.5 million cubic metres of sand) so that some

dredging material is recovered as sand for use in industry. An investigation has to be made of how to deal with this issue in the future.
In the report of van den Berg et al. (2003), it is shown that there is a decrease of the shallow sub-tidal areas, a reduction of the mudflats and extension of the deepwater areas. Parallel to these developments, an increase of high dynamic littoral areas to the detriment of low dynamic counterparts seems to intensify during the second half of the 20^{the} century (Habraken & Parée, 2006). These accelerated changes seem to continue during recent years. The main consequence of these developments is a drastic reduction of the low dynamic intertidal zones that constitute a preferential habitat for most of the macrobenthic fauna (e.g. cockles) and connected food web (birds, fish, seals) (Ysebaert et al., 2003).

Eutrophication

Before the seventies, most effluents from industrial and domestic sources are discharged both directly to the Scheldt River and estuary and indirectly through sewage systems. In the Belgian part of the Scheldt, nutrient concentrations increased at a rate higher than 10% per year during the second half of the seventies (Soetaert et al., 2006). By the end of the seventies, symptoms such as temporal anoxia and massive mortality events are a common feature in the upstream part of the Scheldt. Since that period, major and still ongoing efforts are/are undertaken to reduce this load. Data compiled by Soetaert et al. (2006) show tremendous improvements in the water quality in the Westerschelde over the last two decades. Diffuse sources should continue to supply the estuary with the nutrients needed for the primary production that is and should remain mostly light limited. Indeed, whereas important decreases in nutrient concentrations are observed turbidity remained at a rather unchanged level over the last twenty years (Soetaert et al., 2006), although during recent years transparency has decreased. This means that the `historical` years (70 -80`s) are possibly representative for more disturbed conditions with respect to water quality than more recent years.

Other pressures

In the Westerschelde fishery activities are relatively finite. Professional fishery is done for eel, sprat, sole, shrimps, and cockles. Activities are mainly situated in the western part of the Westerschelde and in the mouth area.
Several micro-pollutants such as some heavy metals, PCBs, PAKs, pesticides, still are present in relatively high concentrations and for several pollutants it is expected that the WFD-standards will not be reached.

The entire extent of human activities in the estuary, occurring concurrently with the rise of the sea level, probably has contributed to an amplification of the tidal energy with elevated current speeds and water levels. The hydro-morphological developments actually observed in the Westerschelde (a change from a sand-importing system to a sand-exporting system, decrease in intertidal areas (especially tidal flats), rising of sand flats, increase of high dynamic areas in the intertidal zone) could then partly result from adjustments to this, at a geological scale, new situation. What the human share represents for the ongoing developments remains uncertain and requires urgently further investigations, but it is a fact that huge areas of intertidal mudflats and shores have been deliberately obliterated.

Main pressures Westerschelde: habitat loss, dredging and eutrophication

4.4.1.3 Habitat typology

The habitat typology of the Westerschelde is based on physical characteristics (ecotopes). This ecotope system has been derived from the ZES-ecotope system (Zoute wateren Ecotopenstelsel) developed by RIKZ for the Dutch coastal and transitional waters (Bouma et al., 2003). This system has a hierarchical structure that includes five strata: salinity < substrate < elevation < hydrodynamics < sediment characteristics. Threshold values defined for each parameter delimit condition classes wherein rather homogeneous benthic communities are expected to occur (see 4.4.1.3.1). Besides, care was also taken to set the parameter thresholds at values close to those in use in existing national and international classifications (EUNIS). Also, a biological validation of those habitats is made by testing if each habitat statistically differs in the benthic community characteristics (biomass, density, species richness, Simpson diversity index and species composition) (Baggelaar et al., 2006; Meesters, 2006; Wijnhoven et al., 2006) (see 4.4.1.3.2). At this moment, three habitat maps for the Westerschelde are made (1996, 2001 and 2004) and validated. The selected habitats of the Westerschelde for the reference and assessment analysis will be given in section 4.4.1.3.3.

4.4.1.3.1 Habitat classification parameters

In this section, the parameters and variables relevant for the habitat classification of the Westerschelde are presented; this follows the ZES classification (for a full description of the ZES – ecotope classification see Bouma et al., 2003).

Salinity

At the highest level, the water body is divided according to its salinity into a marine and brackish area. The threshold between the marine and the brackish areas has been chosen according to the Venice system. Water with an average salinity (ppt) between 5.5 and 18 (3-10 g Cl^-/l) is called brackish (mesohaline), whereas marine water has a salinity higher than 18 (10 g Cl^-/l) and is called marine (poly- and euhaline). Oligohaline waters with a salinity between 0.5 and 5.4 (0.3-3 g Cl^-/l) are not considered within the ZES ecotope system.

Substrate

The second level in the ZES system deals with the nature of the substrate, whether hard substrate or soft-sediment. In the present case only the second category has to be considered.

Elevation

At the third level of the ZES classification distinction is made between areas as function of their vertical position relative to the tidal range. The sub-littoral that is situated under the intertidal zone, remains permanently submerged. The littoral domain corresponds with the area between GLWS and GHWD (each tide submerged) and the supra-littoral is situated above the GHWD (not each tide submerged). Within each of these areas sub-divisions are made according to the vertical position.

Within the sub-littoral a distinction is made between the shallow (5m above GHWS) and the deep (5m below GHWS) sub-littoral. The depth of NAP -718cm, roughly corresponding with the upper edge of the gullies, materializes the limit between two distinct eco-morphological realms. Within the littoral three sub-areas are distinguished according to the frequency of exposure with the lower littoral being exposed from 0 up to 25% of the time, the mid-littoral with emergence frequencies between 25 and 75% and the upper-littoral that are exposed to air more than 75% of the time. Within the supra-littoral, the sub-areas are distinguished based on the frequency of submersion or vegetation zones.

Hydrodynamics

In the present case of a tide dominated system only the linear current speeds by either flood or ebb are considered whereas in other (coastal) environments also wave actions may be considered. The maximum current speed averaged over all tides is used as an indicator for the intensity of hydrodynamics. Average speed below 0.5 m/s is representative for low hydrodynamic conditions whereas high hydrodynamics conditions are depicted with average speeds higher above 0.5 m/s.

Sediment characteristics

At the last level of the ZES classification (also the closest to the benthic fauna), the habitats are divided according to their sediment composition. Four sediment classes are distinguished as muddy, fine and coarse sand and gravel. The distinction between muddy sediment and fine sand is based on the mud content (fraction < 63 µm) and the boundary is 25% mud (>= 25% for muddy sediment and <25% for fine sand. For the Westerschelde maps, the distinction between mud and fine sand is based upon aerial photographs.

Westerschelde soft-bottom habitats

Due to the hierarchical structure of the habitat classification, the combination of the different levels within each stratum (excluding hard substratum) leads to the distinction of up to 40 different habitats (2_{sal} x 1_{subst} x 5_{elev} x 2_{hydr} x 2_{sed}). But within the Westerschelde not all of them exist and due to lumping of the supralittoral habitats (because no differences in benthos characteristics are observed in the WFD data), the amount of habitats can be reduced to 22 habitats for the Westerschelde (Table 32, Figure 44).

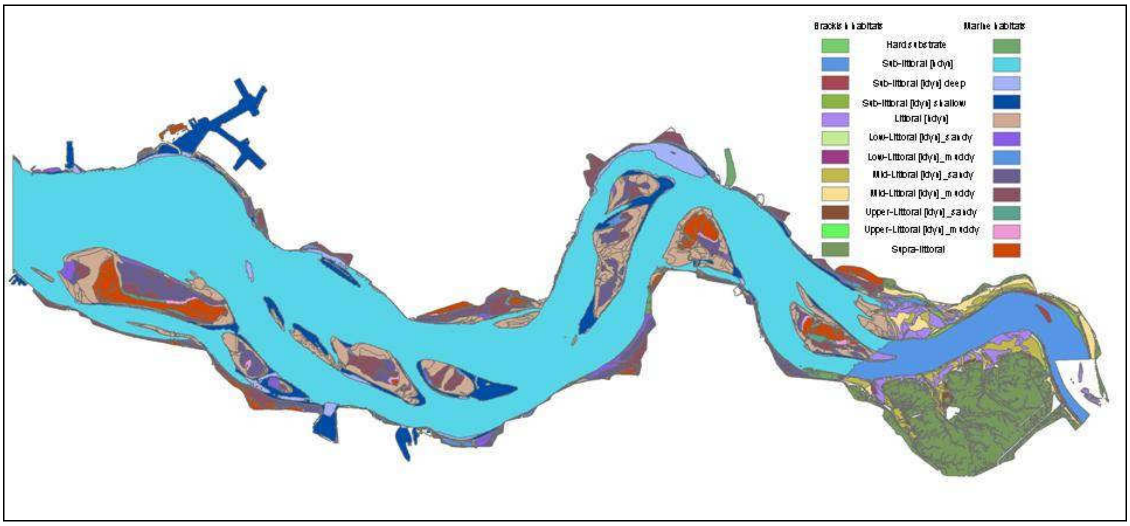


Figure 44. Habitat map of the Westerschelde (2004)

The differentiation between the habitats within the brackish and marine zone is the same. For the sub-littoral zone only a distinction between a high dynamic and a low dynamic (shallow and deep) habitat is made. It has to be noticed that the sublittoral low dynamic deep habitat is maybe not an adequate habitat (an artifact of the current velocity models?), due to the fact that the deep sub-littoral is generally characterized by relatively high currents and sandy environments (gullies of the Westerschelde).

In the intertidal area a distinction is made between high and low hydrodynamic conditions. High hydrodynamic conditions are mostly resulting in mobile sandy environments, therefore no further distinction based on sediment characteristics was considered for this habitat, as also no distinction in elevation. For the low hydrodynamic conditions a further distinction was made on elevation (low-, mid- and upper-) and for each of these zones between sandy and muddy environments, as both environments occur regularly under these conditions.

4.4.1.3.2 Biological validation of habitats

A biological validation of the different habitats is made by testing if the habitats statistically differ from each other in their benthic community characteristics (biomass, density, species richness, Simpson diversity index and species composition) (Baggelaar et al., 2006; Meesters, 2006; Wijnhoven et al., 2006). The major conclusions of these reports are:

- Difference in community characteristics could not be tested for all discerned habitats, due to the low number of samples available for some habitats and for each habitat map of the Westerschelde.
- There was no statistical difference detected in the community characteristics between the overall average of the marine and brackish habitats (Baggelaar et al., 2006). On lower habitat levels, differences are detected. This stresses the importance of a detailed characterization of the habitats within an estuary and not only a typology based on salinity zones (Borja et al., 2004).

In general and in most cases, the sub-littoral[high-dynamic] habitat and littoral[high-dynamic] habitat in the marine and brackish part differ from the other Westerschelde habitats.

- Differences between the sandy and muddy mid-littoral habitats could not be found in every report or investigated period. This lays to the fact that the boundary discerned between sand and mud of the areal pictures (20%) does not match with the biological boundary between sand and mud. There is a significant relation between the benthos density and diversity and mud content (based on samples) found (Wijnhoven et al., 2006). Normally, the muddy mid-littoral habitat are characterized by a higher density, biomass and diversity than the sand mid-littoral habitat. It is important to hold the deviation in muddy and sandy habitats, although the differentiation on the habitat maps is not perfect (but can be improved in the future, due to a better sedimentological characterisation).
- The lower-littoral and upper-littoral habitats are the habitats that are representative for the biological and physical gradient between the sub-littoral and mid-littoral to the supra-littoral. Therefore no clear biological differences are found between the lower and upper-littoral habitats and the other habitats. Another reason is the low amount of data which is available for those habitats and which made a detailed statistical investigation impossible.
- No differences between the different supra-littoral habitats are found and therefore these are pooled as one habitat.

4.4.1.3.3 Discerned habitats

Table 32. The soft-bottom habitats (with their database nummer) of the Westerschelde, with in bold the habitats that are evaluated. The total number of samples and sampling surface for the reference and assessment analysis are indicated. Only the samples taken in summer or autumn are selected for the analysis. [ldyn], low-dynamic; [hdyn], high-dynamic

		Number of samples		Total sampling surface	
habitats	Nr	Reference	Assessment	Reference	Assessment
Brackish littoral[hdyn]	**1**	**55**	**14**	**2.6069**	**0.2897**
Brackish low littoral[ldyn]_Muddy	2	0	3	0.0000	0.0716
Brackish low littoral[ldyn]_Sandy	3	5	4	0.0777	0.0866
Brackish mid-littoral[ldyn]_Muddy	**4**	**34**	**9**	**1.0436**	**0.1881**
Brackish mid-littoral[ldyn]_Sandy	**5**	**24**	**15**	**1.0963**	**0.3047**
Brackish upper-littoral[ldyn]_Muddy	9	3	0	0.1519	0.0000
Brackish upper-littoral[ldyn]_Sandy	22	0	0	0.0000	0.0000
Brackish sub-littoral[hdyn]	**6**	**115**	**44**	**3.4000**	**0.6600**
Brackish sub-littoral[ldyn] shallow	7	16	3	0.3447	0.0450
Brackish sub-littoral[ldyn] deep	20	1	2	0.1050	0.0300
Brackish supralittoral	**8**	**39**	**1**	**1.0826**	**0.0150**
Marine littoral[hdyn]	**10**	**328**	**76**	**7.3821**	**1.3171**
Marine low-littoral[ldyn]_Muddy	11	20	2	0.4042	0.0300
Marine low-littoral[ldyn]_Sandy	12	15	5	0.2271	0.1016
Marine mid-littoral[ldyn]_Muddy	**13**	**202**	**29**	**4.2562**	**0.5943**
Marine mid-littoral[ldyn]_Sandy	**14**	**152**	**65**	**3.0928**	**1.3202**
Marine upper-littoral[ldyn]_Muddy	18	5	4	0.1016	0.0866
Marine upper-littoral[ldyn]_Sandy	19	3	4	0.0450	0.0866
Marine sub-littoral[hdyn]	**15**	**628**	**210**	**27.3028**	**3.2124**
Marine sub-littoral[ldyn] shallow	**16**	**84**	**24**	**3.2162**	**0.3600**
Marine sub-littoral[ldyn] deep	21	10	5	0.1500	0.0750
Marine supralittoral	17	22	8	0.6714	0.1200

The monitoring strategy followed in the Westerschelde is based upon a random sampling strategy in four a priori defined strata. These strata are based on the depth distribution, rather than on a habitat classification. Therefore, it is reasonable to expect that many of the habitats of the Westerschelde will not be quantitatively sampled (i.e. less than 20 records) on a regular base with the current monitoring program. A selection of habitats (possibly lumped) that are quantitatively well represented in the available data will be required. The samples selected for the evaluation of the Westerschelde, are sampled in summer or autumn to exclude seasonal effects. The habitat maps of 1996 and 2004 are used to realize the coupling between respectively the macrofauna reference (<1996, see further) and assessment data (2003-2005) and the 22 habitats. Due to the different availability of data for all habitats, only 10 of them can be considered for setting reference conditions (> 20 samples and total sampling surface of > 1m^2) and only 9 for the assessment analysis (> 3 samples needed) (Table 32).

In Figure 45 and Figure 46, the distribution of respectively the reference and assessment samples was plotted. These samples are taken with a randomized sampling strategy every year, except the sampling points along the intertidal transects of the MOVE monitoring campaign.

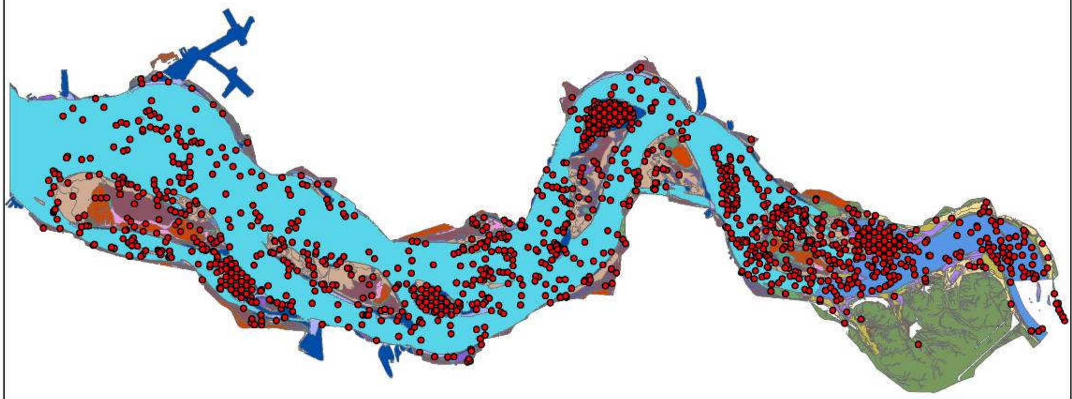


Figure 45. Habitat map of the Westerschelde of 1996 with the distribution of the reference samples

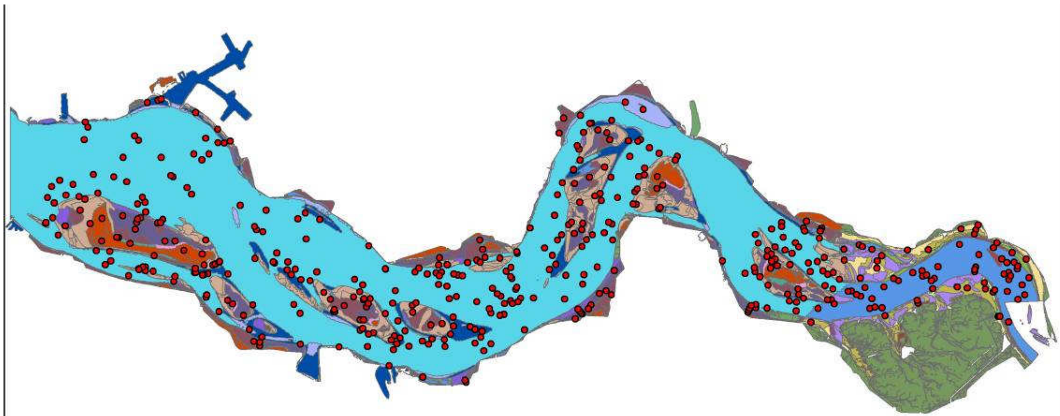


Figure 46. Habitat map of the Westerschelde of 2004 with the distribution of the assessment samples

4.4.1.3.4 Eco-elements

Mussel beds will be included as an eco-element in the ecological quality analysis of the Westerschelde at level 2, due to their ecological importance. Mussel banks provide substratum for epiflora and epifauna, while the mussel matrix provides interstices and refuges for a diverse community of organisms. The build up of mussel deposits under the bed supports a rich and dense infaunal community. Mussel beds are biogenic structures (reefs) that stabilize the sediment, profoundly modify the substratum and increase the turnover of nutrients and organic carbon in estuarine environments. This makes *Mytilus edulis* an ecosystem engineer that could play a key-role in the ecosystem functioning. In the past mussel beds have occurred in the Westerschelde, but nowadays absent.

4.4.1.4 Reference data/settings

4.4.1.4.1 Reference selection

The Westerschelde is considered as a heavily modified water body regarding the records of systematic land-reclamations back to the first millennium of our era (Escaravage et al., 2004). During the last 50 years the anthropogenic pressures, like euthropication and dredging, strongly increased and had an influence on the benthic communities and habitat changes. Unfortunately, the ecological monitoring in general and the monitoring of the benthos in particular started only since the beginning of the 1980’s, i.e. long after the accomplishment of tremendous man-made transformation on the hydrodynamics and the increase of the anthropogenic pressures in the estuary. It is therefore not possible to describe what the natural macrobenthic communities of the Westerschelde should have been today without the influence of man.

In the absence of historical data, another approach is followed, which tried to select out of the available data set, the period that reflects the natural spatial and temporal variability in an area when human activity was at a minimum (or lower than in other periods for which data was available). For the Westerschelde estuary, the main pressures are: land-reclamation and dredging, which affect the geomorphological constraints of the estuary and euthrophication which affects water quality (see above). This made it difficult to define a \`natural\` \`not impacted\` reference period, which does not exist for the Westerschelde estuary. Therefore, the reference data (period) selection for the different levels of the BEQI method will be separately described and mainly based upon a selection of the best available data. For geomorphological information of the estuary, the oldest time-period with available data is 1900 (Van den Bergh et al., 2003), whereas for benthos information the oldest available data are from the beginning of the 1980\`s.

4.4.1.4.2 Reference and boundary values at level 1 (ecosystem)

The boundary settings (high = MEP; good = GEP; moderate; poor; bad) and the optimal reference state (B/P ratio = 1/10) for the evaluation at level 1 of the Westerschelde are the same as set in the overall explanation of level 1 of the BEQI approach (see chapter 2 on the BEQI method) (Figure 3).

4.4.1.4.3 Reference and boundary values at level 2 (habitat)

At level 2 (evaluation of the habitats) of the BEQI approach, there is no general set-up for the different water bodies and therefore the approach at level 2 for the Westerschelde will be in detail described in this section.

A classification in salt marshes, mudflats, sand flats, shallow subtidal areas and deep subtidal areas is used, because this habitat deviation can be tracked back longer (+ 1900; Van den Bergh et al., 2003), than the more detailed habitat classification used at level 3. The determination of the habitats at level 2 is based on other classification parameters and threshold values than the habitat determination used to construct the ecotope maps of the Westerschelde (4.4.1.3.1) and which was used on level 3 of the BEQI approach. It would be advisable to make the two approaches of habitat distinction comparable in the future, so that the evaluation at level 2 and 3 of the BEQI approach is based on the same habitat classification.

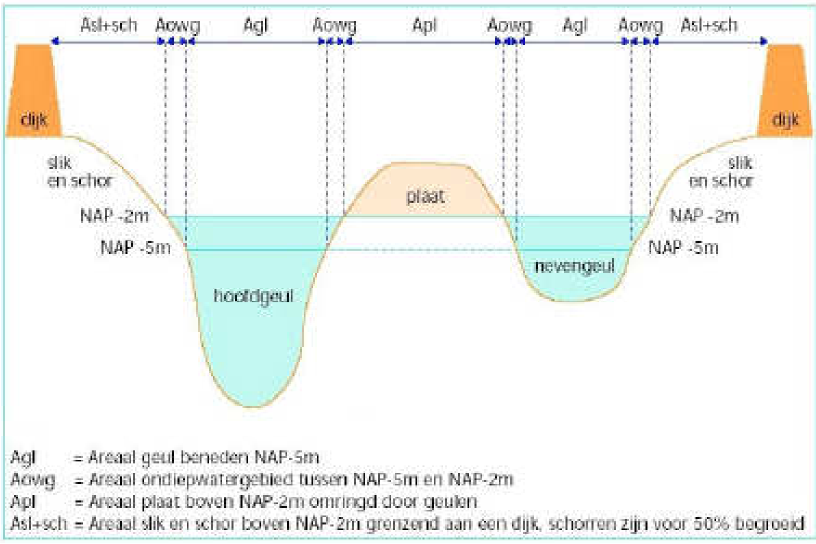


Figure 47. The habitat approach used at level 2 and reported in Holzhauer et al. (2007)

The habitat deviation used at level 2 is based on the following parameters and threshold values (Figure 47):

- The salt marshes are the areas outside the dikes which are covered with plants for more than 50%.

- The mudflats are the areas outside the dikes above the -2m NAP and bordered by a dike or salt marsh.
- The shallow subtidal areas are situated between -2m NAP and -5m NAP.
- The deep subtidal areas are situated below -5m NAP
- The sand flats are situated above -2m NAP and surrounded by gullies.

For the evaluation of the habitat level, the year 1900 could be reasonably used as a `natural` reference for the geomorphologic characteristics for the estuary (see above). Data collected in the reports by van den Berg et al. (2003) and Holzhauer et al. (2007) have been used to construct Table 33 with areal proportions and areas of the main habitats in the Westerschelde around 1900 and in 2004.

Table 33. Main habitat areal proportions and total area as in the Westerschelde around 1900 (van den Bergh et al. , 2003) and 2004 (Holzhauer et al., 2007)

Habitat type	Areal proportion (area in ha) 1900	Areal proportion (area in ha) 2004
Salt marshes	6.5 (2300)	8 (2279)
Mudflats	20 (7350)	9 (2864)
Sand flats	11 (4050)	15 (4536)
Shallow subtidal areas	20 (7350)	10 (2978)
Deep subtidal areas (> NAP -5m)	42.5 (15550)	58 (17590)

It is clear from Table 33 that considerable areas of mudflats and shallow subtidal areas have disappeared, whereas sand flats and especially deep subtidal areas increased during the last century. As a consequence, the proportions of the different habitat types have changed. Ideally, the maximal ecological potential (high status) would be the status of the year 1900, but this is not feasible. It is not possible to go back to the state of the estuary as it was around 1900, as some changes are considered as irreversible (e.g. certain infrastructures).

Nevertheless, Van den Bergh et al. (2003) proposed a series of measures that could partly compensate for the habitat loss that has occurred during the last century. Different measures are proposed, going form constructing sluices to restore connection with inland creeks, digging of old salt marshes, to depoldering, etc. Regarding depoldering, by far the most important measure, Van den Berg et al. (2003) inventory 4610 ha that could be reintegrated to the estuary in the form of salt marshes, mud flats or shallow sub-tidal areas. The proportions of these habitats are not well defined but there are assumed that equal shares evolve into salt marshes, mudflats and shallow sub-tidal areas. The realization of these measures will lead to a new equilibrium state. For the moment it is difficult to estimate what the implementation of all these measures would mean for the morphological developments of the Westerschelde. Jeuken et al. (2004) describe some of the morphological developments under different scenarios of human impacts, but large uncertainties remain.

Table 34. Main habitats areal proportion in 1900 and the MEP for the Westerschelde

Habitat type	Areal proportion 1900	Areal proportion % MEP
Salt marshes	6.5	12
Mudflats	20	15
Sand flats	11	12
Shallow sub-tidal areas	20	15
Deep sub-tidal areas (> NAP - 5m)	42.5	47

Due to these uncertainties, there is decided to use as the maximal ecological potential (MEP) the relative proportions of the different habitats by using the year 1900 as starting point, but taking into account that not all of the areas of mud flats and shallow sub-tidal areas can be restored, but only the surface area proposed by Van den Bergh et al. (2003). The MEP shows as such relatively more salt marshes, less mudflats and shallow sub-tidal areas, more or less equal sand flats and more deep sub-tidal areas as compared to 1900 (Table 34).

Parameters at level 2 and boundary settings

The choice of the habitats to be used as parameters at level 2 was made according to the following rationale. As shown in the study of Van den Bergh et al. (2003), mainly mudflats, sand flats and shallow areas have shown the most intense dynamic changes during the last 200 years whereas the changes in the extent of salt marshes and gullies are less marked at the scale of the estuary. The tremendous changes observed by these habitats result both from active land-claims and from their sensitivity to the changing hydrodynamic conditions that are possibly related to man-made activities. Furthermore these habitats are of high ecological importance for benthic macrofauna and the related higher trophic levels. These three habitats have to be included as parameter at level 2. Salt marshes are for the moment not included into the assessment, as this habitat type is part of another quality element of the Water Framework Directive. However, a link to this quality element could be made in the future. Due to socio-economic considerations that are beyond the scope of the present report, it seems not appropriate to introduce the area of the gullies in the definition of the MEP. Furthermore changes in the surface area of the gullies mostly occur to the profit or detriment of shallow areas and will be consequently indirectly tracked with our parameter. Besides the habitat parameters, also an eco-element parameter will be evaluated, namely mussel beds. There is know that in the Westerschelde, mussel banks have occurred and therefore mussel banks are evaluated as eco-element at level 2 of the BEQI approach. For mussel banks no real historical data exist for the Westerschelde but within the framework of WSV (WaterSysteem Verkenningen) an estimation of 200 ha potential mussel bed area was made and this estimation is further used as the MEP for this parameter.

Boundary settings

Four parameters (3 habitat parameters and 1 eco-element parameter) will then be used at level 2 (habitat), the areal proportion represented by the mudflats, sand flats and shallow areas and the total mussel bank area (Table 35; Escaravage et al., 2004). The divisions separating the different ecological status on this level have been set at constant intervals to allow a representative tracking of the changes.

Table 35. Assessment scale for (Escaravage et al., 2004):
1.-the proportion of both intertidal mudflats and shallow areas in the Westerschelde
2.-the proportion of sand flats in the Westerschelde
3.-the total mussel banks area (ha) in the Westerschelde

(1)

>15% MEP	**15% >…> 12% GEP**	**12% >…> 9% MODERATE**	**9% >…> 6% POOR**	**6%>… BAD**

(2)

>12% MEP	**12% >…> 9% GEP**	**9% >…> 6% MODERATE**	**6% >…> 3% POOR**	**3%>… BAD**

(3)

>200 ha MEP	**200 ha >…> 150 ha GEP**	**150 ha >…> 100 ha MODERATE**	**100 ha >…> 50 ha POOR**	**50 ha>… BAD**

4.4.1.4.4 Reference setting at level 3 (community; within-habitat)

As described above, the changes in the geomorphologic structure and water quality have consequences for the choice of the data representative for the reference state of the benthos, because the increase of physical disturbance still goes on, whereas the water quality improves. For the Westerschelde, benthos data for this project is available from 1979 to 2005. Of this time period, the years before 1997, are selected as the reference period, because:

- The overall strategy of selecting a reference data set is to select approximately the first one third of the years of the available data period.
- The period of the first one third of the years (1979 – 1987) is extended because
  - There are only a low number of samples available for the period 1979-1987.
  - To cover the spatial and temporal variability of an estuary, a longer period and more data is needed.
- The second enlargement of the navigation channels in the Westerschelde was started in 1997 and later on a third is planned, which should have its effect on the geomorphologic structure and finally on the benthos of the Westerschelde. Due to the further changes in geomorphologic structure of the Westerschelde, the later years should not be included in

the reference period. This means that the pressure `dredging` is expected to be lower in the reference period than later on.

- A disadvantage of selecting the years before 1997 is that this period is characterized by a lower water quality (with respect to nutrient levels) than more recent years, but there is assumed that euthropication is not the main problem for the benthos in the Westerschelde and this pressure can not be excluded in the reference data. This means that the pressure `euthropication` is still present in the reference period.
- This reference data set gives a good cover of the spatial and temporal variability in the Westerschelde estuary (Figure 45)

Practically, at this level for each concerned habitat (Table 32) the reference values and related boundaries are calculated for each of the four parameters (number of species, density, biomass and species composition change) in relation to the surface area sampled out of the reference data set (period 1979 – 1996). The samples selected out of the reference period, are sampled in summer - autumn and are coupled to the habitat map of 1996 to link each sample to a certain habitat.

The plots of the reference values in relation to the sampling surface are shown in annex. The reference boundary values used for the assessment are given in Table 38 and Table 39.

4.4.1.5 Assessment

The assessment of the Westerschelde can be done for the three levels of the BEQI method and in section 4.3.1.5.4 an integrated evaluation for the Westerschelde will be made. The years that are selected for the assessment are 2003, 2004 and 2005 (the three last years of the available data period). The samples selected for the assessment of the Westerschelde, are taken in summer – autumn, mainly collected in the framework of the MWTL (BIOMON) and MOVE monitoring and are coupled to the habitat map of 2004 to link each sample to a certain habitat. The data processing that was required for the assessment for each level is further described.

4.4.1.5.1 Level 1: ecosystem

At this level, assessment values for system primary production and average macrofauna biomass is needed to construct the ratio $B_{benthos}/P_{prim}$ and estimate the score and status, based on the proposed boundary settings of Figure 3.

System primary production

A comparison between 1991 and 2001 showed an increase in the primary production in the Westerschelde (Kromkamp & Peene, 2005; Van Damme et al., 2006). Based on these observations, the old estimate (100 gC/m^2 year; Herman et al., 1999) of primary production could be roughly increased with a factor 1.5 to get an estimate of 150 $gC.m^{-2}\ y^{-1}$. On the other hand, a decrease in transparency, as observed for the recent years, has lead to a reduction in primary production, which was shown by model calculations for primary production for the period 1999-2005. Therefore, it is difficult to present a good estimate of system primary production for the assessment period. Also no benthic primary production estimates for the assessment period are available. It is clear that there is a need to further investigate the primary production in the Westerschelde and which parameters influence its behavior.

Average macrofauna biomass

The average macrofauna biomass of the assessment period (2003-2005) was estimated at 11.54 g $AFDW/m^2$. This value corresponds to a plain average of all sampling points considered in the present study without consideration for habitat areas.

Ecological status at level 1

Due to the uncertainties described above about the system primary production for the Westerschelde in recent years, a direct calculation of the primary production : benthic biomass ratio is not possible. However, based on expert judgement and taking into account the reported estimates the status of the Westerschelde for level 1 is currently evaluated as GEP (Good Ecological Potential, average score 0.7).

4.4.1.5.2 Level 2: habitat

Table 36. The Ecological Quality score and status for the parameters at level 2

Level 2: habitat	Mudflat	Sand flat	Shallow areas	Mussel banks
Areal coverage (assessment)	9.50%	15.50%	9.80%	0
EQR score	0.433	1.000	0.453	0.000
EQR status	moderate	MEP	moderate	bad
Min coverage for MEP status	15%	12%	15%	200 ha
Min coverage for GEP status	12%	9%	12%	150 ha
Min coverage for moderate status	9%	6%	9%	100 ha
Min coverage for poor status	6%	3%	6%	50 ha

At this level, assessment values for areal proportions of mudflat, sand flat and shallow areas and area coverage (ha) of mussel banks is needed (Table 33,Table 36). These areal proportion values are calculated for the year 2004 (Holzhauer et al., 2007).

At this level, the parameter mudflat and shallow areas are evaluated as moderate, whereas the sand flat area reached a MEP status. The eco-element parameter, mussel banks, is evaluated as bad, due to the absence of mussel banks. At this level, an overall ecological quality score of 0.472 and a moderate status was obtained.

4.4.1.5.3 Level 3: community (within-habitat)

At this level is tried to evaluate the changes in species richness, species composition change, density and biomass for all habitats of the Westerschelde (Table 32). Only for 10 of these habitats a reference state could be determined and an assessment for the Brackish supra-littoral is not possible, due to the low number of samples (only one).

The results of the assessment of the four parameters are summarized for the brackish and marine habitats in Table 38 and Table 39 respectively. It has to be mentioned that for four habitat assessments, the sampling surface is too low to get an acceptable assessment analysis (assessment precision is unacceptable) (Table 37). Only the assessment precision for the marine littoral[hdyn], marine sub-littoral[hdyn] habitat and marine mid-littoral[ldyn]_sandy habitat is OK, whereas for the Brackish sub-littoral[hdyn] and the Marine mid-littoral[ldyn]_Muddy, the assessment precision is minimal.

Table 37. The minimal and optimal sampling surfaces needed to get an acceptable assessment analysis for the different habitats of the Westerschelde

Habitat	Assessment surface	minimal surface	OK surface	optimal surface	Assessment precision class
Brackish littoral [hdyn]	0.28	0.43	1.14	1.85	unacceptable
Brackish mid-littoral[ldyn]_Muddy	0.18	0.25	0.49	0.77	unacceptable
Brackish mid-littoral[ldyn]_Sandy	0.27	0.37	0.69	-	unacceptable
Brackish sub-littoral[hdyn]	0.65	0.24	1.36	2.22	minimal
Marine littoral[hdyn]	1.31	0.25	0.83	1.60	OK
Marine mid-littoral[ldyn]_Muddy	0.59	0.23	0.65	1.26	minimal
Marine mid-littoral[ldyn]_Sandy	1.30	0.27	0.67	1.57	OK
Marine sub-littoral[hdyn]	3.18	0.44	2.57	7.01	OK
Marine sub-littoral[ldyn] shallow	0.34	0.57	1.69	2.83	unacceptable

Table 38. The assessment of level 3 for the brackish habitats, with indication of the assessment sampling surface, parameter value, the reference boundary values and finally the EQR score and status. [hdyn] = high dynamic; [ldyn] = low dynamic. The habitats with an acceptable sampling surface for assessment are set in grey. No EQR status is defined for the habitats with an unacceptable sampling surface

Brackish habitats	parameters	Assessment surface	Assessment value	Reference boundary values Poor min	Mod min	GEP min	MEP min	Median	MEP max	GEP max	Mod max	Poor max	Max spp.	EQR score	EQR status
littoral[hdyn]	biomass	0.28	2	0	0	0	1	1	2	4	5	6		0.751	
	density	0.28	8090	127	254	382	1385	2584	4588	25205	33606	42008		0.743	
	similarity	0.28	0.75	0.16	0.31	0.47	0.67							0.848	
	species	0.28	28	3	7	10	17						35	0.922	
average of parameters for littoral [hdyn]														0.816	
mid-littoral[ldyn]_Muddy	biomass	0.18	15	1	3	4	7	9	12	20	27	34		0.723	
	density	0.18	34115	3000	6000	9000	16830	22814	27265	42760	57013	71266		0.703	
	similarity	0.18	0.75	0.24	0.48	0.72	0.80							0.671	
	species	0.18	29	6	11	17	22						32	0.940	
average of parameters for mid-littoral[ldyn]_Muddy														0.759	
mid-littoral[ldyn]_Sandy	biomass	0.27	9	0	0	0	1	3	13	32	43	53		0.862	
	density	0.27	17197	307	614	922	2763	4931	22299	54798	73064	91330		0.840	
	similarity	0.27	0.63	0.18	0.36	0.54	0.72							0.705	
	species	0.27	32	5	9	14	19						30	1.000	
average of parameters for mid-littoral[ldyn]_Sandy														0.852	
sub-littoral[hdyn]	biomass	0.65	3	0	0	0	0	1	2	5	7	8		0.712	GEP
	density	0.65	491	72	143	215	399	540	754	3215	4287	5358		0.936	MEP
	similarity	0.65	0.45	0.18	0.35	0.53	0.71							0.515	Moderate
	species	0.65	21	5	9	14	23						44	0.756	GEP
average of parameters for sub-littoral[hdyn]														0.729	GEP

Table 39. The assessment of level 3 for the marine habitats, with indication of the assessment sampling surface, parameter value, the reference boundary values and finally the EQR score and status. [hdyn] = high dynamic; [ldyn] = low dynamic. The habitats with an acceptable sampling surface for assessment are set in grey. No EQR status is defined for the habitats with an unacceptable sampling surface

	parameters	Assessment		Reference boundary values										EQR	
Marine habitats		surface	value	Poor min	Mod min	GEP min	MEP min	Median	MEP max	GEP max	Mod max	Poor max	Max spp.	score	status
littoral[hdyn]	biomass	1.31	9	1	1	2	3	4	5	8	10	13		0.526	Moderate
	density	1.31	5141	505	1011	1516	2202	2658	3284	4838	6451	8063		0.559	Moderate
	similarity	1.31	0.78	0.26	0.53	0.79	0.84							0.596	Moderate
	species	1.31	52	12	25	37	42						63	0.895	MEP
average of parameters for littoral [hdyn]														0.644	GEP
mid-littoral[ldyn]_Muddy	biomass	0.59	47	7	13	20	29	34	41	63	84	105		0.738	GEP
	density	0.59	24775	5042	10084	15126	19652	22153	24728	29540	39387	49233		0.798	GEP
	similarity	0.59	0.78	0.27	0.54	0.81	0.85							0.579	Moderate
	species	0.59	49	12	23	35	41						61	0.880	MEP
average of parameters for mid-littoral[ldyn]_Muddy														0.749	GEP
mid-littoral[ldyn]_Sandy	biomass	1.30	23	2	4	7	9	11	12	16	21	27		0.330	Poor
	density	1.30	17654	1875	3750	5624	7675	8889	10202	12672	16897	21121		0.362	Poor
	similarity	1.30	0.78	0.29	0.57	0.86	0.89							0.542	Moderate
	species	1.30	58	13	26	39	44						57	1.000	MEP
average of parameters for mid-littoral[ldyn]_Sandy														0.558	Moderate
sub-littoral[hdyn]	biomass	3.18	9	0	0	0	1	1	4	11	14	18		0.638	GEP
	density	3.18	742	91	183	274	518	1358	2415	5002	6670	8337		0.869	MEP
	similarity	3.18	0.64	0.21	0.42	0.63	0.73							0.616	GEP
	species	3.18	65	13	27	40	50						86	0.883	MEP
average of parameters for sub-littoral[hdyn]														0.751	GEP
sub-littoral[ldyn]	biomass	0.34	2	0	0	1	3	10	18	37	50	62		0.750	
	density	0.34	1197	94	188	283	1019	4241	10283	20461	27281	34101		0.819	
	similarity	0.34	0.57	0.14	0.29	0.43	0.69							0.707	
	species	0.34	31	6	13	19	29						59	0.813	
average of parameters for sub-littoral[ldyn]														0.772	

Only for one of the brackish habitats, the sub-littoral[hdyn], an acceptable assessment analysis could be done and it shows a GEP ecological quality status, with one parameter (similarity) as moderate. For only one marine habitat, the sub-littoral[hdyn], there seem to be no major changes compared to the reference state (all parameters reached at least a GEP status). The mid-littoral[ldyn]_sandy habitat is evaluated as moderate, due to the fact that the parameters density and biomass reflect a poor status (higher density and biomass in assessment compared to the reference). For the littoral[hdyn] and mid-littoral[ldyn]_Muddy habitat, the overall EQR status is GEP, but some parameters show a moderate status.

When the average is taken of the ecological status scores of the habitats with an acceptable assessment, an overall EQR score for level 3 of 0.686 (GEP status) is obtained.

It has to be mentioned that not all habitats are evaluated due to a lack of data. The overall score at level 3 reflects not the ecological status of all habitats and has to be updated in the future.

4.4.1.5.4 Integration of the three levels

For the overall assessment of the water body the ecological score and status obtained for each of the three levels are averaged into a metric representative for the whole water body. The averaging is done with a weighing factor for each level, 1 for the ecosystem level and 2 for the habitat and community level. The average (Table 40) of the three levels is 0.6, which is at the boundary of the moderate/GEP status for the Westerschelde.

Table 40. Ecological quality score and status obtained by averaging the parameters at each level

	EQR score	EQR status	Remark
Level 1: ecosystem	0.7	GEP	expert judgement
Level 2: habitat	0.472	MODERATE	
Level 3: community	0.686	GEP	
Overall EQR	0.6	MODERATE/GEP	

4.4.1.6 Discussion

Reference settings

The reference settings for the Westerschelde are not based on a `natural, not impacted` period, which does not exist. Alternatively, in essence similar but not or far less modified systems could be used as a proxy for the definition of the state of a ‘free of man’ Westerschelde. Estuaries are complex and variable systems that result from highly specific interfaces between the sea and continental waters. Two estuarine systems reproducing the same abiotic environment are not easily found and it becomes even more difficult when one of them has to be free from anthropogenic pressure, as most of the transitional waters in NW-Europe are impacted to some extent. This task is furthermore hampered by the unbalanced monitoring efforts among the different systems and the difficulty to access monitoring data from elsewhere. As a consequence it was not feasible to find a system that may serve as a reference as a whole for the Westerschelde where man-made influences are reduced at their minimum possible level.

The only way to improve the reference settings is to link the biological data to pressure data to exclude some pressure influence in the reference data. A better characterization of the ecological potentials of the habitats under the present conditions can be a helpful contribution to improve the reference settings of the Westerschelde.

Assessment results

Of all Dutch water bodies, the assessment of the Westerschelde is most complete due to the fact that this area is investigated in detail for many aspects. Therefore primary production estimates are available, estimations of the changes in surface areas of different habitats, the pressures and a lot of recent monitoring data. But the main problem in the assessment of the Westerschelde is that the locations of the monitoring did not fit well with the different habitats. Nowadays the monitoring is mainly fixed in the sub-tidal (2/3 of the samples) compared to the intertidal (1/3 of the samples). The brackish part (as defined for the habitat maps) is not enough sampled, because in the present monitoring a deviation in areas is made and not in salinity zones. Due to these facts only 5 of the 10 habitats have an acceptable assessment surface. Therefore the assessment on level 3 for the Westerschelde is not complete and has to be updated in the near future. Also the estimates of the primary production need further investigation, before the values can be used for determining the ecological status at level 1. An expert judgement is presently used.

The assessment at level 2 shows a moderate status for the Westerschelde, because there are changes observed in the surface area of certain habitats and eco-elements (mussel beds) (van den Bergh et al., 2003; Habraken & Parée, 2006; Holzhauer et al., 2007). The chosen parameters at level 2 reflected these changes linked to the man-made interventions such as dredging and dumping. Caution has to be taken in the future, with further man-made interventions, because the danger exists that the Westerschelde can loose its multiple gully system (see e.g. the changes in the Seine River).

The assessment at level 3 (community) shows that for the analyzed habitats the present state is not really different from the reference state for most parameters. There are also no clear signs that the benthic community has changed drastically in the last 40 years, because the similar species are present now as in the 1960’s (Wolff, 1973). Also no large invasion events of alien species are observed in the Westerschelde. A large invasion of the Japanese oyster (*Crassostrea gigas*), like in the Oosterschelde or the polychaete (*Marenzelleria*), like in the Eems-Dollard is not observed in the Westerschelde (they occur, but are not dominant in the system at present). It seems that there are no real changes in the occurrence of the species, but loss of valuable habitats (mud flats, characterized by high density and biomass) is the main problem. The loss of ecological valuable habitats has an effect on higher trophic levels due to a decrease in available prey items, but this is scored on level 2.

The pressure dredging, which is mainly situated in the gullies, is not reflected in the assessment results, mainly due to the following facts: (1) this pressure was also present in the reference period and affected the reference data, (2) the monitoring is not specifically situated in the dredged areas, which makes it difficult to detect direct effects of dredging and dumping, or (3) the benthos in the Westerschelde is adapted to this pressure.

An overview of the ecological quality status classification of the different habitats is given in Figure 48. There can be concluded that in total surface area, most habitats are evaluated, except in the brackish part.

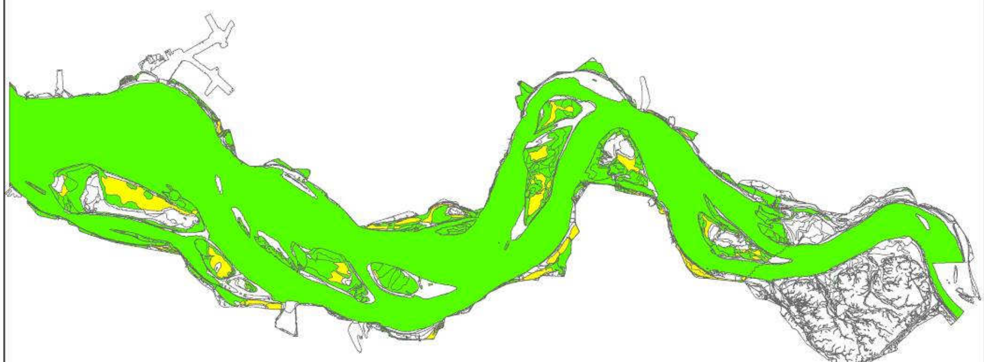


Figure 48. Map of the Westerschelde, with indication of the Ecological quality status of the different habiats: light green: GEP; yellow: moderate and white areas: no assessment possible

Advices

The following advices for the Westerschelde can be formulated:

- To improve the habitat division (especially the division between sand and mud), sedimentological information (mud, sand) has to be included by analyzing the sediment samples taken during the monitoring (not worked out at this moment).
- It would be advisable to make the two approaches of habitat distinction at level 2 and 3 comparable in the future, so that the evaluation of the BEQI approach is based on the same habitat classification.
- To improve the coupling between the monitoring and the discerned habitats (see monitoring section).

4.4.1.7 Conclusion

The Westerschelde:

- Pressure: habitat loss, dredging and euthrophication
- Evaluation:
    - Level 1: MEP; Level 2: Moderate and Level 3: GEP
    - The main pressures on the Westerschelde are evaluated at level 2 and not clearly detected at level 3, due to the absence of `historical` or `natural` benthos reference data. But the situation has not drastically changed over the last 10 years.
- Advices:
    - An operational monitoring program should be installed to evaluate further the impact of the different pressures on the ecosystem of the Westerschelde. This monitoring program should integrate further measurements on primary production (level 1), habitats (level 2) and macrobenthos (level 3). For some specific questions an investigative monitoring might be required.

4.4.2 Eems-Dollard

The Eems-dollard estuary is characterized as a heavily modified water body. Therefore the WFD requires naming the high and good status as respectively the maximal (MEP) and the good ecological potential (GEP). Assessment of this water body should be done in consultation with Germany. The following assessment is proposed by the Netherlands.

4.4.2.1 Short description

The Eems-Dollard estuary (Figure 49) is the only still remaining estuarine system with a natural outflow of fresh water connected to the Dutch sector of the Wadden Sea. It is partly situated on Dutch territory, partly on German territory.

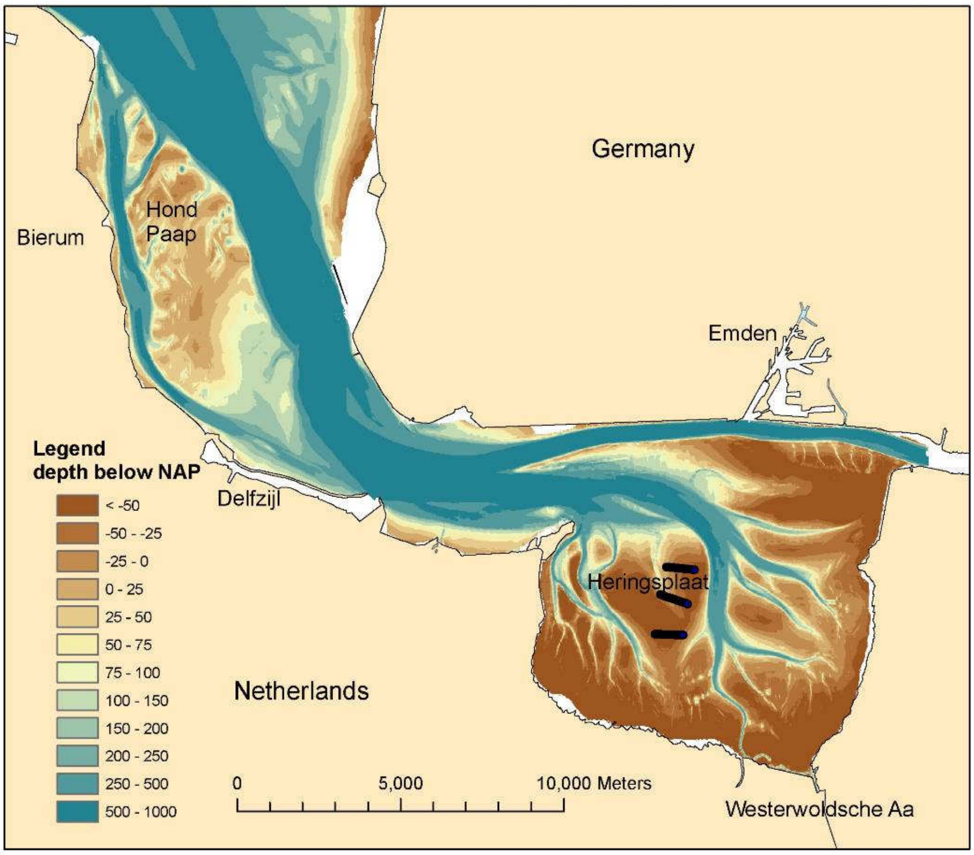


Figure 49. Depth map of the Eems-Dollard Estuary showing the three transects of the Heringsplaat monitoring-series

It is an important area for migratory and resident birds. Specialized foragers of high silt areas like Avocet, Bar Tailed Godwit and Spotted Redshank reach high desities in the Eems-Dollard Area. The salt marshes are major wintering and spring staging sites of the Russion Barnacle Goose population (50 000 which is 10% of the total population). Important haul out sites of harbor seals are found in the area. The Paap is a nowadays a major location where Seagrass *Zostrea nolti* has established itself via natural recruitment.

4.4.2.2 Human pressures and environmental problems

The Ems estuary has constantly changed over the past centuries both from man-made and natural influences. On the time scale of thousands of years, sea level rise has created the estuary and dynamically changed its boundaries. More recently, storm surges created the Dollard sub-basin in the 14th - 15th centuries. Beginning in the 16th century, diking and reclamation of land has greatly altered the surface area of the Ems estuary, particularly in the Dollard. These natural and anthropogenic changes to the surface area of the Ems altered the flow patterns of water, the tidal characteristics, and the patterns of sediment deposition and erosion. Since 1945, reclamation of land has halted and the borders of the Ems estuary have changed little (Talke and de Swart, 2006).

The Dollard area has had a long history of large loads of organically enriched waste water from potato flour and carton industry in the eastern part of the province of Groningen. Organically enriched water from the Westerwoldse Aa entered the Dollard in the south-eastern point of the estuary. In 1977 a sewage outflow was constructed in the Ems further seaward. Hereafter organic loads declined (Figure 50). The large loads of organic material caused low oxygen concentrations in the water. This caused a impoverished benthic fauna near the outflow of the Westerwoldse A.

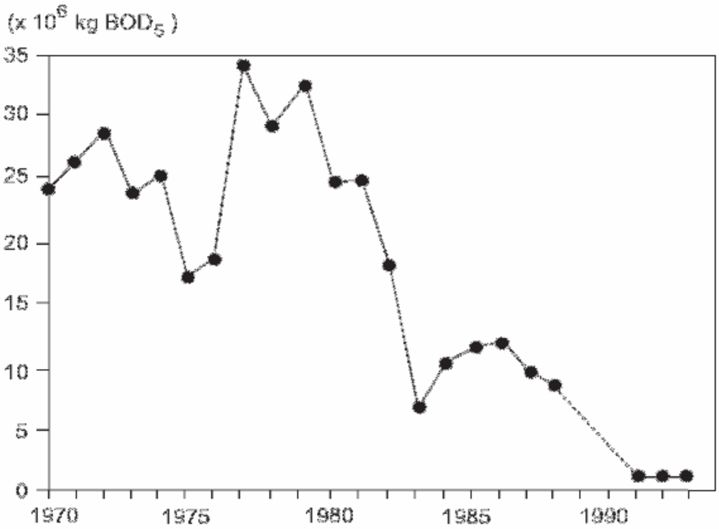


Figure 50. The organic load entering the Dollard in units of biological oxygen demand. From august through March

Benthic nematode community structure close to the sluices of the Westerwoldse Aa responded to the reduction in organic load by a change in species composition and an increase in species richness (Essink and Kiedel 1998). There was a clear direct effect of the outflow on *Nereis diversicolor* within 1.5 km from the outflow, showing very strong changes in distribution after seasonal discharges started in autumn. The macrobenthos on the Heringsplaat further away from the source of enrichment did not show clear responses tot the changes in the organic loads.

Beginning in the late 1950's, dredging activity and construction measures in harbors and shipping channels greatly altered the physical processes in the Ems. Deepening and streamlining the Ems River and shipping channel between the 1960s and 1990s decreased the hydraulic roughness and increased the tidal range in the river above Emden by as much as 1.5 m (Talke and de Swart, 2006). At the turbidity maximum between Emden and Papenburg, concentrations of sediment are currently between 1-2 orders of magnitude larger than in the 1950's, and fluid mud layers of several meters thickness occur (Talke and de Swart, 2006). Other man-made changes, such as gas pipelines and the expansion of harbors, have often caused significant, but more localized, changes to the estuary. The main pressures of today are the dredging activities in the Ems on the German side of the border.

Main pressure Eems- Dollard: dredging

4.4.2.3 Habitat typology

4.4.2.3.1 Habitat classification parameters

A ZES ecotope classification is made for the Eems-Dollard as part of the Wadden Sea ZES ecotope map (Figure 51). This ZES ecotope classification for the Wadden Sea and Eems-Dollard is based on height, emergence time, hydrodynamics and salinity (Wijsman & Verhage 2004). Salinity is an important environmental parameter structuring the macro benthic community. A salinity gradient runs from the Eastern shore towards the North Sea (Figure 52).

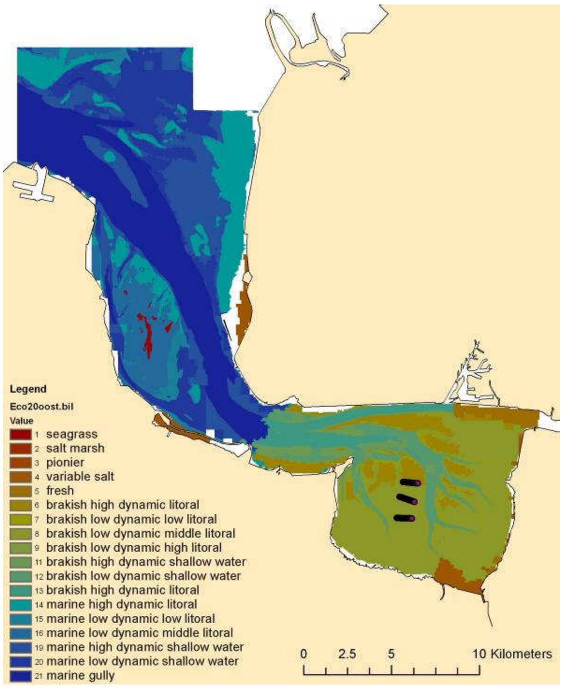


Figure 51. ZES Ecotope map of the Eems-Dollard

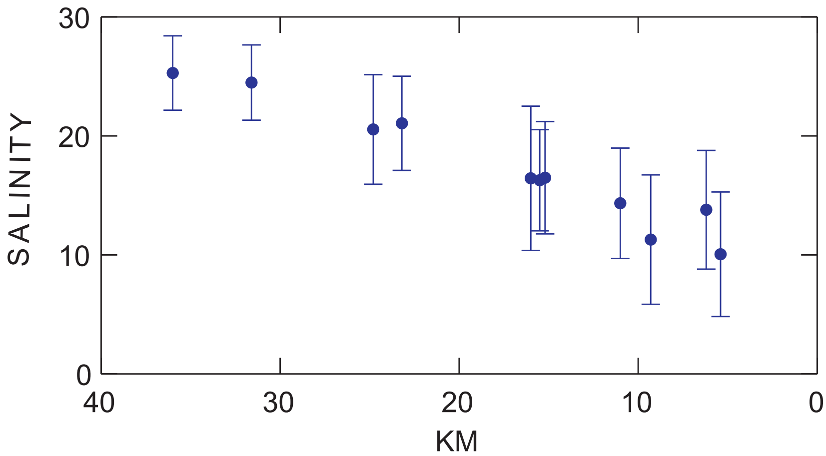


Figure 52. Salinity gradient in the Eems-Dollard. KM 0 starts at the eastern shore of the estuary

Table 41. Zes ecotopes defined in the Eems-Dollard area with their surface areas

nr	description	area (km2)
4	variable salt	9.206
6	brackish high dynamic littoral	14.3524
7	brackish low dynamic low littoral	3.5788
8	brackish low dynamic middle littoral	54.2668
9	brackish low dynamic high littoral	0.1196
11	brackish high dynamic shallow water	7.3304
12	brackish low dynamic shallow water	13.6684
13	brackish high dynamic littoral	10.7296
14	marine high dynamic littoral	32.3192
15	marine low dynamic low littoral	7.0556
16	marine low dynamic middle littoral	19.0292
19	marine high dynamic shallow water	40.7828
20	marine low dynamic shallow water	39.7972
21	marine gully	55.9988

This is a very detailed habitat division. Like in the Wadden Sea it is probably more effective to reduce the number of habitats. However at present there are no data directly available to do a biological validation. This is a point that needs to be addressed in near future.

4.4.2.3.2 Discerned habitats

From the 18 characterized habitats only 1 was represented in the reference and assessment data. Only three transects are monitored in the Eems Dollard area. These transects lay close together on one tidal flat: the Heringsplaat (Figure 51).

Table 42. The number of samples and surface area of reference and assessment samples

		Number of samples		Sample surface m-2	
Habitat	nr	reference	assessment	reference	assessment
Brackish LDyn Middle Littoral	1	48	18	25.92	9.72

4.4.2.3.3 Eco-elements

In the inner part of the Eems-Dollard no mussel or oyster beds are found. In the middle part there is some sea grass, mainly *Zostera noltii*.

4.4.2.4 Reference data/settings

Data are only available from the MWTL monitoring site Heringsplaat. Surveys done by Van Arkel and Mulder in the seventies are only assessable through reports. These data are used in the evaluation but are not in a format detailed enough to be used for reference settings. The Heringsplaat series started in 1977. In the first eleven years the methodology was a little different fromthe period 1988 onwards (see appendix for description of procedures in Dutch). In the earlier years the very common *Heteromastus filiformis* and *Hydrobia ulvae* are not quantified in the samples, only presence or absence was noted. Biomass was not measured for *Mya arenaria* and *Arenicola marina*. Because two numerous species are missing from the density data and two heavy species are missing in the biomass data the data before 1988 are unsuitable to apply to the BEQI method, and are not used.
In 1983 the invasive polychete *Marenzelleria viridis* was first encountered on the Heringsplaat (Essink and Dekker 2002). After an initial lag phase the population increased rapidly and reached maximum levels where it accounted for about half of the total macro benthic biomass (Figure 53). After 1995 the population stabilized at lower levels. It was decided to use data from 1996 to 2003 for the reference description.

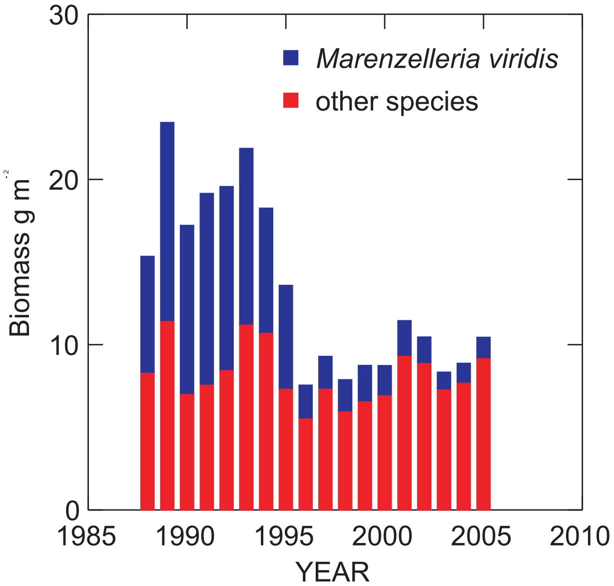


Figure 53. Total biomass of the macro benthos at the Heringsplaat from 1988 until 2005. The bars are divided in the contribution by *Marenzelleria viridis* and the part by the rest of the benthic community

This reference period is not affected by inconsistencies in the methodology and the influence of the invasive species is minimal. Because of the limited number of years both late winter and late summer samples are used for the reference description as well for the assessment (seasonal effect included).

4.4.2.5 Assessment

Assessment is done over the period 2004-2006 and additionally BEQI scores are calculated over three year periods back to 1988.

4.4.2.5.1 Level 1: ecosystem

No recent data are available for system primary production of the Eems-Dollard. Estimates during the nineteen seventies are about 200 g C Y^{-1} m^{-2} (Colijn 1983). At present the biomass estimates are around 10 g m^{-2}, but this estimate is not representative for the whole Eems-Dollard estuary. If primary production is still comparable with the situation 30 years ago the ratio would approximately be 1:20 at the good to moderate level. For an assessment a more recent estimate of the system primary production estimate is required, as well as a better estimate of the overall benthic biomass in the system. System primary production estimates need to be based on measurements of pelagic and benthic production. In the inner part of the Dollard the contribution of the pelagic production is very limited and there benthic production is the major contributor to the system primary production.
Because of these uncertainties no evaluation is made for level 1.

4.4.2.5.2 Level 2: habitat

Table 43. Development of the surface area (ha) in the transitional waters of the Ems-Dollard estuary between Pogum and Dukegat classified for sub-, inter- and supratidal habitats (Herrling & Niemeyer, 2006)

	Subtidal	Intertidal	Salt marshes (supralittoral)
1650	149.8	221.2	63.8
1750	154.1	183.3	37.1
1860	153.6	143.6	31
1960	118.1	136.5	17.9
2005	101.8	140.6	15.9

Historical information (1650, 1750, 1860, 1960) on habitats is available via the Harbasins project (Herrling & Niemeyer, 2006; De Jong, 2006) and compared with the habitats in 2000 (De Jong, 2006) and 2005 (Herrling & Niemeyer, 2006). The subtidal areas in the transitional waters of the Ems-Dollard have remained nearly constant between 1650 and 1860, though the intertidal and particularly the supralittoral areas experienced a significant reduction within that period due to land reclamation (Table 43) (Herrling & Niemeyer, 2006). Obviously there has been a phase shift between the decrease of the catchment area, subsequently reduction of the gullies and ultimate adaption of the subtidal areas to a new equilibrium. Whereas the tidal flat areas remain nearly constant between 1860 and 2005, the subtidal areas decrease by approximately 50% within that

period, particularly between 1860 and 1960. During the whole period between 1650 and 1960 the area of the supralittoral marches decreases continuously with high rates: nearly 42% between 1650 and 1750, about 16% between 1750 - 1860 and again 42% between 1860 – 1960 (Herrling & Niemeyer, 2006). Since 1960 the loss is about 11%. The high losses of supralittoral areas occur mainly due to intensive land reclamation until 1924. When the proportion between the salt marshes, littoral and sub-littoral between 1860 and 2000 is evaluated, most changes are observed near the harbors of Emden and Delfzijl, the smallest changes are near the Hond-Paap (De Jong, 2006). These calculations are a first step and further investigations are needed.
An evaluation at the second level is important in the Eems-Dollard because one of the pressures is dredging and other waterway construction works, mainly in the German part and this will further change the habitat structure in the Eems-Dollard estuary. The data are not yet transformed into a scale that fulfill the requirements of the WFD. But based on this information level 2 is evaluated as moderate, based on expert judgement.

4.4.2.5.3 Level 3: community (within-habitat)

The primary assessment was done over the period 2004-2006. Samples of both spring and autumn sampling are used. Assessment precision class was optimal (Table 44). The overall status at the third level was GEP, with ratings MEP for density and biomass, GEP for number of species and moderate for similarity (Table 45).

Table 44. Sample surface and precision class of the Eems-Dollard assessment

Habitat	Assessment surface	minimal surface	OK surface	optimal surface	Assessment power class
Brackish Ldyn Middle Littoral	9.720	0.54	2.16	3.78	optimal

Table 45. Assessment of the Eems-Dollard Brackish Mid-Littoral Ldyn habitat over the years 2004-2006

	parameter	Assessment		Reference boundary values										EQR	
Habitats		surface	value	Poor min	Mod min	GEP min	MEP min	Median	MEP max	GEP max	Mod max	Poor max	Max spp.	score	status
Brackish Ldyn Middle Littoral	biomass	8.100	9	2	5	7	9	9	10	11	14	18		0.960	MEP
	density	8.100	10846	2558	5116	7675	9299	10283	11343					0.890	MEP
	similarity	8.100	0.90	0.31	0.62	0.93	0.95							0.580	moderate
	species	8.100	17	5	11	16	18						21	0.700	GEP
average of parameters for Brackish Ldyn Middle Littoral														0.783	GEP

Table 46. BEQI scores calculated on quantitative Dollard data from Van Arkel and Mulder (1983)

	parameter	Assessment		Reference boundary values										EQR	
Habitats		surface	value	Poor min	Mod min	GEP min	MEP min	Median	MEP max	GEP max	Mod max	Poor max	Max spp.	score	status
Dollard data Van Arkel and Mulder (1983)	biomass	4.860	7	2	5	7	8	9	10	11	15	19		0.610	GEP
	density	4.860	2213	2276	4551	6827	9037	10365	11699	14220	18959	23699		0.194	bad
	similarity	4.860	0.63	0.30	0.61	0.91	0.94							0.420	moderate
	species	4.860	12	5	10	15	17						21	0.480	moderate
average of parameters														0.426	moderate

4.4.2.5.4 Integration of the three levels

At the first level no assessment was made due to missing data. The second level was rated moderate (0.5) based on expert judgement. The third level was evaluated GEP with a score of 0.78. The combined rating is 0.64, GEP ecological potential. Due to the fact that only one habitat for the Eems-Dollard is evaluate, a extra effort is needed to update the WFD classification for the Eems-Dollard as soon as possible.

4.4.2.5.5 Long term BEQI

The Dollard has been impacted by large loads of organic waste, especially in the nineteen seventies. In the nineteen eighties a new polychaete species invaded the area, *M. viridis*. To study the behavior of the BEQI index under these pressure conditions a series of assessment calculations are done covering the available data period. Over the first five periods there is a steady increase of the overall BEQI score (Figure 54). In the first two periods it was mostly biomass that had very low BEQI values. Also densities gave rise to low scores in the first period. Similarity increased from the start of the series up to the reference period. Number of species has been higher in the earlier periods compared to the later periods.
Overall the BEQI shows a clear signal of a disturbance of the community by the invader *M. viridis*.

Within the period with the highest organic loads (Figure 50) several benthic surveys are made. So one more calculation of a BEQI score is done, based on data from Van Arkel and Mulder (1983) collected on a survey in 1979. Sixty samples are collected in the inner Dollard area with a better spatial coverage than the three transects but still in the Brackish Mid-Littoral Ldyn Muddy habitat. The overall score was moderate (Table 46). Especially the sub-score for density was very low. Also number of species and similarity gave low scores.

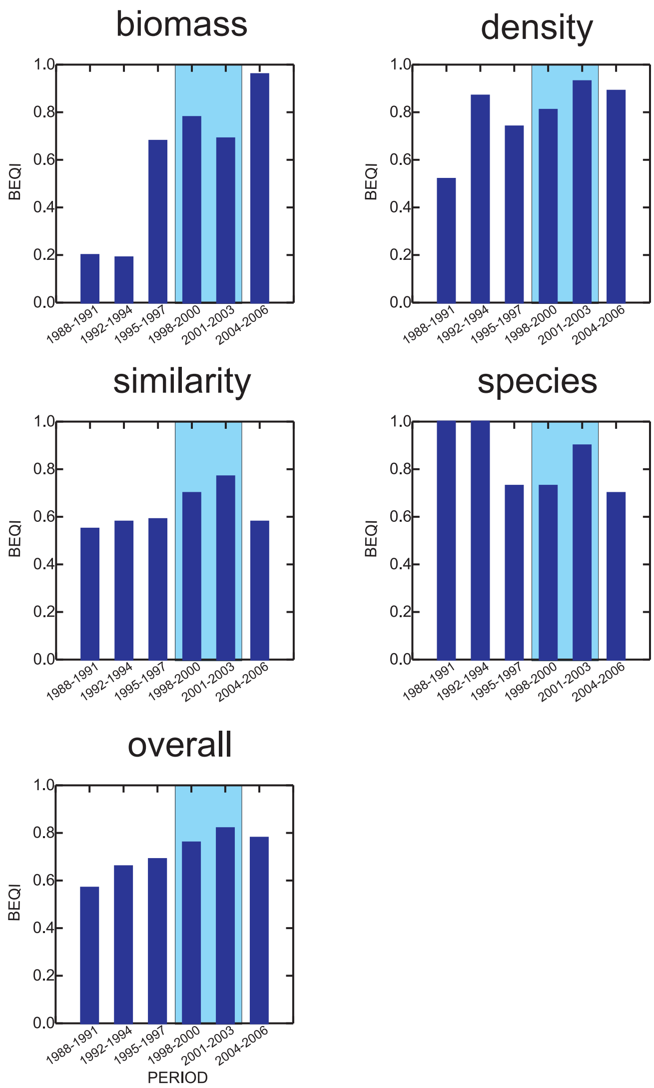


Figure 54. BEQI sub scores and the overall scores for a time series in on the Heringsplaat. The reference period is shown by a blue background

4.4.2.6 Discussion

Reference settings

The reference was chosen in the time window that is thought to be least impacted. Unfortunately no data are available for a reference description on a larger spatial scale. An extra effort is needed to fulfill this requirement.

Assessment analysis

The Eems Dollard region has been impacted by organically enriched wastewater, which was especially serious in the seventies. Near the outflow from the Westerwoldse Aa clear effects are seen in the benthic community. The Heringsplaat was so much further from the discharge point that no strong effects are observed (Essink, 2005). The data from Van Arkel and Mulder (1982) cover the entire Dollard basin. In accordance with the pressure the calculated BEQI score was moderate. The sub-scores for number off species and similarity are moderate. Density was very low leading to a score of 0.2. Intuitively a high density may have been expected as in general disturbance causes systems to shift to few small opportunistic species with high numbers. Possible explanations are that the sampling methodology used was different. Van Arkel and Mulder (1982) sampled during high water using a flushing sampler which may have a low efficiency in collecting small specimens. Applying the same sampling techniques could prevent part of these doubts.

Only one habitat was represented by the available transects. Estuaries with their salinity gradients are very diverse in habitats and communities. At present this is poorly represented in the assessment of the Eems-Dollard. In the whole water bodie several distinctly different communities exists. This is illustrated with presence absence data from a survey in 1978 by Van Arkel en Mulder (1982). A multi dimensional scaling plot (Fig. 55) clearly reveals that in 1978 the inner part had a different community than the middle and outer part of the estuary. The reference from the Heringsplaat used in the present assessment falls together with the samples from the middle section, while the Heringsplaat is located in the inner area. Is this because the communities have shifted? Or did the reduction in organic load lead to a less diverging community? Further investigations are needed.

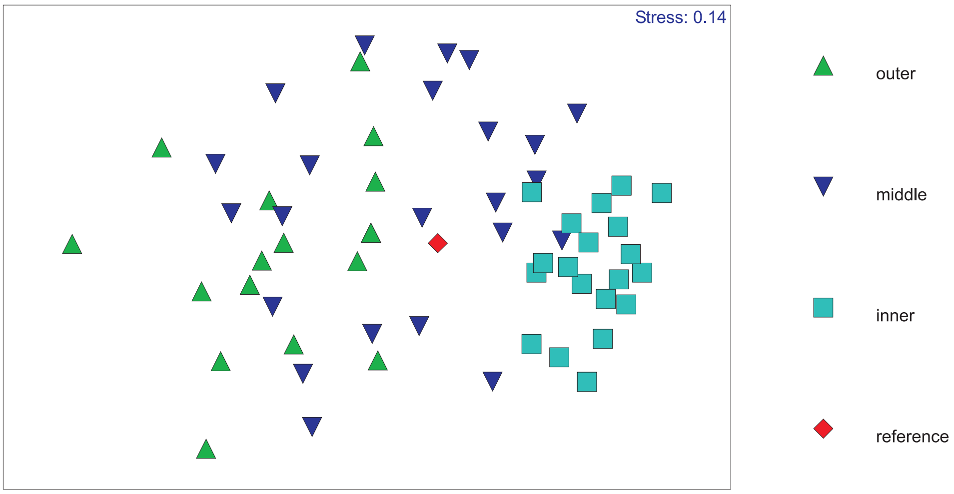


Figure 55. MDS plot of presence/absence data from Van Arkel and Mulder (1982) collected in 1978, and the present reference 1996-2003 from the Heringsplaat

Advices

The following advices for the Eems Dollard can be formulated:

- Increase the spatial coverage of the monitoring series so more habitats and areas are included in the assessment (stratified random sampling).
- A new spatial and temporal survey is needed to determine the reference conditions.
- Further development of level 2 based on the information of the Harbasins project.
- Start estimating primary production (both pelagic and benthic).

4.4.2.7 Conclusion

Eems-dollard:

- pressures: dredging and water works
- Evaluation
  - Level 2: 0.5; level 3: 0.78, overall 0.64 Good Ecological Potential
  - Mainly problems at the habitat level (Level 2)
- Advice:
  - Increase of the spatial covering of monitoring, within and between habitats.
  - Improve information for reference dataset
  - Develop implementation of BEQI level 2 habitats
  - Measure primary production, pelagic and benthic.

4.5 Saline lakes

As a result of the Delta Project, The Netherlands have created two artificial saline lakes, i.e. Lake Veere (Veerse Meer) and Lake Grevelingen, which are classified as heavily modified water bodies. Therefore the WFD requires naming the high and good status as respectively the maximal (MEP) and the good ecological potential (GEP).
The two lakes will be handled separately. After a description of the area, the human pressures will be summarized, followed by the habitat typology. In the next section, the reference settings for the different levels of the BEQI method will be explained, followed by an assessment for the period 2003-2005 for Lake Veere.

4.5.1 Lake Veere

4.5.1.1 Short description

Figure 56. Lake Veere (Google Earth)

Lake Veere (Figure 56) is an artificial lake that was previously part of the Oosterschelde estuary. Due to the flood of 1953 in the Delta area, the government started with the Delta plan, which resulted in the creation of Lake Veere as a lake (for some physical characteristics see Table 47). The construction of the Zandkreekdam (at the Oosterschelde side) started in 1960, and the sea-ward entrance was closed by the construction of the Veerse gatdam in 1961. These dams closed

Lake Veere from water exchange and created an artificial brackish lake. Consequently, Lake Veere has been characterized as a heavily modified water body.

In the lake the water level is kept in an unnatural way: in summer it is higher than in winter. The low water level in winter allows the water from the surrounding polders to be discharged into the lake, whereas in summer the water level is kept high for tourism. Besides tourism and its drainage function the lake has an important function as nature reserve.

Table 47. Physical characteristics of Lake Veere

Surface area Lake Veere	3990 ha
Water surface area NAP -0.10 m	2030 ha
Water surface area NAP -0.70 m	1742 ha
Lake volume (at NAP -0.10 m)	102 million m3
Lake volume (at NAP -0.70 m)	89 million m3
Average water depth	5 m
Maximum water depth	25 m
Length	25 km
Width	0.2 – 1.6 km

4.5.1.2 Human pressures and environmental problems

Over the years, Lake Veere developed into a brackish, eutrophic lake as fresh water input from surrounding polders increased the nutrient load. The high nutrient load results in a high primary production and a huge algal bloom in spring. *Ulva lactuca* (Sea lettuce) accumulates in some years in large decaying packets in some areas, and can cover a surface up to 30% and give rise to huge problems (Holland, 2004). These excessive blooms of Sea lettuce (*Ulva*) have become a frequent nuisance for recreational swimming and boating.

Often stratification occurs in the lake, especially in the elongated gully and deeper parts of the lake (i.e. deeper than 5 m, which is half of the lake's surface). This causes anoxia in the deeper parts of the lake. Stratification is induced by the fact that a water layer with a lower density 'floats' on a layer with a higher density. The salt water stratification that occurs in Lake Veere in summer is further amplified by temperature stratification and by the changes in the water level between winter and summer (Holland, 2004). The formed water layers are stable, allowing almost no oxygen exchange and due to oxygen consuming processes in the lowest layer, oxygen depletion (anoxia) can occur. This oxygen deficient bottom surface can stretch between 8 – 45% of the total bottom surface (Craeymeersh, 2006) and is permanent in the eastern part of the lake. The breakdown of the *Ulva* can amplify the oxygen depletion in the deeper parts of the lake. The oxygen depletion disappears again during autumn – winter. In the years 1990-1993 the percentage of the bottom experiencing a lack of oxygen (<2 mg/l) was less than 10% (Wattel, 1994). During the warm summer months of 1994 this area increased to 36 %.

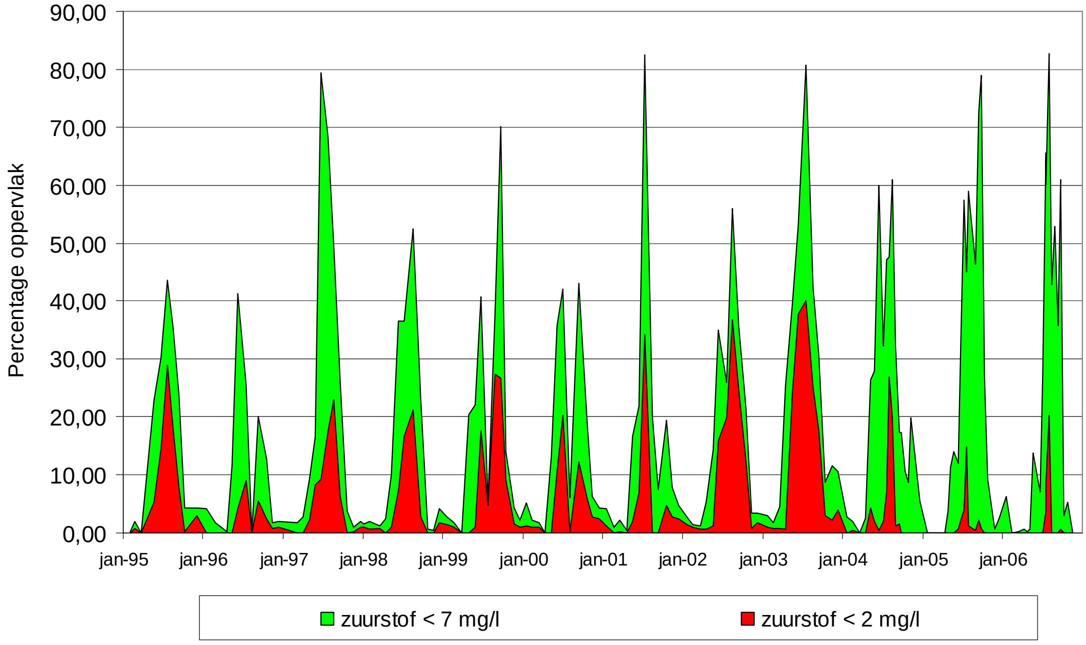


Figure 57. Percentage surface area wit a low oxygen content (< 7 mg/l) and a lack of oxygen (< 2 mg/l) (Craeymeersh, 2006)

The period when areas with low oxygen content (< 7mg/l) and especially the areas with a lack of oxygen (<2 mg/l) occur, fluctuated strongly between 1995 and 2004, but are always higher than in the period 1990-1993 (except for 1996) (Figure 57). After the opening of the sluice `Katse Heule`, the length of the periods and the surface areas with low oxygen content or with a lack of oxygen decreased again.

Therefore, the main environmental problem of Lake Veere is the oxygen depletion caused by the stratification and amplified by euthrophication. Also the transparency of the lake decreased and locally, near the harbors, also polluted sediments occur (Holland, 2004).

Due to these problems (stratification, oxygen depletion, eutrophication), the water quality of Lake Veere drastically declined. In cooperation with Rijkswaterstaat, the Province of Zeeland wants to tackle both the stratification and the eutrophication problems by increasing the exchange of water between Lake Veere and the Oosterschelde. In 1999, the Dutch parliament allocated funds to create an opening – the Katse Heule – in the Zandkreekdam. Through the opening up to 80 m^3/s of water can flow either from Lake Veere to the Oosterschelde or vice versa, depending on the tide in the Oosterschelde. The increased exchange should result in a higher salinity, lower nutrient concentrations and avoid stratification. This sluice was opened in June 2004 and is responsible for the refreshment of Lake Veere with water from the Oosterschelde.

Main pressure Lake Veere: oxygen depletion in the deeper parts due to stratification and euthrophication.

4.5.1.3 Habitat typology

4.5.1.3.1 Habitat classification parameters

The parameters determining the physico-chemical and geomorphological characteristics of Lake Veere are salinity, water level, and depth. The first parameter, which is described, is salinity. Lake Veere was a brackish water system until 2004, in which the chloride concentration fluctuated between 7 g Cl^-/l in winter (due to fresh water inputs from the surrounding land) and 12 g Cl^-/l and more in summer (Craeymeersch, 2006). Due to the water level management and the exchange of salt water at the ship sluice in the Zandkreekdam (until 2004), relatively cold salt water entered from the Oosterschelde. This, in combination with the fresh water input from the surrounding polder land, led to significant vertical differences in chloride concentration and stratification problems (Holland, 2004). The stratification effect decreased with distance from the Zandkreek sluice. This means that there was a small gradient in salinity and stratification along the longitudinal axis of Lake Veere. Due to the construction of the Katse Heule in 2004, the system changed to a more marine system (± 15 g Cl^-/l), which will be better mixed. The observations from 2004-2006 reported a chloride range of 12 – 16.5 g Cl^-/l and the water mixing was better (less stratification) (Craeymeersch, 2006).

The second parameter is the water level, which fluctuates between the winter and summer. Contrary to a water body in a natural state, the water level in Lake Veere is controlled and is maintained at NAP in summer and at NAP -0.70 m in winter. A low water level like that is needed for the drainage of the surrounding land and the groundwater level of the land outside the dikes (Holland, 2004). This water level management is currently operative, but changes are expected in the future.

The third parameter, i.e. the depth distribution, determines the geomorphological structure of Lake Veere. Lake Veere has an average depth of 5 m; only 10% is deeper than 10m (Holland, 2004). There is a central gully existing of eight 'deep wells' with a maximum depth of 15 to 24 m, which are connected with areas of a water depth of 6 to 9 m. Because of the non-tidal conditions in the lake, shallow areas are in principal more sandy due to wave action, whereas the deeper parts are characterized by muddy sediments.

Salinity changes and depth have the most influence on the benthic community composition.

4.5.1.3.2 Discerned habitats

Salinity has an effect on the benthic communities, which are in Lake Veere usually impoverished and consist of brackish and marine species. Due to the fact that there are no salinity zones in Lake Veere, a spatial habitat delineation based on the salinity characteristic is not done. Another determining factor for benthic communities and their habitat distribution is depth, because there are changes in the physical and biogeochemical characteristics along the depth gradient (sediment characteristics, stratification, oxygen depletion (see above).

The present monitoring system for the benthos (MWTL BIOMON program) is based on this depth principle and discerned three depth strata (habitats) (Table 48, Figure 58).

Table 48. The soft-bottom habitats of Lake Veere, with indication of the total number of samples and total sampling surface for the reference and assessment analysis. Habitat 1 was sampled with a flushing sampler (0.02 m^2), for habitats 2 and 3 one core (0.005 m^2) was taken from a box corer (0.068 m^2)

		Number of samples		Total sampling surface	
Habitat	Nr	Reference	Assessment	Reference	Assessment
< 2m t.o.v NAP	1	120	60	2.400	1.200
2 - 6m t.o.v. NAP	2	120	60	0.600	0.300
> 6.m t.o.v. NAP	3	99	60	0.495	0.300

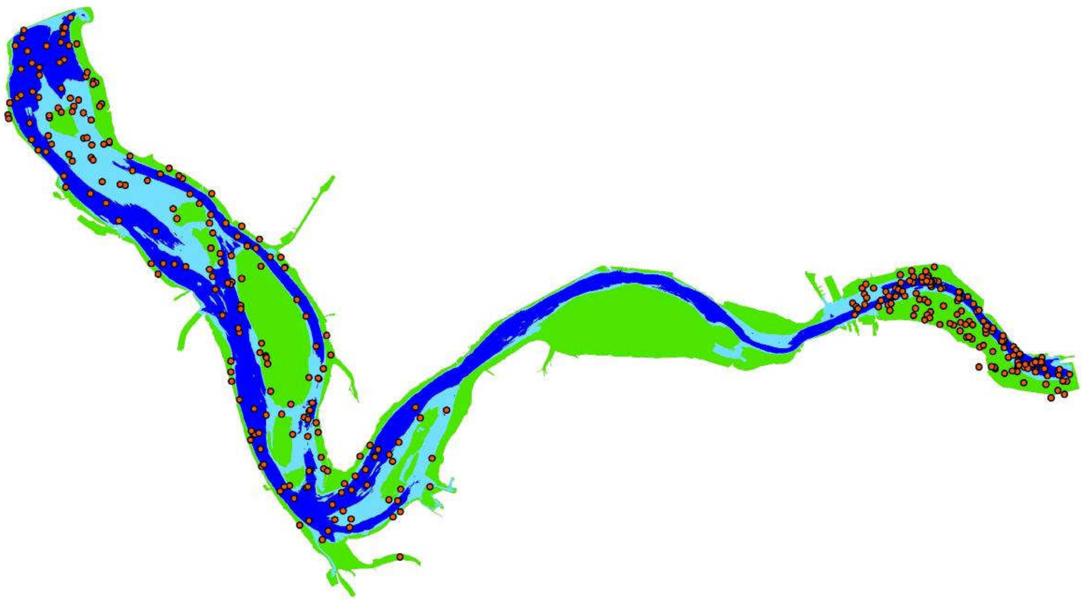


Figure 58. Position of the reference samples (period 1990-1994) in Lake Veere with indication of the three habitats (depth strata): dark blue: > 6m t.o.v. NAP, light blue: 2-6m t.o.v. NAP, green: > 2m t.o.v. NAP

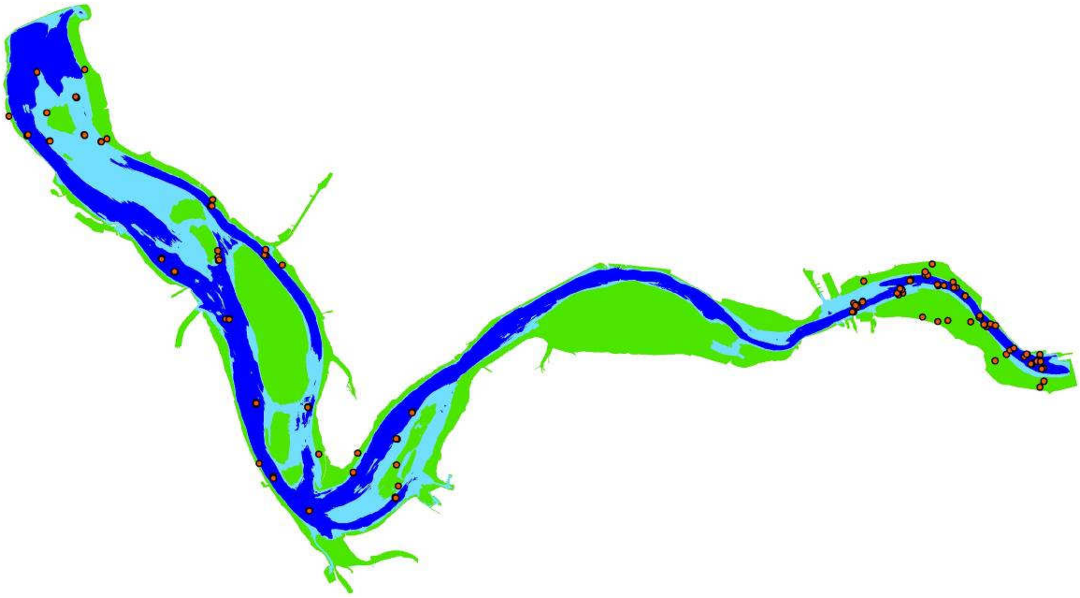


Figure 59. Position of the assessment samples (period 2003-2005) in Lake Veere with indication of the three habitats (depth strata): dark blue: > 6m t.o.v. NAP, light blue: 2-6m t.o.v. NAP, green: > 2m t.o.v. NAP

These three depth strata are used to discern the three investigated habitats for Lake Veere.
It is clear from Table 48, that, for Lake Veere, a large number of samples are taken, however with a low sampling surface per sample (resulting in a low total sampling surface, especially for the deeper parts). In Figure 58 and Figure 59, the distributions of the reference and assessment samples are plotted, respectively. The reference samples are taken based on a randomized sampling strategy every year, whereas the assessment samples are taken at fixed points that are established since 1995.

4.5.1.3.3 Eco-elements

For Lake Veere, some eco-elements are important, because the regional management plan intends to create mussel-beds and to increase the area of sea grass meadows (Craeymeersch, 2006). These could, in the future, be included as parameters at level 2 of the BEQI method for the water body evaluation. On the other hand, an invasion of the Japanese Oyster *Crassostrea gigas* – already very prominently present in the Oosterschelde – could lead to large parts of Lake Veere becoming covered with oyster beds. This should be evaluated as an undesirable evolution.

4.5.1.4 Reference data/settings

The main pressure in Lake Veere is the oxygen depletion due to stratification and euthrophication. The knowledge on pressures is important to determine the reference conditions.

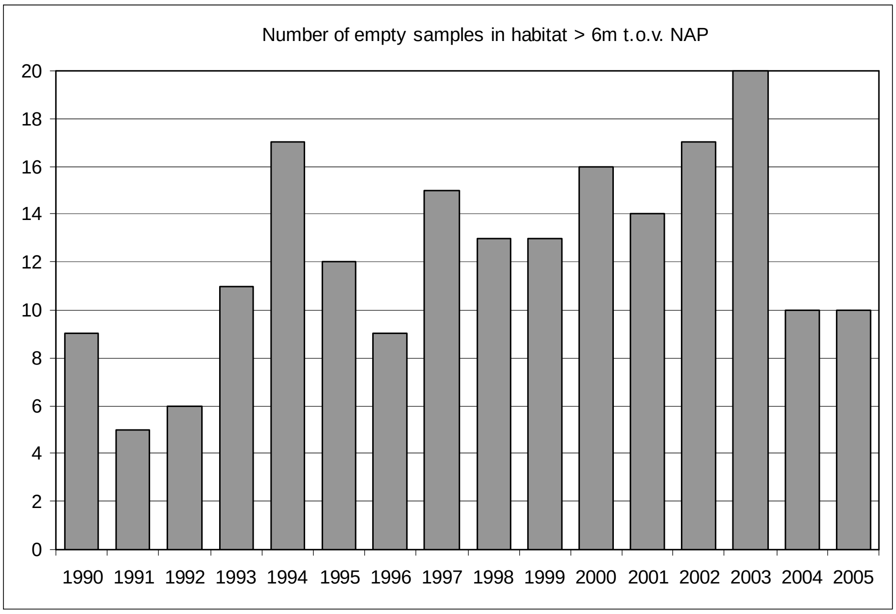


Figure 60. The number of empty samples in the habitat > 6m t.o.v. NAP in the autumn BIOMON monitoring campaign

The occurrence of periods with low oxygen contents (Figure 57) is reflected in the autumn BIOMON monitoring data of Lake Veere. The number of empty samples (no benthic organisms) taken in the habitat of > 6m NAP (the deepest parts of Lake Veere) increased towards 2003 (Figure 60). Before 1994 and after 2003 the chance to take an empty sample in that habitat is lower than 50% (< 10 of the 20 taken in the deep stratum). In the period 1994 – 2003 (except 1996), a huge number of the samples are empty, with in 2003 even all samples. These observations in the monitoring data coincide with the observation in changes in the oxygen content and with the surface area with a lack of oxygen (see above).

The pressures discussed above make it difficult to define a 'natural, not impacted' reference period for Lake Veere. Furthermore, only regular benthic monitoring data from after 1990 are available, and the recent construction of the Katse Heule caused physical changes in the lake, which are reflected in the benthic communities.

Therefore, the benthic data from before 1995 (1990 – 1994) seem to form the best reference based on the following considerations:

- This corresponds with the overall strategy of selecting a reference dataset (approximately the first one third of the years of the available data period).

- Of this period, the year 1994 for the habitat of > 6m NAP is excluded, due to the observed oxygen depletion in the deeper parts of Lake Veere in that year.
- In this reference period, Lake Veere is sampled with a randomized sampling strategy, which improves the spatial knowledge of the system. A good spatial coverage is required for the reference data set.
- This period was already characterized by euthrophication, although it still increased in the later years. Only from 2004 onwards, the euthrophication problem decreased due to the exchange with the Oosterschelde. Therefore, the beginning of the 1990's is at this moment the best choice of the reference state, despite the euthrophication.
- For the new, more saline situation, caused by the construction of the Katse Heule, no reference data are available and therefore those changes are evaluated compared to the old, more brackish situation.

The state of Lake Veere will further change in the near future due to the increase of salinity and this will affect the benthic communities. Therefore, the present reference dataset has to be investigated for its representativeness to the new situation in Lake Veere or another option is to use reference data from a similar system (like the Grevelingen).

Practically, for each discerned habitat (Table 48) the reference values and related boundaries are calculated for each of the four parameters (number of species, density, biomass and species composition change) in relation to the surface area sampled out of the reference data set (period 1990 – 1994, autumn samples only). The plots of the reference values in relation to the sampling surface are visible in the annex. The reference boundary values used for the assessment are given in Table 50.

4.5.1.5 Assessment

The ecological quality assessment of Lake Veere was done for level 1 (ecosystem) and level 3 (community level) of the BEQI method. For level 2, no assessment will be done at this moment, but advices are formulated. The years that are selected for the assessment are 2003, 2004 and 2005 (the last three years for which data are available), of which 2003 is characterized by high anoxia and 2004-2005 are the years where the Katse Heule started to operate. This assessment period is characterized by significant hydrological changes in the lake, and therefore an evolution over the separate years will also be given, next to an overall assessment. The assessment data originate from the autumn BIOMON monitoring campaign.

4.5.1.5.1 Level 1: ecosystem

At this level, assessment values for system primary production and average macrofauna biomass is needed to construct the ratio $B_{benthos}/P_{prim}$ and estimate the score and status, based on the proposed boundary settings of Figure 3.

System primary production

Estimates of phytoplankton primary production in Lake Veere are scarce. Measurements from 1980, 1982 en 1983 give yearly productions between 229 and 377 g C $m^{-2}y^{-1}$ (Wattel, 1984). In Nienhuis (1992) a value of 240 g C $m^{-2}y^{-1}$ is given for phytoplankton primary production and about 60 g C $m^{-2}y^{-1}$ for microphytobenthos production; combined this gives a value of 300 g C $m^{-2}y^{-1}$ for the system primary production. If the macroalgae are included, this increases to 450 g C $m^{-2}y^{-1}$ (Nienhuis, 1992). The estimate of 300 g C $m^{-2}y^{-1}$ was also used by Herman et al. (1999) for their relation between system primary production and macrobenthic biomass and a similar estimate is reported by de Vries et al. (1990). Recent estimates, i.e. from 1990 onwards, are not available.

Average macrofauna biomass

The average macrofauna biomass of the assessment period (2003-2005) was estimated as 29.62 g AFDW/m^2. This value corresponds to a plain average of all sampling points considered in the present study without consideration for habitat areas.

Ecological status at level 1

No recent data are available for system primary production and this makes a direct comparison with the current available benthic biomass difficult, especially because in time changes in the lake system have been described (see above: increased eutrophication, decreased transparency). How the relation will change in the future due to the new management is also unknown. However, based on expert judgement and taking into account the reported estimates the status of Lake Veere for level 1 is evaluated as GEP (Good Ecological Status, score 0.7).

4.5.1.5.2 Level 2: habitat

Due to the fact that Lake Veers is a closed system, no real changes in the surface area of the habitats are expected. For the benthic eco-elements, which are considered to be present in Lake Veere (see 4.5.1.3.3), no data was available. Therefore no evaluation of level 2 (habitat) will be made at this moment, but it is recommended to investigate the occurrence of the eco-elements in the future (see further).

4.5.1.5.3 Level 3: community (within-habitat)

At this level, it was tried to evaluate the changes in species richness, species composition, density and biomass for all discerned habitats of Lake Veere. The results of the assessment of the four parameters are summarized in Table 50. For the three habitats of Lake Veere, the assessment surface was acceptable (Table 49). For the habitat > 6m NAP it is minimal, for the habitat between 2 – 6m NAP it is OK and for the habitat < 2m NAP it is even optimal.

Table 49. Minimal and optimal sampling surfaces needed tot get an acceptable assessment analysis for the different habitats of Lake Veere

Habitat	Assessment surface	minimal surface	OK surface	optimal surface	Assessment power class
< 2m t.o.v NAP	1.20	0.28	0.48	1.04	optimal
2 - 6m t.o.v. NAP	0.3	0.08	0.235	-	OK
> 6m t.o.v. NAP	0.3	0.08	0.36	-	minimal

In Lake Veere, the habitat > 6m NAP reached a poor ecological status where the parameters biomass and density are bad (assessment biomass and density are much lower than in reference dataset). The number of species and species composition are also drastically changed (EQR: poor). The two other habitats reached an ecological quality status of moderate, due to the fact that in the habitat of 2 - 6m NAP, all parameters are classified as moderate, except the number of species. In the habitat < 2m NAP, the moderate status was mainly caused by changes in density (poor status), whereas biomass and number of species are respectively GEP and MEP. When the average is taken of the ecological status scores of the habitats of Lake Veere, an overall EQR score for level 3 of 0.439 (moderate status) is obtained.

Table 50. Assessment of level 3 for the habitats of Lake Veere, with indication of the assessment sampling surface, assessment parameter value, the reference boundary values and finally the EQR score and status. The habitats with an acceptable sampling surface for assessment are set in grey

	parameter	Assessment		Reference boundary values										EQR	
Habitats		surface	value	Poor min	Mod min	GEP min	MEP min	Median	MEP max	GEP max	Mod max	Poor max	Max spp.	score	status
< 2m t.o.v NAP	biomass	1.2	63	18	35	53	65	72	79	95	126	158		0.774	GEP
	density	1.2	6545	5301	10602	15904	18611	20189	21792	25205	33605	42006		0.257	Poor
	similarity	1.2	0.64	0.30	0.60	0.90	0.92							0.428	Moderate
	species	1.2	44	13	26	39	43						52	0.822	MEP
average of parameters for < 2m t.o.v NAP														0.570	Moderate
2 - 8m t.o.v. NAP	biomass	0.3	25	12	23	35	53	65	79	109	145	181		0.442	Moderate
	density	0.3	10427	4518	9035	13555	16771	18739	20875	25851	34467	43083		0.468	Moderate
	similarity	0.3	0.62	0.28	0.57	0.85	0.89							0.435	Moderate
	species	0.3	46	14	27	41	46						59	0.800	MEP
average of parameters for 2 - 8m t.o.v. NAP														0.536	Moderate
> 8m t.o.v. NAP	biomass	0.3	0	1	2	2	7	10	14	25	33	41		0.107	Bad
	density	0.3	327	911	1821	2732	4521	6024	7675	11475	15300	19125		0.155	Bad
	similarity	0.3	0.36	0.26	0.52	0.78	0.87							0.277	Poor
	species	0.3	13	9	17	26	32						40	0.300	Poor
average of parameters for > 8m t.o.v. NAP														0.210	Poor

4.5.1.5.4 Integration of the three levels

For the overall assessment of the water body the ecological score and status obtained for each of the levels (level 1 and 3) are averaged into a metric representative for the whole water body. The averaging is done with a weighing factor for each level, 1 for the ecosystem level and 2 for the community level. For Lake Veere no information is available for level 2. The average (Table 51) of the three levels is 0.53, which corresponds with a moderate status for Lake Veere.

Table 51. Ecological quality Ratio and status for Lake Veere obtained by averaging the parameters at each level

	EQR score	EQR status	Remark
Level 1: ecosystem	0.70	GEP	expert judgement
Level 3: community	0.44	Moderate	
Overal EQR	0.53	Moderate	

4.5.1.5.5 Evolution in level 3 over the last 10 years

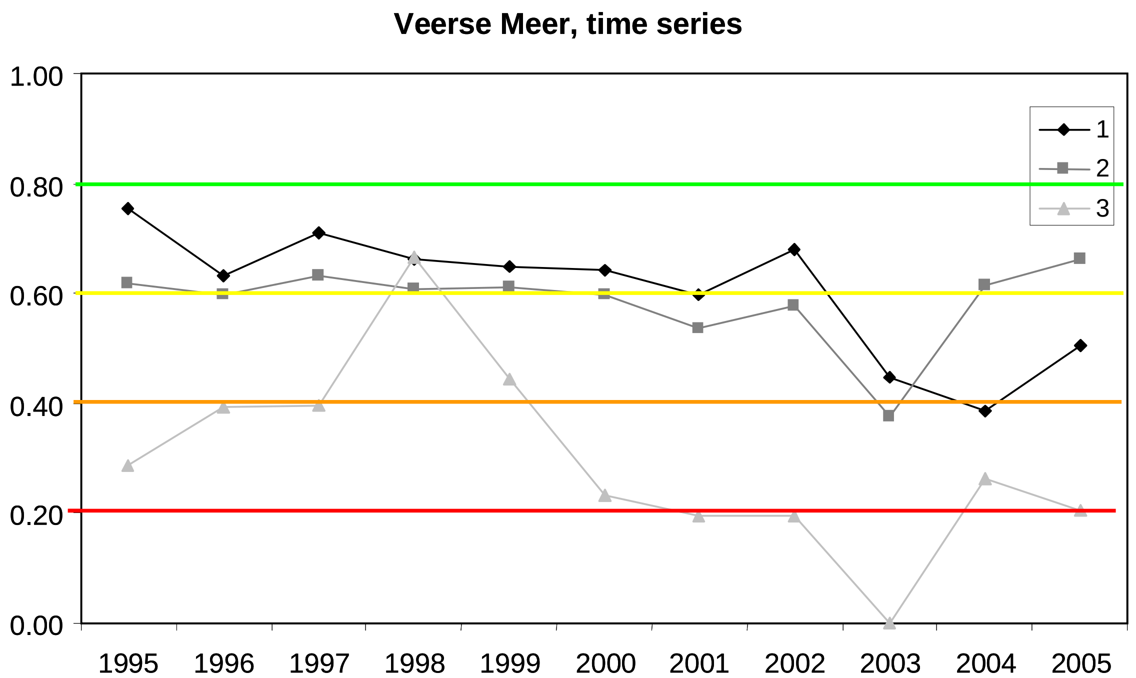


Figure 61. Time series distribution of the overall EQR at level 3 for the three habitats of Lake Veere; 1: < 2m t.o.v. NAP, 2: 2-6m t.o.v. NAP, 3: > 6m t.o.v. NAP. The colored lines show the EQR boundary classes; red: bad-poor boundary, orange: poor-moderate boundary, yellow: moderate-GEP boundary, green: GEP-MEP boundary

Due to the occurrence of oxygen depletion (pressure for the benthos) in Lake Veere, this dataset gives a good opportunity to test how the BEQI method on level 3 reacts to this pressure (Figure

61). After the opening of the Katse Heule, the water quality improved and salinity increased, which should have had an effect on the benthos and the parameters at level 3.

The results in Figure 61 clearly show that the Ecological quality declined from 1995 towards 2003, which coincided with an increase in oxygen depletion (Figure 57). The deepest part of the lake, where most of the oxygen problems occur, clearly shows the lowest EQR values. In 2003, the year with the worst situation with respect to oxygen depletion, the lowest EQR scores are observed for the three habitats. After the opening of the Katse Heule, the EQR increased again for the habitats < 2m NAP and > 6m NAP, whereas for habitat 2 - 6m NAP a higher score was observed only in 2005.

4.5.1.6 Discussion

Reference settings

The reference period of Lake Veere is not free from pressure influences, but it is at this moment the most representative period to determine changes in the state of Lake Veere. This reference has to be changed in the future due to the fact that the management strategy of Lake Veere has changed and a more marine system will be obtained. Therefore, Lake Veere will maybe more related with the Grevelingen in the future, which is an objective in the regional management plan (Craeymeersch, 2006). Investigations on the comparability of the two systems are therefore needed to find a reference dataset which better reflects the new conditions of Lake Veere. At this moment, observations show that there is no disappearance of the brackish species due to the increase of salinity (Sistermans et al., 2006), but it is expected that new, more marine species from the Oosterschelde will enter the system in the future.

Assessment analysis

The ecological functioning of Lake Veere was strongly influenced by the problems of stratification and euthrophication, causing anoxic problems in large parts of the lake. This oxygen depletion problem was tackled by increasing the exchange of water between Lake Veere and the Oosterschelde. This leads to the reduction of euthrophication and also of the occurrence of anoxic periods.

The changes in ecological functioning of Lake Veere are difficult to assess due to the fact that the estimates of the primary production was from the early 1980’s and more accurate and recent estimations are needed. Craeymeersch (2006) estimated that in the period 1993-2001, 20% of the primary production was consumed by the benthos and that for 2003-2004 this will be lower due to lower benthic biomass and the higher primary production (Wetsteyn, 2004) and that the system was no longer in balance at that moment. The benthic biomass is indeed much lower in

2003 - 2004 with respectively 12 gAFDW/m^2 and 14.8 gAFDW/m^2, compared to other years (between 31 and 96 gAFDW/m^2). The biomass increased again in 2005.

The problems in Lake Veere are reflected by the changes in the macrobenthos over the last 15 years (Escaravage & Hummel, 2003). The macrofauna of the deepest parts often completely died off and the occurrence of a lot of species in the more shallow parts was decreasing. Furthermore, a lot of benthic species showed sharp responses to the changing environmental parameters, such as salinity and oxygen (Escaravage & Hummel, 2003). These patterns of benthic changes are also reflected in the assessment with the BEQI method at level 3. The pattern in the parameter over the last 10 years (Figure 61) shows that, for the three habitats, the ecological quality status decreased, coinciding with an increase in oxygen depletion. The EQR score of the habitats increased again after the opening of the Katse Heule and is mainly related to an increase in the number of species parameter (not shown). An increase of species after the opening of the Katse Heule was also observed in Sistermans et al. (2006).

It can be concluded that the parameters at level 3 of the BEQI evaluated the observed changes in the macrobenthos and reacted to the pressures occurring in Lake Veere.

In Figure 62, the ecological quality status at level 3 of the BEQI method is visualized for each habitat of Lake Veere in a map.

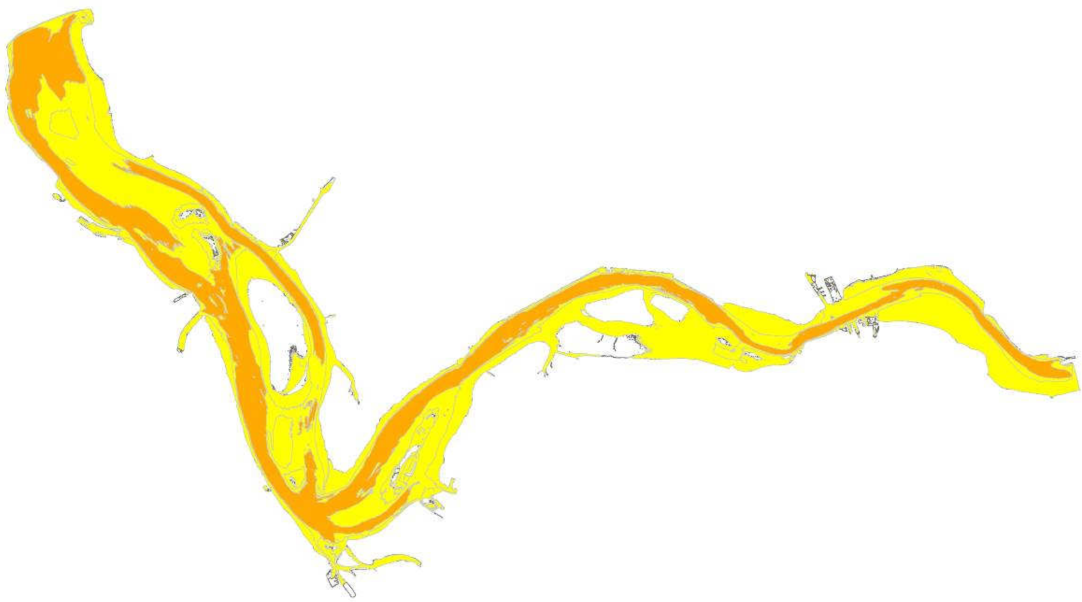


Figure 62. Chart of Lake Veere, with indication of the Ecological quality status of the three habitats: orange: poor EQR status and yellow: moderate EQR status

Advices

The following advices for Lake Veere can be formulated:

- The ecology of Lake Veere is changing under the influence of the changes in water exchange. Therefore, a more detailed investigation on the possibility to find a more adapted reference data set to the new, more marine, character of Lake Veere is advised.
- It is advisable to increase the assessment sampling surface to get a higher assessment precision (see chapter on monitoring for more information).
- To evaluate better the changes on the ecosystem functioning of Lake Veere, it would be advisable to obtain measurements for system primary production.
- The inclusion of an evaluation of the eco-elements (mussels, oysters) is advisable and has to be incorporated in the monitoring.
- Operational monitoring is needed to evaluate the changed management of the lake (changes in oxygen content, nutrients and salinity on the benthos in Lake Veere).

It has to be mentioned that the management authorities of Lake Veere have recognized the main pressure problems (oxygen depletion due to stratification) and that actions are undertaken in 2004 (increase water exchange) to solve these problems. Further monitoring is needed to evaluate the impact of these actions. Another problem (unnatural water level, which have an effect on the banks of the lake) has still to be solved in the near future.

4.5.1.7 Conclusion

Lake Veere:

- Pressures: oxygen depletion and changes in salinity
- Evaluation:
  - Level 1: GEP and level 3: Moderate
  - The evaluation shows that the present situation in Lake Veere is different from the reference period (oxygen problem, changes in salinity) and the overall status is evaluated as moderate.
  - The pressure oxygen depletion on the benthos is reflected in the parameter scores.
- Advices:
  - An operational monitoring program should be installed to evaluate the new management strategy for the lake. This monitoring program should include measurements on primary production (level 1), eco-elements (level 2) and macrobenthos (level 3).

4.5.2 Grevelingen

4.5.2.1 Short description

Figure 63. Grevelingen (Google Earth)

Grevelingen is an artificial lake (for some characteristics see Table 52) and was created in 1971 as a result of the construction of the Brouwersdam at the seaward side (Wattel, 1996). In 1965, the Grevelingendam was already constructed, which closed off the Grevelingen estuary from its connection to the large rivers of the Delta. The salinity is kept on a constant level by means of the sluice in the Brouwersdam (constructed in 1978), so that Grevelingen is a closed salt water lake (on average 16 g Cl^-/l; 29.5 ‰). Compared to Lake Veere (until 2004 a brackish lake), the salinity concentration in Grevelingen is higher and exchange with marine water was made possible. Therefore, the saline lake `Grevelingen` differs in its characteristics from the other saline lake `Lake Veere`, which is also reflected in the benthic species composition.

Due to similar problems (stratification, oxygen depletion) as in Lake Veere, the water quality of Grevelingen has changed over time (Hoeksema, 2002). Therefore, in 1999, it was decided to open the sluice in the Brouwersdam also during summer (this was only done in winter before 1990).

Table 52. Characteristics of Grevelingen (Wattel, 1996)

Water surface area	10800 ha
Surface areas outside the dikes	3120 ha
Surface area of the `afwateringsgebied`	9900 ha
Lake volume	557 million m^3
Length	23 km
Width	4-10 km
Average depth	5.4 m
Maximal depth	48 m

4.5.2.2 Human pressures and environmental problems

The Grevelingen is a heavily modified water body. The main problem in the Grevelingen is the oxygen depletion caused by stratification. In Grevelingen, this stratification is mainly caused by temperature and not by salinity differences (minor). During the summer, temperature stratification occurs in the relatively stagnant and deep water, like in the deep wells of Lake Veere. The sun warms the upper water layer, which becomes less dense than the cold deeper water layer. In absence of turbulence, the water column can not be mixed down to the bottom. A negative effect of stratification is the oxygen depletion in the deeper parts. Before 1994, this stratification and oxygen depletion remained below the accepted management level (not more than 5% of the surface may fall without oxygen). Later on, the years 1997, 1999 and 2002 are characterized by a total surface area of more than 5% without oxygen (figure 3.4 in Hoeksema, 2002). This oxygen depletion occurs mainly from the beginning of May till the end of July, and can last longer in the deeper wells. To reduce this problem, exchange with oxygen rich North Sea water was enabled during the summer. This reduces the stratification problem, but the oxygen depletion still occurs. The breakdown rates of organic matter are increased due to the increased temperature in the deeper parts, which causes a higher oxygen consumption during summer. Moreover, due to the input of sea water, the toxic algae *Phaeocystis* was introduced in Grevelingen (some years), which dies in the stagnant water and sinks to the deeper wells. The breakdown of this organic material also increases the oxygen consumption in the lake (Hoeksema, 2002).

Another important problem in Grevelingen is the increase of the turbidity (transparency of the water has decreased from 0.5 m towards 0.25 m in 2000) (Schaub et al., 2002). This caused changes in the light climate of the deeper parts of the lake, which could have an effect on the growth of benthic micro-algae, macro-algae and sea-grasses. The cause of this increase in turbidity is not known and is possibly influenced by a lot of factors (input of polder water, changes in chlorophyll content, floating dust, …) (Schaub et al., 2002).

Main pressure Grevelingen: oxygen depletion in the deeper parts due to stratification

4.5.2.3 Habitat typology

4.5.2.3.1 Habitat classification parameters

The parameters determining the physico-chemical and geomorphological characteristics of Grevelingen are salinity, water level, and depth. The salinity in Grevelingen is relatively constant over time ($\pm$ 16 g Cl^-/l; min 15 g Cl^-/l and max 19 g Cl^-/l) (Wattel, 1996; Hoeksema, 2002) and is mainly influenced by fresh water input (rain) or a lower salinity concentration at the coastal inlet. The water level of Grevelingen is constant and situated at NAP -0.20m. Grevelingen has an average depth of 5.4m, with a maximum of 48m (Wattel, 1996). The lake is characterized by some deep wells, like the Scharendijke en Den Osse, connected with less deep parts. In the study of Schaub et al. (2002), the relation between bottom parameters (depth, sedimentology) and the macrobenthos was investigated, and it was concluded that depth is the most discriminating parameter. The deeper parts are characterized by a higher mud content.
It can be concluded for Grevelingen that salinity and water level will play no crucial role for the spatial organization of the benthic community in the lake, whereas depth and sediment characteristics do characterize the benthic communities.

4.5.2.3.2 Discerned habitats

Table 53. Soft-bottom habitats of the Grevelingen, with indication of the total number of samples and total sampling surface for the reference and assessment analysis

		Number of samples		Total sampling surface	
Habitat	Nr	Reference	Assessment	Reference	Assessment
< 2m t.o.v NAP	1	115	60	2.266	1.200
2 - 6m t.o.v. NAP	2	120	59	1.800	0.885
> 6m t.o.v. NAP	3	120	57	1.800	0.855

The present monitoring system for the benthos is based on the depth principle and concerns three strata (Table 53). The depth and stratum devision seems the most important structuring parameter for the benthos in Grevelingen (Schaub et al., 2002). Therefore, these three depth strata are selected as habitats for the ecological quality evaluation of Grevelingen (Figure 64). Between the three depth strata, a difference in sedimentology (mud and phi-median) was found

(Schaub et al., 2002). For Grevelingen, a large number of samples are taken, but with a small sampling surface per sample (resulting in a low total sampling surface).

In Figure 64 and Figure 65, the distribution of respectively the reference and assessment samples is plotted. The reference samples are taken with a randomized sampling strategy every year, whereas the assessment samples are taken at fixed points every year.

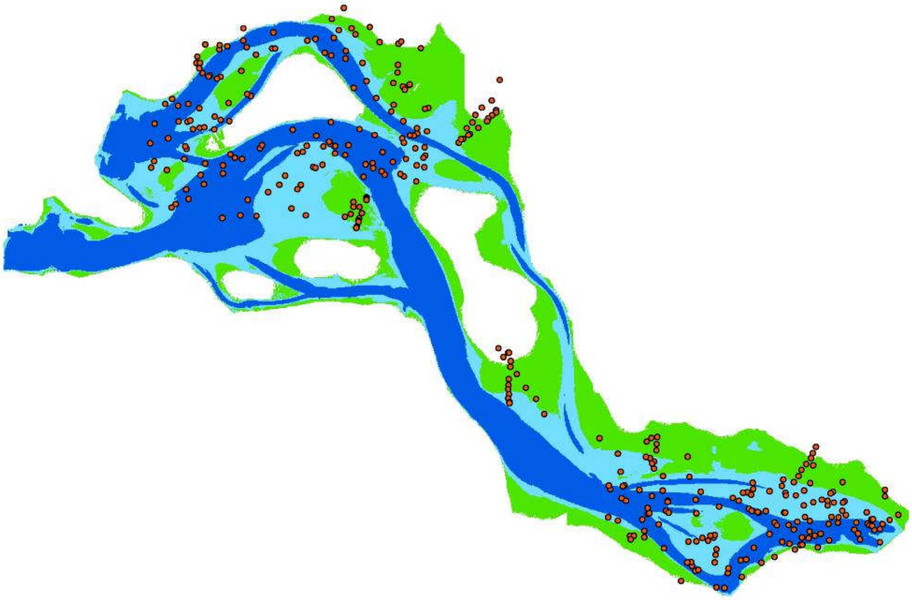


Figure 64. Position of the reference samples in Grevelingen

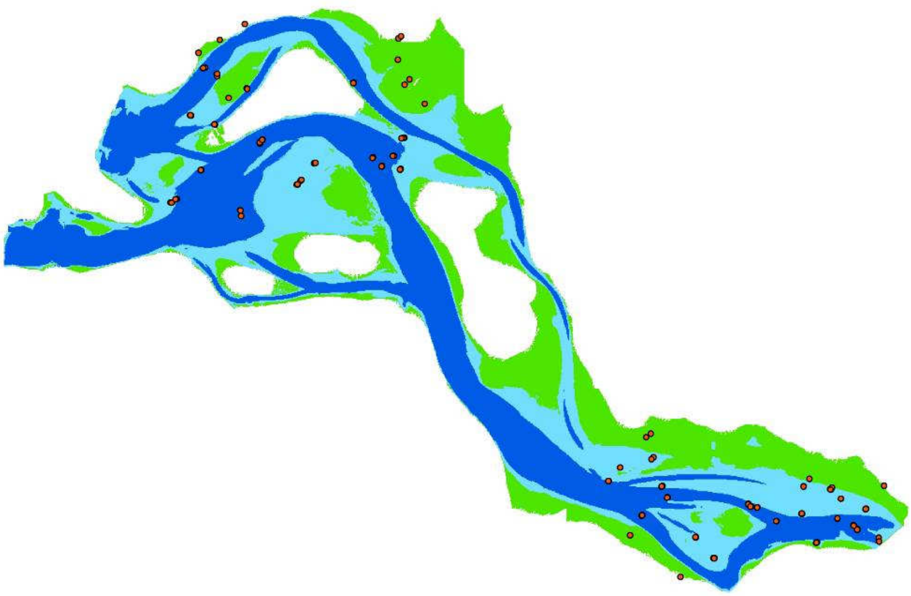


Figure 65. Position of the assessment samples in Grevelingen

4.5.2.3.3 Eco-elements

Grevelingen is the last place where the `Zeeuwse oester` *Ostrea edulis* occurs in relatively high densities and biomass, but their numbers declined during the last 10 years. *Mytilus edulis* also occurred in Grevelingen and equally showed a decline. The `Japanese oyster` *Crassostrea gigas*` was introduced in this area and increased in density and biomass in recent years (from 1998 onwards) (Schaub et al., 2002) and caused problems at some recreation sites. At this moment, no information on the surface area extent of these bivalves is available and therefore no eco-elements are evaluated. The BIOMON monitoring data are not sufficient to deliver information on the extent of the changes concerning these bivalves (eco-elements). In the future, it would be advisable to evaluate the state of these eco-elements.

4.5.2.4 Reference data/settings

The main pressure in Grevelingen is the oxygen depletion and the occurrence of periods with low oxygen concentration is also reflected in the autumn BIOMON monitoring data. The number of empty samples (no benthic organisms) taken in the habitat > 6m t.o.v. NAP increased over the last 15 years (with the most samples in 1999 – 2002) (Figure 66). Before 1995, the occurrence of empty sample is fewer. In the other habitats of Grevelingen, empty samples are only found after 1999 (< 2) (not shown). These observations in the monitoring data coincide with the observation of the oxygen depletion problems. The oxygen depletion problem is less severe in Grevelingen, compared to Lake Veere.

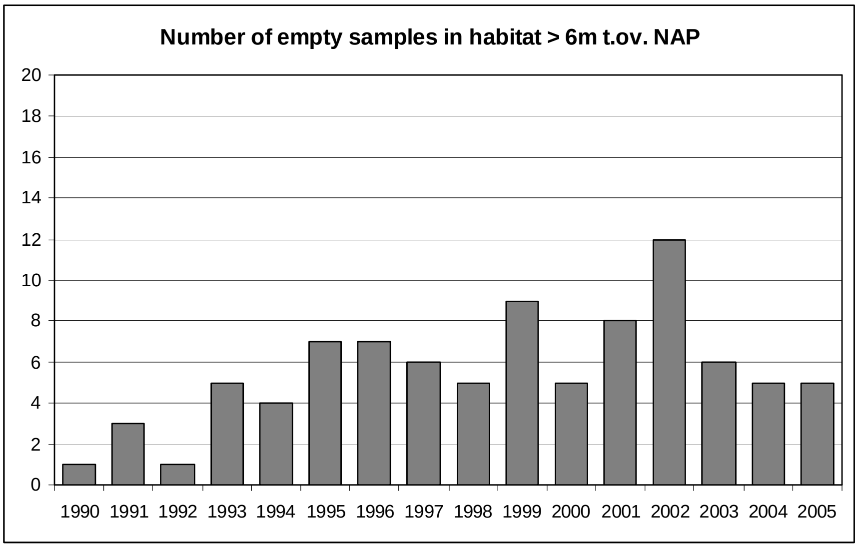


Figure 66. The number of empty samples in the habitat > 6m t.o.v. NAP in the autumn BIOMON monitoring campaign

The problems discussed above make it difficult to define a `natural` `not impacted` reference period for Grevelingen. Additionally, only benthic data of 1990 and thereafter are available.
Therefore, the benthic data from before 1995 (1990- 1994) seem to be the best reference, because:

- This corresponds with the overall strategy of selecting a reference dataset (approximately the first one third of the years of the available data period).
- This period seems to be less affected by oxygen depletion and changes in turbidity of the water than later years.
- In this reference period, the Grevelingen was sampled with a randomized sampling strategy, which improves the spatial knowledge of the system. A good spatial coverage is required for the reference data set.

Practically, for each discerned habitat (Table 53) the reference values and related boundaries are calculated for each of the four parameters (number of species, density, biomass and species composition change) in relation to the surface area sampled out of the reference data set (period 1990 - 1994). The plots of the reference values in relation to the sampling surface are shown in the annex. The reference boundary values used for the assessment are given in Table 55.

4.5.2.5 Assessment

The ecological quality assessment of Grevelingen will be done for level 1 (ecosystem) and level 3 (community level) of the BEQI method. For level 2, no EQR scores can be obtained, but advices are formulated. The years that are selected for the assessment are 2003, 2004 and 2005 and the assessment data originate from the autumn BIOMON monitoring campaign.

4.5.2.5.1 Level 1: ecosystem

At this level, assessment values for system primary production and average macrofauna biomass is needed to construct the ratio $B_{benthos}/P_{prim}$ and estimate the score and status, based on the proposed boundary settings of Figure 3.

System primary production

Only limited data are available on estimates of primary production in Grevelingen. In Nienhuis (1992) a value of about 190 g C $m^{-2}y^{-1}$ is given for phytoplankton primary production and about 70 g C $m^{-2}y^{-1}$ for microphytobenthos production; combined this gives a value of about 260 g C $m^{-2}y^{-1}$ for the system primary production. If sea grasses and some macro-algae are included on the hard substrates this increases to 320 g C $m^{-2}y^{-1}$ (Nienhuis, 1992). Herman et al. (1999) use an estimate of about 300 g C $m^{-2}y^{-1}$ for their relation between system primary production and macrobenthic biomass.

Average macrofauna biomass

The average macrofauna biomass of the assessment period (2003-2005) was estimated as 42.34 g AFDW.m^{-2}. This value corresponds to a plain average of all sampling points considered in the present study without consideration for habitat areas.

Ecological status at level 1

No recent data are available for system primary production and this makes a direct comparison with the current available benthic biomass difficult, especially because in time changes in the lake system have been described (see above, e.g. decreased transparency). However, based on expert judgement and taken into account the reported estimates the status of Grevelingen for level 1 is currently evaluated as GEP (Good Ecological Status, score 0.7).

4.5.2.5.2 Level 2: habitat

Due to the fact that Grevelingen is a closed system, no drastic changes in the surface area of the habitats are expected. For the eco-elements, which are considered to be present in Grevelingen (see 4.5.1.3.3), no data was available. Therefore no evaluation of level 2 (habitat) will be made at this moment, but it is recommended to investigate the occurrence of the eco-elements in the future.

4.5.2.5.3 Level 3: community (within-habitat)

At this level, it was tried to evaluate the changes in species richness, species composition, density and biomass for all discerned habitats of Grevelingen. The results of the assessment of the four parameters are summarized in Table 55. For the three habitats of Grevelingen, the assessment surface was acceptable (Table 54) and falls in the 'OK' assessment precision class.

Table 54. Minimal and optimal sampling surfaces needed to get an acceptable assessment analysis for the different habitats of Grevelingen

Habitat	Assessment surface	minimal surface	OK surface	optimal surface	Assessment power class
< 2m t.o.v NAP	1.20	0.28	0.67	1.32	OK
2 - 6m t.o.v. NAP	0.885	0.24	0.555	1.17	OK
> 6m t.o.v. NAP	0.855	0.285	0.585	1.155	OK

For all habitats, the species composition parameter (similarity) gives a moderate status, which means that there are changes in the species composition in those habitats, whereas the number of species remained high (MEP status). The parameters density and biomass both reached a moderate status for the habitat < 2m t.o.v NAP and 2 - 6m t.o.v. NAP, whereas for the habitat > 6m t.o.v. NAP a MEP and GEP status for respectively density and biomass. After averaging the parameters, only the 2 - 6m t.o.v. NAP habitat reached a moderate status; the others showed a GEP status.

When the average is taken of the ecological status scores of all habitats of Grevelingen, an overall EQR score for level 3 of 0.649 (GEP status) was obtained.

Table 55. Aassessment of level 3 for the habitats of the Grevelingen, with indication of the assessment sampling surface, assessment parameter values, the reference boundary values and finally the EQR score and status. The habitats with an acceptable sampling surface for assessment are set in grey

	parameter	Assessment		Reference boundary values										EQR	
Habitats		surface	value	Poor min	Mod min	GEP min	MEP min	Median	MEP max	GEP max	Mod max	Poor max	Max spp.	score	status
< 2m t.o.v NAP	biomass	1.203	17	6	12	18	26	30	35	46	62	77		0.568	moderate
	density	1.203	3343	606	1212	1818	2215	2456	2672	3232	4309	5387		0.577	moderate
	similarity	1.203	0.67	0.29	0.58	0.87	0.90							0.464	Moderate
	species	1.203	53	16	31	47	52						64	0.817	MEP
average of parametes for < 2m t.o.v NAP														0.606	GEP
2 - 6m t.o.v. NAP	biomass	0.885	61	22	45	67	88	98	112	137	182	228		0.555	Moderate
	density	0.885	5661	2367	4734	7102	8388	9111	9920	11609	15478	19347		0.486	Moderate
	similarity	0.885	0.64	0.29	0.59	0.88	0.91							0.436	Moderate
	species	0.885	72	20	39	59	64						80	0.900	MEP
average of parameters for 2 - 6m t.o.v. NAP														0.594	Moderate
> 6m t.o.v. NAP	biomass	0.855	49	11	23	34	52	61	71	92	123	154		0.769	GEP
	density	0.855	5638	1227	2455	3683	4588	5122	5650	6738	8983	11229		0.804	MEP
	similarity	0.855	0.63	0.29	0.59	0.88	0.91							0.433	Moderate
	species	0.855	66	16	33	49	55						67	0.983	MEP
average of parameters for > 6m t.o.v. NAP														0.747	GEP

4.5.2.5.4 *Integration of the three levels*

For the overall assessment of the water body the ecological score and status obtained for each of the levels (level 1 and 3) are averaged into a metric representative for the whole water body. The averaging is done with a weighing factor for each level, 1 for the ecosystem level and 2 for the community level. The average (Table 51) of the three levels is 0.67, which corresponds with to GEP status for Grevelingen.

Table 56. Ecological quality Ratio and status for Lake Veere obtained by averaging the parameters at each level

	EQR score	EQR status	Remark
Level 1: ecosystem	0.7	GEP	Expert judgement
Level 3: community	0.65	GEP	
Overal EQR	0.67	GEP	

4.5.2.6 Discussion

Reference settings

The reference period (1990-1994) of Grevelingen is not free from pressure influences, but at this moment it is the most representative period to determine changes in the state of Grevelingen. No historical data are available, and no related system exists in neighboring countries to improve the reference conditions. The determined reference conditions can only be changed by expert judgement and by excluding pressure influences on the used data.

Assessment analysis

The assessment at level 1 for the Grevelingen has to be updated with recent primary production estimates. At this moment, this evaluation shows that it is useable for saline lakes, but standard primary production measurements have to be done in the future.

The reports on the macrobenthos of lake Grevelingen did not mention drastic changes in the benthos over the last 15 years (Wattel, 1996; Hoeksema, 2002; Schaub et al., 2002) and it seems that the global patterns in biomass and density still hold. However, clear temporal trends in species composition are observed, whereby the suspension feeders declined, *Hydrobia ulvae* almost disappeared, some polychaetes (e.g. *Nereis succinea*) are more widespread and the commercial bivalves (mussel, oyster, cockle) are disappearing (Schaub et al., 2002). These observations are a signal that there is something happening in Grevelingen.

Our assessment evaluation equally signals that there are changes in the system, in spite of the overall GEP evaluation of the system and for two of the three habitats. In our evaluations, it seems that the biomass and density in the assessment period was decreased compared to the

reference state in all habitats. The changes in Grevelingen are most strongly reflected in the species composition parameter, which is moderate for all habitats and indicates changes over time in macrobenthic community structure. In the number of species, nothing was changed, which is confirmed by the reports concerning Grevelingen.
It can be concluded that the evaluation with the BEQI method reflects the observed changes, but that a more detailed analysis is needed to unravel the changes on species level.

Advices

The following advices for Grevelingen can be formulated:

- There are slight changes in Grevelingen, which need attention and further evaluation.
- It is advisable to increase the total assessment sampling surface to get a higher assessment precision (see chapter on monitoring for more information).
- Investigations on the relation between primary production and the benthic biomass have to be done to evaluate possible changes in ecosystem functioning in Grevelingen. No good estimates of system primary production and benthic biomass (lack of biomass estimates for the mussel and oysters beds) are available for the Grevelingen.
- Inclusion of an evaluation of the eco-elements (mussels and oysters) is also advisable and an appropriate evaluation of these eco-elements has to be included in the monitoring.

It has to be mentioned that the management authorities of the Grevelingen have recognized the main pressure problems (oxygen depletion due to stratification) and that action was undertaken (increase water exchange) to solve this problems. Further monitoring is needed to evaluate the impact of these actions.
If in the future the management of the lake will further change, as situated in the 'Ontwikkelingsschets Zicht op de Grevelingen', further action needs to be taken and an operational monitoring will have to be established to follow up the changes.

4.5.2.7 Conclusion

Grevelingen:

- Pressures: oxygen depletion
- Evaluation:
    - Level 1: GEP and level 3: GEP
    - No drastic changes in the Grevelingen are reported, therefore the evaluation at level 3 shows that the present situation is more or less similar than the reference period. However, some parameters show a moderate status and require further investigations.
- Advices:
    - The current monitoring program should be adapted to include measurements on primary production (level 1), eco-elements (level 2) and macrobenthos (level 3). An operational monitoring program should be installed when a new management strategy for the lake will be adopted.

5. Consequence for monitoring

5.1 Monitoring requirements for the WFD

Article 8 of the Directive establishes the requirements for the monitoring of surface water status, groundwater status and protected areas. Monitoring programmes are required to establish a coherent and comprehensive overview of water status within each river basin district. The programmes must be in accordance with the requirements of Annex V.

Annex V indicates that monitoring information from **surface waters** is required for:

- The classification of status.
- Supplementing and validating the Annex II risk assessment procedure;
- The efficient and effective design of future monitoring programmes;
- The assessment of long-term changes in natural conditions;
- The assessment of long-term changes resulting from widespread anthropogenic activity;
- Estimating pollutants loads transferred across international boundaries or discharging into seas;
- Assessing changes in status of those bodies identified as being at risk in response to the application of measures for improvement or prevention of deterioration;
- Ascertaining causes of water bodies failing to achieve environmental objectives where the reason for failure has not been identified;
- Ascertaining the magnitude and impacts of accidental pollution;
- Use in the intercalibration exercise;
- Assessing compliance with the standards and objectives of Protected Areas; and,
- Quantifying reference conditions (where they exist) for surface water bodies.

Three types of monitoring for surface waters are described in the Guidance Document no. 7 'Monitoring under the Water Framework Directive' as part of the Common Implementation Strategy for the Water Framework Directive:

- For surface water bodies, the Directive requires that sufficient surface water bodies are monitored in **surveillance monitoring** programs to provide an assessment of the overall surface water status within each catchment and sub-catchment within the river basin district.
- **Operational monitoring** is to establish the status of those water bodies identified as being at risk of failing their environmental objectives, and to assess any changes in their status from the programs of measures.

- **Investigative monitoring** will be designed to a specific case or problem being investigated. In some cases it will be more intensive in terms of monitoring frequencies and focused on particular water bodies or parts of water bodies, and on relevant quality elements.

The Directive specifies quality elements for the classification of ecological status that include hydromorphological elements supporting the biological elements and chemical and physicochemical elements supporting the biological elements. For surveillance monitoring, parameters indicative of all the biological, hydromorphological and all general and specific physico-chemical quality elements are required to be monitored. For operational monitoring, the parameters used should be those indicative of the biological and hydromorphological quality elements most sensitive to the pressures to which the body is subject.

The Directive specifies quality elements for the classification of ecological status that include hydromorphological elements supporting the biological elements and chemical and physicochemical elements supporting the biological elements. Supporting means that the values of the physico-chemical and hydromorphological quality elements are such as to support a biological community of a certain ecological status, as this recognises the fact that biological communities are products of their physical and chemical environment.

An important aspect in the design of monitoring programmes is quantifying the temporal and spatial variability of quality elements and the parameters indicative of the quality elements in the surface water bodies being considered. Those that are very variable may require more sampling than those that are more stable or predictable. Alternatively, variability might be reduced or managed by an appropriate targeted or stratified sampling programme which collects data in a limited but well-defined sampling window.

The implementation of the BEQI approach to assess the ecological status of coastal and transitional waters and saline lakes for benthic macrofauna requires a monitoring programme that is targeted to this approach. In this chapter the general monitoring requirements for using the BEQI will be first outlined, and also highlighting the beneficial use of innovative monitoring techniques. Secondly, an overview will be given of the historical and present benthos monitoring that is done in the Netherlands. From a BEQI perspective, advices will be formulated towards a common strategy for monitoring benthic macrofauna in Dutch coastal and transitional waters. It should be stressed that these advices are targeted towards the implementation of WFD and BEQI, and therefore not mean that they are applicable to all other monitoring purpopes.

5.2 Monitoring requirements for BEQI

The Benthic Ecosystem Quality Index (BEQI) uses a multilevel approach. Each level requires a specific monitoring approach.

5.2.1 Metric at ecosystem level: system primary production : benthic biomass

Primary production is an important ecosystem attribute, providing the energy for ecosystem processes. Primary production fuels the food web. The BEQI method uses system primary production for the assessment at the first, ecosystem, level. Although primary production is of overriding importance in ecosystems measurements are at present not included in the MWTL (BIOMON) program.
System primary production in the shallow, soft sediment systems considered is the sum of production of phytoplankton in the water column and microphytobenthos in the top layer of the sediment. The contribution of macrophytes and seagrasses are at present very limited in the Dutch coastal systems. Especially in shallow coastal waters with large intertidal areas the production of the microphytobenthos can be a very significant part of the total system production (Herman et al;, 1999). For example the microphytobenthos-to-phytoplankton ratios of the production m^{-2} $year^{-1}$ in the western Wadden Sea are 0.59, 0.94 and 0.67 for the three years where data are available (1974, 1975 &1981) (Philippart & Cadée 2000).
In the Marsdiep the NIOZ measures phytoplankton parameters from 1974 onwards with a few gaps. A continuous series of yearly estimates of primary production in the Marsdiep tidal inlet started in 1990. Primary production is measured with the ^{14}C technique (Cadée and Hegeman 1974). This series is explained and results are discussed by Cadée and Hegeman (2002) and Philippart et al. (2007). Although important long-term changes could be observed from this measurement series, this single series of primary production measurements of only the pelagic production is not sufficient to cover the needs for the BEQI assessment in particular and to judge the ecosystem processes in all Dutch coastal waters. For benthic production by microphytobenthos it was shown that spatial variability is considerable meaning that estimates need to be made based on an appropriate spatial covering of measurements (Morris et al. 2006). It is remarkable how limited the information is on such a key process driving biological systems. For the BEQI assessment primary production estimates are required for every water body that needs to be rated. With new techniques available, dicussed below, the monitoring of primary production (both pelagic and benthic) in different water bodies and more sites within water bodies is feasible. Both a good temporal and spatial resolution can be achieved with automated measurements and remote sensing techniques. During the process of the development of a

primary production monitoring network knowledge will accumulate for further tuning of methodology and sampling strategy.

Measurement of primary production.

Primary production is the assimilation of inorganic carbon creating new organic matter. Standard technique for the estimation of the primary production rate is measuring the incorporation of radioactive ^{14}C, by phytoplankton or microphytobenthos in an incubation bottle. The ^{14}C is added to the culture through $NaH^{14}CO_3$. Another way is by measuring oxygen production and consumption, because fixing CO_2 means getting rid of oxygen. For phytoplankton this is done in incubation bottles. The primary production of the microphytobenthos can be deduced in situ from oxygen profiles in the sediment. Estimates of oxygen fluxes can be made based on the shape of the oxygen profiles. These profiles are measured with microelectrodes but nowadays also with micro optodes. These are laborious techniques limiting the spatial and temporal resolution of measurements.

Another suit of techniques uses the fluorescent properties of Chlorophyll *a*. Chlorophyll *a* can be exited with specific wave length of light. The resulting fluorescence is then a measure of chlorophyll concentrations. This kind of technique is probably most suited for long term automated recording of phytoplankton biomass.

Fluorescence is also used to estimate primary production. Fluorescence induced by a range of light pulses can be used to estimate the Photosystem II electron transport rates. This photosynthetic rate estimate can then be used to estimate primary production. Two techniques that measure Photosystem II electron transport rates are Pulse Amplitude Modulated fluorometry (PAM) and fast repetition rate fluorometry (FRRF).

Fluorescence based techniques combined with remote sensing are at the moment most promising to give insight in the temporal and spatial variability of primary production. Large scale patterns in biomass of primary producers in the water and on the sediment can be estimated with remote sensing techniques. The conversion of biomass to primary production then has to be made based on relationships between the two. Actual primary production measurements with for instance FRRF or PAM at selected sites have to yield information on the relationships. There is a need for the quantification of primary production in coastal systems for ecosystem assessment but also to advance our understanding of ecosystem functioning. With the new available techniques there is a large potential for systematic estimation of primary production and with this the possibility to fill this gap in the MWTL program.

5.2.2 Metric at habitat level

The qualification at the habitat level is mainly based on abiotic variables, like sediment composition and depth. But also biogenic structures (eco-elements) like mussel and oyster beds are included. A good abiotic description of the water body is an important prerequisite for the application of the BEQI method. Based on the abiotic variables ecotopes are defined, these are the strata which are individually addressed at the third level of the BEQI method. It is important that the classification of the habitats also captures part of the variation in the macrobenthic community. When this is not effectively the case this will have negative consequences for the power of the assessment at the third level. Further it is important to take into account the differences in pressure(s) as for reasons of cost-effective monitoring not all different habitats under equal pressure needs to be monitored. The most sensitive habitat/community types are in this respect sufficient. In this aspect, it is important to realize that for unknown or badly known pressure effects, like climate change, wind mill parks, aquaculture on sea and acidification of the ocean, the monitoring has to be sufficient to detect the effects of these pressures. A detailed investigations is required if monitoring can be reduced to most sensitive habitat/community types, because the danger exist that some unknown pressures will not be picked up.

A good description of the habitat with biologically meaningful parameters is essential. Sediment composition, depth and salinity are among the most important ones. In the present ZES ecotypes also hydrodynamic variables are incorporated, mostly based on model calculations.

Using an indicator at the habitat level in BEQI serves two different purposes. First, it is assumed that different habitat types have different functions in the ecosystem. This is very clear for some of the eco-elements (e.g. mussel beds, seagrass meadows), which may change the system functioning considerably. However, also other habitat types (e.g. coarse sand sediments that may passively filter the water, or mud deposition sites) serve clear and clearly-distinguished functions in the ecosystem. Thus, mapping the habitat types gives an impression of the ‘completeness’ of the benthic functions in the ecosystem. Second, it is assumed that within-habitat variability in ecological characteristics of the macrobenthos is much smaller than between-habitat variability. Different habitats should therefore be distinguishable on the basis of their biological community. If this is not the case, the level 3 parameters (within-habitat community composition) will loose instead of gain power by distinguishing between the habitat types. Combining both aspects, we want the habitats distinguished in this approach to be functionally different subsystems with a clearly distinguishable macrobenthic community.

In practice the number of habitats that can be distinguished is limited by data availability. For some clear habitat types, insufficient sampling has not allowed to characterize the macrobenthic community in several water bodies. A second limitation is that available mapping techniques limit the number of environmental variables on the basis of which one can distinguish the habitats. The first problem may be solved by reconsidering the monitoring sampling effort. The second problem may be solved by adopting new methods for mapping the estuarine habitats.

Monitoring of habitat change requires large scale measurements, which could partly be done with remote sensing techniques. For the subtidal, ship-based measurements are needed using multibeam, side scan sonar, or ADCP techniques. For the intertidal airborne information such as satellite or radar images can be used. Besides these techniques field surveys and ground-truthing are essential.

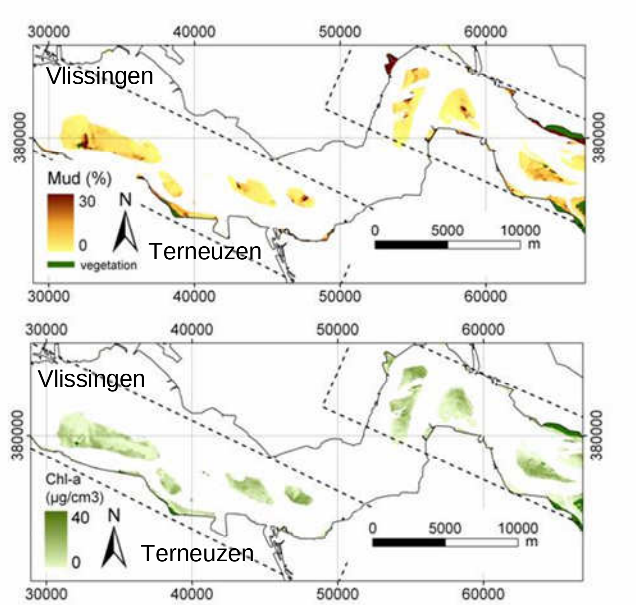
Two examples of using innovative techniques for habitat mapping are presented below:

Mapping of habitats:

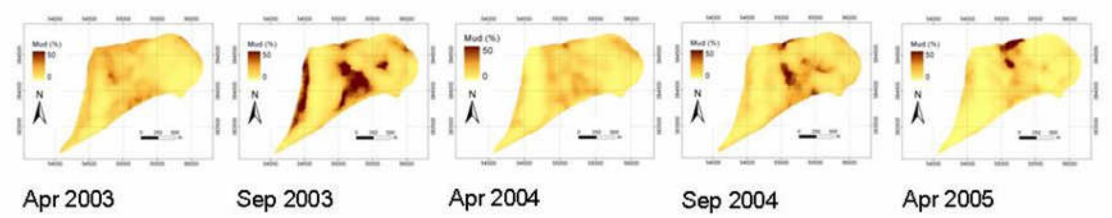
One of the large habitat mapping projects is MESH (Mapping European Seabed habitats) (http://www.searchmesh.net), which aims to generate habitat maps for NW-Europe, develop standards and protocols for habitat mapping and to test this all. There exist a lot of techniques for mapping (for the shoreline and shallow waters (e.g. airborne techniques, satellite imagery, shoreline surveys, ground-truth sampling); for deeper areas (multibeam, sidescan, 3D seismic), ROV's, drop cameras, sediment profile imagery)). This current knowledge makes it possible to map an entire water body, whereas a few years ago only point data and modeling could be used to map the habitats within a water body. With these techniques, not only differences in physical habitats can be determined, but also biogenic structures (e.g. mussel and oyster beds, *Lanice conchilega* beds). When these full coverage maps of the physical habitats are available and if the relationships between the physical and the biological habitat are known, it is also possible to create a full coverage map of the biological habitat (currently in development for the Belgian Continental Shelf; Degraer et al., in prep). These developments have to be started or brought on a wider scale, because in the Netherlands there are no entire habitat maps, based on these new techniques, available for a water body (only test areas).

Remote sensing of intertidal benthic ecosystems (mostly Westerschelde estuary)

Methods are being developed to use optical remote sensing, including satellite remote sensing (Landsat, SPOT, ASTER), Synthetic Aperture Radar (ERS SAR) and airborne hyperspectral remote sensing (CASI, AHS, Hymap) for mapping and monitoring the sediment grain-size and microphytobenthos biomass of the intertidal flats in the Westerschelde. In addition methods are being developed to use synergy of these sources to map and monitor changes in the physical habitats of intertidal flats that can be related to changes in the benthic macrofauna (see e.g., Van der Wal et al., 2004, 2005; Van der Wal & Herman, 2006, Van der Wal & Herman, 2007). The work was applied to the intertidal flats of the Westerschelde estuary.



(a) Mud percentage based on synergy of ERS-2 SAR and CASI (Van der Wal & Herman, 2006), and (b) chlorophyll-a based on CASI data (Van der Wal et al., 2007) of the intertidal flats of the Westerschelde, May 2005. Dashed areas indicate extent of CASI data.



Changes in mud percentage on the Molenplaat, Westerschelde, based on a series of ERS SAR images (Van der Wal & Herman, 2006).

5.2.3 Metric at community (within-habitat) level

At the within-habitat level the ecological quality assessment is based on four measures of the macro benthic community. These are total biomass, total density, number of species and community composition. The four variables are easily estimated from samples of the macrobenthic benthic community. Normally sediment cores are collected at sampling stations with a sampling core at low tide on the intertidal flats or with a device like the Reineck Box corer or Van Veen grab operated from a ship for subtidal stations. The sediment is washed through a 1 mm mesh. Specimens are sorted form the residue, identified to the species level, counted and weighed. Biomass is most accurately measured by the difference between dry weight and ash weight, the ash free dry weight AFDW.

BEQI assesses the state of the macrobenthic community by comparing estimates of biomass, density, number of species and community composition with defined reference conditions. Both the reference conditions and the assessment conditions of the four variables are estimated from benthic field samples. For an unbiased assessment representative unbiased samples of the macrobenthic community are needed for both the assessment situation and the reference situation. Both temporal and spatial variation should be adequately covered.

The design of an experiment or sampling campaign depends largely on the question of interest, or in other words the formulated hypothesis. The BEQI method stratifies systems in habitats, and this spatially stratified approach should also be reflected in the monitoring strategy. Habitats are defined based on (combinations of) abiotic parameters. Part of the spatial variance of the macrobenthic community can be attributed to differences between habitats. Within habitats, the scale of assessment of the benthic community, the variance is reduced compared to that of the total ecosystem. This increases the power to detect changes between assessment and reference conditions. The effectiveness of the use of strata depends on how well the total community variance is explained by the strata. When, theoretically, there is no structuring of the community depending on differences between strata there is not much use in defining strata and sampling accordingly. It will even lead to a reduction in detection power of BEQI because of sample sizes within the strata become smaller compared to the total sample size pooled. Of equal importance is that, in practice, it should be possible to assign samples to the correct stratum. If this is difficult and mistakes are made the variance of the estimate of a stratum increases. For an efficient division of effort over the strata in a stratified sampling program stations are *a priori* selected within each stratum. Thus the strata are defined based on prior knowledge on sediment composition, depth and hydrodynamics. The quality of the prior information will have its limitations in dynamic coastal systems. To check if the station is assigned to the right stratum and to update the environmental information on which the strata are based sediment composition and depth of every station should be measured together with the sampling of the macrobenthos.

A representative sample of the community within a stratum can be collected at randomly assigned stations. Use of randomly selected stations ensures that the sample is representative for the area within which the stations are selected and no systematic bias due to some unknown factors is introduced. Because BEQI assesses differences in time, strata are resampled repeatedly. Randomly assigned stations can be revisited (random-fixed design) or new stations can be selected every time (pure random design). Revisiting stations has the advantage that between-station variation is not obscuring changes in time, increasing the power to detect changes (Van der Meer 1997). Revisiting stations has at least one disadvantage that the representativeness of the stations for the population of interest changes with time. In other words that change in the estimate of population parameters like biomass and density may be caused by changes of local station conditions and are not due to a change in the larger community of interest in general (Armonies 2006). In dynamic systems like estuaries a further disadvantage is that the spatial configuration of the whole system continuously changes, so that a geographically fixed station may actually move along a gradient (of height, sediment composition or another factor). Finally, with limited numbers of samples the spatial coverage of one set of stations is limited. When these stations are subsequently fixed, there is a risk of never representing slightly different conditions still belonging to the same habitat in the data set collected after several years. For all these reasons, we believe that the pure stratified random design is to be preferred as the basic set-up for the monitoring design in traditional waters.

There are, however, two very important points to be added to this advice. One is that some very valuable time series exist in The Netherlands (e.g. the Balgzand series) and it requires the utmost care before anything in the procedure of such a series could be changed. The second is that monitoring should be as 'multi-purpose' as possible. For that reason it could be valuable to keep selected time series based on a random-fixed design to improve resolution in time. However, such series should always be checked thoroughly for any changes in the surrounding landscape, in order to be sure that sampling points remain constant relative to the physical gradients in which they are located.

Temporal variation is another source causing variance of the estimates made. Seasonal variation is part of the temporal variation which is not of interest for the assessment. The seasonal variation is mainly predictable and can largely be excluded from the estimate by collecting samples in the same season every year. Of course there is possibility for bias in such an approach, for instance when phenology of processes changes, e.g. growth seasons starting earlier because of climate change. In WFD surveillance monitoring assessments are required every six years. To prevent strong effects from between year variation an assessment sample should cover the entire assessment period. A continuous sequence of data collected every year is also better suited to detect temporal trends. From a practical point of view it is also good to spread the effort over longer time periods. The capacity for sample collection and processing doesn't need to be as large as in the case of a single sampling campaign and besides that the expertise and facilities to conduct the monitoring are maintained.

5.2.4 Compliance with WFD guidelines

The Netherlands has formulated their own guidelines for monitoring in accordance with the WFD requirements (Van Splunder et al., 2006). This was done to harmonize methods between water bodies to get unequivocal assessments of water bodies. An excerpt of Appendix 4B covering biological monitoring is made as far as it is relevant to monitoring of benthic macrofauna in transitional and coastal waters.

- Minimal requirements for surveillance and operational monitoring are defined. Independent of these minimum requirements, sampling frequency and design should be substantiated. It is likely that the minimum requirements are not sufficient to meet monitoring goals. Monitoring with to low effort leads to noisy data and high risk of misclassification.

  → This is also required for the BEQI method.

- Parameters measured in macrofauna sampling are species composition and abundance. Individuals are identified to the species level and counted. In transitional and coastal waters biomass is also measured. Sub-littoral stations are sampled with a box corer, intertidal stations with a regular sampling corer at low tide.

  → This is also required for the BEQI method.

- Within a water body a representative sample includes the variability in important ecological factors like salinity height in the tidal zone. Every habitat with a significant proportion of surface area is included.

  → This is also required for the BEQI method and certainly sediment and depth information has to be included by the benthos monitoring.

- Sampling takes place in spring (April/May/June).

  → It is more opportune to monitor in autumn for the WFD, because mostly (except for coast and Wadden Sea) the reference data is taken in autumn. This period is also characterised by the highest species richness, density and biomass.

- Because of the large spatial variability of macrofauna within water bodies stratified sampling over relevant strata is suggested.

  → This is also required for the BEQI method.

- Sampling stations should be representative for the water body or stratum. Microhabitats within strata are represented proportional to occurrence.

  → This is also required for the BEQI method, but micro-habitats would maybe better include as separate strata.

- Per water body at least six and per stratum at least three sampling stations are required.

  → This minimum boundary of number of samples is in general not in accordance with the requirements for the BEQI method and is effectively in most water bodies too low. The number of samples depends on the type of habitat (homogenous or heterogeneous) and type of water body. The sampling effort should be adjusted according to the required detectable effect size and equal to the minimal required surface after one year and equal to the maximal required surface after 3 or 6 years, to get an acceptable assessment.

- Minimal measuring frequency for surveillance monitoring is once in six years, for operative monitoring once every three years.

  → A temporal frequency of monitoring of once in six or three years is too low and has several disadvantages. In Box 8, three reasons are given why it is preferred to maintain a yearly program with reduced effort instead of an intensive program with long sampling intervals. Ideally monitoring programs are designed to gain maximum information density with minimal effort. It should be clear that sampling macrobenthic fauna with lower frequencies than once a year does not comply with this principle. A good temporal and spatial coverage at the appropriate scales is essential. Long term monitoring programs are specifically aimed at detecting changes or trends in time. If a reduction of effort is needed it should be considered to reduce the effort per time point and sure not by increasing the sampling interval. The analysis of habitat/community types being the most sensitive will give further arguments for reducing WFD monitoring effort.

Box 8 – Importance of developing a monitoring program, that gives a good temporal and spatial coverage.

The three reasons why it is preferred to maintain a yearly program with reduced effort instead of an intensive program with long sampling intervals.

First, in macrobenthic communities there is a strong year-to-year variability. An estimate based on just one year can deviate far from the system average. An estimate based on the average over several years will be more accurate because year effects will be averaged out. This is illustrated for biomass and density assessment with the Balgzand monitoring program in the mid littoral muddy sand habitat (figure 1). In this example, all estimates are based on eight transects, either eight per year, eight per three or six years, so the effort per estimate is the same, but the spatial coverage is best in the single year estimates and worst in the six year estimates. Single year estimates showed the largest variations. The coefficient of variation of the yearly estimates of biomass (0.33) and density (1.01) is higher than that of the three years (biomass: 0.28 and density: 0.49) and six year (biomass: 0.28 and density: 0.46) estimates. This means that an assessment done on one observation every three or six years is more liable to fluctuate and result in low ratings. However part of this reduction in the variance of the estimates is caused by reducing the effects of spatial variability in the estimates averaged over years. An optimal monitoring program should cover both sources of variability in sufficient detail to minimize the chance of misclassification.

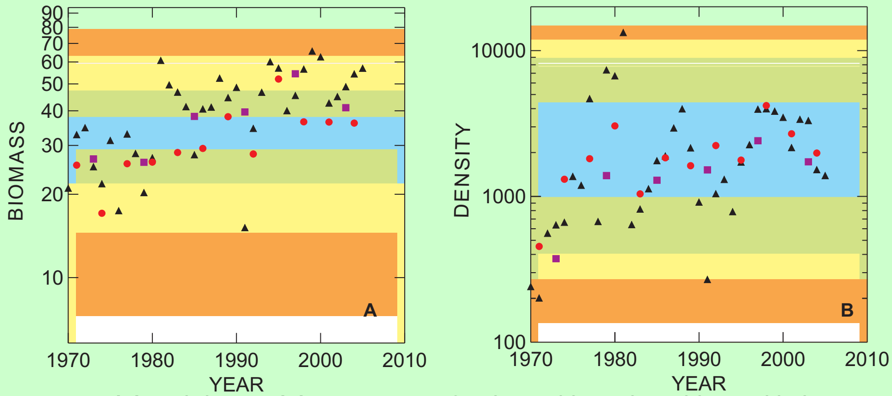


Figure 1: Biomass (A) and density (B) against time for the mid littoral muddy sand habitat at Balgzand. Black triangles (year period) are yearly averages of eight transects. Red dots (three year period) are an average of three fixed transects over three years, following a scheme of all three transects in the first year only two in the second and again all three in the third year. Purple squares (six year period) are averages of two transects (subset of the three transects used for the three year average estimate) re-sampled over 6 years following transect sampling scheme 1,2,1,1,2,1. Colors denote BEQI status, white = bad, orange = poor, yellow = moderate, green = good and blue = high status.

A **second** point of concern is that with a low frequency program power to detect temporal trends will be very low. This is illustrated with the same Balgzand data as in the previous example. The most obvious time trend at Balgzand is an approximate linear increase in biomass (Figure 2A). This trend could be noticed in the course of the eighties (Figure 2B), correlations first becoming significant in 1983. With a low frequency program covering all transects only once every third year the trend would first have been observed in 1995 (Figure 2C). With a six year interval the trend even would have not been noticed up to now (Figure 2C). Using the same sampling effort as in programs with three or six year intervals but now divided over all years would improve the power to detect time trends. When only two or three transects per year would be sampled, which is a similar effort as sampling all transects once every three years, the trend first becomes significant in 1989 and consistently so in 1994 (Figure 2D). Spreading a six year interval program effort over all years would reveal the trend in 1994 (Figure 2D). It is obvious that the power to detect trends with a sampling interval of six years is extremely low. Using the same sampling effort in a yearly program will improve matters strongly.

A **third** more practical considerations is that, sampling routine and expertise is not maintained. This for instance means that extra error is introduced due to subtle differences in methodology applied by different people and institutions.

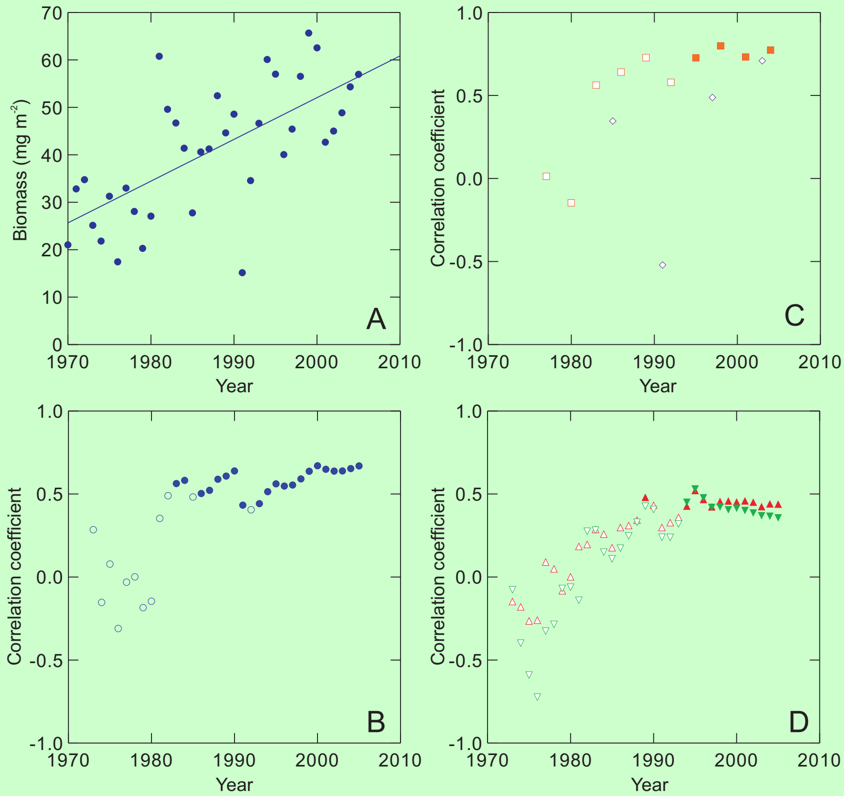


Figure 2. **A**. Development of total macrobenthic biomass at the mid littoral muddy sand ecotope from 1970 onwards at Balgzand. **B**. Correlation coefficient of the biomass year relationship over the past monitoring period. **C**. Correlation coefficient of biomass against year for averages per eight transects once every three years (orange squares) and once every six years (purple diamonds). **D**. Correlation coefficients of averages based on two or three transects per year (red upward pointing triangles) and based on one or two transects per year green downward pointing triangles. In B, C and D significant correlations at the 0,05 level are indicated with filled symbols.

5.3 Benthos monitoring in the Netherlands

5.3.1 Historical monitoring

Water quality issues in the Netherlands have initiated monitoring programs more than 40 years ago. First only chemical and physical parameters are measured in the main rivers Rhine and Meuse. Later estuaries and coastal waters are included in the monitoring programs.

Only in 1990 a biological monitoring program (BIOMON) was set up for Dutch coastal waters (Colijn & Akkermans 1990). This program integrates several trophic levels with components phytoplankton, sea grass, macrobenthic fauna, birds and marine mammals. The BIOMON program is part of the MWTL ("Monitoring van de Waterstaatkundige Toestand des Lands") program of Rijkswaterstaat, the Dutch Directorate for Public Works and Water Management, part of Dutch Ministry of Public Works and Transport. The MWTL program is basically a surveillance monitoring program.

Before 1990 already measurements of macrobenthic community are done on a regular basis. These older activities are partly continued in the MWTL BIOMON program. Besides the MWTL program some additional monitoring takes place, for instance some additional transects on the Balgzand in the Western Wadden by the Royal NIOZ, or the MOVE program an operational monitoring in the Western Scheldt for the study of effects of shipping channel management.

Wadden Sea

At the moment three monitoring series are operational in the intertidal and one in the subtidal Dutch Wadden Sea. All are sampled within the MWTL framework.

One of the longest operational continuous monitoring series of the intertidal macrobenthic community in the Netherlands is that at the Balgzand, which started in 1969. The Balgzand is an intertidal flat area near Den Helder in the Western Dutch Wadden Sea. The Balgzand benthic monitoring series was initiated by Jan Beukema of the Netherlands Institute for Sea Research. The aim of the project was to estimate the production of the macrobenthos (secondary production) at the Balgzand tidal flats. This effort was at that time part of the International Biological Program on productivity of marine communities.

To calculate the secondary production of the macrobenthic community a sequence of data through time are needed to estimate recruitment growth and survival of macrobenthic species. The first measurements are made in the summer of 1968. From 1969 onwards samples are taken in the end of winter, February and March. Beginning in 1980 also in late summer, August, September macrobenthos was sampled. Soon it was evident that the benthic community is characterized by a high temporal variation. For a good estimate of the production of the intertidal area it was necessary to have information over a longer time interval. In an analysis of production of the cockle *Cerastoderma edule* at the Balgzand with a thirty year data set it was concluded that at least a ten year period is needed to get an estimate with reasonable accuracy (Beukema & Dekker 2006).

Another long term monitoring series in the Wadden Sea of the intertidal macrobenthic community is at the Groninger Wad. This series was started in 1969 by Karel Essink (at that time at the University of Groningen) to study the effects of industrial organic sewage discharge from sugar and carton factories. The initial question is different from that of the Balgzand series and with this the sample site selection, sampling methodology and sample treatment.

In 1977 the Dutch Directorate for Public Works and Water Management simultaneously started trend monitoring at two intertidal flats in the Wadden Sea and one intertidal flat in the Ems-Dollard estuary. The new monitoring effort combined with the monitoring activities on the Balgzand and at the Groninger Wad was designed to get a Wadden Sea wide idea of natural and anthropogenic trends. In the Wadden Sea six transects are established, three on the Piet Scheveplaat south of Ameland and three transects on the Ballastplaat near Harlingen. The monitoring at the Ballastplaat ended in 1987, the Piet Scheveplaat series is continued until present. To get comparable results the strategy using transects was adopted from the Balgzand series.

The monitoring efforts in the subtidal areas of the Dutch Wadden Sea are much smaller than those in the intertidal areas. Since 1990 three sub-littoral transects in the Western Wadden Sea are sampled.

Eems-Dollard

In 1977 as part of a Wadden Sea wide initiative monitoring of three transects on the Heringsplaat in the Eems-Dollard area was started. This series is continued up to present.

North Sea

During 1988 and 1989 a large benthic survey was made in the Dutch sector of the North Sea, in a project called MILZON. Based on that program the BIOMON program for the Dutch Continental Shelf was developed and started in 1991. The goal of this program is to study the temporal variation in the North Sea benthic system. In the first few years of the program in total 25 stations along 5 transects perpendicular and one parallel to the coast are sampled. In 1995 the strategy was changed. The effort per station was reduced by 80% and 75 additional stations are selected. The additional stations are randomly chosen divided over four sub areas. Only the stations in the coastal area, 15 in total are relevant for WFD purposes, because the WFD only applies to coastal waters.

Delta area

In the past there has not been a strong long-term monitoring tradition like in the Wadden Sea. . In the 1960’s the whole Delta area has been monitored by researchers like Wim Wolff (see e.g. Wolff, 1973), but this has never been adopted into a long-term monitoring programme. Also for specific projects, such as the building of the storm surge barrier in the Oosterschelde, macrobenthic samples have been taken.

In the BIOMON framework monitoring takes places from 1990 onwards in the Westerschelde, Oosterschelde, Grevelingen and Lake Veere. Sampling strategies differ in the different waters and are further described below. In principle the objective is surveillance monitoring and the approach is the same for all waters in the Delta area.

5.3.2 Present surveillance monitoring

An overview of the MWTL BIOMON macrobenthic fauna monitoring program for 2007 for the different coastal waters, saline lakes and transitional waters is given in Figure 67. The details are explained in Table 57.

Figure 67. MWTL BIOMON macro benthic fauna monitoring program for 2007 with different sampling designs indicated. Details are given in the text

The present Dutch surveillance monitoring can be split up in 3 areas, based on differences in sampling strategy, namely (1) the Delta in the southwestern part of the Netherlands, (2) the Dutch coast and (3) the Waddenzee & Eems-Dollard. Although the macrobenthic fauna monitoring activities in the coastal waters are all unified in the BIOMON program and under the responsibility of one agency (but different offices) the large differences in the methodology are striking. A summary of the sampling strategies are given in Table 57 for the Delta area and in Table 58 for the Dutch coast and Waddenzee & Eems-Dollard.

In the Wadden Sea and Eems-Dollard sampling strategies applied in the BIOMON framework are inherited with the series themselves. The monitoring series are initiated with different motivations, either because of effect studies or to follow natural variation in space and time. Designs are made long before GPS techniques are available for accurate and easy delivery of information on exact position. Collecting samples along transects and at permanent quadrants was a practical solution (Table 58).

The series that are started around 1990 in the Delta area have a larger spatial coverage and use smaller samples per station. Among these newer series there are still differences in methodology. These are mainly in the way stations are selected (random and then revisited, new random stations are generated every campaign) or sampling devices and surfaces (Table 57).

Differences in methodology make comparisons between areas difficult. The BEQI method can account for small variation in sample surface but not for the differences between the Wadden Sea and Eems-Dollard programs on the one side and the Delta and North Sea programs on the other side. A proposal to make the surveillance monitoring more uniform between all water bodies will be given in section 5.4.

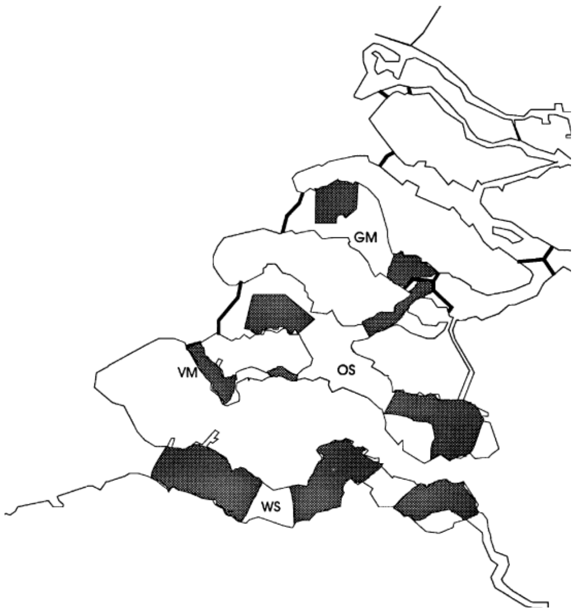


Figure 68 Monitoring sub-areas for the Delta water bodies. GM = Grevelingen. OS = Oosterschelde, VM = Lake Veere and WS = Westerschelde

Table 57. Detailed information about the sampling strategy for the present surveillance monitoring in the different water bodies of the Delta area

Area	**Westerschelde**	**Oosterschelde**	**Lake Veere**	**Grevelingen**
Sub-areas (Figure 68)	3 sub-areas	3 sub-areas	2 sub-areas	2 sub-areas
Strata	4 depth strata: +1 NAP -2 NAP -2 NAP -5 NAP -5 NAP -8 NAP < -8 NAP	4 depth strata: +1 NAP -2 NAP -2 NAP -5 NAP -5 NAP -8 NAP < -8 NAP	3 depth strata: > -2 NAP -2 NAP -6 NAP < -6 NAP	3 depth strata: > -2 NAP -2 NAP -6 NAP < -6 NAP
Random/fixed	Random	Until 1994 random After 1994 fixed	Until 1994 random After 1994 fixed	Until 1994 random After 1994 fixed
# samples	10 samples per stratum	10 samples per stratum	10 samples per stratum	10 samples per stratum
Sampling device	Sub-tidal: Reineck box-corer Intertidal: core	Sub-tidal: Reineck box-corer Intertidal: core	> -2m NAP: flushing sampler Other strata: Reineck box-corer	> -2m NAP: flushing sampler Other strata: Reineck box-corer
Sampling surface	Reineck: 3 x 0.005 m^2 Core: 3 x 0.005 m^2	Reineck: 3 x 0.005 m^2 Core: 3 x 0.005 m^2	Flushing sampler: 0.02 m^2 Reineck: 1x 0.005 m^2	Flushing sampler: 0.02 m^2 Reineck: 3x 0.005 m^2
Frequency	2 x year (spring and autumn)	2 x year (spring and autumn)	2 x year (spring and autumn)	2 x year (spring and autumn)

Table 58. Detailed information about the sampling strategy for the present surveillance monitoring for the Dutch coast, Waddenzee and Eems-Dollard

Area	Wadden sea	Eems – Dollard	Dutch coast
Sub-areas	Balgzand: BZ Groningerwad: GW Piet Scheveplaat: PS Sub-tidal: st	Heringsplaat	Zeeuwse kust (ZK), Noordelijke Delakust (NDK), Hollandse kust (HK), Waddenkust (WK), Eems-Dollard kust (EK)
Random/fixed	BZ: 12 transects random in four areas; 3 quadrants fixed GW: 5 fixed quadrants PS: 3 fixed transects St: 3 fixed transects	Fixed transects	15 Fixed stations: ZK: 3 NDK and EK: 1 HK and WK: 5
# samples	BZ, transects: 50 over 1 km BZ, quadrants: ? GW, quadrants: 20 in 30*30m PS: 3 per 20 stations over 760 m St: 15 over 1.5km	3 per 20 stations over 855 m	1 sample per sampling station
Sampling device	BZ, GW, PS: cores St: Reineck box-corer	Core	Reineck box-corer
Sampling surface	BZ, winter: core of 0.019 m^2, summer: 0.009 m^2 GW: core of 0.0077 m^2 PS: core of 0.009 m^2 St: Reineck of 0.06 m^2	Core of 0.009 m^2	Reineck: 0.078 m^2
Frequency	BZ, GW, PS, St: 2 x year	2 X year	1 x year

Laboratory treatment of samples:

The laboratory treatment of the benthic samples for the Delta area:

The macro benthic species are sorted and identified to the species level. Density is counted per species for determination of numerical densities. Biomass in ash free dry mass is either estimated through mass length relationships in bivalves, conversion of wet weight to ash free dry mass (AFDW) in polychaetes or by the weight difference after drying and after incinerating.

The laboratory treatment of the benthic samples for the Dutch coast:

Biomass is determined through length weight relationships, converting wet weight measurements to AFDW or by actually drying and incinerating of the samples.

The laboratory treatment of the benthic samples for the Wadden sea and Eems-Dollard:

Identification is to the species level. Individuals are counted to calculate numerical densities. Biomass as ash free dry mass is measured per species and for bivalves per length class pooled per transect by the weight differences between dried and incinerated samples. In practice the information on the numerical density and biomass of the benthic fauna and environmental conditions is pooled per transect and quadrant.

5.4 Towards a common monitoring strategy for Dutch coastal and transitional waters and saline lakes

5.4.1 Power consequences for monitoring

The different approaches in the different water bodies have strong implications for the power to detect changes in the macrobenthic community. The description of BEQI reference conditions relies on MWTL monitoring data and additional data which have been collected in similar ways. The description of the reference can be used to estimate the power to detect any changes with a given effect size at a given sample size. The power is the chance to detect a significant difference with a given α normally 0.05 and given effect size and variance. For the reference conditions the variance of the estimate along an axis of increased sampling effort (surface area) is described by the median quartiles and percentiles generated through permutation. Assuming the variance is independent of the population mean the variance distribution of the reference can be used as a predictor of the variance distribution of the assessment. At a certain effect size the chance off detecting a significant difference between reference and assessment is proportional to the part of the variance distribution of the assessment that falls outside the 95 % interval of the variance distribution of the reference. This is further explained with an example in the methods section 2.3. The sampling effort required for a detection of a biomass reduction of 50% (effect size 2) with a power of 75% was calculated separately for every habitat per water body. The results are presented in Fig. 5.4. For the different water bodies and habitats there are large differences in sampling effort to reach the required power. This is in first place caused by the heterogeneity of the habitat. For instance the Noordelijke Deltakust and the Zeeuwse Kust may be more heterogeneous than the Hollandse Kust Waddenkust and Eems-Dollard kust. The Sub-Littoral Ldyn habitat in the Wadden Sea and the Marine Sub-Littoral Hdyn habitat in the Westerschelde must have an extreme heterogeneity because they do not meet the effect size 2 with a power 0.75 with relative large reference samples. Another important factor is the sampling design. The red bars in the graph are based on data from a few transects where large numbers of cores are pulled (up to 1 m^{-2}). The blue bars are estimates from sampling programs with small individual samples from many stations. This tendency for the inefficiency of very large samples compared to a larger number of smaller samples was previously noted for the MWTL monitoring program (Van der Meer 1997).

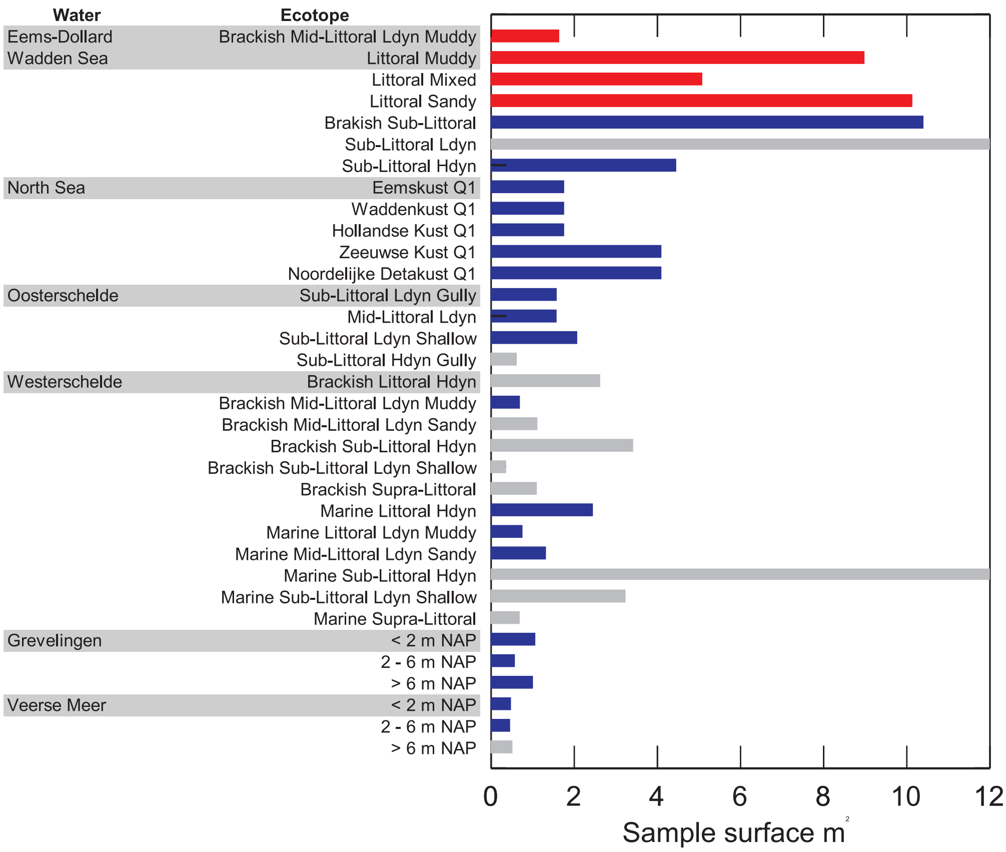


Figure 69. Sample size required to detect a 50% reduction in biomass with a power of 0.75. This is calculated for all habitats where reference conditions are defined. Ecotypes with gray bars have reference samples that are smaller than the sample size required for an assessment detecting an 2 times reduction with a power of 0.75. The gray bars show the total surface of the reference in stead. Red bars are samples collected along transects and in permanent quadrants (many samples pulled). Blue bars apply to sampling strategies with small samples at many stations.

5.4.2 Monitoring proposal

As mentioned in the guidance document no 7, three types of monitoring for surface waters are proposed for the WFD: surveillance, operational and investigative monitoring. In this part, a proposal will be given for the monitoring of the different Dutch water bodies, focusing on the third level of the BEQI method, this is the monitoring and sampling of the macrobenthos itself. As described above, the present benthos monitoring is not uniform and shows some shortcomings (De Jonge et al., 2006). Making changes for the better is however not easy, but unifying of the methods is necessary, especially for the surveillance monitoring of the WFD. This will be described per water body and has to be discussed to adapt present practice towards a new, more uniform WFD proof monitoring for the Netherlands.

Surveillance monitoring is needed for water bodies which are evaluated as GEP or MEP in chapter 3. Operational monitoring is needed for water bodies which are evaluated in chapter 3 as moderate, poor or bad, to investigate the effects of the pressure which is responsible for this evaluation. Investigative monitoring can be used to investigate some specific questions in case of unclear pressure benthos relations.

For the Netherlands, especially in the North, some long-term benthos monitoring occurs, which are very valuable to evaluate effects of e.g. climate changes. This long-term monitoring effort should not be abruptly interrupted, due to its high scientific and management value. However, a thorough additional analysis of some series (e.g. Eems-Dollard) could be done to evaluate if the monitoring strategy used by this long term monitoring is still adequate to obtain the long-term monitoring aims. Changes in protocol will make reference to the past more difficult, thus care has to be taken. Especially for the most impressive one, the Balgzand series which are consistently continued for almost forty years now without a single missing data point, it has to be continued without changes in the sampling protocol, due to its scientific value and for reasons of consistency.

Given the importance of physical structuring of the habitat, it is proposed for all water bodies that sampling for sediment characteristics (grain size distribution, chlorophyll-*a* content) as well as determination of exact height should be a routine part of all macrobenthos sampling efforts, which is already partly done.

The practical recommendations presented below deal with the third level of the BEQI method only. Further discussions between the research community and the local managers are needed to bring these recommendations into practice.

Coastal water bodies

Some coastal water bodies (Zeeuwse kust, Noordelijke Deltakust and Eems-Dollard kust) could not be evaluated due to too few assessment data, others (Wadden sea and Hollandse kust) show

a moderate to poor status. Also for only 1 habitat an evaluation was possible, whereas 4 biological habitats are discerned. Here there is a need to investigate whether the other habitat community types are equally sensitive to the dominant pressures in the coastal zone. This means that for the coastal water bodies an extra effort (increase the number of sampling stations) has to be undertaken to fill these gaps. Perhaps the ongoing PMR project monitoring can be used.
The following strategy for an improved surveillance monitoring of the coastal waters is proposed, where the required sampling effort is based on observed variance in similar habitats:

- stratified per habitat (for Zeeuwse Kust four habitats, the others 2), but first improve habitat description and investigated the sensitivity of the four habitats.
- random sampling within habitat with approximately 20 samples (entire box corers)
- at each sampling station collect additional physico-chemical parameters (e.g. sediment parameters)
- reflect on spatial coverage optimization irrespective of water bodies boundaries

Apart from the requirements for surveillance monitoring, the present state of the coastal waters also calls for an investigative monitoring program. As stated earlier, this should centre on the dynamics of the invasive species *Ensis directus*. At present, it is not known why this species dominates so strongly, what the interaction is with its resources (phytoplankton), with other native bivalve species (e.g. *Spisula*), and, probably most prominently, with human drivers such as fisheries effort or other disturbances.

Finally, for completeness a description of a project monitoring program currently going on in the coastal zone is outlined. This is the PMR monitoring. The Project Main port development Rotterdam (PMR) has been established to offer a solution to the limiting space in the port of Rotterdam, through the construction of a new port area, Maasvlakte 2. In order to compensate for the loss of nature (which is an EU-obligation) a marine reserve is planned in the Voordelta. To examine whether the impact of the extension of the Maasvlakte on the flora and fauna is sufficiently compensated by the provided nature compensations, a 10% increase in the quality of the benthic community through reduction of pressures, a monitoring and evaluation program (MEP-MV2) is started. The first part of MEP-MV2, has 2 main goals: (1) determine the T0 situation and (2) examine if results are sufficient for the subsequent impact studies. A large number of benthic samples (402 samples per campaign) have been taken in 2004 and 2005 to determine the T0 situation (van Steenbergen and Escaravage, 2006). The sampling strategy was configured to minimize the ratio between the effort (number of samples) and the statistical robustness (minimum detectable difference). Apart from standard descriptive statistics a parametric model (GLM) was developed to describe the occurrence of the benthos with a selection of environmental factors. The aim of the habitat approach was to decrease the variation in the data. Despite the large number of samples over this two year period, it was concluded that the ability to observe changes in the marine reserve and reference areas is still limited. Based on these observations the MEP-MV2 now investigates how the minimum detectable difference can be enlarged. This should only be possible when spatial and temporal variations are integrated. This requires an extra sampling year and another way of analyzing the data, including also more abiotic information. These data could be used for the WFD to address sensitivity differences in habitat/community types and to further improve the BEQI approach for the coastal waters.

Transitional waters: Westerschelde

The surveillance/operational monitoring of the Westerschelde is already to a large extent adapted to the requirements of the BEQI evaluation method, but the following improvements have to be made:

- Identify sensitivity of habitat/community types to pressures
- keep stratified random, but stratification no longer based upon depth strata, but on habitats
- subtidal: take larger cores (e.g. a full Reineck box core) due to the heterogeneity of the subtidal environment (otherwise an unpractical amount of cores is needed to get an acceptable assessment surface).
- define extra operational monitoring related to pressures:
    - o e.g. importance of shallow sub-tidal areas, low *vs* high dynamic areas, etc. (what are the needs?)
    - o e.g. dredging pressure

Table 59. Proposal of number of samples needed to get an optimal surveillance monitoring for the Westerschelde

	present monitoring (BIOMON + MOVE)		New monitoring proposal			
	# samples 3 year period	sample surface	# samples	sample surface	total surface 1 year	total surface 3 year
Brackish littoral[hdyn]	14	0.015	20	0.015	0.3	0.9
Brackish low littoral[ldyn]_Muddy	3	0.015	-			
Brackish low littoral[ldyn]_Sandy	4	0.015	-			
Brackish mid-littoral[ldyn]_Muddy	9	0.015	20	0.015	0.3	0.9
Brackish mid-littoral[ldyn]_Sandy	15	0.015	20	0.015	0.3	0.9
Brackish upper-littoral[ldyn]_Muddy	0	0.015	-			
Brackish upper-littoral[ldyn]_Sandy	0	0.015	-			
Brackish sub-littoral[hdyn]	44	0.015	15	0.068	1.02	3.06
Brackish sub-littoral[ldyn]	3	0.015	10	0.068	0.68	2.04
Brackish sub-littoral[ldyn] deep	2	0.015	-			
Brackish supralittoral	1	0.015	10	0.015	0.15	0.45
Marine littoral[hdyn]	76	0.015	30	0.015	0.45	1.35
Marine low-littoral[ldyn]_Muddy	2	0.015	-			
Marine low-littoral[ldyn]_Sandy	5	0.015	-			
Marine mid-littoral[ldyn]_Muddy	29	0.015	30	0.015	0.45	1.35
Marine mid-littoral[ldyn]_Sandy	65	0.015	30	0.015	0.45	1.35
Marine upper-littoral[ldyn]_Muddy	4	0.015	-			
Marine upper-littoral[ldyn]_Sandy	4	0.015	-			
Marine sub-littoral[hdyn]	210	0.015	25	0.068	1.7	5.1
Marine sub-littoral[ldyn]	24	0.015	15	0.068	1.02	3.06
Marine sub-littoral[ldyn] deep	5	0.015	-			
Marine supralittoral	8	0.015	10	0.015	0.15	0.45
TOTAL	527 (176/year)		235/year		6.97	20.91

An example of the number of samples, which are needed to fulfill the assessment requirements for the Westerschelde, is given in Table 59. These numbers are obtained by summarizing the knowledge about the differences in heterogeneity of the habitats, the surface area of the habitats, the minimal assessment surface which are needed and expert judgement. As such the yearly effort yields a sampling surface that fulfils the minimal requirements, and after a three year monitoring an optimal sampling surface is reached for each habitat. The present monitoring efforts are also included in this table and shows that there is an unequal effort between the habitats and it does not correspond with the difference in heterogeneity of the habitats or importance.

Transitional waters: Eems-Dollard

A surveillance monitoring of the Eems-Dollard has to be started. At this moment only a long-term monitoring program exists in only one habitat and no spatial coverage of the Eems-Dollard is reached. Therefore the following sampling strategy for the surveillance monitoring of the Eems-Dollard is advised:

- stratified per habitat, define all habitats in the Eems-Dollard and identify most sensitive
- random sampling within habitat (number of samples can be determined as in the example for the Westerschelde)
- define common program with Germany

It is important to keep the long term monitoring program, but a detailed investigation on the consequence of a possible reduction of the number of transects is needed, because it is observed that the 3 transects are very homogeneous.

Oosterschelde

Not for all habitats of the Oosterschelde an assessment was possible, this means that an extra effort is needed to cover these habitats and assess the sensitivity to dominant pressures. Otherwise, the following advices for the surveillance monitoring of the Oosterschelde can be given:

- stratified per habitat, no longer based upon depth, but on habitats (including eco-elements such as oyster and mussel beds) and identify most sensitive
- random sampling within habitat (number of samples can be determined as in the example for the Westerschelde)
- subtidal: take larger cores (e.g. a full Reineck box core) in stead of single cores due to the heterogeneity of the sub-tidal environment (otherwise an unpractical amount of cores is needed to get an acceptable assessment surface).
- map oyster and mussel banks

Apart from the surveillance monitoring, an operational or investigative monitoring with respect to the pressure 'zandhonger' (decline of the intertidal habitat) and the problem of the rapid spread of the Japanese oyster may be called for. This is subjected for further discussion related to the HMWB status and the analysis of measures possible.

Saline lakes: Lake Veere, Grevelingen

The sampling strategy for the surveillance monitoring could be the same for both saline lakes. The following changes in sampling strategy are proposed:

- stratified per habitat (following the three strata)
- the most sensitive habitats (e.g. Lake Veere in 2003, all habitats effected by oxygen depletion) are those under risk of oxygen depletion
- random within habitat (20 samples per habitat and per year is enough, which is presently done)
- subtidal: take larger cores (e.g. a full Reineck box core) in stead of single cores. This is already done for counting *Mya arenaria*, but it is maybe better to count all benthos in this full Reineck box core. Very abundant species can be counted by taking a subsample.
- map oyster and mussel banks
- Operational monitoring to follow-up the new management strategy (the oxygen depletion problem and the changes in salinity) in Lake Veere is needed and requires maybe a higher number of samples during a certain period.
- Operational monitoring will be needed for Grevelingen if a new management strategy for the system will be adopted.

Wadden sea

A surveillance monitoring of the Wadden sea has to be started (only a long-term monitoring program exist at a few areas in the Wadden sea), because these long-term monitoring sites are not entirely representative for the spatial coverage of the habitats. It is very important to keep the long term monitoring programs of the Wadden sea because there are unique. But the following sampling strategy has to be started to fulfill the needs for the WFD:

- stratified per habitat, including eco-elements, as is common MWTL practice
- random sampling within habitat
- Increase number of stations in general and especially in the subtidal.
- Map besides mussel banks also oyster banks

The observed problem of the changes in primary & secondary production and the benthic biomass can lead to an investigative research project. This is already in the planning of the TOP Carrying capacity NOW research project. Also the effect of invasive species (*Ensis, Crassostrea*) on the biodiversity and functioning of the Wadden sea ecosystem requires further research.

6. Intercalibration

This chapter of the intercalibration is structured, so that it is ready for implementation in the ECOSTAT report (WFD intercalibration technical report, part 3 [coastal and transition waters], section 2 [benthic invertebrates]).

Reference conditions, Class boundary settings are described for both NEA1/26 and NEA 4 followed by a comparison with other Member states methods.

More intercalibration results and examples, with an earlier version of the BEQI method are given in a separated intercalibration report (Van Hoey et al., 2006).

WFD intercalibration technical report

Part 3 – Coastal and Transitional Waters
Section 2 – Benthic invertebrates

2.3 NE Atlantic GIG

2.3.1 Intercalibration approach

2.3.2 National methods that are intercalibrated

In table

Member state	Method	Status
NL	BEQI (Benthic ecosystem quality index)[1]	Agreed

[1)] In the Intercalibration only the level 3 within habitat quality is applied

Full description of this method can be found in Annex 2.1.

It has to be mentioned that the BEQI-method differs from the other international methods, in the fact that the BEQI-method directly evaluated the entire water body, whereas the others only on sample level. Therefore, an intercalibration on sample level is not possible at this moment. To compare (intercalibrate) the Dutch method, the results of the BEQI are compared with an average of the EQR scores of the samples in the water body, obtained by the other methods. The average of the EQR of the samples within a water body is not the proposed evaluation method for a water body of each country, but could be currently used as confirmed at the NEA-GIG benthos expert meeting in Lisbon (February 2007).

2.3.3 Reference conditions and class boundary setting

Type – NEA 1/26
Reference conditions

NL: The Dutch assessment method requires a reference dataset for each habitat within the water body. This is different from the other MS multi-metrics which only require a (maximal) reference value for each parameter of a MS multimetric. The reference values are also related to sampling surface (determined by a randomization procedure) and defined for each WFD boundary. In other words, a reference value is defined, varying with sampling surface, for each boundary, which has to be reached to get the corresponding ecological status. In the table, the reference values for reaching a good status (example: total sampling surface of 1m^2) for the fine muddy sand coastal habitat of the Dutch coast (Hollandse kust, Waddenkust en Eems-Dollard kust) are summarized.

Habitat type	Sampling surface	# species	Similarity	Density (ind/m^2)	Biomass (gAFDW/ m^2)
Dutch coast (fine muddy sand Q1)	1 m^2	60	0.74	2584 and 7975	14.2 and 52.4

Reference conditions correspond to assessment class Good/Moderate. See also explanation in Figures 8 and 9. Reference values are based on the Q1 cluster of stations within six nm from the coast samples between 1983 and 1990.

Class boundary settings

NL: BEQI boundary setting procedure:

The Netherlands has statistically integrated the risk of misclassification at water body level related to sample size in the methodology and boundary setting. Based on permutation calculations, reference values are determined for each component metric and class boundary.

The reference values are calculated per habitat over increasing sampling surfaces. This allows for any given sampling surface, to estimate the reference value .The reference for a 1m^2 sampling surface is based on a set of 2000 artificial random samples.

Out of the randomisation procedure of each component metric (parameter) (density, biomass, species richness, species composition changes), a 5th percentile value is selected as the reference value that has to be reached to get a good status (the value of the good/moderate boundary).

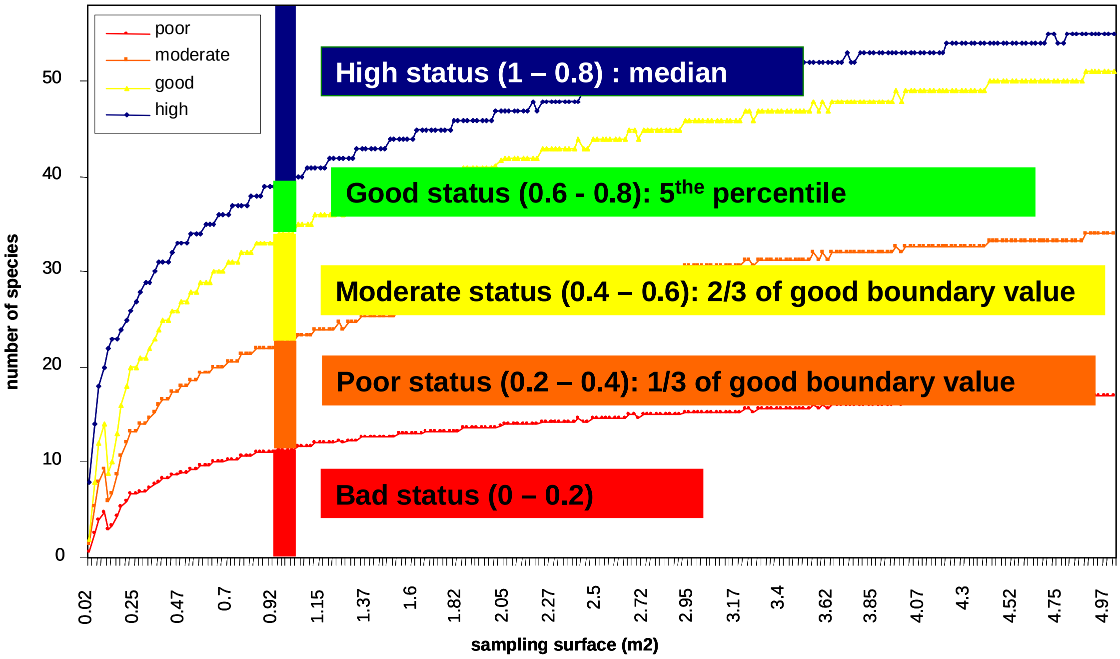


Figure 70. The boundary settings for number of species (this as example in the figure) and similarity in relation to sampling surface

For the parameters density and biomass a two side deviation from the reference values is scored. The other boundary values are adopted from this value (equal intervals), except the high/good reference value which is also directly extracted from the randomisation procedure.

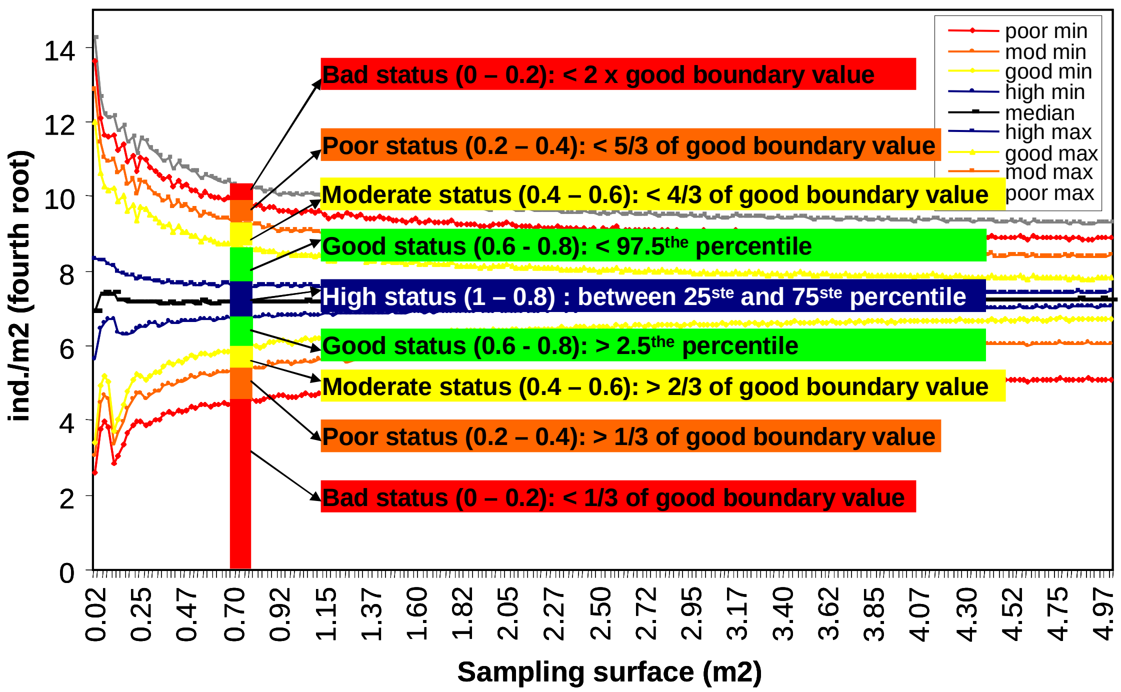


Figure 71. Boundary settings for density in relation to sampling surface. A similar protocol is applied for biomass

The boundaries selected from the randomisation procedure, with the corresponding WFD class boundaries are summarized in the table below.

Boundary	Number of species & Species composition changes	Density & Biomass
High/good: 0.8	median	25 and 75 percentile
Good/moderate: 0.6	5the percentile	2.5^{the} and 97.5^{the} percentile
Moderat/poor: 0.4	2/3 of Good/mod value	2/3 and 4/3 of Good/mod value
Poor/Bad: 0.2	1/3 of Good/mod value	1/3 and 5/3 of Good/mod value

The boundaries set by the BEQI method are statistical significance levels related to a fixed WFD Class boundary value. Thus, adjustment of the boundaries in the harmonisation of ecological status classification, will affect the statistical significance levels for the individual parameters. The BEQI method is considered more sensitive to changes than any other method tested.

Type – NEA 4

Reference conditions

NL: **Reference Conditions for G/M status for 4 habitats of major importance:**

NL	BEQI habitat	Sampling surface	# species	Similarity	Density (ind/m^2)	Biomass (gAFDW/ m^2)
NEA 4	High Littoral Mud	3 m^2	13	0.68	448 and 7643	4.1 and 20.6
	Middle Littoral Muddy Sand	3 m^2	17	0.7	269 and 12063	18.4 and 58.9
	Low Littoral Sand	3 m^2	13	0.6	106 and 7384	4.3 and 24.3
	Brackish Sub Littoral	3 m^2	26	0.82	1810 and 103353	18.7 and 88.8
	m-AMBI	**M**-AMBI reference values used in NL Intercalibration assessment of both Dutch and German sites				
		AMBI	Shannon Wiener	n species		
	High Littoral Mud	2,7	2,16	18		
	Middle Littoral Muddy Sand	0,947	2,34	23		
	Low Littoral Sand	0,393	2,22	17		
	Brackish Sub-Littoral	1,541	2,178	16		

Class boundary settings

NL: see the text for the Netherlands under NEA 1/26

The m-AMBI the boundary setting procedure is described in Borja et al. (2006) For the m-AMBI the original M-AMBI values G/M 0.55 and H/G 0.85 are applied.
Also the optimized class boundaries are taken into account G/M 0.53 and H/G 0.77

The Netherlands has applied both the Dutch BEQI and the German M-AMBI method on both Dutch and German NEA 3 data for 4 different habitats. The m-AMBI was communicated by Germany as the current Germany method to be applied in the first intercalibration round.

2.3.4 Results of the comparison

Type – NEA 1/26

Intercalibration of the Dutch methodology

Comparison of the Dutch (BEQI), Spanish (m-AMBI), UK (IQI), Danish (DKI) and Norwegian (NQI) methodology with Dutch data was done for two water bodies in the North Sea coastal zone Waddenkust and Hollandse Kust. DKI, IQI and M-AMBI scores are calculated with reference settings suggested by the member states (fixed reference) and with reference values determined from the Q1 reference dataset of the North Sea coastal zone (

Table 60). The reference setting is the same for Hollandse Kust and Waddenkust. NQI does not allow for use of local reference values.
The assessment period is 3 years (2002-2004) and total samples are 15 per waterbody. Except for the BEQI method the average of assessment station scores within a water body was taken as the overall water body status.

Table 60. Local reference values used for index calculation. Reference values based on the Q1 cluster of stations within six nm from the coast sampled between 1983 and 1990

	AMBI		Shannon-Wienner		# species		Simpson
	Bad	High	Bad	High	Bad	High	Max
Wadden- and Eemskust	6	0.107	0	2.66	0	31	0.91
Hollandse kust	6	0.246	0	2.83	0	31	0.92
Zeeuwse- and ND kust	6	0.064	0	2.85	0	37	0.94

Assessment results are very similar for both water bodies (Figure 72). DKI, IQI and NQI are hardly different between water bodies. BEQI and M-AMBI show some more variation between Hollandse Kust and Waddenkust. In all cases the outcomes are higher when the local determined reference settings are used, in three out of six cases use of the local reference even results in shift to a higher quality status. Of the five compared methods the BEQI method assigns both water bodies to a lower quality class than the other methods, regardless of the reference used fixed or local. Differences in assessment are as large as between poor and high status. These differences are caused by differences in the applied methodology. In the first place total biomass of the macrobenthic community is included in the BEQI. Biomass has increased considerably in the North Sea coastal zone since the period of reference setting, causing very low scores for the sub-metric biomass and subsequent low overall scores. Also the measure of similarity between reference and assessment only included in the BEQI method was rated not more than moderate at the Hollandse Kust and Waddenkust. Besides the use of different input and sub metrics BEQI relates assessment and reference conditions differently. BEQI explicitly takes spatial and temporal variability into account. The class boundaries are determined by the variance in the reference data. A reference dataset with limited variability will result in much narrower class boundaries than a reference dataset with large variability. In all cases the differences in conditions between BEQI class boundaries are smaller than in the other methods. For instance in M-AMBI the classes cover the whole range of conditions between maximum parameter values and zero values for azoic sediments. This difference causes BEQI assessments to deviate sooner from the high and good status than the other methods.

To increase the number of data for comparison, index scores are calculated for several time intervals (four periods: 91-95, 96-98, 99-01 and 02-04). BEQI responds strongest to the temporal and spatial differences with a coefficient of variation (cv) of 0.12. M-AMBI showed less variation in time and space with a cv of 0.09. The other indices DKI IQI and NQI responded least with cv's of 0.06, 0.05 and 0.05 respectively. M-AMBI, DKI, IQI and NQI are all positively correlated (Fig. 2). This is not really surprising as the indices rely on the same or similar sub metrics. The BEQI correlates poorly with the other methods and the correlations tend to be negative but not significantly.

Based on this very limited comparison between BEQI and four other indices, BEQI seems to rate water bodies lower than the other methods. Besides that the behavior of BEQI is different or more sensitive than that of the other indices being very similar to each other. Further comparisons with a larger dataset with a larger variation in conditions are needed to further evaluate the performance of BEQI relative to the other indices.

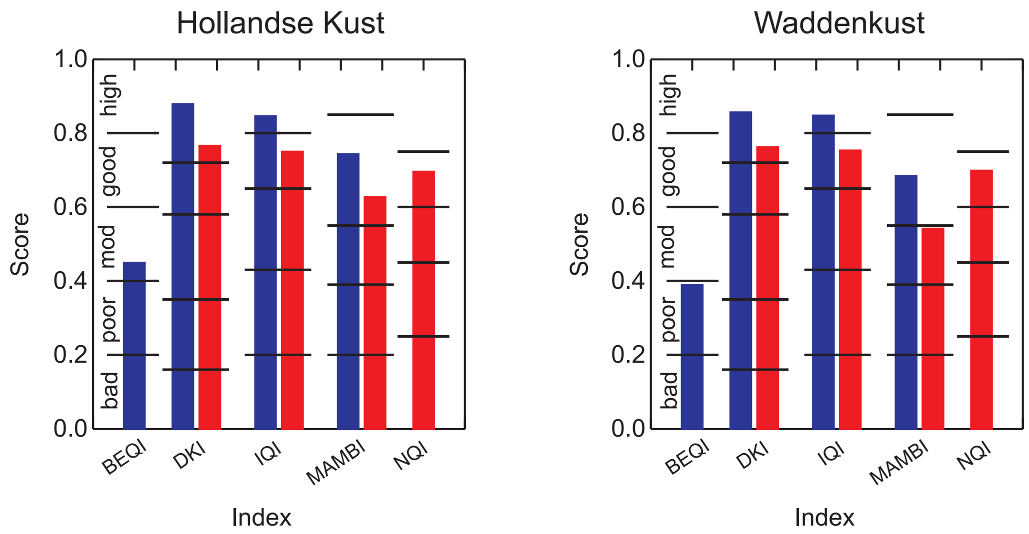


Figure 72. **Bar graph of five metrices calculated for two water bodies of the Dutch coast. Calculations are done using a fixed reference (in red) suggested by the member states and a local reference (blue) determined from a reference data set (Q1 cluster). Boundaries of the indices are indicated with horizontal lines**

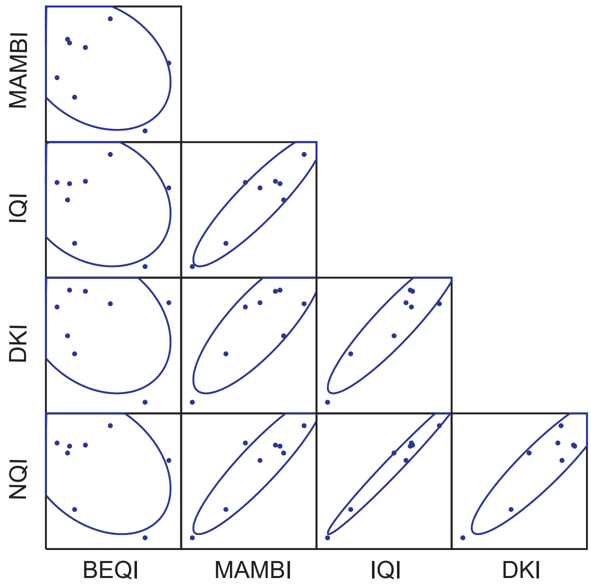


Figure 73. Scatterplot matrix of the index scores of the different methods calculated for two water bodies and four time periods in the coastal zone of the Netherlands, Waddenkust and Hollandse Kust. Sample ellipses are plotted together with the data

Type – NEA 4

At present Germany applies the M-AMBI to the Wadden Sea, type NEA 3/4 waters. A comparison is made between the results of the BEQI method for the different habitats of the Dutch Wadden Sea, reported in paragraph 4.3.2.5.3 and M-AMBI values calculated for the same assessment data. Additionally the BEQI and M-Ambi are calculated for benthos datasets from Norderney and the Leybucht, supplied by Germany for the intercallibration exercise. Norderney data cover a five year period from 1998 to 2002, for the Leybucht data from a longer period are available and there the six most recent years from 1998 to 2003, are selected for the BEQI and M-AMBI calculation. Information on biomass of the macrobenthic species is not available, so only three of the four sub-metrics are calculated.

Results of the BEQI calculations are presented in Table 61. For Norderney reference conditions of Middle Littoral Muddy Sand are used. Leybucht results are relative to the reference for High Littoral Mud. The outcomes of the two areas are similar. Total densities of the macro fauna are higher than the reference. Number of species is very large, in both areas more species are found than described in the reference data. Similarity is rated poor in both areas. The overall score is good for Norderney and moderate for the Leybucht. This difference is due to the larger deviance of total density in the Leybucht. This very strong divergence from the reference conditions is caused by high numbers of *Hydrobia ulvae, Tubificoides benedeni* and *Corophium* sp. This could indicate a disturbed situation, however it is questionable if the reference description is appropriate in this case. No *T. benedeni* or other Oligochaetes are present in the reference set, which might suggest that this group was neglected in the Dutch monitoring, or alternatively this ecotype was not well covered. However, excluding *T. benedeni* in the calculations still leads to a poor density status.

Table 61. BEQI results of three sub-metrics for Norderney (years 1998 to 2002) and Leybucht (years 1998-2003) in the German Wadden

Norderney (Middle Littoral Muddy Sand)

	Assessment		EQR	
Parameters	surface	value	score	status
density	14.34	8206	0.51	moderate
species	14.34	47	1	high
similarity	14.34	0.54	0.38	poor
Overall			0.63	good

Leybucht (High Littoral Mud)

	Assessment		EQR	
Parameters	surface	value	score	status
density	4.98	21784	0.16	bad
species	4.98	32	1	high
similarity	4.98	0.47	0.37	poor
Overall			0.51	moderate

M-AMBI scores are calculated for the same six assessment samples as are used for the BEQI scores, four habitats of the Dutch Wadden Sea presented in paragraph 4.3.2.5.3 and the two German sites discussed above. M-AMBI scores are calculated per station and then averaged per habitat. Reference values are taken from the reference datasets for the habitats in the Dutch Wadden Sea (Table 62).

Table 62. Reference values used for M-AMBI. Reference value for AMBI is the minimum AMBI value of all AMBI scores calculated per sample in the reference data set. Shannon Wiener and number of species are maximum values in the reference data set. N-species is not corrected for differences in sampling surface

Habitat	AMBI	Shannon Wiener	n species
Brackish Sub-Littoral	1.54	2.18	16
Marine Sub-Littoral Low Dynamic	0.38	2.39	16
Marine Sub-Littoral High Dynamic	0.18	2.19	17
High Littoral Mud	2.70	2.16	18
Middle Littoral Muddy Sand	0.95	2.34	23
Low Littoral Sand	0.39	2.22	17

A comparison between the BEQI and M-AMBI scores for the Dutch and German Wadden Sea habitats is made in Table 63. In three out of six cases the status classification is the same for both metrics. In the other three cases BEQI gives a lower quality status, twice moderate versus good and once moderate versus high for the Leybucht. In the Leybucht the difference is mainly caused by the very high abundances of three species with much stronger effect on BEQI than M-AMBI. Moderate BEQI scores in the Dutch Brakish Sub-Littoral and High Littoral Mud habitats are caused by large biomass values relative to the reference and moderate similarities. The submetrics biomass and Bray Curtis similarity are not accounted for in the M-AMBI and therefore not affecting the M-AMBI scores. This explains at last partly the difference in status score.

A more detailed analysis of the comparison between BEQI and M-AMBI results for the Wadden Sea does not seem adequat until the number of cases is substantially larger. In the next section data from the Coastal zone of the North Sea, the Wadden Sea and the Eems-Dollard are combined to increase sample size and variation in assessment conditions for a comparison between the two indices.

Table 63. BEQI and M-AMBI assessment results for the Dutch and German Wadden Sea habitats. The samples in the Dutch Wadden Sea consist of stations in several areas

Country& period	Habitat	Area	BEQI		m-AMBI	
			EQR	status	**EQR**	status
		Netherlands				
2003-2005	Brakish Sub-Littoral	Western Wadden Sea	**0.57**	moderate	**0.79**	good
2003-2005	High Littoral Mud	Balgzand Groninger Wad	**0.56**	moderate	**0.767**	good
2003-2005	Middle Littoral Muddy Sand	Balgzand Groninger Wad Piet Scheveplaat	**0.73**	good	**0.683**	good
2003-2005	Low Littoral Sand	Balgzand	**0.795**	good	**0.8**	good
		Germany				
1998-2002	Middle Littoral Muddy Sand	Norderney	**0.63**	good	**0.66**	good
1998-2003	High Littoral Mud	Leybucht	**0.51**	moderate	**0.85**	high

For further comparison between BEQI and M-AMBI results from the North Sea Coastal waters, Wadden Sea and Eems-Dollard are combined. This generates a larger dataset with a larger range of variation in conditions reflected in the indices. M-AMBI scores are calculated per assessment station and averaged per habitat. M-AMBI reference values are selected from the same reference datasets as are used for the BEQI reference description. Calculation of BEQI scores and results are presented in previous scions of this report.

The comparison was made based on 20 observations from three water bodies. In Figure 74 the AMBI scores are plotted against the BEQI scores. The range of M-AMBI scores is 31% and of BEQI 43% of the scale from 0 to 1. In this limited comparison most M-AMBI assessments fall within a single category, good and a few in high. The BEQI status assessments mainly fall within two categories moderate and good. Overall the two indices are positively correlated with a Pearson correlation coefficient of 0.45 (*P<0.05*). Within water bodies scores are not related. In five out of twenty cases the status is judged similar by BEQI and M-AMBI, in the fifteen remaining cases the M-AMBI status was higher than the BEQI status.

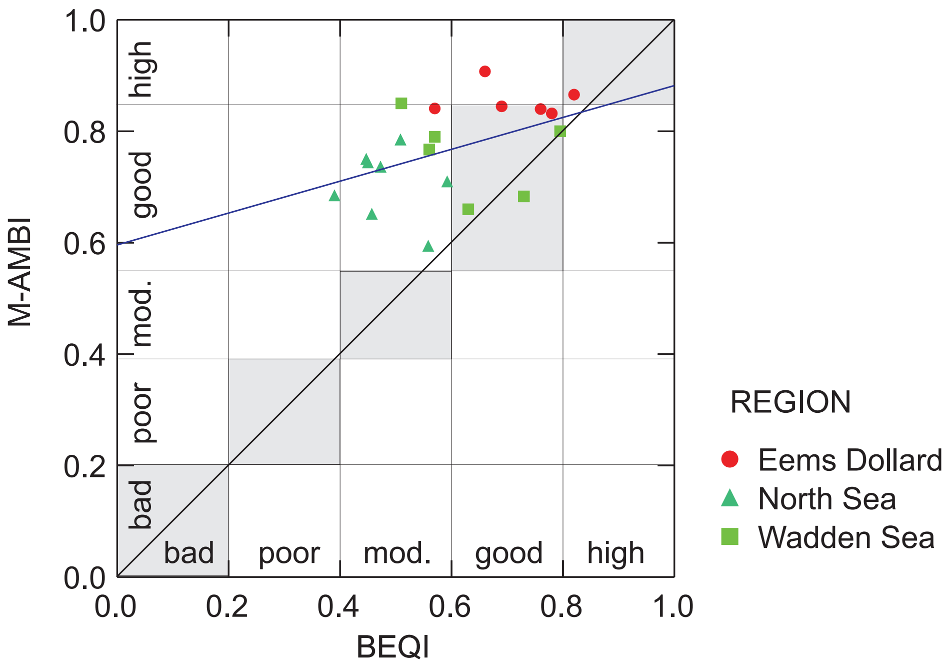


Figure 74. M-AMBI scores plotted against BEQI scores from three water bodies. Quality boundaries are included. Points in the shaded regions have the same status for both indices. The black line is the x=y relationship. The blue line is the linear relationship between M-AMBI and BEQI

BEQI and M-AMBI are indices with several differences in methodology. This leads to differences in the outcomes when applied to the same data. On a large scale there is agreement in the direction of the response of the two methods, shown as a positive correlation. However BEQI is more sensitive to changes in the assessment conditions relative to the reference than M-AMBI.

2.3.5 Results of the harmonization – Boundary EQR values

Type – NEA 1/26 and NEA7

	Good/Moderate	High/Good
Denmark	0.53	0.67
Ireland	0.64	0.75
France	0.53	0.77
Spain	0.53	0.77
Portugal	0.53	0.77
Norway	0.81	0.92
UK	0.64	0.75
NL	0.6	0.8

Type – NEA 3/4

	Good/Moderate	High/Good
The Netherlands	0.6	0.8
Germany	tbc	tbc

Annex 2.1.

The Netherlands:

The BEQI-method on the third level evaluates the state of the benthos within a habitat based on 4 parameters (parameters): number of species, density, biomass and species composition changes. These parameters reflect the normative definitions as defined by the WFD. Any proposed WFD classification scheme must include methodologies that address the following parameters defined for assessing the benthic quality status: `the level of diversity` (BEQI parameter: number of species) and `density of invertebrate taxa` (BEQI parameter: density). The WFD parameter `proportion of disturbance-sensitive taxa` is not entirely the same in the BEQI methodology (BEQI parameter: species composition changes), because it does not classify species in disturbance sensitive taxa classes, but reflects wider species composition changes (see further). The BEQI WFD classification scheme also includes biomass as a parameter, which is not directly required by the WFD, but which can be considered as additional estimate of the abundance. Biomass data reflect the partitioning of resources within the community better than density data and for shallow coastal and estuarine systems the biomass is a relevant parameter to link with the higher trophic levels such as birds and fish. The parameter 'species composition changes' evaluates changes in the species composition of the macrobenthic community belonging to a certain habitat. This approach allows detecting changes in the dominance of species, the disappearance of species from the community and the appearance of new (e.g. invasive) species in the community. In this case, the species are not classified in disturbance-sensitive taxa, but all species are treated similarly. These four parameters are calculated and evaluated per habitat (multisample) within a water body instead of per sample within a water body.

The parameter results strongly depend on the sampling effort (sediment surface) that is deployed. Therefore, the expected reference values for the parameters are calculated per habitat from permutations (KRW program, version 1.0 developed by Peter Herman in FORTRAN) executed over increased sampling surfaces. This allows estimating, for any given sampling surface, the reference value that can be expected. This program is based on bootstrapping with replacement. In statistics, bootstrapping is a modern, computer-intensive, general purpose approach. It is used for estimating the sampling distribution of an estimator (parameter) by re-sampling with replacement from the original samples, most often with the purpose of deriving estimates of standard errors and confidence intervals of a population parameter like the median (Efron, & Tibshirani, 1993).

The basic algorithm of this permutation test works as follows (also illustrated in box 5):

(5) A reference dataset is chosen, that is large enough to represent the main factors of variability in the habitat (temporal and spatial variability). In the reference dataset, all

samples may have the same sample surface or not, but the surface sampled is taken into account in all subsequent calculations

(6) For a range of sample surfaces (the calculation restarts for every level of surface sampled), approximately 2000 random samples are drawn with replacement from the reference database. This is done by choosing a sample (at random) from the reference database, adding its density, biomass etc. and its surface to intermediate storage variables, and continuing this process until the desired surface is reached. For example, when doing the permutation test for an 'assessment' sample of 1 m^2, sufficient samples are drawn from the reference dataset so that their surface amounts to the desired surface of 1 m^2, and sum all species abundance and biomass data into one species list with density and biomass attributes for this one random sample. Subsequently, the procedure is repeated 2000 times for this 1 m^2 surface. The end result is then a set of 2000 artificial random assessment samples with a surface of 1 m^2.

(7) For each of the 2000 random samples with a particular surface, their total density and total biomass is determined, as well as their species richness and similarity to the species composition of the complete reference data set. This yields 2000 values for each of the parameters density, biomass, species richness and similarity. A number of relevant percentiles of this distribution are determined that will serve a purpose in determining critical limits (see below for the procedure for the different variables).

(8) This whole permutation calculation is repeated for the next level of surface of the assessment sample, until an assessment surface equal to the cumulative reference surface has been calculated.

Reference values for each parameter, based on those permutations, are determined for each ecological status class boundary (Figure 75).

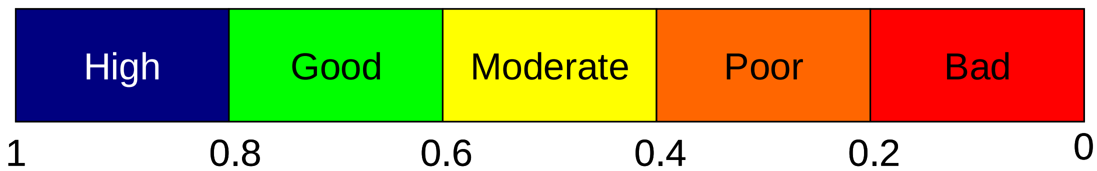
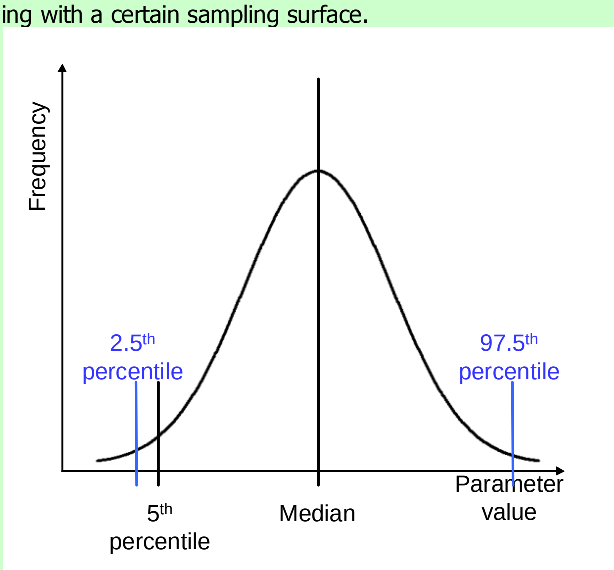


Figure 75. Ecological status class boundaries for the Netherlands

The good/moderate boundary is important within the WFD because it marks the boundary between action and no action to improve the status of the evaluated water body. When the status of the water body is good, no action has to be taken, while some action has to be taken when the water body is evaluated as moderate, poor or bad. For the different parameters, the reference value to be expected in the case of a good status corresponds with the 5th percentile value out of the permutation distribution of each parameter (details in box 5). The 5th percentile is a statistically accepted level, which is not too restrictive and which accounts for the variability within the reference data. Therefore, this level is considered to be suitable as criterion for the important good/moderate boundary, following the WFD. Concerning the boundary setting, it has to be mentioned that in artificial and heavily modified water bodies, the high and good boundary are respectively defined as maximum ecological potential (MEP) and good ecological potential (GEP). In the further analysis, those boundaries are all named as high and good, independent of the water body type.

Box 5 – Randomisation and defining of reference boundaries.

As described higher, for each sampling surface, at random 2000 samples are drawn with replacement from the reference database. Such result is illustrated in the figure below. This distribution is used to define the reference boundary values corresponding with a certain sampling surface.



For each level the 5th percentile is selected as the good/moderate reference boundary, which means that for a one side approach (as for number of species and similarity), the 5% lowest values are deviating from the reference situation. The median value of this distribution is selected as the good/high reference boundary. For a two sided evaluation (as for density and biomass), the 2.5% lowest and highest values are selected as the boundaries at which there is a deviation from the reference situation.
These values of the randomization are calculated for each sampling surface and are visualized in Figure 5 and Figure 6.

The different parameters and the calculation of the reference values and boundaries are described per parameter in the following section and summarized in Table 64.

Number of species (species richness)

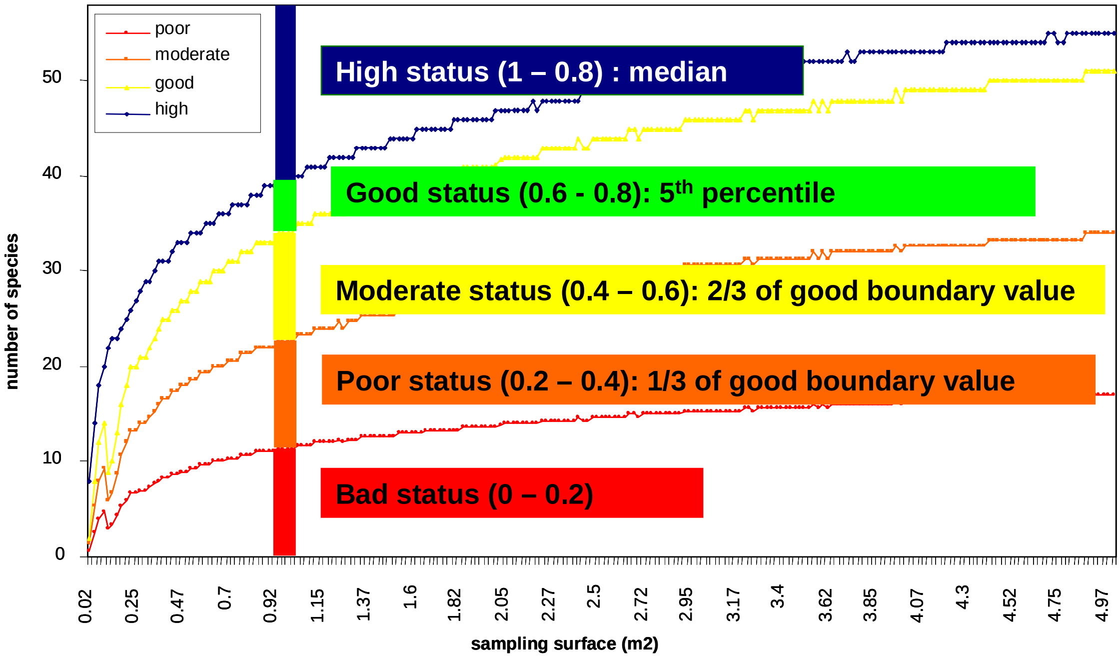


Figure 76. The reference boundary settings for number of species (used as example in the figure) and similarity in relation to sampling surface.

The number of species is an important diversity parameter in evaluating the status of a habitat and is widely used as an indicator for environmental changes or disturbance (Pearson & Rosenberg, 1978; Borja et al., 2007). Therefore, this parameter is selected to evaluate the diversity changes for the different habitats within a water body.

The permutation of the number of species allows estimating the range in the number of species that can be expected to be collected for any given sampling surface (Figure 5). Within this range, the median value is used as the lowest number of species to be expected in the case of a high status. The number of species corresponding with the 5th percentile is used as the lowest number of species to be expected in the case of a good status. The moderate/poor and poor/bad boundaries are scaled in equally intervals relative to the number of species measured for the good/moderate boundary and are respectively 2/3 and 1/3 of the number of species of the good/moderate boundary. The interval of 0.2 between the ecological status boundaries is equally scaled in correspondence to the interval between the number of species of the corresponding reference value boundaries.

Species composition change

This parameter aims at evaluating changes in species assemblage structure (species dominance, occurrence of new species and disappearance of species) between the assessment and reference situation. The evaluation of those changes is based on the Bray-Curtis similarity between the two datasets, after a fourth root transformation. In this case, the species are not classified in disturbance-sensitive taxa, but all species are treated similarly.

A similarity distribution of a certain sampling surface is calculated based on randomization of the reference samples to a selected sampling surface and on the calculation of the similarity of those randomly selected samples to the total reference sample (all samples), which results in a similarity range for a certain sampling surface (Figure 5). The 5th percentile of this similarity range is used as boundary for moderate/good. The median value is used as the lowest similarity to be expected in the case of a high status. The moderate/poor and poor/bad boundaries are scaled in equally intervals relative to the similarity measured for the good/moderate boundary and are respectively 2/3 and 1/3 of the number of species of the good/moderate boundary. The interval of 0.2 between the ecological status boundaries is equally scaled in correspondence to the interval between the similarities of the corresponding reference boundaries.

Density and Biomass

Both the macrofauna density (ind.m^{-2}) and biomass (g AFDW.m^{-2}) are treated in a similar way. The description below refers to density but also fully applies to biomass.

The permutation of density allows estimating the range in density that can be expected to be collected for any given sampling surface (Figure 6). A lower as well as a higher deviation (two sided evaluation) of this range is evaluated as negative, because an increase in density does not always indicate an improvement of the ecosystem. In fact, density increase is classically considered an indicator for organic enrichment problems (e.g. Rosenberg, 2005). Therefore, for each boundary a minimal and maximal density is defined for any given sampling surface. When the average density in the assessment falls between the 25th and 75th percentile of this range, a high status is reached. The range in density that can be expected in the case of a good status corresponds with the 2.5th and 97.5th percentiles. Due to this two-sided evaluation, the 2.5th and 97.5th percentiles are selected to border the good/moderate boundary, instead of the 5th and 95th percentiles to get a same total 5% selection of deviation of the reference state for the four parameters. The moderate/poor and poor/bad boundaries are scaled in equal intervals relative to the minimal or maximal density measured for the good/moderate boundary and are respectively 2/3 and 1/3 of the minimal density value and 4/3 and 5/3 of the maximal of the density value of the good/moderate boundary. The interval of 0.2 between the ecological status boundaries is equally scaled in correspondence to the interval between the minimum or maximum densities calculated for the corresponding reference boundaries.

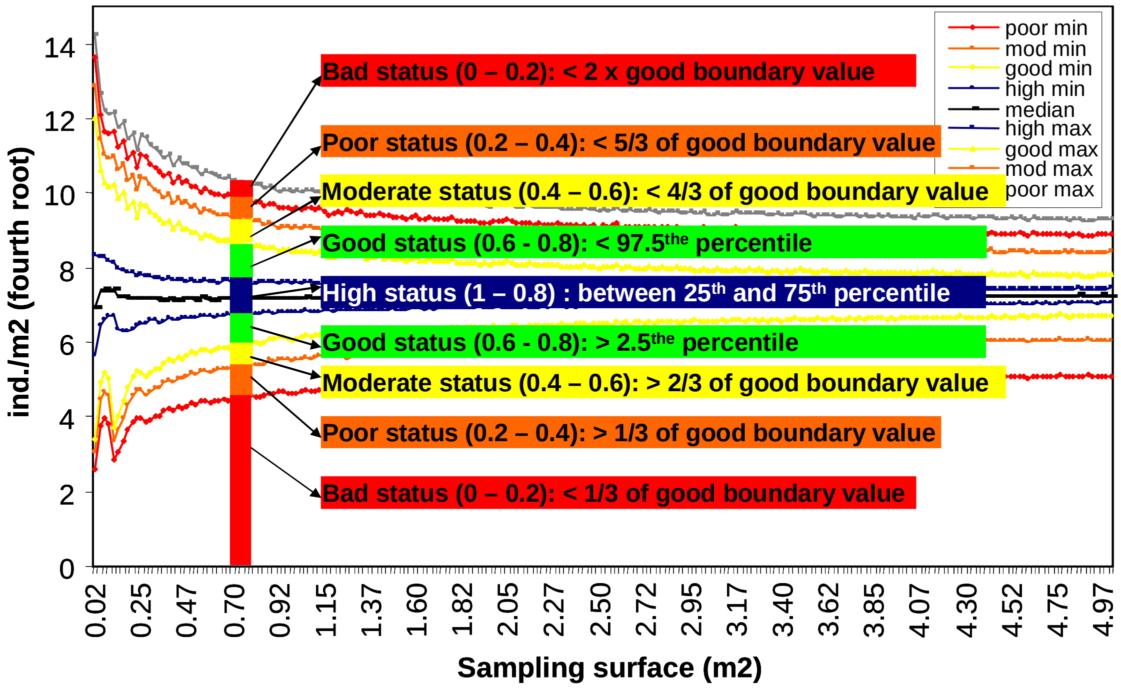


Figure 77. The reference boundary settings for density in relation to sampling surface. A similar protocol is applied for biomass

Table 64. Overview of the reference value boundary settings coupled to the fixed ecological status class boundaries

Ecological status class boundary	**Reference value boundaries**	
	Number of species Species composition changes	Density Biomass
High/good: 0.8	median	25^{th} and 75^{th} percentile
Good/moderate: 0.6	5^{th} percentile	2.5^{th} and 97.5^{th} percentile
Moderate/poor: 0.4	2/3 of Good/mod value	2/3 and 4/3 of Good/mod value
Poor/Bad: 0.2	1/3 of Good/mod value	1/3 and 5/3 of Good/mod value

Risk of misclassification

Precision and confidence of the classification method are crucial elements for decisions to invest large sums to improve the ecological quality (Carstensen, 2007). Within the WFD, the good/moderate boundary is important, because it marks the boundary between action and no action to improve the status of the evaluated water body. When the status of the water body is good, no action has to be taken, while some action has to be taken when the water body is evaluated as moderate, poor or bad. Due to the selection of the 5^{th} percentile, there is a chance of 5% that a parameter is evaluated as bad/poor or moderate, while in fact it is good. Note, however, that this percentage applies to each of the parameters separately, and that it is much more difficult to quantify the risk of misclassification for the combined index. When the four parameters would be independent, the risk of misclassifying all of them is negligible (order 10^{-6}), but independence is unlikely and the real risk will be between virtually zero and 5 %. The probability that the combined index based on the four parameters leads to misclassification is, similarly, well below 5 % but the exact probability level is unknown.

Overall EQR of the community level

The overall Ecological Quality ratio (EQR) of a habitat within a water body is calculated by averaging the Ecological quality ratios of the four parameters (density, biomass, number of species, species composition changes) of that habitat. When the average is situated between 1-0.8, 0.8-0.6, 0.6-0.4, 0.4-0.2 or <0.2, than the Ecological status of the water body at the community (within-habitat) level is respectively high, good, moderate, poor or bad.

Interpreting the EQR values at the community level

The aim of the community level evaluation of the BEQI method is to detect and evaluate changes in the assessment data compared to the determined reference condition for each parameter, within a habitat of a water body. These results are scaled in accordance with the requirements of the WFD.

The outcome of these calculations can be as follows:

→ The overall Ecological Quality ratio (EQR) and the EQR of the parameters of an habitat within a water body are evaluated as good or high.

This should be interpreted as a status quo in the Ecological Quality status in comparison with the reference situation.

→ The overall EQR was evaluated as good or high, but one or more EQR values of the parameters are evaluated as moderate, poor or bad.

This should be interpreted as a first warning of possible changes in comparison with the reference situation. A more detailed investigation (e.g. through operational or investigative monitoring) in the changed parameter is advisable.

→ The overall EQR was evaluated as moderate, poor or bad.

This should be interpreted as a strong change in comparison with the reference situation. This requires action of the government in co-operation with the scientific community to investigate and tackle those changes. Based on a detailed investigation of the results the managing authorities – in close cooperation with the scientific community – can decide to take measures or to install an operational or investigative monitoring program.

7. Research proposal

This chapter contains a first draft research proposal that can serve as input for an international, collaborative research project within e.g. FP7. With the first launch of a draft work program of Theme 6 - Environment (including climate change) – for the next FP7 the different activities, sub-activities and areas are screened for possibilities to initiate or collaborate on a research proposal. One of the topics – assessing the ecological status of water bodies – fits well within the scope of this research proposal. In an annex the description of this topic is given. –

1. Problem definition

With the introduction of legislative initiatives such as the Water Framework Directive (already in operation) and the future European Marine Strategy Directive, the role played by ecological indicators has fundamentally changed. Whereas these indicators are previously used as descriptors of the system state for scientific and public communication purposes, they have now obtained legal value. When the ecological status of certain systems is judged insufficient (based on the use of the accepted indicator), action must be taken and large costs and efforts may be involved. It is thus of utmost importance that the indicators used in the European context are thoroughly tested, well justified from a scientific point of view, and generally accepted throughout the EU.

Much effort has been spent on constructing environmental quality indicators for the Water Framework Directive. In this context, however, the so-called transitional and coastal waters (estuaries and a narrow coastal strip) remain highly problematic from the point of view of indicator development. Indicator systems developed for freshwater cannot be easily transposed to transitional and coastal waters for a number of reasons:

- concentration ranges for nutrients, chlorophyll and other environmental variables are different in freshwater and brackish/marine waters, necessitating different concentration standards.
- virtually all species occurring differ between fresh and brackish/marine waters
- In estuaries and coastal seas, the role of physical forcing in the form of exchange of water with the ocean, tides and currents is far larger than in freshwater systems. This implies that the habitat is shaped to a large extent by these physical forces and that this factor must be taken into account. Furthermore, the relative role played by physical

forcing in different European coastal waters may be very different, compare e.g. a Danish fjord with a megatidal Atlantic estuary.

- The relative role played by internal ecological dynamics, compared to direct external forcing, is much larger in systems with a long water residence time, such as estuaries and coasts, than in flow-through systems such as rivers. Consequently, the indicators will have to capture somehow these internal dynamics and reflect the functioning of the system.
- The threats to coastal and estuarine systems are probably more diverse than those of freshwater systems. In addition to chemical pollution and eutrophication, physical changes to the system (harbour construction, dredging, bottom trawling, coastal defense, land reclamation), changes in hydrology and sediment flow, and climate changes (e.g. the effects of sea level rise) will have to be included into the indicator system.

In this (sub) proposal we focus on the indicators using benthic macro-invertebrates (macrozoöbenthos) data for transitional and coastal waters. At present, a number of indicators have been proposed by the Member States for the Water Framework Directive. These indicators are currently being intercalibrated and implemented with respect to monitoring needs. In a review and intercalibration exercise of the existing indicators, it became clear that the basis of most indicators is rather similar, and rather restricted. Most indicators, including the widely used AMBI index, are based on the classical Pearson-Rosenberg continuum of community composition in response to organic enrichment pressure (Pearson & Rosenberg, 1978). According to this paradigm, benthic species are divided into previously defined ecological groups and then determining the respective proportion of the different groups in the benthic communities. Most indices are complemented with a measure of diversity within the communities, but very often both measures coincide to a large degree (see e.g. Borja et al., 2004, 2007; Rosenberg et al. 2004; Dauvin et al., 2007).

A number of problems can be identified relating to the scientific basis of this class of indicators.

Firstly, although eutrophication remains an important problem in much of the European coastal waters, it is not the only human interference that needs to be identified by the indicator system used. In particular, physical damage to the coastal and estuarine ecosystem, following from land reclamation, dredging, bottom trawling, harbour construction and coastal defense with their consequences needs to be taken into account. It is well known that the composition of benthic communities is highly dependent on factors depending directly or indirectly on hydrography in estuarine and coastal systems (e.g. Ysebaert et al., 2002). One cannot expect the same diversity, biomass, density or species composition in a highly dynamic part of a tidal flat characterized by sand waves and coarse sediment as in a sheltered, muddy part (Herman et al., 1999). A single indicator system could never cope with these two different systems, both of which are a natural part of a tidal estuary, without at least a correction for the physical circumstances. However, merely correcting for the circumstances neglects the problem that, due to anthropogenic stress, the spatial distribution of these habitat types may change fundamentally. An indicator should therefore reflect the distribution of habitat types, as well as the distribution of species within these habitats. Otherwise, it will be insensitive (or worse: evaluate positively) the degradation of estuarine habitats due to physical interferences.

Secondly, all of the recent indices provide information about the relative abundances of the sensitive species faced with increasing organic matter in the sediment and those of the species that are resistant or indifferent to such increases, or even favored by such conditions (the so-called opportunistic species). But the main problem is that all the indices, which aim to determine anthropogenic stress, relate to abundances of stress tolerant species, which may also be tolerant of natural stressors such as in estuaries. Similarly, many of these indices relate to anthropogenically organic-rich systems whereas estuaries are often naturally organic rich systems.

Thirdly, the focus on intrinsic diversity of the benthic community may neglect the important functions of this community in the ecosystem. Diversity is, generally speaking, highest in low to intermediate productivity regions. However, productivity in itself may be an important aspect that cannot be neglected. Intertidal benthic populations are a very important food source for (protected) bird populations, and their productivity is a crucial factor in determining the number of birds that can survive in a particular region. Loss of important populations like cockles has been documented in the Wadden Sea to lead to loss of important numbers of overwintering birds (van Gils et al, 2006; Phillippart et al., 2007). Thus, benthic populations are a key element in the carrying capacity of the estuaries for their diversity of ecosystem functions. Moreover, this productivity cannot be regarded on itself, but must be seen as a function of other changes in the ecosystem. For the Wadden Sea system, it clearly (although non-linearly) relates to changes in nutrient ratios and in particular to the decrease in phosphorus input combined with a rather steady input of nitrogen (Phillippart et al., 2007).

Fourthly, in the current practice benthic indicators are used independently from other indicators in the system (e.g. nutrients, chlorophyll, primary production, composition of the plankton). Although all indicators may eventually be combined in a multi-metric approach, the danger exists that the law of large numbers will apply (some very bad indicators combine with some good, producing an average end number in all cases), or even that intrinsically inconsistent results (different parts of the ecosystem point to quantitatively or qualitatively different stresses) become smothered in an 'average' overall outcome. Insight in the functioning of ecosystems is presently well enough advanced to know that many variants of cause-effect chains are possible (see e.g. Cloern (2001) for an enlightening review on possible scenarios for eutrophication). The time has come to put these conceptual ideas to the test in developing a new generation of indicators that summarize the ecosystem dynamics in a synoptic view. As an example, simultaneously measuring primary and secondary production gives the data to assess the efficiency of energy transfer between lower trophic levels, providing an indication about ecosystem structure and functioning, and biological relationships.

Besides the problems identified above, the current indicator systems are not aimed to identify the effects of climate change. Marine organisms will be influenced by changes in circulation, ventilation, and stratification through changes in temperature, light, and nutrient supply. Alterations of any of these drivers may lead to changes in species abundance and composition, possibly leading to large-scale regime shifts and species migrations. Also habitat loss, resulting from sea level rise, and invasion by non-native species will also perturb marine ecosystems, affecting the health and biodiversity of marine ecosystems.

A final remark in the problem definition concerns monitoring methods and approaches. Any indicator system is crucially dependent on monitoring efforts, and often this aspect is the most expensive part of the total indicator system. Technological innovation in environmental monitoring systems has advanced well in recent years. Nevertheless, most monitoring for legal purposes still depends heavily on traditional monitoring techniques. These techniques typically have discrete sampling at isolated points, restricting the spatial and temporal scales at which changes can be detected. In this (sub)proposal we propose to further develop promising new monitoring techniques, such that a more synoptic view of important ecosystem parameters can be obtained, which can be linked and translated to better indicators through operational mathematical modeling. By combining satellite or airborne observation data with physical and ecological models, currently progress is being made toward the goal of using remote sensing to relate changes in climate and other environmental factors to changes in the distributions of organisms and how they relate to their environments. Improvements in our ability to remotely sense the three-dimensional structure of habitats also help to meet the goal.

Summarizing, we propose to innovate the use of indicators in transitional and coastal waters by:

- focusing on the diversity of human stressors that are important in these systems
- embedding the indicators in an ecosystem view, and dynamically link a number of ecosystem variables into a meaningful indicator system

- deploy new monitoring techniques combined with operational modeling as the basis of the indicator system

By doing so we will develop reliable indicators, including key ecological attributes as well as data-analysis techniques, for measuring the health of the benthos in relation to human and natural stressors.

2. Work proposed

In order to reach the proposal's objectives, work will have to be delivered to (i) develop the conceptual basis for the new generation of indicators (ii) test the indicators and investigate their sensitivity for different contrasting areas in European waters based on (iii) a consistent data management system for a number of test areas; further we have to (iv) link the indicators to novel monitoring methods and (v) develop interfaces to legal authorities and to the general public in order to inform them about the indicators and their results.

This work will be divided in a series of corresponding work packages, the general aim of which will be shortly described below.

2.a. Development of the conceptual basis for the new generation of indicators

This work package is at the core of the current proposal. It aims to analyze the system of indicators presently proposed for or used within the Water Framework Directive, in the light of two different criteria. Firstly: do the indicators cover the full set of relevant anthropogenic threats for transitional and coastal waters, and if not how do they (implicitly) weigh these different pressures. Secondly: are the indicators consistent with current theoretical insights in ecosystem dynamics, are the indicators for different parts of the ecosystem mutually consistent, and do they cover the full range of known reactions of coastal ecosystems to the different relevant anthropogenic pressures.

This critical analysis will be based on two major sources of information. First, literature will be reviewed to extract in a consistent way the existing knowledge about ecosystem reactions to anthropogenic pressures. The particular challenge for the project in this respect is that biodiversity aspects (the 'structure' of the community) will have to be linked to functional aspects (in particular the flows of matter and energy) as well as spatial configurations (the translation of physical forces into the habitat structure). We think that far more information about these aspects is available than what is currently translated into the indicator systems, but the task will nevertheless be quite extensive. Secondly, actual data about a number of example systems, as collected in work package II, will be used to illustrate and check the proposed concepts.

The anticipated results of this work package is a set of indicators that are sensitive to all relevant pressures on the coastal system, that can discriminate between these pressures on the basis of the realized changes in the system, and that are consistent throughout the different trophic levels of the ecosystem.

2b. Test the indicators and investigate their sensitivity for different contrasting areas in European waters

In this work package, first of all a number of European coastal waters will be selected based on data availability on the one hand, and diversity of anthropogenic pressures on the other hand. There are a number of coastal areas that have been quite well monitored over the past decades, and where moreover a considerable body of literature and databases have been collected on different aspects of the ecosystem functioning. We can mention the following examples: Westerschelde estuary (organic loading, pollution, dredging, land reclamation), Wadden Sea (increasing and decreasing nutrient loading, fisheries, variable loading with fine sediment, threat from relative sea level rise), Seine estuary (man-made modifications, dredging, heavy metal pollution), Humber estuary (expanding port complex and extensive industries, etc.), Elbe estuary (harbor development, dredging).

For the selected areas, a consistent data base of available monitoring data will be constructed in Work Package III. Within WP II, emphasis will be on the analysis of these data sets. This analysis will proceed in several steps. First, available meta-data on the systems will be used to make an inventory of the anthropogenic stress. Second, based on the conceptual analysis in Work Package I, as well as on basic ecosystem models for coastal ecosystems, an estimate will be made of how these stressors may translate into the structure and functioning of the ecosystem. Third, these predictions will be tested based on available data, which can be the basis for a re-analysis of the system. Finally, the outcomes of the new indicators, as applied to these systems, will be compared to the results of the previous, more ‘free’ analysis. This will then serve as input to Work Package I, where the conceptual basis of the new indicator systems can be redefined or fine-tuned based on this experience.

2c. Collection, management and publication of relevant databases on the example systems.

This work package will be servicing both WP II as WP IV. Existing data, as well as newly collected data using novel monitoring techniques, will be stored in a central database to the project. Within this WP, not only the collection and management of these data will be central, but also the development of software tools that can be used to query these data and calculate existing and newly developed indicator values. Finally, the work package will also be responsible for the distribution of the results of these calculations, including the software tools and example applications that can be tested in other systems.

2d. Link the new set of indicators to novel monitoring techniques.

Recently, a number of novel monitoring techniques have been developed that are very promising for a further development in the framework of ecosystem quality indicators. Relevant examples of such techniques are: ‘ferrybox’ applications of equipment on board ships of opportunity, allowing continuous and automatic monitoring of nutrients, chlorophyll, phytoplankton productivity, as well as suspended matter concentration and current velocity profiles over the ship’s track. The compilation of such data can allow the measurement of concurrent datasets over broad geographic ranges.
Remote sensing techniques can be used to measure chlorophyll concentrations, even in turbid coastal waters; use of remote sensing techniques to estimate sediment composition, microphytobenthic biomass and production, and to estimate benthic species composition on tidal flats; use of automatic measuring stations that can follow both hydrographic and ecological variables in a fixed position, even making regular vertical profiles; use of advanced sonar-based technology (e.g. multibeam) to estimate the sediment composition and morphology of the seabed; use of ADCP to measure particle load of the water, as well as current profiles and even turbulence levels.

The major challenge posed by these new technologies is not so much in their technicalities (although this is still of concern), but in the question how they can be most optimally used for a regular monitoring servicing relevant indicators of the ecosystem state. Therefore, within the present project this question will be central. It has consequences for the development of indicators (there is little use in developing indicators using variables that cannot be routinely measured), as well as consequences for how to deploy (and possibly further develop) the novel monitoring techniques.

The aim of this work package is to design an optimal combination of indicator qualities and monitoring techniques, such that a maximum of relevant information can be provided for a minimal monitoring effort. Also, the work package aims at developing clear criteria for the further development of the novel methodology.

2e. Link with legal authorities and the public

This work package deals with the outreach of the project. The use of indicators must facilitate communication on environmental issues to stakeholders and policy makers, by promoting information exchange. Routine monitoring, scientific evaluation, and feedback from managers could enable adaptive shifts in management strategies as knowledge about an ecological system grows, and at the same time will provide important opportunities for scientists to test hypotheses about ecosystem responses to environmental change. Substantial improvements in modeling capabilities are also needed to develop and deploy effective options to maintain and enhance the supply of critical goods and services and to evaluate alternative management options under changing environmental conditions.

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9. Appendix

The appendix of this report is delivered on a separated CD. The outline of the appendix on the CD is described below.

9.1 Species lists

The species lists of the different water bodies are grouped in a database, named species lists, and the database can be found on the CD. In the species list per water body is indicated which species are lumped, taxonomically changed and excluded.

e.g. The table structure is illustrated below:

Species list Dutch coast		
species	true name	exclusion
Abludomelita obtusata	Abludomelita obtusata	1

Species: is the name of the species as described in the original database

True name: is the name of the species after adopting the data truncation rules

Exclusion: a 1 means that the species is included in the analysis and a 0 means excluded

9.2 WFD – benthos data

In the database 'benthos database WFD report', all benthos data to do the reference and assessment analysis are grouped per water body. For each water body, four tables are included, which store the species info and metadata for the assessment and reference analysis. This table has to be included in the database 'database for WFD calculations' to calculate the WFD evaluation methods.

This data can only be used for controlling or recalculating the result within this report, because some of the data is exclusively ownership of NIOO or NIOZ. When the data will be used for other reasons, permission to NIOO or NIOZ has to be asked.

9.3 Reference output files and plots

In the database 'Reference randomisation output files' are the results of the permutation program (KRW-program) (percentiles and median) grouped per water body for the four parameters (BIO_OUT [biomass], DENS_OUT [density], SIMILS_OUT [similarity], NSP_OUT [Number of species]).

In the pdf-file ‘Reference plots’, all reference plots, obtained by permutation are visualised per water body, habitat and parameter.

9.4 KRW- program and BEQI- database

The KRW-program can be used to calculate the permutations for the reference settings. In the file KRW_main.f90, the source of the KRW-program is outlined. The input files can be generated from the ‘database for WFD calculations’ and exported via R (freely available software, but include on CD). With the database ‘database for WFD calculations’ it is possible not only to calculate level 3 of the BEQI method, but also the other international methods (m-AMBI, IQI, NQI and DKI). This database is basic, because it is not with a friendly user interface and everything has to be calculated via query-actions. In the appendix, a Dutch version of the guidelines to work with the database for WFD-calculations is included.