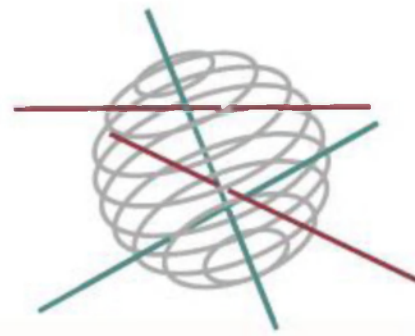# SSD

SCIENCE FOR A SUSTAINABLE DEVELOPMENT

"ECOSYSTEM SENSITIVITY TO INVASIVE SPECIES"

«EnSIS»

J.-S. Houziaux, J. Craeymeersch, B. Merckx, F. Kerckhof,
V. Van Lancker, W. Courtens, E. Stienen, J. Perdon, P.C. Goudswaard, G. Van Hoey,
L. Vigin, K. Hostens, M. Vincx, S. Degraer

ENERGY

TRANSPORT AND MOBILITY

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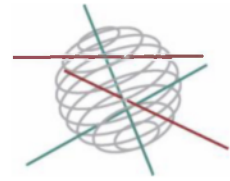
HEALTH AND ENVIRONMENT

CLIMATE

BIODIVERSITY

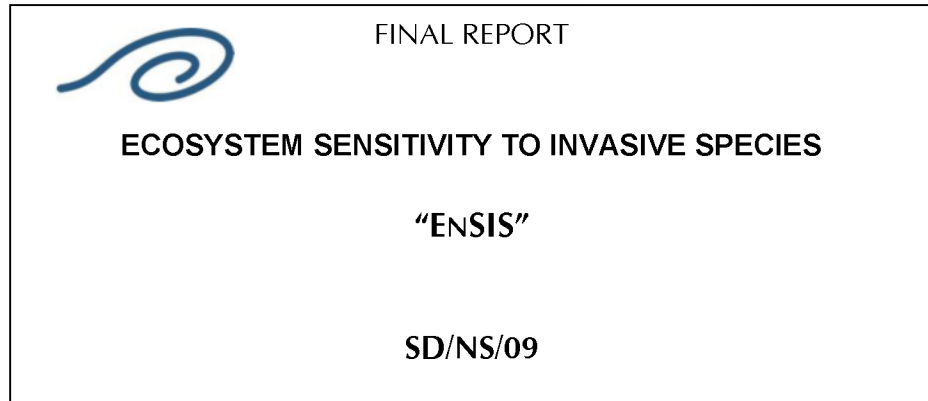ATMOSPHERE AND TERRESTRIAL AND MARINE ECOSYSTEMS

TRANSVERSAL ACTIONS

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North Sea

FINAL REPORT

ECOSYSTEM SENSITIVITY TO INVASIVE SPECIES

"ENSIS"

SD/NS/09

Promotors

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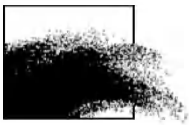(RBINS-MUMM)
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1 Executive Summary

The increasing occurrence of introduced invasive species in coastal marine ecosystems causes concern for the maintenance of local ecological processes, functions and services. Therefore, the global problem of introduced species is a major issues considered within the European Marine Strategy Framework (MSFD; Directive 2008/56/CE) to reach a good environmental status in the European seas. In the North Sea, one species native to the North-Eastern American coast, the American razor clam *Ensis directus*, has established permanent populations since it reached the German Bight in the late 1970s. This now widespread species is present at the Belgian coast since the 1980s, with large amounts of dying specimens and shells periodically washing ashore. So far, it has been little documented due to its ability to dig deep in the sediment, but questions have arisen relative to its population size, distribution and environmental impact. Its abundance also triggers questions on a possible exploitation by specialized fisheries, as is already the case *e.g.* in Dutch waters.

We determined the size-structure of the Belgian and Dutch populations of *E. directus* in the last years. It displays a mode at a length of 11-12 cm and a maximum observed length of 16 cm. Stable adult populations were found around the Nieuwpoort Bank, on the Oostende Bank and on the Northern slope of the Vlakte van de Raan in Belgian waters, and off the Westerschelde mouth, in the Voordelta and off the Frisian Islands in The Netherlands. Abundant recruitment events took place in the southern bight in the last decade, leading to a doubling of observed *E. directus* biomass in Dutch waters. Habitat suitability modeling in the subtidal confirmed that the species tolerates a wide range of environmental conditions but prefers slightly muddy fine sands, in shallow waters where phyto-plankton is present. The lower intertidal was not considered within this research. Acoustic seafloor characterization carried out in areas where this species thrives suggests that it creates a distinctive acoustic signature, together with the tube-dwelling worm *Owenia fusiformis*. No effect of *E. directus* could be detected on the composition of the local macrobenthos, but the assessment was hampered by a lack of appropriate spatio-temporal sampling coverage before and after the invasion. Further analysis including polychaete worms is also desirable before drawing firm conclusions. The results highlight the crucial importance of an *ad hoc* benthos monitoring program to early detect alterations of the marine ecosystem. They tend to confirm that this species colonized an empty ecological niche, because local *Ensis* species generally occurred more offshore, in coarser sediment in the historical (1900) situation. Nevertheless, various observations carried out recently indicate that changes are presently taking place in the macrobenthic species composition. The cause for these changes is not elucidated yet but could relate to the increase in sea surface temperatures noted in the same period.

As a matter of fact, *E. directus* has become a prominent component of the benthic communities of the southern bight of the North sea in terms of distribution, abundance and biomass. Its link with higher trophic levels thus deserves consideration. For the common scoter *Melanitta nigra*, it seems to be a prey item of moderate importance because the birds mostly feed on smaller specimens (less than two years old), the distribution of which likely varies from year to year with hydro-meteorological conditions.

Given the strong bond of scoters with conditions prevailing close to the western coast, *E. directus* likely provides a valuable food source when a high recruitment takes place, but does seem to properly replace the cut trough shell *Spisula subtruncata*. The rarefaction of this species likely explains recent shifts in the distribution of scoters along the Belgian coast. Recent reports also indicated that *E. directus* has become a dominant prey item for flatfishes of the nearby Voordelta area '(dab, plaice, sole). Given the importance of these species in the Belgian part of the North Sea and the observed abundance of *E. directus*, the same situation likely occurs within Belgian waters, an aspect which deserves further research. The findings as well as experience gained on *Ensis* fishery in Dutch waters suggest that at observed recruitment rates, this species could be well-suited for a targeted fishery within Belgian waters. The impact of such a fishery on the ecosystem is expected to be limited if it is carried out on a small-scale, as is the case for the Dutch *Ensis* fishery. Its implementation will require further detailed investigations to balance the expected benefits for the local fishery sector and the potential impacts on the ecosystem.

2 Context

The introduction of invasive species is now considered a major problem to marine ecosystems (Ruiz et al., 1999, Pimentel, 2000). This stressor hence forms a priority item within the European Marine Strategy Framework Directive (MSFD; 2008/56/CE): EU Member States will be obliged to map the problem of invasive species and to develop mitigating measures. The American razor clam *Ensis directus* represents a well investigated and documented example of such invasion in North-West European coastal waters (e.g. Armonies & Reise, 1999). However, many questions, regarding its impact on the indigenous ecosystem remain unanswered (Kerckhof et al., 2007).

Since the first specimens of *E. directus* were discovered in the Belgian part of the North Sea (BPNS) in 1987 (Kerckhof and Dumoulin, 1987), the species rapidly colonized the whole Belgian coastal area and formed extensive populations, also in areas where previously no indigenous *Ensis* spp. occurred. Nowadays vast amounts of shells and dying specimens of *E. directus* yearly wash onto the Belgian beaches (Kerckhof et al 2007), demonstrating the presence of large populations in Belgian nearshore waters. The species tends to often occur in high density and biomass clusters (often 2000 ind./m², up to 15500 ind./m&; Armonies & Reise, 1999; Steenbergen & Escaravage, 2006). These *Ensis* banks display a patchy distribution. However, even though it normally lives in the surface sediment, *E. directus* can dig deep and very rapidly in the sediment (down to 50 cm; Tulp et al, 2010), and is consequenty inadequately sampled with traditional sampling gears such as the Van Veen grab.

Ensis directus occupies a wide range of temperate habitats, is a fast growing species and has far-reaching dispersal capabilities through its pelagic larvae, which explain its success in the North Sea (Beukema and Dekker, 1995; Armonies, 2001). After storms and also perhaps because of overcrowding, a high proportion of the population seem to die few years after settling, forming an idle mass of rotting animals on the seafloor, clogging fishing gear or stranding on the beaches (pers. obs, F. Kerckhof). Most individuals die before reaching the age of five years, while the highest mortality is observed within the first two years. As *E. directus* often occurs in high densities, the hypothesis arose that the species could alter the environment and the local benthic communities (Kerckhof et al. 2007). Given the habitat preferences of the species (*i.e.* muddy sands), this impact is predominantly expected within the "*Abra alba*" and "*Macoma balthica*" macrobenthic communities, (Van Hoey et al., 2004; Degraer et al., 2008), of which the former community has the highest biological value in the BPNS according to Derous et al., 2007. Possible impacts, such as habitat alteration and species shifts were however not yet studied. Prior to the arrival of *E. directus,* two *Ensis* species were common in the BPNS, namely *Ensis arcuatus* and *E. minor* (Backeljau, 1986). The former seems to have withdrawn to areas further offshore, while the latter seems to be extirpated. It is not clear whether the arrival of *E. directus* has affected the fate of local *Ensis* species.

On the other hand, cascading effects can be expected to take place on predators of macrobenthic invertebrates. Eider ducks (*Somateria mollissima*), Sea-ducks (*Melanitta* spp.), seagulls and others have been observed foraging on American razor clams in various places in the North Sea (e.g. Wijsman et al, 2006). Predation by seabirds is mainly directed at smaller individuals, since larger ones cannot be processed by smaller species (Freudendahl & Jensen, 2006). Flatfishes have also been described to forage on this new prey and in some areas, such as the

Voordelta, it has become the dominant component of their diet (Couperus et al, 2009).

Although at present shell fisheries are prohibited in the BPNS, the expected high turn-over rate of the *E. directus* population of the BPNS also triggers discussion about a targeted fishery in Belgian waters. This possibility requires careful consideration about the impact on *E. directus* populations (sustainability and profitability of the activity) as well as on the associated seafloor, macrobenthic communities and food webs.

The spatial distribution, population structure and potential impacts of *E. directus* on the ecosystem in the BPNS remain largely unknown. This project specifically targets filling this gap in knowledge.

<u>Objectives</u>

In the BPNS, the distribution, abundance, population structure and impact of this invader was to date not addressed. This study focuses on three overarching objectives:

1. To describe the ecological features of *E. directus* in Belgian waters (population structure, distribution, habitat preferences).

2. To evaluate the ecological impacts of *E. directus* introduction into Belgian waters, focusing on macrobenthic communities and sea ducks.

3. To assess the feasibility and potential impact of a targeted *Ensis* fishery in Belgian waters.

3 Methodology and results

3.1 Data-sets

3.1.1 Existing data-sets

3.1.1.1 Joint UG-ILVO Macrobenthos database, 1977 – 2008

In the 'Habitat' project (Degraer et al., 2009) a joint database of most of the macrobenthic data collected at ILVO-Fisheries (BIOMON group) and UGent-Marbiol with a Van Veen grab was constructed. This database was checked for species taxonomy and the amount of samples per station and year was reduced to one to exclude temporal variation and to focus on the spatial distribution of the benthic fauna. The benthic fauna not representatively sampled was excluded afterwards (< 5% of occurrence). The database consists of data collected in 30 year and contains benthic data at 1169 stations. Due to low penetration in the sediment (about 15 cm) combined with a small sampled surface (0.1 m²), the Van Veen grab is however considered as inappropriate to track deep-burrowing species as razor clams. This limitation must be taken into account at the data analysis stage.

3.1.1.2 Schelp survey, 2008 – ILVO-Fisheries

In the summer of 2008 (17-25 July), a sampling programme was executed by ILVO-Fisheries onboard the fishing vessel N58 ("SCHELP" survey; Figure 1). A trawled dredge, the standard sampling gear for shellfish surveys in the Netherlands (IMARES, see 2.1.1.3 and Figure 2), was used. The track was executed over a distance of about 150 meters. The exact distance is determined by means of an electronic counter linked to a wheel rolling over the bottom. This measurement is checked by means of the ship geographic positioning system (DGPS) providing information on start and end points of the tow. The DGPS measurement was used to determine the tow length when the electronic counter gave a distance smaller than expected, for instance on very muddy bottoms where the wheel may not operate appropriately. The sampled surface thus amounts at about 15 m².
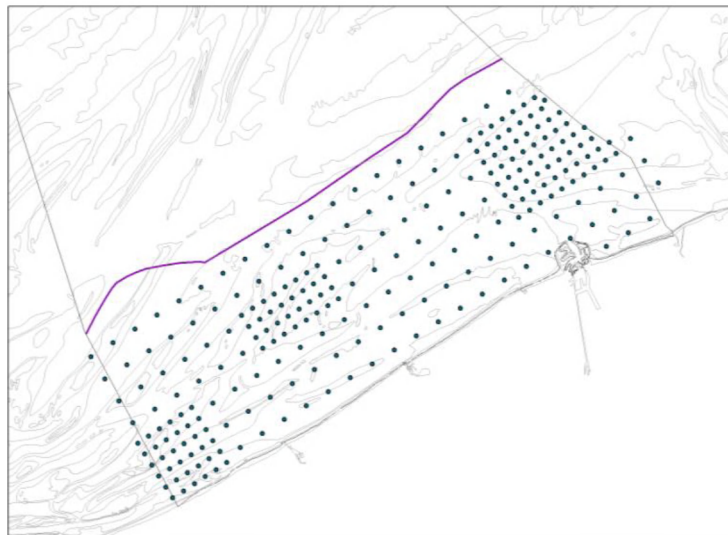
Figure 1. Sampling locations of the SCHELP survey in the Belgian coastal area (within 12 nautical miles).

The sampling points are distributed in a regular grid in the research area. The sampling intensity in each grid cell is determined by the likelihood of finding *Ensis* or *Spisula* shells in this area, as indicated by data described in section 3.1.1.1). In those areas a finer grid was sampled, because higher densities of shell animals could be expected. Three areas were defined for this case, namely the Western coastal area (Smalbank – Trapegeer area), the central coastal area (Kwinte – Middelkerke and Oostende bank) and the Eastern coastal area (Vlakte van de Raan). In the Eastern coastal area, one point per grid cell was systematically skipped during the sampling campaign, due to the habitat homogeneity of this area. The surface area of each grid cell depends on the used grid and is respectively 9, 4.5 and 2.25 km². Finally, 219 locations were sampled within the 12 miles area on the Belgian Part of the North Sea.

Depending on the catch size, all collected specimens or a subsample were sorted out and counted. Intact specimens were weighted (fresh weight, 0.1g accurate). When the specimens of *E. directus* were broken by the sampling device, the shell width was measured at the anterior side (siphon). A distinction was made between 'large' and 'small' specimen using shell width of 16mm as the limit. This width corresponds to a shell length of circa 120 mm. A small fraction of the specimens were too damaged to be measured and were consequently recorded as "non determined".

Figure 2. The trawled dredge (IMARES).

3.1.1.3 *Benthos Monitoring surveys, 1995-2010 – IMARES*

From 1990 onwards, annual surveys have been carried out by IMARES in spring to estimate the standing stock of mussels (*Mytilus edulis*) and cockles (*Cerastoderma edule*) on the tidal flats of the Dutch Wadden Sea and the tidal flats of the Oosterschelde. In 1992 the monitoring was extended to the tidal and shallow subtidal of the Westerschelde estuary. Surveys on the subtidal mussels in the Wadden Sea started in 1992. The coastal area (up to 12 nautical miles offshore) is monitored since 1995, except for the southern part (Voordelta) that is monitored since 1993. The focus is on mussels, cockles, trough shells and, recently, on razor shells. The surveys are mainly carried out as statutory tasks related to Dutch legislation in fisheries management and aquaculture. The main products of the programme are yearly stock assessments reported to the Ministry of Economic Affairs, Agriculture and Innovation (ELI). The inventory of the subtidal mussel beds is the basis for the annual fishing plans, made by the Mussel Producers' Organization as part of a management plan.

The surveys provide information on the spatio-temporal variations in density, biomass and stocks of aforementioned commercially exploited species, but also result in a time-series of abundance and biomassa data for another 25 infaunal and epifaunal species (De Mesel et al 2011).

Most of the stations are sampled either with a trawled dredge, a modified hydraulic dredge or a grab. The trawled dredge (Figure 2), in general, resembles the one described by Bergman & van Santbrink (1994). The dredge is 2 m long, 1m wide and 0.65 m high and weights about 350 kg. Above the cage a depressor plate is fitted to keep the dredge at the seabed surface. The blade has a width of 10 cm. While towed, a strip of sediment is excavated and transported into the cage. The stainless steel cage has a mesh size of 0.5 cm. The back-side of the cage can be opened to collect the sample. The length of the haul is measured by a measuring wheel: a magnetic reed-contact counts the revolutions of the wheel. In the coastal area this is the main sampling device. The modified hydraulic has a sampling width of 20 cm and a mesh width of 0.5 cm. It is trawled over a distance of 100-150 m. Because of the presence of large stones, in a small part of the coastal zone (Texelse stenen) a van Veen grab was used at ± 30 stations. At each of these stations, 3 samples were taken (total surface area 0.15 - 0.3 m²). In 1998, the area north of the Wadden Islands was also sampled with a van Veen grab (one sample per station, surface area 0.167 or 0.182 m²).

Following a stratified approach, samples are taken at 800-1000 sub-tidal stations (Figure 3). The surveyed areas are divided into a number of strata according to prior knowledge of or expectation on the distribution and density of bivalves of commercial interest.
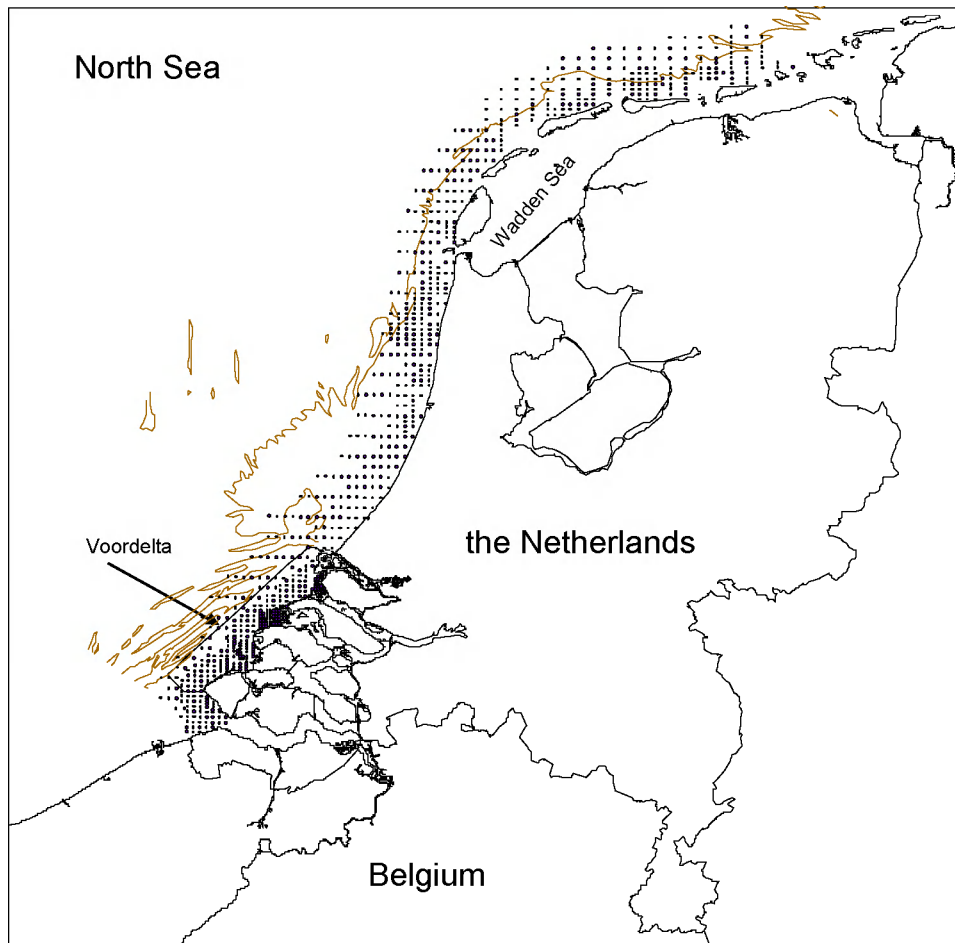
Figure 3. Location of the sampling stations in Dutch coastal waters (year 2001)

The division is based on former surveys, and on information of fishermen and colleagues. Samples are organized within each stratum by cells in a grid. The cell size of the grid is depending on the stratum. In strata with high clam densities, a smaller cell size is used than in areas with low densities of clams. Sampling points are placed at the grid nodes. In the coastal area distances between adjacent sampling points varied between 0.25 and 5 geographical minutes.

Samples were sieved through a 1 mm sieve, sorted and counted. Fresh weight per sample was registered on board. As *Ensis* were often damaged when collected, the siphon tips (anterior side of the shell) were counted and the width of the anterior side of the shell was measured if still present. Measurements on shell-length and weight were, because of the damage on the shells, not possible for the majority of specimens.

3.1.1.4 *Seabird (scoter) data-sets*

The first counts of seaducks were performed from elevated platforms on the coast using binoculars and telescopes, but it was soon realized that this led to a heavy underestimation of the numbers of ducks at sea. To solve this problem, monthly aerial counts were started in 1967 and were conducted until 1977. In 1986, the Research Institute of Nature and Forest (formerly "Institute for Nature Conservation") resumed the aerial counts. Since then, one count was done almost every month during November-March (Offringa & Meire, 1999).

The flights were conducted with a Cessna airplane where the counters are situated under the wings. Each person is counting one side of the plane. With a chronometer, the beginning and end of each transect was timed. Also, every group of sea ducks was timed between the two endpoints, so afterwards the precise location of the birds could be calculated. In this paper, no difference is made between the two scoter-species (Common and Velvet Scoter) because of the difficulty of identifying swimming birds to the species level, and because they often occur in mixed groups and roughly have the same feeding preferences.

The number of counted transects changed in the course of the past decennia. The transects are surveyed in a systematic manner only since the winter of 1996/'97. In order to provide consistent results, we only used the latter counts to create distribution maps and carry on further analysis of distributional data. During each flight, 24 transects covering the area between De Panne and the Zwin Nature Reserve were surveyed (see Figure 51). From 1996 to 2007, 4 to 6 flights were achieved between October and March. From 2008 onwards, only one count per winter was achieved, in March (which generally corresponds with the maximum number of observed birds; see Feys, 2007).

For most winters, additional data from land-based counts were further available (mostly for the transect from De Panne to Ostend, which exhibit maximum numbers). In these counts, ducks were counted from vantage points with a telescope.

The data of the aerial counts since 1967 were used together with the maximum values of the land based counts.

3.1.2 Newly acquired data-sets

Sampling at sea has been organized in spring and autumn 2009, targeting data acquisition with the 'triple-D' trawled dredge and the recently developed 'Semi-grab'. Due to bad weather conditions and technical problems, the Box-corer of the R/V Belgica (NIOZ model) was used in additional surveys carried out in February 2010.

3.1.2.1 Trawled dredge sampling, June 2009

A first campaign at sea was cancelled due to bad weather conditions (March 2009). In June 2009, the routine procedure of IMARES (see section 3.1.1.3) was used to obtain macrobenthic data at 36 stations with the trawled dredge onboard the R/V Zeeleeuw of the Flemish Institute of the Sea (Figure 4). This survey targeted 'filling the gaps' in selected portions of the "SCHELP" survey of the ILVO-Fisheries, in summer 2008 (see section 3.1.1.2.). The data (abundance and biomass of macrobenthic species) were integrated into the database of IMARES to form the joint Belgian-Dutch data-set. The purpose of this data-set was to carry out habitat suitability modelling (HSM) over the entire area.
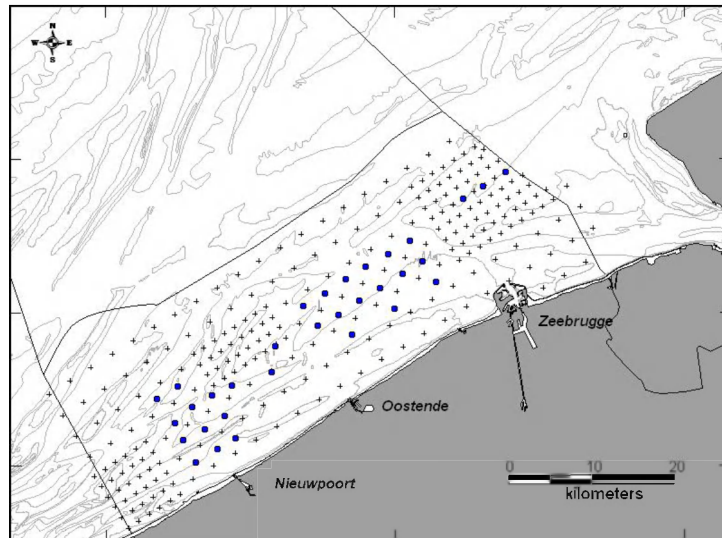
Figure 4. Sampling programme completed on board the R/V Zeeleeuw, June 2009 with the trawled dredge. Black dots: original sampling by ILVO-Fisheries, programme "SCHELP", 2008 (see section 3.1.1.2). Larger blue dots: newly acquired samples.

3.1.2.2 *Semi-grab / Box-Core sampling, October 2009*

One cruise was programmed onboard the R/V Belgica in October 2009 to test the recently developed 'Semi-Grab' benthic sampler (see Craeymeersch et al, 2007) in the BPNS (Figure 5).
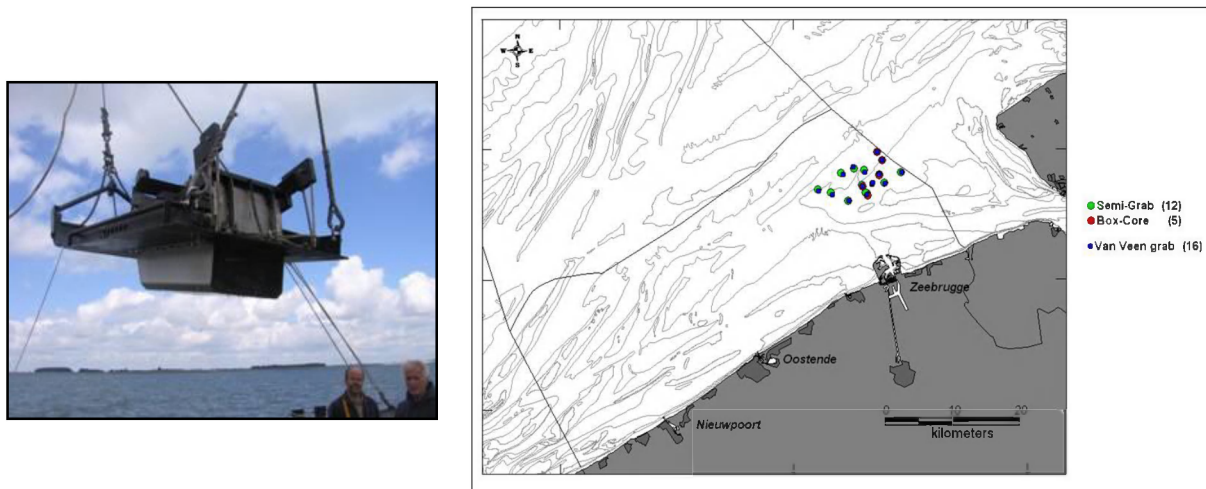
Figure 5. Positions of Semi-Grab (photo), Box-Core and Van Veen grab samples acquired during Belgica cruise 2009-29 (October 2009).

This sampling gear targets the acquisition of a larger volume of sediment including deeper sediment layers, thus providing better likelihoods to collect deep buriers such as larger *Ensis* spp. While the gear was operated successfully by IMARES in the Dutch Wadden Sea, it seems less well suited to the North Sea bottoms due to its dependency on research vessel size, depth and sediment composition. After 12 samples (figure 5), the sampling device broke down probably due to encounter with stones. A NIOZ Box-Corer (surface: 0.082 m²) was deployed and 5 more samples could be obtained. The device went however lost at sea and the cruise was subsequently cancelled. All samples collected were sieved over a 0.5 cm mesh and were stored in an 8% formaldehyde-seawater solution. Their content was sorted out but not further analyzed since new surveys had to be rescheduled (see section 3.1.2.4). An additional Van

Veen grab sample was collected at every station, from which a sub-sample (core) was taken targetting sediment grain-size analysis. All collected samples are stored in MUMM-Oostende and remain available for further analysis.

3.1.2.3 *Acoustic surveys*

Acoustic data were gathered during three cruises in areas where densities of *E. directus* were known to be relatively high (based on the UG-ILVO joint database, section 3.1.1.1. and the SCHELP sampling survey, section 3.1.1.2), *i.e.* on the northern slope of the Vlakte van de Raan and in the surroundings of the Oostende bank. Full-coverage, very-high resolution multibeam echosounder imagery (*RV Belgica*, Kongsberg Simrad EM3002, 300 kHz) was recorded to acoustically characterize the occurrence of *E. directus* beds. Both depth and backscatter intensity of the acoustic signal were recorded.

Overview of acoustic data collected

• 29-30/06/2009: RV Belgica cruise 2009/17. Data were acquired within 2 boxes where densities of *E. directus* were highest during the ILVO sampling in 2008, *i.e.* along the northern flank of the Oostende bank and North of the Vlakte van de Raan area. 25 Van Veen grab samples were collected for sediment characterization and *Ensis* specimen measurements and counts. These data were appended to the joint UG-ILVO data-base (see section 3.1.1.1).

• 16-19/11/2009: RV Belgica cruise 2009/31. Data were acquired within 1 box North of Vlakte van de Raan. Sediment and *E. directus* samples were taken.

• 29-31/03/2010: RV Belgica cruise 2010/08. Area North of Vlakte van de Raan; adverse weather conditions.

Supplementary acoustic data were obtained during campaigns organized in the framework of the "QUEST4D" Belspo project (Van Lancker et al, 2011). Especially, RV Belgica cruise ST1029 (16-19/11/2010) provided valuable data to be correlated with *E. directus* occurrences. In a later "QUEST4D" campaign (ST1108; 21-25/3/2011)), current data (Acoustic Doppler Current Profiling) were obtained along a transect perpendicular to the slope of the Vlakte van de Raan.

3.1.2.4 *Box-core sampling, February 2010*

To meet the project's objectives, new samples were collected in February 2010 with a box-core (NIOZ Box-Core; sampling surface area: 0.082 m²) during three cruises onboard the R/V Belgica (Figure 6). The target was to investigate the *Ensis* population structure in the Belgian waters. Therefore, the survey was focused on areas where higher densities of *E. directus* were found ealier (UG-ILVO data-base, section 3.1.1.1., and SCHELP survey, section 3.1.1.2). At 115 stations (Figure 6), two box-core samples and two Van Veen grab samples were generally collected sequentially to increase the sampled surface. All box-core samples (n=228) were sieved over a 0.5 cm mesh-sized sieve and the residual was fixated with an 8% formaldehyde-seawater solution.
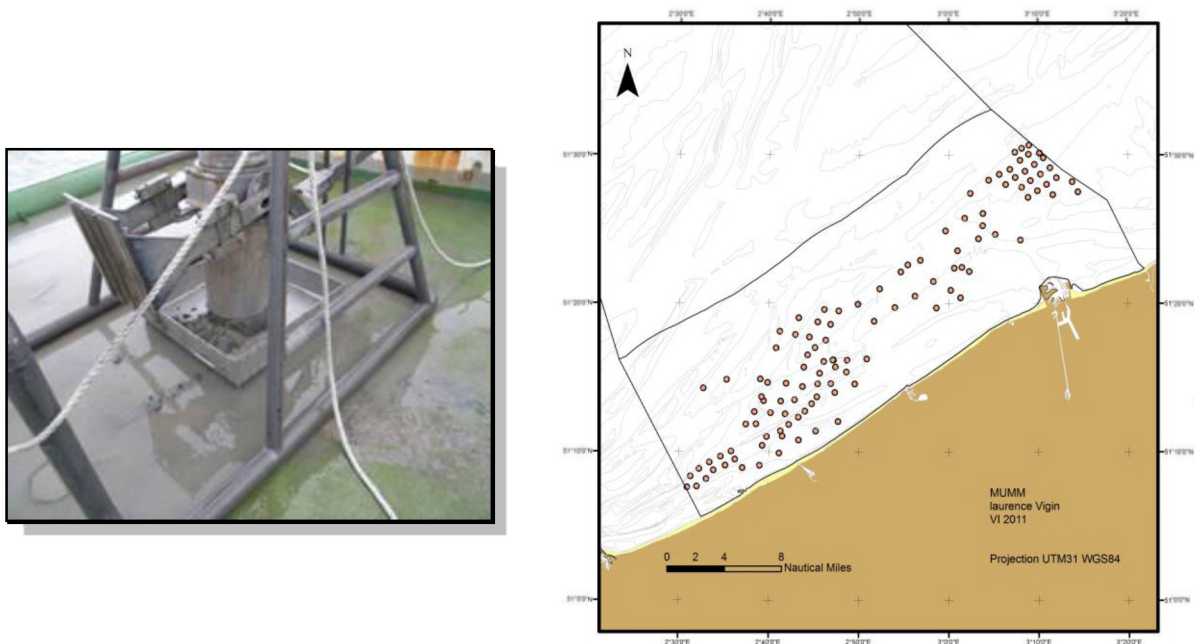
Figure 6. Positions of box-core (photo) sampling stations, February 2010 (cruises Belgica 2010/3, 2010/4a and 2010/5).

In all samples, at the sorting step, a size-based distinction was made between one year-old specimens (<= 6 cm) and the larger specimens. 177 samples were fully sorted out and all taxa were identified to the lowest possible taxonomic level except for most polychaetes, echiurians and sipunculians. All invertebrates except *E. directus*, were transferred to ethanol 70 %. All specimens of *E. directus* (n=4,386) were later extracted from all 228 samples to carry out morphometric measurements and to determine the population size structure. They were left in the formaldehyde solution.

All Van Veen grab sediment samples were analyzed for grain-size distribution. The grain size analysis was done with the Malvern Hydro 2000G Mastersizer. The sample is analysed by low power laser. The particles scatter some of the light characteristic of their size, forming a series of diffraction patterns each consisting of concentric bright and dark rings. This diffraction pattern is collected by a Fourier lens and brought to focus on an array detector. The signal from each detector element is analysed by the computer. From these signals a detailed particle size distribution can be derived within the range of 0 to 2000 μm.

3.1.3 Environmental data-sets

Selected environmental parameters were requested to carry out habitat suitability modelling (HSM) for *E. directus* (see section 3.2). The data were aggregated within grids extending over the Belgian and Dutch zones.

3.1.3.1 *Sediment characteristics and acoustic seafloor maps*

Median grain size (d50) and silt-clay content of the upper seabed of the Belgian part and Dutch continental shelf part of the North Sea were obtained, respectively from BELSPO QUEST4D (Van Lancker et al., 2010) and from TNO, (http://www.tno.nl). In the framework of the "MAREBASSE" project (BELSPO SPSDII; Van Lancker et al, 2007), a number of physical data grids have become available that have been used in the EnSIS project (DVD GIS@SEA, Van Lancker et al., 2007). These data grids (resolution: 80 to 250m) cover the entire Belgian part of the North Sea and comprise (1)

bathymetry (data owner Flemish Authorities, Flemish Hydrography), (2) median grain-size of the sand fraction and (3) proportion of silt-clay. In addition, a number of derivatives of the original data layers were available for modelling, such as data grids on the bathymetric position index (BPI), slope and rugosity as well as proportion of gravel, sand and mud. The GIS@SEA DVD also provided hydrodynamic and sediment transport data grids. Some of these data grids have been upgraded within the BELSPO SSD project "QUEST4D" (Van Lancker et al., 2011). A new sediment database was created containing full grain-size distribution curve data (sediCURVE@SEA; Van Lancker et al., 2009) and was merged with Dutch databases on sediment parameters. The coupling of the datasets was more difficult given the different sampling density in the Belgian and Dutch parts of the Westerschelde estuary. New interpolations were carried out to ensure data fitting.

Acoustic seafloor maps and newly acquired hydrodynamic and sediment transport data within the Vlakte van de Raan area were also made available though the project "QUEST4D".

3.1.3.2 *Hydrodynamic and water quality data*

Data on current speed, bed shear stress and salinity were extracted by Deltares from an archive, using the hydrodynamic modelling framework Delft3D-FLOW, which is DELTARES's software system for simulating three-dimensional hydrodynamic behaviour of free surface flows, and the so-called ZUNO-GROF model schematisation.

A detailed description of the ZUNO-GROF model (including its calibration and validation) can be found in Roelvink et al (2001). The ZUNO-GROF model was used as input to the Delft3D-WAQ (WAQ = water quality) simulations to model current direction and suspended sediment. In this case the version of the hydrodynamic model that was used to force Delft3D-WAQ was that of the 'Impact Sand Extraction Maasvlakte 2' project (van Prooijen et al, 2006). The time step used in the hydrodynamic simulations was 5 minutes, which is sufficiently small, considering the spatial resolution, to obtain accurate model results. The hydrodynamic simulations include a representative tide based on 50 astronomical components, actual river discharges, wind and atmospheric pressure data (van Kessel and Brière, 2006). The relatively short run time of ZUNO-GROF (~ 15 hrs per year simulation) made this model suitable to perform a series of annual runs that differed in their forcing of wind, air pressure and river discharges. For more details on the hydrodynamic simulations see De Goede and Van Maren (2005).

The used version of the ZUNO-GROF model has 12 vertical σ-layers with the highest vertical resolution near the areas were maximum velocity shear is expected, i.e. near-bottom for the tidal flow and near the surface in case of wind forcing. The grid resolution varies from quite coarse far offshore to finer close to the Dutch coast (Figure 7). The bathymetry was derived from a larger number of datasets, including coastal data, data from Dutch Continental Shelf Data (TNO-NITG) and data from the Dutch hydrographic Service. Bathymetric digital terrain models were made available for the Belgian part of the North Sea (Van Lancker et al., 2007), as also for the Dutch continental shelf through TNO, Bouw en Ondergrond through the OpenEarth database (http://www.tno.nl). Belgian and Dutch bathymetric datasets were coupled on a grid with a resolution of 250m.
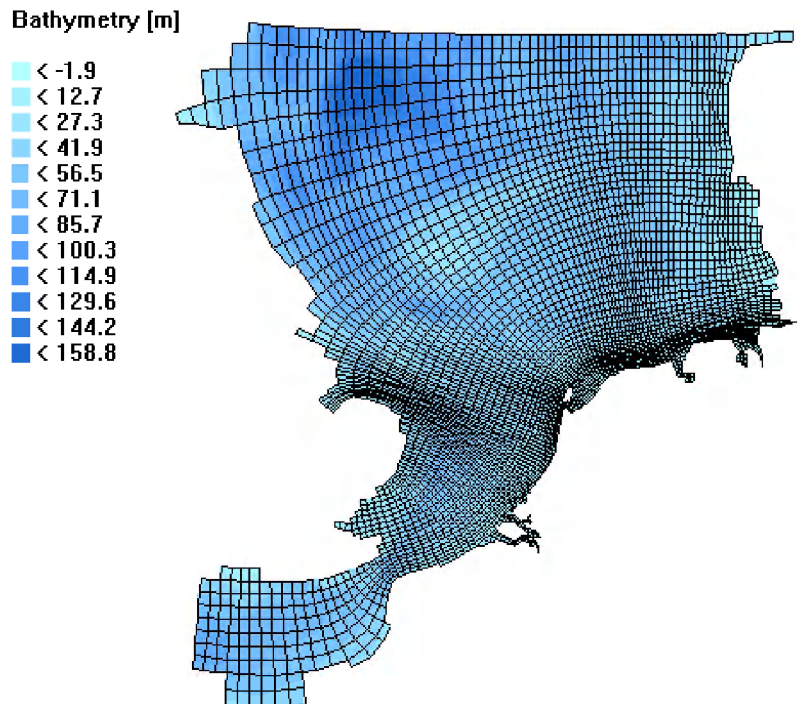
Figure 7. ZUNO-GROF grid, the complete southern part of the North Sea.

Using the results of the hydrodynamics as described above, Delft3D-WAQ was re-run at a time step of 5 minutes, to increase the resolution of the map outputs for total inorganic matter to 2 hours. The SPM model (Delft3D-WAQ) calibrated during the VOP SLIB 2006 (van Kessel and Brière 2006) project was used. It is identical to the ZUNO-GROF hydrodynamic model, without any aggregation, covering the complete southern part of the North Sea. To reproduce the current velocity, salinity and bed shear stress at a higher resolution, these parameters were output as part of the Delft3D-WAQ runs by including the coupling files as segment (forcing) functions.

In the framework of studies for Maasvlakte-2 and Flyland, a calibration of silt transport using the ZUNO-GROF model has been carried out (e.g. Thoolen et al 2001, Boon and van Kessel 2001, Boon and Winterwerp 2001). The optimal parameter settings from those studies were used as a starting point for the calibration of this model. It includes a two-fraction description of sediment characteristics, the sum of which is TIM (Total Inorganic Matter). This is equivalent to SPM (Suspended Particulate Matter) when the organic contribution is neglected. For North Sea SPM the two parameters do not differ much. In the early attempts at HSM within Belgian waters (section 3.2.3.2.1), turbidity values derived from MERIS satellite measurements since 2003 were used (data MUMM). ZUNO-GROF data were used to model the distribution of larger specimens over the entire Belgian-Dutch area.

Wave effects were also taken into account in the as the bed shear stress induced by waves results in sediment resuspension during storms. Wave parameters were computed on the ZUNO grid using SWAN. The presence of waves in the model further improved the simulation of seasonal effects on the SPM concentration along the Dutch coast. For details on the sediment fractions and boundary conditions (tide, salinity, meteorology, discharges and loads) and calibration of this model, we refer to van Kessel and Brière (2006). The calibrated model gives, in general, satisfactory agreement between numerical results and field observations, regarding a number of aspects such as the concentration distributions at specific locations and averaged

over a larger area. Moreover, this agreement holds for both winter conditions and after periods of significant wave activity, and for calm weather periods, exhibiting seasonal patterns, which can be simulated with the model.

The chlorophyll-a and POC concentrations were derived from the Southern North Sea Ecological model (ZUNO-GEM/BLOOM), based on the ZUNO-GROF model hydrodynamics of 10 layers. It consists of a combination of the above described hydrodynamic and water quality models (Delft3D-FLOW and DELFT3D-WAQ), and a phytoplankton model (BLOOM), run at a time step of 30 minutes. BLOOM is a model for predicting the dynamics of phytoplankton communities (see Los, 2009 for details).

For each year of simulation (1996-2003) the calculated data are aggregated to the median value, 10th-percentile and 90th-percentile within each model grid cell. 10th and 90th percentiles are defined as follows: the variable is lower than the 10th percentile during 10 percent of the year and higher than the 90th percentile during 10 percent of the year. For each year of simulation data between 1st of March and end of June were selected to estimate the 90th percentile of Spring chlorophyll-a and Spring particulate organic matter. An overview of all parameters is given in Table 1.

Parameter	Unit	Source
Current velocity	m s^{-1}	ZUNO-GROF (Delft3D-FLOW)
Current direction	degrees	ZUNO-GROF (Delft3D-FLOW)
Bed shear stress	N m^{-2}	ZUNO-GROF (Delft3D-FLOW)
Suspended matter (TIM)	mg l^{-1}	(MERIS measurements – MUMM) ZUNO-GROF (Delft3D-WAQ)
Chlorophyll-a	µg l^{-1}	ZUNO-GROF (GEM/BLOOM)
Spring Chlorphyll-a	µg l^{-1}	ZUNO-GROF (GEM/BLOOM)
Particulate organic matter (POC)	mgC m^{-3}	ZUNO-GROF (GEM/BLOOM)
Spring POC	mgC m^{-3}	ZUNO-GROF (GEM/BLOOM)

Table 1. Parameters derived from the model simulations

The model output files, so-called map files, were converted to ASCII grid files using the program" HabitoolGridCreate" (Jeroen Wijsman, IMARES and Lou Verhage, Deltares). Cell size and left lower corner were set equal to the bathymetry and sediment files. For each grid cell the average value of all parameters was calculated over the years 1996 to 2009 as input for the Habitat Suitability Modelling. Output is in the Dutch RD coordinate system (Rijksdriehoeksmeting). The projection was converted to UTM31NWGS84 in ArcGIS

3.2 Ecological features of *E. directus* in the Belgian-Dutch North Sea

3.2.1 Introduction

The native range of this species is the Eastern American coast from Florida to Labrador (Richards, 1938). Von Cosel (2009) reviewed information about life history traits of *E. directus*. The preferential habitat of this species in its native area was described to be gently sloping beaches with mobile sand, at a depth 5-8 meter, associated to surf clams, but the species can be found intertidally as well as deeper (20-30 m) in mud and gravel, pointing at a large range of potential sub-optimum

habitats, which partly explains its success. In the Voordelta area, the species is fished mainly in the range 8-13 meters, near gullies where food is available (Wijsman et al, 2006). It is found in clean as well as muddy and coarser shelly sands, in dynamic as well as more sheltered waters. "Banks" of this species may occur in areas well-suited for filter-feeding organisms, where food is accumulated by local hydrodynamics (Muir, 2003). *Ensis directus* is found in fully marine conditions and is absent from brackish water areas (Wijsmans et al, 2006). The species is able to carry out passive as well as active migrations at the adult stage through extruding from the sediment. It can dig rapidly in the sediment down to about 50 cm depth (Richards, 1938; Tulp et al, 2010). It is generally acknowledged that the species found an empty ecological niche in European waters (e.g. Von Cosel, 2009). High densities are reported in the literature. A density of more than 15,000 individuals has been observed at the settling stage (Armonies and Reise, 1999). Densities of several thousands juvenile individuals are often observed.

Spawning occurs in may or june, with fertilization occurring in the water column after release of sperm and eggs; sexes are separate (Kenchington et al, 1998). In the Wadden Sea, it was shown that a second spawning may occur in July/August (Armonies, 1996; Wijsmans et al, 2006). The earliest larva measures about 92.78 μm (smallest observed larvae: 80-85 μm length; Von Cosel, 2009). The umbo appears at a size of 115 μm in length. Pelagic larvae settle down after having spent 10 – 27 days in the water column (10 days at 24°C), at a length of 210-270 μm. The duration of the pelagic larva stage depends upon food availability and water temperature (the warmer the water, the shorter the larval phase). Armonies (2001) indicated that considerable distances must be covered by the pelagic larvae, as the species spread in the NorthEast Atlantic coasts at on average 125 km/year northward and 75 km/year southward. However, recent spatially explicit modeling of pelagic larval dispersal also emphasized the strong influence of prevailing hydro-meteorological conditions on the larval dispersion path for the amphipod *Jassa herdmani* (Zintzen, 2007) and the common sole *Solea solea* (Savina et al, 2010), both of which exhibit comparable or even longer pelagic larval phase. A strong year-to-year variability can thus be expected to take place at the larval dispersal stage.

After settlement, the American razor clam grows fast (see Wijsman et al, 2006 and Von Cosel, 2009, and references therein). In their first year, specimens of the German North Sea measure between 1 and 6.4 mm; at year two, 6 to 12.6 mm; at year three, 12.4-14.4 mm and at year four, 13.3 – 15.2 mm. In the U.S.A., the species reaches a maximum length of over 200 mm (Von Cosel, 2009). In Europe, the maximum size so far observed is 184 omm (within Belgian waters; Severijns, 2001). More to the North, along Dutch coasts, specimens seem shorter with a maximum observed of 170 mm. In the German Bight, the maximum estimated length based on the equation of Von Bertalanffy is 166.8 mm. In the North Sea, specimens over 5 years are rarely observed. However, in its native area, one observation pointed at a specimen of 170mm in length with a very thick shell and an estimated age of 20 years (Von Cosel, 2009).

In Belgian waters, specimens of *E. directus* have been regularly encountered in macrobenthic sampling programmes since its introduction. Thus far, these have been generally pooled at the genus level, partly because conventional sampling gears tend to damage them, preventing accurate determination at the species level. On the other hand, macrobenthic community analysis has been focused on local species, excluding the invader from the data-sets. The population structure and distribution as well as environmental preferences remain unknown. In this chapter, we investigate the geographic distribution, size distribution, biomass and

environmental preferences of *E. directus* in the BPNS. Results are further extended to the Dutch zone, where data resulting from regular monitoring programmes are available since 1995.

3.2.2 Material and methods

3.2.2.1 *Size composition of the* **E. directus** *population*

Due to their shape, a high proportion of razor clams are damaged by regular sampling devices. Consequently, the determination of length-based size distribution in the target population, which is useful for fishery assessment purposes, may be hampered. Morphometric measurements were therefore carried out on intact specimens collected in the survey of winter 2010 (see section 3.1.2.4) using a Vernier caliper at a precision of 0.1 mm. The anterior and posterior shell widths were measured on a sub-sample ensuring coverage of a large range of lengths (Figure 8). An additional measurement of shell width was taken at the middle of the ligament, because in few samples, shells were broken both on the anterior and posterior tips. From the measurements, three relationships were thus determined: Total Length (TL) – anterior shell width (SW) (n=427); TL – posterior shell width (FW) (n=621); TL – Ligament shell width (LW) (n=236).
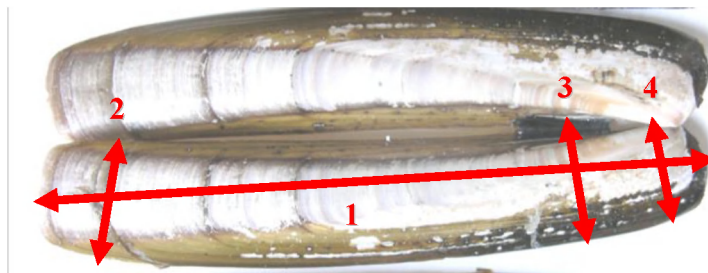
Figure 8. Morphometric measurements carried out on the shell of *E. directus*. 1. Maximum shell length; 2. Anterior (siphon) shell width; 3. Shell width on the ligament; 4. Posterior (foot) shell width. Photo: F. Kerckhof.

All shell width measurements of damaged specimens were converted to total length using these relationships. Total length was always measured on intact specimens (n=2,329 specimens - 53% of the total catch). When large amounts of small specimens (*i.e.* hundreds, total length ≤ 6 cm) were present, a minimum of 50 animals were measured and the size distribution was extrapolated to the total count in the concerned size range. When occurring in these samples, the fewer larger specimens were also measured and added to form the overall count.Weight conversions were carried out through combining the relationship between SW (mm) and fresh weight (g), determined by Goudswaard et al. (2010; $Y_{(g)}=0.0014*X_{(mm)}^{3.2674}$), and the determined TL-SW relationship. The average weight was then determined for every centimetric class center. In every sample, density and weight per square meter were subsequently calculated for all length classes, acknowledging a sampled surface of 0.082 square meters. When more than one sample was obtained at the station, average values were considered. The morphometric measurements carried out enabled to determine total length of all damaged collected specimens on the basis of shell width at different positions on the razor shell (Figure 9). The conversions were subsequently applied to the entire data-set to obtain the full length distribution of the population.
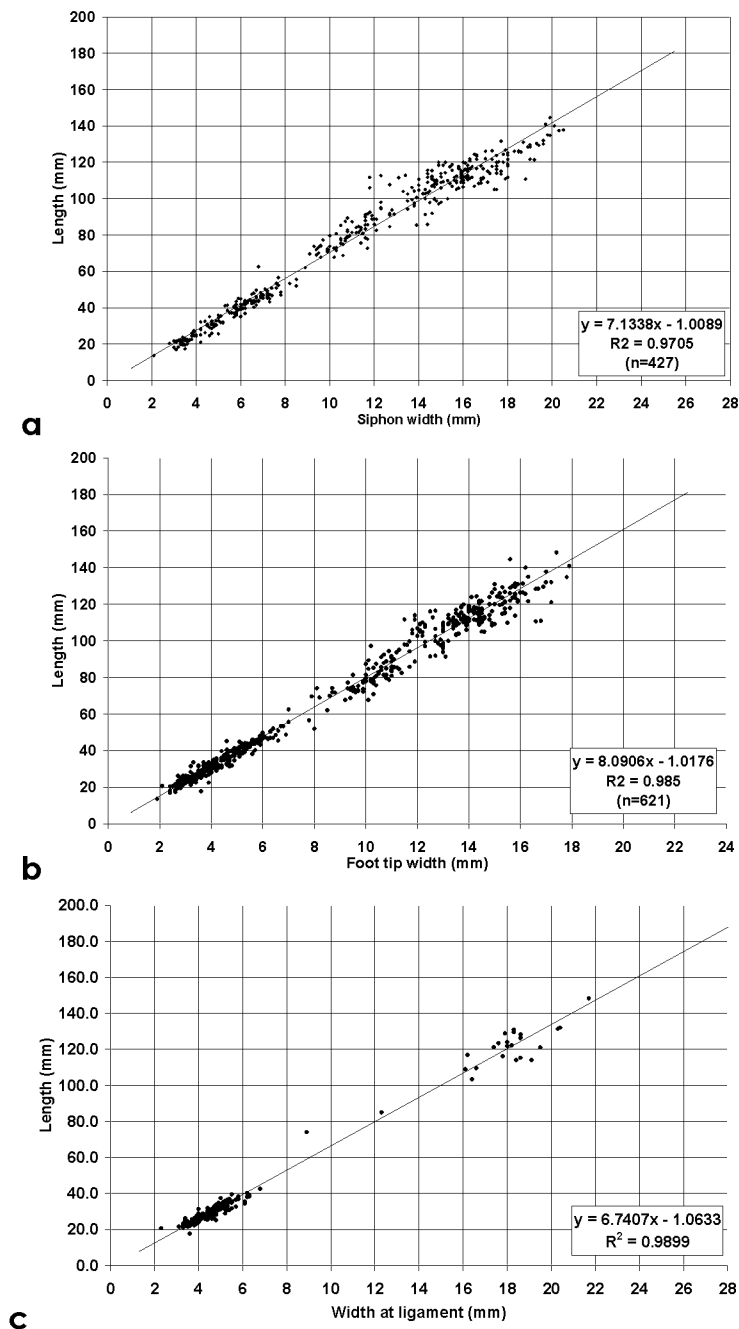
Figure 9. Linear relationships between anterior shell width and total length (a), posterior shell width and total length (b) and shell width at the ligament and total length (c), as determined from morphometric measurements of specimens collected within the BPNS in winter 2010 (EnSIS survey).

Most collected specimens ranged between 20 and 140 mm in length (maximum observed: 148.3 mm). The relationships were further used to determine the relative size-class abundances in the ILVO-SCHELP data-set based on anterior shell width measurements (2008; see section 3.1.1.2.). The relative distribution of sizes displayed by measured specimens was extrapolated to the entire sample content to overcome the problem of non-measured specimens.

3.2.2.2 *Weight conversions*

Combining the anterior shell width – weight relationship determined by Goudswaard et al (2010) and the aforementioned anterior shell width – total length relationship, it is possible to evaluate the weight of a specimen based on its total length (Figure 10). This conversion makes it possible to evaluate the biomass of a given size range for *E. directus* for various purposes in the project. To that purpose, the data were pooled by centimetric classes and the average weight of every class center was determined (Table 2). These values were used to convert abundances into size-based weights in both SCHELP (2008) and EnSIS (2010) surveys.
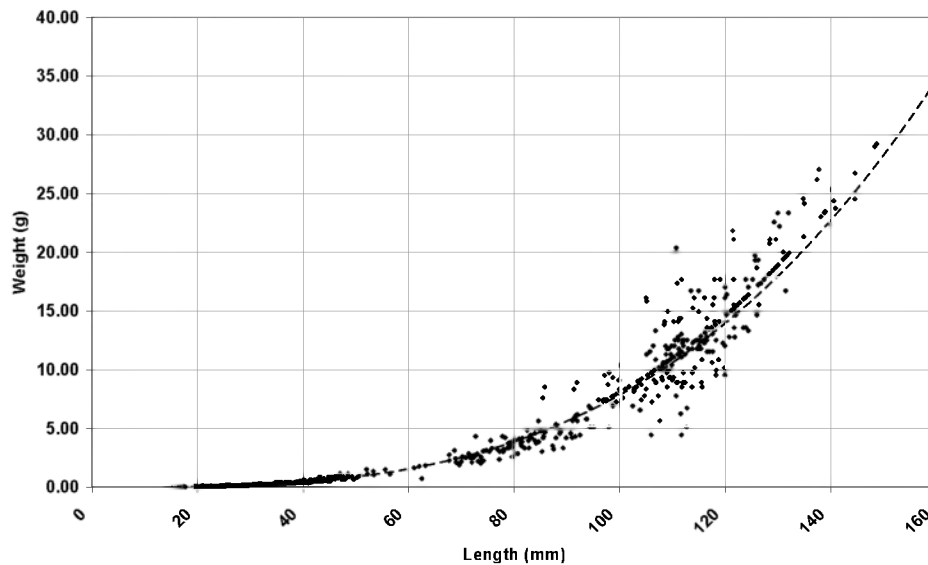
Figure 10. Relationship between total shell length and fresh weight ($Y=0.000004*X^{3.1621}$; $R^2=0.9944$).

Length size-class (cm)	Size-class center (mm)	Fresh weight of class center (g) ($W=0.000004*L^{3.1621}$)	Length size-class (cm)	Size-class center (mm)	Fresh weight of class center (g) ($W=0.000004*L^{3.1621}$)
1-2	15	0.02	10-11	105	9.85
2-3	25	0.11	11-12	115	13.13
3-4	35	0.31	12-13	125	17.09
4-5	45	0.68	13-14	135	21.80
5-6	55	1.27	14-15	145	27.32
6-7	65	2.16	15-16	155	33.74
7-8	75	3.40	16-17	165	41.11
8-9	85	5.05	17-18	175	49.52
9-10	95	7.17			

Table 2. Average weight of centimetric size-classes of *E. directus* used for size-based biomass assessment.

3.2.2.3 *Mapping methodology*

Distribution maps were created using the software ArcGIS. Proportional symbols were first used to reflect the actual value of density of *Ensis* at the location sampled. Stacked bar charts show both the proportions between size-classes and enable the comparison between records displaying extremely high, low or even zero values to be carried out. We applied an Inverse Distance Weighing (IDW) interpolation

method to predict values for pixels in a raster (the pixel is a square with sides measuring 200m). IDW estimated cell values by averaging the values of sample data points in the vicinity of each cell. The search radius was set at 3,400 m and a power of 2 was used. The distance was chosen based on the sampling intensity in 2008 and 2010. All the measured points that fall within the constant radius of 3.4 km were used in the calculation of each interpolated cell. For the data-set of 2008 (ILVO-Schelp campaign) the measured points are spread equally; the input sample points are less regularly spaced for the data set of 2010. Comparison of both resulting maps must thus be carried out taking into account the distribution of sampling stations. A higher power puts more emphasis on the nearest points, creating a surface that is less smooth but has more details.

3.2.2.4 *Habitat Suitability Modelling*

Habitat suitability models (HSMs) reveal information on where the species can potentially be found. The occurrence of a species is explained by the environmental variables (the habitat). When full-coverage maps of the environmental variables are available these models can be used to construct a habitat suitability map of the species. Different modelling techniques have been developed to estimate the potential habitat of a species. Depending on the type of input data, two types of HSMs can be discerned: those based on presence/absence data and HSMs based on presence-only data. Presence/absence data are commonly used for HSMs, but this includes some presumptions about the information. Often the absence of a *Ensis directus* is not 100% sure for different reasons: the species shows a patchy or ephemeral distribution and may not be present at the time of sampling although the habitat is suitable for the species, or the species may not have obtained its full range because of a disturbed environment or because it is an invasive species. Therefore, a presence-only modelling technique is preferred. Maxent has proven its better predictive capacities compared to other presence-only modelling techniques in several independent cases (Hernandez *et al.*, 2006, 2008; Hijmans and Graham, 2006; Pearson *et al.*, 2007; Sergio *et al.*, 2007; Carnaval and Moritz, 2008; Ortega-Huerta and Peterson, 2008; Benito *et al.*, 2009; Roura-Pascual *et al.*, 2009; Reiss et al, 2011) and may compete with or even outcompete presence/absence modelling techniques such as boosted regression trees (BRT), generalised additive models (GAM), generalised linear models (GLM) and multivariate *additive regression* splines (MARS) (Elith *et al.*, 2006; Wisz *et al.*, 2008). These good predictive capacities have been attributed to the l_1-regularisation which prevents the algorithm from overfitting. Other models often do not apply any form of regularisation, and this can cause the observed difference in predictive performance (Gastón and García-Viñas, 2011). A software related advantage is that it allows computerising the calculation of thousands of HSMs by running batch-files. Therefore, Maxent was applied to create habitat suitability models. In spite of these promising features, Maxent models seem to have two major drawbacks: the models may fail to make general predictions (Peterson *et al.*, 2007) and the models may be inaccurate in the presence of biased data (preferential sampling) (Phillips *et al.*, 2009). In this research, these drawbacks are tackled in various ways.

Species with specific habitat requirements and a small spatial range are generally easier to model than common species (Segurado and Araújo, 2004; Evangelista *et al.*, 2008; Reiss et al, 2011) because widely distributed species are not restricted to specific habitats. However, knowing where to find high densities of a common species may be of interest to fisheries. Therefore, a threshold was applied to the density of the species instead of using presence-only data.

In order to reduce over-fitting the dataset was split in five subsets. These subsets were used in a fivefold cross-validation. These cross validation sets were also separated in space: each point from each subset was at least 1 or 5 km from any another point in another subset. In this way, the inflation of the validation statistics (AUC) due to spatial autocorrelation is reduced. The effect of preferential sampling (i.e. some areas where visited more frequently than others) was investigated by comparing null models sampling the actual sampling stations with null models sampling the entire mapping area (Raes and ter Steege, 2007).

3.2.2.4.1 Presence and density data-sets

Two independent databases were analysed: 1) the Joint UG-ILVO Macrobenthos database (see section 3.1.1.1), where Presence/Abence was considered, and 2) data from the EnSIS sampling (winter 2010; see section 3.1.2.4.) hereafter called the "EnSIS database", where densities were considered. The data from the Macrobenthos database was sampled with a Van Veen grab which has a penetration depth of about 7-10 cm (Degraer S., pers. comm.), and the MUMM data with a box corer which has a penetration depth of about 30 cm. Since *E. directus* can easily withdraw up to 50 cm in the sediment (Tulp *et al.*, 2010), actual absence at a station cannot be ruled out when the species is not found in a sample (false zeroes).

The Macrobenthos database contains 869 sampling stations within the 12 miles zone (Figure 1). In 201 stations *Ensis* specimens were found. The EnSIS database holds data of 228 sampling stations, in 137 stations of these stations *E. directus* was found. This database was used prior to the individual measurements described in section 3.2.2.1.
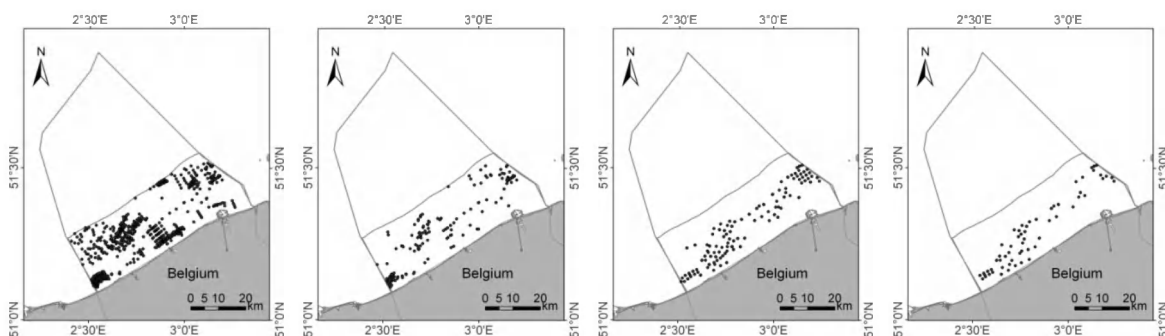
Figure 11. Sampling stations (•) from the MacroDat database (A) within the 12-miles zone of the BPNS and stations where Ensis has been observed (B). Sampling stations from the 2010 box-core data (C) and stations from the 2010 box-core data where 1-year old specimens were found (D).

As explained in section 3.1.2.4, an initial distinction was made to enable distinguishing densities in two main size-classes, namely specimens <= 6 cm (corresponding to one year-old specimens, "D1") and older specimens > 6 cm (named "D2"). The D1-class was found in 94 stations and the D2-class in 78 stations. Only in 37 stations both size classes were found. The sediment data at hand captures the surface sediment composition, where the younger specimens live. It should be noted that the survival of the older, deep-burrowing species, may be influenced by deeper sediment conditions for which no data are available. Therefore, at this early stage, analysis focused on the 1-year old specimens.t

3.2.2.4.2 Density threshold data-sets.

At a later stage of the project, once morphometric measurements were performed and the size-distribution could be determined, HSM was also tentatively carried out above certain thresholds of densities for specific size classes. For the common scoter predation, the optimum size for foraging was determined as being in the range 3-9 cm (see section 3.4.2 for more details), and a density of 50 specimens per square meter was initially considered as minimum requirement for scoters. On the other hand, a unified Be-NL data-set was created using the aforementioned data-sets as well as the IMARES data-base for the years 2008, 2009 and 2010, considering maximum observed densities each year. The results obtained in the BPNS in 2009 (section 3.1.2.1) were included. For this data-set, modeling focused on specimens larger than 10 cm at a densitiy threshold of 10 specimens/m² or more.

3.2.2.4.3 Modeling

Based on the occurrence data we attempted to make two models:

1. A model for the Large Ensis specimens without threshold on density (625 occurrence samples)

2. A model for the large specimens with a density threshold. Whenever large species occurred in densities higher than 10 specimens per square meter, the species was considered to be present. This resulted in 103 occurrence samples

The list and abbreviations of used environmental parameters (see section 3.1.3) are given in Table 3.

Abbreviation	Description
Tsan	Meris measurements of annual Total suspended matter from 2003
Bath	bathymetrical data
chl1	average of the 0.10 percentile values of the modelled chlorophyll-a values in the period over de jaren 1996 - 2003. This can be interpreted as the minimum chlorophyll-a content (Ng.l⁻¹)
Dirm	average of the median values of the modelled direction of the current in the period 1996-2003. This can be interpreted as the average current direction (°N)
dir9	average of the median values of the modelled direction of the current in the period 1996-2003. This can be interpreted as the maximum current direction (°N)
Pocm	average of the median values of the modelled particulate organic carbon content (without chlorophyll) in the period 1996 – 2003. This can be interpreted as the average POC concentration (mg C.l⁻¹)
Poc1	average of the 0.10 percentile values of the modelled particulate organic carbon content (without chlorophyll) in the period 1996 – 2003. This can be interpreted as the minimum POC concentration (mg C.l⁻¹)
Tau9	average of the 0.90 percentile value of the modelled bottom shear stress due to waves and water movement in the period 1996-2003. This can be interpreted as the maximum bottom shear stress (N.m⁻²)

Table 3. Abbreviations of environmental parameters evaluated to explain observed distribution of *Ensis directus* (see also section 3.1.3).

3.2.2.5 *Acoustic seafloor mapping and characterization*

Mutlibeam recordings allowed processing of both depth and backscatter data; data grids were produced at a resolution of <=1m. Variation in backscatter (dB values) can relate to small-scale sediment and terrain variation, as also to bioturbation. Some species, when forming dense colonies, are also able to alter the acoustic response of the seafloor (e.g. Degraer et al., 2008; Van Lancker et al., 2012); these can be distinguished through small-scale terrain analyses. Slope calculations can be performed or rugosity (Benthic Terrain Modeller, ArcGIS tool) can be calculated from digital terrain models. Rugosity, calculated as the ratio of surface area to planar area, is a measure of terrain complexity or 'bumpiness' of the terrain (Oregon State University and NOAA Coastal Services Center). It assists in the identification of areas with higher biodiversity.

3.2.3 Results and discussion

3.2.3.1 *Population structure and distribution*

3.2.3.1.1 Population structure and distribution in Belgian waters

Population structure
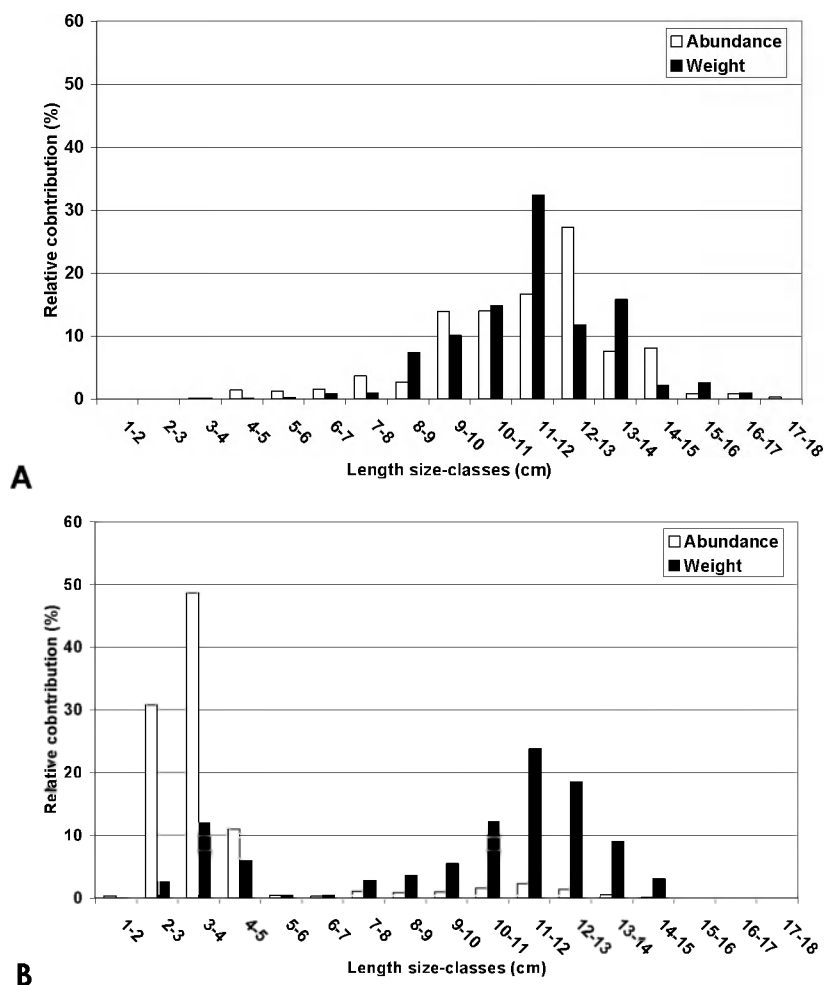
Figure 12. Relative abundance (%) of centimetric size-classes in the sampled population of *Ensis directus*. A: summer 2008, SCHELP survey (NB. "spat" not included). B: winter 2010, EnSIS survey

The two data-sets display important differences in the relative contribution of size-classes to the overall sampled population, mainly due to the prominent contribution of the first year class (recruitment spring 2009, <6 cm: >90 % of collected specimens, with a mode at 2-3 cm) in 2010 (Figure 12). This observation is consistent with that of Goudswaard et al (2010) in Dutch waters, highlighting major recruitment to have taken place in spring 2009 in the Belgian and Dutch waters (see also section 3.2.1.4). Specimens of 6-10 cm also represent a significant fraction of the population, pointing at the recruitment of 2008. In summer 2008, specimens of 6 to 10 cm (year class 1+) represented an important fraction of the population, suggesting that a successful recruitment also took place place in spring 2007.

These observations suggest that recruitment has been high in the area since at least spring 2007. When the juvenile (< 1 year) specimens smaller than 6 cm are removed from the 2010 data-set, the resulting population structure is very similar to that of 2008 (Figure 13), taking into account the fact that sampling took place at different seasons. In the last years, the population seems to have remained fairly stable and reproductively active. Noticeably, specimens over 15 cm (4-5 years) are rare.
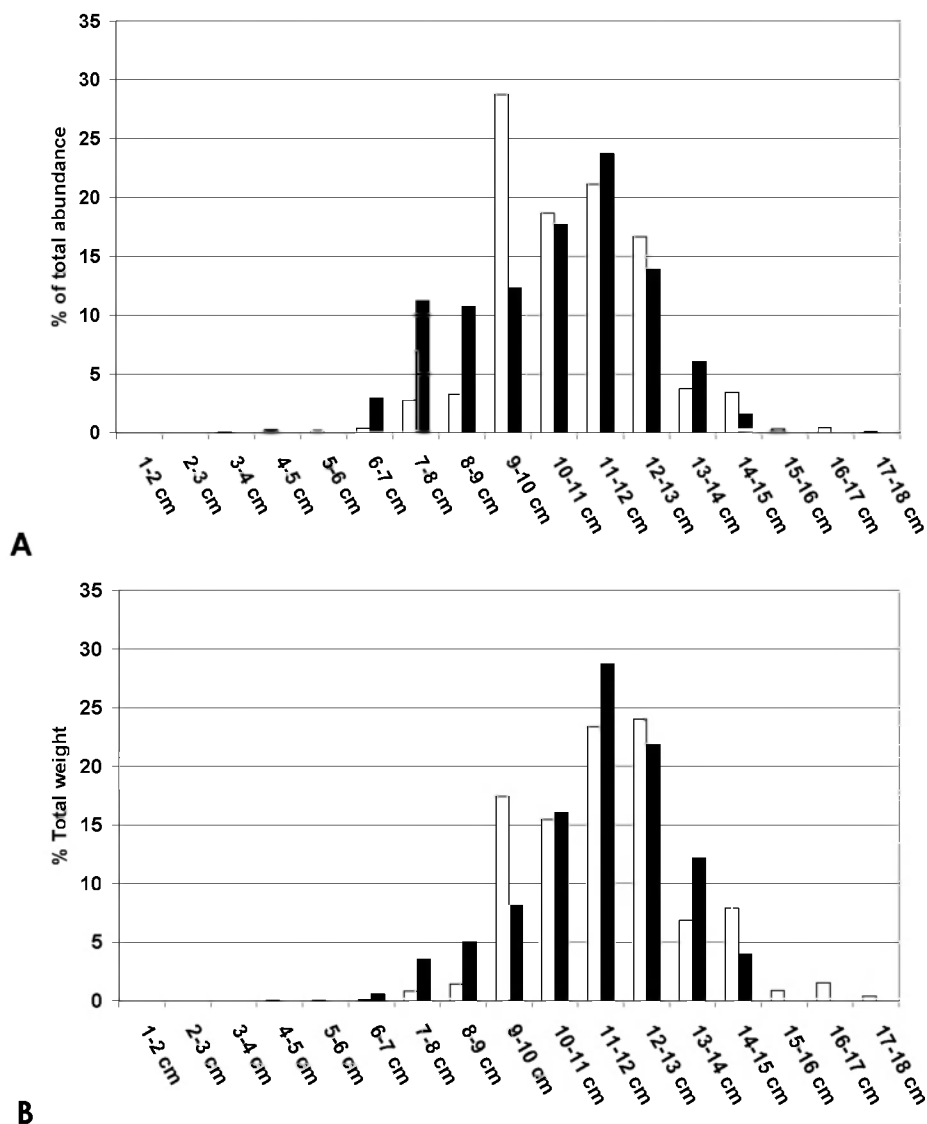
Figure 13. Comparison of population structure of *Ensis directus* in summer 2008 (white bars) and winter 2010 (black bars), when juveniles (<6 cm) are removed from the latter. A: relative abundance. B: relative wet weight.

Geographic distribution

Total densities display different patterns due to the overwhelming influence of juveniles in 2010 (Figure 14). Largest density patches of juveniles surprisingly occur just next to larger density patches of adults, northward.

a

b
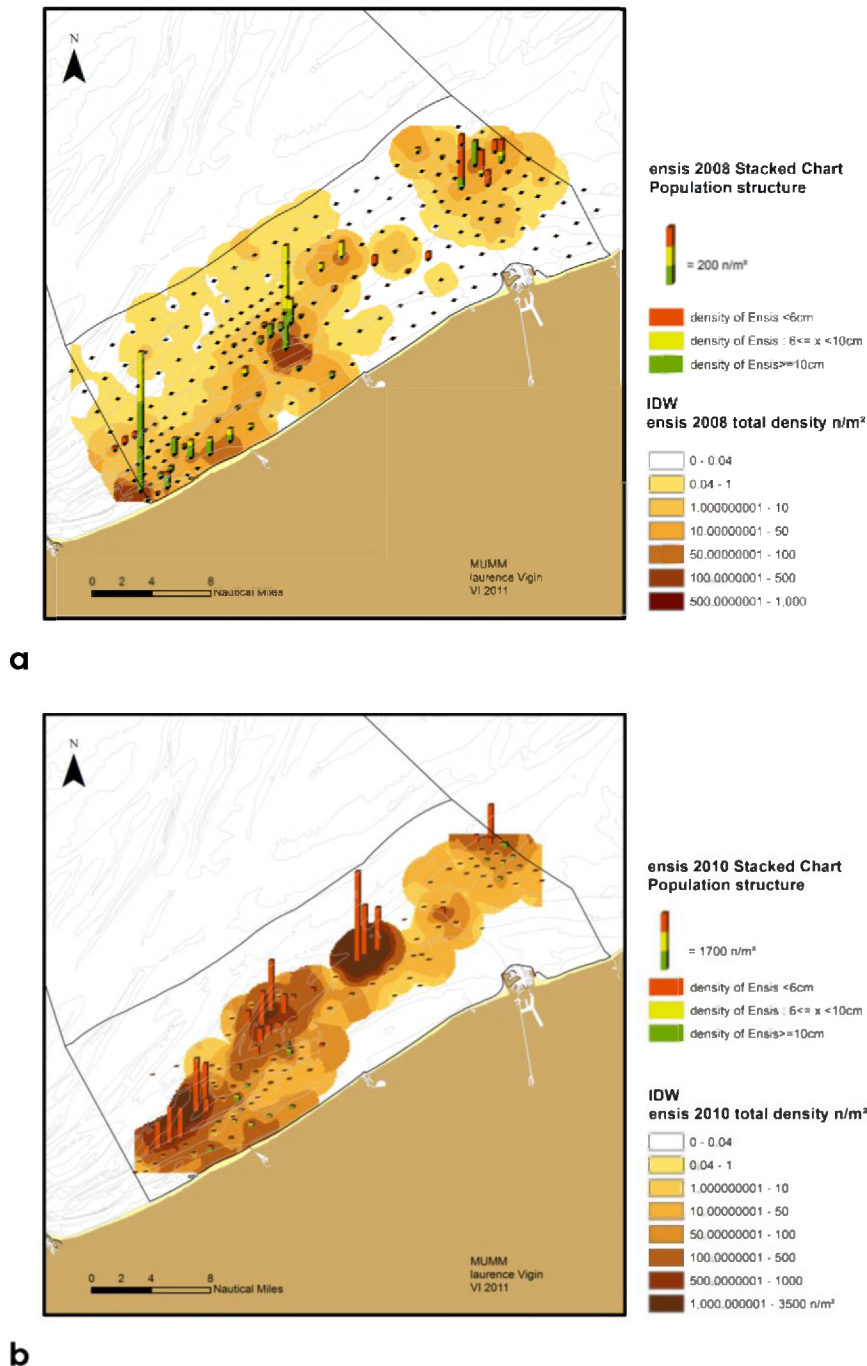
Figure 14. Interpolation maps (Inverse Distance Weighing; search radius 3.4 km) of total densities of *E. directus*, with size composition (three classes) shown as scaled stack bars. a. summer 2008 ; b. winter 2010.

Acknowledging that the pelagic larvae are capable of traveling tens of kilometers with residual currents (Armonies, 2001), the source of propagules is probably located elsewhere than in the nearby adult populations (e.g. French or Dutch waters). The distribution of juveniles near the adult patches would then mirror optimum habitat for

post-settlement survival, pointing at areas suitable and available for colonization. The larvae likely find a suitable ground to grow as older specimens die and are expelled of the sediment. Their reduced representation within higher adult density patches likely mirrors a lack of space for settlement and survival, or predation by adults.

The biomass distribution patterns are more similar (taking into acount differences in sampling intensity), due to the higher contribution of larger specimens (Figure 15). This comparison suggests that the geographic distribution of adult populations remained relatively stable in the period. These are mostly found in coastal waters along the Western coast, on the Oostende bank and north of the Vlakte van de Raan.

a

b
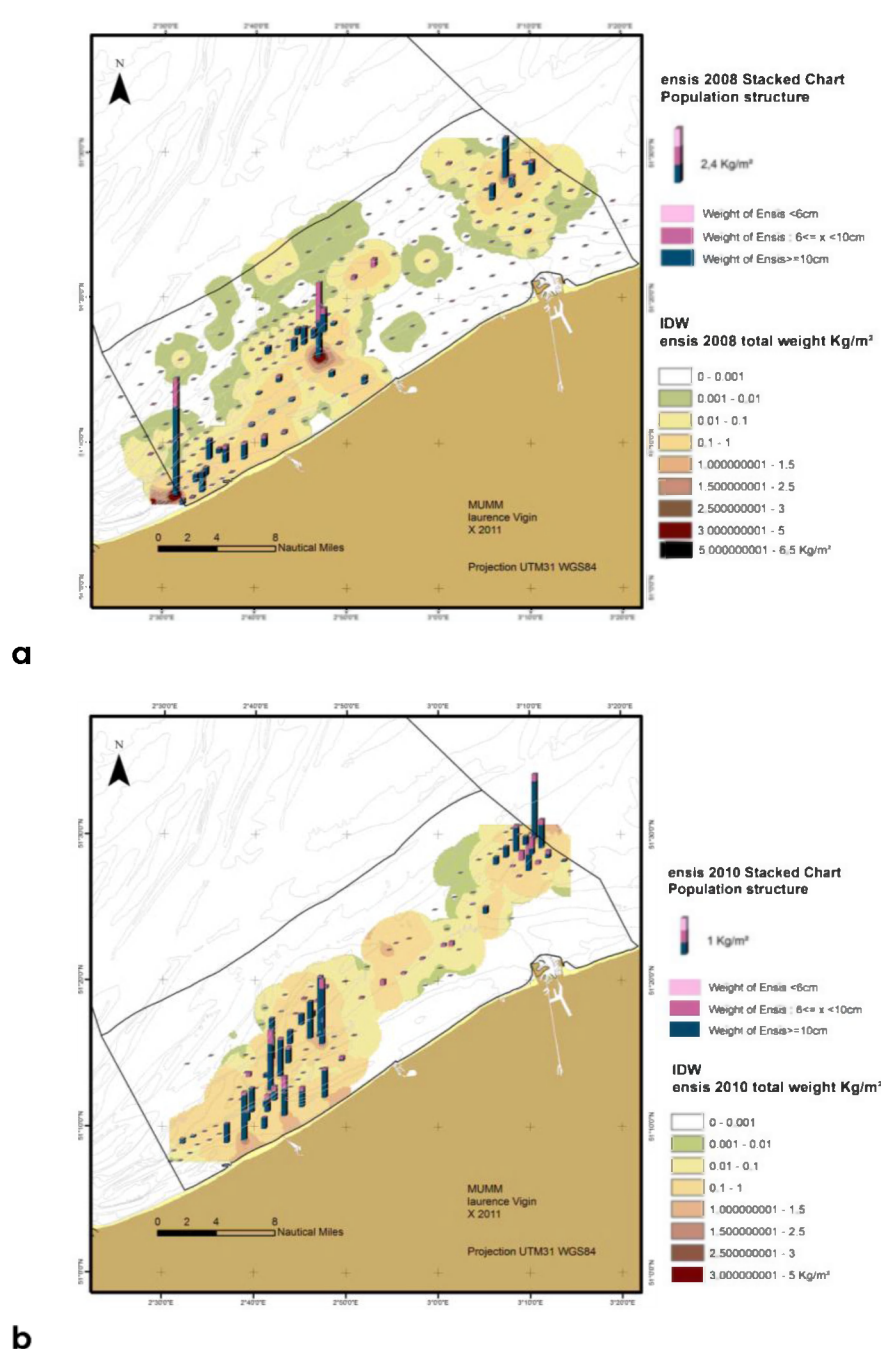
Figure 15. Interpolation maps (Inverse Distance Weighing; search radius 3.4 km) of total biomass of *E. directus,* with size composition (three classes) shown as scaled stack bars. a. summer 2008 ; b. winter 2010.

This distribution partly matches the distribution of the *A. alba* community (Degraer et al, 2008), but noticeably the two surveys indicate a lack of adults north of the Wenduine bank, roughly between Oostende and Zeebrugge. Adult specimens were observed in this area in 2003 in this area (F. Kerckhof, pers. obs.). An important setlement of juveniles was however observed more to the north. Targeted sampling in the coming years should reveal whether the observed juveniles can survive in this area. Thus far, the surveys suggest that it is less suitable for an adult population to develop.

3.2.3.1.2 Population structure and distribution in Dutch waters

The population structure was determined for Dutch waters for springs 2008, 2009 and 2010, on the basis of measured siphon shell widths converted to lengths (Figure 16). The data evidence important recruitments to have taken place, which resulted in high proportions of juveniles, with a mode at 4-5 cm size-class.
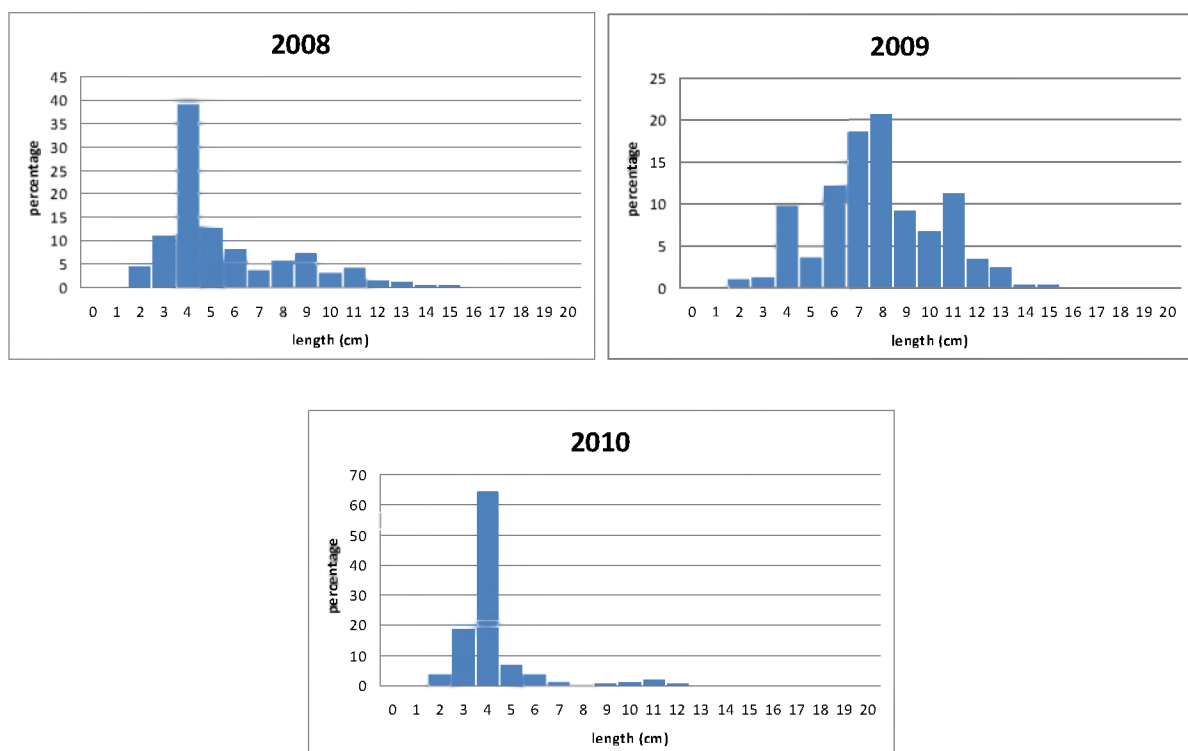
Figure 16. Population structure of *E. directus* (relative abundance of size-classes) in the Dutch waters for the years 2008, 2009 and 2010.

From the start of the time-series onwards, *E. directus* was found all over the coastal area. There were, however, large spatial differences in the density and biomass of *Ensis* along the Dutch coast (Figure 17). Highest densities were found in the Voordelta and adjacent to the Wadden Sea, especially in the last years. These areas have also displayed the most dramatic increase in total macrobenthic biomass in the 2000s.
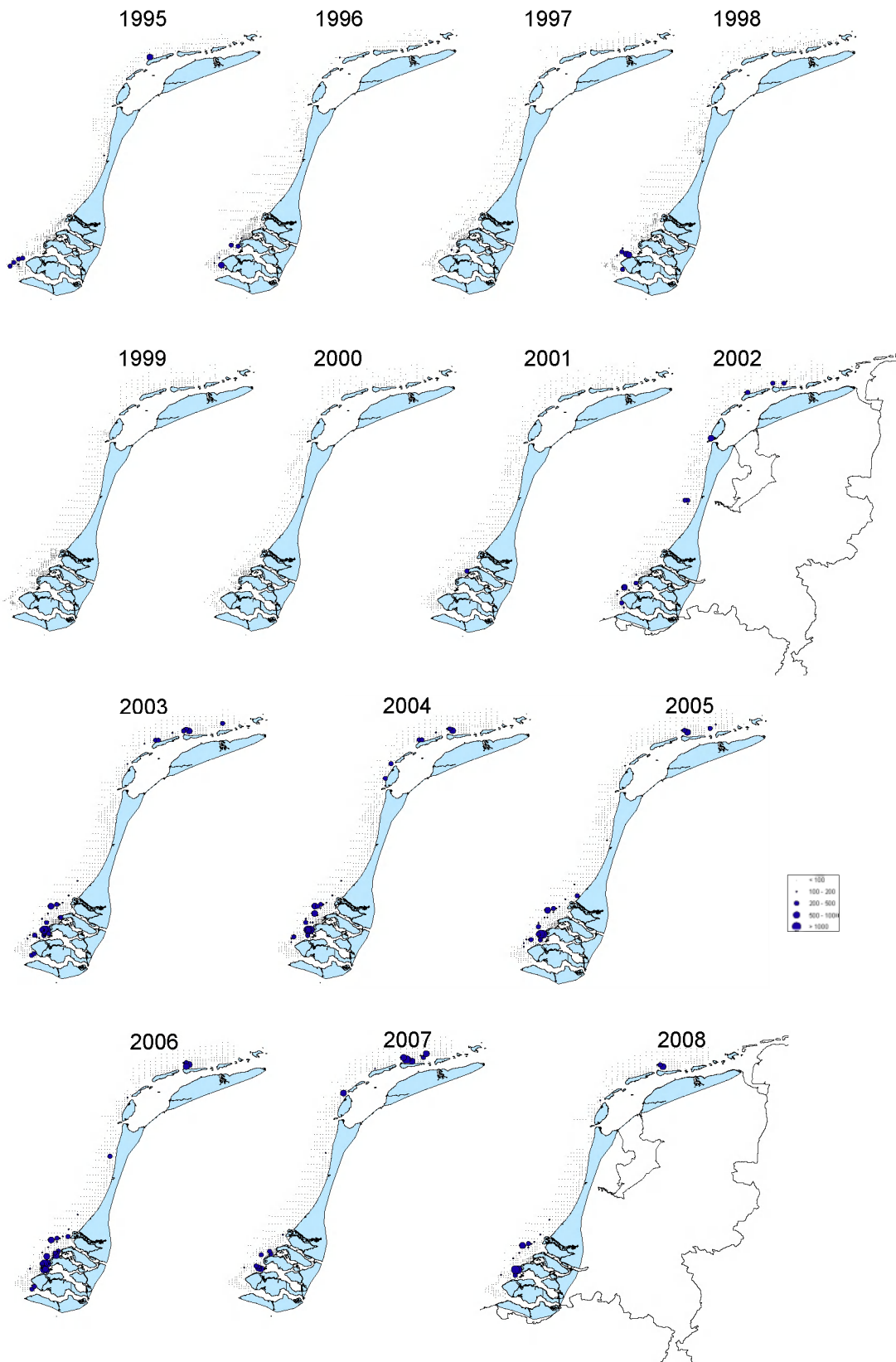
Figure 17. Spatial distribution of density (ind m⁻²) of *Ensis* sp. in the Dutch coastal waters in the period 1995-2008

3.2.3.2 *Environmental preferences: habitat suitability modelling*

3.2.3.2.1 Distribution in the Belgian part of the North Sea

The randomisation exercise (Figure 18) points out that the sampling strategy of the Macrobenthos database is less biased than the EnSIS database. This makes sense, since the recent EnSIS survey was targeted towards larger densities of *E. directus*. Areas where the species is less likely to be found are thus under-sampled.
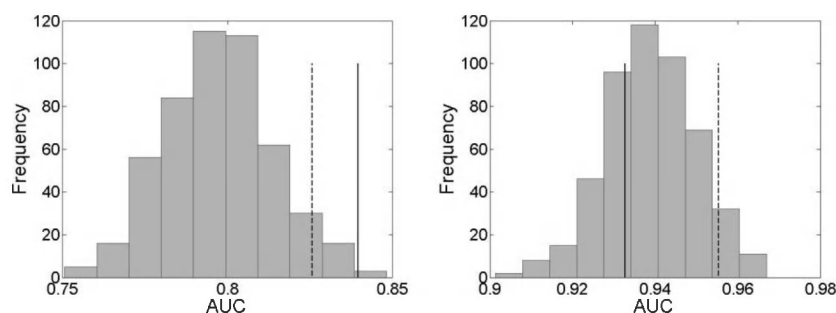
Figure 18. Histograms of the randomisations of the Macrobenthos data-base (left) and the EnSIS data-set (right). The randomisations are based on the actual sampling stations. The 95% quantile value (dotted line) and the AUC of the *Ensis* model (full line) are indicated.

The AUC of the *Ensis* model is 0.93, which is generally considered to be an excellent model (Parolo *et al.*, 2008), but notwithstanding this high AUC the HSM cannot be distinguished from a random model due to preferential sampling. Therefore, we will here focus on the HSM model derived from the data from the Macrobenthos database.

The three models with different distances between training and test data, i.e. 0km, 1km and 5km, were further refined with a variable and feature selection. The final 0 km model uses all features and 10 environmental variables in the final model. This results in overly complex relations between the variable and the model output. For instance two or three optimum values for the minimum bottom shear stress and median grain size are found. The reason why such an overly complex model is selected by cross-validation can be explained by over-fitting and spatial autocorrelation. Since there are no restrictions to the distance between the samples in the training and the test set, it is likely that the samples in the test and training set are spatially close to each other. Therefore the values of both the environmental variables and the output may be very similar for both datasets due to spatial autocorrelation. This means that, notwithstanding the cross-validation test, the model can still be over-fitted because of the similarity between the test and the training set. Hence, the Ensis1km and Ensis5km model will suffer less from over-fitting. Indeed, the 1 and 5 km model are much more realistic, as respectively 5 and 3 variables and only linear and quadratic features are selected (Table 4).

Variable	% contribution	
	Ensis1km	Ensis5km
Maximum bottom shear stress	27.4	39.7
Water depth	22.8	35.9
Sand Fraction	15.9	24.4
Maximum chlorophyll content	29.9	
Minimum chlorophyll content	4.1	

Table 4. Relative contributions of the environmental variables to the final Maxent model.

Both models, Ensis1km and Ensis 5km, select bottom shear stress, water depth and sand fraction as the most important factors (Figure 19). For the Ensis1km model the minimum and maximum chlorophyll content are selected as well. Previous research pointed out that the American jack knife clam is an opportunistic species, with little requirements regarding its environment. It prefers wave- and current-swept clean sands (Beukema and Dekker, 1995) with small amounts of silt (Kennish *et al.*, 2004), but it can also be found in muddy or coarse sediments (Armonies and Reise, 1999) and can thus be independent of sediment characteristics (Dauvin *et al.*, 2007). *Ensis directus* however has a limited tolerance to hypoxia and will thus avoid reduced sediments (Schiedek and Zebe, 1987). The positive relation between the sand fraction and *E. directus* is found in both HSM models. Silt-clay was not selected as a variable. The preference for moving sands (Kenchington *et al.*, 1998) and strong currents is not confirmed by the model; the model indicates that the maximum bottom shear stress should be limited to about 4 N.m^{-2} while a shear stress above 5 N.m^{-2} corresponds to the threshold of sand transport (Mangelsdorf *et al.*, 1990). Likely areas for colonisation are subtidal and intertidal areas (ICES, 2005; Ovcharenko and Gollasch, 2009). The models do not indicate the very nearshore and interidal areas as favourable because no data was available in these very shallow areas in the BPNS. Further specific sampling (using *ad hoc* platforms) should thus be carried out in the future to evaluate the suitability of this zone in the BPNS. Intermediate water depths between 12 and 23 m are optimal according to the models. This is partially in agreement with observations by Armonies and Reise (1999). The influence of chlorophyll a on the Ensis1km seems contradictory: the species is favoured by low minimum chlorophyll a values and high maximum chlorophyll a values. This could mean that the species is preferably found in areas with annually strongly fluctuating chlorophyll a values, however, there are no literature sources supporting this.
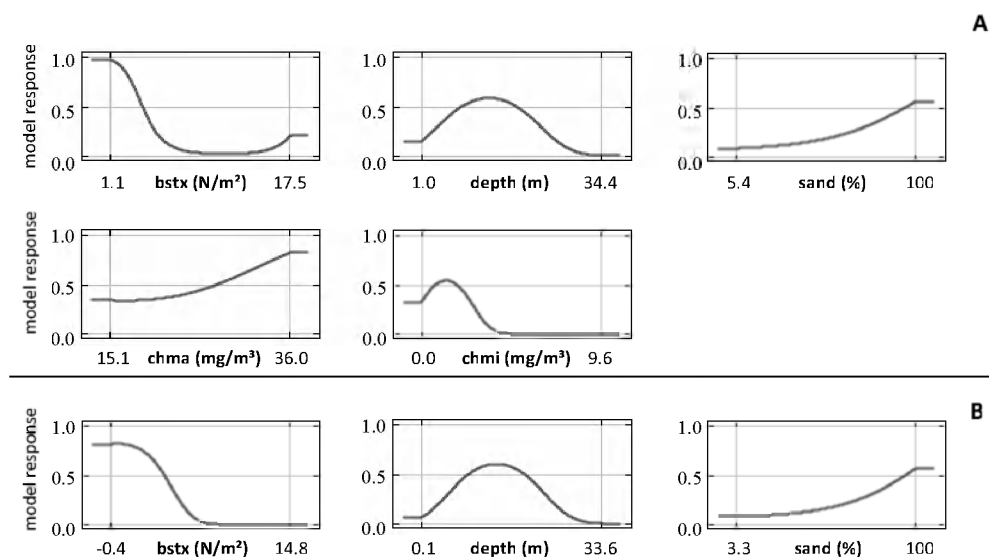
Figure 19. Relation between the environmental variables (see table 4) and logistic output of the Ensis1km (A) and Ensis5km model (B). Each of the curves represents a model created using only the corresponding variable. The minimum and maximum values of the environmental variable are delineated by a vertical line. Before the minimum and after the maximum horizontal markers indicate the starting point and the endpoint of the curve. These markers do not have an ecological meaning. Bstx: maximum bottom shear stress; chma: maximum chlorophyll a content; chmi: minimum chlorophyll a content.

The selected variables do not concord very well with findings in literature. This can be explained by some reasons. Firstly, a lot of the data in literature is collected in intertidal areas. In the present study, information on the intertidal is lacking. Secondly, all observations were taken into account, but *E. directus* is an opportunistic species which may occur in many different sub-optimum habitats. Its presence is therefore only slightly explained by the environmental variables. Thirdly, the accuracy of explanatory variables used in the model may be variable (*e.g.* average values were used, while locally a significant year-to-year variability might be expected to take place in this area where fine-grained sediments are abundantly represented).

The resulting map of the Ensis1km model is shown in Figure 20. This map matches those created on the basis of 2008 and 2010 data-sets, confirming that the distribution of the population remains fairly stable through time in the subtidal areas sampled.
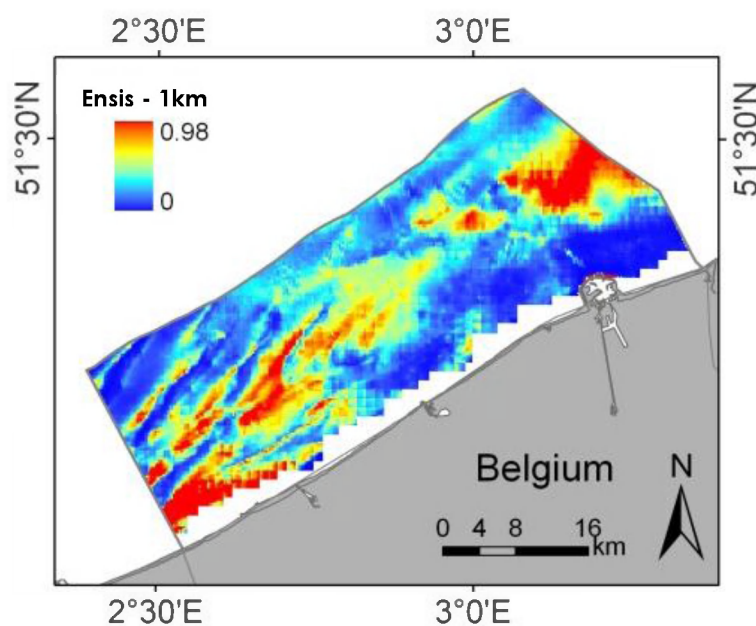
Figure 20. Resulting map of the Ensis1km habitat suitability model

The habitat suitability map indicates whether the habitat is suitable for the species but it does not reveal information about expected densities. Therefore, a density map for *E. directus* was also constructed based on the EnSIS survey. Since there were large differences between the densities, and only a few stations show high densities, the data was log-transformed. Two techniques were applied: OK and RK. For RK, a linear model was constructed but the relation between the log-density of the young cohort (D1) and the environmental variables was not strong. Two variograms were constructed: one for the OK and one for RK, based on the residuals of the linear model. The most important parameters of the variograms can be found in Table 5. The range of the OK variogram is 5.4 km, thus within this range there is a spatial dependency between the density of the samples. The sill and the nugget of the variogram of RK are larger than that of OK, which is unusual. In fact, the linear model should explain a portion of the variation in the data, which would result in a decreasing sill. This observation is also supported by the results in Table 5. Most of the quality parameters, except for the Spearman rank correlation coefficient, perform better for OK. Thus, OK performs better than RK. Hence, the vicinity of high densities can explain better the presence of high densities than the environmental variables,

and no strong relationship is found between the environmental factors and the density of the young cohort of *Ensis*. Spatial autocorrelation and spatial interpolation explains the densities of *E. directus* better than the environmental variables.

	Variogram parameters			Model validation parameters				
	Nugget	sill	Range (km)	MEE	RMSE	Pearson	Spearman	MAEE
Ordinary kriging	0.24	1.50	5.4	-0.02	0.59	0.83	0.84	0.45
Regression kriging	0.90	1.80	9.4	0.03	0.70	0.76	0.87	0.61

Table 5. Variogram parameters and model validation parameters calculated between the predicted and the real values of the test set.

With the data at hand it is not possible to make an area covering the map of the *Ensis* densities (Figure 21). The highest densities were found off the coast and these densities show a patchy distribution. The stations appear as spots on the variance map, since it is assumed that the variance is smallest at the sampling location. To construct relevant maps which give an indication of the species density, we suggest constructing habitat suitability models based on density thresholds, as was done for *L. conchilega* and some nematode species (see Degraer et al, 2009) . Therefore, size and density thresholds were identified for which an impact on the surrounding benthos is expected, or thresholds relevant to fisheries or sea bird analyses.
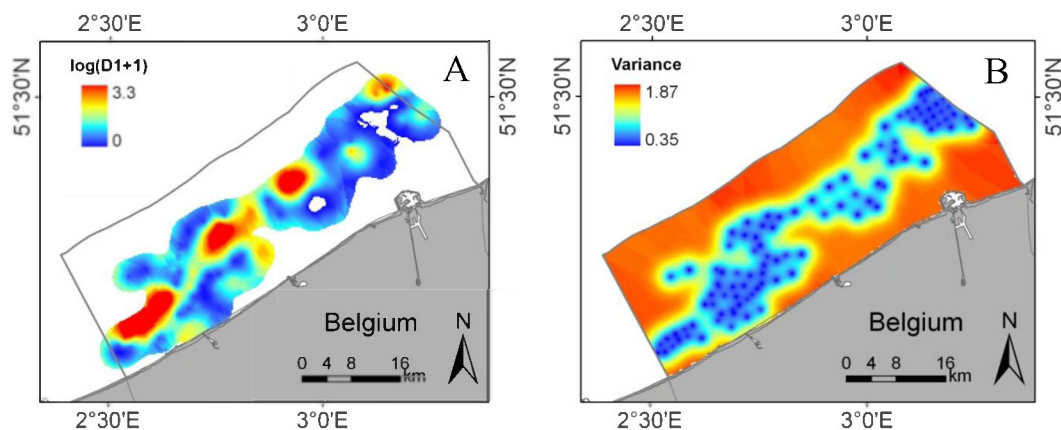
Figure 21. Map of log(D1+1), with D1 the density of the 1-year old cohort (ind.m^{-2}) (A). This map is restricted to a variance smaller than 1.5. On the right side the variance map is shown (B).

3.2.3.2.2 Distribution of adults over the whole Belgian-Dutch continental shelf

The environmental data merged for the entire Belgian-Dutch are were used to draw a larger scale HS map for adult *E. directus*. A randomisation test was performed to test the influence of sampling bias. Since the original data contained 103 samples (Ensis_Large10) and 625 samples (Ensis_Large). The randomisation exercise was run with the same number of stations. A model is considered to be significant if the AUC of the species model is larger than the upper .95 quantile value of the AUC's of the random models. This was not the case for Ensis_Large (Table 6), so this model will not be discussed. The resulting habitat suitability map of the large specimens (>10 cm) appearing in higher densities (>10 spec.m^{-2}) is shown in Figure 22. High densities can be found near the Southern part of the Belgian coast, near the Scheldt estuary and at the Northern part of the Netherlands. The result is fully in line with aforementioned statement that the area from the Belgian-French boundary up to the Delta area is very suitable for this species to thive.

	Upper 95 quantile value of the models of the randomly selected stations	AUC Ensis model
103 samples	0.875	0.934
625 samples	0.738	0.726

Table 6: 0.95 quantile value of the random models and the AUC of the actual *Ensis directus* data.
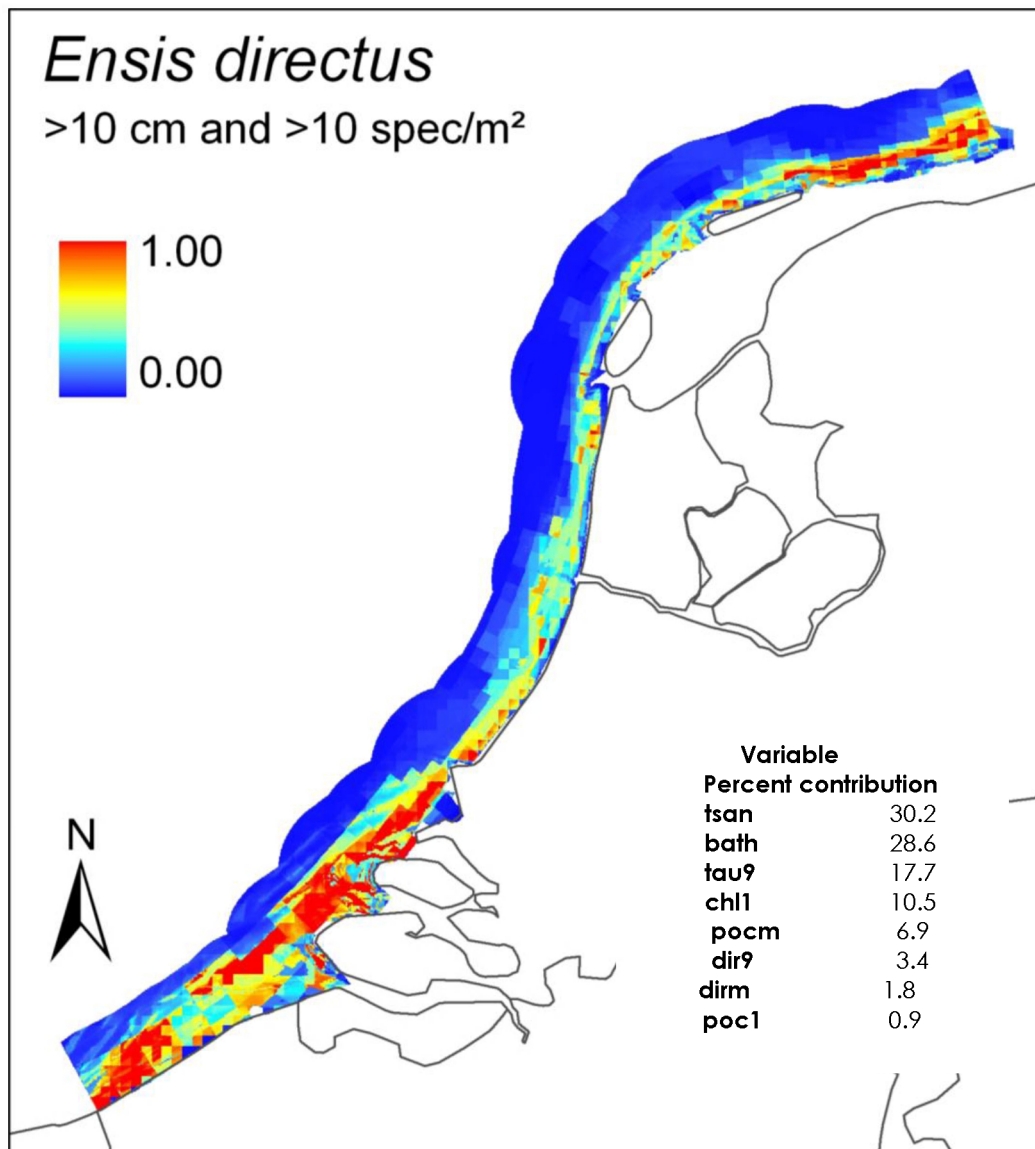
Figure 22. Habitat suitability map of the large specimens (>10 cm) displaying higher densities (>10 spec.m^{-2}). The Ensis density data were derived from maximum values observed across the years 2008, 2009 and 2010, with a density threshold for suitability set at 10 spec. m^{-2}. The relative contributions of the environmental variables to the final Maxent model are given (%). For explanations of the abbreviations, see Table 1.

The Eastern Belgian coastal waters close to the Westerschelde estuary are less suitable, which likely coincides with the occurrence of a turbidity maximum in this area, responsible for larger rates of tidally-induced deposition/resuspension of fines on the seafloor (Fettweis and Van den Eynde, 2003). This area is also hosting high levels of anthropogenic alterations, with in particular the presence of large port

infrastructures and artificially deepened navigation channels, which are suspected to alter the dynamic of fine-grained sediment (Fettweis et al, 2009). HSM maps were created on the basis of averaged environmental conditions. As the position of this turbidity maximum varies in function of dominant winds (Baeye et al, 2010), we can expect that the population structure will display a year-to-year variability at the border of this area, which cannot be determined yet.

The resulting response curves are not smooth (Figure 23). This can be attributed to remaining spatial autocorrelation between test and training set: the cross-validation files were set at 1 km distance. Results are not straightforward. In about 30% o the cases, moderate levels of turbidity are associated to suitable areas. Depth and local geo-morphology, reflected in the bottom shear stress, likely are important drivers to the concentration of adults. There seems to be a minimum requirement for abundance of phytoplankton to feed upon, with minimum levels of Chl a observed to explain 10% of the distribution.
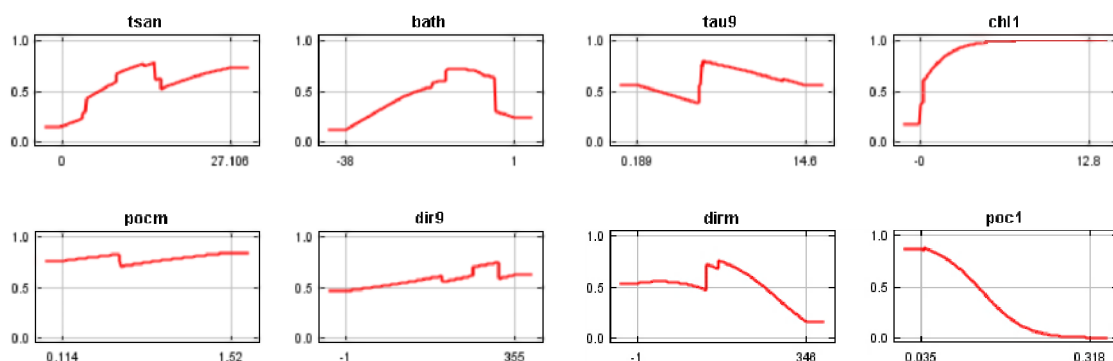
Figure 23. Resulting response curves

3.2.3.3 *Acoustic seafloor characterization*

The use of very high resolution multibeam echosounding technology (300 kHz) was evaluated for the mapping of the distribution of *E. directus*. Correlation with sampling data showed that *E. directus* can be mapped, *directly*, when the species occurs in higher densities (class 100-500 ind/m²). In these areas, an 'acoustic signature' can be established and habitat preferences can be derived. Both can be used for future assessments of the distribution of the species. Data were acquired along the Oostende Bank and the northern slope of the Vlakte van de Raan. Only for the latter area, an acoustic signature could be established (Figure 24).

3.2.3.3.1 Acoustic signature of *Ensis directus* beds

A distinct acoustic facies was observed in the areas where higher densities of *E. directus* were sampled (Figure 24 and Figure 25). Following Van Lancker et al. (2012), the imagery can be described as having a moderate reflectivity, a rough texture with (semi)-circular patterns. This signature cannot be correlated with known bedform types, nor with marks of anthropogenic disturbance (MESH signature catalogue; http://www.rebent.org/mesh/signatures/). Still, varieties of these facies have been associated, previously, with the occurrences of dense aggregations of the tubeworms *Lanice conchilega* (Degraer et al., 2008; Van Lancker et al., 2012) and *Owenia fusiformis* (Van Lancker et al., 2012). The dimensions of the patterns of *E. directus* are around 20 m in diameter, with a height of around 20-40 cm (Figure 26). The patches can be depicted by slope calculations on digital terrain models, with characteristic values of >2° (Figure 27, Figure 28 and Figure 29).

Anomalies within calculated grids of rugosity provide the same areal extent of a rougher seabed (see further). Images are shown from November 2010, when measurement conditions were optimal.
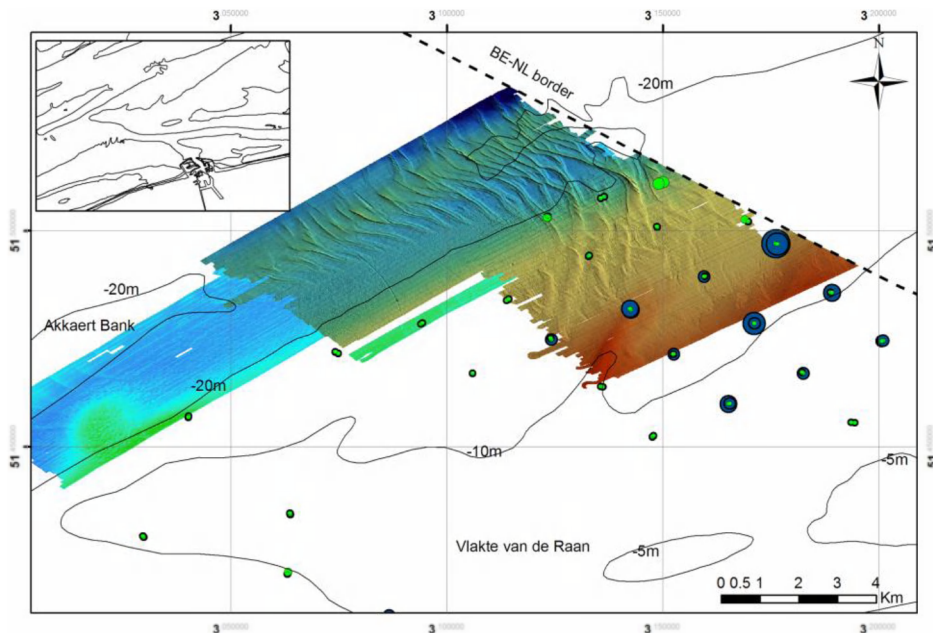
Figure 24. Acoustic seabed imagery north of the slope of the Vlakte van de Raan (BELSPO Quest4D; Van Lancker et al., 2011). Superimposed are the *Ensis* sampling locations of 2010. Green dots represent *E. directus* juveniles; blue dots: adults, scaled with reference to relative densities.
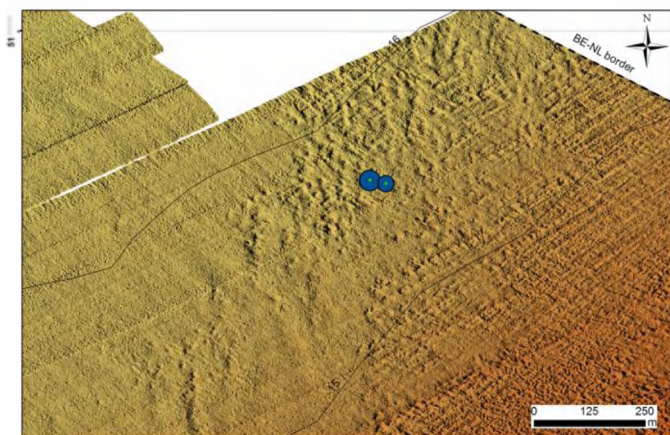
Figure 25. Detailed acoustic seabed image near the BE-NL border, where high densities of adult *E. directus* occur. Blue dots represent samples of adult *E. directus*.
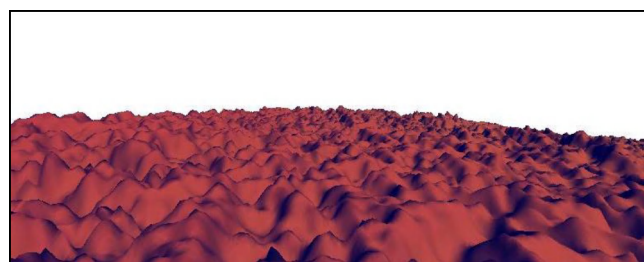
Figure 26. 3D acoustic seabed image (1x1m resolution) of where *E. directus* thrives. Note the rough or bumpy character of the seafloor. Height differences are in the order of 20-40cm. Slope of the Vlakte van de Raan area (RV Belgica ST1029).
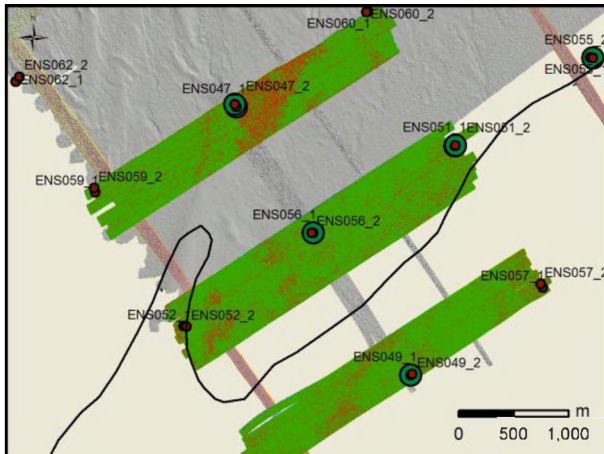
Figure 27. Slope map along northern slope Vlakte van de Raan. Areas with rougher seabed correspond to slope values of more than 2° (red). RV Belgica ST1029; data resolution 1x1m.
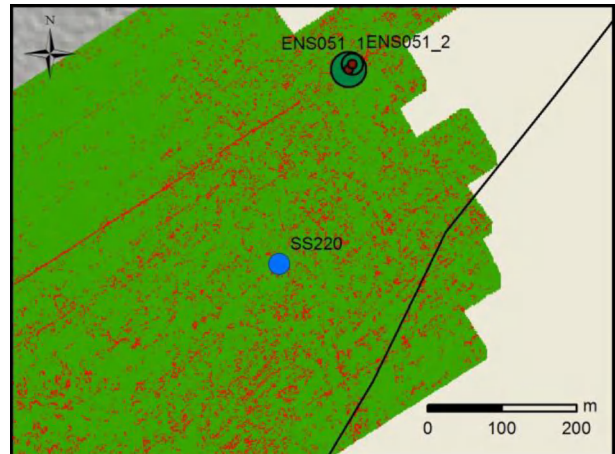
Figure 28. Detail of the slope map around sample location 51 (2010). RV Belgica ST1029; data resolution 1x1m.

In the zone where adult *E. directus* thrives, macrobenthic analysis of samples displayed high densities of both *E. directus* (+/- 341 ind/m²), and *O. fusifomis* (+/- 11.000 ind/m²) (see Tafara Breine, 2011).
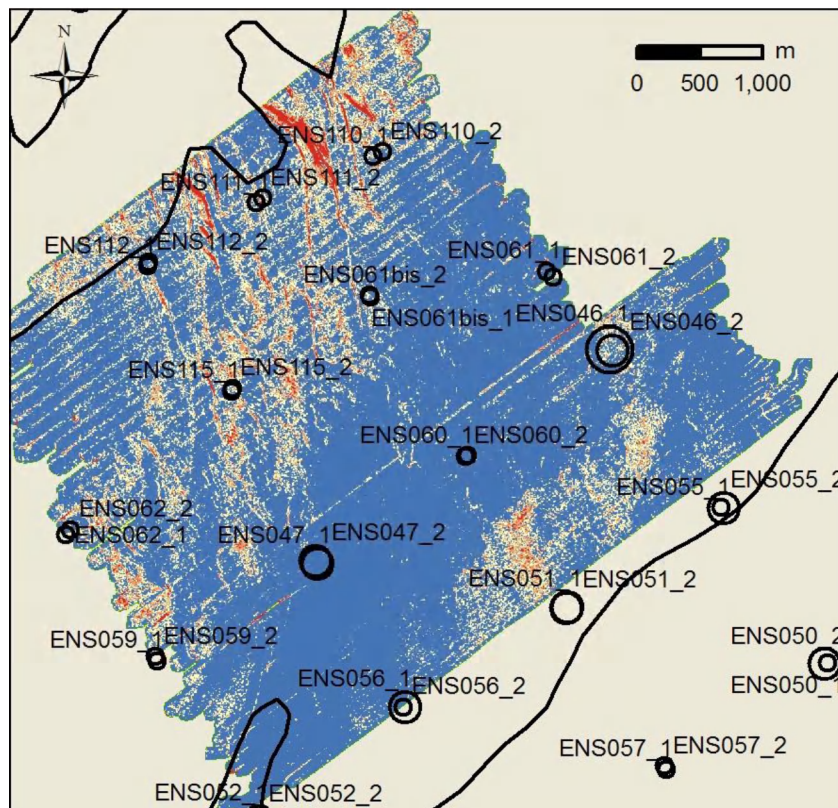
Figure 29. Relative rugosity map (blue to red corresponds to low to high rugosities) along the northern slope of the Vlakte van de Raan (Benthic terrain modeller). Higher rugosity values to the north are related to bedforms (see previous figure). It is hypothesized that the higher rugosity to the south (upper slope) are related to higher densities of both *E. directus* and *O. fusiformis*. Note the relatively higher rugosity near sampling location 51 (2010), where up to 341 ind/m² of *E. directus* were counted.

3.2.3.3.2 Seafloor habitat preferences of *E. directus*: acoustic imagery

From the areas with a distinct acoustic signature, the most relevant habitat related parameters were derived from the acoustic imagery: terrain, substrate and energy characteristics (MESH Guidelines for Seabed Habitat Mapping; http://www.searchmesh.net/).

For *terrain,* the acoustic signature, indicative of *E. directus*, was observed along the mid to upper slope of the Vlakte van de Raan (12 to 16m; Figure 30). This corresponds very well with the occurrences of the adult species of *E. directus* from ground-truthing. Rugosity values in those areas confirmed the rougher texture of the terrain (Figure 29). Relative *substrate* variation was derived from backscatter imagery, in combination with the depth terrain model. High backscatter values (indicative of coarser sediment and/or rougher terrain) could be correlated with the occurrences of the adults. Medium sands of 300-400 µm occur where juveniles occur in the EnSIS data-set (winter 2010), in the gully; fine sands of less than 300 µm are present along the upper slope. Here, adults thrive.

Energy regime was indirectly derived from terrain and substrate variability, in space and in time. Direct measurements of currents (ADCP profiles along transects, BELSPO Quest4D; Van Lancker et al, 2011) were made available, as also hydrodynamics and sediment transport modelling results. The acoustic signature, indicative of *E. directus*, occurred in a zone where bottom stresses are relatively lower than elsewhere along the slope. Transient high density mud suspensions are important in the area where the adults thrive. Current ellipses are more rotary, with magnitudes that do prevent regular deposition. Generally, *Ensis* beds tend to occur away from the bedform areas, hence outside the zones of higher sediment dynamics.
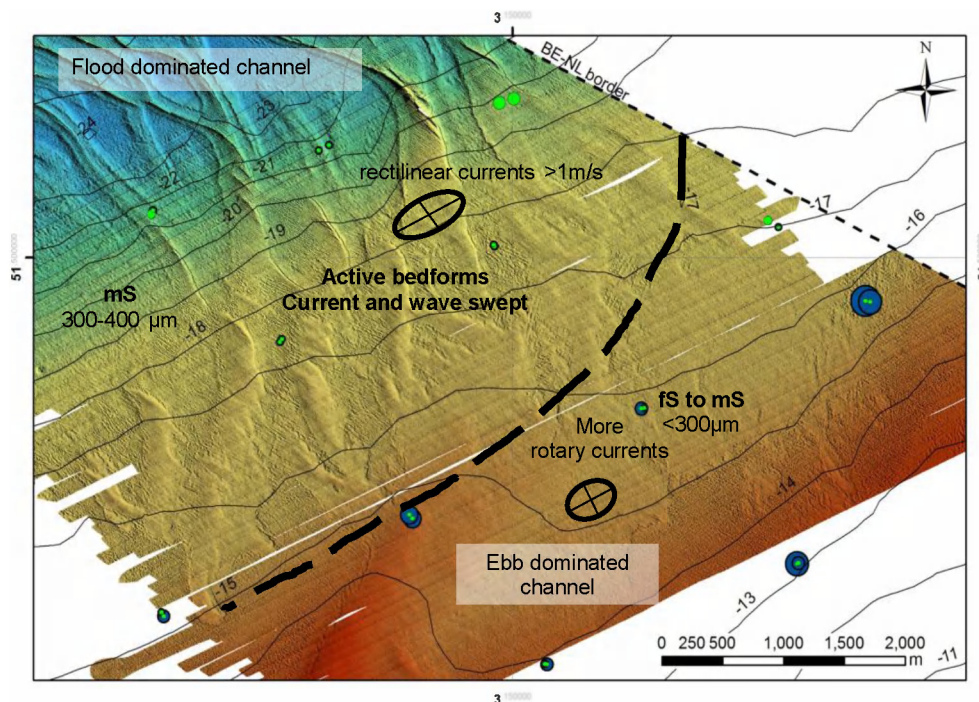
Figure 30. Synthesis figure of habitat preferences of *E. directus* beds, derived from acoustic measurements. Background colour represent larger (blue) to small (red) depths. Green dots correspond with the occurrence of *E. directus* juveniles; they are most abundant near the end of the flood-dominant channel. Medium sands prevail and sediment- and morphodynamics are high. High densities of *Ensis directus* occur outside of this zone (blue dots). Here sediments are finer, currents are more rotary and less in strength. The ebb current is more important in this subarea. RV Belgica ST1029; data resolution 1x1m.

3.2.3.3.3 Anthropogenic disturbance: fisheries' impact

MBES data acquired in June and November 2009 and March 2010, showed a high density of trawl marks along the slope of the Vlakte van de Raan. Overall, the seabed was fully scraped, and locally, even ripped-up (Figure 31).
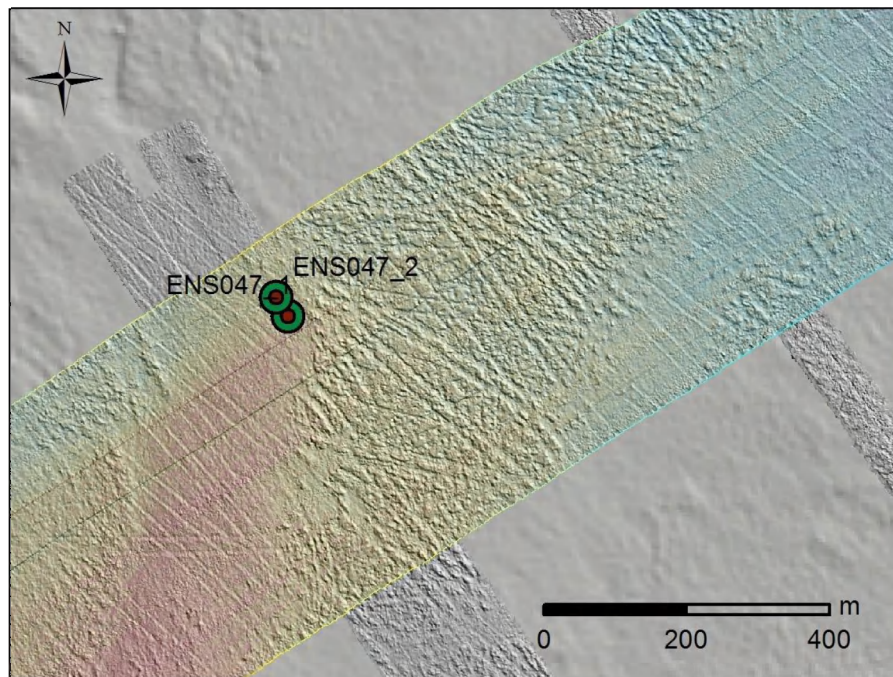
Figure 31. *E. directus* occurrences where the seabed is fully scraped by beam trawling. Similar high intensities of trawling marks were observed during three measurement campaigns (2008-2010), except for campaign ST1029, November 2010, when a sand layer was deposited over these trawled surfaces after a storm.

3.2.3.3.4 Technical recommendations for the visualization of *E. directus* beds

For the direct visualisation of *Ensis* beds, 300 kHz acoustic systems are needed for data acquisition (side-scan sonar or multibeam technology). Data should be processed to data grids of 1x1m to reveal the patchy nature of *Ensis* beds, as also to depict beam trawling intensities. Multibeam technology is preferred since the digital terrain models can be further analysed in terms of slope or rugosity, 2 parameters that assist in the delineation of bioherms. Adequate processing of backscatter data is not straightforward, but remains crucial for characterisaiton of sediment nature. Data interpretation can be hampered by the not always easy differentiation between physically- and biologically-induced structures. More research is needed to distinguish between the acoustic signature of different species. In any case, adequate ground-truthing is needed of the acoustic signal. It needs emphasis that the appearance of patterns can vary also according to the ensonification angle, as well as to different survey conditions. Variations in substrate, sediment availability and energy regime may further alter the way *Ensis* beds are acoustically imaged.

3.2.4 Conclusions

The population structure, distribution and habitat preferences of the American razor clam could be determined in the subtidal domain of the BPNS, and the information could be extended to the entire Belgian-Dutch zone. Populations of adult *E. directus* were found to be relatively stable along the Western coast (Nieuwpoort bank), on the southern side of the Oostende bank, on the northern slope of the Vlakte van de

raan, in the Voordelta area and, more to the North, near the Wadden islands. Important recruitments took place since at least 2007. The status of the population on the northern slope of the Wenduine bank is unclear; as the species was not collected in 2008 and 2010, but used to be formerly present, specific environmental conditions are perhaps making this zone less suitable for the species in the longer run. Sampling procedures onboard research vessels do not enable to determine the species affinity for the shallow subtidal and lower intertidal zones, which might still be suitable for this species. Specific sampling procedures will have to be carried out in the future to complete the picture in these zones. In the subtidal domain, juveniles occupy the seafloor at high initial abundance levels, which indicate release of very large amounts of larvae over the area in spring. A severe mortality rate nevertheless occurs during the two first years, leading to a drop in maximum observed densities of adults, most often lower than 100-200 individuals per square meter. In the Dutch waters, a strong increase in the biomass of *E. directus* was further noted in the recent years, which match results of surveys recently carried out in the BPNS. It cannot be determined yet what factor is responsible for this recent thriving.

Findings of the habitat suitability modelling exercises match information from the literature relative to environmental preferences of this species, namely tolerance to a broad range of values for the considered parameters such as turbidity, depth or sediment structure. The availability of phytoplankton, indicated by chlorophyll a concentrations, as well as moderate levels of turbidity are important factors for this species. Results from acoustic imagery are promising and suggest that this technique could be used to map *Ensis* "banks" in the future, next to other seafloor features. The results further emphasized the strong pressure caused by beam trawling north of the Vlakte van de Raan.

3.3 Impact of *E. directus* on the local macrobenthos

3.3.1 Introduction

Results from section 3.2 and former observations of the amounts of shells of *E. directus* washing ashore every year suggest that the American razor clam occurs at densities susceptible to directly affect local macrobenthic communities as well as their predators. This is suspected in other parts of the North Sea where it is present, but so far no clear impact could be detected on the local benthos. Armonies and Reise (1999) indicated that *E. directus* occurs in poorly faunated as well as densely populated sands. Their findings suggest that on densely populated *Ensis* beds, accumulation of fine sediments and/or fecal pellets alters the habitat. However, only the relative abundance of small polychaetes seemed to have been altered, no further disturbance was evidenced.

On the basis of information available prior to the project, we hypothesized that *E. directus* occurs within the *A. alba* macrobenthic community, considered as the 'most ecologically valuable' of the BPNS in several recent studies (Van Hoey et al, 2004, 2005; Derous et al, 2007; Degraer et al, 2008). The geographic distribution of adult populations confirmed that there is a large overlap between the *A. alba* community and the *E. directus* population. In order to investigate the potential effect of *E. directus* on the local macrobenthos, a series of analyses was carried out with the information available at hand.

3.3.2 <u>Material and methods</u>

<u>Research strategy</u>

The macrobenthic data-set (section 3.1.1.1.) represents the only 'historic' data-set which could provide clues on changes induced by the invader; it was explored during the project but could not be used due to spatio-temporal patchiness in the data distribution (see Van Lancker et al, 2012), preventing any comparison of the situation before and after the introduction of *E. directus*. Furthermore, a large amount of the collected razor clams were recorded as "*Ensis* spp." due to inaccurate determinations, preventing firm conclusions to be drawn on the invader.However, long-term trends in macrobenthos composition can be considered at one station near Oostende, which is regularly monitored since the late 1970s. The historic distribution of local *Ensis* species (Gilson collection, 1899-1914, RBINS-Invertebrates: *E. ensis, E. minor* and *E. arcuatus*) has been made available recently (Van Lancker et al, 2011) and enables to check whether a geographic overlap exists between their natural distribution and that of the recent invader. Finally, given their widespread distribution, the macrobenthic composition of samples gathered in summer 2008 (SCHELP survey; see section 3.1.1.2) and winter 2010 (EnSIS survey; see 3.1.2.4.) can be analyzed to tentatively detect an influence of larger *Ensis* densities on the composition of macrobenthos.

3.3.2.1 *Long-term analysis of macrobenthic changes at station 120*

Long-term variation of the macrobenthos community of the BPNS was assessed at station 120 (51°11'10'' N; 02°42'07'' E). This station was chosen because (1) it is relatively unaffected by anthropogenic activities in comparison with other monitoring stations and (2) the sediment characteristics remained relatively stable; i.e. the average median grain size over the entire time series was 200 µm, fluctuating between 115 and 286 µm, and the mud content (fraction of grains < 63 µm) varied between 2 and 10 % (Van Hoey et al, 2007).

A data-set was compiled using data gathered by the Department of Fisheries (ILVO, DVZ) and the Marine Biology Section (UGent, Marbiol), resulting in a long-term dataset (1979 – 2008) which consists of both spring and autumn data of each year (Wittoeck et al., 2005). No data were available for the periods 1987-1988, 1992-1993 and also the information for spring 1984 and autumn 2007 was lacking. All macrobenthos samples were taken with a Van Veen Grab (sampling surface 0.10 or 0.12 m²) in triplicate, except for spring 2005 (n = 5), spring 1991 (no replicates; samples pooled prior to laboratory analysis) and autumn 2008 (no replicates). The samples were sieved before or after fixation on a 1mm sieve (see Degraer et al, 2006). Non benthic species (e.g. Mysida, crabs) or species which were not adequately sampled with this procedure (Nematoda, Nemertea, Oligochaeta) were excluded for the analysis.

The ash-free dry weight (AFDW) was calculated by converting the blotted wet weight according to Rumohr et al. (1987) and Ricciardi & Bourget (1998). Replicate samples were analyzed separately but were pooled afterwards for the statistical analyses. Analyses were performed in accordance with the guidelines agreed upon within the ICES Benthic Ecology Working Group (see Ruhmor et al, 2009).

3.3.2.2 *Historic distribution of local razor clam species*

Records of local *Ensis* species (*E. ensis, E. arcuatus* and *E. minor*) were extracted from the initial historic data-set (1899-1914; Gilson data) assembled through the projects

"HINDERS (Houziaux et al, 2008) and "Quest4D" (Van Lancker et al, 2011). All records were considered regardless of sampling gear used to maximize available distributional data for these species. The coastal area (up to 10 nautical miles offshore) and the Hinder banks area were intensely sampled with different sampling gears, while the area in-between was mostly sampled with beam trawls; for the latter instrument, most geographic positions have not yet been processed and cannot be mapped (see Houziaux et al, 2008).

3.3.2.3 *Multivariate analysis of macrobenthic composition in 2008 and 2010*

The two data-sets used for this analysis (SCHELP survey, see section 3.1.1.2; EnSIS survey, see section 3.1.2.4) display a regular geographic distribution of samples. However, they differ by the sampling gear (trawled dredge in summer 2008 versus box-core in winter 2010), and in 2010 sampling was more focused on areas exhibiting higher densities of *E. directus*. In both cases, it is searched whether a pattern of species composition can be related to the occurrence of *E. directus* within the sampled area. Its distribution presents an important yet not full overlap with the distribution of the "*Abra alba*" macrobenthic community (70% overlap). We hypothesize that samples where adults of *E. directus* (i.e. specimens larger than 10 cm) are most abundant would display an altered macrobenthic composition within the *A. alba* community. For the 2010 survey, samples with large densities of juveniles were similarly flagged to check for any effect of this massive recruitment on macrobenthos composition

3.3.2.3.1 SCHELP survey, summer 2008

All samples were initially selected except one empty sample. Species selection was done to eliminate as little data as possible. The "Acarina" was removed. "Amphiura" and "*Acrocnida brachiata*" were grouped under "*Amphiura* spp."; swimming crabs were all grouped under *Liocarcinus* spp.; *E. directus* and *Ensis* spp. were grouped as "*Ensis* spp." (*E. directus* overwhelmingly dominate this group; however, as most specimens were damaged by the dredge, it is not possible to know if 100 % of collected specimens indeed belong to this species). The species were then flagged as epi- or endobenthic to carry out separate analyses. For samples, two flags relative to occurrence of large *Ensis* spp. (>= 10 cm in length) were defined: on one hand, presence or absence ; on the other hand, three density levels, namely 0, 0-10 and >= 10 specimens per square meter.

Polychaete species are lacking. For instance, *Owenia fusiformis* was not appropriately recorded. This must be kept in mind when interpreting the multi-species patterns hereafter analyzed.

3.3.2.3.2 EnSIS survey, winter 2010

All processed samples were taken into consideration. Densities were averaged per station when two samples were available. Polychaetes were not determined except *Owenia fusiformis* and *Pectinaria belgica*. The number of species and total count were however determined, enabling the computing of total densitiy and total species richness in all samples.

As for the SCHELP data-set, flags were created to determine whether larger densities of adults could underpin the multivariate ordination of samples based on their similarity (Bray-curtis). As large densities of juveniles were observed as well, density flags were created for juveniles under assumption that large amounts of settling juveniles might as well affect the macrobenthic composition.

3.3.2.3.3 <u>Analysis</u>

Frequency of occurrence and relative abundance were calculated for all species in each data-set and plotted to visualize the overall species composition. The first measure (percentage of stations where the species was found) provides information on how widespread a species is in the data-set, while the second indicates which species numerically dominates. Multivariate analysis was carried out using the Primer-E v. 6 package (Plymouth Marine Laboratory; Clarke and Gorley, 2006). After fourth- or square-root transformation (to reduce influence of numerically dominant species), Bray-curtis similarity matrices were calculated. A first series of CLUSTER analyses with SIMPROF permutation test were carried out on the overall data-set to identify significantly different clusters of benthos composition. This was done in the overall data-set and after removal of *Ensis* spp. A SIMPER procedure was applied where relevant to identify characteristic species of the identified clusters. Samples were ordinated using Multidimensional Scaling (MDS). The analysis was carried out on the entire data-sets as well as on endo- and epibenthic species contents.

Our hypothesis is that higher densities of adult *E. directus* will display the most pronounced impact on macrobenthos composition. Therefore, the ordination of samples was evaluated with reference to major density classes of adults. An ANOSIM permutation procedure was carried out to check to what extent the observed sample ordination matches ordination based on the *à priori* determined classes of adult *E. directus*. We further looked at samples with large abundance of juveniles, as the settlement of thousands of young specimens might in theory as well affect macrobenthos composition.

3.3.3 **Results and discussion**

3.3.3.1 *Long-term changes at station 120 and comparison with the Dutch situation*

Long-term trends were consistent for spring and autumn with, overall, higher species densities, abundances and biomasses in autumn (Figure 32).
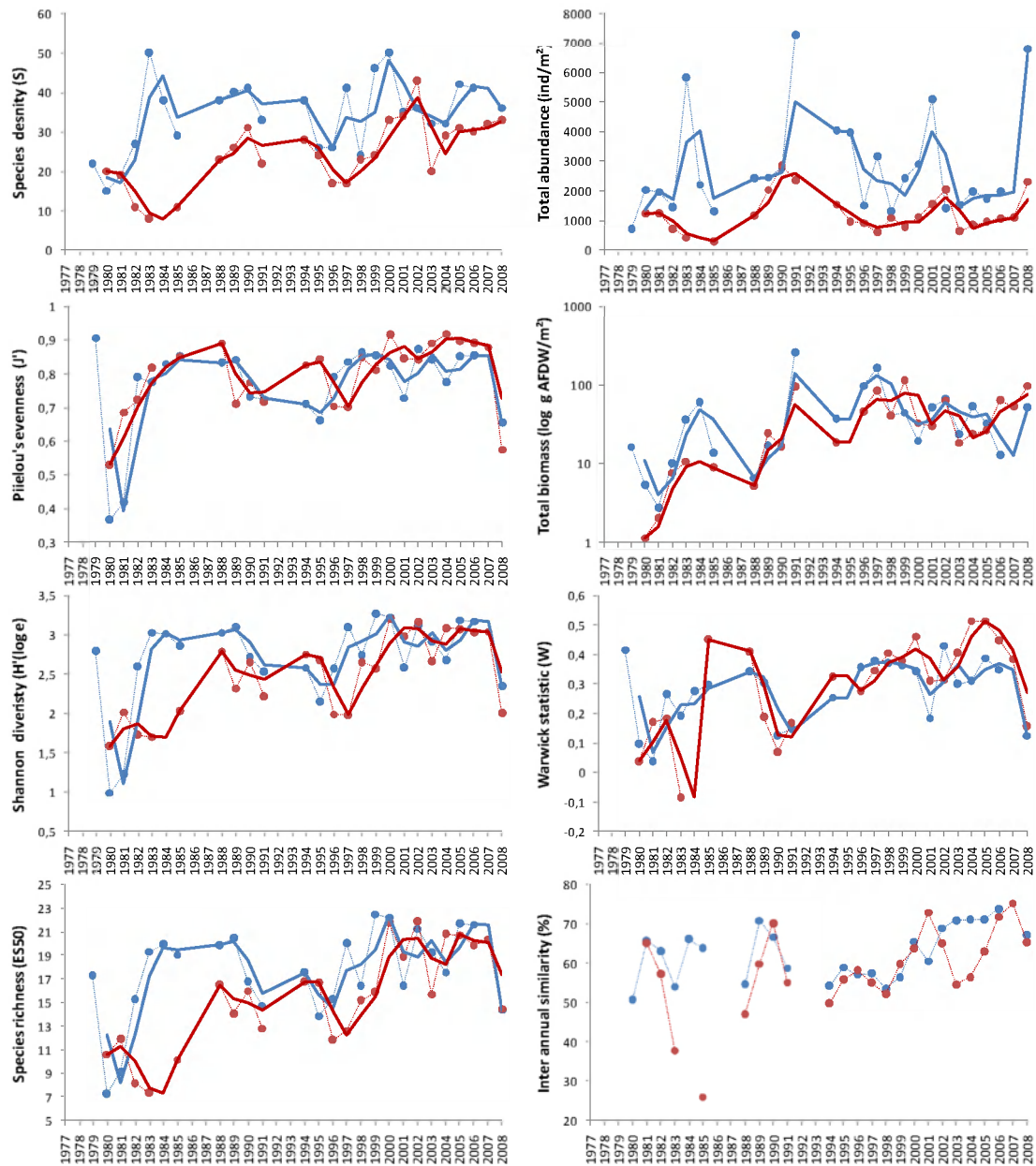
Figure 32. Long-term spring (red) and autumn (blue) variation in species density (total n° of species/sample), Pielou's evenness, Shannon-Wiener diversity (ln), species richness (ES50), abundance (ind./m²), biomass (log g ash-free dry weight/m²), the Warwick statistic and the interannual similarity (bray-curtis similarity between two subsequent sampling occasions based on 4th root transformed community abundance data) between 1979 and 2008 at station 120 in the Belgian Part of the North Sea. Trendlines are running averages (period = 2 sampling occasions).

Over time the community at station 120 became clearly more diverse and consisted of a higher total biomass while a more erratic pattern was found for the abundance. The Warwick statistic increased over time, suggesting an evolution from a community dominated by a small-sized species towards a community where the biomass is dominated by larger and long-lived species.

The increase during the mid-eighties relates to the biomass increases of *Abra alba* and *Tellina fabula*, while the increase during the nineties was due to the increase in biomass of *Ensis directus* and *Spisula subtruncata* (second half nineties). A massive recruitment was observed in summer 1991 for *Ensis directus* and in summer 1995 for *Spisula subtruncata*.

A steep drop in species density, evenness, diversity and species richness occurred in spring after the severe winter of 1995/1996 (Figure 33). Furthermore, in comparison with the autumn of 1979, a steep decrease in species density, species richness, diversity and evenness was noticed in autumn 1980 which suggest a severe effect of the very cold winter of 1979/1980. Following these events, clear drops in interannual similarity for both spring and autumn communities occurred in the periods 1981-1983 and 1996-1998. Furthermore, steep drops in community inter-annual similarity were present in 1990-1991 and 2001-2003 (only for spring samples). As regime shifts, at least in the marine environment, have been defined by de Young et al. (2004, 2008) as 'a change between contrasting persisting states' these periods should be a prime target for further detailed analysis of possible regime shifts in the BPNS.
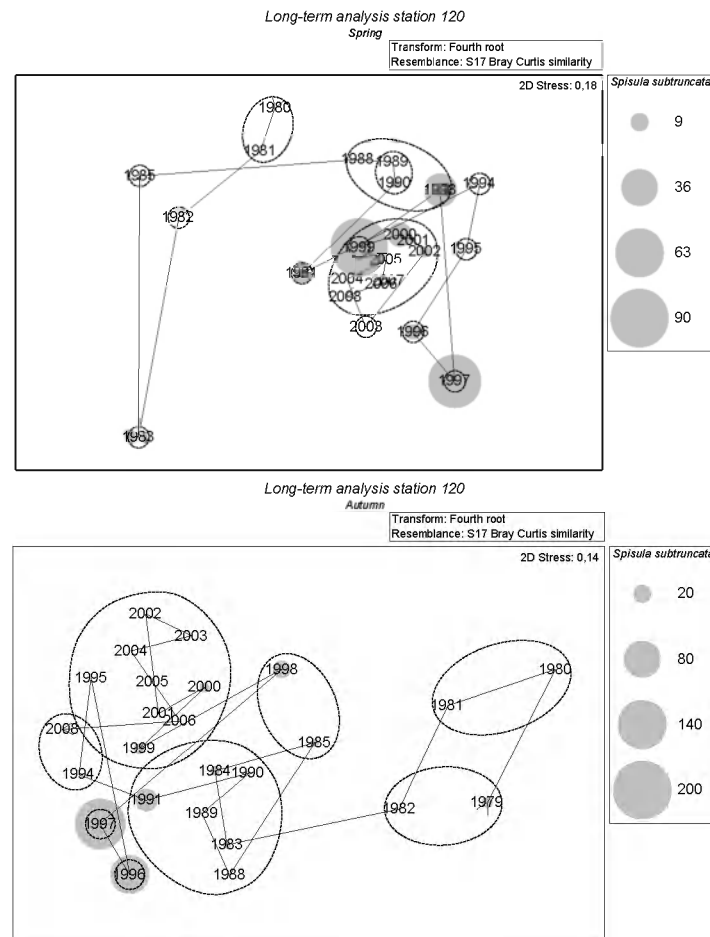
Figure 33. Non-metric multidimensional (MDS) ordination of species assemblages at station 120 over time in spring (upper panel) and autumn (lower panel). Dashed lines represent 60% similarity clusters. Biomasses of Spisula subtruncata are superimposed using grey circles which sizes reflect the magnitude of the total biomass (g AFDW/m²) of this species.

Bivalvia species trend at station 120 and in Dutch waters,

Seventeen bivalve species were found over the period 1979-2008, whereof the most dominant were *Kurtiella bidentata, Tellina fabula, Spisula subtruncata, Abra alba, Ensis directus* and *Venerupis senegalensis*. The number of bivalve taxa found over the period, varies between 5 and 10 from 1982 onwards (Figure 34). The density shows variation over time, with reduced values in the 2000s.
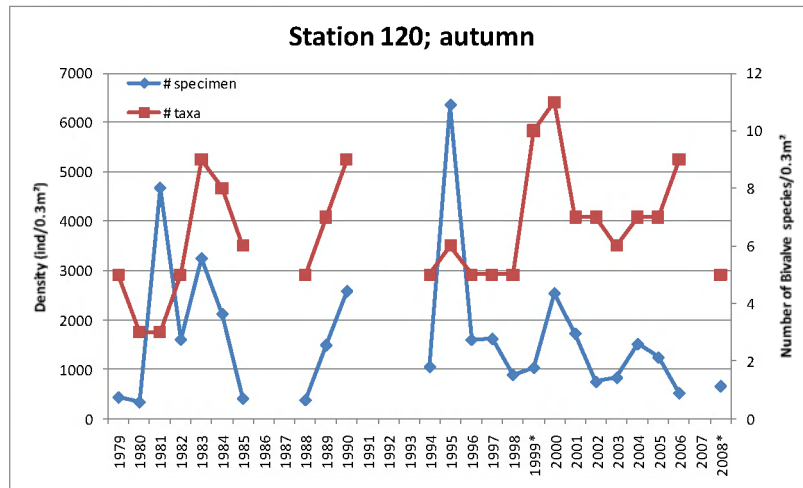
Figure 34. Density and taxa richness pattern of the Bivalvia at station 120 in autumn.

Over 30 years, the dominance and density of the most common bivalve species displays some variability (Figure 35 and Figure 36). Most bivalve species show variability in their contribution to total abundance, most obvious for *Abra alba*. This species dominated over the entire period. The small bivalve *Kurtiella bidentata* shows its highest densities in the period 1980-1984 and shows two smaller peaks in 1996-1997 and 2000-2002. *Tellina fabula* displays very large relative abundances in the early 1980s and became a minor component of the bivalve fauna afterwards. *Spisula subtruncata* was most successful in the period 1995-1997 but became almost absent in the consecutive years, displaying a clear trend to rarefaction.
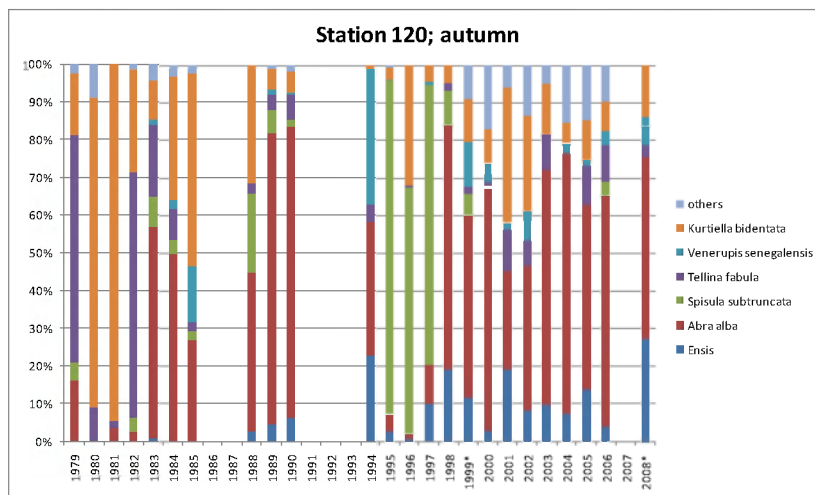
Figure 35. Relative abundance of some Bivalvia taxa at station 120 in autumn.
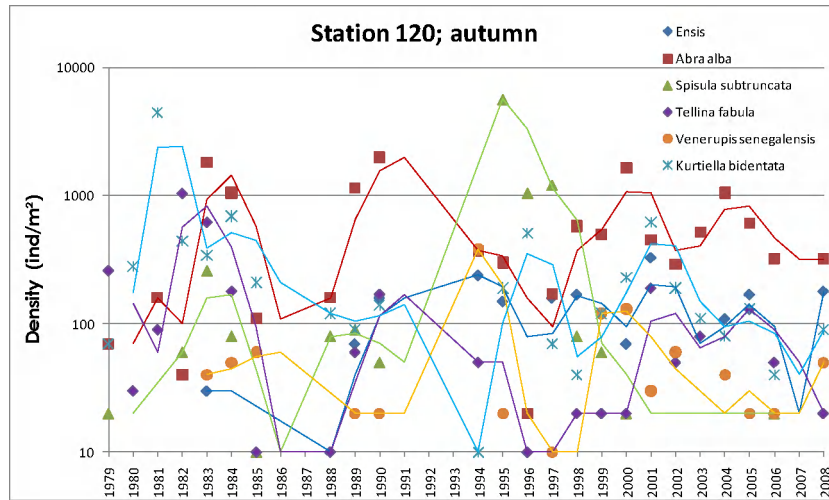
Figure 36. Density pattern of *Ensis* spp., *Abra alba*, *Spisula subtruncata*, *Tellina fabula*, *Venerupis senegalensis* and *Kurtiella bidentata*. Trendlines are running averages (period = 2 sampling occasions).

Noticeably, in the Dutch coastal zone, the total standing stock of shellfish as determined from the monitoring surveys more or less doubled in the course of the past 15 years, strongly dominated by *E. directus* after the 2000s, while a rarefaction of *S. subtruncata* took place (Figure 37). The rarefaction of *S. subtruncata* did not happen at the same moment everywhere (it was earlier in the voordelta area), and therefore it cannot be directly related to the increase of the *E. directus* population; many other causes could be involved in this apparent switch between the two species (such as weather conditions, the global change or interference with human activities). However, the trend in *S. subtruncata* matches that observed at station 120, in the BPNS, suggestive of a common cause, even though no increase was observed in the total macrobenthic biomass nor in the abundance of *E. directus*.
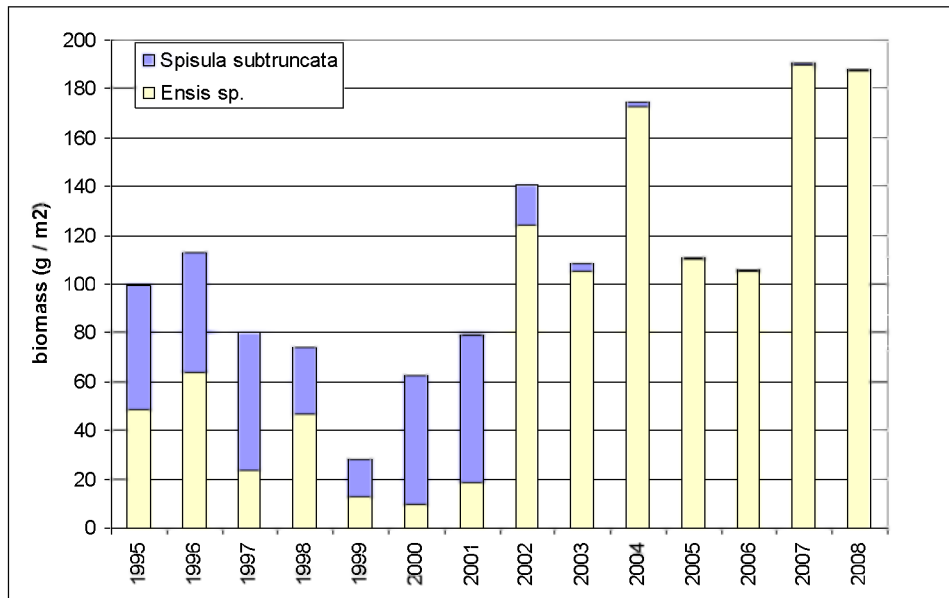
Figure 37. Average biomass (fresh weight; g m^{-2}) of *Ensis* sp. (mostly represented by *E. directus*) and *S. subtruncata* in the Dutch coastal waters betwen 1995 and 2008.

3.3.3.2 *Long-term changes in the distribution of local razor clam species*

Razor clams collected in 1900 are represented by *E. arcuatus* and, to a lesser extent, *E. ensis* and *E. siliqua*. All three species were almost exclusively collected off the Western coast (Figure 38). Even though none of the sampling gear used in these surveys can be considered as accurate for these deep-burying bivalves (Houziaux et al, 2008), their observed distribution suggests that they were most abundant in this area.
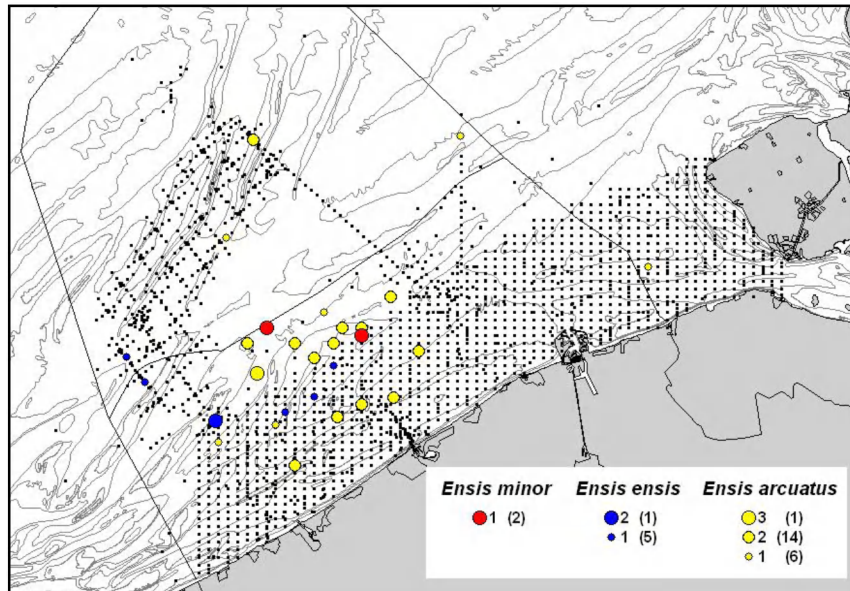
Figure 38. Historic records of local *Ensis* species in the period 1899-1914 (Gilson data-set) within and near the BPNS. Black dots represent sampling events with various sampling gears (sediment sampler, benthic dredge and to a lesser extent trawls, for which little amounts of geographic positions were directly available for mapping). Figures indicate total abundance at the stations, figures between brackets indicate the number of samples at which the abundance was recorded.

The map clearly shows that these species tended to occur further offshore compared to *E. directus* in the present day. Few records located southwards to the Oostende sand bank suggest however that an *E. arcuatus* population did occur there, where *E. directus* is now thriving. This population has most probably disappeared, but it is not known whether this happened before or after the invasion.

More recent records are lacking to investigate a possible competition of these species with the invader for space and/or food. In the 'Habitat' macrobenthic data-base, taxonomic identifications at the species level are generally lacking, preventing any conclusion relatively to the long-term fate of these species. In 2008 (SCHELP survey), a large part of *Ensis* specimens were broken, preventing accurate determinations to be carried out, and it cannot be determined whether local species occurred in the samples. In 2010, when a box-corer was used to obtain deeper sediment samples, only *E. directus* was collected in the surveyed area (coastal waters), suggestive of a disappearance or at least rarefaction of the local species there. However, in 2005, few samples obtained with a 2m chain-mat beam trawl on and around the gravel fields of the Hinder banks provided alive specimens of *E. arcuatus* (Houziaux, unpublished data), evidencing that this species still occurs further offshore, in coarse sediments.

Given their respective historic distributions, it seems unlikely that local species have been directly affected by the invasion itself, although larvae of the invader now certainly dominate over a wide area. Their distribution more offshore also suggests that at least *E. arcuatus* prefers coarser sediment and deeper water. Their abundance was always low, but this observation could be due to low sampling efficiency for such deep-burying species. The species are probably outcompeted by *E. directus* closer to the shore line (e.g. south to the Oostende bank), in finer sediments. This might explain the reduced occurrence of local species on beaches. Further offshore, we have not enough recent data to conclude on the present status of the local species, but their fate should not be influenced by the coastal *E. directus*.

3.3.3.3 *Analysis of macrobenthic composition in 2008 and 2010*

3.3.3.3.1 Summer 2008

General species composition pattern

The overall macrobenthic composition of the samples is summarized in Figure 39, where the numerical dominance as well as the widespread distribution of *A. alba* are evidenced. The trawled dredge caught large amounts of the dominant epibenthic species of the area, namely swimming crabs *Liocarcinus* spp., *Ophiura* spp., *Hinia reticulata*, *Diogenes pugilator*, *P. bernhardus*. Noticeably, *Ensis directus* (represented by "*Ensis* spp", of which it constitutes by far the major component) is the second most abundant and widespread endobenthic bivalve.
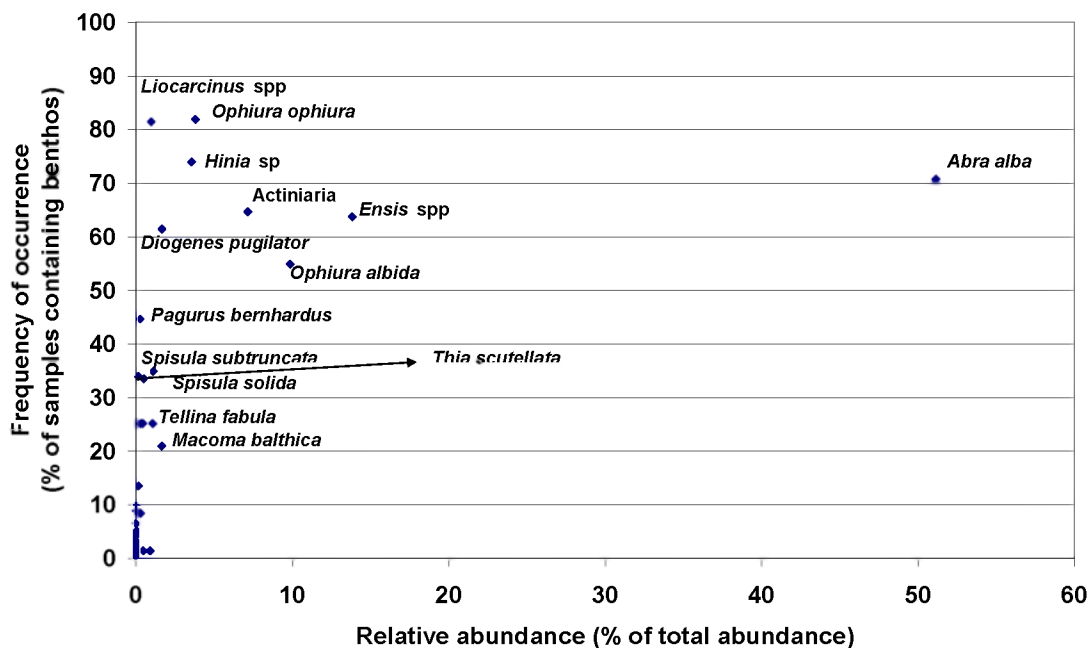
Figure 39. Scatter plot of relative abundance (% of total abundance in samples) and frequency of occurrence (percent of samples where the species is present) for species gathered in summer 2008.

Multivariate analyses

A first series of cluster analyses were carried out on the overall data-set to check whether some structure could be already visible. This was done at two levels of data transforms (square root and fourth root), in the overall data-set and after removal of *Ensis*. In the four cases, samples were grouped in a very similar way in the MDS ordination plots, with some larger dispersion in the fourth root transform case. Larger densities of large *Ensis* were located at the center of the two-dimensional projection of the multivariate dispersion pattern, with *Ensis* spp. included (Figure 40) or without it (not illustrated). The ordination reveals that the central samples also display highest species richness values, while there is a right to left gradient of increased total density. Larger densities of *Ensis* show some affinity with highest total densities. The cluster analysis of the square root transformed data-set including *Ensis* is taken as example to investigate the sample composition. From the cluster tree (not illustrated), large groups of samples identified as significantly different with the SIMPROF procedure were defined at the similarity level of 26. When these clusters were overlaid on the MDS plot, the five major clusters were clearly individualized (Figure 40). Clusters 'f', 'a' and 'b' were composed of only one sample and were considered as outliers.
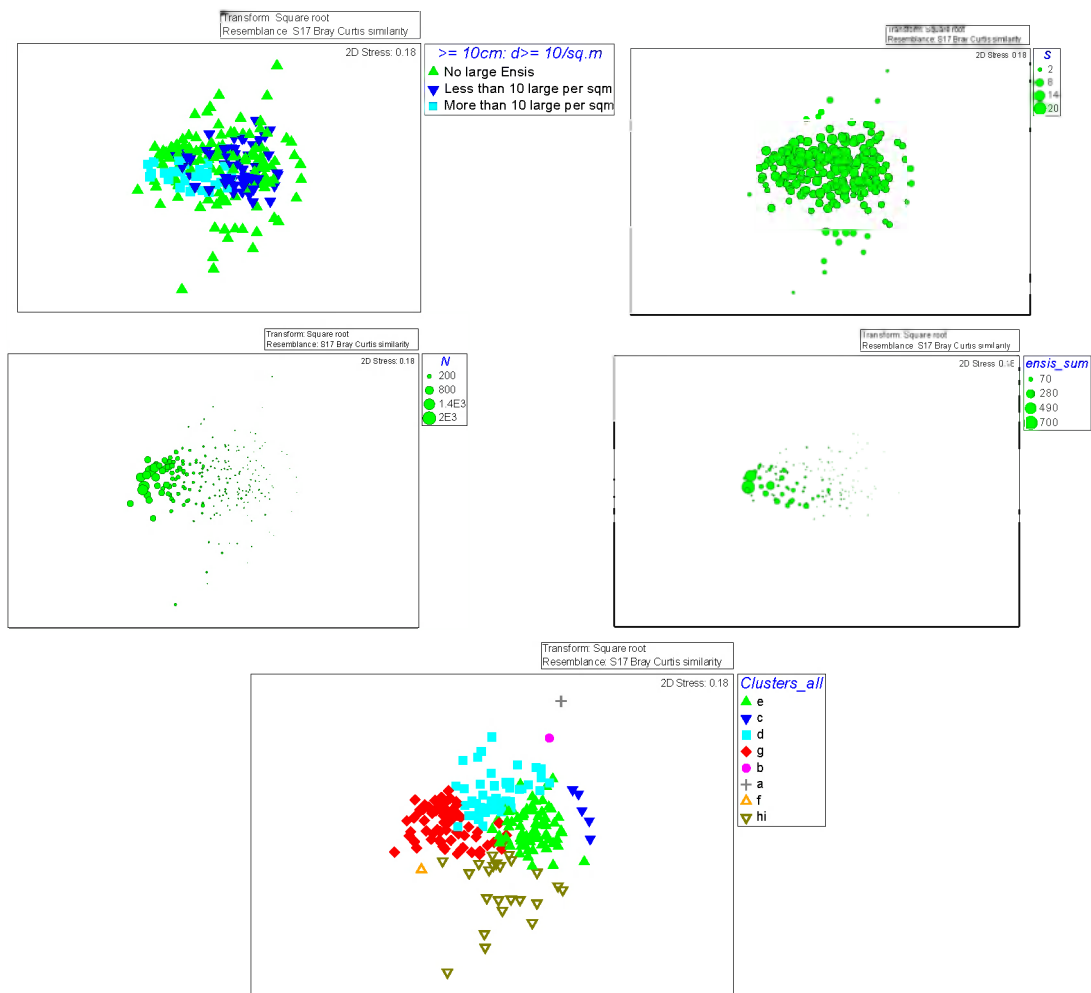
Figure 40. Upper, left: MDS ordination of the overall data-set (data transform: square root; including Ensis spp. Above, right: Bubble plots of species richness (S); Middle, left: Bubble plots of total density (N); Middle, right: Bubble plots of Ensis spp density. Lower: clusters determined at similarity value of 26 based on cluster tree analysis (see text).

The composition of the five clusters was investigated by application of the SIMPER procedure. The clusters were characterized by varying relative densities of some dominant and common epibenthic species (namely: *Diogenes pugilator*, *Ophiura ophiura* and *O. albida*, *Nassarius*, *Pagurus bernhardus*, *Liocarcinus* spp., Actiniria) and some different endobenthic species. Results indicate that the epibenthic content of the samples is relatively homogeneous at the scale of the survey area, and no influence of *Ensis* spp. can be determined when samples are reduced to their epibenthic content (analyses not displayed). The clusters obtained when *Ensis* spp. was removed from the data-set were almost identical. Application of the ANOSIM permutation test on the basis of the categories of density of larger *E. directus* never provided any match with the species composition. The apparent effect of large *Ensis* spp. on the multivariate ordination is largely underpinned by an effect of density.

Endobenthic species composition patterns

The data-set was reduced to its endobenthic species content and samples which became empty were removed. On the square root transformed data-set, B-C similarity and MDS ordination confirmed that adult *Ensis* spp. are present in many samples and strongly contributes to total densities (Figure 41). A cluster analysis was carried out; the SIMPROF test and the overlaying of clusters on the MDS plot revealed that clusters were formed across *Ensis* densities (clusters 'd' and 'b';). When *Ensis* spp data were removed, these clusters remained ordinated similarly, but not the samples with larger *E. directus* (Figure 42). This confirms that densities of larger American razor clams do not underpin the species composition of the samples and that the apparent structuring effect is density-dependant. These clusters are meaningful enough to describe the endobenthic composition of the samples with or without *E. directus*.
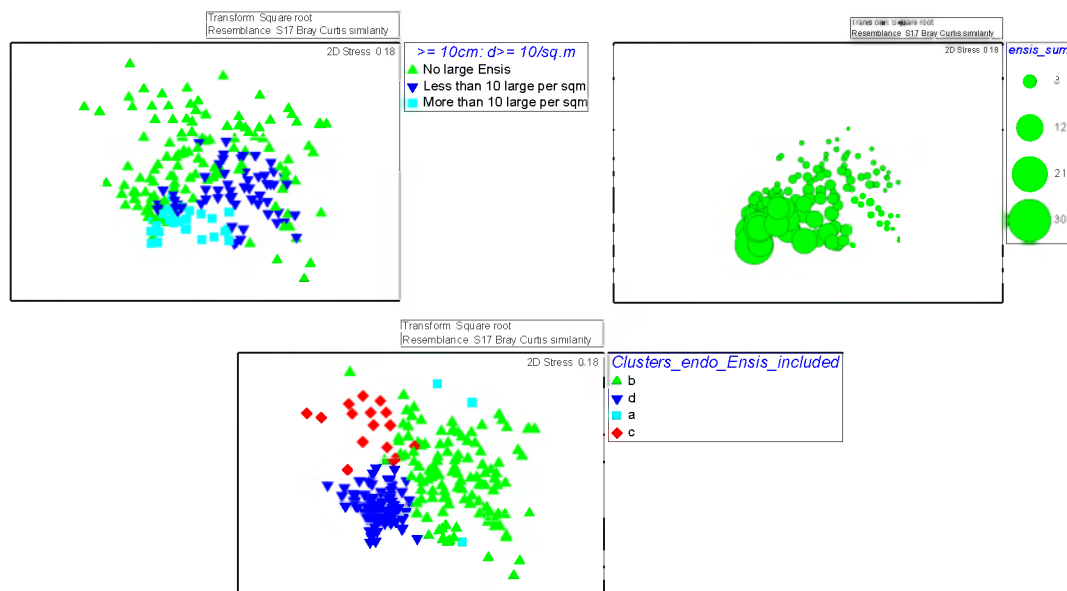
Figure 41. Upper, left: MDS ordination plot of square-root transformed endobenthic data-set including *Ensis* spp. Right: same MDS ordination with densities of Ensis spp. Lower: same ordination with significantly different clusters at similarity level 14.5.
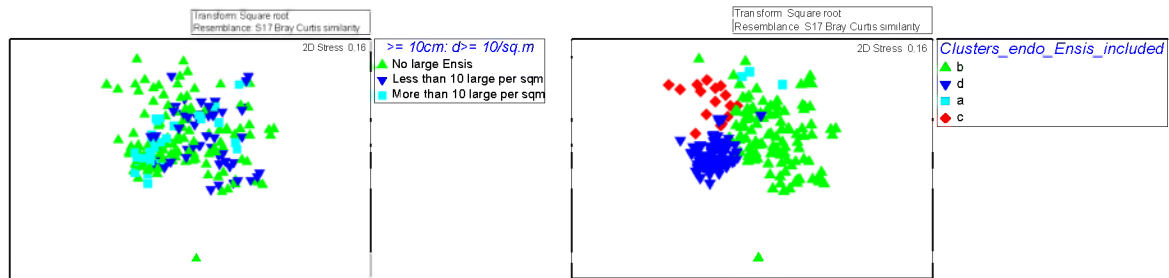
Figure 42. MDS ordination plot of square-root transformed endobenthic data-set after exclusion of *Ensis* spp. left: adult density classes; right: same ordination, with clusters defined based on data including *E. directus* overlaid.

A SIMPER procedure was applied to these clusters to determine their characteristic species (Table 7). Groups 'b' and 'd' form the bulk of the data-set. The group 'b' is a larger, poorly defined cluster where the filter-feeding species *E. directus, S. solida* and *D. vittatus* are on average more represented, but this is not a clear-cut pattern since the deposit-feeder *A. alba*, which strongly characterizes group 'd' (with *S. subtruncata*), is also an important contributor. The group 'd' is characterized by much higher overall densities of endobenthos.

SIMPER analysis: average endobenthic species content of clusters (including Ensis spp)					
Group a (average similarity: 28.33)					
Species	**Av.Abund**	**Av.Sim**	**Sim/SD**	**Contrib%**	**Cum.%**
Spisula subtruncata	1.09	24.73	2.04	87.30	87.30
Donax vittatus	0.16	3.60	0.58	12.70	100.00
Group b (average similarity: 25.49)					
Species	**Av.Abund**	**Av.Sim**	**Sim/SD**	**Contrib%**	**Cum.%**
Ensis directus	**0.81**	**7.94**	**0.67**	**31.14**	**31.14**
Abra alba	0.65	7.35	0.61	28.82	59.96
Spisula solida	0.46	5.74	0.52	22.51	82.47
Donax vittatus	0.25	1.81	0.32	7.10	89.57
Echinocardium cordatum	0.16	1.26	0.28	4.95	94.52
Group c (average similarity: 51.73)					
Species	**Av.Abund**	**Av.Sim**	**Sim/SD**	**Contrib%**	**Cum.%**
Macoma balthica	3.79	42.14	2.48	81.46	81.46
Abra alba	1.31	8.04	0.81	15.54	97.00
Group d (average similarity: 49.53)					
Species	**Av.Abund**	**Av.Sim**	**Sim/SD**	**Contrib%**	**Cum.%**
Abra alba	10.91	35.92	2.18	72.52	72.52
Ensi directus	**4.30**	**8.53**	**0.78**	**17.22**	**89.74**
Spisula subtruncata	1.11	2.18	0.53	4.40	94.14

Table 7: relative contribution of species to cluster similarity levels determined with the SIMPER procedure (see Clarke and Gorley, 2006).

Off the eastern coast, the group 'b' strongly dominates on top of the Vlakte van de Raan, while off the Western coast it is seemingly more interspersed with samples of group 'd' (Figure 43). The map shows a zonation pattern of overall macrobenthic composition in front of the Eastern coast, along the Dutch border. Near the coast, group 'c' is clearly dominated by *M. balthica* (87 % of contribution to similarity) and its distribution matches that of its associated community, typical of muddier sediments (see Degraer et al, 2008). On the Vlakte van de Raan, group "b" dominates, where species such as *D. vittatus* or *S. solida* indicate more exposed conditions and clean fine sand. On the northern slope, group "d" dominate, indicative of muddy sands. Along the Western coast, the decrease in mud content causes group 'c' to disappear and groups 'b' and 'd' are more mixed, with group 'd' being widespread and group 'b' displaying a more patchy distribution. Group 'a' seems to reflect some transition between groups 'b' and 'd'.

From this analysis, *E. directus* appears as a typical species of the macrobenthic communities of the coastal waters of the BPNS, where it finds appropriate conditions to thrive, but it cannot be associated to one specific macrobenthic species association. The findings are consistent with observations by Armonies and reise (1999) that *E. directus* occurs in poorly as well as richly colonized sediments. Its distribution largely overlaps with that of the "*Abra alba*" community as defined by Van Hoey et al (2004), but extends further. This species assemblage is here likely sub-divided into two groups 'b' and 'd', the latter being characterized by much higher endobenthic densities, while clean sand species contribute more to the within-similarity of the former. Group "b" also displays a wider geographic distribution than the "*A. alba*" community *sensu* Van Hoey et al (2004). Those differences are likely partly attributable to the fact that polychaete species were not included in this analysis.
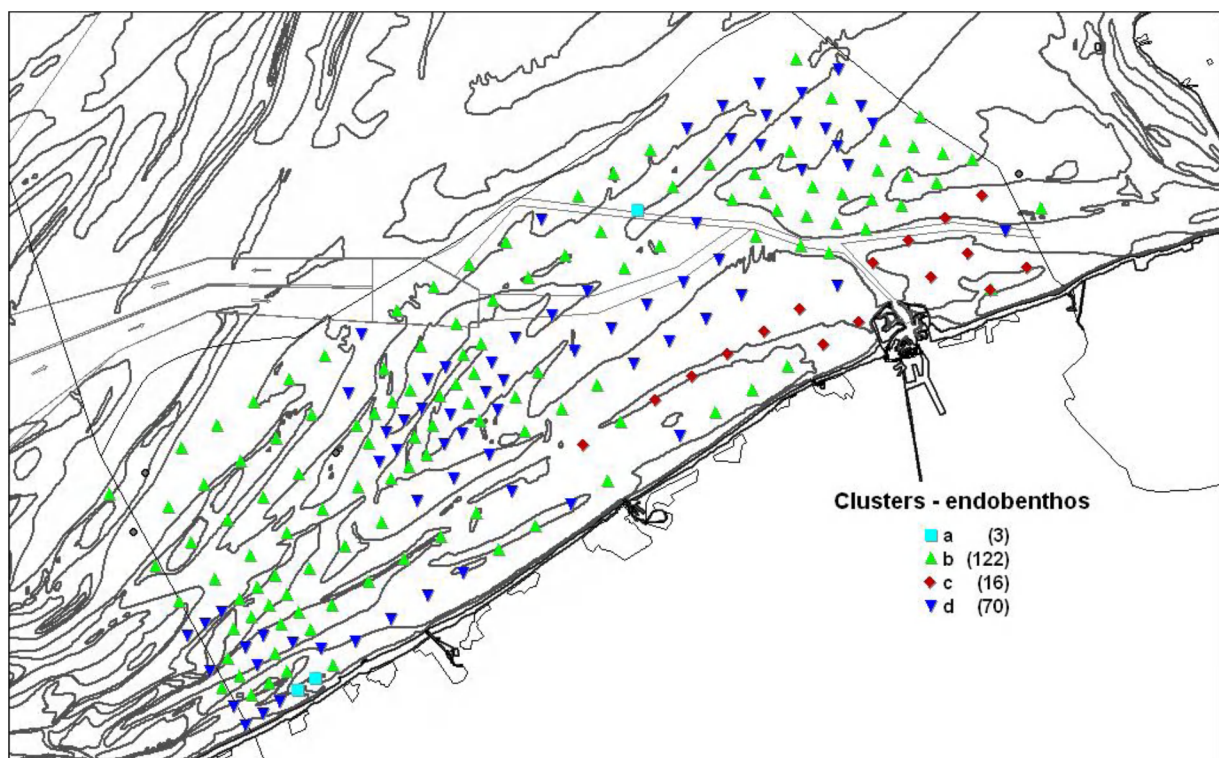
Figure 43. Map of endobenthic clusters determined on the basis of the endobenthic species composition of the samples.

3.3.3.3.2 Winter 2010

In this survey, a large amount of fresh bivalve doublets, empty or filled with black 'mud', were found, indicative of a high mortality perhaps induced by specific winter conditions. Many samples were also characterized by a distinctive smell which could not be defined (Kerckhof and Houziaux, *pers. obs.*). Empty doublets included very large amounts of *E. directus*. As shown in section 3.2.3.1, very large amounts of young live *E. directus* were collected, mostly on the northern border of the sampled area. However, when these juveniles were removed, the population structure appeared to be very much the same as in 2008, and adult specimens were mainly found in similar areas (see Figure 15).

Overall species composition

Figure 44 shows that the sample composition is dominated by *E. directus* and *Owenia fusiformis* (notwithstanding other polychaete species not determined at species level). The strong dominance of *E. directus* can here be explained by the fact that this sampling survey targeted areas where this species was most abundant. On the other hand, a box-corer was employed instead of a trawled dredge, which can be expected to reduce the relative contribution of epibenthos in the samples.
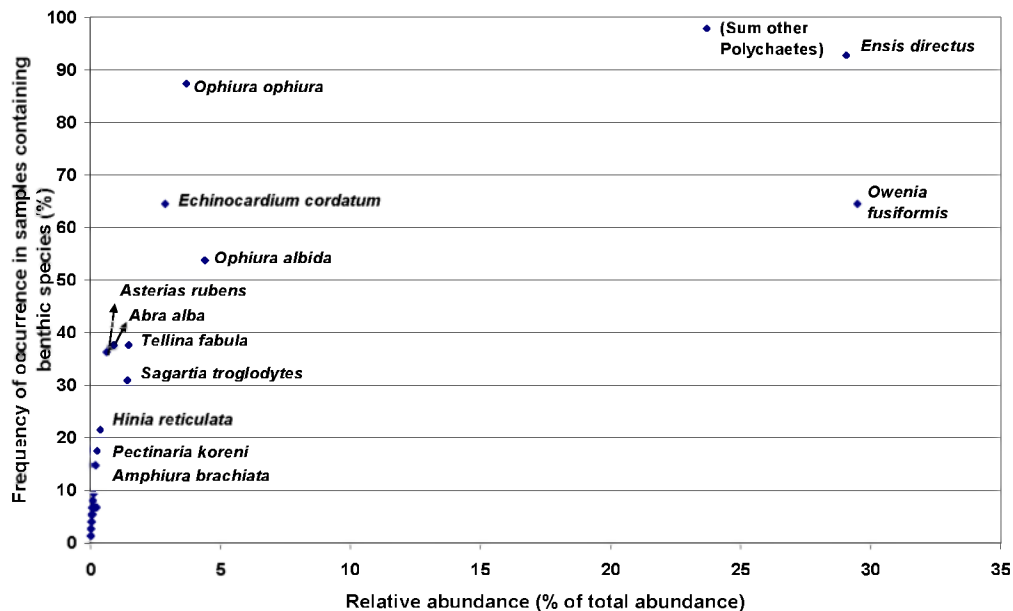
Figure 44. Scatter plot of relative abundance (% of total abundance in samples) and frequency of occurrence (percent of samples where the species is present) for species gathered in winter 2010. Species with frequency of occurrence above 10 % are named.

The polychaete worm *Owenia fusiformis* was found at densities up to 11,000 specimens / m², as observed the same year by Tafara-Breine (2011) on the Vlakte van de Raan. In this survey, highest densities were found on the northern slope of the Wenduine bank, where *E. directus* as well as ophiurid brittle-stars were absent or rare (Figure 45). The green urchin *E. cordatum* was surprisingly well represented in this survey, probably at least partly due to the larger penetration depth of the box-core compared to other sampling gears. The brittle star *Acrocnida brachiata* was found in about 15 % of the sample, pointing at an increase of this formerly rare species in the last years. The sea anemone *Sagartia troglodytes* was noticeably almost always found in large empty shells of *E. directus*, very abundant in most samples.

Compared to the 2008 data-set, the assemblages of epibenthic species are very similar. Endobenthic bivalves are much less abundant. *Abra alba* is the only species present in more than 10% of the samples, and its low relative abundance contrasts with its strong dominance in 2008, despite the fact that the 2010 survey largely overlays areas where this species best thrives (see Degraer et al, 2008).

Multivariate analyses

Bray-curtis similarity was calculated and non-dimensional MDS plots were created (Figure 46). When all taxa are considered including *E. directus*, a clustering (group-average) procedure with SIMPROF test generated few clusters of which one, cluster 'f', dominates the whole survey area. In this cluster, *E. directus* is dominant.
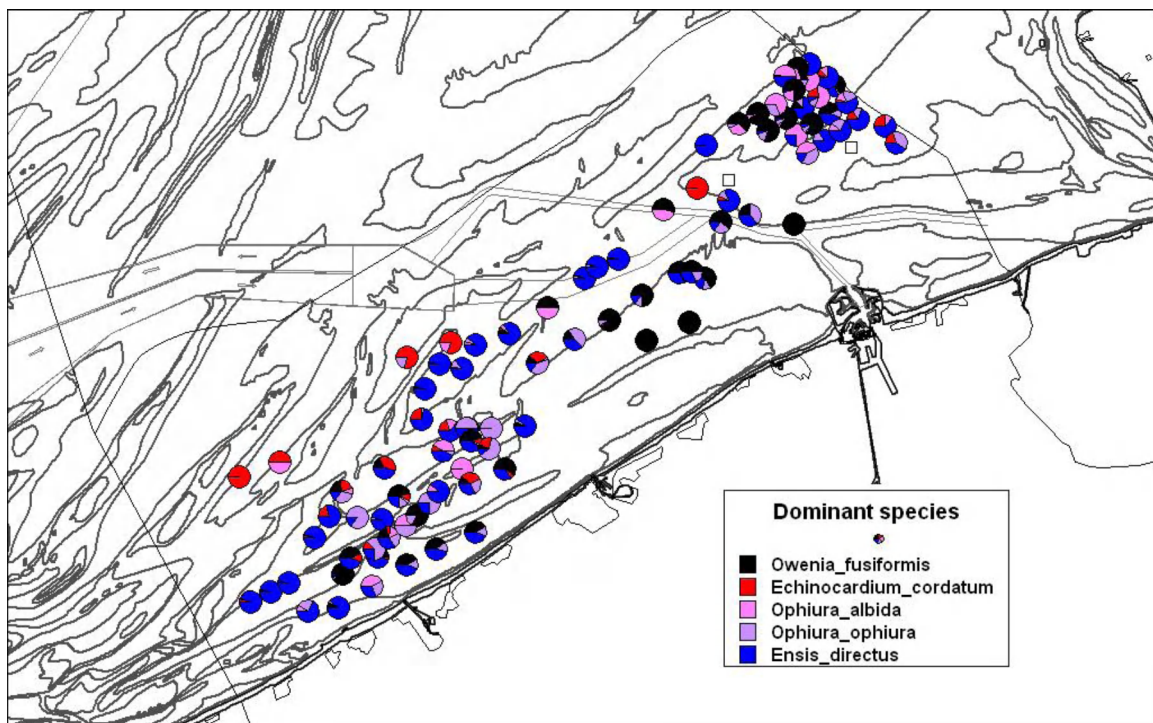
Figure 45. Map of relative numerical contribution of the five dominant species in the winter 2010 survey.

Cluster "e" is characterized by a much larger contribution of the worm *O. fusiformis* (Table 8), which strongly increase total densities (Figure 46). This species assemblage is mainly found to the north of the Wenduine bank and the Vakte van de Raan (see Figure 45). In the MDS ordination of the Bray-curtis similarity matrix, larger densities of adult *Ensis* display no grouped pattern indicative of a possible structuring effect. Samples with more than 1,000 juveniles are clustered close to each other, but their species content does not differ from other samples of the "f" cluster (Figure 47). The lower portion of the ordination is largely driven by some samples bearing very high overall macrobenthic densities, with on one hand, juvenile ensis (cluster "f") and on the other hand, *O. fusiformis* (cluster "e"; see Table 8). Interestingly, the latter samples display variable species richness and Shannon-Wiener diversity index (not illustrated).

When *Ensis* is removed from the data-set prior to Bray-curtis similarity calculation and MDS ordination (Figure 47), the apparent effect of juveniles on sample grouping disappears, evidencing that this grouping results from the higher densities induced by massive juvenile settlement, while species composition remains similar to the rest of the "f" cluster. When only samples of cluster "f" are considered, the same result is obtained (not illustrated). Larger densities of juvenile or adult *E. directus* do not seem to drive the species composition of the local macrobenthos.
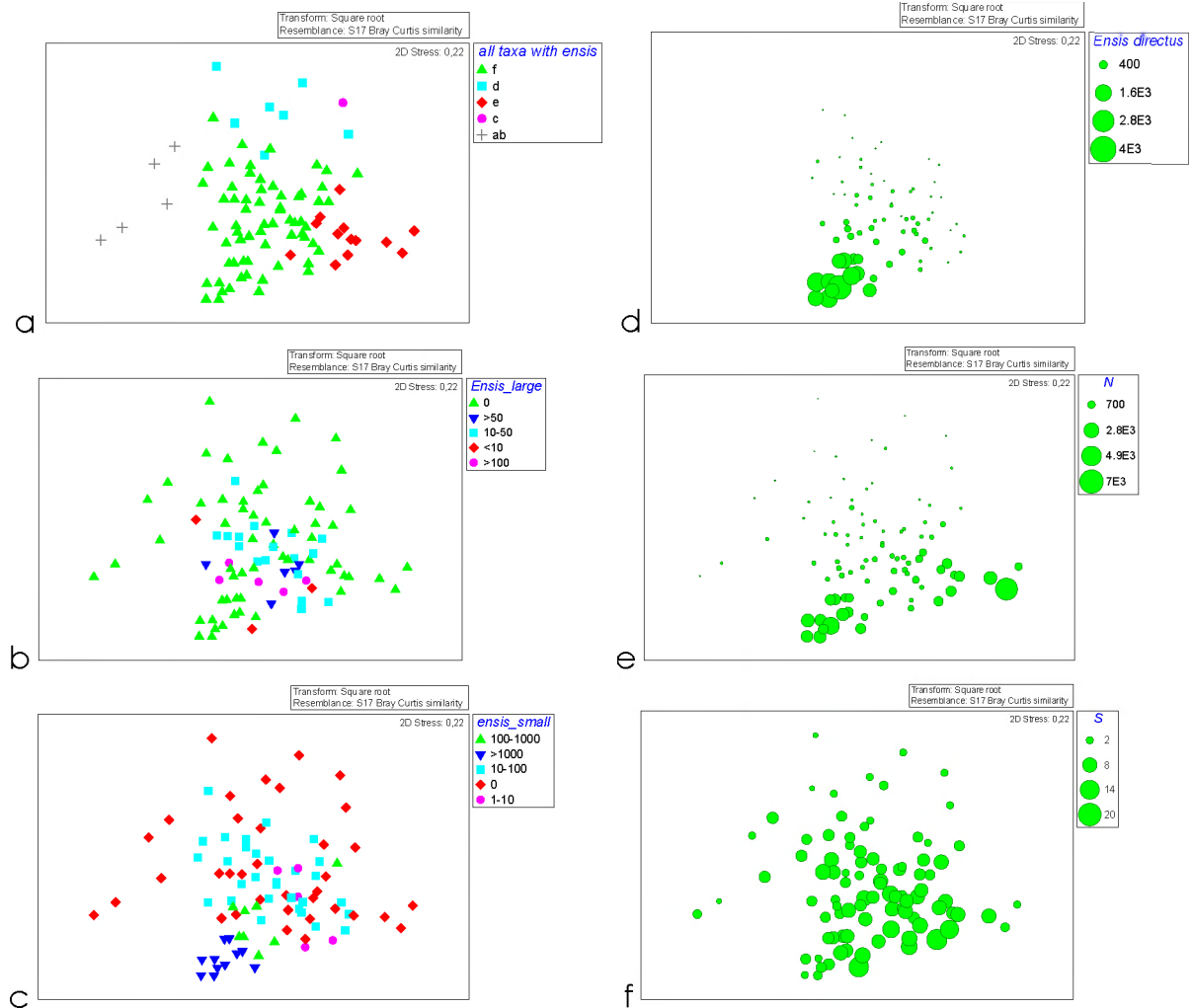
Figure 46. Multivariate ordination (MDS) of Bray-curtis similarities (square-root transformed densities) between the overall macrobenthic content of the samples of the 2010 survey. A. Significantly different clusters of species composition (cluster and Simprof procedure); b. density categories of large (>10 cm) *Ensis*; c. density categories of juvenile (<6 cm) *Ensis*; d. *Ensis* density; e. Total macrobenthic density; f. Species richness.
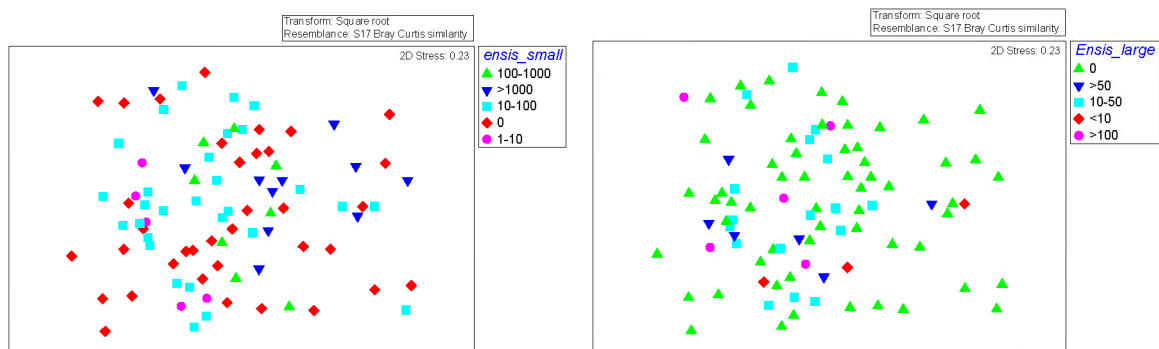
Figure 47. MDS ordination of square root transformed macrobenthic data from 2010 survey, after removal of *E. directus* (Bray-curtis similarity matrix). Left: density categories for juveniles; right: density categories for larger individuals.

Group f (average similarity: 39,41)					
Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Ensis directus	**14,10**	**21,47**	**1,75**	**54,48**	**54,48**
Ophiura ophiura	3,84	7,33	0,99	18,60	73,08
O. fusiformis	3,56	3,77	0,49	9,57	82,65
E. cordatum	2,73	2,52	0,57	6,39	89,03
Ophiura albida	1,95	1,03	0,32	2,61	91,64
Group ab (average similarity: 32,87)					
Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
E. cordatum	5,65	30,37	2,00	92,39	92,39
Group d (average similarity: 46,65)					
Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Ophiura albida	4,69	39,17	3,80	83,97	83,97
Ophiura ophiura	1,99	4,67	0,40	10,02	93,99
Group e (average similarity: 53,02)					
Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
O. fusiformis	**29,16**	**32,81**	**2,64**	**61,88**	**61,88**
Ophiura ophiura	5,28	5,27	1,11	9,95	71,82
Ophiura albida	6,76	4,37	0,67	8,24	80,06
Ensis directus	**3,95**	**2,63**	**0,71**	**4,95**	**85,01**
Tellina fabula	2,89	1,77	0,71	3,34	88,36
Asterias rubens	2,21	1,51	0,69	2,85	91,21

Table 8. Main contributors to the composition of clusters determined with the SIMPROF permutation test.

3.3.4 <u>Conclusions</u>

The determination of the impact of the American razor clam on the local benthic fauna is difficult with the data available in the BPNS, because we lack a homogeneous spatio-temporal coverage before and after the invasion. This is a first important conclusion of this study, as it underlines the importance of a well-designed and regular monitoring programme, as is carried out in the adjacent Dutch waters. Long-term trends of species composition at a station near Oostende shows how variable the species composition of benthos can be in our waters, but no pattern of variability could be specifically associated to the presence of *E. directus* thus far. A major observation is the rarefaction of a typical species of the Belgian-Dutch waters, the cut trough shell *Spisula subtruncata*, on both sides of the border in the mid-2000s. In Dutch waters, it coincides with a strong increase in the biomass of *E. directus*, suggesting a link between the two cases. The local species of razor clams historically thrived further offshore than *E. directus*, in coarser sediment, and *E. arcuatus* at least was still recently collected. The invader likely reduced the geographic distribution of the local species in coastal waters through competition for space and food, but it cannot be held responsible for their apparent rarefaction further offshore in the last decades.

Examination of the data from the two surveys of 2008 and 2010, which both display a broad geographical extent, did not reveal any structuring effect of adults or large amounts of juveniles on the macrobenthos composition. However, further studies including the polychaete fauna, which numerically dominates the local macrobenthos, will have to be carried out to ascertain this conclusion. Side observations were made during the EnSIS survey of winter 2010. A high mortality of bivalves was noted to have taken place, which might be related to particular winter conditions. A clear zonation pattern was observed in front of the eastern coast, thus far little documented, which likely relates to local hydrodynamics and sediment

composition. More unusual was the observed development of the populations of the polychaete worm *Owenia fusiformis*. The widespread distribution and very high densities of *O. fusiformis*, also noted in some other recent studies (Rabaut et al., 2009; Tafara-Breyne, 2011), seem to be recent. These observations are perhaps related to the aforementioned increase of the American razor clam biomass and rarefaction of the cut trough shell, suggesting that a major environmental change took place during the 2000s. The cause for these changes in the local macrobenthos remains to be elucidated.

3.4 Impact of *E.directus* on higher trophic levels: seabirds

3.4.1 Introduction

Since standardised aerial counts started in the sixties, wintering scoters (common scoter *Melanitta nigra* and velvet scoter *M. fusca*) in the Belgian part of the North Sea (BPNS) show major changes both in numbers and distribution. Scoters are diving ducks that feed on prey species that live on or within the upper few centimetres of the substratum (Kaiser *et al.*, 2006). They appear to take any prey that is abundant, that comes in the right size class and that is accessible (Tulp *et al.*, 2010). Bivalves are the main component of the diet with other species incorporated less frequently (Fox, 2003; ICES, 2005). As bivalve stocks may fluctuate strongly between areas and years (e.g. Van Hoey *et al.*, 2007), it can be expected that changes in the distribution, abundance, size/quality and species-composition of bivalves induce changes in the numbers and distribution of scoters.

According to Van Steen (1978), at the end of the 1970s, scoters probably foraged mainly on *Tellina fabula*, *Abra alba* and *Mysella bidentata*, since these prey species were the most abundant bivalves making up the highest biomass. At the beginning of the 1990s, Van Assche (1991) describes that *S. subtruncata*, *Macoma balthica* and *Ensis* spp. made up the highest bivalve biomass along the Belgian coast. Stomach analysis of (oiled) beached birds revealed mostly empty stomachs, bivalve debris found in some stomachs were thought to stem from *S. subtruncata*. This was supported by the presence of some almost intact *S. subtruncata* in the stomach samples. By the end of the nineties, probably *A. alba*, *T. fabula* and *S. subtruncata* made out the most important prey for wintering scoters (Degraer *et al.*, 1999). As *S. subtruncata* is thought to be the staple prey of wintering scoters in the 1990s in Belgium and the Netherlands (Leopold *et al.*, 1995 & 2007; Leopold, 1996; Degraer *et al.*, 2007; Baptist & Leopold, 2009), this prey species can be used to relate to the presence of wintering scoters for that period. Since the start of the 21st century, densities of *S. subtruncata* have decreased dramatically due to failed recruitment (Craeymeersch & Wijsman, 2006, Van Hoey *et al.*, 2007), so sea ducks might have changed their diet towards other bivalves such as *E. directus*.

Since the start of the 21st century, *E. directus* has become very abundant and reaches a very high biomass locally (Goudswaard *et al.*, 2008) so it can be expected to be an important prey source for scoters these days. The first indications that *E. directus* is taken by common scoters in The Netherlands originate from 1995 (Tulp *et al.*, 2010). The first evidence of common scoters effectively feeding on *E. directus* stem from 2003 when scoters were seen diving up *E. directus* at the Brouwersdam in the south of the Netherlands (Leopold & Wolf, 2003; Wolf & Meininger, 2004). Also in Belgium after 2006 scoters were seen to take *E. directus* in near coastal waters.

This section elaborates on the possible link between *Ensis directus* and the occurrence of scoters at the Belgian part of the North Sea. Apart from the distribution and quality of the food, also a number of physical and anthropogenic factors are likely to influence the distribution of scoters. Places where prey is abundant may be unreachable for diving scoters or might only be reached with an expenditure of energy that exceeds the amount of energy gained by foraging there. In this way spots with abundant preys may be less attractive to the birds than places with lower densities of bivalves. Since scoters are extremely sensitive to disturbance (Garthe & Hüppop, 2004), also disturbance by e.g. fishing activities, shipping traffic, (kite-) surfers or military activities might exclude them from potential feeding sites. The presence of larger particles in the sediment, for example, might interfere with foraging efficiency (Kaiser *et al.*, 2006). Along the Belgian coast this seems rather irrelevant since the sediment mostly has a small grain size (Degraer *et al.*, 2008).

In this report these variables will be used to create maps that show spatial variation in the suitability of the habitat for diving sea ducks.

3.4.2 Material and methods

The data-set for scoter was described in section 3.1.1.4. The raw count data of the scoters were converted to a grid of 1x1 km by assigning each observed group to the corresponding grid cell. Because of the marked differences in distribution and numbers before and after the winter of 1998/'99, two distribution grids were created: one for the period 1996/'97 – 1998/'99 and one for 1999/2000 – 2010/'11. No interpolation of numbers was made for this study. Only the grid cells that were intersected by the counting transect were used for further analysis (see Figure 51). Also the data on bivalve densities and physical variables were converted to a 1x1 km grid using the mean and maximum of all raster-cells of the layers of the physical variables within a 1x1 km grid cell.

For determination of food preferences, one weak point in the data has to be pointed out. During the aerial flights neither the side nor the approximate distance to the plane the birds were situated was noted. As such, no exact location of the groups of scoters could be derived. Also, the behaviour (feeding, resting, flying) of the birds was not recorded. Studies of the diet of wintering sea ducks in Belgium and the Netherlands have been hampered by the lack of suitable material. Direct observations of prey consumption by scoters are difficult since most birds feed outside viewing distance. Mostly stomach analyses have been used so far. Several studies have already been conducted to determine the prey of wintering scoters in the Belgian part of the North Sea (Van Steen, 1978; Van Assche, 1991; Degraer *et al.*, 1999). Most studies looked at the composition of the bivalve population underlying the preferred wintering areas of scoters. Direct data from stomach analysis are scarce and data from benthos samples taken exactly at spots where and when the scoters were foraging are as yet not available. This in contrary to for example the Netherlands where more direct data and observations were collected (Offringa, 1991; den Hollander, 1993; Leopold *et al.*, 2010).

For *Spisula subtruncata* and *Ensis directus*, maps predicting the likelihood of occurrence of these species (Habitat Suitability Maps or HSM, Degraer et al., 2008) were available. Only for *E. directus*, interpolated density maps were available through this project (see section 3.2). The likelihood of occurrence of *S. subtruncata* can be used as a proxy for the density of the bivalves and as such for the suitability as a feeding area for the scoters. For both periods an overlay was made between

the scoter distribution grids and the HSM for *S. subtruncata* and *E. directus*. For *E. directus*, an overlay was also made with the density map.

Since it was not possible to create suitability maps for *E.directus* densities because the link with physical parameters proved weak (see section 3.1.5.2), an alternative method had to be followed. In order to be able to model the occurrence and abundance of *E. directus* in relation to the distribution of wintering scoters, a modelling method based on thresholds was used. As for the relevance of *E. directus* for scoters, a threshold for both length and densities had to be determined.

Length threshold

For Belgium, no information on the lengths of *E. directus* eaten by common scoters is available. Koen & van Loon (2007) present data on the length of *E. directus* found in the stomachs of 6 scoters found dead at the Dutch Wadden Sea isles. In total, 23 measurable *E. directus* hinges were collected from these stomachs. The collected shells showed a bimodal length distribution ranging from 3 to 9 cm, with two maxima at 4 and 8 cm (see also Tulp *et al.*, 2010 and Leopold *et al.*, 2008). Skov *et al.* (2008) provide data on 9 measurable *Ensis americanus* found in the stomachs of 29 sampled birds in the Horns Rev II wind park area (Denmark) (see also Freudendahl & Jensen, 2006). Mean length of these was 65.9 mm with a minimum of 45.0 mm and a maximum of 87.8 mm (s.d. 14.1 mm). Also, anecdotic observation of a common scoter suffocated with a specimen of 9.75 cm is available (M. Leopold *in litt.*). Based on the evidence available, it seems that scoters can handle *E. directus* up to at least 9.0 cm long. Occasionally bigger shells might be eaten (e.g. Wolf & Meininger, 2004), but this seems not to be the case very often and as such, 9 cm was chosen as the threshold length for further calculations. Threshold values were calculated for two different length classes: 3.5-6 cm and 6-9 cm.

Density threshold

The density threshold above which foraging on *E. directus* by common scoters becomes profitable is difficult to determine and will largely depend upon prey size and availability. Larger, more energy rich shells are more profitable than smaller shells, but a less accessible (e.g. deep) but large amount of preys can be more profitable than a lower amount in more accessible conditions. Also, the relation between shell length and energy content is not linear but shows a third power function. This implies that a shell that is only a little longer contains much more energy than one that is a little smaller. The scoters constantly must decide whether to eat a lot of small and 'easy-to-swallow' *E. directus* specimens or fewer large *E. directus* with a higher risk of suffocation.

The most important difficulty in determining a profitable threshold of *E. directus* for scoters is the lack of sufficient data on densities of *E. directus* sampled in foraging areas of sea ducks. Here, we based the threshold calculations on Leopold *et al.* (2010) who describe a situation in Noordwijk (The Netherlands) for the winter 2009-2010, where Scoters appeared for several weeks in an area where only *E. directus* was available and left before all available prey was consumed or the winter was over. It is not unlikely that in this situation a threshold was met where foraging was no longer profitable and scoters therefore left the area (M. Leopold *in litt.*).

The Noordwijk area was sampled when the birds were present and after they left. Also two reference areas to the north and to the south of the foraging area were sampled for *E. directus*. After the scoters left the area, *E. directus* with an Ash-free Dry Weight (AFDW) of 9,748 ± 9,453 g/m², a density of 164±154 shells/m² and a median

length of 49,77mm was present. In areas north and south of the area where the scoters resided, *E. directus* was present in respective densities of 174.48±130.50 and 329.96±191.40 ind/m², an AFDW of respectively 9.425±8.128 and 16.95±8.85 g/m², and median lengths of respectively 47,27 and 46,23 mm. Despite the fact the higher densities and AFDW/m² in the southern area, the *E. directus* were apparently too small to be profitable for the scoters. Also, despite the fact that the northern area had an AFDW and shell-length only a little smaller and even occurred in higher densities than in the foraging area, the latter was apparently more profitable. This can possibly be explained by the larger median shell-length (and as such higher energy-content per individual).

As a tentative threshold for *E. directus*, an AFDW of 10 g/m² was taken. Based on the equation for the conversion of shell-length to AFDW in Leopold *et al.* (2010), the density of the different lengths corresponding with this value of AFDW can be calculated. For both classes of *E. directus* (3,5-6 and 6-9 cm), the corresponding densities were calculated. Since the AFDW is a third power function of shell length, we calculated the mean AFDW of shell-lengths in steps of 0,5 cm (35 mm, 40 mm, 45 mm etc) and computed the corresponding density for this mean. For the length class of 3.5-6 cm, the mean AFDW was 0,0482 g/m² and the corresponding density 207 ind/m² (Figure 48). For the length class of 6-9 cm, the mean AFDW was 0,257 g/m² and the corresponding density 39 ind/m². Scoters need around 103 g of AFDW food per winter day (Leopold *et al.*, 2008). This corresponds to about 2000 *E. directus* of 5 cm long (and almost 1900 gr of fresh weight of bivalve flesh) (Leopold *et al.*, 2010).
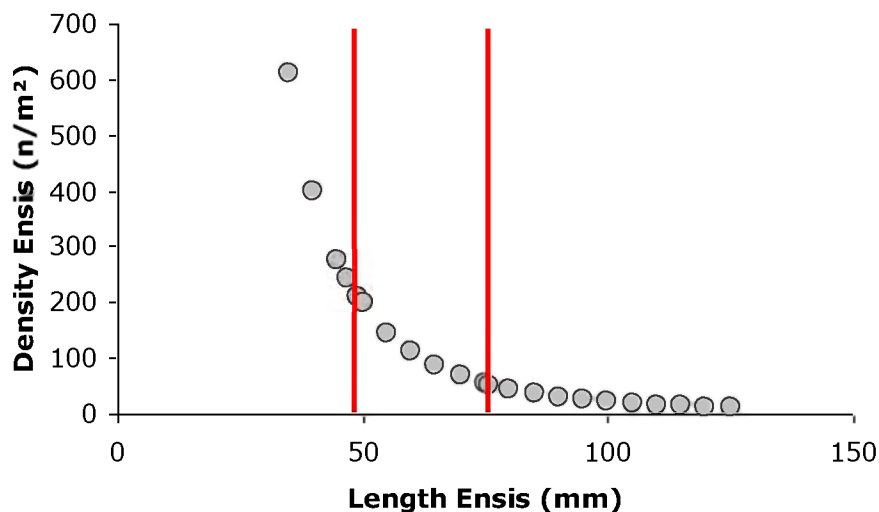
Figure 48. Relation between the length and density of *E. directus* to reach 10 gr of AFDW/m². The red lines give the mean densities for the classes of 35-60 and 60-90 mm.

Only grid cells in which more than 50 scoters were counted between 1996 and 2011 were used for further analysis of the relationship between the occurrence of scoters and physical variables. For prey availability, only grid cells where the average of all groups counted was more than 10 individuals between 1996 and 1999 (*S. subtruncata*) and 2008 and 2011 (*E. directus*) respectively, were retained. The suitability of each 1x1 km grid cell was calculated using a scoring system. For the retained grid cells, the 5, 25, 75 and 95 percentiles were calculated for each variable.

All grid cells in the BPNS of which the value for a certain variable was between the 25 and 75 percentile were scored 1 (most suitable), if the value was between the 5 and 25 or 75 and 95 percentile, it scored a 2 (suitable). Values that were outside the 5 to 95 percentile scored a 3 (least suitable).

For the physical parameter 'depth', suitability scoring was done differently. The maximum diving depth of common scoters is about 20m (Degraer *et al.*, 1999) which in the case of the Belgian coast means that feeding areas of scoters are limited to the more shallow zones. Although the relationship between water depth and energy expenditure might be non-linear (Wilson *et al.*, 2002), water depth directly affects the energy expended during the dive to and from the seabed while foraging. This makes foraging in deeper water energetically potentially less interesting. But of course, water depth over the seabed fluctuates tidally, so prey that is not reachable at high tide might become available during low tide. Grid cells with a mean depth of more than 20 m were given a score of 5 since they are deeper than the maximum diving depth of scoters and as such not suitable as feeding areas. Also, grid cells with a value for depth outside the 5% percentile scored a 2 instead of a 3 since these are quite suitable for scoters because they are very shallow. This was also the case for the depth-averaged current since a lower current speed probably facilitates feeding of scoters since diving ability is not impeded and the displacement speed is low. Grid cells where the likelihood of occurrence of *S. subtruncata* and *E. directus* was higher than 75% all scored 1 since an area with a higher likelihood of occurrence of shellfish also means a more suitable feeding habitat for scoters. For the final suitability map for scoters, the values of all underlying suitability maps were summed. The resulting scores were regrouped and scored from 1 (most suitable) to 3 (least suitable). The percentiles for the different variables are shown in Table 9.

3.4.3 Results and discussion

3.4.3.1 *Numbers and distribution of wintering scoters*

3.4.3.1.1 Fluctuating numbers

Wintering sea ducks have been present along the Belgian coast at least since the first half of the 19th century (Quinet, 1897). Until 1950 common scoters were frequently observed, especially in the western coastal bank area. Velvet scoters and eider ducks seemed to be present in much lower numbers. Numbers for that period are expressed in terms such as 'relatively large numbers' and 'more than in other years' and are as a consequence not very accurate (Van Waeyenberghe *et al.*, 2001). From 1950 onwards, some very cold winters occurred, bringing thousands of common scoters from the northern wintering grounds to the Belgian west coast between Nieuwpoort and the French border. Especially the winter of 1955/'56 was very harsh. As a consequence, in the summer of 1956, between 10.000 and 12.000 scoters on moult migration were counted (VLAVICO, 1989).

From 1965 onwards, better data on numbers and distribution of wintering sea ducks became available both from land based counts and from aerial flights. During the last 47 years, the numbers of scoters on the BPNS have been fluctuating a lot, both between years (Figure 49) and during winters. This was also the case in the Dutch wintering population (e.g. Berrevoets & Arts, 2003) with which the Belgian birds probably form a single wintering population (e.g. Leopold, 1996; Hoekstein *et al.*, 2003). Despite these fluctuating numbers, largely four periods can be distinguished.

From 1965 to 1975, in most winters 3000 to 4000 scoters were counted, with peaks of 7000-8500 individuals in three winters. Between 1975 and 1985, there is a lack of data

since no aerial counts and only few land-based counts were conducted, so the numbers for this period should be treated with caution. During this period the number of wintering scoters were low in most winters, with only several hundreds of birds present. The peak count in that period was more than 2200 in the winter of 1980/'81. Since 1986, systematic aerial counts were organised every winter and as such good data are available (e.g. Maertens *et al.*, 1988 & 1990; Devos, 1990). Between 1986 and 2002, numbers of wintering scoters were relatively high. In most winters more than 5000 birds were counted with peaks of more than 10.000 in 6 winters. After 2002 the number of wintering sea ducks dropped quite dramatically. Only in three winters the maximum count exceeded 1000 birds. In the winters of 2009/'10 and 2010/'11, sea ducks were almost absent with only very small groups observed.
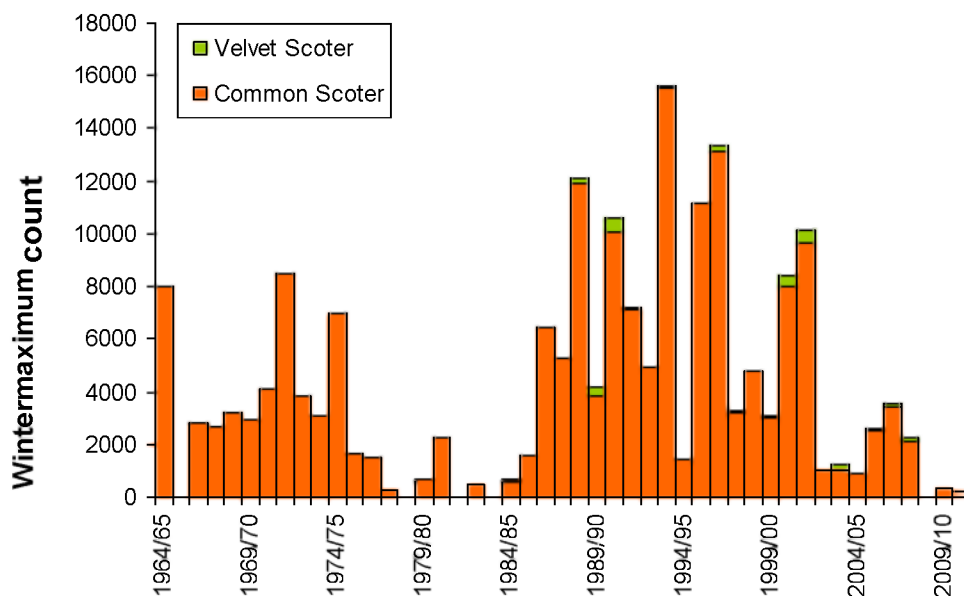
Figure 49. Winter maxima of Common and Velvet Scoter in the Belgian part of the North Sea between 1964/'65 and 2010/'11.

3.4.3.1.2 Changing spatial distribution

Before 1965, data on the distribution of wintering sea ducks are quite scarce. Nevertheless, all available information indicates that the largest concentrations were to be found in the western coastal area, between the French border and Nieuwpoort. Aerial and land based surveys between 1965 and 1978 confirmed the importance of the area between Nieuwpoort and the French border. Only very small numbers were occasionally seen east of Nieuwpoort. Due to the lack of data between 1978 and 1986, nothing can be said about the distribution of sea ducks in this period. Between 1986 and 1999, the most important wintering grounds seem to have shifted to the area between Nieuwpoort and Ostend (Figure 50).
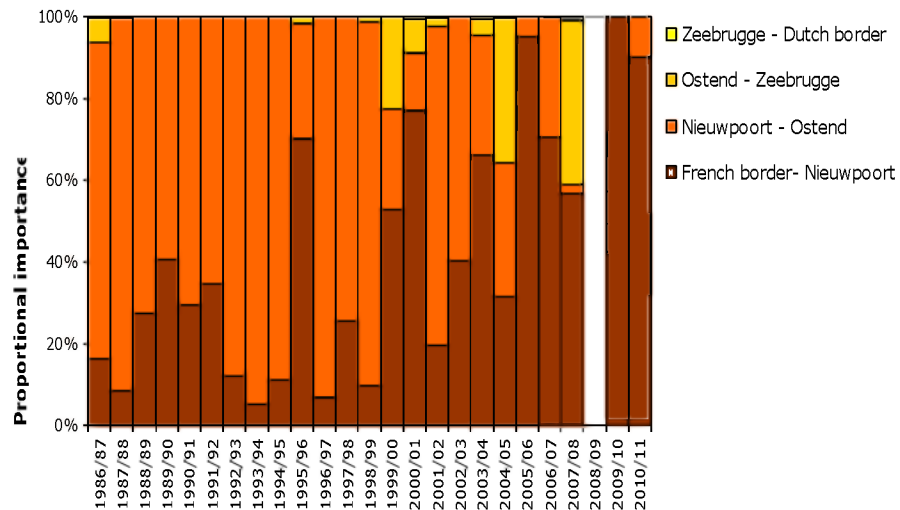
Figure 50. Distribution of wintering scoters *Melanitta sp.* along the Belgian coast between 1986/'87 and 2010/'11.

Especially the Nieuwpoort bank and the area north of the Stroombank held high numbers during that period, while in the western coastal area only few birds were observed. This pattern changed completely from 2000 onwards. The scoters seem to have spread out over most of the coastal area with even some observations of larger groups east of Ostend. Nevertheless, the main wintering areas were still situated west of Ostend. The Nieuwpoortbank lost a lot of importance. The area between Ostend and Nieuwpoort, especially the Stroombank itself seemed to be attractive to scoters. The majority of the scoters, however, were found between the French border and Nieuwpoort with the shallow zone in front of Oostduinkerke often holding large concentrations of wintering scoters. Elsewhere, the scoters were spread out over the western sandbank area with a concentration spot just south of the Smal bank.
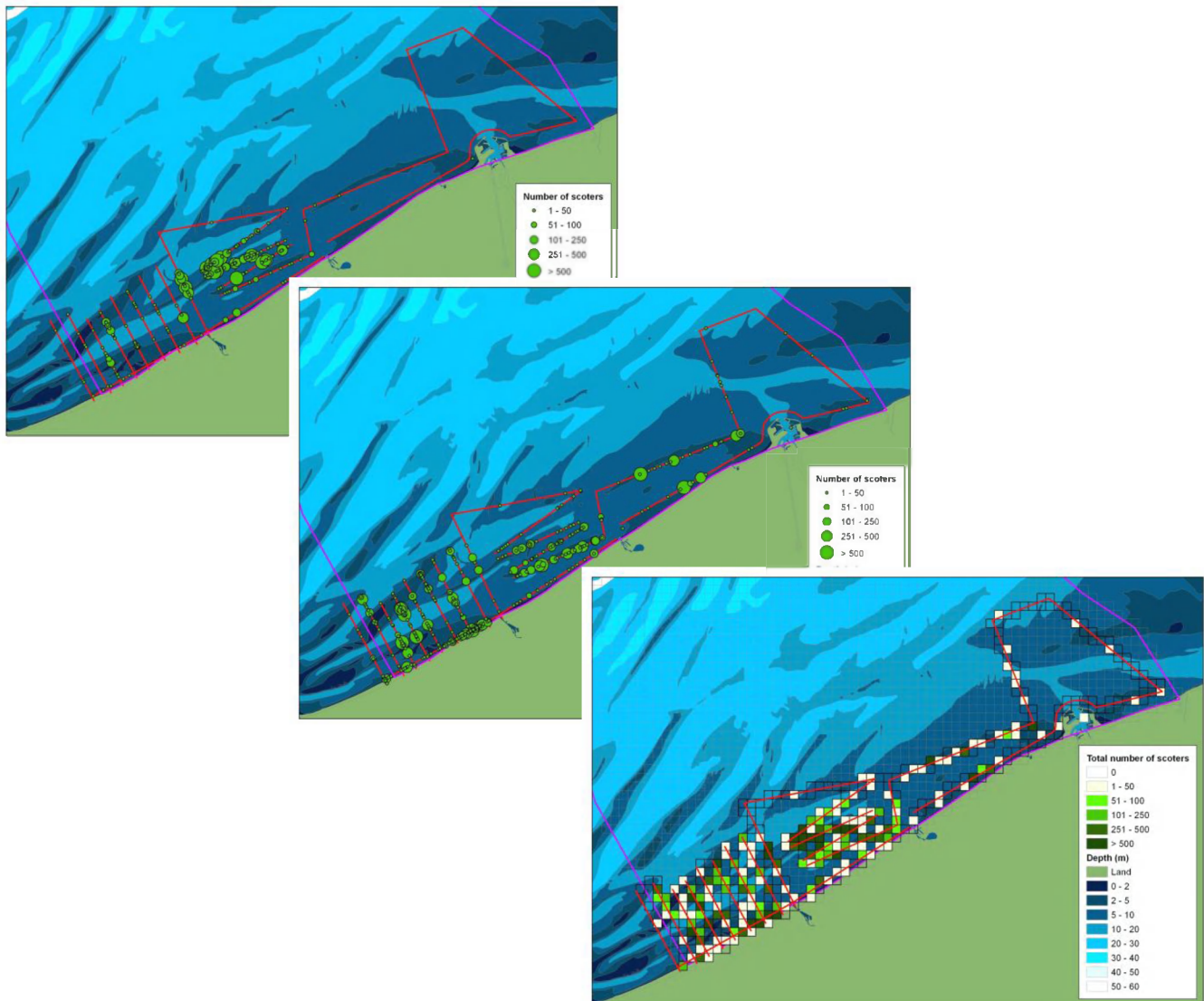
Figure 51. Upper: distribution of wintering scoters *Melanitta* spp. along the Belgian coast between 1996/'97 and 1998/'99. Middle: distribution of wintering scoters *Melanitta* spp. along the Belgian coast between 1999/2000 and 2010/'11. Lower: total of wintering scoters *Melanitta* spp. per km² between 1996/'97 and 2010/'11 along the Belgian coast. The grid-cells used for the analysis are bordered in black. The counting transects are indicated in red.

3.4.3.2 *Relation to prey availability*

3.4.3.2.1 *Spisula subtruncata* as prey for scoters

Before the winter of 1999/2000, there is a clear link between the likelihood of occurrence of *S. subtruncata* and the presence of sea ducks. Scoters preferred grid cells where the likelihood of occurrence of *S. subtruncata* is notably higher than in areas where no scoters were counted (Figure 52; ANOVA p<0,001). Most sea ducks were present in grid cells where the likelihoods of occurrence were between 70 and 85%. The percentiles used to create the suitability map for scoters based on the maximum likelihood of occurrence of *S.subtruncata* are given in Table 9.

Percentile	Current speed (m/s)	Total suspended matter	Mean depth (m)	Spisula (% chance of occurrence)	Ensis (% chance of occurrence)
5	0.75	7.48	1.67	24.6	13
25	0.82	9.64	4.34	65.0	42
50	0.85	11.53	5.94	75.9	77
75	0.91	13.41	7.92	83.0	91
95	0.99	20.50	12.73	89.7	92

Table 9. Percentiles of physical variables in grid cells of 1x1 km where more than 50 scoters *Melanitta* sp. were counted between 1996/'97 and 2010/'11 and percentiles of likelihood of occurrence of *S. subtruncata* and *E. di-rectus* based on grid cells of 1x1 km where the average of all groups counted was more than 10 between respectively 1996/'97 and 2010/'11.

Figure 52 shows the suitability of the BPNS for scoters based on the maximum likelihood of occurrence of *S. subtruncata* per 1x1 km grid cell for the period 1996/'97 to 1999/2000. Most of the larger groups of scoters (> 50 individuals) were present in the area that scores 1 in the suitability map. Nevertheless, several areas with high and medium prey abundance are apparently not used by scoters. Here, one must bear in mind that in this map no physical constraints were incorporated. Most are areas further than 6 nautical miles at sea where the prey is found at depths not available for the scoters.
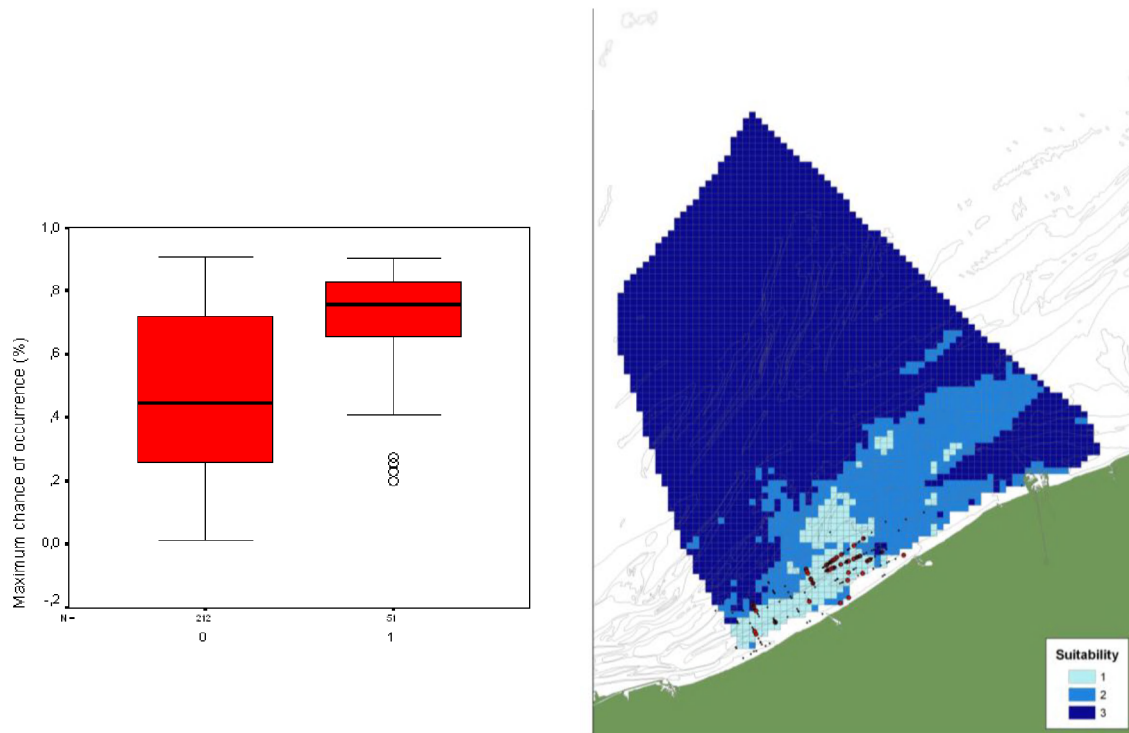
Figure 52. Left: relation between the maximum likelihoods of occurrence of *S. subtruncata* and scoters *Melanitta sp.* in grid cells of 1x1 km between 1996/'97 and 1998/'99 in the Belgian part of the North Sea. Only cells where the average group size was more than 10 birds were retained. Right: suitability for scoters *Melanitta* sp. based on the maximum likelihood of occurrence of *S. subtruncata* between 1996/'97 and 1998/'99. Groups of scoters larger than 50 birds are indicated by red dots, smaller groups by small black dots.

3.4.3.2.2 *Ensis directus* as prey for scoters

Due to the lack of specific data of samples taken in scoter feeding areas, an interpolation of the data of sampling campaigns performed in Belgian coastal waters in 2008 and 2010 was used to compare with the scoter distribution in 2008/'09

– 2010/'11. In grid cells where scoters were present both the maximum likelihood of occurrence and the densities of *E. directus* was higher than in areas where no scoters were observed (Figure 53; ANOVA p<0,01 and p=0.06 when outliers omitted). Figure 54 gives the suitability maps for scoters based on the maximum likelihood of occurrence and densities of *E. directus*.

Figure 53. Relation between the presence of scoters *Melanitta sp.* in grid cells of 1x1 km between 2008/'09 and 2010/'11 in the Belgian part of the North Sea and the maximum likelihood of occurrence (left) and densities of *E. directus* (right). Only cells where the average group size was more than 10 birds were retained.
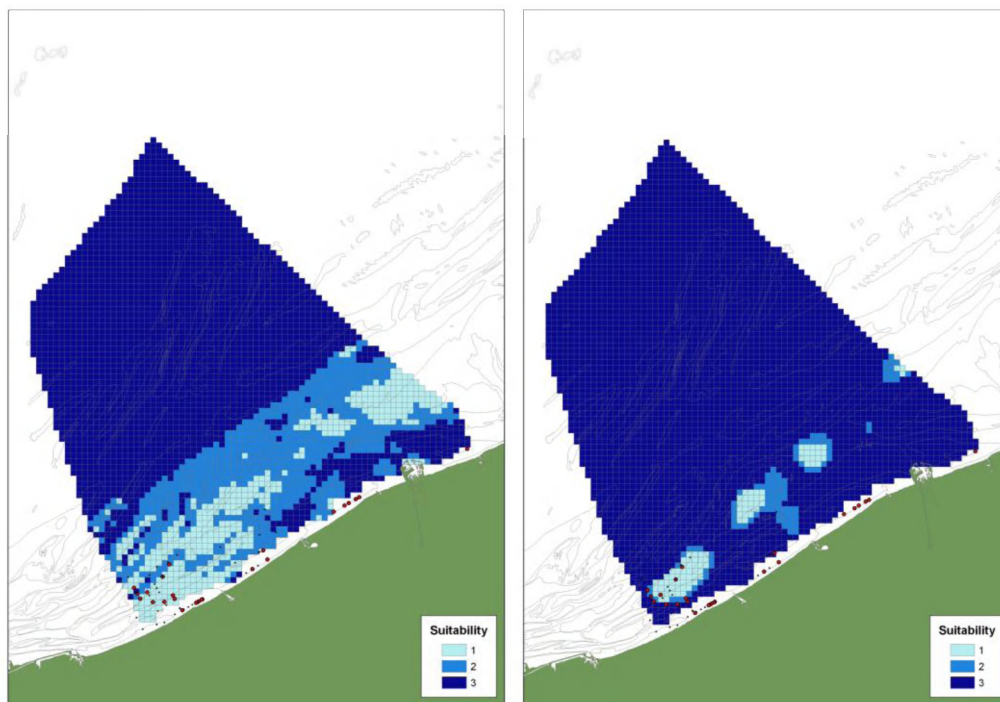
Figure 54. Suitability for scoters *Melanitta sp.* based on the maximum likelihood of occurrence of *E. directus* (left) and densities of *E. directus* (right) between 2008/'09 and 2010/'11 in the Belgian part of the North Sea. Groups of scoters larger than 10 birds are indicated by red dots, smaller groups by small black dots.

Apparently, in a large part of the BPNS the likelihood of occurrence of *E. directus* is high enough to qualify as a feeding area for scoters, but the calculated density thresholds for *E. directus* were almost nowhere met. This might very well explain the

low numbers of wintering scoters in recent years. Apparently feeding thresholds were not met as a result of a lack of sufficient prey in the BPNS.

3.4.3.3 _Relation to physical variables_

Apart from the availability and quality of prey, also physical parameters such as water depth, turbidity and current speed determine the distribution of scoters.

3.4.3.3.1 Water depth

At the Belgian coast, the majority of the scoters between 1996 and 1999 were found in waters of 4 to 9 m deep (Figure 55) with a mean depth of 6,9 m. After 2000, most groups were seen over even shallower waters of 3 to 7 m with a mean of 6,0 m. Since it was not possible to hindcast the real water depth over which the scoters were foraging when they were counted by means of tidal models, these values have to be regarded as a mean since the exact water depth at that moment is not know. A result of this is the occurrence of scoters in waters with a depth of 0 to 2 m, which is probably a bias caused by tidal fluctuations in water depth. Since the suitability map for diving depth (Figure 55) is based on the same bathymetry layer as the one on used to calculate the preferred depth by the scoters, this is not a real problem for this purpose. Table 9 gives the percentiles and suitability scores for water depth.
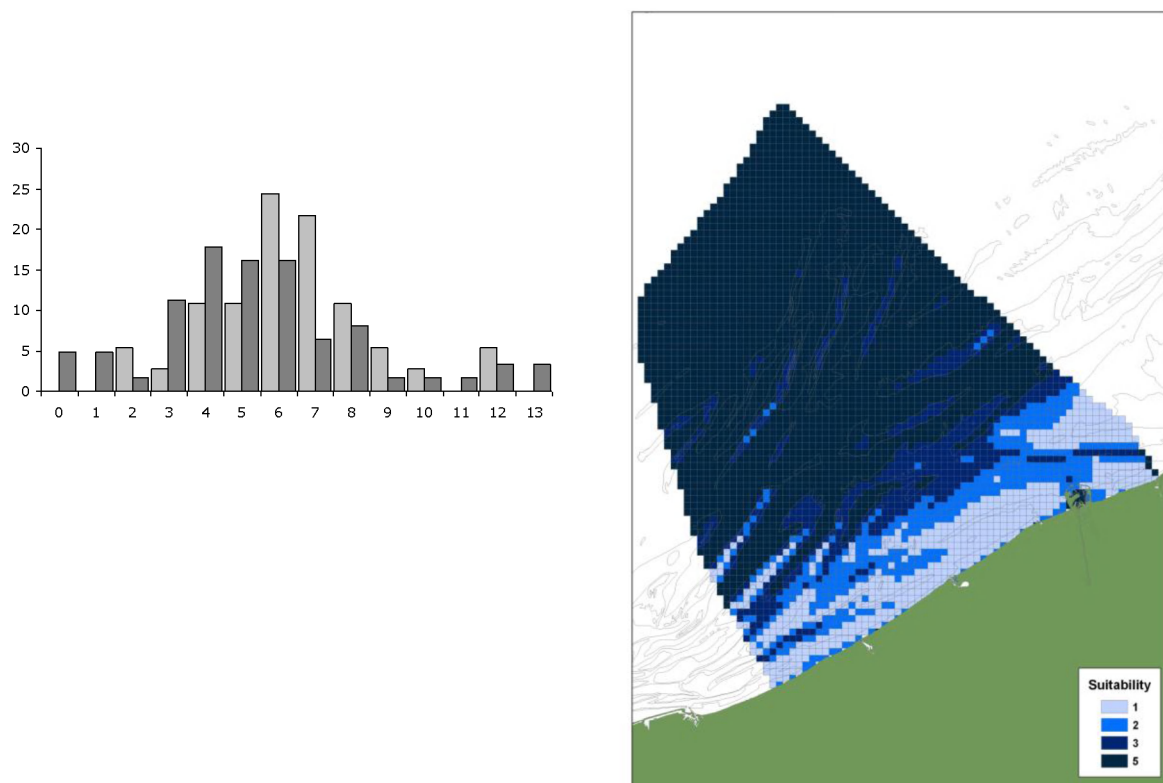
Figure 55. Left: mean water depth of grid cells of 1x1 km where more than 50 scoters were counted between 1996/'97 and 1998/'99 (light grey) and between 1999/2000 and 2010/'11 (dark grey). Right: suitability of the BPNS for foraging scoters based on mean water depth per km².

3.4.3.3.2 Current speed

The occurrence of suitable prey for scoters can be very patchy (e.g. Degraer *et al.*, 2007). As such, the birds have to negotiate the surface current speed, which moves them away from the preferred feeding area. Here, the depth-averaged current was analysed. In the BPNS, the depth averaged current basically varies between 0,5 and 1,5 m/s. There is a significant, although small, difference between the grid cells where scoters occurred and where they were absent (ANOVA p=0,015). The current speed was lower where scoters were present (Figure 56). Table 1 gives the percentiles and suitability scores for depth-averaged current. Figure 56 gives the suitability map for foraging scoters based on depth-averaged current.
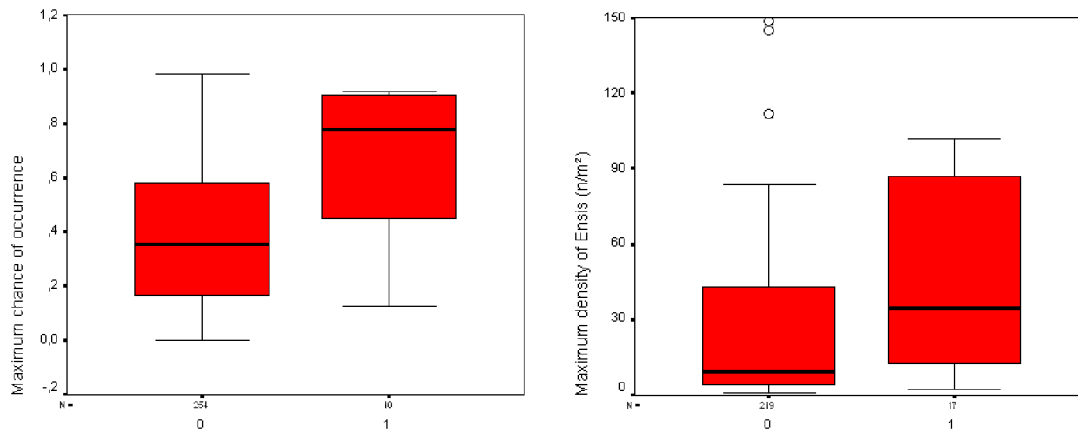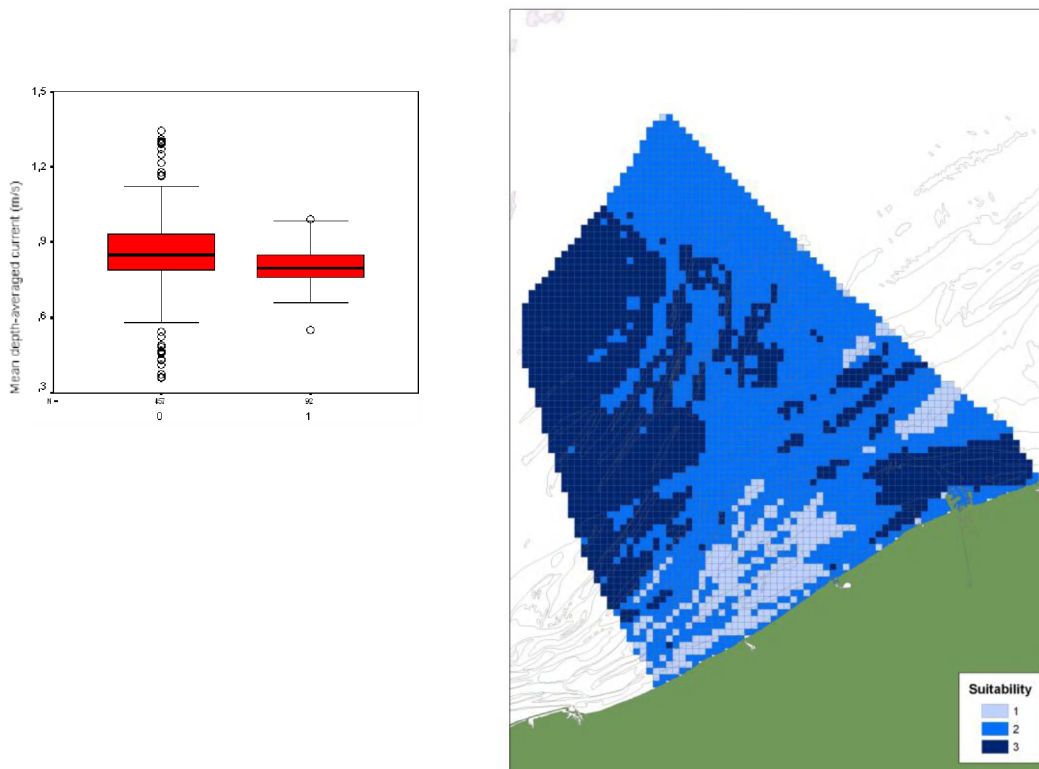

Figure 56. Left: relation between the mean depth averaged current and the occurrence of scoters *Melanitta sp.* in grid cells of 1x1 km between 1996/'97 and 2010/'11 in the Belgian part of the North Sea. Only cells where more than 50 scoters were counted were retained. Right: suitability of the BPNS for foraging scoters based on mean depth averaged current per km².

3.4.3.3.3 Water turbidity

We found a significant relationship between the occurrence of scoters and the mean of the 'total suspended matter' which is used here as a proxy for turbidity (Figure 57; ANOVA p<0,001). Along the Belgian coast, scoters preferred waters with somewhat lower amounts of suspended matter. The exact feeding mechanism of scoters (visual, tactile or both) is unclear. Kaiser *et al.* (2006) state that it is unlikely that scoters are visual feeders since the waters in their study area hold high numbers of feeding scoters although it is very turbid. Probably the birds find their prey by diving at a suitable spot and probing through the mud until they find a suitable prey item.
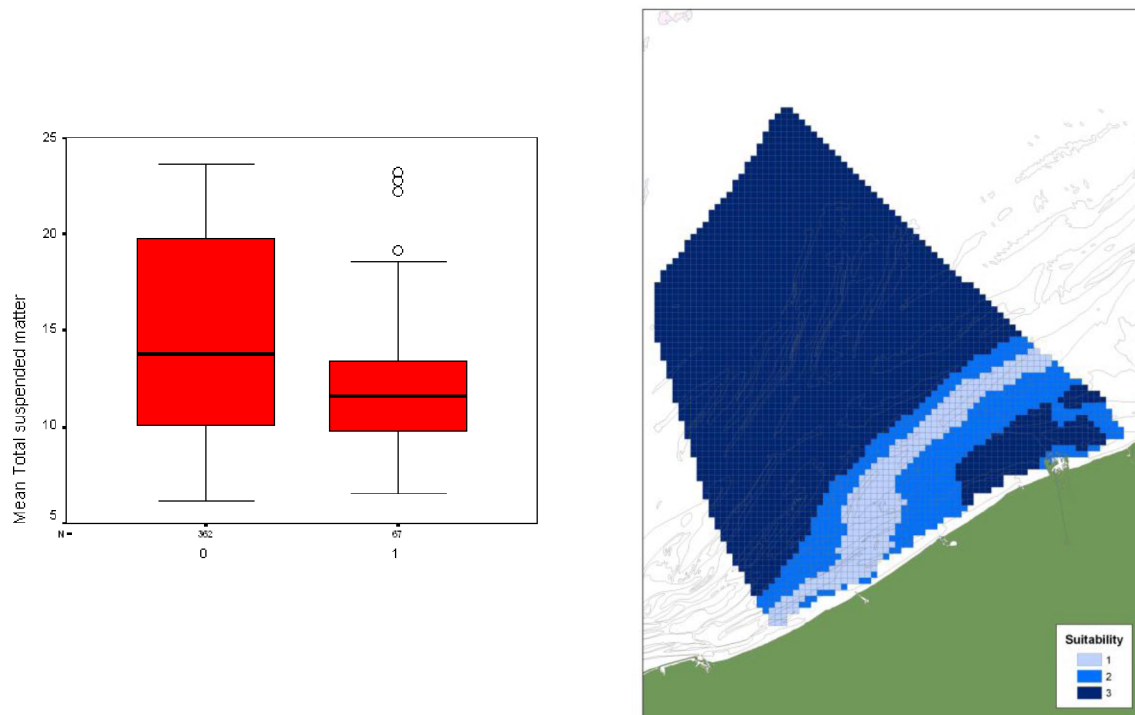
Figure 57. Left: relation between the mean total suspended matter and the occurrence of scoters *Melanitta sp.* in grid cells of 1x1 km between 1996/'97 and 2010/'11 in the Belgian part of the North Sea. Only cells where more than 50 scoters were counted were retained. Right: suitability of the BPNS for foraging scoters based on mean total suspended matter per km².

3.4.3.3.4 Disturbance

Figure 58 shows the biggest shipping lanes (mainly commercial ships such as container vessels, bulk carriers etc) in the BPNS. Scoters are suspected to avoid these areas. Most of the shipping activities (fisheries) and other disturbance factors (wind- and kite-surfing etc) are not represented in this map but can also play a significant role in determining the accessibility of feeding areas. Scoters are among the seabirds that are most sensitive to disturbance by human activities and man-made structures (e.g. windmills, oil platforms). On a scale of 1 (not sensitive) to 5 (very sensitive), common and velvet scoters were the only two seabird species out of 26 to score the maximum in a sensitivity analysis performed by Garthe & Hüppop (2004). Flush distances vary according to flock size. Kaiser *et al.* (2006) found flush distances of 1000-2000 m for larger groups while small flocks flushed at distances of less than 1000 m. Although we acknowledge that disturbance is an important factor to take into account, we do not include this in the final suitability maps due to a lack of distribution data on potentially important sources of disturbance such as recreation activities and fisheries activities.
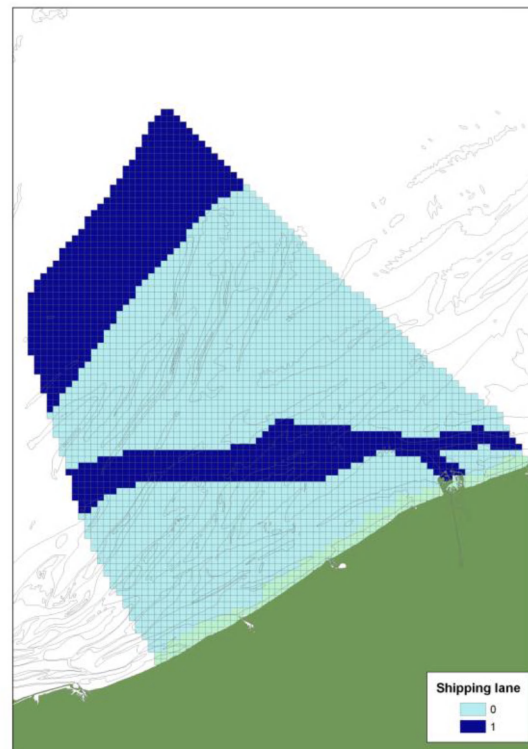
Figure 58. Commercial shipping lanes in the BPNS.

3.4.3.3.5 Integration of physical parameters: suitability for scoters

Integration of mean depth, depth-averaged current, bottom current, maximum total suspended matter and mean total suspended matter was carried out. The suitability map with mean depth, depth-averaged current and mean total suspended matter gives the best fit with the scoter data. Figure 59 gives the resulting suitability map for the considered physical variables. The most suitable part (suitability score of 1) is an area between De Panne and De Haan and about 6 nautical miles from the coast. Almost the entire area within 10 nm (with an exception of a large area in front of Zeebrugge (high current and high level of total suspended matter) and some deeper areas at the west coast) is at least 'medium' suitable and could be expected to attract sea ducks given the prey availability is high enough.

The most suitable zone holds the highest number of groups of scoters (Figure 59). Also the mean number of scoters per group (of all groups and of all groups larger than 25 individuals) occurs in the most suitable zone. In the medium zone also some smaller groups were seen, in the least suitable zone scoters were almost absent.
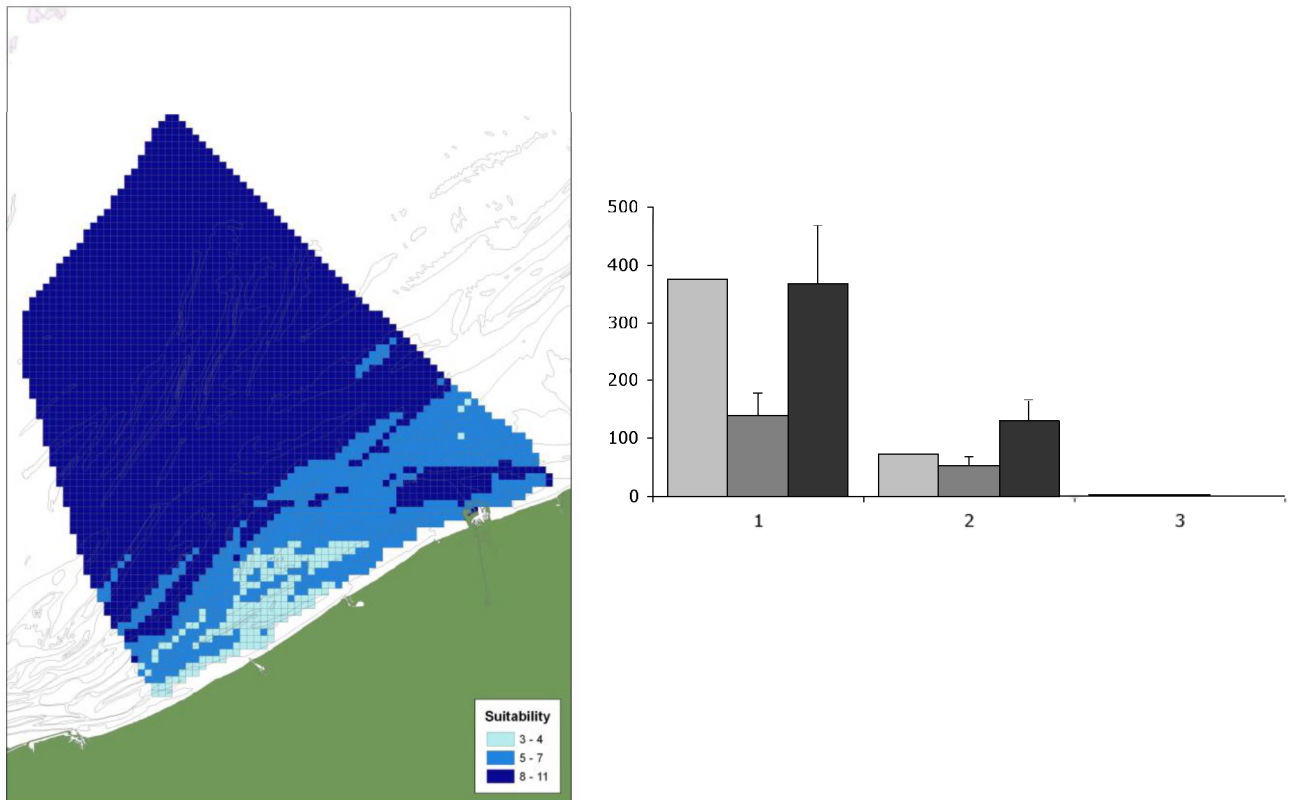
Figure 59. Left: suitability map for scoters *Melanitta sp.* in the BPNS based on integration of physical variables. Right: total number of scoter groups (light grey bar), mean number per group (medium grey bars, all groups, + s.e.) and mean number per group larger than 25 birds (black bars, + s.e.) per suitability class.

3.4.3.4 *Final suitability map*

The final availability maps were made by combining the suitability map based on physical variables and the suitability maps based on the maximum likelihood of occurrence of *S. subtruncata* and *E. directus*. For *E. directus*, an additional availability map was made based on the threshold densities (Figure 60). There is a good overlap between what is expected to be prime foraging area for scoters and the actual occurrence of the birds with only the area between Nieuwpoort and Oostende being really suitable at the end of the nineties. A large area that seems to be good or medium based on the occurrence of *S. subtruncata* is excluded by the physical suitability map. Apparently the likelihood of occurrence is not high enough to overcome the impediments posed by the less-than-optimal physical characteristics of that area.
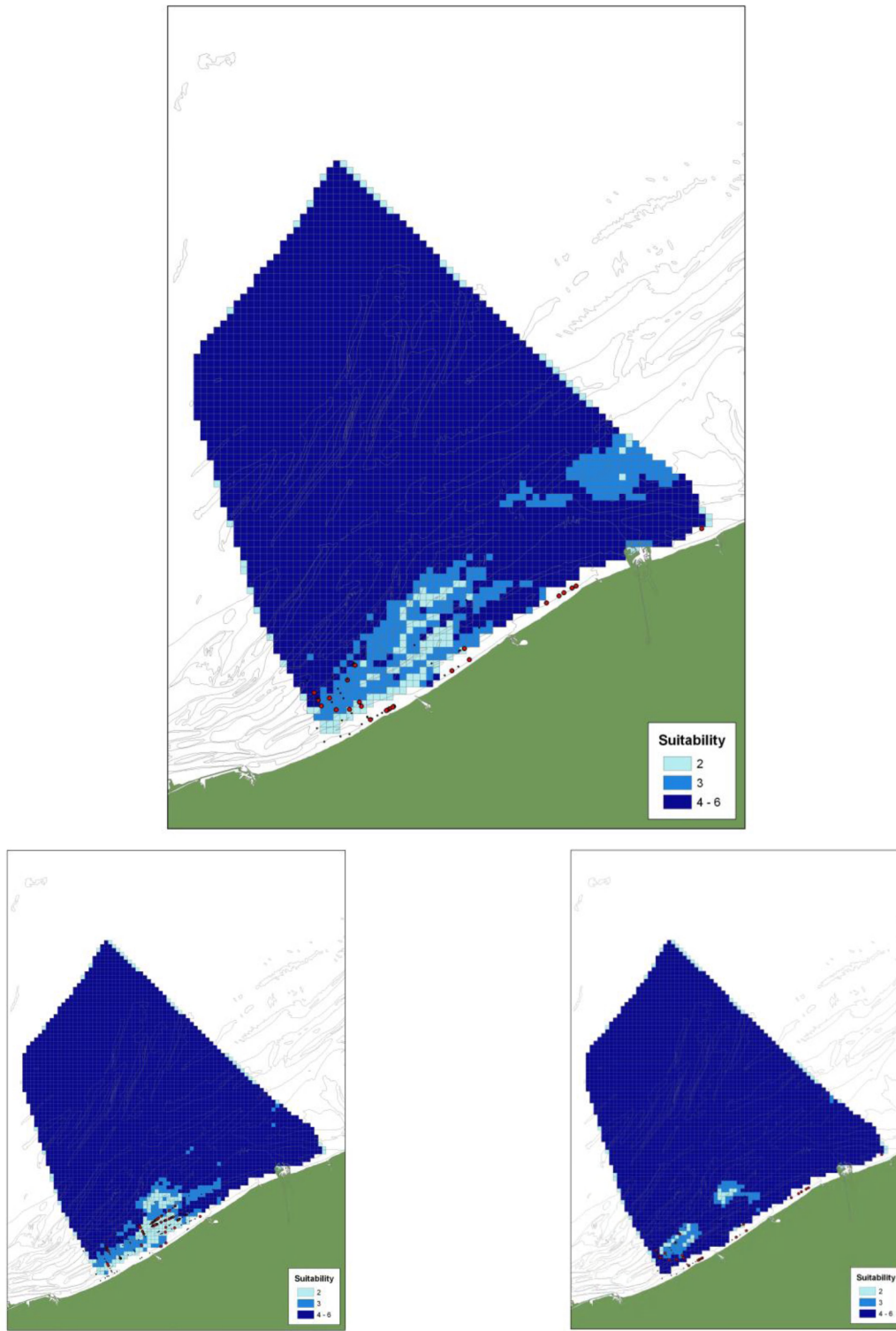
Figure 60. Upper: final suitability maps of the BPNS for scoters *Melanitta sp.* based on physical variables combined with the availability of *E. directus* in 2008 and 2010. Lower, left: final suitability map of the BPNS for scoters *Melanitta sp.* based on physical variables combined with the availability of *S. subtruncata* between 1996/'97 and 1998/'99. Groups of scoters larger than 50 birds are indicated by red dots, smaller groups by small black dots. Lower, right: idem, based on *Ensis* densities as measured in winter 2010. Groups of scoters larger than 10 birds are indicated by red dots, smaller groups by small black dots.

Even when combined with the physical suitability map, a fairly large area of the BPNS between the coast and 10 nm is apparently suitable as a foraging area for scoters. Only when the density thresholds are imposed, it becomes clear that between 2008 and 2010 only in a very small area the densities of *E. directus* are high enough to be suitable for scoters. Unfortunately, the models do not include the near-coastal zone, where more suitable feeding habitat might occur.

3.4.4 <u>Conclusions</u>

Scoters have at least since the first half of the 19th century been present along the Belgian coast, but both numbers and distribution have changed a lot. The highest numbers have always been present to the west of Ostend but the birds showed marked changes in preferred foraging areas. Since scoters are quite opportunistic feeders that forage on the (mostly bivalve) prey that is most abundant and accessible, the driving forces behind these shifts can be expected to be related to changes in the underlying benthic prey community. In the 1990s, *S. subtruncata* was the main prey source for scoters along the Belgian coast. During that period, most scoters were indeed present at places that were physically suitable and with a high change of occurrence of *S. subtruncata*.

After 1998, the *S. subtruncata* stock collapsed and the birds had to look for other prey sources which brought about a change in wintering areas. One possible prey species after 2000 is *Ensis directus*. The distribution of scoters after 2008 coincides with areas with a strong possibility of occurrence and with high densities of *E. directus*. But only very few spots seem to hold profitable amounts of *E. directus* and the few stomachs of Common Scoter that were analysed held no remains of this bivalve. This possible low prey availability coincided with extremely low numbers of scoters along the Belgian coast during the past three years. In contrast, in the same period, the Dutch coastal waters still held high numbers of scoters. Here, much higher densities of *E. directus* that were apparently profitable for foraging ducks were found.

Not only prey availability, but also physical variables such as water depth, current speed and turbidity influence the prey availability and as such the distribution of scoters. The most suitable part of the Belgian part of the North Sea for scoters is an area between De Panne and De Haan between the coast and about 6 nautical miles from shore. Almost the whole area within 10 nm from the coast is more or less suitable (based on physical variables) with an exception of a large area in front of Zeebrugge and some deeper areas at the west coast. Scoters are very sensitive for disturbance and reduction in prey abundance (depletion of bivalve stocks). When looking toward economic exploitation of *E. directus*-stocks, care has especially to be taken when this would be the case in the most suitable part along the Belgian coast.

3.5 Potential for an *Ensis*-targeted fishery in Belgium and preliminary impact assessment

3.5.1 <u>Introduction</u>

The analyses carried out during this project enable to shed some light on two major aspects of the issue associated to the implementation of a targeted *Ensis* fishery, namely the profitability and sustainability of such a fishery (standing stock biomass, expected yields) and the potential impact on the ecosystem.

Next to the scientific assessment of these issues, the practical implementation raises questions by various stakeholders that need to be addressed. For this reason, a round table was held during the final workshop of the project. This chapter summarizes the outcome of the data and literature analysis as well as the outcome of the workshop. Firstly, the experience acquired in Dutch waters with this fishery is used as a reference framework to assess the situation in Belgian waters. The potential yields and the sustainability of such a fishery are assessed. The socio-economic aspects fall outside the scope of this research project. The potential pressure exerted on ecosystem component is also evaluated on the basis of a literature study and the findings of project.

3.5.2 <u>Razor clam fisheries in the North Sea</u>

Thus far, two *Ensis* fisheries exist in the North Sea, in Scotland and in The Netherlands. In Scotland, the fishery was reviewed by Breen et al. (2011). It is practiced on the North Sea (Firth of Forth) as well as on the North Atlantic coasts on the local species *E. arcuatus*. It has been for long mainly operated by divers. Since the 2000s, the fishery is increasingly based on hydraulic dredging (see Hauton et al., 2002). This technique makes use of a dredge, in front of which a pressurized water jet fluidizes the sediment over a height of 30 cm. The fishing gear is towed gently on the seafloor at a very low speed, the ship navigating towards its anchor. As the sediment gets fluidized, it passes through the dredge and larger bivalves are retained. A similar fishery exists in Ireland (Addison et al., 2006). Breen et al (2011) reported on the recent development of electric fishing gears in Scotland. This technique is gaining increased attention because of its reduced impact on the seafloor structure (Woolmer et al, 2011).

The hydraulic dredge is used in coastal Dutch waters (Voordelta and North of the Wadden Sea islands) for the invasive razor shell *E. directus*. Information about the state of this fishery was provided by Keus (2008). Eight fishing licenses are granted, of which five are currently used, and a total of 2,400 hours fishing is allowed per year (equivalent to 200 fishing days of 12 h). Four vessels seem to be active in the Voordelta area, one off the Wadden Islands. The sector has proposed to limit the maximum catch to 8,000 tons per year, which is estimated at a total fished area of 160 hectares (1.6 km²) based on an average catch of 5 Kg/m². Up to 2008, the production ranged between 2,000 and 3,000 tons for an estimated fished surface area of about 0.6 km². This fishery is demand-driven. The minimum landing size imposed by European legislation is 10 cm (EU Council Regulation 850/98).

Stock estimates are carried out yearly by IMARES in the framework of their shellfish monitoring programme. The penetration of the sampling device into the seabed is a crucial factor in measurements of density and biomass of benthic animals especially if they are deep-living or deep-burrowing in sandy sediments (Beukema, 1974; Heip et al., 1985; Lie and Pamatmat, 1965). Based on 400 stations, sampled both with a

box-corer and a trawled dredge in 2004 and 2005 (Craeymeersch et al., 2007), the efficiency of the trawled dredge was estimated to be 50%. As the penetration of the hydraulic dredge and the trawled dredge are about the same, this figure was used for all stations in the calculations of Dutch standing stock. This figure is of the same order of magnitude as found by Beukema (1974) for a van Veen grab for razor shells (*E. ensis*) of about 6 cm in length (60%). In 2004, a total of 71,000 million specimens were assessed to live in the Dutch coastal waters, representing an estimated biomass of 493,000 tons. As individuals of more than 12 cm represent about 12.5 % of the total biomass, the fishable stock thus amounted at about 62,000 tons. Landings thus represent about 0.5 % of the total biomass and 5 % of the Dutch adult stock.

At its present scale, the ecosystem impact of this fishery is considered to be low, owing to the slow speed at which the fishing gear is towed and the fast growth of this invasive razor species (contrary to the local species; see Hauton et a., 2007). Recovery of the sediment after passage of the sampling gear is relatively fast due to its limited width (about 1.2 m) and the sediment composition (sands and muddy sands). Larger densities of *Ensis* being found on clean to slightly muddy sand, the impact of the gear on local turbidity levels is also expected to be low.

3.5.3 A first stock evaluation for Belgian waters

The biomass of larger (> 10 cm) specimens measured in summer 2008 and winter 2010 (see chapter 2) was mapped (Figure 61). Maximum weights per unit surface are found in the same zones in both years, namely: coastal waters between the French border and Oostende (around the Nieuwpoort Bank); the southern part of the Oostende Bank; the northern slope of the Vlakte van de Raan.

A tentative stock assessment for the Belgian coast is given in Table 10, based on overall count of specimens (adults as well as juveniles). It was compared with the spring assessment of the Dutch coast of 2008 (Goudswaard et al., 2008). The Dutch coastal area (delimited by the 12 nautical miles line; 6438 km²) is larger than the Belgian coastal area (1356 km²). The Belgian stock was also compared more specifically with that of the neighboring Voordelta area (1276 km²). Due to the differences in surface area, the comparison was made based on a standardized surface area (1 hectare). The estimated stock at the Belgian Coast (91822 ind/ha) is about half of that of the Voordelta (206615 ind/ha), which displays highest total densities.

	Belgian Coast (1356 km²)		Voordelta (1276 km²)		Dutch Coast (6438 km²)	
	Amount (millions)	Average individuals /ha	Amount (millions)	Average individuals /ha	Total amount (millions)	Average individuals /ha
Small (<12cm)	3884	28627	10121	79307	26431	41054
Big (>12cm)	2101	15486	1472	11539	4486	6968
Siphons	6472	47709	14775	115769	34837	54110
Total	12458	91822	26369	206615	65756	102132

Table 10. Preliminary stock assessment for Belgian waters, on the basis of data from the SCHELP survey (summer 2008) and IMARES survey (Goudswaard et al, 2008).

To evaluate the fishable stock and expected yields, we also focused on areas with larger densities of adults (*Ensis* "banks"). The surface area where measured densities are larger than 50 ind/m² represented approximately 65 million m² (4% of the area within the 12 nautical miles line). Supposing that the discrete samples are

representative of the continuum in the sediment, an average density of 80 ind/ m² can be found in these areas, thus representing a total of about 5,200 million specimens larger than 10 cm (about half of the total population). Taking the modal size-class (11-12 cm; see Figure 12) wet weight of a specimen (0.013 kg) as average weight for the larger size-classes, this would represent a minimum fishable stock of about 70,000 tons. Given the fact that the used sampling gears tend to under-estimate the real stock (Craeymeersch et al, 2007), this is likely a minimum figure.

Keus (2008) took a yield of 5 kg/m² as basis for stock assessment. Using the average weight of the 11-12 cm size-class (0.013 Kg), this would represent a density of 385 ind/m². The surveys evidenced that in Belgian waters, such a density of larger individuals is rarely attained. An intermediate weight of 2 to 3 kg/m² would thus seem a more realistic average yield of marketable specimens across *Ensis* banks in Belgian waters.
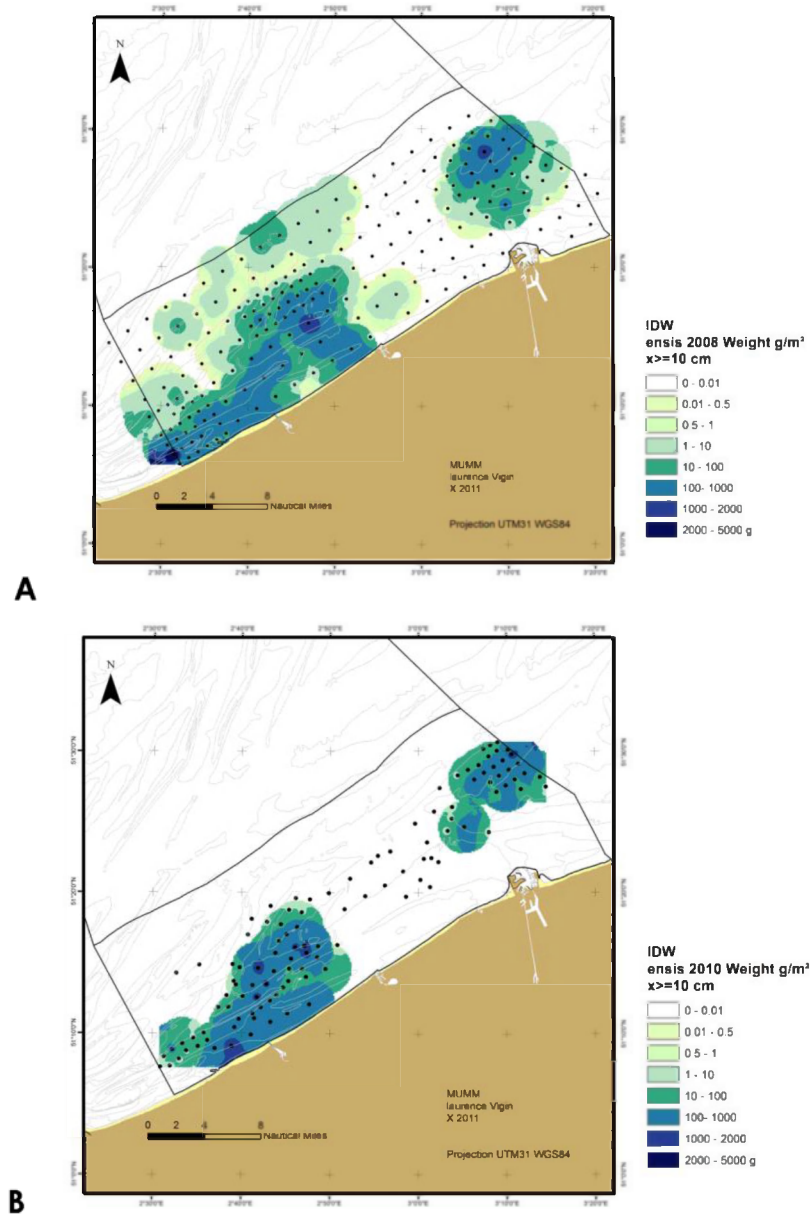

Figure 61. Distribution of wet weight/m² for specimens of *E. directus* larger 10 cm, with sampling locations (interpolation method: Inverse distance Weighing - IDW). A: summer 2008 survey; B: winter 2010 survey.

Kenchington et al (1998) noted that at 10 cm, the animal has spawned twice, but larval production is probably not yet optimal. To ensure sustainability of yields, they suggested that as minimal length of 12 cm should be considered to ensure stock viability in the native area of *E. directus*. In Dutch waters, the monitoring of *Ensis* stocks shows a clear increase in biomass in the last years (see figure 37) despite active fishery ongoing. The targeted fishery at its present scale is thus considered to be within safe limits. Population data acquired within this project indicate that densities of *Ensis* appear to be lower in the Belgian zone, but also that abundant yearly recruitment takes place to renew the stock.

3.5.4 <u>Potential impact on the ecosystem</u>

As outlined by Keus (2008), the potential impact of hydraulic dredging on the ecosystem is to be assessed with reference to the seafloor structure and composition, local hydrodynamics, associated benthos and predators. On the other hand, the scale of the fishery will be a major parameter. Our impact assessment is based on the *Ensis* fishery as is presently carried out in Dutch waters, in particular in the Voordelta area, namely with few fishing vessels.

The project evidenced that higher densities of adult razor clams occur in slightly muddy sands. These patches largely overlap with high trawling intensities in coastal waters (Depestele et al, 2011). In such a trawled area, the potential impact of hydraulic dredging will most probably be negligible. In the absence of trawling, sediment resuspension might become larger than surrounding background levels, calling for an *ad hoc* evaluation. Hydraulic dredging is considered harmful especially for sheltered habitats (Woolmer et al, 12011). The areas where maximum densities of *Ensis* are found can be considered as moderately exposed to disturbance by tidal currents and storm-induced waves, due to their shallowness (see section 3.2.3.3.2). It is however not known to what depth and with what periodicity the seafloor may be disturbed by these natural events. The deep disturbance induced by the hydraulic dredge (about 30 cm in the sediment) needs proper evaluation. However, the fishing gear width is limited (1.25 m) and it is the fishing intensity which will determine whether significant ecosystem impact may take place. The effect on the sediment will also depend on its density. Addison et al. (2006) recommend restricting hydraulic dredging where surface sediment is not too compact, which will favour a better survival of the associated benthos.

For benthos, impact studies indicate a low level of mortality in the dredge track for most fragile burying animals. Hauton et al (2002) indicated 60-70 % survival for the urchin *E. cordatum*, the dominant megafaunal component of their investigated area (Clyde estuary, Scotland), after hydraulic dredging. However, local *Ensis* species as well as the deep-burying common otter shell *Lutraria lutraria* did not survive the process, probably because of their elongated shape and the fact that the siphon tips were damaged in the deepest part of the target sediment, where sand is less fluidized, close to the dredge blade. A slower towing speed was suggested to reduce this impact, which is of interest since only relatively intact *Ensis* are marketable. In this case, razor shells represented only about 30% of the catch, and the associated by-catch is high. Burial rates of dislodged survivors seem high. However, urchins exposed to air during their collection were found to fail in this recolonization process, while dislodged specimens exposed on the seafloor rapidly buried back. Other bivalve species seem to bury back more slowly, which makes them more vulnerable to opportunistic predation (Robinson and Richardson, 1998). Hauton et al (2002) conclude that the obtention of 10 kg of marketable *Ensis* (which

represent about 85 % of the population along the dredge track) can generate 35 kg carrion, which will attract scavengers in the dredge track, and about 7 kg of megafauna able to bury back after dredge-induced dislodgement or discarding. These include under-sized intact *Ensis*.

In the BPNS, the occurrence of *E. directus* displays an overlap of about 70 % with that of the *A. alba* community (B. Merckx, *pers. com.*; see Chapter 2.2. - Ecological features - and Degraer et al, 2008). This community displays high levels of species diversity as well as density and is mostly composed of opportunistic, short-lived species. In the path of the dredge, a large amount of fragile species, including thin-shelled bivalves, will experience high mortality. However, on the basis of aforementioned literature, re-colonization from the sides of the track can be expected to be high. This fishery will however exert a positive effect on their predators, increasing the problem caused by the fishery-induced dominance of scavengers on the seafloor.

Couperus et al (2009) showed that in the Voordelta, Ensis has become a dominant prey item for flat fishes (dab, sole, plaice), but mostly juveniles are preyed upon. Tulp et al (2010) showed that medium-sized *Ensis* represent as well a major prey for some seabird species, including the common scoter (see chapter 3). The hydraulic dredge removing up to 90 % of the *Ensis* specimens present in the sediment (Hauton et al, 2007), this highly efficient fishing technique can in theory lead to a significant reduction of available prey species. However, this effect must be balanced with the relative abundance and distribution of the potential predators (see section 3.4.) and the fact that larger specimens are little foraged upon.

3.5.5 <u>Conclusions</u>

At the light of experience gained with hydraulic dredging in other North Sea countries and the particular ecological features of the BPNS, a small-scale targeted fishery as occurs in Dutch waters (*i.e.* with few ships) is expected to exert limited impact on the ecosystem features, including *Ensis* stocks. This species has been very prolific in the 2000s and obviously takes advantage of the local environmental conditions, which makes it a possible alterative resource for the fishery sector. Contrary to previous evaluations for shellfish fisheries (*Spisula*), the target stocks are abundant and their distribution is now better understood. It must however be stressed that so far, the cause of the increased biomass of *E. directus* remains unknown.

During the workshop, it was put forward that in the present situation, it is not clear yet whether a special authorization for this fishery would encourage local fishermen to adapt to this new resource or attract experienced Dutch vessels. Specific regulations for coastal waters (3, 6 and 12 nautical miles lines) might have to be adapted. A balance must be found between the sector, which must adapt to a reduced fishing area due to recent environmental legislation, and conservation goals for some species (such as scoters). In Dutch waters, a certification by the Marine Stewardship Council (MSC) is currently under evaluation for the Dutch *Ensis* fishery (since August 2011; see http://www.msc.org/nieuws/nieuws/nl-mesheften-eerste-msc-beoordeling-voor-geintroduceerde-soorten). The outcome of this evaluation will be of utmost interest for this issue within Belgian waters.

4 Conclusions and recommendations: policy support

Our research evidenced that the invasive razor clam *E. directus* has become an integral part of the local fauna. From subtidal surveys carried out in summer 2008 and winter 2010, three main adult populations were identified around the Nieuwpoort bank, on the Oostende bank and on the northern slope of the Vlakte van de Raan. These are the areas where they were mostly observed since the invasion. In winter 2010, juveniles displayed a distribution distinct from that of adult populations likely representing sub-optimal areas for settlement and survival. As it takes only two year to reach 9-10 cm in length, the largest part of the population is larger than the minimal fishing length of 10 cm. When high densities of young recruits are removed from the data-set, the size distribution displays a mode at a length of 11-12 cm (corresponding to a two-three year-old specimen), while specimens over 15 cm are rare. The largest part of the populations does thus not live longer than 4-5 years, pointing at a high turnover rate. Further eastwards, in Dutch waters, the distribution extends to the whole Voordelta area. It thrives again in coastal waters to the North of the Wadden Sea. Its distribution was successfully modeled, although its environmental preferences are relatively wide. The species is most abundant at a depth of 10-20m in slightly muddy sand of areas with high levels of primary production.

No direct impact could be detected on the local macrobenthos composition, but we lack appropriate pre- and post-invasion data to form definitive conclusions, and polychaete worms, which numerically dominate the macrobenthos, were not investigated. This apparent lack of impact is in line with the hypothesis that the species occupied an empty ecological niche (Armonies and Reise, 1999). The local *Ensis* species used to occur more offshore, in coarser sediments, hundred years ago, although some overlap with the present-day distribution of *E. directus* existed in the western coastal waters. The rarefaction of local *Ensis* species is probably not related to the invasion. The biomass of *E. directus* has increased in the 2000s in Dutch waters, while that of the cut trough shell *S. subtruncata*, typical of the area, regressed. The timing of this regression was however not even, with earlier decrease in the southern area (Voordelta), and no direct link can be made at this stage.

Important changes have been noticed in the distribution of common scoters in the last years, probably as a result of food shortage during winter months. This species does feed on *E. directus*, on smaller specimens (3-9 cm) which correspond roughly with the size increment of the second growth season. Larvae dispersal being governed by hydro-meteo forcing, there might be more favourable years when larvae occur closer to the shore. The 2010 situation thus indicates that a large amount of new juveniles may have settled outside the feeding range of scoters (*i.e.* towards the offshore). This would be consistent with the idea that *E. directus* does not accurately replace *S. subtruncata* as food item for scoters, as suitable size-classes may not always be available within the feeding range in winter.

A fishery on this species appears, on first analysis, a potential option for the upcoming years. The high levels of recruitment in the last years as well as the occurrence of adult populations within a dynamic environment suggest that if maintained at a small scale, such a fishery would be sustainable and generate limited impact on the ecosystem. Our results suggest that a yield of about 2kg/m² of marketable specimens can be expected on razor clam banks of Belgian waters.

Recommendations for policy support

The case of *E. directus* shows that there is little action to undertake once the invasive species is installed in the subtidal domain. Action needs to be undertaken to limit as much as possible future importation of alien species.

To detect and better understand how invasive species might affect the local ecosystem, it is necessary to dispose of a regular seafloor monitoring programme, where the ecosystem changes are followed up in space and time. Expertise and technical requirements are currently available in the BPNS for the monitoring of specific environmental quality issues such as aggregate extraction or dumping. This spatio-temporal fragmentation of data acquisition was here identified as a problem to analyze long-term patterns. The Dutch experience with respect to shellfisheries, in particular 15 years of intensive monitoring of shellfish stocks, should be considered as a model. The implementation of an integrated monitoring program, combining abiotic and biotic compartments of the seafloor, would ensure that legal obligations of Belgium relative to marine biodiversity are met within the spatial planning of human activities at sea.

The possible implementation of a fishery needs careful evaluation, as was formerly the case for *S. subtruncata* (Degraer et al, 2007). A specific study needs to be carried out to further analyze stock variability in the longer run (relationship with hydro-meteorological forcing) and to better evaluate the ecosystem impact of hydraulic dredging, *e.g.* through targeted experiments. Other, less seabed-disturbing techniques presently under development (electrofishing: Woolmer et al, 2011) should be taken under consideration.

The fishery sector would be in favour for such a shellfishery, since this species has become a problem for trawlers in the areas where it thrives due to damage caused to the nets. How such a fishery would affect local populations of permanent and over-wintering seabirds must be properly assessed along with its impact on macrobenthos and seafloor structure, to evaluate the scale at which this fishery can be carried out.

5 Dissemination and valorization

5.1 ICES annual science conference 2010, 20-24/09/2010, Nantes, France (CM 2010)

5.1.1 <u>Session K – Poster</u>

Houziaux, J.-S.; Kerckhof, F.; Merckx, B.; Vincx, M.; Courtens, W.; Stienen, E.; Van Lancker, V.; Craeymeersch, J.; Van Hoey, G.; Hostens, K.; Degraer, S. (2010). Invasion of the southern bight of the North Sea by *Ensis directus*: ecological consequences and fishery perspectives. ICES CM 2010/K:22

Abstract

The American razor clam *Ensis directus* (Conrad, 1843), first detected in the German Bight of the North Sea in 1978, was observed for the first time in the Belgian sector in 1987. Since then, increasing amounts of specimens wash ashore every year, suggesting that the species forms large nearshore populations likely to significantly alter the local biological communities. A research project was started in 2009 to study this issue, extending the studied area to Dutch waters. On the one hand, the population dynamics will be investigated as well as its impact on the local fauna, making use of pre-1987 macrobenthic data-sets; using knowledge acquired since some years on benthos – sediment relationships in this area, optimum habitats for this alien species will tentatively be determined through a modeling exercise ("habitat suitability modeling") and acoustic seafloor mapping. On the other hand, the impact of altered composition of macrobenthic communities on populations of seabirds will be evaluated. These data will further enable us to evaluate the feasibility, sustainability and ecological impact of a targeted *Ensis* fishery within Belgian waters, incorporating lessons learned from commercial vessels operating in the nearby Dutch waters. In this contribution, our preliminary results are presented and discussed.

5.1.2 <u>Session Q – Oral presentation</u>

Houziaux J.-S.*, Haelters J., Kerckhof F., Degraer S. and Jacques, T. (2010). Baseline benthic biodiversity of the Belgian part of the North Sea: lessons learned and the way forward with ecological restoration goals. ICES CM 2010/Q:14

Abstract

The determination of "baselines" is a necessary step to implement ecological 'restoration'. A baseline can be defined as the ecosystem state under levels of anthropogenic pressure which do not negatively affect its functioning. Historical situations providing us with information on marine ecosystem structure and functioning prior to anthropogenic impacts are thus invaluable to determine the extent of human-induced environmental change and to set targets for the future. The ecology of the Belgian part of the North Sea, now subject to intense human-induced disturbance, has been intensively studied since the 1970s. Earlier, in the first decade of the twentieth century, ecological surveys were also conducted by Gustave Gilson, an early contributor to ICES science. The resulting unpublished archive (specimens and written records), kept at the RBINS, was researched between 2003 and 2009 and was supplemented with an analysis of the pre-1900 literature. The data were compared to the recent situation, targeting sub-tidal sediments and their benthos. Historic "baselines" were drawn for benthic habitats located offshore (Ostrea edulis beds; gravels) and near-shore (coastal sediments) and were placed back in their context of anthropogenic pressure. Observed long-term (> 100 years) biodiversity shifts are consistent with the expected impact of human activities during the 20th century. How such detailed historical data alter our perception of "ecosystem health" thus far mainly based on post-1970 knowledge is discussed and has implications for the definition of meaningful and achievable marine ecological restoration goals (e.g. under the EC 'Habitat' and 'Marine Strategy' framework directives).

5.2 BELSPO project "QUEST4D" - seafloor characterization

Data have been valorized in the context of BELSPO Quest4D (Van Lancker et al., 2011), a research network studying short- to long-term ecosystem changes. Seabed imagery was merged, and datasets were exchanged to optimize interpretations on habitat suitability of *E. directus*. The collaborative work is demonstrated in the Msc thesis of Tafare Breine (2011):

Tafara Breine, N. (2011). Managing biodiversity between estuary and sea: A delicate eternal triangle. M.Sc. Thesis, Universiteit Gent, 40p.

Abstract

A small scale study on macrobenthic diversity was conducted along the delta front of the outer delta of the Scheldt estuary. North of this area, a series of flood and ebb dominated channels are present, mostly hosting large bedforms. The study area is situated at the northeastern end of such a flood channel that borders the deltafront, but also connects to a disposal ground for dredged materials at about 10km to the south-west. High resolution multibeam data was used to visualize bottom morphology and calculate depth and rugosity. Together with vertical current profiling and sediment data they were analyzed for their ability to explain observed patterns in benthic diversity. Biological samples allowed detecting the presence of the *Abra alba* community with very high abundances of the tube dwelling habitat engineer polychaete *Owenia fusiformis*. However, the study area is very heterogeneous and hosts different habitats in close proximity. Species assemblages were most influenced by *O. fusiformis* presence and sedimentological characteristics. The stations in the deeper waters were influenced by a flood channel, and were different from the station along the upper slope that is merely subdued to the ebb current. In between a poor area existed where a medium grained sand-layer was deposited after a storm. No direct proof for the influence of the Western Scheldt or the disposal site was found. To further investigate the habitat mosaic, more elaborated data are needed on water current patterns and velocities on a broader scale, and data on food availability and suspended particulate matter. Within the European framework the acoustic imaging of benthic communities is further valorized as a case study for fine-scale seabed habitat mapping (EU FP7 Research Infrastructure **Geo-Seas**). As an invited talk, results are also presented at an international workshop on Stock Assessments, organized by IMARES (EU+USA).

5.3 PhD thesis: B. Merckx, Universiteit Ghent

Merckx, B. (2011). Modeleren van habitatgeschiktheid en gemeenschapsstructuren van marien benthos (modelling of habitat suitability and community structures of marine benthos). PhD Thesis. Marine Biology Section, Ghent University: Gent. ISBN 978-90-77713-87-7. 309 pp.

Abstract

This PhD focuses on the development of powerful statistical models which cope with modelling pitfalls typical for spatial data and data assembled from various sources, but which are often ignored. In this respect issues such as spatial autocorrelation, sampling bias and overfitting are addressed. The aim of this thesis is two-fold: adapting the current modelling process in such a way that potential pitfalls are revealed and circumvented; and getting insight in the ecology and biodiversity of the taxa under study on a small and large spatial scale. On a small scale 'species assembly rules' have been investigated for the nematode community; on a large spatial scale the diversity of nematode communities is modelled. This research focuses also on habitat suitability modelling of more than 100 nematode species and two macrobenthic species.

5.4 Peer-reviewed publication

Tulp, I., J. Craeymeersch, M. Leopold, C. van Damme, F. Fey & H. Verdaat (2010). The role of the invasive bivalve Ensis directus as food source for fish and birds in the Dutch coastal zone. Estuarine, Coastal and Shelf Science 90: 116-128.

Abstract

Abstract: The razor clam *Ensis directus* was introduced to Europe presumably as larvae in ballast water around 1978. Starting in the German Bight it spread northward and southward along the continental coastline. Currently it is the most common shellfish species in the Dutch coastal zone, where it mainly occurs in the Voordelta and off the Wadden Sea islands. The mean density of *E. directus* in the Dutch coastal zone increased from around 2-5 individuals m^{-2} in the late '90's to around 12-19 individuals m^{-2} from 2002 onwards. Diet studies show that E. directus makes up a significant proportion in the current diet of plaice, sole, dab, flounder and dragonet and in the diet of eider and common scoter. In recent years *E. directus* contributed 20-100% of the total wet weight in fish stomachs. The proportion E. directus in the diet increases with fish length. Based on stomach contents of oiled and beached birds and of faeces samples the recent frequency of occurrence is 85-90% in eider and 26% in common scoter. Also waders and corvids prey on *E. directus* but the contribution to the diet is still unquantified. Because of its great burying depth the species is not easily accessible. Fish either profit from massive die offs that regularly occur, or they extract (probably only the smaller) individuals from the sediment. Sea ducks can extract *E. directus* from the sediment, while shorebirds and gulls feed on dying *E. directus* washing up on the shore. *Ensis directus* is possibly an important food item for fish and seabirds when they occur in high densities and in the right size classes. Since the availability depends greatly on massive die offs, shell size, burying depth and water depth, it is probably not a very reliable food source. Judging from the role *E. directus* currently plays for the higher trophic levels, its introduction must have caused a major change in the food relations in its distribution area.

6 Acknowledgements

7 References

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Annex: end-user committee meetings

End-user committee composition

- Marcel Rozemeijer, Senior Policy Advisor, Rijkswaterstat Waterdienst (NL)

- Willy Versluis, Reder ter Zeevisserij

- Nathalie De Snijder, Natuurpunt vzw

- Danny Huyghebaert, Project coordinator, Stichting Duurzame Visserijontwikkeling vzw (SDVO)

- Marc De Zutter, Voorzitter, Vlaamse Schelpdier – en viscoöperatie cvba (VSVC)

- Elke Van den Broeke, Adjunct van de Directeur, Vlaamse Overheid, Departement Leefmilieu, Natuur en Energie - 3G29, Afdeling Milieu-, Natuur- en Energiebeleid, Dienst Beleidsvoorbereiding en -evaluatie

- Geert Raeymaekers FOD Public Health, Food Safety and Environment DG Environment Marine Environment

Meetings

The "kick-off" meeting of the project was held by 20 January 2010.

An intermediate meeting was held by 3 February 2011.

The final workshop was organized by 16 September 2011. The report of this final meeting is given hereafter.

Final workshop report

The meeting took place at the building of UGMM-BMM, Gulledelle 100, B-1200 BRUSSELS

1. Presentation of results and discussion

The morning was dedicated to the presentation of results gathered by the teams.

J.-S. Houziaux (MUMM) presented results about the distribution and population structure. They confirm that this species is now well-established in this part of the North Sea. Three main adult populations were evidenced along the Western coast (Nieuwpoort bank), on the Oostende bank and on the Vlakte van de Raan, respectively. These populations appear to be stable in the studied period (2008-2010) and a very high recruitment event occurred in 2009.

About 70% overlap was observed with the geographic distribution of the richest soft-bottom macrobenthic community of the BPNS (the "*A. alba*" community). Settlement of juveniles occurs throughout the area, depending on dominant hydro-meteorological conditions; in the EnSIS survey of winter 2010, very large densities of juvenile *Ensis* (70% of collected specimens) were observed fringing the northern edge of the adult populations. The maximum observed densities drop fast from more than 3,000/m² for year one juveniles (2-6 cm) to below 200/m² for older specimens (4-5 year old, > 10 cm).

Based on the data gathered in summer 2008 (ILVO-SCHELP survey) and winter 2010 (EnSIS survey), it seems that adult populations are absent from the area north of the Wenduyne bank. This observation was debated, as adults were observed in the area in 2002-2003 (F. Kerckhof observations). In 2010, however, the survey also evidenced that the segmented worm *Owenia fusiformis* forms dense aggregates nearby the populations of *E. directus*, with some level of overlap. Record densities of this hypoxia-resistant species were observed north of the Wenduyne bank and north of the Vlakte van de Raan (up to 11,000 specimens per square meter). The causes for this phenomenon are not elucidated yet.

J. Craeymeersch (IMARES) and B. Merkx (UG – SMB) presented results on habitat suitability modeling (HSM). Within the BPNS, HSM was applied to the joint UG-ILVO macrobenthos database (1977 – 2010; presence) and to the data-set resulting from the EnSIS survey (winter 2010; presence of 1 year-old). A joint Be-NL database of maximum observed densities was also used to model the distribution of adults (> 10 cm) over the entire area, with a minimum threshold density of 10 specimens/m². It must be born in mind that no data was available for the nearshore and very shallow areas. It was found that the species prefers high levels of Chlorophyll a, reduced levels of mud in the sediment, fine to medium sand, a depth of 12-23 m and moderate hydrodynamics. The relative broad tolerance of the species to environmental variability as well as its high reproductive and dispersal capabilities makes it difficult to predict where it would thrive preferentially. The slight shift in the position of adult populations observed between 2008 and 2010, as well as observed massive recruitment next to adults in 2010, suggests that different areas are colonized as older specimens die within a broad favourable zone. A map could be drawn for the potential occurrence of adult populations in the entire Be-NL coastal waters.

V. Van Lancker (MUMM) presented results on acoustic seafloor characterization aimed at identifying typical features of high-density beds of *E. directus* at the surface of the sediment. High resolution seabed images of the Delta front area revealed a complex system of ebb- and flood-dominated channels from which sediment transport patterns can be deduced. This analysis is rendered difficult by the local hydrodynamics (mobile sand sheets) and very high levels of disturbance induced by beam trawling. Still, distinctive seabed features could be observed in areas where

large amounts of the tube-dwelling polychaete *O. fusiformis* were found. Additionally, the occurrence of biologically induced structures indicative of larger densities of *E. directus* could be deduced but not yet validated by enough macrobenthic samples. The field observations tend to confirm results obtained through environmental modeling.

The determination of the impact of the species on the local marine fauna was presented by J.-S. Houziaux. It is hampered by a relative lack at appropriate data. Macrobenthic data acquired since the mid-seventies within the BPNS do not enable to identify the extent to which the occurrence of *E. directus* has altered the local macrobenthos. The historical (1900) records of the Royal Belgian Institute of Natural Sciences (Dept. of Invertebrates) showed that local species of *Ensis* (*E. arcuatus, E. ensis* and *E. minor*) used to occur predominantly off the western coast, further offshore than E. directus in the present-day. The alien perhaps replaced the local species in coastal waters but cannot be held responsible for their rarefaction further offshore. In the data-sets resulting from the surveys of summer 2008 and winter 2010, no clear pattern could be detected in multivariate ordinations, except perhaps a slight trend in epibenthic invertebrates which could result from the provision of numerous empty shells at the surface of the sediment. It is concluded that no effect of the invader can be evidenced thus far, perhaps partly because the area is altered by various anthropogenic pressures since decades.

W. Courtens (INBO) presented results on the suitability of the BPNS for the common scoter *Melanitta nigra* based on habitat suitability modelling of environmental parameters and food preferences, including occurrence of E. directus. It was shown that especially the western, clearer coastal waters are suitable for this species, for which *E. directus* (6-9 cm size group) now represents an important food item. The population of common scoter however underwent drastic changes recently, which are thought to originate from changes in food availability (drop in populations of S. subtruncata).

2. Afternoon round table: An Ensis-targeted fishery in Belgian waters?

The findings of the project were summarized and examined at the light of experience acquired thus far on commercial fisheries existing in the North Sea, in particular in Dutch waters (*E. directus*) as well as in Scottish waters (local species: *E. arcuatus*). Commercial razor clam fisheries are mainly carried out with hydraulic dredges, which fluidize the sediment down to 30 cm below its surface. The dredge, which generally has a width of less than 2 meters, is hauled extremely slowly, with the ship navigating towards its anchor. Various references state that the impact of this technique on the seafloor and its benthos is very limited, with a fast recovery taking place from the borders of the dredge track within few weeks, and relatively low mortality of fragile species such as the green urchin *E. cordatum*. The limited direct impact owes to the very low speed at which the fishing gear is towed; the overall ecosystem impact will still depend upon the fishing intensity. In Dutch waters,

landings amount at about 3,000 metric tons per year, for a total estimated fished surface of 0.6 km². It has been estimated that the proposed maximum yield (8,000 metric tons) would represent about 0.5 % of the standing stock for a total fished area of 160 ha or 1.6 km². It is thus considered that the Ensis fishery in its present state (total permit granted: 8; 3 to 5 are effective in the Voordelta and further North) has a very limited impact on the ecosystem and the fishery appears to be sustainable. In The Netherlands, maximum yields are proposed by the sector itself.

In the BPNS, the standing stock appears to be similar as in the Voordelta area. Noticeably, *E. directus* biomass seems to have considerably increased the last years in the latter area, while a sharp rarefaction of *S. subtruncata* takes place (data IMARES). A specificity of the Belgian zone is perhaps the increased amounts of mud in the sediment, which could result in higher rates of re-suspension of particulate matter resulting from such a hydraulic dredge fishery. High recruitments are regularly observed at sea and the population structure as determined in 2008 and 2010 appears to be stable. The population is found in the habitat of the *A. alba* macrobenthic community, which in the present-day essentially shelters opportunistic species with high turnover rates and variability, and thus probably a fast recovery. An interaction is expected to occur with other fisheries (mainly beam trawling) through seafloor disturbance. Similarly, the *Ensis* fishery would take place in the area where scoters preferentially feed, and an evaluation of disturbance for scoter populations needs to be carried out.

The seafloor structure itself will be altered by hydraulic dredging. Recent estimations suggest that the particulate matter re-suspension by bottom trawlers may be as high as 30% of the effect of a storm. Hydraulic dredging will put back fine material into suspension. A major concern of such a fishery lays in its potential impact on marine biodiversity, recalling past experience with a (rejected) proposal toward a *Spisula* fishery in the BPNS (two fishing licenses requested). Contrary to then, the population of Ensis is better defined in space (large overlap with the *A. alba* community) and the stock appears to be stable, notwithstanding the unknown longer term variability which needs further monitoring, as in The Netherlands. A clear regulation must be issued concerning the area where the fishery would take place as well as the season (i.e. autumn / winter). On the other hand, it is underlined that contrary to the formerly proposed *Spisula* fishery, an *Ensis* fishery would be carried out with a hydraulic dredge. This species being large and fragile, it is in the interest of the fishermen to operate the gear as carefully as possible to reduce the proportion of damaged specimens. The use of a hydraulic dredge is thus expected to produce a much reduced impact on macrobenthos than the towed dredges formerly considered for the *Spisula* fishery. The survival of disturbed species after the fluidization however requires specific research to be carried out.

The 'duality' of the sector relatively to the now established populations of E. directus was put forward, since coastal beam trawlers (the main métier) don't appreciate *E. directus* beds due to the damages they cause to the nets. It is questioned whether a specific authorization for an *Ensis* fishery would motivate Belgian fishermen to adapt to this new resource, or attract experienced fishers from The Netherlands. It is argued by W. Versluys that the 3 nautical miles zone should be merged with the 6 nautical miles to provide coastal fishers with a living. It is also argued that an *Ensis* fishery could be enabled through a limited amount of personal authorizations, but this would require a modification of the law since shell fisheries are explicitly forbidden in Belgian waters.

From the aforementioned elements, it appears that the ecosystem impact of an *Ensis*-targeted fishery will depend upon its scale; if a similar scale as in the Dutch waters is considered (*i.e.* fished area and percentage of the standing stock), thus far limited impacts are expected to take place. The fishery is however demand-driven, and a rise of demand consecutively to the availability of the species on the market will have to be carefully followed up. As a conclusion, it seems that the implementation of an *Ensis*-targeted fishery could be profitable with limited ecosystem impacts. Complementary research and monitoring on standing stock and potential impacts of hydraulic dredging will be necessary to confirm this view.