Mapping groundwater discharge and assessing the attenuation of groundwater nitrate at a site in the Wadden Sea

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In this paper the hydrogeological and geochemical aspects of the discharge of nitrate containing groundwater is presented for a study site in the northern Wadden Sea (Ho Bay, Denmark). Geophysical investigations using geo-electrical methods both onshore and offshore together with pore water sampling revealed a highly heterogeneous discharge of fresh groundwater out through the intertidal zone. The upper groundwater of the studied catchment contains nitrate concentrations up to 1 mM (mmol/L). However, the groundwater discharging into the sea is generally free of nitrate, except for a few local discharge zones with high nitrate concentrations. Denitrification apparently reduces the flux of nitrate from the coastal aquifer towards the marine environment. Geochemical analysis of flow paths in the groundwater aquifer indicates that nitrate reduction takes place within the groundwater aquifer and the upper surface sediment layer of the intertidal zone. In the groundwater zone nitrate is reduced by the oxidation of pyrite (FeS_2) and sedimentary organic matter, whereas in the intertidal zone sediments the oxidation of organic matter becomes increasingly important. The content of organic matter and pyrite in the sediment of the groundwater zone was found to be small, thus limiting the reduction capacity for nitrate in the groundwater zone. The future consequence of this may be an increase in the nitrate flux towards the marine environment.

Key words: Denitrification, geochemistry, geophysics, monitoring, nitrate, sub-marine groundwater discharge

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Introduction

Past studies of nutrient loadings to the coastal zone have primarily focused on the contribution from rivers and streams. The quantity of nutrients reaching the sea by direct groundwater discharge is poorly known (Conley et al. 2000, Burnett et al. 2003, Slomp & Van Cappellen 2004). While streams and rivers are easily identified and monitoring of their nutrient fluxes relatively straight forward, the identification of zones of direct groundwater discharge and their flux of nutrients is more challenging. For instance, is the direct groundwater discharge for a particular coastline occurring through a diffuse zone along the coast or is the majority of the discharge through discrete vents in the seafloor? Determining the mode of groundwater discharge and locating the major discharge zones thus becomes the first objective.

Geo-electrical methods measuring subsurface resistivity appears promising for mapping the sub-seafloor distribution of freshwater, due to the large resistivity contrast between seawater and freshwater. Geo-electrical techniques employing streamers towed after a boat has the potential for rapid mapping of large areas as opposed to point sampling of pore water (Zektzer et al. 1973, Lee 1985, Lavoie et al. 1988 and Vanek & Lee 1991). However, studies of the sub-seafloor distribution of freshwater saturated sediments utilising geo-electrical methods appear to be rare. In addition most surveys employed a single probe measuring the seawater conductivity or the resistivity of the uppermost layer of the sediment (Zektzer et al. 1973, Lee 1985, and Vanek & Lee 1991). Although the foundations for offshore resistivity measurements were laid decades ago (Zektzer et al. 1973, Lagabrielle & Teilhaud 1981 and Lagabrielle 1983), the application of the Underwater Multi-Electrode Profiling (UMEP) technique of this study to a marine environment, must therefore be considered novel.

The groundwater nitrate content has been increasing steadily over the past 30-40 yrs in northern Europe (Spalding & Exner 1993, Iversen et al. 1998). For coastal areas it is therefore reasonable to expect that the groundwater derived load of nitrate towards the marine environment will also increase. Given the high residence time of groundwater in aquifers (mostly 50-200 years) and because the coastal discharge zone is located at the end of the flow system, the full effect of discharging nitrate-rich groundwater into the marine environment may yet be years ahead. Furthermore, geochemical processes within the aquifer may effectively attenuate nutrients or altogether remove them from the groundwater, notably nitrate removal by denitrification (Postma et al. 1991, Tesoriero et al. 2000, Puckett et al. 2002). The discharge of nitrate free groundwater in the coastal zone may therefore well be a consequence of several factors: the groundwater age (e.g. groundwater infiltrated before the time of intensification of fertilizer application), spatial varying land use, natural nitrate reduction, or a combination hereof. Geochemical analysis of dissolved redox-species and dissolved gasses in the discharging groundwater may be used as a tool to unravel these questions (Postma et al. 1991, Blicher-Mathiesen et al. 1998 and Appelo & Postma 2005). Additionally it can be used to estimate the nitrate load, the degree of nitrate removal by denitrification and identify the electron donors responsible for the nitrate removal.

This paper describes some aspects of the geophysical mapping of groundwater discharge into the Wadden Sea and an assessment of the geochemical attenuation of nitrate in the coastal aquifer. Focus of the paper is on giving a broad overview of some of the qualitative aspects of the employed geophysical and geochemical methods. A more in-depth quantification of groundwater and nitrate fluxes will appear in Andersen et al. a) (in prep.). The results are part of the EC FP5 project NAME: Nitrate from Aquifers and influences on carbon cycling in Marine Ecosystems (NAME project website: http://name.er.dtu.dk). The field site was located in the northern Wadden Sea adjacent to Ho Bay, Denmark (Fig. 1).

Figure 1. Location of the study site (in the Wadden Sea).

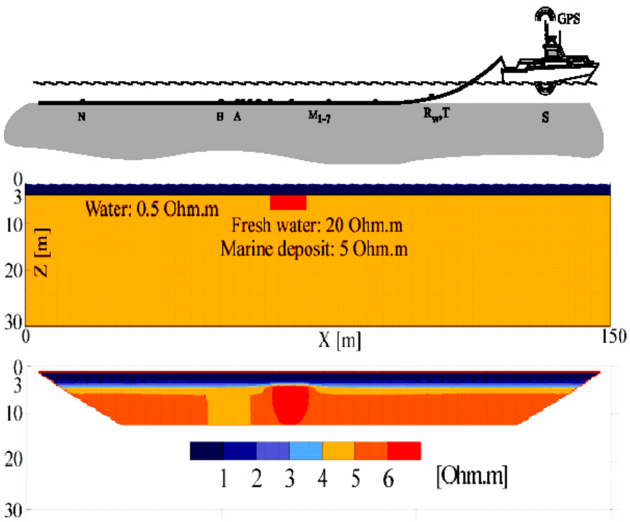

Figure 2. Top: UMEP methodology. On the towed array A and B are current emitting electrodes; N and M1-7 are potential electrodes; T and RW are temperature and seawater resistivity probes; and S are a depth sonar. The position is continuously recorded by GPS. Middle: A hypothetical seabed showing a freshwater body residing in a marine deposit covered by 3 m of seawater. Bottom: Theoretical resistivity response of the UMEP survey.

Materials and Methods

Underwater multi-electrode profiling (UMEP)

The UMEP method employs an array of electrodes towed along the seabed after a boat (Fig. 2). Electrical coupling to the seabed is ensured by the highly conductive seawater. The array is 110 m long with two current electrodes emitting current up to 10 A at 0.2 Hz. The resulting potential field is measured by eight potential electrodes arranged with one electrode 50 m away on one side of the current electrodes and the remaining 7 electrodes on the opposite side of the other current electrode with logarithmical increasing distances. Measurements were done every 1/10 of a second. Continuous measurements were made of the boat location (by GPS), the depth (by sonar), seawater conductivity and temperature.

The theoretical depth of penetration for a homogeneous half space depends of the array type and is about 1/5 to 1/4 of the spacing between the two most external electrodes i.e. a penetration depth of 22 to 28 m (Barker 1989). For a saline homogeneous half space below a seawater layer the depth of penetration reduces to about 1/15 of the total array length, giving a penetration depth of about 7 m. For measurements over freshwater saturated sediments the depth of penetration could increase to up to 25 m. However these simple rules do not apply to a heterogeneous resistivity distribution. Nevertheless the further apart the potential electrodes are from the current electrodes the deeper the measurement penetrates the seabed and the more sensitive the method is in detecting sediments containing freshwater. It is estimated that horizontal surface heterogeneities smaller than 1 m (smallest electrode separation) directly beneath the array can not be resolved. Furthermore the resolution decreases away from the array both vertically and laterally (Barker 1989).

The inversion of the UMEP data was done by a code developed specifically for the employed array configuration. Data was interpreted assuming a 3-layer model. Layer 1 is surface seawater with resistivity fixed by the conductivity probe measurements; layer 2 is surface sediment of varying depth and resistivity; and finally layer 3 is infinitely deep with varying resistivity. The inversion is done for each position along the array. To produce more continuous inverted results laterally along the array, the result of an inversion at a given position is used as the initial condition for the inversion process at the following position.

Multi-electrode profiling (MEP)

MEP was used to map the subsurface geology and on the beach and tidal flats additionally the distribution of fresh and saltwater in the subsurface. The method employs a long array of steel electrodes inserted into the ground with an electrode spacing varying from 1 to 5 m depending on the targeted measuring depth (see Dahlin 1996). An IRIS SYSCAL 48 instrument was used for the measurements. The measured apparent resistivities were inverted using the RES2DINV software (Loke & Barker 1996) to produce a calculated resistivity distribution of the subsurface.

Pore water sampling on the tidal flat

In the intertidal zone of the beach pore water samples were extracted from the sediment using a drive point technique, giving profiles of the upper 1.2 m of the pore water electrical conductivity (EC) and nitrate content. Steel pipes of 1.5 to 2 m in length (0.01 m outer diameter) were in one end fitted with a section of 0.05 m with holes and an outer mesh of polyethylene (PE). The end of the steel pipe was fitted by a pointed PE-tip. The steel pipes were driven into the sediment by a battery powered handheld drilling machine with percussion. At desired depth suction was applied to the pipe by a 60 ml syringe and a 3-way valve. The pipe was flushed with the equivalent of three pipe volumes. EC was measured in a small beaker with a WTW LF-196 conductivity meter and a WTW Tetracon 96 EC probe. The nitrate content was measured using analytical test strips (Merckoquant, range 10-500 mg/L).

Installation of monitoring wells

Permanent monitoring wells were installed at different depth along two transects parallel to the groundwater flow, from the upper beach and out into the intertidal zone. The wells were constructed of either a 0.025 or 0.032 m outer diameter PE pipe and at the bottom fitted with a single 0.12 m section screened with a PE mesh. A Geoprobe 54 DT drill-rig was used to install the well to depths of up to 10 m below the surface.

Groundwater sampling and analysis

From the permanent monitoring wells groundwater samples were extracted by a gas-lift technique using nitrogen gas (see Andersen et al. 2005 and Fetter 1993 for details). The wells were flushed three times and then sampled. A flow cell equipped with probes for O_2, pH and EC (electrical conductivity) was directly mounted on the sampling tube. Dissolved O_2 was measured using a WTW EO 196-1,5 electrode connected to a WTW OXI 196 Oximeter. EC was measured with a WTW Tetracon 96 EC probe and a WTW LF-196 conductivity meter. pH was measured with a WTW SenTix 41 electrode connected to a WTW 196 pH-meter. Samples for all other parameters were filtered through a 0.2 µm Satorius Minisart filter. Samples for the nitrate and sulphate content were frozen and later analysed using Ion Chromatography (HPLC) using a Vydac 3021IC column. Alkalinity was determined in the field by the Gran titration method (Stumm & Morgan 1981). Dissolved ferrous iron (Fe^{2+}) was also measured in the field by the spectrophotometric Ferrozine method (Stookey 1970). Dissolved inorganic carbon (DIC) was calculated on the basis of the measured alkalinity and pH.

Sampling of dissolved gasses

For the sampling of dissolved gasses (N_2, O_2, CO_2 and Ar) a cobber tube was lowered into the screened section of the monitoring wells and pumped, using a peristaltic pump, at a low rate to minimise the drawdown in the well. On the effluent side of the pump the groundwater was passed through a 'bubble-stripper' made of glass, where a gas bubble was equilibrated with the groundwater. After an equilibration period of minimum 20 min. the gas bubble was sampled through a septum using a glass-syringe and immediately analysed. One sub-sample was analysed for N_2, O_2 and Ar on a ML GC 82 gas chromatograph with a Shintzu C.B3A integrator, fitted with a 10 m Haye Sep A column and using helium as carrier gas. The column was cooled to about -20°C using dry ice for the effective separation of O_2 and Ar. CO_2 was determined on another sub-sample on a SRI 8610A gas chromatograph by Thermal Conductivity Detection.

Results and Discussion

Distribution of fresh groundwater at the coastline

The studied catchment at Ho bay, shown in Figure 3, is landwards limited by a groundwater divide located about 1400 m from the coast. No major streams are draining the catchment, so groundwater discharge at the shore face is the only mode for surplus groundwater to leave the aquifer. The aquifer consists of a buried valley cut into a thick Neogene clay deposit and filled with sandy sediments of Quaternary and Neogene age. Secondary buried channels connect the aquifer to the coast as seen on the southwest edge (lower left edge) of the 3D geological model in Figure 3.

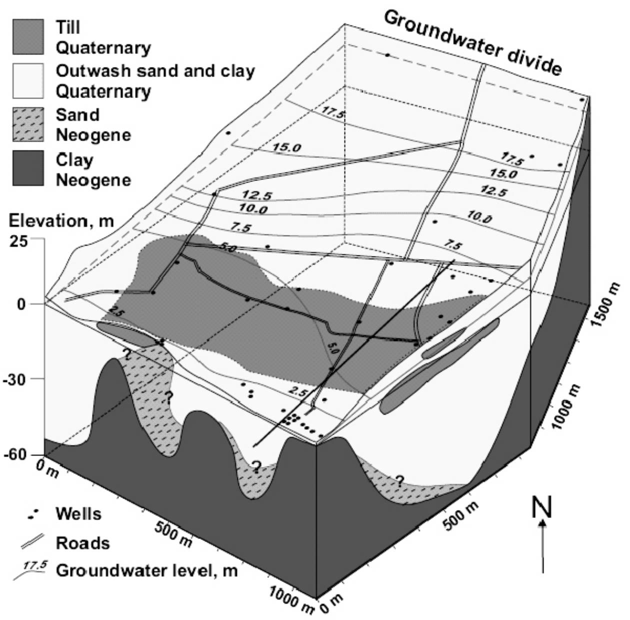

Figure 3. Geological 3D model for the study site. The southwest edge faces the Ho bay.

Figure 4 shows the calculated (or inverted) resistivity in the multi-electrode profile (MEP) done along the beach. The profile shows an upper zone of high resistivity separated by an undulating border from a deeper zone of low resistivity. The low resistivity (< 20 Ω-m) is, based on drillings, interpreted as the Neogene clay layer, but at some locations, it could also be caused by seawater saturated sandy sediments. The zones of high resistivity (30-500 Ω-m) are uniquely caused by freshwater saturated sediments and the secondary buried valleys are situated where these zones reaches deep into the profile (~40 m). Here the aquifer has a significant thickness of freshwater saturated sediments and thus a potentially high discharge of freshwater to the coast.

Figure 5a shows a map of the calculated resistivity of the second layer in the inverted UMEP-data. The map shows how the apparent resistivity varies offshore along the coast with the light grey areas representing a high resistivity, qualitatively indicating zones with freshwater residing in the seabed. These zones are indicated by arrows in Fig. 5a and correlate with the larger zones of high calculated resistivity from the MEP profile (indicated by arrows in Fig. 4).

The presence of freshwater in the seabed was verified by pore water sampling at low tide, measuring the electrical conductivity (EC) at 0.3 to 1.2 m below the sediment surface. This was done within the detailed study area of Figure 5b. Figure 6a show the pore water resistivity (the reciprocal of the EC values) at 0.3 m below the sediments surface with increasing resistivity to the south and landwards. The resistivity also increases with depth (not shown). Only a qualitative comparison of the pore water and UMEP resistivities can be made since the resistivity values derived from the pore water and the UMEP surveys are not directly comparable. This is partly because the pore water resistivity values do not include the effect of the sediment grains, and partly because the depth to the second layer of the UMEP interpretation is not well determined, as opposed to the depths of the pore water data. Despite this the distribution in the pore water resistivity largely confirms the freshwater distribution given by the UMEP (see Fig. 5b). The correlation between the two methods, although the values are not directly comparable, shows that the UMEP method is capable of detecting zones of freshwater saturated sediments within the seabed despite being done in the highly conductive seawater of the bay (resistivity ~ 0.3Ω-m).

The pore water measurements reveal a higher variability with local zones of low resistivity (Fig. 6a). These zones represent seawater present in the sediment near local layers and lenses of more clayey sediments. This shows that although the UMEP method gives a good overall idea of the distribution of freshwater, pore water sampling is necessary for accurately describing the actual freshwater discharge, mixing of sea- and freshwater and the associated chemical reactions.

Distribution of nitrate

The upper groundwater of the catchment generally contains nitrate in concentrations up to about 1 mM (mmol/L). In the groundwater discharging at the

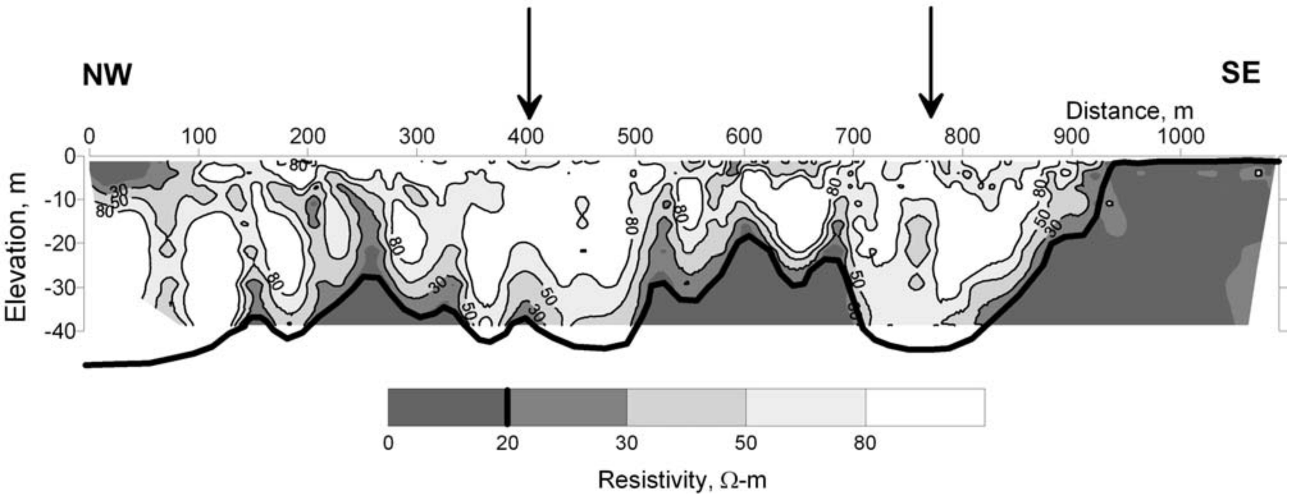

Figure 4. MEP profile along the beach of the study site showing the distribution of freshwater (30-500 Ω-m). Black line is an interpretation of the aquifer bottom and the arrows indicate the major zones of groundwater discharge.

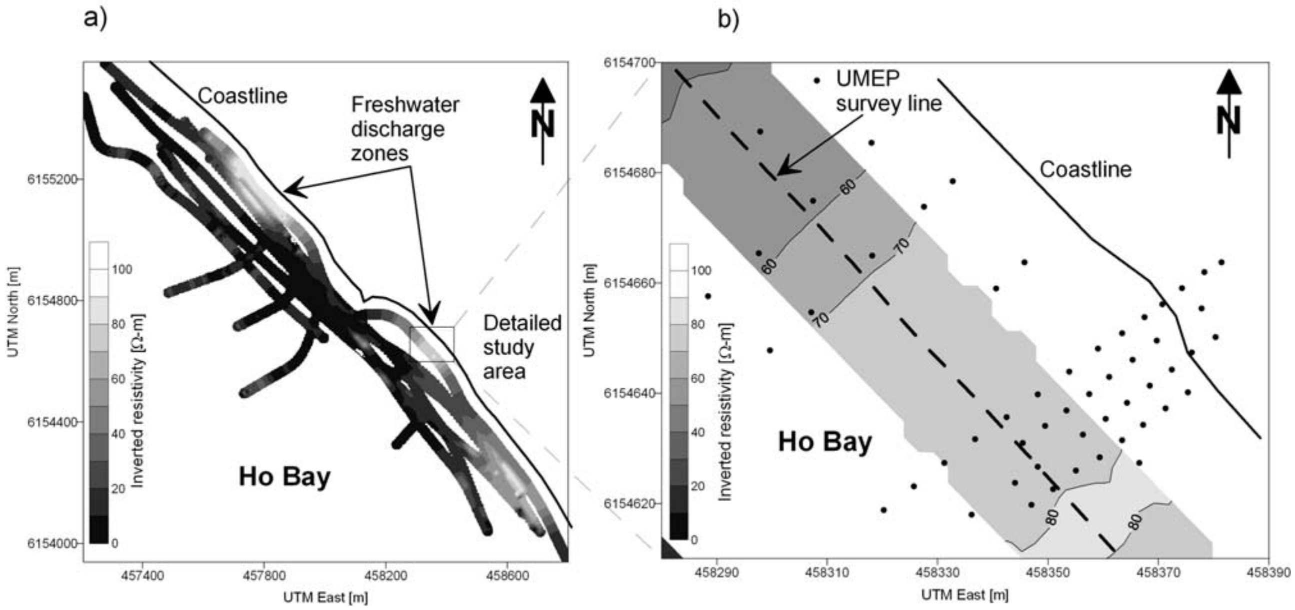

Figure 5. a) UMEP survey lines showing the contoured calculated resistivity (Ω-m) of the second layer of the inverted UMEP data. b) Detailed study area showing contoured calculated resistivity (Ω-m) of one UMEP survey line. Dots are locations for sampling of sediment pore water.

coastline nitrate was only detected at a few locations along the high tide line and only at a few of these; nitrate was found to reach into the intertidal zone. Such a zone is depicted in Fig. 6b which is the same zone as shown for the EC measurements of Fig. 6a. The nitrate concentration at 0.3 m below the sediment surface, reveals a plume of nitrate (up to 0.7 mM) reaching more than 30 m out into the intertidal zone (Fig. 6b). Just 10 meters to the north, nitrate containing groundwater is not emerging in the intertidal zone. A possible explanation for the lack of nitrate in the discharging freshwater of the intertidal zone may be the reduction of nitrate mediated by micro-organisms (denitrification). A way to investigate this is to study the geochemical evolution along a groundwater flow path.

Geochemical evolution along a flow path

Figure 7 shows the water chemistry in a 2D vertical transect (see location Fig. 6b) parallel to the groundwater flow, 10 m north of the nitrate plume of Fig. 6b. The groundwater flow direction was, inferred from head measurements, predominantly horizontal towards the sea with an upward component nearer to the coastline (indicated by the flow path arrow in Fig. 7a). A groundwater travel time of about 1 yr from 0 to 20 m in Fig. 7 was estimated for the flow path based on Darcy's law (Andersen et al. a) in prep.).

Along the flow path nitrate decreases from a maximum of 1.4 mM upstream to 0 mM downstream, near the high tide line (Fig. 7a). Concurrently an increase is seen in both sulphate (Fig. 7b), dissolved inorganic carbon (DIC) (Fig. 7c) and dis-

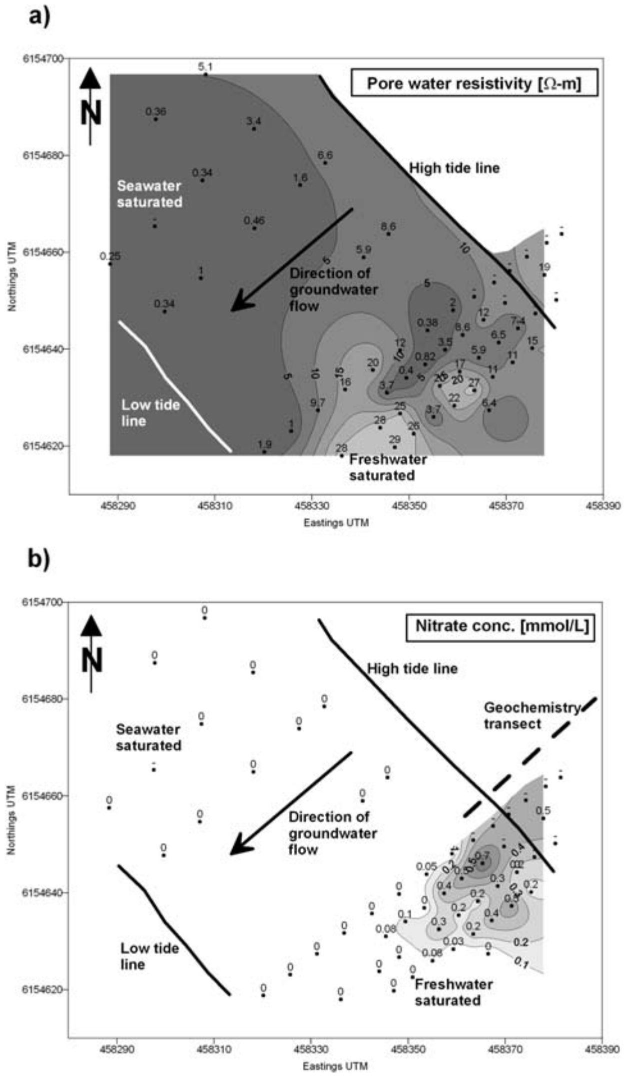

Figure 6. a) Pore water resistivity (Ω-m) based on the electrical conductivity of the pore water samples and b) nitrate distribution on the tidal flat. The dashed line in b) represents a transect of monitoring wells. Both plots cover the same area as in Fig. 5b.

solved ferrous iron (Fe^{2+}) (Fig. 7d). The increase in sulphate and Fe^{2+} indicate that oxidation of pyrite (FeS_2) may play a role in reducing the nitrate according to:

(1)

$$5FeS_2 + 14NO_3^- + 4H^+ \rightarrow 7N_2 + 5Fe^{2+} + 10SO_4^{2-} + 2H_2O$$

In reaction (1) the Fe^{2+} may be further oxidised to Fe^{3+} and precipitate as $Fe(OH)_3$. The increase in DIC (Fig. 7c) indicates that organic matter (CH_2O) also contributes to the reduction of nitrate according to:

(2)

$$5CH_2O + 4NO_3^- \rightarrow 2N_2 + 4HCO_3^- + CO_2 + 3H_2O$$

Both reactions (1) and (2) predict that nitrate is transformed into dinitrogen (N_2). The net nitrate removal by reactions (1) and (2) can uniquely be verified by measuring excess dissolved N_2 (Vogel et al. 1981). Because the background concentration of dissolved atmospheric N_2 may vary due to different processes during groundwater recharge (temperature, excess air etc. (Heaton & Vogel 1981)) argon (Ar) was used as an inert tracer.

Figure 7e shows the measured N_2/Ar-ratio in the transect. In the upstream nitrate rich zone the N_2/Ar-ratio varies between 70 and 90, roughly corresponding to the theoretical N_2/Ar-ratio of 81.8 in water equilibrated with the atmosphere at 8°C. Seawards, into the zone depleted in nitrate, the N_2/Ar-ratio increases to a maximum around 175. This supports that nitrate reduction releasing N_2 is taking place. From the measured N_2/Ar ratio the amount of excess N_2 was calculated using the method suggested by Blicher-Mathiesen et al. (1998). The N_2/Ar data suggests that the amount of nitrate reduced is up to 0.9 mM (Fig. 7f). This roughly equals the nitrate concentration measured in the upstream part of the transect (Fig. 7a). If the 0.9 mM NO_3^- were solely reduced by pyrite it should release 0.64 mM SO_4^{2-}, an increase somewhat higher than the maximum observed increase of 0.5 mM. It should be noted that this rough stoichiometric balance approach does not consider temporal variations in the nitrate concentration along the flow path or the diffusion of nitrate into the deeper flow paths. This may well explain the minor chemical imbalances between the up and downstream parts of the flow path.

To quantify the relative importance of pyrite and sedimentary organic carbon as electron donors in the reduction of nitrate, an electron balance (Postma et al. 1991) was set up along the flow path of Fig. 7a. In the electron balance the number of electrons that participate in a given redox-reaction is multiplied with the aqueous concentration of the relevant reactants or products (Table 1). The electron equivalents obtained in this way can be plotted cumulatively as in Figure 8. Sulphate was corrected for the sea-salt contribution, whereas the DIC was corrected for calcite dissolution. The DIC correction was done by assuming that excess Ca^{2+} (compared to the sea-salt contribution of Ca^{2+}) must largely come from dissolution of carbonate minerals, releasing an equivalent amount of DIC on a molar basis. The sum of electron acceptors, O_2 and NO_3^- in the upstream part of the flow path is roughly matching the downstream increase in the electron donors represented by sulphate and DIC. Pyrite appears to account for about 40% of the increase in the sum of sulphate and DIC whereas organic carbon is responsible for the remaining 60% (Fig. 8).

This raises the question of electron donor availability and the ability of the aquifer to buffer the nitrate load. The sedimentary organic carbon (TOC) and pyrite content was measured on sediment samples along the transect (Andersen et al. b) in prep.).

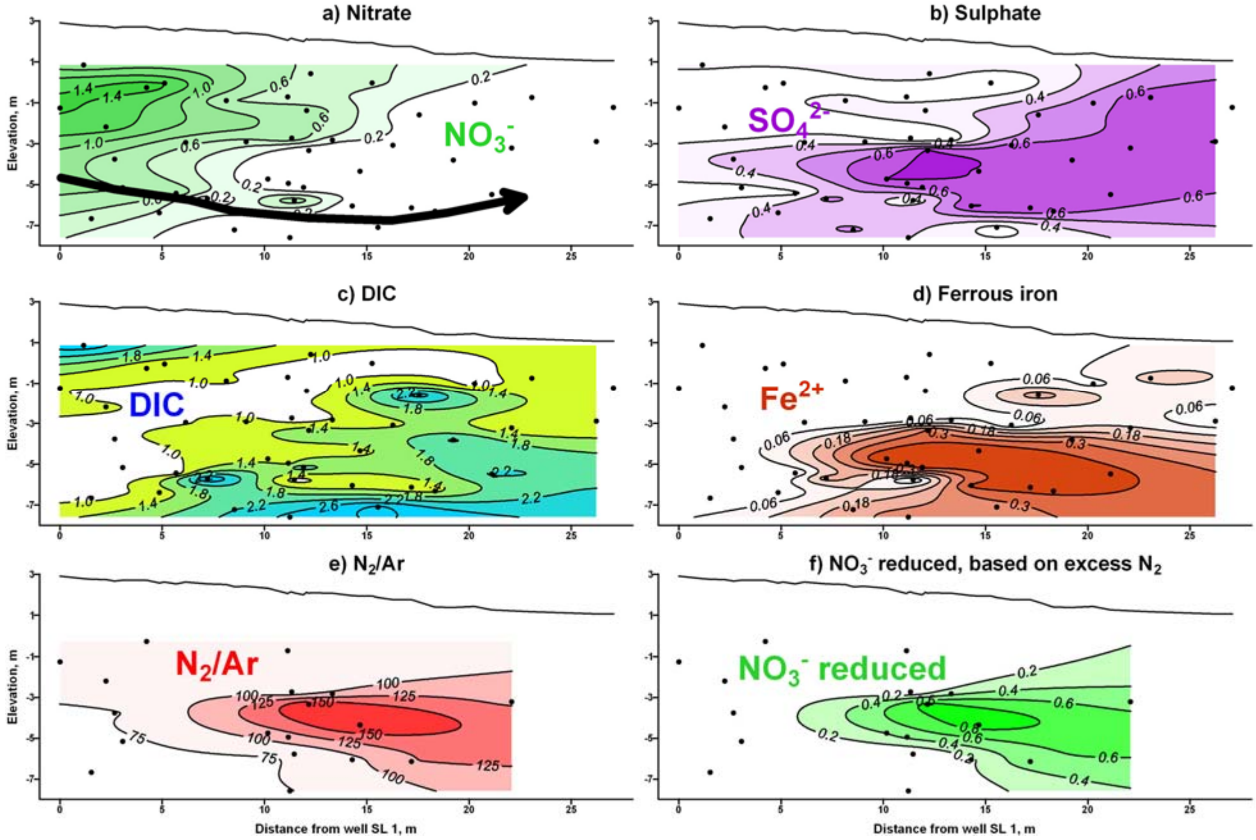

Figure 7. Vertical profiles along a transect showing distributions of a) nitrate (mM), b) sulphate (mM) corrected for sea-salt contribution, c) DIC (mM) corrected for sea-salt contribution and calcite dissolution, d) Ferrous iron (mM), e) the ratio of dissolved N_2/Ar, and f) the amount of nitrate reduced based on excess N_2 (mM). The dots are sampling points and the arrow in a) indicate a groundwater flow path.

Table 1. Principles for the electron balance calculations (from Postma et al. 1991).

Reaction	Transferred electrons	Electron equivalents
$S_{FeS2} \rightarrow SO_4^{2-}$	$-7e^-$	$7\cdot[SO_4^{2-}]$
$CH_2O \rightarrow CO_2$	$-4e^-$	$7\cdot[DIC]$
$Fe_{FeS2} \rightarrow FeOOH$	$-1e^-$	$½\cdot([SO_4^{2-}] - 2\cdot[Fe^{2+}])$
$NO_3^- \rightarrow ½N_2$	$+5e^-$	$5\cdot[NO_3^-]$
$O_2 \rightarrow 2O^{2-}$	$+4e^-$	$4\cdot[O_2]$

The TOC was found to be evenly distributed, averaging 12.5 mmol C/kg (or 47 mmol C/L pore water), but the reactive fraction of this may well be much smaller (the remaining part being in a recalcitrant form, not readily available for the microorganisms). In light of the apparent importance of pyrite as an electron donor, it is surprising that the pyrite content is small ranging between 0.5-2.5 mmol S/kg (or 2-10 mmol S/L pore water). This shows that, at least in the studied portion of the groundwater aquifer, the electron donor pool is limited and can be expected to be exhausted.

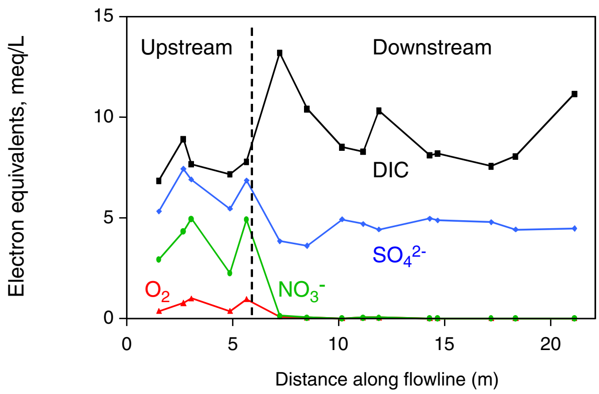

Figure 8. Electron equivalents plotted cumulatively along the flow path in Fig. 7a. The electron equivalents are calculated according to the scheme given in Table 1.

However, in the recent upper sediments (< 1 m) of the intertidal zone the TOC content is orders of magnitudes higher and appears to be more reactive than in the aquifer sediments and here the oxidation of organic matter becomes increasingly important in reducing the groundwater nitrate (Lavik et al. in prep.). This recent sediment layer may possibly retain a renewable reduction potential towards NO_3^- through exchange with the overlying seawater column.

Due to the very heterogeneous conditions at the study site, both in terms of the freshwater discharge and the nitrate distribution, a quantification of the groundwater nitrate flux on the regional scale of the Ho bay is highly uncertain based on these limited data. This said, a "back of the envelope" calculation indicates that the groundwater nitrate flux is probably orders of magnitude lower than the nitrate flux from the nearby river Varde Å.

Conclusions

The UMEP method was found to be fast and efficient for mapping the distribution of freshwater in the seabed of the intertidal zone. The method revealed a very heterogeneous discharge of fresh groundwater out through the intertidal zone.

The flux of nitrate towards the marine environment was likewise highly variable even over short distances along the coast. Although the upper groundwater of the catchment area is generally rich in nitrate, the direct discharge of nitrate rich groundwater occurred only at a few locations. The evolution in geochemistry along sampled flow paths indicates that currently nitrate is attenuated by denitrification before it reaches the marine environment and that pyrite and sedimentary organic matter are responsible for the nitrate reduction. However, the limited amount of pyrite and reactive organic matter in the aquifer sediments, and the fact that nitrate rich groundwater actually does discharge in some locations indicate that the reductive capacity of the aquifer sediments is limited. In the future the groundwater flux of nitrate towards the tidal zone sediments can be expected to increase, if agricultural practices are not changed. A general assessment of groundwater nitrate loads discharging to the coastal zone clearly needs careful consideration of the local variability in geology, hydrogeology and geochemistry.

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Implementation strategy for monitoring of new hazardous substances

Susanne Boutrup

Boutrup, S. 2006: Implementation strategy for monitoring of new hazardous substances. In: Monitoring and Assessment in the Wadden Sea. Proceedings from the 11. Scientific Wadden Sea Symposium, Esbjerg, Denmark, 4.-8. April, 2005 (Laursen, K. Ed.). NERI Technical Report No. 573, pp. 83-87.

Monitoring of hazardous substances is included in TMAP (Trilateral Monitoring and Assessment Program), in the Danish Nationwide Monitoring and Assessment Programme NOVANA as wells as in the Water Framework Directive. Monitoring of hazardous substances in groundwater started in Denmark about ten years ago and was included in monitoring of point sources, air, fresh surface water and marine areas in 1998. The Danish approach in the future is only to include new hazardous substances in the monitoring programme if it has been documented to be relevant. A scheme for the documentation is set up including considerations about analytical methods and preliminary investigations. The preliminary considerations are primarily based on literature. The need for implementation of the list of priority substances in the Water Framework Directive (WFD) in the Danish part of TMAP is discussed in the current paper. The strategy is used as starting point for that discussion.

Key words: hazardous substances, strategy for implementing new substances, preliminary considerations

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Introduction

Monitoring of hazardous substances started up in Denmark about ten years ago with monitoring of pesticides in groundwater. In the previous Danish Environmental Monitoring Programme NOVA-2003 launched in 1998 monitoring of hazardous substances was included as a part of the programme (Danish Environmental Protection Agency, 2000). Monitoring of hazardous substances was included in monitoring of point sources including wastewater and sludge, fresh surface water in addition with sediment, biota and seawater from marine areas as well as continuation of monitoring in groundwater. In addition, heavy metals have been included in air monitoring since 1989.

NOVA-2003 was in 2004 followed by a revised National Monitoring and Assessment Programme for the Aquatic and Terrestrial Environments (NOVANA) (National Environmental Research Institute, 2005). The objectives of NOVANA are to:

- describe sources of pollution and other pressures and their impact on the status of the aquatic and terrestrial environments and identify trends
- generally document the effect of national action plans and measures directed at the aquatic and terrestrial environments – including whether the objectives are achieved and whether the trends are in the desired direction
- meet Denmark's obligations in relation to EU legislation, international conventions and national legislation
- contribute to enhancing the scientific basis for future international measures, national action plans, regional management and other measures to improve the aquatic and terrestrial environments, including contributing to develop various tools.

In 1998 it was not possible to measure a number of the hazardous substances included in NOVA-2003, since analytical methods could not meet the demands for detection limits, analytical uncertainty etc. It meant that some analyses could not be performed in the beginning of the programme period. Another experience from NOVA-2003 was that a number of substances were not detected at all with the used methods and detection limits. The experiences from monitoring of hazardous substances in NOVA-2003 resulted in the conclusion that new hazardous substances would only be included in the subsequent monitoring programme NOVANA if it had been documented to be relevant and within analytical range.

Method

The Danish implementation strategy for monitoring of new hazardous substances can be divided into three main topics: preliminary considerations based on literature, availability of analytical methods and preliminary investigations (fig. 1).

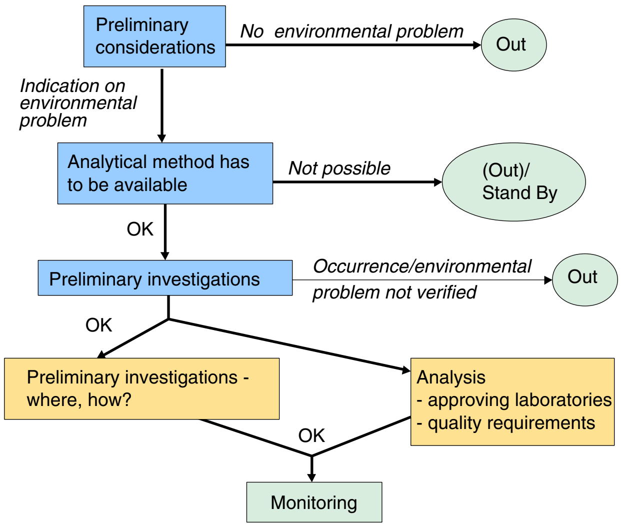

Figure 1. Stepwise approach of the Danish strategy of selecting hazardous substances for monitoring.

Preliminary considerations

In the preliminary considerations information is collected on use and chemical / physical data. Information about solubility and lipophility, persistency and tendency to bioaccumulate are especially important. In addition, any data on occurrence in the environment either within the country or in countries with comparable consumption and environmental conditions are included. This combined information can give a first indication on the presence of the substance in the environment and where it might be found. Besides, when the information is linked to knowledge about toxicology we might get indication on environmental effect.

If the conclusion of the preliminary considerations is that it is likely to find the considered substance in concentrations that might give effects in the environment, the next step has to be taken. It must be considered whether it is sufficient to focus separately on the substance, or if substances with similar characteristics or consumption pattern have to be included. This might be metabolites or substances within the same chemical group, e.g. other phthalates when DEHP is included. Alkylphenols is an example where the metabolic product, for example nonylphenol, nonylphenol-mono-ethoxylate and nonylphenol-di-ethoxylate should be expected to found in higher concentrations than the mother product nonylphenolpolyethoxylates (Danish Environmental Protection Agency, 1995).

Finally, the relevant concentration level must be considered to set up demands for analytical quality. If quality criteria exist, one tenth of those is normally used as demand for detection level. But if no quality criteria exist information about toxicology and consumption have to be used as normative.

Analytical methods

It is essential that analytical methods which meet the demands for detection levels and other quality criteria are available. Normally the analytical work in monitoring programmes is done by accredited laboratories, but since we are talking about analyses of new hazardous substances the analyses are normally not done routinely. For that reason laboratories accredited to the analysis might not exist. It means that it is essential to be even more careful when the demands for analytical quality and demands for documentation of the analytical quality are set up.

It is important that the demanded detection limit is below the concluded relevant concentration level. If the demanded detection level cannot be met we have to reconsider if it is relevant to analyze at all. Analyzing with too high detection limits or unreliable quality is waste of time and money.

As the preliminary considerations could result in inclusion of some extra substances, it is worthwhile considering whether the analytical method provides the opportunity to include additional substances with the same procedure, which renders the total set of analyses to be more easy and cheap. It is of course essential to bear in mind that data handling etc. requires resources.

Preliminary investigations

If the preliminary considerations end up with the conclusion that it might be relevant to include a new substance in the monitoring programme and that a suitable analytical method is available, the next step is to document that inclusion is relevant. We look more closely into that evaluation by a screening. The screening should provide answers to the questions:

- Does the substance occur in the environment?
- If so, in which matrices does it occur?

To get the right answers the screening strategy has to be considered. Included in this is knowledge about the transport of the substance in the ecosystem as basis for considerations about sample matrices and sample locations. It is essential to include the matrices and locations, which are closest to the sources in order to get a positive reply on the first question. If the substance is not found close to the source, it might not be found at all. In addition matrices which are in different ecological distances from the source should be included in order to answer the second question. As an optimum the matrix, which are exactly so far away that the substance doesn't occur, should be included.

Furthermore, the considerations about the strategy also include considerations about sampling strategy. It is essential that the samples are as representative and reproducible as possible. From some matrixes the samples should be composite samples, e.g. samples of sludge, while from other matrixes the samples should be one spot sample, e.g. surface water samples. Wastewater samples should be flow or time proportional, implying that the substance in focus is not volatile.

The number of samples in the screening depends on how homogeneous the substance can be expected distributed. It is necessary to get enough results to be able to make reliable conclusions. Finally a possible seasonal effect has to be considered. Does the sampling time of the year have any influence on the results?

The preliminary investigations may end up with results on which it is concluded that it is not likely to find the substance in the environment in concentrations which give rise to effect, are in conflict with objectives or which exceed the quality criteria. In that case the substance would not be recommended to the monitoring programme. Alternatively, if recommended the substance would have to be included in the monitoring programme. Before the substance is included in the routinely run monitoring programme decisions about matrix, frequency, demands for analytical quality, selection of sampling stations etc. similar to the considerations done in relation to planning a screening have to be taken.

Using the strategy in TMAP revision

The Danish approach to implementation of new hazardous substances in the monitoring will be used in the process of implementing the Danish obligation according to TMAP as well as the Water Framework Directive.

The current list of priority substances in the Water Framework Directive consists of 33 substances (EU, 2001). 10 of these substances are included in the current TMAP monitoring. 14 of the remaining 23 substances are included in NOVANA, 10 in the marine sub-programme and 4 in other sub-programmes. This leaves 9 substances, which are included in neither TMAP nor NOVANA (fig. 2). Among these are 5 substances, which were included in NOVA-2003 resulting in the conclusion that further monitoring is not relevant. The final conclusion is according to the figure below that we don't have sufficient knowledge of 4 of the substances on the Water Framework Directive list of priority substances. In addition to that there were 4 substances of which we don't have knowledge concerning

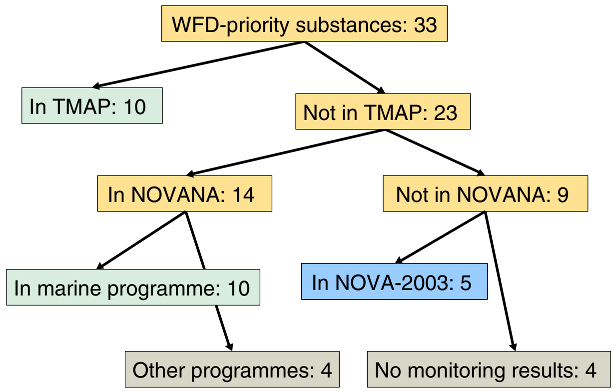

Figure 2. Number of substances on the Water Framework Directive list of priority substances in the TMAP-programme and in the Danish monitoring programmes NOVA-2003 and NOVANA.

occurrence in marine areas. Before implementation of these substances – or any other new hazardous substance - it should be documented that the implementation is relevant. The individual substances in each group are listed in Appendix 1.

Recommendation

When a programme including monitoring of hazardous substances is going to be revised it should be considered whether the monitoring of the current substances should be continued in the revised programme as well as if it is relevant to implement new hazardous substances. Exclusion of some "old" substances for which monitoring is not relevant any longer or for which the frequency could be reduced could give space for new activities. The revision and subsequent monitoring should be done according to the principle "need to know" not "nice to know".

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Appendix 1. Hazardous substances on the Water Framework Directive List of Priority Substances, TMAP (Trilateral Monitoring and Assessment Program), NOVANA (National Monitoring and Assessment Programme for the Aquatic and Terrestrial Environment) and NOVA-2003 (National Monitoring and Assessment Programme for the Aquatic Environment).

CAS nr.		**WFD**	**TMAP**	**NOVANA**	**NOVA-2003**
Metals					
7440-43-9	Cadmium and its compounds	x	x	ps,ms,mb	
7439-92-1	Lead and its compounds	x	x	ps,ms,mb	
7439-97-6	Mercury and its compounds	x	x	ps,ms,mb	
7440-02-0	Nickel and its compounds	x	x	ps,ms,mb	
Pesticides					
15972-60-8	Alachlor	x			
1912-24-9	Atrazine	x		mw	fw, gw
470-90-6	Chlorfenvinphos	x			
2921-88-2	Chlorpyrifos	x			
330-54-1	Diuron	x		mw	
115-29-7	Endosulfan	x			fw
608-73-1	Hexachlorocyclohexane	x	x	ms,mb	
34123-59-6	Isoproturon	x		gw,fw	
122-34-9	Simazine	x		mw, fw,gw	
1582-09-8	Trifluralin	x			fw
Alifatic hydrocarbons					
85535-84-8	C10-13-chloroalkanes	x			
Aromatic hydrocarbons					
71-43-2	Benzene	x		ps	
91-20-3	Naphthalene	x		ps,ms,mb	
Halogenated alifatic hydrocarbons					
107-06-2	1,2-Dichloroethane	x			ps,fw,mw
75-09-2	Dichloromethane	x		ps	
87-68-3	Hexachlorobutadiene	x		ps,ms	
67-66-3	Trichloromethane (Chloroform)	x		ps	
Halogenated aromatic hydrocarbons					
118-74-1	Hexachlorobenzene	x	x	ms,mb	
608-93-5	Pentachlorobenzene	x		ps, ms	
12002-48-1	Trichlorobenzenes	x		ms	
Polyaromatic hydrocarbons					
120-12-7	Anthracene	x	x	ps,ms,mb	
206-44-0	Fluoroanthene	x	x	ps,ms,mb	
n.a.	Polyaromatic hydrocarbons	x	x	ps,ms,mb	

Table is continued on the next page

CAS nr.		WFD	TMAP	NOVANA	NOVA-2003
Phthalates (softeners)					
117-81-7	Di(2-ethylhexyl)phthalate (DEHP)	x		ps,ms,mb	
Alkylphenols (nonionic detergents)					
25154-52-3	Nonylphenols	x	x	ps,ms	
1806-26-4	Octylphenols	x			fw
Brominated flameretardents					
n.a.	Brominated diphenylether	x		ps,ms,mb	
Chlorophenols					
87-86-5	Pentachlorophenol	x		ps, gw, fw	fw,gw
Organotin compounds					
688-73-3	Tributyltin compounds	x		ms,mb	

fw: freshwater, gw: ground water, mw: marine water, ms: sediment, mb: biota, ps: point sources

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Macrophytes in the western Wadden Sea: monitoring, invasion, transplantations, dynamics and European policy

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van Katwijk, M.M., Geerling, G.W., Rašín, R., van 't Veer, R., Bos, A.R., Hermus, D.C.R., van Wieringen, M., Jager, Z., Groeneweg, A., Erftemeijer, P.L.A., van der Heide, T. & de Jong, D. J. 2006: Macrophytes in the western Wadden Sea: monitoring, invasion, transplantations, dynamics and European policy. In: Monitoring and Assessment in the Wadden Sea. Proceedings from the 11. Scientific Wadden Sea Symposium, Esbjerg, Denmark, 4.-8. April, 2005 (Laursen, K. Ed.). NERI Technical Report No. 573, pp. 89-98.

Historic surveys of seagrass beds in the Dutch Wadden Sea were made in 1869, 1931 and 1972/1973. Annual quantitative analysis during 1995-2004 of seagrass-monitorings showed that the beds are highly dynamic. In the Balgzand area (western Wadden Sea), a dominance of eelgrass *(Zostera marina)* was recorded in the 1930s, followed by a dominance of dwarf eelgrass (*Zostera noltii*) in the 1970s. At present, the area is dominated by low densities of widgeon grass *(Ruppia maritima)*, that has invaded the area in 2002 approximately. This sequence of macrophytes might be correlated to increasing soil level due to sedimentation (GIS-analysis of monitoring in 1930s, 1970s and 2000s), but a changed salinity regime may also have been of influence. Near the seaward edge of the *Ruppia* bed, reintroduced dwarf eelgrass (planted in 1993) and eelgrass (planted in 1999, 2003 and 2004) lead a vulnerable existence. The highly variable survival rates underline the importance of spreading of risks of reintroduction programmes, both in time and space. This spreading of risks is also a general population strategy of *Zostera*, and the resulting high population dynamics imply that a large buffer zone around the beds should be protected to allow for new colonisations. This is recommended to be included in EU directives.

Key words: Invasions, monitoring, policy, population dynamics, seagrass, trend analysis, water plants

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Introduction

Seagrass has played an important role in The Netherlands. Until the early twentieth century, hundreds of families earned a living from the collection and harvesting of the robust form of eelgrass that grew around Low Tide level (LT) or deeper. It was used as isolation and filling material, and until the eighteenth century also for dike construction. Due to its former economic importance, historic maps of seagrass distribution from 1869 and 1931 are available (Oudemans et al. 1870, Reigersman et al. 1939). In the early 1930s, the robust form of eelgrass disappeared from the Wadden Sea. This was attributed to a seagrass disease, the closure of the Zuiderzee, two subsequent years of sunshine deficit or a combination of those three (Giesen et al. 1990a,b, den Hartog 1996). In the early 1970s, den Hartog & Polderman (1975) inventoried intertidal seagrass beds in the

Dutch Wadden Sea (dwarf eelgrass and the remaining flexible form of eelgrass). From 1995 onwards (and incidentally in previous years), seagrass beds are monitored on a yearly basis in the Dutch Wadden Sea by the Department of Public Works within the framework of the biological monitoring program (www.zeegras.nl). In 2002, widgeon grass colonised an area of more than 200 ha in the western Wadden Sea, in low densities (less than 1% cover). In this area, seagrasses got extinct in the mid 1970s. Since water clarity improved at the end of the 1980s, possibilities for reintroduction were investigated and transplantations were carried out in 1993, 1998, 2003 and 2004.

In this paper we will relate water plant distribution to location depth (tidal height) using maps from the 1920s onward, to gain insight in the depth distribution of the species eelgrass (two forms), dwarf eelgrass and widgeon grass in the western Dutch Wadden Sea (Balgzand). Secondly, we will analyse the dynamics of natural populations, and thirdly, we will summarise the transplantation results. This will lead to a number of policy recommendations.

Water plants at Balgzand 1931-2002

At Balgzand, in the westernmost part of the Wadden Sea, vegetation was mapped in 1931 by both Reigersman et al. (1939) by boat, and by Harmsen (1936) by foot, presumably also in 1931, or in 1932, the paper is not clear about this. In 1972, den Hartog and Polderman (1975) mapped the area, and in 2002 it was mapped again by van 't Veer, after the invasion of a new species in the area, widgeon grass (Figures 1 and 2).

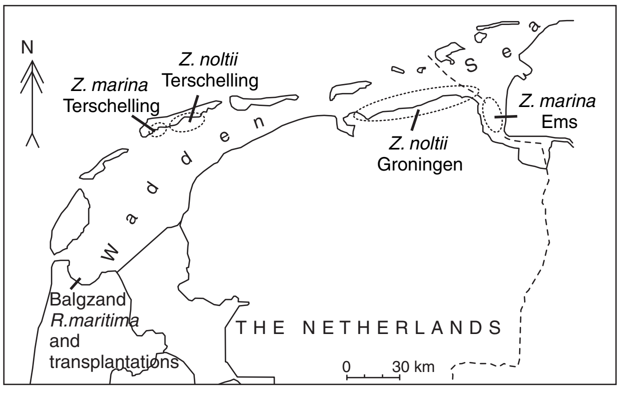

Figure 1. Map showing the Dutch Wadden Sea with present locations of seagrass beds.

The maps were digitised by ArcMap 8.2. These distribution maps were related to tidal depth maps from the Ministry of Transport, Public Works and Water Management. These maps were made by sounding from a boat, and have an accuracy of ± 0.10 m. Tidal depth data were used for the period 1926–1934, 1971–1974, and 1997-2002. Tidal depth maps were converted from ASCII to grid using ArcToolbox 8.2, and subsequently to feature by a spatial analyst (ArcMap 8.2). Grids were 20x20 m.

In the 1930s, in the Wadden Sea, but also in the Thames estuary, the following zonation of *Zostera* species was encountered: in the highest (shallowest) zone dwarf eelgrass occurs, followed by a zone of the flexible form of eelgrass, an un-vegetated zone and a zone of the robust form of eelgrass (Wohlenberg 1935, Harmsen 1936, van Katwijk et al. 2000). In the Balgzand area, Reigersman et al. (1939) and Harmsen (1936) only mapped the eelgrass beds (Fig. 2a and b). Note the difference in areas mapped in 1931/2 by Reigersman et al. (1939) and by Harmsen (1936). The difference is probably due to the different aims and methods: Reigersman et al. had an economical interest, i.e., only in the robust type eelgrass growing around LT and deeper, and mapped the area from a boat; Harmsen (1936) had a botanical interest, and mapped the area by foot and omitted water covered areas (see Fig. 3).

In the 1970s both species of seagrass were mapped (Polderman & den Hartog 1975), dwarf eelgrass appeared to have been slightly dominant over eelgrass. In 2000, for the first time, a few widgeon grass patches had been discovered at Balgzand by Rob Dekker (personal communication), who frequents this area at least yearly since mid-1990's. In 2002 and 2004, 225 and 264 ha of widgeon grass were recorded, respectively (Groeneweg 2004a). Densities were less than 1%. The sequence of water plant species was correlated to tidal depth (Fig. 3).

During this period, the investigated area silted up due to sedimentation, resulting in decreased tidal depths (Table 1). The optimal depth ranges of the water plants in the Dutch Wadden Sea and particularly Balgzand are listed in Table 2, and visualised in Figure 3. Most of the seagrass beds mapped in the western Wadden Sea in 1931 were located subtidally with an optimum depth of around 1 m below MSL (Mean Sea Level) or 0.4 m below LT (Table 2 and 3).

This corresponds with recordings of Feekes (1936 in de Jonge & de Jong 1992). Ninety percent of the seagrass beds were located subtidally. This contrasts with the 44% that de Jonge & Ruiter (1996) calculated on the basis of nautical maps. Perhaps this difference is due to the unavailability of the detailed bathymetric maps at the time of de Jonge and Ruiter's study. Of interest is the higher optimum of the seagrass beds at Balgzand in comparison to the total Wadden Sea, MSL -0.7 versus -1.0 m, respectively (Table 2 and 3). When related to low tide level, the difference is less: LT -0.20 and -0.40 m, respectively. The zone with maximum cumulative wave dynamics roughly corresponds with these depths (van Katwijk & Hermus 2000). Further analysis of the maps in relation to exposure to waves and currents, and in relation to

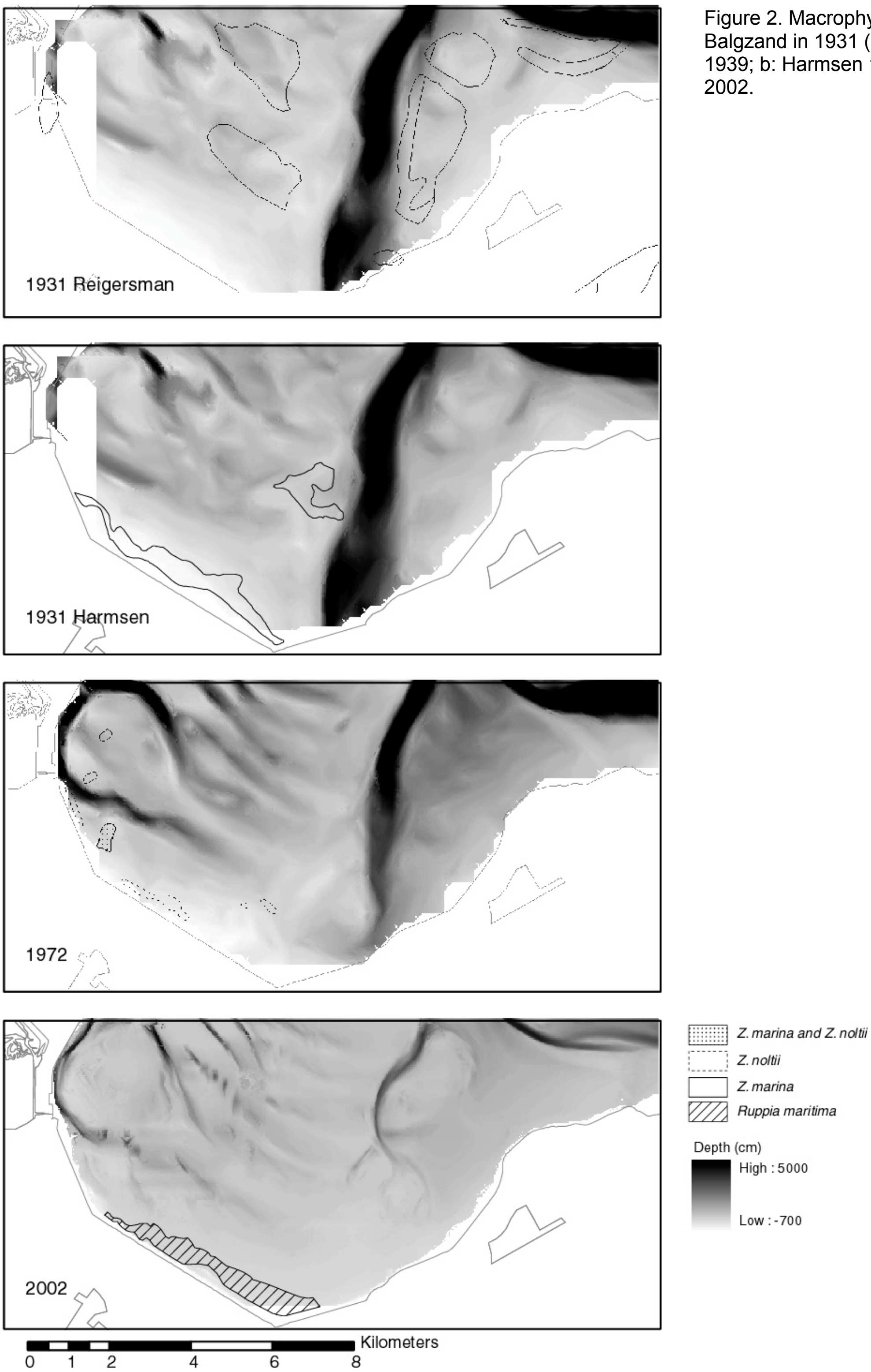

Figure 2. Macrophyte distribution at Balgzand in 1931 (a: Reigersman et al. 1939; b: Harmsen 1936); c: 1972 and d: 2002.

available substratum at each depth level, could offer explanations for the depth distribution of the seagrass beds in the 1930s.

The unvegetated zone that was found during the 1930s at several locations in the Wadden Sea, but also in the Thames estuary (Harmsen 1936, van Katwijk et al. 2000), appeared to have been located between MSL –0.25 and –0.4 m in the Balgzand area. This depth range of the un-vegetated zone as derived from the GIS analysis of seagrass and bathymetric maps, is consistent with field observations noted in literature, i.e. circa -0.20 below MSL and one or two decimetres above LT (van Goor 1920, Wohlenberg 1935, Harmsen 1936, Klok & Schalkers 1980, Boley 1988, van Katwijk & Hermus 2000). This consistency between the notes of eye-witness-scientists and the calculations performed in this study indicates that the data used and the analyses are suffici ently reliable, notwithstanding the inaccuracies in the sounding method and positioning.

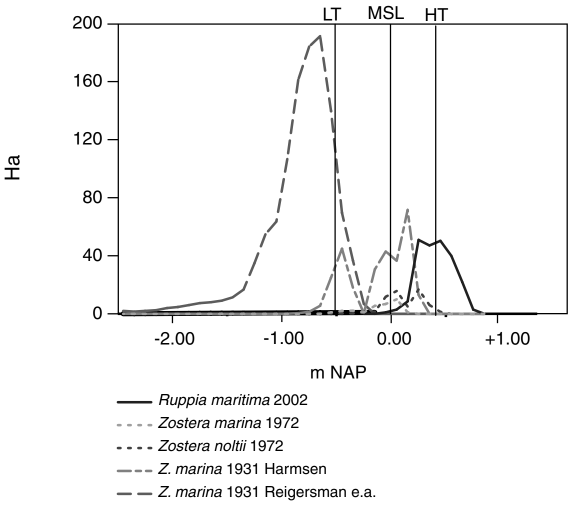

Figure 3. Depth distribution of water plants in Balgzand in 1931 mapped by Reigersman et al. (1939) and by Harmsen (1936), in 1972 mapped by den Hartog and Polderman 1975 and in 2002 by R. van 't Veer (unpubl.). LT: Low tide level, MSL: Mean Sea Level, HT: High tide level (Klok & Schalkers 1980).

Table 1. Depth distribution of the coastal zone of Balgzand compared between the 1930s, 1970s and 2000s. In this GIS-analysis we selected grid cells that (a) had *Ruppia maritima* present in the 2000, and (b) had depth data available in the 1930s.

	area < 0.20 m MSL (ha)	area >= 0.20 m MSL (ha)
1930s	200	1
1970s	43	139
2000s	16	185

Table 2. Tidal depth optima of water plants in the western Wadden Sea, and in Balgzand in particular, on the basis of a GIS-analysis of water plant maps and tidal depth maps.

Macrophyte	Depth optimum in cm MSL	Depth optimum in cm LT
Eelgrass Waddenzee 1931, by boat[1]	-100	-40
Eelgrass Balgzand 1931, by boat[1]	-70	-20
Eelgrass Balgzand 1931 location A[2]	-15 to +14	
Eelgrass Balgzand 1931 location B [2]	-50	
Eelgrass Balgzand 1972[3]	-25 to -5	
Dwarf eelgrass Balgzand 1972[3]	-5 and +20	
Widgeon grass (van't Veer, this study)	-15 to +54	

[1]Vegetation map of Reigersman et al. 1939
[2]Vegetation map of Harmsen 1936
[3]Vegetation map of Polderman & den Hartog 1975

Table 3. Average high tide (HT) and low tide (LT) level in the Balgzand area and in the seagrass beds in the Wadden Sea in 1931 (based on data of Klok & Schalkers 1980 and the seagrass bed map of Reigersman et al. 1939.

	m LT	m HT
Balgzand	-0.50	0.40
Wadden Sea seagrass beds 1931	-0.60	0.40

The settlement and expansion of widgeon grass in recent years may be explained by the decreased tidal depth following sedimentation (table 1); also in Chesapeake Bay and in the Baltic Sea, widgeon grass grows generally shallower than eelgrass (Orth & Moore 1988; Batiuk et al. 1992, Boström & Bonsdorff 2000, Moore et al. 2000). Obviously, the correlation is no indication for causality. The invasion of widgeon grass may indicate a lowered salinity, as this species has a lower salinity optimum than eelgrass (Verhoeven 1979, van Katwijk et al. 1999, Moore et al. 2000, La Peyre 2003). In the 1930's, in the Wadden Sea and Zuiderzee, the salinity range of eelgrass was 10-30 PSU (comparison seagrass maps of Oudemans et al. 1870, Reigersman 1939 with salinity data van der Hoeven 1982). At present, at the Balgzand the salinity drops frequently to 10-15 psu and occasionally as low as 5 psu), as appeared from a continuous monitoring program during 2005. The salinity drops were related to the discharges from Lake IJssel in combination with easterly winds (van Reen 2005). There are no indications that the discharge regime has changed during the last decades, though (van Reen 2005, www.waterbase.nl).

Dynamics in present natural populations in the Dutch Wadden Sea.

Since mid-1990s, four seagrass beds in the Dutch Wadden Sea have been monitored on a yearly basis (e.g. Groeneweg 2004b, Erftemeijer 2005, Fig. 1) by the Department of Public Works within the framework of the biological monitoring program. One of these beds, the eelgrass bed at Terschelling Harbour, had disappeared in 2003 (see also Fig. 4). A new bed has appeared in the Ems estuary, across a channel, 4 km west of the eelgrass population of 'Hond/Paap'. This area, called "Voolhok" is an area with high sedimentation rates. The area probably receives seed from the Hond/Paap beds since long, but only recently, the tidal depths have decreased sufficiently to provide a suitable habitat for germination and bed development. The bed was discovered in 2003 and was not present in 1999. Apart from the beds mentioned above, there are no significant seagrass occurrences in the Dutch Wadden Sea, except the small transplants of eelgrass and dwarf eelgrass at Balgzand, mentioned above.

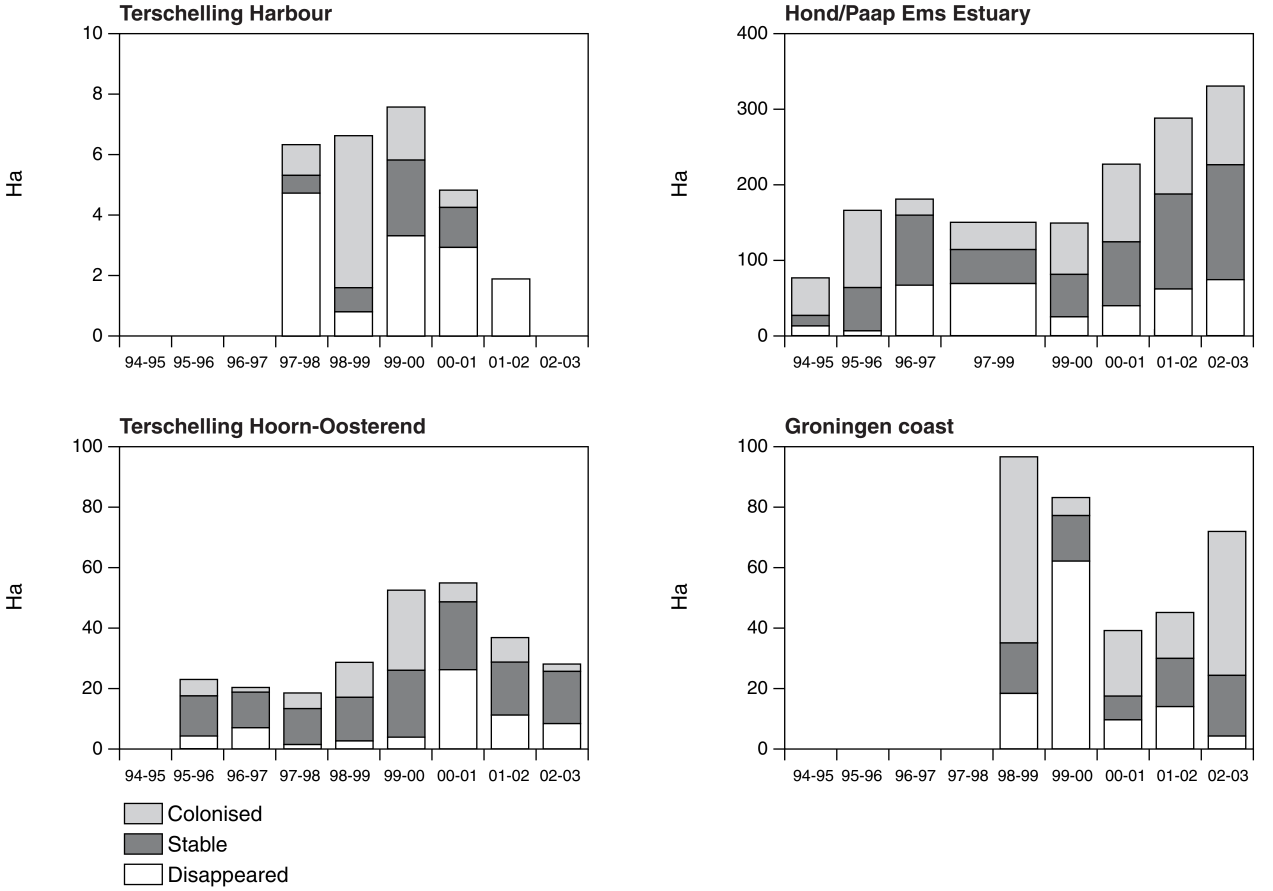

Figure 4. Dynamics of four seagrass beds in the Dutch Wadden Sea. Area of seagrass beds with 'colonised': area that is newly colonised in comparison to the preceding year, 'stable': area that was also vegetated in the preceding year, and 'disappeared': the area that was vegetated in the preceding year but not in the present year

The dynamics of the four seagrass beds in the Dutch Wadden Sea were analysed, using the monitoring data provided by the Ministry of Transport, Public Works and Water Management. Data of the following years were available and have been analysed: Terschelling Hoorn-Oosterend: 1995-2003; Groningen coast: 1998-2003; Terschelling harbour: 1998-2002; Hond/Paap in the Ems estuary: 1994-2003 with 1998 missing; we analysed 1999 data in comparison to 1997 data instead. Inaccuracies in monitoring can rise from the timing of the aerial and ground surveys, the spatial resolution of map data, the consistency in interpretation accuracy of aerial photographs and, for the category 0-5% cover that is hardly visible on aerial photographs, also the limited chance of detection in ground truthing (Frederiksen et al. 2004, Erftemeijer 2005, D. de Jong, pers. comm.)

Using GIS-analysis, we calculated the differences in surface area between two subsequent years, and made a distinction between newly *colonised* areas (areas that were not covered by seagrass in the preceding year), *stable* areas (that had seagrass cover in both years), and areas where seagrass had *disappeared* (local extinction; Fig. 4). The average percentage 'colonised', versus percentage 'stable', was relatively constant per population (no large standard errors of the mean, Fig. 5).

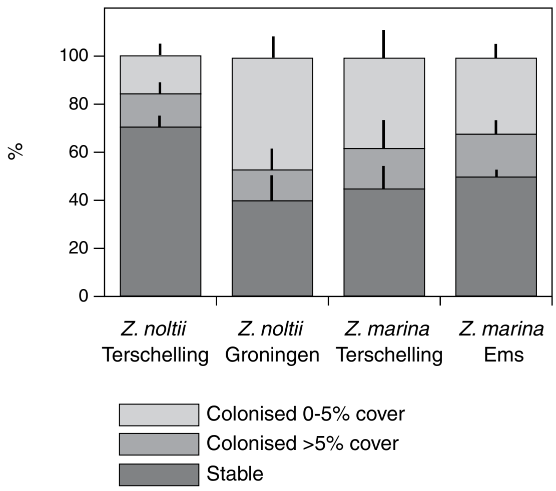

Figure 5. Percentage of the seagrass area that was newly colonised versus the area that was already covered by seagrass in the preceding year in the four seagrass localities present in the Dutch Wadden Sea. Newly colonised areas are subdivided in <5% and >5% seagrass cover.

In three out of four populations the newly colonised area was between 50 and 60% of the total population. The 'colonised' areas mainly have a cover of less than 5%. Only the dwarf eelgrass population of

Terschelling near Hoorn and Oosterend was less dynamic: at average only 30% of a seagrass area in a given year was (re-) colonised and the percentage 'stable' area (i.e. that was also seagrass-covered in the preceding year) was 70% at average. As this population shows no net losses or increases (Erftemeijer 2005), the yearly extinction in the covered area is also about 30%, which can also been seen in Fig. 4. The low extinction percentage may have been due to the compact, stabile sediments at this location (drown salt marshes) in combination with the predominantly perennial strategy of dwarf eelgrass. The low (re-) colonisation percentage indicates that the number of suitable areas in the vicinity is limited.

The high (re-) colonisation area in most seagrass beds of 50-60% yearly, means that if only the existing beds are protected, one would potentially loose 50-60% of the population in the subsequent year. The risk is particularly high in winter and spring, when the new colonisations are not yet visible. Secondly, in summer when new monitoring results are not yet present and available, the risk of overlooking a new colonisation is present when the beds are sparsely covered, i.e. <5%. At average the area with risk of overlooking was calculated to be 30-50% of the seagrass covered area (except for the Hoorn/Oosterend dwarf eelgrass population: 15%; Fig. 5). Then, in summer, a new monitoring is performed and seagrass area (= area of protection) is adjusted to the new situation. When monitoring is not performed on a yearly basis, the risk of losses increases further. This follows from the situation that the protected area will be partly un-vegetated, the unprotected area will be partly vegetated as a consequence of the yearly bed dynamics.

Notably, comparison of historic maps of 1869 and 1931 reveals that also the subtidal, perennial form of seagrass showed large dynamics: 75% of the seagrass beds present in 1931 were a new colonisation compared to 1869, whereas 25% had remained at, or re-vegetated the same location during these 72 years. 55% of the vegetation present in 1869 had disappeared in 1931 (the seeming contradictions in these percentages are due to an increase in the total seagrass cover during the period 1869 and 1931).

Transplantations of eelgrass and dwarf eelgrass at Balgzand, 1993-2004

As part of a large reintroduction programme (van Katwijk et al. 2000, van Katwijk 2003, Bos et al. 2005, Bos & van Katwijk subm.), seagrass transplantations have been carried out in 1993, 1998, 2003 and 2004. In 1993 both eelgrass and dwarf eelgrass were introduced. Dwarf eelgrass transplantations (methods: 1x1 m, 100 plants, planting date May 19th 1993, planting depths between MSL –0.4 and + 0.3 m; survival in 1994 reported between MSL –0.1 and + 0.15 m, Hermus 1995) appeared to have been successful, as they have been and are still spreading since (Fig. 6), though all colonisations are 1-2 m in diameter,

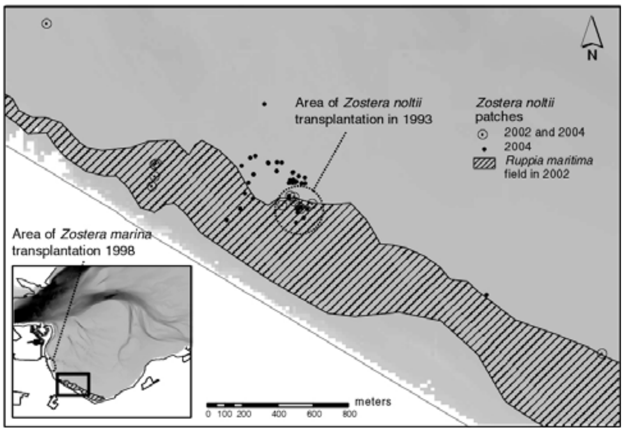

Figure 6. Dwarf eelgrass in 2002 and 2004, transplanted in 1993 at the encircled location; Inserted: location of eelgrass transplantation in 1998.

and do not form beds, yet. In 1993, eelgrass transplantations were successful only during the first growing season and did not survive through the winter, neither by surviving plants, nor through recruitment by seed (van Katwijk & Hermus 2000). Transplantations at the same location in 2003 gave similar results (Bos & van Katwijk 2005). In December 1998, seed bearing shoots (circa 5 kg wet weight) were deposited at another location within the Balgzand area (Fig. 6, details of the area see van Katwijk & Wijgergangs 2004). This transplantation was successful. However, the numbers of plants are highly variable, making this transplantation vulnerable to extinction (Table 4).

Table 4. Development of a transplantion of eelgrass plants at Balgzand, resulting from a donation of seed bearing shoots in December 1998.

year	number of plants	ha
1999	Ca. 100	
2000	Ca. 300	
2001	Ca. 200	
2002	13	
2003	Ca. 800	5.3
2004	50	

Conclusions and recommendations

From monitoring data covering 75 years, we could relate the sedimentation at Balgzand (decreased tidal depth) with a sequence of water plant species dominance in the area (eelgrass dominance, followed by dwarf eelgrass dominance, followed by widgeon grass dominance, Table 1 and 2, Fig. 3).

This temporal relationship corresponds to known spatial relationships (zonation): also in Chesapeake Bay and in the Baltic Sea (Orth & Moore 1988; Batiuk et al. 1992; Boström & Bonsdorff 2000, Moore et al., 2000). *Ruppia* grows shallower than *Zostera*, and dwarf eelgrass is known to occur shallower than eelgrass in the rest of the Wadden Sea, but also in the UK and Brittany, France (Harmsen 1936, Wohlenberg 1935, den Hartog & Polderman 1975, personal observation), with exception of an eelgrass zone with a wintergreen perennial strategy, that occasionally can be encountered in a zone above the dwarf eelgrass zone (van Katwijk et al. 1998). This consistency between the temporal relationship and the zonation patterns suggests that this relationship may be causal. In other words, the invasion of widgeon grass may have been caused by the sedimentation in the Balgzand area. However, based on the depth profiles, one would have expected the invasion sooner: already in the 1970s, the area had silted up considerably (Table 1).

Additionally, other explanations for the observed sequence of macrophyte cover over 75 years are possible. For example, it is known that widgeon grass has a much lower salinity optimum than *Zostera* species (e.g. Verhoeven 1979, Moore et al. 2000). Low salinities have been measured at the Balgzand area (van Reen 2005), offering a possible explanation of the presence of *Ruppia maritima*. However, also in this case an invasion would have been expected earlier as the discharges have not increased or decreased during the last 30 years (www.waterbase.nl). Distributional impairments could have caused the delayed establishment: seeds can travel by birds (Figuerola & Green 2002, Figuerola et al. 2002), but only occasionally this may result in a successful establishment (Clausen et al. 2002). Another means of transportation would be detached shoots bearing seeds (Cho & Poirrier 2005), which can probably travel over large distances, as was found for eelgrass (Harwell & Orth 2002, Reusch 2002, Erftemeijer et al. subm.). Once established, that plants can expand rapidly (Silberhorn et al. 1996, Cho & Poirrier 2005). One may also tentatively speculate that the low general environmental quality during the 1970s and 1980s (high levels of turbidity, eutrophication, heavy metals, toxicants, Marijnissen et al. 2001, de Jonge & de Jong 2002, van Beusekom & de Jonge 2002), may have prevented an earlier establishment of *Ruppia*.

Our GIS-study shows that seagrass bed dynamics are high. Between 1869 and 1931, only 30% of the subtidal beds of the robust type of eelgrass had remained at the same location, whereas 70% had 'moved'. Additionally, the population had expanded with a 25% increase in surface area (note that these are net values: we do not know the dynamics in the intermediate years). Present day seagrass beds, composed of the flexible type of eelgrass and dwarf eelgrass, show yearly shifts of 50-60%. Below, we will elaborate the implications of these dynamics to protection measures.

The importance of regulating shellfish fisheries to effectively protect seagrass beds was shown by de Jonge & de Jong (1992), van Katwijk (2003) and Essink et al. (2003). At the Groningen coast, Essink and co-workers recently recorded the loss of a substantial part of the dwarf eelgrass population following shellfisheries activities. They recommend a buffer zone of 400 m for the Groningen coast. Our study of the observed year-to-year dynamics of the Dutch Wadden Sea seagrass populations indeed urge the need for protection of a larger area than only the present seagrass bed. By doing the latter, one may potentially loose more than 50% of the bed each year (Fig. 5). We recommend that the buffer zone should be established on the basis of the area surrounding the beds that can be considered as suitable for seagrass. The latter can be accomplished by using the habitat suitability model (de Jong et al. 2005, Bos et al. 2005), and on-site expert judgement. Additionally, long-term trend analyses of the bed dynamics (e.g. Erftemeijer 2005, www.zeegras.nl) could help to establish the potentially suitable areas surrounding seagrass beds.

To allow for a larger scale expansion of existing seagrass populations, protection of high potential seagrass areas remote of existing beds is recommended. The recent establishment of a bed more than 4 km remote from an existing bed proves that seagrass is capable to establish outside the direct margins of existing beds. To assign high potential areas, the habitat suitability model for seagrass in the Dutch Wadden Sea can be used (de Jong et al. 2005, Bos et al. 2005). Within the areas appointed by the model, a further refinement should be made at the site, to account for local circumstances. Both the establishment of large buffer zones surrounding seagrass beds and the establishment of protected areas at high potential areas are recommended to be incorporated in EU-regulations and in Wadden Sea management plans.

From the results of transplantations at Balgzand in 1993 (dwarf eelgrass) and 1998 (eelgrass), and from the GIS-analyses in this study, we can conclude that both seagrass transplantations and beds are highly dynamic. A dynamic population strategy is obviously the best strategy in a dynamic environment such as the Wadden Sea. From this, we recommend that transplantation programmes should spread risks in space and time, which is basically what natural populations do as well.

Another recommendation that rises from the observed dynamics is to monitor the abiotic variables in the seagrass beds. Correlations between seagrass dynamics and these environmental variables will provide invaluable insight in the habitat requirements of these threatened species. Variables of interest are, for example depth, salinity and nutrient concentrations during a tidal cycle and over a season (van Katwijk et al. 2000). Additionally, to get insight in the causes of local disappearances that occur in natural beds (Fig. 5) as well as in transplantations (Table 2, Bos et al. 2004, Bos & van Katwijk 2005), continuous visual monitoring is recommended. In monitoring the transplantations, very sudden disappearances during the growing season were detected. When did these plants disappear exactly? After an abundant visit of foraging birds, or after a period of a particular combination of wind speed and direction? Etcetera. Continuous visual monitoring, e.g. using webcams, is a technique that is presently coming into reach to efficiently provide this information.

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Long-term changes in intertidal macrozoobenthos; the wax of polychaetes and wane of bivalves?

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Essink, K.*, Damm-Böcker, S., Dekker, R., Kleef, H., Madsen, P.B. & Ullerup, N. 2006: Long-term changes in intertidal macrozoobenthos; the wax of polychaetes and wane of bivalves? In: Monitoring and Assessment in the Wadden Sea. Proceedings from the 11. Scientific Wadden Sea Symposium, Esbjerg, Denmark, 4.-8. April, 2005 (Laursen, K. Ed.) NERI Technical Report No. 573, pp. 99-105.

At a number of fixed locations in the Dutch, German and Danish Wadden Sea intertidal macrozoobenthos is monitored in the framework of the Trilateral Monitoring and Assessment Programme (TMAP). At the 9th International Scientific Wadden Sea Symposium (Norderney 1996), the results of this long-term trend monitoring programme were used to answer the question whether trends in biomass in different parts of the Wadden Sea were governed by differences in nutrient loads. At that time, that specific question could not unequivocally be answered. An obvious trend, however, was shown of increasing biomass values of polychaetes, whereas bivalve biomass showed strong fluctuations mainly governed by the severity of winter conditions.

In this contribution, new data on the development of bivalve and polychaete populations still show the same picture of 1) fluctuating bivalve biomass and 2) an increase of polychaetes. Individual polychaete species, however, sometimes show different trends, even at a scale of less than one kilometre. This suggests that sediment composition and associated fauna play a yet not understood role in the development of these polychaetes.

Key words: Arenicola marina, biomass, Heteromastus filiformis, monitoring, Macoma balthica, Nereis diversicolor, Scoloplos armiger, Wadden Sea

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Introduction

In a previous paper, presented at the 9th International Scientific Wadden Sea Symposium, November 1996, we presented long-term data on biomass of intertidal macrozoobenthic biomass and addressed the question whether biomass of these species was governed by nutrient loads (Essink et al. 1998). On that occasion we compared data sets from different parts of the Wadden Sea and tried to relate the biomass development with changes in riverine nutrient discharge, nutrient concentrations in the water and chlorophyll concentrations as a measure of phytoplankton biomass. The conclusions were the following:

1. the macrozoobenthic biomass showed large interannual fluctuations, which were mainly caused by bivalves.
2. these fluctuations in biomass were to a great extent synchronised by the character of winters, particularly severe winters which usually are followed by good reproduction of bivalves.
3. the total biomass of polychaetes showed a long-term increasing trend, with the exception however of the Danish Wadden Sea where no trend was discernable.

4. it was not possible to demonstrate a relationship between the increase of polychaete biomass on the one hand side and the trends of nutrient loads in the different parts of the Wadden Sea on the other.

In the present paper an update will be presented, with further data on the development of bivalves and polychaetes living at intertidal flats of the Wadden Sea.

Material and methods

In the international Wadden Sea monitoring programmes of intertidal macrozoobenthos are executed by different Danish, German and Dutch monitoring institutes, and coordinated within the Trilateral Monitoring and Assessment Program (TMAP 2000) (Fig. 1). For this study data on total macrozoobenthic biomass, and the biomass of the separate groups of bivalves and polychaetes were used.

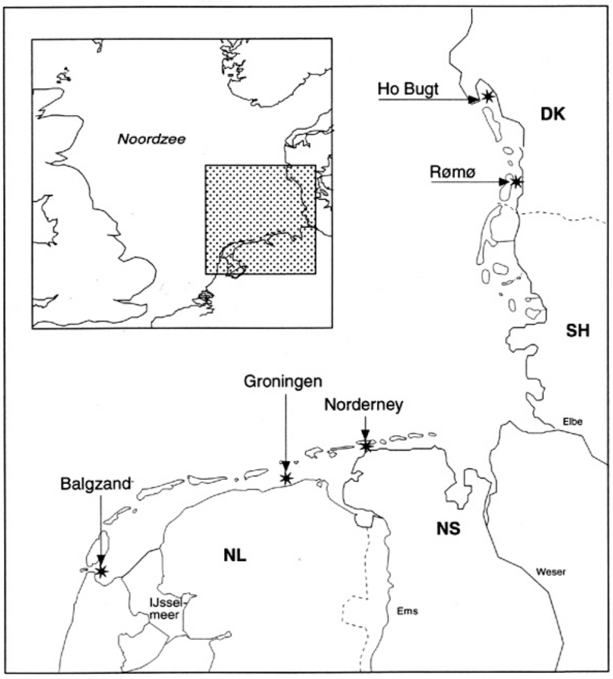

Figure 1. Map of the international Wadden Sea showing the five sub-areas where intertidal macrozoobenthos is monitored within the Trilateral Monitoring and Assessment Program (TMAP). NL: The Netherlands; NS: Niedersachsen (Lower Saxony); SH: Schleswig-Holstein; DK: Denmark.

Macrozoobenthos was monitored at fixed stations and transects, at the end of winter and in late summer/early autumn. From these data annual means were calculated as the average of winter and summer/autumn values and used as indicator of population size. Biomass was expressed in gram ash-free dry weight (AFDW) of the soft parts of the species involved.

Details regarding sampling, sorting of samples and of biomass determination are as given in Essink et al. (1998). It must be noted that for the Danish Wadden Sea 'total polychaete biomass' relates to only the major four species in the community, viz. lugworm *(Arenicola marina)*, ragworm *(Nereis diversicolor)*, *Heteromastus filiformis* and *Scoloplos armiger* (Madsen et al. 2005).

Results

Total biomass and bivalves
To illustrate the long-term development of total biomass and of bivalves data are presented from three sub-areas: western Dutch Wadden Sea (Balgzand), Lower Saxony (Norderney) and Danish Wadden Sea (Rømø area and Ho Bugt) (Fig. 2). It can be seen that the fluctuations of total biomass are virtually determined by the fluctuations in bivalve biomass. On average, bivalves constitute 40 – 60% of the total macrozoobenthic biomass.

The data sets of Balgzand and Rømø area, most clearly the autumn data, show a decreasing trend of bivalve biomass from 1998 onwards. At Norderney and Ho Bugt such a trend is not clear. At Norderney, autumn values are decreasing since 2001; at Ho Bugt the decrease started already in 1988.

Polychaetes
Polychaete biomass shows different trends in the different sub-areas of the Wadden Sea (Fig. 3). At Balgzand there is a continuously increasing biomass (from approx. 6 gram AFDW m-2 in 1980 to more than 20 gram AFDW m-2 in 1999 and 2003). A similar increase was observed at Norderney, whereas in the eastern Dutch Wadden Sea (Groningen) and in Ho Bugt no trend was apparent. In the Rømø area an apparent increase of polychaete biomass (only 4 major species determined) was followed by a steep decrease since 1999.

Single species among the polychaetes do not always show the same development (Fig. 4). The ragworm *(Nereis diversicolor)* shows increasing biomass values at Norderney and Ho Bugt, whereas at Groningen a decreasing trend is present. A different pattern, of an initial increase, followed by a decrease, was observed in the Rømø area. Different developments may occur even at small distances (less than one kilometre) as shown for Groningen in Fig. 5. The decreasing trend which is present at three fixed stations (47-0, 51-2, 54-0) with a more muddy sediment is not present at the more sandy stations 47-1 and 54-1.

The deposit-feeding polychaete *Heteromastus filiformis* shows an increasing biomass at Norderney intertidal flats, and a similar increase may be present in the Rømø area (Fig. 6). At Ho Bugt, no clear trend is visible. Again, the Groningen data show how the development of a polychaete species may be location dependent, with a decreasing trend

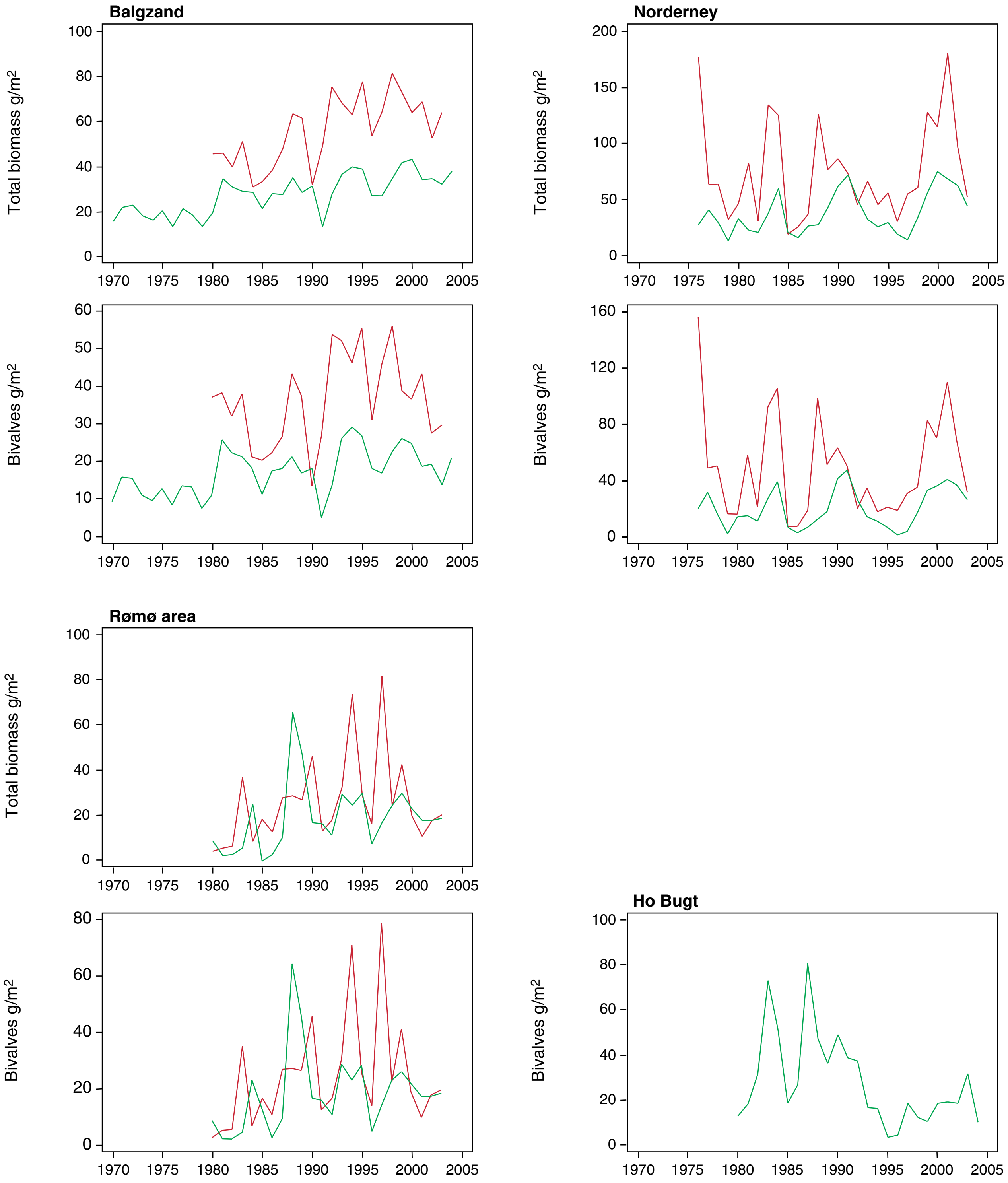

Figure 2. Biomass of total macrozoobenthos and of bivalves at intertidal flats of Balgzand, Norderney, Rømø area and Ho Bugt. Biomass: gram AFDW m-2. Red lines: autumn; black lines: spring. Note: 1) vertical axes have different scales; 2) Ho Bugt graph represents annual mean biomass.

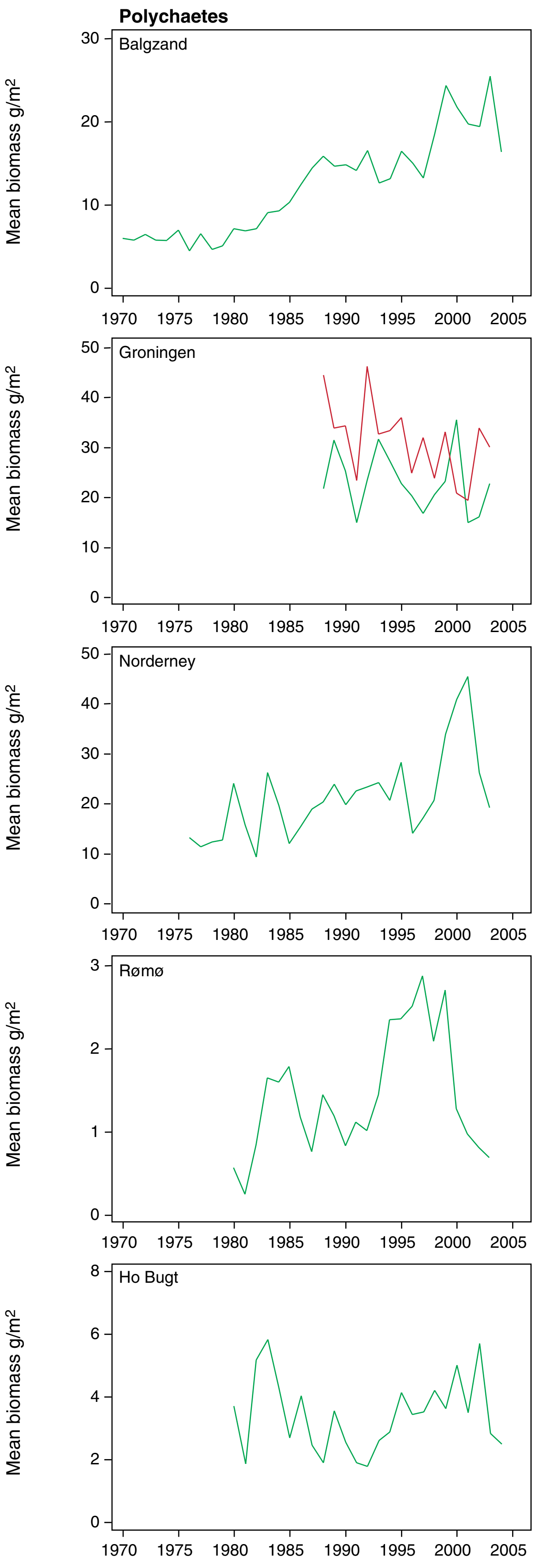

Figure 3. Annual mean biomass of polychaetes (gram AFDW m-2) at intertidal flats of Balgzand, Groningen, Norderney, Rømø area and Ho Bugt. Note: in Rømø area and Ho Bugt only four polychaete species were quantified; Groningen: black line = spring, red line = autumn.

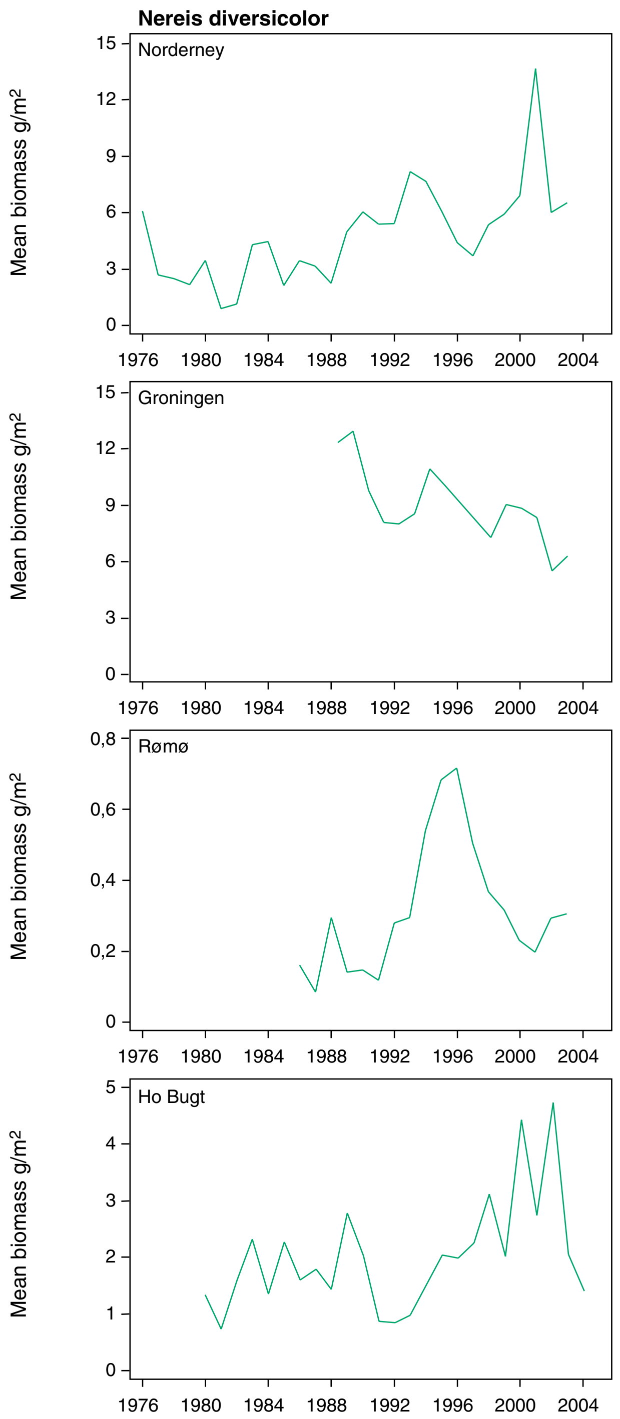

Figure 4. Development of annual mean biomass (gram AFDW m-2) of *Nereis diversicolor* at the intertidal flats of Groningen, Norderney, Rømø area and Ho Bugt. Note: different vertical scales.

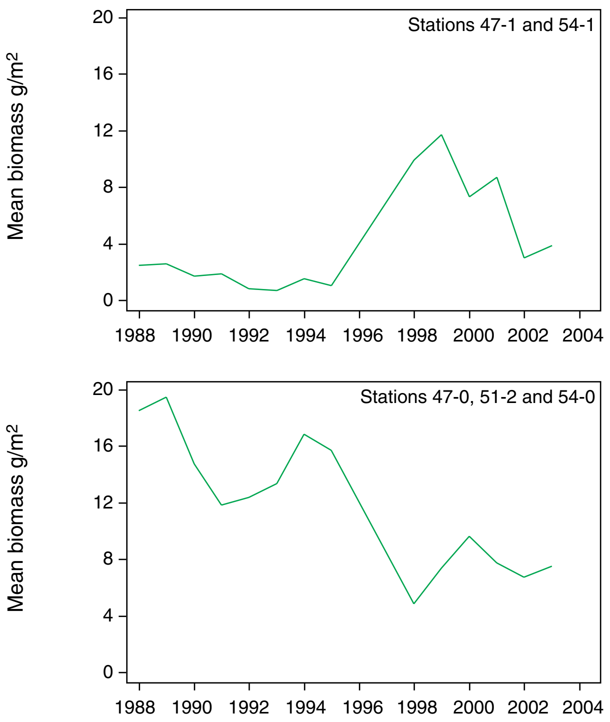

Figure 5. Different trends of annual mean biomass (gram AFDW m-2) of *Nereis diversicolor* at two sandy (top panel) and three muddy stations (lower panel) at Groningen intertidal flats.

at three muddy stations and an appearance since 1995 in two other stations, which are sandy

Polychaete species may even show a quite opposite development, such as *Scoloplos armiger*. This polychaete shows a steady increase at Norderney and a decrease – after 1984 - in the Rømø area (Fig. 7). In Ho Bugt, after 1985, no clear trend is visible.

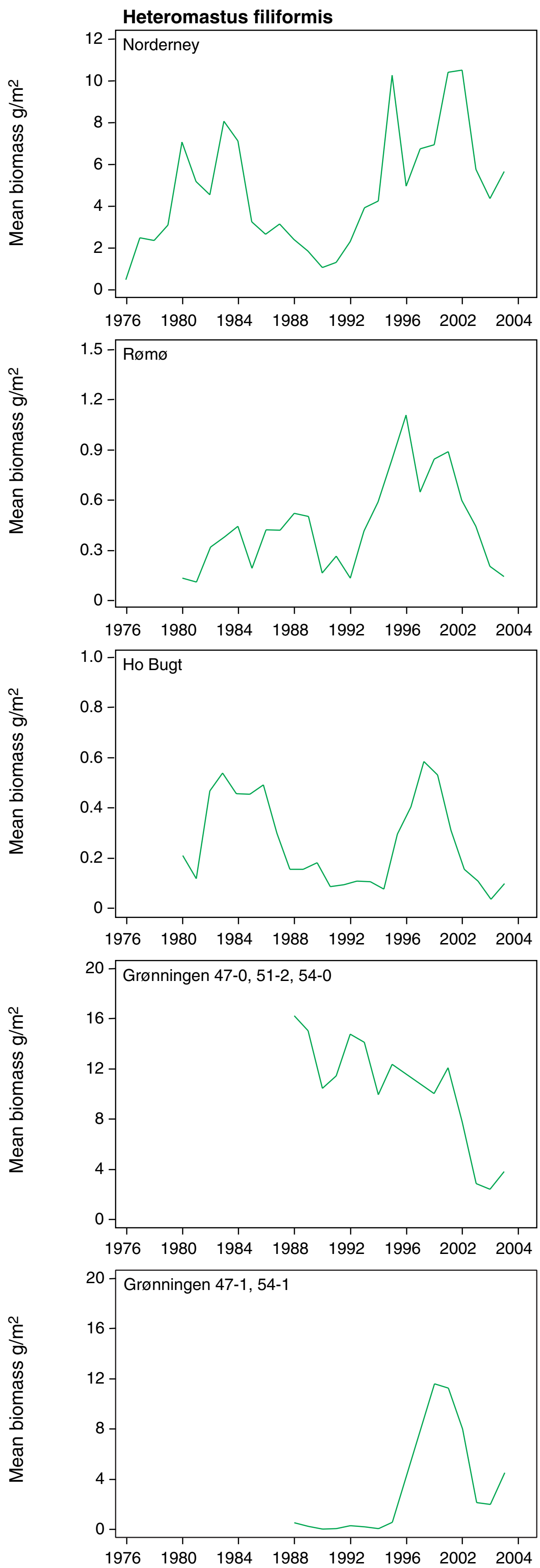

Figure 6. Development of annual mean biomass (gram AFDW m-2) of *Heteromastus filiformis* at the intertidal flats of Norderney, Rømø area, Ho Bugt and Groningen. Note: different vertical scales.

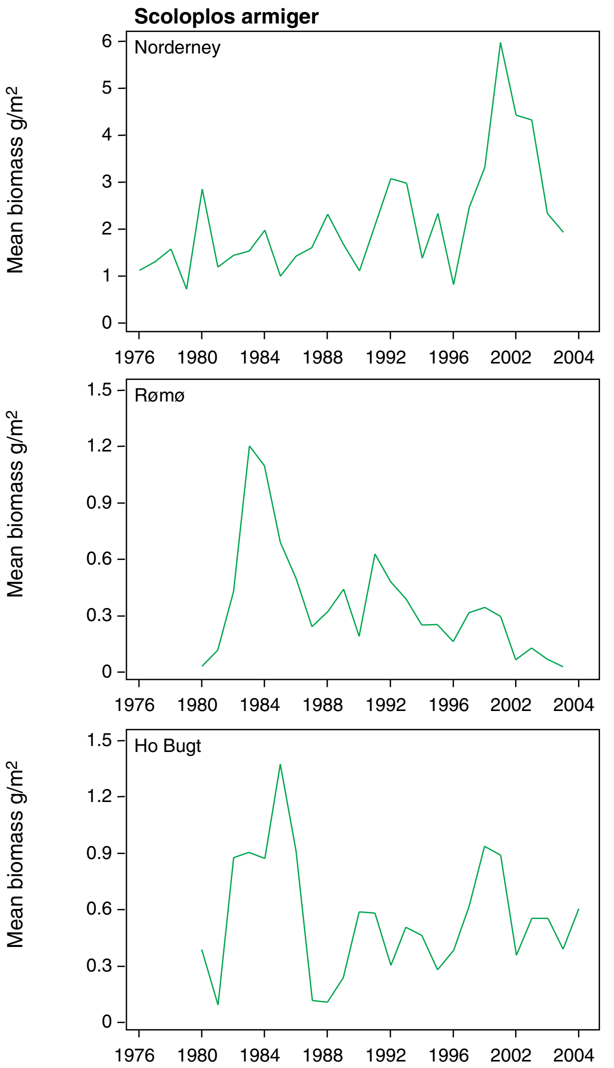

Figure 7. Development of annual mean biomass (gram AFDW m-2) of *Scoloplos armiger* at the intertidal flats of Norderney, Rømø area and Ho Bugt.

Discussion

The pattern of fluctuating bivalve populations shown earlier by Essink et al (1998) did not really change. Bivalves are largely responsible for the fluctuations of the total macrozoobenthic biomass living in intertidal flats of the Wadden Sea,
The total biomass of polychaetes showed continued increase at most sub-areas monitored. A clear relationship of this increase with eutrophication, as hypothesised by Essink et al (1998), does not seem to be present as riverine nutrient discharges into the Wadden Sea have been continuously decreasing (Van Beusekom et al. 2005; Van Beusekom 2006). However, even when assuming a total polychaete biomass in the Rømø area of twice the value of the four major species, biomass values in the northern Wadden Sea were much lower than in the southern part (Balgzand – Norderney).. Most of these polychaetous are deposit feeders, except e.g. the predator *Nephtys hombergii*. This difference in biomass may therefore still be a reflection of the different degree of eutrophication of these parts of the Wadden Sea as demonstrated by Van Beusekom et al (2005) and Van Beusekom (2006). Such a simple explanation, however, may be too easy since within a species opposite trends do occur at different localities. So, apparently different conditions are present, probably also including predator – prey relationships. Similarly, Rachor (2000) described differential development of benthic macrofauna biomass in muddy and sandy sediments in the German Bight during the period of strong eutrophication increase in the 20th century. The importance in local conditions, facilitating or limiting a response of zoobenthic biomass to changes in the trophic condition of the system were also shown by Beukema et al. (2002) for Balgzand intertidal flats.

Kröncke et al. (2001) assume that the observed increase since 1988 (until 1999) of abundance, biomass and species number in sublittoral benthic communities off the island of Norderney may be the result of a change in North Atlantic Oscillation (NAO) causing higher sea surface temperatures in late winter and early spring as well as increased primary production. Such relationships, however, have not yet been investigated with respect to intertidal benthic communities of the Wadden Sea.

The different trends within the same species on a small spatial scale as observed at Groningen intertidal flats suggests an intricate regulation of population size of these polychaetes. To unravel these details a further analysis of the data is necessary.

With respect to bivalves a decreasing trend since 1999 was observed. In fact, bivalves are not doing well lately, as can be illustrated by the Baltic telling *(Macoma balthica)*. Of this species abundance at Balgzand intertidal flats has steadily decreased (Drent et al., in prep) and instantaneous mortality rate has doubled since 1999 (R. Dekker, unpubl.). And at Groningen intertidal flats *M. balthica* does not get as old as it used to, i.e. there is increased adult mortality (P. Tydeman, unpubl.). During the last ca. 15 years at intertidal flats of the Dutch Wadden Sea recruitment failures in *M. balthica* have been more frequent than before (Beukema & Dekker 2005), especially at the lower tidal levels. Further, recruitment of *M. balthica* was shown to be negatively correlated with the abundance of shrimps *(Crangon crangon)* which are known to prey upon bivalve post-larvae. At higher intertidal levels such recruitment failures were not observed. Beukema & Dekker (2005) conclude that the centres of distribution of bivalves in their study area (Balgzand) have shifted to higher intertidal levels with muddier sediments, and that this shift is largely caused by increased epibenthic predation pressure (by shrimps *Crangon crangon)* which is facilitated by the occurrence of milder winters. Most recent data (not shown in Fig. 2), however, show a significant recovery of bivalve biomass at Balgzand in 2005-2006.

One could hypothesise that bivalves and polychaetes compete for food, and that decreasing phytoplankton grazing by decreasing bivalve populations provide a larger food supply to polychaetes in the form of algae eventually being deposited on the sediment. The observations by Beukema et al. (2002) show such a direct dependency of local benthic communities of algal food, but do not suggest any competition between suspension feeders (mainly bivalves) and deposit feeders (largely polychaete). Experiments by Kamermans et al. (1992) could not demonstrate any interspecific competition for food between the deposit feeding bivalve *Macoma balthica* and the suspension feeder *Cerastoderma edule.* It may be worthwhile, however, to use the obtained monitoring data from the Wadden Sea to test Levinton's (1972) hypothesis regarding interspecific competition for food in deposit feeding rather than in suspension feeding communities.

Conclusions

Bivalve biomass, making up 40-60% of total macrozoobenthic biomass, strongly fluctuates, reflecting the effect of the winter (mild or severe) on recruitment success. The total biomass of polychaetes tends to continue to increase. At the same time riverine nutrient discharges into the Wadden Sea decrease, thus not supporting the hypothesis of a causal relationship between polychaete biomass and eutrophication. Polychaete biomass, however, is larger in the more eutrophicated southern, than in the less eutrophicated northern part of the Wadden Sea.

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Winter survival of mussel beds in the intertidal part of the Dutch Wadden Sea

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Every winter, mussel beds in the tidal area of the Dutch Wadden Sea disappear due to storm and predation. The Dutch government only allows fishing of juvenile mussels on newly formed mussel beds if these have a low chance of surviving the following winter. However, at presence the mussel bed area that disappears during winter is unknown. Here, we present the distribution of mussel beds in the Dutch Wadden Sea from 1994 to 2003. We determined the spatial contour of present mussel beds in autumn and spring using GPS. For the first time, we can quantify winter losses and average winter survival. We show that almost 40% of all mussel bed area disappears every winter. Of all newly formed beds, 50% did not survive their first winter. The best areas for development of mature mussel beds are positioned south of Ameland and Schiermonnikoog and at Wierumer Wad along the Frisian coast. Furthermore, we compare average winter survival with a habitat suitability map.

Key words: mussel beds, Mytilus edulis, population dynamics, population stability, Wadden Sea, winter survival

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Introduction

Intertidal beds of the blue mussel *(Mytilus edulis)* are important biogenic structures in the Wadden Sea ecosystem. The beds serve as habitat and as an important food source for a number of species (de Vlas et al. 2005). For example, many birds eat the mussels or rely on the organisms that occur in and around the mussel bed (Dankers et al, 2003).

The larvae or spat of the blue mussel settle in May/June (Van de Koppel et al. 2005). After the settlement of larvae, which is also known as spat fall, a new mussel bed can be formed consisting of a monolayer of mussels. Between August and November a new bed rises 30-40 cm above the surrounding sand flat, due to sedimentation of considerable amounts of fine silt (Hilgerloh et al. 2003). After winter, sand and shells wash in and settle between the mussels. As beds mature, they can develop into a physically solid structure because the accumulated silt consolidates and forms a clay layer. For a detailed review on the development of mussel bed see e.g. Dankers et al. (2001).

The area of mussel beds in the Wadden Sea demonstrates large fluctuations in time due to the erratic amount of spat fall and due to environmental disturbances such as ice cover and storms (Beukema et al. 1993, Nehls & Thiel 1993, Nehls et al. 1997). Especially new mussel beds are vulnerable to these disturbances, because they are easily washed away. Only in sheltered areas, where the impact of storms and ice is less severe, mussel beds can develop into mature beds (Nehls et al. 1997, Nehls & Thiel 1993). When a mussel bed is clearly recognizable over many years, it is considered as a so-called stable bed (Brinkman et al. 2003, de Vlas et al. 2005).

In the early nineties, intertidal mussel beds in the Dutch Wadden Sea almost disappeared due to ongoing fisheries and a low amount of spat fall (Dankers et al. 1999). In 1993, the Dutch government took action to protect important habitats, such as mussel beds and closed permanently 25% of the intertidal area for shellfish fisheries (Ens et al. 2004, Dankers et al. 2001). In 1999, additional 10% of the intertidal area that was believed to be suitable for the development of mussel beds was closed for mussel fisheries (Dankers et al. 2001).

At present, the aim for Dutch Government management plan is a target area of 2000 ha of mussel beds that survived at least one winter in the entire Dutch Wadden Sea. Fisheries on newly formed mussel beds outside the above-mentioned closed areas are only allowed if the total mussel bed area is larger than the target. Even if this condition is met, fishing of juvenile mussels on a specific mussel bed is only allowed if this mussel bed has a low chance of surviving the following winter.

In order to be able to implement this policy, we need to 1) know the area of mussel beds on a yearly basis and 2) have a better understanding on the survival chance of mussel beds. Brinkman et al. (2002) present a habitat suitability map of the intertidal areas of the Dutch Wadden Sea based on the presence of mussel beds in the period 1960-1970 and several environmental characteristics. This map proposes classes of suitability for the natural establishment and survival of mussel beds (see also Brinkman & Bult 2002). Here, we monitored the distribution of mussel beds in the Dutch Wadden Sea from 1994 to 2003. We determined the spatial contour of present mussel beds in autumn and spring using GPS. For the first time, we can quantify winter losses and average winter survival. Now we can also compare average winter survival with the habitat suitability map.

Material and methods

Mussel bed localisation

We located mussel beds in autumn and spring from autumn 1994 to spring 2004. To roughly locate new mussel beds, we performed aerial inspection flights in both spring and autumn and used information from fishermen and fishery inspectors. To measure the precise geographical location, size and shape of each mussel beds, we walked around each mussel bed with a GPS device during low tide. As such, we obtained the spatial contours of all beds.

A mussel bed consists of a collection of smaller patches. Therefore, the boundaries between a mussel bed and the surrounding tidal flat are not always clear (De Vlas et al. 2005). To define the boundaries of a single mussel bed, we used the following criterion conform De Vlas et al. (2005); a group of mussel patches less than 25 meters apart is considered as a bed, but only if at least 5% of the tidal flat is covered by these patches.

Contour reconstruction of unvisited beds

We could not visit every mussel bed during all surveys. Therefore, we needed to reconstruct the spatial contour of the unvisited beds in the missing point of the time-series. Here, we present a summary of the used method of reconstruction. For a detailed description in Dutch see Steenbergen et al. (2003).

We assume that: 1) mussel bed do not change shape during summer, 2) newly formed beds appear in autumn and 3) mussel beds can partially disappear in winter.

If a mussel bed could not be visited in spring, we used the contour of the mussel bed present in autumn of that same year instead. If a mussel bed could not be visited in autumn, we used the contour of the mussel bed present in spring of that same year instead. If a newly formed mussel bed could not be visited in autumn, or an older mussel bed could not be visited in both spring and autumn of the same year, we used the contour of the mussel bed present in spring of the following year instead. In the latter case, there can be a systematic underestimation of the area of mussel beds, because substantial parts of a mussel bed can disappear during winter season. To reduce this underestimation during the surveys, we prioritised the newly formed beds in autumn and those beds that had not been visited in autumn, in spring.

Reconstruction of unvisited beds was therefore possible until spring 2003. In spring, we visited 65% of the total mussel bed area and therefore reconstructed 35%. In autumn, we visited 42% of the total mussel bed area and reconstructed 58%.

Data analysis

We used Arcview version 3.2 (ESRI) to analyse the geographical data. If mussels at a specific location were present in autumn and the subsequent spring, they survived winter. The average winter survival of mussels was calculated by dividing the number of winters that the mussels survived by the number of times mussels settled at the same location. For example, we consider a particular location in the investigated time series where mussels successfully settled in the first year, survived two winters and did not settle again. This location would be classified with an average winter survival of two (two divided by one). If in the fifth year mussels would settle again on this location that survive for three winters, the average winter survival equals 2.5 (five winters divided by two settlements). Because mussel beds almost never disappeared completely, mussel beds were divided into several parts with different average survival.

To calculate the percentage of the lumped area that survived either zero or one winter(s) this area was divided to the total mussel bed area that had been present up to and including autumn 2002. To calculate the percentage of the area that survived (more than) two winters this area was divided to the total mussel bed area that had been present up to autumn 2002. To calculate the percentage of the area that survived three winters this area was divided to the total mussel bed area that had been present up to autumn 2001. And so on. All data are presented as mean±SD.

Winter losses

We assume that the mussel beds are lost only due to natural effects, but we are aware of two fishing activities (De Vlas et al. 2005). Firstly, in the autumn of 1994, some fishery was allowed on newly formed beds of the 1994 spat fall. However, most of the newly formed beds - both fished and un-fished - disappeared completely due to severe storms before the survey in spring 1995 (De Vlas et al. 2005, Ens et al. 2004). Secondly, restricted experimental fisheries were carried out in 2001 on beds that were considered unstable to test the hypotheses that moderate fishery could restore the stability of young beds. Also in this case both fished and unfished beds were destroyed by autumn and winter storms (De Vlas et al. 2005; Smaal et al. 2003). We assume that the above-mentioned two fishing activities have a negligible on the monitored mussel bed area.

Results

From autumn 1994 until spring 2003, 9500 ha of the intertidal area were covered with mussel beds for at least one season. The total mussel bed area ranged from 451 ha in the spring of 1998 to almost 5000 ha in the autumn of 2001 (Fig. 1).

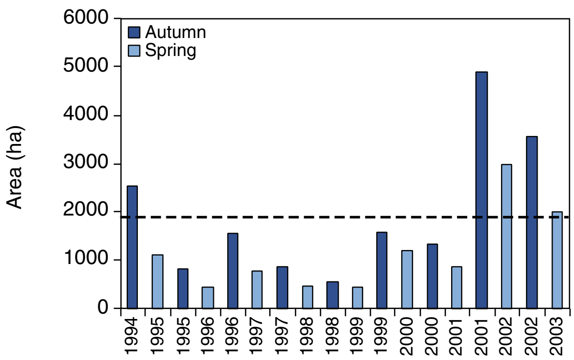

Figure 1. Development of mussel beds in the intertidal part of the Dutch Wadden Sea from autumn (au) 1994 until spring (sp) 2003. Dotted line indicates target area of 2000 ha.

The total mussel bed area in the Dutch Wadden Sea was very low in the nineties; in general there was less than 1000 ha present. In the nineties, the recovery of the mussel bed area started with good spat fall in 1994, but the winter losses were high and only 1000 ha remained. Until 2001 the total mussel bed area was below 2000 ha. The spat fall in autumn 2001 resulted in more than 2000 ha for the first time in seven years.

Almost 5000 ha of mussel beds were recorded of which 60% survived the winter period. The area of winter losses was 39.3 ± 12.6% (n = 9) and varied from 16.5% in the winter of 1998/1999 to 56.6% during the winter of 1994/1995 (Fig. 2). On 17% of the locations, mussels settled more then once.

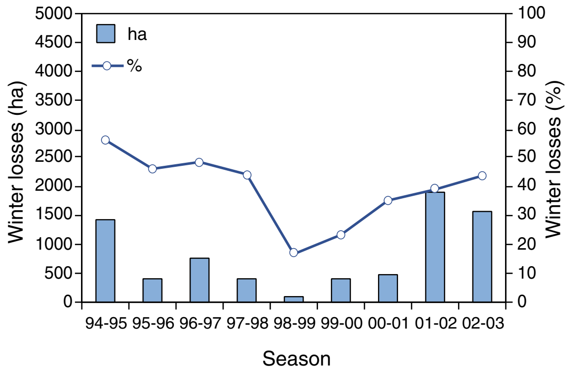

Figure 2. Yearly winter losses of mussel bed area in the intertidal part of the Dutch Wadden Sea since 1994, in hectares (left scale) and percentages (right scale).

The majority of mussel beds are located in the eastern part of the Dutch Wadden Sea (Fig. 3). More than half of the total mussel bed area did not survive the first winter (Fig. 4).

In the subsequent winters, the area roughly decreased by a two-fold: 27.8% survived one winter and 15.8% survived two winters. Of all the mussel bed area that was formed before autumn 2002, 22% survived two or more (2-9) winters. Only 1.3% of the area was formed before autumn 1994 and still present in spring 2003. These 32 ha of mussel beds therefore survived at least 9 winters. The mussel beds with large parts surviving more then four winters were situated at Wierummer Wad along the Frisian coast, or south of the islands Ameland (inset Fig. 3) and Schiermonnikoog.

Discussion

Our surveys show that 39.3 ± 12.6% (n = 9) of the mussel bed area in autumn disappeared during the winter season in the period 1994-2003. In many cases, mussel beds did not disappear completely; parts of beds could survive up to 4 years or more. Especially large parts of newly formed beds disappeared; 51.5% of the mussel bed area did not survive the first winter (Fig. 4). Parts of mussel beds that survived their first winter, still have a large change to disappear during following winters.

Winter losses are probably mainly caused by storms. Zwarts & Ens (1999) suggest that predation of birds can have a substantial effect, especially when only a few mussel beds are present. However, our data do not indicate higher winter loss percentages of mussel bed area in poor years. For example, in 1998-1999 the lowest winter loss (16.5%) co-occurred with the lowest area in autumn (540.5 ha).

We compared the average winter survival with the habitat suitability map of Brinkman et al. (2002).

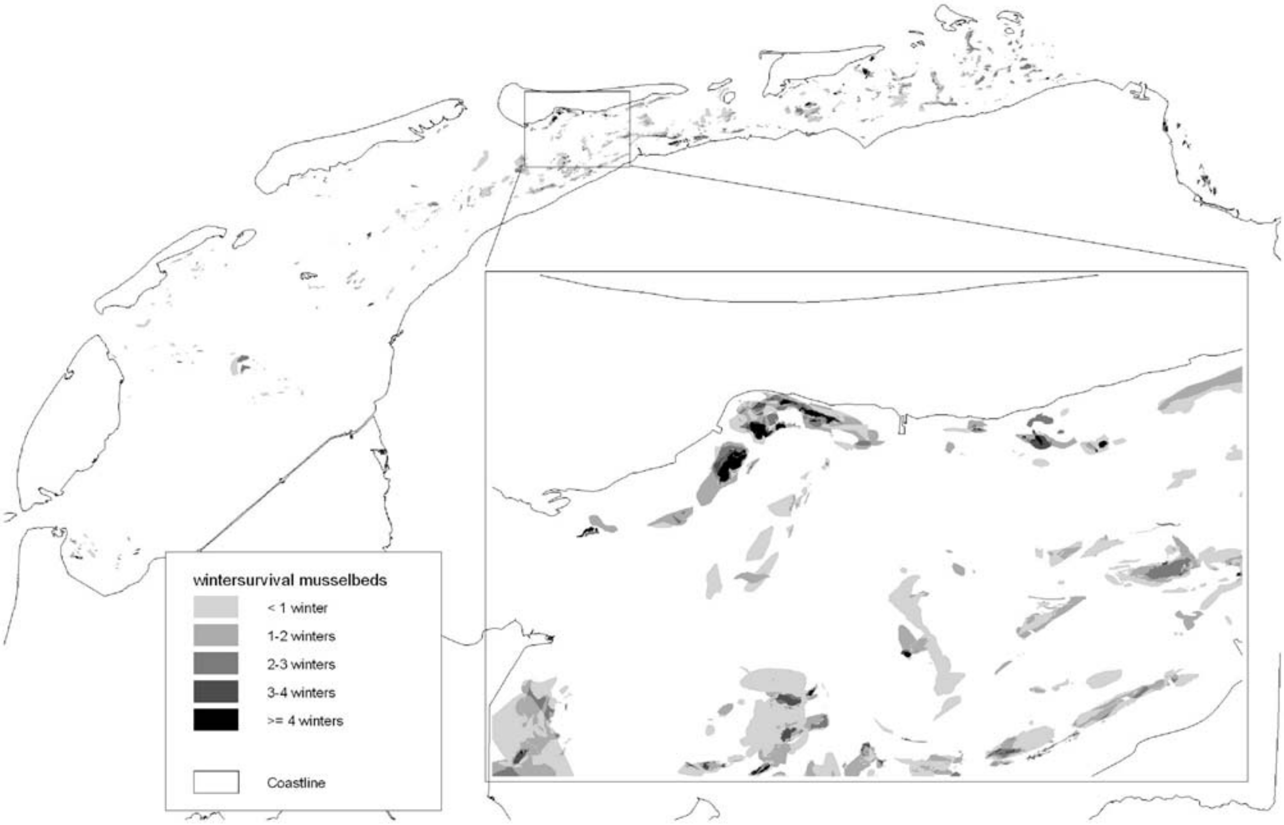

Figure 3. Average winter survival of (parts of) mussel beds in the intertidal of the Dutch Wadden Sea. The islands Ameland (A), Schiermonnikoog (B) and the Wierummer Wad along the Frisian coast (C) are indicated.

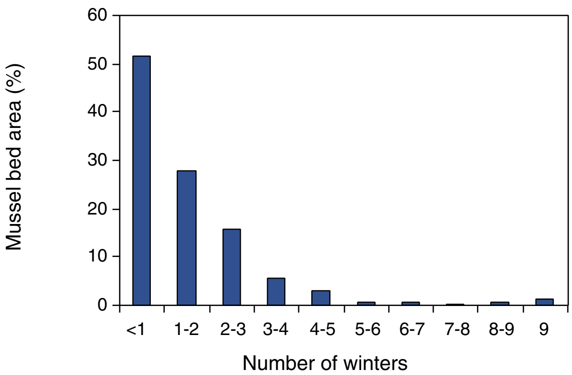

Figure 4. Average winter survival of the total mussel bed area that has been present in the intertidal of the Dutch Wadden Sea between autumn 1994 and spring 2003.

Average winter survival was 2 and 1.5 winters for class 1 and 2, respectively. These first two classes comprise the most suitable areas for the development of mussel beds and cover only 2% of the Wadden Sea. In the remaining part of the Wadden Sea, however, no major fluctuations in average winter survival were found (around 1 winter; Steenbergen et al. 2005). According to our data, the best areas for development of mature mussel beds are the areas south of the eastern islands, and at Wierummer Wad along the Frisian coast. These areas correspond with the 2% best areas suggested by the habitat suitability map.

Nehls et al. (1997) distinguished two types of mussel beds in the Wadden Sea of Schleswig-Holstein, Germany: dynamic or unstable beds and stable beds. Unstable beds were only present for some consecutive years and found in locations that were exposed to storms and ice. Stable beds were present for many years and found only at sheltered locations where the impact of storms and ice was less severe. We can conclude that mussel beds in the Dutch part of the Wadden Sea, which sustained over longer periods of time, all appear to be situated in sheltered areas. This seems consistent with the situation in the Niedersachsen (Germany) part of the Wadden Sea (Herlyn et al, 1999).

Although locations of stable mussel beds have been constant over decades, the total mussel bed area was highly variable in time (Dankers & Koelemaij 1989, Nehls & Thiel 1993). The unstable beds are of interest to mussel fisheries. In the investigated period from 1994 to 2003, a large area (83% of 9500 ha) of the intertidal area was covered with mussels only once due to erratic spat fall. With such a high spatial variability, it is not yet possible to predict future locations of unstable beds based only on observed winter survival.

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The Danish Wadden Sea; fishery of mussels *(Mytilus edulis L.)* in a Wildlife Reserve?

Per Sand Kristensen & Rasmus Borgstrøm

Sand Kristensen, P. & Borgstrøm, R. 2005: The Danish Wadden Sea; fishery of mussels *(Mytilus edulis L.)* in a Wildlife Reserve? In: Monitoring and Assessment in the Wadden Sea. Proceedings from the 11. Scientific Wadden Sea Symposium, Esbjerg, Denmark, 4.-8. April, 2005 (Laursen, K. Ed.). NERI Technical Report No. 573, pp. 113-122.

Mussels have been fished in The Danish Wadden Sea, an International Wildlife Reserve for many years. In the 1980's the mussel stock collapsed, and for a short period the food supply for birds was critically reduced. The number of mussel fishing licenses was drastically reduced, and since 1986 the mussel stocks have been monitored by the Danish Institute of Fisheries Research. Aerial photographs and mussel sampling have been used for estimation of bed area and for monitoring and assessment of the population. The mussel beds have varied between 1,192 ha in 1991 to 632 ha in 1996. In 1999 the total mussel beds were 1,051 ha. On the subtidal beds samples were collected by dredging using a commercial dredge and on the intertidal beds by a large number of frames. Combining these two factors the biomass was estimated. The biomass of mussel in The Danish Wadden Sea has varied considerably over the years. Minimum (5.840 t) and maximum (117.000 t) occurred in 2004 and 1993 respectively. Based on the biomass observed during autumn, the production of mussels for the next year was estimated. The estimated annual production was divided between birds and fishery in a way that at least 10,300 tons were allocated to the birds before a TAC was allocated the fishery. This management plan is intended to supply mussel foraging birds with sufficient food and to prevent overexploitation of the mussel population.

Key words: Mytilus edulis, aerial photography, GIS, image analysis, orthophotos, monitoring, assessment, biomass variation, food for birds, sustainability

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Introduction

The Wadden Sea covers a tidal area of approximately 10,000 km² in the south eastern North Sea between Den Helder in the Netherlands and Ho Bight in Denmark. The Danish part of the Wadden Sea is ca. 900 km². Three islands (Fanø, Mandø and Rømø) divide the Danish Wadden Sea into four tidal areas (Grey Deep, Knude Deep, Juvre Deep and Lister Deep between Rømø and Sylt), thus 500 km² are affected by tides. The blue mussel *(Mytilus edulis)* usually occurs in both subtidal and intertidal areas of the Danish Wadden Sea in beds of sizes from a few thousands m² up to a few km². Blue mussels form clusters on sand and mud bottom. Smaller beds (< one ha) may be very densely crowded with mussels (~ 70% cover). Since 1986 mussels abundance in the Danish Wadden Sea has been monitored using aerial photography and ground sampling (Munch-Petersen & Kristensen 1997 and 1989).

Mussel stocks in the Dutch and German Wadden Sea areas have also been monitored for many years. (Bult et al. 2000; Farke et al. 2004; Nehls et al. 2000; Nehls 2001 and Nehls & Ruth 2004, Stoddard 2003; Dankers et al. 2001; Ens et al. 2004; Herlyn 2005). The technique used in the three countries differs to a certain degree, but all monitoring programs estimate bed sizes (in hectares or square meters) and abundance. These data are used to support sustainable management of the population while ensuring sufficient mussel prey for foraging birds.

In 1992 48% of the Danish Wadden Sea was closed permanently for mussel fishing. The closed areas are ¾ of Ho Bight Juvre Deep and the southern part of Lister Deep at the border to the German Wadden Sea. Mussel fishing was only allowed in Hjerting Stream the south eastern part of Ho Bight, the southern part of Grey Deep and Knude Deep and the northern part of Lister Deep.

Since the beginning of the 1980's the mussels in the Danish Wadden Sea have been exploited at a high rate until the stock collapsed in 1987 (Munch-Petersen & Kristensen 1987, Kristensen & Laursen in prep). Since 1986 the Danish Institute of Fisheries Research (DIFRES) has monitored and assessed the mussel stock for authorities responsible for nature conservation and management of the fishery resources in the Danish Wadden Sea.

The estimation of bed sizes is based on aerial photographs, which have been improved throughout the whole period since 1986. In the early days black/white aerial photos were used, which to a certain extent made it possible to distinguish blue mussels from other organisms in the beds (Munch-Petersen & Kristensen 1987). From 1993 and onwards, colour photos were used to estimate mussel beds. The mussel beds were drawn on transparent plastic-folio and weighed to determine bed sizes (Kristensen 1994).

Introduction of the Geographical Information System (GIS) in 2002 made it possible to estimate the areas with mussels by digital image analysis by use of ortho photos (aerial photos corrected for terrain distortions etc.). Provided that the quality of the ortho photos and the physical parameters such as the tidal situation (low tide), clouds etc. are optimal, it is possible automatically to detect and estimate the sizes of mussel beds.

However, it is necessary systematically to take ground samples of the beds to determine the biomass of mussels alive.

The biomass of mussels in the Danish Wadden Sea has been estimated using these methods during the last 20 years to establish the amount of mussels necessary to sustain birds that depend on mussels for food before allocating mussels to the fishery.

This study describes the development of monitoring techniques and the assessment program performed since 1986 and the introduction and application of new techniques. A key element of the management of the mussel resources in the Danish Wadden Sea is that the population has to observe sustainable and must support a sufficient food supply for birds. In practice, this is accomplished by only allowing exploitation of the annual new mussel production (from October year one (x) to October the following year (x+1)).

Materials and methods

Stock assessment of mussels in the Danish Wadden Sea has been carried out using aerial photographs to estimate the mussel bed areas. These investigations have been combined with ground sampling to determine abundance (n/m²) and biomass (kg/ m²).

The total biomass of mussels in tons in the Danish Wadden Sea is determined by adding all bed size estimates per tidal area based on the aerial photographs and multiplying these figures with the biomass established through the field investigations for each tidal area respectively.

Before 1991 aerial photos in black/white have been used in planning the survey and monitoring of the mussel beds in the Danish Wadden Sea (Munch-Petersen & Kristensen 1987 and 1989) (Fig. 1A). However, some smaller and difficult beds with mussels were not recognizable on the b/w photographs (Munch-Petersen & Kristensen 1989) (Fig. 1B). After 1991 aerial colour photographs have been used to monitor mussels in the Danish Wadden Sea (Kristensen 1994 and 1995). The photos from 1991 until 1996 were taken from an altitude of 1,350 meters. Each photographic slide covered 1,852 * 1,852 m² (i.e. 1 sq. nautical mile).

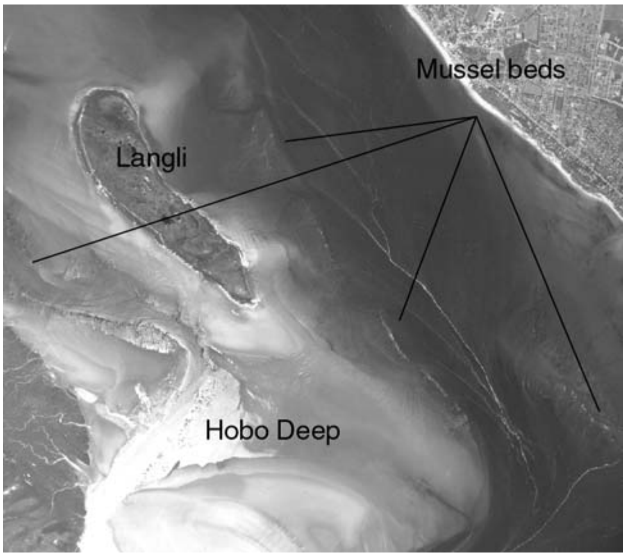


Figure 1A. Black and white aerial photo (1983).

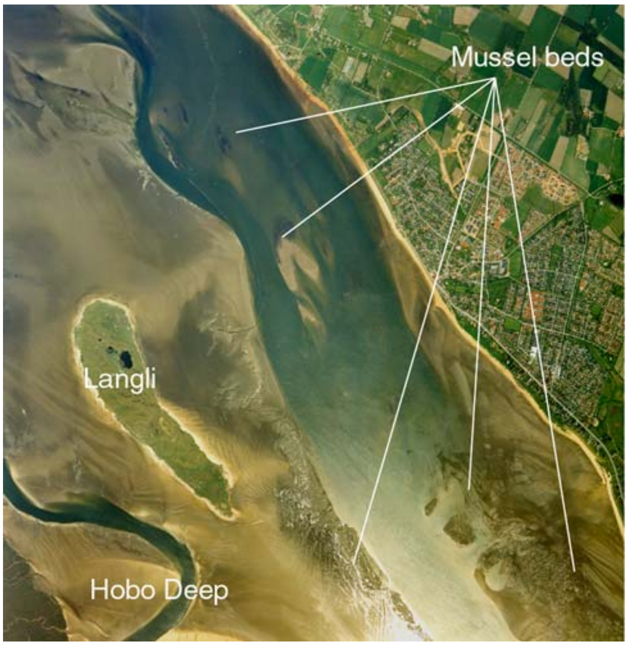


Figure 1B. Aerial colour picture (1999).

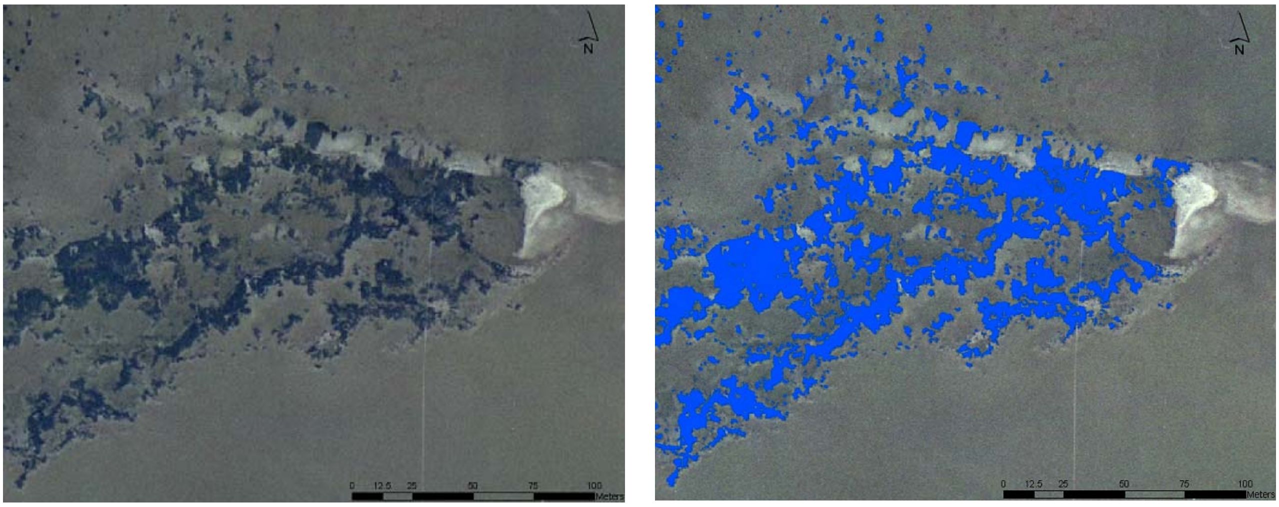


Figure 1C. Scanned ortho photo (2002); the blue areas on the right image indicate areas classified as mussel areas in the GIS analysis. (Note: each pixel in blue is 0.16 m^2)

After 1998 only regular colour ortho photos were taken every 2nd or 4th year in a scale 1:25,000. In 2004 digitisation of the colour photos was introduced to create ortho photos to identify mussel beds in a more precise way (Fig. 1C). The ortho photos were analysed by using GIS and an image classification analysis which made it possible to distinguish between presence and absence of mussels in squares (pixels) of 0.16 m^2 (Fig. 1C). Ground sampling is necessary to distinguish between living mussels and empty mussel shells. Mussels have been collected by using an iron frame (0.5 * 0.5 m^2) pressed over the mussels in the bed (see Fig. 2) or by collecting the mussels by dredging (commercial mussel dredge). The samples were frozen and analysed in the laboratory to estimate the size distribution and individual weights. From the stratified samples the abundance and biomass of mussels per m^2 of mussel bed were estimated.

Since 1990, a swept area sampling technique has been applied. The dredge samples taken by a commercial dredger (catch efficiency of 100 %) have been used to estimate the mussel biomass in the subtidal beds. Sampling stations have been randomly distributed in the fishing area covering between 10 and 12 km^2 respectively in Grey Deep south and north of Esbjerg Harbour (Munch-Petersen & Kristensen 2001). The tracks are of variable length and cover swept areas between 4 – 1,777 m^2 (Kristensen 1995 and 1996; Kristensen & Pihl 1998, 2001 and 2003). An example of dredge tracks and frame samplings in Ho Bight in the 2004 survey is shown in Figure 2. In the Ho Bight an area of 11 km^2 has been randomly dredged to determine the biomass of live mussels on 70 stations giving the mean biomass in kg pr. m^2 of bottom. Similar dredging has been performed south of Esbjerg in Grey Deep and Knude Deep within an area of 10 km^2. The used technique is the same in other Danish fishing areas for mussels (Kristensen & Hoffmann 2004; Kristensen 2001, 2002 and 2003).

In the laboratory samples were sorted into groups: whole living mussels, empty shells of mussels, and other organisms. At least 150 mussels are measured in shell length to nearest mm. The size distribution is recorded for each tidal area. The distribution by frequency is given for each tidal area. The frequency distribution of the mussels by their weight representation in the samples is estimated by transforming numbers to weight by use of the formula:

$W = 0.09076 * L^{2.973726}$

W is total wet weight in grams (shells included) and L is shell length in mm.

This estimation is carried out to establish the weight distribution of mussels in the sampled stock, where the regulation is maximum 10 % of mussels with $L \leq 50$ mm in shell length by weight. The eider ducks are assumed to cover around 60% of their food requirements by mussel meat. The oystercatchers and herring gulls cover on average 17% and 5% respectively of their food requirements by mussel meat (Gross-Custard personal communication). These figures are used in the estimations of the food consumption by birds in the Danish Wadden Sea. The annual average number of bird-days is given as the average bird- counts per day each month of the year over the years 1986 to 1999 multiplied by 365 days (Kristensen & Laursen in press) and (Laursen & Kristensen in press). The food consumption by birds is from the literature (Nehls 1995; Nehls et al. 1997; Kresten & Piersma, 1987; Kersten & Visser 1996; Speakman 1987; Zwarts et al. 1996, 1996a, 1996b and 1996c).

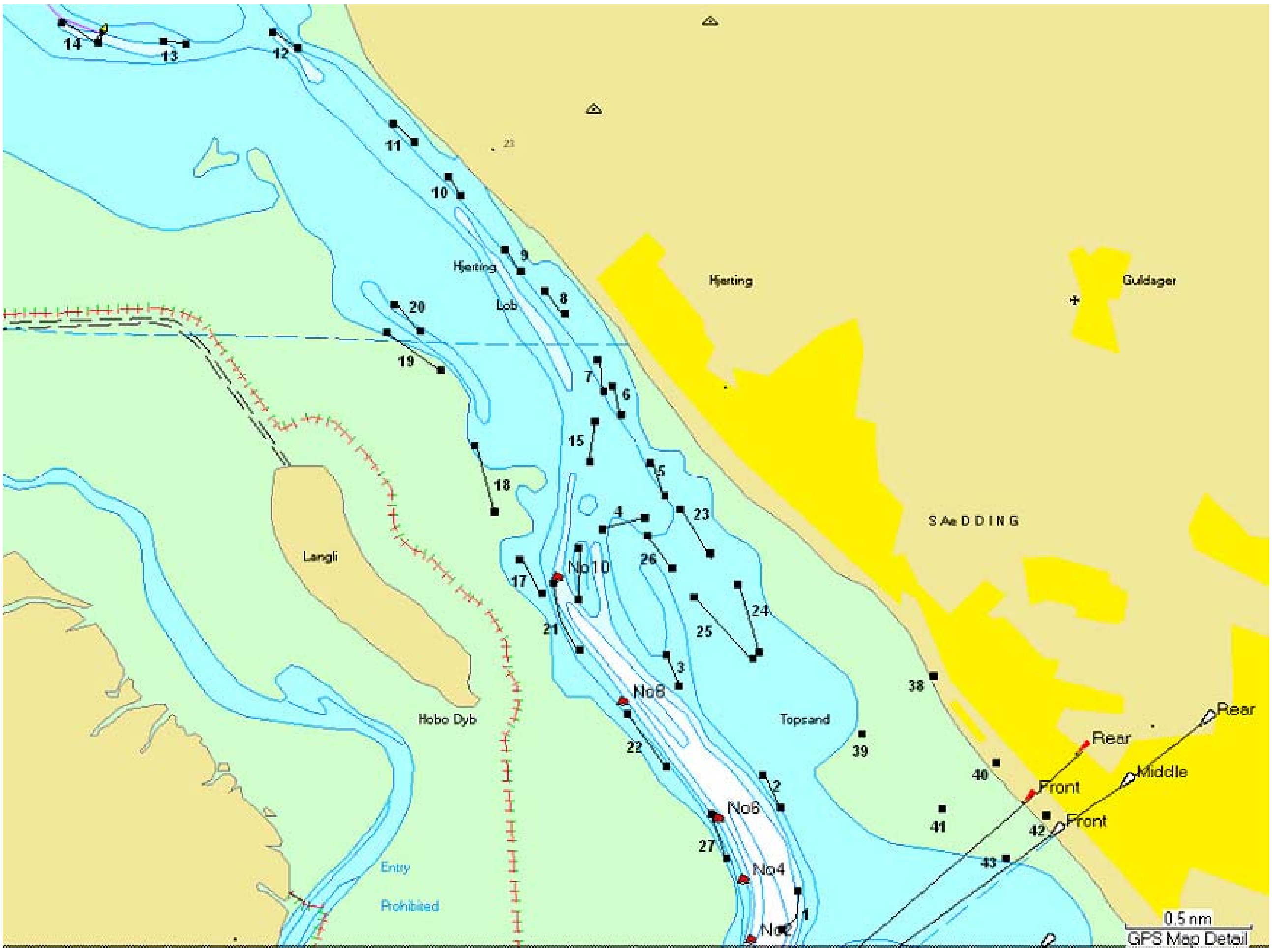


Figure 2. Frame and dredge sampling stations in the survey from September 2004 in Ho Bight. *Frame samples (No. 38-43).*

Results

The bed sizes in m² for each aerial photograph are given as figures for each tidal area (an example for Ho Bight in 2004 is shown in Tab. 1). The total areas with mussels in the Danish Wadden Sea in 2004 are shown in Table 2 (for other results please look Kristensen 1996; Pihl & Kristensen1997; Kristensen & Pihl 1998, 2001 and 2003; Kristensen et al. 2005)

There has been a decrease in the areas with mussels from 1991 to 1996 from 1,192 ha to 632 ha. In 1999 the areas were back to previous sizes of 1,051 ha (Fig. 3).

In 2004 the fishable biomass was estimated to 2,144 tons (Fig. 4). The non fishable biomass was estimated to 1,511 tons (Fig. 4) in the fishing areas. Besides these figures 2,191 tons of mussels were recorded in Juvre Deep (area closed for fishing).

Based on the estimated mean biomass an annual production is then established on basis of measurements of the average mortality and growth of mussels in the Danish Wadden Sea (Munch-Petersen & Kristensen 2001). The estimated biomass (B) in tons in October can be extrapolated to October the following year by multiplying biomass by 0.5 (taken from Munch-Petersen and Kristensen 2001) (Tab.2).

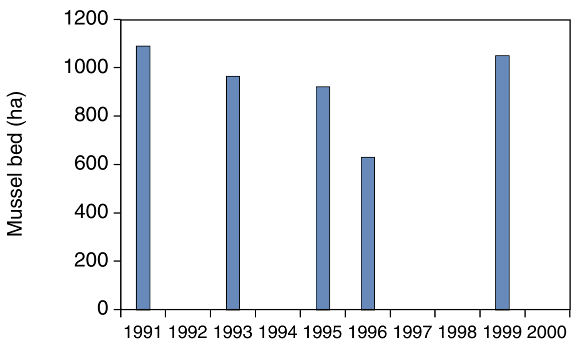


Figure 3. Intertidal and subtidal blue mussel beds (in ha) in the Danish Wadden Sea based on annual surveys in spring.

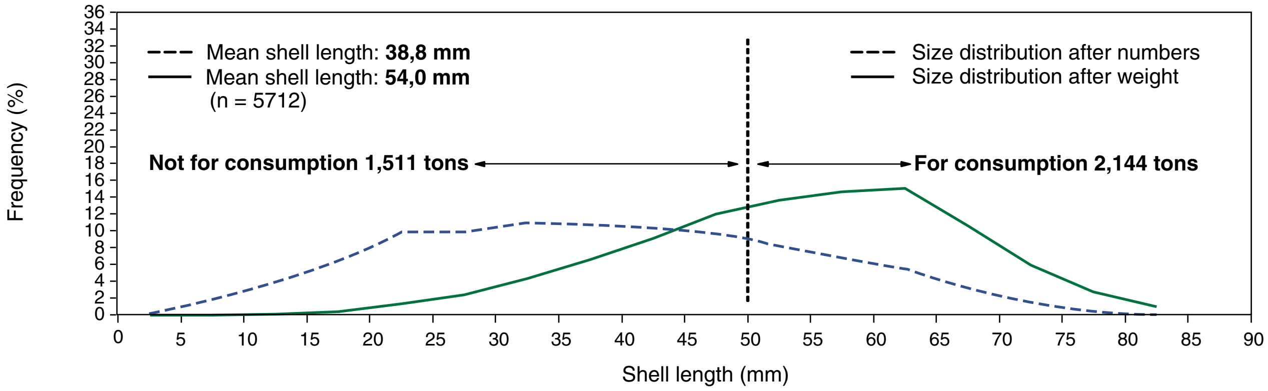
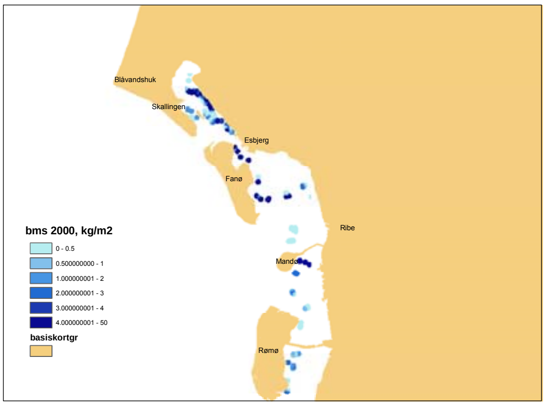


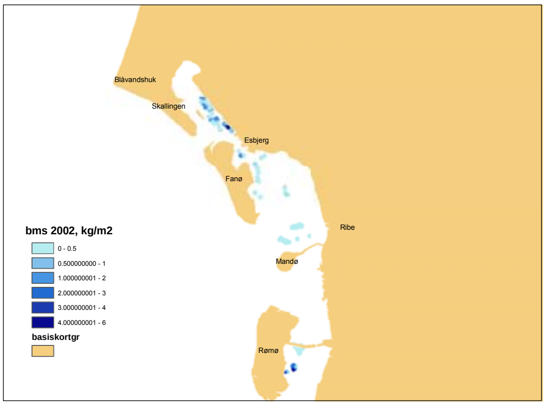
Figure 4. Size distribution of mussels in the Danish Wadden Sea in 2004 by numbers in samples, and by their weight representation in samples.

Table 1. The area (in km^2) and biomass (in kg/m^2) estimations of blue mussels in Ho Bight in 2004. *Both frame and dredge samples are given in the table.*

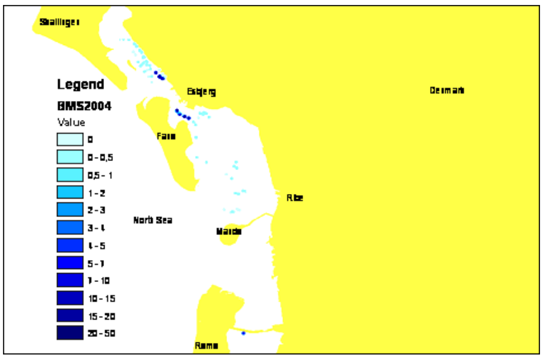
Estimated biomass of common mussels in Ho Bight in September 2004:								**Total mean biomass tons**
	st nr.	Catch in kg	Sample method	Swept area in m^2	Mussels kg/m^2			
	1	0,000	Dregde	660,60	0,00			
	2	4,137	Dredge	504,00	0,01			
	3	0,000	Dredge	462,60	0,00			
	4	25,074	Dredge	577,80	0,04			
	5	4,363	Dredge	482,40	0,01			
	6	93,067	Dredge	401,40	0,23			
	7	60,373	Dredge	448,20	0,13			
Area	8	27,512	Dredge	430,20	0,06			
in km^2:	9	12,282	Dredge	426,60	0,03			
11	10	89,658	Dredge	295,20	0,30			
	11	85,849	Dredge	401,40	0,21			
all samples are dredged	12	13,067	Dredge	421,20	0,03			
where dredging takes place	13	37,751	Dredge	300,60	0,13			
in areas both with and without	14	35,466	Dredge	464,40	0,08			
mussels giving a low biomass	15	45,313	Dredge	527,40	0,09			
	16	0,000	Dredge	694,80	0,00			
	17	144,041	Dredge	554,40	0,26			
	18	90,589	Dredge	959,40	0,09			
	19	0,000	Dredge	889,20	0,00			
	20	0,000	Dredge	504,00	0,00			
	21	26,263	Dredge	1002,60	0,03			
	22	48,790	Dredge	878,40	0,06			
	23	70,673	Dredge	703,80	0,10			
	24	75,174	Dredge	964,80	0,08			
	25	83,492	Dredge	1173,60	0,07			
	26	107,918	Dredge	552,60	0,20			
	27	0,000	Dredge	615,60	0,00			
	28	26,471	Dredge	23,58	1,12			
	29	0,000	Dredge	648,00	0,00			
	30	0,000	Dredge	455,40	0,00			
						Mean. biom. (kg/m^2):	**Standard error.**	
						0,112	0,078	
						Mean. biom.	1.232	1.232
Area	Mussels in the littoral beds september 2004 (frame samples)					max:	2.087	
in km^2:	(Note area surveyed applying GIS estimation (counting of pixels))					min:	377	
0,032706	0-samples are therefor omitted in the estimations							
	38	0,000	Frame	0,25				
Only the densly populated	39	3,416	Frame	0,25	13,66			
mussel beds are detected	40	0,000	Frame	0,25		Mean. biom. (kg/m^2):	**Standard error.**	
0-samples are omitted!	41	4,352	Frame	0,25	17,41			
	42	0,000	Frame	0,25		**15,018**	2,934	
	43	3,496	Frame	0,25	13,98	**Mean. biom.**	491	491
						max:	587	
						min:	395	
						All mean:		1.723
						max:	2.674	
						min:	772	



A: Mean biomass 49,107 tons



B: Mean biomass 16,601 tons



C: Mean biomass 5,840 tons

Figure 5. The development in the biomass of mussels in the Danish Wadden Sea in 2001 (A), 2002 (B), and 2004 (C).

The changes in the total biomass of mussels in the Danish Wadden Sea are shown in Figure 5A, 5B and 5C. There is a dramatic decrease from around 50,000 tons in 2000 (Fig. 5A) to only around 6,000 tons in 2004 (Fig. 5C).

Tabel 2. The total area (in km^2) monitored for mussels and the total estimated biomass of mussels alive (in tons) in the Danish Wadden Sea in September 2004. *The management related consequences for the fishery are mentioned.*

Year 2004	
Survey data:	
Swept area (dredging)	21,0 km^2
Aerial photographs (frame)	0,3 km^2
Annual production of mussels*	0,5
Biomass estimations:	
Average biomass	0,27 kg/m^2
Total biomass	5,840 tons
Biomass production	2,920 tons
Error of estimations	35%
Management planning:	
Quota for the fishery (season 2004-2005)	0 tons
Food for birds (one season)	8,760 tons

*From Munch-Petersen & Kristensen 2001

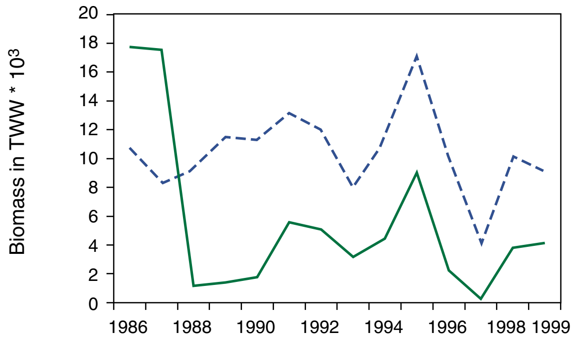


Figure 6. The removal pattern of mussels in total wet weight in the Danish Wadden Sea by birds (dashed line) and fishery (solid line) between 1986 and 1999.

Subtracting the estimated biomass year one from the extrapolated biomass year two gives a figure for the production (P) of mussels over one year. This production is allocated to birds and the fishery according to official management policy (Ministry of Food Agriculture and Fisheries). The estimated minimum ration for birds is at least 10,300 tons total wet weight, corresponding to 463 tons of AFDW of mussel meat, which on average can sustain at least 17 million bird days per year (Kristensen & Laursen in prep).

Since 1988 the management policy based on the survey and assessment of the mussel stock has resulted in a clear change in the exploitation and elimination of the mussels in the Danish Wadden Sea from a much higher exploitation level by the birds compared to the fishery as before 1987, where it was the other way around (Fig 6). The exploitation by fishery decreased substantially in 1986/1987 and has since varied between 38 tons in 2004 to 9,481 tons in 1995 (Fig. 7).

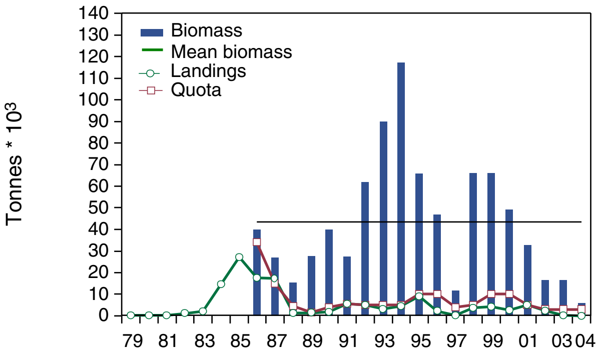


Figure 7. The development in the mussel biomass, landings, and fishing quota from the Danish Wadden Sea (1979-2004).

At the same time the consumption by birds has increased from 4,000 to 17,100 tons (Fig. 6). Since 2003 landings have been sporadic and lower than 260 tons annually. Due to the very low standing stock in the 2004 assessment and accordingly a very low production potential, the quota for the fishery in 2005 was set to zero.

Discussion

The main reason for monitoring and estimating the mussel stock in the Danish Wadden Sea has been to advise the Danish Ministry of Food Agriculture and Fisheries and the Ministry of the Environment on the total allowable catch and quotas for the fishery. At the same time the purpose was to take into account the sustainability of the mussel stock and to make sure that a sufficient food supply for bird species was not jeopardized by the fishery in the Danish Wadden Sea.

The biomass of mussels estimated at a certain time of the year (September - October) will produce a certain amount of mussels in coming years. The production estimate is based on knowledge of the growth and mortality of mussels in the Wadden Sea (Munch-Petersen & Kristensen 2001). Birds are ensured a minimum level of food and have been allocated at least 50% of the estimated production and not less than 10,300 tons per year during the last 15 years (Kristensen & Laursen in prep).

After 1990 the quota for the fishery has been between 2,000 and 10,000 tons. Between 1990 and 2004 the fishery has in average landed ca. 3,300 tons annually (between 38 and 8,931 tons) equal to around 6% of the estimated mean biomass. The exploitation has been higher in periods but very seldom over 10%. The data material on the stock assessment is delivered to the Directory of Fisheries and Nature and Forest Agency by the Danish Institute of Fisheries Research. In the last couple of years the landings have been very low and below 260 tons annually.

In 1990 the number of licenses was reduced from forty to five. In addition, in 1992 48% of the Danish Wadden Sea was closed for mussel fishing. The fishery was subjected to obey a new regulation law in which an annual, daily and weekly quota was set for the fishery negotiated between the Ministry of the Environment (The Forest and Nature Agency) and The Ministry for Food, Agriculture and Fisheries (Directory of Fisheries).

Similar tasks are also important in the other Wadden Sea countries as The Netherlands and Germany. However the mussels in the Dutch and the German Wadden Sea are primarily cultured, in which newly settled seed mussels are transplanted from the settling beds to culture beds. The transplantation has the purpose to improve production, growth and meat yield in the cultured mussels.

The program and management plans for the mussel stocks in the Danish Wadden Sea follow monitoring and recording tasks similar to the programs performed in the Dutch and the German Wadden Sea. There are only minor differences between the monitoring programmes between the three Wadden Sea countries. Aerial photographing is the same although the scaling varies between the countries. These variations are discussed and adapted in the Trilateral Monitoring and Assessment Program group at annual meetings (Marencic 2001). There are also minor differences in the field survey programs between the three Wadden Sea countries.

However, the largest step forward in the techniques used in monitoring of mussels in the Wadden Sea, was made by digitising the aerial photos to ortho photos to make them much more applicable for estimating the sizes of the mussel beds. However, it is important to stress that the most reliable results by using digitised aerial photos can only be achieved on intertidal beds. Measuring bed sizes in the subtidal beds by using the same technique depends on the tidal situation (low tide only), the position of the sun (the angle with the surface of the earth), reflection from the sea surface and cloud formations etc. Fortunately the Danish Wadden Sea is relatively shallow with a maximum water depth < 5 meters at low tide in the main parts, which makes it possible to interpret the sizes of the subtidal mussel beds in large parts of the Danish Wadden Sea using aerial photos. The digitising technique now makes it possible to zoom in and out in a scale, making it easier to draw a more precise line around the bed with mussels, excluding the application of cover percentages. Satellite images have been considered in the Danish monitoring program for mussels in the Wadden Sea for producing images closer to the sampling time in the field. However the cost, resolution and other problems have so far postponed the use of that technique (Munch-Petersen & Kristensen 1989).

To have the best reliable measurement of the mussel bed sizes it is important to employ in situ field sampling in combination with aerial photographs. This is especially important if an objective is to monitor the annual and seasonal changes in the mussel beds. In this study the estimation of the biomass of the standing stock and annual production is the most essential objective. In that context, only the relative bed sizes were important in the estimation of the total biomass of mussels in The Danish Wadden Sea combined with the measured biomass of live mussels per square meter of mussel bed based on in situ samplings.

Aerial photography and sampling in the field are easy to conduct on intertidal mussel beds. Monitoring of mussels in the subtidal beds demands other methods. In the Danish Wadden Sea we have used dredging on random subset of sampling stations within stable areas.

It is essential that the monitoring and assessment programs for the bivalve species in the Wadden Sea are coordinated to enable comparison of changes and developments in these stocks through the whole of the Wadden Sea, which is one of the most important marine wetlands in Europe. Coordinated assessment facilitates the exploitation of mussels in the Danish Wadden Sea by humans and the bird populations. Many bird species depend on the Wadden Sea as resting and feeding areas either permanently as residents or as migrants from their winter residence to their summer residence and visa versa. The declines in the bivalve stocks of cockles and mussels have had great importance for birds especially in the Dutch Wadden Sea, where high mortality rates have been observed among birds especially eiders (Ens et al., 2004). There has been less mortality of birds in the German and the Danish Wadden Sea. The exploitation level of cockles and mussels has in many years been higher in the Dutch Wadden Sea compared to the German and the Danish Wadden Sea (Dijkema 1997 and Seaman & Ruth, 1997). During the last 15 years the exploitation level of mussels in The Danish Wadden Sea has been limited to the production which the monitored standing stock was capable of producing over the coming year. This allocation rule has been applied to safeguard sufficient food supply for the bird species depended of mussels and cockles as food. No such estimations are used in The Netherlands or in Germany. This is probably due to the different exploitation forms in Germany and in The Netherlands, where the new settled blue mussels spat are transplanted and cultured on culture lots. In Denmark fishery is only allowed on the natural beds and culture is prohibited.

Some concern can be expressed for the future, if one looks at the present situation with the lowest biomass observed in the last 20 years in spite of the strict management program, which has been applied during the last 15 years in the Danish Wadden Sea. The lack of recruitment and scanty spat fall the last years in the whole of the Wadden Sea can result in lower biomasses of mussels and cockles and accordingly less food for birds and less mussels and cockles to fish. Knowledge on stock developments and factors affecting production and survival of mussels will in the future be more important than previously.

As in the other Wadden Sea countries, the Danish Wadden Sea is a Wildlife Reserve and appointed as a Ramsar area, Bird protection area under the EU-Bird Directive, and a NATURA-2000 area under the EU-Habitat Directive. A strong management of the exploitation of fishable stocks, which have been implemented the last fifteen years in the Danish Wadden Sea, is therefore essential for all the involved authorities. The new techniques applied may improve the assessment and reduce variations and uncertainties on estimates of stock size and annual production rates. These parameters are used to advice the authorities on TAC and quotas for the fishery.

As the question raised in the title implies, there may be problems fishing mussels in a nature Wildlife Reserve such as the Danish Wadden Sea. It is however the authors opinion, that it is possible to fish mussels in a Wildlife Reserve if the fishery is conducted at a level which does not affect interaction with other species or the eco-system.

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Foreland development along the Advanced Seawall at Højer, the Danish Wadden Sea

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Vestergaard, P. 2006: Foreland development along the Advanced Seawall at Højer, the Danish Wadden Sea. In: Monitoring and Assessment in the Wadden Sea. Proceedings from the 11. Scientific Wadden Sea Symposium, Esbjerg, Denmark, 4.-8. April, 2005 (Laursen, K. Ed.). NERI Technical Report No. 573, pp. 123-132.

As part of the advanced seawall at Højer, built in 1979-1981, a marsh foreland, 150 m wide and sloping from +2.45 m DNN to +0.20-0.30 m DNN, was designed. The aim was to provide protection of the seawall and to create a green environment as a replacement of the salt marsh areas, which would be lost between this and the old seawall. The foreland was founded by marine sand, and the innermost part was sown with grasses. To enhance sedimentation, a coherent row of sedimentation fields was established along the foreland in 1986-1988. The development of the foreland has been monitored with irregular intervals since 1981. In this paper I present observations of changes in profile and vegetation, and I compare the state of the foreland in 2004 with mature marsh forelands.

The main trends observed so far have been: 1) the species richness of vascular plants has increased, 2) Glasswort *(Salicornia)* and Common Cord-Grass *(Spartina)* have established widely on tidal flats in the sedimentation fields, 3) At the inner part of the tidal flat a Common Salt-Marsh-Grass *(Puccinellia maritima)* salt marsh has gradually established, 4) the foreland landwards to the tidal flat has currently been narrowed by erosion, but has also been influenced by sand accretion, 5) the vegetation of the outer part of the foreland is still open and characterized by beach and dune species, 6) the vegetation of the inner part of the foreland is slowly developing towards a typical Wadden Sea high marsh.

In conclusion, the planned foreland has not yet been achieved after 23 years, and the development of the foreland has differed somewhat from what was originally expected. This is probably caused by the circumstance that the advanced seawall was built on a relatively low-lying tidal flat. The vegetation zonation developed so far indicates, however, that a typical marsh foreland may be formed along the advanced seawall during the coming decades on the basis of the technical support carried out until now and a continuation of the current management of the foreland. Establishment of extra brushwood groynes in the existing sedimentation fields may, however, speed up the marsh formation.

Key words: Foreland, geomorphology, management, monitoring, salt marsh, tidal flat, vegetation

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Introduction

After several flood incidents in 1976, the Danish Government decided in 1977 to protect the existing coastline at the old Højer seawall behind a new, advanced, 8.6 km long seawall starting from the Danish-German border in the south to Emmerlev in the north. As a part of the seawall, it was decided to create a green, 150 m broad foreland like the marsh forelands, which typically are developed elsewhere in the Wadden Sea (e.g. Raabe 1981). The aim of this foreland should be partly to protect the new seawall, partly to replace the former marsh foreland, the Ny Frederikskog Forland, which would be lost behind the advanced seawall. The advanced seawall and foreland were built in 1979-1981. Behind the seawall, a saltwater lake was established in the Margrethekog, (Fig. 1).

In 1979, the Danish Scientific Research Council initiated a project, aiming to monitor the vegetational development of the new foreland. The project was carried out in 1981-2004. The investigation has included the total foreland, but since 1989 especially the foreland north of the Vidå Sluice has been in focus.

The objectives of the present paper have been, based on results from the northern part of the foreland, to discuss the development up to now of the morphology and vegetation of the foreland, partly in relation to the original foreland design, partly in relation to typical, mature marsh forelands elsewhere at the Wadden Sea.

The foreland: construction and management

According to the project plan, the foreland was founded by marine sand and established with a gently sloping surface from the original tidal flat level at +0.20-0.30 m DNN (Danish Ordnance Datum) to the dike at +2.45 m DNN (Fig. 2). The sand was supplied from a pit on the tidal flat, parallel to the dike in a distance of 800 m. In order to create the basis of a coherent plant cover, the innermost 50 m of the foreland in 1980 and 1981 was fertilized and sown with a mixture of grasses: Red Fescue *(Festuca rubra)* 68%, Tall Fescue *(F. arundinacea)* 8%, Sheep's Fescue *(F. ovina)* 12% and Italian Rye-grass *(Lolium multiflorum)* 12%. Due to erosion and sand accretion in the following winters, it was necessary to supply the sowing in 1982 and 1983. Since 1981, the current management of the foreland has included seaward bulldozing of sand, accreted during winter, additional grass sowing until 1992, grazing by sheep (at the northern part of the foreland until 1990) and occasional mowing (Svend Petersen and Aksel Pedersen, pers. comm., Vestergaard 1997).

In order to enhance sedimentation on the tidal flat and thereby to enhance stabilisation of the foreland, a coherent row of sedimentation fields, about 200 x 200 m, was established in 1986-1988 (Fig. 3). After the establishment, the sedimentation fields were ditched (Danish: *grøblet*) for several years in order to drain and thereby to improve conditions for terrestrial vegetation to establish.

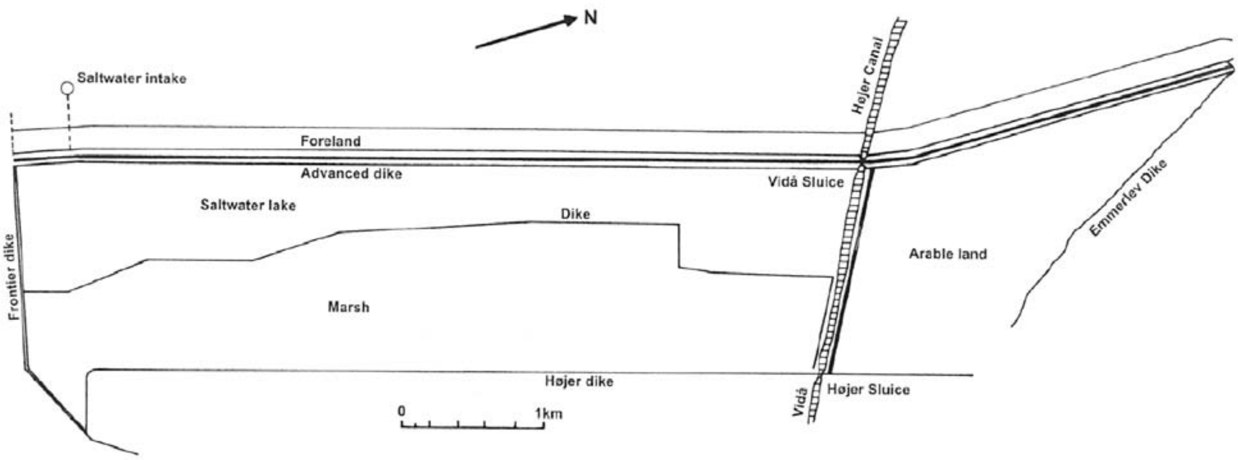

Figure 1. The Margrethekog wit h the advanced seawall and foreland. After Vestergaard 1984.

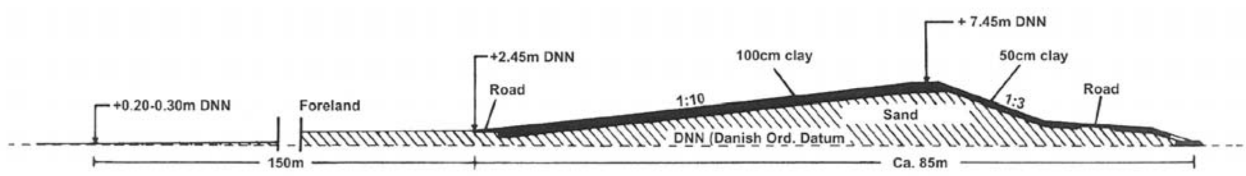

Figure 2. Schematic cross-section of the advanced seawall with the planned, 150 m broad foreland. After Jespersen & Rasmussen (1984).

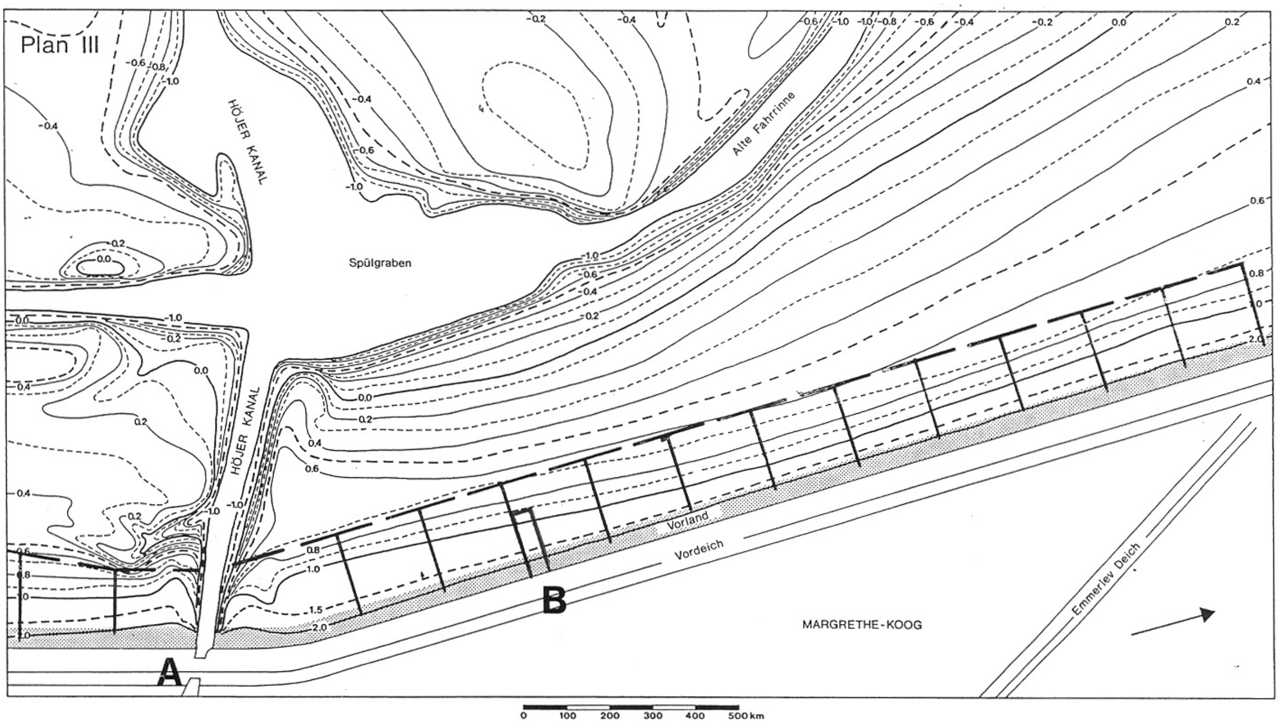

Figure 3. Contour map of the tidal flat and foreland north of the Vidå Sluice 1983/1984, with the sedimentation fields, established in 1988, inserted. A: The Vidå Sluice. B: Permanent study area used in the present study. After Jespersen & Rasmussen 1989.

Methods

The botanical investigations in the northern part of the foreland, 2.5 km, cover the period 1981-2004 with irregular intervals. Based on the morphology, the elevation above DNN and the character of the plant cover, the foreland was divided into coast-parallel zones, A, B, C, D, D´, explained in Figure 5. The width of the individual zones varied from year to year. In each zone vascular plant species were recorded (nomenclature according to Hansen 1981), and their abundance were estimated, using a 1-3 abundance scale: 1: scattered, 2: common, 3: very common to dominating.

About 800 m north of the Vidå Sluice, a representative, permanent study area, 50 x 120 m, was established (Fig. 3). Each monitoring year, the structure of the study area was mapped, based on the above-mentioned coast-parallel zones. In each zone vascular plant species were recorded, and the total plant cover was estimated. A fixed transect line, perpendicular to the dike line, was levelled for every 5th m in relation to the elevation of the road at the base of the seawall (Fig. 2). The levelling data were expressed in relation to DNN.

In this paper, I present the main trends in the development of morphology and vegetation of the foreland, based on results from selected years.

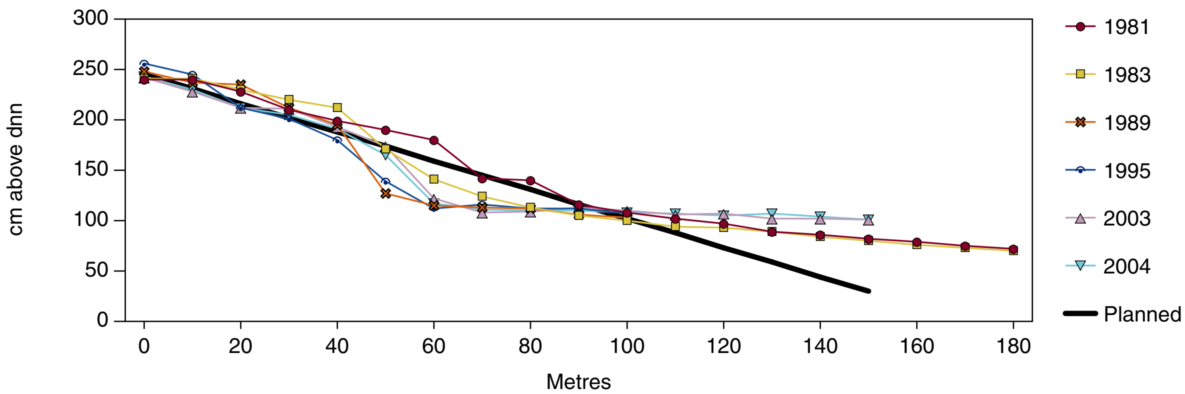

Figure 4. Change in profile of foreland and tidal flat 1981-2004, levelled along a transect line, seaward from the dike (0 m), in the permanent study area. The originally planned profile is also shown.

Results

The permanent study area: changes in profile

When compared with the planned profile, the morphology of the study area was strongly modified during the study period due to varying spatial and temporal influence of erosion, accretion and sedimentation processes (Fig. 4). Already in 1981, the study area could be divided into an upper, terrestrial *foreland* from 0-80 m, with a slope not very different from the planned slope, and a *tidal flat* below +1.40 m DNN. Apart from the innermost rather steep 10 m, the slope of the tidal flat was less steep than the planned slope. This main structure was largely retained throughout the study period, and so was the +1.40 m DNN level as being the vertical limit between foreland and tidal flat. But due to erosion, the inner border of the tidal flat gradually mowed landwards at the expense of the foreland.

Since the establishment of the sedimentation fields in 1988, the level of the tidal flat has raised several decimetres, and the width of the foreland has been rather constant, i.e. about 50 m, which corresponds to the initial grass sowing area. The micro relief of the foreland has, however, varied from year to year due to accretion of sand during winter storms, and subsequent seaward bulldozing of the sand. This has inhibited the establishment of a coherent, permanent plant cover, especially in the outer part of the foreland.

The permanent study area: changes in plant cover

The development of plant cover in the study area has reflected the changing profile and surface morphology. On the foreland, two parts could be defined: An inner, more stable part (zone A) with a more or less coherent plant cover, and an outer, less stable part (zone B+C), with a scattered plant cover.

In 1981, the plant cover at the inner part of the foreland was dominated by two of the sown species, Red Fescue and Italian Rye-grass, but included also scattered individuals of a few halophytic species, e.g. Halberd-leaved Orache *(Atriplex prostrata)*, Glasswort *(Salicornia* sp.*)* and Sea Spurrey *(Spergularia salina)*. During the following years this zone gradually narrowed (Fig. 5), and the species composition changed. In 1989, the number of species had increased to 19 with Red Fescue and Tall Fescue as dominants. In 2004, zone A was restricted to a strip, less than ten metres broad, between the road and a low bank of sand.

In 1981 the outer part of the foreland stretched from 50 to 80 m. The plant cover consisted of scattered individuals of halophytes. In 1989, this part of the foreland was replaced by tidal flat, and the seaward part of the former zone A had changed into outer foreland, zone B+C with a sparse plant cover. In 2004, the number of species in this zone had increased to 33 species, of which especially beach and sand dune species dominated.

On the tidal flat, the landward expansion and gradual raise of the level was accompanied by strongly increasing abundance of Glasswort and Common Cord-Grass *(Spartina anglica)*, especially after the establishment of the sedimentation fields. On the inner part of the tidal flat also Common Salt-Marsh-Grass *(Puccinellia maritima)* and Sea Aster *(Aster tripolium)* expanded down to about +1.10 m DNN.

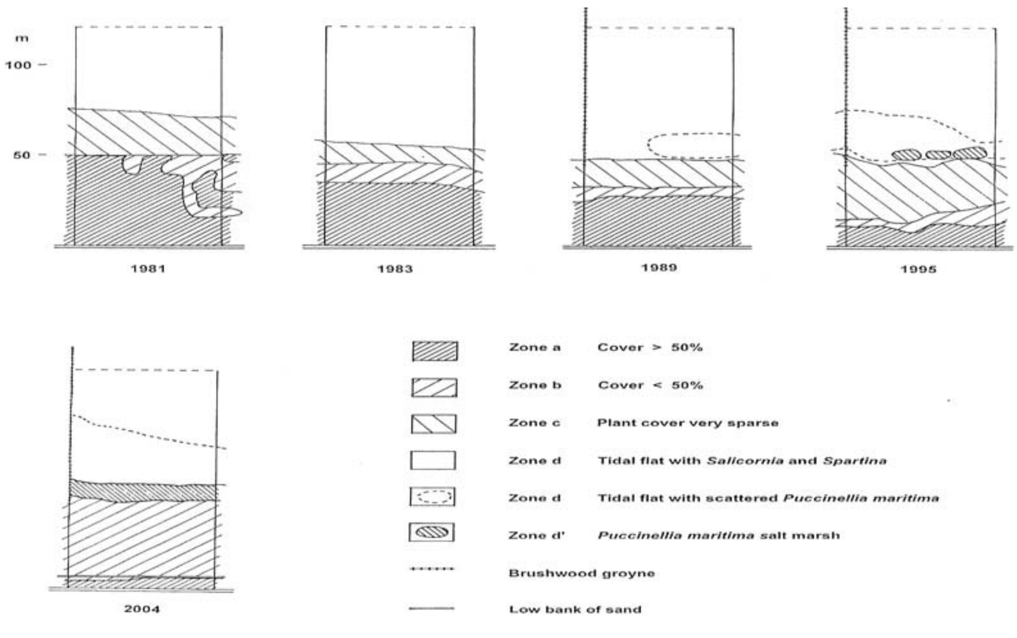

Figure 5. Change in vegetation zones in the permanent study area 1981-2004.

In 1995, scattered "islands" of Common Salt-Marsh-Grass marsh had developed on the innermost 10 m of the tidal flat within the vertical interval of about +1.15-1.40 m DNN (Fig. 6). In 2004 this belt had closed into a coherent salt-marsh community.

Figure 6. Salt marsh (Common Salt-Marsh-Grass, *(Puccinellia maritima)* developing on the inner part of the tidal flat in the permanent study area in 1995.

Changes in species richness and species abundance in foreland and tidal flat

The structure and vegetation of the entire foreland and tidal flat north of the Vidå Sluice largely repeat the structure of the permanent study area. Theoretically, only the four sown grass species were present on the foreland in 1980. The species richness increased, however, very fast (Fig. 7). Already in 1981, the number of species had increased to 33, and in 1995, to 73. The increase in number of species was mostly due to species that immigrated into the foreland. About half of these species are exclusively found in coastal habitats. The remaining species are mostly from arable land and other open inland habitats.

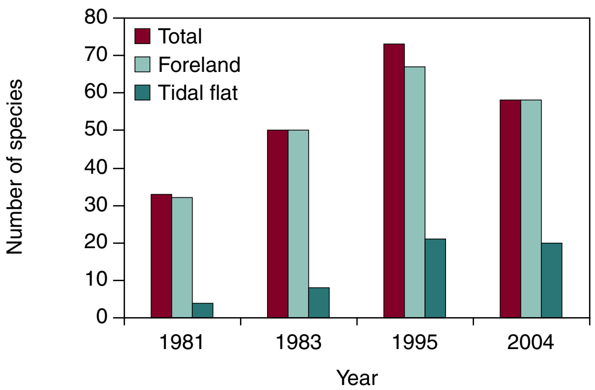

Figure 7. Change in species richness of foreland, tidal flat incl. the salt marsh, and in total, 1981-2004.

In the tidal flat, the increase in species number was mostly due to species, inhabiting the Common Salt-Marsh-Grass salt-marsh belt that gradually developed in the inner part.

The relative proportions of high and low abundance species changed over time (Fig. 8).

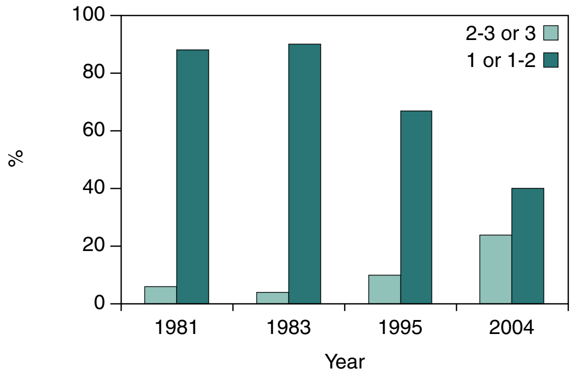

Figure 8. Changes in the proportion of high abundance species (abundance 2-3 or 3) and low abundance species (abundance 1 or 1-2) on foreland and tidal flat incl. salt marsh, 1981-2004.

In the first years after the establishment of the foreland, only the sown grass species were high-abundant (Table 1).

Table 1. Species with high abundance (2-3 or 3) in one or several zones in different years.

		1981	1983	1995	2004
Red Fescue	*Festuca rubra*	x	x		x
Italian Rye-grass	*Lolium multiflorum*	x	x		
Tall Fescue	*Festuca arundinacea*			x	
Saltmarsh Rush	*Juncus gerardi*			x	
Creeping Bent	*Agrostis stolonifera*			x	x
Sea Couch x Couch	*Elytrigia pungens x repens*			x	x
Spiny Rest-Harrow	*Ononis spinosa*			x	x
Silverweed	*Potentilla anserina*			x	x
Common Salt-Marsh-Grass	*Puccinellia maritima*			x	x
Strawberry Clover	*Trifolium fragiferum*			x	x
Sea Aster	*Aster tripolium*				x
Hybrid Sea-Couch	*Elytrigia junceiforme x pungens*				x
Couch	*Elytrigia repens*				x
Perennial Rye-grass	*Lolium perenne*				x
Black Medick	*Medicago lupulina*				x
Glasswort	*Salicornia* sp.				x
Common Cord-Grass	*Spartina anglica*				x
White Clover	*Trifolium repens*				x

The non-sown species were dependent on accidental dispersal from the surroundings. A large majority of these species occurred very scattered and largely independent of the environment. In course of the development, however, the abundance of an increasing proportion of the species increased due to generative and vegetative, internal dispersal and formation of stands in specific environments. This indicates an increasing competition between the species and an incipient formation of plant communities.

Vegetation zonation in foreland and tidal flat in 2004

In 2004, a zonation with four zones had developed: 1. Tidal flat, 2. Salt marsh, 3. Outer part of foreland, 4. Inner part of foreland (Table 2).

The tidal flat, below about +1.15 m DNN, was especially characterized by Glasswort and Common Cord-Grass, which were present everywhere in the sedimentation fields, Cord-Grass mostly forming smaller and larger clones.

In the salt marsh belt, about +1.15-1.40 m DNN, twenty species were present. Besides Common Salt-Marsh-Grass, also Sea Aster and Red Fescue could dominate (Table 2), and from 1995, Sea Purslane *(Halimione portulacoides = Atriplex portulacoides)* was present. Towards the northern part of the foreland, where the seawall approaches the old coastline, also Sea Club-Rush *(Scirpus maritimus)* and Common Reed *(Phragmites australis)* were abundant, probably due to fresh water influence. The surface of the salt marsh areas was about 20-30 cm above the surrounding sand flat, and the uppermost 5-10 cm of the sandy soil contained some organic matter (2-4%) (Vestergaard 1997), probably originating from sedimentation at high tide.

Table 2. Species composition in yr 2004 of tidal flat, salt marsh belt, outer and inner foreland.
Abundance scale: x: scattered; xx: common; xxx: very common-dominating.

		Tidal flat	Salt marsh	Outer foreland	Inner foreland
Yarrow	*Achillea millefolium*			x	xx
Creeping Bent	*Agrostis stolonifera*		xx	xxx	xxx
Sea Wormwood	*Artemisia maritima*			x	
Mugwort	*A. vulgaris*			xx	x
Sea Aster	*Aster tripolium*	x	xx	x	x
Grass-leaved Orache	*Atriplex littoralis*			xx	
Halberd-leaved Orache	*A. prostrata*		x	xx	x
Sea Rocket	*Cakile maritima*			x	
Saltmarsh Serge	*Carex extensa*		x		
Little Mouse-ear	*Cerastium semidecandrum*				x
Creeping Thistle	*Cirsium arvense*			xx	xx
Wild Carrot	*Daucus carota*			x	
Sand Couch	*Elytrigia junceiforme*			xx	
Hybrid Sea-Couch	*E. junceiforme x pungens*			xxx	
Sea Couch	*E. pungens*			x	xx
Sea Couch x Couch	*E. pungens x repens*		x	xxx	xx
Couch	*E. repens*			xx	xxx
Tall Fescue	*Festuca arundinacea*			xx	x
Red Fescue	*F. rubra*		xx	xxx	xxx
Sea Milkwort	*Glaux maritima*		xx	x	
Sea Purslane	*Halimione portulacoides*		x	x	
Sea Sandwort	*Honckenya peploides*			xx	x
Saltmarsh Rush	*Juncus gerardi*		xx		x
Sea Pea	*Lathyrus japonicus*			x	
Autumn Hawkbit	*Leontodon autumnalis*			x	xx
Lyme Grass	*Leymus arenarius*			x	x
Common Toadflax	*Linaria vulgaris*			x	x
Perennial Ryegrass	*Lolium perenne*				xxx
Common Birdsfoot Trefoil	*Lotus corniculatus*			xx	xx
Black Medick	*Medicago lupulina*			xx	xxx
Red Bartsia	*Odontites verna*				xx
Spiny Rest-Harrow	*Ononis spinosa*			xx	xx
Common Reed	*Phragmites australis*		x	x	
Buckshorn Plantain	*Plantago coronopus*				xx
Ribwort Plantain	*P. lanceolata*			x	xx
Ratstail Plantain	*P. major*			x	xx
Sea Plantain	*P. maritima*		xx	xx	xx
Annual Meadow-Grass	*Poa annua*				x

Table is continued on next page

		Tidal flat	Salt marsh	Outer foreland	Inner foreland
Smooth Meadow-Grass	*P. pratensis*				x
Knotgrass	*Polygonum aviculare*			x	xx
Silverweed	*Potentilla anserina*			xx	xxx
Common Salt-Marsh-Grass	*Puccinellia maritima*	x	xxx	x	
Curled Dock	*Rumex crispus*			xx	
Knotted Pearlwort	*Sagina nodosa*				xx
Glasswort	*Salicornia sp.*	xxx	xx		
Sticky Groundsel	*Senecio viscosus*			x	
Groundsel	*S. vulgaris*			x	x
Sea Club-Rush	*Scirpus maritimus*		xx	x	x
Bladder Campion	*Silene cucubalus*			x	x
Corn Sow-Thistle	*Sonchus arvensis*			xx	xx
Sea Spurrey	*Spergularia marina*	x	x	*x*	
Greater Sea Spurrey	*S. media*		xx		
Common Seablite	*Suaeda maritima*		xx	x	
Tansy	*Tanacetum vulgare*			xx	x
Dandelion	*Taraxacum sp.*			xx	x
Strawberry Clover	*Trifolium fragiferum*		x	xx	xxx
Red Clover	*T. pratense*			x	x
White Clover	*T. repens*			xx	xxx
Sea Arrow-Grass	*Triglochin maritimum*		xx		
Scentless Mayweed	*Tripleurospermum inodorum*			x	x
Coltsfoot	*Tussilago farfara*			x	x
Tufted Vetch	*Vicia cracca*			x	x

At the foreland, between +1.40 and +2.45 m DNN, the development of vegetation has been less directional than in the lower zones. In the outer part, 50 species were present in 2004. The total cover was mostly low, < 50%, and the sand surface was instable. The dominants were Creeping Bent *(Agrostis stolonifera)*, Red Fescue, Sea Couch x Couch *(Elytrigia pungens x repens)* and Hybrid Sea-Couch *(E. junceiforme x pungens)*, of which especially the latter formed coherent, high swards over large areas (Fig. 9). The other species were beach and sand dune species, salt marsh species and species from dry inland habitats.

In the inner part of the foreland the plant cover was somewhat more stable and coherent, but the sandy soil was still very low in organic matter. The number of species was 43. The vegetation was dominated by more or less salt-tolerant grasses: Creeping Bent, Couch *(Elytrigia repens)*, Red Fescue, and Perennial Rye Grass *(Lolium perenne)*, but also other grasses and a couple of dicotyledons were important (Table 2).

Figure 9. Dominance of Hybrid Sea-Couch *(Elytrigia junceiforme x pungens)* in the outer part of the foreland in 2004.

Discussion and conclusion

One of the intentions of the planners of the advanced seawall was to create a 150 m green foreland in front of it. However, profile as well as vegetation of the foreland has shown a development, which differs from that expected. This is probably because the advanced dike was built on a relatively low-lying tidal flat, as pointed at by Jespersen & Rasmussen (1989). This caused already from the very beginning a strong modification of the foreland profile by erosion and accretion of sand during winter, as also observed by Jespersen & Rasmussen (1984, 1989) on the basis of levelling in 1982 and 1983 of 15 foreland profiles.

But never-the-less many traits indicate, that the development of the foreland at Højer largely follows the same direction as known from tidal flats and salt marshes elsewhere in the Wadden Sea (e.g. Jakobsen 1964, Veenstra 1980, Raabe 1981).

The lower limit of the foreland was planned at a level of +0.20-0.30 m DNN, corresponding to a distance of 150 m from the dike. However, the actual level at 150 m, measured in 1981-1983, was much higher, about +0.80 m DNN (Fig. 4, and Jespersen & Rasmussen 1984). Thorough studies by Jakobsen (1964) from the Danish Wadden Sea have shown that the tidal flat community with Glasswort *(Salicornia)* develops within the vertical interval between the mean high water level (MHW) and 20-30 cm below that level. According to calculations by Vestergaard (1984), on the basis of sea-level fluctuation data from the Vidå Sluice 1982 and 1983, the MHW level at Højer is about +1.15m DNN. It should therefore be expected, that Glasswort would invade the tidal flat from about +0.90 m DNN. And this was in fact observed already during the first years. Also according to Jakobsen (1964), salt marsh development, based on Common Salt-Marsh-Grass *(Puccinellia maritima)*, is stabilizing at about MWH. This was confirmed by the results from the Højer foreland as well, which showed that the initial marsh belt developed from +1.15 m DNN.

The colonisation of Glasswort (and Common Cord-Grass, *Spartina*) on the tidal flat proceeded very slowly during the first years of the study period. This may be ascribed to low sedimentation rate, which again may be due to partly the presence of the deep sand pit on the tidal flat 800 m from the dike, and partly to the lack of sedimentation fields until 1988. After 1988, the colonisation of Glasswort and Common Cord-Grass on the tidal flat increased, apparently due to increasing sedimentation caused by the sedimentation fields, but probably also because of gradual filling-up of the sand pit, allowing material to be sedimented on the tidal flat, which was earlier sedimented in the sand pit. So, according to measurements by Jespersen & Rasmussen (1989), the water depth in the sand pit declined from 1982 to 1989, and in 2004 the sand pit seems to be completely filled by sediment (Svend Petersen, pers. comm.).

The development up to now as well as the possible future vegetation development of the foreland at Højer, can be judged by comparing the zonation observed with salt marsh zonation studied elsewhere in the Wadden Sea. According to Joenje et al. (1976), Raabe (1981), Pedersen (1983) and Dijkema (1983), the vegetation zonation on the mostly grazed salt marshes at the Wadden Sea generally includes four vertical zones: Tidal flat with Glasswort and Common Cord-Grass; a low marsh, from about the MHW level, dominated by Common Salt-Marsh-Grass; a middle marsh, from about 30 cm above the MHW level, dominated by Red Fescue *(Festuca rubra)*, Saltmarsh Rush *(Juncus gerardi)*, Thrift *(Armeria maritima)*, Creeping Bent *(Agrostis stolonifera)*, Sea Milkwort *(Glaux maritima)* and Sea Wormwood *(Artemisia maritima)*, and a high marsh, dominated by Red Fescue, Perennial Ryegrass *(Lolium perenne)*, White Clover *(Trifolium repens)*, Tall Fescue *(Festuca arundinacea)* and Spiny Rest-Harrow *(Ononis spinosa)*.

This zonal structure is also valid for ungrazed marshes, but here Sea Purslane *(Halimione portulacoides)* is often dominant in the lower marsh, and Sea Couch *(Elytrigia pungens)* and Couch *(E. repens)* in the higher marsh (Esselink 2000, Bakker et al. 2003).

At Højer, development of three of these zones is indicated by the observations so far. The vegetation on the tidal flat is well developed. A low marsh, i.e. the Common Salt-Marsh-Grass belt, is developing; most of the typical species are present, but the area is still small. Also a high marsh is developing in the inner foreland. The above-mentioned typical species are present, but the plant cover is not coherent, the content of organic matter in the soil is low, and the species composition varies rather much, even within small distances. The relative recent appearance of Sea Purslane in the low marsh, and the role of Couch and Sea Couch in the high marsh agrees well with the, at least since 1990, ungrazed state of the foreland.

A middle marsh is, however, not yet present at the Højer foreland. Even if many of the typical middle marsh species are present, the strong dominance of Hybrid Sea-Couch *(Elytrigia junceiforme x pungens)* (Fig. 9) and the presence of sandy beach and sand dune species indicate a still very unstable environment, characterized by mobile sand. At the foreland south of the Vidå Sluice trials with artificial establishment of Common Salt-Marsh-Grass sward were carried out in 1986-1988 at the vertical level +1.70-1.80 m DNN (Jespersen & Rasmussen 1989), i.e. at a higher level than that occupied by the spontaneously developed Common Salt-Marsh-

Grass belt. Probably due to erosion, sand coverage and drought these swards, however, largely had disappeared in 1994 (Vestergaard 1997).

In conclusion, the planned marsh foreland has not yet been achieved after 23 years, and the development has until now differed from the originally expected. The development of vegetation zonation so far indicates, however, that a typical marsh foreland may be formed along the advanced seawall during the coming decades. A factor, which may delay this development, is that the accretion of sand on the foreland still demands measures like bulldozing. Reduction of this negative factor awaits the formation of a broader, protecting Common Salt-Marsh-Grass salt marsh belt in front of the foreland. To speed up this process, an increasing sedimentation rate, which probably can be achieved by establishment of extra, coast-parallel brushwood groynes in the existing sedimentation fields, would be positive.

The monitoring method used in this study, which combines a reiterate survey of a larger area on the basis of functional, non-fixed zones, and a more intensive reiterate study and mapping of a representative study area, large enough to include most of the variation in the larger area, seems to be a suitable and not very expensive method for long-term monitoring of an extensive coastal area.

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Assessment of breeding birds in SPAs in Danish Wadden Sea marshland

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Laursen, K. 2006: Assessment of breeding birds in SPAs in Danish Wadden Sea marshland. In: Monitoring and Assessment in the Wadden Sea. Proceedings from the 11. Scientific Wadden Sea Symposium, Esbjerg, Denmark, 4.-8. April, 2005 (Laursen, K. Ed.). NERI Technical Report No. 573, pp. 133-141.

Six SPAs (Special Protection Areas) have been designated according to the EU-Birds Directive in the Danish Wadden Sea marshland (marshland areas separated from the Wadden Sea by a seawall). The objective is to protect wetlands of international importance, and to manage them in a way that promote their conservation status, including bird species mentioned on Annex 1 of the EU-Birds Directive (birds species that shall be protected in the EU member states). In this study the number of breeding birds has been evaluated during 1983-2001. Species considered are those that were included in the foundation description of the SPAs when they were designated. The species listed for each SPA include both species on the Annex 1 as well as additional species that contributed to describe the characteristic bird fauna of the SPA. During 1983-2001 in total 24% of the breeding bird species on Annex 1 have increased, and so did also 21% of the additional breeding bird species mentioned. However, 53% of the bird species on Annex 1 decreased in numbers together with 46% of the additional species. A national action plan has been accepted for threatened meadow birds in important grassland areas in Denmark to stop the decreasing trend and improve conditions for breeding birds. Future monitoring will reveal if this action plan had relieved the decreasing trends.

Key words: Breeding bird, EU-Bird Directive, historical data, numbers, SPA, trend

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Introduction

The Forest and Nature Agency appointed in 1983 six SPAs (Special Protection Areas) in the Danish Wadden Sea marshland according to the EU-Birds Directives. SPAs shall be designated in each EU member state in areas of international importance, and the member states are obliged to manage the areas in a way that protect or promote their conservation status. In addition the Annex 1 of the EU Birds Directives species are mentioned that are particular threatened in the member states, and the SPAs appointed shall include one or more of these species to safeguard their status.

The six SPAs in the Danish Wadden Sea marshland are (see Fig.1):

SPA no. 51: Ribe Holme and meadows along the Ribe Å and the Kongeåen (in short Ribe).
SPA no. 52: Mandø.
SPA no. 53: Fanø.
SPA no. 60: Vidåen, Tønder Marsh and the Margrethe Koog incl. Saltvandssøen (in short Tønder Marsh).
SPA no. 65: Rømø.
SPA no. 67: Ballum and Husum Enge and the Kamper Salt meadows (in short Ballum).

These SPAs are described in the books 'EF-fuglebeskyttelsesområder' (Skov- og Naturstyrelsen 1983) and 'EF-fuglebeskyttelsesområder og Ramsarområder' (Skov- og Naturstyrelsen 1995).

For each SPA a bird species list is given for species on the Annex 1 for which the area has been designated. Besides, this list also includes additional species that are particularly numerous in the area or are scarce in number in Denmark. These additional species are also mentioned for each SPA because they contribute to characterise the bird fauna in the area. Several of such species are included on the national Red List and Yellow List (Stoltze & Pihl 1998, Stoltze 1998).

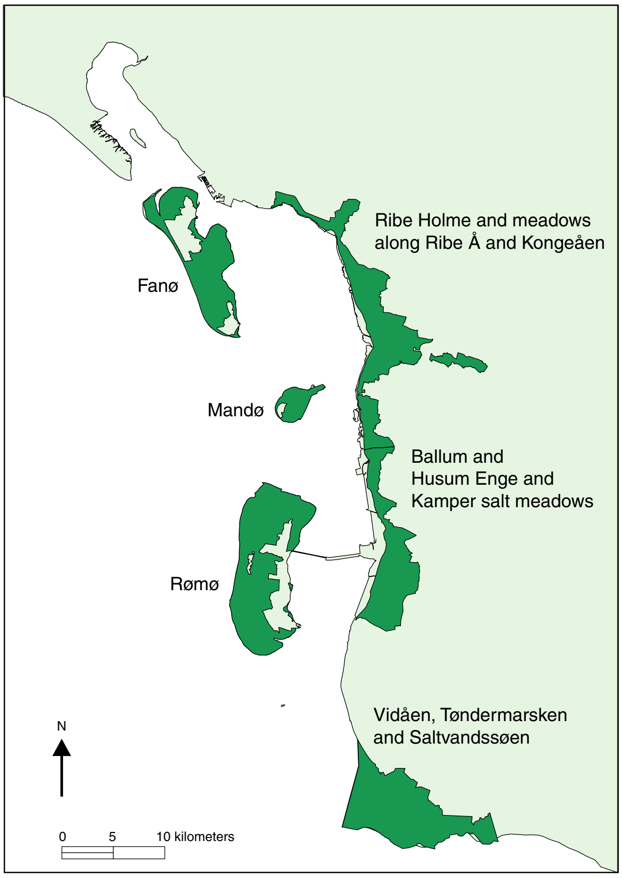

Figure1. Map showing the Danish Wadden Sea. The six marshland SPAs treated in the report are indicated.

Material and methods

The basic description of the species for which the SPAs has been designated contains information of the species number from 1983 when the SPAs were established and for the following years or periods 1987-1988 and 1993-1994 (Skov- og Naturstyrelsen 1995). For this study the breeding birds numbers are brought up to date by including the two later total counts of breeding birds in the Wadden Sea in 1996 (Rasmussen & Thorup 1998) and 2001 (Rasmussen 2003).

The information on breeding birds for the years before 1988 is based on local reports. In the years after the Trilateral Monitoring and Assessment Program (TMAP) was established, total counts of breeding birds were performed every five years in 1991, 1996 and 2001 (Rasmussen 2003). For these counts a detailed method was described (Hälterlein et al 1995).

In this paper the trend of the breeding bird is evaluated from 1983 to 2001, and grouped into the following categories: Increasing (I), stable (S), decreasing (D) or accidental (A). The latter refers to species only recorded one or two times (see below).

The survey intensity was low during the first years than later, which means that there could be more birds present during the first surveys than actually recorded. On the other hand if a species was not recorded or only recorded in small numbers during the last years, it is likely that the species had disappeared or occurring in smaller numbers than during the former period.

This raises some methodology problems that the survey intensity changed during the study period. However, histological figures can contribute to valuable information. This is a common problem and Boyd (2003) claimed that considering management aspect, 'pragmatism must take precedence over perfectionism'. In our study the material can not be treated using statistical methods due to both methodological problems and because the figures for some species are small and given with an unknown uncertainty. Due to these circumstances conclusions have not been drawn for single species, nor have any conclusions been drawn for a single SPA. Conclusions have only been drawn when more species showed the same tendency and when a tendency was seen in more SPAs.

Some species occur in small numbers as e.g. dunlin *Calidris alpina* and ruff *Philomachus pugnax*. However, these species are targets of considerable interest both due to their presence on the Annex 1 of the EU-Bird Directive and on the Danish Red List (Stoltze & Pihl 1997, Thorup 2004). Therefore it is important to include them in the analysis, and to formulate the criteria for trend evaluation in a way that minority species could be included.

The trend of the species is evaluated using the following criteria:

1) The trend is only evaluated if there are data from 4 out of the 5 years/periods and data from 2001 shall be present.

2) The trend is decreasing if a species is recorded by a + (present) or a number of pairs in the first year (or in the second year if the first year is lacking), and the species is not recorded in the following years.

3) The trend is decreasing if the number of pairs in 2001 is lower than during the former years. For species with more than 10 pairs the difference shall be >20%. However, if a species is only recorded in 2001 with one pair, the occurring species is considered as accidental.

4) The trend is increasing if the numbers of pairs in the first year or period is lower than during the following years. For species >10 pairs the difference shall be >20%.

5) The trend has changed if a species was recorded with higher or lower numbers during the 2-3 latest counts compared to the 2-3 earlier counts. For species with >10 pairs the different shall be >20%.

6) The occurrence of a species is accidental if the number is 0-1 or 0-2 pairs during the period, and a trend was not evaluated.

The trilateral monitoring program does not cover all species mentioned in the designation criteria of the SPAs, e.g. bittern *Botaurus stellaris*, marsh harrier *Circus aeruginosus* and pigeons, owls and passerines. Therefore ornithologists with local experiences have been contacted for supplementary information on these species. For further information on the methods see Laursen (2005).

Results

Results from each of the six SPAs are presented, with the species on the Annex 1 of the EU-Bird Directives mentioned first and afterwards the additional species. The number of pairs for each species is shown in Appendix 1-3 including the evaluated trend (increasing: I; constant: C; decreasing: D; and accidental: A) in the right column. The species are not commented in the text, for which an evaluation is not possible according to the criteria.

SPA no. 51: Ribe
Marsh harrier has increased in numbers (Appendix 1). The occurrence of avocet *Recurvirostra avosetta* and short-eared owl *Asio flammeus* is evaluated as constant, however, the number of ruff had decreased and it was not recorded since 1993-94. The occurrence of hen harrier *Circus cyaneus* is considered as accidental. For the additional species garganey *Anas querquedula*, shoveler *A. clypeata*, and black-tailed godwit *Limosa limosa* a reduction in numbers was observed. Gargeney and shoveler have not been recorded since 1993-94.

SPA no. 52: Mandø
Common tern *Sterna hirundo* has increased in numbers; avocet, dunlin and arctic tern *Sterna paradisaea* have been constant, while ruff has decreased in numbers and was not recorded after 1991 (Appendix 1). The occurrence of kentish plover *Charadrius alexandrinus* and little tern *Sterna albifons* is evaluated as accidental. For the additional species the trend was increasing for black-tailed godwit and constant for teal *Anas crecca* and eider duck *Somateria mollissima*.

SPA no. 53: Fanø
Bittern, marsh harrier and kentish plover have increased in numbers, while avocet and dunlin have been constant. However, montagu's harrier *Circus pygargus*, ruff, arctic tern, little tern and short-eared owl were reduced in numbers, and have not been recorded since 1983 and 1988-89 respectively (Appendix 1). Hen harrier has only bred once and its occurrence is considered as accidental. For the additional species the number of curlew *Numenius arquata* has increased, however, the number of teal, shoveler and grasshopper warbler *Locustella naevia* has decreased and these were not recorded since 1993-94.

SPA no. 60: Tønder Marsh
This area is treated as three separate areas (Magisterkogen, Ydre Koge and Margrethe Kog), because they are managed in different ways.

Magisterkogen: The number of bittern, white stork *Ciconia ciconia*, marsh harrier, montagu's harrier, spotted crake *Porzana porzana* and black tern *Chlidonias niger* has decreased (Appendix 2). The occurrence of common tern is evaluated as accidental. For the additional species the number of gadwall *Anas strepera* and pintail *Anas acuta* has increased, the trend of teal was constant, while the numbers of garganey and black-tailed godwit have decreased.

Ydre Koge: The number of bittern, marsh harrier, montagu's harrier and black tern have decreased (Appendix 2). For the additional species the number of gadwall has increased, the number of garganey, black-tailed godwit has decreased, while the occurrence of widgeon *Anas penelope* and teal is evaluated as accidental.

Margrethe Kog: The number of avocet has increased and numbers of common terns is considered as constant. However, marsh harrier, montagu's harrier, kentish plover, dunlin, arctic tern and little tern experienced decreases (Appendix 2). For the additional species the number of gadwall and black-tailed godwit has increased, the number of widgeon and pintail was constant, while the number of teal and garganey has decreased.

SPA no. 65: Rømø
The number has increased for kentish plover, arctic tern and little tern, it has been constant for avocet, while it decreased for dunlin, ruff, common tern, sandwich tern *Sterna sandvicensis* and short-eared owl (Appendix 3). For the additional species the number was stable for curlew and black-tailed godwit.

SPA no. 67: Ballum
Numbers decreased for ruff, which has not been reported since 1983 (Appendix 3). For the additional species the number was stable for lapwing *Vanellus vanellus*. However the number decreased for black-tailed godwit.

For the six SPAs taken together an evaluation of common trends was made for each species included in the Annex 1 of the EU-Bird Directive. In this trend evaluation species-specific trends for each SPA were considered and increases and decreases were weighted against each other (e.g. three areas with increases and one area with a decrease give an increase as a common trend of the species considered). The analyses of the common trend show that four species (24%) had increased in number (bittern, marsh harrier, kentish plover and common tern), three species (17%) were stable in number (white stork, water rail *Rallus aquaticus* and avocet), nine species (53%) had decreased in number (montagu's harrier, spotted crake, ruff, dunlin, arctic tern, sandwich tern, little tern, black tern and short-eared owl) and for one species (6%) the common trend was regarded accidental (hen harrier) (Fig. 2A). For the additional species there was a similar result; 21% had increased in number, 29% was stable, 36% had decreased and for 14% the trend was considered as accidental (Fig. 2B).

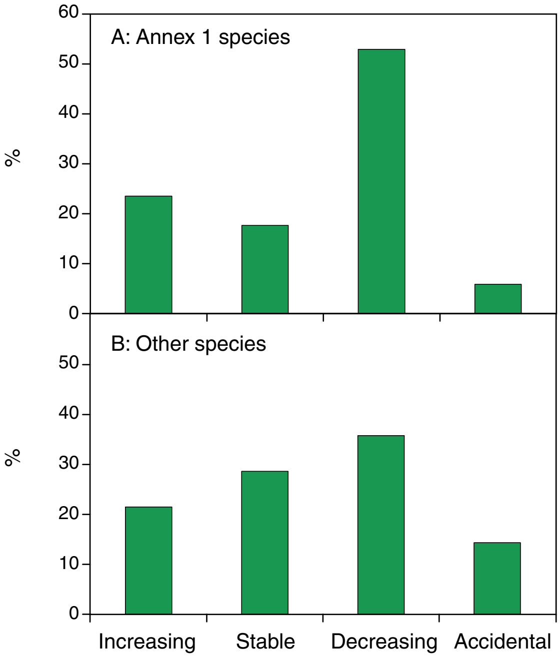

Figure 2. Trends of breeding birds (in %) in the marshland SPAs in the Danish Wadden Sea distributed in categories: Increasing, Stable, Decreasing and Accidental. A: Species on the Annex 1 of the EU-Bird Directive. B: Additional species mentioned in the appointment description of the SPAs.

However, it can be difficult to evaluate a common trend of a species, if the species has increased in two areas, has been constant in one area and has decreased in a fourth area. Therefore all trends in Appendix 1 for the two groups of species (the species in the Annex 1 of the EU-Bird Directive and the additional species, respectively) have been summed up for all areas (e.g. all evaluations of trends: I, S, D and A had been added). The summation of the trends for all Annex 1 species of the EU-Bird Directive showed that 21% of the trends were increasing, 25% of the trends were stable, 46% of the trends were decreasing and 8% was considered as accidental (Appendix 1-3). For the additional species there were increases for 17% of the trends, 30% of the trends were stable, 44% of the trends were decreasing and 9% of the trends were considered as accidental.

The results of the two calculation methods used for the Annex 1 species and the additional species respectively, show small differences. However, the results of both methods showed the same tendencies for the species.

Discussion

Trends in the International Wadden Sea

Trends of breeding birds in the International Wadden Sea have been analysed during 1991-2001 (Koffijberg et al. 2005). Nine species on the Annex 1 of the EU-Bird Directive are in common in both that report and in the present study. Three of these species were either increasing or stable in the Danish Wadden Sea marshland, and in the study covering the entire Wadden Sea two species were also stable or increasing (avocet, common tern) and one species were decreasing (kentish plover). In the Danish Wadden Sea six species were decreasing in the marshland areas, while four of these species were increasing or stable in the entire Wadden Sea (sandwich tern, arctic tern, little tern, short-eared owl) and two species were decreasing in both studies (dunlin, ruff). It is important to stress that the two studies do neither cover the same period or the same area. The study in the Danish Wadden Sea goes further back in time using historical data compared to the study covering the entire Wadden Sea. Besides, the first mentioned study focus on the marshland areas behind the seawalls, while the results from the entire Wadden Sea covers all areas including the saltmarsh areas in front of the seawalls. The results show that more species had decreased in the Danish Wadden Sea marshland than in the entire Wadden Sea.

Factors affecting breeding birds

Several factors are influencing the number of breeding birds (Koffijberg et al. 2005). Agricultural activities, including mowing and livestock-grazing are important to maintain a breeding population for several species. Experiences from several sites in Denmark (Tøndermarsh, Tipperne, Vejlerne, and Saltholm), Germany and the Netherlands have shown this (Beitema et al 1995, Nehls 1998, Rasmussen& Laursen 2000, Thorup 2004). It is also known that most of the bird species connected to meadows and marshland are depending on the intensity of the farming activity; e.g. common snipe *Gallinago gallinago* breeds in areas with a very low farming intensity, while on the other hand oystercatcher *Haematopus ostralegus* has its highest densities on more intensively used grassland (Beintema et al 1995). Lapwing is positioned in-between these two species. However, the intensity of the agricultural activity in the SPAs in the Danish Wadden Sea is not known. On the other hand analysis of aerial photographs in 1999 showed that only about 30% of the areas were permanent grassland (defined as grassland more than seven years old) in the SPA no. 51: Ribe and SPA no. 67: Ballum, while it was higher in the SPA no. 65: Rømø (53%), the SPA no. 52: Mandø (72%) and in some of the areas in the Tøndermarsh, Margrethe Kog (75%) and the Ydre Koge (85%) (Kampsax 2001).

Fields that are ploughed every year have only a small density of breeding marshland birds compared to permanent grasslands. Most meadow birds do not breed on fields ploughed every year. However, oystercatcher and to a lesser extent also lapwing may use these types of agriculture areas, but often with poor breeding results (Ettrup & Bak 1985, Falk et al. 1991). Montagu's harrier has changed breeding habitat during the last years from reed bed areas to fields with winter cereals (Ehmsen 2004). Grass areas can be re-laid (re-seeded) and studies on these new established grass areas indicate that they have a lower number of marshland birds than old grass areas (Falk et. al 1991). It lasts probably about 10-20 years before the density of breeding birds is at the same level on the two types of fields (Clausen pers. com.)

Grazing animals may destroy a large part of the meadows birds nest. On pastures with a density of three young cattle per ha, about 80% of the clutches were destroyed (Nielsen 1996). In general there is a positive relationship between the density of cattle and the destruction of nests. Likewise there is a positive relationship between the number of days with grazing cattle and the number of nests destroyed (Nielsen 1996).

Mowing can be an advantage for some of the breeding meadow birds. In mowed fields the nests are not destroyed by cattle and not otherwise disturbed before the mowing takes place. In this way the breeding success can be improved for species that prefer to breed in high vegetation (Thorup 1998). However, it is important, that most of the birds haves finished breeding (including the chick-rearing period) before mowing occurs Otherwise, nests are destroyed by mowing and chicks are killed.

Other factors that influence the breeding densities of meadow birds are ground water level during springtime and the presence of predators (Rasmussen & Laursen 2000, Kahlert et al. 2003, Olsen 2004). However, these parameters will not be considered in detail here, but studies of breeding birds in the Wadden Sea indicate that there is an increasing predator pressure on nests in the mainland areas (Koffijberg et al. 2005).

Management

The management of the SPSAs is not known. However, the proportion of permanent grassland defined as grass areas older than seven years have been mapped (Kampsax 2001). The results indicate that there is a relationship between permanent grassland and the number of breeding birds. For example the proportion of species with either a positive or stable trend is large (> 40%) in SPAs with permanent grasslands larger than 50% (e.g. Margrethe Kog, Rømø and Mandø). However, a large proportion of grasslands is obviously not the only factor to safeguard an increasing or stable trend for the species, since a large part of the species are decreasing in both Magisterkogen and Ydre Koge. Both of these SPAs have permanent grasslands covering more than 85% of the area but they show different trends in the breeding bird populations. Therefore it is likely that management of the areas plays an important role for the species trend, as shown for Margrethe Kog and Ydre Koge in the Tøndermarsh (Rasmussen & Laursen 2000). Thorup (2004) made a review on the development of breeding marshland birds in Denmark and concluded that species were only stable or increasing in areas with a management plan that include the demands of breeding marshland birds. As a follow up to that review the Forest and Nature Agency under the Ministry of Environment, have made an action plan for threatened marshland bird species. The action plan focus on 25 of the most important breeding bird areas in Denmark and describe elements in an active management for each area (Asbirk & Pitter 2005). The SPAs of the Wadden Sea areas dealt with in this paper are included in the action plan, and future monitoring results will show if the negative trend has halted for the species presented in this report.

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Appendix 1

Number of breeding birds in SPAs (Ribe, Mandø, Fanø) 1983-2001. Species names in bold: Specific species on Annex 1 of the Birds Directive for which the SPA is appointed. Species names in italic: Species on the Annex 1, which are also listed in the description of the SPA. Species names in normal: Additional species listed in the description of the SPA. + indicate that the species was recorded. – No information. ** One pair in 1994.

SPA no. 51, Ribe	1983	1987-88	1993-94	1996	2001	Trend
Garganv	-	2-3	2	0	0	D
Shoveler	-	3-5	5-10	0	0	D
Montagu's harrier	-	0-3	1-2	-	3*	-
Marsh harrier	-	0-2	2-4	3-4	>4	**I**
Hen harrier	0	0	0	0	1	**A**
Avocet	+	150-200	300	190	163	**S**
Ruff	-	0-5	5-10	0	0	**D**
Black-tailed godwit	10	12-20	15	25	6	D
Short-eared owl	+	0	0-2	1	0	**S**
Yellow wagtail	-	10	+	+	+	-

SPA no. 52, Mandø	1983	1987-89	1991	1996	2001	Trend
Wigeon	-	0-1	-	0	0	-
Gadwall	-	0-2	-	0	0	-
Teal	-	0-1	1-2	0	0	S
Pintail	-	0-1	-	0	0	-
Shelduck	-	-	19	-	-	-
Eider duck	400	120-210	286	+	403	S
Marsh harrier	+	1-3	2	+	2-3	-
Avocet	-	7-71	14	21	81	**S**
Kentish plover	-	0-1	0	0	0	**A**
Dunlin	+	0-1	1	0	1	**S**
Redshank	-	-	136	317	161	-
Lapwing	-	-	172	166	174	-
Oystercatcher	-	-	454	1086	569	-
Ruff	-	8-16	12	0	0	**D**
Black-tailed godwit	10	21-27	30	22	68	I
Turnstone	-	-	0**	0	0	-
Gull-billed tern	-	-	1	0	0	-
Common tern	-	25-35	37	143	43	**I**
Arctic tern	-	90-370	235	87	144	**S**
Little tern	-	0-1	0	0	0	**A**
Short-eared owl	-	-	0	3	3	-

SPA no. 53, Fanø	1983	1988-89	1993-94	1996	2001	Trend
Bittern	-	3-4	3-4	5-6	7	**I**
Greylag goose	-	-	1	5-6	10	-
Teal	-	2-8	3-5	0	0	D
Shoveler	-	2-4	0-2	0	0	D
Montagu's harrier	-	2-3	0	0	0	**D**
Marsh harrier	-	3-4	3-4	3-4	9	**I**
Hen harrier	0	0	0	1	0	**A**
Avocet	10	19-38	12-38	18	9	**S**
Kentish plover	+	4-10	19-21	16	16	**I**
Dunlin	-	9	6-11	7	6	**S**
Ruff	+	0	0	0	0	**D**
Curlew	8	26-33	23-30	10	22	I
Arctic tern	-	55-90	11-28	47	9	**D**
Little tern	20-30	30-42	65-75	53	7	**D**
Stock dove	-	16-22	20-25	+	20-30	-
Short-eared owl	-	1-2	0	0	0	**D**
Bearded tit	-	0-10	10-50	+	5-10	-
Grasshopper warbler	-	6-7	3-5	0	0	D

Appendix 2

Number of breeding birds in SPA, Tønder Marsh (Magisterkogen, Ydre Koge and Margrethe Kog) during 1983-2001. Species names in bold: Specific species on Annex 1 of the Birds Directive for which the SPA is appointed. Species names in italic: Species on the Annex 1, which are also listed in the description of the SPA. Species names in normal: Additional species listed in the description of the SPA. + indicate that the species was recorded. – No information. * Figures from 2002.

SPA no. 60: Tønder Marsh						
Magisterkogen	1983	1988	1993	1996	2001	Trend
Bittern	14	4	3	3	6	**D**
White stork	1	1	1	1	0	**D**
Gadwall	1	0	2	3	7	I
Teal	1	0	1	0	0	S
Garganey	35	4	9	10	8	D
Pintail	0	0	0	1	1	I
Marsh harrier	25	11	19	18	15	**D**
Montagu's harrier	6	12	6	0	3	**D**
Spotted crake	8	5	7	2	4	**D**
Black-tailed godwit	13	8	5	3	0	D
Common tern	0	0	1	3	0	**A**
Black tern	4	3	0	3	0	**D**
Bluethroat	-	-	1-4	1	19*	-
Ydre Koge	1983	1988	1993	1996	2001	Trend
Bittern	7	4	5	1	2	**D**
Wigeon	0	0	1	0	0	A
Gadwall	0	4	8	14	18	I
Teal	0	0	1	0	0	A
Garganey	76	17	53	12	24	D
Marsh harrier	7	4	5	1	2	**D**
Montagu's harrier	4	3	1	1	0	**D**
Black-tailed godwit	202	92	75	61	82	D
Black tern	72	21	36	46	12	**D**
Margrethe Kog	1983	1988	1993	1996	2001	Trend
Wigeon	0	5	3	2	3	S
Gadwall	3	14	15	23	22	I
Teal	3	5	5	0	2	D
Garganey	19	6	5	4	5	D
Pintail	0	8	3	2	1	S
Marsh harrier	2	1	0	1	0	**D**
Montagu's harrier	2	1	0	0	0	**D**
Avocet	108	423	482	268	143	**I**
Kentish plover	34	16	0	1	2	**D**
Dunlin	1	2	0	0	0	**D**
Black-tailed godwit	23	32	45	31	42	I
Common tern	0	105	124	68	10	**S**
Arctic tern	38	7	34	54	0	**D**
Little tern	9	3	0	0	0	**D**

* in 2000

Appendix 3

Number of breeding birds in SPAs (Rømø, Ballum) during 1983-2001. Species names in bold: Specific species on Annex 1 of the Birds Directive for which the SPA is appointed. Species names in italic: Species on the Annex 1, which are also listed in the description of the SPA. Species names in normal: Additional species listed in the description of the SPA. + indicate that the species was recorded. – No information. `Figures from 1994.

SPA no. 65, Rømø	1983	1988	1991	1996	2001	Trend
Bittern	1	5-9	2-3`	+	+	-
Gadwall	-	4-5	-	0	0	-
Teal	-	9-11	-	0	0	-
Pintail	-	0-2	-	2	0	-
Garganey	-	3	-	0	0	-
Shoveler	-	10-12	-	0	0	-
Montagu's harrier	+	6-7	9`	+	1	-
Hen harrier	-	-	1`	2	0	-
March harrier	+	7	-	+	>5	-
Spotted crake	-	0-1	-	0	0	-
Avocet	30-50	53	10-15`	67	47	**S**
Dunlin	20-25	23	22	18	12	**D**
Ruff	40-50	15-19	12	0	3	**D**
Kentish plover	25	20-22	10	38	68	**I**
Curlew	25	34	36	26	20	S
Black-tailed godwit	80	79	83	62	73	S
Common tern	40-50	25	50	0	2	**D**
Arctic tern	170	300	109	630	281	**I**
Sandwich tern	+	0	0	0	0	**D**
Gul-billed tern	-	-	2-3`	8	1	-
Little tern	8-12	18	10	136	162	**I**
Short-eared owl	1-2	1	0	5	0	**D**
Grashopper warbler	-	11-13	-	-	-	-
SPA no.67, Ballum	1983	1989	1990	1996	2001	Trend
Garganey	-	1-3	-	0	0	-
Teal	-	5-8	-	0	0	-
Shoveler	-	1-3	-	0	0	-
March harrier	-	-	2`	+	>15	-
Montagu's harrier	-	-	4`	-	15	-
Lapwing	-	550	299	602	409	S
Black-tailed godwit	+	19	15	42	12	D
Redshank	-	14	-	207	160	-
Ruff	min. 10	0	0	0	0	**D**
Short-eared owl	+	0	-	0	0	-

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